Climate, air quality, and health benefits of a carbon fee-and-rebate bill in Massachusetts, USA

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Abstract

State and regional climate policies in the United States are becoming more prevalent. Quantifying these policies’ health co-benefits provides a local and near-term rationale for actions that also mitigate global climate change and its accompanying harms. Here, we assess the health benefits of a carbon fee-and-rebate policy directed at fuel use in transport, residential and commercial buildings and industry in Massachusetts. We find that the air pollution reductions from this policy would save 340 lives (95% CI: 82–590), 64% of which would occur in Massachusetts, and reduce carbon emissions by 33 million metric tons, with 2017 as an implementation year, through 2040. When monetized, the benefits to health may be larger than the benefits from climate mitigation, but are sensitive to valuation methods, discount rates, and the leakage rate of natural gas, among other factors. These benefits derive largely from lower transportation emissions, including volatile organic compounds from gasoline combustion. Reductions in oil and coal use have relatively large benefits, despite their limited use in Massachusetts. This study finds substantial health benefits of a proposed statewide carbon policy in Massachusetts that carries near-term and direct benefit to residents of the commonwealth and demonstrates the importance of co-benefits modeling.

Introduction

The transportation and building sectors contribute heavily to greenhouse gas emissions that drive climate change, and to other harmful air pollutants. In 2015, the transportation sector emitted 1700 million metric tons of CO₂—52% of total US CO₂ emissions. Residential and commercial buildings emitted 570 million metric tons of CO₂—10% of total US CO₂ emissions (US Environmental Protection Agency 2017). Road transportation is the largest contributor to air pollution-related mortality in the US (Caiazzo et al 2013), followed by electric power generation and commercial/residential buildings. Residential combustion alone has been associated with 10 000 deaths per year in the US (Penn et al 2017). Policies that mitigate climate change by reducing fossil fuel consumption can also have health benefits, often termed ‘health co-benefits’, by improving air quality (Wilkinson et al 2009, Nemet et al 2010, Perry et al 2014, Plachinski et al 2014, Mittal et al 2015, Thompson et al 2016, Haines 2017, Li et al 2018).

Research on co-benefits of carbon policies has found that health benefits can be substantial. The magnitude depends on sectors affected, policy design, and the geographical relationship between source locations and populations exposed to air pollution. In the electrical sector, health co-benefits are greatest for policies that principally displace coal (Siler-Evans et al 2012, Driscoll et al 2015, Buonocore et al 2015, 2016a, 2016b, Li et al 2018). Other studies have modeled hypothetical nation, state, or province-level carbon policies and found that policies targeting multiple economic sectors can have higher co-benefits than policies targeting just one sector, and that health benefits are generally higher if high-emission sources
are reduced (Saari et al. 2014, Thompson et al. 2016, Li et al. 2018). While these studies provide insight into drivers of benefits, they do not examine the benefits of specific state-level policies. This may become increasingly relevant, since with the proposed US withdrawal from the Paris Agreement, 11 states and 271 cities and counties have committed to actions that will meet or exceed greenhouse gas targets required to meet the Paris goals (Sanderson and Knutti 2016, Hepburn 2017, Tollefson 2017, we Are Still In n.d.).

Here, we assess the climate and health benefits of a model carbon fee-and-rebate bill applied to fossil fuel consumption in the transportation, residential, commercial, and industrial sectors in Massachusetts. The fee schedule starts at $10/ton in 2017, and increases by $5 yr⁻¹ until it reaches a plateau price of $40 per ton in 2023, a price similar to many other carbon prices worldwide (Benson n.d., Barrett n.d., The World Bank n.d.). These bills, and our study, exclude electrical generation, since the electrical grid in Massachusetts is covered by the Regional Greenhouse Gas Initiative. We assess benefits from 2017 through 2040, assuming that the carbon price was implemented in 2017. We quantify the health co-benefits in economic terms and compare with monetized climate benefits.

Methods

We developed a model framework that links the following components (figure 1):

- CO₂ emissions and fuel use reductions for Massachusetts from the Carbon Tax Assessment Model (CTAM).
- Emissions of SO₂, NOₓ, PM₂.₅, and VOCs for relevant sectors within Massachusetts.
- A Community Multi-Scale Air Quality Model Direct Decoupled Method (CMAQ-DDM)-based impact-per-ton methodology providing state-resolution health benefits per ton of emissions reduced.

Using this framework, we estimated the reductions in fossil fuel use, CO₂ emissions reductions, the reductions in emissions of NOₓ, SO₂, PM₂.₅, and VOCs and the consequent health benefits, comparing a ‘business-as-usual’ (BAU) scenario with a policy scenario, put in monetary terms using standard valuation methodologies.

Modeling the reductions in fuel use and CO₂ emissions

To estimate reductions in fossil fuel use and CO₂ emissions, we used output from CTAM (Nystrom and Zaidi 2013, Breslow et al. 2014). CTAM is an economic...
model that uses price elasticities, differing by fuel type and sector, along with state-specific economic data, including GDP growth, number of households, and existing policies, to estimate how fuel use will change, sector-wide, in response to changes in the price of the fuel. This model relies on future energy use forecasts from the National Energy Modeling System from the US Energy Information Administration, and existing literature on price elasticities of energy.

The carbon emissions from each fuel type and sector are calculated using standard emissions factors. The carbon fee is then applied to these carbon emissions under a BAU scenario, and the expected reduction in fuel consumption in response to the increase in price is calculated using the price elasticity—how fuel consumption changes in response to the change in price. From this fuel use reduction, we calculate the emissions reductions and health benefits of those reductions. To estimate the monetary value of the reductions in CO2 emissions, we use the social cost of carbon (SCC), a metric for measuring benefits of reducing carbon emissions, and put them in monetary, net-present-value terms. The specific values used here change over time to account for the increase in the marginal impact per ton of carbon emitted in future years, due to nonlinear response of the climate system to increased emissions. They start at $45 per ton (2017 USD) in 2017, and end at $70 per ton (2017 USD) in 2040 (Interagency Working Group on Social Cost of Carbon 2016). We present the benefits of CO2 reductions undiscounted and discounted at 3% yr−1, to be consistent with the discount rate of future impacts internal to each year’s estimate of the SCC of an emission in that year.

**Estimating reductions in emissions of non-GHG air pollutants, and health benefits**

To estimate the reductions in PM2.5, SO2, NOx, and VOCs from the carbon fee-and-rebate policy, we used emissions data from the US EPA National Emissions Inventory (NEI) from 2014 as our BAU emissions. We used these emissions data along with the BAU fuel consumption from CTAM to create sector-and-fuel type-wide emissions factors for fuel use within Massachusetts. We then used these emissions factors to calculate BAU emissions in future years. The matching between CTAM and NEI fuel types and sectors are described in detail in the supplemental (table S1 is available online at stacks.iop.org/ERL/13/114014/mmedia). To estimate the health benefits of the emissions reductions, in terms of mortality avoided, we used a CMAQ-based impact-per-ton methodology that gives source state-resolution estimates of mortality avoided per ton of pollutant reduced, across multiple source sectors (Levy et al 2016, Penn et al 2017). CMAQ is a complex atmospheric chemistry, fate, and transport model that is often used by the US EPA to simulate the air quality impacts of air pollution and energy policies, and is commonly used for research purposes (Buonocore et al 2014, Stackelberg et al 2013, Roy et al 2007, Foley et al 2010, US Environmental Protection Agency, Office of Air Quality Planning and Standards, Air Quality Assessment Division 2012, Fann et al 2011, Appel et al 2017, Arunachalam et al 2011, Byun and Schere 2006).

This impact-per-ton methodology is based on a series of runs with the CMAQ-DDM (Levy et al 2016, Penn et al 2017). DDM allows for the influence of given emissions sources and pollutants across the modeling domain to be tracked within a given simulation (Napelenok et al 2006, Wagstrom et al 2008, Itahashi et al 2012). The impact-per-ton methodology used here was developed using CMAQ-DDM to individually model the sensitivities of ambient concentrations of PM2.5 and ozone to emissions of SO2, NOx, PM2.5, and VOCs in both winter and summer for multiple sectors within each source state in the US (Cohan et al 2005, Bergin et al 2012, Itahashi et al 2012, Levy et al 2016, Penn et al 2017). These sensitivities, providing estimates of the air quality impact per ton emitted for each precursor pollutant, were then used to model the health impacts using a concentration-response function of a 1% increase in all-cause mortality per 1 μg m−3 increase in annual average ambient PM2.5 levels (95% CI: 0.2−1.8) and a 0.4% increase in daily mortality per 10 ppb increase in ozone concentrations (95% CI: 0.14−0.66), population data from the US Census, and baseline mortality data from the Centers for Disease Control and Prevention. These concentration-response functions were similar to those used in previous studies (Roman et al 2008, Driscoll et al 2015, Levy et al 2016, Buonocore et al 2016a, Penn et al 2017, Centers for Disease Control and Prevention n.d., Bell 2004, Bell et al 2005, Schwartz 2005), and the 95% confidence intervals encompass the variability in concentration-response functions from major epidemiological studies of air pollution (Bell 2004, Bell et al 2005, Schwartz 2005, Lepeule et al 2012, Driscoll et al 2015, Levy et al 2016, Penn et al 2017). For transportation-related sources, we applied the average of the summer and winter sensitivities; for building-related sources, we used the sensitivities for winter, which assumes that most of the fuel use in these sectors is for heating (US Energy Information Administration 2018).

To put a monetary value on the lives saved from the emissions reductions, we used the value of statistical life (VSL) (Dockins et al 2004). This is a willingness-to-pay methodology commonly used in regulatory impact analysis and other policy research applications that captures most of the value of the health benefits from emissions reductions (Dockins et al 2004, US Environmental Protection Agency Office of Air Quality Planning and Standards 2011, Siler-evans et al 2012, Thompson and Selin 2012, Siler-Evans et al 2013, Thompson et al 2014, US Environmental Protection Agency, Office of Air and...
Radiation, Office of Air Quality Planning and Standards 2015, Thompson et al 2016, Buonocore et al 2016a). To account for the delay between PM$_{2.5}$ exposure and mortality, we use a standard cessation lag (US Environmental Protection Agency Office of Air Quality Planning and Standards 2011, US Environmental Protection Agency, Office of Air and Radiation, Office of Air Quality Planning and Standards 2015). The VSL for PM$_{2.5}$-related mortality is $8.5 million; the VSL for an ozone-related mortality case is $9.4 million, since there is not a significant lag between ozone exposure and mortality. We calculate the stream of health benefits both undiscounted and discounted at 3% yr$^{-1}$, to be consistent with the values used in the SCC. All monetary values are presented in 2017 USD.

**Results**

**Changes in fuel use and emissions reductions**

Transportation had the highest reduction in energy use, followed by residential buildings. Gasoline was the fuel type with the highest reduction in use, followed by natural gas (figure 2). Reductions vary over time given the increasing carbon price through 2023, with differential responses to rising carbon prices among sectors. Peak reductions in motor vehicle gasoline use occur in 2025 and begin to decrease after 2025. For other affected sources, this peak occurs in 2035 (figure 2).

The air pollutant with the most reductions is NO$_x$, followed by VOCs (figure 3(a)). Reductions in all air pollution emissions except VOCs grow rapidly until 2023, grow more slowly between 2023 and 2025, plateau or decrease between 2025 and 2035 (figure 3(a)); VOCs reach their peak in reduction in 2025 and taper off afterwards (figure 3(a)). The CO$_2$ reductions follow a time trend where they reach a peak in 2025, plateau from 2025 to 2035, and then begin to taper off after 2035 (figure 3(b)). Reductions in NO$_x$ and primary PM$_{2.5}$ emissions are largely from reduced use of gasoline, oil, and diesel in the transportation sector (figure 3(a)). Reductions in SO$_2$ are largely from reduced oil use in residential and commercial buildings, and from reduced coal use in the industrial sector (figure 3(a)). VOC reductions are mostly from reduced gasoline use in the transportation sector (figure 3(a)). CO$_2$ emissions reductions are largely driven by reductions in gasoline use in the transportation sector from implementation to 2025 (figure 3(b)). CO$_2$ reductions from gasoline use taper off after 2025, while reductions from reduced natural gas use grow, becoming roughly equal to reductions from gasoline use in 2035 (figure 3(b)).
Figure 3. (a): Emissions reductions of NOx, PM2.5, SO2, and VOCs due to a model carbon fee between 2017 and 2040, by source sector and fuel type. (b): Emissions reductions of CO2 due to a model carbon fee between 2017–2040, by source sector and fuel type.
Figure 4. (a): Aggregate NO\textsubscript{x}, PM\textsubscript{2.5}, VOCs, and SO\textsubscript{2} emissions reduced per unit energy consumption reduced, by source sector and by fuel type. (b): Aggregate CO\textsubscript{2} emissions reduced per unit energy consumption reduced, by source sector and by fuel type.
Emissions avoided per unit of energy use reduced are very high for the use of oil in transportation, and high across all other sectors (figure 4(a)). Coal has the highest SO2 emissions reductions per unit energy reduced (figure 4(a)). Gasoline and diesel use in the transportation sector also had fairly high VOC emissions avoided per unit energy reduced (figure 4(a)). CO2 reductions per unit of energy reductions are much more even across fuel types and source sectors than for air pollutants (figure 4(b)).

### Health and climate benefits

Between 2017–2040, this policy saves approximately 340 (95% CI: 82–590) lives (table 1, figure 5). The health benefits roughly parallel emissions reductions over time, with a value of $2.9 billion (95% CI: $0.66–$5.2 billion) undiscounted, and a value of $2.0 billion ($0.45–$3.5 billion) when discounted at 3% yr\(^{-1}\) (table 1, figure 5). The health benefits of this policy are mainly driven by NO\(_x\) reductions contributing to both ozone and PM\(_{2.5}\), primary PM\(_{2.5}\), and emissions from VOCs, which contribute to both PM\(_{2.5}\) and ozone formation (figure 5). Reductions in SO2 emissions had a comparatively small contribution (figure 5). Emissions reductions in the transportation sector were the largest contributor (85%) to total health benefits, with health benefits initially driven by reductions in gasoline, but gradually becoming nearly evenly split among diesel, gasoline, and oil as the carbon fee goes into effect (figure 5, table 2). The lives saved mainly occurred within Massachusetts, with 63%, and in the surrounding states—12% in New Hampshire, 8% in New York, 4% in Rhode Island, 4% in Connecticut, 4% in New Jersey, and 2% in Maine (table 2).

Reduction in oil use, largely in buildings and in transportation, also contributed highly to health benefits. Reductions in NO\(_x\) emissions from buildings, largely from lesser oil and natural gas use, predominantly occurred in the winter and produced slight increases in ozone. This slightly reduced the total health co-benefits (figure 5). Reduced oil consumption for transportation also generally has the highest lives saved per unit of energy reduced, especially in transportation, but reduced coal use in industry is also high, largely driven by SO2 reductions (figure 7). When monetized, oil generally has the highest benefits per unit of energy consumption reduced, especially in transportation. Between 2017–2040, the policy would reduce CO2 emissions by approximately 33 million metric tons (figure 2(b)). The CO2 reductions have a value of $2.0 billion undiscounted, and $1.3 billion when discounted at 3% yr\(^{-1}\) (table 1, figure 2(b)).

The monetized benefits are slightly higher for health than for climate, with marked variation across sectors and fuel types (table 1, figure 6). The monetized benefits are largely driven by transportation, followed by the buildings sectors. The benefits of reduced fuel consumption in transportation were driven by health, with a more even split between health and climate for commercial buildings. Residential and industrial benefits were mostly based on greenhouse gas mitigation (figure 6). Across fuel types, the leading contributor to benefits is reductions in gasoline use, followed by diesel, oil and natural gas (figure 6). Health benefits generally contributed more to total benefits than climate for gasoline, diesel, and oil, while climate benefits were greater for natural gas than health (figure 6).

### Discussion

A model carbon fee-and-rebate bill in the state of Massachusetts would advance greenhouse gas emissions’ reductions, while providing substantive, health gains, mainly within state, that have comparable magnitude to the climate benefits when monetized.

The peak reductions occur after the peak price due to the time required for turnover, which is longer for buildings than the vehicle fleet. Reductions in fuel consumption and pollutant emissions reach a plateau and then fall since the model carbon fee is not tied to inflation.

Our results here indicate that when monetized, the health benefits are fairly similar in magnitude to the climate benefits, comparable to findings in previous studies focusing on renewable energy and energy efficiency (Siler-Evans et al 2012, 2013, Buonocore et al 2015, 2016b, Levy et al 2016). Between 2017–2040, this policy has approximately $88 in health benefits per ton of CO2 reduced, and saves 10.3 lives per million tons of CO2 reduced, similar to estimates from an economy-wide cap-and-trade program implemented in the northeast US (Thompson et al 2016), and slightly higher than a carbon standard in the electrical sector in the US (Driscoll et al 2015).
economy, and energy efficiency in buildings, but CTAM also does not model how the changes in fuel demand are being implemented, what the costs are, and which actors incur those costs (Washington State Department of Commerce n.d., Breslow et al. 2014). CTAM does employ a lag structure in the price elasticity, so it is able to appropriately reflect longer-term changes due to, for example, turnover time of building stock or vehicle fleet (Washington State Department of Commerce n.d., Breslow et al. 2014). Since CTAM does not explicitly model how the reductions in fuel use are occurring, it cannot model what changes are temporary and reversible, like switching from driving a personal vehicle to using public transportation or maintaining buildings at a lower indoor temperature; or permanent, like purchasing a more fuel-efficient vehicle or improving home insulation (Washington State Department of Commerce n.d., Breslow et al. 2014). Future changes in emissions due to air quality regulations, changes in combustion efficiency or fuel mix, changes in the use of air pollution controls or other technology changes are not captured. While the use of CTAM and reliance on emissions from NEI may not capture all relevant economic, regulatory, and technological effects that may determine emissions reductions and health gains from a carbon price, the basic framework does reasonably at capturing the main drivers of the benefits of a carbon fee-and-rebate policy, including economic effects associated with price elasticity (Nystrom and Zaidi 2013, Breslow et al. 2014). This modeling framework here is fairly similar to the economic modeling in previous studies (Thompson et al. 2016, Li et al. 2018). Although these studies contain more explicit linkages to other elements of the economy, linkages to trade, and other economic factors, while the model used here exclusively uses in-state changes (Thompson et al. 2016, Li et al. 2018). Use of CTAM alone is also insufficient to understand broader economic impact of a carbon policy, in part, because the economic effects of a carbon price depend on what is done with the revenue (Nystrom and Zaidi 2013, Breslow et al. 2014, Ambasta and Buonocore 2018, California Air Resources Board 2018).

Our health model operates at state-level resolution with average values for winter and summer, so may not perfectly capture effect variability due to population proximity to sources, timing of emissions, or changes in population or baseline population health status over time. Across Massachusetts, the county-level mortality rate varies by a factor of two (872 to 1662 per 100 000), and the social costs of NOx, SO2, and PM2.5 emissions also do not vary substantially (Centers for Disease Control and Prevention n.d., Heo...
The model may still not perfectly capture possible geographical clustering of emissions reductions and areas with high population or more vulnerable populations, introducing some uncertainty. Additionally, the use of residential combustion impact-per-ton values is an imperfect proxy for transportation-related emissions, but the county-level emissions from each source type are reasonably correlated (Pearson correlation coefficients: 0.65 for SO₂, 0.71 for VOCs, 0.73 for PM₂.₅, and 0.98 for NOₓ).

Using emissions and impact rates from a historical base year to estimate the impacts of future changes in air pollution is common (Buonocore et al 2014, 2015, 2016a, 2016b, Driscoll et al 2015, Levy et al 2016, Heo et al 2016a, 2016b, 2017, Penn et al 2017). Ignoring population growth and aging would likely result in an underestimate of benefits. Technology changes over time could result in reduced emissions per unit energy and a corresponding overestimate of the total climate and health benefits. The model framework employed here does not have as high spatial and temporal resolution as other research, but our impact-per-ton functions are based on an advanced atmospheric modeling platform and allow for decomposition of benefits by sector and by emission type.

This analysis excludes several morbidities associated with air pollution including, like cardiovascular and respiratory disease, asthma, heart attacks, stroke, premature birth, low birth weight, lost days of school and work, and neurocognitive diseases (Gilliland et al 2016a, 2016b).
We also do not include ecological benefits of reduced air pollution, like increased productivity for crops and timber, and decreases in eutrophication (Jaworski et al 1997, Aldy and Kramer 1999, Wittig et al 2007, Van Dingenen et al 2009), or health impacts from the reduction in hazardous air pollutant (Sunderland et al 2016). We also do not include other health and environmental impacts of extraction of the fuels whose use is avoided, including methane leaks from natural gas infrastructure, strain on underground natural gas storage facilities, air and water pollution and consequent health impacts from natural gas extraction, and possible impacts from accidents in oil and gas extraction or transportation (Adgate et al 2014, Brandt et al 2014, McKenzie et al 2014, Subramanian et al 2015, Brandt et al 2016, Peres et al 2016, Torres et al 2016, Butkovskyi et al 2017, Harriman et al 2017, Michanowicz et al 2017).

The climate benefits may be quite sensitive to methane leakage across the natural gas supply chain. A leak rate of 2.3% and a social cost of methane of $1200 per ton would result in an additional ~$138 million in benefits from climate mitigation, increasing the total benefits due to reduction in natural gas consumption by around 18% (Interagency Working Group on Social Cost of Carbon 2016, Phillips et al 2013, Alvarez et al 2018). Using the global average social cost of methane of $4600 per ton, which incorporates health impacts, makes benefits of reduced leakage closer to $530 million, or a 31% higher (Shindell 2015). Higher leakage rates, or higher weighting of near-term climate impacts would increase this sensitivity substantially. Our SCC may be an underestimate, as it may miss some important health impacts, effects on economic growth, and not appropriately deal with the distribution of climate impacts across space or time (Arrow et al 2013, Diaz and Moore 2017).

This study adds to existing literature on the health co-benefits of efforts to mitigate climate change. We examine the effects of a multi-sector fee-and-dividend bill, not including electricity, whereas much of the existing health co-benefits literature applies to electricity generation. It also examines an individual state plan, as opposed to regional or national policies. That said, the benefits of applying a carbon price to electricity generation can be around 3.9 lives per million tons of CO2 emissions reduced (Driscoll et al 2015). Another study examining nationwide clean energy standards found that the value of the health benefits
can exceed the policy implementation costs (Thompson et al. 2014).

With two years remaining to change the global trajectory of carbon emissions to meet the goals set forward by the Paris Agreement (Figueres et al. 2017), and the US having announced its intention to leave the agreement (Tollefson 2017), regional, state, and local carbon pricing initiatives have become more important to the nation as a whole (Hepburn 2017). Understanding the health co-benefits of different policy options can aid in the design of policies, help incentivize their implementation (Petrovic et al. 2014, Bain et al. 2015), and help sub-national governments contribute to the world achieving the goals of the Paris Agreement.

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References

Adgate J L, Goldstein B D and McKenzie L M 2014 Potential public health hazards, exposures and health effects from unconventional natural gas development Environ. Sci. Technol. 48 8307–20
Aldy J E and Kramer R A 1999 Environmental equity and the conservation of unique ecosystems: an analysis of the distribution of benefits for protecting southern appalachian spruce-fir forests Soc. Nat. Resour. 12 93–106
Alvarez R A et al. 2018 Assessment of methane emissions from the U.S. oil and gas supply chain Science 361 186–8
Ambasta A and Buonocore J 2018 Carbon pricing: a win-win environmental and public health policy Can. J. Public Health 1–3 (https://doi.org/10.17269/s41997-018-0099-5)
Anderson H R, Favarato G and Atkinson R W 2011 Long-term exposure to air pollution and the incidence of asthma: meta-analysis of cohort studies Air Qual. Atmos. Health 6 47–56
Appel K W et al. 2017 Description and evaluation of the Community Multiscale Air Quality (CMAQ) modeling system version 5.1 Geosci. Model Dev. 10 1703–32
Arrow K et al. 2013 Determining benefits and costs for future generations Science 341 349–50
Arunachalam S, Wang–B, Davis N, Baek B H and Levy J J 2011 Effect of chemistry-transport model scale and resolution on population exposure to PM2.5 from aircraft emissions during landing and takeoff Atmospheric Environ. 45 3294–300
Bain P G et al. 2015 Co-benefits of addressing climate change can motivate action around the world Nat. Clim. Change 5 226
Barrett M J An Act Combating Climate Change (https://malegislature.gov/Bills/190/SD1021/District)
Bell M L 2004 Ozone and short-term mortality in 93 US urban communities, 1987–2000 J. Am. Med. Assoc. 292 2372
Bell M L, Dominici F and Samet J M 2005 A meta-analysis of time-series studies of ozone and mortality with comparison to the national morbidity, mortality, and air pollution study Epidemiology 16 436–45
Benson J E An Act to Promote Green Infrastructure, Reduce Greenhouse Gas Emissions, and Create Jobs (https://malegislature.gov/Bills/190/H11726)
Bergin M S, Russell A G, Odman M T, Cohan D S and Chameides W L 2012 Single-source impact analysis using three-dimensional air quality models J. Air Waste Manage. Assoc. 58 1351–9
Brandt A R, Heath G A and Cooley D 2016 Methane leaks from natural gas systems follow extreme distributions Environ. Sci. Technol. 50 12512–20
Brandt A R et al. 2014 Methane leaks from north american natural gas systems Science 343 733–5
Butkovskyi A, Bruning H, Kools S A E, Rijnaarts H H M and van Wezel A P 2017 Organic pollutants in shale gas systems Clim. Change
Caiazzo F, Ashok A, Waizt I A, Yim S H L and Barrett S R H 2013 Air pollution and early deaths in the United States: I. Quantifying the impact of major sectors in 2005 Atmos. Environ. 79 198–208
California Air Resources Board 2018 California Climate Investments Using Cap-and-Trade Auction Proceeds (https://arcb.ca.gov/cc/capandtrade/auctionproceeds/2018_cci_annual_report.pdf)
Centers for Disease Control and Prevention CDC Wonder (https://wonder.cdc.gov)
Roman H A, Walker K D, Walsh T L, Conner L, Richmond H M, Hubbell B J and Kinney P L 2008 Expert judgment assessment of the mortality impact of changes in ambient fine particulate matter in the US Environ. Sci. Technol. 42 2268–74
Roy B, Mathur R, Gilliland A B and Howard S C 2007 A comparison of CMAQ-based aerosol properties with IMPROVE, MODIS, and AERONET data J. Geophys. Res. 112 1–17
Saari R K, Selin N E, Rausch S and Thompson T M 2014 A self-consistent method to assess air quality co-benefits from US climate policies J. Air Waste Manage. Assoc. 65 74–89
Sanderson B M and Knutti R 2016 Delays in US mitigation could rule out Paris targets Nat. Clim. Change 7 92–4
Schwartz J 2005 How sensitive is the association between ozone and gas reporting program protocol
Sanderson B M and Knutti R 2016 Delays in US mitigation could rule out Paris targets Nat. Clim. Change 7 92–4
Roy B, Mathur R, Gilliland A B and Howard S C 2007 A comparison of CMAQ-based aerosol properties with IMPROVE, MODIS, and AERONET data J. Geophys. Res. 112 1–17
Saari R K, Selin N E, Rausch S and Thompson T M 2014 A self-consistent method to assess air quality co-benefits from US climate policies J. Air Waste Manage. Assoc. 65 74–89
Sanderson B M and Knutti R 2016 Delays in US mitigation could rule out Paris targets Nat. Clim. Change 7 92–4
Schwartz J 2005 How sensitive is the association between ozone and daily deaths to control for temperature? Am. J. Respir. Crit. Care Med. 171 627–31
Shindell D T 2015 The social cost of atmospheric release Clim. Change 130 313–26
Siler-Evans K, Azevedo I L and Morgan M G 2012 Marginal emissions factors for the US electricity system Environ. Sci. Technol. 46 4742–8
Siler-Evans K, Azevedo I L, Morgan M G and Apt J 2013 Regional variations in the health, environmental, and climate benefits of wind and solar generation Proc. Natl Acad. Sci. USA 110 11768–73
Stackelberg von K, Buonocore J, Bhave P V and Schwartz J A 2013 Public health impacts of secondary particulate formation from aromatic hydrocarbons in gasoline Environ. Health 12 19
Subramanian R et al 2015 Methane emissions from natural gas compressor stations in the transmission and storage sector: measurements and comparisons with the EPA greenhouse gas reporting program protocol Environ. Sci. Technol. 49 3232–61
Sundler E M et al 2016 Benefits of regulating hazardous air pollutants from coal and oil-fired utilities in the United States Environ. Sci. Technol. 50 2117–20
Talbot E O, Arena V C, Rager J R, Clougherty J E, Michanowicz D R, Sharma R K and Stacy S L 2015 Fine particulate matter and the risk of autism spectrum disorder Environ. Res. 140 414–20
The World Bank Carbon Pricing Dashboard (https://carbonpricingdashboard.worldbank.org)
Thompson T M, Rausch S, Saari R K and Selin N E 2014 A systems approach to evaluating the air quality co-benefits of US carbon policies Nat. Clim. Change 4 917–23
Thompson T M, Rausch S, Saari R K and Selin N E 2016 Air quality co-benefits of subnational carbon policies J. Air Waste Manage. Assoc. 66 988–1002
Thompson T M and Selin N E 2012 Influence of air quality model resolution on uncertainty associated with health impacts Atmos. Chem. Phys. 12 9753–62
Tollefson J 2017 Trump pulls United States out of Paris climate agreement Nature 546 198–198
Torres L, Yadav O P and Khan E 2016 A review on risk assessment techniques for hydraulic fracturing water and produced water management implemented on onshore unconventional oil and gas production Sci. Total Environ. 539 478–93
US Energy Information Administration 2018 Monthly Energy Review (https://eia.gov/totaledenergy/data/monthly/)
US Environmental Protection Agency 2017 Inventory of US Greenhouse Gas Emissions and Sinks: 1990–2015—Main Text (https://epa.gov/sites/production/files/2017-02/documents/2017_complete_report.pdf)
US Environmental Protection Agency, Office of Air and Radiation, Office of Air Quality Planning and Standards 2015 Regulatory Impact Analysis for the Clean Power Plan Final Rule vol 314 (https://3.epa.gov/tnntcasi1/docs/ria/utilities_ria_final-clean-power-plan-existing-units_2015-08.pdf)
US Environmental Protection Agency Office of Air Quality Planning and Standards 2011 Regulatory Impact Analysis for the Final Mercury and Air Toxics Standards (https://www3.epa.gov/tnntcasi1/regdata/RIAs/matsrifinal.pdf)
US Environmental Protection Agency 2012 Air Quality Modeling Technical Support Document: 2017-2025 Light-Duty Vehicle Greenhouse Gas Emission Standards Final Rule Office of Transportation and Air Quality, Air Quality Assessment Division pp 1–153 (EPA-420-R-12-004, August 2012)
Van Dingenen R, Dentener F J, Raes F, Krol M C, Emberson L and Cofala J 2009 The global impact of ozone on agricultural crop yields under current and future air quality legislation Atmos. Environ. 43 604–18
Wagstrom K M, Pandis S N, Yarwood G, Wilson G M and Morris R E 2008 Development and application of a computationally efficient particulate matter apportionment algorithm in a three-dimensional chemical transport model Atmos. Environ. 42 5650–9
Washington State Department of Commerce Carbon Tax Assessment Model (https://commerce.wa.gov/growing-the-economy/energy/washington-state-energy-office/carbon-tax/)
We Are Still In Who’s (https://wearestillin.com/signatories)
Wilkinson P et al 2009 Public health benefits of strategies to reduce greenhouse-gas emissions: household energy Lancet 374 1917–29
Wittig V E, Ainsworth E A and Long S P 2007 To what extent do current and projected increases in surface ozone affect photosynthesis and stomatal conductance of trees? A meta-analytic review of the last 3 decades of experiments Plant, Cell Environ. 30 1150–62
Zanobetti A, Franklin M, Koutrakis P and Schwartz J 2009 Fine particulate air pollution and its components in association with cause-specific emergency admissions Environ. Health 8 1860
Zanobetti A and Schwartz J 2009 The effect of fine and coarse particulate air pollution on mortality: a national analysis Environ. Health Perspect. 117 898–903