Air Quality and Health Co-Benefits of a Carbon Fee-and-Rebate Bill in Massachusetts

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Executive Summary

Two bills have been introduced in the Massachusetts legislature – S.1821 *An Act Combating Climate Change* (Barrett 2017), and H.1726 *An Act to Promote Green Infrastructure, Reduce Greenhouse Gas Emissions, and Create Jobs* (Benson 2017). Both bills would put a fee on greenhouse gas emissions, except for those from electrical generation. Revenue collected from the fee in S.1821 would be deposited in a fund sequestered from general revenue, from which all residents and employers in Massachusetts would receive a rebate, with a slightly higher rebate going to residents of rural areas since residents of these areas drive more. Twenty percent of the revenue from H.1726 would go into a "green infrastructure fund" that would support development in transportation, climate resilience, energy efficiency, and renewable energy. The remaining revenue would be returned to residents of Massachusetts as rebates, weighted toward lower-income households and rural residents; employers would receive rebates based on their number of employees.

These two bills, directed at mitigating greenhouse gas emissions, may also provide substantial co-benefits to health, similar to many other efforts to reduce greenhouse gas emissions. A proposed Massachusetts carbon fee may yield health benefits by reducing emissions of air pollutants like fine particulate matter ($PM_{2.5}$), nitrogen oxides (NO_x), volatile organic compounds (VOCs), and sulfur dioxide (SO_2), which are emitted when fossil fuels are burned. Exposure to air pollution can have a variety of health effects, including premature mortality (Roman *et al* 2008), heart attacks (Madrigano *et al* 2013, Mustafić 2012b), hospital admissions for respiratory and cardiovascular disease (Zanobetti *et al* 2009, Levy *et al* 2012b), asthma exacerbations (Anderson *et al* 2013), stroke (Shin *et al* 2014), lost days of school and work (Lei Chen, Brian L. Jennison, Wei Ya 2000, Gilliland *et al* 2001), and possibly autism spectrum disorder (Talbott *et al* 2015) and Alzheimer's disease (Jung *et al* 2015, Cacciottolo *et al* 2016).

Here, we build a model to calculate the health co-benefits that would result from a carbon fee in Massachusetts. The framework of this model is similar to models used for previous co-benefits studies (Buonocore *et al* 2015, 2016b, Thompson *et al* 2012, Saari *et al* 2015, Thompson *et al* 2016, Levy *et al* 2016, Plachinski *et al* 2014), and uses well-established values and methods from prior research (Penn *et al* 2017, Levy *et al* 2016, Driscoll *et al* 2015, Buonocore *et al* 2016a). This model combines the results of an economic model of the fuel use and carbon emissions reductions that would result from this rule (Breslow *et al* 2014, Nystrom and Zaidi 2013a), emissions data from the U.S. Environmental Protection Agency (EPA) (U.S. Environmental Protection Agency 2017), health impact functions from the scientific literature (Levy *et al* 2016, Penn *et al* 2017, Driscoll *et al* 2015), and standard health benefit valuation metrics (Dockins *et al* 2004).

We model the co-benefits of a carbon fee consistent with the fee schedule specified in S.1821 that starts at $10/ton, and increases by $5 per year until it reaches a plateau of $40 per ton. We find the following cumulative benefits from a rule with this structure from an implementation year of 2017 through 2040:

- 340 lives saved
- 26 respiratory hospitalizations avoided
- 28 cardiovascular hospitalizations avoided
- 20 heart attacks avoided
- $2.9 billion ($2017 USD) of cumulative health benefits between 2017 and 2040, worth $2.0 billion ($2017 USD) if discounted to 2017 at 3% per year

Additional benefits from reduction in air pollution emissions were not quantified here. These include reductions in asthma attacks (Jacquemin *et al* 2015, Brauer *et al* 2002, Anderson *et al* 2013), fewer lost days of school and work (Lei Chen, Brian L. Jennison, Wei Ya 2000, Gilliland *et al* 2001), premature birth and low birth weight (Kloog *et al* 2012, Darrow *et al* 2011, Li *et al* 2016), autism spectrum disorder (Talbott *et al* 2015) and Alzheimer's disease (Jung *et al* 2015, Cacciottolo *et al* 2016), along with benefits to crop productivity, farming, forestry, and reductions in acid rain (Committee on Health, Environmental 2010, Chestnut and Mills 2005, Wittig *et al* 2007, Joseph E. Aldy *et al* 1999).

The results indicate that the health co-benefits of a carbon fee in Massachusetts can be substantial, even though the policy was not designed as a public health measure. The annual health benefits of the carbon fee grow as the carbon fee rises, and peaks in 2035 with this fee schedule.

This work provides further evidence to the growing body of research demonstrating that policies designed to mitigate climate change can also produce substantial health co-benefits (Siler-Evans *et al* 2013, Plachinski *et al* 2014, Saari *et al* 2015, Thompson *et al* 2016, Driscoll *et al* 2015). Understanding these health co-benefits can be important to policy decisions and policy design, since health co-benefits of climate policies can be a powerful motivator for action, due to the fact that the benefits occur in the short-term (Bain *et al* 2015, Petrovic *et al* 2014). Additionally, since the health co-benefits generally begin immediately after emissions reductions take place and in the same region as the emissions reductions, these considerations can become quite relevant for the decision-making process, especially for state and local decision-makers (Driscoll *et al* 2015, Buonocore *et al* 2015, 2016b, Siler-Evans *et al* 2013, West *et al* 2013, Nemet *et al* 2010).

Background

In August of 2008, the Global Warming Solutions Act (GWSA) was signed into law in Massachusetts (Massachusetts Executive Office of Energy and Environmental Affairs 2017a). The GWSA requires Massachusetts to achieve a 10-25% reduction of greenhouse gas (GHG) emissions reductions below 1990 levels by 2020, and an 80% reduction below 1990 levels by 2050 (Massachusetts Executive Office of Energy and Environmental Affairs 2017a).

Massachusetts' most recent GHG emission inventory demonstrates that Massachusetts has made progress in achieving its GHG reduction goals since 2006. The state emitted 75.8 million tons of CO_2-equivalent (CO_2e) in 2013, a 19.7% reduction in emissions since 1990, with all sectors except for the commercial sector achieving reductions compared to business as usual projections (Massachusetts Executive Office of Energy and Environmental Affairs 2017b).

Two bills that would place a fee on carbon dioxide and other GHG emissions have been introduced in the Massachusetts legislature – S.1821 *An Act Combating Climate Change* (Barrett 2017), and H.1726 *An Act to Promote Green Infrastructure, Reduce Greenhouse Gas Emissions, and Create Jobs* (Benson 2017). These two bills would place a fee on GHG emissions from fossil fuel consumption in the state, except for their use for electricity generation and assist in attaining Massachusetts' GHG emission goals (Barrett 2017, Benson 2017).

Under S.1821, the fee would start at $10 per ton CO_2e, and increase by $5 per year, until it reaches a fee of $40 per ton CO_2e (Barrett 2017). The revenue generated from fees from household use would fund a rebate paid to all Massachusetts residents, with rural residents receiving a higher motor vehicle fuel rebate since they are more reliant on vehicles for transportation (Barrett 2017). Under H.1726, the fee would start at $20 per ton CO_2e, and increase at $5 per ton CO_2e until it reaches $40 per ton CO_2e (Benson 2017). Under this bill, the revenue generated from household-related fees would be split: 20% would go into a green infrastructure fund, which would support transportation development, climate resilience, energy efficiency and renewable energy, and the remainder would be distributed to all Massachusetts residents, with greater allotments for lower-income residents, rural residents, and low-income energy assistance. In both bills, fees due to sales to businesses and other employers would be returned to this sector, in proportion to number of employees, except that under H.1726 20% would go to a green infrastructure fund.

The fees set up by each bill would largely be incurred by carbon emissions from fuel use in the transportation sector, along with heating fuel use in commercial, residential, and industrial buildings. In addition to GHGs, fuel use in transportation and buildings also produces air pollution, which harms human health and the environment. This study models the averted air pollution emissions, and associated health damages that would arise from the proposed carbon fee.

The Health Impacts of Air Pollution

Reductions in fossil fuel use can reduce GHG emissions as well as release of harmful air pollutants (Fann *et al* 2013, 2012a, 2012b), including sulfur dioxide (SO_2), nitrogen oxides (NO_x), volatile organic compounds (VOCs), fine particulate matter ($PM_{2.5}$), and others. Some of these chemicals react to form ground level ozone, commonly known as smog. These pollutants also contribute to acid rain, impair crop and timber productivity, and damage ecosystems (Wittig *et al* 2007, Joseph E. Aldy *et al* 1999, Committee on Health, Environmental 2010, Chestnut and Mills 2005). While this report focuses on the health benefits from reductions in these other pollutants, ecosystem benefits would also be expected with reductions in these pollutants.

Exposure to air pollution has been associated with many health effects, including the following:

- Premature mortality (Roman *et al* 2008, Schwartz *et al* 2008, Lepeule *et al* 2012, Krewski *et al* 2009, Pope III and Dockery 2006)
- Heart attacks (Madrigano *et al* 2013, Mustafić 2012a)
- Hospital admissions due to respiratory causes (Zanobetti *et al* 2009, Levy *et al* 2012a)
- Hospitalizations due to cardiovascular causes (Zanobetti *et al* 2009, Levy *et al* 2012a)
- Asthma exacerbations (Anderson *et al* 2013)
- Stroke (Shin *et al* 2014)
- Lost days of school and work (Lei Chen, Brian L. Jennison, Wei Ya 2000, Gilliland *et al* 2001)
- Premature birth and low birth weight (Kloog *et al* 2012, Darrow *et al* 2011, Li *et al* 2016)
- Autism spectrum disorder (Talbott *et al* 2015) and Alzheimer's Disease (Jung *et al* 2015, Cacciottolo *et al* 2016).

The health burden of air pollution from fossil fuels in the United States is substantial. In the U.S., the health burden of air pollution in 2005 was between 130,000 and 320,000 premature deaths per year (Fann *et al* 2012b), largely due to emissions from generating electricity, vehicular exhaust, and area sources such as mining, industry, and burning heating fuels (Fann *et al* 2013). From 2005 to 2016, the contributions from electricity generation and mobile source emissions were expected to decrease substantially due to regulatory requirements and technology change, while contributions from other air pollution sources were expected to stay the same or rise (Fann *et al* 2013).

A recent study estimated that burning fuels for home heating in 2005 contributed to around 10,000 excess deaths each year in the U.S., and that burning fuels for electricity contributed to around 21,000 deaths annually (Penn *et al* 2017). This study also calculated the health burden from these sources on a state-by-state basis, and found that the use of fuels in homes in Massachusetts, mainly for heating, is responsible for 390 deaths yearly. Of these 390 deaths, 64% (250) happen within Massachusetts, while 36% occurred out-of-state due to migration of air pollution across state borders(Penn *et al* 2017).

Health Co-Benefits of Greenhouse Gas Reductions

Policies and projects are often developed with the intent to reduce GHG emissions through disincentivizing fossil fuel consumption. However, many studies have shown that these policies can also improve near term health outcomes and confer so-called health "co-benefits".

Examples of policies and projects aimed at greenhouse gas mitigation but which also provide co-benefits range from clean energy standards and cap-and-trade standards (Thompson *et al* 2014, 2016), to national-scale carbon emissions policies (Driscoll *et al* 2015, Buonocore *et al* 2016a), to implementation of renewable energy and energy efficiency (Buonocore *et al* 2015, 2016b, Siler-evans *et al* 2013).

Health benefits accrue from reduced burdens of air pollution associated disease, including the previously mentioned health impacts. When quantified, these co-benefits may be important to decisions that appear unfavorable based upon a limited assessment of climate benefits. Co-benefits are both near-term and apply to the region in which actions are taken to curb GHG emissions. They extend beyond human health and include improvements to timber and crop yields, as examples (Cuevas and Haines 2016, Kennedy *et al* 2015, Watts *et al* 2015, Committee on Health, Environmental 2010, Chestnut and Mills 2005).

In this analysis, we focus on quantifying co-benefits from avoided mortality, heart attacks, and hospitalizations from respiratory and cardiovascular causes.

The Health Co-benefits of a Carbon Emissions Fee in Massachusetts

Regional Economic Models, Incorporated (REMI) has performed a series of analyses on the economic effects of a carbon price in Massachusetts. (Nystrom and Zaidi 2013a, Breslow *et al* 2014, Nystrom 2016) The most recent analysis mirrors the carbon fee schedule over time in S.1821 (Barrett 2017), starting in 2017. It then projects the resulting changes in fuel consumption and GHG emissions through

2040. The fee schedule starts at $10/ton in 2017, and increases by $5/ton each year until it reaches a plateau price of $40/ton in 2023. Here, we built a model using the output from the REMI analysis to estimate the reductions in emissions of other pollutants (SO_2, NO_x, VOCs, and $PM_{2.5}$) that would result from the fuel use reductions from the proposed Massachusetts carbon fee bills, and the consequent health benefits of these pollution reductions.

To calculate the pollutant emissions reductions from this rule, we used the U.S. EPA National Emissions Inventory (NEI) data for 2014 (U.S. Environmental Protection Agency 2017) to estimate SO_2, NO_x, $PM_{2.5}$, and VOC emissions from these sources without the rule. We matched the 2014 NEI emissions data to the 2014 energy consumption from the REMI analysis by sector and fuel types. We then used this emissions data to calculate a statewide average emissions rate for $PM_{2.5}$, SO_2, NO_x, and VOCs, and applied this across all modeled years to calculate emissions. We then used the reductions in fuel consumption predicted by REMI (Breslow *et al* 2014, Nystrom and Zaidi 2013a) to estimate reductions in air pollutant emissions from the carbon fee.

To estimate reductions in premature mortality, we use health impact functions specific for residential combustion of fuels in Massachusetts from previous studies (Levy *et al* 2016, Penn *et al* 2017). These health impact functions were based on an atmospheric model typically used by the U.S. EPA to model the benefits of air pollution control strategies (Byun and Schere 2013), combined with population data and background mortality rate (Centers for Disease Control and Prevention 2015) and evidence from meta-analyses or large cohort studies on the health impacts of air pollution (Roman *et al* 2008, Driscoll *et al* 2015, Schwartz *et al* 2008). To estimate reductions in hospitalizations and heart attacks, we use the results from an analysis of carbon emissions standards for power plants and derive ratios of hospitalizations and heart attacks avoided per life saved for Massachusetts (Driscoll *et al* 2015, Zanobetti *et al* 2009, Levy *et al* 2012a, Mustafić 2012a).

Reductions of Fossil Fuel Use:

REMI model output (Breslow *et al* 2014, Nystrom and Zaidi 2013a) provided reductions in the use of fossil fuels by fuel type and sector, compared to a "business as usual" case without the rule (Figure 1). From 2017 to 2040, the greatest reduction in fossil fuel use comes from reduction in motor gasoline use, followed by the use of natural gas for heating, use of diesel for transportation, and the use of oil for heating. For all sources, the amount of the reduction grows until 2023, when the carbon fee reaches its peak of $40/ton. For motor gasoline, the reductions due to the fee peak in 2025 and then taper off. For the other affected sources, the reductions peak in 2035. The reductions due to the carbon fee taper after reaching a peak amount because the fee is not tied to inflation. The reductions in fuel use reach their peak some years after the fee peaks due to a time lag between the fee and when people respond to it. Transportation reaches its peak sooner than buildings, because the vehicle fleet has a shorter turnover time than buildings and HVAC equipment, so this sector can respond to price signals sooner.

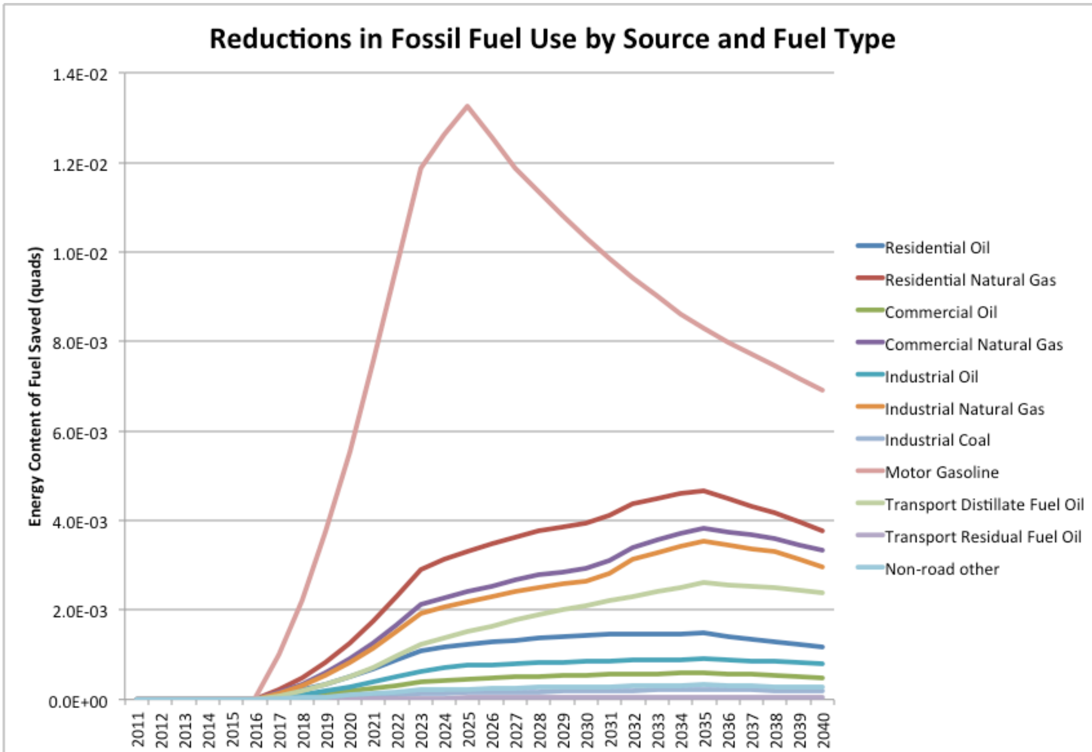

Figure 1: Reductions in the use of fossil fuels due to a carbon fee starting in 2017, by sector and fuel type, in terms of total energy content of fuel, until 2040.

Reductions in Air Pollutant Emissions:

Our model uses emissions factors derived from the U.S. EPA NEI data on emissions of $PM_{2.5}$, NO_x, SO_2, and VOCs (our pollutants of interest) for 2014 (the most recent year with available data) (U.S. Environmental Protection Agency 2017) and the 2014 "business as usual" fuel use from the economic model. We use these values of emissions per unit of fuel consumed by fuel and sector to calculate the "business as usual" emissions. We then calculate the emissions reductions of $PM_{2.5}$, NO_x, SO_2, and VOCs as a result of the fee, compared to "business as usual", using the changes in fuel consumption. Emissions of air pollutants decrease in proportion to the reductions in fuel use (Figure 2). Reductions in NO_x emissions are higher than those of SO_2 and $PM_{2.5}$ since NO_x is the dominant air pollutant for the sources and fuel types affected by the carbon fee. Reductions due to the carbon fee increase from the implementation year through 2035, consistent with the reductions in fuel use. The emissions reductions slow somewhat in 2023, when the fee reaches its peak, and taper further in 2025, when the reductions in the use of motor gasoline decrease.

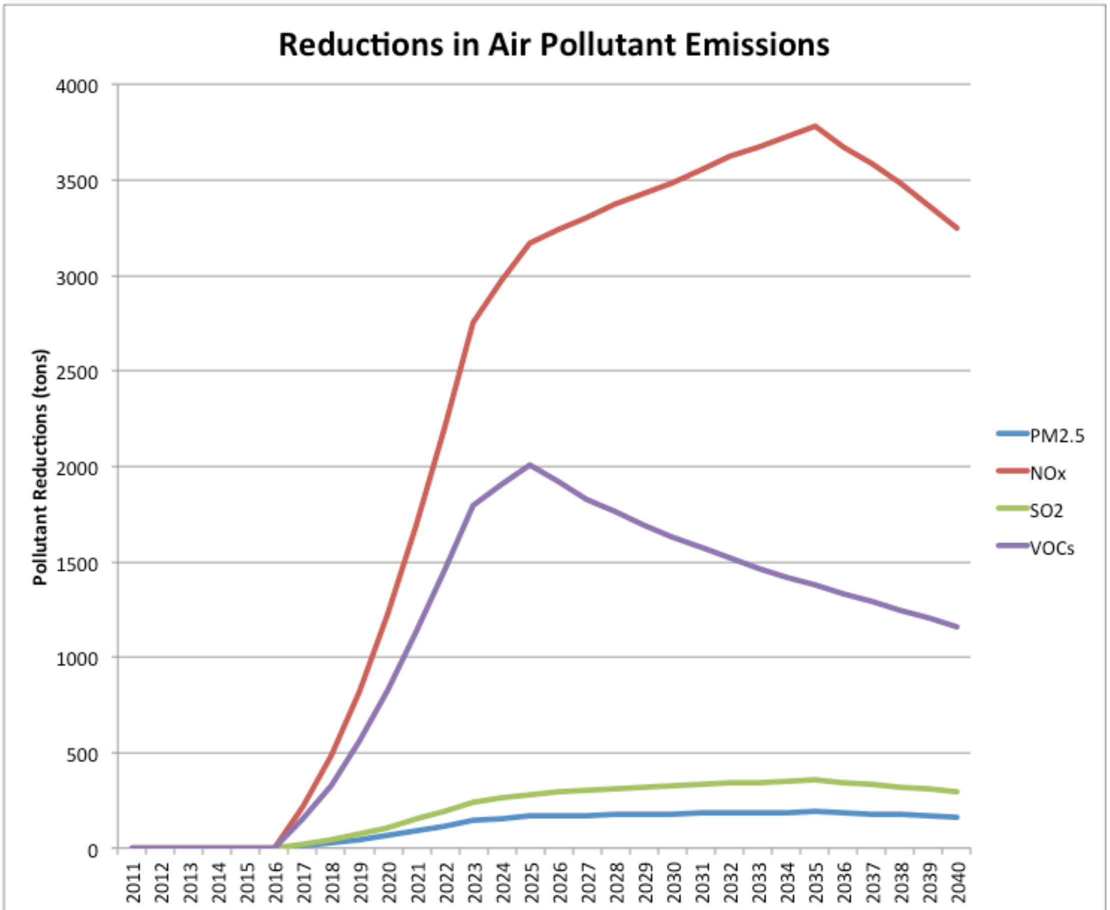

Figure 2: Reductions in key air pollutants due to a carbon fee implemented in 2017, until 2040.

Health Benefits of the Carbon Fee:

Our model uses health impact per ton estimates from peer-reviewed literature to estimate the number of lives saved due to the emissions reductions (Penn *et al* 2017, Levy *et al* 2016). The model then uses ratios between mortality and other health impacts from previous studies (Driscoll *et al* 2015) to estimate other health benefits of the carbon fee. To put a monetary value on the health benefits, this model uses a standard valuation methodology used by the U.S. EPA, other government agencies, many other co-benefits studies, and other research, called the "value of statistical life" (VSL) (Buonocore *et al* 2015, 2016b, Siler-Evans *et al* 2013, Thompson *et al* 2014, 2016, Dockins *et al* 2004, Levy *et al* 2009). The VSL puts a value on health benefits using "willingness-to-pay", or the value a person is willing to pay, for a reduction in their risk of death (Dockins *et al* 2004, Viscusi and Aldy 2003). Our model provides a central estimate, along with 95% confidence intervals, to characterize uncertainty in the estimates (Table 1). The results discussed refer to these central estimates, unless otherwise stated.

Our model shows that compared to a “business as usual” scenario, the cumulative health benefits of this carbon fee from 2017 through 2040 will be:

- 340 lives saved
- 26 respiratory hospitalizations avoided
- 28 cardiovascular hospitalizations avoided
- 20 heart attacks avoided
- $2.9 billion ($2017 USD) of cumulative health benefits between 2017 and 2040, worth $2.0 billion ($2017 USD) if discounted to 2017 at 3% per year
- Reductions in other health outcomes that were not quantified here

Table 1: Estimated cumulative health benefits due to a carbon fee in Massachusetts, compared to business as usual. Central estimates are displayed along with 95% confidence intervals. Values are rounded to two significant digits.

Cumulative Health Benefits between 2017 (implementation year) and 2040	**Central estimate (95% Confidence Intervals)**
Total Lives Saved	340 (82 - 590)
Respiratory Hospitalizations Avoided	26 (14 - 39)
Cardiovascular Hospitalizations Avoided	28 (19 - 36)
Heart Attacks Avoided	20 (12 - 28)
Value of health benefits (billion $2017 USD), undiscounted	$2.9 billion ($0.71-$5.2 billion)
Value of health benefits (billion $2017 USD), discounted to 2017 value at 3% per year	$2.0 billion ($0.49-$3.5 billion)

The health benefits of the carbon fee vary over time, and in a pattern that is consistent with the emissions reductions (Figure 3). The health benefits peak in the year 2035, and rise more slowly between 2023 and 2025, since the emissions reductions slow during this period.

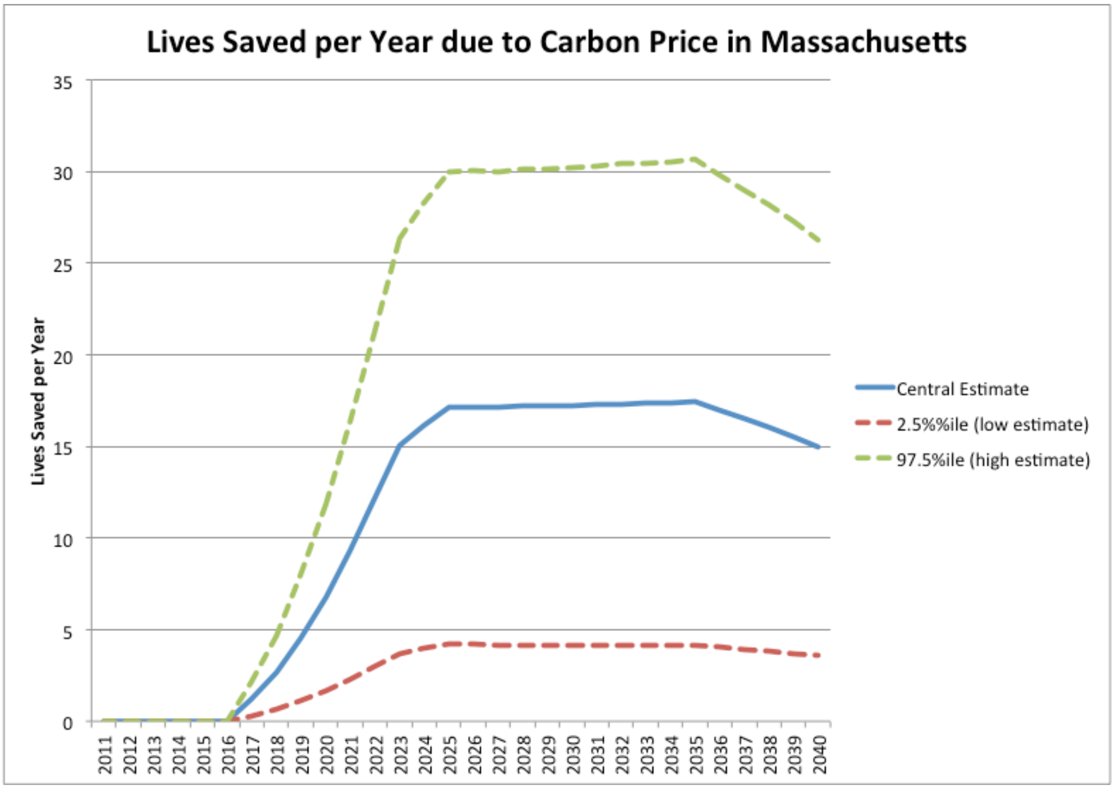

Figure 3: Lives saved each year due to carbon fee. The values displayed are the central estimate and the low and high bounds of the 95% confidence interval.

The lives saved per year by pollutant are shown in Figure 4. Most of the public health benefits are from reductions in NO_x, VOC and $PM_{2.5}$ emissions, with a smaller contribution from SO_2 reductions. The health benefits of avoided SO_2 are low because comparatively little SO_2 is emitted from the fuels affected by the carbon fee. Even though much higher NO_x emissions are avoided than $PM_{2.5}$, the health benefits of avoiding these emissions are fairly similar, because on a per-ton-emitted basis, $PM_{2.5}$ has a much higher impact than NO_x does. NO_x and VOCs both contribute to health impact estimates through formation of $PM_{2.5}$ and O_3 in the atmosphere, but these formation processes take some time and occur most readily only under certain conditions related to season, weather, and emissions of other background sources. Emissions of $PM_{2.5}$ do not need to go through any chemical transformation, and can have higher impacts closer to the source of emissions.

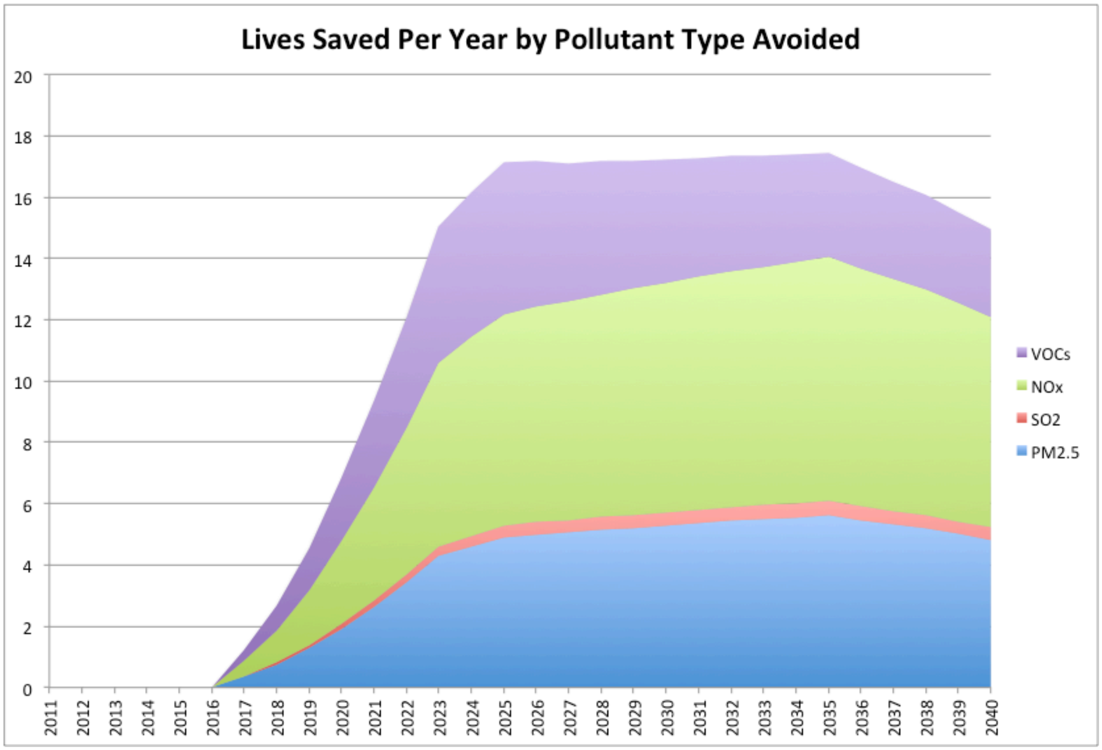

Figure 4: Lives saved per year by avoided emissions type.

Lives saved due to the carbon fee, broken down by sector and fuel type, are shown in Figure 5. The health benefits are largely driven by reductions in the transportation sector, followed by commercial, industrial, and then residential buildings. This is consistent with the reductions in fuel use, and also the emissions profile of each fuel and source type.

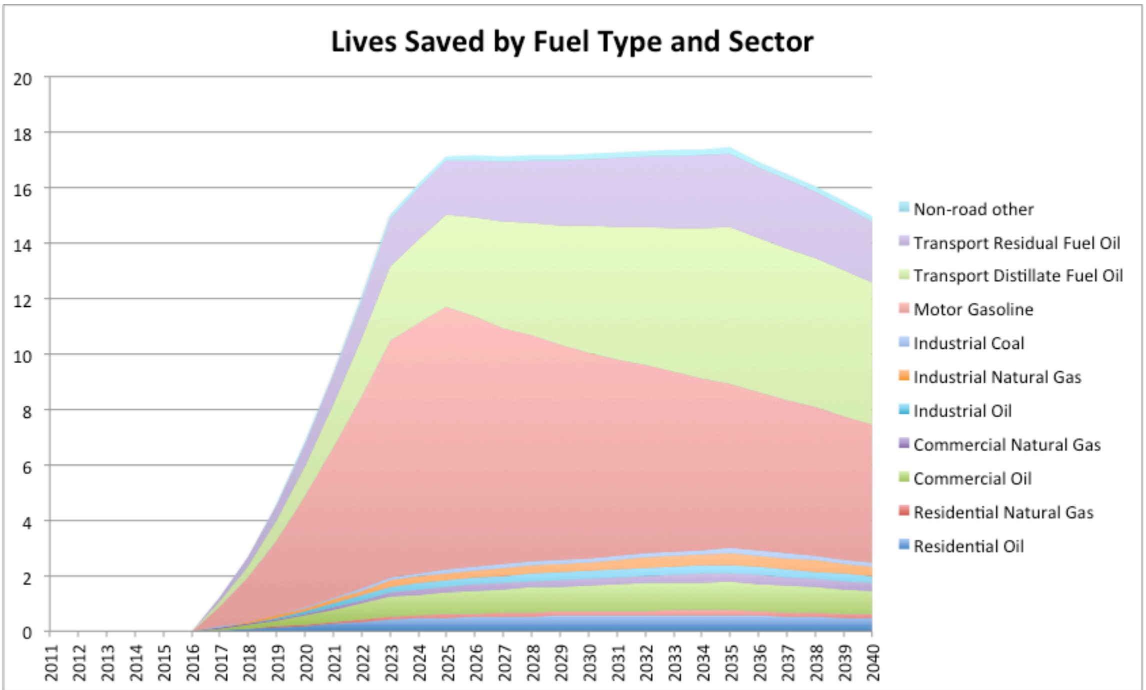

Figure 5: Annual lives saved per year, by source sector and fuel type.

Value of Health Benefits

Our model calculates the economic value of the health benefits of the carbon fee using the value of statistical life (VSL), a standard methodology from the U.S. EPA (Dockins *et al* 2004) applied to each premature death (Figure 6). We do not calculate the monetary benefits for other health outcomes, but prior research shows that VSL captures most of the benefits of air pollution reductions (Buonocore *et al* 2016a, Fann *et al* 2009). The estimated cumulative health co-benefits of the carbon fee are approximately $2.9 billion. The trend over time in undiscounted value of health benefits is slightly different from the trend in lives saved over time (Figure 7). This is due to a time lag between $PM_{2.5}$ exposure and mortality, since the estimates are based on long-term exposure studies that found a time lag. This is not the case for ozone exposure and mortality, since the estimates are based on short-term exposure studies.

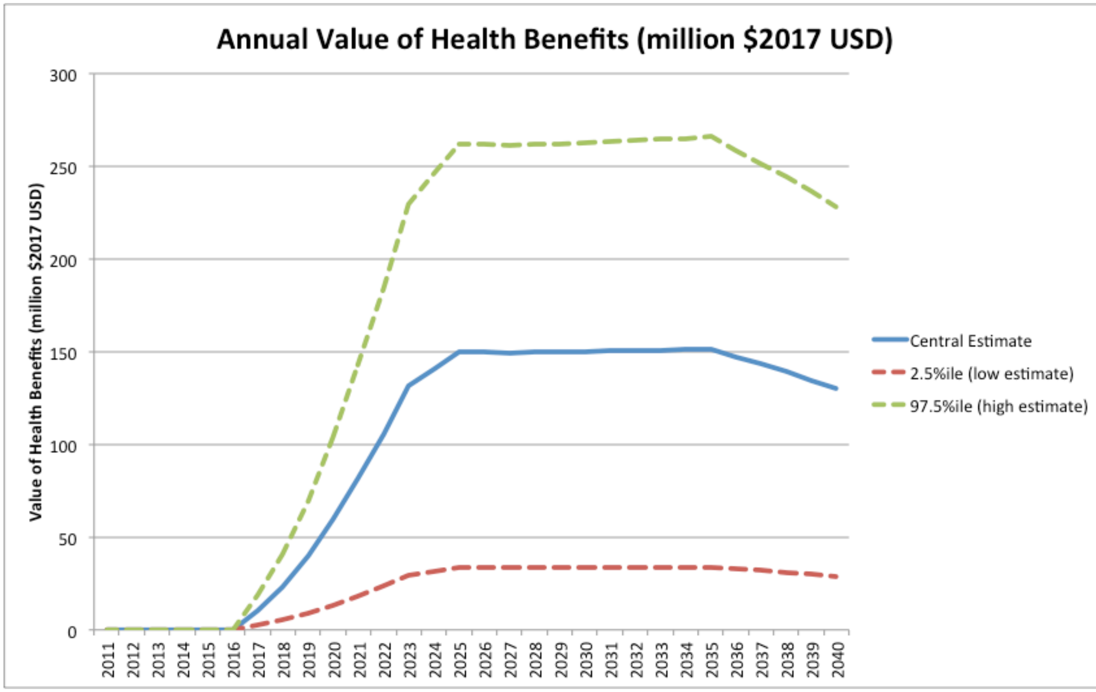

Figure 6: Annual monetized health benefits of a carbon fee, from 2017 to 2040. Values shown are the central estimate, and lower and upper bounds of the 95% confidence interval. Results shown are undiscounted.

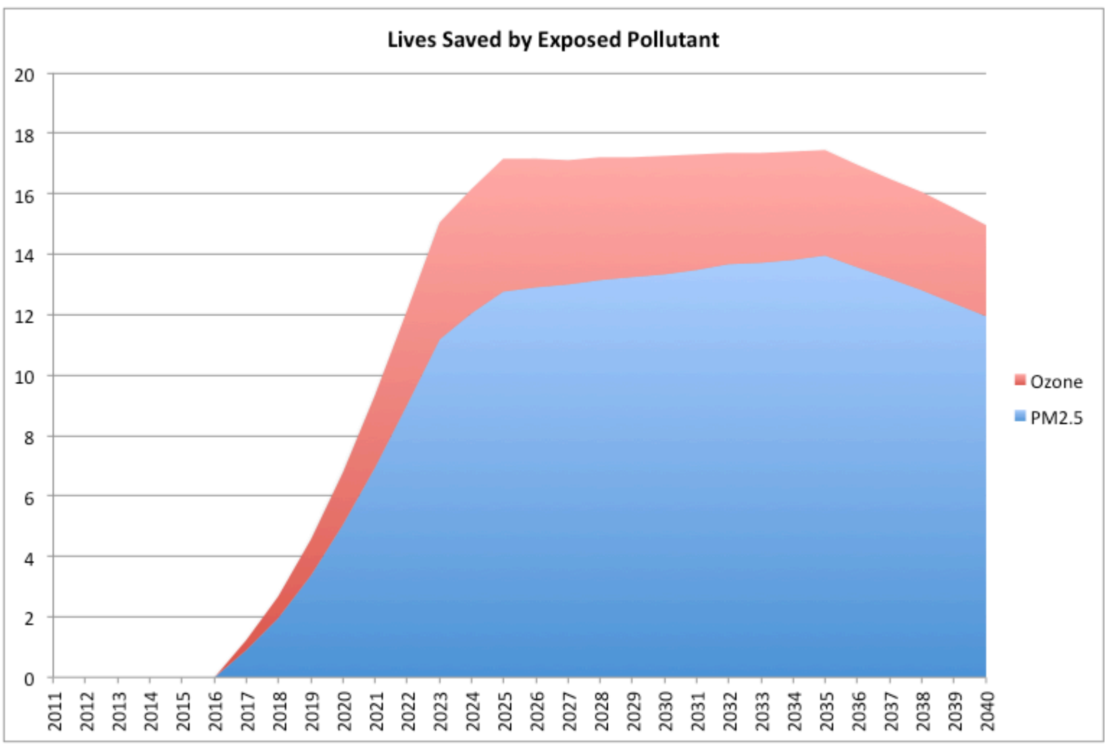

Figure 7: Annual lives saved due to a carbon fee, by reductions in exposure to ozone and to $PM_{2.5}$.

Summary and Conclusion

Here, we build a model designed to calculate the health co-benefits of a carbon fee in Massachusetts. This model used output from a economic analysis by REMI (Nystrom and Zaidi 2013a, Breslow *et al* 2014), estimated emissions of other air pollutants using data from the U.S. EPA NEI (U.S. Environmental Protection Agency 2017), and used a series of existing methods to estimate the co-benefits from the carbon fee and put them in monetary terms (Levy *et al* 2016, Penn *et al* 2017, Dockins *et al* 2004, U.S. Environmental Protection Agency (EPA) 2015, Driscoll *et al* 2015).

We found that the cumulative health co-benefits of a proposed carbon fee in Massachusetts from 2017 through 2040 are as follows:

- 340 lives saved
- 26 respiratory hospitalizations avoided
- 28 cardiovascular hospitalizations avoided
- 20 heart attacks avoided
- $2.9 billion ($2017 USD) of cumulative health benefits between 2017 and 2040, worth $2.0 billion ($2017 USD) if discounted to 2017 at 3% per year

In addition to these benefits, additional health benefits from reductions in air pollution were not quantified here, such as asthma attacks (Jacquemin *et al* 2015, Brauer *et al* 2002, Anderson *et al* 2013), lost days of school and work (Lei Chen, Brian L. Jennison, Wei Ya 2000, Gilliland *et al* 2001), premature birth and low birth weight (Kloog *et al* 2012, Darrow *et al* 2011, Li *et al* 2016), autism spectrum disorder (Talbott *et al* 2015) and Alzheimer's disease (Jung *et al* 2015, Cacciottolo *et al* 2016), along with benefits to crop productivity, farming, forestry, and reductions in acid rain (Committee on Health, Environmental 2010, Chestnut and Mills 2005, Wittig *et al* 2007, Joseph E. Aldy *et al* 1999).

The health co-benefits of the carbon fee increase along with increases in the proposed carbon fee and peak in 2035. There is a lag between fee increase and health co-benefits due to the time required for turnover in the vehicle and building HVAC fleet. Additionally, health co-benefits peak in 2035 since the fee is not tied to inflation. These findings indicate a relationship between the carbon fee amount and the magnitude of health co-benefits – as the amount of the fee increases, so too do the health co-benefits.

The health benefits of the carbon fee would largely be driven by reductions in emissions from transportation and from buildings, and people nearer these sources would experience much of the health benefits. The health damage function model used in this study examined the state as a whole, using 2005 population and underlying disease rates, and it was beyond the scope of this study to evaluate the geographical distribution of these benefits.

In Massachusetts, transportation fossil fuel use is the largest source of GHG emissions, followed by fossil fuel use in buildings (Massachusetts Executive Office of Energy and Environmental Affairs 2017b). These sources are also major sources of other air pollutants, which have a substantial burden on public health (U.S. Environmental Protection Agency 2017, Fann *et al* 2012a, 2013). The proposed bills, Massachusetts S.1821 (Barrett 2017) and Massachusetts H.1726 (Benson 2017), are both expected to reduce GHG emissions in the state and make substantial progress towards Massachusetts' goals under the Global Warming Solutions Act (Massachusetts Executive Office of Energy and Environmental Affairs 2017a). The results of this study underscore that a carbon fee will also substantially lessen health burdens of air pollution within Massachusetts and throughout the region.

Health co-benefits from air quality improvements begin shortly after emissions reductions, increase with fee increases, and occur roughly where emissions reductions occur, which can make them highly relevant for decision-making (Driscoll *et al* 2015, Buonocore *et al* 2015, 2016b, Siler-Evans *et al* 2013, West *et al* 2013, Nemet *et al* 2010, Bain *et al* 2015, Petrovic *et al* 2014). Health co-benefits can be a critical factor in decisions around climate mitigation (Petrovic *et al* 2014, Bain *et al* 2015), have been an important consideration for national-level climate policy (Buonocore *et al* 2016a, Driscoll *et al* 2015), renewable energy development (Siler-Evans *et al* 2013, Buonocore *et al* 2015, 2016b, Plachinski *et al* 2014) and could be an important consideration in debates around a carbon fee in Massachusetts, and in other efforts to mitigate climate change.

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Methodology:

To calculate the health co-benefits of this policy, we developed a methodology that linked a series of models:

- We started with output from a REMI analysis using the economic model CTAM (Carbon Tax Assessment Model) (Washington State Department of Commerce), calibrated for Massachusetts (Breslow *et al* 2014, Nystrom and Zaidi 2013a). This model provided the reductions in CO_2 emissions and fuel consumption for relevant economic sectors in Massachusetts.
- We linked this to the U.S. Environmental Protection Agency National Emissions Inventory data for Massachusetts (U.S. Environmental Protection Agency 2017) to determine the emissions of SO_2, NO_x, $PM_{2.5}$, and VOCs in 2014 from the relevant sectors and then calculate the expected reductions in these emissions due to the policy.
- We estimated the number of lives saved from this policy using mortality per ton emitted functions for SO_2, NO_x, $PM_{2.5}$, and VOCs, developed specific to Massachusetts (Levy *et al* 2016, Penn *et al* 2017).
- We then estimated the reductions in hospitalizations and heart attacks using the ratio between the reduction in these health outcomes and lives saved, specific to Massachusetts (Buonocore *et al* 2016a, Driscoll *et al* 2015).

The details of the REMI analysis are available elsewhere (Nystrom and Zaidi 2013a, Breslow *et al* 2014), but briefly, this analysis relies on CTAM (Carbon Tax Assessment Model), which is a economic model that uses data from the U.S. Energy Information Administration (EIA) and price elasticities for fuels (how fuel consumption changes with a change in fuel price). In addition to Massachusetts, this model has been used to simulate the effect of carbon emissions policies in Arkansas, Rhode Island, Vermont, Oregon, and Washington (Nystrom and Zaidi 2013a, Nystrom 2015a, 2015b, Nystrom and Suite 2014, Liu and Renfro 2013, Nystrom and Zaidi 2013b, Mori 2011). Forecasts from the National Energy Modeling System (NEMS) run by the EIA provide forecasts of future fuel use. The carbon emissions from each fuel source are calculated using standard emissions factors for fuels, based on EIA consumption data and forecasts from NEMS. The carbon emission fee from the policy is then applied to these carbon emissions to calculate the expected cost of emissions. The price elasticities are then used to calculate the expected reductions in consumption of each fuel, and the consequent reduction in carbon emissions. The model was calibrated for conditions specific to Massachusetts, including GDP growth, number of households, and existing policies.

The model output used here estimates the carbon emissions under a "business as usual" baseline without a carbon fee for individual fuel categories used in the residential, commercial, and industrial building sector, and transportation sector. Using the estimated carbon emissions, and the policy price trajectory, the model then calculates the sector-wide response to this policy in terms of reduced use of each fuel type, the resulting reduction in carbon emissions, and the resulting reduction in the use of these fossil fuels. As this model is built from a series of price elasticities, effects like fuel switching, technological change, and changes in future regulations are not explicitly captured in the model, except as reflected in the underlying NEMS forecasts.

CTAM only provides the reductions in fossil fuel use and carbon emissions. To determine the reductions in $PM_{2.5}$, SO_2, VOCs, and NO_x, we used the "business-as-usual" fossil fuel consumption and the emissions from the U.S. EPA National Emissions Inventory for 2014 to develop emissions factors for the sectors in this model, providing emissions per unit of fossil fuel used. We then used these emissions per unit of fossil fuel consumed values to calculate expected emissions in future years, under our business-as-usual case. Using the best-available information, we matched CTAM output to the NEI as described in Table 2. Since our emissions inventory is based on the year 2014, changes in the emissions profile due to fuel type switches, changes in fuel quality, improvements in control technology, and changes in air quality regulations are not fully captured by this model.

Table 2: Sector and fuel type matching between the Carbon Tax Assessment Model and the U.S. EPA National Emissions Inventory

CTAM	NEI
Residential	**Residential**
Kerosene, Distillate Fuel Oil	Oil
Natural Gas, Liquefied Petroleum Gases	Natural Gas
Commercial	**Commercial**
Distillate Fuel Oil, Residual Fuel Oil, Motor Gasoline	Oil
Natural Gas, Liquefied Petroleum Gases	Natural Gas
Industrial	**Industrial**
Distillate Fuel Oil, Residual Fuel Oil, Other Petroleum, Motor Gasoline	Oil
Natural Gas, Liquefied Petroleum Gases	Natural Gas
Transportation	**Transportation**
Motor Gasoline	On-Road non-Diesel Heavy Duty Vehicles, On-Road non-Diesel Light Duty Vehicles, Non-Road Equipment-Gasoline
Distillate Fuel Oil	On-Road Diesel Heavy Duty Vehicles, On-Road Diesel Light Duty Vehicles, Locomotives, Non-Road Equipment-Diesel
Residual Fuel Oil	Commercial Marine Vessels
Liquefied Petroleum Gases, Pipeline Fuel Natural Gas, Other Petroleum	Non-Road Equipment - Other

With the calculated emissions of $PM_{2.5}$, SO_2, VOCs, and NO_x, we then used previously developed health damage functions (in cases of mortality per 1,000 tons emitted) to calculate the mortality burden of these emissions under business as usual, through 2040 (Levy *et al* 2016, Penn *et al* 2017). The health damage functions are based on emissions from residential combustion of fuels simulated using the Community Multiscale Air Quality Model Direct Decoupled Method (CMAQ-DDM). CMAQ is a complex atmospheric chemistry, fate, and transport model commonly used by the U.S. EPA and others to simulate the air quality impacts of air pollution policies and individual sources, and has been calibrated using observations from ground-based monitors and satellites (U.S. Environmental Protection Agency Office of Air Quality Planning and Standards Health and Environmental Impacts Division 2011, U.S. Environmental Protection Agency 2015, Buonocore *et al* 2014, Aiyyer *et al* 2007, Roy *et al* 2007, Foley *et al* 2010). The results used here were developed using CMAQ-DDM, which allows the calculation of the sensitivity of pollutant concentrations to emissions from a given source or set of sources (Levy *et al* 2016, Penn *et al* 2017, Itahashi *et al* 2012). The sensitivities between air pollution emissions and air quality were then used in tandem with concentration-response functions and data from the U.S. Census and Centers for Disease Control and Prevention to calculate the health impacts of emissions (Centers for Disease Control and Prevention, U.S. Census Bureau, Levy *et al* 2016, Penn *et al* 2016).

The health damage functions were based on 2005 data for population and underlying health status (Penn *et al* 2017, Levy *et al* 2016). We did not adjust for changing population over time, likely underestimating future health benefits given population growth and aging, but allowing us to generate a conservative estimate of benefits based directly on peer-reviewed estimates. Additionally, the sensitivities for emissions from residential combustion were used to calculate emissions from all buildings and for transportation. This is a reasonable proxy since most buildings do not have extremely tall stacks, transportation emissions generally occur near the ground, and emissions from residential combustion and from transportation both tend to cluster in population centers.

To calculate the burden of respiratory hospitalizations, cardiovascular hospitalizations, and heart attacks from these emissions, we used ratios between these events and the number of deaths from a previous analysis of the health benefits of carbon emissions standards in 2020, and applied these ratios to our mortality estimates (Driscoll *et al* 2015). This ratio is based on an analysis of the electricity sector for a different year, but since the air pollutants are fairly regionally dispersed and the same populations are exposed, it can still represent a reasonable proxy method to calculate these other health impacts.

This method provides a baseline scenario, with "business as usual" estimates of fossil fuels consumed, carbon emissions, emissions of SO_2, NO_x, VOCs, and $PM_{2.5}$, and the health burden in terms of mortality cases, respiratory hospitalizations, cardiovascular hospitalizations, and heart attacks due to fossil fuel consumption in Massachusetts from 2017 to 2040. We then used the expected reductions in fossil fuel use from CTAM to calculate the expected reductions in the other air pollutants that would occur under the carbon price, along with the expected health benefits (Nystrom and Zaidi 2013a).

To calculate monetary estimates of the health benefits of the carbon fee, we use a standard method called the value of statistical life (VSL) (Dockins *et al* 2004). This metric is commonly used in policy research and by regulatory agencies to place a value on reductions in the risk of mortality and generally captures most of the value of health benefits (Dockins *et al* 2004, U.S. Environmental Protection Agency Office of Air Quality Planning and Standards Health and Environmental Impacts Division 2011, U.S. Environmental Protection Agency 2015, Thompson *et al* 2012, Saari *et al* 2017, Garcia-Menendez *et al* 2015, Saari *et al* 2015, Buonocore *et al* 2016a, 2016b, 2015). Here, we use the standard "cessation lag" structure to account for the delay between exposure to $PM_{2.5}$ and mortality and adjust the values for income in 2017 (U.S. Environmental Protection Agency Office of Air Quality Planning and Standards Health and Environmental Impacts Division 2011, U.S. Environmental Protection Agency 2015). This results in a VSL of $8.5 million (2017 USD) for mortality from $PM_{2.5}$ exposure, accounting for the cessation lag between $PM_{2.5}$ exposure and mortality, and a VSL of $9.4 million (2017 USD) for mortality due to ozone, since there is no substantial delay between exposure to ozone and mortality given evidence derived from time-series studies (U.S. Environmental Protection Agency Office of Air Quality Planning and Standards Health and Environmental Impacts Division 2011, U.S. Environmental Protection Agency 2015). We then calculate the stream of future health benefits in both undiscounted and discounted terms, using a 3% discount rate per year.