Spring 1-1-2016

The Benefits of Internalizing Air Quality and Greenhouse Gas Externalities in the Us Energy System

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B.A., University of California Berkeley, 2010
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A thesis submitted to the

Faculty of the Graduate School of the

University of Colorado in partial fulfillment

of the requirement for the degree of

Doctor of Philosophy

Environmental Engineering Program

2016

This thesis entitled:

The Benefits of Internalizing Air Quality and Greenhouse Gas Externalities in the US Energy System

written by Kristen E Brown

has been approved for the Environmental Engineering Program

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Jana Milford

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Michael Hannigan

*Date*_______________

The final copy of this thesis has been examined by the signatories, and we find that both the content and the form meet acceptable presentation standards of scholarly work in the above mentioned discipline.

Brown, Kristen E. (Ph.D., Environmental Engineering)

The Benefits of Internalizing Air Quality and Greenhouse Gas Externalities in the US Energy System

Thesis directed by Professors Daven Henze and Jana Milford

The emission of pollutants from energy use has effects on both local air quality and the global climate, but the price of energy does not reflect these externalities. This study aims to analyze the effect that internalizing these externalities in the cost of energy would have on the US energy system, emissions, and human health. In this study, we model different policy scenarios in which fees are added to emissions related to generation and use of energy. The fees are based on values of damages estimated in the literature and are applied to upstream and combustion emissions related to electricity generation, industrial energy use, transportation energy use, residential energy use, and commercial energy use. The energy sources and emissions are modeled through 2055 in five-year time steps. The emissions in 2045 are incorporated into a continental-scale atmospheric chemistry and transport model, CMAQ, to determine the change in air quality due to different emissions reduction scenarios. A benefit analysis tool, BenMAP, is used with the air quality results to determine the monetary benefit of emissions reductions related to the improved air quality. We apply fees to emissions associated with health impacts, climate change, and a combination of both.

We find that the fees we consider lead to reductions in targeted emissions as well as co-reducing non-targeted emissions. For fees on the electric sector alone, health impacting pollutant (HIP) emissions reductions are achieved mainly through control devices while Greenhouse Gas (GHG) fees are addressed through changes in generation technologies.

When sector specific fees are added, reductions come mainly from the industrial and electricity generation sectors, and are achieved through a mix of energy efficiency, increased use of renewables, and control devices. Air quality is improved in almost all areas of the country with fees, including when only GHG fees are applied. Air quality tends to improve more in regions with larger emissions reductions, especially for $PM_{2.5}$.

Acknowledgements

The work presented in this paper was funded through the NASA Applied Sciences Program grant NNX11AI54G, a University of Colorado seed grant from the Renewable and Sustainable Energy Institute and the NSF GK-12 program at the University of Colorado. I would like to thank the people without whom this work would not be possible. My advisors, Jana Milford and Daven Henze, were instrumental in the direction and implementation of this research. I would also like to thank Michael Hannigan, Shelly Miller, and Greg Frost for their insight and effort as part of my defense committee. I thank Matt Turner and Shannon Capps for their assistance with the CMAQ model as well as Dan Loughlin, Jeff McLeod, and Azadeh Keshavarzmohammadian for assistance in using and modifying the MARKAL model. Finally, I would like to thank my parents for their support throughout my academic career.

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Introduction

1.1 Motivation

Energy use in the US is influenced by many factors, but not all consequences are considered when choosing electricity sources. The emission of pollutants has effects on both local air quality and global climate. Externalities are activities that affect the well-being of an unrelated group or individual outside the market mechanism. Damages are the monetary value of externalities, such as the value of medical bills from adverse health effects. For example, the health related damages from electricity generation in the US in 2005 have been estimated at $62 billion from coal plants and $740 million from natural gas plants (National Research Council Committee on Health, Environmental, and Other External Costs and Benefits of Energy Production and Consumption, 2010). Greenhouse gas (GHG) related damages from electricity generation in 2005 were $118 billion if calculated with the social cost of carbon (Interagency Working Group on Social Cost of Carbon, 2013) for the year 2010 with a 2.5% discount rate. This study aims to determine how incorporating these damages into energy costs would impact energy use, emissions, and health in the US.

Using fees to incorporate damages into the cost of energy encourages practices that reduce externalities (Pigou, 1962). These are sometimes referred to as Pigouvian taxes. According to economic theory, the most efficient policies to address externalities are directed at the externality such as a fee on emissions rather than a fee on electricity. If fees are applied to electricity then there is an incentive to use less electricity, but there is no financial benefit to change the way the electricity is produced, even if that would further reduce emissions. Emissions fees are likely to lower emissions rates, which reduces the negative effects on air quality and climate and in turn on human health and welfare. Emissions can be reduced due to a switch in the fuels used or application of emission control technologies. By considering fees based on damages instead of an emission or technology goal, even if fees cause an increase in the price of electricity, the overall societal cost related to electricity will decrease because externality costs are lowered.

1.2 Objectives

In this thesis, we evaluate how incorporating externality costs into the cost of energy would change energy use, emissions, air quality, and health in the US. A range of damage estimates from the literature is used to construct policy scenarios, all of which prescribe a set of damage-based emission fees. The fee scenarios consider damages associated with GHGs and HIPs across a range of estimates. The EPA US 9 region MARKAL (MARKet ALlocation) model is used to evaluate changes to the US energy system through the year 2055 under these policies compared to a case with no policies. With MARKAL, we can determine how much emissions change, what level of co-reductions we can expect, and what type of technologies are used to reduce emissions. We also compare MARKAL results for cases with fees applied only to electricity and cases where fees are applied to all energy-use sectors. The case with all sectors considered should have more options to reduce emissions. The CMAQ (Community Multiscale Air Quality) model is used to evaluate the air quality implications of emissions changes projected for 2045 for a subset of the fees. With CMAQ, we can determine what degree of air quality improvement can be expected from the emissions reductions and whether air quality is degraded anywhere despite emissions reductions. We can also examine the distribution of air quality improvements. BenMAP-CE (the Environmental Benefits Mapping and Analysis Program-Community Edition) is then used to determine the change in human health that would occur due to the emissions reductions and the economic value associated with that change. Using BenMAP, we can determine whether the benefits outweigh the costs of policy and compare benefits between fee cases. We also compare the benefits against emissions scaled by the fees to see how closely this simple method matches the more complex calculation.

1.3 Background

In addition to air quality and climate benefits, policies based on fee structures may also be able to provide additional economic benefits. The US Congressional Budget Office (CBO) evaluated a carbon tax (Congressional Budget Office, 2013b) and concluded that a tax of $25/metric ton of CO_2-e increasing at 2% per year would increase federal revenue by $1.06 trillion during the first decade. Another report (Congressional Budget Office, 2013a) says the revenues from such a tax could be used to reduce deficit or offset other taxes, which would both reduce total cost to the economy of the fee. The report states that using the revenue to provide relief to those burdened disproportionately would not reduce the total economic costs, but could be a use for at least a portion of the revenue.

There are frequently air quality co-benefits associated with meeting climate goals. Chen et al. (2013) found 199 million year 2009 USD of total social benefits in the U.S., not including the cost of premature mortality, associated with industrial energy conservation designed to reduce GHG emissions. Zapata et al. (2013) modeled the air quality resulting from successful compliance with California's AB32 GHG reduction policy. They found a 6.2% reduction in annual $PM_{2.5}$ related mortality in 2030, although some parts of California saw an increase in concentrations of $PM_{2.5}$ due to a shift in location of combustion created by using renewable fuels. Kleeman et al. (2013) considered the effects of transportation related GHG reduction in California on $PM_{2.5}$. They found that population weighted $PM_{2.5}$ concentrations in California would be reduced by up to 9.5% by shifting transportation to a system with lower GHG emissions. Thompson et al. (2014) use the United States Regional Energy Policy (USREP) model to determine the costs and approximate emissions changes associated with three different climate policies: a clean energy standard, a cap and trade regulation, and a transportation fuel standard. They then use CMAQ with scaled emissions to determine ambient concentrations of O_3 and $PM_{2.5}$ and calculate monetized health co-benefits with BenMAP. They find that more flexible policies are less costly, and that the air quality co-benefits may be larger than the costs of the policy, which will have further climate benefits. Saari et al. (2015) also find that health co-benefits achieved as a result of GHG emission reductions may fully offset the costs of reducing GHG emissions, although the distribution of the health benefits is not uniform across the US.

In this thesis, we evaluate how incorporating externality costs into the cost of energy would change energy use, emissions, and air quality in the United States. A range of damage estimates from the literature is used to construct policy scenarios, all of which prescribe a set of damage-based emission fees. The fee scenarios consider damages associated with GHGs and HIPs across a range of estimates. The EPA US 9 region MARKAL model is used to evaluate changes to the US energy system through the year 2055 under these policies compared to a case with no policies. The CMAQ model is used to evaluate the air quality implications of emissions changes projected for 2045 for a subset of the fees. BenMAP is then used to determine the change in human health that would occur due to the emissions reductions and the economic value associated with that change. The modeled policies apply fees uniformly, without regard to the location of the emissions, but the air quality results from CMAQ and health effects from BenMAP-CE are examined to determine if the policy benefits some areas more than others.

Fees are not the only potential source of energy system changes; emissions also change based on changes in the fuel supply. Although natural gas is touted as a bridge fuel (Moniz et al., 2011) to a low carbon future, climate policy is needed to reduce radiative forcing with increased natural gas use. McJeon et al. (2014) studied the effect of increased supply and decreased price of natural gas, intended to represent a global increase in hydraulic fracturing, and found that this would not lead to a reduction in CO_2 emissions. In the absence of additional climate policy, the increased use of natural gas would reduce CO_2 emissions from coal, but also lead to lower use of renewable and conservation technologies, resulting in similar CO_2 emissions to the baseline projection. When the increase in fuguitive methane emissions is considered, abundant, cheap natural gas causes an increase in radiative forcing, although this may be small depending on the rate of fugitive methane emissions, which varies widely in the literature. If climate policies are added to the cheap gas scenario, reductions in emissions are possible. Shearer et al. (2014) analyzed the effect of abundant natural gas for the US specifically. They also find that without additional policy, the increased supply of natural gas does not reduce GHG emissions, even when methane leakage rates are assumed to be low. More natural gas slows the implementation of renewables without additional policy specifically targeted at renewables (not just a carbon tax or cap). Also, the low price of natural gas leads to lower energy prices, which leads to increased fuel use. McLeod et al. (2014) evaluated the effect of natural gas price and supply and found that GHG emissions were not affected by these assumptions at the national level, but there are differences at a regional level. They analyzed the Rocky Mountain region, which has ample renewable resources so there are changes in the GHG emissions within that region for different assumptions of future natural gas cost.

There are some types of benefits we do not consider in this dissertation. West and Fiore (2005) found that global exposure to O_3 is reduced when methane emissions are reduced, leading to health benefits. While we examine the health co-benefits of greenhouse gas fees due to co-reduced emissions, the spatial scale for our analysis is not designed to consider the effect of methane on O_3 concentrations. We also do not consider the climate impacts of other so-called short-lived climate forcers, such as black carbon and sulfate aerosol. Anenberg et al. (2012) analyzed the effect of emissions controls designed to reduce short lived climate forcers and found that black carbon reductions can lead to significant health benefits. Shindell (2015) calculated damages in a slightly different way to that considered here by defining the "social cost of atmospheric release." His global-scale evaluation included both climate and air quality effects for various pollutants, including climate impacts of aerosols for which we consider only the health impacts in this dissertation. Shindell found that the air quality related health effects dominate for aerosol related species, which indicates that our method is capturing much of the damages. The health and climate damages for a pollutant are not always in the same direction, so it is possible that when fees reduce emissions of SO_2, there is a climate disbenefit that is not being accounted for. While we consider health effects of O_3 and $PM_{2.5}$, we do not consider benefits from lowering emissions of toxic pollutants such as mercury and benzene. These pollutants can cause negative health impacts including birth defects and cancer. In addition, there are externalities associated with exposure pathways other than air. Runoff of waste products from resource extraction can impact water quality, leading to contaminated drinking water or waterways that are unsafe for fish and wildlife. There are also additional ecosystem impacts both for pollutant categories that we consider and those that are outside the scope of this study. Some of these include ozone damage to plants, reduced visibility, and altered chemistry of waterways.

1.4 Damages

Several studies have investigated the source and values of externalities associated with electricity generation. Burtraw et al. (2012) and the Interagency Working Group on the Social Cost of Carbon (2010) both studied the external cost of CO_2 emissions. The European ExternE (European Commission, 1995) project calculated external costs of energy to the environment. The Stern (2007) review came to the conclusion that the benefits of action to avoid climate change and the related externalities outweigh the costs. Levy et al. (2009) modeled the damages from coal fired power plants in the US, and Muller et al. (2011) estimated US air pollution damages from all sectors. Several of these studies suggest that the damages they report could be applied as fees on emissions to internalize externalities.

1.4.1 Health Impacting Pollutant Damages

First, we will discuss health impacting pollutant (HIP) damages. We consider NO_x, $PM_{2.5}$, PM_{10}, SO_2 and VOC as HIPs. For health impacts, the damage values are location dependent because emissions that are located near population centers will often affect more people than emissions located in rural areas (e.g., Fann et al. 2009). Although damages are not varied by location in this study, some of the location dependence is captured by using different damage values for different sectors, for instance industrial emissions tend to be closer to population centers than electricity generation emissions, therefore industrial damages tend to be higher than electric sector damages per ton of emission. The majority of the damages from HIPs are due to premature mortality associated with chronic exposure to $PM_{2.5}$ (Pope et al., 2002). Other health effects include chronic bronchitis (Abbey et al., 1993), and respiratory and cardiovascular hospitalization (Burnett et al., 1999).

There are a few reasons for discrepancies in published damage estimates. We therefore surveyed several HIP damage studies in the literature and consider three sets of fees (low, mid, high). Whether or not age is taken into account when applying the Value of Statistical Life (VSL) to mortalities has a large effect on the damage value. Using a uniform VSL can produce 50% higher marginal damages than differentiating by age (Muller et al., 2011). Muller et al. (2007; 2011) chose to differentiate VSL based on age because the estimates of VSL are based on a working age population but mortalities from air pollution affect the elderly more often. There is no consensus regarding how age, life expectancy and other factors should be applied when differentiating the VSL for mortality (Levy et al., 2009) so the other studies do not take age into account.

Another factor that can cause differences in marginal damage estimates is what sources of pollution are considered. The studies by Muller et al. (2007; 2011) and Fann et al. (2009; 2012) consider many sources of emissions while NRC2010 focuses on EGUs combusting coal and natural gas. Muller et al. (2011) distinguish the sources of emissions by stack height rather than source type. Variations in population exposure also contribute to differences among damage estimates. In urban areas, a ton of pollutant would affect more people and have higher damages than in a rural area. Muller and Mendelsohn (2007) consider urban and rural damages separately, while Fann et al. (2009) only consider urban effects. Fann et al. (2012) also consider the benefit achieved by a reduction of pollution, rather than the damages caused by an additional quantity of pollution.

Since mortality from $PM_{2.5}$ is a significant component of the damage estimates, the choice of the corresponding concentration-response functions is also important. The National Research Council (2010) and Muller et al. (2011) used Pope et al. (2002) and Woodruff et al. (2006) to relate $PM_{2.5}$ exposure to mortality; Fann et al. (2012) used Krewski et al. (2009).

1.4.2 Greenhouse Gas Damages

Global climate change is caused in large part by energy related emissions of CO_2, methane, and other GHGs (Ciais et al., 2013). These emissions can lead to global changes in temperature, weather patterns, and water systems (Kirtman et al., 2013). Although there is high uncertainty about the speed and degree with which changes might take place, the changes are still a serious concern. Severe weather changes are predicted and some are already observed, including decreases of snow and ice cover (Vaughan et al., 2013), changes in precipitation patterns (Boucher et al., 2013), rises in sea level (Rhein et al., 2013), and shifts in ecosystem structure such as altered food webs or migration of animals (Ciais et al., 2013). There are also effects to human health and welfare, which are not uniform throughout the world (Kirtman et al., 2013). Due to some of the ecological changes mentioned earlier there will be a shift in water availability that can cause crop loss and disease, especially in regions already stressed for water (Dasgupta et al., 2014). The combination of temperature and water availability changes will have profound effects on agriculture. While yields will increase in some places, some regions that currently depend on agriculture may find the land unable to continue to support their crops (Porter et al., 2014). These effects are important externalities for energy use.

Given the broad range of impacts of climate change, occurring on vastly different temporal and spatial scales, and uncertainties associated with estimating these impacts, there are large discrepancies in the estimated value of damages from climate change. The National Research Council (2010) found that almost all variation in marginal damage estimates derives from differences in assumptions of discount rate and the magnitude of damages from climate change, especially whether unlikely but catastrophic effects were considered. The greenhouse gas damages used in this study are taken from the Social Cost of Carbon (SCC) used in regulatory impact analysis by the US government (Interagency Working Group on Social Cost of Carbon, 2013). The reason for evaluating these cost estimates is so that various government agencies can evaluate the cost or benefit of regulations they are considering that might alter CO_2 emissions. The SCC is designed to be used for “regulatory actions that have ‘marginal’ impacts on cumulative global emissions.” (Interagency Working Group on Social Cost of Carbon, 2010) The first set of estimates was released in 2010 and a revised set of estimates was released in 2013. Both estimates were created using the same methodology, but the updated estimates use more recent versions of the integrated assessment models. The SCC documentation recommends considering the full range of values they report because there are many uncertainties involved. We have considered the four sets of damage values from the 2013 SCC in fee scenarios. The fees were applied to CO_2 and CH_4. The fees were adjusted for CH_4 using the 100 year global warming potential (GWP) of 28 (Myhre et al., 2013).

1.5 Methods and Organization

Three models are used to evaluate the effects of several possible fee scenarios, as depicted in Figure 1.1. The MARKAL model is used first to evaluate the effects of fees on emissions from energy use. The emission results from MARKAL are then used to determine future emissions inputs to CMAQ. CMAQ calculates the changes in air quality resulting from the reduced emissions. BenMAP takes the air quality changes as inputs to determine the change in health associated with the policies and the monetary value associated with those changes.

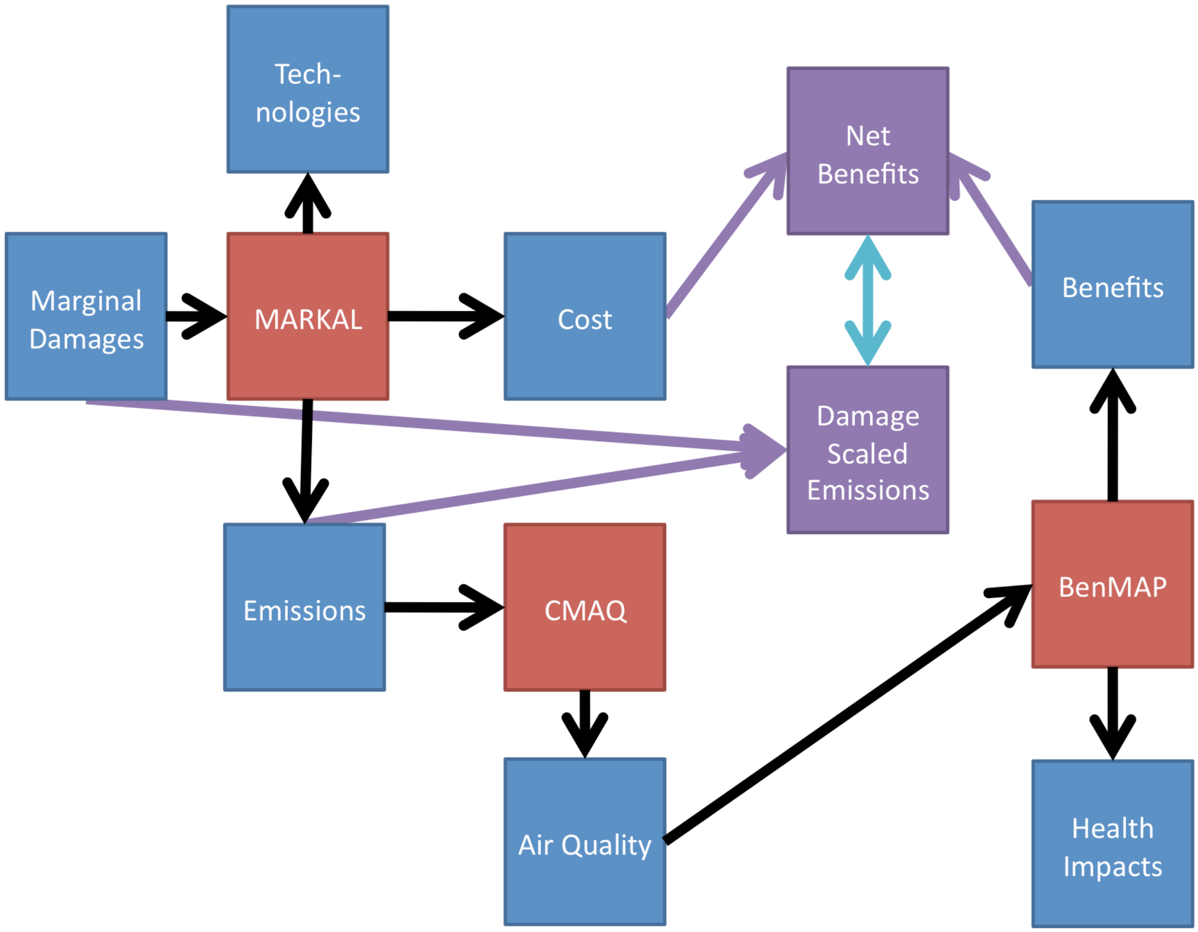

Figure 1.1 The relationship between the models (red), their inputs and outputs (blue), and calculations performed with the results (purple).

1.5.1 The MARKAL Model

The first portion of this study attempts to determine how implementing damage-based emissions fees would change how energy is used in the US in the coming decades. This is done using the MARKAL model, which is an energy system model that is used here to compare energy scenarios. MARKAL is chosen because of its thorough description of the US energy system and the ability to model a variety of fees (Loulou et al., 2004). MARKAL considers the economic advantages of different technologies and allows users to incorporate external damages to be considered as costs. Chapter 2 of this thesis presents the energy and emissions changes when damages are internalized only in the electric power sector. Chapter 3 uses a more recent database with the MARKAL model to discuss the energy and emissions resulting from fees applied to all energy use sectors in MARKAL. Once the effect of the modeled policies on emissions is determined using MARKAL, the effect of these policies on ambient air quality are modeled using CMAQ. Emissions information by itself, does not give policy makers a complete view of the effects of policy, but a chemical transport model can translate predicted emissions into air quality.

1.5.2 The CMAQ Model

CMAQ is a three-dimensional Eulerian chemical transport model (Byun and Schere, 2006) that simulates the chemical and physical processes that alter and transport pollutants through the air. An Eulerian model uses a grid that remains fixed in space, as opposed to a Lagrangian model that follows an air parcel or plume. The Eulerian framework is best suited for evaluating air quality across a wide range of different locations simultaneously. As a chemical transport model, meteorology is pre-calculated and used as an input to CMAQ, thus enabling simulation of multiple pollutants and their complex chemical interactions at once. In Chapter 4, the details of the CMAQ simulations for $PM_{2.5}$ and O_3 are further described, and model estimates of the distributions of these species are compared to in situ measurements.

1.5.3 The BenMAP-CE Model

BenMAP-CE was created by the US EPA to estimate health impacts and economic benefits related to changes in air quality (US EPA, 2013a). The model includes a variety of health impact functions that can be chosen by the user. These functions relate changes in air pollutant concentrations with changes in health. This calculation considers the exposed population, the baseline incidence rate of the effect, the concentration of the pollutants considered, and the concentration-response function chosen to relate the pollutants to the health effects. BenMAP-CE considers health effects associated with ozone and $PM_{2.5}$. The pollutant concentrations from CMAQ will be used as an input to the BenMAP-CE model. After determining the air quality under various policies using CMAQ, we can determine the monetary value of the benefits of the emissions reduction using BenMAP-CE. In this dissertation it will be used to determine the benefit of implementing damage based fees in the US. Chapter 4 discusses the air quality and health impacts associated with the changes in Chapter 3, and Chapter 5 presents the overall conclusions from this work as well as a discussion of uncertainty and possible topics for future research.

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Chapter 2

Accounting for Climate and Air Quality Damages in Future US Electricity Generation Scenarios

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2.0 Abstract

This study analyzes how assessing fees based on damages caused by air pollution might transform US electricity generation. Damages from life cycle emissions are internalized in the EPA-MARKAL model through fees per ton of SO_2, nitrogen oxides (NO_x), particulate matter, and greenhouse gases (GHG) from 2015 through 2055. Electricity production, fuel type, emissions controls, and emissions produced under various fees are examined. A shift in fuels results from \$30/ton CO_2-equivalent GHG fees or from criteria pollutant fees that correspond to the higher end of the range of published damage estimates, but not from criteria pollutant fees based on low or mid-range damage estimates. With mid-range criteria pollutant fees assessed, SO_2 and NO_x emissions are lower than the business as usual case (52% and 10%, respectively), with larger differences in the western US than the eastern US. GHG emissions are not significantly impacted by mid-range criteria pollutant fees alone; conversely, with only GHG fees, NO_x emissions are reduced by up to 11%, yet SO_2 emissions are slightly higher than in the business as usual

case. Therefore, co-reductions of GHG and criteria pollutant emissions for the electricity sector can be enhanced by deliberate policy design.

2.1 Introduction

Electricity production in the US is influenced by many factors, but not all consequences of electricity generation are considered when planning capacity expansions. The emission of pollutants from electricity generating plants affects both local air quality and global climate. For example the Natural Research Council Committee on Health, Environmental, and Other External Costs and Benefits of Energy Production and Consumption estimated damages (the monetary value of adverse effects of pollution) associated with criteria pollutants (NO_x, SO_2, and particulate matter (PM)) from electricity generation in the US in 2005 at $62 billion from coal plants and $740 million from natural gas plants (National Research Council Committee on Health, Environmental, and Other External Costs and Benefits of Energy Production and Consumption, 2010). That damages owing to climate and air quality are linked to common sources can potentially be exploited; policies targeting greenhouse gas mitigation may lead to reduction in levels of conventional air pollutants (Rive, 2010; Burtraw et al., 2003). Here we examine how incorporating damages associated with air quality and climate into the cost of electricity would impact the way in which electricity is generated in the US and the amount of associated emissions produced.

Several studies have investigated the source and values of externalities associated with electricity generation. Burtraw et al. (2012) and the Interagency Working Group on the Social Cost of Carbon (2010) both studied the external cost of CO_2 emissions. The European ExternE (European Commission, 1995) project calculated external costs of energy to the environment. The Stern (2007) review came to the conclusion that the benefits of action to avoid climate change and the related externalities outweigh the costs. Levy et al. (2009) modeled the damages from coal fired power plants in the US and Muller et al. (2011) estimated US air pollution damages from all sectors. Several of these studies suggest that the damages they report could be applied as fees on emissions to internalize externalities.

Incorporating damages into the cost of electricity is expected to encourage practices that reduce externalities. Markets with associated externalities are not efficient unless the damage costs are internalized or the externalities are otherwise considered during decision making (Longo et al., 2008). According to economic theory, the most efficient policies to internalize damages are directed at the externality, such as a fee on emissions rather than a fee on electricity. Emissions fees are expected to lower emissions rates, reducing negative effects on air quality and in turn on human health and welfare. By considering policies based on damages instead of an emission or technology goal, even if fees cause an increase in the price of electricity, the overall social welfare cost related to electricity will decrease because externality costs are lowered.

Previous studies have assessed the potential impacts of incorporating externalities into energy systems outside the US. The MARKAL (MARKet ALlocation) model is an economic-optimization model that finds the least cost way to satisfy specified energy demands over multi-year time horizons (Loulou et al., 2004). Nguyen (2008) used MARKAL to examine the effect of internalizing damages in Vietnam and found that although the cost of electricity production increased by 2.6¢/kWh, external costs of 4.4¢/kWh were avoided so the true cost of electricity decreased. Rafaj and Kypreos (2007) used MARKAL to examine the effects of including externality costs in the price of electricity on the global electricity mix. In their study, electricity demand was reduced, a different mix of generating technologies was used, and emissions control technologies were installed. Rafaj and Kypreos (2007) found that when internalizing externalities for global regions, the increase in electricity price is larger for regions relying on coal-based technology. Klaassen and Riahi (2007) performed a similar study with MESSAGE-MACRO, internalizing criteria pollutant externalities from the global energy system. They saw reduced electricity demand and use of fossil fuels and increased production from renewable sources and concluded that internalizing criteria pollutant externalities can also reduce GHG emissions.

While previous studies have examined this approach for other countries or the global electricity system, to our knowledge, the impact of internalizing estimated damages on the suite of fuel types and technologies used to generate electricity, and their associated emissions, has not been studied in detail for the US electricity system. Here we use the MARKAL energy systems model to investigate scenarios in which damages from criteria pollutants and greenhouse gases (GHGs) are accounted for in long-term electricity generation decisions by applying emissions fees equal to the estimated damages. Significant externalities occur due to direct fuel combustion by power plants; we extend the MARKAL model to also account for those occurring during upstream stages of the electricity production life cycle including resource extraction, equipment manufacturing, and transportation. The analysis thus builds on previous literature that determined the value of marginal damages of electricity generation. The study considers the interplay of GHG and criteria pollutant fees and examines differences in responses across US regions with different pre-existing emissions control requirements.

2.2 Methodology

2.2.1 MARKAL model

The MARKAL energy systems model is used here to compare future electricity generation scenarios. MARKAL is used with the US Environmental Protection Agency's (EPA) 9-region database that describes the US energy system, including electricity, transportation, residential, commercial, and industrial sectors. The model determines the least cost way to satisfy future end-use demand for electricity within specified constraints (Loulou et al., 2004). Electricity generation and conservation technologies are included, so demand can be met by using more efficient end-use devices or by generating more electricity.

2.2.2 EPA database

The EPA US 9 region database (EPAUS9r_2010_v1.3)(U.S. EPA Office of Research and Development, 2010) is used as a basis for all scenarios considered. The database represents the US energy system for the years 2005-2055 in 5 year increments. While other sectors are also represented in the database, this study focuses on the electricity system. The database includes the projected demand for electricity in each of the nine census regions in the US as well as the currently installed generation capacity, by type, in each of these regions (Shay et al., 2008). Resources and energy are traded and transported between regions through modeled pipelines, transmission lines, and import and export parameters.

The database includes several traditional and advanced electricity generation technologies that are in use or available to install. Fuel types used include solar, wind, hydro, geothermal, municipal solid waste, biomass, nuclear, oil, natural gas and coal. The new generation technologies available for coal are integrated gasification combined cycle (IGCC) and supercritical steam. Natural gas generation methods include combined cycle, steam, and combustion turbine. All coal and natural gas technologies have the option of applying carbon capture and storage (CCS). The existing coal steam plants have the option of applying controls on NO_x, SO_2 and PM. The SO_2 controls are flue gas desulphurization (FGD), and the optional PM controls are a cyclone, an electrostatic precipitator, or a fabric filter. To control NO_x, the residual coal steam plants can use low NO_x burners either alone or in combination with either selective non-catalytic reduction (SNCR) or selective catalytic reduction (SCR). Residual control technologies on existing generation are also included in the model. FGD, NO_x, and PM controls are standard for natural gas and new coal-fired power plants. Controlled emission rates are from the Inventory of US Greenhouse Gas Emissions and Sinks (U.S. EPA, 2010), eGRID (E. H. Pechan & Associates, Inc., 2010), and the Documentation of EPA Modeling Applications (US EPA, 2003). A fixed lifetime is specified for all methods of electricity generation; hurdle rates for the cost of capital are applied to approximate barriers to investing in new technologies (U.S. EPA Office of Research and Development, 2010) .

Fuel supply curves used in the EPA database are based on the National Energy Modeling System (NEMS) outputs used for the Energy Information Administration's (EIA) Annual Energy Outlook (AEO) 2010 report (U.S. Energy Information Administration, 2010). The cost of electricity production covers steps from fuel extraction to transportation to end use. Costs include capital equipment and financing, operation and maintenance, and fuel costs. The capital cost for solar declines over time, but the capital cost for wind stays roughly consistent throughout the time period. Availability factors for wind (Energy

Information Administration, 2010) and solar (Government of Canada, 2012) are differentiated by region, season, and time of day. The fuel costs of coal and natural gas increase with time. Data on existing power plants including capacity, lifetime, availability, and operating costs come from EIA Forms 860, 767, 759/906, and Form1 (Energy Information Administration). The estimated cost and efficiency for new electricity generating units (EGUs) is obtained from the 2010 AEO (U.S. Energy Information Administration, 2010).

The EPA database also includes predicted electricity demand for all years modeled. MARKAL treats demand by specifying an end use demand for a service such as lighting and then providing a range of technologies to fulfill the demand, for example a selection of light bulbs. The types, cost, and efficiency of the end use technologies are derived from AEO (U.S. Energy Information Administration, 2010). Demand for the commercial sector is derived from the NEMS Commercial Sector Demand module output (Energy Information Administration, 2010). Most of the demand data for the industrial sector come from the Manufacturing Energy Consumption Survey database (Energy Information Administration, 2007). Demand in the residential sector is derived from AEO Table A4: Residential Sector Key Indicators and Consumption (U.S. Energy Information Administration, 2010).

State renewable portfolio standards referenced from the 2010 AEO (U.S. Energy Information Administration, 2010) are aggregated to the nine regions in the database, by weighting each state's requirements by their portion of the region's historic electricity generation. There are no GHG regulations represented in the current database. The database also includes emissions constraints based on existing or pending regulations. The Cross State Air Pollution Rule (CSAPR) is represented by placing an upper bound on SO_2

and NO_x emissions from EGUs in the eastern US. Although the CSAPR has been overturned (*EME Homer City Generation v. EPA*, 2012), the limits in the model should approximate the eventual regulations limiting emissions from EGUs. All emissions are also capped at historical levels from the US EPA Clean Air Markets Division's database (Clean Air Markets Division of US EPA, 2012), and limited to comply with the Clean Air Act Amendments (Shay et al., 2008). Existing coal-fired EGUs are also constrained to use at least one NO_x control method in 2020 and beyond. While emissions are constrained to historical levels, the method of achieving these levels is not specified, so utilization of control technologies in the model does not necessarily match historical use.

Electricity generation emissions in the EPA MARKAL database generally include only direct combustion emissions. For this study we added emissions from other life cycle stages. Transportation of materials, land use changes, disposal of worn out materials, and electricity use in equipment production are among the emissions sources upstream of generation (America's Energy Future Panel on Electricity from Renewable Resources and National Research Council, 2010). GHG, SO_2, NO_x, and PM_{10} upstream emissions are introduced for all electricity generating technologies. See the supporting information for upstream emissions estimates.

2.2.3 Damages

The primary fees applied are based on the marginal damage estimates from Hidden Costs of Energy (National Research Council Committee on Health, Environmental, and Other External Costs and Benefits of Energy Production and Consumption, 2010), hereinafter referred to as NRC2010. These values consider emissions throughout the entire US, with damages specific to EGU emissions. The criteria pollutant damages are estimated in NRC2010 using the Air Pollution Emission Experiments and Policy (APEEP) model (Muller and Mendelsohn, 2006), accounting for the contribution of current EGU emissions to ambient concentrations in each county in the contiguous US using emissions data from EPA's 2002 National Emission Inventory (Office of Air Quality Planning and Standards, 2002). These county level concentrations are multiplied by the population of each county to determine exposure. The damages are calculated based on the US population circa 2000 and are not recalculated for predictions of future population. To estimate health effects, NRC2010 compared these exposure levels to concentration response functions from peer reviewed health studies (Pope et al., 2002; Abbey et al., 1993; Burnett et al., 1999). They then monetized the effects to determine damage estimates. The value of a statistical life (VSL) used is approximately $6 million, which NRC2010 applied uniformly to all lives lost. The value of market goods was determined by market price, and the value of illness was derived from nonmarket valuation literature (Chestnut and Rowe, 1990). To calculate marginal damages, NRC2010 used APEEP to estimate total damages due to all sources in the model. From these baseline emissions, an additional ton of pollution was added from one source at a time and total damages were recomputed; the marginal damage is the difference. This was repeated for each pollutant and each source. The majority of the damages from criteria pollutants are due to human health effects, specifically premature mortality associated with chronic exposure to $PM_{2.5}$ (C. A. Pope et al., 2002). Other health effects include chronic bronchitis (Abbey et al., 1993), and respiratory and cardiovascular hospitalization (Burnett et al., 1999). Environmental externalities include changes in crop and timber yields and reduced visibility (Pye, 1988; Reich, 1987; Muller and Mendelsohn, 2006; Rawlings et al., 1990).

Table 2.1. Marginal damage estimates presented in the literature (values converted from original units to year 2005 USD/metric ton pollutant).

Source	NRC 2010	NRC 2010	Muller and Mendelsohn 2007	Fann et al. 2009
Fuel type/ sector	Coal EGUs	Natural gas EGUs	Average for all sources	Average for EGUs
SO_2	6000	13000	1500	80000
NO_x	1700	2300	370	15000
$PM_{2.5}$	9800	33000	2700	
$PM_{10-2.5}$	470	1800	440	

Table 2.1 compares the NRC2010 damage estimates to other recent values published in the literature. There are a few primary reasons for discrepancies in the damage estimates. Marginal damage estimates may be 50% higher if the VSL is applied uniformly (Muller et al., 2011) as opposed to differentiating based upon age (Muller and Mendelsohn, 2007). Another important factor is which sources are considered. The study by Muller and Mendelsohn (2007) considered all sources of emissions while NRC2010 and Fann et al. (2009) focused solely on EGUs, and NRC2010 further separated coal and natural gas units. Variations in population exposure also contribute to the differences across damage estimates. Muller and Mendelsohn (2007) considered damages in urban and rural areas separately, while Fann et al. (2009) only considered effects in urban areas. Since mortality from $PM_{2.5}$ is a significant component of the damage estimates, the choice of the corresponding concentration-response functions is also important. NRC2010 and Muller and Mendelsohn (2007) used Pope et al. (2002) and Woodruff et al. (2006) to relate $PM_{2.5}$ exposure to mortality; Fann et al. (2009) used Laden et al. (2006).

The fees used for direct combustion and upstream emissions of greenhouse gases are set at \$30/ton CO_2-equivalent (CO_2-e). This value is chosen to represent the central tendency of the damage estimates in the literature (National Research Council Committee on Health, Environmental, and Other External Costs and Benefits of Energy Production and Consumption, 2010; Burtraw et al., 2012). There are large discrepancies in the estimated value of damages from GHG. NRC2010 found that almost all variation derives from differences in assumptions of discount rate and the magnitude of damages from climate change, especially whether unlikely but catastrophic effects were considered. Emissions of CO_2 are assumed to dominate the direct combustion emissions of GHG. For upstream emissions, the GHG fee is also applied to methane emissions from production of natural gas and landfill gas. A factor of 21 is used to convert methane emissions on a mass basis to CO_2-e.

The damage estimates described above are internalized as fees on emissions. These fees are applied in the years 2015 through 2055. All of the fees are applied per metric ton of pollutant emitted and all values are given in year 2005 USD. The NRC2010 values are used as the mid-range fees for criteria pollutants. Distinct fees are applied to emissions from coal and natural gas fired power plants corresponding to the separate damage values provided by NRC2010. For upstream emissions and biomass combustion emissions, the damages per ton are the same for each generation type and are determined by an average of the coal and natural gas combustion damage estimates from NRC2010. In addition to examining the effect of fees based on the mid-range NRC2010 estimates, sensitivity cases use Muller and Mendelsohn's (2007) values as lower estimates and Fann et al.'s (2009) values as higher estimates. In these sensitivity cases, the same fees are applied to both combustion and upstream emissions regardless of generation type. In separate cases, fees are considered on criteria pollutants only, GHG emissions only, and both criteria pollutants and GHGs. In the cases presented here, fees are applied to full life cycle emissions. We also examined the impact of only applying fees to direct combustion emissions, with results shown in the supporting information.

Due to uncertainty in the predicted cost and supply of natural gas, a sensitivity analysis was run to represent the fuel supply curves predicted for natural gas in the 2012 AEO,(U.S. Energy Information Administration, 2012) which correspond to increased supply and lower gas prices than the 2010 AEO forecast used in the EPA MARKAL database. This modification is modeled by adding a subsidy of $1 million/petajoules ($1.09/thousand cubic feet) on natural gas in the MARKAL database for all model years, approximating the price difference between the 2012 and 2010 AEO projections.

2.3 Results

Figure 2.1 shows the electricity generation by fuel type in four cases: business as usual (BAU) with no fees added, life cycle fees on criteria pollutants based on NRC2010 damage estimates, life cycle fees on GHG, and life cycle fees on criteria pollutants and GHG. In the BAU case, the amount of generation from coal remains relatively constant, but as total generation increases the percentage from coal declines from 48% in 2010 to 37% in 2055. Natural gas generation increases significantly, with its contribution increasing from 20% in 2010 to 33% in 2055. With little change in absolute generation from nuclear power, its contribution drops from 18% to 13%. The share from hydropower drops from 6% to 5%. Finally, the absolute amount of generation from solar, geothermal, and wind increases over time, with wind playing the largest role and contributing 7% of generation in 2055.

When mid-range fees are applied to the criteria pollutant life cycle emissions, there are only slight changes to the total electricity use and generation mix. More significant changes result when only GHG fees are applied. There is up to 7% less total electricity generated with GHG fees in place compared to the BAU case, and up to 16% less electricity generated from coal. There is up to 9% more generation from natural gas, 7% more from wind and up to 13% more from hydro. When GHG fees and mid-range criteria pollutant fees are combined, limited additional reductions in electricity generation and coal use occur, with up to 7% less total electricity and 18% less electricity from coal than in the BAU case. With the combination of fees there is yet more expansion of natural gas and wind use, with increases of up to 10 and 11% compared to BAU, respectively.

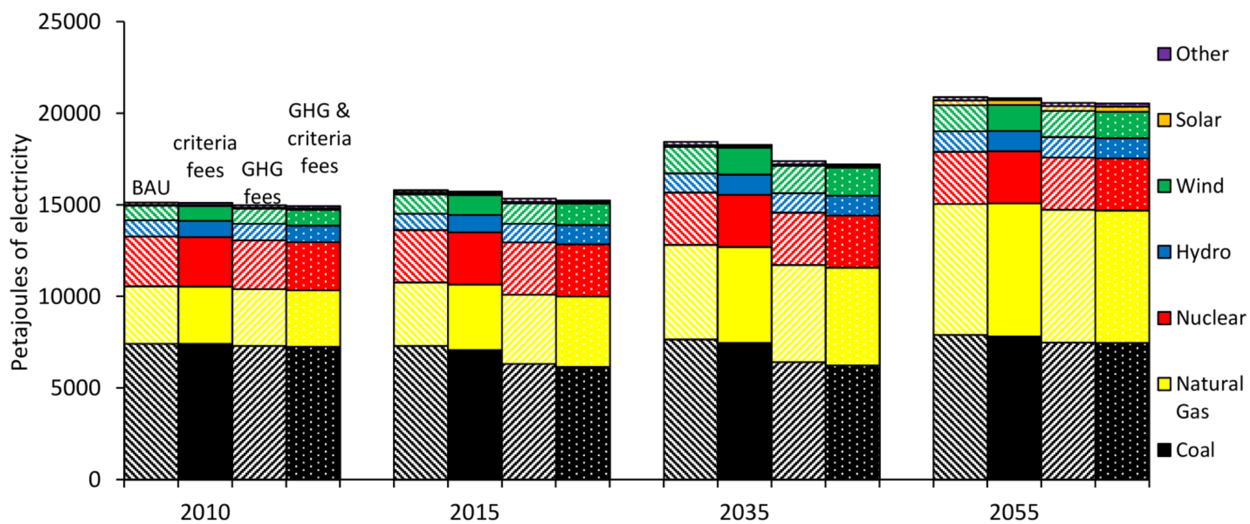

Figure 2.1. Electricity production by fuel type in four cases: BAU, life cycle criteria pollutant fees, life cycle GHG fees, and life cycle criteria pollutant and life cycle GHG fees. “Other” includes oil, biomass, waste, and geothermal generation.

Figure 2.2 shows emissions of SO_2 and NO_x from each of the four main cases. The emissions are divided into those occurring in the eastern and western US, as these two regions have existing emissions regulations that are distinctly different. Figure 2.2 also shows results from application of criteria pollutant fees based on low and high damage estimates presented as "error bars" on the national total emissions and discussed in the next section. PM changes are typically small when fees are applied to criteria pollutants, GHG or both; results for PM are shown in the supporting information.

In the BAU case, emissions control technologies are applied to EGUs due to existing regulations, so that by 2035 total SO_2 and NO_x emissions are respectively 78 and 27% lower than in 2005. In the BAU case, the model applies few new controls on SO_2 until 2015, after which 65-70% of the electricity generation from coal has FGD technologies applied. Most coal plants have both low NO_x burners and SNCR equipment in place in all years. Control technologies selected in the BAU case are expected to differ somewhat from those applied in the real world due to the model's simplified treatment of costs and constraints.

When mid-range fees are applied to the criteria pollutant life cycle emissions, there are only slight changes to the electricity mix, as shown in Figure 2.1, but the changes in emissions are more substantial. The reduction in SO_2 and NO_x emissions is due to an increase in the control technologies applied. As shown in Figure 2.2, nationwide SO_2 emissions are 47-53% lower in the case with mid-range criteria pollutant fees than in the BAU case. With mid-range criteria pollutant fees, 87% of coal plants across the country have FGD controls after 2020 and every plant has at least one NO_x control and typically two. SO_2 emissions in the western region are 70% lower than in the BAU case, while those in the eastern region are 36-42% lower. NO_x emissions are reduced by up to 10% overall, and in the western region are as much as 21% lower than in the BAU case. The regional discrepancy in additional reductions occurs because more controls were installed in the eastern region prior to application of fees.

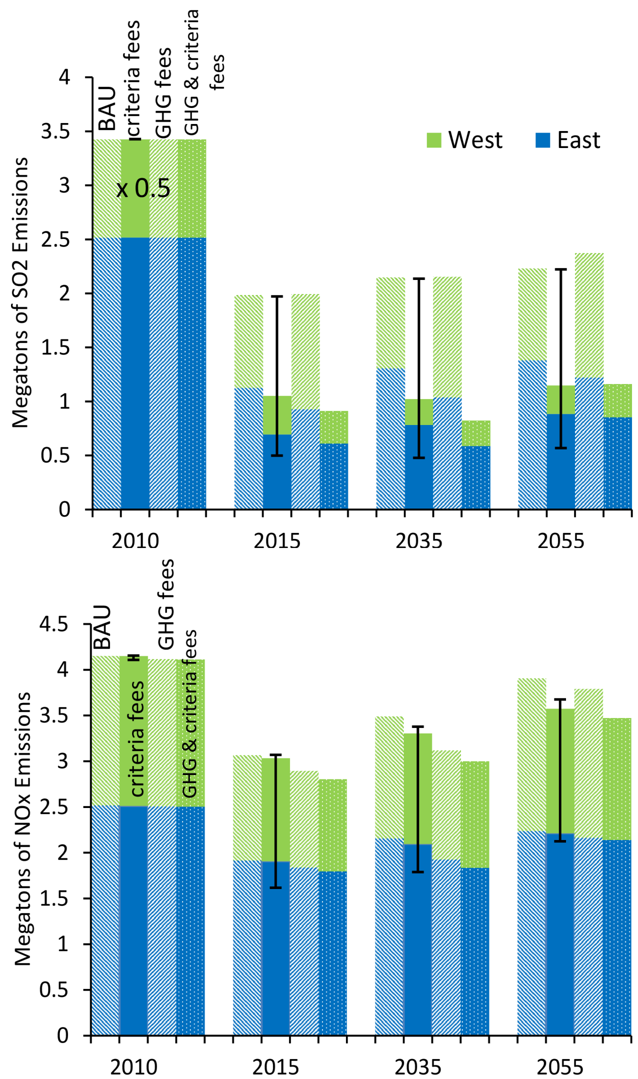

Figure 2.2. Emissions of SO_2 and NO_x from electricity generation in eastern and western regions of the US. The error bars represent national total emissions with high and low sensitivity fees in place.

When fees are applied to GHG emissions alone, without fees on criteria pollutants, the NO_x emissions are lower than in the BAU case for all years due to a reduction in the use of coal to generate electricity. NO_x emissions are as much as 11% less than in the BAU case. As shown in Figure 2.2, the SO_2 emissions are actually slightly greater than in the BAU case due to a reduction in the number of SO_2 controls applied, with a difference of 6% by 2055.

Adding GHG fees to the mid-range criteria pollutant fees results in additional reductions in criteria pollutant emissions. NO_x emissions are reduced as much as 14% from the BAU case and 12% compared to the case with fees on criteria pollutants alone, due to the combined use of control technologies and a reduction in coal use. SO_2 emissions are as much as 63% lower than in the BAU case and 22% lower than the mid-range criteria pollutant fee case. With GHG fees combined with criteria pollutant fees, 86% of generation from coal goes through FGD emissions controls after 2020. This is a slightly lower percentage than in the case with only criteria pollutant fees because more reductions are achieved through reduced coal combustion.

Figure 2.3 shows GHG emissions results for the four main cases we considered. As in Figure 2.2, results are also shown as error bars for cases with criteria pollutant fees based on low and high damage estimates. In the BAU case, GHG emissions increase over time, but more slowly than total electricity generation as technologies with lower emissions contribute a larger portion of the electricity. When mid-range criteria pollutant fees are applied without including GHG fees, there is a slight decrease of at most 2% in GHG emissions relative to the levels in the BAU case. When fees are only applied to GHG emissions, those emissions are as much as 14% less than in the BAU case. The maximum impact of GHG fees on GHG emissions occurs about 20 years out, and declines in the later years due to an increase in electricity demand that is not fully satisfied using low GHG intensity generation. Finally, the combination of fees on GHG and mid-range fees on criteria pollutants leads to slightly lower GHG emissions than the GHG fees alone, with reductions of up to 16% compared to the BAU case.

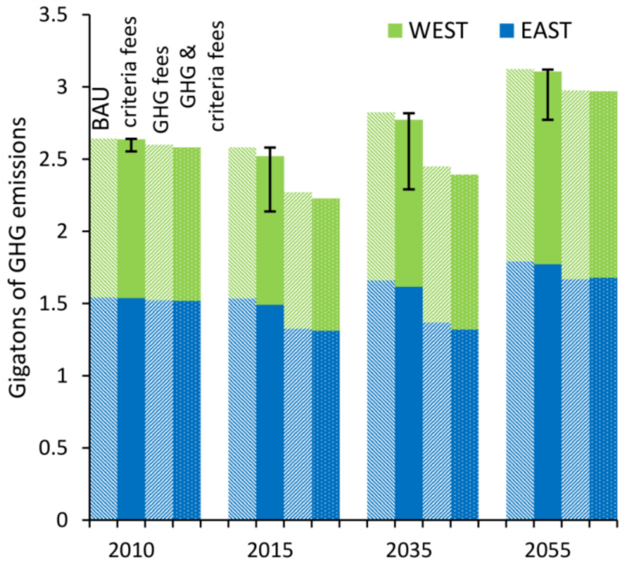

Figure 2.3. Lifecycle GHG emissions (in CO_2-e) in four different cases where the error bars represent the total emissions with high and low criteria pollutant fees.

2.3.1 Sensitivity Analysis

2.3.1.1 High Damage Estimates

One of the largest sources of uncertainty in the results shown above is the value of the external damages. To investigate how robust our findings are to this uncertainty, sensitivity cases are considered using fees based on high and low estimates of damages. For SO_2 and NO_x, respectively, the high criteria pollutant damages shown in Table 2.1 are approximately 13 and 9 times higher than the NRC 2010 estimates for coal-fired power plants, and 6 and 6.5 times higher than the NRC estimates for gas-fired plants. As shown in the Supporting Information (Figure 2.4) when criteria pollutant fees are set to correspond to these higher damage estimates, changes to the electricity system are much more pronounced than with mid-range criteria pollutant fees. Compared to the mid-range fees, high life cycle criteria pollutant fees lead to as much as 23% less electricity generated from coal. There is up to 5% less total electricity compared to the mid-range fees, and there is an increase in natural gas and wind power of up to 21% and 11% respectively. Thus the electricity mix may be altered by criteria pollutant fees alone, if the fees are high enough.

More emissions reductions are seen with high fees on criteria pollutants, including reductions in GHG emissions. There is up to 18% less CO_2 generated with high fees on criteria pollutants than with the mid-range fees (Figure 2.3). Fewer FGD controls are applied in this case than with the mid-range fees and the scrubbers are implemented more slowly, in part because the reduction in coal use also reduces emissions. The NO_x emissions are as much as 49% lower than with mid-range fees, and there is up to a 54% additional reduction in SO_2 emissions (Figure 2.2).

2.3.1.2 Low Damage Estimates

The low damage estimates given in Table 2.1 for SO_2 and NO_x are, respectively, about one-fourth and one-fifth as large as those presented in NRC2010 for coal plants. Correspondingly, when only the low criteria pollutant fees are applied the magnitude of changes to the generating system is significantly smaller than in the case with mid-range fees. As shown in the Supporting Information (Figure 2.4), there is very little change from the BAU case in total electricity generation or the mix of generating technologies. As shown in Figure 2.2, the SO_2 emissions are nearly the same as for the BAU case or more than double those in the mid-range fee case. FGD is installed on only 67% of the units across the country. With low fees, NO_x emissions are up to 8% lower than in the BAU case, whereas with mid-range fees NOx emissions were reduced by up to 12%. The GHG emissions levels are close to those for the median fee case.

2.3.1.3 Low Natural Gas Prices

In the cases in which natural gas is given a subsidy of $1 million/petajoules to represent the AEO 2012(U.S. Energy Information Administration, 2012) forecast of lower natural gas prices, some electricity generation is shifted from coal to natural gas, but the effect on emissions is small. With the subsidy, natural gas use is increased by up to 5% in the BAU case (without fees) and by up to 6% in the case with mid-range fees applied to criteria pollutants compared to the same fees without the subsidy. There are slightly lower GHG and SO_2 emissions (2% and 4% respectively) with low natural gas prices and criteria pollutant fees due to less coal combustion. In the case without fees, GHG and SO2 emissions are about 1% lower than in the BAU case. NO_x emissions show very little change with low natural gas prices.

2.4 Discussion

These results suggest that imposing criteria pollutant fees at levels that correspond to low to mid-range damage estimates would have little effect on total electricity production, the generation mix, or GHG emissions, but would substantially reduce criteria pollutant emissions through implementation of control technology. In contrast, criteria pollutant fees based on upper range damage estimates not only reduce criteria pollutant emissions but also affect the amount and method of electricity production as well as GHG emissions. When GHG fees of $30/ton CO_2-e are applied without criteria pollutant fees, total electricity generation, generation from coal, and GHG and NO_x emissions are all reduced, but not SO_2 emissions. When GHG fees are combined with mid-range criteria pollutant fees, the electricity portfolio is changed the most from the BAU case and both GHG and criteria emissions are reduced more than in the other cases.

The model determines which technologies to use based on the relative prices of the available options. In the MARKAL database, FGD controls for coal-fired power plants have typical cost-effectiveness of $2,000-4,000/ton. Therefore many coal plants have FGD installed with mid-range fees of $6,000/ton, but not with low fees of $1,500/ton. Cost-effectiveness estimates for both SNCR and low-NO_x burners in the MARKAL database are as low as $1,100/ton, which is higher than the low fees for NO_x of $370/ton but cheaper than the mid-range fees of $1,700/ton. Depending on conditions for a particular EGU, cost effectiveness of SCR in the database can be as much as $9,000/ton, which is still cheaper than high-end NO_x fees at $15,000/ton. Similar comparisons can be made between generating technologies if fees expressed in $/ton are expressed in $/MWh for specific units. As one example, MARKAL estimates that electricity generated at a certain existing coal plant costs about $10/MWh while a new wind turbine nearby would produce electricity that costs about $70/MWh. With GHG and mid-range criteria pollutant fees, the coal plant would have an additional cost imposed of $76/MWh.

Some technologies that are available in the model are never used in the cases presented here. CCS is an optional control technology but is not utilized by any of the scenarios considered here, as the GHG fee of $30/ton is less than the cost of CCS in the MARKAL database, which is about $140/ton. Integrated gasification combined cycle plants are not used either. Although the fuel is still inexpensive and some emissions (and fees) could be avoided, the capital cost of building these new plants is too high for them to be selected for capacity expansion.

While natural gas price projections are shifting rapidly, the results of our study are not highly sensitive to these forecasts. Natural gas use increases with time in all cases and its use increases over the BAU scenario when high criteria pollutant fees, GHG fees, or both are in place. Reducing the effective price of natural gas by $1 million/petajoules does increase natural gas use and decrease coal use, but the decrease in emissions from this change is small compared to the impact of fees. Thus, while less expensive natural gas may help to reduce particulates, fees are still needed to minimize externalities from electricity generation.

Some of the results shown here support key conclusions of previous studies for the US and other areas. Burtraw et al. (2003) found that in the US in 2010, a GHG fee would lead to health benefits in addition to the climate benefits, and with a GHG fee applied here, criteria pollutant emissions are reduced. Rive (2010) found that meeting the Kyoto protocol in Europe reduced the emissions controls needed to meet air quality policy targets. Here we see a similar result for the interaction of GHG and criteria pollutant emissions fees in the US, and the impact of internalizing GHG damages alone generally reduces criteria pollutants excluding a slight increase in SO_2 emissions in some cases In their global scale assessment, Klaassen and Riahi (2007) found that application of fees on criteria pollutants could also reduce GHG emissions. We also found this to be true for the US, but the GHG emissions reductions were small except when the highest level of criteria pollutant fees were applied. Thus, while the potential co-reductions of GHG and criteria pollutant emissions are evident, maximizing such benefits may require judicious design of emissions control strategies.

Further work is warranted to reconcile and refine damage estimates for criteria pollutants as well as greenhouse gases particularly given the range of values presented in the literature. Damage-based fees could also be modified for future years based on population growth projections, or varied regionally to reflect differences in damage estimates. In future work, the model could also be refined to include opportunities to directly reduce upstream emissions through controls or changes in production methods at upstream stages. In this study, upstream emissions were only modified through shifts at the electricity generation stage. Finally, the version of MARKAL used here models in-elastic end-use demand, so that all reduction in electricity demand comes from more efficient end-use devices. Future work with a fully elastic model would be needed to assess how end-use demand might change if fees increase the price of electricity.

2.5 Acknowledgements

We would like to thank the EPA for the use of the EPA US 9 region MARKAL database version EPAUS9r_2010_v1.3_052112_ucb. The results and analysis presented here were derived independent of EPA input. We would also like to thank Nicholas Flores and Garvin Heath for their helpful comments during early stages and Dan Loughlin for his assistance in using the MARKAL model. This research was supported by a University of Colorado seed grant from the Renewable and Sustainable Energy Institute and the NASA Applied Sciences Program NNX11AI54G.

2.6 Supplemental Information

2.6.1 Electricity Portfolio and Emissions in Additional Cases

Figures 2.4-2.9 below show the electricity and emissions generated under various assumptions in MARKAL in the year 2035. Though not shown here, the trend across years is similar for all cases with emissions and generation decreasing once the fees are applied, but increasing again as the demand increases due to population growth. This variation can be seen in Figures 2.1-2.3 in the text. The year 2035 is shown here because it represents a high level of variability between the cases since earlier years have not fully adjusted to the fees and later years become more constrained by the demand. Figure 2.4 shows the electricity generated by each fuel type in the cases run. High criteria pollutant fees and GHG fees have a relatively large effect on the amount of electricity generated and how the electricity is generated. Cases with mid-range and low criteria pollutant fee levels have a generation profile similar to the BAU case. The influence of reduced natural gas prices is relatively small.

As shown in Figure 2.5, SO_2 emissions exhibit a sharp response to mid-range and high criteria pollutant fees. Alone or in combination with GHG fees, they reduce SO_2 emissions by 50% or more, due to the application of controls. This is because the control technologies for SO_2 are inexpensive compared to the fee level. Since the control technologies only reduce combustion emissions in the current model, upstream emissions are only changed in cases where the fuel used to generate electricity changes. Upstream emissions are discussed further in the next section. As shown in Figure 2.7, the NO_x emissions response is smaller for low or mid-range fees, but there are lower emissions in these cases due to an increase in control technology and a decrease in combustion. With high fees on NO_x there are even more stringent controls used, which further reduces the emissions. The higher fee is required for further NO_x reductions because NO_x controls beyond those existing are relatively expensive compared to the mid to low fees. The upstream NO_x emissions show little change because in the model they cannot be reduced by the application of control technologies. As seen in Figure 2.8, GHG emissions are reduced either with GHG fees or with high-level fees on criteria pollutants. These fees do not result in direct application of CO_2 controls (i.e., carbon capture and sequestration), which is estimated in MARKAL to be relatively expensive. However, GHG fees or high-level criteria pollutant fees are high enough to stimulate some changes in fuel use, which reduce GHG emissions. It can be seen in the figure that the majority of the GHG emissions are from combustion, especially since no CCS is used to reduce those combustion emissions in these cases.

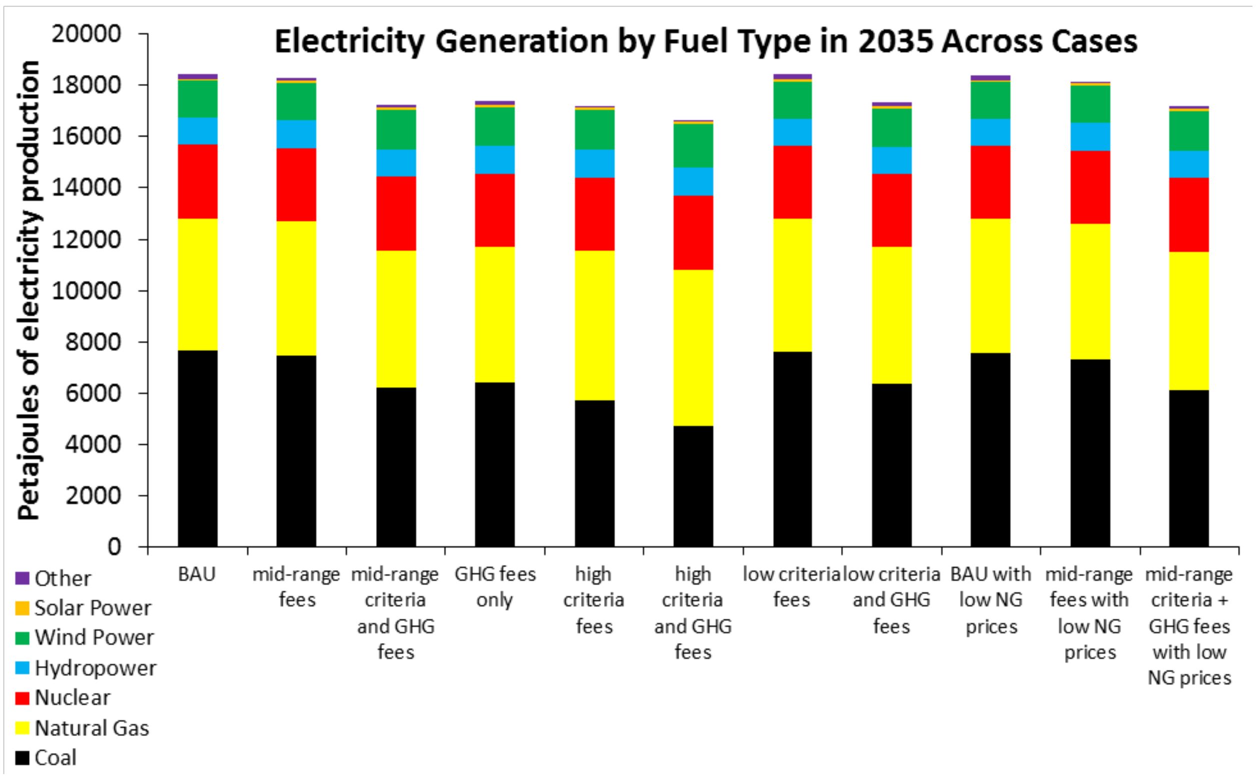

Figure 2.4. Electricity generated by fuel type in 2035 for the cases run. The “other” category includes oil, biomass, MSW, and geothermal.

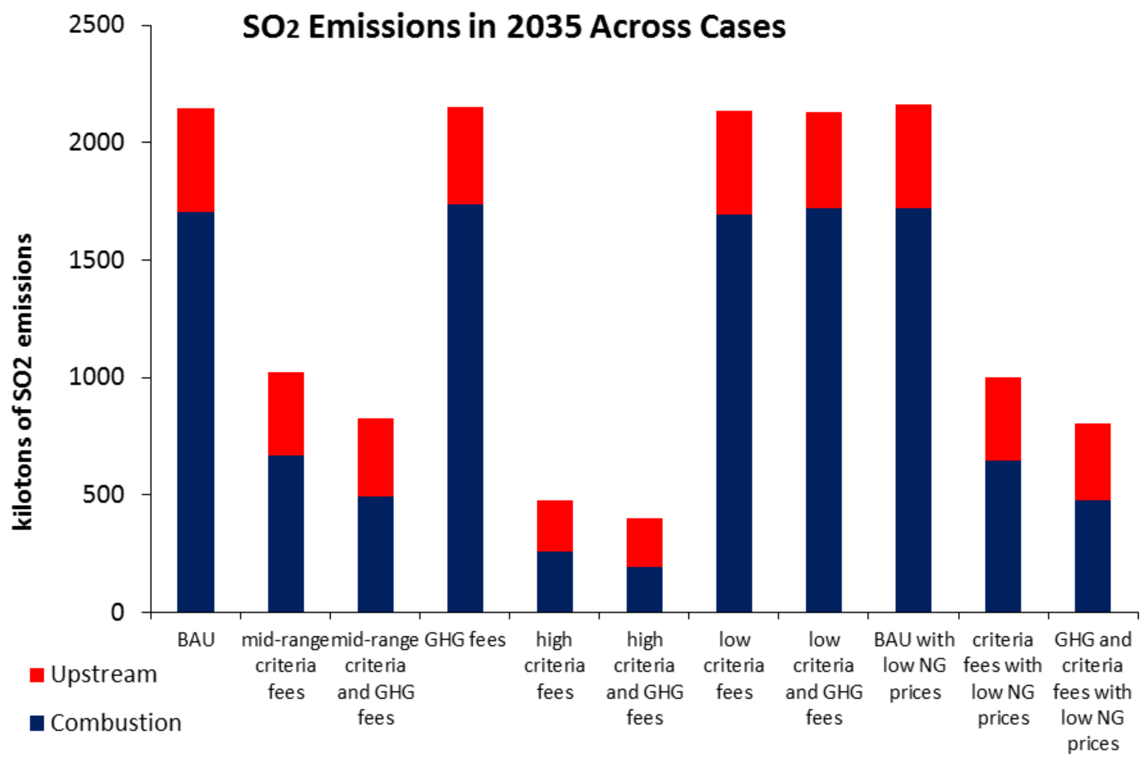

Figure 2.5. SO_2 emissions generated due to electricity production in 2035 across cases.

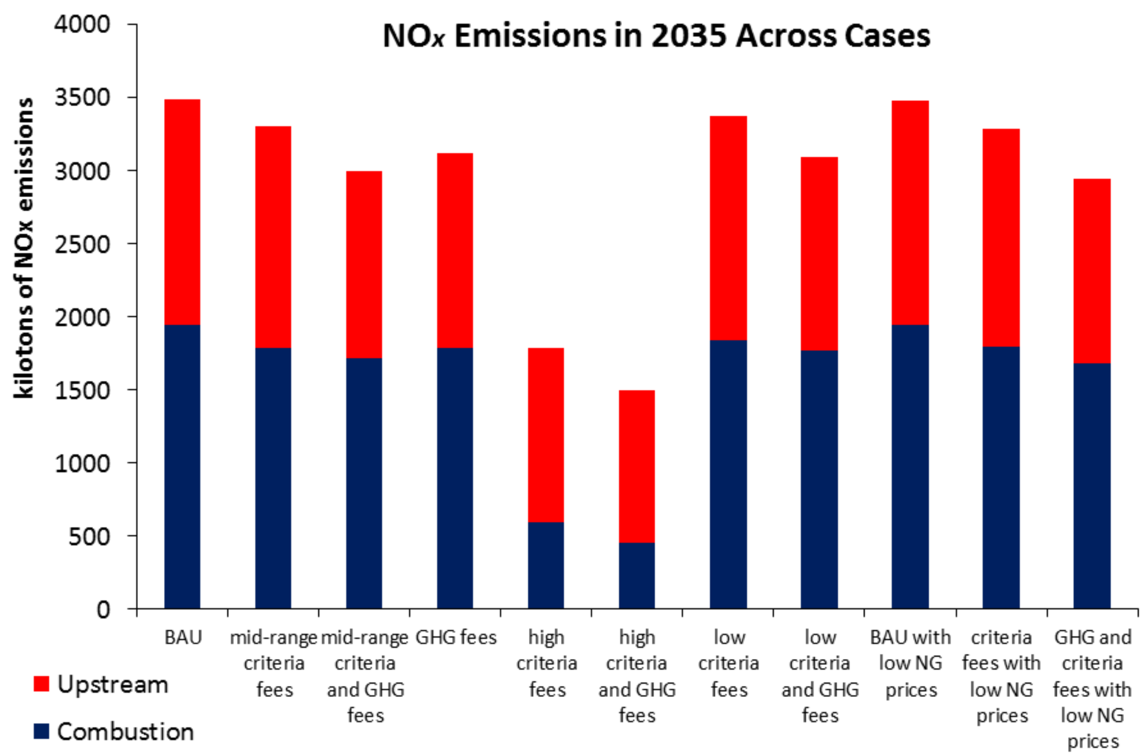

Figure 2.6. NO_x emissions generated due to electricity production in 2035 across cases.

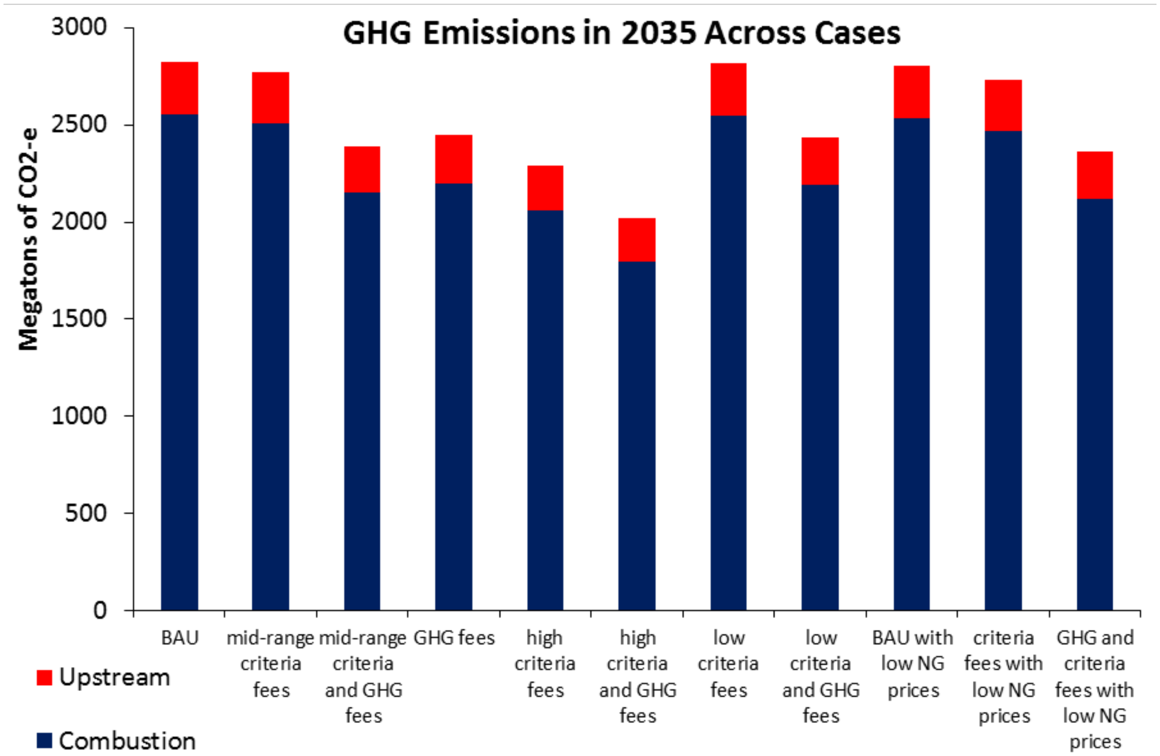

Figure 2.7. GHG emissions produced due to electricity generation in 2035 across cases.

Emissions of primary particulate matter (PM) are less responsive to fees compared to other emissions (see figure 2.9). As for GHG emissions, reductions in PM emissions are mostly due to changes in fuels as opposed to the use of additional control technologies. The combustion PM emissions in the model are already controlled in all cases including the BAU, and while additional controls are available they are more expensive than the fee on PM. Since the controlled PM emissions are already relatively low, the changes in PM emissions are driven by the fuel changes due to fees on other emissions. When mid-range or low criteria pollutant fees are applied to criteria pollutant life cycle emissions, PM_{10} emissions are very similar to the BAU case. When fees are applied to GHG emissions (alone or in combination with criteria pollutant fees), PM_{10} emissions are lower than the BAU case. The reduction in this case is due to the shift away from combustion technologies to reduce GHG emissions. With high fees on criteria pollutants, PM_{10} emissions are lower than the BAU and mid-range fee cases.

Figure 2.9 shows that the majority of the PM10 is emitted upstream. The values for the upstream coal PM emissions are highly uncertain because estimates come primarily from a single study published in 1999 (Spath et al., 1999). This study found that PM emissions are dominated by limestone quarrying for use in coal mining and sulfur controls. The data available for life cycle PM emissions for all generation types is limited, therefore the topic of lifecycle PM emissions for electricity generation warrants further study.

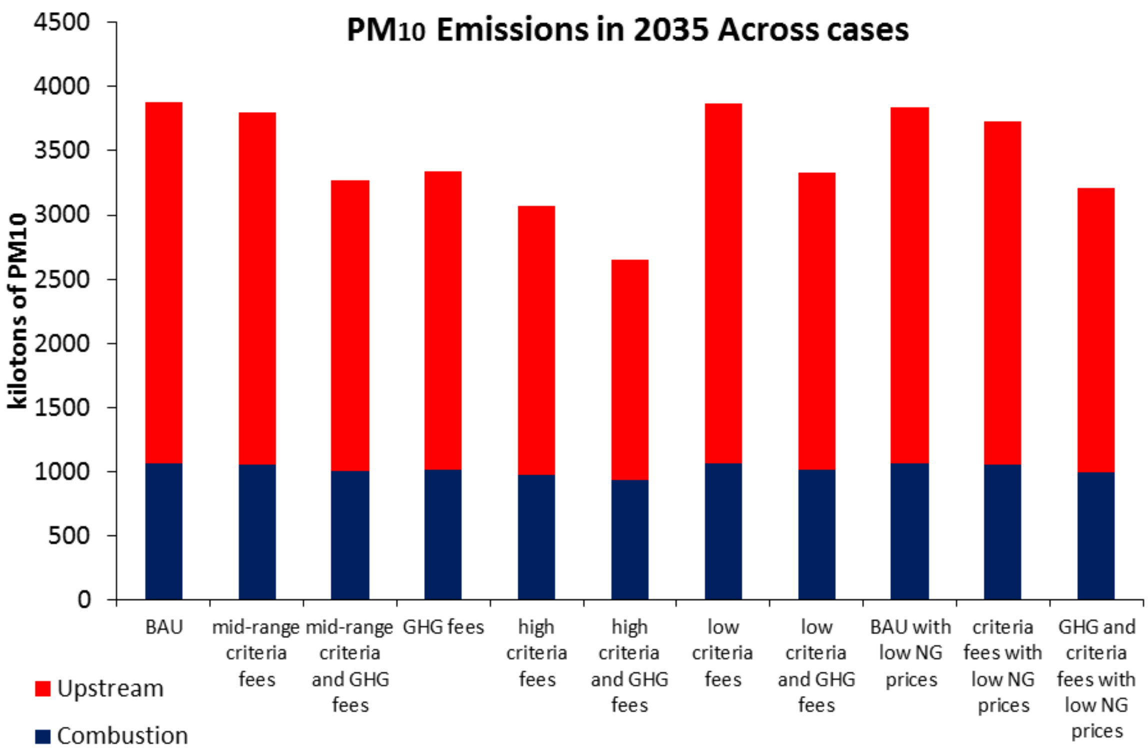

Figure 2.8. PM_{10} emissions produced due to electricity generation in 2035 across cases.

2.6.2 Life Cycle Emissions and the Effect of Incorporating Upstream Fees

In order to consider the effects of incorporating fees on life cycle emissions, not just direct fuel combustion emissions, emissions values representing the upstream emissions of GHG, NO_x, PM_{10}, and SO_2 for different electricity generating technologies are added to the MARKAL database. The electricity generating technologies considered are biopower, geothermal, hydropower, wind, nuclear, natural gas, coal, municipal solid waste (MSW), landfill gas (LFG), photovoltaic (PV), concentrated solar thermal power (CSP), fuel oil, and diesel oil.

The upstream emissions are added by creating new emission types (separate from combustion emissions) for each type of electricity generation named CO2EU, NOXEU, PM10EU, and SO2EU. For combustion based generation, the upstream emissions are given in terms of heat input of the fuel type where possible since the upstream emissions occur based on the amount of fuel used. In these cases the new emissions are added in terms of kt/PJ heat input. For generation types that do not rely on combustion, the new emissions are added in terms of kt/PJ electricity.

When such values are available, the upstream emissions are gathered from sources that estimated the upstream emissions separately as part of a life cycle analysis. Those upstream emission values are presented in table 2.2, as added to the MARKAL database.

Table 2.2 Upstream Emissions in kt/PJ

Fuel/generation type	**PJ in terms of**	**GHG**	**NO_x**	**PM_{10}**	**SO_2**
Coal	Heat input	5.0	0.052	NA [1]	0.006
Concentrated solar power (CSP)	Electricity	5.3	0.028	0.0072	0.014
Diesel	Heat input	28	0.056	0.022	0.13
Fuel oil	Heat input	28	0.089	0.025	0.12
Geothermal	Electricity	36	0.0056	0.008	0.75
Hydropower	Electricity	5.3	0.0097	0.014	0.008
Landfill gas ICE	Heat input	283	NA	NA	NA
Landfill gas turbines	Heat input	213	NA	NA	NA
Natural gas	Heat input	7.5	0.024	NA	0.0056
Nuclear	Electricity	8.3	0.011	0.0022	0.008
Photovoltaic	Electricity	12	0.029	0.036	0.061
Wind power	Electricity	2.8	0.0083	0.0028	0.01

For the following emissions, the upstream emissions values were determined by subtracting the combustion emissions in the EPA MARKAL database from the estimate of the life cycle emissions in the literature. Table 2.3 presents the entire life cycle emissions for those technologies.

Table 2.3**.** Life cycle emissions in kt/PJ(heat input)

Fuel type	**GHG**	**NO_x**	**PM_{10}**	**SO_2**
Biomass	2.9	0.0034	0.0043	0.014
Coal	See table 2.2	See table 2.2	0.13	See table 2.2
NG	See table 2.2	See table 2.2	0.0076	See table 2.2

More information on specific sources of emissions estimates is given below including the reference from which the data was taken. Data sources were chosen that

[1] NA = data not available

provided values for each of the four pollutants. Where possible we selected sources that separate combustion emissions from other life cycle emissions and used estimates specific to the US. The sources from which the upstream emission values were taken are listed below, as well as any emissions specific to that generation type. Landfill gas and natural gas GHG emissions include methane, not just CO_2; the upstream CO_2 emissions parameter CO2EU is used to represent all upstream GHG emissions in CO_2-e.

Biomass: NO_x and SO_2 upstream emissions come from Mann and Spath (1997) while CO_2 and PM life cycle emissions are from the National Research Council (2010). For biomass, the combustion emissions of CO_2 do not have fees applied because in a life cycle analysis the growth and combustion of biomass should have net zero CO_2 emissions. There are some CO_2 emissions from other processes associated with biopower and these are treated in the upstream emissions. The CO_2 emissions from biomass are affected by the feedstock used, whether it is waste residues or cultivated as an energy crop, and how the feedstock is grown and harvested. Land use change is not considered in this estimate.

Coal: Separate upstream PM emissions are not available so the PM values in table 2.3 represent whole life cycle PM_{10} from the average for the values reported by the National Research Council (2010). To get upstream values, the combustion emissions in MARKAL are subtracted from the life cycle emissions. Due to this calculation method and the limited number of PM estimates in the literature, there is a high level of uncertainty in the upstream PM estimates here. The PM upstream estimates are derived in part from a 1999 study that does not distinguish PM by size so this value may be an overestimate of total PM_{10} emissions. All other values represent only upstream emissions from Jaramillo et al. (2011). The NREL harmonization study (Whitaker et al., 2012) also published a report on coal. The life cycle coal values reported in the harmonization study have a median of 1001g CO_2-e/kWh electricity and the full life cycle emissions considered here are about 342 g CO_2/kWh heat input. Assuming a 30% or 40% efficiency the life cycle emissions would be 1140 or 855 g CO_2-e/kWh respectively, which is validated by the harmonization values.

CSP: Upstream emissions values are taken from the NEEDS study (New Energy Externalities Development for Sustainability, 2008), which considers a current trough system with salt as the working fluid and only using solar power to generate electricity at the plant. The solar thermal harmonization study from NREL (Burkhardt et al., 2012) provides a median value of 26 g CO_2-e/kWh and this study has a value of 19 g CO_2-e/kWh. This is within the interquartile range reported in the harmonization study.

Fuel oil and Diesel Oil: Santoyo-Castelazo et al. (2011) provide life cycle emissions for these fuels. Since the combustion damages are not separately considered in the model for these technologies, the upstream emissions added to the database consider the entire life cycle. The SO_2 emissions mainly come from the combustion of sulfur-containing fuels. Production of the fuels contributes to CO_2 due to gas flaring during extraction and combustion of the fuels also creates GHG emissions. Particulates and NO_x are generated in combustion.

Geothermal: The upstream emissions for geothermal are taken from Santoyo-Castelazo et al. (2011) because this report presents all the desired values. The CO_2 and SO_2 values presented in this report are within the range of values considered in Frick et al. (2010) There can be differences in life cycle emissions values based on each specific site, which can alter the level of emissions created while constructing the below-ground portion of the plant, which is the largest contributor to the life cycle impacts.(Frick et al., 2010) The values from Santoyo-Castelazo et al. (2011) are used here for all geothermal plants, but in practice, the emissions measured should be considered separately for each plant before levying a fee, due to the geologic differences that can affect emissions levels.

Hydropower: The average values of the upstream emissions from National Research Council (2010) are used. In their assessment, the largest source of GHG emissions from hydropower is from the flooding of biomass during initial hydropower development. More studies are needed to better understand this impact.

Landfill Gas (LFG): LFG life cycle emissions are already considered in the EPA MARKAL database. The upstream criteria pollutant emissions for LFG criteria pollutants are set equal to the values already in the database. CO_2 is treated as CO_2-equivalent for LFG to account for the high levels of methane emissions from this technology. The GHG emissions from LFG are from Kaplan et al. (2009) The GHG emissions are high for LFG technologies because landfills are a major source of CH_4 emissions and the emissions continue for a long period of time in contrast to waste to energy where the emissions are instantaneous (Kaplan et al., 2009).

Municipal Solid Waste (MSW): The EPA MARKAL database already considers MSW emissions over the whole life cycle. The upstream emission types for MSW are set equal to the emission levels in MARKAL. The emission fees for this technology are the upstream emission fees set to the total life cycle emissions of this technology.

Natural Gas: The PM values are taken from the average of the whole life cycle PM emissions from the National Research Council (2010). The combustion PM values from the EPA MARKAL database are then subtracted from the life cycle emissions to give the upstream PM emissions that are added to the database. All other values represent upstream emissions only and come from Jaramillo et al. (2011).

Nuclear: The average values of the upstream emissions from National Research Council (2010) are used. For GHGs this value is 30 gCO_2e/kWh which is higher than the median value of 12 g CO_2e/kWh given in the NREL harmonization study (Warner and Heath, 2012), but well below the high estimate in that study which was 110 g CO_2e/kWh, so this value is reasonable compared to other values in the literature.

PV: The average values of the upstream emissions from National Research Council (2010) are used which are within the range of values for PV given in the NREL harmonization study (Kim et al., 2012), but on the high end of the values. Since the harmonization assumed solar irradiance typical of the southwest, the higher value of CO_2/kWh used here makes sense. The emissions values are dependent on the conversion efficiencies of the panels, the energy mix and intensity used during manufacturing, and also the solar insolation at the site of installation.

Wind: The average values of the upstream emissions from National Research Council (2010) are used. This value is also justified in the NREL harmonization study (Dolan and Heath, 2012) where the median GHG emission estimate is 11 g CO_2-e/kWh, which for this study is 10g CO_2-e/kWh. Factors affecting life cycle CO_2-e emissions for wind include wind speed and generation capacity.

2.6.3 The Effect of Life Cycle Versus Combustion Fees

It is important to consider the full life cycle effects when comparing technologies, because while a single stage of the life cycle may dominate the emissions for one technology (for example the combustion of coal) other technologies may have emissions at other points in the life cycle that could have effects on human health and welfare. In this study cases were run considering just combustion emissions fees as well as full life cycle emissions fees. The overall conclusions hold whether life cycle or combustion fees are considered, that emissions fees lead to a reduction in emissions, and GHG and high criteria pollutant fees cause significant differences in the electricity mix to contribute to those reductions. There are some differences in the results when using combustion-only fees. The main differences occur for technologies with upstream emissions that are relatively high compared to their combustion emissions, which mostly concerns renewable electricity generation methods.

Renewable electricity generation changes differently when fees are applied to life cycle emissions than when applied to combustion emissions alone. Renewable technologies such as geothermal that seem inexpensive with combustion only fees are relatively more expensive when life cycle fees are in place because they have high upstream fees compared to their combustion emissions. When only combustion fees are in place, upstream emissions may increase due to the increased use of technologies with higher life cycle emissions. When upstream fees are also applied, technologies such as wind with low emissions across the entire life cycle are used and the upstream emissions are lower than the BAU and combustion fee cases. It can be seen in figure 2.10 that the overall fuel use is similar when the same fee is applied to combustion or life cycle emissions, but as shown in figure 2.11 the renewable portfolio changes. The cases shown represent no fees, fees on criteria pollutants only, fees on GHGs only, and fees on both criteria pollutants and GHGs. The change is most obvious when looking at the difference in wind and geothermal electricity generation between two cases where the same fee is applied to the combustion or full life cycle emissions.

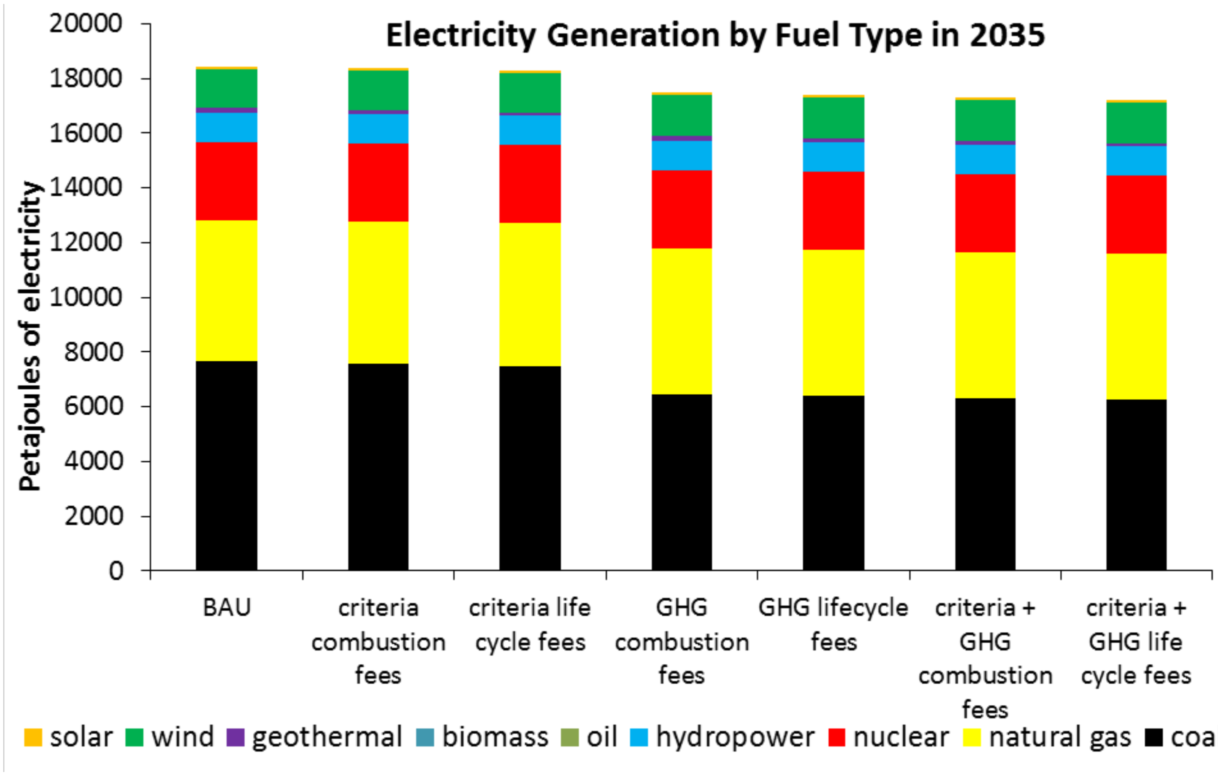

Figure 2.9 Fuel Use in 2035 for cases with fees on combustion and life cycle fees.

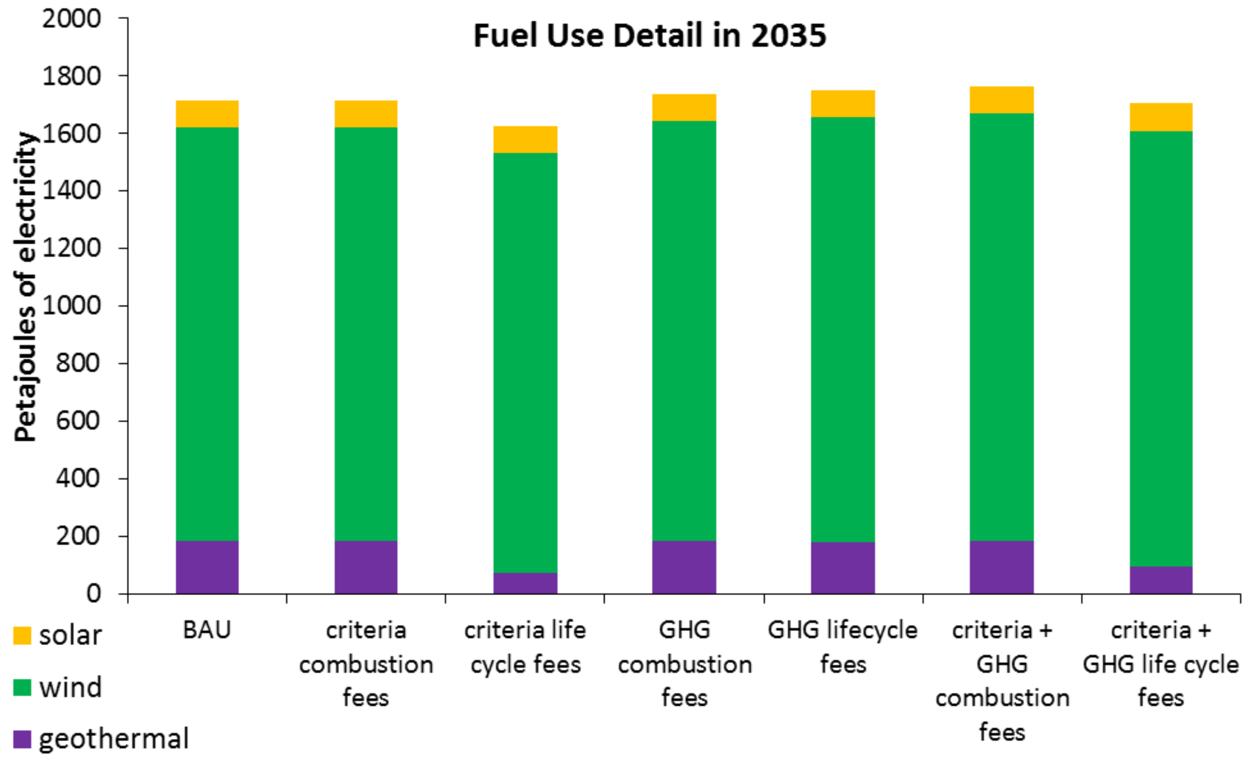

Figure 2.10 Detail of fuel use in 2035 for cases with fees on combustion emissions and life cycle emission fees.

The emissions are also different in these different cases. With criteria combustion fees, NO_x combustion emissions are reduced by up to 17% compared to the 16% reduction with life cycle fees, but the upstream emissions are up to 3% greater than in the BAU case with combustion fees and up to 3% less with life cycle fees. The total NO_x emissions are also up to 1% higher than the BAU case with criteria combustion only fees. SO_2 combustion emissions are up to 62% lower than in the BAU case with life cycle fees and up to 60% lower with criteria combustion fees, but the upstream emissions are 20% lower with life cycle fees and up to 12% higher with combustion only fees. Overall SO_2 emissions are lower than the BAU case for both cases. For the combined fee cases, the conclusions are similar, but the reductions are higher. For both combustion and life cycle GHG fees, the combustion GHG emissions are up to 14% lower than in the BAU case, but the upstream GHG emissions are up to 6% less than the BAU case with combustion fees and up to 9% less with life cycle fees.

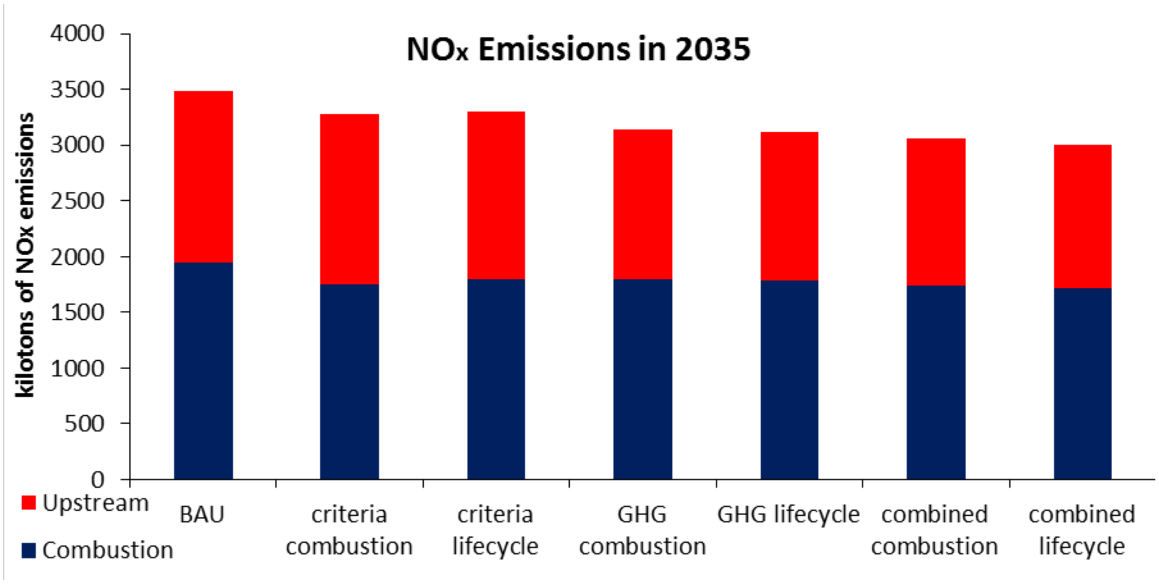

Figure 2.11 NO_x emissions in 2035 in cases with fees applied to combustion and life cycle emissions.

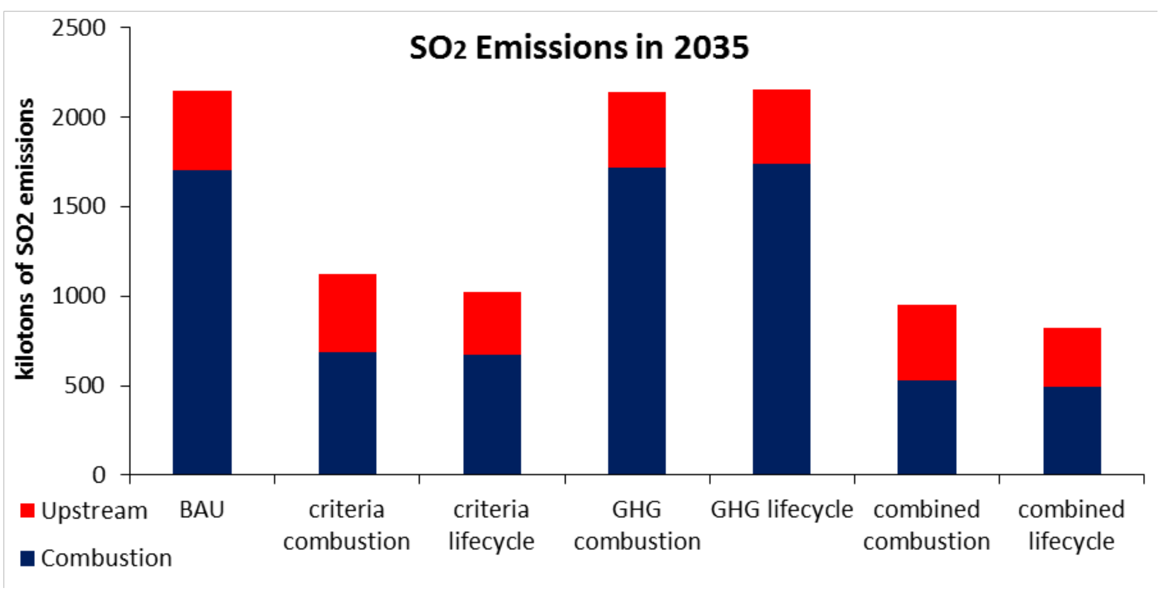

Figure 2.12 SO_2 emissions in 2035 in cases with fees applied to combustion and life cycle emissions.

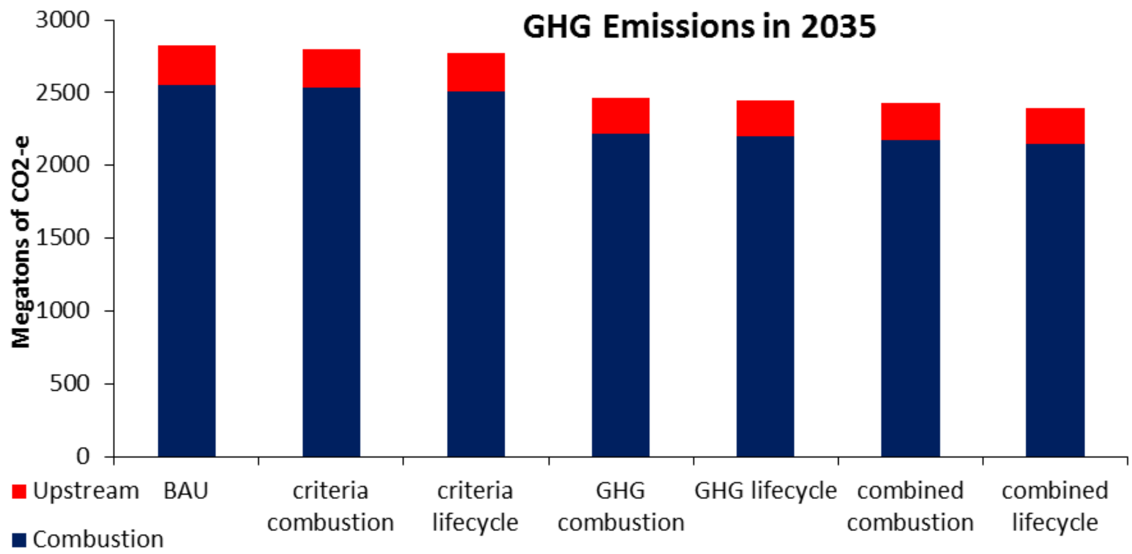

Figure 2.13 GHG emissions in 2035 in cases with fees applied to combustion and life cycle emissions.

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Chapter 3

How Accounting for Climate and Health Impacts of Energy Use Could Change the US Energy System

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3.0 Abstract

This study aims to determine how incorporating damages into energy costs would impact energy use in the US. Damages from health impacting pollutants or HIPs (NO_x, SO_2, particulate matter PM, and O_3) as well as greenhouse gases (GHGs) are accounted for by applying emissions fees equal to the estimated external damages associated with life cycle emissions. We determine that in a least-cost framework, fees reduce emissions, including those not targeted by the fees. Emissions reductions are achieved through the use of control technologies, energy efficiency, and shifting the fuels and technologies used in energy conversion. The emissions targeted by fees typically decrease, and larger fees lead to larger reductions. Compared to the base case with no fees, in 2045, SO_2 emissions are reduced up to 70%, NO_x emissions up to 30%, $PM_{2.5}$ up to 45%, and CO_2 by as much as 36%. Emissions of some pollutants, particularly VOCs and methane, sometimes increase when fees are applied compared to the case without fees. The co-benefit of reduction in non-targeted pollutants is not always larger for larger fees, as HIPs are reduced less with the largest GHG fees than two of the other GHG fee cases considered. NO_x emissions are reduced by 8% with the largest GHG fees, but 13% with the second largest. The degree of

co-reduced emissions is dependent on the technology pathway used to achieve emissions reductions, including the mix of energy efficiency, fuel switching, and emissions control technologies.

3.1 Introduction

The emission of pollutants associated with energy production and use has effects on both local air quality and global climate. For example, the health related damages from electricity generation in the US in 2005 have been estimated at over $62 billion (National Research Council Committee on Health, Environmental, and Other External Costs and Benefits of Energy Production and Consumption, 2010). Greenhouse gas (GHG) related damages from electricity generation in 2005 were $118 billion if calculated with the social cost of carbon (Interagency Working Group on Social Cost of Carbon, 2013) for the year 2010 with a 2.5% discount rate. The direct health impacts of air pollution include effects such as premature mortality (Krewski et al., 2009) and asthma exacerbation (Mar et al, 2004). Global climate change has effects on temperature and weather patterns (Kirtman et al., 2013), with consequences such as loss of crops and an increase in the prevalence of diseases.

These consequences can be viewed as externalities, i.e., effects on the well-being of an unrelated group or individual outside the market mechanism that controls the price of energy. Damages are the monetary value of externalities, such as the value of medical bills from adverse health effects resulting from exposure to air pollution. Incorporating damages into the cost of energy would encourage practices that reduce the externalities. The most efficient policies to address externalities are those that are directed at the externality itself, such as a fee on emissions rather than a fee on electricity. This allows the policy to most effectively reduce the externality instead of reducing the surrogate. By considering fees based on damages instead of an emission or technology goal, even if fees cause an increase in the price of electricity, the overall social cost related to electricity will decrease because external costs are lowered.

Guided by these general economic principles, several previous works have explored how energy systems might develop in response to application of fees to internalize external damages. These studies (Klaassen and Riahi, 2007; Nguyen, 2008; Pietrapertosa et al., 2009; Rafaj and Kypreos, 2007) have used integrated energy system models to make predictions of changes to their energy usage and production if fees are applied. These papers found that internalizing externalities might reduce energy consumption, change the types of generation technologies, and increase the use of control technologies. They also found that emissions of un-taxed pollutants are sometimes reduced as well. A previous analysis (Brown et al., 2013) focused on internalizing damage costs in the electric sector in the US, but did not consider how the system would respond to fees implemented across all energy sectors. Applying fees more broadly could yield greater emissions reductions and health benefits, or allow for more cost effective emissions reductions. This will also ensure that reduced emissions in the electric sector are not outweighed by increased emissions elsewhere.

The dual impacts of air pollution on human health and climate, and differences between the regulatory frameworks designed to address health impacting pollutants (HIP) versus greenhouse gases (GHGs), raises questions regarding how fees on emissions of one set of species may impact the other. Different pathways to emissions reductions can have different co-benefits or even regional disbenefits, as shown by Driscoll et al. (2015) in their analysis of Clean Power Plan scenarios targeting GHG reduction from electricity generation in the US. Carbon policies that allow reductions from multiple sectors have been estimated to achieve larger co-benefits and reduce the cost of compliance (Saari et al., 2015; Thompson et al., 2014). Other studies (Chen et al., 2013; Zapata and Muller, 2013; Kleeman et al., 2013; Nam et al., 2013) have evaluated how air quality and climate goals might be met symbiotically. These studies found that energy efficiency and fuel switching measures usually lead to co-benefits. On the other hand, Leinert et al. (2013) found that Ireland might actually see more NO_x emissions when reducing GHG emissions than they would under a BAU scenario due to a shift from gasoline to diesel vehicles.

In this paper, we evaluate how incorporating external damage costs into the cost of energy could change energy use and emissions in the US. A range of damage estimates from the literature is used to construct various policy scenarios, all of which prescribe a set of damage-based emission fees associated with emissions of GHGs and HIPs. A modified version of the EPA US 9 region MARKAL (MARKet ALlocation) model is used to evaluate changes to the US energy system through the year 2055 under these policies. This framework is applied to evaluate how emissions reductions from the energy sector might be achieved in the US by considering different methods of reduction such as control technologies, changing fuels or conversion technologies, and improved efficiency. Further, we compare the emissions reductions possible with fees directed at HIPs and GHGs alone and simultaneously directed at both. We examine co-reductions and increases in non-targeted pollutants as well as reductions in targeted pollutants. Our fee structure and modeling system are specific to the energy system (extending from fuel extraction through processing, energy conversion, and end use); we hence do not consider possible emissions reduction pathways in other economic sectors such as agriculture, waste disposal, or most industrial processes. Most anthropogenic emissions in the US are associated with energy production, conversion, or use including 83% of anthropogenic GHG emissions (US EPA, 2016), 95% of anthropogenic NO_x emissions, 60% of VOC emissions, 48% of primary $PM_{2.5}$ emissions, and 91% of SO_2 emissions (Office of Air Quality Planning and Standards, 2015); therefore, although we restrict our analysis to the energy sector, we capture the majority of anthropogenic emissions.

3.2 Methods

3.2.1 Health Related Damages

The HIP emissions considered here are NO_x, $PM_{2.5}$, PM_{10}, SO_2 and volatile organic compounds (VOCs). Although Hazardous Air Pollutants (HAPs) can also cause health impacts, we do not consider these here. Three sets of sector-specific, damage-based fees are used in this paper, shown in Table 3.1. Damage values for pollutants should be location dependent because emissions that are located near population centers will often affect more people than emissions located in rural areas. Although damages are not varied by location in this study, the location dependence is captured by using different damage values for different sectors, for instance industrial emissions tend to be closer to population centers than electricity generation emissions, therefore industrial damages tend to be higher than electric sector damages per ton of emission.

Table 3.1. Health impacting pollutant damages used as fees. (All values in $/t unless otherwise specified.) The asterisk (*) represents values that were taken from a different literature source than the rest of that set of fees, see sources in text.

$/ton	Sector	NOx	PM10	PM2.5	SO2	VOC	Natural Gas Use M$/PJ
Low Sector Specific Fees	Electric	364	195	2261	1866	240	
	Industrial	547	378	4343	2274	436	
	Transportation	593	444	5147	2476	510	
	Upstream	501	339	3917	2205	395	
	Refinery	547	378	4343	2274	436	
	Residential						0.059
	Commercial						0.025
Mid Fees	all	1970	1115	21520	9750	1720*	
High Sector Specific Fees	Electric	4700	4110*	117100	31500	2330*	
	Industrial	5500	4110*	234300	35100	2330*	
	Transportation	6600	4110*	324400	17100	2330*	
	Upstream	5600	4110*	225267	27900	2330*	
	Refinery	5900	4110*	279300	59500	2330*	
	Residential	11700	4110*	324400	87400	2330*	
	Commercial						0.579

We designed the three sets of fees used here after reviewing the literature and choosing values that capture the low-end, high-end, and mid-range level of values reported. The low sector-specific fees are derived from Muller et al. (2011). The mid-range fees are from National Research Council (2010) hereafter called NRC2010, except that the mid-range VOC fee is based on the geometric mean of the VOC damages from Muller and Mendelsohn (2007) and Fann et al. (2009). The high fees are based on damages from Fann et al. (2012) except that the VOC damages from the high fee case are based on Fann et al. (2009). The VOC damage estimates do consider VOC as a precursor to $PM_{2.5}$. The PM_{10} values in the high fee case are based on the NRC2010 values multiplied by a factor representing the average increase of Fann et al. (2012) over NRC2010. These adjustments were made so that the same set of pollutants has fees applied in all cases. Damages for the residential and commercial sectors are sometimes only applied as fees to natural gas used in these sectors. This is because fewer studies analyzed damages for these sectors and they are often reported in terms of natural gas use instead of tons of emissions. When emissions specific values were available, they were used. Most energy use in these sectors is electricity or natural gas, so we assume that these estimates capture most of the damages. The damages in Table 3.1 in the natural gas use column are derived from NRC2010 by multiplying by a ratio as described above form PM_{10}.

There are a few reasons for discrepancies in damage estimates from previous studies, mostly related to differences in assumptions when calculating the damages. Whether or not age is taken into account when applying the Value of Statistical Life (VSL) to mortalities has a large effect on the damage value. Using a uniform VSL can produce 50% higher marginal damages than differentiating by age (Muller et al., 2011). Only Muller et al. (2011) differentiate VSL based on age. Another factor that can cause differences in the calculated marginal values is what sources are considered. The studies by Muller et al. (2011) and Fann et al. (2012) consider all sources of emissions while NRC2010 focuses on EGUs combusting coal and natural gas. Variations in population exposure also contribute to differences among damage estimates, so the areas considered in each study have an effect as well. Since mortality from $PM_{2.5}$ is a significant component of the damage estimates, the choice of the corresponding concentration-response functions is also important. NRC2010 and Muller et al. (2011) used results from Pope et al. (2002) and Woodruff et al. (2006) to relate $PM_{2.5}$ exposure to mortality; Fann et al. (2012) used the health impact functions from Krewski et al. (2009). Fraas and Lutter (2013) discussed the uncertainties underlying the published damage estimates, and found that the uncertainty in the concentration-response functions may be larger than that encompassed by the range of studies considered here, Buonocore et al. (2014) analyzed the variability of damage estimates between individual facilities; such spatial variability of marginal damages is important for evaluating the benefits of alternative energy technologies (e.g., Siler-Evans et al., 2013). Although we do not have the ability to incorporate this level of variability into our modeling framework, this is partially accounted for in our work by using sector specific damages and having a multi-region model. In this paper we assume that location difference is captured by applying fees based on the sector from which the emissions are produced.

3.2.2 GHG related damages

The greenhouse gas damages used in this study are taken from the Social Cost of Carbon (SCC) estimates developed for use in regulatory impact analysis by the US government (Interagency Working Group on Social Cost of Carbon, 2013). The SCC documentation recommends considering the full range of values they report because there are many uncertainties involved. We have considered all four sets of values, which increase with time and are reported in the appendix (Table 3.3). Fees are applied to CO_2 and CH_4. The SCC values are adjusted for CH_4 using the 100 year global warming potential (GWP) of 28 (Myhre et al., 2013). Three of the sets of fees are determined by averaging the results of several models considering different discount rates: 5%, 3% and 2.5%. The fourth set of fees represents the 95th percentile of the ensemble of estimates for the 3% discount rate.

Some studies have found damage estimates larger than those used here. Moore and Diaz (2015) described how SCC may be underestimated because the effect of warming is compounded over time as GDP is reduced due to climate impacts. When Dietz and Stern (2015) incorporated endogenous growth into the DICE model, allowed for damages to increase rapidly with respect to temperature, and explored the climatic response to GHG emissions, they found the range of damages exceeds those used here. Lontzek et al. (2015) created a stochastic version of DICE and found that the carbon costs were higher than projected from the deterministic version. Also, Howard (2014) reported that the SCC is probably biased low because some impacts are not included. NRC2010 found that almost all variation in marginal damage estimates derives from differences in assumptions of discount rate and the magnitude of damages from climate change, especially whether unlikely but catastrophic effects were considered.

3.2.3 The MARKAL model

We use the MARKAL (MARKet ALlocation) energy system model to compare the fee scenarios (Brown et al., 2013; US EPA, 2013b; ETSAP, 1993; Loughlin et al., 2011; Loulou et al., 2004). MARKAL uses linear optimization to determine the lowest cost set of technologies required to meet energy demand. All end use demands must be satisfied, through either generation or conservation technologies, and constraints such as emissions regulations can be applied (Loulou et al., 2004). MARKAL considers the economic advantages of different technologies. Incorporating external damages as costs ensures that the environmental costs are also considered.

We use a modified form of the EPA US 9 region 2014 v 1.1 database (US EPA, 2013b). The database represents the US energy system for the years 2005-2055 in 5 year increments with a system-wide 5% discount rate. Predictions have been made for the US energy system under the “business as usual” (BAU) case in the Annual Energy Outlook (AEO). The EPA release of the MARKAL database is benchmarked to the 2014 AEO (U.S. Energy Information Administration, 2014). The database includes the projected demand for energy services as well as the existing technologies in each of the nine US census regions. Demand is specified in terms of heat, lighting, and other end use services instead of the amount of energy required so that efficiency options are available. The projected demand in the database changes over time based on AEO projections.

The database also defines technologies that are available to install, including both traditional and advanced options. All technologies are defined by capacity, efficiency, cost, and emissions rate. Some technologies also have hurdle rates defined, which are discount rates that are higher than the standard 5% and reflect behavioral or non-economic barriers to investment in new technologies. Typical values for hurdle rates are 5-20%. The database includes investment costs, fuel costs, and operation and maintenance costs. Costs also change with time, especially for newer technologies for which learning curves cause the cost of the technology to decrease over time. Energy sources available in the database include waste, solar, wind, natural gas, coal, geothermal, hydropower, biomass, nuclear, and oil. The database includes supply curves for natural gas, coal, biomass, and oil. The model also includes end use efficiency options to meet the end use demand, such as lighting, with a reduced energy demand. Resources and energy are traded between regions where pathways such as transmission lines are defined between those two regions.

The database includes several energy sectors, including transportation, residential, commercial, and industrial end use sectors as well as the refinery and electricity sectors that convert energy to meet the end use demand. The database also captures the energy cycle including resource extraction, processing, electricity generation, electricity and direct fuel use in the other sectors. Control technologies are also available in the model, including carbon capture and storage (CCS) to remove CO_2, flue gas desulfurization (FGD) to remove SO_2 emissions, and a variety of PM and NO_x removal technologies.

The database also includes some existing regulations. In the electric sector, SO_2 and NO_x emissions are constrained to comply with the Clean Air Interstate Rule (CAIR), although this has since been replaced by the Cross State Air Pollution Rule (CSAPR), the regulations are very similar at the level of resolution represented in the database. The database also includes the Mercury Air Toxics Rule (MATS) as well as other Clean Air Act-based regulations. State level renewable fuel standards are also represented. Current US policy directed at reducing GHG emissions includes the Clean Power Plan (CPP), which focuses on electricity generation, and transportation sector policies including Corporate Average Fuel Economy (CAFE) standards. There is also proposed policy targeting methane emissions from oil and gas extraction (Bureau of Land Management, Interior, 2016; US EPA, 2015b), but these are not represented in the database. The Clean Power Plan is not included in MARKAL, but is discussed in the appendix. In the transportation sector, compliance with the CAFE standards requiring 54.5 mpg by 2025 and Tier III emissions regulations is required. In the industrial sector, the Industrial/Commercial/Institutional (ICI) boiler Maximum Achievable Control Technology (MACT) rule is represented.

The flexibility and completeness of MARKAL make it well-suited for this study. However, MARKAL does have limitations in what it can evaluate. While this model can determine the types of responses to the fee that might be likely including whether and how emissions are reduced and whether co-benefits or dis-benefits are likely, it would be inaccurate to represent the results as precise emissions forecasts. MARKAL is not designed to determine how novel or disruptive technologies might contribute to the energy system; it is possible that emissions fees would spur innovation in technologies that are not represented in the model. Although some energy sectors are more capable of responding to these fees in both the model and real life, certain sectors may not have enough technologies modeled to respond to the fees even if such options are available in the real world, especially if newer financing or technology options gain traction in sectors such as residential, commercial, and transportation. MARKAL provides a picture of the possible responses to emissions fees and shows that emissions reductions to a certain level are possible, but the results do not show the only possible pathway to emissions reductions.

3.2.4 MARKAL Database Changes

3.2.4.1 Emissions

Starting from the EPA US 9 region database described above, we modify certain aspects of the database to better suit analysis of the possible methods of reducing emissions. A more comprehensive document detailing the changes can be found in the appendix, as well as a comparison of the EPA base case with the base case used here. Additional emissions tracking parameters are added to be able to analyze emissions from each sector (electricity generation, refining, industry, transportation, commercial, and residential). We also improve treatment of upstream emissions by both adding previously uncounted emissions as well as adding the ability to track these emissions separately. In addition to adding emissions, CO_2 uptake for biomass production was added as well, as described in the appendix. The additional values and sources are given in Table 3.5.

3.2.4.2 Technology Changes

Most of our changes to the EPA database are for the industrial sector, because many technologies that could be used to reduce emissions in this sector are not included in the original EPA database. The EPA database includes demand for both heat and energy in the industrial sector. There are different fuel options available to meet these demands, but no emissions control technologies. The only efficiency improvement defined in the EPA database is for natural gas boilers. Due to the complex and varied nature of the industrial sector, it is not possible to represent the full range of methods available to reduce emissions, as some emissions reduction options are only available for small subsectors of industrial energy use. We expand the technologies modeled in the industrial sector to create a representative picture of the types of responses available (i.e., fuel switching, efficiency improvements and control technologies), which allows us to better analyze which emissions reduction techniques might be important.

For this study, we add the option of improving boiler efficiency by one percentage point at an associated capital cost of 1.082 million USD per PJ (US EPA, 2010) for boilers not fueled by natural gas. This is not an additional option for natural gas boilers because the ICI boiler MACT rule already requires similar efforts for them.

Emissions control technologies for NO_x, SO_2, and PM are added to boiler and process heat energy use. The technologies added are based on CoST (Misenheimer et al., 2010) modeling[2]. For boilers, two levels of NO_x controls and one level of SO_2 controls are added. For process heat, options are added to control SO_2, PM and NO_x. All controls become available in 2015 with a 40-year lifetime. Penetration of controls is constrained to at most 80% of the possible level. For boilers subject to the boiler MACT regulations or for which other controls are assumed in the EPA database, additional controls are not modeled on affected pollutants, with the exception that in some cases Low NOx Burners (LNB) exist for boilers but selective catalytic reduction (SCR) and selective non-catalytic reduction (SNCR)

[2] CoST modeling performed by Julia Gamas at the EPA

controls could still be applied. No PM controls are added to boilers because where they are feasible they have already been used to comply with the MACT regulations. No VOC controls are added because the controllable sources do not fit well with the technologies in MARKAL, because they are either not strictly part of the energy system or are applied to specific processes not explicitly defined in MARKAL.

For refineries, new emissions control options are added using performance and cost estimates determined based on data in the Mid-Atlantic Regional Air Management Association report (MARAMA, 2007). A wet scrubber provides the SO_2 and PM combined control; two NO_x reduction options are added, and advanced leak detection and repair (LDAR) techniques are represented to reduce VOC emissions.

Although the US currently does not utilize much solar heat in the industrial sector, other countries have found solar energy to be a cost efficient source of industrial heat (Islam et al., 2013). We define a flat plate solar technology that is only available for the food sector based on temperature requirements. We add the option of using parabolic trough technology, with performance characteristics adapted from data from the System Advisor Model (Blair et al., 2014) for concentrating solar power, and the investment cost based on a Public Interest Energy Research (PIER) demonstration facility in California (Ruby and American Energy Assets, California L.P., 2012). The use of these technologies is constrained, because solar thermal energy requires space and certain temperature demands. The constraints implemented are based on Lauterbach et al. (2012) and hurdle rates were set to 20%.

We also added Carbon Capture and Sequestration (CCS) control technology options for cement process heat. CCS is included in the original database only for electricity generation. The emissions for cement process heat are also altered to take into account the large portion of CO_2 that occurs due to calcination in addition to the CO_2 emissions related to fuel combustion, because these emissions can alter the economics of using the controls.

3.2.4.3 Other changes

Constraints are added to ensure that older coal fired power plants are retired after they have been in operation for about 75 years, because in the EPA database most mid twentieth century coal fired power plants are not retired within the time frame of the model. The CO_2 removal efficiency is lowered for biomass Integrated Gasification Combined Cycle (IGCC) with CCS based on the kgC/kWh rate given in Rhodes and Keith (2005).

3.2.5 Scenario Description

We run ten fee cases and a no fee (aka base) case. Three HIP fee cases and four GHG fee cases are run using the fees described above (Table 3.1). Three combined fee cases are run, one with mid range HIP fees and 3% average GHG fees (the lowest combined fee case), one with high HIP fees and 3% average GHG fees (the intermediate combined fee case), and one with high HIP fees and 2.5% average GHG fees (the largest combined fee case). Fees start in 2015 in all cases. In some figures, results from the base case are presented for 2010 to show the pre-fee level and 2045 to compare to the other results.

3.2.6 Current energy system

The current US electricity system uses mostly coal and natural gas to generate electricity. There are some technologies already in place to reduce emissions of health impacting pollutants, but more stringent control technologies are available. The industrial sector uses mostly natural gas with some actions taken to reduce emissions, but again further emissions reductions are possible. The transportation sector uses mostly gasoline and diesel fuel, and existing emissions controls are already widely used. Further emissions reductions could be achieved in this sector from advanced vehicle technologies, including fuel cell and electric vehicles. There is increasing demand for energy services in all sectors as population is expected to increase with time. Figure 3.1 gives a sense of this increase for most sectors in the base case, although it does not explicitly show the increase in demand for the transportation sector (roadway vehicle miles traveled is expected to increase 43% in all MARKAL cases from 2015 to 2055) as both miles traveled and fuel efficiency increase.

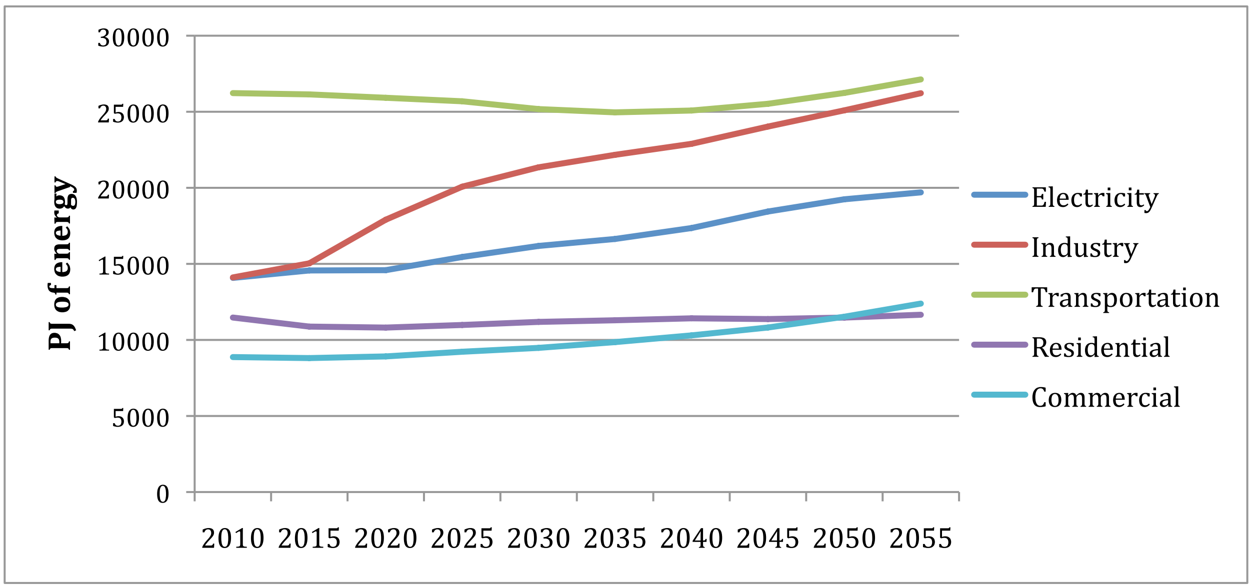

Figure 3.1. Total fuel used or electricity produced by each sector across time in the base case as a proxy for increase in demand for energy services. Vehicle miles traveled also increase, but transportation fuel use is mitigated by increased efficiency.

3.3 Results

3.3.1 Emissions

Emissions of HIPs decrease over time, in spite of increased demand, due to current emission reduction policies for transportation and electric sector emissions. In the base case, NO_x emissions are reduced 39% from 2010 levels in 2045. This is achieved by a 64% reduction in transportation NO_x emissions and a 30% reduction in electric sector NO_x and in spite of a 33% increase in industrial NO_x emissions. $PM_{2.5}$ emissions are 21% lower in 2045 in spite of a 54% increase in industrial emissions, due to a 63% reduction in transportation sector emissions and a 70% reduction in electric sector emissions. Base case SO_2 emissions in 2045 are 52% less than in 2010. Although industrial emissions increase 22% in this time period, electric sector emissions decrease 75%. VOC emissions are 31% lower in 2045, mostly due to a 74% decrease in transportation sector VOC emissions.

When considering either HIP or GHG fee-driven scenarios, we find that NO_x emissions reductions from fees (see Figure 3.2) are the greatest in the electric sector, followed by the industrial sector. Electric sector emissions are reduced by up to 82% with high HIP fees, but only a few percent with low HIP fees, and 11-33% with mid range fees across different years. As HIP fees increase, more stringent electric sector NO_x controls are used. With mid-range or high HIP fees, refinery NO_x emissions are reduced by 56% compared to the base case. Industrial sector NO_x emissions decrease 9-42% for HIP fee cases across fee levels and years and industrial boiler NO_x controls are used with mid-range and high HIP fees. Transportation emissions are reduced by at least 12% by 2040 in all cases with HIP fees and even earlier with high HIP fees.

GHG fees also cause HIP emissions reductions. Electric sector NO_x emissions are reduced by up to 37% with 2.5% average GHG fees, but with 3% 95th percentile fees the reduction is only a few percent, because the technologies used in this case are quite different from the other GHG fee cases. Industrial sector NO_x emissions decrease 3-24% in

GHG fee cases. With GHG fees, transportation NO_x is reduced 12% in 2040 and beyond, except for 5% average GHG fees in which only a 3% reduction is achieved.

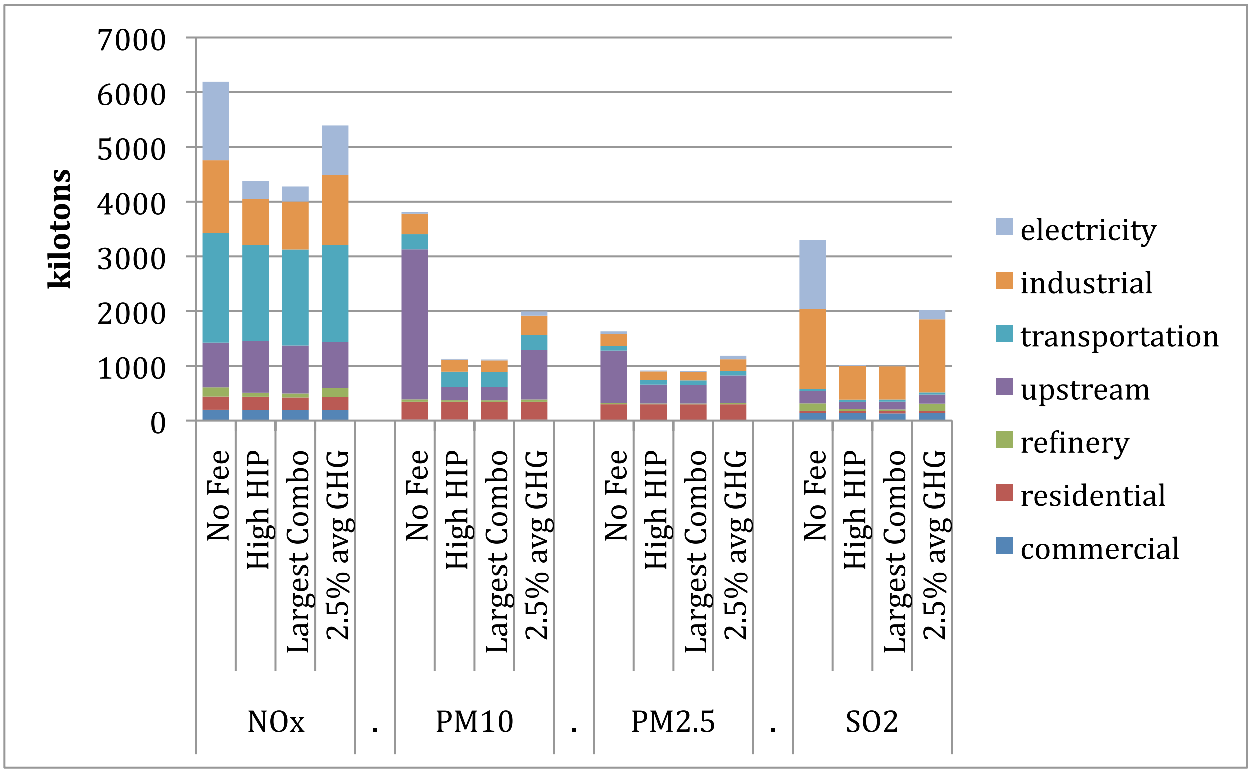

Figure 3.2. HIP emissions in 2045 for selected cases. The results for all cases can be found in the appendix in Figures 3.17-20. The largest combined fee case includes high HIP fees and 2.5% average GHG fees.

For PM emissions, over half of the emissions in both size fractions in the base case are from upstream processes, because control devices are used in most end use and electricity generation scenarios, but the model does not have any controls for upstream processes. Upstream PM is also the most responsive to fees, which can be traced to a reduction in mining emissions associated with reduced coal use. With high HIP fees, upstream $PM_{2.5}$ emissions are up to 67% less than the base case in the same year. For PM_{10}, the response is even larger, with 14-92% reduction after 2020 in HIP fee cases. With all HIP fees, the refinery $PM_{2.5}$ is 30-32% lower than the base case (32-29% for PM_{10}). Industrial sector $PM_{2.5}$ is 25-33% lower than the base case for any cases with mid-range or high HIP fees (29-42% for PM_{10}), but only 8-13% lower with low HIP fees (18-23% for PM_{10}). Emissions controls are applied in the industrial sector, with more stringent control technologies being used as HIP fees increase. With high HIP fees, electric sector $PM_{2.5}$ and PM_{10} is 53-79% less than the base case. With mid-range fees, the $PM_{2.5}$ reduction is 34-57% (42-74% for PM_{10}) across different years. With low HIP fees, the reduction is never more than 5%. Fabric filters are applied in response to fees to reduce electric sector PM emissions. In all GHG fee cases, upstream $PM_{2.5}$ and PM_{10} emissions are always at least 58% less than the base case. With GHG fees, the industrial PM does not diverge from the base case until 2035, after which the emissions are still within 10% for $PM_{2.5}$ and 16% for PM_{10} except for the 3% 95th percentile GHG fee case, which has up to 15% less $PM_{2.5}$ and 21% less PM_{10}. Electric sector PM changes the most between cases. With GHG fees, PM from the electric sector actually increases (up to several times) due to the increased use of biomass co-firing. With the use of additional controls, this impact could be mitigated, but additional regulation may be required before such technology is installed.

SO_2 emissions are highly tied to coal use, so they are heavily affected by fuel choice, but also by control technologies. With high HIP fees, the electric sector SO_2 emissions are reduced by at least 98% in 2025 and beyond and at least 63% starting in 2015; mid range HIP fee reductions are 76-81% after 2020. With low HIP fees, electric sector SO_2 emissions are within 30% of the base case. In the electric sector FGD scrubbers are used to reduce SO_2 in all cases, but reduced coal use is also a large factor with mid or high HIP fees. Industrial SO_2 emissions are 35-62% less than the base case in different years with mid or high HIP fees. In the mid and high HIP fee cases, SO_2 controls are used for process heat and boiler emissions in the industrial sector. GHG fees corresponding to the 2.5% average or 3% 95th percentile levels reduce electric sector SO_2 emissions by at least 83% after 2025, while the smaller and larger GHG fees lead to lower reductions. With GHG fees, the industrial SO_2 emissions are 2-33% less than the base case.

VOC emissions do not typically decrease with fees, and may even increase with fees because there are limited control technologies modeled to reduce these emissions and the fees on VOCs are smaller than on other pollutants. VOCs are discussed further in the appendix.

As shown in Figure 3.3, CO_2 emissions are lower than in the base case in all fee cases. Most of the reductions occur in the electric sector. With mid range HIP fees, the electric sector CO_2 emissions are 8-12% less, and with high fees 38-47% less across different years. There are also reductions from the industrial sector. With high HIP fees the reduction is up to 14%, or 10% with mid range fees. Electric sector emissions in 2040 are 9-13% lower than in the base case with 5% average GHG fees, 26-35% lower with 3% average GHG fees, 37-46% lower with 2.5% average fees, and 89% lower with 3% 95th percentile fees. CCS is used in the electric sector to reduce CO_2 emissions in the 2.5% average and 3% 95th percentile cases. The sequestered CO_2 is represented in Figure 3.3 as an outlined box, because it does not contribute to the total emissions. 2.5% GHG fees reduce industrial CO_2 by 5% in most years, while 3% 95th percentile fees cause reductions up to 12% and 3% average fees cause reductions of at most 6%.

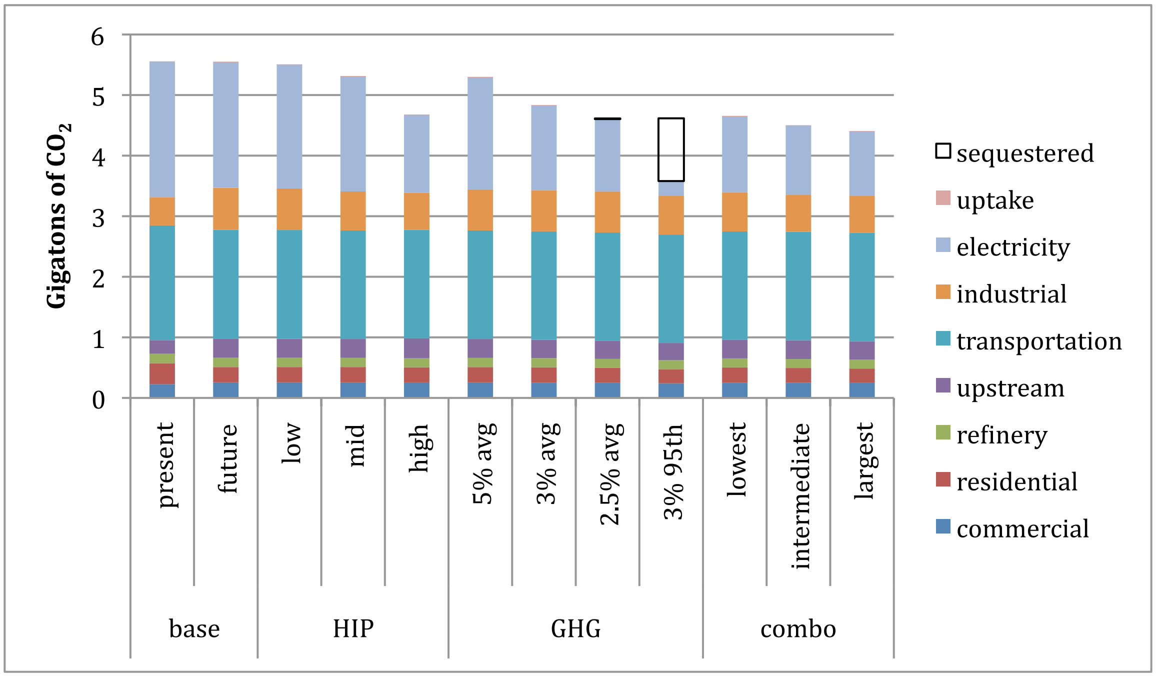

Figure 3.3 CO_2 emissions by energy use sector in 2045 for various fee cases. The lowest combined (combo) fees include 3% average GHG fees and mid-range HIP fees, the intermediate combined fees include 3% average GHG fees and high HIP fees, and the largest combined fees include 2.5% average GHG fees and high HIP fees.

Methane emissions are almost entirely upstream (see Figure 3.21). Differences in methane emissions between cases are small and mostly related to the changes in use of natural gas, because control options are not modeled for upstream methane emissions. In the high HIP fee case, methane emissions are actually 6% higher than in the base case; with mid range HIP fees they are 2% higher than in the base case. They are 6% less than in the base case with 2.5% average GHG fees, and 14% less with 3.5% 95th percentile fees.

Combined fees reduce total emissions of a pollutant more than either set of fees alone while using fewer control technologies, although the emissions from a particular sector may be slightly more. With combined fees, industrial NO_x controls are used less because more fuel switching occurs that reduces controllable emissions. Industrial sector NO_x emissions are slightly higher in combined fee cases compared to HIP fee cases. After

2020, combined fees reduce electric sector NO_x by at least 60% and up to 83%, which is more than with HIP fees alone. The lowest combined fee has larger reductions in SO_2 than either set of fees alone, 89-94% lower electric sector emission. The intermediate and largest combined fee cases have similar reductions to the high HIP fee case. In the combined fee cases, industrial SO_2 controls are used less because more emissions are reduced through fuel switching. The PM in the combined fee cases behaves similarly to the HIP fee cases, except that the upstream PM in the lowest combined fee case is reduced by about 70% after 2025, much more than with the mid-range HIP fees. The combined fee cases have slightly larger reductions of CO_2 than either set of fees alone. CCS is not used in any combined fee cases. With combined fees methane emissions are reduced as well, up to 5%, which is less than with GHG fees alone, but avoids the increase seen with only HIP fees.

3.3.2 Industrial sector technologies

In all cases in the industrial sector, natural gas and biomass-fired boilers are added to keep up with the increased demand over time. Electricity, LPG, and natural gas use increase to meet the additional process heat demand. Increased natural gas, biomass, and electricity use also help meet increased demand for other industrial energy needs, although energy products used for feedstock include an increase in oil use.

The representation of many of the energy uses in in the industrial sector in MARKAL are still fairly inflexible, despite the modifications made for this study, but the additions to the model discussed above allow for responses to the fees. We only discuss the boiler and process heat energy uses because there are not enough technology options in the rest of the industrial sector for changes to occur. The one exception is that the “other” energy use category has some flexibility; as HIP fees increase, the use of petroleum coke and coal decreases and the use of electricity and oil increases to maintain the same level of energy provided, see Figure 3.24 in the appendix.

For process heat (see Figure 3.22), the main change with fees is for the cases with mid or high HIP fees, where coal use is significantly less (none at all in some years with high HIP fees and at most 16% of the base use with high HIP fees or 63% with mid-range fees) and consequently the use of electricity is increased. The same effect is seen to a much smaller degree for the 3% 95th percentile GHG fee case (63-94% as much coal is used compared to the base case). The lowest combined fees use slightly more coal than the mid-range HIP fees, while both combined fee cases with high HIP fees use less coal than the high HIP case alone. The solar thermal technologies are used in all cases. Flat plate solar is used to the maximum extent allowed by the constraints in all cases including the base case as it is such an inexpensive technology. Concentrated solar heat is used to some extent in all cases, but with high HIP fees or any GHG fees the use of this technology is higher than in the base case. Due to the constraints, this represents a small fraction of the overall energy use. In cases where use of concentrated solar is greater, less LPG is used.

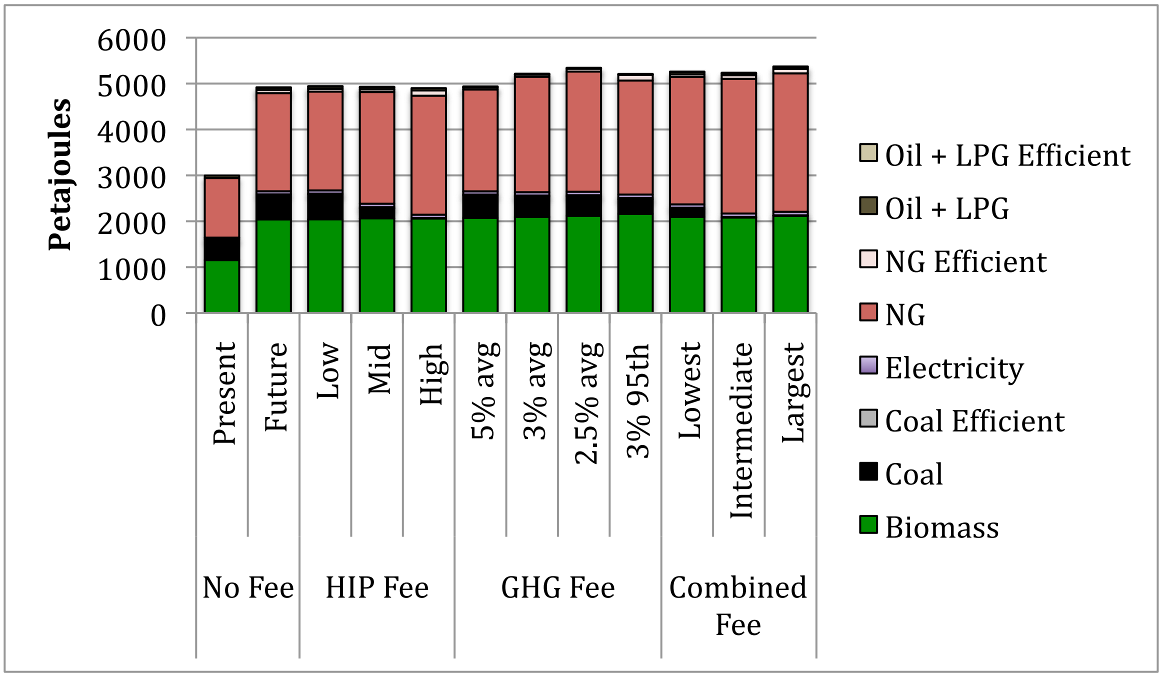

Figure 3.4 Fuel use and energy efficiency in industrial boilers in 2045.

The largest share of technology changes in response to fees in the industrial sector occurs with boilers, see Figure 3.4. As with process heat, coal use decreases with mid-range or high HIP fees and 3% 95th percentile GHG fees, compared with the other cases. Natural gas use increases as HIP fees increase. Efficient natural gas boilers are used more in the high HIP case in all years, while mid range HIP fees lead to earlier implementation with similar final levels. In the base case and low and mid HIP fee cases efficient LPG boilers are used for about half of all LPG boilers through 2040, after which their use trails off. With high HIP fees, the efficient LPG boilers are not used, but all LPG boilers are used less than half as much in these cases. The efficiency retrofits for coal boilers are used in low and mid HIP fee cases, for less than 10% of coal boilers. Up to 9% of the coal boilers are upgraded in the mid-range HIP case, so this case is building new, more efficient coal fired boilers as well as improving the existing ones. Natural gas use is higher with all GHG fees, although the natural gas use is lower for the 3% 95th percentile case than the 2.5% average case. In the

3% 95th percentile GHG fee case, efficient natural gas boilers are used more than in the base case in all years, while in the mid and high GHG fee cases the efficient natural gas boilers are used more in earlier years, but at a similar level to the base case after 2040. The boiler retrofits for LPG are used for just over half (57%) of the LPG boilers in all GHG fee cases. The new upgraded coal boilers are also used in all of the GHG fee cases, but always at less than 10% of all coal boilers. The use of boilers increases with GHG fees, but the use of fuels for unspecified industrial needs decreases in these cases so this is not an inefficiency, but a shift in energy use that is not apparent from the subset of results presented here. Natural gas boiler use is higher for the combined fee cases than for the corresponding single fee cases, but the efficient natural gas boilers are used to a similar extent in the combined and single fee cases. Some discussion of industrial control technologies occurs in the previous section on emissions, and an additional figure 3.23 and more detailed discussion of these technologies can be found in the appendix.

3.3.3 Electric sector technologies

Electricity generation and use increases over time in all cases. Most of the increase is met through an increase in generation with natural gas combined cycle (NGCC) technology. Wind generation increases sharply through 2035 although it remains small in comparison to natural gas. Solar, albeit a small percentage of generation, also increases throughout the time period. The investment cost of wind and solar is projected to come down over time as is the investment cost for the most efficient NGCC plants. Hydropower and nuclear generation remain quite constant across time and cases due to the hurdle rates in place to represent regulatory and other difficulties of constructing these new facilities combined with their large upfront costs. Investment in new hydroelectric capacity is not allowed in the model due to the difficulty of siting new facilities.

For most fees, the largest change in electricity generation technologies occurs for coal and natural gas, see Figure 3.5. By 2020, electricity generated from coal is 75% less than in the base case for high HIP fees, but for mid range HIP fees it is only 24% less. Much of this coal is displaced by NGCC. In cases with high HIP fees, some offshore wind is used, but this is less than 1% of total wind use. With 2.5% average GHG fees, coal is used 75% less than in the base case after 2030. With 3% average GHG fees, generation from coal is 60% less than in the base case. In the GHG fee cases, more renewable technologies are used, particularly wind, which is used approximately twice as much as in the base case in the 2.5% GHG fee case after 2025.

The electricity mix in the case with 3% 95th percentile GHG fees is much different from that in other cases. Although electricity from coal is only slightly (5%) lower than for 2.5% average GHG fees, electricity from natural gas is also lower than in all other cases by 2045. Although natural gas has lower carbon intensity than coal, CO_2 and methane are still released. Three and a half times more electricity is generated from wind power compared to the base case in 2045 (60-77% more than with 2.5% average GHG fees), and solar thermal use is much larger than in other cases (up to 10 times as much). Solar thermal use increases more than PV, which has a lower capital cost, because the capacity factor for solar thermal is higher due to the thermal energy storage built into this generation type. Biomass co-fired with coal is also much higher in this case than in the others, up to 33 times more than in the base case, although this falls off in later years as coal use decreases. This shift away from natural gas and towards a larger increase in renewables with large fees shows that the changes do not scale linearly with fees, as fees beyond a certain level lead to responses that are quite different from fees lower than that level.

Table 3.2. The cost of the fees for lifecycle emissions translated to $/kWh for two electricity generating technologies in 2045.

Taxed pollutant	Level	Coal	NGCC
GHG	5% avg	0.02	0.01
GHG	3% avg	0.07	0.03
GHG	2.5% avg	0.09	0.04
GHG	3% 95th	0.21	0.08
HIP	low	0.01	0.000
HIP	mid	0.05	0.001
HIP	high	0.18	0.002

The emissions fees are altering the economics of energy use in a complex system of energy prices. Natural gas prices are higher in the case with high HIP fees than the other cases, due to increased use of natural gas and correspondingly higher costs on the supply curve. Because coal has much higher emissions than natural gas, high fees can make coal the more expensive fuel to use to generate electricity. Even accounting for the low efficiency, the fuel and facility costs for existing coal plants are lower than for new natural gas plants; however, the fees on coal emissions with high fees outweigh the other costs of coal generation. Table 3.2 shows the additional cost due to fees for an existing coal fired power plant and a new NGCC plant assuming no additional control technologies, both operating in 2045, where it is obvious that the higher efficiency and lower emitting fuel used in NGCC contribute to much lower emission fees associated with that technology. See the SI for a comparison of these results with the Clean Power Plan, a discussion of CCS assumptions, and a comment on constraints on renewables.

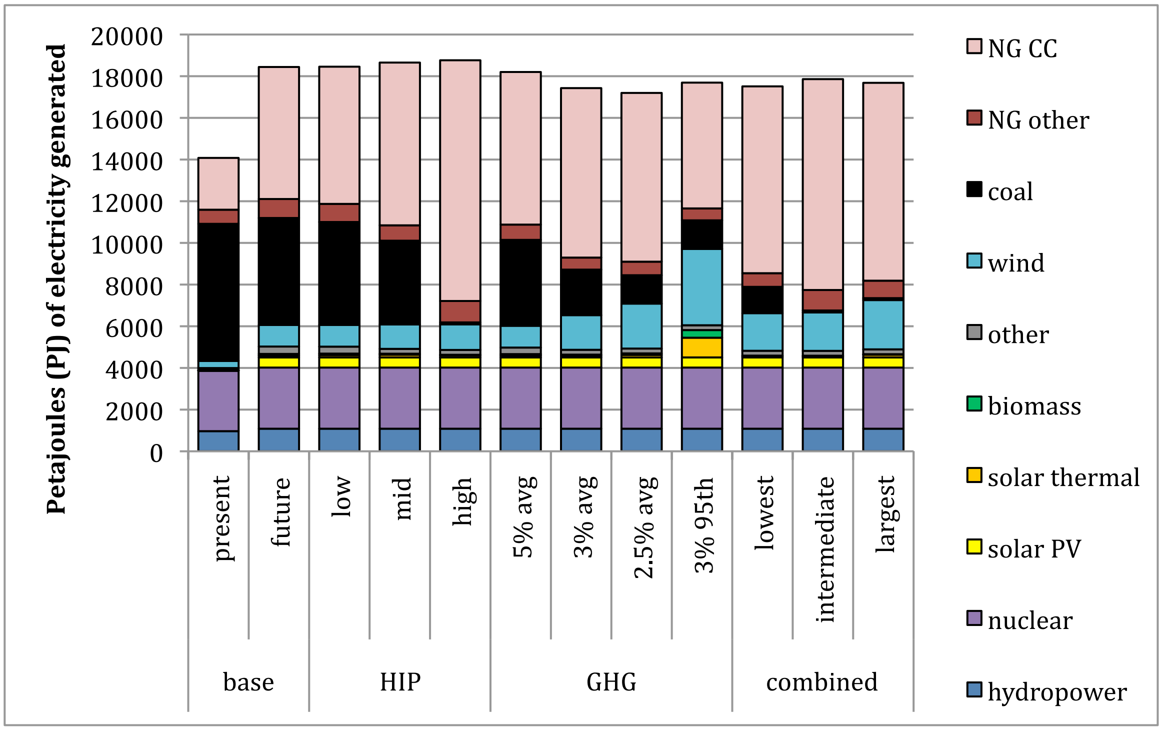

Figure 3.5 Fuels and technologies used to generate electricity. The "other" category includes oil, geothermal, and waste energy. The lowest combined (combo) fees include 3% average GHG fees and mid-range HIP fees, the intermediate combined fees include 3% average GHG fees and high HIP fees, and the largest combined fees include 2.5% average GHG fees and high HIP fees.

3.5. Discussion

All fee cases lead to emissions reductions, but the degree of reduction and the technologies used to achieve those reductions differs for different fees studied. HIP emissions generally decrease with time in all cases, including the base case, and decrease as HIP fees increase. The decrease in emissions over time is due to policies already in place to reduce emissions such as CAIR and CAFE. Many of the HIPs are reduced with GHG fees as well. Although HIPs are often reduced with GHG fees, the co-reductions tend to be less with 3% 95th percentile GHG fees than with 3% or 2.5% average GHG fees, because this case uses a different fuel mix than the lower GHG fee cases. Energy efficiency tends to be more important for cases with GHG fees. Less total electricity is produced in GHG fee cases than other cases, with most of the reduction in demand coming from the industrial sector followed by the residential sector.

Although the fees vary by sector, the sectors with the most emissions reductions tend to be driven more by the availability and price of technologies than the value of the damages across sectors. The fees are typically lowest for the electric sector, but this is the sector with the largest emissions reductions. This sector also has a wealth of technology options defined in the model that can be used to generate electricity. Many of the technologies are also less expensive than similar technology options in other sectors due to economies of scale. The highest fees are always in the transportation or residential sectors, two sectors that show very little response to fees. Figure 3.6 displays the fees collected in each case, and the transportation sector accounts for at least 30% of the fees collected in GHG fee cases, but there is almost no reduction in those emissions with fees. Similarly, with HIP fees the revenue collected from the residential sector comprise about 20% of the fees collected in those cases, with very little change in emissions across cases. In the model, there are fewer technology options in the transportation, residential, and commercial sectors; between upfront costs and high discount rates it is much less likely that alternative options will be chosen. The higher costs in these sectors are realistic, and the high discount rates used in these sectors represent observed behavior such as the hesitance or inability of individual or smaller scale consumers to spend money on large upfront investments and less familiar technologies (e.g., US EPA, 2013b). There is a possibility that additional technologies could be developed or that increased demand for options would bring down the cost of existing but expensive options in these sectors, for instance novel financing has increased residential PV installations (Coughlin and Cory, 2009). This is an example where the model might not fully describe the real world response to the fees, if the price or options of technologies in sectors with high fees were to change in response to the policy. The industrial and refinery sectors have costs that are higher than the electric sector, but lower than the other sectors and have responses in line with this explanation, lower reductions than the electric sector, but higher than other sectors. The upstream emissions are harder to target for reductions, because taking actions that lower overall emissions may sometimes increase upstream emissions.

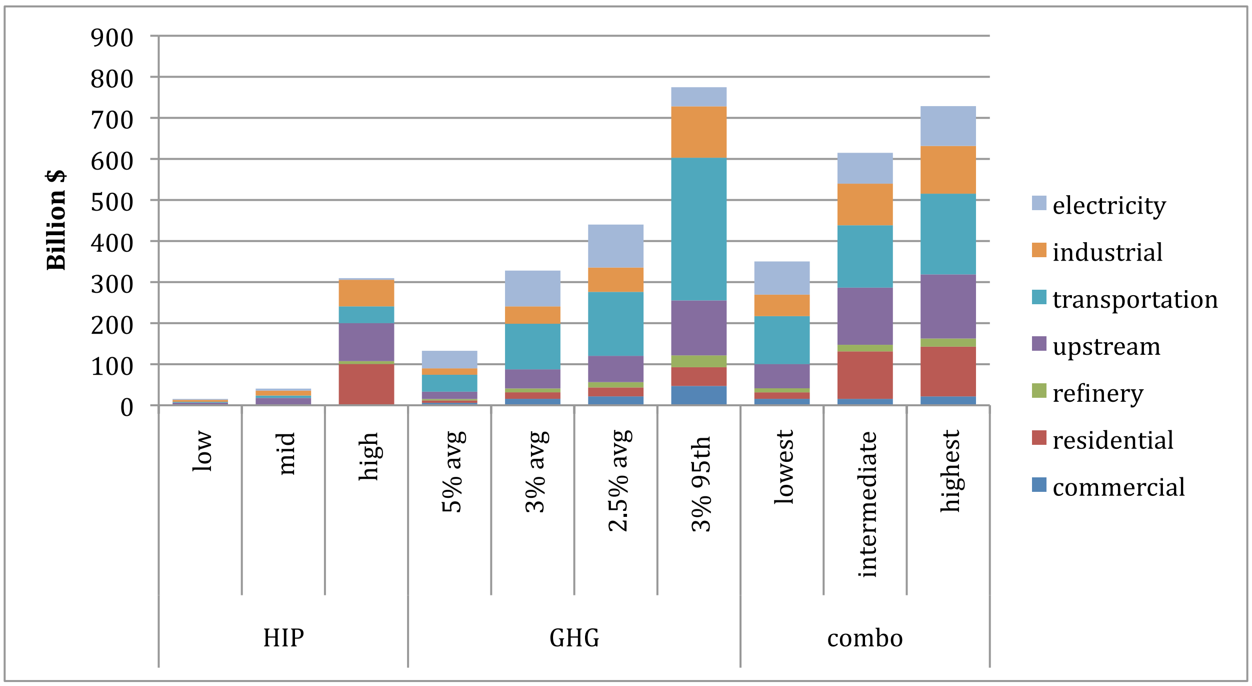

Figure 3.6. The fees collected by sector in each case in 2045 in year 2005 USD. The lowest combined (combo) fees include 3% average GHG fees and mid-range HIP fees, the intermediate combined fees include 3% average GHG fees and high HIP fees, and the largest combined fees include 2.5% average GHG fees and high HIP fees.

Compared with previous work (Brown et al., 2013), the fuels and technologies used to generate electricity are more likely to change with HIP fees. Some of this is likely due to changes in the cost of newer technologies compared to previous predictions. In particular, the investment cost and discount rate for natural gas EGUs are both lower than for previous model versions, and the natural gas supply is less expensive as well. One change from previous studies using MARKAL is that the investment costs and hurdle rates in the model have been re-evaluated based on factors such as readiness, public acceptability, and uncertainty about future policies so that new natural gas plants are more likely to be built than before.

As expected, fees reduce the targeted pollutant with larger reductions as the fees increase. For most of the fees, the non-targeted pollutants of interest are also increasingly reduced as fees increase. Although the 3% 95th percentile GHG fees lead to a much larger reduction in targeted emissions than the next largest fees (2.5%), there are actually more HIP emissions in this case than the 2.5% GHG fee case, see Figure 3.2. This is due to a shift away from natural gas and an increase in biomass combustion with large fees, which shows that the response to different levels of fees is more complicated than a simple scaling, where fees beyond a certain level lead to responses that are quite different from fees lower than that level. The use of CCS also decreases co-benefits because this technology greatly reduces the overall efficiency of the electricity generation system, which leads to an increase in pollutants other than CO_2 for that facility. This reversal of co-benefits for the highest GHG fees shows that it is important to consider all effects of interest for each policy proposal and not just assume that the relationship between emissions of different pollutants is linear.

In most cases the behavior of the combined fees is similar to that of the GHG or HIP specific fees. The emissions reduction is generally greater than or at least equal to that of the single component fee case with fees directed at that pollutant. The method of reducing emissions is sometimes different, however. Particularly when controls are used with HIP fees, more reductions are achieved through efficiency or fuel switching in combined fee cases because those typically reduce all emissions instead of a subset. For instance, fewer industrial controls are used in the combined fee cases, but more electricity is generated using wind so that total HIP emissions are reduced. The involvement of two different sectors in this change between the combined and HIP fee cases shows that there is interplay in terms of where emissions reductions occur, and that including more sectors leads to more options for emissions reduction. Because the emissions reductions are often larger in combined fee cases than the corresponding GHG and HIP fee cases (see Figure 3.7) and follow less targeted emissions reduction technology trajectories, coordinating GHG and HIP fee policies may allow for more economical reductions in emissions than two separate policies.

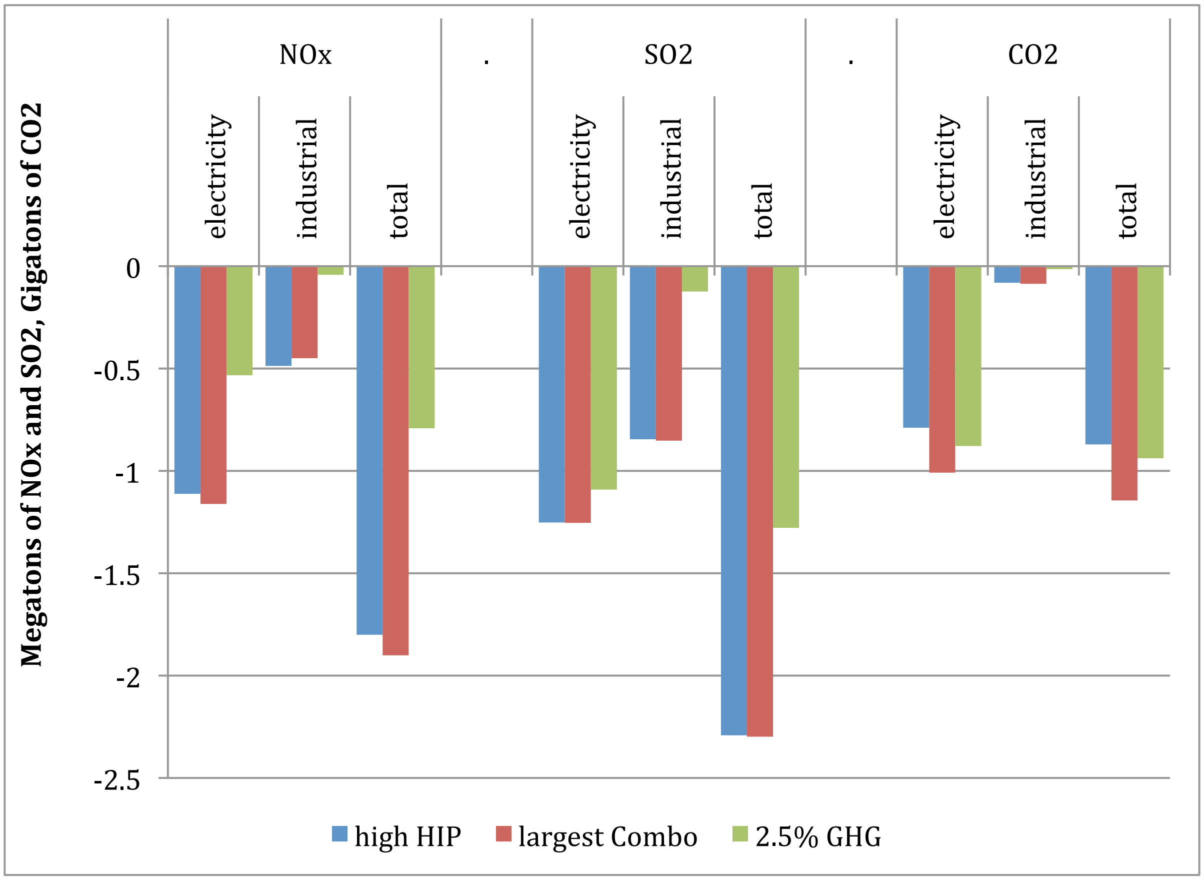

Figure 3.7 The reduction in emissions in 2045 compared to No Fee case emissions in 2045 for the electric and industrial sectors as well as total emissions reductions for that case.

The co-reduced emissions in Figure 3.7 present an interesting story. Total emissions reductions in the combined fee cases are larger than for the component fee cases, as the largest combined fee case (in red) includes both high HIP fees and 2.5% GHG fees. The SO_2 reductions in the combined fee case are nearly identical to the HIP fee case and the NO_x reductions are only 5% more with combined fees than HIP fees alone. The CO_2 emissions are reduced an additional 22% compared to the GHG fee case with combined fees. This shows that while combined fees lead to additional reductions for both categories of pollutants concerned, the GHG benefit is larger. The co-benefit of individual fees is also of interest. While the co-benefits of HIP fees are total CO_2 reductions that are 93% of those achieved with GHG fees alone, the total NO_x and SO_2 emissions reductions from GHG fees are only 44% and 56% of those achieved by HIP fees, respectively. Although this finding is sensitive to which two sets of fees are compared, the high HIP fees lead to similar or larger reductions in CO_2 emissions than most of the GHG fee cases, while NO_x and SO_2 reductions with any GHG fees are smaller than all but the low HIP fee case. Damage based emissions fees lead to reduced emissions, but the degree of reduction and pollutants reduced depends on the specific fees. Targeting all pollutants of interest will ensure those pollutants are reduced, whereas optimal emissions levels may not be reached if policies rely on co-benefits to produce emissions reductions for some species.

3.6 Appendix

3.6.1 Social Cost of Carbon

Table 3.3 GHG damages used as fees, from the social cost of carbon (Interagency Working Group on Social Cost of Carbon, 2013).

	2005M$/Mt CO2				2005M$/kt CH4			
year	5% avg	3% avg	2.5% avg	3% 95th	5% avg	3% avg	2.5% avg	3% 95th
2015	11	36	55	103	0.32	1.0	1.5	2.9
2020	11	40	61	121	0.32	1.1	1.7	3.4
2025	13	45	66	136	0.37	1.3	1.8	3.8
2030	15	49	72	150	0.42	1.4	2.0	4.2
2035	18	54	76	166	0.50	1.5	2.1	4.6
2040	20	58	82	181	0.55	1.6	2.3	5.1
2045	23	62	87	194	0.63	1.7	2.4	5.4
2050	25	67	92	208	0.71	1.9	2.6	5.8
2055	25	67	92	208	0.71	1.9	2.6	5.8

3.6.2 Effect of Database Changes

The base case used here is compared to the database released by the EPA to show the effect of the changes described in section 3.7.

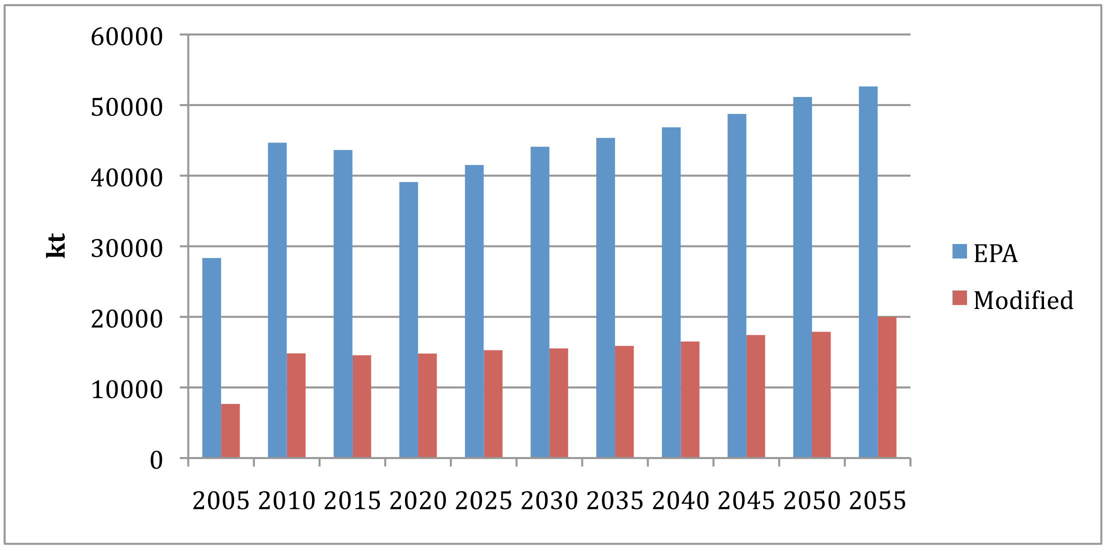

Figure 3.8 Methane emissions in the base case using two database versions.
The reason that the methane emissions (Figure 3.8) are so much higher in the EPA database is that very large values were used for upstream coal methane.

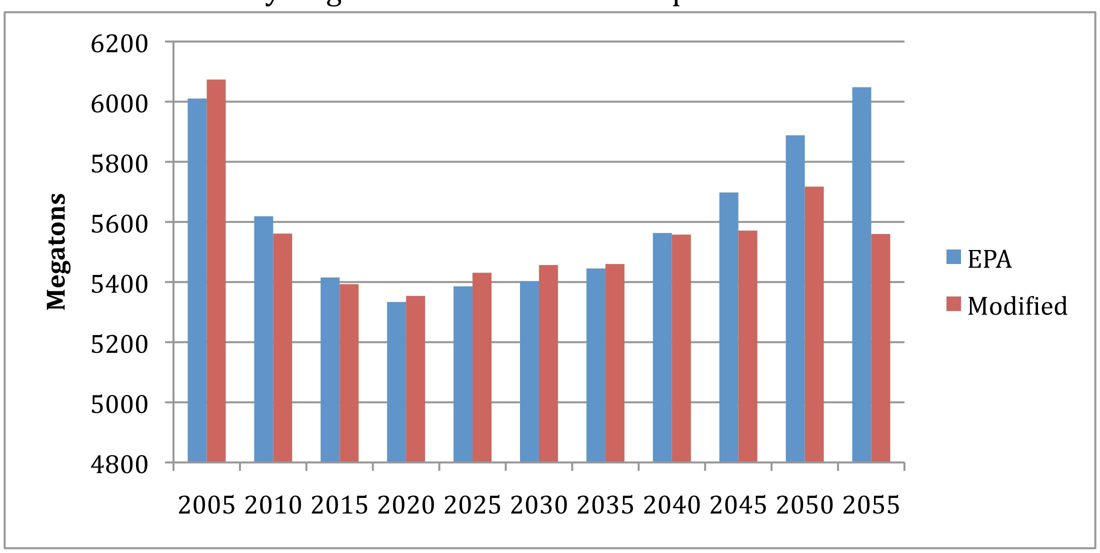

Figure 3.9 CO_2 emissions in the base case using two different database versions.

Where CO_2 emissions (Figure 3.9) are larger in the modified base case, this is due to the lifecycle emissions added. In later years, the coal retirement constraints affect the modified case and are the reason for the lower emissions in those years.

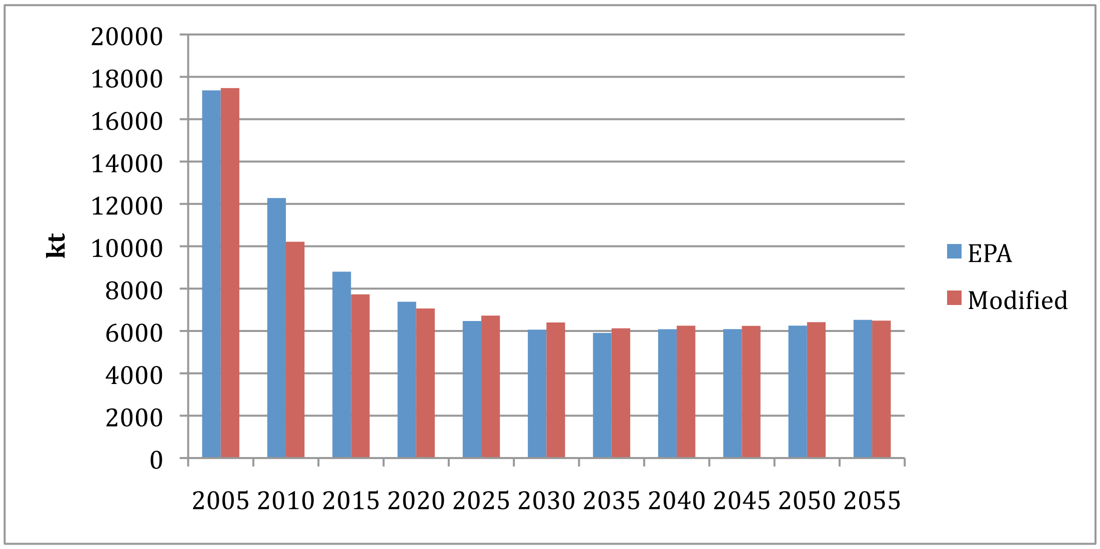

Figure 3.10 NO_x emissions in the base case using two different database versions.

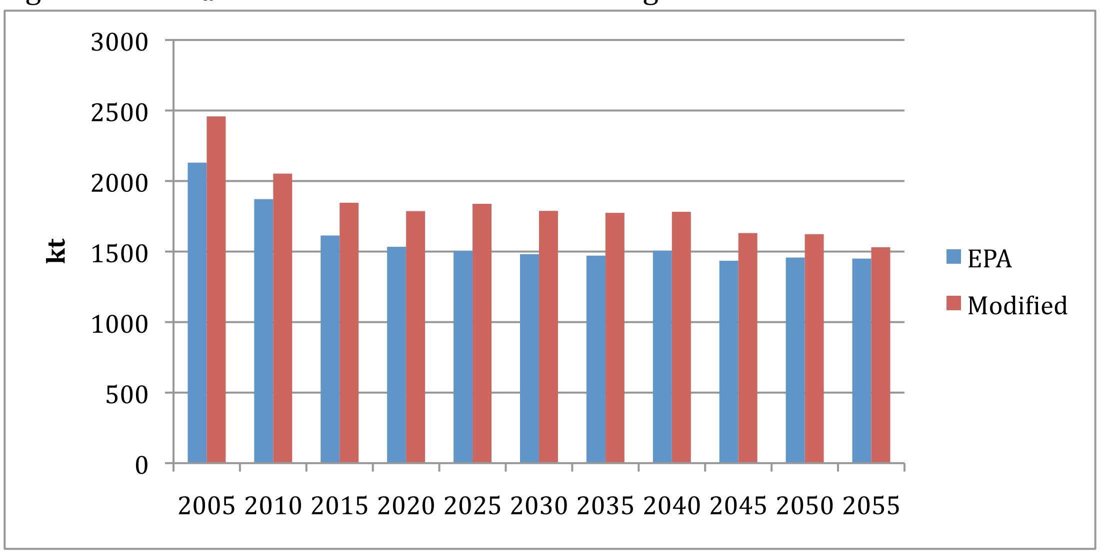

Figure 3.11 $PM_{2.5}$ emissions in the base case using two different database versions.

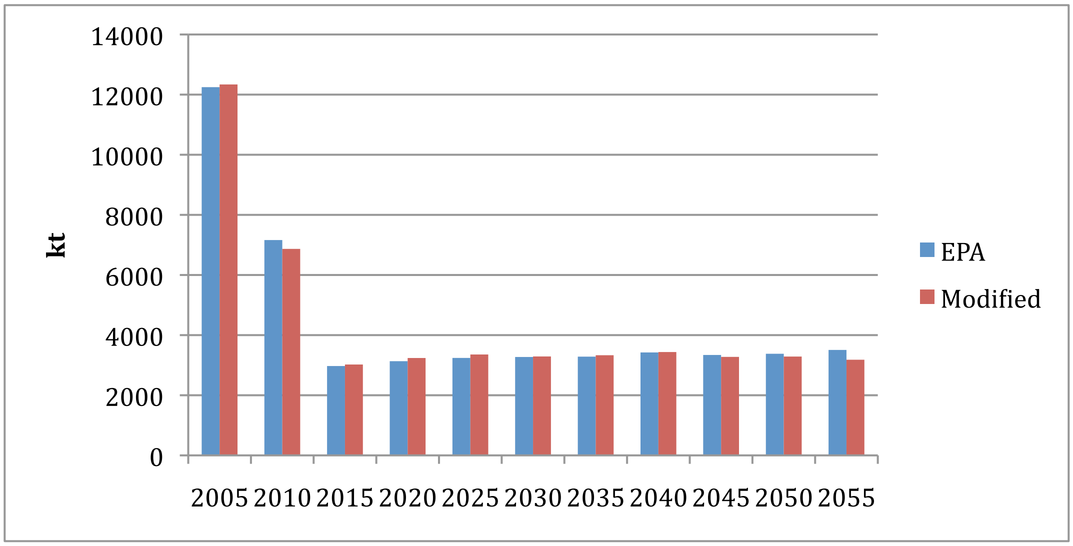

Figure 3.12 SO_2 emissions in the base case using two different database versions.

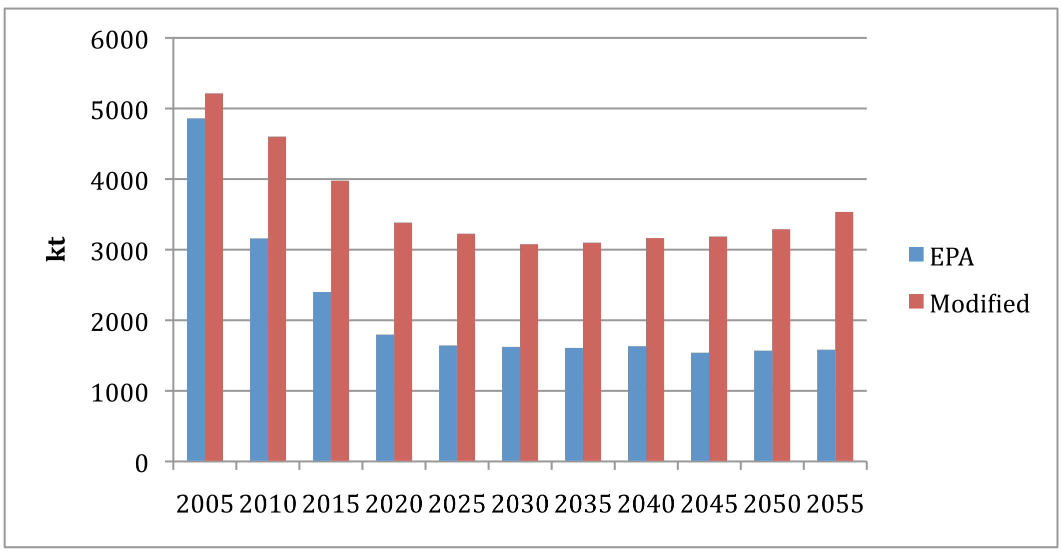

Figure 3.13 VOC emissions in the base case using two different database versions.

The HIP emissions (Figures 3.10-3.13) are higher in our base case because of the additional life cycle emissions tracking that was not considered in the EPA base case.

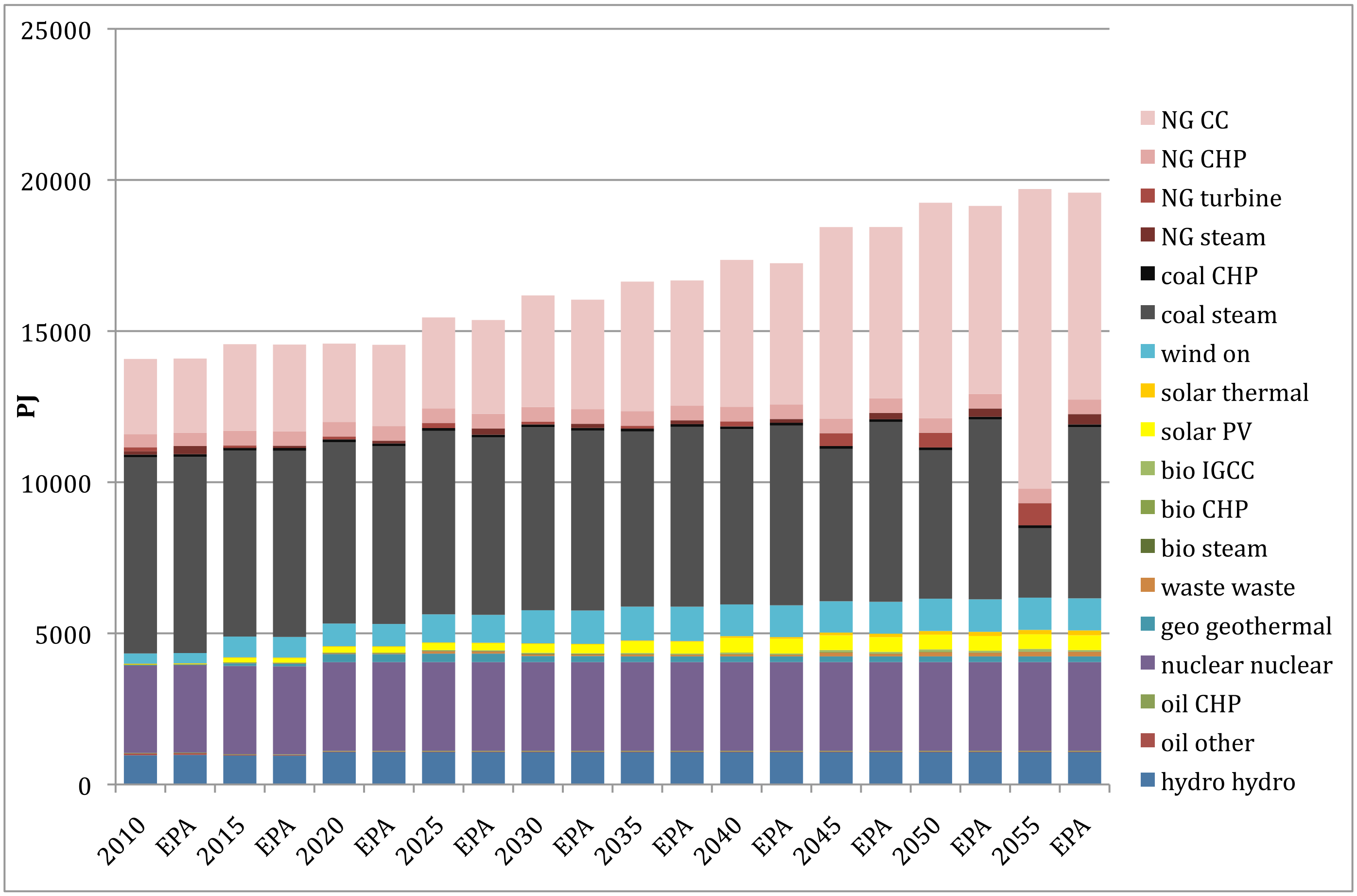

Figure 3.14 Electricity generation by fuel type in the base case using two different database versions, the EPA release and the modified version used here. The left column for each year presents the results from the modified database, and the right column presents the EPA database results for the same year.

The largest change in the electricity use (Figure 3.14) is the requirement for older coal fired power plants to be retired in our case.

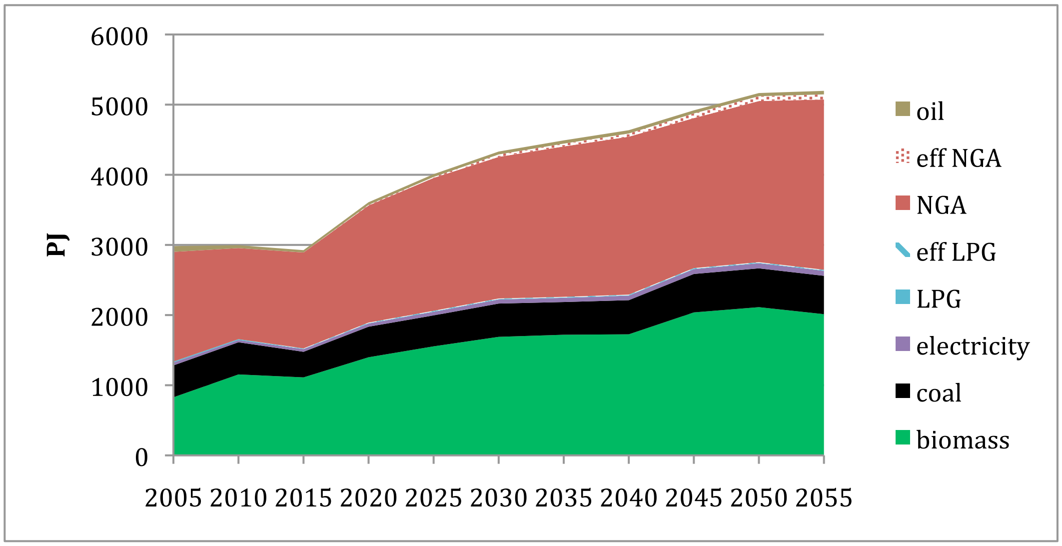

Figure 3.15 Industrial boilers in the modified base case used in this analysis.

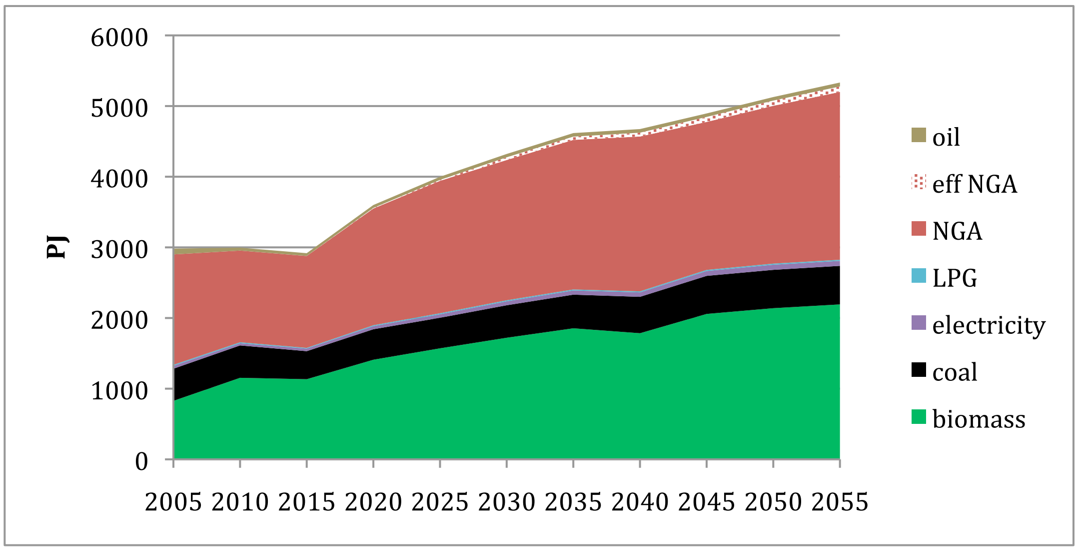

Figure 3.16 Industrial boilers in the EPA released base case.

There are additional efficiency improvements for boilers that are used even in our base run (Figure 3.15-3.16). The EPA base case only has a more efficient option for natural gas. The industrial solar process heat technologies used are discussed in the main paper. Since there are no comparable technologies in the EPA base case a further comparison is not shown here.

3.6.3 Additional Results

3.6.3.1 Emissions

Here we present emissions results from all fee cases, for comparison of the effect of fees across the range of damage estimates considered. Some of these results are discussed in the main paper.

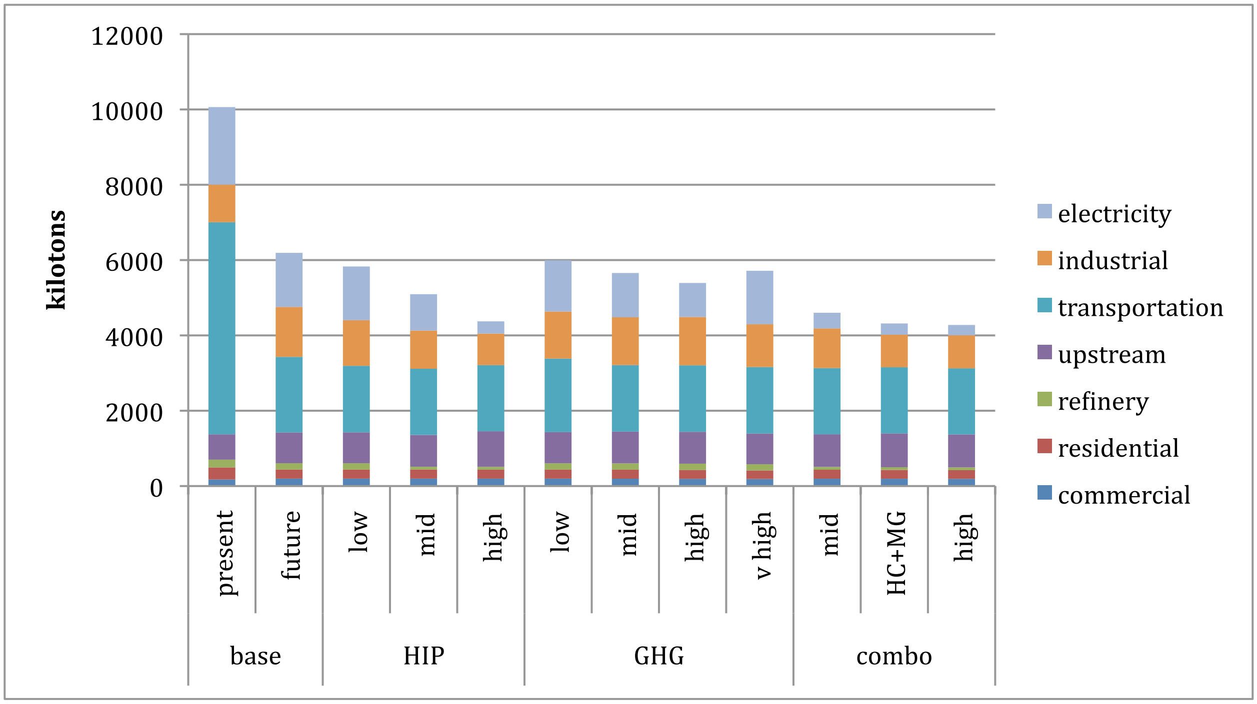

Figure 3.17 NO_x emissions in 2045 for all cases and in 2010 for the base case.

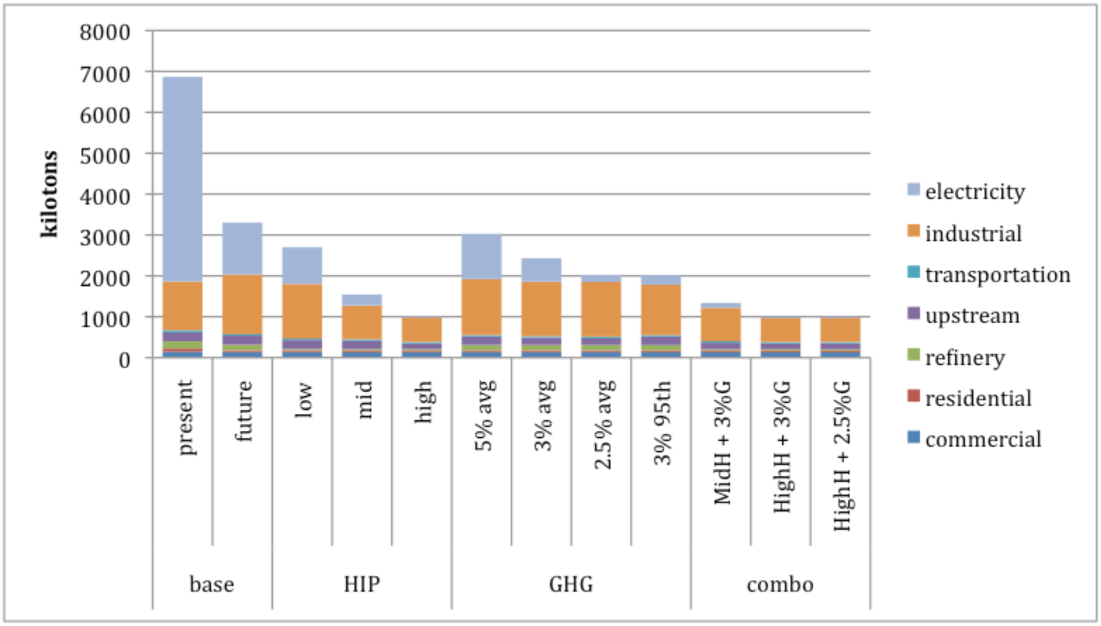

Figure 3.18 SO_2 emissions in 2045 for all cases and in 2010 for the base case.

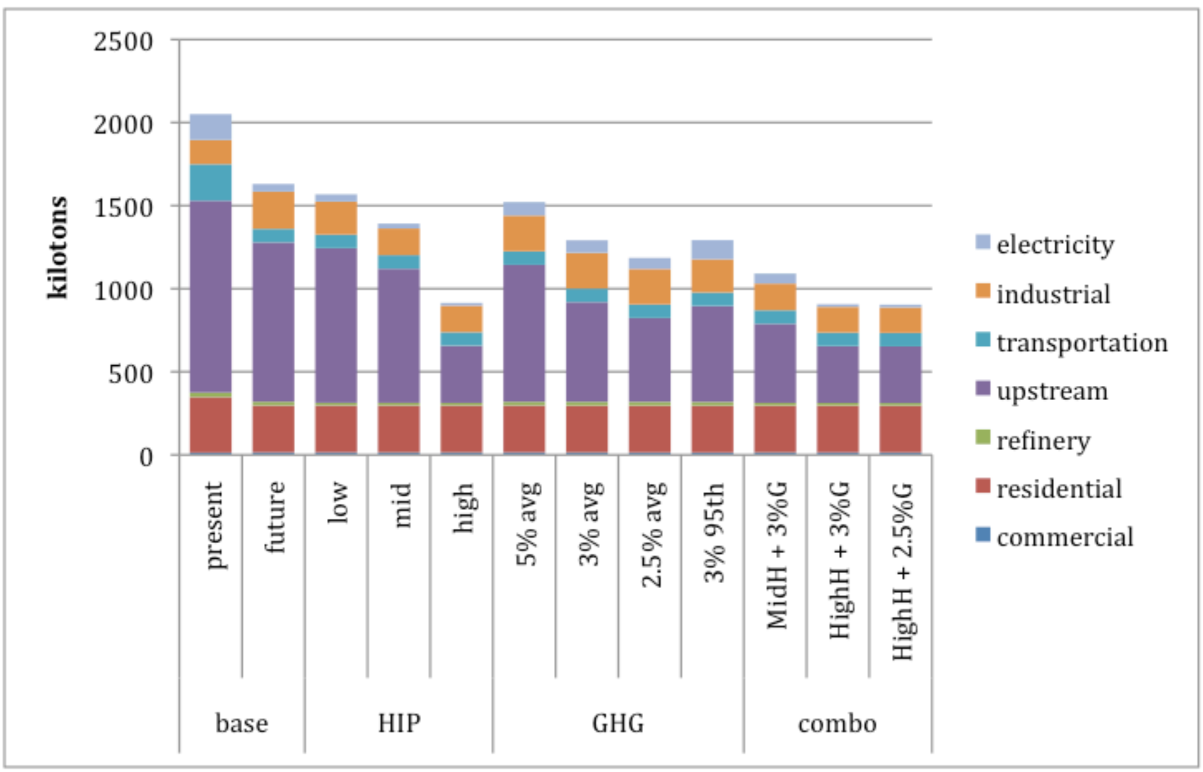

Figure 3.19 $PM_{2.5}$ emissions in 2045 for all cases and in 2010 for the base case.

Figure 3.20 VOC emissions in 2045 for all cases and in 2010 for the base case.

As the only sector with VOC specific controls, refinery emissions are an important source of reductions for VOCs. In all cases with mid or high HIP fees, VOCs from refineries are reduced 8-11% compared to the base case. Industrial VOCs are fairly similar across cases, sometimes higher or lower than the base case, but the low HIP fee case does have higher emissions than the other cases, up to 14% more than the base case. Electric sector VOC emissions are reduced with all fees.

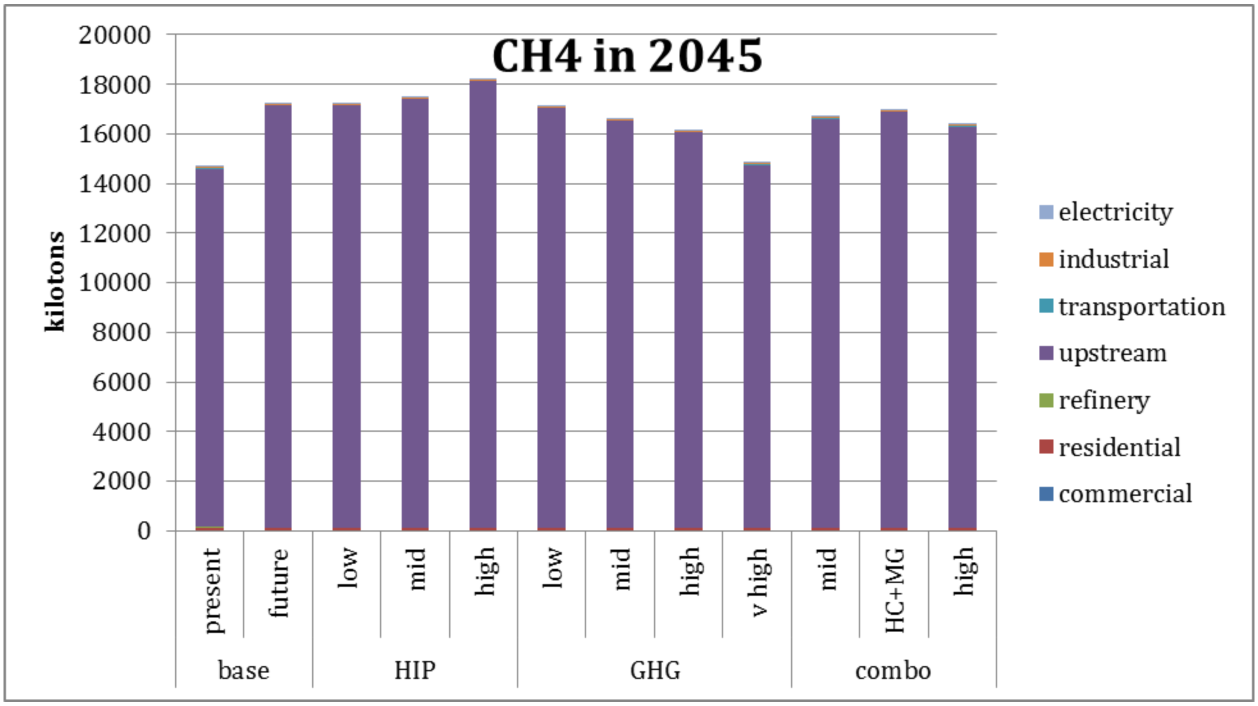

Figure 3.21 CH_4 emissions in 2045 for all cases and in 2010 for the base case.

3.6.3.2 Transportation, Residential and Commercial

The transportation sector is not very responsive to the policies modeled here. The policies modeled here may not be the best ones to affect the transportation system. High hurdle rates in the transportation sector, designed to represent the consumer's hesitance to buy less familiar technologies or to spend more on the initial investment, mean that the initial cost of lower emitting technologies is weighed more heavily than later savings on fuel or emissions fees. The future vehicle technology options are also held constant across all cases in this model.

The commercial sector fuel use also shows little change. There is slightly less natural gas used with GHG fees, up to 8% less than the base case with very high GHG fees. This is due to use of more efficient devices in this sector and not a shift between fuels. In the residential sector, less electricity is used with GHG fees, due to the use of more efficient devices. With very high GHG fees the use of electricity for space heating is closer to the HIP fee cases to offset the decrease in natural gas use.

3.6.3.3 Industrial

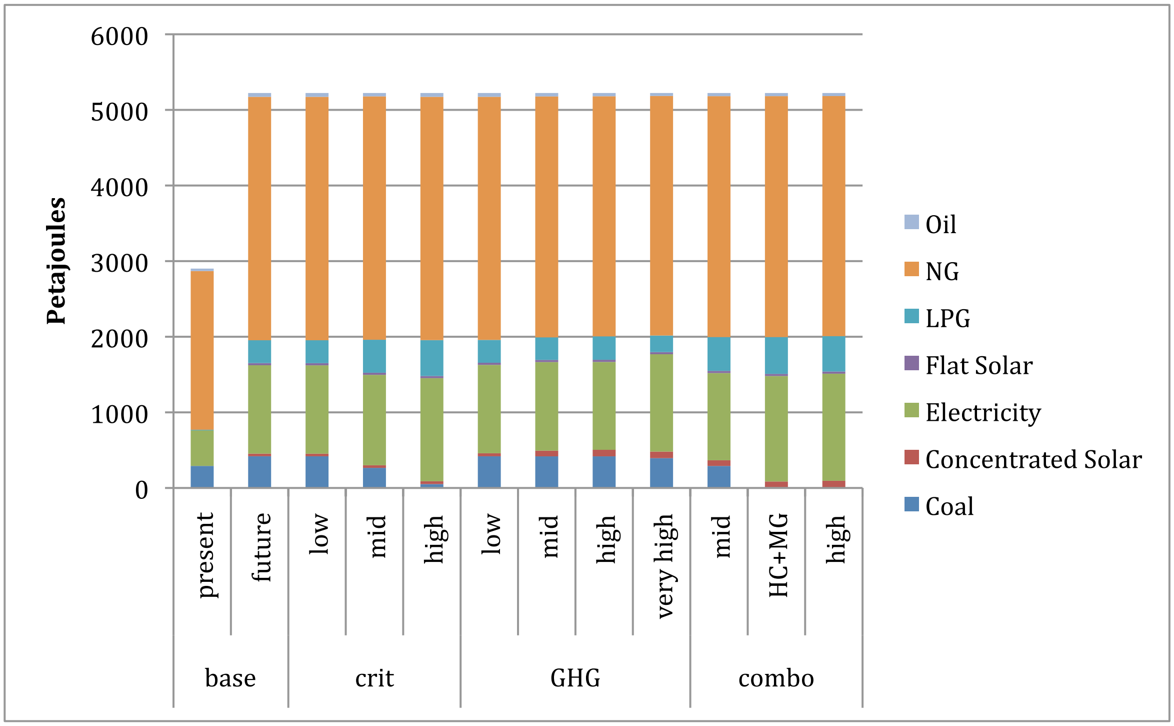

Figure 3.22 Industrial process heat fuel use in 2045 for all cases and in 2010 for the base case.

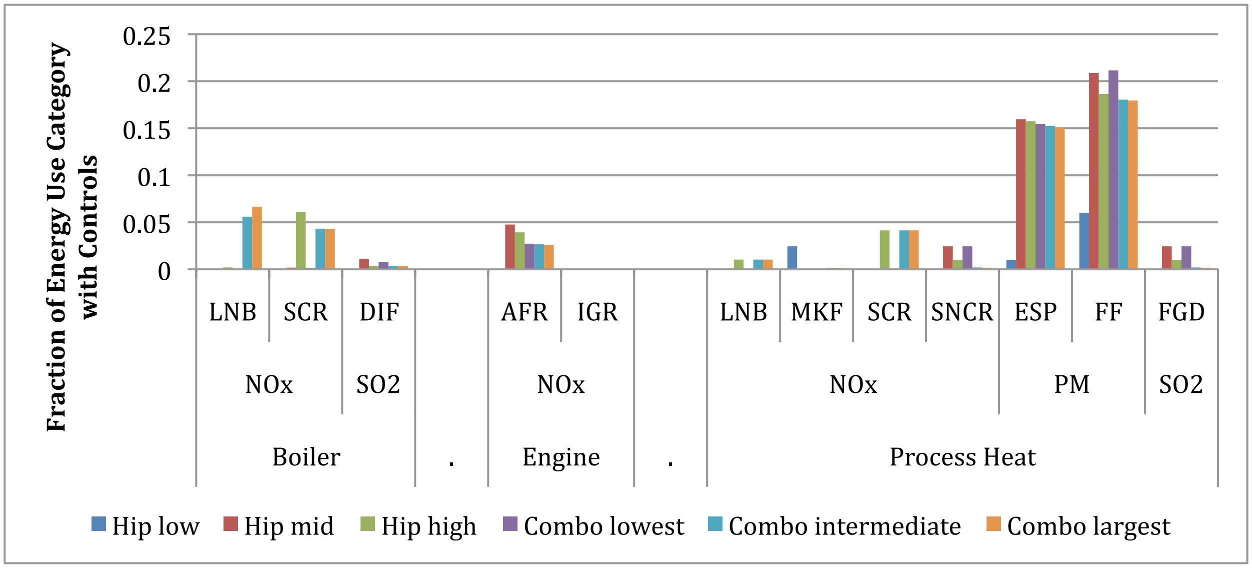

Figure 3.23 The control technologies used in the industrial sector as a fraction of the energy use category that the specified control is applied to. LNB refers to Low NOx Burners, SCR refers to Selective Catalytic Reduction, DIF refers to a dry injection/fabric filter system, AFR refers to adjusting the air to fuel ratio, IGR means retarding the engine, MKF refers to mid kiln firing, ESP is an electrostatic precipitator, FF is a fabric filter, and FGD is flue gas desulfurization.

With mid and high HIP fees, SO_2 controls are used for process heat and boiler emissions. In the combined fee cases the controls are used less because emissions are reduced through fuel switching. NO_x boiler controls are used with mid range and high HIP fees. With high fees two types of controls are used, while only one is used with mid range fees. Again there is more fuel switching in the combined fee cases, which leads to slightly lower levels of control technology use, but the controllable emissions are similar across fee levels. NO_x controls are also used for process heat. With low HIP fees, the mid-kiln firing option is used, but it is not used significantly in the other cases. With mid range fees (alone or combined), SNCR (Selective Non-Catalytic Reduction) is used to remove about 1/3 of those emissions. With high fees, LNB (low NO_x burners) and SNCR used. The SCR (Selective Catalyic Reduction) use is similar in all cases, but the SNCR use is lower in the combined cases where the emissions are lower. PM is reduced through the use of both ESP

(Electrostatic Precipitator) and fabric filters. In the low fee case most of the reductions come from fabric filters, but as the fees increase fabric filters are used less and ESP is used more, combined with a decrease in controllable emissions as the fees increase. This trend follows for combined fees as well.

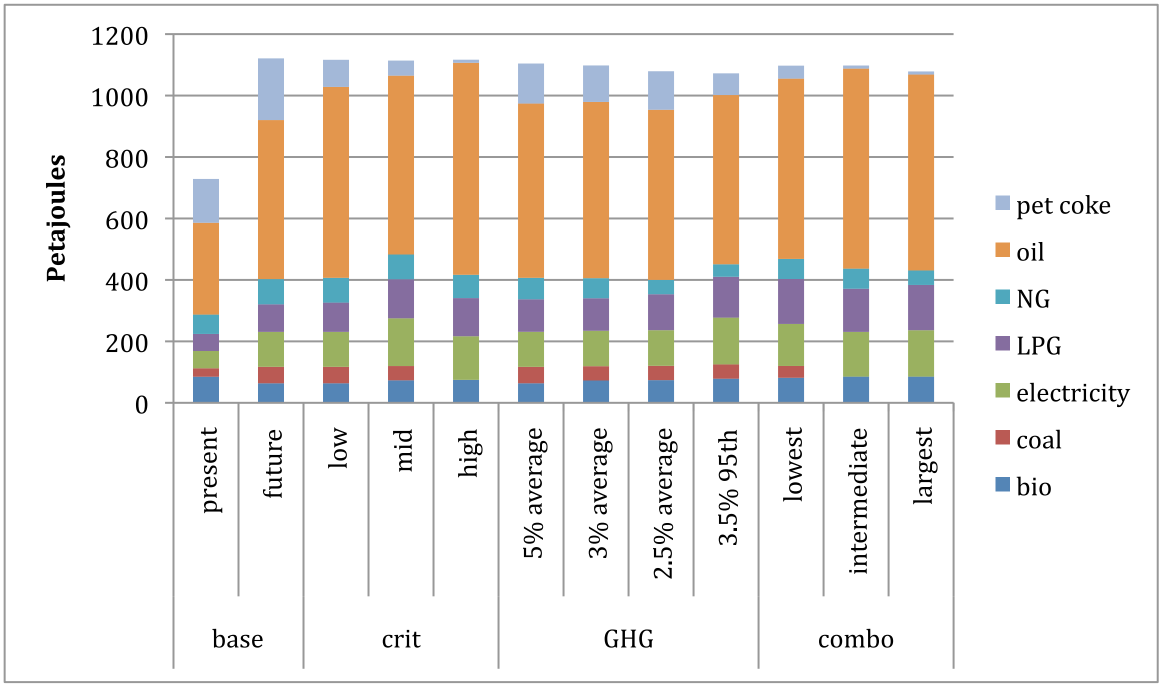

Figure 3.24 Fuel use for “other” industrial energy needs for all cases in 2045 and for the base case in 2010.

3.6.3.4 Electricity Generation

In the electric sector, controls were applied in response to fees. Fabric filters were applied to reduce PM emissions. FGD (Flue Gas Desulfurization) scrubbers are used to remove SO_2. As the fees on HIP increase, fewer emissions are removed through control technologies, because these technologies are only available for generation from coal, which decreases as the fees increase. One exception to this trend is that SCR is used more in the mid range HIP fee case than the low fee case. This is because the low fee case uses more of the less efficient SNCR technology. Also, the combined fee cases use fewer controls than the HIP fee only cases for the same reason.

3.6.4 Sensitivity analysis

3.6.4.1 Clean Power Plan

Given that there is uncertainty in how states will choose to follow the Clean Power Plan (CPP), there is uncertainty in these emissions reductions. There is more uncertainty in the changes to the energy sector, so we will not compare technologies expected under the plan to the results presented here, only the changes in emissions. We compare changes in emissions from the electric sector alone, as well as the total energy system emissions reductions. The CPP projects emissions reductions only from the electric sector as it is a more targeted policy. The electric sector CO_2 emissions in the policy cases presented here are lower than projected in the RIA for the CPP (US EPA, 2015) for 2025 and 2030 for all cases except low and mid HIP fees and 5% GHG fees. When CO_2 reductions from all sectors are considered, only the low HIP fee case has lower emissions reductions than the CPP. All of the GHG cases and the low HIP fee case have larger electric sector NO_x emission than the clean power plan, and only in the 2.5% GHG fee case of those do the emissions reductions from other sectors lead to larger total NO_x reductions than the CPP. The SO_2 emissions from the electric sector are lower than than CPP in all cases but low HIP fees and 5% GHG fees. When reductions from other sectors are counted, the reductions are larger than the CPP in all cases but the 5% GHG fees.

The CPP aims to reduce CO_2 emissions from electricity generation by 32% from 2005 levels by 2030. A similar reduction could be achieved by a fee level somewhere between the 5% average and 3% average GHG fees, which reduce CO_2 from the electric sector by 23 and 42% from 2005 levels in 2030, respectively. Similar CO_2 reductions are also reached in the mid and high HIP fee cases, which reduce CO_2 from electricity generation by 21 and 46% from 2005 levels in 2030, respectively. This provides further evidence that the plan should be able to achieve the large co-reductions of pollutants promised as well as that the reduction in CO_2 can be achieved economically.

3.6.4.2 CCS cost assumptions

The MARKAL database used here has a lower CO_2 removal efficiency from biomass IGCC with CCS. The EPA database removes 412 kt CO2/PJ biomass IGCC with CCS used, while the modified database removes 142 kt CO2/PJ biomass IGCC with CCS. When the EPA database CO_2 removal efficiency is used it mostly changes the 3% 95th percentile GHG fee case. The 2.5% GHG fee case with the original biomass CCS tech uses that technology to remove 60 kt of CO_2 in 2030 and beyond. With 3% 95th percentile GHG fees about 1800 PJ of electricity is generated this way from 2030 on, but close to 800 megatons of CO_2 is sequestered from the technology. When biomass IGCC with CCS is used with 3% 95th percentile GHG fees the electric sector has a negative contribution to CO_2 emissions in 2025 and beyond, and less electricity is generated from other biomass technologies, coal, NGCC, solar thermal, and wind. Sarica et al (2013) modified the biomass representation in MARKAL beyond the scope of this paper, and found that these changes are important when studying a renewable fuel standard, but that with coal and biomass cofiring the biomass and energy prices are similar. The use of biofuels is typically small in the cases presented here.

3.6.4.3 Wind and solar cost and constraints

The installation and fixed O&M cost of utility scale solar PV was lowered to better match the values from the NREL/Black and Veatch (Black and Veatch, 2012) report. The constraints on the use of wind are binding in the 3% 95th percentile GHG fee case, so if there is more capacity for wind in that case even more wind might be used. In all cases, the assumptions for the availability of the best wind or solar resource may have an effect on how much of these renewable technologies are used.

3.7 Changes Documentation

Updates to MARKAL from base of EPAUS9r_2014v1.1

Downloaded on December 19, 2014 from the Environmental Science Connector.

References to "EPA database" refer to this version

To incorporate these changes:

- The .fil file needs to be imported to Answer to make some constraints work.
- The relevant updated workbooks need to be imported.

3.7.1 Emissions Updates

3.7.1.1 Sector specific classification

When "emissions types" are added, the existing emissions values are used to create an additional subset of emissions so that sector specific emissions can be tracked and fees can be applied to specific sectors. This does not change the overall emissions, but rather allows for more targeted fees and more thorough analysis. Although sector-specific pollutant tracking already existed, the revised database tracks a larger set of pollutants, including $PM_{2.5}$ and VOC emissions as well as additional sectors. The pollutants treated in this manner are NO_x, PM_{10}, $PM_{2.5}$, VOC, SO_2, CO_2, and CH_4. The sectors considered (and corresponding workbooks affected) are:

E = electricity = ELC

F = 'fuels' = REF, H2, UNCNV

I = industrial = IND

T = transportation = TRN_LDV, TRN_HDV, TRN_OH

C = commercial = COM

R = residential = RES

3.7.1.2 Upstream emissions

Some upstream emissions are accounted for in the EPA database, athough they are not explicitly defined as upstream. Emissions associated with extraction of fossil fuels are considered, but other upstream emissions are not included in the EPA database. The modified database has an expanded treatment of upstream emissions to ensure that decisions in the model consider all emissions associated with a technology. Using literature sources and emissions values already in the EPA database, upstream emissions are explicitly accounted for in this version of MARKAL. Where the EPA database did not account for these upstream emissions, a literature search was performed to add in these emissions values. This allows upstream emissions to be tracked separately and for fees to be applied to that emissions category separately. Upstream emissions are defined for coal, natural gas, and oil extraction, biomass production, and the industrial, electric power, residential, and commercial sectors for technologies that do not have upstream representation in the model, such as renewable technologies without a fuel extraction pathway. The last four scenarios have upstream emissions mostly for renewable generation technologies where there is not a fuel pathway where the accounting can be done.

Table 3.4 The upstream emissions added by technology and pollutant. All values are in kt/PJ and values with a grey background are CO2 equivalents because separate CH4 and CO2 values were not available. This table does not include upstream emissions that are included in the EPA database, but does include those from these sources: (America's Energy Future Panel on Electricity from Renewable Resources and National Research Council, 2010; Arvesen and Hertwich, 2012; Burkhardt et al., 2012; Kaplan et al., 2009; Koroneos and Nanaki, 2012; New Energy Externalities Development for Sustainability, 2008; Santoyo-Castelazo et al., 2011; Solli and Reenaas, 2009; Wang et al., 2012).

literature source	technology	CO2	CH4	NOx	PM10	PM2.5	SO2	VOC
America's Energy Future Panel on Electricity from Renewable Resources 2010	PV	12.222		0.029	0.036		0.061	
	hydropower	5.278		0.010	0.014		0.008	
	wind	2.778		0.008	0.003		0.010	
	nuclear	8.333		0.011	0.002		0.008	
Burkhardt--2012r	CSP	7.784		0.023	0.007		0.019	
Kaplan 09	landfill gas--ICE	10.000	13.000	0.223	0.039		0.055	
	landfill gas--turbines	7.528	9.786	0.168	0.030		0.041	
	MSW	29.556		0.077	0.003		0.012	
Santoyo-Castelazo 2011	geothermal	36.111	0.006	0.006	0.008		0.753	0.001
NEEDs 2008	industrial PTC	3.611	0.010	0.017	0.003	0.001	0.004	0.004
Solli 2009	residential wood	7.222	0.583	0.153	0.000	0.611	0.022	0.694
Koroneos 2012	flat plate solar	0.020	0.000	0.000			0.000	
Wang et al. 2012	corn	0.086	0.142	0.160	0.009	0.008	0.024	0.021
	stover/ag residue	-0.008	0.078	0.092	0.005	0.005	0.013	0.012
Arvesen & Hertwich 2012	offshore wind	3.472	0.009	0.010	0.005		0.010	0.003
		CO2-e						

The values for upstream emissions that were added to the database are shown in Table 3.4. Note that for two of the literature sources, the upstream GHG emissions are only given in terms of CO_2-e and not separate CO_2 and methane (CH_4) values. All values in this table are in kt/PJ. Also the values shown for (Kaplan et al., 2009) are full life cycle values, but the added upstream values are the table values less the existing combustion emissions values. For more on the biomass emissions see below.

Other emissions changes include a change in the emissions of upstream coal methane to 0.152 kt/PJ (Venkatesh, 2011), and a modification of upstream natural gas values based on McLeod (2014). The upstream emissions considered here only consider emissions from fuel extraction, processing and delivery. This would include the fuel pathway of gasoline, but not the vehicle that uses it. Production of a solar collector is considered as part of the fuel related emissions because these emissions are considered to be similar to extraction, whereas the manufacture of a vehicle is related to the energy use, not the collection of energy. Upstream emissions for the construction of coal and natural gas combustion facilities are not included.

3.7.1.2.1 General Procedure

Upstream emissions are accounted for in the upstream fuel pathway, where one exists. The EPA database has some existing upstream emissions, so where those emissions are already present, the same values were used in a duplicate emissions parameter ending in *U to denote upstream emissions. For renewable generation technologies where there is not a fuel pathway where the accounting can be done, the upstream emissions are accounted for in the definition of the energy conversion technology. Also, for these technologies the specific technology can be more important in determining the upstream emissions as the construction of the technology (for example the construction of a PV panel versus a parabolic trough) is the major source of the upstream emissions and can vary quite a bit for different technologies using the same energy source. When new emissions values were added, the emissions were added to the upstream accounting and the overall emissions accounting to ensure that fees are applied correctly in all cases. In general, the added upstream emissions are related to renewable and unconventional technologies or less common fuels that had less robust upstream characterization in the existing model. The added upstream emissions values and the literature sources are presented in the table above.

The upstream emissions considered here only consider emissions from fuel extraction, processing and delivery. This would include the fuel pathway of gasoline, but not the vehicle that uses it. Production of a solar collector is considered as part of the fuel related emissions because these emissions are considered to be similar to extraction, whereas the manufacture of a vehicle is related to the energy use, not the collection of energy. The construction of coal and natural gas combustion facilities is not included in the upstream emissions values.

3.7.1.2.2 Coal

For coal we use the existing emissions in MARKAL, with one change from the 2014 v1.1 release. We are using the value 0.152 kt/PJ for methane emissions from coal extraction across the board. This number is taken from "Uncertainty in Life Cycle GHG Emissions from US Coal" (Venkatesh, 2011) as recommended in an email.[3]

3.7.1.2.3 Geothermal

Although the values used are not a perfect representation of all sources, these emissions can vary among geologic formations and the Santoyo-Castelazo et al.(2011) values were used to represent that there are associated upstream emissions. Due to the coarse spatial resolution in MARKAL, it is not the correct tool to choose which geothermal locations would be best choice to both meet electricity demand and reduce emissions.

[3] Personal correspondence with Carol Lenox May 28, 2015

3.7.1.2.4 Natural Gas

The emissions for natural gas extraction have been updated compared to the EPA database release. Another researcher working with US 9 region MARKAL has altered the natural gas extraction emissions to better represent the differences between shale gas and conventional resource extraction (McLeod, 2014), and those changes are incorporated into the database used here.

The EPA 9-region database has only a single pathway for natural gas and does not separate shale and conventional resources. The modified natural gas representation estimates shale gas as a fraction of total natural gas production by region based on projections from IHS Global Insight (Larson et al., 2012) and AEO 2013 (Energy Information Administration, 2013). Revised emissions factors for upstream natural gas were used. A single source is still modeled for natural gas extraction in each region, but the emissions factors were weighted by regional shale gas production percentages.

These production percentages had to be specified for each time step in the model, so modeling results from two other studies were used in the calculations: unconventional gas production forecasts to 2035 by IHS Global Insight (Larson et al., 2012), and total dry NG production forecasts from EIA's AEO 2013 (Energy Information Administration, 2013). In 2010, shale gas production percentages were derived from EIA Natural Gas Gross Withdrawals data: Tri-state (R2): 80%; Great lakes (R3): 50%; Midwest (R4): 24%; South Atlantic (R5): 44%; East Gulf (R6): 0%; Texas & West Gulf (R7): 41%; Rocky Mountain (R8): 6%; Pacific & Alaska (R9): 3%. These percentages represent the percentage of total NG production in each region from shale gas. Growth in shale gas production was projected and included in the model. The growth represents steady increases that leveling off in shale gas production fraction.

CO_2 and CH_4 emissions values are derived from Weber and Clavin (2012), and distinct values are used for conventional and shale gas. The NO_x, VOC, SO_2, and CO emissions factors are derived from results of the West-wide Jumpstart Air Quality Modeling Study (ENVIRON International Corporation, 2013), which covers the main energy-producing basins in the Rocky Mountain region. The production-weighted average of emissions per unit of production across these basins was assumed to apply across the country.

Future emissions are affected by New Source Performance Standards (NSPS) and National Emissions Standards for Hazardous Air Pollutants (NESHAP) rules (Macpherson, 2012) finalized in 2012 for oil and gas production, which are expected to significantly reduce VOC and CH_4 emissions associated with upstream natural gas production (Macpherson, 2012; U.S. EPA, 2012a). The technical support documents for the rules provide estimates of emissions reductions associated with each rule. Emissions reductions were calculated from information in the technical support documents and extended to 2030 using a straight line phase-in. Linear interpolation was used to calculate reductions for the intervening time periods. After 2035 the emissions factors were held constant, assuming that the effects of the regulations will be fully accounted for by that point. More details about adjusting emissions factors to represent the increase in hydraulic fracturing can be found in McLeod (2014).

3.7.1.3 Commercial and Residential Sectors

Changes made in the commercial and residential sector include the upstream emissions and sector-specific emissions described above. Solar water heaters and photovoltaic panels are the commercial technologies that required upstream emissions. Some zero values were also added to the residual technologies to fix an issue with importing the workbook to the database.

3.7.147 Biomass CO_2 Uptake

Another set of emissions factors was added to the database to account for CO_2 uptake from biomass growth. These CO2UB emissions are accounted for on biomass collector technologies, and this accounting was not included in the EPA database. These values were derived from Wang et al. (2012). Uptake emissions are only applicable to biomass from stover and agricultural residue, not corn. These emissions are represented as negative CO_2 emissions and positive CO2UB emissions. The CO2UB emission type serves two purposes: first that the specific uptake emissions can be analyzed, and second that emissions fees can be applied to CO2U without causing errors in MARKAL.

For each MARKAL biomass feedstock collector technology, (XCBCRN, XCBSTV, XCBAGR), the emissions for land use change (LUC), diesel fuel use, and natural gas use are added together to determine the total upstream emission value of that fuel. The emissions are shown in Table 3.5. These emissions factors are derived from Wang et al. (2012) and include emissions from diesel use from agricultural equipment, natural gas used to create fertilizer, and emissions associated with land use change. For the fuels shown in the table, the emissions are added to the database, but the fuel use itself is not. The life cycle emissions are added as total and *U emissions, aside from the uptake emissions discussed above.

		Fuel PJ/Mt	CO2 t/t	VOC kt/Mt	NOx	PM10	PM25	SO2	CH4	CO2U	CO2UB
Feedstock production-collection energy corn	LUC		0.08							82.38	
	diesel	0.38	0.00	0.02	0.14	0.01	0.01	0.00	0.01	0.94	
	NG	0.48	0.00	0.00	0.02	0.00	0.00	0.02	0.13	2.62	
Implemented Values			**0.09**	**0.02**	**0.16**	**0.01**	**0.01**	**0.02**	**0.14**	**85.94**	**0.00**
Feedstock production-collection energy corn stover	LUC		-0.01							-9.59	
	diesel	0.22	0.00	0.01	0.08	0.00	0.00	0.00	0.00	0.54	
	NG	0.26	0.00	0.00	0.01	0.00	0.00	0.01	0.07	1.43	
Implemented Values			**-0.01**	**0.01**	**0.09**	**0.01**	**0.00**	**0.01**	**0.08**		**7.61**
Feedstock production-collection energy ag residues - other	LUC		-0.01							-9.59	
	diesel	0.22	0.00	0.01	0.08	0.00	0.00	0.00	0.00	0.54	
	NG	0.26	0.00	0.00	0.01	0.00	0.00	0.01	0.07	1.43	
Implemented Values			**-0.01**	**0.01**	**0.09**	**0.01**	**0.00**	**0.01**	**0.08**		**7.61**

Table 3.5 The Emissions values implemented in the modified version of MARKAL for biomass growth, as well as the intermediate values. LUC stands for Land Use Change, NG stands for natural gas. The Fuel use values are used as intermediate values to calculate emissions, but the fuel used in growing biomass is not considered in the MARKAL fuel pathway. CO2UB refers to the CO_2 uptake from biomass growth. All emissions except for CO_2 are in kt/Mt biomass, while CO_2 is represented as tons of CO_2 per ton of biomass. The CO2U and CO2UB parameters are in kt/Mt.

Additionally, CO_2 uptake negative emissions of 7.61kt/Mt derived from Wang et al. (2012) were added to account for CO_2 uptake from biomass growth from stover and agricultural residue. Uptake emissions are not applicable for corn. This emission type allows uptake emissions to be analyzed, and also allows GHG fees to be applied without causing errors. To account for GHG fees, a subsidy was applied to growth of stover and agricultural residue, as shown in Table 3.6.

Table 3.6 Subsidy for biomass CO_2 uptake in million $ per Megaton of biomass.

3% 95th	2.5% avg	3% avg	5% avg	
-0.7806	-0.4153	-0.2721	-0.0859	**2015**
-0.9238	-0.4655	-0.3079	-0.0859	**2020**
-1.0312	-0.5013	-0.3437	-0.1003	**2025**
-1.1386	-0.5442	-0.3724	-0.1146	**2030**
-1.2603	-0.5800	-0.4082	-0.1361	**2035**
-1.3749	-0.6230	-0.4440	-0.1504	**2040**
-1.4752	-0.6588	-0.4726	-0.1719	**2045**
-1.5826	-0.7018	-0.5084	-0.1933	**2050**

3.7.2 Refineries

3.7.2.1 Fuel Emission type

The EPA database was modified to apply the "fuel" emissions category to emissions from refineries, H2 production, and unconventional energy production such as a coal to liquids plant. This included redefining some *I industrial emissions as *F. This distinction was made because these emissions sources could be considered as either upstream or industrial. The emissions that were previously considered as industrial emissions are now considered in a new emission type with the same values.

3.7.2.2 Control Technologies

Emissions control options are also added for refineries. Control options added were determined based on information in the Mid-Atlantic Regional Air Management Association report (MARAMA, 2007). A wet scrubber option on the Fluid Catalytic Cracking Unit (FCCU) provides the SO_2 and PM combined control; two NO_x reduction options, one each on the FCCU and boiler are added, and advanced leak detection and repair (LDAR) techniques are represented to reduce VOC emissions. Refinery efficiency improvements were not added because I was unable to find data on efficiency improvements that are technologically available but not already commonly in place.

3.7.3 Old Coal

An additional scenario was added to ensure that older coal fired power plants were retired after they had been in operation for ~75 years. A set of filters was created to define coal by decade. A set of constraints uses those filters to retire each decade's investment at the appropriate time step. For example, coal EGUs built in the 1940s are no longer allowed to be used starting in 2025. (Note for those trying to use this database version: The filters are importable, but the version of the electricity scenario included here does not include the constraints. If this is a desired change, I can provide instructions or a different workbook that does.)

3.7.4 Electricity

Changes made in the electric sector include sector-specific emissions definitions and upstream emissions for waste, hydropower, geothermal, wind, solar, and nuclear electricity generation. Cost information was also updated for solar photovoltaics. One carbon capture and sequestration technology was also altered.

3.7.4.1 Biomass CCS

We changed the CO_2 removal efficiency for biomass IGCC with CCS. In both the modified and EPA versions of the database, biomass CO_2 from combustion is assumed equal to uptake. This means that the carbon captured by CCS on biomass is considered negative CO_2 emissions. In the EPA release of the database, the CO_2 removed from biomass IGCC with CCS is about twice that of coal, due to the low combustion efficiency of biomass. We have altered the CO_2 removal rate based on the kgC/kWh rate given in Rhodes and Keith (2005) and confirmed by IPCC numbers (Abanades et al., 2005). The EPA database removes 412 kt CO_2/PJ biomass IGCC with CCS used, while the modified database removes 142 kt CO_2/PJ biomass IGCC with CCS. As this technology is still new, however, this assumption should still be examined. We also researched cost assumptions, and decided that keeping the cost used in the EPA database with the more modest CO_2 removal rate fit best with the literature.

Since the policy mechanism we are using includes fees on CO_2 emissions, this technology can have a large effect on results, because a large enough CO_2 fee could lead to the EBIOIGCCCS technology to have a negative net cost even if the electricity generated from the technology is not large. This is why we have paid particular attention to this technology.

3.7.4.2 Solar PV cost

(Note: if looking for changes within the workbooks: these changes were not made in an imported worksheet, but in a calculations page. Changes were made in the electric sector workbook on the solar raw data page, row 25.)

The base PV cost in $/kW was changed from the EPA database. For each region, class, cost category combination there is a cost adder to determine the actual cost of solar PV investment. The cost was changed to the values below based on the average, centralized installation cost from the NREL/Black and Veatch (Black and Veatch, 2012) report converted to 2005 USD. This represents a lower installation cost than in the EPA database. Costs were not changed in residential and commercial sectors, because in those sectors the investment cost was similar to the NREL values. The cost was not changed for CSP because the CSP costs were also similar to those in the NREL report. The PV FIXOM parameter in the electric sector was also changed to 37.57 $/kW-year and based on the average across years from the Black & Veatch comparisons.

Table 3.7 The modified investment costs values for solar PV electricity generation in $/kW of capacity.

Year	2010	2015	2020	2025	2030	2035	2040	2045	2050	2055
Investment Cost ($/kW)	3091	2568	2418	2304	2196	2129	2057	1990	1930	1868

3.7.4.3 Electricity trade

The INVCOST parameter for electricity import infrastructure (from Mexico and Canada) and the INVCOST of domestic electricity trade infrastructure was raised to $1500. This change was originally applied by McLeod (2014); we have retained it with the new version of the workbook. This change is to represent increasing the cost of new electricity transmission capacity to match assumptions from NREL (National Renewable Energy Laboratory, 2012).

3.7.5 Industrial Sector Changes

The industrial sector contains most of the changes compared to the EPA database. In addition to the typical sector-specific and upstream emissions, further control technologies and energy conversion technologies were added. Several emissions reduction options have been added to this version of the MARKAL database, but the full spectrum of possible responses is not represented. This is due in large part to the very inhomogeneous nature of the industrial sector and the rough resolution of the model. Since many emission control and efficiency improvements are not available for all industrial facilities, they cannot be considered in MARKAL. However, we think the modifications provide a more representative picture of types of responses available (i.e., fuel switching, efficiency improvements, control technologies), allowing the model to respond more fully.

3.7.5.1 Efficient boilers

There is some representation of industrial efficiency improvements in the industrial sector in the EPA database, but it is limited. In order to model a more robust industrial system, we have added more energy efficiency technologies in parallel to the existing technologies. Efficiency improvements were added to industrial boilers. The efficiency improvement option represented is based on optimization of the boiler, but the range of costs and improvements associated with optimization are similar to the ranges associated with “reducing slagging and fouling of heat transfer surfaces.” Optimization refers to testing the boiler and establishing the parameters, such as temperature and air flow, under which the boiler will produce the most energy output per energy input. The representative option could represent either optimization or reduced fouling of heat transfer surfaces being implemented on an individual boiler. This improves efficiency by one percentage point and the associated capital cost is 1.082 million USD per Petajoule. This efficiency improvement reduces emissions because emissions are defined per PJ of fuel used for industrial boilers. This measure is assumed to be a one time action so there is no operation and maintenance cost. This option is available as a retrofit technology that takes an input of .99PJ of fuel and outputs 1 PJ of fuel or as a new boiler which has a heat rate one percentage point lower than the original heat rate but with the same lifetime and O&M costs as the existing boiler options, but with the capital cost increased by 1.082M$/PJ. Although the retrofit efficiency improvement is slightly less than one percentage point, the efficiency improvement is within the range suggested in the EPA white paper, and retrofitting an existing boiler may not be able to reach the same efficiency as better initial planning in a new boiler. This data is from a white paper by the EPA on GHG reduction methods (US EPA, 2010). The cost we implement in MARKAL is one of the higher values in the white paper, because the paper reported costs for facilities that have already optimized their boilers, and this is more likely to have been done at facilities where the cost was lower.

Because the ICI boiler MACT rule requires an annual tune-up for natural gas boilers in lieu of emissions limits, no more efficient technologies were added for that fuel. Also, efficient boilers were not added where the activity level of the existing technology was low in the base case. When a new boiler is needed, there are efficient and standard options. When an existing boiler is being used, there is a passthrough option that continues use of the boiler as is and a retrofit option that improves the efficiency at a cost. The naming convention follows the EPA database technologies, but the fourth character is R for retrofit or U for upgrade (instead of N for new or E for existing). The start date, lifetime, capunit, discrate, and AF parameters are based on technologies defined in the EPA database. Retrofit and retrofit passthrough technologies do not need an AF to be defined, because those technologies are add-ons to the existing technology. For new and upgraded technologies, the (fixed and variable) operation and maintenance costs are the same as for the regular technologies, and for the retrofit technology, those costs are accounted for in the existing technology. The technology pathway is as follows: retrofit commodity fuel in → retrofit or passthrough→interim commodity→EPA technology →out Energy.

3.7.5.2 Industrial solar

There is potential for solar heat to be used in the industrial sector. Although the US currently does not utilize this technology very much, other countries have found solar energy to be a cost efficient source of industrial heat (Islam et al., 2013). Industrial solar technologies are not represented in the EPA database, but I added two types of solar heat; flat plate solar, which satisfies lower temperature heat needs, and parabolic trough solar, which can be used for higher temperature heat demand but is more expensive. The flat plate technology was created based on the flat plate solar heat technology available in the commercial sector. The parabolic trough technology option was defined in part from data from the System Advisor Model (Blair et al., 2014) for electric parabolic trough, with the investment cost based on a PIER demonstration facility in California (Ruby and American Energy Assets, California L.P., 2012). The use of these technologies is constrained, because solar thermal energy requires space and certain temperature demands. The constraints implemented were based on Lauterbach et al. (2012) The constraints are defined as the maximum percentage of process heat that the technology could satisfy for each industrial sub-sector, and the values are defined below. Hurdle rates for this technology are set to 0.20 based on evaluation of electric, residential, and commercial sector hurdle rates for solar technologies. The upstream emissions factors for these technologies are taken from a NEEDS report on solar thermal power plants (New Energy Externalities Development for Sustainability, 2008) for criteria pollutants and from Burkhardt et al. (2012) for GHGs.

Table 3.8 The multipliers used to define constraints on parabolic trough solar heating used in the industrial sector.

Sector	Abv.	Constraint multiplier
Food	FD	0.04
Pulp	PL	0.15
Paper	PA	0.15
Paperboard	PB	0.15
Other paper	PO	0.15
Organic chemicals	CO	0.04
Inorganic chemicals	CI	0.04
Plastic, fiber, resin	CP	0.13
Ag chemicals	CA	0.04
Other chemicals	CT	0.04

The constraints for the parabolic trough heat were determined by multiplying the sector specific multiplier above (determined from Lauterbach et al. (2012)) by the industrial demand for the sector and region and by the % of process heat for that sector and region. The constraint multiplier values (in Table 3.8) include a 0.4 multiplier to represent the assumption that only 40% of industrial facilities with the appropriate heat demand have the space and ability to install a concentrating solar structure. Further determination of appropriate sectors and temperature values is described in Lauterbach et al. (2012). The flat plate solar thermal is only allowed in the food sector because of the low temperature required in that sector. The constraint for that sector is 0.08 multiplied by the industrial demand multiplied by the percentage of process heat in the sector. These constraints were determined to limit the use of solar process heat to facilities that could get by with only solar thermal, because the economics of a combined system would be different. Solar is only allowed in industries where we know from the literature that it is feasible, because some sectors require higher temperatures than this technology can achieve.

3.7.5.4 Control Technologies

Industrial energy use in MARKAL includes boiler, electrochemical, feedstock, machine drive, process heat, facility, and other. I have added emissions controls to the boiler and process heat energy use but such controls are not applicable for electrochemical energy or for fuels used as feedstock. The facility energy use and other energy use categories are too diverse to add control technologies; the energy used in these "technologies" (as a technology is defined in MARKAL) includes a wide variety of energy use. Many of these uses do not have applicable controls and the number of different technologies included under these umbrellas does not have any universally applicable control technologies

The technologies added are based on CoST modeling done by Julia Gamas at the EPA (Gamas, 2015). These control technologies were compared with the technologies in MARKAL to see which available controls fit into the MARKAL framework and are not already assumed to be in place. Most controls are added to process heat for a specific sector or boilers. Only controls that could be mapped to technologies explicitly modeled in MARKAL were added and only for the industrial sector. No VOC controls were added because the controllable sources don't fit well with the technologies in MARKAL. For boilers subject to the boiler MACT regulations or for which other controls are assumed in the EPA database, additional controls were not modeled on affected pollutants, with the exception that in some cases LNB was added to boilers but S(N)CR controls could still be applied. No PM controls were added to boilers because if these controls were feasible they would have already been added to comply with the MACT regulations. Additional SO_2 controls should be avoided if wet scrubbers are already used, based on Table 100 in the BoilerMACT worksheet of the IND workbook.

A constraint was added to limit controls to 80% of the possible level for all industrial controls because there are likely a small number of facilities that cannot or would not apply controls that are generally applicable within a technology category. When the technologies were applied, the cost was defined as a VAROM based on the average of the ann_cost_per_ton output from CoST.

The control framework for control options (except cement process heat, which is slightly different and described above) is:

Existing MARKAL technology→ pre-control commodity→NOx control options→interim commodity→ SO2 control options→interim commodity2→ PM control options→original MARKAL output commodity

- "Options" are 2-5 technologies including a passthrough technology option.
- Not every pathway will include all steps, because not all emissions controls are valid for all technologies.
- A control option will be defined with a negative emissions factor and costs based on the CoST file.
- All controls become available in 2015 with a 40 year lifetime.
- Each control option will be defined for a specific "Existing MARKAL technology" because different fuels have different emissions factors and different industrial sub-sectors need different commodity outputs.

3.7.5.4.1 Applicability

In the database of control technologies we are using from the CoST model, EPA eliminates controls where the cost per ton of emission reduction is above some threshold. Many small sources of emissions are not included in the pool of controllable sources. For this reason rule penetration limits are added to ensure that cheap controls are not being added in situations where they do not exist. Some sources are just not included as technologies that can be controlled in MARKAL, i.e. “area source” boilers and “micro-turbines.” Additionally, technologies were constrained so that only 80% of applicable sources could be controlled. This was done by creating filters on control categories (e.g. NOx boiler controls not including LNB) and controllable technologies (e.g. boilers to which controls could be applied) and defining rule-based constraints that limit the controls to be 80% of the energy conversion technologies.

For boilers that are already subject to emissions controls, additional control options may not be available. Exceptions include NG and LPG-fueled boilers, for which only NO_x emissions are affected by MACT related controls, so controls can be applied (all new NG already has LNB so that option is still not available. SO_2 controls can be applied except for coal and Tomlinson Black Liquor boilers.

PM boiler controls will not be added because the boiler MACT standards already take care of this. LNB is already installed on new NG and all boilers subject to the MACT rule, so will not be added as a new control. Further SO_2 controls won’t be added on coal or Tomlinson boilers subject to MACT because they are already controlled.

The controls and controllable technologies are detailed in the tables below.

3.7.5.4.2 Boiler Controls

AFR	air fuel ratio	applies to ICE gas
IGR	ignition retard	applies to ICE gas, ICE oil, REG oil
LNB	low NOx burner	applies to NG CCT
BSI	biosolid injection	applies to cement PH
MKF	mid kiln firing	applies to cement PH
S(N)CR	selective (non)catalytic reduction	applies to cement PH
CPH	cullet preheat	applies to glass PH
LNB	low NOx burner	applies to glass PH
OXY	oxyfiring	applies to glass PH
SCR	selective catalytic reduction	applies to glass PH

Table 3.9 NO_x control naming. ICE stands for internal combustion engine, PH stands for process heat, REG refers to reciprocating engine, NG represents natural gas, and CCT refers to combustion turbines.

Constraint Naming

Filters

Boilers:

Table 3.10 The boiler controls and conversion technologies to which they are applied. The * refers to the 2 letter sector designations used in the EPA release of MARKAL, e.g. FD for the food subsector. Filters were designed to apply the constraints with names as follows. Filter names: COABOLCON, COABOLLNB, COABOLNOX, COABOLSO2; RFLBOLCON, RFLBOLLNB, RFLBOLNOX, RFLBOLSO2; DSTBOLCON, DSTBOLLNB, DSTBOLNOX, DSTBOLSO2; NGBOLCON, NGBOLLNB, NGBOLNOX.

Fuel	Conversion tech	LNB controls	NOx controls	SO2 controls
Coal	I*BOLCOA, I*BPVCOA, I*BFBCOA, I*BSOCOA	S*BOCLNB	S*BOCSCR, S*BOCSNR	S*BOCDIF
Residual oil	I*BOLRFL, I*BHLRFL	S*BORLNB	S*BORSCR, S*BORSNR	S*BORDIF, S*BORWET, S*BHRWET
Distilate oil	I*BOLDST, I*BLLDST	S*BODLNB	S*BODSCR, S*BODSNR	S*BODWET, S*BLDWET
Natural Gas	I*BSPNGA, I*BOLNGA	S*BONLNB	S*BONSCR, S*BONSNR	NA

CCT*:* The names for combustion turbines follow this pattern, as for other names the * represents the two letter code for the industrial subsector. Energy Conversion Technologies: I*CCTNGA; Control Technologies: S*CTNLNB; filter names CCTNGA, CCTLNB.

REG

Table 3.11 The engine controls grouped by fuel type, which are constrained by the following filters: REGRFL, REGDST, REGNGA, REGOIL, REGLPG; REGCTRLRFL, REGCTRLDST, REGCTRLNG, REGCTRLOIL, REGCTRLLPG.

fuel	Conversion tech	Control techs (AFR and IGR)
resid	I*REGRFL	S*EGR*
dist	I*REGDST	S*EGD*
NG	I*REGNGA	S*EGN*
oil	I*REGOIL	S*EGO*
LPG	I*REGLPG	S*EGL*

BST

Table 3.12 The control technologies for steam turbines by fuel and conversion technologies. As for other names, the * represents the two letter code associated with each industrial subsector. The filter names associated with this table are: RFLBST, RFLBSTLNB, RFLBSTNOX, RFLBSTSO2; DSTBST, DSTBSTLNB, DSTBSTNOX, DSTBSTSO2; NGBST, NGBSTLNB, NGBSTNOX; COABST, COABSTLNB, COABSTNOX, COABSTSO2; LPGBST, LPGBSTLNB, LPGBSTNOX, LPGBSTSO2; OILBST, OILBSTLNB, OILBSTNOX, OILBSTSO2.

Fuel	Conversion tech	LNB control	NOx control	SO2 control
Residual oil	I*BSTRFL, I*CHLRFL	S*BCRLNB	S*BCRSCR, S*BCRSNR	S*CHRWET, S*BSRWET, S*BSRDIF
Distilate	I*BSTDST, I*CLLDST	S*BCDLNB	S*BCDSCR, S*BCDSNR	S*CLDWET, S*BSDWET
Natural Gas	I*BSTNGA	S*BCNLNB	S*BCNSCR, S*BCNSNR	
Coal	I*BSTCOA, I*CPBCOA, I*CFBCOA	S*BCCLNB	S*BCCSCR, S*BCCSNR	S*BSCDIF
LPG	I*BSTLPG	S*BCLLNB	S*BCLSCR, S*BCLSNR	S*BSLWET
Oil	I*BSTOIL	S*BCOLNB	S*BCOSCR, S*BCOSNR	S*BSOWET

Process Heat controls

Table 3.13 The controls for industrial process heat based on conversion technology and industrial subsector (category) to which they are applied. For this table the filter names are shown within the table itself. For process heat applicability varies by industrial subsector, but is applicable to all combustion fuels within the subsector. Electricity and concentrated solar power are not combustion fuels and therefore the control technologies are not applicable to process heat using these fuels.

Category	*tech*	*Tech filter*	*control tech*	*Control Filter*
Paper SO2	IP*PRH* -ELC, -CSP	IPPH	SP*FGD	IPPHFGD
Cement SO2	SINDPN*CO2, -SINDPNECO2	INCPH	SNC*FGD	INCPHFGD
Other nonmetal SO2	INOEPRH*-ELC	INOPH	SNO*FGD	INOPHFGD
Paper PM	IP*PRH* -ELC, -CSP or PTC (repeated filter)	IPPH	SP*ESP	IPPHESP
Chemical PM	IC*PRH*-ELC, -CSP	ICPH	SC*ESP	ICPHESP
Cement PM	SINDPN*CO2 (repeated filter)	INCPH	SNC*FFF	INCPHFF
Other nonmetal PM	INOEPRH*-ELC (repeated filter)	INOPH	SNO*FFF	INOPHFF
Steel PM	IMS*PRH*-ELC, IMT*PRH*-ELC	IMSPH	SMS*ESP, SMS*FFF, SMT*ESP, SMT*FFF	IMSPHPM
Other metal PM	IMA*PRH*-ELC, IML*PRH*-ELC, IMO*PRH*-ELC	IMPH	SMA*FFF, SML*FFF, SMO*FFF	IMPHPM
Cement NOx	SINDPN*CO2 (repeated filter)	INCPH	SNCPH*BSI, SNCPH*MKF, SNCPH*SCR, SNCPH*SNR	INCPHNOX
Glass SO2	INGEPRH*-ELC	INGPH	SNGEPR*FGD	INGPHSO2
Glass NOx	INGEPRH*-ELC (repeated filter)	INGPH	SNGPH*CPH, SNGPH*OXY, SNGPH*SCR, SNGPH*LNB	INGPHNOX

Different PM controls being used may not indicate a choice as much as they indicate options in particular subsectors. Also the percentage of controllable PM10 removed can be greater than 80% because the constraint isn't on the emissions (it's on activity).

3.7.5.3 Cement CCS

Carbon Capture and Sequestration control technology options were added to the industrial sector for cement process heat. The emissions for cement process heat were also altered to take into account the large portion of CO_2 that occurs due to calcination in addition to the CO_2 emissions related to fuel combustion. Accounting for these emissions is important, because even though they are not strictly energy related emissions, the benefit of removing these emissions may alter the economics of using CCS in the cement industry. The existing CO_2 emissions for cement process heat were multiplied by 3 for the controllable technologies, because the IEA technology roadmap(International Energy Agency, 2009) says that calcination emissions are 60-65% of total emissions while fuel combustion generates the rest. CO_2 emissions when the process heat is created using electricity were determined based on the emissions for the other fuels, 2/3*average of CO_2 for the other technologies.

The energy carriers added to support this inclusion are named ICCS_*. The pathway in MARKAL is as follows: ind energy → emis accounting → placeholder energy → PT or CCS → placeholder E → process heat → INCPRH

The CCS controls are named SINDPN*CCS where the * refers to the fuel used. The cost and energy penalty for these technologies is based on the values used for the electric sector in the EPA database. These technologies become available in 2020. The emissions accounting is also moved to new technologies (SINDPN*) that are "upstream" of the process heat technologies. Carbon Capture and Sequestration (CCS) options are added between the emissions accounting and process heat technologies. After the process heat technology, the PM, SO2, and NOx control options are applied as specified above.

3.7.6 Transportation

In addition to adding sector specific emissions, the representation of CAFE was also changed. The existing CAFE calculations were changed to refer to the LHV instead of HHV. For 2005 the HHV is used because altering this year causes difficulties because of constraints to historical data.

3.7.7 Other Changes

In two scenarios (IND and TRN_OH) the definition of CO2 in the commodities tab was changed to Mt because it was erroneously kt. This is just a small change for consistency.

3.7.8 Changes by workbook

Industrial

Solar thermal process heat

Efficiency options

Controls on PH and boilers with constraints

Emissions types and upstream

Electricity

Upstream

Emissions types

PV cost

CCS for biomass

Coal

Emissions upstream

Coal methane

TRN LDV

Emissions

CAFE LHV

TRN OH

Emissions

NG

Upstream emissions

Refinery

Controls added

Emissions types

Commercial

Emissions

Set residuals to zero

Residential

Emissions types

Set residuals to zero

Biomass

Upstream emissions

TRN HDV

Emissions

Oil

Upstream emissions

TRD_ELC

Cost of adding electric transmission capacity increased

Unconventional

Emissions

H2

Emissions

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Chapter 4

Evaluating the Health Benefits of Internalizing Air Quality and Climate Externalities through Fees in the US Energy System Using 3D Air Quality Modeling

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4.0 Abstract and Preamble

4.0.1 Abstract

Emissions from energy production, conversion, and use are associated with negative impacts on human health and climate. We use an energy system model to determine the impact of fees designed to mitigate these negative impacts. We then use air quality and health benefit models to quantify the effect of three emissions policies for the year 2045. The policies are composed of emissions fees designed to internalize some of the externalities associated with health impacting pollutant and greenhouse-gas emissions. The net policy benefits in 2045 are $413 billion with fees based on health related damages, and $238 billion with climate-based fees. A combined policy, which includes fees on emissions of both greenhouse gases and health impacting pollutants, has net benefits of $439 billion. Concentrations of both $PM_{2.5}$ and ozone are reduced with all three policies, therefore air quality co-benefits exist with climate policy targets, but the air quality improvements and health benefits are larger when the fees including those targeting health impacting pollutants.

4.0.2 Preamble

The results presented in this chapter are preliminary. We intend to modify the air quality simulations to remove some of the uncertainty associated with the methodology that we use to scale the emissions to represent the results of the MARKAL runs. The new simulations will better account for the non-energy related fraction of emissions that are captured in the emissions reductions described below. We also intend to reconsider the way upstream $PM_{2.5}$ emissions are handled, due to the large portion of benefits associated with these emissions and the uncertainty surrounding how their location is treated. We also hope to improve our ability to evaluate the accuracy of our 2007 modeled $PM_{2.5}$ concentrations, as our estimates do not correspond well with the observed air quality. Viewing the results presented here is instructive, and we expect some conclusions to hold related to regional differences and relative differences between cases, but we will continue to seek ways to improve this analysis, particularly pertaining to scaling factors for emissions reductions and $PM_{2.5}$ model evaluation. Due to these uncertainties, the quantitative results presented here are tentative, particularly calculations of total benefits and absolute future air quality. There are still qualitative conclusions that can be drawn from the results, such as which areas of the country or fees are likely to have the most improvement.

4.1 Introduction

Energy production, conversion, and use produce emissions that have negative consequences for human health and the earth's climate. Ozone (O_3), which is formed from emissions of NO_x and VOCs (volatile organic compounds), and $PM_{2.5}$ (particulate matter less than 2.5 microns in diameter), which is emitted from combustion sources as well as formed from emissions of NO_x, SO_2 and VOCs, have many impacts on human health, particularly on cardiovascular and respiratory systems, which can lead to societal and medical costs. High concentrations of $PM_{2.5}$ are associated with health impacts including asthma exacerbation (Ostro et al., 2001; Mar et al., 2004), heart attacks (Pope et al., 2006), and other illnesses of the cardio-respiratory system (Kloog et al., 2012; Peng et al., 2009; Bell, et al., 2008; Dockery et al., 1996; Moolgavkar, 2000; Ostro and Rothschild, 1989). Exposure to high concentrations of O_3 is associated with health impacts including chronic asthma (Mortimer et al., 2002; Schildcrout et al., 2006; O'Connor et al., 2008), respiratory related hospital visits (Mar and Koenig, 2009; Glad et al., 2012). GHG emissions lead to climate change including increases in temperature and changes in precipitation patterns (Kirtman et al., 2013). Among other impacts, climate change can reduce crop yields affecting food supply and alter the availability of water (Dasgupta et al., 2014). These negative health and climate effects are externalities because they are not included in the price of energy.

Air quality modeling and health impact assessment tools are frequently employed to evaluate the monetized benefits of air pollution regulations. Fann and Risley (2011) used observed concentrations with the BenMAP (Benefit Mapping and Analysis) tool to determine the health benefits of historical air quality trends under the influence of the NAAQS. They found an overall reduction in mortality from both O_3 and $PM_{2.5}$ in the time period from 2000 to 2007. Fann et al. (2012) estimated that over 100,000 premature mortalities were associated with modeled 2005 levels of air pollution in the US. As another example, a Regulatory Impact Analysis (RIA) (U.S. EPA, 2012b) was performed by the EPA when the National Ambient Air Quality Standard (NAAQS) for $PM_{2.5}$ was strengthened from

15 to 12 $\mu g/m^3$. The RIA used the Community Multiscale Air Quality (CMAQ) model, a continental-scale atmospheric chemistry and transport model and the Benefits Mapping and Analysis Program (BenMAP), software designed to calculate air pollution related health impacts, to estimate the effects of the regulation. EPA estimated net benefits, which include the cost of additional emissions controls and monetized health benefits, in 2020 from $3.7 billion to $9 billion assuming a 3% discount rate. For the RIA accompanying the revision of the ozone standard from 75 ppb to 70 ppb (US EPA, 2015a), the EPA found that a reduction in NO_x emissions of 237-556 thousand tons per year for the contiguous US excluding California and a VOC reduction of 20-105 thousand tons per year over the same domain in 2025 would lead to net annual benefits of $2.4 -12 billion (year 2011 USD). The total benefits include $PM_{2.5}$ co-benefits of reduced morbidity and mortality, which were calculated by using existing benefit per ton values for precursor emissions.

Analysis of the monetized health benefits of air quality regulations, in conjunction with estimates of emissions changes that drove these improvements, has led to calculations of the marginal damages of health impacting pollutants, and how these vary by location and sector. Marginal damages refer to the external cost associated with an additional unit of emissions, such as the value of mortality and other health impacts associated with an additional ton of emissions of SO_2. Marginal damages for criteria pollutants are calculated in several papers for emissions from different sources including electricity generation, residential and commercial energy use (National Research Council Committee on Health, Environmental, and Other External Costs and Benefits of Energy Production and Consumption, 2010), and several energy and industrial sources (Fann et al., 2012). Marginal damages for health impacting pollutants (HIPs) are heavily affected by the proximity of the emissions source to populations, so damages tend to be larger for industrial and transportation sources that and lower for electricity generation. The largest component of the damage values is $PM_{2.5}$ related mortality. Those damages are estimated using concentration-response functions determined by Krewski et al. (2009), Laden et al. (2006), Pope et al. (2002) and Woodruff et al. (2006). Turner et al. (2015) found that marginal damages of black carbon emissions owing to premature deaths in the US are larger near the location of premature deaths. Buonocore et al. (2014) discuss the influence of location and find the population distribution downwind of emission sources to be important in calculating marginal damages.

Future marginal damages of air pollutant emissions will depend on future meteorology and boundary conditions. Other studies have considered the effect of altered meteorology and boundary conditions on modeled air quality. Simulating air quality using future meteorology assuming climate change results in higher O_3 and $PM_{2.5}$ concentrations, although when emissions are also reduced this effect is mitigated (Trail et al., 2014; Garcia-Menendez et al., 2015; Fiore et al., 2012). There is uncertainty in projecting this future meteorology, so it is important to be aware of possible effects, but aggressive climate policy could also mitigate these effects (Hogrefe et al., 2004).

Some previous research has coupled energy systems modeling with air quality modeling. Saari et al. (2015) used a coupled economic and air quality modeling system comprised of the United States Regional Energy Policy (USREP) model and the Comprehensive Air Quality Model with Extensions (CAMx) to explore the air quality co-benefits from two climate policies. With a Cap-and-Trade climate program, the air quality co-benefits completely offset the cost of the reduction, although these co-benefits are not distributed uniformly across the US. Thompson et al. (2014) used USREP to determine the costs and approximate emissions changes associated with three different climate policies: a clean energy standard, a cap and trade regulation, and a transportation fuel standard. They then used CMAQ with scaled emissions to determine ambient concentrations of O_3 and $PM_{2.5}$ and calculated monetized health co-benefits with BenMAP. They found that more flexible policies are less costly, and that the air quality co-benefits alone may be larger than the costs of the policy, which will have further climate benefits. They report that the air quality co-benefits can offset up to 1000% of the cost of the climate policies, but for some policies the cost of the policy will outweigh the air quality co-benefits. Rudokas et al. (2015) analyzed several climate policies using MARKAL (the MARKet ALlocation energy economic model), and found that SO_2 and NO_x emissions are lower with many climate policies, although sometimes they are slightly higher due to reduction in investment in controls associated with shorter lifetimes of power plants. Trail et al. (2015) continued this work, coupling the MARKAL results with CMAQ. They present that O_3 and $PM_{2.5}$ decrease with most CO_2 policies considered, but the increased NO_x emissions in some cases lead to increased O_3, and in some urban areas there is an increase in $PM_{2.5}$ associated with fuel switching.

Previous studies have also considered the effect of internalizing emissions externalities on the total cost of the system. Nguyen (2008) show, using MARKAL for Vietnam for 2005-2025, that internalizing health and climate externalities in Vietnam would lead to an increase in electricity price of 2.6 cents/kWh, but the avoided external costs would be 4.4 cents/kWh, so the total cost of electricity decreased. Pietrapertosa et al.

(2010) found that by internalizing health and climate externalities in Italy, modeled using the multi-region NEEDS-TIMES model through 2050, the total system cost including the energy system cost and externalities cost was reduced by up to 1.7% compared to the projected costs without internalizing externalities.

In this paper, we perform an integrated assessment of the air quality and health benefits of applying damage-based fees to emissions from energy production, conversion, and use. We combine energy system, air quality, and benefits modeling to determine the extent to which internalizing externalities through fees reduces externalities at a cost lower than the damages. The MARKAL model described in section 4.1 is used to project how future fuels, energy technologies, and emissions controls might change in response to application of emissions fees, described in section 4.2, that are designed to internalize externalities by incorporating damage costs into the cost of energy. One set of fees applied corresponds to health damages from air pollution, a second set is focused on climate change impacts, and a third has fees accounting for both. The CMAQ model, described in section 4.3, is used to evaluate the air quality in 2045 for a base case as well as the three fee cases. We detail how the emissions for each case are calculated in section 4.4, and BenMAP-CE is described in section 4.5. We present the air quality results in section 4.3.

Theoretically, multiplying the avoided emissions by the damage value of the emissions should approximate the benefit of a policy that reduces emissions. In this paper, we use additional models (CMAQ and BenMAP) to calculate the benefits of emissions reductions corresponding to each fee case, and compare them to damage scaled emission (estimated by multiplying the reductions (tons) by the damage value ($/ton)). We expect the additional models to provide a more accurate benefit estimate than the damage scaled emissions, because the marginal damages represent an average value, while the benefit of reducing emissions is not constant across emissions locations (Fann et al., 2012). We present benefits calculated both ways in section 4.3. This methodology allows us to compare the reduced form approach to evaluating benefits of a policy to the more time- and resource-intensive benefit calculation. We discuss the differences and other conclusions in section 4.4.

4.2 Methods

4.2.1 MARKAL

MARKAL is an energy system model that can be used to determine the lowest cost system of fuels and processing, conversion and end-use technologies that will satisfy a specified level of demand for energy services. We use a modified version of the EPA US 9 region database (US EPA, 2013b), which is benchmarked to the 2014 Annual Energy Outlook (U.S. Energy Information Administration, 2014). The modifications made for this study include an increase in emissions control and solar technologies as well as efficiency improvements in the industrial sector, more thorough accounting of life cycle emissions, and cost updates for renewable technologies (Brown et al., 2013; Brown et al., 2014). MARKAL solutions must satisfy demand for energy services and other constraints such as emissions limits, but can do so through the use of a wide range of fuels, production, emissions control or energy conservation technologies (Loulou et al., 2004).

Our MARKAL simulations include a no fee case and three fee scenarios for the US from 2005 to 2055 in five-year time steps. We use a 5% discount rate. The difference in total system cost in a year with a fee enforced compared to cost in the no fee case simulation without fees is interpreted as the cost of the policy. The total system cost each year includes the annualized investment in new technologies, operation and maintenance costs, the cost of importing and producing fuels, revenue from energy resource exports, and costs to transport the fuel.

4.2.2 Emission Fee Cases

In this study, we generate four emissions scenarios for the year 2045 by considering application of fees to emissions produced in the US energy system and simulating their impact on emissions using MARKAL. Unless otherwise stated, all currency values in this paper are presented in year 2005 USD.

The first case we consider represents a business as usual scenario, including existing policies to reduce emissions and influence energy choices such as the renewable portfolio standards, the Corporate Average Fuel Economy standards, and the Clean Air Interstate Rule, which has since been replaced by the Cross State Air Pollution Rule, but with no implementation of emissions fees. This is referred to as the No Fees case.

The other cases all include emissions fees in addition to the existing policies represented in the No Fee case. The fees used are based on estimates of damages associated with the emissions. In the HIP, or Health Impacting Pollutant, case, fees are levied on health impacting pollutants (NO_x, SO_2, PM_{10}, $PM_{2.5}$, and VOCs) from energy related emissions sources. The fees for this case are presented in Table 4.1. The fees vary by energy sector to capture some of the location dependence of the damages, except for PM_{10} and VOC, for which sector specific values were not available. For instance, industrial emissions tend to be located near population centers and will lead to more health impacts than electric sector emissions that are often located further from dense populations. The fees used here are based on damage estimates from Fann et al. (2012) with the following exceptions: VOC fees are based on Fann et al. (2009), and the PM_{10} and commercial sector values are based on National Research Council (2010) estimates, scaled to match the generally higher estimates in Fann et al. (2012). Fees for the commercial sector are applied to natural gas use (not emissions) because emissions specific damage estimates were not available for this sector; most commercial energy is from natural gas used for heat or from electricity. The second fee case is designed to address climate externalities, by imposing fees on greenhouse gas (GHG) emissions (methane and CO_2) from energy. The fees in this case vary across years because the impact increases as total carbon load increases with time; in 2045 the CO_2 fee is \$86.57/ton CO_2 and the methane fee is \$2,420/ton methane. The GHG fees are based on Social Cost of Carbon (Interagency Working Group on Social Cost of Carbon, 2013) for the average result of a 2.5% discount rate. Methane fees are determined from the Social Cost of Carbon using a 100 year global warming potential (GWP) of 28 (Myhre et al., 2013). Lastly, we also consider a combined fee case in which both the HIP and GHG fees are applied simultaneously.

Table 4.1: Fees applied in HIP policy case. (All values in $/ton except natural gas use which is in million $ per petajoule.) The Natural Gas use fee only applies for the commercial sector, because emissions specific damages were not available for this sector.

Sector	NO_x	PM_{10}	$PM_{2.5}$	SO_2	VOC	Natural Gas Use M$/PJ
Electric	4700	4110	117100	31500	2330	
Industrial	5500	4110	234300	35100	2330	
Transportation	6600	4110	324400	17100	2330	
Upstream	5600	4110	225267	27900	2330	
Refinery	5900	4110	279300	59500	2330	
Residential	11700	4110	324400	87400	2330	
Commercial						0.579

4.2.3 CMAQ Simulations and Performance Evaluation Data

The Community Multiscale Air Quality (CMAQ) model is a three-dimensional Eulerian air quality model (Byun and Schere, 2006). CMAQ includes treatment of gaseous, aqueous, and solid phase chemistry and aerosol dynamics in order to simulate the interaction of pollutants with each other and the environment. The physical processes modeled by CMAQ include advection and diffusion both horizontally and vertically as well as wet and dry deposition. This includes the formation of secondary aerosols such as sulfate and nitrate. For this project we use CMAQ v 5.0.2 with the 36-kilometer resolution CONUS domain, which extends over the contiguous 48 states of the US and beyond with 24 vertical layers. Meteorology input data are obtained using version 3.1 of the Weather Research Forecasting model (WRF) (Skamarock et al., 2005); and boundary and initial conditions are calculated using GEOS-Chem version 8.02.03 with a 2.0^{o} x 2.5^{o} grid resolution (U.S. EPA, 2012b). Biogenic emissions are from BEIS 2.14 (Pierce and Waldruff, 1991); wildfire emissions are obtained using the SMARTFIRE2 system (Ewer et al., 1999); and sectoral emissions use the Sparse Matrix Operator Kernel Emissions modeling system (Coats and Houyoux, 1996) v 3.5.1 including EGU and non-EGU industrial combustion sources. The default historical emissions inputs we use in CMAQ are based on 2007 National Emissions Inventory (NEI) emissions. We used the carbon bond gas phase chemical mechanism with active chlorine chemistry (CB05CL) and the fifth-generation aerosol mechanism (AE5). Simulations are performed for four months of the year (February, May, August, and November) to capture seasonal differences across the year with minimized computation time. A spinup time of one week is run before each of these months but is not part of the analyzed model results.

We first ran CMAQ for 2007 without any inclusion of scaling factors from MARKAL. We compare the results from this 2007 case against observations for 2007 obtained from the AQS Data Mart (US EPA, 2016). The AQS Data Mart contains all measured values the EPA collects via the national ambient air monitoring program, including 8 hour and daily averages. When we refer to "annual" comparisons we mean comparing observations from February, May, August, and November to the 2007 model results from the same months. We compare total $PM_{2.5}$ and O_3 as well as the sulfate and nitrate components of $PM_{2.5}$, as those are important for health impacts. We created plots of the 2007 model results with the observed concentrations plotted as the monitor location, as presented in Figure 4.2 and Appendix figures 4.7 - 4.16. We also calculated the statistical relationship between observed concentrations and model concentrations at the same location.

4.2.4 Emission Scenarios

In order to use CMAQ to estimate the air quality impacts of the policies modeled in MARKAL, we need to map the output of the emissions from MARKAL to inputs for CMAQ. The same 2007 emissions have been used at the 12km resolution in previous studies (Turner et al., 2015; Turner et al., 2015). Here we modify these emissions using scaling factors for the year 2045 for each policy scenario treated by MARKAL. Scaling factors are calculated for nine regions of the country, corresponding to the nine regions in MARKAL, mapped in Figure 4.1. This methodology assumes that the spatial distribution of emissions at scales finer than these regions remains fixed. Although this assumption is unlikely to be precisely true as older facilities are retired and newer ones built, many emissions will be in similar locations as demands will still be near the same cities, resources, or other industries. Scaling factors are applied only to emissions within the US, not to the boundary conditions or the emissions in the parts of Canada and Mexico included in our modeling domain. A limitation of the analysis is that non-energy related emissions within the US are affected by the emissions factors for the scaled species. Non-anthropogenic emissions are not affected by the industrial and electric sector scaling factors. The species scaling factors are applied to all emissions of a species. VOC scaling factors are not applied to VOCs that are mainly biogenic including isoprene, terpenes,, and sesquiterpenes. For $PM_{2.5}$, we do not apply the scaling factor to sea salt or water in the particle phase. For PM_{10}, the soil emissions are not perturbed, nor are the salt, water, or sulfate emissions; only the unspeciated PM_{10} emissions are altered because the other emission, coarse soil, is not energy related.

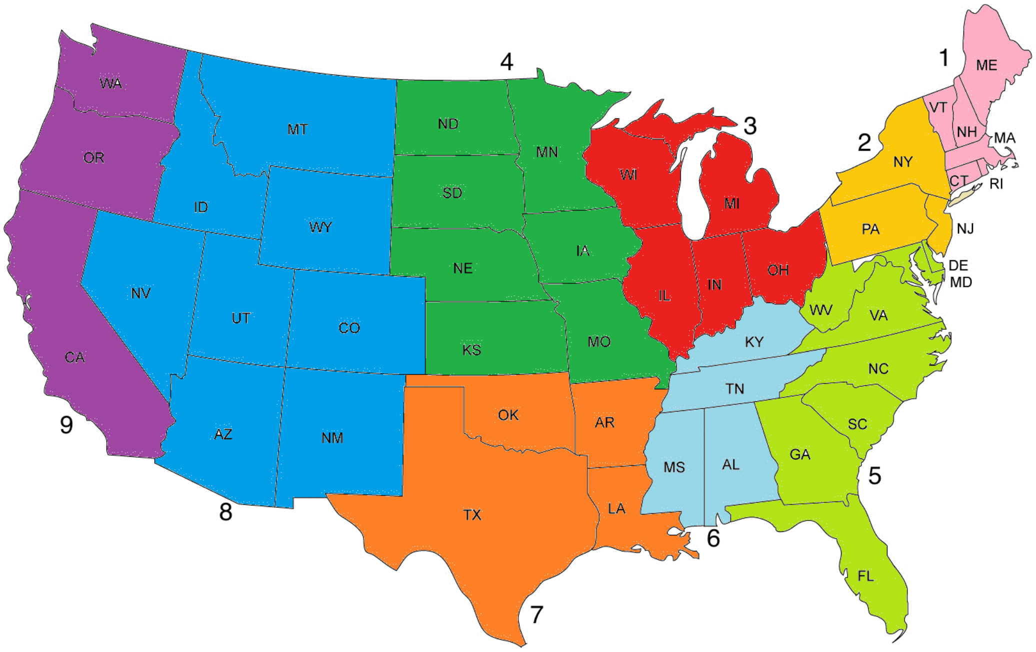


Figure 4.1 The nine MARKAL regions, for which emissions changes are calculated for CMAQ inputs. This figure is modified from the MARKAL US 9 region database documentation (US EPA, 2013b).

Table 4.2 Scaling factors for pollutants and sectors for each case and region.

No Fee									
	R1	R2	R3	R4	R5	R6	R7	R8	R9
NOx	0.5	0.6	0.6	0.6	0.5	0.6	0.8	0.6	0.6
CO	0.5	0.5	0.5	0.5	0.6	0.5	0.6	0.6	0.6
SO2	0.8	0.5	0.4	0.5	0.4	0.3	0.6	0.8	1.3
VOC	0.6	0.8	0.5	0.8	0.7	0.8	0.8	0.6	0.7
PM2.5	0.8	0.8	0.7	0.8	0.8	0.7	0.9	0.8	0.9
PM10	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8
IND	2.0								
ELC	0.6								

HIP									
	R1	R2	R3	R4	R5	R6	R7	R8	R9
NOx	0.4	0.5	0.4	0.4	0.4	0.4	0.6	0.4	0.5
CO	0.5	0.4	0.4	0.5	0.5	0.5	0.6	0.5	0.5
SO2	0.5	0.2	0.1	0.2	0.1	0.1	0.2	0.1	0.4
VOC	0.5	0.9	0.5	0.7	0.8	0.7	0.9	0.7	0.8
PM2.5	0.4	0.4	0.5	0.4	0.6	0.3	0.5	0.3	0.5
PM10	0.2	0.2	0.3	0.2	0.3	0.2	0.3	0.2	0.3
IND	2.6								
ELC	0.2								

GHG									
	R1	R2	R3	R4	R5	R6	R7	R8	R9
NOx	0.5	0.5	0.5	0.5	0.5	0.4	0.7	0.5	0.5
CO	0.5	0.4	0.4	0.5	0.5	0.5	0.6	0.6	0.5
SO2	0.7	0.4	0.2	0.3	0.3	0.2	0.4	0.4	0.7
VOC	0.5	0.8	0.5	0.7	0.7	0.7	0.8	0.6	0.7
PM2.5	0.5	0.5	0.6	0.5	0.7	0.5	0.7	0.5	0.6
PM10	0.3	0.4	0.4	0.4	0.5	0.4	0.5	0.4	0.4
IND	2.8								
ELC	0.7								

Combined									
	R1	R2	R3	R4	R5	R6	R7	R8	R9
NOx	0.4	0.5	0.4	0.4	0.4	0.3	0.6	0.4	0.4
CO	0.5	0.4	0.4	0.5	0.5	0.5	0.6	0.5	0.5
SO2	0.5	0.2	0.1	0.2	0.1	0.1	0.2	0.1	0.4
VOC	0.5	0.8	0.5	0.7	0.7	0.7	0.9	0.6	0.6
PM2.5	0.4	0.4	0.5	0.4	0.6	0.3	0.5	0.3	0.5
PM10	0.2	0.2	0.3	0.2	0.3	0.2	0.3	0.2	0.3
IND	2.6								
ELC	0.2								

Two categories of scaling factors are calculated, one for each of NO_x, CO, SO_2, VOCs, $PM_{2.5}$, and PM_{10} by region, and then two additional scaling factors per case for industrial and electricity generation related emissions, because these sectors change more significantly in the MARKAL results. We use two categories of scaling factors because our emissions are separated by species and by source category. We can multiply a scaling factor by either subset of emissions, species or sector, but when species-specific scaling factors are applied they apply to all emissions of that species regardless of sector. When sector specific scaling factors are applied, emissions are altered by the scaling factor regardless of species.

To calculate the pollutant and sector scaling factors ($A_{p,R}$), the 2045 MARKAL emissions are compared to the 2010 No Fee emissions from MARKAL. The values are presented in Table 4.2 and calculated using Equation 1.

$$A_{p,R} = \frac{(M_{p,R} + \frac{1}{9}M_{o,p})}{(M_{p,R} + \frac{1}{9}M_{o,p})_N} \qquad (1)$$

where $M_{p,R}$ is the MARKAL emissions for pollutant p (p = NO_x, VOC, SO_2, PM_{10}, $PM_{2.5}$, CO) in region R (R = 1-9), and $A_{p,R}$ is the scaling factor used to alter the CMAQ emissions. The subscript N refers to emissions from 2010 in the no fee case. All other results are from 2045 for the case being analyzed. 2010 is chosen as a base year because MARKAL only calculates results in 5 year time steps so there is not an exact match to 2007. $M_{o,p}$ is emissions that MARKAL does not specify as being in a particular region, typically emissions associated with upstream processes; these emissions are evenly distributed across the regions for the purpose of this analysis. The $A_{p,R}$ values align with each pollutant and region in Table 4.2. Scaling factors for the industrial and electric sectors are calculated as follows. First we define $B_{p,s}$,

$$B_{p,s} = \frac{K_{p,s}}{K_{p,s,N}} \tag{2}$$

where $K_{p,s}$ is the emissions of a pollutant within either the industrial or electricity generating sector. $B_{p,s}$ is calculated as an interim value for each pollutant, *p*, and relevant sector (s = industrial and electric). As in Equation 1, *N* refers to the 2010 No Fee case. We next calculate an inflation factor C_s,

$$C_s = < \frac{B_{p,s}}{< A_{p,R} >_R} >_p \tag{3}$$

where $<A_{p,R}>_R$ is the average of the quantity over the regions, weighted by the quantity of emissions in that region, calculated separately for each pollutant. C_s is calculated separately for the electric and industrial sectors, and the fraction within the outer brackets is calculated for each of a subset of the pollutants (NO_x, SO_2, PM_{10}, $PM_{2.5}$). Then these four values for each sector are averaged as indicated by the outer brackets. The C_s values are denoted as "IND" and "ELC" in Table 4.2.

There are two formulae that can be used to determine the ultimate emissions factor that is used in CMAQ given the 2007 emissions, as written in Equations 4 and 5.

$$E_{s,p,R} = F_{s,p,R} A_{p,r} C_s \tag{4}$$

$$E_{p,R} = O_{p,R} A_{p,R} \tag{5}$$

where $E_{s,p,R}$ is the modified emission value used in CMAQ, $F_{s,p,R}$ is the historical 2007 CMAQ emissions, $O_{p,R}$ is the 2007 emissions of any anthropogenic point source emission not coming from electricity generation or industry, and $E_{p,R}$ is the modified emission values for emissions that are not electricity generating or industrial sources. For emissions that are from an electric or industrial sector source, Equation 4 is used, while Equation 5 is used for other modified emissions. Industrial emissions do refer to both refinery and industrial emissions.

A table with all of the scaling factors can be found in Table 4.2. In most cases, the scaling factor represents a decrease in emissions from the historical level, even for the No Fee case. The exceptions to this are that SO_2 emissions in region 9 (corresponding to the west coast states) increase from their very low initial level without fees, and that industrial sector emissions increase over time due to economic growth and are therefore always higher than the 2007 emission (always multiplied by a factor greater than 1). This industrial factor is also large because emissions are decreased by the species factor before being increased by the industrial factor.

4.2.5 BenMAP-CE

BenMAP-CE version 1.1, the Environmental Benefits Mapping and Analysis Program-Community Edition, was created by the US EPA to estimate health impacts and economic benefits related to changes in air quality (US EPA, 2013c). BenMAP-CE includes a variety of health impact functions, population data, and baseline health data. These functions relate changes in air pollutant concentrations with changes in health. This calculation considers the exposed population, the baseline incidence rate of the effect, the concentration of the pollutants considered, and the concentration-response function chosen to relate the pollutants to the health effects. BenMAP-CE considers health effects associated with ozone and $PM_{2.5}$. BenMAP-CE does not consider health impacts for NO_2. There are uncertainties in the results from BenMAP-CE, which stem from uncertainties within the health impact functions, modeled pollutant concentrations from CMAQ, and projections of population.

The pollutant concentrations from CMAQ are used as an input to the BenMAP-CE model. These models are often used together (Hubbell et al., 2009; Fann et al., 2011; Nowak et al., 2013). BenMAP-CE is used to calculate benefits for each of the three fee cases compared to the No Fee case in 2045. We use a currency year of 2005 to correspond with the MARKAL cost estimates. The health impact functions and valuation functions are selected based on the functions used for the O_3 and $PM_{2.5}$ NAAQS regulatory impact analysis (US EPA, 2015a; U.S. EPA, 2012b). The mortality estimates in Table 4.4 are calculated based on Krewski et al. (2009) and Jerrett et al. (2009). A complete list of health impact functions used can be found in the appendix. We apply EPA's standard value of statistical life, based on 26 value-of-life studies, which has an average value of $6.6 million year 2005 USD. The 2040 population data at 12 km resolution (the latest available in BenMAP-CE) are used to evaluate the impacts. This may lead to a slight underestimate of health impacts, but some of the health impact functions depend on demographic breakdowns, and trying to project this information to the future is beyond the scope of this work. The total population is projected to increase only 2% from 2040 to 2045 according to US Census projections (Colby and Ortman, 2014), as population growth is projected to slow over the next few decades. Baseline mortality incidence is based on the 2020 projections, which is the latest available in BenMAP-CE. Baseline incidence data for other health impacts is from 2007, except school loss days and acute bronchitis, which are from 2000 and the asthma prevalence dataset from 2008. EPA standard income growth was projected to 2020, again the latest year available. Regional health impacts are calculated at the CMAQ 36 km grid level, then aggregated to state level, and then summed by the regions in Figure 4.1.

In addition to the multiple categories of benefits, there are multiple ways to determine the value of those benefits. In addition to calculating benefits using BenMAP-CE, we also calculate damage-scaled emissions, which should approximate benefits. When we refer to damage-scaled emissions in this paper, we are referring to the quantity obtained by multiplying emissions reductions from the No Fee case by the HIP damage values in Table 4.1, adding reductions in commercial natural gas multiplied by the fee since damages for the commercial sector are in terms of natural gas use instead of emissions.

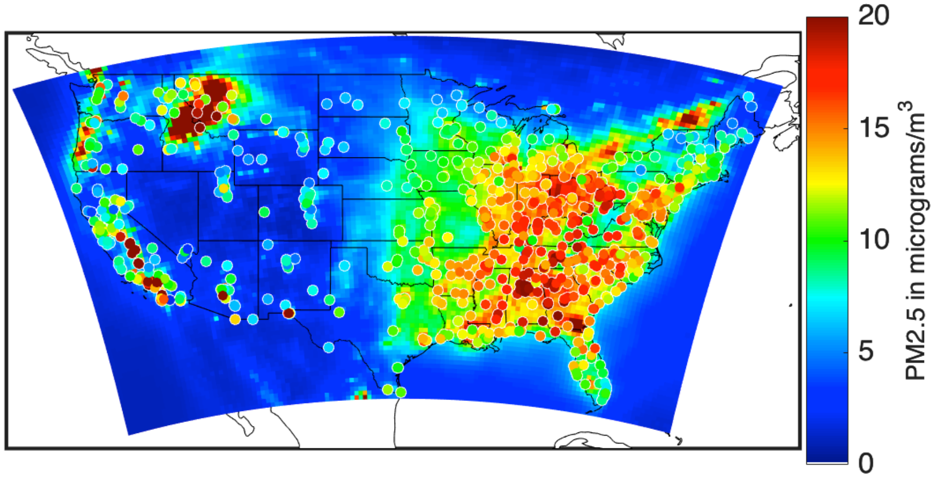
$$N = \sum_{p,s} \Delta H_{p,s} D_{p,s} + \Delta G D_G \tag{6}$$

where N is the damage-scaled emission value, ΔH_{ps} is the change in emissions of pollutant p in sector s compared to the No Fee case, D_{ps} is the damage value for that pollutant and sector, ΔG is the change in commercial natural gas use from the No Fee case and D_G is the damage value for commercial natural gas. The avoided GHG damages are calculated using the same method but multiplying by the GHG fees. Net benefits can be calculated in two ways, depending on how total costs are determined with respect to treatment of fees. Fees can be considered part of the cost or can be assumed to be neutral within the energy-environment-health system in which we are internalizing externalities. The benefits include the avoided GHG damages and all avoided mortality and non-mortality benefits.

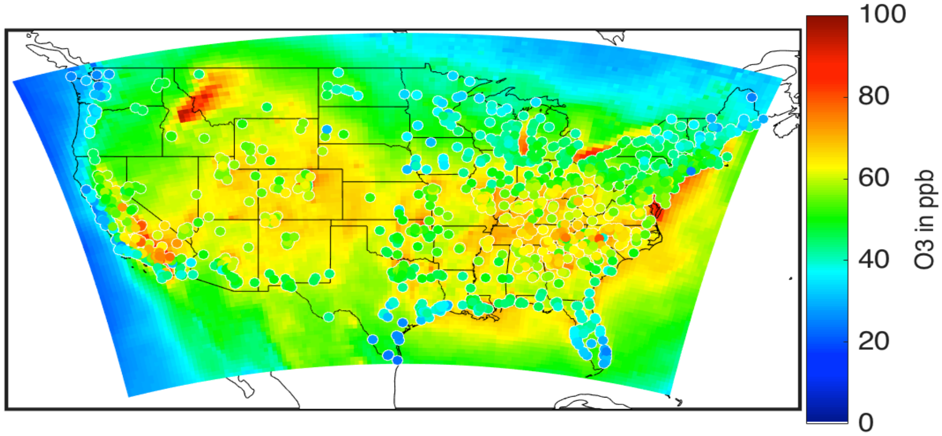
4.3 Results

4.3.1 Air Quality

Baseline CMAQ model simulations of O_3 and $PM_{2.5}$ for 2007 are first evaluated using in situ measurements. Figure 4.2(a) is a plot of the modeled estimate of the annual average $PM_{2.5}$ concentration for 2007 (based on the four modeled months), while the overlaid dots show the observed concentrations from AQS data (averaged over the same four months) for comparison. The correlation coefficient between the modeled and observed annual average concentration of $PM_{2.5}$ is 0.55, and the root-mean-square error (RMSE) is 4.93 $\mu g/m^3$. Figure 4.2(b) presents the daily maximum one-hour average O_3 concentrations across the US, averaged over the month of August, overlaid with AQS observations. We present August results for O_3 because this is the most relevant month for health impacts and policy compliance, since O_3 is generally highest in summer. Figure 4.15 in the appendix presents a scatter plot of daily 1hr maximum concentrations for model-observation pairs, including the occurrence of higher values expected in summer. The correlation coefficient for August ozone is 0.66, and the RMSE is 18 ppb. Additional evaluation plots and a statistical summary of model performance are included in the Appendix.



(a)



(b)

Figure 4.2 (a) Modeled and observed $PM_{2.5}$ concentrations in 2007. The average of the 24hr average over February, May, August, and November is plotted. (b) Modeled and observed ozone concentrations in 2007. The plot values are the average of daily 1 hour max (MDA1) for the month of August.

Modeled $PM_{2.5}$ concentrations are highest in November and February and lowest in May. Geographically, pollutant concentrations are higher in the eastern half of the US, the central valley of California, and Idaho. Locally, there are some fires that lead to high $PM_{2.5}$ concentrations during the time period they are active, sometimes in excess of 40 $\mu g/m^3$. For the fire events, the $PM_{2.5}$ is composed of approximately 45% primary organic carbon, 10% elemental carbon, and up to 40% unspeciated $PM_{2.5}$ mass. Elemental carbon concentrations are also high in urban centers where they make up 10-20% of the total $PM_{2.5}$ mass. $PM_{2.5}$ is otherwise mostly composed of sulfate (20-40%), nitrate (frequently around 30%), and ammonium (10-20%) aerosols. Secondary organic aerosol (SOA) is important in some areas, primarily the southern coastal states, where it may make up as much as a quarter of the $PM_{2.5}$ mass.

The emissions changes from present day are represented in the scaling factors (Table 4.2). The scaling factors are applied to all emissions, including non-energy related emissions, except for the excluded species mentioned above. The emissions changes for each fee case compared to the No Fee case are presented, by region, in Table 4.3. The emissions reductions presented in Table 4.3 are approximate, because they only account for the species specific reductions and do not include the additional reductions from the electric and industrial sector. In all future cases, concentrations of both ozone and $PM_{2.5}$ are reduced compared to the original 2007 concentrations, as we would expect with the reduced emissions. There are a few small areas where O_3 increases slightly, but the average MDA1 for August is never more than 3 ppb greater in the 2045 No Fee case compared to the 2007 concentrations.

$PM_{2.5}$ concentrations are lower across all regions with fees compared to the No Fee case. The reductions in concentration are smallest with GHG fees. The geographical distribution of concentration reductions is similar across cases. The largest absolute reductions in concentration occur with combined fees, but the combined fee case is very similar to the HIP fee case. The largest reductions in concentrations of both pollutants occur in region 3 (the Midwest, see Figure 4.1) because without fees this region has many highly polluting energy sources, including coal-fired power plants. For these emission-intensive sources, shifting to lower emitting technologies is less expensive than using the existing facilities and paying emissions fees. The concentration reductions, as a percentage of the No Fee concentration, tend to be largest in regions where the emissions reductions were largest. Figure 4.4 presents both the percent reduction in emissions compared to the No Fee case as well as the percent difference in annual average $PM_{2.5}$ concentrations from the No Fee case by region. This is also presented in table form in appendix Table 4.8. Region 8 (the Intermountain West) has the largest percentage decreases in $PM_{2.5}$ concentrations as well as the largest decrease in SO_2 emissions and large decreases in NO_x emissions. Although the large PM concentrations in Oregon are due to fires, there are reductions in the fee cases in Figure 4.3. This is due to the method of scaling emissions, which leads to reductions in non-anthropogenic emissions. In August, a large fire in Idaho also affects air quality (see appendix Figure 4.10c).

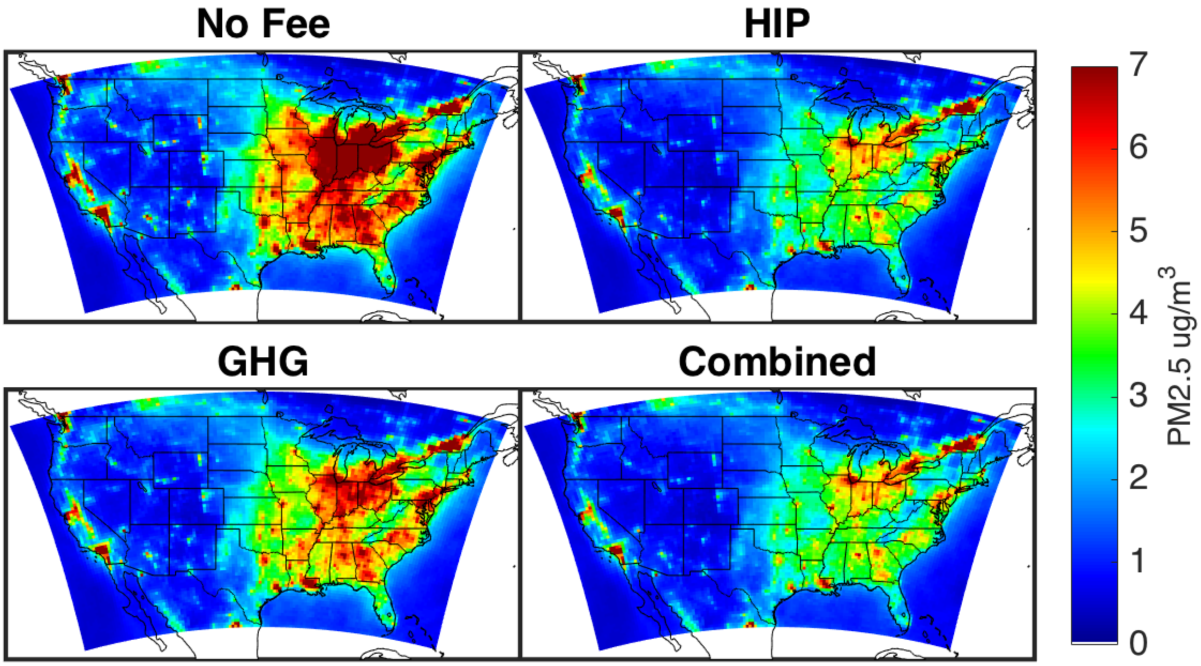


Figure 4.3 Annual average modeled $PM_{2.5}$ concentrations

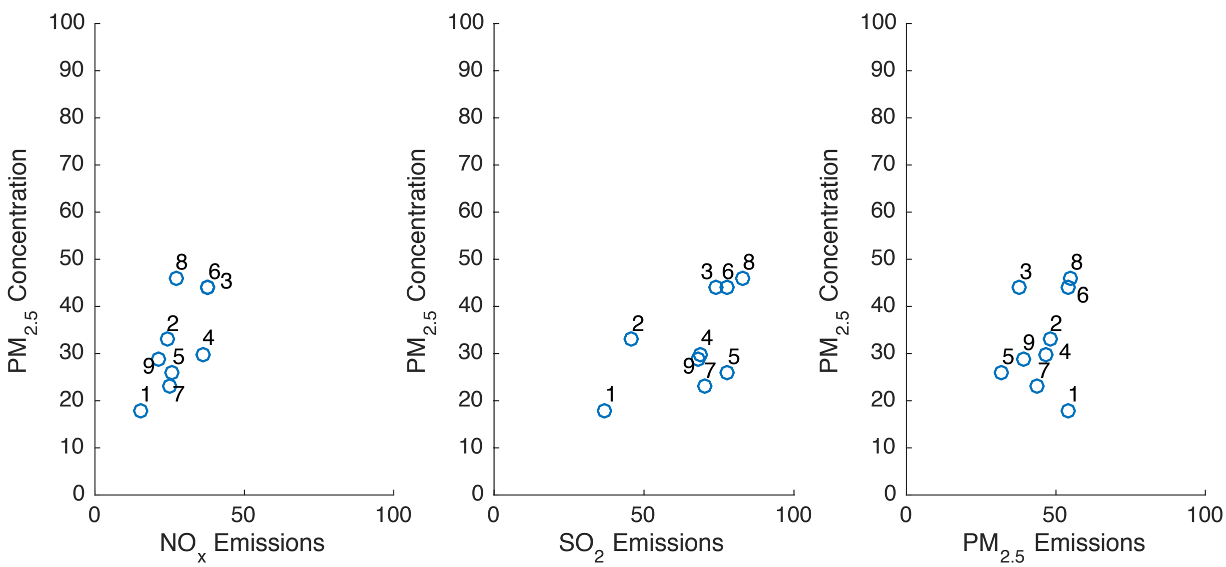


Figure 4.4 The relationship between percentage of emissions reductions and percentage reductions in $PM_{2.5}$ concentrations by region. The numbers label the region associated with each point. The emissions reductions are based on total MARKAL emissions reductions for each species by region and may not accurately represent the emissions reductions in CMAQ do to the inclusion of non-energy sector emissions in CMAQ.

Figure 4.5 displays the highest hourly average O_3 concentration per day (MDA1) averaged over the month of August for the No Fee case and the differences caused by fees. It is obvious from this that the ozone concentrations are reduced with all fees, and reduced further with HIP fees. While O_3 concentrations decrease over most of the US, there are some areas with a slight disbenefit. At the regional level, the smallest decrease in O_3 concentration is in region 8 with the large NO_x reductions mentioned above. Conversely, regions 3 and 6 have the largest reductions in NO_x and also the largest reductions in O_3 concentrations. This is an example of why CMAQ provides more information on the effect of policies than emissions projections alone. Due to non-linearities in O_3, decreases in emissions can sometimes lead to increases in concentrations. Regardless, O_3 concentrations in the majority of the country are still reduced by reducing emissions through damage based fees. The reductions in O_3 may be overstated in the results here, because the scaling factors that reduce NO_x and VOC emissions apply to some emissions that are not associated with energy and would therefore not be reduced by energy related fees, specifically in Idaho.

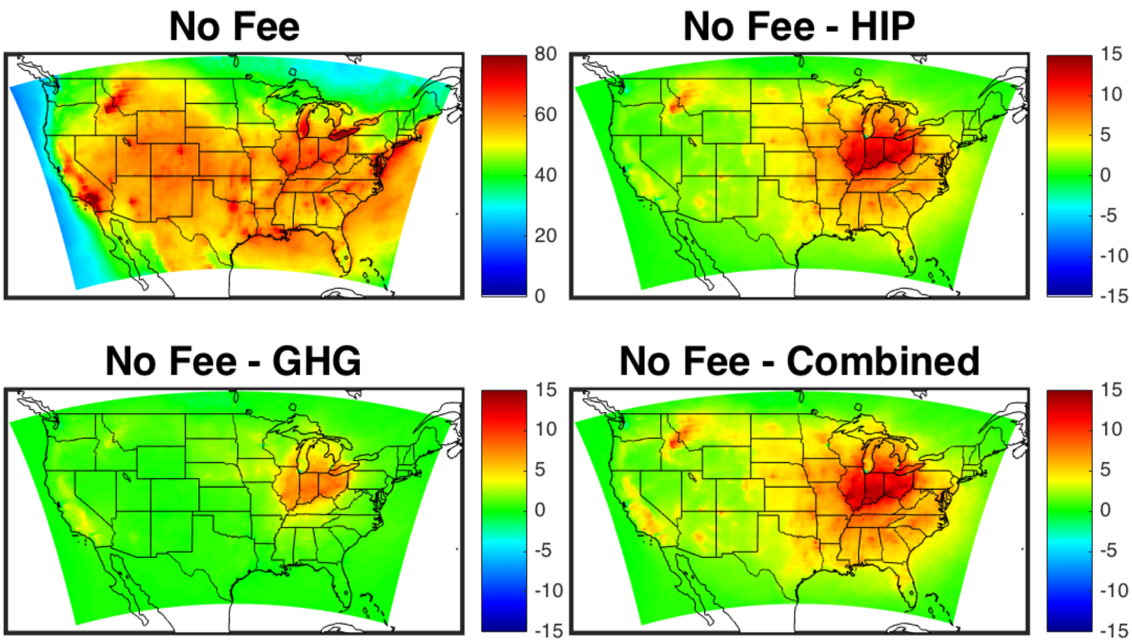


Figure 4.5 Average of the highest hourly average O_3 concentration each day, or MDA1, in August in ppb for the No Fee case and the difference in ppb of each fee case compared to the No Fee case.

We also present results for ozone in terms of frequency with which maximum daily eight hour average (MDA8) concentrations exceed the standard of 70 ppb in August. The number of days where the ozone MDA8 over land exceeds 70 ppb decreases when fees are applied, as shown in Figure 4.6. In all fee cases, there are fewer locations that exceed 70 ppb compared to the No Fee case. The GHG fee case has more exceedances than the cases where fees are applied to HIPs. With HIP or combined fees, there is an increase of up to three days per year in the number of days of O_3 exceedances in Los Angeles. With GHG fees, there is also an increase of up to two days per year in number of days with exceedances in Los Angeles, as well as near the Texas and Louisiana coast and a few other grid cells.

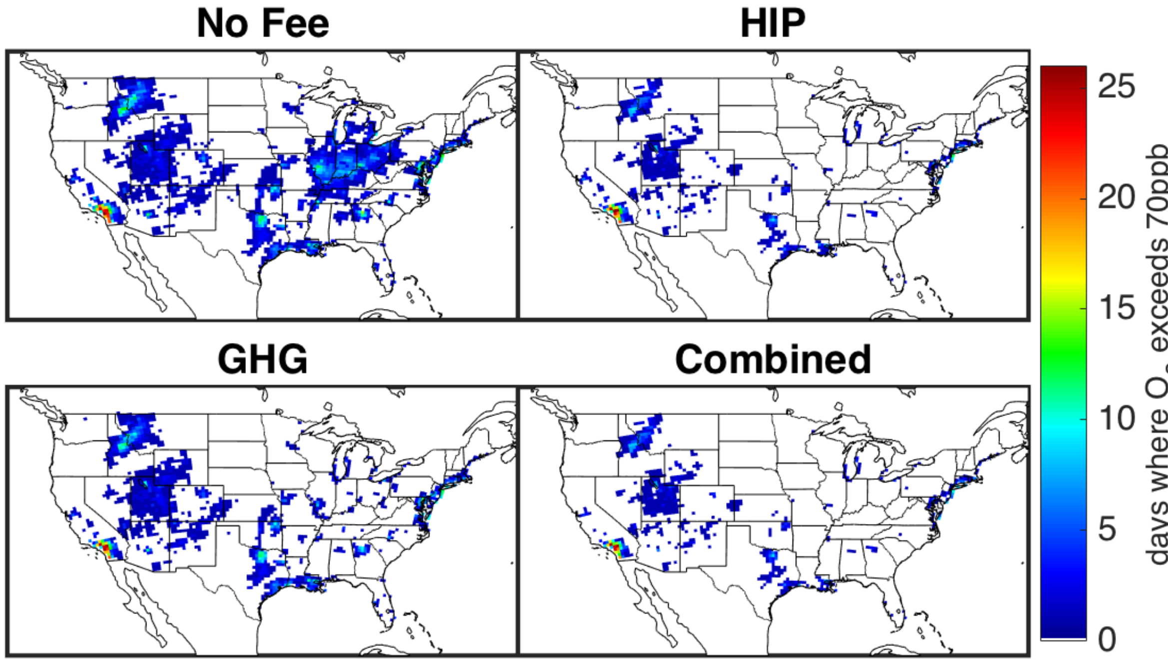


Figure 4.6 Number of days in August where O_3 MDA8 exceeds 70 ppb, only when such exceedances occur over the US land areas. For the No Fee case there were 5106 total exceedances for the 31 days across all 6019 applicable grid cells, with the most in a single cell being 26. With HIP fees, 1577 total exceedances occur in August with the largest in any one cell being 26. For GHG fees, 25 is the largest in any single location and 3031 total exceedances occur. For combined fees 25 is the largest in any single cell and 1428 exceedances occur.

4.3.2 Costs and Benefits

The changes in costs from the MARKAL results are presented in Table 4.3. "Cost Increase" represents the increase in cost of the energy system in 2045 when fees are applied compared to the base case, presented in 2005 USD. The costs in Table 4.3 are calculated with investment costs annualized over the lifetime of the technology. Revenues collected in the form of fees are not included in this cost increase, because this expense could be used in many ways, including offsetting the cost of emissions reductions in the energy system or using them to reduce care costs for the remaining externalities.

Therefore, we present this value separately in the "Fees Collected" column. The cost increases represent only 1-2% of the total system cost, and the fees collected increase the total energy system cost by 6-15%. We do not present regional cost results in MARKAL, because not all costs are assigned to a region. End use processes are assigned to a region, but upstream processes such as fuel extraction can have large costs that are not allocated to a region. Also, inter-regional trading is allowed so investment in a power plant in one region may supply demand in a different region.

Table 4.3 Energy system costs associated with fees from MARKAL in million 2005 USD.

	Cost Increase	Fees Collected
HIP	*$76,322*	$271,783
Combined	*$91,273*	$691,387
GHG	*$48,315*	$439,874

We analyze the mortality benefits by region, with our findings presented in Table 4.4. The benefit results for each fee case indicate benefits compared to the 2045 No Fee case. Region 3 is projected to have large concentration reductions (see Figs. 4.3-4.6), and, as displayed in Table 4.4, there are large mortality related benefits for this region as well. In region 9 for the HIP fee case, there is an ozone disbenefit associated with concentration increases in highly populated areas of southern California. To accommodate such disbenefits, one possible approach could be that money collected from the fees could be used to offset this disparity by helping to subsidize medical bills that may be related to high ozone concentrations. Region 8 has a large percentage decrease in $PM_{2.5}$ concentrations, but the low concentration in the No Fee case and the low population density lead to low benefits. The percent reduction of ozone is largest in region 6, but the reduction in health impacts for ozone is much larger in region 5 because there are reductions in more populated areas where there is a larger number of potentially impacted people.

Following the trend for air quality, the benefits are largest for the combined fee case and lowest with GHG fees. The $PM_{2.5}$ benefits for combined fees are only 3% larger than those for HIP fees, but there is more distinction in the O_3 benefits, where there is a 22% increase in O_3 related mortality benefits with combined fees compared to HIP fees alone. The air quality related co-benefits of GHG fees are much lower. The O_3 related mortality benefits with GHG fees are only 24% as large as the O_3 benefits with HIP fees. The $PM_{2.5}$ mortality related co-benefits are 49% as large as the HIP fee $PM_{2.5}$ benefits. As the O_3 benefits are so much smaller than the $PM_{2.5}$ benefits, the difference in total mortality reduction is within one percent of the difference in $PM_{2.5}$ benefits.

Table 4.4 Mortality related benefits by region in millions of USD.

		Nat'l	R1	R2	R3	R4	R5	R6	R7	R8	R9
HIP	PM2.5	$403,381	$9,956	$41,830	$98,881	$26,900	$76,628	$31,877	$46,553	$19,355	$51,402
	O3	$8,998	$100	$350	$995	$780	$3,275	$1,610	$1,136	$1,011	-$258
Combined	PM2.5	$414,323	$10,136	$42,592	$100,008	$27,283	$78,603	$32,551	$47,615	$19,819	$55,717
	O3	$10,970	$121	$448	$1,094	$829	$3,569	$1,691	$1,278	$1,237	$703
GHG	PM2.5	$198,403	$4,808	$16,579	$56,610	$14,447	$27,275	$15,741	$18,471	$11,260	$33,213
	O3	$2,172	$27	$135	$262	$187	$512	$452	$68	$283	$248

Although the benefit valuation is dominated (99%) by reduced premature mortality, there are other avoided health impacts associated with the fee cases analyzed here. The non-mortality health impacts, presented in Table 4.5, are much smaller in monetary terms than the mortality impacts. While we present total valuation here, incidence and values for individual health endpoints can be found in the Appendix.

Table 4.5 The benefits associated with the fee cases in Millions of USD. The Damage-scaled emissions are calculated using Equation 4.6 for GHG and HIP damages using the emissions changes from MARKAL.

	Damage Scaled Emissions		Mortality Benefit			Non-Mortality Benefit	
	GHG Scaled MARKAL Emissions	HIP Scaled MARKAL Emissions	$PM_{2.5}$ adult mortality	$PM_{2.5}$ infant mortality	O_3 mortality	O_3 non-mortality	$PM_{2.5}$ non-mortality
HIP	$73,057	$256,775	$402,739	$642	$8,998	$137	$3,761
Combined	$101,186	$260,412	$413,664	$659	$10,970	$169	$3,857
GHG	$83,839	$154,782	$198,093	$309	$2,172	$31	$1,879

By comparing the damage scaled emissions to the benefits we can evaluate how well the less computationally intensive scaling calculation performs in terms of estimation of nation-wide benefits. The main source of the differences between the damage scaled emissions and the calculated benefits is due to population increase. The Fann et al. (2012) values are based on population in 2016. In the Fann et al. (2012) paper they calculate the benefits per ton of emissions for 2005 and 2016 and find an average increase of 36% in benefit per ton over only eleven years. If we calculate benefits for a 2016 population for the HIP fee case keeping all else equal, we find adult mortality values based on Krewski et al. (2009) of $233 billion, infant mortality based on Woodruff et al. (2006) of $548 million and total $PM_{2.5}$ benefits of $235 billion. This is actually slightly less than our damage scaled emissions calculation. It illuminates that there is a 73% increase in benefits due only to the increase in population. This indicates the importance of re-evaluating fees over time.

We calculate benefits using the same mortality health impact functions used by Fann et al. (2012), which was the basis of our fee estimates, and many of the same health impact functions for the non-mortality health impacts. We also consider some health impact functions that were not available when the damage estimates were calculated. Fann et al. (2012) used CAMx with 12km grid resolution to calculate the air quality associated with their emissions scenarios, so there may be differences between CAMx and CMAQ. We also aggregate some of the industrial sectors in determining our fees as not all of the industrial sectors that benefits are calculated for have exact matches in the MARKAL database. For instance, iron and steel manufacturing has higher benefits/ton, which might be captured in our benefits estimates, but is not captured in our damage-scaled emissions. The Fann et al. (2012) study on which the fees are largely based only calculated damages for $PM_{2.5}$, so all O_3 benefits are expected to be in addition to the damage scaled emissions. Another reason for the discrepancy in benefits calculated with BenMAP-CE compared to the damage scaled emissions has to do with the difference in impacts of emissions from different facilities. Although we calculate emissions reductions by region, the location of 2007 facilities within the regions provides more local information than just reducing concentrations by a certain percentage across the region. With the damage scaled emissions approach, the location dependence is only captured by the source type, as the marginal damages used are the same across the country, although population density across the country is not. Using sector based damage estimates captures the typical proximity to large urban areas, but the combined CMAQ and BenMAP-CE methodology goes a step further to also account for the size of those urban areas. Also, more of the non-linearities associated with ozone are captured using CMAQ and BenMAP-CE, although this contributes a smaller fraction of the total damages.

Analyzing damage scaled emissions also allows us to trace the benefits to emissions of specific pollutants. Figure 4.7 presents the damage scaled MARKAL emissions for the HIP fee case separated by pollutant and sector. Although $PM_{2.5}$ emissions are small compared to emissions of other pollutants, the damage scaled emission value for $PM_{2.5}$ is large because the marginal damages associated with $PM_{2.5}$ are significant. This is because $PM_{2.5}$ concentrations lead to large health effects, and all $PM_{2.5}$ emissions contribute to $PM_{2.5}$ concentrations as opposed to emissions of precursor species where only a fraction contributes to $PM_{2.5}$ concentrations. There is significant uncertainty in the benefit of $PM_{2.5}$ emissions reductions, however, especially considering the large fraction associated with upstream emissions. Most of the $PM_{2.5}$ upstream emissions reductions are associated with extraction processes such as mining, which are likely to occur further away from urban centers. In the damage scaled emission reductions, this means that the marginal damage value used may overestimate the marginal damages associated with mining, as the damage value assumes that upstream emissions are averaged across electricity generation, industrial use, and transportation. Similar uncertainties may affect the benefits estimated using CMAQ and BenMAP. In the CMAQ modeling, upstream emissions reductions are applied to all $PM_{2.5}$ emissions sources although the bulk of the reductions occur at a few remote sources. The upstream emissions rates are also likely to be much less certain than emissions from sources such as electricity.

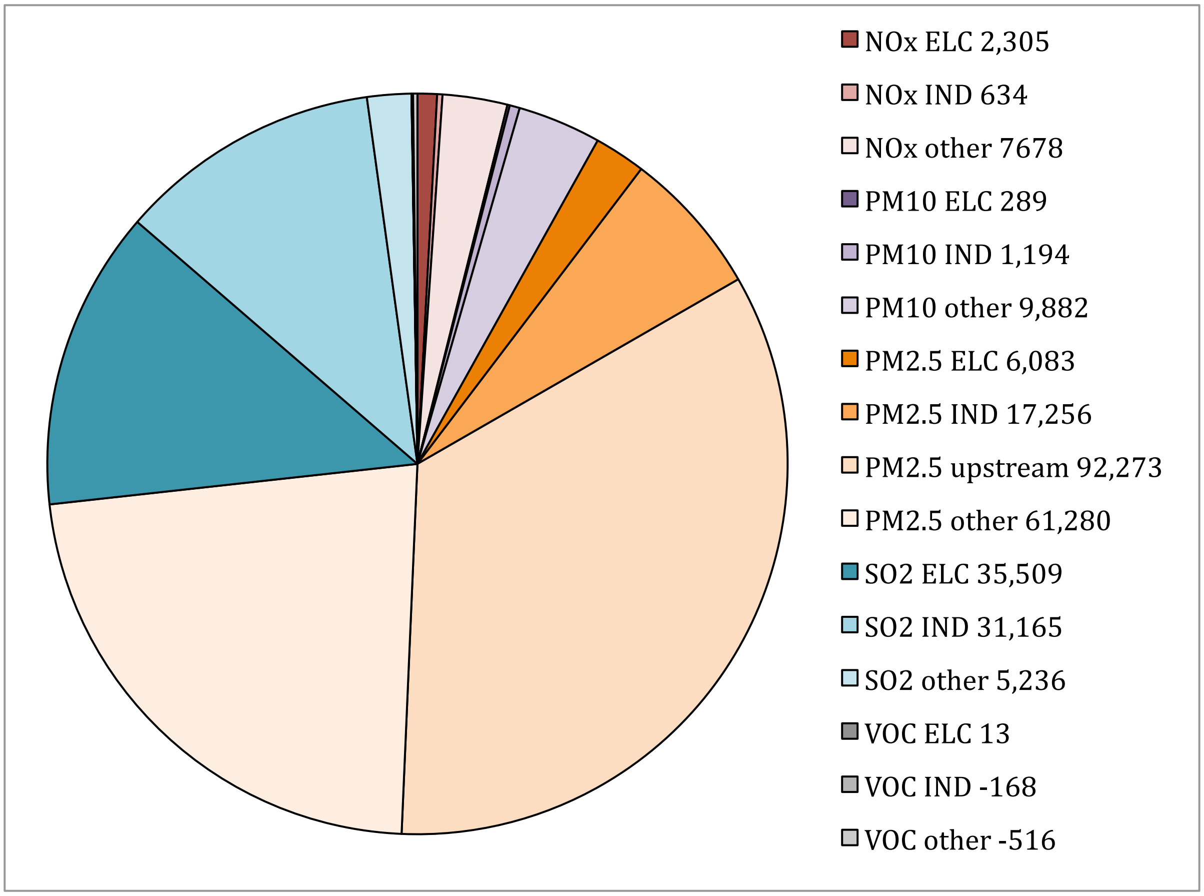


Figure 4.7 Damage scaled emissions by pollutant and emission source for the HIP fee case. Values are in Million USD. IND emissions are for the industrial sector and ELC emissions are for the electric sector.

We next consider the net benefits of the air quality and climate policies. The net benefits are the sum of the health and climate benefits less the cost increase from MARKAL. To calculate the health benefits, we sum over the mortalities and O_3 and $PM_{2.5}$ non-mortality benefits from BenMAP-CE in Table 4.5. For climate benefits, we use the GHG damage scaled emissions value in Table 4.5, and the costs can be read from the "Cost Increase" column in Table 4.3. For 2045, the total health benefits from the HIP fee case are \$416 billion, the climate benefits are \$73 billion, and the net benefits are \$413 billion. The total health benefits from the GHG fee case are \$202 billion, the climate benefits are \$84 billion, and net benefits are $238 billion. The total health benefits for the combined fee case are $439 billion with climate benefits of $101 billion and net benefits of $338 billion.

4.4 Discussion

When damage-based fees on emissions are included in the US energy system, air quality improves for almost all areas of the country. As expected, the health related benefits and air quality improvements are largest for the combined fee case and lowest for the GHG fee case. However, the estimated benefits are only slightly higher for the combined fee case than the HIP fee case, and the net benefits are highest for the HIP fee case. This is because of additional cost increases associated with reducing GHG emissions that are included in the combined fee case and smaller benefits associated with GHG emission reductions. Pollutant concentrations are typically reduced most in the regions where the emissions are reduced most, but in some regions O_3 concentrations change very little or increase locally despite decreased emissions. As there are few options to reduce VOC emissions in the MARKAL model, the unchanging O_3 may occur in areas that are VOC limited, in which case the reduction in NO_x emissions has little effect on the O_3 concentrations.

When fees are not considered as part of the cost, benefits are larger than the costs of the policy. Although the fees paid increase the energy price, the associated emissions also affect the well-being of the population to a similar degree. If the fees collected are put toward reducing the cost of externalities by those affected, it could further reduce the cost of the larger energy-environment-health system. Otherwise, the fees paid will be government revenue and may offset other taxes paid or would otherwise improve social welfare. Due to the large quantity of fees collected, it is worth careful consideration of how that revenue is used.

The GHG damage scaled emissions provide the climate benefit or co-benefit of the policies. The climate co-benefit is always smaller than the air quality benefit, but some studies (Moore and Diaz, 2015; Dietz and Stern, 2015; Lontzek et al., 2015) suggest that climate damages may be an underestimate as there are still many uncertainties surrounding climate change predictions. Even in the HIP fee case, the climate co-benefits are only 13% less than the climate benefits of the GHG fee case. The health benefits for the GHG policy here are similar to those found in Thompson et al. (2014) for health co-benefits of climate policies, which studied a wider range of climate policies, but did not investigate air quality or combined policies.

The benefits associated with $PM_{2.5}$ in this paper are larger than those found in the PM NAAQS RIA (U.S. EPA, 2012b) because the changes in air quality are larger and occur across more of the US and we are analyzing a year further in the future where the population is larger, so there are more people that could be affected by pollution. The O_3 related impacts are similar to those found in the Ozone NAAQS RIA (US EPA, 2015a).

The largest component of the benefit estimate is mortality associated with $PM_{2.5}$ due to the large monetary value associated with mortality as well as the large decreases in modeled $PM_{2.5}$ concentrations. The lowest measured level (LML), or lowest measured $PM_{2.5}$ concentration, for determining the health impact functions is higher than some of the concentrations modeled here, which means that there is uncertainty in the estimates for $PM_{2.5}$ benefits because the shape of the health impact functions may be different at these lower concentrations. For Krewski et al. (2009) the LML is 5.8 $\mu g/m_3$. It is obvious from Figure 4.3 that $PM_{2.5}$ concentrations in 2045 in some regions are often projected to be below this level, thus there is uncertainty in the $PM_{2.5}$ related benefits estimates in these regions. There are also uncertainties in the health impact functions themselves for concentrations above this threshold. The 95th percentile confidence interval for the valuation of Krewski et al. (2009) benefits given the population and air quality in this study ranges by an order of magnitude.

Due to complex chemistry and transport as well as uneven population distribution, we expected a significant difference between the benefits calculation and the damage scaled emissions. As exemplified by the effect of population growth, the population exposed to emissions is highly important in evaluating benefits. We expected that by combining CMAQ and BenMAP we would be better able to capture the differences in marginal damages across emission locations, since the damages present an average value. One reason the differences may be small is that, within a region, the emissions from similar source types are reduced equally, so if an individual facility has much different marginal damages that is not captured here. We also evaluated aggregated results to regional and national levels, and the difference between the two methods may be larger for individual counties. It is possible that there are more differences if the difference between the two methods were to be analyzed at a finer resolution, either by reducing emissions more specifically than by the nine regions used here, or if the air quality modeling and benefit calculation were to be performed at a higher resolution, which may better capture localized dense populations (Punger and West, 2013). Similarly, some location information is captured in the damage scaled emissions by using different values for different sectors. An analysis using a single marginal damage estimates for all emissions may not match the more rigorously calculated values as closely.

The treatment of emissions changes through scaling factors introduces some uncertainty in the results as well. Because this methodology reduces some emissions not related to energy use, our results likely over estimate the air quality and health benefits associated with the fees. One example of this is the improvement in air quality near biomass burning events, which in practice would not occur due to fees. The MARKAL modeling also introduces uncertainties because it is an inelastic model; therefore, we cannot capture the effect fees may have on reducing demand for energy services, such as a reduction in travel due to higher gasoline prices with fees. There is also the possibility that the fees will lead to the development of newer, less expensive technologies that reduce emissions. The model captures similar processes by, for example, decreasing solar costs over time, but does not capture more novel technologies or the possibility of faster decreases in cost.

The results of this research can inform policy decisions about similar fee structures. One important point that future policy makers should take away from these results is that for this policy to be cost-neutral, the ultimate use of the fees is important. In some cases, small areas experience air quality dis-benefits from the fees, in which case the revenue could be put toward off-setting some of the costs associated with that worsened air quality. Another important result relevant to policy is the impact of population on the benefits. Given that this is such an impactful component, fees should be re-evaluated over time. Also, in calculating the benefits of any policy, whether that policy uses fees or not, the population change should be factored into the valuation of benefits. This is typically done when evaluating policy benefits with BenMAP, but if using damage scaled emissions to estimate benefits of future policy, the damages should be scaled or take the population increase into account in some way. For instance, if comparing the benefit of several policies in a future year, damage scaled emissions could be used if the damages are calculated based on the future year of interest.

Although there are uncertainties associated with estimating the effect of future emissions reductions, it is still an important exercise to undertake when considering policies. Additional information about likely locations of concentration reductions associated with a policy leads to more informed benefit calculations. Analyzing the concentration changes associated with emissions changes is also useful considering the non-linearities in air chemistry particularly for ozone benefits. We also find that the changing population with time has a large impact on the benefit of emissions reductions.

4.5 Appendix

4.5.1 CMAQ validation

We ran CMAQ before applying the factors shown in Table 4.1 and compared these concentrations to measured concentrations for 2007 to validate the model run. The observations and model results are compared for ozone, $PM_{2.5}$, sulfate, and nitrate below. The monthly average $PM_{2.5}$ concentrations at each location are compared and the daily 1st max hour concentration of O_3 are compared. The observation data comes from EPA's AQS Data Mart (O. US EPA, 2016). Values listed as annual here refer to comparisons of February, May, August, and November model/observation pairs all at once.

4.5.1.1 $PM_{2.5}$

Table 4.6 $PM_{2.5}$ evaluation statistics (MdnB is the median bias in μg/m^3, MdnE is the Median error in μg/m^3, NMdnB is the normalized median bias in %, NMdnE is the normalized median error in %, NMB is the normalized mean bias in %, NME is the normalized mean error in %, RMSE is the root mean squared error in μg/m^3, and R is Pearson's correlation coefficient.)

PM2.5 total	MdnB	MdnE	NMdnB	NMdnE	NMB	NME	RMSE	R
February	5.98	6.64	54.68	60.67	51.63	67.69	9.63	0.37
May	-3.05	3.63	-24.90	29.66	-23.59	35.30	5.68	0.58
August	-3.20	4.15	-22.15	28.70	-18.19	37.98	8.80	0.47
November	4.74	5.60	39.76	46.97	35.55	50.98	8.18	0.41
Annual	0.28	2.88	2.16	22.19	5.09	28.04	4.93	0.55
Sulfate	MdnB	MdnE	NMdnB	NMdnE	NMB	NME	RMSE	R
February	0.44	0.46	168.64	175.53	158.68	168.24	0.70	0.48
May	0.43	0.50	69.39	73.43	84.08	93.83	1.00	0.62
August	0.28	0.36	35.02	45.16	45.45	71.51	1.22	0.58
November	0.47	0.47	100.92	102.14	107.96	116.99	0.72	0.60
Nitrate	MdnB	MdnE	NMdnB	NMdnE	NMB	NME	RMSE	R
February	1.12	1.12	329.83	329.83	273.99	280.70	1.66	0.54
May	0.10	0.12	81.80	100.73	168.12	191.56	0.71	0.73
August	0.00	0.08	1.81	95.01	160.99	211.67	0.68	0.54
November	1.09	1.13	447.20	463.97	333.91	345.58	1.88	0.46

The equations used to calculate the values in the above table (and the table for O_3 below) are:

$$
\begin{aligned}
\text{MdnB} &= \text{median}(C_M - C_o)_N && (7)\\
\text{MdnE} &= \text{median}|C_M - C_o|_N && (8)\\
\text{NMdnB} &= \frac{\text{median}(C_M - C_o)_N}{\text{median}(C_o)_N} \times 100\% && (9)\\
\text{NMdnE} &= \frac{\text{median}|C_M - C_o|_N}{\text{median}(C_o)_N} \times 100\% && (10)\\
\text{NMB} &= \frac{\sum_1^N (C_M - C_o)}{\sum_1^N C_o} \times 100\% && (11)\\
\text{NME} &= \frac{\sum_1^N |C_M - C_o|}{\sum_1^N C_o} \times 100\% && (12)\\
\text{RMSE} &= \sqrt{< (C_M - C_o)^2 >} && (13)
\end{aligned}
$$

where C_M is the modeled conc, C_o is the observed concentration, N is the number of model-observation pairs. MdnB is the median bias, MdnE is the median error, NMdnB is the normalized median bias, NMdnE is the normalized median error, NMB = normalized mean bias, NME = normalized mean error, RMSE = root mean square error, and R is Pearson's correlation coefficient. The angled brackets represent an average.

The model validation for our 2007 case does not perform as well as the 2007 case that EPA used in the Regulatory Impact Analysis (RIA) for the PM National Ambient Air Quality Standards (NAAQS) revision. We use a more recent CMAQ version here (5.0.2 compared to 4.7.1). We also use a lower resolution of 36km compared to the EPA 12 km. In the RIA, sulfate model predictions were biased low compared to observations. They found a normalized mean bias of -25%. Our modeled sulfate is less biased, but the normalized mean error for each season is higher than that found in the RIA. For nitrate, the bias in the RIA varies by season and location, and the average normalized mean error is 71%. The normalized mean error for nitrate in our evaluation is higher, but not above the range of values seen in the RIA on a regional basis.

The scatter plots and histograms show each day in comparison, e.g. Feb 1 24hr average observation and Feb1 24 hr average modeled concentration at that location, for each observation location and day modeled. This section presents validation plots for total $PM_{2.5}$ concentration, speciated results are presented later.

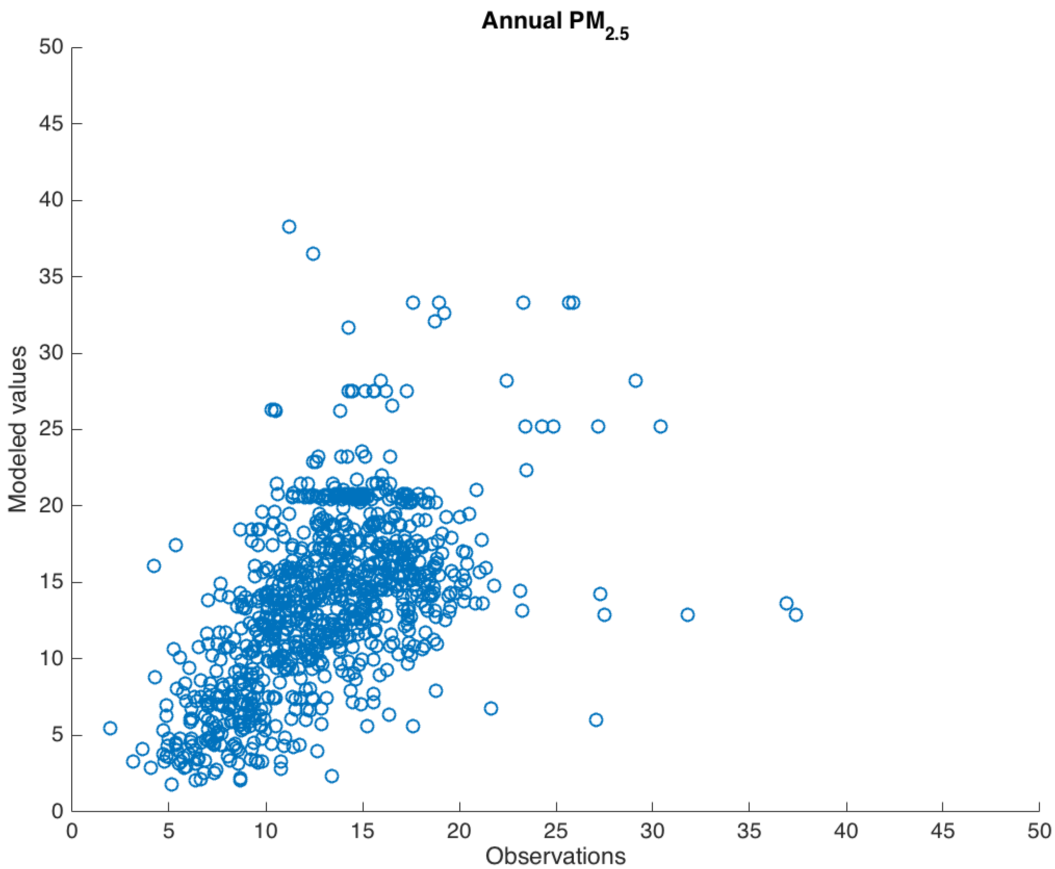


Figure 4.8 $PM_{2.5}$ validation scatter plot

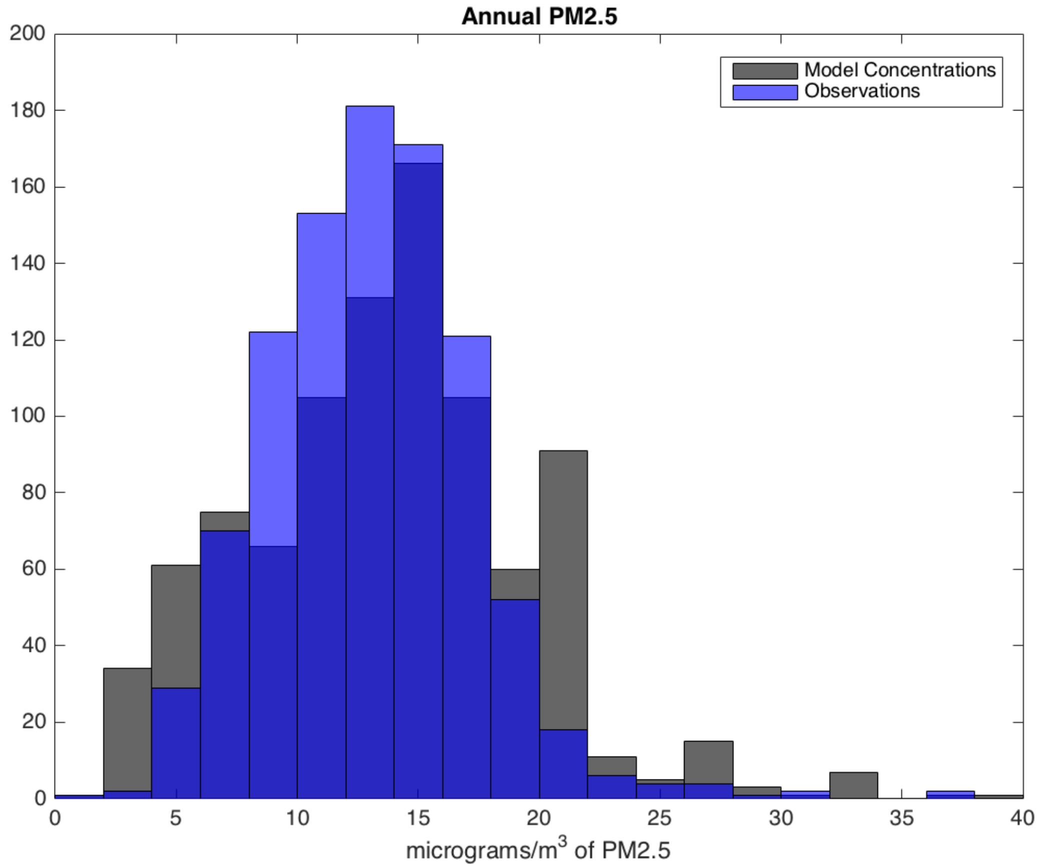
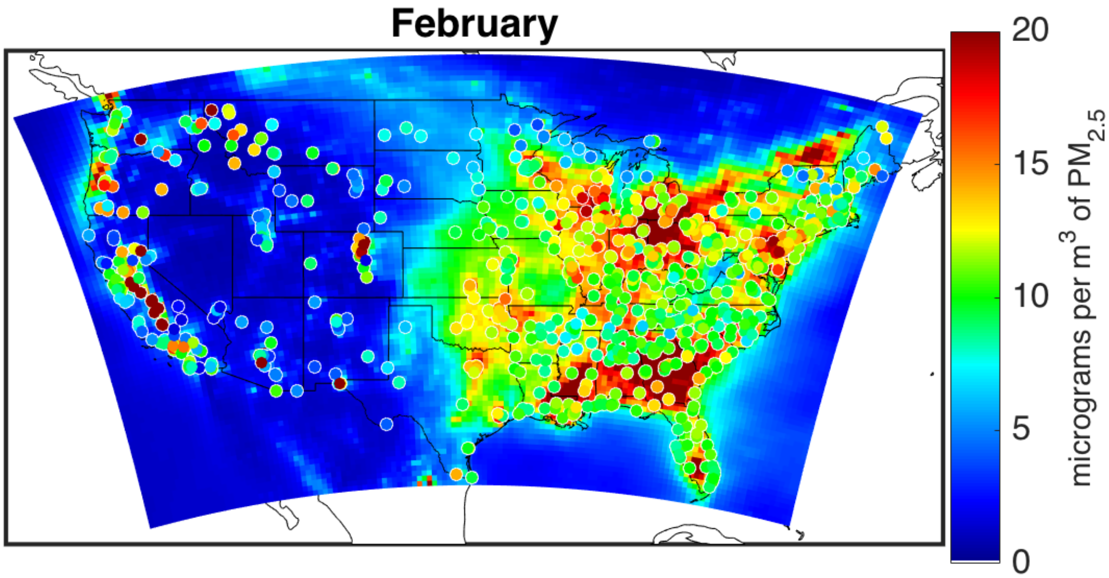
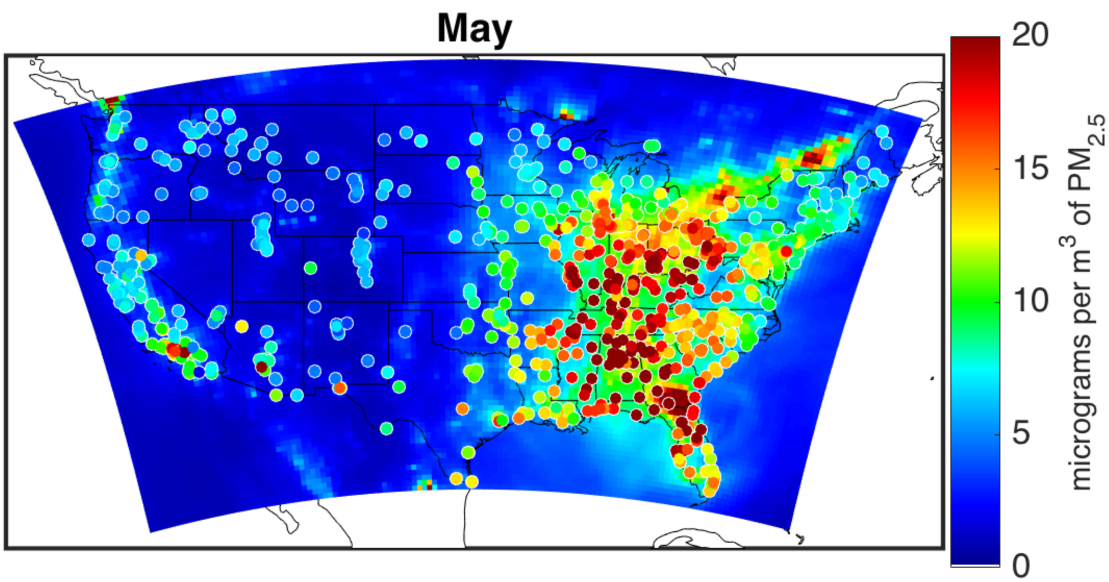


Figure 4.9 $PM_{2.5}$ validation histogram.

The map plot shows the mean value of the 24 hour average concentrations across the month.


February
20
15
10
5
0
micrograms per m3 of PM2.5


May
20
15
10
5
0
micrograms per m3 of PM2.5

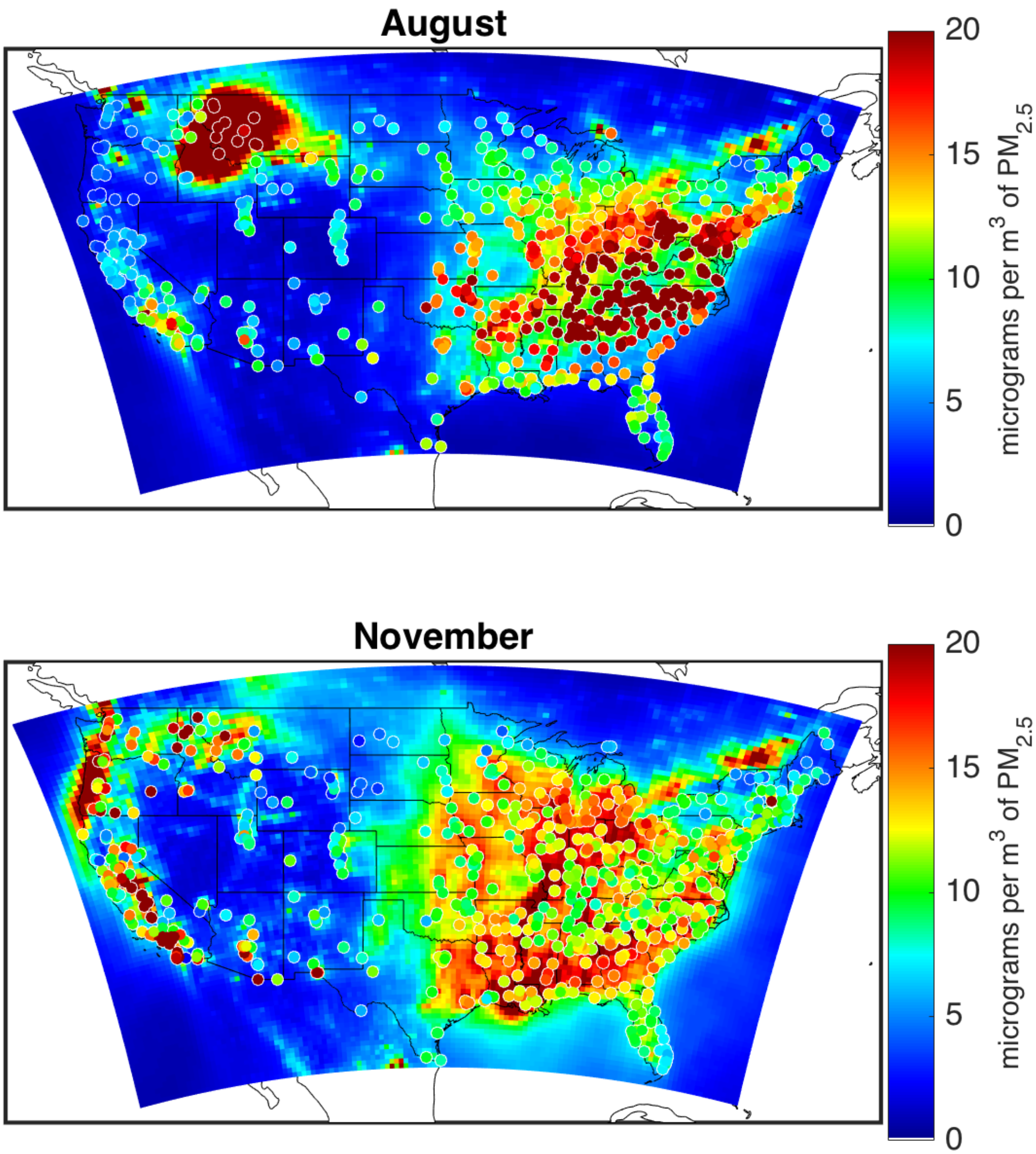


Figure 4.10(a-d) Monthly maps of modeled and observed $PM_{2.5}$ for 2007.

We also looked at individual species performance for sulfate and nitrate.

Sulfate

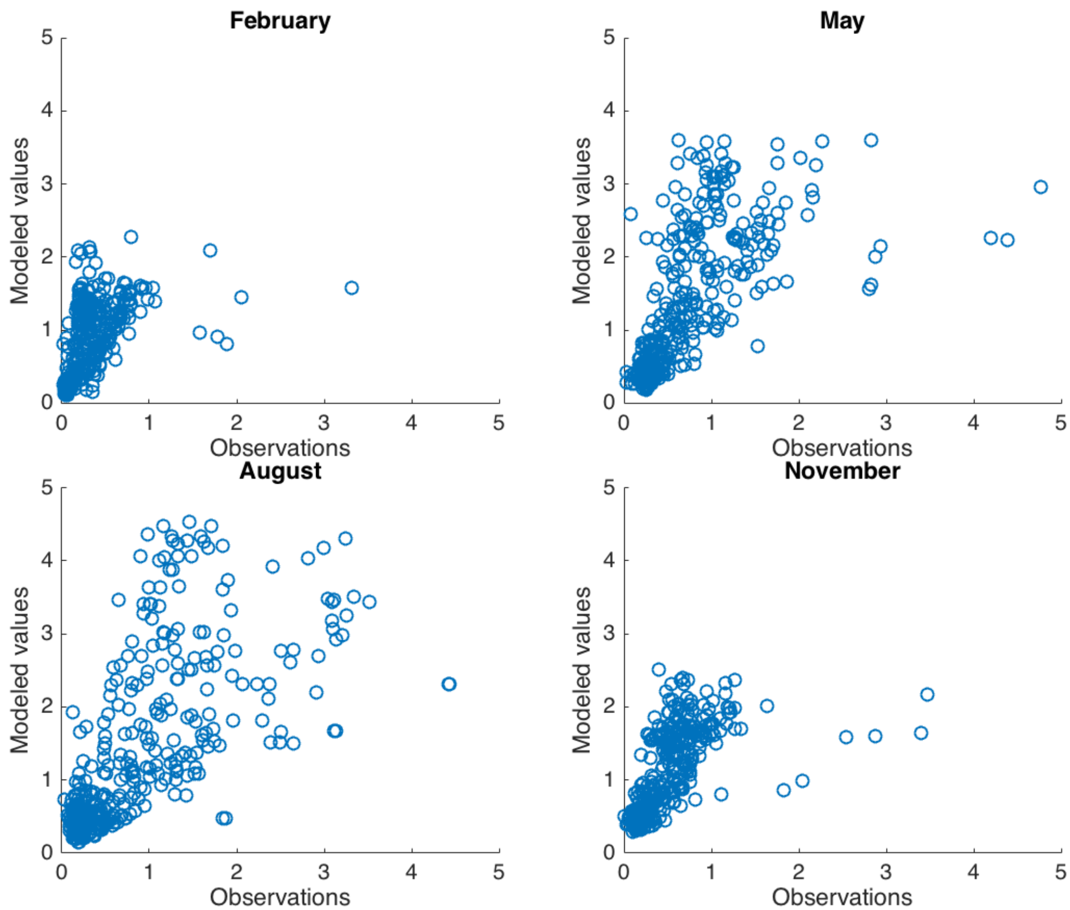


Figure 4.11 Sulfate validation scatter plots.

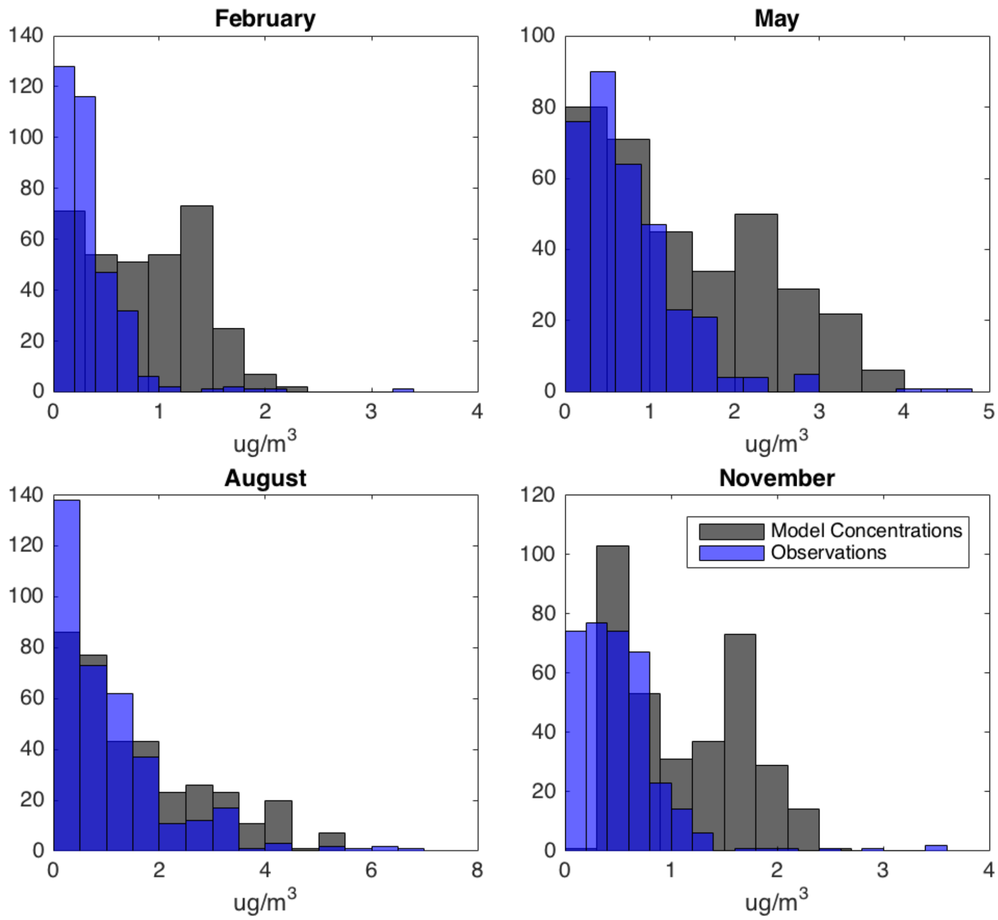


Figure 4.12 Sulfate validation histogram.

Nitrate:

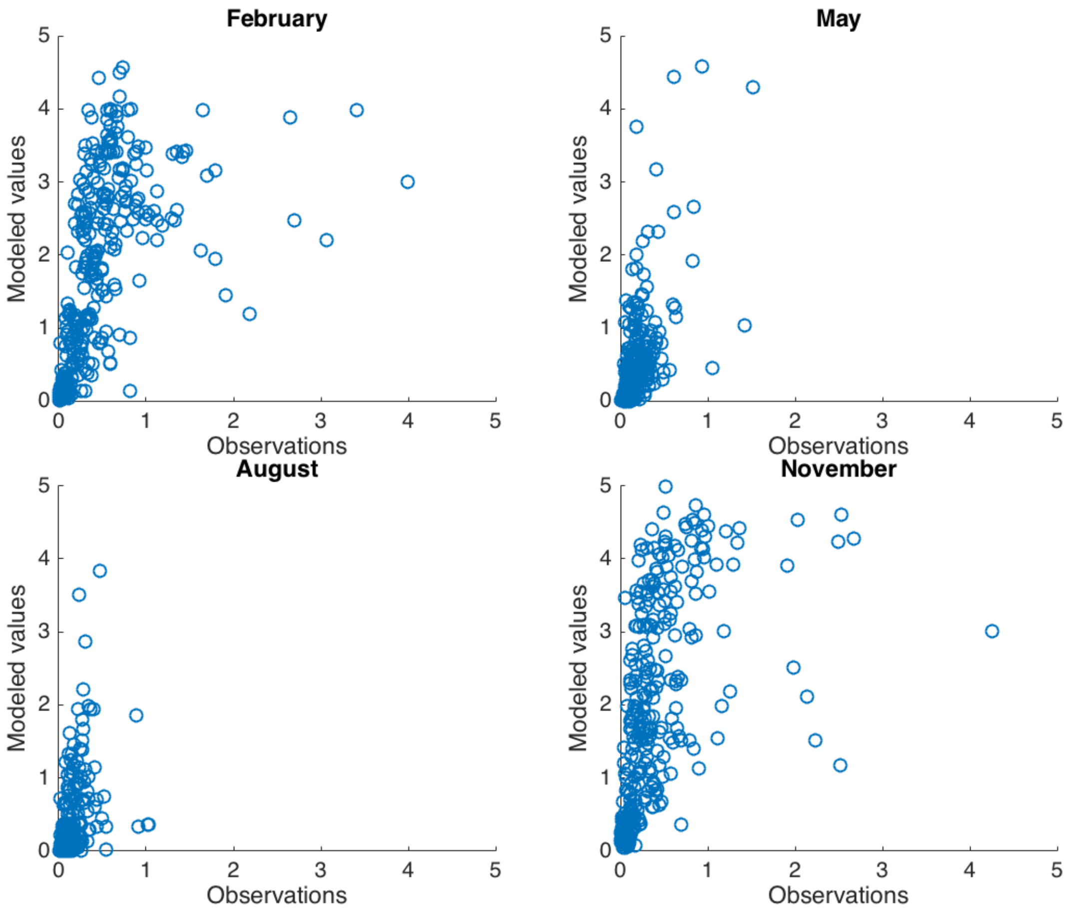


Figure 4.13 Nitrate validation scatter plots.

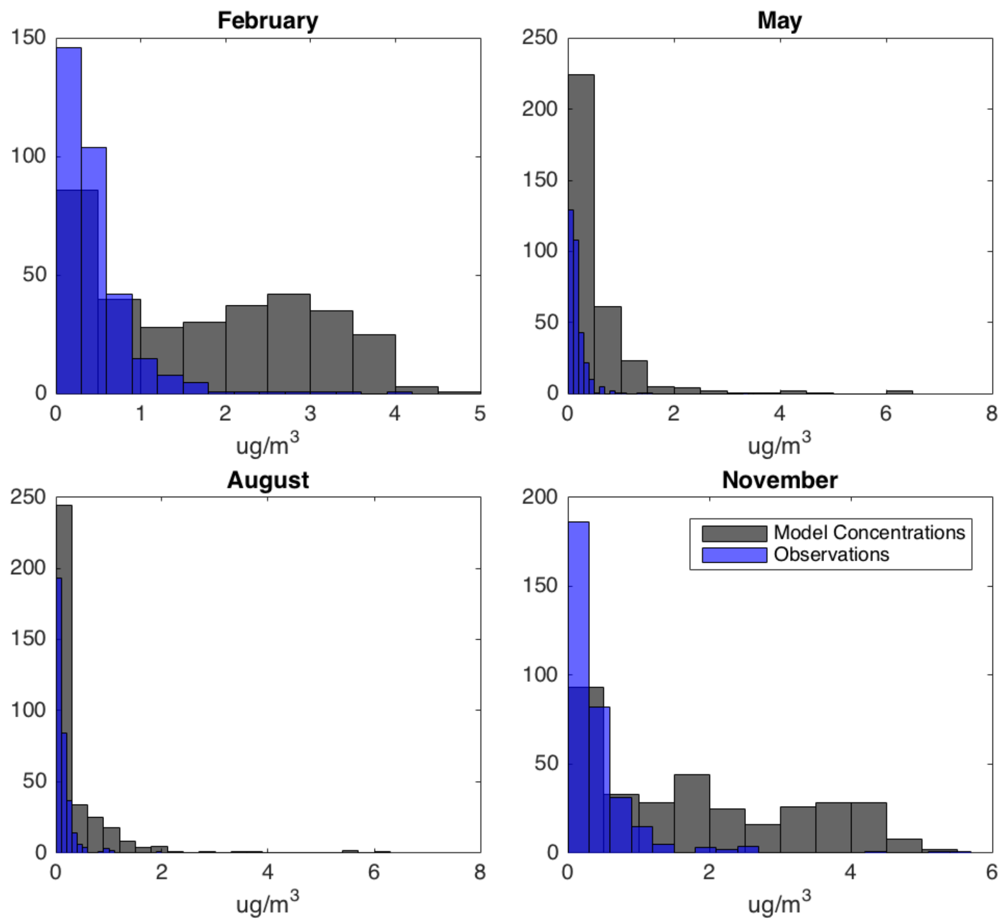


Figure 4.14 Nitrate validation histograms.

4.5.1.2 Ozone

The same analysis is also performed for ozone.

Table 4.7 Ozone evaluation statistics (MdnB is the median bias in ppb, MdnE is the Median error in ppb, NMdnB is the normalized median bias in %, NMdnE is the normalized median error in %, NMB is the normalized mean bias in %, NME is the normalized mean error in %, RMSE is the root mean squared error in ppb, and R is Pearson's correlation coefficient.)

O3	MdnB	MdnE	NMdnB	NMdnE	NMB	NME	RMSE	R
February	3.68	4.79	10.47	13.62	8.80	15.05	6.49	0.65
May	6.94	7.02	12.94	13.08	13.57	14.51	9.12	0.65
August	14.18	14.18	27.95	27.95	30.74	31.09	17.95	0.66
November	6.99	7.01	21.11	21.15	22.89	23.54	9.10	0.72

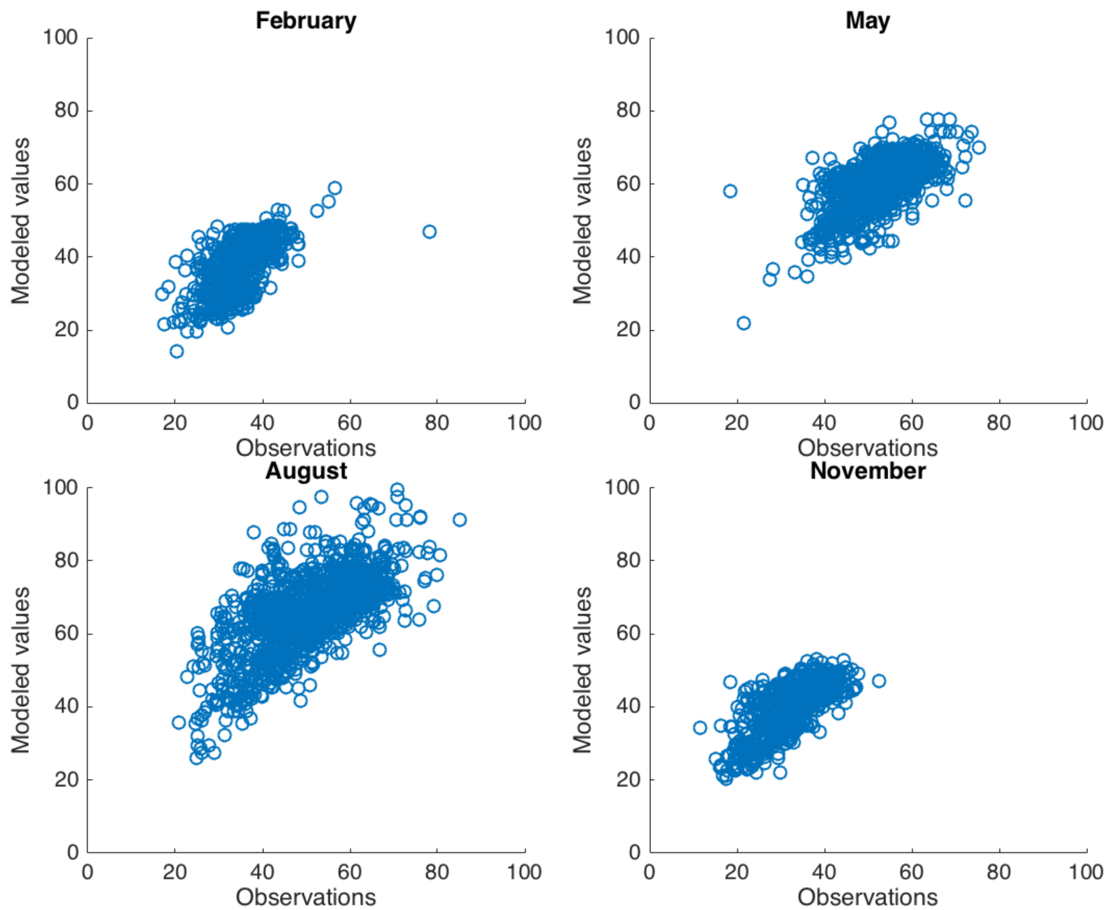


Figure 4.15 O_3 validation scatter plots (ppb).

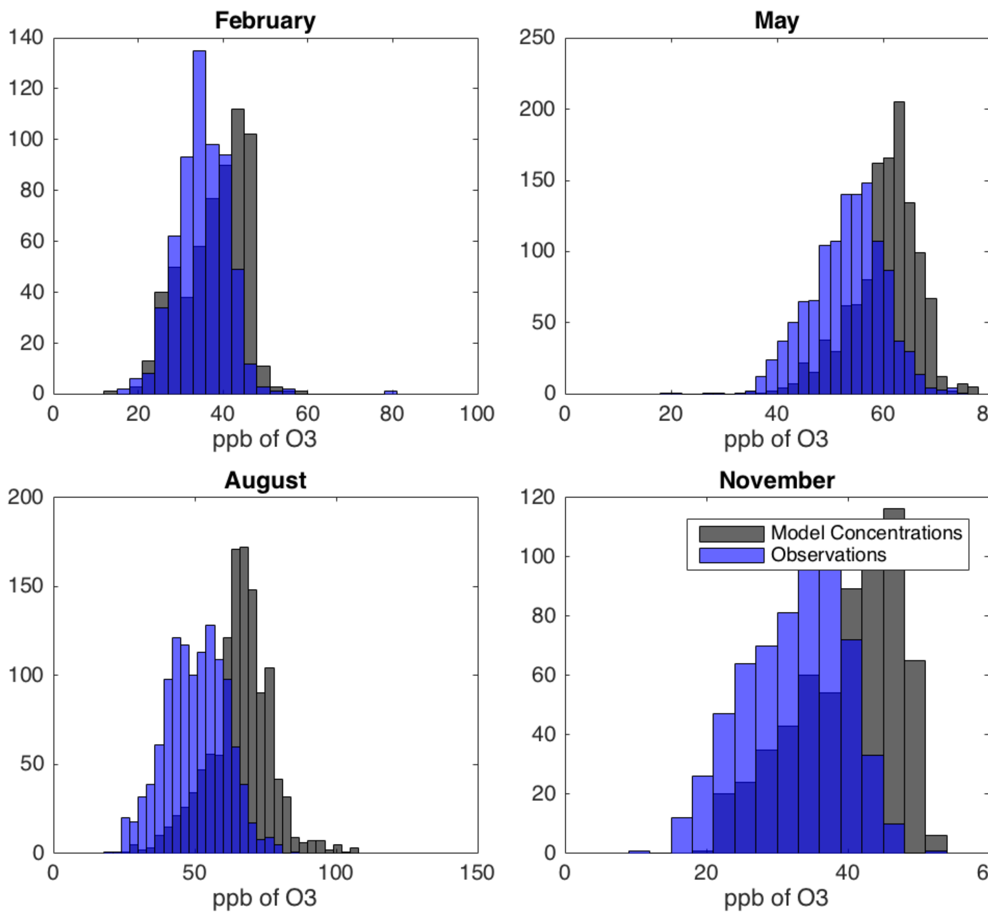
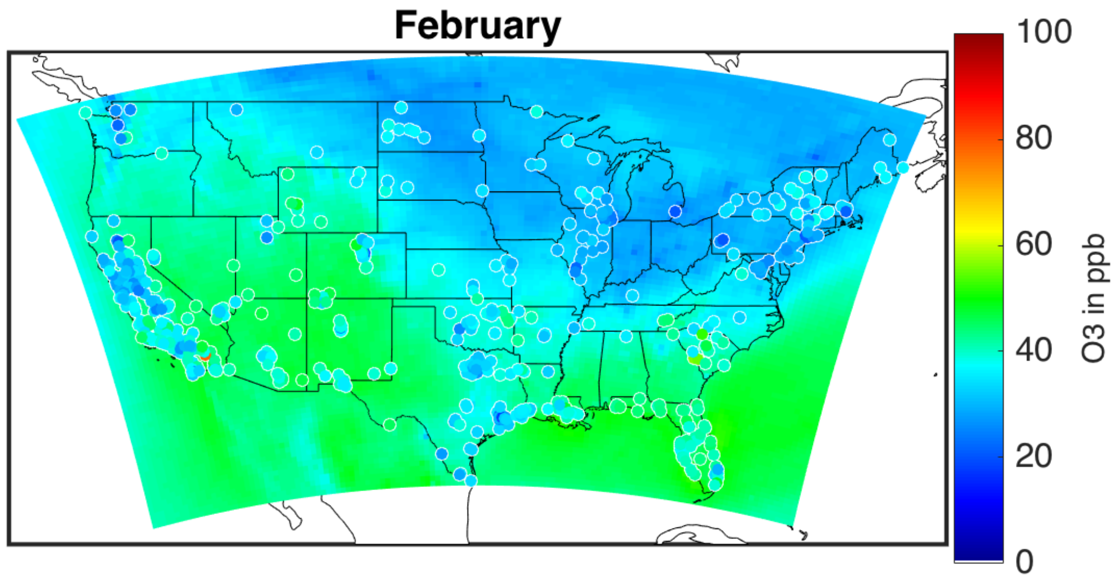
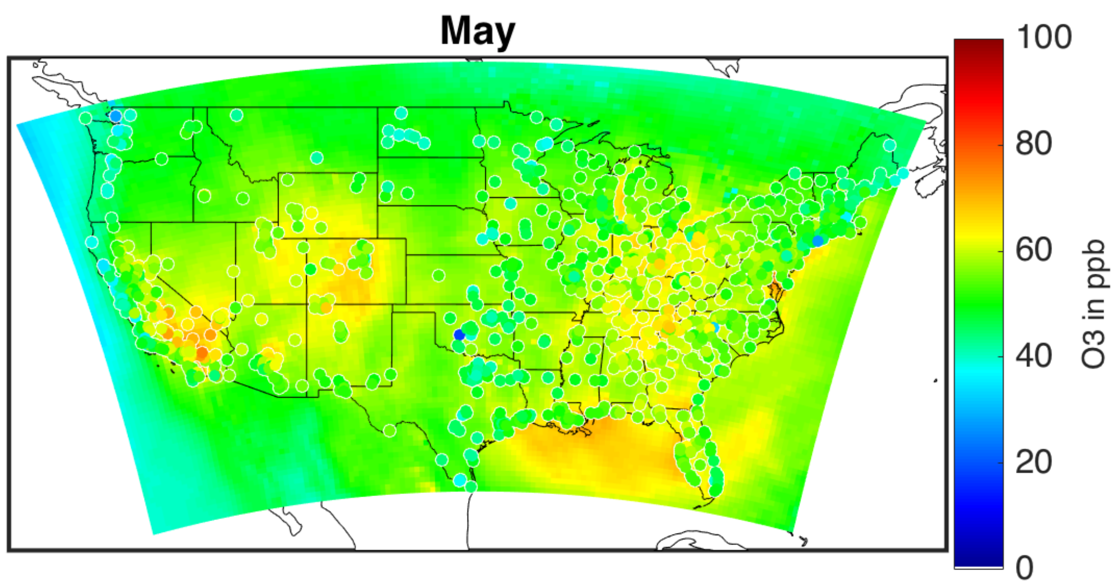


Figure 4.16 O_3 validation histograms.


February
100
80
60
40
20
0
O3 in ppb


May
100
80
60
40
20
0
O3 in ppb

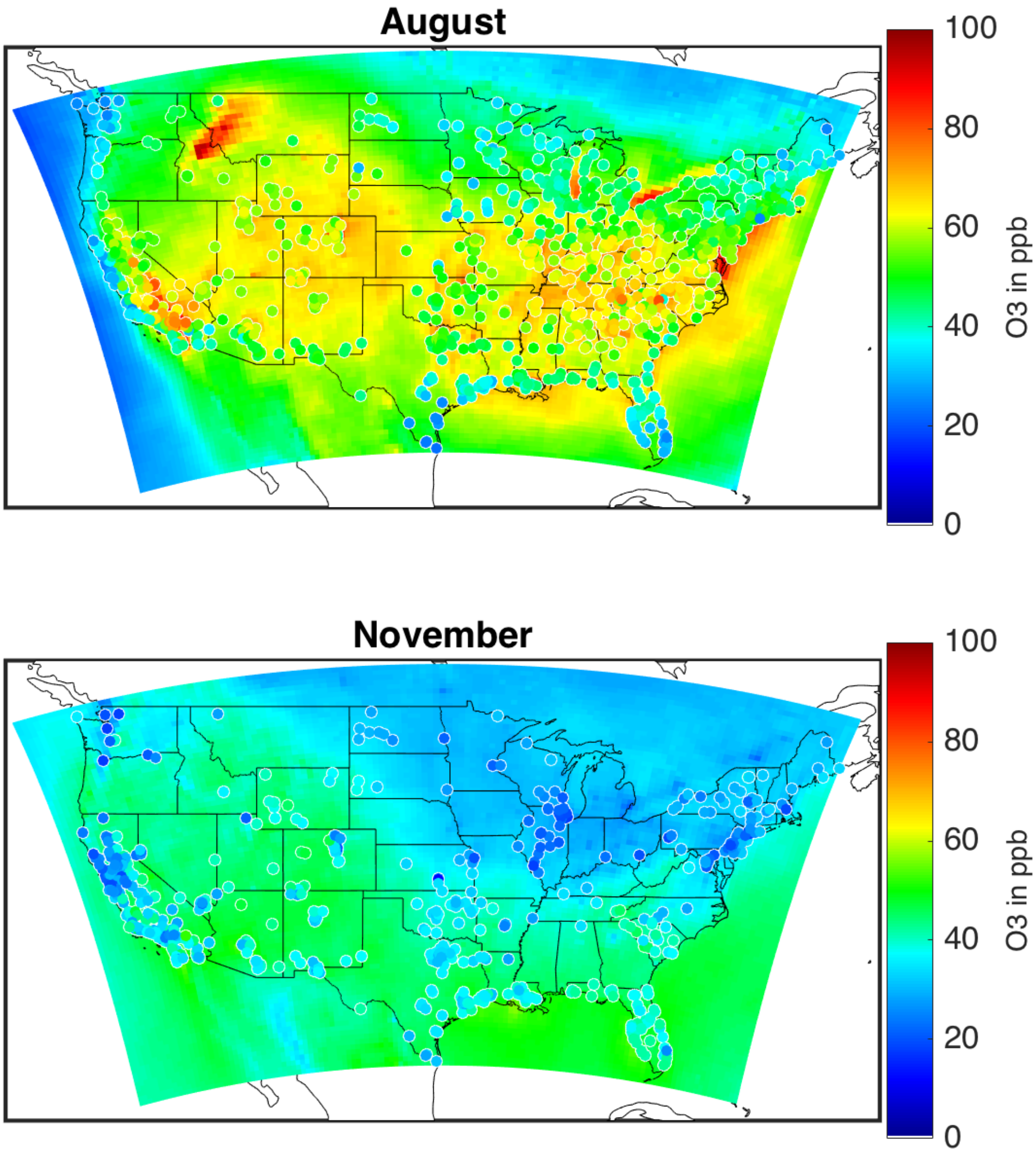
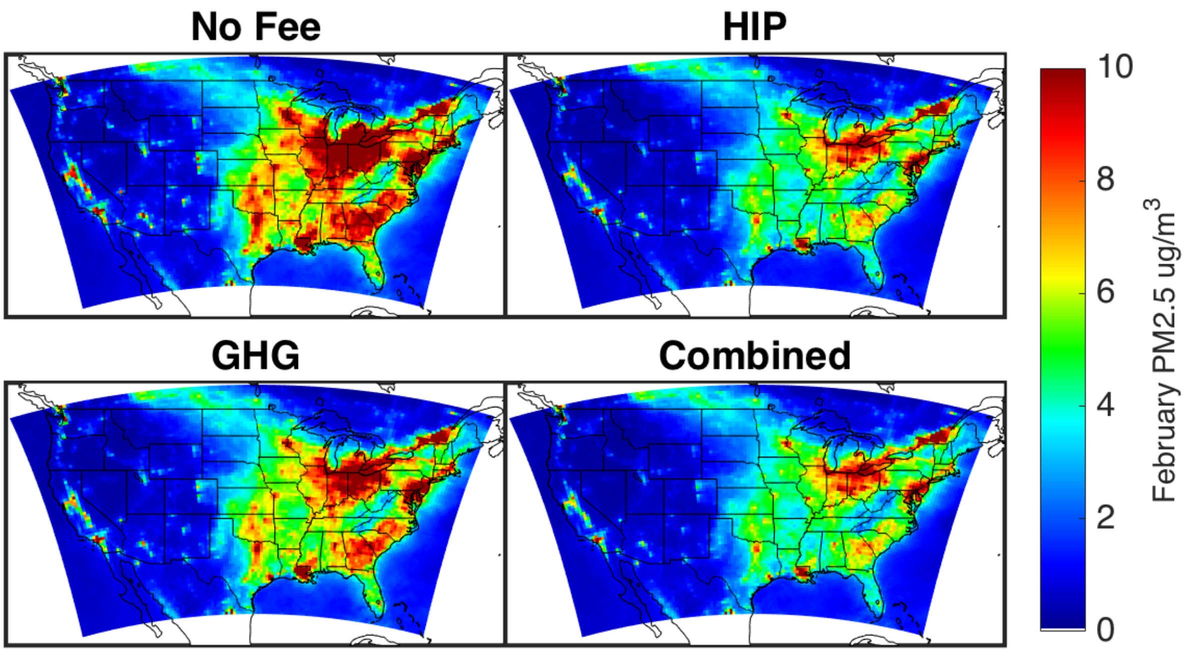


Figure 4.17 (a-d) Modeled and observed O_3 concentrations for 2007.

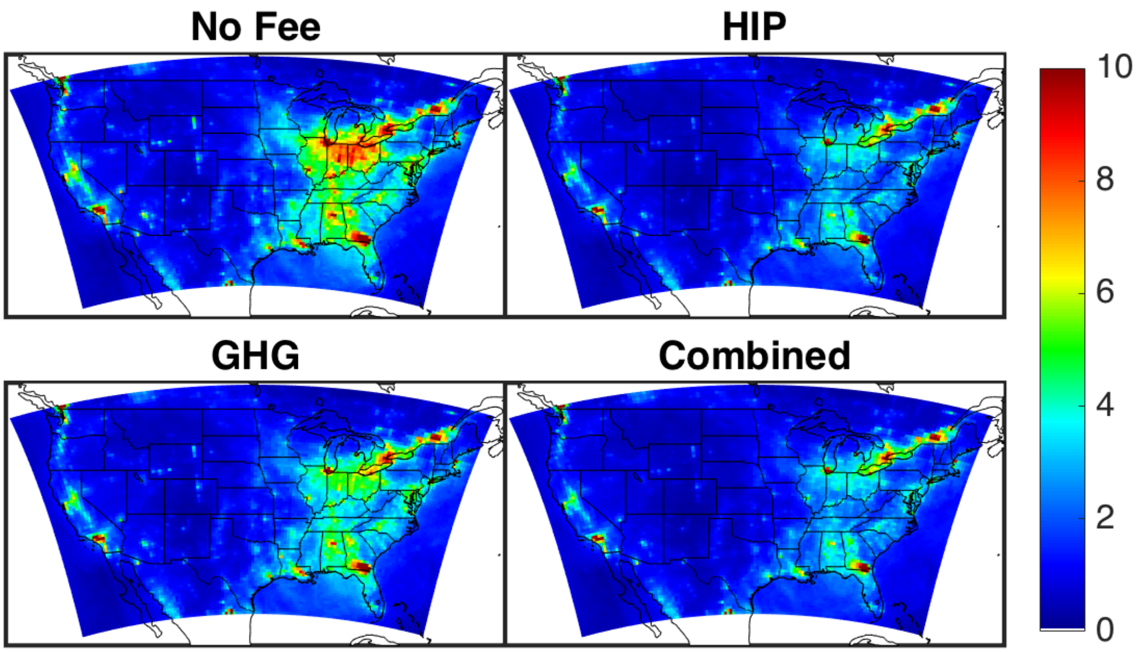
4.5.2 Seasonal CMAQ PM results

Here we present the median of the 24 hour average $PM_{2.5}$ concentration in each modeled month. Figure 4.4 in the main text presents the annual average results.

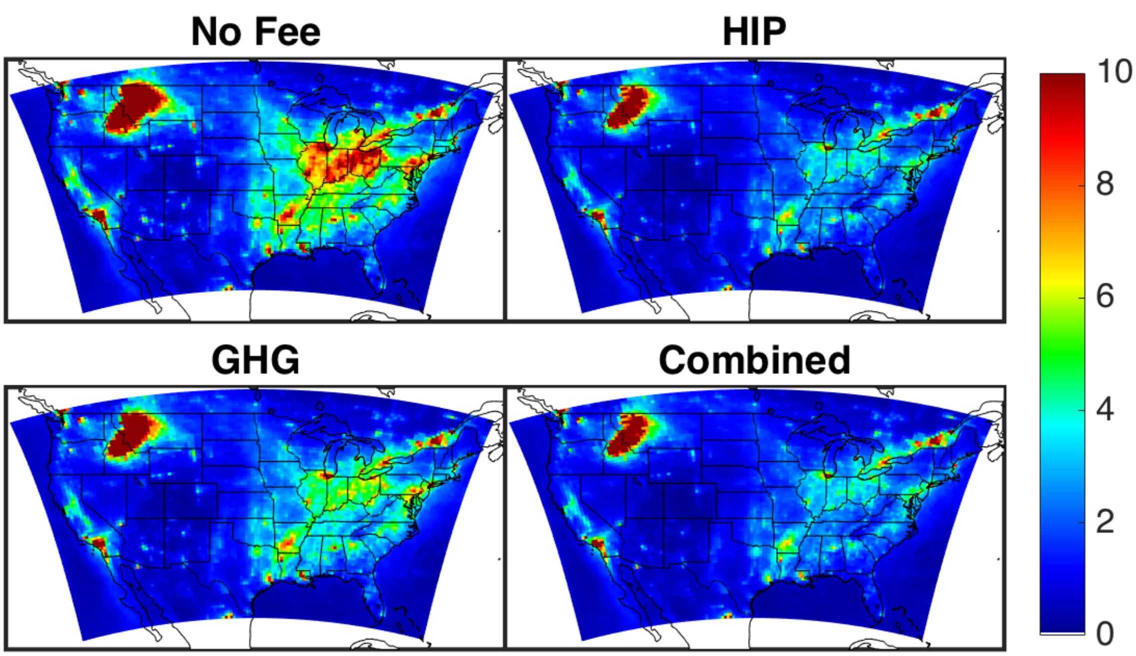
(a)



(b)


No Fee
HIP
GHG
Combined
May PM2.5 ug/m3
10
8
6
4
2
0

(c)


No Fee
HIP
GHG
Combined
August PM2.5 ug/m3
10
8
6
4
2
0

(d)

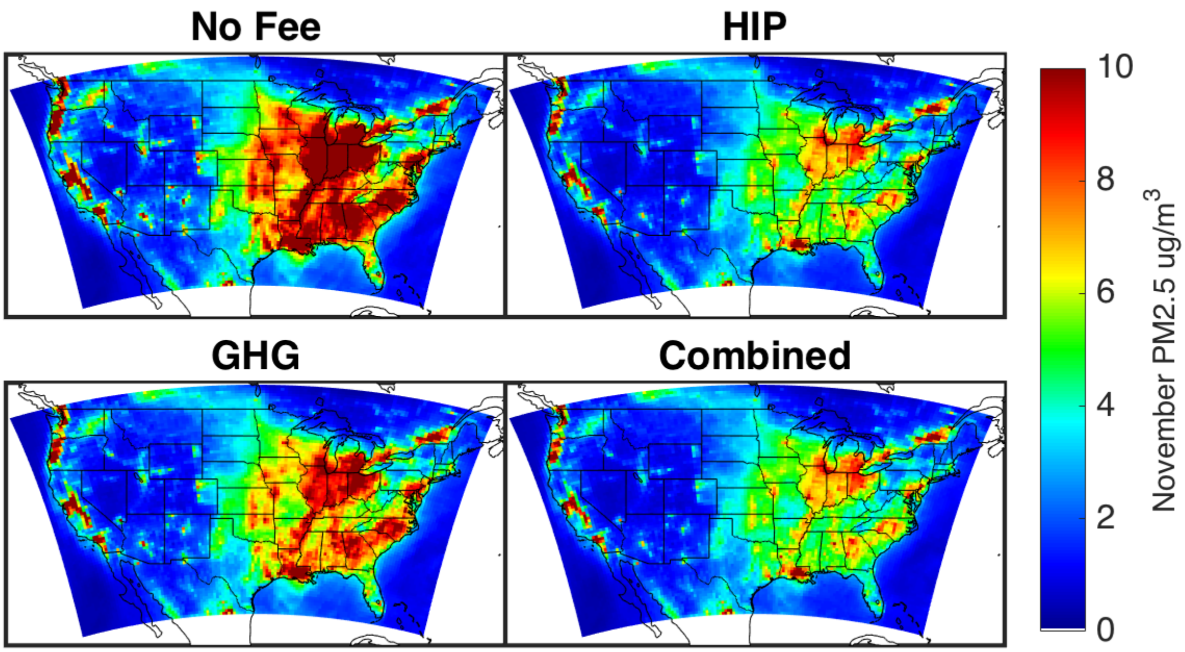


Figure 4.18 (a-d) Monthly average $PM_{2.5}$ concentrations.

4.5.3 Emissions and Concentration Reductions

Table 4.8 The approximate percent change in emissions from the No Fee case as well as the percent change in concentration from the No Fee case. The emissions reductions are based on total MARKAL emissions reductions for each species by region and may not accurately represent the emissions reductions in CMAQ do to the inclusion of non-energy sector emissions in CMAQ.

	NOx Emissions			SO2 Emissions			PM2.5 Emissions			O3 concentration			PM2.5 Concentration		
	HIP	Combo	GHG	HIP	Combo	GHG	HIP	Combo	GHG	HIP	Combo	GHG	HIP	Combo	GHG
Nat'l	-29	-30	-13	-69	-70	-39	-44	-45	-27	-6	-6	-2	-26	-27	-13
R1	-15	-16	-6	-37	-38	-11	-54	-55	-38	-7	-7	-1	-18	-18	-10
R2	-24	-25	-9	-46	-47	-19	-48	-48	-30	-7	-7	-4	-33	-33	-15
R3	-38	-38	-16	-74	-74	-48	-38	-38	-21	-14	-14	-9	-44	-45	-29
R4	-36	-37	-16	-69	-68	-36	-47	-47	-29	-3	-3	-1	-30	-30	-16
R5	-26	-28	-7	-78	-79	-39	-32	-32	-16	-4	-4	-1	-26	-27	-11
R6	-38	-39	-21	-78	-78	-50	-54	-55	-36	-19	-20	-13	-44	-44	-24
R7	-25	-26	-10	-70	-70	-36	-44	-45	-26	-2	-2	-1	-23	-23	-11
R8	-27	-30	-9	-83	-82	-51	-55	-56	-35	0	0	0	-46	-47	-29
R9	-21	-27	-19	-68	-70	-47	-39	-40	-27	-4	-5	-4	-29	-31	-19

4.5.4 Health Impacts and Valuation

This section includes further details on choices made in BenMAP-CE. While the main text discusses the mortality functions as these have the largest impact on valuation, a range of health impacts is considered for this analysis. These are the functions that were used to calculate health impacts in BenMAP-CE.

Table 4.9 Health Impact Functions for $PM_{2.5}$.

Endpoint	**Affected Ages**	**Author/Year**	**citation**
Acute bronchitis	8--12	Dockery et al 1996	(Dockery et al., 1996)
Asthma Exacerbation, Cough	6--18	Mar et al 2004	(T. F. Mar et al., 2004)
Asthma Exacerbation, Cough	6--18	Ostro et al 2001	(B. Ostro et al., 2001)
Asthma Exacerbation, Shortness of Breath	6--18	Ostro et al 2001	(B. Ostro et al., 2001)
Asthma Exacerbation, Shortness of Breath	6--19	Mar et al 2004	(T. F. Mar et al., 2004)
Asthma Exacerbation, Wheeze	6--18	Ostro et al 2001	(B. Ostro et al., 2001)
ER visits, asthma	0-99	Mar et al. 2010	(Therese F. Mar et al., 2010)
ER visits, asthma	0-99	Glad et al. 2012	(Glad et al., 2012)
ER visits, asthma	0-99	Slaughter et al. 2005	(Slaughter et al., 2005)
HA asthma	0-17	Babin et al. 2007	(Babin et al., 2007)
HA, Chronic Lung Disease	18-64	Moolgavkar 2000	(Moolgavkar, 2000)
Hospital Admissions, All cardiovascular (less AMI)	65-99	Peng et al. 2009	(Peng et al., 2009)

Hospital Admissions, All cardiovascular (less AMI)	65-99	Bell et al. 2008	(Bell, Ebisu, et al., 2008)
Hospital Admissions, All cardiovascular (less AMI)	18-64	Moolgavkar 2000	(Moolgavkar, 2000)
Hospital Admissions, all respiratory	65-99	Kloog et al. 2012	(Kloog et al., 2012)
Hospital Admissions, all respiratory	65-99	Zanobetti et al 2009	(Zanobetti et al., 2009)
Hospital Admissions, asthma	0--17	Sheppard 2003	(Sheppard et al., 1999)
Lower Respiratory	7--14	Schwartz and Neas 2000	(Schwartz and Neas, 2000)
Minor Restricted Activity Days	18-64	Ostro and Rothschild 1989	(Bd Ostro and Rothschild, 1989)
Mortality	0-0	Woodruff et al. 1997	(T J Woodruff et al., 1997)
Mortality	30-99	Krewski et al 2009	(Krewski et al., 2009)
Mortality	25--99	Lepeule et al. 2012	(Lepeule et al., 2012)
Nonfatal Acute Myocardial Infarction	18-99	Peters et al. 2001	(Peters et al., 2001)
Nonfatal Acute Myocardial Infarction	18-99	Pope et al. 2006	(C. A. Pope et al., 2006)
Nonfatal Acute Myocardial Infarction	18-99	Zanobetti and Schwartz 2006	(Zanobetti and Schwartz, 2006)
Nonfatal Acute Myocardial Infarction	18-99	Zanobetti et al 2009	(Zanobetti et al., 2009)
Nonfatal Acute Myocardial Infarction	18-99	Sullivan et al. 2005	(Sullivan et al., 2005)
Upper Respiratory Symptoms	9--11	Pope et al. 1991	(C. A. R. Pope and Dockery, 1991)
Work loss days	18-64	Ostro 1987	(Bd Ostro, 1987)

Table 4.10 Health impact functions for O_3.

Endpoint	**Affected Ages**	**Author/Year**	**citation**
Asthma Exacerbation	6--18	Mortimer et al. 2002	(Mortimer et al., 2002)
Asthma Exacerbation	6--18	Schildcrout et al. 2006	(Schildcrout et al., 2006)
Asthma Exacerbation	6--18	O'Connor et al. 2008	(G. T. O'Connor et al., 2008)
Emergency Room Visits, Asthma	0--17	Mar and Koenig 2009	(Therese F. Mar and Koenig, 2009)
Emergency Room Visits, Asthma	0--99	Sarnat et al. 2013	(Sarnat et al., 2013)
Emergency Room Visits, Asthma	0-99	Ito et al. 2007	(Kazuhiko Ito et al., 2007)
Emergency Room Visits, Asthma	18-99	Mar and Koenig 2009	(Therese F. Mar and Koenig, 2009)
Emergency Room Visits, Asthma	0-99	Glad et al. 2012	(Glad et al., 2012)
Emergency Room Visits, Asthma	0-99	Peel et al. 2005	(Peel et al., 2005)
Emergency Room Visits, Asthma	0-99	Wilson et al. 2005	(Wilson et al., 2005)
HA, All Respiratory	65-99	Katsouyanni et al. 2009	(Katsouyanni et al., 2009)
Minor Restricted Activity Days	18-64	Ostro and Rothschild 1989	(Bd Ostro and Rothschild, 1989)
Mortality	0-99	Zanobetti and Schwartz 2008	(Zanobetti and Schwartz, 2008)

Mortality	0-99	Smith et al. 2009	(Smith et al., 2009)
Mortality	0-99	Jerrett et al. 2009	(Jerrett et al., 2009)
School Loss Days	5--17	Chen et al 2000	(L. Chen et al., 2000)
School Loss Days, All Cause	5--17	Gilliland et al. 2001	(Gilliland et al., 2001)

The valuation was calculated based on costs in BenMAP-CE. The value of mortality is a value of statistical life based on 26 value of life studies and is equivalent to $6.6 million in 2005 USD. All hospital admissions are valued based on cost of illness data combined with wage loss. Acute, lower, and upper respiratory symptoms are based on willingness to pay data assuming a duration of one day. Work loss days are based on county-specific median daily wages. Asthma exacerbation is based on willingness to pay data. Visits to the emergency room are based on cost of illness. Acute bronchitis valuation is based on willingness to pay data and assumes a six day illness.

4.5.5 Full BenMAP-CE results

Table 4.11 shows national level incidence and valuation results for the individual health impacts considered. Where there are multiple citations for a health impact that means that several health impact functions were used to calculate the incidence and then equally weighted to determine a single value.

Table 4.11 Non-mortality health impacts (HA stands for Hospital Admissions, AMI stands for Acute Myocardial Infarction, ER stands for Emergency Room).

			Incidence			Value (Million USD)		
Pollut-ant	Health Endpoint	C-R functions	HIP	combo	GHG	HIP	combo	GHG
O3	Asthma Exacer-bation	(F. O'Connor et al., 2010; Schildcrout et al., 2006; Mortimer et al., 2002)	497,209	616,344	112,213	$26	$32	$6
	ER Visits Respiratory	(Sarnat et al., 2013; Peel et al., 2005; Glad et al., 2012; Wilson et al., 2005; Therese F. Mar and Koenig, 2009; K. Ito et al., 2005)	1,422	1,749	311	$1	$1	$0
	HA Respiratory	(Katsouyanni et al., 2009)	592	721	130	-	-	-
	Minor Restricted Activity Days	(Bd Ostro and Rothschild, 1989)	1,190,145	1,476,564	262,450	$73	$90	$16
	School Loss Days	(L. Chen et al., 2000; Gilliland et al., 2001)	435,441	536,709	98,790	$38	$46	$9
PM2.5	Acute Bronchitis	(Dockery et al., 1996)	48,755	50,056	24,387	$21	$22	$10

	AMI Nonfatal	(Peters et al., 2001)	43,921	45,029	22,079	$2,799	$2,869	$1,404
	AMI Nonfatal	(C. A. Pope et al., 2006; Sullivan et al., 2005; Zanobetti et al., 2009; Zanobetti and Schwartz, 2006)	4,936	5,066	2,439	$315	$323	$155
	Asthma Exacer-bation	(B. Ostro et al., 2001; T. F. Mar et al., 2004)	723,471	744,838	345,285	$38	$39	$18
	ER Visits Respiratory	(Glad et al., 2012; T. F. Mar et al., 2004; Slaughter et al., 2005)	5,757	5,904	2,820	$2	$2	$1
	HA Cardio-vascular (no AMI)	(Moolgavkar, 2000)	1,414	1,450	682	$48	$49	$23
	HA Cardio-vascular (no AMI)	(Zanobetti et al., 2009; Bell, Ebisu, et al., 2008; Peng et al., 2009)	3,265	3,350	1,608	$103	$106	$51
	HA Respiratory	(Zanobetti et al., 2009; Kloog et al., 2012)	3,029	3,110	1,501	$80	$82	$40
	HA Asthma	(Babin et al., 2007; Sheppard et al., 1999)	304	312	145	$4	$4	$2

	HA Chronic Lung Disease	(Moolgavkar, 2000)	657	672	316	$11	$11	$5
	Lower Respiratory Symptoms	(Schwartz and Neas, 2000)	204,706	210,191	102,324	$4	$4	$2
	Respiratory Restricted Activity	(Bd Ostro and Rothschild, 1989)	8,098,199	8,321,386	4,005,074	$494	$508	$245
	Upper Respiratory Symptoms	(C. A. Pope et al., 2002)	302,979	311,363	149,321	$10	$10	$5
	Work Loss Days	(Bd Ostro, 1987)	1,374,944	1,413,009	678,469	$196	$201	$97

Table 4.12 Mortality incidence for O_3 and $PM_{2.5}$ by health impact function.

		Avoided PM2.5 Mortality			Avoided O3 Mortality		
		Krewski et al.	Lepule et al.	Woodruff et al.	Jerrett et al.	Zanobetti and Schwartz	Smith et al.
HIP	***Nat'l***	***46,953***	***105,154***	***75***	***1,430***	***622***	***668***
	R1	1,160	2,605	1	173	6	7
	R2	4,872	10,923	5	262	24	26
	R3	11,510	25,638	17	226	64	70
	R4	3,131	7,006	5	149	52	56
	R5	8,919	20,032	15	172	237	256
	R6	3,708	8,319	8	14	111	119
	R7	5,415	12,168	13	60	82	88
	R8	2,253	5,060	4	292	64	68
	R9	5,986	13,403	7	82	-19	-21
Combined	***Nat'l***	***48,227***	***107,981***	***77***	***1,744***	***757***	***814***
	R1	1,181	2,652	1	187	7	8
	R2	4,960	11,120	5	279	31	34
	R3	11,642	25,926	17	252	71	77
	R4	3,175	7,105	6	172	56	60
	R5	9,149	20,545	15	192	259	279
	R6	3,787	8,493	8	161	117	125
	R7	5,538	12,443	13	73	93	99
	R8	2,307	5,181	4	333	79	84
	R9	6,488	14,516	8	95	44	47
GHG	***Nat'l***	***23,095***	***8,688***	***36***	***345***	***142***	***153***
	R1	560	1,260	0	48	2	2
	R2	1,931	4,348	2	60	9	10
	R3	6,590	14,762	10	25	15	16
	R4	1,681	3,779	3	-12	12	13
	R5	3,175	7,162	5	72	36	39
	R6	1,831	4,128	4	49	31	33
	R7	2,148	4,851	5	6	4	5
	R8	1,311	2,952	2	78	18	19
	R9	3,868	8,688	5	19	15	16

Mortality incidence should not be summed across all studies to get total mortality. One $PM_{2.5}$ mortality estimate (excluding Woodruff et al. (2006)), one O_3 mortality estimate, and the estimate from Woodruff et al. should be summed to determine estimated mortality.

To determine the value of the avoided mortalities, multiply the incidence total by $6.6 million USD, which is the value of statistical life used for discussions in the main text. Multiple overlapping health impact functions are presented to provide information about uncertainty in mortality due to the large impact on the total monetary benefit of policies. The Woodruff et al. study estimates infant mortality and does not overlap with the other $PM_{2.5}$ adult mortality functions. That is also why there are fewer incidences; there is a smaller population to which this function is applicable.

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Chapter 5

Conclusions

5.1 Summary and Conclusion

In this thesis we analyzed application of damage-based fees in the US energy system to determine scenarios for energy use and associated emissions. The objective was to internalize externalities such that health and climate costs are considered when choosing how to produce and use energy. We analyzed changes to the energy system incurred by application of damage-based fees by calculating the costs and benefits of business as usual and fee driven systems to determine how emissions reductions were achieved and whether the benefits of the emission reductions outweighed the costs.

All fees led to reductions in emissions, but the method of reduction and the level of reduction varied. We added fees in the MARKAL (the MARKet ALlocation) model with a modified version of the US EPA's nine-region energy system database to determine the level of emissions and mix of technologies that would be used if fees were in place. When fees were applied to the electricity sector alone, we found that the lower HIP (Health Impacting Pollutant) fees led to emissions reductions achieved almost exclusively by applying control technologies. Higher HIP fees additionally led to a different technology mix and a lower demand for electricity. GHG (Greenhouse Gas) fees alone led to a reduction in electricity and coal use, reducing both GHG and NO_x emissions, but not SO_2 emissions. Although coal use was reduced with GHG fees, fewer SO_2 controls were used on the remaining coal fired power plants in the GHG fee case, leading to a slight increase in SO_2

emissions. Combined fees led to the largest change in the electricity system and the largest reductions in emissions of both HIPs and GHGs.

When fees were applied to all energy sectors, similar but not identical patterns of emission reduction were found. Control technologies were still used more in response to HIP fees while energy efficiency and fuel switching were more prevalent responses to GHG fees. Although fees based on sector specific damages were used, emissions reductions were highly driven by the number and affordability of emissions reduction options by sector. Therefore, emissions were still reduced the most from the electric sector, despite the relatively low damage per ton of emissions in this sector. The electric sector was also responsible for a significant portion of emissions, so a low unit cost across a large quantity resulted in reductions from this sector that were still quite impactful. With fees on all sectors, we noted less delineation between control technologies for HIP fees and fuel switching for GHG fees. This is in part due to the inclusion of other sectors, but the updated database used for the multi-sector analysis also had lower costs for CCS and renewable fuels, which impacted this shift as well.

We also evaluated the effect of these emission changes on air quality using the CMAQ (Community Multiscale Air Quality) model. When emissions reductions associated with fees were included in CMAQ, air quality improved in almost all areas of the country, even for GHG focused fees, because the efficiency improvements used to achieve the GHG emissions reductions also reduced all co-emitted pollutants. As we would expect, the air quality improved more with HIP fees than GHG fees, and the air quality was best with the combined fees (although the distinction from the HIP fee case was small). Pollutant concentrations were typically reduced most in regions where emissions were reduced most, but this was true more often for $PM_{2.5}$ than O_3. In some cases O_3 concentrations changed very little despite a change in emissions. This may be because NO_x emissions were reduced in areas where O_3 was VOC limited. The control options in MARKAL meant that VOC emissions were not changed significantly across cases. The reductions in VOC emissions with fees were small or even increased compared to the No Fee case. Biogenic emissions can be very important for O_3 production as well. There were some small areas with slightly worse air quality due to redistribution of emissions. This may be an argument for using the fees collected to offset some of the increased costs that might be incurred in these areas.

Using the MARKAL model coupled with CMAQ provided a more detailed estimate of how regional emissions may respond to fee-based policies than would have been made through simply reducing all CMAQ emissions by a certain percentage. For instance, region 3, the Midwest, had very large air quality improvements, as this was a region where large amounts of coal EGUs (Electricity Generating Units) were retired early with fees. In contrast there was little change in concentrations of pollutants in region 1, the Northeast, which even without fees used mostly low emitting electricity generation such as NGCC (Natural Gas Combined Cycle), renewables, and nuclear, and had fewer industrial facilities than many other regions.

We calculated the monetary benefit of the fees using BenMAP (the Benefit Mapping and Analysis Program) for health impacts and damage scaled emissions for health and climate impacts. Between 2016 and 2045, the total population will grow, and the fraction of older people who are more susceptible to many of the health impacts of air pollution will increase. We found that the population increase and demographic shifts increased the value of damages per ton over time, and therefore fees should be re-evaluated accordingly with time. The change in population was very important in determining the magnitude of benefits for 2045, as it increased the value by 77% compared to a calculation with 2016 population.

We also found that if population change is accounted for, the damage scaled emissions closely approximate the benefits calculated using the coupled model system. For the HIP fee case, the damage scaled emission estimate was within 10% of the benefits calculated with the same population. Future studies could thus be streamlined, computationally, by calculating benefits per ton of emissions for the year(s) of interest in policy analysis and comparing damage scaled emissions across a variety of cases. This would be particularly useful in comparing a wide range of possible policy options, because the damage calculations (i.e., running an air quality model such as CMAQ or CAMx and thenBenMAP) themselves will require time.

In 2045, the health related benefits for the HIP fee case were $416 billion, for the GHG fee case health benefits are $202 billion, and for the combined fee case they were $429 billion. The climate benefit was $73 billion with HIP fees, $83 billion with GHG fees, and $101 billion with combined fees. The costs were $76 billion with HIP fees, $48 billion with GHG fees, and $91 billion with combined fees. Summing health and climate benefits and subtracting costs led to net benefits of $413 billion for HIP fees, $238 billion for GHG fees, and $439 billion for combined fees for the year 2045. All values in this chapter were presented in year 2005 USD. The damage-based fees always had a net benefit, which was expected as this was the reason to use damages to define the fees. The climate benefit was always smaller than the air quality benefit, even if only GHG fees were applied.

5.2 Sources of Uncertainty and Ideas for Future Analysis

We next consider sources of uncertainties in these calculations, such as those associated with damage estimates, projections of future population, and each analysis tool. We discuss these uncertainties, their potential impacts on our results, and highlight targets for future research.

There are several sources of uncertainty associated with damage estimates, including uncertainty in the concentration-response functions, the monetization of mortality, and the location dependence of the calculated value. We have accounted for some of this uncertainty by considering a range of damage estimates in the MARKAL analyses. The estimates used here are chosen because they represent the range of values found in the literature (National Research Council Committee on Health, Environmental, and Other External Costs and Benefits of Energy Production and Consumption, 2010; Levy et al., 2009; Muller et al., 2011; Muller and Mendelsohn, 2007; Levy et al., 1999; Fann et al., 2009; Fann, et al., 2012). We present the average damages per ton for the studies considered in Table 5.1. For any species, the damages for a pollutant range by at least an order of magnitude across studies. Different authors used different modeling tools and considered different sources, which also affect the results. The APEEP model was used in the NRC (2010) study and Muller et al. (2007; 2011) studies while Fann et al. (2012) used CAMx with BenMAP and Fann et al. (2009) use a response surface model derived from

CMAQ. Also, the marginal damages from the NRC (2010) study are for electricity generation from fossil fuels while the other studies also include emissions from other sectors.

Table 5.1 Average marginal damage estimates by species from several studies in $/ton.

	NO_x	SO_2	PM_{10}	$PM_{2.5}$	VOC
Fann et al. 2012	7,100	42,800		250,100	
Fann et al. 2009	11,200	68,300			2,300
NRC2010	2,000	9,800	1,100	21,500	
Muller et al. 2011	500	2,200	300	3,900	400
Muller & Mendelsohn 2007	400	1,900	600	4,100	600

Although this table presents average values across all sectors, not all studies considered all sectors. Fewer studies have evaluated the effect of residential and commercial emissions. This is an area that should be explored further in the future as these emissions, while smaller in magnitude than other sectors, are necessarily located very close to potentially sensitive groups, which means their marginal damages may be large, as seen in the Fann et al. (2012) study which does provide benefit estimates for the residential sector. The HIP fee case considered in Chapter 4 is the high HIP fee case, based mostly on the Fann et al. (2012) study. This case has the largest emissions reductions; lower fees would lead to fewer health benefits.

There are different sources of uncertainty for GHG damages, including discount rate and the magnitude of impacts. We again capture some of this uncertainty in the estimates we use. The Social Cost of Carbon was calculated using an ensemble of models to help reduce bias toward any particular set of assumptions. We also use a range of values to account for uncertainty in the appropriate discount rate. Some studies (Moore and Diaz, 2015; Dietz and Stern, 2015; Lontzek et al., 2015) suggest that these damages may be an underestimate as there are still many uncertainties surrounding climate change predictions. The damage estimates are unable to account for impacts that scientists are aware of but cannot yet quantify, such as ocean acidification. We use the 2.5% average GHG fee case in our analysis in Chapter 4. This case has the largest HIP co-benefits, so other GHG fees will likely lead to smaller health benefits.

There will always be uncertainties surrounding future projections. Projecting population level and related energy demand into the future requires many assumptions. The population growth rate for the US has been decreasing, but changes in immigration, life expectancy, or birth rate projections could alter the population. This affects both our projections of future demand as well as the calculation of future benefits in a way that will likely compound. If the population in 2045 is larger than projected, there will be more energy demand which will likely lead to larger emissions and worse air quality. This worse air quality would affect more people and therefore compound the increase in damages.

There are also uncertainties with MARKAL results, most of which are again associated with difficulties in predicting future values. If a new technology is used more, the cost of that technology might decrease faster than what is predicted in the MARKAL database, but for technologies with slow adoption the cost might be higher than what we see here. For instance, if solar PV use increases faster than projected, the learning curve would be affected, leading to lower PV costs. In the past decade, we have seen how the change in natural gas price has affected the energy system compared to prior projections. Also new technologies or financing methods in the energy market could alter the energy system. Financing methods could lead to lower hurdle rates for some technologies as the barrier to using those technologies is diminished. Further, if new technologies are developed that can provide energy with lower emissions or a lower price than current options, they would displace technologies used in the model results here. It is possible that such innovation would be more likely with fees as there would be more incentive to reduce energy costs while also reducing emissions. Even for newer technologies that are included, if they are largely unused at present, the costs and operating parameters are more uncertain. This can be particularly important for Carbon Capture and Sequestration (CCS), which could be an important emission reduction mechanism with GHG fees depending on the cost and CO_2 removal efficiency of the technology.

Specific choices made when developing MARKAL also create uncertainties. The version of MARKAL used here is inelastic with respect to demand for energy services. Specifying demand for energy services and allowing options that improve end-use efficiency mitigates the effect that inelastic demand for energy would have, but it does limit the ability to respond to fees. Inelasticity means that measures such as reduced travel with high transportation costs are not options to respond to the fees. Although we have added some technologies to allow for more energy efficiency, this is still a weakness in MARKAL. The options for efficiency are still limited in the commercial and residential sectors. The complexity of the industrial sector is not fully captured, and individual facilities have more emissions reduction options than those presented here. The representation of upstream processes could also be improved. In MARKAL, many of the upstream emissions are modeled without a source, so the specific location of those emissions is unknown. The breakdown of upstream emissions between industrial, transportation, or extraction sources would allow us to better couple the MARKAL results with CMAQ.

As MARKAL is not a dispatch model, reliability issues with using intermittent technologies such as wind and solar are accounted for through the use of constraints on those technologies. As more information is gathered about system performance using intermittent technologies, these constraints may be altered. If future energy systems have more storage or other integration breakthroughs are made, these constraints could lead to an underestimate in use of renewables. There are also other reasons for low penetration of renewables such as cost and the availability of ideal locations.

Some of the sources of uncertainty with CMAQ are based on the modeling and analysis methodology. We made some decisions that reduce the computational intensity of our calculation, but may increase uncertainty. Our analysis is calculated with a 36 km grid resolution, although finer resolution is available. Several studies (TThompson and Selin, 2012; Thompson et al., 2014; Punger and West, 2013) have evaluated air quality and benefits at multiple grid resolutions to determine the impact that resolution has on the results. Ozone tends to be overestimated using 36 km resolution compared to smaller grid scales, and $PM_{2.5}$ tends to be underestimated. The uncertainty due to grid resolution is smaller than the uncertainties within the concentration-response functions for calculating benefits.

We also evaluate only four months of the year instead of all twelve. Turner et al. (2015) used one quarter of the weeks of the year to represent a full year and found the results were representative of the annual average BC concentration, and we chose months specifically to capture seasonality. We also do not use future meteorology. Any choice of meteorology for the future will introduce uncertainty due to the uncertainty surrounding climate change, but including the effects of climate change is likely to lead to higher concentrations of O_3 and $PM_{2.5}$ (Trail et al., 2014; Garcia-Menendez et al., 2015; Fiore et al., 2012). The benefits are based on the difference between air quality in two future cases, so the effect of neglecting climate change on benefits estimates may be smaller than the effect on air quality.

The scaling factor method of modeling emissions reductions introduces some uncertainties into the CMAQ analysis as well. This method may lead to an overestimation of the benefits of upstream $PM_{2.5}$ emissions reductions, as we treated upstream $PM_{2.5}$ emissions reductions as being evenly distributed across all sources whereas in reality most of the upstream emissions occur at more remote locations such as mining sites. Also, there are emissions reductions modeled for non-energy related sources that lead to an over-prediction of the air quality improvement in response to the fees modeled. Also, our 2007 $PM_{2.5}$ modeled concentrations underestimate summer concentrations and overestimate winter concentrations compared to observations (US EPA, 2016). This may lead to uncertainties in our projected future concentrations, but as we calculate health benefits based on the difference in concentration from the No Fee case, this may mitigate the uncertainty somewhat. We intend to recalculate the air quality and related health benefits with a modified version of the scaling factor approach that avoids reductions in non-energy related emissions and more realistically distributes the upstream $PM_{2.5}$ emissions. This latter change is particularly important as they drive much of the health benefits in the current results.

There are also uncertainties in the calculations done with BenMAP-CE. We are particularly concerned with uncertainties related to population projections and mortality from $PM_{2.5}$, because these contribute most to the benefit value. We use 2040 population data because 2045 data are not available for BenMAP-CE. The total population is projected to increase only 2% from 2040 to 2045 according to US Census projections (Colby and Ortman, 2014), as population growth is projected to slow over the next few decades. There are further uncertainties associated with demographics, however.

Many of the BenMAP uncertainties are ideal targets for future research, such as uncertainties in the concentration-response (C-R) functions. As concentrations decrease with time, additional epidemiology studies can be performed to increase confidence in the concentration response relationship at low concentrations. Krewski et al. (2009) found that there could be a 58% increase in risk at low concentrations if a different functional form is used for the concentration response relationship. The 90^{th} percentile confidence interval for the $PM_{2.5}$ adult mortality (Krewski et al., 2009) benefits given the population and air quality in this study ranges by an order of magnitude. Fraas and Lutter (2013) examine a range of C-R functions and find that the spread in the distribution across C-R functions is larger than the uncertainty ranges presented for individual functions. If there are damages from categories not considered here, the damages here would present an underestimate. The benefit of $PM_{2.5}$ reductions may vary in either direction depending on the impact of composition on $PM_{2.5}$ health impacts.

The value of statistical life (VSL) used is based on 26 studies from the literature, but there is uncertainty in how to treat this. This could potentially be a large source of error due to the dominance of mortality in the valuation. Using a uniform VSL can produce 50% higher marginal damages than differentiating by age (Muller et al., 2011). Fraas and Lutter (2013) examine the mortality valuation literature and exclude all labor market studies as healthy, working age adults are not the population typically affected by PM mortality. This leads them to estimate a much lower VSL of $2.8 million.

This paper presents the technology changes, air quality improvements, and health benefits that can be expected with damage based fees. There are further issues that must be considered when drafting such a policy. One consideration is how to use the fees. For instance, if consumers pay less in income tax but more for electricity, this might lead to a net benefit overall, but a net disbenefit if the system considered is confined to energy, health, and the environment. Using fees collected to offset emissions reduction costs or medical costs would be two ways to avoid having net disbenefits within the smaller system boundary. Emissions fees would lead to higher energy costs, which will disproportionately affect low-income households. Although there is a benefit gained through the fees, households where energy costs already form a large percentage of the household expenditure may have difficulty meeting their needs with higher energy costs, so consideration of this effect would be prudent. Also, implementation of the fees as described here may not be practical for some sectors. For the transportation sector, an alternative standard based on fuel use should be considered. Another option would be adding fees or subsidies to the cost of the vehicle based on expected emissions over the vehicle life. Due to the high hurdle rates associated with vehicle purchases, subsequent fuel purchases are not a strong factor in vehicle purchase decisions. This is likely to have lower cost of implementation than attempting to measure emissions from all vehicles, but would still encourage emissions reductions through engine efficiency and possibly reduced mileage traveled. For the residential and commercial sectors, a fee on natural gas, as is modeled in some cases, may be an option to implement the fee in these sectors. Applying a fee or subsidy to the upfront cost of HVAC or other equipment might be more efficient. This would eliminate the need for additional emissions measurements as well as additional damage calculations to determine emissions specific fees. Although fuel based fees do not encourage emissions removal these options are less likely to be considered for the residential, commercial, and transportation sectors. Different fuel types could be taxed at different rates as well, which would capture the benefit of fuel switching. This would be particularly relevant for the transportation sector, which is the most likely of the three to have fuel-switching options. Given the health and climate benefits that can be achieved from damage-based fees, it is worth considering this policy option.

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Chapter 6

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