THE EFFECT OF US STATES' ENVIRONMENTAL POLICIES ON THEIR OWN CARBON EMISSIONS

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THE EFFECT OF US STATES' ENVIRONMENTAL POLICIES ON THEIR OWN CARBON EMISSIONS

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DECLARATION OF ORIGINALITY

I, Colby Kennedy, certify that

- I am the sole author of this thesis and that I have fully acknowledged and documented in my thesis all sources of ideas and words, including digital resources, which have been produced or published by another person or institution;
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ABSTRACT

The Effect of US States' Environmental Policies on Their Own Carbon Emissions

Reducing carbon emissions has been a longstanding goal of policy makers, who have implemented a variety of taxes, incentives, and mandates to achieve emissions reductions in line with their own goals as well as international climate agreements. In the United States, both command-and-control policies and market-based policies have been implemented in different combinations and with different design choices under different circumstances. Understanding the effect of these policies individually as well as the benefits of each approach generally is critical for states to design and implement effective climate policies. This thesis studies the effect of three of these policies; carbon markets, environmental review, and renewable portfolio standards together in order to both compare the effects of each policy as well as to better separate the effects of each policy from the total effect of all policies in the context of the United States. We find that carbon market schemes in the US are effective in accelerating the transition to a low carbon economy, while renewable portfolio standards and environmental review policies are not effective or may even be impairing the transition to a low carbon economy.

ÖZET

Türkçe Tez Başlığı

Çeşitli vergiler, teşvikler ve yetkiler uygulayarak karbon emısyonları azaltmak, hem ülkelerin kendi hedefleri hem de uluslararası iklim anlaşmaları doğrultusunda politika yapıcıların uzun süredir olmuştur. Amerika Birleşik Devletleri (ABD)'nde, hem komuta ve kontrol politikaları hem de piyasaya dayalı politikalar, farklı kombinasyonlarda ve farklı tasarım seçenekleriyle uygunlanmaktadır. Bu politikaların ayrı ayrı etkisini ve her bir yaklaşımın faydalarını anlamak, genel olarak etkili iklim politikaları tasarlamak ve uygulamak isteyen devletler için kritik öneme sahiptir. Bu tez, bu politikalardan üçünün–karbon piyasalarının, çevresel incelemenin ve yenilenebilir portföy standartlarının-etkisini, ABD özelinde incelemektedir. Amaç hem politikaların etkilerini karşılaştırmak, hem de her bir politikanın etkisini tüm politikaların toplam etkisinden daha iyi ayırmaktır. Bulgular, karbon piyasası planlarının düşük karbon ekonomisine geçisi hızlandırmada etkili olduğunu, yenilenebilir portföy standartlarının ve çevresel inceleme politikalarının ise etkili olmadığını ve hatta düşük karbon ekonomisine geçişi engelleyebileceğini göstermektedir.

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CHAPTER 1

INTRODUCTION

1.1 Motivation

In its 2020 Human Development Report, the UN Development Programme (UNDP) introduced a major update to its most important and influential metric for assessing cross-country development levels with the inclusion of a Planetary Pressures-adjusted Human Development Index (PHDI) alongside their well established, unadjusted Human Development Index (HDI) (United Nations Development Programme, 2020). Along with this new metric a question was posed: how can we bring countries to a point where they have reduced their environmental impact after reaching a high level of development. Given the numerous policies available to governments aiming to accelerate their countries “green transition”, an important question is how effective are these different policies in accelerating this transition.

In pursuit of a "green transition" policymakers have implemented a variety of taxes, incentives, and mandates to achieve emissions reductions in line with their own goals as well as international climate agreements. These can be classified based on their approach, the most common of which are command-and-control, and market-based policies.

Command-and-control policies generally attempt to meet a goal by imposing certain, specific requirements on firms, government agencies, or some other actor. Progress is typically monitored by a regulatory agency and fines or other penalties are issued for non-compliance. These policies can take the form of maximums or bans on certain pollutants, minimums or requirements for a certain activity or investment, or any other fixed target for a business or government body (Stavins, 2020).

While command-and-control schemes in theory offer a high level of control for policymakers to achieve climate goals in an organized and efficient way, they require a well-organized and well-funded regulatory regime, and also have difficulty addressing scenarios in which responsibility is diffuse or when causes of a single issue are multiple. Additionally, these policies can potentially require firms to use less cost-effective means of achieving the desired goals than if they were simply incentivized for doing so (Karp & Gaulding, 1995). While these concerns could be mitigated thorough careful design of the requirements to reflect the situation for each firm or location, in practice this is often not achieved, imposing an inefficient solution on firms and the economy generally (Stavins, 2020).

In response to these perceived shortcomings, market based solutions were developed and implemented which aim to achieve policy goals through incentive structures involving subsidies or taxes on desirable or undesirable activities, respectively, or market-based systems to reduce a specific behavior where it is least costly to do so by assigning a marginal cost to that behavior and, in theory, motivating actors whose marginal benefit of the environmentally harmful behavior to cease that behavior. A large share of environmental policy enacted from the 90s onward has been market-based, including carbon markets, in which a set number of permits to emit a certain amount of carbon are issued in a given year either through auction, assignment based on pre-policy emissions, or a mix of both, and are then traded freely among emitters in a formal market. However, market based policies have also received criticism on the basis that they attempt to commoditize the human and environmental cost of polluting activity, effectively acting as "licenses to pollute" (Stavins, 2020). Additionally, firms could potentially find methods to comply with market-based policies which would allow them to continue polluting, potentially by moving the location of the polluting activity out of the jurisdiction of the enforcing agency.

1.2 The United States' environmental policy landscape

From the late 60s to the early 80s the majority of major US environmental legislation was enacted, including the creation of the US Environmental Protection Agency (EPA) as well as the National Environmental Policy Act (NEPA). This period of legislation was primarily focused on reducing and controlling air, ground, and water pollutants and policies generally took the form of command-and-Control schemes in which a regulatory agency would directly monitor and enforce targets or limitations on pollutants (Karp & Gaulding, 1995). Command-and-Control policies include environmental review policies, in which an environmental agency has the right to review and prohibit the construction of a public or private sector project based on its potential negative impacts on the environment; as well as renewable portfolio standards (RPS) in which utilities must source a certain, legislatively fixed proportion of their electricity from renewable sources, although the definition of renewable varies from policy to policy.

More recently, market-based policies, such as carbon market schemes, have received attention as a way to accelerate decarbonization while maintaining flexibility. However, these have not generally been adopted as widely as command-and-control policies, as their later conception and promotion have coincided with a period of higher skepticism of climate policy in the United States, restricting the implementation of these policies to a few regions where climate change receives more attention from voters and politicians.

In the United States, each state has been largely left to implement its own decarbonization policies, resulting in wide varieties of program types, designs, and combinations between each state, as well as differences in overall emissions intensity, as seen in Figure 1; and distribution of generator types, as seen in Figure 2. These facts make the US an important candidate for study, both as a large carbon emitter and as an environment in which different policies have been implemented and can have their effectiveness measured against one another.

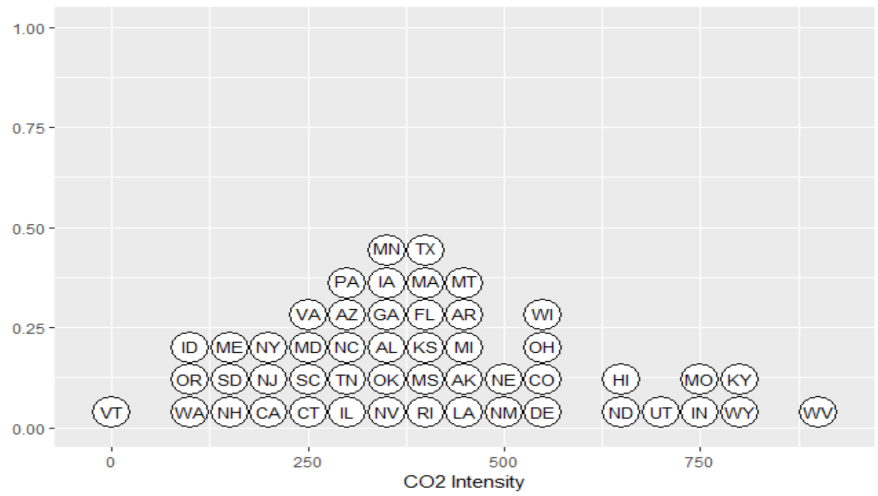

Figure 1. Histogram of states binned by the amount of CO2 emitted per MWh generated (Source: US Energy Information Administration, 2019a)

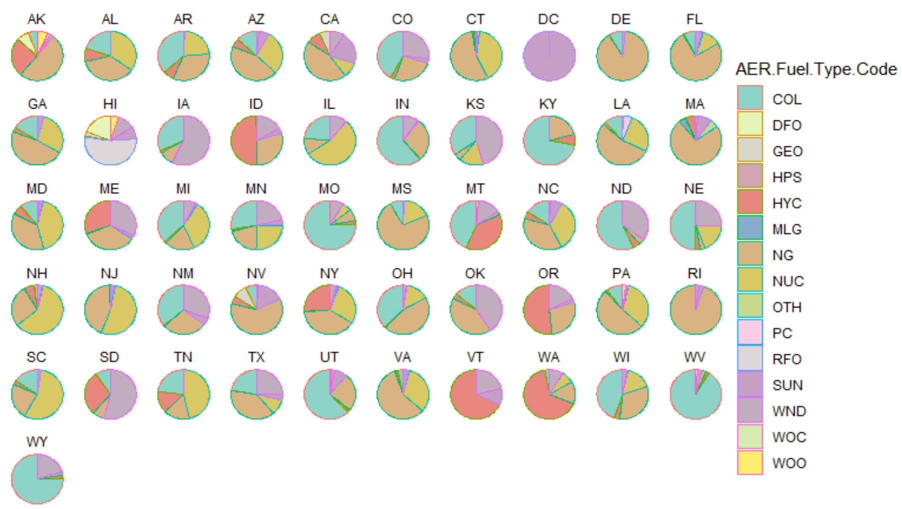

Figure 2. Electrical generation in each state by fuel type (Source: US Energy Information Administration, 2019a)

One reason for this variety in emissions is the variation in geography and climate between states, making each state more or less suitable for different categories of energy generation. The states where coal mining is concentrated, Kentucky, West Virginia, and Wyoming, have the highest emissions per megawatt-hour owing to the large share of coal-fired plants in their electrical generation stock. Oregon and Idaho, despite the significantly more stringent environmental policies of the former versus the latter, have nearly identical emissions per MWh owing to the significant hydroelectric potential in each state.

Another influential factor is the economic growth of the state. While growth in GDP in theory increases demand for electricity, in actuality states with healthier economics have been more able to build out new, cleaner generation facilities to replace older, more polluting generators. Even states with climates unfavorable to renewables, such as New York, have been able to reduce their emissions greatly by replacing their more polluting coal plants with less polluting natural gas plants.

Another important factor, and the focus of this thesis, are the states' policy regimes and their effects on emissions. As mentioned already, states have full discretion on whether or not to implement an emissions reduction policy, and given the variety of policies available for this purpose, these policies have been mixed and matched and tailored in strength by each state according to their situation and priorities. While some policies, such as carbon markets, typically are enacted in states politically dominated by Democrats; others, such as renewable portfolio standards, have seen more broad enactment, although the stringency of the policy is generally higher in states which have seen more Democratic domination; and some policies, such as environmental review, have no obvious partisan lean in the states which have adopted it.

Together, these factors create different combinations of policies, geographies, climates, and economic development and specialization, which allow for an evaluation of the effect of the policies while controlling for other factors which can affect carbon emissions in a given year.

From this point, this study will focus on the effect of state-level environmental policies on the electrical generation sector in the United States, which accounts for 25% of US CO_2 emissions, has a high level of information available to analyze, and has high decarbonization potential, making it a good candidate for study in terms of importance and workability. Also, the States of the United States, whose economies are more similar to each other than the economies of any similarly sized group of countries, are a good environment for examining the effect of these policies ceteris paribus.

The structure of the thesis is as follows. After this introduction, Section 2.1 will discuss the policy landscape in the United States; Section 2.2 will review the literature on alternative ways of assessing the effect of carbon abatement policies; Sections 3.1, 3.2, and 3.3 will discuss the data used in this study, its sources, and the nature of the data and overall trends that they illustrate; Section 4.1 will describe the model used in this study; Section 5.1 will introduce and evaluate the results of the model; and Chapter 6 will conclude this study.

CHAPTER 2

LITERATURE REVIEW

2.1 Environmental policy design in the United States

In pursuit of climate impact abatement, a great variety of policies have been designed and implemented. With goals such as emissions reduction, ecosystem and biodiversity preservation, and air and water quality management. While a single policy can help to achieve multiple different goals, our area of focus is on emissions reduction policies. In the United States specifically, North Carolina State University's Database of State Incentives for Renewable Energy (DSIRE) lists 45 different types of policies implemented by different US government entities.

2.1.1 Command-and-control environmental policies in the United States

As in much of the world, the first wave of environmental legislation in the 1970s in the United States took the form of command-and-control policies, involving enforcement of specific targets and restrictions through a state institution. The subsections below introduce two of these used in the United States – renewable portfolio standards and environmental review policies.

2.1.1.1 Renewable portfolio standards in the United States

Renewable portfolio standards mandate that utilities in a given jurisdiction source a certain amount of their power, as a share of total megawatts, from sources that meet certain requirements including power source as well as other factors which vary by policy. While in theory each policy is seeking to achieve a similar goal through similar means, there are major differences in implementation which forces us to look beyond the nameplate requirement in order to correctly understand how these policies function, or fail to function, in the real world (Wiser et al., 2008).

Already in the name we have our first area of policy differentiation, what exactly is "renewable"? In the case of RPSs, this is defined within each piece of RPS legislation and is influenced by factors unique to the time and place of passage. One of the most important variations is whether, and to what degree, hydroelectric power can count towards a utility's RPS requirements (Wiser et al., 2008). While some US states have no or nearly no hydroelectric generation due to their unsuitable geography, for others hydroelectric is the largest source of or even over half of their total energy output in a given year (US Energy Information Administration, 2019a). While hydroelectric is often considered renewable due to the fact it generates energy from a theoretically non-exhaustible source, it causes other environmental harms which have made its expansion unpopular in the present day, resulting in only a 2% increase in hydroelectric capacity between 2009 and 2021 (US Energy Information Administration, 2019a). Also, the purpose of an RPS is ostensibly to spur new renewable capacity installation, and if utilities could fulfill their renewable requirement with existing hydroelectric stock it would make the policy ineffective for this purpose. For this reason, policymakers often choose either to exclude hydroelectric energy from qualifying sources, or to restrict qualifying installations to ones that meet certain requirements, such as those built before or after a certain year, those which are under a certain capacity, or those which use "run of the river" systems which have less impact on ecosystems NC Clean Energy Technology Center, 2023. This already can make the nameplate policy requirement misleading when taken at face value, as an RPS with a 20% target which can't be met with hydroelectric could be more effective than a 50% target that can be met with hydroelectric if that state already has 30% or more existing hydroelectric capacity (Yin & Powers, 2010).

Adding to this, other sources which may not be conventionally considered renewable, such as landfill gas, are often included as a qualifying source for the purposes of an RPS (NC Clean Energy Technology Center, 2023). In the case of landfill gas, there is some justification for this, as 50% of the gas released from a landfill is methane, and burning it converts it to CO_2, which has a lower environmental impact than releasing the unburnt methane (US Environmental Protection Agency, 2023). In some states however, sources that are not renewable and whose inclusion is less justifiable qualify for the fulfillment of an RPS requirement. One notable and controversial example is Maryland's RPS, which allows waste incineration to qualify for the RPS requirement (NC Clean Energy Technology Center, 2023). As a result, 19% of Maryland's "renewable" RPS qualifying energy stock is from this polluting source of generation (US Energy Information Administration, 2019a).

While this example is extreme, many states allow non-renewables to satisfy their renewables requirement, including Pennsylvania, which allows waste gas from coal mining, and North Carolina, which allows pig waste incineration to contribute towards fulfilling their RPS requirement. While these examples involve converting a harmful by-product to a less harmful one, whether they also induce the activities which generate these by-products by subsidising them is a potential source of policy weakness, and also introduces issues in comparing policies that allow these polluting sources with those whose definition of renewables is more conventional (Yin & Powers, 2010).

Many states also require specific sources to contribute a certain amount for full compliance, with a specific solar requirement the most common implementation. For example, New Jersey's RPS required that, of the 16.675% total requirement, 4.3 percentage points must be from solar sources (NC Clean Energy Technology Center, 2023), and many other states also have similar solar requirements (Wiser et al., 2008).

In terms of enforcement, RPSs differ in how they penalize non-compliance, generally forcing utilities that fail to source enough energy from renewable sources to purchase alternative compliance payments (ACPs) which costs a fixed amount per MWh of missing renewable supply (Heeter et al., 2014).

Another difference between RPS policies is whether investor-owned and other types of utilities, such as cooperative or community-owned utilities, face different requirements. Several states have higher requirements for investor-owned utilities which, combined with different market shares of non-investor-owned utilities, can have an impact on the overall final effect of a policy. In addition, some states have additional requirements for utilities of a certain size, or legislation which is designed to regulate a specific utility or utilities without explicitly naming them (NC Clean Energy Technology Center, 2023).

Beyond policy design, the percentage requirement of an RPS must be compared against the existing renewables stock of the utilities which fall under its regulation. For example, in 2007 North Dakota passed their RPS, with a requirement of a 10% renewable share by 2015 (NC Clean Energy Technology Center, 2023). However, North Dakota is a state with high natural wind power potential, and by 2010 the state had already exceeded the RPS target, and by 2015 the share of qualifying renewables had reached 17% (US Energy Information Administration, 2019a). While it's impossible to say in this case whether the policy had no effect, it's a possibility that the goal would have been achieved whether or not this legislation had been enacted. In several states there are instances of policies whose goals may have been achieved regardless of the passage of the policy, and several states with no RPS targets which have achieved a relatively high level of decarbonization through the use of other policies or almost no environmental incentives whatsoever.

One important design aspect is whether to allow out-of-state renewable energy sources to contribute to a utilities RPS requirement. While states which allow this forbid double-counting, in which the same megawatt is used to satisfy RPS requirements in two different states, buying electricity from states which do not have an RPS can dilute the effect of the requirement, as there is no in-state demand for these renewables leaving them free to be exported to states with more stringent requirements. Indeed, generally utilities do not have to necessarily buy the electricity, but can instead buy RPS credits created by renewable sources which are often sold separately from the generated electricity (Bird & Lokey, 2008).

Previous studies on the effects of renewable portfolio standards in the US have focused primarily on whether they increase renewable energy capacity, not whether they contribute to reducing emissions. One of these, Yin and Powers (2010) , used the state RPS requirement minus the share of renewables in that state as the dependent variable in order properly measure the effect of a policy on new renewables development. In addition, the authors look at several policy design choices and assess their effects, finding that RPSs which mandate utilities provide an option for users to purchase their electricity from only renewable sources are more effective at stimulating renewables development than those that do not. They also found that policies which allow purchasing of renewable credits from out of state are less effective, and that those which have higher alternative compliance payment prices are more effective.

Another study by Greenstone and Nath (2020) found that while RPSs are successful at both reducing CO2 per MWh and at increasing renewable generation capacity, but are associated with increases in electricity costs of between \$190 and \$464 per ton of carbon dioxide abated, higher than the EPAs estimate of the social cost of carbon emissions at \$51 per ton of CO2. They also remark that the effects are sensitive to specification, although the results suggest that there is some effect overall on carbon emissions.

A third study by Joshi (2021), which used a binary variable for the presence of an RPS, found that RPSs had increased renewable capacity installation by over 50% in the states in which they were implemented between 1990 and 2014, with the impact increasing in later years.

As mentioned earlier, all of these studies, and nearly all of the literature on RPS effect, is concerned with their effect on renewable capacity and not directly with their effect on emissions. RPSs are used both as investment stimulation tools and emissions abatement policies (Heeter et al., 2014), and this approach doesn't directly evaluate RPSs on this second criteria. Many studies take the amount of replaced emitting generation as the RPSs carbon abatement, which assumes that promoting renewables specifically is the most effective way to reduce emissions.

2.1.1.2 Environmental review in the United States

Environmental Review is a policy in which a government agency, such as an environmental regulator or permit issuing institution, is given the power to prevent construction of or refuse to issue permits for projects which are determined to have environmental impacts in excess of some criteria. While these criteria vary from policy to policy and regulator to regulator, they generally consider aspects such as air and water quality, threats to wildlife, and carbon emissions. In addition, some policies also consider aspects such as quality of life or noise pollution (US Department of Housing and Urban Development, 2023).

While some policies are only applicable to projects carried out by the state, others apply to all private sector initiatives which require government permission to carry out. This includes any project which requires a change in zoning, access to government owned utilities infrastructure, access to government owned lands, or other government action to complete (Sud & Patnaik, 2022).

While environmental review policies are intended to reduce the impact of new development on the environment, there is a growing debate around whether they achieve this or not (Russo, 2020). In the US, several high-profile renewables projects have faced delays or cancellation due to the regulatory burden imposed by environmental review requirements, which can either make pursuing the project financially unfeasible or can block the project outright on the basis of environmental impacts (Reed, 2021).

This has led to criticism claiming that environmental review policies have a "status quo bias" which puts new projects and construction at an undue burden but allows existing environmentally harmful buildings and infrastructure to exist unaffected, and many activists and policymakers are calling for these policies to be adjusted or repealed to allow projects the projects necessary for decarbonization such as new solar and wind installations as well as high efficiency transmission lines, to be constructed (Reed, 2021).

Currently, little to no research directly assessing the effect of environmental review policies on carbon emissions exists despite increased public scrutiny and calls from legislators in the United States for "permitting reform" to streamline the process by which new green energy installations are approved (Kaufmann, 2023).

2.1.2 Market based environmental policies in the United States

2.1.2.1 Carbon markets in the United States

In the United States, three separate carbon markets exist, one of which is the Regional Greenhouse Gas Initiative (RGGI), and two state level markets, one in California and one in Washington state. No carbon tax schemes have been implemented in the United States. All of these programs operate in a similar way; each year, carbon permits are auctioned on an open market, and can be traded between firms at a market price in order to meet compliance goals. In addition to this, a certain number of emissions permits are issued to polluters based on their emissions prior to the start of the program. In the case of the RGGI, only 2.5% of permits were directly assigned in 2022 (RGGI, 2022).

Each year, a certain number of permits are retired in order to gradually shrink the total carbon emitted within each jurisdiction. This is pre-determined, but can be updated to be more or less stringent according to developments which arise after policy implementation. For example, the RGGI member states, after a period of relatively lax carbon budgets characterized by low carbon market prices and unsold emissions permits from 2009-2014, lowered the carbon budget for subsequent years, initiating an increase in carbon prices and stronger incentives for emitters to reduce their emissions (Gittings & Roach, 2020). As with most carbon markets, there is a penalty per unpermitted ton of carbon emitted, which acts as a maximum permit price given that a firm would have no reason to pay more for a permit than they would be fined for non-compliance. However, the RGGI states individually set penalties for non-compliance (Environmental Defense Fund, 2015), so the relationship between a theoretical maximum allowance price and these varied penalties is potentially complex. In practice the RGGI has a goal minimum and maximum carbon permit price per ton of carbon emitted, which were, respectively, $6.42 and $13.91 in 2022 (New Hampshire Department of Environmental Services and New Hampshire Department of Energy, 2022) . Prices falling outside of this range would likely trigger the issuance of additional permits in the case of high, or the retirement of additional permits the following year in the case of low prices.

In the RGGI, each state auctions its assigned carbon permits and is allowed to spend the proceeds how they see fit. In practice, a majority of the proceeds are spent on programs which aim to reduce emissions, or achieve other environmental goals. In 2019, the RGGI states spent 40% of auction revenue on energy efficiency programs, 18% on clean and renewable energy, and 15% on greenhouse gas abatement. The remainder mostly goes to electricity bill assistance for low-income households, as well as a small share spent on program management and maintenance (Environmental Defense Fund, 2015).

Currently, 11 states participate in RGGI, and one state, Pennsylvania, is currently undergoing the process of joining. While RGGI participation has proved popular in many of the member states, in others various groups have criticized the policy on the basis of its potential harms outweighing the potential benefits of reducing carbon emissions (Navarro, 2011). This has had political consequences, such as delaying Pennsylvania's and Virginia's entry process, and motivating the then-governor of New Jersey, Chris Christie, to pull his state out of the RGGI in 2011, after which it wouldn't rejoin until 2020. It is notable that, unlike all of the policies discussed here, which have been enacted in many geographically distant locations, carbon markets have only been implemented in western coastal states and northeastern states, both of which are regions where support for climate legislation is generally stronger (Gallup, 2019). This may suggest that voters and policymakers perceive prices on carbon as more of an economic burden than programs such as RPSs and Feed-in-Tariffs, which can be more convincingly described as a program to stimulate growth in the renewable energy industry. These barriers have slowed the adoption of carbon markets in the United States, and only one state, Washington in 2023, has implemented a new carbon market structure since 2010.

2.2 Environmental policy effect modeling literature review

In designing a model for assessing the effects of carbon abatement policies, there are a variety of choices to be made. Aside from choices about control variables, evaluation of existing research suggests these four design choices as the most important to design such a model. First, I will discuss emissions data, then policy data, and then, finally, control variable data.

Table 1. Carbon reduction policy effect model design choices

Means of Quantifying Policy	Model Specification	Sample of Countries or Provinces	Means of Qualifying Emissions
As a dummy variable for having or not having a given policy Nominal tax\price\subsidy amount Revenue raised or amount spent as policy amount Market price for emissions trading scheme Sector by sector analysis of policy amount	Panel Data Differences-in-Differences	Use all countries for which data is available Use countries in a given region with similar economic profiles use provinces in a single country	Absolute Emissions Interperiod Change in Emissions Rate of convergence to a steady-state

2.3 Model specification

One important choice is which model specification to use. In current research the two most common choices are panel regressions and differences-in-differences regressions. Panel regressions have some advantages in that they would, in theory, allow multiple policies to be compared in one regression, whereas a differences-in-differences model can only accommodate a single treatment variable. As there is some lack of clarity as to whether the effect of a policy in a given study could be due to the effect of other policies, being able to control for this with the inclusion of multiple policies would be an advantage (Haites, 2018). As countries tend to enact certain policies together, such as renewable energy subsidies and carbon taxes, including each policy in one regression would allow for the individual effect of each policy to be better understood, as not considering the presence of other policies could cause a positive bias in the estimated effect.

A differences-and-differences approach would allow for a simpler regression specification, but would make controlling for different types of lagged effects difficult and requires a well-constructed control group which may be difficult to identify and implement.

If a differences-in-differences approach is used, a treatment group and a control group must be selected. This poses several challenges, the first of which is that the number of countries which have enacted any given environmental policy is limited. For example, the number of countries which have enacted any kind of carbon tax in the year 2023 is only 25, and many of these have only been enacted recently and would not have sufficient data for use in a differences-in-differences model. This is compounded by the fact that many countries which have adopted a policy have done so to a trivial degree, for example, compare the EU carbon market with a 2023 carbon price of $86.53/ton versus the 2023 Kazakhstan carbon market price of $1.08/ton (The World Bank, 2023). This means the number of countries which have a significant enough policy to affect emissions may be lower. However, where a policy would begin to be effective is not fully understood.

Alternatively, a solution used by Rivers and Schauefele (2015) is to use each country's amount of tax or subsidy or market price in t=1 and t=2 as the amount of treatment instead of a dummy variable for treatment/no treatment to account for the varying stringency in policy implementations in a panel regression.

2.4 Selection of countries or provinces

Regarding the selection of countries for study, some studies such as Lin and Li (2011) perform their analysis on countries in a certain region with similar economic characteristics, in this case, and in most such cases, high-wealth EU members, which have similar wealth levels, developmental characteristics, and political systems as well as often comparable carbon policies. Lin and Li specifically used five countries with the policy, Denmark, Finland, Norway, the Netherlands, and Sweden, as the treatment group, and the remaining EU countries as the control group in a differences-in-differences analysis.

Another potential solution is to use provinces in a country where a policy has been implemented as the treatment group and all other provinces as the control group. This has advantages in that the control and treatment groups have key similarities that make them useful for this purpose, as well as easing the standardization of data. However, the number of countries where such an approach could be used is limited, and the ability to generalize the results from such a homogenous group is questionable.

Unsurprisingly, the countries where certain policies have been enacted are not evenly distributed geographically, with a high concentration in Europe, a few in Asia and Oceania, and a handful in other locations (Haites, 2018). This means that ensuring results are due to the effect of the policy and not factors specific to the regions where the policy has been enacted may be difficult.

2.5 Means of quantifying policies

Next, a means to quantify the "amount" of each policy must be determined. While policies can usually be quantified through some Dollar amount, for example a Carbon Market with a market price of $20/ton, each policy is constructed differently with different carve outs, exceptions, and differential rates for different types of consumption. Common variations include rates for commercial, industrial, and residential carbon consumption, exemptions for sectors such as residential or transport, or exemptions for petroleum. This means that the effect of two policies that have an equivalent nominal tax value or other requirement may have substantially different levels of effectiveness.

In addressing how to represent the value of different policies in the data, some studies such as Lin and Li (2011) perform a differences-in-differences in which the implementation of a carbon tax is the variable of interest, meaning there is no differentiation of the amount of tax, simply in whether there is such a policy or not. While the simplicity of this approach is appealing, and removes uncertainty in specifics of the policy being measured across countries, it could create a misleading conclusion, as the study might include inadequate policies in the treatment group, reducing the apparent effectiveness of all carbon taxes. The authors in this case mitigate this by only using EU countries with similarly high levels of carbon tax in their treatment group, but this may not be feasible in all settings.

Aydin and Esen (2018) address this by using environmental tax revenue as a share of GDP as the regressor. This approach does give a seemingly simple way of standardizing policy size across countries, but ignores how different policy setups may be more or less effective in reducing overall emissions. In addition, tax revenue depends not only on policy size, but amount of taxable emissions as well, meaning that a country that emits more per capita than another would have an apparently stronger policy than one with an equivalent policy but less emissions, although this could potentially be controlled for.

Saether (2021) uses the market price of a carbon market as the dependent variable. This is arguably the most straightforward approach, which ignores structural differences in which sectors are exempted and uses the nominal policy amount directly.

Another consideration is whether to limit the analysis to a single sector and regress the effect of a policy amount for a certain sector on emissions from that sector. While this does allow for more dynamic analysis of different policy designs and would provide potentially insightful sector-specific policy effect estimations, it could potentially increase the complexity of the analysis and could introduce other sources of bias depending on which countries have exempted or reduced burdens on certain sectors. Another issue would be finding sector-by-sector emissions data to perform the regression with.

2.6 Means of quantifying emissions

In much of the existing research, absolute or log-adjusted emissions are used as the dependent variable of interest, but other alternatives exist. Another common choice is to use the interperiod change in emissions, as Best, Burke, and Jotzo (2020) used in their study on carbon pricing efficacy. A less common but potentially useful option used by Lin and Li (2011) in their study on carbon taxes in the EU is to use a beta-convergence test to determine if the rate of convergence to a hypothetical steady-state is slower due to a certain policy.

Another choice is whether to examine emissions in the economy generally, or those of a certain sector, typically electricity or transportation, to identify the effect of these policies. The latter has certain advantages by both controlling for differences in the relative sizes of sectors across countries as well as simplifying the comparison of different policies across countries, as exceptions and carve-outs typically are applied on a sector-by-sector basis. While this does not measure the effect of a policy on emissions generally, given the relatively high share of emissions from electrical generation and transportation in many countries it directly explains at least a large share of potential emissions and could be generalized to the overall economy.

2.7 Previous policy effect studies

Gittings and Roach (2020) performed a differences-in-differences analysis with two treatment variables, one for the beginning of the Regional Greenhouse Gas Initiative in 2009, and another for the start of the period where the lowered emissions cap was implemented in 2014. The significance of the lowered emissions cap in a carbon exchange market is substantial, as if the supply of carbon permits is too high, prices will not be high enough to deter emissions and spur the intended investment in lower carbon alternatives. This seemingly did result in high permit prices, and 2014 was the first year since 2009 where there were no unsold emissions permits.

One dynamic of interest is that of carbon spillover, when a jurisdiction with a carbon market begins importing power from jurisdictions without it, resulting in the pollution "leaking" from the carbon market and reducing the effectiveness of the carbon market in incentivizing carbon reducing investment. In addition to assessing the effect of the RGGI on carbon emissions, Gittings and Roach (2020) also attempted to determine the scale of leaked emissions and found the measured effect insignificant. Overall, the paper found that CO2 emissions from coal generators were reduced by the RGGI, primarily after the cap tightening, but found no significant effect for reductions from natural gas generators.

Another study by Yan (2021) performed a similar differences-in-differences regression on coal and natural gas consumption in the RGGI states, finding that the RGGI reduced both coal and natural gas consumption. To test for leakage, a dummy variable was added in for Pennsylvania and Ohio, as these two states, due to their proximity to the RGGI market and interconnection with RGGI states, would be ideal candidates for leakage to occur in. This method found that proximity to the RGGI increased natural gas construction in those states, but also found a reduction in coal consumption. The authors suggest this may be due to a reduction in natural gas prices increasing natural gas investment and thereby accelerating the phasing out of coal. They also found a significant increase in electricity transfers to the RGGI market states during the program period. Of note is that this study includes no control variables, such as GDP growth, commonly used in studies on carbon markets.

Regarding the effect of renewable portfolio standards, one study by Shirmali, Chan, Jenner, Groba, and Indvik (2015) found that a more stringent RPS is effective in increasing renewables deployment. The study evaluated the effect of various policy design choices such as caps on electricity increases due to the policy and whether out-of-state power can be used to satisfy the requirement. The primary policy effect is constructed by taking the percentage RPS requirement, multiplied by the electricity which is subject to the policy, multiplied by the total electrical generation, and subtracting the existing renewable stock from that to get the difference between current stock and the goal stock. This study performs its analysis on a year by year basis, without any way to account for investment spurred by a future RPS target.

In addition to econometric studies, much of the literature takes the form of analytical models of the effects of environmental policies, often carbon taxes or carbon markets. In his influential paper, Goto (1995) uses sector energy use and emissions data to calculate the required tax to achieve a given emissions target and the resulting economic impact of this tax. He found that in order to maintain Japan's 1990 level of emissions in 2010, a difference of 26.47%, a tax of $200/Tonne of CO2 would be necessary. Despite how high this is relative to other existing carbon taxes, he also estimated that a tax of this magnitude would only negative impact GNP by .11%. However, few such studies on the United States have been conducted in the last decade, likely due to the fact that existing carbon market schemes provide an opportunity for empirical analysis.

CHAPTER 3

DATA

3.1 Generator data

To assess the effect of each of our policies on the electrical generation sector of each state, information about electrical generation, fuel usage, location, and energy source was gathered from EIA forms 923 and 860. Aside from generators which aren't required to report their information separately, whose information is reported in aggregate on the state level, all information is on the level of individual generators, allowing us to track the activity of individual plants, or aggregate certain fuel types together to study other trends.

To calculate emissions, emissions factors provided by the US Energy Information Administration (2023) are multiplied with each generator's fuel consumption in MMBtu in a given year (US Energy Information Administration, 2019a). While this may not allow us to fully capture the effects of any carbon scrubbing technology implementation, it still reflects increases in generator efficiency.

From this data certain trends become apparent which will affect how our model will be designed. One important trend is that in general, fossil fuel generators do not typically shut down production gradually, but will cease production over the span of one or two years. This is likely because once operating the facility becomes unprofitable due to competition with cheaper, newer, or more environmentally friendly facilities there's no reason to continue operating. While some plants display some downward trend in production, a sudden shutdown is also common in the data. To illustrate this, the annual generation data of plants in Alabama in figure 3 displays this tendency in some plants, such as plant 8, which produced its peak generation in 2018 and then shut down in 2019.

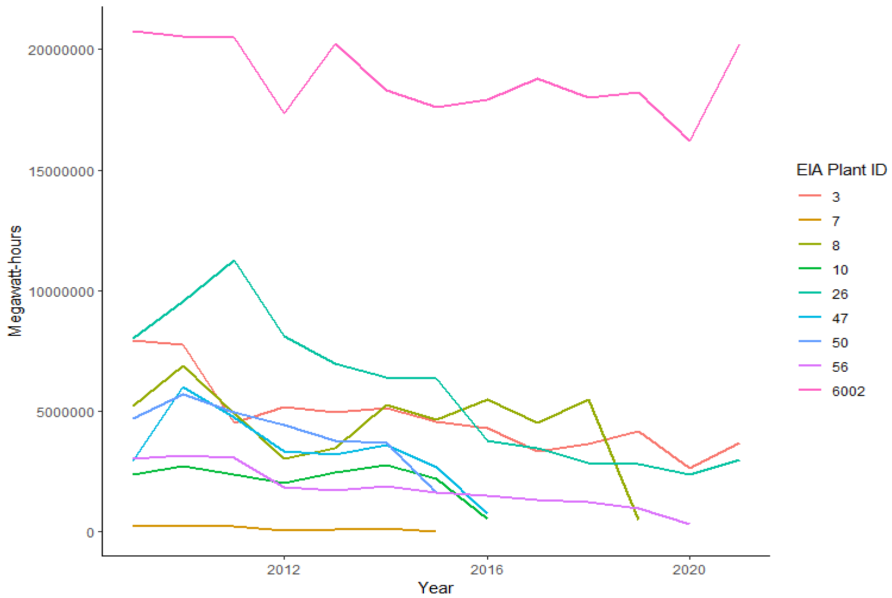

Figure 3. Annual generation of coal burning power plants in Alabama (US Energy Information Administration, 2019a)

This pattern, which can be seen across the United States, makes trying to measure the effect of a policy on the plant level fraught with issues; if the effect of a policy is applied gradually, as is generally the case, and if plants are shut down in response to the policy all at once, which is often the case, the effect would be difficult to capture through typical linear models.

In contrast, the aggregated production by fuel source at the state level shows clear, linear trends which are themselves illustrative as well as suitable for a variety of analyses. Figure 4, which shows the total generation by source in Alabama, illustrates a common transition typology. Even though Alabama has none of the environmental policies previously discussed or included in our analysis, coal generation has markedly decreased over the period of study and has been replaced primarily by increases in natural gas generation and nuclear power. While neither of these replacement sources are renewable, they do emit less, or nothing, per MWh of electricity generated.

Renewable generation facilities are nearly non-existent in Alabama, and along with nearly no solar power generation, Alabama is one of the few states with no wind power whatsoever as of 2021. Unfavorable geography for wind and solar, combined

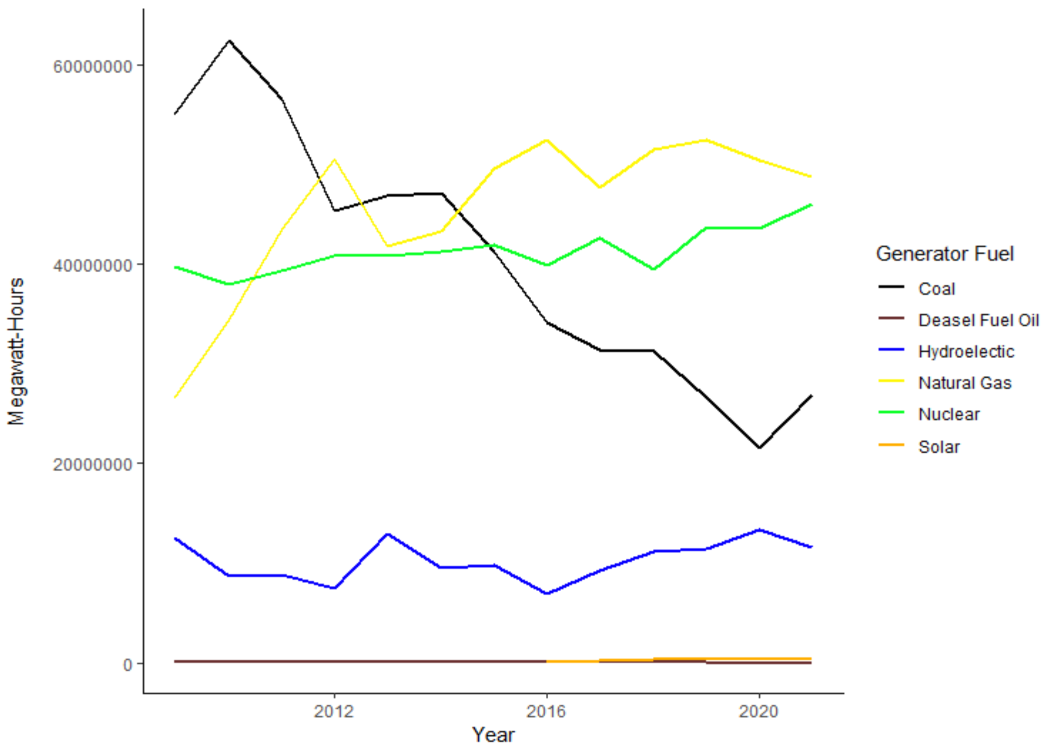

Figure 4. Annual generation by fuel source in Alabama (US Energy Information Administration, 2019a)

with governments which have not prioritized renewable energy, have both contributed to one of the lowest shares of wind and solar generation in the United States, although it still has a higher than average share of non-emitting sources owing to its large hydroelectric and nuclear capacity.

Where geography allows, the declining share of coal power has been replaced by renewables, often with or without the mandate or support of the state. One notable example is Oklahoma, which has extremely favorable geography for wind energy, and has nearly entirely replaced its coal generation capacity with wind power, as can be seen in figure 12. Here the trend is stark; coal has almost entirely been replaced by wind power. Oklahoma is another state where the state government has implemented none of the policies examined in this study, and one which is generally skeptical of environmental policy and climate change generally. However, economic forces such as the inefficiency of coal plants and low cost of wind where suitable have eliminated most coal production regardless.

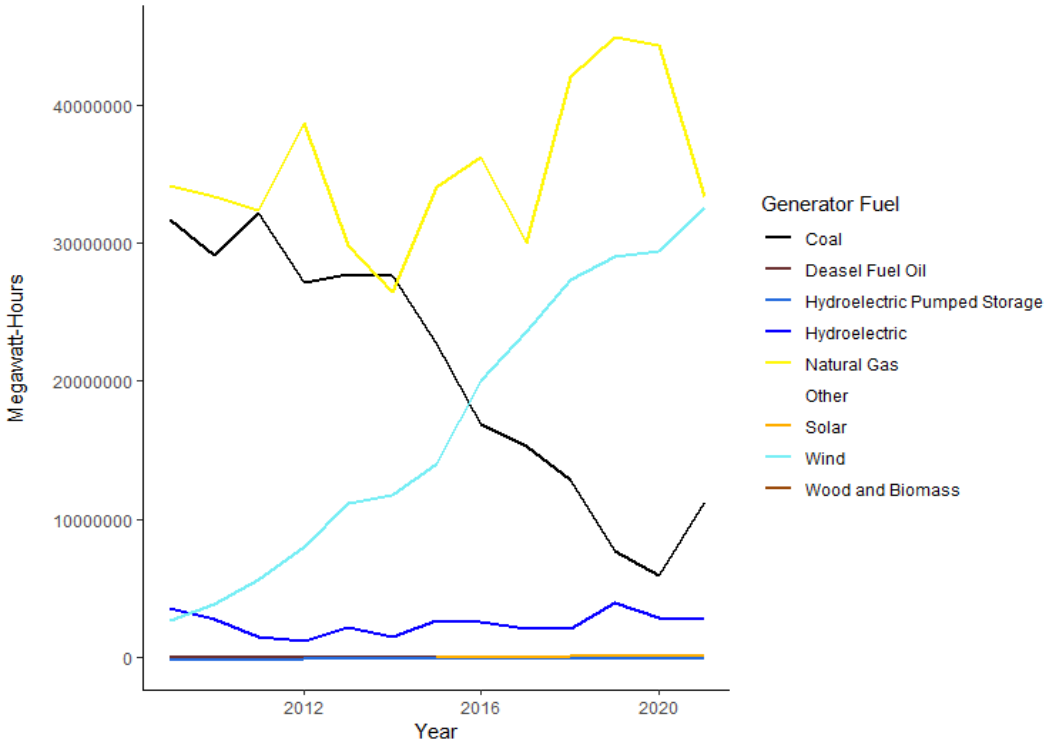

Figure 5. Annual generation by fuel source in Oklahoma (US Energy Information Administration, 2019a)

Electrical generation decarbonization in the United States, where it takes place, generally takes one of three forms. One, coal generation is eliminated and does not need to be fully replaced due to efficiency improvements and slow economic growth. This group includes states such as Alabama. The second group includes states such as Oklahoma, where local conditions allow for the use of wind or solar, and coal plants' generation capacity can be replaced wholesale by a renewable source. The third group includes states such as Pennsylvania (see Figure 13) where phased out coal capacity is being replaced with natural gas generators.

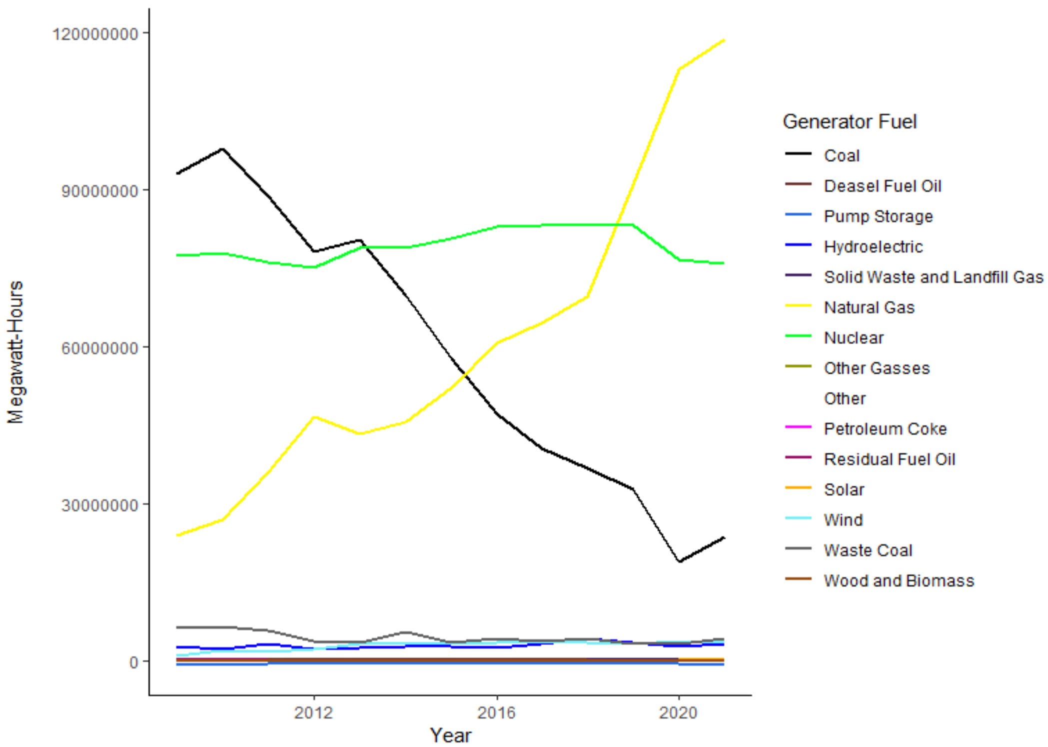

Figure 6. Annual generation by fuel source in Pennsylvania (US Energy Information Administration, 2019a)

3.2 Policy data

3.2.1 Environmental review

To determine whether a state has environmental review (see figure 7), a list of state policies from NEPA.gov (2023), the website for the national environmental policy act was used. All policies were passed before the period of study and none have been repealed (NEPA.gov, 2023). Despite the potential hoped-for environmental benefits and feared economic harms, environmental review policies face little legislative scrutiny after their passage, while at the same time there has been little effort to expand it to states beyond those which adopted the policy in the 1970s. This makes environmental review policies unique, as they have not experienced the expansions, revisions, or repeals other policies discussed here have been.

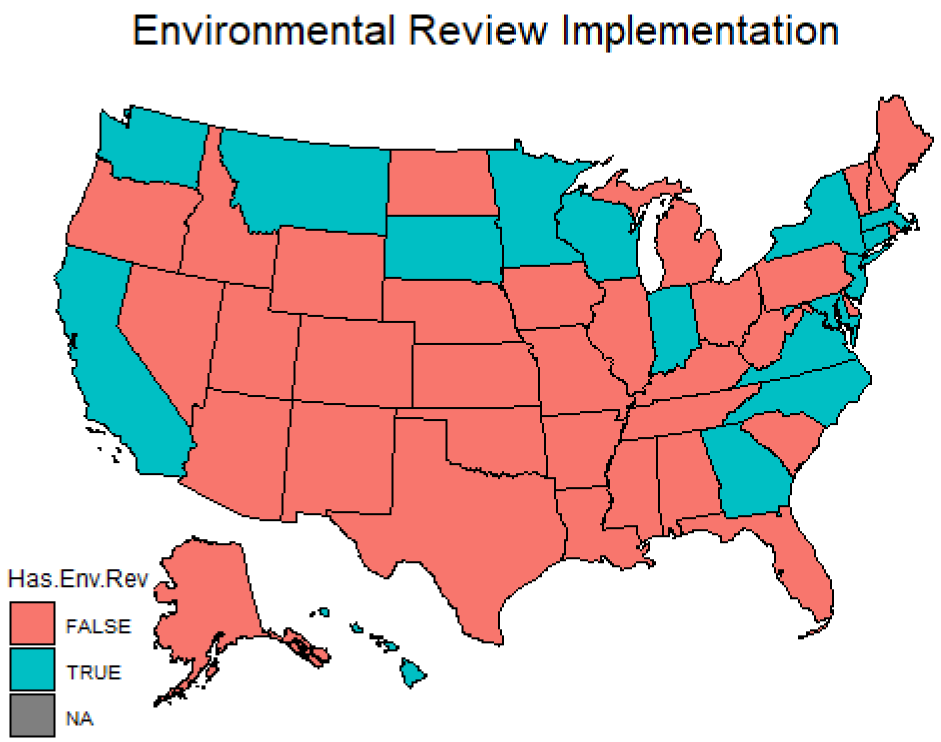

Figure 7. Map of states with and without environmental review policies

For the purposes of this analysis, whether a state has environmental review or not is represented as a binary variable. This does overlook the complexity and variety of environmental review systems, and the actual regulatory burden can vary greatly between states with environmental review. However, capturing the differences between these policies can be extremely difficult, as often differences take the form of bureaucratic habit which is not reflected in the text of the law or reflected in other sources. Another difficulty are the differences between each state's existing permitting process, and how each has chosen to integrate environmental concerns into that process. Further difficulties arise when considering that policies may have different requirements for projects of different sizes or in different sectors.

For this reason, this analysis currently treats the presence of environmental review as a dummy variable. While this could suffer from treating these varied policies identically, in practice all of these policies impose some additional burden on renewable investment, and even with this method a significant effect could be measured from our data.

3.2.2 Carbon markets

Data on carbon market prices was sourced from RGGI.org (see figure 8), the website for the regional greenhouse gas initiative. The RGGI is ideal for our purposes as it imposes identical policies on several states, giving us a highly controlled sample from which to measure the effect of the policy. Our variable amount is in $ per short ton of carbon dioxide, and is calculated by averaging the auction prices for all auctions for permits which can be used for compliance in a given year.

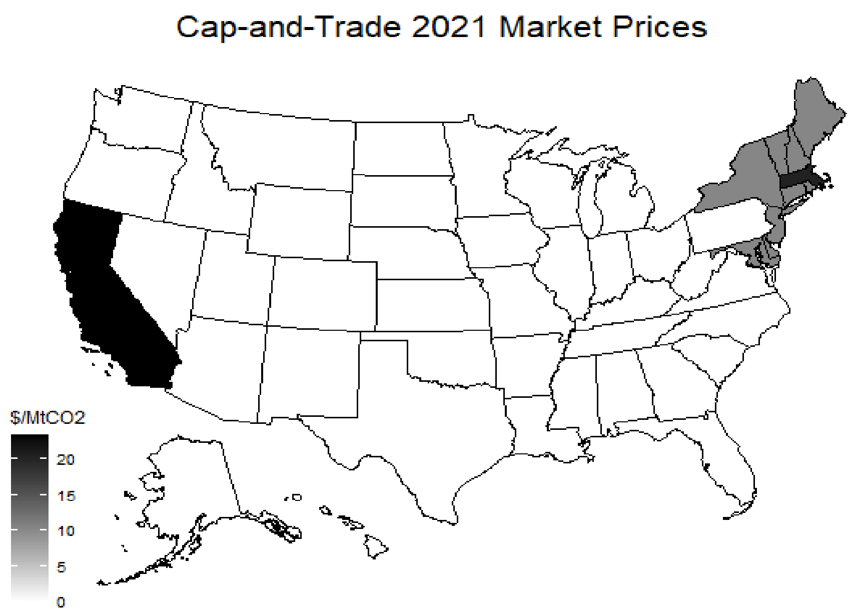

Figure 8. Carbon market prices by state in the US (RGGI, 2022) (California Air Resources Board, 2022)

Two other states have different carbon prices. One of which is Massachusetts, which imposes its own, stricter requirement on the power industry in addition to the

RGGI requirements by forcing polluting generators to buy additional permits that other polluting facilities do not need to purchase. The other is California, which has an independent carbon market from the RGGI. Because California's carbon market differs from the RGGI system in several ways, such as how allowances are distributed in auctions versus assignments, California has not been included in the model in order to maintain the consistency of the RGGI policy's effect as represented in our study.

Table 2. Carbon market prices by year and market (RGGI, 2022) (California Air Resources Board, 2022)

Year	California	RGGI	RGGI + MA
2009	0	3.03	3.03
2010	0	2.11	2.11
2011	0	2.08	2.08
2012	0	2.13	2.13
2013	0	3.22	3.22
2014	0	5.27	5.27
2015	12.28	6.73	6.73
2016	12.73	4.92	4.92
2017	13.71	3.77	3.77
2018	14.84	4.87	15.93
2019	16.41	5.98	18.31

3.2.3 Renewable portfolio standards

Renewable portfolio standards are also designed in a variety of ways which could impact how a policy would affect emissions in practice. One study by Yin and

Powers (2010) performed an analysis on the effect of RPSs on renewable investment, specifically looking at which policy design choices had significant impacts on renewable investment. Of the factors evaluated, three were found to be significant: ease of trading renewable credits across state lines, whether the state mandated that consumers have the option to purchase only renewable energy from their utility, and existing renewable generation as defined by the policy.

The first of these, whether and to what extent renewables in other states can count towards fulfilling an RPS requirement, is not necessarily as important to to the effect of a policy on a state's carbon emissions. While the authors found that allowing cross-state trade reduced in-state investment, in terms of carbon emissions a coal plant that is replaced by a renewable source is a reduction in carbon, regardless of which state the replacement generator is located in.

The second, a mandatory green power option, is not directly connected to the effect of an RPS, and theoretically utilities should meet their obligations regardless of whether consumers have a green option or not. As an example, Virginia passed a mandatory green power option in 2007, but only passed an RPS in 2020. While a mandatory green option is a potentially useful policy, it is not connected with the functioning of an RPS directly, although they are often passed together.

The third, existing generation capacity, is critical to whether an RPS is strong, or even acting on utilities' decisions. Several RPSs have requirements well below the existing renewable capacity in their state, effectively providing no incentive for additional renewable energy investment (US Energy Information Administration, 2019a). In order to measure this effect properly, existing renewable capacity in a state as defined by its RPS must be calculated and then subtracted from the following year's RPS target to calculate the gap between the goal and current production, which is then used as the effect of the policy for that year. While it is often not possible to fully distinguish which generators meet the criteria for a given policy with the data available, for major sources of power which different policies include or exclude such as hydroelectric, plants have been included in the state's RPS qualified generation capacity where the policy allows it to count towards the policy requirement.

Table 3. RPS requirements, qualifying generation, and inter-period difference in 2019

Year	State	RPS	Qualifying Energy	$RPSEffect_{i,t}$
2019	AZ	10.000%	5.34%	4.66
2019	CO	30.000%	21.71%	8.29
2019	CT	29.000%	2.25%	26.75
2019	DE	20.000%	5.98%	14.02
2019	HI	30.000%	12.20%	17.80
2019	IL	16.000%	8.30%	7.70
2019	IN	7.000%	9.02%	0.00
2019	KS	20.000%	41.70%	0.00
2019	MD	28.000%	3.57%	24.43
2019	ME	14.000%	45.27%	0.00
2019	MI	12.500%	9.37%	3.13
2019	MN	21.500%	22.57%	0.00
2019	MO	10.000%	3.93%	6.07
2019	MT	15.000%	8.75%	6.25
2019	NC	10.000%	8.02%	1.98
2019	ND	10.000%	27.37%	0.00
2019	NH	19.200%	9.14%	10.06
2019	NJ	16.029%	3.10%	12.93
2019	NM	20.000%	23.71%	0.00
2019	NY	26.440%	28.29%	0.00
2019	OH	5.500%	3.00%	2.50
2019	OK	15.000%	34.01%	0.00
2019	OR	20.000%	13.48%	6.52
2019	PA	7.500%	3.82%	3.68
2019	RI	16.000%	6.15%	9.85
2019	SD	10.000%	73.80%	0.00
2019	VT	59.000%	99.92%	0.00
2019	WA	15.000%	8.06%	6.94
2019	WI	10.000%	5.17%	4.83

3.3 Control variables

3.3.1 GDP Per capita

Data on GDP is sourced from the Bureau of Economic Analysis, which provides GDP figures for each state in a given year. In our study, we use the percentage growth between the previous year and current year as a control variable, as our dependent variable is change in emissions.

3.3.2 Solar potential

Data on solar power potential is calculated by the Nebraska Energy Office based on data from the National Renewable Energy Lab's (NREL) renewable energy dataset. By this metric Nevada has the highest potential with 1.19, Washington the lowest with .67, and California indexed at 1 (Nebraska Department of Environment and Energy, 2006).

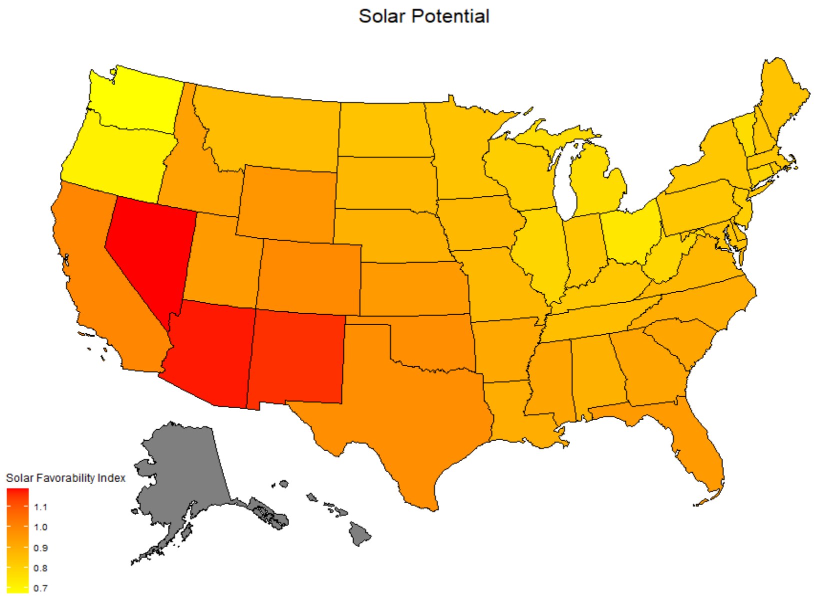

Figure 9. Solar energy index

3.3.3 Wind potential

Wind energy potential is provided by the US Department of energy as Megawatt hours which could be generated in a given state at different Technology-Resource Groups (TRG), which categorize wind in an area by its distribution of wind speeds.

Most wind projects in the United States are constructed in areas in TRG classes 2-7, owing to the rarity of ideal class 1 sites and unfavorability of class 8-10 sites (National Renewable Energy Laboratory, 2022). For this reason, our data uses the ratio of Megawatts of potential wind energy in classes 1-7 to total existing generation as the control for wind energy potential.

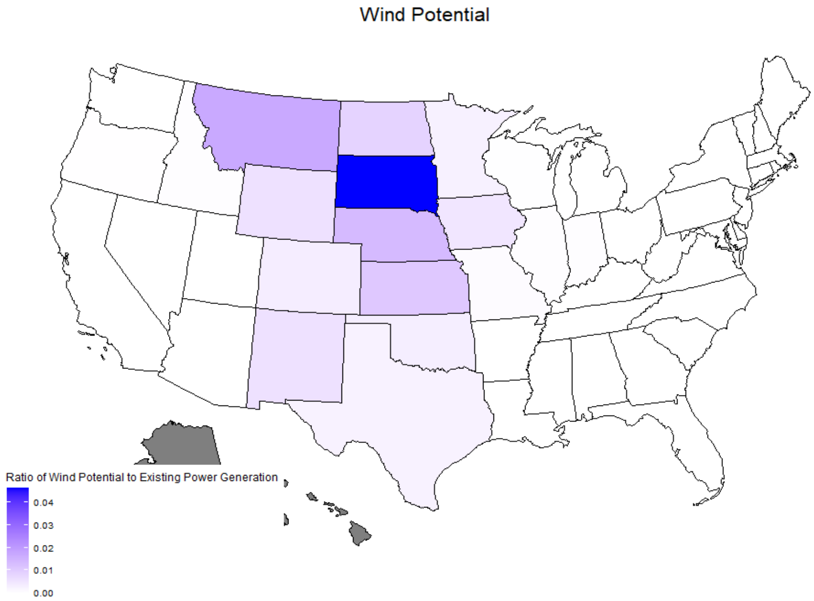

Figure 10. Ratio of TRG class 1-7 wind energy potential to total power consumption

3.3.4 Share of generation from coal

From the EIA generator data, the share of all power generation from coal is calculated for each year and is included as a control. As typically new renewable energy and natural gas sources replace existing coal power, a high share of coal energy can be more easily replaced with a lower emissions source, which would allow for a faster rate of decarbonization.

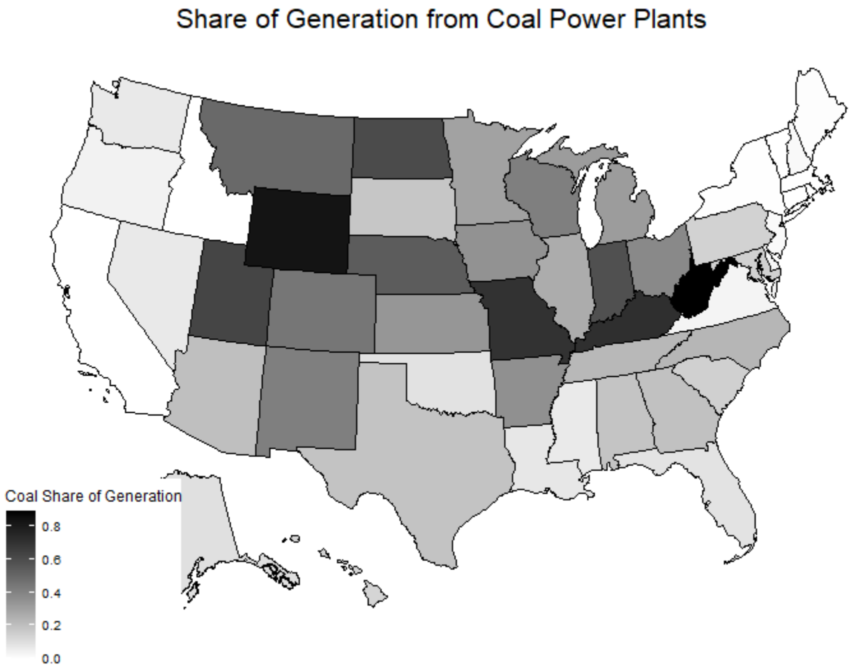

Figure 11. Share of state power generation in MWh from coal power plants

CHAPTER 4
MODEL

4.1 Regression model

Panel regressions are a well-established model for assessing the effect of environmental policies. For assessing RPSs, Yin and Powers (2010) built on existing literature through their panel regression which included a variety of policy design choices and related policies, and found that the positive effect on renewable investment was significant when RPSs were measured as the difference between the policy target and existing production.

In measuring the effect of carbon markets, several studies covering both the RGGI as well as other markets have used a panel regression to identify their effect. Saether (2021) looked at the effect of carbon market prices on CO2 intensity in several European Countries using a fixed effects panel regression and found no significant effect from the carbon market on CO2 intensity. Important for us is that this does not necessarily indicate that carbon markets did not reduce emissions, but rather that they did not make the energy sector more efficient in terms of CO2 per KWh. The study by Murray, Maniloff, and Murray (2015) on the effect of the RGGI on capacity utilization found that participation in the RGGI increased gas generator capacity utilization and reduced coal capacity utilization and attributed RGGI with a 40% drop in emissions in the policy region, although their study did not attempt to find the effect on carbon emissions directly through the model.

This study builds upon these by applying the panel regression approach directly to the growth in emissions in each state, allowing us to take advantage in the changes in policy to better understand the relationship between policy strength and effect on emissions. Measuring the effect on emissions specifically is of great importance, as reducing emissions overall is one of or the most important goal of these policies, and growth in renewable energy capacity is not an inherent benefit from this perspective if it is not associated with actual reductions in emissions. It would be conceivable that a policy could grow renewable energy installation without reducing emissions, so measuring the effect directly might reveal important policy implications.

Our regression is therefore constructed as follows:

$$\left(\frac{CO2_{i,t}}{CO2_{i,t-1}} - 1\right) = \alpha_i + \alpha_t +$$
$$\beta_1(RGGIPrice_t) + \beta_2(RPSGap_{i,t}) + \beta_3\left(\frac{GDP_{i,t}}{GDP_{i,t-1}} - 1\right) +$$
$$\gamma_1(EnvRev_i) + \gamma_2(SolarPotential_i) + \gamma_3(WindPotential_i)$$

Given

$$RPSGap_{i,t} = \begin{cases} 0 & if \quad RPSTarget_{i,t} < RPSGen_{i,t-1} \\ RPSTarget_{i,t} - RPSGen_{i,t-1} & if \quad RPSTarget_{i,t} > RPSGen_{i,t-1} \end{cases}$$

Where $RPSTarget_{i,t}$ is the nameplate policy requirement in year t, and $RPSGen_{i,t-1}$ is the existing capacity in year $t-1$ which qualifies for the RPS in state i.

Several of our control variables and one policy variable are dummy variables, as they are either indices of renewable suitability, which does not vary, or is the presence of environmental review, which has not been passed or repealed in any state during the policy period, making it a dummy variable. Of course, in a model with individual random effects these cannot be included, so this is one reason to perform and compare models both with and without individual effects.

One more consideration is that our generator data includes industrial co-generation facilities, which are facilities which generate electricity using waste heat form industrial processes. These have emissions associated with them in the data, but these are the emissions of whatever industrial process which generates the heat used, so efficiency varies greatly between installations. While these generators often appear as the most polluting per KWh, they are only taking advantage of an industrial facility that is already operational, and shutting down these facilities wouldn't eliminate the carbon emissions associated with them in reality, but would in the data. Because of the different incentives these facilities face and because their growth and decline is influenced by multiple exogenous factors such as policies on other industries and demand for common sources of co-generation energy, including oil refineries and smelting operations, these sources of energy may not be as sensitive to the policies being evaluated. For these reasons, these facilities have been removed from the state emissions totals in the dependent variable, although they are included for the calculation of RPS effects where applicable. These facilities account for 7.8% of generation capacity, and removing them will allow us to better control for the effect of these policies on the energy generation sector.

CHAPTER 5

ANALYSIS AND RESULTS

5.1 Results

The results of the regression are reported below in Table 4. Three models, one with individual and time effects, one with time fixed effects, and one with time fixed effects as well as our dummy variables are included.

Table 4. Regression results: log difference in annual emissions

	Panel Regression		
	all with fixed year effects		
	Fixed Individual	Only Fixed Time	With Dummy Variables
$CPrice$	-0.015^{**}	-0.016^{***}	-0.016^{***}
	(0.007)	(0.004)	(0.004)
$GDPGrowth$	0.001	0.001	0.001
	(0.003)	(0.003)	(0.003)
$RPSEffect$	-0.001	0.004^{**}	0.004^{**}
	(0.004)	(0.002)	(0.002)
$CoalRatio$	0.475^{**}	-0.029	-0.029
	(.152)	(0.026)	(0.027)
$HasEnvRev$			0.005
			(0.016)
$SolarPotential$			-0.018
			(0.066)
$WindPotential$			0.097
			(0.907)
F-test	3.26036^{**}	4.6685^{***}	2.69162^{***}
T	10	10	10
N	47	47	47

Nota: $^{*}p < 0.1$; $^{**}p < 0.05$; $^{***}p < 0.01$

In most cases, the sign of the effect on change in carbon emissions, where a negative sign indicates a decrease in carbon emissions associated with that variable, aligns with expectations. In all models, higher carbon prices are associated with greater decreases in emissions. The sign on the ratio of energy from coal generation in each period is positive and significant in the first specification, and negative but insignificant in the other two. GDP growth is associated with an increase in emissions, although not significantly. Solar potential has a negative sign and Wind potential has a positive sign, but both are highly insignificant. The sign for environmental review is also positive, but is not significant.

In our model, the effect of carbon price is significant and negative across all specifications. This is as expected, and is in line with other studies such as Murray et al. (2015) which found that the RGGI reduced carbon emissions in participating states, and with Haites (2018), whose meta-analysis found that carbon markets were effective tools to reduce emissions across different studies and markets.

Regarding environmental review, as discussed earlier the actual effects of environmental review policies and the burdens they place on renewable energy installations is rapidly becoming a prominent issue in policy circles. As far as we are aware, this is the first study to attempt to directly link the effect of this policy on decarbonization of the energy sector. Our results are insignificant, but have a positive sign. This could suggest that the effect of environmental review is outweighed by other factors, and that reducing this burden would not itself affect investment decisions substantially. However, our study didn't control for the myriad of differences between policies, such as which agency is responsible for the review, which impacts are considered, and permit processing periods. It's possible that, once these effects are controlled for, the effect would become significant, but as there is no existing database for any of this information it would itself require significant research in order to evaluate. Still, given the growing attention these policies are receiving, there is potential for more detailed research on the effects of environmental review policies in the future.

Of interest is the sign on RPS effect, which is our difference between target and actual renewable generation. While in the fixed individual effects model the sign was negative but insignificant, in the other models it was both significant and positive. This suggests that a higher renewable portfolio standard relative to existing renewables decreases the speed of decarbonization, which is unintuitive and would be considered a failure of the policy if true.

There are a few possible explanations for this. One is that there are individual effects not accounted for by our controls but which are captured in the state fixed effects. This is possible, and indeed our model with state fixed effects has a negative, but insignificant, estimate for our RPS effect. Even in that case though, there is, at best, no evidence for RPSs having an effect in either direction.

Another possible explanation is that, while RPSs do increase renewable capacity, increasing renewable capacity is not a more effective way of reducing carbon emissions than other strategies, namely replacing existing coal and other, more polluting facilities with natural gas generators.

Table 5. Cost and efficiency of different generator types

Generator Type	Emissions Factor	LCOE	Capital Cost
Onshore Wind	0	56.5	40.2
Photovoltaic	0	62.5	50.2
Conventional CC	0.97	46.8	9.1
Advanced CC	0.97	41.6	7.4
Coal	2.26	NB	NB

Nearly all states have some portion of coal power generation, and in 2009 this was even higher. In pursuit of decarbonizaiton, the ideal strategy would be to replace these generators first. At the same time, rising coal prices have driven the cost of operating coal power plants, most of which are nearing the end of their lifespan. This means that, regardless of policy regime, coal power plants are naturally being phased out due to market forces, although the speed of this could be accelerated through policy. The question then is what sources should replace this generation capacity.

While a new renewable facility will have no emission, replacing a coal generator with natural gas generation will eliminate some emissions. Specifically, the EIA estimates that the average natural gas generator in the United States emits 0.97 pounds (440 grams) of carbon per kWh, versus 2.26 pounds (1025 grams) per kWh for the average coal generator currently operational, meaning that replacing a coal generators annual generation with natural gas removes 57% of the carbon previously emitted (US Energy Information Administration, 2023).

Of course, replacing a coal generator with wind or solar would eliminate 100% of that generator's emissions, but with the same capital budget at least 300% more coal power can be replaced with natural gas (US Energy Information Administration, 2019b), and doing so would eliminate at least 128% more carbon emissions than the equivalent initial cost wind or solar installation.

Of course, this strategy produces carbon reduction at a fast pace, but building new natural gas capacity would create a stock of new, profitable generation capacity that would have to be retired early in order to fully decarbonize. This approach would then save on capital cost initially, but might take more time and investment in order to completely decarbonize. Additionally, these prices are 2018 estimates for plants opening in 2023, and solar and wind capital costs and LCOE have both been steadily decreasing. It is possible that eventually the capital cost difference between natural gas and renewables installations could shrink enough to make the upfront cost more manageable and the LCOE lower than for natural gas, especially if natural gas prices rise above their current historic lows.

CHAPTER 6
CONCLUSION

This thesis aimed to understand which state policies in the United States were accelerating decarbonization, and to understand which approach, command-and-control or market-based, is more effective generally. Our results suggest that, in the United States, carbon price schemes have been met with success, while renewable portfolio standards have actually impeded decarbonization. This second result is in contrast to some existing literature, such as Yin and Powers (2010) which performed a similar study and found that RPSs were effective in increasing renewables investment in the United States, for example. Another study by Bersalli, Menanteau, and El-Methni (2020) found that RPSs increased investment in Europe and Latin America, although the effectiveness of these policies varied by region, with RPSs in Europe stimulating an estimated 2 to 4 times more development than policies in Latin America.

Also, as discussed earlier, the capital cost of new natural gas generation is 1/4th or less of an equivalently sized solar or wind facility, and the lifetime cost of energy is also lower for natural gas generation. In Europe, the higher and more volatile price of natural gas may reduce the cost-effectiveness of natural gas to the point where differences in operating costs make the higher capital costs of renewable installation more bearable.

This is further highlighted by state-level data as well. Consider that Oklahoma, a state in which none of the environmental policies discussed have been implemented, and where support for decarbonization policies is very low, has been far more successful at growing its renewable stock than Pennsylvania, a state which is both a participant in RGGI as well as having an RPS. While both states have natural gas resources, in Oklahoma favorable geography for wind energy has prompted rapid investment in wind energy despite a lack of decarbonization policy, while Pennsylvania has had difficulties meeting their RPS targets due to the relative unfavorability of renewable generation.

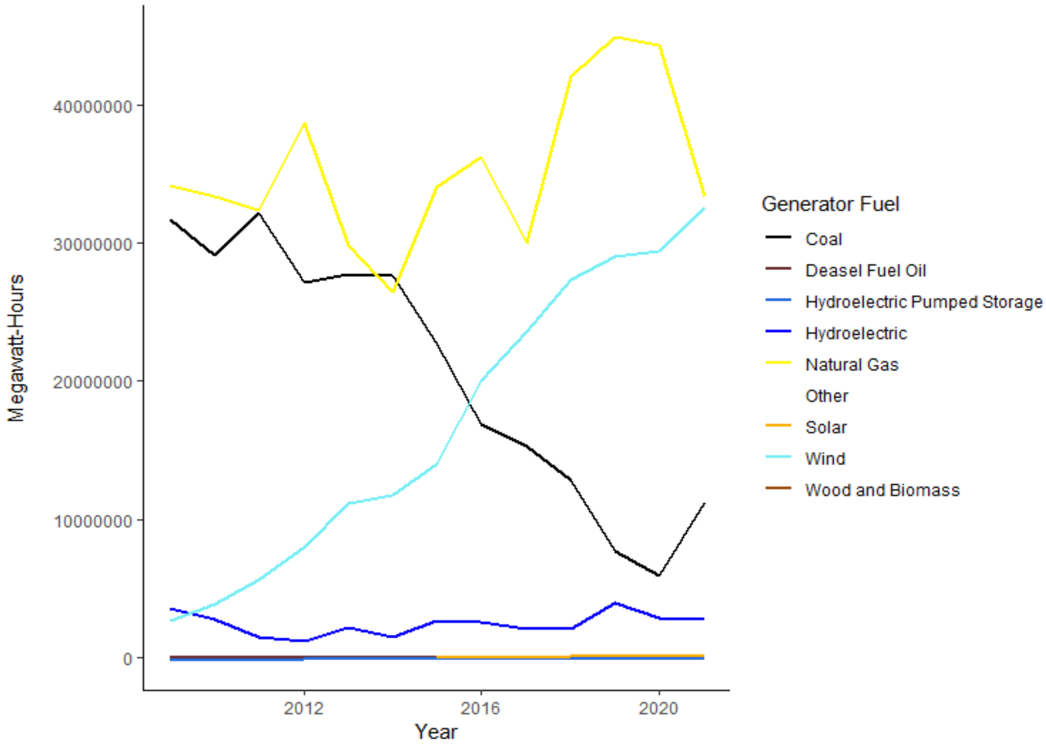

Figure 12. Generation by power source over time in Oklahoma

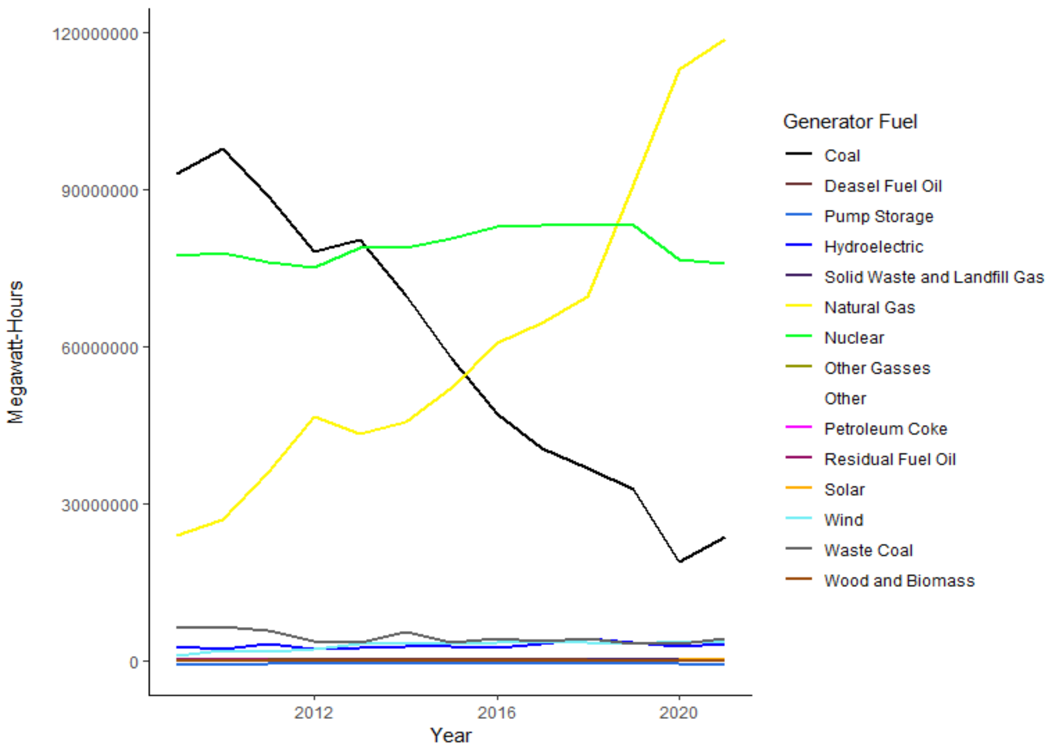

Figure 13. Generation by power source over time in Pennsylvania

Unlike renewable portfolio standards, which, as a command-and-control policy, specifies how decarbonization should be achieved, carbon markets simply impose a penalty for all emissions, and thereby incentivize utilities to reduce their emissions in the way they deem most effective. In states with poor renewable energy potential with existing technologies, this means replacing coal with natural gas facilities, which can be scaled up at a lower cost than renewables and abate more carbon. In states where renewable resources are most suitable, existing coal generation is being replaced regardless of policy, so in these locations any policy which incentivizes carbon reduction would likely also lead to more rapid renewables investment.

Ultimately, factors such as wind speed and consistency, solar exposure, and natural gas prices, dictate which energy sources should be used to replace more polluting sources of generation. While market-based policies which simply incentivize emissions reductions allow local actors to select the most cost-effective way to reduce emissions, command-and-control policies must be carefully constructed to suit local conditions in order to have the maximum decarbonization effect. A context-sensitive approach is needed when considering whether and how to adapt policies successful in other regions to a new location with different economic, climactic, and social factors.

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