Abstract

This dissertation consists of three essays related to environmental justice. The first essay examines the impact of decarbonization of the US electric grid on air quality and assesses how the health benefits of better air quality will be distributed among people of different ages and races. This work was done for the contiguous US at the county level. These benefits are estimated through three regulatory-grade models: Integrated Planning Model (IPM), Community Multiscale Air Quality Modeling System (CMAQ), and Environmental Benefits Mapping and Analysis Program (BenMAP). Air quality improvements and health gains (premature deaths avoided) are reported for the years 2030, 2040, and 2050. Most of the $PM_{2.5}$ and O_3 reductions are concentrated in the Eastern US. Black communities experience the largest improvement in air quality compared to all other races. For health benefits, I find that Whites have the largest benefits in terms of absolute numbers, but when appropriate race-specific mortality incidence rates are used and population-weighted race-age decomposition is conducted, Blacks have 20% larger gains compared to Whites in age group 25-74. Moreover, when premature deaths averted are converted to life years, I find that disparity in health benefits between age groups is sharply reduced, shifting 2.86 percentage points of the total gains from Whites to Blacks. Age-race decomposition analysis for decarbonization of US electric grid thus suggests improvement in environmental justice. The finding from this paper can help policymakers understand how health disparities are reduced with respect to age and race due to decarbonization.

In the second essay I examine how improved air quality due to the decarbonization of the US power sector can reduce asthma exacerbation among children disaggregated by poverty status, race, and geography. These benefits are estimated through three regulatory-grade models: Integrated Planning Model (IPM), Community Multiscale Air Quality Modeling System (CMAQ),

and Environmental Benefits Mapping and Analysis Program (BenMAP). Using spatial datasets that differentiate asthma prevalence by income, race, and state, I find that children in living in households with income below the poverty line receive a disproportionate share of the benefits. Within each racial group, households with child poverty have 50% larger reductions in asthma exacerbations than households without childhood poverty. Furthermore, Black people, both above and below the poverty line, have larger health gains than all other races and income groups. I also provide general methodological insights for quantifying the environmental justice impacts of regulatory policies.

Environmental collaboration has become an increasingly common approach to the management of natural resources. Scholars and practitioners have tried to understand how collaborative structures impact performance using a multitude of single case studies and comparative studies. However, despite calls for the evaluation of collaborative performance, minimal quantitative research explores the connections between collaborative structures and performance using a large sample for analysis. I address this gap in my third essay by carrying out a fixed effects analysis that is used to examine the impact of several structural variations, including collaboration form, number and representational diversity of participants, and contributions of in-kind resources, on the cost-effectiveness of collaborative watershed projects in Oregon. The data for this project come from the Oregon Watershed Restoration Inventory (OWRI). My results indicate that collaboration form, participant numbers, and resource contributions affect cost-effectiveness, but representational diversity among participants does not. The findings from this article can help sponsoring and implementing agencies execute collaborative projects more cost-effectively. They also indicate the need for additional research exploring the relationship between collaborative structures, outputs, and outcomes.

Three Essays on Environmental Justice

by

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For my beloved family, intellectual and spiritual gurus, sheer coincidences, and BDD

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Chapter 1: Introduction

Environmental justice (EJ) is an important concern for regulatory policies in the US. This dissertation has three essays on EJ and its roles and ramifications in energy transition and environmental management. For this thesis, I rely on the definition of EJ used by the US Environmental Protection Agency (EPA). The EPA defines EJ as the “fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income, with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies.” As such, this dissertation contains two essays on how regulatory reforms of the power sector emissions are fair in the sense of distributing health benefits across the entire US population. The third essay leverages a unique dataset from Oregon and predicts how the meaningful involvement of diverse stakeholders led to the cost-effective implementation of watershed management projects to protect salmon. Each of these three essays are previewed in more detail next.

In this dissertation's first essay (Chapter 2) I examine the impact of decarbonizing the US electric grid on air quality. I assess how the premature deaths averted due to better air quality will be distributed among people of different ages and races. This work was done for the contiguous US at the county level. A clean energy standard scenario is used to model the changes in emissions and generation. These emissions patterns are used to model changes in ambient air quality for $PM_{2.5}$ and ozone. Air quality improvements and health gains (premature deaths averted) are reported for 2030, 2040, and 2050. Significant health benefits are observed for all races and age groups. Over 30 years, from 2020-2050, almost 315,000 premature deaths would be averted. Moreover, the findings of this chapter regarding resident health have EJ implications.

Decarbonization of the electric grid results in greater premature deaths averted per 100,000 for Blacks compared to Whites in the age group 25-74.

These results highlight how failing to use race-specific mortality rates can significantly underestimate health benefits for Black communities, a critical issue in existing literature. In several prior studies health benefits are analyzed in terms of premature deaths averted by reducing air pollution from decarbonizing the power sector in different parts of the US (Driscoll et al. 2015; Luo et al. 2022; Millstein et al. 2018; Penn et al. 2017). However, very few studies conduct age-race decomposition, using race-specific incidence rates at a county-level resolution (Luo et al. 2022).

In the second essay I construct a unique asthma prevalence dataset by poverty status, race, and state from CDC's *Behavioral Risk Factor Surveillance System* (*BRFSS*) surveys from 2008-2021. These prevalence rates are used to model the reductions in asthma exacerbations due to improvements in air quality from decarbonizing the US power sector. Compared to the first essay, the second essay offers a methodological extension as it differentiates the prevalence and population by poverty status. This analytical advancement is important because asthma has historically been much more prevalent in poor and Black children (Pate et al. 2021). I find substantial health benefits for children in households with income below the poverty line, especially Black children in the Eastern half of the US. Nationally I find that 190,317 asthma exacerbation cases are averted in 2040 for households with children living above the poverty line and 45,175 amongst households with child poverty.

In the final essay in this dissertation I explore EJ along a distinctly different but nevertheless immensely important dimension of policymaking and policy implementation. Rather than continuing to focus on the EJ considerations of those affected by public problems and their relevant policies, in this third chapter I apply an EJ lens to investigate how diversity in those collaborating to address public problems affects the outcomes of these collaborations. This analysis relies on a unique data repository of environmental projects managed by the Oregon Watershed Enhancement Board (OWEB). In the third essay I study these projects to understand how collaborative structures encourage diverse participation and affect the performance of environmental restoration projects in Oregon watersheds. I examine the impact of several structural variations, including collaboration form, number and representational diversity of participants, and contributions of in-kind resources, on the cost-effectiveness of collaborative watershed projects in Oregon. My results indicate that collaborative form, participant numbers, and resource contributions affect cost-effectiveness. Moreover, the shape and structure of collaborative arrangements—collaborative form, size, and contributions—affect the cost-effectiveness of projects. The results suggest that Collaborative Governance Regimes (CGR) are more cost-effective for larger, longer, and more complex projects. At the same time, *ad hoc* collaborations are more cost-effective for smaller, shorter, and simpler projects.

Thus, while in two of the three essays I evaluate health impacts and offer methodological insights on how health outcomes can be better measured for EJ, in the third essay I emphasize the importance of collaboration to reach cost-effective solutions that the communities perceive as legitimate and sustainable. Together, these essays offer insights for the policymakers on how some aspects of EJ can be better measured and achieved concerning “fair treatment and meaningful involvement” of disadvantaged communities. More than just filling gaps in the literature, in this dissertation I provide a profound reflection on the health and economic effects of regulatory policies on populations that have been historically disadvantaged, such as poor and Black communities that bear a larger share of air pollution or economically disadvantaged Native American tribes and rural communities in Oregon.

Chapter 2: How Much Environmental Justice is Achieved if the US Decarbonizes Its Electric Grid

Introduction

Urban centers in the US often have a disproportionately higher share of panethnic populations and poor air (Lichter, 2018; Martin, 2017; Stronsnider et al. 2017). These metropolitan cities tend to have far worse air quality due to their proximity to O_3 and $PM_{2.5}$ emission sources (Stronsnider et al. 2017). This pattern leads to a disproportionate burden on ethnic minorities and people of color (PoC). This disproportionate share of negative environmental consequences falling on PoC raises concerns about environmental justice (EJ). Using EPA's definition, I in this paper define EJ as "the fair treatment and meaningful involvement of all individuals regardless of their race, ethnicity, socio-economic status, or country of origin, with respect to the development, implementation and enforcement of environmental laws, regulations, and policies." (EPA 2021).

In this paper I focus on one source of emissions in particular: the electricity generation sector. In the past few decades, the US has raised the share of renewable energy sources in the generation mix and implemented stringent air quality protocols like the Acid Rain Program (ARP) under the Clean Air Act (CAA). Compared to 1995, in 2021 electricity generation has increased, while the emissions of pollutants like sulfur dioxide (SO_2), nitrogen oxides (NO_x), and carbon dioxide (CO_2) have decreased substantially (EPA 2021). These achievements are certainly worthy of celebration, but electricity generation is still causing pollution. This higher exposure to air pollution can lead to asthma, chronic obstructive pulmonary disease, lung cancer, and cardiovascular disease. In addition to PoC, other populations such as people with low socioeconomic status (SES), children and adults over the age of 65 are also at risk.

While the need to decarbonize the US electric grid is recognized, the exact policy pathway and its environmental justice (EJ) implications are not fully understood. In this paper I conduct a race-age decomposition analysis using air quality data from a clean energy standard policy scenario with net-zero carbon emissions by 2040, and model the health outcomes by age and race for 2030, 2040, and 2050. Particularly, to model premature deaths averted and life-years saved due to decarbonization I employ race and age specific mortality rates. In the US, mortality rates are higher for the younger Black population compared to the younger White population. Due to this, mortality improvements are larger for young Black populations than for young White populations and the results show how using age and race specific mortality rates is important for accurate understanding of EJ impacts.

Background and Literature Review

In 1982, the predominantly Black community in Warren County, North Carolina, led a non-violent protest of the disposal of soil laced with the toxin polychlorinated biphenyl (PCB) in their community. The protest failed to stop the dumping of toxic waste but has now become iconic as “the first major milestone in the national movement for environmental justice” (Skelton and Miller 2016). The protest drew national media attention and revealed a pattern: pollution-emitting sources and dumping sites were often situated near economically disadvantaged communities of color (Bullard 1994; Skelton and Miller 2016). Almost a decade of social mobilization and lobbying resulted in President Clinton issuing Executive Order 12898 (1994), which laid the legal foundation for all federal agencies to develop EJ strategies to address the disproportionately high rate of adverse health effects of pollution on PoC and low SES communities. This section will discuss relevant literature that has tried to measure the burden of emissions on EJ from the power sector.

Though decades have passed, many pollution-emitting sites still operate near PoC and economically disadvantaged communities. Across the US, PoC and low SES populations have a higher likelihood of living near industries such as coal-fired power plants (Collins et al. 2016; Hii et al. 2021; Schulz et al. 2016). Emissions from these plants contribute to negative health outcomes (Collins et al. 2011: Martenies et al. 2019; McDaniel, Paxson and Waldfogel 2006; O'Neill 2003; Rice et al. 2014; Rhee et al. 2019), and are disproportionately distributed by SES and ethno-racial dimensions (Ash et al. 2013; Ash and Boyce 2018; Cushing et al. 2015, Knight et al. 2017; Miranda et al. 2011). In addition to the disproportionately higher pollution exposure, PoC and low SES communities often have inadequate access to health infrastructure and other resources to negate the effects of higher pollution exposure (Thind et al. 2019; Schulz & Northridge 2004).

Since the inception of the EJ movement, regulations and environmental policies have been introduced in nearly every state to reduce the emissions exposure for all ethno-racial groups and SES communities. The most iconic of these federal regulations is the SO_2 *allowance-trading program* established under Title IV of the 1990 Clean Air Act Amendments which led to a great deal of improvement in air quality across the US (Chen et al. 2012; Schmalensee and Stavins 2013). This was arguably the world's first large-scale cap-and-trade system for pollution. Despite a 26% increase in coal generation from 1990-2007, thanks to the program SO2 emissions decreased 43% (Stavins 2012). This improvement in air quality has been linked to improvement in health outcomes for numerous communities (Casey et al. 2020; Ostro 1999; Chestnut 2005; Gauderman 2015).

In addition to these policy changes, changes in supply have resulted in the electricity sector shifting to less polluting sources of energy. Due to fracking, natural gas prices have decreased substantially. This technology has led to natural gas increasing from 24% of the US energy mix in 2011 to 35% in 2021 (EIA 2021). This increase in natural gas has led to a corresponding decline in total coal generation as part of the US energy mix from 44% in 2011 to 25% in 2021. Additionally, the development of more advanced, efficient, and affordable renewable energy generation sources has led its usage to increase from 4% in 2011 to 14% in 2021. These improvements certainly spell a promising future for EJ issues, but there are still substantial disparities in emission exposure and health outcomes between White people and PoC.

Electricity generation can produce a variety of emissions that contribute to poor air quality including NO_x, SO_x, and volatile organic compounds (VOCs). These can undergo chemical reactions in the atmosphere to produce fine particulate matter with a diameter of less than 2.5 micrometers ($PM_{2.5}$), which is particularly dangerous for human health (Hodan et al, 2004). Additionally, reactions between NO_x and VOCs in the troposphere produce ground-level ozone (O_3), which is an extremely harmful pollutant when inhaled (Sillman and He, 2002). Reducing power-sector emissions would decrease ground-level O_3 and $PM_{2.5}$, leading to a decline in premature deaths, heart attacks, asthma, and hospitalization (Driscoll et al. forthcoming; Thind et al. 2019). Additionally, there are environmental benefits to reducing O_3 and $PM_{2.5}$ levels, such as reductions in acid rain and eutrophication, improvements in tree and crop productivity, and improvements in visibility.

The number of premature deaths in 2005 in the US due to EGU emissions has been estimated to be between 38,000 and 52,000 (Fann et al. 2013, Dedoussi et al. 2014, Caiazzo et al. 2013). Over time the number has generally fallen as air quality has improved: 17,050 in 2010 (Leliveld et al. 2015), 10,400 in 2014 (Tessum et al. 2019), and 17,000 in 2016 (Fann et al. 2013). Penn et al. (2017) find 21,000 premature mortalities per year in 2005 from EGU emissions for both O_3 and $PM_{2.5}$. Variation in the number of estimated deaths, as seen in the estimates from 2005, is due to differences in models and methods. But in general mortality is falling over time, corresponding with decreases in emissions.

Public health scholars have extensively researched the impacts of power plant emissions on health outcomes. For example, Buonocore et al. (2014) estimated the monetized health benefits of reducing primary emissions from individual power plants in Mid-Atlantic and Lower Great Lakes regions of United States to be $130,000 per ton of $PM_{2.5}$, $28,000 per ton SO_2 and $16,000 per ton of NO_x. Levy et al. (2009) estimated the health benefits from reducing emissions from coal-fired power plants to be $72,000 per ton of $PM_{2.5}$, $19,000 per ton of SO_2 and $4,800 per ton of NO_x. Fann et al. estimated the benefits to be $100,000 per ton of $PM_{2.5}$, $27,000 per ton of SO_2 and $3,800 per ton of NO_x. These estimates vary due to geographical scale of the analysis, the usage of different concentration response functions, the power plants included, and other parameters that influence the health benefits.

The burden of air pollution differs by demographic group (Ash et al. 2013; Ash and Boyce 2018; Cushing et al. 2015, Knight et al. 2017; Miranda et al. 2011; Spiller et al. 2021). Particularly for the power sector, Thind et al. (2019) estimate that, across the population of the US, there are 5.3 premature deaths due to $PM_{2.5}$ exposure per 100,000 people. However, the rate is not uniform across demographic groups. They found that the premature death rates were 6.6 per 100,000 for Black people, 5.9 for White, non-Latino people, and 3.6 averaged across the remaining racial and ethnic groups. Additionally, Tessum et al. (2019) find that while PoC have higher exposure to air pollution, they consume less energy compared to the US average, and as such, they are less responsible for the generation of these emissions.

In this paper I use a high emission-reduction policy scenario, an aggressive Clean Energy Standard (CES), to determine the health benefits of decarbonization. A CES mandates a certain percentage of energy generation through “clean” sources, such as renewable energy sources like solar and wind power, which produce few pollutants. Under a CES, even fossil fuels like natural gas and coal, when combined with carbon capture and storage (CCS), can be defined as clean if they meet certain emission factor criteria established in legislation. Here, the emission factor is the quantity of pollutants released into the atmosphere by burning a unit of fossil fuel in a power plant. Some states already have a CES in place. For example, the New York Public Service Commission implemented a CES in 2016 which supports existing nuclear facilities as a bridge toward making the grid 50% renewable by 2030. As a CES classifies some non-renewable sources as clean in the generation mix (like nuclear and gas with CCS), it provides more flexibility and lower costs than a strict renewable portfolio standard (RPS), which requires a certain percentage of energy be produced solely from renewable sources. Coffman et al. (2012) found that in Hawaii, adopting a CES compared to an RPS would decrease the costs of achieving a given emissions target by 90%. Much like an RPS, in a CES, the standard "percentage" of the amount of energy produced based on renewable energy ramps up over time. Utilizing the Haiku electricity market model, Paul et al. (2011) determined that increasing the clean standard target from an initial 12.3% to 57% by 2035 would decrease carbon emissions by 30 percent.

It is critical to understand how these policies which achieve complete decarbonization will affect health across the population. Considerable interest has been shown in a high-ambition hybrid CES policy called "CES40B". This scenario CES40B has the following characteristics: (a) a target of 100% clean energy generation by 2040, (b) use of banked credits being allowed until 2050, (c) the initial carbon intensity benchmark being set at 0.82 tons MWh^{-1} , and (d) natural gas partial crediting being allowed until 2040, based on the emission rate of each plant. Under the above provisions, the policy attains 80% clean energy by the year 2030.

In this paper I determine the health outcomes, broken down by age and race, that would result from changes in $PM_{2.5}$ and O_3 exposure in 2030, 2040, and 2050 under CES40B. After decomposing by age and race, I find that the largest gains in air quality improvement and health benefits are for Black people between the ages of 25 and 74, followed by White people, particularly over the age of 75. Unlike previous scholarship which has averaged across age and race to determine county-level mortality rates, this paper is unique as it breaks down mortality rates within each county by age and race. The results reveal a larger disparity in health outcomes between White people and PoC. In this paper I show that stringent decarbonization strategies that aim for zero carbon emissions by 2040 will improve the health outcomes of the entire nation and would specifically benefit the communities currently bearing a disproportionate air pollution burden.

Models & Methods

To model the health benefits of decarbonizing, I use three existing regulatory grade models in my analysis. In this section I will describe these models, datasets, and vital assumptions for the analysis.

Air pollution impacts were taken from two scenarios in the Clean Energy Futures (CEF) database[1]: the CES40B policy scenario and the 'Business as Usual' (BAU) policy scenario. The BAU policy scenario is based on a run developed in 2020 by the Natural Resources Defense Council (NRDC) and it assumes no new regulations in the power sector. BAU (base case) and CES40B (decarbonizing case) are described in detail in the CEF project documentation (Driscoll et al. 2021)[2]. The CES40B scenario assumes a CES policy is in place and a goal of net-zero carbon emissions by 2040. The CEF project forecasts changes in electricity generation by source –

[1] For more information visit: https://cleanenergyfutures.syr.edu/
[2] Total generation is held constant in BAU and CES40B. Some key CES40B assumptions are: (a) a target of 100% clean energy generation by 2040, (b) use of banked credits being allowed until 2050, (c) the initial carbon intensity benchmark being set at 0.82 tons MWh^{-1}, and (d) natural gas partial crediting being allowed until 2040, based on the emission rate of each plant. Under the above provisions, the policy attains 80% clean energy by the year 2030.

nuclear, wind, and coal – in 2030, 2040, and 2050 using the Integrated Planning Model (IPM). This model has been extensively used by the Environmental Protection Agency (EPA) to analyze the effects of environmental regulations on the electricity generation sector. IPM is an engineering-economic dynamic linear programming model that projects various outcomes at the plant level. Some of the outcomes modeled are: electricity generation; levels of CO_2, NO_x, SO_2, and mercury (Hg); fuel costs; capital costs; operations and maintenance costs; and investments in new and old generation facilities. The model attempts to minimize overall system costs when determining generation patterns or when adding capacity to the electricity grid. EPA's 2011 National Emission Inventory (NEI) is used as the baseline emissions data in the model.

Emission outputs from IPM were used to estimate air quality using the Community Multiscale Air Quality (CMAQ) model. CMAQ is an open-source, dynamic air quality model developed by EPA that combines atmospheric science and multiple air quality models to predict the concentrations of ozone and $PM_{2.5}$ (Byun and Schere 2006; USEPA 2014). For this work, CMAQ version 5.0.2 and Appel et al.'s (2011) physical and chemical schemes for air quality across the contiguous US were utilized, and climate change projections were held constant. Stack heights for the fossil-fuel based EGUs are maintained in CMAQ. Due to high stack heights, pollutants are dispersed over larger areas downwind. CMAQ outputs air quality results for a 36 km x 36 km raster grid for the contiguous US. While I discuss the modeled outputs briefly here, for greater details on these runs and CMAQ modeling please refer to Vasilakos et al. (2022).

Due to high stack heights, CMAQ models the air quality changes due to decommissioning fossil fuel at a regional level. As such, the health benefits modeled are not only for the communities living close to the EGUs but also communities down wind. The benefits in ambient air quality can be observed across state borders. The health benefits are for everyone, but they are larger for those close to the EGUs, than those further downwind.

The results of the CMAQ runs were entered into BenMAP CE v.1.5 (US EPA 2015), a Geographic Information System (GIS) model designed to calculate the health impacts of air pollution. I used BenMAP to estimate mortality and morbidity risks based on the difference in air quality between the CES40B scenario and the BAU scenario. For mortality estimates due to changes in O_3 I used a concentration-response function (CRF) for adults 25 years and older from Turner et al (2016) based on the seasonal average of the 8-hour daily maximum O_3 concentration. For mortality estimates due to changes in the $PM_{2.5}$ I used a CRF for adults 25 years and older from Vodonos et al. (2018) based on the annual average of 24-hour average $PM_{2.5}$ concentrations. Population projections for 2030, 2040, and 2050 were taken from Woods and Poole Economic, Inc, as provided in BenMAP CE v 1.5. Historical mortality rates by age and race (White people and PoC), which were used to calculate baseline risk in this analysis, were taken from the CDC Wonder database for the years 2006 through 2017. CDC does not report mortality for counties by race-age classification for counties with less than 10 deaths per 100,000. For these suppressed data, mortality rate cannot be calculated. To compute the mortality rates I rely on EPA's methodology highlighted in the BenMap manual. EPA computes the mortality rates for suppressed counties through the following calculation:

$$R_{s,i,r} = \frac{D_{s,i,r} - Du_{s,i,r}}{P_{s,i,r}}$$

Here $R_{s,i,r}$ is the state average suppressed mortality rate for state s, age group i and race r. r classifies two race groups: Whites and non-Whites. Many counties and even states have very small Asian, Black, and Native populations, and due to this they do not report deaths for these groups. Therefore, in BenMAP EPA aggregates all non-White races into a group. $D_{s,i,r}$ is the total state s death count for age group i and race r. $Du_{s,i,r}$ is the aggregated total state s unsuppressed death count for age group i and race r in all the counties. And $P_{s,i,r}$ is the aggregated total state s population for age group i and race r in all its counties. For the counties with suppressed data, EPA set it equal to the state specific mortality rate $R_{s,i,r}$. I adjusted the incidence data with the projected change in mortality for each age group in 2040 as provided in BenMAP manual (2023). EPA uses the Census Bureau's projected life tables to create these adjustments ratios.

Several studies have used similar approaches to estimate premature mortality and health benefits under decarbonization policies (Driscoll et al. 2015; Millstein et al. 2018; Penn et al. 2017; Levy et a. 2001); however, they generally do not analyze by race and age, and do not break down mortality rates accordingly. Spillers et al. (2021) do attempt a race decomposition analysis of mortality due to $PM_{2.5}$ pollution. They employ unique CRFs for each race, but their research focuses only on people aged 65 and older. While it would be useful to have CRFs for every race and age, to my knowledge these do not yet exist in the literature for $PM_{2.5}$ and O_3. As a result, for this analysis, I determine changes in mortality rates by age for two broad groups: White people and PoC as a whole.

The health impact function (HIF) used to calculate the number of premature deaths averted D for both $PM_{2.5}$ and O_3 is given below, where i indicates the county r indicates a demographic race (Blacks or Whites), and a represent demographic age group.

$$D_{i,r,a} = \left(1 - \frac{1}{exp\,(\beta *\ \delta Q_i)}\right) . \mu_{i,r,a} . \rho_{i,r,a}$$

In this equation, β is a constant that is derived from epidemiological studies. For Turner et al.'s (2016) O_3 study it is 0.002 and for Vodonos et al.'s (2018) $PM_{2.5}$ study it is 0.0129. δQ_i is the difference in air quality between the CES40B scenario and the BAU scenario for each county. μ_{ir} is the mortality incidence or baseline risk in county i for age-race group a and r (estimated from CDC data), and ρ_{ira} is county i's population of race r in age group a. In this equation age group a has following age groups: 25-34, 44-45, 45-54, 55-64, 65-74, 75-84, and 85-99. The race groups are Blacks, and Whites. This approximation is carried out for both the pollutants $PM_{2.5}$ and O_3 by year.

National deaths averted by race are estimated by this calculation:

$$D_r = \sum_i \sum_a D_{i,r,a}$$

Health benefits are also computed in life-years saved *Y* to investigate how the disparity between age and race group affects. For this I use the following calculation:

$$Y_{i,r,a} = D_{i,r,a}.(Exp_a - Mid_a)$$

In this equation, premature deaths avoided are national aggregate by race-age groups *r*. For more precise estimation race-age groups *r* and *a* is in 10-year intervals (25-34, 35-44, 45-54, 55-64, 65-74, 75-84, and 85-99) and I assume that each of these age groups has a mid-point mean age Mid_a. For example, for the group aged 25-34 the midpoint is 30 years. *Exp* is the life expectancy by age group. These were obtained from US Census Bureau's projection in 2040. For environmental justice ethical considerations, I assume a similar life expectancy by race.

Life years saved for race *r* are estimated through the following calculation:

$$Y_r = \sum_i \sum_a Y_{i,r,a}$$

The population of people of race *r*, P_r, is obtained through the following calculation:

$$P_r = \sum_i \sum_a P_{i,r,a}$$

Deaths averted nationally per 100,000 people of race is estimated by:

$$\hat{D}_r = 1e5 \cdot \frac{D_r}{P_r}$$

Life-years saved nationally per 100,000 people of race is estimated by:

$$\hat{y}_r = 1e5 \cdot \frac{Y_r}{P_r}$$

Results

Here I will present the results for the three regulatory grade models. I will briefly discuss the IPM and CMAQ outputs as Driscoll et al. (2022) and Vasilakos et al. (2022) have analyzed them in much detail. Furthermore, the main contribution of this paper is modeling the health benefits using age-race decomposition at county level. Therefore more attention is given to the health benefit analysis.

Predictions from IPM suggest that under CES40B coal and gas generation decrease over time. The changes in electricity generation with respect to technology are shown in area plots in Figure 1. Particularly, coal generation is removed from the electric grid by 2030. There is some gas generation that lingers, but it is largely replaced by gas with carbon capture and sequestration. Under CES40B, there is large expansion in solar and wind generation over time. Hydro and nuclear generation facilities remain mostly unchanged. CO_2, Hg, NO_x, SO_2, and direct PM emissions in 2040 under CES40B compared to BAU are shown in Figure 2. All the pollutants experience drastic reductions in emissions due to decarbonization of the US electric grid. These changes in emissions were used to determine the change in ambient air quality with respect to O_3 and $PM_{2.5}$ via CMAQ simulations.

Broadly, CMAQ simulations suggest that $PM_{2.5}$ and O_3 concentrations decrease across the contiguous US in the CES40B scenario compared to the BAU scenario (Figure 3 and Figure 4). $PM_{2.5}$ concentrations decrease up to 15%, and O_3 concentrations decrease up to 5%. Most of the $PM_{2.5}$ reductions were concentrated in Midwest, West Virginia, and Texas. O_3 reductions were more widespread and covered parts of the Eastern Seaboard, Midwest, and Texas. For both pollutants, the largest reductions are observed in the Eastern US. $PM_{2.5}$ and O_3 levels remained largely unchanged in the Western US under CES40B and BAU scenarios. However, this pattern is not surprising because there are substantially fewer fossil fuel powered EGUs (especially coal plants) in the Western US than in the Eastern US.

To summarize changes in exposure to $PM_{2.5}$ and O_3 by race I constructed the kernel density plots shown in Figure 5 and Figure 6. This analysis suggests that non-Hispanic Black people are concentrated in areas that experience relatively large reductions in pollution. The peak density for Black people is around values of 0.35 $\mu g/m^3$ for $PM_{2.5}$ and 1 ppb for O_3. Positive numbers represent an improvement in air quality: they are computed as the BAU concentration minus the CES40B concentration. In contrast, large shares of other populations live in communities that experience low reductions in pollution (to the left of each diagram) although a somewhat lower peak at roughly the same reduction in emissions is observed for White people. These results are comparable to those of Thind et. al (2019) and Goforth and Nock et al. (2022) who find that the largest beneficiaries of $PM_{2.5}$ reductions are Black people, followed by White people.

A general summary of average population-weighted reductions in $PM_{2.5}$ and O_3 with respect to race in 2030, 2040 and 2050 is shown in Figure 7. In both cases, the gains from CES40B relative to BAU are largest for non-Hispanic Black people and White people. In 2040 non-Hispanic Blacks have 0.25 μg/m^3 reductions in $PM_{2.5}$ and 0.69 ppb reduction in O_3. Non-Hispanic Whites in the same year have 0.22 μg/m^3 reductions in $PM_{2.5}$ and 0.60 ppb reduction in O_3.

As with $PM_{2.5}$ and O_3 concentrations, the number of premature deaths due to air pollution from electricity generation is broadly reduced across the contiguous US. In 2040 alone, the CES40B scenario results in more than 13,384 (±7,503) fewer premature deaths than the BAU scenario (Figure 8). Of these 11,882 (±6,662) are attributed to Whites, and 1,864 (±1,043) to Blacks. Most of the reductions in premature deaths are found along the Eastern Seaboard, in the Midwest, and in Texas. Metropolitan centers like Chicago, Detroit, Dallas, Pittsburgh, and Houston see the largest benefit (Figure 8). Approximately two-thirds of these premature deaths averted are attributed to reductions in $PM_{2.5}$ and approximately one-third are attributed to reductions in O_3. Cumulatively, premature deaths averted for both the pollutants from 2020 to 2050 is approximately 315,000 lives.

Using BenMAP, I found that the variable changes in $PM_{2.5}$ exposure levels for different races/ethnicities led to differences in the number of deaths averted for each group. The premature deaths per 100,000 averted by state and race in 2040 is shown in Figure 9. The redline in the scatter plot demarcates 1:1 between the two races. For both the races on a state-by-state level, Kentucky (KY), Arkansas (AR), Indiana (IN), Ohio (OH), and Tennessee (TN) stand out as the states with the largest premature deaths averted per 100,000 individuals.

However, for every state except the District of Columbia (DC) premature deaths averted per 100,000 are higher for White people than for Black people. On a surface level, this pattern suggests that Whites have overall larger numbers of premature deaths averted compared to the Black population. On a national aggregate, the average number of premature deaths averted per

100,000 people is 4.87 across all people. However, nationally an average of 5.31 premature deaths per 100,000 are averted for White people and 4.49 premature deaths per 100,000 are averted for Black people.

However, the age groups among each racial/ethnic group show different patterns in terms of premature deaths averted (Figure 10). Disaggregating premature deaths averted by age reveals that for age groups 25-34, 35-44, 45-54, 55-64, 65-74, 75-84 and 85-99, Black people tend to have the largest health benefits per capita across the age groups. As shown in Table 1, premature deaths averted in the CES40B scenario compared to the BAU scenario for Black people in comparison to White people are 23% higher in 25-34 age group, 23% higher in 35-44 age group and, 24% higher in 45-54 age group, 24% higher in 55-64 age group, 21% higher in 65-74 age group, 19% higher in 75-84 age group, and 8% higher in 85-99 age group.

Table 1: Number of premature deaths averted per 100,000 by race and age for CES40B compared with BAU.

		Age						
		25-34	35-44	45-54	55-64	65-74	75-84	85-99
Race	Blacks	0.28	0.45	1.11	2.71	6.60	18.09	46.20
	Whites	0.23	0.36	0.90	2.18	5.46	15.25	42.79

The fact that overall premature deaths averted are higher for Whites than for Blacks can be explained by the population distribution and age-specific mortality rates. As shown in Figure 11, older age groups skew White. Moreover, on average the older age groups have higher probability to die compared to younger age groups. Thus, the high numbers of premature deaths averted for the aging White population results in higher premature deaths averted when averaged across age groups. Therefore, it is important to examine mortality patterns both by race and by age.

Comparison of Results with and without White and PoC mortality baseline classification

In this section I examine the impact of accounting for race-specific baseline mortality rates. The benefits of decarbonizing the electric grid calculated using equal baseline and race-specific mortality rates is shown in Figure 12. Due to scale, the results may seem homogenous by age and race. However, there are some interesting results that need to be unpacked to understand how the relative risk for Black people changes when appropriate race-specific mortality rates are used.

The comparison of results with equal baseline mortality rates and race-specific mortality rates is shown in Table 2. The lives saved per 100,000 are compared by race and the two different mortality rates. In column 6 and column 7 the relative mortality risk of Blacks versus Whites is shown with and without race-specific mortality rates. These relative risks are calculated by dividing the Black population gains by White population gains for both equal baseline and race-specific mortality rates and show in Figure 13.

The relative risk of mortality between these two approaches is calculated in column 8, which is the difference between column 6 and column 7. When appropriate race-specific mortality baselines are used, compared to average mortality rates, I find that for Blacks: (i) within the 25-34 age group, the relative risk is larger by 0.17; (ii) within the 35-44 age group it is 0.19 larger; (iii) within 45-54 age group the ratio is 0.21 larger; and (iv) within 55-64 age group the ratio is 0.27 larger; (v) within the 65-74 age group it is 0.15 larger; (vi) within 75-84 age group the ratio is 0.02 smaller; and (vii) within 85-99 age group the ratio is 0.15 smaller. These results show that for the age groups from 25 to 74, using average baseline mortality rates produces results that underestimate the impact of air quality changes on Blacks and overestimate the impact on Whites.

For ages 75-84, the rate is largely unaffected using race-specific mortality, and for ages 85 to 99, results using average baseline mortality underestimate the impact on Whites and overestimate the impact on Blacks. Moreover, Figure 10 was developed with the relative risk values computed in Table 2. For the younger five age groups I find that the relative risk is about 20% higher for race-specific mortality compared to equal baseline mortality.

Table 2: Comparison of premature deaths averted per 100,000 with and without race-specific baseline mortality for CES40B relative to BAU

(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Age Range	**Premature deaths averted, equal baseline mortality (Blacks)**	**Premature deaths averted, race-specific baseline mortality (Blacks)**	**Premature deaths averted, equal baseline mortality (Whites)**	**Premature deaths averted, race-specific baseline mortality (Whites)**	**Relative Black risk, equal baseline mortality**	**Relative Black risk, race-specific baseline mortality**	**Change in relative risk**
25-34	0.28	0.31	0.23	0.22	1.23	1.40	0.17
35-44	0.45	0.51	0.36	0.36	1.23	1.42	0.19
45-54	1.11	1.27	0.90	0.88	1.24	1.45	0.21
55-64	2.71	3.17	2.18	2.10	1.24	1.51	0.27
65-74	6.60	7.41	5.46	5.47	1.21	1.36	0.15
75-84	18.09	18.24	15.25	15.60	1.19	1.17	-0.02
85-99	46.20	40.72	42.79	43.70	1.08	0.93	-0.15

Comparison of Results with Race-Age Decomposition by Life-Years Saved

Table 2 showed that older age groups, such as 65-74 year olds, have substantially larger mortality risk reductions compared to younger age groups, such as 35-44 year olds. However, these results do not consider that a premature death averted by a younger individual has significantly more life-years saved compared to an older individual. Viewing health benefits in form of life-years saved reduces this disparity substantially. To control for this disparity in life years saved, I conducted a more in-depth race-age decomposition for 2040 under CES40B.

The analysis suggests that in 2040 for Blacks 1,864 premature deaths are averted and 26,422 life-years are saved, and for Whites 11,882 premature deaths are averted and 134,698 life-years are saved. These results are shown in two panels in Figure 14, one each for the aggregated number of lives and life-years saved in 2040. The first panel in Figure 14 suggests that decarbonization leads to greater number of lives saved for Blacks compared to Whites in age groups 25-84. Due to scaling of vertical axis, which is driven by mortality late in life, it is difficult to see the difference between Black and White outcomes at younger ages. However, the second panel shows the larger relative gain for Blacks in terms of life years. This pattern suggests that the disparity between groups in life-years saved is smaller than in lives saved. This difference arises because the share of the population that is Black is larger in the younger age groups in Figure 11. By assigning a larger weight of health benefits in form of life-years saved on the younger groups, this approach highlights that health benefits for Black population are relativity larger compared to White population. The aggregated normalized life-years saved per 100,000 for both White and Black population are provided in Table 3.

Table 3: Comparison of life-years saved per 100,000 with race-specific baseline mortality rates of CES40B relative to BAU

(1)	(2)	(3)
Age Range	**Normalized Life-Years Saved per 100,000 for Blacks**	**Normalized Life-Years Saved per 100,000 for Whites**
25-34	16.65	11.91
35-44	22.48	15.86
45-54	44.03	30.31
55-64	81.36	53.92
65-74	129.00	95.16
75-84	189.78	162.34
85-99	222.95	239.26

To summarize why it is important to use life-years as an assessment measure instead of premature deaths averted, I find that nationally, under CES40B, the Black share of premature deaths averted is 13.56%. But when considering life-years saved, the share for the Black population is 16.40%. Thus, using life year gains shifts 2.86 percentage points of the total gains from Whites to Blacks.

Discussion and Conclusion

Decarbonization of the US electric grid has health benefits for all age groups and race. In this paper I find that health improvements attributed to air quality due to the decarbonization of the US electric grid are larger per capita for younger age groups and communities of color. On average Whites have the largest aggregated benefits in terms of absolute numbers, but when population-weighted age decomposition is conducted and race-based mortality rates are used, in per capita terms the disparity is much smaller and Blacks aged 25-74 actually gain more than

Whites. Thus, health benefits for non-White groups are often underestimated by standard methodology.

Other scholars have used reduced complexity models to predict changes in ambient $PM_{2.5}$ (Thind et al. 2019; Goforth and Nock 2022). In this paper we employ the more sophisticated chemical transport model, CMAQ, to predict changes in ambient air quality for both $PM_{2.5}$ and O_3. CMAQ finds comparable mean reductions of 0.2 $\mu g/m^3$ with respect to $PM_{2.5}$ due to decarbonization. Similarly, other researchers find that Blacks have the largest air quality and health gain compared to other race groups (Thind et al. 2019; Goforth and Nock 2022). A strength of this paper is that I use age and race-specific mortality rates to improve the accuracy of the age-race decomposition. However, unlike others, it does not decompose by poverty status because the CDC does not disaggregate the mortality incidence by income.

In this paper I find that two adjustments in the methodological approach should help avoid underestimating health benefits for younger groups and communities of color. Firstly, other things equal, race-specific mortality rates should be used in calculating health benefits. Compared to health impact functions (HIFs) with average population mortality rates, when correct race-based mortality rates are applied, Blacks between 25 and 74 years old have a positive change in relative risk of mortality compared to Whites. However, for the Blacks in age group 85-99, I find a negative change in relative risk of mortality. This highlights that using average population mortality rates will underestimate the impacts on some groups and overestimate them on others, and these errors can obscure impacts that are important for understanding environmental justice impacts.

Secondly, an equally important methodological insight from this paper is the significance of using life-years saved estimates. The Black share of the US population is 13.6% (Census 2022) and the Black share of premature deaths averted in 2040 is quite similar to this number at 13.56%.

However, when I modeled the health benefits in life-years, I find that this share for the Black population is 16.40%. This disproportionate increase in health benefits arises because the Black population of the US is relatively younger than the White population. Assigning a younger individual a larger weight via the number of life-years saved shows that decarbonization is more favorable in terms of EJ than it appears from the number of lives saved. In fact, I find that using life year gains shifts 2.86 percentage points of the total gains from Whites to Blacks. I find that standard methodology underestimates the impacts on Blacks and skews the apparent health gains towards the Whites and older people. Using appropriate methodology can correct these biases and give a more accurate picture of the EJ impacts of air quality policies.

Future research should examine the impact on these results of using: (1) alternate models and modeling approaches such as other Chemical Transport Models and Reduced-complexity models; (2) smaller grid resolution that would enable a fine-scale analysis; (3) alternate health CRFs for endpoints such as asthma and cancer, especially as it is well known that asthma affects children and teenagers disproportionately more than older adults; (4) updated emission inventories from more recent years; (5) including effects of increase in energy demand from projected electrification of US transportation fleet.

The results of this paper are useful for helping scientists and policymakers better understand how decarbonization can reduce disparities in air pollution exposure and premature deaths or other health outcomes by race, age, and geography. It is important to note that all race-age groups experience health gains, but these gains are skewed towards Black people due to historical socio-economic prejudices and environmental injustices that causes them to live in areas with relatively high levels of air pollution. Thus, reductions in EGU emissions of $PM_{2.5}$ and O_3 would not only save lives but also can reduce environmental and health inequalities.

Figures

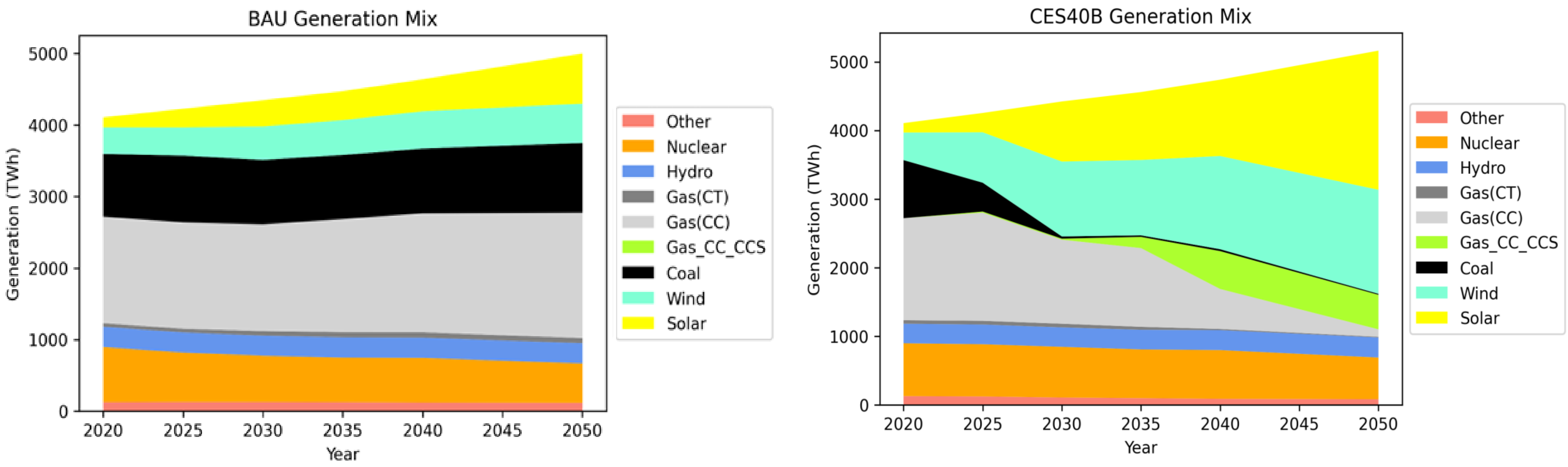

Figure 1: Electricity generation by source for BAU and CES40B

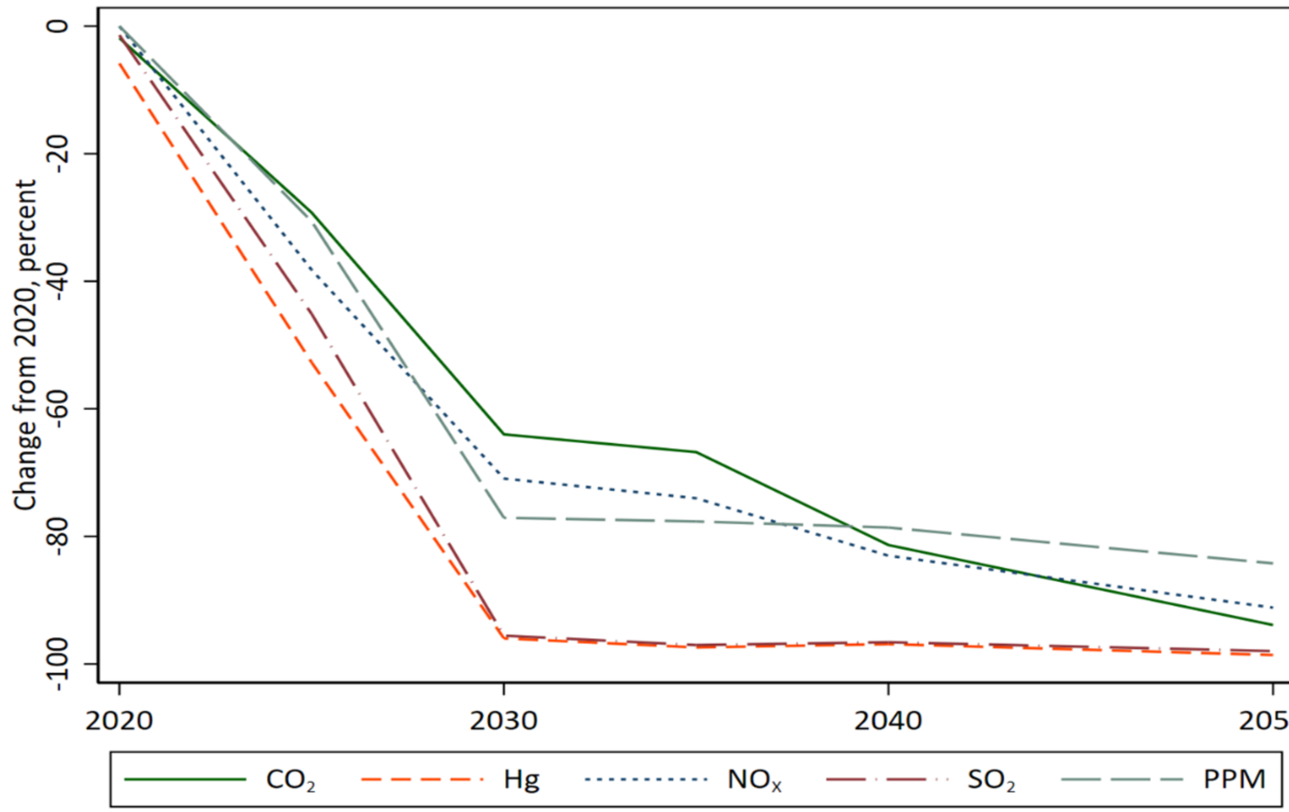

Figure 2: Projections of percent change in direct emissions of CO2, Hg, NOx, SO2, and direct PM from 202 to 2050 under CES40B compared to BAU

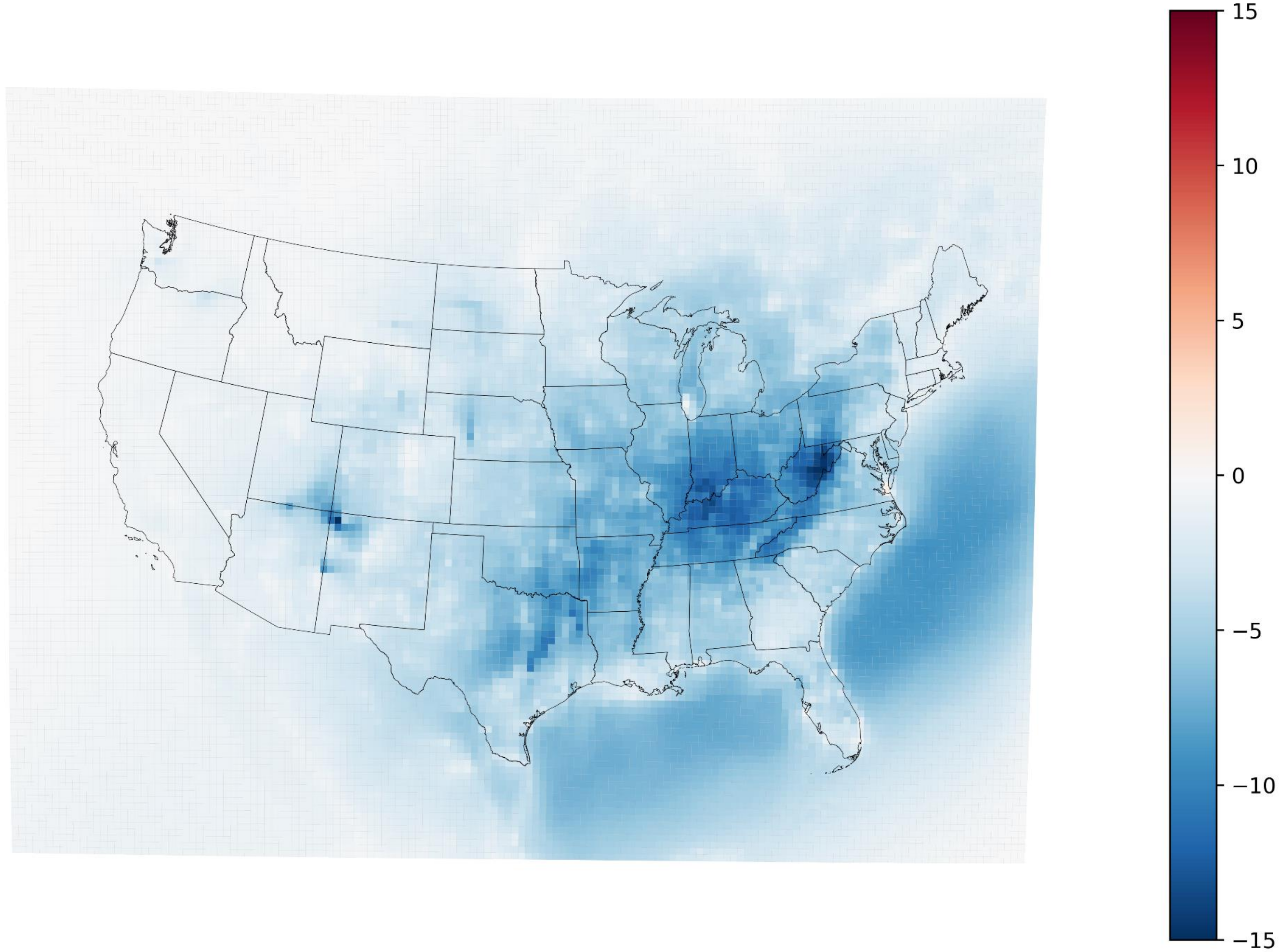

Figure 3: Projections of percentage change in fine particulate matter (PM2.5) concentrations in 2040 in the CES40B scenario compared to the BAU scenario for annual average 24-hour concentrations, in micrograms per cubic meter (µg/m3).

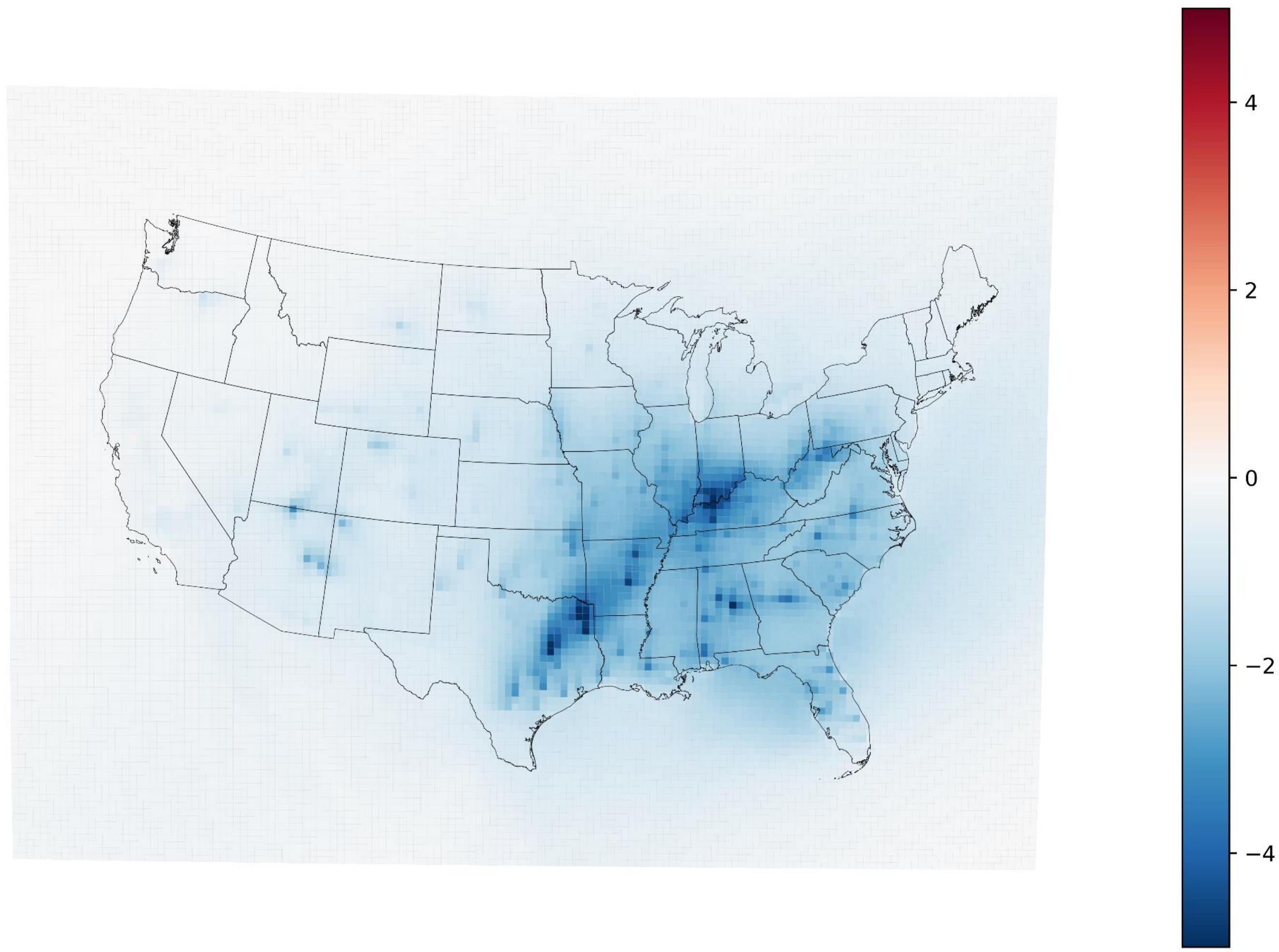

Figure 4: Projections of percentage change in ozone (O3) concentrations in 2040 in the CES40B scenario compared to BAU scenario for seasonal average maximum 8-hr concentrations in parts per billion (ppb).

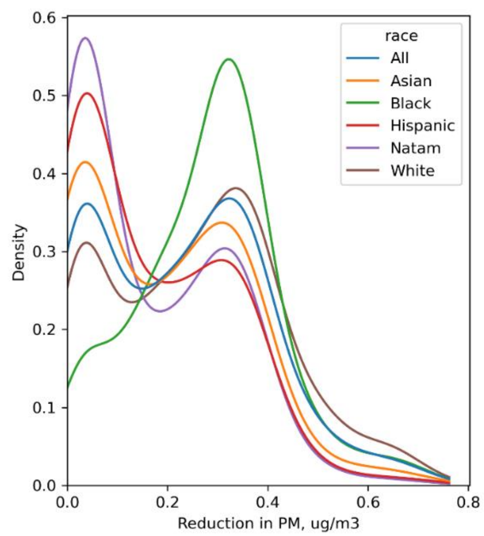

Figure 5: Fraction of population experiencing different degrees of $PM_{2.5}$ reduction in air pollution by race under CES40B relative to BAU for 2040

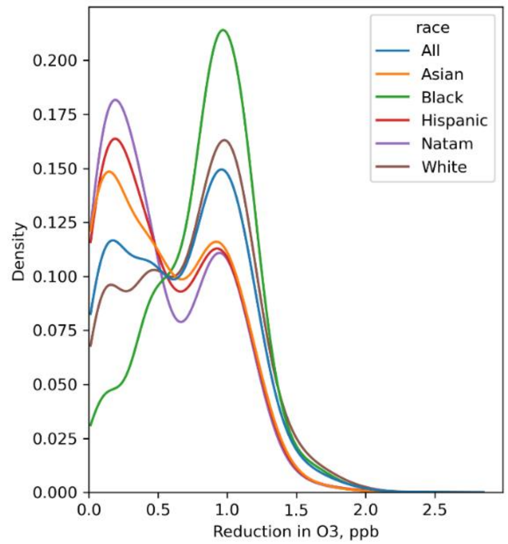

Figure 6: Fraction of population experiencing different degrees of O_3 reduction in air pollution by race

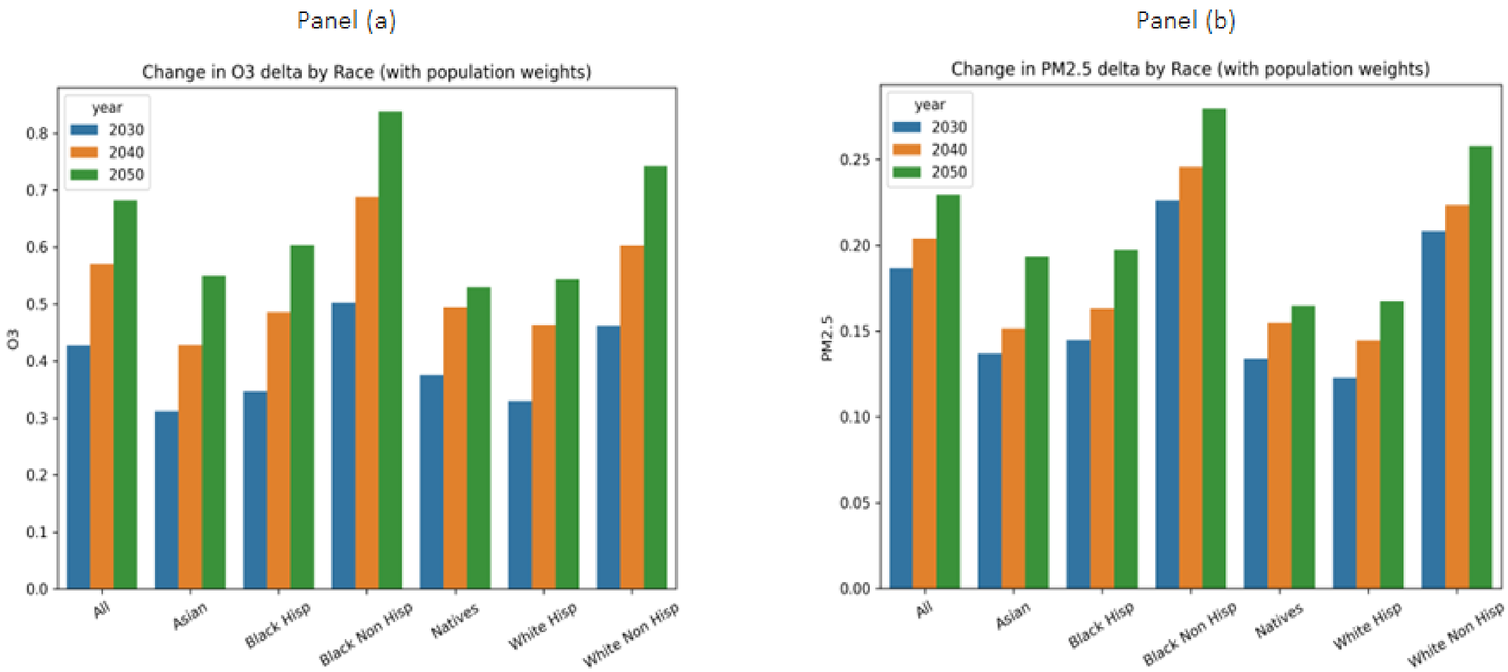

Figure 7: Projections of average reduction in O_3 (a) and $PM_{2.5}$ (b) under CES40B compared to BAU for 2030, 2040 and 2050, by race

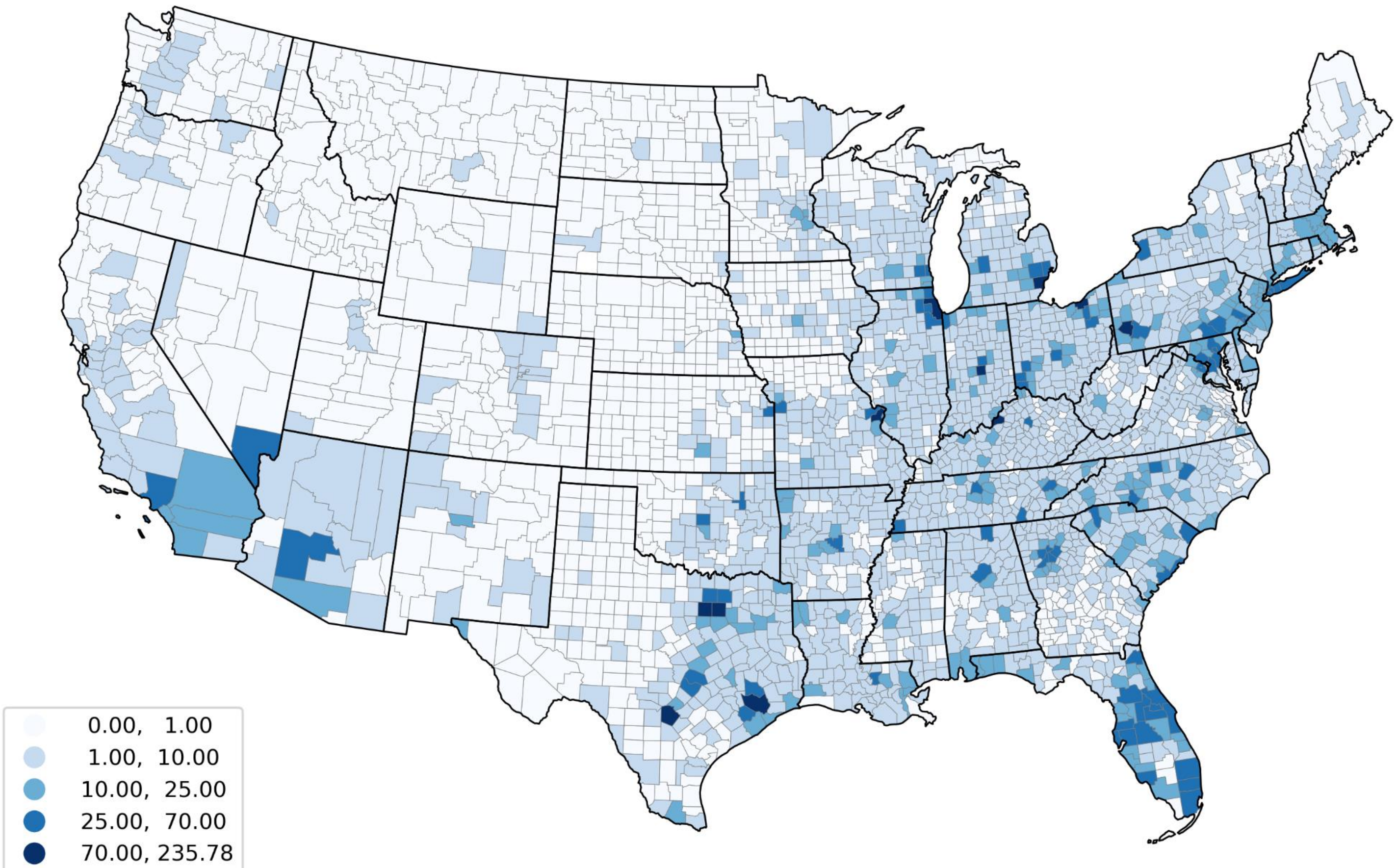

Figure 8: Projections of premature deaths averted by 2040 under CES40B compared to BAU

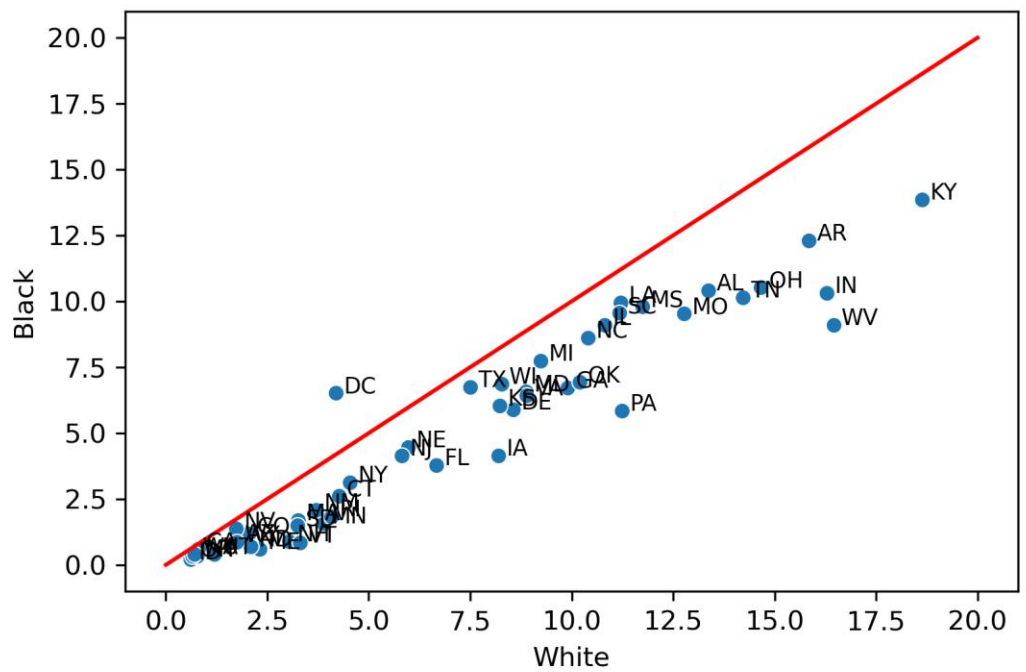

Figure 9: Projections of premature deaths averted in 2040 per 100,000 individuals. The vertical axis shows values for Blacks and the horizontal axis shows values for Whites, and each point represents and individual state. The red line shows where rates are equal between the groups.

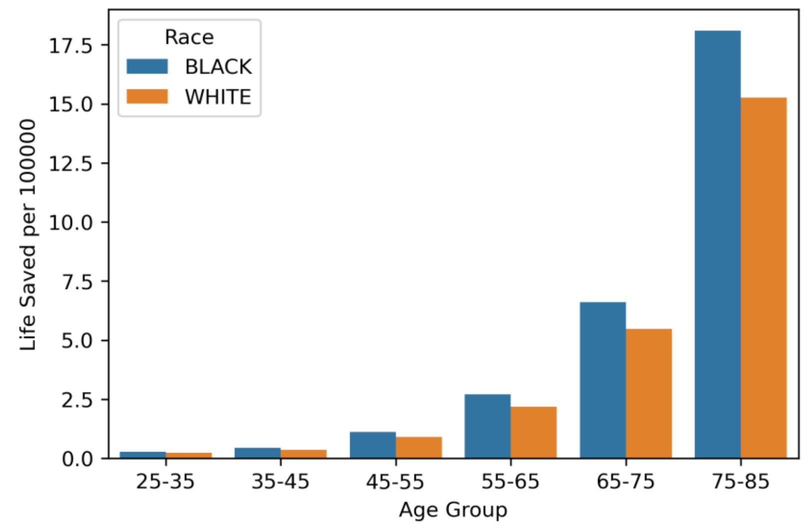

Figure 10: Projections of premature deaths averted in 2040 by race and by age under CES40B compared with BAU

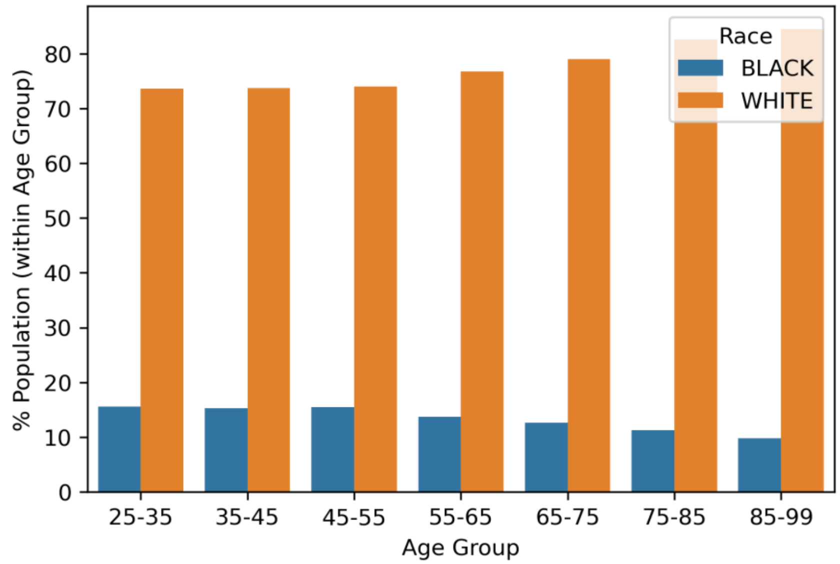

Figure 11: Population distribution by race and age in 2040 from BenMAP (Wood and Poole Economics, Inc)

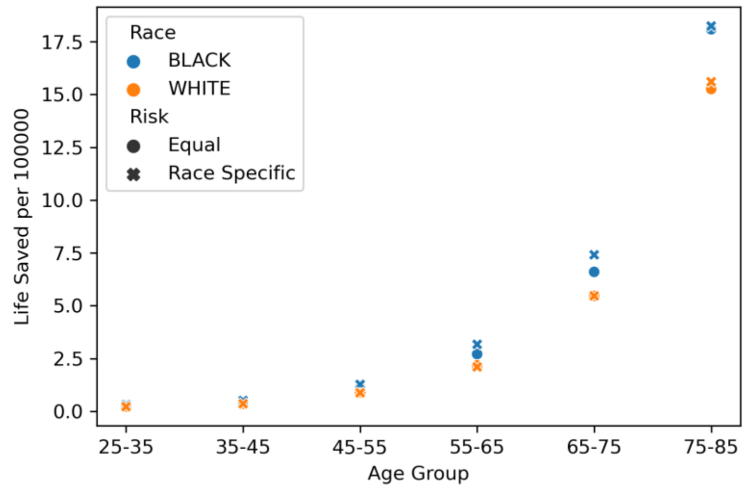

Figure 12: Projections of premature deaths averted in 2040 by race, risk, and age under CES40B compared with BAU

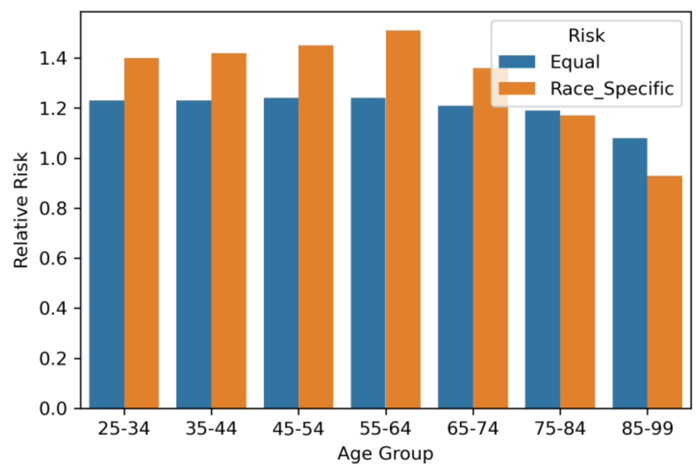

Figure 13: Projections of changes in relative mortality risk in 2040 by age under CES40B compared with BAU, under equal and race-specific baseline mortality

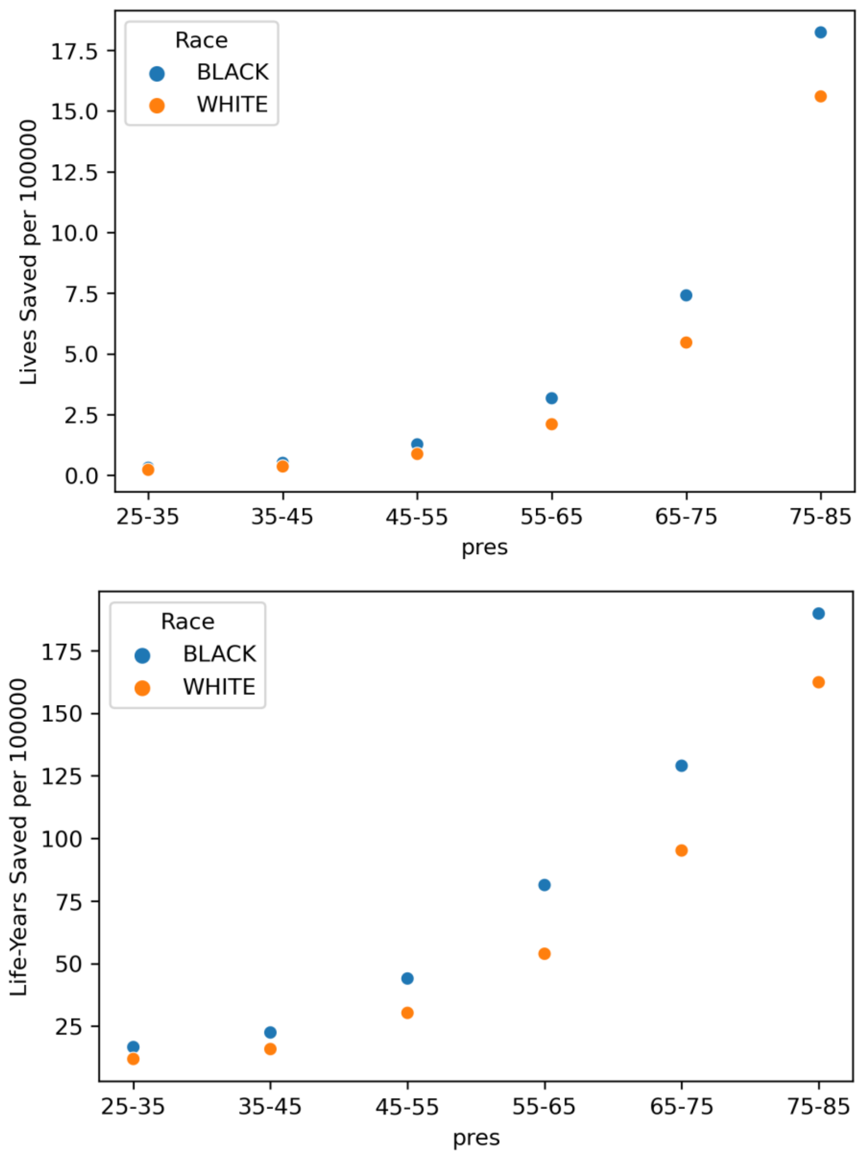

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Figure 14: Projections of lives saved and life-years saved in 2040 per 100,000 by race and age under CES40B compared with BAU

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Chapter 3: Assessing the Environmental Justice Implications of Decarbonizing the US Electric Grid: Estimating Changes in Asthma Exacerbation by Race and Income

Introduction

Asthma is a chronic respiratory condition characterized by inflammation and narrowing of the airways. Individuals with asthma are subject to periodic exacerbations of the condition, commonly known as asthma attacks, during which the airways become more constricted, causing increased difficulty in breathing and decreased lung function. During an asthma exacerbation, the individual experiences symptoms such as wheezing, coughing, chest tightness, and shortness of breath. Asthma exacerbation can be classified as mild, moderate, and severe, and the last two often require emergency department visits or hospitalization (Pollart et al. 2011). They can be very serious: asthma contributes to thousands of deaths yearly, costing the US over $80 billion annually (CDC 2022: AAFA 2021). Even non-fatal exacerbations have significant impacts: in 2019 asthma caused over 1.8 million emergency department visits, of which 44% were for children under 18 (CDC 2021).Asthma is an important issue for environmental justice. Results from National Health Interview Survey (NHIS) 2006-2018 show that the prevalence of asthma and asthma exacerbations is higher for people experiencing poverty and people of color (PoC) than for the rest of the population (Barnthouse and Bridgette 2019; CDC 2022; Grant et al. 2022; Pate et al. 2021; Stern et al. 2020). Poverty increases exposure to viral infections, allergens, and pollution, which can increase the risk of asthma exacerbation (Forno and Celedón 2012; Weinberg 2000), as do environmental conditions associated with poverty such as smoking and poor housing conditions (Rona 2000). Moreover, research suggests that after adjusting for socioeconomic status, asthma is more common among Black children than Latinos, and that among Black children in particular, the prevalence of asthma is significantly higher in households with low socioeconomic status (Thakur et al. 2012; Carroll 2013; Assari and Lankarani 2018). Finally, although the risk of asthma declines with increasing income for both Black and White children, the decline is weaker for Black children (Assari and Lankarani 2018). This suggests that policies that reduce the racial gap in socioeconomic status may not be enough to eliminate the disproportionate prevalence of asthma among Black children.

In addition to race and income, it is also important to consider how the burden of asthma is heavily borne by children, especially those living below the poverty line. Asthma among children can also lead to limitations in physical activity, impaired quality of life, and negatively impact the educational outcome of children (Gilraine and Zheng 2022; Mullen et al. 2020; Requia et al. 2022; Yan et al. 2022). Each year asthma exacerbations are responsible for more than 19 million missed school days, and for a loss of 500 age-standardized disability-adjusted life years per 100,000 (Akinbami et al. 2011; Moonie et al. 2006; Nurmagambetov et al. 2018, Zhang and Zheng et al. 2022). Furthermore, child poverty rates are higher in the US compared to other developed countries, and they differ significantly by state and race (McCarty 2016). Moreover, data from the American Community Survey (ACS) shows that the poverty rate is higher for families with children than those without children: in 2021, 15.7% of children lived in poverty, while the share of adults between 19 and 64 years old was 10.5%, and the rate for those 65 and over was only 10.3% (Kaiser Family Foundation 2023). Therefore, in order to accurately model the EJ implications for the effects of a regulatory policy on asthma, it is crucial to analyze the population by both income or poverty status, and give special attention to children.

In this paper I show how improvements in air quality from decarbonizing the US electric grid would significantly reduce asthma exacerbation for poor children, especially in Black households. Equally important, I demonstrate why using race and poverty status based prevalence functions is critical for understanding the distribution of health improvements and evaluating whether policies contribute to environmental justice goals. To obtain these results, I employ a number of different federally administered datasets to calculate baseline childhood asthma prevalence by state, race and income. I then use these results to examine how changes in air quality under CES40B relative to BAU impact asthma exacerbations among children. Finally, I conclude by discussing how cross-agency collaborations can help researchers develop higher-resolution datasets that would allow improved precision in determining changes in asthma exacerbations.

Air Quality and Asthma

Poor asthma control, individual susceptibility, viral infections, allergen exposure, physical activity, environmental tobacco smoke exposure, and outdoor pollution are some risk factors identified for asthma exacerbation (Forno and Celedón 2012). Public health scholars and epidemiologists have highlighted exposure to air pollution as a key critical risk factor for asthma and asthma exacerbations (Guarnieri and Balmes 2014; Rosenquist et al. 2020; Toskala and Kennedy 2015). Several meta-analyses have studied the effects of PM, SO_x, NO_x, CO, and O_3 on higher hospitalization, incidence, and prevalence of asthma (Bowatte et al. 2017; Fan et al. 2016; Gasana et al. 2012; Orellano et al. 2017; Zheng et al. 2015). Fine particulate matter below 2.5 micrometers ($PM_{2.5}$), in particular, is an important risk factor for asthma exacerbation (Glad et al., 2012; Mar and Koenig, 2009; Rosenquist et al., 2020) and has been shown to be associated with cough, shortness of breath, and wheezing (Ostro et a. 2001; Mar et al. 2004). Ostro et al. (2001) studied the relationship between air pollution in Los Angeles and asthma exacerbation among African American children (8 to 13 years old).

An important source of air pollutants is electricity generation. It can produce a variety of emissions that contribute to poor air quality, including direct emissions of $PM_{2.5}$ as well as NO_x, SO_x, and volatile organic compounds (VOCs). These can undergo chemical reactions in the atmosphere to produce $PM_{2.5}$. $PM_{2.5}$ causes impacts on human health beyond asthma (Hodan et al., 2004) and there is an extensive literature modeling the impact of power sector emissions on premature mortality (Fann et al. 2013; Penn et al. 2017; Thind et al. 2019) including decompositions of the impact by race and income (Luo et al. 2022; Thind et al. 2019; Zhu et al. 2022).

To my knowledge, only a few studies examine how the retirement of fossil fuel electricity generation units (EGUs) affects asthma (Casey et al., 2020; Milner et al., 2022). This paper contributes to that literature by modeling the asthma exacerbation-related health benefits due to decarbonizing of the US electric grid for the contiguous US in 2030, 2040, and 2050. Moreover, to understand the environmental justice implications of the transition, I perform a detailed decomposition by state, race, and income. I focus on children under the age of 18 because asthma has been the leading chronic illness amongst children and most of the prior scholarship related to the health benefits of decarbonization has overlooked it by focusing on premature deaths averted among adults (Fann et al. 2013; Penn et al. 2018; Thind et al. 2019).

For modeling the reductions in asthma exacerbation due to improvements in air quality, I use EPA's Environmental Benefits Mapping and Analysis Program (BenMAP).[3] BenMAP has been extensively used to model health benefits in the US and worldwide (Aldegunde et al. 2023; Chen et al. 2017; Davidson et al. 2007; Driscoll et al. 2015; Sacks et al. 2018). BenMAP includes health impact functions for a wide range of endpoints, incidence rates, baseline prevalence, and population statistics by race, gender, and age. The version I use has population projections for 2030, 2040, and 2050. BenMAP does not disaggregate the population by income; as discussed below I extend it to do so since poverty is a significant risk factor for asthma.

[3] Specifically, the open source community edition BenMAP-CE v1.6, which I will refer to as simply BenMAP.

Asthma Prevalence by Demographic and Geographic Characteristics

Discussion on the literature in the previous discussion suggests that poverty and asthma prevalence rates vary by race and state in the US. In this section, I detail how poverty and asthma prevalence rates were estimated and present the variations in these variables by income, race and state.

To determine the number of children by state, race (Black, White and all) and poverty status I use data for 2021 from American Community Survey (ACS) conducted by the US Census. Information was not available for states with small Black populations like North Dakota, South Dakota, and Wyoming. I did not include these states in the analysis.

BenMAP has asthma exacerbation prevalence rates by age group provided by the American Lung Association and based on the National Health Interview Survey from 2008 (EPA 2023). For all races, it is 6.14% for children below 5, 10.7% for the age group 5-17, and 9.41% overall for children under 18. For Blacks the rates are higher: 9.98% for children below 5, 17.76% for the age group 5-17, and 15.53% overall for children below 18. Recent reports analyzing asthma datasets report differences by income, race, and geography (Pate et al. 2021; AAFA 2021). To account for these differences, I build state-specific prevalence rates following a procedure similar to that used by EPA to build BenMAP's race-specific mortality rates for 2007-2016 (EPA, 2023).

I use data from Behavioral Risk Factor Surveillance System (BRFSS) operated by the Centers for Disease Control to build asthma prevalence by state, race and poverty status. BRFSS is a system of health-related telephone surveys regarding health-related risk behaviors, chronic health conditions, and use of preventive services. Certain parts of the survey are implemented across all the states, while others are assigned to only some states. The asthma section of the survey is conducted in a subset of states every year.

As mentioned in the previous section, asthma prevalence varies by poverty status and decreases for households with income above the poverty line (Assari and Lankarani 2018; Pate et al. 2021; Zahran et al. 2018). Fortunately, BRFSS gathers considerable socio-economic and demographic information on households. To estimate child poverty based on household income and household size of interviewees in BRFSS, I use the federal poverty thresholds for each respective year provided by the Census to determine each household's poverty status. These thresholds are updated annually, and they take into account household size, and are used to assess whether family's income falls below the poverty line.

The percentages of children living in poverty by race and state are mapped in Figure 1. There are some pronounced differences among the three groups in this analysis. Throughout the US, Black children have substantially higher poverty rates compared to White children and children overall. Data from 2021 ACS shows that poverty rates also vary by state. More than 25% of the Black households with children with income below the poverty line live in Alabama, Kentucky, Louisiana, Michigan, Mississippi, Ohio, and Pennsylvania. These are also states which have sizeable Black populations. Relative to this pattern, the percentage of White children in households with income below the poverty line are lower, with the largest concentrations of poverty in New Mexico, Kentucky, and West Virginia. Among children of all races there is a significant concentration of poverty in the Southern US, like in Alabama, Arkansas, Louisiana, and Mississippi. Across the US, 14-33% of Black households with children are living below the poverty line (the rate varies by state); for White households with children the rates are substantially lower: 4-15%.

Similar to the ACS, in BRFSS, certain states like Idaho, North Dakota, or Wyoming have low Black populations and results for Black residents are not reported; I did not include these states in my analysis. Moreover, the asthma portion of BRFSS is not conducted in South Carolina, Colorado, Arkansas, and South Dakota, so I dropped these states from the analysis as well. For the rest of the US, I pooled the BRFSS survey from 2008-2021 and computed asthma prevalence by poverty status, race, and state.

There are two crucial child asthma prevalence-related questions in BRFSS. The first is "Has a doctor, nurse, or other health professional EVER said that the child has asthma?" and the second is "Does the child still have asthma?". As mentioned earlier, BRFSS also provides households' annual income and household size. We eliminate the observations where respondents refused to answer or answered 'Don't Know.' After estimating the poverty status of the household, I use the response to the second question to estimate the prevalence, $\mu_{i,r,s}$, by poverty status, race, and state using the following calculation:

$$\mu_{i,r,s} = \frac{Y_{i,r,s}}{Y_{i,r,s}+N_{i,r,s}} \times 100$$

where $Y_{i,r,s}$ is the number of households who reported 'Yes' to the question and $N_{i,r,s}$ is number of households who reported 'No.'

The CDC considers BRFSS results to be unreliable when the sampled population is smaller than 50 (2018). Following this methodology, we dropped states with small sample sizes for two or more race and income groups. However, to include as many states as possible, we kept states that had adequate samples for all but one race and income group. For example, I include Oregon in the sample even though it has a small sample of Black households below the poverty line because it has adequate samples for White households below the poverty line and both Black and White households above the poverty line.

Before the prevalence results are discussed, it is essential to note that there were not enough BRFSS observations for White households below the poverty line in North Carolina and District of Columbia. Furthermore, the samples for Black households in poverty are small for Arizona, Delaware, Minnesota, North Carolina, Oregon, Utah, and West Virginia. Finally, some states in the Rockies and New England had adequate samples in the BRFSS data but had to be eliminated because the ACS does not report the number of Black households in poverty in those states.

A set of maps are presented in Figure 2 for the prevalence of asthma by poverty status and race for Black, White, and all respondents. States in grey were not part of the BRFSS asthma survey, and states with diagonal hatching have small samples for the corresponding subgroup. In Figure 2 states with darker shades of red have higher asthma prevalence rates among children. There are some evident patterns. Firstly, consistent with the prior literature, asthma prevalence is higher across all the race groups for households in poverty. Second, it is also important to note that for both income groups, Black households have higher prevalence rates than White households and or households overall.

As mentioned earlier, EPA reports a national average asthma prevalence rate of 9.41% for all children and 15.53% for Black children in BenMAP. These numbers are comparable to those reported in Figure 1 and Table 1-6. Prevalence is substantially higher than the overall national average amongst households in poverty. However, the share of this population is relatively small. This emphasizes the earlier point that asthma prevalence is high in economically vulnerable households, especially for those identifying as Black. Detailed results by state, race, and poverty status for baseline prevalence (and asthma exacerbations averted, as discussed in the following section) are presented in Tables 1-4 in the Appendix. Next, I discuss how these poverty and prevalence rates were used to model changes in asthma exacerbation due to air quality improvements in 2030, 2040, and 2050.

Modeling the Impacts of Air Quality Improvements:

This paper shares the methodological approach of essay 1. The Integrated Planning Model (IPM) was used to predict changes in generation and emissions for the contiguous US under a clean energy standard policy (CES40B). These changes were then used to determine changes in ambient $PM_{2.5}$ concentrations using the Community Multiscale Air Quality (CMAQ) Model. To avoid repetition from essay 1, details regarding the use of both IPM and CMAQ are omitted here and I focus only on the modeling of asthma. I used the change in ambient air quality for $PM_{2.5}$ to model reductions in asthma exacerbation (cough, wheezing, and shortness of breath) in BenMAP. In this section I detail the modifications that were used to allow BenMAP to determine reductions in asthma exacerbation by poverty status, race and state under CES40B.

BenMAP's health impact functions have four sets of inputs: the change in air quality, parameters for the corresponding concentration response functions, the population, and the baseline prevalence. For comparison, I also ran BenMAP with nationally aggregated prevalence as defined in BenMAP (Based on American Lung Association 2010) to investigate how the results differ when both the population and prevalence are split by income, race, and state. The health impact function (HIF) for the change in asthma exacerbations attributable to $PM_{2.5}$ has the form below:

$$E_{i,r,s} = \left(1 - \left(\frac{1}{(1-A).\exp(\beta * \delta Q_c) + A}\right)\right).A\,.\mu_{i,r,s}.\rho_{i,r,c}$$

where $E_{i,r,s}$ is asthma exacerbations averted by *'i'* income, *'r'* race and state *'s'*. Following BenMAP (EPA 2023), I take β and A from Ostro et al. (2001). Several studies have used their associations to model asthma-related health benefits for transportation and energy policies (Aldegunde et al. 2023; Abel, 2019; Coomes et al. 2022). The term δQ is the change is ambient $PM_{2.5}$ due to decarbonization. CMAQ provides output for a grid of 36 x 36 km cells and I aggregate these values to county (c). $\mu_{i,r,s}$ is the asthma prevalence by poverty status *'i'*, race *'r'*, and state *'s'* as discussed in the previous sections. Finally, $\rho_{i,r,c}$ is the population projection by poverty status, race, and county. I split the population projections in BenMAP by child poverty rates extracted from the 2021 ACS**.** Finally, note that this HIF was evaluated for each of the three asthma exacerbation endpoints (wheezing, cough, and shortness of breath) as reported in BenMAP and the underlying literature. In the results below I assume that these exacerbations are independent and equally harmful and sum them to get an overall change in asthma exacerbations.[4]

Asthma exacerbation cases averted per 100,000 people of poverty status i, race r, and state s is given by:

$$\hat{D}_{i,r,s} = 1e5 \,.\, \frac{E_{i,r,s}}{P_{i,r,s}}$$

where $P_{i,r,s}$ is the corresponding population. I obtain national values of avoided exacerbations by poverty status and race, $E_{i,r}$, as well as the corresponding populations, $P_{i,r}$, by summing across states. Asthma exacerbation cases averted nationally per 100,000 people of each income-race group is then given by:

$$\hat{D}_{i,r} = 1e5 \,.\, \frac{E_{i,r}}{P_{i,r}}$$

[4] The assumption that they are equally harmful is consistent with the valuations used in BenMAP, which are based on Dickie and Ulery (2002).

Changes in Asthma Exacerbation

Before discussing the asthma exacerbation results, I first discuss the distribution of changes in air quality by income and race. In essay 1 I suggest that Black children, on average, experience the largest reductions in pollution. White children have the second largest reductions, and the other races have comparatively smaller gains. This pattern is primarily because of the distribution of the population in the US. Black children are concentrated in the Midwest and South. In essay 1 I show that the Eastern half of the US experiences the most significant improvements in air quality under CES40B compared with BAU. Comparatively, there is a more significant concentration of Native American and Asian populations in the Western US and parts of the Rockies. These regions do not experience significant gains in air quality because, to begin with, they are less reliant on electricity from fossil-fueled EGUs and coal fired units in particular. A kernel density plot is shown in Figure 3 for the reduction in exposure to $PM_{2.5}$ by income and race. This figure was produced for the whole US population, using the change in pollutant concentration, δQ, and county-level population. There are two interesting patterns here. Firstly, for all respective races, children living below the poverty line have a relatively higher share of larger reductions in exposure (measured as the concentration in CES40B less the concentration under BAU) than children living above the poverty line. Secondly, the share of Black children experiencing small reductions in pollution (<0.18 $\mu g/m^3$) is relatively low but the share experiencing moderately large reductions (0.18-0.33 $\mu g/m^3$) is relatively high. Only at higher reductions (>0.35 $\mu g/m^3$) do Whites and all races together have slightly larger population shares than Blacks.

Nationally I find that 190,317 asthma exacerbation cases are averted in 2040 under CES40B compared with BAU for children above the poverty line, with 46,313 for Black children, 95,229 for Whites, and the remainder in families not identifying as Black or White. Below the poverty line 45,175 cases are averted, with 18,376 for Black children and 14,421 for Whites. Of the total cases averted, 49% are attributable to wheezing, 31% to shortness of breath, and 21 % to cough.

Asthma exacerbations averted nationally under CES40B compared to BAU in 2030, 2040, and 2050 by race and income and normalized per 100,000 individuals are provided in Figure 4. When the results are aggregated to the national level, across all race groups, children living below the poverty line have substantially more significant gains compared to children living above the poverty line. Furthermore, Black children below the poverty line have almost 986 per 100,000 asthma exacerbation cases averted in 2040. White children have almost 679 cases averted per 100,000. Interestingly I find that the number of per capita asthma exacerbations averted for all races is less than the per capita numbers for White and Black children: 612 per 100,000 in 2040. A similar pattern can be observed for children above the poverty line. Per 100,000 individuals, Black children have 742 cases averted, White children have 393, and all races together have 377. This pattern suggests that groups other than non-Hispanic Blacks and Whites have significantly smaller gains for both income levels.

For comparison, I ran BenMAP with its built-in dataset, which lacks income disaggregation and uses a constant asthma prevalence value applied across the nation. The results of this analysis are as follows: 719 exacerbations avoided per 100,000 for Black children, 675 per 100,000 for White children, and 591 per 100,000 for all races together. All of these values lie between the health benefits reported above for children in households above and below the poverty line. This analysis suggests my methods for disaggregating the population by poverty status, race, and state are plausible. However, note that several of BenMAP's HIF parameters—those that come from epidemiological studies—are not disaggregated by poverty status, race, and state. Due to this limitation, the current study may not be capturing the full EJ implications of the policy.

A set of maps are provided in Figure 5 detailing the asthma exacerbation cases averted per 100,000 by race and poverty status. Results suggest that there is large concentration of health benefits for states in the eastern half of the US. In absolute terms, the largest benefits are in Florida, Georgia, Illinois, Indiana, and Texas. Please refer to Tables 1-4 in the appendices for the specific numerical values. There is great variation among states in asthma exacerbations averted per 100,000 by income, race, and geography under CES40B compared with BAU. For example, Black children in Kentucky living below and above the poverty line show 2,586 and 1,745 cases averted, respectively, per 100,000 individuals. Gains for White children in Kentucky living below and above poverty are projected to be 1,617 and 979 cases averted per 100,000. However, results in California are starkly lower: Black children living above and below the poverty line in California are projected to experience 63 and 81 cases averted per 100,000 while White children living above and below the poverty line are projected to experience 28 and 35 cases averted per 100,000.

States with large gains in asthma cases averted per 100,000 are Alabama, Indiana, Kentucky, Ohio, Missouri, Pennsylvania, Texas, and Wisconsin. Across all states, children living below the poverty line have larger health benefits than children above the poverty line. Much like the national aggregation, for many states, both Black children living above and below the poverty line have more significant gains than White children and children of all races together living above and below the poverty line under the decarbonization scenario.

In Figure 6a asthma cases averted per 100,000 individuals under decarbonization for Black children above and below the poverty line are shown for all states. The red line indicates equal outcomes for both income groups. For almost all states, children below the poverty line (the vertical axis) show greater improvements. There are some states close to the origin, primarily Western states like California, Oregon, Nevada, and Washington, where air quality is not substantially affected by the policy. Most states are close to the red line, with similar outcomes above and below the poverty line, but some, like Kentucky, Indiana, Wisconsin, Alabama, and Pennsylvania, have much more significant gains for children below the poverty line. Figure 6b is similar to 6a but for White children. However, unlike 6a, children below the poverty line receive larger improvements than those above it for all states. Interestingly, for both Black and White children (Figures 6a and 6b), in most states other than those in the West, children below poverty line have 50-75% larger gains compared to children above poverty line.

Figure 6c compares the asthma improvements under decarbonization for Black and White children above poverty line. Apart from the Western states, the gains for Black children above the poverty line are 50-100% larger than the gains for White children above the poverty line. Some prominent states with the most pronounced differences are Kentucky, Indiana, Missouri, Ohio, Illinois, and Wisconsin. Figure 6d compares Black children above poverty line with White children below the poverty line. Here are some fascinating results. Firstly, there are many states with outcomes close to the line, although some are above it (better outcomes for high income Black children) and others are below it (better outcomes for low income White children). This is a mixed case of results, where some states like Kentucky, Indiana, Ohio, Missouri, Oklahoma, Illinois, and Tennessee are projected to experience more significant gains for Black children above the poverty line under decarbonization than White children below the poverty line. At the same time, in Alabama, West Virginia, Virginia, and New York, White children below the poverty line have more considerable gains than Black children above the poverty line.

In the Figure 6e, reductions in asthma exacerbations in 2040 for Black children below the poverty line are compared those for with White children above the poverty line. There are some stark differences between the two groups. The gains for Black children in most states are often 50-200% larger than those for White children. The states with the largest EJ gains (improvements for low income Black children exceeding those for high income White children) are Kentucky, Indiana, Missouri, Ohio, Alabama, Pennsylvania, Illinois, Mississippi, and Wisconsin.

Finally in Figure 6f the health benefits for Black children below the poverty line is compared to that for White children below the poverty line. In this comparison, the gains for Blacks under decarbonization in many states are 50-100% larger than those for Whites. States like Kentucky, Indiana, Ohio, Missouri, and Wisconsin are observed to have substantial improvements in asthma cases under decarbonization. The only states where Whites with below the poverty line have more considerable gains than Blacks below the poverty line are West Virginia, Virginia, and New York. For West Virginia, the gains are almost 100% larger for White children compared to Black children. However, note that prevalence rates for Black children below the poverty line in West Virginia are unreliable.

Discussion

I had two objectives for this paper. Firstly, I attempt to quantify asthma exacerbation cases averted by poverty status, race, and state due to the decarbonization of the US electric grid. Secondly, I developed a methodology for building a unique set of prevalence and poverty rates by race and states using BRFSS, Census's poverty thresholds, and the ACS. The results highlight how these variables can help improve understanding of EJ related to health impacts due to decarbonization policies for the electricity sector.

Much like essay 1, in this paper I observe that health benefits are concentrated in the Eastern half of the US, with a larger share of benefits for Black children and children living below the poverty line. These factors can be attributed to the spatial pattern of emissions reductions and to existing socioeconomic inequity. Firstly, the Eastern half of the US is more reliant on fossil energy (especially coal) compared to the Western half of the US. Therefore, these regions also accrue the largest improvements in air quality, particularly parts of the Ohio Valley and southern US (Richmond-Bryant et al. 2020; Vasilakos et al. 2022; Driscoll et al. 2022). Secondly, due to historical and socio-economic circumstances, Black households with children have a relatively higher poverty burden than other races (Koball, Moore, and Hernandez. 2019). This pattern can be observed across almost all the states in the US. However, there are some nuances. Some states, like West Virginia, have been reliant on the coal mining industry, which is in decline due to fracking. White communities in the area have not progressed economically, and have been hit hard by the opioid crisis, and they have a large share of the population with living below the poverty line (Baker 2022; Young et al. 2023). Moreover, Eastern states also have a more significant concentration of the Black population than elsewhere in the US. In contrast, Asians are more concentrated in the Western US, and Native Americans are more spread out in their ancestorial lands in different parts of the US like the Rockies, Alaska, Oklahoma, and New Mexico. Finally, it is also essential to consider that PoC tend to be younger than the White population (Colby et al. 2015; Census 2021).

In this paper I highlight how using race and poverty status specific prevalence and population parameters results in health benefits from decarbonization policies that are disproportionately higher for Black children and children living below the poverty line. Premature deaths are the central piece of health benefits from decarbonization. However, it is vital also to consider the health benefits for youth because asthma can also lead to impaired quality of life over the long-term for children (Gilraine and Zheng 2022; Mullen et al. 2020; Requia et al. 2022; Yan et al. 2022). While some scholars have analyzed premature deaths attributed to $PM_{2.5}$ using race-specific CRFs, and have decomposed health benefits by income, race, and geography (Spiller et al. 2021; Thind et al. 2019), there is considerable work needed to develop unique income, poverty-status, race, and geography-based incidence, and prevalence rates, especially for non-mortality endpoints. It is reasonable to state that incidence and prevalence rates for many other diseases are likely higher for disadvantaged communities and PoC. For this reason, it is essential that when health benefits are estimated by income, economic status, race, and geography, specific disease incidence and prevalence rates are used to understand the full EJ implication of energy policies. In this paper I offer some methodological insights concerning asthma, and hope that other scholars will consider developing similar methodologies for other diseases.

To fully understand the EJ implications of asthma-related health benefits due to the decarbonization of the US electric grid, I relied on four unique federal datasets and three regulatory grade models. Prevalence rates are calculated from the CDC's BRFSS (2008-2021), child poverty status from the Census's ACS (2021), poverty thresholds and household size by year from the Census, and population and epidemiological parameters from EPA's BenMAP. There is considerable publicly accessible data which can be used to create high-resolution incidence, prevalence, and population parameters for different socio-economic groups and diseases. To complement these high-resolution details, I believe that EPA should consider updating the prevalence and incidence parameters for different diseases and population groups. So far, there is no income-based parameter in BenMAP. However, EPA does recognize the significance of income concerning EJ. EPA has built the EJScreen tool, which tries to capture the distribution of at-risk populations, and it defines low-income at-risk populations as earning less than two times the poverty threshold (EPA 2022). Unfortunately, I could not analyze income groups beyond simply poverty status because the publicly available ACS API does not provide that information. I believe cross-agency collaboration among Federal agencies could enable the usage of restricted data that was not possible with publicly available ACS and BRFSS data. This approach will help EPA measure EJ impacts of regulation policies with much better accuracy.

These first sets of estimates are subject to many vital assumptions and uncertainties in constructing parameters for the underlying HIFs. I discuss some of these below, and anticipate that future scholarship will further advance the input data and methods. Children living in a household with income below the poverty line have the highest asthma prevalence, followed by households earning between one to two times the poverty threshold, and households with income greater than twice the poverty threshold have the least asthma prevalence (Pate et al. 2021). However, I was unable to evaluate distinguish between the two income groups above the poverty line because publicly accessible ACS APIs do not provide such high resolution data at the state level. Scholars should consider using other datasets like Current Population Survey (CPS), Annual Social and Economic Supplements (ASEC), and Integrated Public Use Microdata Series (IPUMS) to understand population distribution by income, race, and geography.

In addition to BRFSS, the National Health Interview Survey (NHIS) and the National Survey of Children's Health (NSCH) ask many important questions concerning health status and access. However, the NHIS suppresses the results by state in the publicly available dataset. Pooling the two databases would give reliable observations and information on states like Arkansas, Colorado, and South Carolina, which are not surveyed, or states like North Carolina and Massachusetts, which are not frequently sampled in BRFSS.

An interesting aspect of the results is the similarity of outcomes in the urban Northeast and the more rural Southeast. While there is no publicly available dataset that measures asthma prevalence at the county level, scholars have found that asthma rates for all ages are higher in medium metropolitan areas (8.5%) and small metropolitan areas (8.4%) than in large central metropolitan areas (7.3%) (Pate et al. 2021). Simultaneously, the South has the largest poverty gap between rural and urban areas (Weeks 2018). Ideally, county-level data could clarify the relationship, and CMAQ can model the changes in air quality at the county level. However, due to data limitations for asthma prevalence, I had to aggregate results at the state level. Due to this, I am unable to project reductions in asthma exacerbation at the county level and discuss urban-rural and regional effects of decarbonizing the power sector. It is likely that poor and rural counties have larger health benefits in the South, but there will be regional differences as racially diverse and poor communities are more concentrated in urban centers in Midwest. Future scholarship should consider exploring this in a greater detail.

Another key aspect of the study is that I chose to use the CRFs estimated by Ostro et al. (2001), which are available in BenMAP. The Ostro et al. (2001) study sample exclusively consisted of Black children. As discussed above, asthma prevalence is higher in Black communities, so my estimate may overestimate the health benefits for non-Black race groups. Also, key CRF parameters are uncertain, and can at times vary in a single study. I could not find a meta-analysis that reported parameters for all three types of asthma exacerbation (cough, wheezing, and shortness of breath). Future scholars should consider examining other endpoints like asthma emergency room visits, hospitalizations, acute respiratory illnesses, myocardial infractions, etc. Scholars should also experiment with other air quality measures like O_3, SO_x, and NO_x differentiating the affected population by income, poverty status, race, age, gender, and geography. Machine learning may be a promising way to distinguish between these as drivers of asthma and other endpoints.

Lastly, childhood poverty and asthma prevalence are slowly decreasing over time. The ACS reports that child poverty reached a record low in 2021, although that may have been a temporary outcome due to tax credits put in place during the coronavirus pandemic. Pate et al. (2021) find that decreasing trends in asthma attacks are observed for adults but not children. They also report that prevalence has remained unchanged when aggregated across all age groups, but there are some small changes amongst children over time. To account for these changes, the existing study could be extended. Firstly, assumptions could be made about how these factors might influence asthma prevalence in 2030, 2040, and 2050. Secondly, it may also be essential to disaggregate the results by gender because boys have a higher asthma prevalence than girls (Pate et al. 2021). I avoided making assumptions about future changes in poverty and prevalence to focus more on the significance of disaggregating the HIF parameters and results by poverty status, race, and state. Future scholars can relax those assumptions.

I hope that EPA, other federal agencies, and future scholars will consider the significance of using income, race, and geography-specific incidence, prevalence, and population parameters as HIF input for asthma and other health functions. These can significantly help public health experts, policy practitioners, and scholars fully understand the EJ implications of federal and state regulations.

Conclusion

In this paper I quantify the asthma exacerbation cases averted amongst children by poverty status, race, and state if the US decarbonizes its power sector. I build asthma prevalence rates by race and income from BRFSS (2008-2021) and build the share of the population by race and poverty status from the ACS (2021). All races experience substantial reductions, but there is much variation by poverty status, race group, and state. The benefits are primarily concentrated in the Eastern US states, where substantial reductions in $PM_{2.5}$ are observed. Health gains are most prominent for Black children and children in households with income below the poverty line. This paper has general methodological insights for estimating EJ benefits due to air quality regulations. Using high-resolution datasets that differentiate the incidence, prevalence, and mortality by income, poverty status. race, and state, immense benefits can be observed for the most vulnerable households.

Figures

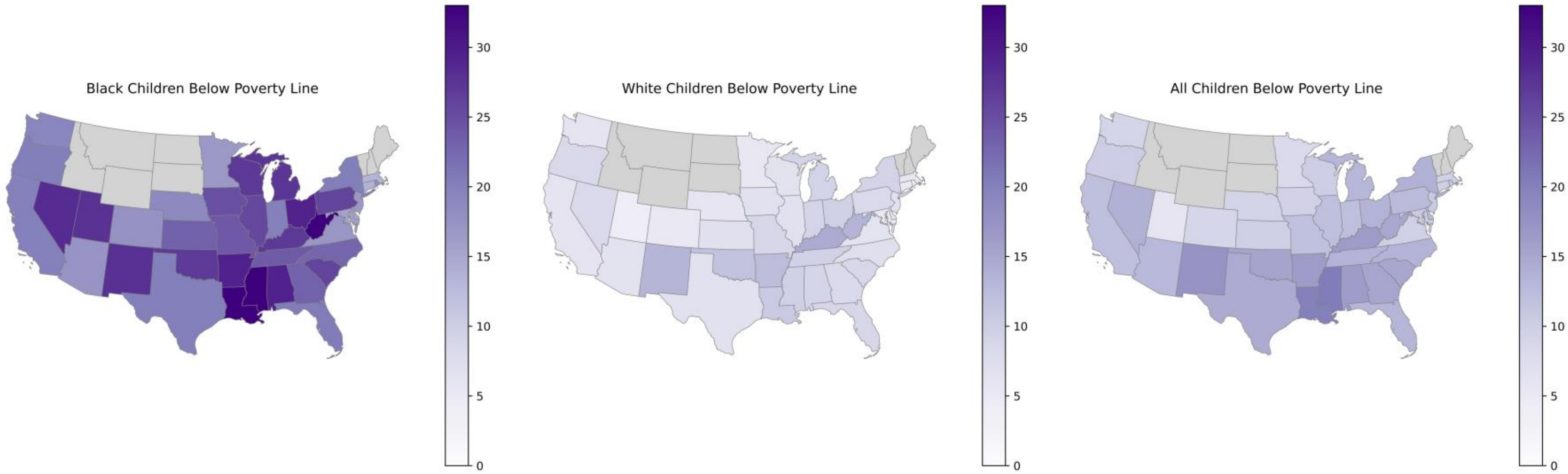

Figure 1: Percentage of children by race living below the poverty line (author's calculations from ACS data).

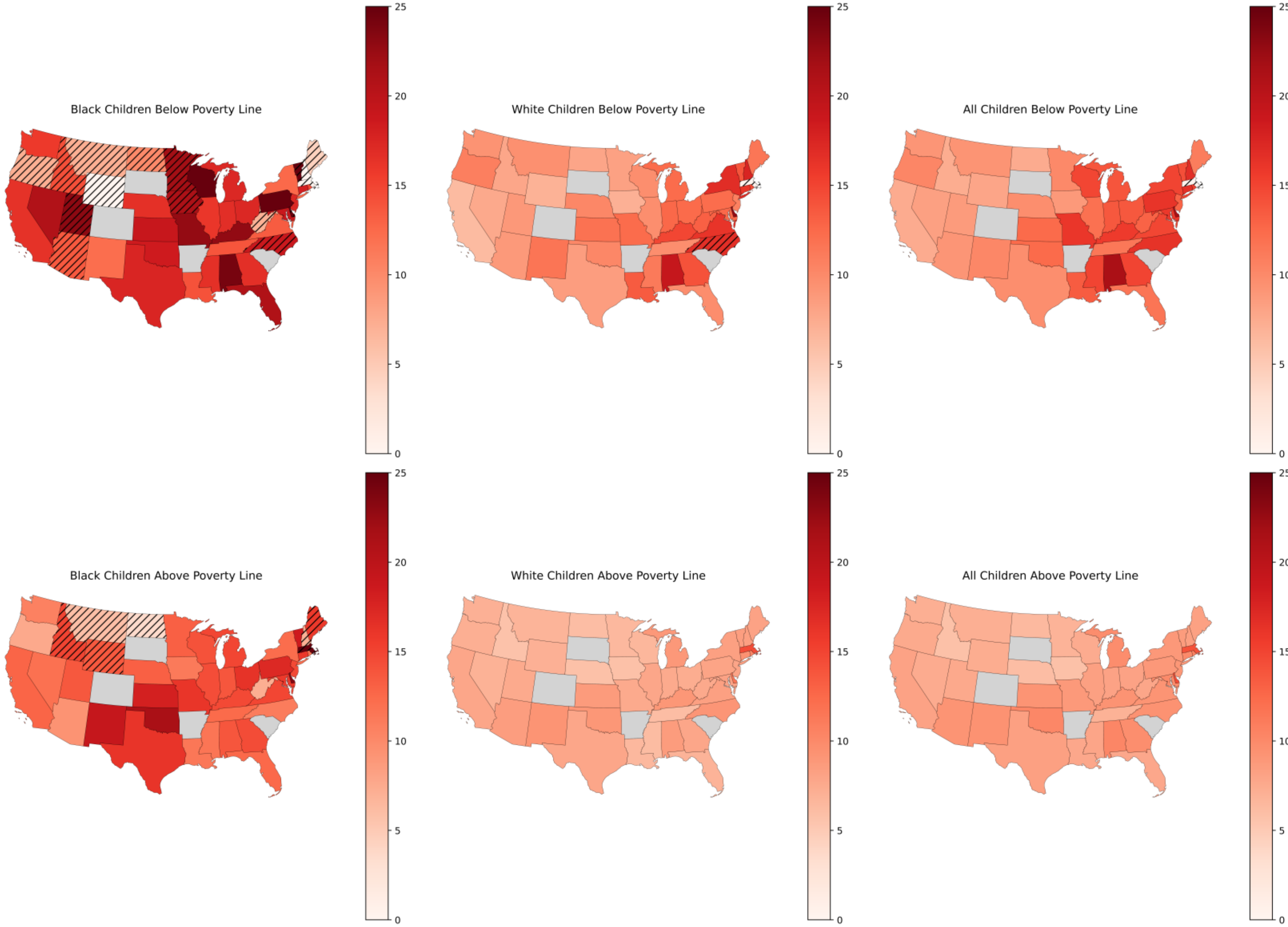

Figure 2: Asthma prevalence in children, by household race and poverty status (author's calculations from ACS data).

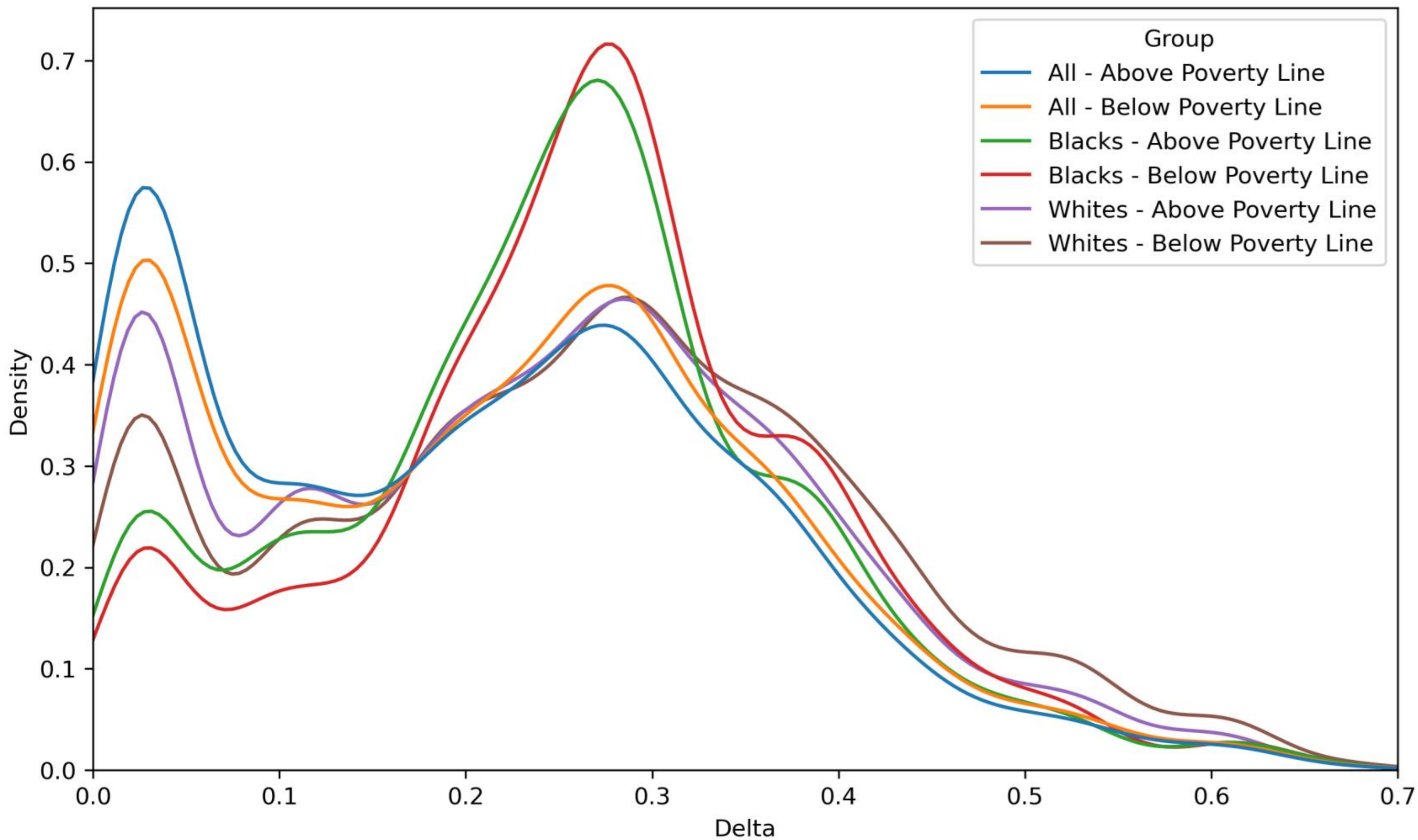

Figure 3: Kernel density plot showing the fraction of each population (vertical axis) experiencing a given reduction in $PM_{2.5}$ in $\mu g/m^3$, (horizontal axis), by poverty status and race

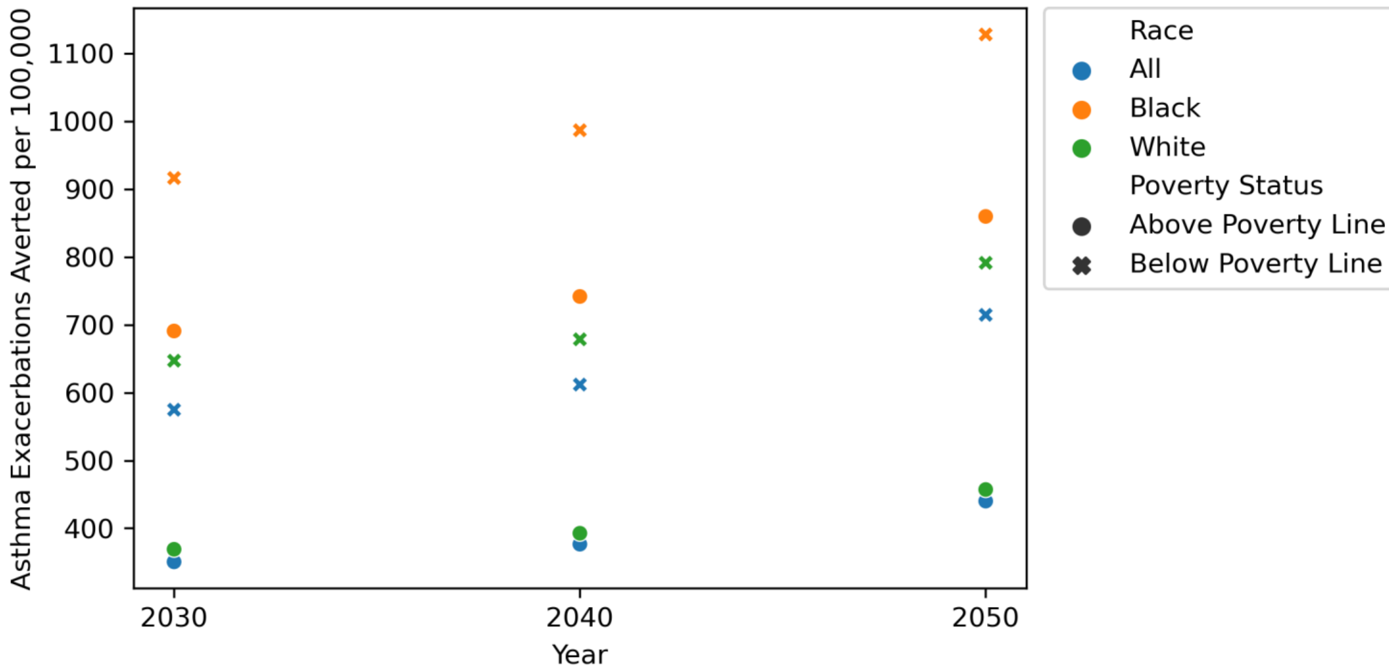

Figure 4: Projections of asthma exacerbations averted nationally under CES40B compared to BAU in 2030, 2040, 2050, by poverty status and race

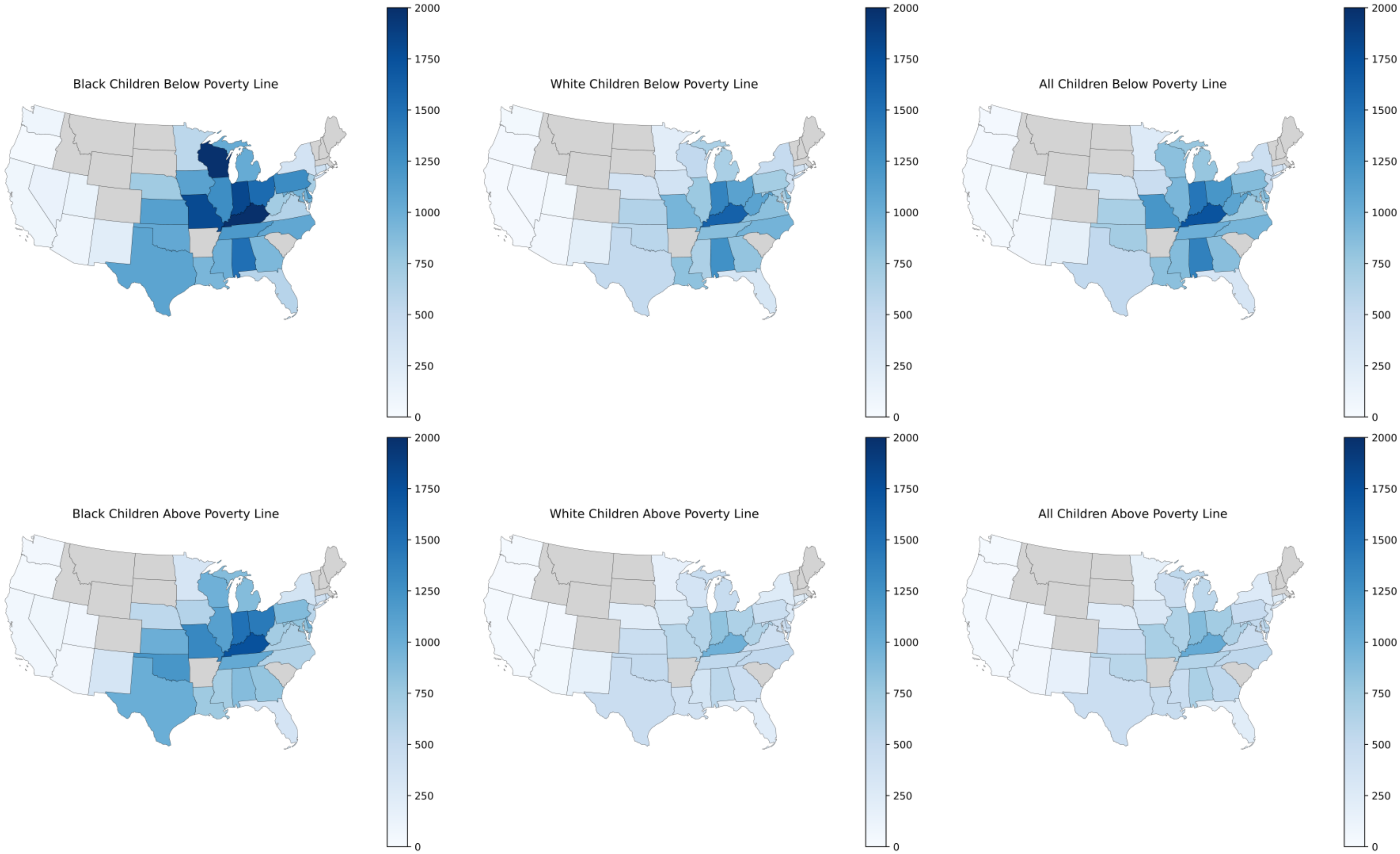

Figure 5: Projections of asthma exacerbation cases averted per 100,000 under CES40B relative to BAU by poverty status, race, and state in 2040

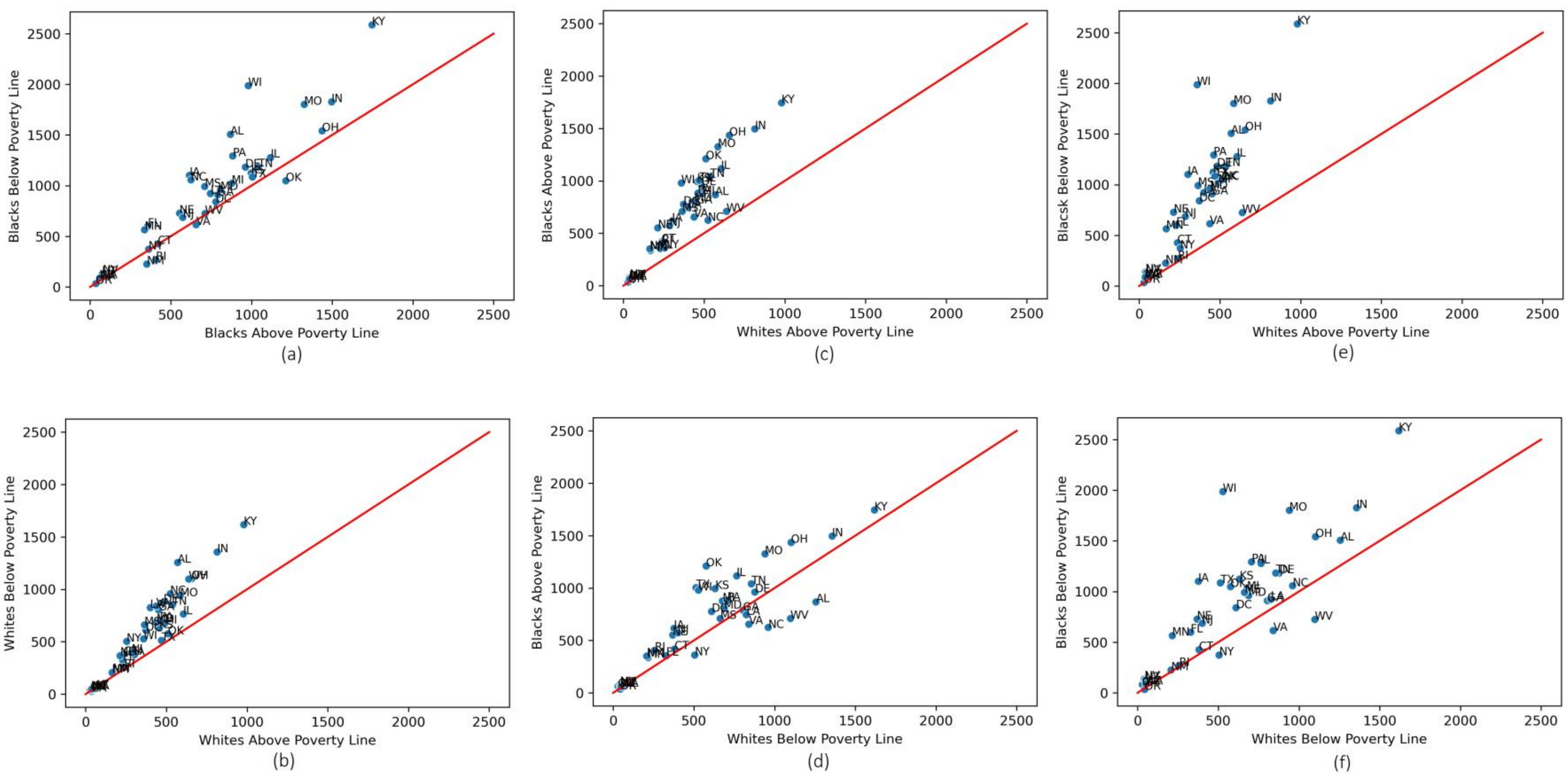

Figure 6: Projections of asthma exacerbations averted per 100,000 under CES40B relative to BAU, by poverty status, race and state

Appendix

Table 1: Population and prevalence (below poverty line)

State	**Percentage of Population**			**Prevalence**		
	All	Blacks	Whites	All	Blacks	Whites
AL	16	29	9	21	24	19
AZ	13	17	6	9	14*	9
CA	12	20	6	7	16	6
CT	10	14	5	17	18	16
DC	18	30	1	20	20	7
DE	12	15	6	21	27*	23*
FL	13	21	9	12	21	10
GA	15	23	8	15	17	14
IA	9	25	7	9	21*	7
IL	12	25	7	12	16	10
IN	12	20	9	14	17	13
KS	10	23	7	12	19	12
KY	16	27	15	16	23	15
LA	20	33	11	13	14	14
MD	10	15	5	16	19	13
MI	13	27	9	14	17	12
MN	8	17	6	10	22*	8
MO	12	24	9	16	23	12
MS	21	34	10	15	17	11
NC	13	23	7	16	19*	17*
NE	9	19	6	10	17	10
NJ	11	17	7	13	17	11
NM	17	28	13	10	12	12

NV	14	28	8	8	21	8
NY	14	20	9	15	13	17
OH	14	29	10	14	17	12
OK	15	27	12	12	18	10
OR	10	20	8	10	7*	11
PA	13	26	8	16	26	12
RI	11	18	4	15	15	15
TN	13	24	9	11	14	10
TX	15	20	7	10	18	9
UT	6	28	4	9	23*	9
VA	10	17	6	15	13	16
WA	9	19	6	9	16	9
WI	10	27	6	15	29	10
WV	15	33	13	13	7*	13

Unreliable prevalence rates are marked with ‘*’

Table 2: Population and Prevalence (above poverty line)

State	Percentage of Population			Prevalence		
	All	Blacks	Whites	All	Blacks	Whites
AL	84	71	91	10	14	9
AZ	87	83	94	9	9	9
CA	88	80	94	8	13	8
CT	90	86	95	10	16	9
DC	82	70	99	11	18	6
DE	88	85	94	15	23	12
FL	87	79	91	8	13	7
GA	85	77	92	10	15	8
IA	91	75	93	6	11	6
IL	88	75	93	8	14	8
IN	88	80	91	8	14	8
KS	90	77	93	9	18	9
KY	84	73	85	9	15	9
LA	80	67	89	8	11	7
MD	90	85	95	10	16	8
MI	87	73	91	10	15	9
MN	92	83	94	7	13	7
MO	88	76	91	9	16	8
MS	79	66	90	8	12	6
NC	87	77	93	9	11	9
NE	91	81	94	6	13	6
NJ	89	83	93	9	15	8
NM	83	72	87	9	19	9
NV	86	72	92	7	12	7
NY	86	80	91	9	12	8
OH	86	71	90	8	16	7

OK	85	73	88	10	21	9
OR	90	80	92	8	8	7
PA	87	74	92	9	17	8
RI	89	82	96	11	12	10
TN	87	76	91	7	12	6
TX	85	80	93	8	16	8
UT	94	72	96	8	14	8
VA	90	83	94	9	15	8
WA	91	81	94	7	11	7
WI	90	73	94	7	14	7
WV	85	67	87	8	7	8

Unreliable prevalence rates are marked with '*'

Table 3: Asthma exacerbation cases averted (below poverty line)

State	Asthma exacerbation			Standard deviation			Asthma exacerbation averted per 100,000		
	All	Blacks	Whites	All	Blacks	Whites	All	Blacks	Whites
AL	2040	1198	569	598	351	167	1382	1508	1256
AZ	122	9	21	36	3	6	59	86	57
CA	337	65	29	99	19	9	36	81	28
CT	227	41	48	66	12	14	389	428	380
DC	167	98	2	49	29	0	810	840	609
DE	196	84	44	57	25	13	894	1183	878
FL	1737	975	420	509	286	123	343	598	329
GA	2837	1427	637	831	418	187	852	909	804
IA	254	91	112	74	27	33	465	1102	376
IL	2508	1111	576	735	326	169	924	1279	765
IN	2222	568	1079	651	166	316	1460	1828	1356
KS	398	125	173	117	37	51	684	1125	631
KY	2436	675	1533	713	198	449	1733	2586	1617
LA	1526	1037	397	447	304	116	837	923	824
MD	1033	532	162	303	156	48	832	960	687
MI	1696	803	632	497	235	185	781	1024	676
MN	235	76	92	69	22	27	262	566	213
MO	1686	687	685	494	201	201	1218	1802	940
MS	1039	835	176	304	244	52	878	992	662
NC	2748	1205	768	805	353	225	938	1057	960
NE	129	30	57	38	9	17	360	727	368
NJ	808	231	155	237	68	45	494	688	399
NM	150	6	26	44	2	8	179	225	207
NV	40	17	7	12	5	2	48	136	41
NY	1975	370	657	579	108	193	438	372	505

OH	3315	1534	1479	971	449	433	1212	1542	1102
OK	930	221	303	272	65	89	715	1050	575
OR	34	2	17	10	0	5	42	36	43
PA	2374	989	764	696	290	224	881	1293	705
RI	54	7	8	16	2	2	280	270	257
TN	1838	751	709	538	220	208	990	1185	854
TX	5378	1551	690	1575	455	202	526	1087	513
UT	24	7	12	7	2	4	48	125	50
VA	1216	409	440	356	120	129	748	616	840
WA	64	15	23	19	4	7	47	87	48
WI	911	554	247	267	162	72	843	1988	528
WV	494	41	392	145	12	115	1087	724	1097

Table 4: Asthma exacerbation Cases averted (above poverty line)

State	Asthma exacerbation			standard deviation			Asthma exacerbation averted per 100,000		
	All	Blacks	Whites	All	Blacks	Whites	All	Blacks	Whites
AL	5079	1658	2634	1488	486	772	668	869	569
AZ	833	30	298	244	9	87	59	59	54
CA	2700	201	560	791	59	164	39	63	35
CT	1367	256	601	401	75	176	250	420	235
DC	470	215	121	138	63	35	487	778	372
DE	930	381	401	272	112	117	582	962	482
FL	7538	2246	3049	2208	658	893	226	357	229
GA	10191	4152	3998	2985	1216	1171	547	789	451
IA	1622	155	1229	475	45	360	306	615	300
IL	12824	2839	6254	3757	832	1832	638	1118	604
IN	9906	1860	6537	2902	545	1915	877	1495	815
KS	2605	369	1645	763	108	482	482	997	457
KY	7459	1232	5465	2185	361	1601	1035	1745	979
LA	3518	1697	1630	1031	497	477	485	745	399
MD	5540	2444	1906	1623	716	558	519	807	437
MI	7922	1818	4562	2321	533	1336	550	877	489
MN	1806	226	1237	529	66	362	178	337	167
MO	7096	1599	4465	2079	468	1308	685	1326	584
MS	2156	1183	904	632	347	265	470	710	364
NC	10239	2422	5289	3000	709	1549	542	624	524
NE	811	100	520	237	29	152	227	553	213
NJ	4620	954	1587	1354	280	465	334	574	285
NM	648	26	133	190	7	39	164	351	163
NV	221	24	75	65	7	22	43	78	38
NY	7459	1406	3280	2185	412	961	263	361	253

OH	12633	3473	8273	3701	1017	2423	722	1436	656
OK	4149	695	2052	1216	204	601	583	1212	510
OR	223	6	119	65	2	35	31	36	29
PA	9176	1945	5575	2688	570	1633	492	883	461
RI	369	45	204	108	13	60	245	406	239
TN	7249	2106	4238	2123	617	1242	605	1040	535
TX	27149	5665	8635	7953	1660	2530	453	1005	468
UT	339	11	237	99	3	70	43	74	42
VA	7056	2113	3366	2067	619	986	461	657	437
WA	509	43	272	149	12	80	37	59	37
WI	4194	739	2416	1229	216	708	431	979	359
WV	1710	81	1465	501	24	429	661	711	638

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Chapter 4: Saving the Salmon: Examining the Cost-Effectiveness of Collaboration in Oregon

Introduction

Environmental collaboration, in which a group of diverse stakeholders work across their boundaries to restore, protect, and otherwise govern the commons and resolve shared dilemmas, has become an increasingly common approach to the management of natural resources (Conley and Moote 2003; Heikkila and Gerlak 2005; Karkkainen 2002; Koontz et al. 2004, 2020; Lubell 2004; Ostrom 1990; Wondolleck and Yaffee 2000). The logic is simple: "Collaboration can lead to better decisions that are more likely to be implemented and, at the same time, better prepare agencies and communities for future challenges. … it is a means to several ends: building understanding, building support, and building capacity" (Wondolleck and Yaffee 2000, 23). Moreover, because collaboration gives the government more legitimacy in the eyes of citizenry (Spear, Huybrechts, and Nicholls 2013), it can reduce conflict, particularly over property rights (Buckles 1999). Likewise, because collaboration requires dialog and repeated interactions, actors can build trust and develop a shared understanding, which in turn can lead to cost-effective, environmentally sustainable solutions for collective action problems (Berkes 2007).

Today, environmental collaboration is widespread and myriad approaches exist, ranging from comparatively simple, short-term or temporary, project-based collaborations to more sophisticated, longer-term or sustained collaborative governance regimes (CGRs). As environmental collaboration has grown in use, so too have scholars and practitioners become interested in how collaborative structures impact performance. However, despite calls for the evaluation of collaborative performance generally (e.g., Emerson and Nabatchi 2015a; Koontz and Thomas 2006; Newig and Fritsch 2009; O'Leary and Bingham 2003; Thomas and Koontz 2011; Thomson, Perry, and Miller 2009), and in environmental settings specifically (Hardy and Koontz 2008; Koontz and Newig 2014; Scott 2015, 2016), little research explores the connections between collaborative structures and performance. Moreover, although scholars have produced a multitude of single case studies (e.g., Koontz et al. 2004; Lubell 2002; Ulibarri 2015; Wester, Minero, and Hoogesteger 2011) and several comparative case studies (e.g., Clarke and Fuller 2010; Heikkila and Gerlak 2005; Ostrom 1990; Ulibarri and Scott 2017), they have conducted few large-sample analyses (Scott 2015).

I address this gap by assessing the impact of several structural variations on the cost-effectiveness of over 958 collaborative watershed restoration projects. Specifically, I draw on a unique dataset from the Oregon Watershed Enhancement Board (OWEB) to examine how variations in collaborative structures, including collaboration form, number of participants, representational diversity among participants, and resource contributions, affect the cost-effectiveness of projects. To this end, in this article I first review existing literature and present hypotheses about the relationship between collaborative structures and cost-effectiveness. Next, I explain the methods of analysis, including the data and key variables. Finally, I present the results and discuss their relevance and implications for collaborative governance.

Environmental Collaboration

Collaboration literally means to work together, or co-labor. It involves the "pooling of appreciations and/or tangible resources (e.g., information, money, labor) by two or more stakeholders to solve a set of problems which neither can solve individually" (Gray 1985, 912). Collaborative governance refers to multiorganizational arrangements in which actors work across boundaries—whether sectoral, jurisdictional, geographic, or otherwise—to address public problems that cannot be easily addressed by a single organization or a single sector alone (e.g., Agranoff and McGuire 2003; Ansell and Gash 2008; Carlson 2007; Emerson and Nabatchi 2015b). Today, collaborative approaches to addressing public issues can be found in nearly every policy arena, but “[n] owhere has the collaboration trend been more evident than in the environmental sector” (Koontz and Johnson 2004, 186), where collaborative efforts are “broadly promoted as promising ways to deal with complex and contentious natural resource issues” (Conley and Moote 2003, 371; see also Jager et al. 2020; Koontz, Jager, and Newig 2020; Mandarano 2008). Environmental collaboration is particularly common in the domain of watersheds (Benson et al. 2013; Hardy and Koontz 2008; Lurie and Hibbard 2008), where the assumption is that local people have a better understanding of the context and can use their knowledge to make more optimal environmental decisions (Berkes 2004). As collaboration builds legitimacy (Spear, Huybrechts, and Nicholls 2013), it also helps achieve environmental justice (Dobbin and Lubell 2021; Lee 2002; Mendez 2020).

As the practice of collaboration has expanded, so too has its evaluation—“hundreds if not thousands of studies have investigated collaborative conservation efforts around the world” (Koontz, Jager, and Newig 2020, 443). Researchers have examined process characteristics and criteria (e.g., inclusiveness, representation, decision-making methods, accountability), outputs (e.g., plans, policy recommendations, agreements), social and behavioral outcomes (e.g., trust, economic development, networking, social learning), and environmental outcomes (e.g., the implementation of collaborative outputs and their effects on social and ecological conditions) (Conley and Moote 2003; Koontz, Jager, and Newig 2020; Mandarano 2008). However, not all collaborations are alike, and more research is needed to assess how variations in collaborative structures affect performance. As Koontz and Johnson (2004, 189) note, “[i]f systematic differences exist in the types of groups most appropriate for different situations, then we should be able to detect such differences through empirical analysis” (Koontz and Johnson 2004, 189).

Collaborative structures, that is, the organization or configuration of collaborative arrangements, can vary across a number of dimensions, including collaboration form, sponsors and conveners, collective purpose, locus of action, geographic scale, participant selection and recruitment, nature and locus of decision-making, and degree of formalization, among others (for a discussion, see Emerson and Nabatchi 2015b; see also Bryson, Crosby, and Stone 2006; Huxham and Vangen 2005; Margerum 2011). These variations likely account for at least some of the differences in process, outputs, and outcomes identified in research.

In this article, I investigate the relationships among four aspects of collaborative structure (collaboration form, number of participants, representational diversity, and resource contributions) on cost-effectiveness. While some studies have examined the links between group membership and results (e.g., Bidwell and Ryan 2006; Koontz and Johnson 2004; Korfmacher 2000; Moore and Koontz 2003; Steelman and Carmin 2002), I found no studies that focused on these other aspects of collaborative structure. Moreover, I found no studies that examined cost-effectiveness vis-à-vis collaborative structure. This deficiency is surprising as many advocates claim that collaborative approaches to environmental management (and other policy problems) are more cost-effective than traditional, hierarchical approaches (e.g., Emerson, Nabatchi, and O'Leary 2017; O'Leary and Bingham 2003; Prager 2015), and some research backs them up (e.g., Bodin et al. 2020; Booher 2004; Imperial and Hennessey 2000; McDonald et al. 2020).

Collaboration Form

Collaboration can take many forms. In this study, I focus on cross-boundary collaboration, a generic term that describes "the activity of collaboration among people from different organizations, sectors, or jurisdictions" (Emerson and Nabatchi 2015a, 16; see also Agranoff and

Rinkle 1986; Bardach 1998; Thomas 2002). In particular, I examine two types of crossboundary collaboration: (1) *ad-hoc* collaborations and (2) collaborative governance regimes (CGRs).

We define *ad-hoc* collaborations as collaborative efforts created or done for a particular end or goal or formed or used for a specific or immediate problem or circumstance without consideration of wider application or sustained interaction. An *ad-hoc* collaboration is one of the simplest types of cross-boundary collaboration. It forms in response to an immediate need or instrumental purpose and is temporary, existing only for the duration of a well-defined, narrow and bounded, one-off project. Once a project is fully executed, the *ad-hoc* collaboration is over, although a successful project could spur additional efforts or more sophisticated forms of collaboration.

In the context of watershed restoration, for example, an *ad-hoc* collaboration might form to address issues pertaining to non-native invasive species. Landowners and local government, perhaps with the support of a local environmental nonprofit, might come together to remove an invasive plant in a particular riparian zone that crosses several private and public property lines. Given the multiple properties involved, none of the actors can remediate the problem on their own, but once the plant has been removed and the project is complete, the actors disband, and the collaboration is over.

In contrast, a CGR is a system in which cross-boundary collaboration represents the predominant mode for conduct, decision-making, and activity between autonomous participants who have come together to achieve some collective purpose defined by one or more target goals" (Emerson and Nabatchi 2015b, 18). This approach is one of the most complex types of crossboundary collaboration. A CGR is a governing arrangement, a system "imbued with a set of explicit and implicit principles, norms, rules, and decision making procedures" during which participants engage in repeated interactions sustained over the longer term (Emerson and Nabatchi 2015b, 18–19). Moreover, a CGR typically addresses multifaceted problems that require a well-developed theory of change, replete with multiple, interconnected projects.

In the context of watershed restoration, for example, a CGR may involve actors from state and local government, nonprofit and community organizations, private landowners, and private companies. Together, the actors in this CGR may conduct a needs assessment for an entire watershed, identifying and prioritizing problems, and developing a strategic plan for addressing issues over the longer term. As a first step, they might engage in a riparian revegetation project, planting trees, shrubs, and other flora to restore natural habitats, improve and maintain water quality, and prevent erosion. As this work is being done (or once it is complete), the CGR may move on to another project, such as installation of a fish ladder to next to a dam to assist with spawning and migration. In short, as collaborative actions are taken, the CGR continues to work in a sustained and organized way to address watershed threats, restoration, and maintenance issues, which often are identified through assessment or planning processes.

Theoretical and empirical research suggests that the quality of relationships among partners is the key to collaboration (Huxham and Vangen 2005; Leach and Sabatier 2005; Margerum 2011; Ostrom 1998). Collaborative relationships that are built on respect and trust, that foster fair and civil discourse, and encourage innovation and open information exchange help build social capital among the parties (Koppenjan, Koppenjan, and Klijn 2004). In turn, social capital provides a foundation for the sharing and leveraging of scarce resources, such as funding, equipment, technical, logistical, or administrative support, and analytic or other expertise (Ansell and Gash 2008; Emerson and Nabatchi 2015b; Emerson, Nabatchi, and Balogh 2012; Thomson and Perry 2006). Some literature explores how collaborative governance can foster cost-saving, economies of scale, and effectiveness of both time and resources (Hoornbeek, Beechey, and Pascarella 2016; Kim and Darnall 2016; Lindsay et al. 2021; Mitchell, O'Leary, and Gerard 2015).

Given their nature, *ad-hoc* collaborations (which are shorter in duration and tend to have site-specific projects) are less likely to build social capital, and therefore leverage resources, than CGRs (which are longer in duration and tend to have systems-level goals). This leads to the first hypothesis, which is about collaboration form:

Hypothesis 1: Projects convened by collaborative governance regimes will be more cost-effective than projects convened by *ad-hoc* collaborations.

Number and Representational Diversity of Participants

The relationship between the size of a collaboration and the outcomes it produces has received limited theoretical and empirical attention in the literature. Newig et al. (2018) theorize five clusters of causal mechanisms in the relationship between participation and environmental outcomes, including (1) opening up of decision-making to environmental concerns, (2) incorporation of environmental relevant knowledge, (3) group interaction, learning, and mutual benefits, (4) acceptance and conflict resolution for implementation, and (5) capacity building for implementation and compliance.

Empirically, some studies have examined the links between group membership and results (e.g., Korfmacher 2000; Moore and Koontz 2003; Steelman and Carmin 2002). For example, a study of watershed groups in Ohio found that group composition affected a number of outputs—those "with a broader array of participants tend to excel in watershed plan creation, identifying/prioritizing issues, and group development and maintenance" while those with "narrower membership … focus more on pressuring government for policy change" (Koontz and

Johnson 2004, 185). A study of Oregon's watershed partnerships found several relationships between group composition (i.e., organizational affiliations) and group activities, strategies for prioritizing action, and outcomes (Bidwell and Ryan 2006). A meta-analysis of collaborative watershed partnership studies revealed that while diverse, broad membership contributed to success in many cases, in other cases such breadth created serious problems (Leach and Pelkey 2001). Finally, a study using agent-based modeling found that more participants lead to a decrease in the expected level of group agreement (Scott, Thomas, and Magallenes 2019).

To my knowledge, scholars have not assessed the effect of group size and composition on cost-effectiveness, though research suggests having "an optimal number of collaborators changes evaluation of the collaborative output" and that "the investment of more resources enhances the quality of task outputs" (Maglio et al. 2020, 241, citations omitted). Knowledge, "the currency of collaboration" (Emerson and Nabatchi 2015b, 71), is a key resource in collaborative governance and the lynchpin in the next two hypotheses.

Knowledge, or more accurately, lack of knowledge and uncertainty, drives the formation of a collaborative endeavor and shapes its performance over time (Emerson and Nabatchi 2015b; Emerson, Nabatchi, and Balogh 2012; Hurlbert and Gupta 2015; Newig et al. 2018). Hence, many scholars assert that collaborative arrangements will be more effective when the participants are sufficiently knowledgeable about the issue at hand (e.g., Beierle and Cayford 2002; Geissel 2009; Newig et al. 2018; Saint-Onge and Armstrong 2004; Sirianni 2009). This approach is particularly effective when the needed knowledge is diverse and specialized and the institutional frameworks are more complex, as is the case in environmental collaborations (Ansell and Gash 2008; Margerum 2011; Wondolleck and Yaffee 2000). Not only are more participants more likely to generate the requisite knowledge than fewer participants, but representational diversity (i.e., the types of organizations represented by participants) also is likely to matter. Representation not only helps "assure balanced voices, public buy-in, and legitimacy, but also generates a useful diversity of perspectives and ideas, which can lead to a deeper and more thoughtful and comprehensive consideration of issues and potential solutions" (Emerson and Nabatchi 2015b, 5).

However, while some assume that more is better, others offer a countervailing argument about size and diversity. Specifically, while there is the "belief that more collaborators help in producing a better output. … people intuit domain-dependent maxima that cap the extent to which more people strengthen a particular cause and, beyond which, those additional collaborators start to compromise it" (Maglio et al. 2020, 240). Moreover, representational diversity may "hamper social learning in collaborative governance settings … [It can be] threatening, leading stakeholders to react to new information defensively, which impedes knowledge assimilation and belief change and potentially thwarts collective action that could have resulted from shared understanding cultivated through the collaborative process" (Siddiki, Kim, and Leach 2017, 871).

Thus, the benefits of participant numbers and representational diversity may be subject to diminishing returns. While each additional participant might contribute a new and different kernel of knowledge that can make the project more cost-effective, at a certain point, the knowledge pool will become saturated, and "too many cooks will spoil the broth" (Maglio et al. 2020). Moreover, as the size of the group grows, so too do the challenges of resolving conflicts, which can lead to higher costs and less cost-effective work (Franks et al. 2014; Head 2008). Recognizing that an optimal figure or range of participant numbers and representational diversity likely exists, I offer the second and third hypotheses:

Hypothesis 2: The relationship between the number of participants in a project and cost-effectiveness will be quadratic, that is, cost-effectiveness will improve up to a point and then deteriorate with additional participants.

Hypothesis 3: The relationship between representational diversity in a project and cost-effectiveness will be quadratic, that is, cost-effectiveness will improve up to a point and then deteriorate with additional representational diversity.

Contributions of Resources

The potential for sharing and leveraging scarce resources is one of the most recognized benefits of collaboration (Emerson and Nabatchi 2015b; Thomson and Perry 2006). Indeed, the contribution of resources by participants has been shown to be of critical importance in collaborative arrangements (Provan and Milward 1995), including those in environmental settings (Yaffee and Wondolleck 2010). Beyond the obvious resource of cash funding, myriad types of in-kind contributions exist including meeting space; information, communication, and other technologies; materials and equipment; technical, administrative, and organizational assistance; and specialized expertise such as those needed for data gathering and analysis, pooling of internal resources by participants, and implementation functions (Emerson and Nabatchi 2015b).

Although such in-kind contributions are not without cost to the contributing organizations, they may reduce overall project expenses by limiting the need to purchase materials, services, and other items from external sources. Thus, collaboratives with more in-kind resources might be better prepared to implement projects cost-effectively since they are ready to hit the ground running and do not have to purchase—or have to purchase fewer—inputs before launch. Moreover, in-kind donations may represent a substantial and substantive commitment to the collaborative endeavor and nurture a sense of ownership over projects (Lee and Restrepo 2015), which in turn may incentivize participants to ensure that projects are cost-effective. This sets the stage for the final hypothesis:

Hypothesis 4: Projects will be more cost-effective if participants contribute in-kind resources.

Methods

In this section, I explain the methods used to test the hypotheses. I begin with a brief discussion of the empirical setting. Next, I explain the data sources, analytical approach, and variables of interest.

Empirical Setting

The Pacific Northwest of the United States has several different species of salmon and steelhead with genetically distinct runs in different river systems. By the 1990s, decades of overfishing, degradation of watershed conditions, pollution, lack of regulation, and building of dams and other fish barriers threatened numerous salmonid species in the Pacific Northwest, resulting in the listing of several salmon and steelhead populations under the Endangered Species Act. Given that salmonid species are vital for area ecosystems and economies, the National Ocean and Atmospheric Agency (NOAA), in partnership with state and local actors, has been counting spawning salmon ("spawners") in 12 rivers of the Northwest United States since 1955. Figure 1 shows that the spawner population trended downward from the mid-1950s until the mid-1990s, at which point it began to increase. Many scholars and practitioners attribute this turnaround to changes in state and federal strategies for environmental and species conservation (e.g., Ford et al. 2011; Katz et al. 2007; Morishima and Henry 2020).

Specifically, in the mid-1990s, Oregon's legislature recognized that native salmonid species contributed to environmental benefits, cultural values, and economic gains, and that to revitalize and protect the salmon, attention had to be given to the aquatic ecosystems in which they lived[5]. The Oregon state legislature and then Governor John Kitzhaber pushed to protect the fish through the use of environmental collaboration between different stakeholders, including public agencies, tribal governments, private businesses, nonprofit organizations, citizen groups, and others (OWEB n.d.c).

In 1997, the state passed legislation that implemented the Oregon Plan for Salmon and Watersheds (OPSW) which, in part, provided formal mechanisms for local governments to establish watershed councils that were to operate as CGRs with the goal of "restoring land and water from 'ridgetop to ridgetop'" (Network of Oregon Watershed Councils n.d.; see also OWEB n.d.c). The state also created the Oregon Water Enhancement Board (OWEB), a public agency authorized to administer dedicated funds for fish and wildlife restoration and protection and provide grants "to help protect and restore healthy watersheds and natural habitats that support thriving communities and strong economies" (OWEB n.d.a).

Among OWEB grants are those supporting collaborative watershed restoration projects that aim to improve watershed functions, water quality, or fish habit and show "direct evidence of collaboration between stakeholders and agencies" (as opposed to single party projects) (OWEB n.d.b). In other words, these grants require environmental collaboration. OWEB encourages diverse collaborations, both in terms of numbers and representation, asserting that diversity can generate greater impacts (Arnold 2017). OWEB awards these grants based on the collaborative

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[5] 1 Watersheds, areas of land where precipitation collects and drains off into a common outlet such as a river or lake (Anderson and Anderson 2010), are critical to the wellbeing and survival of salmon. Salmon spawn their eggs in upstream watersheds, and these salmon eggs hatch in shallow streams. The young fish spend 1–2 years in the watersheds. This time is crucial in the life of the fish, and health of the watershed in terms of water temperature, water quality, and nutrient availability dictates their survival. After a couple years, the fish physiologically adapt to saltwater and journey downstream to spend their adult lives in the ocean. After 2–5 years in the ocean, they return upstream—back to the watersheds—to lay their eggs. Barriers in the river and high-water velocity impede the upstream migration of the fish. Thus, once again, watershed health influences the survival of the salmon.

and technical merits of the proposed projects. Since its creation, OWEB has dispersed thousands of grants to support collaborative watershed restoration projects, some of which were conducted by watershed councils (i.e., CGRs) and others by multiorganizational, project-based collaborations (i.e., *ad hoc* collaborations).

Data Sources

The data for this project come from the Oregon Watershed Restoration Inventory (OWRI). OWRI, managed by OWEB, is easily the largest watershed restoration database in the United States, with information for over 150 different kinds of projects aimed at restoring habitat and improving watershed conditions in Oregon. Most projects are organized by the habitat in which they are implemented (i.e., riparian, upland, wetland, and estuary), while some projects are tracked by priority activity (e.g., fish passage or road improvements). OWRI provides tabular and spatial information on projects, including finances, collaborators, environmental outputs, and timelines. It is intended to support watershed assessment and future restoration planning and provide a public platform for monitoring regional and statewide restoration efforts.

OWRI includes more than 19,000 projects in total. Among these are 7,057 projects funded by OWEB and carried out between 1997 and 2017. The scale of watershed efforts across the state is highlighted in Figure 2, which shows the areas in Oregon covered by watershed councils (in teal)[6] and the areas that have been home to a restoration project (in dark green). It is within this context that I test the hypotheses.

OWEB requires that funded restoration projects report their data to OWRI (reporting is voluntary for projects not funded by OWEB), and OWEB staff review the data to ensure its

[6] Watershed council jurisdictions do not align with county jurisdictions because watershed councils are bound by hydrological unit “watersheds” rather than by county lines.

accuracy. In this study, I focus on the OWEB funded projects because they have been subjected to review by OWRI staff.

Many of the projects in OWRI have multiple activities, and consequently have more than one type of output variable. For example, a single project could report two watershed restoration treatments: (1) riparian trees planted, where the output variable is reported in acres, and (2) culverts removed, where the output variable is reported in instream miles made accessible. However, the database does not provide information on how resources are allocated between the treatments within a single-activity type, so we only focus on single-activity projects. This reduced the sample to 4,451 projects. From these I selected the two broad categories of projects with the largest number of individual entries: (1) riparian planting projects (n = 659) and (2) instream barrier removal projects (n = 393). Each project category is discussed briefly below.

Riparian planting projects aim to improve habitat by restoring riparian zones through the planting of trees[7]. Riparian zones are streambanks and areas adjacent to a river. Healthy riparian zones with appropriate tree and plant species can improve habitat through variety of mechanisms. Trees and plants in the riparian zones provide shade, which lowers the temperature of the stream and makes it more hospitable for native fish. They create a natural boundary between upland areas and the stream, which helps filter pollutants and makes the water cleaner for the instream species. They also help reduce stream bank erosion and maintain stable stream channel geomorphology. Additionally, old, large trees fall into the stream and provides habitat, by decreasing the water velocity and creating features that fish prefer to fulfill their life cycle.

[7] Included among the riparian planting projects in the database are those that planted hardwood trees (n = 77), conifer trees (n = 64), conifer and hardwood trees (n = 513), and other trees (n = 5).

Instream barrier removal projects involve replacing barriers like culverts and fords to make access and passage easier for fish[8]. The removal of instream barriers is particularly important in the salmon breeding season, when salmon swim upstream to access spawning grounds. These barriers also can impede access downstream. The negative impact of fish passage barriers and instream infrastructure on natural hydrology and habitat are well known—barriers obstruct the flow of water and route of migratory fishes, reptiles, and water-based mammals (O'Hanley et al. 2013). For this reason, many of the programs aimed at improving habitat for in-stream fish try to remove or change the shape of barriers to make passage easier.

Project Characteristics

OWRI contains numerous variables for each project. Below I describe the dependent variable (cost per output), independent variables (collaborative form, number of prior projects, participant count, representational diversity, and in-kind share), and control variables used in the analysis and present summary statistics for the sample.

Cost per output is the quotient of cost divided by output and is normalized across the two different project types. The total cost of a project is the sum of the OWEB grant, the in-kind monetary contributions (i.e., cash), and the in-kind nonmonetary contributions (e.g., equipment, labor, technical expertise, materials). All costs are in 2017 dollars. Output is the measure of area treated. For riparian planting projects, output is measured in acres, and for barrier removal projects, output is measured in miles. I recognize that all acres of restoration and all miles of accessible stream are not equivalent (e.g., an acre located directly on water may be more meaningful than an acre far removed from the stream, and a mile of a larger stream with greater salmon habitat

[8] Included among the instream barrier removal projects in the database are those where culverts/structures/fords were replaced with bridges (n = 141), with embedded culverts (n = 174), with fords (n = 2), with open bottom arch culverts (n = 69), and with weir/baffle culverts (n = 7).

potential may be more meaningful than a mile of a smaller stream with less salmon habitat potential). However, the data do not allow me to capture such nuances and normalizing output by cost helps to randomize the measurement error between projects of different sizes.

Collaborative governance regime captures collaboration form. To code this variable, I categorize all grant projects led by a watershed council as taking the form of a CGR, and all other grant projects as taking the form of an *ad-hoc* collaboration. Thus, this is a binary variable, where "1" indicates that the project was executed by a CGR (i.e., a watershed council), and "0" indicates that the project was executed by an *ad-hoc* collaboration. While this is a simplified bifurcation, it captures a key distinction of interest: a formalized, "official," and sustained collaborative effort versus an informal, extemporaneous, and temporary collaborative effort. Of course, this distinction does not capture the quality of collaboration within the two forms—some *ad-hoc* collaborations may be more collaborative in practice than some watershed council CGRs—however, it does speak to the nature of the collaborative form. Moreover, OWEB staff supported this coding approach for the purposes of this analysis.

Number of prior projects is the count of OWEB-funded past projects completed by each CGR and each *ad-hoc* collaboration. Number of prior projects, in part, captures the experience collaboratives may have working on watershed projects, and such experience may relate to cost-effectiveness. I also include an interaction term between collaborative form and the number of prior projects completed.

Participant count (PC) is the number of collaborators in the project. Each collaborator is a distinct organization. I include both linear and quadratic measures of participant count.

Representational diversity is the number of different types of organizations represented by participants involved in the project. OWEB categorizes organizations into eight broad types:

citizen groups (including nonprofits), federal government agencies, state government agencies, local/ city/county government agencies, tribes, private industrial companies, private nonindustrial companies, and other (e.g., educational institutions, volunteers, and extension services). Like participant count, I allow for a quadratic relationship between cost-effectiveness and the representational diversity of participants. I expect a correlation between participant count and diversity and discuss the risk of confounding later.

In-kind share is the ratio of nonmonetary in-kind contributions to the project's total cost. I use in-kind contributions because the dataset neatly distinguishes between nonmonetary in-kind contributions and monetary expenses for any particular project. OWEB encourages implementing organizations to have diverse sources of funding for their project beyond the grant (Arnold 2017). I recognize that organizations may inflate their self-reported estimates of in-kind contributions, however, OWEB questions grant applicants at initial stage if the in-kind numbers are not reasonable and flags projects that report unreasonable numbers.

Table 1: Descriptive Statistics for Riparian Planting and Barrier Removal Projects

		Riparian Planting Projects		Barrier Removal Projects	
		Mean	Standard Deviation	Mean	Standard Deviation
Dependent variables (and related measures)	Cost per output	12,625	64,854	105,599	267,420
	log(Cost per output)	10.6	1.27	4.61	0.55
	Total cost	7,090	18,454	117,165	173,321
	Output	2.2	7.7	2.9	4.5
Independent variables	CGR	0.75	0.42	0.62	0.48
	No. prior of projects	68.4	51.4	5.8	19.0
	Participant count	8.7	3.8	4.8	1.5
	Representational diversity	4.5	1.4	2.7	0.95
	In-kind share	0.32	0.19	0.14	0.18
Control variables	Duration in months	15.5	12.6	4.4	7.8
	Goals	4.5	2.2	1.5	1.3
	Public	0.067	0.25	0.42	0.49
	Complimentary collaborations	54.8	44.5	4.0	3.7
	Permit			0.75	0.43
	No. observations	594		364	

I also include a number of control variables that are likely to affect costs. Duration captures the length of project in months. Longer projects may require more resources than shorter projects. Goals measures the number of aims associated with project completion. Projects with more goals may be more challenging and resource intensive than projects with fewer goals. Complementary collaborations are a count of sister projects done in an OWEB grant. Grants with multiple similar projects may achieve economies of scale and make projects more cost-effective. Public is a binary variable, where a project done on public land is coded 1, and 0 otherwise[9]. Projects on public land may face more regulation and bureaucratic oversight and therefore require more resources that projects on private land. Permit is a binary variable, where a project requiring a permit is coded 1, and 0 otherwise. Projects that require permits may have higher technical and financial needs than projects that do not require permits. In the data, only some barrier removal projects required permits. Riparian planting projects do not require permits.

Descriptive statistics for the variables are shown in Table 1. Several observations are worth noting. First, the projects have large cost differences. The mean cost of barrier removal projects ($117,165) is more than 16 times higher than the mean cost of riparian planting projects ($7,090). Similarly, the average unit cost for barrier removal projects ($105,599 per stream mile) is more than eight times higher than the average unit cost for riparian planting projects ($12,625 per acre). Barrier removal projects typically require more machinery, materials, and engineering and construction expertise than riparian planting.

[9] Inclusion of this variable required us to drop projects that were done on both public and private land. To test the impact of this restriction, I removed this variable and reran the regression on the larger set of observations. This increased our sample to 610 riparian planting projects and 379 instream barrier removal projects but did not meaningfully change the results.

Second, the projects have scope and duration differences. Riparian planting projects tend to have a higher participant count than barrier removal projects (8.7 organizations compared to 4.8 organizations), as well as more representational diversity (4.5 organizational types compared to 2.7 organizational types)[10]. This pattern is likely because the many actors who directly and indirectly affect the health of a riparian zone, including farmers, cattle ranchers, home or land owners, local communities, businesses, government agencies, and others, need to be engaged in the project. Similarly, riparian planting projects last an average of 15.5 months (standard deviation 12.6) and barrier removal projects last an average of 4.5 months (standard deviation 7.8). Riparian planting projects tend to last longer because it may take one season to prepare the site and another to nurture the seedlings (Withrow-Robinson, Bennett, and Ahrens 2017).

Third, the partners in the project have different collaborative experience in terms of the number of prior projects completed and complementary collaborations. Specifically, those engaged in riparian planting projects have, on average, completed 68.4 prior projects and 54.8 complementary collaborations, whereas those engaged in barrier removal projects have, on average, completed 5.8 prior projects and 4.03 complementary collaborations. Together, these factors suggest that riparian planting projects may have unique collaborative needs compared to barrier removal projects.

Fourth, the standard errors for both types of projects are large, suggesting that the data are skewed, with many projects being quite small with very little output and cost and others being

[10] To the check the robustness of our findings, I tried alternate specifications. I found a high correlation (0.70) between participation count and diversity, which raises concerns of spurious association with the independent variable or simple confounding. To test for confounding, I reran the regression without representational diversity. I did not find meaningful differences in the results.

quite large[11]. Moreover, some projects are community funded with limited support from OWEB and others are completely funded by OWEB. Some have more experienced leadership and greater representative diversity, while others have less. This variation in scale of the projects generates large standard errors for the descriptive statistics of some of the variables. To correct for this skewness, I take the log of the dependent variable. For the logged dependent variable, the range of standard deviations reduces substantially with respect to the mean (Table 1).

Finally, collaboratives that conduct riparian planting projects rarely do barrier removal projects (Figure 3). Riparian planting projects are clustered in a few watersheds along the coast and Willamette River Basin, while barrier removal projects are more evenly spaced across the western third of the state. The areas without projects are places that are (a) public lands, (b) sparsely populated, and/or (c) get limited precipitation (e.g., the arid areas east of the Cascade Mountains). Similarly, collaboratives that conduct barrier removal projects do fewer projects compared to those collaboratives that carry out riparian planting projects. These patterns also result in different descriptive statistics for the two project types.

Results

To test the hypotheses, I use the following regression model:

$$\ln(\text{Costperunit}_i) = \beta_0 + \beta_1 CG + \beta_2 CG * \text{Experience} + \beta_3 PC_i + \beta_4 PC_i^2 + \beta_5 \text{Diversity}_i + \beta_6 \text{Diversity}_i^2 + \beta_7 \ln(\text{Inkindshare}_i) + \beta_8 X + \gamma_b + \delta_t + \varepsilon_i$$

where i denotes each individual project, X is the set of control variables, γ_b is a vector of basin fixed effects, and δ_t is vector of time fixed effects. Basin fixed effects help control for differences

[11] This is likely from when a single OWEB grant funds a large-scale riparian planting effort in multiple watersheds or on many properties. OWRI reports at the worksite level, that is, each site should be reported as a separate OWRI project. Not all respondents follow this reporting protocol. This difference in the scale of reporting makes it difficult to compare similar activities.

in watershed geography and time fixed effects control for time-invariant characteristics. I use the log of the dependent variable, to provide a more meaningful interpretation of the regression coefficients. Specifically, logging allows the coefficients on independent variables to be interpreted as percentages, that is, a one unit change in the independent variable i leads to an x percent change in cost-effectiveness. The results are presented in Table 2. Columns 1 (for riparian planting projects) and 2 (for barrier removal projects) include fixed effects and are the preferred models. Columns 3 (for riparian planting projects) and 4 (for barrier removal projects) provide the results without fixed effects for comparison. I organize and review the results by hypothesis.

Table 2: Regression Coefficient Estimates for the Log of the Cost per Unit for Riparian Planting and Barrier Removal, With and Without Fixed Effects

	(1)	(2)	(3)	(4)
	Riparian Planting	**Barrier Removal**	**Riparian Planting**	**Barrier Removal**
Participant count (PC)	−0.469** (0.014)	−0.559** (0.020)	−0.899*** (0.000)	−0.548** (0.017)
Participant count2 (PC2)	0.037*** (0.003)	−0.054** (0.009)	0.064*** (0.000)	−0.056** (0.005)
Collaboration governance regime	−0.312 (0.103)	0.356** (0.017)	−0.103 (0.585)	0.362** (0.006)
CGR*No. of prior projects	−0.006*** (0.007)	−0.013** (0.015)	−0.009*** (0.002)	−0.016** (0.008)
Representational diversity	0.311 (0.540)	0.488 (0.161)	0.147 (0.799)	0.463 (0.168)
Representational diversity^2	−0.015 (0.844)	−0.080 (0.150)	0.051 (0.550)	−0.079 (0.150)
In-kind share	−1.630*** (0.000)	−1.112*** (0.002)	−2.130*** (0.000)	−1.23*** (0.000)
No. of prior projects	−0.001 (0.046)	0.0160***(0.003)	−0.001 (0.018)	0.018*** (0.002)
Duration	−0.036*** (0.000)	0.028*** (0.000)	−0.036*** (0.000)	0.029*** (0.000)
Goals	−0.061** (0.004)	0.08 (0.250)	−0.061** (0.041)	0.074 (0.217)
Complementary collaborations	−0.009** (0.342)	0.0137 (0.349)	−0.011*** (0.004)	0.0026 (0.878)
Public	0.294 (0.218)	0.77*** (0.000)	0.407 (0.153)	0.789*** (0.000)
Permit		0.183 (0.259)		0.218 (0.118)
Period FE	x	x		
Basin FE	x	x		
r^2	0.59	0.27	0.45	0.26
N	594	364	594	364

p values are reported in the parenthesis.
**Significance level $p < .05$.
***Significance level $p < .01$.

The first hypothesis posits that projects convened by CGRs will be more cost-effective than projects convened by *ad-hoc* collaborations. The CGR coefficient and the interaction term are jointly significant in all four models. Among riparian projects, those convened by CGRs are 31%

(standard error is 19%) more cost-effective than those convened by *ad-hoc* collaborations, and this cost-effectiveness increases with the number of prior projects executed. However, among instream barrier removal projects, CGRs are 35% (standard error is 14%) less cost-effective than *ad-hoc* collaborations. Moreover, the interaction term shows that CGRs become more cost-effective when the number of prior projects increases. Thus, the results for riparian planting are consistent with the first hypothesis, but the results for instream barrier removal are not.

The second hypothesis posits that the relationship between the number of participants in a project and cost-effectiveness will be quadratic, that is, cost-effectiveness will improve up to a level and then decrease with additional participants. Participant count (PC) and its quadratic term (PC2) are jointly significant for both riparian planting and barrier removal projects. To provide more interpretation of the quadratic coefficients, I plotted cost-effectiveness ($\hat{y}$) against the participant count for both project types[12]. The relationship for riparian planting projects is shown in Figure 4 and the relationship for instream barrier removal projects is shown in Figure 5. For riparian planting projects, the relation suggests that projects are most cost-effective with participation by eight organizations. The cost per output tends to be highest when participation is much lower (e.g., 3 organizations) or much higher (e.g., 13 organizations). On average, projects tend to be 16% more cost-effective when closer to the optimal number of participants than to either extreme. The difference is smaller for barrier removal projects, which are, on average, 4% less expensive with optimal participation compared to suboptimal participation. For barrier removal projects, the relationship suggests that projects are most cost-effective with participation by five to six organizations. The cost per output tends to be highest when participation is much lower than

[12] I ran a series of regression with different model specifications—linear and quadratic for participation and/or diversity—and computed Bayesian information criteria (BIC) for these regressions. For these alternate linear models, I found higher BIC scores, which suggests that the quadratic regressions offer a better fit.

2 organizations or much higher than 11 organizations. Overall, these results are consistent with the second hypothesis.

The third hypothesis posits that the relationship between representational diversity and cost-effectiveness will be quadratic, that is, cost-effectiveness will improve up to a level and then decrease with additional representational diversity. The results suggest a positive, albeit insignificant, relationship between representational diversity and cost-effectiveness. This pattern also suggests that the optimal amount of representational diversity for both types of projects is four types of organizations. Specifically, on average, riparian planting projects tend to be 20% more cost-effective when closer to the optimal amount of representational diversity (four different organization types) than to either extreme (Figure 6). Similarly, on average, barrier removal projects tend to be 4% more cost-effective when closer to the optimal amount of representational diversity (four different organization types) than to either extreme (Figure 7). However, as these results are not statistically significant, we reject the third hypothesis.

The fourth hypothesis posits that projects will be more cost-effective if participants contribute in-kind resources. As mentioned earlier, the dependent variable is log of cost per output, which allows for interpretation of the regression coefficient for in-kind share as a percentage change in cost-effectiveness. For interpretation, I estimate the regression coefficient with 10% nonmonetary in-kind shares, which simply is multiplying the regression output for in-kind share by a factor of 10. In the fixed effects model, riparian projects with 10% nonmonetary in-kind share are, on average, 16.3% more cost-effective than those without nonmonetary in-kind shares. Similarly, for barrier removal projects, those with 10% nonmonetary in-kind shares are, on average, 11.1% more cost-effective than projects without nonmonetary in-kind shares. The results for both project types show strong and significant relationships, which is consistent with the fourth hypothesis[13].

Discussion and Conclusion

In recent years, environmental collaboration has received considerable attention. Despite calls for research on collaborative performance, scholars have not examined how the structural characteristics of collaboration affect the cost-effectiveness of projects. This analysis begins to fill this gap. The results suggest that collaboration form, participant numbers, and share of in-kind contributions affect cost-effectiveness, but that representational diversity does not.

The results for the first hypothesis about collaboration form (CGR versus *ad-hoc*) diverge for the two project types. Specifically, watershed councils, which operate as CGRs, are more cost-effective than *ad-hoc* collaborations for riparian planting projects, but less cost-effective than *ad-hoc* collaborations for instream barrier removal projects. This result is likely a function of the differences between riparian planting projects and barrier removal projects, both in terms of technical and collaborative characteristics. More specifically, by their very nature, riparian planting projects take more time and involve more partners, and therefore tend to have more complexity, than do instream barrier removal projects. Another possible explanation may be that CGRs have more staffing in place to manage longer-term projects such as those involving riparian planting, while *ad-hoc* groups do not. However, many of the *ad-hoc* collaboratives are led by private companies, which have staff, though they may not be trained or have experience working with and executing multistakeholder collaborative projects.

[13] The data have measures for permit, species count, and number of barriers only for instream barrier removal projects. I ran the regressions with and without these controls. The results did not change.

Still, that *ad-hoc* collaborations generate more cost-effective barrier removal projects than CGRs, raises questions about the staffing explanation. Research suggests that as project complexity increases, so too does the need for social capital among the partners, and such social capital is likely to be better generated through a CGR than through an *ad-hoc* collaboration (Emerson and Nabatchi 2015b; see also Bryson, Crosby, and Stone 2006; Huxham and Vangen 2005; Margerum 2011). This interpretation leads to an interesting proposition: while many projects may require collaboration, not all projects necessarily require a CGR. In future research, scholars should attend to this proposition and explore which collaborative forms—whether CGR, *ad-hoc*, or otherwise— are best suited for what types of conditions. In the case of watershed restoration projects, those conditions may involve issues such as duration, scope, or technical complexity. These and other conditions may extend to other kinds of projects and activities.

The results for the second hypothesis suggest there is a sweet spot in terms of the size of a collaborative effort (cf. Franks et al. 2014; Head 2008; Maglio et al. 2020; Scott, Thomas, and Magallenes 2019). On the one hand, too few participants reduce cost-effectiveness, which may be due to the collaboration lacking in-kind resources or information that could have been provided by additional participants. On the other hand, too many participants also reduce cost-effectiveness, perhaps because of higher transaction costs or the need to satisfy too many demands. Interestingly, however, the optimal size of a collaboration depends on the kind of project being undertaken. My analyses show that the ideal size of riparian planting projects is eight participants, while the ideal size of instream barrier removal projects is five to six participants. This result raises an interesting question: what conditions contribute to the ideal size of a collaborative effort? Scholars should attend to questions of collaborative size and project conditions in future research.

The results for the third hypothesis suggest that representational diversity is unimportant, at least in terms of cost-effectiveness. Various scholars have asserted both the positives (e.g., Emerson and Nabatchi 2015b) and negatives (e.g., Siddiki, Kim, and Leach 2017; Ulibarri and Scott 2017) of greater stakeholder diversity in collaborative arrangements, and research finds that group composition can affect various outputs and outcomes (e.g., Bidwell and Ryan 2006; Koontz and Johnson 2004; Korfmacher 2000; Leach and Pelkey 2001; Moore and Koontz 2003; Steelman and Carmin 2002). I find neither an upside nor a downside effect in terms of the impact of representational diversity on cost-effectiveness. This finding is perhaps the most surprising result given the attention to stakeholder diversity in the literature.

Although I recognize that increasing representational diversity could lead to conflicts over problem definitions and preferred solutions, it also seems plausible that a diverse set of organizational types would provide access to a broader range of in-kind resources and information, which, if pooled, would result in more cost-effective outcomes. This pattern appears to be different from the cases with these watershed restoration projects. These results may be a function of the measure of representational diversity used, which may not reflect the actual heterogeneity of the collaborators. An OWEB report suggests that the board members of watershed councils have diverse, multisectoral backgrounds, which could enable them to bring diverse knowledge to these collaborative projects (Arnold 2018). Moreover, the findings might be due to missing variable bias (i.e., unmeasured belief conflicts that counter technical effectiveness). Regardless, this result does not imply that representational diversity is unimportant to process aspects or other substantive collaborative outcomes beyond cost-effectiveness. Thus, scholars should continue to explore the effect of different kinds of diversity on collaborative efforts.

Finally, the results for the fourth hypothesis show contributions of in-kind resources increase the cost-effectiveness of projects. This finding makes intuitive sense: in-kind resources are resources that do not need to be paid for out of grant funding. Individual participants are likely to want their contributed resources to be used in as cost-effective a manner as possible, which may improve the likelihood that projects are implemented efficiently and in a timely manner. One caveat is in order here: cost-effectiveness is not the only criterion that matters, and funders should be cautious about assuming that projects with in-kind resources are “better” than projects without such contributions. Assumptions about in-kind contributions, as well as other questions about funding and budgeting for collaborative arrangements, should be explored in future research.

Together, these findings provide valuable insights about the design of collaborative arrangements that should be useful both to those who are engaging in and those who are studying collaboration. First, and perhaps most obviously, design matters. The shape and structure of collaborative arrangements—collaborative form, size, and contributions—affect the cost-effectiveness of projects. The results seem to suggest that CGRs are more cost-effective for larger, longer, and more complex projects, while *ad-hoc* collaborations are more cost-effective for smaller, shorter, and simpler projects.

Second, scholars should explore further the connections between collaborative structures, processes, outputs, and outcomes. They should examine whether these findings hold in other natural resource projects, as well as in other policy sectors such as education, public safety, and health. They should look beyond form, size, representational diversity, and in-kind contributions, to examine other aspects of collaborative structure that are likely to matter, such as sponsors and conveners, leadership, mandates, and voluntariness, among others. They should examine how collaborative structures affect collaboration dynamics (Emerson and Nabatchi 2015b) and collaborative performance in terms of process and productivity (Emerson and Nabatchi 2015a). In doing so, they must examine measures beyond cost-effectiveness, including things like community and organizational capacity, social capital, and impacts on the ground, which arguably are the most important of all potential collaborative outcomes.

Finally, to produce the research described above, scholars must not only continue to generate single and small sample size cases studies (albeit with more attention to the details of structure, process, outputs, and outcomes), but also work toward building large-sample size datasets like the Atlas of Collaboration and Collaborative Governance Case Database. Indeed, improving our understanding of where, when, why, and how collaboration does (and does not) work requires multilevel, multisite, and multitype data that are only available in large datasets.

Such research is the key to advancing the practice of collaborative governance. As the use of collaborative arrangements in the environmental sector and in other policy areas grows, so too must we work toward identifying design characteristics and features that contribute to the achievement of specific outputs and outcomes. Simply stated: scholars should seek to understand which collaborative arrangements work best under what circumstances.

Acknowledgements:

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Figures

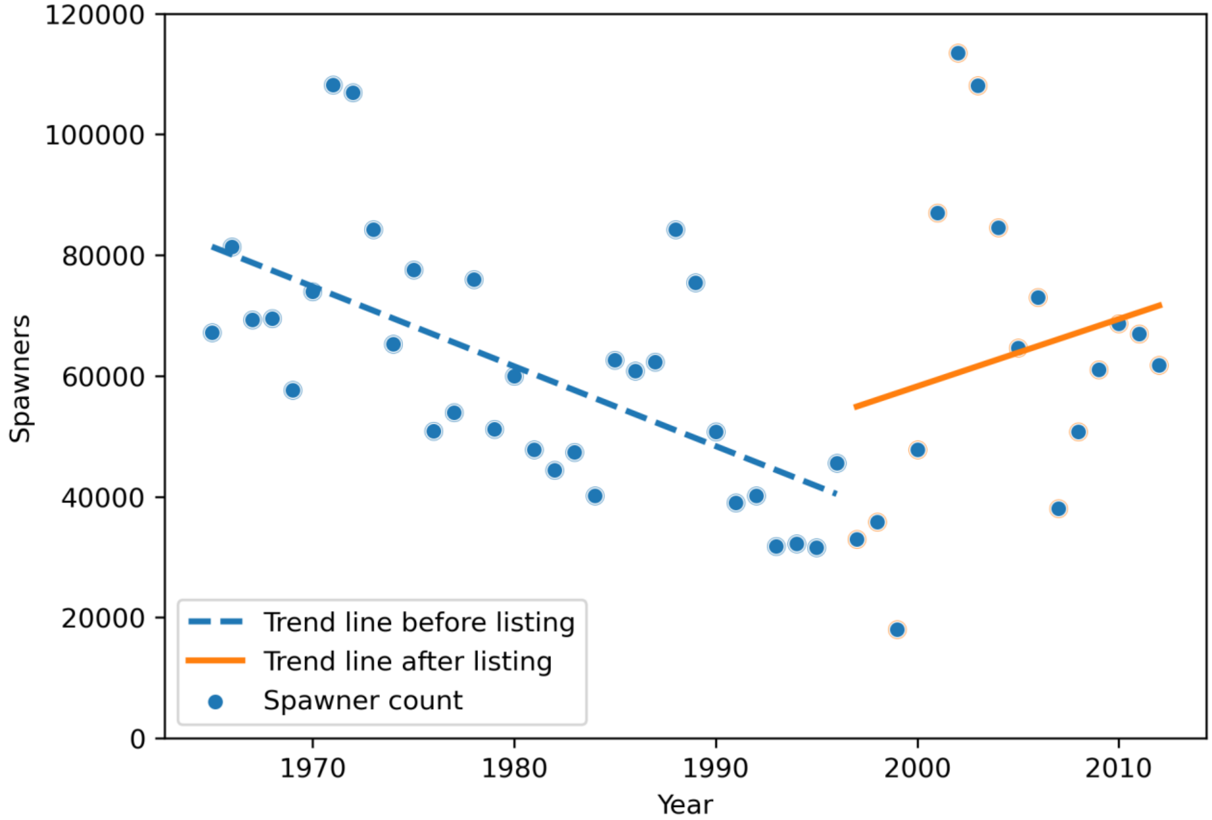

Figure 1: Time series data on the salmon spawner population in the Northwest United States. Note: This plot was generated with the NOAA Salmon Population Summary (SPS) dataset and includes Chinook Salmon, Coho Salmon, and Steelhead species. The last update, published in April 2016, has data through 2012 or 2014 depending on the salmon population of interest. A more current update was not available at the time of this analysis

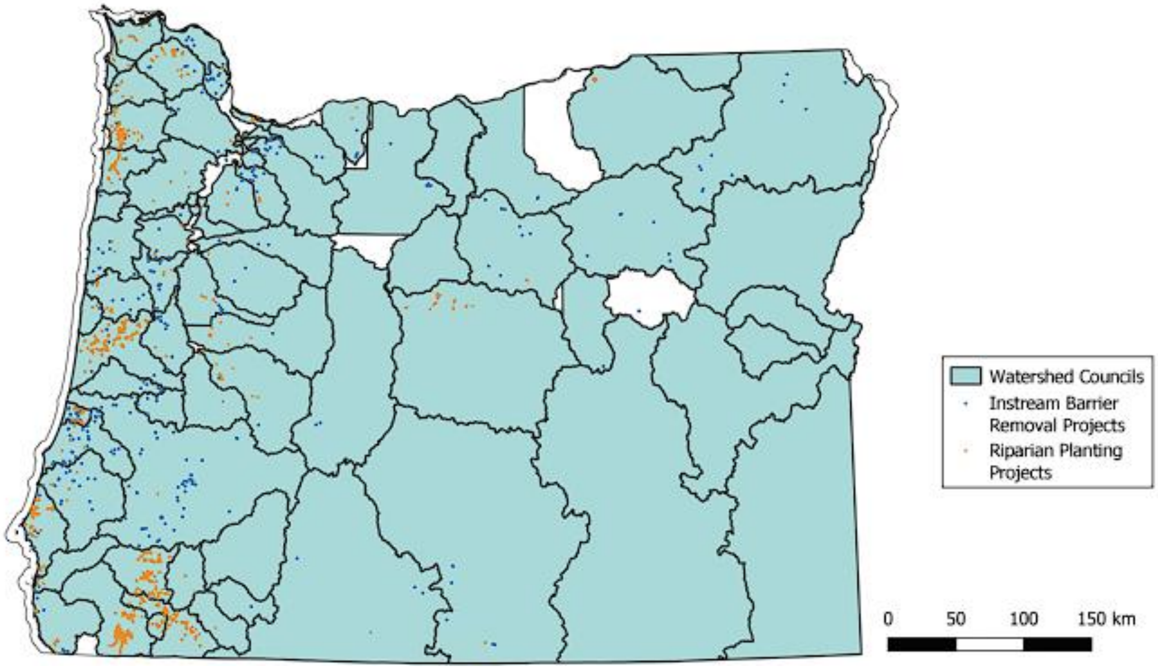

Figure 2: Geographic Distribution of Oregon Watershed Councils and Restoration Efforts

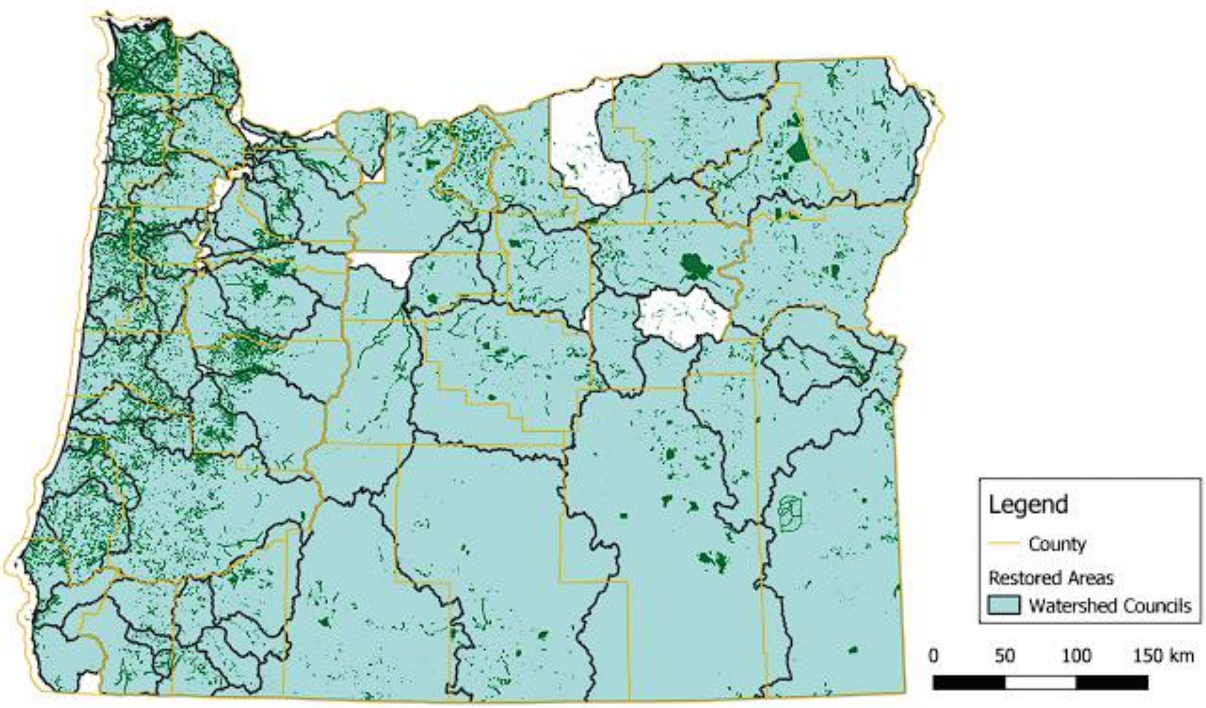

Figure 3: Geographic Distribution of Riparian Planting and Barrier Removal Projects.

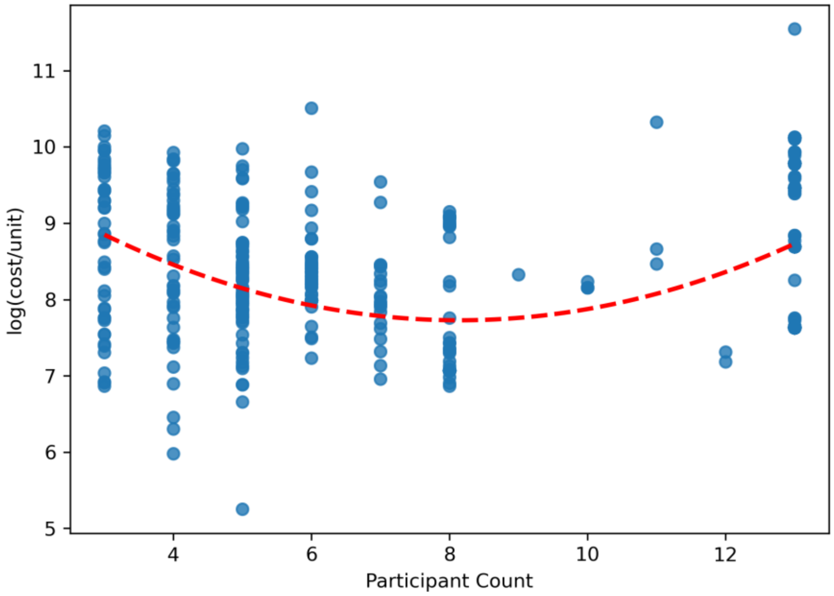

Figure 4: Log of unit cost as a function of participant count for riparian planting projects.

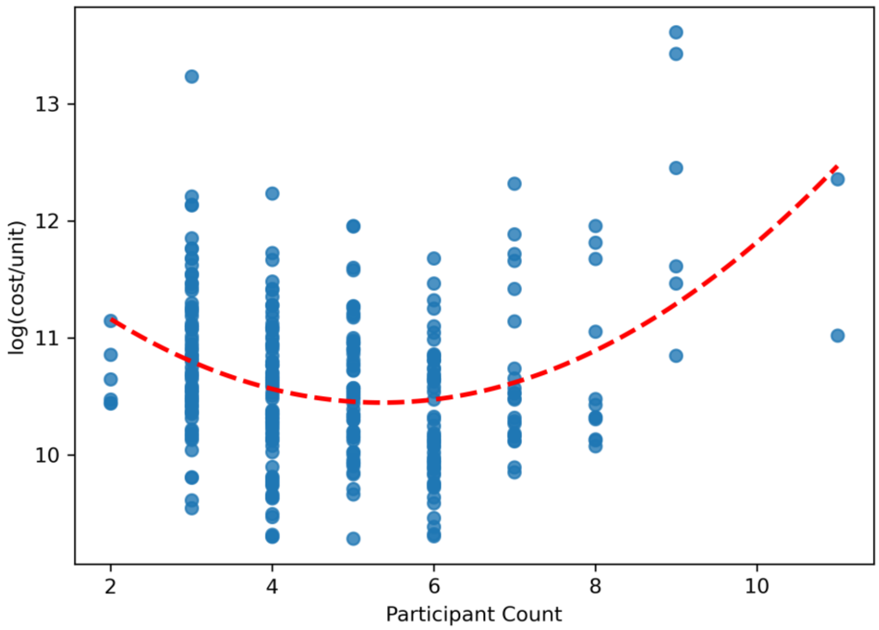

Figure 5: Log of unit cost as a function of participant count for barrier removal projects.

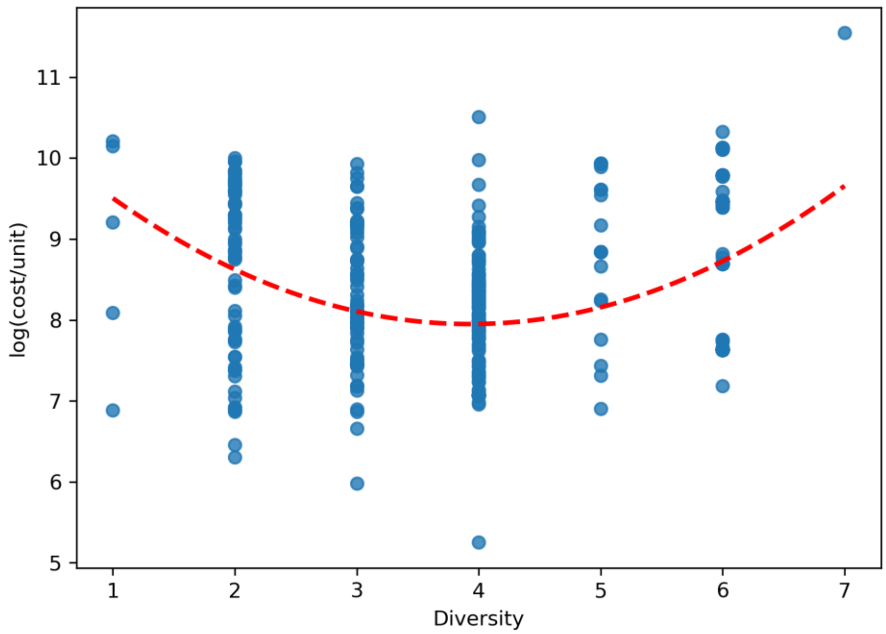

Figure 6: Log of unit cost as a function of representational diversity for riparian planting projects.

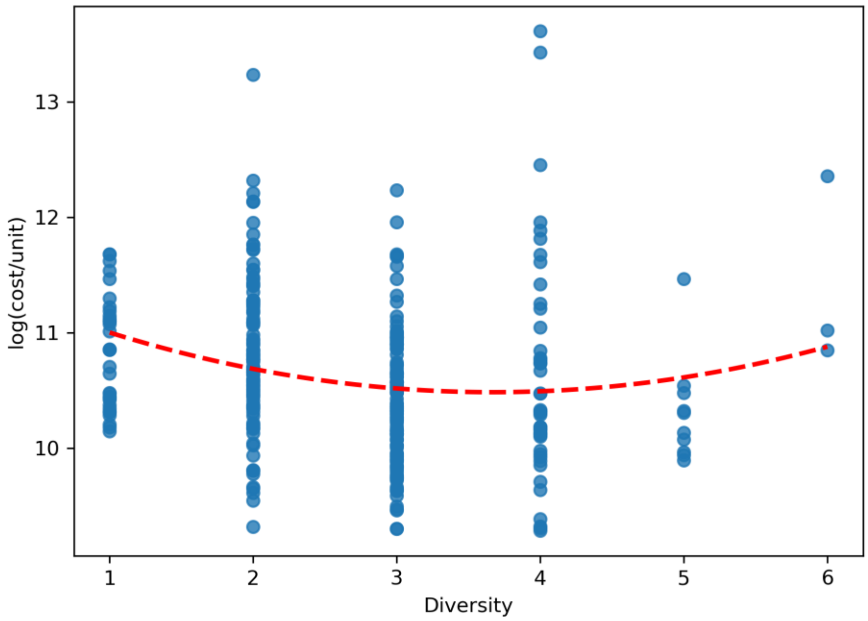

Figure 7: Log of unit cost as a function of representational diversity for barrier removal projects

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Chapter 5: Conclusion

In the first two essays in this dissertation I model the health impacts of decarbonizing the US power electric sector and provide methodological insights on how these health improvements and their contributions to environmental justice can be better understood and disaggregated by income, race, age, and geography. More specifically, in the first essay I use three regulatory grade models to show how reductions in emissions from fossil fuel EGUs improve ambient air quality for $PM_{2.5}$ and ozone. The reduced exposure results in a substantial reduction in premature mortality for all races and age groups. The normalized gains are largest for Blacks, followed by Whites aged 25-74. The differences in race-specific mortality incidence significantly impact the health results and suggest greater EJ-improving outcomes.

In the second essay I extend the methodology from the first essay to estimate reductions in asthma exacerbation among children (age group 6-18). This work builds on the contribution of the first essay by adding a poverty dimension to the race, age and state decomposition. Prevalence rates by race and poverty were estimated through the 2008-2021 BRFSS surveys, poverty status from the 2021 ACS, and poverty thresholds provided by the Census. When the population and prevalence rates are disaggregated by poverty status, I find substantially larger health benefits for poor and predominantly Black children.

In the last essay I address how diverse and broad participation in complex projects can increase stakeholder engagement, reducing the costs for environmental collaboration to protect natural resources. This analysis relies on a unique data repository of environmental projects managed by OWEB. I examine the impact of several structural variations, including collaboration form, number and representational diversity of participants, and contributions of in-kind resources, on the cost-effectiveness of collaborative watershed projects in Oregon. My results indicate that collaboration form, participant numbers, and resource contributions affect cost-effectiveness.

In this dissertation I provide scholars and policymakers with high-resolution methodologies to understand the impacts of regulatory policies on environmental justice. Historic prejudices have created environmental injustices that have lingered on for decades and have deeply scarred divided, and even destroyed communities and species. The first two essays pertain to healing the damages from emissions from the fossil fuel plants in diverse and poor communities, and in the third essay I discuss how that healing can be made cost effective by analyzing environmental projects in Oregon that have been carried out in past two decades to protect the salmon species.

Finally, it is important to emphasize that environmental injustices are not just against humans but often against other species and even environments. No matter how daunting and costly the endeavor – whether it be decarbonizing the US power sector, saving the salmon in Pacific Northwest, protecting the coral reefs in the Indian Ocean, or safeguarding baobab trees in Madagascar – wherever these injustices are, it is vital that irrespective of socioeconomic status, nationality, geography, race, or even specie, we must work collaboratively to protect the environment better and heal our communities and planet from historical prejudices of our own making.

Curriculum Vitae

EDUCATION

Maxwell School of Citizenship and Public Affairs, Syracuse University *2018-2023*
PhD in Public Administration

- Dissertation: Three Essays on Environmental Justice
- Advisor: Peter Wilcoxen
- Committee: Tina Nabatchi, David Popp, Dallas Burtraw, Charles Driscoll, Rebecca Schewe

College of Engineering and Computer Science, Syracuse University *2020-2023*
Master of Environmental Engineering

- Dissertation: Modelling changes in deposition patterns of pollutants under decarbonization of US electric grid
- Advisor: Charles Driscoll
- Committee: Ted Russell, Cliff Davidson, and Peter Wilcoxen

Maxwell School of Citizenship and Public Affairs, Syracuse University *2017-2018*
Master of Public Administration

Lahore University of Management Sciences *2010-2014*
Bachelor of Arts (Honors)

Honors and Awards

- Research Excellence Doctoral Funding (REDF) Fellowship 2022
- Ajello Fellowship 2020-2021
- EMPOWER Fellowship 2019-2021
- EMPOWER-NSF Seed Grant ($2500) 2020
- Syracuse Water Initiative Fellowship 2019
- Convocation Student Speaker – Brady K Howell Speech 2018
- Maxwell Merit Scholarship 2017

RESEARCH & PUBLICATIONS

Peer-reviewed journal articles

- **Mehdi, Qasim** and Tina Nabatchi, "Saving the salmon: Examining the efficiency of collaboration." *Journal of Public Administration Research and Theory* (2022).
- Vasilakor, Petros, Huizhong Shen, **Qasim Mehdi**, Peter Wilcoxen, Charles Driscoll, Kathy Lambert, Dallas Burtraw, and Armistead G. Russell, "US Clean Energy Futures – air quality benefits of zero-carbon energy policies." *Atmosphere* (2022).

Working papers and ongoing research projects

- **Mehdi, Qasim**, and Peter Wilcoxen, "How much environmental justice is achieved if US decarbonizes its electric grid."

- **Mehdi, Qasim**, and Marwah Maqbool, "Understanding attitudes, knowledge and practices of farmers pertaining to agricultural crop burning in Punjab Pakistan."
- **Mehdi, Qasim**, and Peter Wilcoxen, "Difference in air quality and health benefits with and without unrestricted small gas-electric generators units (EGU)."
- **Mehdi, Qasim**, Petros Vasilakos and Charles Driscoll. "Modelling changes in deposition patterns of pollutants under complete decarbonization of US electric grid."
- **Mehdi, Qasim**, and Peter Wilcoxen, "Modelling changes in asthma prevalence by race under complete decarbonization."
- Osterling, Nick, **Qasim Mehdi**, and Tina Nabatchi, "Organizational stability and diversity in collaborative governance regimes."

Reports

- Driscoll, Charles, Kathy Lambert, Peter Wilcoxen, Armistead Russel, Dallas Burtraw, **Qasim Mehdi**, Maya Domeshek, Shen Huizhong and Petros Vasilakos, Clean Energy Futures. "An 80x30 clean electricity standard: Carbon, costs, and health benefits." (2021).
- Cochran, Bobby, Kathy Fields, Bryce Barthuly, Tina Nabatchi, Rebecca McLain, and **Qasim Mehdi**. "The Atlas of Collaboration: Oregon Volume: Version 1.0." (2019).

CONFERENCES & SEMINARS

- How much environmental justice is achieved if US decarbonizes its electric grid, Association for Public Policy Analysis & Management, 2022
- Decarbonizing the US Electric Grid through a Clean Electricity Standard: Carbon, Costs, and Health Benefits. NC Breathe, Catawba College, 2022
- Modelling changes in deposition patterns of pollutants under decarbonization of US electric grid. National Atmospheric Deposition Program, University of Wisconsin-Madison, 2021
- Saving the salmon: Examining the efficiency of collaboration. Public Management Research Association, Online, 2020
- Saving the salmon: Examining the efficiency of collaboration. American Society for Public Administration, Online, 2020
- Saving the salmon: Examining the efficiency of collaboration. Program for Advancement of Research on Conflict and Collaboration, Syracuse University, 2019

PROFESSIONAL EXPERIENCES

Researcher

Centre for Policy Research, Maxwell School, Syracuse University (SU)	*2018-*
Washington State Department of Fish and Wildlife (WSDFW)	*2018*
Campbell Public Affairs Institute, Maxwell School, Syracuse University (SU)	*2017-2018*
Lahore University of Management Sciences (LUMS)	*2013-2016*
World Wildlife Fund, Pakistan	*2014*

Project Manager
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SERVICES

Reviewer for *Journal of Public Administration Research and Theory* *2021-*
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PROFESSIONAL ORGANIZATION MEMBERSHIP

- Association for Public Policy Analysis and Management
- American Society of Public Administration
- Public Management Research Association

SKILLS

Technical

- Python, Stata, R, Tableau, ArcMap, QGIS, and BenMAP