UNIVERSITY OF CALIFORNIA,
IRVINE

Assessment of Emerging Regional Air Quality (AQ) and Greenhouse Gas (GHG) Impacts and Potential Mitigation Strategies in U.S. Energy Sectors

DISSERTATION

Submitted in partial satisfaction of the requirements
for the degree of

DOCTOR OF PHILOSOPHY

In Environmental Engineering

by

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2015

DEDICATION

I dedicate this work to those that have gone before us to prepare the way

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LIST OF ACRONYMS

AB	Assembly Bill
AEO	Annual Energy Outlook
AES	Advanced Energy Storage
ANL	Argonne National Laboratory
AP	Air Pollution
AQ	Air Quality
AQS	Air Quality Study
ASES	American Solar Energy Society
AWEA	American Wind Energy Association
BAU	Business As Usual
BC	British Columbia
BEV	Battery Electric Vehicle
BIGCC	Biomass Integrated Gasification Combined Cycle
BPA	Beaumont-Port Arthur
CA	California
CAA	Clean Air Act
CAES	Compressed Air Energy Storage
CAFE	Corporate Average Fuel Economy
CAISO	California Independent Systems Operator
CARB	California Air Resources Board
CCS	Carbon Capture and Storage
CESA	California Energy Storage Alliance

CHE	Cargo Handling Equipment
CHP	Combined Heating and Power
CMAQ	Community Multi-scale Air Quality Model
CNG	Compressed Natural Gas
CO	Carbon Monoxide
CSAPR	Cross-state Air Pollution Rule
CSP	Concentrated Solar Power
CT	Combustion Turbine
CTL	Coal-to-Liquid
CTMS	Chemical Transport Modeling System
CV	Conventional Vehicles
DC	Direct Current
DEC	Dedicated Energy Crops
DOE	Department of Energy
DOT	Department of Transportation
EE	Energy Efficiency
EGRID	Emissions and Generation Resource Integrated Database
EGS	Enhanced Geothermal System
EGU	Electricity Generating Unit
EIA	Energy Information Administration
EMS	Emissions Modeling System
EPA	Environmental Protection Agency
EPRI	Electronic Power Research Institute
ERCOT	Electricity Reliability Council of Texas

EV	Electric Vehicle
FCEV	Fuel Cell Electric Vehicle
FCGT	Fuel Cell Gas Turbine Hybrid System
GHG	Greenhouse Gas
GM	Goods Movement
GMERP	Goods Movement Emission Reduction Plan
GREET	Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation Model
GW	Gigawatt
GWP	Global Warming Potential
HAP	Hazardous Air Pollutants
HC	Hydrocarbon
HDV	Heavy Duty Vehicle
HEV	Hybrid Electric Vehicle
HGB	Houston-Galveston-Brazoria
HVAC	Heating, Ventilation and Air Conditioning
ICE	Internal Combustion Engine
IEA	International Energy Administration
IGCC	Integration Gasification Combined Cycle
IMO	International Marine Organization
IPCC	Intergovernmental Panel on Climate Change
kW-hr	Kilowatt-hour
LA	Los Angeles
LCA	Life Cycle Analysis

LDV	Light Duty Vehicle
LFG	Landfill Gas
LNG	Liquefied Natural Gas
LPG	Liquefied Petroleum Gas
MARKAL	Market Allocation Model
MDV	Medium Duty Vehicle
MISO	Midcontinent Independent System Operator
MJ	Megajoule
MM	Million Metric
MMTCO2	Million Metric Tons Carbon Dioxide
MSW	Municipal Solid Waste
MW	Megawatt
MW-hr	Megawatt-hour
NA	Not Applicable
NAAQS	National Ambient Air Quality Standards
NCEP	National Centers for Environmental Prediction
NEI	National Emissions Inventory
NEUS	Northeastern United States
NGCC	Natural Gas Combined Cycle
NJ	New Jersey
NMHC	Non-methane Hydrocarbons
NO	Nitrogen Oxide
NRC	National Research Council
NREL	National Renewable Energy Laboratory

NRGGI	Northeast Regional Greenhouse Gas Initiative
NY	New York
NYC	New York City
OGV	Ocean Going Vessel
ORNL	Oak Ridge National Laboratory
PA	Pennsylvania
PAN	Peroxyacetyl Nitrate
PC	Pulverized Coal
PFI	Petroleum Fuel Infrastructure
PHA	Port Houston Authority
PHEV	Plug-in Hybrid Electric Vehicle
PM	Particulate Matter
PNNL	Pacific Northwest National Laboratory
PV	Photovoltaic
ROG	Reactive Organic Gasses
RPS	Renewable Portfolio Standard
SCC	Source Classification Code
SCR	Selective Catalytic Reduction
SF	San Francisco
SIP	State Implementation Plan
SJV	San Joaquin Valley
SMOKE	Sparse Matrix Kernel Operator Emissions Model
SMR	Steam Methane Reformation
SNCR	Selective Non-catalytic Reduction

SO2	Sulfur Dioxide
SOx	Sulfur Oxides
SOCAB	South Coast Air Basin
TES	Thermal Energy Storage
TEU	Thousand Twenty Toot Equivalent Units
TPY	Tons Per Year
TSP	Total Suspended Particles
TW	Terawatt
TX	Texas
UCD	University of California, Davis
UCI	University of California, Irvine
US	United States
USA	United States of America
USEPA	United States Environmental Protection Agency
UTC	Coordinated Universal Time
VMT	Vehicle Miles Traveled
VOC	Volatile Organic Compounds
WECC	Western Electricity Coordinating Council
WRF-ARF	Advanced Research Weather Research and Forecasting
WWTP	Wastewater Treatment Plant

ACKNOWLEDGEMENTS

My ability to complete this work was directly facilitated by several key individuals and groups who supported, believed in, and inspired me to do something I would not have suspected I had the capability to do.

My wife is named Sarah and she accompanied me during this journey. She celebrated the successes with me and quietly and consistently carried me through the failures. She truly believed in me and I always knew it. She put up with the poor lifestyle of a graduate student for far too long with no complaints. Well, perhaps a few minor complaints and those only occasionally. She is selfless and giving and always takes care of me. I can't thank her enough really. My only wish is that this degree leads to a better life for her and our children.

My mother is named Cheryl and she planted the seeds for this work many years ago. She saw potential in me that I strongly denied for a very long time. In fact when almost any sane person would have accepted that I wasn't anything special, she would not. She is also the reason I survived a much harder journey simply to have the chance to embark on this one. She is the bedrock of my family, and most everything good in me comes from her.

My sister is named Christina and she is very special. She supports me and encourages me honestly, without reservation. She was the first true academic in our family, and it came from passion. This is an important distinction, and one I am glad I witnessed firsthand. She is also fervently believes that individuals can change the world, no matter the enormity of the entity they seek to alter. I need to see that sometimes. She is another vital reason I made it this far to begin with.

The rest of my family is deserving of personal acknowledgement but, similarly to the weight of their contribution to my life and this work, I simply cannot fit it all here. Suffice to say, my family (all of them) is the greatest gift I've ever been given.

During this work I was in the fortunate and unique position of having multiple examples of outstanding leadership of both a professional and personal nature. I won't be able to fully capture their impact on this work and my life with these words but I will try nonetheless. Just note that the following persons deserve more accolades than I can give in this small section.

My boss is named Dr. Jack Brouwer. He gave me the chance to do this work when most would not have. To this day I'm still driven to show him he made the right choice. He pushed me when I needed motivation, supported me when I needed support, and was patient with me throughout. He can be tough but he is always fair. He gave me the freedom to explore ideas that captivated me and allowed me to find my strengths in the field of environmental engineering. There are many many other things that I could mention here but instead I will simply thank him for everything.

My mentor is named Dr. Jack Brouwer. Apart from work, he very much shaped parts of my character by teaching (or preaching depending on how you look at it) and by personal example. Characteristics he demonstrates include emphasizing the importance of family, being open and honest about good things and bad, and standing up for your beliefs while respecting others. He cares about how students are more than the journal articles and conference presentations they may complete, this is not always the case in our profession. He doesn't realize how important our relationship is to me because it isn't easy to talk about these things but it truly is something I am deeply thankful for. Again, I must thank him for everything.

I must gratefully acknowledge the contribution of Professor G. Scott Samuelsen to this work, to my life, and to my future. He is a pillar (a term not used lightly) in the field I studied and the experience of pursuing an education under his guidance was quite simply, invaluable. The research that I completed for this work was directly facilitated by his ability to create and maintain a world-leading research center with unparalleled opportunities for students. Observing his interactions with students, colleagues, and the public was a critical lesson in how a professional in our field should behave – with integrity, intelligence, and kindness. His vision for energy and the environment has been passed on to me with a significance I'm only now beginning to understand. This is again a situation where countless other positive comments could be made, but I will stop and thank him for all he has done for me.

I was extremely fortunate to have the opportunity to work with Dr. Donald Dabdub during my entire graduate career. As a world leading air quality expert, his insight and technical understanding directly raised the level of my own research through the feedback I received. He also provided enormous contributions to this work by serving as an active member on my committee. However, what I will perhaps remember most is his advice on how to holistically achieve an effective and successful balance of coursework, research, and life. I will always think back to the air quality meetings led by Dr. Dabdub where we didn't discuss air quality but rather how everyone was doing in all aspects of their life as being so valuable. I would like to acknowledge his imperative contribution here.

Dr. Marc Carreras-Sospedra is perhaps the most important technical contributor to this work. Working directly with Marc has been nothing but a pleasure as he is an exceptional researcher, engineer, colleague, and friend. His knowledge and ability quite literally provided me the tools I used to conduct this research. He was patient with my constant requests and mistakes and worked extremely hard to facilitate the model capabilities. Additionally, his advice on all things was critical to me time after time. And it was nice being able to talk soccer (football) with someone who understood how hard it is being a Liverpool fan.

My research group is named APEP. It has been an incredible and inspiring place to work and grow. Those words are accurate but fail to truly capture the importance of APEP in terms of my successful graduate school completion. The education I was provided at APEP goes far beyond a simple environmental engineering degree. I was immersed at the cutting edge of the entire spectrum of advanced energy topics and was granted an unbelievable amount of knowledge in so many fascinating and relevant subjects. The opportunities I was afforded here are beyond what I deserve. All of this reflects again the leadership and vision of Professor Samuelsen as this is the world he has built, and the continued support of the APEP staff including Professor Brouwer. I would also like to acknowledge the APEP staff including Jeff Wojciechowski, Jean Grigg, Will Decker, and Rich Hack for their unfailing assistance when it was needed. I didn't know how fortunate I was when I was granted a place here, but I do now.

My lab mates are an assorted group of exceptional individuals. So many of them have had an imprint on me that will never leave. Some that must simply be mentioned – Gia Nguyen sat next to me on the first day of this work and on the last. In between he inspired me with unfailing positivity and cheerfulness, assisted me with high level mathematics, and became a great friend. Li Zhao blazed the trail a few years before me and provided reliable and valuable insights. He proved to be an outstanding coffee companion and friend. Kersey Manliclic walked the trail directly alongside me and we shared the ups and downs together. He helped me enormously with some of the hardest parts of this journey and I couldn't have done it without him. His loyalty to his friends and co-workers should be noted and appreciated by all. So many others, Adrian Narvaez, Howard Lee, Brendan Shaffer, Derek McVay, Chris Hartley, Guillermo Gomez, Analy Castillo-Munoz, Alicia Tan, Brian Tarroja, Dimas Avila, Aria Etemadieh, Katie Leong, Chris Cogan, James Kast. The list goes on and on, I'm sure I forgetting many who deserve acknowledgement here. Thank you so much.

I would like to thank Dr. Steven J. Davis for his important contribution to this work by serving on my committee. His insight and contribution was highly valuable. Additionally, Professor Sunny Jiang provided an important contribution by serving on my qualifying exam committee.

I would also like to thank the many various ectotherms that provided me a fascinating escape when I needed to recharge and allowed me to spend countless hours in some of the most wild and beautiful natural landscapes that this world has to offer. *Crotalus cerastes* for all the exciting days spent in the ditch. *Lampropeltis zonata*, which proved to be and still is a backbreaking and entirely frustrating pursuit with an occasional joyous payoff. At least their pursuit resulted in days spent in some of the most outstanding mountain ranges in the world. *Crotalus mitchellii*, particularly those screamers in the TAs, which provided so much excitement and awesomeness. *Bothrops asper* for those incredible nights in the jungle. And many more.

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- M. Mac Kinnon, M. Carreras-Sospedra, J. Brouwer, D. Dabdub. "*Evaluating the Emerging Regional Air Quality and Greenhouse Gas Impacts of U.S. Energy Sectors*". Platform Presentation at the 107th Air and Waste Management Association Annual Conference and Exhibition. Long Beach, CA. June 2014.
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- M. Mac Kinnon, M. Carreras-Sospedra, J. Brouwer, D. Dabdub. "*Air Quality Impacts of GHG Mitigation Strategies in the Power Generation and Transportation Sectors*". Poster presentation at the 9th AirUCI Collaborative Workshop in Chemistry at Interfaces. Laguna Beach, CA. January 2013.
- M. Mac Kinnon, M. Carreras-Sospedra, J. Brouwer, D. Dabdub. "*Effects of Climate Change and Greenhouse Gas Mitigation Strategies on Air Quality*". Presented to the U.S. EPA. Durham, NC. September 2012.

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ABSTRACT OF THE DISSERTATION

Assessment of Emerging Regional Air Quality (AQ) and Greenhouse Gas (GHG) Impacts and Potential Mitigation Strategies in U.S. Energy Sectors

By

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Doctor of Philosophy in Environmental Engineering

University of California, Irvine, 2015

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The current domestic reliance on high-emitting fossil fuels for energy needs is the key driver of U.S. greenhouse gas (GHG) and pollutant emissions driving both climate change and regional air quality (AQ) concerns. Moving forward, emission sources in U.S. energy sectors will be subjected to changes driven by numerous phenomena, including technology evolution, environmental impacts, sustainability goals, and socioeconomic factors. This evolution will directly affect emissions source-related impacts on regional AQ that effective emissions control strategies must account for, including relative source contributions. Though previous studies have evaluated the emissions and AQ impacts of different sectors, technologies and fuels, most previous studies have assessed emissions impacts only without

using advanced atmospheric models to accurately account for both spatial and temporal emissions perturbations and atmospheric chemistry and transport. In addition, few previous studies have considered the integration of multiple technologies and fuels in different U.S. regions.. Finally, most studies do not project emissions several decades into the future to assess what sources should be targeted with priority over time. These aspects are critical for understanding how both emissions sources and potential mitigation strategies impact the formation and fate of primary and secondary pollutants, including ground-level ozone and particulate matter concentrations. Therefore, this work utilizes a set of modeling tools to project and then to spatially and temporally resolve emissions as input into a 3-D Eulerian AQ model to assess how sources of emissions contribute to future atmospheric pollutant burdens. Further, analyses of the potential impacts of alternative energy strategies contained in potential mitigation strategies are conducted for priority targets to develop an understanding of how to maximize AQ benefits and avoid unforeseen deleterious tradeoffs between GHG reduction and AQ. Findings include changes in the relative contribution to AQ that elevate the importance of addressing emissions from all sectors and sources including some that may be more difficult to control, including industry, petroleum refineries, and non-light duty vehicle transportation sources. Additionally, mitigation strategies must consider the full range of life cycle and system effects in order to avoid AQ tradeoffs spatially and temporally.

Chapter 1: INTRODUCTION

1.1 OVERVIEW

It is becoming evident that the environmental concerns associated with energy conversion to meet societal demands must be addressed to ensure sustainability for forthcoming generations. Climate change, driven by anthropogenic emissions of greenhouse gasses (GHG), is expected to significantly affect a multitude of global capacities with potentially devastating consequences [1, 2]. It is estimated that deep reductions in GHG emissions (e.g., 50 to 80% below 2005 levels by 2050) will be required from developed nations, including the U.S., to stabilize atmospheric concentrations at levels obligatory for prevention of detrimental changes [3, 4]. Emphasizing the enormous challenge of meeting these goals, it has been suggested that stabilizing climate may require the complete de-carbonization of energy sectors [5]. Similarly of concern, pollution in the air is expected to represent the largest global cause of environmentally related premature mortality by 2050, surpassing both inadequate sanitation and contaminated water[2]. Indeed many regions of the U.S. currently experience air quality (AQ) challenges including levels of pollutants in excess of Federal health-based standards. Thus, both GHG emissions and AQ represent areas requiring action in the U.S. to support advancement of societal health and well-being.

Epidemiological studies have linked air pollution with increased incidence of premature mortality and morbidity, incurring significant health care costs and loss of productivity [6-8]. In recent decades the U.S. has made significant progress in addressing AQ concerns, including photochemical smog and particulate matter (PM), through regulatory controls and technological advancements targeting reductions in emissions of gaseous and particulate pollutants[9]. However, the U.S. is only now beginning to address the mitigation of GHG emissions in efforts to reduce or avoid the harmful effects of changes in global climate. Though no federal regulation exists, various state and local governments have initiated plans to reduce future GHG emissions. In 2006 California implemented Assembly Bill 32 (AB 32); ground breaking climate change legislation targeting a return to 1990 levels of GHG emissions by 2020, and an 80% reduction below 1990 levels by 2050[10]. Further, a group of states in the Northeastern U.S. (NEUS) has agreed to participate in a cap-and-trade program for power generation emissions called the Northeast Regional Greenhouse Gas Initiative (NRGGI) [11]. Such measures will encourage GHG mitigation from different sectors of the U.S. economy and spur the implementation of cleaner and more efficient alternative technologies and fuels.

In 2009 U.S. GHG emissions totaled 6,633.2 million metric tons (MMT) of carbon dioxide equivalents (CO_2eq), representing an increase of 7.3% from 1990[12]. Simply put, mitigating GHG emissions is an energy problem, since energy-related activities accounted for 87% of total U.S. GHG emissions in 2009[12]. Combustion of carbon-based fossil fuels in the various energy sectors is by far and away the largest single source of U.S. GHG emissions; in 2009 fossil fuels provided 83% of total U.S. energy conversion and accounted for nearly 95% of total energy related CO_2 emissions[12]. Energy related activities were also responsible for 49% and 13% of total U.S. emissions of CH_4 and N_2O respectively. It is clear that any substantial efforts to reduce emissions of GHGs will require dramatic changes in the way energy is generated and utilized in the provision of energy services.

Further, energy-related activities are major contributors to U.S. AQ challenges. Combustion processes, particularly those associated with fossil fuels; result in release to the atmosphere of both gaseous and particulate pollutants which have both primary and secondary impacts on AQ. Direct criteria pollutants of concern include carbon monoxide (CO), which is generally emitted in urban areas and carries health risks. Oxides of sulfur (SO_x), including sulfur dioxide (SO_2), have negative impacts on human health and are precursors to acid rain and atmospheric particulate. Important indirect criteria pollutants include volatile organic compounds (VOCs), and oxides of nitrogen (NO_x), which react in the presence of sunlight via a series of photochemical reactions involving hydroxyl-, peroxy-, and alkoxy radicals to form oxidants including tropospheric ozone and peroxyacetyl nitrate (PAN) [13]. Particulate matter (PM) is both directly emitted and formed through secondary processes and is regulated under the Clean Air Act (CAA). Of particular concern are PM with diameters of less than 10 microns (PM_{10}) and PM with diameters less than 2.5 microns ($PM_{2.5}$), as exposure has been linked with serious health consequences. In 2005 energy related-sources were responsible for 95% of anthropogenic NO_x, 92% of anthropogenic SO_2, and 10% of PM_{10} emissions in the U.S.[14]. In addition, fossil fuel combustion exhaust can contain various species designated as air toxics which have associated health concerns[15].

Energy-associated anthropogenic GHG emissions occur as a result of activity in various energy sectors, including industrial, residential, transportation, and electric power generation. The two sectors responsible for the highest energy conversion and associated air emissions of GHG and pollutants, both nationally and globally, are the electric power generation and transportation sectors. The electricity sector is the largest GHG emitting sector, responsible for 33% of total U.S. GHG emissions and about 40% of total CO_2 emissions [12]. The transportation sector is responsible for 27% of total U.S. GHG emissions, second only to the electric power sector. In addition, both sectors feature prominently in the majority of proposed GHG mitigation strategies, including AB 32.

Mitigation of GHG emissions will require more efficient generation, transmission, and distribution of energy as well as the deployment of alternative technologies and fuels. Altering the current U.S. electricity and transportation sectors via technological and behavioral shifts will have significant and wide ranging effects outside of reducing emissions of GHG. Emissions of GHGs and criteria pollutants are highly correlated as a result of shared generating sources, particularly with regards to fossil fuel combustion. Adjustments in technologies utilized for energy production and consumption in concert with deviations in net, economy-wide energy conversion will have various potential AQ impacts resulting from spatial and temporal perturbation of criteria pollutant emissions. GHG mitigation strategies that seek to curb combustion of fossil fuels will concurrently reduce other emissions associated with direct and indirect air pollutants including NO_x, SO_x, PM, CO, Ozone, and air toxics such as mercury and lead. Further, deployment of clean energy technologies has other ancillary benefits; including negating the need for costly pollution control technologies utilized on conventional energy technologies to achieve compliance with AQ regulations.

Economic estimates of the monetary co-benefits of AQ from climate change mitigation in the literature ranged from $2-196 per ton of mitigated CO_2 (mean value of $49), demonstrating the importance in considering AQ in climate change policy decision making [16].

Currently, climate change legislation is a source of controversy in the U.S., and measures to curb emissions face uncertainty in both the public and private sectors. Implementing policy that can simultaneously address AQ and GHG concerns will assist law makers in maximizing health and economic co-benefits, and assure that GHG mitigation strategies do not have unforeseen negative impacts on AQ. Thus, with a primary focus on identification of potential for improving AQ the present study seeks to evaluate GHG mitigation strategies in the electric power generation and transportation sectors.

The regions selected for AQ assessment in include California (CA), Texas (TX) and a five state region representing the northeastern U.S (NEUS). Texas includes nonattainment areas for both ozone and $PM_{2.5}$, particularly the Houston-Galveston-Brazoria (HGB) and Beaumont-Port Arthur (BPA) areas. The HGB and BPA areas are characterized by elevated ozone levels resulting from high temperatures and intensive solar radiation much of the year, a stagnating land-sea breeze circulation that traps pollutants, and significant emissions of VOC and NO_x from urban and industrial activities [17-19]. Further, large emissions of primary PM and secondary PM precursors from industrial and urban sources lead to elevated levels of $PM_{2.5}$ [20].

The various strategies that may be employed to reduce GHG emissions (or enhance sustainability and energy independence) will also affect the regional sources of pollutants and vice versa. Such shifts could potentially be in quantity (e.g., net reduction or increase) and/or chemical composition (e.g., different pollutant species). Additionally, and with high importance to the formation and fate of secondary pollutants, emission distributions may be altered spatially and temporally. Impacts on emissions from a given generation strategy are directly dependent on the strategy that it displaces, which is complicated by the significant variation in technologies, fuels, and demands that comprise different regional power grids.

Ambient pollutant concentrations are determined by multiple complex factors, including the quantity, location, and timing of direct emissions from sources, and various atmospheric processes including transport and dilution. Further, multifaceted atmospheric chemistry defines formative species level impacts. Therefore, AQ conditions are dependent on natural factors, including topography, meteorology, biogenic emissions and climate; in addition to the local emissions signature associated with anthropogenic sources. Variances in AQ problems arising from differences in regional emissions patterns and naturally occurring conditions necessitate the formation and deployment of regionally specific mitigation strategies.

The formation of tropospheric ozone is determined by complex interrelationships between characteristics of precursor emissions, atmospheric chemical reactions, and meteorology which impacts dispersion and chemical reaction rates[21, 22]. The chemistry relating to ozone production is non-linear and results in time lags occurring between emissions and ozone formation which introduces spatial complexity. Ozone formation varies regionally due to the complex mix of factors responsible for its production, including the production potential and spatial and temporal effects of VOC and NO_x precursor emissions [23].

Further, the dynamics of atmospheric pollutant formation are often complex and predicting how technologically-driven pollutant perturbations translate to changes in atmospheric concentrations is often complicated. Using OZONE as an example, nitrogen oxides emitted as nitric oxide (NO) rapidly react in the atmosphere to produce nitrogen dioxide (NO_2), which further reacts with VOCs in the presence of sunlight via a series of photochemical reactions involving hydroxyl-, peroxy-, and alkoxy-radicals to form oxidants including tropospheric ozone and peroxyacetyl nitrate (PAN) [13]. Additionally, regions can be either NO_x- or VOC-limited (generally VOC limited areas encompass urban centers with high anthropogenic emissions and NO_x-limited areas include rural locations), which directly impacts resulting variation in ozone from reductions or increases in emissions [24, 25]. Similarly multifaceted relationships exist between direct emissions and atmospheric PM concentrations [26, 27]. Thus, predicting how implementation of a mitigation strategy will impact ozone or PM based off approximations of emission reductions or increases is comprehensively limited. In fact, a thorough assessment of such impacts requires characterization of technological information to develop detailed spatially and temporally resolved pollutant emission fields to serve as input for advanced AQ models to account for chemical and physical atmospheric processes, e.g., mixing, transport, photochemistry.

Reducing the ambient levels of atmospheric pollutants and thus limiting human exposure to health-damaging compounds has important value to society, including monetary worth. Historical improvements in AQ from the regulation of emissions have garnered net public health benefits in the U.S. estimated in excess of $1 trillion[9]. Therefore GHG mitigation-related technology shifts that likewise reduce pollutant emissions and improve AQ provide monetary value to society that, although potentially similar in magnitude to abatement costs, often go unaccounted for or underestimated for various reasons [28, 29]. Further, select power generation mitigation strategies (e.g., CCS, large renewable capacities) that may be most necessary in providing deep sector GHG abatement face significant cost barriers[3, 30]. The extent to which a given strategy can reduce emissions is correlated with attained market penetrations and large-scale reductions may require alternative technologies to be more cost competitive with current options. Accurate identification and assessment of co-benefits can assist in determining the true costs of GHG abatement associated with various technologies and fuels and identify opportunities to optimize mitigation strategy development and deployment. Additionally, climate change legislation continues to represent a source of controversy and measures to curb GHG emissions face societal hurdles. Implementing policy that can simultaneously address AQ and GHG concerns will assist law makers in maximizing co-benefits, and assure that GHG mitigation strategies do not have unforeseen deleterious impacts on AQ[16]. Further, developing integrated policy can allow currently existing and mature AQ regulatory statutes to support climate mitigation in the near- to mid-term[31]. Thus, there is a need for additional information regarding the AQ impacts of low-carbon generation methods to better understand and assess mitigation strategies in the domestic power sector.

1.2 GOAL

The goal of this dissertation is to delineate how deploying alternative energy strategies in U.S. regions can best improve regional AQ in tandem with GHG emissions reductions. The results of this work will advise and support an enhanced understanding of how decision makers can shape emission mitigation strategies in different U.S. regions to best attain AQ and GHG benefits 2055. To achieve the dissertation goal, the following objectives are required:

1.3 OBJECTIVES

Objective 1. Develop an understanding of the key drivers of U.S. regional AQ and GHG concerns and identify and characterize potential mitigation strategies,

Objective 2. Project energy system evolution to the year 2055 in the regions of study

Objective 3. Develop and apply a modeling platform to evaluate the spatial and temporal distribution of emissions and AQ impacts for three U.S. regions in future years,

Objective 4. Simulate atmospheric chemistry and transport to assess contributions of energy sector emission sources to pollutant burdens,

Objective 5. Develop and evaluate alternative energy provision strategies to determine AQ and GHG impacts of potential mitigation measures, and

Objective 6. Determine the mitigation measures in each sector that most effectively improve AQ in tandem with GHG emission reductions in modeled regions.

Chapter 2: BACKGROUND

2.1 TRANSPORTATION SECTOR

It is becoming evident that the environmental concerns associated with energy conversion to meet societal transportation demands must be addressed to ensure sustainability for forthcoming generations. Representing one key sustainability tenant, pollution in the air is expected to represent the largest global cause of environmentally related premature mortality by 2050, surpassing both inadequate sanitation and contaminated water[2], as stated previously. Many regions of the U.S. currently experience AQ challenges that include levels of atmospheric pollutants in excess of Federal health-based standards[51]. Energy-related activities, including the energy conversion of the transportation sector, are responsible for the bulk of pollutant emissions driving current U.S. regional AQ concerns, including ground level concentrations of ozone and $PM_{2.5}$[14]. In recent decades the U.S. has made significant progress in addressing such concerns, including photochemical smog and PM, through regulatory controls and technological advancements targeting reductions in emissions of gaseous and particulate pollutants[9]. However, in the absence of targeted and comprehensive adjustment, the emissions and atmospheric pollution associated with U.S. transportation could potentially intensify as the demand for various transportation services rises in tandem with population and economic growth.

Transportation encompasses the movement of persons or goods by various technology types and is typically categorized into sectors that include light duty vehicles (LDV), medium duty vehicles (MDV), heavy duty vehicles (HDV), rail, ship, aircraft, and other vehicles (e.g., off road equipment). Typically, the LDV classification designates passenger cars and light-duty trucks, (e.g., sport utility vehicles, pickup trucks) while MDV and HDV incorporate commercial technologies (e.g., tractor trailers, school and transit buses) designated by gross vehicle weight. While LDVs are primarily utilized for personal transport, MDVs and HDVs span a range of uses with construction, agriculture, and transport of freight the foremost end-uses[52]. Off-road vehicles include a large and diverse spectrum of technologies including agricultural equipment (e.g., tractors, mowers, combines), airport ground equipment, construction and mining equipment (e.g., pavers, backhoes, drill rigs), industrial equipment (e.g., forklifts, terminal tractors), logging equipment, railroad maintenance vehicles, and recreational equipment (e.g., off-road motorcycles, all-terrain vehicles, golf carts). Additionally, the transportation sector includes air travel, however impacts from aircraft are not considered here due to modeling limitations associated with the distribution of source emissions.

Currently, the combustion processes associated with the bulk of conventional transportation technologies and fuels results in atmospheric releases of both gaseous and particulate pollutants. Additionally, the transportation sector is a key contributor of domestic greenhouse gas (GHG) emissions and a foremost target for any meaningful U.S. mitigation effort[53]. In 2012 over 28% of total U.S. energy conversion and GHG emissions were attributable to transportation sources, second only to power generation[54]. Similarly, emissions of criteria air pollutants from transportation comprise a large fraction of domestic totals, including 54%, 59%, and 23% of carbon monoxide (CO), oxides of nitrogen (NO_x), and volatile organic compounds (VOC), respectively[52]. Additionally, transportation sources emit important amounts of sulfur oxides (SO_x) and particulate matter; including fine particles ($PM_{2.5}$) with considerable human health risk [55, 56]. While emissions from transportation directly impact society via induced health effects, materials degradation, aesthetics etc., further contribution to these burdens occurs via the formation of health and materials damaging secondary pollutant species, including ground-level ozone, and secondary particulate matter (PM). Ozone forms in the troposphere via chemical interactions between emissions of NO_x and VOCs in the presence of sunlight and represents one of the most challenging pollutants to mitigate. Indeed many regions of the U.S. currently experience nonattainment for Federal criteria pollutant regulatory standards for both ozone and $PM_{2.5}$ [51]. Further, exposure to elevated concentrations is known to induce a range of detrimental health outcomes[57] and meeting the health-based standards have been shown to provide significant societal benefits [58]. Similarly, $PM_{2.5}$ has been shown to increase a number of serious disease burdens and represents a foremost regional AQ concern [59, 60].

The transportation sector is responsible for 27% of total U.S. GHG emissions, largely resulting from the combustion of petroleum based fuels which accounted for 94% of sector energy conversion in 2009[12, 61]. The transportation sector includes on-road vehicles, rail, marine, and air transport. GHG emissions are directly correlated with energy conversion by each sub-sector as all modes are currently dependent on petroleum fuels with similar carbon contents. On-road vehicles are responsible for the majority of U.S. transportation related CO_2 emissions, emitting 77% of total transportation associated emissions and consuming

80% of sector energy conversion in 2008. Within on-road vehicles, passenger cars and light-duty trucks, which constitute the light duty vehicle fleet (LDV), contribute the greatest amount of emissions (60%).

Accounting for the transportation sector's fractional share of total emissions to meet a 60-80% reduction of 1990 GHG emissions by 2050 net U.S. LDV emissions must decrease by about 80%[62]. In the same time period factors including economic and population growth are expected to increase demand for the transportation of persons, goods and services, further increasing the difficulty of meeting targeted reductions.

The growth in GHG emissions from transportation is expected to continue in the absence of mitigation efforts. It has been projected that in 2050 30-50% of total global CO2 emissions will arise from transport, compared to 20-25% currently[63]. Historically the transportation sector has experienced the highest growth rate in energy conversion and associated emissions of all U.S. energy sectors, with exceptions including short temporal periods of decline in response to economic crises [64]. Extrapolation of the 2011 EIA Annual Energy Outlook reference case to the year 2050 yields a net increase in total transportation GHG emissions of about 7% [65]. However, other estimates of future LDV GHG emissions have included much higher values [62]. For example, Melaina et al., 2010 assumed an emissions growth of 62% if no further policy designed to reduce emissions is implemented [66].

Transportation sector strategies to mitigate GHG emissions center on three broad approaches; reducing fuel carbon intensity, reducing fuel consumption, and reducing vehicular miles traveled (VMT). These strategies require the development and deployment of alternative fuels and vehicle technologies, as well as behavioral changes to decrease travel demand intensity. Recent studies examining the U.S. and California transportation sectors have demonstrated that it is unlikely a single mitigation mechanism will have the ability to provide the desired emissions reduction goals and multiple complimentary strategies implemented in parallel will be required to meet long-term emissions reduction goals [66-69].

The current, almost total reliance of the transportation sector on petroleum-based fuels that emit significant amounts of GHGs and criteria pollutants when combusted gives shifting to fuels with lower life cycle GHG emissions critical importance in mitigation efforts [70]. The entrenchment of petroleum based liquid fuels is partly a result of their inherent advantages for meeting transportation demand, including a high energy density which allows for desired vehicle range in tandem with low on-board storage requirements. Further, petroleum fuels have a firmly established production and distribution infrastructure which assists in lowering fuel costs. Overcoming the inertia of the petroleum fuel-based infrastructure represents a major challenge, however many alternative fuels are being developed and pursued that could provide vehicle power with lower GHG and criteria pollutant emissions.

Alternative fuels with the potential to meet a large portion of transportation sector demand in 2050 include electricity, hydrogen, and biomass-derived liquid fuels such as ethanol and biodiesel. Other alternative fuels include liquefied petroleum gas, and synthetic fuels but these options were not considered in this evaluation it is unknown if high levels of use in the LDV sector will be achievable in the study horizon. Natural gas provides moderate life cycle GHG reductions (15%) relative to gasoline LDVs, however potential benefits are constrained by interactions with the electric power generation and residential sectors; as such natural gas was not considered here [64]. The choice of fuel will be closely related to vehicle technology, as some alternative fuels, such as ethanol (up to a 10% blend), can be used in today's vehicles while others, such as hydrogen and electricity, will require a modified powertrain. Further, some alternative fuels, i.e. hydrogen, will require a novel production, distribution, and refueling infrastructure and others, i.e. electricity, will require upgrades to current distribution systems.

The carbon intensity of a fuel is determined by quantifying GHG emissions on a life cycle basis per unit of energy contained in the fuel. Life cycle stages with the potential for GHG and pollutant emissions include feedstock extraction or production, refining processes, and distribution activities including transportation and transmission. Additional emissions can occur from ancillary activities, for example the production of fertilizers and pesticides required for biofuel feedstock growth. For an accurate, robust assessment of GHG mitigation and AQ impacts from the production and use of alternative transportation fuels net emissions from all relevant stages must be considered. For example, FCEVs and BEVs have no tailpipe emissions of GHGs or criteria pollutants; however air emissions occur during various stages associated with the production of hydrogen and electricity utilized in vehicle propulsion.

The carbon intensity of motor gasoline is about 93.8 CO_2eq per megajoule (MJ). The carbon intensity of gasoline and distillate fuels will likely grow over time as limited supplies of conventional crude oil lead to greater use of unconventional fossil resources such coal, shale, heavy oil, and tar sands. For example, production of gasoline from tar sands and coal-to-liquid (CTL) technology, can have significantly (27-77%) higher life cycle GHG emissions than traditional gasoline[71]. LCA analysis has demonstrated that switching to heavy oil and tar sands could increase the GHG emission intensity of petroleum fuels up to 40% and increase refinery emissions by 200-300%[72, 73]. A move away from petroleum to alternative fuels is therefore essential if major reductions in transportation GHG emissions are to be achieved.

Criteria pollutant concentrations, including ozone, and particulate formation and transport will be impacted by changes in vehicle technology, miles traveled, and direct emissions occurring from implementation of GHG mitigations strategies, as well as atmospheric chemistry and transport. Many of the transportation strategies to mitigate GHG emissions offer the potential co-benefit of improved AQ; however others have the potential to increase life cycle criteria pollutant emissions which could negatively impact AQ. Further, many individual strategies could have beneficial or deleterious impacts on both GHG and criteria pollutant emissions dependent on the evolution of other sectors (e.g., electric vehicle deployment concurrent with renewable resources versus coal fired generation).

In addition to direct release of emissions from the various vehicle technologies, the production of the required fuels is associated with substantial amounts of criteria pollutants

[74, 75]. The current reliance of the transportation sector on petroleum fuels requires the existence of an extensive petroleum fuel production and distribution system in the U.S., including the regions of study in this work. Industrial process plants utilized in the processing and refining of crude oil feedstock are generally labeled petroleum refineries. Refineries range in complexity, and use numerous processes (e.g., distillation, reforming, hydrocracking, coking, blending) to produce an assortment of products including gasoline, aviation fuel, distillate fuel, and residual fuels (additional intermediates are produced including hydrogen)[76]. The large, sprawling industrial processes typified by refinery complexes generate a diverse range of pollutant emissions distributed across a spatial area including CO, NO_x, PM, SO_2, VOCs, and numerous air toxic compounds, e.g., benzene, toluene[77]. It follows then that emissions associated with the production, storage, transport, and distribution of conventional petroleum fuels are known to be major contributors to regional AQ problems. For example, industrial activities around the Greater Houston area, which includes some of largest concentrations of petrochemical facilities in the U.S., contribute to regional non-compliance with Federal AQ standards[17]. Further heightening concern, reported emissions from petrochemical facilities may be underreported and therefore AQ impacts currently underestimated [78-80]. Thus, there is a need for more information regarding the potential effects on regional AQ from producing, transporting, and distributing petroleum transportation fuels in coming decades.

2.1.1 Advanced Light Duty Vehicle Technologies

Alternative LDV technologies offer potential increases in vehicle efficiency compared to CVs and the use of fuels with reduced environmental impacts compared to petroleum fuels[68]. Currently, those considered for large-scale deployment include some degree of propulsion system electrification. Potential evolutionary pathways from current CVs are demonstrated in Figure 3. Hybrid electric vehicles (HEVs) utilize an electric traction motor and battery bank in combination with an internal combustion engine (ICE). Plug-in hybrid electric vehicles (PHEVs) utilize batteries that can recharge from an external power source, displacing a portion of liquid fuel with grid electricity. Two zero emission vehicles currently being pursued by major automakers are the fuel cell electric vehicles (FCEVs) powered by hydrogen and battery electric vehicles (BEVs) powered by grid electricity. The absence of an ICE achieves efficient vehicle propulsion and zero GHG and AQ emissions at the tailpipe.

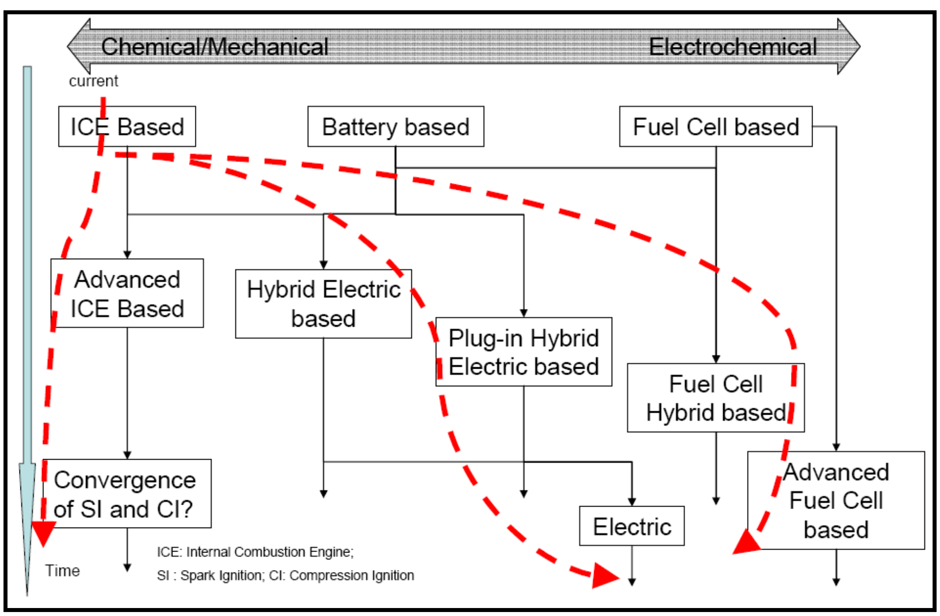


Figure 1: LDV Propulsion System Evolutionary Pathways. Source: [68]

2.1.1.1 Fuel Cell Electric Vehicles (FCEV)

Electric vehicles deriving propulsion power from onboard hydrogen fuel cells (FCEVs) have experienced significant consideration as a strategy for reducing transportation related GHG and criteria pollutant emissions. Hydrogen, when utilized in a fuel cell to provide vehicle propulsion, offers the benefits of no direct vehicle GHG or criteria pollutant emissions and can be produced from a variety of supply chain strategies, including several with the potential for high sustainability and energy security benefits.

By replacing the onboard ICE with a fuel cell for propulsion, vehicle efficiency is increased 2-3 times relative to current and future CVs, reducing fuel use and associated emissions [68, 320, 321]. FCEVs have several characteristic advantages relative to other advanced vehicle technologies, including BEVs. Thomas, et al. (2009) reported that for vehicle range greater than 100 miles fuel cells are superior to batteries for vehicle propulsion in terms of mass, volume, cost, and refueling time[322]. The current FCEV ranges per refill (190-250 miles) are much higher than those for BEVs per full battery charge and FCEV allow for more complete decoupling from petroleum fuels usage than PHEVs[321]. FCEVs can also be deployed in heavier duty applications (e.g., goods movement). These advantages may be most important with regards to consumer acceptance of a novel technology, as in some regards FCEVs can be considered more similar to current vehicle technologies than PHEVs and BEVs. These advantages of FCEV technology are enabled by the fundamental decoupling of the energy storage device from the energy conversion device allowing independent sizing of the power and energy capacities of an FCEV.

Challenges for large scale deployment of FCEVs include durability and cost issues associated with automotive fuel cell systems and the incremental costs of FCEV could remain higher than CVs for several decades[323, 324]. However, fuel cell performance has improved and costs have decreased significantly in recent years, and increased market penetrations of FCEVs will further facilitate mass commercialization of fuel cell technology, potentially lowering costs dramatically [68, 325, 326]. On-board fuel storage is a hurdle due to hydrogen's low volumetric energy density and major technological advancement in hydrogen storage could simultaneously improve vehicle range while lowering costs. Perhaps most challenging, significant displacement of CVs will require deployment of a novel production, distribution, and refueling infrastructure to support a fleet of FCEVs. If these and other challenges can be successfully overcome FCEVs have the potential to significantly mitigate GHG emissions in the LDV sector and could offer substantial AQ improvement benefits.

Upstream emissions associated with energy utilized in the production, compression, liquefaction, and delivery of hydrogen must be accounted for in an accurate analysis of GHG and AQ impacts of hydrogen fuel cell vehicle deployment. The majority of the hydrogen is produced today from fossil fuel primary energy sources, resulting in air emissions of GHG and criteria pollutants (96% of global hydrogen is produced from fossil fuels, with more than 75% produced from natural gas feedstock) [327]. Hydrogen supply chain strategies include centralized or distributed production processes followed by compression and storage and/or transportation to dispensing locations via pipeline or truck. Currently the lowest cost and most widely used production method is steam methane reformation (SMR) of natural gas which accounts for 80% and 40% of U.S. and global hydrogen production respectively[328].

Though hydrogen supply chain strategies involving SMR result in significant emissions of GHG and pollutants, estimated reductions in life cycle GHG emissions are approximately 40-50% for FCEVs compared to gasoline powered CVs of similar size[329, 330]. This is due to the relatively low GHG intensity of hydrogen production from natural gas (e.g., compared to gasoline combustion) coupled with the high efficiency and zero GHG emissions of the FCEV. If carbon capture and storage (CCS) technology is co-deployed with hydrogen production from SMR emissions reductions of up to 90% could be possible [323]. *On the Road in 2035* estimates lifecycle GHG emissions for FCEV operating on hydrogen produced from distributed steam methane reformation and compressed to 10,000 psi are 119 g CO_2eq per kilometer (km), which is equivalent to a 33% reduction from a 2035 CV[68]. As would be expected, GHG estimates for FCEVs operating on renewable hydrogen demonstrate increased GHG reductions. Granovskii et al. (2007) estimates that substituting renewable hydrogen in FCEVs for gasoline in CVs reduces GHG emissions by 12 to 23 times and 5 to 8 times for hydrogen derived from wind and solar respectively[331]. Further, the authors estimate air pollution reductions of 76 and 32 times for FCEVs operating on wind and solar derived hydrogen.

Net transportation sector emissions impacts resulting from FCEV deployment are directly related to the level of vehicle fleet penetration reached and the mix of deployed hydrogen supply chain strategies to meet the resultant fuel demand. Complete replacement of gasoline vehicles with FCEVs has been shown to offer significant GHG reductions in total transportation emissions nationally (14-23%) and across the passenger vehicle fleet regionally (84%), including scenarios with reliance on fossil fuel supply chain strategies[293, 332]. Although a 100% replacement level of all on-road vehicles is unlikely in the 2050 horizon, these results are illustrative with regard to potential impacts. Future penetrations of FCEVs in the LDV fleet are dependent on various factors (i.e. future policy and subsidies, meeting of technology targets, etc.), and high penetrations have been reported as feasible in the literature. A comprehensive study conducted by an NRC committee estimated that the maximum practical number of FCEVs that could be on the road in 2050 was about 200 million, representing 60% of the LDV fleet, with a corresponding reduction in GHG emissions of 60% and 22% relative to a LDV reference case and total transportation emissions respectively [326]. Deployment and subsequent development of an advanced hydrogen infrastructure can offer efficiency gains and facilitate greater usage of less carbon intensive hydrogen pathways. A U.S. DOT analysis assuming the same market penetration levels reported reductions ranging from 52-74 MMT CO_2eq per year in 2030 for hydrogen produced from distributed SMR and 388-474 MMT CO_2eq per year in 2050 for hydrogen produced from a generic advanced pathway.

2.1.1.2 Plug-in Hybrid Electric and Battery Electric Vehicles (PHEVs and BEVs)

In the near term PHEVs are likely to facilitate electricity as a vehicle fuel. PHEVs alternate between a battery supplied with grid electricity and an on-board ICE which displaces a portion of gasoline consumption. PHEVs have a battery supply that provides an electric range typically 30 to 60 km [333]. Because most LDV trips cover short distances it has been estimated that a PHEV with a 40 mile battery range (PHEV40) can operate on electricity for approximately 60% of vehicle miles driven[334]. Further, PHEV efficiencies could be 74% and 58% higher than 2005 and 2035 CVs, respectively.

Technical, economic and social barriers exist to PHEVs gaining a considerable market penetration. Currently technical challenges include the performance of batteries, reliability, and cost among others. BEV and PHEV technology gaining market success will depend on further advancement in battery performance including reduction in battery costs and improvements in battery lifetimes. Due to the availability of the ICE, PHEVs do not face range limitations that BEVs face.

Benefits of BEVs include no direct air emissions and highly efficient drive trains. BEVs face near-term challenges of high costs and energy storage issues. Currently the Nissan Leaf is the only commercially available BEV with a reported range of 100 miles, although testing has reported a 73 mile range under real-world conditions[337]. With current battery technology, BEVs with ranges of greater than 200 miles would be heavier and costlier than competing CVs and HEVs as a consequence of the large required battery system [68]. Also presenting a challenge to consumer acceptance is associated charging time with a 200 mile range, estimated to be 7 to 30 hours if 240V or 120V current is used. If range limitations are not overcome BEVs could be appropriate for short distance commuting, particularly in urban settings. BEVs could also be useful second cars. If battery performance improvements are realized and vehicle costs reduced it is possible that BEVs could offer ranges similar to today's CVs, allowing high marketplace penetration.

Electricity has benefits relative to other alternative fuels in that it has an existing transmission and distribution infrastructure and is currently widely available. Studies have shown that that penetrations (20%) of BEVs could be supported by the existing grid without the need for further capacity additions [338]. An analysis by Kintner-Meyer demonstrated that up to 73% of the existing LDV fleet could be supported as PHEVs from the existing power grid with a gasoline displacement potential of 6.5 million barrels of oil equivalent per day [339]. Increased deployment of electric vehicles could also allow utility companies to use excess power capacity during off-peak hours or when excess renewable power is available. Regional work conducted for the Northeastern U.S. reported that unutilized, nighttime base-load coal power available from the current grid could meet the charging demands of a 20% penetration of PHEVs if vehicle charging occurred at night[340]. Of course, using coal power to charge PHEVs would not necessarily reduce GHG emissions. Without control or incentives for charging at specific times electric vehicle use could increase peak power demand, highlighting the importance of temporal charging impacts. A light duty vehicle (LDV) fleet comprised of electric vehicles could be used to manage the grid and provide regulation of supply to load intermittency issues associated with renewable such as wind power [148]. The dispatchable load associated with PHEV charging could provide other benefits such as reducing overall systems cost by increasing the minimum system load, increasing the utilization of base load units during off-peak hours, and decreasing plant cycling if charging profiles are optimized [341, 342].

Electricity Greenhouse Gas Impacts

Power generated for vehicle charging can come from low emitting sources including renewable and nuclear energy, moderate emitting sources such as natural gas, or high emitting sources including coal. Thus, GHG emissions per mile for electric vehicles vary depending upon the region, time of day, and season. Life cycle emissions depend upon the carbon intensity of grid power for charging, the GHG life cycle intensity of the displaced fuel used in the ICE (PHEV), and emissions associated with battery manufacture[343]. Operation in all-electric mode reduces direct emissions from vehicles and upstream emissions from petroleum fuel production and distribution. Consequently, increasing all-electric ranges will increase emissions benefits unless the electricity is carbon intensive, such as from coal power plants. Scenario-based projections of grid evolution, power dispatch and geographic generation and distribution can model how sources of electricity could be generated, providing a more robust analysis of GHG impacts.

The literature that assesses emission impacts of EV deployment includes PHEVs due to the potentially increased near-term deployment relative to BEVs. A series of studies showed that PHEVs increased electricity consumption and emissions from the power sector in the absence of carbon policy; however these were offset by direct vehicle emissions reductions that resulted in a net reduction in CO_2 emissions[344]. A study that examined nine national PHEV implementation scenarios spanning three different carbon intensity values for electricity concluded that even for the worst case GHG emissions would be reduced relative to a baseline scenario [345]. Estimates of GHG emissions for PHEVs using current carbon intensities for electricity estimated net emissions could be reduced by approximately 15% using the average 2006 U.S. power generation mix and 33-68% in California [346, 347]. Studies analyzing GHG emissions of PHEVs with electricity from non-fossil (i.e., renewable energy sources, nuclear power) or less carbon intensive pathways have shown large reductions (up to 63%) compared to CVs and HEVs [148, 339, 346, 348-350]. A subset of studies is displayed in Table 1. The reductions are greater if a low carbon liquid fuel such as E85 (20-80%) or renewable hydrogen (25-90%) is used in place of petroleum or diesel fuel [348, 349].

Table 1: GHG Impacts Associated with PHEV Deployment Reported in the Literature

Study	GHG Reduction	PHEV Range	Charging Electricity Assumption
Kliesch 2006	57.6%, 78.3%	PHEV 40	2006 U.S. average gen. mix, 2006 CA average gen. mix
Kintner-Meyer 2007	0-40%, 27% U.S. overall	PHEV 33	Regional/U.S. average generation mix
Kempton 2007	60% Eastern U.S.	N/A	Off-shore wind generation
EPRI 2007	40-65%	PHEV 10,20,40	Varied carbon intensity of generation mix
Samaras 2008	38-41%	PHEV 30-90	U.S. average generation mix
Stephan 2008	25-50%	N/A	Current and expected future dispatchable units
Elgowainy 2010	-10-15%	PHEV 40	Large coal share generation mix
	25-40%	PHEV 40	Large NGCC share generation mix

	20-25%	PHEV 40	U.S. average generation mix

BEV GHG impacts are determined by the carbon emissions and efficiency of generating and transmitting the electricity used for vehicle charging. The U.S. DOT reports that using the current national electricity grid mix and assuming an efficiency of 0.4 kWh/mile, BEVs today reduce emissions by 33% relative to CVs[335]. DOT estimates based on fuel-specific life-cycle emissions report a 6% increase in emissions for coal derived power, a 45% decrease for natural gas, and a 99% reduction for nuclear power.

Electricity AQ Impacts

The magnitude and direction (positive or deleterious) of AQ impacts from EV deployment depend upon the regional grid mix, temporal charging profiles, and regulatory constraints [340, 351, 352]. Wide spread use of PHEVs and BEVs can change the way electricity is managed in the transportation and power sectors, shifting the spatial and temporal emissions of pollutants from distributed tailpipes to centralized generation locations. Central facilities can be located inside or outside of urban airsheds and spatial shifts in emissions could increase or decrease population exposure. Replacing CVs with PHEVs reduces direct pollutant emissions by reducing startup and total operation time of the ICE, although life cycle considerations must be made [292, 343, 353]. When power emissions are considered in a life cycle approach, increases and decreases in NO_x emissions have been shown dependent on whether or not increased coal generation is used to meet demand, however emissions of SOx did not increase as existing AQ standards prevent it

[339]. Decreases in CO and VOCs have been associated with EVs. Increases in total PM have been shown for EVs but spatial shifts result in decreases in urban PM emissions[292].

Spatial and temporal AQ analysis performed using detailed atmospheric transport and chemistry models assessing PHEV deployment have generally shown small but significant, spatially dependent improvements in AQ in tandem with localized areas of worsening. Negative AQ impacts are attributed to shifting emissions from tail pipes to high-emitting coal power plants. A comprehensive study on a national scale found high PHEV deployment led to small but significant decreases in ozone (61% of the study area) and PM; although ozone concentrations increased locally (1% of the study area) if no further constraints are assumed in future power generation[351]. Similar trends have been observed on regional scales assuming 100% PHEV penetration in the LDV fleet [352, 354]. Other studies have shown both improvements and worsening of AQ indicating overall impacts are complex and related to both grid dynamics and atmospheric processes [340]. Table 2 summarizes the literature for both primary and secondary pollutant impacts from the use of PHEVs, highlighting the spatial and temporal complexity of evaluating impacts.

Table 2: Air Quality Impacts of PHEV Deployment Reported in the Literature

Study	NO_x Emissions	SO_x Emissions	PM Emissions	PM Concentration	Ozone Concentration
EPRI 2007	D	D	I-10%	D	D -61% area I -1% area
Kintner-Meyer 2007	I- Coal D-No coal	I-75% area	I-67% area	N/A	N/A
Parks 2007	D- small	D-some I-some	N/A	N/A	N/A
Thompson 2009	I-night D- day	I-Potential	I-Potential D- Potential	I- SOx related D- Potential	D- 2-6 ppb

					Localized I < 8 ppb
Brinkman 2010	D	N/A	N/A	N/A	D - <2-3 ppb I- small

*** I denotes an increase in emissions; D denotes a decrease in emissions. Area indicates the total area considered in the study**

2.2 POWER GENERATION SECTOR

The generation of electricity in the U.S. greatly contributes to the aforementioned difficulties; contributions that are heightened by the magnitude of domestic electricity demand and include emissions of GHGs and pollutants with resulting AQ detriments. It is clear that any meaningful U.S. GHG mitigation efforts contain methods to support and achieve substantial changes to current domestic electrical supply chains. On a sector basis, conversion of primary energy carriers to electricity accounts for the largest current share of total CO_2 emissions (35%) in the U.S. [81]. The ability of the power sector to become decarbonized is then essential in attaining climate goals due to its status as the highest emitting U.S. sector and to support reductions via electrification of other sectors (e.g., transportation, industrial, building demands)[82]. Moving forward the power sector is likely to receive a heavy focus in established U.S. GHG mitigation policies or plans, perhaps even disproportionately so relative to other sectors. This is because (1) power is the highest emitting sector currently, (2) many strategies exist to generate electricity with little to no GHG emissions, (3) electrification can achieve reductions from additional sectors, and (4) emissions from many sector sources, e.g., large power plants, are concentrated and thus more suitable for emissions control applications, including carbon capture and sequestration [83]. Demonstrating this, the U.S. Environmental Protection Agency (EPA) is in the process of developing guidelines for regulating CO_2 emissions from both existing and future power plants[84].

In line with GHGs, energy-related activities are responsible for the bulk of pollutant emissions driving many current U.S. regional AQ concerns, including ground level concentrations of ozone and $PM_{2.5}$. The combustion processes associated with many conventional generation technologies and fuels result in atmospheric releases of both gaseous and particulate pollutants. Demonstrating the substantial contribution, in 2005 energy related-sources were responsible for 95% of anthropogenic nitrogen oxides (NO_x), 92% of anthropogenic sulfur dioxide (SO_2), and 10% of coarse particulate matter (PM_{10}) emissions in the U.S.[14]. Specifically, electricity generation is a key contributor of emissions. For example, stationary fossil fuel combustion in the power sector is by far and away the largest emitter of SO_2, responsible for 83% of total 2010 emissions[81]. In addition, emissions of GHGs and pollutant species are highly correlated as a result of shared generation sources; most importantly fossil fuel combustion. The deployment of alternative low carbon technologies and fuels, in addition to changes in demand, will alter the magnitude of emissions of criteria pollutant species, including NO_x, PM, volatile organic compounds (VOCs), carbon monoxide (CO) and SO_2. Furthermore, spatial and temporal shifts in emission patterns will influence the formation and fate of secondary pollutants that carry human health consequences, including ozone and $PM_{2.5}$. Thus, an important opportunity exists to simultaneously address U.S. GHG and AQ concerns by deploying alternative, low emitting power generation strategies.

In the absence of targeted and comprehensive adjustment the emissions and atmospheric pollution associated with the U.S. power sector will potentially increase in proportion to the demand for power to support population and economic growth. According to the most current U.S. EIA Reference Case (AEO2014) representing business-as-usual (BAU) progression, total electricity consumption is projected to increase from 3,826 billion kWh in 2012 to 4,954 billion kWh in 2040, an overall growth exceeding 29%. Thus, the U.S. faces a difficult situation entailing a need to dramatically reduce emissions coinciding also with a need for increased total generation.

Currently the central strategy for large-scale power generation comprises the combustion of fuel to produce heat which can be used directly or via the formation of steam to provide mechanical energy to a turbine-driven generator to produce electricity (current nuclear generation also follows the steam turbine method however heat is produced via nuclear fission rather than combustion). Additionally, fuel combustion to power a turbine or reciprocating engine that directly drives an electrical generator is often utilized, particularly in smaller, distributed-scale applications. In accordance with all energy sectors, current fuels most utilized in electricity procurement are fossil-based, with coal and natural gas accounting for 37% and 30% of total U.S. generation and oil and other liquids meeting an additional 1%[85]. The majority of the remaining U.S. power generation share is provided by lower CO_2 emitting technologies, including nuclear power (19%) and renewable energy resources (12% in aggregate).

Natural gas use for domestic power generation has experienced rapid expansion in recent years and additional growth is expected, largely as a result of sustained development of shale gas production[85]. In recent years coal and natural gas have experienced divergent trends with the percentage of coal declining in response to reductions in natural gas costs and environmental regulations targeting emissions from coal power plants. In the AEO2014 gas-fired generation surpasses coal as the dominant power sector fuel by 2035. Nevertheless, despite a reduced market share the total amount of coal consumed annually increases and generation from coal meets an important fraction of electricity demand in 2040. The AEO2014 projects generation with non-hydropower renewable energy will increase from 2012 to 2040 (1.9 quadrillion Btu to 4.5 quadrillion Btu), with wind and biopower respectively accounting for 39% and 27% of renewable growth. However, renewable energy provides a modest share (i.e., 16%) of total U.S. power generation in the Reference Case. Reflecting societal concerns, nuclear power capacity remains level to 2040 (i.e., 102 GW) and, despite a 5% increase in total generation, the share of overall generation falls from 19% in 2012 to 16%. It should be noted that the reference case assumes no measures explicitly targeting significant mitigation of GHG emissions and implementation of such strategies (e.g., carbon tax, cap-and-trade) could result in heightened interest in low carbon options, including nuclear capacity [85]. Therefore, both the current and expected mix of U.S. power strategies is largely comprised of combustion strategies that generate GHG and pollutant emissions when fossil fuels are combusted.

Historically, coal has represented the dominant fuel for power generation due to large domestic resources translating to high availability and low costs and the ease with which we can move coal throughout society due to its high gravimetric and volumetric energy density. However, continued use of coal represents a threat to global climate and regional AQ as coal is associated with a disproportionate contribution of emissions relative to capacity levels. Coal has the highest carbon content of commonly utilized fuels, giving rise to the largest output rate of CO_2, and coal combustion results in emissions of additional forcing gasses including CH_4 and N_2O[86]. It follows then that the largest single source of U.S. CO_2 emissions in 2010 was the stationary combustion of coal, responsible for 81% of power sector emissions (the bulk of the remainder arose from natural gas, emitting 18% of total sector CO_2)[81]. Coal-fired power plants are also the largest source of SO_2 in the U.S., contributing 60% of total U.S. emissions, and are second only to motor vehicles in emissions of NO_x, accounting for 18% of the U.S. total[70]. Coal plants also emit primary PM and air toxic emissions, including lead and mercury [70]. Furthermore, the atmospheric oxidation of SO_2 and NO_2 emissions results in acidic secondary PM formation. Thus, displacing generation from new or existing coal-fired electric generators is essential to meeting future GHG mitigation goals and will also obtain important pollutant emission and AQ co-benefits[33].

The current global and domestic reliance on high-emitting fossil fuels for power is the key driver of sector emissions; and the widespread use of novel technologies and fuels will be required to achieve deep mitigation. While most current strategies are responsible for substantial emissions a diverse range of approaches offer the potential for meeting future needs in tandem with reduced emissions. Low-carbon technologies with the ability to generate significant amounts of electricity include renewable energy resources including technologies associated with wind, solar, geothermal, ocean, hydropower, and biopower, nuclear energy, and fossil generation equipped with carbon capture and storage (CCS). Additionally, switching from higher to lower emitting fossil fuels (e.g., coal to natural gas) can potentially represent an effective mitigation option. Furthermore, methods to reduce electricity demand via improvements in the efficiency of power generation, transmission, distribution and the end-use of electricity can result in less primary generation; lowering total fuel consumption and reducing emissions while meeting demands. Thus, options to curb emissions from the sector exist even at the dramatic scale required. However, such shifts represent an extraordinary societal challenge, with a host of economic, technological, and behavioral barriers to overcome.

Considered technologies and fuels vary widely with regard to technological maturity. Some strategies are mature (e.g., nuclear power, wind) and can assist in off-setting emissions in both the near- and long-term while others will require technological advancement before contributing important reductions (e.g., advanced biopower pathways, CCS). Significant uncertainty exists with regards to numerous aspects of power sector GHG mitigation, including future technology development, fuel costs, and policy implementation. Further, many studies have demonstrated that no one technology can provide necessary reductions, and an optimized portfolio of low carbon strategies will be required to meet climate goals [41, 53, 82].

To fully cognize emission impacts from GHG mitigation strategies in the power sector an understanding of the complex system spanning electricity generation, transmission/distribution, and end-use must be developed. Regional electricity grids are characterized by different combinations of base load, load following, and peaking units operating on an assortment of fuels that must be dispatched to appropriately balance load and demand while achieving minimization of generation cost. Generally, base load units are designed to operate continuously at capacity with low operational costs and include coal-fired, nuclear, natural gas combined cycle (NGCC), geothermal and hydrological (hydro) power plants[87]. In contrast, load following and peaking units are generally natural gas or petroleum-fired and operate with enhanced flexibility to fill the remaining gap, with peaking units operating only during peak demand periods. This task is complicated by the variability of demand which fluctuates instantaneously, hourly, daily, and seasonally and the current relative lack of large-scale energy storage capability which necessitates complex dispatch strategies based upon demand forecasts, unit availability, costs, etc. [87]. Additionally, generation from certain mitigation strategies, i.e., wind and solar power, can be considered non-dispatchable in that resource variability currently inhibits reliable availability. The generation and release of GHG and pollutant emissions varies by generator and is impacted by many factors including technological attributes, fuel and fuel quality, efficiencies, generator age, applied pollutant control technologies, and others.

In addition to the specific mix of generation technologies and fuels, the dynamics of grid systems directly impact emissions attributable to electricity. The term marginal generation is generally used to describe the plants comprising the last units dispatched to meet the load for a given time and thus generally the first to respond to any new events[88]. Mitigation strategy deployment often then impacts marginal generation rather than base load, which are normally categorized by peaking and/or load following units operating on natural gas or petroleum fuels, i.e., a simple cycle gas turbines[87]. Such units are generally characterized by lower efficiencies and higher emissions than systems regularly used for base load power provision, e.g., a NGCC plant. Consequently, in many regions of the U.S. periods of peak power demand coincide with the utilization of generators with elevated emission rates leading to increased emissions of both GHGs and pollutants[89]. Therefore, strategies that can offset marginal power demand and/or replace needed generation can achieve enhanced emission reductions and potentially higher GHG and AQ co-benefits.

The complexity of the structure and operation of regional power grids translates to emissions, which can vary spatially (e.g., by region or specific location) and temporally (e.g., by season or time of day). As such, it may not be appropriate in all cases to simply compare embodied emissions between generation strategies. For example, replacing a natural gas turbine with wind turbines would show a significant reduction in total GHG and pollutant emissions as wind power is free of point-of-use emissions. However, such an outlook would fail to account for the response of the grid to the intermittency of wind generation which can impact emissions negatively from additional generators; potentially lessening AQ benefits and/or introduce and novel AQ problems. Similarly, deploying a wind turbine in the NEUS may show a larger AQ benefit than utilizing the same turbine in California due to a higher amount of coal generation being offset; however superior wind resources in California may support improved performance and lower costs. Such considerations are essential for properly assessing the regional AQ impacts of technological shifts.

2.2.1 Renewable Resources

One of the most widely proposed groups of alternative power generation technologies for mitigating emissions of criteria pollutant and GHGs includes renewable resources. Renewable power involves the provision of electricity from sources that are naturally replenished and is conventionally defined to include technologies power generation associated with wind, solar, geothermal, hydro, biopower (i.e., biomass and biogas), and ocean resources. For various reasons, including energy independence/security, improved AQ and climate change mitigation, the amplification of renewable power has garnered significant attention and is being targeted by policy at various levels of government, e.g., 30 states and the District of Columbia have Renewable Portfolio Standards (RPS) or similar laws[85].

The renewable power pathways considered include solar (i.e., PV an CSP) wind, geothermal, and biopower because (1) they are expected or able to contribute significantly to future U.S. power needs, (2) they have potential for impacts on AQ and GHGs, and (3) there is a significant body of research regarding emissions from each. It should be noted generally that common conversion pathways and technologies are more prominently considered, e.g., algae fuels could be important in the future but requires significant technical and economical advancements thus currently utilized biopower pathways receive more attention, including solid biomass and biogas resources.

Technologies comprising renewable pathways and, subsequent associated challenges and impacts, are extremely diverse and wide ranges are reported across technologies for different indicators (e.g., cost, performance, environmental impact)[90]. In general, renewable technologies have higher associated costs and lower power densities than current fossil alternatives, and many (e.g., wind, solar) are inherently variable (i.e., seasonally, diurnally) and require the co-deployment of complementary technologies to achieve acceptable systems-level dynamics (balancing of generation with fluctuations in load demand) [41, 91, 92]. Due to these and other challenges, dramatically increasing the capacity of renewable power will require key changes to the current U.S. power grid due to the intermittency, spatial distribution, and scalability of resources [35, 93, 94]. Further, many locations with high resource potential are not adjacent to population centers, necessitating upgrade of existing and/or construction of new transmission infrastructure[95, 96]. Demonstrating the impact of these, and other characteristics, a study examining different renewable pathways required to meet an 80% reduction in GHG emissions from 1990 levels in California concluded that a high-renewable scenario required increased installed capacity, transmission infrastructure, and energy storage relative to scenarios involving high-nuclear or CCS[82].

AQ and GHG benefits from renewables will be directly related to the total amounts installed and array of products utilized in U.S. energy sectors (i.e., in addition to power renewable resources can provide fuels and process heat). Thus, AEO2014 Reference Case projections represent a conservative estimate. The available resource base for renewable energy is vast, with potential feasible penetration levels far in excess of current levels. It has been reported that accelerated deployment of a mix of technologies, including hydroelectric, could provide 20% of total U.S. electricity in 2020 and non-hydroelectric pathways could meet 20% of total domestic generation by 2035[35]. Examination of potential contributions towards meeting domestic climate stabilization scenarios produced estimates ranging from 10 to 50% of total primary energy and up to 70% of total power generation by 2050[97]. A study examining the implications and challenges of high (30-90%) renewable electricity generation concluded that 2050 U.S. electricity demand could be met with 80% renewables and a mix of flexible conventional generation and grid storage, transmission additions, increased load responsiveness, and power system operational changes[94]. A key finding from the study was that multiple technology pathways were available to reach the 80% level and that power supply and demand could be balanced in every hour of the year for all regions. Though the resource base could support a complete (100%) domestic reliance on renewable energy, it is unlikely in the study horizon due to key hurdles (e.g., intermittency, cost, transmission constraints). For example, renewables could meet a maximum of 74% of California energy in 2050, despite the State's substantial and diverse potential resource base, and such a level would require effective forecasting, rapid and significant advancements in energy storage and dramatic load shifts from smart EV charging[82].

Generally, any available low marginal cost renewable power will displace higher marginal cost fossil generation and, as a result, traditional EGUs will respond by reducing output. However, current grid operation must accommodate all load demand conditions irrespective of the availability of renewable power and, due to the rapidity at which intermittent resources come online and/or dropout, additional reserve capacity must be constantly available. As a result, renewable power dictates rapid responses by traditional EGUs, including the co-deployment of complementary technologies and/or the dynamic operation of existing generators; both of which can have important emissions consequences that should be considered for a complete understanding of impacts. Thus, for renewable energy AQ and GHG impact assessment it is important to distinguish between those renewable resources whose intermittency necessitates additional measures to maintain system reliability and those with some degree of dispatch capability.

2.2.1.1 Intermittent Renewable Resources

Solar Power

Power generation from solar energy includes both photovoltaic (PV) and various forms of concentrated solar power (CSP). PV involves the direct conversion of solar irradiation into electricity and includes a wide range of technologies and semiconductor materials (e.g., multicrystalline silicon, monocrystalline silicon, ribbon silicon, thin-film cadmium telluride (CdTe)). PV installations have a wide range of sizes and applications, and are suitable for residential-, commercial-, and utility-scaled deployment. Some CSP systems use lenses to focus solar power onto small multi-junction PV cells. These systems must track the sun in order to keep the concentrated solar power focused upon the PV cell. The more typical CSP systems involve large mirror arrays that concentrate solar irradiation onto a heat transfer medium used to generate steam directly or via a heat exchanger to drive a turbine. Current designs include parabolic troughs, constituting roughly 90% of current CSP, and linear Fresnel reflectors, power towers, and Stirling thermal systems[98]. Further, some plants incorporate thermal energy storage (TES) or complimentary back-up generation to smooth output fluctuations from cloud episodes and to extend generation. A key benefit of CSP is the provision of dispatchable and potentially base load power dependent on the inclusion of TES systems with appropriate temporal limits. Barriers to solar power deployment include high costs, intermittency/variability of solar irradiation, and for some solar power (i.e., CSP, utility-scale PV) transmission from remote locations to population centers. Advanced PV technologies (e.g., organic solar cells, thin film PV modules, concentrating PV, nano-scale technologies) could potentially address barriers by increasing conversion efficiencies and lowering costs.

Solar Power GHG Impacts

Emissions per unit generated power vary with respect to individual technologies (i.e., achieved efficiencies, required manufacturing processes) and regional deployment characteristics (i.e., insolation, meteorology); which impact total emissions and power output. Estimates in the literature report values of 19-95 for various thin film PVs (e.g., CdTe, a-Si, CIS) and 20-104 g CO_2eq/kWh for crystalline technologies (e.g., m-Si), although prospective advances in manufacturing could lower emissions to levels comparable to other renewable technologies [99-111]. For PV systems the bulk of emissions occur upstream during processes involved with raw material acquisition and material processing, e.g., emissions from using fossil energy in the production of materials for solar cells, modules, and systems, and from smelting, production, and manufacturing facilities. Further, total emissions are correlated to the generator grid mix localized to manufacturing facilities and vary by pathway for material and fuel processing. Demonstrating the direct relationship, CdTe is currently the least carbon intensive PV technology as manufacturing processes require lower total energy input during module production than other PV forms. Though life cycle emissions for PV technology are among the highest for renewable technologies the bulk of reported values are considerably lower than any coal or natural gas technology.

Though less information is available for CSP, studies have reported a range of 12-284 g CO_2e/kWh for various CSP technologies with upper range values for facilities that incorporate natural gas-fired complimentary generation [112-117]. Highlighting the significant fraction contributed by gas generation to total emissions, studies reported a range of 30-149 g CO_2e/kWh for parabolic trough technology when 3-25% natural gas back up was considered and 26-28 g CO_2e/kWh for solar only processes[116, 118]. Life cycle emissions for a 50 MW trough plant with 7.5 hours of TES were reported at 33 g CO_2e/kWh[115], which is similar to the reference plant design in Reference [116]. Life cycle GHG emissions for CSP technologies are impacted by plant design including utilized cooling technologies, fuel and operating characteristics of any backup generation, and heat transfer medium [116]. Deployment of wet cooling is associated with the lowest life cycle emissions due to increased plant performance; however constrained water resources in many areas with high CSP potential may necessitate the use of dry- or hybrid-cooling strategies. If TES is utilized, emissions are impacted by both the overall design and the specific heat transfer medium (e.g., synthetic vs. natural thermal salts). In general synthetic salts are associated with higher life cycle emission than mined salts while thermocline design reduces emissions relative to a two tank system[116]. Despite having life cycle emissions in the higher range for renewables, particularly if gas-fired backup is utilized, CSP emissions are roughly 3 to 7% of gas- and coal-fired generation[33].

Further, emissions reductions from solar thermal technologies can be achieved through strategies outside of power generation, including HVAC system designs, solar water heating and cooking. For example, solar water heating systems can displace natural gas combustion required for conventional water heating and deployment of such systems is being actively pursued in California and is projected to yield reductions of 0.14 MMT CO_2eq in 2020 [119]. In addition, solar heat could potentially be utilized directly in various processes and sectors including industrial and commercial applications to displace fossil fuel use.

Studies have demonstrated important potential for GHG mitigation from large-scale deployment of solar power, highlighting the importance of solar technologies towards meeting future climate change goals [41, 120-122]. It has been proposed that 69% of national electricity could be generated by PV and CSP in 2050, displacing 300 coal-fired and 300 gas-fired plants and mitigating 1.7 billion tons of GHG emissions [122, 123]. A fleet of electric vehicles charged from the grid would mitigate another 1.9 billion tons from gasoline vehicles, lowering total domestic GHG emissions in 2050 to 62% below 2005 levels. Though the study takes highly favorable outlooks on almost every aspect of future solar technologies and should be considered an upper bound, it does illustrate the large GHG abatement potential for solar power. A more conservative, though still optimistic approach projects that 30-80 GW of CSP and 200 GW of PV could be deployed by 2030, annually mitigating 73-143 MMT CO_2eq [120].

Solar Power Air Quality Impacts

Since solar power technologies generate no point-of-use pollutants, life cycle emissions are determined by manufacturing processes and the energy mix used to meet manufacturing, delivery and installation demand[34]. Rates of SO_2 emissions for PV installations in the U.S. have been reported to vary from 158 to 540 mg/kWh for a range of technologies[34]. A review of PV production data from 2004 to 2006 from four major commercial PV types (multicrystalline silicon, monocrystalline silicon, ribbon silicon, and thin-film cadmium telluride) reported a range of SO_2 emissions from 158 to 378 mg/kWh[105]. Life cycle emissions of NO_x for PV technologies are reflected by the grid mix of utilized energy in material production and have been estimated to total between 40 to 260 mg/kWh[34, 105]. A review of 5 LCAs for PM emissions associated with PV electricity generation in the U.S. found only one that was greater than 100 mg/kWh, reported to be 610 mg/kWh[34]. The high range in the studies represents an area with low insolation rates and a greater reliance on coal for electricity generation, while regions more favorable to PV deployment or with cleaner grid mixes would achieve higher efficiencies and lower emissions. Natural gas back-up generation integrated into CSP design could produce direct emissions; however the spatial and temporal operation of such facilities would lessen opportunities for worsening of urban AQ as areas with high CSP potential are often in remote locations (e.g., Mohave desert of California) and many current and proposed CSP facilities are located far from population centers.

Future large-scale deployment of solar technologies is not expected to have any negative AQ impacts. Despite life cycle pollutant emission estimates for PV and CSP being highest among renewables, reported values are significantly lower than any fossil source and offer significant mitigation benefits when displacing fossil power. For example, it is estimated that a minimum of 89-98% of air emissions (GHGs, criteria pollutants, heavy metals, and radioactive species) associated with electricity generation could be avoided if electricity generated from PV replaces average grid electricity[105]. Localized impacts are possible if PV manufacturing facilities are located in urban air sheds and emissions from industry related activities should be considered in plant permitting and siting. However, the majority of current global production occurs outside of the U.S. lessening concerns over localized or regional AQ impacts[124]. However, variability inherent with electricity generated by PV, and to a lesser degree CSP, requires back up generation that has emissions and can potentially lead to localized AQ concerns.

Wind Power

The conversion of wind energy into electricity through devices including turbines is well recognized as an environmentally beneficial source of power[125]. Wind power technologies are commercially developed and currently provide power on a utility-scale in many U.S. regions, including large wind farms consisting of numerous individual turbines. Additionally, the economics of wind power can be favorable relative to other renewable technologies[39]. However, the variable nature of wind resources can represent a negative factor for power grid reliability[126]. Thus, impacts from systems-level dynamics (e.g., altered generator operation, complementary generation) must be considered in assessments of total impacts and will be addressed in subsequent sections.

Wind power has the technical potential to contribute significant amounts of electricity to the future U.S. grid, e.g., the technical wind resource is estimated at 10,000 GW in the contiguous U.S. [94, 127] and 11 thousand TWh/year could be produced from the continental U.S. wind resource base, roughly 2.75 times total estimated U.S. generation in 2007[128]. Studies have demonstrated that expansion of wind power technologies is feasible and could meet much greater amounts of U.S. electricity than is projected in AEO2014. A 2007 study modeled 20% of U.S. electricity supply from wind power by 2030, representing a growth of approximately 300 GW of new wind power capacity, 50 GW of which would be installed offshore [129]. The report concludes the significant costs, challenges, and impacts associated with high capacities of wind generated electricity are offset by substantial benefits in many areas, although such a scenario is unlikely to occur without a major national commitment to renewable resources. Further, the study considered additional capacity of wind power only and not the impact of EE measures. It is more likely that a combination of renewable energy technologies will be deployed in concert with increased efficiency measures if high national priority is placed on lowering electricity GHG emissions. A study considering this more plausible outcome determined wind energy would provide the greatest amount of renewable electricity by 2030, with new capacity additions of 245 GW needed to meet a 20% penetration after efficiency improvements were considered[120]. Other estimates have suggested that 100 GW of new wind capacity could be added to the U.S. electric grid by 2030, off-setting substantial amounts of CO_2 emissions[41]. The results of these studies demonstrate that high penetrations of wind energy into the future U.S. electric grid are feasible, offering the potential for significant reduction of emissions.

Wind Power GHG Impacts

There is no question that wind power represents a foremost GHG mitigation strategy. Wind generated electricity involves no point-of-use emissions and very low life cycle emissions of GHGs and criteria pollutants. Emissions do occur during upstream processes, including manufacture and transport of turbines, which have been estimated to contribute 84.4% of total emissions[130]. Life cycle emissions per unit generation are site specific and dependent on many factors including turbine size, wind conditions, and turbine lifetime. For instance, off- shore turbines have a higher carbon footprint than onshore turbines of equal capacity factors due to emissions associated with foundation, connection, and construction of off-shore turbines. Similarly, larger wind farms located in areas with large wind resources have lower emissions than wind farms located in areas with poorer wind resources. Estimated LCA emissions for wind turbines range from 3-40 and 3-22 CO_2e/kWh for onshore [38, 108, 131-139] and off-shore [38, 109, 131, 134, 139-142] turbines respectively, although turbine technology improvements could further reduce net emissions. Despite some variation, GHG emissions for wind technologies are among the lowest for all renewables and any replacement of fossil generation with wind power has been shown to have important GHG reduction benefits. For example, relative to an average value for fossil power generation, total net avoided GHG emissions have been estimated at 35,265 and 122,961 tons for a 850 kW and 3.0 MW turbine over a 20-year service life, respectively[132]. In addition, wind power avoids other harmful environmental impacts associated with fossil generation including mining, drilling, and pollutant generation for both air and water.

Very low life cycle emissions associated with wind power can translate to important GHG reductions depending on future wind turbine capacities in the U.S. The electricity generated from the entire fleet of U.S. wind turbines installed and operating at the end of 2009 was estimated to offset over 57 MMT of CO_2 annually based on the 2009 average U.S. grid mix[143]. Meeting 20% of U.S. power with wind reportedly displaced about 50% and 18% of electric utility natural gas and coal consumption respectively, reducing emissions by 825 MMT CO_2 (25% from a no-new-wind scenario) in 2030 and achieving cumulative reductions of 7,600 and 15,000 MMT in 2030 and 2050, respectively[144]. Similarly, as part of a portfolio of renewables and energy efficiency measures wind power could provide mitigation of 1100 to 1780 MMT of CO_2eq per year with an equivalent penetration level[120]. A study examining the feasibility of meeting 20% and 30% of the projected annual electricity requirements in 2024 for a subset of the eastern U.S. electrical grid reported reductions in CO_2 emissions of 4% and 18% respectively[145]. The increased reduction from the 30% scenario resulted from higher implementation of natural gas EGUs compared to coal due to grid flexibility requirements. It was estimated that supplying 20% to 30% of the electricity needed for the U.S. portion of the Eastern Interconnection would necessitate the deployment of approximately 225 to 330 GW of wind generation capacity. A similar study examining the operational effects of high penetrations of wind in a subset of the western electrical grid concluded that meeting 30% and 5% of the energy needs of the system with wind and solar technologies respectively was feasible and could potentially reduce 45% of CO_2 emissions annually in 2017[146]. The deployment of large, offshore wind resources in the NEUS to take advantage of strong winter offshore winds could attain important emissions reductions benefits from fossil fuel power plants[147]. Similarly, it was demonstrated that off shore wind resources of the Mid-Atlantic region of the U.S. could mitigate up to 68% of the region's CO_2 emissions[148]. **Table 3** contains summaries of reported future penetrations of wind power and corresponding GHG reductions.

Table 3: Studies reporting future penetrations of wind power and associated emissions reductions.

Study	Horizon Year	% of total U.S. Electricity	Emission Reduction
AWEA 2011[149]	2009	1%	57 MMT CO_2
U.S. DOE [144]	2030 2050	20%	7,600 MMT CO_2 15,000 MMT CO_2
ASES 2007[120]	2030	20%	1100-1780 MMT CO_2
NREL 2011[145]	2024	20-30%	4-18%

		(Subset of Eastern Grid)	
NREL 2010[146]		30% & 5% solar (Subset of Western Grid)	45% CO_2, 50% SO_x, 30% NO_x
EPRI 2009[41]	2030	100 GW new	Substantial reductions

Wind Power Air Quality Impacts

Criteria pollutant emissions associated with all life cycle stages of wind generated power are the lowest of applicable technologies, estimated to total less than 100 mg/kWh for SO_2, NO_x, and PM[34]. These values are far below those associated with fossil generation; and are lower than life cycle estimates for other renewable technologies. No large-scale emissions of HAPs or other compounds of concern have been reported for wind turbine manufacturing or installment. Studies of high levels of wind power deployment have also reported important reductions in total NO_x and SO_2[146]. Additionally, wind farms have been estimated to reduce SO_2, NO_x and $PM_{2.5}$ from natural gas power plants[150]. Thus, no direct increases in pollutant emissions are expected to occur in tandem with significant wind power deployment, and AQ impacts would be expected to be positive under the majority of deployment scenarios. Indeed, it has been estimated that wind power has the ability to provide significant human health benefits from reductions in ambient $PM_{2.5}$ levels[150]. The ability of wind power to provide utility-scale power virtually free of emissions yields significant GHG and AQ mitigation potential and justifications of wind-related government subsidies often sites the societal benefits of reducing air pollution[151].

Emission Impacts of Intermittent Renewable Resource Grid Integration

For renewable energy AQ and GHG co-benefit assessment it is perhaps most appropriate to distinguish between those whose intermittency necessitates additional measures (e.g., complementary generation) to maintain system reliability and those with some degree of dispatch capability. Though generation from some renewables (e.g., wind, solar) is void of direct emissions their integration can have system-level impacts with emissions consequences. In general, low marginal cost renewable power displaces higher marginal cost fossil and traditional EGUs respond by reducing output. However, grid operation must accommodate all load demand conditions irrespective of net renewable power and, due to the rapidity at which intermittent resources come online and/or dropout, additional reserve capacity must be constantly available. As a result, renewable power dictates rapid responses by traditional EGUs, including the co-deployment of complementary technologies and the dynamic operation of existing generators; both of which can have significant emissions impacts.

Output from wind power can't be currently forecasted with complete certainty, even for short lead times[152]. Though generally less intermittent than wind, solar power can experience fluctuations in output from unforeseen factors (e.g., cloud cover) that could require additional power come on-line rapidly. Consequently, integrating large capacities of wind and solar power into regional power grids may require additional reserve capacity of compensatory fast-ramping generators (i.e., gas-fired CT plants generally characterized by low efficiencies and high emissions) to be on-line continuously to respond in the case of an unanticipated loss of output. Contrastingly, if available renewable power increases, output from conventional generators may be unutilized though generators continue to operate. Both situations can result in additional emissions of both GHGs and pollutants that degrade some fraction of the expected emissions benefits from renewable resources.

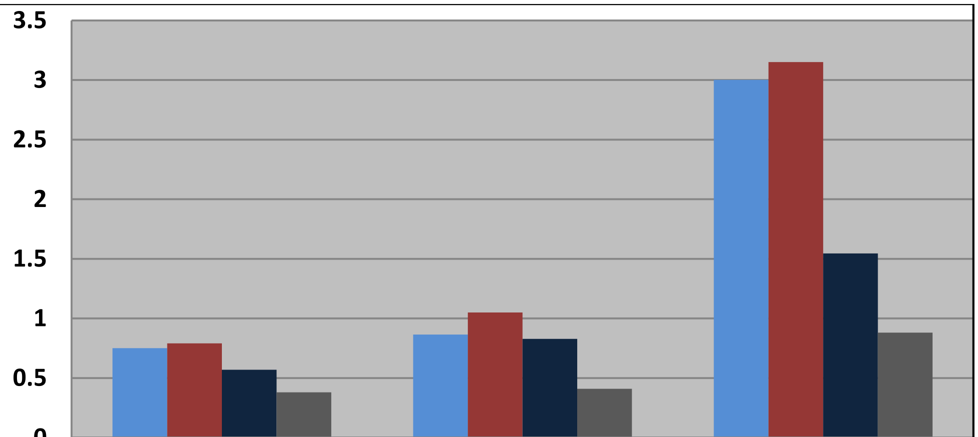
The difference in expected and achieved impacts can be substantial and, when emissions from back up generation are incorporated into estimates, reduced emissions benefits have been observed [153, 154]. Modeling of a wind or solar PV system incorporating two different gas generators over a wide range of renewable penetration levels reported actual emissions reductions were roughly 80% of expected for CO_2 and 30-50% of expected for NO_x [37]. Perhaps most concerning, when dry NO_x control was utilized, a 2 to 4 times net increase in NO_x emissions was reported, suggesting that localized AQ dis-benefits were possible from wind power. However, some have questioned the methodologies utilized in the study as being inappropriate and suggested that the results are at best an upper bound for emissions benefit degradations [155]. Further, other studies have reported significantly lower degradation impacts. A model-based methodology used to evaluate the reserve requirements (i.e., natural gas power plants) necessary to backup large-scale wind power in compensation for wind forecast errors estimated 6% or less degradation in GHG emissions benefits[156]. It has also been shown that renewable generation may have varying impacts on GHGs and pollutants at the species level. A dispatch model analysis reported that due to impacts on power system operation wind generation alone was not effective in reducing SO_2 and NO_x emissions from conventional power plants, but was effective in lowering CO_2 [157]. Though studies offer differing conclusions, the results demonstrate both the complex power system dynamics associated with large-scale renewable power integration and the importance in managing emissions from compensating generators. Further, in contrast to large point source EGUs generally located outside of urban areas, emissions from complimentary generation may occur at distributed locations within urban air sheds which could heighten the importance of AQ impacts[158].

Displaced emissions from the substitution of fossil with renewable generation govern attained GHG mitigation and AQ benefits. Often, it is assumed that emission reductions are proportional to replacing an average unit of regional generation with an emissions free unit, particularly for wind or solar power. However, the system dynamics described above result in the marginal offset emissions not being equivalent to the average emissions for the offset unit of generation[159]. A comprehensive analysis of the impacts from assimilating wind power into the Electric Reliability Council of Texas (ERCOT) grid via modeling based on generator-level patterns of dynamic interactions between wind-generated and conventional EGUs concluded one MWh of wind power offset approximately 1.36 kg of SO_2, .45 kg of NO_x, and more than .75 tons of CO_2 [151]. A key finding of the study was that wind power reduced emissions from coal baseload power plants in the service territory, in addition to the commonly assumed fast-ramping gas plants utilized to provide load balancing and peaking services. Thus, a unit of generation displaced by a unit of wind is not equivalent to a proportional unit of the ERCOT generation mix (i.e., though gas and coal account for 43% and 37% of actual generation wind power primarily displaced gas generation (72%) with coal (28%) meeting the balance). Essentially, the authors conclude 1 MWh of wind power integrated into the ERCOT grid offsets 0.72 MWh of gas and 0.28 MWh of coal.

Similarly, marginal emission rates, rather than averages, must be considered from any displaced generation. Substantial emissions occur during the startup of offline fossil EGUs and dynamic operation effects (e.g., cycling, ramping) in response to stochastic variation in wind generation has been shown to impact the emissions per unit energy output [160]. Fossil EGUs have lower emission rates during steady operation at designed levels of output and when cycled up or down to accommodate wind power EGUs incur losses of efficiency, thereby increasing emission rates[160]. Startup emissions and increased cycling or ramping, both in magnitude and frequency, can potentially reduce emissions benefits from renewable energy. A unit commitment and economic dispatch model used to evaluate various penetrations of wind power reported significant emissions effects from increased cycling and start-ups of thermal power plants[161]. Additionally, failure to account for marginal emission rates can result in incorrect evaluation of reductions, i.e., though [151]determined units of marginal generation offset by wind, average emission rates for displaced EGUs were used to calculate emission offsets, potentially overestimating emission savings[151].

Further, a range of factors influence the composition and operation of regional power grids and emissions from displaced generation are regionally and temporally (e.g., seasonally, diurnally) dependent. Significant and systematic spatial and temporal variation exists with regards to the magnitude of avoided emissions attributable to the addition of renewable generation[162]. The potential of wind power to reduce emissions associated with conventional electricity generation was examined using systemic hourly wind data and plant emissions for the ERCOT, CAISO (California), and MISO (Upper Midwest) electrical grids[163]. The study revealed significant variation in the magnitude of emissions benefits between the study regions, reflecting differences in utilized technologies and fuels (i.e., gas generation dominates CAISO while coal is heavily used in MISO). In California, emissions reductions per MWh of integrated wind generation was equivalent to 0.042 lbs/MWh SO_2, 0.357 lbs/MWh NO_x, and 0.195 tons/MWh CO_2. These results confirm that renewable generation from different regions has dissimilar emission benefits and the magnitude of displaced emissions varies, both at and across the species level. Therefore, the GHG and AQ impacts of renewable energy will directly depend on regional temporal integration patterns.

To highlight the importance of the factors discussed above, the results from [151], [162], and [163] for total avoided emissions from the integration of 1 MWh of wind power into the ERCOT power grid are displayed in **Figure 2** and compared relative to estimates derived from average factors for ERCOT reported in the U.S. EPA eGRID database for the year 2007. The use of average grid factors underestimates emissions benefits relative to estimates of marginal generation provided by the detailed dispatch model used by Cullen. This is intuitive as eGRID emission factors are net averages and include power from low-emitting technologies (e.g., wind, nuclear) that generally don't provide marginal generation. However, inclusion of marginal emission rates in [162] results in significantly lower estimated benefits and is more representative of actual generator emissions. Further reductions in benefits were reported in [163] due to differences in assumptions, further emphasizing the dependence and importance of model construct. These results validate the complex systems-level interactions that must be considered when evaluating wind power GHG and AQ impacts and demonstrate the inaccuracy of assuming wind power is equivalent to a unit of emissions free electricity.



	CO2 tonsCO2e/MWh	NOx lb/MWh	SO2 lb/MWh
eGRID Year 2007	0.751	0.8642	3.0015
Cullen 2010	0.79	1.05	3.15
Novan 2011	0.569	0.828	1.545
Kaffine 2011	0.38	0.41	0.88

Figure 2: Estimated emission reductions from displacement of 1 MWh of average grid electricity with one MWh of wind power for ERCOT. Adapted from [151, 162-164].

2.2.1.2 Dispatchable Renewable Resources

Renewables not constrained by variability and characterized by some degree of dispatch capability, include hydroelectric, geothermal, and biopower. Large-scale hydroelectric requires significant land-use and appropriate settings and is not discussed in this work. In addition, energy storage can be dispatched when needed to assist in providing a link for variable renewable resources to meeting fluctuating electricity demand. This allows for management of renewables similar to current dispatchable fossil fuel plants, providing power when needed and allowing for ramping in response to demand fluctuations. Though many forms of energy storage are not technically renewable by the established definition, direct linkages make discussion of energy storage appropriate in the renewables section of this work. Biopower involves a complex array of different energy resources and pathways and consequently entails a wide range of potential impacts, including large benefits and dis-benefits. Due to the large and diverse body of research available and the significance of potential impacts the biopower section of this work is presented as a stand-alone section.

Geothermal Power

Conventional geothermal is a commercially proven technology that harnesses naturally occurring heat in the Earth's crust. Geothermal energy can be used for power generation, heat pumps, or other direct uses. Three major conventional geothermal technologies utilized to provide power include dry/direct steam plants, flash steam plants, and binary-cycle plants. Additionally, enhanced geothermal power (EGS), involving the use of advanced drilling and fluid injection methods to add water and permeability in locations where heat is available, could increase potential resource bases.

Geothermal power plants typically operate with capacity factors in excess of 90%, higher than most renewables, and can provide base load power free from intermittency concerns. High expansion potential exists for geothermal energy, for example the U.S. Geological Survey estimates that current technologies could utilize about 40,000 MW of resource[165]. However, deployment prospects vary regional and most available resources are located in the western U.S. Other barriers to furthered geothermal deployment include initial capital risk from exploration and investment uncertainties, remote geographic distribution necessitating transmission infrastructure, and permitting delays. Despite these and other barriers, a panel of experts concluded that EGS alone could reach an installed capacity of 100,000 MW by 2050, equivalent to about 33% of current installed U.S. coal capacity[166].

Replacing fossil generation with geothermal is an effective GHG mitigation strategy, as life cycle emissions for all technology types are significantly less than col and gas [166]. Life cycle estimates available in the literature range from 5-57 g CO_2e/kWh for various plant designs [108, 138, 167, 168]. Some hydrothermal reservoirs contain trace amounts of dissolved GHGs which are released to the atmosphere from direct and flash steam geothermal plants[169]. Though emissions of lithospheric CO_2 can be significant; emissions vary widely with respect to particular geothermal fields and average emissions are still much lower than any fossil energy source (**Table 4**). Binary-cycle plants utilize a closed loop cycle and lack air emissions. An estimate of life cycle CO_2 emissions from geothermal power, including all stages from plant construction to decommissioning, reported associated emissions of 5.6 g CO_2/kWh, however the value did not include emissions associated with normal operation which have been estimated to be about 30 g CO_2/kWh [138]. The comparative results of the study demonstrate that geothermal power results in slightly higher CO_2 emissions per unit electricity than wind and hydroelectric power, however geothermal also had the highest embodied energy. Further, replacing the generation from an average 500 MW coal plant with geothermal would result in reduced emissions totally 3 MMT CO_2eq annually. Fugitive CO_2 emissions are also a concern; however a study of emissions from hot dry-rock geothermal electricity calculated a CO_2 emissions factor of 37.8 g CO_2/kWh with fugitive emissions included[109].

Table 4: Emissions from various power plant technologies relative to geothermal power plants. Adapted from Tester et al. (2007)[166].

Plant Type	CO_2 [Kg/MWh]	SO_2 [Kg/MWh]	NO_x [Kg/MWh]	PM [Kg/MWh]
Coal	994	4.71	1.955	1.012
Oil	758	5.44	1.814	NA
N.G.	550	0.0998	1.343	0.0635
Geothermal (flash-steam)	27.2	0.1588	0	0
Geothermal (The Geysers)	40.3	0.000098	0.000458	negligible
Geothermal (binary-cycle)	0	0	0	negligible
EPA average (All U.S. plants)	631.6	2.734	1.343	NA

Expansion of geothermal power is not expected to be associated with any AQ concerns; and impacts could be particularly beneficial if fossil base load generation is displaced as direct emissions, including NO_x and SO_2, are extremely low compared to fossil power plants (**Table 4**). Previous AQ concerns caused by H_2S emissions associated with geothermal generation have been successfully mitigated by commercially available control technologies. Further, it is expected that geothermal technology will move away from hydrothermal towards larger EGS developments which have reduced environmental risks. Geothermal power derived from closed loop binary-cycle plants produce no air emissions of criteria pollutants and can be considered an emissions free source of electricity.

Energy Storage

The systems-level impacts of variable renewables, including emissions from complimentary generation, could be limited by the development and implementation of low-cost, effective electrical energy storage. Storage of electricity can be accomplished via various forms of energy, including chemical, kinetic, and/or potential, that can later be utilized to provide needed power. Potential technologies include hydrogen, compressed air, pumped hydro, flywheels, capacitors, thermal, superconducting magnetic and advanced battery technologies (i.e., flow batteries, Lithium-ion, sodium-sulfur). Energy storage technologies and their characteristics are considerably diverse; however most are currently constrained by economic and technological barriers and wide-spread commercialization will require these challenges be overcome[170].

Energy storage can provide three important power system needs: load shifting by supplying power to meet peak demand from periods of lower demand, load balancing to match supply and demand, and displacing the need for increased grid capacity[171]. Studies have demonstrated that generation from renewables at levels necessary for significant GHG mitigation will require the development and large-scale deployment of cost effective, functional, energy storage [41, 82, 122, 172, 173]. For example, results from dispatch simulations analyzing the carbon abatement potential of high penetrations of wind turbines, PV, and CSP in California suggest that deep GHG reductions (90-100%) will require energy storage with capacities minimum 65% of the peak load, and large enough to permit seasonal energy storage[174]. Similarly demonstrating the facilitated increase in renewable capacity, an analysis of the interrelationship between wind power and energy storage projected that co-deploying 30 GW of energy storage would allow an additional 50 GW of installed wind capacity by 2050, equating to a 17% increase relative to a no-energy storage case[175]. The studies highlight the importance of the ability of large-scale energy storage to decouple demand from real-time available renewable power. Without available energy storage conventional EGUs must be available for dispatch during periods when available renewable power isn't equivalent to demand.

Energy storage technologies could have significant GHG and AQ benefits by reducing emissions associated with load balancing and spinning reserve via displacement of the commonly fossil complimentary EGUs. Reported life cycle GHG values for energy storage in the literature include estimates for pumped hydro, CAES, and battery systems. Pumped hydro is extremely low at 5.6 g CO_2eq/kWh; and battery system values ranged from 32.6 to 40 g CO_2eq/kWh[176]. CAES had the highest carbon footprint at 292 g CO_2eq/kWh, largely due to the use of natural gas to reheat air in adiabatic CAES systems. With the possible exception of CAES, such options offer lower life cycle emissions than most complimentary grid technologies. In addition, CAES could represent a viable mitigation option if advanced systems without natural gas consumption are developed, e.g., a 56% reduction in CO_2 per kWh of generation was estimated when CAES was utilized in place of a gas-fired turbine to balance wind power[177].

In addition to direct reductions, energy storage has the potential to reduce emissions by time-shifting loads from temporal periods of peak demand, reducing the generation from dirtier, less efficient peaking EGUs and utilizing more efficient, cleaner base-load generation. Climate benefits from reductions in GHG emission could be substantial from energy storage peak shifting. The reduction of CO_2 per MWh shifting from peak to off-peak is estimated to be 26 to 33% for the three major California utilities[178]. A major caveat to GHG mitigation impacts of energy storage is the co-deployment with renewable or other low-carbon sources of power. Utilization of energy storage could potentially increase GHG emissions if power is supplied by lower cost, higher-carbon generation (e.g., coal) to displace more expensive but cleaner peaking power (e.g., natural gas).

Similarly to GHG mitigation, emission impacts could also offer improvements in regional AQ. A California assessment concluded energy storage could provide reductions of 55% CO_2, 85% NO_x, 77% SO_x, 84% PM_{10} and 96% CO per MWh of on-peak electricity delivered from an average gas peaking plant[179]. Any AQ impacts from energy storage will be dependent on the spatial and temporal emission offsets, but will likely be beneficial. In particular, if energy storage precludes the use of fossil generation in or near urban airsheds AQ and subsequent human health benefits could be significant.

Biopower

Generation pathways encompassing fuels derived from biomass sources (i.e., organic material produced by a biological process) are an attractive renewable option due in part to the flexibility of energy provision, which includes an extensive assortment of feedstocks with various opportunities and availabilities across a broad range of geographic areas[180]. Prospective resources include wood and woody wastes (e.g., residuals from timber harvesting, sawmilling, and pulp and paper production), trees, plants, grasses, aquatic plants and algae, agricultural residues (e.g., wheat straw, corn stover), industrial wastes, sewage sludge, animal wastes, organic waste materials and municipal solid wastes (MSW)[181]. A key distinction for feedstocks differentiates dedicated energy crops (DEC) grown intentionally for use as a biopower resource from those representing waste streams. A simplified diagram of potential biomass energy utilization pathways is provided in **Figure 3**. It should be noted the production of liquid fuels are generally considered for application in the transportation sector but could also be utilized in traditional combustion devices to produce power and heat. Good overviews of various biopower challenges can be found in [181, 182]

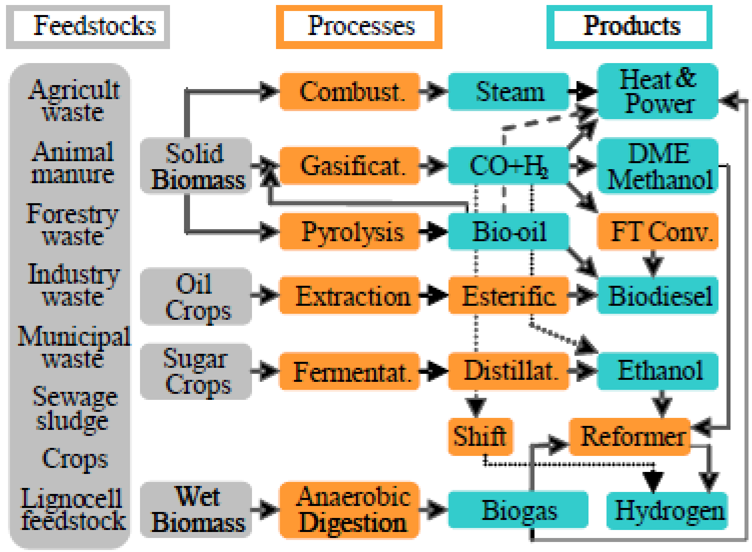


Figure 3: Potential biopower feedstocks and energy conversion pathways. Source: IEA 2007 http://www.iea.org/techno/essentials3.pdf

Fundamental generation strategies include direct combustion, co-firing, gasification, pyrolysis, and anaerobic digestion. Direct chemical conversion of gaseous fuels is also possible via fuel cells. Additionally, gaseous fuels can be upgraded and injected into existing natural gas pipelines to provide a source of flexible renewable natural gas which can be utilized in various applications; including as a low carbon transportation fuel[183, 184]. Most current biopower systems include direct combustion of resources to produce heat for the mechanical and/or thermal work required to produce electricity from generation devices, e.g., a common method for solid biomass involves combustion to provide heat to produce high-pressure steam which is expanded through a generation turbine [185]. Similarly, gaseous biopower fuels are often utilized directly as a fuel for reciprocating engines or turbines. Currently utilized conversion devices are commercially available and demonstrated at domestic- and utility-scale but generally entail lower efficiencies and higher pollutant emissions[186]. Feedstocks can also be co-fired with fossil fuels in traditional power plants, displacing some fraction of the original fossil generation, e.g., solid biomass co-firing with coal, biogas co-firing with natural gas. Benefits of co-firing include high overall electrical efficiency due to the economy of scale of the existing plant, low investment cost, and directly avoided emissions which can be significant if coal is displaced[187]. Gasification has been proposed as a method for effectively converting biomass into a useful fuel for power generation, CHP applications, H_2 production, and liquid fuel production[182]. Gasification differs from combustion in that solid fuels are partially oxidized in an O_2 starved (but not absent) environment at high temperatures to produce carbon char, and a flexible fuel gas composed of hydrogen, CO, CO_2, and CH_4[188]. Differing from gasification, pyrolysis is conducted around 500° C without any O_2 and can produce solid (char), liquid (tar), and gas products[188]. Gasification pathways Further, gasification pathways can improve generation by employing gas-Brayton cycles in higher efficiency turbine engines, i.e., applications of biomass integrated gasification (BIGCC) in gas-turbine plants, but are currently limited by cost and require further development and demonstration at commercial scale[189].

Further, biomass-derived gasses (biogas) with energy potential can be produced via intentional or unintentional anaerobic digestion or fermentation (i.e., the conversion of organic matter via bacteria in the absence of O_2) of biodegradable organic matter including manure, sewage sludge, and MSW[190]. Major sources of available organic waste streams suitable for digestion include wastewater treatment plants (WWTP), agricultural activities (e.g., animal and crop wastes), and industrial wastes (e.g., food processing) (Figure 4). MSW facilities including landfills generate collectible biogas from natural decomposition of organic material. Biogases range in composition but generally include 50-80% CH_4, with CO_2 largely providing the balance, and have significant energy value[191]. Additionally, small amounts of N_2, O_2, H_2S and a variety of organic and element-organic compounds are present which can lead to emissions of criteria and hazardous air pollutants depending on the conversion device [192]. Fuel gas produced from digestion can be utilized in a variety of methods to provide useful products, including power, heat, and chemical fuels for both power generation and transportation (Figure 4). Potential conversion devices for biogas energy pathways can include boilers, turbines, reciprocating engines, Stirling engines, and fuel cells[190]. The potential for domestic biogas energy is significant; in 2010 U.S. agricultural operations alone included 162 anaerobic digesters generating 453 million kWh of energy, enough to power 25,000 average homes[193].

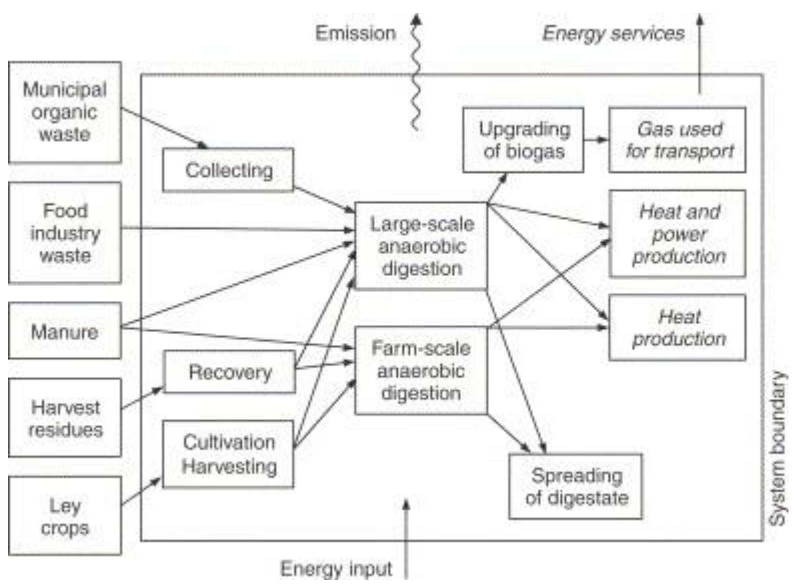


Figure 4: Overview of potential biogas systems from resource to energy services. Arrows demonstrate material, energy, and emission flows. From [194].

Utilization of fuel cells to directly convert chemical energy from resources into power, heat, and fuels represents an important opportunity to reduce emissions[195]. Suitable gaseous fuels include biogas and syngas produced from gasification of solid biomass. In addition to low emissions, benefits of using biogas in high temperature fuel cells include high electrical efficiencies, water neutrality, and the generation of multiple useful products (e.g., waste heat, industrial steam, chemical fuels) [196]. Fuel cells have been demonstrated with landfill gas[197, 198] and anaerobic digester gas from municipal WWTPs [199] and agricultural wastes [196]. Additional advanced conversion technologies suitable for biogas applications with improved environmental performance include Stirling engine, Organic Rankine Cycle, and micro gas turbines[200].

The diversity of biomass and biogas energy systems yields substantial differences in emissions impacts which complicate the drawing of broad conclusions regarding GHG and AQ co-benefits[194]. The net GHG impact of generating power from biomass or biogas is dependent on emissions from the specific deployed system and the emissions from the energy system that is displaced. Systems must be evaluated individually on a case-by-case basis accounting for the specific resources, technologies, and end-uses of generated products. Estimates of emissions available in the literature are generally conducted using LCA methodologies, e.g. see [194, 200-202], and reported in totals for each LCA category. Such studies are appropriate for discussing GHG impacts. **Figure 5** compares carbon cycles for solid biomass and fossil fuel energy systems. Key stages contributing emissions include feedstock growth, production and transport, utilized conversion technologies, and the desired product and end use. Combustion of solid biomass releases CO_2 that was removed from the atmosphere during feedstock growth and, if rates from uptake during growth offset those from combustion, the potential for carbon neutrality exist (or even negative net emissions if CCS is co-deployed)[203]. However, feedstock growth and combustion sites often differ spatially and temporally and, depending on determined system boundaries, can be challenging to account for in emission estimates[203]. Further, net environmental impacts of biopower depend on a wide range of diverse factors, including direct and indirect land-use changes if feedstocks are grown intentionally, i.e., dedicated energy crops (DEC). In particular, collection, distribution, and handling of feedstocks are an important source of emissions for both biomass and biogas systems[194]. Current biopower facilities are often large-scale and centralized due to cost and efficiency considerations and require collection and distribution of resources potentially across broad areas. The development and deployment of modular systems is being investigated for distributed applications as the use of conversion devices spatially located nearby feedstocks can negate the need for collection and transport to central locations, avoiding associated emissions and costs. However, distributed technologies offer incur higher emissions and reduced efficiencies at scale relative to larger systems[204]. The use of scalable fuel cells for distributed applications is particularly attractive due to the potential for increased efficiencies and reduced emissions relative to other distributed conversion devices [205]. Additionally, accurate emissions characterization must include any avoided emissions which can represent an important component of net impacts, e.g., utilization of forestry and agricultural residues is a promising resource pathway as it can negate traditional disposal methods which are associated with significant emissions, i.e., prescribed or open-field burning, natural decomposition, transport to landfills [206]. Similarly, collection and use of biogas for generation can offset emissions of methane associated with organic decomposition of feedstock. Further, if produced gas is collected flaring activities necessitated by regulatory standards designed to address methane release generate emissions that can be offset by energy pathways. However, identifying and establishing offsets quantitatively and qualitatively (i.e., spatial and temporal characteristics) is difficult and often neglected in biopower evaluations which can lead to underestimation of environmental benefits.

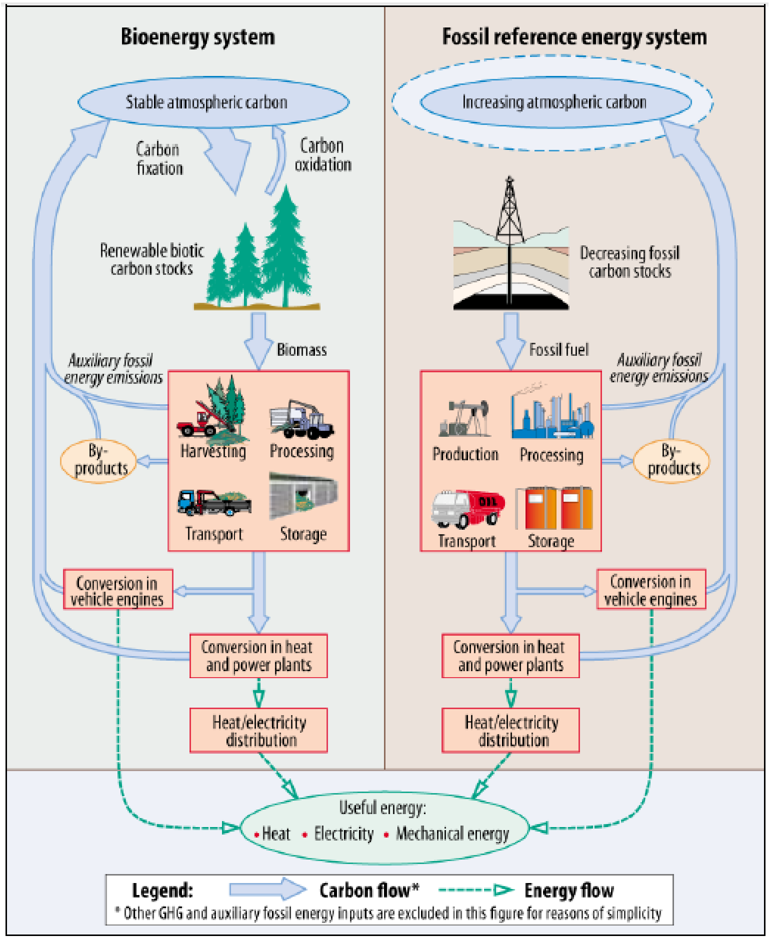


Figure 5: Diagram of CO_2 balances for biopower and fossil energy systems. Source: IEA Bioenergy Task 38, Greenhouse Gas Balances of Bioenergy and Bioenergy Systems, 2002. Adapted by [207].

Reported life cycle emissions from biopower generation systems vary dramatically across and within energy provision pathways and depend on numerous factors, including feedstock source and conversion technologies [208]. In addition to variability in conducted LCA methodologies, estimates are complicated by assumptions regarding system design and performance, technology development, resource quality, agricultural practices and land-use changes; and can lead to reported values that differ widely even for the same resource and pathway [5, 11, 12]. Life cycle GHG emissions reported in the literature for biopower systems utilizing solid biomass demonstrate significant range for different resources (**Table 5**). Estimates for woody crops range from 4 to 160 g CO_2eq/kWh depending on conversion pathway [109, 209-215]. Similarly, a survey of multiple studies on wood-based biomass reported life cycle emissions from 35 to 99 g CO_2e/kWh [103]. Improved life cycle emissions, including the potential for net negative emissions, have been reported for feedstocks generated by waste/residue streams, ranging from -633 to 320 g CO_2e/kWh [109, 209-223]. Biomass power generation can also include the co-deployment of CCS technology such as those currently being developed for coal plants which allows for substantially carbon-negative electricity by effectively removing carbon from the natural cycle [224, 225]. Literature estimates for CCS pathways range from -1368 to 594 g CO_2e/kWh [223, 226]. As a result, biopower co-deployment with CCS technology potentially offers the largest GHG mitigation benefits among all considered technologies; an important consideration for future allocation of limited biomass resources. The future technical performance and commercial viability of such systems is currently uncertain however, and it is unknown if pathways such as large-scale biomass gasification facilities with CCS will be available in the study horizon [227]. In order to better understand the wide variation in reported biopower life cycle GHG assessments NREL conducted a comprehensive review of studies published between 1980 and 2010[228]. The study evaluated estimates for solid biomass utilized in co-firing (with coal), direct combustion, gasification, and pyrolysis technologies. The majority of life cycle emissions were reported between 16 and 74 g CO_2eq/kWh with a high value of 360 g CO_2eq/kWh. Avoided GHG emissions from landfilling and CCS deployment allowed some pathways to achieve net negative emissions. An important limitation of the study was the exclusion of land use change impacts in estimates, which generally reduce benefits when accounted for.

Emissions from biogas energy pathways vary dramatically between systems and depend upon properties of digested raw material, efficiency and characteristics of gas production, deployed end-use technology, efficiency of generation (electricity and thermal), and other parameters including the utilization of by-product heat[200]. For example, the fuel-cycle emissions from various biogas systems can vary by a factor of 3-4, and even by up to 11 times, for different biogas systems providing equivalent energy[194]. A major factor in life cycle GHG emissions from biogas systems include any losses of CH_4 due to leakage or venting which, if present, can add substantial warming potential[194]. The general environmental impact from displacement of reference energy systems via novel biogas system introduction has been deemed beneficial on a life cycle basis, with indirect environmental benefits (e.g., reduced emissions of ammonia and CH_4) sometimes in excess of direct benefits (e.g., reduced emissions of CO_2 and pollutants)[202]. Deployment of biogas energy conversion pathways (e.g., the treatment of animal manure using anaerobic digestion, collection and utilization of LFG) can significantly reduce GHGs by offsetting both the release of potent GHGs and the need for grid electricity, displacing emissions from traditional power generation. For example, traditional manure disposal methods (e.g., lagoons, outdoor storage) generate and emit CH_4 and N_2O during the decomposition process. Similarly, LFG is composed of 50-80% methane and the utilization of LFG via the installation of conversion technologies, largely as a result of regulations addressing recovery for flaring or energy conversion, has resulted in significant GHG emission reductions in the U.S. [229]. Further, the total potential for domestic biogas production and utilization is much higher than the current levels. Complete conversion of available domestic livestock manure into biogas alone could contribute about 88 billion kWh using standard microturbines; equivalent to roughly 2.4% of annual U.S. electricity consumption[230]. Assuming coal is replaced and manure GHG emissions avoided, such a strategy would achieve a net GHG emission decrease of 99 MMT; equivalent to about 3.9% of annual U.S. power sector emissions, however this study should be treated as an upper bound. Demonstrating achievable benefits if biogas pathways are utilized in coming decades, an assessment concluded that by 2050 manure digestion could mitigate 151 MMT of CO_2e from methane abatement and an additional 31 MMT from displaced electricity emissions[231].

Despite the dissimilarity present in literature estimates of GHG emissions the majority report reductions from current fossil power generation, including coal and natural gas. A series of life cycle assessments conducted for biopower systems using both biomass residues, DEC and several traditional fossil pathways (e.g., PC coal, NGCC)reported significant reductions in global warming potential (**Figure 6**)[232]. Reported emissions for IGCC plants utilizing DEC offered significant reductions from both fossil alternatives and the use of biomass residue resulted in net negative GHG emissions when direct-fired. Extremely low GHG emissions for woody biomass relative to fossil generation have been reported using a dynamic LCA approach, however biomass pathway emissions were in high relative to other renewable sources [109]. Co-firing with coal reduces CO_2 emissions relative to single firing with coal and could be an effective near term GHG mitigation strategy[233]. A life cycle assessment of biomass co-firing at rates of 5% and 15% by heat input in a coal-fired power plant reported reductions in GHG emissions of 5.4 and 18.2% respectively[234]. In general, these results demonstrate that life cycle GHG emissions for biopower systems are comparable to other renewables and reduced from common fossil generation systems.

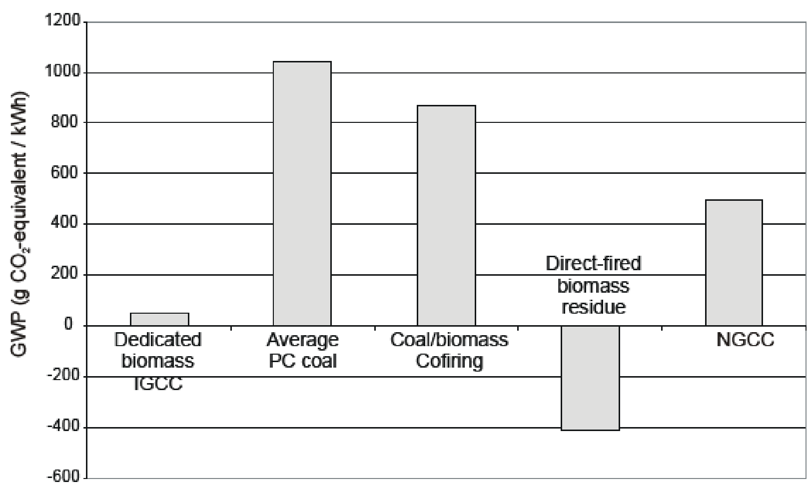


Figure 6: Life cycle GHG emissions for power generation from biomass and fossil pathways. From Bain et al. (2003) [232].

Life cycle impacts for biopower using DEC are difficult to interpret as required alterations to land and water resource uses affect a variety of complex sequences (e.g., food resources, hydrologic cycles, biodiversity, etc.) which greatly complicate assessments. DEC can also have emissions from upstream processes including agricultural processes (e.g., emissions from fertilizer use, farming vehicles) and the harvesting and transport of feedstock to conversion locations. The complexity is highlighted by widely varying GHG estimates for the same energy crop species [235]. The use of some DEC, including food crops, can limit GHG benefits and larger reductions are attained from waste products or low-input, high diversity perennial plants grown on degraded or marginal lands[236, 237]. In general, pathways that include appropriately selected DEC grown on specific land categories have the potential to sequester carbon [238, 239]. Assessment of the life cycle GHGs of several potential energy crops (i.e., switchgrass, giant reed, and hybrid poplar) for generation in an IGCC system reported reductions relative to a coal gasification system[240]. Interestingly, GHG benefits were greater than those from conversion to biodiesel and ethanol, which would support resource utilization in the power sector. Similarly, reported life cycle GHG emissions for short rotation coppice (willow/poplar) and miscanthus showed benefits for coppice per unit produced energy [241]. However, miscanthus was favorable per unit of land highlighting the tradeoffs and challenges that must be considered when considering the deployment of DEC.

In contrast, the utilization of waste/residue streams can achieve substantial reductions due to the avoidance of decomposition and/or treatment of feedstocks, including gasses with higher GWP than CO_2 (e.g., methane, N_2O). Further, analyses have shown that biomass residue is preferable to DEC use for biopower application in terms of both net energy ratio and GHG emissions across various feedstock and conversion technology pathways [232, 242]. In light of these and other prominent environmental concerns associated with DEC (e.g., land-use, water resources, soil, aquatic toxicity, competition with food) , it has been proposed that only certain feedstocks be considered for renewable biopower including perennial plants grown on degraded lands abandoned from agricultural use, crop residues, sustainably harvested wood and forest residues, double crops/mixed cropping systems, and municipal and industrial wastes[243].

Table 5: Life cycle GHG estimates reported in the literature for various biopower pathways.

Biopower Pathway	LCA GHG Emissions [g CO_2eq/kWh]	Reduction from Average Gas	References
Total	-1368-360	26->100%	[109, 133, 209-223, 244-246]
Woody Crops	4-360	26-99%	[109, 209-215]
Waste Stream	-633-320	34->100%	[109, 209-223]
Direct Combustion	22-120	75-95%	[211, 215, 217, 221, 245, 246]
Engine	14-110	77-97%	[133, 244]
Deployment with CCS	-594-(-1368)	>100%	[223, 226]

Biopower differs from many renewable resources in that some pathways (e.g., combustion, gasification) can include direct pollutant emissions (e.g., PM, CO, VOCs, NO_x, SO_x, acid gasses, heavy metals) comparable or in excess of conventional fossil generation with potentially adverse impacts on regional and local AQ. Pollutant emissions generated per unit biomass energy are directly related to the specific energy provision pathway and vary with respect to consumed feedstock, utilized conversion technology, and the use of co-deployed pollutant controls. Wide variation in biomass resource types (e.g., solid biomass, biogas), characteristics (e.g., energy, ash, moisture content), utilized conversion technologies, and end-uses yield substantial deviation in generated emissions. For assessment of regional AQ impacts it is necessary to understand and account for emissions both directly from source contributors (e.g., exhaust of conversion devices, machinery used to collect/transport feedstock) and from those avoided (e.g., decomposition, flaring) in entirety.

The combustion of solid woody biomass is associated with the generation of PM, including smaller fractions correlated with significant human health impacts, i.e., particulate matter less than 2.5 microns ($PM_{2.5}$), and particulate matter less than 1 micron (PM_1)[247]. Relative to emissions from a fossil system, e.g., heavy fuel oil, wood-fired boilers generated substantially more PM_1 and total suspended particles (TSP), although the use of control technologies including electrostatic precipitators yielded similar or reduced PM_1 (Figure 7)[248]. Additionally heavy fuel oil combustion generated more SO_2 and NO_x emissions than wood-fired boilers with potentially greater secondary PM atmospheric burdens. Thus, biopower systems may have multi-faceted impacts on emissions and AQ when compared to reference fossil systems.

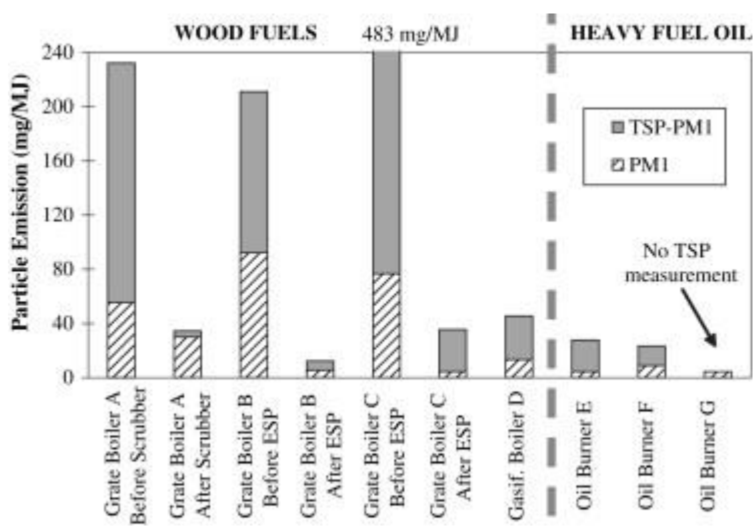


Figure 7: Average emissions of PM_1 and total suspended particles (TSP) from heating units utilizing wood fuels and heavy fuel oils. From [248].

Relative to coal, direct pollutant emission impacts from solid biomass combustion are generally favorable. Emissions of CO are of concern concentrations can be higher than coal combustion in units designed for coal and unmodified for biomass co-combustion[249]. In addition, biopower facilities using boilers emit significant amounts of PM_{10} and CO; but both can be reduced if different combustion or gasification systems are used[232]. Emissions of SO_x are generally reduced for biomass relative to coal as most resources contain less sulfur and some SO_2 produced is captured by alkaline ash [250]. NO_x emissions may increase, decrease, or remain the same relative to coal, depending on fuel, firing conditions, and operating conditions[251]. NO_x levels can be lower as the fuel nitrogen content in some, but not all, biomass sources is typically lower than coal. Biopower pathways with fluidized bed combustion or gasification emit NO_x similar to NGCC plants; however, if selective catalytic reduction (SCR) is co-deployed NGCC emit less [232]. Co-firing biomass with coal reduces the total emissions of NO_x, CO, NMHC, PM, and SO_2 relative to single firing coal in currently existing pulverized-coal fired power plants, potentially offering a strategy for attenuating AQ impacts of coal power plants near-term [234, 252]. However, in many regions of the U.S. comparing emissions from biomass systems to coal is of limited value, e.g., California uses very little coal for generation and a more apt comparison would be to natural gas technologies. Further, the construction of a biopower facility results in the introduction of a novel emissions source into a region which could have detrimental impacts on regional and local PM and ozone concentrations, as well as levels of toxic air emissions including heavy metals.

In addition to direct facility emissions, significant upstream emissions occur as a result of activities required to produce, process, and transport feedstock to facility locations.

For example, emissions from transportation activity (e.g., PM associated with diesel combustion) associated with gathering and transporting feedstock could impact local communities along major transportation corridors. Life cycle modeling of various biopower systems operating on short rotation coppice and miscanthus demonstrated that up to 44% and 70% of total NOx and particulate emissions occurred from upstream processes [253]. Impacts vary across pollutant species however, as the study also concluded that CO and hydrocarbon emissions were predominantly associated with plant activities. Further, impacts of upstream emissions can differ for waste or residue feedstocks relative to DEC as typically such pathways have reduced emissions upstream of the conversion facility.

When emissions are considered over a life cycle both species-level increases and decreases are observed relative to fossil power. Bain et al. (2003) reported life cycle pollutant emissions relative to an average PC coal and NGCC plants for different biopower pathways, including direct residue, dedicated biomass IGCC, and co-firing with coal[254]. As can be seen in **Figure 8**, emissions of PM, NO_x, and SO_x are highest for coal pathways while CO and NMHC was highest for the biomass residue and IGCC pathways. However, in contrast to GHGs, the regional nature of AQ dictates spatial and temporal emission patterns as the determinants of impacts and biopower systems require evaluation on a site-specific basis, rather than solely a life cycle approach.

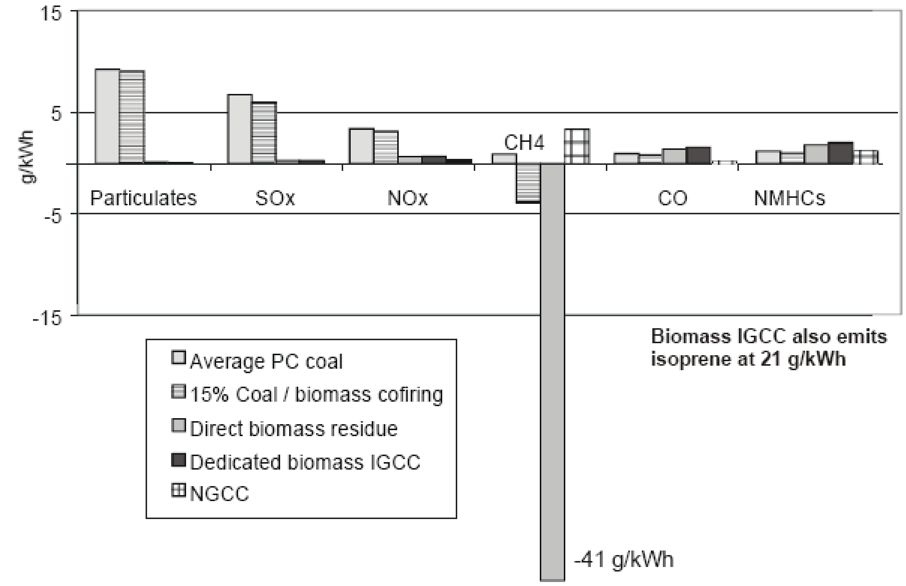


Figure 8: Life cycle pollutant emissions for fossil and biopower generation pathways. From [254].

Emissions from biogas systems vary considerably and are affected by the digested feedstock characteristics, efficiency of gas production, and the status of the end-use technology. Fuel-cycle emissions can vary between two biogas systems that provide equivalent energy services by factors of 3 to 11[194]. Thus, in order to accurately assess emission impacts the specific utilized energy pathways (i.e., gas production system, end-use technology) must be established. The current uses of biogas often involve heat engines (reciprocating, gas turbines) with enhanced economic viability but significant pollutant emissions. Table 6 shows pollutant emissions from stationary turbines and engines fired with natural gas and digester gas from agricultural sources in California. Emissions from the more commonly utilized biogas-fired reciprocating internal combustion engines vary widely and depend upon digester configuration, generation capacity, and others[255]. Additionally, significantly varying H_2S content in biogas (i.e., 4 ppm to 1,586 ppm) directly impacted variation in SO_x emissions[255]. Further, while average values for ICE pollutant emissions are less than natural gas estimates at in the higher range are significantly more for NO_x, CO and SO_x which could potentially worsen regional concentrations of ozone and PM. In some regions, this has precluded resource utilization due to stringent emissions regulations necessitated by regional AQ concerns, e.g., the San Joaquin Valley (SJV) of central California. Such challenges often prompt the use of flaring in place of conversion (e.g., to meet NO_x limits in the SJV selective catalytic reduction must be utilized) to prevent the uncontrolled release of methane. Flaring removes VOC emissions but produces emissions of other pollutants including NO_x and PM, though to lesser degree than combustion, with no added energy benefits. Implementation of advanced conversion technologies (e.g., fuel cells, microturbines) could provide biogas pathways that achieve maximum GHG and AQ co-benefits [256].

Table 6: Pollutant emission factors for stationary turbines and engines fired with natural gas and digester gas from California agricultural activity. Adapted from [255].

System	**Emissions (lb/MMscf) **All values represent uncontrolled emissions****				
	NO_x	CO	VOC	PM	SO_x
Stationary Turbine – Biogas	96	10.2	3.48	NA	3.9
Stationary Turbine – Natural Gas	336	86.1	2.21	6.9	3.6
Engine – Biogas	324 (10 to 918)	546 (222 to 948)	NA	NA	870 (6 to 3,180)
Engine – Natural Gas	588	892.5	NA	NA	0.6

As with biomass, biogas energy pathways have the potential for avoided pollutant emissions, including from flaring excess gas and the offset of grid electricity, which must be considered to accurately assess AQ impacts. Additional reductions can occur from the offset of conventional waste disposal practices, e.g., currently livestock manure is collected and stored outdoors to decompose during which releases of ammonia, H_2S, methane, and PM occur[257]. Biological decomposition of any biomass feedstock can produce methane and non-methane VOC emissions with implications for ozone and PM formation. In biogas energy pathways such emissions are eliminated via collection and conversion of gas in energy recovery devices; however generally with increases in other pollutants (e.g., NO_x, CO) if combustion of the biogas is utilized. Further, the use of digesters can reduce the quantity of waste requiring disposal; reducing emissions associated with collection and transport (e.g., diesel transportation technologies) in addition to direct decomposition emissions. Thus, the true effects on AQ of deploying biogas conversion methods include both direct emissions from utilized conversion devices and avoided emissions from traditional feedstock infrastructure.

In order to assess biopower accurately three environmental impact categories must be considered: *direct impacts* arising from the energy system used, *indirect impacts* arising from upstream activities that provide ancillaries or fuel used by the system, and *avoided impacts* from displacement of conventional biomass/biogas management strategies[258]. For biopower to sustainably contribute to GHG and AQ goals, selected energy pathways must be economically viable, environmentally beneficial, and socially acceptable[259]. The diversity and range of pathways yields the potential for both resources with carbon benefits and pathways that could increase net emissions. In addition, some resources (e.g., DEC) could be both beneficial and/or harmful depending on different environmental endpoints and the complexity of evaluating life cycle impacts may prevent a clear understanding. If widely deployed, biopower could have high mitigation potential, reflecting prospective future capacity additions and availability of low carbon pathways. For example, *Tackling Climate Change* reported that biomass electricity generation could result in emission reductions of 75 MMT CO_2eq per year in 2030[120]. Similarly, combusting biomass or biogas results in pollutant emissions which can have localized AQ impacts, while other pathways offer reduced or near-zero emissions that could improve AQ if generated power is used to offset grid-provided electricity. Utilization of resource waste streams and advanced conversion devices (e.g., fuel cells) can provide generation with potentially net negative GHG and no pollutant emissions, representing a mitigation strategy with high co-benefits. Conversely, the use of DEC and some traditional conversion devices (e.g., reciprocating engines, direct boilers) could potentially increase GHGs relative to natural gas and worsen localized AQ. In general, technologies involving some degree of combustion (e.g., boilers, heat engines) can worsen local AQ depending on spatial and temporal emission patterns and displaced fossil generation. Impacts could be important in areas currently experiencing poor AQ; a concern in many U.S. regions including those studied in this work. Thus, careful consideration of resources and conversion pathways will be necessary to avoid GHG and/or AQ dis-benefits.

2.2.1.3 Renewable Resources Conclusion

Renewable resources are consistently proposed to meet power demands with minimal environmental cost. This supposition is somewhat accurate for AQ and GHG co-benefits as the displacement of coal or gas with a combination of renewable technologies will almost certainly reduce life cycle GHG and direct pollutant emissions. However, the intermittencies of some (i.e., wind, solar) requires operation of additional technologies (largely comprised by emitting gas-fired EGU) to ensure adequate grid dynamics, thus blunting GHG benefits and potentially introducing localized worsening of ozone and PM. Thus, maximizing renewable co-benefits, including facilitating capacity levels and avoiding novel emissions, will require the co-deployment of both low- or zero emitting complementary technologies including commercial scale AES (i.e., pumped hydro, batteries) and upgrades to the current power infrastructure seeking a SMART grid.

As can be seen in **Table 7**, technologies comprising the majority of renewable pathways on a life cycle basis produce very low GHG emitting power. In addition, the majority of technologies comprising renewable pathways have no or very low point-of-use pollutant emissions and can potentially offer regional AQ benefits. A prominent exception includes some biomass/biogas pathways which can result in emissions of GHGs and pollutants that could potentially increase localized atmospheric concentrations and will be discussed in detail in a subsequent portion of this work. Further, co-deployment of energy storage can reduce the negative impacts of complimentary generation and dynamic EGU operation in response to systems-level integration of variable renewables. GHG and pollutant emission rates for conventional fossil EGU vary extensively by technology, fuel, and age and the magnitude and timing of emissions impacts from renewable integration depend upon many diverse factors including displaced EGU, any dispatched compensating EGU, and the transient emissions rates of dispatched compensating EGU. Thus, replacing one unit of conventional generation with one unit of zero emissions renewables will not necessarily provide the same generation, maintain reliability or decrease emissions as much as would be expected from nameplate power ratings. In addition, the need for available spinning reserve in response to unexpected or rapid declines in renewable output yields emissions from un-utilized generation. Further, the spatial and temporal aspects of emission perturbations are important for any resulting AQ impacts.

Assessments of emission reductions have demonstrated that portfolios of diverse renewable sources generally perform better than high penetrations of individual technologies [260]. Benefits of installing a range of technologies include increased system flexibility, minimization of intermittency issues via resource aggregation, and reduced curtailment, among others. Results from a set of California dispatch simulations for high penetration levels concluded combining wind and solar power improved GHG abatement performance relative to individual build-out scenarios [174]. For example, a portfolio of 30% wind and 70% solar achieved a maximum carbon abatement potential of 79%, while wind and solar alone achieved 58% and 56%. A study examining the operational effects of high penetrations of wind in a subset of the western electrical grid concluded that meeting 30% and 5% of the energy needs of the system with wind and solar technologies respectively was feasible and could potentially reduce 45% of CO_2, 50% of SO_x and 30% of NO_x emissions annually in 2017[146].

Table 7: Life cycle GHG emissions and corresponding reductions from advanced NGCC plant (i.e., 358 gCO_2e /kWh) reported in the literature.

Renewable Technology	LCA Emissions (g CO_2e/kWh)	Potential Reduction (Average gas)	References
Wind (offshore)	3-22	94-99%	[38, 109, 131, 134, 139-142]
Wind (onshore)	3-40	89-99%	[38, 108, 131-139]
Solar-P.V. Thin Film (CdTe, a-Si, CIS)	19-95	73-95%	[100, 105-107, 261]
Solar-P.V. Crystalline (m-Si)	20-104	71-94%	[99, 101, 106, 108-111]
Solar- CSP	12-241	33-97%	[112-117]
Geothermal	5-57	84-99%	[108, 138, 167, 168]
Ocean-tidal and wave	2-56	84-99%	[138, 262, 263]
Hydropower	1-39	89-99%	[133, 134, 176, 264-266]
Biopower	-633-360	0-277%	[109, 133, 209-223, 244-246]
Biopower with CCS	-1368 to -594	266-482%	[223, 226]
Energy Storage	6-292	21-98%	[176]

2.3 INDUSTRIAL SECTOR

The domestic industrial sector includes facilities, processes, or equipment utilized to create, process, and/or assemble commodities and wares. About a third of 2012 U.S. energy was delivered to the industrial sector for activities including manufacturing, construction, mining, and agriculture; with manufacturing dominate in terms of energy conversion (i.e., ≈80%) [85, 267]. Within manufacturing, energy usage is highest for bulk chemical production and additional energy-intensive industries include the production of iron, steel, aluminum, fertilizers, glass, food products, cement and lime, paper and petroleum refining [85]. Indeed bulk chemicals, refining, paper, steel, and food industries are responsible for about 60% of total sector consumption[267]. Less-energy-intensive industries include the manufacture of transportation equipment, computers, and additional durable metal products

Highlighting the current and intensifying importance to energy-related impacts, the 2014 U.S. Energy Information Administration's Annual Energy Outlook projects industry will become the largest energy consuming sector by 2018 and remain so through 2040, a total sector growth of nearly 28%[85].

As with all current U.S. sectors, industry currently relies on fossil fuels for delivered energy including coal, natural gas, and petroleum. Electricity represents roughly a third of sector consumption, with petroleum and natural gas accounting for 27% and 26%, respectively[267]. Use of coal directly meets only about 6%, although indirectly coal generates a portion of necessary power in many U.S. regions.

Industrial emissions arise from (1) direct and indirect consumption of fossil fuels for heat, power, and steam, (2) non-energy conversion of fossil fuels, e.g., chemical processing, metal smelting, and (3) non-fossil fuel related processes, e.g., cement manufacture.

Reflecting consumed energy and current fossil fuel dependence, the industrial sector is a key source of domestic GHG emissions, contributing 20% of total U.S. emissions in 2012[54]. However, when indirect and direct emissions from consumed electricity are included the industry share rises to 28%, representing the second largest sector behind only transportation[54]. The bulk of industrial GHGs originates directly from the combustion of fossil fuels to generate steam and needed process heat. Moreover, emissions occur as by-products of industrial processes, i.e., not directly from energy conversion to facilitate the process. Such emissions constitute 5.1% of the 2012 U.S. total, with the substitution of ozone depleting substances such as hydrofluorocarbons representing the dominant source[54]. Additionally, iron and steel, metallurgical coke, cement and nitric acid production contributed notable fractions of non-energy emissions.

Befitting the enormous diversity of industrial emission sources, extensive options exist that could mitigate both GHG and pollutant emissions. Many strategies are industry specific, e.g., improved flaring practices at refineries, switching from wet to dry kilns in cement manufacture; however general categories can be identified including improved energy efficiency, fuel switching, and energy recovery including heat, power, and fuel.

Energy Efficiency – It has been reported that, while challenging to quantify, industrial processes often operate with low energy efficiencies and thus significant opportunity exists to reduce emissions via reduced energy conversion[268]. Key factors in industrial efficiency include choice and optimization of technology, operating procedures and maintenance, and capacity utilization[268].

Fuel Switching – Industrial fuels used to provide steam or heat are generally determined by cost, availability, and regulatory drivers[268]. Additionally, some industrial fuels are determined by individual process needs and may or may not be feasible for substitute requiring individual assessment.

Energy Recovery – Techniques or methods to recover or exchange energy from one component of system (e.g., heat from exhaust streams, fuels from waste streams) for use in another can improve system efficiencies consequently reducing energy conversion and emissions. Three common methods for industrial applications include heat, power, and fuel recapture and use.

Potential strategies to reduce industrial energy conversion and emissions include energy efficiency and savings strategies, e.g., heat recovery systems in process heating[269]. It has been proposed that the most feasible near-term mitigation strategy for the industrial sector encompasses energy efficiency.

Due to its complexity industrial energy conversion and associated environmental impacts represents one of the most challenging end-use sectors to assess and project[270]. For example, estimating the potential for energy efficiency gains in the global industrial sector to 2030 was prevented by substantial knowledge gaps and uncertainties[271]. Despite relative uncertainty, it is clear that large-scale implementation of mitigation strategies in the industrial sector can achieve major GHG reductions. An assessment of the Brazilian industrial sector concluded energy efficiency measures, materials recycling and cogeneration, transitions from fossil fuels, and eliminating the use of biomass from deforestation could achieve eliminate 1.5 billion tonnes CO_2 from 2010 to 2030[272].

Improving energy efficiency of various industries can have important GHG reductions, particularly for carbon intensive industries. For example, deploying a suite of energy efficiency technologies and measures could reduce US iron and steel industry CO_2 emissions by 19% from 1994 levels[273].

2.3.1 Major Industrial Sub-sectors

Resulting AQ impacts from industrial sector emissions can differ from others, e.g., power generation, due to spatial, temporal, and composition variation. For example, ozone formation dynamics differ downwind of power plants and industrial sources due to discrepancies in NO_x and VOC emission rates and ratios [79, 274, 275].

2.3.1.1 Cement

The production of cement includes large heating demands which result in its manufacturing being both energy- and emissions-intensive. Cement manufacture is highly energy intensive and involves the grinding and mixing of raw materials as well as chemical alteration via intense heating in a high-temperature kiln. Fuels used in in cement plant kilns are often lower quality and high emitting due to economic drivers. In general coal is the most commonly utilized fuel due to low cost and the contribution of coal ash to product. For example, the most common fuel mixtures in California are coal and petroleum coke which is enhanced with scrap tires, dried sludge, or biomass fuels depending on the individual facility[276]. In California the cement industry is the largest consumer of coal state-wide. California produces the most cement of any U.S. state accounting for 12% of U.S. cement production in 2005 and was responsible for 8.7 $MMTCO_2e$ in 2008[276].

Emissions of CO_2 from cement production are primarily generated through fuel combustion and the calcination of limestone. The combustion of fuel to meet kiln needs is estimated to produce around 40% of direct GHG emissions with the remaining 60% is arises from calcination[276]. Calcination involves the high temperature conversion of raw ingredients to produce clinker via the chemical transformation of $CaCO_3$ into CaO and CO_2. Table 8 displays the emission intensity (lb CO_2/MMBtu) for fuels commonly utilized in cement kilns.

Table 8: CO_2 emissions intensity (lbCO_2/MMBtu) for fuels combusted at cement kilns. From [277].

	Common Cement Kiln Fuels					
	Natural Gas	**Heavy Fuel Oil**	**Western Sub-bit. Coal**	**Tires**	**Eastern Bit. Coal**	**Petroleum Coke**
CO_2 Emissions (lb CO2/MMBtu)	105.02	169.32	186.83	187.44	199.52	212.56

Improving the energy efficiency of cement production has been commonly proposed as an emissions reduction strategy [277-280]. Opportunities to improve efficiency range considerably and include shifts to more energy efficient processes, e.g., from wet/long dry to preheater/precalciner kiln systems. A complete description of all potential efficiency measures is outside the scope of this document but thorough reviews can be found in [277, 278, 280].

In addition to methods that can reduce the generation of CO_2, the removal of CO_2 from flue gas via CCS can significantly reduce emissions. The cultivation of algae to produce biofuels and other useful products using recycled concentrated CO_2 streams from cement plants also represents a potential GHG mitigation strategy. Closed algal cultivation systems will require further development prior to commercialization but the concept has been demonstrated successfully in Canada[278].

Additionally, novel techniques for meeting building material demands may provide significant GHG and AQ benefits. One such process has been developed and demonstrated by Calera involving the removal of CO_2 from the stack of a 10 MW coal-fired power plant which is then converted to solid calcium carbonate cement. The Calera process provides significant GHG benefits by preventing atmospheric release of CO_2 in tandem with offset cement production needs at traditional, high-emitting plants[277].

Pollutant emissions from cement plants include NO_x, SO_x, CO, PM, and VOCs as well as various toxic air contaminants. In particular, high concentrations of NO_x are generated in kiln gases via high combustion temperatures and, to a lesser degree, fuel nitrogen content (in contrast to CO_2 the majority of NO_x emissions occur from combustion in kilns)[281]. Total amounts of NO_x generated per ton clinker vary by kiln system characteristics and energy efficiency[281]. Table 9 demonstrates uncontrolled NO_x emissions in pounds per ton clinker for various kiln types from data obtained for eight of the top U.S. states in terms of clinker capacity and AP-42 emission factors. As can be seen there significant variance and overlap between kiln types although average values generally correspond to the lowest emissions from preheater kilns and highest from wet kilns. Variation in kiln emissions can be attributed to differences in kiln technology and the properties of the raw material being processed[282].

Table 9: Summary of updated uncontrolled NOx emissions data. Adapted from [281]

	AP-42	State Data	
Cement Kiln Type	[lb/ton clinker]	Average [lb/ton clinker]	[Range lb/ton clinker]
Wet kiln	7.4	6.2	1.9 – 13.4
Long dry kiln	6.0	4.5	2.5 – 7.1
Preheater kiln	4.8	1.7	0.4 – 3.7
Precalciner kiln	4.2	2.9	1.1 – 5.6

Prospective emission control strategies include selective catalytic reductions (SCR) which can reduce NOx from cement kilns by 80-95%[282]. SCR is attractive for cement plant deployment as gas temperatures between kiln and stack are within necessary bounds, although concerns include high dust levels which must be managed to avoid catalyst plugging or wearing. Additional NOx control strategies and potential reductions are displayed in Table 10. The injection of biosolids (i.e., treated sludge from a waste water treatment plant) has been shown to reduce NO_x by 50% from a facility in California[281]. Additionally, the CemStar process, involving the introduction of steel furnace slag as feed material into kiln, has proven effective at reducing thermal NO_x via kiln firing temperature reduction. Additionally, CemStar can improve kiln production in tandem with emission reductions.

Table 10: NO_x control strategies and potential reductions for cement kilns. Adapted from [282].

Reduction Strategy	Potential NO_x Reduction
LNG with indirect firing	14 to 47%
Low NOx precalciners	30 to 40%
Mid-kiln firing	33 to 40%
CemStar	20 to 60%
SNCR	10 to 90%
SCR	80-95%
Biosolids injection	50%

2.3.1.2 Chemicals

The bulk chemical industry includes producers of various goods, including among other things the manufacture of organic and inorganic chemicals, resins, ceramics, petrochemicals, agrochemicals, polymers, explosives, fragrances, flavors and synthetic rubber and fibers.

Turnaround operation of chemical plants (i.e., start-ups, shutdowns) result in large quantities of compounds which must be flared, producing large emissions of CO2, CO, NOx, VOCs, and air toxics [283, 284]. Controlling both continuous and episodic NO_x and VOC emissions from industrial flaring has been shown to significantly reduce average peak ozone concentrations in an urban air shed [285].

2.3.1.3 Petroleum Refining

The refining process generates emissions of a diverse range of pollutants including CO, NO_x, PM, SO_2, VOCs, and numerous air toxic compounds, e.g., benzene, toluene[77].

Emissions of NO_x occur from refinery processes including combustion of fossil fuels for necessary power and heat generation and from flaring; generally represented by a few large sources [79]. In contrast, refinery VOC emission sources are more numerous and diffuse and include an extensive assortment of compounds from stacks, process vents, flares, cooling towers, and leaks from storage tanks, pipes, and valves[286].

Refinery locations are associated with plumes containing both high levels of NO_x and reactive VOCs; conditions typically associated with rapid and efficient (i.e., per NO_x molecule) ozone formation[79, 275]. Indeed, net ozone formation rates and yields per molecule NO_x have been estimated in excess of those for power plants[79].

Episodic emissions from petroleum refineries and other chemical processing facilities, including highly reactive VOCs, can contribute profoundly to regional ozone levels[287]. For example, plumes of highly concentrated ozone (i.e., peaks of 180-200ppb) have been observed downwind of large emission events[288]. Emissions from a single refining unit, i.e., the fluidized-bed catalytic cracking unit, were shown to contribute to elevated $PM_{2.5}$ levels during a concurrent regional haze episode in Houston, TX[289].

Further heightening concern, reported emissions from petrochemical facilities may currently be underreported and thus AQ impacts underestimated [78-80].

In addition to routine (expected) emissions, excess event (upset) emissions generated by unforeseen/uncontrolled activities or failures (e.g., tank ruptures, compressor failures, startups, shutdowns, maintenance) have been shown to be important in terms of quantity and occurrence and may be equivalent to .

2.3.1.4 Primary and Secondary Metals

Metal industries include smelting and refining of both ferrous and non-ferrous metals, rolling, drawing, and alloying metals, manufacturing castings, nails, spikes, wire and cable, production of coke. Sources of metal industry emissions include foundries, blast furnaces, and rolling and finishing mills.

Secondary metal processing industry includes secondary magnesium, aluminum, lead, copper, and zinc processing and iron and steel foundries. Emissions from secondary metal processing occur from material handling and storage, scrap pretreatment, and metal melting, refining, forming and finishing. Control techniques for emissions include scrubbers for PM and acid gasses, incinerators for organic compounds, and cyclone filters, electrostatic precipitators, and fabric filter for PM.

2.3.1.5 Additional Industry Sub-sectors

The previous industry sub-sectors were discussed in more detail due to the significant energy conversion and/or emission impacts on regional AQ. Additional industry sub-sectors considered for the AQ assessment portion of this work include food processing, primary and secondary metal production, mining, chemicals production, and natural gas recovery and production. Cases were developed accounting for the removal of all emissions associated with the aforementioned industry sub-sectors and assessed for resulting impacts on ground-level ozone and $PM_{2.5}$.

2.3.2 Air Quality Impacts of Industrial Sub-sectors

The complexity and variation that exists amongst different industrial sub-sectors extends to the emissions and subsequent AQ impacts associated with each. Further, regional industrial energy sectors are very different including the major industries present in each region. Thus, the future AQ impacts of industry will differ from region to region and industry to industry.

To elucidate the AQ impacts of industry energy conversion in 2055 Cases were evaluated comprising the removal of emissions from a singular industry sub-sector. Impacts on ozone and $PM_{2.5}$ were quantified and discussed.

2.4 SUMMARY

Transitions to low-carbon generation technologies and fuels will alter pollutant emission patterns from U.S. energy sectors and subsequently impact regional AQ. In many cases these impacts could be beneficial; however some mitigation strategies could potentially worsen AQ. In addition, AQ impacts of alternative energy pathways are not generally as well understood as those for conventional technologies. Thus, there is a need for further understanding of how GHG mitigation strategies can be designed and deployed to maximize the human health and monetary co-benefits of reducing ambient levels of air pollutants.

A number of studies have examined life cycle GHG and life cycle or direct emissions associated with various alternative technologies (e.g., see [32-41]) or linkages between the physical impacts of climate change on emissions and AQ, e.g., rising mean temperatures driving emissions growth from amplified power generation needed for cooling[42], impacts on ambient PM[43] and ozone levels[44, 45]. The important relationship between climate change mitigation and AQ has been noted by previous researchers, but much of the available literature is focused on broadly quantifying and/or monetizing AQ co-benefits in total [46-50]. Few, if any, studies discuss the consequential pollutant emission and AQ impacts corresponding to specific technological, fuel, and behavioral perturbations included in mitigation strategies. Additionally, studies that are available generally assess only on a singular endpoint, e.g., GHG emissions **or** pollutant emissions. Further, many studies solely report potential *emission* perturbation magnitudes without considering spatial and temporal dynamics or the impacts to atmospheric primary and secondary pollutant levels. While this is an important step, understanding how emission perturbations arising from technological transitions affect concentrations of secondary pollutants, e.g., ozone or PM, is essential to accurately account for the true benefits and costs of carbon mitigation.

Chapter 3: Approach

The first step in this research is to outline priority areas and determine a methodology to feasibly achieve all the goals of the research. The following tasks directly address the established objectives for this work.

3.1 TASKS

The first task (Task 1) develops an understanding of which technological options should be considered for this work. The second and third tasks (Task 2 and Task 3) develop a modeling platform to facilitate the required AQ assessment. Task 3 provides a detailed and quantified assessment of various emissions sources to provide justification for the mitigation strategies chosen for assessment in the final three tasks (Tasks 4-6).

Task 1. UNDERSTAND MAJOR DRIVERS OF AQ AND GHG CONCERNS AND CHARACTERIZE MITIGATION STRATEGIES

The first task of this work requires the establishment of competency in the field by conducting a literature review of relevant areas of research. To study potential impacts from specific mitigation strategies a detailed understanding of the incorporated technologies and the factors regarding implementation must be gained. First, current energy system sources of emissions must be assessed and characterized. Next, potential alternative energy pathways must be identified and evaluated for all relevant factors pertaining to deployment status in 2055. Next, potential emissions (GHG and pollutant) and AQ impacts for strategies must be understood and accounted for. A thorough literature review is conducted accounting for all things including quantification of potential GHG reductions from base technologies, alterations to pollutant emissions both in quantity and spatial and temporal pattern, and reported resulting effects on atmospheric levels of pollutants, including ozone and $PM_{2.5}$.

Task 1.1. Conduct a literature review for GHG impacts of potential mitigation strategies

Task 1.2. Conduct a literature review for criteria pollutant and AQ impacts of potential mitigation strategies

Task 2. PROJECT ENERGY SYSTEM EVOLUTION TO 2055

Estimation of energy system evolution requires knowledge of deviations in primary energy resource supplies, energy conversion technologies, end-use demands, and various technological options to meet specified demands. Thus, fuel consumption and emissions data can be generated and applied to base year (2005) emissions via multiplicative emissions growth factors developed at the state-, technology-, fuel-, and chemical species-level (The base year emissions for this work will correspond the EPA-developed National Emissions Inventory from 2005). Regional energy system projections, including emission factors, will be derived by output from the Market Allocation (MARKAL) Model [31]. MARKAL is an energy optimization model designed to evaluate future energy systems evolution under given demands, constraints, and available technologies and fuels. Energy system details embodied in the model framework include primary energy resource supplies, energy conversion technologies, end-use demands, and various technological options to meet specified demands in power generation, residential, commercial, industrial, and transportation sectors. Model outputs include technologies, fuel use and emissions for projected future years. Information from energy sector projection can then be used to quantify changes in emissions from all relevant regional sources and utilized to grow a base year emissions inventory to a targeted future date under various potential scenarios.

Task 3. DEVELOP MODELING PLATFORM TO ASSESS AQ AND GHG IN 2055

Examining AQ metrics in future years requires the selection of regions for study, projection of baseline emissions, spatial and temporal resolution of emissions, and simulations of atmospheric chemistry and transport, the methodology is broken up into several stages as follows:

a. **Selection of Study Regions** - The regional nature of AQ coupled with differences in technologies and fuels comprising regional energy systems give importance to evaluating impacts in multiple areas of the U.S. The challenges faced in the U.S. with regards to AQ and GHG are determined by region-specific factors, including differing meteorological conditions, demands, technologies, fuels, policy, etc. Selection criterion will include existing/expected AQ challenges, distinctive energy systems, and the likelihood of alternative energy strategy deployment.

b. **Spatially and Temporally Resolve Emissions –** In addition to projecting evolution, the development of AQ model-ready emission fields must include spatial and temporal allocation and resolution, chemical speciation, generation of biogenic emission estimates and control of area-, mobile-, and point-source emissions. Fields will be developed for both baseline and alternative cases examining various technological outcomes via a scenario approach. For this work the Sparse Matrix Operator Kernel Emissions (SMOKE) Modeling System. SMOKE is an emissions processing system that develops appropriately formatted (gridded, speciated, and hourly) emissions for input into AQ models using a series of matrix calculations that allows for rapid and flexible processing of emissions data[290]. SMOKE carries out the core functions of emissions processing including spatial and temporal allocation, chemical speciation, generation of biogenic emission estimates and control of area-, mobile-, and point-source emissions. Growth factors for emission categories are entered into a SMOKE growth and control factor file and SMOKE then grows the base-year inventory via disaggregation of emissions into their constituent chemical species via a library of SCC-specific chemical speciation profiles. Spatial and temporal allocation into a 3-D modeling grid is accomplished through spatial surrogates and SCC-specific temporal allocation profiles. Point source emissions are allocated directly to the grid cell in which each source's coordinates are given. Non-point emission sources are characterized at the county level and emissions are allocated to grid cells via spatial surrogates. Further, SMOKE uses temporal activity profiles to allocate emissions to hour of day. Source-specific information used in allocation methodologies includes factors such as land use, census data, employment information, and others. SMOKE outputs an appropriately gridded file that can be used within an AQ model to simulate the impacts on AQ for a given region.

c. **Assess Air Quality Impacts–** Evaluation of emission impacts on ambient atmospheric pollutant concentrations requires simulations of atmospheric chemistry and transport via advanced models. Necessary inputs include meteorological conditions, initial and boundary conditions, land use and land cover information, and anthropogenic and biogenic source emissions. The AQ model used in this work will include the Community Multiscale Air Quality model (CMAQ), a comprehensive modeling system developed by the U.S. EPA and widely used for AQ regulatory and research purposes. The source code and technical formulation of the model are available from the CMAQ website: www.cmaq-model.org. CMAQ is designed from the "one atmosphere" perspective and is used for studies on tropospheric ozone, particulate matter, acid deposition and visibility. The CMAQ system includes a meteorological modeling system (MM5), emissions modeling system (EMS) and chemical transport modeling system (CTMS). Outputs of AQ models can be used to analyze technological shifts for impacts on primary and secondary pollutants, including ozone and fine particulate matter ($PM_{2.5}$).

Task 4. ASSESS SOURCES IN ENERGY SECTORS FOR AQ IMPACTS

Due to differences in key drivers, different energy sectors have significant variation in emission profiles, i.e., spatially and temporally, in magnitude, with importance for resulting AQ impacts. Additionally, regional variation in utilized fuels and technologies to meet energy demands result in relative differences in sector impacts across regions. The methodology developed in Task 2 and 3 will be utilized to study the impacts of major energy sectors (i.e., power generation, transportation, industry, residential, commercial) on GHG emissions and concentrations of secondary air pollutants, including ozone and $PM_{2.5}$. Sectors will be evaluated for impacts in each region to identify both intra- and inter-regional variation in atmospheric pollutant impacts occurring from emissions. The results will assist in identifying important sources and sectors as drivers of regional air pollution and climate change in 2055 that will be considered for mitigation strategy deployment.

Task 4.1. ASSESS IMPACTS FROM VARIOUS TRANSPORTATION SUB-SECTORS INCLUDING BOTH LDV AND NON-LDV TECHNOLOGIES

Of particular interest due to important contributions to regional AQ and GHG burdens, the transportation sector will be assessed for impacts in 2055. Transportation includes a large collection of very diverse emission sources including both on-road (LDV, HDV) and non-road (ship, rail, air, off-road) categories.

Reflecting dissimilar characteristics (technological, fuel, demands, spatial and temporal operation) emissions from transportation sub-sectors differ widely, including by composition, intensity, and spatial and temporal patterns. For example, ocean going vessels generally operate on low quality petroleum fuels with high emissions restricted to coastal areas. In contrast, LDVs have fewer emissions per unit fuel but have a tremendous demand in urban areas with high populations. Additionally, future emissions will be a product of demand growth and alterations in utilized technologies. Thus, impacts of transportation sub-sectors on regional AQ and GHG emissions will change to 2055, both relative and in aggregate, and thus further investigation is warranted. In this task emission from all transportation sources will be spatially and temporally resolved and then assessed for contributions to atmospheric pollutant burdens.

Task 5. DEVELOP AND EVALUATE POTENTIAL MITIGATION MEASURES

Results from Task 1 and additional insights gained in Tasks 2-4 will be used to identify both important opportunities and strategies to mitigate emissions in the sectors and sources of interest. Mitigation measures will be assessed for impacts on GHG emissions and AQ impacts, including reductions in ozone and PM2.5. Specific categories of assessment are directly related to those of high impact noted in Task-4 and include:

Task 5.1. ASSESS MITIGATION STRATEGIES FOR TRANSPORTATION

Assess identified key GHG mitigation strategies in the LDV sector for impacts on ozone and $PM_{2.5}$ to determine the preferred technological, fuel, and behavioral strategies in the attainment of co-benefits in selected study regions in 2055. Assess the potential GHG and AQ impacts of future deployment of hydrogen fuel via a scenario approach including the impacts of various hydrogen supply chain strategies and vehicle market penetration levels. Assess the potential GHG and AQ impacts of future deployment of electric vehicles via a scenario approach including the impacts of various power sector responses to vehicle charging demand including the co-deployment of carbon capture and storage.

The contributions to ozone and $PM_{2.5}$ of additional non-LDV transportation sub-sectors (HDVs, Offroad, Marine, Rail) will have high importance in 2055. In particular, emissions from marine vessels are with significant AQ impacts in all study regions, particularly in locations of major international shipping activity, i.e., Ports of L.A., Houston, and New York/New Jersey. Furthermore, the importance of addressing emissions from all transportation sectors will increase due to the evolution of the LDV sector in response to regulatory drivers. This work will analyze additional transportation sub-sectors for impacts on ozone and $PM_{2.5}$ to determine the preferred technological, fuel, and behavioral strategies in the attainment of co-benefits. The goal of this task is to assess impacts on AQ in 2055 to determine effective strategies to reduce environmental impacts of U.S. shipping ports. The substantial goods movement activity in 2055 in study regions, including the locations of several major shipping ports, merits investigation of the impacts on AQ. An analysis of changes in emissions and subsequent impacts on AQ from mitigating strategies designed to target high-emitting sources at major U.S. shipping ports;

including changes in technologies, fuels, and infrastructure. Scenarios will be developed to evaluate contributions to regional ozone and $PM_{2.5}$ levels from individual sources including ships, trains, heavy duty trucks, and offroad equipment. Additionally, emission mitigation strategies will be recognized and evaluated for sources identified as having a high impact, including ocean going vessels. The marine sector is expected to substantially impact AQ in terms of both ozone and $PM_{2.5}$ in study regions in 2055. The deployment of a range of efficiency improvement and emission reduction strategies will be evaluated to identify strategies that can improve AQ in tandem with CO_2 reductions.

Task 5.2. ASSESS MITIGATION STRATEGIES FOR POWER GENERATION

The power sector is a key contributor to GHG and air pollution burdens in all regions, although specific drivers differ due to underlying variation in regional power grids (i.e., demands, fuels, technologies, regulatory statutes). Important emission sources identified in Task 2 will be delineated and suitable mitigation strategies in the power generation sector for impacts on ozone and $PM_{2.5}$ to determine the preferred technological, fuel, and behavioral strategies in the attainment of co-benefits. Potential strategies include various renewable resources, nuclear generation, and carbon capture and storage (CCS) in addition to improvements in efficiency and conservation. For regions containing significant generation from coal-based generation pathways the AQ and GHG impacts of replacement with low-carbon technologies and fuels relative to the deployment of CCS will be analyzed.

Task 5.3. ASSESS MITIGATION STRATEGIES FOR INDUSTRY

Industrial energy divisions are characterized by a wide range of varying needs and processes and correspondingly utilized fuels and technologies represent a diverse range, including in regards to emissions. Resulting AQ impacts from industrial sector emissions can differ from others, e.g., power generation, due to spatial, temporal, and composition variation. For example, ozone formation dynamics differ downwind of power plants and industrial sources due to discrepancies in NO_x and VOC emission rates and ratios [20, 28, 33]. Thus, modeling is required to better understand spatial distributions of impacts from alterations to baseline levels of industrial sector activity.

First, industrial sub-sectors will be evaluated for regional impacts and appropriate mitigation strategies identified and assessed that can achieve reductions in emissions and improve AQ. Next this work will assess the impacts on ozone and $PM_{2.5}$ from identified industrial sector mitigation strategies using spatially and temporally resolved emission fields followed by simulations of atmospheric chemistry in CMAQ. The AQ impacts of various mitigation measures deployed in the domestic industrial sector will be evaluated for important regional sources of industry. Results will be used to identify the most effective strategies to reduce harmful impacts on AQ from industrial sector activity in study regions in the future.

a. In particular, impacts of the petroleum fuel production industry will be assessed. Emissions from the production, distribution, and storage of petroleum-based fuels contribute significantly to regional levels of ozone and $PM_{2.5}$ and enhance the AQ benefits of transitioning to alternative transportation fuels. Further, areas most effected, e.g., localized to refineries adjacent to the major urban populations or ports, often coincide with communities currently experiencing serious deleterious health impacts from poor AQ. Thus, displacing petroleum refinery emissions represents an important opportunity to maximize the AQ benefits of alternative transportation technologies and fuels.

Task 6. DETERMINE OPTIMAL MITIGATION MEASURES TO IMPROVE AQ IN TANDEM WITH GHG EMISSION REDUCTIONS

The results from previous tasks will be used to develop insights into construction of comprehensive multi-sector strategies to best improve AQ in regions of study in the future. Potential mitigation strategies will be evaluated for the ability to improve AQ and reduce GHG emissions and recommended for given regions. Important sources of emissions contributing to levels of ozone and $PM_{2.5}$ will be identified in various energy sectors and discussed. Mitigation strategies with effectiveness in reducing ozone and $PM_{2.5}$ with region specificity will be discussed as will strategies that have success across regions (e.g., applicable at the national level).

Chapter 4: Evaluate Contributions to Regional Air Quality Burdens from Emissions Associated with Energy Sector Sources

4.1 INDIVIDUAL ENERGY SECTOR GHG AND AIR QUALITY IMPACTS

Future efforts to mitigate climate change will include transitions to alternative technologies and fuels seeking reductions in GHG emissions from U.S. energy sectors. In addition, displacement of conventional energy strategies will impact emissions of pollutant species directly influencing regional AQ due to common generation processes and sources. Currently, sectors of paramount GHG concern include transportation and power generation, which combined total over half of domestic GHG emissions and account for the bulk of emissions driving regional AQ concerns in many U.S. regions including ambient concentrations of ozone and particulate matter[291]. As such, regulatory efforts often focus on reducing emissions from currently significant sources in the transportation and power generation sectors including power plants and light duty vehicles. However, future emissions from additional sectors (i.e., industrial) and sources (i.e., ships, rail, offroad) could intensify in importance to AQ, particularly in light of expected demand growths and technological advancement.

The work in this section evaluates the AQ and GHG impacts of U.S. energy sectors in 2055 to identify emerging sources with the potential to profoundly impact AQ and to assist in determining the preferred GHG mitigation strategies that concurrently improve AQ. Three regions of the U.S. with typically poor AQ are evaluated; California, Texas, and the Northeastern US (NEUS). Baseline AQ is assessed accounting for the evolution of major emission drivers, including increases in key sector activity occurring from population and economic growth. Next, alternative scenarios are developed accounting for the spatial and temporal distribution of key sources to evaluate impacts on ambient pollutant concentrations from emission perturbations, including on ozone and $PM_{2.5}$. Though previous studies have evaluated the emissions[292, 293] or AQ[294, 295] impacts of individual sectors, technologies or fuels, few have analyzed comparative impacts of multiple sectors across several U.S. regions using advanced models to account for spatial and temporal emissions perturbations in tandem with simulations of atmospheric chemistry and transport.

4.1.1 Individual Sector GHG Impacts

Figure 9 displays the relative share of total regional GHG emissions for energy sectors in 2055 in the Base Case. In line with the present, power generation and transportation continue to represent the highest emitting sectors, although regional differences exist (i.e., the highest emitting sector in TX is power while transportation dominates in CA and the NEUS). However, emissions from additional sectors have importance, including industrial emissions in TX and residential and commercial emissions in the NEUS. Additionally, the importance of addressing GHG emissions from all energy sectors is heightened in light of the dramatic reductions identified as necessary for climate change mitigation[5].

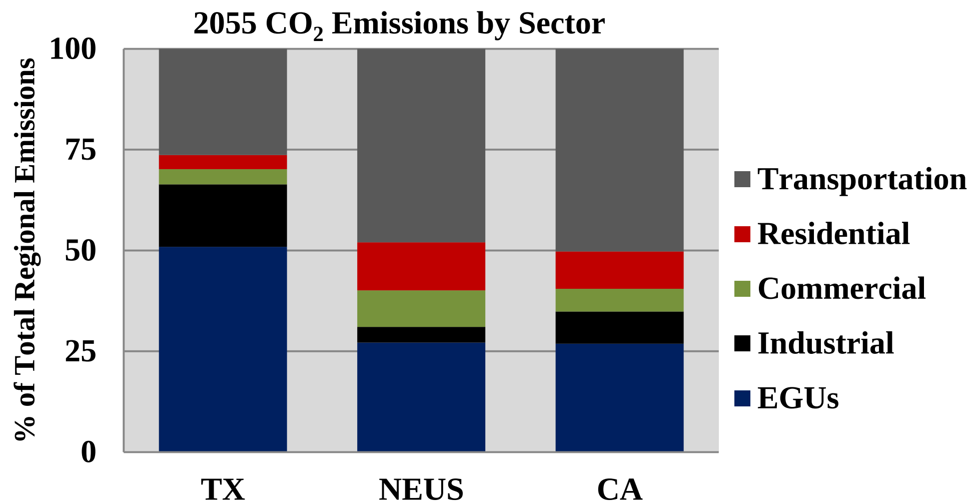
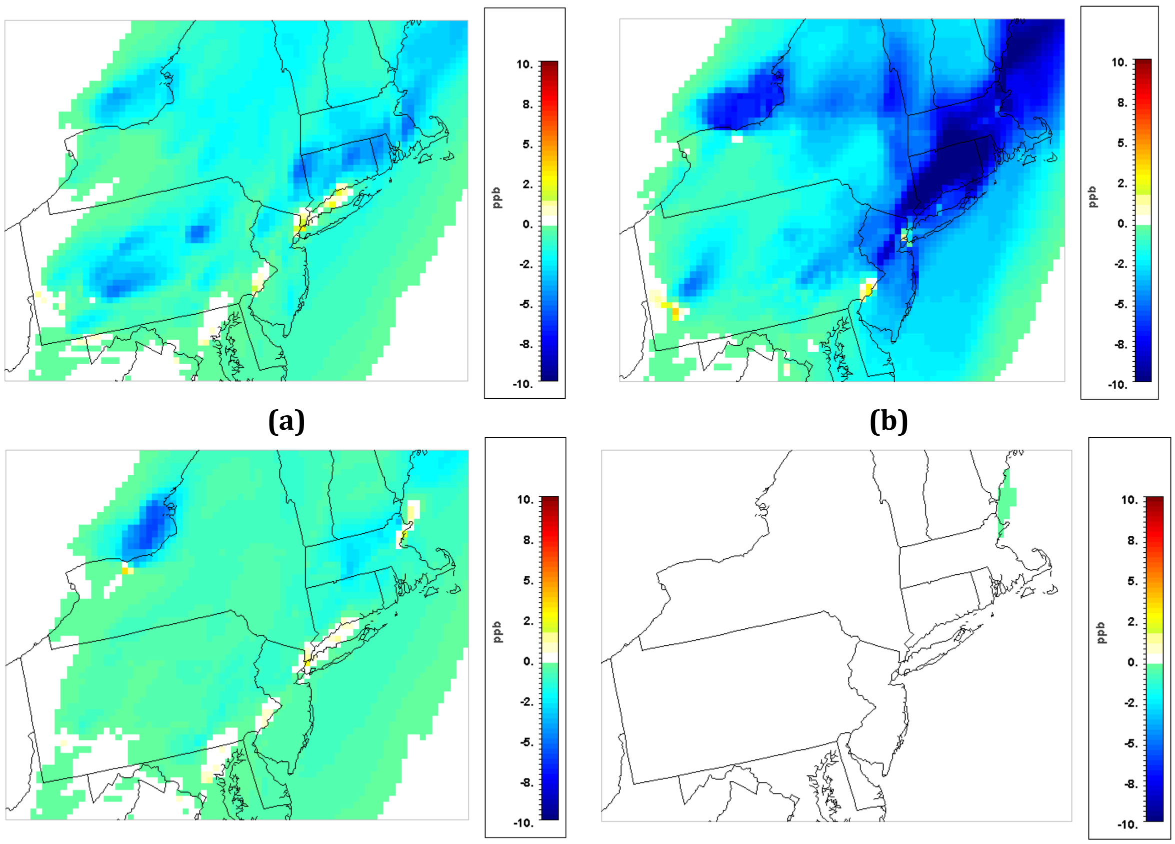


Figure 9: Share of Total Regional CO_2 Emissions by Energy Sector in 2055

4.1.2 Individual Sector Air Quality Impacts

In 2055 variation in energy sector profiles, demands, and constraints between study domains produce differences in regional emission patterns with corresponding disparities in ozone and $PM_{2.5}$ impacts. For all regions evaluated removing emissions from transportation activity contributes to the largest reductions in ground-level ozone in terms of magnitude and spatial area. As can be seen in Figure 10, improvements in maximum 8-hr ozone levels for the NEUS exceed 13 ppb in heavily impacted locations including areas upwind of the New York City metropolitan area. Similar trends are seen in TX and CA, with peak reductions 2 to 3 times greater than those from any other sector and important impacts occurring downwind of major urban centers. Power sector impacts in TX and the NEUS display localized plumes characterized by significant reductions while impacts on ozone from power generation in California are modest due to a relatively clean generator mix. Additionally, the industrial sector was shown to have comparable impacts to power generation on ozone in all regions, although spatial differences occur in the distribution of resulting reductions. Reductions in ozone from the removal of commercial and residential sector emissions was also evaluated and found to be significantly less than the three previously discussed sectors.



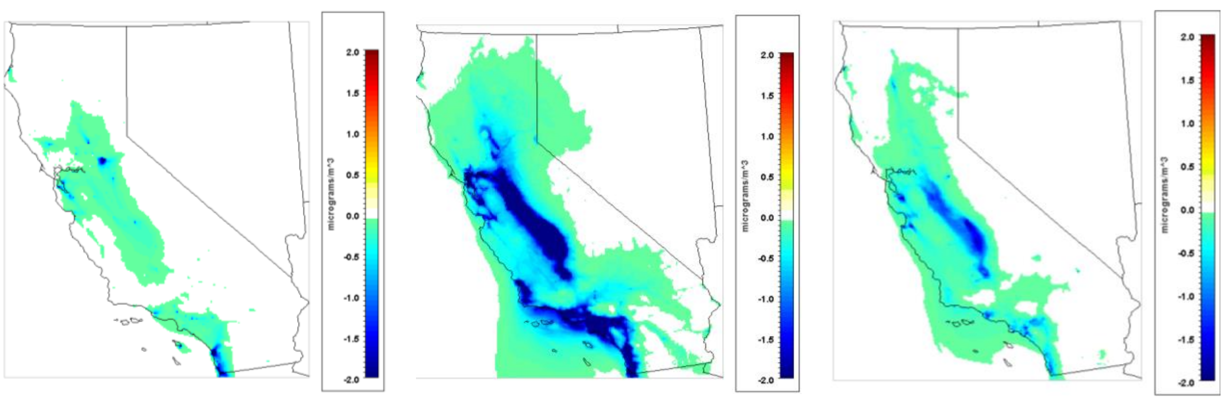
(a) (b) (c) (d)

Figure 10: Impacts on maximum 8-hr ozone from the removal of emissions from the (a) Power, (b) Transportation, (c) Industrial, and (d) Commercial Sectors in the NEUS

Maximum improvements in regional 24-hour $PM_{2.5}$ levels are displayed in **Table 11.** As can be seen, sector impacts on $PM_{2.5}$ display more regional variability than ozone, e.g., transportation emissions are the largest contributor to ambient $PM_{2.5}$ concentrations in TX and CA while power sector emissions dominate in the NEUS. It should also be noted that considering peak reduction magnitudes alone fails to capture the totality of AQ impacts. For example, though power sector relative to industrial emissions in CA yields a significantly larger peak improvement (-25.8 vs. -2.6 μg/m^3), impacts spatially are highly localized to generator sites (Figure 11). In contrast, reductions from industry are distributed over a larger area and include regions currently represented by non-compliance with Federal NAAQS, including the San Joaquin Valley and South Coast Air Basins[296].

Table 11: Peak reductions in 24-h $PM_{2.5}$ from Sector Emissions Removal in 2055

Sector	Transportation	Industrial	Power
Region	**Δ $PM_{2.5}$ 24-hr [μg/m^3]**	**Δ $PM_{2.5}$ 24-hr [μg/m^3]**	**Δ $PM_{2.5}$ 24-hr [μg/m^3]**
TX	-4.7	-2.1	-2.3
NEUS	-2.3	-7.2	-5.1
CA	-44.6	-2.6	-25.8



(a) (b) (c)

Figure 11: Impacts on 24-hr $PM_{2.5}$ from the removal of emissions from the (a) Power, (b) Transportation, and (c) Industrial Sectors in CA

The evolution of regional energy systems to 2055 will directly influence emission patterns and subsequent GHG and AQ impacts. As current, the power and transportation sectors contribute the largest sector shares of GHG emissions, though with varied individual importance. In terms of ozone, transportation represents the sector with the highest potential for achieving AQ co-benefits from GHG mitigation in all regions. Sector impacts on $PM_{2.5}$ experience enhanced regional variation for co-benefits, i.e., the industrial sector in the NEUS and power sector in TX. Further, industrial sector emissions were found to have important effects on ozone and $PM_{2.5}$ in all regions and should be considered equitable to the power sector for mitigation in 2055, particularly in CA. These results further demonstrate the importance of considering regionally-specific AQ and GHG mitigation strategies to maximize co-benefits.

4.2 SECTOR COMBINATION AIR QUALITY IMPACTS

Though important information can be obtained regarding the relative implications of sector-level emission impacts on AQ, such analyses fail to capture interactions of emissions from different sectors which are critical determinants in the formation and fate of secondary pollutants. In order to investigate impacts and linkages from emissions across sectors and to facilitate comparison, cases involving emission reductions from multiple sectors in different combinations were assessed. The TX study region was chosen for analysis because regional emissions from the three major energy sectors (power, transportation, and industrial) are more balanced from a magnitude standpoint, e.g., in CA emissions from the transportation sector dominate AQ impacts relative to power generation.

4.2.1 Sector Combination Scenario Development

The three major energy sectors identified in Section 4.1.2 as having the most significant impacts on TX regional AQ include power generation, transportation, and industrial activity. To investigate cross-sector impacts, cases were developed to account for the 50% co-reduction of emissions from two sectors in tandem while leaving the third at baseline levels. For example, the 50E/50T Case corresponds to a 50% reduction in power generation and transportation emissions only, with industrial sector activity remaining constant. Similarly, the 50E/50I Case involves power generation and industrial emissions being halved while transportation emissions are not perturbed.

In addition, to examine the impacts of moderate emission reductions from all sources, a case was developed representing 25% emission reductions for all species across all sectors relative to the 2055 Base Case. The All 25 Case can be considered representative of deploying strategies that increase the efficiency of energy conversion in all sectors, although decreases for some sources are not based on feasible or expected values reported in the literature. Further, the All 25 Case further provides a baseline to compare the impacts of major energy sectors relative to each other and to other emission sources.

4.2.2 Sector Combination Emission Results

As can be seen in Figure 12, from a quantitative standpoint emission impacts between cases are not equivalent and vary by sector and chemical pollutant species. In TX, the 50E/50T Case results in the largest GHG improvement, with the 50E/50I Case achieving the next highest reduction. Further, the All 25 Case achieves a greater carbon reduction than the 50T/50I Case. These results demonstrate the importance of power generation in TX with respect to GHG mitigation efforts as all cases involving electricity provision achieve greater GHG reduction than the one case which does not.

However, quantitative impacts on pollutant emissions differ by sector and chemical species. Impacts on NO_x are greatest for the 50T/50I Case which achieves over a 40% reduction from baseline levels and the 50E/50T Case at over 30% removal. The 50E/50I and All 25 Cases have similar NO_x impacts, around 25%. Impacts on the emission of VOCs follow similar trends, with both cases involving transportation corresponding to the largest improvements and emissions in the All 25 Case reduced relative to the 50E/50I Case. The results demonstrate that, for ozone-precursor pollutant emissions, the transportation sector is the quantitatively dominant sector in TX and should be targeted for reductions. Interestingly, the transportation and industrial sector cases had larger ozone improvement potential relative to power generation.

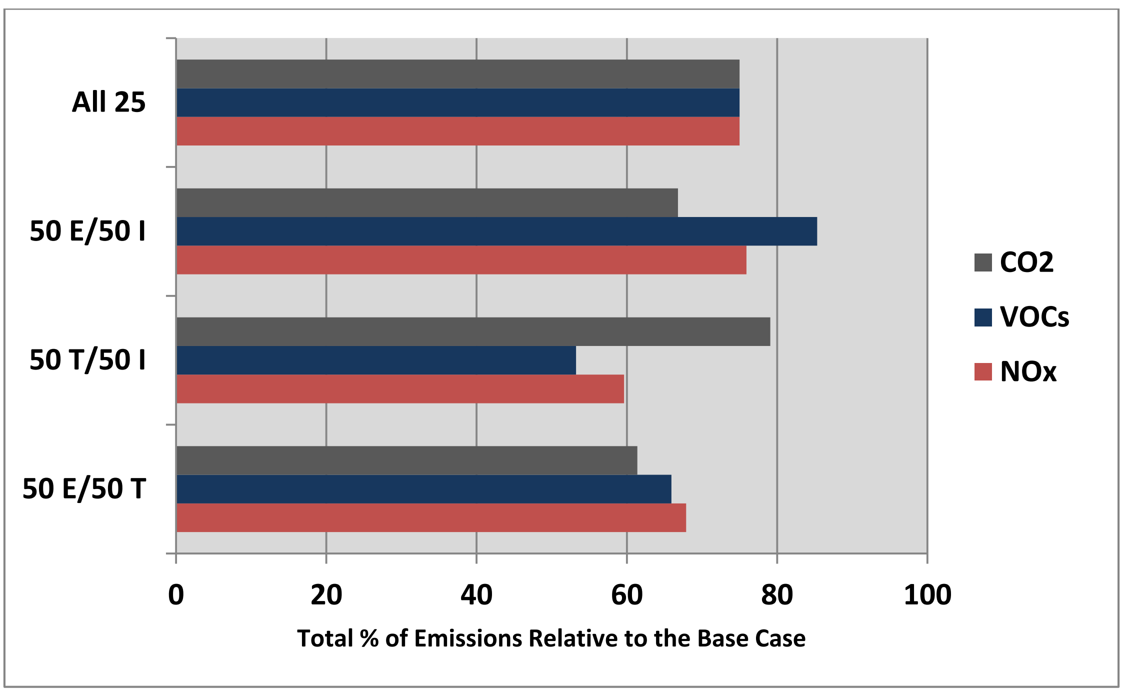


Figure 12: Emission impacts in sector combination scenarios by relative % of Base Case

The results presented in Figure 12 give some initial information regarding the relative contributions of emissions by sector. However, analysis based on quantitative perturbations only fails to account for the spatial and temporal characteristics of ozone formation. Thus, cases were evaluated for emission perturbations via SMOKE to determine spatial and temporal emission distributions including geographic patterns and relative rates.

Spatially, NO_x emission patterns are similar for all cases and demonstrate the significant activity in urban locations attributable to all three sectors, including Greater Houston, Dallas-Ft. Worth, Austin, and San Antonio. Interestingly, the largest spatial impacts occur in the 50E/50I Case which has a distributed profile for emission reductions including source locations not present in other cases.

Despite spatial similarities, relative NO_x reduction emission rates vary dramatically amongst cases. The largest peak reduction occurs in the 50T/50I Case with removal of 24-h average NO_x over 850 kg/hr in some grid cells including the Greater Houston area. Similarly, the 50T/50E Case reduces emissions over 800 kg/hr with similar areas of impact. The All 25 Case results in peak reductions in NO_x roughly half that of the cases involving the transportation sector. Interestingly, despite a larger area of impact the 50E/50I Case results in relative emission reductions much lower than all other cases, with peak impacts reaching 58 kg/hr and highlights the large contribution from transportation.

These results reflect the quantitative and qualitative disparity that exists in regards to sectoral emissions and potential AQ impacts. For example, despite a larger area of impact the 50E/50I Case is associated with significantly lower reduction rates. Contrastingly, both cases involving the transportation sector achieve dramatic reductions in locations associated with urban centers, particularly in combination with the industrial sector. Further, implications for AQ effects include the potential for improvements in regional ozone levels near large population centers.

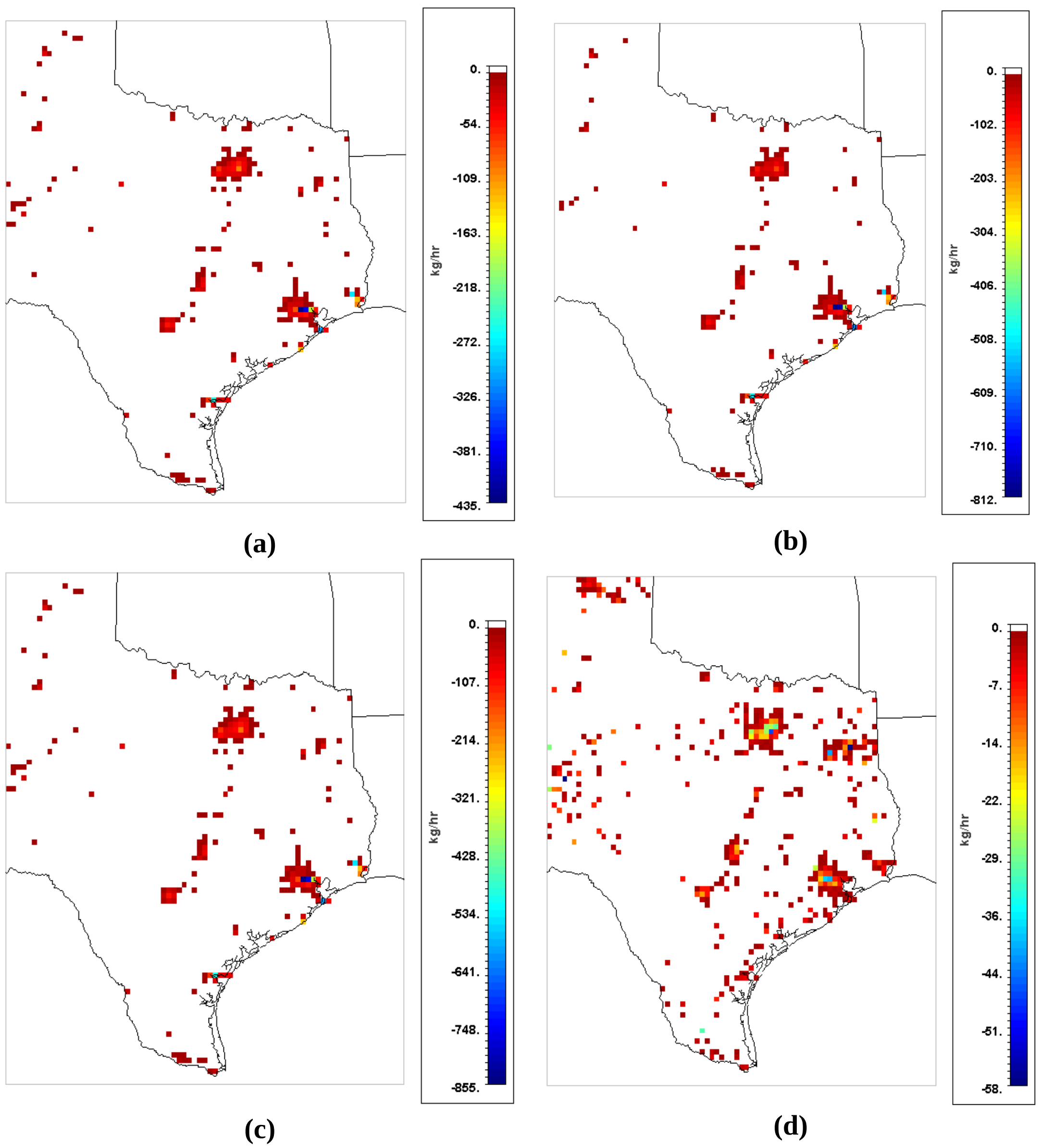


Figure 13: Impacts on 24-h NO_x for the (a) All 25, (b) 50 T 50E, (c) 50T 50I, and (d) 50E 50I Cases

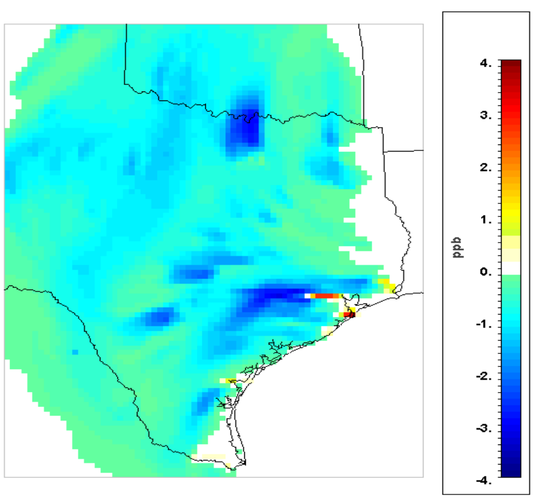
4.2.3 Sector Combination Air Quality Simulation Results

As can be seen in Figure 14 and Figure 15, the spatial dimensions of pollutant impacts differ between cases, reflecting differing source distributions and emission patterns. Reducing emission from all sectors by 25% in the All 25 Case moderately improves ground-level ozone across large areas of TX with peak reductions reaching 3.2 ppb downwind of major urban centers. Improvements in $PM_{2.5}$ include peak reductions up to 1.8 μg/m^3 occurring in the center of the region and downwind of major urban areas. Improvements also occur in the more sparsely populated northeastern, middle and western portions of the state. These results are intuitive as emissions are lowered from all sources and locations corresponding to reduced concentrations of secondary pollutants that follow spatial and temporal patterns of formation and fate established in the Base Case. In addition, larger improvements are seen for sectors and areas that have high emissions, as evident in overlap with transportation sector cases.

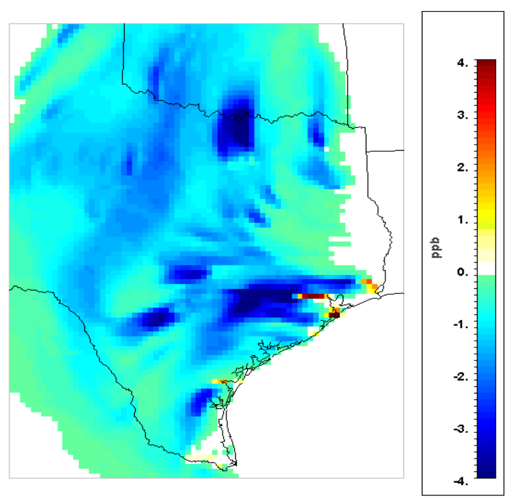
Impacts in the 50T/50I Case include the largest ozone reductions of 6.7 ppb, including downwind of Dallas-Ft. Worth and Houston, during periods of maximum background levels. Reductions in peak $PM_{2.5}$ reach 4.9 μg/m^3 and occur with similar patterns to ozone. Additionally, reductions localized to industrial areas in coastal regions occur, likely as a result of emissions from petroleum fuel refining and additional sector sources adjacent to petrochemical refinery complexes.

Reducing emissions from transportation and power sector activity by 50% in the 50T/50E Case reduces peak ozone and $PM_{2.5}$ over 5 ppb and 4 μg/m^3 (Figure 14 and Figure 15). Major impacts on ozone are visible downwind of urban regions reflecting the removal of ozone-precursor emissions from transportation sector activity. Additionally, improvements are seen in the northeast region of the State corresponding to the locations of large coal power generators. Reductions in $PM_{2.5}$ occur west of Greater Houston, potentially from interactions between transportation and coal fired power plant emissions. Additionally, impacts from petroleum refineries are evident. Interestingly, $PM_{2.5}$ levels are improved in the 50T/50E Case but with less robustness than for cases involving the industrial sector.

Impacts of co-reductions from industrial and power generation emissions result in modest ozone improvements relative to other cases. Spatially, reductions in ozone occur over the entire study region during periods of peak formation. However, peak impacts are represented by narrow plumes originating from major power plants with reductions reaching 3.2 ppb. Contrastingly, improvements in 24-h average $PM_{2.5}$ are larger than the All 25 and 50T/50E Cases and similar in magnitude to the 50T/50I Case. Improvements occur over the middle of the state associated with urban areas and peak at 2.7 μg/m^3 over the Dallas/Ft. Worth area. In addition, plumes of reduced $PM_{2.5}$ concentrations, though moderate in magnitude, can be observed downwind of Greater Houston. These results demonstrate that interactions between transportation and industrial emissions have a role in the formation of secondary PM, particularly downwind of areas supporting both large urban populations and industrial activity such as Greater Houston.



(a)



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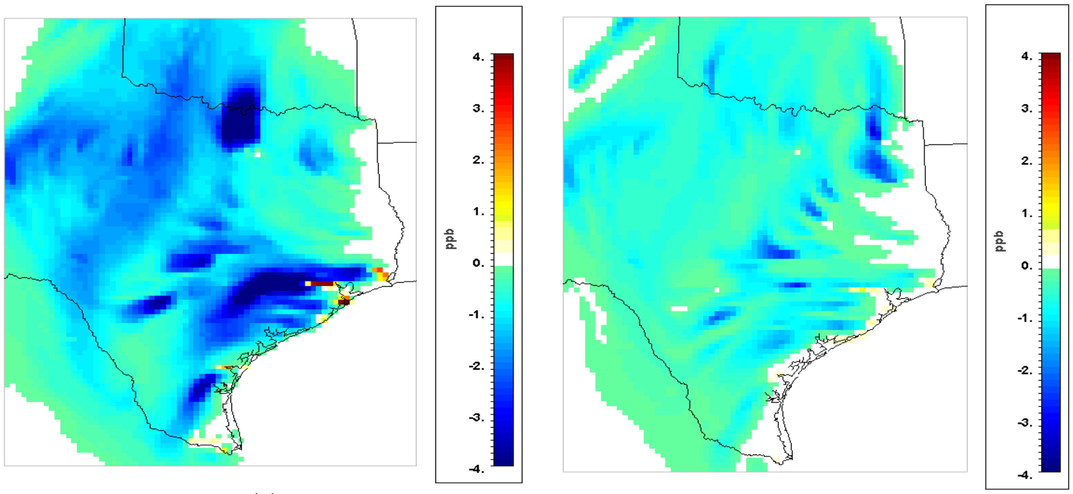
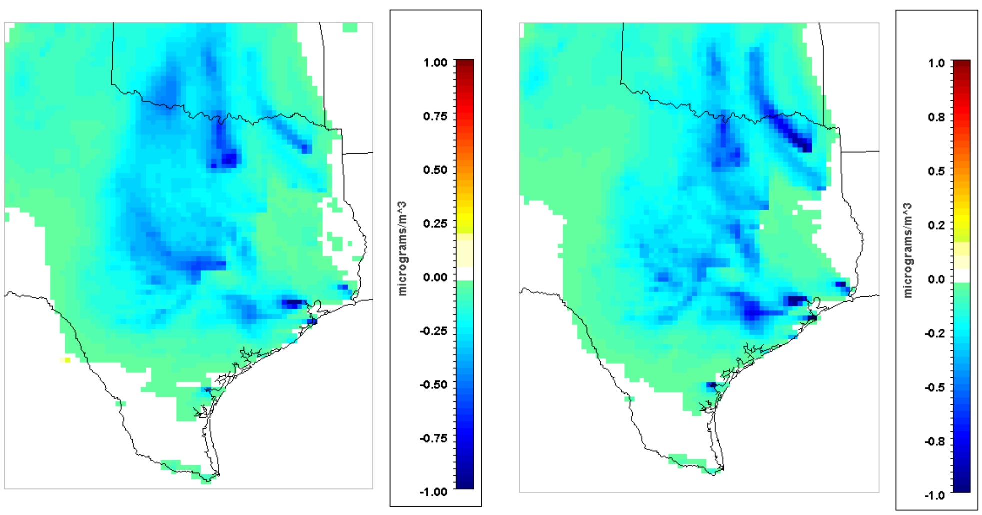
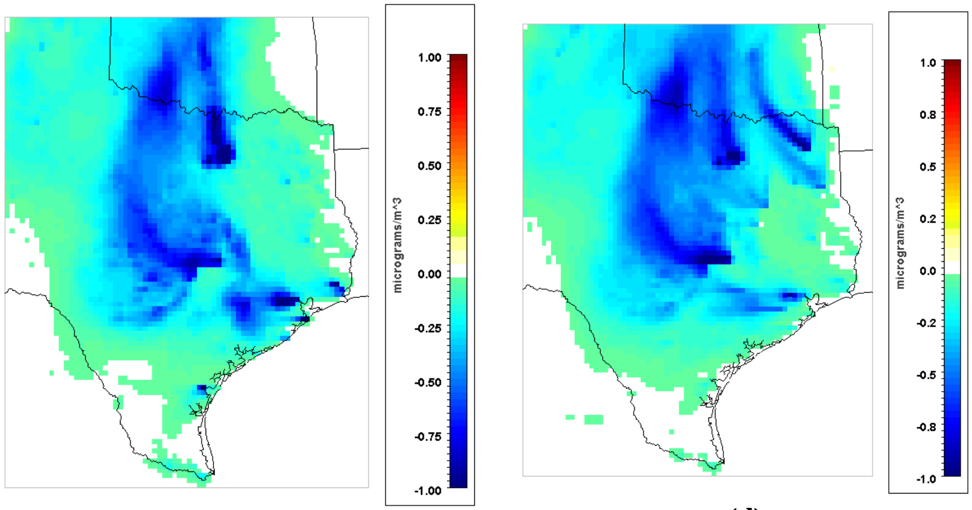


(c) **(d)**

Figure 14: Impacts on peak ozone in the (a) All 25, (b) 50T/50E, (c) 50T/50I, and (d) 50E/50I Cases



(a) **(b)**



(c) (d)

Figure 15: Impacts on 24-h $PM_{2.5}$ for the (a) All 25, (b) 50T/50E, (c) 50T/50I, and (d) 50E/50I Cases

4.2.3.1 Sector Combination Conclusions

Maximum improvements in 24-h average NO_x and VOC emissions and resulting improvements in peak ozone and 24-h average $PM_{2.5}$ are displayed in Table 12. The 50T/50I Case achieves the largest improvements in both peak $PM_{2.5}$ and ozone. It should be noted that peak values correspond to one model grid cell where the highest reduction occurs and thus fails to capture spatial impacts. For example, though the 50T/50I Case has a larger peak $PM_{2.5}$ improvement it is only slightly more than the 50E/50I Case which entails a greater area of impact. Thus, it could be argued the electricity and industrial sector case is the most important from a $PM_{2.5}$ standpoint.

In addition to further validating the importance to AQ of industrial sector emissions in TX, these results show the differences in spatial and temporal impacts on secondary pollutants that are associated with emissions from various energy sectors. Thus, reducing equivalent quantities of direct emissions from different sectors can have very different impacts on the formation and fate of secondary pollutants. Further, these results demonstrate that mitigation strategies which reduce emissions in important sectors achieve greater improvements than balanced reductions of a lower magnitude across all sectors, i.e., comparatively the 50T/50E and 50T/50I Cases yield significant improvements in ozone relative to the All 25 Case. In addition, the temporal pattern of ozone and $PM_{2.5}$ formation and fate differ with maximum impacts for the 50T/50I and 50T/50E Cases occurring in densely populated regions with human health implications. Contrastingly, the 50E/50I and 50T/50I Cases were associated with the most significant impacts on regional $PM_{2.5}$ concentrations and emphasize the importance of the industrial sector in improving particulate levels.

These results also demonstrate the disparity in TX with regards to sectors of importance for GHG and regional AQ. Due to this, achieving maximum reductions in GHG emissions may not necessarily correspond to paramount AQ improvements. From a GHG standpoint, strategies that target low- or zero-carbon power generation should be pursued while transportation sector strategies with low- or zero-NO_x emissions would be most effective in addressing regional ozone concerns. Contrastingly, targeting $PM_{2.5}$ reductions would be best served by addressing sources in the industrial and power sectors. For example, as can be seen in Figure 12, the 50E/50T scenario achieves the greatest reduction in GHG emissions for the TX study region despite the 50T/50I scenario maximally improving AQ in some locations. However, the 50E/50T case does significantly lower concentrations of ozone and $PM_{2.5}$ and offers AQ co-benefits. Thus, decisions prioritizing targeted sectors and fuels for strategy deployment must balance the effectiveness of GHG mitigation and AQ improvement in deciding which endpoint is more desirable in a region with regards to political and societal goals.

Table 12: Impacts on 24-h average NOx and VOC emissions and peak ozone and 24-average PM2.5

Scenario	Δ 24-hr NO_x [kg/hr]	Δ 24-hr VOC [moles/s]	Δ Peak Ozone [ppb]	Δ 24-h $PM_{2.5}$ [μg/m³]
All 25	-435.3	-1.5	-3.2	-1.6
50T/50E	-811.8	-2.9	-5.5	-2.5
50T/50I	-855.0	-2.9	-6.7	-3.0
50E/50I	-58.4	-0.05	-3.2	-1.2

4.3 TRANSPORTATION SECTOR

Various transportation sub-sectors differ extensively in features determining consequent distributions and intensities of emissions including purpose, technological and fuel characteristics, spatial and temporal patterns of operation, regional demands, etc. Further, evolution patterns to 2055 of major emission drivers will not be equivalent and thus some sub-sectors may grow in relative importance to regional AQ while others may lessen e.g., alternative, low-emitting technologies may be easier to develop and apply in the LDV sector than those applicable to replace current ship or rail technologies. As such, there is a need for more insight into how each area of transportation contributes to regional AQ challenges, particularly in coming decades, as insights can be gained into how to best develop and deploy mitigation strategies. Additionally, identification of priority targets for reducing emissions can assist decision makers in formulating effective legislation aimed at reducing the environmental impacts of transportation.

The variation amongst transportation sector emission sources is significant with regards to spatial and temporal patterns and intensities. Further, differences in key factors will result in dissimilar patterns of evolution including changes in total emissions Thus, resulting impacts on AQ in 2055, including ground-level concentrations of ozone and $PM_{2.5}$, will differ amongst transportation sources and from current. The following work is designed to evaluate the contributions from the individual transportation sub-sectors

Sources of transportation emissions attributable to on-road sources include vehicles traveling on roadways for the purpose of transporting passengers and/or freight. Categories of on-road sources include LDVs, MDVs, HDVs, and motorcycles. Currently, and in the 2055 Base Case, the majority of on-road vehicles are powered by the combustion of petroleum fuels (i.e., motor gasoline, diesel) which produce significant emissions of pollutants and GHGs. Emissions from non-road sources include those from vehicles, engines, and equipment used for a variety of different purposes including construction, agriculture, transportation, recreation, and many others. Non-road sources are examined under transportation sector impacts as they are generally grouped as mobile sources for emission inventories. Included are emissions associated with marine shipping and port related activity. In addition, emissions from rail transportation include those from locomotives.

The complexity of ozone formation and fate (e.g., ground-level concentrations depend on quantity, transport, and spatial/temporal profiles of precursor emissions, meteorological conditions, regional topography, etc.) requires detailed, 3-D atmospheric models be used to simulate regional AQ in predicting tropospheric concentrations[297]. Similarly, atmospheric modeling is required to assess how direct emissions from transportation effect penultimate regional $PM_{2.5}$ levels, accounting for spatial and temporal distributions of both primary and secondary particulate[298]. This work evaluates contributions of various transportation sources to regional AQ burdens in three important U.S. regions by predicting how direct emissions translate to atmospheric concentrations of ozone and $PM_{2.5}$. To assess regional AQ impacts in 2055 of transportation-related sources, emissions must be justifiably projected from current levels and spatially and temporally resolved to facilitate input into an advanced model of atmospheric chemistry and transport. Geographic regions selected for study include California (CA), an aggregate of five Northeastern U.S. states (NEUS), and Texas (TX) due to the regional nature of AQ coupled with significant differences in regional energy infrastructures (e.g., demands, utilized technologies and fuels, regulatory constraints). Baseline AQ is established in the year 2055 accounting for business-as-usual continuation of current technological, energy, and economic trends via output from a data-intensive, energy systems economic optimization model, the MARket ALlocation (MARKAL) model. Emissions are then grown to 2055 from current levels and spatially and temporally resolved to account for direct perturbations using an emissions processing tool, the Sparse Matrix Operator Kernel Emissions (SMOKE) modeling system. Finally, simulations of atmospheric processes are conducted using the Community Multi-scale Air Quality model (CMAQ) version 4.7, with the Carbon Bond 05 chemical mechanism to establish fully developed distributions of atmospheric concentrations of pollutants of ozone and $PM_{2.5}$.

To assess the impacts of each transportation sub-sector (LDV, HDV, off-road, ships, rail) scenarios are constructed accounting for the removal of emissions from a given sub-sector while holding the other constant with the baseline. Atmospheric modeling is performed for 2055 baseline and transportation sub-sector scenarios for summer ozone episodes defined by a seven day period to dissipate the effect of initial conditions. The ozone and $PM_{2.5}$ impacts of removing emissions from transportation sub-sectors is quantified by determining differences in concentrations between the baseline and alternative scenarios. The results are displayed as difference plots with the specified scenario minus the baseline scenario reported as maximum 8 hour average ozone and 24 hour average $PM_{2.5}$. Additionally, peak reductions in ozone and $PM_{2.5}$ observed in study region for a given scenario are reported in Table 17 (It should be noted that while useful, peak quantitative impacts lack information regarding spatial distribution of impacts and a comprehensive assessment requires consideration of both). The goal of these spanning scenarios is to provide collective insight into the AQ impacts of different components of transportation in the year 2055.

4.3.1 Light Duty Vehicles

The operation of LDVs has been correlated with a variety of regional and local AQ problems and addressing vehicle emissions has been a key U.S. pollutant control strategy since the 1950s. Currently, LDV-related emissions continue to be a major regulatory focus, in part due to substantial existing and projected demands. In order to elucidate the future relationship between LDVs and regional AQ, particularly in relation to both other transportation sub-sectors and technologies in other sectors, scenarios were established to determine resulting impacts on pollutant concentrations from removing or altering the fleet compositions

To evaluate regional AQ impacts attributable to LDV operation, spatially and temporally resolved emission fields must be developed accounting for any associated with or arising from regional LDV demands. Direct vehicle emissions must be removed from onroad mobile sources comprising LDV technologies; which occur on various roadway forms and generally follow distinct diurnal patterns that must be adjusted to accurately represent a given scenario.

In addition to those from vehicle tailpipes, emissions corresponding to stationary point and area sources associated with the production, storage and distribution of consumed fuel must be accounted for. Though a portion of LDV demand is met with alternative fuels (e.g., electricity, ethanol) in the Base Case, total amounts are small relative to overall consumption and petroleum fuel (e.g., motor gasoline, diesel) use is dominant. As such, in this work only fuel pathway emissions associated with sectoral petroleum fuel production and consumption are considered.

Using SMOKE, emissions are adjusted for all LDV technologies operating on every roadway category (e.g., rural interstate, principal arterial, major collector). In addition, petroleum fuel pathway emissions were adjusted to account for alterations in traditional LDV fuel consumption. Emissions from petroleum refineries are assumed to be proportional to product outputs and are allocated based on the fraction of a given fuel (e.g., motor gasoline) relative to other products. Refinery output distributions differ by region and year and 2055 values for each study region are available in MARKAL outputs corresponding to the Base Case. The determined perturbation is then combined with the initial 2055 growth factor, yielding a final multiplicative factor that can be applied to appropriate sources. For example, gasoline is 52% of 2055 CA refinery outputs and complete removal of LDVs results in an equivalent reduction. In the Base Case refinery emissions are reduced 15% from 2005 levels thus reducing the remaining 85% by 52% yields a final factor of .408. Though in actuality refinery emissions are not precisely proportioned by output, this methodology represents an acceptable method for emissions accounting, particularly in light of the many sources of uncertainty inherent with refinery emissions.

Though some alternative fuel use (e.g., electricity, ethanol) is attributable to LDVs in the Base Case, amounts relative to total consumption are very low and gasoline continues to dominate consumption. As such, only LDV emissions associated with petroleum fuel production, distribution, and storage are considered and reduced according to sectoral removal parameters.

An initial scenario was developed for all study regions involving the complete removal of LDV and associated fueling infrastructure activities (No LDV Case). Though achievement of a zero-emissions LDV sector is likely unrealistic in the study horizon, important information can be gained including the establishment of upper bounds for impacts on various pollutant species. In addition, insights gained from No LDV Cases will assist in the development of more realistic scenarios involving alternative technologies and fuels in subsequent work.

Complete removal of LDV activity significantly impacts total emissions in all regions, although quantitative differences occur and reflect differences in technologies and fuels driven by state regulations, i.e., fleet-wide efficiencies, alternative fuel deployment levels. For example, the CA study region is characterized by lower-emitting vehicle technologies that dramatically reduce onroad vehicle emissions in the 2055 Base Case, i.e., -88% total NO_x from 2005 levels. Though the LDV fleet compositions for TX and the NEUS have higher emission intensities relative to CA, and thus have a larger potential for AQ improvement in 2055, total fleet emissions are reduced from 2005 levels in both regions. In addition to reductions in direct vehicle emissions, the Base Case also is characterized by reductions in emissions from fueling infrastructure, i.e., petroleum refinery NO_x is reduced by 12-15%.

Complete removal of emissions attributable to LDV operation has sizable impacts in all study regions. As can be seen in Figure 16 (a), spatial allocation of emission impacts via SMOKE in TX demonstrates substantial decreases in large metropolitan areas and major roadways with direct vehicle perturbations dominating total impacts. In addition, many refinery complexes are co-located in urban regions and contribute significantly to reductions (e.g., the greater Houston region). Temporal emission patterns associated with periods of LDV travel dictate the timing of reductions, with maximum reductions associated with afternoon peak travel demands. Additionally, the dramatic reductions in ozone precursor emissions (i.e., NOx and VOCs) support further investigation of secondary pollutant impacts via AQ modeling.

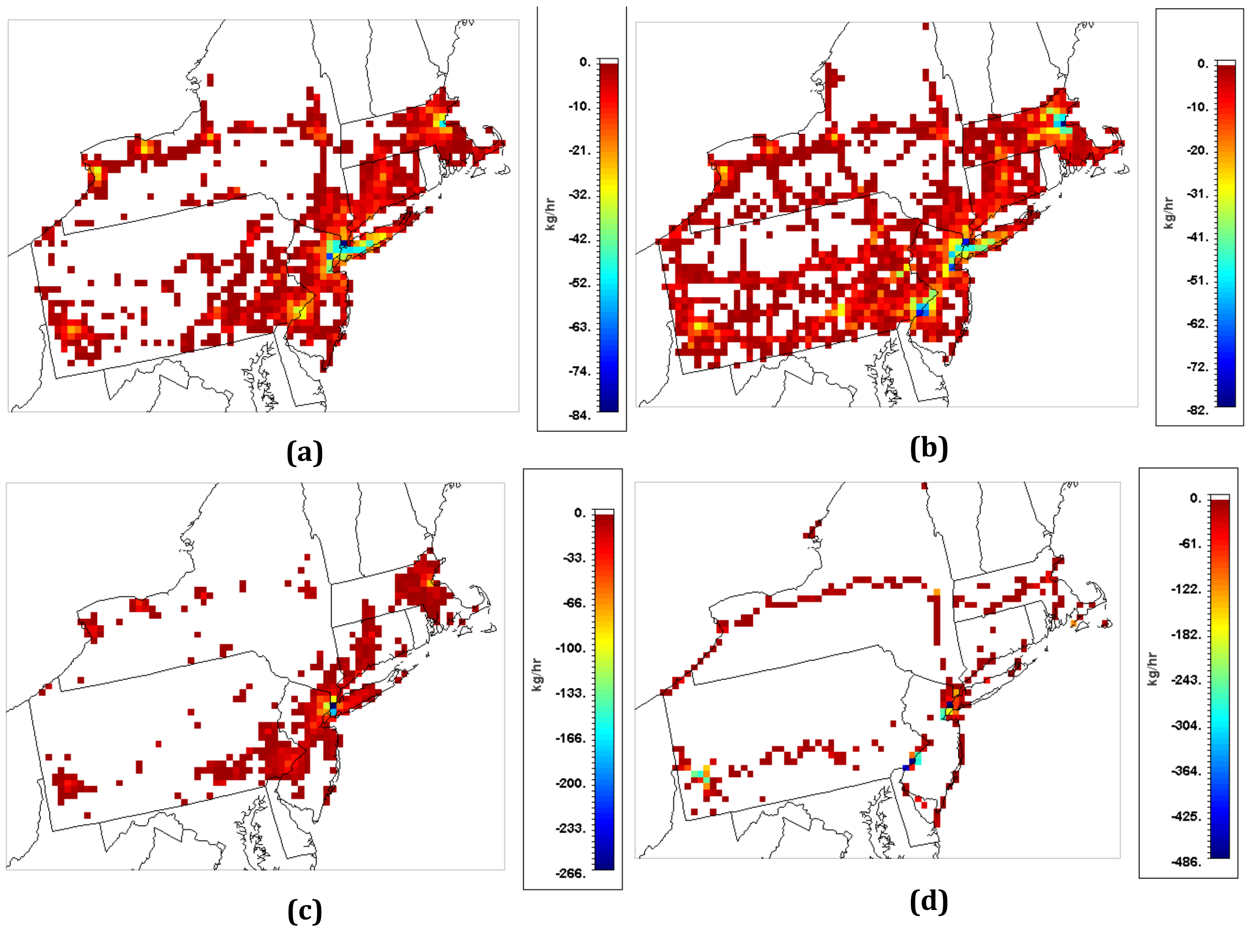


Figure 16: Impacts on 24-h average NO_x emissions from (a) LDV, (b) HDV, (c) Offroad, and (d) Marine and Rail Sources

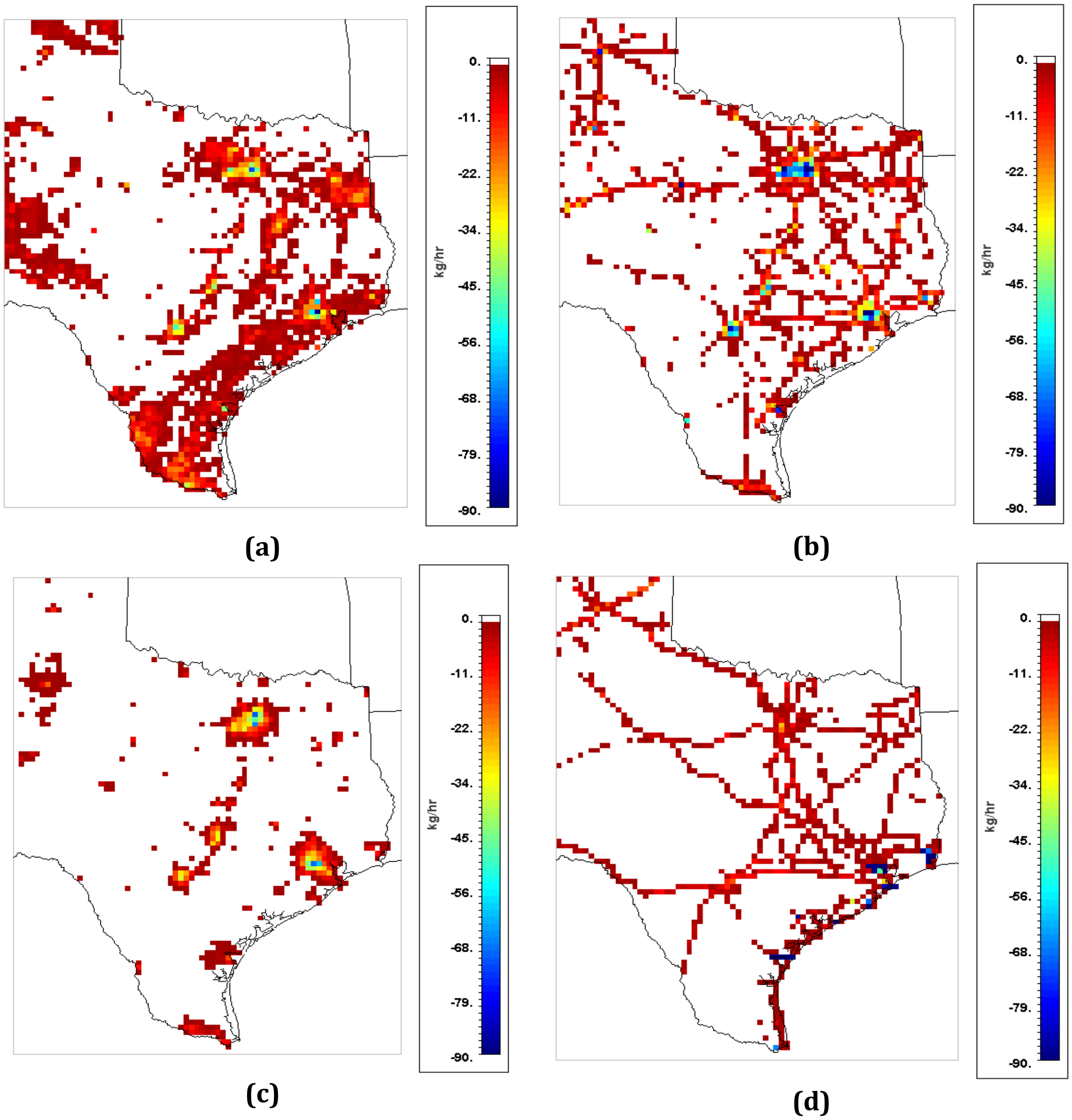
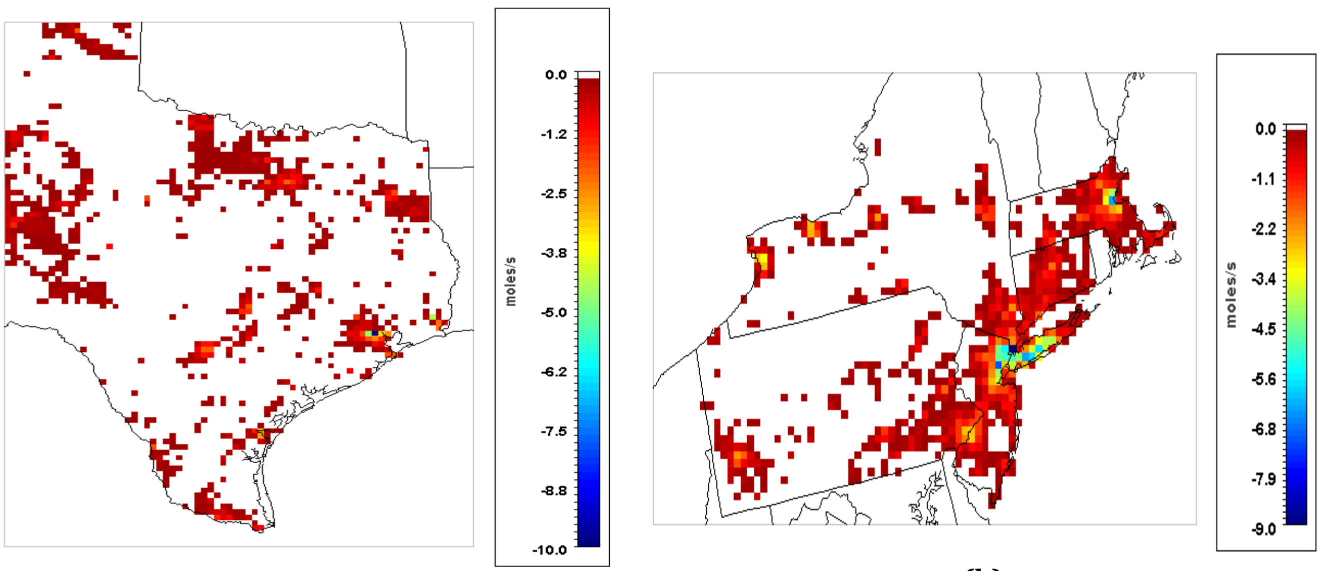


Figure 17: Impacts on 24-h average NO_x emissions from (a) LDV, (b) HDV, (c) Offroad, and (d) Marine and Rail Sources. ** Scale has been normalized to facilitate comparison across cases.

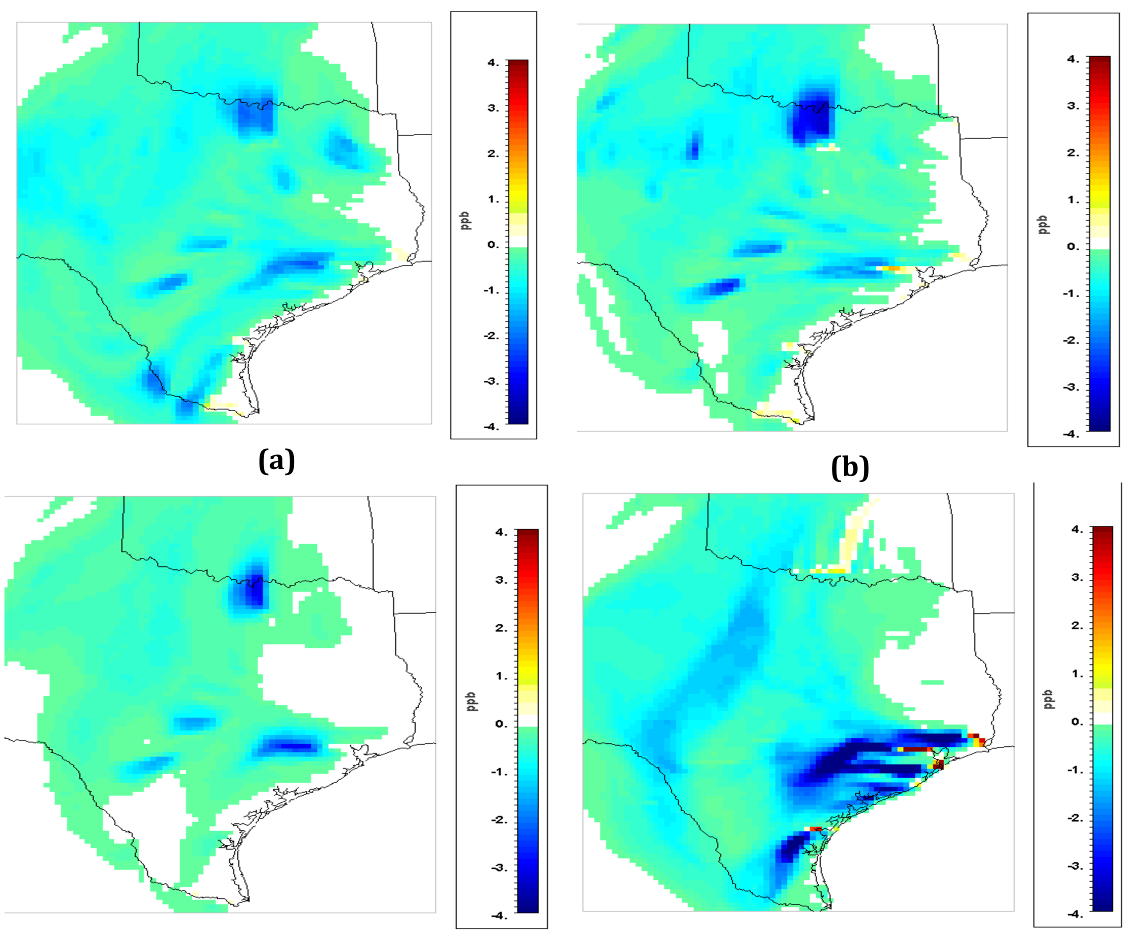
VOC emission reductions in the No LDV Case follow similar spatial and temporal patterns as NO_x with magnitudes being similar between regions. The TX and NEUS regions experience peak reductions of roughly -9 and -10 moles/s, respectively. Similarly, emissions of VOCs are reduced by nearly 2 moles/s in CA. Urban areas experience the largest improvements and diurnal patterns result in greater improvements during the daytime hours. Refinery complex locations experience significant improvements in emissions. For example, in the TX study domain impacts along the coast coincide to the locations of major refinery activity.



(a) (b)

Figure 18: Peak reductions in VOC emissions from Base for No LDV Scenario in (a) TX and (b) NEUS

The resulting impacts on peak ozone from removal of emissions associated with LDV-fleet activity in 2055 are displayed in Figure 19 (a), Figure 20(a), and Figure 21(a), for TX, NEUS, and CA, respectively. The removal of LDV-related emissions in CA results in improvements in peak ozone (-4 ppb) and $PM_{2.5}$ (-6.5 μg/m^3) that are correlated with urban areas associated with high levels of vehicle traffic, e.g., SoCAB and the Bay area. Additionally, impacts from refinery activity are visible as a result of emission reductions from reduced gasoline production. In the NEUS improvements in peak ozone and $PM_{2.5}$ from the Base Case reach 5.8 ppb and 1.5 μg/m^3. Impacts on ozone are larger in magnitude than other study regions, particularly upwind of the New York metropolitan area as the modeled meteorological episode progresses. This trend is due to the intense cluster of LDV emissions and refinery activity located in the NYC area. However, other NEUS sub-regions (i.e., Pittsburgh, Philadelphia) experience reductions in ozone that are more similar to those experienced in other study regions. In TX, reductions in peak ozone concentrations of 2.6 ppb and $PM_{2.5}$ of 1.5 μg/m^3 are observed and correspond to Houston, Dallas-Ft. Worth, San Antonio, and Austin. AQ impacts are larger for both the TX and NEUS regions relative to CA as the respective LDV fleets retain greater emissions intensities and thus have a larger impact when removed.



(c) **(d)**

Figure 19: Impacts in TX on peak ozone from (a) LDV, (b) HDV, (c) Offroad, and (d) Marine and Rail

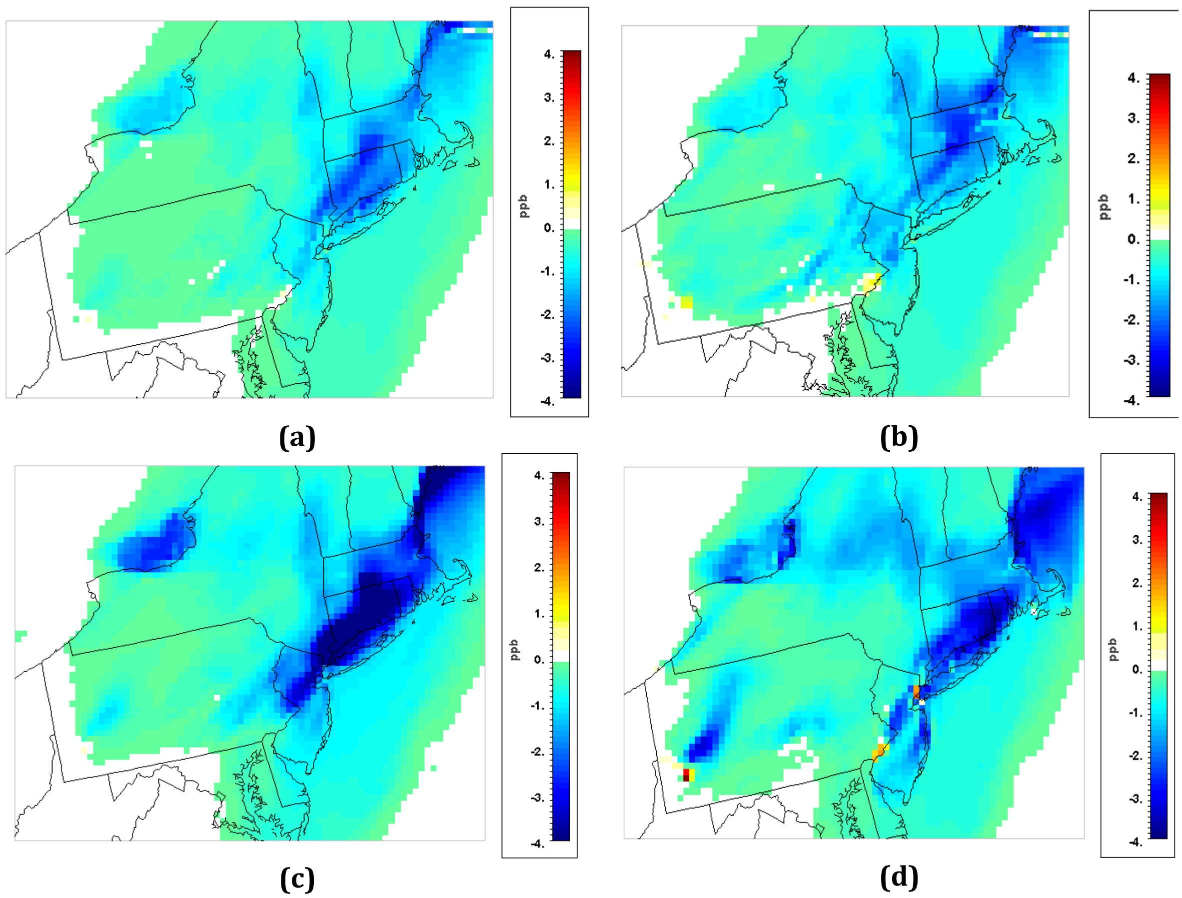


Figure 20: Impacts in NEUS on peak ozone from (a) LDV, (b) HDV, (c) Offroad, and (d) Marine and Rail

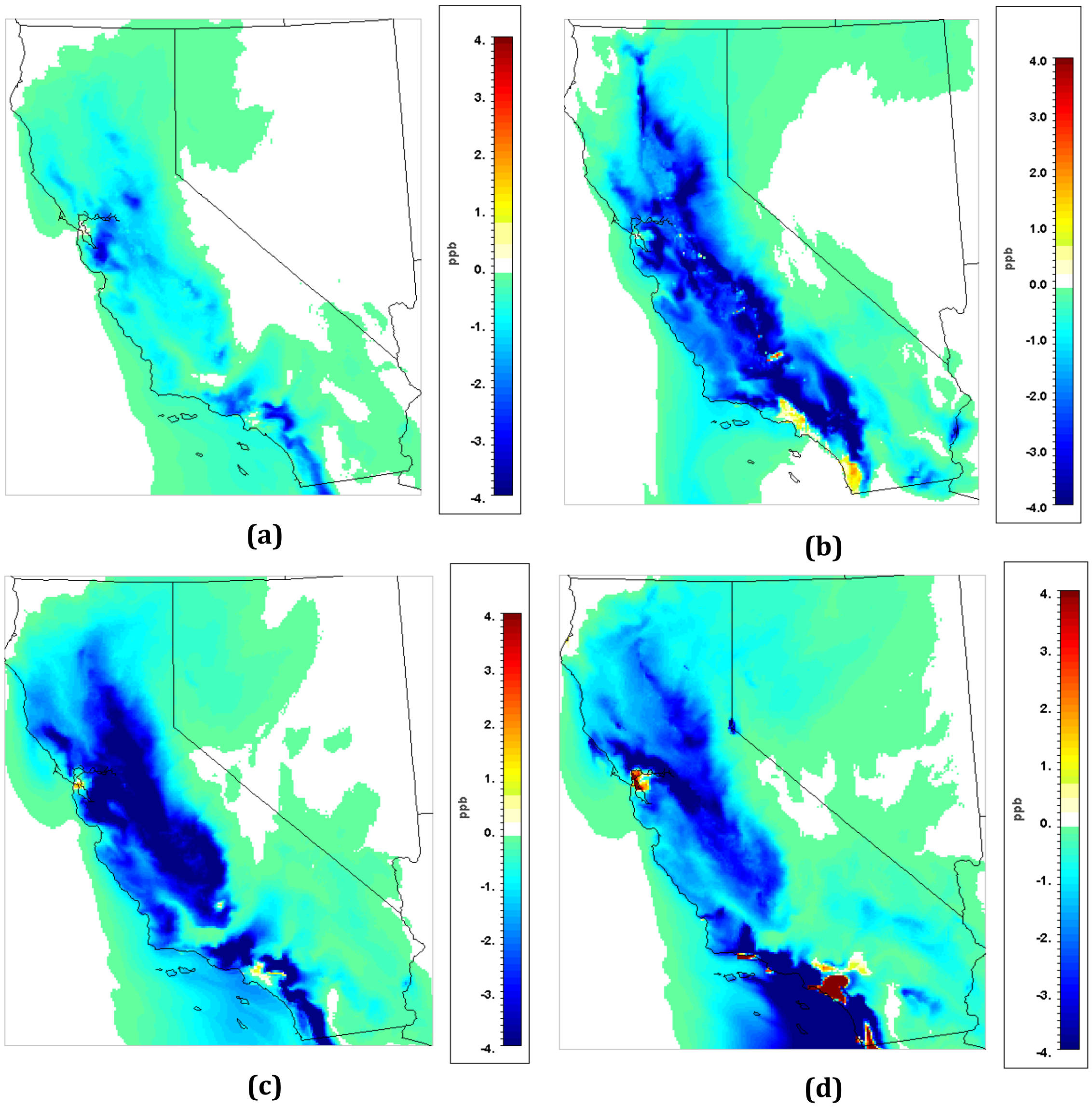


Figure 21: Impacts in CA on peak ozone from (a) LDV, (b) HDV, (c) Offroad, and (d) Marine and Rail

****Scale normalized across sub-sector cases****

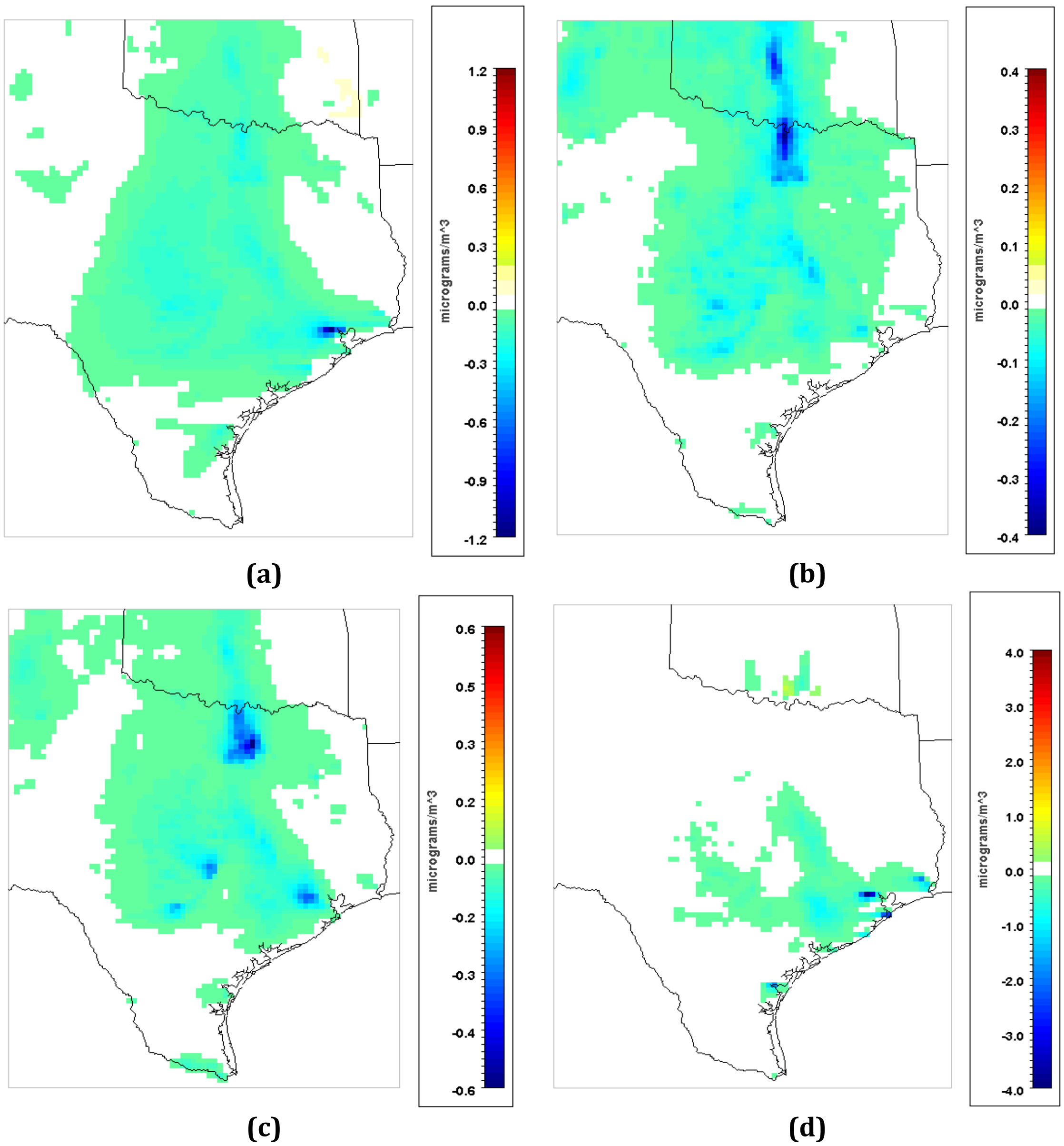


Figure 22: Impacts in TX on 24-h PM2.5 from (a) LDV, (b) HDV, (c) Offroad, and (d) Marine and Rail

****Scale not normalized across sub-sector cases****

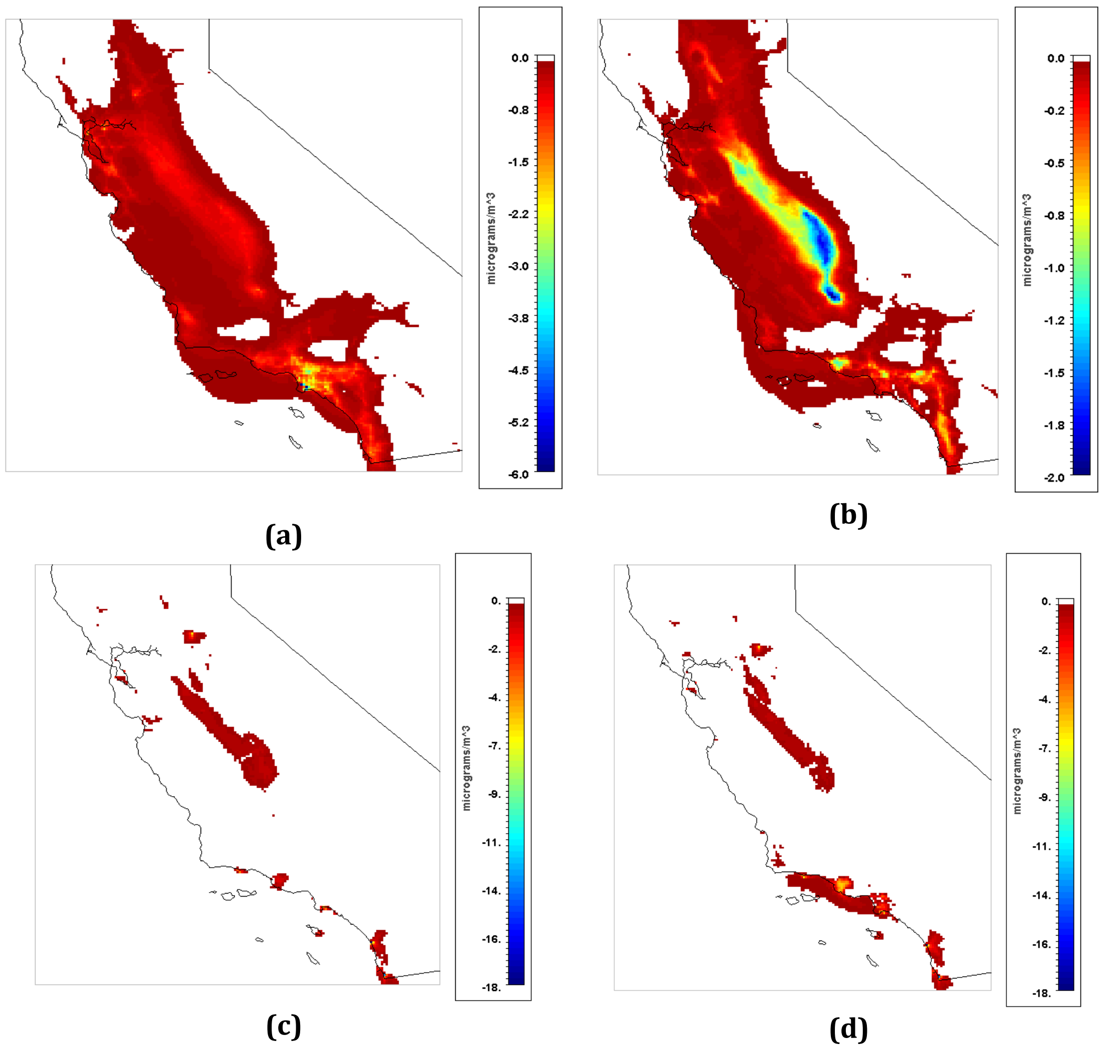
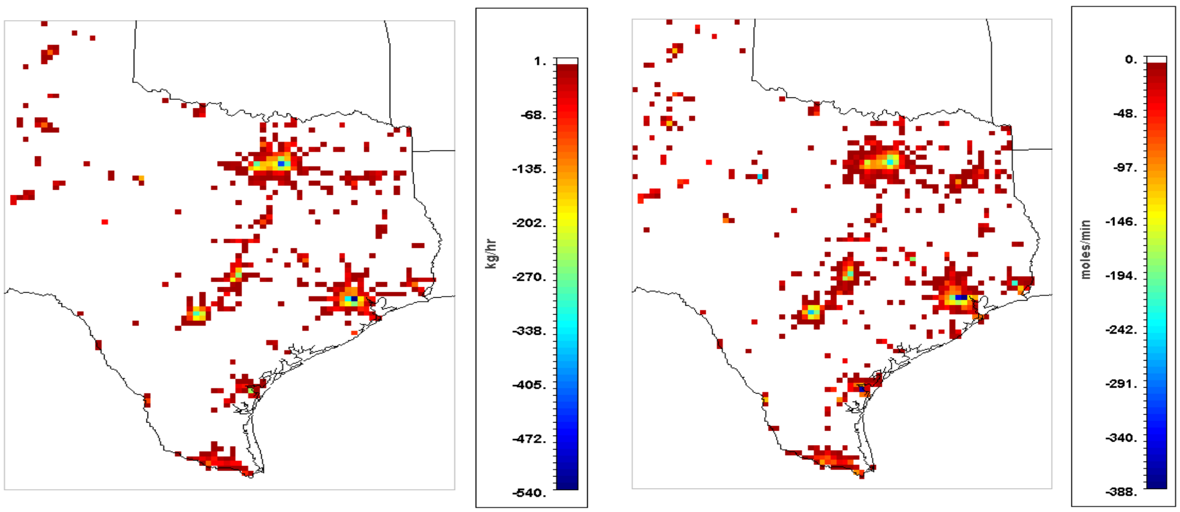


Figure 23: Impacts in CA on 24-h $PM_{2.5}$ from (a) LDV, (b) HDV, (c) Offroad, and (d) Marine and Rail

4.3.1.1 Evolution of LDV Impacts from 2005 Levels

Due to significantly reduced emissions as a result of regulatory and economic drivers (e.g., 2055 fleet-wide NO_x emissions are less than 80% of 2005 levels for all regions); relative LDV AQ impacts are reduced in magnitude in the study horizon. Figure 24 demonstrates reductions in 24-h average NO_x and VOC emissions from removal of the LDV fleet in 2005 in the TX study region with peak reduction rates reaching 540 kg/hr NO_x in the center of urban areas. The impacts from 2005 are significantly higher than peak NO_x removal rates for the same case in 2055 (i.e., Figure 17) of 89.7 kg/hr.



(a) **(b)**

Figure 24: Reductions in 24-h average (a) NO_x and (b) VOC emissions from removal of LDV in 2005

As would be expected, the evolution of the LDVs characterized by a lower emitting fleet reduces the relative impacts on ambient AQ, i.e., reduced improvements in ozone and $PM_{2.5}$ when activity is removed. Figure 25 displays the resulting relative concentrations of ozone and $PM_{2.5}$ when LDV fleet activity is removed from the region in 2005, with ambient concentrations reduced by almost 7 ppb and 1.2 μg/m^3. When compared to values from the 2055 case (Figure 19), ozone improvements are nearly three times and $PM_{2.5}$ reductions are twice as large. These results are intuitive due to the magnitude difference in pollutant emissions from 2005 to 2055 as a result of significant efforts to address LDV AQ impacts that is assumed to result in a cleaner vehicle fleet.

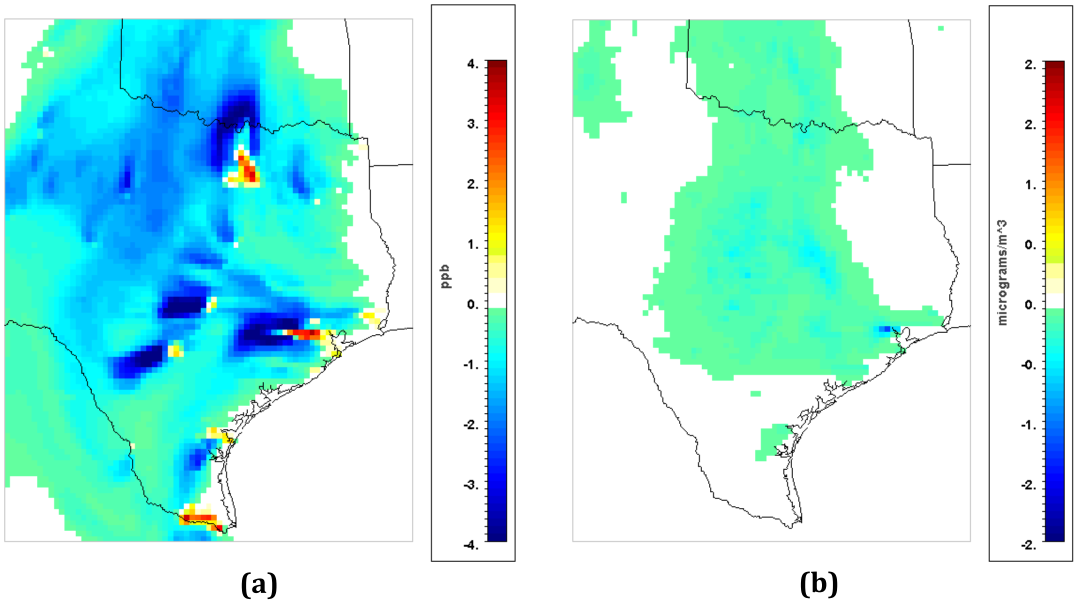


Figure 25: Impacts on (a) peak ozone and (b) 24-h $PM_{2.5}$ from the removal of the LDV fleet in 2005

4.3.2 Heavy Duty Vehicles

Removing HDV activity results in substantial emission reductions in all study regions. As can be seen in Figure 26, impacts on 24-h NO_x include peak reductions of 155, 82, and 60 kg/hr in TX, the NEUS, and CA, respectively. The value for TX is particularly important and is second only to Marine and Rail in terms of removal rate magnitude among sub-sectors. Spatially, impacts are clustered in urban regions with the largest impacts occurring in the center of major cities. Additionally, spatial roadway patterns are more distinct than LDV impacts and demonstrate the important major transportation corridors utilized heavily for freight truck transport, i.e.,, interstate highways.

It follows then that removing HDV emissions improves regional AQ with regards to ozone and $PM_{2.5}$ levels. In TX the No HDV case has peak reductions in ozone and $PM_{2.5}$ of up to 2.6 ppb and 0.4 μg/m³ and in the NEUS improvements include -5.6 ppb and -2.5

μg/m^3relative to the Base Case. In CA impacts on peak ozone are significant, equivalent to almost 13 ppb reductions in some locations, while $PM_{2.5}$ improves nearly2 μg/m^3.

Impacts on emissions and, subsequently AQ, for HDVs relative to LDVs are of significant, e.g., in CA removing HDVs yields an additional 55.4 kg/hr NO_x in some locations relative to removing LDVs (Figure 26). Contrastingly, the removal of LDVs yields a deeper increase in directly emitted PM (e.g., 16.5 kg/hr in some locations), which is somewhat surprising given the compression ignition diesel-fuel vehicles associated with HDVs are generally associated with PM emission concerns. Figure 27 displays difference plots for ozone and $PM_{2.5}$ for the No HDV Case relative to the No LDV Case in CA. As can be seen, ozone concentrations are lower for the No HDV Case over much of the region and during peak formation periods. However, exceptions include major urban areas (e.g., SoCAB) which experience localized improvements for the No LDV case. This is intuitive as urban regions are characterized by high levels of LDV fleet activity which result in high levels of emissions from personal vehicle travel. Contrastingly, 24-h $PM_{2.5}$ levels are higher for the No HDV Case relative to the No LDV Case in much of the state, with the exception of a region in the central valley encompassing Bakersfield. As with direct PM, these results are unexpected as it is indicative that LDVs have a greater impact on both direct and secondary PM than HDVs. Additionally, similar patterns for ozone and $PM_{2.5}$ were observed in TX but not for the NEUS. In the NEUS ozone levels are lower for the No HDV scenario in some areas and higher in others. Further, $PM_{2.5}$ levels are modestly improved from removing HDVs relative to LDVs.

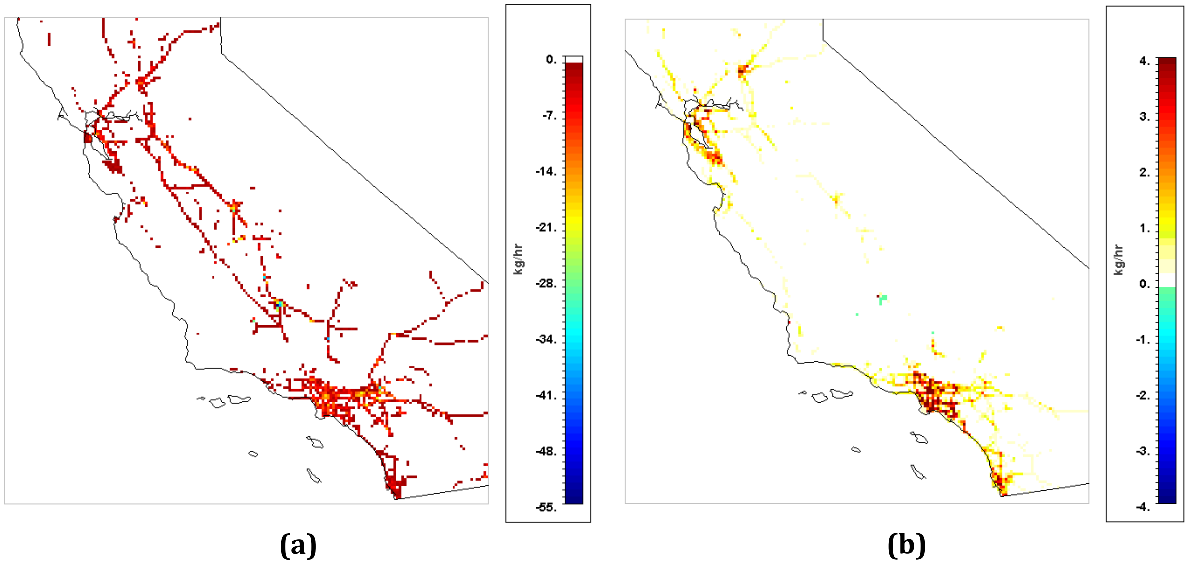


Figure 26: Impacts in CA on (a) 24-h NO_x and (b) 24-h direct PM from HDVs relative to LDVs

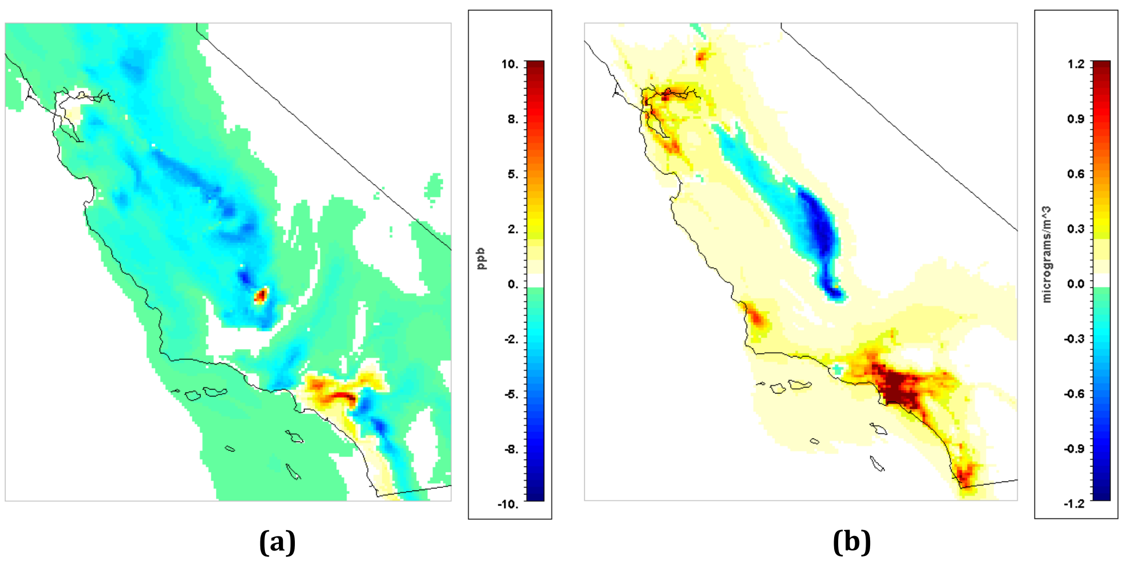


Figure 27: Impacts in CA on (a) peak ozone and (b) 24-h PM2.5 from HDVs relative to LDVs

4.3.3 Offroad Sources

Emissions from offroad sources independent of those from marine and rail include those from cargo handling equipment, large spark ignition engines (e.g., industrial equipment, forklifts, portable generators), small spark ignition engines (e.g., lawn and garden equipment), compression ignition engines (e.g., construction and agricultural vehicles), and offroad recreational vehicles (e.g., motorcycles, all-terrain vehicles). Mobile sources from agriculture include tractors, harvesters, combines, balers, swathers, sprayers, forklifts, and all-terrain vehicles. Off-road compression-ignition engines burning diesel fuels are found in a range of applications including agriculture, construction, and industrial. Common examples include tractors, excavators, dozers, scrapers, portable generators, transportation refrigeration units, irrigation pumps, welders, compressors, scrubbers, and sweepers.

As can be seen in Figure 16 and Figure 17, the removal of offroad emissions yields dramatic reductions in 24-h NO_x with peak reductions particularly important in the NEUS, i.e., -266 kg/hr, second only to Marine and Rail in terms of peak impacts. In TX the removal of offroad sources is significant but represents the lowest reduction relative to other sectors at -70 kg/hr. Spatially, emission reductions are largest in urban areas with peak reductions generally associated with the center of a major urban center, i.e., Houston and Dallas-Ft. Worth in TX and NYC in the NEUS. This is expected given the distribution of various offroad sources throughout urban regions.

The removal of emissions from offroad categorical sources has a profound impact on AQ in all study regions, particularly on ambient ozone levels. Peak levels are reduced in CA by nearly 12 ppb in some locations, significantly more than impacts from both LDV and HDVs. Similarly, in the NEUS and TX peak reductions exceed 8 and 3 ppb, respectively, and represent the second highest sub-sector specific improvement. Additionally, as marine and rail sources are combined it is likely that offroad sources represent the single most important sub-sector with regards to regional ground-level ozone.

Offroad emission impacts on secondary particulate levels vary between regions. In CA, 24-h average $PM_{2.5}$ levels are improved dramatically (-18 μg/m^3) with removal and represent the largest case improvement in-tandem with marine and rail. Contrastingly, in TX and the NEUS reduction magnitudes are minor (less than 1 μg/m^3) and represent the lowest sub-sector case perturbation.

Spatially, AQ improvements in study regions are widespread and of particular importance in urban areas as a result of large concentrations of sources that are distributed throughout urban air basins. For example, minor impacts on ozone in TX cover large portions of the state while concentrated areas of large improvements occur downwind of the State's major urban centers. Similarly, in CA dramatic improvements in ozone are visible throughout the state with the most profound improvements occurring in heavily populated areas. Additionally, the areas most affected include those with pre-existing ozone concerns. The area source nature (i.e., opposed to impacts of mobile sources localized to major roads) of the offroad sector is visible and could be important in terms of exposure levels for communities not traditionally considered for impacts (i.e., those not directly to major roadways) in addition to those with higher risk for AQ-related health effects.

4.3.3.1 Air Quality Impacts of Individual Offroad Sectors

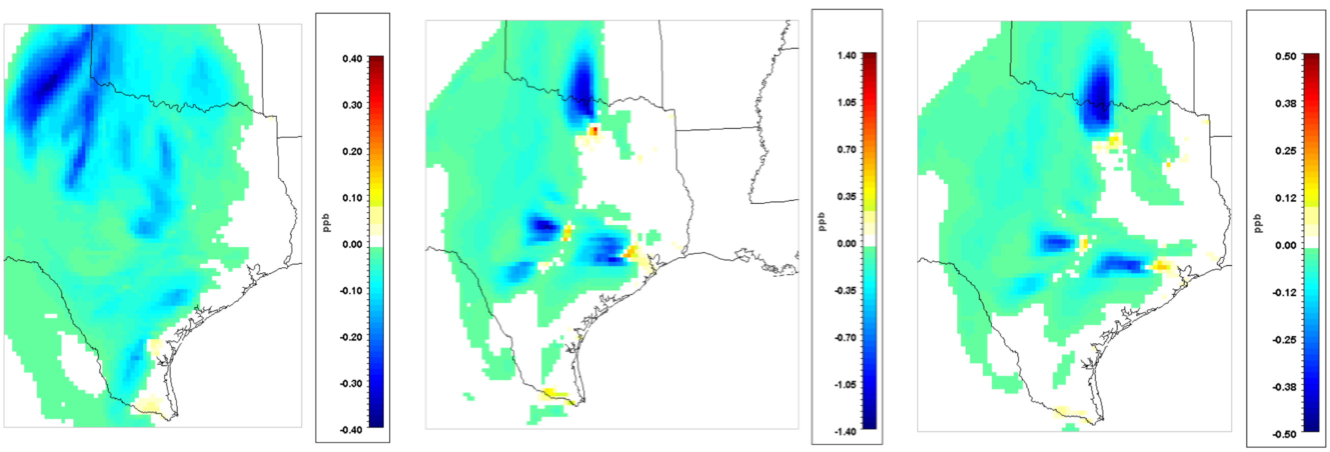
The importance of emissions from offroad sources to AQ in 2055 is demonstrated by the notable resulting reductions in ozone and $PM_{2.5}$ from removal. The diversity and breadth of technologies comprising the sector generate some uncertainty regarding the contribution from each. To elucidate how individual offroad sources impact AQ in study regions cases were developed to evaluate the contributions of emissions to ground-level ozone and $PM_{2.5}$. Emissions from offroad sources used for agriculture, construction and mining, and industrial equipment were removed individually and AQ simulations were conducted to evaluated pollutant perturbations. The three chosen sub-sectors (agriculture, construction and mining, industrial equipment) were chosen as they generally represent the largest sources of offroad energy conversion and emissions. The TX and CA study regions were selected to model AQ and assess impacts on AQ.

Table 13 displays the resulting perturbations in direct vehicle emissions for offroad sector cases in TX in 2055. As can be seen, the largest direct reductions in NO_x and PM occur in the case where construction and mining activity is removed (-52 and -7.5 kg/hr respectively). Industrial equipment removal results in peak reductions of -21 and -1 kg/hr of NO_x and PM. Agricultural equipment achieves a lesser peak impact of about -2 kg/hr NO_x and -0.3 kg/hr PM.

Table 13: Direct emission impacts from mobile offroad source categories in TX

Source Category	Δ 24-h NO_x [kg/hr]	Δ 24-h Direct PM [kg/hr]
Agricultural Equipment	-1.99	-0.33
Industrial Equipment	-21.19	-0.95
Construction/mining	-52.23	-7.492

Peak impacts on ozone and $PM_{2.5}$ from the offroad source category cases in TX are displayed in Table 14 and difference plots for ozone are shown in Figure 28. Mirroring direct emissions, the largest peak improvements occur from the removal of construction and mining mobile equipment with reductions of 1.4 ppb and 0.46 μg/m³. Impacts from the additional two cases were significantly less in terms of peak magnitude, with industrial equipment associated with a -0.48 ppb and -0.05 μg/m³. Spatially, impacts in the agriculture case occur in the Northwest and North central portions of the State. In contrast, reductions peak downwind of both the construction and mining and industrial equipment cases.



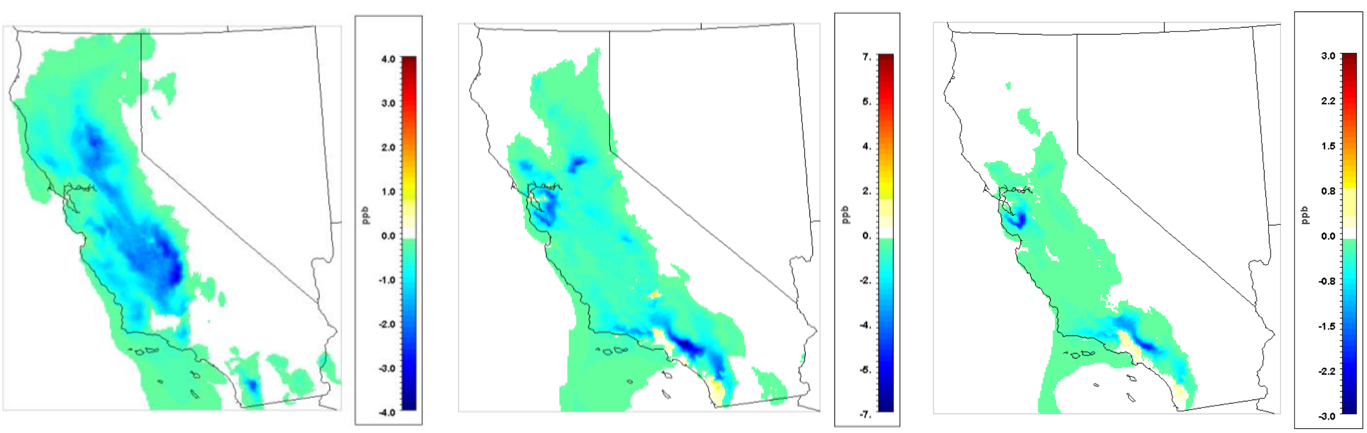
(a) **(c)** **(e)**

Figure 28: Impacts on peak ozone from mobile emission sources in (a) agriculture, (b) construction and mining, and (d) industrial activity in TX

Table 14: AQ Impacts from mobile offroad source categories in TX

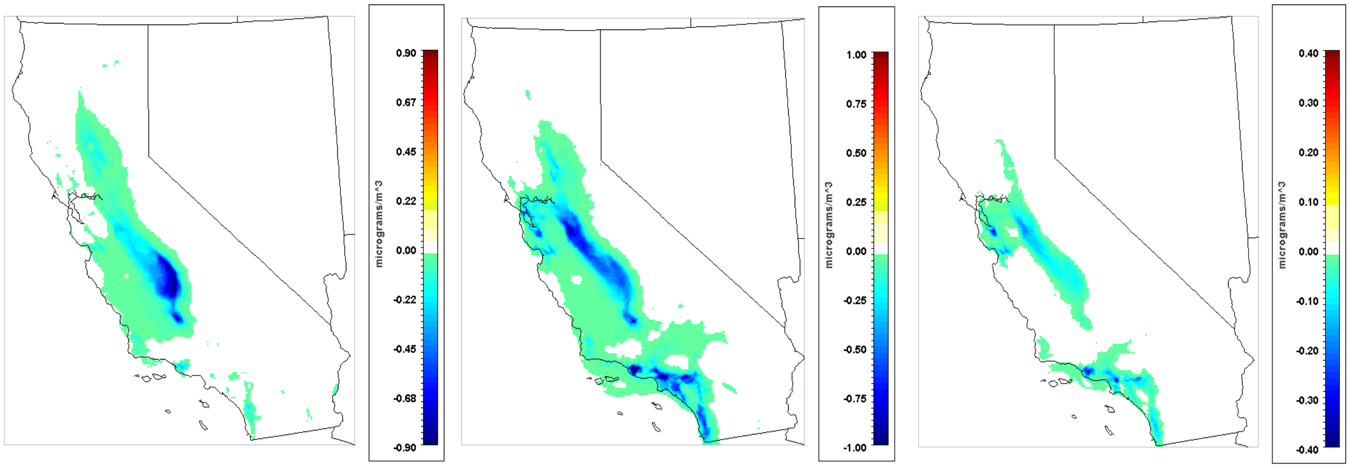
Source Category	Δ Peak Ozone [ppb]	Δ PM 24-h [μg/m³]
Agricultural Equipment	-0.37	-0.07
Construction & Mining Equipment	-1.42	-0.46
Industrial Equipment	-0.48	-0.05

Figure 29 and Figure 30 display difference plots for peak ozone and 24-h $PM_{2.5}$ contributions from mobile offroad sources. In addition, peak impacts on ozone and $PM_{2.5}$ from the offroad source category cases in CA are displayed in Table 15. As in TX, the largest peak impacts occur for the construction and mining equipment case and are most pronounced in urban areas including the SF Bay Area, SoCAB, and the greater Sacramento area. Impacts on $PM_{2.5}$ are also pronounced in the Central Valley in certain regions. The agricultural equipment case also is associated with significant impacts on ozone and $PM_{2.5}$ with peak reductions over 4 ppb and almost 1 μg/m^3. In addition, the spatial distribution of reductions is larger than the other two sub-sectors and largely covers the Central Valley. The results are intuitive as significant agricultural activity is associated with the State. Furthermore, high agriculture activity occurs in the Central Valley and AQ impacts are visible in these regions. Improvements to ground-level concentrations of air pollutants have importance due to existing AQ challenges. Mobile sources associated with industry also have an important AQ impact in CA with peak reductions higher than those for TX, although this is likely a result of differences in grid cell resolution. Improvements in ozone and $PM_{2.5}$ are notable in SoCAB and the SF Bay Area.



(a) (c) (e)

Figure 29: Impacts on peak ozone from mobile emission sources in (a) agriculture, (b) construction and mining, and (d) industrial activity in CA



(a) **(b)** **(c)**

Figure 30: Impacts on 24-h $PM_{2.5}$ from mobile emission sources in (a) agriculture, (b) construction and mining, and (d) industrial activity in CA

Table 15: AQ Impacts from mobile offroad source categories in CA

Source Category	Δ Peak Ozone [ppb]	Δ PM 24-h [μg/m³]
Agricultural Equipment	-4.63 ppb	-0.95
Construction & Mining Equipment	-7.00 ppb	-1.26
Industrial Equipment	-2.93 ppb	-0.35

4.3.4 Marine and Rail

In order to elucidate effects of marine and rail transport, a case for each region was evaluated involving the emissions absence of both sectors. While a combination of sectors in a single case can complicate the understanding of the observed impacts, the spatial boundaries of ship impacts along coasts and rail transportation extending inland assists in clarification. Further, the structure of the current U.S. cargo transportation system entails important situations where both sectors are actively co-located (e.g., major coastal shipping ports) and thus combined impacts can yield important information in addition to simplifying analysis techniques.

From solely a magnitude standpoint the removal of marine and rail emissions yields the largest ozone improvement of all transportation sub-sectors in all three regions, i.e., the highest recorded reduction in one cell from the Base Case. In part this is a function of two sectors in combination relative to only one in the other cases. However, the observed reductions are dramatic and emphasize the importance of the two sectors to AQ in 2055. Peak reductions in are particularly dramatic in CA, with some areas experiencing over a 20 ppb reduction in ambient ozone levels, an improvement of roughly 9 ppb higher than any other case. In TX and the NEUS the impacts are reduced but continue to be in excess of other sectors, with reductions of over 4 and 10 ppb, respectively. The discrepancy between other sectors is also not as large as it is for CA. Additionally, the lower value in TX is somewhat surprising given the significant port activity that occurs in the region although it should be noted that trends between sectors are equivalent to the NEUS and CA.

Impacts on $PM_{2.5}$ concentrations are similarly significant from a magnitude standpoint, although spatially the impacts are more highly localized than for ozone impacts. The marine and rail cases achieve the highest peak impacts amongst transportation sub-sectors for both CA (-18.9 $\mu g/m^3$) and TX (-4.1 $\mu g/m^3$) and the second highest for the NEUS (-1.2 $\mu g/m^3$). In particular, impacts in TX are significantly higher than other related cases and demonstrate the presence of many active shipping ports.

The distinct spatial distribution of emissions from shipping (i.e., coastal and inland waterways) and rail (i.e., established railways) sources drives resulting area of impact from an AQ perspective. Additionally, as both sources are co-located the largest impacts are observed at or down-wind of major shipping ports. In CA the largest impacts occur along the southern coast, including adjacent to the Ports of Long Beach and L.A., and the San Francisco Bay Area which includes among others the Ports of Oakland, San Francisco and Richmond. Notable ozone reductions extend downwind and encompass regions with high population density including San Francisco, SoCAB, and San Diego. Interestingly, the areas immediately adjacent to the L.A and Long Beach ports experience an increase in ozone during peak formation periods due to the titration mechanism discussed above. Impacted locations in CA also include the central valley, although to a lesser degree, and some inland locations associated with rail transport. Similar impact distributions are observed for TX and the NEUS, including impacts downwind from major shipping ports like those found in Houston-Galveston-Brazoria, NY-NJ, and Philadelphia. Reductions also extend from the Port of Pittsburgh and demonstrate the importance to AQ of shipping emissions from inland waterways in addition to coastal ocean ports.

4.3.5 Transportation Sub-sector Conclusion

The evolution of various transportation subsectors to 2055, including alterations in demand and characteristic technologies and fuels, adjusts emissions quantitatively relative to baseline levels. Transitions in the LDV fleet to more efficient and lower-emitting vehicles, and to a lesser degree the use of alternative vehicle technologies, significantly reduces the total pollutant emissions attributable to personal vehicle travel, despite a large increase in total regional demands. Similarly, HDV activity increases are offset by more efficient and cleaner fleet technologies. In contrast, emissions from other transportation sub-sectors (e.g., offroad, marine, rail) increase in the study horizon as demand growth offsets any efforts to lessen impacts. These trends are visible in Table 16 which displays peak 24-h NO_x emission reductions for sub-sector removal. For all regions the Marine and Rail Case represents the largest improvement in emissions. Similarly, the LDV case represents the lowest value in TX and CA and in the NEUS LDVs are second only slightly to HDVs. For all regions the offroad case is associated with important reductions, with the NEUS experiencing a particularly dramatic peak improvement. In the offroad sector emissions from construction and mining mobile sources were shown to be a significant contributor to regional AQ burdens in TX and CA. In addition, mobile agriculture-related sources had an important impact in CA.

Table 16: Δ 24-hr NO_x [kg/hr] impacts from removal of transportation sub-sector emissions

Region	LDV	HDV	Offroad	Marine & Rail
TX	-89.7	-155.6	-70.6	-679.9
NEUS	-84.8	-82.5	-266.0	-486.7
CA	-13.1	-59.9	-45.5	-823.5

As would be expected, all regions experience similar trends with regards to secondary pollutant species. Table 17 reports peak impacts on ozone and 24-h $PM_{2.5}$ values experienced in each region when the associated sub-sector emissions are removed. Due to significantly reduced precursor emissions (e.g., fleet-wide NO_x and VOCs are reduced by more than 80% and 75%); AQ impacts of LDVs decrease relative to the baseline year in terms of both ozone and $PM_{2.5}$. Thus, removal of LDVs in 2055 is generally associated with the lowest improvements in AQ, (in the NEUS impacts on ozone are slightly better than HDVs). The largest regional improvements in ozone are associated with marine and rail emissions, with the offroad or HDV sub-sector second depending on region, i.e., in TX Offroad is second while in CA HDVs are the next highest. Improvement in peak 24-h $PM_{2.5}$ varies and is highly region-dependent. In TX, the Marine and Rail Case has the largest improvement, followed in sequence of highest to lowest by the LDV, Offroad, and HDV Cases. In CA, the Marine and Rail Case and Offroad cases are equally associated with the highest reduction, followed by LDVs and HDVs. In the NEUS the deepest peak reduction occurs from HDVs, followed by LDVs, Marine and Rail, and Offroad.

Increases in ozone occur in some scenarios as a result of reduced ozone scavenging from reductions in NO_x emissions (e.g., ship scenarios in all regions) and similar effects have been reported in the literature for both transportation [299] and other sectors[300]. Effects most often occur in VOC-limited urban areas with high levels of anthropogenic NO_x (e.g., SoCAB, NYC) and are not generally considered a deleterious effect as reductions often occur following transport in key locations during peak formation periods. An example includes the ships case for CA in which NO_x reductions from the Ports of L.A. and Long Beach yield improvements in ozone in Riverside and San Bernardino coinciding with peak afternoon ozone concentrations despite initial increases over Los Angeles. As the highest ground-level concentrations of ozone in SOCAB occur in those locations the results should be considered a benefit to the region rather than a detriment. This phenomenon further demonstrates the importance of utilizing atmospheric modeling to assess AQ impacts as solely quantifying emissions perturbations would not generally facilitate such insights.

Table 17: Peak impacts on ozone and 24-h $PM_{2.5}$ from removal of transportation sub-sector emissions

Region	LDV		HDV		Marine and Rail		Offroad	
	Δ Ozone [ppb]	Δ 24-h $PM_{2.5}$ [μg/m³]	Δ Ozone [ppb]	Δ 24-h $PM_{2.5}$ [μg/m³]	Δ Ozone [ppb]	Δ 24-h $PM_{2.5}$ [μg/m³]	Δ Ozone [ppb]	Δ 24-h $PM_{2.5}$ [μg/m³]
TX	-2.4	-1.2	-2.6	-0.4	-4.3	-4.1	-3.3	-0.6
NEUS	-5.8	-1.5	-5.6	-2.5	-10.4	-1.2	-8.2	-0.8
CA	-4.0	-6.5	-12.9	-1.9	-20.6	-18.9	-11.8	-18.9

As noted, impacts on ozone and $PM_{2.5}$ from transportation emissions in the NEUS include several important regions largely associated with major metropolitan regions and ports. As would be expected from the large, concentrated presence of sources, ozone and $PM_{2.5}$ impacts are most notable downwind of the New York City Metropolitan region (NYC) with improvements over much of Massachusetts, Connecticut, and Rhode Island.

In the NEUS removing emissions associated with off-road sources has a high impact on ozone and $PM_{2.5}$ in 2055. Spatially, peak impacts are centered in NYC and New Jersey and extend over much of the States located upwind. Additionally, notable impacts occur along the Lake Ontario coastline and extending from Pittsburgh. HDV and LDV emissions have comparable impacts for both studied pollutants with areas of peak impact upwind of NYC. Removing ship emissions results in areas of significant improvement along the New Jersey and New York coastlines from ports associated with the region, including the Ports of NY/NJ, Philadelphia, and others. Additionally, shipping activity along inland waterways yields improvements extending from the Port of Pittsburgh and along the coast of Lake Ontario. Impacts of emissions from rail sources are most notable in northern New York State and central Pennsylvania, coinciding with major railway infrastructure, and achieve moderate impacts on ozone and $PM_{2.5}$.

Important areas of CA with respect to regional AQ include the South Coast Air Basin (SoCAB) encompassing much of Los Angeles, Orange County, Riverside, and San Bernardino Counties, the Bay Area, the San Joaquin Valley (Central Valley) and Sacramento area as these regions are currently affected by harmful levels of airborne pollution; including ozone and $PM_{2.5}$ [51].

Removal of LDV emissions in CA in 2055 results in reductions that reflect the distribution of vehicles, with reductions over much of the state and most pronounced downwind of urban regions including SoCAB. HDV emissions removal has a significantly higher impact than LDV emissions and achieves important reductions throughout much of CA. Similarly, off-road emissions have substantial effects on both ozone and $PM_{2.5}$ over large regions of the State. Notable impacts that occur are associated with major urban locations including SoCAB, the Bay Area, Central Valley, Sacramento and San Diego. With similarity to HDV, $PM_{2.5}$ impacts are important in the Central Valley, although removing off-road sources achieves larger concentration reductions in SoCAB. The removal of ship emissions results in substantial reductions in both ground-level ozone and $PM_{2.5}$. Areas of improvement are notable in SoCAB and result from heavy ship traffic at the Ports of L.A. and Long Beach. Additional impacts occur as a result of ship activity at the Ports of Richmond, Oakland, San Francisco and San Diego. Effects are also visible in the northern section of the Central Valley as a result of the emission reductions from port-activity in the Bay Area. Removing rail emission achieves moderate reductions in ozone and $PM_{2.5}$ in 2055 in CA.

In the TX region notable areas of impact on ozone and $PM_{2.5}$ that occur as a result of removing transportation source emissions comprise regions upwind of major metropolitan areas, including Houston, Dallas-Ft. Worth, Austin, and San Antonio. In particular, concentration improvements in the Greater Houston (Houston-Galveston-Brazoria) and Dallas-Ft. Worth regions are beneficial as both currently experience difficulty meeting 2008 Federal ozone standards[51].

Removing LDV emissions achieves a minor impact on ozone and $PM_{2.5}$ relative to other transportation technologies, with impacts notable downwind of major metropolitan areas. HDV emissions have a moderately larger impact than LDVs and off-road sources for ground-level ozone concentrations, particularly upwind of the Dallas-Ft. Worth metropolitan area. Off-road sources have a slightly lower impact on ozone than HDVs but have a higher impact on 24-h $PM_{2.5}$ concentrations. Spatially, off-road emissions result in peak impacts downwind of major metropolitan areas. Similar to the other regions of study, the AQ impacts of ship emissions have high importance in the TX in 2055. When removed, the highest peak improvements attributable to any single transportation source in ozone and $PM_{2.5}$ are evident starting at sites of major port locations along the Gulf Coast (e.g., the Ports of Houston/Texas City/Galveston, Beaumont/Port Arthur/Orange, and Corpus Christi) and extending through the region. Peak impacts on $PM_{2.5}$ from ship emissions are particularly notable as being an order of magnitude higher than any other transportation sub-sector. Rail-related emissions have minor impacts on $PM_{2.5}$ and ozone in 2055 in TX.

LDV emission impacts are moderate in all regions for both ozone and $PM_{2.5}$ relative to other sub-sectors. Similar effects in regards to future ozone impacts of LDV emissions have been reported for CA [299]. This is not surprising given the major reduction in many criteria pollutants fleet-wide in 2055 from current levels, e.g., baseline NO_x emissions are over 80% lower for all three regions despite significant growth in total demand for LDV travel. Reductions are a product of various efforts to improve LDV performance and reduce emissions and reflect a current regulatory focus designed to amplify such areas at both the Federal and State levels. Additionally, the baseline scenario assumes a slight increase in lower emitting LDV technologies and fuels, including various electric vehicles. Thus, the 2055 LDV fleet is responsible for less influence on ozone and $PM_{2.5}$ relative to current day impacts. The results further emphasize the importance of addressing emissions from non-LDVs, e.g., ships, off-road, HDVs, in efforts to improve regional AQ through emission reduction plans.

The moderate AQ impacts attributable to LDVs in 2055 relative to other transportation sources should be evaluated in the context that; (1) improvements in ozone and $PM_{2.5}$ occur in populated urban regions and thus have human health implications, and (2) LDVs will continue to be an important source of domestic GHG emissions despite reducing pollutant emission rates. Additionally, the considerable effects on ozone and $PM_{2.5}$ of producing and distributing motor gasoline should be considered with those directly from vehicles in totality of impact. Thus, LDVs will continue to represent an important opportunity for alternative low-emitting technologies and fuels in coming decades. However, it may be more effective to pursue mitigation strategies on the basis of GHG reductions.

The magnitude and spatial dimension of both primary and secondary pollutant impacts occurring from ship emissions highlights water-borne vessels as a primary target for future mitigation strategies seeking to improve AQ in the study regions. Additionally, contributions to ground-level ozone and $PM_{2.5}$ warrant attention for the development and deployment of cleaner alternative technologies and fuels for off-road vehicles. In particular, locations of major shipping ports emerge as perhaps the dominant contributor of transportation-related regional air pollution in 2055 when considering the convergence of emission sources comprising goods movement strategies including ships, off-road, HDV, and rail. A current understanding of the harmful AQ impacts of port activity exists and programs and policies are in place seeking emission reductions from the aforementioned technologies (e.g., CA's Goods Movement Emission Reduction Plan[301]). However, expected growth in demand for global shipping in tandem with reduced emissions from other sectors including LDVs will increase the prominence of meeting or exceeding current emission reduction goals. Further, operational and other constraints increase the difficulty of deploying alternative strategies for some goods movement technologies. Thus, these results support the near-term development of research, development, and deployment plans for advanced, Port-related technologies.

4.3.6 AQ Impacts of Goods Movement Sector Emissions

It has been noted that air pollution impacts from the goods movement sector represent a major public health concern and contributions to regional AQ problems could increase in coming decades[302]. Pollutant emissions from goods movement activity have been shown to be a major contributor to local AQ problems in the regions of study. For example, it is estimated that 70% of the diesel PM pollution in California in 2001 occurred from activities in the goods movement sector [303]. Similarly, a substantial portion of the NO_x and SO_x emitted in SoCAB is associated with goods movement activity[304]. The current importance of minimizing goods movement contributions to GHG and regional AQ will only be enhanced in coming years; driven by, among other things, increased demand for transport of goods and reduced emissions from other sources resulting from regulatory constraints (e.g., LDVs, power plants). For example, container traffic at California ports has increased dramatically in recent decades and it is projected that activity at California ports will grow up to 250% by 2020[119, 303].

The current technologies and fuels utilized in the goods movement sector are associated with the production and release of exhaust with serious human health impacts, including increased risk for cancer, premature mortality, and other disease burdens [305, 306]. Direct emission of diesel PM have been linked to increased health risks for communities surrounding ports and can reach communities substantially downwind during regional transport events[307, 308]. Additionally, the goods movement sector has been explicitly linked to deleterious human health effects in study regions, including exacerbation of respiratory disease in children[304].

Of particular importance with regards to the regional AQ impacts of the goods movement sector are locations of major ports, which encompass activity from all relevant mobile sources. Further, emissions of NO_x, reactive organic gasses (ROG), and SO_x at ports are substantial and contribute to regional air pollution formation, including elevated levels of ozone and $PM_{2.5}$ [309, 310]. In the absence of mitigation port-related emissions are projected to increase; mirroring growth in demand. In 2023 activities from the Ports of Los Angeles and Long Beach are the single largest source of emissions in Southern California, projected to account for 60% of SO_x, 27% of NO_x, and 6% of $PM_{2.5}$[311]. Indeed, emissions of SO_x from ship traffic are the only regulated pollutant projected to experience total in-basin emissions growth to 2020. Emphasizing the importance in meeting regional AQ regulatory standards, mitigation of port-related emissions is a focus of regulatory efforts in study regions. For example, California has targeted port emission reductions with high priority in both the 2007 State Implementation Plan (SIP) and the Goods Movement Emission Reduction Plan (GMERP).

At present, the bulk of goods movement technologies operate via combustion of distillate and/or residual fuels by means of compression ignition engines; including on-road heavy-duty trucks, locomotives, marine vessels, and cargo handling equipment used to shift containerized and bulk cargo, i.e., yard trucks, side-picks, rubber tire gantry cranes, and forklifts. Ship emissions include main and auxiliary engine emissions from ocean-going vessels, both at-berth and over-water, as well as emissions from harbor craft such as tugboats, fishing vessels and passenger ferries. Further, various industrial sources, including petrochemical refinery complexes, are often located near sites of significant goods movement activity, i.e., major shipping ports, and contribute significant point source and area emissions.

In terms of sources, ships generally represent the greatest challenge with regards to pollutant reduction goals. Ocean-going vessels that transport cargo in and out of ports have little to no emissions control and operate on high pollutant (NO_x, SO_x, PM) emitting distillate or residual fuels. SO_x emissions are particularly high as marine fuels contain high levels of sulfur (2.7%, world average) relative to on-road diesel fuels (.0015%, CA limit)[309]. Emissions from ocean-going vessels include those occurring during transiting, maneuvering and hoteling (i.e., auxiliary engine operation to provide necessary power while at berth) and represent the largest emissions source at ports [312]. Emissions associated with hoteling are of particular concern, as they make up a large fraction of both ocean-vessel and total port emissions and various strategies are being examined to displace the need for auxiliary engine operation at-berth.

Mitigation strategies for reducing ship emissions include the deployment of new, cleaner engines and fuels, add-on emission controls, and operational changes. Examples of such strategies include speed reductions and switching to low sulfur fuels. Further, the use of shore-based electrical power while in port (cold ironing) can offset substantial emissions by reducing hoteling. HDV strategies include efficiency improvements via acceleration of fleet turn-over rates, eliminating older, higher-emitting vehicles, deploying retrofit controls, and a transitions to alternative propulsion systems and fuels, including biodiesel, hydrogen, and CNG. Potential locomotive strategies include upgrading current engines and utilization of alternative propulsion systems and fuels, including hydrogen fuel cell /gas turbine hybrid configurations (FCGT). Off-road vehicle strategies are numerous and reflect the widely varying technologies comprising port-related fleets, and include upgrading to newer, lower-emitting models and the use of alternative propulsion systems and fuels, including electric battery, hydrogen fuel cell, and CNG applications.

4.3.6.1 Methodology

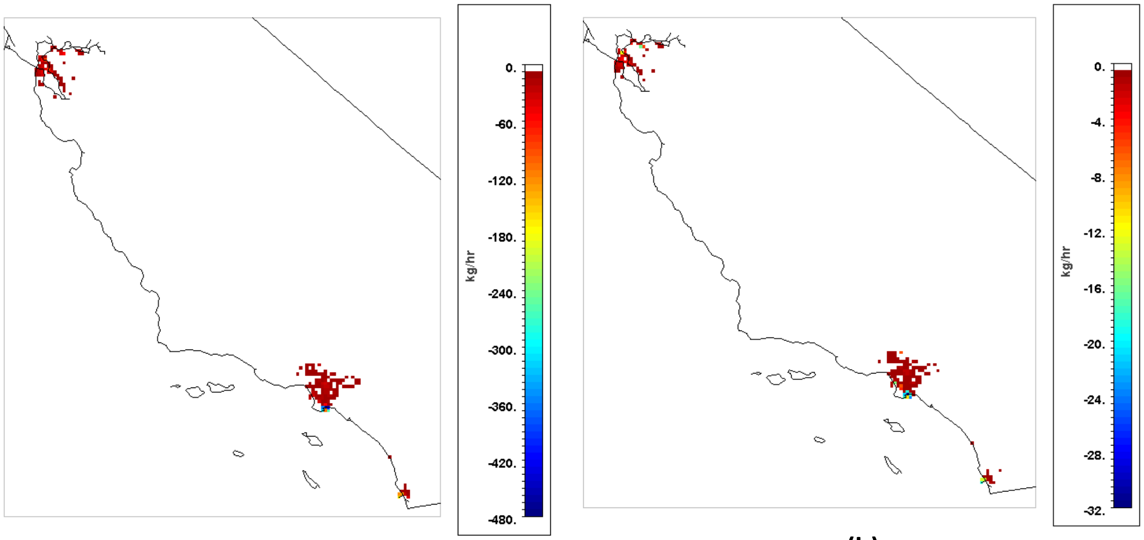
The substantial goods movement activity in 2055 in the study regions, including the locations of several major shipping ports, merits investigation of the impacts on AQ of meeting goods movement demands. Scenarios were developed to evaluate contributions to regional ozone and $PM_{2.5}$ levels from arising from deployment of GHG mitigation strategies associated with goods movement sector activities in study regions. Mobile sources evaluated in the Goods Movement sector include ships, trains, heavy duty trucks, and off-road equipment which largely utilize petroleum fuels. Thus, reductions in necessary fuel were accounted for by decreasing emissions from petroleum fuel production pathways including refineries.

Criteria pollutant emission perturbations were applied at specific activity sector levels. SCC codes representing technologies responsible for port-related mobile emissions were identified, including for various off-road vehicles, vessels, locomotives, and heavy-duty vehicles. Appropriate SCC codes were identified for technologies representing major point- and area-source emissions, including petroleum refineries co-located near major shipping ports. Multiplication factors were applied to corresponding SCC codes to adjust pollutant emissions to reflect impacts of mitigation strategies directed at reducing port-related GHG emissions. Due to the heavy confluence of sources and particular importance to regional AQ problems, scenarios included the manipulation of emissions in counties supporting major ports only.

In the 50% Goods Movement (GM 50) scenario it is assumed that all major mobile source technologies are converted to emissions-free sources of power. For example, it is assumed that all off-road equipment is replaced with technologies that provide services without producing air emissions, e.g., fuel cell operated fork lifts. Additionally, emissions from HDVs, ships, and locomotives are removed. Similarly, the 25% (GM 25) and GM 50 scenarios incorporate emission reductions equivalent to a quarter or half of the 2055 Base emissions.

4.3.6.2 Emission Impacts

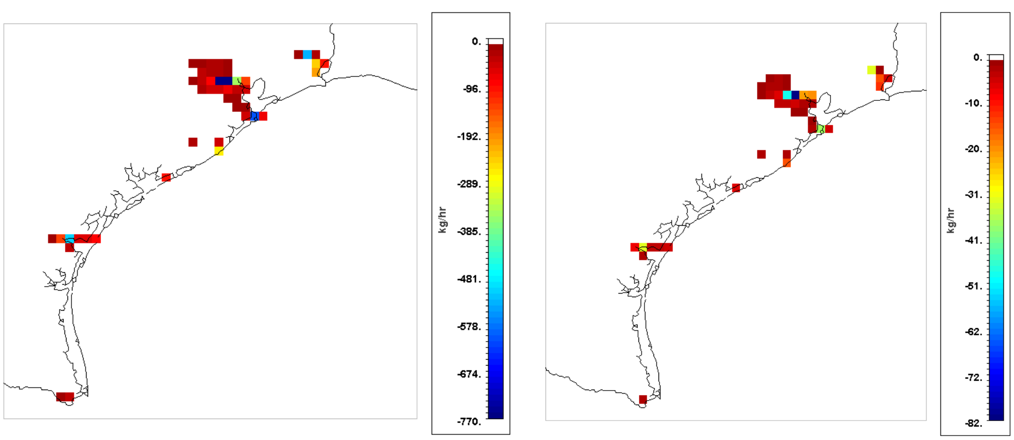
The impacts on emissions of removing 50% of goods movement sector emissions from counties in CA supporting major ports is shown in Figure 31. Reductions in NO_x and PM peak at around 480 kg/hr and 32kg/hr. Additionally, impacts on VOCs include significant reductions in similar locations (Figure 33). Spatially, locations of particular impact include the Long Beach/L.A. Port system.



(a) (b)

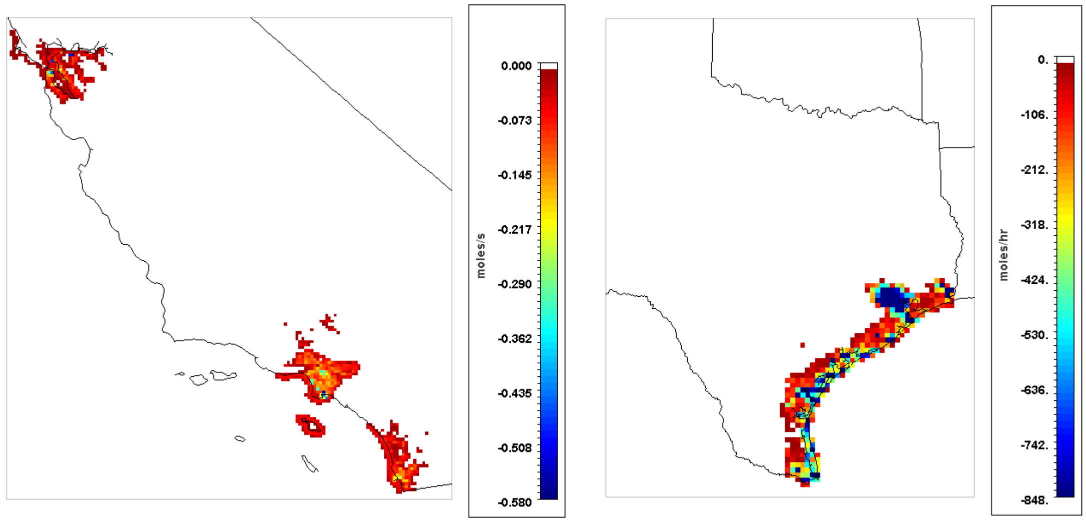
Figure 31: Impacts on 24-h (a) NO_x and (b) PM emissions for the GM 50 Case in CA

In TX the GM 50 Case achieves maximum reductions of 771 and 82 kg/hr NO_x and PM and 2.4 moles/second in VOC emissions. The coastline of TX supports significant goods movement sector activity and the impacts presented in Figure 32 and Figure 33 show reductions throughout the region. As can also be seen, the most significant reductions are associated with the Houston Ship Channel.



(a) (b)

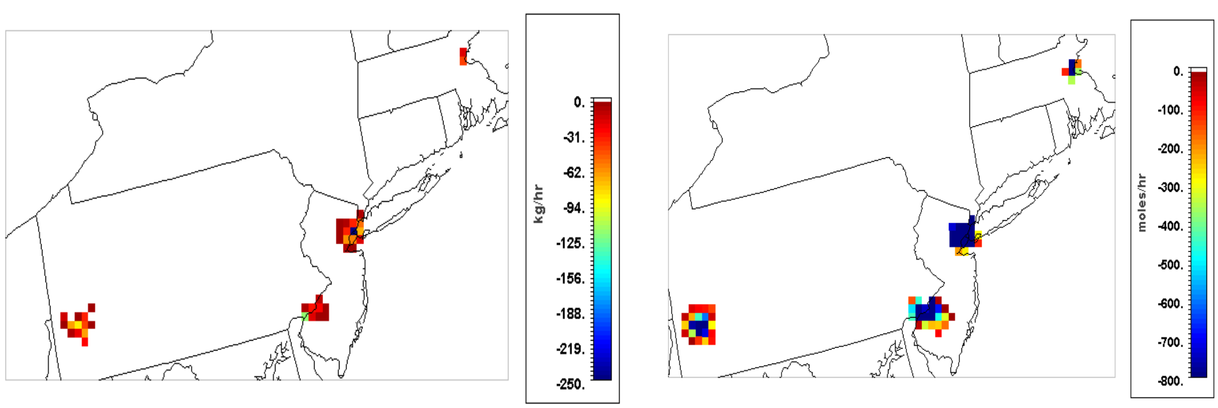
Figure 32: Impacts on 24-h (a) NOx and (b) PM emissions for the GM 50 Case in TX



(a) (b)

Figure 33: Impacts on 24-h VOC Emissions in (a) CA and (b) TX for the GM 50 Case

The resulting reductions in NO_x and VOC emissions from the GM 25 Case in the NEUS are displayed in Figure 34. As can be seen, large reductions are visible for the Philadelphia and NY/NJ sea ports as well as for the Boston Harbor. In addition, a large amount of goods movement along the Ohio River system necessitates adjustment of emissions for the Pittsburgh area.

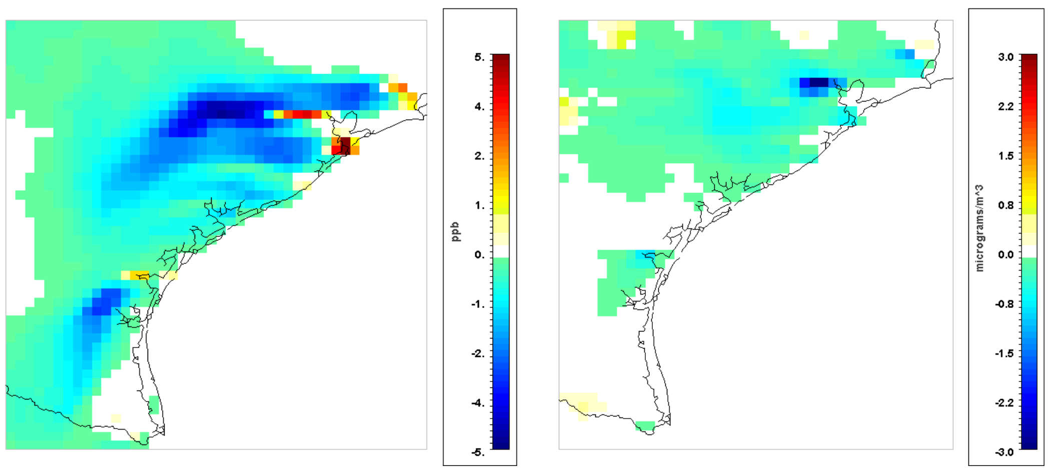


(a) (b)

Figure 34: Impacts on 24-h (a) NO_x and (b) PM emissions for the GM 25 Case in NEUS

4.3.6.3 Air Quality Impacts

In TX the GM 50 Case results in peak ozone reductions up to 4.7 ppb in some locations, with areas downwind of Greater Houston experiencing the largest impact. Ambient concentrations of $PM_{2.5}$ are also significantly impacted, with reductions from the Base Case exceeding 3.5 μg/m^3 as shown in Figure 35.



(a) (b)

Figure 35: Difference in (a) ozone and (b) $PM_{2.5}$ for the GM 50 Case in TX

In the NEUS the GM 25 Case improves ozone levels upwind of the NY/NJ and Pittsburgh areas, and to a lesser degree upwind of Philadelphia as shown in Figure 36. Surprisingly, the impacts on $PM_{2.5}$ include increases over much of the study region, with the exception of Massachusetts including Boston. Increases are most substantial for the NY/NJ region and may be related to impacts of reducing NO_x and SO_x.

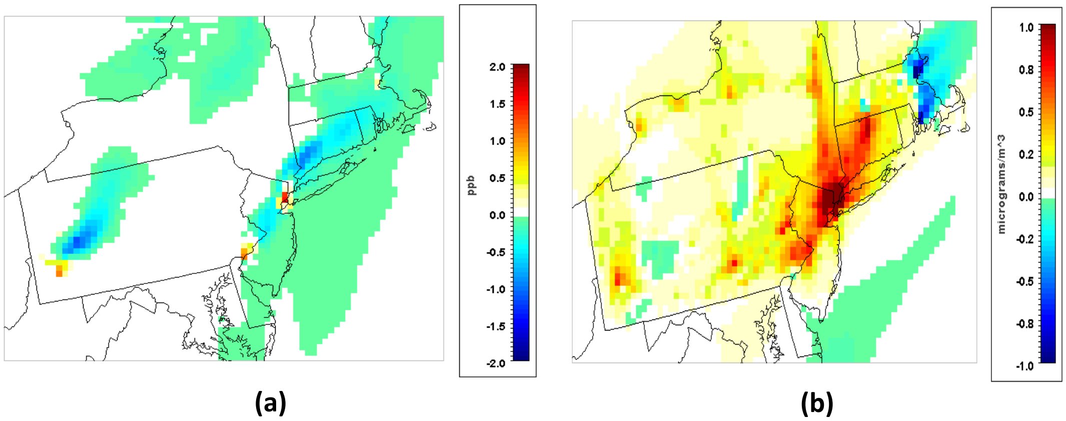


Figure 36: **Difference in (a) ozone and (b) $PM_{2.5}$ for the GM 25 Case in the NEUS**

In CA the GM 25 Case is associated with peak reductions in ozone up to 6.5 ppb as shown in Figure 37. In the GM 50 Case reductions reach 8.9 ppb. Impacts on 24-h $PM_{2.5}$ in the Ports 25 and Port 50 Case include reductions exceeding 10 and 20 μg/m^3, respectively, as shown in Figure 38.

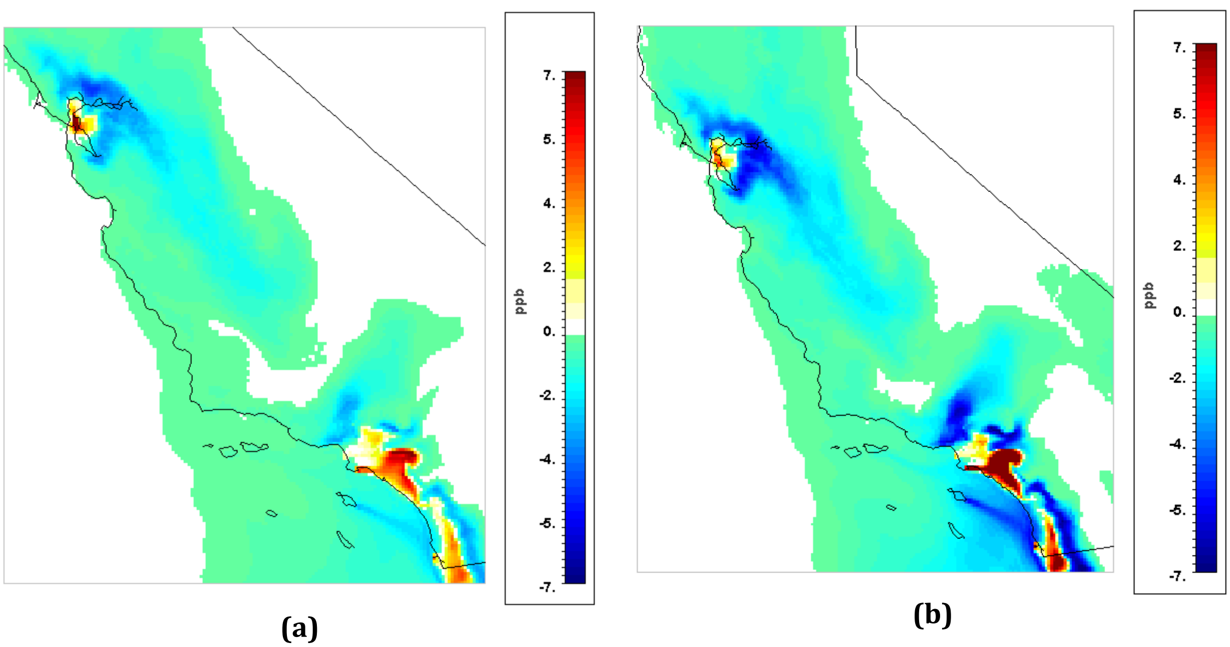
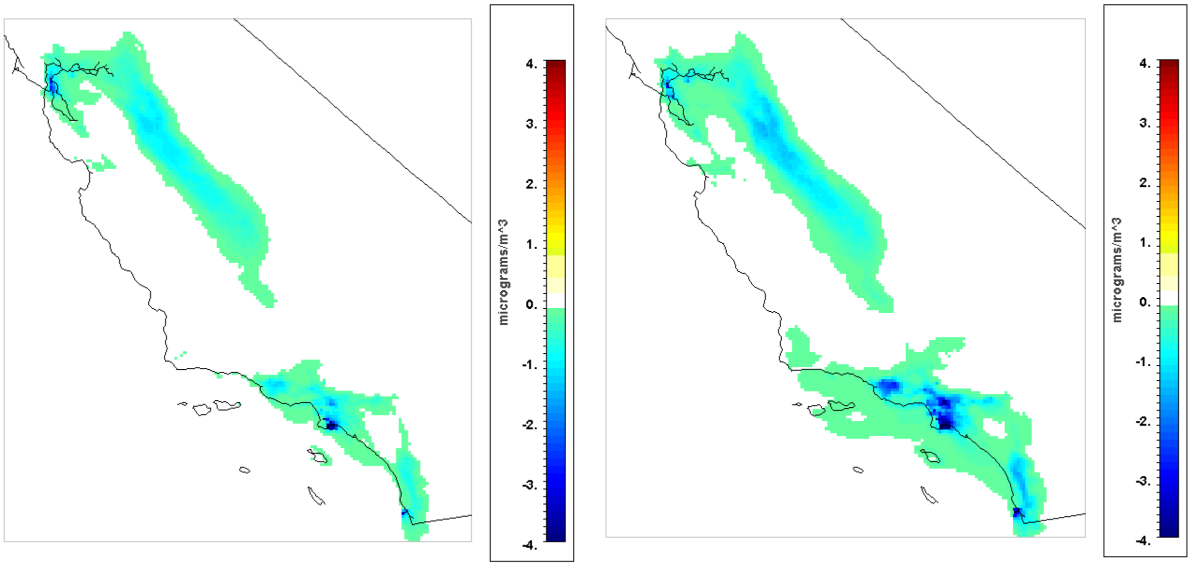


Figure 37: **Impacts on CA ground-level ozone in the (a) GM 25 and (b) GM 50 Cases**



(a) (b)

Figure 38: Impacts on CA $PM_{2.5}$ in the (a) GM 25 and (b) GM 50 Cases

Removing emissions associated with sources contained in the Goods Movement Sector in areas of study regions supporting substantial sector activity results in significant improvements in AQ. Even a 25% reduction in activity from ships, rail, HDVs, and off-road sources corresponds to notable improvements in ozone and $PM_{2.5}$. While it should be noted that factors were applied at the county level and thus not entirely attributable solely to port activity, the impacts of major ports are clearly visible in the spatial patterns of improvement in secondary pollutant species.

In particular, emissions from ships have a major impact on ozone and $PM_{2.5}$ concentrations and should be considered for future AQ improvement strategies. Strategies to address both GHG and pollutant emissions from large ocean going vessels were identified and, if deployed at maximum levels, can achieve important GHG and AQ co-benefits in all study regions.

Finally, areas of improvement tend to be associated with major urban areas with correspondingly high importance to human health. For example, the impacts of port emissions on the health of citizens in communities surrounding major shipping ports are a concern in many regions of the U.S., including those under study in this work. Reductions in emissions from GHG mitigation strategies yield AQ benefits that could assist in improving heavily-impacted communities, i.e., those surrounding Long Beach port activity

4.4 IMPACTS OF PETROLEUM FUEL INFRASTRUCTURE

The current reliance of the transportation sector on petroleum fuels requires the existence of an extensive petroleum fuel production and distribution system in the U.S., including in the regions of study for this project. Industrial process plants utilized in the processing and refinement of crude oil feedstock are generally labeled petroleum refineries. Refineries range in complexity, and use a myriad of processes (e.g., distillation, reforming, hydrocracking, coking, blending) to produce an assortment of products from petroleum. Refineries receive petroleum and petroleum products required for regional fuel production from terminal facilities; and consequently finished products are transported to distribution centers, via tanker, barge, pipeline, truck and/or rail. As the majority of U.S. imported petroleum arrives by tanker, a significant portion of terminals are marine and thus located at or near major U.S. ports and refineries are often co-located near such terminals.

The large, sprawling industrial processes typified by refinery complexes can result in a significant and diverse range of pollutant emissions over a large spatial area. Further, refining is generally a large scale, high capacity process and many facilities are operated continuously; resulting in the steady generation of large amounts of pollutant emissions for extended periods (e.g., months to years) and differing from the defined temporal emissions patterns in other sectors. It follows then that emissions associated with the production, storage, transport, and distribution of conventional petroleum fuels are known to be major contributors to regional AQ problems. For example, industrial activities around the Greater Houston area, which includes some of largest concentrations of petrochemical facilities in the U.S., is known to contribute significantly to regional non-compliance with Federal AQ standards[17]. In particular, it has been shown that petroleum refining facilities co-emit large quantities of NO_x and hydrocarbons (i.e., ethane, propene) which contribute significantly to the rapid and efficient formation of high concentrations of ozone [313].

4.4.1.1 Base Case Petroleum Fuel Production

Refinery products include gasoline, aviation fuel, distillate fuel, and residual fuels; however other intermediates are produced including hydrogen. The relative output of products can vary by facility and season; about half of current CA refinery output is gasoline, with aviation fuel, distillate fuel, and residual fuel representing 12%, 13%, and 9% respectively[314]. Base Case product output percentages in 2055 differ by region (Table 18) with implications for GHG and AQ mitigation co-benefits. For example, distillate fuel makes up a larger portion of products in TX and could be responsible for a greater portion of emissions relative to gasoline. Therefore, alternative fuel usage in the TX HDV sector to offset GHG emissions could potentially have a larger refinery emissions reduction co-benefit than in other study regions.

Table 18: Relative product output for Base Case petroleum refineries

Region	Gasoline [%]	Diesel [%]	Jet Fuel [%]	LPG [%]	Other [%]
NEUS	58	29	6	3	3
TX	38	32	16	11	3
CA	52	27	18	2	3

Base Case refinery-related emissions are substantial and make up a significant portion of total pollutant emissions in regions that support refinery activity (i.e., TX, CA, and NEUS R2). For example, Figure 39 demonstrates that refinery activity is responsible for roughly 1.5 and 2.5% of total 2055 NO_x and VOC emissions, respectively, in the TX region. In addition, refineries contribute up to 3.3% and 3.9% of regional SO_2 and PM_{10}.

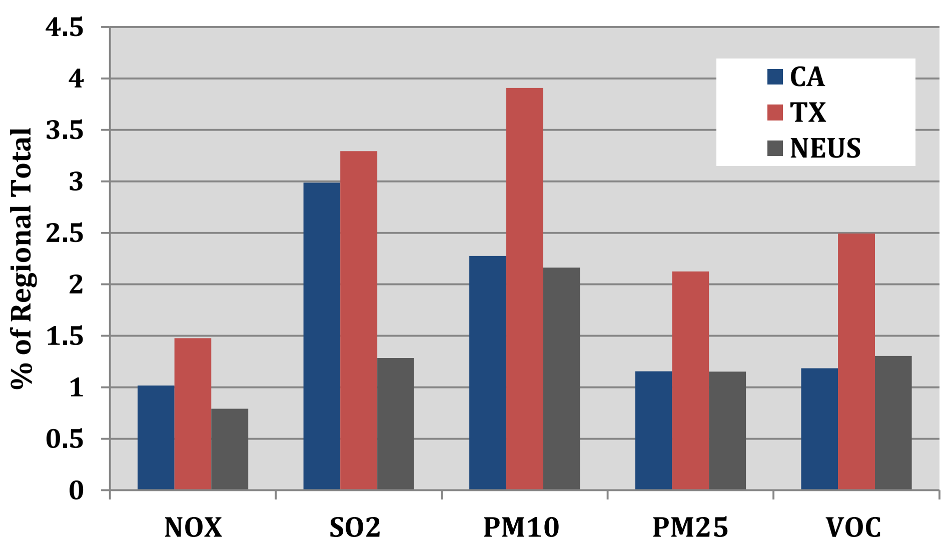


Figure 39: 2055 Base Case regional pollutant emissions from petroleum refinery activity

Despite the important contribution to total emissions, net emissions from refinery related activities experience a slight reduction in all regions from 2005 to 2055. Figure 40 shows emissions of NO_x from regional refinery activity declining modestly through the study horizon. In addition, the importance of refinery activity in the TX study region with regards to emissions and potentially AQ is evident. Reductions in emissions are driven by substantial decreases in gasoline demand due to increased LDV efficiencies and alternative fuel use.

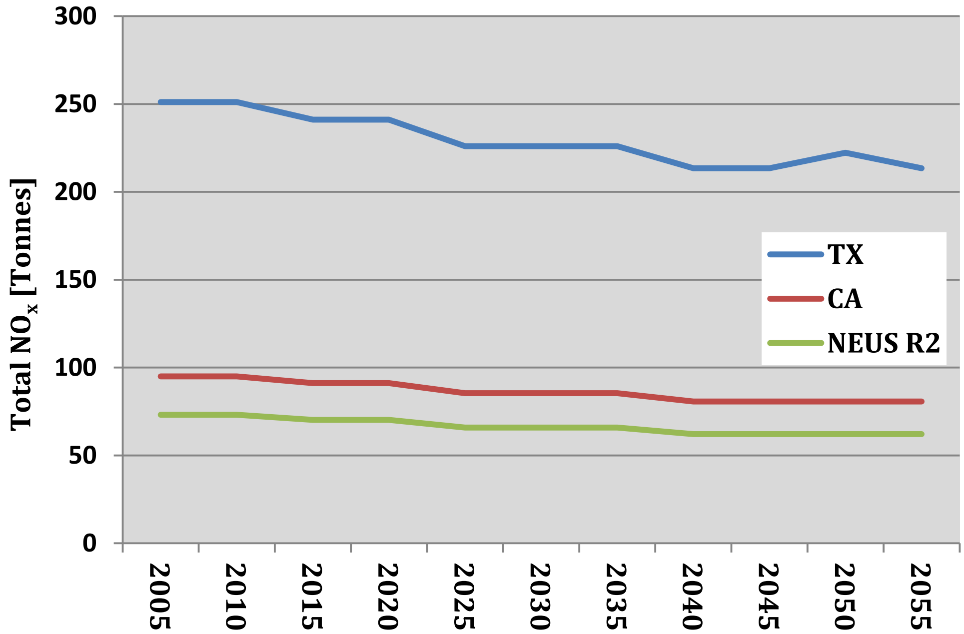
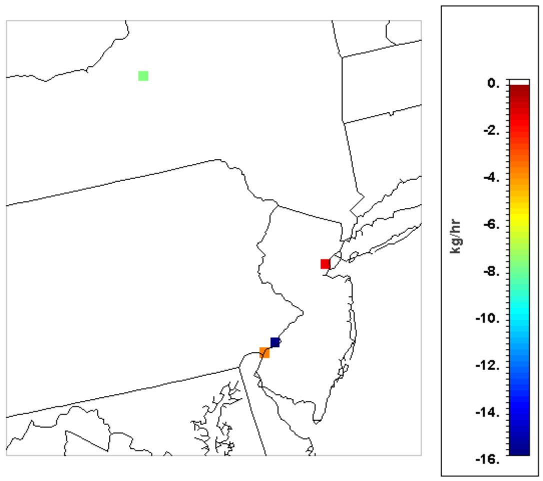


Figure 40: Base Case total NOx emissions from petroleum fuel refining activity

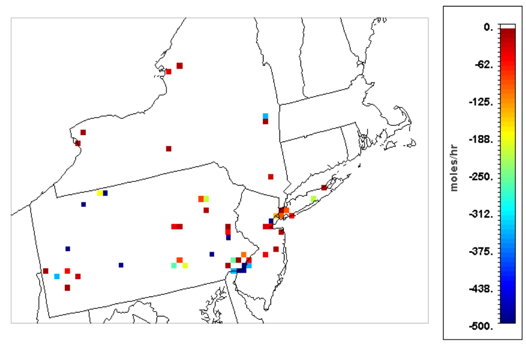
4.4.2 Impacts of Petroleum Fuel Production and Distribution on Air Quality

To elucidate the impacts of producing and distributing petroleum transportation fuels (e.g., motor gasoline, distillate fuels) scenarios were developed and evaluated involving the removal of petroleum fuel infrastructure (PFI) related emissions. Adjusted sources include all point and area source emissions occurring at refinery complexes, including industrial process (e.g., fuel combustion for heat) and evaporative emissions from storage tanks, etc. Moreover, emissions from activities related to the transport and storage of vehicle fuels to dispensing locations are removed, including emissions associated with vehicle refueling activity. All additional emissions, including those directly from vehicles, are left unchanged. Thus, the PFI Case solely demonstrates the contribution to regional AQ burdens of producing, storing, transporting, and distrusting petroleum transportation fuels.

Removing petroleum refinery activity and associated emissions in the NEUS yields improvements in 24-h NO_x of over 157 kg/hr. As can be seen in Figure 41 (a), impacts are largely directed to sites of major refineries, e.g., near Philadelphia, PA and Camden, N.J. Similar impacts are seen for direct P.M. with reductions exceeding 53 kg/hr coinciding with refinery locations. In contrast, impacts on VOC emissions are more widespread as they include fueling station impacts; although the largest impacts (594 moles/hr) are co-located with refineries. Displayed in Figure 41 (b), additional reductions occur throughout the study region, likely as a result of decreases in VOC releases from fuel storage tanks, including those that occur during vehicle refueling.



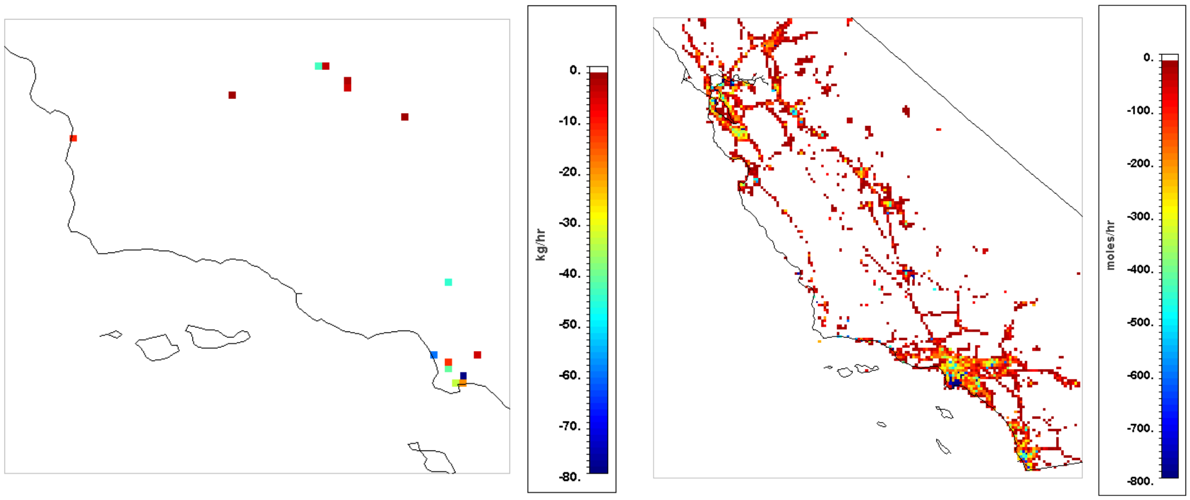
(a)



(b)

Figure 41: Impacts on 24-h average (a) NO_x and (b) VOC emissions from removing petroleum fuel production and storage in the NEUS

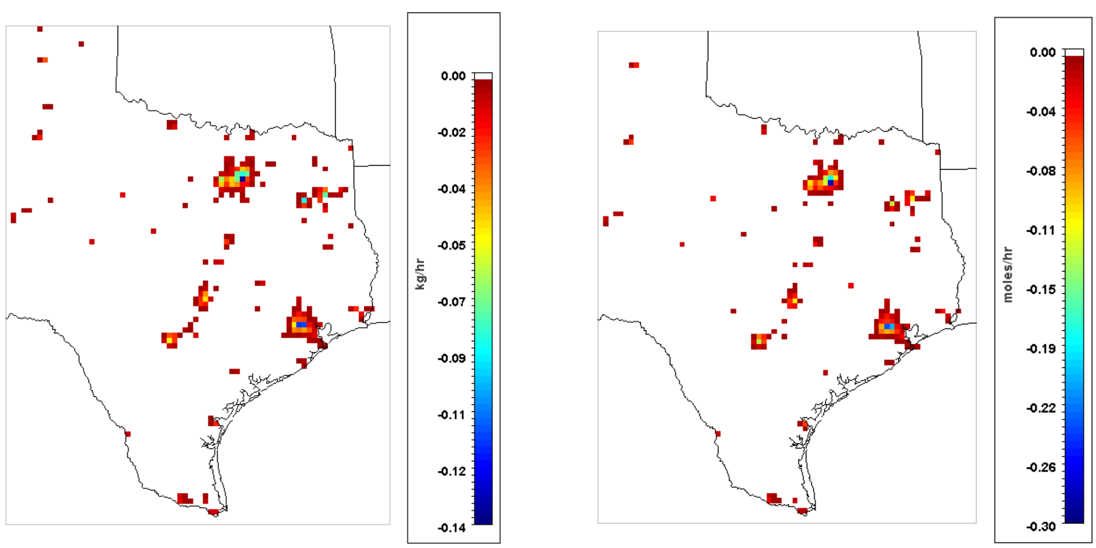
In CA eliminating petroleum fuel production, distribution, and storage results in major reductions in NO_x in three key areas: the S.F. Bay Area, Central Valley, and SoCAB. As can be seen in Figure 42 (a) impacts in southern California include the highest reductions in NO_x (89 kg/hr) coinciding with the sites of major refineries located adjacent to the Long Beach and L.A. Ports. Similar spatial improvements in direct P.M. are observed with maximum reductions reaching 36 kg/hr. Reductions in emissions of VOCs are more distributed throughout the state, as seen in the NEUS, with peak reductions coinciding with large refineries Figure 42 (b).



(a) **(b)**

Figure 42: Impacts on 24-h average (a) NO_x in central and southern CA and (b) VOC emissions state-wide from removing petroleum fuel production

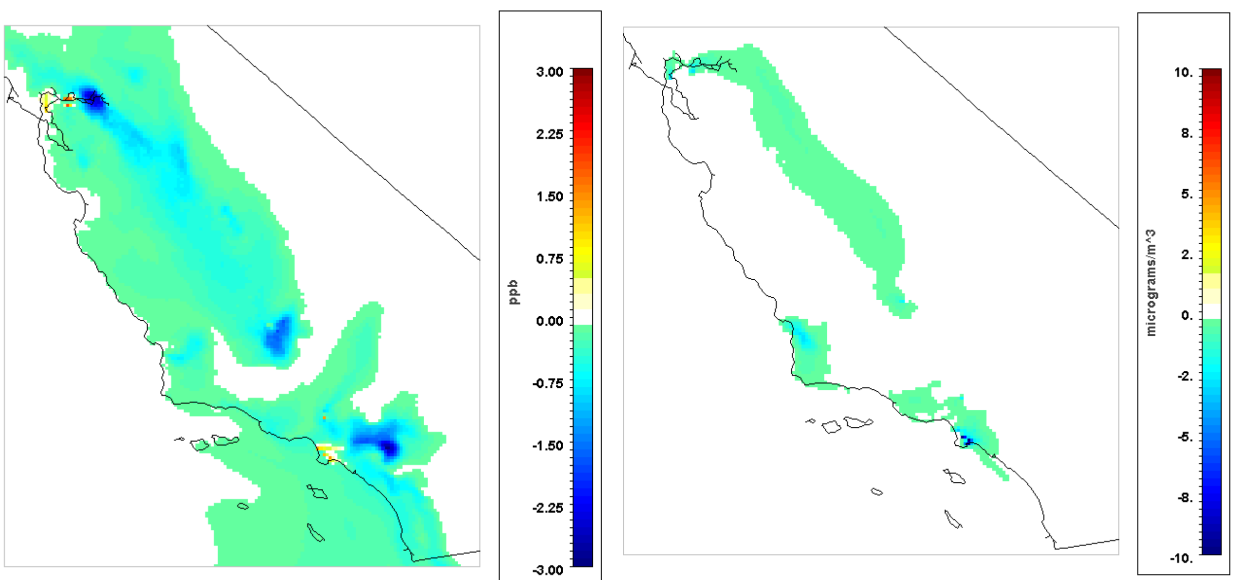
In TX impacts from removing petroleum emissions are roughly three orders of magnitude lower than in CA and the NEUS. Spatially impacts are as expected and congregate most heavily near the industrial complexes located on the Houston ship channel.



(a) **(b)**

Figure 43: Impacts on 24-h average (a) NO_x and (b) VOC emissions from removing petroleum fuel production in TX

The largest impacts from removing petroleum fuel production-related emissions occur in CA, with reductions in peak ozone exceeding 3.5 ppb in areas of greatest effect. Important ozone reductions are observed in three major areas associated with petroleum fuel infrastructure: the SF Bay Area, Bakersfield, and the SoCAB. Reductions in 24-h $PM_{2.5}$ are particularly high in magnitude in SoCAB with peak impacts in excess of 14.6 μg/m^3 occurring near the Long Beach and L.A. Ports; which include communities heavily impacted by PM levels.



(a) **(b)**

Figure 44: Impacts on (a) peak ozone and (b) 24-h $PM_{2.5}$ from petroleum fuel production in CA

Despite the relatively lower emissions reductions in TX compared to CA and the NEUS, AQ impacts are still evident and with high magnitude. As shown in Figure 45, removing the emissions from petroleum fuel production infrastructure reduces peak ozone by over 1.3 ppb and 24-h $PM_{2.5}$ by 3.4 μg/m^3 in TX. Impacts on ozone include reduction plumes originating from refining activity along the coastline of the region including the Houston Ship Channel. Similarly, perturbations to $PM_{2.5}$ are most pronounced directly adjacent to the Houston Ship Channel as well with moderate reductions spread across a large portion of the region.

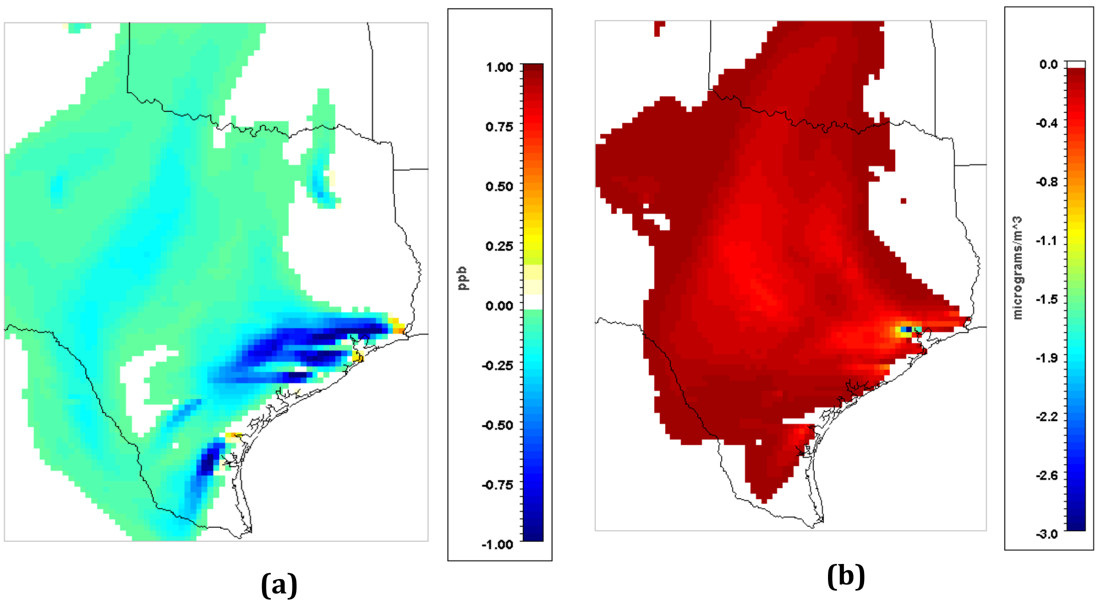
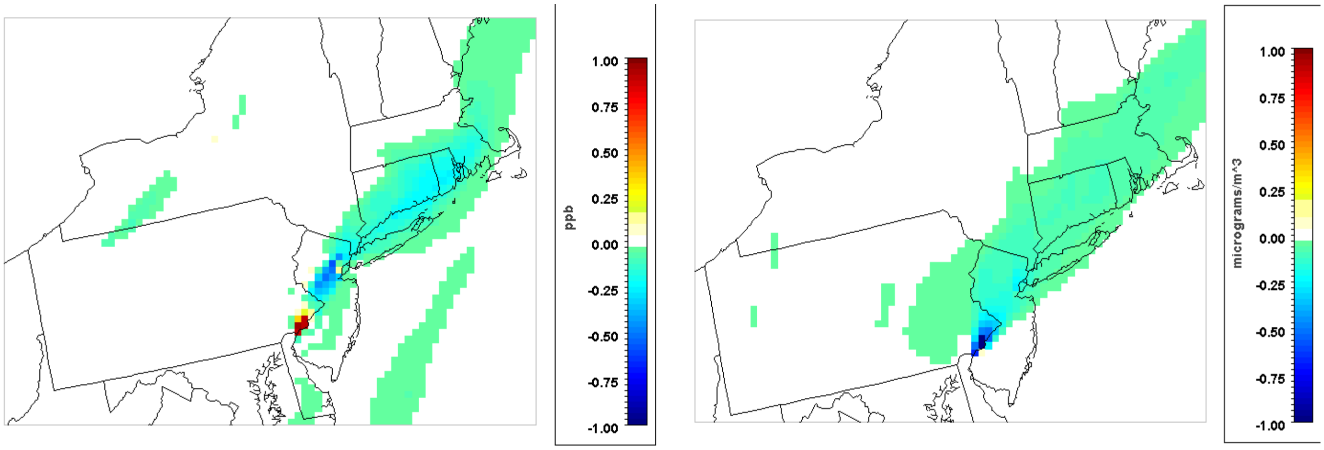


Figure 45: Impacts on (a) peak ozone and (b) 24-h $PM_{2.5}$ from petroleum fuel production in TX

In the NEUS impacts on peak ozone are less than TX and CA with maximum reductions of .56 ppb. However, reductions in $PM_{2.5}$ are more pronounced including reductions exceeding 1.7 μg/m^3 in some locations. Spatially, impacts for both pollutants are similar with improvements in NY and NJ as well as upwind of the NYC metropolitan area in regions experiencing high population densities.



(a) (b)

Figure 46: Impacts on (a) peak ozone and (b) 24-h $PM_{2.5}$ from petroleum fuel infrastructure in the NEUS

Emissions from petroleum fuel production and distribution activity are substantial and represent a significant opportunity for AQ and GHG co-benefits in the study regions. As can be seen in Table 19, removing the pertinent PFI activity achieves major reductions in emissions of NO_x, PM, and VOCs in regions of study. Specifically, improvements in PM emissions could have direct health benefits for populations positioned near areas of large refinery activity. Further, large reductions in NO_x and VOC emissions would be expected to lower levels of secondary pollutants, including ozone, in addition to providing a direct health benefit. In addition, the largest impacts on NO_x and PM are co-located with major refinery complex locations while VOC emission reductions are more distributed throughout regions. Locations of refineries near major ports could increase the importance of reducing emissions due to additive effects with emissions from other sources

Table 19: Peak emission reductions from removal of regional petroleum fuel infrastructure

Region	Δ 24-hr NO_x [kg/hr]	Δ 24-hr PM [kg/hr]	Δ 24-hr VOC [moles/hr]
TX	-.137	-.015	-.277
NEUS	-157.6	-53.0	-594.5
CA	-89.9	-36.7	-893.9

As can be seen in Table 20, displacing emissions from the production and distribution of petroleum fuels significantly improves AQ in all three study regions, with particular impact in CA. It is interesting to note the large area of resulting AQ improvements observed in the study regions considering emissions are generally reduced from a few select locations. For example, in TX major emission sources are located near coastal ports but improvements in ozone and $PM_{2.5}$ extend throughout the entire state. Additionally, locations of greatest impact, e.g., reductions in PM near the Long Beach and L.A. Ports, often coincide with areas encompassing large populations that are currently plagued by poor AQ. Further, the significant improvements further support the deployment of strategies to reduce emissions from transportation as all sub-sectors rely on petroleum fuels. Thus, addressing petroleum refinery emissions represents an important opportunity to maximize the AQ benefits of alternative transportation fuels.

Table 20: Peak impacts on ozone and 24-h $PM_{2.5}$ from removal of petroleum fuel infrastructure

Region	Δ Peak Ozone [ppb]	Δ $PM_{2.5}$ 24-hr [μg/m³]
TX	-1.34	-3.42
NEUS	-0.56	-1.77

CA	-3.55	-14.6

The impacts described here are consistent with the current understanding in the literature; particularly for large refinery complexes. Refining facilities co-emit large quantities of NO_x and reactive VOCs[275]; conditions that are typically associated with rapid and efficient ozone formation[79], and have been shown to contribute to elevated ozone and $PM_{2.5}$ levels in urban regions [287, 289, 313]. Emissions of NO_x generally occur from a few large sources processes, e.g., combustion of fuel for power and heat, flaring, [79] while refinery VOC emission sources are typically more diffuse and include an assortment of compounds from stacks, process vents, flares, cooling towers, and leaks from storage tanks, pipes, and valves[286]. Further, refining is generally a large scale, high capacity process and many facilities are operated continuously resulting in the steady generation of large quantities of emissions for extended periods (e.g., months to years) which differs from the defined temporal emissions signatures from other sectors.

4.5 POWER GENERATION SECTOR

4.5.1 Coal Power Plant and Mitigation Strategy Impacts

The significant emission impacts of coal-fired power generation give displacement the high potential for AQ and GHG co-benefits. Despite reductions in total capacity and generation from 2005 levels, coal continues to maintain a presence in the Base Case regional generation profiles for TX and the NEUS R2, meeting 25% and 36% of total generation respectively. In addition, the NEUS R1 receives 11% of its power from coal.

Potential mitigation strategies examined for the displacement of coal include nuclear power plants and carbon capture and storage (CCS), which share similar base load and large centralized operating characteristics. Though both strategies can significantly reduce GHGs, impacts on pollutant emissions differ. Nuclear power plants operate virtually free of pollutant emissions while CCS deployment can have species-dependent increases and decreases depending upon the specific technical system that is utilized.

4.5.1.1 Methodology

In order to examine the GHG impacts of replacing coal power plants with nuclear plants versus deploying CCS, the TX generation profile for 2055 was determined by fuel and technology (coal to existing steam met 27% of regional power, gas to combined-cycle supplied 54%, and gas to combustion turbine provided 4%). Using average emission factors for grams CO_2 per unit power (derived from literature review) the total regional GHG emissions were determined including fractional contributions of fuels. Demonstrating the disproportionate contribution, coal plants emit 47% of GHG emissions despite supplying 27% of the power.

For the 100 nuclear deployment scenarios it is assumed that all coal power plants are replaced with nuclear power with an LCA value of 26 g CO_2/kWh which is an average value derived from the literature review. Natural gas generation continues to meet Base Case demands and has equivalent GHG emissions. Perturbations to Base Case emission levels are presented in Figure 47. The resulting reduction in GHG is approximately proportional to the contribution of coal, i.e., a 46% reduction in power sector emissions and over a 19%

reduction in total regional emissions. In addition, emissions of total regional NO_x , SO_2, and CO are removed by 14%, 11%, and 4%, respectively. PM emissions are also significantly reduced. Gas generation was not removed in the 100 Nuclear Case due to the different dynamic operation characteristics typically employed by such generators as nuclear plants typically are base load while gas generation can have increased flexibility such as ramping capabilities. It should be noted that significant use of nuclear power could offset some future gas and thus this scenario represents a conservative estimate of potential nuclear impacts in that regard.

For the 100 CCS deployment scenarios it is assumed that post combustion amine capture systems are deployed on all coal and natural gas fired generation. Average values identified in the literature (Table 21) are assumed for capture efficiencies and energy penalties. As can be seen in Figure 47, GHG emissions are reduced by over 75% from the power sector and nearly 35% from regional totals. The larger removal of carbon emissions relative to the 100 Nuclear Case is a result of the capture of gas-fired generation emissions and demonstrates the ability of CCS to reduce CO_2 emissions from power generation. In addition, emissions of SO_2 and PM decrease as a result of deploying post-combustion capture technologies. In contrast, emissions of some pollutant species increases relative to the Base Case as a result of increased fuel combustion necessary to meet energy needs of CCS systems. Most notably, emissions of NO_x increase 24% in the power sector, translating to a 3% increase in regional totals. Additionally, CO increases proportionately.

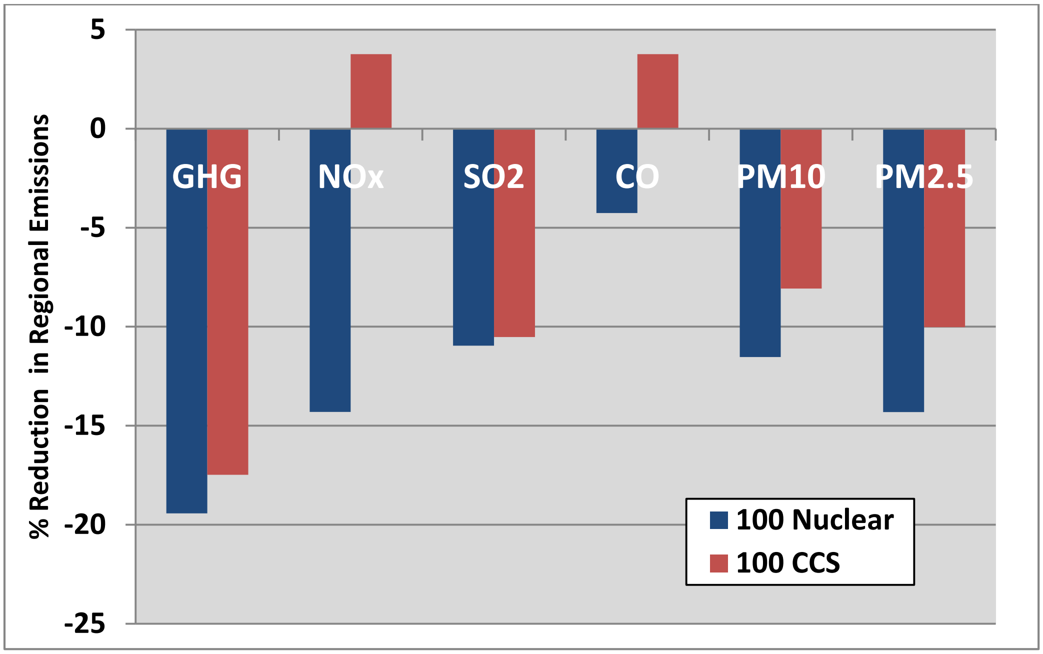


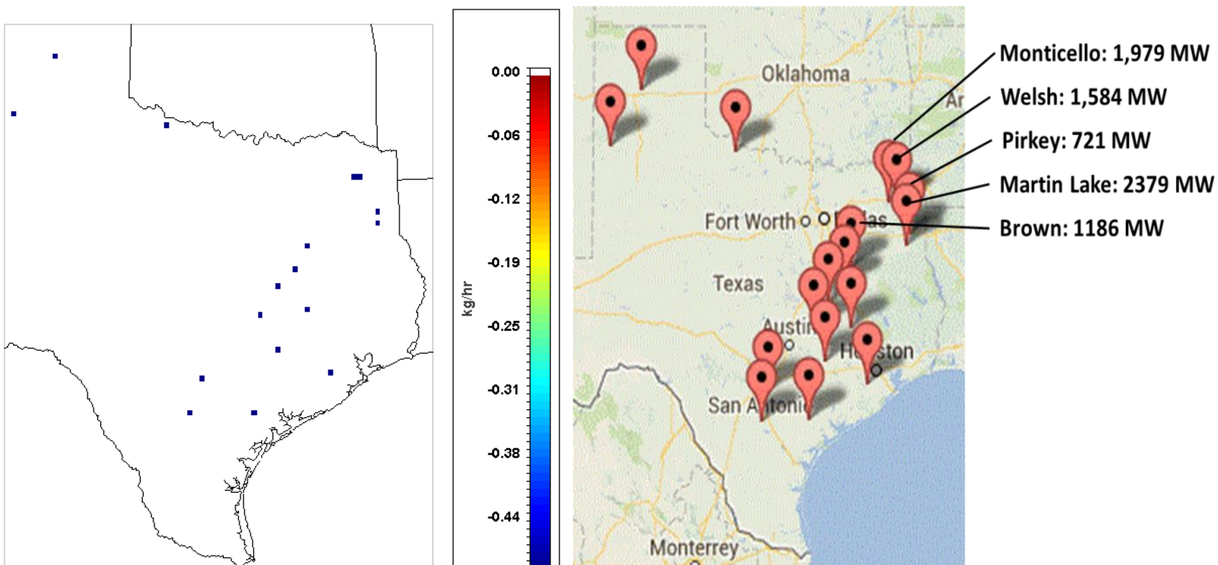
Figure 47: Alterations to Base Case emissions in the 100 Nuclear and CCS Cases

4.5.1.2 Nuclear Power Air Quality Impacts

Deploying nuclear power plants in place of existing coal power plants in 2055 is fundamentally similar to removing direct emissions from coal plants since nuclear plants do not directly emit GHGs or pollutants. Indirect emissions from plant-related activities are assumed to be comparable between plant types and not altered in the scenarios. In addition, no changes are made to natural gas generation as nuclear power is traditionally operated as base load generation and may not be appropriate to displace more flexible gas generation in some situations, e.g., for load-balancing.

The reductions in NO_x from removing coal power plant emissions are shown in Figure 48 along with the known locations of current major coal power stations in TX. As can be seen, emission impacts match closely with plant locations and provide validation for the methodology. In particular, a cluster of high capacity plants in the Northeast corner of the

State are noticeable and provide important emission reductions when removed. Similar impacts are noticeable through the center of the State correlating to major population centers including Austin, San Antonio and Houston.



(b)

(a)

Figure 48: (a) Impacts on 24-h NO_x emissions from replacing coal power plants with nuclear power plants in TX and (b) locations of existing coal power plants in TX

In TX, removing coal power plants via replacement with nuclear generation yields improvements in peak ozone that exceed 7 ppb in some areas (Figure 49 (a)). As would be expected, emission reductions from large, point source coal plants translate to ozone reductions that extend downwind in a plume-type structure. Impacts on 24-h $PM_{2.5}$ levels are spatially similar to ozone trends and include reductions of 2.3 $\mu g/m^3$. Visible in the results is the importance to AQ of the cluster of coal generators located in the northeast region of the state as the origins of the plumes with the largest improvements correspond to the location.

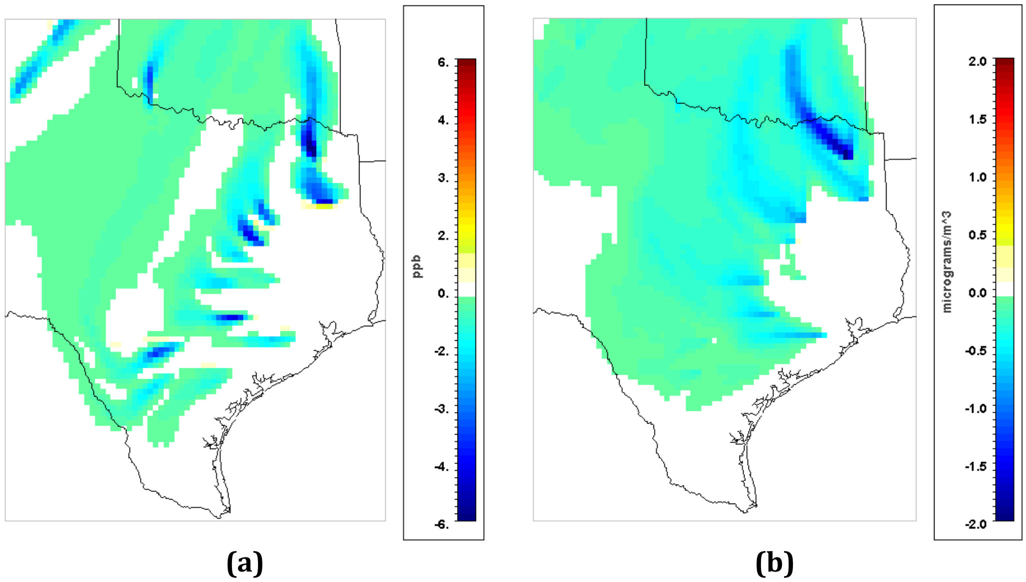


Figure 49: Impacts on (a) ozone and (b) $PM_{2.5}$ from replacement of coal power plants with nuclear power plants in TX

4.5.1.3 CCS Deployment Air Quality Impacts

For CCS deployment scenarios, it is assumed that CCS technology is applied to both coal and natural gas-fired generation. Impacts on criteria pollutant emissions are calculated by applying species-level factors identified from a literature review and displayed in Table 21. Factors vary for technologies and fuels and are applied accordingly to the portions of regional generation met with coal and gas.

Table 21: Emissions from power plants equipped with CCS relative to baseline plants

Technology	CO_2 [/kWh]	CO_2 [Total]	NO_x [Total]	SO_2 [Total]	P.M. [Total]	References
P.C.	-82-84%	-75-89%	+(13-79)%	-96 to +20%	-(29-35)%	[40, 315-317]
Super-critical P.C.	-72-87%	---	+(25-44)%	- (61-95%)	-(35-49)%	[40, 316, 318]
IGCC	-81-88%	-79-83%	-16 to +20%	+ (10-19%)	-(0-41)%	[40, 316-318]

NGCC	-59-83%	-51 to 80%	-50 to +17%	+ (0-100%)	-42 to +25%	[40, 315-319]

To determine the impacts on regional AQ from deploying CCS technology in the power generation sector Base Case emissions were adjusted to reflect the species-level effects in TX. Deploying CCS technology on all applicable sources increases 24-h NO_x emissions with peak impacts of 1.5 kg/hr, although most impacts are less than 0.6 kg/hr (Figure 50 (a)). In contrast, SO_2 emissions are reduced by up to .27 kg/hr in peak locations (Figure 50 (b)). Reflecting the point source nature of power stations impacts are distributed throughout the State and localized; with a cluster of sources present in the Houston and Austin areas. It is interesting that the largest impacts are not observed from large coal power plants in the northeast portion of the State. It should also be noted that SO_2 emissions are already assumed to be dramatically reduced in the Base Case due to implementation of pollutant legislation. Thus, impacts on SO_2 from CCS deployment could have increased importance if such legislation fails to be implemented or if coal use associated with high sulfur content coal experiences increased generation.

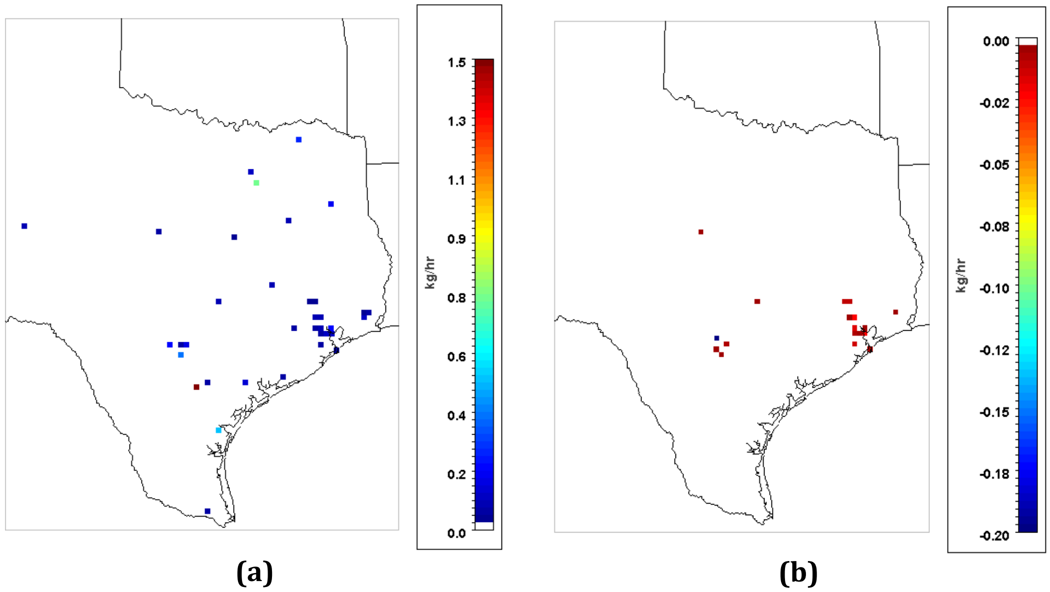


Figure 50: Impacts on 24-h (a) NO_x and (b) SO_2 from CCS on all power plants in TX

The increased NO_x emissions penalty from utilizing CCS technology yields significant increases in ozone, with peak impacts exceeding 2.5 ppb in some areas as shown in Figure 51. Escalations manifest as plumes that extend downwind from major point sources in the inverse of reductions observed in the 100 nuclear case.

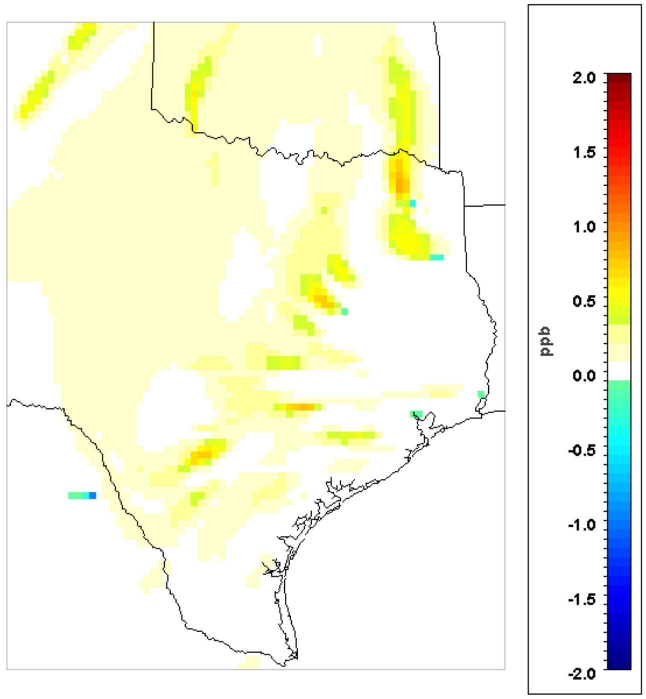


Figure 51: Impacts on peak ozone from deploying CCS on all power plants in TX

Deploying CCS on all fossil generators is unlikely given the enormous cost and technical challenges required to do so, including retro-fit of existing plants and locating appropriate storage locations. However, the results are useful in establishing an upper-bound for possible AQ impacts and for demonstrating the possible trade-off between reducing carbon emissions and worsening AQ. Additionally, the CCS evaluation capabilities developed in this section are utilized in following sections to examine interactions between the transportation and power generation sectors, e.g., the deployment of BEVs under various charging scenarios.

4.5.1.4 Coal Power Mitigation Conclusions

The use of coal for electricity generation at substantial levels in the Base Case will continue to give importance to addressing GHG and AQ impacts of coal power plants. Two potential strategies to mitigate carbon emissions (nuclear power and CCS technology) were evaluated for AQ and GHG impacts. Though both strategies can achieve significant GHG reductions, impacts on pollutant emissions differ. Nuclear power is generated free of direct pollutant emissions while CCS can have species-level increases, e.g., NO_x, and decreases, e.g., SO_2. Thus, CCS represents a GHG strategy with the potential for both AQ benefits and negative consequences (dis-benefits) that requires the investigation of spatial and temporal effects on secondary pollutants for robust assessment.

Significant reductions in 2055 power sector GHG emissions in TX and the NEUS will require addressing those from coal power plant. Potential strategies for deep cuts include deploying CCS and replacing coal generation with nuclear generation. In the TX Base Case, replacing all coal plants with nuclear achieves regional GHG reductions of almost 20% while deploying CCS on all fossil generation can reduce GHGs by 35%. Impacts on emissions of pollutants differ significantly however, as a result of efficiency penalties and operational impacts of CCS systems. For example, the 100 Nuclear Case removes NO_x by over 14% while the 100 CCS Case increases NO_x by 4%.

4.6 INDUSTRIAL SECTOR

The complexity of the emissions from industry sources makes generalized conclusions at the sector level difficult. Various industries exist which can potentially impact regional pollutant levels currently and in 2055. In terms of emissions signatures industries vary with regards to quantities, chemical compositions, temporal periods, source characteristics, etc. Additionally, industry varies from region to region, which together with geography and meteorology determines the overall impact on regional AQ. Industrial sources emit from different combinations of technologies, including external and internal combustion devices such as boilers or engines, stationary source combustion, and in-process fuel uses which can vary in characteristic between industries. Table 22 displays the difference in peak ozone and 24-hour $PM_{2.5}$ when emissions from the aforementioned source types that were removed, demonstrating the differences that each can have on regional AQ. As can be seen, the largest impacts on ozone and $PM_{2.5}$ in terms of industrial source emissions come from external combustion boilers. Additionally, internal combustion engines contribute over 3 ppb and nearly 1 μg/m^3.

Table 22: Impacts from Various Industrial Sector Combustion Types in TX

SCC Code	Source Category	Ozone Delta Max	PM 24-h average
10200000	External Combustion Boilers	-7.79 ppb	-2.86
20200000	Internal Combustion Engines	-3.19 ppb	-0.72
2102001000	Stationary Source Combustion	-1.85 ppb	-0.97
2390000000	In-process Fuel Use (Non-point)	N/A	N/A
390000000	In-process Fuel Use (Point)	-.037	N/A

Cases were developed and assessed for impacts on ozone and $PM_{2.5}$ from different industry sub-sectors in CA, TX, and the NEUS. Industry sub-sectors that were considered include chemical manufacturing, food processing, paper and pulp, primary and secondary metals production, and oil and gas production.

Figure 52 displays the resulting impacts on peak ground-level ozone concentrations in CA from removing emissions associated with chemical manufacturing (-3.9 ppb), food processing (-6.46 ppb), oil and gas production (-6.8 ppb), and primary and secondary metals

(-0.63 ppb). Spatial differences in industry locations throughout the State translate to variations in the location of ozone impacts for Cases. Chemical manufacturing has an important impact in the SoCAB, food processing has penultimate impacts in the San Joaquin Valley (Central Valley), and peak impacts from oil and gas production occur in and around Bakersfield. Primary and secondary metal production has a minor impact centered in SoCAB. Additionally, Table 23 presents peak impacts for ozone and $PM_{2.5}$ as well as the SCC codes that were considered in the case development.

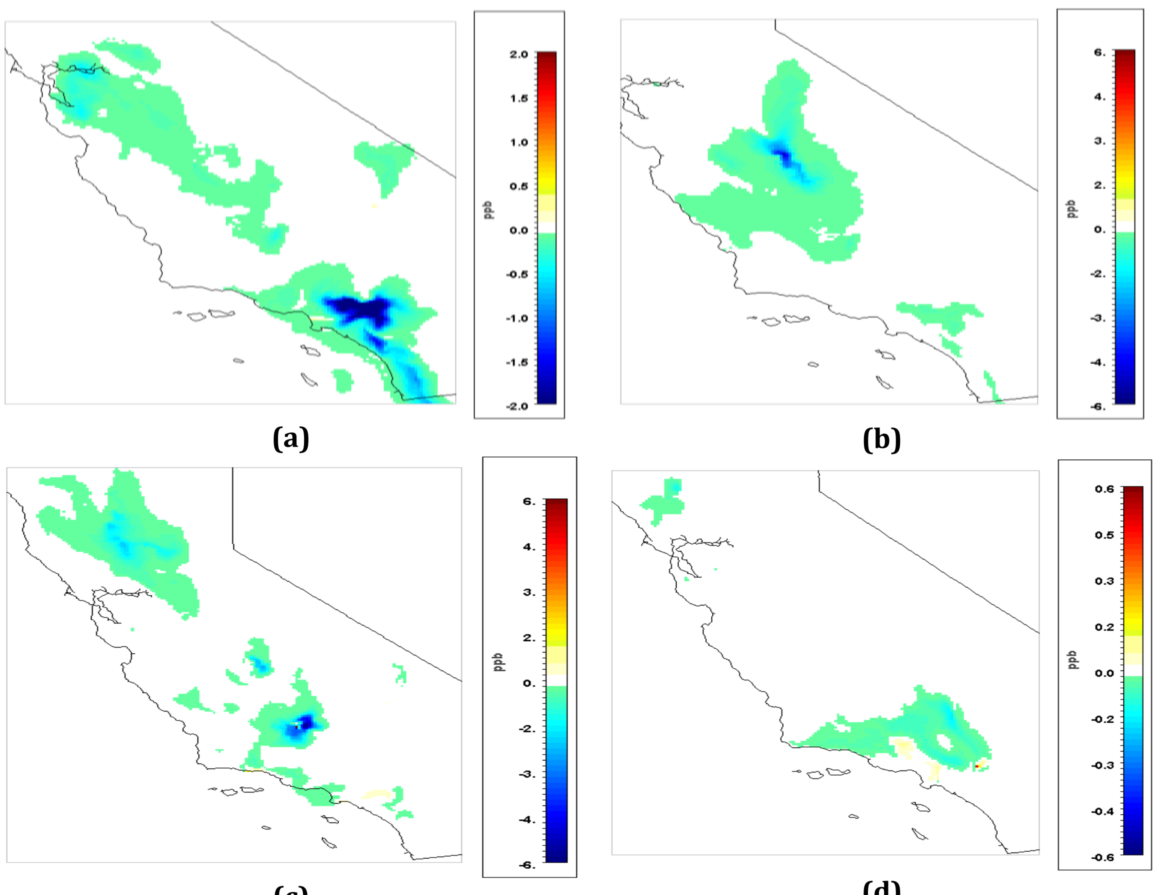
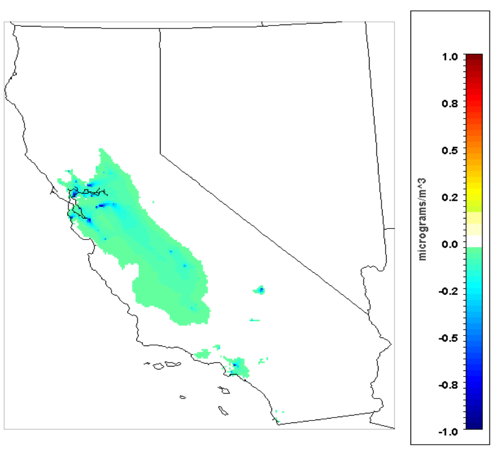
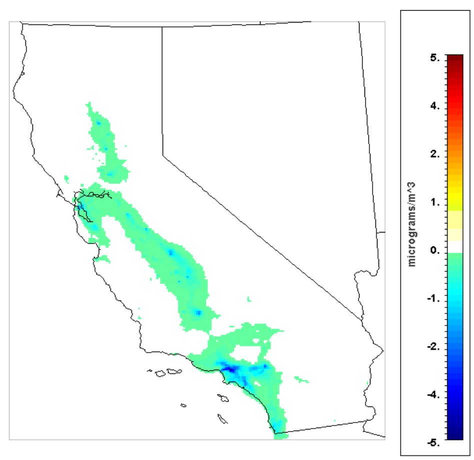


Figure 52: Impacts on peak ozone from (a) chemical manufacturing, (b) food processing, (c) oil and gas production, and (d) primary and secondary metals production

Figure 53 displays the resulting impacts on 24-hour $PM_{2.5}$ concentrations in CA from removing emissions associated with chemical manufacturing (-2.7 μg/m³), food processing (-5.8 μg/m³), paper and pulp (-9.0 μg/m³), and primary and secondary metals (-3.3 μg/m³). Of interest, the location of the most significant PM2.5 impacts can vary even for the same industry, e.g., peak food processing impacts on $PM_{2.5}$ include the SoCAB while ozone impacts are centered in the Central Valley. Paper and pulp impacts are most notable in the Northern California. In other Cases, including oil and gas production and chemical manufacturing, impacts on ozone and $PM_{2.5}$ are spatially similar.



(a)



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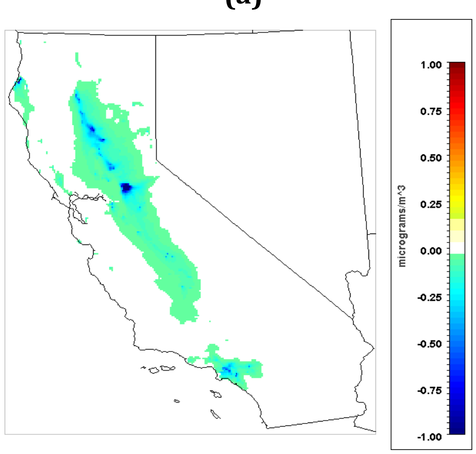
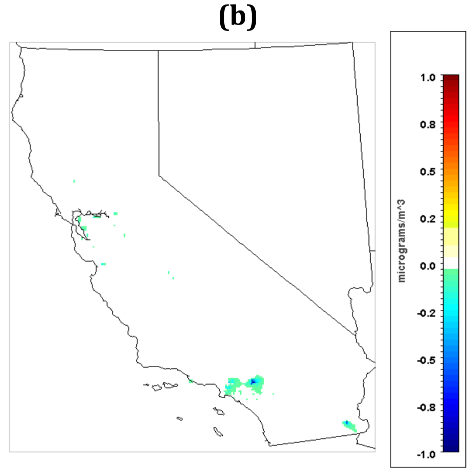


(c)



(d)

Figure 53: Impacts on 24-hour average $PM_{2.5}$ from (a) chemical manufacturing, (b) food processing, (c) paper and pulp, and (d) primary and secondary metals production

Table 23: Impacts from Industrial Sector Source Categories in CA

SCC Code	Source Category	Ozone Delta Max	PM 24-h average
30100000	Chemical Manufacturing	-3.99 ppb	-2.66
31000000	Oil and Gas Production	-6.81 ppb	-2.66
30300000, 30400000	Primary and Secondary Metal	-0.63 ppb	-3.32
30700000	Paper and Pulp	-1.07 ppb	-9.00
30200000	Food Processing	-6.46 ppb	-5.82
30500000	Mining/Mineral Products	-6.54 ppb	-16.40

Figure 54 displays the resulting impacts on peak ground-level ozone concentrations in TX from removing emissions in the Base Case associated with chemical manufacturing (-1.44 ppb), food processing (-0.78 ppb), paper and pulp (-0.05 ppb), and primary and secondary metals (-0.43 ppb). Essentially, the peak impact is represented by the maximum difference in concentration observed in the Base Case and the alternative case where emissions have been removed. Chemical manufacturing impacts are significant, originating in and around the Houston Ship Channel and propagating throughout the State. The remaining three sectors are more localized in regards to the resulting reductions. Additionally, Table 24 contains peak impacts for ozone and $PM_{2.5}$ as well as the SCC codes that were considered in Case development.

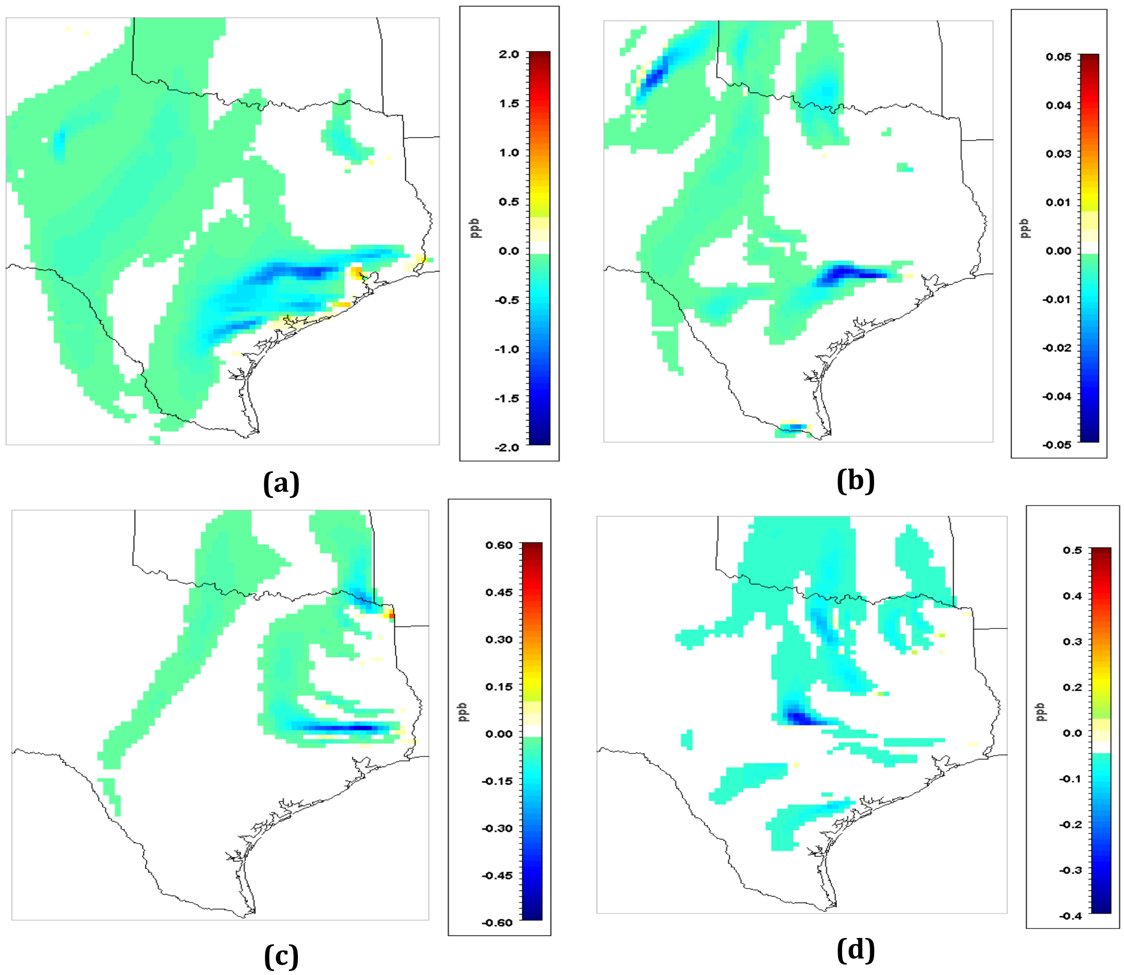


Figure 54: Impacts on peak ozone from (a) chemical manufacturing, (b) food processing, (c) paper and pulp, and (d) primary and secondary metals production

Figure 1 displays the resulting impacts on 24-hour $PM_{2.5}$ concentrations in TX from removing emissions associated with chemical manufacturing (-1.7 μg/m^3), food processing (-2.1 μg/m^3), paper and pulp (-9.0 μg/m^3), and primary and secondary metals (-2.2 μg/m^3). In particular, chemical manufacturing has an important impact on $PM_{2.5}$ with peak impacts occurring from sources in and around the Houston Ship Channel. Impacts from food processing are significant in magnitude but are highly localized in contrast to chemical manufacturing.

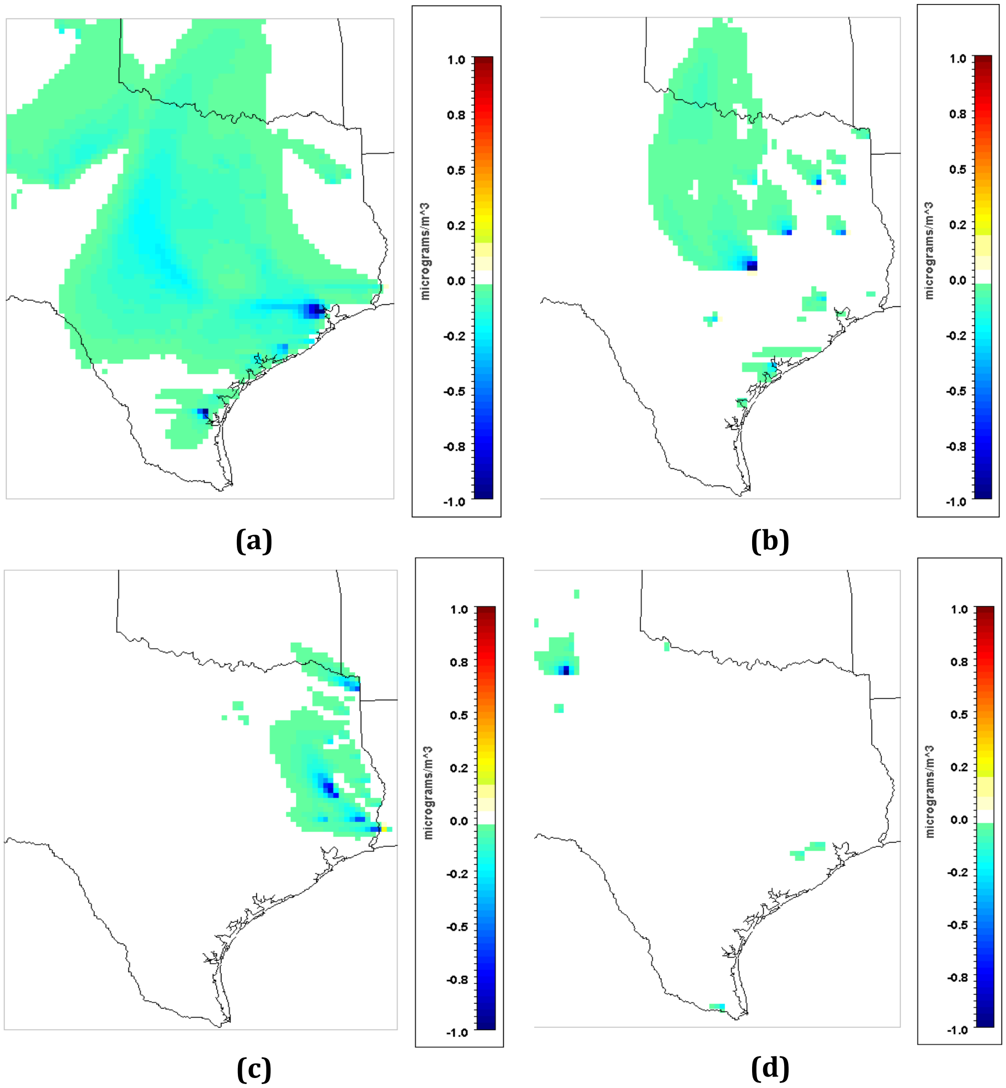
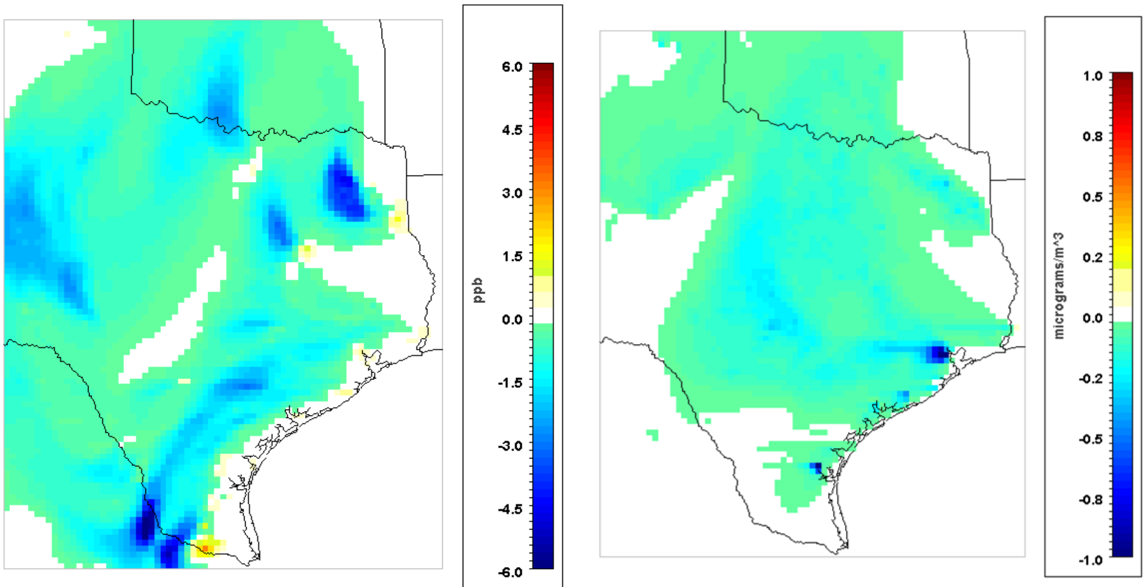


Figure 55: Impacts on 24-hour average $PM_{2.5}$ from (a) chemical manufacturing, (b) food processing, (c) paper and pulp, and (d) primary and secondary metals production

Figure 56 displays the impacts on peak ozone and 24-h $PM_{2.5}$ (-6.4 ppb and -1.7 µg/m^3 from oil and gas production in TX. Reductions in ozone and $PM_{2.5}$ when emissions from oil and gas production are removed are sizeable and occur throughout the State. The results are reasonable when considering the large oil and gas industry present in the State.



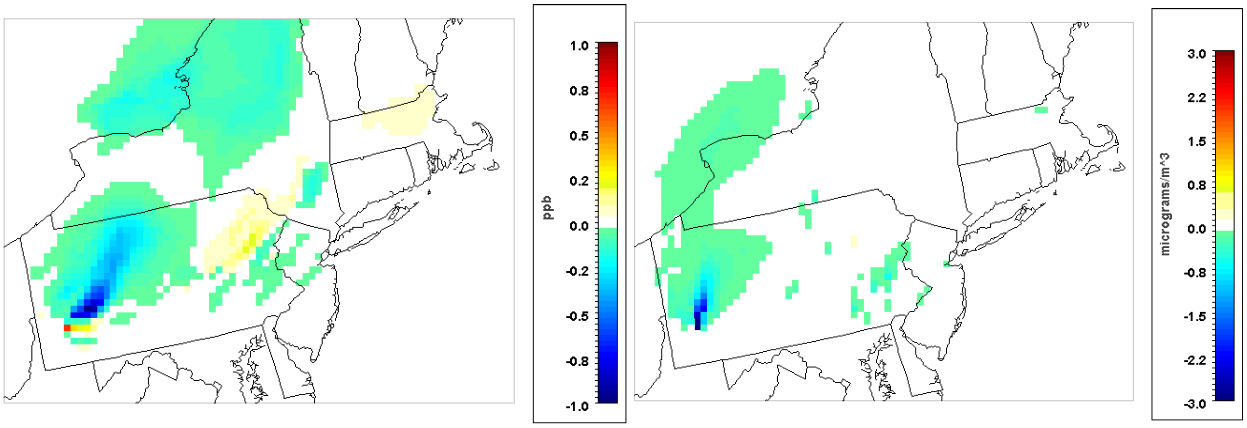
(a) **(b)**

Figure 56: Impacts on (a) peak ozone and (b) $PM_{2.5}$ from oil and gas production

Table 24: Impacts from Industrial Sector Source Categories in TX

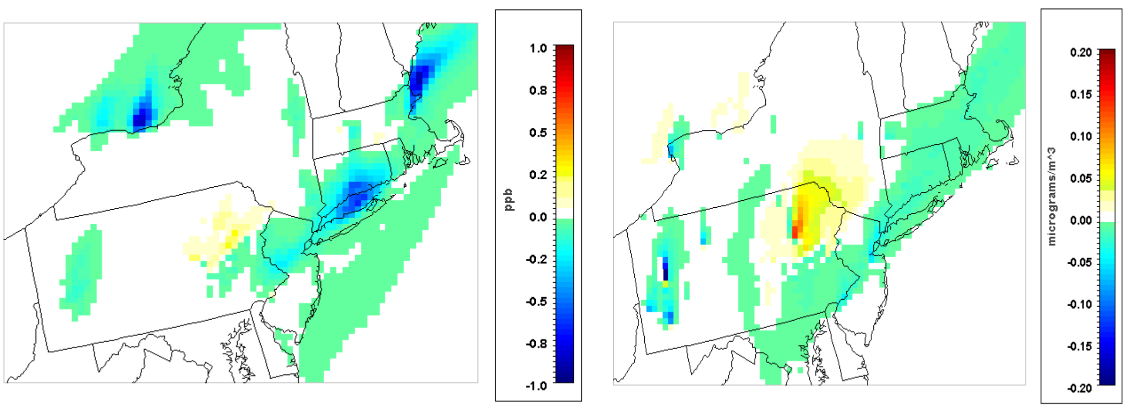
SCC Code	Source Category	Ozone Delta Max	PM 24-h average
30100000	Chemical Manufacturing	-1.44 ppb	-1.68
31000000	Oil and Gas Production	-6.37 ppb	-1.68
30300000, 30400000	Primary and Secondary Metals	-0.43 ppb	-2.18
30700000	Paper and Pulp	-0.78 ppb	-0.81
30200000	Food Processing	-0.05 ppb	-1.08

Figure 57 displays the resulting impacts on ozone and $PM_{2.5}$ concentrations in the NEUS from primary and secondary metal production (-1.1 ppb and -2.2 μg/m^3) and Figure 58 displays impacts from chemical manufacturing (-0.9 ppb and -0.3 μg/m^3). The largest impacts from metal production occur in the Pittsburgh area for both ozone and $PM_{2.5}$. Chemical manufacturing has important impacts on ozone upwind of NYC and Boston and $PM_{2.5}$ in and around Pittsburgh. Additionally, Table 25 displays the peak impacts, SCC codes, and peak temporal impact periods for the NEUS Cases.



(a) (b)

Figure 57: Impacts on (a) peak ozone and (b) 24-h $PM_{2.5}$ from primary and secondary metal production in the NEUS



(a) (b)

Figure 58: Impacts on (a) peak ozone and (b) 24-h $PM_{2.5}$ from chemical manufacturing in the NEUS

Table 25: Impacts from Industrial Sector Source Categories in NEUS

SCC Code	Source Category	Ozone Delta Max	Ozone Peak	PM 24-h average
30100000	Chemical Manufacturing	-0.92	17 UTC	-0.25
31000000	Oil and Gas Production	-0.27	13 UTC	-0.25
30300000, 30400000	Primary and Secondary Metal	-1.06	20 UTC	-3.58
30200000	Food Processing	-0.46	17 UTC	-3.0
30500000	Mining/Mineral Products	-8.52	19 UTC	-3.3

Chapter 5: Air Quality Impacts of Advanced Light Duty Vehicle Technologies

5.1 AIR QUALITY IMPACTS OF FUEL CELL ELECTRIC VEHICLES

Transportation sources account for an important fraction of total emissions driving regional AQ concerns in many U.S. States, including ambient concentrations of ozone and particulate matter (PM) associated with detrimental human health outcomes[1]. In California (CA) emissions from the combustion of fossil fuels by transportation sources including light duty vehicles (LDV) have been shown to be major contributors to total regional pollutant burdens [2]. It follows then that shifts to cleaner alternative propulsion systems is being pursued in CA to reduce the environmental impacts of the LDV transportation sector, including on regional AQ [3, 4]. A key strategy to reduce emissions of greenhouse gasses (GHG) and criteria pollutants includes hydrogen in tandem with fuel cell electric power trains (FCEVs) as FCEVs produce no direct (i.e., tail pipe) emissions during operation[5-7]. In addition, FCEV technologies offer the benefits of high efficiencies [8, 9], similar ranges and refueling times compared to combustion engines [6, 10, 11], and the promotion of domestic energy independence via displacement of petroleum fuels (hydrogen can be produced from a range of domestically available feedstocks) [12].

The deployment of FCEVs can reduce total LDV emissions across a wide range of hydrogen infrastructure options from the potential for very low lifecycle GHG and criteria pollutant emissions compared to current and future conventional LDVs, including oxides of nitrogen (NO_x), volatile organic compounds (VOC), PM, and carbon monoxide (CO) [13-17].

Replacing the current on-road LDV fleet with FCEVs reduced net GHG emissions in the U.S. [18, 19] and CA [20] and similar findings have been reported for pollutant emissions at various scales [21-23]. In particular, FCEVs supplied with hydrogen produced from renewable power-provided electrolysis have the potential for achieving large reductions in total emissions from transportation [24]. However, assessing how using FCEVs in place of conventional LDVs impacts regional AQ is multifaceted and requires more than simply quantifying emission perturbations.

The complexity associated with the formation and fate of atmospheric pollutant species complicates an understanding of how FCEVs deployed in select counties will impact regional AQ generally in CA air basins. In particular, the dynamics associated with the production of ground-level ozone from pre-cursor emissions lessens the value of solely quantifying emission reductions in pursuit of AQ outcomes[25]. Similarly, atmospheric levels and compositions of PM in CA are governed by a large range of factors including sources of particulate and atmospheric processes that control particle formation that could lead to spatial and temporal variation in source-related impacts and potential mitigation strategies[26]. Therefore, detailed atmospheric models must be used to accurately account for the spatial and temporal distribution of pollutant concentrations in order to conduct a detailed assessment of how FCEVs may affect tropospheric ozone and $PM_{2.5.}$ While some studies have shown that FCEV driven reductions in direct emissions result in improvements in secondary air pollutants, including ground-level ozone and $PM_{2.5}$ [27, 28], the body of available literature is limited. Use of a novel methodology for future hydrogen infrastructure development in the South Coast Air Basin (SoCAB) of CA reported substantial reductions emissions including NO_x for the majority of Cases [23] translating to significant AQ improvements (e.g., reductions in peak 8-h-averaged ozone and 24-h-averaged $PM_{2.5}$ concentrations) [20]. However, these studies were spatially restricted to select regions of CA (e.g., the SoCAB, Sacramento) or the entire U.S. Further, no study has considered secondary pollutant impacts arising from emission perturbations associated with county level deployment.

In addition to direct emissions from LDVs, the production and distribution of petroleum fuels (most notably motor gasoline) incurs emissions of criteria pollutants[29, 30]. A widespread petroleum fuel infrastructure (PFI) exists in the U.S., including large refinery complexes which process and refine crude oil feedstock into finished products including LDV fuels [31]. Point and area source emissions released from refineries comprise numerous pollutant and air toxics including CO, NO_x, PM, SO_2, VOCs, benzene, and toluene[32]. Further, emissions associated with various life cycle stages of petroleum fuels are known contributors to regional AQ burdens including non-compliance with Federal AQ standards[33] and may be underreported [34-36]. With particular importance to CA AQ, petroleum refining facilities co-emit large quantities of NO_x and VOCs which contribute to the rapid and efficient formation of tropospheric ozone [37] and have been linked with pollutants in CA air basins [38]. Deploying FCEVs in CA will reduce the consumption of gasoline and other petroleum fuels and could potentially offset emissions from PFI sources. However, the specific response of, for example, large in-state refinery complexes are unknown and how PFI emission perturbations impact AQ is subject to the same uncertainty as those from direct vehicles. For example, ozone formation dynamics differ downwind of power plants and industrial sources due to discrepancies in NO_x and VOC emission rates and ratios [20, 28, 33]. Thus, further information is needed regarding the importance of PFI emissions to regional AQ in CA, notably in regards to interactions with state goals for alternative LDV strategies.

The true consequences of transitioning to hydrogen as a vehicle fuel will be determined by the full life cycle of deployed vehicle and fuel pathways[39]. Hydrogen is an energy carrier, not a primary energy source, and can be produced from a variety of primary energy sources, including fossil and renewable sources. Currently, low cost and widely used supply chain strategies to produce hydrogen are fossil-based options such as steam methane reformation (SMR) of natural gas resulting in the generation and release of both GHG and pollutant emissions [40]. However, hydrogen production via methods with enhanced sustainability will likely increase as GHG, AQ and additional environmental goals drive technological development and deployment [12]. Future options could include centralized and distributed electrolysis of water using power generated from renewable sources (e.g., wind and solar power) to achieve near-zero carbon hydrogen production and a potential pathway for FCEVs to deeply reduce GHG and criteria pollutants relative to current strategies [41]. Additional prospective renewable hydrogen systems include various processes (e.g., gasification, pyrolysis, fermentation, anaerobic digestion) associated with biomass or biogas feedstock and additional routes incorporating solar energy such as thermochemical splitting of water [42]. Furthermore, the integration of hydrogen production with the future electric grid could have benefits by essentially providing complementary services in the form of energy storage and can allow for greater penetrations of renewables, particularly those plagued by intermittency challenges [43].

Due to this potential, hydrogen has been proposed as an important complement to the implementation of wind energy in part as a means of coupling GHG mitigation strategies in the utilities and transportation sectors [44-46]. Similar concepts and conclusions have been reported for solar hydrogen production [47]. As many places around the world (including CA) are pursuing greater procurement of renewable energy in coming decades, including significant amounts expected from intermittent wind and solar technologies, the incorporation of hydrogen energy systems to provide fueling for vehicles and stationary sources could represent an important opportunity to maximize AQ and GHG benefits and maintain grid reliability.

5.1.1 Methodology

5.1.1.1 Regional Energy System Projection

The sources, magnitudes, and spatial/temporal distributions of future anthropogenic emissions will be affected by a wide-ranging assortment of drivers, including population growth and migration, economic growth and evolution, availability and depletion of various energy resources, climatic changes, technology development and deployment, future policy implementation, and human behavior[48]. Assessing regional AQ impacts of FCEVs in 2055 then requires emissions projection of all relevant anthropogenic sources by consistent methods. Of foremost importance is the comprehensive accounting of regional emissions evolution under business-as-usual (BAU) conditions to provide a Reference Case for comparison with FCEV and HDV Cases. The approach for the developed 2055 Reference Case follows the methodology described by Loughlin et al., 2011[48]. Energy system progression and the evolution of emissions in major economic sectors is estimated using output from the Market Allocation (MARKAL) model. MARKAL is a data-intensive energy systems economic optimization model utilizing EPA developed and maintained regional databases that serve to characterize regional energy systems to the year 2055. Energy system details accounted for in model framework include primary resources, conversion technologies, end-use demands, and technological pathways to meet future energy demands. Model outputs include demands, technologies, fuel use and emissions of pollutants from current to 2055. Emissions from energy sectors are reported for CA and utilized to develop growth factors specific to various source classification codes (SCC) assigned to technology and fuel pathways. The MARKAL run used to produce the Reference Case integrates future constraints, including a representation of the recent federal CAFE standard (54.5% by 2025), the Cross-State Air Pollution Rule (CSAPR), and the Mercury and Air Toxics Standards. Factors were generated using the 9-region MARKAL database, version 1.3, with updated electric sector and CSAPR representations (v1.5_052112_dhl and 1.3_052212_dhl, respectively). With the exception of the CAFE standard, the assumptions used to project the Reference Case were calibrated to the 2010 Energy Information Administration's Annual Energy Outlook.

5.1.1.2 Development of Emissions Fields

Construction of spatially and temporally resolved emission fields by the Sparse Matrix Operator Kernel Emissions (SMOKE) Modeling System is necessary to generate inputs representative of assessed Cases. SMOKE is an emissions processing tool that develops appropriately formatted inputs for AQ models using a series of matrix calculations [49]. SMOKE accomplishes the core functions of emissions data needs such as spatial and temporal allocation, chemical speciation, biogenic emission estimates and control of area-, mobile-, and point-source anthropogenic emissions. Additionally, growth and control factors generated as output from MARKAL were applied to the 2005 base-year inventory via SMOKE, including disaggregation of emissions into constituent chemical species via SCC-specific chemical speciation profiles. Spatial and temporal allocation of both point and area-source emissions into a 3-D modeling grid is performed via source coordinates and spatial surrogates at the county level and SCC-specific temporal allocation profiles. Source-specific information used in allocation methodologies includes land use, census data, employment information, and others.

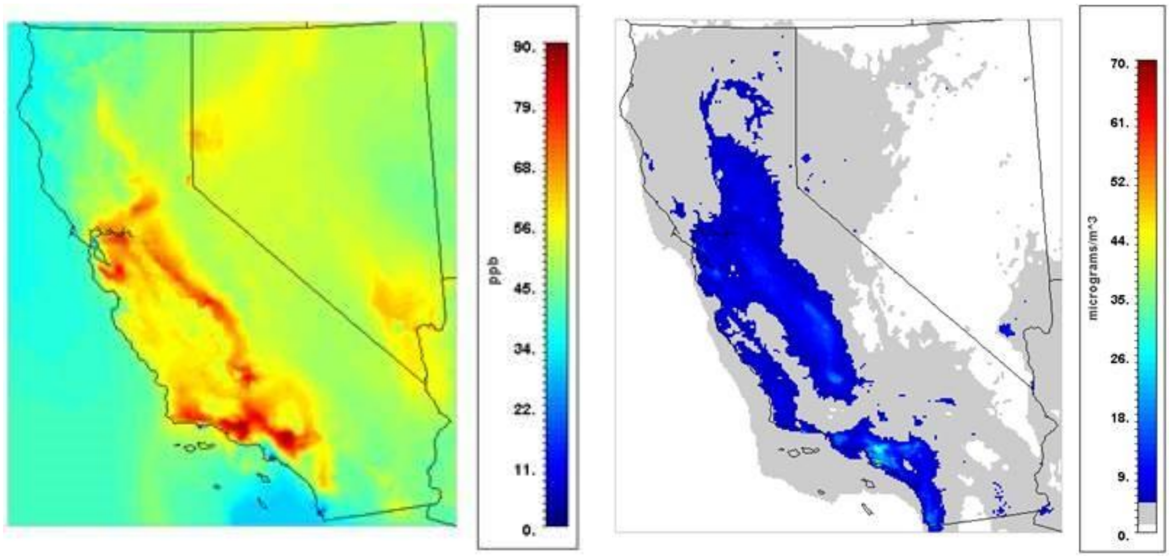
5.1.1.3 Atmospheric Modeling

Simulations of atmospheric chemistry and transport are accomplished via the Community Multi-scale Air Quality model (CMAQ) version 4.7, with the Carbon Bond 05 chemical mechanism [50]. CMAQ is a comprehensive AQ modeling system developed by the US Environmental Protection Agency (EPA) and widely used for a various AQ needs, e.g., regulatory simulation applications [51, 52]. The source code and technical formulation of the model are available from the CMAQ website: www.cmaq-model.org. CMAQ is designed from the "one atmosphere" perspective and is used for studies on tropospheric ozone, PM, acid deposition and visibility. The CMAQ system includes a meteorological modeling system, emissions modeling system and chemical transport modeling system. Model inputs include meteorological conditions, initial and boundary conditions, land use and land cover information, and anthropogenic and biogenic source emissions. The chemical mechanism used is the CB05, which includes the photochemical formation of ozone, oxidation of volatile organic compounds and formation of organic aerosol precursors. For the simulations presented in this report, the spatial resolution of control volumes is 4 km × 4 km, and a vertical height of 10,000 meters above ground, with 30 layers of variable height based on pressure distribution. Meteorological input data for CMAQ was obtained from the Advanced Research Weather Research and Forecasting Model, WRF-ARW. The National Centers for Environmental Prediction (NCEP) Final Operational Global Analysis 1° × 1° grid data are used for WRF-ARW initial and boundary conditions.

Simulations are conducted for the week of July 7-13 as this period encompasses conditions typically associated with high tropospheric ozone formation in many CA regions, including high temperatures, an abundance of sunlight, lack of natural scavengers, and the presence of inversion layers [53]. The first six days of simulations are used to dissipate the effects of the initial conditions as this has been shown to be sufficient[53]. Results are obtained from the seventh day of simulation (July 13) and reported as maximum 8-hr average ozone and 24-hr average $PM_{2.5}$.

5.1.1.4 Reference Case

In the Reference Case the LDV sector follows current trends and is predominantly comprised of gasoline combustion engine technologies (a moderate to minor amount of LDV demand is assumed to be met with alternative technologies and fuels including electricity and E-85). The HDV sector is similarly assumed to be reliant on fossil-based fuels including distillate fuels, compressed natural gas, etc. Simulated ground-level concentrations of ozone and $PM_{2.5}$ are shown in Figure 59 for the Reference Case in 2055. As can be seen, some regions of the State experience greater ground-level concentrations which heighten the importance of reductions, including the SoCAB, the San Francisco (SF) Bay Area, the Central Valley, and the Greater Sacramento area. These areas currently experience high levels of ground-level ozone that often exceed Federal health-based standards and contain large urban populations [56]. Thus, improvements in these areas are desirable to the State in terms of mitigating deleterious human health outcomes from air pollution[57]. The Reference Case serves as a basis for comparison for FCEV and HDV Case with results presented as difference plots for pollutant distributions.



(a) **(b)**

Figure 59: Predicted ground-level concentrations of (a) max 8-hr average ozone and (b) 24-hr average $PM_{2.5}$ for the Reference Case during a typical summer day in CA in 2055. Projected peak levels exceed 90 ppb and 78 µg/m^3.

5.1.2 Statewide FCEV Deployment

5.1.2.1 Case Development

To evaluate changes in spatial and temporal distributions of ozone and $PM_{2.5}$ from FCEV deployment a set of Cases are developed and analyzed in California for the year 2055. Assessment of Cases comprises the construction of spatially and temporally resolved emission fields appropriately accounting for all mobile and stationary source perturbations followed by simulations of atmospheric chemistry and transport. Output from atmospheric modeling is then assessed for changes in ground-level maximum 1 hour (1-hr) average ozone and 24 hour (24-h) average $PM_{2.5}$ concentrations relative to the baseline Case (i.e. gasoline internal combustion engine dominant) for the same year.

Table 26 displays the emission impacts for the various sources comprised in the assessed Cases. All Cases encompass a FCHV penetration of 90% of all LDVs in California in 2055 and direct (i.e., tailpipe) LDV emissions are correspondingly reduced fleet-wide, i.e., all Cases exhibit a 90% reduction for all direct pollutants including NO_x, PM, VOC, etc. Reductions are applied via SMOKE and occur over all road-way types throughout the State.

Reductions in gasoline consumption are assumed to translate to reductions in petroleum fueling infrastructure (PFI) emissions including those from large refineries, gasoline storage, fueling stations, etc. It should be noted that one Case is included without PFI reductions (FCEV 38 NoTurn) to demonstrate the impact of PFI emissions relative to vehicle tailpipe emissions. The largest source of PFI emissions occur from large refinery complexes that produce a range of products in addition to motor gasoline (e.g., distillate fuels, kerosene, jet fuel). Hence, the reduction in PFI emissions is assumed to correspond only to the fraction of output attributable to motor gasoline, i.e., in the CA region in 2055 gasoline comprises 42% of net refinery production. Thus, a 90% reduction in gasoline consumption is applied as a 38% reduction in total refinery emissions. Contrastingly, emissions from additional PFI sources (e.g., fueling stations) are associated predominantly with gasoline vehicles and are reduced accordingly (i.e., taxable gasoline sales comprise 80% of total with diesel thus fueling station emissions are reduced by 80% in all Cases expect for the FCEV 38 NoTurn (http://energyalmanac.ca.gov/transportation/summary.html#fuel).

Furthermore, the FCEV 75 – HDV Case is established to provide an upper bound on the potential impact that heavy duty vehicle (HDV) trucking of hydrogen may have if it was widely used (corresponding to a 10% increase in HDV emissions).

Table 26: Impacts on emissions representing evaluated Cases

Case	Power Emissions	LDV Emissions	HDV Emissions	Refinery Emissions
FCEV 21	-21%	-90%	---	-38%
FCEV 38	-38%	-90%	---	-38%
FCEV 38 -No Turn	-38%	-90%	---	---
FCEV 75	-75%	-90%	---	-38%
FCEV 75 -HDV	-75%	-90%	+10%	-38%
FCEV 85	-85%	-90%	---	-38%

Synergies are possible between advanced alternative vehicle fueling pathways and the electric grid that can assist in maximizing GHG and pollutant emission reductions. For example, electrolysis production of hydrogen can be constructed and managed to allow the grid to absorb enhanced levels of variable generation from various wind and solar technologies. Increased renewable generation could then be substituted in place of fossil generation, notably natural gas power plants in the State. Thus, for the assessed Case it is assumed that increases in generation from renewable resources, including electrolysis production of hydrogen to support vehicle fueling, results in decreases in output and subsequent emissions from existing California generators. Table 27 displays the determined reduction in output of natural gas-fired power plants occurring from displacement by renewable resources. In the Low Renewable Case it is assumed that 205 Gigawatts (GW) of renewable resources are deployed resulting in turn down of gas generators equal to 21% of the Base Case. For the High Renewable Case 425 GW are deployed corresponding to reductions of 38%, 75%, and 85% from gas generators dependent on vehicle characteristics and fueling infrastructure. The results are generated via a modeling methodology developed to examine the impacts of deploying various advanced alternative LDVs and charging/fuel infrastructure in tandem with the feasibility of meeting the 2050 GHG goal dictated in California by Executive Order S-21-09. The methodology integrates detailed models of the California electric grid operations and the State's light duty transportation sector. Comprehensive explanation of the method and results can be found at http://www.apep.uci.edu/3/ResearchSummaries/pdf/SustainableTransportation/ElectricGrid_VehicleIntegrationGHGimpacts.pdf.

Table 27: Percent reduction in MWh output of gas-fired generators from renewable resource deployment

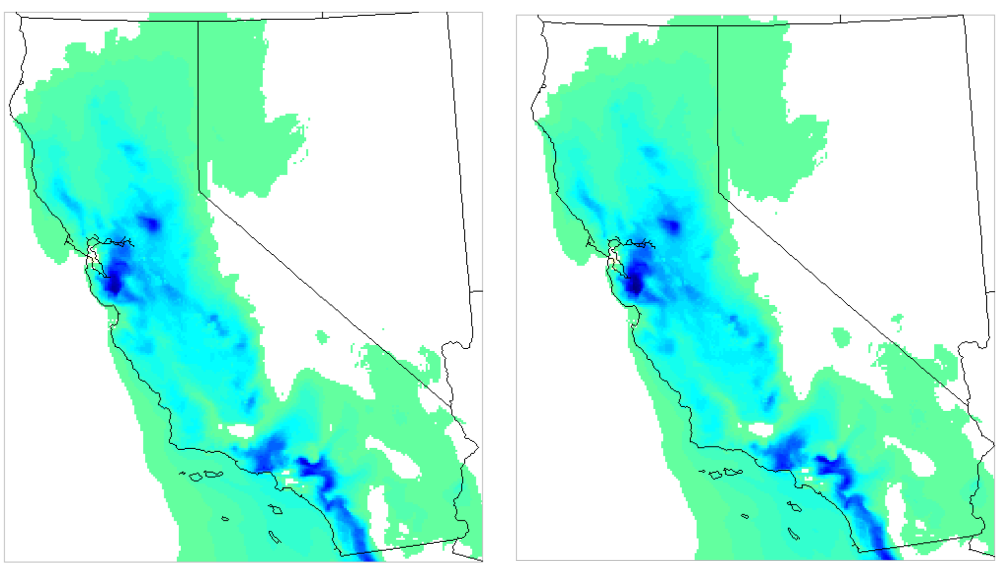
	Low Renewable Case 205 GW	High Renewable Case 425 GW
FCEV 21	0.211	---
FCEV 38	---	0.375
FCEV 75	---	0.746
FCEV 85	---	0.848

The development of hydrogen fueling infrastructure includes a diverse range of potential technological and operational options, including some that represent new emissions sources, e.g., deployment of a steam methane reformation facility. A major assumption for all Cases is vehicle fueling pathways are optimized to absorb fully variable renewable generation on the electric grid resulting in pathways that do not introduce new emission sources into the State. This is assumed to be accomplished by various methods including electrolysis of water via renewable power that is then provided by pipeline to fueling stations. While this represents a highly optimistic case for hydrogen fuel production and provision to meet demands from a large vehicle fleet, it is a reasonable potential outcome given the horizon period (2055) considered. Further, dramatic reductions required to meet California's 2050 GHG goals will require novel energy strategies and pathways in transportation and power generation that could reach the high deployment levels described in this work [37, 38]. Thus, this work can provide spanning information on AQ impacts.

5.1.2.2 FCEV Results

For all Cases evaluated, deploying FCEVs and renewable resources at high levels directly contributes to reductions in ground-level ozone and $PM_{2.5}$. As shown in Figure 60, improvements in maximum 1-hr ozone levels exceed 4 ppb for the FCEV 21, FCEV 38, FCEV 75, and FCEV 85 Cases, including in important regions of California in terms of AQ, e.g., in the SoCAB, San Francisco Bay Area, and the Central Valley. Reductions in ground-level concentrations peak in urban regions associated with high vehicle populations and the presence of PFI. These areas currently experience high levels of ground-level ozone that often exceed Federal health-based standards and contain large urban populations [39]. Thus, improvements in these areas are desirable to the State in terms of mitigating deleterious human health outcomes from air pollution.

Impacts do not vary considerably amongst the Cases and emphasize the more significant contribution of direct vehicle and PFI emissions reductions to ozone concentration reductions compared to those from the power sector. Peak reductions observed in all of the Cases for ozone and $PM_{2.5}$ are displayed in Table 28. As can be seen, when emissions from the power sector are reduced while maintaining emissions reductions from LDVs and PFI increased ozone benefits are achieved (see Figure 60). However, the magnitude is minor relative to the magnitude of the overall impact. For example, peak reductions increase from -4.31 ppb in the FCEV 21 Case (21% power sector emissions reduction) to -4.75 ppb in the FCEV 85 (85% power sector emissions reduction).



(a) (b)

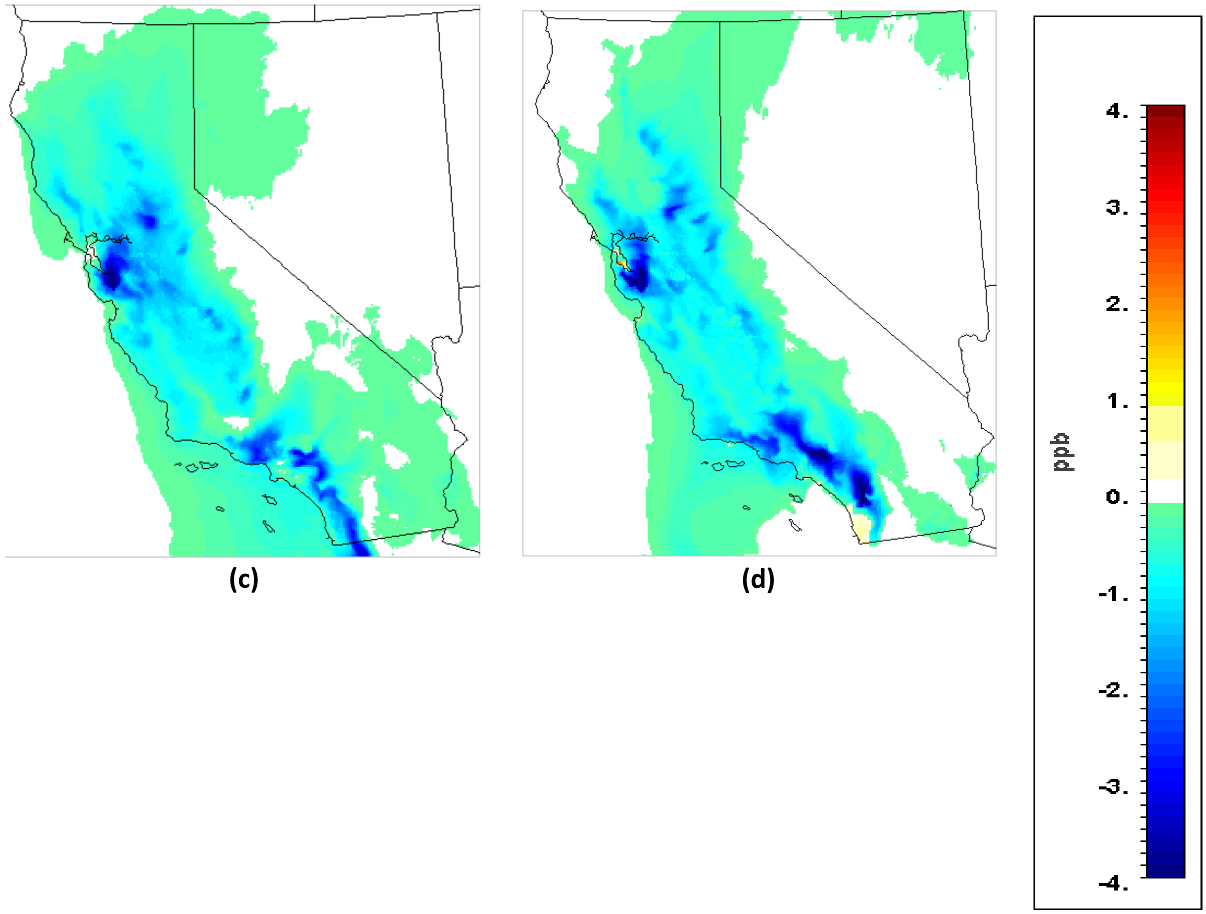


Figure 60: Differences in maximum 1-hr ozone from the Base Case for the (a) FCEV 21, (b) FCEV 38, (c) FCEV 75, and (d) FCEV 85 cases in California

Reductions in regional 24-hour $PM_{2.5}$ levels for all of the Cases are displayed in Figure 61. Ambient $PM_{2.5}$ concentrations in CA are reduced by over 4 μg/m^3 in some locations for all Cases. Notable regions of impact correspond to those for ozone, i.e., the SoCAB, S.F. Bay Area, and the Central Valley. Additionally, reductions show correspondence with the locations of major petroleum refinery complexes including from locations in Long Beach, Los Angeles, and Santa Maria. Also, similar to the ozone results, only minor changes occur amongst the Cases despite significant variance in power sector emissions (i.e., peak reductions of -4.17 ppb in the FCEV 21 Case relative to -4.20 ppb for the FCEV 85 Case). Thus, these particulate matter results demonstrate that the dominant contributor to $PM_{2.5}$ impacts originates from direct vehicle and PFI emissions.

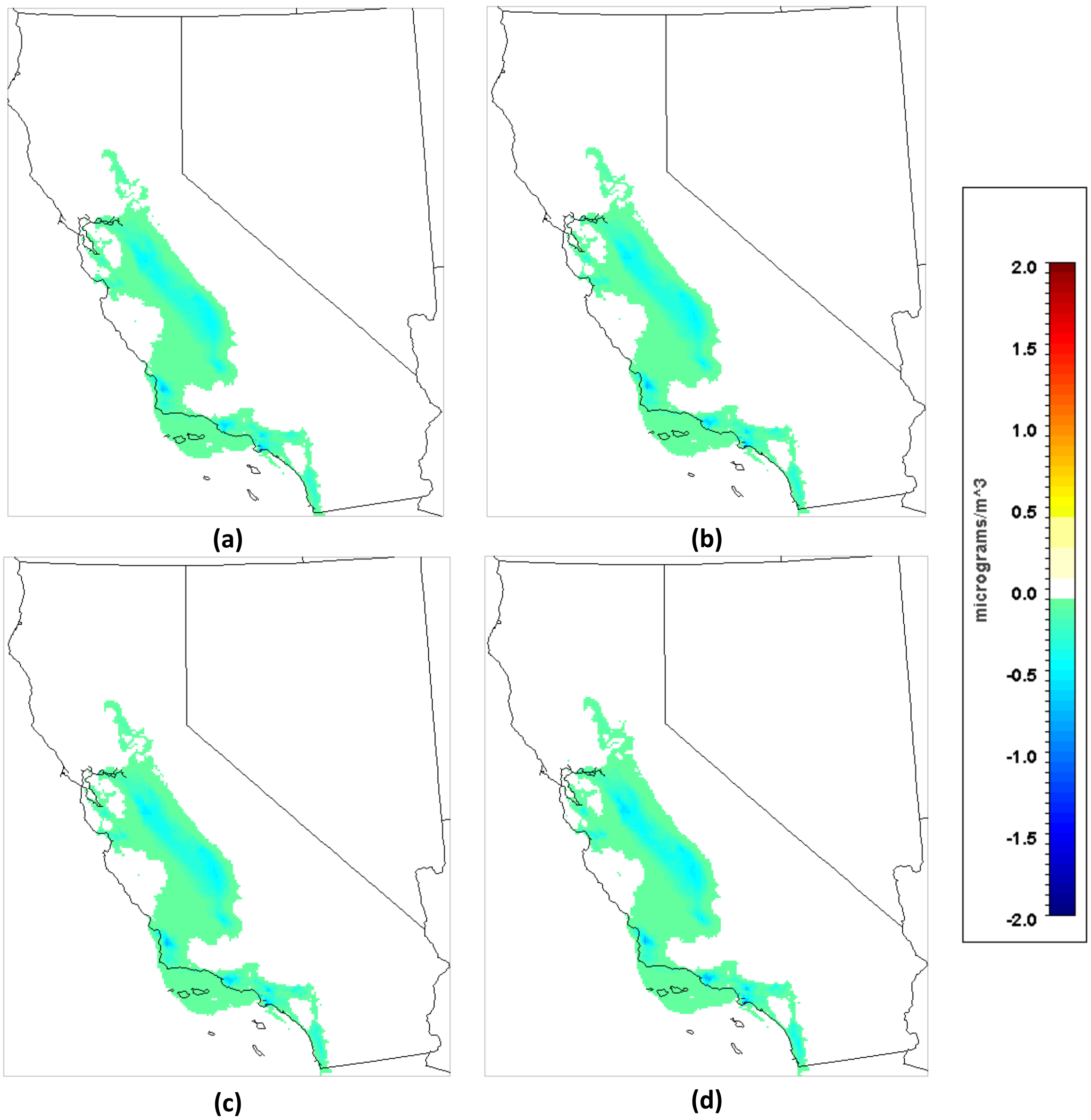


Figure 61: Differences in maximum 24-hr PM2.5 from the Base Case for the (a) FCEV 21, (b) FCEV 38, (c) FCEV 75, and (d) FCEV 85 cases in California

To provide spatial information on the impacts attributable to the power sector, Figure 62 shows a difference plot for ozone and $PM_{2.5}$ between the FCEV 21 and FCEV 85 Cases. As all other emission source perturbations remain constant, differences show the effect of varying power sector emissions, i.e., -21% vs. -85%. Peak differences between the Cases reach -1.78 ppb and -0.17 μg/m³. Relatively speaking, differences in ozone impacts are larger than $PM_{2.5}$ impacts in terms of magnitude. The spatial distribution of ozone impacts corresponds to generator locations and includes some impacts in regions discussed previously (e.g., the SF Bay Area) as well as some regions that do not experience the highest background levels (e.g., northern portions of the State, San Diego).

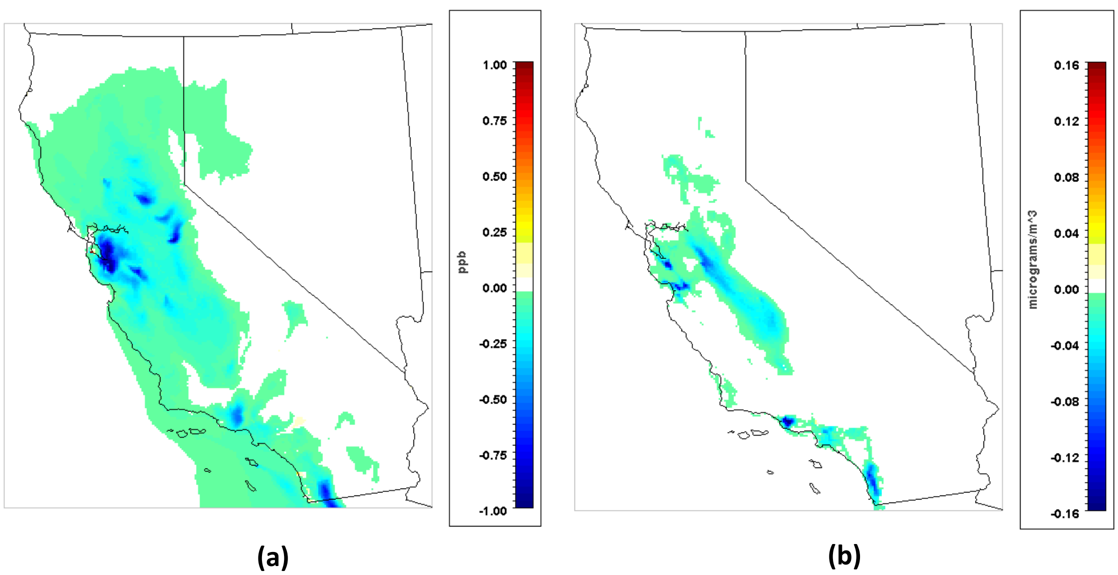


Figure 62: Impacts on (a) 1-hr ozone, and (b) 24-hr PM2.5 for the FCEV 85 from the FCEV 21 Case

Impacts of PFI Emissions Turn-down

In order to assess the impact of PFI emissions on AQ the FCEV 38 No Turn Case is compared relative to the FCEV 38 Case. As all other emission perturbations remain constant, i.e., direct vehicle and power plant reductions, the difference in ground-level concentrations is attributed solely to the difference from reducing PFI emissions associated with gasoline production and distribution. Additionally, the FCEV 38 No Turn Case is compared to the Base Case to provide information regarding the contribution of PFI emissions to overall observed impacts.

Figure 63 presents the data in terms of a difference plot between the FCEV 38 vs. the FCEV 38 No Turndown such that the results demonstrate the enhanced reduction from the PFI reductions present in the FCEV 38 Case. Ozone impacts peak at -1.28 ppb in the SF Bay Area, SoCAB and Bakersfield areas and are attributable to the presence of large refinery complexes. Additional benefits of a lesser magnitude occur in other regions of the Central Valley and San Diego. Impacts on ozone from PFI emissions are important in regards to both magnitude and spatial distribution. Reductions exceeding 1 ppb in several areas are prominent as peak impacts in the FCEV 38 Case relative to the Base Case exceeded 4 ppb. Further, the spatial distributions of reductions are important due to the high baseline concentrations experienced in those areas.

The impacts of PFI emissions are significant for $PM_{2.5}$. Reductions in concentrations between the Cases peak at 4.75 $\mu g/m^3$ in an area downwind of a large refinery located in Santa Maria. Additional lesser benefits occur in the SoCAB, SF Bay Area, and Central Valley. Table 11 lists reductions in $PM_{2.5}$ from the Base Case for the FCEV 38 No Turn Case peaking at -0.55 $\mu g/m^3$; while all other Cases containing PFI turn-down achieve reductions greater than 4 $\mu g/m^3$. Therefore, the largest peak reductions in ground-level concentrations of $PM_{2.5}$ from FCEV deployment result from the assumed reduction in PFI output and emissions. It should be noted that information regarding spatial distribution of impacts is not considered solely from comparing peak impacts. For example, the difference plot presented in Figure 4 shows that the peak impacts on $PM_{2.5}$ occur over a relatively small area with moderate reductions visible across a greater expanse. Nonetheless, results highlight the importance of considering emissions from PFI in reaching maximum AQ benefits from the deployment of advanced alternative LDV technologies.

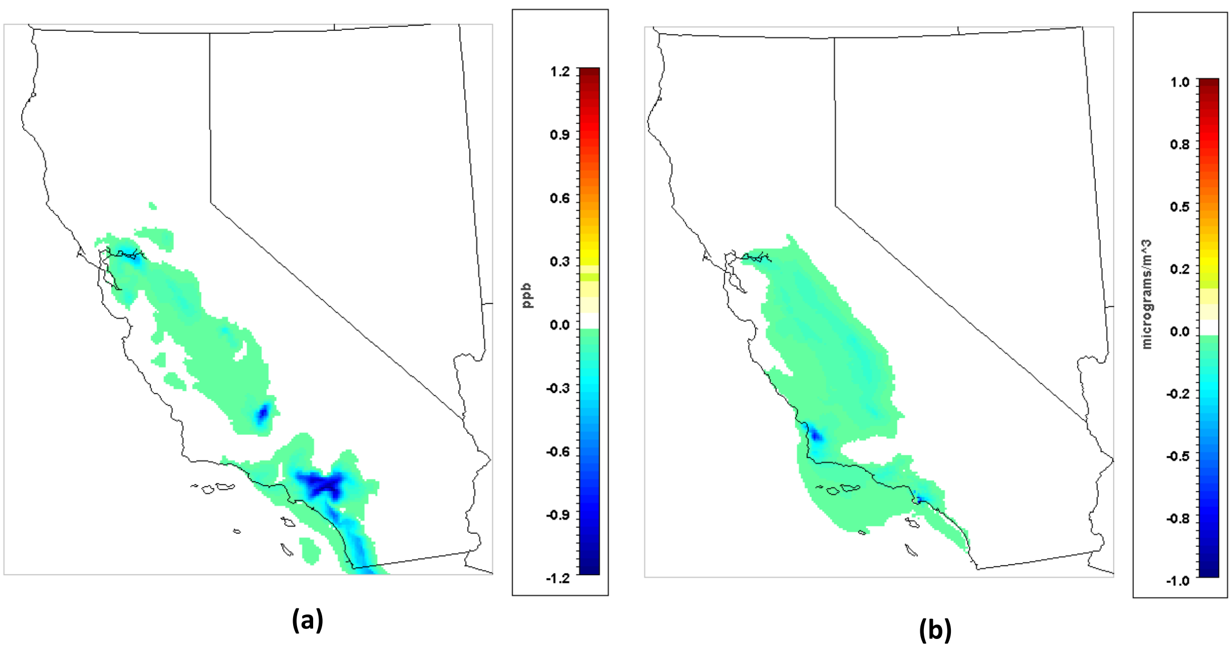
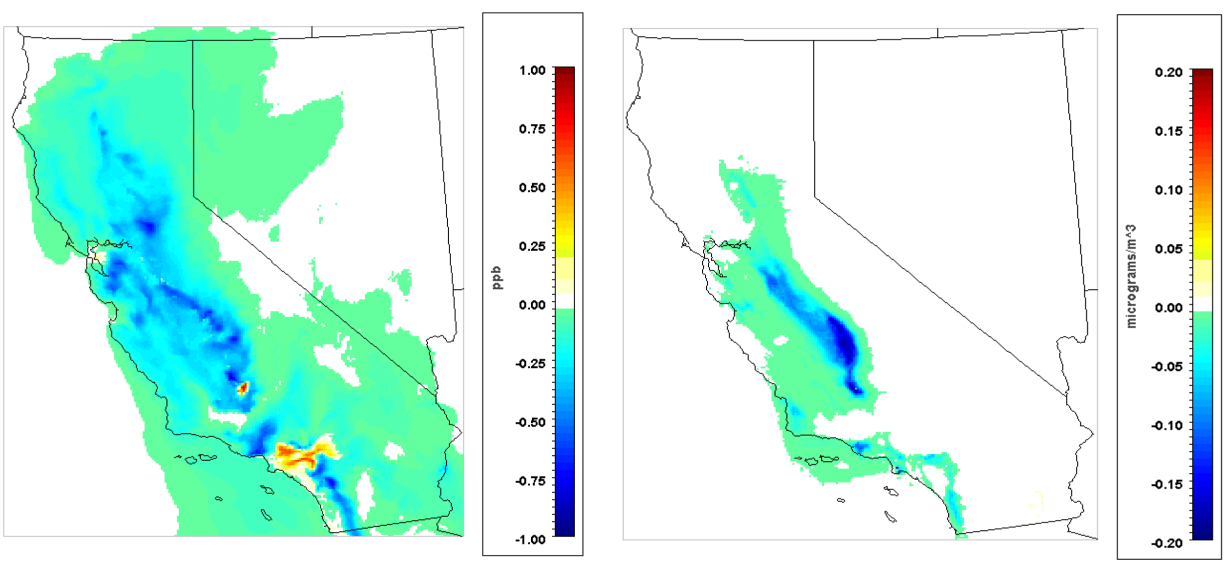


Figure 63: Impacts on (a) 1-hr ozone, and (b) 24-hr PM2.5 for the FCEV 38 from the FCEV 38 No Turn Case

Impacts of HDV Emission Increases

If HDV were used to truck hydrogen throughout society there would be significant emissions associated with such trucking. Figure 64 displays the impacts on ozone and $PM_{2.5}$ for the FCEV 75 Case relative to the FCEV 75 HDV Case. An increase state-wide of 10% in all HDV tail-pipe emissions (a possible upper-bound for trucking emissions for hydrogen delivery) results in ground-level ozone and $PM_{2.5}$ concentration increases of approximately 1 ppb and 0.3 μg/m³, respectively. Additionally, the largest impacts are co-located in important areas of the State in terms of AQ as previously indicated. Furthermore, when the FCEV 75 HDV Case is compared to the Base Case peak reductions of -3.88 μg/m³ (vs. -4.19 μg/m³ for the FCEV 75 Case) demonstrating that the increased HDV traffic to distribute fuel can erode some of the AQ benefits of FCEVs (Table 28). While the 10% increase in HDV emissions modeled here does not correspond directly to any quantified hydrogen amount or spatial distribution of trucking routes, the Case represents a spanning outcome to be interpreted as an upper bound for AQ impacts. Accordingly, results demonstrate the importance to AQ of constructing hydrogen infrastructure that seeks better distribution pathways compared to HDV truck delivery (e.g., pipeline delivery, on-site generation) rather than providing predictive information.



(a) **(b)**

Figure 64: Impacts on (a) 1-hr ozone, and (b) 24-hr PM2.5 for the FCEV 75 from the FCEV 75 HDV Case

5.1.2.3 CA FCEV and HDV Deployment Scenarios Summary

The deployment of high levels of FCEVs (i.e., 90% LDV sector penetration) in tandem with renewable resources achieves significant benefits to AQ in California, including reductions in ground-level concentrations greater than 4 ppb ozone and 4 μg/m^3 $PM_{2.5}$. The greatest AQ impacts occur in key regions of the state where high urban populations are located and where poor AQ conditions are already occurring including the SoCAB, SF Bay Area, and the Central Valley. The impacts of not reducing PFI emissions corresponding to reduced motor gasoline production and distribution are demonstrated in lesser peak reductions, particularly for $PM_{2.5}$ (-0.55 μg/m^3 vs. 4.18 μg/m^3). Similarly, increasing HDV emissions lowers peak reductions for both ozone and $PM_{2.5}$. Despite reduced magnitudes, both the FCEV 38 No Turn (which retains PFI emissions) and FCEV 75 HDV (which introduces new HDV emissions for hydrogen trucking) Cases achieve overall AQ benefits relative to the Base Case.

Table 28: Peak reductions in ground-level concentrations of ozone and $PM_{2.5}$

Case	Δ 1-hr Ozone [ppb]	Δ 24-hr $PM_{2.5}$ [μg/m^3]
FCEV 21	-4.31	-4.17
FCEV 38	-4.37	-4.18
FCEV 38 No Turn	-4.13	-0.55
FCEV 75	-4.58	-4.19
FCEV 75 HDV	-4.05	-3.88
FCEV 85	-4.75	-4.20

The AQ benefits of all of the Cases are largely driven by vehicle and PFI emissions with moderate changes attributed to power sector impacts, e.g., the difference in peak ozone and $PM_{2.5}$ between a 21% and 85% reduction in generator emissions are -1.71 ppb and 0.17 $\mu g/m^3$. Additionally, peak impacts are generally located in less populated areas relative to those from vehicles. The composition of the California power generation sector directly impacts results as California has a lower emitting mix of electric power generators relative to other regions of the U.S. including a near complete lack of coal power generation. Thus, the benefit of reducing power sector emissions may achieve significantly higher AQ benefits in regions that deploy large coal power generation fleets.

Emission impacts from PFI supporting motor gasoline production and distribution are important factors that affect ozone and $PM_{2.5}$ in California. Peak reductions of 1.28 ppb and 4.75 $\mu g/m^3$ between Cases with and without turn down are predicted. Impacts on $PM_{2.5}$ are particularly notable as being the major driver of peak impacts for all Cases. Moreover, improvements in AQ occur in regions of the State currently experiencing poor AQ, which heightens the importance of the results, such as ozone improvements in Bakersfield and SoCAB. However, while programs and policies are in place to promote the deployment of alternative, low or zero-emitting LDV technologies that will concurrently reduce gasoline consumption, e.g., California's Zero Emission Vehicle Program, it is unknown if emission will also decrease from PFI. A potential outcome is that gasoline production at California refineries may remain constant with excess product exported. Thus, designing and deploying LDV adoption strategies seeking maximum AQ and GHG benefits should consider also reductions from sources associated with gasoline production and storage; most notably large petroleum refinery complexes.

The Cases evaluated in this section represent a positive outcome for FCEV technology marketplace success including a penetration of 90% in the LDV sector. Assumptions regarding hydrogen fuel infrastructure are highly optimistic in that it is assumed that all of the hydrogen production is renewable and facilitated by increased renewable power penetrations in the future. Thus, results from this work should be considered as an upper bound for the AQ benefits of FCEVs in California.

5.1.3 CA County Level FCEV Deployment

For the first time this work uses advanced atmospheric modeling to examine how deploying FCEVs in CA counties that are expected to be early adopters of novel vehicle technologies may impact the spatial and temporal distribution of primary and secondary pollutants in the basin. Although previous studies have evaluated the emissions [18, 21] or AQ [27, 28] impacts of FCEVs, few have utilized detailed 3D Eulerian AQ models to account for spatial and temporal emissions perturbations and atmospheric chemistry and transport processes to produce ground level ozone and particulate matter distributions in CA. A central question of this work is whether FCEV deployment will achieve environmental benefits amongst CA communities irrespective of socio-economic factors impacting deployment levels, e.g., county level mean income. Baseline AQ in the horizon year (2055) is established accounting for changes in various emission drivers, including demand growth in economic sectors, efficiency improvements, and utilized technologies and fuels according to a business-as-usual progression. Cases are developed for FCEV deployment accounting for spatial and temporal distribution of fundamental sources to evaluate impacts on ambient pollutant concentrations from emission perturbations, including ozone and $PM_{2.5}$.

5.1.3.1 Case Development

In order to assess resulting perturbations in direct and secondary atmospheric pollutants a set of Cases representing FCEV deployment in LDVs are developed and analyzed at various penetration levels in select CA counties in the year 2055. The counties are chosen based on plug-in hybrid and battery electric vehicle new vehicle registration data, as well as mean income data, as it is likely to correspond to early adoption of FCEVs (Table 29). Additionally, selected counties tend to be located in CA air basins that currently experience existing AQ challenges, including non-compliance with federally mandated levels of ozone and PM. Case assessment comprises the development of spatially and temporally resolved emission fields appropriately accounting for all mobile and stationary source perturbations followed by simulations of atmospheric chemistry and transport. Resulting output is assessed for alterations of ground-level maximum 8 hour (8-hr) average ozone and 24 hour (24-hr) average $PM_{2.5}$ relative to the baseline gasoline dominated vehicle Case.

Table 29: Selected counties for study for early adoption of FCEVs.

County	Population [persons]	PHEVs [vehicles]	BEVs	Combined	Mean Income [dollars]	CA Air Basin
Los Angeles	9,999,897	6385	4737	11121	81,416	SoCAB
Santa Clara	1,873,242	2477	3828	6305	120,718	SF Bay
Orange	3,110,069	3109	1898	5007	101134	SoCAB
San Diego	3,141,267	1329	2343	3672	84889	San Diego
Alameda	1,554,285	1267	1619	2887	96982	SF Bay

San Mateo	717,063	678	1438	2116	126129	SF Bay
Contra Costa	973,517	783	741	1523	106018	SF Bay

Developed Cases encompass penetrations of 1%, 10%, 30%, 50% and 100% of the total LDV fleet in counties in Table 29 in 2055 and are labeled accordingly, i.e., FCEV 1, FCEV 50, etc. Correspondingly, direct (i.e., tailpipe) LDV emissions are reduced fleet-wide in the counties of deployment across all road way types. Further, declines in gasoline consumption are assumed to translate to reductions in baseline PFI emissions including those from refineries, gasoline storage, fueling stations, etc. The largest single source of PFI emissions occur from large refinery complexes which produce a range of products in addition to motor gasoline (e.g., distillate fuels, kerosene, jet fuel). Hence, the reduction in PFI emissions is assumed to correspond only to the fraction of output attributable to motor gasoline, i.e., in the CA region in 2055 gasoline comprises 52% of net refinery production. Thus, a 100% reduction in gasoline consumption is applied as a 52% reduction in total refinery emissions. Contrastingly, emissions from additional PFI sources (e.g., evaporative emissions from fueling stations) are predominantly associated with gasoline vehicles and are reduced accordingly. It should be noted that one FCEV Case is included without PFI reductions (FCEV 50 No Turn) to provide comparison of the impact of PFI emissions relative to vehicle tailpipe emissions. Finally, increases in renewable resources in support of electrolysis production of hydrogen to support vehicle fueling results in decreases from power plant emissions which are also accounted for. One Case, the FCEV 50 No Electric (NoE) Case, was assessed with power plant emissions held constant to the baseline to provide insight into the comparative impacts relative to other sources.

In addition, Cases are analyzed for the removal of emissions from the heavy duty vehicle (HDV) fleet in the same counties at 1%, 50%, and 100% to facilitate comparison of the AQ impacts of addressing non-LDV transportation sources, i.e., HDV 1, HDV 50, and HDV 100 Cases. It should be noted that HDV Cases do not include reductions in emissions from power generation or PFI and account for tail pipe reductions only. However, to facilitate comparison with FCEV a Case (HDV 50 Turn) with the corresponding PFI and power sector emissions reductions from the FCEV 50 case is included. Table 30 displays the emission perturbations associated with both FCEV and HDV Cases considered in this section.

Table 30: Emission reductions from the Reference Case for notable sources in analyzed Cases in 2055.

Case	Impact on Power Emissions	Impact on LDV Emissions	Impact on HDV Emissions	Impact on Refinery Emissions
FCEV 1	-0%	-1%	---	-1%
FCEV 10	-0%	-10%	---	-5%
FCEV 30	-0%	-30%	---	
FCEV 50	-35%	-50%	---	-26%
FCEV 50 No Turn	-35%	-50%	---	-0%
FCEV 50 NoE	-0%	-50%	---	-26%
FCEV 100	-70%	-100%	---	-52%
HDV 1	---	---	-1%	---
HDV 50	---	---	-50%	---
HDV 50 Turn	-35%	---	-50%	-26%
HDV 100	---	---	-100%	---

5.1.3.2 FCEV Results

For all Cases evaluated, the use of FCEVs in selected counties contributes to improvements in ground-level ozone and $PM_{2.5}$ as displayed in Figure 65 for the FCEV 1, FCEV 50, and FCEV 100 Cases relative to the Reference Case. Concentrations of pollutants for a typical episode in 2055 are modeled using the Reference Case assumptions for all economic sectors and expected changes in stationary and mobile sources between now and 2055 are accounted for. The use of FCEVs as assumed in this work will further reduce pollutant emissions from various sources and consequently when pollutant formation in FCEV Cases is compared to that of the Reference Case improvements in ground-level ozone and $PM_{2.5}$ are expected.

With respect to peak ozone, improvements occur in the S.F. Bay Area and SoCAB including locations of peak baseline concentrations including San Bernardino and Riverside Counties. These areas currently experience high levels of ground-level ozone that often exceed Federal health-based standards and contain large urban populations [56]. Thus, improvements in these areas are desirable to the State in terms of mitigating deleterious human health outcomes from air pollution. Reductions are also visible in the Central Valley for Cases with higher FCEV penetrations including the FCEV 50 and FCEV 100 Cases. Peak impacts for all Cases are listed in Table 31.

Quantitatively, maximum reductions in ozone range from -0.27 ppb in the FCEV 1 Case to -3.17 in the FCEV 100 Case. The locations of the most pronounced impacts occur as a result of the displacement of emissions from conventional LDVs, petroleum refineries, and additional PFI emissions sources. Large urban air sheds such as the SoCAB and S.F. Bay Area contain large numbers of such sources and experience significant reductions in total emissions which occur in areas with high importance to secondary pollutant formation. However, the largest reductions occur some distance from the highest levels of emission reduction as a result of the temporal period required for the formation of ozone resulting in the transport of precursor emissions[58]. For example, the largest ozone benefits occur in the northeastern section of the SoCAB including portions of San Bernardino and Riverside County while peak emission reductions occur in Los Angeles County. In addition, to the impacts of displaced emissions from power generators can be seen as plumes of reduction extending from point sources northeast of the SF Bay Area in the FCEV 50 Case.

Reductions in emissions also improve ambient $PM_{2.5}$ concentrations in CA in regions associated with both high LDV populations and/or the presence of large petroleum refineries including the SoCAB, San Diego, SF Bay Area, and Central Valley. LDVs emit $PM_{2.5}$ directly from tailpipes [59] as well as NO_x emissions that contribute to the formation of secondary $PM_{2.5}$. Thus, reducing emissions from both sources results in improvements in ground-level $PM_{2.5}$ concentrations. Quantitatively, peak impacts range from -0.05 to -5.82 $\mu g/m^3$ for the FCEV1 and FCEV 100 Cases, respectively. Relative to ozone, $PM_{2.5}$ reductions occur with increased localization to source emissions as evident in reductions corresponding with locations of major petroleum refinery complexes in Long Beach, Los Angeles, and Santa Maria. Improvements in ground-level $PM_{2.5}$ are attributable to both the reduction of directly emitted PM and secondary PM, which has been shown to largely form from NO_x conversion to nitrate aerosol in Southern CA[2].

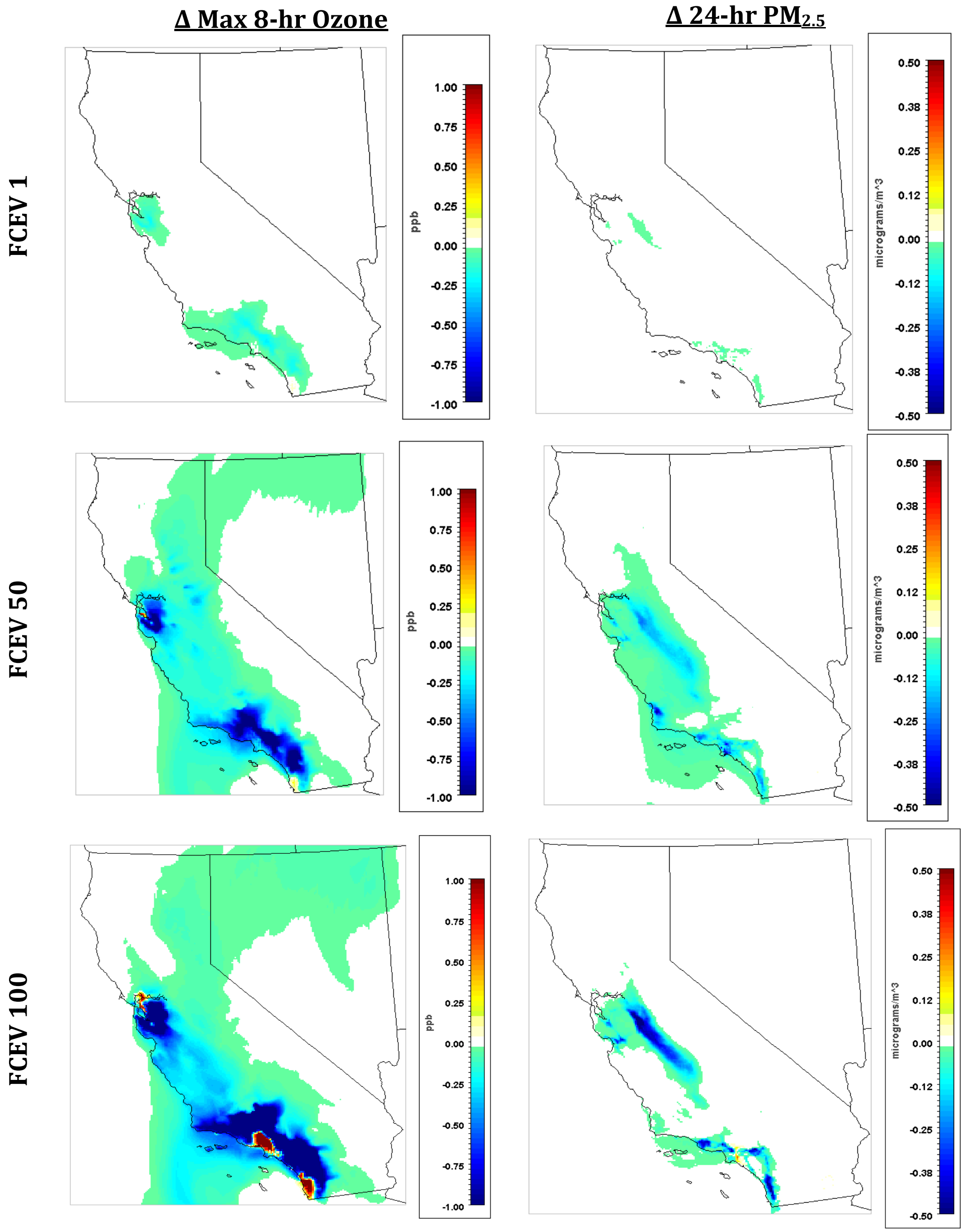
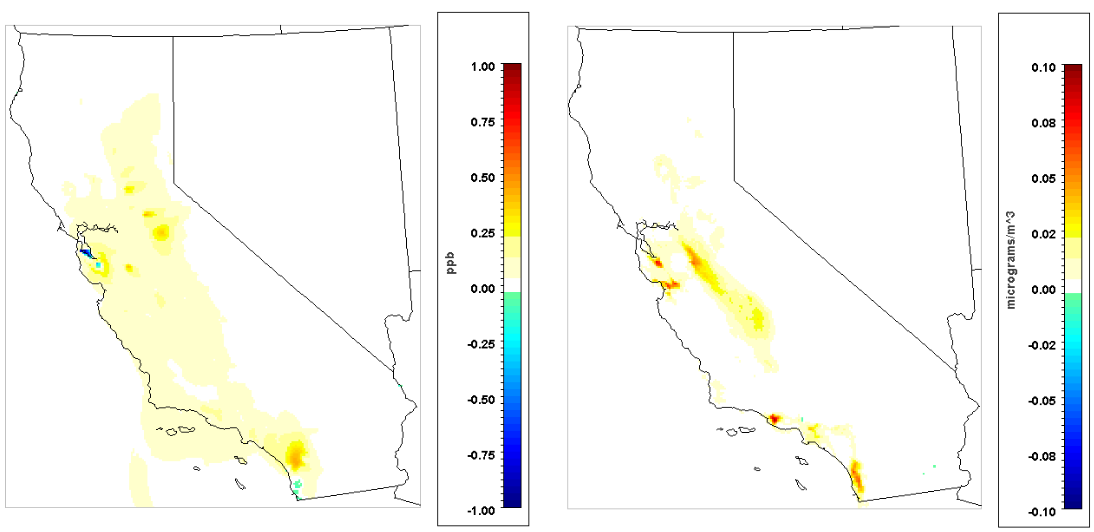


Figure 65: Predicted differences in ground-level maximum 8-hr average ozone and 24-h average $PM_{2.5}$ between the FCEV 1, FCEV 50 and FCEV 100 Cases and the Reference Case.

FCEV No Electricity Reduction Case

Figure 66 displays difference plots for ozone and $PM_{2.5}$ for the FCEV 50 No Electric Case and the FCEV 50 Case to show the impact of the assumed emission reductions from the power sector. Increases in concentrations result from maintaining baseline power sector emission profiles in place of the reduction assumed in the FCEV 50 Case (-35%) as no other emission source is altered. Generally, impacts are moderate with peak increases for maximum 8-h ozone exceeding 1 ppb, although most impacts are less than 0.25 ppb. Similarly, small differences in 24-h $PM_{2.5}$ are observed that peak at -0.09 μg/m³. Given the significant reduction in total sector emissions and the moderate observed AQ perturbations; electricity generators can be considered a minor driver of the total impacts of Cases. This is expected given the mix of resources providing power to the California grid, which includes high levels of low PM-emitting natural gas, renewables, and nuclear [60].

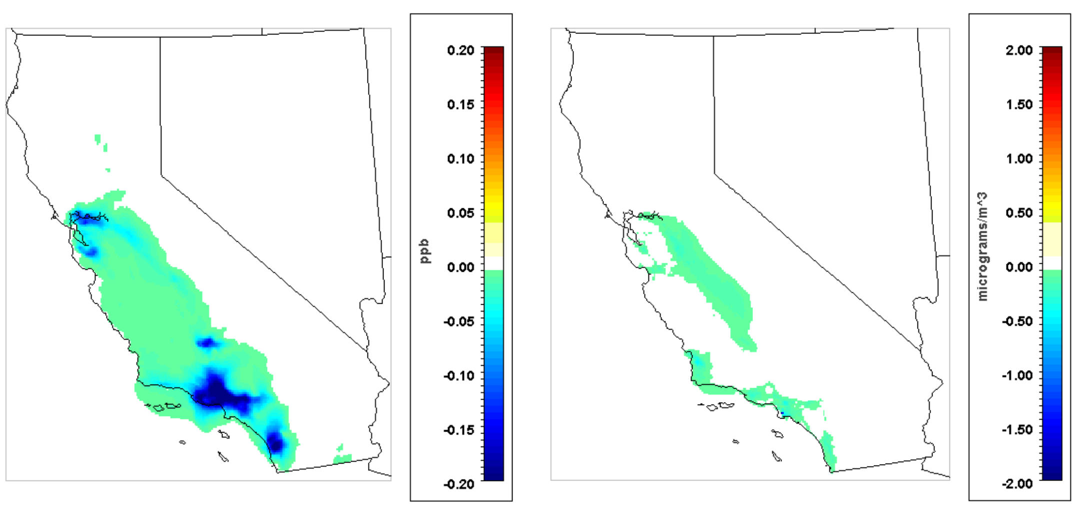


(a) (b)

Figure 66: Predicted differences in ground-level (a) maximum 8-hr average ozone and (b) 24-h average $PM_{2.5}$ between the FCEV 50 No Electric Case and FCEV 50 Case.

FCEV No Turndown Case

Figure 67 presents the data in terms of a difference plot between the FCEV 50 vs. the FCEV 50 No Turn Cases to show impacts from the PFI emission reductions present in the FCEV 50 Case. As emissions from all other sources remain constant (i.e., direct vehicle and power plant reductions), variance in ground-level concentrations can be attributed solely to gasoline production, distribution and storage. In particular, PFI emissions contribute a notable fraction of the total $PM_{2.5}$ benefits observed in the FCEV 50 Case. Localized reductions in concentrations between the cases exceed 2 μg/m^3, including areas directly adjacent with the Long Beach-area refinery complexes. Smaller reductions also occur in the SF Bay Area and Central Valley. Ozone impacts from PFI emissions are lesser with peak differences equivalent to -0.2 ppb in the SF Bay, SoCAB, San Diego and Bakersfield and associated with the presence of large refinery complexes in those regions. Additional reductions of lesser magnitude occur across large areas of the State and likely result from the distributed impacts of fueling stations.



(a) **(b)**

Figure 67: Maximum difference in (a) 8-hr ozone and (b) 24-h $PM_{2.5}$ between the FCEV 50 and FCEV 50 No Turn Cases.

Table 31:Peak reduction in ground-level ozone and $PM_{2.5}$ from the reference case for FCEV Cases.

Case	Δ 8-hr Ozone [ppb]	Δ 24-hr $PM_{2.5}$ [μg/m³]
FCEV 1	-0.27	-0.05
FCEV 30	-0.92	-3.22
FCEV 50	-1.62	-2.82
FCEV 50 No Turn	-1.48	-0.30
FCEV 50 No Electric	-1.52	-2.80
FCEV 100	-3.17	-5.82

HDV Impacts

Removing emissions from HDVs in counties under study results in improvements in AQ with regards to both ozone and $PM_{2.5}$. It should be noted that the only emission source altered in the HDV Cases is direct vehicle emissions and comparison with FCEV Cases should consider this caveat. Removing 1% of HDV activity in counties of study yield minor AQ improvements similar in spatial dimension to those from FCEVs, i.e., less than -0.02 ppb and -0.01 μg/m³. Figure 68 shows impacts on ozone and $PM_{2.5}$ for the HDV 50 and 100 Cases relative to the Reference Case. Reductions in ozone peak are approximately 2 and 4 ppb while reductions in $PM_{2.5}$ reach -0.31 and -0.62 μg/m³ in the HDV 50 and HDV 100 Cases, respectively. Reductions in ground-level ozone concentrations are wide-spread through the State, including areas distant from the counties of emission displacement. Despite a lack of emission reduction from PFI and power generators the HDV 50 and HDV 100 Cases achieves comparable or enhanced ozone benefits relative to corresponding FCEV Cases (i.e., -4.23 ppb in the HDV 100 and -3.17 in the FCEV 100). These results highlight the importance of HDV emissions to regional ozone burdens in 2055. Relative to the FCEV Cases, lesser impacts on PM are observed and attributable to the lack of refinery turn down in the bulk of the HDV Cases.

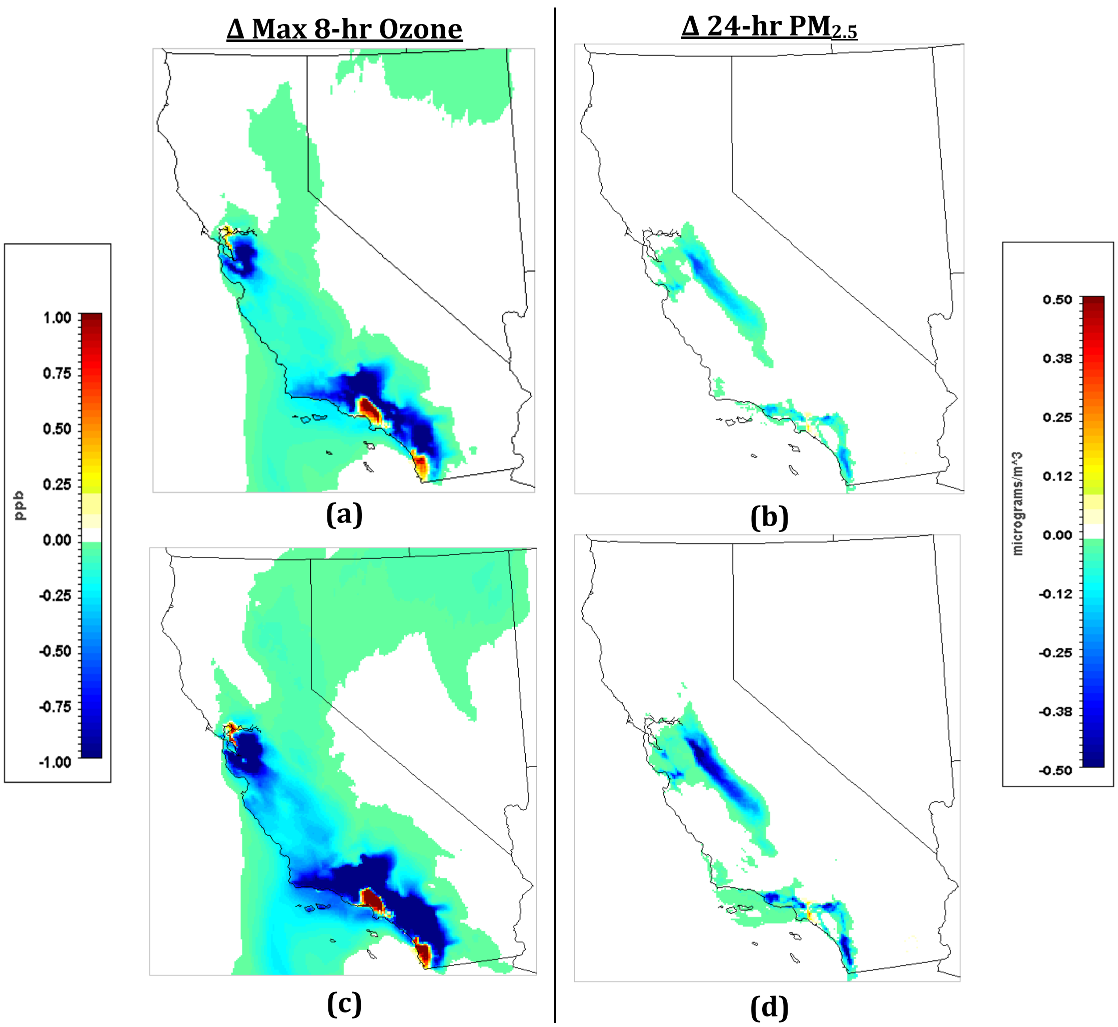
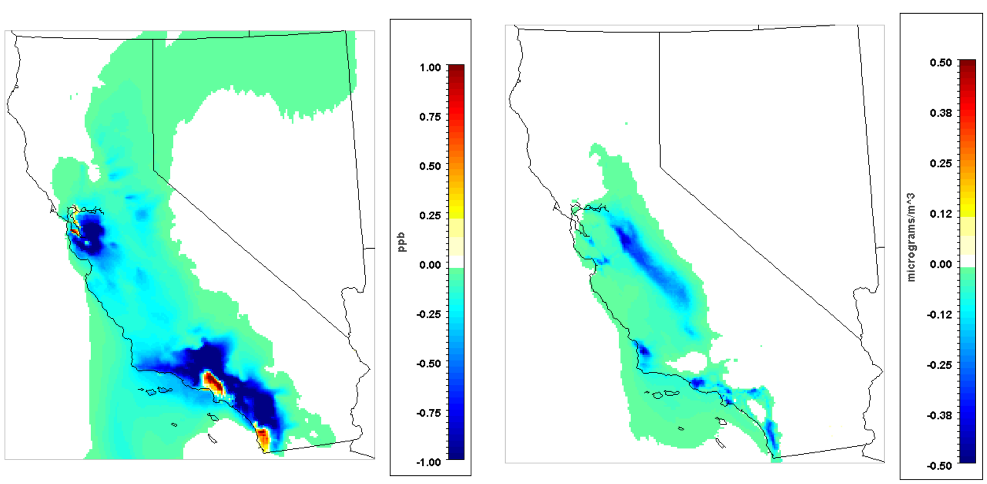


Figure 68: Peak impacts on 8-hr ozone and 24-h $PM_{2.5}$ for the HDV 50 ((a) and (b)) and HDV 100 ((c) and (d)) Cases

HDV Turndown Case

With similarity to the FCEV Cases involving PFI emissions, the HDV 50 Turn Case was constructed to evaluate the impacts of HDV tailpipe emissions in concert with petroleum fuel production. All appropriate PFI and power sector emissions are reduced accordingly with the FCEV 50 Case as the relative fractions of diesel fuel for HDVs is unknown and reductions were chosen to facilitate a direct comparison with the FCEV 50 Case in terms of LDV and HDV emissions. While quantifiable reductions do not correspond directly with HDV fuel production and consumption, the HDV 50 Turn Case demonstrates the impacts of HDV tailpipe and PFI emissions. Figure 69 shows the resulting perturbation in ozone and $PM_{2.5}$ for the HDV 50 Turn Case relative to the Reference Case with peak impacts exceeding -2 ppb and -2.8 $\mu g/m^3$. Relative to the HDV 50 Case, benefits to $PM_{2.5}$ are significantly enhanced by the addition of PFI reductions (i.e., peak reductions of -2.85 vs. -0.31 $\mu g/m^3$). Additional ozone reductions also occur from the removal of power plant emissions in the northern central area of the State with similarity to the FCEV 50 Case.



(a) (b)

Figure 69: Maximum difference in (a) 8-hr ozone and (b) 24-h $PM_{2.5}$ for the HDV 50 Turn and Reference Case

Table 32: Maximum reduction in ground-level ozone and $PM_{2.5}$ from the Reference Case for HDV Cases

Case	Δ 8-hr Ozone [ppb]	Δ 24-hr $PM_{2.5}$ [μg/m³]
HDV 1	-0.02	-0.01
HDV 50	-1.98	-0.31
HDV 50 Turn	-2.32	-2.85
HDV 100	-4.23	-0.62

Summary of HDV Scenarios

Meeting a large fraction of the LDV fleet with FCEVs (e.g., 50-100%) in selected counties in tandem with high renewable penetration of the power grid achieves AQ benefits in CA, including reductions in ground-level ozone and $PM_{2.5}$, e.g., reductions in maximum 8-hr average ozone and 24-h $PM_{2.5}$ exceed 3 ppb and 5 μg/m³ for complete LDV fleet penetration by FCEVs**.** FCEV impacts on ozone tend to be driven by direct vehicle tail pipe reductions while $PM_{2.5}$ levels are affected most by reductions occurring in PFI emissions. Emission impacts from the power sector have a lesser degree of impact and reflect the relatively clean nature of CA's power grid. While FCEV deployment achieves AQ benefits, the magnitude of observed concentration reductions can be considered moderate when viewed in full context. Cases experiencing notable AQ improvements represent a considerably successful outcome for FCEV technology, i.e., a penetration of 50-100% in the LDV sector. Further, assumptions regarding hydrogen fuel infrastructure are highly optimistic in that hydrogen production is renewable and does not incur an emissions introduction from new hydrogen production facilities. Thus, results from this work very much represents an upper bound for the AQ benefits of FCEVs in CA.

The results show the complex relationship between the formation and fate of atmospheric pollutants and the spatial distribution of direct emissions, most notably in regards to ground-level ozone. Due to the temporal period required for ozone production from precursor emissions the spatial distributions of reductions are not necessarily directly correlated with sites of emission subtraction from vehicle operation and others, e.g., maximum ozone reductions occur in Riverside and San Bernardino Counties from emission reductions in Los Angeles and Orange Counties. However, as the highest background levels of ozone occur in the affected areas which support large urban populations the observed ozone impacts are beneficial. Similarly, $PM_{2.5}$ benefits were most prominent in areas adjacent to large refinery complexes which may not necessarily located in counties of FCEV deployment. This has importance to the State in terms of Environmental Justice as questions have been raised regarding the fairness of deploying FCEVs in affluent areas.

Increases in efficiency and improved pollutant control technologies in LDVs result in significantly lower-emitting fleet in 2055 relative to current in the Reference Case, despite an increase in vehicle-miles-traveled. In contrast, some non-LDV transportation technologies can have a proportionately larger impact on AQ in 2055, including HDVs. Demonstrating this, reducing HDV emissions in the early adoption counties achieves similar or enhanced AQ benefits relative to the FCEV Cases despite the lack of reductions from PFI and power generation. Further, when similar reductions are assumed (i.e., the HDV 50 Turn) the corresponding reductions in ozone and $PM_{2.5}$ are greater both quantitatively and spatially. Thus, incorporating fuel cell technologies in all transportation sub-sectors, including HDVs, can help mitigate current AQ challenges. Thus, all transportation technologies should be considered for fuel cell incorporation for propulsion power moving forward seeking the attainment of AQ improvement.

Emission impacts from PFI supporting motor gasoline production and distribution are important factors affecting ozone and $PM_{2.5}$ in CA in 2055. Impacts on $PM_{2.5}$ are particularly notable as being the driver of peak impacts for both FCEV and HDV Cases. Moreover, improvements occur in regions of the State currently experiencing poor AQ which heightens the importance of the results, including ozone reductions in Bakersfield region, SF Bay Area, and SoCAB and $PM_{2.5}$ in the SoCAB. However, while programs and policies are in place to promote the deployment of alternative, low or zero-emitting LDV technologies that will concurrently reduce gasoline consumption, e.g., CA's Zero Emission Vehicle Program[3], it is unknown if emission will also decrease from PFI. A potential outcome is that gasoline production at CA refineries may remain constant with excess product exported. Thus, designing and deploying LDV adoption strategies seeking maximum AQ and GHG benefits should consider also reductions from sources associated with gasoline production and storage; most notably large petroleum refinery complexes.

5.2 AIR QUALITY IMPACTS OF ELECTRIC VEHICLES

In order to assess AQ impacts from transitions to electricity in the LDV sector a methodology was developed to account for spatial and temporal emission shifts correlated to different deployment cases. Vehicle technologies considered include PHEVs with differing battery performance, e.g., 20- and 40-mile all-electric range, and BEVs operating solely on electricity. Additionally, capabilities include a range of potential power sector responses to increases in load demand attributable to vehicle charging with altered emission consequences. Finally, various penetration levels are possible to address the uncertainty of future deployment and examine impacts across a range of outcomes.

The deployment of BEVs and PHEVs will shift emissions from distributed vehicle tailpipes to sites of power generation. In addition, utilization of electricity reduces the need for petroleum fuel production and distribution and correspondingly emissions. However, PHEVs operate on electricity for only a portion of total travel and rely on liquid fuel combustion for the remaining fraction. Thus, spatially and temporally resolved pollutant emission fields for EV cases must account for perturbations to emissions from on-road vehicles, electric generators, and petroleum fuel pathways.

To define the supplanting of direct vehicle and petroleum fuel production pathway emissions total amounts and types of EVs must be determined. The Base Case includes very modest penetrations of BEVs and PHEV-20s in all regions, with electricity meeting 6 to 8% of total LDV fuel demands. Additionally, emission impacts of meeting BAU demand are previously accounted for in power sector emission factors. Thus, to avoid double counting, increased EV deployment is calculated in excess of the levels intrinsic in the Base Case.

As BEVs have no tailpipe emissions, their deployment reduces LDV fleet on-road emissions proportionately (e.g., a 50% increase in fleet wide BEV levels reduces total emissions by half). To estimate the portion of PHEV travel that is met with electricity, and is therefore emissions free, average factors are used corresponding to vehicle all-electric range. For example, a PHEV-40 is assumed to operate on electricity during 50% of total vehicle operation. The remaining fraction of VMT is met with an ICE and thus produces emissions (e.g., a 100% fleet of PHEV-40s would have direct emissions during 50% of the total VMT).

Impacts on petroleum fuel pathways were accounted for relative to the calculated amount of gasoline displaced by electricity in a given case. Associated emissions from gasoline refining, storage, transport, and distribution are also adjusted as discussed for the FCEV scenarios.

Impacts on power sector emissions from increased demand require the estimation of the magnitude and power generation strategy. The amount of total required power is determined by estimating the total portion of regional VMT that is met by electric vehicles and then applying a vehicle efficiency factor corresponding to battery requirements per distance traveled. Calculated power requirements for vehicle charging are then added to regional loads to establish a new regional demand.

Corresponding emissions perturbations from increased power generation are case specific and determined by the assumed evolution of regional generation technologies and fuels. For cases utilizing non-emitting renewable resources (e.g., wind, solar) and nuclear power for vehicle charging no additional emissions are included. For cases assuming that regional average grid mixes provide the power (i.e., the technologies and fuels remain the same but generate more) emissions are increased proportionately to increased demand. Contrastingly, cases that involve the deployment of generation with emissions that differs from BAU grid mixes (e.g., replacement of coal with natural gas combined cycle plants, the deployment of CCS technologies) require calculation accounting the for the fuel and technology switch. For such cases the impacts on power sector emissions are first estimated due to the change in generation characteristics alone; and then emissions are adjusted to reflect the increased generation from EVs.

5.2.1 100 BEV Cases

To verify the developed methodology and study impacts of deploying electric drive LDVs at high levels; cases were examined involving the complete replacement of ICE LDVs with BEVs, i.e., 100 BEV Cases. Though such a dramatic penetration level in 2055 is unlikely, insight can be gained as to the range of possible AQ impacts, i.e., both upper and lower bounds and to compare differences in regional responses to BEV deployment.

In order to elucidate emission trade-offs between emission reductions from vehicles and any increases from necessary generation several power sector responses were developed and included. In the 100 BEV Base Case, additional power requirements are met by increasing generation from existing sources, thus proportionately increasing emissions. Contrastingly, the 100 BEV R Case assumes increased generation is met with emissions free power sources or strategies, i.e., nuclear, renewables, energy efficiency measures, and thus no emission penalty are incurred. Additionally, a case was developed to consider the deployment of CCS technologies in tandem with BEVs. The 100 BEV CCS Case entails both a proportionate emissions increase from augmented generation and perturbations in emissions per unit generation due to the impacts of CCS. Thus, emissions of NO_x in the 100 BEV CCS Cases represents the highest power sector emission penalty while SO_2 emissions are significantly lower. Resulting emissions perturbations from the 100 BEV Cases are displayed in Table 33, Table 34, and Table 35 for the three study regions.

Table 33: Emissions perturbations from complete replacement of LDVs with BEVs in TX

	Impact on Power Demand	Impact on Power Emissions			Impact on LDV Emissions	Impact on Refinery Emissions
TX		**NO_x**	**SO_2**	**PM**		
100 BEV Base	+14.6%	+15%	+15%	+15%	-100%	-38%
100 BEV R	--	--	--	--	-100%	-38%
100 BEV CCS	+14.6%	+45 %	-96%	+15%	-100%	-38%

Table 34: Emissions perturbations from complete replacement of LDVs with BEVs in the NEUS

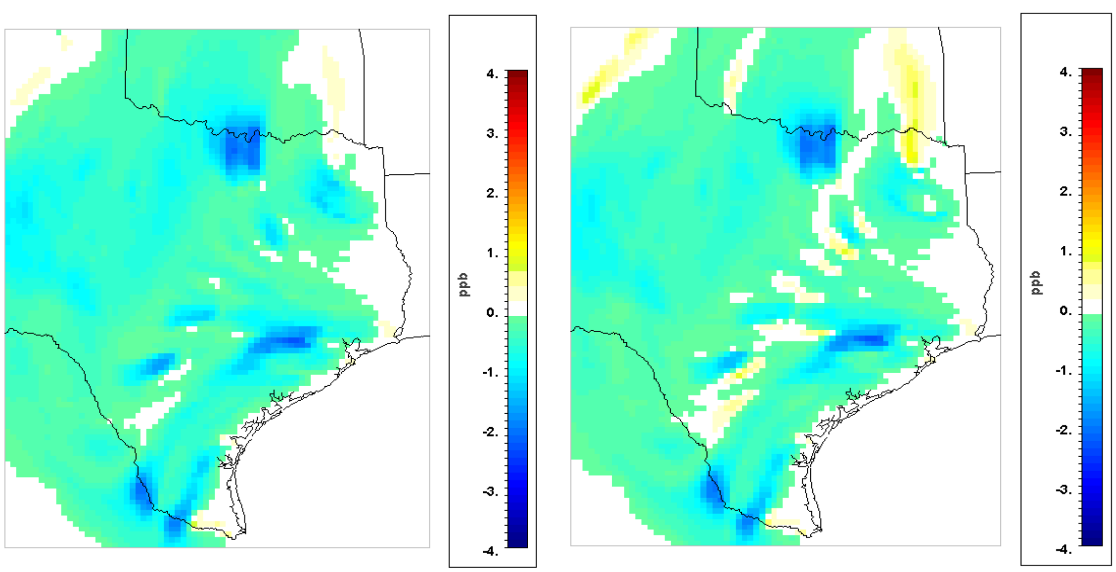
	Impact on Power Demand	Impact on Power Emissions			Impact on LDV Emissions	Impact on Refinery Emissions
NEUS R1		**NO_x**	**SO_2**	**PM**		
100 BEV Base	+10.6%	+11%	+11%	+11%	-100%	--
100 BEV R	--	--	--	--	-100%	--
100 BEV CCS	+10.6%	+36 %	-96%	+11%	-100%	--
NEUS R2		**NO_x**	**SO_2**	**PM**		
100 BEV Base	+4.6%	+5%	+5%	+5%	-100%	-51%
100 BEV R	--	--	--	--	-100%	-51%
100 BEV CCS	+4.6%	+29%	-96%	+5%	-100%	-51%

Table 35: Emissions perturbations from complete replacement of LDVs with BEVs in CA

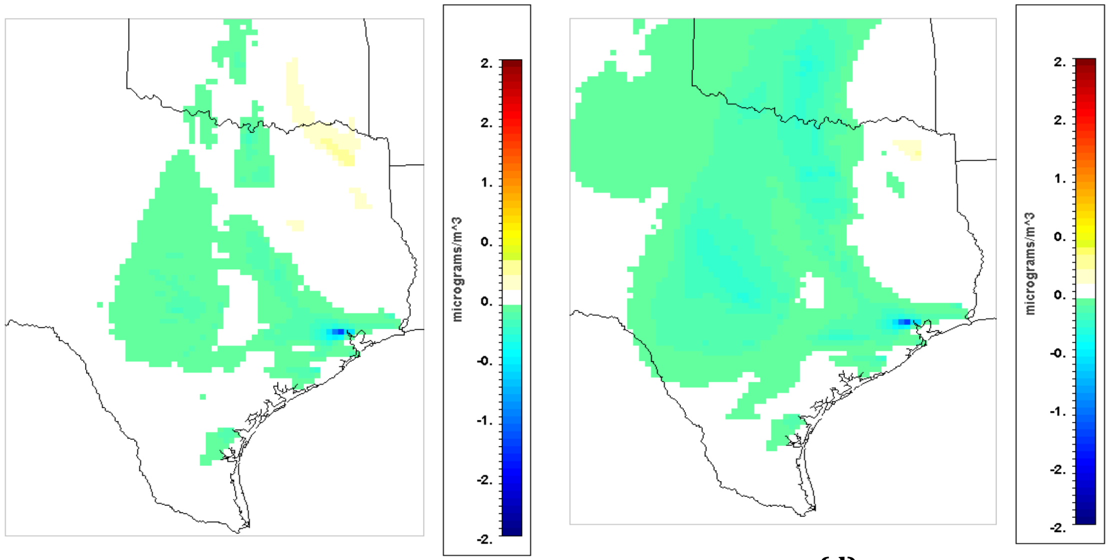
	Impact on Power Demand	Impact on Power Emissions			Impact on LDV Emissions	Impact on Refinery Emissions
CA		**NO_x**	**SO_2**	**PM**		
100 BEV Base	+27%	+27%	+27%	+27%	-100%	-52%

100 BEV R	**--**	**--**	**--**	**--**	**-100%**	**-52%**
100 BEV CCS	**+27%**	**+56%**	**-96%**	**+27%**	**-100%**	**-52%**

The results from the 100 BEV Base and 100 BEV CCS Cases in TX are presented in Figure 70. As can be seen, both cases largely improve levels of ozone and $PM_{2.5}$ throughout much of the region. In particular, ozone levels improve downwind of prominent urban areas of the state in response to significant decreases in tail pipe emissions of NO_x and VOCs, in addition to reductions from petroleum fuel production and distribution. In contrast, small regions of increased ozone are visible from the impacts of increasing major power plant emissions to meet vehicle charging needs. Impacts are similar in spatial dimension but more pronounced in the CCS Case relative to the Base Case, as would be expected from a larger emission increase. Concentrations of $PM_{2.5}$ are most significantly reduced from refineries in the Houston industrial area while the most dramatic increases occur from major coal plants in the northeast corner of the state.



(a) **(b)**



(c) **(d)**

Figure 70: Impacts on peak ozone (a) 100 BEV Base and (b) 100 BEV CCS and 24-h $PM_{2.5}$ (c) 100BEV Base and (d) 100BEV CCS

5.2.2 Electric Vehicle Case 1 (EV 1)

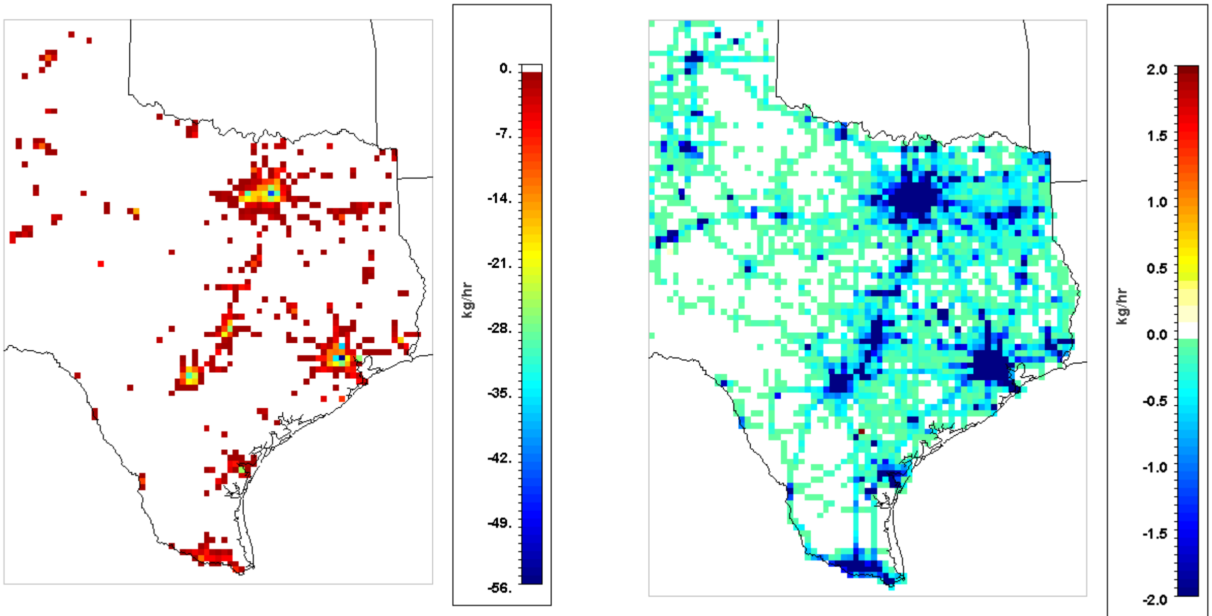
The 100 BEV Cases provide information regarding upper bounds of AQ impacts from electric vehicle deployment but may not represent realistic deployment levels in 2055 due to various factors. In order to examine more feasible penetration levels the EV 1 case was developed. In the EV 1 case BEVs represent 50% of LDVs while PHEVs with a 40 mile all-electric range (PHEV-40s) comprise an additional 25% the fleet. The cases were evaluated for emission and AQ impacts in the TX study region as sector impacts are balanced (i.e., more equally contribute to secondary pollutant levels) and the TX power sector contains high emitting coal plants which could have negative impacts in response to increased power demand for vehicles.

EV 1 Cases followed similar power generation responses as was considered in the 100 BEV Cases, i.e., in the EV 1 Base Case, additional power requirements are met by increasing the base generators and the EV 1 R Case assumes increased generation is met with emissions free strategies. Additionally, a Case was developed to consider the deployment of CCS, the EV 1 CCS Case. The impacts on sector emissions from the EV 1 Cases are displayed in Table 36.

Table 36: Emissions perturbations from the EV 1 Cases in TX

	Impact on Power Demand	Impact on Power Emissions			Impact on LDV Emissions	Impact on Refinery Emissions
TX		**NO_x**	**SO_2**	**PM**		
EV 1 Base	**+9.2%**	**+9%**	**+9%**	**+9%**	**-63%**	**-24%**
EV 1 R	**--**	**--**	**--**	**--**	**-63%**	**-24%**
EV 1 CCS	**+9.2%**	**+34 %**	**-96%**	**+9%**	**-63%**	**-24%**

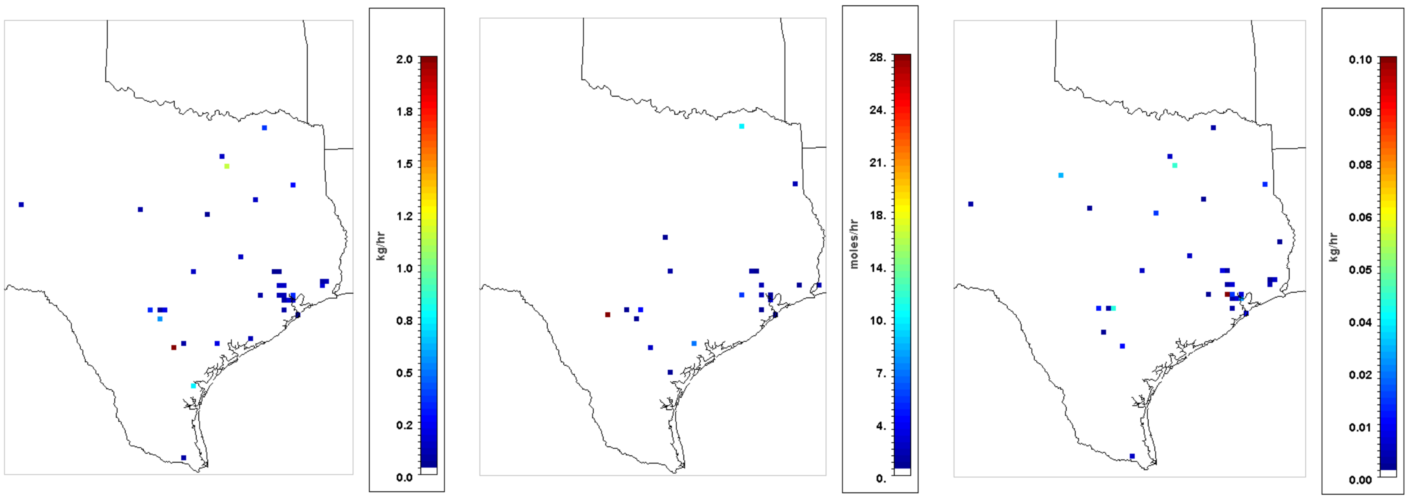
Relative to the Base Case impacts on NO_x emissions are dominated by reductions from on-road vehicles and petroleum fuel infrastructure. As can be seen in Figure 71, for the EV1 Base Case reductions in 24-h average NO_x exceed 56 kg/hr in peak locations. In contrast, impacts from power generation result in localized increases in some areas from power sector emissions. However, as can be seen in Figure 71 (b) reductions dominate over much of the region. Additionally, reductions occur over heavily populated urban centers and major roadways. Directly emitted PM experience similar spatial distribution, however only small differences are seen between cases.



(a) **(b)**

Figure 71: Impacts on 24-h NO_x in the (a) EV1 Base and (b) EV1 CCS Cases relative to the Base Case

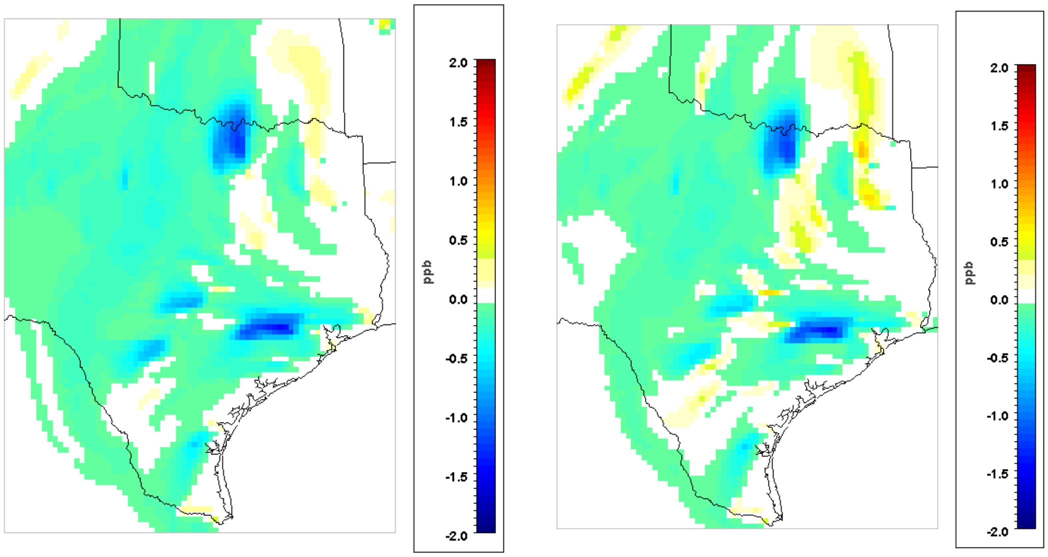
Maximum emission perturbations from power generators can be evaluated by comparing the EV 1 CCS Base Case to the EV 1 R Case. Difference plots for NO_x, PM, and VOCs are shown in Figure 72, which demonstrates the differences in 24-h average emission rates. In general, impacts attributable to different power sector strategies are fairly modest, particularly considering the upper bound nature of the current scenarios. For example, the maximum difference between the EV 1 R Case and the EV 1 CCS Base Case is equivalent to .106 kg/hr. Similarly, though spatially distributed throughout the region the maximum difference in NO_x exceeds 2 kg/hr. This is somewhat surprising given the presence of coal plants in the region and may be a result of assumed implementation of control devices to meet regulatory limits by 2055 that are incorporated in the Base Case.



(a) **(b)** **(c)**

Figure 72: Difference in 24-h (a) NO_x (b) VOCs and (c) PM emissions for the EV1 CCS Case Less the EV1 R Case

The impacts on peak ozone in the EV1 Base and EV1 CCS Cases are displayed in Figure 73. As can be seen, improvements peak around 2 ppb in both cases downwind of major urban areas, mirroring reductions in NO_x. The largest benefits occur from emissions reductions in Dallas-Ft. Worth and Houston, reflecting the significant LDV activity present in both locations. In contrast, worsening is visible in plumes that extend downwind from major coal plant locations. Impacts are similar to the 100 BEV spatially, as would be expected.

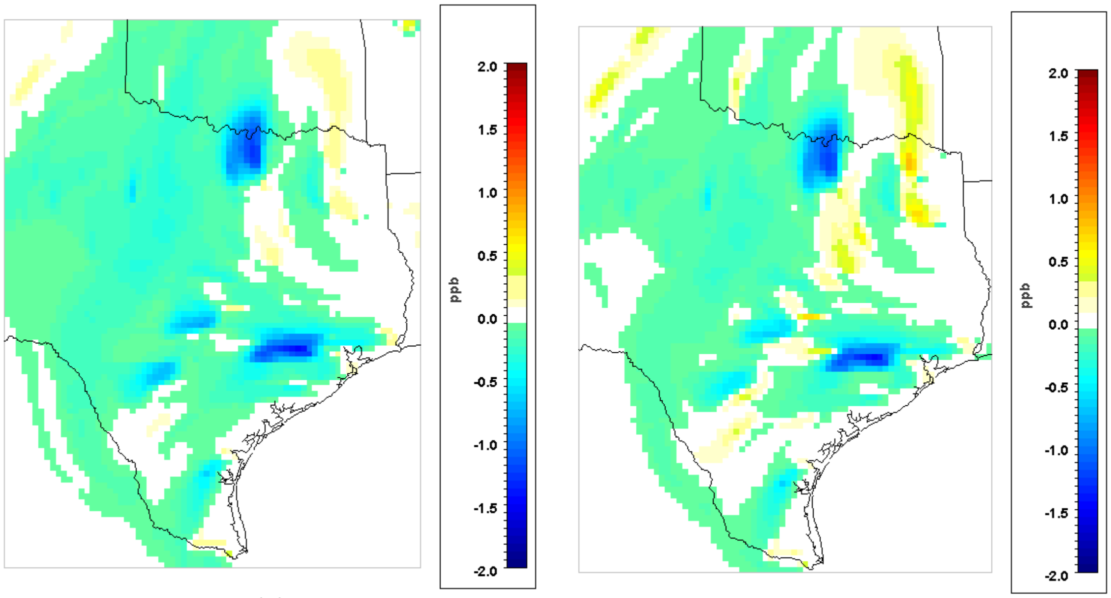


(a) **(b)**

Figure 73: Impacts on peak ozone in the (a) EV1 Base and (b) EV1 CCS Cases relative to the Base Case

5.2.3 Additional Cases

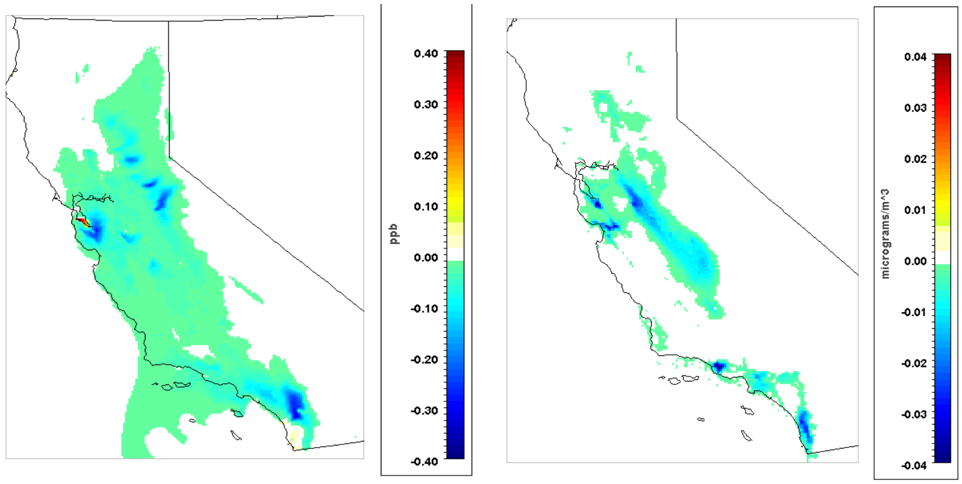
In order to examine the impacts of transitioning to electricity as an LDV fuel cases involving combinations of Battery Electric and Plug-in Hybrid Electric Vehicles were developed and evaluated for ozone and $PM_{2.5}$. Cases involved various combinations of vehicle charging strategies, including the co-deployment of CCS in the power sector which could represent an outcome of high GHG mitigation at the cost of deleterious effects on AQ. In general, direct vehicle emission removal tend to dictate regional AQ impacts, with reductions generally having a higher magnitude and encompassing a greater area than any increases from power generators. However, the co-deployment of CCS with high penetrations of electric vehicles was shown to worsen ozone and $PM_{2.5}$ significantly in areas localized to coal power plants (Figure 74) and such interactions should be considered for regions with significant coal generation.



(a) **(b)**

Figure 74: Impacts on ozone for 30% battery electric and 30% plug-in hybrid electric 40 mile range vehicle deployment with charging demand met (a) without and (b) with the co-deployment of CCS

Figure 75 shows the impacts on AQ from increasing baseline power plant emissions to meet vehicle charging demands relative to meeting demands with zero emitting resources in CA. The results yield relatively minor differences between charging scenarios, equivalent to less than .5 ppb peak ozone, although impacts occur in locations of importance including SoCAB, SF Bay Area, and Sacramento. In addition, impacts on 24-h PM occur but are slight, reaching only -0.04 μg/m^3 in peak locations.



(a) (b)

Figure 75: Impacts on (a) peak ozone and (b) 24-h $PM_{2.5}$ for the EV 1 R compared to the EV 1 Base Scenario in CA

The results demonstrate a noteworthy difference in regards to AQ impacts for EV deployment in CA relative to other regions resulting from a cleaner overall power mix, including a lack of coal. Therefore, an increase in electricity generation from vehicle battery charging does not result in as large of an emissions increase and subsequently increases in ozone and $PM_{2.5}$ as TX. These results suggest that EV deployment in CA could represent an opportunity to simultaneously address GHG and AQ, even if charging strategies include primarily fossil power generation. This is in conflict with TX which experiences significant worsening of ozone and $PM_{2.5}$ levels localized to coal plant locations that must be considered for human health impacts. It should also be noted that these results shouldn't encourage the deployment of EVs in the absence of strategic (smart) charging and low carbon power generation strategies as both will be required to maximize GHG reductions.

Chapter 6: Air Quality Impacts of Mitigation Strategies for Ocean Going Vessels

6.1 INTRODUCTION

Emissions from goods movement sector activity at locations of major ports have been shown to be important drivers of regional AQ with potential human health consequences [355-357]. At present, the bulk of port technologies operate via combustion of distillate and/or residual fuels by means of compression ignition engines; including on-road heavy-duty trucks (HDV), locomotives, marine vessels, and cargo handling equipment (CHE) used to shift containerized and bulk cargo(i.e., yard trucks, side-picks, rubber tire gantry cranes, and forklifts). Exhaust from such systems has been associated with various detrimental human health outcomes including increased risk for cancer, premature mortality, and other disease burdens including the exacerbation of respiratory disease in children [305, 306]. Emphasizing importance towards meeting Federal AQ regulatory standards, addressing port-associated emissions remains a focus of AQ planning in many U.S. regions. For example, California has targeted port emission reductions in both the 2007 State Implementation Plan (SIP) and additional planning documents including the Goods Movement Emission Reduction Plan (GMERP)[301, 311].

In terms of port-associated sources, ships including ocean going vessels (OGV) often represent the greatest challenge with regards to both AQ impacts and pollutant reduction goals. OGV that transport cargo in and out of ports generally lack stringent emissions control and operate on high pollutant emitting distillate or residual fuels[358]. Substantial emissions from OGV include nitrogen oxides (NO_x), particulate matter (PM), volatile organic compounds (VOC), carbon monoxide (CO) and sulfur oxides (SO_x), which all carry direct health risks and contribute to the formation of secondary pollutants, including ozone and $PM_{2.5}$ [358-361]. SO_x emissions are particularly high as marine fuels contain high levels of sulfur (2.7%, world average) relative to on-road diesel fuels (.0015%, CA limit)[309]. It follows then that emissions from OGVs have been linked with events of AQ worsening including the formation of ozone, PM, and radical species [309, 310, 362, 363]. Further, direct emission of PM from OGVs has been shown to impact both communities surrounding ports and those substantially downwind during regional transport events[307]. As a result, emissions from ships represent a significant global threat to human health including coastal regions of the U.S. [308, 364].

Further, the current significance of minimizing environmental impacts of OGV including on regional AQ will be enhanced in coming years by increased demand for transport of goods and reduced emissions from other sources resulting from regulatory constraints (e.g., light-duty vehicles, power plants). It is expected that marine shipping and associated emissions will increase dramatically to 2050[365]. For example, container traffic at California ports has increased dramatically in recent decades and is projected to continue to grow [119, 303]. For example, in 2023 activities from the Ports of Los Angeles and Long Beach are expected to comprise the largest source of emissions in Southern California accounting for 60% of SO_x, 27% of NO_x, and 6% of $PM_{2.5}$ [311]. Indeed, emissions of SO_x from ship traffic are the only regulated pollutant projected to experience total in-basin emissions growth. Thus, in the absence of mitigation, growth in port-associated emissions will have rising importance to regional AQ in the coming decades.

6.2 OGV EMISSIONS

An OGV is defined as a commercial vessel greater than or equal to 400 feet in length or 10,000 gross tons; or propelled by a marine compression ignition engine with a displacement of greater than or equal to 30 liters per cylinder[366]. Generally, OGVs have two primary engine types to meet vessel needs; main and auxiliary. Main engines are commonly large and provide power during transit and maneuvering while auxiliary engines provide power for needs other than propulsion. Further, OGVs have auxiliary boilers that provide steam for vessel needs including fuel and water heating that operate during vessel operations at slow speeds during most port activities (most vessels can use recovered exhaust gas at sea for heat but recovery systems are not effective under a set vessel speed).

OGV emissions include those from main and auxiliary engines and auxiliary boilers both at-berth and over-water. Emissions include those occurring during all operational phases including transiting, maneuvering and hoteling (auxiliary engine emissions at berth) and generally represent the largest single emissions source at ports for many pollutants including NO_x, PM, and SO_x [312]. For example, Table 37 and Table 38 display emissions associated with the Port of Houston and the Port of Los Angeles by mobile source with OGV emissions highest for NOx, VOC/HC, SO_2 and PM. Thus, mitigating emissions from OGV represents a foremost target for mitigation.

Table 37: Sources of Port of Houston Authority Associated Emissions. OGV: Ocean Going Vessel, HDV: Heavy Duty Vehicle, CHE: Cargo Handling Equipment

Pollutant	OGV	HDV Diesel	CHE	Locomotives	Harbor Craft
NO_x	36%	35%	15%	13%	1%
VOC	31%	31%	22%	15%	1.4%
CO	16%	46%	26%	11%	<1%
SO_2	99%	0.1%	0.7%	0.5%	<1%
PM	61-66%	12-14%	14-16%	7-8%	<1%
CO_2	31%	41%	18%	9%	<1%

Table 38: Sources of Port of Los Angeles Associated Emissions. Adapted from [367].

Pollutant	OGV	HDV Diesel	CHE	Locomotives	Harbor Craft
NO_x	47%	18%	11%	12%	11%
HC	45%	14%	15%	11%	15%
CO	21%	18%	32%	10%	19%
SO_2	98%	1%	<1%	<1%	<1%
$PM_{2.5}$	51%	8%	10%	16%	15%
CO_2	24%	45%	17%	8%	6%

Emissions occurring during periods of hoteling make up a large fraction of both total OGV and port emissions and various strategies are being pursued to displace the need for auxiliary engine operation at-berth. For example, in 2012 emissions from auxiliary engines were responsible for 50%, 45%, and 42% of total OGVs at the Port of Los Angeles (Table 39).

Table 39: 2012 emission estimates for OGVs at the Port of LA by engine type. Adapted from [367] TPY=Tons per year.

Engine Type	PM_{10} TPY	$PM_{2.5}$ TPY	NO_x TPY	SO_x TPY	CO TPY	CO_2e tonnes
Main	40	37	1533	132	251	49514
Auxiliary Engine	48	44	1709	245	156	84690

Auxiliary Boiler	18	16	161	245	16	69642
Total	106	97	3402	621	423	203846

6.3 OGV EMISSIONS MITIGATION STRATEGIES

Mitigation strategies for reducing ship emissions include the deployment of new, cleaner engines and fuels, add-on emission controls, and operational changes. Examples of such strategies include speed reductions and switching to low sulfur fuels. Further, the use of shore-based electrical power while in port (cold ironing) can offset substantial emissions by reducing hoteling. Compliance of an ocean-going vessel with vessel speed reduction program and a switch from high- to low-sulfur fuel demonstrated significant reductions in emissions of CO_2, SO_2,and PM[368].

Mitigation strategies include deployment of advanced cargo transportation technologies, including those utilizing linear induction motor technologies operating on electricity, to transport containers to and from ports thereby reducing locomotive and heavy-duty truck emissions. Strategies to mitigate harbor craft emissions include accelerated replacement of existing crafts with models including advanced engine technologies and requiring cleaner fuels.

6.3.1 Cold Ironing

The mitigation strategy Cold Ironing or Shore-to-ship Power (i.e., providing shore power to OGV in place of diesel auxiliary engine operation) has received significant attention by major Ports and regulatory agencies to reduce emissions from OGVs at Ports. For example, the California Air Resources Board (CARB) passed regulatory measures (Airborne Toxic Control Measures for Auxiliary Diesel Engines Operated on Ocean-Going Vessels At-Berth in a California Port) in 2007 requiring operators of container, passenger, and refrigerated cargo OGVs to reduce NO_x and PM from auxiliary engines while docked at CA ports. The requirements include reducing fleet-wide power generated by auxiliary engines by 80% by 2020.

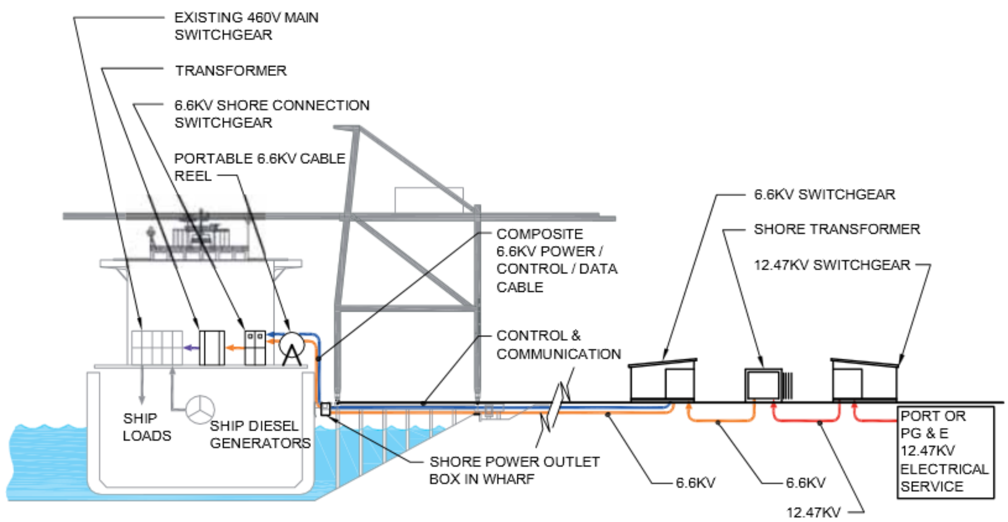


Figure 76: Graphic displaying cold ironing at the Port of Oakland. From[369]

Removing auxiliary engine operation in place of shore power can significantly reduce in-port emissions from vessels, i.e., reductions can be 48-70% CO_2, 3-60% SO_2, and 40-60% NO_x[370]. Further, net reductions have been shown even when considering emissions from local grid mixes and localized shore-side generating options, e.g., liquefied natural gas generators[371].

6.4 AIR QUALITY IMPACTS OF COLD IRONING OGV

6.4.1.1 Scenario Development

To assess the full range of AQ impacts from utilizing cold ironing techniques at major ports all emission perturbations must be accounted for; including both direct reductions from vessel engines and potential increases from electricity generators to support the novel load imposed on the grid. First, the amount of power that will be required by ships at berth for auxiliary needs must be determined for the year 2050. Next, the impact of generating the needed power on emissions from the supporting electrical grid must be accounted for, including both quantity and spatial and temporal perturbations. Pollutants of importance include NO_x, CO, PM, SO_x, and VOCs. Similarly, the reduction in ship emissions from avoiding auxiliary engine operation while at berth must be accounted for to construct detailed emissions fields representing plausible future outcomes. Finally, developed emissions fields must be utilized as input into advanced atmospheric models to conduct simulations of chemistry and transport and determine final distributions of pollutant species of concern including ozone and P.M.

OGV Activity at California Ports

California has 11 major ports including both ocean and inland water ways. Ports selected for assessment included those (1) supporting significant OGV activity and (2) with available data regarding vessel activity.

Data required for scenario development involved accounts of vessels statistics including types, average main and auxiliary engine and boiler sizes, total port calls, average at-berth periods, and emissions. Data was acquired for vessel activity in the Ports of Long Beach and Los Angeles for the year 2012 from published air emission inventories[367, 372]. Similar data was gathered for the Ports of Oakland[373], San Diego[374], Richmond[375], San Francisco[376], Redwood City[377], and Benicia[378]. Additionally, data was obtained from a California Air Resources Board (CARB) document describing emissions estimation methodology for OGVs and reporting average 2005 data for California ports[366]. Data was prioritized by most current year and when possible specific port reported data was utilized in preference over the CARB document. For example, detailed vessel statistics and emissions information was available for the Ports of L.A. and Long Beach in 2012 via port-specific documents (e.g., [367, 372]) and this data was used with priority. In contrast, the Port of Hueneme had limited public data available and thus average 2005 values from [366] were used to represent Port of Hueneme in scenario development. Table 40 displays vessel activity and characteristic data for the Port of Long Beach in 2012.

Table 40: OGV activity data for the Port of Long Beach in 2012

Vessel Type	Vessel Arrivals	% of Total Arrivals	Vessel Departures	Avg. Auxiliary Engine Load at Berth (kW)	Average Time at Berth (hours)
Auto Carrier	154	7.56	154	1181	14.7
Bulk	192	9.43	202	208	50.5
Bulk - Heavy Load	6	0.29	8		68
Bulk – Self Disch.	9	0.44	9	272	34.3
Bulk - Wood Chips	1	0.05	2	179	98.2
Container - 1000	146	7.17	146	710	32.1
Container - 2000	98	4.81	98	1039	49.8
Container - 3000	58	2.85	58	641	46.3
Container - 4000	216	10.61	218	1136	58.4

Container - 5000	173	8.50	171	1128	40.6
Container - 6000	4	0.20	4	804	60.2
Container - 7000	36	1.77	36	845	73.3
Container - 8000	164	8.06	163	1008	85.4
Container - 9000	33	1.62	34	1030	75.6
Container - 11000	17	0.83	17	1500	84.4
Container - 12000	3	0.15	3	2000	113
Container - 13000	6	0.29	6	2500	104
Cruise	160	7.86	160	5445	12.7
General Cargo	95	4.67	94	575	40.3
Ocean Tugboat	15	0.74	16	99	40.7
Miscellaneous	9	0.44	8	467	1122.7
Reefer	0	0.00	2	1091	32.4
Tanker - Aframax	89	4.37	88	632	55.4
Tanker - Chemical	136	6.68	136	738	46.2
Tanker - Handysize	4	0.20	2	605	35.9
Tanker - Panamax	97	4.76	94	683	37.4
Tanker - Suezmax	82	4.03	84	778	26.3
Tanker - ULCC	24	1.18	24	1171	33.3
Tanker - VLCC	9	0.44	9	1171	29.5
Total	*2036*		*2046*		

Cold Ironing Electricity Requirements

To study emissions impacts from cold ironing on the electric grid the novel additional load that will be imposed on the grid must be determined. The amount of electricity for auxiliary vessel needs can be calculated by determining the auxiliary engine load and time of operation, i.e., vessel at-berth periods. At-berth ships do not require their auxiliary engines operate at full load, thus auxiliary engine loads are obtained by applying average load factors. For example, Table 41 displays average load factors for auxiliary engines for various vessel classes under different operational parameters from [366]. For data reporting only auxiliary engine sizes it is assumed that vessels require the average auxiliary load from Table 41 for the duration of their time at berth; equivalent to the average value reported for each visit. It should be noted that some auxiliary engine data is reported with the engine load factor previously applied, e.g., the Port of Long Beach emissions inventory[372].

Table 41: Ocean Going Vessel Auxiliary Load Characteristics. From [366]

Vessel Type	Load Factor (%)		
	Hoteling	Maneuvering	Transit
Auto Carrier	26%	45%	15%
Bulk Carrier/Gen. cargo	10%	45%	17%
Container Ship	18%	50%	13%
Passenger	16%	64%	80%
Reefer	32%	45%	15%
Tanker	26%	33%	24%

The total demand for ship-to-shore power at a given port can be obtained by multiplying the average auxiliary engine loads and at-berth times by total vessel calls in a given year. Table 6 displays the results from applying the methodology for the Port of L.A. The various vessel types differ with regards to number of calls, auxiliary engine size, engine load factor, and average at-berth periods. Thus, significant differences in total ship-to-shore power demand exist for various vessel types. For example, cruise ships have high auxiliary power requirements but are at-berth for shorter periods. In contrast, container ships have a longer at-berth period on average, which leads to high power demand despite, in some cases, smaller auxiliary engine loads. Additionally, the distribution of vessel types by number of calls strongly impacts the results. The Port of L.A. is a major port for container cargo traffic and a thus large amount of total power needs are associated with container ships.

Table 42: 2012 vessel activity data, characteristics, and ship-to-shore power needs for the Port of L.A. Data adapted from [367], Power calculated using developed methodology

Vessel Type	Vessel Calls	Average Auxiliary Engine Size (kW)	Average Time at Berth (hours)	Auxiliary Engine Load Factor	Total Power (kW-hr)
Auto Carrier	100	3169	23.8	0.26	1961
Bulk	89	na	66.6	0.10	0
Bulk - Heavy Load	2	na	49.2	0.10	0
Bulk - Wood Chips	3	na	82.1	0.10	0
Container - 1000	41	4421	24.1	0.18	786
Container - 2000	256	4649	26.8	0.18	5741
Container - 3000	46	3919	53.1	0.18	1723
Container - 4000	289	7058	36	0.18	13218
Container - 5000	232	8228	40.6	0.18	13950
Container - 6000	291	10631	75.1	0.18	41820
Container - 7000	19	10771	73.5	0.18	2708
Container - 8000	93	10911	71.8	0.18	13114
Container - 9000	98	11520	76.2	0.18	15485
Container - 11000	5	15196.5	79.1	0.18	1082
Cruise	98	18873	9.5	0.16	2811
General Cargo	73	3286	53.2	0.10	1276
Ocean Tugboat	38	na	37		0
Miscellaneous	1	na	76.9	0.10	0
Reefer	30	3245	21.3	0.32	664
Tanker - Aframax	3	2040	53.1	0.26	84
Tanker - Chemical	71	2400	33.2	0.26	1471
Tanker - Handysize	32	1650	35.5	0.26	487
Tanker - Panamax	43	2040	45.5	0.26	1038
Total	*1953*				*119418*

Table 43 displays the results for total power required for each port to provide ship-to-shore power for all vessels calls in a given base year. The three largest demands are associated with the Port of L.A., the Port of Long Beach, and the Port of Oakland. It should be noted that the Ports of Oakland, Richmond, San Francisco, Redwood City, and Benicia are all located in the same geographic area (i.e., the Bay Area) and in terms of power system and AQ impacts may be considered together.

Ports differ significantly in primary function which yields variance in vessel characteristics which impact results. For example, the Port of Richmond is predominated by auto carrying vessels with shorter temporal at-berth periods while the Port of San Francisco hosts a high number of cruise ships which maintain large auxiliary engine loads relative to other types of OGVs. Similarly, vessels docking at the Port of Oakland maintain shorter berth periods on average than those at the Port of L.A. and Long Beach. Thus, though total vessel call numbers are similar the total demand for shore power is significantly higher at the Port of L.A. and Long Beach.

Table 43: Total power required to provide ship-to-shore power for all vessel calls in a given base year by individual port

Port	Base Year	Total OGV Calls	Total MW-hours
Los Angeles	2012	1953	119,418.95
Long Beach	2012	2036	91,952.29
Oakland	2012	1812	55,890.87
San Diego	2006	506	15,841.06
Hueneme	2005	403	15,743.49
San Francisco	2005	121	7,019.10
Richmond	2005	178	2,732.96
Redwood City	2005	75 (barges excluded)	1,176.67
Benicia	2005	63	717.04
Total			*310,492.43*

Cold Ironing Vessel Projection

To assess the impacts in coming decades of deploying cold ironing strategies vessel activity must be projected from the baseline year. In some cases, specific data was available for ports including the Ports of L.A. and Long Beach, e.g., see Table 44 which displays projected container vessel calls by class to the year 2023 from a document projecting emissions growth[379].

Table 44: Container ship forecast for the Ports of L.B. and L.A. by class to 2023. From [379]

Container Vessel Class	Port of Long Beach			Port of Los Angeles		
	2005	**2014**	**2023**	**2005**	**2014**	**2023**
Container 1,000-1,999	203	208	52	199	0	0
Container 2,000-2,999	320	286	156	180	286	156
Container 3,000-3,999	181	182	260	285	182	260
Container 4,000-4,999	281	407	468	377	633	728
Container 5,000-5,999	170	357	416	205	267	312
Container 6,000-6,999	61	368	468	128	204	260
Container 7,000-7,999	57	166	208	49	250	312
Container 8,000-9,999	111	213	260		255	312
Container 10,000-12,000		104	260		104	260
Total	**1,384**	**2,291**	**2,548**	**1,423**	**2,181**	**2,600**

Additionally, projected growths in vessel activity for non-container OGVs must be accounted for, i.e., the projections in Table 44 only apply for Container OGVs. Table 45 shows estimated projections in non-container cargo and cruise ships for the Port of L.A. and Long Beach. The available projection data for non-container vessels was in units of tonnes cargo (and number of passengers for cruise vessels) rather than vessel calls. Future growth in vessel calls was estimated from percent growth in cargo or passengers applied to the vessel calls in the base year, i.e., 2012 for the Port of L.A.

Table 45: Non-container cargo projections for San Pedro Bay Ports (i.e., L.A. and L.B.). From [379]

Commodity		2005 (tonnes)	Forecasted Cargo 2014 (tonnes)	2023 (tonnes)
Dry Bulk		17,369	26,443	30,141
Liquid Bulk		23,594	31,403	35,164
General Cargo & Break Bulk		5,469	8,597	11,113
Auto		896	1,200	1,560
Reefer		476	633	733
Cruise LA	Passengers:	1,218,739	1,406,036	1,727,710

Using growth trends from the available data for predicted vessel call and cargo growth to 2023, container vessels and non-container cargo vessel activity data was projected to 2050. Table 46 displays projection estimates for the Port of L.A. For vessel types that result in sum total negative values the 2023 value was held constant (e.g., Container – 2000 class OGV). As can be seen, significant growth in calls occurs for some container ship classes including Container 4000 and 7000 OGVs. Overall, total vessel calls increase in excess of three times to 2050 from 2012. While it is unlikely the Port of L.A. could support the projected number of vessel calls without major expansion to its facilities and/or improved cargo handling methods, the estimates still provide a reasonable assessment of potential OGV activity in 2050.

Table 46: Projected Vessel Call Activity for the Port of L.A.

Vessel Type	2012 Calls	2023 Calls	Growth	Years	Calls/ year	Years	New Calls	2050 Calls
Auto Carrier	100.00	130.00	30.00	11	2.73	38	103.64	203.64
Bulk	89.00	101.45	12.45	11	1.13	38	43.00	132.00
Bulk - Heavy Load	2.00	2.24	0.24	11	0.02	38	0.83	2.83

Bulk - Wood Chips	3.00	3.88	0.88	11	0.08	38	3.03	6.03
Container - 1000	41.00	0.00	-41.00	11	-3.73	38	-141.64	0.00
Container - 2000	256.00	156.00	-100.00	11	-9.09	38	-345.45	156.00
Container - 3000	46.00	260.00	214.00	11	19.45	38	739.27	785.27
Container - 4000	289.00	728.00	439.00	11	39.91	38	1516.55	1805.55
Container - 5000	232.00	312.00	80.00	11	7.27	38	276.36	508.36
Container - 6000	291.00	260.00	-31.00	11	-2.82	38	-107.09	183.91
Container - 7000	19.00	312.00	293.00	11	26.64	38	1012.18	1031.18
Container - 8000	93.00	312.00	219.00	11	19.91	38	756.55	849.55
Container - 9000	98.00	260.00	162.00	11	14.73	38	559.64	657.64
Container - 11000	5.00	0.00	-5.00	11	-0.45	38	-17.27	0.00
Cruise	98.00	120.42	22.42	11	2.04	38	77.45	175.45
General Cargo	73.00	94.36	21.36	11	1.94	38	73.80	146.80
Ocean Tugboat	38.00		-38.00	11	-3.45	38	-131.27	38.00
Miscellaneous	1.00	1.16	0.16	11	0.01	38	0.55	1.55
Reefer	30.00	34.74	4.74	11	0.43	38	16.37	46.37
Tanker - Aframax	3.00	3.36	0.36	11	0.03	38	1.24	4.24
Tanker - Chemical	71.00	91.78	20.78	11	1.89	38	71.78	142.78
Tanker - Handysize	32.00	41.60	9.60	11	0.87	38	33.16	65.16
Tanker - Panamax	43.00	49.79	6.79	11	0.62	38	23.47	66.47
Totals	*1953.00*	*3274.78*	*1321.78*	*11*	*120.16*	*38*	*4566.14*	*6519.14*

Using the number of vessel calls projected for 2050 the growth in load to meet cold ironing needs can be calculated. It is assumed that the vessel characteristics for each vessel type (i.e., auxiliary engine size, load) remain the same to 2050. However, to accommodate growth in total cargo it is expected that improvements in cargo loading and unloading discharge rates will occur via strategies including terminal densification measures and the addition of cargo handling equipment, e.g., cranes. These changes will reduce temporal hoteling periods per vessel call and thus lower required shore power. To account for changes efficiency improvements estimated for 2023 in [379] were used (Table 47). Additionally, the same efficiency improvements were applied to all non-cargo vessels as well to account for technological and behavioral improvements that will be realized in coming decades.

Table 47: Projected hoteling efficiency gains for the Port of L.A. From [379]

Vessel Capacity	Average Hours POLA 2005	Average Hours POLB 2005	2014 Assumed Efficiency	2023 Assumed Efficiency
Container 1,000-1,999	36.5	23.2	0%	0%
Container 2,000-2,999	38.4	40.3	0%	0%
Container 3,000-3,999	41.6	44.7	11%	14%
Container 4,000-4,999	44.2	47.6	11%	14%
Container 5,000-5,999	73.7	72.4	11%	14%
Container 6,000-6,999	66.1	105.5	11%	14%
Container 7,000-7,999	63.5	74.0	11%	14%
Container 8,000-9,999	36.2	100.9	11%	14%
Container 10,000-12,000	N/A	N/A	0%	0%

The total load required to provide ship-to-shore for all projected vessel calls in 2050 is the product of the number of calls, average hoteling periods, auxiliary engine load and load factor, and the efficiency improvement factor. Table 48 displays the resulting projections for the Port of L.A. in 2050. The total MWh required for providing ship-to-shore power for all vessel calls is significant, equivalent to an increase over four times the power needed in 2012. The majority of power needs are for container OGV, particularly large vessel classes, i.e., 7000 to >9000. This reflects both the transition to larger OGVs with higher capacity and the characteristic deep ship channel facilitating larger OGV access to the Port.

Table 48: Projected 2050 vessel calls and required ship-to-shore power for the Port of L.A.

Vessel Type	2050 Calls	Efficiency Factor	Total MWh	% of Total
Auto Carrier	203.64	0.86	3434.21	0.71
Bulk	132.00	0.86	0.00	0.00
Bulk - Heavy Load	2.83	0.86	0.00	0.00
Bulk - Wood Chips	6.03	0.86	0.00	0.00
Container - 1000	0.00	0.86	0.00	0.00
Container - 2000	156.00	0.86	3008.78	0.62
Container - 3000	785.27	0.86	25296.55	5.24
Container - 4000	1805.55	0.86	71017.20	14.71
Container - 5000	508.36	0.86	26288.50	5.44
Container - 6000	183.91	0.86	22729.41	4.71
Container - 7000	1031.18	0.86	126371.62	26.17
Container - 8000	849.55	0.86	103025.94	21.34
Container - 9000	657.64	0.86	89364.33	18.51
Container - 11000	0.00	0.86	0.00	0.00
Cruise	175.45	0.86	4328.56	0.90
General Cargo	146.80	0.86	2207.06	0.46
Ocean Tugboat	38.00	0.86	0.00	0.00
Miscellaneous	1.55	0.86	0.00	0.00
Reefer	46.37	0.86	882.07	0.18
Tanker - Aframax	4.24	0.86	102.73	0.02
Tanker - Chemical	142.78	0.86	2543.86	0.53
Tanker - Handysize	65.16	0.86	853.47	0.18
Tanker - Panamax	66.47	0.86	1379.49	0.29
Total	*7008.78*		*482833.77*	*100.00*

Again, it should be noted that the projected OGV calls and cargo loads are in most cases greater than current or expected port capacities. For example, projecting growth in cargo tonnage for the Port of Oakland yields a total of 7642 thousand twenty foot equivalent units (TEUs) in 2050 while capacity for the Port of Oakland in 2020 is estimated at 3817 thousand TEUs (Figure 77) [380]. It is possible that existing port facilities may expand or novel port facilities constructed. Additionally, as discussed above technological and behavioral improvements in cargo transport and handling will facilitate increased throughput in terms of TEU per time, allowing for more cargo to be processed at existing facilities.

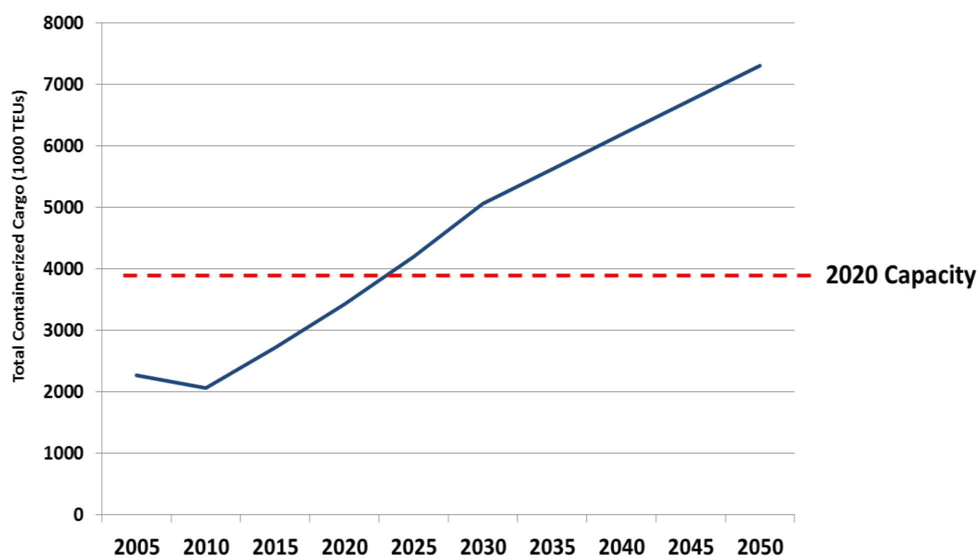


Figure 77: Actual and projected total containerized cargo throughput for the Port of Oakland. Additionally, estimated maximum cargo capacity in 2020 is shown.

Utilizing cargo projection trends to predict growth in vessel calls for non-cargo vessels (e.g., cruise ships) may not be appropriate as demand for vessel activity may be impacted differently by various drivers. For example, cruise ship activity is highly correlated to local economies and societal factors including development and safety of port destinations. For some California ports cruise ship activity represents a significant portion of auxiliary power needs, e.g., the Port of San Diego. Thus, for some ports cruise ship traffic was projected using available specific data, e.g., Figure 78 displays data projected to 2040 from [381] under three different growth assumptions which is then extrapolated to 2050. Using the trends from the published data cruise ship vessel calls were projected from current levels for the Port of San Diego to estimate required auxiliary power needs for cruise ships in 2050 (note that the medium growth scenario was used to account for variability in future factors). For ports lacking specific projection data for cruise ships, e.g., Port of Hueneme, San Francisco, the number of 2055 cruise vessel calls was maintained at baseline levels to avoid over counting.

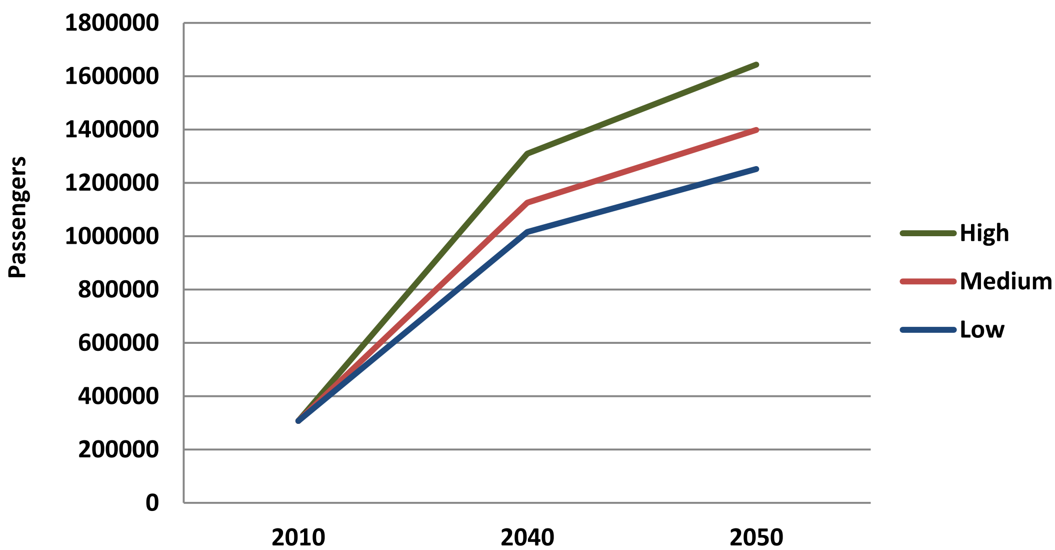


Figure 78: Projected growth in cruise ship passengers for the Port of San Diego.

For ports lacking appropriate projection data for shipping cargo demand, e.g., the Port of San Diego, projections were based on output from the Market Allocation (MARKAL) model, a data-intensive energy systems economic optimization model with an EPA developed and maintained 9-region MARKAL allowing for national and regional energy system characterization. Figure 79 displays the projected growth in cargo shipping demand for the region containing California. Factors for percentage growth to 2055 were determined from the MARKAL data and applied to estimate growth in vessel calls.

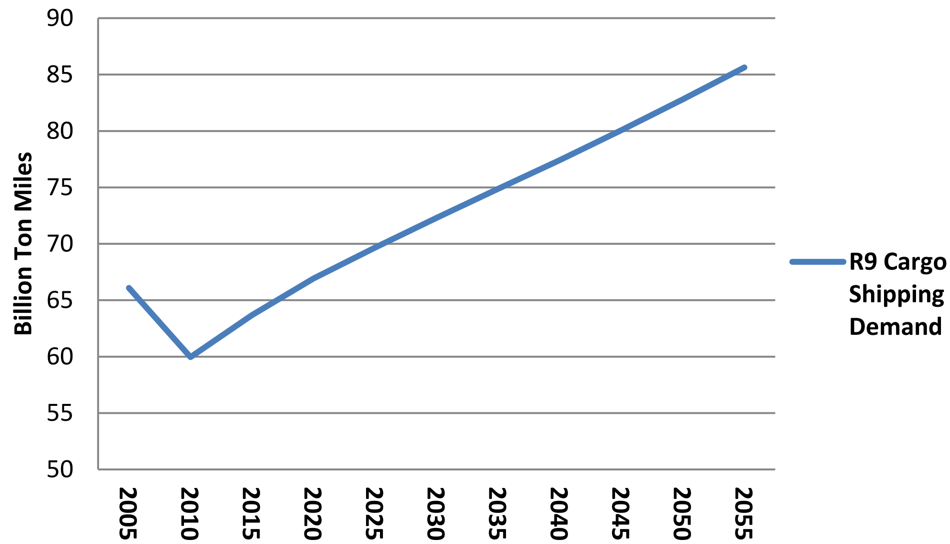


Figure 79: MARKAL projected data for Region 9 cargo shipping demand

The projected 2055 OGV calls and required MWh for auxiliary power needs are displayed in Table 43. Total annual vessel calls exceed 21,000 for all vessel types in aggregate. Total power demand to provide ship to shore power for all calls exceeds 1200 gigawatt hours (GWh). Though the amount of novel power is significant it is within reason to consider for addition to the grid, e.g., the total value is equivalent to less than .5% of the 2013 total California power system from data available at http://energyalmanac.ca.gov/electricity/total_system_power.html.

Table 49: Total power required to provide ship-to-shore power for all vessel calls in 2050

Port	2055 OGV Calls	Total 2055 MWh
Los Angeles	7008	482833.77
Long Beach	7415	503357.10
Oakland	5647	149633.39

San Diego	655	33164.47
Hueneme	522	17647.55
San Francisco	132	35563.51
Richmond	231	11712.48
Redwood City	97	1311.12
Benicia	82	798.97
Total	*21,789*	*1236022*

6.4.1.2 Cold Ironing Emissions Impact

Power Sector Emissions

In order to fully assess emissions and AQ impacts of deploying ship to shore power impacts on emissions from power generators responding to novel demands must be considered. The additional load required to meet ship auxiliary demands can be met by (1) grid provided electricity and (2) distributed technologies including fuel cells and natural gas generation devices.

Power Sector Projection to 2055

For cases assuming the power sector emissions increase in proportion to the growth in projected load to meet novel power demand it is assumed that the existing CA power grid provides the entire load to port locations. Further, it is assumed that existing generation technologies increase operation proportionately in response to power demand. Thus, power sector emissions increase proportionately to demand and are spatially allocated to current plant locations.

In order to determine the quantitative increase in emissions, the relative increase in power generation in 2055 must be determined. Mirroring population and economic growth, the CA power system will grow in total size from current day, including in both demand and generation. For this assessment, power systems growth was projected using MARKAL output from the same run that provided estimates of shipping cargo demand in Chapter 4:. However, MARKAL data is grouped at the census region level, thus reported quantities are representative of the total of several states rather than just CA. In order to accurately estimate the CA grid in 2055, reported data for 2013 was used to estimate the portion of the MARKAL total attributable to CA. This percentage was then utilized to determine the total fraction of CA power in 2055 http://energyalmanac.ca.gov/electricity/total_system_power.html), assuming all contributions are held equal in proportion. The factor was determined to be 66.48% of R9 demand, which is a reasonable value given CA's large size relative to the additional R9 states. Using the factor it was determined that the total annual power system load for CA in 2055 was 477,786.96 GWh including all imported sources of power (Figure 76). Using this value as a baseline the relative increase in demand arising from additional power needed for ships can be calculated. For the Cold 100 Case the novel load for auxiliary power was equivalent to an increase of 0.25% total annual CA power consumption. Emissions from the power sector were determined to be proportionate to the increase in demand, i.e., 0.25% for the Cold 100 Case. It is assumed that all generator locations and grid dynamics remain constant. Increases were applied to all area and point sources associated with electricity in CA via appropriate source classification (SCC) codes in the SMOKE model.

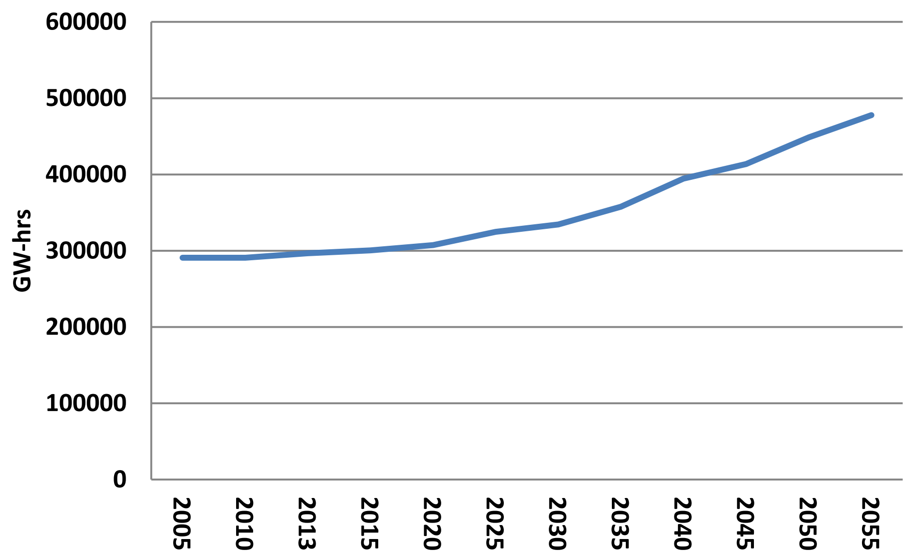


Figure 80: Projected California net power consumption

Ship Emissions

Next, the reduction in emissions from the avoidance of auxiliary engine operation must be calculated and applied to developed emission fields. Auxiliary engine emissions represent only a portion of OGV port-related emissions and thus the contribution to the total must be determined. Emissions in 2012 by OGVs at the Port of L.A. are displayed by mode in Figure 81. Emissions of NO_x from auxiliary engine and boiler operation during hoteling while at berth are equivalent to 56% of total OGV NO_x emissions. It should be noted that emissions from hoteling during anchorage were not reduced as ships at anchorage are positioned away from docks and it is unknown if ship-to-shore power will be available. Thus, emissions of NO_x from OGVs at the Port of L.A. were reduced by 56% to account for the cold ironing of auxiliary power equipment on vessels. Relative emission contributions were not equivalent amongst various ports. For example, the Port of L.B. 2012 air emissions inventory reported NO_x emissions from hoteling at berth equivalent to 39% of total OGV port-related NO_x emissions. However, constraints in the Smoke model require that emission impacts for a given pollutant and technology be applied at the county level and for the Ports of L.A. and L.B. a net factor must be calculated as they share the same county (Los Angeles). Thus total emissions and emissions from hoteling at berth were summed for both ports in 2012 and utilized to determine an overall reduction factors for Los Angeles County (i.e., the total reduction in OGV NO_x emissions from cold ironing is equivalent to 38.16%. Equivalent methods were utilized to determine OGV emission reductions for the remaining Ports and pollutants in the assessment and the results are displayed in Table 50. The only port for which an emissions inventory was accessible was the Port of Hueneme. Emissions were allocated for the Port of Hueneme by applying an average factor calculated from the available data.

Mode	Engine Type	PM_{10} tpy	$PM_{2.5}$ tpy	DPM tpy	NO_x tpy	SO_x tpy	CO tpy	HC tpy	CO_2e tonnes
Transit	Main	33.0	30.4	32.3	1,316.0	124.2	204.7	104.9	46,571
Transit	Aux	9.0	8.3	9.0	319.5	46.3	28.9	10.5	15,664
Transit	Auxiliary Boiler	1.1	1.0	0.0	12.3	13.1	1.2	0.6	5,350
Total Transit		**43.1**	**39.7**	**41.3**	**1,647.8**	**183.6**	**234.8**	**116.0**	**67,585**
Maneuvering	Main	6.9	6.4	6.9	217.0	7.5	46.3	39.1	2,943
Maneuvering	Aux	3.6	3.4	3.6	130.5	18.5	11.8	4.3	6,417
Maneuvering	Auxiliary Boiler	0.4	0.3	0.0	3.7	4.5	0.4	0.2	1,585
Total Maneuvering		**10.9**	**10.1**	**10.5**	**351.2**	**30.5**	**58.5**	**43.6**	**10,945**
Hotelling - Berth	Main	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0
Hotelling - Berth	Aux	31.5	29.1	31.5	1,125.0	157.0	103.4	37.6	56,126
Hotelling - Berth	Auxiliary Boiler	15.2	13.6	0.0	135.0	209.6	13.6	6.8	58,555
Total Hotelling - Berth		**46.7**	**42.7**	**31.5**	**1,260.0**	**366.6**	**117.0**	**44.4**	**114,681**
Hotelling - Anchorage	Main	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0
Hotelling - Anchorage	Aux	3.9	3.7	3.9	133.6	22.8	11.9	4.3	6,483
Hotelling - Anchorage	Auxiliary Boiler	1.2	1.0	0.0	9.6	17.6	1.0	0.5	4,152
Total Hotelling - Anchorage		**5.1**	**4.7**	**3.9**	**143.2**	**40.4**	**12.9**	**4.8**	**10,635**
Total		**106**	**97**	**87**	**3,402**	**621**	**423**	**209**	**203,846**

Figure 81: 2012 Port of L.A. OGV emissions by mode. From [367]

Table 50: Net emission impacts at the county level for OGVs in the High Scenario. **Port of Hueneme represents an average value calculated for all ports**

County and Port	NO_x	PM_{10}	$PM_{2.5}$	SO_2	CO	VOC
Los Angeles – L.A. & L.B.	38.16%	45.91%	45.60%	60.14%	29.74%	23.75%
San Diego – S.D.	39.82%	45.88%	45.97%	49.24%	29.97%	37.46%
Alameda – Oakland	31.84%	35.42%	35.42%	41.18%	32.32%	20.45%
Ventura – Hueneme**	33%	35%	35%	40%	31%	25%
San Francisco – S.F.	44.52%	45.18%	45.18%	46.54%	44.37%	42.10%
Cont. Costa – Richmond	42.14%	36.67%	36.67%	41.92%	39.62%	27.27%
San Mateo – Redw. City	42.87%	43.49%	43.49%	52.27%	44.64%	34.18%
Solano – Benicia	25.22%	24.75%	24.75%	28.21%	24.80%	18.36%
Average	*33%*	*35%*	*35%*	*40%*	*31%*	*25%*

6.4.1.3 Explanation of Cases

The uncertainty inherent with projection of energy systems to 2055 necessitates the use of scenario approach to span a range of potential outcomes (i.e., identification of trends in relation to technological shifts is the primary goal rather than exact prediction of impacts). Thus, a set of cases encompassing different combinations of results for both the power sector and OGV emissions was developed to account for a range of plausible situations in 2055 for cold ironing at ports.

Cold 100 Projected

The Cold 100 Projected Case essentially represents the standard case for cold ironing using the developed methodology. In the Cold 100 Projected Case power sector impacts are determined by the scaling method previously described, i.e., in CA power sector demand and emissions rise by 0.25% from the baseline distributed equally across all generators. Similarly, ship emission reductions are determined by the method described previously and equivalent to the values listed in Table 50. Thus, the Cold 100 Case incorporates the results from the utilized methodology and should be interpreted as the baseline cold ironing case.

Cold 100 Renewable

The Cold 100 Renewable Case is designed to examine the impact on AQ from if the increase in power sector emissions that may come from increased power demand can be avoided. Thus, in the Cold 100 Renewable Case the power sector emissions do not change and remain equivalent to the baseline. Ship emission reductions are equivalent to those in the Cold 100 Case. Essentially the Cold 100 Renewable Case could be viewed as a best case outcome for cold ironing in 2055 in that significant reductions are achieved in the absence of any increases elsewhere. While such an outcome is optimistic, several real world situations could be represented including the use of renewable power, demand response, or fuel cells to provide ship-to-shore power with zero or near-zero emissions.

Cold 100 High Electric Case

In contrast, the Cold 100 High Electric Case is constructed to examine the AQ impacts of cold ironing if major effects occur on the power system in terms of increased emissions, i.e., a worst case outcome for emission trade-offs. In the Cold 100 High Electric Case it is assumed that emissions from power generators increase by 10% in response to novel power demands for providing ship-to-shore power. The ship emission impacts in the Cold 100 High Electric Case are equivalent to the Cold 100 Projected and Cold 100 Renewable Cases. It should be noted that the 10% value is significantly greater that what is likely or expected (i.e., a 10% increase in State-wide power demand in 2055 would be equivalent to almost 48 terawatt-hours). However, by utilizing the 10% value confidence can be gained that any power sector emission impacts that occur in 2055 will likely be reduced from the 10% increase and the Cold 100 High Electric Case can effectively function as a "worst case" scenario in that regard.

Cold 50 Projected Case

The Cold 50 Projected Case assumes that only 50% of the projected reductions in OGV emissions are achieved from the deployment of cold ironing strategies. The power sector emission response is assumed to be that of the Cold 100 Projected Case, i.e., an increase of

0.25% in total load corresponding emissions. The Cold 50 Projected Case is designed to account for the outcome that various challenges prevent the full realization of cold ironing at ports and subsequent emission reductions. By maintaining the projected increase in generator emissions the scenario can evaluate the AQ impacts of cold ironing in the eventuality that ship emissions do not decrease as expected.

6.4.1.4 Emissions Results

The resulting emissions perturbations for both OGV and the power sector were applied via SMOKE to develop spatially and temporally resolved emissions fields representative of the various scenarios.

Table 51 displays the maximum quantitative change in emissions of PM and NO_x for several key Cases. The impact of power sector emission is fairly minor relative to impacts from auxiliary ship engines. The difference in emissions of NO_x between the Cold 100 High Electric Case and the Cold 100 Renewable Case (i.e., a comparison between the highest possible impact on power sector emissions and no impact) yields increases from a few spatial locations throughout the state corresponding to natural gas power plants. The peak impact observed in the state is +7.99 kg/hr; although the majority of impacted locations experience impacts below +3 kg/hr. In contrast, reductions in PM and NO_x in the Cold 100 Projected Case relative to the Base Case exceed 640 kg/hr and 350 kg/hr, respectively, and reflect the reductions occurring from ships. Thus, from a quantitative standpoint it would be expected that improvements in AQ would be observed despite the increase from power sector emissions. However, the spatial distribution of impacts complicates an understanding of resulting changes to ground-level concentrations of ozone and $PM_{2.5}$ and thus AQ simulations must be performed to estimate them.

Table 51: Peak reductions from Base Case in 24-h average emissions

Case	Less Case	Δ Direct PM 24-hr [kg/hr]	Δ NO_x 24-hr [kg/hr]
Cold 100 Proj	**Base**	-641.20	-352.84
Cold 100 HE	**Cold 100 Ren**	+9.26	+7.99

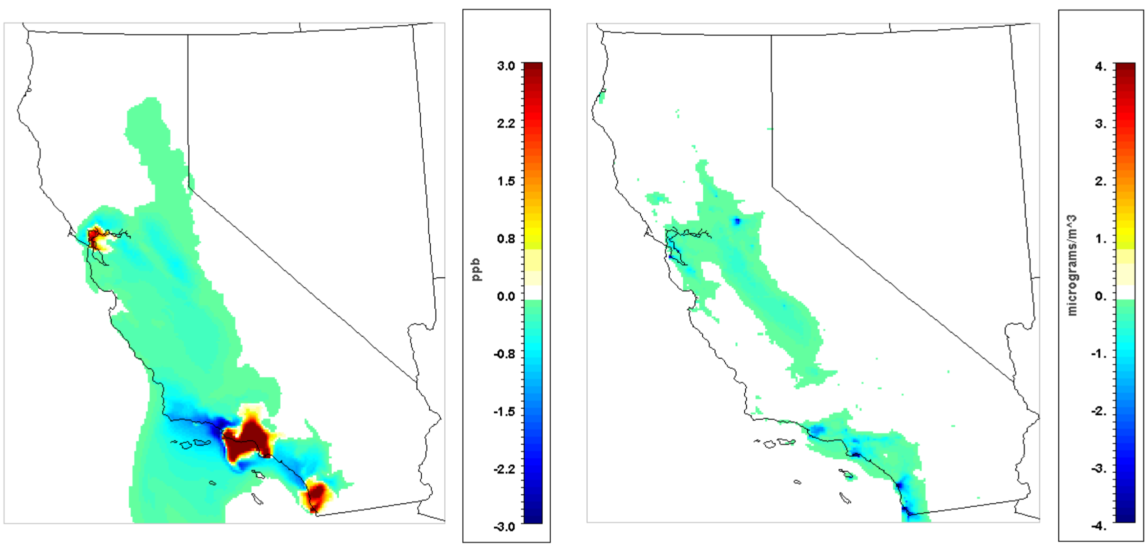
6.4.1.5 Cold Ironing Air Quality Results

Table 52 displays the peak reductions in maximum 8-hr ozone and 24-hr $PM_{2.5}$ from the Cold Ironing Cases evaluated in this section.

Table 52: Peak reductions from Base Case in ozone and $PM_{2.5}$

Case	Less Case	Δ Max 8 Hour Ozone [ppb]	Δ $PM_{2.5}$ 24-hr [μg/m³]
Cold 100 Proj	**Base**	-3.38 to +7.43	-18.34 to +0.011
Cold 100 Ren	**Base**	-3.39 to +7.46	
Cold 100 HE	**Base**	-3.36 to +7.48	-18.34 to +0.010
Cold 100 HE	**Cold 100 Ren.**	-0.13	-0.047 to +0.026
Cold 50 Proj	**Base**	-1.65 to +3.44	-18.25 to +0.011

Figure 82 displays the resulting differences in (a) maximum 8-hour average ozone and (b) 24-hour average $PM_{2.5}$ for the Cold 100 Projected Case and the Base Case. Impacts on ozone range from -3.38 ppb to +7.43 ppb; but, note that the increase results from titration effects (usually at night) associated with ozone formation and not as a result of increased emissions from power generators. Thus, impacts on ground-level ozone are largely beneficial and occur over much of the State with peak impacts including urban areas adjacent to major ports. Increases in ozone due to the effects of titration are visible in areas associated with the largest NOx reductions and include the SoCAB, San Diego, and the S.F. Bay Area. Despite localized increases, reductions occur over regions of the highest baseline concentrations including Eastern portions of San Bernardino and Riverside Counties. Impacts on $PM_{2.5}$ are more straight forward and are characterized by reductions that range from -18.3 to +0.01 μg/m^3. Peak impacts occur in the SoCAB and San Diego although lesser reductions also occur in Northern portions of the Central Valley and the SF Bay Area. No areas of worsening are observed at the scale shown and thus the Cold 100 Projected Case can be considered largely beneficial from an AQ standpoint.

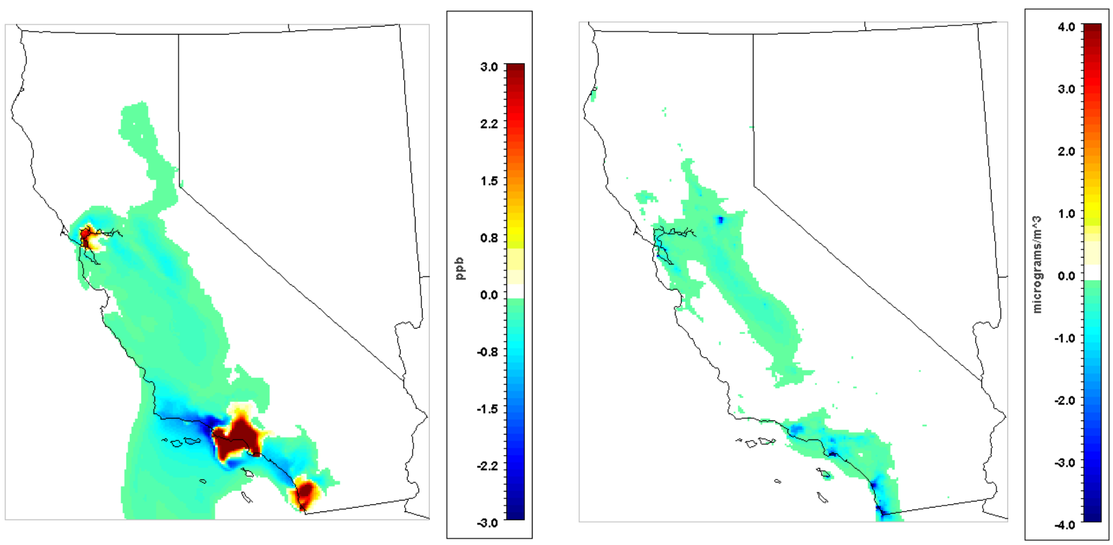


(a) (b)

Figure 82: Difference in (a) max 8-hr ozone and (b) 24-h $PM_{2.5}$ for the Cold 100 Projected Case relative to the Base Case

Figure 84 displays the resulting differences in (a) maximum 8-hour average ozone and (b) 24-hour average $PM_{2.5}$ for the Cold 100 High Electric Case and the Base Case. Impacts on ozone range from -3.36 ppb to +7.48 ppb and follow similar spatial distributions to those from the Cold 100 Projected Case. Similarly, $PM_{2.5}$ impacts closely resemble those in the Cold

100 Projected Case and range from -18.34 to +0.010 μg/m^3 with peak reductions occurring in the SoCAB and San Diego. The results support the conclusion that the largest driver of AQ impacts in Cases occurs from the ship emissions with only minor impacts from those from the power sector. For example, despite a significant difference in assumed increases from generator emissions (i.e., +0.25% vs. +10%) the peak impacts of both the Cold 100 Projected and Cold 100 High Electric Cases have only minor differences.



(a) **(b)**

Figure 83: Difference in (a) max 8-hr ozone and (b) 24-h $PM_{2.5}$ for the Cold 100 High Electric Case relative to the Base Case

The true impact of increasing power sector emissions can be shown by a difference plot between the Cold 100 Renewable (+0%) and the Cold 100 High Electric (+10%) as all other emissions sources remain constant. Thus, any perturbation observable between the two Cases can be directly attributed to the increase assumed in the High Electric Case. Figure 84 displays the resulting differences in (a) maximum 8-hour average ozone and (b) 24-hour average $PM_{2.5}$ for the Cold 100 High Electric Case and the Cold 100 Renewable Case. Impacts are minor and include peak increases of +0.13 ppb ozone and +0.03 μg/m^3 $PM_{2.5}$. Spatially, impacts are associated with generator locations and include the SoCAB, S.F. Bay area, and sources throughout Greater Sacramento and Northern Central CA. While increases in ground-level concentrations are of concern to adjacent populations, the minor magnitude lessens this somewhat. This is particularly true when considering the overall reduction occurring from the Base Case in some of the same locations with large, dense urban populations, i.e., SoCAB and the S.F. Bay Area. Thus, the results show the importance of removing ship emissions even if consequently increases occur from the power sector. It should be considered that the results may be more applicable for CA as the State has a relatively clean mix of power generators.

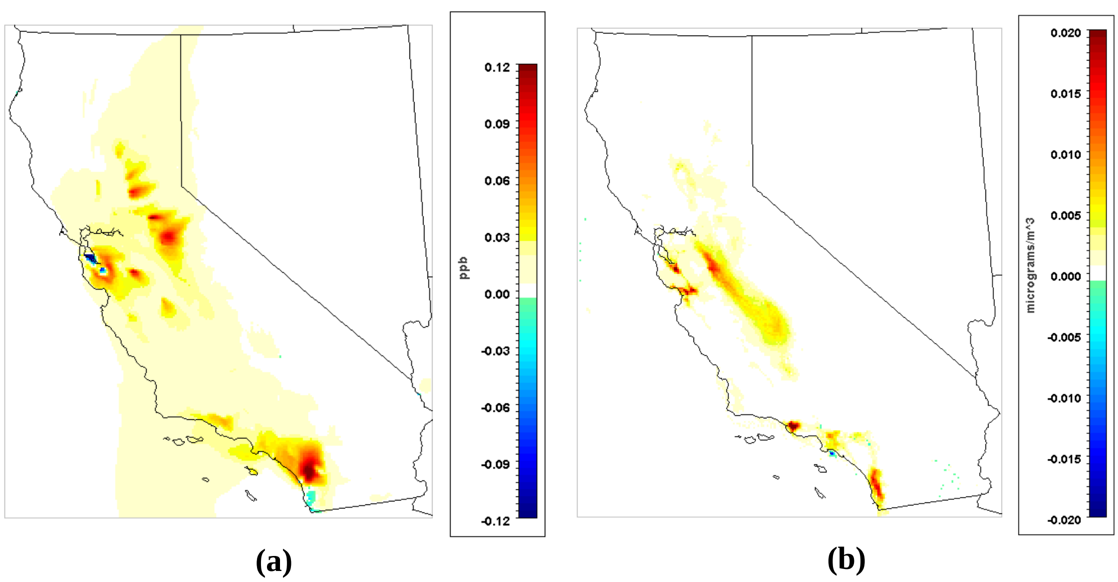
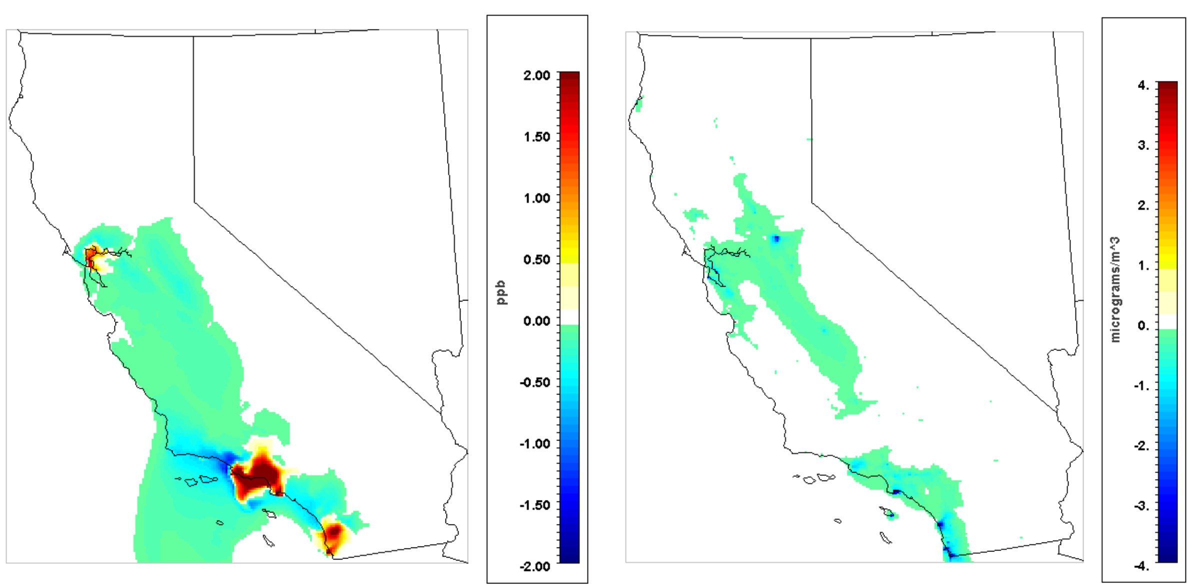


Figure 84: Difference in (a) max 8-hr ozone and (b) 24-h $PM_{2.5}$ for the Cold 100 High Electric Case relative to the Cold 100 Renewable Case

Figure 16 displays the resulting differences in (a) maximum 8-hour average ozone and (b) 24-hour average $PM_{2.5}$ for the Cold 50 Projected Case and the Base Case. Peak impacts on ozone and $PM_{2.5}$ range from -1.65 to +3.44 ppb and -18.25 to +0.01 μg/m^3, respectively. Spatially, the distribution of impacts is similar to the additional Cases including the Cold 100 Projected Case. The Cold 50 Projected Case is designed to examine the outcome of ship emission reductions not reaching projected levels while the increase in power sector remains constant. Thus, the results of the Case can help to further answer the question of if power sector emission increases could potentially represent a dis-benefit to the State for cold ironing. As can be seen, impacts continue to represent a benefit to the State and further demonstrate the tremendous magnitude and significance of ship emissions relative to those from the power sector.



(a) (b)

Figure 85: Difference in (a) max 8-hr ozone and (b) 24-h $PM_{2.5}$ for the Cold 50 Projected Case relative to the Base Case

6.4.1.6 Cold Ironing Summary

The results of the Cold Ironing Cases show that cold ironing OGVs in 2050 represents an opportunity to improve regional AQ in CA. Improvements in ozone and $PM_{2.5}$ occur from completely cold ironing suitable vessels despite potential increases in emissions from the power sector as a result of new generation required to meet vessel needs and exceed 3 ppb and 18 $\mu g/m^3$ for most Cases. Indeed, the difference in ozone and $PM_{2.5}$ occurring between no increase and a 10% increase in power emissions is minimal and in the most sensitive areas still represents a reduction overall from the Base Case. Further, the locations of greatest impact are associated with large urban populations including the SoCAB, S.F. Bay Area, and San Diego due to the large ports located in those regions. Thus, the State should pursue cold ironing as a potential strategy to meet Federal ambient AQ standards and provide health benefits to residents.

Chapter 7: Impacts on Air Quality of Renewable Resources and Electrification

7.1 INTRODUCTION

California (CA) must reduce greenhouse gas (GHG) emissions as mandated by Assembly Bill 32; including an aggressive 2050 goal of 80% below 1990 levels[10]. In order to meet these goals significant changes will be required in all CA energy sectors, including the power sector. As discussed in Section 2.2 renewable resources are a key strategy being pursued by CA and the U.S. to reduce environmental impacts of power generation including reducing GHGs and improving regional AQ. However, the deep reductions required by AB 32 will likely require the co-deployment of additional emissions mitigation strategies including increased efficiencies, electrification of additional sectors, demand response, and energy storage[82].

Electrification is defined as the deployment of devices utilizing electricity in place of those that would otherwise convert a fuel to meet the energy demand. For example, the replacement of a water heater operating on natural gas with an electric water heater is an electrification strategy in the Residential Sector. The result of electrification is to remove the emissions from the fuel-converting device (e.g., a gas water heater) in tandem with increases in electricity demand and subsequent emissions from power generators. The goal of electrification is to reduce net emissions of both GHGs and criteria pollutants which can improve AQ, especially if it is accompanied with increased electricity production from renewable resources.

Indeed some of these electrification strategies may facilitate increased penetrations of renewable technologies while minimizing emissions penalties from operational impacts. Electrification represents an opportunity to provide low carbon energy to CA sectors in parallel with renewable resource integration of the grid. With this premise, transition to electricity from conventional energy strategies in energy sectors (e.g., gasoline in transportation, natural gas in commercial or residential) could result in net reductions in GHG emissions, particularly as CA's grid becomes integrated with higher levels of low carbon renewable power. In addition, reductions in pollutant emissions could also result in AQ benefits. However, assessing the AQ impacts of such shifts is not as simple as quantifying total emissions. The complexity of regional pollutant formation and fate requires an understanding be gained of how perturbations in pollutant emissions from both the power and electrified sector translate to changes in spatial and temporal emissions and how these are converted by atmospheric chemistry and transport into ground-level pollutant concentrations. Thus, there is a need for more information regarding how renewable resources in tandem with increased electrification of energy sectors could impact emissions and AQ in CA.

As discussed in Section 2.2.1, assessment of the AQ impacts from increasing level of renewable resources requires an analysis of the entire power system, including accounting of spatial and temporal emissions from all generators. Electrification of additional sectors in tandem with renewable resource deployment will increase power demand with potential changes to spatial and temporal distributions. Thus, a Base Case for a given year (2020, 2030, and 2050) must be developed accounting for targeted renewable resource capacities, load growth, baseline demands, etc. Next, Cases are constructed accounting for the electrification of different sectors (Residential, Commercial, Industrial, and Transportation) with changes to power demand in a horizon year (2020, 2030, and 2050). Changes to baseline generator emissions must be estimated and spatially and temporally resolved via a detailed representation of the future CA electrical grid. Conversely, reductions in emissions from technologies in energy sectors that are electrified must also be accounted for. Finally, simulations of atmospheric chemistry and transport are conducted and assessed for impacts on maximum 8-hour ozone and 24-hour $PM_{2.5}$ relative to the concentrations in the Base Case.

7.1.1 Sectors

The following section discusses the sectors considered for electrification in this work including residential, commercial, industrial, and transportation energy conversion. A full description of the sectors, including electrification potentials and characteristics, can be found in Reference [382].

7.1.1.1 Residential

The amount of energy consumed for residences in CA is less than the national average as a result of a moderate climate in the State and stricter building codes. Major energy conversion purposes include space heating and cooling, water heating, appliances, electronics, and lighting. It should be noted that electrification only applies to non-electric devices, e.g., an electric technology cannot be electrified. The main fuel used in the CA residential energy sector other than electricity is natural gas; with other fuels including biomass, LPG, and residual fuel oil at small amounts. Most of the natural gas consumption is attributable to water and space heating; with cooking, pools, and dryer operation also included.

Potential electrification technologies in the residential sector include electric space heating via a variety of technologies. Additionally, heat pumps can transfer thermal energy when needed. Resistance heating and heat pumps can also be used to supply heated water when required in place of gas water heaters. Electricity can also be utilized in ovens and stoves for cooking needs.

7.1.1.2 Commercial

Significant consumption of energy in the commercial sector includes large and small office buildings, educational buildings, retail, health, lodging, restaurants, food stores, and refrigerated warehouses. Major commercial consumption of natural gas (and can thus potentially be electrified) includes cooling, heating, water heating, and cooking. In particular, cooking represents an important opportunity for electrification.

7.1.1.3 Industrial

The industrial sector includes facilities, processes, and equipment utilized in the production of goods in CA. Major consumers of natural gas in industry include process heating, thermal processing units, boilers, HVAC, and miscellaneous uses. As discussed in Section 4.6, the industrial sector is complex and contains many specific processes that cannot be easily replaced with alternative strategies. Thus, electrification of the industrial sector has higher uncertainty and constraints compared to other sectors.

7.1.1.4 Transportation

The transportation sector represents a major opportunity to electrify in CA as the vast majority of its energy conversion is associated with non-electric petroleum-based fuels. The largest single sub-sector in terms of energy conversion is LDVs and the dominant fuel for LDVs is gasoline. Thus, there is significant potential to offset gasoline consumption by utilizing electricity to power LDVs.

Potential technologies for electrification of LDVs including PHEVs and BEVs as discussed in Section 5.2. Barriers for electricity as a vehicle fuel include battery range, lack of charging infrastructure, high capitol cost of vehicles, and others. However, the use of the vehicle fleet as potential energy storage could offer significant benefits to grid operation and renewable resource integration.

7.2 APPROACH

The approach used to develop and analyze electrification cases for AQ impacts involving various energy sectors are briefly described in this section. A more exhaustive explanation can be found in Reference [382].

Cases of electrification implementation in end-use sectors are developed for 2020, 2030, and 2050. First, electrification potential is developed based on BAU electric penetration, electrification potential, and feasibility of available electric technologies. For example, higher electrification penetrations are assumed in the 2030 and 2050 Cases due to a higher possibility of avoiding limitations and supportive policies. Cases are also developed for electrification in combinations of end-use sectors. In addition, a BAU Case is developed for each horizon year to serve as a comparison for the electrification outcomes (the Base Case). The Base Case includes projections from MARKAL[383].

Electrification in end-use sectors created new power demand that must be accounted for in the development of load profiles for each Case. The load profile is determined based on perturbations between the baseline and new electric penetrations while considering energy efficiency ratios.

In order to determine complementary technology and renewable dispatch a model developed at UCI, HiGRID, is utilized. HiGRID determines the temporal power profiles of demand response, energy storage, distributed generation, and electric LDVs required to balance intermittencies of renewables [384]. The dispatch of utility generators is accomplished using an electric grid simulation model, PLEXOS [385]. The temporal distribution of pollutant emissions including both centralized utility and distributed generators are determined through temporal generation profiles in combination with emission factors. Next, spatial information regarding the locations of existing generators is utilized to develop spatial distributions of emissions from power generators. End-use emissions are reduced via SMOKE based on the net change in fuel consumption translating to emissions reductions[386].

Finally, developed emissions fields are utilized as input for simulations of atmospheric chemistry and transport. The resulting output files are assessed for ground-level concentrations of ozone and $PM_{2.5}$ to determine overall AQ impacts. Cases are assessed for both summer and winter in order to capture both high ozone (summer) and high PM (winter) episodes.

7.3 ELECTRIFICATION AIR QUALITY RESULTS

A set of scenarios are analyzed, developing spatially and temporally resolved emissions and simulating the resulting AQ accounting for the electrification of various energy sectors in tandem with renewable resource integration in the power sector. The reference case – Base Case – is used as the baseline for the analysis of the other cases. The baseline emissions inventory used for the analysis presented here is based on the National Emissions Inventory (NEI) for 2005, developed by United States Environmental Protection Agency (USEPA, http://www.epa.gov/ttn/chief/emch/index.html). The 2005 emissions are then projected to 2020 using statewide growth and control factors reported by the California Air Resources Board (http://www.arb.ca.gov/app/emsinv/fcemssumcat2013.php). All scenarios represent power demand and generation for the horizon year (i.e., 2020, 2030 or 2050) as well as reductions in emissions from technologies that are electrified, e.g., natural gas ovens, space heating. A summer and a winter one-week episode are evaluated for each case, in order to analyze the effects of changing emissions on high ozone (summer) and high particulate matter (winter) formation conditions.

The following abbreviations are used to denote Cases: Residential = Res, Commercial = Com, Industrial = Ind, Transportation = Tra. Further, Cases involving combinations of sectors are abbreviated as such, Residential, Commercial and Transportation = ResComTra.

Power sector emissions are calculated directly in the modeling methodology and thus are not listed here.

7.3.1 Cases for 2020

Table 53 displays the list of developed cases and emission reductions inherent in each case by sector excluding the power sector. As can be seen, variation in electrification potential for technologies and fuels is not equivalent across sectors and thus significant differences exist in emission reductions, i.e., the residential and commercial sectors can support a larger penetration of electric technologies and thus achieve higher reductions than the industrial sector. The transportation sector case assumes a moderate penetration of electric light duty vehicles (≈10%) and thus achieves the lowest reduction in sector emissions.

Table 53: Reductions in energy sector emissions for 2020 Cases

2020	Sector Emissions Reduction			
Case	**Residential**	**Commercial**	**Industrial**	**Transportation**
Res 2020	29.44%	----	----	----
Com 2020	----	30.77%	----	----
Ind 2020	----	----	5.25%	----
Tra 2020	----	----	----	5.11%
ResCom 2020	29.44%	30.77%	----	----
ResComTra 2020	29.44%	30.77%	----	5.11%
ResComTraInd 2020	29.44%	30.77%	5.25%	5.11%

7.3.1.1 Residential Sector 2020 Case

Summer

Figure 86 displays the difference in maximum 8 hour average ozone in the Summer Residential 2020 from the Base Case. Generally, impacts on ozone are minor and range from -0.84 to 0.21 ppb; although the majority of perturbations fall between + or - 0.5 to 0.2 ppb. Emission reductions from residential natural gas combustion technologies translate to small improvements that cover large areas of the State. Contrastingly, increased emissions from existing gas power generation results in localized areas of worsening that have higher magnitude than improvements but are highly limited in spatial terms. Additionally, effects of the ozone formation dynamics lead to small increases in tropospheric ozone levels in VOC-limited areas (i.e., the South Coast Air Basin (SoCAB) despite decreases in NO_x emissions; however these are generally not regarded as an AQ detriment.

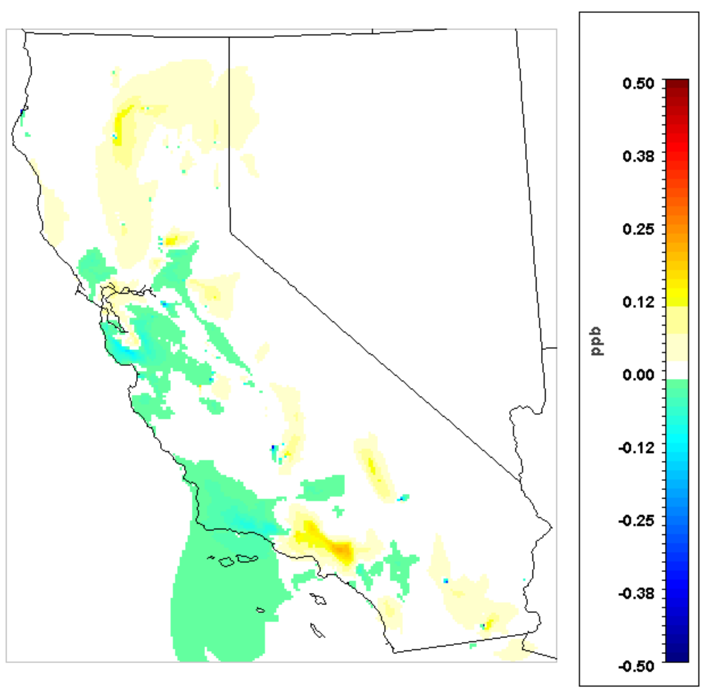


Figure 86: Difference in max 8-h average ozone in the Summer Residential 2020 Case from the Base Case

Figure 87 displays the difference in 24 hour fine particulate matter (24-h $PM_{2.5}$) in the Summer Residential 2020 Case from the Base Case. In general, impacts are relatively moderate (i.e., range of -0.08 to +0.38 μg/m^3) reflecting the low PM emitting nature of both California's power grid (the lack of coal and high reliance on natural gas) and residential natural gas technologies. However, improvements are observed in the northern portion of the Central Valley and parts of SoCAB which have importance as both regions currently experience challenges associated with meeting the health-based Federal standards for $PM_{2.5}$. Conversely, with similarity to spatial ozone patterns localized areas of worsening occur throughout the State, most notably the central valley in the region of Bakersfield which experiences the largest impacts in terms of magnitude. This is a concern as the area currently is plagued by poor AQ.

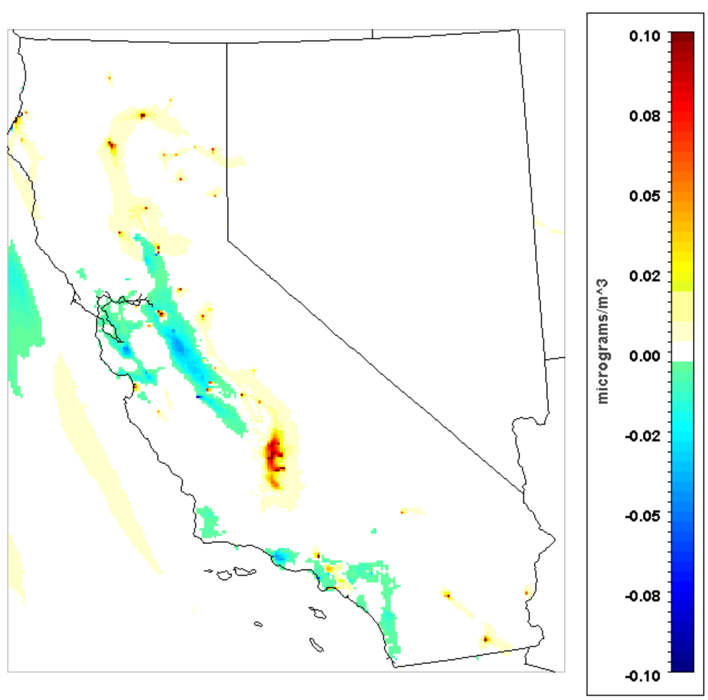


Figure 87: Difference in 24-h average $PM_{2.5}$ in the Summer Residential 2020 Case from the Base Case

Winter

Figure 88 displays the difference in maximum 8 hour average ozone in the Winter Residential 2020 Case from the Base Case. Quantitatively, impacts range from -0.36 to +0.75 ppb. In general, trends are similar to the Summer Case, e.g., moderate improvements over large areas and localized areas of worsening with higher magnitude but reduced area of impact. However, reflecting the complexity of grid dynamics localized areas of worsening occur at dissimilar locations in winter relative to summer, e.g., the winter case experiences an increase in ozone in SoCAB and the Bay Area while the summer case experiences worsening in the northern central part of the State. Contrastingly, reductions in concentration are observable in the Central Valley of the State. Essentially, emission reductions in winter are associated with worsening and increased emissions yield reductions in ground-level ozone. Differences in impacts on ozone are expected for summer and winter episodes due to differences in ozone formation dynamics including reduced photolysis resulting from shorter days and lower ambient temperatures. As a result, ambient ozone concentrations are typically much lower in winter compared to summer.

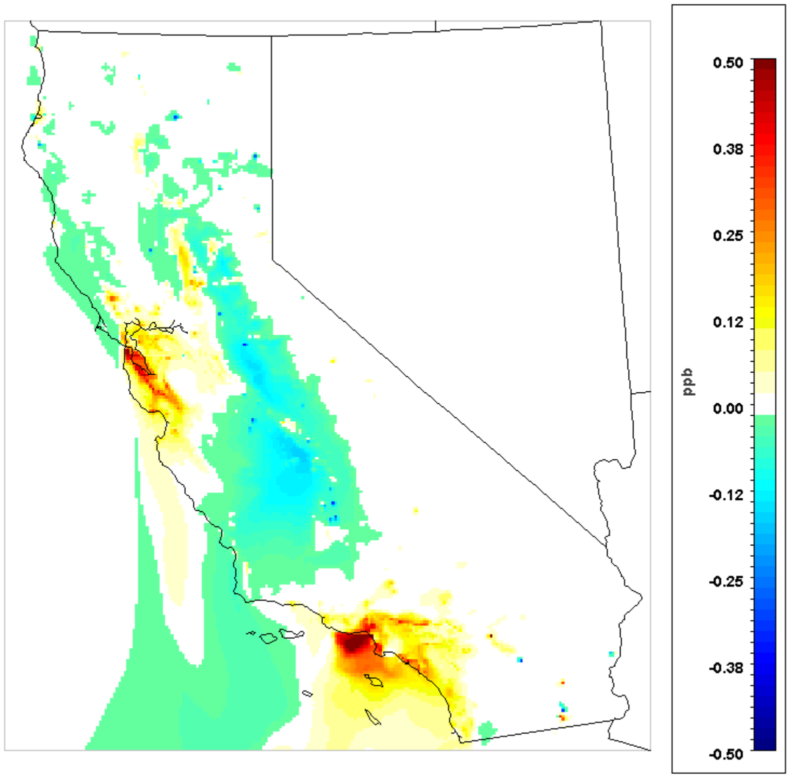


Figure 88: Difference in max 8-h average in the Winter Residential 2020 Case from the Base Case

Figure 89 displays the difference in 24-h $PM_{2.5}$ in the Winter Residential 2020 Case from the Base Case. Impacts are similar to those from the Summer Case and include small to moderate improvements in the central valley in tandem with small areas of worsening co-located to gas generators. Additionally, with similarity to ozone results, differences between summer and winter are evident in locations of worsening and reflect different grid dynamics between the two seasons as well as variation in impacts from residential sector demands and technologies, e.g., heating is required in winter while cooling is required in summer, which changes the emissions and resulting AQ.

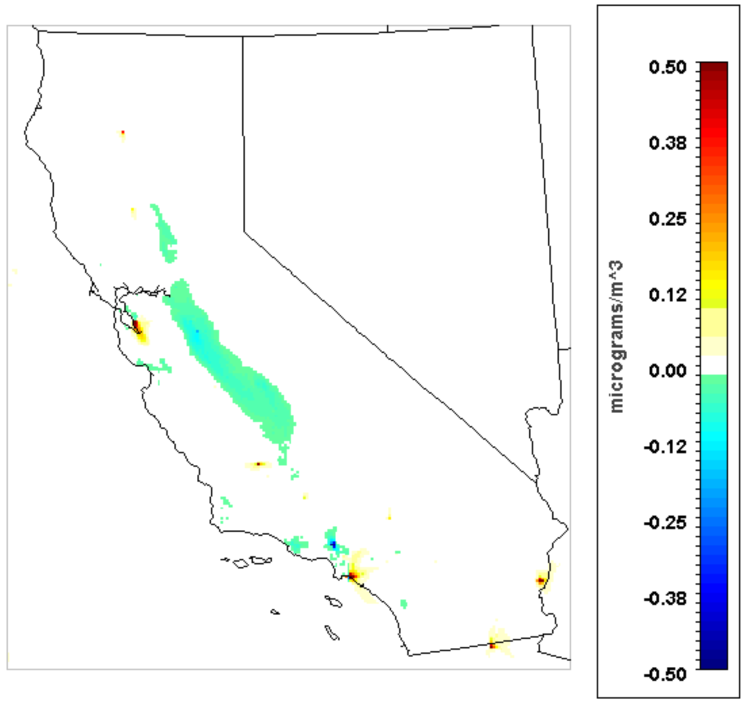


Figure 89: Difference in 24-h $PM_{2.5}$ in the Winter Residential 2020 Case from the Base Case

7.3.1.2 Commercial Sector 2020 Case

Summer

Figure 90 shows the difference in maximum 8-hour average ozone in the Summer Commercial 2020 Case from the Base Case. Quantitatively, impacts range from -0.34 to +0.78. Impacts are similar in spatial distribution to those from the Residential Case, i.e., reductions in concentrations occur over large areas of the study region while some local areas experience worsening. Most notably, impacts are observed in the Bakersfield region including increases in concentrations which are followed by decreases in surrounding areas. This could be a result of reduced ozone scavenging reflecting significant reductions in emissions from the commercial sector in the area. Similar impacts are also observed in SoCAB. Relative to the Summer Residential 2020 Case impacts include larger reductions over much of the State as a result of higher emissions occurring from commercial sector sources. Contrastingly, worsening in and around Bakersfield is higher in the Commercial 2020 Case.

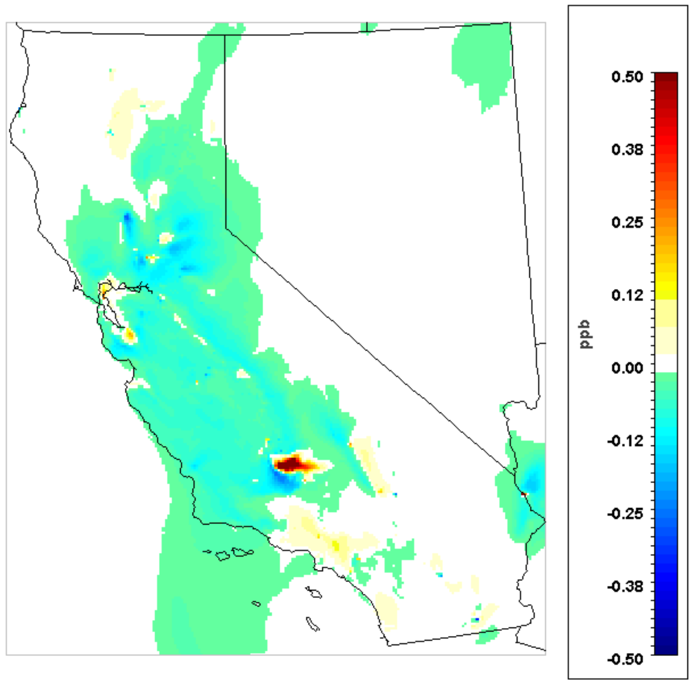


Figure 90: Difference in max 8-h average ozone in the Summer Commercial 2020 Case from the Base Case

Figure 91 displays the difference in 24-h $PM_{2.5}$ in the Summer Commercial Case from the Base Case. Impacts are characterized largely by reductions in concentrations over large areas of the State. Quantitatively, notable impacts range from -0.38 to +0.37 $\mu g/m^3$. Regions of improvement are observable in the Central Valley, SoCAB, the Bay Area, and the Sacramento metropolitan area, reflecting the importance of commercial sector emissions to PM and the low PM emitting nature of California's fossil power generators. As noted for the

Summer Residential 2020 Case, reductions in PM in the Central Valley have high importance to the State.

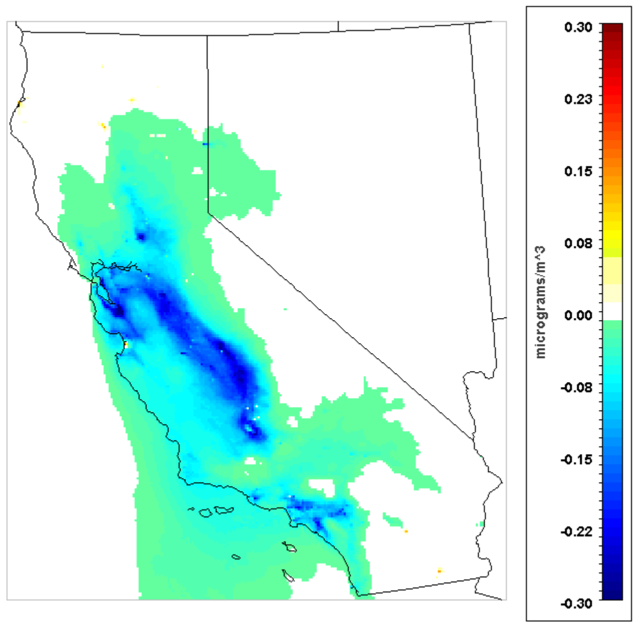


Figure 91: Difference in 24-h $PM_{2.5}$ in the Summer Commercial 2020 Case from the Base Case

Winter

Figure 92 displays the difference in maximum 8-hour average ozone in the Winter Commercial Case from the Base Case. Peak impacts in the scenario reach -0.21 to +1.50 ppb. Impacts differ significantly relative to the Summer Commercial 2020 Case as a result of variation in ozone formation in winter, as discussed above. Contrasting with the Summer Commercial Case, impacts largely include worsening, particularly in and around Bakersfield and highlight how electrification and renewable resource deployment can have varying AQ impacts due to grid dynamics during different seasons. It should again be noted that though ambient concentrations increase from the baseline, overall concentrations remain much lower than those during the modeled Summer episode and thus may not carry the same level of concern. It is also interesting to note that the reductions in commercial sector emissions do not yield improvements in winter in ground-level ozone.

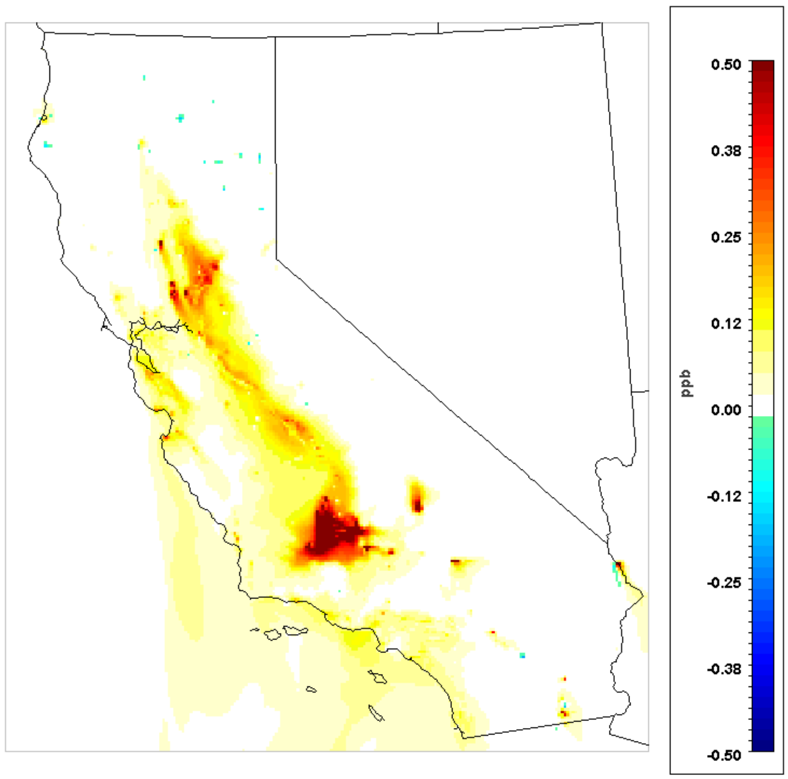


Figure 92: Difference in max 8-h average in the Winter Commercial 2020 Case from the Base Case

Figure 93 displays the difference in 24-h $PM_{2.5}$ in the Winter Commercial Case from the Base Case. Quantitatively, impacts range from -1.09 to +0.24 μg/m^3. Impacts are similar to PM impacts in the Summer Commercial Case including notable improvements in the Central Valley which may have increased importance in winter due to the difficulty many areas in the region experience meeting Federal PM Standards during winter months. Improvements are also observable in the Bay Area and SoCAB.

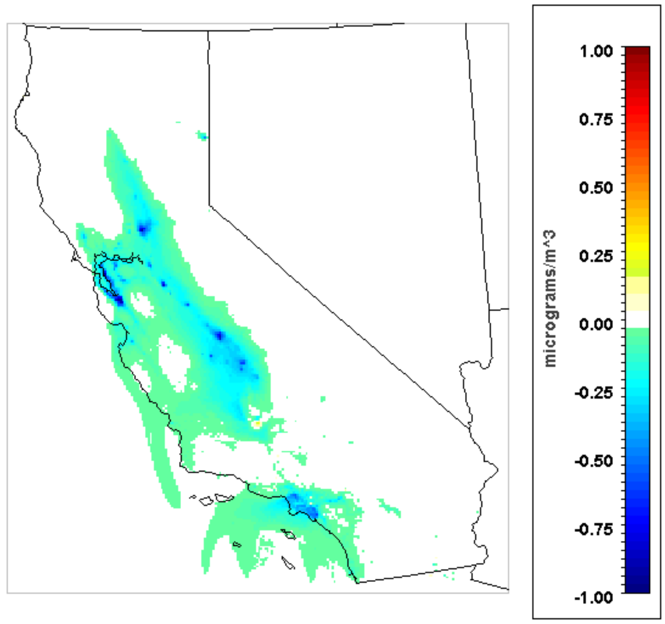


Figure 93: Difference in 24-h $PM_{2.5}$ in the Winter Commercial 2020 Case from the Base Case

7.3.1.3 Residential and Commercial (ResCom) 2020 Case

Summer

Figure 94 displays the difference in maximum 8 hour average ozone in the Summer ResCom 2020 Case from the Base Case. Quantitatively, impacts range from -1.00 to +0.83 ppb. As would be expected, the results are fairly additive in terms of both the Summer Residential and Summer Commercial Cases. Larger reductions, although still fairly moderate, occur across the State. Increases in ground-level concentrations adjacent to Bakersfield and in SoCAB are heightened as the scenario includes reductions from both residences and commercial buildings together and thus has a larger power demand.

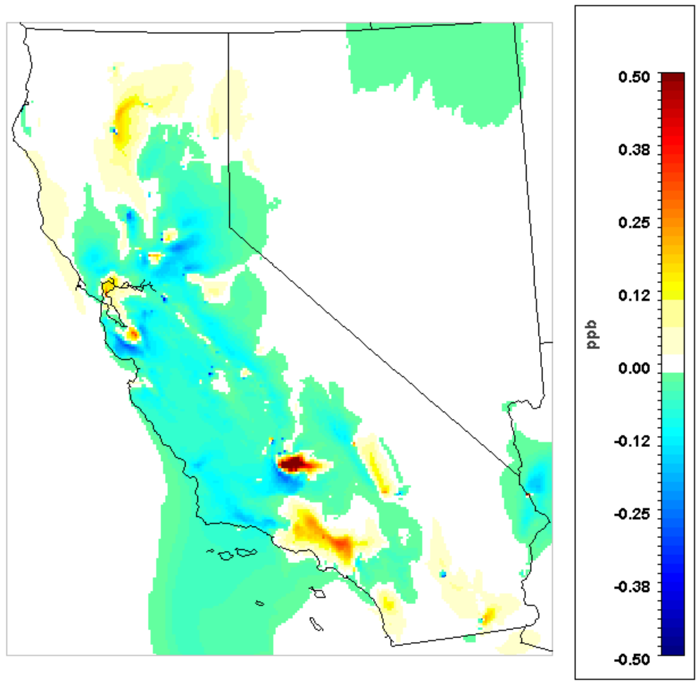


Figure 94: Difference in max 8-h average ozone in the Summer ResCom 2020 Case from the Base Case

Figure 95 displays the difference in 24-h $PM_{2.5}$ in the Summer ResCom 2020 Case from the Base Case. Following ozone trends, $PM_{2.5}$ impacts are generally additive when considering both Cases combined and include significant improvement in the Central Valley, SoCAB, Bay Area, and Sacramento regions. Localized worsening is visible in isolated grid cells representing generator locations throughout the State and reflects a larger increase in emissions than in singular cases due to higher new power demand combined. Quantitatively, impacts range from -0.43 to +0.43 μg/m^3.

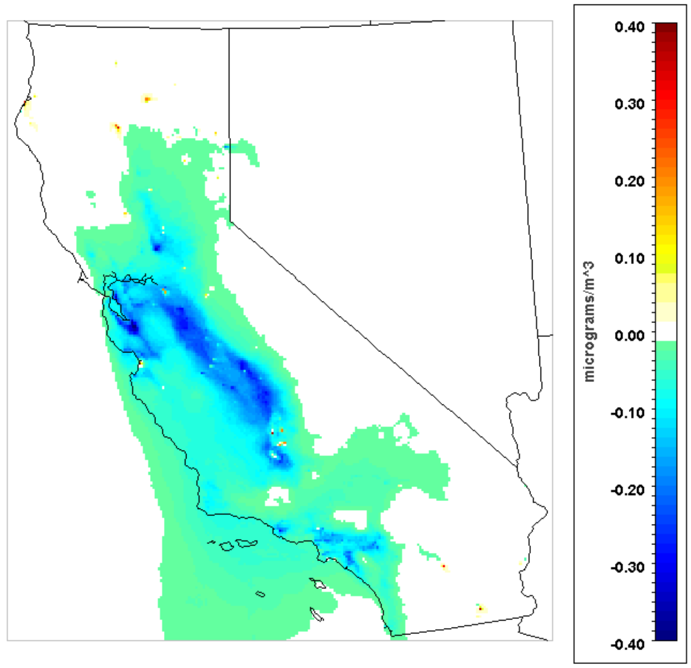


Figure 95: Difference in 24-h $PM_{2.5}$ in the Summer ResCom 2020 Case from the Base Case

Winter

Figure 96 displays the difference in maximum 8 hour average ozone in the Winter ResCom 2020 Case from the Base Case. Impacts range from -0.53 to +1.60 ppb ozone although the majority of impacts fall under + or -0.8 ppb. Results demonstrate an additive nature with regards to the individual winter Cases in that worsening in ozone concentrations are observed across the State, including several important areas. However, the reduced concern in terms of winter ozone levels mitigates some concerns as discussed above for the individual cases.

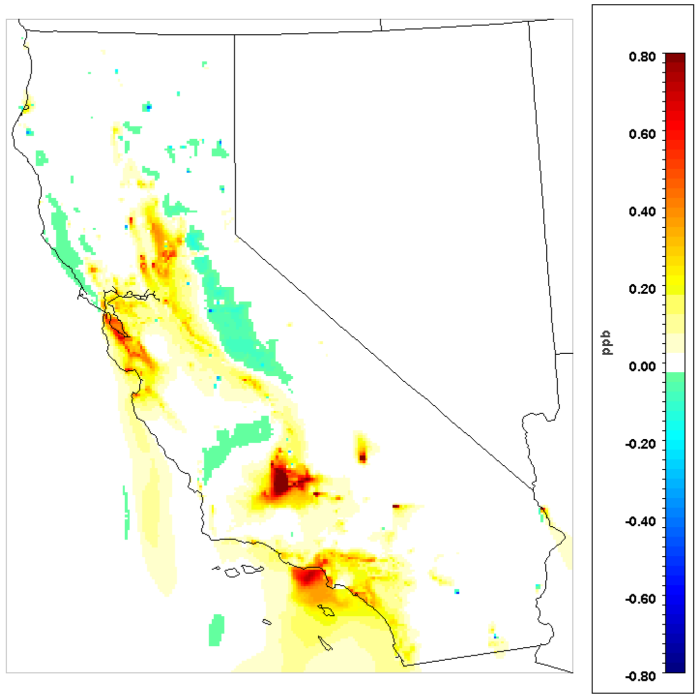


Figure 96: Difference in max 8-h average in the Winter ResCom 2020 Case from the Base Case

Figure 97 displays the difference in 24-h $PM_{2.5}$ in the Winter ResCom 2020 Case from the Base Case. Impacts include significant reductions that occur throughout the State and peak at -7.67 μg/m^3 with a range to +0.24 μg/m^3. Reductions are large enough to offset any increases from generators and thus the Case achieves a significant AQ benefit to the State in terms of improved winter PM concentrations. As previously stated, the benefits in the Central Valley are particularly important during winter months as PM levels often exceed health-based standards.

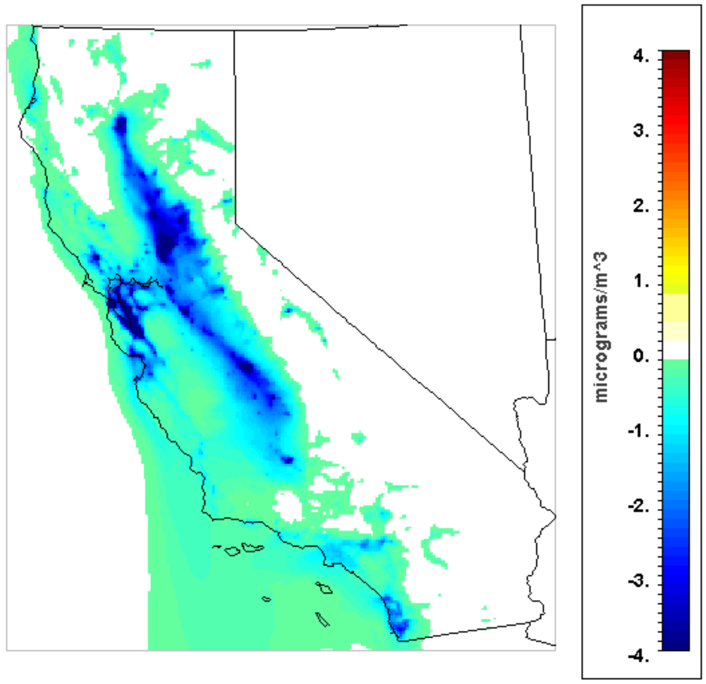


Figure 97: Difference in 24-h $PM_{2.5}$ in the Winter ResCom 2020 Case from the Base Case

7.3.1.4 Industrial Sector 2020 Case

Summer

Figure 98 displays the difference in maximum 8 hour average ozone in the Summer Industrial 2020 Case from the Base Case. The range of ozone perturbations is equivalent to -0.78 to +0.39 ppb. However, the majority of impacts are relatively minor in magnitude reflecting lesser emission reduction potential for the industrial sector compared to others, i.e., sector emissions are reduced by only 5% relative to the 30% and 29% reductions observed for the Commercial and Residential Cases. Impacts are also highly localized for both reductions and worsening and thus must be considered in terms of local communities that could be impacted.

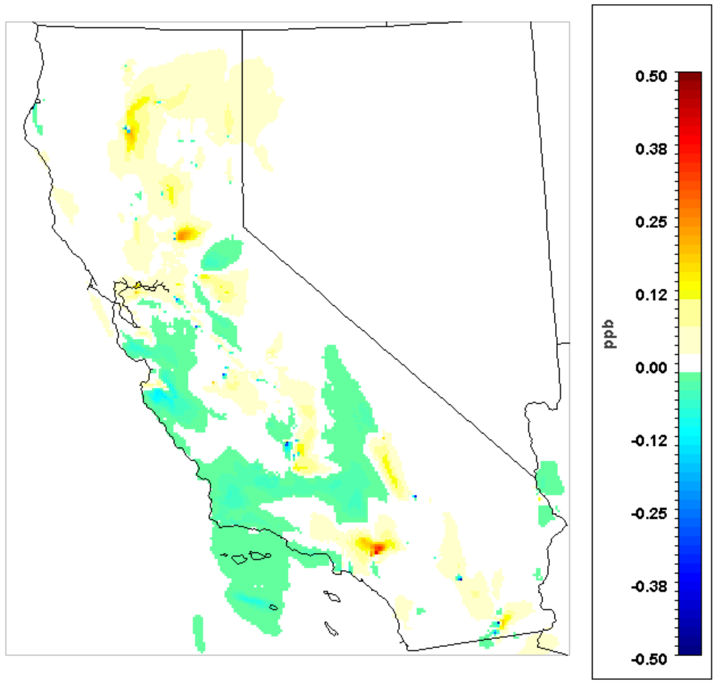


Figure 98: Difference in maximum 8-h average ozone in the Summer Industrial 2020 Case from the Base Case

Figure 99 displays the difference in 24-h $PM_{2.5}$ in the Summer Industrial 2020 Case from the Base Case. Spatially, improvements and worsening are observed in many areas of the state in highly localized patterns. Generally, areas of improvement include the Bay Area and SoCAB and areas of worsening include the lower Central Valley and in northern areas of the State including Sacramento. Quantitatively, impacts are fairly minor and range from -0.11 to +0.59 μg/m^3.

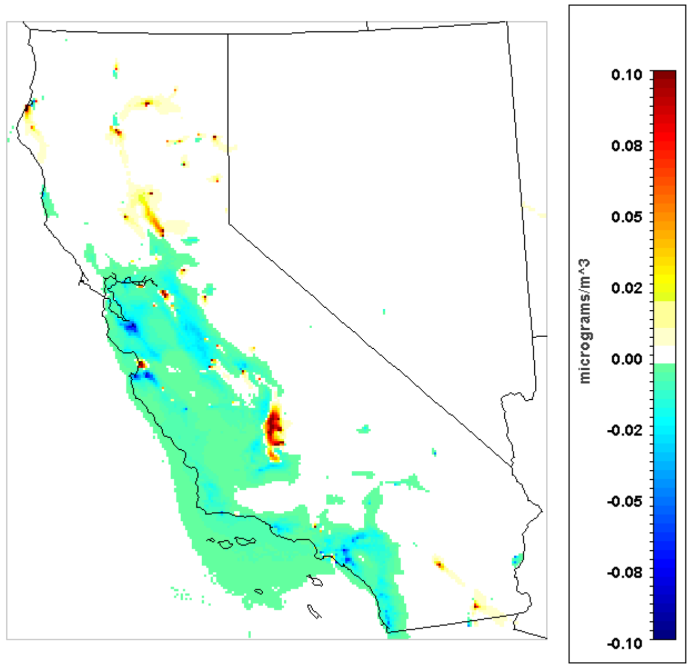


Figure 99: Difference in 24-h $PM_{2.5}$ in the Summer Industrial 2020 Case from the Base Case

Winter

Figure 100 displays the difference in maximum 8-hour average ozone in the Winter Industrial 2020 Case from the Base Case. Impacts are fairly minor as a result of small emission reductions and increases that arise from potential industrial sector electrification. In general, worsening trends are observed at minor magnitudes, i.e., ranging from -0.50 to +0.40 ppb although most impacts are generally less than 0.5 ppb.

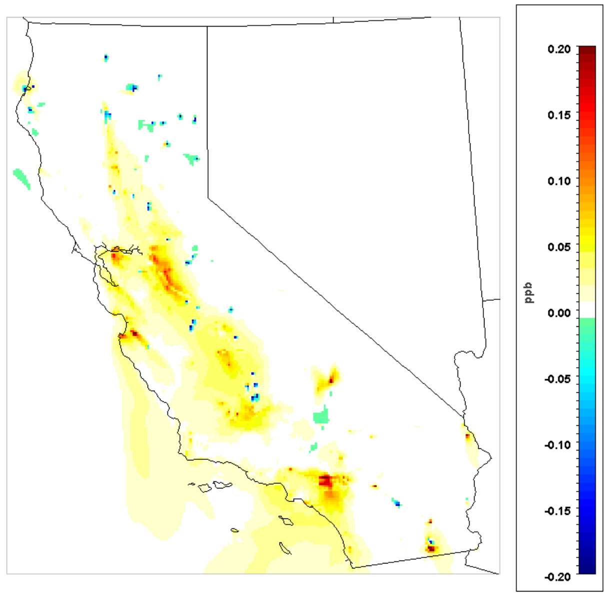


Figure 100: Difference in max 8-h average in the Winter Industrial 2020 Case from the Base Case

Figure 101 displays the difference in 24-h $PM_{2.5}$ in the Winter Industrial 2020 Case from the Base Case. Though emission reductions are relatively minor, impacts on PM include improvements in the central valley, most notably in and around Bakersfield. Localized areas of improvement also include the Bay Area and SoCAB. The magnitude of improvements is generally less than 0.2 $\mu g/m^3$ although some localized areas experience reductions around 0.3 $\mu g/m^3$. Given the small emission removal impacts are fairly substantial and occur in important areas. Thus, electrification of the industrial sector can provide AQ benefits in terms of winter PM reduction.

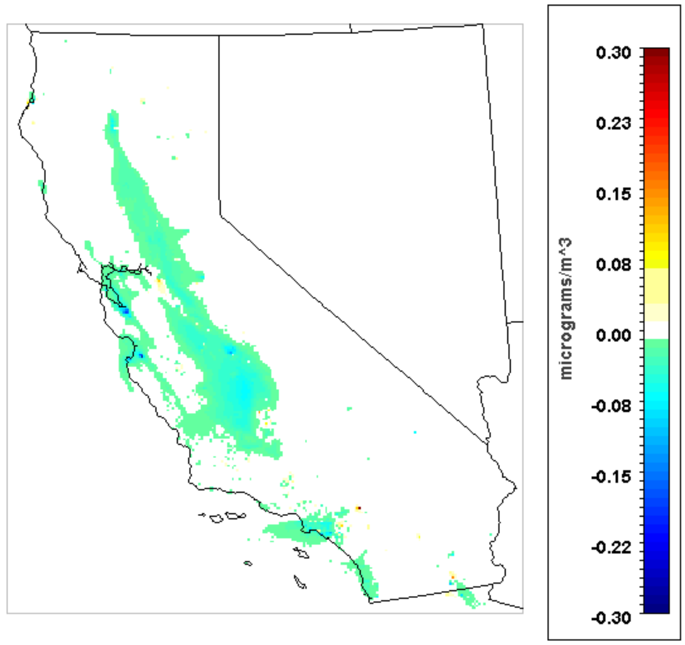


Figure 101: Difference in 24-h $PM_{2.5}$ in the Winter Industrial 2020 Case from the Base Case

7.3.1.5 Transportation Sector 2020 Case

Summer

Figure 102 displays the difference in ozone in the Summer Transportation 2020 Case from the Base Case. Quantitatively, impacts on maximum 8 hour average ozone range from -0.86 to +0.34 ppb. However, the magnitude of impact is generally moderate with the majority of reductions occurring at -0.5 ppb or less. The penetration of electric light duty vehicles (LDV) at fairly moderate levels (i.e., a 4% reduction in emissions) yields reductions in ozone in many regions of the State, including urban areas of SoCAB and the Bay Area. Bakersfield also experiences improvement although maximum impacts are may be attributed to the refinery complexes rather than vehicle tail pipe reductions. This is a reflection of both the moderate penetration level of electrification and the improvement in the traditional gasoline internal combustion engine LDVs. Additionally, assumed reductions in petroleum fuel infrastructure including refinery complexes may be the largest driver of AQ benefits.

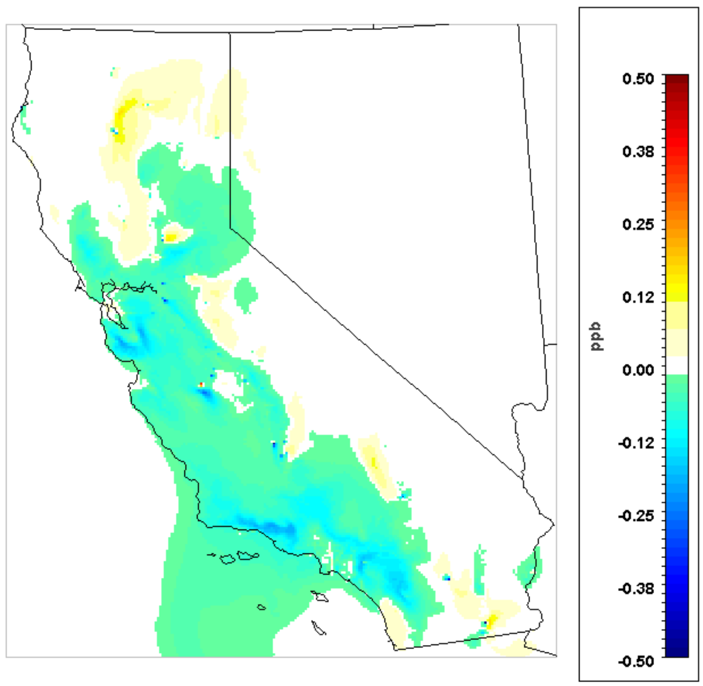


Figure 102: Difference in max 8-h average ozone in the Summer Transportation 2020 Case from the Base Case

Figure 103 displays the difference in 24-h $PM_{2.5}$ in the Summer Transportation 2020 Case from the Base Case. Impacts on PM are fairly minor and range from -0.26 to 0.43 μg/m^3 with the majority under -0.1 μg/m^3 or +0.1 μg/m^3. This is expected as LDVs and CA power generators in general do not emit large amounts of PM. Improvements are observed in SoCAB, San Diego, the Bay Area, and the northern area of the central valley. In contrast, localized worsening is observed in Bakersfield and other places in-state. Similarly to ozone, emission from petroleum fuel infrastructure should be considered as a primary driver of impacts in terms of reductions in addition to reductions from vehicle exhaust.

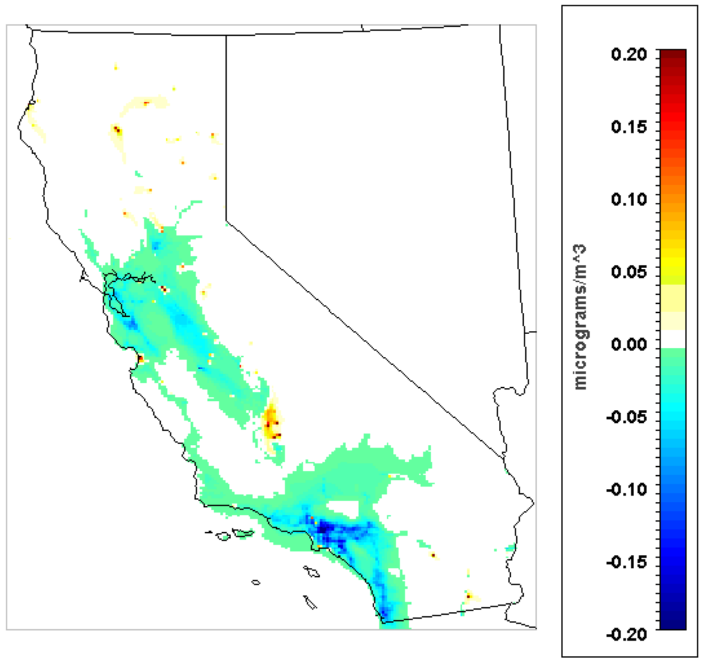


Figure 103: Difference in 24-h $PM_{2.5}$ in the Summer Transportation 2020 Case from the Base Case

Winter

Figure 104 displays the difference in maximum 8-hour average ozone in the Winter Transportation 2020 Case from the Base Case. Impacts range from -0.40 to +0.39 ppb, however the majority of perturbations are very minor. Generally, slight worsening occurs in the Bay Area and SoCAB but likely does not represent a significant concern due to winter ozone characteristics. The moderate impacts in the scenario reflect the small penetration of LDVs projected for 2020.

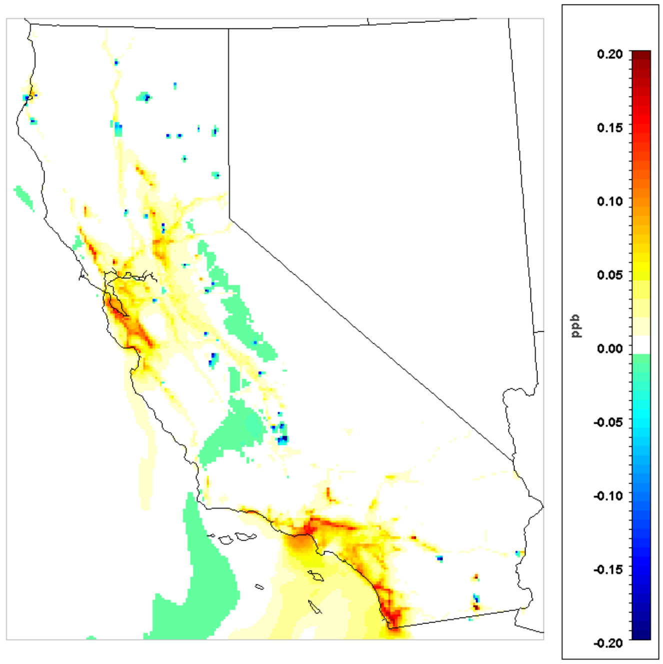


Figure 104: Difference in max 8-h average in the Winter Transportation 2020 Case from the Base Case

Figure 105 displays the difference in 24-h $PM_{2.5}$ in the Winter Transportation 2020 Case from the Base Case. In contrast to ozone impacts, significant improvements in $PM_{2.5}$ are observable across all three key regions of the state. Impacts range from -0.37 to +0.29 μg/m^3 although improvements dominate any localized worsening from generator increases. Similar to the summer episode, the improvements are relatively substantial relative to the small direct emission reduction inherent in the scenario and it would be expected that a larger penetration of electric vehicles could yield even greater AQ benefits.

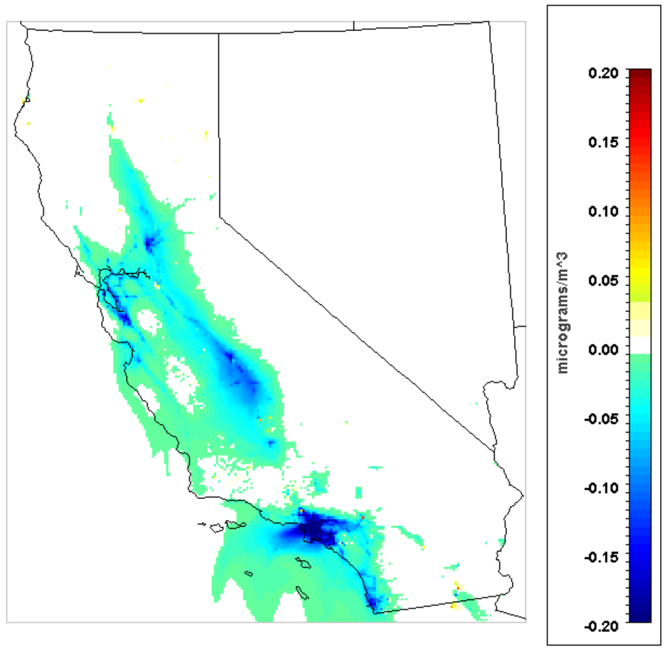


Figure 105: Difference in 24-h $PM_{2.5}$ in the Winter Transportation 2020 Case from the Base Case

7.3.1.6 Residential, Commercial, and Transportation (ResComTra) 2020 Case

Summer

Figure 106 displays the difference in maximum 8 hour average ozone in the Summer ResComTra 2020 Case from the Base Case. Impacts range from -1.54 to +0.80 ppb although most fall between -1 to +1 ppb. Spatially, impacts follow similar trends to those observed in the individual cases and include moderate improvements over large areas of the state while some areas experience worsening that has higher quantitative values but cover less total area. In particular, the southern Central Valley (i.e., adjacent to the Bakersfield region) experiences the highest increase in ground-level ozone concentrations which is a concern given the existing poor AQ conditions the area experiences.

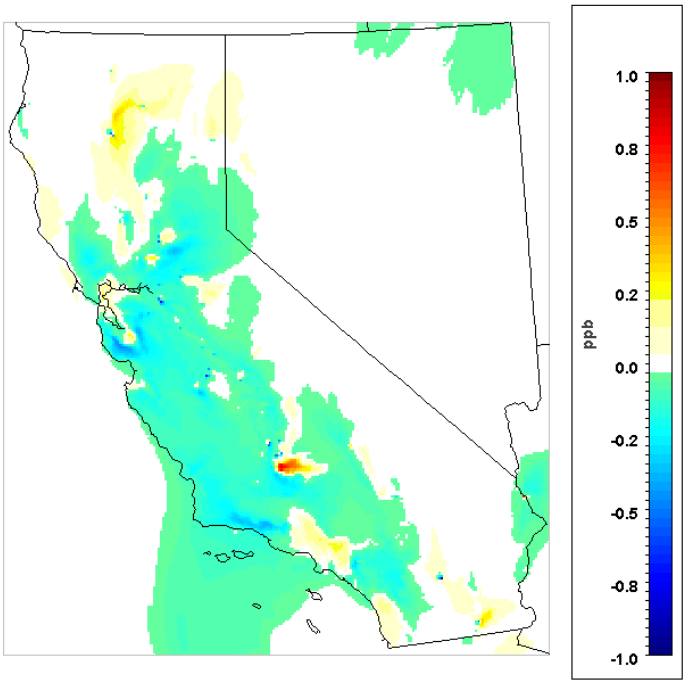


Figure 106: Difference in max 8-h average ozone in the Summer ResComTra 2020 Case from the Base Case

Figure 107 displays the difference in 24-h $PM_{2.5}$ in the Summer ResComTra 2020 Case from the Base Case. Impacts range from -0.54 to +0.74 μg/m^3. Much of the State experiences improvements although some localized worsening is evident. The key areas in terms of AQ, i.e., SoCAB, Bay Area, and Central Valley all experience general improvements.

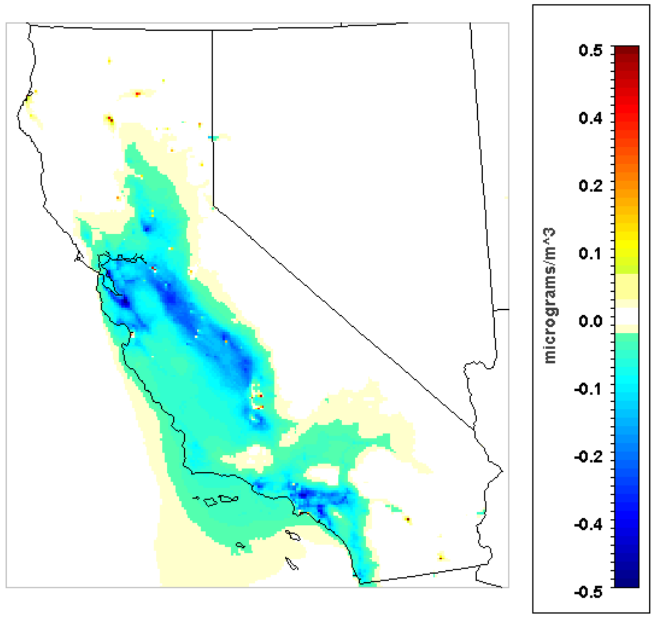


Figure 107: Difference in 24-h $PM_{2.5}$ in the Summer ResComTra2020 Case from the Base Case

Winter

Figure 108 displays the difference in maximum 8 hour average ozone in the Winter ResComTra 2020 Case from the Base Case. Impacts are generally characterized by small to moderate increases in ozone concentrations with a range of -0.91 to +1.61 ppb. Impacts are additive in nature in relation to the individual cases and include generalized worsening in the SoCAB, Bakersfield, Bay Area, and Sacramento regions. Small areas of moderate reductions also occur.

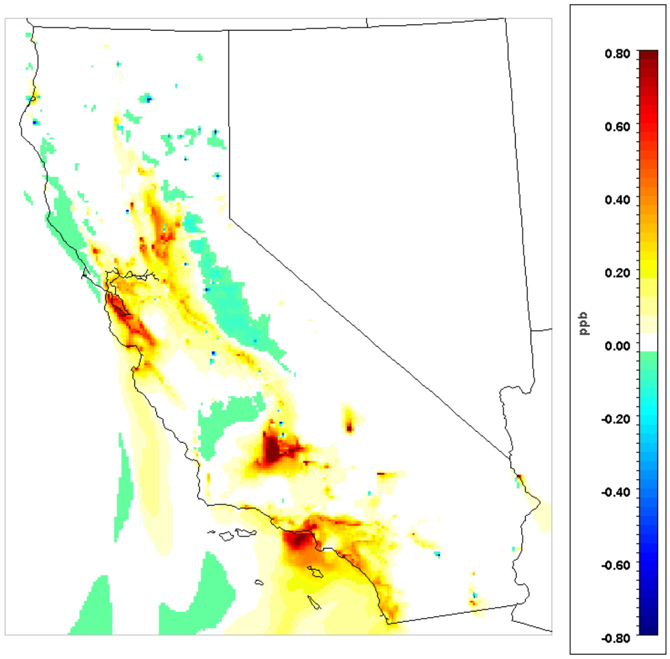


Figure 108: Difference in max 8-h average in the Winter ResComTra 2020 Case from the Base Case

Figure 109 displays the difference in 24-h $PM_{2.5}$ in the Winter ResComTra 2020 Case from the Base Case. Impacts are characterized largely by significant reductions in PM with notable improvements in the Central Valley, Bay Area, and Sacramento regions. Impacts range from -7.78 to +0.23 μg/m^3.

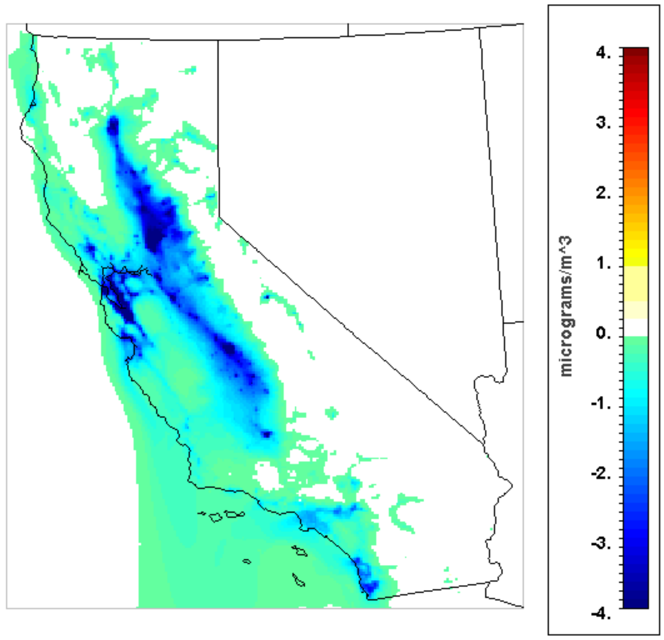


Figure 109: Difference in 24-h $PM_{2.5}$ in the Winter ResComTra 2020 Case from the Base Case

7.3.1.7 Residential, Commercial, Transportation, and Industrial (ResComTraInd) 2020 Case

Summer

Figure 110 displays the difference in maximum 8-h average ozone in the Summer ResComTraInd 2020 Case from the Base Case. Impacts range from -2.24 to +0.93 ppb. Spatially impacts resemble the patterns of individual cases that have been previously discussed, although with higher magnitude in terms of increases and decreases.

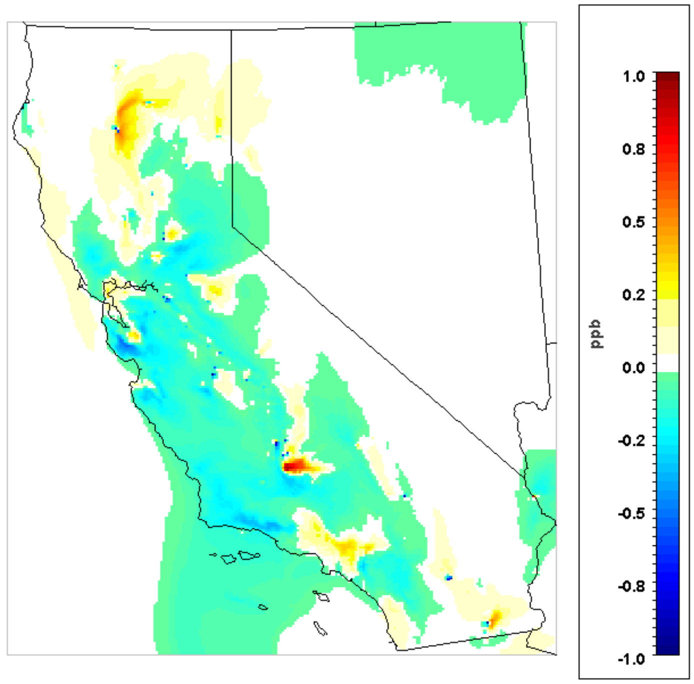


Figure 110: Difference in max 8-h average ozone in the Summer ResComTraInd 2020 Case from the Base Case

Figure 111 displays the difference in 24-h $PM_{2.5}$ in the Summer ResComTraInd 2020 Case from the Base Case. As would be expected, the impacts are generally additive for cases and include larger areas of improvement concurrent with localized areas of worsening. Significant areas of improvement with importance include the Bay Area, Central Valley, and SoCAB. Quantitatively, impacts range from -0.64 to +1.27 μg/m^3.

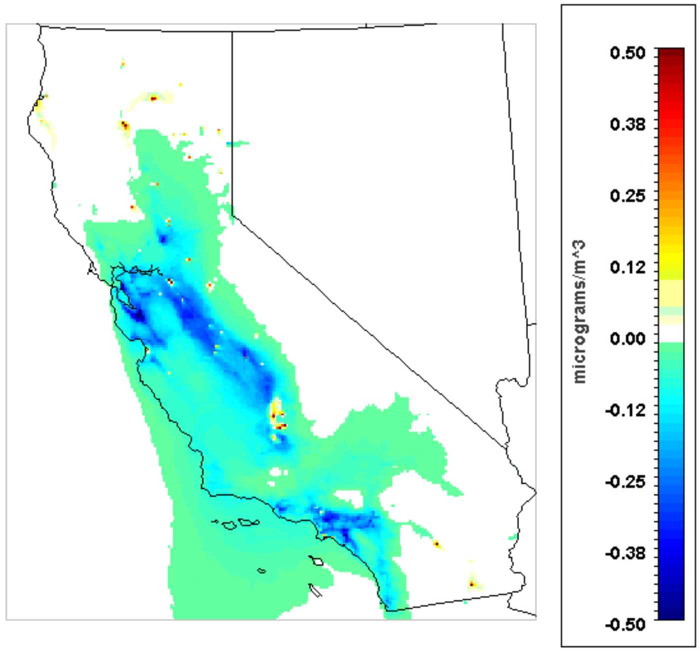


Figure 111: Difference in 24-h $PM_{2.5}$ in the Summer ResComTraInd 2020 Case from the Base Case

Winter

Figure 112 displays the difference in maximum 8 hour average ozone in the Winter ResComTraInd 2020 Case from the Base Case. Impacts range from -1.37 to 1.69 ppb with the majority of impacts including slight to moderate increases in ground-level ozone. Spatially, impacts resemble those from individual cases and are represented generally as additive.

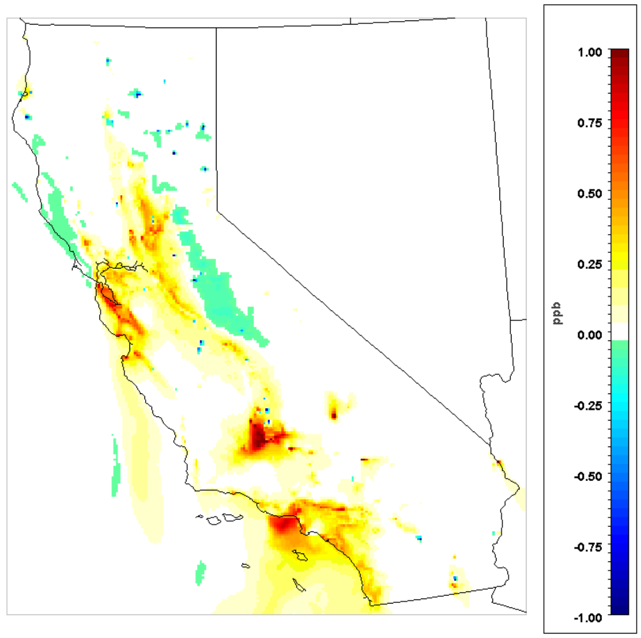


Figure 112: Difference in max 8-h average in the Winter ResComTraInd 2020 Case from the Base Case

Figure 113 displays the difference in 24-h $PM_{2.5}$ in the Winter ResComTraInd 2020 Case from the Base Case. Impacts are generally characterized by significant improvements that range from -7.83 to +0.37 μg/m^3. With similarity to the Winter ResComTra 2020 Case, the effects of emission reductions in the Winter ResComTraInd 2020 Case yield significant improvements in PM levels in important regions of California.

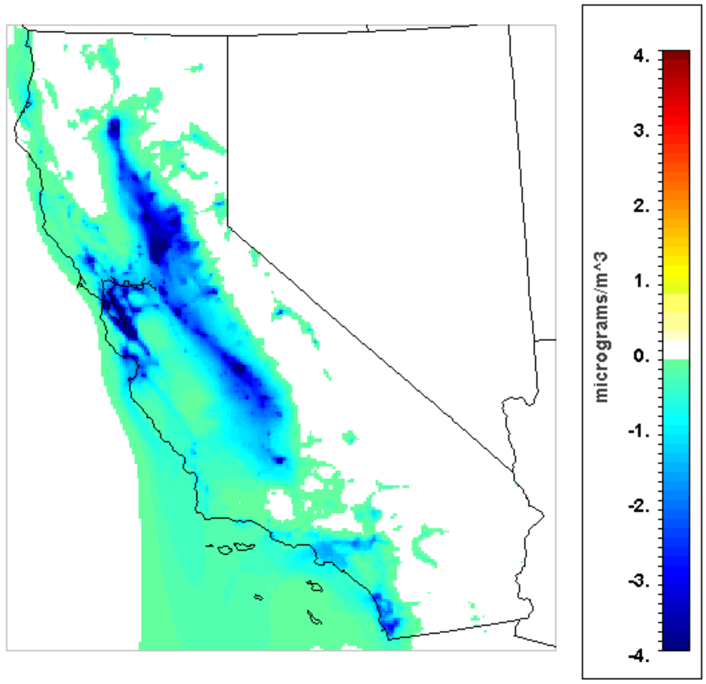


Figure 113: Difference in 24-h $PM_{2.5}$ in the Winter ResComTraInd 2020 Case from the Base Case

7.3.1.8 Summary of 2020 Cases

Table 54 displays the peak impacts on 8 hour maximum average ozone and 24 hour average $PM_{2.5}$ for the Summer 2020 scenarios.

Table 54: Summary of peak impacts on 8-hr max ozone and 24-h $PM_{2.5}$ for Summer 2020 Cases

Summer Case	8-hr Ozone [ppb]	24-hr $PM_{2.5}$ [$\mu g/m^3$]
Res 2020	-1.92 to +1.14	-0.08 to +0.38
Com 2020	-0.97 to +1.75	-0.38 to +0.37
ResCom 2020	-2.47 to +1.78	-0.43 to +0.43
Ind 2020	-2.23 to +1.26	-0.11 to +0.59
Tra 2020	-1.98 to +1.78	-0.26 to +0.43
ResComTra 2020	-4.23 to +1.78	-0.54 to +0.74
ResComTraInd 2020	-6.20 to +1.89	-0.64 to +1.27

Table 55 displays the peak impacts on 8 hour maximum average ozone and 24 hour average $PM_{2.5}$ for the Winter 2020 scenarios.

Table 55: Summary of peak impacts on 8-hr max ozone and 24-h $PM_{2.5}$ for Winter 2020 Cases

Winter Case	8-hr Ozone [ppb]	24-hr $PM_{2.5}$ [$\mu g/m^3$]
Res 2020	-0.36 to +0.75	-7.30 to +0.06
Com 2020	-0.21 to +1.51	-1.09 to +0.24
ResCom 2020	-0.53 to +1.60	-7.67 to +0.24
Ind 2020	-0.50 to +0.40	-0.31 to +0.34
Tra 2020	-0.40 to +0.39	-0.37 to +0.29
ResComTra 2020	-0.91 to +1.61	-7.78 to +0.23
ResComTraInd 2020	-1.37 to +1.69	-7.83 to +0.37

Impacts on PM and ozone are fairly minor for all electrification scenarios in 2020 and reflect a moderate electrification potential from current for many sectors of study. As would be expected, combination cases achieve both the largest improvements from reductions in emissions occurring from multiple sectors but also the largest increases from novel power generation due to larger loads (e.g., the Summer ResComTraInd Case experiences both a 6 ppb reduction and 1.2 ppb increase).

Sectors identified as having higher potential for electrification in 2020 include residential and commercial energy conversion. The electrification of the residential and commercial sectors in tandem with renewable resource deployment moderately improves ozone and $PM_{2.5}$ over some areas of the State in 2020 via emission reductions from gas-fired technologies in those sectors. In particular, residential electrification improves winter-time PM in northern and central California.

Impacts vary between sectors both in terms of magnitude and spatial area of impact, reflecting source distributions, emissions intensities of displaced technologies, electrification potential, etc. In 2020 the industrial sector case achieves the largest reduction in terms of peak max 8-hr ozone while the commercial sector case experiences the largest peak reduction in 24-h $PM_{2.5}$. However, peak impacts fail to demonstrate differences in spatial distribution of impacts, i.e., the transportation case achieves more widespread improvements in ground-level ozone than the industrial case despite having a lower peak value. In the absence of complementary technologies/strategies designed to mitigate increased electricity loads areas of AQ worsening occur for both ozone and $PM_{2.5}$ as a result of increased generator emissions in 2020.

The results highlight the difference in season and pollutant for impacts in scenarios, i.e., PM impacts in winter scenarios are largely beneficial while ozone impacts are often associated with worsening. Contrastingly, both summer ozone and PM benefits and worsening occur. Thus, electrification strategies should take into account seasonal factors to maximize AQ benefits.

7.3.2 Cases for 2030

Table 56 displays the list of developed cases and emission reductions inherent in each case by sector excluding the power sector for 2030. The longer horizon facilitates larger penetrations of electrification technologies in major energy sectors and thus greater emission reductions. The Residential and Commercial Sectors experience the greatest impacts in relative terms. In addition to the Tra 2030 Case, a Case involving the smart charging of vehicles is examined to assess how controlled vehicle charging can be used to avoid emission penalties incurred from the immediate charging scenario. For example, immediate charging of vehicles results in increased power demand during peak demand hours which can result in increased ramping and emissions from peaking units. The smart charging Case is designated the Tra Smart 2030 Case.

Table 56: Reductions in energy sector emissions for 2030 Cases

2030		**Sector Emissions Reduction**		
Case	Residential	Commercial	Industrial	Transportation
Res 2020	64.94%	----	----	----
Com 2020	----	64.58%	----	----
Ind 2020	----	----	27.57%	----
Tra 2020	----	----	----	10.65%
ResCom 2020	64.94%	64.58%	----	----
ResComTra 2020	64.94%	64.58%	----	10.65%
ResComTraInd 2020	64.94%	64.58%	27.57%	10.65%

7.3.2.1 Residential Sector 2030 Case

Summer

Figure 114 displays the difference in maximum 8 hour average ozone in the Summer Residential 2030 Case from the Base Case. Quantitatively, impacts range from -0.96 to +1.85 ppb. In general, impacts are fairly moderate with improvements in regions associated with large urban populations coinciding with high concentrations of residential source emissions and worsening in areas corresponding with generator locations. Concentration reductions are notable in the Bay Area and SoCAB while Northern California experiences worsening.

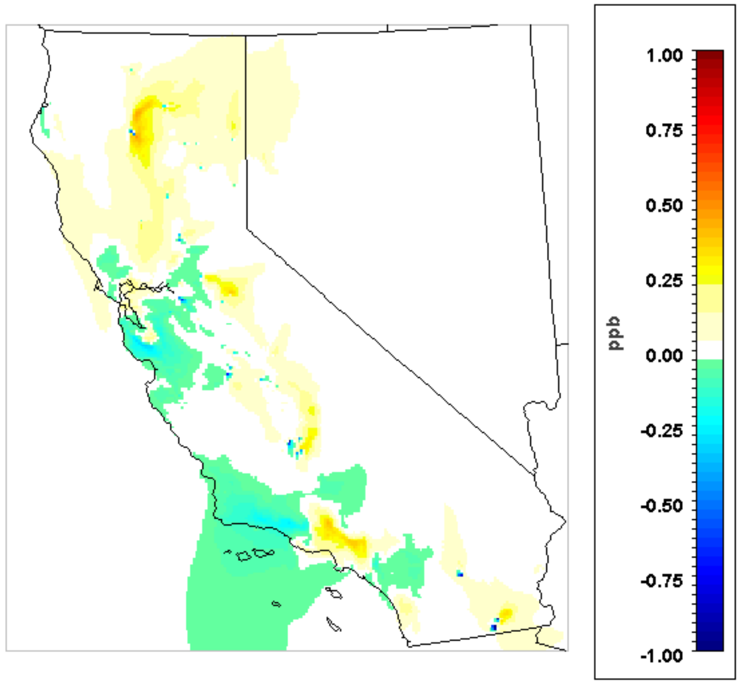


Figure 114: Difference in maximum 8 hour average ozone in the Summer Residential 2030 Case from the Base Case

Figure 115 displays the difference in 24-h $PM_{2.5}$ in the Summer Residential 2030 Case from the Base Case. Quantitatively, impacts range from -0.19 to +1.13 μg/m^3. Impacts, both increases and decreases, are fairly minor throughout the State. A notable area of worsening occurs in the Bakersfield region.

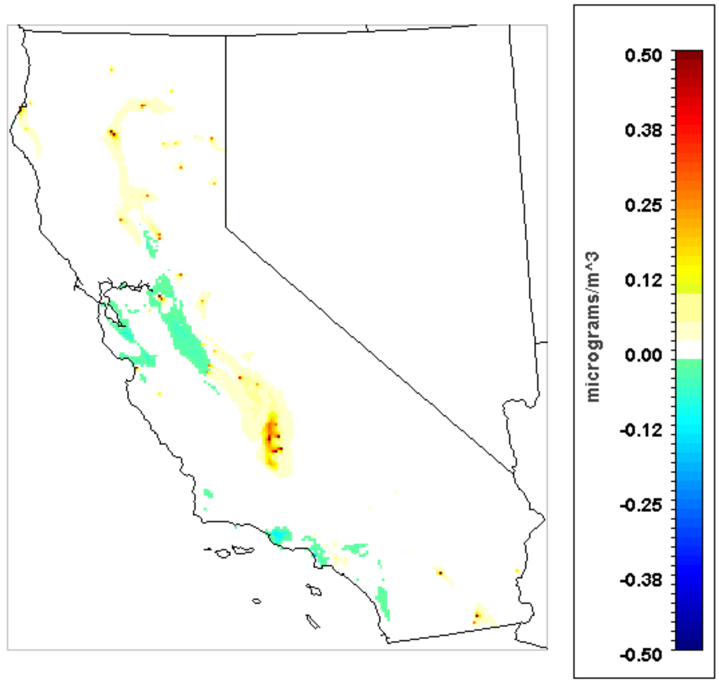


Figure 115: Difference in 24-h $PM_{2.5}$ in the Summer Residential 2030 Case from the Base Case

Winter

Figure 116 displays the difference in maximum 8-hour average ozone in the Winter Residential 2030 Case from the Base Case. Quantitatively, impacts range from -0.78 to +1.82 ppb. Notable areas of concentration increases occur in the SoCAB and the Bay Area. Contrastingly, reductions occur throughout the Central Valley. Generally, impacts tend to occur around 1 ppb and reflect winter ozone formation dynamics.

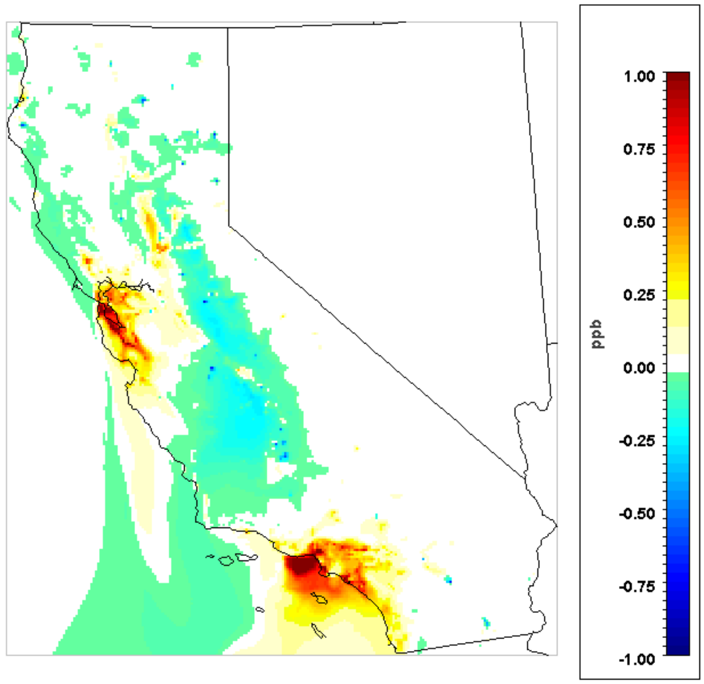


Figure 116: Difference in maximum 8 hour average ozone in the Winter Residential 2030 Case from the Base Case

Figure 117 displays the difference in 24-h $PM_{2.5}$ in the Winter Residential 2030 Case from the Base Case. Quantitatively, impacts range from -13.33 to +0.25 μg/m^3. Spatially, reductions are most notable in Central California, beginning in Bakersfield and continuing north through the Bay Area and Sacramento. The magnitude of peak reductions is substantial (-13 μg/m^3) while no notable areas of worsening occur. Further, as previously mentioned for 2020 Cases reductions in PM in many of these areas is desirable due to currently high winter time PM levels. Thus, the AQ benefits of the Winter Residential 2030 Case are prominent.

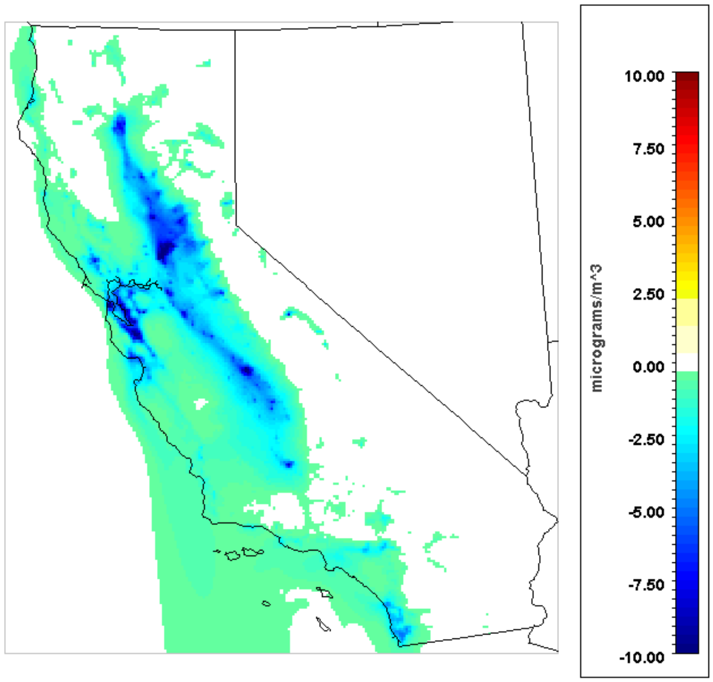


Figure 117: Difference in 24-h $PM_{2.5}$ in the Winter Residential 2030 Case from the Base Case

7.3.2.2 Commercial Sector 2030 Case

Summer

Figure 118 displays the difference in maximum 8 hour average ozone in the Summer Commercial 2030 Case from the Base Case. Quantitatively, impacts range from -1.33 to +0.44 ppb. Significant improvements are visible throughout the State including Sacramento and the Bay Area. SoCAB experiences reductions, although at a lesser magnitude. With similarity to the ozone results, the Bakersfield region experiences worsening, including peak concentration increases due to effects on generator emissions.

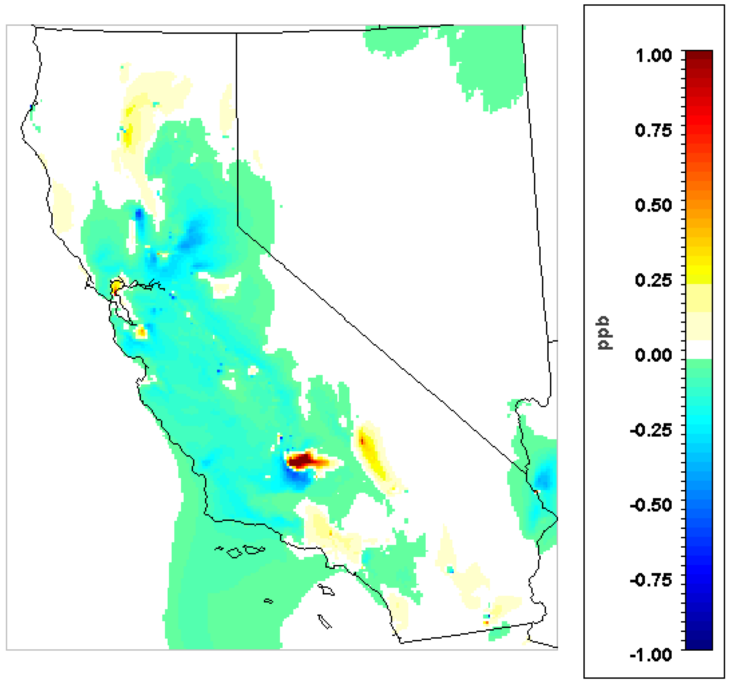


Figure 118: Difference in maximum 8 hour average ozone in the Summer Commercial 2030 Case from the Base Case

Figure 119 displays the difference in 24-h $PM_{2.5}$ in the Summer Commercial 2030 Case from the Base Case. Quantitatively, impacts range from -0.99 to +0.48 µg/m^3. Generally, impacts are described by reductions in concentrations that include many important areas of the State, i.e., Bay Area, SoCAB, Central Valley, Sacramento. In particular, reductions in the Central Valley cover a large area and include areas experiencing peak reductions.

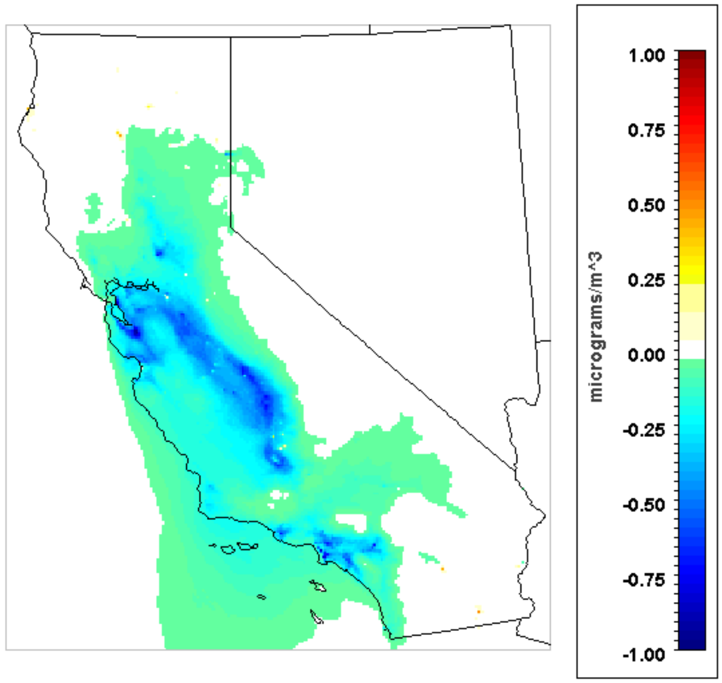


Figure 119: Difference in 24-h $PM_{2.5}$ in the Summer Commercial 2030 Case from the Base Case

Winter

Figure 120 displays the difference in maximum 8 hour average ozone in the Winter Commercial 2030 Case from the Base Case. Quantitatively, impacts range from -0.49 to +3.19 ppb. Generally, impacts include an area of worsening centered in and around Bakersfield and extending north through the Central Valley, the Bay Area, and Sacramento.

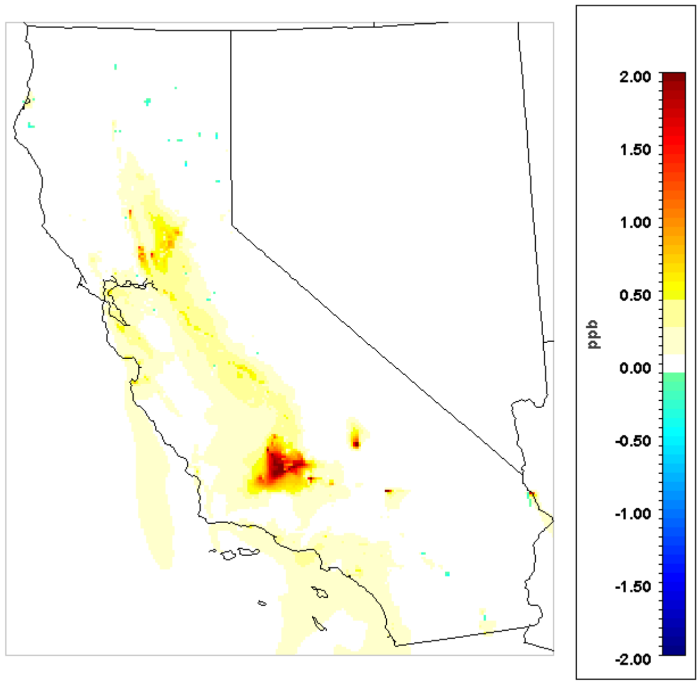


Figure 120: Difference in maximum 8 hour average ozone in the Winter Commercial 2030 Case from the Base Case

Figure 121 displays the difference in 24-h $PM_{2.5}$ in the Winter Commercial 2030 Case from the Base Case. Quantitatively, impacts range from -2.69 to +0.19 μg/m^3. Spatially, reductions are most prevalent in the Central Valley, Bay Area, and the SoCAB. While less than the same case for the Residential sector, no significant areas of worsening occur and the outcome of the Winter Com2030 case largely represents an AQ improvement for $PM_{2.5}$. The case also serves as a good example of the reversal of impacts in winter relative to summer for both ozone and PM.

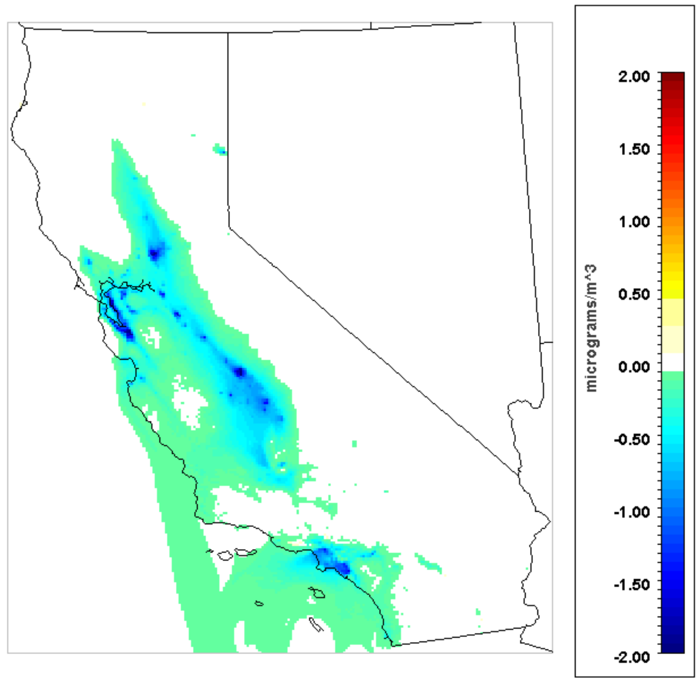


Figure 121: Difference in 24-h $PM_{2.5}$ in the Winter Commercial 2030 Case from the Base Case

7.3.2.3 Residential and Commercial (ResCom) 2030 Case

Summer

Figure 122 displays the difference in maximum 8 hour average ozone in the Summer ResCom 2030 Case from the Base Case. Quantitatively, impacts range from -2.24 to +1.93 ppb. Impacts are generally additive to the two singular cases, i.e., residential and commercial, and include large areas of improvement with a lower magnitude (less than or equal to 0.5 ppb). In contrast, localized worsening occurs with a higher magnitude (1 to 2 ppb). A notable area of increase includes Bakersfield and the Northern area of the state.

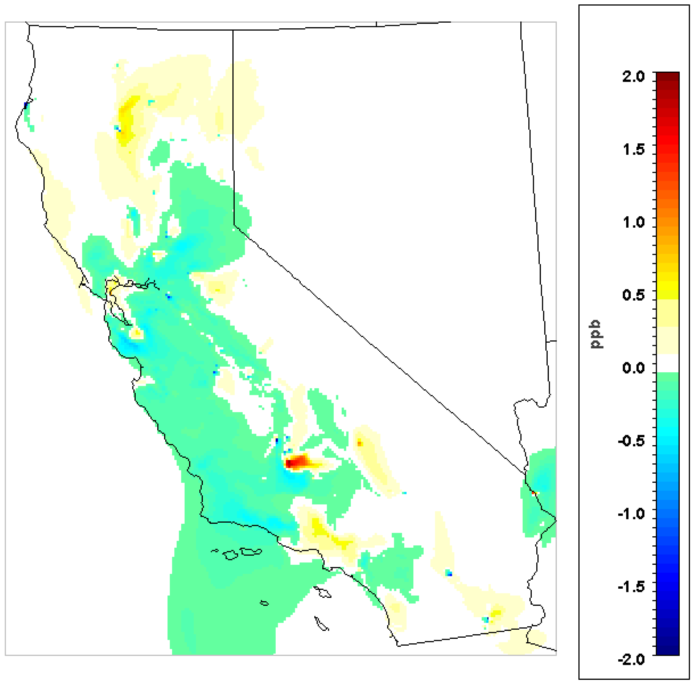


Figure 122: Difference in maximum 8 hour average ozone in the Summer Residential Commercial 2030 Case from the Base Case

Figure 123 displays the difference in 24-h $PM_{2.5}$ in the Summer ResCom 2030 Case from the Base Case. Quantitatively, impacts range from -1.07 to +1.42 μg/m^3. Impacts are largely characterized by improvements over large areas of the State, including the SoCAB, the Central Valley, and the Bay Area. Small, localized increases occur in the same location as ozone increases but are dominated by improvements.

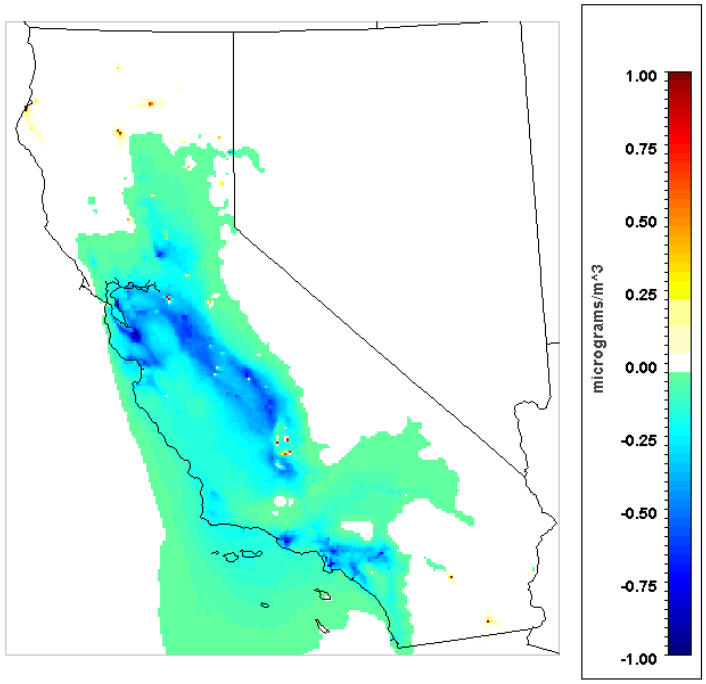


Figure 123: Difference in 24-h $PM_{2.5}$ in the Summer Residential Commercial 2030 Case from the Base Case

Winter

Figure 124 displays the difference in maximum 8 hour average ozone in the Winter ResCom 2030 Case from the Base Case. Quantitatively, impacts range from -3.56 to +3.31 ppb. Spatially, impacts are fairly additive for the two cases and include prominent areas of worsening in many of the regions of the State that currently experience poor AQ, i.e., SoCAB, Bay Area, Bakersfield, and Sacramento. However, the winter time ozone impacts are less of a concern due to seasonal differences discussed in the results section regarding the 2020 Winter scenarios.

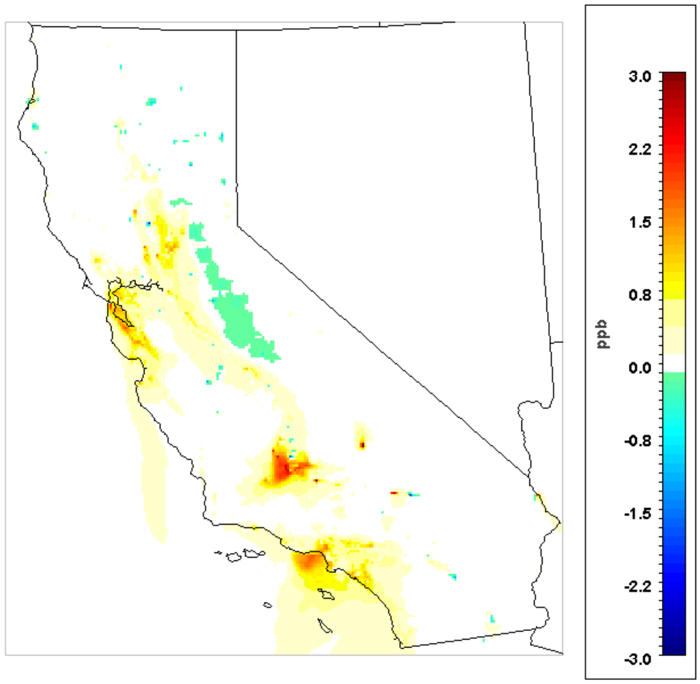


Figure 124: Difference in maximum 8 hour average ozone in the Winter Residential Commercial 2030 Case from the Base Case

Figure 125 displays the difference in 24-h $PM_{2.5}$ in the Winter ResCom 2030 Case from the Base Case. Quantitatively, impacts range from -14.51 to +0.45 μg/m^3. As the ResCom Case is largely additive, the largest driver of impacts is associated with emission reductions in the Residential sector. As discussed in the Residential 2030 Case, the magnitude of improvements are substantial and occur in important areas for winter time PM levels, i.e., the Central Valley. Thus, the Winter ResCom2030 Case achieves important improvements in AQ in terms of $PM_{2.5}$.

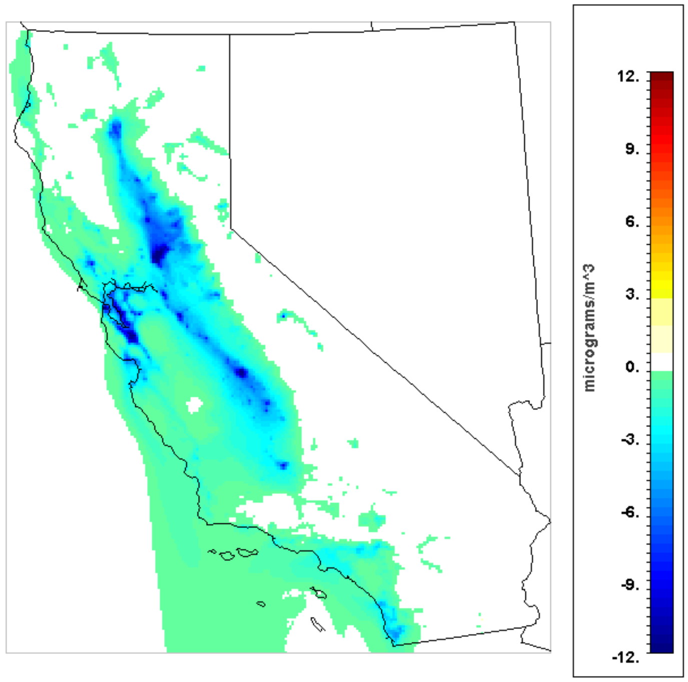


Figure 125: Difference in 24-h $PM_{2.5}$ in the Winter Residential Commercial 2030 Case from the Base Case

7.3.2.4 Industrial Sector 2030 Case

Summer

Figure 126 displays the difference in maximum 8 hour average ozone in the Summer Industrial 2030 Case from the Base Case. Quantitatively, impacts range from -4.63 to +1.50 ppb. The significant amounts of novel power required to electrify the industrial sector yield large NO_x emission increases from generators that drive worsening of AQ in some areas of the State. Spatially, impacts include significant areas of worsening in northern sections of the state including the Central Valley, Bay Area, and Sacramento. For the most, part concentration reductions occur in the southern areas of the state including SoCAB from reductions in NO_x associated with major industrial sources.

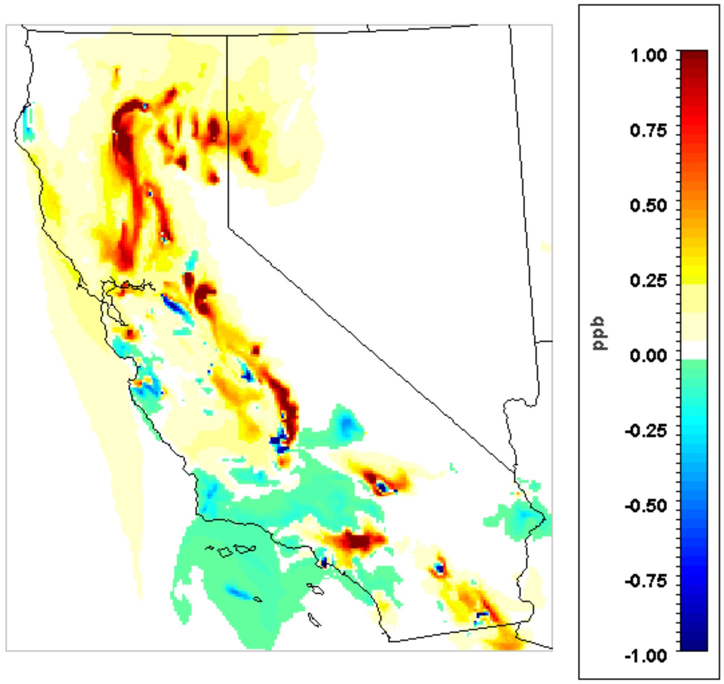


Figure 126: Difference in maximum 8 hour average ozone in the Summer Industrial 2030 Case from the Base Case

Figure 127 displays the difference in 24-h $PM_{2.5}$ in the Summer Industrial 2030 Case from the Base Case. Quantitatively, impacts range from -0.48 to +4.34 μg/m^3. Slight reductions in $PM_{2.5}$ occur in SoCAB and the Bay Area, largely along the coast. A notable area of increased $PM_{2.5}$ includes the Bakersfield region. Additional increases occur localized to generators in locations throughout the State.

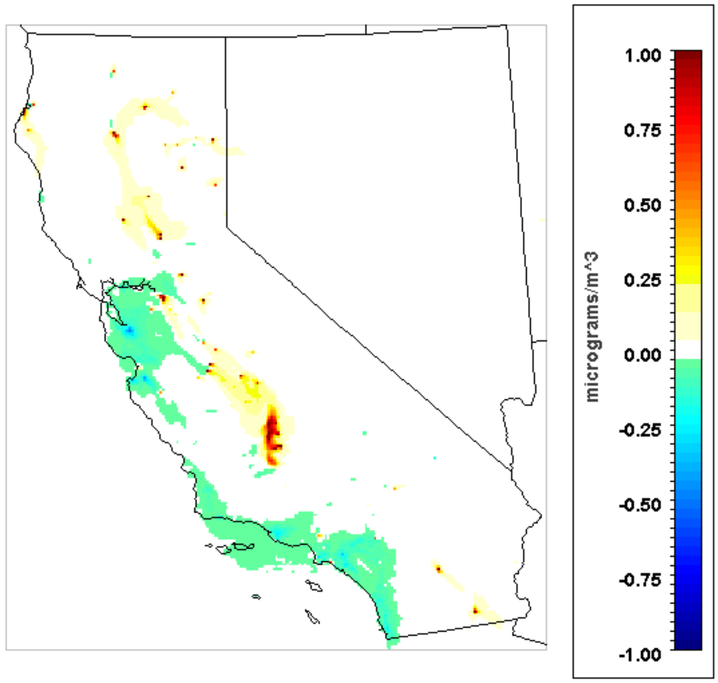


Figure 127: Difference in 24-h $PM_{2.5}$ in the Summer Industrial 2030 Case from the Base Case

Winter

Figure 128 displays the difference in maximum 8 hour average ozone in the Winter Industrial 2030 Case from the Base Case. Quantitatively, impacts range from -10.18 to +1.52 ppb. Although peak reductions are highest in terms of achieved reductions, impacts are spatially highly localized to both generator and industrial emission sites. Additionally, worsening occurs in several regions including SoCAB.

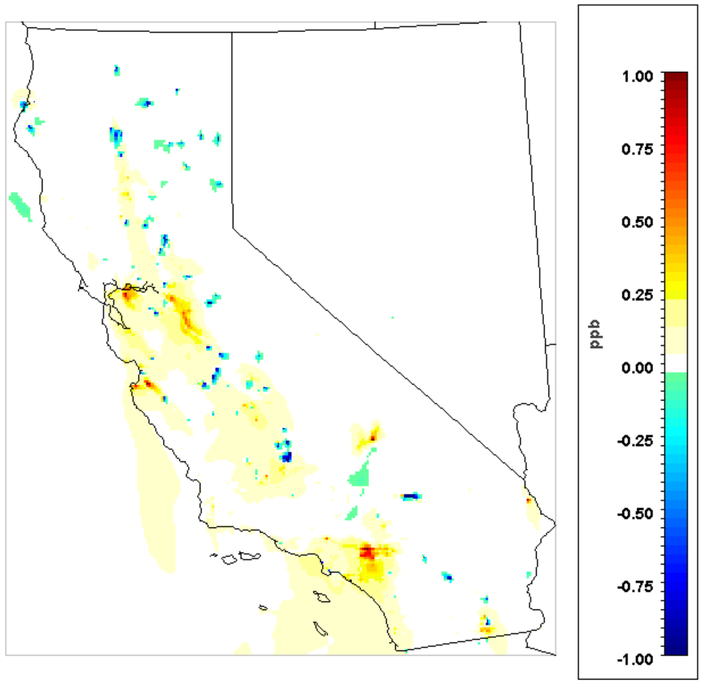


Figure 128: Difference in maximum 8 hour average ozone in the Winter Industrial 2030 Case from the Base Case

Figure 129 displays the difference in 24-h $PM_{2.5}$ in the Winter Industrial 2030 Case from the Base Case. Quantitatively, impacts range from 1.21 to 1.29 μg/m³. Generally, impacts are fairly minor and include some reductions in the Bay Area, Central Valley, and coastal parts of SoCAB. Worsening is limited to generators near Bakersfield and some other generator locations distributed throughout the State.

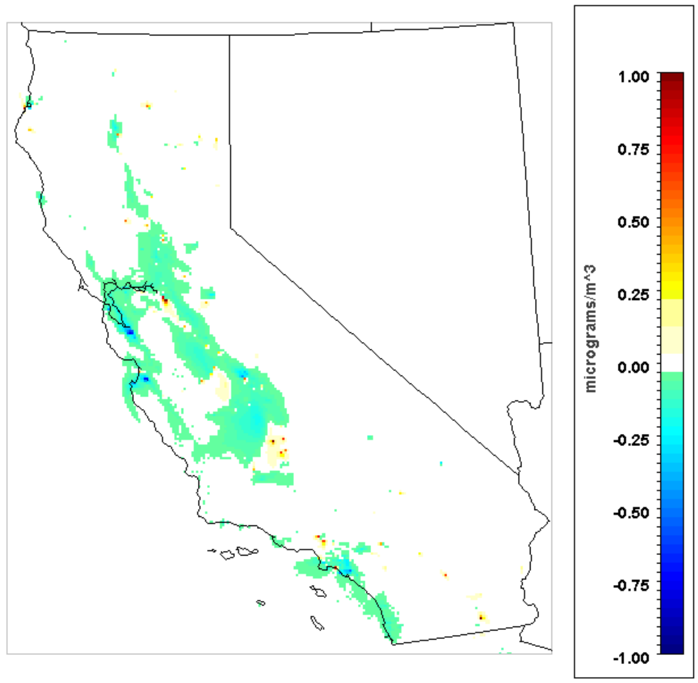


Figure 129: Difference in 24-h $PM_{2.5}$ in the Winter Industrial 2030 Case from the Base Case

7.3.2.5 Transportation Sector 2030 Case

Summer

Figure 130 displays the difference in maximum 8 hour average ozone in the Summer Transportation 2030 Case from the Base Case. Quantitatively, impacts range from -2.41 to +0.92 ppb. Improvements in ozone occur in areas associated with high vehicle traffic including SoCAB and the Bay Area. Worsening is visible associated with generators in Northern California and some areas of the Central Valley.

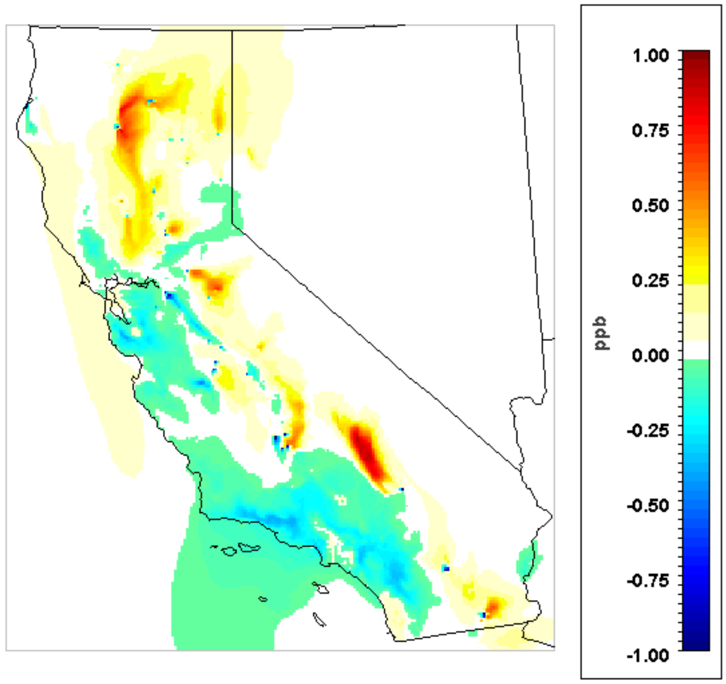


Figure 130: Difference in maximum 8 hour average ozone in the Summer Transportation 2030 Case from the Base Case

Figure 131 displays the difference in 24-h $PM_{2.5}$ in the Summer Transportation 2030 Case from the Base Case. Quantitatively, impacts range from -0.76 to +2.04 μg/m^3. Spatial patterns of impacts are similar to those for ozone and include reductions in urban regions, i.e., SoCAB, Bay Area, occurring in tandem with increases in the Central Valley and Northern California.

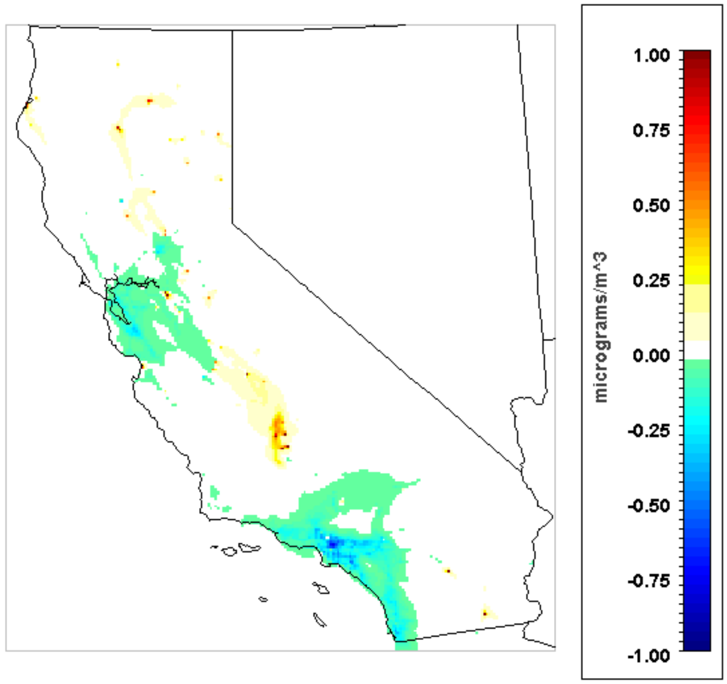


Figure 131: Difference in 24-h $PM_{2.5}$ in the Summer Transportation 2030 Case from the Base Case

Winter

Figure 132 displays the difference in maximum 8 hour average ozone in the Winter Transportation 2030 Case from the Base Case. Quantitatively, impacts range from -1.63 to +0.67 ppb. Generally, impacts on ozone are fairly moderate and include areas of worsening in the SoCAB and Bay Area and areas of improvement localized to generators sites. Essentially, the winter-time dynamics of ozone result in increases in areas of emission reductions and decreases in locations that experience NO_x increases.

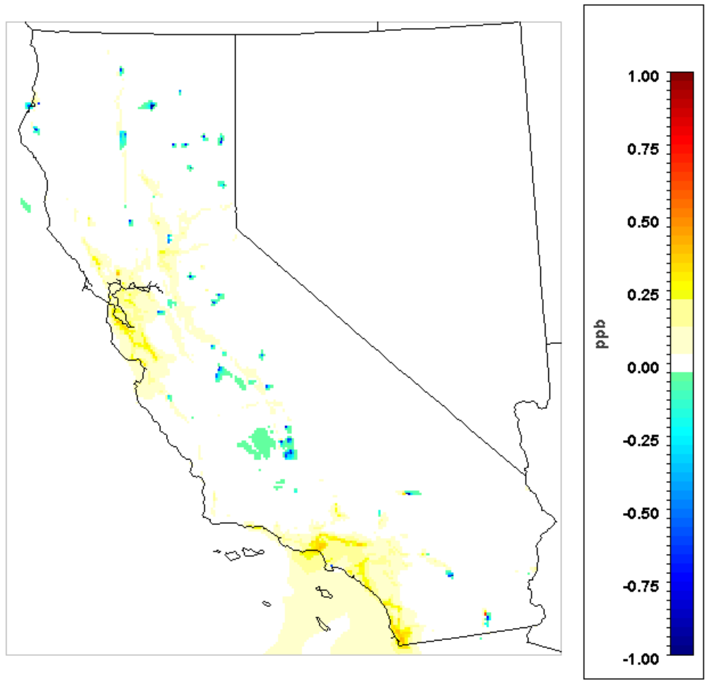


Figure 132: Difference in maximum 8 hour average ozone in the Winter Transportation 2030 Case from the Base Case

Figure 133 displays the difference in 24-h $PM_{2.5}$ in the Winter Transportation 2030 Case from the Base Case. Quantitatively, impacts range from -1.08 to +0.60 µg/m^3. Impacts are largely beneficial and include reductions in many areas of the State. Most notably, peak reductions occur in SoCAB as a result of direct vehicle and petroleum refinery emission reductions and represent an important benefit. Additionally, the Central Valley, Bay Area, and Sacramento experience benefits.

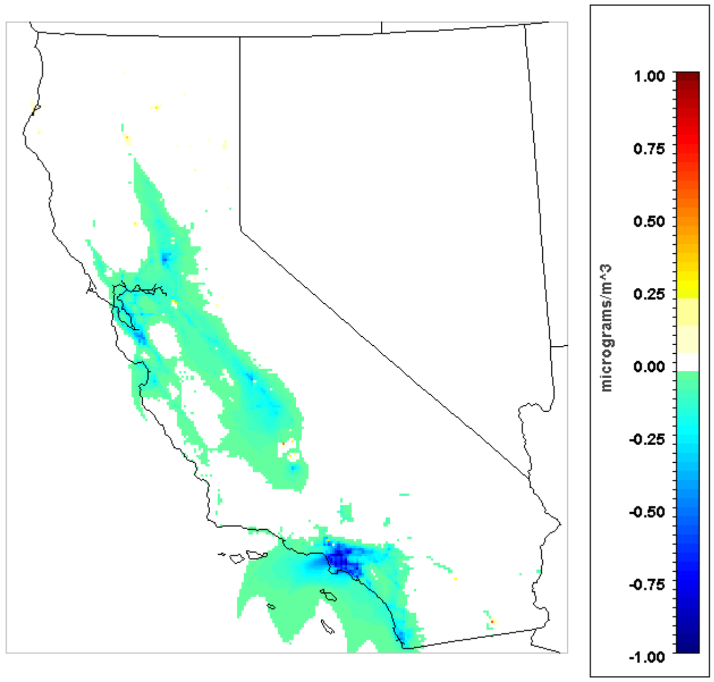


Figure 133: Difference in 24-h $PM_{2.5}$ in the Winter Transportation 2030 Case from the Base Case

7.3.2.6 Transportation Smart Charging 2030 Case

Summer

Figure 134 displays the difference in maximum 8 hour average ozone in the Summer Tra Smart Charging 2030 Case from the Base Case. Quantitatively, impacts range from -1.89 to +0.63 ppb. Generally, impacts are beneficial and include improvements along coastal urban regions supporting large vehicle populations such as SoCAB, the SF Bay Area, and some portions of the Central Valley. Two notable areas of concentration increases occur, with one being in the northern portion of the state and the other originating from natural gas generators near Bakersfield.

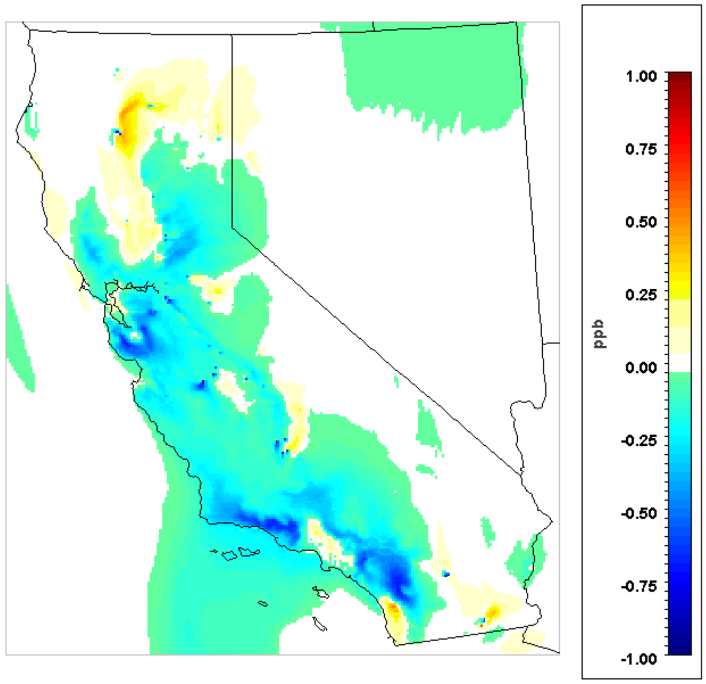


Figure 134: Difference in maximum 8 hour average ozone in the Summer Transportation Smart Charging 2030 Case from the Base Case

Figure 135 displays the difference in 24-h $PM_{2.5}$ in the Summer Transportation Smart Charging 2030 Case from the Base Case. Quantitatively, impacts range from -0.96 to +1.02 μg/m^3. Impacts are largely characterized by improvements over large areas of the State, including the SoCAB, the Central Valley, and the Bay Area. In particular, reductions in the SoCAB represent the largest impact in the Case. Small, localized increases occur in the same location as ozone increases but are dominated by improvements.

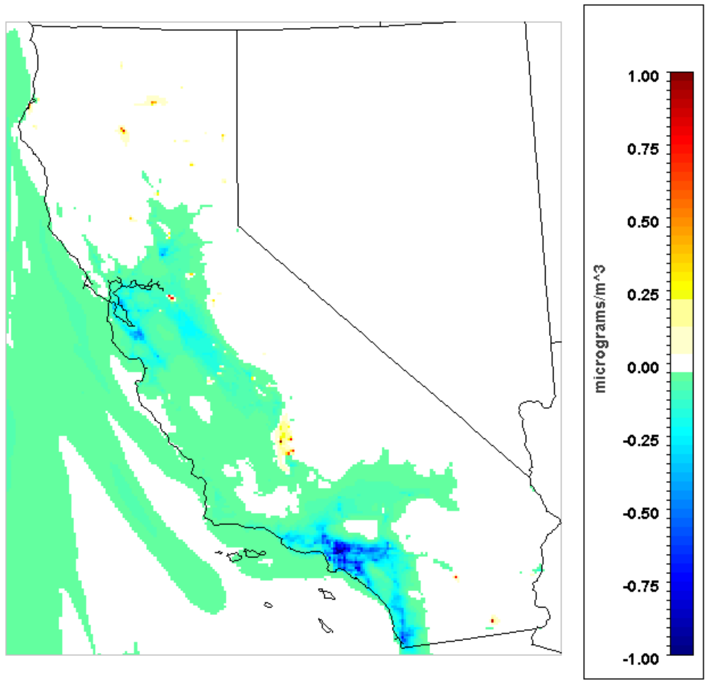


Figure 135: Difference in 24-h $PM_{2.5}$ in the Summer Transportation Smart Charging 2030 Case from the Base Case

Winter

Figure 136 displays the difference in maximum 8 hour average ozone in the Winter Transportation Smart Charging 2030 Case from the Base Case. Quantitatively, impacts range from -0.81 to +0.69 ppb. Spatially, impacts display fairly minor areas of worsening in many of the regions of the State that currently experience poor AQ, i.e., SoCAB, Bay Area, Bakersfield, and Sacramento. However, the winter time ozone impacts are less of a concern due to seasonal differences discussed in the results section regarding the 2020 Winter scenarios.

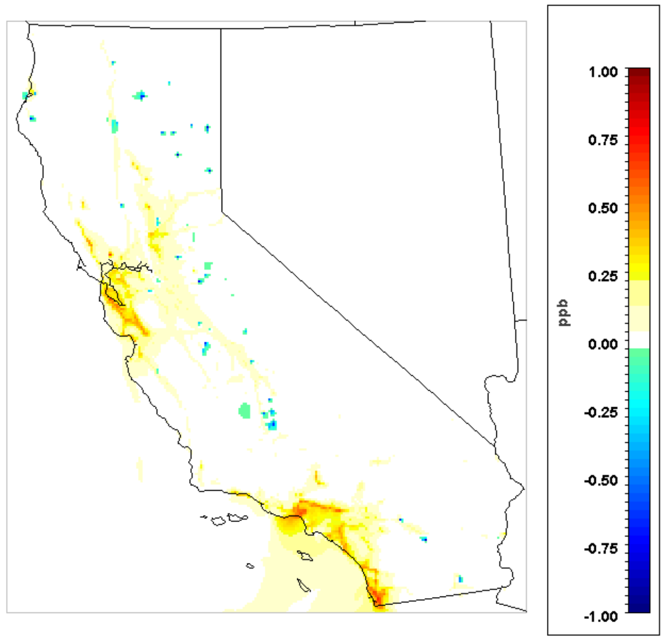


Figure 136: Difference in maximum 8 hour average ozone in the Winter Transportation Smart Charging 2030 Case from the Base Case

Figure 137 displays the difference in 24-h $PM_{2.5}$ in the Winter Transportation Smart Charging 2030 Case from the Base Case. Quantitatively, impacts range from -1.65 to +0.39 $\mu g/m^3$. The magnitude of improvements are substantial and occur in important areas for winter time PM levels including the SoCAB, Central Valley, SF Bay Area, and Sacramento Area. Additionally, increases in concentrations are minor to reductions. Thus, the Winter Smart Charging Transportation 2030 Case achieves important benefits to AQ in 2030.

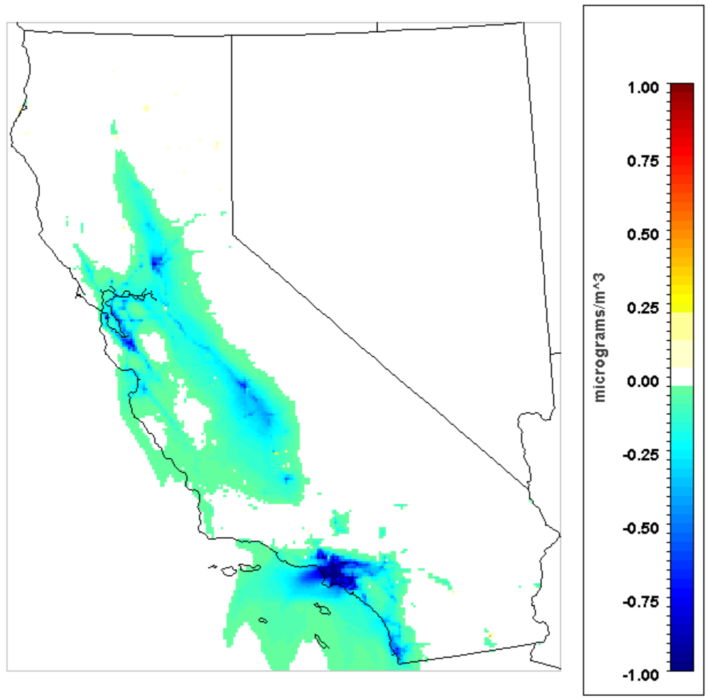


Figure 137: Difference in 24-h $PM_{2.5}$ in the Winter Transportation Smart Charging 2030 Case from the Base Case

Comparison of Immediate and Smart Charging Air Quality Impacts

In order to assess the AQ impacts of smart charging relative to immediate charging difference plots were generated for the Transportation Smart Charging 2030 Case relative to the Transportation 2030 Case which assumes immediate charging of vehicles. Thus the following figures display spatial and temporal distributions of pollutants such that negative values represent enhanced reductions and positive values represent increased concentrations when smart charging is deployed. It should be noted that the Smart Charging Case involves a greater penetration of EVs than the Immediate Charging Case and thus a direct comparison should include that caveat.

Figure 138 shows the difference in maximum 8-hr ozone from smart charging for the Summer 2030 Transportation Case. Quantitatively, impacts range from -0.96 to +1.03 ppb. Impacts are generally represented by improvements over most of the State. Peak improvements occur in the Central Valley with notable impacts in Sacramento and the SF Bay Area additionally. It is perhaps most notable that despite an increase in required electricity for vehicles the Smart Charging Case does not experience higher areas of worsening from power plants. This is due to the charging strategy which avoids charging during peak times and subsequent emissions associated with power generation at those times. Thus, the smart charging of vehicles can achieve improved AQ benefits relative to immediate charging in terms of summer ozone levels.

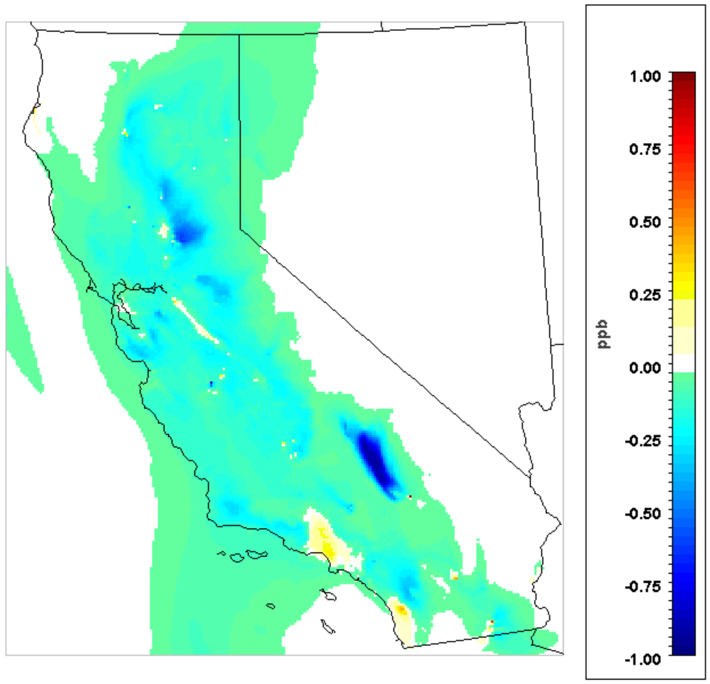


Figure 138: Difference in maximum 8-hr ozone between the Smart and Immediate Charging Summer Transportation 2030 Cases

Figure 139 shows the difference in 24-hr $PM_{2.5}$ from smart charging for the Summer 2030 Transportation Case. Quantitatively, impacts range from -0.85 to +0.41 μg/m^3. With similarity to the ozone difference, impacts are characterized by improvements in many areas of the State including the SoCAB, Central Valley, and SF Bay Area.

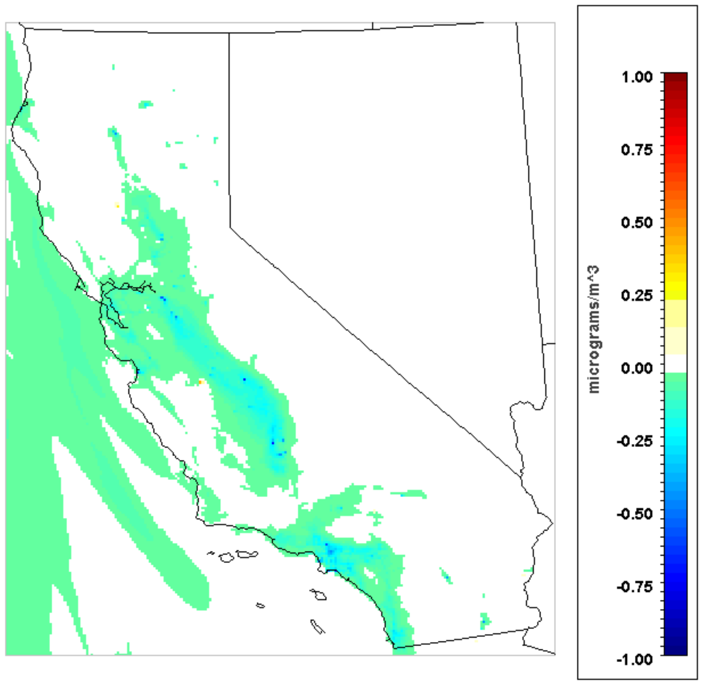


Figure 139: Difference in 24-h $PM_{2.5}$ between the Smart and Immediate Charging Summer Transportation 2030 Cases

Figure 140 shows the difference in maximum 8-hr ozone from smart charging for the Winter 2030 Transportation Case. Quantitatively, impacts range from -0.82 to +1.68 ppb. Impacts are largely characterized by moderate increases throughout the State. Despite increases, the winter ozone dynamics limit the importance of the effects.

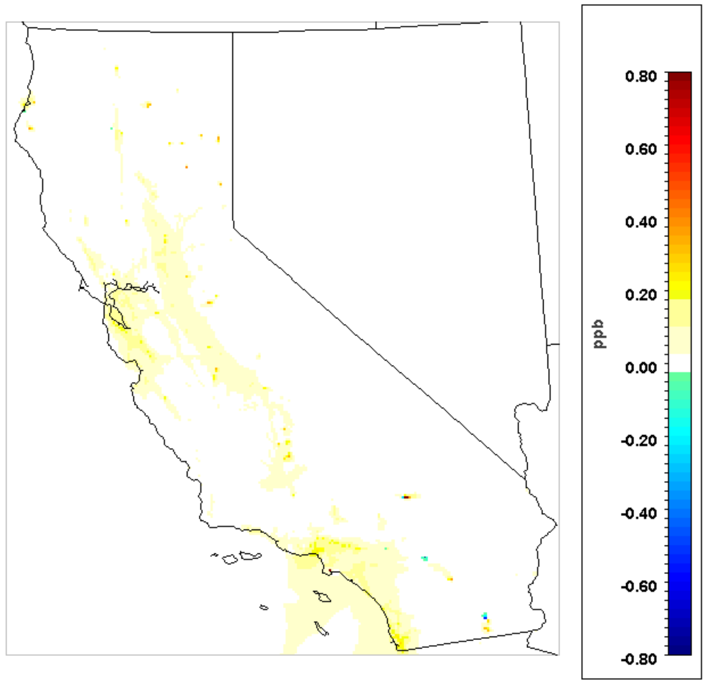


Figure 140: Difference in maximum 8-hr ozone between the Smart and Immediate Charging Winter Transportation 2030 Cases

Figure 141 shows the difference in 24-hr $PM_{2.5}$ from smart charging for the Winter 2030 Transportation Case. Quantitatively, impacts range from -1.10 to +0.34 μg/m³. Impacts are largely characterized by improvements in several key areas of the State. Peak impacts occur in the SoCAB which experiences significant improvements in ground-level concentrations. Additional areas of improvement occur in the Central Valley.

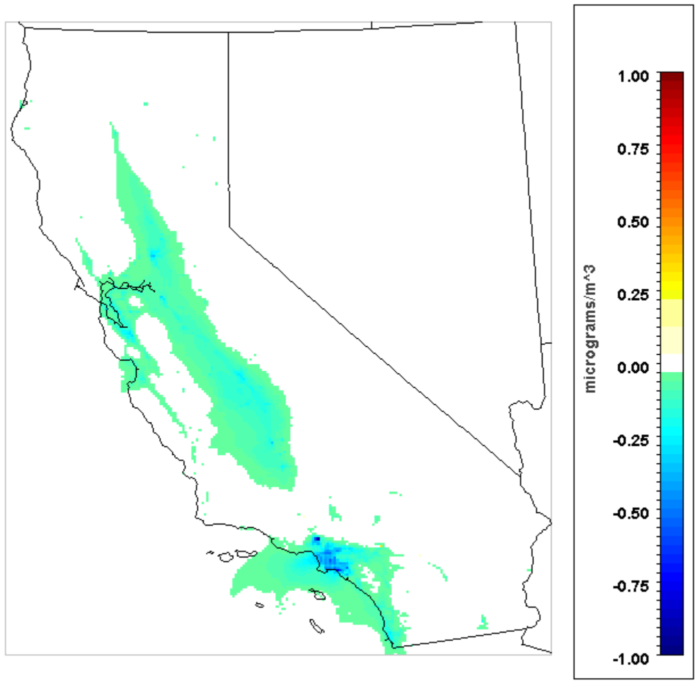


Figure 141: Difference in 24-h $PM_{2.5}$ between the Smart and Immediate Charging Winter Transportation 2030 Cases

7.3.2.7 Residential, Commercial, and Transportation (ResComTra) 2030 Case

Summer

Figure 142 displays the difference in maximum 8 hour average ozone in the Summer ResComTra 2030 Case from the Base Case. Quantitatively, impacts range from -4.49 to +1.92 ppb. Impacts are spatially similar to the individual cases and are characterized by areas of improvement throughout the State, including SoCAB and the Bay Area. Increased concentrations occur from generator locations, most notably in Northern California.

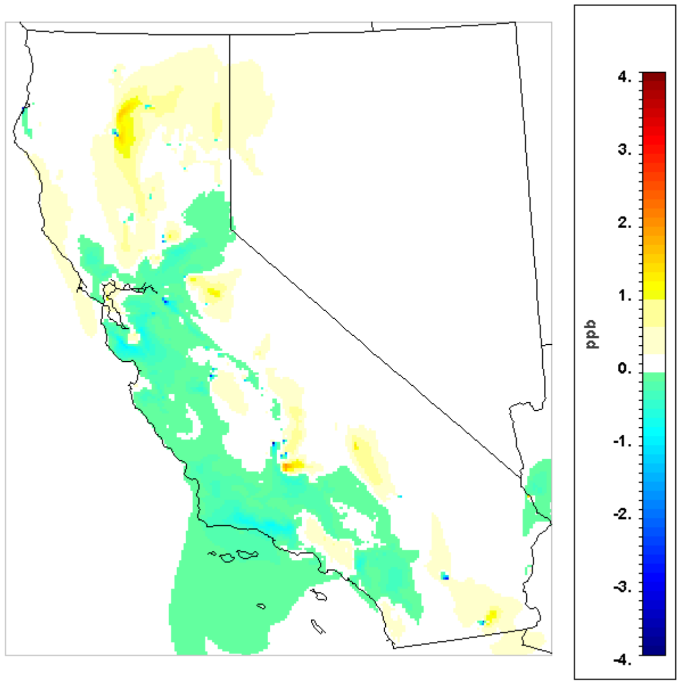


Figure 142: Difference in maximum 8 hour average ozone in the Summer ReComTra 2030 Case from the Base Case

Figure 143 displays the difference in 24-h $PM_{2.5}$ in the Summer ResComTra2030 Case from the Base Case. Quantitatively, impacts range from -1.41 to 3.66 μg/$m^3$and are largely beneficial to the State. Large areas of concentration reduction occur in the Bay Area, Central Valley, and SoCAB. Localized areas of worsening do occur adjacent to some generator locations.

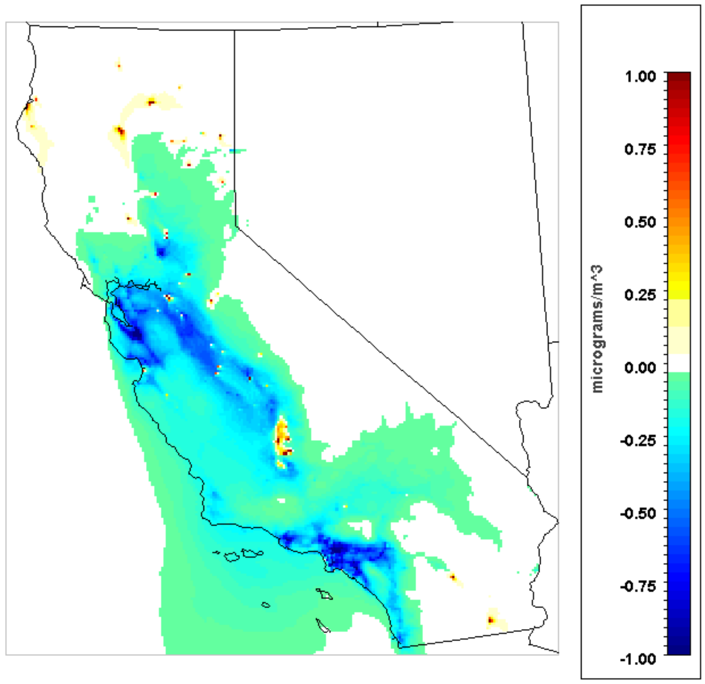


Figure 143: Difference in 24-h $PM_{2.5}$ in the Summer ResComTra 2030 Case from the Base Case

Winter

Figure 144 displays the difference in maximum 8 hour average ozone in the Winter ResComTra 2030 Case from the Base Case. Quantitatively, impacts range from -2.62 to +3.34 ppb, although generally much of the State experiences increases in ground-level concentrations.

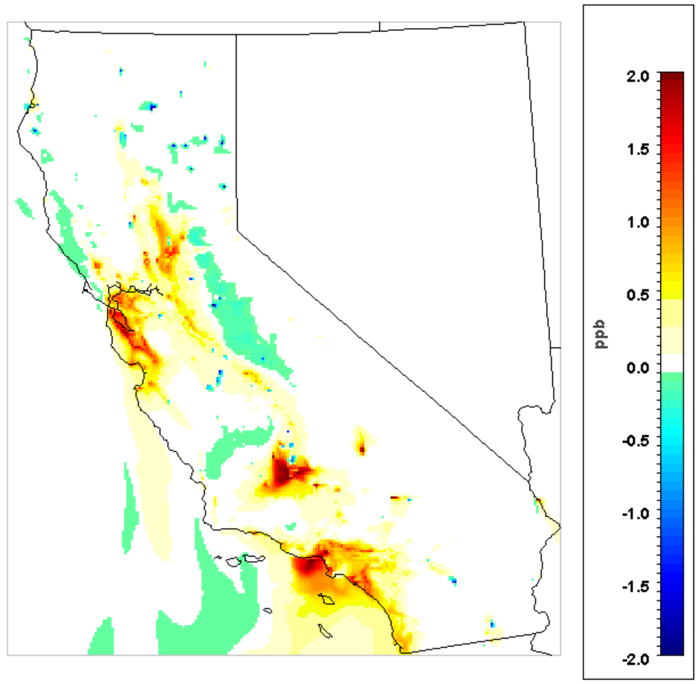


Figure 144: Difference in maximum 8 hour average ozone in the Winter ResComTra 2030 Case from the Base Case

Figure 145 displays the difference in 24-h $PM_{2.5}$ in the Winter ResComTra 2030 Case from the Base Case. Quantitatively, impacts range from -15.09 to +1.11 µg/m^3. Impacts are characterized by significant improvement in ground-level concentrations throughout many regions of the State.

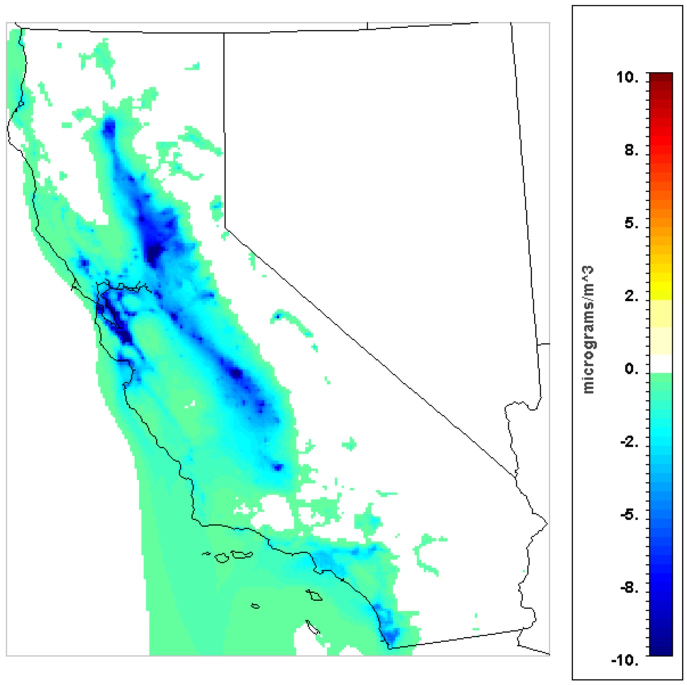


Figure 145: Difference in 24-h $PM_{2.5}$ in the Winter ResComTra 2030 Case from the Base Case

7.3.2.8 Residential, Commercial, Industrial and Transportation (ResComIndTra) 2030 Case

Summer

Figure 146 displays the difference in maximum 8 hour average ozone in the Summer ResComIndTra 2030 Case from the Base Case. Quantitatively, impacts range from -8.63 to +2.91 ppb. The impacts of the industrial sector electrification are evident in the plumes of worsening that occur mirroring the Industrial Sector Case in isolation.

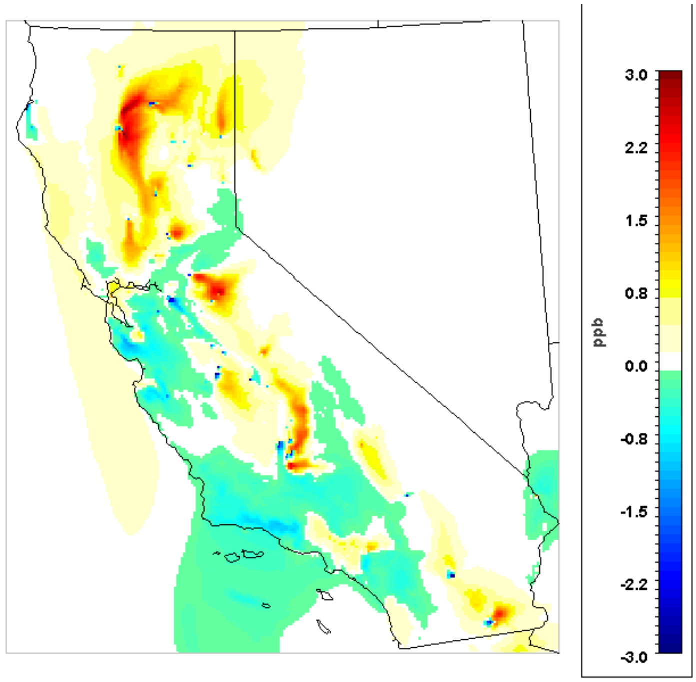


Figure 146: Difference in maximum 8 hour average ozone in the Summer ResComIndTra 2030 Case from the Base Case

Figure 147 displays the difference in 24-h $PM_{2.5}$ in the Summer ResComIndTra 2030 Case from the Base Case. Quantitatively, impacts range from -1.77 to +8.60 μg/m^3. Impacts are largely beneficial including improvements from emission reductions in SoCAB and the SF Bay Area. Contrastingly, increases occur from gas generator emissions in Bakersfield and some generators in Northern California.

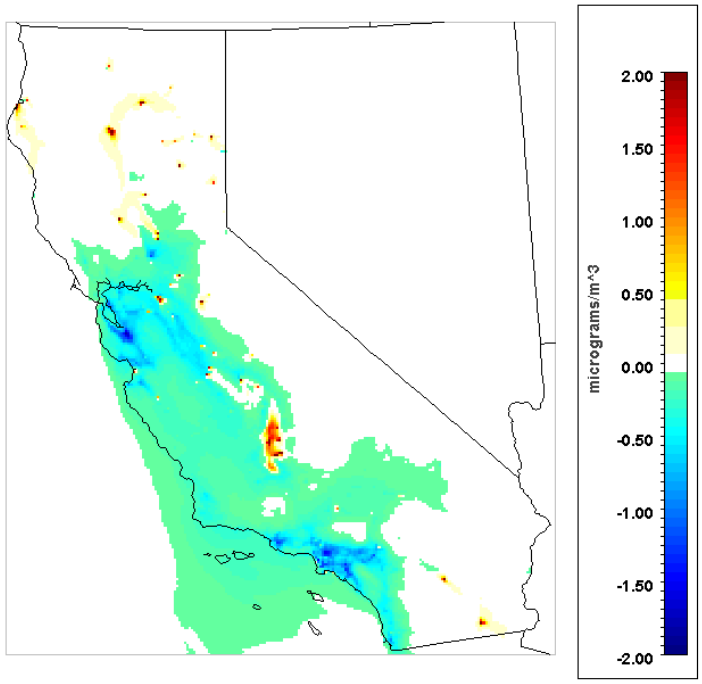


Figure 147: Difference in 24-h $PM_{2.5}$ in the Summer ResComIndTra 2030 Case from the Base Case

Winter

Figure 148 displays the difference in maximum 8-hour average ozone in the Winter ResComIndTra 2030 Case from the Base Case. Quantitatively, impacts range from -12.34 to +3.65 ppb. The majority of impacts are represented by increases in ground-level concentrations including in Bakersfield, SoCAB and the SF Bay Area.

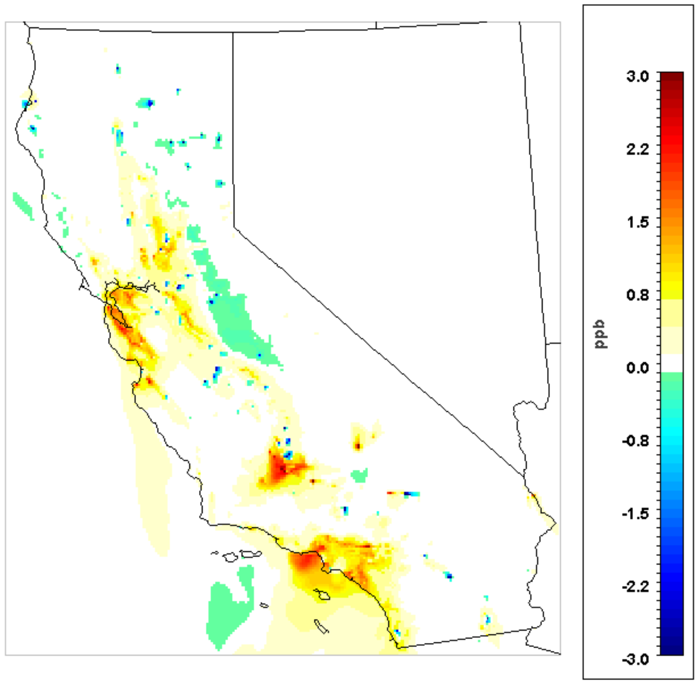


Figure 148: Difference in maximum 8 hour average ozone in the Winter ResComIndTra 2030 Case from the Base Case

Figure 149 displays the difference in 24-h $PM_{2.5}$ in the Winter ResComIndTra 2030 Case from the Base Case. Quantitatively, impacts range from -15.26 to +5.79 μg/m^3. Impacts are almost exclusively beneficial and include dramatic improvements in the SF Bay Area, Central Valley, and Greater Sacramento areas. Reductions in emissions from winter time residential scenarios appear to contribute to the overall impacts.

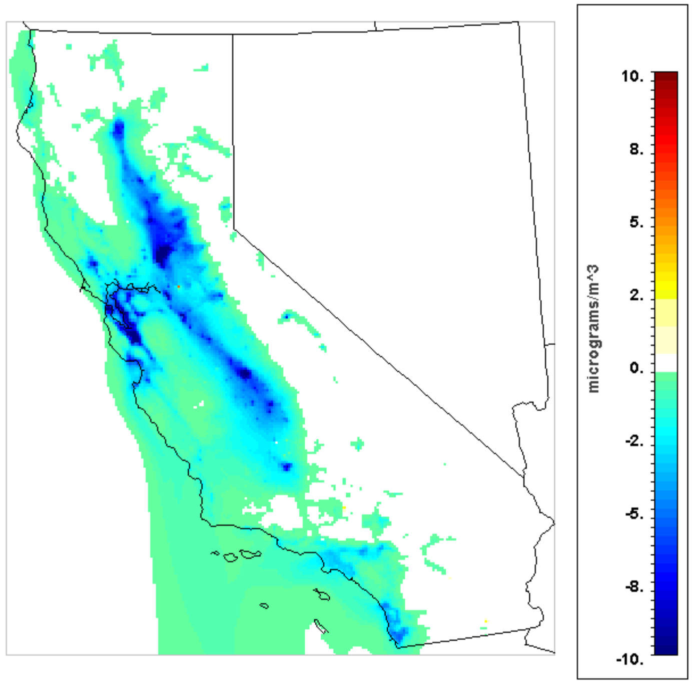


Figure 149: Difference in 24-h $PM_{2.5}$ in the Winter ResComIndTra 2030 Case from the Base Case

7.3.2.9 Summary of 2030 Cases

Table 57 displays the peak impacts on 8 hour average ozone and 24 hour $PM_{2.5}$ for the Summer 2030 Cases relative to the Base Case. Table 58 displays the peak impacts on 8 hour average ozone and 24 hour $PM_{2.5}$ for the Winter 2030 Cases relative to the Base Case. Impacts on $PM_{2.5}$ and ozone are moderate to substantial for all electrification scenarios in 2030 and reflect a higher electrification potential from 2020 for many sectors of study. Impacts on max 8-hr ozone range from -12.34 in the Winter ResComIndTra Case to +2.91 in the same Case.

Table 57: Summary of peak impacts on 8-hr max ozone and 24-h $PM_{2.5}$ for Summer 2030 Cases

Summer Case	8-hr Ozone [ppb]	24-hr $PM_{2.5}$ [µg/m^3]

Res 2030	-1.33 to +0.44	-0.19 to +1.13
Com 2030	-0.96 to +1.85	-0.99 to +0.48
ResCom 2030	-2.24 to +1.93	-1.07 to +1.42
Ind 2030	-4.63 to +1.49	-0.48 to +4.34
Tra 2030	-2.41 to +0.92	-0.76 to +2.04
Tra Smart 2030	-1.89 to +0.63	-0.96 to +1.02
ResComTra 2030	-4.49 to +1.92	-1.41 to +3.66
ResComIndTra 2030	-8.63 to +2.91	-1.77 to +8.60

Table 58: Summary of peak impacts on 8-hr max ozone and 24-h $PM_{2.5}$ for Winter 2030 Cases

Winter Case	8-hr Ozone [ppb]	24-hr $PM_{2.5}$ [$\mu g/m^3$]
Res 2030	-0.78 to +1.82	-13.33 to +0.25
Com 2030	-0.49 to +3.19	-2.69 to +0.19
ResCom 2030	-3.56 to +3.31	-14.51 to +0.45
Ind 2030	-10.18 to +1.52	-1.21 to +1.29
Tra 2030	-1.63 to +0.67	-1.08 to +0.60
Tra Smart 2030	-0.81 to +0.69	-1.65 to +0.39
ResComTra 2030	-2.62 to +3.34	-15.09 to +1.11
ResComIndTra 2030	-12.34 to +3.65	-15.26 to +5.79

7.3.3 Cases for 2050

7.3.3.1 Residential Sector 2050 Case

Summer

Figure 150 displays the difference in maximum 8 hour average ozone in the Summer Residential 2050 Case from the Base Case. Quantitatively, impacts range from -2.12 to +1.07 ppb. Scenario results are characterized by moderate improvements in some areas of the SoCAB and Bay Area. In contrast, generator emission increases results in moderate worsening in the Central Valley and Sacramento areas.

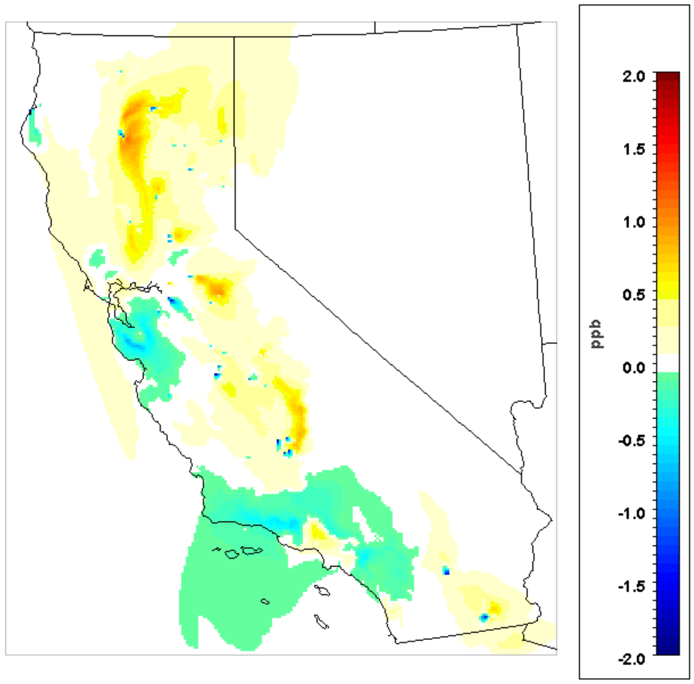


Figure 150: Difference in maximum 8 hour average ozone in the Summer Residential 2050 Case from the Base Case

Figure 151 displays the difference in 24-h $PM_{2.5}$ in the Summer Residential 2050 Case from the Base Case. Quantitatively, impacts range from -1.02 to +3.37 μg/m^3. Scenario results are characterized largely by moderate worsening from several regions of the State as a result of increased generator emissions. While impacts are highly localized, magnitudes of increases are high (i.e., peak increases of 3.37 μg/m^3) and warrant some concern; especially as areas of peak worsening include the Central Valley.

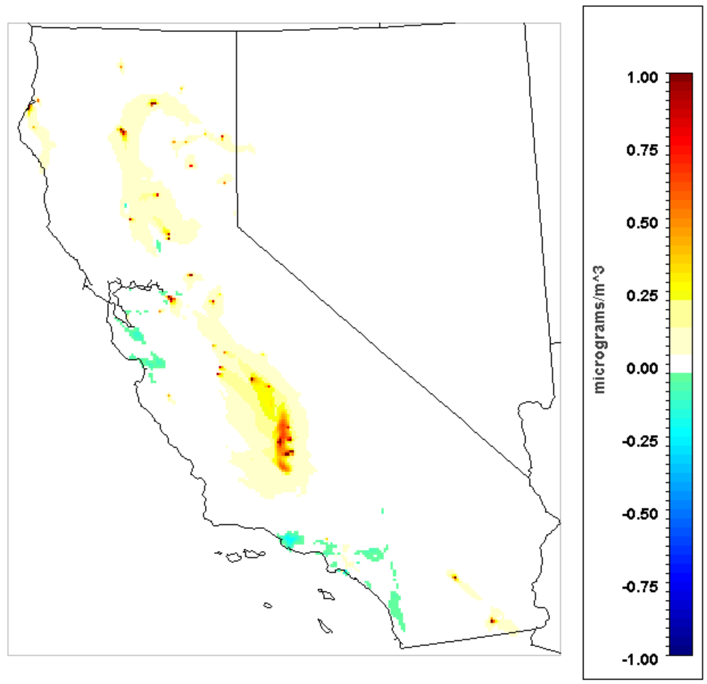


Figure 151: Difference in 24-h $PM_{2.5}$ in the Summer Residential 2050 Case from the Base Case

Winter

Figure 152 displays the difference in maximum 8 hour average ozone in the Winter Residential 2050 Case from the Base Case. Quantitatively, impacts range from -10.02 to +3.64 ppb. Impacts arise from the impacts of winter ozone formation and include increases in SoCAB and the Bay Area. Moderate reductions occur in other areas of the State.

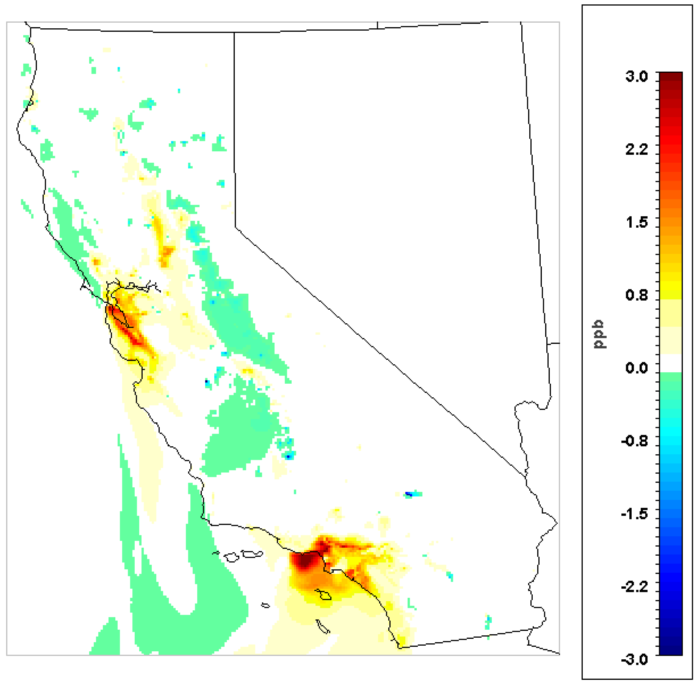


Figure 152: Difference in maximum 8 hour average ozone in the Winter Residential 2050 Case from the Base Case

Figure 153 displays the difference in 24-h $PM_{2.5}$ in the Winter Residential 2050 Case from the Base Case. Quantitatively, impacts range from -3.36 to +0.96 μg/m^3. Impacts are almost solely associated with improvements in ground-level concentrations. Areas of notable improvement include the SF Bay Area, Greater Sacramento, and the Central Valley. The impacts could be associated with reductions in emissions including residential combustion of wood for heating which generates significant amounts of PM.

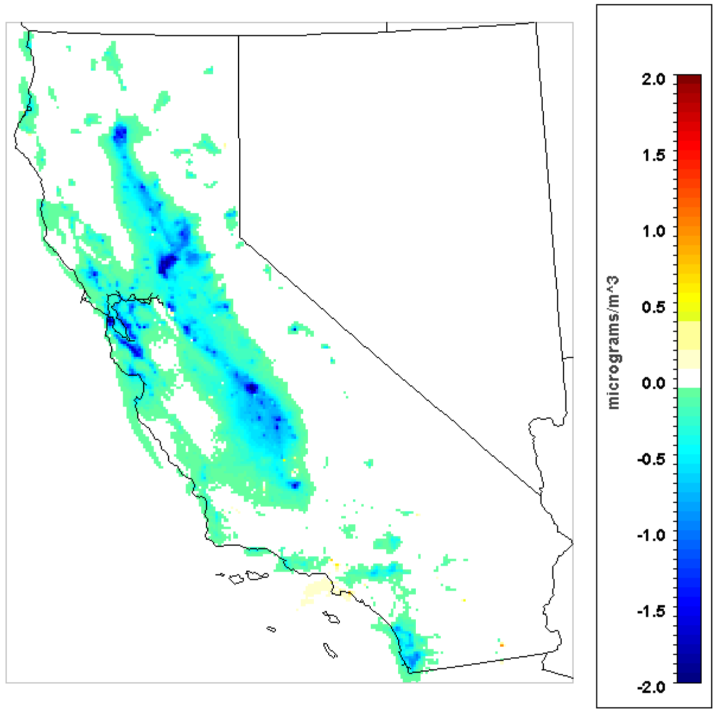


Figure 153: Difference in 24-h $PM_{2.5}$ in the Winter Residential 2050 Case from the Base Case

7.3.3.2 Commercial Sector 2050 Case

Summer

Figure 154 displays the difference in maximum 8-hour average ozone in the Summer Commercial 2050 Case from the Base Case. Quantitatively, impacts range from -2.0 to +5.29 ppb. Impacts include significant areas of reductions in ozone levels; with peak impacts occurring in the Bay Area, Central Valley, and SoCAB. Areas of worsening also occur in the Central Valley, with a localized area near Bakersfield experiencing significant increases (+5.29 ppb). The northern portion of the State also displays some worsening due to generator emissions.

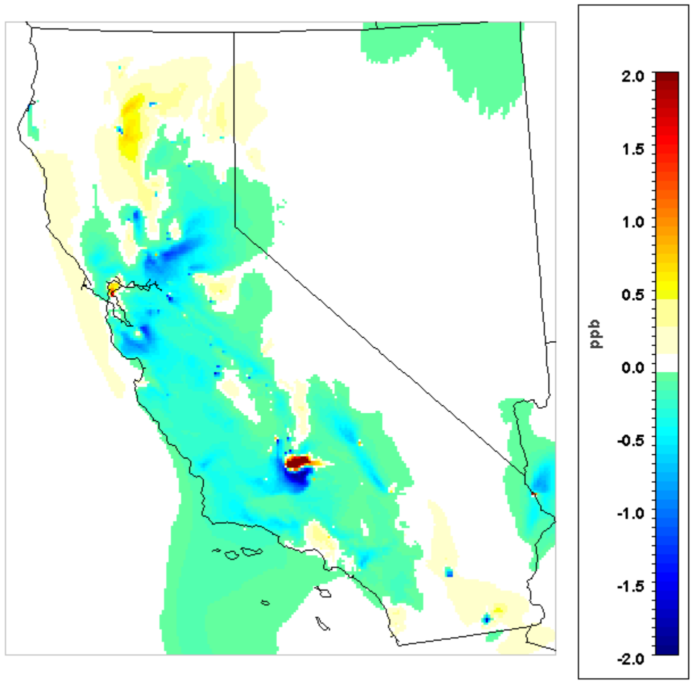


Figure 154: Difference in maximum 8 hour average ozone in the Summer Commercial 2050 Case from the Base Case

Figure 155 displays the difference in 24-h $PM_{2.5}$ in the Summer Commercial 2050 Case from the Base Case. Impacts range from -1.37 to +2.06 μg/m^3. Impacts are largely characterized by improvements, with peak effects occurring in key areas of the State (i.e., SoCAB, Central Valley, and Bay Area). Some localized worsening occurs at sites of fossil fuel generators, however overall impacts are largely beneficial.

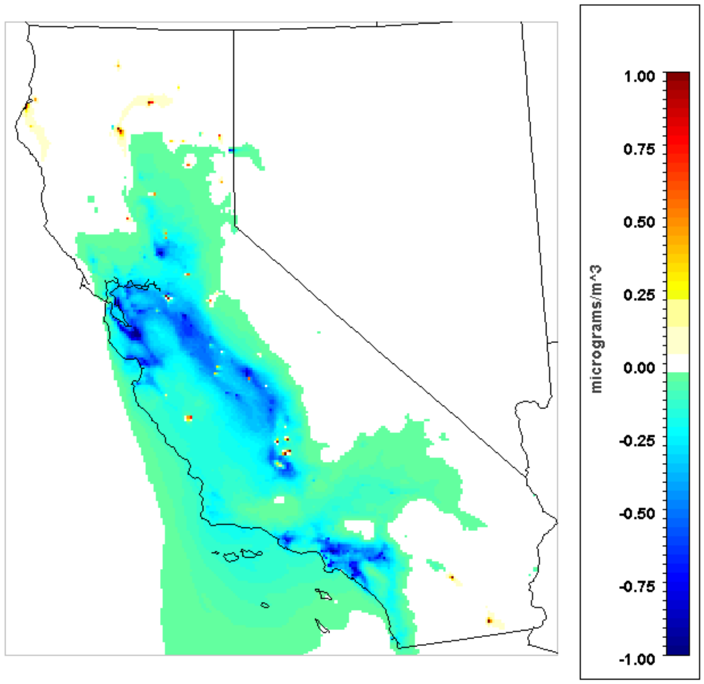


Figure 155: Difference in 24 hour average $PM_{2.5}$ in the Summer Commercial 2050 Case from the Base Case

Winter

Figure 156 displays the difference in maximum 8 hour average ozone in the Winter Commercial 2050 Case from the Base Case. Quantitatively, impacts range from -1.60 to +7.27 ppb. Impacts are largely described by increases in ground-level ozone concentrations across the State. The winter time dynamics associated with ozone formation and fate result in increases in areas that experience NO_x reductions.

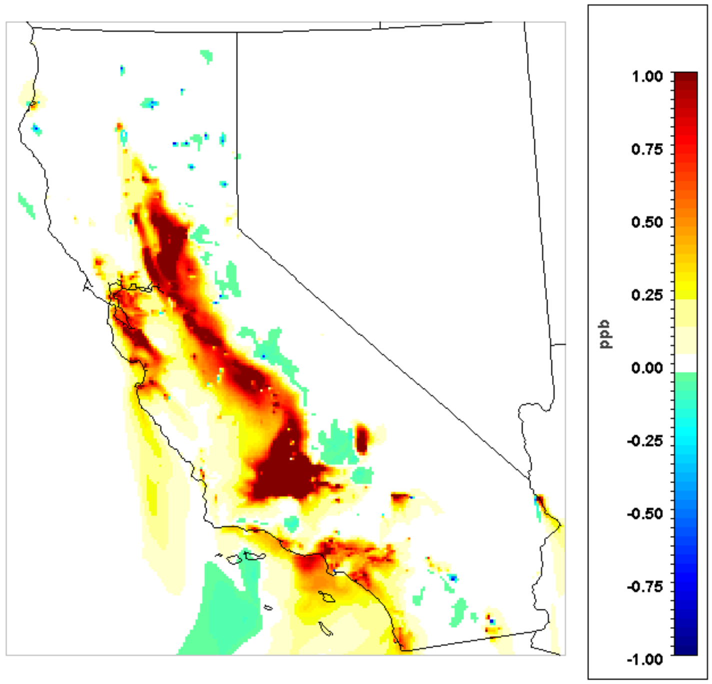


Figure 156: Difference in maximum 8 hour average ozone in the Winter Commercial 2050 Case from the Base Case

Figure 157 displays the difference in 24-h $PM_{2.5}$ in the Winter Commercial 2050 Case from the Base Case. Quantitatively, impacts range from -1.64 to +0.87 μg/m^3. Generally, impacts are fairly moderate with peak worsening occurring in the Bakersfield due to increased gas generator emissions. Contrastingly, improvements occur in other areas of the State including SoCAB and the SF Bay Area.

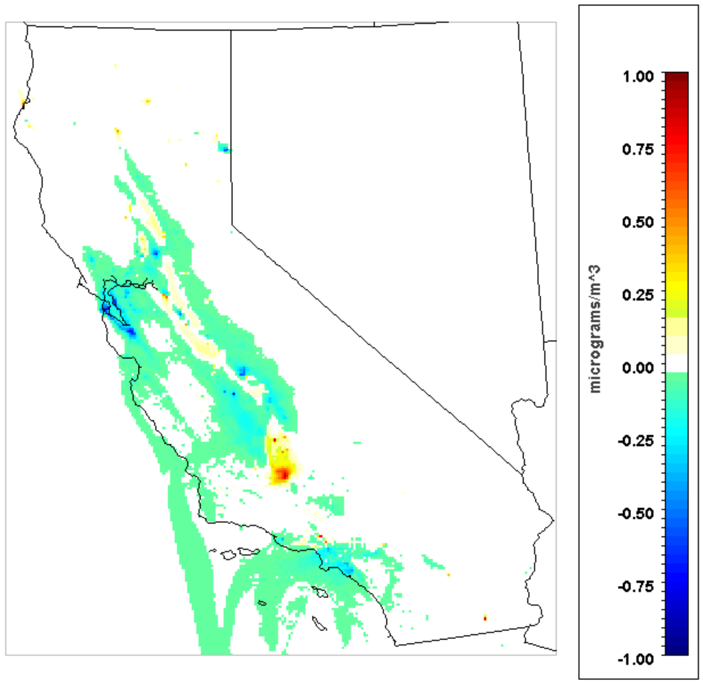


Figure 157: Difference in 24 hour average $PM_{2.5}$ in the Winter Commercial 2050 Case from the Base Case

7.3.3.3 Residential and Commercial (ResCom) 2050 Case

Summer

Figure 158 displays the difference in maximum 8 hour average ozone in the Summer ResCom 2050 Case from the Base Case. Impacts range in magnitude from -3.83 to +5.48 ppb. NO_x increases from generators result in notable areas of worsening in Bakersfield and, to a lesser degree, some northern areas of the State. Reductions in NO_x from residential and commercial sectors yield improvements in many others regions of the State including SoCAB and the SF Bay Area.

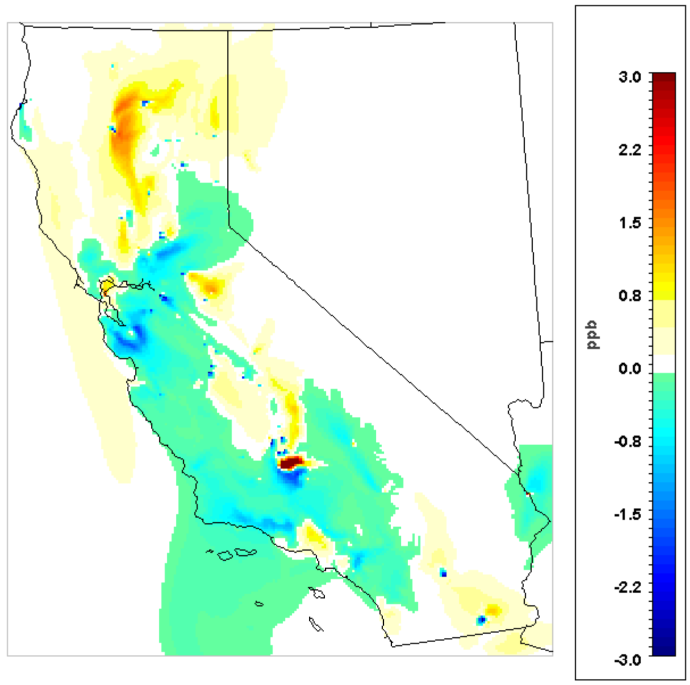


Figure 158: Difference in maximum 8 hour average ozone in the Summer ResCom 2050 Case from the Base Case

Figure 159 displays the difference in 24-h $PM_{2.5}$ in the Summer ResCom 2050 Case from the Base Case. Quantitatively, impacts range from -1.41 to +5.61 μg/m^3 and represent an additive outcome relative to the individual cases. Largely, reductions occur over large areas of the State with peak impacts in the SF Bay Area and SoCAB.

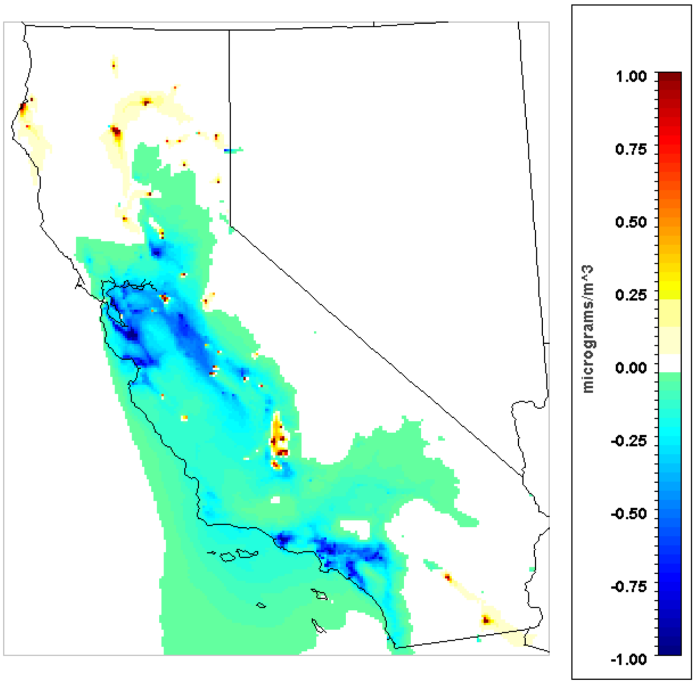


Figure 159: Difference in 24 hour average $PM_{2.5}$ in the Summer ResCom 2050 Case from the Base Case

Winter

Figure 160 displays the difference in maximum 8 hour average ozone in the Winter ResCom 2050 Case from the Base Case. Quantitatively, impacts range from -13.45 to +7.46 ppb. In general, impacts are moderate to minor and result from the winter-time ozone formation dynamics.

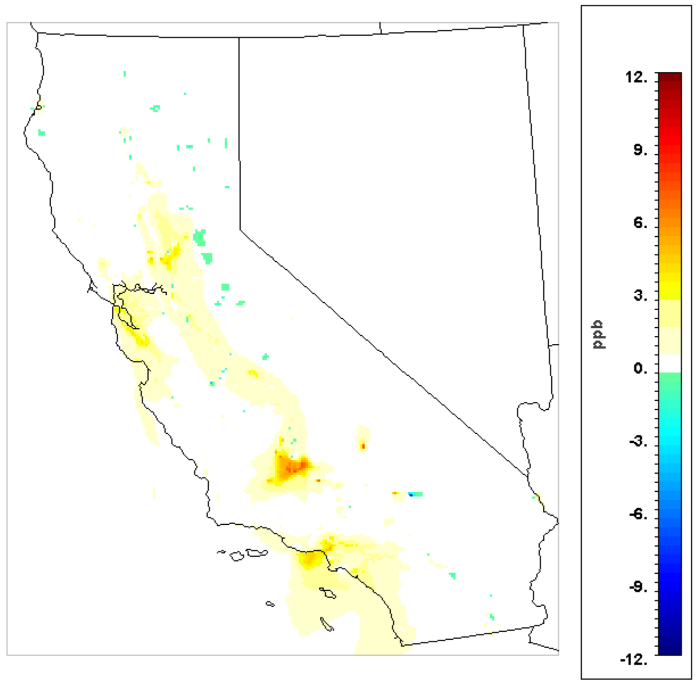


Figure 160: Difference in maximum 8 hour average ozone in the Winter ResCom 2050 Case from the Base Case

Figure 161 displays the difference in 24-h $PM_{2.5}$ in the Winter ResCom 2050 Case from the Base Case. Quantitatively, impacts range from -20.62 to +1.77 μg/m^3. Impacts are highly beneficial and include large reductions throughout the State. The largest benefits occur in the SF Bay Area and northern portion of the Central Valley.

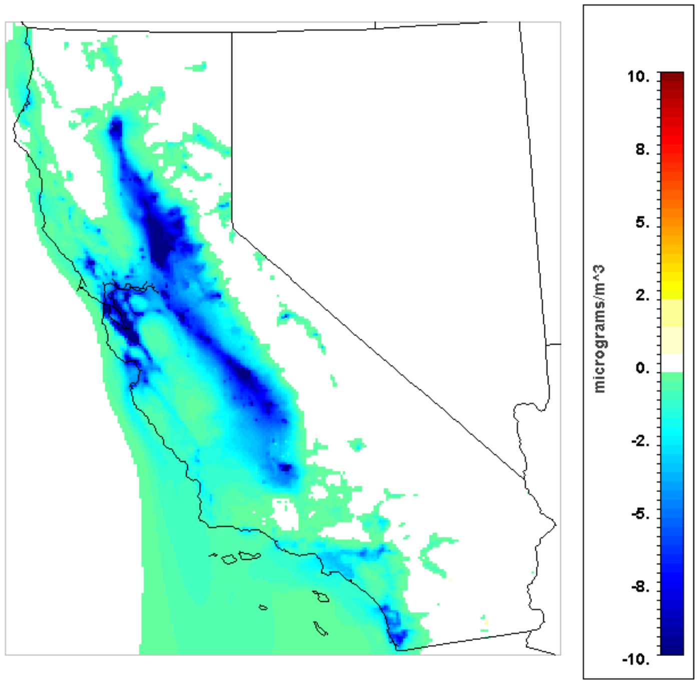


Figure 161: Difference in 24 hour average $PM_{2.5}$ in the Winter ResCom 2050 Case from the Base Case

7.3.3.4 Industrial Sector 2050 Case

Summer

Figure 162 displays the difference in maximum 8 hour average ozone in the Summer Industrial 2050 Case from the Base Case. Quantitatively, impacts range from -7.10 to +3.58 ppb. Impacts from electrification of industrial sources largely include significant worsening across the Central Valley and northern regions of the State from increased NO_x emissions from generators. Additionally, areas of localized worsening occur in the southern portion adjacent to the border with Mexico. Areas of improvement are lesser in magnitude and include some areas of SoCAB and Bakersfield as a result of reduced NO_x from large industrial sources. Additionally, one site near the Bay Area experiences a plume of reduction with high magnitude. However, impacts on ozone are generally deleterious for the scenario and demonstrate the high increase in electricity needed to meet sector electrification needs as a result of the replacement of efficient technologies.

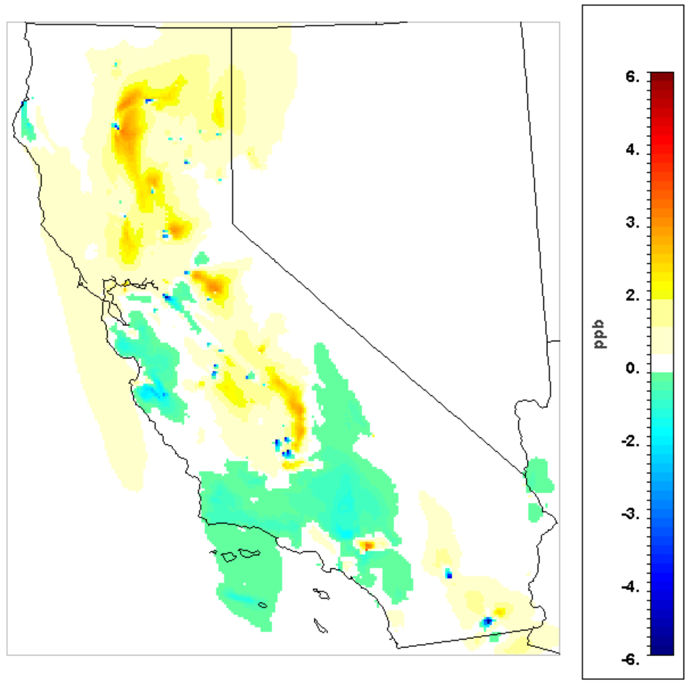


Figure 162: Difference in maximum 8 hour average ozone in the Summer Industrial 2050 Case from the Base Case

Figure 163 displays the difference in 24-h $PM_{2.5}$ in the Summer Industrial 2050 Case from the Base Case. Quantitatively, impacts range from -1.10 to +12.40 μg/m^3. Impacts for $PM_{2.5}$ are similar to those for ozone in that increases in ground level concentrations are the dominant effect, including in the Central Valley and northern part of the State. Additionally, improvements are observed in the Bay Area and SoCAB although at a lesser magnitude.

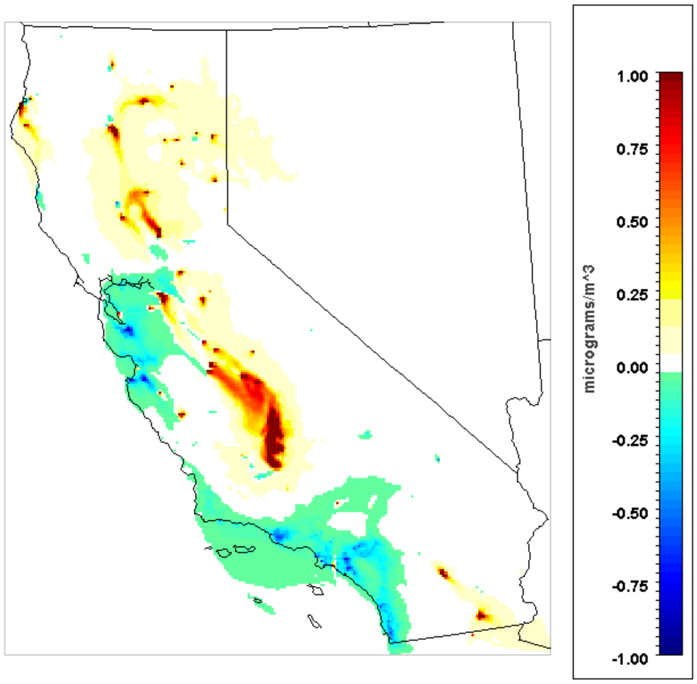


Figure 163: Difference in 24 hour average $PM_{2.5}$ in the Summer Industrial 2050 Case from the Base Case

Winter

Figure 164 displays the difference in maximum 8 hour average ozone in the Winter Industrial 2050 Case from the Base Case. Quantitatively, impacts range from -13.61 to +5.55 ppb. Impacts largely include worsening of ground-level concentrations including peak impacts in SoCAB due to the inverse NO_x relationship observed in winter. Generally, impacts are fairly minor in spatial coverage relative to other cases.

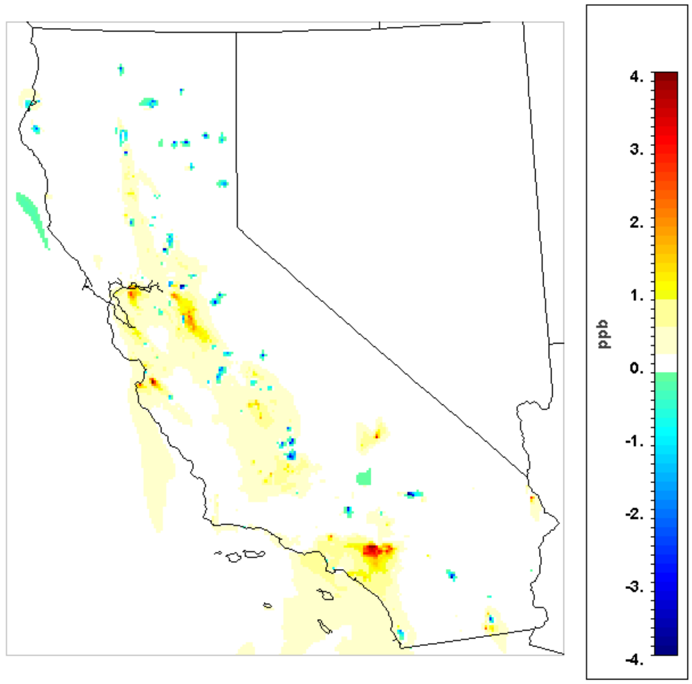


Figure 164: Difference in maximum 8 hour average ozone in the Winter Industrial 2050 Case from the Base Case

Figure 165 displays the difference in 24-h $PM_{2.5}$ in the Winter Industrial 2050 Case from the Base Case. Quantitatively, impacts range from -3.22 to +12.28 μg/m^3. Overall, impacts are fairly minor for both reductions and increases. As can be seen, impacts tend to be localized with the largest area of improvement occurring in the SF Bay Area and Central Valley. Some improvement is also seen from NO_x reductions in SoCAB. Contrastingly, some localized worsening occurs, notably from the Bakersfield area gas generators.

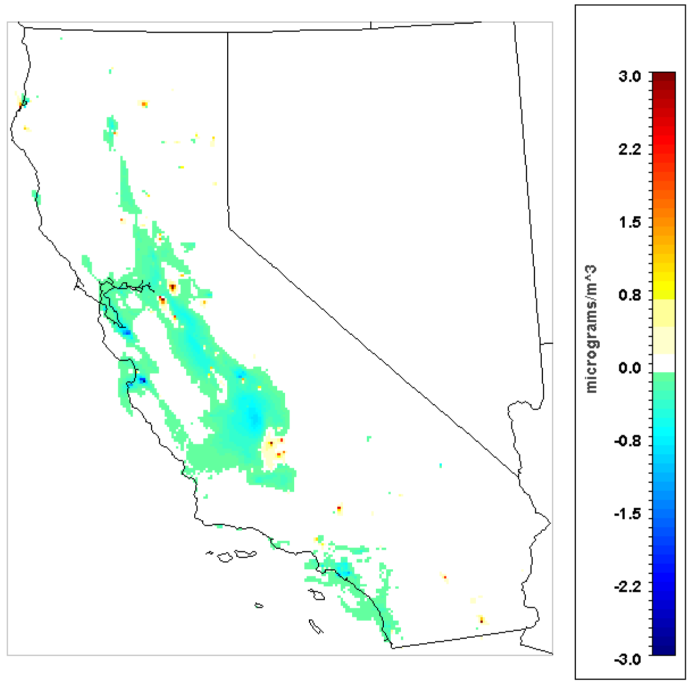


Figure 165: Difference in 24 hour average $PM_{2.5}$ in the Winter Industrial 2050 Case from the Base Case

7.3.3.5 Transportation Sector 2050 Case

Summer

Figure 166 displays the difference in maximum 8 hour average ozone in the Summer Transportation 2050 Case from the Base Case. Quantitatively, impacts range from -2.61 to +1.69 ppb. Spatially, impacts on ozone include improvements in the Bay Area and SoCAB and worsening seen in plumes in the Central Valley and across the northern parts of the State. As would be expected, reductions occur as a result of reduced NO_x from vehicle tailpipes and petroleum fuel refineries while increases result from additional generator NO_x. However, the magnitude of peak reductions is greater than concentrations increases and covers a larger spatial area. Additionally, it should be considered that the improvements occur in large urban areas with high populations as a result of concentrated vehicle presence and thus are important in terms of health impacts. Contrastingly, much of the worsening occurs in the northern regions of the State with lower population density. Thus, the results from this scenario would generally be viewed as an AQ benefit to the State, although additional strategies to limit the increase in generator NO_x from the Bakersfield-area plants should be pursued.

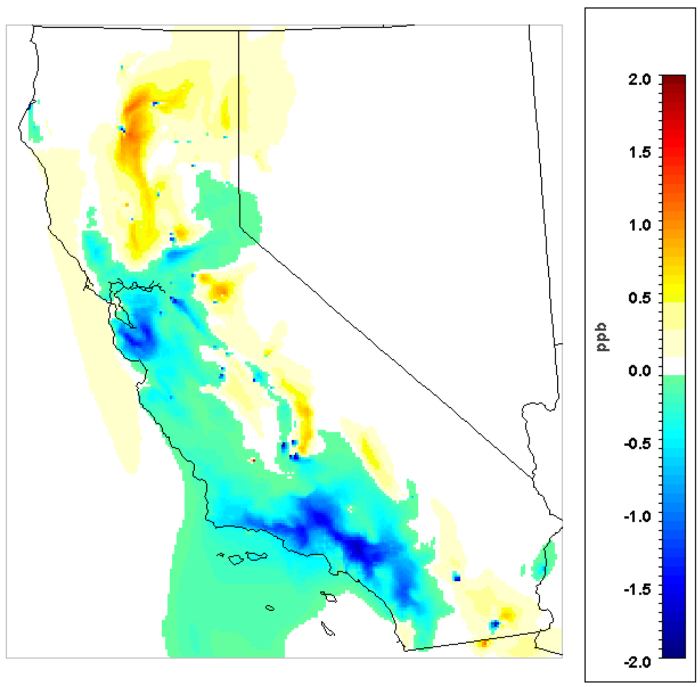


Figure 166: Difference in maximum 8 hour average ozone in the Summer Transportation 2050 Case from the Base Case

Figure 167 displays the difference in 24-h $PM_{2.5}$ in the Summer Transportation 2050 Case from the Base Case. Quantitatively, impacts range from -1.85 to +3.98 μg/m^3 and are similar in spatial effect to those observed for ozone. Notable improvements occur in the SoCAB and the Bay Area while worsening occurs in the Central Valley and northern region.

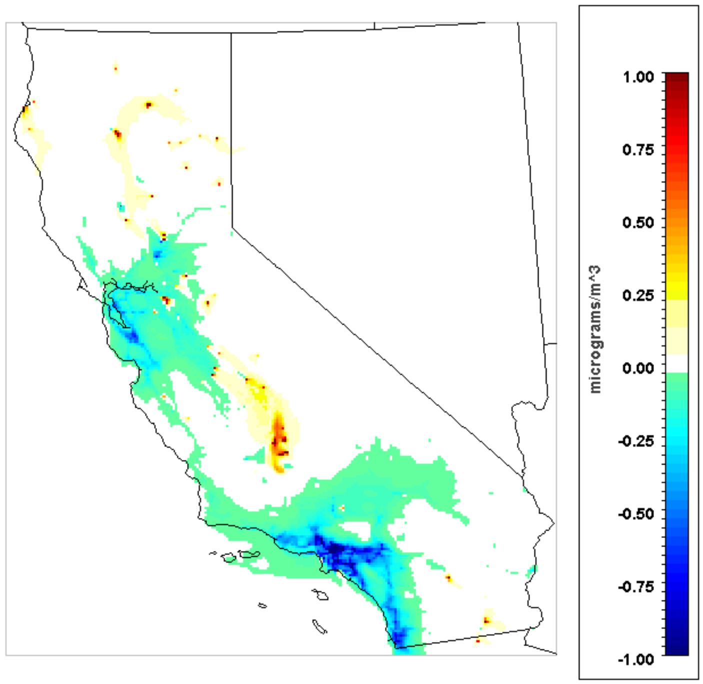


Figure 167: Difference in 24 hour average $PM_{2.5}$ in the Summer Transportation 2050 Case from the Base Case

Winter

Figure 168 displays the difference in maximum 8 hour average ozone in the Winter Transportation 2050 Case from the Base Case. Quantitatively, impacts range from -30.28 to +12.86 ppb. Impacts are significant, both in magnitude and in spatial coverage and largely are characterized by reductions in ground level concentrations. Areas of improvement include SoCAB, Central Valley, and the Bay Area. The wintertime dynamics of ozone raise questions however as relationships are generally inverse to NO_x emissions which is not observed in this scenario.

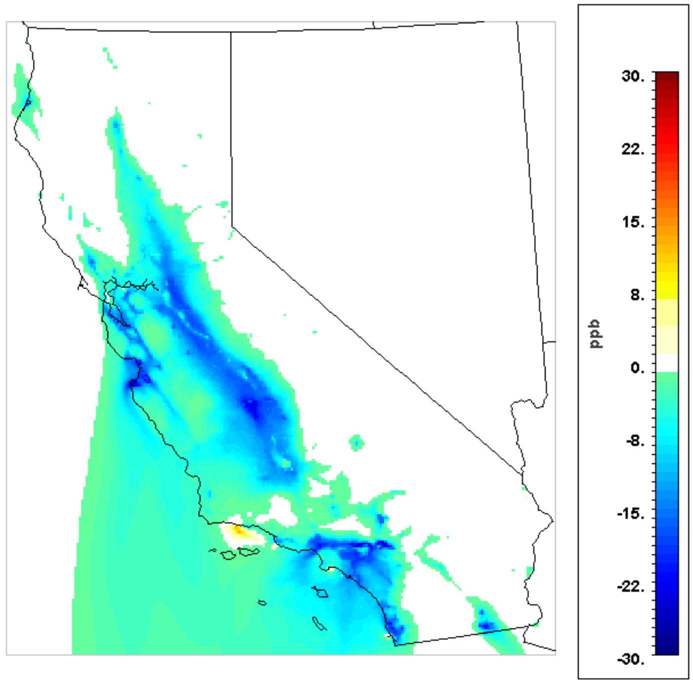


Figure 168: Difference in maximum 8 hour average ozone in the Winter Transportation 2050 Case from the Base Case

Figure 169 displays the difference in 24-h $PM_{2.5}$ in the Winter Transportation 2050 Case from the Base Case. Quantitatively, impacts range from -5.89 to +16.32 μg/m^3. The relative lack of PM emissions from LDV tailpipe generally results in worsening as increased generator emissions dominate total impacts. The Central Valley experiences a major area of worsening which is a concern due to existing AQ challenges – particularly winter PM levels.

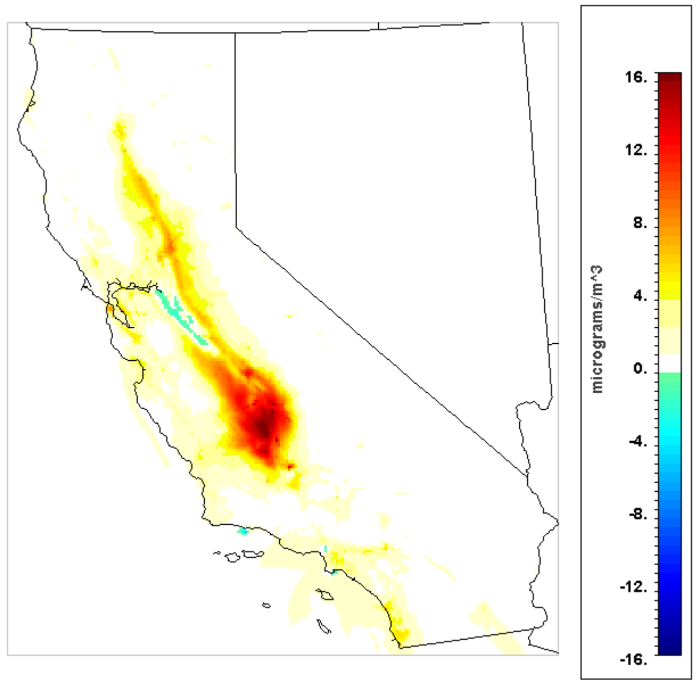


Figure 169: Difference in 24 hour average $PM_{2.5}$ in the Winter Transportation 2050 Case from the Base Case

7.3.3.6 Transportation Smart Charging 2050 Case

Summer

Figure 170 displays the difference in maximum 8 hour average ozone in the Summer Tra Smart Charging 2050 Case from the Base Case. Quantitatively, impacts range from -2.55 to +1.53 ppb. Significant reductions in ground level concentrations are observable across many regions of the Sate including SoCAB, the SF Bay Area, and much of the Central Valley. One notable area of concentration increase occurs in the northern portion of the state. However, these increases are on top of very low baseline ozone concentrations so that overall impacts on ozone are favorable as improvements occur in many urban regions and would thus offer important health benefits.

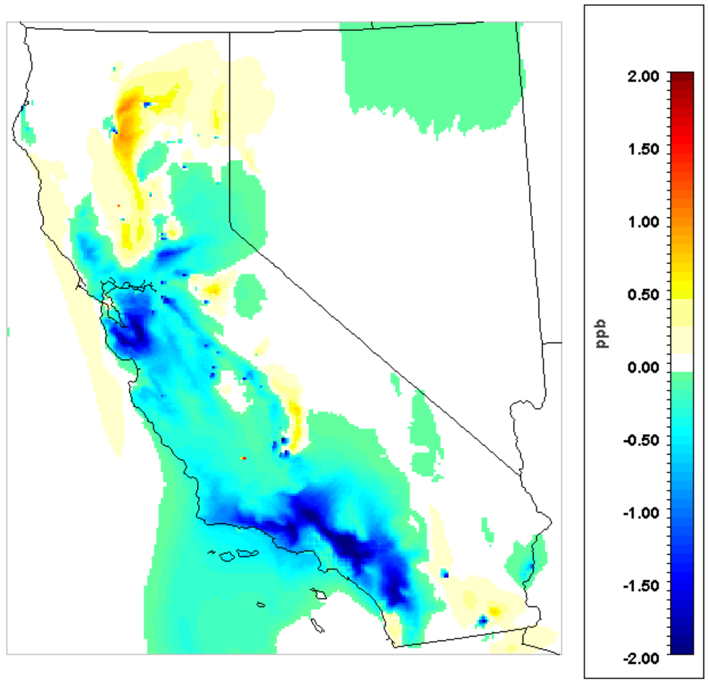


Figure 170: Difference in maximum 8 hour average ozone in the Summer Transportation Smart Charging 2050 Case from the Base Case

Figure 171 displays the difference in 24-h $PM_{2.5}$ in the Summer Transportation Smart Charging 2050 Case from the Base Case. Quantitatively, impacts range from -4.17 to +3.61 $\mu g/m^3$. Impacts are largely characterized by improvements in the SoCAB and the SF Bay Area. In particular, reductions in the SoCAB represent the largest impact in the Case. Small, localized increases occur in the same location as ozone increases but are dominated by improvements.

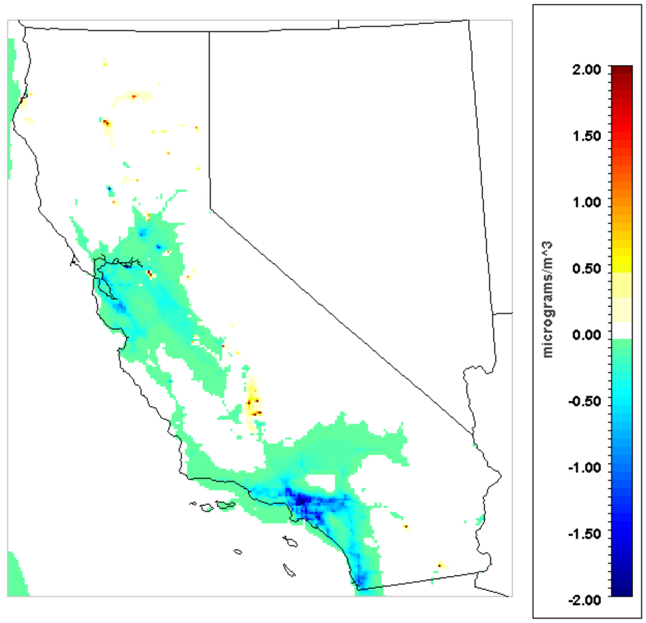


Figure 171: Difference in 24-h $PM_{2.5}$ in the Summer Transportation Smart Charging 2050 Case from the Base Case

Winter

Figure 172 displays the difference in maximum 8 hour average ozone in the Winter Transportation Smart Charging 2050 Case from the Base Case. Quantitatively, impacts range from -1.15 to +2.26 ppb. Spatially, impacts include areas of worsening in the SoCAB, Bay Area, and Sacramento. Areas of improvement include in and around Bakersfield. However, the winter time ozone impacts are less of a concern due to seasonal differences discussed in the results section regarding the 2020 Winter scenarios.

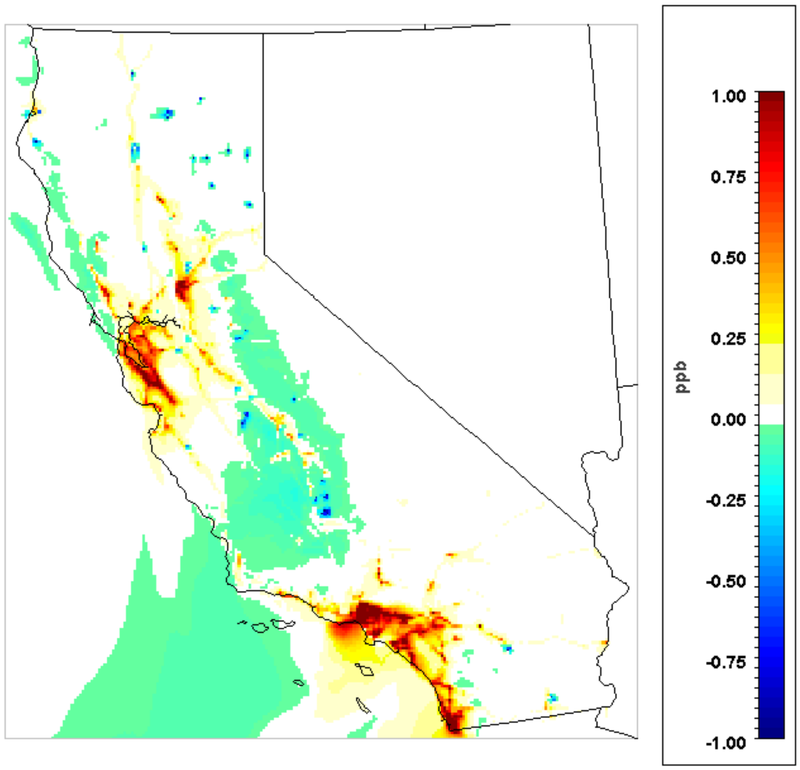


Figure 172: Difference in maximum 8 hour average ozone in the Winter Transportation Smart Charging 2050 Case from the Base Case

Figure 173 displays the difference in 24-h $PM_{2.5}$ in the Winter Transportation Smart Charging 2050 Case from the Base Case. Quantitatively, impacts range from -3.16 to +1.13 $\mu g/m^3$. The magnitude of improvements are substantial and occur in important areas for winter time PM levels including the SoCAB, Central Valley, SF Bay Area, and Sacramento Area. Additionally, increases in concentrations are minor to reductions and thus not visible at the given scale. Thus, the Winter Smart Charging Transportation 2050 Case achieves important benefits to AQ in 2050.

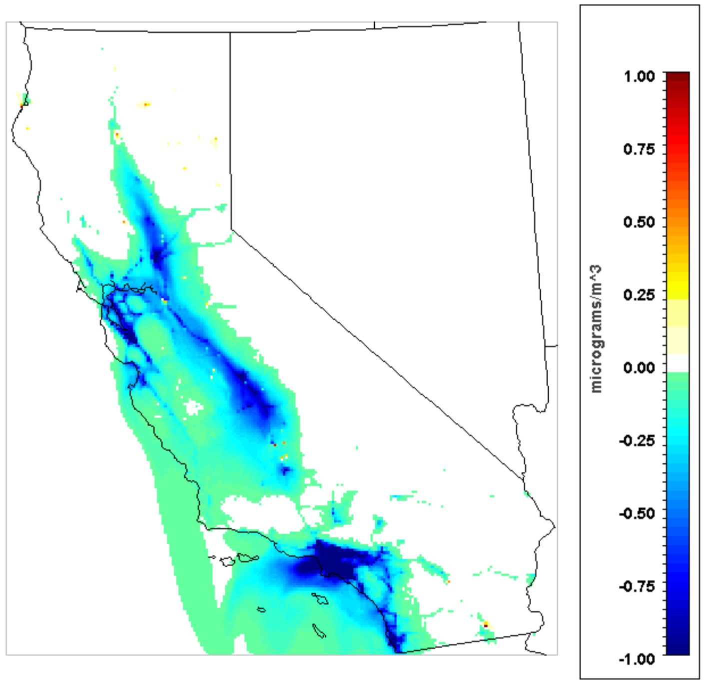


Figure 173: Difference in 24-h $PM_{2.5}$ in the Winter Transportation Smart Charging 2050 Case from the Base Case

Comparison of Immediate and Smart Charging Air Quality Impacts

In order to assess the AQ impacts of smart relative to immediate charging difference plots were generated for the Transportation Smart Charging 2050 Case relative to the Transportation 2050 Case which assumes immediate charging of vehicles. Thus the following figures display spatial and temporal distributions of pollutants such that negative values represent enhanced reductions and positive values represent increased concentrations when smart charging is deployed.

Summer

Transitioning to smart charging of electric vehicles significantly improves ozone concentrations relative to immediate charging. Figure 174 shows the difference in maximum

8-hr ozone from smart charging for the Summer 2050 Transportation Case. Quantitatively, impacts range from -1.05 to +0.71 ppb. The peak reductions are particularly significant given that the peak reduction of the Summer Tra2050 Case is -2.61 ppb. Peak improvements occur in many part of the State including SoCAB, SF Bay Area, and Central Valley. Evident are reductions from increased vehicle levels leading to reductions in direct emissions and reductions in emissions from power plants from the avoidance of ramping. Thus, the smart charging of vehicles can achieve important improvements in AQ benefits relative to immediate charging in terms of summer ozone levels in 2050.

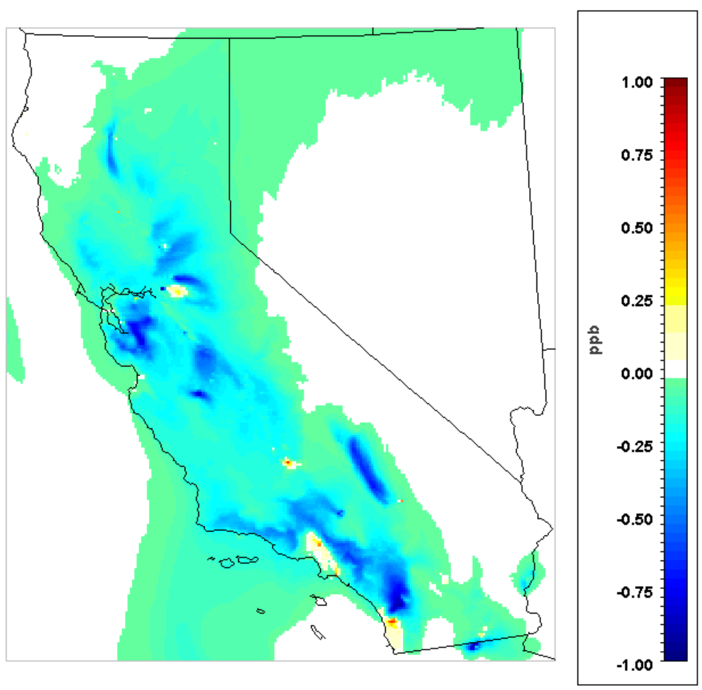


Figure 174: Difference in maximum 8-hr ozone between the Smart and Immediate Charging Summer Transportation 2050 Cases

Figure 175 shows the difference in 24-hr $PM_{2.5}$ from smart charging for the Summer 2050 Transportation Case. Quantitatively, impacts range from -5.13 to +0.59 μg/m³. With similarity to the ozone difference, impacts are characterized by improvements in many areas of the State including the SoCAB, Central Valley, and SF Bay Area. While peak reductions reach high levels, the majority of impacts are lesser. Still, a transition to smart charging achieves notable improvements in summer $PM_{2.5}$ levels from reductions in vehicle and power plant emissions.

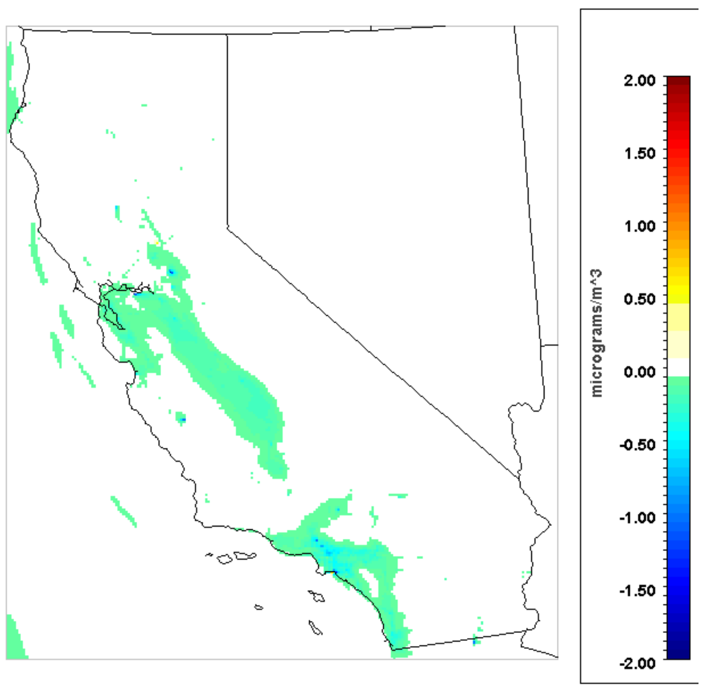


Figure 175: Difference in 24-h $PM_{2.5}$ between the Smart and Immediate Charging Summer Transportation 2050 Cases

Winter

Figure 176 shows the difference in maximum 8-hr ozone from smart charging for the Winter 2050 Transportation Case. Quantitatively, impacts range from -14.53 to +29.59 ppb.

Impacts are largely characterized by significant increases throughout the State. Despite increases, the winter ozone dynamics limit the importance of the effects.

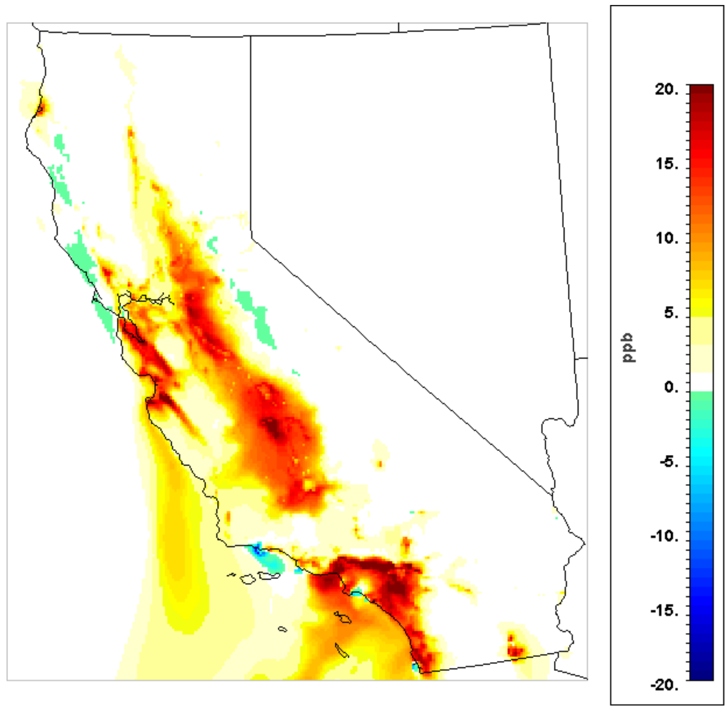


Figure 176: Difference in maximum 8-hr ozone between the Smart and Immediate Charging Winter Transportation 2050 Cases

Figure 141 shows the difference in 24-hr $PM_{2.5}$ from smart charging for the Winter 2050 Transportation Case. Quantitatively, impacts range from -17.31 to +5.78 μg/m^3. Impacts are largely characterized by dramatic improvements in several key areas of the State. The largest area of reduction is concentrated in the Central Valley and extends northward through Sacramento. The magnitude of the difference is particularly large.

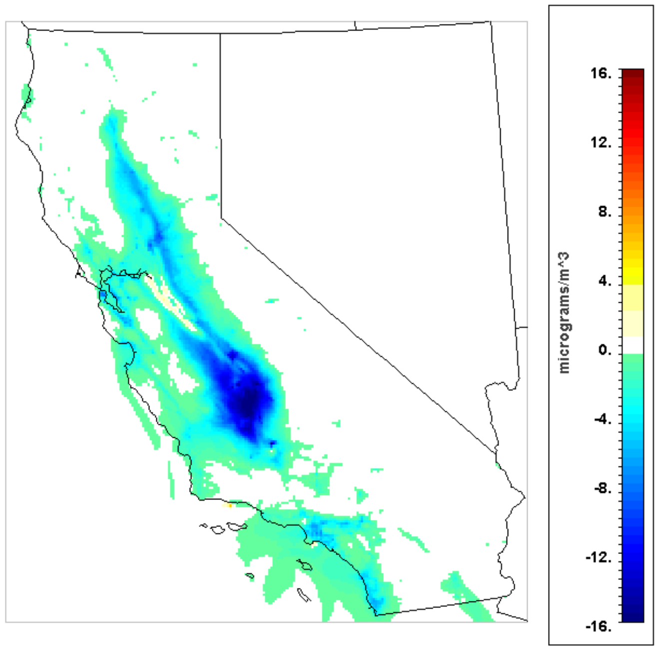


Figure 177: Difference in 24-h $PM_{2.5}$ between the Smart and Immediate Charging Winter Transportation 2050 Cases

7.3.3.7 ResComTra 2050 Case

Summer

Figure 178 displays the difference in maximum 8 hour average ozone in the Summer ResComTra 2050 Case from the Base Case. Quantitatively, impacts range from -3.78 to +4.76 ppb. Results are generally additive and are characterized by large areas of improvement throughout the State. One notable area of worsening occurs over Bakersfield as a result of increased NO_x from large gas generators located in the region. However, overall impacts are generally favorable.

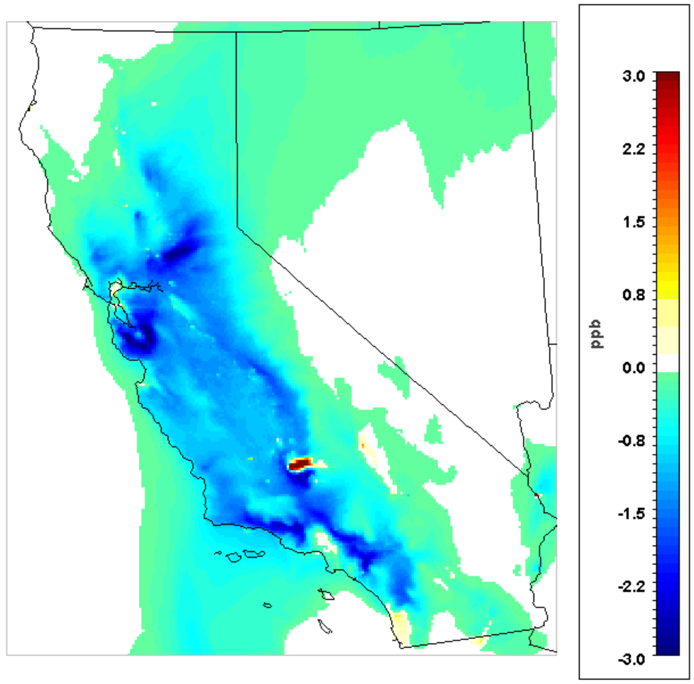


Figure 178: Difference in maximum 8 hour average ozone in the Summer ResComTra 2050 Case from the Base Case

Figure 179 displays the difference in 24-h $PM_{2.5}$ in the Summer ResComTra 2050 Case from the Base Case. Quantitatively, impacts range from -3.82 to +1.79 μg/m^3. Impacts are largely characterized by improvements throughout the State including SoCAB, the SF Bay Area, and the Central Valley, and the case represents an opportunity to improve summer AQ in terms of $PM_{2.5}$.

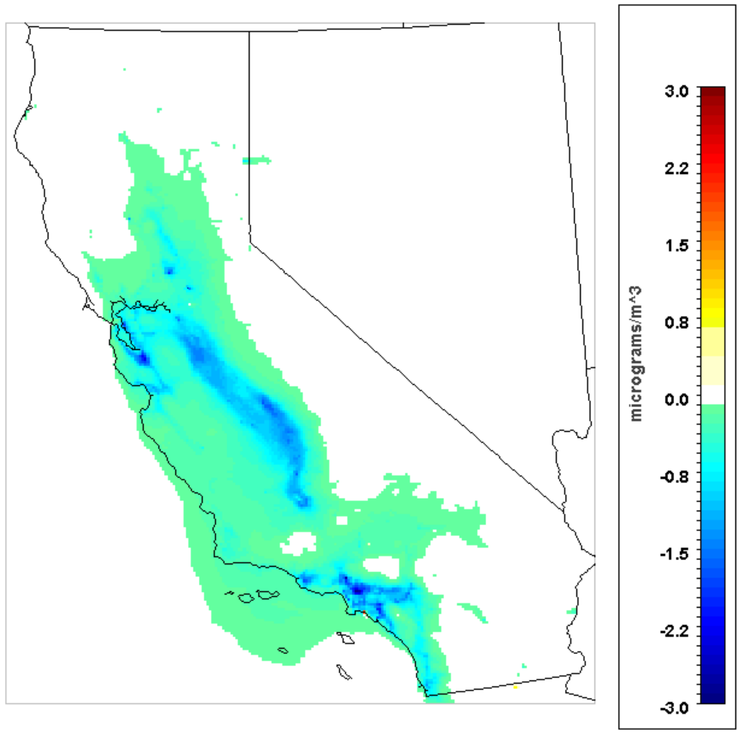


Figure 179: Difference in 24 hour average $PM_{2.5}$ in the Summer ResComTra 2050 Case from the Base Case

Winter

Figure 180 displays the difference in maximum 8-hour average ozone in the Winter ResComTra 2050 Case from the Base Case. Quantitatively, impacts range from -14.13 to +31.69 ppb.

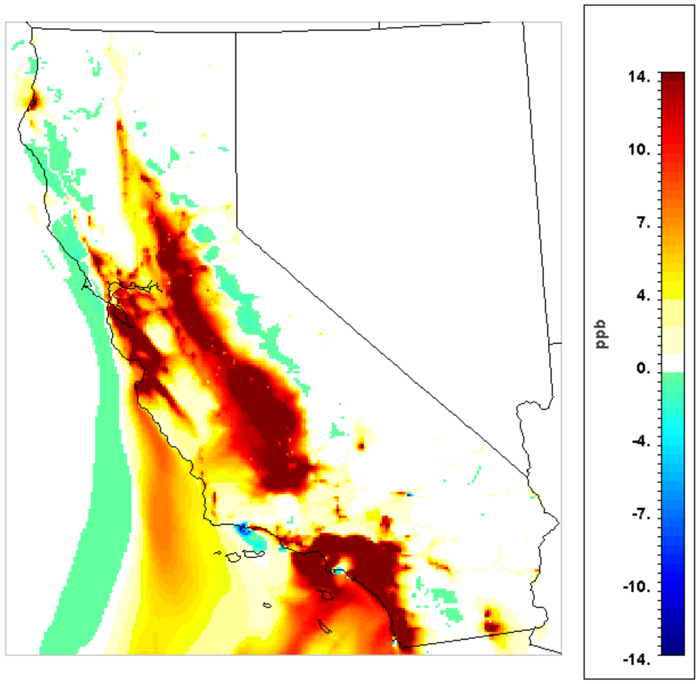


Figure 180: Difference in maximum 8 hour average ozone in the Winter ResComTra 2050 Case from the Base Case

Figure 181 displays the difference in 24-h $PM_{2.5}$ in the Summer ResComTra 2050 Case from the Base Case. Quantitatively, impacts range from -22.34 to +0.20 µg/m^3.

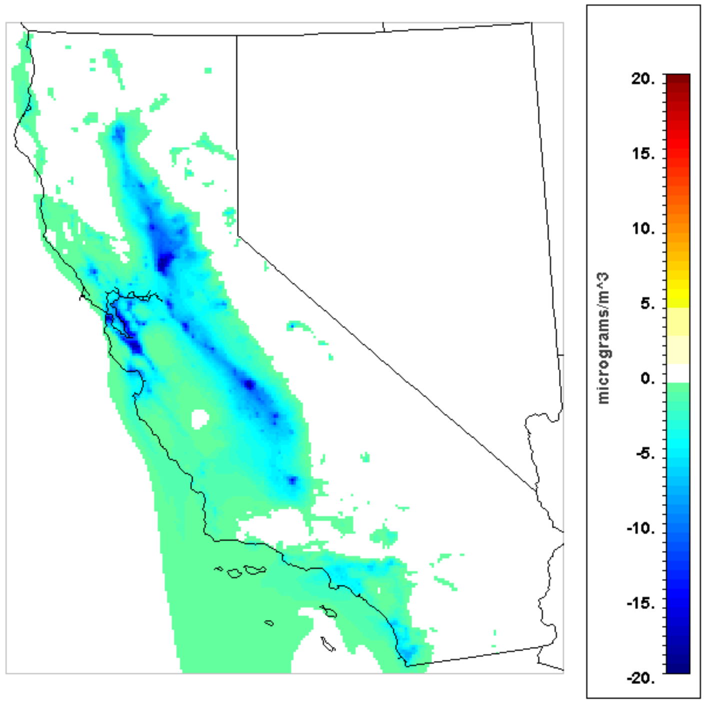


Figure 181: Difference in 24 hour average $PM_{2.5}$ in the Winter ResComTra 2050 Case from the Base Case

7.3.3.8 Summary of 2050 Cases

Table 57 displays the peak impacts on 8 hour average ozone and 24 hour $PM_{2.5}$ for the Summer 2050 Cases relative to the Base Case. Table 58 displays the peak impacts on 8 hour average ozone and 24 hour $PM_{2.5}$ for the Winter 2050 Cases relative to the Base Case. Impacts on $PM_{2.5}$ and ozone are substantial for all electrification scenarios in 2050 and reflect a very high electrification and renewable penetration for the sectors of study. Impacts on max 8-hr ozone range from -30.28 in the Winter Tra Case to +12.86 in the same case.

Table 59: Summary of peak impacts on 8-hr max ozone and 24-h $PM_{2.5}$ for Summer 2050 Cases

Summer Case	8-hr Ozone [ppb]	24-hr $PM_{2.5}$ [$\mu g/m^3$]
Res 2050	-2.12 to +1.07	-1.02 to +3.37
Com 2050	-2.00 to +5.29	-1.37 to +2.06

ResCom 2050	-3.83 to +5.48	-1.41 to +5.61
Ind 2050	-7.10 to +3.58	-1.10 to +12.40
Tra 2050	-2.61 to +1.69	-1.85 to +3.98
Tra Smart 2050	-2.55 to +1.53	-4.17 to +3.61
ResComTra 2050	-3.78 to +4.76	-3.82 to +1.79

Table 60: Summary of peak impacts on 8-hr max ozone and 24-h $PM_{2.5}$ for Winter 2050 Cases

Winter Case	8-hr Ozone [ppb]	24-hr $PM_{2.5}$ [$\mu g/m^3$]
Res 2050	-10.02 to +3.64	-13.33 to +0.25
Com 2050	-1.60 to +7.27	-4.43 to +0.79
ResCom 2050	-13.45 to +7.46	-20.62 to +1.77
Ind 2050	-13.61 to +5.55	-3.22 to +12.28
Tra 2050	-30.28 to +12.86	-5.89 to +16.32
Tra Smart 2050	-1.15 to +2.26	-3.16 to +1.13
ResComTra 2050	-14.13 to +31.69	-22.34 to +0.20

Chapter 8: Summary and Conclusions

8.1 SUMMARY

Many options exist for addressing emissions and reducing the regional AQ impacts of sources in energy sectors moving forward. Assessments were conducted with the understanding that lowering concentrations of atmospheric pollutants must occur in tandem with the continued meeting of societal energy needs. Further, analyses were performed in future years in three different regions to include the impacts of regional energy system evolution and variation.

A methodology was developed for use in assessing the impacts on regional AQ of alternative energy strategies deployed to address both GHG and pollutant emissions from key sources. First, major sources of pollutant emissions are evaluated for potential impacts on ozone and $PM_{2.5}$ to ascertain key sources that should be considered with priority for mitigation strategies. Next, mitigation strategies for identified sources are considered and assessed for their potential to actually improve AQ using an AQ model in each region. The following sections highlight the important findings from this work. These results can be used by government organizations, industry sources, community leaders, or additional stake holders whose goal is to develop and deploy strategies to achieve regional AQ improvement in tandem with GHG emissions reductions.

Figure 182 displays the relative findings from this work for the AQ and GHG emission impacts of advanced power generation strategies considered in this work. Potential impacts on regional AQ correspond to the y axis while impacts on GHG emissions correspond to the x axis; with "high" representing potential for improvement in the case of AQ and reduction in life cycle emissions for GHGs. The ellipse surrounding the strategies represents the range of potential impacts a given strategy can have. It should be noted that the impacts are qualitative in nature to facilitate discussion.

As can be seen, renewable resources (with biopower considered separately) can provide significant benefits to AQ and GHG emissions but maximization will require the co-deployment of advanced complementary strategies including energy storage, demand response, vehicle-2-grid, etc. Integration of high levels of intermittent resources can even yield localized worsening of AQ in the absence of such strategies. Biopower is highly complex with the potential for very large GHG benefits if waste streams and advanced conversion devices are utilized. Contrastingly, the use of energy crops and traditional conversion devices could have AQ and GHG impacts with limited benefit or even dis-benefit. Nuclear power is a key strategy for providing power with very low life cycle GHG emissions and benefits to AQ, particularly if coal power plants are replaced. While currently limited by societal concerns, nuclear power offers reliable and reasonably cost-effective electricity and should be considered for low carbon energy portfolios. CCS technology can provide significant reductions in GHG emissions, although less than renewable and nuclear power. However, CCS could potentially increase emissions from power plants fleet-wide with AQ dis-benefits, although it should be noted that traditional technologies were considered here and advanced CCS technologies could avoid such impacts. Efficiency measures are important to GHG and AQ improvement planning but will only be able to achieve a modest improvement if deployed singularly as a GHG mitigation strategy. Similarly, advanced NG combustion can offer AQ and GHG benefits of a limited nature when replacing coal plants.

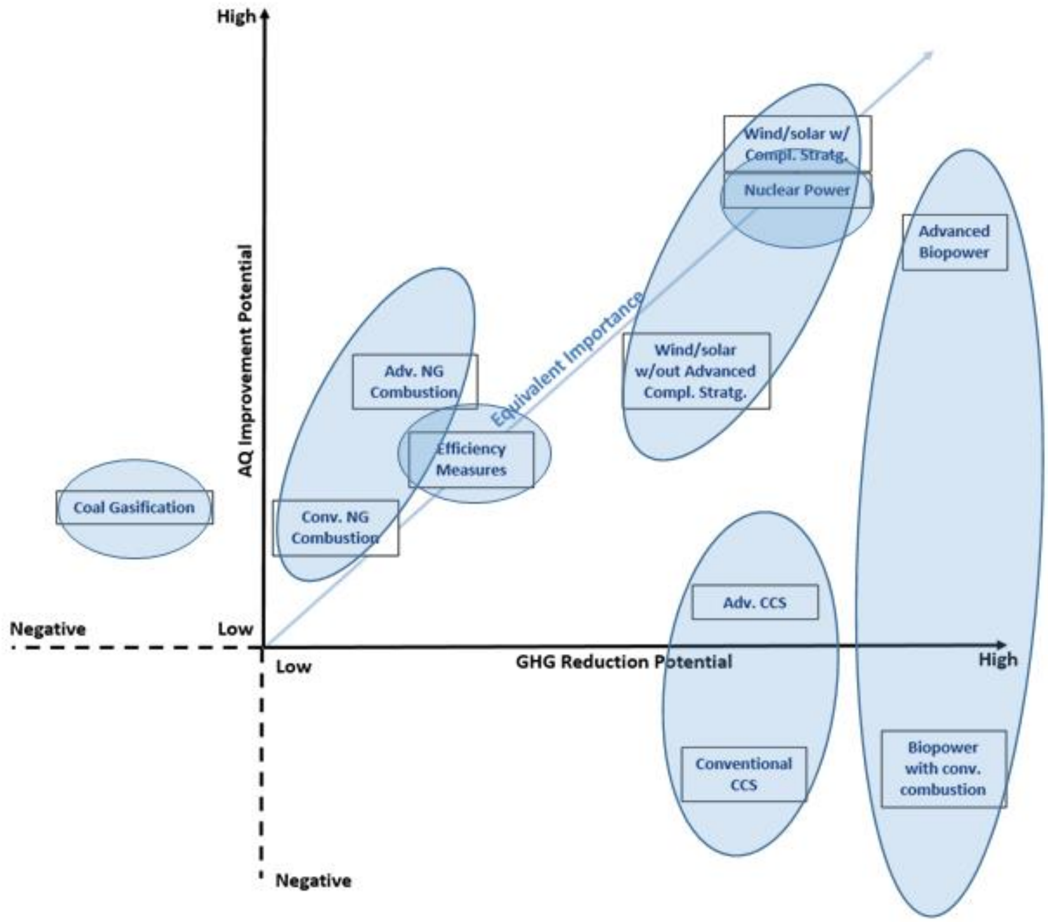


Figure 182: Potential relative impacts on air quality (AQ) and greenhouse gas emissions (GHG) from alternative electricity generation strategies

Figure 183 displays the relative findings from this work for the AQ and GHG emission impacts of advanced LDV energy strategies with the same parameters described above for Figure 182. The two strategies with the highest potential for AQ and GHG co-benefits include electricity in all-electric battery powered vehicles (BEV) and hydrogen in fuel cell electric vehicles (FCEV). However, the numerous potential pathways for generating hydrogen and electricity yield a significant range of possible impacts. For example, when renewable energy is utilized for either fuel the maximum AQ and GHG co-benefits are achieved. Contrastingly, if coal power is utilized benefits are still possible but significantly reduced. Current common methods including an average grid and SMR production result benefits. The use of biofuels results in a wide range of impacts reflecting the complexity of biofuel production similar to those discussed above for biopower. If waste streams are utilized, biofuels can provide high GHG benefits although if combustion engines are utilized the benefits are lesser because of direct tail pipe emissions. Efficiency measures to traditional LDV technologies (i.e., internal combustion engines) can reduce emissions modestly. The use of CNG in combustion engines achieves a minor benefit relative to gasoline.

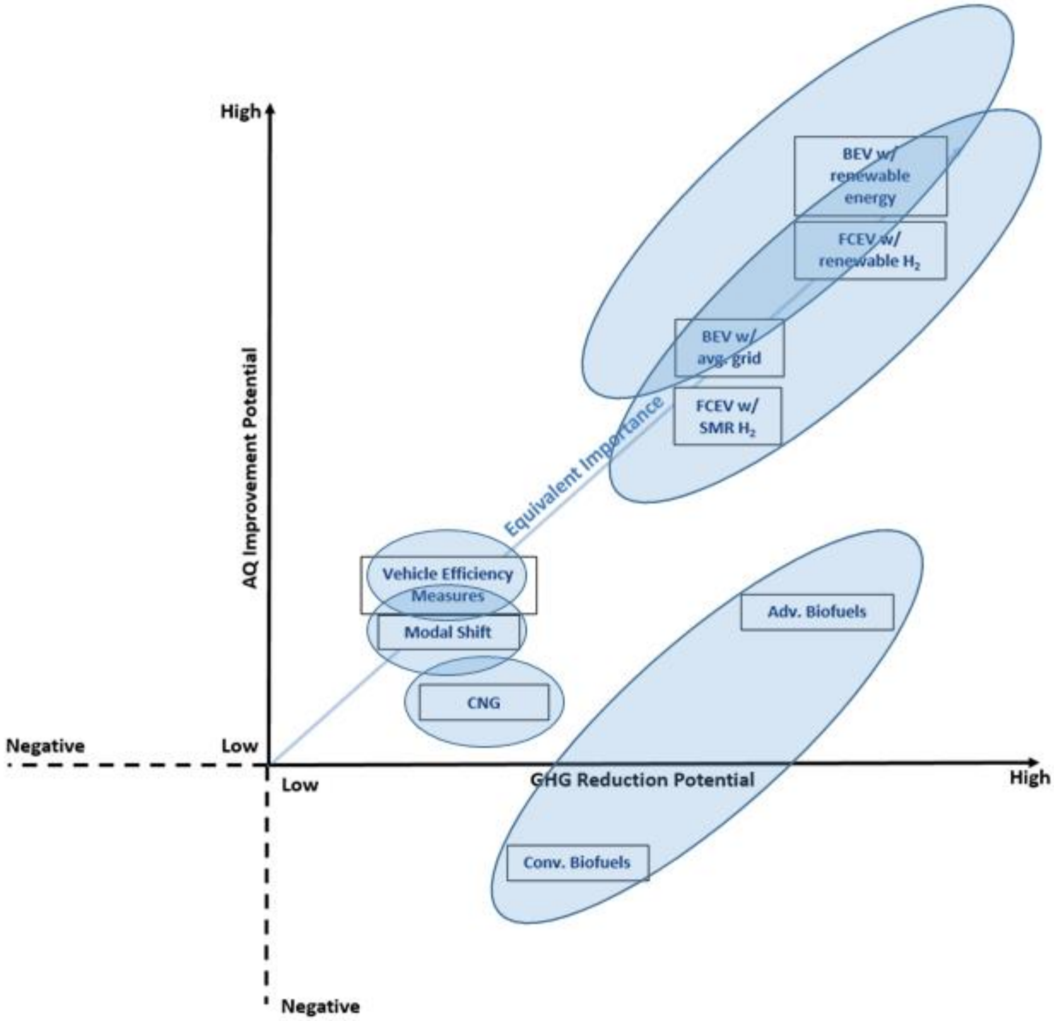


Figure 183: Potential relative impacts on air quality (AQ) and greenhouse gas emissions (GHG) from alternative Light Duty Vehicle (LDV) energy strategies

8.2 CONCLUSIONS

8.2.1 Assessment of Impacts from Key Sources

Various sources of emissions from regional energy sectors are evaluated for resulting impacts on primary and secondary pollutant atmospheric concentrations in 2055 including various transportation sub-sectors, coal power plants, petroleum fuel infrastructure, and industries.

- **Mitigating AQ impacts from regional energy sectors will require consideration of different quantitative, spatial, and temporal effects on pollutants**

Assessing regional AQ impacts to determine mitigation efforts must consider spatial and temporal emissions impacts in addition to quantifying concentration reductions. For example, although the 50T/50I Case has a larger peak $PM_{2.5}$ improvement it is only slightly more than the 50E/50I Case which encompasses a greater area of impact. Thus, it could be argued that the electricity and industrial sector case is the most important from a $PM_{2.5}$ standpoint. The results also show the differences in spatial and temporal impacts on secondary pollutants that are associated with emissions from various energy sectors. Thus, reducing equivalent quantities of direct emissions from different sectors can have very different impacts on the formation and fate of secondary pollutants. Further, these results demonstrate that mitigation strategies which reduce emissions in important sectors achieve greater improvements than balanced reductions of a lower magnitude across all sectors, i.e., comparatively the 50T/50E and 50T/50I Cases yield significant improvements in ozone relative to the All 25 Case. In addition, the temporal pattern of ozone and $PM_{2.5}$ formation and fate differ with maximum impacts for the 50T/50I and 50T/50E Cases occurring in densely populated regions with human health implications.

- **Optimal emission reduction technology targets for GHG mitigation may differ from those for improving regional AQ**

The results demonstrate that certain sectors are more important for GHG reductions while sometimes other sectors are more important for regional AQ. Due to this, achieving maximum reductions in GHG emissions may not necessarily correspond to the best AQ improvements. From a GHG standpoint, strategies that target low- or zero-carbon power generation should be pursued while transportation sector strategies with low- or zero-NO_x emissions would be most effective in addressing regional ozone concerns. Contrastingly, targeting $PM_{2.5}$ reductions would be best served by addressing sources in the industrial and power sectors. For example, the 50E/50T scenario achieves the greatest reduction in GHG emissions for the TX study region despite the 50T/50I scenario maximally improving AQ in some locations. However, the 50E/50T case does significantly lower concentrations of ozone and $PM_{2.5}$ and offers AQ co-benefits. Thus, decisions prioritizing targeted sectors and fuels for strategy deployment must balance the effectiveness of GHG mitigation and AQ improvement in deciding which endpoint is more desirable in a region with regards to political and societal goals.

- **Transportation sub-sectors will experience different evolution patterns which impacts emissions and resulting relative AQ impacts**

Future impacts on ozone and $PM_{2.5}$ are not equivalent to current impacts with some sources increasing in relative importance and others becoming less important. In particular, the current focus on reducing fuel consumption and emissions from LDVs at nearly all levels of U.S. and state governments results in a baseline fleet in 2055 that is low-emitting and therefore has moderate to minor AQ impacts relative to other sources. In contrast, emissions from ships have a major effect on ground-level concentrations of pollutants, particularly for regions supporting major ports. Additionally, HDV and off-road emissions result in significant atmospheric pollution in all regions. Thus, regional AQ planning should consider non-LDV sources with high priority in the development of mitigation strategies.

- **The future regional AQ impacts of LDVs may be moderate but LDV should still be considered for alternative technology deployment in GHG mitigation**

The moderate AQ impacts attributable to LDVs in 2055 relative to other transportation sources should be evaluated in the context that, (1) improvements in ozone and $PM_{2.5}$ occur in populated urban regions and thus have human health implications, and (2) LDVs will continue to be an important source of domestic GHG emissions even if they become less important to meeting AQ goals by significantly reducing pollutant emission rates. Additionally, the considerable effects on ozone and $PM_{2.5}$ of producing and distributing motor gasoline should be considered together with those directly emitted from vehicles. Thus, LDVs will continue to represent an important opportunity for alternative low-emitting technologies and fuels in coming decades. However, it may be more effective to pursue mitigation strategies on the basis of GHG reductions.

- **For regions supporting major ports mitigating associated ship emissions is a priority for regional AQ improvement**

The magnitude and spatial dimension of both primary and secondary pollutant impacts occurring from ship emissions highlights the fact that water-borne vessels should be a major and required target for future mitigation strategies that seek to improve AQ in the current study regions. In particular, the current work shows that ship activities in locations of major shipping ports emerge as the dominant contributors to transportation-related regional air pollution in 2055, particularly in TX and CA.

- **Emissions from goods movement represents a key source of air pollution currently and in the future, particularly for regions containing ports**

Goods movement is facilitated by many of the technologies identified as having important impacts on ozone and $PM_{2.5}$ (ships, HDVs, off-road and rail) which compound deleterious AQ impacts in study regions. If one additionally considers the co-location of emission sources comprising the entire goods movement sector including ships, off-road, HDV, and rail, port activities emerge as the most important of activities to address in order to meet AQ goals. A current understanding of the harmful AQ impacts of port activity exists and programs and policies are in place and/or under development to seek emission reductions from the aforementioned technologies. However, expected growth in demand for global shipping in tandem with reduced emissions from other sectors (e.g., LDVs) will increase the importance of reducing port emissions. Further, operational and other constraints (e.g., fuel energy density requirements of ships) increase the difficulty of deploying alternative strategies for some goods movement technologies. Thus, the current results support the urgent need for research, development, and deployment plans for advanced, low emissions port-related technologies.

- **Emissions from producing and distributing petroleum fuels should be prioritized in transportation emissions mitigation efforts with the same priority as vehicle tailpipe emissions**

Significant impacts on ozone and $PM_{2.5}$ occur from petroleum fuel production and distribution sources and, as a result, these sources merit investigation for mitigation strategies, particularly for large refinery complexes. In particular, activities associated with fuel production in CA and TX contribute greatly to regional AQ burdens e.g., in TX reductions in ozone levels from removing petroleum fuel pathway emissions exceed those from removing direct emissions from individual transportation sub-sectors. Further, in all alternative LDV cases impacts from PFI emissions were a key driver of overall AQ impacts, e.g., the largest impacts on $PM_{2.5}$ from deploying FCEVs and BEVs in CA arise from emission reductions from petroleum fuel infrastructure. Thus, maximizing the AQ benefits of deploying alternative transportation technologies and fuels will require corresponding reductions in emissions from petroleum fuel production. While increasing numbers of alternative fueled vehicles will reduce gasoline consumption, it is unknown if refineries will reduce output or emissions. The results here demonstrate that achieving high AQ benefits from LDV emissions mitigation strategies will likely require deployment of both alternative LDV technologies and policies and programs to reduce PFI emissions.

- **Industrial sector sources should be targeted in regional AQ improvement planning with high priority. However, significantly more information is needed to identify optimal technology and behavioral pathways to maximize AQ and GHG benefits.**

Contributions of emissions from industry to regional pollutant burdens were significant in 2055 for all regions studied, e.g., in CA industry had a larger effect than power generation. Thus, developing and applying mitigation strategies that can remove emissions from industry should be considered with priority by entities seeking regional AQ improvements. However, the industrial sector is highly complex, composed of many different technologies, processes and energy demands that limit the current understanding of effective and feasible emission reduction opportunities. Further, factors such as economics and operational parameters impact possible replacement technologies and strategies. Thus, more information is needed to better understand how to address industrial AQ impacts and assessments of individual industry sectors are required to begin this process.

- **Regional variation in industry requires regional- and local-scale consideration for AQ improvement planning strategies**

The results from this work demonstrate differences in AQ impacts of industry sub-sectors amongst regions that require regional- and local-scale consideration for control strategy development. Some industries uniformly impact AQ across regions including chemical manufacturing, while others exhibit enhanced regional variation including primary and secondary metals production. In addition, within States different industries have spatial and temporal differences in their effects on AQ, e.g., in CA the chemical industry most impacts ozone in SoCAB while oil and gas production has peak impacts in and around Bakersfield. Thus, when developing effective control strategies for industrial emissions consideration must be made at the regional- and local-scale.

- **The production of crude oil and natural gas should be considered when assessing the AQ and GHG impacts of downstream fuels (e.g., gasoline), particularly in regions supporting significant natural gas and oil recovery**

The recovery, processing, and transport of natural gas and petroleum feedstocks is associated with emissions that contribute significantly to regional AQ burdens in 2055. This is particularly true for regions that support high levels of oil and gas industry, including TX. For example, removing oil and gas production emission in TX improves AQ across the State with peak impacts reaching -6.4 ppb and -1.7 $\mu g/m^3$. These impacts should be considered when assessing the overall impacts of produced fuels on AQ and GHG emissions. For example, the impacts of gasoline combustion engines in LDV on emissions and AQ is well understood but the upstream impacts of recovering the crude oil resources is generally not considered when evaluating the replacement of current LDV with advanced, low-emissions technologies.

- **Coal power generation will continue to have major deleterious impacts on AQ and GHG emissions even if natural gas surpasses coal as the dominant fossil resource.**

The continued use of coal for power generation represents a major target for regional AQ and GHG improvements and should be addressed with high priority The high emissions of both GHG and criteria pollutants associated with coal power plants are such that even expected reductions in total power generation resulting from displacement with natural gas and renewables is not enough to offset harmful impacts. Thus, coal power represents a foremost target to address in terms of regional AQ improvement and GHG reductions for regions supporting any level of coal utilization in the future.

- **Carbon Capture and Storage represents a GHG mitigation strategy with the potential for AQ dis-benefits and should be examined further for methods to avoid increases in atmospheric pollution**

Efficiency penalties associated with post-combustion amine capture CCS technologies could increase emissions leading to increases in ozone over 2 ppb if deployed on all fossil power generators in TX. Nuclear power represents a strategy for mitigating coal plant GHG emissions with similar operating characteristics that would achieve regional AQ benefits in tandem. It is estimated that using nuclear plants in place of coal could achieve reductions in ozone and $PM_{2.5}$ around 7 ppb and 2.3 μg/m^3, respectively.

8.2.2 Mitigation of LDV Air Quality Impacts

Mitigation of the GHG and pollutant emissions from LDVs is an important target for future efforts to mitigation climate change and regional AQ. Transitions to electricity in EVs and hydrogen in FCEVs represent a foremost strategy in displacing conventional vehicles. Various scenarios involving BEVs and FCEVs were assessed for impacts on ozone and $PM_{2.5}$ including variations in associated penetration levels and infrastructure.

- **The deployment of FCEVs at high levels can achieve important AQ benefits in urban areas, particularly if low emitting hydrogen pathways are utilized**

The deployment of high levels of FCEVs (i.e., 50-90%) LDV sector penetration achieves significant benefits to AQ in California including reductions in ground-level concentrations potentially greater than 4 ppb ozone and 4 $\mu g/m^3$ $PM_{2.5}$. The greatest AQ impacts occur in key regions of the state where high urban populations are located and where poor AQ conditions already occur, including the SoCAB, SF Bay Area, and the Central Valley.

- **In CA, AQ impacts of alternative LDVs are driven by tailpipe and petroleum refinery emissions with lesser impacts from the power sector**

The largest impacts on ozone and $PM_{2.5}$ in BEV and FCEV scenario assessments occurred from changes in direct vehicle and petroleum fuel infrastructure emissions. In contrast, alterations to emissions from power generators (both positive and negative) had a lesser impact. However, CA's power system is low emitting compared to other U.S. regions and the same conclusion may not be appropriate for all States.

- **The integration of FCEV deployment and renewable resources via renewable powered electrolysis is an effective strategy for maximizing regional AQ benefits and providing benefits to the grid**

The utilization of renewable power generation to produce hydrogen via electrolysis is a foremost strategy in integrating the power and transportation sectors in pursuit of GHG reductions and AQ improvements. In addition to providing potential benefits to both the grid and transportation system, the coupling of hydrogen fueling and renewable energy can provide maximal benefits to atmospheric pollution from FCEV deployment.

- **Concerns regarding environmental justice in terms of advanced LDV should consider that pollution benefits from FCEV and EV deployment often occur in sites distant from vehicle deployment**

The deployment locations of vehicles are not necessarily correlated with areas of AQ improvement and benefits in priority areas of concern can be achieved even in the absence of vehicle deployment in those locations. The dynamics of atmospheric pollutant formation and fate govern spatial and temporal variance in sites of emissions and regions of greatest final impact. This is particularly true for concentrations of tropospheric ozone due to the transport occurring during the temporal period associated with photochemical reactions. Thus, emissions reductions from deployed FCEVs can yield reductions in ozone in non-adjacent locations. An example of this includes the peak reductions in ozone that occur in San Bernardino and Riverside Counties when FCEVs are deployed in Orange and Los Angeles Counties.

- **Policies targeting emissions reductions from petroleum fuel infrastructure can maximize the benefits of deploying alternative LDV technologies**

For all scenarios involving alternative non-gasoline LDVs, assumptions regarding emissions from petroleum fuel infrastructure were a major determinant of overall AQ impacts. For example, differences between turning down refinery emissions and leaving them baseline reach -1.2 ppb ozone and 4.75 $\mu g/m^3$ $PM_{2.5}$ for a 90% penetration of FCEVs. CA has developed programs and policies supporting the increased penetration of alternative, low-emitting vehicle technologies including EVs and FCEVs. However, it is unknown if reduced gasoline consumption State-wide will translate to reductions in output or emissions from petroleum fuel infrastructure. Thus, additional policies designed to address such emissions can assist the State and maximizing the emissions and AQ benefits of LDV programs.

- **The deployment of electric LDVs largely improves regional AQ, even in regions supporting higher emitting generators, although localized worsening from large coal plants should be considered**

The large scale deployment of electric LDVs achieves reductions in ozone and $PM_{2.5}$ that peak downwind of urban areas. The results suggest that EV deployment in CA could represent an opportunity to simultaneously address GHG and AQ, even if charging strategies include primarily fossil generation. This differs in TX which experiences some worsening of ozone and $PM_{2.5}$ levels localized to large coal plant locations that must be considered for human health impacts. However, impacts in TX are still beneficial in many areas including It should also be noted that these results encourage the deployment of EVs in tandem with strategic charging and additional complementary strategies as both will be required to maximize GHG reductions.

8.2.3 Cold Ironing of Ocean Going Vessels

Emissions from ocean going vessels (OGV) were shown to have a major impact on regional ozone and $PM_{2.5}$ levels in 2055. A prominent strategy to reduce emissions from OGVs includes cold ironing involving the displacement of auxiliary engine operation in favor of shore-to-ship electricity. A projection methodology was developed and utilized to assess the air quality impacts of increasing net electricity loads to support cold ironing of vessels at all major CA ports in 2055.

- **Pursuing Cold Ironing as a strategy for OGV emissions reductions should consider in tandem potential impacts on the grid and emissions**

Pursuing Cold Ironing as a strategy for OGV emissions reductions could potentially impact the power sector as vessel calls increase. The large demand for OGV calls at CA ports results in significant required power to meet auxiliary needs if engine operation is displaced by local power generation on land. The power demand will also grow in coming decades with expected growth in cargo shipping e.g., meeting all OGV calls in 2055 at major CA ports results in growth in load demand of roughly 0.25%. The level of impact is not negligible and could impact the grid from both an operational and emissions standpoint.

- **The avoidance of auxiliary engine operation of OGV should be mandated at all CA ports in pursuit of regional AQ improvements**

Emissions from auxiliary engine operation represent a significant portion of total OGV emission at CA ports. In terms of total OGV emissions at CA ports, those from auxiliary engine operation represent a significant fraction. This was determined to range from 18 to 45% depending on the specific CA port. Given the significant total OGV emissions, it follows that auxiliary engine operation is responsible for a high level of emissions and important effects on ozone and $PM_{2.5}$. Thus, Cold Ironing does represent an important strategy for improving CA AQ.

- **CA pursuance of cold ironing at ports is justified and will obtain significant AQ benefits to the State**

Cold ironing OGVs represents an opportunity to improve regional AQ in CA. Improvements in ozone and $PM_{2.5}$ occur from completely cold ironing suitable vessels despite potential increases in emissions from the power sector as a result of new generation required to meet vessel needs and exceed 3 ppb and 18 μg/m^3 for most Cases. Further, the locations of greatest impact are associated with large urban populations including the SoCAB, S.F. Bay Area, and San Diego due to the large ports located in those regions. Thus, the State should pursue cold ironing as a potential strategy to meet Federal ambient AQ standards and provide health benefits to residents.

- **In CA cold ironing will not have a detrimental impact in terms of power generation emissions**

Impacts on ozone and $PM_{2.5}$ from potential increased electrical loads from Cold Ironing are minor compared to improvements from auxiliary engine reductions. For all Cases considered reductions from auxiliary engines dominated any worsening from generator emission increases. Indeed, the difference in ozone and $PM_{2.5}$ occurring between no increase and a 10% increase in power emissions is minimal and in the most sensitive areas still represents a reduction overall from the Base Case. With similarity to the conclusions for LDV Cases, the lower emitting nature of the CA electric grid is responsible for the minor impacts and regions with coal power should target strategies to avoid increasing emissions from such generators.

8.2.4 Renewable resources and Electrification

The implementation of higher penetrations of renewable resources into the CA electrical grid can replace emitting generators but may also result in dynamic impacts that increase emissions per unit electricity, e.g., ramping, start/stop. Electrification of combustion technologies in other energy sectors will reduce emissions but may increase power sector emissions from new electrical demand. Further, if loads are increased when the grid is already constrained by high renewable penetration dynamics, then increased emissions consequences may result. The result is the potential for both increases and decreases in emissions and atmospheric pollutant concentrations from electrification and high use of renewable resources in CA.

Various Cases of electrification of energy sectors (residential, commercial, industrial, and transportation) are developed and assessed for perturbations in ground-level ozone and $PM_{2.5}$ in three horizon years (2020, 2030, and 2050)

- **Electrification will likely result in both improvements and worsening in AQ dependent upon multiple factors that vary spatially, temporally and in magnitude**

Transitions to electrification will reduce emissions from current combustion-based technologies in various energy sectors but will likely increase emissions from power generators supporting new electric loads from increased consumption. Thus, the net AQ impacts must be assessed by accounting for all spatial and temporal changes including reductions and increases in atmospheric pollution levels in different places simultaneously.

AQ impacts vary markedly by pollutant, sector, horizon year, season, and location. Contrastingly, increased electricity demand from electrification and altered grid dynamics from intermittent renewable penetration can result in localized worsening of AQ at sites of emitting power generators. Increases generally tend to be point sources while decreases occur from both point and area sources. The difference in characteristics between emission sources results in differing impacts on the spatial distribution of resulting perturbation to ozone and $PM_{2.5}$, i.e., point source impacts are often represented as plumes with higher peak values but over lesser area.

- **Resulting changes to ozone and $PM_{2.5}$ vary depending upon multiple factors that should be considered in regional AQ improvement planning**

Impacts on AQ differ by season as a result of differing generation profiles, demands, and resource availability. Although the major trends remain similar and AQ improvements are more likely than AQ dis-benefits of electrification, different impacts are observed in terms of spatial impacts and to a lesser degree, reduction and increase quantities. It should be noted that different emission profiles for different sectors arise as a result of various factors that impact the results. For example, the residential sector demands are highest in winter due to space heating requirements and thus electrification, emissions, and AQ impacts for those cases are higher in winter than in summer.

Seasonal impacts are also important, e.g. an additional area of ozone increase is observed between the Summer Commercial and Winter Commercial Cases. For Winter ozone cases, the formation dynamics associated with ozone result in an inverse relationship with NO_x emissions, i.e., increases attributed to sites of decreased emissions and vice versa. However, generally ozone is not a concern during this season due to low solar insolation rates which limit photochemical ozone formation. These differences should be considered when considering electrification and renewable resource deployment as strategies to maximize ozone and $PM_{2.5}$ benefits and limit or avoid worsening may not be equivalent from summer to winter. For example, availability of intermittent resources varies from season to season and may be managed differently in terms of temporal integration and balancing strategies at certain times.

- **Further assessment, including exposure and human health based analyses, should be utilized to fully maximize the AQ and GHG benefits of electrification and renewables while avoiding any dis-benefits**

The complexity of emissions perturbations in electrification Cases results in concurrent AQ improvements and AQ worsening in each individual Case. While information can be gained by considering the quantity of pollutant changes and the relative spatial and temporal pattern of the effects, and considering baseline concentrations in the individual locations impacted, determining the overall AQ impacts can be difficult. Thus, further assessment is required to fully assess the AQ impacts and this assessment should consider health-based exposure estimates to better determine the benefits and consequences of electrification.

- **Electrification of the Residential and Commercial sectors could be particularly important for mitigating CA PM levels during winter months**

The electrification of the residential and commercial sector demonstrated minor to significant impacts on ozone and $PM_{2.5}$ depending on pollutant, horizon year, and season. Summer impacts are fairly minor as the majority of demand occurs in winter as a result of space heating. In contrast, winter $PM_{2.5}$ impacts demonstrate major improvements throughout the State – particularly in the northern central area of the State. Additionally, commercial sector cases also achieve significant benefits in $PM_{2.5}$ for summer cases. Thus, electrification of the residential and commercial sector should be considered in tandem with high renewable resource deployment, particularly to address $PM_{2.5}$ concerns.

- **Electrification of the industrial sector is more complex than other sectors and further assessment is needed to identify and assess opportunities while avoiding any dis-benefits**

Electrification of the industrial sector is difficult due to the variation, complexity, and specific nature of industry energy demands, technologies, and processes. As a result of large demands and inefficiencies of replacing technologies the electrification of the industrial sector yields significant increases in criteria pollutant emissions from power generators in support of new loads relative to other sectors. As result, AQ impacts are similarly complex with localized areas of worsening and improvement in both ozone and $PM_{2.5}$. Additionally, in the 2030 and 2050 Cases the demand for electricity from high electrification of industry results in significant deleterious impacts on AQ. It must be considered that electrification in this report pertains to boiler emissions only –industrial process emissions are not reduced.

Thus, the results may underestimate the AQ benefits if electrification can be utilized for processes.

The complexity of the industrial sector in terms of technologies and processes also complicates an understanding of advantageous opportunities to replace combustion sources with electricity. However, as discussed previously, the industrial sector is an important target for future regional AQ mitigation. Therefore, further evaluation of the industrial sector is needed to identify potential opportunities to implement electric technologies and processes in place of combustion-based ones. Additionally, alternative dispatch strategies should be investigated that can reduce the increase in generator emissions to support industrial sector loads.

- **Electrification of the Transportation (LDV) Sector should be considered with priority in terms of AQ improvement relative to other sectors**

Electrification of the LDV transportation sector at high levels results in moderate improvements in ozone and $PM_{2.5}$ that often occur in important regions including SoCAB and the SF Bay Area. Impacts on AQ are also observable from petroleum fuel infrastructure emission reductions in key regions, e.g., Bakersfield, Long Beach. Contrastingly, some worsening occurs from power plant emissions, although impacts are generally in regions of the state with less population density. In all horizon years studied the transportation sector cases achieved benefits relative to other cases and the electrification of LDVs could represent an important strategy to improve AQ in tandem with renewable resource deployment.

- **Electrification of the LDV sector should highly consider co-deployment of smart or controlled charging strategies to maximize AQ benefits and avoid dis-benefits**

The deployment of smart charging achieves a significant AQ benefit relative to immediate charging. It is perhaps most notable that despite an increase in required electricity for vehicles the Smart Charging Case does not experience higher areas of worsening from power plants; instead the emissions decrease notably. This is due to the charging strategy which avoids charging during peak times and during times of renewable power absence resulting in substantial reductions in emissions. Further, reductions in emissions from generators occur due to the avoidance of ramping during peak periods. The reductions in emissions translate to enhanced reductions in ozone and $PM_{2.5}$ in 2030 of about 1 ppb and 1 μg/m^3 and in 2050 of -1 ppb and 5.13 μg/m^3 relative to the immediate vehicle charging cases. Thus, smart charging of electric vehicles can allow for greater vehicle penetrations in tandem with reduced worsening of AQ from power plants and electrification of the LDV sector should include smart or controlled charging strategies to garner AQ benefits and support the utility grid network.

- **Additional advanced complementary strategies should be considered in tandem with electrification to mitigate AQ worsening from power plant emissions in many locations throughout the State**

The deleterious impacts associated with power sector emissions increases may be mitigated by advanced complementary strategies (e.g., advanced energy storage, demand response, vehicle-to-grid) and should be considered for co-deployment. Further, such strategies have a range of additional energy and environmental benefits including enhanced levels of renewable power and support of the utility grid network.

- **Electrifying multiple sectors in tandem enhances the impacts in magnitude, including benefits and worsening**

Combinations of Cases generally result in impacts that are additive relative to individual sector Cases. Generally, combination Cases enhance benefits to ozone and $PM_{2.5}$, although cases including the industrial sector can still encompass deleterious impacts (e.g., Summer ResComIndTra 2030). The results highlight the challenges associated with industrial sector electrification and further demonstrate the worsening that can occur from increased generator emissions. The results suggest that if multiple sectors experience electrification the largest co-benefits are possible; however the importance of complementary strategies to mitigate worsening also grows.

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