The Good, the Bad, and the Ugly: Economic and Environmental Implications of Using Natural Gas to Power On-Road Vehicles in the United States

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Abstract

Currently, in the United States, on-road vehicles are primarily powered by petroleum fuels (gasoline and diesel). These vehicles have caused serious climate change effects from emissions of greenhouse gas (GHG) and health and environmental impacts from criteria air pollutant (CAP). The recent success of shale gas development has brought industry interest in using natural gas to power on-road vehicles. In addition to low costs and wide availability of this national fuel source, natural gas is a common feedstock to produce alternative fuels. The question arises of whether using natural gas for transportation could help or hinder the environment.

In this dissertation, I study the economic and environmental effects of a wide range of natural gas fuel pathways for a selection of light duty (LDV) and medium and heavy duty (MHDV) vehicle types. I choose to focus on two environmental metrics: GHGs and CAPs emitted over the life cycle of each potential pathway for natural gas use. First in Chapters 2 and 3, I use life-cycle analysis to understand the emissions of GHGs from different natural gas pathway for LDVs and MHDVs. Then in Chapter 4 I focus on the CAP emissions from these vehicles.

Overall, I find that none of the natural gas pathways eliminate life cycle air emissions. In fact, only *a few* pathways reduce life cycle GHG emissions and/or life cycle air pollution damages compared to baseline petroleum fuels (gasoline for light-duty vehicles (LDVs) and diesel for heavy-duty vehicles (HDVs)).

For the cases of light duty vehicles (LDVs) and transit buses, battery electric vehicles (BEVs) powered by natural gas-based electricity provide significant reduction in life cycle GHG emissions and life cycle air pollution damages (for almost all counties) compared to the baseline petroleum fuels. However, the actual electricity that charges BEVs may not be natural gas-based electricity in most parts of the U.S. When powered by U.S. grid electricity (using average emission factors for 2010 and 2014), BEVs reduce life cycle GHG emissions to a lesser extent but increase life cycle air pollution damages significantly. Compressed natural gas (CNG), while reducing GHG emissions and CAP emissions (except CO) at tailpipe, are more likely to increase

life cycle GHG emissions and increase life cycle air pollution damages in the majority of U.S. counties.

For heavy-duty trucks, CNG sparking-ignition (SI) trucks and liquefied natural gas (LNG) high-pressure direct ignition (HPDI) trucks have mixed environmental impacts. While they are unlikely to reduce life cycle GHG emissions compared to diesel, they reduce life cycle air pollution damages in 76-99% of U.S. counties for local-haul tractor-trailers and in 32-71% of U.S. counties for long-haul tractor-trailers.

In Chapters 5 and 6, I examine the economic impacts of natural gas fuel pathways for two vehicle types, tractor-trailers and transit buses. I study the economic feasibility of a national natural gas refueling infrastructure for long-haul trucks in U.S., which is a prerequisite for natural gas tractor-trailers. I find that a transition to natural gas fuels in long-haul trucks is more expensive when the shares of natural gas trucks are below 5% because of low refueling demands and over-capacity of the refueling infrastructure to ensure network coverage. At higher shares of natural gas trucks, both the total refueling capacity and the net economic benefits of the national refueling infrastructure increase almost linearly as adoption increases.

Finally, in Chapter 6, I provide an economic-technology assessment for transit buses by considering both life cycle ownership costs and life cycle social costs due to GHG emissions and CAP emissions. Transit buses are early adopters of alternative fuel technologies because of funding supports and operation characteristics (such as high fuel consumption and private refueling infrastructure). I find that the availability of external funding is crucial for transit agencies to adopt any alternative fuel option. Without external funding, only rapid-charging battery electric buses (BEBs) have lower ownership & social costs than conventional diesel buses. When external funding is available to reduce bus purchase costs by 80%, BEBs become much more cost-effective. In this case, life cycle ownership and social costs of BEBs are 37-43% lower than conventional diesel buses. Including life cycle social costs does not change the ranking of alternative fuel options.

The findings in this dissertation suggest different strategies of using natural gas for different vehicle markets. Natural gas is best used in electric power generation than to produce gaseous or liquid fuels for powering on-road LDVs. The use of CNG and LNG for heavy-duty trucks may continue as there are less alternative fuel options but issues such as methane leakage should be addressed to avoid important climate change effect. Finally, natural gas-based transportation fuels can at best partially mitigate climate change or air pollution damages, so other mitigation strategies in the transportation sector are ultimately needed to achieve sustainable transportation.

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List of Abbreviations

ABTRC	Altoona Bus Research & Testing Center
AFV	Alternative Fuel Vehicle
APPEP/AP2	Air Pollution Emission Experiments and Policy model
B100	pure Biodiesel
B20	A blend of biodiesel (20% in volume) and diesel (80% in volume)
BCF/d	Billion Cubic Feet per Day
BEB	Battery Electric Bus
BEV	Battery Electric Vehicle. If a number # follows BEV, it means the all-electric range of the PHEV is # kilometers.
CAFE	Corporate Average Fuel Economy standard (a U.S. federal standard on on-road vehicles)
CAP	Criteria Air Pollutant
CCS	Carbon Capture and Sequestration
CEMS	Continuous Emission Monitoring System
CH_4	Methane
CI	Compression Ignition
CNG	Compressed Natural Gas
CO	Carbon Monoxide
CO_2	Carbon Dioxide
CPI	Consumer Price Index
DGE	Diesel Gallon Equivalent
DME	Dimethyl Ether
DOE	Department of Energy (a U.S. government agency)
DOT	Department of Transportation (a U.S. government agency)
DPF	Diesel Particulate Filter
DPM	Diesel Particular Matter
E85	A blend of ethanol (85% in volume) and gasoline (15% in volume)
EASIUR	Estimating Air pollution Social Impact Using Regression model
EER	Energy Economy Ratio
EIA	Energy Information Administration (a U.S. government agency)
EIO-LCA	Economic input-output LCA

EPA	Environmental Protection Agency (a U.S. government agency)
FAF	Freight Analysis Framework
F-T liquids	Fischer-Tropsch liquids
FCEV	Fuel Cell Electric Vehicle
FFV	Flex-Fuel Vehicle
FHWA	Federal Highway Administration (a U.S. government agency)
FTA	Federal Transit Administration (a U.S. government agency)
gCO_2-eq	Gram of Carbon Dioxide Equivalent
GH_2	Gaseous Hydrogen
GHG	Greenhouse Gas
GIS	Geographic Information System
GREET	Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation model (developed at the Argonne National Laboratory)
GPS	Global Positioning System
GTFS	General Transit Feed Specification
GVWR	Gross Vehicle Weight Rating
GWP	Global Warming Potential
H_2	Hydrogen
HDV	Heavy-Duty Vehicles
HEB	Hybrid Electric Bus
HEV	Hybrid Electric Vehicle
HPDI	High-Pressure Direct Injection
ICEV	Internal Combustion Engine Vehicle
IPCC	Intergovernmental Panel on Climate Change
IRR	Internal Rate of Return
ISO	International Organization for Standardization
lbs.	Pound
LCA	Life Cycle Assessment
LCFS	Low Carbon Fuel Standard
LDV	Light-Duty Vehicle
LH_2	Liquid Hydrogen
LHV	Lower Heating Value
LNG	Liquefied Natural Gas

LPG	Liquefied Petroleum Gas
M85	A blend of methanol (85% in volume) and gasoline (15% in volume)
Mcf	Thousand Cubic Feet
MHDV	Medium- and Heavy-Duty Vehicle
MILP	Mixed-Integer Linear Programming
MJ	Megajoule
MPG	Mile Per Gallon
MPGGE	Mile Per Gallon Gasoline Equivalent
MPGDE	Mile Per Gallon Diesel Equivalent
N_2O	Nitrous Oxide
NEI	National Emissions Inventory (collected and published by U.S. EPA)
NERC	North American Electric Reliability Corporation
NGCC	Natural Gas Combined Cycle power plant
NH_3	Ammonia
NHTS	National Household Travel Survey
NHTSA	National Highway Traffic Safety Administration (a U.S. government agency)
NO_x	Nitrogen Oxides. NO_x is a generic term for the mono-nitrogen oxides NO and NO_2.
NPV	Net Present Value
NRC	National Research Council (part of the U.S. National Academies)
O_3	Ozone
O&M	Operation and Maintenance
PAAC	Port Authority of Allegheny County (in the Commonwealth of Pennsylvania, U.S.)
PHEV	Plug-In Hybrid Electric Vehicle. If a number # follows PHEV, it means the all-electric range of the PHEV is # kilometers.
PM_{10}	Particular Matters less than 10 micrometers in diameter
$PM_{2.5}$	Particular Matters less than 2.5 micrometers in diameter
RNG	Renewable Natural Gas
SCC	Social Cost of Carbon
SCR	Selective Catalytic Reduction
SI	Sparking Ignition
SOC	State of Charge

SO_x	Sulfur Oxides. When x=2, it is sulfur dioxide.
SUV	Sport Utility Vehicle
TEDB	Transportation Energy Data Book
VMT	Vehicle Mile Traveled
VOC	Volatile organic compounds
VSL	Value of Statistical Life

Chapter 1. Introduction and Background

1.1. Motivation

The successful combination of petroleum-based fuels and internal combustion engine vehicles (ICEVs) is one of the greatest successes in the Industrial Revolution. It has greatly increased human's mobility and capability to move goods and materials. It is such a success that the U.S. is called *the country on the wheels*. However, the dominance of petroleum in the transportation sector also takes its toll on society. As a fossil fuel, petroleum, when combusted to power vehicles, contributes to 23.5% of greenhouse gas (GHG) emissions in the U.S.[1], which could lead to severe climate change impacts that last for centuries.[2] On-road vehicle fleets are the largest sources of carbon monoxide (CO) and nitrous oxides (NO_x) emissions and they also emit sulfur dioxide (SO_2), volatile organic compounds (VOCs), and particular matters less than 2.5 um ($PM_{2.5}$), all of which are precursors to ground-level ozone and smog.[3] These criteria air pollutants (CAPs) increase human mortality and morbidity risks (for instance lung and heart diseases), and lead to environmental impacts such as soil and water acidification, reduced tree growth, reduced agricultural yields, and impaired visibility. It is not until recently that quantitative estimates of these health and environmental damages are available. U.S. National Research Council (NRC) estimated that the vehicle sector imposed a health and environment toll of more than $100 billion.[4] The "hidden costs" or negative externalities of petroleum used to power ICEVs are significant.

Alternative fuels (i.e. transportation fuels that are *alternatives* to petroleum fuels, such as compressed natural gas (CNG), liquefied natural gas (LNG), electricity, hydrogen, methanol, ethanol, liquefied petroleum gas (LPG), and, Fischer-Tropsch liquids) and advanced vehicle technologies (i.e. non-ICEV vehicle technologies, such as hybrid electric vehicles (HEVs), plug-in hybrid electric vehicles (PHEVs), battery electric vehicles (BEVs), and fuel cell electric vehicles (FCEVs)) have thus carried high hopes to reduce the reliance of transportation service on petroleum fuels and mitigates the negative impacts associated with petroleum fuels. Compared to petroleum fuel-powered ICEVs, alternative fuels and advanced vehicle technologies can partially or completely (in the case of BEVs and FCEVs) mitigate GHG and CAP emissions that directly emit from vehicle tailpipes. Some alternative fuels can also be produced from renewable sources which have the potential to completely cut away from fossil fuel resources and eliminate all GHG and CAP emissions during the life cycle of fuel use. For instance, electricity can be produced from renewables powers such as wind, solar, geothermal and hydropower, hydrogen can be produced via electrolysis of water with renewable electricity sources, and renewable natural gas (RNG) can be produced using anaerobic digestion or thermochemical gasification from landfills, livestock operations, and wastewater treatment. However, high economic costs, low adoption rates, and limited refueling or charging infrastructure limits the widespread adoption and use of alternative fuels and advanced vehicles.

Thanks to the shale gas revolution, natural gas has become a cost-effective feedstock to produce alternative fuels.[5–8] The past decade has seen a significant increase in U.S. natural gas productions due to the technological success in extracting natural gas from unconventional resources. While in 2005 the United States (U.S.) shale gas production was negligible, by 2012 it reached 25.7 billion cubic feet per day (BCF/d),[9] and today it accounts for 40% of total dry natural gas production in the U.S.[10] The U.S. Energy Information Administration (EIA) forecasts that shale gas production will reach 45.8 BCF/d by 2040.[11] The rapid increase of natural gas supply has led to a large decrease in wellhead prices, which dropped from $7.97 per thousand cubic feet (Mcf) in 2008 to $2.66/Mcf in 2012.[12] More importantly, natural gas is able to produce all alternative fuels efficiently. In fact, natural gas is already the second largest primary fuel used in electricity generation, the largest feedstock used to produce hydrogen and methanol (which are primarily used for industrial purposes), the second largest resources to produce LPG and the dominant feedstock to produce CNG and LNG.

The economic advantages, the U.S. abundance to improve energy security, the plethora of pathways to use natural gas in the transportation sector, and the potentials to reduce negative environmental impacts, all drive industrial interests to use natural gas in the transportation sector. Furthermore, while natural gas is traditionally used significantly in the electricity sector (for electric power generation), industrial sector (for heat generation and for chemical production), and residential & commercial sectors (for space heating, water heating, and cooking), the use of natural gas in the transportation sector is very little and is concentrated in powering natural gas pipelines. Thus, in the eyes of private interests, using natural gas to power on-road vehicles, the primary transportation mode in the U.S., means developing new natural gas markets that not only boost natural gas consumption but also stabilize natural gas prices in the face of strong supply of conventional natural gas and shale gas.

The rapid developments in shale gas revolution and the high industry interests in using natural gas to power on-road vehicles, while intriguing at first sight, imposes important questions on the society: what does using natural gas to power on-road vehicles mean economically and environmentally for the society as a whole? What are the economic and environmental implications of using natural gas to power on-road vehicles?

Indeed, notwithstanding the contributions of existing literature,[4,13–49] a clear gap remains between high industrial interests of tapping natural gas to power on-road vehicles and little understandings of the economic and environmental implications of using natural gas to fuel road transportation. After all, the phenomenal success of shale gas production was completely out of sight as recently as in 2008. The research focus in the sustainable transportation field in the 2000s has largely focused on hydrogen fuel cell electric vehicles, biofuels (especially ethanol), hybrid electric vehicles and plug-in electric vehicles (including PHEVs and BEVs). The most recent literature that performed a comprehensive environmental assessments on natural gas-based transportation fuels dates back to 2000.[15] Since then significant technological changes have happened in the natural gas supply chains as well as in the vehicle technologies, spurred by industrial interests and market forces. In addition, new evidence, such as the issue of methane leakage, has emerged questioning the environmental advantages of natural gas. Existing studies fail to account for all of these important changes and are not able to answer the questions I have asked earlier regarding the economic and environmental implications of using natural gas to power on-road vehicles. Thus, in this dissertation, I propose a study to investigate the economic and environmental implications of using natural gas to power on-road vehicles in the U.S.

1.2. Dissertation Overview and Research Questions

In the previous section (*Motivation*), I have introduced the leading question of this dissertation: (1) what are the economic and environmental implications of using natural gas to power on-road vehicles? The findings on this question will naturally address the following two questions: (2) Can natural gas reduce environmental and economic impacts compared to petroleum fuels? (3) How should we use natural gas to power on-road vehicles (which natural gas pathway is preferred and why)?

To answer these overarching questions, I designed four separate studies (in *Chapters 2-5*), each of which addresses one or multiple society-wide economic and environmental implications for one or multiple vehicle types. Following the introduction (this chapter), the dissertation is divided into five research chapters and a concluding chapter, followed by appendices and references. The main chapters and their research questions are outlined below:

Chapter 2: ***Comparison of Life Cycle Greenhouse Gases from Natural Gas Pathways for Medium and Heavy-Duty Vehicles*****,** and **Chapter 3:** ***Comparison of Life Cycle Greenhouse Gases from Natural Gas Pathways for Light-Duty Vehicles.*** These two chapters together focuses on the life cycle GHG emissions from natural gas pathways and petroleum pathways (for comparison) in a range of representative vehicles (passenger cars, sports utility vehicles (SUVs), pick-up truck/van, parcel delivery van, box truck, transit bus, refuse truck, local-haul tractor-trailer, and long-haul tractor trailer). Key research questions include:

- Which natural gas pathways in which vehicle type provide greenhouse gas emissions reduction compared to petroleum fuel pathways?
- What are key uncertainties? How sensitive are the results to these uncertain variables?

Chapter 4: ***Life Cycle Air Pollution Damages of Petroleum and Natural Gas Pathways for Powering Light-Duty Vehicles and Heavy-Duty Vehicles.*** This chapter focuses on the life cycle air pollution damages due to CAP emissions from natural gas pathways and petroleum pathways (for comparison) in a range of representative vehicles (passenger cars, sports utility vehicles (SUVs), local-haul tractor-trailer, and long-haul tractor trailer). Key research questions include:

- What are the life cycle air pollution damages from petroleum fuel pathways in each of the vehicle type considered?
- What are the life cycle air pollution damages from natural gas fuel pathways in each of the vehicle type considered?
- Which natural gas pathway reduces air pollution damages compared to petroleum fuel pathways in each of the vehicle type considered?
- What are the best and the worst fuel pathway for each vehicle type in each county?
- How sensitive are the results to emissions data and marginal damage model used?

Chapter 5: ***Should We Build A National Infrastructure to Refuel Natural Gas Powered Trucks?*** This chapter focuses on the economic feasibility of a national natural gas refueling infrastructure for long-haul trucks in the U.S. Key research questions include:

- How many refueling stations have to be built to form a national natural gas refueling infrastructure for long-haul trucks in the U.S.?
- What are the economic costs (capital investments) and economic benefits (annualized profits) of such a refueling infrastructure?
- How do the capacities and economic status of the refueling infrastructure correspond to different shares of natural gas trucks in the long-haul truck fleet?
- How sensitive are the capacities and economic status of the refueling infrastructure to different vehicle range assumptions of natural gas truck?
- How sensitive is the economic status of the refueling infrastructure to different economic assumptions?
- Is there a better way to build a national natural gas refueling infrastructure for long-haul trucks in light of the results?

Chapter 6: ***Life Cycle Ownership and Social Costs of Alternative Fuel Options for Transit Buses.*** This chapter estimates the life cycle ownership costs (private costs) and synthesizes social costs associated with air emissions to compare different alternative fuel options for transit buses for a local transit agency in Pittsburgh, PA. Key research questions include:

- Which alternative fuel technology has the lowest costs when only considering life cycle ownership costs and life cycle ownership & social costs?
- Which alternative fuel technology has the lowest life cycle social costs?
- Does the inclusion of life cycle social costs change the rankings of alternative fuel technologies?

- Does government funding play an important role in selecting the best alternative fuel technology?
- How sensitive are the life cycle ownership and social costs sensitive to economic and operation assumptions?
- Do transit buses a major contributor to air pollutions in the hotspot areas in Pittsburgh, PA?
- Which alternative fuel technology provides largest reduction benefits in the hotspot areas in Pittsburgh, PA?
- Are there any factors that the life cycle cost framework cannot account for? How to address these factors?

1.3. Backgrounds

This section provides brief introduction and background information on selected topics that are important in this dissertation. The goal of this section is not to be comprehensive but to be informative. Necessary discussion and important assumptions are available in each individual study (*Chapters 2-6*) and in the corresponding Supporting Information (*Appendices A-E*).

1.3.1. Life Cycle Assessment

This dissertation uses the Life Cycle Assessment (LCA) to quantify the environmental implications of using natural gas to power on-road vehicles. Life cycle analysis (LCA) is a widely used method to assess the environmental effects of a product or service from production to end of life (usually termed ‘cradle to grave’ or ‘cradle to cradle’ for a product, and ‘well to wheel’ for on-road vehicles).[50] Simply put, LCA accounts the direct and indirect environmental impacts to a final product or service. There has been considerable progress in the standardization of LCA methods. In particular, the International Organization for Standardization (ISO) has published two guidelines on the principles and framework for LCA, ISO 14040 and ISO 14044.[51,52] In general, there are four steps to perform a LCA: (1) define the goal and system scope (*goal and scope definition*); (2) build the environmental inventory (*inventory analysis*); (3) assess the environmental impacts based on the inventory (*impact assessment*); and (4) interpret the impacts and ways to reduce them (*interpretation*). LCA requires comparing functionally-equivalent products or services that has the same functional unit.

There are multiple types of LCA methods. One way to distinguish LCAs is based on the system scope especially when evaluating a new or an emerging product or service: if then LCA treats the product or technology system as 'static, context independent, and average',[53] then it is called *attributional* LCA. In contrast, *consequential* LCA is "dynamic, context specie and marginal"[53] in that it considers the marginal changes brought by the new product or service of interest and considers the resulting market responses in its scope.

Another way to differentiate LCAs is based on the modeling strategy. A top-down or an economic input-output LCA (EIO-LCA)[54] uses the input-output data to model the relationships between a particular product or service with all the other economic sectors, whereas a bottom-up or a process-based LCA models the life cycle of a product or a service based on the actual physical processes. The EIO-LCA has the strengths of being comprehensive and modeling general equilibrium but it is usually unable to model product or service details because the input-output data is based on economic sectors. The process LCA, on the other hand, excels in modeling process-specific details but may not include all economic relationships due to time and data constraints.

In this dissertation, I use the *attributional* bottom-up LCA to model environmental implications of using natural gas to power on-road vehicles to provide a consistent and comprehensive framework and to account for important technical details. The limitations of such an approach and future work to address these limitations are available in *Chapter 7*.

1.3.2. On-Road Vehicles

On-road vehicles are not the same; in fact, vehicles have different sizes, weights, performances, and fuel consumptions depending on their functions. Broadly speaking, the function of on-road vehicles is to move people (such as passenger cars, shuttles, and transit buses) or to move goods (such as package delivery trucks and tractor-trailers). Of all the different attributes, vehicle weight (including the payloads of people or goods) is the most important because it affects the vehicular power needed to propel vehicles as well as determines the road infrastructure needed. U.S. federal agencies, such as U.S. Federal Highway Administration (FHWA), and U.S. Environmental Protection Agency (EPA), thus classify on-road vehicles using the weight-based classification method.[55] The weight-based classification method divides on-road vehicles into eight classes based on gross vehicle weight rating (GVWR) where a higher class number (Class #) corresponds to a higher GVGR. A relevant but more general vehicle classification method is to divide on-road vehicles into light-duty vehicles (LDVs), and medium- and heavy-duty vehicles (MHDVs). Depending on the context, MHDVs may also be called heavy-duty vehicles (HDVs). Here LDVs correspond to Class 1 and Class 2a and MHDVs/HDVs include all the remaining classes (Class 2b-Class 8). The classification of on-road vehicles are important to environmental and economic assessment such as this dissertation, because LCA requires comparing fuel pathways for functionally equivalent vehicles for the same functional unit.[56] This dissertation thus follows the weight-based classification method.[55]

1.3.3. Environmental Impacts from Air Emissions

Conventional on-road vehicles that are powered by petroleum fuels emit GHGs (carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)) and CAPs (NO_x, CO, SO_2, VOC, $PM_{2.5}$) over the life cycle of vehicle operation. Emissions of GHGs and CAPs cause serious health and environmental damages on the surrounding environment and exposed population. Specifically, GHG emissions and CAP emissions increase air concentrations of GHGs (eventually CO_2) and CAPs (eventually $PM_{2.5}$ and ground-level ozone) due to physical and chemical processes (accumulation, dispersion and removal process). There are multiple mechanisms linking concentration changes to physical impacts: elevated concentrations of GHGs affect the energy balance of the earth, which could lead to climate change, such as temperature increase, precipitation change, sea level rise, and ocean acidification;[2] and increased levels of $PM_{2.5}$ and ground-level ozone due to CAP emissions impose higher risks of mortality and morbidity on the exposed human population, and contribute to soil and water acidification, reduced tree growth, reduced agricultural yields, and impaired visibility.[30,57]

While GHGs and CAPs are both air emissions, several fundamental differences exist between their impacts. The first difference lies in the lifetime. While $PM_{2.5}$ and ground-level ozone rank only last a few days to a few weeks, the lifetime of GHGs span from a few years and a few decades (most non-CO_2 GHGs) to tens of thousands of years (CO_2). The second difference lies in the impacted regions. Whereas $PM_{2.5}$ and ground-level ozone usually affect the environment within hundreds of miles of its emissions, climate change impacts due to GHG emissions in one county can cause global change that affects everyone on Earth. Finally, the health and environmental impacts of $PM_{2.5}$ and ground-level ozone are immediate (in several days to a few years), whereas climate change takes place in a much longer time frame (in decades to centuries). In short, while GHGs and CAPs all cause real health and environmental impacts on our society and the natural environment, we may have different perceptions and mitigation actions towards these emissions. Whereas CAPs are quick to remove, have immediate impacts in the local region, GHGs are very hard to remove, have long-lasting impacts on Earth.

Luckily, previous research has showed that there are synergies between GHG emission mitigation and CAP emission mitigation because both types of emissions come from the same fossil fuel combustion process. In this dissertation, in addition to quantifying the life cycle GHG emissions and the life cycle CAP emissions from petroleum and natural gas fuel pathways separately, I am also interested in the synergies and differences between GHG emissions and CAP emissions across petroleum and natural gas fuel pathways.

Chapter 2. Comparison of Life Cycle Greenhouse Gases from Natural Gas Pathways for Medium and Heavy-Duty Vehicles

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2.1. Abstract

The low-cost and abundant supply of shale gas in the United States has increased the interest in using natural gas for transportation. This chapter compare the life cycle greenhouse gas (GHG) emissions from different natural gas pathways for medium and heavy-duty vehicles (MHDVs). For Class 8 tractor-trailers and refuse trucks, none of the natural gas pathways provide emissions reductions per unit of freight-distance moved compared to diesel trucks. When compared to the petroleum-based fuels currently used in these vehicles, compressed natural gas (CNG) and centrally-produced liquefied natural gas (LNG) increase emissions by 0-3% and 2-13%, respectively, for Class 8 trucks. Battery electric vehicles (BEVs) powered with natural gas-produced electricity are the only fuel-technology combination that achieves emission reductions for Class 8 transit buses (31% reduction compared to the petroleum-fueled vehicles). For non-Class 8 trucks (pick-up trucks, parcel delivery trucks, and box trucks), BEVs reduce emissions significantly (31-40%) compared to their diesel or gasoline counterparts. CNG and propane achieve relatively smaller emissions reductions (0-6% and 19%, respectively, compared to the petroleum-based fuels), while other natural gas pathways increase emissions for non-Class 8 MHDVs. While using natural gas to fuel electric vehicles could achieve large emission reductions for medium-duty trucks, the results suggest there are no great opportunities to achieve large emission reductions for Class 8 trucks through natural gas pathways with current technologies. There are strategies to reduce the carbon footprint of using natural gas for MHDVs, ranging from increasing vehicle fuel efficiency, reducing life cycle methane leakage rate, to achieving the same payloads and cargo volumes as conventional diesel trucks.

2.2. Introduction

In recent years, the successful combination of technologies such as hydraulic fracturing, horizontal drilling, and seismic mapping have led to significant production of unconventional natural gas resources, which in turn has attracted industrial interests in using natural gas as a transportation fuel.[5–8,58–71] While economic considerations have dominated this discussion, environmental impacts of natural gas-based fuels are likely to be of interest to multiple stakeholders.[37,70,72] A recent National Research Council (NRC) report[70] analyzed the impacts of natural gas to fuel medium- and heavy-duty vehicles (MHDVs) and concluded that "more studies and data are needed to determine the well-to-tank GHG emissions of NG vehicles."

There are several approaches to evaluate the GHG emissions of MHDVs. Both vehicle simulation[34,73–80] and vehicle tests[33,35,36,38,81–89] provide estimates of emissions from the use phase. These tests are limited in that they fail to account for emission sources beyond tailpipe. Thus, vehicle simulations and tests may not be appropriate for making generalized recommendations regarding GHG emissions. Life cycle assessment (LCA) studies[27,28,38–42,44–46,48,61,90–96] overcome this shortcoming as they account for several phases of the vehicle life and can include data from vehicle simulations and tests. However, existing studies were generally focused on specific fuel pathways for some MHDVs, especially compressed natural gas (CNG) and liquefied natural gas (LNG) for transit buses and heavy-duty trucks), relied on outdated data, ignored payload differences, and presented contradictory conclusions. Outdated data about methane emissions from the natural gas sector is particularly concerning as there are several recent field studies[97–102] that performed on-site measurements to estimate methane leakage rates. Similarly, natural gas vehicle technologies have undergone recent improvements in fuel efficiency,[92] which previous studies could not account for. Finally, the contradictory results from recent studies suggest further analysis is needed: TIAX[95] found more than 20% reductions for CNG and LNG trucks compared to diesel; Meyer et al.[46] found a 5% reduction for CNG trucks; Santini et al.[92] found an 8% reduction for LNG trucks and a 3% reduction for CNG trucks; Volvo[96] found a 2-30% increase for lean-burn CNG trucks. For transit buses, conclusions from the same consulting agency were contradictory when one study[48] reported CNG buses emit slightly less than diesel buses while the other[38] reported the opposite. Clark et al.[42] found CNG and ultra-sulfur diesel were comparable for year 2007, but a more recent load-based life cycle GHG emission calculator[93] found a 14% reduction for CNG buses.

A fundamental characteristic of the MHDV market is that MHDV fleets are extremely heterogeneous and their environmental performance is highly dependent on the use patterns (such as truck configurations, payloads, drive cycles, etc.).[45,55,70,103,104] The complexity of modeling the MHDV market has posed serious barriers to understanding the magnitude of life cycle GHG emissions attributed to MHDVs. Existing studies generally differed from each other by considering different MHDV segments, or by using different vehicle configurations, payloads, or drive cycles. Moreover, MHDVs have only recently been added to the Corporate Average Fuel Efficiency (CAFE) standards,[105] so researchers and policymakers are still learning how to characterize MHDVs emissions and how to assess the consequences of different fuels and technologies in this market. Unlike light-duty vehicles (LDVs), where authoritative sources (such as fueleconomy.gov) provide comparable fuel economy estimates for the same drive cycle and test specifications, test-based fuel economy estimates of MHDVs are limited. Furthermore, a non-negligible portion of existing studies neglected methane emissions from natural gas trucks (in the form of incomplete combustion and direct leaks from MHDVs),[35] though some recent work attempts to bridge this gap.[106]

This chapter aims to fill a specific knowledge gap in terms of GHG emissions estimates from MHDVs. More specifically, I evaluate the relative comparison of different ways of using natural gas for different types of MHDVs. To achieve this goal, I perform a LCA on a comprehensive set of natural gas-derived fuels, engine technologies, and vehicle types. The contribution of this chapter is not methodological; instead it addresses an important gap in current policy discussions such as the Low Carbon Fuel Standard (LCFS)[107] in California, U.S. and the CAFE standards set to reduce fuel consumption and GHG emissions of MHDVs in the U.S.[108] While the CAFE standards for MHDV only consider use phase emissions, it is of key relevance to identify whether the best strategies in terms of emissions reductions still hold when one accounts for the full life cycle emissions in order to avoid unintended negative consequences that may be derived from a use-phase-only policy design – as it becomes apparent in the Results section. I follow a bottom-up attributional LCA approach with latest available data and a consistent system analysis boundary. I perform detailed reviews regarding the assumptions related to natural gas production, fuel production, fuel delivery, and vehicle specifications, to ensure consistency and transparency throughout the analysis. I include a Monte-Carlo analysis to explicitly account for the variability and uncertainty in emissions along the life cycle of natural gas pathways. In addition, I estimate the break-even life cycle methane leakage rates for CNG and LNG pathways that would make them net emissions reducers or net emission contributors to understand the relative importance of methane leakage and vehicle fuel efficiency.

2.3. Methods

2.3.1. System Boundary

I define a *pathway* as a way of using natural gas for road transportation. **Figure 2.1** illustrates the different pathways considered. I assume that natural gas used to produce these transportation fuels is derived from shale gas resources in the U.S., as shale gas is expected to account for the majority of natural gas produced in coming decades.[109] The baseline fuel pathways are conventional gasoline (Class 2b) and conventional (ultra-low-sulfur) diesel (Class 3-8 MHDVs). The geographic scope of the study is the contiguous U.S.

2.3.2. Emissions Inventory

The LCA boundary starts at natural gas extraction and ends with the use of the natural-gas-derived fuel during vehicle operation. In general, there are four stages in the life cycle of any fuel pathway: feedstock (natural gas) production and transport, transportation fuel production, transportation fuel delivery, and vehicle use. In pathways that rely on distributed transportation fuel production, the natural gas transport stage of the life cycle includes both interstate and distribution pipelines. Pathways that include centralized production of the transportation fuel only account for GHG emissions from the interstate pipeline network, where natural gas is assumed to be drawn directly from. Emissions related to manufacturing of batteries and fuel cells for electric vehicles are included, while emissions with manufacturing of other vehicle components are assumed to be similar among pathways. Emissions from building the infrastructure needed to deploy different fuels and vehicle end-of-life are outside of the scope of this study. Existing studies found that emissions associated with infrastructure construction and decommissioning contribute to less than 1% of the life cycle emissions for electricity and hydrogen production,[110,111] and I anticipate that the values for the natural gas infrastructure would be in the same ballpark.

This chapter focuses on estimating emissions of three GHGs, CO_2, methane (CH_4), and N_2O, which are converted to CO_2-equivalent emissions using the probabilistic distribution for the latest global warming potential (GWP) values. I build a bottom-up model in accounting for all emissions defined in this system boundary. Details of the LCA model and discussions on the quality of the data sources can be found in Appendix A.

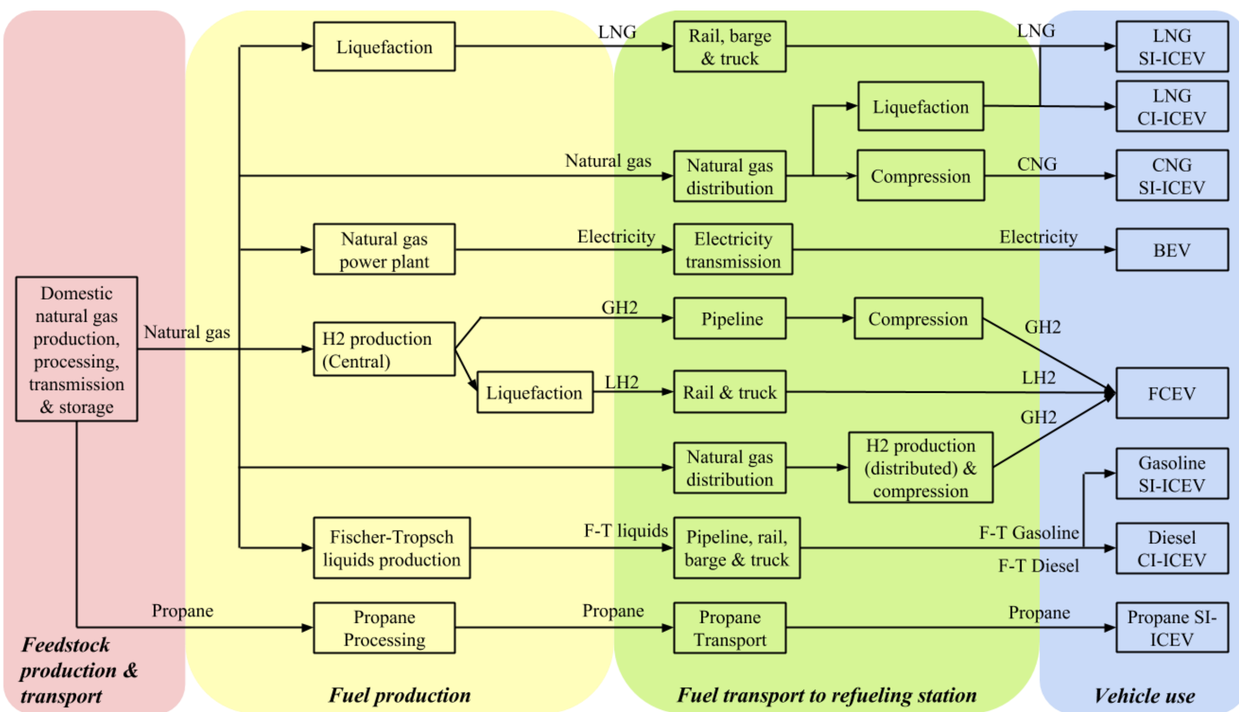


Figure 2.1. Study system boundary of natural gas pathways. Different colored areas correspond to different life cycle stages: natural gas upstream (pink), fuel production (yellow), fuel transport (green), and vehicle operation (blue) (indicated by engine technologies). Both feedstock and energy carriers are marked along each pathway. LNG = liquefied natural gas; CNG = compressed natural gas; H2 = hydrogen; GH2 = gaseous hydrogen; LH2 = liquid hydrogen; F-T = Fischer-Tropsch; LPG = liquefied petroleum gas, or propane; ICEV = internal

combustion engine vehicle; SI = sparking ignition; CI = compression ignition; BEV = battery electric vehicle; FCEV = fuel cell electric vehicle.

2.3.3. Vehicle Specifications

Given that natural gas-powered MHDVs are still emerging, this chapter models new vehicles available in the market rather than existing vehicles. I use functionally equivalent vehicles for different fuel pathways within a specific vehicle segment.[72] I follow the weight-based classification method for on-road vehicles,[55] which is used by industry and U.S. federal agencies (e.g. U.S. Federal Highway Administration (FHWA), and U.S. Environmental Protection Agency (EPA)). I consider seven types of MHDVs:[55,112] Class 2b pick-up truck, Class 4 parcel delivery truck, Class 6 box truck (such as beverage delivery truck), Class 8 transit bus, Class 8 local-haul tractor-trailer, Class 8 long-haul tractor-trailer, and Class 8 refuse truck. Finally, I include five vehicle engine technologies: sparking ignition internal combustion engine vehicle (SI-ICEV), compression ignition internal combustion engine vehicle (CI-ICEV), hybrid electric vehicle (HEV), battery electric vehicle (BEV), and fuel cell electric vehicle (FCEV). The interaction between fuel pathways, vehicle engine technologies and types of MHDVs are shown in **Table 2.1**, along with key parameters, such as vehicle fuel efficiencies and vehicle payloads. The Appendix A includes a detailed discussion of vehicle-side assumptions (such as fuel economy, payload, lifetime, battery and fuel cell sizes, and tailpipe methane and N_2O emissions).

2.3.4. Functional Unit

I use two functional units: vehicle distance traveled (gCO_2-eq/km), and freight-distance moved (gCO_2-eq/km-metric-ton). The first functional unit is simple but fails to reflect the functionality of MHDVs. While heavier trucks have lower fuel economy than their lighter substitutes, they are more efficient in moving the same weight of load, thus getting lower load-normalized fuel economy (gallons per cargo-ton-mile) than lighter vehicles.[47,55] I thus include the second functional unit to address this issue, at the expense of adding an additional set of assumptions (payloads of MHDVs).

Table 2.1. Vehicle specifications for different fuel pathways and different vehicle applications.

		Class 2b Pick-up truck/van	Class 4 Parcel delivery van	Class 6 Box truck	Class 8 Transit bus	Class 8 Refuse truck	Class 8 Tractor trailer local-haul	Class 8 Tractor trailer line-haul
								
Unit of fuel economy+		MPG/ L/100km	MPG / L/100km (diesel gallon/liter equivalent)					
Gasoline (SI-ICEV)		14.0/16.8*	-	-	-	-	-	-
Diesel (CI-ICEV)		16.1/14.6	11.5/20.5*	7.0/33.6*	4.0/58.8*	3.3/71.3*	4.3/54.7*	6.5/36.2*
Gasoline-HEV(SI-ICEV)		16.8/14.0	10.9/21.5	-	-	-	-	-
Diesel-HEV (CI-ICEV)		19.3/12.2	14.4/16.4	9.3/25.3	4.8/49.0	3.6/64.8**	5.2/45.6	7.2/32.9
CNG (SI-ICEV)		14.0/16.8	10.8/21.8	6.6/35.7	3.6/65.3	2.9/81.0	3.9/60.8	5.9/40.2
LNG (SI-ICEV)		-	-	-	3.6/65.3	2.9/81.0	3.9/60.8	5.9/40.2
LNG (CI-ICEV)		-	-	-	-	-	4.2/55.8	6.4/36.9
Propane (SI-ICEV)		14.0/16.8	-	-	-	-	-	-
BEV		42.0/5.6	34.5/6.8	21.0/11.2	16.8/14.0	-	-	-
H_2-FCEV		-	-	-	7.6/30.9	-	-	-
Conventional	Gross weight (lbs.)	8,501-10,000	16,000	19,501-26,000	39,980	60,000	80,000	80,000
	Empty weight (lbs.)	5,000-6,300	9,700	11,500-14,500	27,730	16,627	30,500	35,550
	Payload (lbs.)	3,700	6,300	11,500	12,150	43,373	49,500	44,450
Alternative fuels	Weight penalty for payloads (lbs.)	Gasoline/Diesel HEV: 350 CNG: 200 BEV: 600	Gasoline/Diesel HEV: 0 CNG: 0 BEV: 200	HEV: 1200 CNG: 515 BEV: 200	Gasoline/Diesel HEV: 0/750 CNG: 900 LNG: 1150 H_2-FCEV: 5,400 BEV: 4,800	Hybrid: 400 CNG: 915 LNG (SI): 265	HEV: 880 CNG: 502 LNG (SI): 252 LNG (CI): 1249	HEV: 880 CNG: 2042 LNG (SI): 1142 LNG (CI): 2541
	Volume penalty	Reduced cargo space	No difference (except for transit buses) because fuel tanks are mounted on the chassis, behind the cabin, or atop the vehicle. For transit buses, volume penalty has been factored into weight penalty.					

*+ Different vehicle segments have different baseline petroleum fuels (gasoline for Class 2b and diesel for Class 3-8) so the same 'gallon' has a different meaning in different vehicle segments. * The baseline petroleum fuel pathway is marked and highlighted in gray. ** A diesel refuse truck with hydraulic hybrid system is assumed. # Details on how to determine weight and volume penalties in payloads of alternative fuel pathways are discussed in the Appendix A. The vehicle cartoon figures come from NREL (2013).[113] Acronyms. CNG = compressed natural gas. LNG = liquefied natural gas. SI-ICEV = sparking ignition internal combustion engine vehicle. CI-ICEV = compression ignition internal combustion engine vehicle. HEV = hybrid electric vehicle. BEV = battery electric vehicle. H_2-FCEV = hydrogen fuel cell electric vehicle. MPG = mile per gallon.*

2.4. Results

2.4.1. Life Cycle Emissions for Fuel Pathways

Methane emissions have been shown to play an important role in the life cycle emissions of natural gas but the methane leakage rate in the U.S. natural gas systems remains a subject of debate. In particular, there is a wide gap between bottom-up studies (including this study) and top-down studies.[114] To account for a potential bias in methane leakage rate estimates, and also to account for choices to use GWPs with different time frames, I consider four scenarios: (1) a baseline methane estimate with 100-year GWP (baseline scenario), (2) a baseline methane estimate with 20-year GWP, (3) a pessimistic estimate with 100-year GWP, and (4) a pessimistic estimate with 20-year GWP.

For the baseline estimate, the mean estimate of natural gas upstream GHG emissions is 17.2 gCO_2-eq/MJ_{LHV}, with a 95% confidence interval of 10.2-29.3 gCO_2-eq/MJ_{LHV}. This estimate uses 100-year GWPs, and implies a methane leakage rate of 1.0-2.2% for a 95% confidence interval. The distribution of natural gas upstream emissions is right-skewed, which is likely the results of superemitters.[114] The baseline mean estimate falls within the range of other recent bottom-up estimates.[115] However, to account for the differences between bottom-up and top-down estimates,[114,116] I multiply the baseline methane emission estimate by 1.5. The Appendix A includes a detailed description of the data and assumptions used to develop the life cycle inventory.

Figure 2.2 shows the life cycle GHG emissions (also called “carbon intensity”) of the natural gas-based fuels that can be used in MHDVs. It should be noted that this figure is not meant to be used for a fuel comparison, as the carbon intensity is not functionally equivalent or comparable unless the efficiency of end-use technologies is considered. Thus, this figure is only meant to summarize the range of estimates for each pathway.

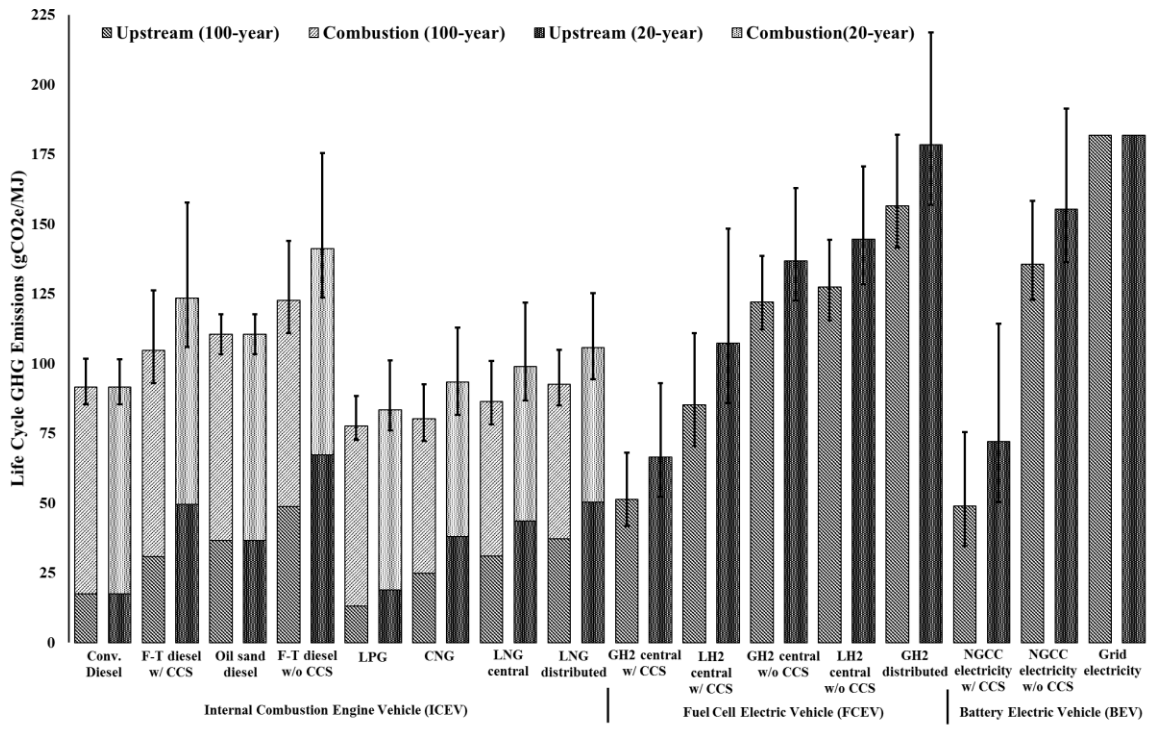


Figure 2.2. Life cycle GHG emissions from natural gas-derived fuels and existing liquids with 100-year GWP (left bar in each pair) and 20-year GWP (right bar in each pair). The functional unit is 1 MJ (lower heating value) of fuel delivered to end use. Upstream emissions include all emission sources until the fuel is dispensed into the vehicle. Combustion emissions are estimated based on fuel characteristics, as described in the Appendix A. Error bars represents the 95% confidence interval of the life cycle GHG emissions.

2.4.2. Life Cycle Emissions for Fuel/Vehicle Pathways

I report results of the life cycle GHG emissions of natural gas-based transportation fuels for MHDVs in **Figure 2.3**. These results are based on 100-year GWPs, the baseline estimate of natural gas upstream emissions, and are presented in the functional unit considering payloads of MHDVs (gCO_2-eq/km-cargo-metric-ton). In the Appendix A, I present bar plots and cumulative distribution plots for results in other scenarios (with both functional units).

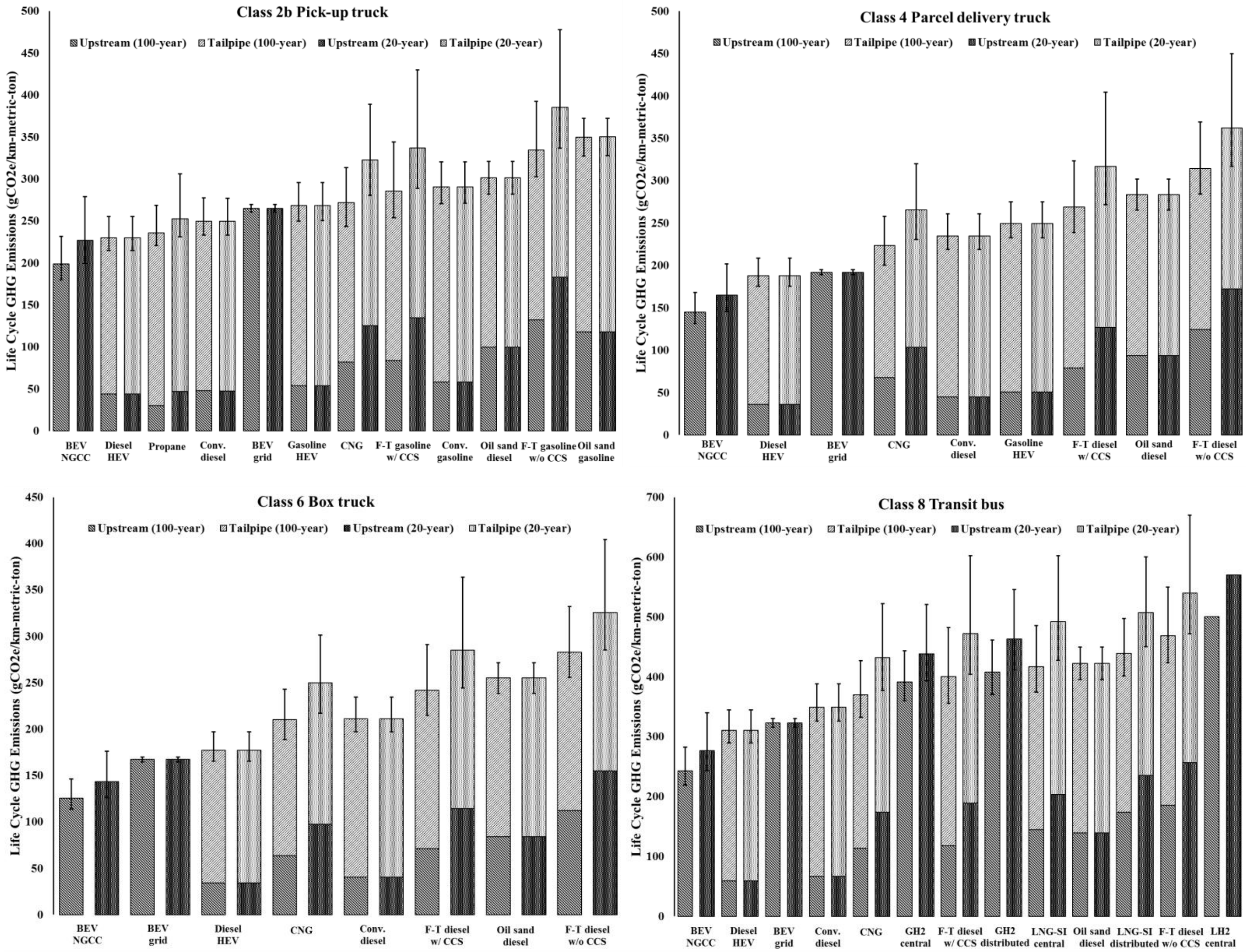
Class 2b, Class 4, and Class 6 vehicles are ‘medium-duty trucks’ with similar life cycle GHG emissions rankings among fuel pathways. As shown in **Figure 2.3**, BEVs with natural gas-based electricity achieve the largest (31-40%) mean emission reductions compared with the baseline petroleum fuels (gasoline for Class 2b, and diesel for Class 4 & 6). CNG trucks achieve 0-6% mean emission reductions for these three MHDV segments, and are better than the baseline petroleum pathway for over 80% of time (except Class 6). In the LCA model, propane is only available for Class 2b pick-up trucks, for which it achieves the largest emission reductions (19% on average) among natural gas pathways with ICEV technologies. However, the supply of propane may be regional (where wet natural gas is abundant) and could be limited due to competitions from other demands (such as residential heating).[117]

For Class 8 transit buses, BEVs with natural gas electricity emit the lowest life cycle GHG emissions, achieving 31% reductions compared to diesel bus. Thanks to the large fuel efficiency benefits (3.2 times better than diesel), BEVs powered with U.S. current grid electricity still achieve 8% emission reductions. Other natural gas pathways that are available for transit buses, such as CNG, hydrogen FCEVs, LNG, and F-T liquids, increase GHG emissions by 6-43% on average compared to conventional diesel. For Class 8 trucks, CNG emits lowest among natural gas pathways but it cannot reduce emissions (0-3% higher for three types of Class 8 trucks) on average compared with conventional diesel. LNG and F-T liquids increase GHG emissions by 2-34% for Class 8 trucks when compared to the baseline.

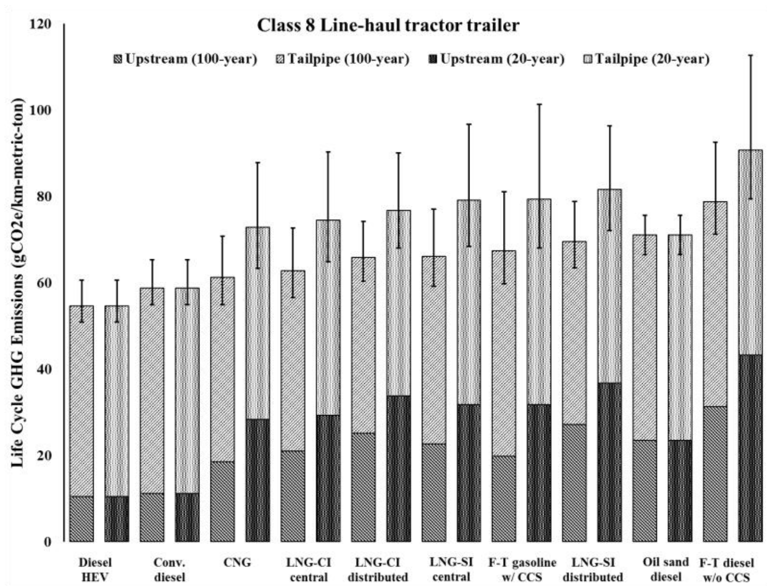
The distributions of life cycle emissions from natural gas pathways are found to be wider than those from petroleum pathways and exhibit highly asymmetrical shapes skewed to the right.

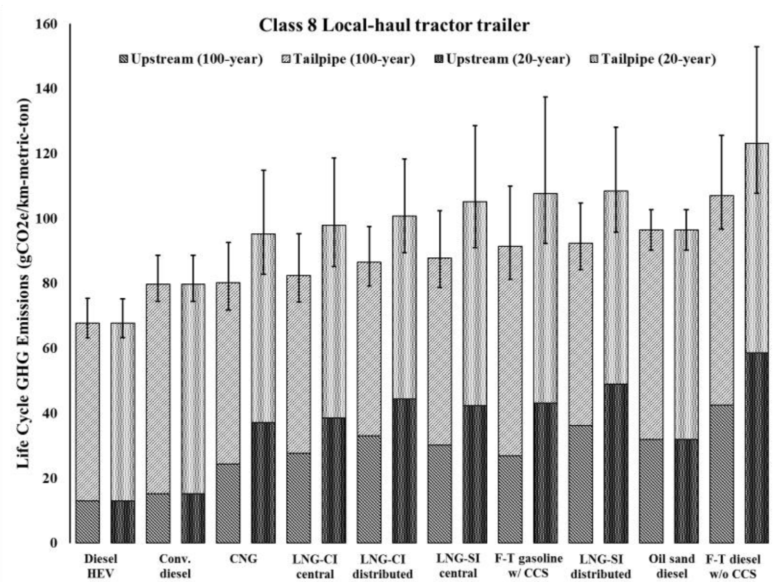
Thus, when calculating relative emission changes compared to petroleum fuels, the resulting distributions are also skewed to the right. An important factor in determining the relative benefits of natural gas pathways is the choice of baseline fuel and vehicle technology. While conventional gasoline and diesel used in ICEs still appear to be appropriate baselines for MHDVs, I also include hybrid technologies (7-21% less emissions than baseline) and crude oils derived from Canadian oil sands (21% more emissions than baseline).

Moving payloads or passengers is the primary goal of MHDVs, and the differences in payloads from different pathways appear to be important. I find that all natural gas fuel pathways incur payload penalties for all MHDVs (**Table 2.1**) and the issue of payload loss is more severe for pick-up trucks and transit buses than for other MHDVs. For pick-up trucks, any changes in the payload are relatively large since the bassline payload is small. For transit buses, alternative fuel buses see large drops (40-45%) in the maximum number of bus riders (determined from vehicle tests) compared to diesel buses.



Class 2b Pick-up truck
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km-metric-ton)
BEV NGCC
Diesel HEV
Propane
Conv. diesel
BEV grid
Gasoline HEV
CNG
F-T gasoline w/ CCS
Conv. gasoline
Oil sand diesel
F-T gasoline w/o CCS
Oil sand gasoline
Class 4 Parcel delivery truck
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km-metric-ton)
BEV NGCC
Diesel HEV
BEV grid
CNG
Conv. diesel
Gasoline HEV
F-T diesel w/ CCS
Oil sand diesel
F-T diesel w/o CCS
Class 6 Box truck
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km-metric-ton)
BEV NGCC
BEV grid
Diesel HEV
CNG
Conv. diesel
F-T diesel w/ CCS
Oil sand diesel
F-T diesel w/o CCS
Class 8 Transit bus
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km-metric-ton)
BEV NGCC
Diesel HEV
BEV grid
Conv. diesel
CNG
GH2 central
F-T diesel w/ CCS
GH2 distributed
LNG-SI central
Oil sand diesel
LNG-SI distributed
F-T diesel w/o CCS
LH2 central





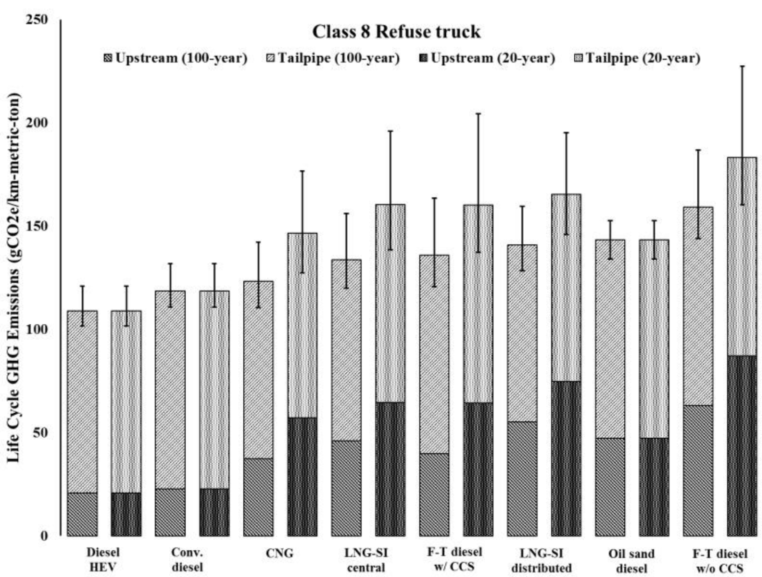


Figure 2.3. Life Cycle GHG emissions of MHDVs (unit: gCO_2-eq/km-metric-ton), with baseline methane emission estimate. In each figure, results with 100-year GWP (left bar in each pair) and 20-year GWP (right bar in each pair) are shown side by side. Error bars are based on the 95% confidence interval of life cycle GHG emissions.

The implications of payload differences depend on the actual operations of MHDVs. For instance, transit buses might only operate at full loads in certain time periods and along specific routes, in which case the functional unit that includes the payload is most appropriate. At other times, especially in non-rush hours, all transit buses should be able to operate functionally the same, in which case the maximum payload is not the limiting factor and the other functional unit (gCO_2-eq/km) is more representative. In the bus example, when payload differences do not affect service levels (such as in non-rush hours), hydrogen FCEVs using gaseous hydrogen could achieve an emission reduction of more than 35%, hydrogen FCEVs using liquid hydrogen could reduce emissions by 20%, and CNG could reduce emissions by 2% compared with diesel buses for mean estimates.

As for trucks, highway statistics[118] show that not all on-road Class 8 trucks reach the federal weight limits (i.e. carrying full payloads). For trucks that are limited by the cargo space rather than cargo weight,[63] considering payload differences may result in biased results. Moreover, the consideration of payload differences not only determines which functional unit is better but may also change the operation schedules of MHDVs (for instance, less payloads mean more trips) and thus affect total GHG emissions from freight movement. The attributional LCA framework does not account for these system responses. As a result, this chapter is limited to reporting the results for both functional units.

In addition to payload, the choice of GWPs and methane emission estimates are other important factors for absolute emission levels and relative rankings of natural gas fuel pathways. Using 20-year GWPs instead of 100-year GWPs increases life cycle GHG emissions by 7-21% for natural gas pathways. While the pessimistic estimates of methane leakage from the natural gas system increase baseline methane emission estimates by 50%, this effect is attenuated to only 5-7% for the life cycle emissions since the majority of GHG emissions are emitted during vehicle operations. While more studies are needed to improve the understandings of battery and fuel cell manufacturing emissions and tailpipe methane emissions, both emission sources are small (1-4%) for BEVs as well as CNG and LNG pathways across all possible MHDVs.

While carbon capture and sequestration (CCS) technologies are not mature, they may be available in the future for some of the fuel pathways in this analysis. I include CCS technologies for natural gas electricity generation, central hydrogen production, and F-T liquids production. When comparing life cycle GHG emissions for pathways with CCS and without CCS, there are significant reductions for electricity generation (64% for mean estimate) and hydrogen production (46% for liquid hydrogen, and 58% for gaseous hydrogen), but much smaller reductions for F-T liquids (**Figure 2.2**). As a result, F-T liquids even with CCS technologies still increase emissions compared to conventional diesel (**Figure 2.3**). On the other hand, if CCS technologies are available, BEVs and gaseous hydrogen FCEVs could reduce emissions by 67% and 53%, compared to petroleum-based systems, for transit buses considering the payload differences.

2.4.3. Break-Even Life Cycle Methane Leakage Rates

One of the key uncertainties that drives natural gas pathways to be net emissions reducers or not is the assumed methane leakage. The previous analyses presented the LCA results across fuels, vehicle engine technologies, and MHDVs, but important insights on the trade-off between vehicle fuel efficiency and methane leakage rate may have been buried behind the scene. Here I present a break-even analysis on methane leakage rates for two pathways, CNG and distributed LNG, as these fuels seems to currently be the focus on intense interest.[5,60,64,65,70,92] Break-even methane leakage rate is defined as the mole or volume percentage of all dry natural gas produced that is lost through fugitive emissions at which the life cycle GHG emissions of the natural gas-based transportation fuels are comparable to the life cycle GHG emissions of incumbent petroleum fuels. I find that a linear relationship exists between break-even methane leakage rate and the relative fuel efficiency of the vehicles (Appendix A includes mathematical derivations). As shown in **Figure 2.4**, distributed LNG allows for a smaller break-even methane leakage rate than the CNG pathway for the same relative vehicle fuel efficiency. I find that there are lower bounds on the relative vehicle fuel efficiency for CNG and LNG pathways (77.5% and 91%, respectively) below which carbon dioxide emissions in the life cycle would already make CNG and LNG pathways worse than incumbent petroleum fuels. If the carbon intensity of the baseline petroleum pathway changes, then break-even methane leakage rates will shift accordingly.

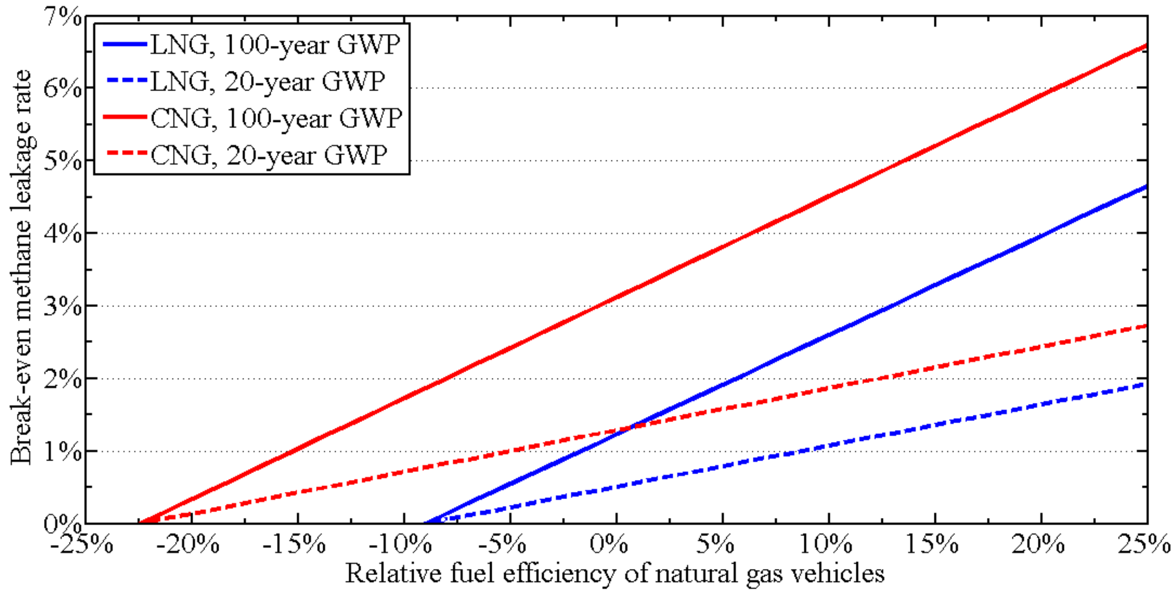


Figure 2.4. Relationship between break-even life cycle methane leakage rate and relative fuel efficiency of natural gas vehicles. The break-even life cycle methane leakage rate identifies the methane leakage rate for the life cycle at which the natural gas fuel would have the same life cycle emissions as the incumbent petroleum fuel, which in this case is conventional gasoline or conventional diesel. Further, the leakage rate is calculated as the volumetric percentage of natural gas produced that is lost through venting or fugitive leaks in the life cycle. Finally, the relative fuel efficiency is represented as the percentage difference between the efficiency of the petroleum-based vehicle and the natural gas-fueled vehicle. A negative relative fuel efficiency means the efficiency of the natural gas vehicle is lower than the efficiency of the petroleum-based vehicle.

2.5. Conclusions and Policy Recommendations

This chapter systematically analyzes the life cycle GHG emissions of natural gas pathways for a comprehensive combination of natural gas-derived fuel, engine technologies, and vehicle types of MHDVs using a bottom-up LCA approach. The contribution of this chapter is not methodological. Instead, the paper addresses an important gap in current policy discussions such as the Low Carbon Fuel Standard (LCFS) in California and the CAFE standards of MHDVs in the U.S. To understand the sensitivity of the emission reductions, I calculate break-even methane leakage rates of the CNG and LNG pathways as a function of the relative fuel efficiency of natural gas vehicles (compared to baseline petroleum fuels).

The emissions reduction potentials of natural gas pathways vary sharply between non-Class 8 MHDVs (e.g. pick-up trucks, parcel delivery trucks, and box trucks), Class 8 transit buses and Class 8 MHDVs (e.g. refuse trucks and tractor-trailers). BEVs, LPG and CNG pathways could reduce life cycle GHG emissions for non-Class 8 MHDVs compared to the baseline petroleum fuels. Similarly, BEVs achieve emission reductions for transit buses. On the other hand, none of natural gas pathways - CNG, LNG, and F-T liquids - achieve any emission reductions for Class 8 trucks compared to conventional diesel.

Choice of natural gas pathway, relative fuel efficiency of natural gas vehicles (relative to petroleum counterparts), and life cycle methane leakage rate are important factors determining rankings of natural gas pathways. Payload losses in natural gas-fueled MHDVs compared to conventional MHDVs are also an important consideration. For instance, transit buses with alternative fuels see large drops in payloads (measured by the maximum numbers of bus riders). Excluding these payload differences in the comparison may incorrectly result in larger emission reduction potentials than could actually be achieved. While the payload losses considered might only occur in certain conditions, the results still highlight the importance of considering payload differences when assessing emissions of MHDVs. Furthermore, choices of baseline petroleum fuels and global warming metrics play important roles in determining emission reduction potentials of natural gas pathways for MHDVs.

The results could be important inputs to current policy debates such as the LCFS[107] in California, the CAFE standards of MHDVs,[108] as well as methane regulations in the U.S. In addition to the exact emission estimates, large uncertainties shown with natural gas pathways should be considered and discussed in the LCFS-type regulations. In terms of methane regulations, more transparent reporting requirements (such as U.S. EPA's GHGRP program[98]) and more on-site measurements on natural gas systems and natural gas vehicles (such as EDF's efforts[106]) are crucial to solve the ongoing debates regarding methane leakage and to identify emission reduction opportunities which can then be implemented via cost-effective technologies or stringent regulations.[119–123]

There are several limitations to this study. The analysis focuses on GHG emissions and I use the global warming potential of non-CO_2 gases. Recent literature suggests that GWP has serious limitations. For instance, GWP treats all emissions as if they are pulse emissions at the beginning of the time horizon considered, thus completely ignoring different effects of emissions happening at different time.[28,124–127] Further, while GWP is closely related to radiative forcing, GWP does not consider other drivers of climate change, such as the rate of change, and variations in surface temperature response.[128] Some research is ongoing to develop more appropriate climate impact metrics,[124–127] but there is no consensus about the use of these metrics for LCA and a comparison of such metrics is beyond the scope of this study. In the future, as more appropriate metrics are identified, I can use the inventory results in this chapter to re-evaluate the climate impacts of natural gas-based transportation fuels.

This analysis is also limited by my inability to consider real-world conditions in actual operations of MHDVs, especially the drive cycles (e.g., speed, idling, road grade)[38,93,104] and payload profiles,[45,93] as such information is limited. As more vehicle tests and innovative methods to factor duty cycles into the assessment of vehicle emissions become available, further analysis could refine my estimates of the life cycle GHG emission of natural gas-based transportation fuels.

Finally, while this chapter focuses on GHG emissions, there are other environmental benefits from using natural gas for road transportation, such as health benefits from reduced air pollutants and lower operating noises,[33,44,46,95,129] which could be significant. There are also other types of MHDVs beyond those included in this chapter; for instance, I do not include school buses, port drayage trucks, and all off-highway MHDVs. I also exclude dual-fuel pathways (such as CNG and diesel, and plug-in hybrid electric vehicles) because of limited data, though these vehicles may serve as near-term options to meet the long-term goals of oil independence and emission reductions. While this chapter is the most up-to-date and comprehensive analysis of the potential environmental benefits of natural gas-based transportation fuels for the MHDV fleet, future analysis should be performed as data becomes available and analytical methods improve.

Chapter 3. Comparison of Life Cycle Greenhouse Gases from Natural Gas Pathways for Light-Duty Vehicles

This chapter is based on the published work: *Tong, F.; Jaramillo, P.; Azevedo, I. Comparison of Life Cycle Greenhouse Gases from Natural Gas Pathways for Light Duty Vehicles. Energy & Fuels* ***2015****, 29, 6008–6018. http://pubs.acs.org/doi/abs/10.1021/acs.energyfuels.5b01063.*

3.1. Abstract

Low prices and abundant resources open new opportunities for using natural gas, one of which is the production of transportation fuels. In this chapter, I use a Monte Carlo analysis combined with a life cycle analysis framework to assess the greenhouse gas (GHG) implications of a transition to natural gas-powered vehicles. I consider six different natural gas fuel pathways in two representative light-duty vehicles: a passenger vehicle and a sport utility vehicle. A battery electric vehicle (BEV) powered with natural gas-based electricity achieves around 40% life cycle emissions reductions when compared to conventional gasoline. Gaseous hydrogen fuel cell electric vehicles (FCEVs) and CNG vehicles have comparable life cycle emissions with conventional gasoline, offering limited reductions with 100-year global warming potential (GWP) yet leading to increases with 20-year GWP. Other liquid fuel pathways (methanol, ethanol, and Fischer-Tropsch liquids) have larger GHG emissions than conventional gasoline even when carbon capture and storage technologies are available. Life cycle GHG emissions of natural gas pathways are sensitive to the vehicle fuel efficiency, to the methane leakage rates of natural gas systems, and to the GWP assumed. With the current vehicle technologies, the break-even methane leakage rates of CNG, gaseous hydrogen FCEV, and BEV are 0.9%/2.3%, 1.2%/2.8%, and 4.5%/10.8% (20-year GWP/100-year GWP). If the actual methane leakage rate is lower than the break-even rate of a specific natural gas pathway, that natural gas pathway reduces GHG emissions compared to conventional gasoline; otherwise, it leads to an increase in emissions.

3.2. Introduction

The past decade has seen a significant increase in U.S. natural gas productions due to the technological success in extracting natural gas from unconventional resources. While in 2005 the United States (U.S.) shale gas production was negligible, by 2012 it reached 25.7 billion cubic feet per day (BCF/d),[9] and today it accounts for 40% of total dry natural gas production in the U.S.[10] The U.S. Energy Information Administration (EIA) forecasts that shale gas production will reach 45.8 BCF/d by 2040.[11] The rapid increase of natural gas supply has led to a large decrease in wellhead prices, which dropped from $7.97 per thousand cubic feet (Mcf) in 2008 to $2.66/Mcf in 2012.[12] As a result of the emergence of this domestic natural gas resource, there is a growing interest in using natural gas for electricity generation, for producing transportation fuels, for petrochemical manufacturing, and also for exports.[11]

Light duty vehicles (LDV) are the largest providers of mobility services to the U.S. population. More than 90% of U.S. families have at least one vehicle, and, on average, each household owns more than two vehicles.[3] Currently, there are more than 244 million LDVs in use in the U.S and each year around 15 million new LDVs are sold.[3] In 2013, more than half (54%) of the new LDVs were gasoline-powered passenger vehicles, while the other half were gasoline-powered sport utility vehicles (SUV) (32%), and pick-up trucks (11%).[3] By comparison, there are less than 1.2 million alternative fuel vehicles (AFV) in use,[3] representing only 0.5% of the LDV fleet. In the transportation sector, gasoline and distillate fuel from petroleum meet more than 90% of energy consumption.[3] The emergence of natural gas supply may open the opportunity for use of natural gas for transportation.[5–8,58–60,62,65,66] If so, several different pathways can be used. For example, natural gas could be used directly as a transportation fuel through compression or liquefaction, or it can be converted into other transportation fuels such as hydrogen, electricity, and even gasoline and diesel via the Fischer-Tropsch process.

Life cycle analysis (LCA) is a widely used method to assess the environmental effects of a product or service from production to end of life.[50] There is an extensive body of research about the life cycle greenhouse (GHG) emissions of alternative transportation fuels, including hybrid electric vehicles (HEV), plug-in hybrid electric vehicles (PHEV), battery electric vehicles (BEV), and hydrogen fuel cell electric vehicles (FCEV).[4,13–20,22,23,25–29,130–137] Similarly, another large body of work has analyzed the life cycle GHG emissions of using natural gas to meet end uses (including transportation).[15,27,29,114–116,135–147] In 1999, Wang et al.[15] evaluated the life cycle GHG emissions of nine natural gas-based fuels, compressed natural gas (CNG), liquefied natural gas (LNG), liquid petroleum gas (LPG), electricity, methanol, gaseous hydrogen, liquid hydrogen, Fischer-Tropsch diesel, and dimethyl ether (DME), and found that the "use of NG-based fuels can help reduce per-mile fossil energy use considerably and eliminate petroleum use in most cases; all [but near-term M85 FFVs] help reduce GHG emissions." More recently, Venkatesh et al.[27] used a Monte Carlo analysis to characterize uncertainty of the life cycle GHG emissions of CNG and gasoline HEVs for passenger vehicles. The authors found that both HEVs and CNG vehicles achieve emission reductions over conventional gasoline vehicles (on average 25% reduction for HEV and 5% reduction for CNG), but with some probabilities that either pathway is worse than conventional gasoline vehicles. Similarly, Curran et al.[136] used the GREET model (version 2012) to analyze the well-to-wheel energy use and GHG emissions from natural gas pathways. They specifically compared CNG vehicles and electric vehicles charged with natural gas-based electricity and found that the latter is better. Dai et al.[137] considered other environmental impacts (air pollutants, toxicity, land use, and water consumption) in addition to GHG emissions and found that BEVs and FCEVs with natural gas-based electricity and hydrogen reduce environmental impacts. Luk et al.[29] analyzed the life cycle GHG emissions and ownership costs of CNG and natural gas-derived electricity in BEVs. They found that CNG is more cost-effective in reducing GHG emissions than BEVs. While these studies reached similar conclusions, they also shared common limitations. Except Venkatesh et al.[27] and Luk et al.,[29] both of which focused only on CNG and electricity from natural gas, other studies reported point estimates and largely ignored the uncertainty and variability in the life cycle of natural gas pathways, especially the uncertainty in methane emissions from natural gas systems. In addition, they used outdated global warming potential (GWP) values that do not reflect the most recent Intergovernmental Panel on Climate Change (IPCC) estimates, and they did not include SUVs in the analysis.

In this chapter I address these shortcomings by performing an LCA coupled with a Monte Carlo analysis to estimate GHG emissions from a broad set of potential fuel pathways that use natural gas directly or indirectly to power passenger vehicles and SUVs. This study uses scenario analysis and break-even analysis to understand the implications of policy-relevant choices (such as the GWP timeframe) and highly uncertain variables (such as methane emissions from natural gas systems).

3.3. Methods

3.3.1. System Boundary

Figure 3.1 illustrates the natural gas pathways and engine technologies considered in this study. The analysis includes six different types of transportation fuels: CNG, natural gas-based electricity, natural gas-based hydrogen, natural gas-based Fischer-Tropsch liquids (gasoline and diesel), natural gas-based methanol, and ethane-based ethanol. This study evaluates six vehicle technologies: a sparking ignition internal combustion engine vehicle (SI-ICEV), a flex fuel vehicle (FFV), a compression ignition internal combustion engine vehicle (CI-ICEV), a hybrid electric vehicle (HEV), a plug-in hybrid electric vehicle (PHEV), a battery electric vehicle (BEV), and a fuel cell electric vehicle (FCEV). For both passenger vehicles and SUVs, the functional unit is one vehicle kilometer traveled.

This study considers GHG emissions from the full life cycle for each pathway. For instance, GHGs result from the combustion of natural gas and other fossil fuels used to provide energy in the production and transport of natural gas, natural gas flaring, non-combusted emissions (such as vents and fugitive methane and CO_2 emissions), and land use (associated with well pad and well constructions). Downstream activities include production of transportation fuels from natural gas, transportation of final fuels from plants to fueling stations, and use in vehicles. In addition to these fuel-related GHG emissions, I also include emissions from vehicle manufacturing. I exclude emissions associated with building the infrastructure needed to deploy different fuel pathways. Recent literature suggests that they contribute to less than 1% of the cradle-to-gate (from extraction of feedstock to the finished product from the production facilities) GHG emissions in the case of oil and natural gas production as well as the production of electricity and hydrogen from natural gas.[110,111,148]

I convert emissions of different GHGs into CO_2-equivalent emissions by multiplying the mass of emissions and their GWP. I consider fossil methane, and model the uncertainty in GWP using a normal distribution[2,128] (**Table 3.1**). I use the latest GWP values with inclusion of climate-carbon feedbacks reported in the Fifth Assessment Report of the IPCC[2] and assume a normal distribution. All else being equal, the choice of time horizon for GWP greatly changes the equivalent CO_2 emissions of methane, which has a much higher GWP over 20 years than over 100 years. While most life cycle studies used 100-year GWP, short-term implications of methane emissions are increasingly of interest.[142,149] This study thus reports GHG emissions using GWP with both 100-years and 20-years. There are limitations in using GWP, such as ignoring the timing of emissions,[2,28,124–127] but taking account of alternative climate metrics is beyond the scope of this chapter.

Table 3.1. Global Warming Potential (GWP) values (climate-carbon feedbacks of non-CO_2 gases are considered).

Greenhouse gas	100-yr	20-yr
CO_2	1	1
CH_4 (fossil)	Norm (36, 8.5)	Norm (87, 15.9)
N_2O	Norm (298, 52.5)	Norm (268, 34.2)

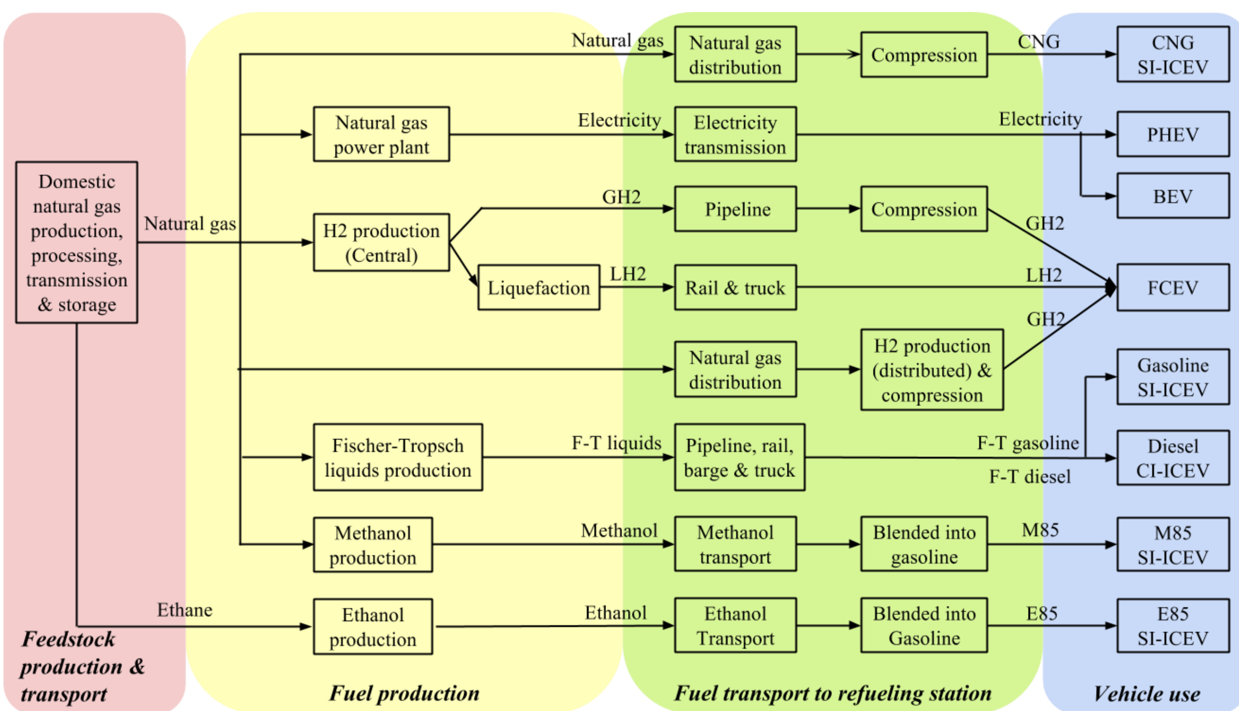


Figure 3.1. Study system boundary of natural gas pathways. Colored areas correspond to different life cycle stages: natural gas upstream (pink), fuel production (yellow), fuel transport (green), and vehicle operation (blue) (indicated by engine technologies). Both feedstock and energy carriers are marked along each pathway. CNG = compressed natural gas; H2 = hydrogen; GH2 = gaseous hydrogen; LH2 = liquid hydrogen; F-T = Fischer-Tropsch; M85 = a blend of methanol (85% by volume) and gasoline (15%); E85 = a blend of ethanol (85% by volume); ICEV = internal combustion engine vehicle; SI = sparking ignition; CI = compression ignition; PHEV = plug-in hybrid electric vehicle; BEV = battery electric vehicle.

3.3.2. Natural Gas Upstream Emissions

U.S. production of natural gas in 2013 came from four sources, including conventional natural gas (38%), shale gas (40%), associated gas as a co-product of crude oil (18%), and coal-bed methane (5%).[10] This study focuses on shale gas given its prevalence in the U.S.[11] Differences in life cycle GHG emissions from conventional natural gas and shale gas are small.[115,150]

This study uses a bottom-up life cycle assessment framework[27,115,139,142,151] to estimate natural gas upstream estimations. The bottom-up framework divides the natural gas systems in five stages - preproduction, production, processing, transmission and storage, and distribution – following the U.S. Environmental Protection Agency (EPA)'s GHG emissions inventory.[152] For each stage, I model key emission sources individually, which require emission factors (emission per unit of activity) and corresponding activity data (total units of activity). The emissions model accounts for two types of emissions sources, combustion emissions and non-combustion (i.e. fugitive) emissions. Combustion sources include well drilling, transportation of hydraulic liquids and wastewater, lease fuel use, plant fuel, and pipeline fuel use. I use previous studies[27,139,152,153] to model these combustion sources. Non-combustion sources include intentional venting and non-intentional leakage. Sources of intentional venting include well completion, well workover, liquid unloading, and blowdowns and upsets. Non-intentional sources include leaks from production devices and pipelines. Existing studies usually rely on the U.S. EPA's GHG emissions inventory for emissions factors and activity data. However, since U.S. EPA still uses outdated emission factors originating from a field campaign in 1990s, recent research efforts have focused on updating these emission factors with on-site measurements.[97–99] This framework thus uses recent field measurements (such as well completion, well work-over, and liquid unloading[97,99]). In addition, I use the U.S. EPA's GHG emissions inventory[152] for emissions sources that recent studies have not evaluated.

I use Monte-Carlo analysis to account for variability and uncertainty of emissions factors and activity data of emissions sources considered in the bottom-up framework. Some recent studies suggest that there is a 50% difference between top-down estimates and bottom-up estimates of methane emissions from natural gas systems.[114,116] Unfortunately, there is no sufficient information to develop meaningful probability distributions that account for bottom-up and top-down estimates. Instead, to account for potential bias in the baseline bottom-up estimate, the emissions model includes a pessimistic scenario of methane emissions. In this pessimistic scenario, I multiply the distribution of the baseline bottom-up emissions estimates by 1.5. I refer the readers interested in the technical details of the bottom-up framework and Monte Carlo analysis for the upstream natural gas emissions in Appendix A.

3.3.3. Natural Gas-Derived Fuel Production and Distribution Emissions

Dry natural gas and ethane (the feedstock for ethanol production) go through different conversion and distribution processes to produce transportation fuels. I assume that the first four upstream stages (from preproduction to transportation) are common to all fuels produced from dry natural gas, while the last stage (distribution) is only included in pathways in which fuel production takes place at refueling stations, such as CNG and distributed gaseous hydrogen (**Figure 3.1**). Ethane, which is used to produce ethanol, shares the first three upstream stages with dry natural gas.

Electricity. Although it is difficult to identify individual power plants for specific electricity consumptions, it is likely that increasing electricity demand from electric vehicles, as well as stringent regulations on new coal power plants,[154,155] will drive further deployment of natural gas combined cycle (NGCC) plants. In fact, nearly half of the new power plant capacity in U.S. in 2013 was NGCC.[156] In this study, I assume that natural gas-based electricity powers centrally-located fuel production plants (hydrogen, ethanol, methanol, and Fischer-Tropsch liquids) as well as the charging of BEVs and PHEVs. Other, smaller, electricity consumers (such as fuel production at refueling stations) rely on the U.S. grid (with an emission factor of 612 gCO_2-eq/MJ_{LHV}[157]). I use NETL (2013)[158] to model NGCC power plants without and with carbon capture and sequestration (CCS) technologies (energy efficiencies in lower heating value are 55% and 47.5% and carbon capture rates are 0% and 88.2%, respectively). I assume transmission and distribution losses to be 6.5% of generated electricity.[159] I also assume that the charging efficiency of BEVs and PHEVs follows a uniform distribution of 85%-88%.[23,134,159]

CNG. The CNG pathway relies on natural gas pipelines to deliver natural gas to refueling stations, where compression occurs to “produce” CNG. There are two types of compressors: electric compressors and natural gas-fueled compressors, with electric compressors being the prevalent choice.[159] I thus only consider electric compressors, which have an energy efficiency that follows a uniform distribution of 0.94 to 0.98.[27]

Hydrogen (H_2). Current industry practice uses steam methane reforming technology to produce gaseous hydrogen from natural gas. I consider three configurations of a hydrogen supply chain: centrally-produced gaseous hydrogen (GH_2); centrally-produced hydrogen used in the liquid phase (LH_2); and distributed production of GH_2. I also consider CCS technologies in central hydrogen production plants. **Table 3.2** summarizes hydrogen plant assumptions.[111,159,160] Liquid hydrogen has a higher energy density than gaseous hydrogen but requires an energy-intensive liquefaction process and suffers from boil-off leakage. I assume loss factors of 0.3%, 0.16% and 0.5% at liquefaction plant, transport & distribution, and storage of LH_2 after accounting for an 80% rate of capture and reuse of boil-off gas.[159]

Fischer-Tropsch liquids. In the Fischer-Tropsch liquid production process, natural gas delivered through the transmission system undergoes thermo-chemical transformation to produce liquid fuels (gasoline and diesel) similar to those produced in an oil refinery. I use the process-level data of a Fischer-Tropsch plant (see Jaramillo et al. (2008)[161] for details) and perform an energy-based emissions allocation. Upon production, Fischer-Tropsch liquids are transported in existing petroleum-product infrastructure and used in current petroleum ICEVs.

Methanol. The late 1990s and early 2000s saw a growing interest in using methanol as an alternative fuel in the U.S. and Canada.[162,163] While there are currently no methanol-fueled vehicles in the market, high performance of methanol attracts some niche markets.[162] I model a M85 (85% methanol and 15% gasoline in volume) pathway since pure methanol suffers from cold start issue and has safety concerns (such as invisible flame and erosion of mechanical systems).[162] Methanol can be produced from natural gas in a centralized methanol production plant using steam reforming technologies. It is a two-step process, where the first step is to produce synthesis gas from natural gas, and the second step is the catalytic synthesis of methanol from the synthesis gas.[162] **Table 3.3** summarizes the assumptions in the M85 model.[22,159,164,165]

Ethanol. While the production of ethanol has already transitioned to biomass-based pathways (such as corn grain, sugarcane, and cellulosic biomass[166]), ethanol can be produced from fossil fuel-based naphtha and ethane, a co-product of methane. There are two steps to produce ethanol from ethane – the first step is to produce ethylene through ethane cracking[144] and the second step is to produce ethanol using catalytic ethylene hydration.[167] The Appendix B provides specific details about the technical specifications of this two-step process.

Table 3.2. Hydrogen production plant profile for one MJ of hydrogen produced.[111,159,160]

Key parameters	Central hydrogen plant (without CCS)		Central hydrogen plant (with CCS)[160]		Hydrogen production at refueling stations	
	Distribution	Distribution parameters	Distribution	Distribution parameters	Distribution	Distribution parameters
Energy efficiency*	triangular**	(0.72, 0.74, 0.79)	triangular	(0.71, 0.73, 0.78)	triangular	(0.71, 0.72, 0.74)
Electricity share of all inputs	triangular	(0.007, 0.012, 0.044)	triangular	(0.018, 0.026, 0.03)	triangular	(0.024, 0.050, 0.083)
Natural gas share of all inputs	1- (electricity share)		1- (electricity share)		1- (electricity share)	
Process GHG emission factor (gCO_2-eq/MJ of H_2)+	uniform++	77-79	uniform	7.7-7.9	uniform	77-77.2

**The energy efficiency is the ratio of the energy contents of all outputs to those of all inputs (natural gas as feedstock, natural gas as fuel and electric power). + Process GHG emission factor include GHG emissions within the hydrogen production plant but doesn't include emissions embodied in electricity inputs and upstream emissions of natural gas inputs. **Parameters of the triangular distribution are lower limit, mode, and upper limit. ++Parameters of the uniform distribution are lower limit and upper limit.*

Table 3.3. Methanol production plant profile for one MJ of methanol produced.[22,159,164,165]

Key parameter	Distribution type	Distribution parameters
Energy efficiency*	Triangular+	(0.41, 0.57, 0.68)
Natural gas share of all inputs	Triangular+	(0.994, 0.999, 1.000)
Electricity share of all inputs	1- (natural gas share of all inputs)	
Feedstock share of natural gas inputs	Triangular+	(0.63, 0.78, 0.88)
Fuel share of natural gas inputs	1- (feedstock share of natural gas inputs)	

**The energy efficiency is the ratio of the energy contents of all outputs to those of all inputs (natural gas as feedstock, natural gas as fuel and electric power).*
+Parameters of the triangular distribution are lower limit, mode, and upper limit.

3.3.4. Petroleum Upstream and Combustion Emissions

The baseline fuel for LDVs is conventional gasoline in the U.S. I rely on existing studies[159,168–170] to model GHG emissions from its life cycle, which includes oil production, oil transport, oil refining, gasoline transport, and combustion during vehicle use (**Table 3.4**). I also include conventional diesel and one specific type of unconventional oil,[171] Canadian oil sand-derived crude, which accounts for the largest fraction of imported oil in the U.S.[172]

Table 3.4. Life cycle GHG emissions of gasoline and diesel (Unit: gCO_2-eq/MJ of fuel delivered).

Fuel	Stage	Distribution	Mean	S.D.⁺	95% C.I.⁺
Conventional gasoline	Upstream	Venkatesh et al. (2011)[169*]	18.6	4.0	12.6-28.0
	Combustion	Triangular (71.0, 72.7, 74.9)[159,168–170]	72.9	0.8	71.4-74.4
	Life cycle	Upstream plus combustion	91.5	4.1	85.2-101.0
Conventional diesel	Upstream	Venkatesh et al. (2011)[169**]	17.5	4.3	11.5-27.7
	Combustion	Triangular (72.6, 74.1, 75.2)[159,168–170]	74.0	0.5	72.9-74.9
	Life cycle	Upstream plus combustion	91.5	4.3	85.3-101.7
Oil sand-derived gasoline / diesel	Life cycle	Uniform (103,118)[171]	110.5	4.3	103.4-111.6

*Notes: I assume that upstream emissions of conventional gasoline and conventional diesel follow the same distributions as in Venkatesh et al. (2011). *Conventional gasoline upstream emissions follow the difference between a shifted log-logistic distribution (mu=2.2, alpha=0.2, delta=80) and a triangle distribution (68.2, 70.2, 74.6). **Conventional diesel upstream emissions follow the difference between a shifted log-logistic distribution (mu=2.3, alpha=0.2, delta=82) and a triangle distribution (73.6, 75.3, 76.6). ⁺ S.D. stands for standard deviation; C.I. stands for confidence interval.*

3.3.5. Vehicle Specifications

This study models new vehicles available in the market instead of the existing fleet. I use functionally equivalent vehicles across fuel pathways - compact passenger vehicles and compact SUVs - to eliminate the bias of vehicle choices.[72] There is at least one vehicle model currently offered in the market for all natural gas and petroleum pathways except for M85. The fuel economy assumptions of these vehicles are from the U.S. Department of Energy (DOE) and the U.S. EPA[173] (see **Table 3.5**). I rely on the literature for fuel economy assumptions of M85 vehicles and PHEVs.[132,134,159] I further rely on the National Household Travel Survey (NHTS)[174] to estimate the fraction of electric and gasoline driving for any ride of PHEVs (PHEV30 and

PHEV60 with 30 km and 60 km electric-only range, respectively). On-road fuel economy may be different from measurements due to factors such as speed, weight, age, road gradient, and ambient temperate.[104,175] While it is beyond this study's scope to consider these factors, they should be carefully studied in future studies that focus on regional variations. The Appendix B includes the technical specifications of the vehicles and assumptions for M85 vehicles and PHEVs.

Table 3.5. Fuel economy assumptions (Unit: Miles per Gallon of gasoline equivalent).

Pathway		Passenger Vehicle	SUV	Source
Gasoline (baseline)		33	25	fueleconomy.gov[173]
Diesel		32.3	26.2	
Gasoline HEV		45	33	
PHEV30	Charging sustaining	43.8	-	Karabasoglu et al. (2013)[134]
	Charging depleting	112	-	
PHEV60	Charging sustaining	42.4	-	
	Charging depleting	105	-	
BEV		110	76	fueleconomy.gov[173]
CNG dedicated		31	-	
M85 dedicated		35.3	-	GREET (v.2014)[159]
E85 flex fuel vehicle		31.6	24.7	fueleconomy.gov[173]
Hydrogen fuel cell vehicle		59	49	

Vehicle tailpipe emissions include CO_2, CH_4, and N_2O emissions. I calculate tailpipe CO_2 emissions using the fuel economy of the vehicle and combustion emissions of the fuel (based on the fuel's carbon content). I model CH_4 and N_2O emissions using emission factors from the GREET model.[159] I assume that electric vehicles (both BEVs and FCEVs) have zero GHG emissions at tailpipe. As a result of incomplete combustion of natural gas, CNG vehicles have much higher CH_4 emission factors than conventional gasoline vehicles.

This study includes emissions from vehicle manufacturing. I assume all ICEVs have similar vehicle manufacturing emissions.[159] I rely on the GREET model[159] for manufacturing emissions of ICEVs and FCEVs. For HEVs, PHEVs, and BEVs, I assume they have incremental emissions associated with battery manufacturing, which I calculate using emission factors[176] and activity data (battery size and number of batters per vehicle lifetime), compared to the ICEVs (see the Supported Information for details).

3.4. Results

3.4.1. Comparisons between Natural Gas Pathways and Conventional Gasoline

Figure 3.2 shows the main results: the life cycle GHG emissions of natural gas pathways for passenger vehicles and SUVs in gCO_2-eq/km and the associated uncertainty and variability in the results. For light-duty vehicles, the median results suggest that BEVs powered by electricity generated by a natural gas plant provide the lowest GHG emissions across all technologies and pathways considered. This is due to the fact that the high efficiency of BEVs outweighed the emission penalty of electricity generation and battery manufacturing. PHEVs, either with a 30 or 60-km range, when powered by natural gas electricity, have the second lowest average emissions. Both BEVs and PHEVs provide large (more than 20%) emissions reductions compared to conventional gasoline but none of them is a dominant strategy when compared to gasoline HEVs. Gaseous hydrogen FCEVs and CNG vehicles have comparable life cycle emissions with conventional gasoline, offering limited reductions with 100-year GWP yet leading to increases with 20-year GWP. All other fuel pathways (E85, M85, and Fischer-Tropsch liquids) have larger GHG emissions than conventional gasoline.

Compared to passenger vehicles, SUVs have larger life cycle GHG emissions for all natural gas pathways. For example, a hydrogen-powered fuel-cell SUV has life cycle GHG emissions that are at least 19% higher than a fuel-cell passenger vehicle, while a battery electric SUV has life cycle GHG emissions that are 41% higher than a battery electric passenger vehicle. While SUVs provide advantages such as larger cargo space and better road accessibility, their functions are not significantly different from passenger vehicles in most applications but they have larger carbon footprints.

When looking at emissions changes between natural gas pathways and conventional gasoline, I find that all but two natural gas pathways have the same sign (emission increase or emission reduction) under all scenarios considered (i.e., 20 and 100-year GWP; baseline and pessimistic methane emissions). The exceptions are gaseous hydrogen and CNG for which the time horizon of GWP determines whether they reduce or increase emissions when compared to gasoline. Still, GWP has non-negligible effects on the absolute levels of GHG emissions. Using the 20-year GWP, the life cycle GHG emissions of natural gas pathways increase by 6-17% compared to emission estimates with 100-year GWP.

I find that the benefits of natural gas pathways largely depend on two factors, vehicle fuel efficiency and carbon intensity of the fuel (cradle-to-gate GHG emissions per MJ of fuel delivered, see the Appendix B for visualizations on emissions estimates). Pathways that run on electric vehicles (except liquid hydrogen) have smaller emissions than pathways that run on ICEVs. Within each vehicle technology group, if vehicles' fuel efficiencies are comparable, pathways with higher supply-chain efficiency emit less than pathways with lower supply-chain efficiency (e.g., CNG vs. methanol or gaseous hydrogen vs. liquid hydrogen). To assess the contribution of these two effects, I developed a break-even analysis that shows the trade-offs between vehicle fuel efficiency and methane leakage rate that influences the carbon intensity of a natural gas-based fuel.

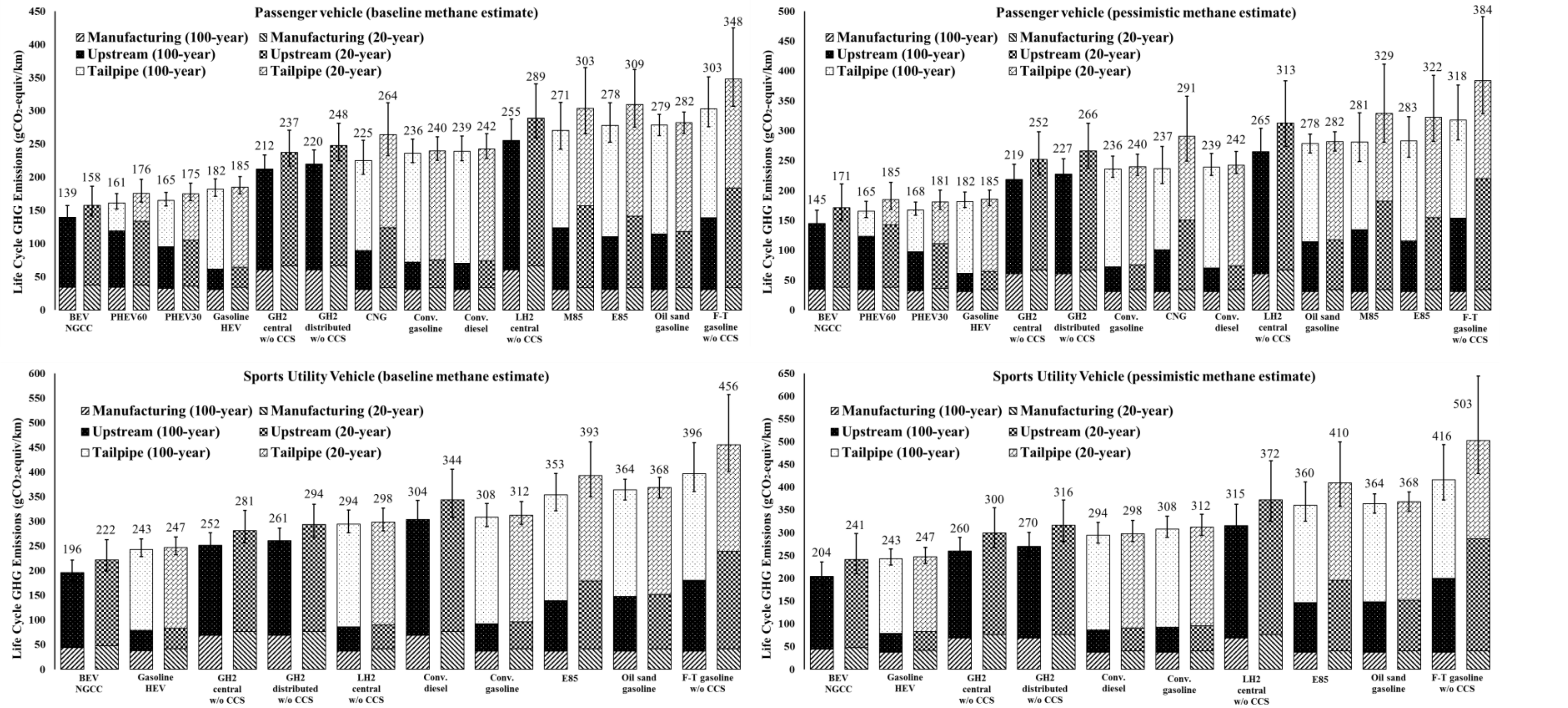


Figure 3.2. Life cycle GHG emissions of natural gas pathways for passenger vehicles (top panels) and SUVs (bottom panels) in gCO_2-eq/km. Here I assume both baseline (left panels) and pessimistic estimates (right panels) of methane emissions from natural gas systems. Error bars represent the 95% confidence interval of life cycle GHG emissions, which comprise three sources, vehicle manufacturing, upstream (well-to-pump), and tailpipe (pump-to-well) emissions. Upstream emissions include all use-related emissions from primary energy extraction to dispensing the fuel into vehicles. Tailpipe emissions include all use-related emissions from vehicle operation. Estimates using both 100-year GWPs (left bars) and 20-year GWPs (right bars) are presented side by side for each pathway. Pathways are sorted based on life cycle emissions with 100-year GWP. Data labels represent mean life cycle GHG emissions.

3.4.2. Break-Even Methane Leakage Rates

I perform a break-even analysis for three pathways, CNG, GH_2 FCEV, and BEV, as their emissions are highly dependent on fugitive methane emissions. Further, these pathways are the closest to commercial deployment. I conduct a parametric analysis where I determine the methane leakage rate at which life cycle GHG emissions from each of these natural gas pathways equal that of conventional gasoline, hereafter called the *break-even* rate (For technical details regarding the break-even analysis, please refer to the Appendix B). If the actual methane leakage rate from the natural gas systems is lower than the calculated break-even rate of a specific pathway, that pathway has a lower life cycle emissions than conventional gasoline. Further, the higher the break-even rates, the larger emissions reduction comes from that pathway.

The analysis shows that the break-even rate depends on vehicle fuel efficiency (expressed as the ratio of the gasoline equivalent fuel economy of a natural gas-powered vehicle relative to a conventional gasoline vehicle, hereafter denoted as the *energy economy ratio*, EER) and GWP (**Figure 3.3**). There is a linear relationship between break-even rate and the vehicle's EER, i.e., all else being equal, the break-even rate increases with a higher EER. See, for instance, the relationship between the break-even rate and the EER of the CNG pathway using a 100-year GWP: increasing the EER of the CNG vehicle from 100% (i.e., the same fuel efficiency as its gasoline counterpart) to 110% (i.e., 10% more efficient than its gasoline counterpart) allows for an 1.4 percent point increase in methane leakage rate (from 3.1% to 4.5%).

Figure 3.3 suggests that current CNG vehicles offer emissions reductions if the life cycle methane leakage rate is lower than 2.3% (using the 100-year GWP) or 0.9% (using the 20-year GWP). Shorter time horizon (such as 20 years), which considers a higher warming potential of methane, requires a lower break-even rate than a longer time horizon (such as 100 years). Current FCEVs offer emission reductions if the life cycle methane leakage rate is lower than 2.8% (100 years) or 1.2% (20 years). Of the three pathways considered, BEVs have largest break-even rates with current vehicle technologies, 10.8% (100 years) and 4.5% (20 years). This is consistent with the previous findings that BEVs achieve largest emissions reductions (see **Figure 3.2**). For comparison, the baseline estimate of methane leakage is 1.3%, and the pessimistic estimate is 2.0%, which mean that CNG vehicles, FCEVs and BEVs could achieve emissions reduction with 100-year GWP but not with 20-year GWP, as shown in **Figure 3.2**.

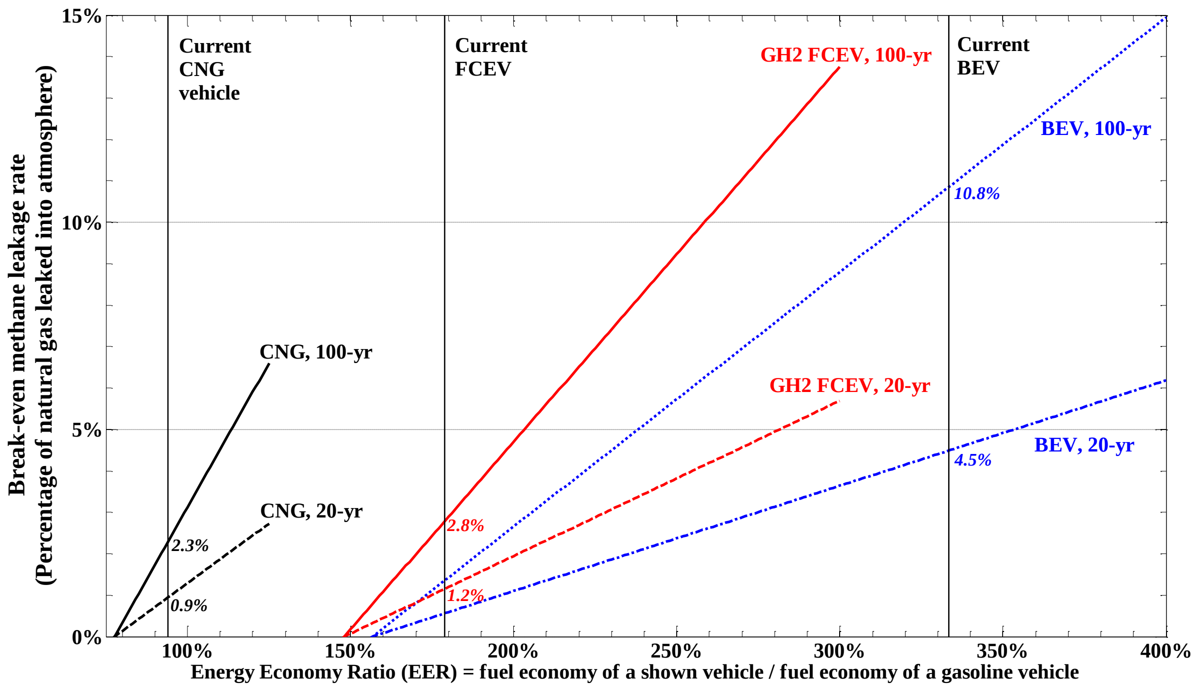


Figure 3.3. Break-even methane leakage rates of CNG, distributed gaseous hydrogen FCEV, and BEV pathways. At the break-even rate (defined as volumetric percentage of natural gas that leaks directly into atmosphere in the life cycle), life cycle GHG emissions from these natural gas pathways are comparable with conventional gasoline. If the actual methane leakage rate from the natural gas systems is lower the calculated break-even rate of a specific pathway, that pathway has a lower life cycle emissions than conventional gasoline. Current EER for these vehicles are marked with vertical lines.

3.4.3. Other Important Factors

While vehicle manufacturing emissions account for less than 30% of life cycle GHG emissions across all vehicles, manufacturing emissions from FCEVs, which are almost double the manufacturing emissions of ICEVs, merit further discussion. In the sections presented above, I assumed that fuel cells work for the entire lifetime of the vehicle. However, if consumers need to replace fuel cells during vehicle lifetime, increased manufacturing emissions of FCEVs would cause hydrogen pathways to have larger life cycle GHG emissions than gasoline vehicles. Although there is not enough data to quantify the actual lifetime of fuel cells as they are still in the demonstration phrase, U.S. DOE has pushed for increasing the durability and reliability of the fuel cell system.[177]

3.4.4. CCS Technology

One potential way to further reduce GHG emissions from natural gas pathways is to capture the carbon emitted from centralized fuel production facilities and store it in geological structures. I consider a scenario in which carbon CCS is available at centralized hydrogen production facilities, natural gas combine cycle power plants, and Fischer-Tropsch plants. CCS technologies reduce the cradle-to-gate emissions of these fuels significantly (37% for Fischer-Tropsch liquids, 38-46% for hydrogen, and 64% for NGCC electricity, compared to the same fuel pathway without CCS technologies). Compared to conventional gasoline passenger vehicles, when CCS technologies are available at fuel production facilities, BEVs and FCEVs have much lower emissions than conventional gasoline (71%, 47% and 29% reductions for BEVs, gaseous hydrogen, and liquid hydrogen, respectively), but Fischer-Tropsch liquids still have higher emissions than gasoline vehicles.

3.5. Conclusions and Policy Recommendations

In this chapter, I find that the use of natural gas to produce electricity to then charge BEVs has the lowest life cycle GHG emissions of all natural gas-based fuels considered in this study, and it achieves large emission reductions compared to conventional gasoline. On the other extreme, E85, M85 and Fischer-Tropsch liquids, which have low requirements for new infrastructure, are most likely to lead to increases in GHG emissions. Hydrogen and CNG pathways have the ability to reduce life cycle GHG emissions in LDVs, but such reductions require that methane leakage rates decrease from their current levels.

I find larger uncertainty and variability in life cycle GHG emissions of natural gas pathways than that of conventional gasoline. There is stochastic dominance among natural gas pathways (see Appendix B) so I simplified the discussions by referring to mean emissions. However, it is important that policymakers consider uncertainty and variability when they set policy goals based on relative or absolute emissions.[178]

While this chapter focuses on GHG emissions, natural gas-based fuels may provide other environmental benefits, such as the reduction of other air pollutants. On the other hand, the adoption of natural gas has to include consumers as part of the equation, namely in what concerns costs, fueling convenience, performance and safety. These issues are outside the scope of this work and will be addressed in future research.

Increasing vehicle fuel efficiency and reducing methane emissions from the natural gas system are promising strategies to reduce GHG emissions from using natural gas for road transportation. For the same mobility service, a higher vehicle fuel efficiency leads to lower life cycle emissions or translates into a higher allowable break-even methane leakage rate. Recent studies find evidences of "super emitters" in natural gas systems – a small number of emission sources that lead to a significant share of methane emissions.[97,98,100,101,114,179,180] There are cost-effective technologies to reduce these emissions[119,120,181,182] that would reduce the methane leakage rate and provide better opportunities for reducing GHG emissions from light duty-vehicles by using natural gas as a transportation fuel.

Chapter 4. Life Cycle Air Pollution Damages of Petroleum and Natural Gas Pathways for Powering Light-Duty Vehicles and Heavy-Duty Vehicles

This chapter is based on the working paper: Tong, F.; Jaramillo, P.; Azevedo, I. *Life Cycle Air Pollution Damages of Petroleum and Natural Gas Pathways for Powering Light-Duty Vehicles and Heavy-Duty Vehicles*; Carnegie Mellon University: Pittsburgh, PA, 2016.

4.1. Abstract

This chapter estimates the life cycle air pollution damages due to criteria air pollutant (CAP) emissions from light-duty vehicles (LDVs) and heavy-duty vehicles (HDVs) in the U.S using gasoline, diesel, compressed natural gas (CNG), liquefied natural gas (LNG), increased penetrations of natural gas-based electricity (replacing existing coal-fired electricity generation), and current electricity grid. I compare air pollution damages estimates using two recent emissions data for primary energy production and oil refineries, and two state-of-the-art marginal damage models. Battery electric vehicles (BEVs) with increased penetrations of natural gas-based electricity achieve the lowest damages for passenger cars, sports utility vehicles, and transit buses, while CNG, LNG high-pressure direct-ignition (HPDI) and diesel hybrid-electric trucks each achieves lowest damages for tractor-trailers in part of the U.S. This work shows that spatial differences in marginal damages of SO_2 and NO_x lead to different rankings of fuel pathways in the Rocky Mountain, western Texas, and New England. Large differences in emissions factors of oil refineries from two emissions source exist but the findings remain robust. Better data collection on CAP emissions from key energy facilities and continued advancement on the marginal damage estimates should improve our understandings of air pollution damages from transportation use in the future.

4.2. Introduction

On-road vehicles have greatly improved the mobility of people and goods, but they have also produced negative externalities such as air pollutants, car accidents, and noise. The transportation sector accounts for more than half of carbon monoxide (CO) and nitrogen oxides

(NO_x) emissions in the U.S., as well as nearly a quarter of volatile organic compounds (VOCs) and 6% of particulate matter with a diameter of 2.5 μm or less ($PM_{2.5}$).[3] Increased concentration of these pollutants leads to poor urban air quality, and increases the risks of mortality and morbidity in exposed popultaions.[183–185]

In 2010, the National Research Council (NRC) provided a consistent and comprehensive assessment of externalities from major energy uses.[4] In particular, NRC (2010) found large air pollution damages associated with on-road vehicles, totaling $56 billion. Since 2010, a number of studies have examined the air pollution damages of petroleum fuels and alternative fuels in U.S., motivated by the increased public awareness of negative externalities from modern technologies, and spurred by progress in emission measurements and in the understandings of emissions' health impacts. Specifically, these studies focused on biofuels,[32,186–188] CNG vehicles,[29,32] diesel vehicles,[188–190] plug-in hybrid electric vehicles (PHEVs) and/or battery electricity vehicles (BEVs),[26,29,32,191,192] as well as gasoline vehicles (all the above studies except[189,190]). **Table 4.1** summarizes the key characteristics (scope, emissions' data sources, and damages used) in these recent studies. I find that recent studies focused on passenger cars, and studies that estimated the life cycle damages used the GREET model[159] for their emission estimates. The common attributes in these studies, while enabling comparative analysis between the different studies, have some limitations. First, none of the studies published after the NRC report[4] estimates the air pollution damages for other vehicle types. In the last two decades, the sales of sports utility vehicles (SUVs) are comparable in size with passenger cars but no study has included SUVs.[3] While light duty vehicles (LDVs) outnumber heavy-duty vehicles (HDVs) by about 100 to 1,[3] larger per-mile emissions and greater mileage could result in disproportionally larger damages from HDVs. Second, the reliance on the GREET model for the emissions inventory may lead to systematic biases. Recent work by Jaramillo and Muller,[193] for example, found that using U.S. Environmental Protection Agency (EPA)'s National Emissions Inventory (NEI)[194] to estimate the emissions factors for primary energy production resulted in values that differed significantly from the emissions factors reported in the GREET model.[159]

Table 4.1. Comparison of existing studies on air pollution damages related to vehicle use.

Study	Transportation pathway	Emission inventory (the geographic scope is U.S. unless otherwise mentioned).	Marginal damages (species considered; spatial scale; impact endpoints).	Life cycle air pollution damages of conventional gasoline (mean), or other major conclusions.
Jacobson et al. (2007)[186]	Gasoline; Ethanol E85.	Tailpipe emissions only (Los Angeles & U.S.).	GATOR-GCMOM and literature (CO, NO_x, SO_x, VOC, NH_3, $PM_{2.5}$, PM_{10}; nested global/CA/Los Angeles & nested global/US; cancer risk and ozone-related mortality and mobility)	"E85 may increase ozone-related mortality, hospitalization, and asthma by about 9% in Los Angeles and 4% in the United States […] relative to gasoline."
Keefe et al. (2008)[188]	Gasoline; Gasoline hybrid; Advanced diesel; Corn-based E85	GREET (version unknown)	NHTSA Regulatory Impact Analysis for CAFE standards (CO, NO_x, SO_x, VOC, $PM_{2.5}$; health and environmental impacts)	"The results from the societal analysis are qualitatively similar to the private analysis."
Hill et al. (2009)[187]	Gasoline; Corn ethanol (different process fuel feedstock); Cellulosic ethanol (corn stover, switchgrass, diverse prairie, miscanthus).	GREET (version unknown).	Response Surface Model (RSM) and BenMAP. (NO_x, SO_x, NH_3, $PM_{2.5}$; 36 km * 36 km; $PM_{2.5}$ health and environmental impacts)	1 cent/VMT
NRC (2010)[4]	Most light-duty technology and fuel combinations.	GREET (fuel cycle v1.8b, vehicle manufacturing 2.7a)	APEEP model (NO_x, SO_2, VOC, $PM_{2.5}$ & PM_{10}; county; $PM_{2.5}$, PM_{10}, ozone, NO_2, SO_2 health and environmental impacts).	1.2-1.7 cents/VMT
Michalek et al. (2011)[26]	Gasoline; HEV; PHEV20 & PHEV60. BEV240.	GREET (fuel cycle v1.8d, vehicle manufacturing 2.7a)		1 cent/VMT
Tessum et al. (2014)[32]	Gasoline; Gasoline hybrid; Diesel; CNG;	GREET (fuel cycle v1.8d, vehicle manufacturing; version 2012).	WRF-Chem v3.4 & literature for damage assessment and economic valuation (NO_x, SO_x, NH_3, VOC, $PM_{2.5}$, PM_{10}; 12*12 km; O_3 and $PM_{2.5}$ health impacts).	1.7 cent/VMT (gasoline)

<table>
<tr><td></td><td>Corn grain/corn stover ethanol;
EV grid – average, coal, natural gas, corn stover, and renewable.</td><td></td><td></td><td></td></tr>
<tr><td>Luk et al. (2015)[29]</td><td>Gasoline;
CNG;
CNG hybrid;
NGCC-BEV.</td><td>GREET (fuel cycle & vehicle manufacturing; version unknown).</td><td rowspan="4">AP2 model (NO_x, SO_2, VOC, $PM_{2.5}$; county; county; $PM_{2.5}$ & ozone mortality, morbidity, crop and timber yields, degradation of buildings and material, and reduced visibility and recreation).</td><td>0.4 cent/VMT.</td></tr>
<tr><td>Weis et al. (2016)[192]</td><td>Gasoline;
Gasoline hybrid;
PHEV35 & PHEV65;
BEV265.</td><td>GREET (fuel cycle version 2013, vehicle manufacturing; version 2013).
PJM interconnection region in U.S.</td><td>2.7 cent/VMT (including climate change damages).</td></tr>
<tr><td>Holland et al. (2015)[191]</td><td>Gasoline;
BEV.</td><td>Tailpipe emissions from gasoline vehicles (GREET); power plant emissions for BEVs.</td><td>1.3-2.9 cent/VMT (multiple vehicle models)</td></tr>
<tr><td>Holland et al. (2016)[190]</td><td rowspan="2">2009-2015 Volkswagen diesel vehicles. (Excessive NO_x emissions).</td><td rowspan="2">Tailpipe NO_x emissions only.</td><td>Damages of
$430 million and 46 excess expected deaths.</td></tr>
<tr><td>Barrett et al. (2015)[189]</td><td>GEOS-Chem
adjoint-based rapid air pollution exposure model (50km*50km; $PM_{2.5}$ and ozone mortality).</td><td>Damages of
$450 million and 59 (95% confidence interval is 10 to 150) early deaths in the US.</td></tr>
</table>

I analyze the air pollution damages of petroleum pathways and natural gas pathways for representative LDVs and HDVs in the U.S. To understand the relative effects of different emissions factors, I use and compare the emissions data in U.S. EPA's NEI and those from the GREET model.[159] Likewise, I compare the impacts of two recent estimates on CAP marginal damages, the AP2 model,[30,31] and the EASIUR model.[57,195] In short, I estimate the air pollution damages of petroleum and natural gas pathways for different vehicles types with the goal of understanding if the findings are robust when using different emissions data and marginal damage estimates.

4.3. Methods

4.3.1. System Scope

This chapter estimates the air pollution damages from criteria air pollutants (CAPs) emitted over the life cycle of vehicles used in the transportation sector. I consider five vehicle types: passenger cars, sports utility vehicles (SUVs), transit buses, local-haul (also called short-haul) tractor-trailers, and long-haul (also called line-haul) tractor-trailers. I evaluate five fuel pathways: conventional petroleum fuels (gasoline or diesel), CNG, LNG, natural gas-based electricity, and the U.S. grid electricity. Six vehicle technologies are paired with fuel pathways based on fuel characteristics and market availability (**Table 4.3**). Gasoline works with conventional sparking-ignition (SI) internal combustion engine vehicles (SI-ICEVs) in LDVs. Similarly, diesel powers conventional compression-ignition ICEVs (CI-ICEVs) in HDVs. CNG can be used in dedicated SI-ICEVs in LDVs or transit buses. For tractor-trailers, LNG powers SI-ICEVs as well as high-pressure direct ignition (HPDI) engines, which operate similar to diesel CI-ICEVs. Finally, electricity powers BEVs in LDVs and transit buses.

The geographical scope is the contiguous U.S. (48 states). Specifically, the functional unit is one vehicle mile travelled (VMT) in each of the 3,109 counties in the contiguous U.S. I note, however, that not all assumptions have explicitly considered county-level spatial resolutions, as summarized in **Table 4.2**. Compared to existing literature, this chapter relies on the most recent data to reflect the changing U.S. energy landscape. In particular, the data sources on primary energy sources and oil refinery,[193] U.S. electricity grid,[196,197] and vehicles[146,198] are for year 2011, 2014, and 2014, respectively. For the estimated air pollution damages, I use 2010 U.S. dollars and convert all other dollars using the Consumer Price Index (CPI) inflation calculator from the U.S. Bureau of Labor Statistics.[199]

Finally, I assume the health and environmental damages from one vehicle mile traveled (VMT) are marginal so that the damages of an activity can be calculated as the product of emission factors per unit of activity and the marginal damages of CAPs emissions for the same county. I also assume that vehicles are used in the same county where they are refueled. I ignore the economy-wide market responses and behavioral changes in driving patterns that may occur when alternative fuels or advanced technologies replace baseline petroleum fuels.

4.3.2. Marginal Damages of CAP Emissions

As summarized in NRC (2010),[4] marginal damage models rely on the "damage function approach", where a multi-step process is used to link the marginal CAP emissions with its resultant monetary damages. Specifically, any model should be able to calculate ambient concentration changes due to marginal emissions; to consider which fraction of population and/or what part of the ecosystem are affected by the ambient pollutant concentrations (particulate matter or ozone); to account for the health or environmental damages resulting from the exposure (via dose response functions); and to monetize these damages using market prices or nonmarket price estimates such as value of statistical life (VSL).

I use two state-of-the-art marginal damage models: the AP2 model[30,31] and the EASIUR model.[57,195] Both models follow the damage function approach outlined above and use similar concentration-response relationships with regard to $PM_{2.5}$.[183,184] However, several key differences exist in terms of emissions data year, spatial and temporal resolutions, pollutants included, and damage endpoints considered (**Table 4.2**). One key methodological difference is how they calculated airborne concentration changes due to marginal emissions: the AP2 model uses a source-receptor matrix framework derived from a Gaussian Plume model while the EASIUR model uses regression methods to derive reduced-form models from a tagged chemical transport model. Another major difference is that the AP2 model considers both health and environmental impacts due to primary and secondary $PM_{2.5}$ and ozone, while the EASIUR model only includes the health impacts from primary and secondary $PM_{2.5}$. As a result, there are significant differences in the spatial distributions on CAP marginal damages (See Heo et al.[57] for details).

Both the EASIUR and AP2 models estimate marginal damages for ground-level and elevated emission sources. I use the elevated-level marginal damages for fossil fuel power plants and oil refineries and the ground-level marginal damages for all other emissions sources. VSL characterizes the willingness to pay to reduce the risk of death and is used to monetize the negative externalities.[4] To eliminate the effect of differing assumptions about VSL, I adjust marginal damages to use the U.S. EPA's official VSL estimate ($8 million in 2010 dollars).[200] Finally, while CO is known to cause cardiovascular effects and is linked to secondary effects from ground-level ozone,[26] neither model estimated the marginal damages of CO emissions. I thus use the national-average CO damage estimate from Matthews et al. (2000),[201] which is $520/t in 1992 dollars.

Table 4.2. Characteristics of the marginal damage models used in this study.

Model	Data Year (for emissions, and for population)	Approach	Spatial resolution	Temporal resolution	Pollutant	Damages
AP2	Emissions: 2011 Population: 2011	Source-receptor matrix and damage functions.	County centroid (3,109 counties)	Annual average	$PM_{2.5}$, SO_x, NO_x, NH_3, VOC	Health (short-term and long-term mortality and morbidity) and environmental impacts due to primary and secondary $PM_{2.5}$ and ozone.
EASIUR	Emissions: 2005 Population: 2010	Reduced form models of chemical transport models, and damage functions.	Grid cell size of 36 km × 36 km (148*112 cells)	Seasonal and annual average	$PM_{2.5}$, SO_x, NO_x, NH_3	Health impacts (long-term mortality) due to primary and secondary $PM_{2.5}$.

4.3.3. Emissions Inventory

Table 4.3 illustrates the scope of the emissions inventory considered in this chapter. The life cycle boundary includes primary energy extraction, fuel production and transportation, and vehicle use. In addition, I include the manufacturing process of lithium-ion batteries for hybrid-electric vehicles (HEVs) and BEVs. I assume all other vehicle components are similar across vehicle technologies for a given vehicle type.

Table 4.3. Life cycle steps for petroleum (gasoline and diesel), natural gas (CNG, LNG, and natural gas-based electricity), and grid electricity pathway. Background colors are used to illustrate geographic details employed in the analysis – yellow means one estimate across the U.S.; green means estimates by NERC regions; blue means estimates by each county in the U.S.

Fuel pathway / Life cycle stage	**Gasoline**	**Diesel**	**CNG**	**LNG**	**Natural gas-based electricity**	**Grid electricity**
Primary energy extraction	Crude oil production and transportation		Natural gas production, processing, and transmission			Fossil fuels
Fuel production and transportation	Oil refinery		Natural gas compression	Natural gas liquefaction	NGCC power plants	All power plants
	Petroleum product transportation		[Assuming at refueling stations]		Losses over transmission & distribution lines.	
Vehicle operation	Vehicles are used in each of the 3109 counties in the Contiguous U.S.					
	Conventional ICEVs and HEVs.		SI-ICEVs	SI-ICEVs; HPDI-ICEVs	BEVs	
Vehicle type availability	Passenger cars; SUVs.	Transit buses; tractor-trailers.	Passenger cars; transit buses; tractor-trailers.	Tractor-trailers.	Passenger cars; SUVs; Transit buses.	

Primary energy extraction. Following Jaramillo and Muller (2016),[193] I estimate the air pollution damages of extracting and transporting crude oil, natural gas, and coal in the U.S. using U.S. EPA's NEI and the AP2 or EASIUR model. Specifically, I first multiply the county emissions data on primary energy extraction with marginal damages of CAPs for the same county from the AP2 or EASIUR model to calculate the total damages for producing one primary energy product. I then calculate the weighted-average damages of an energy product weighted by CAP emissions from each county. Because the trade flows of energy products are not available, I use the weighted-average damages across the U.S. Most primary energy products are produced domestically in the U.S. with the exception of crude oil.[202] I assume that imported crude oil has similar air pollution damages with U.S. crude oil. While this may not be the case for now, the share of U.S. crude oil import is declining rapidly.[202] In addition, I account for air pollution damages due to electricity used to move crude oil pipelines and natural gas pipelines using pipelines' electricity intensity data[3,203] and electricity damages calculated in this chapter.

The GREET model also provides weighted-average emissions factors for oil and natural gas extraction and coal mining. I estimate air pollution damages from primary energy extraction by multiplying the GREET's emissions factors with the weighted-average marginal damages of CAPs assuming the GREET model has the same spatial emissions inventory as the U.S. EPA's NEI.[193] I find that air pollution damages of crude oil and natural gas production align well between the two data sources while air pollution damages of coal mining differ by 1.6 or 2.8 (with GREET model-based damages being higher than those from NEI).

Fuel production. Similar to primary energy extraction, I use emissions data from U.S. EPA's NEI (county-level) and the GREET model (U.S.-average) to estimate air pollution damages from oil refining. Because the emissions data are normalized to oil refinery capacity (barrel of crude oil inputs), I convert it to the actual outputs of petroleum products such as gasoline and diesel using the energy allocation method[204] and the utilization rate of refinery capacity.[202,205] The oil refining damages differ significantly depending on the emissions data used: GREET-based damages (0.06-0.08 cent/MJ) are much larger than those based on U.S. EPA's NEI (0.01 cent/MJ). Following *Chapter 2*, *Chapter 3*, and Tong et al.,[146,198] I model distributional CNG and LNG pathways in this chapter. I assume both compression and liquefaction processes take place at refueling stations, that they use grid electricity, and that their energy efficiency are 96% and 90%, respectively.

I estimate air pollution damages from grid electricity for each NERC (North American Electric Reliability Corporation) region[206] as I assume the electricity is balanced within the NERC region for simplicity.[4,26,29,32,187,188,192] I combine U.S. EPA's Continuous Emission Monitoring System (CEMS)[196] data (net generation and annual SO_2, NO_x, and, $PM_{2.5}$ emissions for fossil fuel power plants) and U.S. Energy Information Administration (EIA)'s Form-923[197] data (net generation for non-fossil fuel power plants) to characterize U.S. grid electricity in 2014. I calculate the air pollution damages from on-site combustion by multiplying the CAP emissions with the marginal damages of CAPs in the county where the power plant is located. I use the U.S. EIA's Form-923 to calculate fossil fuel consumptions by NERC region and allocate air pollution damages of primary energy extraction to electricity generated. Finally, I account for line losses using U.S. EPA's eGRID data.[206] I also consider a scenario of increased natural gas-based electricity in the U.S. electric power grid. In particular, I assume that each existing coal-fired power plant (identified in the CEMS data) is replaced by a new natural gas combined cycle (NGCC) power plant[158] that generates the same amount of electricity in the same county.

Vehicle operation. CAP emissions result from combustion of fossil fuels as well as tire and break wears during vehicle operation. The resulting air pollution damages are calculated by multiplying the CAP emissions with the marginal damages of CAPs in the county where the vehicle is driven. For passenger cars and SUVs, I use the GREET model for vehicle operation emissions. For transit buses, I use the chassis emission tests of CO, NO_x, and VOC from the Altoona Bus Research & Testing Center (ABTRC) reports[207] and use the GREET model for $PM_{2.5}$ and SO_2 emissions. For tractor-trailers, I use the chassis emission tests of NO_x, CO, and $PM_{2.5}$ from Thiruvengadam et al.[208] and use the GREET model for SO_2 and CO emissions. Both the Altoona Bus Test Center and Thiruvengadam et al.[208] tested transit buses and tractor-trailers that comply with the U.S. EPA's 2010 emissions standards for heavy-duty engines.[209] Following Thiruvengadam et al.,[208] I include two diesel trucks, one with a diesel particulate filter (DPF) and selective catalytic reduction (SCR), and one with only DPF, to compare the impact of emission control technologies. I assume the diesel truck with DPF and SCR as the baseline truck. In addition, I include low-NO_x CNG transit buses (whose NO_x emissions are assumed to be 1/10 of the conventional diesel bus) to account for the 2015 California Optional Low NO_x standard.[210] I find that my assumptions on HDV's vehicle operation emissions are in good agreement with GREET model's assumptions (See Appendix C for details). Finally, I summarize vehicle operation emissions from the technology perspective. BEVs reduce all tailpipe CAP emissions.

All the other technologies reduce some CAP emissions compared to conventional gasoline or diesel, but there are three exceptions. First, stoichiometric SI natural gas engines increase CO emissions significantly in the chassis emission tests.[208] Second, diesel HEV buses have 50% higher NO_x emissions than conventional diesel buses. Third, diesel tractor-trailers with DPF have higher $PM_{2.5}$ and CO emissions than diesel tractor-trailers with both DPF and SCR.

Vehicle fuel efficiency determines the amount of fuels used to move one vehicle mile. A higher fuel efficiency means lower CAPs related to upstream activities of extracting primary energy as well as producing and transporting fuels for the same VMT. I use the fuel economy assumptions from *Chapter 2*, *Chapter 3*, and Tong et al.[146,198] I assume air pollution damages of battery manufacturing in the U.S. are $8.68/kWh following Tessum et al.[32] I use the same assumptions on vehicle uses (life cycle mileage), as well as battery sizes and battery replacements for HEVs and BEVs, as reported in *Chapter 2*, *Chapter 3*, and Tong et al.[146,198]

4.4. Results

In the main text, I include results with upstream emissions data from U.S. EPA's NEI and using the EASIUR model. In the Supporting Information we include the additional results using alternative emissions data (from the GREET model) and using the AP2 model for the marginal damages.

4.4.1. Life Cycle Air Pollution Damages Across the U.S.

Figure 4.1 shows the reduction in life cycle air pollution damages of alternative petroleum and natural gas pathways replacing the baseline petroleum pathways across U.S. counties. **Table C.26** in Appendix C reports the percentage of counties that see reduction from these alternative pathways. The Appendix C also shows map visualizations of the reduction from each alternative fuel pathway in each county. We find that BEVs powered by increased penetrations of natural gas-based electricity provide the largest damage reduction in almost all counties (100%, 99%, and 75-93% of counties) for passenger cars, SUVs, and transit buses. However, BEVs powered by current grid electricity see largest increase in damages in the majority of counties for passenger cars, SUVs, and transit buses (81-83%, 85-86%, and 97-99% of counties). CNG is more likely to increase rather than reduce damages compared to gasoline when used in passenger cars and SUVs - only 18-22% of counties see reductions from CNG. However, CNG has the largest damage reduction potential when replacing diesel in tractor-trailers. LNG-HPDI has similar life cycle damages with CNG but LNG-SI is worse due to energy efficiency penalties. Indeed, CNG and LNG-HPDI trucks achieve damage reductions in 76-99% of counties for local-haul tractor-trailers but in only 32-71% of counties for long-haul tractor-trailers. Diesel HEVs reduce life cycle damages in all counties in all vehicle types except transit buses. This exception is due to higher tailpipe NO_x emissions from diesel hybrid-electric buses. Diesel trucks with only DPF achieve damage reductions in almost all counties when used in local-haul tractor-trailers but not in long-haul tractor-trailers, which again traces back to assumptions on tailpipe NO_x emissions.

In **Figure 4.1**, we also include the life cycle air pollution damages of the baseline petroleum pathways. We find systematic differences in the life cycle damages of petroleum pathways across vehicle types. Passenger cars have the lowest damages per VMT of all vehicle types while median per-VMT life cycle damages from SUVs, transit buses, and tractor-trailers are around 1.5 times, 3 times, 5 times, and 10 time larger, respectively. Except BEVs, we find that the majority of counties see larger damages from vehicle operation than from upstream activities for all other fuel pathways.

Figure 4.1 highlights large spatial variations in life cycle air pollution damages of baseline petroleum pathways and in damages reduction from alternative fuel pathways across U.S. counties. Furthermore, **Table C.26** highlights the importance of considering the life cycle scope beyond vehicle operation. Except diesel hybrid-electric tractor-trailers, fewer counties find damage reductions from replacing petroleum fuels in the life cycle scope than only considering vehicle operations. The differences are particularly salient for CNG LDVs, BEVs charged with grid electricity, and LNG long-haul tractor-trailers.

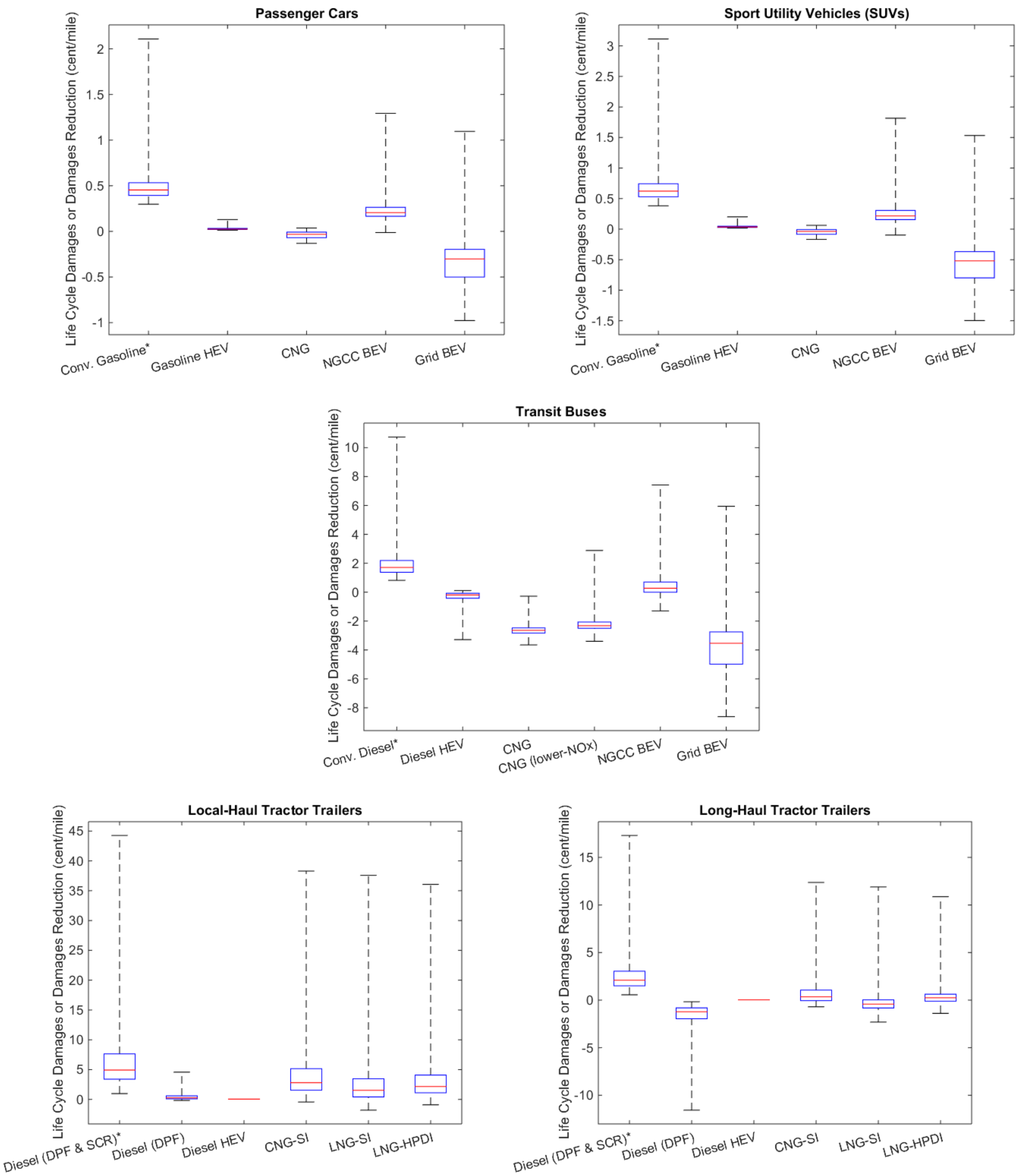
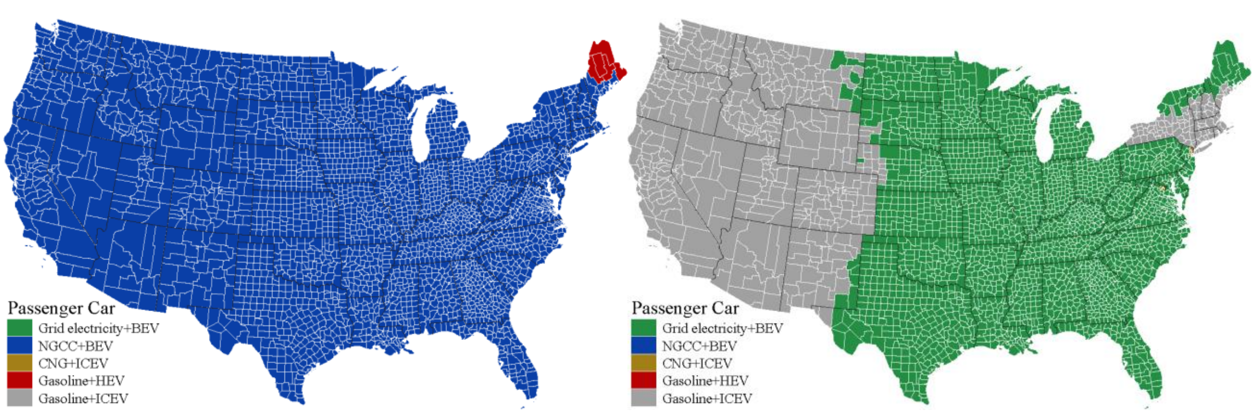


Figure 4.1. Life cycle air pollution damages of baseline petroleum pathways (conventional gasoline for light-duty vehicles, and conventional diesel for heavy-duty vehicles, marked with *), and reduction in life cycle damages of alternative petroleum and natural gas pathways replacing the baseline petroleum pathways. We calculate the damages reduction in each county across the U.S. Negative values in damages reduction suggest lower damages from the alternative pathways compared to the baseline petroleum pathways. Marginal

damage estimates of CAPs come from the EASIUR model. The box plots show the minimum, 25th percentile, median, 75th percentile, and maximum.

4.4.2. Best/Worst Pathway in Each County

Figure 4.2 shows the pathway that achieves the lowest ('best') and highest ('worst') life cycle damages in each county for each vehicle type. BEVs charged with increased penetrations of natural gas-based electricity are the best pathway for LDVs and transit buses in almost all counties. Diesel HEVs are the best bus technology in the Rocky Mountain regions and western Texas. LNG-HPDI is the best pathway for tractor-trailers in West Coast and Rocky Mountain regions, as the electricity damages of the liquefaction process are lower in these regions. CNG-SI and diesel HEVs are the best pathways for tractor-trailers in other parts of U.S. The baseline petroleum fuels are the worst pathway for all vehicle types except transit buses in part of the U.S. (West Coasts for LDVs and HDVs, and eastern U.S. for trucks), suggesting that replacing petroleum fuels with any alternative fuel is likely to reduce life cycle air pollution damages. BEVs charged with grid electricity have the highest damages for much of the eastern U.S. due to significant damages of the grid electricity. New England and western U.S. have cleaner electricity grid so BEVs charged with grid electricity are not the worst pathways in these regions. CNG-SI transit bus is the worst in the western U.S. because of significant tailpipe CO emissions. LNG-SI is the worst in tractor-trailers in parts of the U.S. (western Texas, western Midwest, and Rocky Mountain) because of the shift in tailpipe emissions (more than 90% NO_x reduction compared to diesel buses while increasing CO emissions by a factor of 8) and high electricity damages in the liquefaction process.



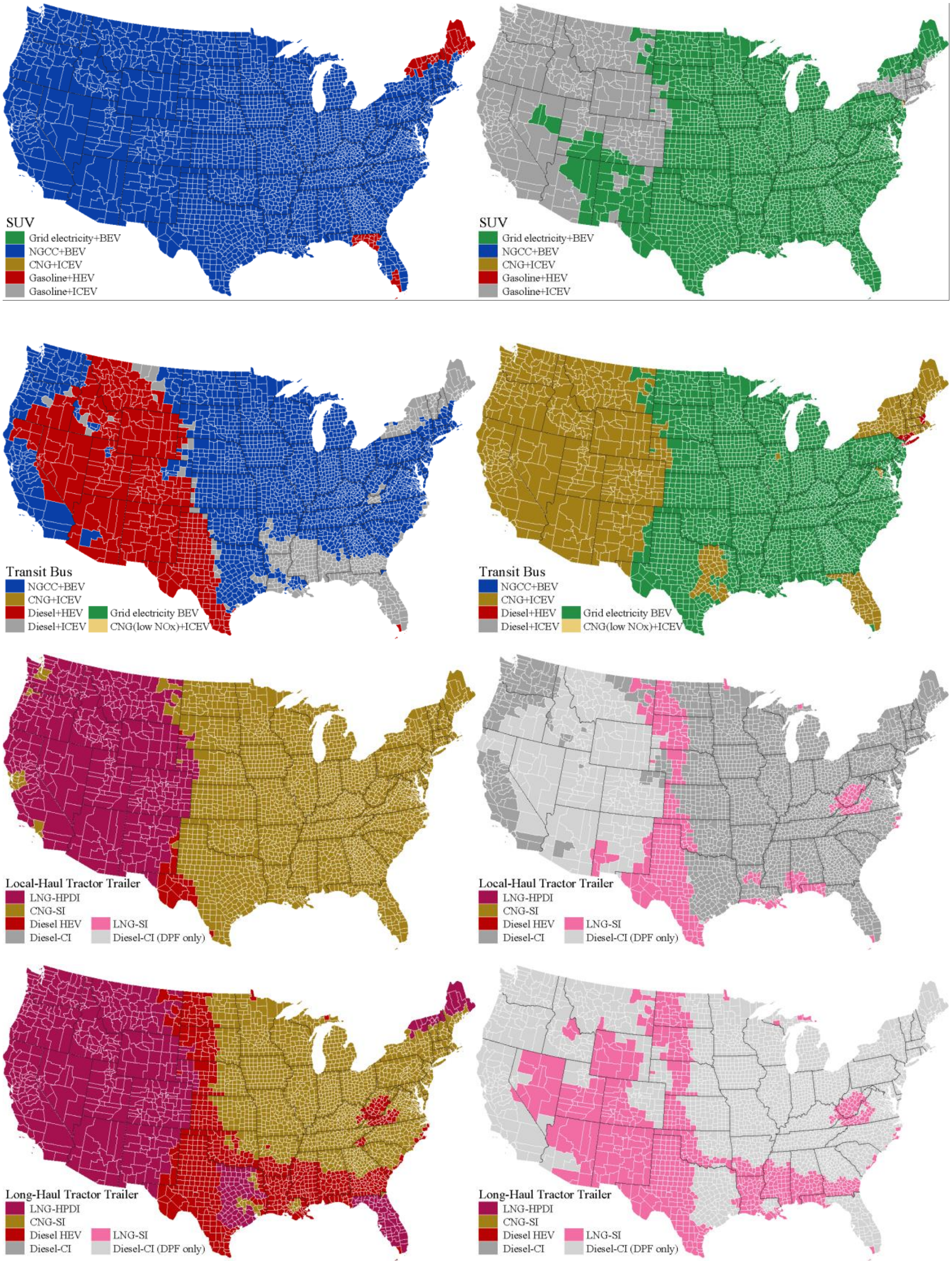

SUV
Grid electricity+BEV
NGCC+BEV
CNG+ICEV
Gasoline+HEV
Gasoline+ICEV
SUV
Grid electricity+BEV
NGCC+BEV
CNG+ICEV
Gasoline+HEV
Gasoline+ICEV
Transit Bus
NGCC+BEV
CNG+ICEV
Diesel+HEV
Grid electricity BEV
Diesel+ICEV
CNG(low NOx)+ICEV
Transit Bus
NGCC+BEV
CNG+ICEV
Diesel+HEV
Grid electricity BEV
Diesel+ICEV
CNG(low NOx)+ICEV
Local-Haul Tractor Trailer
LNG-HPDI
CNG-SI
Diesel HEV
LNG-SI
Diesel-CI
Diesel-CI (DPF only)
Local-Haul Tractor Trailer
LNG-HPDI
CNG-SI
Diesel HEV
LNG-SI
Diesel-CI
Diesel-CI (DPF only)
Long-Haul Tractor Trailer
LNG-HPDI
CNG-SI
Diesel HEV
LNG-SI
Diesel-CI
Diesel-CI (DPF only)
Long-Haul Tractor Trailer
LNG-HPDI
CNG-SI
Diesel HEV
LNG-SI
Diesel-CI
Diesel-CI (DPF only)

Figure 4.2. The best pathway (left panel) and worst pathway (right panel) for each vehicle type in each county. Here the best/worst means achieving the lowest/highest life cycle air pollution damages. Air pollution damages are calculated using the EASIUR model. Similar figures based on the AP2 model are shown in the Appendix C.

4.4.3. Robustness of the Results to the Use of Different Emissions Inventories

As discussed in the Methods section, GREET model's emission factors on primary energy extraction and on oil refinery are higher than U.S. EPA's NEI emissions factors. While the differences on emissions from primary energy extraction do not significantly affect the result, the five-times-larger air pollution damages associated with oil refining increase life cycle damage of petroleum fuels significantly (0.2-2 cents/mile across vehicle types). As petroleum fuels are the baseline fuels in all vehicle types, higher damages from petroleum fuels make all the other pathways more attractive when using the emissions factors from the GREET model. In particular, CNG has lower life cycle damages than gasoline or diesel for LDVs and tractor-trailers in all counties. LNG-SI and LNG-HPDI trucks reduce air pollution damages compared to diesel trucks in almost all counties. However, previous findings on the best/worst pathways remain robust. The best technologies are now solely natural gas pathways - BEVs powered by increased natural gas-based electricity for LDVs and transit buses, and CNG and LNG-HPDI for tractor-trailers while diesel pathways become the worst tractor-trailers technologies in almost all counties.

4.4.4. Robustness of the Results to Different Marginal Damages Models for CAPs

I find notable differences between life cycle air pollution damages using the EASIUR model and the AP2 model. The marginal damages in the AP2 model are generally higher than those in the EASIUR model (likely due to different emissions data used) and significant spatial differences exist between the two models. In particular, the AP2 model reported higher marginal damages on SO_2, which leads to higher life cycle damages in BEVs, CNG and LNG that use grid electricity significantly. Furthermore, the spatial differences in the marginal damages of CAPs by the two model (see Heo et al.[57] for detailed discussions) have key implications on the previous findings in certain regions. For instance, because tailpipe NO_x emissions differ significantly across vehicle technologies, the comparison of life cycle damages across pathways is prone to different marginal damages of NO_x. Indeed, the AP2 model estimated higher (2-10 times) NO_x damages in the Rocky Mountains, but significantly lower damages in New England compared to EASIUR model.[57] As a result, the best and the worst pathways found using the two marginal damage models differ in western Texas, the Rocky Mountain region, the Appalachian Mountain region for transit buses and tractor-trailers, and in New England for all vehicle types (except passenger cars). I find that life cycle damages of CNG and LNG pathways are sensitive to the marginal damage of CO because of high tailpipe CO emissions from stoichiometric sparking-ignition natural gas engines. This study used a marginal damage of CO published more than a decade ago, so an update on the CO's marginal damage may change damages estimates for CNG and LNG pathways.

4.4.5. Comparison between the Results and Those in the Literature

This section compares my results with the literature for both the absolute damage estimates as well as the ranking of fuel pathways. Existing studies[4,26,29,32,186,187,189–192] find the air pollution damages of gasoline passenger cars to be 0.4-2.9 cents/VMT (see **Table 4.1** for details). In this chapter, the median damage estimates for gasoline cars are 0.5-0.7 and 0.6-0.7 cents/VMT (using U.S. EPA's NEI and the GREET model). Tessum et al.[32] and Luk et al.[29] both find that CNG vehicles to have lower air pollution damages than gasoline vehicles but differed in the comparison between CNG vehicles and BEVs with natural gas-based electricity. Tessum et al.[32] find lower damages from BEVs with natural gas-based electricity whereas Luk et al.[29] concludes the opposite. This chapter found that CNG vehicles are more likely to increase rather than reduce air pollution damages in the majority of U.S. counties using U.S. EPA's NEI emissions data. However, CNG vehicles reduce damages in all counties if the GREET model's emissions data is used. I find BEVs charged with increased penetrations of natural gas-based electricity reduces air pollution damages significantly regardless of emissions data and marginal damage model used. Previous studies reached mixed conclusions regarding air pollution damages for battery manufacturing. In particular, Tessum et al.[32]'s battery manufacturing damage estimates are lower than Michalek et al.[26], Luk et al.[32], and Weis et al.[192] I use the estimate from Tessum et al.[32] in this chapter so my BEV results are subject to the same potential bias. Finally, NRC (2010)[4] and this chapter report similar ranges in air pollution damages from diesel heavy-duty trucks.

4.5. Conclusions and Policy Recommendations

This paper provides a comprehensive estimate of life cycle air pollution damages of petroleum and natural gas pathways for LDVs and HDVs in the contiguous U.S. We find that BEVs charged with increased penetrations of natural gas-based electricity achieve the lowest air pollution damages in passenger cars, SUVs, and transit buses, while CNG, LNG-HPDI, and diesel HEVs achieve lowest damages in tractor-trailers (LNG-HPDI in the western U.S, CNG and diesel HEVs in the eastern U.S.). However, not all natural gas-based fuels do not necessarily reduce air pollution damages. In particular, BEVs charged with current grid electricity, CNG transit buses, and LNG-SI tractor-trailers would all lead to the largest air pollution damages in each vehicle type in some U.S. regions.

The findings in this paper suggest a divergent use strategy of using natural gas to power LDVs and heavy-duty freight trucks. For LDVs and transit buses, natural gas is better used in electric power generation than as CNG fuel because of the high energy efficiency and low or zero emissions of NGCC power plants and BEVs. For heavy-duty trucks, cleaner tailpipe emissions from CNG and LNG-HPDI trucks help reduce life cycle air pollution damages. The divergence among natural gas fuels in different vehicle types is due to different vehicle use purposes, physical and technological attributes of natural gas fuels, and market availability of engines and vehicles. We note, however, that a comprehensive decision making on how to power on-road vehicles should include other fuel pathways (e.g. biofuels and renewable energy sources) and other considerations such as economics, consumer behavior, infrastructure needs, and climate change impacts.

We find that natural gas fuel pathways at best partially reduce life cycle air pollution damages when replacing the baseline petroleum fuels. Natural gas fuels alone are not going to get us to the ultimate goal of eliminating air pollution damages. Renewable energy sources, such as wind, solar, biomass, and renewable natural gas from landfills, livestock operations, and wastewater treatment, should be studied to see if they offer deeper or complete reduction in air pollution damages.

Our work highlights the importance of the life cycle perspective when estimating air pollution damages from the transportation sector. In particular, we showed that ranking of natural gas fuel pathways and the baseline petroleum fuels changes when using life cycle scope instead of the vehicle tailpipe scope. Currently, federal and state policies regulates CAP emissions from vehicle operation phase.[209,210] We find that the majority of counties see larger damages from vehicle operation than from upstream activities for fuel pathways other than BEVs. However, with more adoption of BEVs in the U.S.,[211] there will be a shift in air pollution damages across life cycle stages. Systematic analysis like ours is needed to illustrate the comparison of life cycle air pollution damages across pathways and to identify major sources for emissions and damages.

Our work is one of the first to systematically assess how the use of two different emissions data and two different marginal damage models affects life cycle air pollution damages of transportation technologies. We find that our results (especially the best/worst pathway analysis) for the majority of U.S. counties (except the Rocky Mountains, western Texas, and New England) are robust to emissions data and marginal damage models used. We do not have robust findings for some regions (Rocky Mountains, western Texas, and New England) because of systematic spatial differences in the marginal damages of CAP species (in particular, SO_2 and NO_x) in the two models used. Furthermore, we find large discrepancy in the emissions factor of oil refining, which has an impact on the ranking of fuel pathways. We notice that marginal damages of CO and VOCs, for which transportation sector is responsible for more than half and over 25% emissions in the U.S., are relatively poor. We encourage future research to work on these unresolved issues. These limitations notwithstanding, our paper estimates and compares life cycle air pollution damages of natural gas transportation pathways and petroleum fuels. Our results show that the analysis and knowledge of air pollution externalities is essential to achieve transportation sustainability.

Chapter 5. Should We Build A National Infrastructure to Refuel Natural Gas Powered Trucks?

This chapter is based on the working paper: Tong, F.; Azevedo, I. M. L.; Jaramillo, P. *Should We Build A National Infrastructure to Refuel Natural Gas Powered Trucks*; Carnegie Mellon University: Pittsburgh, PA, 2016.

5.1. Abstract

Low natural gas prices offer an opportunity to expand the use of this fuel in new sectors, such as powering trucks in the transportation sector. The use of natural gas for a portion of the heavy-duty fleet would require a large new refueling infrastructure, for which the costs and benefits have not yet been assessed. This study performs an analysis on the economic feasibility of a national natural gas refueling infrastructure for long-haul trucks in U.S. The model prioritizes building refueling stations at highway intersections and ensures highway network coverage. For small shares of the truck fleet powered by natural gas (1-5%), a national natural gas refueling infrastructure requires 92-203 standard refueling modules and a capital investment of $230-508 million. However, at adoption rates lower than 12.5%, building a national natural gas infrastructure may lead to economic losses for baseline economic assumptions. Higher fuel price margin and lower capital cost of a refueling module have large impacts on the economic viability of the refueling infrastructure investment while discount rate and infrastructure lifetime are less important. If the goal is to meet the same refueling demand, a regional refueling infrastructure that covers California and Texas has better economic returns than a system covering the entire country. Technology innovation or policy actions that reduce capital costs of refueling modules or increase fuel price margin improve the economic viability of the refueling infrastructure.

5.2. Introduction

Trucks form the backbone of the U.S. freight transportation system. In 2012, trucks in the U.S. transported more than 12.5 billion tons of goods, or about three quarters of all good shipments across the U.S.[212] The U.S. Department of Transportation (DOT) projects a more-than-40% increase in shipment weight by trucks between 2012 and 2040.[212] While heavy-duty trucks and other heavy-duty vehicles represent less than 5% of registered on-road vehicles in the U.S., they account for 24% of energy consumption and 25% of greenhouse gas (GHG) emissions associated with on-road vehicles.[3,211] As a result, the U.S. Environmental Protection Agency (EPA) and the U.S. National Highway Traffic Safety Administration (NHTSA) have jointly issued regulations to reduce fuel consumptions and GHG emissions from medium- and heavy-duty vehicles (MHDVs) by 2018. With modest compliance burdens, U.S. EPA and U.S. NHTSA expect fuel saving to be six times higher than the costs of compliance, as well sizable environmental benefits from mitigated GHG emissions, reduced ambient concentrations of particulate matter and ground-level ozone, and improved energy security.[103] While the agencies expect the majority of fuel reductions will come from efficiency-improving technologies such as improved aerodynamics, low-resistance tires, vehicle weight reduction, extended engine idling technologies, and reduction of vehicle speeds, alternative fuels, including natural gas fuels and biodiesel, could also play a role.[103]

The successful development of shale gas resources has not only increased natural gas supply dramatically, but also lowered and stabilized natural gas prices. In addition to price advantages, natural gas fuels, such as liquefied natural gas (LNG) and compressed natural gas (CNG), have lower tailpipe GHG and air pollutant emissions than diesel fuels,[146,198,208] further increasing their attractiveness to the trucking industry that faces stringent environmental regulations. Recognizing these benefits, industry has invested in natural gas fuels. UPS has purchased hundreds of LNG-powered trucks.[213] Vehicle manufacturers such as Cummings, Westport, and Volvo have either developed or are developing new natural gas engines for heavy-duty trucks.[70] Fuel suppliers such as Clean Energy, BluLNG, and Shell have started building CNG or LNG refueling stations along major freight corridors and at key freight distribution centers.[63] Furthermore, the U.S. Energy Information Administration (EIA) and energy consulting firms all forecast a significant penetration of CNG and LNG in heavy-duty trucks in the next two decades.[8,109,112] An expanded use of natural gas for heavy-duty vehicles would require a nationwide refueling infrastructure. Indeed, there are only 110 LNG refueling stations and 1,039 CNG refueling stations in the U.S. and not all of these refueling stations are able to refuel heavy-duty trucks.[214,215] The lack of natural gas refueling stations is obvious in contrast to the approximately 160,000 gasoline stations and 5,000-10,000 diesel stations for heavy-duty trucks in the U.S.[67,216]

A number of existing studies have estimated the capacities and/or locations of alternative fuel refueling infrastructure using two distinct methods. The first group of studies relies on simple metrics such as numbers of vehicles to estimate the number of refueling stations.[217,218] These estimates are easy to use but they are only valid for order-of-magnitude estimation purposes and cannot provide detailed information such as locations and capacities for individual refueling stations. The second group of studies uses mathematical models such as mixed-integer linear programming (MILP) to optimize capacities and locations of refueling stations. Depending on data availability and how to model refueling demands, three types of MILP models - median-based models (such as p-median, fixed charge models), covering-based models (such as set covering, max covering, p-center), and flow capturing models - can be used.[219,220] Compared to the first group, these studies are able to provide more detailed estimates but their estimates come at the cost of increased data needs (road network distances and topology, vehicle flows or origin-destination matrix) and increased computational complexity that limits their use for complex road networks. A particular type of MILP models, the flow-capturing refueling model (FRLM) proposed by Kuby et al.[221] and its variations have been used to optimize alternative fuel refueling infrastructure for highway networks in Orlando, FL,[220] Arizona,[222] Florida,[223] and Pennsylvania,[224] and in European countries.[225] Still, few studies have rigorously examined what a national alternative fuel refueling infrastructure means and how much economic benefits or costs it entails.

This chapter contributes to the body of literature by conducting an economic feasibility analysis of a national natural gas refueling infrastructure for long-haul trucks. This chapter develops a refueling model that prioritizes building refueling stations at highway intersections and ensures highway network coverage. This chapter applies the model to determine the locations and capacities of a natural gas refueling network for long-haul trucks in U.S. This chapter also estimates economic costs and benefits from the perspective of refueling infrastructure owners.

5.3. Methods

5.3.1. Overview

I develop a model to assess the locations, capacities, and costs of refueling infrastructure for long-haul trucks powered by natural gas fuels in major U.S. highways. The model has three characteristics: (1) it relies on truck flow estimates; (2) it ensures refueling coverage of any truck traveling within the selected highway network no matter where it starts and ends; (3) it prioritizes building refueling stations at highway intersections.

I consider two scenarios: (i) a national scenario where the infrastructure is built in the entire U.S., and (ii) a regional scenario which focuses solely on Texas and California. I include a separate scenario for California and Texas as both states have high truck flows as well as state incentives for the deployment of natural gas trucks.[226] Furthermore, natural gas in both states enjoy larger price advantages over diesel, thus motiving the conversion of diesel trucks to natural gas trucks.[227,228]

I model refueling demands of natural gas fuels based on the existing truck flows in U.S. highways, and adoption rates of natural gas truck flows. Specifically, the adoption rate is defined as the percentage of annual average truck flows that run on natural gas fuels (CNG or LNG). If a natural gas truck has the same average annual vehicle mile traveled (VMT) as a diesel truck, then the adoption rate also represents the share of natural gas trucks in the total truck fleet. Furthermore, I assume a uniform adoption of natural gas trucks in the highway network. For both the national and regional scenarios I vary the adoption rate parametrically from 1% to 100% to assess how refueling infrastructure corresponds to different levels of refueling demands. **Figure 5.1** shows the flow diagram of the refueling infrastructure model. The model has three major steps. The first step is to decide the boundary of the highway network and prepare the truck flow data. The second step is to determine the locations and capacities of refueling stations. Finally, the last step is to examine the economics of the refueling infrastructure. Before detailing the refueling infrastructure model, I discuss the assumptions natural gas trucks and refueling stations.

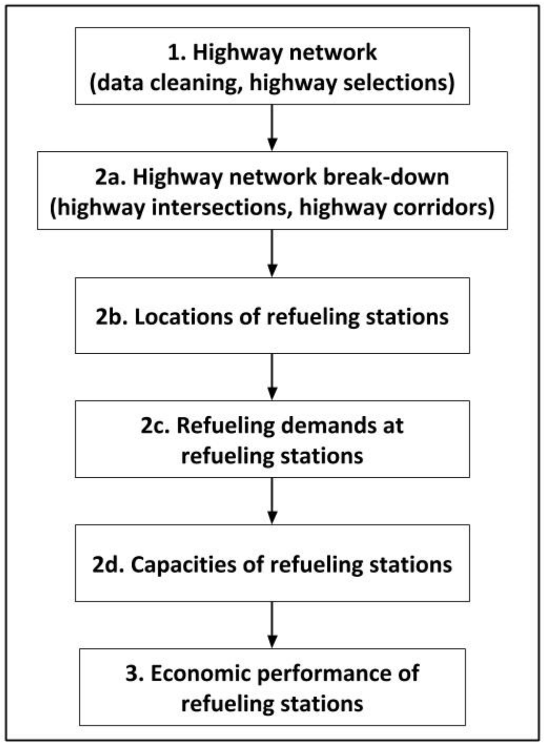


Figure 5.1. Flow diagram of the refueling infrastructure model used in this study.

5.3.2. Truck Characteristics

I assume long-haul trucks account for all the "long-distance truck flows" estimated in the Freight Analysis Framework version 3 (FAF3). According to **Table 5.1**, natural gas trucks have a vehicle range (with a full tank of fuels) of 300-700 miles, depending on fuel tank specifications. The actual distance between two stops in a truck trip, however, may be less than the vehicle range due to factors such as trip planning, and legal regulations on driving hours of truck drivers.[229] I thus assume the range of the trucks either 200, 300, or 600 miles and test the effect of these assumptions in the sensitivity analysis.

Table 5.1. Truck specifications (adopted from *Chapter 3*, Tong et al.[146] and Deal et al.[64]).

Fuel	**Diesel**	**CNG**		**LNG**	
Engine technology	Compression Ignition (CI)	Sparking Ignition		High-Pressure Direct Ignition (HPDI)	
Fuel economy (MPGde)	5.8	5.3		5.7*	
Fuel tank (DGE)	150	80	140	60	120
Vehicle range (miles)**	870	335	670	340	680
Payload (lbs.)	44,450	43,748	42,408	43,202	41,910

Price premium ($ premium relative to diesel)	0	35,000	62,000	70,000	89,000

*Note: * LNG-HPDI engine uses diesel to ignite the compression ignition, so only 95% of fuel consumption is LNG. ** Vehicle range = fuel economy × fuel tank × fraction of usable fuel. The fraction of usable fuel for LNG and CNG are assumed to be 0.95 and 0.9.*

5.3.3. Refueling Station Characteristics

I assume refueling stations are built in standardized refueling modules, so the capacity distribution of refueling stations is not continuous but discrete. I assume each LNG refueling module has two refueling lanes with two dispensers (each of which refuels 15 diesel gallon equivalent (DGE) per minute), a 15,000 gallon LNG storage tank, and other supporting equipment.[224] The maximum capacity of a refueling module is 7.3 million DGEs/year assuming continuous refueling at both refueling lanes for 700 minutes per day. The capital cost of a refueling model is $2.5 million.[224]

5.3.4. Highway Network and Truck Flow Estimates

I use truck flow estimates from the FAF3 developed by the U.S. DOT.[230] The most recent version, FAF3 estimates freight flows based on the 2007 U.S. Commodity Flow Survey, trade freight flows, and other data sources, and then allocates freight flows to truck movement in the U.S. road networks.[230] The original FAF database models freight trucks on all types of roads in the U.S. The use of natural gas fuels is unlikely to happen in all of these roads. It is also computationally inefficient to consider all roads in the U.S. I thus chose a reduced boundary of the national highway network to include a selection of interstate highways and a state highway (California S99) in U.S. The selection criteria include the ranking of weighted-average truck flow and coverage of major cities and freight centers. **Figure 5.2** shows the FAF truck flows on the selected highway network.

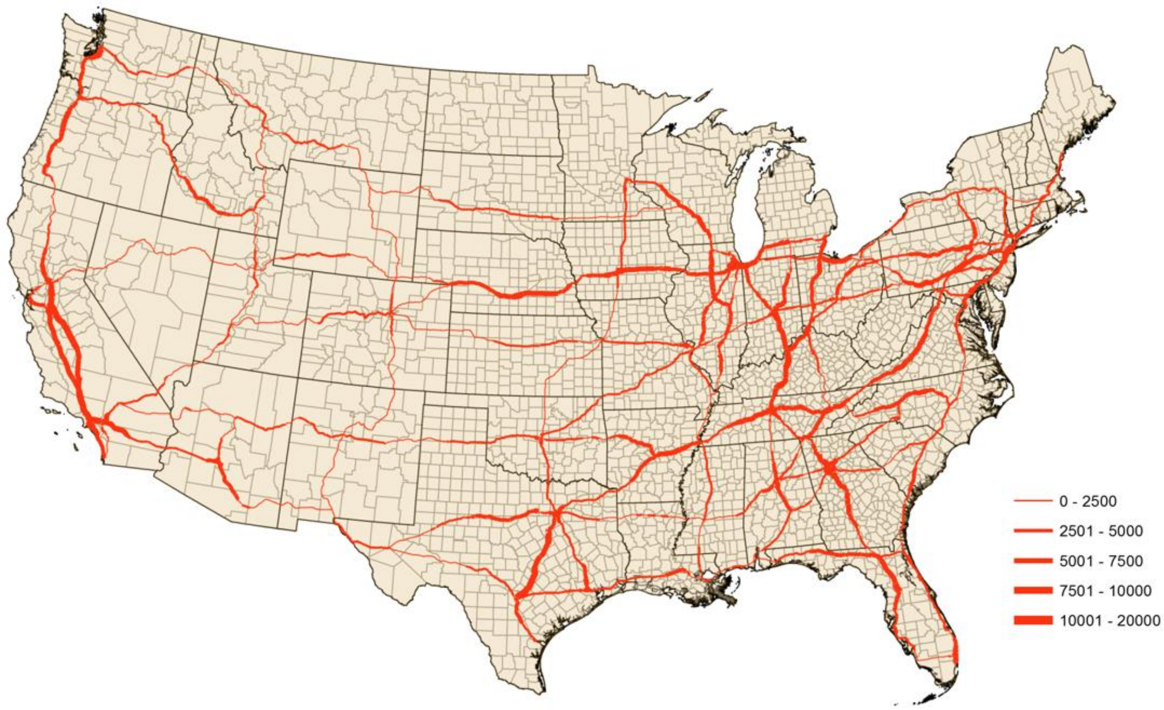


Figure 5.2. U.S. truck flows in selected interstate and state highways for year 2007. Unit: number of trucks per day. The truck flow estimates come from FAF3.[230]

FAF data reports the number of long-haul trucks per day (for the annual average in 2007) for any road segment across U.S. road networks. A road segment is defined in the FAF database and has a variable road length on the order of 0.1-10 miles. I convert FAF truck flows into vehicle mile traveled (VMT) estimates for any road segment (Eqn. 5.1). With fuel economy assumptions of diesel trucks (**Table 5.1**), I further estimate fuel consumption of long-haul trucks for all road segments across the U.S. (Eqn. 5.2). Since FAF3 is a snapshot of highway truck flows in 2007, I assume it does not include natural gas trucks. Finally, Eqn. 5.3 calculates fuel consumption of natural gas trucks for any road segment at a given adoption rate.

$$VMT_{road\ segment\ s} = Number\ of\ trucks_s \times road\ length_s \tag{5.1}$$

$$fuel\ consumption_{road\ segment\ s,\ diesel} = VMT_s / fuel\ economy_{diesel} \tag{5.2}$$

$$fuel\ consumption_{road\ segment\ s,\ NG\ fuels} = adoption\ rate_{NG} \times VMT_s / fuel\ economy_{NG} \tag{5.3}$$

Table 5.2 shows the estimates of VMTs and fuel consumption of long-distance trucks. The only available estimate on VMTs and fuel consumption for trucks in the Transportation Energy Data Book (TEDB) from the Oakridge National Laboratory.[3] The TEDB estimates are 3.5 times higher than my estimates because of different scopes considered. For instance, the truck flows estimated in the FAF database only include long-distance trucks; the selected highways only cover 57% of truck flows in the FAF3; the FAF3 data was for 2007 while the TEDB data was year 2013.

Table 5.2. VMT and fuel consumption estimates in this study and other sources.

Variable	**Transportation Energy Data Book**[3]	**FAF3**[230]	**This study**
Vehicle type	Class 7-8 combination trucks	long-distance trucks	
Year	2013	2007	
Road distance (1000 miles)	1100 (urban) + 2600 (public paved roads)[231]	448	30
Number of trucks (million)	2.5	1.2**	0.7**
VMT (billion miles)	163	81	46
Fuel consumption (billion diesel gallon equivalent)*	28	14	8

*Note: * Assuming the only fuel consumed by the long-distance truck fleets is diesel. Fuel consumption is back-calculated using the fuel economy of an average diesel Class 7-8 combination truck, which is 5.8 mile per gallon (MPG).[3] ** Assuming the average VMT per truck remains the same.*

5.3.5. Refueling Infrastructure Model

The refueling infrastructure model takes refueling demands of natural gas trucks as the input and determines the locations and capacities of natural gas refueling stations. For simplicity, I assume refueling stations refuel trucks moving in both directions. The siting of refueling stations should ensure that any truck with any origin and any destination within the selected highway network has enough fuel at any point during its trip. In other words, the maximum road distance between two adjacent refueling stations should be less than the range of a truck. The model prioritizes building a refueling station at every major highway intersection to capture high refueling demands. Existing research found that refueling stations at road intersections see significantly larger vehicle visits than other refueling stations.[232] As a result, the model determines the locations of refueling stations in two steps. First, it builds refueling stations at highway intersections. I call the portions of highways defined by two adjacent intersections *corridors*. Second, if the length of a corridor is longer than the range of a truck, a minimum number of equally spaced refueling stations are built within the corridor to make sure the nearest distance between two refueling stations is no larger than the vehicle range. For instance, if a corridor has a length of 900 miles and the truck range is 300 miles, then two refueling stations are built in the corridor at a distance of 300 miles from each end point of the corridor.

The model then proceeds to determine capacities of refueling stations. I assume that trucks refuel at every refueling station along their trips until they reach the destination, and trucks refuel the same amount of fuel consumed since their last refuel. I assume trucks have a full tank when they enter the highway network. The FAF3 truck flow data aggregates truck flows over moving directions. Without further information, I assume symmetrical truck flows in two directions. Finally, Eqn. (5.4) calculates the refueling demands for any refueling stations.

$$\begin{aligned} &\textit{Refueling demand}_{\textit{refueling station } r} \\ &= \frac{1}{2} \times \sum_{\substack{\textit{any refueling station } r' \textit{ that have} \\ \textit{a highway connection with } r}} \; \sum_{\substack{\textit{any highway segment } s \textit{ that} \\ \textit{falls between } r' \textit{ and } r.}} \textit{fuel consumptions}_{s,\, \textit{NG fuels}} \end{aligned} \tag{5.4}$$

Where the first summation sums up refueling demands from truck flows in all highways that intersect at refueling station *r*, the second summation sums up refueling demands between refueling station *r* and one of its adjacent refueling station, and “1/2” accounts for the driving direction of truck flows. The capacity of a refueling station is chosen as the minimum number of refueling modules that have a total maximum capacity higher than its refueling demand (Eqn. 5.5). I find that all high-demand refueling stations are located at highway intersections, which confirms my hypothesis (see Appendix D for details).

$$\begin{aligned} &\textit{Refueling capacity}_{\textit{refueling station } r} \\ &= \left\lceil \frac{\textit{Refueling demand}_r}{\textit{Capacity of a refueling module}} \right\rceil \times \textit{Capacity of a refueling module} \end{aligned} \tag{5.5}$$

Where $\lceil \; \rceil$ is the ceiling function, and defined as $\lceil x \rceil = \min\{n \in \mathbb{Z} \mid \mathrm{n} \geq x\}$ and the capacity of a refueling module is 7.3 million DGEs/year, as previously discussed.

5.3.6. Economic Analysis

I perform the economic analysis from the perspective of refueling infrastructure owners. I treat all natural gas refueling stations as though they are owned by one single owner. This situation will not happen in reality because of anti-trust law, but it provides an overall account of the economic viability of natural gas refueling infrastructure. I use Eqn. 5.6-5.9 to calculate the capital investment, net present value (NPV) of the net profit, and internal rate of return (IRR). **Table 5.3** lists key assumptions used in the economic analysis. In addition to baseline values, I include wide sensitivity ranges for each assumption to analyze their effect on the economic viability of the refueling infrastructure.

$$\begin{aligned} &Capital\ investment = \sum_{all\ refueling\ stations} capital\ cost\ of\ a\ refueling\ station \\ &= number\ of\ refueling\ modules * capital\ cost\ of\ a\ refueling\ module \end{aligned} \tag{5.6}$$

$$\begin{aligned} &Net\ profit\ NPV \\ &= capital\ investment + \sum_{t=1}^{lifetime} \sum_{refueling\ station\ i} fuel\ sale\ profits_{i,\ year\ t} \times (1 + discount\ rate)^{-t} \\ &= capital\ investment + \sum_{t=1}^{lifetime} \sum_{refueling\ station\ i} fuel\ sales_{i,t} \times fuel\ price\ margin \times (1 + discount\ rate)^{-t} \\ &= capital\ investment + Annuity\ factor_{lifetime,discount\ rate} \times \sum_{refueling\ station\ i} fuel\ sales_{i,t} \times fuel\ price\ margin \end{aligned} \tag{5.7}$$

$$Annuity\ factor_{lifetime,discount\ rate} = \frac{1 - (1 + discount\ rate)^{-lifetime}}{discount\ rate} \tag{5.8}$$

$$Internal\ rate\ of\ return\ (IRR) = the\ discount\ rate\ that\ results\ in\ a\ net\ profit\ NPV\ of\ zero. \tag{5.9}$$

Table 5.3. Key economic assumptions.

Variable	Baseline value	Sensitivity range
Discount rate	7%[233]	3%-11%
Fuel price margin (unit markup less operation and maintenance costs for refueling stations)	\$0.2/DGE[7]	\$0.1-1.0/DGE
Lifetime	20 years[7]	10-30 years
Natural gas refueling module capital cost	\$2.5 million[224]	\$0.5-4.5 million

5.3.7. The Differences between CNG and LNG

The model works for either CNG or LNG refueling infrastructure, whose major differences are truck range, capital cost of a refueling station, and fuel price margin. Compared to the LNG counterpart, a typical CNG truck has a lower range (**Table 5.1**) while a typical CNG refueling station has a higher fuel price margin because of lower costs to produce CNG than LNG (see Appendix D for details),[7,234] and a lower capital cost for a smaller capacity.[235] While the baseline assumptions of the model are specific for LNG trucks and LNG refueling stations, I expect the sensitivity analysis covers the CNG case as well.

5.4. Results

5.4.1. Locations and Capacities of Refueling Stations

The locations of refueling stations are determined by highway topology and vehicle range. The selected national highway network has 86 highway intersections, so there are 86 refueling stations at intersections. The number of refueling stations built within highway corridors depends on vehicle range: if the assumed truck range is 600 miles, only six more refueling station sites are chosen to ensure corridor coverage, but if the truck range is reduced to 200 miles, then 80 more refueling stations have to be built (**Table 5.4**).

I find that the capacity of the refueling infrastructure (i.e. number of refueling modules) is largely driven by the portion of long-haul trucks powered by natural gas fuels (**Figure 5.3**). When the share of natural gas trucks is higher than 5%, the total refueling capacity grows almost linearly with respect to the adoption rate. However, if the share of natural gas trucks is less than 5%, the total refueling capacity needed is fairly constant. Specifically, 92-106, 124-130, and 166-168 refueling modules are needed if the range of natural gas truck is 200 miles, 300 miles, or 600 miles. The relative constant number of refueling stations when adoption is less than 5% is a result of the discrete sizes for refueling models and the assumption of network refueling coverage. This suggests that at lower adoption rates (i.e. 1%) the refueling stations built would have excess capacity. Furthermore, the lower the trucks' vehicle range is, the more severe the overcapacity issue is at lower adoption rates. Finally, the impact of vehicle range on the total refueling capacity becomes less salient at high adoption rates than at low adoption rates.

Faced with a very low adoption rate of natural gas trucks (1%), a national natural gas refueling infrastructure would require only 92-166 refueling modules, depending on the vehicle range assumption (see **Figure 5.3** and **Table 5.4**). On the other hand, a complete conversion of existing truck fleet to natural gas trucks would require a natural gas refueling infrastructure consisting of 1,231-1,263 refueling modules, accounting for vehicle range assumptions from 200 miles to 600 miles. To put these numbers in context, there are about 5,000-10,000 heavy-duty diesel refueling stations on the interstate highways.[67,216] The number of existing diesel refueling stations is significantly larger than the model's estimate at 100% adoption rate because the model does not account for competitive market behavior of refueling infrastructure investors.

In **Figure 5.3** and **Table 5.4**, I report the estimated locations and capacities of a regional natural gas refueling infrastructure in California and Texas. Because these two states have much smaller and relatively denser highway networks than the entire U.S., fewer refueling station sites are chosen – 10-14 refueling stations depending on vehicle range assumptions. However, 12% of the U.S. total trucks' VMTs take place in California and Texas. As a result, to serve the same amount of VMTs driven by natural gas trucks (e.g. 1.2% of the total truck VMT), a national refueling infrastructure requires building 92-166 fueling stations (for a 1.2% adoption rate), while a regional refueling infrastructure in California and Texas would require only 16-18 refueling stations (for a 10% regional adoption rate) at different vehicle range assumptions (200-600 miles). This implies that building refueling stations in high-demand regions such as California and Texas would have a higher benefit-cost ratio than building a national refueling infrastructure.

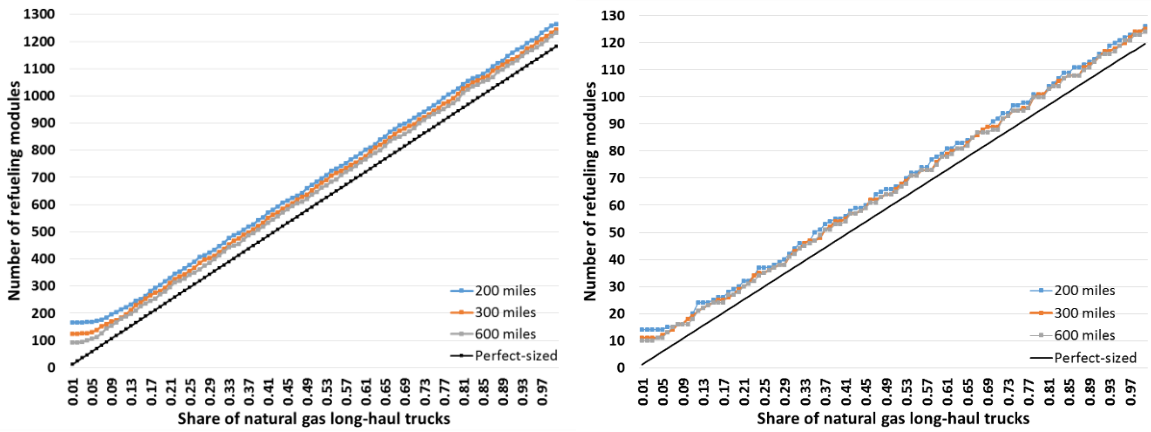


Figure 5.3. Number of refueling modules as functions of adoption rate and vehicle range for the national (left) and the regional (right) scenarios. 'Perfect-sized' scenario is calculated as a ratio between total refueling demands and maximal refueling capacity of a refueling module. Thus perfect-sized scenario calculates the lower bound on the number of refueling modules.

Table 5.4. Number of refueling stations to serve 1%, 5% and 10% of the total truck fleet in U.S., and in California and Texas.

Geographic scope and vehicle range		Number of refueling station sites			Number of refueling modules (adoption rate: 1%/5%/10%)		
		At intersections	Within corridors	Total	At intersections	Within corridors	Total
U.S. national	200 miles	86	80	168	86/88/118	80/80/85	166/168/203
	300 miles		38	124	86/92/130	38/38/43	124/130/173
	600 miles		6	92	86/100/157	6/6/8	92/106/165
California & Texas	200 miles	10	4	14	10/10/13	4/4/5	14/14/18
	300 miles		1	11	10/11/16	1/1/2	11/12/18
	600 miles		0	10	10/11/16	0/0/0	10/11/16

Note that according to ***Table 5.2****, 10% penetrate rate corresponds to 70 thousand trucks, 4.6 billion miles per year, and 0.8 billion DGEs of fuels per year.*

5.4.2. Economic Analysis

For 1-10% national adoption rates of natural gas trucks, the initial investment on the refueling infrastructure ranges from $230 million to $508 million (for 92-203 refueling modules). However, to capture 1% of the nationwide VMT, the capital investment of developing a refueling infrastructure in California and Texas is an order-of-magnitude lower ($40 to $45 million) than a national LNG refueling system. Jaffe et al. estimated that "the costs to provide dedicated coverage for LNG across California are estimated to be less than $100 million",[227] which is in the same order of magnitude as this study's estimate. However the authors did not state the highway coverage or adoption levels for LNG trucks, so a more detailed comparison is not possible.[227] The National Petroleum Council estimates that an investment of $10 to $20 billion (2008$) would be needed for a national LNG refueling infrastructure to replace one third of the diesel fuel market.[7] For that level of natural gas truck adoption, my estimates range from $1.1 to $1.2 billion. As the NPC study did not disclose how they estimated the number of refueling stations, a detailed discussion is not possible. Furthermore, these figures can be compared with the investment in the gasoline and diesel refueling infrastructure. Following Melaina et al.'s estimates on remodeling costs of gasoline refueling stations,[236] I estimate that annual remodeling investments to maintain existing diesel refueling stations are $150 million. In comparison, the capital costs to build a national natural gas refueling infrastructure for 1-10% adoption rates are 1.5-3.4 times higher.

While the capital costs of building a natural gas refueling infrastructure are relatively low (for 1-10% adoption rates), the net profits (NPV) of the refueling infrastructure are more likely to be negative for low adoption rates. For the baseline economic assumptions (**Table 5.3**), the net present value (NPV) of all refueling stations at 5% adoption rate range from -$192 million to -$37 million (for different vehicle range assumptions). At a 10% adoption rate, the NPV ranges from -$51 million to $44 million. At adoption rates higher than 12.5%, the baseline estimates show positive NPVs for all baseline assumptions.

Figure 5.4 and **Figure 5.5** provide the estimates of NPV of the refueling infrastructure for a variety of sensitivity scenarios on economic variables. Each figure displays a sensitivity analysis by showing the trade-offs between a pair of variables (refueling module capital cost, fuel price margin, discount rate, and lifetime) at an adoption rate (5% or 15% of the truck fleet powered by natural gas fuels). While the baseline NPV show different signs (negative and positive) for a 5% or a 15% adoption rate, the effects of economic variables on the economic returns of the refueling infrastructure remain similar. Scenarios with either low fuel price margins, high capital costs of a refueling module, or high discount rates, result in net economic losses. The NPV of the refueling infrastructure investment is more sensitive to fuel price margin and capital cost of a refueling module than discount rate or lifetime, as shown by the slopes of the contour lines. Lifetime alone does not have a large effect on the economic viability of the refueling infrastructure.

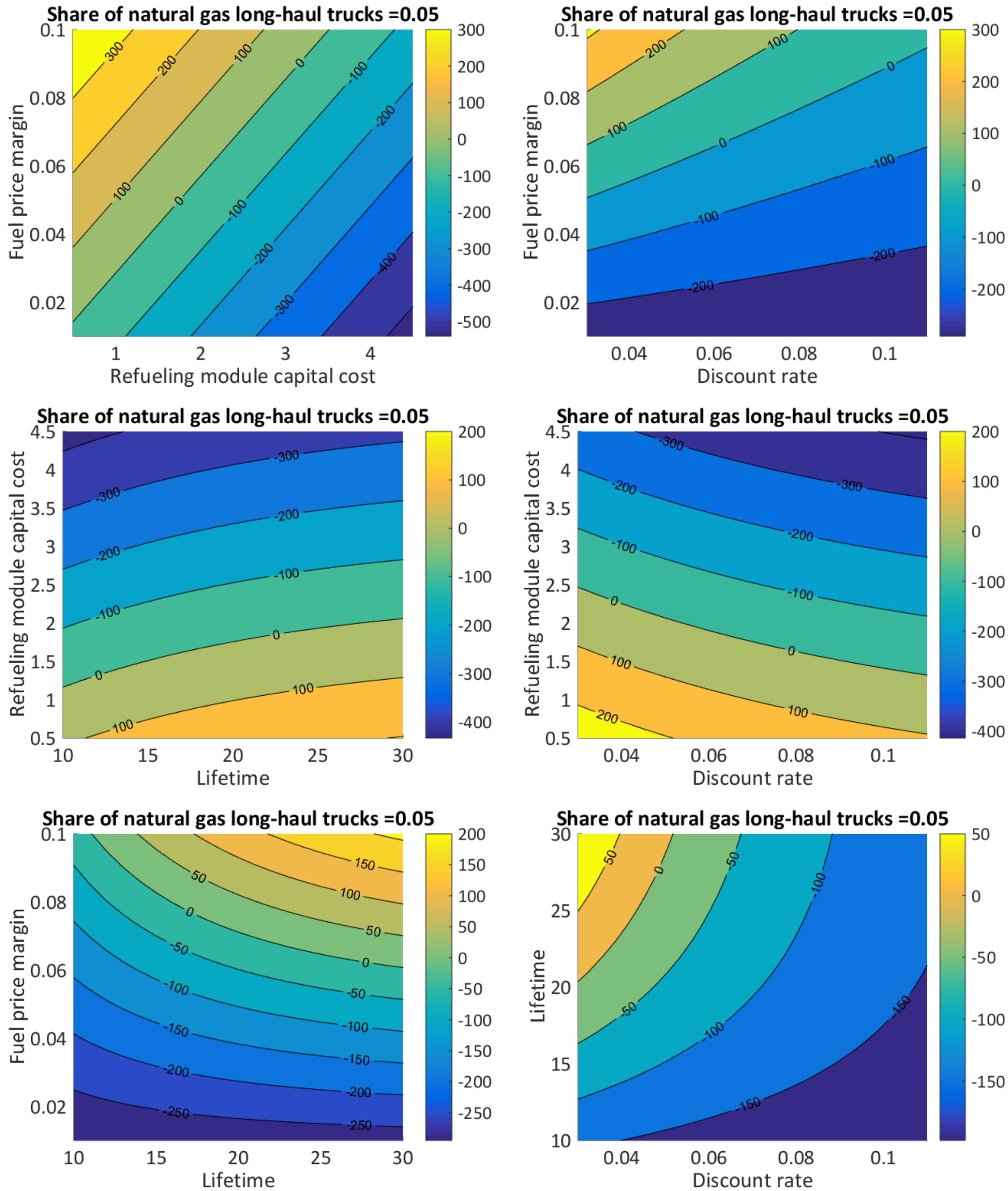


Figure 5.4. Net present value of net profits (in million $) of natural gas refueling stations for 5% share of long-haul trucks in the national scenario at various values of refueling module capital cost, discount rate, fuel price margin, and lifetime. A 300-mile vehicle range is assumed. Baseline values for these assumptions are shown in Table 5.3. Contour lines (that achieve the same net profits) are marked. One color is filled in each interval of the net profits. The same color may represent different net profits in different figures.

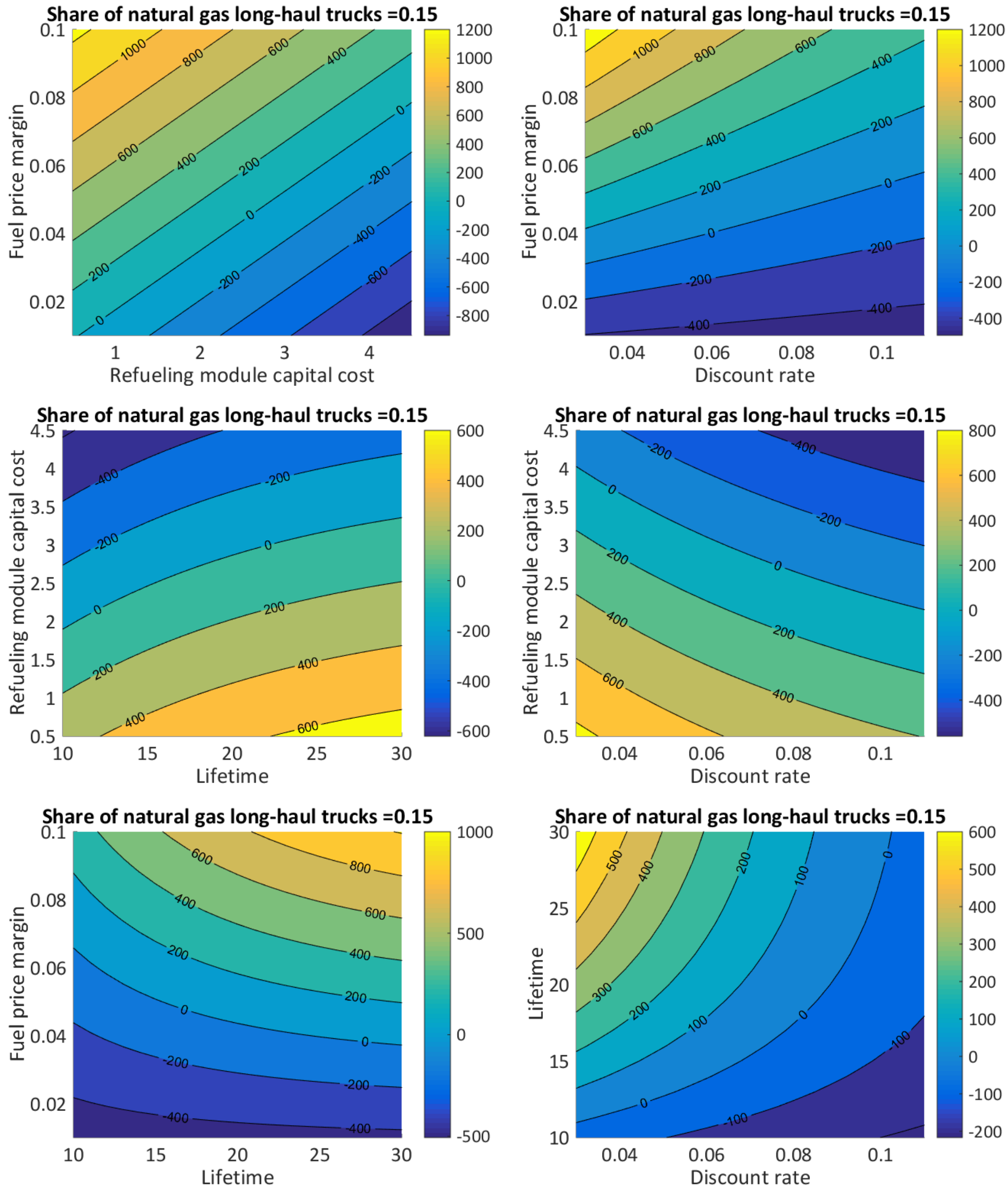
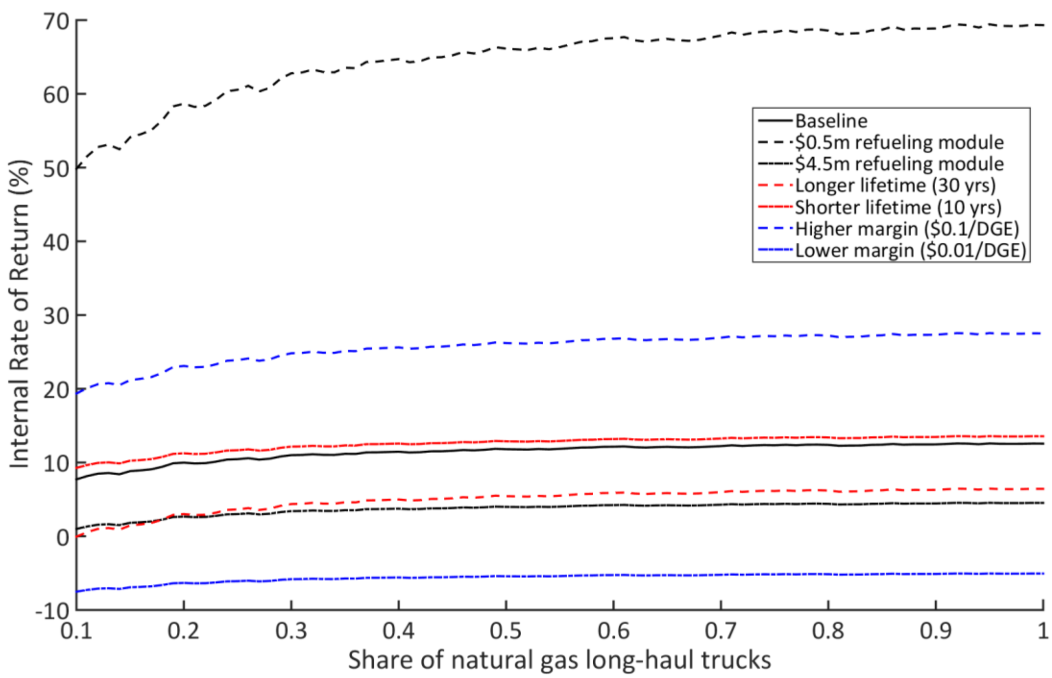


Figure 5.5. Net present value of net profits (in million \$) of natural gas refueling stations for 15% share of long-haul trucks in the national scenario at various values of refueling module capital cost, discount rate, fuel price margin, and lifetime. A 300-mile vehicle range is assumed. Baseline values for these assumptions are shown in Table 5.3. Contour lines (that achieve the same net profits) are marked. One color is filled in each interval of the net profits. The same color may represent different net profits in different figures.

Finally, I estimate the internal rate of return (IRR) of the refueling infrastructure for different adoption rates and for the national and regional scenarios (**Figure 5.6**). I consider the baseline economic assumptions as well as sensitivity cases where one economic assumption (refueling module capital cost, lifetime, and fuel price margin) is changed upwards or downwards. **Figure 5.6** shows that the IRRs are about 10% for all adoption rates when the baseline economic assumptions and a 300-mile vehicle range are assumed. The IRR curve is slightly non-linear with respect to low adoption rates. Again this shows the negative effect of building over-capacity at very low adoption rates. Assumptions on fuel price margin and capital cost of a refueling module shift the IRR curve significantly. In particular, a low fuel price margin would result in negative economic returns for the refueling infrastructure for all adoptions rates for both the national and the regional scope. On the other hand, lifetime of the refueling infrastructure has a smaller impact compared to fuel price margin and capital cost of a refueling module.



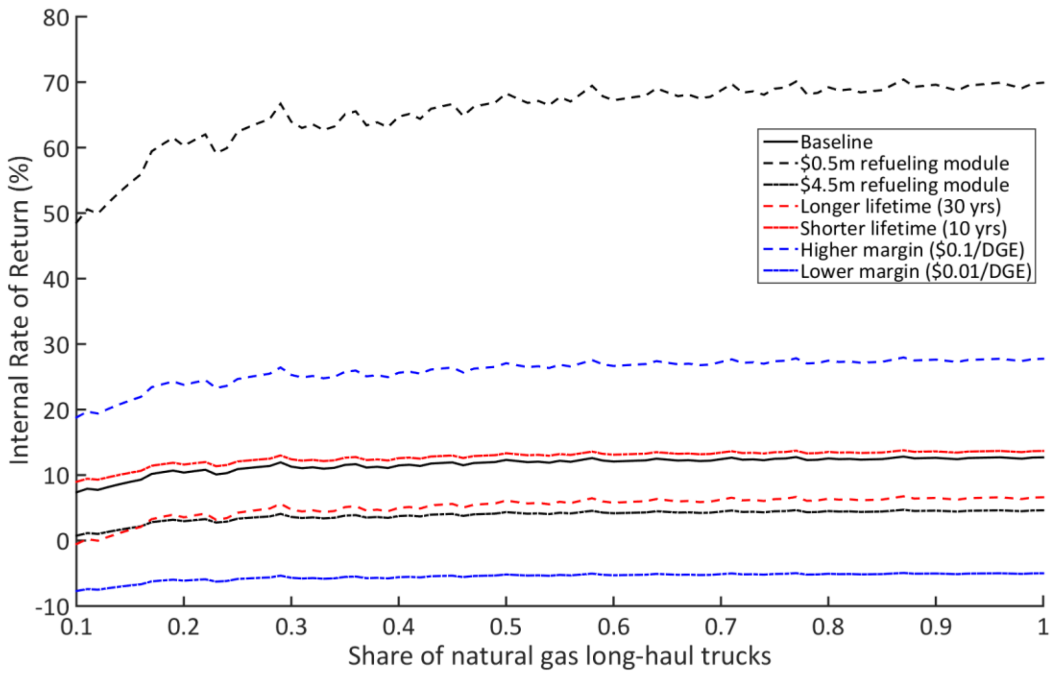


Figure 5.6. Internal rate of return (IRR) of natural gas refueling stations at different adoption rates for the national scenario (top) and the regional scenario (below). Baseline assumptions in Table 5.3 are used unless otherwise stated. A 300-mile vehicle range is assumed.

5.5. Conclusions and Policy Recommendations

This chapter develops a refueling infrastructure model to determine the locations, capacities, and economic performance of a natural gas refueling network for long-haul trucks in U.S. I find that the total refueling capacity needed is relatively constant when the share of natural gas trucks is 1-5% of the total long-haul truck fleet, and that for higher shares, the total refueling capacity is almost linear with respect to the share of natural gas trucks. For small shares of the truck fleet powered by natural gas fuels (1-10%), a national natural gas refueling infrastructure would require only 92-203 refueling modules and a capital investment of $230-508 million. However, at adoption rates lower than 12.5%, building a national natural gas infrastructure may lead to economic losses for baseline economic assumptions. Higher fuel price margin and lower capital cost of a refueling module can improve the economic viability of the refueling infrastructure investment.

5.5.1. Modeling Characteristics and Limitations

The refueling infrastructure model proposed in this chapter assumes a perfect coordination between the refueling behavior of natural gas trucks (demand side) and the build-up of the natural gas refueling infrastructure (supply side). In reality, such coordination is hard to achieve because both sides are independent decision makers, each of which faces unique objectives and constraints. At the same time, the trucking industry and the refueling industry have strong interdependencies and they use market tools such as long-term contracts to lock in refueling demands and fuel supplies. Federal and state governments have also stepped in to help overcome market and non-market barriers and encourage the use of natural gas fuels on both the supply and the demand side.[5,7,59]

The refueling infrastructure model prioritizes building refueling stations at highway intersections. While this decision rule may forbid the model to find the optimal solution, it captures market behaviors. Indeed, market force has produced towns (e.g. Breezewood, PA) at highway intersections whose sole existence is to provide highway services ranging from refueling stations, restaurants, convenience stores, to motels and hotels.

This work only considers engineering cost estimates for building refueling stations. Other factors, such as land acquisition, supporting infrastructure, and refueling market competition may lead to different refueling infrastructure designs than assumed in this study. Fuel price differences across regions, congestion at refueling stations or truck stops, driving hour regulations, are also ignored in this work and should be the focus of future work.

5.5.2. Economic Viability of the Refueling Infrastructure

Our results show that a transition to natural gas fuels in long-haul trucks requires a high adoption rate (>5%) of natural gas trucks and a high fuel price margin of the natural gas fuels (~$0.1/DGE). Both these factors are highly influenced by the diesel fuel price (see Appendix D for comparison on diesel and natural gas fuel prices). With a high diesel price, truck fleets are more likely to switch to natural gas fuels for economic savings, and refueling infrastructure owners can ask for a higher price margin. However, the current diesel price is at the lowest point since 2005.[237] Further, the adoption of natural gas is still below 3% of the heavy-duty trucks and buses fleet.[214] These facts do not bode well for the economic viability of the existing and new natural gas refueling infrastructure.

However, opportunities of natural gas fuels may still exist in regional markets. Indeed, the analysis on regional refueling infrastructure shows that much fewer refueling stations are needed to ensure highway coverage in California and Texas. Furthermore, regional factors that have not been taken into account yet – such as higher price differential between natural gas fuels and diesel,[227,228] attractive state incentives,[226] and strong environmental regulations – would help a swifter adoption of natural gas fuels in California and Texas. Finally, technology innovation or policy interventions that reduce capital costs of refueling modules or increase fuel price margin improve the economic viability of the refueling infrastructure, especially for low shares of natural gas trucks in the total truck fleet.

5.5.3. Social Costs of Natural Gas Trucks and Refueling Infrastructure

It is interesting to take a look at the social costs of natural gas trucks and refueling infrastructure and see whether a transition to natural gas fuels reduce social costs compared to diesel. Here, social costs include lifetime ownership costs of trucks, economic performance of the refueling infrastructure, and the social costs of air emissions. For the social costs of air emissions, findings in Chapter 2 and Chapter 4 show that natural gas fuels provide limited reduction in air pollution damages in certain counties and may increase GHG emissions at 20-year global warming potential. In addition, social costs of air emissions are small compared to vehicle and infrastructure costs, as shown in Chapter 6 for transit buses. The private costs of the refueling infrastructure are sensitive to fuel price margin, refueling module capital cost, and adoption rate of natural gas trucks. Fuel price margin and adoption rate of natural gas trucks are in turn largely influenced by diesel price. While I have not estimated the ownership costs of natural gas trucks, literature shows that the ownership costs are driven by the diesel price.[7] So putting these items together, we see that at the current diesel price (lowest since 2005),[237] a transition to natural gas fuels are more likely to lead to net economic losses in terms of social costs.

Chapter 6. Life Cycle Ownership and Social Costs of Alternative Fuel Options for Transit Buses

This chapter is based on the working paper: Tong, F.; Hendrickson, C.; Biehler, A.; Jaramillo, P.; Seki, S. *Life Cycle Ownership and Social Costs of Alternative Fuel Options for Transit Buses*; Carnegie Mellon University: Pittsburgh, PA, 2016.

6.1. Abstract

This chapter assesses alternative fuel options for transit buses. I consider the following options for a 40-foot and a 60-foot transit bus: a conventional bus powered by either diesel or a biodiesel blend (B20 or B100), a diesel hybrid-electric bus, a sparking-ignition bus powered by Compressed Natural Gas (CNG) or Liquefied Natural Gas (LNG), and a battery electric bus (BEB) (rapid or slow charging). I estimate life cycle ownership costs (buses and infrastructure) and social costs caused by greenhouse gases (GHGs) and criteria air pollutants (CAPs) emitted from the life cycle of bus operations. Without external funding only rapid-charging BEBs reduce ownership & social costs compared to diesel, while other options increase ownership & social costs. When external funding is available to pay for 80% of vehicle purchase expenditures, BEBs yield large reductions (39-45%) in ownership & social costs compared to diesel. Factors such as annual mileage, diesel price, discount rate, per-bus infrastructure cost, and electricity price, determine BEBs' cost reduction potential without external funding. But when external funding is available, BEBs' advantages are robust. BEBs are able to reduce CAP emissions significantly in Pittsburgh's hotspot areas, where existing bus fleets contribute to 1% of particular matter emissions from mobile sources. There are still practical barriers for BEBs, e.g. range limits, land to build the charging infrastructure, and coordination with utilities. However, favorable trends such as better battery performance and economics, cleaner electricity grid, and more experience likely favor use of BEBs where feasible.

6.2. Introduction

Transit buses provide short-distance public transportation service with multiple stops along fixed routes to serve citizens' mobility needs. Currently, there are 653 transit agencies operating in urbanized areas and 525 transit agencies in rural areas in the U.S.[238] In 2013, these 1,178 transit agencies operated a fleet of 65,950 active buses, which travelled 2.2 billion vehicle miles, and served 19.4 billion passenger miles. Altogether, transit buses consume 79 trillion Btu's of energy, or about 0.4% of energy consumed by on-road vehicles in the U.S.

Alternative fuels and advanced technologies have the potential to reduce petroleum consumption and to mitigate unintended environmental consequences including climate change damages caused by greenhouse gases (GHGs) and health and environmental damages caused by criteria air pollutants (CAPs) by substituting for conventional vehicles powered by petroleum fuels. Transit agencies are more willing, compared to mainstream private vehicle owners, to adopt alternative fuel vehicles. This is not only because they have a different cost structure (fueling costs are more important due to high mileages), but also because they have higher awareness and obligations to funding agencies to pursue fuel diversity and/or environmental sustainability.[5] In the past two decades, there has been an increase in the penetration of alternative fuels in the transit bus market. APTA reported that 20% of U.S. transit buses were powered by compressed natural gas (CNG) and liquefied natural gas (LNG) and blends in 2013. In addition, 13% of transit buses were diesel hybrid electric buses (HEBs) and another 7% used biodiesel. Zero-emission buses, such as battery electric buses (BEBs) and fuel cell electric buses, have also emerged in some regional markets (notably, California), as encouraged by state-level environmental regulations and incentive programs.[239]

There is a growing literature that assesses alternative fuel options for transit buses.[42,48,240–252] **Table 6.1** provides a summary of the scope and conclusions of selected U.S. studies. I find that existing studies estimated lifetime ownership costs of purchasing and operating diesel, diesel HEBs, CNG, B20 (a liquid blend of 20% biodiesel and 80% diesel), and BEBs. All of these studies considered capital investment and lifetime operation costs related to bus purchases and uses, and most studies included capital investment related to supporting infrastructure such as refueling stations and garage modifications. A few economic assessment studies also conducted separate assessments on GHG and CAP emissions from the life cycle of bus use, [42,48,247,248] and two recent studies monetized the impacts of GHGs or CAPs.[247,248] Furthermore, as summarized in *Chapter 3* and Tong et al. (2015),[146] a number of studies examined solely life cycle GHG emissions for the same set of fuel options.

Some insights emerged from **Table 6.1**. First, the focus of alternative fuel options has changed from studies published a decade ago (where CNG and diesel HEB are the primary focuses) to more recent studies (where BEBs are included), which clearly reflects the changing technology landscape. Second, baseline assumptions, in particular, diesel prices, assumed in these studies have changed over time to reflect market dynamics. This in turn changes conclusions from these studies because diesel prices impact life cycle costs of conventional diesel buses significantly (see, for instance, Clark et al. 2007 & 2008[42,241]). Finally, I find that technology assessments on transit buses still largely focused on ownership costs from transit agencies' perspectives. No study has included externalities or social costs caused by by-products of bus operation, such as GHGs and CAPs in addition to ownership costs to estimate full societal costs. In the literature review, only two recent studies[247,248] assessed social costs, but their assessments are incomplete. Bi et al. (2016)[248] only included social costs related to climate changes, but recent studies have showed that CAP-related health and environmental costs from electricity generation are significant.[193,253] Ercan et al. (2015)[247] only considered social costs of CAPs and they used national-average damage estimates of CAPs, which may be inaccurate because CAP impacts are local.

This chapter estimates both life cycle ownership costs as well as life cycle social costs of GHGs and CAPs for alternative fuel options for transit buses. In addition to a complete estimate of life cycle social costs using up-to-date emission inventories and state-of-art marginal damage estimates, contributions of this chapter also include a comparison between two types of BEBs (slow-charging and rapid-charging) and separate assessments for 40-foot buses and 60-foot buses. I believe that contributions in this chapter could help transit agencies, bus manufacturers, and policymakers gain a better understanding of benefits and costs of alternative fuel options. In addition, I also estimate the contributions from transit buses to CAP emissions inventory in hotspot areas of Pittsburgh, PA to understand the environmental impacts of bus operations at a finer geographic scale.

Table 6.1. Summary of alternative fuel assessment studies for transit buses in the U.S.

Study	Cost components [a]	Fuel options	Conclusions
Lowell et al. (2007)[240]	Vehicle costs (purchase, fuel, O&M excluding fuel) and operator's labor costs.	Diesel, diesel HEB, CNG, hydrogen fuel cell electric bus, hydrogen fuel cell hybrid bus.	The net present value of projected total life cycle costs of fuel cell electric buses and fuel cell hybrid buses are higher than diesel, CNG, or diesel HEB buses for all scenarios considered.
Clark et al. (2007)[42]	Vehicle costs (purchase, fuel, O&M excluding fuel) and infrastructure costs (refueling stations). Separate emission estimates are available in Clark et al. (2007).	Diesel, diesel HEB, CNG, B20.	"Diesel buses are still the most economic technology. In the case where only 20% of the bus procurement cost was considered, as a result of subsidies, the four bus types had a sufficiently similar life cycle cost."
Clark et al. (2008)[241]		Diesel, diesel HEB, CNG, B20.	This report updated the results in Clark et al. (2007a) using (higher) fuel costs in 2008. CNG buses are the most economic technology in four fuel price scenarios, and diesel HEBs are the least economic technology.
Clark et al. (2009)[242]	Vehicle costs (purchase, fuel, O&M excluding fuel) and infrastructure costs (refueling stations and garages).	Diesel (pre-2007 and 2007), diesel HEB, gasoline HEB, CNG	"Each technology could possibly be a best choice in a real procurement and operation scenario, even when default values are used." Key factors include bus speed, annual mileage, cost assumptions, fuel prices, and purchase incentives may impact the comparison.
Johnson (2010)[243]	Vehicle costs (purchase, fuel, O&M excluding fuel) and infrastructure costs (refueling stations and garages).	Diesel, CNG	CNG is profitable for large transit bus fleets (>75 vehicles) unless one or multiple factors (such as diesel prices, CNG bus maintenance costs, bus annual mileage, and vehicle incremental costs) become unfavorable.
Science Applications International Corporation (2011)[244]	Vehicle costs (purchase, fuel, O&M excluding fuel) and infrastructure costs (refueling stations and garages).	Diesel, biodiesel, gasoline, ethanol, CNG, LNG, hydrogen ICE, propane, dimethyl ether, electric trolleybus, BEB, diesel HEB, hydrogen fuel cell electric bus	"This guidebook begins with an overview of how to choose a transit bus fuel, followed by 13 chapters, each addressing one particular fuel or powertrain type." It also has an accompanying spreadsheet-based life cycle costs model, FuelCost2.

Gladstein Neandross & Associates (2012)[245]		Diesel, CNG	"The overall economic feasibility to convert one bus depot to support CNG buses appears to be attractive."
Lowell (2012)[48]	Vehicle costs (purchase, fuel, O&M excluding fuel) and infrastructure costs (refueling stations). Separate emission estimates.	Diesel, CNG	"The pay-back period on the incremental purchase cost of CNG buses and fueling infrastructure, compared to diesel buses, is between five and eight years. CNG buses have 14% reduction in annual fuel costs compared to diesel buses."
McKenzie and Durango-Cohen (2012)[249]		Diesel, diesel HEB, CNG, hydrogen fuel cell bus.	"We find that the alternative fuel buses reduce operating costs and emissions, but increase life-cycle costs. The infrastructure requirement to deploy and operate alternative fuel buses is critical in the comparison of life-cycle emissions."
Trillium CNG (2014)[246]	Vehicle costs (purchase, fuel, O&M excluding fuel) and infrastructure costs (refueling stations and garages).	Diesel, CNG	The payback periods of a small (50 vehicles) and a large (200 vehicles) fleet are 3.7/5.7 years and 2.0/4.0 years (without/with federal funding for bus purchase).
Ercan et al. (2015)[247]	Vehicle costs (purchase, fuel, O&M excluding fuel), infrastructure costs (no details), and social costs (GHGs and CAPs).	Diesel, diesel HEB, B20, CNG, LNG, BEB	"This study finds an optimal bus fleet combination for different driving conditions to minimize life cycle cost, greenhouse gas emissions, and conventional air pollutant emission impacts. In heavily congested driving cycles such as the Manhattan area, the battery electric bus is the dominant vehicle type, while the hybrid bus has more balanced performances in most scenarios because of its lower initial investment comparing to battery electric buses."
Bi et al. (2016)[248]	Vehicle costs (purchase, fuel, O&M excluding fuel), infrastructure costs (chargers), and social costs (GHGs).	Diesel, diesel HEB, plug-in charging BEB, wireless charging BEB.	"The wireless charging bus system has the lowest life cycle cost of US$0.99 per bus-kilometer among the four systems and has the potential to reduce use-phase carbon emissions attributable to the light-weighting benefits of on-board battery downsizing compared to plug-in charging"

*Note: a. These papers have different details in estimating these cost components. * Acronyms explained: HEB, hybrid-electric bus; CNG, compressed natural gas; LNG, liquefied natural gas; BEB, battery electric bus; B20, A blend of 20% biodiesel and 80% petroleum diesel; B100, biodiesel (pure); O&M, operation and maintenance; GHG, greenhouse gas; CAP, criteria air pollutant.*

6.3. Methods

6.3.1. Study Scope

I model a 40-foot bus and a 60-foot articulated bus separately. I consider new transit buses in Model Year 2015 with the following fuel options: a conventional diesel bus, a diesel HEB, a sparking ignition natural gas bus powered by CNG, a sparking ignition natural gas bus powered by LNG, a conventional diesel bus with B20, a conventional diesel bus with B100, a BEB with slow charging in a garage, and a BEB with rapid charging along a bus route. The two types of BEBs differ in onboard batteries and the charging infrastructure. **Table 6.2** lists key assumptions used in this study.

The system boundary for ownership costs is not limited to a bus itself, but also includes refueling infrastructure and a maintenance garage. This is because transit agencies use private refueling stations located within their property. In deploying alternative fuel buses, transit agencies should co-optimize bus fleets and refueling infrastructure (even though it is contracted and owned by a third party) to maximize investment return. The metric used to compare across options is annualized costs evaluated over a bus lifetime of 12 years. I use a 1% discount rate following the Office of Management and Budget.[233] I use 2015 U.S. dollars and convert all other dollars using the Consumer Price Index (CPI) inflation calculator from the U.S. Bureau of Labor Statistics.[199]

This chapter chooses the Port Authority of Allegheny County (PAAC) in Pennsylvania for a case study. PAAC currently operates a transit bus fleet of 704 clean diesel buses and 32 hybrid diesel-electric buses.[254] Some assumptions used in this chapter are specific to PAAC (such as annual bus mileage, diesel price, electricity price, and GHG and CAP emissions of grid electricity in Allegheny County). While some PAAC-specific assumptions apply for other parts of the country (such as annual bus mileage and diesel prices), some assumptions may not (electricity price, and emissions associated with the grid electricity).

Table 6.2. Key technical and economic assumptions used in this study.

Variables	Bus size	Conventional diesel	Diesel HEB	CNG	LNG	Rapid-charging BEB	Slow-charging BEB	B20	B100
Fuel economy (MPGDE)	40-foot	4.8	5.76	4.3	4.3	22.1	18.9	4.8	4.8
	60-foot	3.3	3.96	3	3	15.2	13.0	3.3	3.3
Battery size (kWh/bus)	40-foot	0	5	0	0	88	324	0	0
	60-foot	0	5	0	0	102	377	0	0
Vehicle price ($/bus)[a]	40-foot	$485,000	$758,000	$525,000	$525,000	$800,000	$800,000	$485,000	$485,000
	60-foot	$600,000	$1,114,754	$800,000	$800,000	$1,200,000	$1,200,000	$600,000	$600,000
Vehicle O&M cost (excluding fuel cost) ($/mile)	-	$1.0	$1.0	$1.0	$1.0	$0.3	$0.3	$1.0	$1.0
Battery replacement (probability during lifetime)	-	0%	50%	0%	0%	50%	50%	0%	0%
Range (mile)[b]	40-foot	690	720	600	640	41	130	690	690
	60-foot	475	565	480	510	33	104	475	475
Fuel cost ($/gallon of diesel equivalent)	-	$2.3	$2.3	$1.5	$2.1	$2.1	$2.1	$2.4	$3.0
Per-bus infrastructure cost ($/bus)	-	$0	$0	$50,000	$50,000	$45,000	$55,000	$0	$0
Discount rate	-	1%							
Bus annual mileage	40-foot	36,400 miles/year (minimum 9,882 miles/year, maximum 69,889 miles/year)							
	60-foot	32,719 miles/year (minimum 16,726 miles/year, maximum 44,912 miles/year)							
Bus lifetime	-	12 years							

*Note: a. All vehicles (except 60-foot BEBs) are available on the market. The prices of 60-foot BEBs are calculated from the 40-foot buses assuming the same relative costs with regard to conventional diesel. The battery sizes of 60-foot BEBs are calculated to achieve 80% of the range of the 40-foot BEBs. b. Range is calculated based on fuel economy, the size of fuel tanks/batteries, and usable fuel per tank/battery. * Acronyms explained: HEB, hybrid-electric bus; CNG, compressed natural gas; LNG, liquefied natural gas; BEB, battery electric bus; B20, A blend of 20% biodiesel and 80% petroleum diesel; B100, biodiesel (pure); O&M, operation and maintenance.*

6.3.2. *Life Cycle Ownership Costs*

This section estimates life cycle ownership costs for a transit agency when a fleet of alternative fuel buses are deployed and the supporting infrastructure is built. Life cycle ownership costs consist of four components: bus purchase costs, fuel costs, operation and maintenance (O&M) costs (except fuels), and upfront infrastructure costs (including building refueling facilities unless they already exist and garage modifications). These costs are then summed and converted into annualized costs using Eqn. 6.1-6.2.

$$Annualized\ ownership\ cost = \frac{Vehicle\ \&\ infrastructure\ capital\ cost}{Annuity\ \ factor_{lifetime,\, discount\ rate}} + Annual\ O\ \&\ M\ \ cost \tag{6.1}$$

$$Annuity\ \ factor_{lifetime,\, discount\ rate} = \frac{1-\left(1+discount\ rate\right)^{-lifetime}}{discount\ rate} \tag{6.2}$$

I obtain bus purchase costs from bus manufacturers and published literature.[255,256] One factor that may change bus purchase costs from a transit agency's perspective is the availability of external funding. For instance, the U.S. Federal Transit Administration (FTA) Section 5307 provides funding that may cover up to 80% of bus purchase costs.[42,255] Thus, I present two life cycle cost estimates which assume external funding that pays for 80% bus purchase costs exists or no external funding. I assume there is 50% probability that HEBs and BEBs will need to replace their batteries once in year 6, following *Chapter 3* and Tong et al.[146] I note some studies[247] assumed a higher number of battery replacements during the bus lifetime but their assumptions are likely to be an underestimate of battery lifetime. I assume a $700/kWh battery cost for battery replacement.[255] Fuel costs over a given period are calculated based on annual mileage, fuel economy, and fuel prices.[234] I do not account for fuel price changes over the bus lifetime as the actual fuel price trajectory is hard to project. Instead, I run a sensitivity analysis on fuel prices to understand their impacts. Fuel economy assumptions are drawn from Altoona Bus Research & Testing Center[207] and Tong et al.,[146] which also included a review of recent studies on fuel economy. O&M costs (except fuels) are from recently published technology evaluation studies. [255]

Infrastructure costs are estimated using an engineering economics approach. A key step is to examine if alternative fuel buses require new refueling infrastructure and/or garage modifications (such as CNG, LNG, and BEBs) or if they work well with existing infrastructure (such as diesel HEBs and biodiesel). The infrastructure costs for natural gas buses are taken from a recent PAAC design study.[245] Here I assume a high utilization rate of the natural gas infrastructure, which supports 100 natural gas buses. If the actual utilization rate is lower than assumed, each bus's share of the infrastructure cost will increase. I estimate charging infrastructure costs for BEBs based on communications with officials at PAAC, which has invited major BEB manufacturers to present and demonstrate their buses. I emphasize that I only include direct equipment costs for infrastructure costs, as with most studies listed in **Table 6.1**. Indirect equipment costs, such as capital investment to update grid connections (which might be needed for CNG/LNG refueling stations and BEB chargers), are not included because these costs are case-specific. Similarly, labor costs associated with the design and construction of infrastructure are not included.

6.3.3. Life Cycle Social Costs

Transit buses emit GHGs (carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)) and CAPs (nitrogen oxides (NO_x), carbon monoxide (CO), volatile organic compounds (VOC), particulate matter (PM), and sulfur dioxide (SO_2)) over the life cycle of bus operation. The life cycle components consist of bus operation (tailpipe exhaust, tire and brake wear), the process to produce and deliver fuels used to power a bus, and upstream activities that extract primary energy and feedstock used in fuel production processes. In addition, I include GHGs and CAPs from manufacturing additional lithium-ion batteries for HEBs and BEBs.

This chapter characterizes health and environmental damages caused by GHGs and CAPs using the damage function approach.[4] Emissions increase air concentrations due to physical and chemical processes (accumulation, dispersion and removal process). There are multiple mechanisms linking concentration changes to physical impacts: elevated concentrations of GHGs affect the energy balance of the earth, which could lead to climate change, such as temperature increase, precipitation change, sea level rise, and ocean acidification;[2] and increased levels of

$PM_{2.5}$ and ground-level ozone due to CAP emissions impose higher risks of mortality and morbidity on the exposed human population, and contribute to soil and water acidification, reduced tree growth, reduced agricultural yields, and impaired visibility.[30,57] All of these physical effects are valued in monetary terms using market prices or estimated price proxies (such as willingness-to-pay) of non-market goods.[30,31,57,195]

In this chapter, I assume that GHGs and CAPs emitted by transit buses are marginal. So I estimate social costs by multiplying the amount of emissions (by species and by location) with the marginal damage from a unit emission (of the same species emitted in the same location). There is a key distinction between GHGs and CAPs. GHGs are globally mixed so their marginal damages are the same around the world, but CAPs are locally mixed thus their marginal damages vary from region to region. For example, it is problematic to compare a ton of CAP emissions in New York City to a ton in Pittsburgh. The formulas to calculate climate change damages and air pollution damages are as follows.

$$Climate\ change\ damages = life\ cycle\ GHG\ emissions \times Social\ cost\ of\ carbon \tag{6.3}$$

$$\begin{aligned} &Air\ pollution\ damages_{life\ cycle\ stage} \\ &= \sum_{CAP\ species} CAP\ emission_{CAP\ species,\ location,\ life\ cycle\ stage} \times Marginal\ Damages_{CAP\ species,\ location} \end{aligned} \tag{6.4}$$

$$\begin{aligned} &Life\ cycle\ air\ pollution\ damages \\ &= air\ pollution\ damages_{vehicle\ operation} + air\ pollution\ damages_{battery\ manufacturing} \\ &+ \frac{air\ pollution\ damages_{upstream\ activities}}{vehicle\ fuel\ efficiency} \end{aligned} \tag{6.5}$$

I use life cycle GHG emissions estimates in *Chapter 3* and Tong et al. (2015)[146] with adjusted fuel economy assumptions. In addition, I assume that B100 reduces life cycle GHG emissions by 50% compared with conventional diesel.[257] I convert all GHGs to CO_2-eqivalent emissions using Global Warming Potential (GWP).[2] I use both 100-year and 20-year GWP, the latter of which leads to higher CO_2-equivalent emissions per unit of methane than the former. Appendix E includes the life cycle GHG emission estimates. The marginal damage from a unit of carbon emission is called the social cost of carbon (SCC). A U.S. interagency group published SCC estimates for use in decision-making process.[258] The SCCs are estimated using integrated assessment models that model earth's physical systems and economic systems. The most recent SCC estimates range from \$13 to \$120 for a metric ton of CO_2 emitted in 2015 (in 2015 dollars). I use a median estimate of \$41 per metric ton of CO_2 emitted.

I use life cycle CAP emissions and the resulting air pollution costs estimated in Chapter 4 and Tong et al. (2016)[253] with adjusted fuel economy assumptions (See Appendix E for specific assumptions used in this chapter). Chapter 4 and Tong et al. (2016)[253] constructed a spatial life cycle CAP emission inventory by U.S. counties. It used data sources such as EPA's National Emissions Inventory (NEI),[194] EPA's Continuous Emissions Monitor System (CEMS),[196] Altoona Bus Research & Testing Center,[207] and the GREET model[159] to characterize CAP emissions from energy production processes, electric power grids, and bus operations in the U.S. Chapter 4 and Tong et al. (2016)[253] used two state-of-the-art models, the AP2 model[30,31] and EASIUR model[57,195] to estimate social costs of CAPs. Both models estimate environmental and health damages resulting from one unit of CAP emission in every county in the contiguous U.S. The two models differ primarily by how they model the link between CAP emissions and concentration changes of $PM_{2.5}$ and ground-level ozone (See Chapter 4, Tong et al. (2016),[253] and Heo et al. (2016)[57] for details). Since PAAC's bus fleet primarily operates within Allegheny County in Pennsylvania, I use the CAP emission inventory and marginal damages for Allegheny County to estimate air pollution costs for PAAC's bus fleet. Because Chapter 4 and Tong et al. (2016)[253] did not include biodiesel, so I assume comparable air pollution costs between conventional diesel and B100 due to a lack of recent literature on this issue. I believe research is needed to clarify biodiesel's air pollution costs.

6.3.4. Criteria Air Pollutant Emissions in Hotspot Areas

While literature has shown that air pollution costs vary within the county boundary, it is currently computationally impossible to estimate air pollution impacts with a grid size smaller than 10 km by 10 km. So this chapter models CAP emissions from PAAC's bus fleets in hotspot areas in Pittsburgh, PA to estimate PAAC's contributions at a finer geographic scale than a county. There are currently no real-time emission monitoring systems on mobile sources (including transit buses) due to the size and cost of monitoring devices. Instead, I calculate emissions based on vehicle operation emissions measured during bus tests, which are used in social cost estimates as well as estimated bus fleet mileage in hotspot areas. The hotspot areas include the Downtown, North Shore, Station Square, and Oakland areas in Pittsburgh, PA. (**Figure 6.1**)

The bus fleet mileage in hotspot areas are calculated as the total bus miles from all bus trips within hotspot areas over a calendar year. The bus mileage in hotspot areas for any bus route is calculated using ArcGIS software and bus route shapefile files.[259] **Figure 6.1** shows bus routes and bus stops in the hotspot areas. The number of bus trips for any bus route in a calendar year is calculated using bus schedule files (General Transit Feed Specification (GTFS) files).[259] In this analysis, I do not account for planned and unplanned bus service changes during holidays.

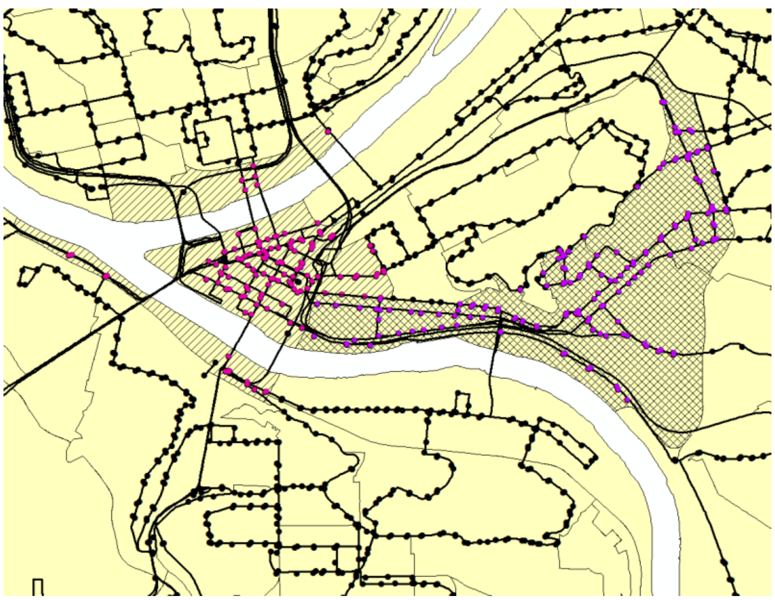


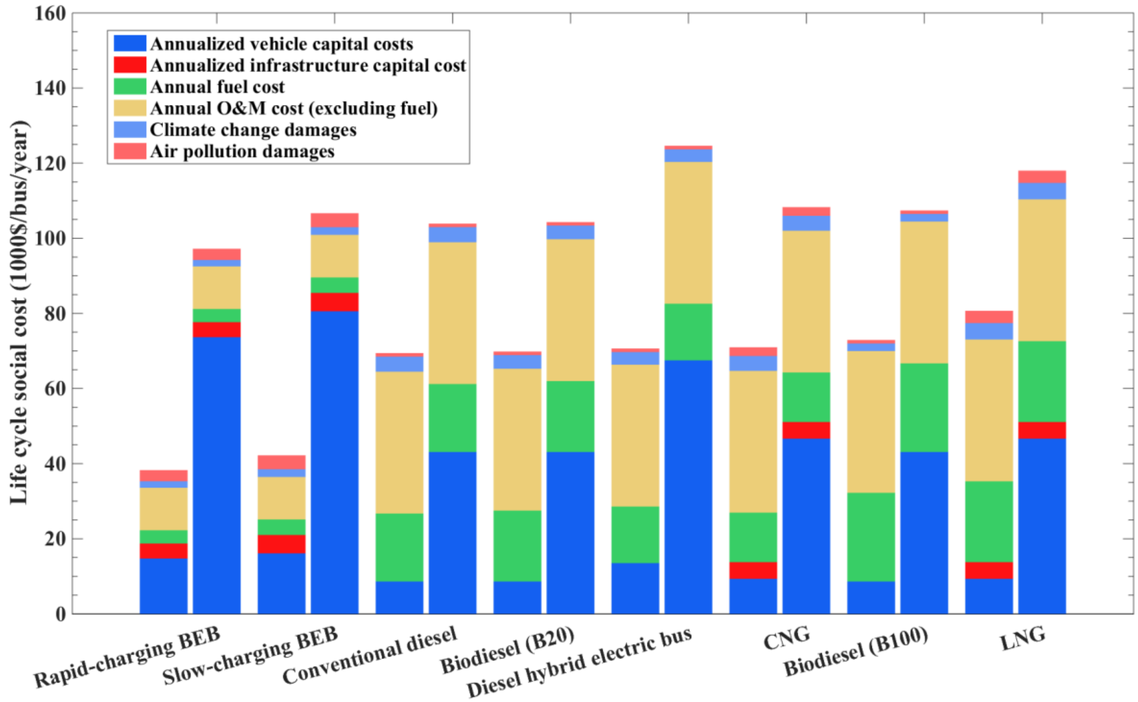
Figure 6.1. PAAC's transit bus routes (black solid lines) and stops (pink and purple dots) in hotspot areas (shaded areas) in Pittsburgh, PA. The hotspot areas include the Downtown, North Shore, Station Square, and Oakland areas in Pittsburgh, PA.

6.4. Results

6.4.1. Life Cycle Ownership Costs

I consider two scenarios for baseline results, one where external funding that pays for 80% of bus purchase costs is available, and the other where external funding is not available. I note that external funding (such as the FTA funding) can have other competing uses, such as retrofitting existing buses and upgrading bus garages, so its availability for bus purchases may be less than assumed. However, upon communication with PAAC, external funding is currently sufficiently available.

Figure 6.2 shows life cycle ownership & social costs as well as cost breakdowns for 40-foot and 60-foot transit buses in two scenarios (with or without external funding). I find that the availability of external funding is crucial for transit agencies to adopt any alternative fuel option. Without external funding, conventional diesel is among the cheapest in terms of both life cycle ownership costs and life cycle ownership & social costs. For a 40-foot transit bus, only rapid-charging BEBs have lower ownership & social costs than a conventional diesel bus without external funding. The advantages of BEBs are their high vehicle efficiency, low electricity rates in PA, and low O&M costs for BEBs as fewer mechanical devices and pollution control devices are needed. Surprisingly, diesel HEBs have the largest ownership & social costs among all technologies considered. This is because of a high bus purchase price premium and a low return in reduced fuel costs over the lifetime. When external funding is available to reduce bus purchase costs by 80%, BEBs become much more cost-effective. In this case, life cycle ownership & social costs of BEBs are 37-43% lower than conventional diesel buses. Other bus options still cost more than a conventional diesel bus in terms of life cycle ownership & social costs (+1%, +2%, +2%, +5%, and +16% for B20, diesel HEB, CNG, B100, and LNG respectively).



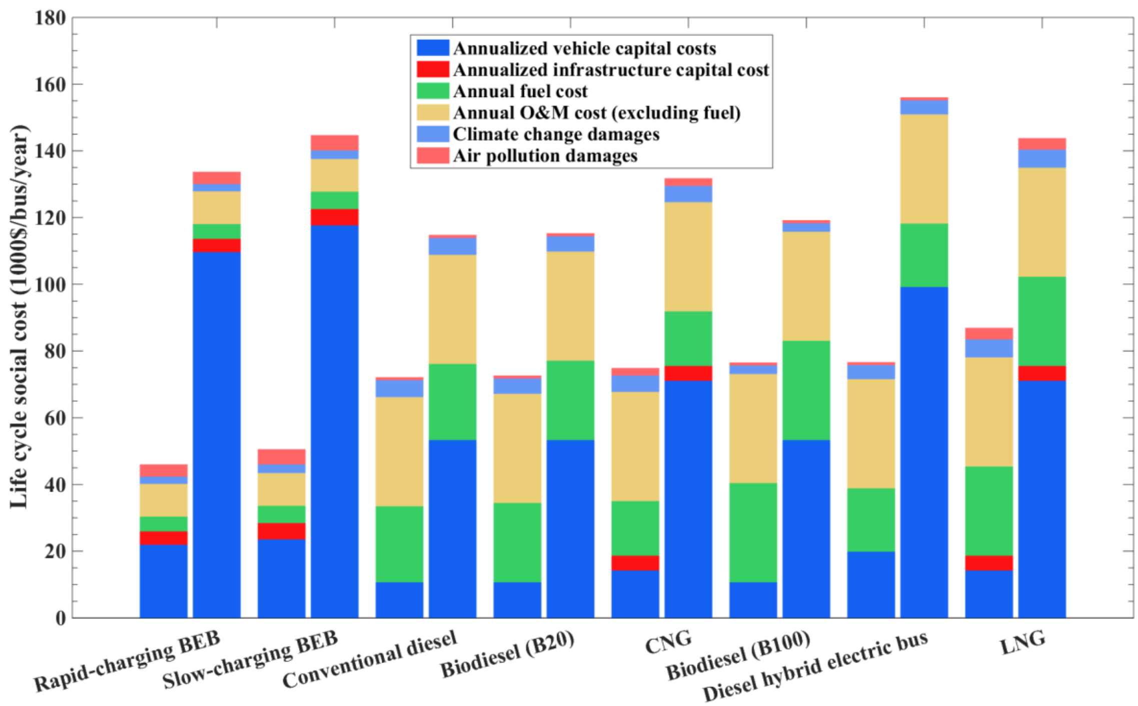


Figure 6.2. Annualized life cycle ownership & social costs for a 40-foot transit bus (top) and a 60-foot bus (bottom). In each figure, left bars assume reduced vehicle purchase costs (80% paid by external funding) and right bars consider full vehicle purchase costs without external funding. The project lifetime is assumed to be the same as the lifetime of a bus (12

years) and I assume 1% discount rate. Social costs include climate change damages (using 100-year global warming potential (GWP)) and air pollution damages (using AP2 model).

A 60-foot bus is more capital-intensive and has a lower fuel economy than its 40-foot counterpart, but it carries more riders during one trip. When evaluating the two options in terms of ownership costs or ownership & social costs, a 60-foot bus is more expensive than a 40-foot bus. When external funding is available, the rank of technology options is similar to that of the 40-foot transit bus (except that diesel HEBs become relatively worse). The rank of technology options changes compared with the 40-foot case when external funding is not available. In this case, none of the alternative fuel options has lower ownership & social costs than a conventional diesel bus. B20 has comparable ownership & social costs with conventional diesel, because the only differences are fuel price and life cycle GHG emissions. BEBs have 16-26% higher ownership & social costs compared to conventional diesel. Furthermore, CNG, LNG, and diesel HEBs all have larger increases in ownership & social costs than their 40-foot counterparts. Two reasons explain the relatively poor performance of alternative fuel options for 60-foot buses versus 40-foot buses. First, 60-foot transit buses have some unfavorable conditions compared to 40-foot buses – they are relatively more expensive because of a smaller demand; they have worse fuel efficiency because of heavier weight; and they have lower annual mileage as they are used less often on weekends and holidays. Second, the metric ($/bus/year) does not account for the additional service provided by 60-foot transit buses compared to 40-foot buses. Alternative metrics such as passenger-miles and seat-miles may favor 60-foot transit buses. While 60-foot transit buses are more valuable in rush hours, they are less valuable in non-rush hours.

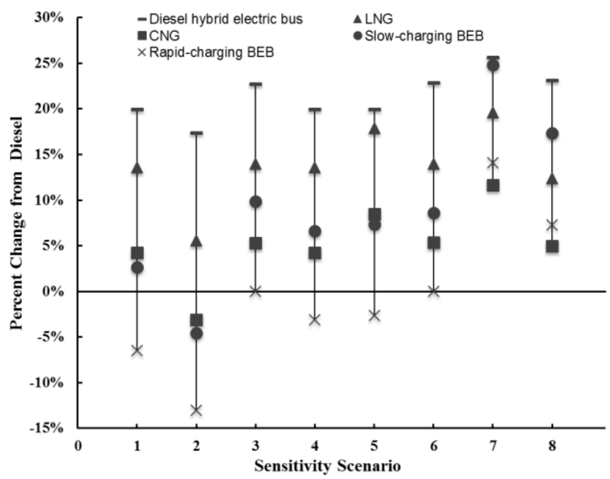
6.4.2. Factors That Change the Ranks of Alternative Fuel Options

Figure 6.3 shows sensitivity analysis results for a 40-foot transit bus on higher diesel prices, lower annual bus mileage, higher electricity rates, higher infrastructure costs, and higher discount rates. **Table 6.3** lists sensitivity scenarios considered. I consider these five factors because they are uncertain and are likely to impact the ranks of transit bus technologies. For each of the five factors, I determine a likely value different from the baseline assumption. I then run the analysis holding all other assumptions the same as the baseline scenario.

I find that all five factors, independently or jointly, do not change the conclusions that BEBs achieve large reductions in ownership & social costs compared to conventional diesel, when external funding is available. When external funding is not available, I find that lower annual mileage and a higher discount rate have large impacts on the cost reduction potential of BEBs. When annual mileage is lower than 31,600 miles/year or discount rate is higher than 4.15%, the benefits of rapid charging BEBs with regard to conventional diesel diminish. On the contrary, doubling electricity rates or doubling per-bus infrastructure costs (either through higher infrastructure costs or smaller bus fleets to use the rapid charging infrastructure) do not change the (+ or -) sign of the comparison between rapid-charging BEBs and conventional diesel buses, even though BEBs' cost reductions become smaller.

Table 6.3. Scenario descriptions for sensitivity analysis. Baseline assumptions are used unless otherwise stated.

Scenario	Assumptions
1 – Baseline	Annual mileage of 37,761miles/year and 1% discount rate
2 – Higher diesel price	Diesel price $1/gallon higher the baseline
3 – Reduced annual mileage	Annual mileage reduced to 31,600 miles/year
4 – Doubling electricity price	Double electricity price from the baseline
5 – Doubling infrastructure cost	Double the per bus infrastructure cost from the baseline
6 – Higher discount rate	Increase discount rate to 4.15%
7 – Combine scenarios 3, 4, 5, 6	See above
8 – Combine scenarios 2, 3, 4, 5, 6	See above



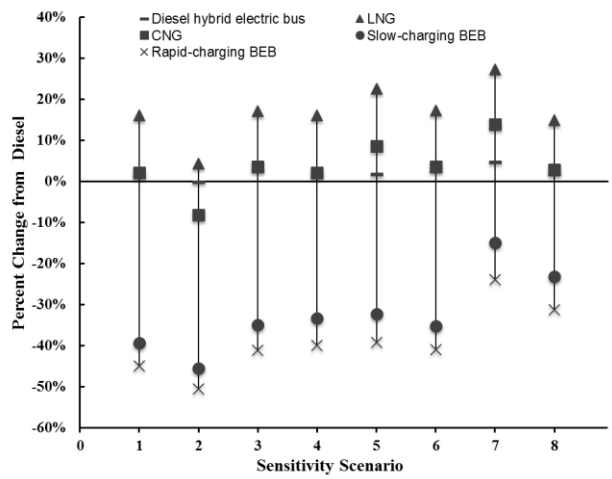


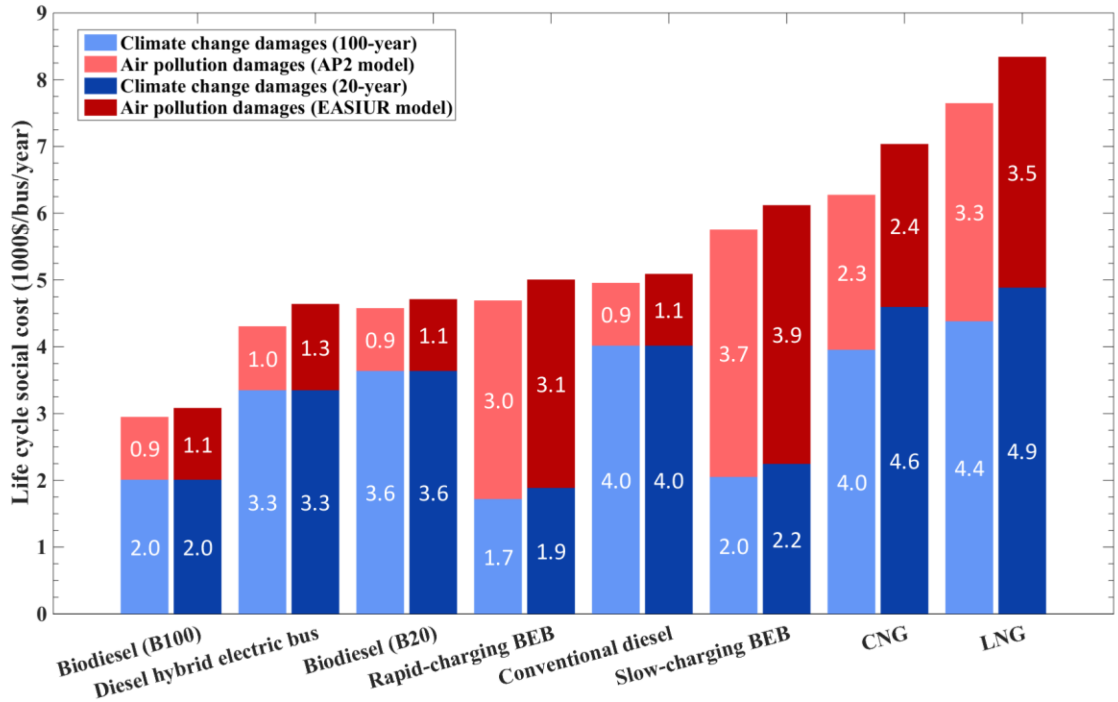
Figure 6.3. Sensitivity analysis results for 40-foot transit buses without external funding (left) and with external funding (right). Percentages are calculated as differences between life cycle ownership & social costs of alternative fuel options and conventional diesel. Negative percentages mean lower ownership & social costs from alternative fuel options.

The diesel price is currently low due to a combination of strong supply and weak demand in global crude oil and refined product markets.[237] In the baseline scenario, I assume the diesel price to be $2.30/gal based on PAAC's data and recent diesel markets.[234] I note, however, the large variability in diesel prices in the last decade (2007-2016), where diesel prices ranged between $2.00/DGE and $4.70/DGE.[237] Because the conventional diesel bus serves as the baseline in the assessment, changes in diesel prices significantly affect the comparison between alternative fuel options. As the diesel price is currently at a decade-low point,[237] I expect the diesel price to rebound slightly back as global market adjusts towards equilibrium. In the sensitivity analysis, I consider a diesel price of $3.30/gal. I note that higher diesel prices can happen in the future. At the diesel price of $3.30/gal, both types of BEBs and CNG achieve lower ownership & social costs compared to conventional diesel, with or without external funding. The estimates show that this diesel price is not high enough to balance out all of the jointly unfavorable conditions (lower annual bus mileage, higher electricity rate, higher infrastructure costs, and higher discount rate) for BEBs. Further analysis shows that the break-even diesel price is around $4.50/gal, or almost a doubling of the baseline diesel price to cancel out all of the unfavorable conditions for rapid-charging BEBs.

6.4.3. Life Cycle Social Costs

For the baseline results (**Figure 6.2**), I find that including life cycle social costs does not change the rank of technologies. This is because social costs are small compared to ownership costs. For 40-foot buses, the ratio between social costs and ownership costs fall between 4% and 7% (without external funding), or 6% and 16% (with external funding) -- with biodiesel and conventional diesel on the lower end and BEBs on the higher end. A similar pattern exists for 60-foot buses although the range of ratios becomes 3-7% (without external funding) or 5-16% (with external funding). Nevertheless, technology assessments that ignore these social costs are incomplete because these are actual costs paid by people not just the emitter.

If I limit the scope to include only social costs, I find that biodiesel (B100 and B20) and diesel HEBs have lower social costs compared to conventional diesel for both 40-foot and 60-foot buses (**Figure 6.4**). On the other hand, LNG, CNG and slow-charging BEBs have higher social costs than conventional diesel, and LNG 60-foot buses more than double social costs of conventional diesel. The leading source for high social costs of these pathways is the grid electricity, which has high SO_2 and NO_x emissions. Specifically, BEBs charge and store the grid electricity to power buses, and liquefaction and compression of natural gas are both intensive in electricity use. In addition, LNG also has higher GHG emissions than conventional diesel, contributing further to high social costs. Contrary to LNG, BEBs have low GHG emissions due to their high vehicle efficiency. While BEBs completely reduce CAP emissions from tailpipe exhausts, damages associated with CAP emissions from grid electricity are quite significant, and more than cancel out BEBs' climate change mitigation benefits.



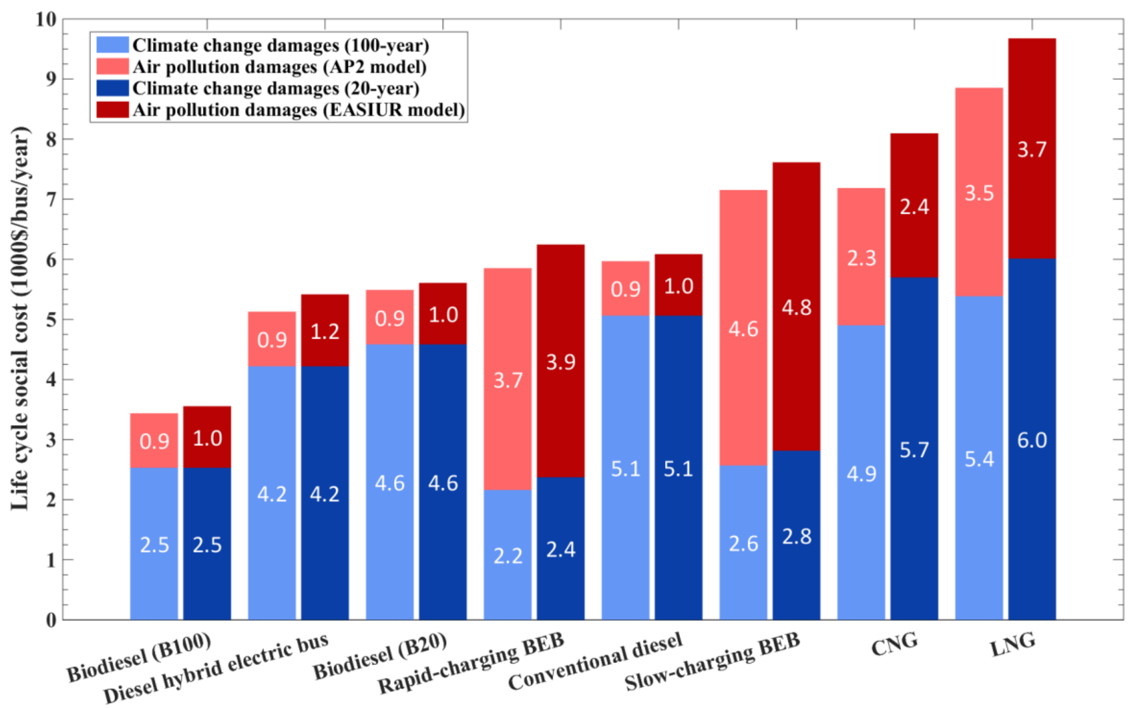


Figure 6.4. Life cycle social costs for a 40-foot (top) and a 60-foot transit bus (bottom). Left bars represent climate change damages (based on 100-year time horizon) and air pollution

damages (based on AP2 model). Right bars represent climate change damages (based on 20-year time horizon) and air pollution damages (based on EASIUR model). The left bars and right bars show lower bounds and upper bounds of social costs using different time horizons of global warming potential (GWP) and criteria air pollutant (CAP) marginal damage models.

6.4.4. Criteria Air Pollutant Emissions in Hotspot Areas

PAAC currently operates 100 bus routes including 2 temporary routes to make up for reduced light rail service. 83 of these 100 bus routes serve Downtown Pittsburgh, and 89 bus routes serve either Downtown or the Oakland area. Over a calendar year, these 89 routes make more than 900,000 bus trips, or 94% of all PAAC's bus trips, in the hotspot areas (Downtown & Oakland). The bus fleet mileage within hotspot areas is 2.7 million miles per year, or roughly 10% of PAAC's total bus mileage. The actual emissions in hotspot areas are calculated using fleet mileage in hotspot areas and weighted-average emission factors of PAAC's bus fleet [260]. I find that PAAC's bus fleet emitted 135 metric tons of NO_x and 2.2 metric tons of $PM_{2.5}$ in 2015 (**Table 6.4**). Around 10% of these emissions happened in hotspot areas.

To compare emissions reduction potential of alternative fuel options, I calculate an emission proxy using emission factors of new buses. In other words, the emission proxy represents emissions if the whole bus fleet is composed of new buses. Although this is an unlikely scenario, without referring to a complex bus turnover model, the emission proxy should help identify relative benefits of alternative fuel options. **Table 6.4** shows that BEBs can eliminate all tailpipe emissions (but still have $PM_{2.5}$ emissions from break and tire wear), achieving the best emissions reduction potential of all technologies considered. Diesel HEBs reduce SO_2, VOC, and CO emissions but increase NO_x emissions by 50% relative to new diesel buses. LNG and CNG buses reduce SO_2, NO_x, and VOC emissions but increase CO emissions significantly by a factor of 64.

Michanowicz et al. (2012)[261] estimated that 224 tons of $PM_{2.5}$ were emitted from mobile sources in Allegheny County in 2009 and 43% (or 96.3 tons) came from diesel vehicles. Thus PAAC's bus fleet only contributes to slightly more than 1% of $PM_{2.5}$ emissions from all mobile sources in Allegheny County. However, it is worth noting that reduction of $PM_{2.5}$ emissions are important to human health. Literature shows that diesel particular matter (DPM) is the leading additive cancer risk air toxic in Downtown Pittsburgh and in Allegheny County.[262] Thus alternative fuels (CNG, LNG, BEBs) have the added benefit of reducing cancer risk by replacing diesel buses in Downtown Pittsburgh and Allegheny County.

Table 6.4. Estimated criteria air pollutant (CAP) emissions from PAAC's bus fleet in the hotspot areas in 2015. Unit: metric ton/year. Note only emissions directly from vehicle operation are included. Emission proxies (*) are calculated assuming the whole bus fleet is composed of new buses. N/A means not available.

Scope	PAAC all					Hotspot areas				
	$PM_{2.5}$	SO_2	NO_x	VOC	CO	$PM_{2.5}$	SO_2	NO_x	VOC	CO
Existing fleet	2.7	N/A	135	N/A	N/A	0.3	N/A	13.7	N/A	N/A
New diesel*	0.9	0.4	24.8	3.0	13.2	0.1	0.04	2.5	0.3	1.3
New diesel HEBs*	0.9	0.3	39.0	2.1	5.0	0.1	0.03	4.0	0.2	0.5
New CNG*	0.9	0.3	15.6	1.9	844	0.1	0.03	1.6	0.2	86.0
New LNG*	0.9	0.0	15.6	1.9	844	0.1	0.00	1.6	0.2	86.0
New BEBs*	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

6.5. Conclusions and Policy Recommendations

This chapter compares ownership & social costs for alternative fuel options, and estimates CAP emissions from PAAC's bus fleet in hotspot areas. I find that BEBs have the best performance in both assessments. If external funding is available, purchasing and operating BEBs results in significant savings compared to diesel buses. Furthermore, the results regarding battery electric buses (BEBs) are robust when the external funding is available. When external funding is not available, I find that lower annual mileage and a higher discount rate have large impacts on the cost reduction potential of BEBs. I find that rapid-charging BEBs achieve lower costs than slow-charging BEBs due to double dividends from smaller batteries used in rapid-charging BEBs. The battery replacement costs are smaller, and rapid-charging BEBs are lighter in weight, achieving better fuel efficiency.

6.5.1. Regional Variations

I emphasize that the results and findings are limited by the assumptions used. As mentioned in the Method section, some PAAC-specific assumptions, such as electricity rates and emissions associated with grid electricity vary from region to region. Performing the same assessments with region-specific electricity-related assumptions may yield different conclusions. For instance, average electricity rates across utilities are $0.08-0.28/kWh for slow charging and $0.14-0.44/kWh for fast charging in California.[263] These electricity rates are significantly higher than electricity rates in Pittsburgh, PA, because utilities in CA have demand charges and dynamic pricing.

I expand the sensitivity analysis to test impacts of these electricity rates. I find that without external funding, an average electricity rate higher than $0.165/kWh, or tripling the baseline electricity rate, would cause BEBs to have higher life cycle ownership & social costs than conventional diesel. If external funding is available, however, rapid-charging BEBs still have lower ownership & social costs than diesel for an electricity rate as high as $0.50/kWh. Further, when demand charges and dynamic pricing are in place, slow-charging BEBs may result in lower ownership & social costs than rapid-charging BEBs, because slow-charging BEBs are subject to lower average electricity rates. The electricity grid of the Midwest/Mid-Atlantic region has the largest share of coal-fired power plant plants in the country.[264] So other regions could find BEBs achieve lower social costs than conventional diesel if electricity grids in those regions are cleaner.

Finally, fuel economy assumptions and GHG & CAP emissions from vehicle operation may also vary across region, because of varying factors such as speed, weight, road grade, and weather.[104,175,265] Indeed, the measured fuel economy values from Altoona Bus Research & Testing Center[207] may not apply for extreme weather conditions. A previous study has identified large variations in fuel economy of light-duty vehicles under extreme weather.[175] Similar studies on transit buses are needed when there are more data.

6.5.2. *Practical Challenges for BEBs*

While BEBs are estimated to have the smallest life cycle ownership costs and/or social costs, both types of BEBs face practical challenges to immediate operation for a typical bus route. First, BEBs have limited ranges (33-41 miles and 104-130 miles for rapid-charging and slow-charging BEBs), which are significantly smaller than other bus technologies (**Table 6.2**), and would require special routes or specialized planning and scheduling. Indeed, rapid-charging BEBs require tight control of bus schedules to ensure a bus is charged at a specific bus stop and time. Even though buses are operated on a planned schedule, the actual schedule is determined by traffic congestion, weather and other road factors. As a result, bus routes on dedicated bus lanes or fixed busways may be more feasible for rapid-charging BEBs. Additionally, BEBs require dedicated charging infrastructure, which, in addition to higher capital expenditures and O&M costs, require land to install and coordination with local utilities. Finally, charging infrastructure for BEBs are currently not compatible among bus manufactures.

6.5.3. *Favorable Trends for BEBs*

Several trends may make BEBs more attractive in the near future. BEBs will become more technologically mature as more buses are delivered and operated across the country. The costs of batteries are declining rapidly while the performance is improving quickly[266] due to increased battery deliveries in light-duty vehicle markets. Thus, future BEBs will have better economics and longer range.

Equally important are federal and state energy policies such as U.S. EPA's Clean Power Plan and state-level Renewable Portfolio Standards (RPS).[267,268] They will lead to more renewable energy sources and less coal-fired power plants in U.S. electricity grids in the next two decades. In particular, U.S. EIA[264] projected a 26% decline in direct CO_2 emissions from the electricity grid in the Midwest/Mid-Atlantic region from 2015 to 2030 (in the reference case in Annual Energy Outlook 2016) as a result of a more than 40% reduction in coal-fired electricity generation during the same period. As coal-fired power plants have large CAP emissions, I expect a similar reduction in direct CAP emissions from electricity grid.

If I assume a 26% reduction in social costs from grid electricity in the Midwest/Mid-Atlantic region, and assume conventional diesel's social costs remain the same over the next 15 years, then BEBs in 2030 will result in lower life cycle social costs than conventional diesel for all GWP's time horizons and CAPs marginal damage models. If battery and other technology improvements are considered, BEBs advantages will be even larger. Finally, I note that BEBs are easier to integrate with intelligent control technologies. For instance, BEBs already have the capability to communicate their key information (such as battery's state of charge (SOC) and GPS locations) to a control room to facilitate scheduling, charging, and operation.[269] In the future, sensing and communication capacities of BEBs could help build a smart transportation system where connected & automated vehicles dominate.

6.5.4. Uncertainty in Social Cost Estimates

While I have used the most recent data to build emissions inventories and used state-of-art marginal damage estimates of GHGs and CAPs, I emphasize that there are high uncertainties in social cost estimates due to conflicting emissions estimates and evolving scientific understandings of health and environmental impacts of GHGs and CAPs. First, Chapter 4 and Tong et al. (2016)[253] found that upstream (well-to-pump) air pollution costs from petroleum fuels would increase by a factor of 4 using GREET model's emissions data rather than using U.S. EPA's NEI (used in this chapter), and would increase life cycle air pollution costs by 87% for PAAC's bus fleet. However, because of the relatively low ratios between social costs and ownership costs, using alternative social cost estimates does not change the ranking of fuel options in terms of ownership & social costs. Second, the SCC has a large range of estimates from a few to hundreds of dollars per metric ton of CO_2 emission. The two-order-of-magnitude difference is mainly due to different assumptions used on discount rates and various climate change damage functions.[4] Third, CAPs' social damage estimates do not include all known health impacts due to data and methodological issues. In particular, currently available marginal damage estimates of VOCs and CO are likely to be underestimates,[253] and cancer risks of diesel particular matter are not monetized at all.[262] Furthermore, current estimates of CAPs' social damages cannot go smaller than a 10-km-by-10-km resolution, which is still too large to accurately characterize CAPs' damages.

6.5.5. Policy Implications

The analyses on alternative fuel options for transit buses indicate that BEBs are promising technology options. While BEBs were not included in previous assessments, they exhibit high fuel efficiency, clean exhaust emissions, and resulting low life cycle ownership & social costs. BEBs should attract attention and strong interest from transit agencies, bus manufacturers, and public officials who want to maximize public interests. As highlighted in the previous discussion, BEBs could help transit agencies operate in more intelligent transportation systems that are likely in the near and medium futures. I note, however, that any transit agency that plans to operate BEBs should prepare for changes in planning and scheduling, operation and maintenance, fuel procurement, and supporting infrastructure.

This chapter extends the framework and method of economic assessments on alternative fuel options by including life cycle social costs of unintended air pollutants. While the inclusion of social costs does not change the rank of fuel options, it provides more accurate accounts of private and social impacts caused by transit buses. Furthermore, this chapter highlights the uncertainty and methodical limitations of state-of-the-art damage function approaches and points out potential research directions. In addition, this chapter estimates emissions from bus fleets in hotspot areas to show the implications of high-resolution emissions estimates. I believe that this updated framework of life cycle ownership & social costs will help transit agencies, and other interested audiences to determine the best alternative fuel option, and to maximize private and social net benefits.

Chapter 7. Conclusions and Future Work

This chapter concludes the findings in this dissertation, and discusses policy implications and future research directions learnt from this work. I also briefly discusses the deliverables originating over the course of this dissertation.

7.1. Research Questions Revisited

This summarizes the findings from five studies in this dissertation by providing brief answers to the research questions outlined in *Chapter 1 Introduction and Background*.

Dissertation: *The Good, the Bad, and the Ugly: Economic and Environmental Implications of Using Natural Gas to Power On-Road Vehicles in the United States.*

- *What are the economic and environmental implications of using natural gas to power on-road vehicles? Can natural gas reduce environmental and economic impacts compared to petroleum fuels?*

This work analyzed the economic and environmental impacts of a wide range of natural gas fuel pathways for a selection of vehicle types. I choose to focus on two environmental impacts: greenhouse gas (GHG) and criteria air pollutant (CAP) emitted over the life cycle of the pathway. None of the natural gas pathways eliminates life cycle air emissions. In fact, only *a few* pathways reduce life cycle GHG emissions and/or life cycle air pollution damages compared to baseline petroleum fuels (gasoline for light-duty vehicles (LDVs) and diesel for heavy-duty vehicles (HDVs)).

For LDVs and transit buses, battery electric vehicles (BEVs) powered by natural gas-based electricity provide significant reduction in life cycle GHG emissions and life cycle air pollution damages (for almost all counties) compared to the baseline petroleum fuels. However, the actual electricity that charges BEVs may not be natural gas-based electricity in most parts of the U.S. When powered by U.S. grid electricity (using average emission factors), BEVs reduce life cycle GHG emissions to a lesser extent but increase life cycle air pollution damages significantly. Compressed natural gas (CNG), while reducing GHG emissions and CAP emissions (except CO) at tailpipe are more likely to increase life cycle GHG emissions and life cycle air pollution damages.

For heavy-duty trucks, CNG sparking-ignition (SI) trucks and liquefied natural gas (LNG) high-pressure direct ignition (HPDI) trucks have mixed environmental impacts. While they are unlikely to reduce life cycle GHG emissions compared to diesel, they reduce life cycle air pollution damages in 76-99% of counties for local-haul tractor-trailers and in 32-71% of counties for long-haul tractor-trailers.

Except hydrogen fuel cell electric vehicles (FCEV) and propane, all the other natural gas fuel pathways (including methanol, ethanol, and Fischer-Tropsch liquids) increase life cycle GHG emissions. Hydrogen fuel cell electric vehicles provide small reduction in life cycle GHG emissions, but factors such as methane leakage rate, global warming potential uses, and the assumption on fuel cell replacement may shrink the emissions reduction potential.

I examined the economic impacts of natural gas fuel pathways for two vehicle types, tractor-trailers and transit buses. The economic returns of natural gas tractor-trailers versus diesel tractor trailers have been studied extensively and a stable fuel price discount is needed to make natural gas tractor-trailers more attractive. In this dissertation, I studied the economic feasibility of a national natural gas refueling infrastructure for long-haul trucks in U.S., which is a prerequisite for natural gas tractor-trailers. I found that a transition to natural gas fuels in long-haul trucks is the hardest in the beginning (when the shares of natural gas trucks are below 5%) because of low refueling demands and over-capacity of the refueling infrastructure to ensure network coverage. At higher shares of natural gas trucks, both the total refueling capacity and the net economic benefits of the national refueling infrastructure increase almost linearly as adoption increases.

Finally, I considered both life cycle ownership costs and life cycle social costs due to GHG emissions and CAP emissions for transit buses. Transit buses are early adopters of alternative fuel technologies because of funding supports and operation characteristics (such as high fuel consumption and private refueling infrastructure). I found that the availability of external funding is crucial for transit agencies to adopt any alternative fuel option. Without external funding, only rapid-charging battery electric buses (BEBs) have lower ownership & social costs than a conventional diesel bus. When external funding is available to reduce bus purchase costs by 80%, BEBs become much more cost-effective. In this case, life cycle ownership & social costs of BEBs are 37-43% lower than conventional diesel buses. While including life cycle social costs does not change the ranking of alternative fuel options, a high-resolution estimate on the bus fleet's emissions in hotspot areas shows emissions reduction benefits from alternative fuel technologies (especially BEBs).

- *How should we use natural gas to power on-road vehicles (which natural gas pathway is preferred and why)?*

The findings in this dissertation suggest different strategies of using natural gas for different vehicle markets. Natural gas is best used in electric power generation than to produce gaseous or liquid fuels for powering on-road LDVs. The use of CNG and LNG for heavy-duty trucks may continue as there are less alternative fuel options but issues such as methane leakage should be addressed to avoid unintended environmental impacts. However, natural gas-based transportation fuels can at best partially mitigate climate change or air pollution damages, so renewable sources are ultimately needed to achieve sustainable transportation.

Chapter 2: ***Comparison of Life Cycle Greenhouse Gases from Natural Gas Pathways for Medium and Heavy-Duty Vehicles***.

- *Which natural gas pathways in which vehicle type provide greenhouse gas emissions reduction compared to petroleum fuel pathways?*

The emissions reduction potentials of natural gas pathways vary sharply between non-Class 8 medium- and heavy-duty vehicles (MHDVs) (e.g. pick-up trucks, parcel delivery trucks, and box trucks), Class 8 transit buses and Class 8 MHDVs (e.g. refuse trucks and tractor-trailers). battery electric vehicles (BEVs) with natural gas-based electricity provides at least 30% emissions reduction compared to petroleum pathways for non-Class 8 MHDVs and Class 8 transit buses. In addition, propane and compressed natural gas (CNG) pathways could reduce life cycle greenhouse gas (GHG) emissions for non-Class 8 MHDVs compared to the baseline petroleum fuels. On the other hand, none of natural gas pathways - CNG, liquefied natural gas (LNG), and Fischer-Tropsch liquids - achieves any emission reductions for Class 8 trucks compared to conventional diesel.

- *What are key uncertainties? How sensitive are the results to these uncertain variables?*

Key factors that determine the GHG emission reduction potentials of natural gas pathways include the choice of natural gas pathway, relative fuel efficiency of natural gas vehicles (relative to petroleum counterparts), life cycle methane leakage rate of natural gas pathways, global warming metrics, choices of baseline petroleum fuels, and payload losses in natural gas-fueled MHDVs compared to conventional MHDVs. Of these factors, life cycle methane leakage rate of natural gas pathways and global warming metrics are highly uncertain.

While there has been significant research progress on measuring methane leakage from natural gas systems as well as from natural gas vehicles, there are still large differences between bottom-up and top-down methane leakage rates measured for much of the U.S. As a result, I use the scenario analysis (baseline estimates vs. pessimistic estimates) to analyze the likely impacts from different methane leakage assumptions. In the meantime, I calculate the break-even methane leakage rates for CNG and LNG pathways. The break-even rates for CNG are 1-2.5% (100-year global warming potential, GWP), or 0-1% (20-year GWP); 0-1% for LNG (100-year GWP). These results show that CNG and LNG cannot provide emissions reduction at the current levels of methane leakage rates (likely 1-3%).

Global warming metrics (such as GWP) is uncertain because the time horizon used should be determined by the policymakers or the society as a whole. Because methane is a short-term climate forcer, a good practice is to use GWP both in 20 years and in 100 years. The impact of global warming potential increases with a higher methane leakage rate. Except BEVs with natural gas-based electricity and propane, all the other natural gas pathways see large increases when using 20-year GWP than with 100-year GWP. CNG's emissions reduction potential only exists with 100-year GWP.

Chapter 3: ***Comparison of Life Cycle Greenhouse Gases from Natural Gas Pathways for Light-Duty Vehicles.***

- *Which natural gas pathways in which vehicle type provide greenhouse gas emissions reduction compared to petroleum fuel pathways?*

A battery electric vehicle (BEV) powered with natural gas-based electricity achieves around 40% life cycle emissions reductions when compared to conventional gasoline. Gaseous hydrogen FCEVs and CNG vehicles have comparable life cycle emissions with conventional gasoline, offering limited reductions with 100-year global warming potential (GWP) yet leading to increases with 20-year GWP. Other liquid fuel pathways (methanol, ethanol, and Fischer-Tropsch liquids) have larger greenhouse gas (GHG) emissions than conventional gasoline even when carbon capture and storage technologies are available.

- *What are key uncertainties? How sensitive are the results to these uncertain variables?*

Similar to the findings in *Chapter 2,* life cycle methane leakage rate of natural gas pathways and global warming metrics are highly uncertain. Again, I have calculated the break-even methane leakage rates to understand what the maximum methane leakage rate is. With the current vehicle technologies, the break-even methane leakage rates for compressed natural gas (CNG) vehicles, gaseous hydrogen FCEVs, and BEVs are 0.9%/2.3%, 1.2%/2.8%, and 4.5%/10.8% (20-year GWP/100-year GWP). These results show that hydrogen fuel cell electric vehicles (FCEVs) and BEVs charged with natural gas-base electricity can provide emissions reduction at the current levels of methane leakage rates (likely 1-3%). In fact, BEVs are able to provide emissions reductions at even higher methane leakage rates.

The emissions reduction potentials of FCEVs is also sensitive to the assumptions on fuel cells replacement during vehicle life time. For the baseline assumptions, I assume fuel cells work for the entire lifetime of the vehicle. However, if consumers need to replace fuel cells during vehicle lifetime, increased manufacturing emissions of FCEVs would cause hydrogen pathways to have larger life cycle GHG emissions than gasoline vehicles.

Chapter 4: ***Life Cycle Air Pollution Damages of Petroleum and Natural Gas Pathways for Powering Light-Duty Vehicles and Heavy-Duty Vehicles.***

- *What are the life cycle air pollution damages from petroleum fuel pathways in each of the vehicle type considered?*

Systematic differences exist in the life cycle air pollution damages of petroleum pathways across vehicle types. Passenger cars have lowest damages per vehicle mile travelled (VMT) of all vehicle types while median per-VMT life cycle damages from SUVs, transit buses, and tractor-trailers are around 1.5 times, 3 times, 5 times, and 10 time larger, respectively. In the meantime, there is also high spatial variation in life cycle damages across counties.

The median, minimum, and maximum life cycle damages (using the EASIUR model, or using the AP2 model) for passenger cars, sports utility vehicles (SUVs), transit buses, long-haul tractor-trailers, and local-haul tractor-trailers are 0.45 (0.3-2.11) or 0.5 (0.32-4.25) cent/VMT, 0.62 (0.38-3.11) or 0.66 (0.41-4.82) cent/VMT, 1.71 (0.82-10.73) or 1.75 (1.01-8.04) cent/VMT, 2.09 (0.56-17.31) or 2.07 (0.65-16.24) cent/VMT, and 4.93 (0.98-44.27) or 4.54 (0.99-42.25) cent/VMT.

- *What are the life cycle air pollution damages from natural gas fuel pathways in each of the vehicle type considered? Which natural gas pathway reduces air pollution damages compared to petroleum fuel pathways in each of the vehicle type considered?*

To better compare air pollution damages from natural gas fuel pathways and petroleum fuel pathways, I discuss the relative reductions in air pollution damages when replacing petroleum fuels with natural gas fuel pathways. I find that battery electric vehicles (BEVs) powered by increased penetrations of natural gas-based electricity provide largest damage reductions in almost all counties (100%, 99%, and 75-93% of counties) for passenger cars, SUVs, and transit buses. However, BEVs powered by current grid electricity see largest increases in damages in the majority of counties for passenger cars, SUVs, and transit buses (81-83%, 85-86%, and 97-99% of counties). Compressed natural gas (CNG) is more likely to increase rather than reduce damages compared to gasoline when used in passenger cars and SUVs - only 18-22% of counties see reductions from CNG. However, CNG has the largest damage reduction potential when replacing diesel in tractor-trailers. Liquefied natural gas (LNG) high-pressure direct ignition (HPDI) has similar life cycle damages with CNG but LNG-SI is worse due to energy efficiency penalties. Indeed, CNG and LNG-HPDI trucks achieve damage reductions in 76-99% of counties for local-haul tractor-trailers but in only 32-71% of counties for long-haul tractor-trailers.

- *What are the best and the worst fuel pathway for each vehicle type in each county?*

BEVs charged with increased penetrations of natural gas-based electricity are the best pathway for light-duty vehicles (LDVs) and transit buses in almost all counties. Diesel hybrid electric vehicles (HEVs) are the best bus technology in the Rocky Mountain regions and western Texas. LNG-HPDI is the best pathway for tractor-trailers in West Coast and Rocky Mountain regions, as the electricity damages of the liquefaction process are lower in these regions. CNG sparking-ignition (SI) and diesel HEVs are the best pathways for tractor-trailers in other parts of U.S. The baseline petroleum fuels are the worst pathway for all vehicle types except transit buses in part of the U.S. (West Coasts for LDVs and heavy-duty vehicles (HDVs), and eastern U.S. for trucks), suggesting that replacing petroleum fuels with any alternative fuel is likely to reduce life cycle air pollution damages. BEVs charged with grid electricity have the highest damages for much of the eastern U.S. due to significant damages of the grid electricity. New England and western U.S. have cleaner electricity grid so BEVs charged with grid electricity are not the worst pathways in these regions. CNG-SI transit bus is the worst in the western U.S. because of significant tailpipe CO emissions. LNG-SI is the worst in tractor-trailers in parts of the U.S. (western Texas, western Midwest, and Rocky Mountain) because of the shift in tailpipe emissions (more than 90% NO_x reduction compared to diesel buses while increasing CO emissions by a factor of 8) and high electricity damages in the liquefaction process.

- *How sensitive are the results to emissions data and marginal damage model used?*

I find that the two emissions data sources (U.S. EPA's National Emissions Inventory and GREET model) result in significant differences in life cycle air pollution damages of petroleum fuels. Because petroleum fuels are the baseline fuels for all vehicle types, such a change affects the damage reduction potentials of alternative fuel pathways. In particular, the percentage of counties that find lower damages from CNG than gasoline in LDVs increases from 18-22% (depending on the marginal damage model used) to 100% when higher damages of petroleum fuels based on the GREET model are used. Similarly, LNG-SI and LNG-HPDI trucks see a large increase in the percentage of counties that find their damages to be lower than diesel. The sensitivity of pathway comparisons on emissions data calls for better data collection on criteria air pollutant (CAP) emissions from key facilities (such as oil refineries).

I find systematic differences in the marginal damages of CAP species (in particular, SO_2 and NO_x) between the AP2 model and the EASIUR model have large impacts on the findings for some regions of the U.S., such as the Rocky Mountains, western Texas, and New England. I emphasize that marginal damages of CO and volatile organic compounds (VOCs), for which transportation sector is responsible for more than half and over 25% emissions in the U.S., are relatively poor. Currently I cannot fully examine the impacts due to marginal damages of CO and VOCs.

Chapter 5: ***Should We Build A National Infrastructure to Refuel Natural Gas Powered Trucks?***

- *How many refueling stations have to be built to form a national natural gas refueling infrastructure for long-haul trucks in the U.S.?*

The total refueling capacity needed is relatively constant when the share of natural gas trucks is 1-5% of the total long-haul truck fleet, and that for higher shares, the total refueling capacity is almost linear with respect to the share of natural gas trucks. Faced with a very low adoption rate of natural gas trucks (1%), a national natural gas refueling infrastructure would require only 92-166 refueling modules, depending on the vehicle range assumption. On the other hand, a complete conversion of existing truck fleet to natural gas trucks would require a natural gas refueling infrastructure consisting of 1,231-1,263 refueling modules, accounting for vehicle range assumptions from 200 miles to 600 miles.

- *What are the economic costs (capital investments) and economic benefits (annualized profits) of such a refueling infrastructure?*

For small shares of the truck fleet powered by natural gas fuels (1-10%), a national natural gas refueling infrastructure would require only 92-203 refueling modules and a capital investment of $230-508 million. However, at adoption rates lower than 12.5%, building a national natural gas infrastructure may lead to economic losses for baseline economic assumptions. Higher fuel price margin and lower capital cost of a refueling module can improve the economic viability of the refueling infrastructure investment.

- *How do the capacities and economic status of the refueling infrastructure correspond to different shares of natural gas trucks in the long-haul truck fleet?*

The capacity of the refueling infrastructure (i.e. number of refueling modules) is largely driven by the portion of long-haul trucks powered by natural gas fuels. When the share of natural gas trucks is higher than 5%, the total refueling capacity grows almost linearly with respect to the penetration rate. However, if the share of natural gas trucks is less than 5%, the total refueling capacity needed is fairly constant. Specifically, 92-106, 124-130, and 166-168 refueling modules are needed if the range of natural gas truck is 200 miles, 300 miles, or 600 miles. The flat slopes of refueling capacities suggest that refueling infrastructure needed to ensure refueling coverage has overcapacity issues for low shares of natural gas trucks.

- *How sensitive are the capacities and economic status of the refueling infrastructure to different vehicle range assumptions of natural gas truck?*

There is an inverse relationship between number of refueling station sites and vehicle range. The longer the vehicle range, the sparser the refueling network can be. I find that the impact of vehicle range on the number of refueling station sites is less salient at high penetration rates than at low penetration rates. As a result, vehicle range has negligible impacts on the economic viability of the refueling network at high adoption rates (≥ 10%).

- *How sensitive is the economic status of the refueling infrastructure to different economic assumptions?*

In the sensitivity analysis, I examined the following sensitivity variables, refueling module capital cost, fuel price margin, discount rate, and lifetime at different adoption rates. Scenarios with either low fuel price margins, high capital costs of a refueling module, or high discount rates, result in net economic losses. The NPV of the refueling infrastructure investment is more sensitive to fuel price margin and capital cost of a refueling module than discount rate or lifetime, as shown by the slopes of the contour lines. Lifetime alone does not have a large effect on the economic viability of the refueling infrastructure.

- *Is there a way to build a national natural gas refueling infrastructure for long-haul trucks in light of the results?*

My analysis shows that a transition to natural gas fuels in long-haul trucks requires a high adoption rate (>5%) of natural gas trucks and a high fuel price margin of the natural gas fuels (~$0.1/DGE). Both these factors are highly influenced by the diesel fuel price. With a high diesel price, truck fleets are more likely to switch to natural gas fuels for economic savings and refueling infrastructure owners can ask for a higher price margin. However, the current diesel price is at the lowest point since 2005.[237] As a result, the adoption of natural gas is still below 3% of the heavy-duty trucks and buses fleet.[214] These facts do not bode well for the economic viability of the existing and new natural gas refueling infrastructure.

However, opportunities of natural gas fuels may still exist in regional markets. Indeed, the analysis on regional refueling infrastructure shows that much less refueling stations are needed to ensure highway coverage in California and Texas. Furthermore, regional factors that have not been taken into account yet – such as higher price differential between natural gas fuels and diesel,[227,228] attractive state incentives,[226] and strong environmental regulations – would help a swifter adoption of natural gas fuels in California and Texas. Finally, technology innovation or policy interventions that reduce capital costs of refueling modules or increase fuel price margin improve the economic viability of the refueling infrastructure, especially for low shares of natural gas trucks in the total truck fleet.

Chapter 6: *Life Cycle Ownership and Social Costs of Alternative Fuel Options for Transit Buses.*

- *Which alternative fuel technology has the lowest costs when only considering life cycle ownership costs and life cycle ownership & social costs?*

Without external funding, conventional diesel is among the cheapest in terms of both life cycle ownership costs and life cycle ownership & social costs. For a 40-foot transit bus, only rapid-charging battery electric buses (BEBs) have lower ownership & social costs than a conventional diesel bus without external funding. The advantages of BEBs are their high vehicle efficiency, low electricity rates in PA, and low operation and maintenance (O&M) costs for BEBs as fewer mechanical devices and pollution control devices are needed. Surprisingly, diesel hybrid electric buses (HEBs) have the largest ownership & social costs among all technologies considered. This is because of a high bus purchase price premium and a low return in reduced fuel costs over the lifetime. When external funding is available to reduce bus purchase costs by 80%, BEBs become much more cost-effective. In this case, life cycle ownership & social costs of BEBs are 37-43% lower than conventional diesel buses. Other bus options still cost more than a conventional diesel bus in terms of life cycle ownership & social costs (+1%, +2%, +2%, +5%, and +16% for B20, diesel HEB, compressed natural gas (CNG), B100, and LNG respectively).

- *Which alternative fuel technology has the lowest life cycle social costs?*

Biodiesel (B100 and B20) and diesel HEBs have lower social costs compared to conventional diesel for both 40-foot and 60-foot buses. On the other hand, liquefied natural gas (LNG), CNG and slow-charging BEBs have higher social costs than conventional diesel, and LNG 60-foot buses more than double social costs of conventional diesel. The leading source for high social costs of these pathways is the grid electricity, which has high SO_2 and NO_x emissions. Specifically, BEBs charge and store the grid electricity to power buses, and liquefaction and compression of natural gas are both intensive in electricity use. In addition, LNG also has higher greenhouse gas (GHG) emissions than conventional diesel, contributing further to high social costs. Contrary to LNG, BEBs have low GHG emissions due to their high vehicle efficiency. While BEBs completely reduce criteria air pollutant (CAP) emissions from tailpipe exhausts, damages associated with CAP emissions from grid electricity are quite significant, and more than cancel out BEBs' climate change mitigation benefits.

- *Does the inclusion of life cycle social costs change the rankings of alternative fuel technologies?*

For the baseline results, I find that including life cycle social costs does not change the rank of technologies. This is because social costs are small compared to ownership costs. For 40-foot buses, the ratio between social costs and ownership costs fall between 4% and 7% (without external funding), or 6% and 16% (with external funding) -- with biodiesel and conventional diesel on the lower end and BEBs on the higher end. A similar pattern exists for 60-foot buses although the range of ratios becomes 3-7% (without external funding) or 5-

16% (with external funding). Nevertheless, technology assessments that ignore these social costs are incomplete because these are actual costs paid by people not just the emitter.

- *Does government funding play an important role in selecting the best alternative fuel technology?*

Yes, the availability of external funding is crucial for transit agencies to adopt any alternative fuel option. Currently, the U.S. Federal Transit Administration (FTA) Section 5307 provides funding that may cover up to 80% of bus purchase costs. Without external funding only rapid-charging BEBs reduce ownership & social costs compared to diesel, while other options increase ownership & social costs. When external funding is available to pay for 80% of vehicle purchase expenditures, BEBs yield large reductions (39-45%) in ownership & social costs compared to diesel. In addition, B20, diesel hybrid electric bus, and CNG buses all have similar life cycle ownership & social costs with conventional diesel when external funding is available.

- *How sensitive are the life cycle ownership and social costs sensitive to economic and operation assumptions?*

I performed a sensitivity analysis for a 40-foot transit bus on higher diesel prices, lower annual bus mileage, higher electricity rates, higher infrastructure costs, and higher discount rates. I find that all five factors, independently or jointly, do not change the conclusions that BEBs achieve large reductions in ownership & social costs compared to conventional diesel, when external funding is available. When external funding is not available, I find that lower annual mileage and a higher discount rate have large impacts on the cost reduction potential of BEBs. When annual mileage is lower than 31,600 miles/year or discount rate is higher than 4.15%, the benefits of rapid charging BEBs with regard to conventional diesel diminish. On the contrary, doubling electricity rates or doubling per-bus infrastructure costs (either through higher infrastructure costs or smaller bus fleets to use the rapid charging infrastructure) do not change the (+ or -) sign of the comparison between rapid-charging BEBs and conventional diesel buses, even though BEBs' cost reductions become smaller.

- *Do transit buses a major contributor to air pollutions in the hotspot areas in Pittsburgh, PA?*

No. Port Authority of Allegheny County (PAAC)'s bus fleet only contributes to slightly more than 1% of $PM_{2.5}$ emissions from all mobile sources in Allegheny County. However, it is worth noting that reduction of $PM_{2.5}$ emissions are important to human health. Literature shows that diesel particular matter (DPM) is the leading additive cancer risk air toxic in Downtown Pittsburgh and in Allegheny County.

- *Which alternative fuel technology provides largest reduction benefits in the hotspot areas in Pittsburgh, PA?*

BEBs can eliminate all tailpipe emissions (but still have $PM_{2.5}$ emissions from break and tire wear), achieving the best emissions reduction potential of all technologies considered. Diesel HEBs reduce SO_2, volatile organic compounds (VOCs), and CO emissions but increase NO_x emissions by 50% relative to new diesel buses. LNG and CNG buses reduce SO_2, NO_x, and VOC emissions but increase CO emissions significantly by a factor of 64. Furthermore, alternative fuels (CNG, LNG, BEBs) have the added benefit of reducing cancer risk by replacing diesel buses in Downtown Pittsburgh and Allegheny County.

- *Are there any factors that the life cycle cost framework cannot account for? How to address these factors?*

Yes. While BEBs are estimated to have the smallest life cycle ownership costs and/or social costs, both types of BEBs face practical challenges to immediate operation for a typical bus route. First, BEBs have limited ranges (33-41 miles and 104-130 miles for rapid-charging and slow-charging BEBs), which are significantly smaller than other bus technologies, and would require special routes or specialized planning and scheduling. Indeed, rapid-charging BEBs require tight control of bus schedules to ensure a bus is charged at a specific bus stop and time. Even though buses are operated on a planned schedule, the actual schedule is determined by traffic congestion, weather and other road factors. As a result, bus routes on dedicated bus lanes or fixed busways may be more feasible for rapid-charging BEBs. Additionally, BEBs require dedicated charging infrastructure, which, in addition to higher capital expenditures and O&M costs, require land to install and coordination with local utilities. Finally, charging infrastructure for BEBs are currently not compatible among bus manufactures.

One way to address these issues is to carefully select bus routes that BEBs are feasible to use. In the meantime, any transit agency interested in BEBs can wait or experiment with BEBs before significant transition to BEBs. Several trends may make BEBs more attractive in the near future. BEBs will become more technologically mature as more buses are delivered and operated across the country. The costs of batteries are declining rapidly while the performance is improving quickly 267 due to increased battery deliveries in light-duty vehicle markets. Thus, future BEBs will have better economics and longer range.

7.2. Research Contributions

This dissertation provides an up-to-date estimate on the economic and environmental implications of using natural gas to power on-road vehicles. It also sheds light on how to reduce environmental externalities from on-road vehicles, as well as how to best use natural gas for transportation sector from the society-wide perspective.

This work (*Chapter 2-Chapter 4*) contributes to the literature on the environmental externalities caused by life cycle GHG emissions and CAP emissions from on-road vehicles in the following ways. First, compared to existing studies, this work is first to provide a complete estimate on life cycle GHG emissions and life cycle air pollution damages for vehicle types other than passenger cars. Second, this work is the first to systematically compare the impacts of emissions data sources and different marginal damages on the life cycle air pollution estimates whereas most of the existing studies used the GREET model and the APEEP model. Third, this work extends 'break-even methane leakage rate idea' proposed in Alvarez et al. (2012)[28] and is the first to quantify the relationship between break-even methane leakage rate with respect to relative vehicle fuel efficiency. Forth, this work extends the use of Monte-Carlo simulation in the life cycle assessment to account for uncertainty and variability explicitly, following Venkatesh et al. (2011).[27]

Chapter 6 extends the framework and method of economic assessments on alternative fuel options by including life cycle social costs of unintended air pollutants and uses transit buses as a case study to illustrate how to build and the impacts of the extended framework. In addition, *Chapter 6* estimates emissions from bus fleets in hotspot areas to show the implications of high-resolution emissions estimates

Chapter 5 contributes to the literature on alternative fuel refueling infrastructure in the following ways. First, it is one of the first to estimate a national refueling infrastructure for long-haul trucks. Second, it proposes a heuristics of building refueling stations at highway intersections. The analysis in *Chapter 5* confirms that such locations at highway intersections see much higher truck flows and thus refueling demands than other locations. Finally, *Chapter 5* highlights the nonlinear impact of adoption rate of natural gas trucks (share in the total truck fleet) on the number of refueling station sites, capacities and economics of the refueling infrastructure. This finding may motivate future researchers to examine the market dynamics at low adoption rates where the refueling infrastructure faces over-capacity issue and is likely to suffer from economic loss.

7.3. Discussion and Policy Implications

Because the nature of the research questions addressed by this dissertation, the findings as well as the analytical approaches in this dissertation have ample policy implications for a wide range of audiences.

7.3.1. Accounting for Environmental Externalities.

This dissertation shows that environmental externalities, especially caused by GHG and CAP emissions, can and should be accounted. While the exact damage estimates still remain uncertain and are sensitive to emissions data (for instance, methane leakage rate for GHGs, and oil refinery emissions for CAPs) and impact metrics (for instance, global warming potential for GHGs, and marginal damage estimates for CAPs), the emissions and damages estimates fall in the same ballpark. In addition, the research progress in these areas is improving fast. Data, models, and tools have become more available, accessible, and refined than ever.

Use the life cycle scope. Findings in *Chapter 2-Chapter 4* highlight the need to design policies that consider the life cycle of vehicle use rather than focuses on just one stage (typically vehicle use) of the life cycle. For instance, some counties that see reduction in life cycle air pollution damages in vehicle operation from replacing petroleum fuels with an alternative fuel pathway may find an increase in life cycle air pollution damages. Similarly, while most natural gas fuel pathways reduce GHG emissions from the use phase (vehicle tailpipe), only a few pathways reduce life cycle GHG emissions compared to the baseline petroleum fuels.

***Account for uncertainty and variability*.** In this dissertation, I find that the environmental impacts of natural gas pathways have large uncertainty and variability. While improved data collections and analytical tools such as sensitivity analysis, scenario analysis, Monte-Carlo simulation, and bounding analysis (break-even analysis) can help address uncertainty and variability, there does not exist a set of magic estimates. As a result, policymakers should consider uncertainty and variability when they set policy goals based on relative or absolute emissions.[178] For instance, policy goals should be robust if an alternative emissions data is used. It is for the same reason that the discussion in this dissertation emphasized reduction potentials and factors that change key findings rather than focusing on specific numbers in the results.

7.3.2. Implications for Current Policies

The findings in *Chapter 2-Chapter 4* are immediately relevant to current policy debates such as the Corporate Average Fuel Economy (CAFE) standards for light-duty vehicles (LDVs) and medium- and heavy-duty vehicles (MHDVs),[108] the emissions standards for light-duty engines and heavy-duty engines,[209] and the Low Carbon Fuel Standard (LCFS)[107] in California. In addition, the break-even analysis in *Chapter 2* and *Chapter 3* provides policy-relevant bounds on the methane leakage rates from the perspective of transportation technologies. These break-even estimates could of interests to policy discussions on methane regulations in oil and natural gas systems in U.S. While the exact numbers in this dissertation may not be directly used by these regulatory agencies, the analytical framework, the assumptions, and insights from this dissertation are of value.

Furthermore, the findings in this dissertation may suggest areas that are worth policy interventions. For instance, the high uncertainty and sensitivity of life cycle GHG emissions on methane leakage rate highlights the need for better data collections. More transparent reporting requirements (such as U.S. EPA's GHGRP program[98]) and more on-site fugitive methane measurements on natural gas systems and natural gas vehicles (such as EDF's efforts[106]) are crucial to solve the ongoing debates regarding methane leakage and to identify emission reduction opportunities which can then be implemented via cost-effective technologies or stringent regulations.[119–123] Similarly, CAP emissions data, especially those related to large energy facilities (such as oil refineries) are important to estimate and understand the life cycle air pollution damages of transportation technologies.

Finally, I emphasize that this dissertation did not study the design or implementation of specific policies, which require estimating emissions or damages reduction as well as figuring out the costs of emissions mitigation.

7.3.3. Implications for Refueling Infrastructure Investors

The findings in *Chapter 5* help refueling infrastructure investors and other interested audiences to understand the economics implications of building a national natural gas refueling infrastructure. In particular, the results highlight the impacts of the adoption rate of natural gas trucks on the economic viability of the refueling infrastructure. In addition, it illustrates the impact of vehicle ranges on the locations and capacities of refuelling infrastructure. The finding in this analysis suggests that refueling demands from natural gas trucks are the key for the fueling infrastructure. The finding suggests prioritizing building refueling stations at highway intersections and starting regional refueling coverage in states such as California and Texas before building a national refueling infrastructure.

7.3.4. *Implications for Transit Agencies*

Chapter 6 provides an assessment framework that can be used (with updated assumptions) to compare alternative fuel technologies for transit buses. In the meantime, it embodied a consulting effort for Port Authority of Allegheny County (PAAC) whose specific data are used as the baseline assumptions in the analysis. In addition, *Chapter 6* discusses in detail the practical challenges and potential future benefits of operating battery electric buses (BEBs) in addition to those benefits and costs already included in the life cycle ownership and social costs. For instance, I highlighted the challenges brought by limited vehicle range and the infrastructure investment and modifications related to electricity supply and land use. On the other hand, I believe BEBs could help transit agencies operate in more intelligent transportation systems that are likely to happen in the near and medium futures.

7.4. Future Work

In this section, I briefly discuss future work that addresses the limitations in this dissertation. There are, of course, much more future work that can directly or indirectly stem from this dissertation. For instance, a system analysis on alternative fuel technologies that considers fuel supply chains, vehicle technologies, refueling infrastructure, and consumer behaviors on the adoption and use of alternative fuels would benefit from the frameworks, results, and insights generated in this dissertation.

7.4.1. *Real-World Factors and Variation in Vehicle Fuel Efficiency*

In this dissertation I have examined the effect that economic variables, emissions factors, emissions activities, marginal damages of emissions, process and vehicle technologies, and vehicle use uses have on overall emissions. I have used point estimates for vehicle fuel economy for a selection of alternative fuel vehicles for a given vehicle type. In particular, I compare vehicles from the same vehicle manufacturers and the same vehicle design as much as possible to eliminate bias in vehicle designs and production. I rely on measured (where possible) and estimated (reported in literature) fuel economy of new vehicles for fuel economy assumptions used in this thesis (details on assumptions are available each chapter and Appendices A and B).

However, if the goal had been a representation of the mix of current and future vehicle fleet, the variation in fuel efficiency would play a role in the overall emissions. Two key factors would need to be taken into account: (i) within each vehicle type ('comparable vehicles'), vehicle efficiency varies widely for different vehicle designs. For instance, the fuel economy of conventional gasoline subcompact passenger vehicles ranges from 15 to 37 MPG.[270] This variation of more than 100% would shadow the potential benefits of alternative fuel pathways as no natural gas pathway provides deep (>50%) reduction (with the exception of air pollution damages in certain counties). If I were to assume that the comparative vehicle have an MPG of 37 MPG (a more efficient baseline vehicle) then only electric vehicles (BEVs, FCEVs, and HEVs) achieve reduction in life cycle GHG emissions. (ii) the vehicle fuel efficiency of a vehicle varies under different use conditions, such as drive cycle, terrain, payload, and weather. While these factors are outside the scope of my analysis, I note that Reyna et al. (2014) examined the impact of drive cycle (speed, and congestion), road grade and road type, and vehicle age on GHG and CAP emissions.[104] They find that the variability in GHG and CAP emissions for LDVs are -2% to 11% and -47 to 228% when compared with the average characteristics of the U.S. driving condition. For HDV, the variability in GHG and CAP emissions is -21 to 55% and -32 to 174%, respectively. Yuksel et al. (2015) find that "annual energy consumption of BEVs can increase by an average of 15% in the Upper Midwest or in the Southwest compared to the Pacific Coast due to temperature differences".[175] What is more, these factors may affect emissions in ways other than impacting the fuel efficiency of the vehicles. For instance, air pollutant emissions from a cold-start vehicle is significantly higher than a hot-start vehicle,[271] which shows the impact of drive cycle on vehicles. Payload affects per-payload emission metrics significantly if the actual payload information is available.[45,265]

In sum, the variability in vehicle fuel efficiency of the current and future fleet, and real world driving conditions are likely to affect life-cycle emissions in significant ways and should be the focus of future studies.

7.4.2. Consequential Life Cycle Assessment

Because alternative fuel technologies are emerging rather than existing technologies, the penetration of alternative fuel technologies are likely to cause system-wide changes. For instance, charging of BEVs cause additional electricity demand which would dispatch otherwise offline power plants to generate electricity. As a result, it is more appropriate to use consequential LCA or at least marginal emission factors instead of average emission factors to estimate actual emissions impact from the grid. While this dissertation did not assume marginal generation from the electricity grid, several studies have estimated marginal emission factors or used marginal emission factors to estimate the health, environmental, and climate benefits of vehicle and renewable electricity technologies.[192,272–274]

7.4.3. Alternative Global Warming Metrics

In *Chapter 2* and *Chapter 3*, I used global warming potential (GWP) to convert mass emissions of non-CO_2 gases to CO_2 equivalent emissions. Recent literature suggests that GWP has serious limitations. For instance, GWP treats all emissions as if they are pulse emissions at the beginning of the time horizon considered, thus completely ignoring different effects of emissions happening at different time.[28,124–127] Further, while GWP is closely related to radiative forcing, GWP does not consider other drivers of climate change, such as the rate of change, and variations in surface temperature response.[128] Some research is ongoing to develop more appropriate climate impact metrics,[124–127] but there is no consensus about the use of these metrics for LCA and a comparison of such metrics is beyond the scope of this study. In the future, as more appropriate metrics are identified, I can use the inventory results in these two chapters to re-evaluate the climate impacts of natural gas-based transportation fuels.

7.4.4. Improved Marginal Damage Estimates of CAPs

In *Chapter 4*, I found that systematic differences in the marginal damages of CAP species (in particular, SO_2 and NO_x) between the two models (AP2 model and EASIUR model) have large impacts on the 'best'/ 'worst' pathway analysis for some regions of the U.S. In addition, the marginal damage of CO used in this work is likely to be outdated and literature suggests high uncertainty of the marginal damages of VOCs.[57] Last but not the least, cancer risks of diesel particular matter are not monetized at all.[262] When updated marginal damages on these CAPs become available, an update on the life cycle air pollution damages is certainly needed.

One way to improve marginal damage estimates is to develop methods and tools to estimate CAPs' health and environmental impacts in higher spatial and temporal resolutions. *Chapter 4* explored in this direction by building a high-resolution emissions inventory using operation and schedule data of the transit bus fleet and emissions factors of transit buses. Future work is needed to link high-resolution emissions inventories with exposed population and health impact end points. Such a high-resolution marginal damage model will better characterize the heterogeneity of air pollutant impacts, particularly in urban areas.

7.4.5. Refueling Infrastructure Modeling

There are multiple ways to improve the refueling infrastructure model. For instance, factors such as land acquisition (or conversion of existing diesel refueling stations or truck stops), supporting infrastructure, and refueling market competition, once taken into account, may lead to different refueling infrastructure location strategies. Fuel price differences across regions, congestion at refueling stations or truck stops, driving hour regulations, are also ignored in the current work and can be the focus of future work. In addition, the refueling infrastructure model can be extended to include production and logistics of natural gas transportation fuels.

7.4.6. Comparison of Air Emissions' Social Costs across Freight Modes

This dissertation analyzed the economic and social impacts of one freight movement mode, long-haul freight trucks. In reality, there are other modes, such as ships and rail, to compete for freight movement. Compared to freight trucks, ships and rails receive less public and regulatory attention. While ships and rail have lower energy intensity, they could have large environmental externalities, especially local air pollutions near urban fright center. Furthermore, natural gas fuels (especially LNG) may play a role due to its potential for cost savings and emissions reduction. An analysis on the social costs of air emissions from ships and rail would address this knowledge gap.

Appendix A. Supporting Information for Chapter 2

A.1. Units and Metrics

A.1.1 Global Warming Potential (GWP)

This study considers the following greenhouse gases (GHGs): carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). I convert emissions of different GHGs into the CO_2-equivalent (CO_2-eq) emissions by multiplying the mass of emission to the global warming potential (GWP) of each gas as reported in the 5th Assessment Report of the Intergovernmental Panel on Climate Change (IPCC AR5),[2] and as listed in **Table A.1**. I consider fossil fuel methane emissions, and include the uncertainty in GWPs using a normal distribution.[128] The mean and standard deviations are estimated from the IPCC AR5.[2]

Table A.1. Global Warming Potential (GWP) values used.[2]

Greenhouse Gas	100-yr	20-yr
CO_2	1	1
CH_4	Norm (36, 8.5)	Norm (87, 15.9)
N_2O	Norm (298, 52.5)	Norm (268, 34.2)

Note: Values include climate-carbon feedbacks of non-CO_2 gases and fossil methane.

A.1.2 Fuel Properties and Emission Factors from Combustion

In this section, I report properties (such as energy density, mass density), and emissions factors of energy carriers used in this study (**Table A.2**). There are two reported values for energy contents, LHV (lower heating value) and HHV (higher heating value). MacLean and Lave[72] suggested using LHV for mobile use (such as in vehicles) and HHV for stationary use (such as in power plants and fuel production plants). To be consistent, I use LHV for all energy sources in this study.

I rely on the GREET model[159] for all properties of energy carriers to maintain consistency, even though there is a noticeable difference in the energy content (HHV) for dry natural gas used in existing studies: 1,089 BTU/cf in the GREET model;[159] 1,030 BTU/cf in Jaramillo et al. (2007),[138] Venkatesh et al. (2001),[27] and Jiang et al. (2011).[139]

I calculate the combustion emission factor of natural gas based on Venkatesh et al. (2011).[27] The combustion emission factor of natural gas reported in Venkatesh et al. (2011)[27] follows a normal distribution with a mean of 50 gCO_2-eq/MJ_{HHV} and a standard deviation of 0.7 gCO_2-eq/MJ_{HHV}. To convert it to a LHV-basis, the ratio of natural gas HHV and LHV has to be applied.

Table A.2. Energy content and emission factor for different energy carriers.

Fuel	Energy density	Mass density	Combustion emissions factor
Unit	BTU/cubic foot or BTU/gallon	Unit: gram/cubic foot or gram/gallon	gCO_2-eq/MJ_{LHV}
Liquid Fuels (at 32F and 1atm)			
Conventional Gasoline	112,194	-	See section A.3. Life Cycle GHG Emissions of Gasoline and Diesel
Oil sand gasoline		-	
Conventional Diesel	128,450	-	
Oil sand diesel		-	
Methanol	57,250	3,006	68.4[b]
Ethanol	76,330	2,988	70.9[b]
LPG	84,950	1,923	64.5[b]
Ethane	20,295 (Btu/lb)[a]	-	-
Butane	94,970	2,213	-
n-Hexane	105,125	2,479	-
Gaseous Fuels (at 32F and 1atm)			
Natural gas	983	22.0	Normal dist. (50, 0.7)/983*1089[c]
Pure methane	962	20.3	-
Gaseous hydrogen	290	2.6	-

Note: a. The energy density of ethane is used in modeling the ethane steam cracking process. And the source of the value is http://www.engineeringtoolbox.com/heating-values-fuel-gases-d_823.html. b. The combustion emission factor of methanol, ethanol, and LPG are calculated using energy density, mass density, and carbon weight ratio from the GREET model[159]. c. The HHV of natural gas is 1,089 BTU/cubic foot.[159]

I distinguish different natural gas sources by their methane composition. Generally speaking, as natural gas flows from the well site to end users, its methane composition increases due to various processing and purification processes. **Table A.3** shows the methane composition for four types of natural gas. I note, however, that methane composition of natural gas varies by region,[152] so region-specific analysis may have slightly different results to those presented in this study.

Table A.3. Methane composition of natural gas.

Fuel	Methane composition	Reference
Natural gas (production)	0.894	U.S. EPA (2014)[152]
Natural gas (pipeline)	0.934	U.S. EPA (2014)[152]
CNG	0.934	Assumed to be the same as pipeline-quality natural gas
LNG	0.95	Foss (2007)[275]

A.2. Natural Gas Upstream GHG Emissions

A.2.1 Estimation Methods

Currently, the supply of natural gas in the U.S. is produced from four sources, conventional natural gas (roughly 43% in 2012), shale gas (roughly 35%), associated gas as a co-product of crude oil (17%), and coal-bed methane as a co-product of coal (5%).[10] As discussed in the main text, I focus on the shale gas upstream GHG emissions because of the increasing importance of shale gas in the U.S. natural gas supply.[109]

To estimate natural gas upstream GHG emissions, I follow the bottom-up life cycle assessment framework in existing studies.[27,115,139,142,151] The baseline year for this analysis on natural gas upstream emissions is 2011, the most recent year that data permits. I estimate emissions for the following five stages in the natural gas system: preproduction, production, processing, transmission and storage, and distribution. I assume that the first four stages are common to all fuels produced from dry natural gas, while the last stage (distribution) is only included in pathways with distributed fuel production designs, such as the CNG pathway, the distributed gaseous hydrogen pathway, and the distributed LNG pathway. In addition, natural gas liquids, from which ethanol and propane are produced, share the first three upstream stages as co-products and I allocate emissions between dry natural gas and natural gas liquids based on energy contents.

Preproduction Stage. In this framework, the preproduction stage refers to the well construction activities that happen at the beginning of the lifetime of a natural gas well. Emissions from the preproduction stages need to be normalized by the ultimate total recovery of natural gas over the well lifetime. Following Jiang et al. (2011),[139] I assume that the preproduction stage includes the following four stages: (1) well pad and access roads construction, (2) well drilling, (3) hydraulic fracturing, and (4) well completion. For the first three stages, I used Jiang et al. (2011)[139] estimates, as there is little evidence that GHG emissions from these activities have changed significantly. A recent paper by Caulton et al. (2014)[276] suggests high emissions during drilling operations, but that work relies on air-borne ambient air tests in South Western Pennsylvania. As the authors mention in the paper,[276] this area is well known for high availability of coal-bed methane that likely contributed to the higher concentration of air-borne methane. It is thus unclear how appropriate those estimates are for shale gas drilling and fracturing. I did update Jiang et al.'s (2011)[139] well completion estimates, as explained below.

Well completion refers to the stage between the end of well drilling and the start of routine production of a well, where fluids and natural gas flow back to the surface through the wellbore.[151] Various existing studies[97,115,151,152,277] have identified well completion as one of the most uncertain sources of methane emissions in the natural gas upstream stages. Furthermore, U.S. Environmental Protection Agency (EPA) has been revising the method used in how to estimate GHG emissions from well completions. While this effort had been ongoing for the past few years, U.S. EPA has not completed this process yet. In addition, there are policy changes in regulating the use of Reduced Emission Control (REC) technologies, or so-called "green completion," at both federal and state levels in the U.S.[121,122] To reflect the changing industry practices in well completion activities, I use the latest data set that is available from Allen et al. (2013).[97] Allen et al. (2013)[97] performed recent, facility-level emission tests in major U.S. shale gas plays. Although the data set only included 27 well completion events, the authors covered a large range of industry practices and technologies, as well as different geographic areas. Other facility-level data set, while suffering the same potential sample biases due to small samples and cooperation with industries,[114] is either outdated,[278] or less transparent.[98,99,152]

Allen et al. (2013)[97] performed on-site measurements of methane leakage in production sites over major natural gas production basins. The authors found that there are six types[i] of GHG

[i] To be more specific, the emission categories reported in Allen et al. (2013)[97] are "Flowback to open top tank; gases vented", "Atmospheric Vent from Tank handling liquid HC stream from Completion Separator", "Controlled (combusted) Vent from Tank handling liquid HC stream from Completion Separator", "Atmospheric Vent from Tank handling liquid water stream from Completion Separator", "Controlled (combusted) Vent from Tank handling liquid water stream from Completion Separator", and "Gas from overhead of completion separator, sent to flare (assumed 2.0% of methane is uncombusted in flare)."

emissions during well completion. Based on the type of GHGs eventually emitted, I combine these six types of emissions into two categories: natural gas that is vented, and natural gas that is flared. The total amount of natural gas that is either vented or flared can be calculated based on the *flowback rate*[i] and *flowback duration* data from Allen et al. (2013).[97] *Flowback rate* refers to the rate of natural gas coming out of the wellbore (in volume per hour) during well completion while *flowback duration* is the time length of well completion (hours).

I estimate the *flaring rate*, i.e., the percentage of the volumes of natural gas to be either flared or combusted, compared to the total volumes of natural gas from the well during well completion[151] for each well completion event. There are two distinct groups of data samples of well completion events in the study by Allen et al. (2013).[97] For the first group (Type I, 14 events), there is no flaring and all natural gas from well completion is vented directly into atmosphere; for the other group (Type II, 13 events), natural gas is mostly flared (the average flaring percentage is 0.94 with bounds being 0.80 and 0.98). Thus I model these two categories of well completion emissions separately and use a probability mixture model to combine them (assuming the probability of each category is equal[ii]), as shown in Eqn. A.1.

$$\begin{aligned} \textit{Well completion GHG emissions} \; = \; & 50\% \times \textit{Well completion GHG emissions} \; (\textit{Type I}) \; + \\ & 50\% \times \textit{Well completion GHG emissions} \; (\textit{Type II}). \end{aligned} \qquad \text{(A.1)}$$

The two categories of well completion emissions share the same estimation structure but each category has its own distributions for key variables. I model potential natural gas emissions from well completion with the *flowback rate*, and *flowback duration*. I then calculate the amount of natural gas flared or combusted as well as the amount of natural gas vented based on the *flaring rate* assumptions. Finally, I calculate the resulting GHG emissions of flared or combusted natural

[i] Allen et al. (2013)[97] did not report *flowback rate*, so we calculated it by dividing total natural gas emitted over *flowback duration* for each event.
[ii] The percentage of type-I wells at the U.S. national well is a key uncertain variable in estimating emissions from well completion. In the baseline scenario, we assume a 50%:50% spilt for type-I and type-II well completion events based on the percentage of those events[ii] in the dataset in Allen et al. (2013)[97]. Given that we only have a small sample from Allen et al. (2013)[97], this ratio is likely to change when more data is available. Furthermore, a probability mixture model of well completion emission factors for different types of well completion events is preferred if the Estimated Ultimate Recovery (EUR) data for different types of wells are known.

gas using the *combustion emission factor* of natural gas, the *combustion efficiency*,[i] and the *methane composition*[ii] *of natural gas*. I also calculate vented methane emissions based on *combustion efficiency*, *methane composition of natural gas*, and *methane density*. Eqn. A.2 shows the expression used to calculate GHG emissions from well completion.

$$
\begin{aligned}
&\text{Well completion GHG emissions } (\text{Type } i) = \\
&(\text{flowback rate} \times \text{flowback duration} \times \text{methane composition}) \times \\
&\left\{ \begin{array}{l} \text{flaring rate} \times \left[\begin{array}{l} \text{combustion efficiency} \times \text{methane combustion emission factor} \times GWP_{CO_2} + \\ (1 - \text{combustion efficiency}) \times \text{methane density} \times GWP_{CH_4} \end{array} \right] + \\ (1 - \text{flaring rate}) \times \text{methane density} \times \text{GWP of methane} \end{array} \right\}. \\
&(i = I, II).
\end{aligned} \tag{A.2}
$$

I summarize all the assumptions for the well completion stage (including those calculated from Allen et al. (2013)[97]) in **Table A.4**. I assume that well completions are done once per well lifetime, following the assumption from existing studies.[115,139,151]

Table A.4. Assumptions for GHG emissions from well completion.

Variables	I. Venting only	II. Venting and flaring
Flowback duration (hour)	Truncated lognormal (4.3, 1.0)[iii]	Truncated lognormal (3.6, 1.0)[iv]
Flowback rate (scf/hour)	Truncated lognormal (6.3, 1.7)[v]	Truncated lognormal (10.6, 1.6)[vi]
Flaring rate	0%	Uniform distribution (80%, 98%)
Methane composition in natural gas (by volume)	89.4%	
Methane density	20.3 gram/cubic foot	
Combustion efficiency	N/A	98%
Percentage of each type	50%	50%

[i] We assumed 2% of methane is uncombusted in flare, following Allen et al. (2013)[97].
[ii] Methane composition of natural gas from hydraulic fracturing wells is 89.4% for the U.S. average in year 2011[152].
[iii] The distribution is truncated at the minimal sample, and 10 times the maximal sample (14-3,390 hours). After truncation, the mean and standard deviation are 122 hours, and 146 hours, respectively.
[iv] The distribution is truncated at the minimal sample, and 10 times the maximal sample (10-1,640 hours). After truncation, the mean and standard deviation are 57 hours, and 69 hours, respectively.
[v] The distribution is truncated at the minimal sample, and 10 times the maximal sample (30-42,000 scf/hour). After truncation, the mean and standard deviation are 2,000 scf/hour, and 4,100 scf/hour, respectively.
[vi] The distribution is truncated at the minimal sample, and 10 times the maximal sample (1,900-3,800,000 scf/hour). After truncation, the mean and standard deviation are 130,000 scf/hour, and 270,000 scf/hour, respectively.

We compare the assumptions and the resulting emission factors for the well completion stage in this study and the existing literature in **Table A.5**. My estimates for both type-I and type-II well completion emission factors are smaller and have a narrower uncertainty range than those reported in existing studies. Compared to the existing literature, the type-I well completion events[i] have very small vented/flared methane emissions, suggesting increased use in REC technologies[ii]; the type-II well completion events have similar levels of natural gas emissions to existing studies but the percentage of flared natural gas is much higher. Even though my results are generally smaller than other studies, my estimated well completion emission factor is about 70% larger than the green completion scenario in Weber et al. (2012),[115] which accounted for the effects of U.S. EPA's New Source Performance Standards (NSPS) and National Emission Standards for Hazardous Air Pollutants (NESHAP).[121] As U.S. EPA's NSPS and NESHAP rules took effect in August 2012 and require RECs (or "green completion") and other emission reduction practices by January 1, 2015, the industry is expected to adapt and comply with the rule. In addition, some states, such as Colorado,[122] require the natural gas industry to detect and repair leaks from tanks, pipelines, and other drilling and production processes. By contrast, all the existing studies are based on data that was measured prior to this rule making, thus being unable to capture the recent changes.

U.S. EPA revised its estimates of GHG emissions from well completions for year 2011 in the most recent release of the national GHG Inventory[152] (**Table A.6**). By relying on the industry-submitted data,[98] the new release of the national inventory reported a nearly three-quarters decrease of methane emissions from well completion events compared to the previous inventory.[277] As several existing studies[140,150,279,280] relied on the previous U.S. EPA inventory[281] to estimate total vented and flared emissions, their estimated emission factors would decrease significantly when using the updated U.S. EPA estimates.

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[i] It seems surprising at first that non-flaring Type-I well completion events have lower methane emissions than Type-II well completion events. This is because Type-I well completion events have much lower (two orders of magnitude lower) flowback rates than Type-II well completion events. See footnote 12 and 13 for more details.
[ii] Allen et al. (2013)[97] calculated the ratio of measured methane emissions over potential methane emissions and the average ratio is 0.014. In other words, the majority of potential methane emissions from well completion is either combusted or flared.

To summarize, while the amount of GHG emissions from well completion is still an open question, there has been a trend of emissions reductions due to more stringent regulations at both federal and state level as well as increasing information of profitable opportunities to reduce emissions, and these trends are likely to continue in the future.

Table A.5. Comparison of assumptions and GHG emissions due to well completions. Assumptions from existing literature are taken from Weber et al. (2012).[115] The well completion emission factor is normalized by the estimated ultimate recovery (EUR) of a natural gas well.

Source		**Total vented/flared**	**Flaring rate**	**Estimated ultimate recovery (EUR)**	**Well completion emission factors**
Unit		metric ton CH_4/well	percentage	BCF	gCO_2-eq/MJ_{LHV}
Jiang et al. (2011)[139]		400 (26-1,000)	76% (51%-100%)	2.85 (0.5-91)	1.2 (0.1-9.2)
Skone et al. (2011)[279]		177	15% (12%-18%)	3 (2.1-3.9)	1.3 (1.0-1.9)
Hultman et al. (2011)[280]		139	15%	0.54	5.2
Stephenson et al. (2011)[150]		177 (52-385)	51% (0%-100%)	2 (1-3)	1.6
Burnham et al. (2011)[140]		177 (13.5-385)	41% (37%-70%)	3.5 (1.6-5.3)	0.75
Howarth et al. (2011)[141]		74-3,610	0%	1.2-7.4	8.6
Weber et al. (2012)[115] Best estimate		Triangular (13.5, 177, 38.5)	Triangular (15%, 41%, 100%)	Triangular (0.5, 2, 3.5)	1.2 (0.2-3.4)
Weber et al. (2012) [115] Green Completion		Not given	Not given		0.20 (0.04, 0.6)
This study	Type I	4.4 (0.04-32)*	0%	2 (0.8-3.2)*	0.09 (0.0006, 0.7)* $^+$
	Type II	135 (1-935)*	80-98%		0.57 (0.004, 4.0)* $^+$
	Weighted average	N/A	N/A		0.33 (0.001, 2.4)* $^+$

** These ranges are calculated based on the 95% confidence interval calculated from the Monte-Carlo simulation model used in this study. $^+$ 100-year GWPs are used.*

Table A.6. Comparison of well completion GHG emissions in 2011 by U.S. EPA GHG inventory[152,277] and in Allen et al. (2013).[97] Unit: Gg of methane per year.

Emissions sources	**U.S. EPA GHG Inventory 2013[277]**	**U.S. EPA GHG Inventory 2014[152]**	**Allen et al. (2013)[97]**

Well completion and workover (wells with hydraulic fracturing)	796.9*	218.1*	-
Well completion (wells with hydraulic fracturing)	654**	179***	18 (5-27)+

*The net emissions reported in U.S. EPA GHG Inventory are from well completion and well workover. ** Allen et al. (2013)[97] estimate that net emissions from well completion based on the data from U.S. EPA GHG Inventory 2013[277] are 654Gg. *** I calculate the net emissions from well completion based on data from U.S. EPA GHG Inventory 2014[152] by applying the same ratio (654Gg/796.9Gg) from the U.S. EPA GHG Inventory 2013[277]. + The range is based on the 95% confidence interval.*

Estimated Ultimate Recovery (EUR). GHG emissions from preproduction are normalized by the EUR of a natural gas well to get to gCO_2-eq/MJ_{LHV}. Existing literature[115,139] reports that EUR is a major factor for GHG emissions from well completions. As most shale gas wells have been developed in the past five years, EUR is unknown. Most studies (as listed in **Table A.5**) have relied on parametric analysis to explore the effect of EUR on preproduction GHG emission factors. I follow the assumptions put forth by Weber et al. (2012)[115] as the paper provides a review of EUR assumptions from existing bottom-up studies. More specifically, I assume the EUR of a natural gas well follows a triangular distribution with minimum, mode, and maximum of 0.5, 2, 3.5 billion cubic foot (bcf) per year.

For natural gas production, processing, transmission & storage, and distribution, I follow the estimation framework of Venkatesh et al. (2011)[27] with updated natural gas flow data from U.S. Energy Information Administration (EIA)[10,153,282] and emissions data from U.S. EPA GHG Inventory.[152,277] Besides substantial data updates, this study includes notable changes in the estimation and uncertainty analysis methods compared with Venkatesh et al. (2011)[27], such as performing emission allocations between dry natural gas and natural gas liquids (NGLs). In addition, I estimate GHG emissions for U.S. domestic shale gas only to reflect changes in natural gas supply in the United States.

Production Stage. In the production stage, natural gas is routinely produced from the wellbore, and transported through gathering lines to the processing stages, or to the transmission pipelines directly. I divide emissions from natural gas production into the following five sources: lease fuel use, well workover, liquid unloading, flaring emissions, other fugitive CO_2 emission (except for those from liquid unloading, well completion, and well workovers), and other fugitive CH_4 emissions (except for those from liquid unloading, well completion, and well workovers). For emission categories in the production stage, I normalized all emission factors using natural gas volumes produced at well sites[i] (that is, prior to use and prior to any leaks of natural gas that occur in the production stage), which was 25.1 trillion cubic foot (tcf) for year 2011.

Lease fuel use. U.S. EIA[10] reports lease fuel use for every natural gas producing state. I calculate the volumetric percentage of natural gas used as lease fuel for each state and fit distributions. Emission factors of lease fuel use are then calculated by multiplying combustion emission factors of natural gas. I assume the combustion is complete, so lease fuel use is an emission source of CO_2.

Flaring emissions. Flaring is used to convert stranded methane into carbon dioxide for safety considerations, and for reducing methane emissions.[283] U.S. EIA[10] reports flaring and vented estimates for every natural gas producing state. I first calculate the volumetric percentage of natural gas flared for each state and then fit distributions. Emissions factors for flaring are then calculated by assuming a 98% combustion efficiency (same as flaring well completion). The U.S. EPA GHG Inventory[152,277] also reports flaring emissions, and they are in the same range as those in this study[ii] after normalization by natural gas production volume at well sites.

Well-workover. Well workover refers to "the second (or more) hydraulic fracturing of a well to stimulate production" over the well lifetime.[115] Due to lack of more accurate information, it is often assumed to have the same emission factor as well completion.[115,140,280] The number of well walkovers (or additional hydraulic fracturing events) over the well lifetime is highly uncertain because most shale gas wells are still in their early years. Thus, existing studies rely on parametric analysis to explore the impacts of well workover. I followed Weber et al. (2012)[115] in assuming a discrete probability distribution – equal probability of 0, 1, and 2 workovers over well lifetime. As well workovers are assumed to be similar to well completions, they share the same set of uncertainty factors (except the number of well workovers).

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[i] This amount of natural gas coming out of wells can be calculated by subtracting 'repressuring" from 'gross withdraw', both of which are reported for each state by EIA[10].
[ii] EPA uses a 100-year global warming potential (GWP) of 21 for methane[382]. When using the same GWP, the EIA data and EPA data produce similar emission factors (less than 5 percent difference).

Liquid unloading. Liquid unloading refers to the "removal of accumulated fluids from well bore either by venting or using artificial lift techniques (e.g. plunger lifts)".[284] There are two types of liquid unloading, those with plunger lift and those without plunger lift. Plunger lift is used to lower the GHG emissions during liquid unloading.[285] Allen et al. (2013)[97] report 9 events of liquid unloading without plunger lift and they did not reported any liquid unloading events with plunger lift. Even though the sample size is very small, their yearly methane emissions from liquid unloading per well span a very large range, from 1,900 cubic feet to nearly 1.4 million cubic feet. The estimation procedure for liquid unloading without plunger lift is straightforward: (1) fit a distribution to simulate the yearly methane emissions from liquid unloading (per well); (2) calculate total methane leakage by multiplying the fitted distribution with the number of wells without plunger lift. For the case of liquid unloading with plunger lift, I instead rely on the data from API and ANGA survey.[99] This data set (which includes 24 samples) reports the aggregated methane emissions on a sub-basin level (lumping over several wells together). However, without better information, I treat them as if they were data samples for individual wells. I follow the same estimation procedure listed above. Numbers of wells equipped with plunger lift and without plunger lift are from U.S. EPA.[277] Finally, I combine methane emissions from two types of liquid unloading before normalizing it by total natural gas production for the U.S. I compare the annual methane emissions per well across my estimates and U.S. EPA GHG Inventory 2014:[152] for wells with plunger lifts, my mean estimate is 0.27 MMscf, which is the same as U.S. EPA's estimate; for wells without plunger lifts, my mean estimate is 0.42 MMscf while U.S. EPA's point estimate is 0.14 MMscf. My estimation results are in the same range as U.S. EPA's but the measurement data in Allen et al. (2013)[97] suggests that U.S. EPA may be underestimating methane emissions from wells without plunger lifts.

I have discussed the major combustion and flaring sources, as well as major methane leakage sources in the production stage. For all the other smaller but negligible sources[i] of carbon dioxide and methane emissions from the production of natural gas, I rely on the U.S. EPA GHG emission inventory.[152] Specifically, these "other fugitive CO_2 emissions" and "other fugitive CH_4 emissions" (**Table A.7**) are calculated by adding up the emissions estimates in the U.S. EPA's

[i] Such as field separation equipment, compressors, pneumatic devices, pumps, condensate tanks, blowdowns and upsets.

"fugitive CO_2 emissions" and "fugitive CH_4 emissions" categories (excluding well completions, well workovers, and liquid unloading, since I use other sources of data), respectively. I calculate "other fugitive CO_2 emissions" (2,90Gg/year) and "other fugitive CH_4 emissions" (1,536Gg/year) from U.S. EPA's GHG emissions inventories released in 2013 and 2014[152,277] and find little difference in these two inventories.

In addition to the point estimates, U.S. EPA also reports the 95% confidence intervals for the total CO_2 emissions and total methane emissions from natural gas production – both with the lower bound as 19% lower than the point estimate and the upper bound as 30% higher than the point estimate. I thus assume "other fugitive CO_2 emissions" and "other fugitive CH_4 emissions" follow triangular distributions, with the minimum, mode and maximum as 81%, 89%, and 130% of the point estimates calculated above (**Table A.7**). The parameters are chosen to make sure the mean of the triangular distribution match the point estimate reported by U.S. EPA.[152,277] Fugitive emissions are normalized by the volume of natural gas produced at the well site. **Table A.7** summarizes the key assumptions for my model of emissions from production stage.

Table A.7. Assumptions of GHG emissions from natural gas production stage.

Variable		Unit	Distribution
Lease fuel use		volumetric share of natural gas produced at well site	Truncated lognormal (-3.7, 0.7)[i]
Well workover	Emission factor	gCO_2-eq/MJ_{LHV}	0.33 (0.001, 2.4)
	Activity	Number of well workover per well lifetime	Discrete distribution: {0, 1, 2} with equal probability
Liquid unloading	Without plunger lift	methane emitted per year per well (cubic foot)	Truncated lognormal (-3.0, 2.3)[ii]
	With plunger lift		Truncated lognormal (-2.7, 1.7)[iii]
	Without plunger lift	number of wells	35,828
	With plunger lift		22,866
Flaring and vented		volumetric share of natural gas produced at well site	Truncated lognormal (-5.4, 2.3)[iv]
Other fugitive CO_2 emissions		Gg/year	Triangular distribution: 290× (81%, 89%, 130%)

[i] The distribution is truncated at the minimal sample, and 2 times the maximal sample (0.005-0.6). After truncation, the mean and standard deviation are 0.033, and 0.026, respectively.
[ii] The distribution is truncated at the minimal sample, and 10 times the maximal sample (0.002-13 MMscf). After truncation, the mean and standard deviation are 0.41 MMscf, and 1.17 MMscf, respectively.
[iii] The distribution is truncated at the minimal sample, and 10 times the maximal sample (0.001-59 MMscf). After truncation, the mean and standard deviation are 0.27 MMscf, and 0.93 MMscf, respectively.
[iv] The distribution is truncated at the minimal sample, and 2 times the maximal sample (0.0005-0.6). After truncation, the mean and standard deviation are 0.009, and 0.014, respectively.

Other fugitive CH_4 emissions (excluding well workover and liquid unloading).	Gg/year	Triangular distribution: 1536 × (81%, 89%, 130%)

Processing Stage. In the processing stage, natural gas is processed to remove impurities, reduce the compositions of CO_2, and be separated from natural gas liquids (NGLs) to produce pipeline-quality dry natural gas, which is then transported to consumers. We rely on U.S. EPA's Greenhouse Gas Reporting Program[i] (GHGRP)[98] (reporting data from Sep. 1, 2011 to Aug. 31 2012) to estimate GHG emissions from natural gas processing plants and referred to the latest U.S. EIA 757 survey[286] (for year 2012)[ii] for the corresponding production flows of natural gas processing plant. I match these two sources of data using the NACIS (North American Industry Classification System) code and get 217 matched data pairs (GHG emissions and production rate) at the level of individual natural gas processing plants. The matched set of natural gas processing plants have a total capacity of 48.6 billion cubic feet per day, which is nearly three quarters of natural gas processing plant capacity in the Lower 48 states in the U.S.

With the matched dataset, I first calculate the emission factors (separately for different GHGs, CO_2, CH_4 and N_2O) by normalizing total GHG emissions by production rates of each plant; then fitted distributions on each GHG. **Table A.8** shows the fitted distribution parameter for each GHG emitted from processing plants.

Table A.8. Assumptions of GHG emissions from natural gas processing plants.

Variable	Unit	Distribution
CO_2 emission factors	gram of each GHG per cubic foot of natural gas processed	Truncated lognormal (0.38, 0.89)[iii]
CH_4 emission factors		Truncated lognormal (-5.8, 1.4)[iv]
N_2O emission factors		Truncated lognormal (-13.0, 1.3)[v]

[i] EPA mandates "facilities that emit 25,000 metric tons or more per year of GHGS" to report their GHG data annually to the GHGRP since 2010[98].

[ii] Following Huricane Karina in 2005, EIA set up a triennial survey of natural gas processing plants (EIA-757) to collect information on the capacity, status, and operations of natural gas processing plants[383]. The latest EIA-757 survey covered 517 active natural gas processing plants with a total capacity of 65.5 billion cubic feet per day in the Contiguous U.S.[383].

[iii] The distribution is truncated at the minimal sample, and twice the maximal sample (0.2-395). After truncation, the mean and standard deviation are 2.2, and 2.4, respectively.

[iv] The distribution is truncated at the minimal sample, and twice the maximal sample (0.0001-1). After truncation, the mean and standard deviation are 0.008, and 0.02, respectively.

[v] The distribution is truncated at the minimal sample, and twice the maximal sample (0-0.1). After truncation, the mean and standard deviation are 6e-6, and 1e-5, respectively.

Allocation. During the natural gas processing stage, natural gas liquids (NGLs) are separated from dry natural gas, and upstream GHG emissions have to be allocated across dry natural gas and NGLs. I perform an energy-based allocation, so the upstream GHG emissions are the same per unit of energy (such as MJ) of dry natural gas and NGLs. With the natural gas flow information for year 2011 from U.S. EIA,[287] and energy intensity information from U.S. EIA,[288] I also calculate the share of preproduction production, and processing emissions across dry natural gas and NGLs (**Table A.9)**. Through this allocation, the majority of upstream GHG emissions are due to dry natural gas production.

Table A.9. Assumptions of GHG emissions from natural gas processing plants.

Energy carrier	**Volume**		**Energy Content**		**Allocation**
	Value	**Unit**	**Value**	**Unit**	
Dry natural gas	16,566,883	Million cubic feet	983	BTU per cubic foot	84.5%
Ethane	337,972	Thousand barrel	3.08	Million BTU per barrel	5.4%
Propane	230,227		3.84		4.6%
Normal butane	57,399		4.33		1.3%
Isobutane	76,983		4.33		1.7%
Pentanes Plus	106,284		4.62		2.5%

Transmission and Storage Stage. In the transmission and storage stages, processed natural gas is transmitted through interstate pipelines to the consumers. The division between transmission stage and distribution stage is the city gate in the pipeline system. While consumers and residential users are typically serviced by utilities that operate distribution pipelines, while interstate pipeline operators serve large industrial users, such as power generators and chemical plants. Following Venkatesh et al. (2011a),[27] I divide GHG emissions from transmission and storage stage into the following three categories: fuel use, fugitive CO_2 emissions, and fugitive CH_4 emissions. For estimates of fuel use from natural gas pipelines, I rely on U.S. EIA data for "Pipeline & Distribution Use" and the volumes of natural gas coming into transmission Stage (see **Table A.12**) to calculate volumetric shares of delivered natural gas used as fuel for each state and fitted distributions on these samples. We assume the electricity used to power natural gas along the pipelines is grid average electricity. Fugitive CO_2 and CH_4 emissions are estimated based on the U.S. EPA GHG Inventory[152] and are modeled with triangular distributions[i]. These

[i] For more details on distributions, see relevant discussions on fugitive emissions in the Production Stage.

fugitive emissions are then normalized by the volume of natural gas coming into transmission stage and converted to gCO_2-eq/MJ_{LHV} through unit conversion. **Table A.10** summarized the resulting distribution parameters.

Table A.10. Assumptions for GHG emissions from natural gas the transmission and storage stages.

Variable	Unit	Distribution
Transmission and distribution fuel use (natural gas)	volumetric share of natural gas produced	Truncated lognormal: (-3.81, 0.79)[i]
Transmission and distribution fuel use (electricity)	Million kWh/year	3098.6[3]
Fugitive CO_2 emissions	Gg/year	Triangular distribution: 65 × (81%, 89%, 130%)
Fugitive CH_4 emissions	Gg/year	Triangular distribution: 2153 × (81%, 89%, 130%)

Distribution Stage. Combustion emissions from fuel used in the distribution stage have been accounted for in the transmission and storage stage. Thus, in the distribution state, I consider only fugitive CO_2 and CH_4 emissions. Fugitive CO_2 and CH_4 emissions are estimated based on the U.S. EPA GHG Inventory[152] and are modeled with triangular distributions[ii]. These fugitive emissions are then normalized by the volume of natural gas coming into Distribution Stage and converted to gCO_2-eq/MJ_{LHV} through unit conversion. **Table A.11** summarizes the resulting distribution parameters.

Table A.11. Assumptions for GHG emissions from the natural gas distribution stages.

Variable	Unit	Distribution
Fugitive CH_4 emissions	gram per cubic foot of natural gas delivered	Triangular distribution: 40 × (81%, 89%, 130%)
Fugitive CO_2 emissions	gram per cubic foot of natural gas delivered	Triangular distribution: 1311 × (81%, 89%, 130%)

Alignment of emissions factors in natural gas upstream stages. The emission factors for carbon dioxide and methane calculated above are normalized to one mega joule of natural gas coming into each stage (for example MJ of gas coming into the processing stage or MJ of gas

[i] The distribution is truncated at the minimal sample, and twice the maximal sample (0.001-0.4). After truncation, the mean and standard deviation are 0.03, and 0.03, respectively.
[ii] For more details on distributions, see relevant discussions on fugitive emissions in the Production Stage.

coming into the transmission stage). However, the functional unit of interest is one mega joule of natural gas delivered to the end use (for instance, out of the distribution pipelines). I calculate the *loss factor*[i], defined as the ratio between the total natural gas flow coming out of a natural gas upstream stage and the total natural gas flow coming into a natural gas upstream stage. A loss factor of one means there is no loss of natural gas (either as fuel or as methane leak) for the given natural gas upstream stage. One the other extreme, a loss factor of zero means that all natural gas is lost. In this case, no matter how large emission factors of previous upstream stages are, the emission factor with regard to one unit of natural gas delivered to the end use should be infinite because no natural gas can be delivered to the end use. From the perspective of natural gas end uses, a less-than-one loss factor is a summary of the fact that one unit of natural gas at the end of the supply chain requires more than one unit of natural gas produced at the well site because some fraction of natural gas produced is either vented into the atmosphere or combusted.

I use U.S. EIA's data on natural gas flows[10,153,282] to estimate loss of natural gas used as fuel in upstream natural gas stages[ii] (consistent with the bottom-up model) and I use the mean of methane leakage rate calculated from the bottom-up model to estimate loss of natural gas leaked from the natural gas system. (**Table A.12**). I then calculate loss factor using Eqn. A.3.

$$\begin{aligned} Loss\ factor_{Stage\ X} &= 1 - \frac{Total\ loss\ of\ natural\ gas_{Stage\ X}}{Natural\ gas\ flow_{prior\ to\ Stage\ \mathrm{X}}} \\ &= 1 - \frac{Volume\ of\ natural\ gas\ used\ as\ fuel_{\ Stage\ X}}{Natural\ gas\ flow_{prior\ to\ Stage\ \mathrm{X}}} - \frac{Volume\ of\ natural\ gas\ leaked_{Stage\ X}}{Natural\ gas\ flow_{prior\ to\ Stage\ \mathrm{X}}} \\ &= 1 - Volumetric\ ratio_{fuel,\ Stage\ X} - Volumetric\ tratio_{leak,\ Stage\ X} \end{aligned} \tag{A.3}$$

Table A.13 shows the resulting loss factors of natural gas upstream stages. Two existing studies[279,289] also calculate loss factor[iii], while the other studies implicitly assume that the loss factor is 1 for all upstream stages. My estimate for the loss factor is 0.91, while Skone et al. (2011)[279] report 0.87, and Logan et al. (2012)[289] report 0.93.

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[i] We assume that the heat content of natural gas in any natural gas upstream stage remains the same, and the loss factor represent the "energy efficiency" for the upstream stage ($MJ_{\text{out of Stage X}}/ MJ_{\text{prior to Stage X}}$).

[ii] As a simplification, we ignored the differences in energy intensity (energy content per volume) of natural gas from different upstream stages. Thus, the energy efficiency is simplified as the volumetric ratio between natural gas that come out of and natural gas that come into a designated upstream stage.

[iii] To be more specific, they estimate an equivalent metric, 'loss of produced natural gas'.

Table A.12. Natural gas flow used in this study (year 2011) (Unit: trillion cubic feet) (Terminology used by U.S. EIA is underlined).

<table>
<tr><th>Upstream stage</th><th>Flow</th><th>Formula</th><th>Value</th></tr>
<tr><td rowspan="2">Production</td><td>Coming into the stage</td><td><u>Gross Withdraws</u> – <u>Repressuring</u></td><td>25.1</td></tr>
<tr><td>Loss (as fuel use or leaks)</td><td><u>Lease Fuel</u></td><td>0.9</td></tr>
<tr><td rowspan="2">Processing</td><td>Coming into the stage</td><td><u>Gross Withdraws</u> - <u>Repressuring</u> - <u>Lease Fuel</u> - <u>Vented and flared</u></td><td>24.0</td></tr>
<tr><td>Loss (as fuel use or leaks)</td><td><u>Plant fuel</u></td><td>0.4</td></tr>
<tr><td rowspan="3">Transmission, storage and distribution</td><td>Coming into the stage</td><td>Sum of the two rows below (this number differs from those coming out of the Processing stage because of changes in natural gas storage and exports/imports)</td><td>23.2</td></tr>
<tr><td>Loss (as fuel use or leaks)</td><td><u>Pipeline & Distribution Use</u></td><td>0.7</td></tr>
<tr><td>Coming out of the stage</td><td><u>Volumes Delivered to Consumers</u></td><td>22.5</td></tr>
</table>

Table A.13. Loss factor in natural gas upstream stages.

Upstream stage	This study	Skone et al. (2011)[279]	Logan et al. (2012)[289]
Preproduction	1.00	-	-
Production	0.96	0.98	0.98
Processing	0.98	0.90	0.96
Transmission and storage	0.97	0.99	0.99
Distribution	1.00	1.00	1.00
Natural gas system (total)	0.91	0.87	0.93

Upstream total. Total GHG emissions from upstream stages of the natural gas life cycle are the sum of the emissions from each upstream stage divided by the energy ratios between one mega joule of natural gas delivered to the end use and one mega Joule of natural gas coming into each upstream stage. The total GHG emissions are calculated using Eqn. A.4.

$$\begin{aligned} &\textit{Life cycle GHG emissions} \\ &= \sum_{X \in \textit{Natural gas upstream stages}} \frac{\textit{GHG emissions}_{\textit{Stage } X}}{\prod_{Y \in \{\textit{All upstream stages not prior to Stage } X\}} \textit{Loss Factor}_{\textit{Stage } Y}} \end{aligned} \tag{A.4}$$

Here, *Stage X* represents each of the natural gas upstream stages discussed, ranging from the preproduction stage to the distribution stage. As discussed in the main text, and *Section A.1*

Global Warming Potential (GWP) in this Appendix, I use the GWPs from IPCC[2] to convert GHGs to the same unit (gCO_2-eq/MJ_{LHV}).

Distribution fitting. I fit distributions for key variables in the Monte-Carlo simulation. Compared to existing studies (for instance, Jiang et al. (2011)[139] and Venkatesh et al. (2011)[27]), I improve the distribution fitting by considering: (1) distribution truncation, and (2) weighted distribution. I apply a distribution truncation to minimize the effects of extreme simulated samples on the simulation result[i]. More specifically, I truncate the distributions generated in the range of minimum data sample and 10 times of the maximum data sample. Much of the raw data I use are at the state-level, or shale gas well-level data, which have different production levels. I apply a weighted distribution fit with the natural gas productions being the weights. In choosing distributions, I compare distribution candidates from uniform distribution, triangular distribution, lognormal distribution, beta distribution, and bootstrapping. The criteria include statistics such as weighted mean, weighted standard deviation and confidence interval, and metrics such as the Akaike Information Criterion (AIC) and the Bayesian Information Criterion (BIC).

A.2.2 Estimation Results

Here I provide detailed results with breakdowns in GHGs and in processes in each upstream stage (**Table A.14**, **Table A.15**, **Figure A.1** and **Figure A.2**). In accordance with the main text, I consider four scenarios: baseline methane estimate with 100-year GWP; baseline methane estimate with 20-year GWP; pessimistic methane estimate with 100-year GWP; and pessimistic methane estimate with 20-year GWP. My estimates show that there is a very large uncertainty range of natural gas upstream GHG emissions. The mean total GHG emission is 17.4 gCO_2-eq/MJ_{LHV} and the 95% confidence interval is 10.3-29.5 gCO_2-eq/MJ_{LHV}. I find that the distribution of total upstream GHG emissions is highly asymmetrical, as shown in **Table A.14.**

[iii] On the other hand, we tried our best to not influence Monte-Carlo simulations through bounding the simulated variables. Thus, we cut off generated distributions at 10 times of the maximum data sample to allow for an order-of-magnitude freedom. For the normalized volumetric percentage of natural gas used as fuels (lease fuel, plant fuel, and pipeline use), we cut off generated distributions at 2 times of the maximal data sample because these information is more certain and a 10-times bound is too large (more than 100%).

The fitted distribution[i] is chosen based on maximizing the negative of the log likelihood, Bayesian information criterion (BIC), and Akaike information criterion (AIC).

For the breakdown of upstream stages, well-site production (including both preproduction and production) and pipeline transportation (including transmission and distribution) contribute most to GHG emissions. By comparison, natural gas processing is responsible for a much lower share of GHG emissions and only 79% of produced natural gas have to be processed at processing plant,[10,282] While methane emissions from the preproduction and production stages have attracted most attention from regulators and industry,[121–123] methane emissions from the pipeline system should receive more attention as well. First, methane emissions from the pipeline system are going to be more important when cost effective technologies[120] to reduce methane emissions from well sites are put into place. Second, methane emissions in urban areas pose serious security risks to surrounding properties.

For the relative contribution of carbon dioxide and methane, methane is found to be more important than carbon dioxide (**Figure A.1** and **Figure A.2**). For the baseline scenario, methane contributes slightly more than carbon dioxide in terms of global warming potential. For the 20-year GWP scenario, the contribution of methane is nearly two times more than carbon dioxide. While carbon dioxide is quite evenly distributed across natural gas upstream stages (mostly as result of fuel combustion), methane emissions are more concentrated. Most methane emissions occur either at the well site (liquid unloading, well completion and well workover), or in the pipeline system (fugitive emissions). In addition, there is evidence that a small share of super-emitters are responsible for a larger share of GHG emissions (as summarized in Brandt et al. (2014);[114] and reflected in the right-skewed distribution shown in this study).

Existing studies of the upstream GHG emissions for natural gas have spanned a wide range with a 95% uncertainty range of 11.0-21.0 gCO_2-eq/MJ_{LHV} (compiled using estimates from six individual bottom-up studies).[115] These ranges result from the limitations in available data and

[i] Candidate distributions include Beta, Birnbaum-Saunders, Exponential, Extreme Value, Gamma, Generalized Extreme Value, Generalized Pareto, Inverse Gaussian, Logistic, Log Logistic, Lognormal, Nakagami, Normal, Rayleigh, Rician, t location-scale, and, Weibull.

different assumptions about emission factors and activities in the natural gas system. While my baseline estimates (mean values) align well with existing studies, there are two major differences in terms of emission structures. First, I found smaller emissions from natural gas preproduction, production, and processing stages,[i] and larger emissions from natural gas pipeline systems compared to most existing studies (Howarth et al. (2011)[141] is an exception). Second, I find a lower methane leakage rate from natural gas systems and higher carbon dioxide emissions. I think these two observations are partially the results of reduced methane emissions in well completions and well workovers due to better industry practices and stringent regulations on well completions. I do acknowledge two general limitations of bottom-up LCA studies. First, as discussed in Brandt et al. (2014),[114] bottom-up studies are likely to be conservative in nature. Second, any LCA study, including this study, is constrained by the accuracy and representativeness of the data sources used (see *Section A.9 Data Quality* for more discussion).

[i] See Table SI-5 in the Supporting Information of Weber et al. (2012)[115] for a summary of GHG emissions from preproduction, production & processing, and transmission stages.

Table A.14. Natural gas upstream emissions with breakdown of upstream stages and GHGs. Both 100-year and 20-year GWP estimation results are shown as well as the pessimistic case (methane emissions are multiplied by 1.5). Mean estimate and the 95% confidence interval (in parenthesis) are shown in table entries.

Stage	GHG emissions breakdown		Total GHG emissions			
	CO_2 (baseline)	**CH_4 (baseline)**	**100-year GWP (baseline)**	**100-year GWP (pessimistic)**	**20-year GWP (baseline)**	**20-year GWP (pessimistic)**
Unit	gram/MJ_{LHV}		gCO_2-eq/MJ_{LHV}			
Pre-production	1.3 (0.5-3.1)	0.006 (0-0.05)	1.5 (0.6-4.2)	1.7 (0.6-4.9)	1.9 (0.6-6.3)	2.2 (0.6-8.2)
Production	2.6 (0.7-7.3)	0.09 (0.06-0.23)	6.0 (2.8-13.3)	7.7 (3.7-17.1)	10.7 (5.7-24.1)	14.8 (7.9-34.2)
Processing	2.2 (0.3-8.6)	0.008 (0-0.05)	2.5 (0.4-9.2)	2.7 (0.4-9.6)	2.9 (0.5-10.5)	3.3 (0.5-12)
Transmission	1.8 (0.4-6.1)	0.09 (0.08-0.11)	5.2 (2.7-9.6)	6.9 (3.7-11.7)	9.9 (6.2-15.2)	14 (8.8-20.7)
Distribution	0.002 (0.001-0.002)	0.06 (0.05-0.07)	2.0 (1.1-3.1)	3.0 (1.6-4.7)	4.9 (3.0-7.1)	7.3 (4.5-10.7)
Upstream total	8.0 (3.6-16.9)	0.26 (0.20-0.43)	17.2 (10.2-29.3)	22.0 (12.9-36.7)	30.3 (19.3-49.7)	41.7 (26.3-68.8)
Fitted distribution for upstream total	Generalized extreme value (‘shape’=0.13, ‘scale’=2.28, ‘location’ = 6.33)	Generalized extreme value (‘shape’=0.30, ‘scale’=0.024, ‘location’ = 0.23)	Log logistic (‘log location’ = 2.80, ‘log scale’ = ‘0.15’)	Log logistic (‘log location’ = 3.37, ‘log scale’ = ‘0.13’)	Log logistic (‘log location’ = 3.05, ‘log scale’ = ‘0.14’)	Log logistic (‘log location’ = 3.69, ‘log scale’ = ‘0.13’)

Table A.15. Natural gas upstream emissions with the breakdown of processes in each upstream stage (Unit: gCO_2-eq/MJ_{LHV}). Both 100-year and 20-year GWP estimation results are shown as well as the pessimistic case (methane emissions are multiplied by 1.5). Mean estimate and the 95% confidence interval (in parenthesis) are shown in table entries.

Stage	Process	100-year GWP (baseline)	100-year GWP (pessimistic)	20-year GWP (baseline)	20-year GWP (pessimistic)
Pre-production	Wellpad Construction	0.2 (0.1-0.6)			
	Well Drilling	0.4 (0.1-1.0)			
	Hydraulic Fracturing	0.6 (0.2-1.5)			
	Well Completion	0.3 (0.0-2.4)	0.4 (0.0-3.2)	0.7 (0.0-4.8)	0.9 (0-6.8)
Production	Lease Fuel Use	2.0 (0.4-6.2)			
	Flaring	0.6 (0.0-2.8)	0.7 (0.1-3.1)	0.8 (0.1-3.7)	0.9 (0.1-4.4)
	Liquid Unloading	0.7 (0.0-4.7)	1.0 (0.0-7.0)	1.6 (0.0-11.2)	2.4 (0-16.9)
	Well Workover	0.3 (0.0-2.5)	0.4 (0.0-3.4)	0.7 (0.0-5.0)	0.9 (0-7.1)
	Other Fugitive Emissions	2.4 (1.2-3.6)	3.5 (1.8-5.5)	5.7 (3.5-8.2)	8.6 (5.3-12.4)
Processing	CO_2	2.2 (0.3-8.6)			
	CH_4	0.3 (0.0-1.8)	0.4 (0.0-2.7)	0.7 (0.0-4.3)	1.1 (0-6.4)
	N_2O	< 0.01			
Transmission	Fuel Use – Natural gas	1.7 (0.3-6.0)			
	Fuel Use – Electricity	0.1 (0.1-0.1)			
	Fugitive Emissions	3.4 (1.7-5.2)	5.1 (2.6-7.8)	8.1 (5.0-11.8)	12.2 (7.5-17.7)
Distribution	Fugitive Emissions	2.0 (1.1-3.1)	3.0 (1.6-4.7)	4.9 (3.0-7.1)	7.3 (4.5-10.7)
Upstream total emissions		17.4 (10.3-29.5)	22.0 (12.9-36.7)	30.3 (19.3-49.7)	41.7 (26.3-68.8)
Implicit methane leakage rate		1.3% (1.0%-2.2%)		2.0% (1.6%-3.3%)	

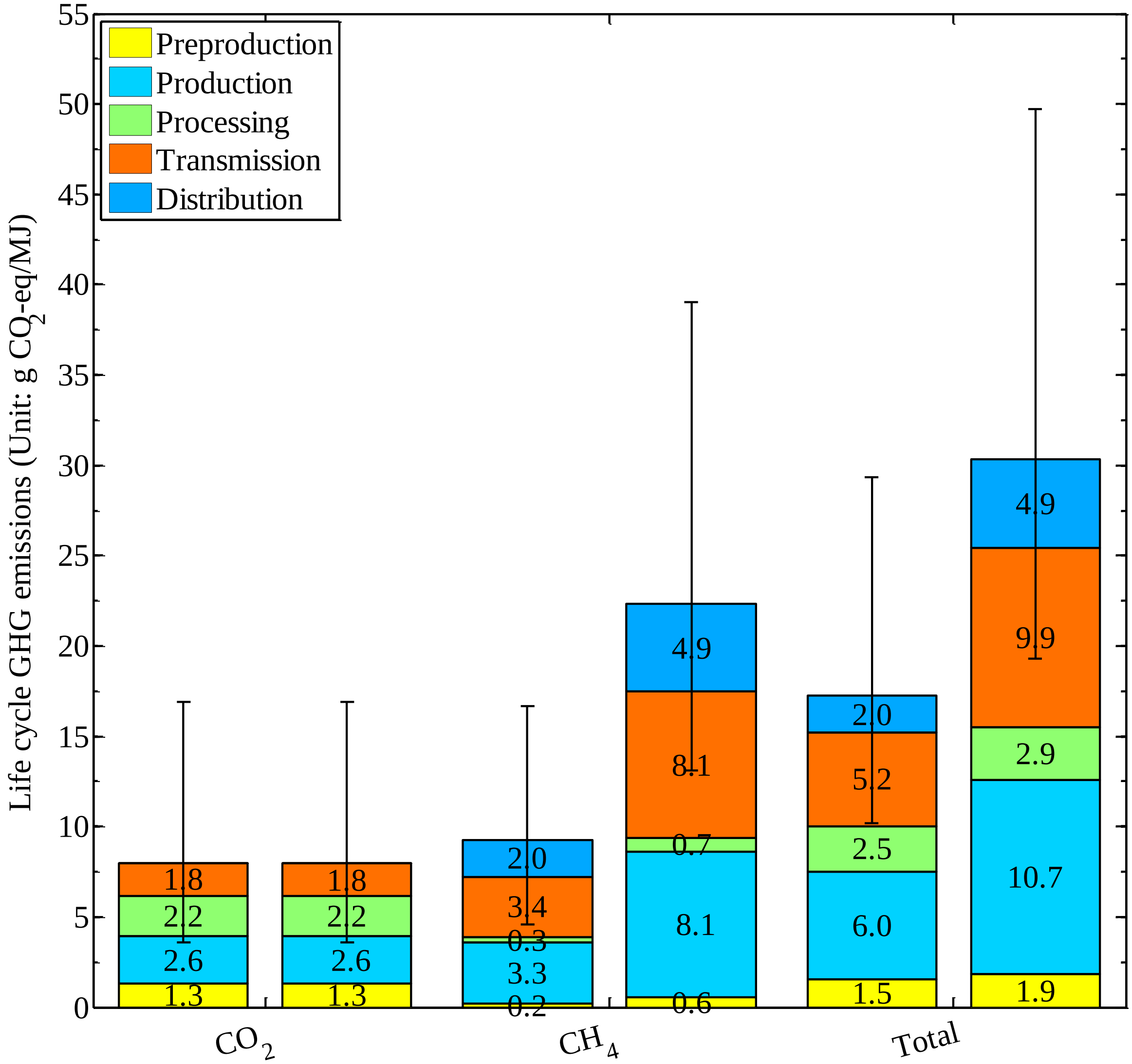


Figure A.1. Breakdown of natural gas upstream GHG emissions by greenhouse gas and by upstream stages. Error bar is calculated based on the 95% confidence interval of the total emissions for each GHG. Estimates with 100-year GWP (left bars) and 20-year GWP (right bars) are shown side by side.

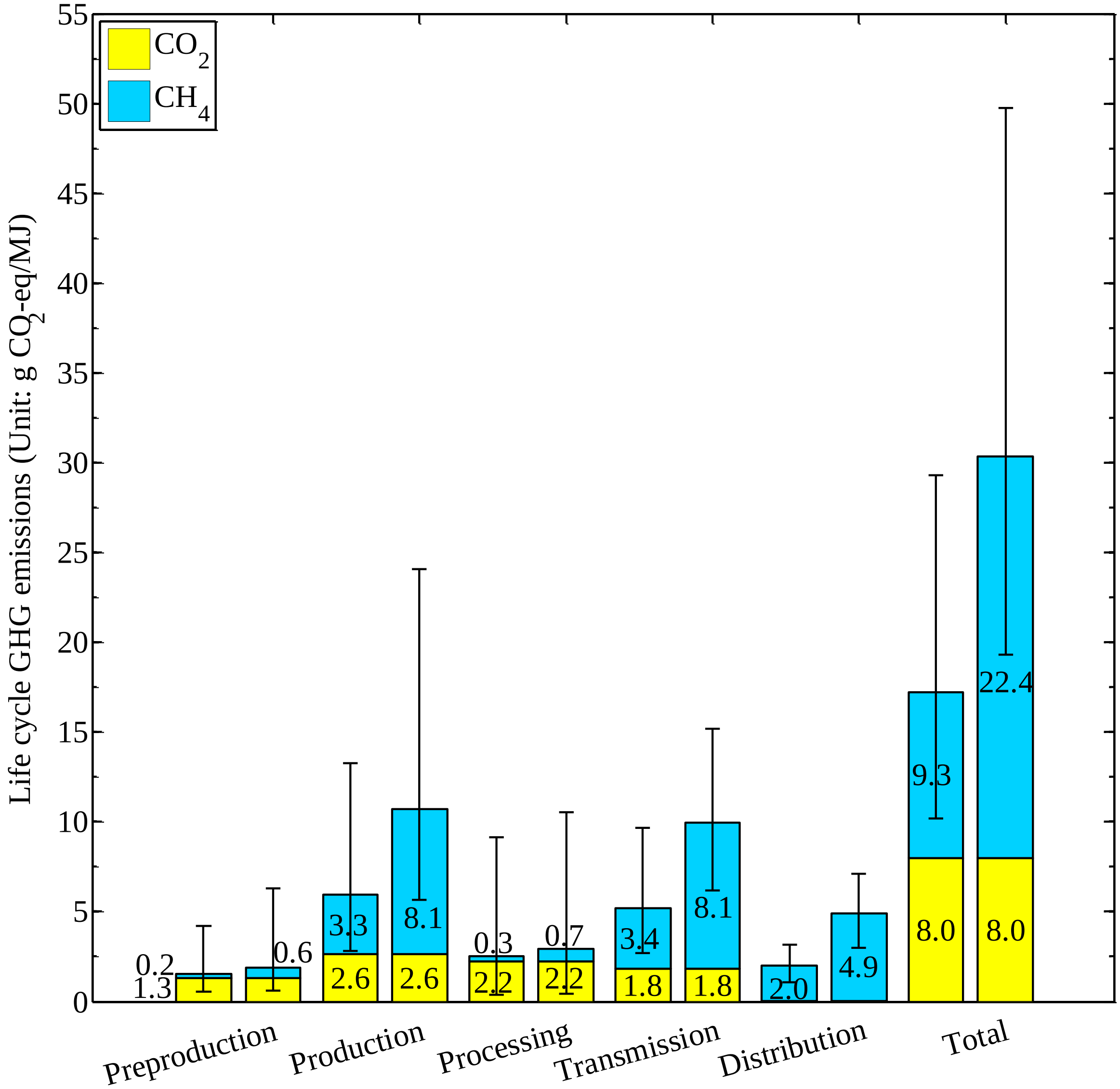


Figure A.2. Breakdown of natural gas upstream GHG emissions by upstream stages and by greenhouse gases. Error bar is calculated based on the 95% confidence interval of the total emissions for each GHG. Estimates with 100-year GWP (left bars) and 20-year GWP (right bars) are shown side by side.

A.3. Life Cycle GHG Emissions of Gasoline and Diesel

We perform a literature review on conventional gasoline, conventional diesel, oil sand-based gasoline, and oil sand-based diesel to characterize the emission factors of the baseline fuel, gasoline or diesel, with which natural gas pathways have to compete. There are noticeable differences across existing studies regarding the combustion emission factors of conventional gasoline and diesel.[168–170,290,291] I model conventional gasoline and diesel in the following way: (1) upstream emissions and its uncertainty characterization are from Venkatesh et al. (2011);[169] (2) combustion emissions of conventional gasoline and diesel are assumed to have triangular distributions based on the range in the literature; (3) life cycle GHG emissions are the sum of upstream emissions and combustion emissions. **Table A.16** shows a summary of the modeling parameters used in this study.

I also consider gasoline and diesel that are refined from oil sands imported from Canada. For these fuels, I rely on Englander et al.[171] for life cycle GHG emissions and assume the combustion emissions are the same as conventional source. Compared to conventional gasoline and diesel, the GHG emissions from oil sand-based gasoline and diesel span over a larger range, and are sensitive to numerous factors, including scope of the LCA, oil sand production method, assumptions about energy intensity and fuel mix, treatment of secondary non-combustion sources, and assumptions with land-use associated emissions[290]. The data in **Table A.16** comes existing studies[171,290,292] and is used in this study.

Table A.16. Life Cycle GHG emissions of gasoline and diesel (Unit: gCO2-eq/MJ of fuel delivered). S.D. stands for standard deviation. C.I. stands for confidence interval.

Fuel	Stage	Distribution	Mean	S.D.	95% C.I.
Conventional gasoline	Upstream	Venkatesh et al. (2011)[169]	18.6	4.0	12.6-28.0
	Combustion	Triangular (71.0, 72.7, 74.9)	72.9	0.8	71.4-74.4
	Life cycle	Upstream plus combustion	91.5	4.1	85.2-101.0
Conventional diesel	Upstream	Venkatesh et al. (2011)[169]	17.5	4.3	11.5-27.7
	Combustion	Triangular (72.6, 74.1, 75.2)	74.0	0.5	72.9-74.9
	Life cycle	Upstream plus combustion	91.5	4.3	85.3-101.7
Oil sand-derived gasoline/diesel	Life cycle	Uniform (103,118)	110.5	4.3	103.4-111.6

A.4. GHG Emissions from the Production of Natural Gas-Based Fuels

Production and distribution of natural gas-based fuels. All natural gas pathways share the same GHG emissions from natural gas systems (with the exception of fuels produced from natural gas liquids[i]). Different natural gas pathways, however, have different emissions associated with their production, transportation and distribution, as I describe below.

A.4.1 NGCC Electricity

In this study I am interested in evaluating how growing natural gas use in the electricity sector affects the life cycle of different fuels. I assume that electricity produced by dedicated Natural Gas Combined Cycle (NGCC) power plants supplies electricity for centrally-located fuel production plants as well as for charging BEVs and PHEVs. For the remaining electricity use (such as demand for electricity at fuel production facilities in refueling stations), I assume grid average electricity. Although it is very difficult to identify individual power plants that produce the electricity used for charging, it is likely that increasing electricity demand from electric vehicles, as well as stringent regulations on new coal power plants,[154,155] will drive further deployment of NGCC plants. In fact, nearly half of the new power plant capacity in U.S. in 2013 was NGCC.[156] In a sensitivity analysis, I evaluate the effects of using grid average emissions instead.

To assess GHG emissions from NGCC electricity in this study, I perform a literature review on GHG emissions of NGCC electricity. Existing studies found that life cycle GHG emissions (upstream and on-site GHG emissions) of NGCC electricity could range between 439 to 529 gCO_2-eq/kWh.[280,293] For studies that reported a heat rate or energy efficiency of NGCC power plant, I find a range from 48.8% to 53.0% (on a HHV basis). Combustion emissions of NGCC

[i] Natural gas liquids are the liquid fuels produced along with dry natural gas. Although the majority of energy produced from natural gas developments is derived from dry natural gas, natural gas liquids have attracted the interests of natural gas developers due to their high economic returns[384]. Typical natural gas liquids include ethane, propane, butane, and iso-butane[287].

electricity vary less compared to upstream emissions as the combustion process is well-known and optimized.

I use the technical assumptions from NETL (2013)[158] for NGCC power plant with and without carbon capture and sequestration (CCS) technologies. The energy efficiencies (on a LHV basis, to be consistent with the heating values used in the paper) are 55.7% and 47.5% for NGCC without and with CCS technologies (the corresponding HHV energy efficiencies are 50.2% and 42.8%). **Table A.17** shows my estimate for life cycle GHG emissions from NGCC power plant, which is similar to emission estimates from existing studies. The life cycle GHG emissions of NGCC without CCS technology have a mean of 456 gCO_2-eq/kWh (with a 95% C.I. of 413-533 gCO_2-eq/kWh). If CCS technology is installed, then life cycle GHG emissions are over 60% lower (165 gCO_2-eq/kWh with a 95% C.I. of 116-254 gCO_2-eq/kWh). For U.S. grid average electricity generations in 2010 (the most recent year that has available data), the mean life cycle GHG emissions factor is 612 gCO_2-eq/kWh.[157] In addition, I assume that 6.5% of generated electricity is lost through transmission and distribution.[159]

Table A.17. GHG emissions for electricity generation (Unit: gCO2-eq/kWh). Mean estimates and the 95% confidence interval (in parenthesis) are shown in table entries.

Electricity source	NGCC without CCS	NGCC with CCS	Grid average (Year 2010) (Cai et al. (2013)[157])
Assumptions			
Energy efficiency	55.7%	47.5%	N/A
Per-kWh Capture Rate+	N/A	88.2%	N/A
Results			
Upstream	98 (57, 174)	115 (67, 204)	48
Combustion	358 (348, 368)	50 (48, 51)	564
Total (at power plant gate)	456 (413, 533)	165 (116, 254)	612

+ Per-kWh capture rate is calculated as the relative changes between combustion emission factors (on net power output basis) of NGCC with CCS (43 kg/MWh) and those of NGCC without CCS (365 kg/MWh).

A.4.2 Compressed Natural Gas (CNG)

The CNG pathway relies on the natural gas pipeline system to deliver natural gas to refueling stations, where CNG is produced by compression. There are two types of compressors for use at refueling stations: electric compressors and natural gas-fueled compressors, with electric compressors being the prevalent choice.[159] I thus assume the use of electric compressors at the refueling stations with the U.S. grid average electricity supply. I further model the energy efficiency of the electric compressor with a uniform distribution of 0.94 to 0.98.[27]

A.4.3 Liquefied Natural Gas (LNG)

LNG is similar to CNG in that natural gas does not undergo chemical conversion. Instead, natural gas is liquefied into LNG through a liquefaction process that increases compression beyond what is achieved for CNG, resulting in a higher energy density of LNG when compared to CNG. The liquefaction process, however, is more energy-intensive. For this study, I consider two configurations for the LNG pathway: distributed production at fueling stations and centralized production. I assume the same liquefaction profile[19,159,294] – a uniform distribution for the energy efficiency of the process, ranging from 89.3% to 91%, with electricity as the additional energy input. The differences between the distributed and centralized liquefaction processes are the source of electricity used and the source of natural gas feedstock. For centralized production, I assume that natural gas combined cycle (NGCC) power plants provide electricity. For distributed production pathway, I assume that grid electricity is used. While distributed LNG production draws natural gas from distribution pipelines, I assume that centralized LNG production draws natural gas from interstate pipelines. In other words, centralized LNG pathway needs to consider fuel transport emissions of LNG while distributed LNG pathway accounts for emissions from natural gas distribution pipelines. To estimate the emissions from this additional transportation stage for centralized LNG pathway, I use the transportation emission factor from the GREET model,[159] as detailed in **Table A.23**. Further, I include methane losses associated with the boil-off effect of LNG, as described in **Table A.21**.

A.4.4 Natural Gas-Based Hydrogen (H_2)

There are three potential configurations for hydrogen supply: centrally-produced gaseous hydrogen (GH_2) used in the FCEV; centrally-produced GH_2 which is then transported and dispensed as liquid hydrogen (LH_2) in the FCEV; and distributed production of GH_2 used in the FCEV. These configurations differ in hydrogen production, transport, storage, and dispensing so there are trade-offs between environmental performance, costs, and implementation difficulties.

There are also three types of hydrogen production plants (using steam methane reforming technologies to convert natural gas to hydrogen), central hydrogen production plant without and with CCS technologies, as well as distributed hydrogen production plant at refueling stations. **Table A.18- Table A.22** summarize the assumptions for these three types of plants. From the perspective of GHG emissions, the central hydrogen plant without CCS has the highest overall energy efficiency; the distributed hydrogen plant has a lower energy efficiency and a wider range of GHG emission factors; the central hydrogen plant with CCS has slightly lower energy efficiency than those without CCS but has a much lower GHG emission factor (almost 90% reduction). The hydrogen pathways also include loss factors to account for the boil-off effect of LH_2 during liquefaction, transport, and storage, as described in **Table A.21**. The manufacturing emissions of fuel cells are listed in **Table A.27**.

Table A.18. Central hydrogen plant profile (without carbon capture and sequestration (CCS)) for one unit of energy (MJ) of hydrogen produced.

Key parameters	**Review of existing studies**				**This study**	
	Spath et al. (2001)[111]	**H2A 3.0**[160]		**GREET 2013**[159]	**Distribution**	**Distribution parameters**
		Current	Future			
Energy efficiency*	0.79	0.72	0.72	0.72	triangular	(0.72, 0.72, 0.79)
Electricity share (of all inputs)	0.007	0.012	0.012	0.044	triangular	(0.007, 0.012, 0.044)
Natural gas share (of all inputs)	0.993	0.988	0.988	0.956	1 – share of electricity as input	
Process GHG emission factor (gCO_2-eq/MJ of H_2)+	N/A	77	77	79	uniform	77-79

**The energy efficiency is the ratio of product output to all energy inputs to the facility (natural gas as feedstock, natural gas as fuel and electric power). + Process GHG emission factor include all GHG emissions within the hydrogen production plant but doesn't include combustion emissions of electricity inputs.*

Table A.19. Central hydrogen plant (with carbon capture and sequestration (CCS)) profile for one unit of energy (MJ) of hydrogen produced.

Key parameters	**Review of existing studies (H2A 3.0**[160]**)**		**This study**	
	Current	**Future**	**Distribution**	**Distribution parameters**
Energy efficiency*	0.72	0.72	point	0.72
Electricity share (of all inputs)	0.012	0.013	uniform	0.012-0.013

Natural gas share (of all inputs)	0.988	0.987	1 – share of electricity as input	
Process GHG emission factor (gCO_2-eq/MJ of H_2)+	8	8	point	8

**The energy efficiency is the ratio of product output to all energy inputs to the facility (natural gas as feedstock, natural gas as fuel and electric power). + Process GHG emission factor include all GHG emissions within the hydrogen production plant but doesn't include combustion emissions of electricity inputs.*

Table A.20. Hydrogen production profile at refueling stations for one energy unit (MJ) of hydrogen produced.

Key parameters	Review of existing studies			This study	
	H2A Case study[160]		GREET, 2013[159]	Distribution	Distribution parameters
	Current	Future			
Energy Efficiency*	0.71	0.74	0.71	triangular	(0.71, 0.71, 0.74)
Electricity share (in all inputs)	0.024	0.050	0.083	triangular	(0.024, 0.050, 0.083)
Natural gas share (in all inputs)	0.976	0.950	0.917	1 – share of electricity as input	
Process GHG emission factor (gCO_2-eq/MJ of H_2)+	77	72	81	triangular	(72, 77, 81)

**The energy efficiency is the ratio of product output to all energy inputs to the facility (natural gas as feedstock, natural gas as fuel and electric power). + Process GHG emission factor include all GHG emissions within the hydrogen production plant but doesn't include combustion emissions of electricity inputs.*

A.4.5 Boil-Off Effects of LNG and Liquid H_2

LNG liquefaction, transport, and storage result in boil-off of natural gas. A similar effect occurs in liquefaction, transport, distribution, and storage of liquid hydrogen. I use loss factors to take into account for the boil-off effect of LNG during liquefaction, transport, and storage (**Table A.21**) with an 80% capture and reuse rate of boil-off gas.[159]

Table A.21. Loss factor used in the LNG and Liquid H_2 (LH_2) pathways[159] (80% capture rate of boil-off natural gas is applied).

Loss factor	Liquefaction	Transport and distribution	Storage
LNG central	1.001	1.0005	1.0016
LNG distributed	1.001	-	-
LH_2	1.003	1.0016	1.005

A.4.6 Fischer-Tropsch (F-T) Liquids

In the F-T liquid production process, natural gas is delivered through the transmission system and undergoes thermo-chemical transformation into liquid fuels similar to those produced in an oil refinery. **Table A.22** summarizes process-level data for an F-T plant and I include a case with a carbon capture rate of 90%. Following Jaramillo et al. (2008),[161] I perform an emission allocation to liquid co-products on an energy basis. Upon production, F-T liquids are transported using existing petroleum-product infrastructure and their combustion emissions are comparable to those of conventional gasoline and diesel.

Table A.22. Production profile for centralized F-T liquids production plant (Jaramillo et al. (2008)).[161]

Input					Output			
Natural gas (million m^3/hr)	**Butanes (metric tons/day)**	**Electricity (MWh/day)**			**Propane /LPG (TJ/day)**	**Gasoline (TJ/day)**	**Diesel (TJ/day)**	**Carbon lost (metric tons/day)**
		No CCS	**90% CCS**					
			min	**max**				
0.6	32	-590*	-155*	170+	6.2	77	137	1,651

**Negative values for electricity suggest that the F-T plant sells electricity to the grid. +I assume on-site generation of this electricity.*

A.4.7 Propane

While heating is the current primary use, propane can also be used as a transportation fuel. In fact, propane (also known as liquefied petroleum gas, LPG) is the third most used alternative transportation fuel in the U.S. after ethanol and CNG.[295] To estimate the life cycle emissions of propane, I allocate energy-based GHG emissions associated with natural gas preproduction, production, and processing. The fueling infrastructure of propane is similar to that for gasoline and diesel, and the fueling station costs are much cheaper than CNG and LNG.[296] The conversion from propane feedstock to propane fuel has an energy efficiency of 96.5%[159] and the additional energy inputs are natural gas (96%), electricity (3%) and diesel (1%).[159]

A.5. GHG Emissions from Fuel Transport

All the pathways in this study can be summarized in three groups in terms of transportation between where the fuel is produced and where the fuel is pumped into vehicles (fueling station):

distributed pathways (CNG, GH_2 distributed, LNG distributed), electricity transmission, and liquid pathways (conventional gasoline, conventional diesel, oil sand-based gasoline, oil sand-based diesel, E85, M85, LPG, F-T liquids, GH_2 central, LH_2 central, and LNG central). For distributed pathways, natural gas transportation is accounted in the natural gas upstream (see *Section A.2 Natural Gas Upstream GHG Emissions* in this Appendix). For electricity transmission, I assume a 6.5% loss.[159] For gasoline and diesel pathways, GHG emissions from transportation are accounted in the "upstream" emissions, as discussed in the previous section. For the remaining liquid pathways, I rely on the transportation emission factors reported by the GREET 2013.[159,297] I summarize liquid fuel transport assumptions and their resulting emission factors from the GREET 2013[159] in **Table A.23** and **Table A.24**. The GREET 2013 model[159] does not have a F-T gasoline pathway, so I assume that F-T gasoline has the same transportation emission factor as F-T diesel transportation emission factors are generally very small (see **Table A.24**).

Table A.23. Fuel transport assumptions (GREET 2013[159]). Note that fuel transport has two stages, the first of which consists of barge, pipeline, and rail, and the second stage is truck.

Fuel		F-T Gasoline/Diesel				Propane/LPG			
Transportation mode		Barge	Pipeline	Rail	Truck	Barge	Pipeline	Rail	Truck
Distance (mile, one-way)		200	308	490	30	520	400	800	30
Share of transportation mode		48.5%	46.4%	5.1%	100%	6%	60%	34%	100%
Fuel share for consumed energy	diesel			100%	100%			100%	100%
	residual oil	100%				100%			
	electricity		100%				100%		
Fuel		**GH_2 central**		**LH_2 central**			**LNG central**		
Transportation mode		Pipeline	Truck	Barge	Pipeline	Truck	Barge	Pipeline	Truck
Distance (mile, one-way)		750	30	520	750	30	520	750	30
Share of transportation mode		100%	0%	50%	50%	100%	50%	50%	100%
Fuel share for consumed energy	diesel		100%			100%			100%
	residual oil			100%			100%		
	electricity	100%			100%			100%	

Table A.24. Fuel transport emission factors (Unit: gCO_2-eq/MJ_{LHV}) (GREET 2013[159]).

Fuel	F-T liquids	GH_2 central	LH_2 central	LNG central
CO_2	1.0	4.8	0.6	0.8
CH_4	0.002	0.014	0.001	0.001
N_2O	1×10^{-5}	6×10^{-5}	9×10^{-5}	1×10^{-5}

A.6. Vehicle Assumptions

For this analysis, I model new vehicles available in the market rather than existing fleets. I use functionally equivalent vehicles for different fuel pathways within a specific vehicle segment to eliminate the bias of vehicle choices.[72] **Table A.25** summarizes the vehicle specifications for the different fuel pathways and vehicle specifications included in this analysis. I discuss the vehicle specifications in detail below.

Table A.25. Vehicle specifications for different fuel pathways and different vehicle applications[i].

		Class 2b Pickup truck/van	Class 4 Package delivery van	Class 6 Beverage truck	Class 8 Transit bus	Class 8 Refuse hauler	Class 8 Tractor trailer	
							local-haul	line-haul
								
Fuel economy								
Unit of fuel economy+		(MPGGE)	(MPGDE)	(MPGDE)	(MPGDE)	(MPGDE)	(MPGDE)	(MPGDE)
Gasoline/FTG		14*	-	-	-	-	-	-
Diesel/FTD		16.1	11.5*	7.0*	4.0*	3.3*	4.3*	6.5*
Gasoline-HEV		16.8	10.9	-	4.0	-	-	-
Diesel-HEV		19.3	14.4	9.3	4.8	3.6**	5.2	7.2
CNG		14	10.8	6.6	3.6	2.9	3.9	5.9
LNG SI		-	-	-	3.6	2.9	3.9	5.9
LNG CI		-	-	-	-	-	4.2	6.4
LPG/Propane		14	-	-	-	-	-	-
BEV		42	34.5	21.0	16.8	-	-	-
H_2-FCEV		-	-	-	7.6	-	-	-
Vehicle weight and payload								
Conventional	Gross weight (lbs.)	8,501-10,000[55]	16,000[298]	19,501-26,000[55]	39,980[299]	60,000[300]	80,000[64]	80,000[64]
	Empty weight (lbs.)	5,000-6,300[55]	9,700[298]	11,500-14,500[55]	27,730[299]	16,627[300]	30,500[64]	35,550[64]
	Payload (lbs.)	3,700[55]	6,300[298]	11,500[55]	12,150[299]	43,373[300]	49,500[64]	44,450[64]

*+ Different vehicle segments have different baseline petroleum fuels (gasoline for Class 1-2 and diesel for Class 3-8) so the same 'gallon' has different meanings in different vehicle segments. * Baseline petroleum fuel-engine pathway is marked and highlighted in gray. ** A diesel refuse truck with hydraulic hybrid system is assumed.*

[i] In this study, we ignore the effects of natural gas composition on fuel economy and vehicle exhaust emissions of natural gas vehicles. Refer to existing studies[385,386] on this issue.

A.6.1 Fuel Economy

There are many vehicle segments in MHDVs,[4,70,112,301] and I restrict the focus to the following vehicles: Class 2b pick-up truck, Class 4 package delivery van, Class 6 box truck (for instance, beverage delivery truck), Class 8 transit bus, Class 8 refuse truck, and Class 8 tractor-trailer. The baseline petroleum pathways are conventional gasoline (for Class 2b) and conventional diesel (for Class 4-8), as shown in **Table A.25**.

For Class 2b pick-up truck or van, I consider the Ford F-250[302] or similar trucks (such as GMC 2500 series, or the RAM 2500) as the representative vehicle. The fuel economy information for Class 2b pickup is taken from TIAX[112] and the Clean Energy Coalition.[303] Conventional gasoline is assumed to be the baseline pathway. I assume that diesel truck increase fuel economy by 15%. I further assume that gasoline HEVs and diesel HEVs will increase fuel economy by 20% compared to conventional pick-up trucks with the same type of fuel.[74,304,305] BEV will increase fuel economy by 200% compared to the baseline gasoline truck.[112] In addition, propane or LPG is also a viable option on the market, and its fuel economy is the same as a gasoline truck.[306]

Package delivery vans are step-vans are designed to deliver packages. Package delivery vans span over Class 3, 4 and 6 (10,000-26,000 lb. gross vehicle weight ratings). In addition to payload differences, Class 4 and Class 6 package delivery trucks differ in the available choices of alternative fuels. While both Class 4 and Class 6 trucks have the same diesel engine, gasoline hybrid and battery electric engines are currently only available for Class 4 but not for Class 6.[307–312] As a result, I focus on a Class 4 step vans for this study. The Freightliner P70D step van[307,308,310] and the Smith Electric Newton Step Van[313,314] are the representative vehicles for diesel and electric pathways, respectively. I rely on TIAX[112] for fuel economy estimates, but revise the fuel economy benefit of diesel hybrid compared to a diesel truck to 25%[312] (i.e. the fuel economy of diesel hybrid is 25% higher than conventional diesel). For gasoline hybrid, I assume its fuel economy is 95% that of the conventional diesel truck,[298,315] on the diesel gallon basis. For BEVs, I follow TIAX by assuming a 200% efficiency improvement relative to conventional diesel van, while existing studies showed a range of 150-330%.[87,313] The fuel economy of CNG truck is assumed to be 6% lower.[112]

Box trucks are usually used to deliver food and beverage in urban areas. I model a Class 6 box truck, for which representative technical specifications can be found in TIAX.[301] I further rely on TIAX[112] to estimate the fuel economy of different engines. For BEVs, I use the battery size and vehicle specification information from Smith[316] and NREL.[317,318]

Transit buses carry passengers mostly in urban areas and have a low fuel economy due to its severe operational characteristics - low-speed and a large share of stop-and-go operations.[319–321] Compared to other MHDVs, air emissions of transit buses are relatively well studied.[38,42,83,90,320,322–328] The typical size of a transit bus is either 40-foot long or 60-foot long, with the 40-foot bus being the dominant choice.[319–321] I model the 40-foot long buses - New Flyer XD40, New Flyer C40LF, and BYD electric bus as the reference transit buses[38,299,329] for diesel, CNG, and BEV, respectively. For fuel economy estimates, I rely once again on TIAX[112] with some modifications.

- There is a wide range (from 73% to 109%) of relative fuel economy of the CNG-SI bus compared to the conventional diesel-CI.[5,38,42,48,112,243,320,330–336] The lowest relative fuel economy is reported for the New York City bus duty cycle, where engines operate at low speed and low load for most of the time, a worse scenario for SI engines than for CI engines.[129,334,336] In this study, I assume the mid-point of the range found in the literature[38,48,112,243] (90%) as the relative fuel economy of a CNG-SI bus when compared to a conventional diesel-CI bus.
- CNG-SI and LNG-SI share the same engine (SI-ICE), though fuel systems are slightly different. I assume the CNG-SI bus and LNG-SI have the same fuel economy, thus ignoring any differences brought by fuel systems.[5,70,320]
- The efficiency improving benefits of diesel-HEVs differ across studies. The drive cycle is one of the most influencing factors among all. Hallmark et al.[328] find a range from 10% to 36% for efficiency-improving benefits from their review of existing studies. I assume a

20% increase for diesel-HEVs over conventional diesel-CI, which aligns well with MJB&A,[38] a recent and technology-rich study that summarizes results from vehicle tests.

- The fuel economy of hydrogen FCEV buses is assumed to be 90% higher than that of conventional diesel buses, based on a reported range of efficiency increase of 82% to 131%.[177]
- The fuel economy of BEV buses is assumed to be 320% higher than that of conventional buses based on actual vehicle tests.[299,337,338] I note that there are variations among new BEVs (BYD bus's fuel economy is 290% higher than New Flyer's diesel bus; but Proterra's fuel economy is 350% higher).[299,337,338] The increase in relative fuel economy is also highly dependent on drive cycle; it ranges from 230% for commuter phase (COM cycle) to 270% for arterial phase (ART cycle), and 420% CBD (Central Business District) Phase.[299,337,338] In other words, BEVs could achieve larger efficiency benefits in more demanding drive cycles, such as CBD.

Refuse haulers/trucks form a niche fleet market and their average fuel economy is only about 2.8 MPGde (miles per gallon diesel equivalent)[339] due to their stop-and-go drive cycle and the need to provide "power take off" (PTO) operation. There is already a promising CNG market thanks to cost savings and reduced tailpipe air emissions.[129,339–341] I rely on TIAX[112] for fuel economy estimates for diesel and diesel-hybrid refuse trucks. Reference refuse trucks include AutoCar ACX, Peterbilt 320, and Mack TerraPro.[129,339,342] Existing studies indicate a range of relative fuel economy for CNG refuse trucks from 83% to 96%[33,88,89,243,330,342] (an outlier 59% was found in Sandhu et al.[89]), and thus I assume the relative fuel economy of CNG and LNG refuse trucks to be 88% of the conventional diesel fuel efficiency, similar to those reported by recent in-use results.[89,342] The hybrid refuse truck is assumed to be a conventional diesel truck with an onboard hydraulic system, and has a 10% increase in energy efficiency.[84,343]

Tractor-trailers haul more freight ton-miles than any other transportation mode.[344] They also represent one of the fastest-growing road transportation segments in terms of fuel consumptions and GHG emissions.[109,345] I consider two types of tractor-trailers: line-haul, which is assumed to haul freight along interstate highways; and local-haul, which has a return-to-base operation schedule.[34,64] I assume that both types of tractor-trailers have the same fuel pathways, but fuel economy estimates vary due to different payload and drive cycle specifications. I rely on TIAX[112] for fuel economy estimates for diesel tractor-trailers. I consider that CNG-SI and LNG-SI trucks have a 10% energy efficiency penalty[34,64,112,346,347] and LNG-CI truck has a 2% energy efficiency penalty[34,64,82,85,86,112] compared to diesel trucks. This reference LNG-CI truck uses the Westport High-Pressure Direct Injection (HPDI) technology, and 5% diesel consumption is needed for pilot ignition.[64,82,85,86,348] For hybridization, the assumed fuel efficiency benefit is larger (20%) for local-haul and smaller (10%) for line-haul due to different shares of stop-and-go and interstate highway in their drive cycles.[74–76,78,79]

A.6.2 Vehicle Payloads

To calculate payload penalties of natural gas fuel pathways compared to the baseline gasoline/diesel pathway, I reviewed literature and summarized the analysis into **Table A.26** (vehicle tare weight[i] and gross vehicle weight assumptions for the baseline gasoline/diesel pathway are summarized in **Table A.25**). For baseline pathway, I determined their payloads in the following ways: (1) for transit buses and refuse trucks, I determined the payload based on actual vehicle specifications; (2) for tractor trailers, I assumed the gross vehicle weight (GVR) is at the federal weight limit on Interstate highways, which is 80,000 lbs. (certain western states, however, have grandfather rights to allow longer combination vehicles weighing more than 80,000 lbs.);[349] (3) for other MHDVs, I relied on the recent NRC report.[55] It should also be noted that transit buses are used to move people rather than freight, and the payload of a transit bus is approximated by the total weights of bus riders (assuming each bus rider weighs 150 lbs.) and assuming all the seats and free floor space are taken, as determined by the bus tests at the Thomas D. Larson Pennsylvania Transportation Institute.[299]

I assume non-electric fuel pathways are equipped with fuel tanks large enough to have the same amount of usable fuels (measured as diesel equivalent gallon (DGE)) as the conventional gasoline or diesel pathways for the same vehicle class. Vehicle range depends on vehicle fuel

[i] Tare weight, also called curb weight or unladen weight, is the weight of an empty vehicle. Gross vehicle weight (laden weight) equals the sum of vehicle tare weight and the weight of the goods carried (the net weight, or payloads).

economy and the percentage of usage fuels in a full fuel tank[i]. Thus different driving cycles may result in different vehicle ranges even if the fuel tanks are full.

For the CNG and LNG pathways, I calculate the differences between tare weights of natural gas vehicles and those of baseline diesel vehicles when vehicle weight information (e.g. vehicle tests, vehicle specification, etc.) is not available. I assume that the payload loss is the same as incremental curb weight. In these bottom-up calculations, I consider the following vehicle components: engine, fuel system (fuel tank with fuels at full capacity), and after-treatment system. I rely on Deal et al.[64] for weight estimates of these components. Most current natural gas engines, Cummings Westport 8.9L ISL G, and Cummings Westport 11.9L ISX G, can be used with either CNG or LNG, use sparking ignition (SI) engine cycle, and don't require after-treatment system (such DPF, SCR, etc.) to meet the U.S. EPA 2010 emissions standard. The LNG-CI engine Westport 15L ISX, however, has to use both LNG and diesel (as a pilot for ignition), and requires the same after-treatment system as conventional diesel engine to meet the emission standard. These differences are considered in the weight analysis.

For HEVs, I assume the payload losses are equal to the extra weight of the hybrid electric system (electric motor, battery, etc.) unless better information (such as those from vehicle tests) is available. For BEVs, I rely on estimates of curb weight increases from similar vehicles (for pick-up trucks), and vehicle tests (for transit buses).

Finally, I have to emphasize that these payload comparisons are generalized and may not represent the actual operations of HDVs. For instance, only 35% of tractor trailers are 'weight out' (carrying its maximal payload) while the other 65% are 'cube out' (carrying freight until there is no empty space)[63]. Similarly, Zhao et al. find that "Class 8 combination trucks … [use] 77% of the 80,000 lbs. weight allowed" by examining the federal highway data.[79] In view of these data, what I consider might be an extreme case for estimating payload losses in tractor trailers by assuming all the tractor trailers are 'weight out'.

Table A.26. Vehicle payload penalties relative to baseline gasoline/diesel pathways.

[i] CNG and LNG vehicles have lower percentages of usable fuels than conventional diesel vehicles.

Vehicle	Fuel	Payloads for baseline pathways (highlighted in grey) or payload penalties for other pathways (lbs.)
Class 2b pick-up truck	Conventional gasoline (baseline)	3,700 lbs.[55]
	Conventional diesel, F-T diesel, oil sand diesel, LPG/Propane, F-T gasoline, oil sand gasoline	0 lbs. (assumed to be the same as baseline)
	CNG	200 lbs.[350]
	Gasoline/Diesel HEV	350 lbs.[305] (only considering hybrid fuel system)
	BEV	600 lbs. (a conservative estimate based on the differential in curb weights of the gasoline version and all-electric version Toyota RAV4)
Class 4 package delivery truck	Conventional diesel (baseline)	6,300 lbs.[298] (30 DGE usable tank size)
	Conventional gasoline, F-T gasoline, oil-sand gasoline	0 lbs. (assumed to be the same as baseline)
	F-T diesel, oil-sand diesel.	
	CNG	0 lbs. (450 lbs. weight reduction due to no after-treatment system weight, no difference in engine weight, and 450 lbs. weight increase from a 30-DGE fuel system (a side rail tank); based on [64])
	Gasoline HEV	0 lbs.[315]
	Diesel HEV	0 lbs.[310]
	BEV	200 lbs. (payload difference between the BEV in Lee et al.[313] and conventional diesel in [315])
Class 6 beverage delivery truck	Conventional diesel (baseline)	11,500 lbs.[55] (75 DGE usable tank size)
	F-T diesel, oil-sand diesel.	0 lbs. (assumed to be the same as baseline)
	Diesel HEV	1,200 lbs.[315]
	CNG	515 lbs. (450 lbs. weight reduction due to no after-treatment system weight, 175 lbs. weight reduction in engine weight, and 1140 lbs. weight increase from a 74-DGE fuel system (two 41.2 DGE side-rail mounted tanks); based on [64])
	BEV	200 lbs. (assumed to be the same as Class 4 BEV)
Class 8 transit bus	Conventional diesel (baseline)	12,150 lbs.[299] (150 DGE usable tank size)
	F-T diesel, oil-sand diesel.	0 lbs. (assumed to be the same as baseline)
	Gasoline HEV	0 lbs. (assumed to be the same as baseline)[351]
	Diesel HEV	750 lbs.[352]
	CNG	900 lbs.[329]
	LNG (SI)	1150 lbs. (450 lbs. weight reduction due to no after-treatment system weight, 175 lbs. reduction in engine weight, and 1780 lbs. weight increase from a 150-DGE fuel system; based on [64])
	H_2 FCEV	5,400 lbs.[353]
	BEV	4,800 lbs.[337]

Class 8 refuse truck	Conventional diesel (baseline)	43,373 lbs.[300] (75 DGE usable tank size)
	F-T diesel, oil-sand diesel.	0 lbs. (assumed to be the same as baseline)
	Diesel hybrid	400 lbs.[343]
	CNG	915 lbs. (450 lbs. weight reduction due to no after-treatment system weight, 175 lbs. reduction in engine weight, and 1780 lbs. weight increase from a 75-DGE fuel system (one 150 gallon side-rail tank)[i]; based on [64])
	LNG (SI)	265 lbs. (450 lbs. weight reduction due to no after-treatment system weight, 175 lbs. reduction in engine weight, and 1780 lbs. weight increase from a 75-DGE fuel system (one 150 gallon side rail tank); based on [64])
Class 8 local-haul tractor trailer	Conventional diesel (baseline)	49,500 lbs.[64] (75 DGE usable tank size)
	F-T diesel, oil-sand diesel.	0 lbs. (assumed to be the same as baseline)
	Diesel hybrid	880 lbs.[76]
	CNG	502 lbs. (450 lbs. weight reduction due to no after-treatment system weight, 188 lbs. reduction in engine weight, and 1140 lbs. weight increase from a 75-DGE fuel system (two 41.2-DGE side rail tanks); based on [64])
	LNG (SI)	252 lbs. (450 lbs. weight reduction due to no after-treatment system weight, 188 lbs. reduction in engine weight, and 890 lbs. weight increase from a 75-DGE fuel system (one 150-DGE side rail tank); based on [64])
	LNG (CI)	1249 lbs. (43 lbs. reduction in engine weight, and 1292 lbs. weight increase from a 75-DGE fuel system (one 120-gallon LNG side rail tank); based on [64])
Class 8 line-haul tractor trailer	Conventional diesel (baseline)	44,450 lbs.[64] (150 DGE usable tank size)
	F-T diesel, oil-sand diesel.	0 lbs. (assumed to be the same as baseline)
	Diesel hybrid	880 lbs.[76]
	CNG	2042 lbs. (450 lbs. weight reduction due to no after-treatment system weight, 188 lbs. reduction in engine weight, and 2680 lbs. weight increase from a 142-DGE fuel system (two 41.2-DGE side rail tanks and five 15-DGE behind cab tanks); based on [64])
	LNG (SI)	1142 lbs. (450 lbs. weight reduction due to no after-treatment system weight, 188 lbs. reduction in engine weight, and 1780 lbs. weight increase from a 150-DGE fuel system (two 150-DGE side rail tanks); based on [64])
	LNG (CI)	2541 lbs. (43 lbs. reduction in engine weight, and 2584 lbs. weight increase from a 120-DGE fuel system[ii] (two 120-gallon LNG side rail tanks); based on [64])

[i] The volume size of the tank is 150 gallon, or 87 DGEs. However, the LNG tank has a lower utilization rate than diesel tank[64]. So the usable tank size is 75 DGEs. The same explanation works for other CNG and LNG trucks.
[ii] Note that the LNG-CI truck has only a usable tank of 120 DGE rather than 150 DGE (the baseline assumption for line-haul tractor trailers). According to Deal et al.,[64] 150-gallon tanks cannot work with the LNG HPDI engine technology.

A.6.3 Battery and Fuel Cell Manufacturing Emissions

I consider battery and fuel cell manufacturing emissions for HEVs, BEVs, and FCEVs as they are 'new' emission sources with these pathways. I rely on Dunn et al. (2012)[176] and GREET 2013[159] for battery and fuel cell manufacturing emission factors. I collect assumptions related to vehicle lifetime mileage and battery/fuel cell sizes from a number of sources. Details in terms of lifetime replacements of batteries and fuel cells are in **Table A.27**.

Table A.27. Battery and fuel cell stack specifications.

Application	Technology	Lifetime	Battery/Fuel cell power plant (FCPP) type	Numbers of batteries/FCPP per vehicle lifetime
Hybrid and battery electric vehicles				
Class 2b pick-up truck	Gasoline/diesel HEV	15 years, 200,000 miles[112]	1.8 kWh Li-Ion battery[305]	equal probability, 1, 2 (author's judgment)
	BEV		40 kWh Li-Ion battery[354]	
Class 4 parcel delivery truck	Gasoline HEV	20 years, 400,000 miles[112]	2.45 kWh Li-Ion[315]	3 (assuming battery lifetime is 7 years[i])
	Diesel HEV		1.8 kWh Li-Ion[309]	
	BEV		100 kWh Li-Ion[87]	
Class 6 box truck	Diesel HEV	15 years, 400,000 miles[112]	2 kWh[355]	2 (assuming battery lifetime is 7 years)
	BEV		120 kWh[316,356]	
Class 8 transit bus	Diesel HEV	12-15 years, 500,000 miles[48,112,177,357]	6 kWh[357]	equal probability, 1, or 2[ii]
	BEV		323 kWh Li-Ion[358]	
Class 8 local-haul tractor trailer	Diesel HEV	6 years, 240,000 miles[64,112]	15 kWh Li-Ion[34]	1 (assuming battery life is 7 years)
Class 8 line-haul tractor trailer	HEV	4 years, 480,000 miles[112 iii]	5 kWh Li-Ion[75]	1 (assuming battery life is 7 years)
Fuel cell electric vehicles (FCEV)				
Class 8 Transit bus	FCEV		120 kW fuel cell[177]	Equal probability 4, or 5 (based on

[i] NREL reports that "the service life of the battery is estimated by Eaton at more than 7 years"[312]. Electric truck manufacturer Smith finds that "[lithium ion] battery should still have a minimum of 80% capacity after 3,000 cycles [of fully charging and discharging]"[387], suggesting a 10-year lifetime. We assume a battery lifetime of 7 years.

[ii] The electric transit bus manufacturer BYD states that one battery per lifetime is achievable[358]. However, according to the Federal Transit Administration (FTA), "the longest warranty coverage of a current bus equipped with LIB [Lithium Ion Batteries] is 5-6 years"[357]. Since the lifetime of a transit bus is much longer than the battery lifetime assumed (7 years), we assume a 50% probability of battery replacement.

[iii] There is wide range of lifetime mileages for Class 8 tractor trailers. A case study of a regional city-to-landfill hauls of waste (80% interstate highway) reported a truck lifetime of 6 years, or 600,000 miles, which is 25% higher than the assumption in our paper.

		12-15 years, 500,000 miles[112,177,357]		lifetime operation hours)
			21 kWh Li-Ion battery[177]	Equal probability, 1, or 2 (assumed to be the same as for hybrid or battery electric buses).
Manufacturing emissions				
Battery manufacturing emissions		5.1 kg CO_2-eq/kg[176]		
Battery specific energy	HEV	0.11 kWh/kg of battery[176]		
	BEV	0.13 kWh/kg of battery[176]		
Fuel-cell manufacturing emissions		23.7 kg CO_2-eq/kW (GREET 2013[159])		
PHEV/BEV charging efficiency		Uniform distribution: 85%-88%[23,134,159]		

A.6.4 Tailpipe Methane and N_2O Emissions

Incomplete combustion in the vehicles results in emissions of methane and N_2O from the tailpipe. Current estimates on methane leakage from vehicle operation do not agree with each other[28,35,36,359–362] thus calling for the need for additional tests on methane leakage from vehicle's tailpipe.[35,106] I use tailpipe CH_4 and N_2O emission factors from U.S. EPA[362] for MHDVs (except CNG and LNG vehicles). Electric vehicles (both BEVs and FCEVs) are assumed to have zero tailpipe emissions.

Historically, CNG and LNG vehicles have much higher CH_4 emission factors than the baseline petroleum pathways because of incomplete combustion and fuel system leaks (a potentially severe problem for LNG due to the boil-off effect). For instance, Chandler et al. (2000) found that an emission rate of methane at 17.7 gram/mile from LNG truck tests.[363] After reviewing vehicle emission tests undertaken at West Virginia University (WVU)'s mobile testing facility, Davies et al. (2005)[35] argued that because "combustion emissions of CH_4 […] are less directly related to fuel composition […] can therefore not be easily derived and instead must be determined through use of published emission factors for each combination of fuel, end-use technology, combustion conditions, and emissions control system." Davies et al. (2005)[35] subsequently recommended "the preferred method of calculating these emissions is based on mileage." Thus, I represent tailpipe methane emissions from CNG and LNG vehicles as leakage rates from vehicle operations (expressed as a percentage of fuel use).

I first performed a review of existing studies on tailpipe methane emission factors (**Table A.28-Table A.30**). I found that tailpipe methane emissions are closely related to the vehicle engine technologies, and vehicles model years. While older studies found higher methane emissions factors, more recent studies reported emission factors that are an order of magnitude lower. During the same period, there have not only been changes in terms of vehicle emission regulations, but also major improvements in natural gas engine technologies. Since my focus is on the new natural gas vehicles on the market, I decided to use recent studies (as listed in **Table A.31** and **Table A.32**) as the basis for my assumptions.

Table A.28. Methane emission factors of Class 4 parcel delivery trucks.

Study	Fuel (Model Year)	Methane Emission Factor (Unit: g/mile)	Source of Data
Chandler et al. (2002)[364]	CNG (1996)	7.2	Vehicle test.

Table A.29. Methane emission factors of transit buses. Recent transit bus technologies are marked with an asterisk *.

<table>
<tr><th>Study</th><th>Fuel (Model Year)</th><th>Methane Emission Factor (Unit: g/mile)</th><th>Source of Data</th></tr>
<tr><td rowspan="2">Graham et al. (2008)[36]</td><td>Diesel (2006)</td><td>0.076 g/km</td><td rowspan="2">Literature review on emission measurements by Environment Canada, Emissions Research and Measurement Division (ERMD). Note that this study was done for Canada.</td></tr>
<tr><td>CNG (2006)</td><td>6.26 g/km</td></tr>
<tr><td rowspan="2">Hesterberg et al. (2008)[33]</td><td>Diesel</td><td>0.03
0.02 (oxidation catalysts)
0.00 (catalyzed particulate filters (traps))</td><td rowspan="2">Literature review on studies from various sources.</td></tr>
<tr><td>CNG</td><td>9.97
20.98 (oxidation catalysts)
2.75 (three-way catalysts)</td></tr>
<tr><td rowspan="3">Weigel et al. (2010)[327]</td><td>Diesel</td><td>0.0051</td><td rowspan="3">The Climate Registry. General Reporting Protocol. Version 1.1. http://www.theclimateregistry.org/downloads/GRP.pdf. Accessed July 7, 2009.</td></tr>
<tr><td>Gasoline</td><td>0.2356</td></tr>
<tr><td>CNG</td><td>1.966</td></tr>
<tr><td rowspan="2">MJB&A (2012)[48] *</td><td>Diesel (2012)</td><td>0.002</td><td rowspan="2">Based on MOVES</td></tr>
<tr><td>CNG (2012)</td><td>1.080</td></tr>
</table>

MJB&A (2013)[38] *	Diesel (2011)	0.48 (Manhattan Cycle) 0.27 (Orange County Cycle) 0.38 (UDDS cycle)	Based on actual vehicle test
	CNG (2011)	0.61 (Manhattan Cycle) 0.37 (Orange County Cycle) 0.20 (UDDS cycle)	
GHGenius v4.03 (2013)[94] *	ULSD	0.08 (g/km)	Not explicitly mentioned.
	CNG	1.27 (g/km)	
	LNG	1.25 (g/km)	

Table A.30. Methane emission factors for heavy-duty trucks. Recent transit bus technologies are marked with an asterisk *.

Study	Fuel (Model Year)	Methane Emission Factor (Unit: g/mile)	Source of Data
Chandler et al. (2000)[363]	LNG	17.7; (or equivalently) 2.7 percent of vehicle fuel supply.	Based on actual vehicle test
TIAX (2008)[95]	ULSD	3.03	Not explicitly mentioned
	CNG/LNG	0.009	
Graham et al. (2008)[36]	Diesel	Not measured	Based on actual vehicle test. Used by Arteconi et al. (2010).
	LNG (HPDI)	2.62 (g/km)	
Weigel et al. (2010)[327]	Diesel	0.0051	The Climate Registry. General Reporting Protocol. Version 1.1. http://www.theclimateregistry.org/downloads/GRP.pdf. Accessed July 7, 2009.
	CNG/ LNG	1.966	
Meyer et al. (2011)[46]	ULSD	<0.02	Not explicitly mentioned.
	CNG	0.3	
	LNG	1.0	
Alvarez et al. (2012)[28]	CNG	0.15% (leakage rate, as of total natural gas produced).	Meyer et al. (2011).
Santini et al. (2013)[92] *	Diesel	0.013	Not explicitly mentioned.
	LNG (HPDI)	0.451	
	CNG/LNG (SI)	0.546	
Meier et al. (2013)[44] *	LNG (HPDI)	1.6 g/mile; (or equivalently 0.34 percent of vehicle fuel supply).	Based on information from engine manufacturer.
GHGenius v4.03 (2013) [94] *	ULSD	0.07 (g/km)	Not explicitly mentioned. Note that this study was done for Canada.
	CNG	1.27 (g/km)	
	LNG	1.25 (g/km)	

I then calculate the methane leakage rate as a percentage of fuel supply for the corresponding methane emission factor. I use the fuel economy assumptions from the original studies to ensure consistency. The calculated tailpipe methane leakage rates are shown in **Table A.31** and **Table A.32**.

Of these five studies listed, only MJB&A (2013)[38] is directly based on actual emission tests of modern natural gas vehicles. MJB&A (2012)[48] is based on U.S. EPA's MOVES model and there are not enough details on their other assumptions (such as drive cycle and speed profile). GHGenius (2013)[94] is developed for Canada, which doesn't have the same emission regulations on MHDVs as U.S. For transit buses, I thus use a simple average of leakage rates from two vehicle tests with three drive cycles as the methane leakage rate for CNG and LNG transit buses in this study, which is 0.06%. The remaining two studies[44,92] are used to estimate methane emission factors for other MHDVs. These two studies' methane emission factors differ from each other by a factor of four. Without any other data available, I use the average emission factor, which is 0.25%. By comparison, GREET model[159] assumes a 0.21% leakage rate for CNG passenger vehicles.

I assume that CNG and LNG vehicles (with sparking ignition technologies) have the same emission factors since the only differences between these vehicles are in the fuel supply system. Boil-off effects of LNG could be a driver for higher methane leakage rates from the vehicle. Here I assume the fuels are used quickly after refueling rather than waiting for days so boil-off effects should be minimal.[64,92] Existing studies do not differentiate between emission factors for CNG and LNG vehicles either.

Ideally, methane emission factors should be specific in terms of vehicles types, drive cycles, or even ambient environment conditions. Given the data availability, I have to assume that all CNG and LNG MHDVs (except transit buses) have the same methane emission factors. I thus call for more emission tests to be performed for modern CNG and LNG MHDVs to improve our understandings of methane leakage directly from the vehicle. Finally, I summarize the assumptions on tailpipe methane emissions in **Table A.33**.

Table A.31. Tailpipe methane leakage rates of transit buses, calculated from recent studies.

Study	Fuel (Model Year)	Methane Emission Factor (Unit: g/mile)	Fuel Economy (Unit: MPGDE)	Methane Leakage Rate (Unit: % of vehicle fuel supply)
MJB&A (2012) [48]	Diesel (2012)	0.002	3.27	n/a
	CNG (2012)	1.080	3.00	0.13%
MJB&A (2013) [38]	CNG (2011) (Vehicle #1)	0.61 (Manhattan Cycle) 0.37 (Orange County Cycle) 0.20 (UDDS cycle)	2.77 4.09 5.51	0.07% 0.06% 0.04%
	CNG (2011) (Vehicle #2)	0.48 (Manhattan Cycle) 0.27 (Orange County Cycle) 0.38 (UDDS cycle)	2.82 4.17 5.44	0.05% 0.05% 0.08%
GHGenius v4.03 (2013)[94]	ULSD	0.08 (g/km)	5.93	n/a
	CNG	1.27 (g/km)	5.82	0.48%
	LNG	1.25 (g/km)	5.82	0.46%

Table A.32. Tailpipe methane leakage rates of heavy-duty trucks, calculated from recent studies.

Study	Fuel (Model Year)	Methane Emission Factor (Unit: g/mile)	Fuel Economy (Unit: MPGDE)	Methane Leakage Rate (Unit: % of vehicle fuel supply)
Santini et al. (2013)[92]	ULSD	0.013	6.5	n/a
	LNG (Westport HPDI)	0.451	6.24	0.11%
	CNG/LNG (SI)	0.546	5.62	0.12%
Meier et al. (2013)[44]	LNG (Westport HPDI)	1.6	6.0	0.38%*
GHGenius v4.03 (2013)[94]	ULSD	0.07 (g/km)	5.93	n/a
	CNG	1.27 (g/km)	5.82	0.48%
	LNG	1.25 (g/km)	5.82	0.46%

** The calculated methane leakage rate is slightly higher than those calculated by Meier et al. (2013), which is 0.34%.*

Table A.33. Tailpipe methane emissions for HDV (Class 2-8) pathways (Unit: gr/km).

Fuel	Diesel	Gasoline	BEV / H_2 FCEV	CNG/LNG		Other fuel pathways
				Transit Bus	Other MHDVs	
CH_4	0.005	0.03	0	0.06%	0.25%	Same as diesel
N_2O	0.005	0.03	0	Same as diesel		

A.7. Break-Even Methane Leakage Rate Analysis

My results, as well as those in the literature,[28,44] suggest that methane leakage rate and relative fuel economy of natural gas vehicles are the most important factors influencing whether natural gas fuel pathways achieve net emission reductions. Given the importance of these variables, I derive the relationship between break-even life cycle methane leakage rate and relative fuel economy of natural gas vehicles.

Here I restrict the analysis for CNG and distributed LNG pathways because they are currently the focus of much interest. Eqn. A.5-A.10 describe the estimation model (note that I ignored tailpipe methane and N_2O emissions for the first-order approximation). In Eqn. A.5, life cycle GHG emissions from incumbent petroleum pathways are calculated using life cycle GHG emissions of gasoline or diesel and the corresponding vehicle fuel efficiency (or equivalently, fuel economy). In Eqn. A.6, life cycle GHG emissions of the CNG or distributed LNG pathway is calculated as a sum of two parts, life cycle CO_2 emissions and life cycle methane emissions, which are then expressed as a function of life cycle methane leakage rate, and relative fuel efficiency of natural gas vehicles compared to gasoline/diesel vehicles.

$$Petroleum\ life\text{-}cycle\ emissions = \frac{Petroleum\ life\text{-}cycle\ GHG\ emission\left[{}^{gCO_2e}\!/\!_{MJ}\right]}{Petroleum\ vehicle\ energy\ efficiency\left[{}^{mile}\!/\!_{MJ}\right]} \tag{A.5}$$

$$\begin{aligned} &CNG\ vehicle\ life\text{-}cycle\ emissions \\ &= \frac{Natural\ gas\ pathway\ life\text{-}cycle\ CO_2\ emission\left[{}^{gCO_2e}\!/\!_{MJ}\right]}{Natural\ gas\ vehicle\ energy\ efficiency\left[{}^{km}\!/\!_{MJ}\right]} + \frac{methane\ leakage\ rate \times methane\ density\left[{}^{gram}\!/\!_{cf}\right] \times GWP_{methane}}{Natural\ gas\ vehicle\ energy\ efficiency\left[{}^{km}\!/\!_{MJ}\right] \times energy\ \text{intensity}_{Natural\ gas}\left[{}^{MJ}\!/\!_{cf}\right]} \end{aligned} \tag{A.6}$$

To calculate break-even methane leakage rate, I equaled Eqn. A.5 and Eqn. A.6, arranged terms, and reached the following expression of break-even methane leakage rate:

$$Break\text{-}even\ methane\ leakage\ rate = \left(\alpha_{Fuel\ economy} \times Petroleum\ life\text{-}cycle\ GHG\ emission - NGV\ life\text{-}cycle\ CO_2\ emission\right) \times \frac{\beta_{methane}}{GWP_{methane}} \tag{A.7}$$

Where

$$\alpha_{Fuel\ economy} = \frac{Natural\ gas\ vehicle\ energy\ efficiency\left[{km}/{MJ}\right]}{Petroleum\ vehicle\ energy\ efficiency\left[{km}/{MJ}\right]} = \frac{Natural\ gas\ vehicle\ fuel\ economy\left[MPG\right]}{Petroleum\ vehicle\ fuel\ economy\left[MPG\right]} \quad (A.8)$$

is the relative fuel economy of the CNG vehicle relative to baseline petroleum vehicle, and is a constant related to natural gas:

$$\beta = \frac{Energy\ \text{intensity}_{Natural\ gas}\left[{MJ}/{cf}\right]}{Methane\ density\left[{gram}/{cf}\right]} \quad (A.9)$$

In the derivations above, I made two simplifications to get a linear relationship. In Eqn. A.6, I assume the life cycle carbon dioxide emissions do not change with methane leakage rate by ignoring the impacts of methane leakage rate on carbon dioxide emissions from natural gas upstream developments. In Eqn. A.5 and Eqn. A.6, I ignore tailpipe N_2O emissions to simplify derivation. Since tailpipe N_2O emissions from petroleum pathways and CNG pathways are similar (**Table A.33**), this approximation barely has any effects on the analysis presented here. I calculate average life cycle GHG emissions of the petroleum pathways, and average life cycle CO_2 emissions of the CNG pathway through the Monte-Carlo simulation, and substitute these estimates into Eqn. A.7. The baseline petroleum pathway is conventional diesel (91.5 gCO_2/MJ_{LHV}). **Figure 2.4** in the main text shows the resulting relationships.

Assuming a CNG vehicle has 5% fuel efficiency penalty compared to petroleum vehicles, the break-even methane leakage rate for the CNG pathway is 1.0% for 20-year GWP, and 2.4% for 100-year GWP (**Table A.34**). As methane has a much shorter lifetime compared to CO_2,[2] a longer time frame allows for a much higher leakage rate than the short-term time frame. The relationship between break-even methane leakage rate and relative fuel economy of CNG vehicle is linear: all else being equal, a lower relative fuel economy of CNG vehicle is likely to cause the natural gas pathways to emit more than the baseline petroleum pathway; on the other hand, an efficiency-improving vehicle technology, such as hybridization[34,76] and electrification, achieves emission reductions at even larger leakage rates.

Table A.34. Break-even methane leakage rate for natural gas pathways (the baseline fuel pathway is conventional gasoline or conventional diesel).

Relative fuel economy	CNG		LNG	
	100-year GWP	20-year GWP	100-year GWP	20-year GWP
90%	1.7%	0.7%	N/A	N/A
95%	2.4%	1.0%	0.6%	0.2%
100%	3.1%	1.3%	1.2%	0.5%

A.8. Additional Results

A.8.1 Formula to Calculate Life Cycle GHG Emissions

Similar to Eqn. A.5 and Eqn. A.6 in the break-even analysis, the life cycle GHG emissions of any vehicle pathway (with functional unit of gram/km) can be calculated using Eqn. A.10.

$$\begin{aligned}
&\text{Life cycle GHG emission}\,(fuel_i, class_k, GHG_j) = \\
&\quad \frac{\text{Well-to-pump emission of } fuel_i\,(GHG_j)}{\text{Vehicle fuel economy}(fuel_i, class_k) \big/ \text{Energy intensity of baseline fuel}} + \\
&\quad \text{Tailpipe emission}\,(fuel_i, class_k, GHG_j) + \\
&\quad \text{Battery / Fuel cell manufacturing emission}\,(fuel_i, class_k, GHG_j)
\end{aligned} \tag{A.10}$$

Here, *fuel*$_i$ represents different natural gas pathway, *class*$_k$ represents different classes of MHDVs, *GHG*$_j$ represents three GHGs (CO_2, CH_4, and N_2O). *Energy intensity of baseline fuel* refers to the energy intensity of conventional gasoline (for Class 2b) or conventional diesel since vehicle fuel economy is represented in gasoline equivalent (for Class 2b) or diesel equivalent (for Class 3-8). *Well-to-pump emission of fuel*$_i$ refers to all emissions attributed to fuel i before the fuel is dispensed into the vehicle. It is also called carbon intensity in some policy contexts (such as the Low Carbon Fuel Standard in California). *Tailpipe emissions* only include methane and N_2O emissions. *Battery and fuel cell manufacturing emissions* are only applicable to certain pathways (HEV, BEV, and FCEVs). I finally apply unit conversions are need to convert the results from per-mile to per-km.

I use the GWP metric to convert emissions of different GHGs into CO_2-equivalent metric (gCO_2-eq/km).

$$Life\ cycle\ GHG\ emission\ (fuel_i, class_k, CO_2\text{-}eq) = \sum_{j=CO_2, CH_4, N_2O} Life\ cycle\ GHG\ emission\ (fuel_i, class_k, CO_2\text{-}eq) \times GWP(GHG_j) \quad \text{(A.11)}$$

I calculate life cycle GHG emissions with the other functional unit (gCO_2-eq/km-metric-ton) by dividing the previous result by vehicle payload.

$$Life\ cycle\ GHG\ emission\ (fuel_i, class_k, CO_2\text{-}eq) = \frac{Life\ cycle\ GHG\ emission\ (fuel_i, CO_2\text{-}eq)}{Vehicle\ \ payload\ (fuel_i, class_k)}$$

(A.12)

The Monte-Carlo simulation model is implemented in MATLAB (Version R2012b). The sample size of Monte-Carlo simulation model is one million.

A.8.2 Scenario Analysis

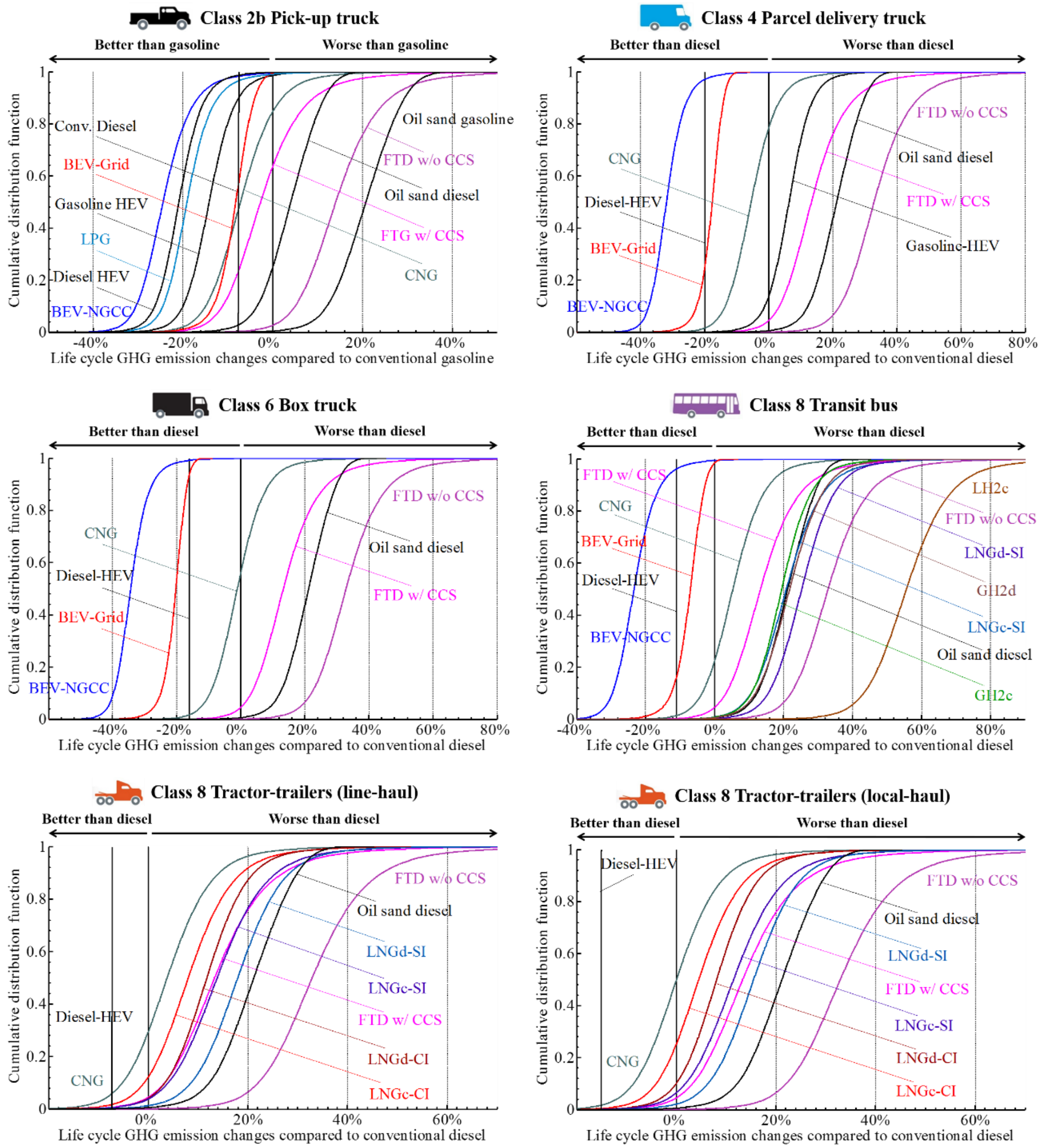
I use scenarios to compare the effects of different methane leakage rate assumptions, different choices of GWP metrics, and different functional units. Specifically, I consider the following scenarios with both functional units (gCO2-eq/km, and gCO2-eq/km-metric ton).

(1) Baseline natural gas upstream estimates, 100-year GWP with NGCC electricity;
(2) Baseline natural gas upstream estimates, 20-year GWP with NGCC electricity;
(3) Pessimistic natural gas upstream estimates, 100-year GWP with NGCC electricity;
(4) Pessimistic natural gas upstream estimates, 20-year GWP with NGCC electricity;
(5) Baseline natural gas upstream estimates, 100-year GWP with grid-average electricity.

In the main text, I discuss how GHG emissions of the vehicle life cycle change if I use a different time horizon for GWPs or if I don’t consider payloads (and thus payload differences) in various pathways for MHDVs. Here, I provide additional results, cumulative distribution function (CDF) in **Figure A.3**-**Figure A.6**, and bar plots in **Figure A.7-Figure A.8**. There are trade-offs in these two types of data visualization: bar plots are clear but they fail to represent the shape of the distribution; CDF plots capture the key information from the shape of the distribution but they are less intuitive. The distribution of life cycle emissions from natural gas pathways are found to be wider than those from petroleum pathways and exhibit a highly asymmetrical shape skewed to the right. Thus when I calculate relative emission changes compared to petroleum fuels, the resulting distributions are also skewed to the right (**Figure A.3**-**Figure A.6**).

Using 20-year GWPs instead of 100-year GWPs increases life cycle GHG emissions by 7-21% (when baseline methane emissions are used) for natural gas fuel pathways where pathways with higher methane emissions, such as CNG, LNG and F-T liquids, face larger emission increases. Comparatively, using pessimistic methane emission assumptions (which increase methane emissions from natural gas systems by 50%) only increase life cycle GHG emission by 5-7% across natural gas pathways. However, if 20-year GWPs and pessimistic methane emission assumptions are jointly considered, life cycle GHG emissions increase between 13-33% across natural gas pathways.

I find that all alternative fuel pathways carry some payload penalties for all MHDVs (**Table A.26**) and the issue of payload penalty is more severe for pick-up trucks and transit buses than for any other MHDVs. For pick-up trucks, their relative light payloads make them very sensitive to any changes in vehicle payloads. For transit buses, alternative fuel buses see a large drops in the maximum number of bus riders (determined from real vehicle tests) compared to diesel buses. These large reductions in payloads from alternative fuel buses cancel out most of their emission reduction potentials (with a functional unit that doesn’t consider payloads), resulting in small, if any, emissions reductions for the payload-normalized scale (with a functional unit that considers payloads).



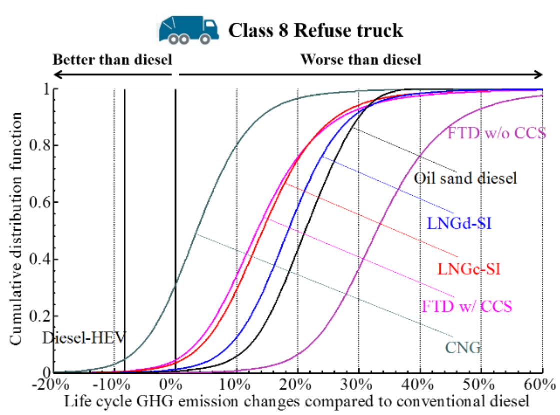
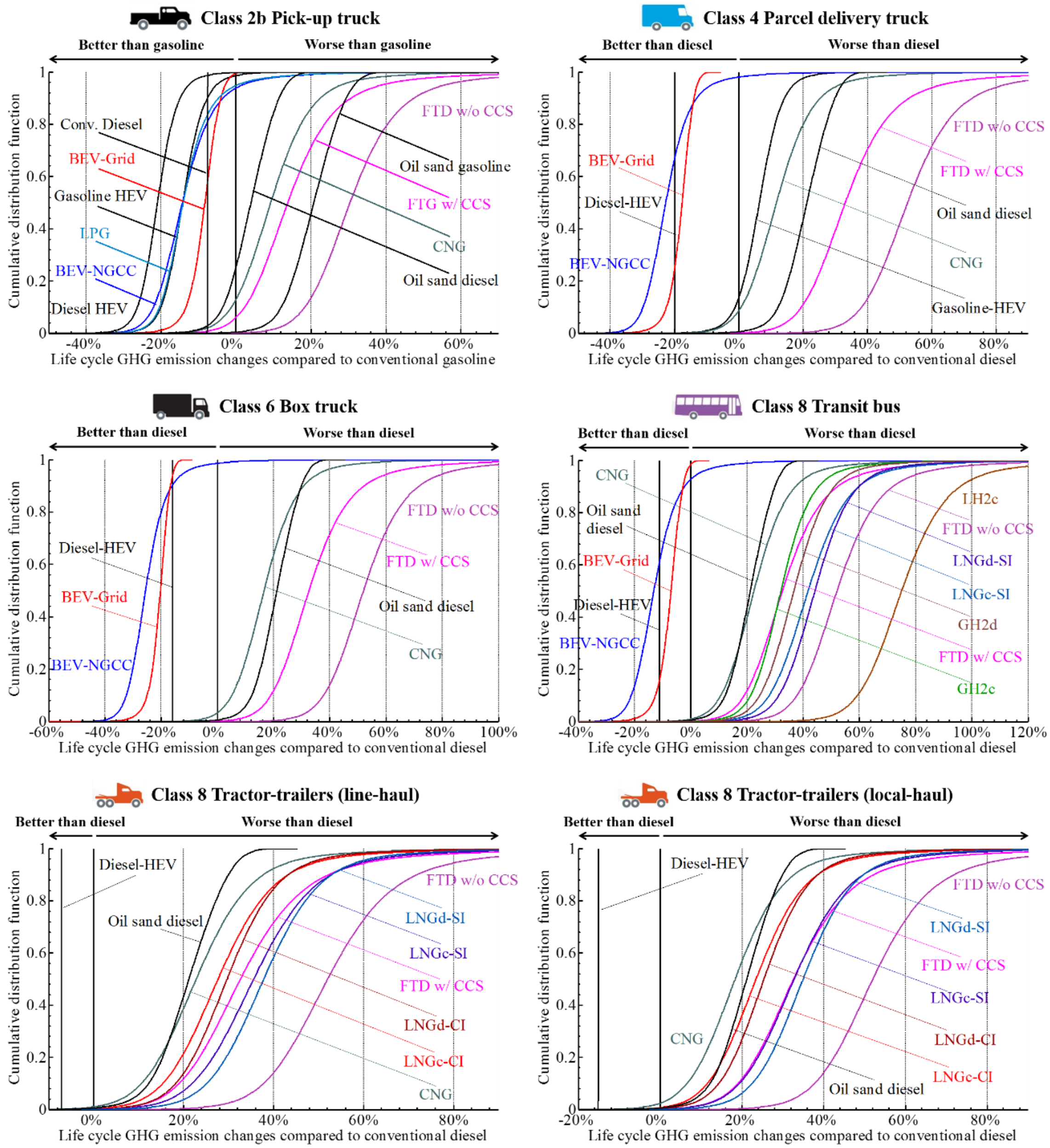


Figure A.3. Cumulative probability distributions (CDF) of life cycle GHG emission changes compared to the baseline petroleum pathways (gasoline for Class 2b, and diesel for all remaining classes) for MHDVs (100-year GWPs) (Vehicle payloads are factored in).



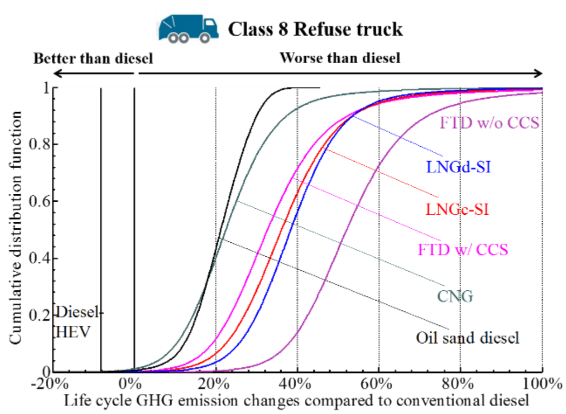
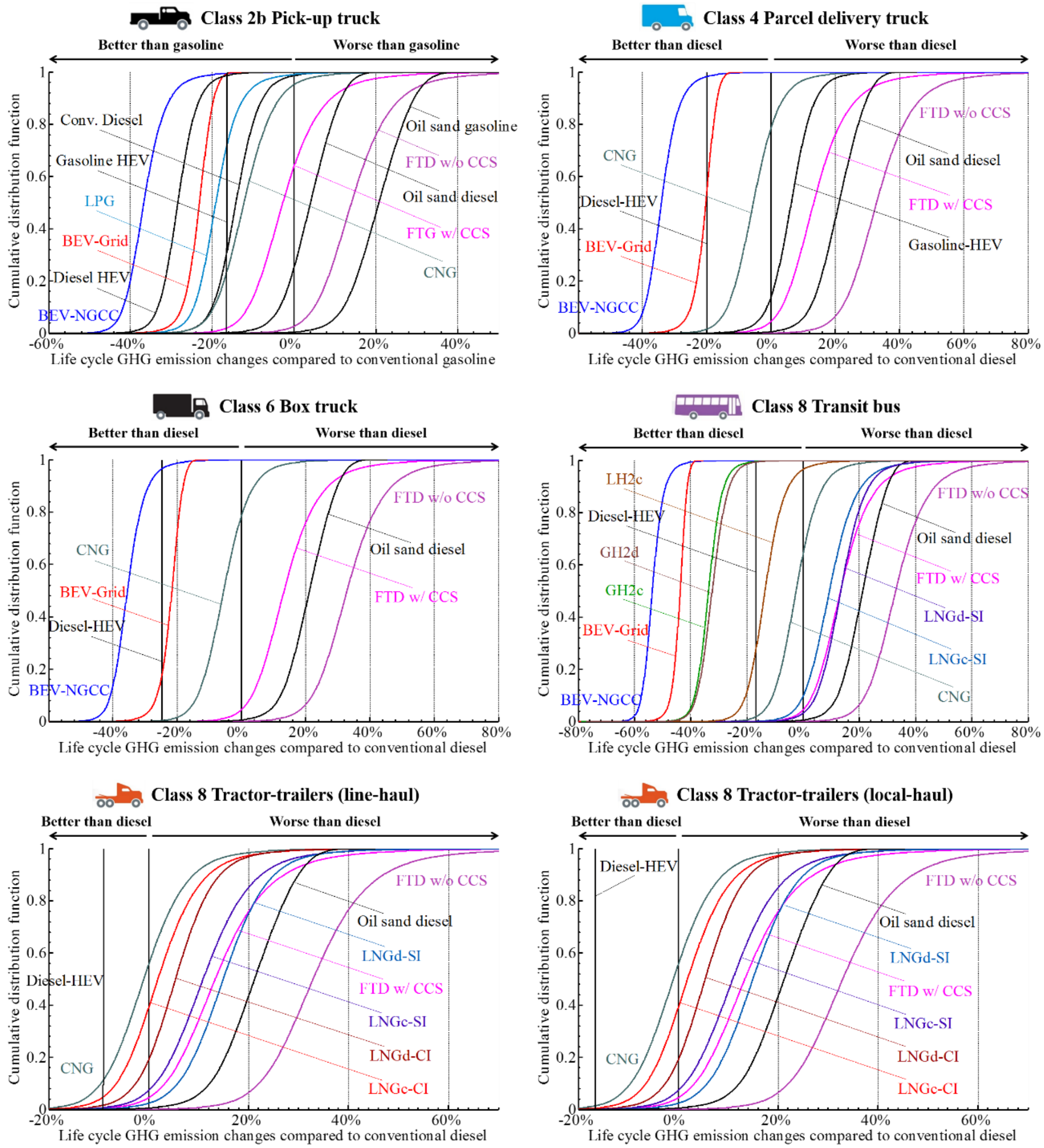


Figure A.4. Cumulative probability distributions (CDF) of life cycle GHG emission changes compared to the baseline petroleum pathways (gasoline for Class 2b, and diesel for all remaining classes) for MHDVs (20-year GWPs) (Vehicle payloads are factored in).



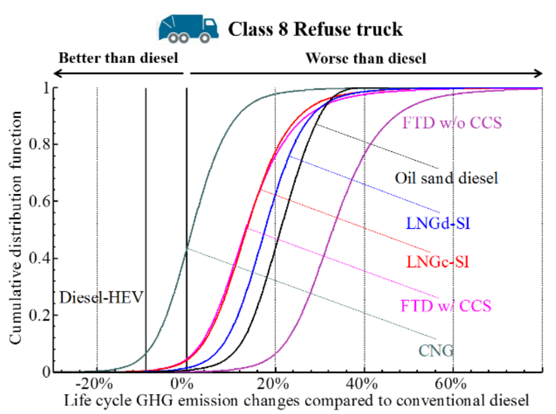
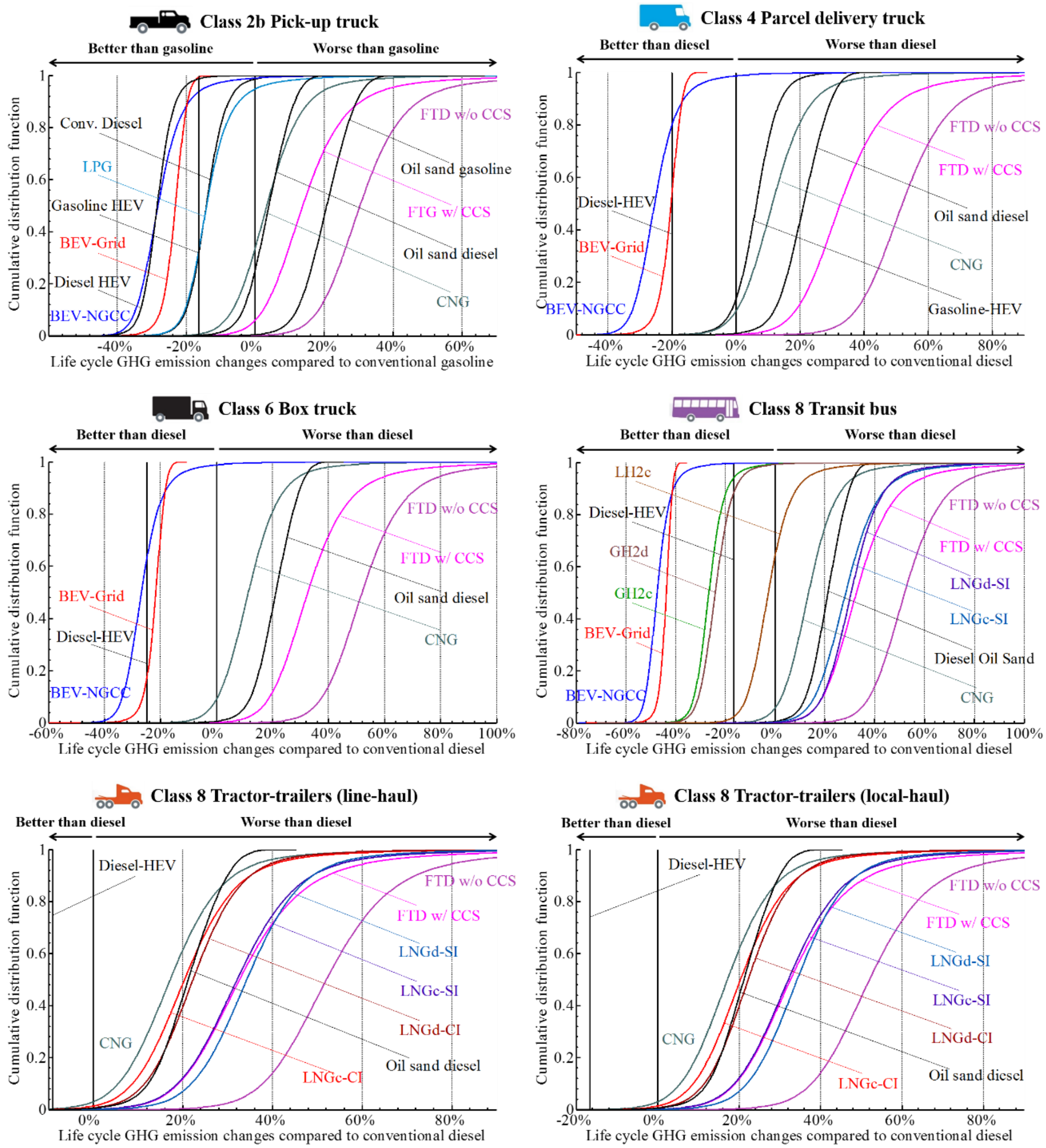


Figure A.5. Cumulative probability distributions (CDF) of life cycle GHG emission changes compared to the baseline petroleum pathways (gasoline for Class 2b, and diesel for all remaining classes) for MHDVs (100-year GWPs) (Vehicle payloads are not factored in).



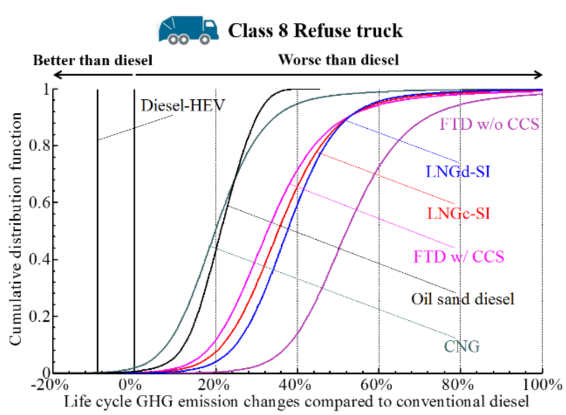
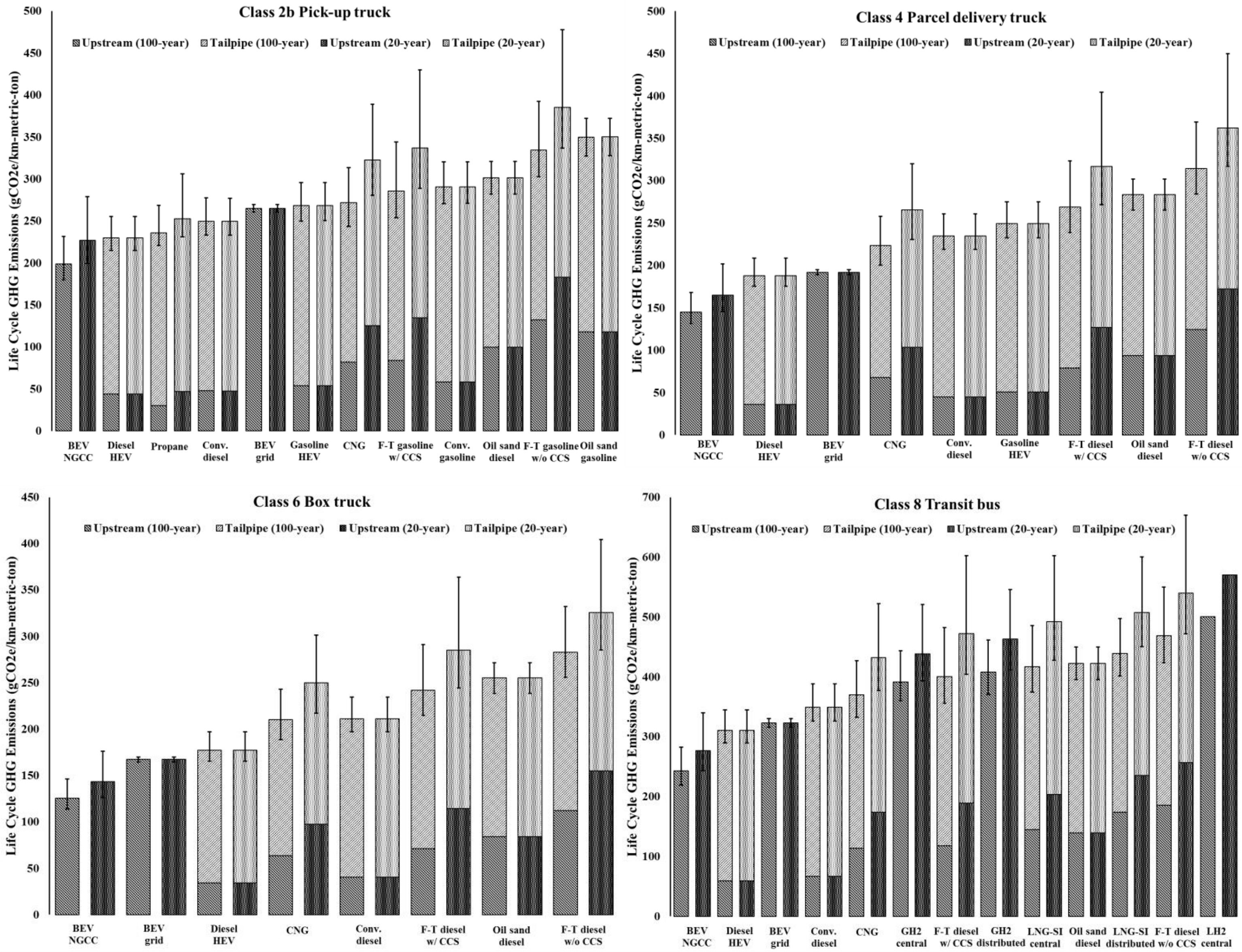
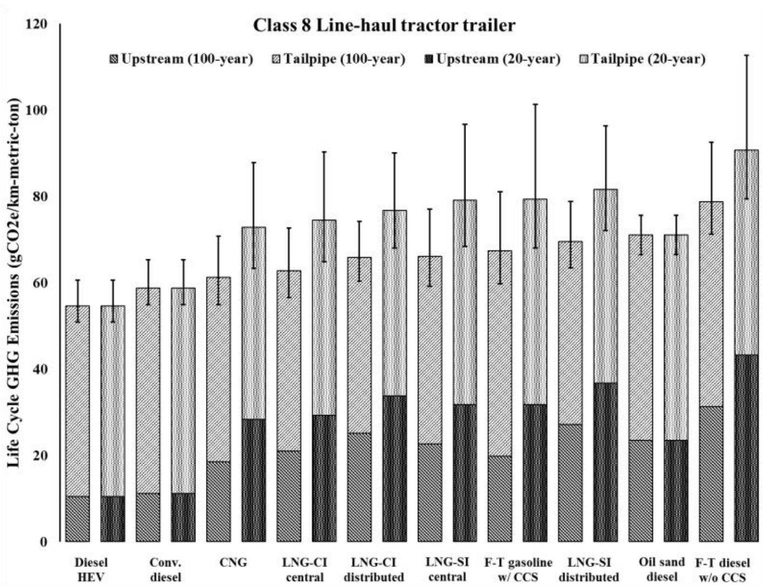
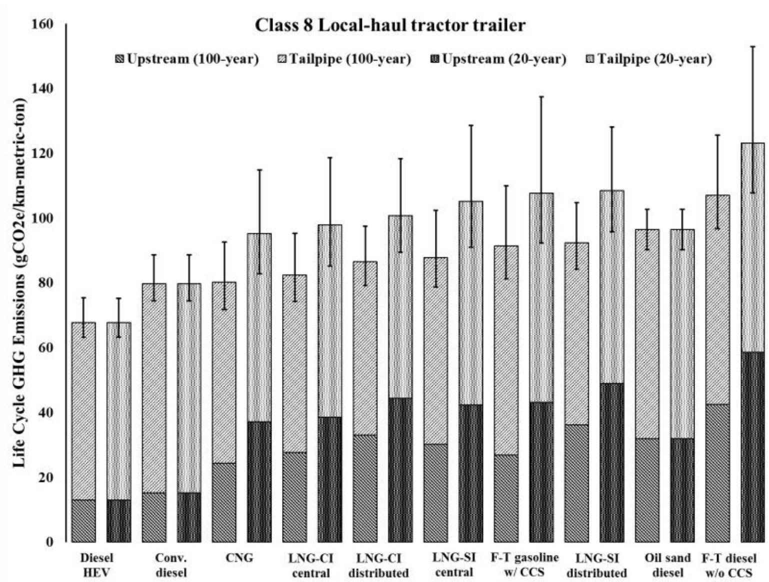


Figure A.6. Cumulative probability distributions (CDF) of life cycle GHG emission changes compared to the baseline petroleum pathways (gasoline for Class 2b, and diesel for all remaining classes) for MHDVs (20-year GWPs) (Vehicle payloads are not factored in).


Class 2b Pick-up truck
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km-metric-ton)
0
50
100
150
200
250
300
350
400
450
500
BEV NGCC
Diesel HEV
Propane
Conv. diesel
BEV grid
Gasoline HEV
CNG
F-T gasoline w/ CCS
Conv. gasoline
Oil sand diesel
F-T gasoline w/o CCS
Oil sand gasoline
Class 4 Parcel delivery truck
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km-metric-ton)
0
50
100
150
200
250
300
350
400
450
500
BEV NGCC
Diesel HEV
BEV grid
CNG
Conv. diesel
Gasoline HEV
F-T diesel w/ CCS
Oil sand diesel
F-T diesel w/o CCS
Class 6 Box truck
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km-metric-ton)
0
50
100
150
200
250
300
350
400
450
BEV NGCC
BEV grid
Diesel HEV
CNG
Conv. diesel
F-T diesel w/ CCS
Oil sand diesel
F-T diesel w/o CCS
Class 8 Transit bus
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km-metric-ton)
0
100
200
300
400
500
600
700
BEV NGCC
Diesel HEV
BEV grid
Conv. diesel
CNG
GH2 central
F-T diesel w/ CCS
GH2 distributed
LNG-SI central
Oil sand diesel
LNG-SI distributed
F-T diesel w/o CCS
LH2 central





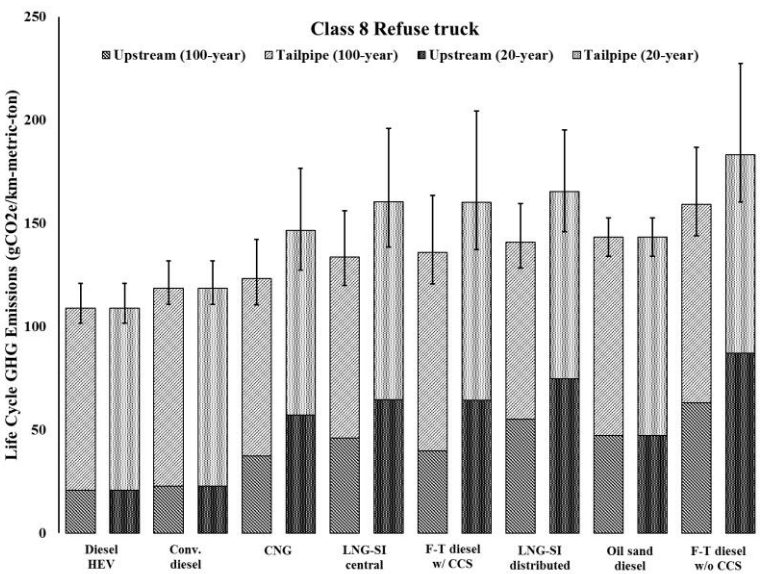
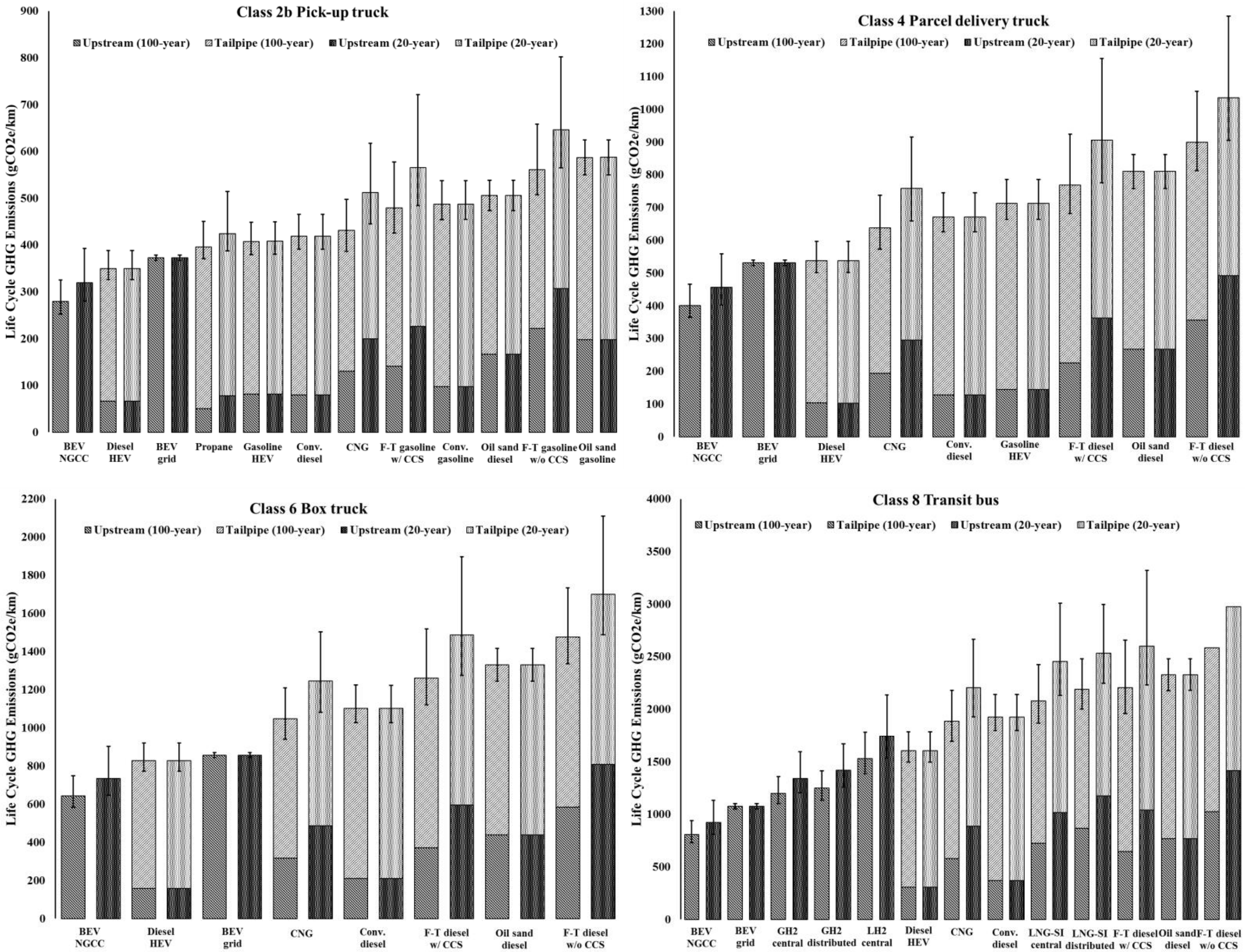
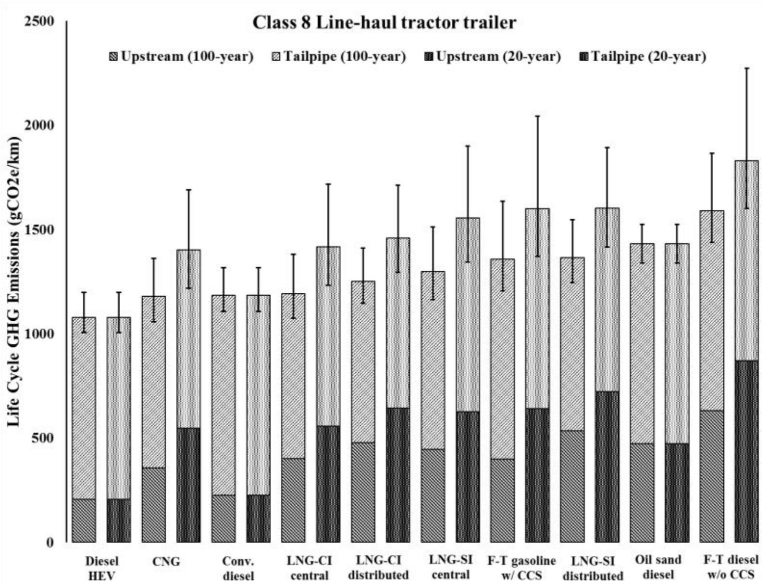
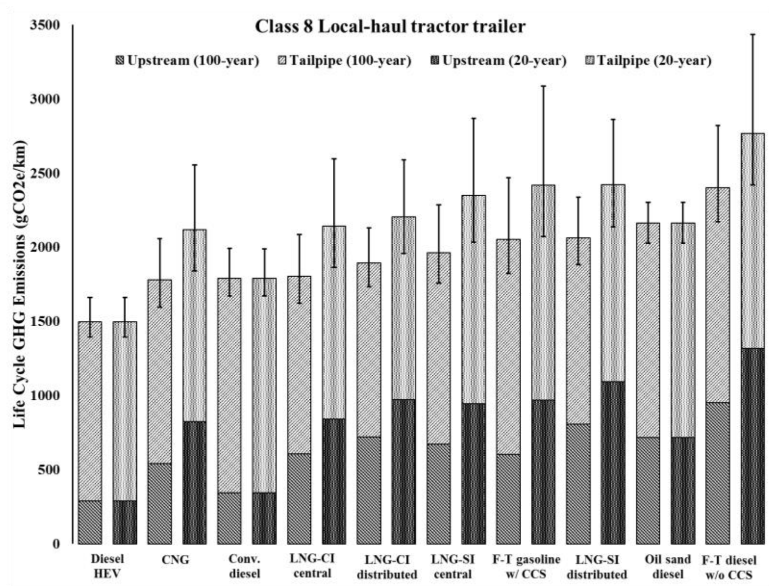


Figure A.7. Life Cycle GHG emissions of MHDVs (unit: gCO2-eq/km-metric-ton), with baseline methane emission estimate. In each figure, results with 100-year GWP and 20-year GWP are shown side by side. Error bars are based on the 95% confidence interval of life cycle GHG emissions.


Class 2b Pick-up truck
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km)
0
100
200
300
400
500
600
700
800
900
BEV NGCC
Diesel HEV
BEV grid
Propane
Gasoline HEV
Conv. diesel
CNG
F-T gasoline w/ CCS
Conv. gasoline
Oil sand diesel
F-T gasoline w/o CCS
Oil sand gasoline
Class 4 Parcel delivery truck
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km)
0
100
200
300
400
500
600
700
800
900
1000
1100
1200
1300
BEV NGCC
BEV grid
Diesel HEV
CNG
Conv. diesel
Gasoline HEV
F-T diesel w/ CCS
Oil sand diesel
F-T diesel w/o CCS
Class 6 Box truck
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km)
0
200
400
600
800
1000
1200
1400
1600
1800
2000
2200
BEV NGCC
Diesel HEV
BEV grid
CNG
Conv. diesel
F-T diesel w/ CCS
Oil sand diesel
F-T diesel w/o CCS
Class 8 Transit bus
Upstream (100-year)
Tailpipe (100-year)
Upstream (20-year)
Tailpipe (20-year)
Life Cycle GHG Emissions (gCO2e/km)
0
500
1000
1500
2000
2500
3000
3500
4000
BEV NGCC
BEV grid
GH2 central
GH2 distributed
LH2 central
Diesel HEV
CNG
Conv. diesel
LNG-SI central
LNG-SI distributed
F-T diesel w/ CCS
Oil sand diesel
F-T diesel w/o CCS





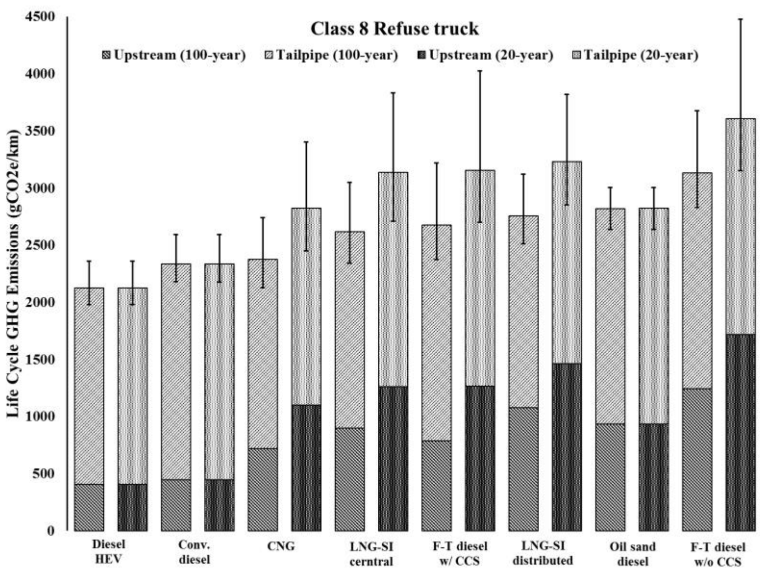


Figure A.8. Life Cycle GHG emissions of MHDVs (unit: gCO2-eq/km), with baseline methane emission estimate. In each figure, results with 100-year GWP and 20-year GWP are shown side by side. Error bars are based on the 95% confidence interval of life cycle GHG emissions.

A.8.3 Sensitivity Analysis

To test the sensitivity of assumptions related to vehicle on-board battery use and vehicle tailpipe methane emissions, I calculated the shares of battery manufacturing emissions in the life cycle GHG emissions from BEVs and the shares of tailpipe methane emissions in the life cycle GHG emissions from CNG pathways. I found that both emission sources are of small shares (1-4%) of the life cycle GHG emissions (**Table A.35** and **Table A.36**). In general, they play much smaller roles in life cycle GHG emissions compared to other factors discussed, such as choice of vehicle fuel pathway, vehicle fuel economy, and methane leakage rate.

Table A.35. Percentages of battery manufacturing emissions in life cycle GHG emissions from BEV pathways (with natural gas electricity). Shown in the table are mean values and 95% confidence intervals (in parenthesis).

MHDVs	Baseline methane estimate		Pessimistic methane estimate	
	100-year GWP	20-year GWP	100-year GWP	20-year GWP
Class 2	1.8% (1.5-1.9%)	1.5% (1.2-1.7%)	1.7% (1.4-1.9%)	1.4% (1.1-1.6%)
Class 4	4.6% (3.9-5.0%)	4.0% (3.3-4.5%)	4.4% (3.7-4.9%)	3.7% (2.8-4.2%)
Class 6	2.3% (2.0-2.5%)	2.0% (1.6-2.3%)	2.2% (1.8-2.4%)	1.8% (1.4-2.1%)
Transit bus	2.9% (1.7-4.2%)	2.5% (1.5-3.7%)	2.8% (1.6-4.0%)	2.3% (1.3-3.5%)

Table A.36. Percentages of tailpipe methane emissions in life cycle GHG emissions from CNG pathways. Shown in the table are mean values and 95% confidence intervals (in parenthesis).

MHDVs	Baseline methane estimate		Pessimistic methane estimate	
	100-year GWP	20-year GWP	100-year GWP	20-year GWP
MHDVs (except transit bus)	2.0% (1.1-2.8%)	4.1% (2.8-5.2%)	1.9% (1.1-2.6%)	3.7% (2.6-4.6%)
Transit bus	0.5% (0.3-0.7%)	1.0% (0.7-1.3%)	0.5% (0.3-0.6%)	0.9% (0.6-1.2%)

A.9. Data Quality

In this study, I perform a LCA on a comprehensive set of natural gas-derived fuel, engine technologies, and vehicle types to evaluate the relative comparison of different ways of using natural gas for different types of MHDVs. In doing so, I relied on a large number of data sources for different parts of the LCA model, ranging from units and metrics, natural gas upstream stage, production of natural gas-based fuels, fuel transport, to vehicle assumptions. In this Appendix, I have discussed the details behind these different parts of the model individually.

In this section I provide the corresponding discussions on data quality for each of these parts. Specifically I discuss the type of source and how it was used. In terms of data sources, I divide all data sources into the following categories: peer-reviewed journal papers (including peer-reviewed conference proceedings), thesis, conference presentations, academic working papers, government sources (including those authored or contracted by national laboratories), vehicle manufacturer specifications, and industry consulting reports. For several inputs, no peer-reviewed data sources exist, in which case I rely on alternative data sources. In addition to data sources, I also include the information regarding the nature of data for natural gas upstream emissions and vehicle assumptions. For instance, are they based on actual emission measurements or vehicle tests? Direct data sources, such as actual measurements or tests are used whenever available.

Finally, I use a break-even analysis, scenario analysis, and sensitivity analysis to explore the sensitivity of the results to these various assumptions. I present these results in the main text, and in this Appendix (*Section A7 Break-Even Methane Leakage Rate Analysis* and *Section A.8 Additional Results*). In short, I find that vehicle fuel economy, methane leakage rate from the natural gas system, choice of global warming metrics, and vehicle payloads play important roles in the relative comparisons of natural gas fuel pathways and baseline petroleum pathways. On the other hand, the results are less sensitive to changes in vehicle tailpipe methane emissions, battery and fuel cell manufacturing emissions, and fuel transport assumptions because of their small shares (less than 4%) in the life cycle GHG emissions.

A.9.1 Units and Metrics

I summarize the reviews on data sources (used in *Section A.1 Units and Metrics*) in **Table A.37**. Here we used the authoritative sources wherever possible. We use the fuel properties and combustion emission factors from the GREET model[159] because it is the mostly-widely used LCA model in the transportation sector and some of their periodic updates are peer-reviewed.

Table A.37. Review of data sources used in *Section A.1 Units and Metrics*.

Type	Data Source	Type of source
Global warming potential	IPCC AR5[2]	Peer-reviewed inter-government report.
Fuel properties and combustion emission factors (except for natural gas)	GREET[159]	Government or national laboratory report.
Methane composition in natural gas	U.S. EPA (2014)[152]	Government or national laboratory report.
	Foss (2007)[275]	Non peer-reviewed academic institute source.

A.9.2 Natural Gas

I summarize data sources used to model natural gas upstream and combustion GHG emissions in **Table A.38**. Almost all data sources used here are peer-reviewed journal papers or government sources (U.S. EPA). In building the LCA model, I use recent on-site methane measurements,[97–99] which are likely to be more representative of the changing industry practices. I compare the assumptions and emission estimates for major fugitive methane sources such as well completion and liquid unloading with other estimates and discussed the differences. As I have emphasized in the main text and previous sections in this Appendix, the largest uncertainty in natural gas upstream emissions is the gap on methane leakage rates between top-down measurements and bottom-up measurements.[114,116] Since no evidence has emerged about which estimates are more appropriate, I rely on scenario analysis to explore their impacts.

Table A.38. Review of data sources used in *Section A.2 Natural Gas Upstream GHG Emissions*.

Type	Data Source	Type of source
Natural Gas Preproduction Stage		

Preproduction emissions (excluding those from well completion)	Jiang et al.'s (2011)[139]	Peer-reviewed journal paper (life cycle assessment).
Well completion emissions (data used)	Allen et al. (2013)[97]	Peer-reviewed journal paper (on-site measurements).
Well completion emissions (existing studies to compare results with)	Jiang et al. (2011)[139], Skone et al. (2011)[279], Hultman et al. (2011)[280], Stephenson et al. (2011)[150], Burnham et al. (2011)[140], Howarth et al. (2011)[141], Weber et al. (2012)[115]	Peer-reviewed journal papers.
	U.S. EPA GHG Inventory 2013[277] and 2014[152]	Government or national laboratory reports.
Estimated ultimate recovery (EUR)	Weber et al. (2012)[115]	Peer-reviewed journal paper.
Natural Gas Production, Processing, Transmission, and Distribution Stage		
Natural gas flow data (volume of natural gas produced, processed, and transported)	U.S. EIA[10,153,282,286]	Government or national laboratory report.
Electricity used for natural gas pipelines	Davis et al. (2013)[3]	Government or national laboratory report.
Natural Gas Production Stage		
Lease fuel use, and flaring emissions	U.S. EIA[10]	Government or national laboratory report.
Liquid unloading	Allen et al. (2013)[97]	Peer-reviewed journal paper (on-site measurements).
	API and ANGA survey[99]	Non peer-reviewed industry report (on-site measurements).
Other CO_2 and CH_4 emissions	U.S. EPA GHG Inventory 2013[277] and 2014[152]	Government or national laboratory report.
Natural Gas Processing Stage		
Emissions reported by processing plants	U.S. EPA Greenhouse Gas Reporting Program[i] (GHGRP)[98]	Government or national laboratory report.
Natural Gas Transmission, Storage, and Distribution Stage		
Fugitive CO2 and CH4 emissions	U.S. EPA GHG Inventory 2014[152]	Government or national laboratory report.
Natural Gas Combustion		
Combustion GHG emissions	Venkatesh et al. (2011)[27]	Peer-reviewed journal paper.

A.9.3 Gasoline and Diesel

I relied on peer-reviewed journal papers for upstream and combustion emissions of conventional and oil-sand derived gasoline and diesel, as summarized in **Table A.39**.

[i] U.S. EPA mandates "facilities that emit 25,000 metric tons or more per year of GHGS" to report their GHG data annually to the GHGRP since 2010[98].

Table A.39. Review of data sources used in *Section A.3 Life Cycle GHG Emissions of Gasoline and Diesel*.

Type	Data Source	Type of source
Upstream emissions for conventional gasoline and diesel	Venkatesh et al. (2011)[169]	Peer-reviewed journal paper.
Combustion emissions for gasoline and diesel	ref. [171,290,292]	Two peer-reviewed journal papers, and one government or national laboratory report.
Oil sand-derived gasoline/diesel	Englander et al.[171]	Peer-reviewed journal paper.

A.9.4 Fuel Production

I relied on peer-reviewed journal papers and reports from the national laboratories to model fuel production profiles (**Table A.40**). I avoid using single data source for validation issues as well as for constructing distributions.

Table A.40. Review of data sources used in *Section A.4 GHG Emissions from the Production of Natural Gas-Based Fuels*.

Type	Data Source	Type of source
NGCC electricity		
Energy efficiency	NETL (2013)[158]	Government or national laboratory report.
Existing studies to compare the life cycle GHG emissions	ref. [280,293]	Peer reviewed journal paper.
U.S. grid average electricity	Cai et al. (2013)[157]	Government or national laboratory report.
Compressed Natural Gas (CNG)		
Energy efficiency of electric compressors	Venkatesh et al. (2011)[27]	Peer reviewed journal paper.
Liquefied Natural Gas (LNG)		
Energy efficiency of liquefaction process	ref. [19,159,294]	Two peer-reviewed journal papers, and one government or national laboratory report.
Natural Gas-Based Hydrogen (H2)		
Central hydrogen plant profile (w/o CCS)	ref. [111,159,160]	Government or national laboratory reports.
Central hydrogen plant profile (w/ CCS)	H2A 3.0[160]	Government or national laboratory report.
Distributed hydrogen plant profile (w/o CCS)	ref. [159,160]	Government or national laboratory reports.
Boil-Off Effects of LNG and Liquid H2	GREET 2013[159]	Government or national laboratory report.
Fischer-Tropsch (F-T) Liquids		

Centralized F-T liquids production plant	Jaramillo et al. (2008)[161]	Peer-reviewed journal paper.
Propane		
Processing process from propane feedstock	GREET 2013[159]	Government or national laboratory report.

A.9.5 Fuel Transport

I relied on the GREET model[159] for fuel transport assumptions for liquid fuels (**Table A.41**). GREET assumes a national-average transportation profile in terms of transportation modes, energy intensities, and distances. While regional variations exist in terms of transportation profiles, the shares of transportation emissions in life cycle GHG emissions are very small.

Table A.41. Review of data sources used in *Section A.5 GHG Emissions from Fuel Transport*.

Type	Data Source	Type of source
GHG emission factors of fuel transport for F-T liquids, propane, GH_2 (central), LH_2 (central), and LNG (central)	GREET 2013[159]	Government or national laboratory report.

A.9.6 Vehicle

I summarize data sources for vehicle fuel economy assumptions in **Table A.42**. Given the availability of data, I used some industry sources that are not peer-reviewed. It should also be noted that most GHG emissions (except battery and fuel cell manufacturing emissions, as well as tailpipe methane and N_2O emissions) are inversely proportional to vehicle fuel economy. Once better sources of vehicle fuel economy become available, readers should be able to update the life cycle GHG emissions without causing large biases.

Table A.42. Review of data sources for vehicle fuel economy.

Type	Data Source	Type of source
Class 2b pick-up truck		
All pathways unless noted below	TIAX[112]	Industry consulting report.
Baseline pathway (gasoline)	TIAX[112] and the Clean Energy Coalition[303]	One industry consulting report; and one government (city government) contracted report.
Hybrid electric vehicles (HEV)	ref. [74,304,305]	Two non-peer reviewed government sources; and one vehicle manufacturer specification.

Propane/LPG	ref. [306]	Vehicle manufacturer specification.
Class 4 parcel delivery truck		
All pathways unless noted below	TIAX[112]	Industry consulting report.
Diesel HEV	ref. [312]	Government or national laboratory report (vehicle in-use evaluation).
Gasoline HEV	ref. [298,315]	Government or national laboratory reports (vehicle in-use evaluation).
BEV	ref. [87,313]	One peer reviewed journal paper; and one industry consulting report (vehicle in-use evaluation).
Class 6 box truck		
All pathways unless noted below	TIAX[112]	Industry consulting report
BEV	TIAX[112], Smith[316] and NREL[317,318]	Two government or national laboratory reports, one vehicle manufacturer specification, and one industry consulting report. Three of four sources are vehicle in-use evaluations.
Class 8 transit bus		
All pathways unless noted below	TIAX[112]	Industry consulting report
CNG/LNG (SI)	ref. [5,38,42,48,70,112,243,320,330–336]	Ten government or national laboratory reports, and five industry consulting reports Five of these 15 sources are vehicle in-use evaluations.
H_2 FCEV	ref. [177]	Government or national laboratory report (vehicle in-use evaluations).
BEV	ref. [299,337,338]	Government or national laboratory reports (vehicle tests).
Class 8 refuse truck		
All pathways unless noted below	TIAX[112]	Industry consulting report
CNG/LNG	ref. [33,88,89,243,330,342]	One peer-reviewed journal publication, two academic conference presentations, and three government or national laboratory reports (including contracted). Five of these 6 sources are vehicle in-use evaluations.
Hybrid	ref. [84,343]	One peer-reviewed conference publication, and one industry consulting report. One source is vehicle in-use evaluation.
Class 8 tractor-trailer		
All pathways unless noted below	TIAX[112]	Industry consulting report.
CNG/LNG (SI)	ref. [34,64,112,346,347]	One Ph.D. dissertation, one conference paper (vehicle simulation), one academic working paper, and two industry consulting reports.
LNG (CI)	ref. [34,64,82,85,86,112]	One government or national laboratory report, one conference paper (vehicle simulation), one academic working paper, and three industry consulting reports.

		Three of these 6 sources are vehicle in-use evaluation.
Diesel HEV	ref. [74–76,78,79]	Three peer-reviewed journal publications, and two government or national laboratory reports.

I summarize data sources for vehicle payload assumptions in **Table A.43**. As noted in *Section A.6.2 Vehicle Payloads*, I assume that all MHDVs will carry maximal payloads so their payload penalties are equal to the changes in vehicle tare weights. I discussed this assumption in length in the main text. In addition, I present two sets of results where one set considers payload differences while the other does not.

Table A.43. Review of data sources for vehicle payloads.

Type	Data Source	Type of source
Class 2b pick-up truck	ref. [55,305,350]	One government or national laboratory report, and two vehicle manufacturer sources (actual vehicle information).
Class 4 parcel delivery truck	ref. [64,298,310,313,315]	One peer-reviewed journal paper, three government or national laboratory reports (actual vehicle information), and one academic working paper.
Class 6 box truck	ref. [55,64,315]	Two government or national laboratory reports (one of them is actual vehicle information), and one academic working paper.
Class 8 transit bus	ref. [64,299,329,337,351–353]	Six government or national laboratory reports (all are actual vehicle information), and one academic working paper.
Class 8 refuse truck	ref. [64,300,343]	One peer-reviewed journal paper, one academic working paper, and two vehicle manufacturer sources (actual vehicle information).
Class 8 line-haul tractor-trailer	ref. [64,76]	One peer-reviewed journal paper (vehicle simulation), and one academic working paper.
Class 8 local-haul tractor-trailer	ref. [64,76]	One peer-reviewed journal paper (vehicle simulation), and one academic working paper.

I summarize data sources for battery and fuel cell manufacturing emissions in **Table A.44**. Emission factors of battery and fuel cell manufacturing are from peer-reviewed journal papers and the GREET model.[159] Assumptions related to the battery and fuel cell sizes and their lifetime replacements are taken from a number of data sources that are largely non-peer reviewed. There is an issue of limited data because HEVs and BEVs are not widely used in MHDVs. However, sensitivity result shows that battery and fuel cell manufacturing emissions only contribute to less than 4% of life cycle GHG emissions.

Table A.44. Review of data sources for battery and fuel cell manufacturing emissions.

Type	Data Source	Type of source
Emission factors		
Battery manufacturing emissions	ref. [176]	Peer-reviewed journal paper.
Battery specific energy	ref. [176]	Peer-reviewed journal paper.
Fuel cell manufacturing emissions	ref. [159]	Government or national laboratory report.
PHEV/BEV charging energy efficiency	ref. [23,134,159]	Two peer-reviewed journal papers and one government or national laboratory report.
HEV/BEV: lifetime, and battery size		
Class 2b pick-up truck	ref. [112,305]	One vehicle manufacturer specification, and one industry consulting report.
Class 4 parcel delivery truck	ref. [87,112,309,315]	Two government or national laboratory reports, one vehicle manufacturer specification, and one industry consulting report. Three of four sources are vehicle in-use tests.
Class 6 box truck	ref. [112,316,355,356]	One vehicle manufacturer specification, one industry consulting report, and two industry news reports. Three of four sources are actual vehicle information.
Class 8 transit bus	ref. [48,112,177,357,358]	Two government or national laboratory reports, two industry consulting reports, and one vehicle manufacturer specification. Four of five sources are actual vehicle information.
Class 8 local-haul tractor trailer	ref. [34,64,112]	One government or national laboratory report (vehicle simulation), one academic working paper, and one industry consulting report.
Class 8 line-haul tractor trailer	ref. [75,112]	One government or national laboratory report (vehicle simulation), and one industry consulting report.
Fuel cell electric vehicle (FCEV) : lifetime, and battery size		
Class 8 transit bus	ref. [112,177,357]	Two government or national laboratory reports (actual vehicle in-use tests), and one industry consulting report.

I summarize data sources for Tailpipe methane and N_2O emissions in **Table A.45**. I find disagreements on methane leakage rates for CNG and LNG MHDVs (refer to ***Section A.6.4*** *Tailpipe Methane and N2O Emissions* for more information). However, sensitivity result shows that tailpipe methane emissions only contribute to less than 4% of life cycle GHG emissions.

Table A.45. Review of data sources for tailpipe methane and N_2O emissions.

Type	Data Source	Type of source
Gasoline and diesel MHDVs	ref. [362]	Government or national laboratory report.
Methane emission factors for CNG/LNG MHDVs		
Class 4 parcel delivery truck	Chandler et al. (2002)[364]	Government or national laboratory report (actual in-use vehicle test).

Class 8 transit bus	ref. [33,36,38,48,94,327]	Two peer-reviewed journal papers, one master's thesis, and three industry consulting reports. Three of these six sources are actual vehicle tests.
Heavy-duty Truck	ref. [28,36,44,46,92,94,95,327,363]	Three peer-reviewed journal papers, one master's thesis, one academic working paper, two government or national laboratory reports, and two industry consulting reports. Five of these nice sources are actual vehicle tests.

Appendix B. Supporting Information for Chapter 3

B.1. Fuel Properties and Emission Factors from Combustion

Table B.1 summarizes properties (e.g., energy density and mass density) and emissions factors of energy carriers used in this study. There are two reported values for energy contents, LHV (lower heating value) and HHV (higher heating value). MacLean and Lave[72] suggested using LHV for mobile use (such as in vehicles) and HHV for stationary use (such as in power plants and fuel production plants). To be consistent, I use LHV for all energy sources in this study. Furthermore, I rely on the GREET model[159] for all properties of energy carriers to maintain consistency, even though there is a noticeable difference in the energy content (HHV) for dry natural gas used in existing studies: 1,089 BTU/ft^3 in the GREET model;[159] 1,030 BTU/ ft^3 in Jaramillo et al. (2007),[138] Venkatesh et al. (2001),[27] and Jiang et al. (2011).[139]

I calculate the combustion emission factor of natural gas based on Venkatesh et al. (2011).[27] The combustion emission factor of natural gas reported in Venkatesh et al. (2011)[27] follows a normal distribution with a mean of 50 gCO_2-eq/MJ_{HHV} and a standard deviation of 0.7 gCO_2-eq/MJ_{HHV}. To convert it to a LHV-basis, I used the ratio of natural gas HHV (1,089 BTU/cf) and LHV (983 BTU/cf).[159]

Table B.1. Energy content and emissions factors for different energy carriers.

Fuel	**Energy density**	**Mass density**	**Combustion Emissions factor**
Unit	BTU/cubic foot or BTU/gallon	gram/cubic foot or gram/gallon	gCO_2-eq /MJ_{LHV}
Liquid Fuels (at 32F and 1atm)			
Conventional Gasoline	112,194	-	See Chapter 2
Oil sand gasoline		-	
Conventional Diesel	128,450	-	
Oil sand diesel		-	
Methanol	57,250	3,006	68.4[b]
Ethanol	76,330	2,988	70.9[b]
LPG	84,950	1,923	64.5[b]

Ethane	20,295 (Btu/lb)[a]	-	-
Butane	94,970	2,213	-
n-Hexane	105,125	2,479	-
Gaseous Fuels (at 32F and 1atm)			
Natural gas	983	22.0	Normal dist. (50, 0.7)/983*1089[c]
Pure methane	962	20.3	-
Gaseous hydrogen	290	2.6	-

Note: a. The energy density of ethane is used in modeling the ethane steam cracking process. And the source of the value is http://www.engineeringtoolbox.com/heating-values-fuel-gases-d_823.html. b. The combustion emission factor of methanol, ethanol, and LPG are calculated using energy density, mass density, and carbon weight ratio from the GREET model[159]. c. The HHV of natural gas is 1,089 BTU/cubic foot[159].

Generally speaking, as natural gas flows from the well site to end users, its methane composition increases due to various processing and purification processes. **Table B.2** summarizes methane composition of four types of natural gas. I note, however, that the methane composition of natural gas varies by region,[152] so a region-specific analysis may have slightly different results to those presented in this study.

Table B.2. Methane composition of natural gas.

Fuel	Methane composition (volume)	Reference
Natural gas (production)	0.894	U.S. EPA (2014)[152]
Natural gas (pipeline)	0.934	U.S. EPA (2014)[152]
CNG (Compressed Natural Gas)	0.934	Assumed to be the same as pipeline-quality natural gas
LNG (Liquefied Natural Gas)	0.95	Foss (2007)[275]

B.2. Fuel Productions Assumptions

This section additional assumptions on fuel production and transport.

B.2.1 Ethanol Production

While the production of ethanol has already transitioned to biomass-based pathways (such as corn grain, sugarcane, and cellulosic biomass[166]), ethanol was historically produced from fossil fuel-based naphtha and ethane. I consider ethane produced along with natural gas, and I assume that separation of ethane and other natural gas liquids takes place at natural gas processing plants. I perform an energy-based allocation to assign greenhouse gas (GHG) emission associated with natural gas preproduction, production, and processing to ethane.

There is a two-step process to produce ethanol from ethane, and both steps are well studied. The first step is ethane cracking, from which ethylene is produced.[365] I rely on Posen et al.[144] to model the ethane cracking process, as detailed in **Table B.3**. The second step is catalytic ethylene hydration, where a mixture of gaseous steam water and gaseous ethylene react over phosphoric acid catalysts.[366–368] The conversion rate of the process is very low (around 4%) so the unreacted feedstock is recycled until the overall conversion rate is economically favorable (usually more than 95%).[167] I model this ethylene hydration process using data from the Ecoinvent database,[167] as shown in **Table B.4**. According to the Ecoinvent database,[167] the intermediate co-products (butane, and acetaldehyde) are burned as fuel and the final outputs of this process include ethanol and diethyl ether. The Ecoinvent database also suggests a mass-based emission allocation for these co-products (99.2% ethanol and 0.8% diethyl ether). The inputs of the process include ethane as a feedstock, as well as dry natural gas to provide steam and electricity. I assume that the steam boiler has an energy efficiency of 80%.[159] I also calculate carbon dioxide emissions from the process by performing the carbon balance between the inputs and outputs. The ethanol produced is anhydrous ethanol,[167] so I do not account for a dehydration process of ethanol. After production, ethanol is transported to refueling stations. I assume that natural gas-based ethanol and methanol have the same transportation emission factor (per one unit energy transported), as listed in **Table B.5**, because these two pathways are likely to operate with similar infrastructure.

Table B.3. Ethylene steam cracking (ethylene production) profile.[144]

Key parameter	Distribution parameters	Units
Specific Energy Required	Uniform (15,25)	GJ/ metric ton ethylene
Ethylene Produced	Triangular (764, 803, 840)	kg/metric ton ethane
Propylene Produced	Triangular (14.1, 16, 29.9)	
Butadiene Produced	Triangular (17.4, 19.9, 23)	
Aromatics Produced	Uniform (0, 19.9)	
Hydrogen Produced	Triangular (57.9, 60, 89.7)	
Methane Produced	Triangular (58.8, 61, 70.1)	

C4 Components Produced	Triangular (0, 6, 8.1)	
C5 and C6 Components Produced	Uniform (0, 26)	
Product Losses	Uniform (3, 20)	
Methane leakage	Triangular (5.45,6,6.6)	

Table B.4. Ethylene hydration profile for one energy unit (MJ) of ethanol.[167]

Key parameter	Distribution type	Distribution parameters
Molecular conversion rate	Uniform	(0.968, 0.971)
Total energy demand (MJ/kg ethanol produced)	Uniform	(2.85, 2.98)
Steam share	Uniform	(0.88, 0.98)
Electricity share	1 – share of steam as energy input	
Co-product: ethylene	99.2% (by weight)	
Co-product: diethyl ether	0.8% (by weight)	

B.2.2 Transport and Distribution of Liquid Fuels

All the natural gas pathways fall into two groups in terms of where the fuel is produced and where the fuel is pumped into vehicles (i.e. fueling stations): (1) distributed production pathways where alternative fuels are produced at distributed refueling stations (CNG and gaseous hydrogen from distributed productions (GH_2d)), and (2) centralized production pathways where alternative fuels are produced at centralized locations and then transported to refueling stations (E85, M85, Fischer-Tropsch liquids, GH_2 central, LH_2 central, and electricity).

For distributed pathways, emissions related to delivering natural gas to refueling stations are included in the natural gas upstream emissions. For centralized pathways (as well as gasoline pathways), the final fuels are transported either as electricity or liquids. I assumed a 6.5% loss[159] in electricity transmission. For other fuel pathways, I rely on the emission factors reported in the GREET model (version 2013).[159,297] I summarize transportation emission factors in **Table B.5**. The GREET model[159] does not have a natural gas-based ethanol pathway or a Fischer-Tropsch gasoline pathway, so I made two further assumptions: (1) the ethanol pathway has the same transportation emission factor as methanol; (2) Fischer-Tropsch gasoline has the same transportation emission factor as Fischer-Tropsch diesel.

Table B.5. Emission factors of fuel transport[159] (Unit: gram/MJLHV).

Fuel pathways	Fischer-Tropsch gasoline or diesel	GH_2 central	LH_2 central	Methanol/Ethanol
CO_2	1.0	4.8	0.6	1.7
CH_4	0.002	0.014	0.001	0.003
N_2O	1×10^{-5}	6×10^{-5}	9×10^{-5}	3×10^{-5}

B.3. Vehicle Assumptions

B.3.1 Fuel Economy

For this analysis, I model new vehicles available on the market rather than existing fleets. I use functionally-equivalent vehicles for different fuel pathways for two vehicle types, a compact passenger vehicle and a compact Sports Utility Vehicle (SUV), to eliminate bias.[72] I use the official fuel economy estimates published by U.S. Department of Energy (DOE) and U.S. Environmental Protection Agency (EPA),[173] who test all vehicles on the same duty cycle. There are currently no methanol vehicles in the market, so I rely on the literature for fuel economy estimates. I also rely on the literature to model "standardized" plug-in hybrid vehicle (PHEVs) - PHEV30 and PHEV60 that have 30 or 60 km of all electric range (AER) - since PHEVs offered by vehicle manufacturers differ in their AERs. **Table B.6** and **Table B.7** summarize fuel economy assumptions, and representative vehicle models as well as their specifications (engine, transmission, range, and weight) for passenger vehicles and SUVs included in this analysis.

Table B.6. Passenger vehicle fuel economy assumptions.

Pathway		Representative vehicle	Source	MPGge[i]	Range (km)
Gasoline vehicle (baseline)		2015 Honda Civic (1.8 L, 4 cyl, Automatic (variable gear ratios))	fueleconomy.gov[173]	33	702
Diesel vehicle		2015 BMW 328d (A-S8, 2.0 L, 4cyl)	fueleconomy.gov[173]	32.3[ii]	840
Gasoline hybrid electric vehicle (HEV)		2015 Honda Civic Hybrid (1.5 L, 4 cyl, Automatic (variable gear ratios))	fueleconomy.gov[173]	45	956
PHEV30	CS*	Karabasoglu et al. (2013)[134]	Karabasoglu et al. (2013)[134]	43.8	-
	CD*			112	30

[i] MPGge, miles per gallon of gasoline equivalent.
[ii] The fuel economy of BMW 328d is 37 MPG, which is equivalent to 32.3 MPGge using the energy intensity of diesel and gasoline.

PHEV60	CS*		Karabasoglu et al. (2013)[134]; weight-fuel economy relationship from Shiau et al. (2009)[132].	42.4	-
	CD*			105	60
Battery Electric Vehicle (BEV) 130[i]		2015 Ford Focus Electric	fueleconomy.gov[173]	105	122
		2015 Nissan Leaf	fueleconomy.gov[173]	114	135
		My assumptions	-	110	-
CNG dedicated		2015 Honda Civic Natural Gas (1.8 L, 4 cyl, Automatic 5-spd)	fueleconomy.gov[173]	31	311
M85 dedicated		N/A	GREET 2013[159] (7% more energy efficient than conventional gasoline vehicle)	35.3	-
E85 flex fuel vehicle (FFV)		2015 Ford Focus FWD FFV (2.0 L, 4 cyl, Auto (AM6))	fueleconomy.gov[173]	31.6[ii]	459
Hydrogen fuel cell electric vehicle (FCEV)		2014 Honda Clarity FCX	fueleconomy.gov[173]	59	372

** CS stands for Charge-Sustaining, and CD stands for Charge-Deleting.*

Table B.7. Sports utility vehicle assumptions. All vehicle economy and range estimates are taken from fueleconomy.gov.[173]

Pathway		Representative vehicle	Specifications	MPGge[iii]	Range (km)	GVW (lbs.)
Gasoline vehicle (baseline)	Marketed vehicles	2015 Hyundai Tuscon 2WD,	2.0 L, 4 cyl, Automatic 6-spd	25	615	3294
		2015 BMW X3 xDrive28i,	2.0 L, 4 cyl, Automatic (S8), Turbo	24	665	4150
		2015 Toyota RAV4	2.5 L, 4 cyl, Automatic (S6)	26	684	3700
		2015 Lexus NX 200t	2.0 L, 4 cyl, Automatic (S6), Turbo	25	641	3940
	My assumptions	N/A	N/A	25	N/A	N/A

[i] Since Honda Civic does not have an all-electric version, we use a comparable BEVs from other manufacturers. The Nissan Leaf has an AER of 84 miles with a 24 kWh battery and the Ford Focus Electric has an AER of 76 miles and a 23kWh battery (from fueleconomy.gov). While the Nissan Leaf and the Ford Focus have roughly the same all-electric range, their equivalent fuel economy differs by 10%. To be more conservative, we assume that the representative BEV has the average parameters of Nissan Leaf and Ford Focus Electric, i.e. 100 MPG with an AER of 130 km (80 miles) and a 24 kWh battery.

[ii] The fuel economy of Ford Focus FFV is 23 MPG, which is equivalent to 31.6 MPGge using the energy intensity of E85 and gasoline. Note that the GREET model (versions 2013)[159] achieves that FFV has the same MPGge as conventional gasoline vehicle.

[iii] MPGge, miles per gallon of gasoline equivalent.

Diesel vehicle	2015 BMW X3 xDrive 28d	2.0 L, 4 cyl, Automatic (S8), Turbo	26.2[i]	855	4230
Gasoline HEV	2015 Lexus NX 300h	2.5 L, 4 cyl, Automatic (S6)	33.0	785	4055
E85 flex fuel vehicle	2015 Chevrolet Captiva FWD	2.4 L, 4 cyl, Automatic 6-spd	24.7[ii]	563	3801
Hydrogen fuel cell vehicle	2015 Hyundai Tucson Fuel Cell	Fuel cell power (max): 100 kW	49.0	426	-
BEV165	2014 Toyota RAV4 EV	Automatic (variable gear ratios)	76.0	166	4032

B.3.2 Tailpipe Methane and N_2O Emissions

Table B.8 summarizes tailpipe CH_4 and N_2O emissions from different vehicle technologies. The main text explains key assumptions and data sources.

Table B.8. Tailpipe methane emissions of light-duty vehicles (LDV).[159]

GHG	Absolute values (Unit: g/km)		Relative values (percentage as gasoline pathway emissions)				
	Gasoline	**Diesel**	**CNG**	**M85**	**HEV**	**PHEV**	**BEV / FECV**
CH_4	0.014	0.006	1,000%	100%	47%	47%	0%
N_2O	0.007	0.007	100%	100%	100%	100%	0%

B.3.3 PHEV-specific Assumptions

This analysis includes two types of PHEVs for passenger vehicles: PHEV30 and PHEV60 with an AER of 30 and 60 kilometers, respectively. The operation of PHEVs is categorized into two modes depending on the battery state of charge (SOC): charge-depleting (CD) mode, in which the vehicle receives some or all of its propulsion energy from the battery; and charge-sustaining (CS) mode, in which gasoline provides all propulsion energy.[134] There are two control strategies for the CD mode, all-electric control (or extended-range) and blended control, which differ if the CD mode uses any non-electric energy sources (such as gasoline).[23,132–134] For simplicity, I assume an all-electric control strategy for the CD mode, which is used first until the battery is depleted to predefined SOC.

[i] The fuel economy of 2015 BMW X3 xDrive 28d given at fueleconomy.gov is 30.0 MPG. We have converted the number to MPGge using the energy intensity of diesel and gasoline.
[ii] The fuel economy of 2015 Chevrolet Captiva FWD given at fueleconomy.gov is 18.0 MPG. We have converted the number to MPGge using the energy intensity of E85 and gasoline.

Following Samaras et al.,[23] I approximate the fractions of vehicle trips powered by electricity and gasoline using National Household Travel Survey (NHTS) 2009.[174] I assume that the fraction of a PHEV (with an AER X km)'s electric drive equals the probability of daily vehicle kilometer traveled less than X km. Furthermore, I assume PHEVs are charged once every day (in the night). **Figure B.1** shows the calculated cumulative distribution, while **Table B.9** shows the selected fractions of electric drive for different types of PHEVs. For PHEV30, this electric range share is 0.44, while for PHEV60 it is 0.68 (**Table B.9**).

I use a probability mixture model to combine the GHG emissions from CD and CS mode (i.e. electric and gasoline drives). I rely on Karabasoglu et al.[134] for the technical specifications of the PHEVs examined. Shiau et al.[132] reported that the additional battery weight associated with increasing the All Electric Range (AER) by 10 mile reduces CD-mode and CS-mode efficiencies by 0.10 mile/kWh and 0.68 MPGge (mile per gallon gasoline equivalent), respectively. Thus, I adjust the fuel economy of the PHEV60, and assume the battery weight effects are accounted for in the fuel economy estimates of HEV and PHEV30.

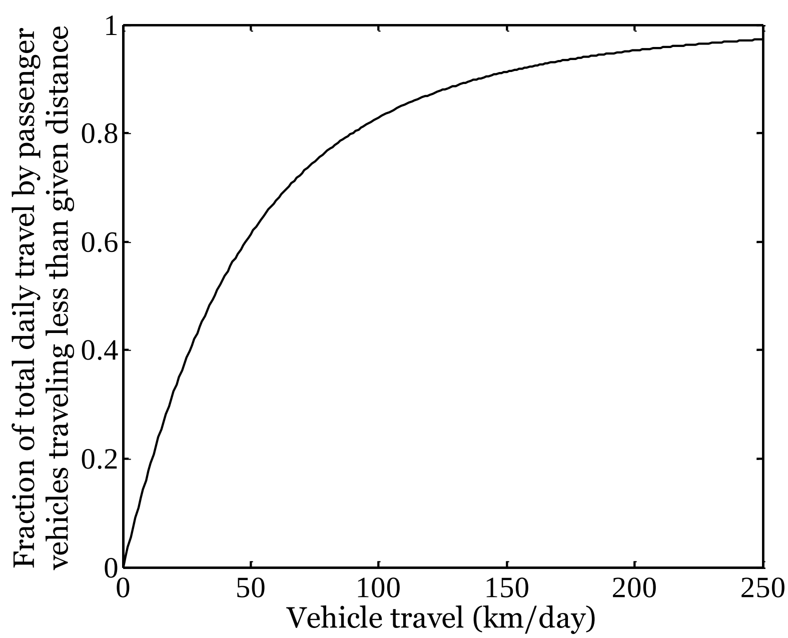


Figure B.1. Cumulative distribution of daily passenger vehicle travel (km/day). The distribution is constructed with data from the National Household Travel Survey 2009.[174]

Table B.9. Fraction of vehicle kilometers powered by electricity in PHEVs if PHEVs are charged overnight (based on Figure B.1).

Range	(mile)				(km)			
	20	30	40	60	20	30	60	90
Probability	0.47	0.60	0.70	0.82	0.32	0.44	0.68	0.80

B.3.4 Vehicle Manufacturing Emissions

Table B.10 summarizes relevant assumptions used to calculate a vehicle's emissions. The main text explains key assumptions and data sources.

Table B.10. Vehicle manufacturing emissions.

Application	Technology	Battery/Fuel cell power plant (FCPP) type and size	Numbers of batteries/FCPP per vehicle lifetime
Hybrid and plug-in electric vehicles			
Passenger Vehicle	Gasoline HEV	1.3 kWh Li-Ion battery	1[23,134,159]
	PHEV30	9.9 kWh Li-Ion battery	
	PHEV60	19.9 kWh Li-Ion battery	
	BEV130	24.0 kWh Li-Ion battery	
SUV	Gasoline HEV	1.6 kWh Ni-Mh battery	
	BEV165	41.8 kWh Li-Ion battery	
Fuel cell electric vehicles (FCEV)			
Passenger vehicle	FCEV	100 kW fuel cell + Li-ion battery[369]	1[159]
SUV	FCEV	100 kW fuel cell	
Vehicle lifetime travel distance			
LDV (passenger vehicles & SUVs)		150,000 miles, or 240,000 km[23,134]	
Vehicle manufacturing emission factors			
Passenger vehicle	Internal Combustion Engine Vehicle (ICEV)	$[CO_2, CH_4, N_2O]$ = [45.2, 0.1, 0.011] gram/mile[159]	
	PHEV/BEV	Calculated as the sum of vehicle manufacturing emissions (same as the ICEV) and additional battery manufacturing emissions	
	FCEV	$[CO_2, CH_4, N_2O]$ = [89.3, 0.2, 0.002] gram/mile[159]	
SUV	ICEV	$[CO_2, CH_4, N_2O]$ = [55.4, 0.2, 0.001] gram/mile[159]	
	PHEV/BEV	Calculated as the sum of vehicle manufacturing emissions (same as the ICEV) and additional battery manufacturing emissions	
	FCEV	$[CO_2, CH_4, N_2O]$ = [102.2, 0.2, 0.002] gram/mile[159]	
Battery manufacturing emission factors			
Battery manufacturing emission factors		5.1 kg CO_2-eq/kg[176] (Li-Ion battery)	
	HEV	0.11 kWh/kg of battery[176]	

Battery specific energy	BEV	0.13 kWh/kg of battery[176]

B.4. Break-Even Methane Leakage Rate Analysis

I derive the closed-form formulas to estimate the break-even life cycle methane leakage rate with respect to relative vehicle fuel economy (normalized to that of a conventional gasoline vehicle), global warming potential (GWP), and the baseline fuel choice (conventional gasoline). I estimate the break-even rate for three pathways: CNG, distributed gaseous hydrogen FCEVs, and BEVs powered with NGCC (Natural Gas Combined Cycle Power Plants) electricity. The system boundary is the same for the break-even analysis and the Monte-Carlo simulations.

$$\begin{aligned}&\textit{gasoline life cycle emissions}\left[gCO_2e/\mathrm{km}\right]\\&=\frac{\textit{gasoline life cycle GHG emissions}\left[gCO_2e/MJ\right]}{\textit{energy efficiency}_{\textit{gasoline vehicle}}\,[\mathrm{km}/MJ]}+\textit{vehicle manufacturing}_{\text{gasoline vehicle}}\left[gCO_2e/\mathrm{km}\right]\end{aligned} \tag{B.1}$$

Eqn. B.1 calculates the life cycle emissions for conventional gasoline vehicles. Here the life cycle include upstream, combustion, and manufacturing emissions but exclude tailpipe methane and N_2O emissions, both of which contribute little to the life cycle emissions. I include the unit in the bracket in equations to help readers understand the unit conversions.

$$\begin{aligned}&\textit{CNG life cycle emissions}\\&=\frac{\textit{CNG life cycle } CO_2 \textit{ emissions}}{\textit{efficiency}_{CNGV}}+\frac{\textit{CNG life cycle methane emissions}}{\textit{efficiency}_{CNGV}}\times GWP_{methane}+\textit{vehicle manufacturing}_{CNGV}\end{aligned} \tag{B.2}$$

Eqn. B.2 calculates the life cycle emissions for the CNG pathway (with a dedicated CNG vehicle). Here I split the use-related life cycle emissions into two parts, CO_2 emissions and methane emissions. I calculate CO_2 emissions and methane emissions as sum of emissions from natural gas upstream, compression, and vehicle tailpipe. Later I will represent methane emissions as a function of leakage rate (percentage of natural gas lost into the atmosphere) and then back calculate the break-even methane leakage rate.

$$
\begin{aligned}
& BEV\ life\ cycle\ GHG\ emissions \\
&= BEV\ life\ cycle\ CO_2\ emissions + BEV\ life\ cycle\ methane\ emissions + vehicle\ manufacturing_{BEV} \\
&= \frac{Natural\ gas\ upstream\ CO_2\ emissions + Natural\ gas\ combustion\ CO_2\ emissions}{efficiency_{power\ plant} \times \left(1 - loss_{transmission\ line}\right) \times efficiency_{charging} \times efficiency_{BEV}} + \\
& \frac{Natural\ gas\ life\ cycle\ CH_4\ emissions}{efficiency_{power\ plant} \times \left(1 - loss_{transmission\ line}\right) \times efficiency_{charging} \times efficiency_{BEV}} \times GWP_{methane} + vehicle\ manufacturing_{BEV}
\end{aligned}
\tag{B.3}
$$

Eqn. B.3 calculates the life cycle emissions for NGCC-electricity-BEV pathway. Again, I split the use-related life cycle emissions into CO_2 and methane emissions. The pathway of generating electricity through NGCC power plants is quite straightforward; natural gas is produced and transported to NGCC power plants (natural gas upstream emissions), then combusted to generate electricity, then transmitted to charging stations where electricity is used to power a vehicle.

$$
\begin{aligned}
& Gaseous\ hydrogen\ FCEV\ life\ cycle\ GHG\ emissions \\
&= GH_2d\ life\ cycle\ emissions\ (except\ natural\ gas\ feedstock) + \\
& Natural\ gas\ feedstock\ for\ hydrogen\ production\ emissions + vehicle\ manufacturing_{FCEV} \\
&= \frac{GH_2d\ life\ cycle\ emissions\ (except\ natural\ gas\ feedstock)}{efficiency_{FCEV}} + \\
& \left(\frac{Natural\ gas\ upstream\ CO_2\ emissions}{efficiency_{FCEV}} + \frac{Natural\ gas\ upstream\ methane\ emissions}{efficiency_{FCEV}}\right) \times Hydrogen\ plant\ feedstock_{natural\ gas} \\
& + vehicle\ manufacturing_{FCEV} \\
&= \frac{GH_2d\ life\ cycle\ emissions\ (except\ natural\ gas\ feedstock)}{efficiency_{FCEV}} + \\
& \frac{Natural\ gas\ upstream\ CO_2\ emissions}{efficiency_{FCEV}} \times Hydrogen\ plant\ feedstock_{natural\ gas} + \\
& \frac{life\ cycle\ methane\ emissions}{efficiency_{FCEV}} \times Hydrogen\ plant\ feedstock_{natural\ gas} + vehicle\ manufacturing_{FCEV}
\end{aligned}
\tag{B.4}
$$

Eqn. B.4 calculates the life cycle emissions for a gaseous hydrogen distributed FCEV pathway. Because hydrogen production plants involve multiple inputs and outputs, I split the hydrogen life cycle emissions in a different way. The life cycle emissions of the gaseous hydrogen pathway include emissions associated with the natural gas feedstock (only natural gas upstream emissions), emissions associated with electricity input (assumed to be grid-average electricity), hydrogen production process emissions, and hydrogen compression emissions.

In Eqn. B.1 through Eqn. B.4, I can further represent vehicle energy efficiency and methane emissions using Eqn. B.5 and Eqn. B.6, respectively. In Eqn. B.5, fuel economy is in mile per gallon of gasoline equivalent (MPGge), as shown in **Table B.6** and **Table B.7**. In Eqn. B.6, methane emissions are calculated as a function of methane leakage rate (the percentage of produced natural gas leaked into the atmosphere).

$$energy\ efficiency_{vehicle}\left[{}^{km}\!/\!{}_{MJ}\right]=\frac{fuel\ economy_{vehicle}\left[{}^{mile}\!/\!{}_{gge}\right]}{energy\ intensity_{gasoline}\left[{}^{MJ}\!/\!{}_{gallon}\right]\times conversion\left[{}^{mile}\!/\!{}_{km}\right]} \tag{B.5}$$

$$\begin{aligned}&methane\ emission\left[{}^{g\mathrm{CO_2}e}\!/\!{}_{MJ}\right]\\&=\frac{methane\ leakage\ rate\times natural\ gas\ composition_{methane}\times methane\ density\left[{}^{\mathrm{gram}}\!/\!{}_{cf}\right]\times GWP_{methane}\left[{}^{g\mathrm{CO_2}e}\!/\!{}_{gram}\right]}{energy\ intensity_{Natural\ gas}\left[{}^{MJ}\!/\!{}_{cf}\right]}\end{aligned} \tag{B.6}$$

The break-even methane leakage rate is defined as the methane leakage rate at which natural gas-based fuels' life cycle emissions equal conventional gasoline's life cycle emissions. To calculate the break-even methane leakage rate, I equaled Eqn. B.1 with Eqn. B.2-B.4 separately, re-arranged terms, and reached the following Eqn. B.7-B.9 for the three natural gas pathways.

$$\begin{aligned}&Break\text{-}even\ methane\ leakage\ rate_{CNG}\\&=\left(EER_{CNG\ vehicle}\times gasoline\ life\ cycle\ GHG\ emissions-CNG\ life\ cycle\ CO_2\ emissions\right)\times\frac{energy\ density_{methane}}{GWP_{methane}}\end{aligned} \tag{B.7}$$

$$\begin{aligned}&Break\text{-}even\ methane\ leakage\ rate_{NGCC-BEV}\\&=\begin{bmatrix}EER_{BEV}\times EF_{natural\ gas\text{-}electricity}\times\left(gasoline\ life\ cycle\ GHG\ emissions-\Delta vehicle\ manufacturing_{BEV}\times efficiency_{gasoline\ vehicle}\right)\\-\left(Natural\ gas\ upstream\ CO_2\ emissions+Natural\ gas\ combustion\ CO_2\ emissions\right)\end{bmatrix}\\&\times\frac{energy\ density_{methane}}{GWP_{methane}}\\&where\ EF_{natural\ gas\text{-}electricity}\equiv efficiency_{power\ plant}\times\left(1-loss_{transmission\ line}\right)\times efficiency_{charging}\end{aligned}$$

(B.8)

$$
\begin{aligned}
&\textit{Break-even methane leakage rate}_{GH_2d\text{-}FCEV} \\
&= \left[\begin{aligned} &\frac{EF_{natural\ gas\text{-}electricity}}{Hydrogen\ \ plant\ \ feedstock_{natural\ gas}} \times \left(gasoline\ life\ cycle\ GHG\ emissions - \Delta vehicle\ manufacturing_{BEV} \times efficiency_{gasoline\ vehicle} \right) \\ &- \frac{1}{Hydrogen\ \ plant\ \ feedstock_{natural\ gas}} \times GH_2d\ life\ cycle\ emissions\ (except\ natural\ \ gas\ \ feedstock) - Natural\ \ gas\ upstream\ CO_2\ \ emissions \end{aligned} \right] \\
&\times \frac{energy\ density_{methane}}{GWP_{methane}}
\end{aligned}
$$

(B.9)

Here I define the energy economy ratio (EER) as the ratio between a vehicle's gasoline-equivalent fuel economy to another vehicle's gasoline-equivalent fuel economy.

$$
Energy\ Economy\ Ratio\ (EER)_{vehicle\ A} = \frac{fuel\ economy_{vehicle\ A,\ gasoline\ equivalent}}{fuel\ economy_{gasoline\ vehicle}} \tag{B.10}
$$

and the energy density of methane can be calculated as,

$$
energy\ density_{methane}\left[{MJ}/{gram} \right] = \frac{energy\ intensity_{Natural\ gas}\left[{MJ}/{cf} \right]}{natural\ \ gas\ composition_{methane} \times methane\ density\left[{\mathrm{gram}}/{cf} \right]} \tag{B.11}
$$

Eqn. B.7-B.11 give the exact formulas to calculate the break-even methane leakage rates for the selected natural gas pathways. All the inputs used to solve these equations are from the Monte-Carlo simulation model but I use the average estimates instead of distributions. Some of the key inputs are as follows.

- Methane density is 20.3 g/ft^3, and a cubic foot (ft^3) of natural gas has 93.4% methane on average. The energy density of natural gas is 1.037 MJ/ ft^3 (or 983 BTU/ ft^3). The energy intensity of gasoline is 118.4 MJ_{LHV}/gallon or 112,194 BTU/gallon.

- The conventional ICEV vehicle has a fuel economy of 33 MPG and an energy efficiency of 0.28. Conventional gasoline has life cycle GHG emissions of 91.5 gCO_2-eq/MJ_{LHV} (for both 20-year and 100-year time periods).
- CO_2 emissions from natural gas upstream and combustion are 7.99 gCO_2-eq/MJ_{LHV} and 55.39 gCO_2-eq/MJ_{LHV}.
- The energy efficiency of NGCC power plants is 55.7% (LHV basis), the average transmission and distribution line loss is 6.5%, and the average BEV charging efficiency is 86.5%. In other words, the overall energy efficiency from the delivery of natural gas at the front gate of the NGCC power plant to the electricity in the BEV is 45%.
- The input of natural gas feedstock for a hydrogen production plant is 1.3 MJ per MJ of hydrogen produced. The life cycle emissions of hydrogen include the cradle-to-bus-bar emissions of the natural gas input and electricity input (13.2 gCO_2-eq/MJ_{LHV}), emission factors of hydrogen production plant (77.1 gCO_2-eq/MJ_{LHV}), and hydrogen compression emissions (14.8 gCO_2-eq/MJ_{LHV}), which brings the subtotal to be 127.6 gCO_2-eq/MJ_{LHV}.
- While all the above assumptions are fuel-specific and are transparent to vehicle types, vehicle manufacturing emissions are different for passenger vehicles and SUVs. For passenger vehicles, compared to conventional ICEVs, the additional vehicle manufacturing emissions of a BEV (with a battery size of 24 kWh) are 6.3 gCO_2-eq/mile and the additional vehicle manufacturing emissions of a FCEV are 48.1 gCO_2-eq/mile.

With all these assumptions and values, the break-even formulas can be simplified to the following expressions, where the break-even methane leakage rates are only dependent on the EER and the GWP.

$$\begin{aligned}
&\textit{Break-even methane leakage rate}_{CNGV} = \left(91.5 \times EER_{CNGV} - 71.0\right) \times \frac{0.055}{GWP_{methane}} \\
&\textit{Break-even methane leakage rate}_{NGCC-BEV} = \left(40.4 \times EER_{BEV} - 63.4\right) \times \frac{0.055}{GWP_{methane}} \qquad \text{(B.12)} \\
&\textit{Break-even methane leakage rate}_{GH_2d\text{-}FCEV} = \left(59.3 \times EER_{FCEV} - 87.5\right) \times \frac{0.055}{GWP_{methane}}
\end{aligned}$$

I plot the break-even methane leakage rates with regard to EERs in the main text, and calculate the break-even methane leakage rates for selected EERs in **Table B.11**. Note that current vehicle technologies have an EER of 94%, 179%, and 333% for a CNGV, a FCEV, and a BEV, respectively.

Table B.11. Break-even methane leakage rates for natural gas pathways (the baseline fuel pathway is conventional gasoline).

EER	CNG		NGCC-BEV		GH_2d-FCEV	
	100-year GWP	20-year GWP	100-year GWP	20-year GWP	100-year GWP	20-year GWP
90%	1.7%	0.7%	-	-	-	-
95%	2.4%	1.0%	-	-	-	-
100%	3.1%	1.3%	-	-	-	-
110%	4.5%	1.9%	-	-	-	-
120%	5.9%	2.4%	-	-	-	-
150%	-	-	-	-	0.2%	0.1%
175%	-	-	1.1%	0.5%	2.5%	1.0%
200%	-	-	2.6%	1.1%	4.7%	2.0%
225%	-	-	4.2%	1.7%	7.0%	2.9%
250%	-	-	5.7%	2.4%	9.2%	3.8%
275%	-	-	7.2%	3.0%	11.5%	4.8%
300%	-	-	8.8%	3.6%	13.7%	5.7%
325%	-	-	10.3%	4.3%	-	-
350%	-	-	11.9%	4.9%	-	-
375%	-	-	13.4%	5.5%	-	-
400%	-	-	14.9%	6.2%	-	-

B.5. Additional Results

B.5.1 Natural Gas Upstream GHG Emissions

Here I provide detailed results with breakdowns by GHGs and by processes in each upstream stage (**Table B.12**, **Table B.13**, **Figure B.2** and **Figure B.3**). In accordance with the main text, I consider four scenarios: baseline methane estimate with 100-year GWP; baseline methane estimate with 20-year GWP; pessimistic methane estimate with 100-year GWP; and pessimistic methane estimate with 20-year GWP. The estimates show that there is a very large uncertainty range of natural gas upstream GHG emissions. The mean upstream total GHG emission is 17.4 gCO_2-eq/MJ_{LHV} and the 95% confidence interval is 10.3-29.5 gCO_2-eq/MJ_{LHV}. I find that the distribution of total upstream GHG emissions is highly asymmetrical. In addition, I fit distributions to the GHG emissions from the natural gas system (the functional unit is one mega joule of natural gas delivered at the end of distribution pipelines) in **Table B.12.** I chose the fitted distribution[i] based on maximizing the negative of the log likelihood, Bayesian information criterion (BIC), and Akaike information criterion (AIC).

In the breakdown of upstream stages, well-site production (including both preproduction and production) and pipeline transportation (including transmission and distribution) contribute most to GHG emissions and all these stages have large methane emissions. By comparison, natural gas processing is responsible for a much lower share of GHG emissions and only 79% of produced natural gas has to be processed at processing plant.[10,282]

I find that the relative contribution from methane is higher than carbon dioxide (**Figure B.2** and **Figure B.3**). For the baseline scenario, methane contributes slightly more than carbon dioxide. For the 20-year GWP scenario, the contribution of methane is nearly two times more than carbon dioxide. While carbon dioxide is quite evenly distributed across natural gas upstream stages (mostly as result of fuel combustion), methane emissions are more concentrated. Most methane emissions occur either at the well site (liquid unloading, well completion and well workover), or in the pipeline system (fugitive emissions). In addition, there is evidence that a small share of super-emitters is responsible for a larger share of GHG emissions (as summarized in Brandt et al.[114] and reflected in the right-skewed distribution shown in this study).

Existing studies of natural gas GHG emissions span a wide range with a 95% uncertainty range as of 11.0-21.0 gCO_2-eq/MJ_{LHV} (compiled using estimates from six individual bottom-up studies)[115]. While the baseline estimate (mean value) aligns well with existing studies, there are two differences. First, I find smaller emissions from natural gas preproduction, production, and processing stages[ii] while larger emissions from natural gas pipeline systems compared to most

[i] Candidate distributions include Beta, Birnbaum-Saunders, Exponential, Extreme Value, Gamma, Generalized Extreme Value, Generalized Pareto, Inverse Gaussian, Logistic, Log Logistic, Lognormal, Nakagami, Normal, Rayleigh, Rician, t location-scale, and, Weibull.
[ii] See Table SI-5 in the Supporting Information of Weber et al. (2012)[115] for a summary of GHG emissions from preproduction, production & processing, and transmission stages.

existing studies (Howarth et al.[141], is an exception). Second, I find a lower methane leakage rate from natural gas systems and higher carbon dioxide emissions. I think these two observations are partially the results of reduced methane emissions in well completions and well workovers that have been the result of better industry practices and stringent regulation on well completions.

Table B.12. Natural gas upstream emissions with breakdown of upstream stages and GHGs. Both 100-year and 20-year GWP estimation results are shown as well as the pessimistic case (methane emissions are multiplied by 1.5). Mean estimate and 95% confidence interval (in parenthesis) are shown in table entries.

Stage	**GHG emissions breakdown**		**Total GHG emissions**			
	CO_2 (baseline)	**CH_4 (baseline)**	**100-year GWP (baseline)**	**100-year GWP (pessimistic)**	**20-year GWP (baseline)**	**20-year GWP (pessimistic)**
Unit	gram/MJ_{LHV}		gCO_2-eq/MJ_{LHV}			
Pre-production	1.3 (0.5-3.1)	0.006 (0-0.05)	1.5 (0.6-4.2)	1.7 (0.6-4.9)	1.9 (0.6-6.3)	2.2 (0.6-8.2)
Production	2.6 (0.7-7.3)	0.09 (0.06-0.23)	6.0 (2.8-13.3)	7.7 (3.7-17.1)	10.7 (5.7-24.1)	14.8 (7.9-34.2)
Processing	2.2 (0.3-8.6)	0.008 (0-0.05)	2.5 (0.4-9.2)	2.7 (0.4-9.6)	2.9 (0.5-10.5)	3.3 (0.5-12)
Transmission	1.8 (0.4-6.1)	0.09 (0.08-0.11)	5.2 (2.7-9.6)	6.9 (3.7-11.7)	9.9 (6.2-15.2)	14 (8.8-20.7)
Distribution	0.002 (0.001-0.002)	0.06 (0.05-0.07)	2.0 (1.1-3.1)	3.0 (1.6-4.7)	4.9 (3.0-7.1)	7.3 (4.5-10.7)
Upstream total	8.0 (3.6-16.9)	0.26 (0.20-0.43)	17.2 (10.2-29.3)	22.0 (12.9-36.7)	30.3 (19.3-49.7)	41.7 (26.3-68.8)
Fitted distribution for upstream total	Generalized extreme value (‘shape’=0.13, ‘scale’=2.28, ‘location’ = 6.33)	Generalized extreme value (‘shape’=0.30, ‘scale’=0.024, ‘location’ = 0.23)	Log logistic (‘log location’ = 2.80, ‘log scale’ = ‘0.15’)	Log logistic (‘log location’ = 3.37, ‘log scale’ = ‘0.13’)	Log logistic (‘log location’ = 3.05, ‘log scale’ = ‘0.14’)	Log logistic (‘log location’ = 3.69, ‘log scale’ = ‘0.13’)

Table B.13. Natural gas upstream emissions with the breakdown of processes in each upstream stage (Unit: gCO_2-eq/MJ_{LHV}). Both 100-year and 20-year GWP estimation results are shown as well as the pessimistic case (methane emissions are multiplied by 1.5). Mean estimate and 95% confidence interval (in parenthesis) are shown in table entries.

Stage	Process	100-year GWP (baseline)	100-year GWP (pessimistic)	20-year GWP (baseline)	20-year GWP (pessimistic)
Pre-production	Wellpad Construction	0.2 (0.1-0.6)			
	Well Drilling	0.4 (0.1-1.0)			
	Hydraulic Fracturing	0.6 (0.2-1.5)			
	Well Completion	0.3 (0.0-2.4)	0.4 (0.0-3.2)	0.7 (0.0-4.8)	0.9 (0-6.8)
Production	Lease Fuel Use	2.0 (0.4-6.2)			
	Flaring	0.6 (0.0-2.8)	0.7 (0.1-3.1)	0.8 (0.1-3.7)	0.9 (0.1-4.4)
	Liquid Unloading	0.7 (0.0-4.7)	1.0 (0.0-7.0)	1.6 (0.0-11.2)	2.4 (0-16.9)
	Well Workover	0.3 (0.0-2.5)	0.4 (0.0-3.4)	0.7 (0.0-5.0)	0.9 (0-7.1)
	Other Fugitive Emissions	2.4 (1.2-3.6)	3.5 (1.8-5.5)	5.7 (3.5-8.2)	8.6 (5.3-12.4)
Processing	CO_2	2.2 (0.3-8.6)			
	CH_4	0.3 (0.0-1.8)	0.4 (0.0-2.7)	0.7 (0.0-4.3)	1.1 (0-6.4)
	N_2O	< 0.01			
Transmission	Fuel Use – Natural gas	1.7 (0.3-6.0)			
	Fuel Use – Electricity	0.1 (0.1-0.1)			
	Fugitive Emissions	3.4 (1.7-5.2)	5.1 (2.6-7.8)	8.1 (5.0-11.8)	12.2 (7.5-17.7)
Distribution	Fugitive Emissions	2.0 (1.1-3.1)	3.0 (1.6-4.7)	4.9 (3.0-7.1)	7.3 (4.5-10.7)
Upstream total emissions		17.4 (10.3-29.5)	22.0 (12.9-36.7)	30.3 (19.3-49.7)	41.7 (26.3-68.8)
Implicit methane leakage rate		1.3% (1.0%-2.2%)		2.0% (1.6%-3.3%)	

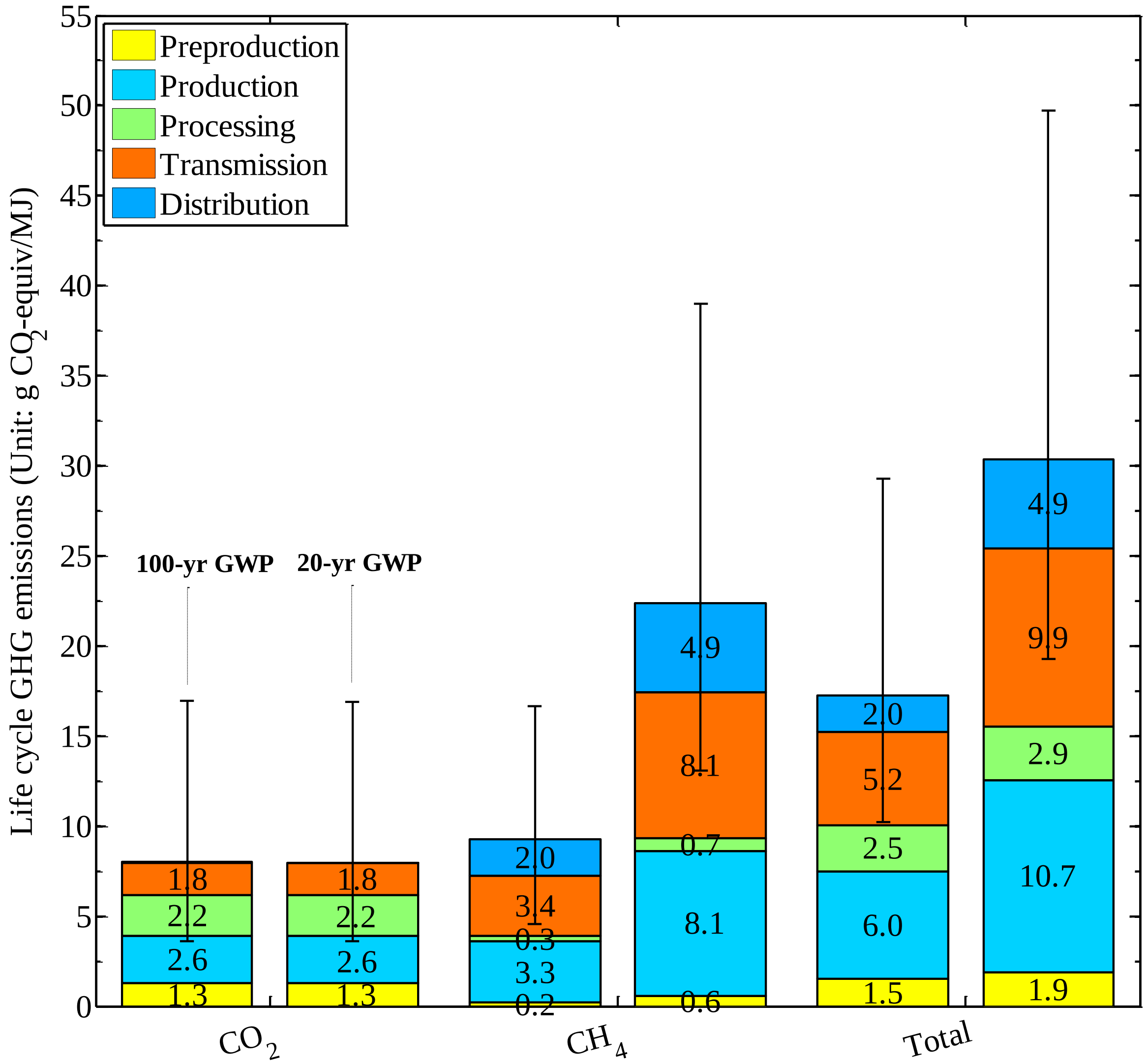


Figure B.2. Breakdown of natural gas upstream GHG emissions by greenhouse gas and by upstream stages. Error bar are based on the 95% confidence interval of the total emissions for each GHG. Estimation results with 100-year GWP (left bars) and 20-year GWP (right bars) are shown side by side.

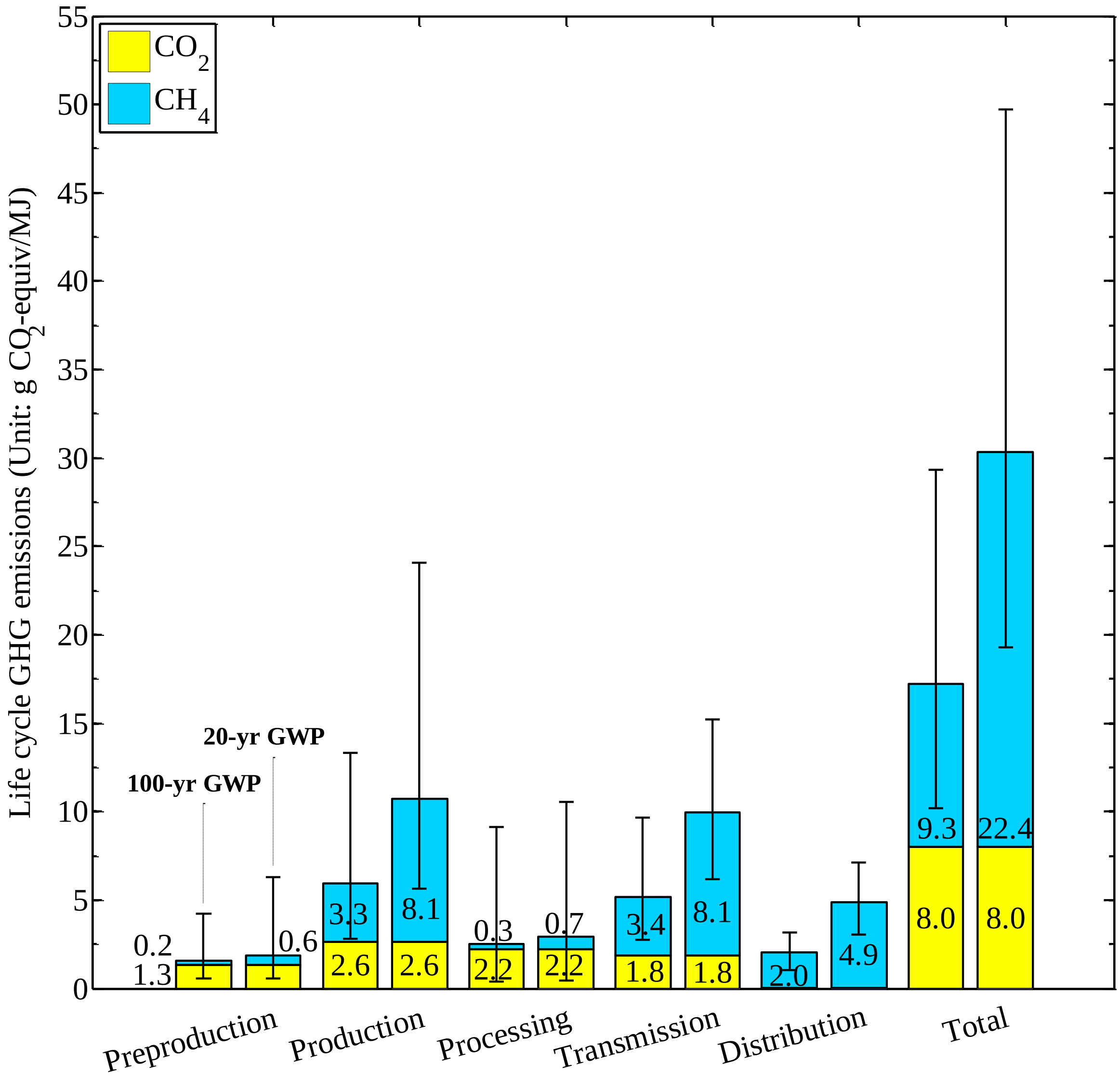


Figure B.3. Breakdown of natural gas upstream GHG emissions by upstream stages and by greenhouse gases. Error bars are based on the 95% confidence interval of the total emissions for each GHG. Estimation results with 100-year GWP (left bars) and 20-year GWP (right bars) are shown side by side.

B.5.2 Carbon Intensity of Fuel Pathways

Figure B.4 and **Figure B.5** shows the carbon intensity of all fuel pathways considered in this study. The figure shows the breakdown of GHG emissions from upstream activities (well-to-pump) and fuel combustion. **Table B.14** summarizes the carbon intensity of power generation in a different unit. A spreadsheet Supplemental Data file is available online and includes numbers behind the figures. While fuel carbon intensity is of interest to policymakers (for instance, California sets the Low Carbon Fuel Standards for transportation fuels), we caution that fuel carbon intensity should not be directly compared unless the efficiency of end use technology is considered. These results, however, may be useful to other researchers who wish to compare my estimates with other sources or wish to evaluate a wide range of end-use technologies beyond those included in this study.

Table B.14. GHG emissions for electricity generation (Unit: gCO2-eq/kWh). Shown in the table are mean estimates and 95% confidence interval (in parenthesis) from this study.

Life cycle stage	NGCC without CCS	NGCC with CCS	Grid average (2010)[157]
Upstream	98 (57, 174)	115 (67, 204)	48
Combustion	358 (348, 368)	50 (48, 51)	564
Total (at power plant gate)	456 (413, 533)	165 (116, 254)	612

Note: Here I report energy efficiency in lower heating value.

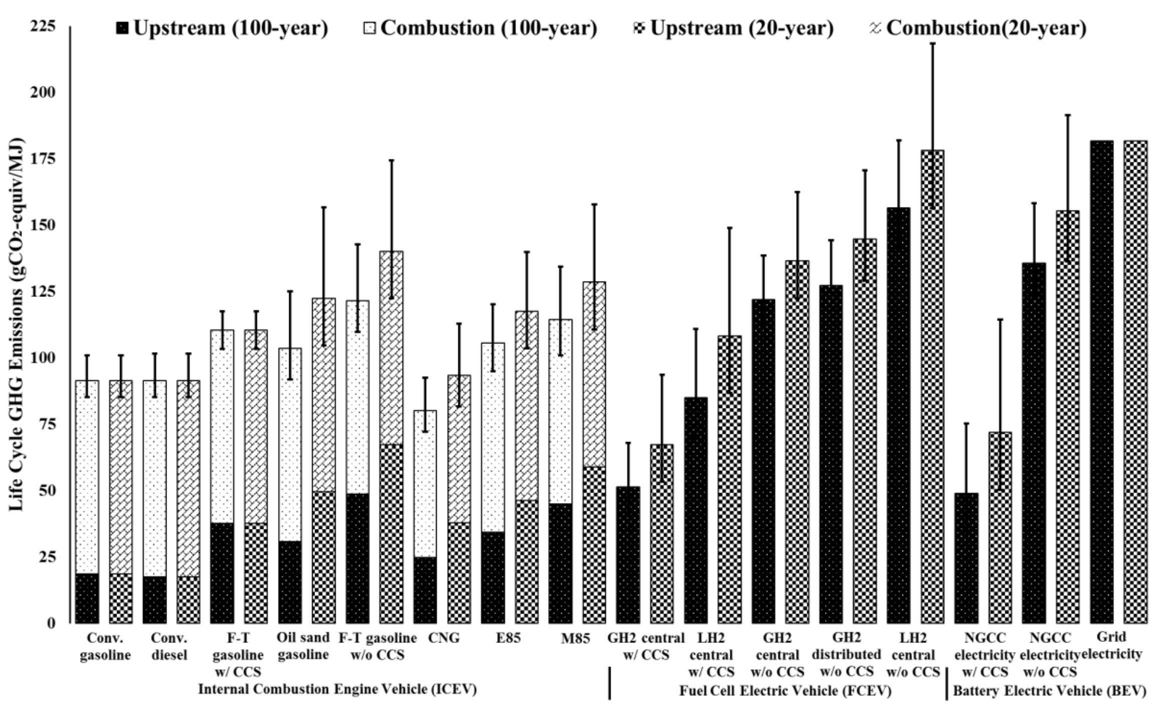


Figure B.4. Life cycle GHG emissions ('carbon intensity') of natural gas-derived fuels and petroleum fuels with 100-year and 20-year GWP. Baseline methane emissions estimates are assumed for natural gas pathways. The functional unit is 1 MJ (lower heating value) of fuel delivered to end use. These estimates do not account for vehicle fuel efficiencies or tailpipe

CH_4 and N_2O emissions, as they depend on vehicle technologies. Error bars represents the 95% confidence interval of the life cycle GHG emissions. See the spreadsheet Supplementary Data file for numbers.

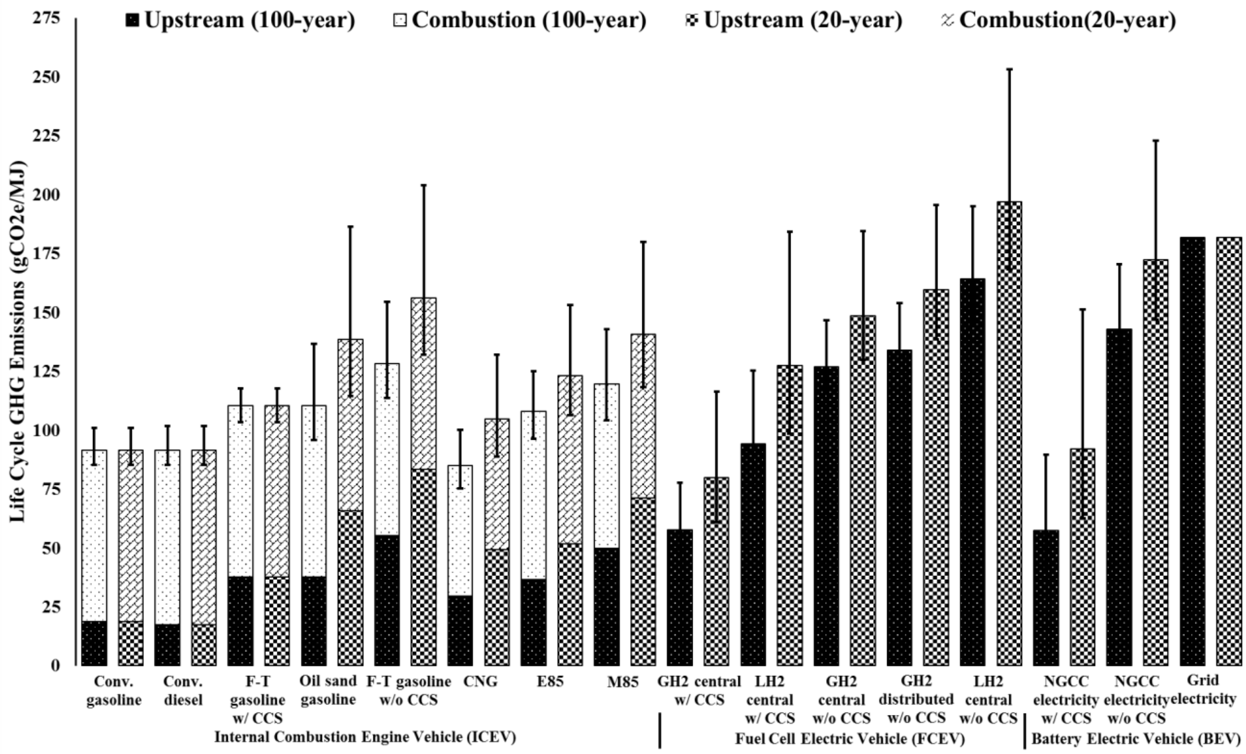
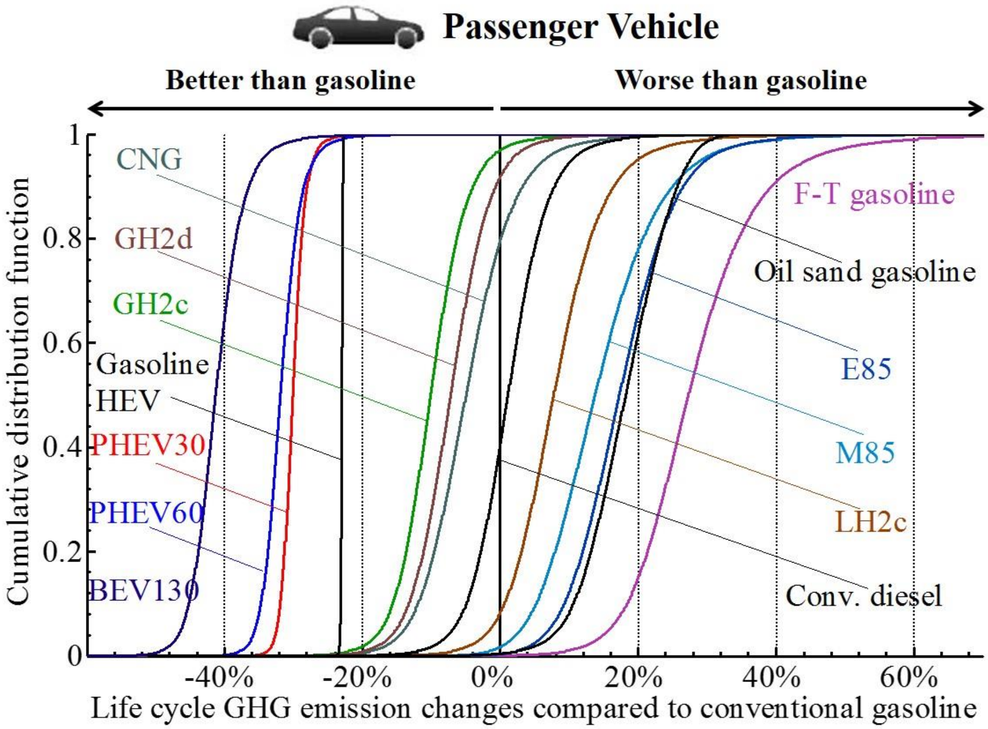


Figure B.5. Life cycle GHG emissions ('carbon intensity') of natural gas-derived fuels and petroleum fuels with 100-year and 20-year GWP. Pessimistic methane emissions estimates are assumed for natural gas pathways. The functional unit is 1 MJ (lower heating value) of fuel delivered to end use. These estimates do not account for vehicle fuel efficiencies or tailpipe CH_4 and N_2O emissions, as they depend on vehicle technologies. Error bars represents the 95% confidence interval of the life cycle GHG emissions. See the spreadsheet Supplementary Data file for numbers.

B.5.3 Additional Results from the Monte-Carlo Simulations

Figure B.6- Figure B.9 shows the cumulative distribution function of the relative changes between life cycle GHG emissions of natural gas pathways and that of conventional gasoline. These figures highlight that natural gas pathways have larger uncertainty and variability than the conventional gasoline pathway. I also find strict stochastic dominance among natural gas pathways, which allows us to use the average GHG emissions to determine the ranks of natural gas pathways. Note that a spreadsheet Supplemental Data file is available online and includes numbers behind the figures.



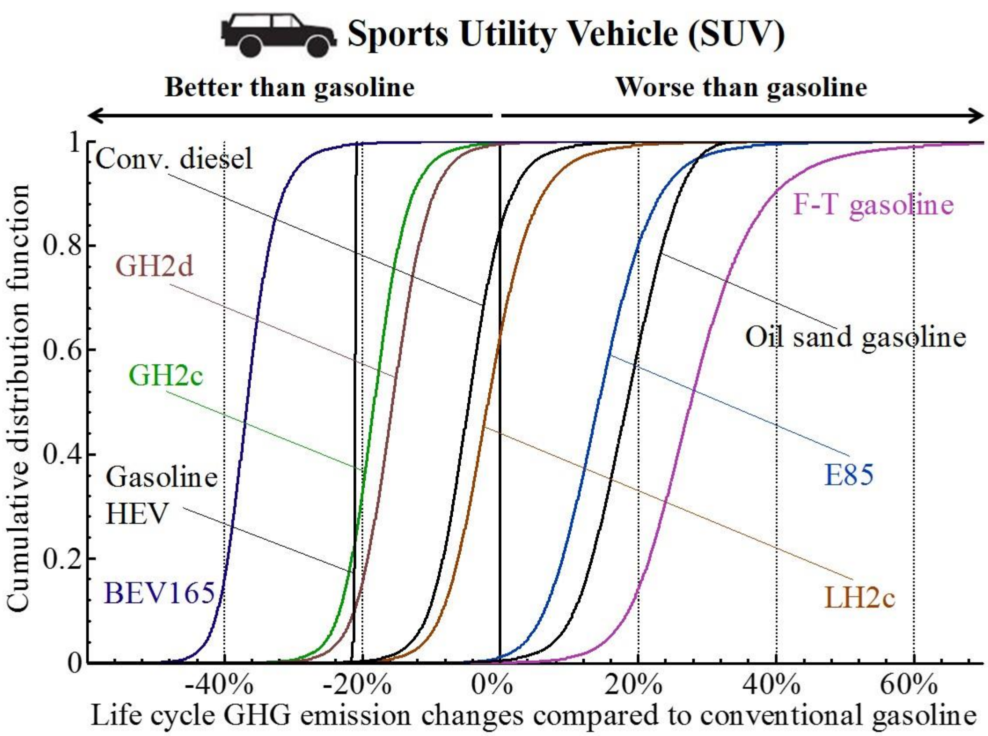


Figure B.6. Cumulative probability distributions (CDF) of life cycle GHG emission changes compared to the baseline gasoline pathways for LDVs with 100-year GWPs and baseline methane emissions estimate. (The cartoon icons are from the Alternative Fuels Data Center; http://www.afdc.energy.gov/.)

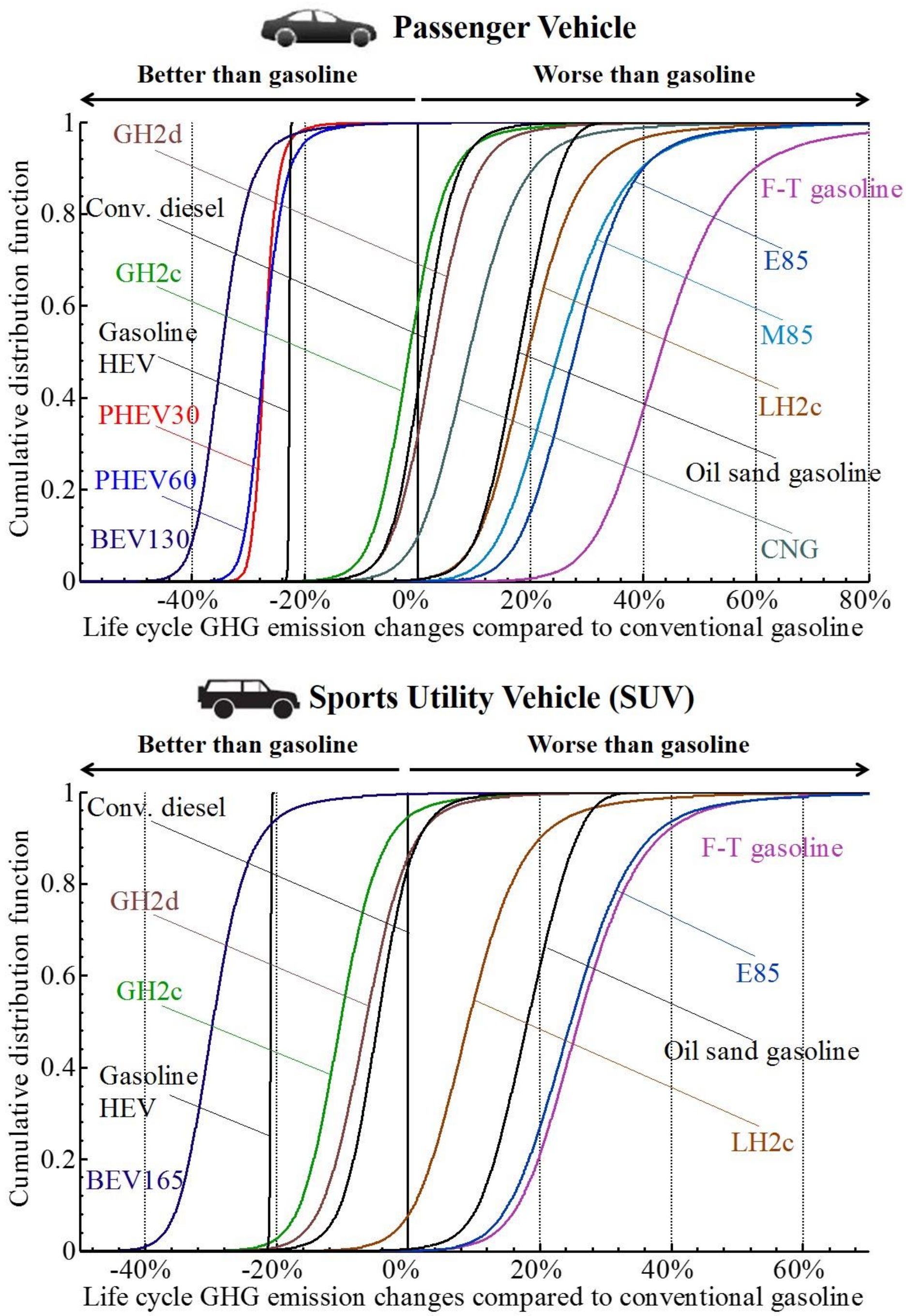


Figure B.7. Cumulative probability distributions (CDF) of life cycle GHG emission changes compared to the baseline gasoline pathways for LDVs with 20-year GWPs and baseline methane emissions estimate. (The cartoon icons are from the Alternative Fuels Data Center; http://www.afdc.energy.gov/.)

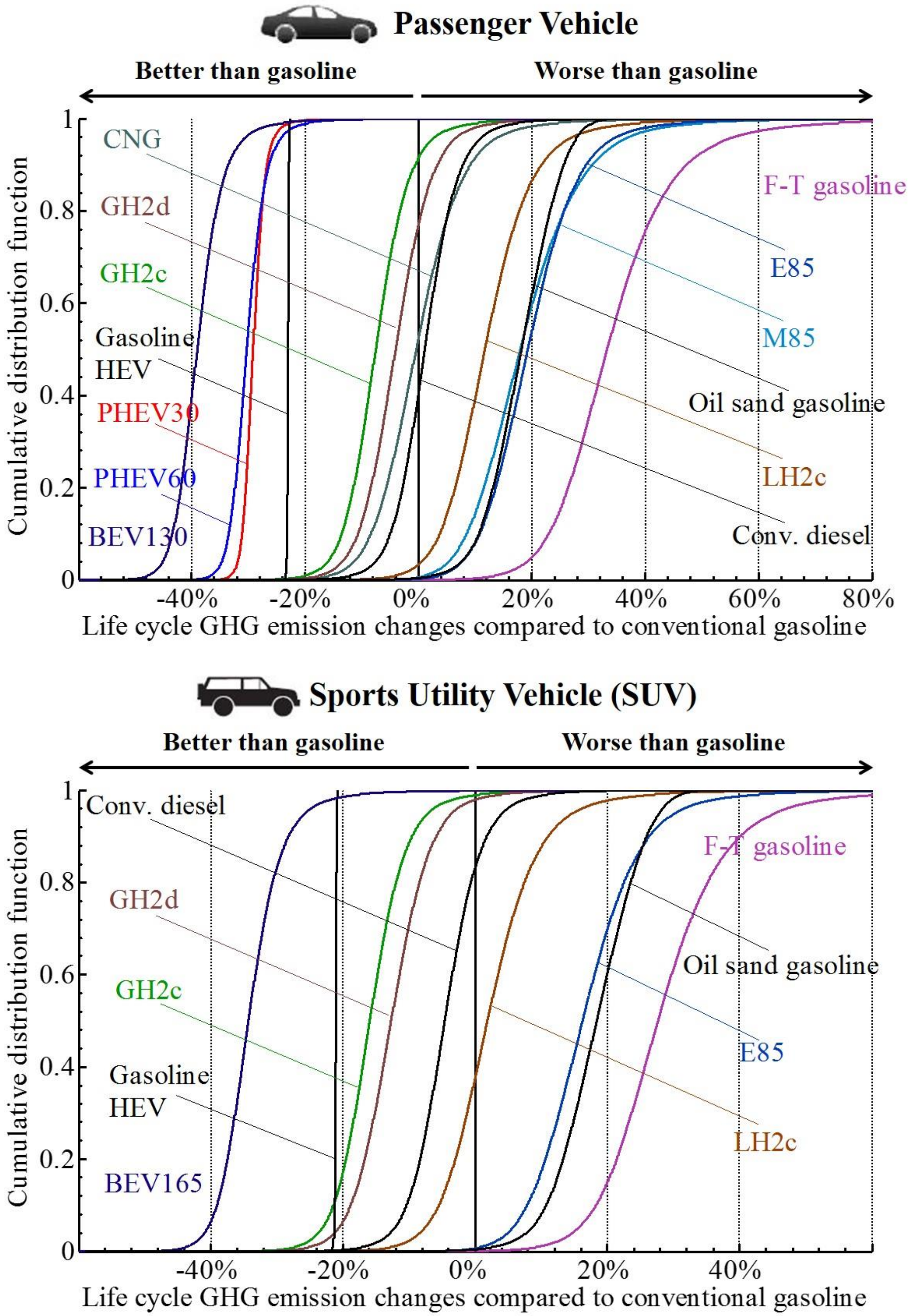


Figure B.8. Cumulative probability distributions (CDF) of life cycle GHG emission changes compared to the baseline gasoline pathways for LDVs with 100-year GWPs and pessimistic methane emissions estimate. (The cartoon icons are from the Alternative Fuels Data Center; http://www.afdc.energy.gov/.)

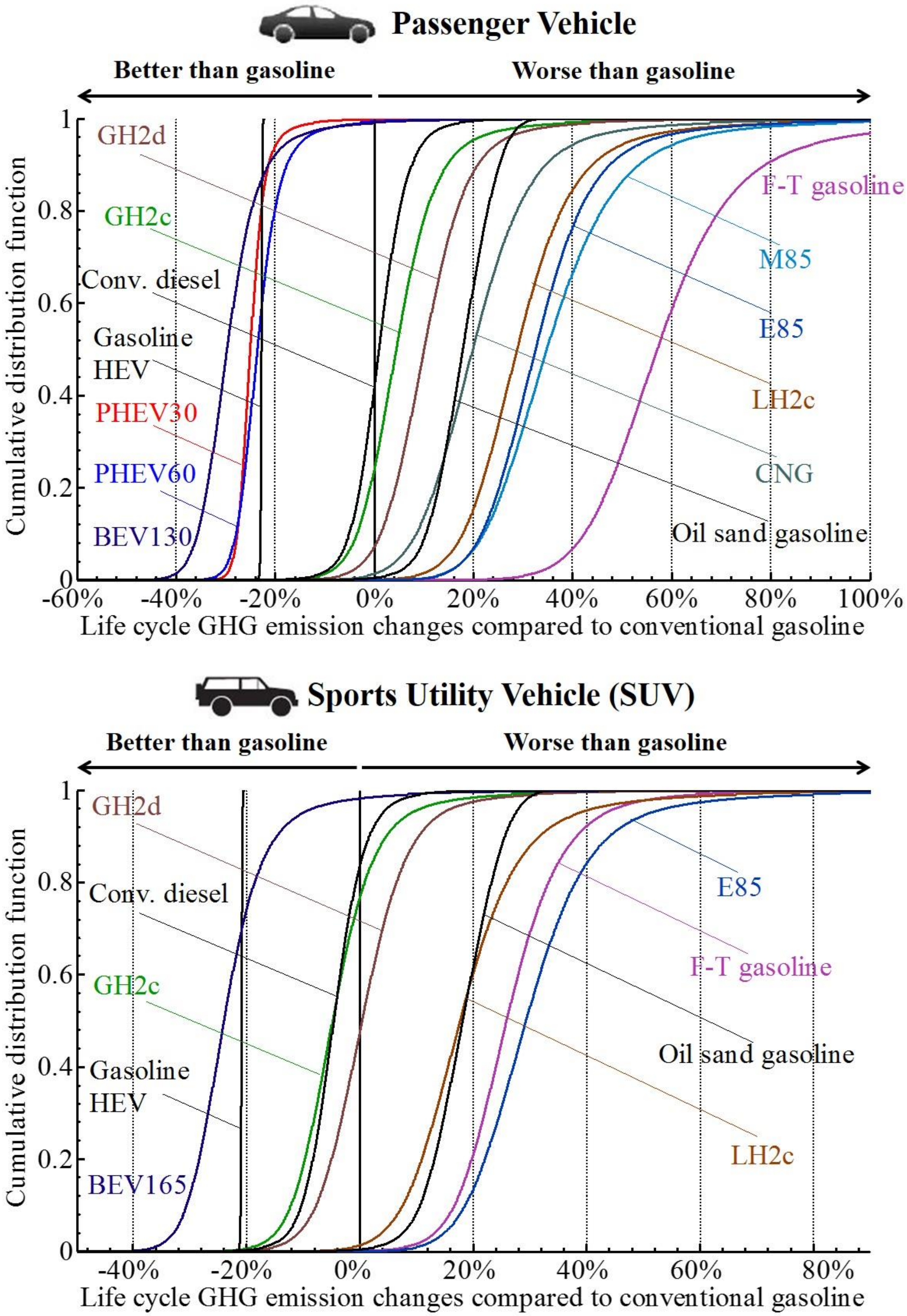


Figure B.9. Cumulative probability distributions (CDF) of life cycle GHG emission changes compared to the baseline gasoline pathways for LDVs with 20-year GWPs and pessimistic methane emissions estimate. (The cartoon icons are from the Alternative Fuels Data Center; http://www.afdc.energy.gov/.)

B.5.4 Impact of the Carbon Intensity of Electricity on GHG emissions of Electric Vehicles

Figure B.10 shows that life cycle GHG emissions from HEVs, PHEVs, and BEVs are linear functions of the carbon intensity of electricity sources. Here, I use the same system boundary and formulas with the Monte-Carlo simulation model except that I treat electricity input as a parameter. I use the 100-year GWP[2]. I find that with current U.S. grid, BEV130 and PHEV30 are slightly better than gasoline HEVs in terms of life cycle GHG emissions, while PHEV60 is worse. If carbon capture and storage (CCS) technology is available at NGCC power plants, BEVs reduce GHG emissions by nearly 75% compared to gasoline HEVs. With NGCC electricity, BEV130 is much better than gasoline HEVs as I have shown in the main text. **Table B.15** summarizes the tipping points of carbon intensity of electricity inputs where these electric pathways have the same emissions. My estimates of the tipping points are different from Samaras et al. (2008)[23] in that I account for incremental vehicle manufacturing emissions and include non-CO_2 GHG emissions from the vehicle tailpipe. In addition, I use updated assumptions on fuel economy of electric vehicles and battery manufacturing emissions. Samaras et al. (2008)[23] report a tipping point of roughly 690 gCO_2-eq/kWh, below which PHEV60 emits less GHGs than PHEV30. I find that a much cleaner electricity source is needed[i] (520 gCO_2-eq/kWh) in order for PHEV60 to emit less than PHEV30.

[i] An interesting fact in our study is that the tipping point between PHEV30 and PHEV60 is around 510 gCO_2-eq/kWh which is also the mean life cycle GHG emissions of NGCC electricity.

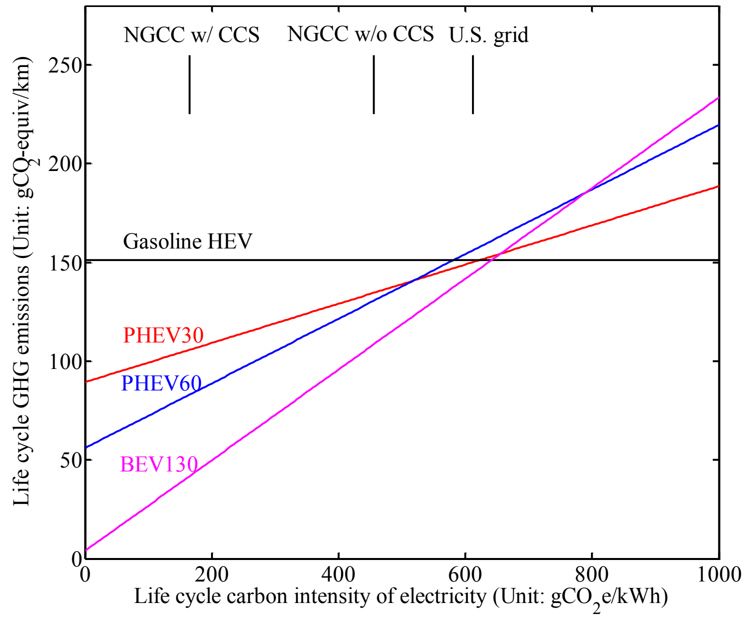


Figure B.10. Life cycle GHG emissions from HEV, PHEV, and BEV as a function of life cycle carbon intensity of electricity generation.

Table B.15. Break-even carbon intensity of electricity inputs for plug-in electric vehicles.

Index	Break-even between		Break-even carbon intensity of electricity (Unit: gCO_2-eq/kWh)
1	PHEV30	BEV130	790
2	HEV	BEV130	640
3	HEV	PHEV30	625
4	HEV	PHEV60	580
5	PHEV30	PHEV60	520

B.6. Comparison with Existing Studies

I compare my results with the GREET model[159] which has been widely used in the literature.[15,16,20,22,26,29,55,136] I choose two recent versions of the GREET model, version 2013 and version 2014, which differ primarily in updated assumptions on fugitive methane emissions from natural gas systems and GWP values. While the version 2013 is not most up-to-date, it provides fuel production and transport assumptions used in this study. The GREET model by default focuses on existing LDVs and assumes the U.S. grid as the electricity source. I thus modified the GREET model settings to use fuel economy assumptions in this study (**Table B.6**) and to use NGCC electricity for plug-in electric vehicles.

Figure B.11 shows the comparisons for life cycle GHG emissions for natural gas pathways in passenger vehicles. I find that my results are comparable with those from the GREET models. The point estimates of most natural gas pathways (except the M85 and E85 pathways) from the GREET models fall in the 95% confidence intervals of life cycle emissions from my model. In fact, for most of these pathways, my average estimates are comparable with the point estimates in the GREET model. I find smaller differences between my average emission estimates with the GREET model Version 2014 than with Version 2013. Version 2014 updated the fugitive methane emissions from natural gas systems and used the new GWPs in the IPCC AR5 report[2], thus having higher emission estimates for all natural gas pathways.

There are differences between my model and the GREET model for M85 and E85 pathways. For the M85 pathway, the differences are solely from different estimates in upstream (well-to-pump) emissions of M85: 34.9 gCO_2-eq/MJ in my model compared to 17.5 gCO_2-eq/MJ. My assumptions on methanol production are less optimistic than those used in the GREET model. If I use GREET's assumptions of energy efficiency and input shares, my upstream emissions of M85 change to 27.9 gCO_2-eq/MJ, bringing the differences in life cycle emissions to less than 6%. In addition, my natural gas upstream emissions are slightly higher than the GREET model. The differences for the E85 pathway are much easier to explain, as the GREET model does not a natural gas-based E85 pathways but I include the corn-based E85 for comparison.

As I have discussed in the main text, although existing studies used the GREET model to estimate the emission reduction potentials and the cost-effectiveness of natural gas pathways compared to petroleum fuels, they failed to include a comprehensive set of pathways, used outdated data with regard to natural gas upstream emissions and GWP, and largely ignored uncertainty and variability, especially those related to fugitive methane emissions from natural gas systems. This study addresses these limitations and provides an independent emission inventory in addition to the GREET model.

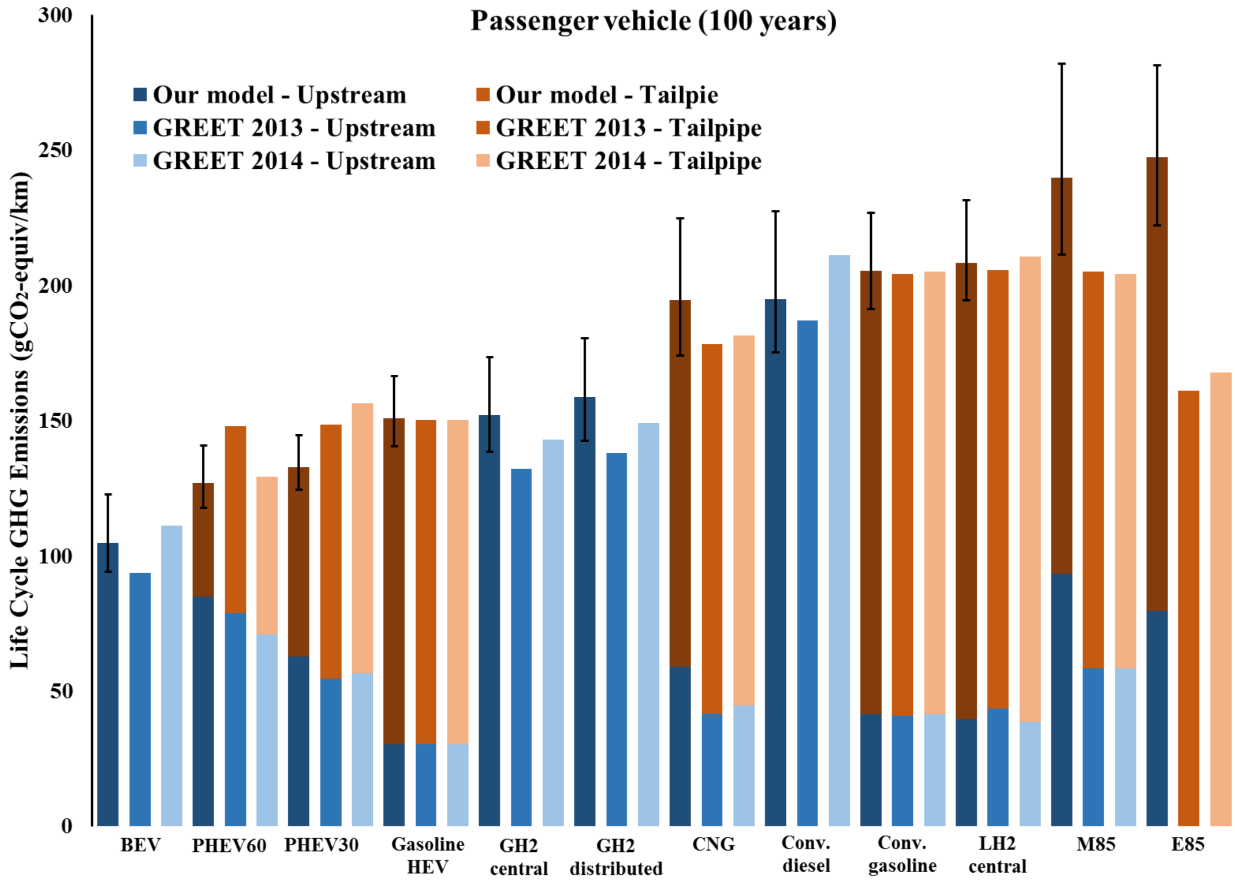


Figure B.11. Comparison of life cycle GHG emissions of natural gas pathways for passenger vehicles from this study, the GREET model (version 2013), and the GREET model (version 2014). Results from this study are from the baseline scenario (baseline methane estimate and 100-year GWPs). Results from the GREET model (version 2013 and 2014) use the same fuel economy assumptions and assume NGCC electricity for plug-in electric pathways. Error bars represent the 95% confidence interval of life cycle GHG emissions. The E85 pathway in the GREET model assumes corn as the feedstock input.

B.7. Data Quality

In this study, as with other LCA work, I relied on many data sources. In the main text and this Appendix, I discussed how I used these data sources and what my model leads to. Here, I discuss the data quality of these sources. Specifically I discuss the type of source and where I used it in my model. I divide data sources into the following categories: peer-reviewed journal papers (including peer-reviewed conference proceedings), thesis, conference presentations, academic working papers, government sources (including those authored or contracted by national laboratories), vehicle manufacturer specifications, and industry consulting reports. For several model assumptions, peer-reviewed data sources do not exist, in which case I relied on alternative data sources. In addition to data sources, I also include the information regarding the nature of data for natural gas upstream emissions and vehicle assumptions. For instance, are they based on actual emission measurements or vehicle tests? Direct data sources, such as emission measurements or vehicle tests are preferred over indirect sources.

B.7.1 Units and Metrics

I summarize data sources in **Table B.16**. Here I used the authoritative sources wherever possible. I use the fuel properties and combustion emission factors from the GREET model[159] because it is the mostly-widely used LCA model in the transportation sector and some of their periodic updates are peer-reviewed.

Table B.16. Review of data sources related to units, metrics and fuel properties.

Type	Data Source	Type of source
Global warming potential	IPCC AR5[2]	Peer-reviewed inter-government report.
Fuel properties and combustion emission factors (except for natural gas)	GREET[159]	Government or national laboratory data source (LCA model).
Methane composition in natural gas	U.S. EPA (2014)[152]	Government or national laboratory report.
	Foss (2007)[275]	Non peer-reviewed academic institute source.

B.7.2 Gasoline and Diesel

I rely on peer-reviewed journal papers for upstream and combustion emissions of conventional and oil-sand derived gasoline and diesel, as summarized in **Table B.17**.

Table B.17. Review of data sources used to estimate GHG emissions of gasoline and diesel.

Type	Data Source	Type of source
Upstream emissions for conventional gasoline and diesel	Venkatesh et al. (2011)[169]	Peer-reviewed journal paper.
Combustion emissions for gasoline and diesel	ref. [159,168–170]	Two peer-reviewed journal papers, and one government or national laboratory report.
Oil sand-derived gasoline/diesel	Englander et al.[171]	Peer-reviewed journal paper.

B.7.3 Fuel Production

I rely on peer-reviewed journal papers and reports from the national laboratories to model fuel production profiles (**Table B.18**). I use multiple data sources for validation purposes as well as for constructing distributions for the Monte Carlo model.

Table B.18. Review of data sources for fuel production assumptions.

Type	Data Source	Type of source
Electricity		
NGCC Energy efficiency	NETL (2013)[158]	Government or national laboratory report.
U.S. grid average electricity	Cai et al. (2013)[159157]	Government or national laboratory report.
Compressed Natural Gas (CNG)		
Energy efficiency of electric compressors	Venkatesh et al. (2011)[27]	Peer reviewed journal paper.
Natural Gas-Based Hydrogen (H2)		
Central hydrogen plant profile (w/o CCS)	ref. [111,159,160]	Government or national laboratory reports.
Central hydrogen plant profile (w/ CCS)	H2A 3.0[160]	Government or national laboratory report.
Distributed hydrogen plant profile (w/o CCS)	ref. [159,160]	Government or national laboratory reports.
Boil-Off Effects of Liquid H_2	GREET[159]	Government or national laboratory report.
Fischer-Tropsch Liquids		
Centralized Fischer-Tropsch liquids production plant	Jaramillo et al. (2008)[161]	Peer-reviewed journal paper.
Methanol		
Methanol production profile	ref. [22,159,164,165]	One government or national laboratory data source (LCA model), two scientific society reports, and one consulting report
Ethanol		
Ethane steam cracking production profile	ref. [144]	Peer-reviewed journal paper.
Ethylene hydration production profile	ref. [167]	A LCA database provided by Universities and national laboratories in Switzerland

B.7.4 Fuel Transport

I rely on the GREET model[159] for fuel transport assumptions for liquid fuels (**Table B.19**).The GREET model assumes a national-average transportation profile in terms of transportation modes, energy intensities, and distances. While regional variations exist in terms of transportation profiles, the shares of transportation emissions in life cycle GHG emissions are very small.

Table B.19. Review of data sources used for fuel transportation assumptions.

Type	Data Source	Type of source
GHG emission factors of fuel transport for F-T liquids, GH_2 (central), LH_2 (central), methanol and ethanol.	GREET[159]	Government or national laboratory data source (LCA model).

B.7.5 Vehicle

I summarize data sources on vehicle fuel economy assumptions in **Table B.20**. Most natural gas pathways have new vehicles offered by original equipment manufacturers (OEM). U.S. EPA regulates, monitors and publishes fuel economy information through standardized vehicle tests.

Table B.20. Review of data sources for vehicle fuel economy assumptions.

Type	Data Source	Type of source
Passenger vehicle		
Gasoline vehicle, diesel vehicle, gasoline HEV, BEV, CNG vehicle, E85 FFV, Hydrogen FFV	fueleconomy.gov[173]	Government or national laboratory data source (vehicle tests).
PHEV	ref. [132,134]	Peer-reviewed journal papers.
M85	GREET[159]	Government or national laboratory data source (LCA model).
Sports Utility Vehicle (SUV)		
All pathways	fueleconomy.gov[173]	Government or national laboratory data source (vehicle tests)

I summarize data sources used for calculating vehicle manufacturing emissions in **Table B.21**. Emission factors of battery and fuel cell manufacturing are from peer-reviewed journal papers and the GREET model[159]. Assumptions related to the battery and fuel cell sizes are collected from vehicle specifications of literature if actual vehicles do not exist. I note that emissions from FCEVs vehicle manufacturing are significantly larger than those of conventional ICEVs. In the main text, I mentioned that if the fuel cells have to be replaced even once during the life of the vehicle, all hydrogen pathways would increase life cycle GHG emissions compared to the conventional gasoline pathway. Given that current FCEVs have only been available in certain regions for less than a decade, it is still early to say much about the life time of fuel cells.

Table B.21. Review of data sources for battery and fuel cell manufacturing emissions.

Type	Data Source	Type of source
Emission factors		
Battery manufacturing emissions	ref. [176]	Peer-reviewed journal paper.
Battery specific energy	ref. [176]	Peer-reviewed journal paper.
Vehicle manufacturing emissions	GREET[159]	Government or national laboratory data source (LCA model).
PHEV/BEV charging energy efficiency	ref. [23,134,159]	Two peer-reviewed journal papers and one government or national laboratory report.
PHEV use patterns		
Household travel survey	ref. [174]	Government or national laboratory report (national survey).
LDV lifetime travel distance		
LDV lifetime travel distance	ref. [23,134]	Two peer-reviewed journal papers.
EV battery and fuel cell size and replacement		
Gasoline HEV and BEV battery size	fueleconomy. gov[173]	Government or national laboratory data source (vehicle tests)
PHEV battery size	ref. [132,134]	Peer-reviewed journal papers.
FCEV fuel cell size	fueleconomy. gov[173]	Government or national laboratory data source (vehicle tests)
Battery replacement	ref. [23,134,159]	Two peer-reviewed journal papers, and one government or national laboratory data source (LCA).
Fuel cell replacement	GREET[159]	Government or national laboratory data source (LCA model).

I summarize data sources for tailpipe methane and N_2O emissions in **Table B.22**. Tailpipe emissions are relatively small and comparable across natural gas pathways. The only exception is CNG, which has a tailpipe methane emission factor that is 10 times higher than the conventional ICEVs. However, tailpipe emissions only represent 2% of life cycle emissions of the CNG pathway.

Table B.22. Review of data sources for tailpipe methane and N_2O emissions.

Type	Data Source	Type of source
Tailpipe methane and N_2O emissions	GREET[159]	Government or national laboratory data source (LCA model).

Appendix C. Supporting Information for Chapter 4

C.1. Geographical Details

This study uses three levels of geographical resolutions in the Contiguous U.S. (or the lower 48 states of the United States). The coarsest level treats the lower 48 states as a whole, either because technology assumptions remain the same across the U.S. or because a lack of data prevents more detailed analysis. For instance, I estimate the weighted average air pollution damages related to primary energy extraction for the U.S. because I don't know the trade flows between primary energy supplies and demands. The U.S. electric power grid is interconnected, so I model the grid electricity-associated health damages based on the NERC (North American Electric Reliability Corporation) region.[206] There are eight NERC regions in the Contiguous U.S., namely FRCC, MRO, NPCC, RFC, SERC, SPP, TRE, and WECC.[206] The NERC regions do not follow state boundaries. **Figure C.1** is a map of NERC regions from the U.S. Environmental Protection Agency (EPA).

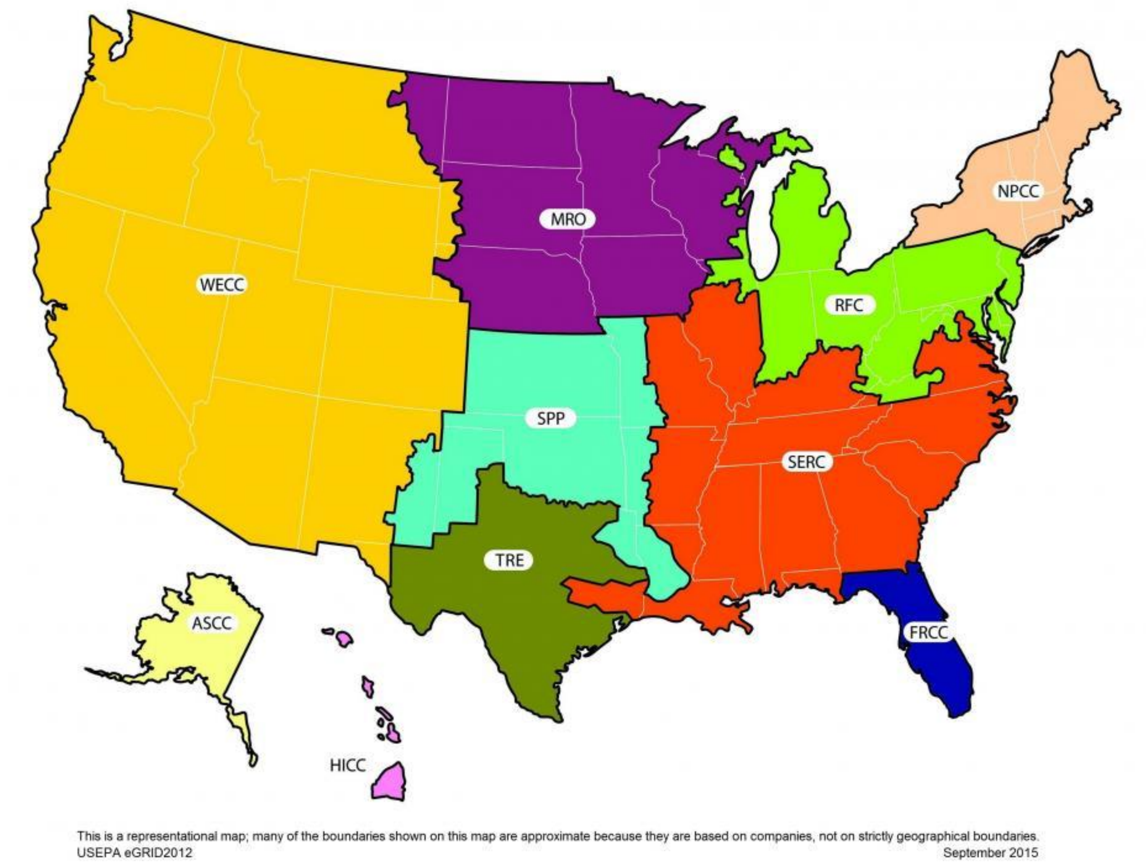


Figure C.1. Map of the NERC regions in the U.S.[206]

Some data, such as the marginal air pollution damages of criteria air pollutants (CAPs) are available for each county in the Contiguous U.S., which is the most refined resolution in this analysis. I use the latest county definitions from the U.S. Census Bureau.[370]

A hierarchical structure exists between counties, states, and the lower 48 states. Simply put, any county belongs to one and only one state. There does not, however, exist a similar hierarchical structure between counties and NERC regions. A county may be served by one or multiple NERC regions. The relationship between a county and its primary NERC region can also change from time to time. In the absence of better data, I determine the county's primary NERC region as the NERC region that serves the largest fraction of population in the county. Specifically, I use the relationship between the ZIP Code Tabulation Area (ZETA) and NERC sub regions provided in the U.S. EPA's Power Profiler tool[371] and the relationship between ZETA and counties from the U.S. Census Bureau[372] to calculate the fraction of population in each county served by each NERC region. **Figure C.2** shows the resulting map of the primary NERC region for counties in the Contiguous U.S. There are some differences in the boundaries of NERC regions between **Figure C.1** and **Figure C.2**. This is likely due to differences between the protocols I used compared to the ones U.S. EPA used to determine the primary NERC region for counties that are served by more than one NERC region.

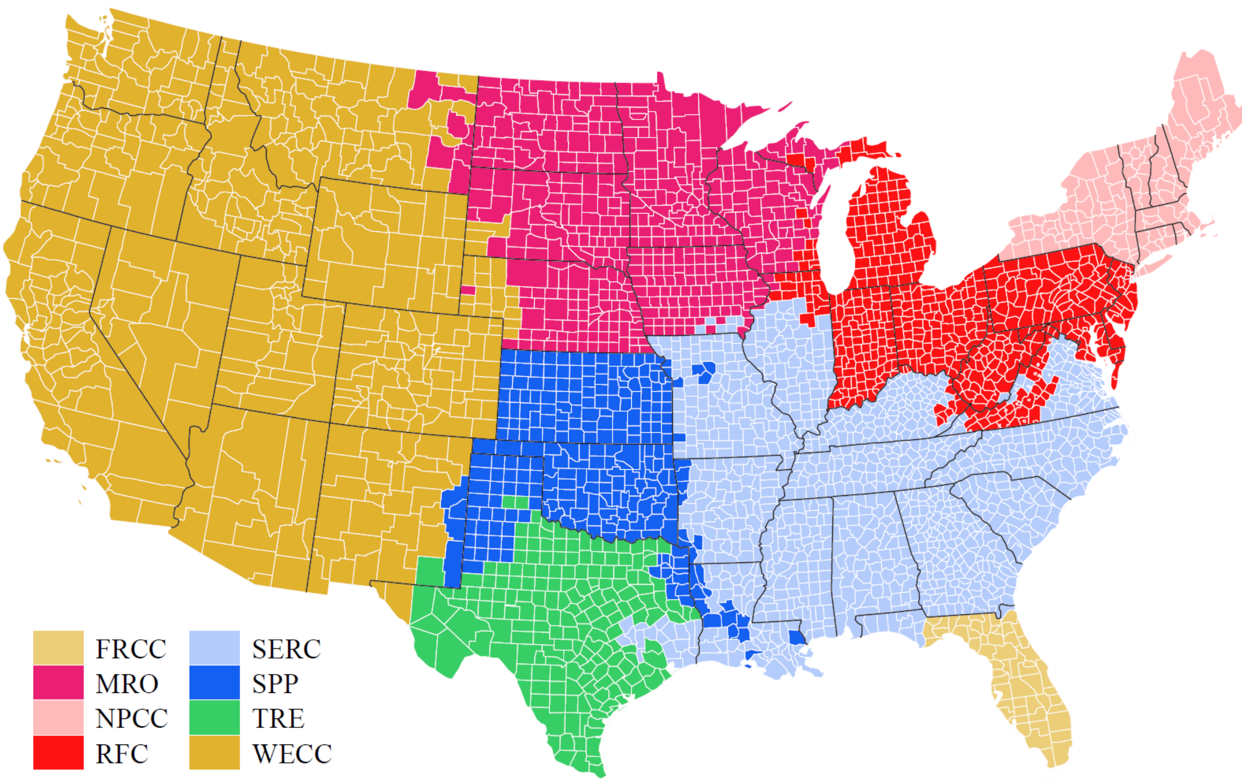


Figure C.2. Map of the primary NERC region for U.S. counties (used in this study).

The EASIUR model[57,195] relied on the same county definition I used but the AP2 model used an earlier county definition. I corrected the differences by updating the FIPS code for Dade County, Florida now FIPS # 12086 (formerly Miami-Dade County and FIPS code 12025). Similarly, AP2 does not include Broomfield County, Colorado (FIPS code 08014), which was created from the nearby four counties (formerly FIPS codes 08001, 08013, 08059, and 08123). Without further information, I calculate the marginal damages of each CAP species for the new county as the simple average of marginal damages of the same CAP species for the four nearby counties. Finally, the formerly independent city of Clifton Forge, VA (former FIPS code 51560) is now part of Alleghany County, VA (FIPS code 51005). I calculate marginal damages of CAPs for the

Alleghany County as the simple average of the two counties' marginal damages in the AP2 model.

C.2. Fuel Specifications

Table C.1 lists the fuel specifications used in this study. In **Table C.1**, I use the lower heating value (LHV) for liquid fuels and higher heating value (HHV) for coal.

Table C.1. Fuel specifications used in this study.

Fuel	Energy density	Unit	Reference
Coal	19,620,000	BTU/short ton	U.S. EIA[373]
Conventional gasoline	112,194	BTU/gallon	GREET (version 2015)[159]
Conventional diesel	128,450		
Crude oil	5,800,000	BTU/barrel	U.S. EIA[374]

C.3. Primary Energy Production Processes: Assumptions and Air Pollution Damages

C.3.1 Primary energy production processes (on-site damages)

I use two emissions data sources to estimate the air pollution damages associated with primary energy production processes, U.S. EPA's National Emission Inventory (NEI),[194] and the GREET model (version 2014).[159]

Following Jaramillo and Muller (2016),[193] I estimate the air pollution damages of extracting and transporting crude oil, natural gas, and coal in the U.S. using U.S. EPA's NEI. Specifically, I first multiply the county emissions data on primary energy extraction with the marginal damages of CAPs at the county level to calculate the total damages for producing one primary energy product. I then calculate the weighted-average damages of an energy product weighted by CAP emissions from each county (Eqn. C.1). Because I do not know the trade flows of energy products, I use the weighted-average damages across the U.S. Because I do not know the trade flows of energy products, I use the weighted-average damages across the U.S. **Table C.2** and **Table C.3** summarize the weighted-average emission factors and weighted-average air pollution damages for energy production processes based on emissions data in U.S. EPA's NEI.[194]

$$
\begin{aligned}
&\textit{Air pollution damages per unit of energy product}_{\textit{energy product } p,\, \textit{model } m} \\
&= \sum_{\textit{CAP species } s} \frac{\sum_{\textit{County } c} \textit{CAP emissions}_{p,s,c} \times \textit{marginal damages}_{m,s,c}}{\sum_{\textit{County } c} \textit{energy output}_{p,c}} \\
&= \sum_{\textit{CAP species } s} \frac{\sum_{\textit{County } c} \textit{CAP emissions factor}_{p,s,c} \times \textit{energy output}_{p,c} \times \textit{marginal damages}_{m,s,c}}{\sum_{\textit{County } c} \textit{energy output}_{p,c}}
\end{aligned}
\tag{C.1}
$$

Where CAP emissions factors come from U.S. EPA's NEI,[194] and energy outputs come from Jaramillo and Muller (2016),[193] and marginal damages$_{m,s,c}$ come from the AP2 model or the EASIUR model.

Jaramillo and Muller (2016)[193] reported weighted-average emissions factors on energy production processes calculated from the GREET model (version 2014). Assuming that the GREET model's underlying emissions inventory and the NEI have the same spatial distributions on county-level CAP emissions associated with energy production processes, I calculate the GREET-based normalized air pollution damages of energy production processes by multiplying the weighted-average marginal damages of CAPs for energy production processes (Eqn. C.2) and the GREET model's weighted-average CAP emission factors on energy outputs (Eqn. C.3).

$$
\begin{aligned}
&\textit{Air pollution damages of CAPs}_{\textit{energy product } p,\, \textit{model } m,\, \textit{CAP species } s} \\
&= \frac{\sum_{\textit{County } c} \textit{CAP emissions}_{p,s,c} \times \textit{marginal damages of CAPs}_{m,s,c}}{\sum_{\textit{County } c} \textit{CAP emissions}_{p,s,c}} \\
&= \frac{\sum_{\textit{County } c} \textit{CAP emissions factor}_{p,s,c} \times \textit{energy output}_{p,c} \times \textit{marginal damages}_{m,s,c}}{\sum_{\textit{County } c} \textit{CAP emissions factor}_{p,s,c} \times \textit{energy output}_{p,c}}
\end{aligned}
\tag{C.2}
$$

Where CAP emissions factors come from U.S. EPA's NEI,[194] energy outputs come from Jaramillo and Muller (2016),[193] and marginal damages$_{m,s,c}$ come from the AP2 model or the EASIUR model. **Table C.4** shows the weighted-average air pollution damages of CAPs for energy extraction processes.

$$Air\ pollution\ damages\ per\ unit\ of\ energy\ product_{energy\ product\ p,\ model\ m,\ GREET's\ emissions\ data} = \sum_{CAP\ species\ s} Air\ pollution\ damages\ of\ CAPs_{p,m,s} \times CAP\ emissions\ factor_{p,s,GREET\ model} \quad (C.3)$$

Where CAP emissions factors come from the GREET model, as reported in Jaramillo and Muller (2016),[193] and **Table C.2**. Finally, **Table C.3** summarizes the air pollution damages for energy extraction processes using the GREET model's emissions data. I find that GREET-based air pollution damages are uniformly higher than marginal damages estimated using emissions data from U.S. EPA's NEI for all energy production processes considered. While the differences for crude oil production and natural gas production are quite small (GREET-based damages are 32-58% and 6-16% higher), GREET-based air pollution damage estimates for oil refinery, and coal production are 494-593% and 154-275% higher than those based on U.S. EPA's NEI. (Here the range of differences are from air pollution damages using two marginal damage models.) Because petroleum fuels (gasoline and diesel) are the baseline fuels for all vehicle types, the large difference in the estimated oil refinery damages has serious impacts on the comparison and rankings of fuel pathways.

Table C.2. Production-weighted average emissions factors for energy production processes (Reproduced from Table 2 and Table S9 in Jaramillo and Muller (2016)[193]).

Energy Process	Air Pollutant	U.S. EPA's NEI 2011	GREET
Coal mining and transportation (short ton/million short ton of coal)	NO_x	54.01	44.9
	$PM_{2.5}$	12.48	8.06
	SO_2	3.38	29.4
	VOC	2.39	4.1
	NH_3	3.16	N/A
Crude oil extraction and transportation	NO_x	46.11	68.3
	$PM_{2.5}$	1.65	3.46

(short ton/million BOE)	SO_2	13.44	17.3
	VOC	26.14	3.86
	NH_3	0.61	N/A
Natural gas extraction and transportation (short ton/million BOE)	NO_x	46.11	102
	$PM_{2.5}$	1.65	1.87
	SO_2	13.44	3.65
	VOC	26.14	4.94
	NH_3	0.61	N/A
Oil refineries and petroleum fuel transportation (short ton/million barrel capacity)	NO_x	14.95	88
	$PM_{2.5}$	4.02	12.6
	SO_2	8.51	76.3
	VOC	12.47	20.4
	NH_3	0.56	N/A

Table C.3. National-average air pollution damages for energy production processes using emissions data in U.S. EPA's NEI and the GREET model.

Energy Process	Unit	Data Source	U.S. EPA's NEI		GREET	
		Data Year	2011		Unknown	
		Marginal damage model	AP2	EASIUR	AP2	EASIUR
Coal mining and transportation	$_{2000}/short ton of coal produced	NO_x	$0.14	$0.08	$0.15	$0.08
		$PM_{2.5}$	$0.31	$0.44	$0.21	$0.29
		SO_2	$0.10	$0.05	$1.19	$0.54
		VOC	$0.01	N/A	$0.02	N/A
		NH_3	$0.01	$0.02	N/A	N/A
		Total	$0.57	$0.59	$1.57	$0.91
Crude oil extraction and transportation	$_{2000}/barrel-of-oil-equivalent (BOE) extracted	NO_x	$0.25	$0.13	$0.37	$0.19
		$PM_{2.5}$	$0.08	$0.10	$0.16	$0.21
		SO_2	$0.27	$0.15	$0.36	$0.20
		VOC	$0.08	N/A	$0.01	N/A
		NH_3	$0.01	$0.00	N/A	N/A
		Total	$0.69	$0.38	$0.91	$0.60
Natural gas extraction and transportation	$_{2000}/barrel-of-oil-equivalent (BOE) extracted	NO_x	$0.25	$0.13	$0.56	$0.28
		$PM_{2.5}$	$0.08	$0.10	$0.09	$0.11
		SO_2	$0.27	$0.15	$0.08	$0.04
		VOC	$0.08	N/A	$0.01	N/A
		NH_3	$0.01	$0.00	N/A	N/A
		Total	$0.69	$0.38	$0.73	$0.44
Oil refinery and petroleum products transportation	$_{2000}/barrel-of-oil-equivalent (BOE) refining capacity	NO_x	$0.06	$0.11	$0.37	$0.64
		$PM_{2.5}$	$0.20	$0.35	$0.61	$1.08
		SO_2	$0.33	$0.16	$2.92	$1.39
		VOC	$0.04	N/A	$0.07	N/A
		NH_3	$0.05	$0.02	N/A	N/A

		Total	$0.67	$0.63	$3.97	$3.11

Table C.4. Weighted-average marginal damages of criteria air pollutants (CAPs) weighted by the corresponding energy activity for each energy process.

Energy Process	Air Pollutant	AP2 model	EASIUR model
Coal mining and transportation ($\$_{2010}$/short ton)	NO_x	$3,412	$1,866
	$PM_{2.5}$	$25,713	$36,529
	SO_2	$40,456	$18,240
	VOC	$4,871	N/A
	NH_3	$20,474	$88,112
Oil extraction and transportation ($\$_{2010}$/short ton)	NO_x	$5,445	$2,787
	$PM_{2.5}$	$47,424	$59,629
	SO_2	$20,811	$11,849
	VOC	$3,025	N/A
	NH_3	$26,3481	$32,677
Natural gas extraction and transportation ($\$_{2010}$/short ton)	NO_x	$5,445	$2,787
	$PM_{2.5}$	$47,424	$59,629
	SO_2	$20,811	$11,849
	VOC	$3,025	N/A
	NH_3	$26,3481	$32,677
Oil refineries and petroleum product transportation ($\$_{2010}$/short ton)	NO_x	$4,173	$7,217
	$PM_{2.5}$	$48,800	$86,064
	SO_2	$38,286	$18,255
	VOC	$3,376	N/A
	NH_3	$88,745	$39,182

C.3.2 Electricity use in natural gas, crude oil, and petroleum product pipelines

Electricity is used to pump natural gas, crude oil, and petroleum in pipelines. But previous analysis using U.S. EPA's NEI or GREET model's emissions factors does not account for the air pollution damages from generating such electric power in their damage estimates. In this section, I calculate the air pollution damages of electricity used for pipelines based on the electricity energy intensity of pipelines (**Table C.5**) and the national-average air pollution damages of power electric generation in the U.S. (see *Section C.4.4 Grid electricity*). The electricity intensity of natural gas pipelines is 0.0486 MJ electricity consumed per MJ natural gas transported, based on the Transportation Energy Data Book.[3]

Table C.5. Electricity intensity for crude oil and petroleum product pipelines.

Variable	Crude oil	Petroleum Product	Reference
Energy intensity (percentage of energy input per unit of crude oil or petroleum products moved over 1000 miles)	0.7% per 1000 miles	0.8% per 1000 miles	Table S-3 in Hooker (1981)[203].
Average movement distance	687.2 miles	266.6 miles	Table 9.3 and Table 9.4 in Hooker (1981)[203]; and U.S. EIA[202].
Energy intensity (percentage of energy input per unit of crude oil or petroleum products moved)	0.48%	0.21%	Author's calculation (the first row times the second row).
Share of electricity in the input	0.52	0.56	Author's calculation based on Hooker (1981)[203].
Electricity intensity (percentage of electricity input per unit of crude oil or petroleum products moved)	0.25%	0.12%	Author's calculation (the third row times the fourth row).

C.3.3 Air pollution damages for energy production processes

Finally I calculate air pollution damages of coal, natural gas, and crude oil using the following formulas.

$$\begin{aligned}&\text{Air pollution } damages\ per\ 1\ MJ\ of\ crude\ oil\ produced\\&= \frac{marginal\ damage\ estimates\ for\ crude\ oil\ production\ and\ transportation}{energy\ intensity\ of\ crude\ oil}\\&+ air\ pollution\ damages\ from\ electricity\ use\ for\ crude\ oil\ pipelines.\end{aligned} \tag{C.4}$$

$$\begin{aligned}&\text{Air pollution } damages\ per\ 1\ MJ\ of\ natural\ gas\ produced\\&= \frac{marginal\ damage\ estimates\ for\ natural\ gas\ production\ and\ transportation}{energy\ intensity\ of\ natural\ gas}\\&+ air\ pollution\ damages\ from\ electricity\ use\ for\ natural\ gas\ pipelines.\end{aligned} \tag{C.5}$$

$$\begin{aligned}&\text{Air pollution } damages\ per\ 1\ MJ\ of\ coal\ produced\\&= \frac{marginal\ damage\ estimates\ for\ coal\ production\ and\ transportation}{energy\ intensity\ of\ coal}\end{aligned} \tag{C.6}$$

Where marginal damage estimates for energy processes come from **Table C.3**, and the air pollution damages from electricity use for crude oil and natural gas pipelines are calculated by multiplying the air pollution damages per unit of electricity consumption with electricity intensity of pipelines. Finally, **Table C.6** summarizes the estimated air pollution damages with energy production processes.

Table C.6. Air pollution damages for energy production processes. Unit: $cent_{2010}$/MJ.

Emissions data	**U.S. EPA's NEI**		**GREET**	
Marginal damage model	**AP2**	**EASIUR**	**AP2**	**EASIUR**
Coal production	0.003	0.003	0.008	0.004
Crude oil production	0.013	0.008	0.017	0.011
Natural gas production	0.012	0.006	0.012	0.007

C.4. Fuel Pathway Assumptions and Air Pollution Damages

C.4.1 Petroleum fuels

The emissions data in Jaramillo & Muller (2016)[193] on oil refineries are in the unit of refining capacity (barrel of crude oil inputs), so a conversion from refining capacity to the actual petroleum fuel output (per MJ petroleum fuels produced) is needed.

The U.S. Energy Information Administration (EIA) reports the refining capacity (atmospheric crude oil distillation capacity) in the Refinery Capacity Report[205]. In year 2011, the refining capacity was 17.7 million barrels per day while the total crude oil input is 5,422,000 thousand barrels[202]. Thus the utilization rate of the refining capacity was 84%.

I use energy allocation[204] to convert the input unit, barrel of crude oil, to the output unit, MJ of petroleum fuel products. U.S. EIA reports that in 2013 one barrel crude oil yielded about 45 gallons of petroleum products because of refinery processing gain.[375] **Table C.7** shows the breakdowns and energy intensity of refinery outputs (petroleum products) from one barrel of crude oil input. Overall, one barrel of crude oil produces 5.9 million BTUs of petroleum outputs. Because I used the energy allocation method, air pollution damages from oil refining per 1 MJ of gasoline or diesel are the same. Finally, air pollution damages from oil refinery and petroleum product transportation are calculated using Eqn. C.7.

$$Air\ pollution\ damages\ per\ 1\,MJ\ of\ petroleum\ products\ produced$$
$$= \frac{Marginal\ damage\ estimates\ for\ oil\ refinery}{5.9\ [million\ BTU / 1\,BOE\ refining\ capacity]} \times \frac{unit\ conversion\ between\ MJ\ and\ million\ BTU}{refining\ capacity\ utilization\ rate}$$
$$+\ air\ pollution\ damages\ from\ electricity\ use\ for\ petroleum\ products\ pipelines. \qquad (C.7)$$

Table C.7. Energy allocations of refinery outputs (petroleum products) from one barrel of crude oil input.

Oil refinery	Petroleum products	Volume	Heat content
		Gallon	Million Btu/barrel
Input	Crude oil (1 barrel)	42	5.80
Output	Gasoline	19	5.301
	Diesel/Ultra-low sulfur distillate	11	5.774
	Jet fuel	4	5.670
	Hydrocarbon gas liquids*	2	3.968
	Heavy fuel oil (residual oil)	1	6.287
	Other distillates (heating oil)	1	5.817
	Other products	7	5.825
	Total	45	5.90

** Note that hydrocarbon gas liquids are previously referred as liquefied petroleum gases (LPG) by U.S. EIA. ** Heat convent is taken from U.S. EIA, http://www.eia.gov/totalenergy/data/monthly/pdf/sec13_1.pdf*

C.4.2 CNG

Following Tong et al.,[146] I assume that compression of natural gas takes place at refueling stations. In this study, I further assume that compression of natural gas takes place in the same county where the CNG vehicle is used. The energy efficiency of the compression process is 96%.[146] In addition to natural gas as the input, and CNG as the output, additional energy input is grid electricity. I calculate the air pollution damages from the compression process by multiplying the electricity consumption per 1 MJ of CNG produced with the air pollution damages of the grid electricity in the county.

C.4.3 LNG

Tong et al.[146] considered two LNG pathways, one where liquefaction process takes places at a centralized plant and one where liquefaction process takes place at refueling stations. In this study, I only consider the distributed LNG pathway and assume that liquefaction process takes place in the same county where the LNG vehicle is used. The energy efficiency of the liquefaction process is 90%.[146] In addition to natural gas as the input, and LNG as the output, additional energy input is grid electricity. I calculate the air pollution damages from the liquefaction process by multiplying the electricity consumption per 1 MJ of LNG produced with the air pollution damages of the grid electricity in the county.

C.4.4 Grid electricity

I use annual emissions from U.S. electricity grid in 2014 to characterize emissions associated with electricity charged by battery electric vehicles (BEVs). Previous studies have analyzed the differences between marginal emissions and average emissions as well as impacts of charging on grid operations.[192,272,376] In this study, I aim to compare electric pathways with petroleum and natural gas pathways across U.S. counties, so an average model of grid electricity is sufficient. I assume that the grid electricity is balanced in each NERC region. While the actual balancing area is arguably smaller than the scope of a NERC region, issues such as data availability and model capacities prohibits modeling emissions factors at the level of each actual balancing area in this study. Section A.1 discusses how to determine the primary NERC region for any U.S. county.

U.S. EPA produces the eGRID data set which provides consistent emissions and generation data but the most recent eGRID data is only available for year 2012.[206] To better reflect grid mix changes due to low natural gas prices and expansions of wind and solar capacity, I use the U.S. EPA's Continuous Emission Monitoring System (CEMS)[196] to get emissions data from fossil fuel power plants and U.S. EIA Form-923[197] to get electricity generation and fuel consumption data for all power plants for year 2014.

I find some consistency issues between U.S. EIA Form-923 data and U.S. EPA's CEMS data for year 2014 (**Table C.8**). In particular, net generations by coal-fired, natural gas-fired, and oil-fired power plants differ between the two data sources.

Table C.8. Net electricity generation from fossil fuel power plants – U.S. EIA's Form-923 data and U.S. EPA's CEMS data. Unit: MWh/year.

NERC region	U.S.EIA Form-923			U.S. EPA CEMS		
	Coal	Natural gas	Oil	Coal	Natural gas	Oil
FRCC	4.8E+07	1.4E+08	2.7E+05	4.9E+07	1.2E+08	7.0E+06
MRO	1.4E+08	8.8E+06	2.4E+05	1.5E+08	7.6E+06	1.5E+04
NPCC	9.6E+06	1.0E+08	1.2E+06	1.1E+07	1.0E+08	9.7E+05
RFC	4.7E+08	1.5E+08	2.0E+06	4.9E+08	1.3E+08	6.1E+05
SERC	4.6E+08	2.8E+08	2.1E+06	5.1E+08	2.5E+08	2.7E+04
SPP	1.2E+08	5.9E+07	1.3E+05	1.3E+08	4.5E+07	2.8E+03
TRE	1.2E+08	1.7E+08	8.3E+04	1.3E+08	1.5E+08	0.0E+00
WECC	2.0E+08	2.2E+08	2.9E+05	2.2E+08	2.0E+08	0.0E+00
Contiguous U.S.	1.6E+09	1.1E+09	6.3E+06	1.7E+09	1.0E+09	8.6E+06

Upon further research, I found that the U.S. EIA Form-923 data matches better with the net generation in 2014, reported by U.S. EIA's Electric Power Monthly[377]. As a result, I perform the following adjustments on the coal-fired and natural gas-fired power plants in the CEMS data so that the net power generation by coal and by natural gas in each NERC region from the adjusted CEMS data are the same as those calculated using U.S. EIA Form-923 data. The emissions of coal-fired and natural gas-fired power plants in the CEMS data are scaled up or down according to changes in net generation.

$$Adjustment\ factor_{NERC\ region\ r,\ fuel\ type\ f} = \frac{net\ generation\ in\ EIA\ Form\text{-}923_{r,f}}{net\ generation\ in\ EPA\ CEMS_{r,f}}$$

(C.5)

Where fuel type equals coal for natural gas. For any power plant in the CEMS data, the annual net power generation, NO_x emissions, SO_2 emissions, and $PM_{2.5}$ emissions of the power plant are then multiplied by the adjustment factor (matching the fuel type and the NERC region of the power plant) to get the adjusted data.

$$data\ j_{Power\ plant\ i, NERC\ region\ r, fuel\ type\ f,\ adjusted} = data\ j_{i,r,f,original\ CEMS} \times Adjustment\ factor_{r,f} \quad (C.6)$$

Where fuel type equals coal or natural gas, i is the power plant index, and data j refers to net generations, NO_x emissions, SO_2 emissions, or $PM_{2.5}$ emissions. Finally, I combine the adjusted CEMS data (that includes coal-fired, natural gas-fired, oil-fired and biomass-fired power plants) with the net generation data of nuclear and renewables sources in U.S. EIA Form-923 data to form the data set I used in this study (referred as "adjusted CEMS + EIA Form-923" hereafter). To provide an idea of the adjusted CEMS + EIA Form-923, **Table C.9** reports the net power generation by energy source and by NERC region. **Table C.10** reports the total and normalized CAP emissions by NERC region.

Table C.9. Net electricity generation by NERC region in 2014 (Adjusted CEMS + EIA Form-923). Unit: MWh/year.

NERC region	Coal	Gas	Oil	Biomass	Nuclear & Renewables	Total
FRCC	4.8E+07	1.4E+08	7.0E+06	6.5E+05	3.1E+07	2.2E+08
MRO	1.4E+08	8.8E+06	1.5E+04	4.7E+05	7.8E+07	2.2E+08
NPCC	9.6E+06	1.0E+08	9.7E+05	5.9E+05	1.2E+08	2.4E+08
RFC	4.7E+08	1.5E+08	6.1E+05	3.7E+05	3.1E+08	9.2E+08
SERC	4.6E+08	2.8E+08	2.7E+04	5.5E+05	3.3E+08	1.1E+09
SPP	1.2E+08	5.9E+07	2.8E+03	0.0E+00	4.0E+07	2.2E+08
TRE	1.2E+08	1.6E+08	0.0E+00	1.9E+05	7.7E+07	3.6E+08
WECC	2.0E+08	2.2E+08	0.0E+00	0.0E+00	3.0E+08	7.3E+08
Contiguous U.S.	1.6E+09	1.1E+09	8.6E+06	2.8E+06	1.3E+09	4.0E+09

Table C.10. Total and normalized CAP emissions by NERC region in 2014 (Adjusted CEMS + EIA Form-923).

Metric	Total emissions (short ton/year)			Normalized emissions (lb./MWh)		
Air pollutant	**$PM_{2.5}$**	**SO_2**	**NO_x**	**$PM_{2.5}$**	**SO_2**	**NO_x**
FRCC	2.0E+04	8.5E+04	5.7E+04	0.18	0.77	0.51
MRO	1.4E+04	2.3E+05	1.4E+05	0.13	2.08	1.27
NPCC	7.6E+03	2.4E+04	2.4E+04	0.06	0.20	0.20
RFC	7.2E+04	1.1E+06	4.6E+05	0.16	2.41	1.00
SERC	6.3E+04	8.0E+05	4.0E+05	0.12	1.50	0.74
SPP	1.1E+04	2.0E+05	1.1E+05	0.10	1.85	1.04
TRE	1.6E+04	2.7E+05	8.9E+04	0.09	1.48	0.49
WECC	3.0E+04	1.4E+05	2.6E+05	0.08	0.39	0.70
Contiguous U.S.	2.3E+05	2.9E+06	1.5E+06	0.12	1.44	0.77

I estimate the air pollution damages from power generation by multiplying the CAP emissions from the individual power plants in the adjusted CEMS + EIA Form-923 data with the corresponding marginal damages of CAPs in the same county where the power plant is located. I then calculate the total air pollution damages by NERC region by adding up air pollution damages of all power plants in the NERC region. Finally, I calculate the normalized air pollution damages in each NERC region by dividing the total damages with the total electricity generation. **Table C.11** and **Table C.12** report the total and normalized air pollution damages from electricity generation using the EASIUR model and the AP2 model.

Table C.11. Air pollution damages from electricity generation in 2014, based on the marginal damage estimates from the EASIUR model. Only on-site CAP emissions are included.

Metric	Total air pollution damages ($\$_{2010}$/year)				Normalized damages ($\$_{2010}$/MWh)			
Air pollutant	**$PM_{2.5}$**	**SO_2**	**NO_x**	**All**	**$PM_{2.5}$**	**SO_2**	**NO_x**	**All**
FRCC	1.2E+09	1.2E+09	1.8E+08	2.5E+09	5.4	5.2	0.8	11.4
MRO	7.0E+08	3.8E+09	9.3E+08	5.5E+09	3.1	17.2	4.2	24.4
NPCC	1.3E+09	6.0E+08	4.2E+08	2.3E+09	5.6	2.6	1.8	10.0
RFC	6.8E+09	2.4E+10	3.9E+09	3.4E+10	7.3	25.7	4.3	37.3
SERC	3.8E+09	1.3E+10	1.6E+09	1.8E+10	3.5	12.0	1.5	17.0
SPP	4.3E+08	2.7E+09	4.1E+08	3.5E+09	2.0	12.1	1.9	15.9
TRE	7.2E+08	3.4E+09	2.3E+08	4.4E+09	2.0	9.4	0.6	12.0
WECC	7.9E+08	1.2E+09	3.7E+08	2.3E+09	1.1	1.6	0.5	3.2
Contiguous U.S.	1.6E+10	4.9E+10	8.1E+09	7.3E+10	3.9	12.4	2.0	18.3

Table C.12. Air pollution damages from electricity generation in 2014, based on the marginal damage estimates from the AP2 model. Only on-site CAP emissions are included.

Metric	Total air pollution damages ($\$_{2010}$/year)				Normalized damages ($\$_{2010}$/MWh)			
Air pollutant	**$PM_{2.5}$**	**SO_2**	**NO_x**	**All**	**$PM_{2.5}$**	**SO_2**	**NO_x**	**All**
FRCC	7.1E+08	2.1E+09	1.1E+08	2.9E+09	3.2	9.3	0.5	13.0
MRO	2.9E+08	5.6E+09	8.7E+08	6.8E+09	1.3	25.2	3.9	30.4
NPCC	9.7E+08	6.6E+08	2.3E+07	1.7E+09	4.1	2.8	0.1	7.0
RFC	4.1E+09	4.6E+10	1.7E+09	5.2E+10	4.4	50.2	1.9	56.4
SERC	2.2E+09	2.4E+10	1.7E+09	2.8E+10	2.1	22.7	1.6	26.3
SPP	2.0E+08	3.5E+09	6.6E+08	4.4E+09	0.9	15.9	3.0	19.8
TRE	3.5E+08	5.3E+09	4.5E+08	6.1E+09	1.0	14.5	1.2	16.7
WECC	6.8E+08	2.2E+09	1.1E+09	4.0E+09	0.9	3.0	1.5	5.5

Contiguous U.S.	9.4E+09	9.0E+10	6.6E+09	1.1E+11	2.4	22.5	1.7	26.5

In addition, I calculate the air pollution damages associated with extraction and transportation of fossil fuels to the power plants. **Table C.13** reports the weighted-average energy efficiency for coal-fired, natural gas-fired, and oil-fired power plants in each NERC region from U.S. EIA's Form-923. The shares of the coal-fired, natural-fired, and oil-fired electricity generation are already available in **Table C.9**. **Table C.6** shows the estimated air pollution damages associated with extraction and transportation of fossil fuels. Finally, I consider the electricity losses over transmission and distribution lines, based on the eGRID data (for year 2012)[206] (**Table C.13**).

Table C.13. Weighted-average energy efficiency for coal-fired, natural gas-fired, and oil-fired power plants by NERC region.

NERC region	Weighted-average energy efficiency (output energy/input energy)			Electricity losses over transmission and distribution lines[206]
	Coal	Natural gas	Oil	
FRCC	33%	45%	33%	9.17%
MRO	32%	40%	26%	
NPCC	33%	42%	38%	
RFC	33%	42%	30%	
SERC	33%	44%	30%	
SPP	32%	41%	27%	7.03%
TRE	33%	46%	29%	5.76%
WECC	33%	45%	31%	8.33%
Contiguous U.S.	33%	44%	31%	-

Finally, I report the air pollution damages from the life cycle of grid electricity in **Table C.14**. Note that these air pollution damages are likely to underestimate the actual air pollution damages from grid electricity because VOC and CO emissions from the U.S. power plant fleet are not accounted due to data availability. However, power plants are not primary emission sources for VOC and CO in the U.S., so the bias should be limited.

Table C.14. Life cycle air pollution damages from electricity generation in the Contiguous U.S. in 2014. Unit: $\$_{2010}$/MWh electricity delivered to end use.

NERC region	EASIUR model			AP2 model		
	Upstream	On-site combustion	Total	Upstream	On-site combustion	Total

FRCC	0.5	12.5	13.0	0.8	14.3	15.1
MRO	0.2	26.9	27.1	0.3	33.5	33.7
NPCC	0.3	11.0	11.3	0.5	7.7	8.2
RFC	0.3	41.1	41.3	0.3	62.1	62.4
SERC	0.3	18.8	19.1	0.4	28.9	29.4
SPP	0.4	17.5	17.9	0.5	21.8	22.3
TRE	0.4	12.9	13.3	0.6	17.9	18.5
WECC	0.3	3.4	3.6	0.4	5.8	6.2
Contiguous U.S.	0.3	20.0	20.3	0.4	28.9	29.4

C.4.5 Natural gas-based electricity

In this study, I consider a hypothetical scenario in which natural gas-based electricity replaces electricity generated by all existing coal-fired power plants in the adjusted CEMS + EIA Form-923 data. Specifically, for each coal-fired power plant in the adjusted CEMS + EIA Form-923 data, I assume a new natural gas combined cycle (NGCC) power plant is built in the same county and produces the same net generation as the existing coal-fired power plant. I calculate the total and normalized air pollution damages from the 'new' electricity grid using the same method as for the current U.S. grid. I assume that all NGCC power plants have the same emission factors no matter how much electricity is generated (**Table C.15**).[158]

Table C.15. CAP emission factors of a new NGCC power plant. Unit: gram/MWh.

Sources	**$PM_{2.5}$**	**SO_x**	**NO_x**	**NH_3**	**VOC**	**CO**
NETL (2013)[158]	0	0	27	0	0	0

Table C.16 reports the estimated life cycle air pollution damages from the increased natural gas electricity. Several interesting findings emerge when I compare air pollution damages from the current grid and from the increased natural gas-based electricity. First, I find that several NERC regions (MRO, RFC, SERC, SPP, and TRE) see an order-of-magnitude smaller air pollution damages from the increased natural gas-based electricity than the current electricity grid. In particular, the RFC region and the MRO region, both of which have a large share of coal-fired power plants, find significantly larger reductions in air pollution damages if all existing coal-fired power plants are replaced by new NGCC power plants. On the other hand, air pollution damages from electricity in the NPCC region only reduce 20-30% (depending on which marginal damage model is used) when all coal-fired power plants are replaced by new NGCC power plants. This suggests that only renewable electricity sources can significantly low the air pollution damages in this region. Finally, I find that in the increased natural gas electricity scenario, air pollution damages from upstream activities (extraction and transportation of fossil fuels) are comparable or even larger than air pollution damages with on-site combustion in several NERC regions (MRO, SPP, TRE, and WECC). This suggests the importance to estimate the life cycle damages when the grid electricity becomes cleaner in the future.

Table C.16. Life cycle air pollution damages from the hypothetical scenario of increased natural gas-derived electricity in the Contiguous U.S. Unit: $\$_{2010}$/MWh electricity delivered to end use.

NERC region	EASIUR model			AP2 model		
	Upstream	On-site combustion	Total	Upstream	On-site combustion	Total
FRCC	0.4	5.4	5.8	0.8	3.8	4.6
MRO	0.3	0.6	0.9	0.6	0.3	0.9
NPCC	0.2	8.2	8.5	0.4	5.6	6.0
RFC	0.3	2.8	3.2	0.6	1.9	2.5
SERC	0.4	2.1	2.5	0.6	1.7	2.3
SPP	0.4	0.8	1.2	0.7	0.9	1.6
TRE	0.4	1.1	1.5	0.7	0.7	1.4
WECC	0.3	0.9	1.1	0.5	0.9	1.4
Contiguous U.S.	0.3	2.3	2.7	0.6	1.7	2.3

C.4.6 Air pollution damages for fuel pathways

I summarize the life cycle (well-to-pump) air pollution damages of petroleum fuels, CNG, and LNG in **Table C.17**. Life cycle (well-to-socket) air pollution damages of grid electricity and increased penetrations of natural gas-based electricity are in **Table C.14** and **Table C.17**.

Table C.17. Life cycle (well-to-pump) air pollution damages of petroleum fuels, CNG, and LNG. Unit: $cent_{2010}$/MJ. The table reports the damages using both emissions data and both marginal damage models.

Emissions data	Marginal damage model	EASIUR model			AP2 model		
	NERC region	Petroleum fuels	CNG	LNG	Petroleum fuels	CNG	LNG
	FRCC	0.020	0.021	0.045	0.027	0.028	0.056

U.S. EPA's NEI	MRO		0.038	0.090		0.050	0.115
	NPCC		0.019	0.040		0.021	0.036
	RFC		0.054	0.133		0.084	0.203
	SERC		0.028	0.064		0.045	0.101
	SPP		0.027	0.061		0.037	0.079
	TRE		0.021	0.046		0.032	0.067
	WECC		0.010	0.017		0.018	0.030
	Contiguous U.S.		-	-		-	-
GREET model	FRCC	0.071	0.022	0.046	0.094	0.029	0.056
	MRO		0.039	0.090		0.051	0.116
	NPCC		0.020	0.041		0.021	0.036
	RFC		0.055	0.134		0.084	0.204
	SERC		0.029	0.065		0.046	0.102
	SPP		0.028	0.061		0.038	0.080
	TRE		0.022	0.047		0.033	0.068
	WECC		0.011	0.018		0.019	0.030
	Contiguous U.S.		-	-		-	-

C.5. Vehicle Assumptions

C.5.1 Fuel economy

This study models new vehicles instead of existing vehicle fleets. I use the same fuel economy assumptions (**Table C.18**) as reported in Tong et al.,[146,198] since these papers provide the most comprehensive assumptions of new vehicles for vehicle types considered in this analysis. Using the same fuel economy also facilitates the comparison of estimated climate change damages in Tong et al.[146,198] and the health and environmental damages reported in this study.

Table C.18. Fuel economy assumptions. Baseline vehicle technologies are highlighted in grey. Acronyms are explained in the main text.

Vehicle type	**Passenger car**	**SUV**	**Transit bus**	**Tractor-trailer**	
				local-haul	**long-haul**
Unit	MPGGE (mile per gallon gasoline equivalent)		MPGDE (mile per gallon diesel equivalent)		
Gasoline (SI-ICEV)	33	25	-	-	-
Diesel (CI-ICEV)	-	-	4.0	4.3	6.5
Gasoline-HEV(SI-ICEV)	45	33	-	-	-
Diesel-HEV (CI-ICEV)	-	-	4.8	5.2	7.2
CNG (SI-ICEV)	31*	23.5*	3.6	3.9	5.9

LNG (SI-ICEV)	-	-	3.6	3.9	5.9
LNG (HPDI-ICEV)	-	-	-	4.2	6.4
BEV	110	76	16.8	-	-

*Note: * Currently there are no CNG-dedicated passenger vehicles or SUVs in the market. For CNG-dedicated passenger vehicles, I use the fuel economy test results from the Honda Civic CNG although its production stopped in 2015. For CNG-dedicated SUVs, I assume that its relative fuel economy to a gasoline SUV is the same as the relative fuel economy of a CNG passenger car to a gasoline passenger car.*

C.5.2 Battery manufacturing air pollution damages and vehicle use

In this study, I assume that the air pollution damages for different vehicle technologies for a specific vehicle type are the same except the additional lithium-ion batteries in hybrid electric vehicles (HEVs) and battery electric vehicles (BEVs). I assume the air pollution damages of battery manufacturing in the U.S. are $8.68/kWh of battery capacity following Tessum et al. (2014).[32] As discussed in Tessum et al. (2014), their estimates on air pollution damages from battery production are lower than Michalek et al. (2011).[26] I use the same assumptions on vehicle uses (life cycle mileage), as well as battery sizes and battery replacements for HEVs and BEVs as in Tong et al.[146,198] (**Table C.19**).

Table C.19. Assumptions on vehicle use, battery size battery size as well as battery replacements over life time for HEVs and BEVs.

Vehicle type	Technology	Lifetime mileage	Battery size	Numbers of batteries per vehicle lifetime
Passenger car	HEV	150,000 miles	1.3 kWh	1
	BEV		24 kWh	
SUV	HEV		1.6 kWh	1
	BEV		41.8 kWh	
Transit bus	HEV	500,000 miles	6 kWh	1
	BEV		323 kWh	1.5
Local-haul tractor-trailer	HEV	240,000 miles	15 kWh	1
Long-haul tractor-trailer	HEV	480,000 miles	5 kWh	1

C.5.3 CAP emissions from vehicle use

Table C.20-Table C.22 report the assumptions on the CAP emissions from vehicle use. While most CAP emissions occur through vehicle tailpipes, a fraction of $PM_{2.5}$ emissions are created through tire and break wear. I rely on the GREET model[159] (version 2015, vehicle model year 2015) for vehicle use emissions for passenger cars and SUVs since there is no better data. For transit buses, the Altoona Bus Research & Testing Center (ABRTC) test the tailpipe CO, NO_x, and total hydrocarbon (THC) emissions for three drive cycles (Manhattan, Orange County, and UDDS) on new buses before they are offered on the market. I use the ABRTC test results for three vehicle technologies, conventional diesel buses (New Flyer XD40, XD60), diesel HEV (New Flyer XDE40, XDE60), and CNG buses (New Flyer XN40 and XN60).[207] I use the test results from the same vehicle manufacturer to eliminate potential biases from vehicle designs and manufacturing. All these buses comply with the U.S. EPA's 2010 emissions standards for heavy-duty engines. For each vehicle technology (conventional diesel, diesel HEV, and CNG), I use the average emissions factors in the Manhattan Cycle and Orange County Cycle for the 40-foot and the 60-foot buses. I calculate the VOC emissions from the THC emissions by using the conversion factors suggested by U.S. EPA[378]. I rely on the GREET model[159] (version 2015, vehicle model year 2015) for $PM_{2.5}$ and SO_2 emissions.

For tractor-trailers, Thiruvengadam et al.[208] reported the chassis dynamometer emission testing of diesel trucks with a diesel particulate filter (DPF) and selective catalytic reduction (SCR), diesel trucks with DPF and without SCR, natural gas trucks (stoichiometric natural gas engine with a three-way catalyst (TWC)), and LNG HPDI (high-pressure direct ignition) trucks with advanced exhaust gas recirculation (EGR), DPF, and SCR. All of these vehicles comply with the U.S. EPA's 2010 emissions standards for heavy-duty engines. I use the mean emission factors of NO_x CO, and PM from the local drive cycle and the regional drive cycle for the local-haul and long-haul tractor-trailers. I rely on the GREET model[159] (version 2015, vehicle model year 2015) for SO_2 and CO emissions. Neither Thiruvengadam et al.[208] or the GREET model considered the diesel hybrid-electric truck, so I assume that diesel HEVs and diesel trucks with DPF and SCR have the same vehicle use emissions.

It is worth mentioning that the GREET model[159] also reported vehicle operation emissions for transit buses and tractor-trailers (**Table C.23** and **Table C.24**). I find that the assumptions in this study and GREET's assumptions are largely in agreement with each other. The notable differences for transit buses are that (1) this study assumes higher NO_x emissions from diesel

HEVs than conventional diesel buses whereas the GREET model assumes lower NO_x emissions from diesel HEVs; (2) this study assumes almost 50% higher CO emissions than in the GREET model. For tractor-trailers, the only notable differences are relatively higher CO emissions (compared to diesel) from my assumptions than the GREET model.

Table C.20. Vehicle use emissions from for passenger cars and SUVs. Unit: gram/mile. Source: GREET model (version 2015, vehicle model year 2015)[159].

Air Pollutant	Passenger car				SUV			
	Gasoline	Gasoline HEV	CNGV	BEV	Gasoline	Gasoline HEV	CNGV	BEV
$PM_{2.5}$	0.0093	0.0093	0.0093	0.0046	0.0136	0.0136	0.0136	0.0073
SO_2	0.0018	0.0013	0.0010	0.0000	0.0056	0.0042	0.0014	0.0000
NO_x	0.1105	0.0928	0.1105	0.0000	0.1728	0.1451	0.1728	0.0000
VOC	0.2192	0.1652	0.1682	0.0000	0.2256	0.1680	0.1754	0.0000
CO	2.5679	2.5679	2.5679	0.0000	3.1996	3.1996	3.1996	0.0000

Table C.21. Vehicle use emissions from transit buses. Unit: gram/mile. Source: ABRTC[207] and GREET model (version 2015, vehicle model year 2015)[159].

Air Pollutant	Diesel	Diesel HEV	CNGV	CNGV (low NO_x engine)	BEV
$PM_{2.5}$	0.0335	0.0335	0.0335	0.0335	0.0124
SO_2	0.0160	0.0114	0.0093	0.0093	0.0000
NO_x	0.9175	1.4450	0.5775	0.0918	0.0000
VOC	0.1121	0.0787	0.0695	0.0695	0.0210
CO	0.4900	0.1850	31.2750	31.2750	0.0000

Table C.22. Vehicle use emissions from passenger cars and SUVs. Unit: gram/mile. Source: Thiruvengadam, et al.[208] and GREET model (version 2015, vehicle model year 2015)[159].

Air Pollutant	Diesel DPF	Diesel DPF/SCR	Diesel HEV	CNGV	LNGV	LNG HPDI
Local-haul tractor-trailer						
$PM_{2.5}$	0.0258	0.0238	0.0238	0.0218	0.0218	0.0228
SO_2	0.0090	0.0090	0.0090	0.0050	0.0000	0.0000
NO_x	5.4900	6.2600	6.2600	0.3200	0.3200	0.6520
VOC	0.1031	0.1031	0.1031	0.1031	0.1031	0.1031
CO	2.7500	0.3230	0.3230	7.5800	7.5800	0.0490
Long-haul tractor-trailer						

$PM_{2.5}$	0.0252	0.0240	0.0240	0.0226	0.0226	0.0215
SO_2	0.0092	0.0092	0.0092	0.0050	0.0000	0.0000
NO_x	3.8900	2.1500	2.1500	0.1720	0.1720	0.4680
VOC	0.4128	0.4128	0.4128	0.4128	0.4128	0.4128
CO	1.2700	0.1000	0.1000	8.4500	8.4500	0.0600

Table C.23. Vehicle use emissions of transit buses in GREET model (version 2015, vehicle model year 2015)[159]. Unit: gram/mile. Note that this table is for comparison only and is not used in damage estimates.

Air Pollutant	Diesel	Diesel HEV	CNGV	LNGV	BEV
$PM_{2.5}$	0.0335	0.0335	0.0335	0.0335	0.0124
SO_2	0.0160	0.0115	0.0093	0.0000	0.0000
NO_x	1.1669	1.1669	0.5834	0.5834	0.0000
VOC	0.0919	0.0919	0.0919	0.0919	0.0437
CO	0.5158	0.2579	23.0000	23.0000	0.0000

Table C.24. Vehicle use emissions of local-haul and long-haul tractor-trailers in GREET model (version 2015, vehicle model year 2015)[159]. Unit: gram/mile. Note that this table is for comparison only and is not used in damage estimates.

Air Pollutant	Local-haul tractor-trailer				Long-haul tractor-trailer			
	Diesel	CNGV	LNGV	LNG HPDI	Diesel	CNGV	LNGV	LNG HPDI
$PM_{2.5}$	0.0443	0.0443	0.0443	0.0417	0.0571	0.0571	0.0571	0.0534
SO_2	0.0090	0.0050	0.0000	0.0000	0.0092	0.0050	0.0000	0.0000
NO_x	1.2806	0.6403	0.6403	0.6403	3.4855	1.7427	1.7427	1.7427
VOC	0.1031	0.1031	0.1031	0.1031	0.4128	0.4128	0.4128	0.4128
CO	0.5213	8.0000	8.0000	0.2606	1.4241	23.0000	23.0000	0.7120

C.6. Additional Results

C.6.1 Upstream emissions data from U.S. EPA's NEI

This sections shows additional results using the emissions data from U.S. EPA's NEI,[194] as summarized in **Table C.25**.

Table C.25. A summary of additional results using emissions data from U.S. EPA's NEI.[194]

<table>
<tr><th colspan="2">Graphics topics</th><th>Boxplots on relative reductions*</th><th>Percentage of counties that see damage reductions*</th><th>“Best”/ “Worst” technology</th><th>-</th><th>-</th></tr>
<tr><td rowspan="2">Marginal damage model</td><td>EASIUR</td><td>Figure 4.1</td><td rowspan="2">Table C.26</td><td>Figure 4.2</td><td>-</td><td>-</td></tr>
<tr><td>AP2</td><td>Figure C.3</td><td>Figure C.4</td><td>-</td><td>-</td></tr>
<tr><td colspan="2">Graphics topics (vehicle types)</td><td>Passenger car</td><td>SUV</td><td>Transit bus</td><td>Local-haul tractor-trailer</td><td>Long-haul tractor-trailer</td></tr>
<tr><td rowspan="2">Metrics*</td><td>Relative reductions</td><td>Figure C.5</td><td>Figure C.6</td><td>Figure C.7</td><td>Figure C.8</td><td>Figure C.9</td></tr>
<tr><td>Absolute reductions</td><td>Figure C.10</td><td>Figure C.11</td><td>Figure C.12</td><td>Figure C.13</td><td>Figure C.14</td></tr>
</table>

** when replacing the baseline petroleum pathways with alternative fuel pathways*

Table C.26. Percentage of counties that see damage reductions from petroleum and natural gas pathways compared to the baseline petroleum fuels (conventional gasoline for passenger cars and SUVs; conventional diesel for transit buses and tractor-trailers) using emissions data from U.S. EPA's NEI. Damage of a pathway is calculated based on CAP emissions from life cycle or the vehicle operation stage, using the AP2 or the EASIUR model.

Method	Boundary	Gasoline HEV	CNG	NGCC + BEV	Grid + BEV	-
Passenger cars						
AP2	Life Cycle	100%	22%	100%	19%	-
	Vehicle Operation	100%	100%	100%	100%	-
EASIUR	Life Cycle	100%	18%	100%	17%	-
	Vehicle Operation	100%	100%	100%	100%	-
SUVs						
AP2	Life Cycle	100%	22%	99%	14%	-
	Vehicle Operation	100%	100%	100%	100%	-
EASIUR	Life Cycle	100%	18%	99%	15%	-
	Vehicle Operation	100%	100%	100%	100%	-
Method	**Boundary**	**Diesel HEV**	**CNG**	**CNG (low-NO_x engine)**	**NGCC + BEV**	**Grid + BEV**
Transit buses						
AP2	Life Cycle	25%	0%	1%	93%	1%
	Vehicle Operation	4%	0%	1%	100%	100%
EASIUR	Life Cycle	12%	0%	1%	75%	3%
	Vehicle Operation	0%	0%	1%	100%	100%
Method	**Boundary**	**Diesel (DPF)**	**Diesel HEV**	**CNG-SI**	**LNG-SI**	**LNG-HPDI**
Local-haul tractor-trailers						
AP2	Life Cycle	89%	100%	91%	57%	76%
	Vehicle Operation	89%	0%	98%	98%	100%
EASIUR	Life Cycle	87%	100%	99%	88%	96%
	Vehicle Operation	87%	0%	98%	98%	100%
Line-haul tractor-trailers						
AP2	Life Cycle	0%	100%	51%	14%	32%
	Vehicle Operation	0%	0%	81%	82%	100%
EASIUR	Life Cycle	0%	100%	71%	26%	68%
	Vehicle Operation	0%	0%	83%	84%	100%

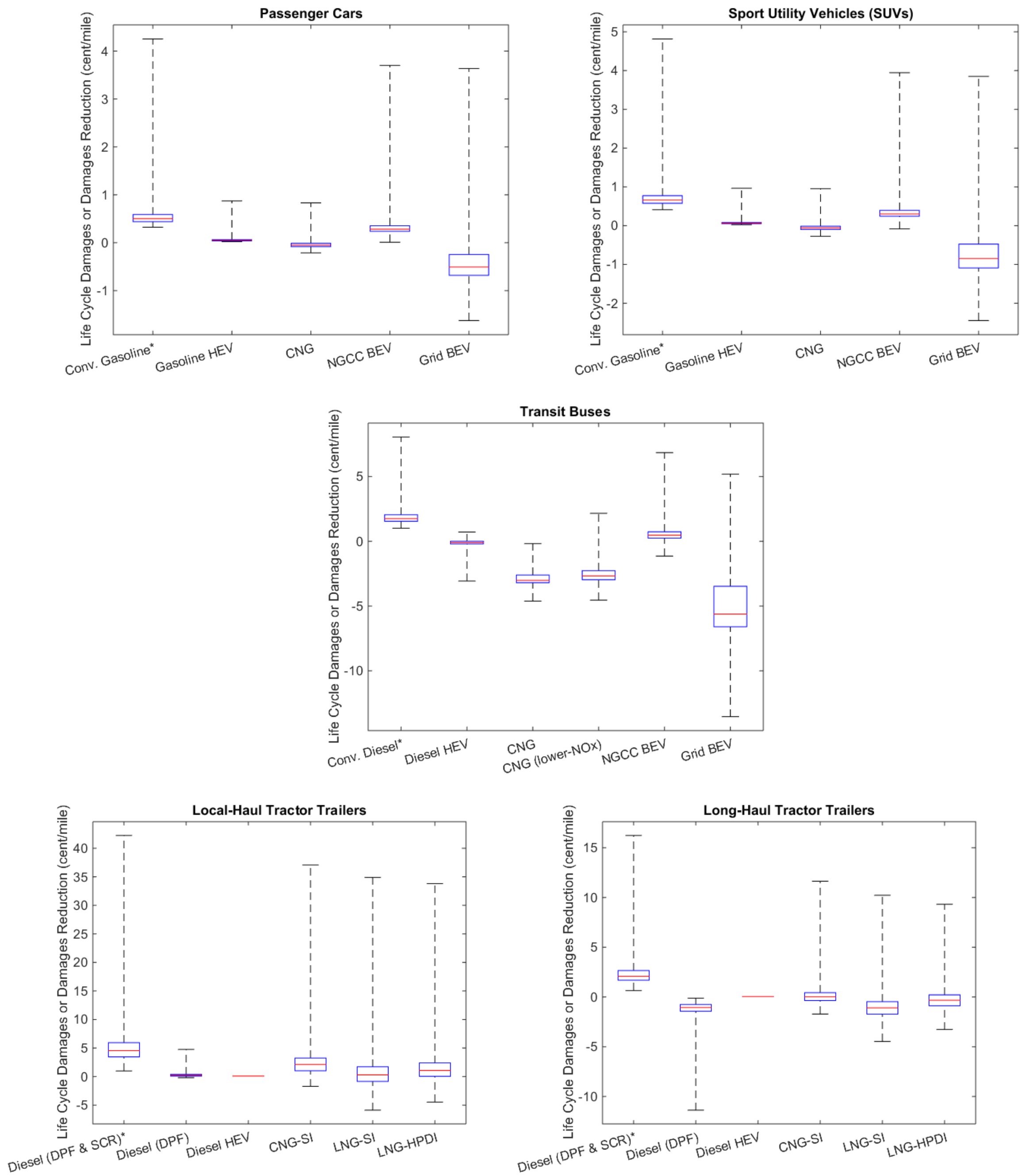
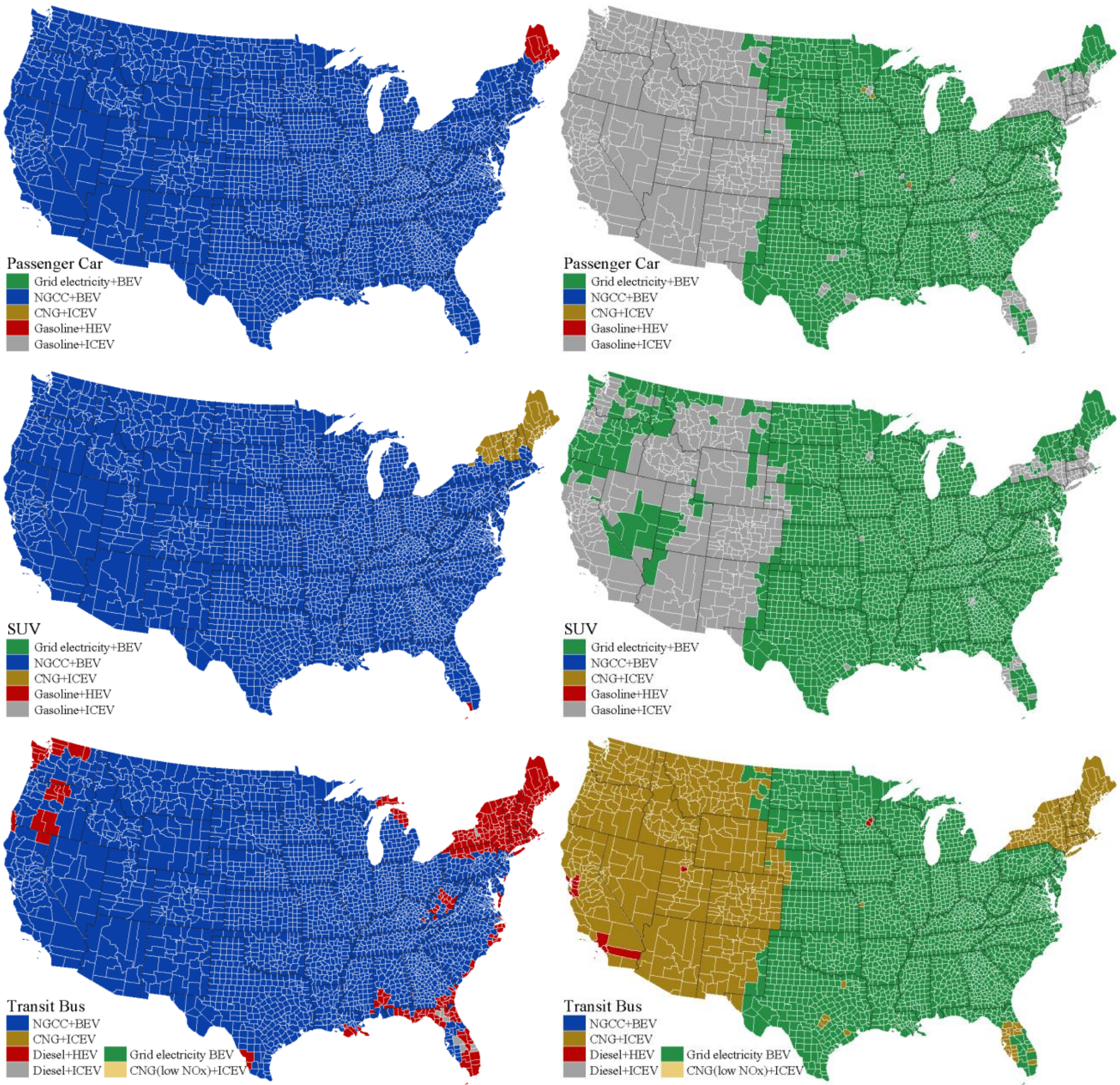


Figure C.3. Life cycle air pollution damages of baseline petroleum pathways (conventional gasoline for light-duty vehicles, and conventional diesel for heavy-duty vehicles, marked with *), and reduction in life cycle damages of alternative petroleum and natural gas pathways replacing the baseline petroleum pathways. I calculate the damages reduction in each county across the U.S. Negative values in damages reduction suggest lower damages from the alternative pathways compared to the baseline petroleum pathways. Emissions data comes from U.S. EPA's NEI. Marginal damage estimates of CAPs come from the AP2

model. The box plots show the minimum, 25th percentile, median, 75th percentile, and maximum.



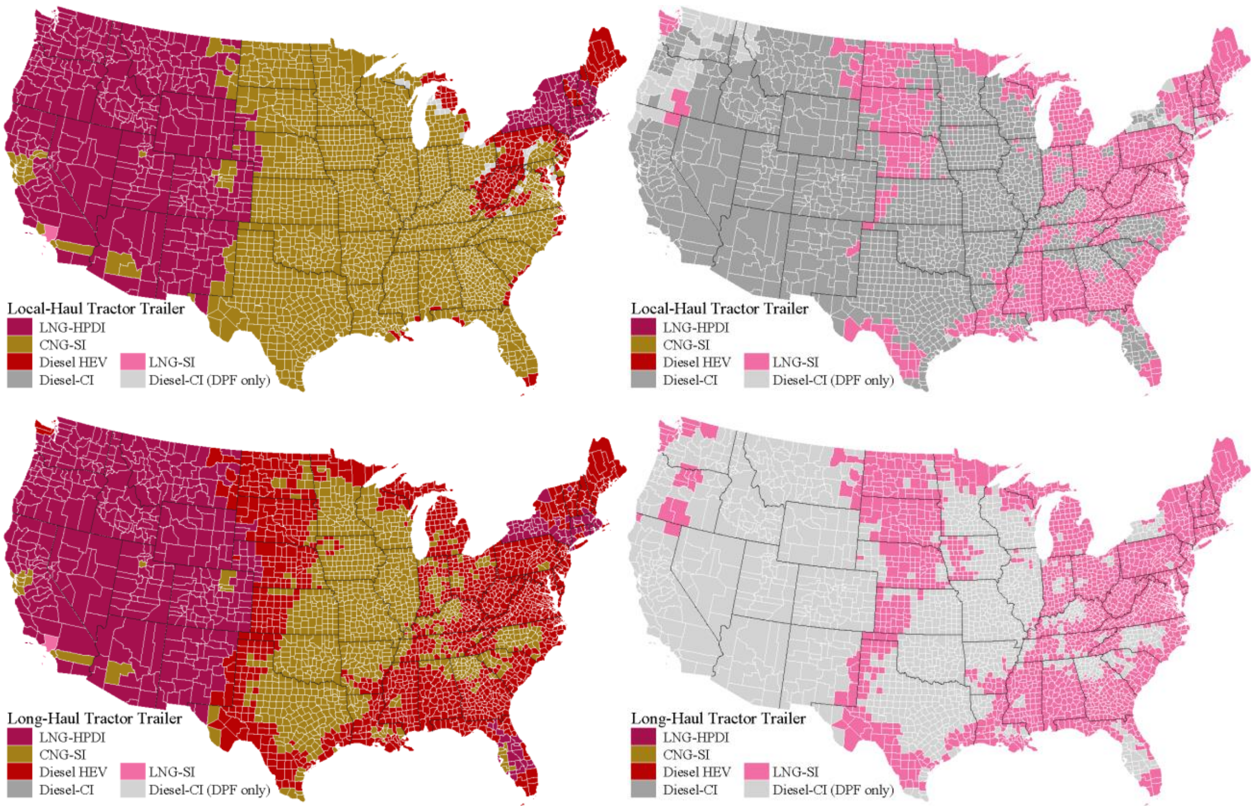
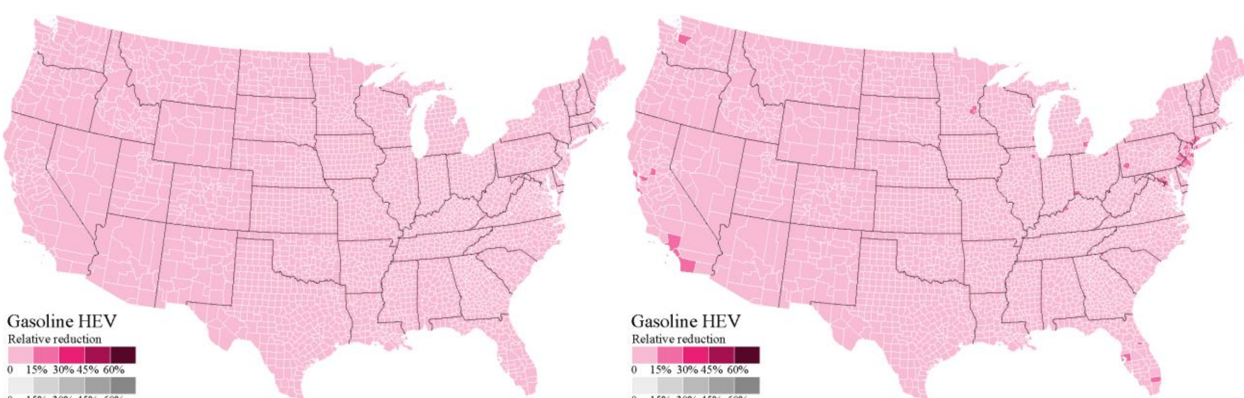


Figure C.4. The best pathway (left panel) and the worst pathway (right panel) for each vehicle type in each county using emissions data from U.S. EPA's NEI. Here the best/worst means the achieving lowest/highest life cycle air pollution damages. Air pollution damages are calculated using the AP2 model.



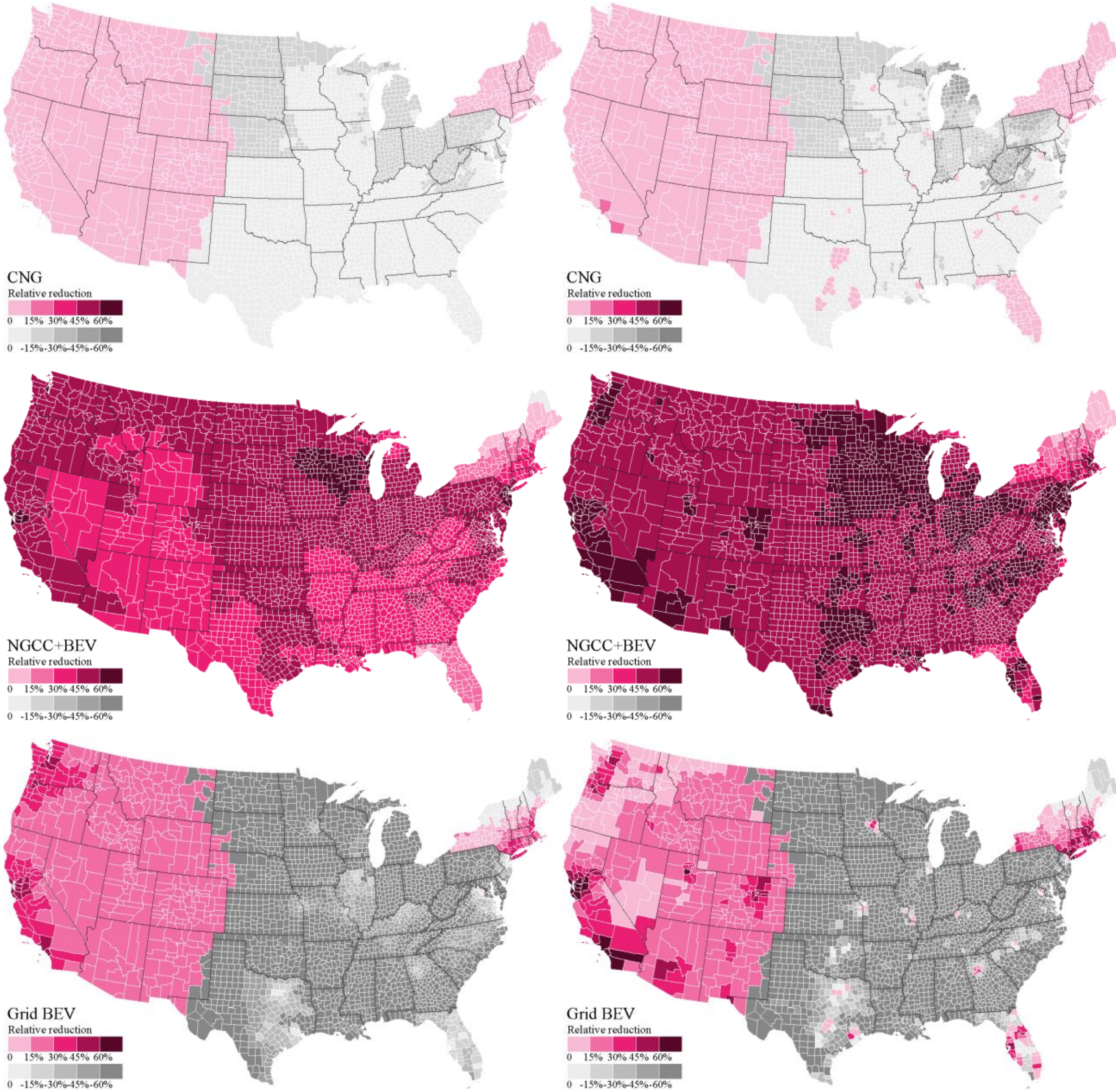


Figure C.5. Relative reduction in life cycle air pollution damages from replacing conventional gasoline with alternative fuel pathways for passenger cars using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional gasoline.

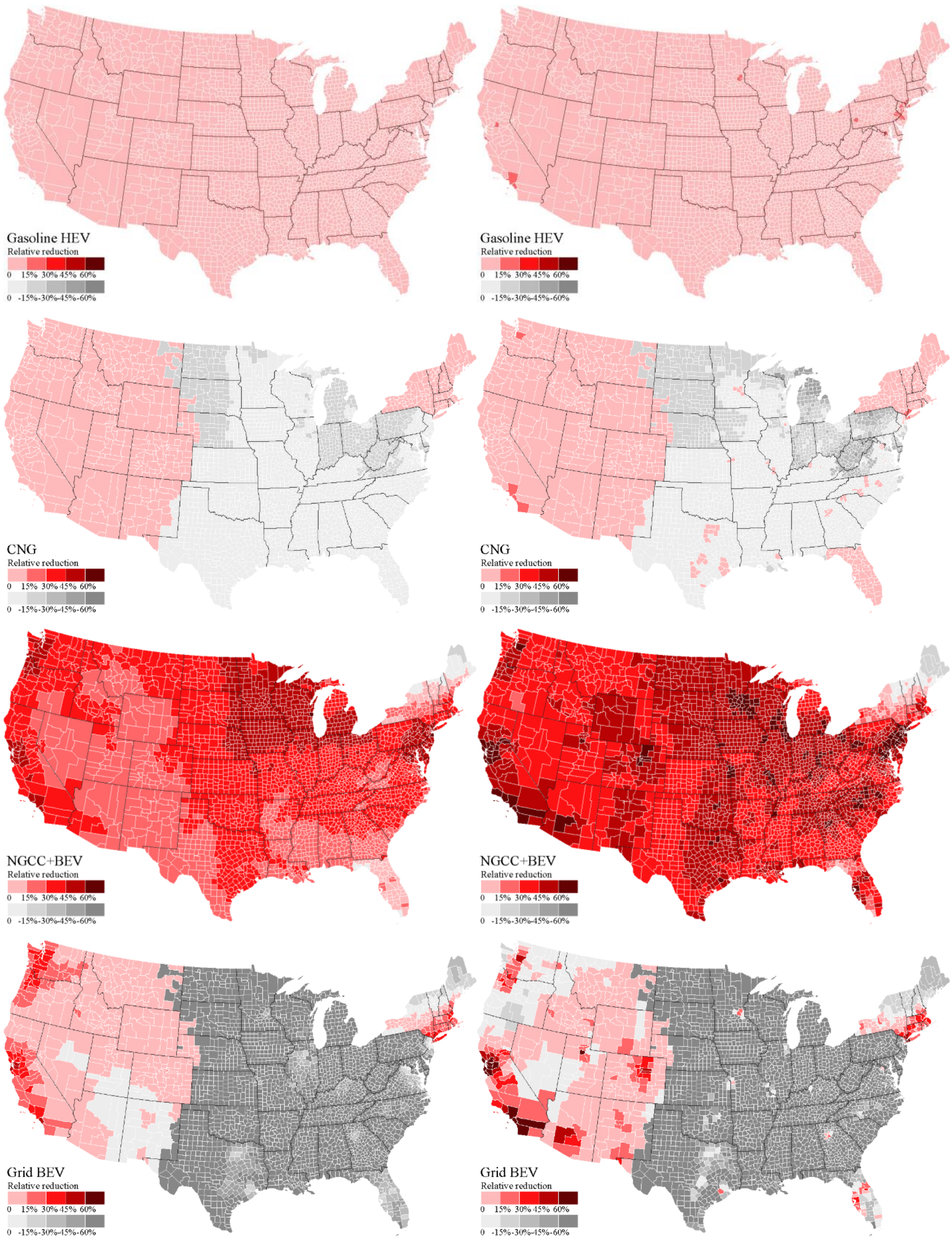
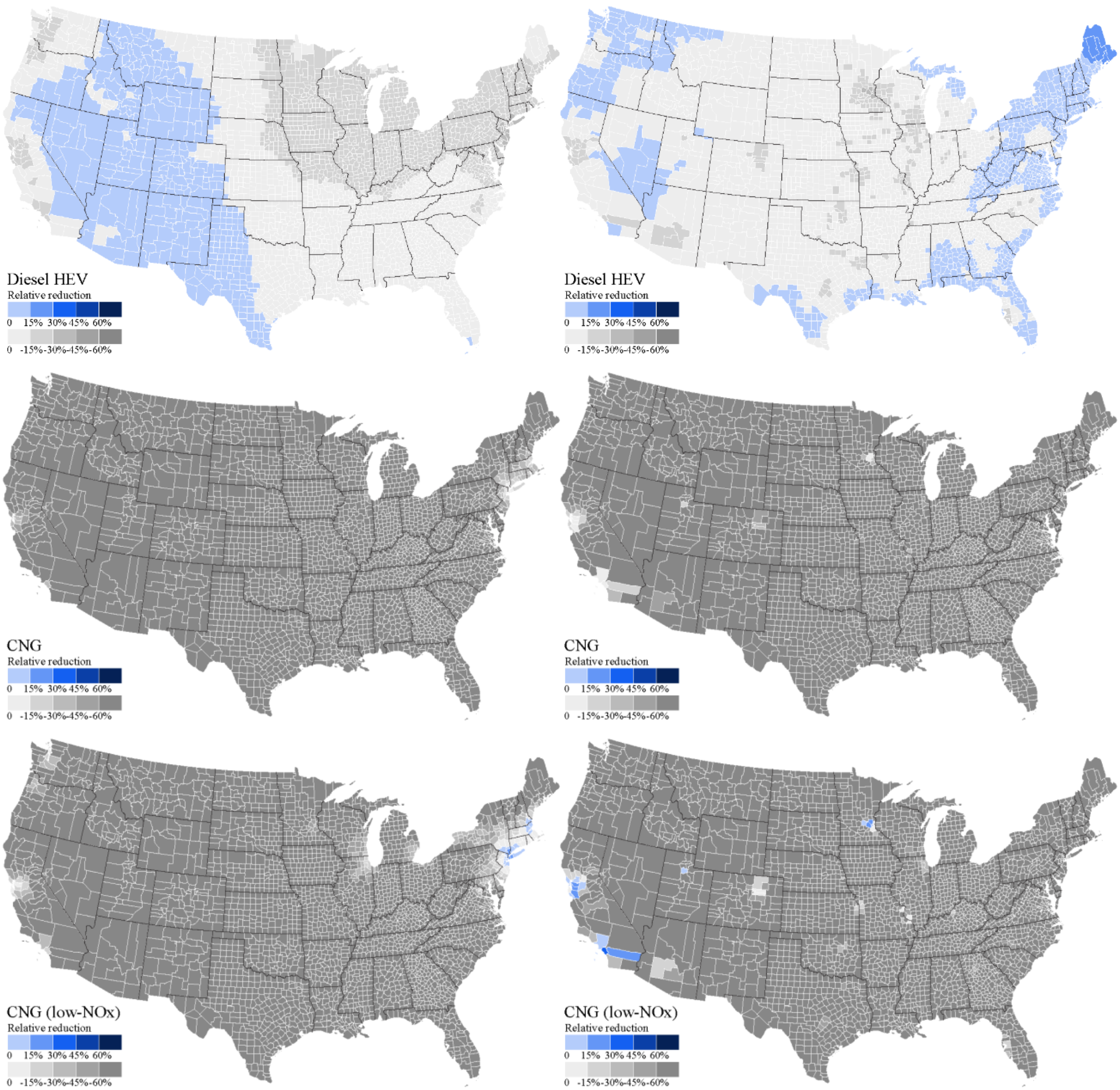

Gasoline HEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Gasoline HEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
CNG
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
CNG
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
NGCC+BEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
NGCC+BEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Grid BEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Grid BEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%

Figure C.6. Relative reduction in life cycle air pollution damages from replacing conventional gasoline with alternative fuel pathways for SUVs using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional gasoline.



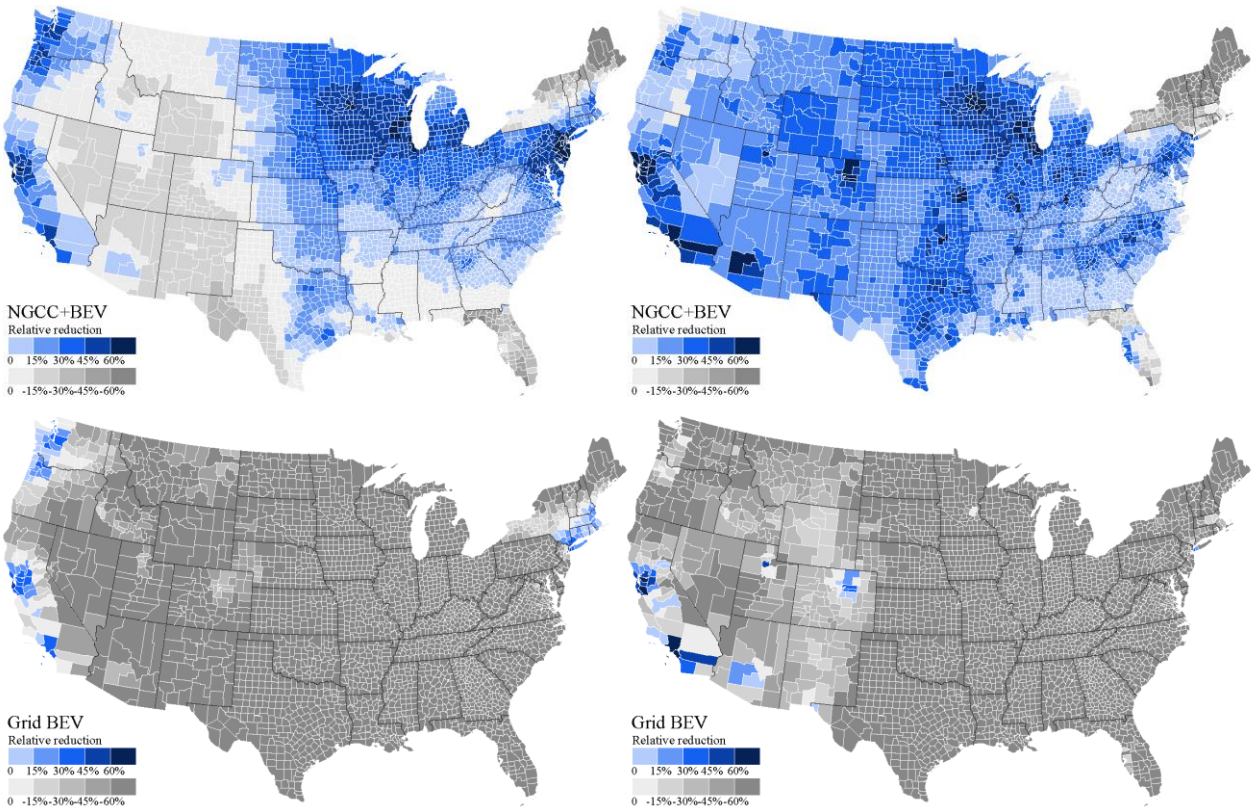
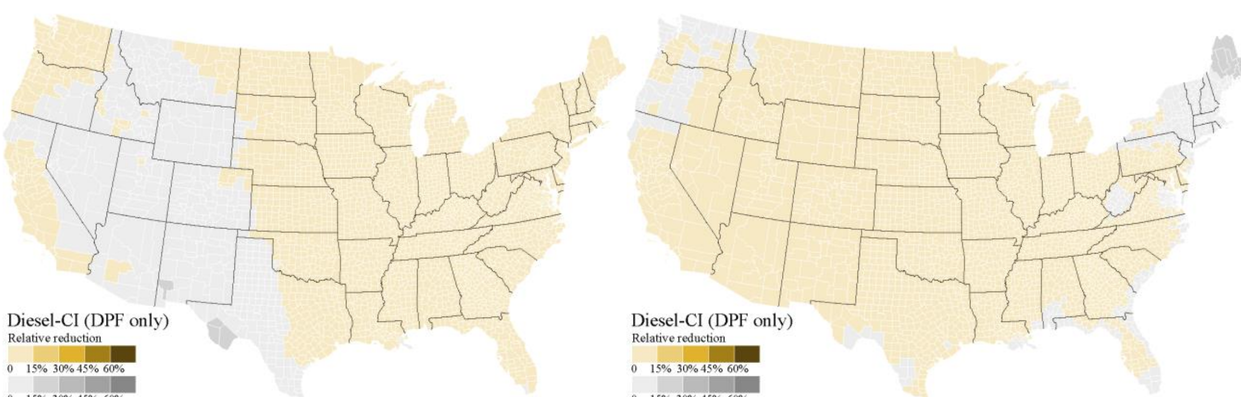


Figure C.7. Relative reduction in life cycle air pollution damages from replacing conventional diesel with alternative fuel pathways for transit buses using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.



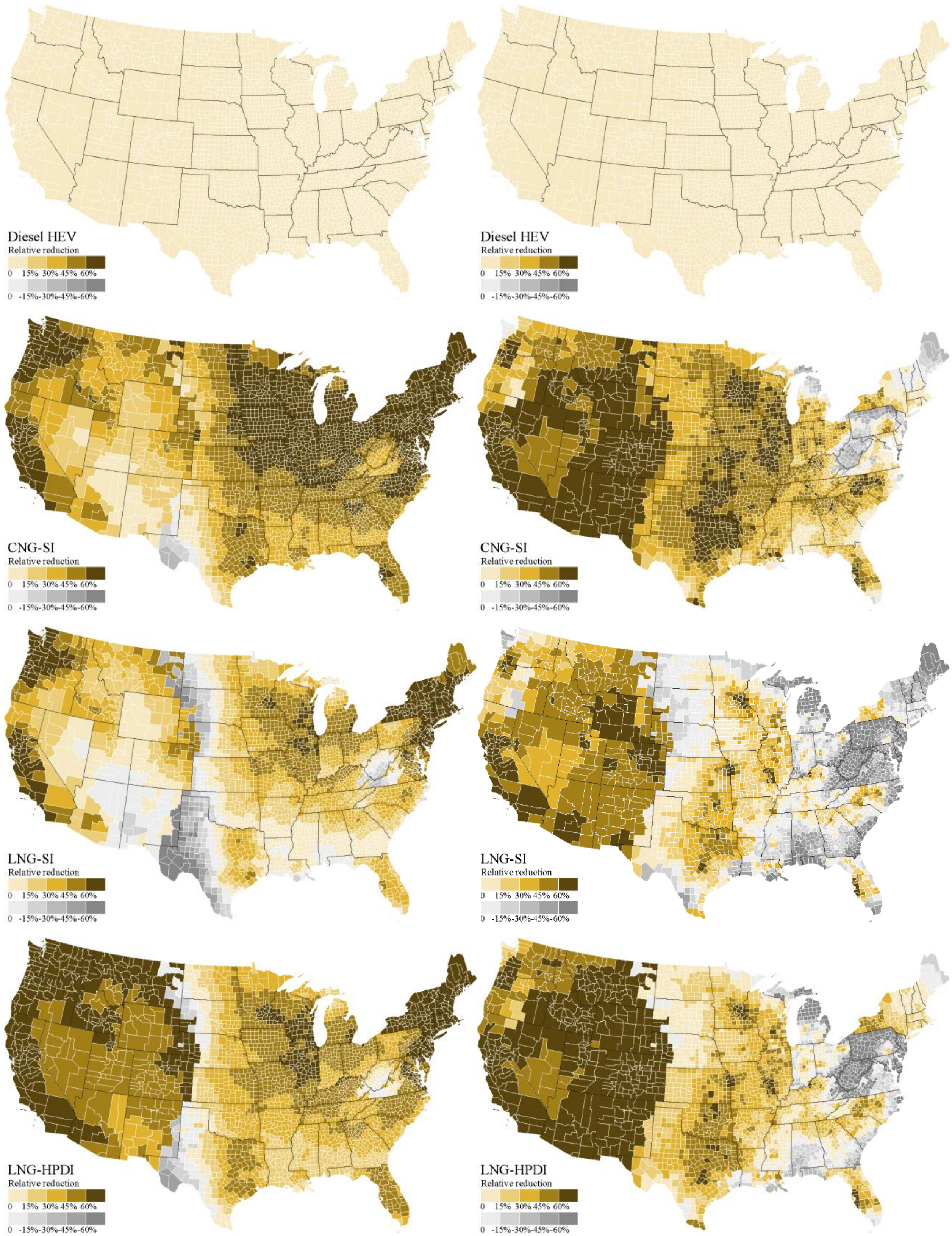
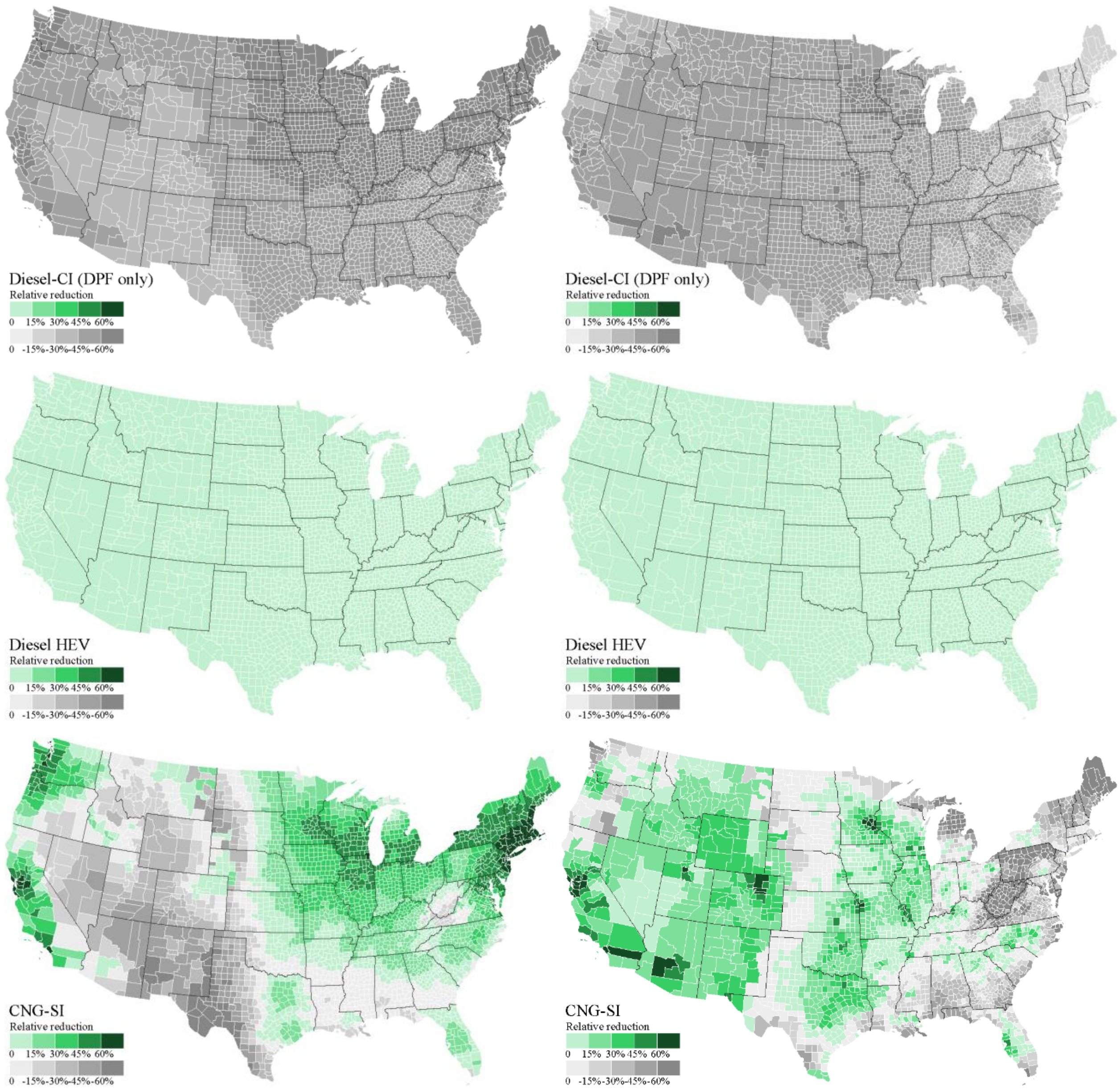

Diesel HEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Diesel HEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
CNG-SI
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
CNG-SI
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
LNG-SI
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
LNG-SI
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -15% -60%
LNG-HPDI
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
LNG-HPDI
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%

Figure C.8. Relative reduction in life cycle air pollution damages from replacing conventional diesel with alternative fuel pathways for local-haul tractor-trailers using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.



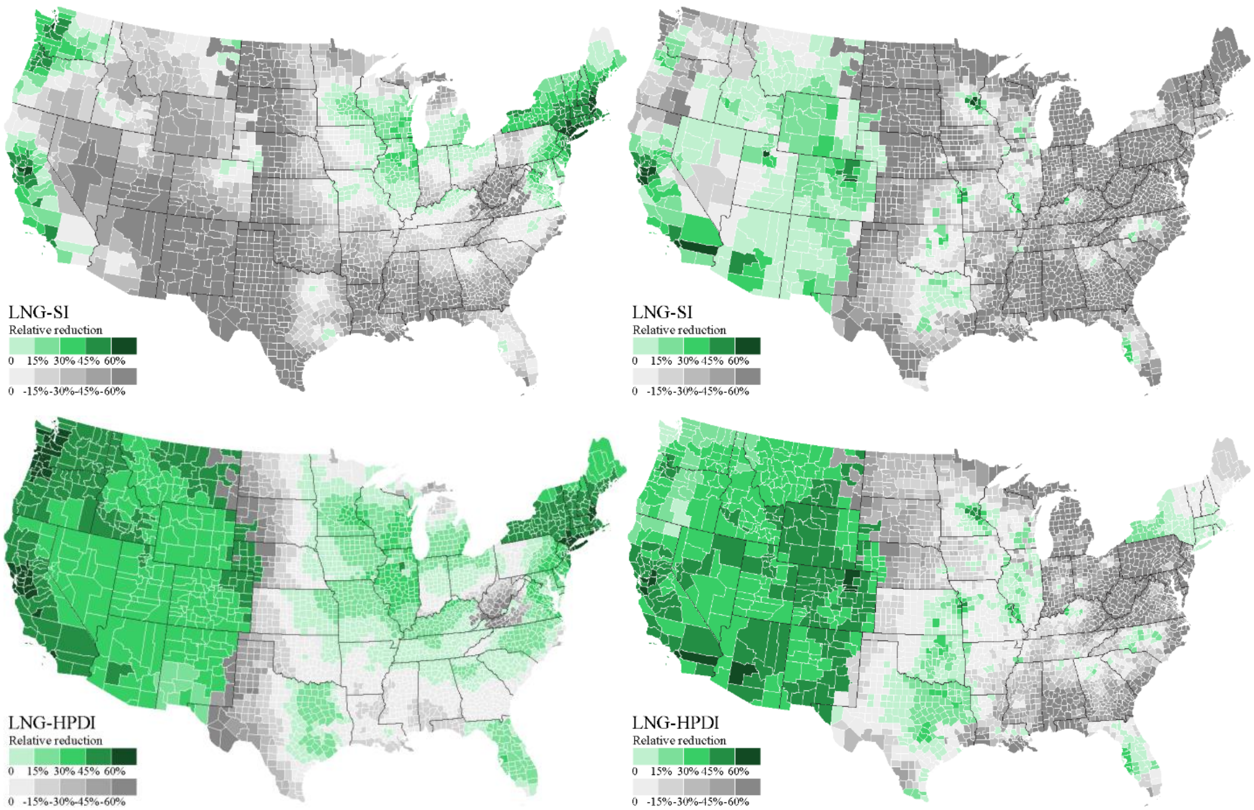
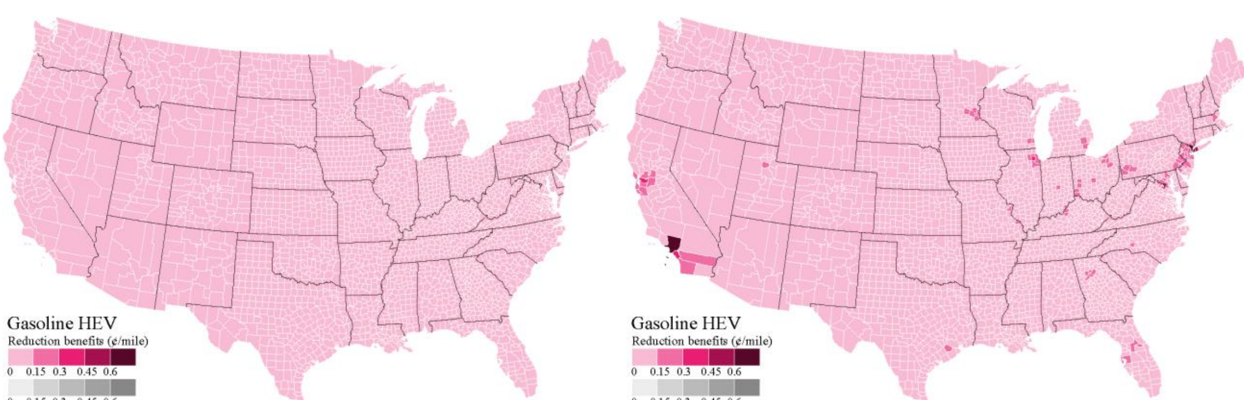


Figure C.9. Relative reduction in life cycle air pollution damages from replacing conventional diesel with alternative fuel pathways for long-haul tractor-trailers using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.



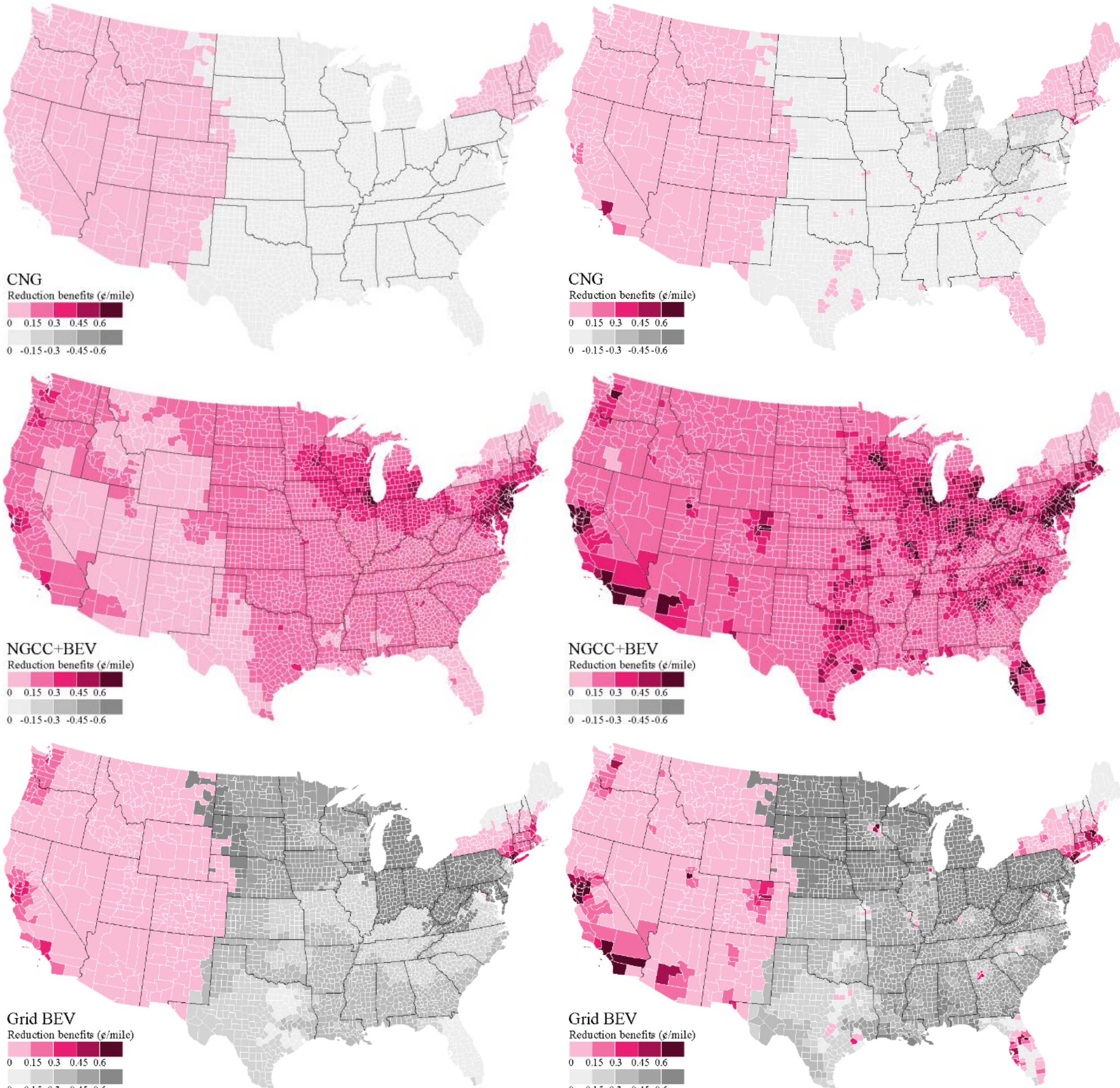
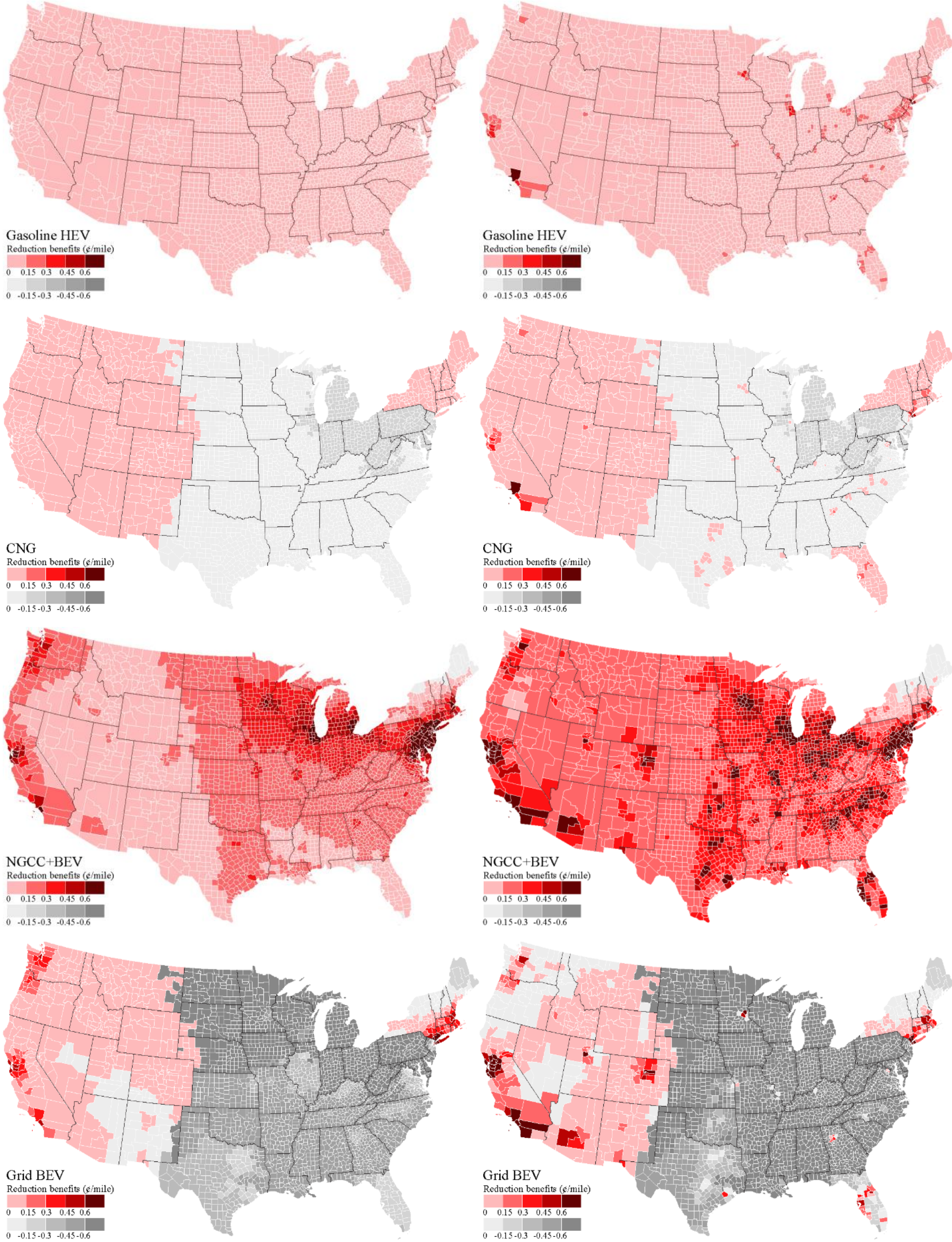
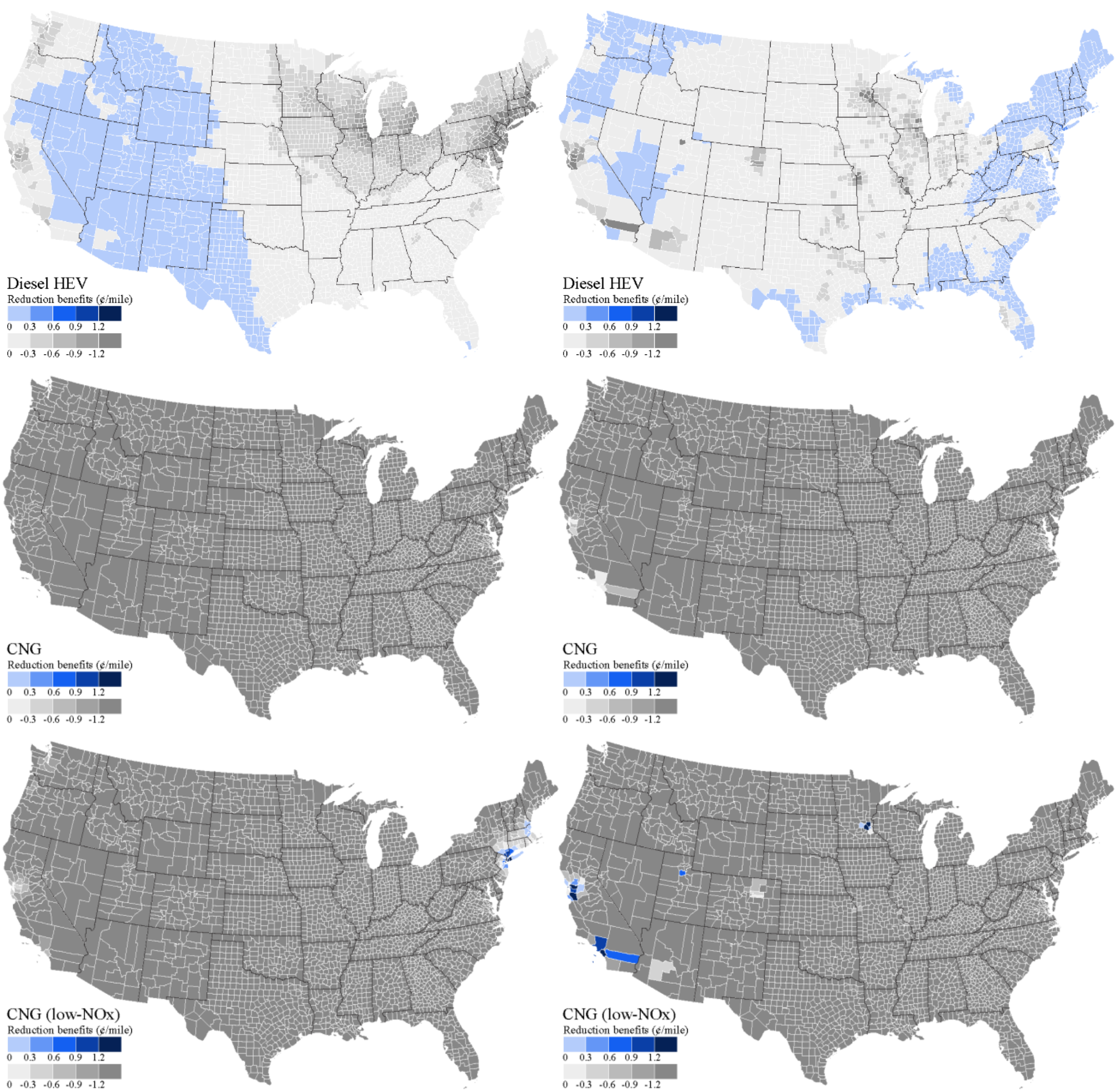


Figure C.10. Life cycle air pollution damage reduction benefits from replacing conventional gasoline with alternative fuel pathways for passenger cars using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; bottom panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional gasoline.



Gasoline HEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
Gasoline HEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
CNG
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
CNG
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
NGCC+BEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
NGCC+BEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
Grid BEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
Grid BEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6

Figure C.11. Life cycle air pollution damage reduction benefits from replacing conventional gasoline with alternative fuel pathways for SUVs using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional gasoline.



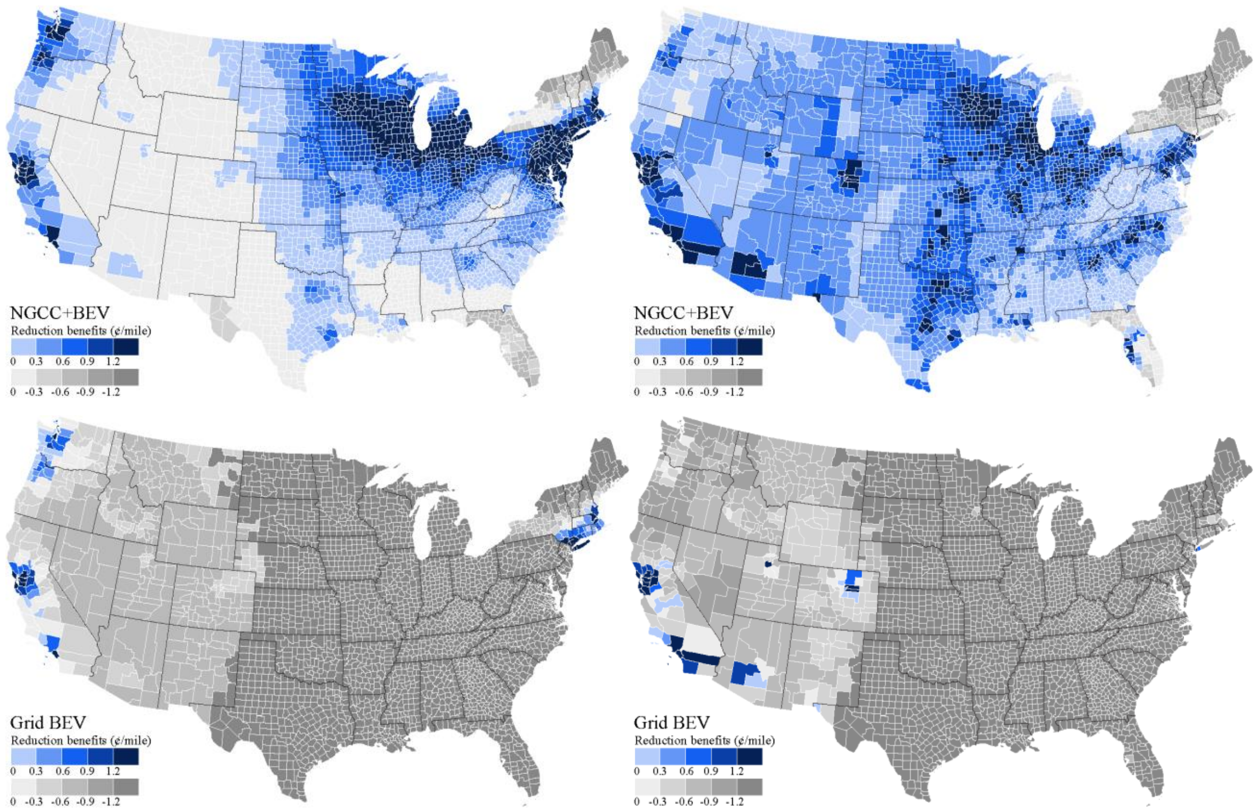
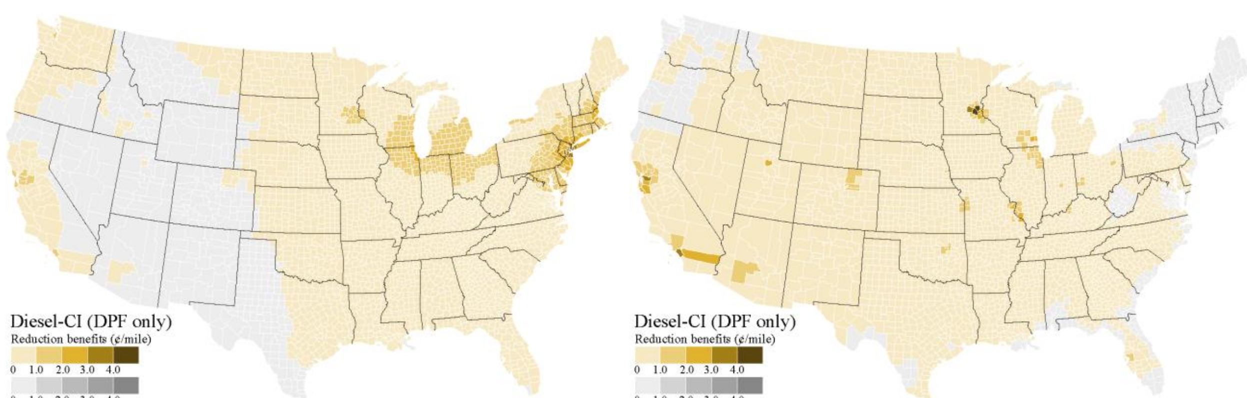
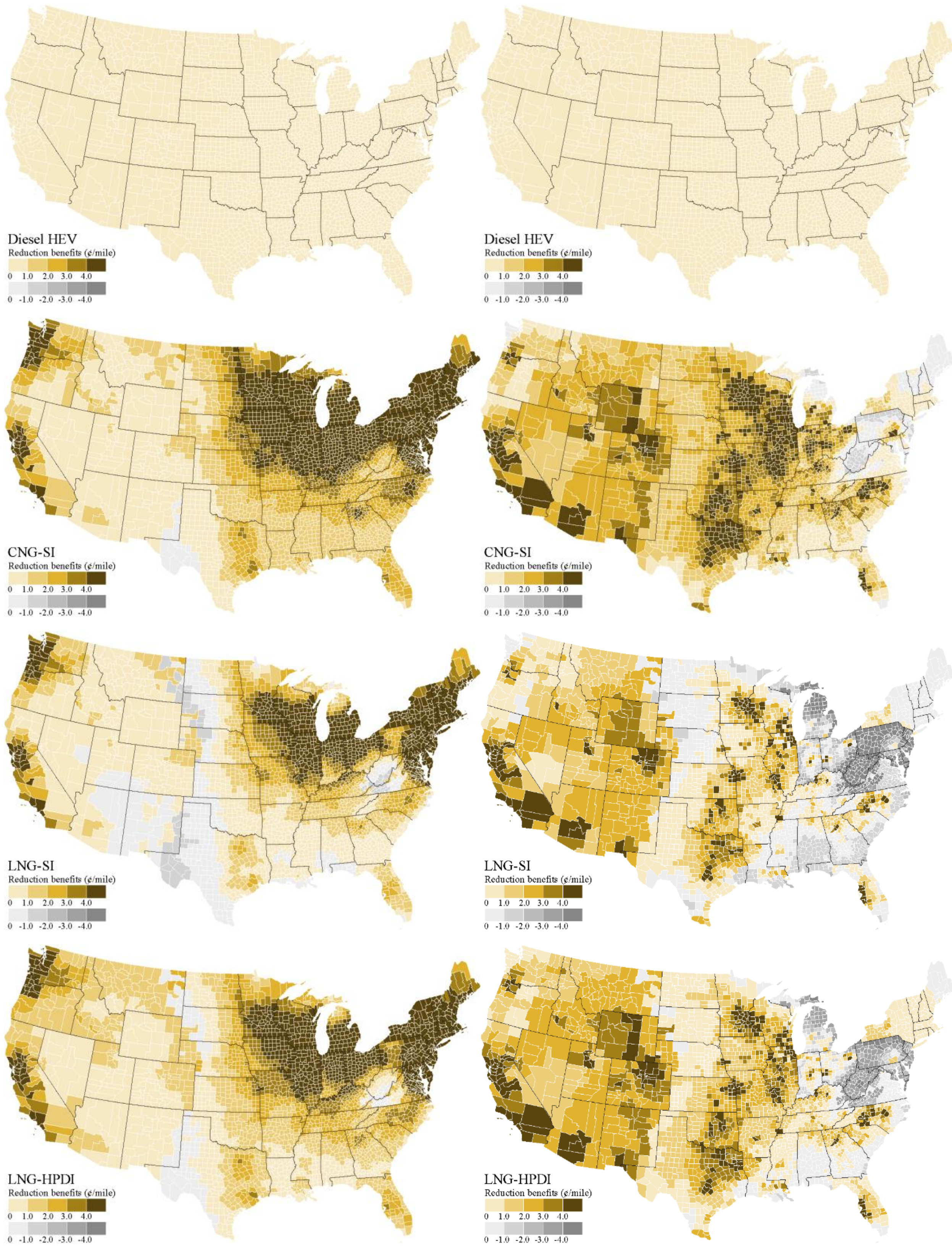


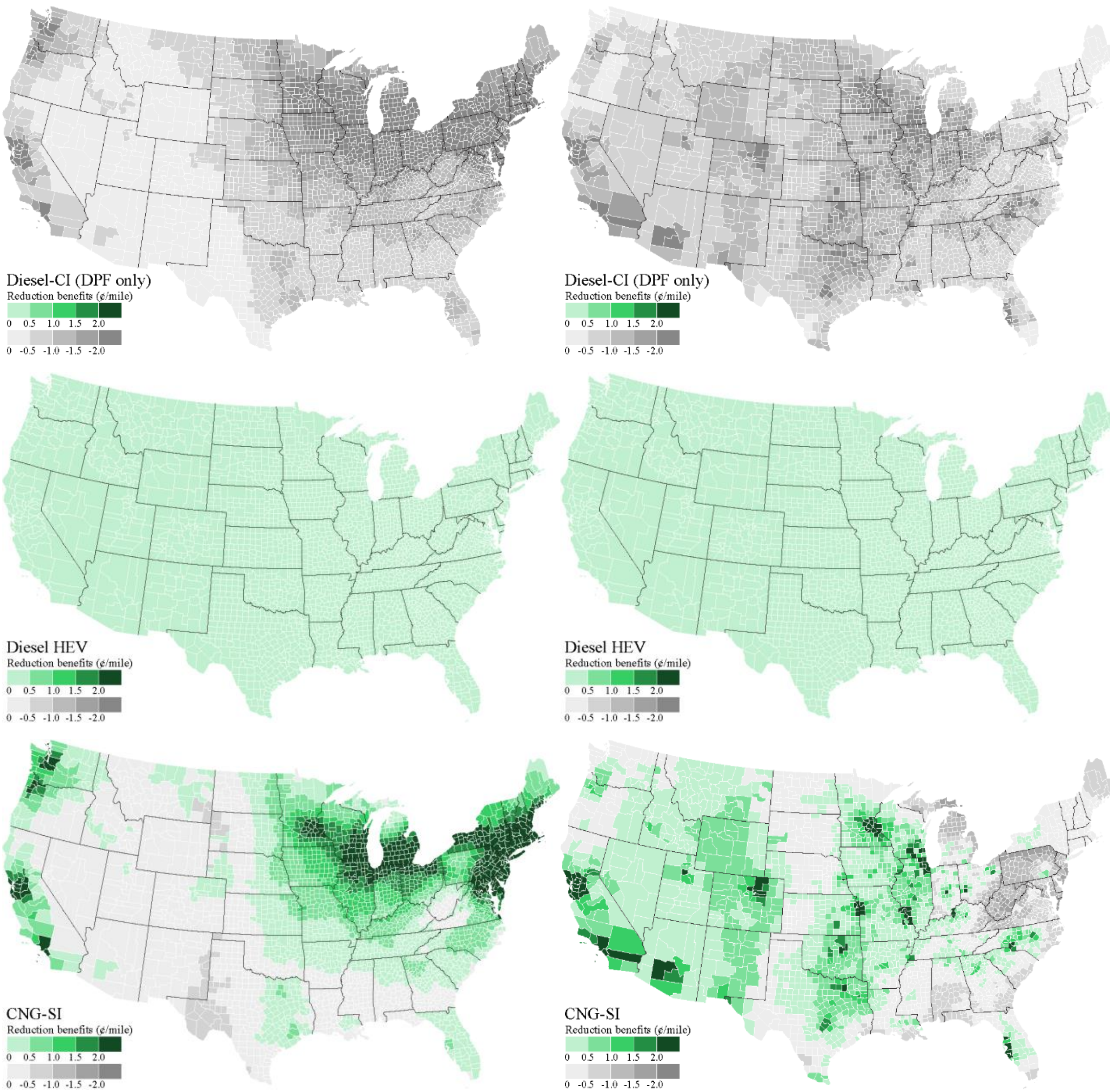
Figure C.12. Life cycle air pollution damage reduction benefits from replacing conventional diesel with alternative fuel pathways for transit buses using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.





Diesel HEV
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
Diesel HEV
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
CNG-SI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
CNG-SI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
LNG-SI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
LNG-SI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
LNG-HPDI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
LNG-HPDI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0

Figure C.13. Life cycle air pollution damage reduction benefits from replacing conventional diesel with alternative fuel pathways for local-haul tractor-trailers using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.



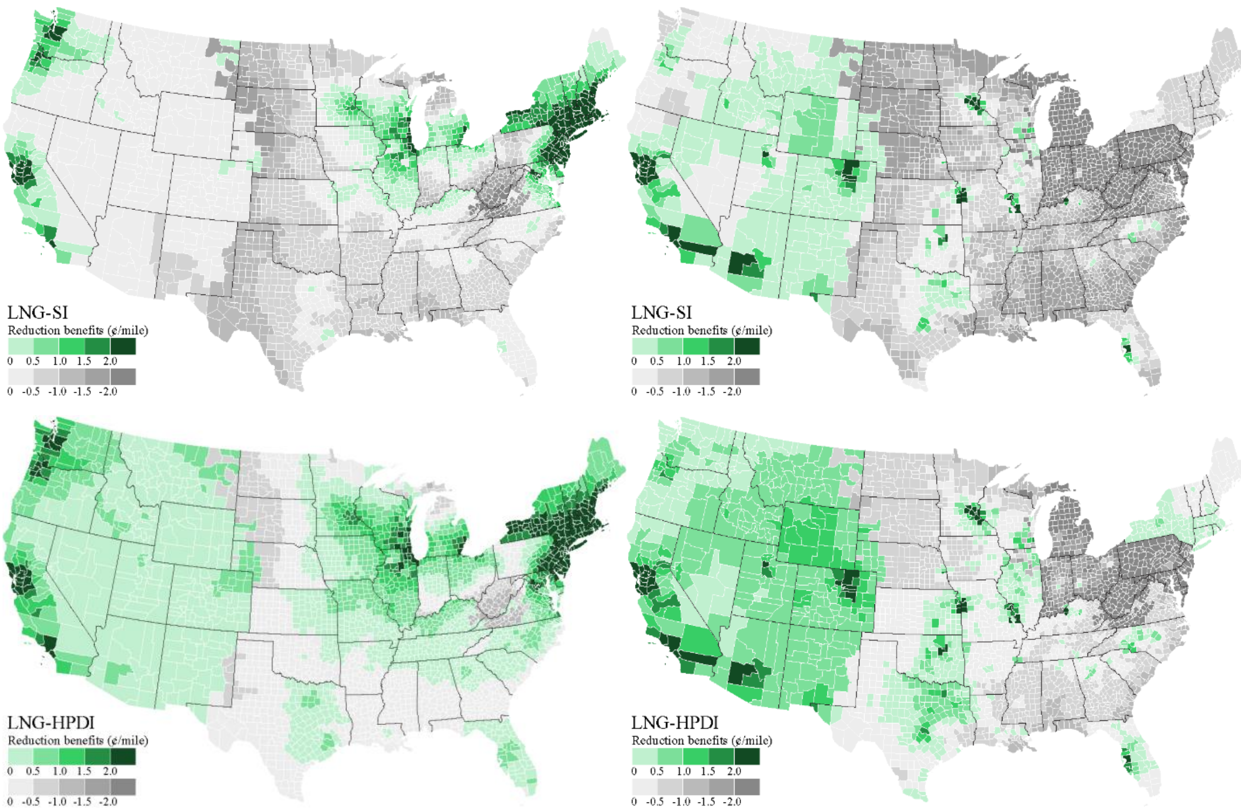


Figure C.14. Life cycle air pollution damage reduction benefits from replacing conventional diesel with alternative fuel pathways for long-haul tractor-trailers using emissions data from U.S. EPA's NEI. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.

C.6.2 Upstream emissions data from the GREET model

This sections shows additional results using the emissions data from the GREET model, as summarized in **Table C.27**.

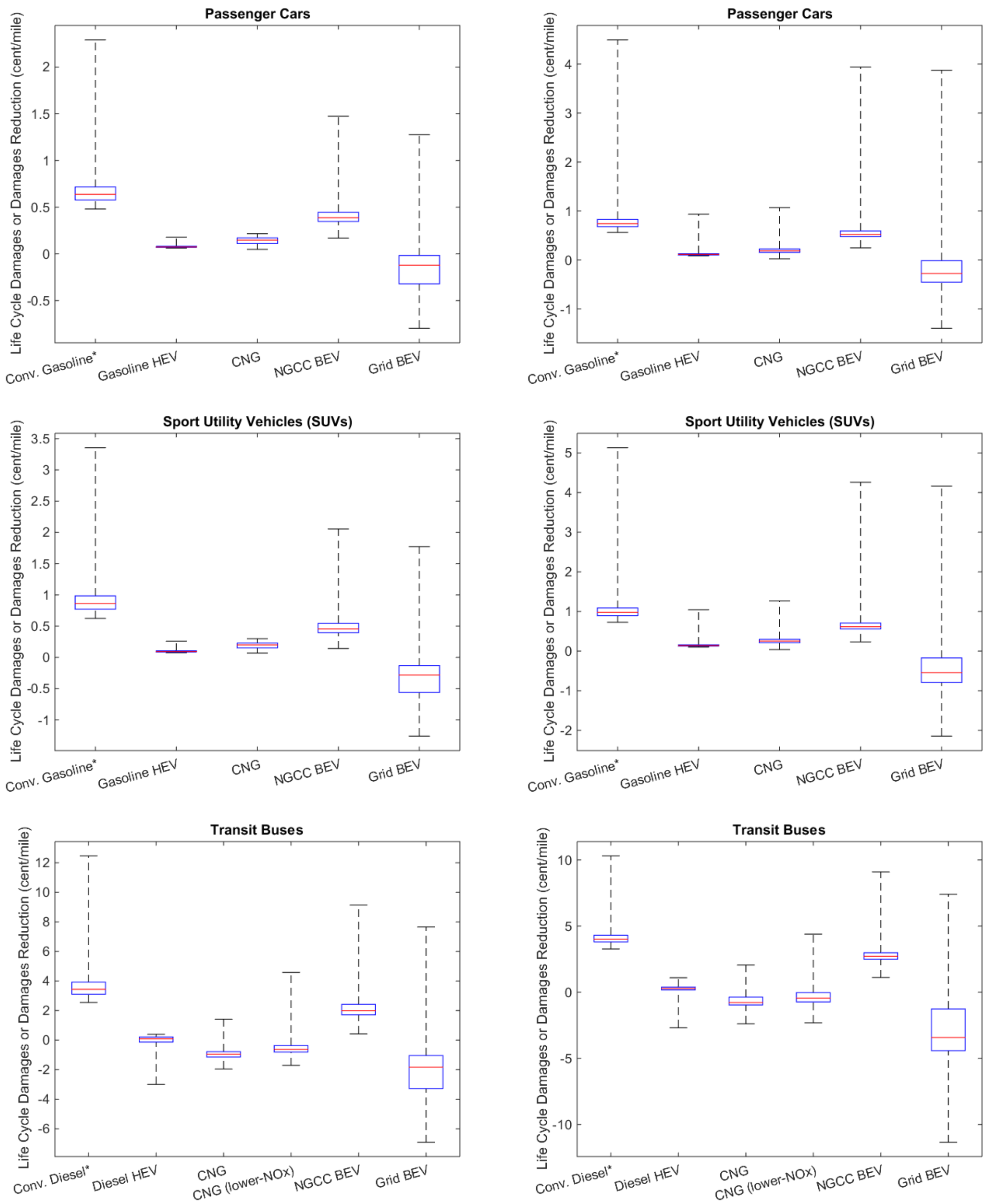
Table C.27. A summary of additional results using emissions data from GREET model.

<table>
<tr><th colspan="2">Graphics topics</th><th>Boxplots on relative reductions*</th><th>Percentage of counties that see damage reductions*</th><th>“Best”/ “Worst” technology</th><th>-</th><th>-</th></tr>
<tr><td rowspan="2">Marginal damage model</td><td>EASIUR</td><td rowspan="2">Figure C.15</td><td rowspan="2">Table C.28</td><td>Figure C.16</td><td>-</td><td>-</td></tr>
<tr><td>AP2</td><td>Figure C.17</td><td>-</td><td>-</td></tr>
<tr><th colspan="2">Graphics topics (vehicle types)</th><th>Passenger car</th><th>SUV</th><th>Transit bus</th><th>Local-haul tractor-trailer</th><th>Long-haul tractor-trailer</th></tr>
<tr><td rowspan="2">Metrics*</td><td>Relative reductions</td><td>Figure C.18</td><td>Figure C.19</td><td>Figure C.20</td><td>Figure C.21</td><td>Figure C.22</td></tr>
<tr><td>Absolute reductions</td><td>Figure C.23</td><td>Figure C.24</td><td>Figure C.25</td><td>Figure C.26</td><td>Figure C.27</td></tr>
</table>

** when replacing the baseline petroleum pathways with alternative fuel pathways*

Table C.28. Percentage of counties that see damage reductions from petroleum and natural gas pathways compared to the baseline petroleum fuels (conventional gasoline for passenger cars and SUVs; conventional diesel for transit buses and tractor-trailers) using emissions data from GREET model. Damage of a pathway is calculated based on the CAP emissions from life cycle or the vehicle operation stage, using the AP2 or the EASIUR model.

Method	Boundary	Gasoline HEV	CNG	NGCC + BEV	Grid + BEV	-
Passenger cars						
AP2	Life Cycle	100%	100%	100%	24%	-
	Vehicle Operation	100%	100%	100%	100%	-
EASIUR	Life Cycle	100%	100%	100%	23%	-
	Vehicle Operation	100%	100%	100%	100%	-
SUVs						
AP2	Life Cycle	100%	100%	100%	21%	-
	Vehicle Operation	100%	100%	100%	100%	-
EASIUR	Life Cycle	100%	100%	100%	19%	-
	Vehicle Operation	100%	100%	100%	100%	-
Method	**Boundary**	**Diesel HEV**	**CNG**	**CNG (low-NO_x engine)**	**NGCC + BEV**	**Grid + BEV**
Transit buses						
AP2	Life Cycle	93%	16%	23%	100%	18%
	Vehicle Operation	4%	0%	1%	100%	100%
EASIUR	Life Cycle	63%	2%	9%	100%	18%
	Vehicle Operation	0%	0%	1%	100%	100%
Method	**Boundary**	**Diesel (DPF)**	**Diesel HEV**	**CNG-SI**	**LNG-SI**	**LNG-HPDI**
Local-haul tractor-trailers						
AP2	Life Cycle	89%	100%	100%	89%	91%
	Vehicle Operation	89%	0%	98%	98%	100%
EASIUR	Life Cycle	87%	100%	100%	100%	100%
	Vehicle Operation	87%	0%	98%	98%	100%
Line-haul tractor-trailers						
AP2	Life Cycle	0%	100%	96%	60%	85%
	Vehicle Operation	0%	0%	81%	82%	100%
EASIUR	Life Cycle	0%	100%	100%	89%	98%
	Vehicle Operation	0%	0%	83%	84%	100%



Passenger Cars
Life Cycle Damages or Damages Reduction (cent/mile)
Conv. Gasoline*
Gasoline HEV
CNG
NGCC BEV
Grid BEV
Passenger Cars
Life Cycle Damages or Damages Reduction (cent/mile)
Conv. Gasoline*
Gasoline HEV
CNG
NGCC BEV
Grid BEV
Sport Utility Vehicles (SUVs)
Life Cycle Damages or Damages Reduction (cent/mile)
Conv. Gasoline*
Gasoline HEV
CNG
NGCC BEV
Grid BEV
Sport Utility Vehicles (SUVs)
Life Cycle Damages or Damages Reduction (cent/mile)
Conv. Gasoline*
Gasoline HEV
CNG
NGCC BEV
Grid BEV
Transit Buses
Life Cycle Damages or Damages Reduction (cent/mile)
Conv. Diesel*
Diesel HEV
CNG
CNG (lower-NOx)
NGCC BEV
Grid BEV
Transit Buses
Life Cycle Damages or Damages Reduction (cent/mile)
Conv. Diesel*
Diesel HEV
CNG
CNG (lower-NOx)
NGCC BEV
Grid BEV

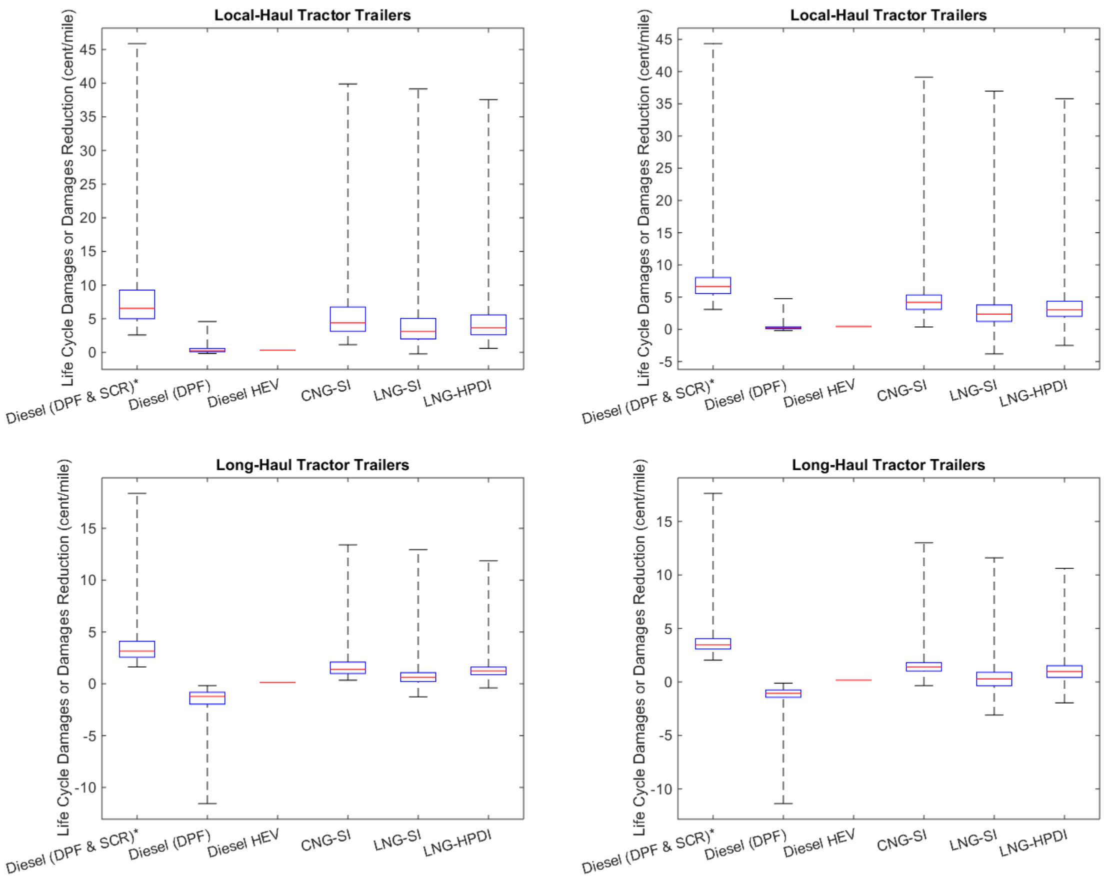
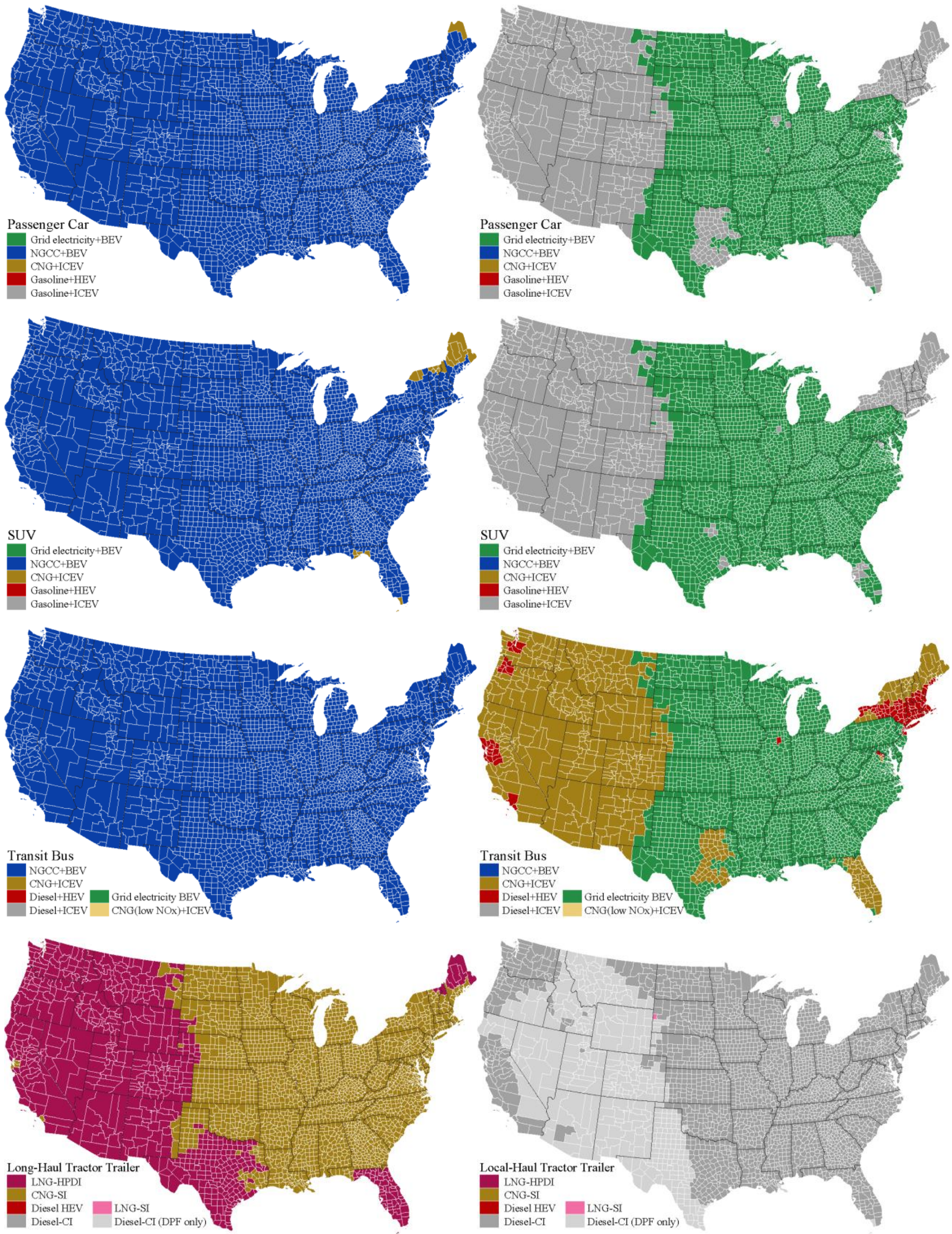


Figure C.15. Life cycle air pollution damages of baseline petroleum pathways (conventional gasoline for light-duty vehicles, and conventional diesel for heavy-duty vehicles, marked with *), and reduction in life cycle damages of alternative petroleum and natural gas pathways replacing the baseline petroleum pathways. Emissions data comes from the GREET model. Left panel uses the marginal damages from the EASIUR model; right panel uses the marginal damages from the AP2 model. We calculate the relative reductions in each county across the U.S. Negative values in damages reduction suggest lower damages from the alternative pathways compared to the baseline petroleum pathways. The box plots show the minimum, 25th percentile, median, 75th percentile, and maximum.


Passenger Car
Grid electricity+BEV
NGCC+BEV
CNG+ICEV
Gasoline+HEV
Gasoline+ICEV
Passenger Car
Grid electricity+BEV
NGCC+BEV
CNG+ICEV
Gasoline+HEV
Gasoline+ICEV
SUV
Grid electricity+BEV
NGCC+BEV
CNG+ICEV
Gasoline+HEV
Gasoline+ICEV
SUV
Grid electricity+BEV
NGCC+BEV
CNG+ICEV
Gasoline+HEV
Gasoline+ICEV
Transit Bus
NGCC+BEV
CNG+ICEV
Diesel+HEV
Grid electricity BEV
Diesel+ICEV
CNG(low NOx)+ICEV
Transit Bus
NGCC+BEV
CNG+ICEV
Diesel+HEV
Grid electricity BEV
Diesel+ICEV
CNG(low NOx)+ICEV
Long-Haul Tractor Trailer
LNG-HPDI
CNG-SI
Diesel HEV
LNG-SI
Diesel-CI
Diesel-CI (DPF only)
Local-Haul Tractor Trailer
LNG-HPDI
CNG-SI
Diesel HEV
LNG-SI
Diesel-CI
Diesel-CI (DPF only)

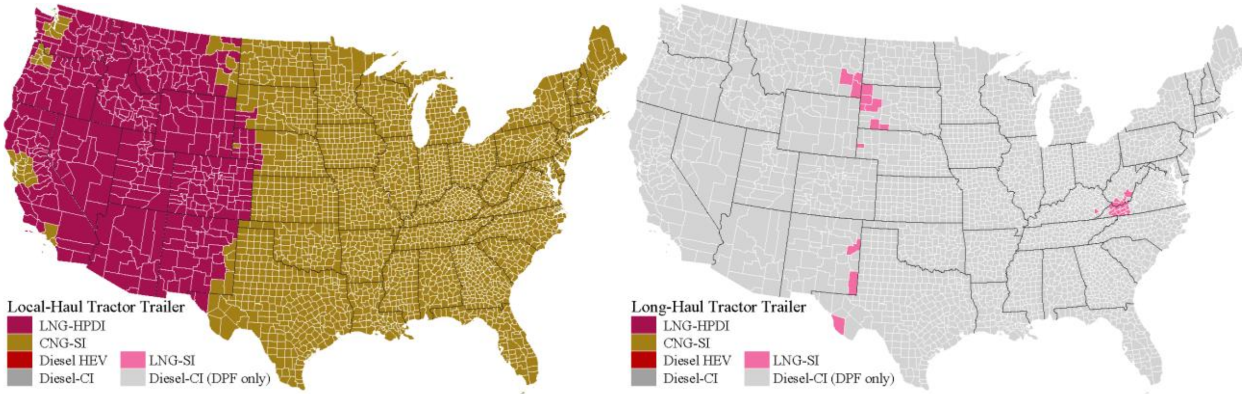
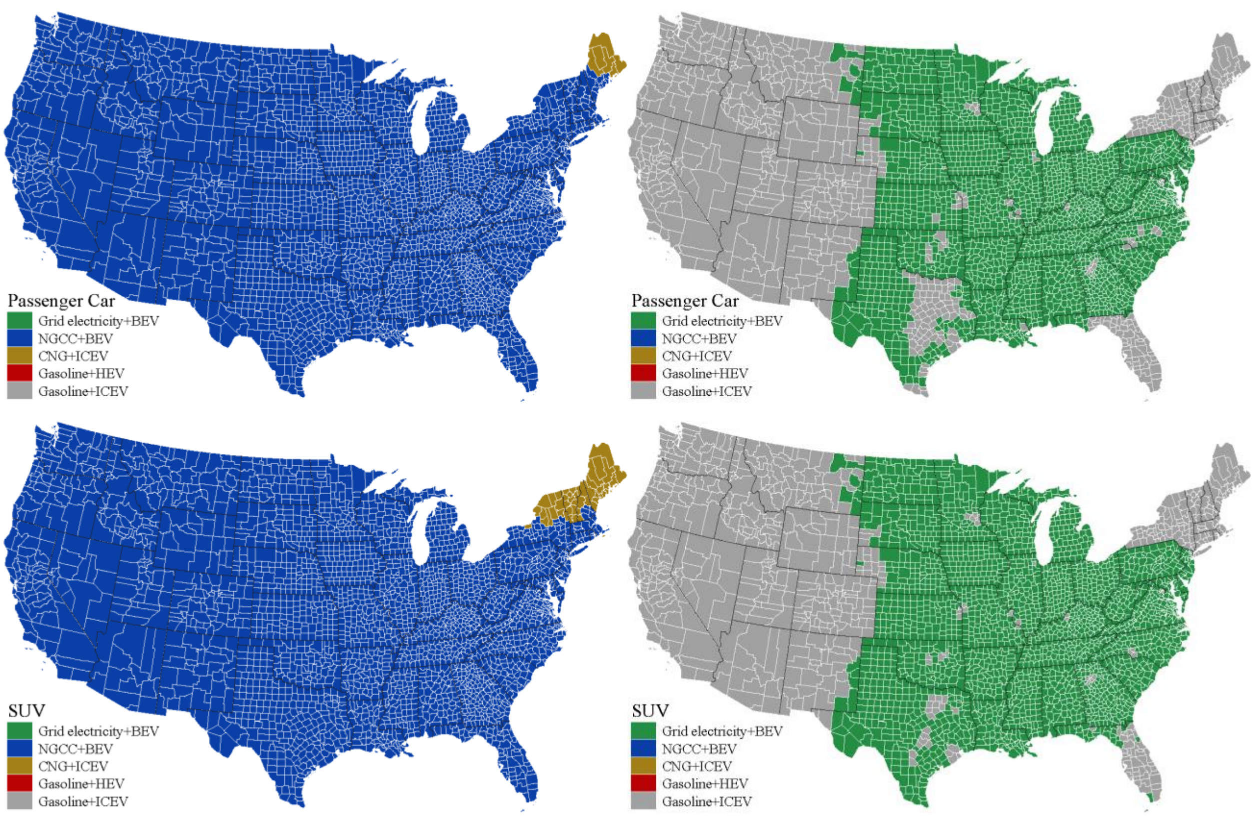


Figure C.16. The best pathway (left panel) and the worst pathway (right panel) for each vehicle type in each county using emissions data from GREET model. Here the best/worst means achieving the lowest/highest life cycle air pollution damages. Air pollution damages are calculated using the EASIUR model.



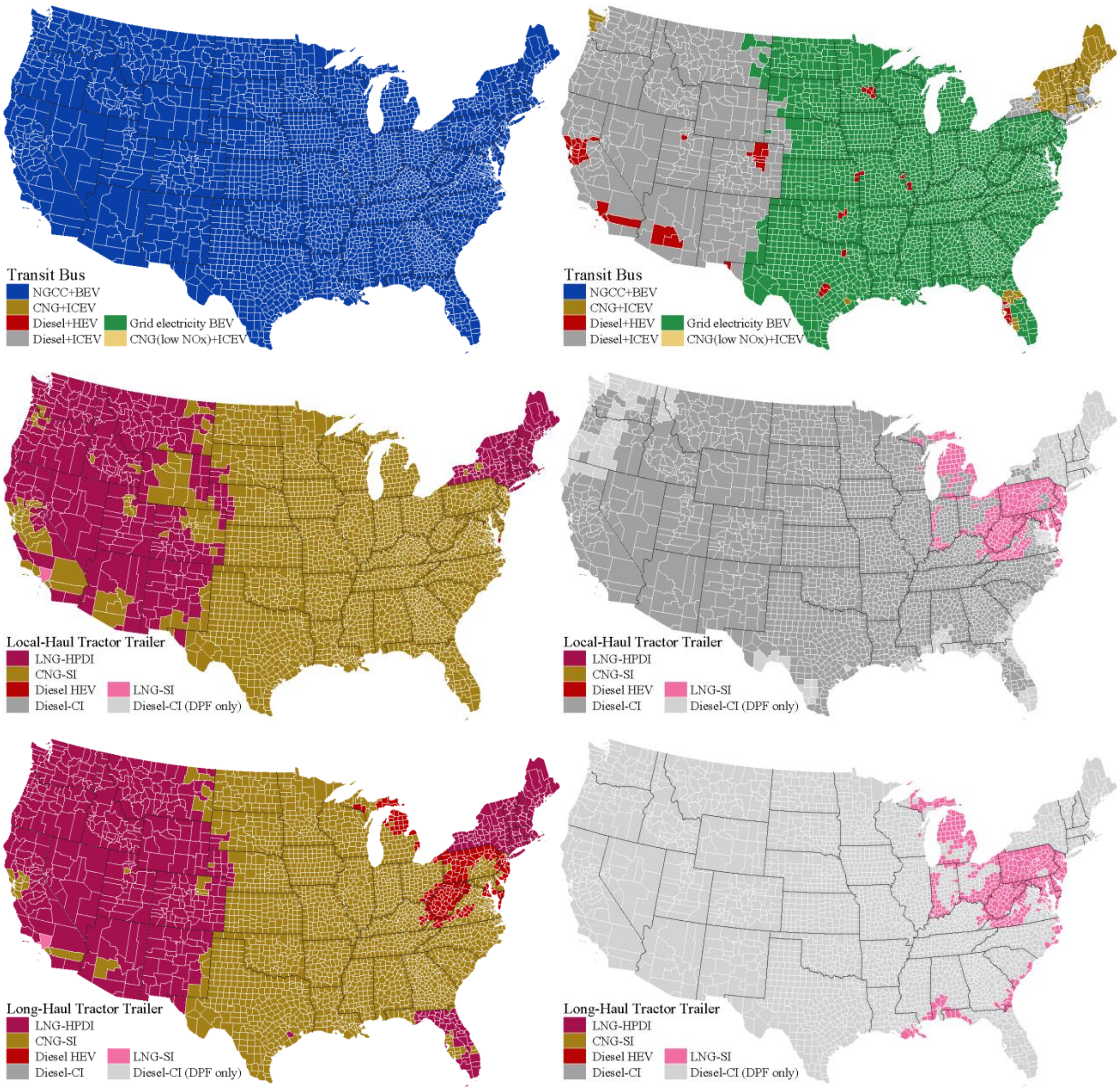


Figure C.17. The best pathway (left panel) and the worst pathway (right panel) for each vehicle type in each county using emissions data from GREET model. Here the best/worst means achieving the lowest/highest life cycle air pollution damages. Air pollution damages are calculated using the AP2 model.

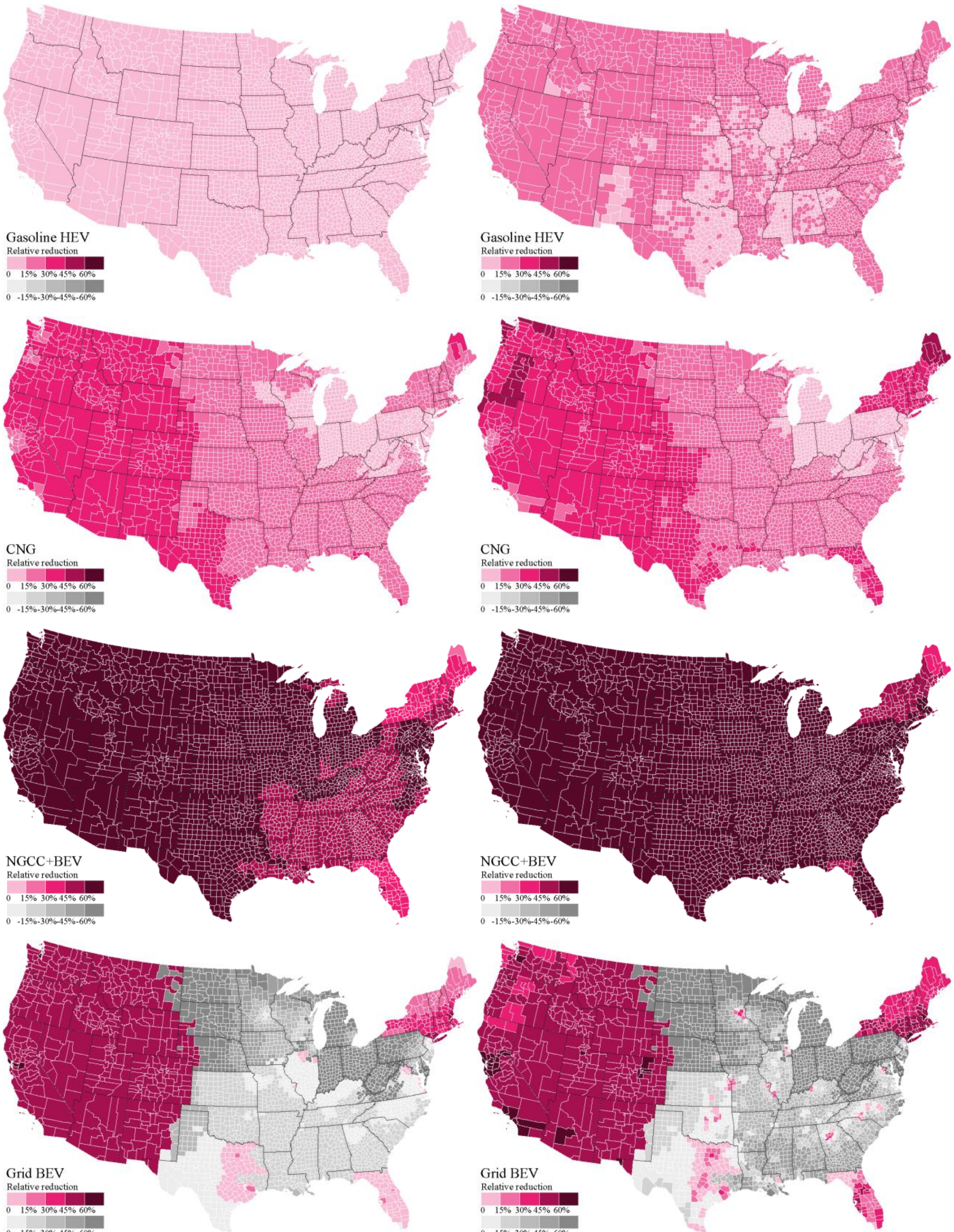
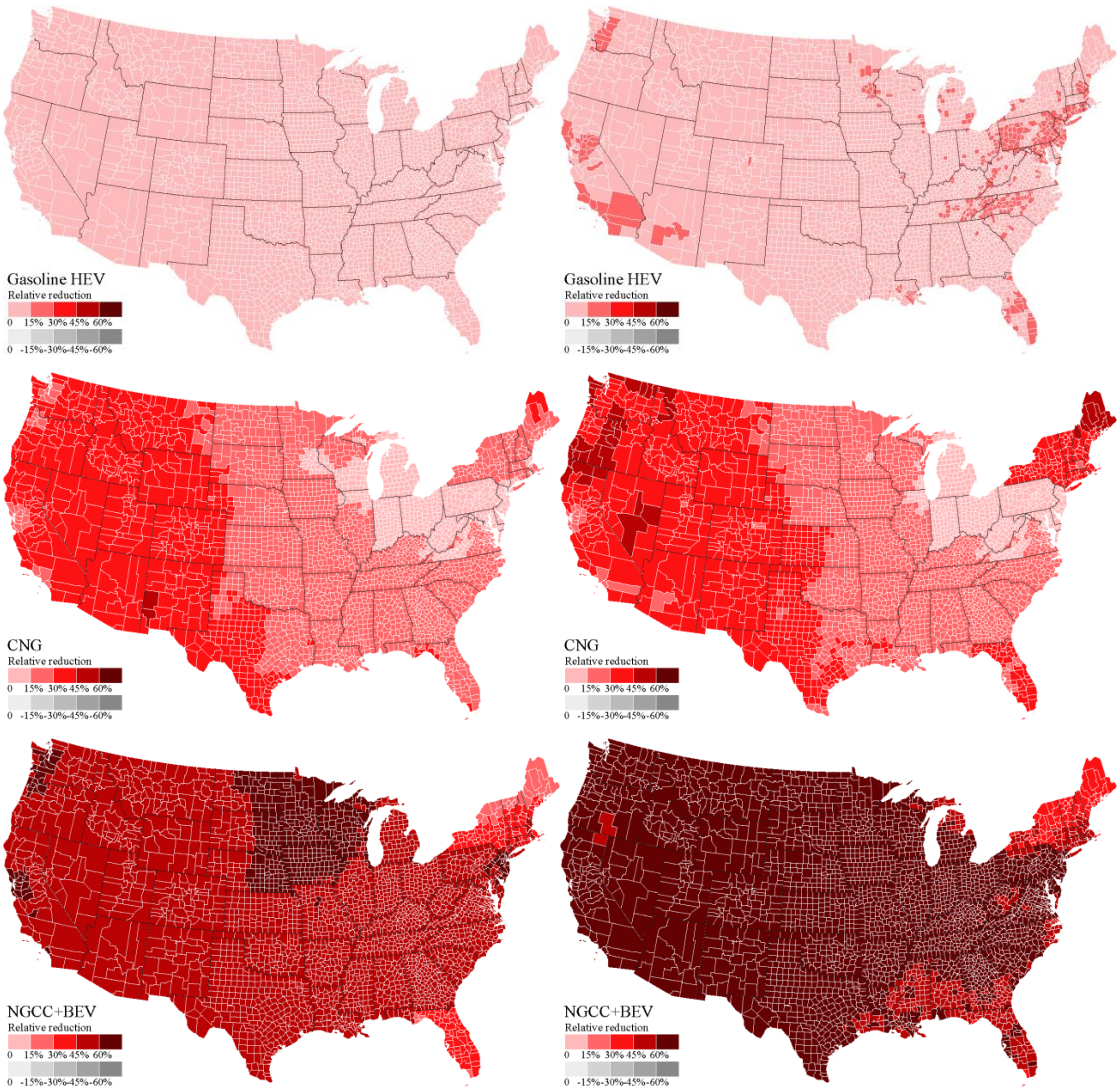

Gasoline HEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Gasoline HEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
CNG
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
CNG
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
NGCC+BEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
NGCC+BEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Grid BEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Grid BEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%

Figure C.18. Relative reduction in life cycle air pollution damages from replacing conventional gasoline with alternative fuel pathways for passenger cars using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional gasoline.



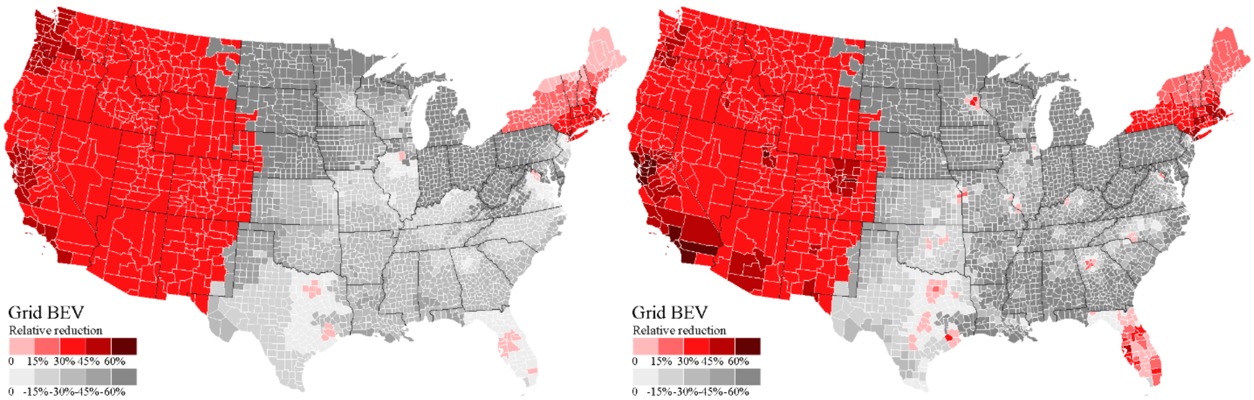
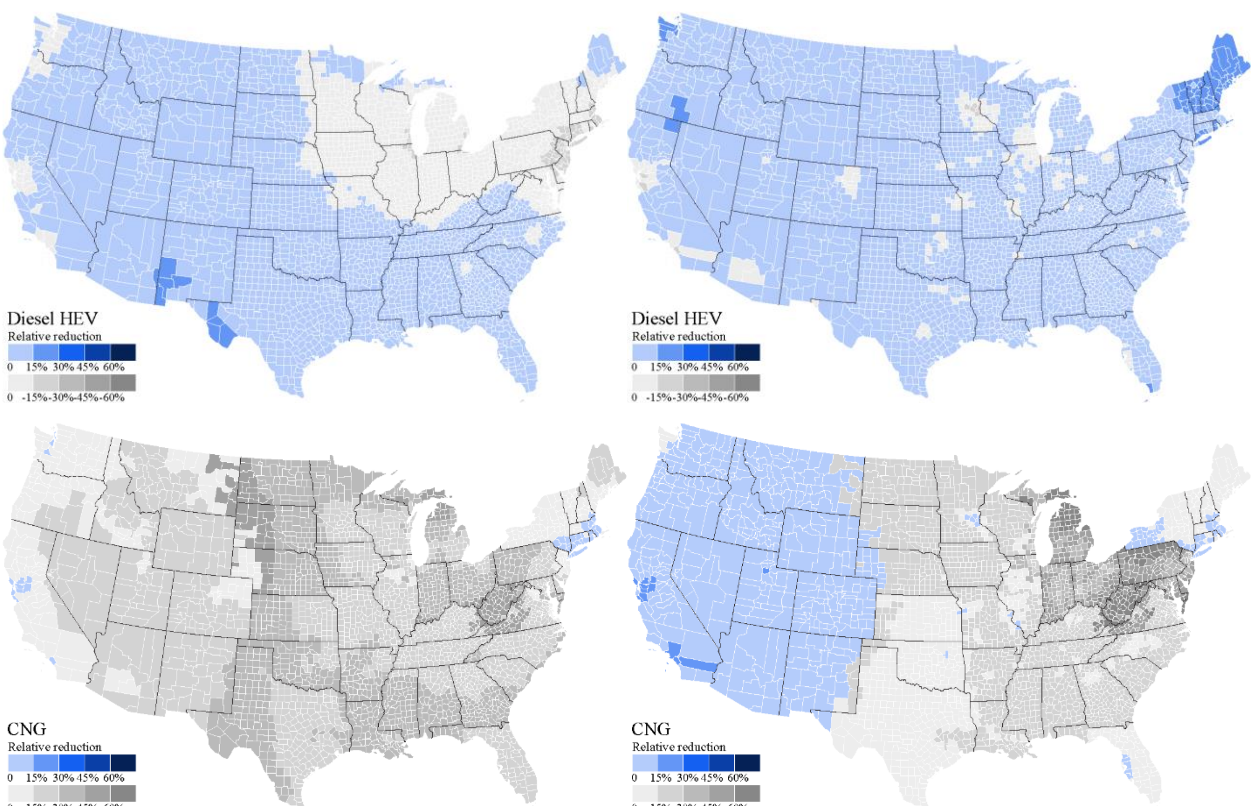


Figure C.19. Relative reduction in life cycle air pollution damages from replacing conventional gasoline with alternative fuel pathways for SUVs using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional gasoline.



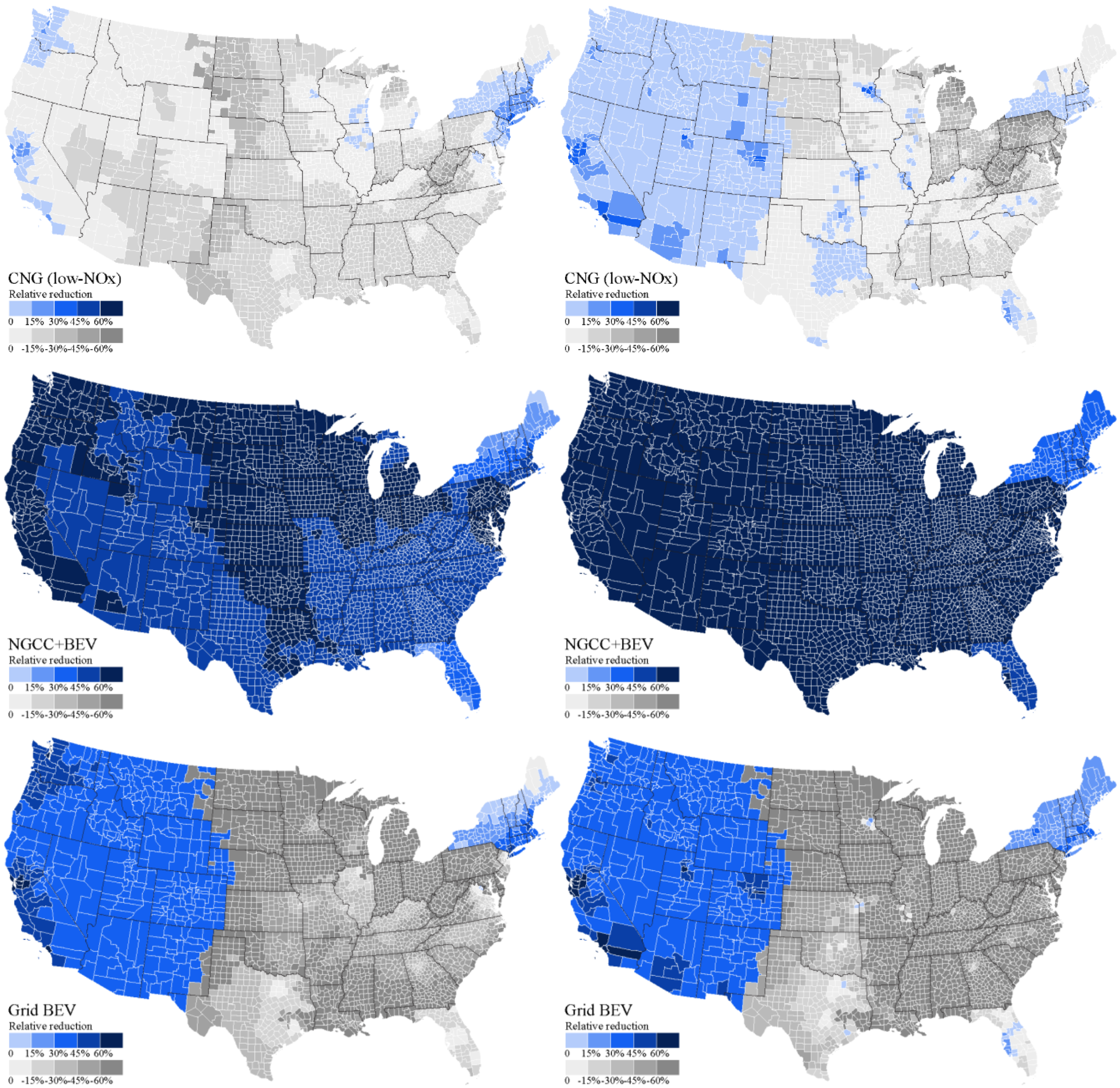
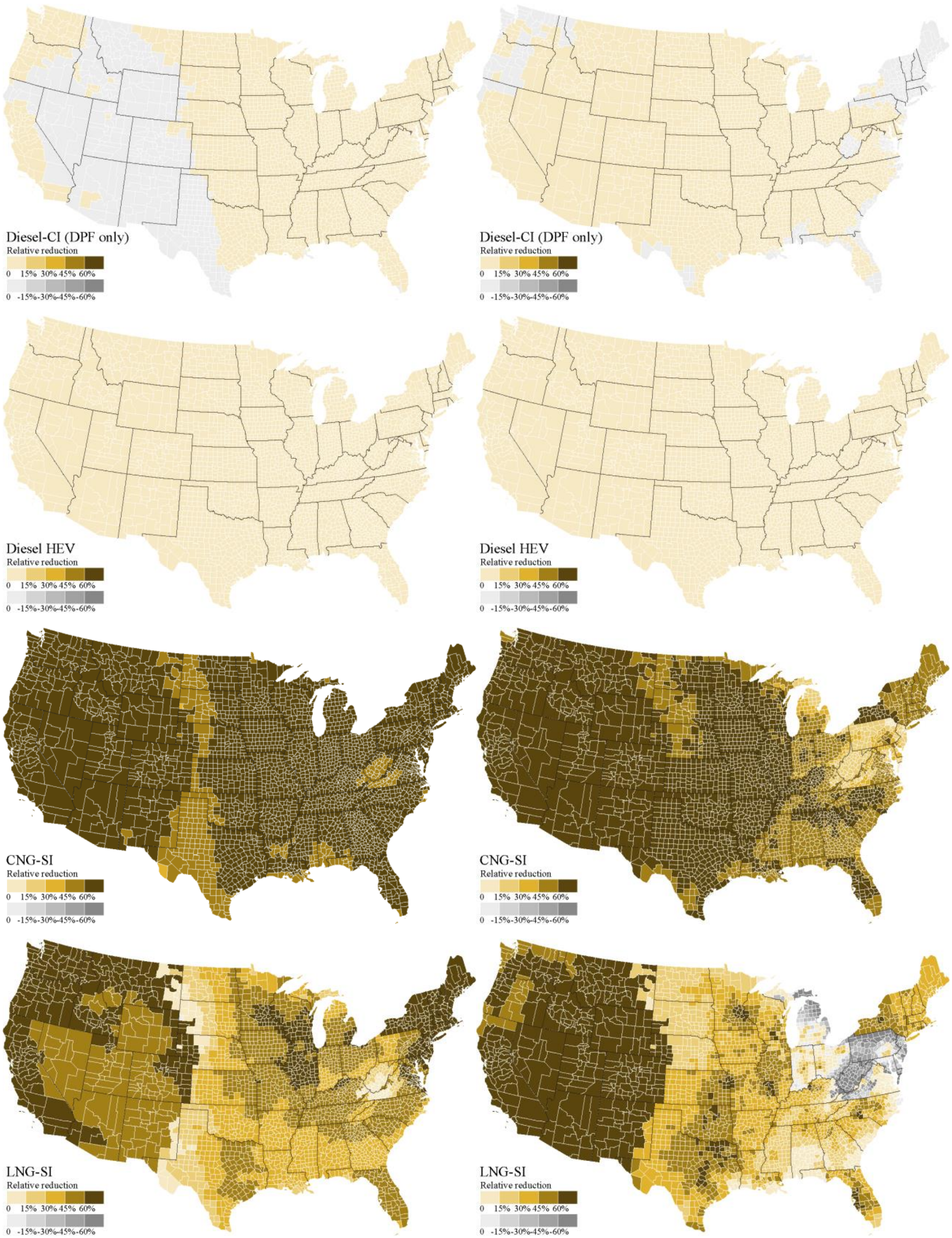


Figure C.20. Relative reduction in life cycle air pollution damages from replacing conventional diesel with alternative fuel pathways for transit buses using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.


Diesel-CI (DPF only)
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Diesel-CI (DPF only)
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Diesel HEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
Diesel HEV
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
CNG-SI
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
CNG-SI
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%
LNG-SI
Relative reduction
0 15% 30% 45% 60%
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LNG-SI
Relative reduction
0 15% 30% 45% 60%
0 -15% -30% -45% -60%

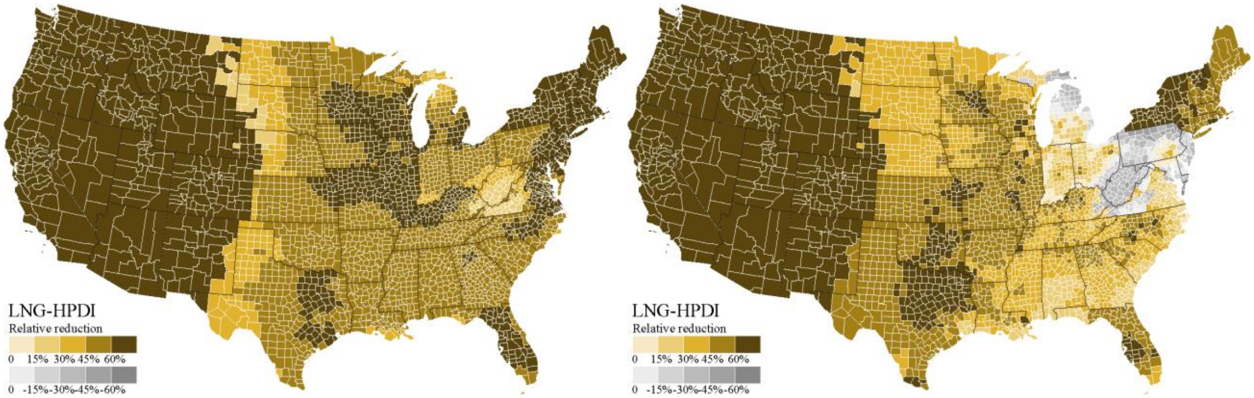
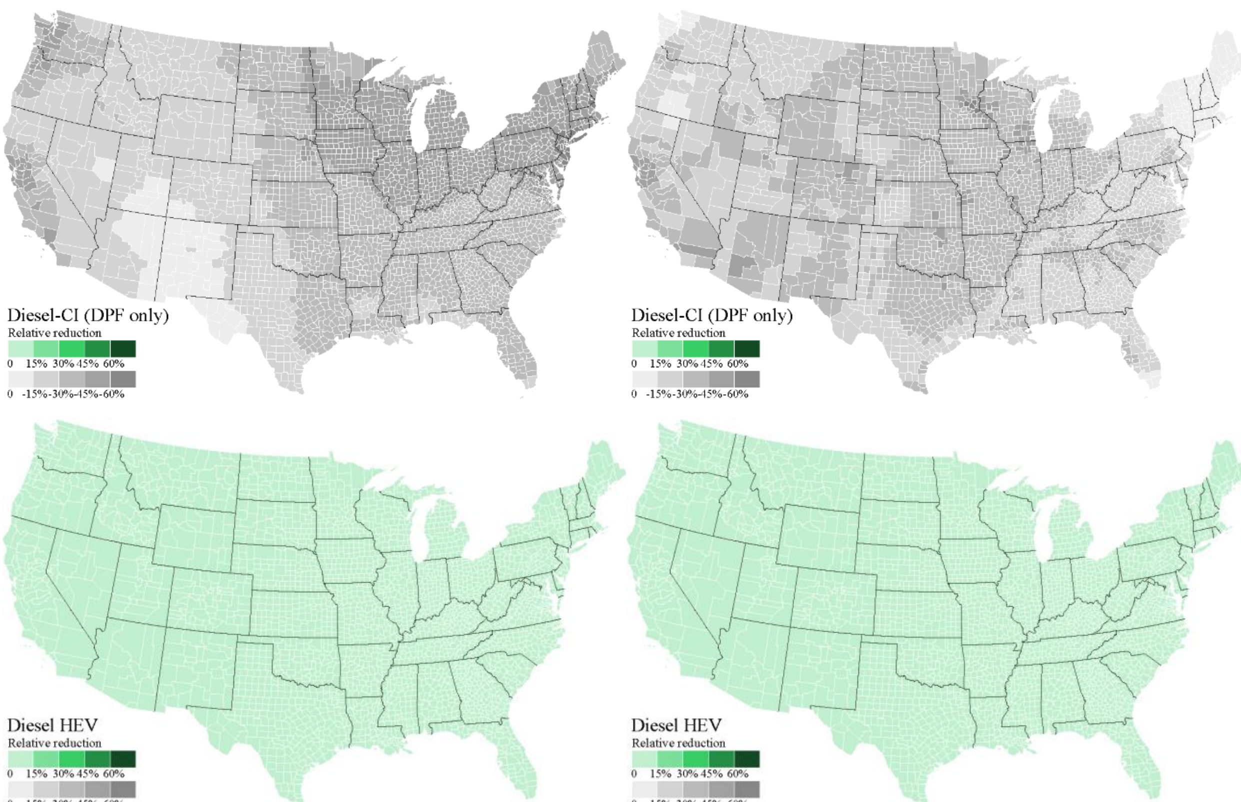


Figure C.21. Relative reduction in life cycle air pollution damages from replacing conventional diesel with alternative fuel pathways for local-haul tractor-trailers using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.



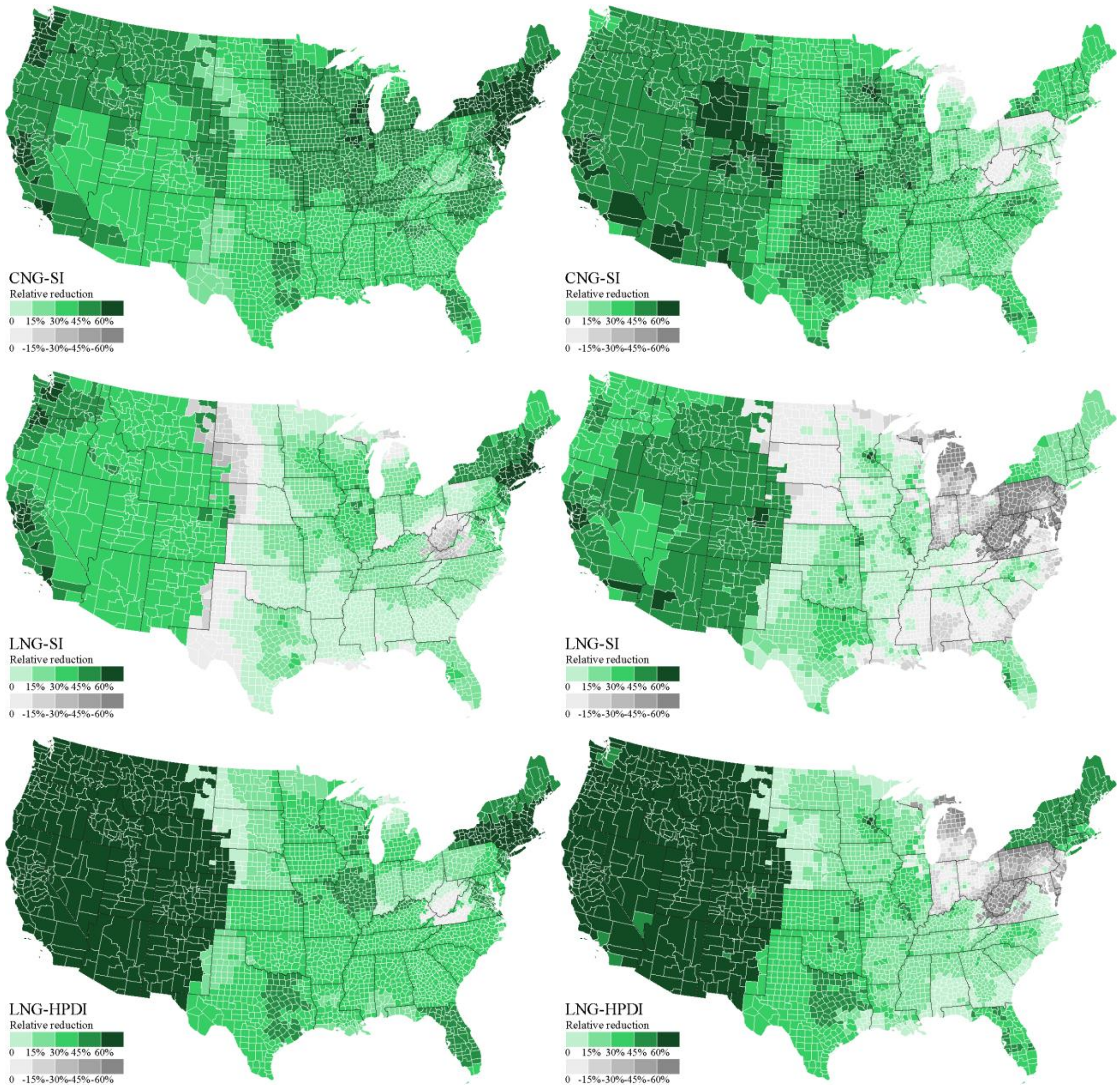


Figure C.22. Relative reduction in life cycle air pollution damages from replacing conventional diesel with alternative fuel pathways for long-haul tractor-trailers using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.

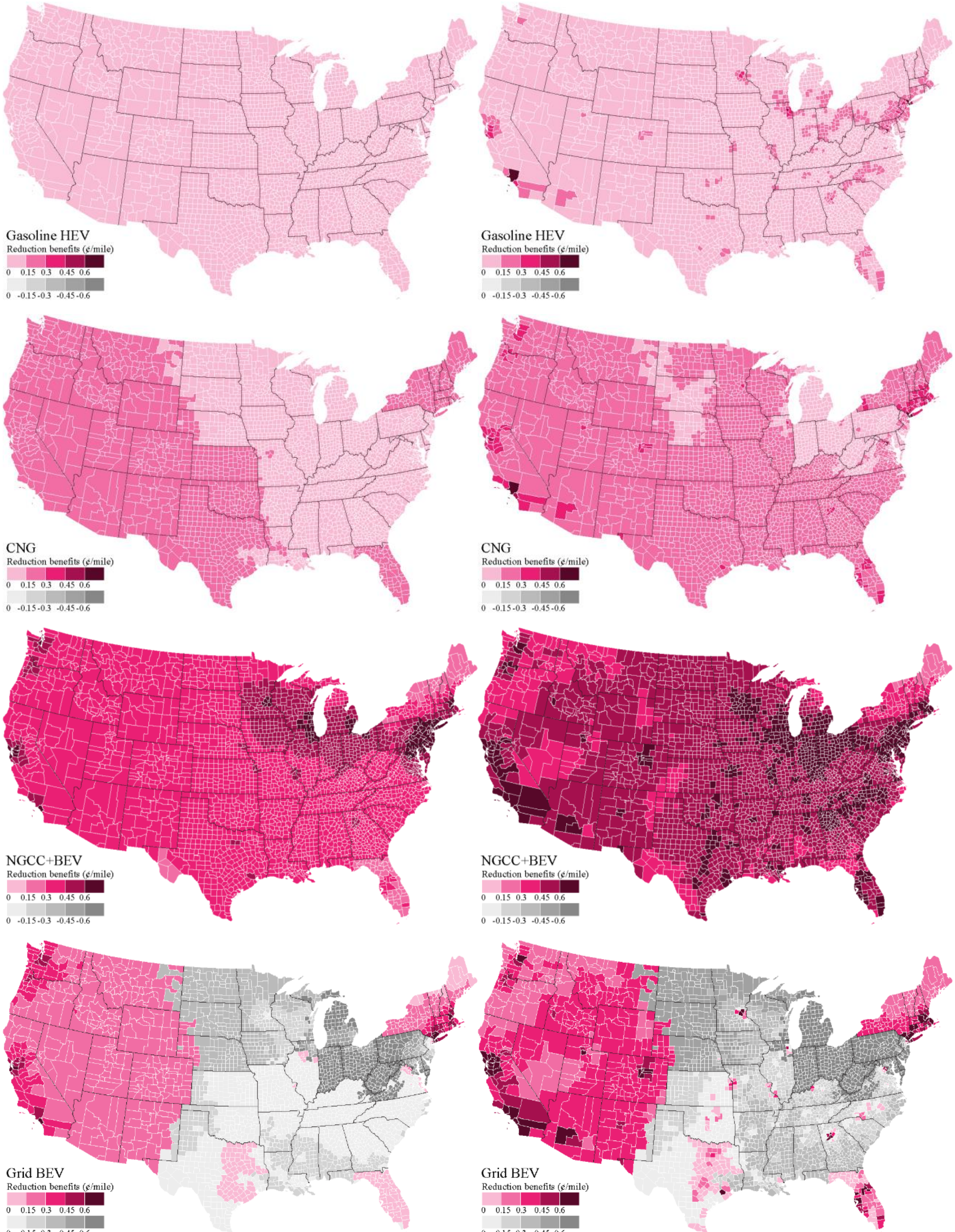
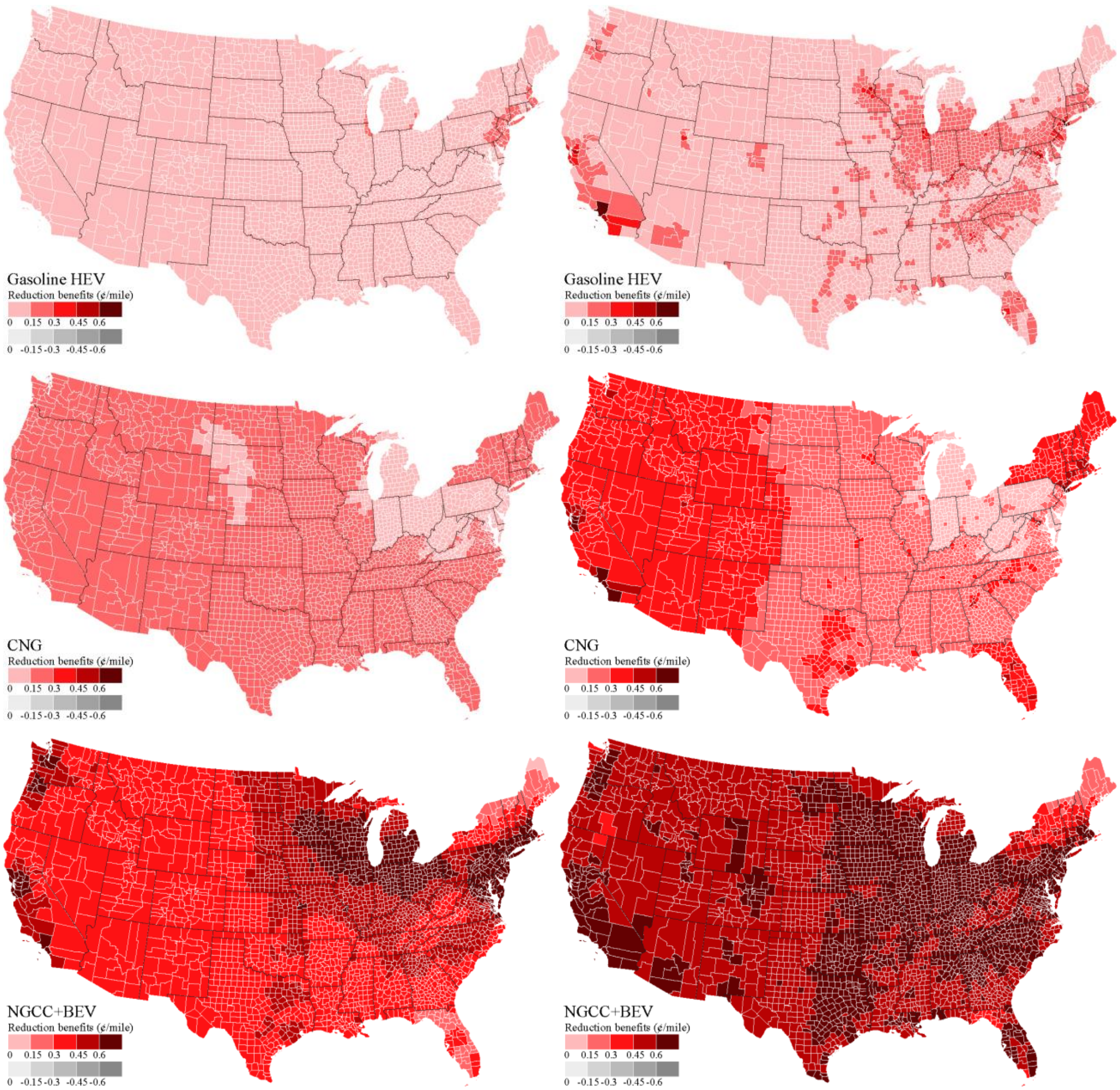

Gasoline HEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
Gasoline HEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
CNG
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
CNG
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
NGCC+BEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
NGCC+BEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
Grid BEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6
Grid BEV
Reduction benefits (¢/mile)
0 0.15 0.3 0.45 0.6
0 -0.15 -0.3 -0.45 -0.6

Figure C.23. Life cycle air pollution damage reduction benefits from replacing conventional gasoline with alternative fuel pathways for passenger cars using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional gasoline.



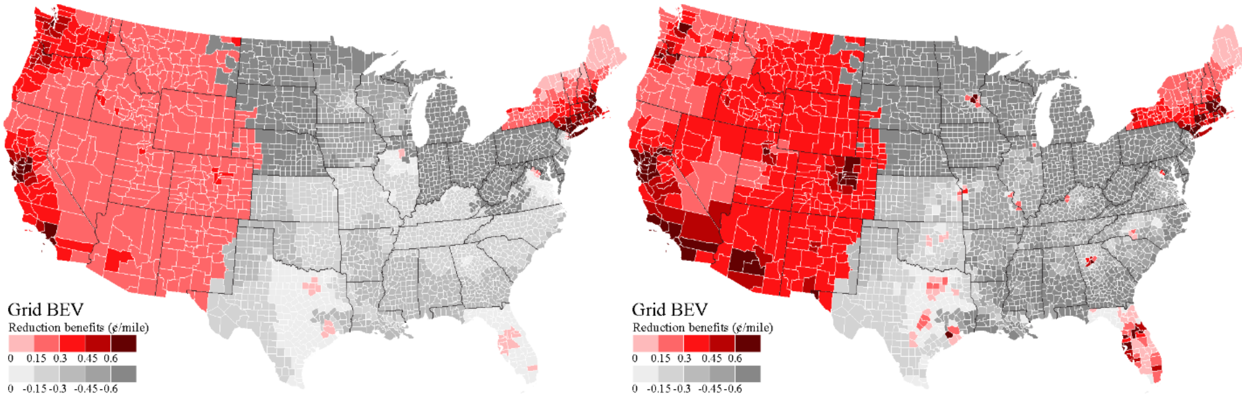
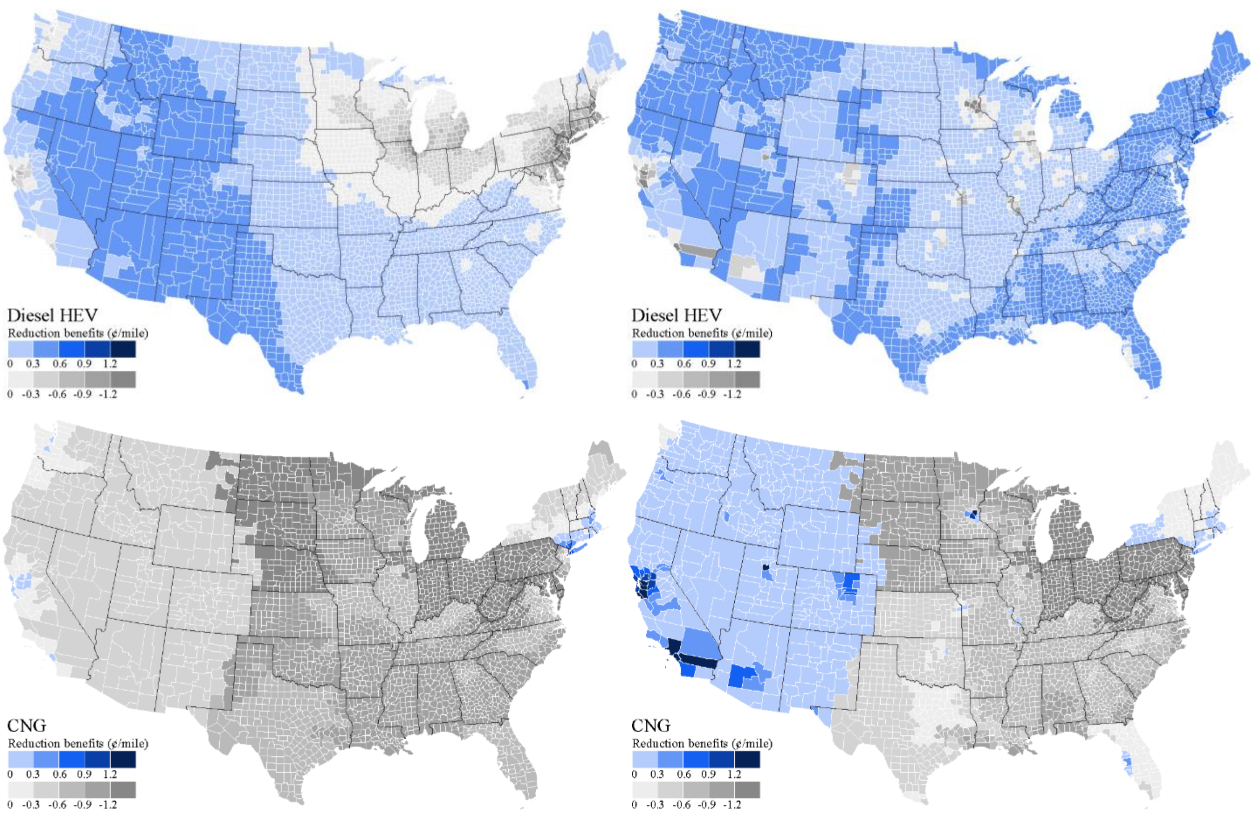


Figure C.24. Life cycle air pollution damage reduction benefits from replacing conventional gasoline with alternative fuel pathways for SUVs using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional gasoline.



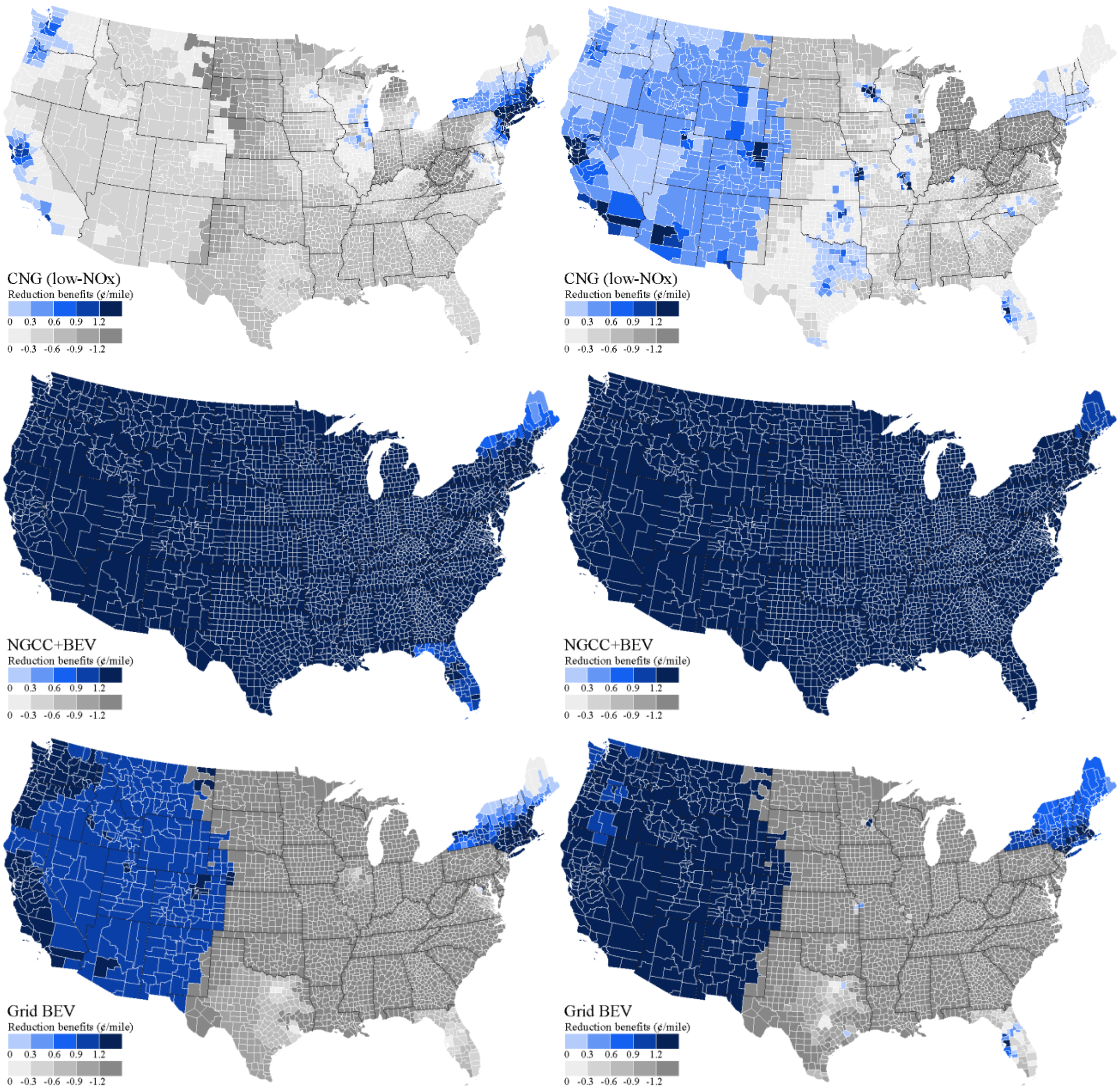
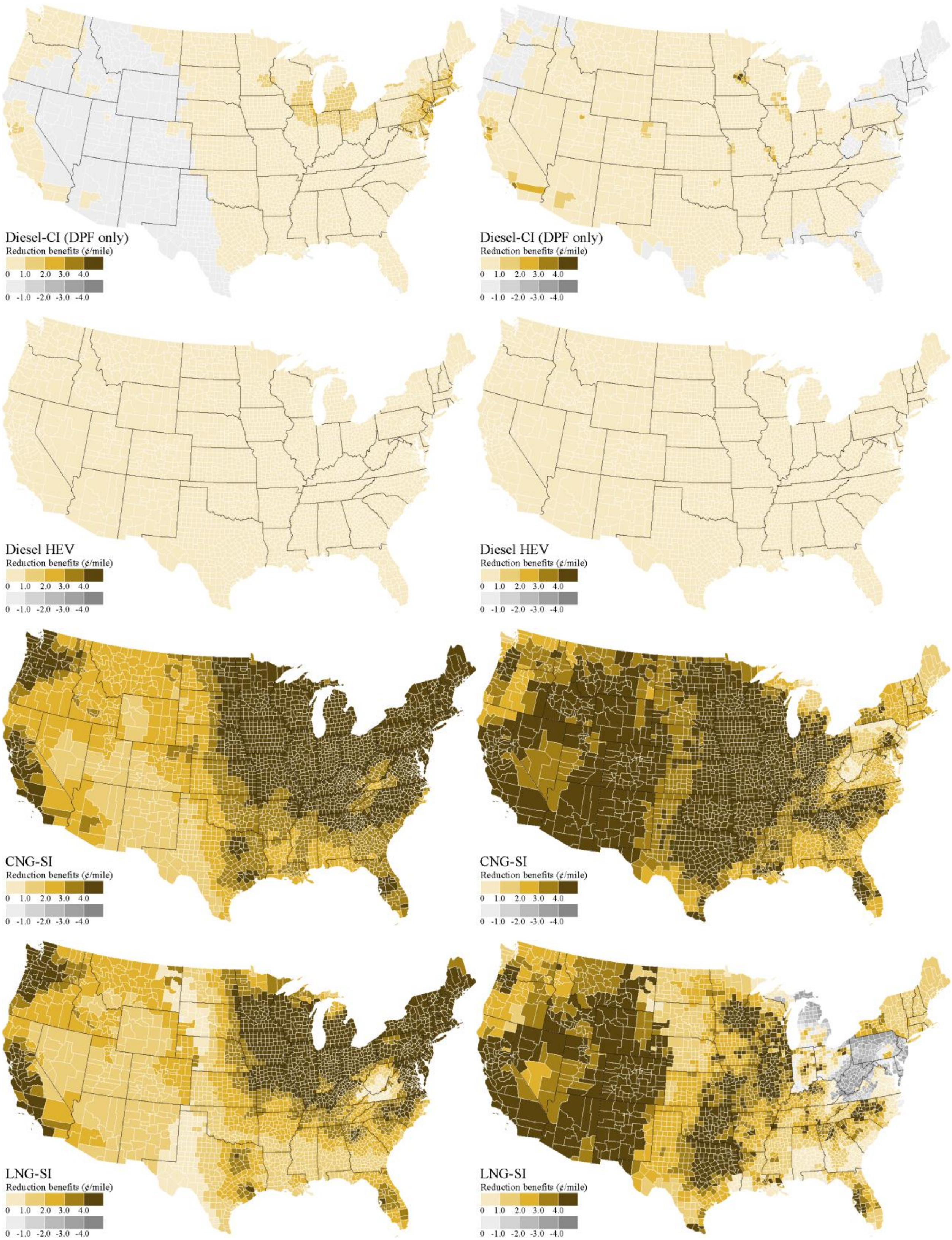


Figure C.25. Life cycle air pollution damage reduction benefits from replacing conventional diesel with alternative fuel pathways for transit buses using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.


Diesel-CI (DPF only)
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
Diesel-CI (DPF only)
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
Diesel HEV
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
Diesel HEV
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
CNG-SI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
CNG-SI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
LNG-SI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0
LNG-SI
Reduction benefits (¢/mile)
0 1.0 2.0 3.0 4.0
0 -1.0 -2.0 -3.0 -4.0

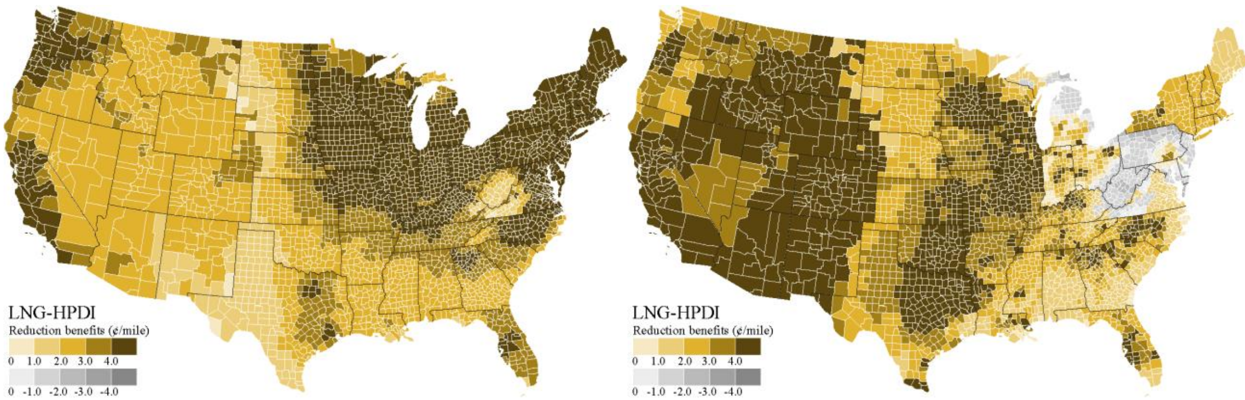
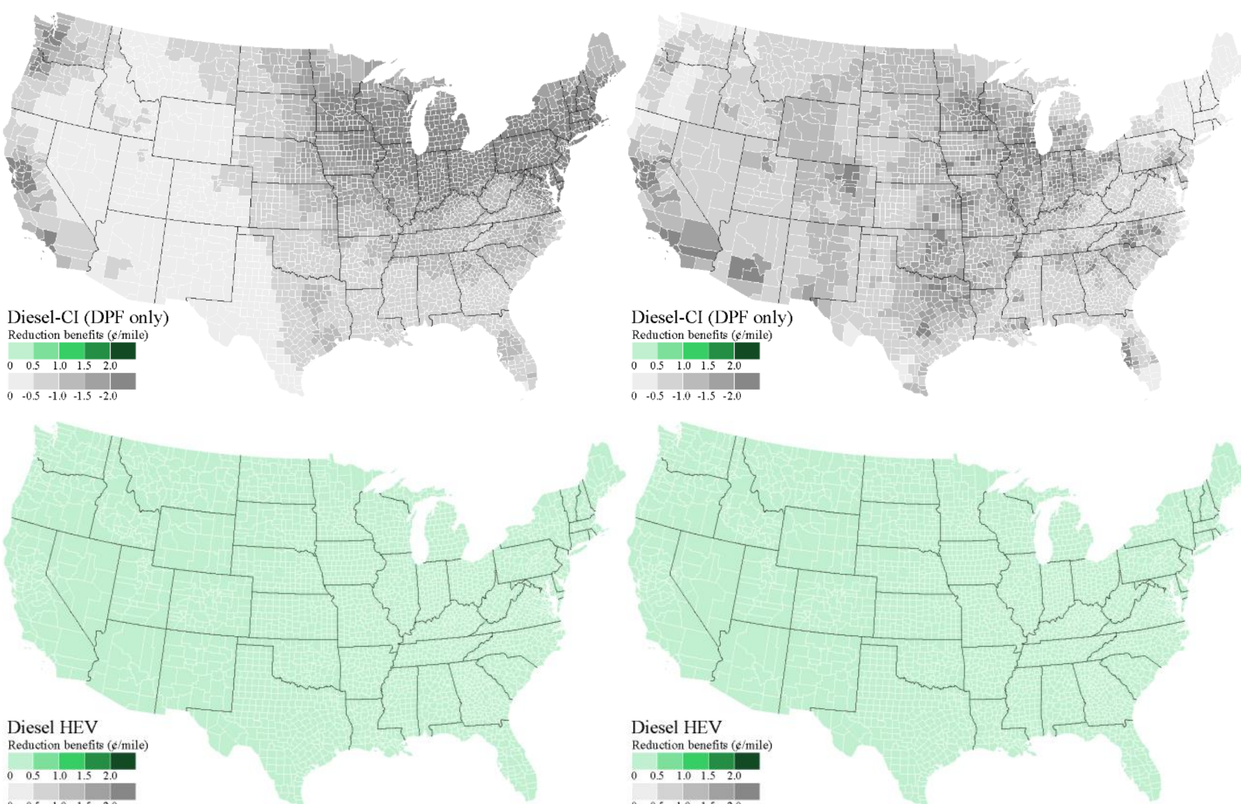


Figure C.26. Life cycle air pollution damage reduction benefits from replacing conventional diesel with alternative fuel pathways for local-haul tractor-trailers using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional diesel.



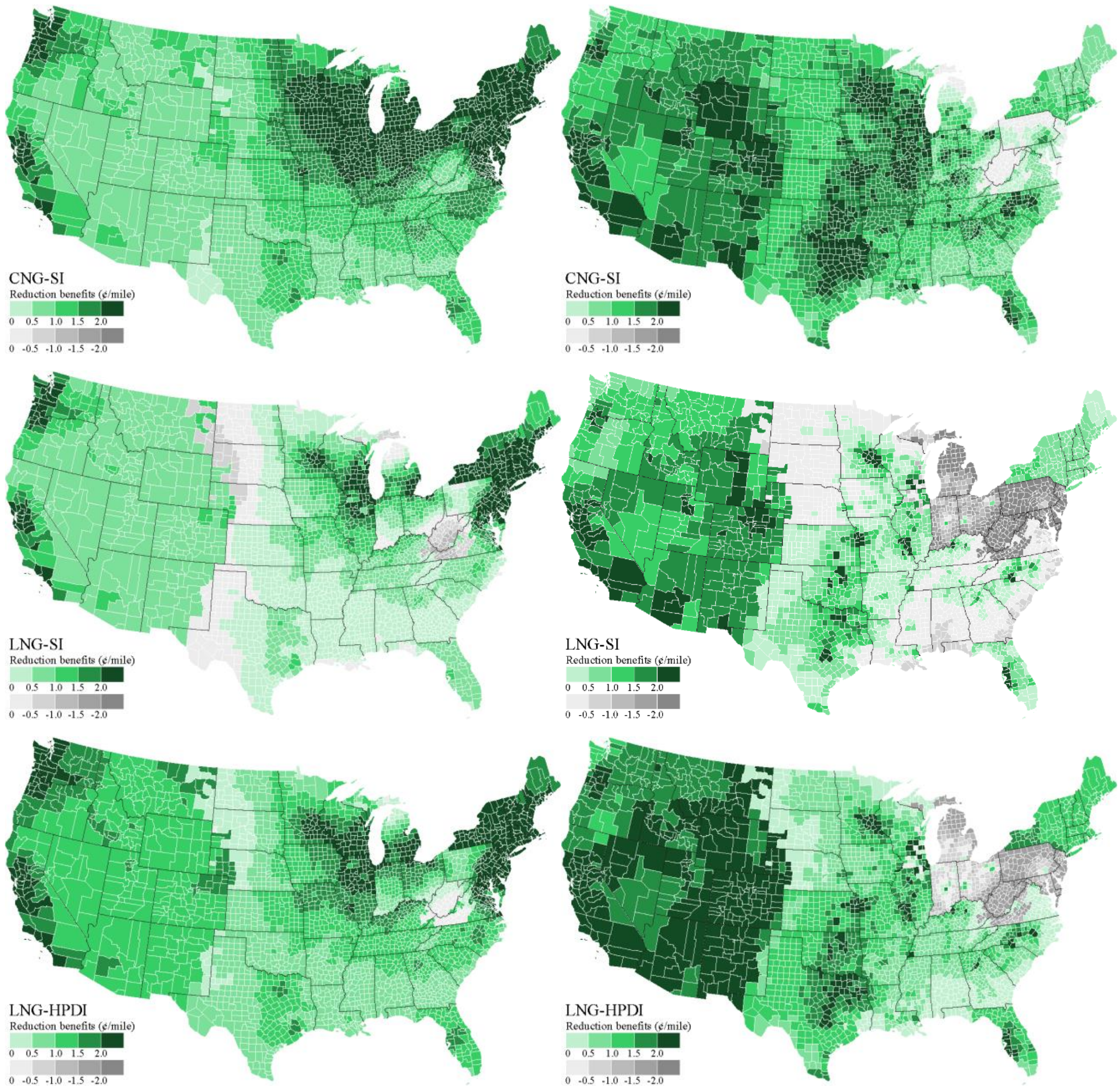


Figure C.27. Life cycle air pollution damage reduction benefits from replacing conventional diesel with alternative fuel pathways for long-haul tractor-trailers using emissions data from GREET model. Left panel: damages are based on the EASIUR model; right panel: damages are based on the AP2 model. Negative values represent low life cycle damages from conventional gasoline.

Appendix D. Supporting Information for Chapter 5

D.1. Comparison of Fuel Prices

Figure D.1 compares historical price trends between diesel and LNG. The fuel price differential between diesel and LNG stays around $1-$1.5/DGE between early 2011 and late 2014. But the price differential can be as low as -$0.5/DGE (early 2016), which is not a good sign for natural gas trucks. Indeed, a necessary condition of for the trucking industry to make a transition to natural gas fuels is a positive price differential between diesel and LNG. I find that the volatility of diesel price (which traces back to the volatility of crude oil price) drives the changes in the fuel price differential between diesel and LNG. Changes in the federal exercise tax, and the opportunity to apply tax breaks could also change the price differential between diesel and natural gas fuels. **Figure D.1** also shows the composition of the LNG fuel price. Essentially, the refueling industry buys LNG from fuel suppliers at a price that pays for production and delivery costs and profit margins of fuel producers. The refueling industry then sells LNG at the retail price which adds operation costs, fuel price margins, and the exercise tax. Since it is easier to make CNG than LNG, CNG is generally cheaper than LNG.[7,234]

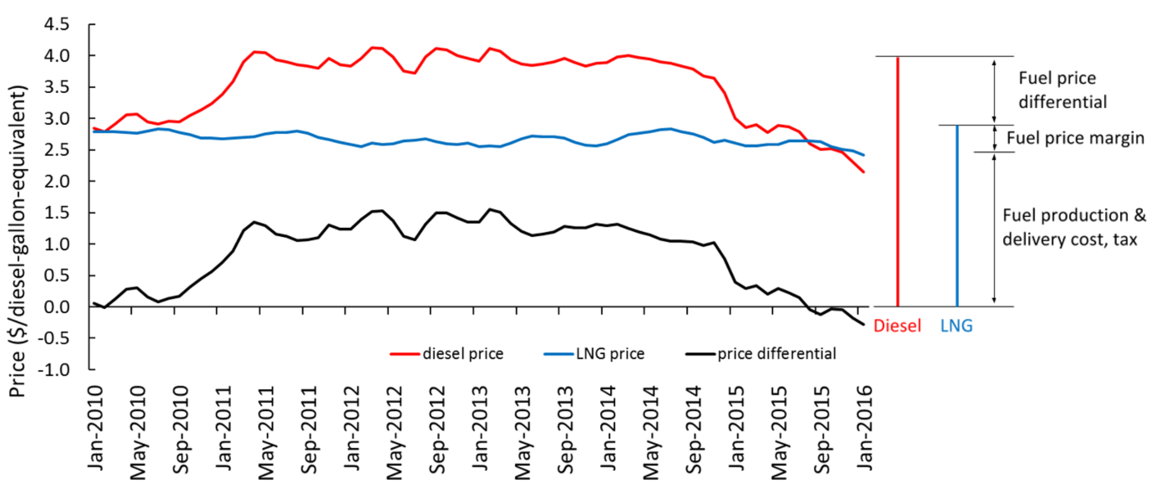


Figure D.1. U.S. average diesel price, reconstructed LNG price and price differentials between diesel and LNG in 2010-2016. The figure on the right illustrates fuel price

differential between diesel and LNG, fuel price margin (net profit for refueling stations), and fuel production & delivery cost as well as tax.

In **Figure D.1**, I get diesel price estimates from U.S. Energy Information Administration (EIA).[379] The LNG price is estimated using Eqn. D.1 since U.S. EIA only provides LNG price data recently. I use the natural gas commercial price to estimate the price of natural gas feedstock.[380] I assume that the processing and delivery cost is \$0.9/diesel gallon equivalent (DGE).[7] I assume the market markup is \$0.3/DGE. The federal exercise tax was \$0.413/DGE for LNG until the end of 2015 and becomes \$0.243/DGE starting 2016 (the same as diesel).[381]

$$\begin{aligned} &\textit{LNG price} = \\ &\quad \textit{natural gas commercial price } (\$/\textit{MMBtu}) \;/\; 8 + \\ &\quad \textit{processing } \& \textit{ delivery cost} + \\ &\quad \textit{market markup} + \\ &\quad \textit{federal tax.} \end{aligned} \tag{D.1}$$

D.2. Refueling Demands at Refueling Stations

Figure D.2 shows the distribution of refueling demands faced by natural gas refueling stations at three adoption rates (1%, 5%, and 10%), as well as the distribution of fuel sales at existing diesel refueling stations by an industry survey.[218] The histograms of refueling demands are highly right-skewed. The majority of refueling stations have similar refueling demands while a small fraction of refueling stations face high refueling demands that are five times higher than the mode. I find that all high-demand refueling stations are located at highway intersections, which confirms my hypothesis. In particular, I find Nashville (where I-24, I-40, I-65, and I-81 intersect) and Dallas (where I-20, I-30, I-35, and I-45 intersect) see the highest refueling demands consistently. The distributions of refueling demands in my model are different from that of diesel refueling stations. This is due to the modeling assumption of bundling refueling modules at one location together. If I allow several refueling stations to exist and compete at one location, I would see a distribution closer to that of existing diesel refueling stations.

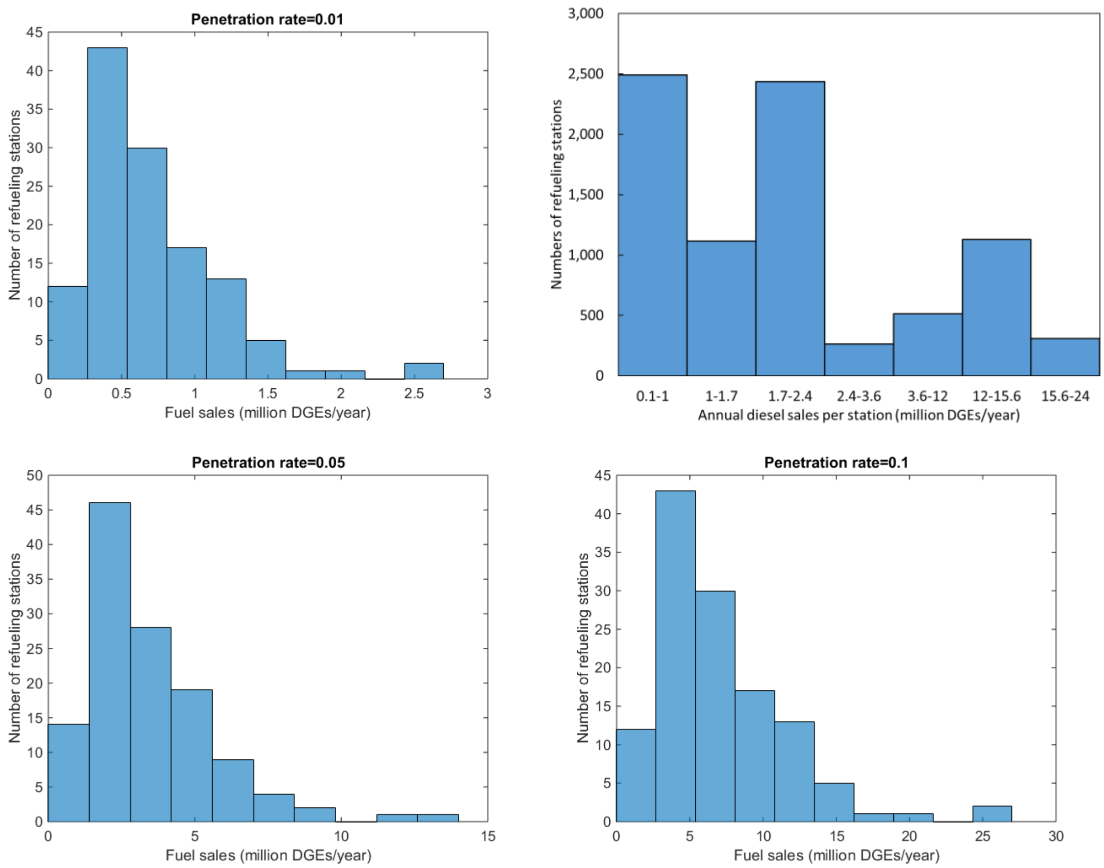


Figure D.2. Distribution of refueling demands faced by natural gas refueling stations at adoption rates of 1% (top-left), 5% (bottom-left), and 10% (bottom-right). Top-right shows the annual diesel sales at existing diesel refueling stations that have an annual throughput larger than 0.1 million DGEs per year (TIAX, 2012).[218] Note that the surveyed diesel refueling stations are not always along highways.

Appendix E. Supporting Information for Chapter 6

E.1. Life Cycle GHG emissions for Transit Buses

I relied on *Chapter 3* and Tong et al. (2015)[146] to estimate life cycle GHG emissions from diesel and alternative fuel transit buses, as shown in **Table E.1** and **Table E.2**. Climate change damages are then estimated using Eqn. 6.3 in *Section 6.3.3 Life Cycle Social Costs*.

Table E.1. Life cycle GHG emissions of diesel and alternative fuel 40-foot transit buses. Unit: gCO_2-eq/mile. Source: *Chapter 3* and Tong et al. (2015).[146]

Fuel pathway	Upstream	Tailpipe (Combustion)	Tailpipe (Non-combustion)	Battery manufacturing	Life cycle
100-year Global Warming Potential (GWP)					
Conventional diesel	494	2089	2	0	2585
Diesel HEB	412	1741	2	1	2155
CNG	782	1746	16	0	2544
LNG	978	1746	96	0	2820
Rapid-charging BEB	1095	0	0	10	1106
Slow-charging BEB	1281	0	0	38	1319
20-year Global Warming Potential (GWP)					
Conventional diesel	494	2089	2	0	2585
Diesel HEB	412	1741	2	1	2155
CNG	1195	1746	16	0	2957
LNG	1302	1746	96	0	3144
Rapid-charging BEB	1203	0	0	10	1214
Slow-charging BEB	1407	0	0	38	1445

Table E.2. Life cycle GHG emissions of diesel and alternative fuel 60-foot transit buses. Unit: gCO_2-eq/mile. Source: *Chapter 3* and Tong et al. (2015).[146]

Fuel pathway	Upstream	Tailpipe (Combustion)	Tailpipe (Non-combustion)	Battery manufacturing	Life cycle
100-year Global Warming Potential (GWP)					
Conventional diesel	719	3039	2	0	3759

Diesel HEB	599	2533	2	1	3134
CNG	1121	2502	16	0	3640
LNG	1402	2502	96	0	4000
Rapid-charging BEB	1593	0	0	12	1605
Slow-charging BEB	1863	0	0	44	1908
20-year Global Warming Potential (GWP)					
Conventional diesel	719	3039	2	0	3759
Diesel HEB	599	2533	2	1	3134
CNG	1713	2502	16	0	4232
LNG	1867	2502	96	0	4465
Rapid-charging BEB	1750	0	0	12	1762
Slow-charging BEB	2046	0	0	44	2091

E.2. CAP emission estimates and marginal damages

Life cycle air pollution damages are calculated using Eqn. 6.4-6.5 in Section *6.3.3. Life Cycle Social Costs.* Key inputs for Eqn. 6.4-6.5 are presented in **Table E.3**, **Table E.4**, and **Table E.5**. In addition, social costs of CAP emissions from battery manufacturing are $\$_{2015}$ 8.7/kWh, and marginal damages of CO are $\$_{2015}$ 878/metric ton. See Chapter 4 and Tong et al. (2016)[253] for details.

Table E.3. Vehicle operation CAP emissions from transit buses. Unit: gram/mile. Due to data availability, I assume CAP emissions from vehicle operation are the same for a 40-foot and a 60-foot transit bus. Source: Chapter 4 and Tong et al. (2016).[253]

Air pollutant	Diesel	Diesel HEB	CNG	LNG	BEB
$PM_{2.5}$	0.0335	0.0335	0.0335	0.0335	0.0124
SO_2	0.0160	0.0114	0.0093	0.0000	0.0000
NO_x	0.9175	1.4450	0.5775	0.5775	0.0000
VOC	0.1121	0.0787	0.0695	0.0695	0.0210
CO	0.4900	0.1850	31.2750	31.2750	0.0000

Table E.4. Marginal social damages of CAP emissions from ground-level sources in Allegheny County, PA.[30,31,57,195] Unit: $\$_{2015}$/metric ton.

Marginal damage model	$PM_{2.5}$	SO_2	NO_x	VOC
AP2 model	$270,596	$84,823	$5,422	$25,912
EASIUR model	$272,885	$27,439	$13,309	N/A

Table E.5. Marginal social damages of CAP emissions from upstream activities of fuel pathways for Allegheny County, PA. Unit: $\$_{2015}$/MJ. Source: Chapter 4 and Tong et al. (2016).[253]

Life cycle stage	Diesel [a]	CNG [b]	LNG [b]	Electricity [c]
AP2 model				
Energy/feedstock production & transportation	0.01	0.01	0.01	0.01
Fuel production & transportation	0.01	0.05	0.13	1.18
Upstream total	0.02	0.06	0.14	1.19
EASIUR model				
Energy/feedstock production & transportation	0.01	0.01	0.01	0.01
Fuel production & transportation	0.01	0.05	0.14	1.24
Upstream total	0.02	0.06	0.15	1.25

Note: a. Social damages from diesel are estimated for U.S.-average diesel due to data availability. b. Social damages from CNG and LNG are estimated for the ReliabilityFirst (RF) region as defined by the North Amercian Electric Reliability Corporation (NERC). ReliabilityFirst (RF) region includes Midwest/Mid-Atlantic states such as DE, IN, MD, MI, NJ, OH, PA, WV and parts of IL, KY, VA, and WI. See Appendix C for geographical boundaries. c. Social damages from electricity are estimated for average electricity delivered in the ReliabilityFirst (RF) region.

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