Carnegie Mellon University

CARNEGIE INSTITUTE OF TECHNOLOGY

THESIS

SUBMITTED IN PARTIAL FULFILLMENT OF THE REQUIREMENTS

FOR THE DEGREE OF Doctor of Philosophy

TITLE **Quantitative Modeling Under Uncertainty to Inform Effective Energy and Environmental Policies**

PRESENTED BY **Leslie Abrahams**

ACCEPTED BY THE DEPARTMENTS OF

Engineering and Public Policy & Civil and Environmental Engineering

W. Michael Griffin	April 28, 2016
ADVISOR, MAJOR PROFESSOR	DATE
Douglas Sicker	May 2, 2016
DEPARTMENT HEAD	DATE

APPROVED BY THE COLLEGE COUNCIL

Vijayakumar Bhagavatula	May 5, 2016
DEAN	DATE

Quantitative Modeling Under Uncertainty to Inform Effective Energy and Environmental Policies

Submitted in partial fulfillment of the requirements for the degree of
Doctor of Philosophy
in
Engineering & Public Policy
Civil & Environmental Engineering

Leslie Abrahams
B.S. in Chemical Engineering, Washington University in St. Louis
M.S. in Engineering & Public Policy, Carnegie Mellon University

Carnegie Mellon University
Pittsburgh, Pennsylvania 15213

May 2016

DOCTORAL THESIS COMMITTEE

Thesis Chair

W. Michael Griffin
Associate Research Professor, Engineering and Public Policy
CARNEGIE MELLON UNIVERISTY

Thesis Advisor

H. Scott Matthews
Professor, Civil and Environmental Engineering
CARNEGIE MELLON UNIVERISTY

Thesis Advisor

Constantine Samaras
Assistant Professor, Civil and Environmental Engineering
Carnegie Mellon University
CARNEGIE MELLON UNIVERISTY

Thesis Reader

Paulina Jaramillo
Assistant Professor, Engineering and Public Policy
CARNEGIE MELLON UNIVERISTY

Acknowledgments

This research was made possible through support from The National Science Foundation Graduate Research Fellowship Program under Grant No. (DGE1252522), the Climate and Energy Decision Making (CEDM) center through a cooperative agreement between the National Science Foundation (SES- 0949710), and the Carnegie Mellon Department of Engineering and Public Policy.

Thank you to my advisor and committee chair, Mike Griffin, for his dedication to his students and his commitment to thoughtful, thorough, and meaningful research. Thank you also to my co-advisors, Scott Matthews and Cost Samaras, for their insights and contributions to this work. I would also like to thank my thesis reader, Paulina Jaramillo, for her time and helpful perspective.

Finally, thank you to my family for their unwavering support and encouragement and to my friends for being such a special group of thoughtful, selfless, intelligent, and fun individuals.

Abstract

Natural gas production in the United States has increased significantly over the past decade. This is largely due to advancements in hydraulic fracturing, horizontal drilling, and seismic monitoring capabilities that have enabled extraction of natural gas from shale resources to be economically viable. While natural gas is an important global energy resource and may result in fewer emissions than coal for electricity generation, it is important to recognize that extraction of natural gas has the potential to cause local and regional environmental damages. Successfully managing these risks is critical in order to ensure natural gas consumption has a net positive environmental footprint. The second and third chapters of this dissertation use quantitative modeling to assess how policies can address and mitigate potential environmental impacts in a cost-effective manner. Specifically, this work focuses on minimizing incremental fragmentation in critical core forest ecosystems resulting from natural gas infrastructure and on managing wastewater byproducts from natural gas extraction.

The second chapter finds that in the case study of a core forest region in Bradford County, Pennsylvania, the number of core patches of forest, an indicator of fragmentation, could as much as double throughout the life time of the Marcellus Shale play (from 80 to 160 core patches) without any regulatory intervention. However, through unitization and collaborative planning, and by designating that gathering pipelines must follow the route of pre-existing roads in forests whenever possible, natural gas infrastructure can be developed in a manner that would both prevent incremental fragmentation from occurring and reduce pipeline construction costs for operators as a result of reduced infrastructure redundancies. The third chapter finds that approximately 1.3 million gallons of wastewater, called produced water, are generated by each well. Across Pennsylvania, 67% of the time Class II disposal is the least cost option, 25% of the time CWT is the least cost option, and 8% of the time on-site treatment is the least cost option. The corresponding average costs are \$5.80/bbl (\$0.015/Mcf), \$7.80/bbl (\$0.020/Mcf), and \$8.40/bbl (\$0.021/Mcf), respectively. In addition to cost, however, there are several technical, ecological, regulatory, and logistical issues that also affect the relative feasibility of these three produced water management strategies. If regulators

could capture producers willingness to pay to dispose of water rather than treat the water, that money could be invested in treating other water quality issues in Pennsylvania such as coal mine drainage, which can be treated for $0.064/bbl on average, or agricultural runoff, which could be prevented at an average cost of $0.08/bbl.

The last two chapters in this dissertation explore how quantitative modeling can inform policy making on a national and global scale. Chapter 4 does this by characterizing the life cycle greenhouse gas impact of United States natural gas exports. This study finds that mean landed (pre-combustion) life cycle GHGs for exported U.S. LNG after regasification at the importing country were found to be 37 g CO_2-equiv/MJ with a range of 27 to 50. The net global impact of these emissions depends on the global warming potential time scale, methane leakage rate, end use, and the fuel it displaces. On a 100 year time scale, life cycle emissions from exported LNG were found on average to be 655 g CO_2-equiv/kWh for electricity generation, a 45% reduction over life cycle emissions from coal consumption. However, because of the spatial shift in emissions generation, although there is a global GHG benefit to US natural gas exports, the United States should consider the implications of this given that emissions calculations are based on CO_2-eq emitted within a country's borders rather than based on the net global impact of those emissions. The fifth chapter continues to explore international trade policy by focusing on the global crude trade as a case study. This chapter considers how shifting trade patterns can influence global costs and greenhouse gas emissions using a linear optimization model. The baseline 2014 crude trade system had a global cost of $3T and resulted in 16.5 Gt of CO_2. Minimizing by cost would save $6T and increase emissions by 4 Gt CO_2, while minimizing by emissions would increase cost by $0.5T and decrease emissions by 5.4 Gt. This chapter then explores the interaction between climate policies including carbon accounting methods, a designated global carbon cap, and unilateral country specific emissions allocations. There is a 40% higher allowable consumption under a strict global carbon cap without country-specific emissions allocations (1100 Mmt) than with country-specific emissions caps (770 Mmt). These results illustrate cooperative international climate policy could be more cost-effective in mitigating carbon emissions than countries acting individually.

Contents

List of Tables

List of Figures

Chapter 1. Introduction and Background

1.1 Motivation

Natural gas is often discussed as a 'bridge fuel' to a lower carbon energy system, replacing coal until a transition to large-scale reliable renewable energy sources is feasible. Although use of natural gas results in greenhouse gas (GHG) emissions, life cycle emissions from the natural gas electricity generation supply are estimated to generally be lower than those from coal electricity generation[1,2]. The viability of natural gas as a bridge fuel depends on assumptions about stabilization objectives, emissions, and technological change[3-6]. Additionally, natural gas extraction, especially the development of unconventional resources (shale gas), has other social costs such as air, water, and road impacts[7-9]. Therefore, the net environmental benefit of natural gas as a bridge fuel depends on policy's aptitude to successfully manage these local and regional environmental impacts. This dissertation uses quantitative modeling to demonstrate the capacity to mitigate environmental damages from natural gas extraction.

The Marcellus is the most expansive shale gas play in the United States[10], covering 14-25 million hectares (ha) (35-62 million acres) at a depth of 600-3,000 m[11]. The economically viable region of the Marcellus crosses southern New York, much of Pennsylvania, northern West Virginia, and eastern Ohio. Because of the continuous nature and vast expanse of the shale deposit, developers have some flexibility in locating wells. Currently in PA the vast majority of well pads are drilled with little consideration for land use preservation. In 2008 alone, half of the wells were developed in PA forests[12]. This destruction of core forest habitat poses a significant risk to forest ecosystem health and biodiversity. Additionally, deforestation results in an increase in impervious surface, which is a threat to stream health, the integrity of headwater watersheds, and quality of drinking water[13].

In addition to land use concerns, waste streams generated by natural gas extraction also need to be managed in an economically and environmentally sound manner. The largest by-product by volume accompanying energy extraction is wastewater[14]. The volume and quality of produced water is dependent upon the geographic location of the field, the geologic formation from where the water was produced, and the type of hydrocarbon being produced[14]. The significant increase in produced water accompanying new Marcellus wells between 2008 and 2009 led to a seven-fold increase in the total volume of produced waters sent to surface-discharging treatment plants[15]. Additionally, the produced water from the Marcellus can have salinity levels up to 360,000 mg/L[16]. The combination of volume and quality of produced water makes managing the byproduct especially challenging in the region.

Beyond potential negative environmental impacts of natural gas development, natural gas consumption can potentially have global greenhouse gas benefits when compared to substitutable fuels such as coal. Therefore, natural gas production in the U.S. could contribute to reduced global GHG emissions when exported to regions with high coal consumption. Prior to 2008, U.S. domestic natural gas production did not meet projected demand growth, and the national natural gas debate centered on LNG imports. As unconventional natural gas production (shale gas) became economically viable, U.S. technically recoverable natural gas reserves increased by 665 Tcf, which represents an increase in total U.S. natural gas resources by 38%[17]. As domestic natural gas production increased, supply flooded the market and wellhead prices in today's dollars in the U.S. dropped from close to $9/Mcf in 2008 to under $3/Mcf in 2012[18]. Because of higher natural gas prices in other regions, producers are hoping to sell incremental quantities of natural gas to economically attractive Asian and European markets[19]. This has prompted a debate in the United States on the policy of liquefied natural gas (LNG) exports regarding whether additional LNG exports to non-FTA countries should be approved, and if so to what capacity.

International trade poses a challenge for climate change mitigation. While traditional climate policies tend to focus on unilateral mitigation measures (actions taken by individual countries acting independently), international transport of goods and services is a global sector. As a result, the responsibility for reducing GHG emissions associated with shipping do not fall within the jurisdiction of any individual country[20]. Additionally, climate policies such as a global carbon cap, unilateral climate mitigation objectives, and carbon accounting practices can influence the total cost and life cycle greenhouse gas emissions of international trade. However, some climate policies may have unintended consequences for global trade, thereby potentially reducing the cost-effectiveness of greenhouse gas mitigation efforts.

1.2 Research Questions

This dissertation uses a variety of methods to address the issues discussed above. The research questions explored and answered in each of the following four chapters are as follows:

Chapter 2: Assessment of policies to reduce core forest fragmentation from Marcellus Shale development in Pennsylvania

- What is the historical impact of natural gas development on Pennsylvania's forest ecosystem based on the principals of landscape ecology?
- How is future development likely to impact PA's forest ecosystem?
- What development strategies could be used to mitigate the cumulative impact of natural gas development?

Chapter 3: A systems level perspective on Marcellus wastewater management in PA

- What volume of non-reusable wastewater will likely be generated throughout the lifetime of the Marcellus Shale play? Is Marcellus representative of other shale plays?
- What are the wastewater management options available to Marcellus producers and what are the economic tradeoffs between them? What is the least cost option for the average Marcellus well?
- What are other important water quality issues in Pennsylvania and how much would it cost to address these sources of pollution?

Chapter 4: Life Cycle Greenhouse Gas Emissions from U.S. Liquefied Natural Gas Exports: Implications for End Uses

- What are the greenhouse gas implications of the natural gas export landed life cycle, including upstream extraction and pipeline transmission, liquefaction, shipping, and regasification?

- What are the GHG cost or savings of displacing the traditional fuel sources of domestic coal (extracted in the importing country) or Russian natural gas transmitted via pipeline?
- What are the environmental implications of LNG exports from the US perspective?

Chapter 5: Assessment of GHG mitigation opportunities in the global crude trade

- What is the current global production, consumption, and trade of crude and what are the implications of different carbon accounting methods on characterizing country specific emissions burdens?
- What is the potential for cost and greenhouse gas savings in an optimized global crude system given current demand structure?
- What would be the impacts on global crude consumption given a severe carbon budget? How could carbon accounting strategies incentivize more effective mitigation behavior across the crude trade network?

1.3 Background on Oil and Gas

Fossil fuels have been the core of the global energy system and world economic activity since the 1700s, and consumption of energy resources has continued to increase annually. A recent study of onshore hydrocarbon wells estimates that over 4 million on shore wells have been developed worldwide[21]. Global oil and gas production totaled over 10,000 million tons oil equivalent (Mtoe) in 2013[22]. In particular, shale gas is becoming an increasingly important contributor to the global energy supply. Technological advances have recently made unconventional gas extraction economically viable in many regions of the world. A 2013 study by The U.S. Energy Information Administration estimated globally technically recoverable shale resources to be about 7,300 trillion cubic feet[23].

1.3.1 Natural Gas

1.3.1.1 The U.S. Natural Gas Landscape

Currently the United States is the largest producer of natural gas. The natural gas landscape in the United States has experienced a significant transition over the past decade. Technically recoverable reserves have increased by approximately 40%, and domestic production is projected to reach 36 trillion cubic feet (Tcf) by 2040 in the EIA reference case scenario, representing approximately a 50% increase over 2012 production[24]. Historically, consumption of natural gas outpaced domestic production and the US was poised to become the largest importer of natural gas. Currently, however, the U.S. is the largest producer of natural gas in the world and was a net liquefied natural gas exporter for the first time in 2015[24]. This trend has primarily been enabled by advances in hydraulic fracturing combined with horizontal directional drilling technologies and improved seismic monitoring techniques, which has made the extraction of unconventional (shale) resources economically viable.

There are several productive shale gas formations in the United States including the Barnett and Eagle Ford plays in Texas, the Haynesville play that spans part of Texas, Louisiana, and Arkansas, and the Bakken play the spans parts of North Dakota and Montana. The Marcellus Play, however, is the most expansive shale gas play in the United States[10], covering 14-25 million hectares (ha) (35-62 million acres) at a depth of 600-3,000 m[11]. The economically viable region of the Marcellus crosses southern New York, much of Pennsylvania, northern West Virginia, and eastern Ohio. While Pennsylvania (PA) has a rich history of oil and gas development, historically the industry targeted conventional gas, coal bed methane, and PA grade crude oil. Over 350,000 wells have been drilled in the Commonwealth since the first successful oil well in 1859[25]. Beginning in 2007, companies began to extract natural gas from the Marcellus Shale. Unconventional permits rose from 76 to 476 in 2008 alone, and between 2009 and 2010, permit applications again increased by 67%[26]. This general trend has continued, and through 2013 over 7,400 unconventional wells have been drilled in Pennsylvania[27].

A well is characterized by its estimated ultimate recovery (EUR) and its decline rate. The EUR specifies the total volume of gas that an operator can expect to extract from a well over its lifetime. The decline rate describes how the rate of gas recovery changes from its highest level of productivity immediately after hydraulic fracturing occurs, to when the well is no longer producing. Knowledge of EURs and decline rates are important for producers because they indicate how frequently the producer needs to develop new wells in order to keep their production constant. EURs can be in the range of billions of cubic feet, with 50-75% of the volume being recovered in the first year of production[28], but the average productivity of a well varies across plays. For example, Haynesville wells have a typical EUR of 6.5 Bcf and a decline rate of 85%, while Marcellus wells have an average EUR of 4.4 Bcf and a decline rate of 75% (Figure 1)[29].

The dramatic increase in unconventional natural gas development both in the United States and abroad is an important opportunity for access to additional energy resources. However, there is a broad set of issues associated with shale development including economics, security, and infrastructure development that need to be considered as the industry advances[30]. Furthermore, there are key public health and environmental concerns such as water use, water contamination, air quality, and land use that need to be addressed to ensure natural gas extraction is done safely and sustainably.

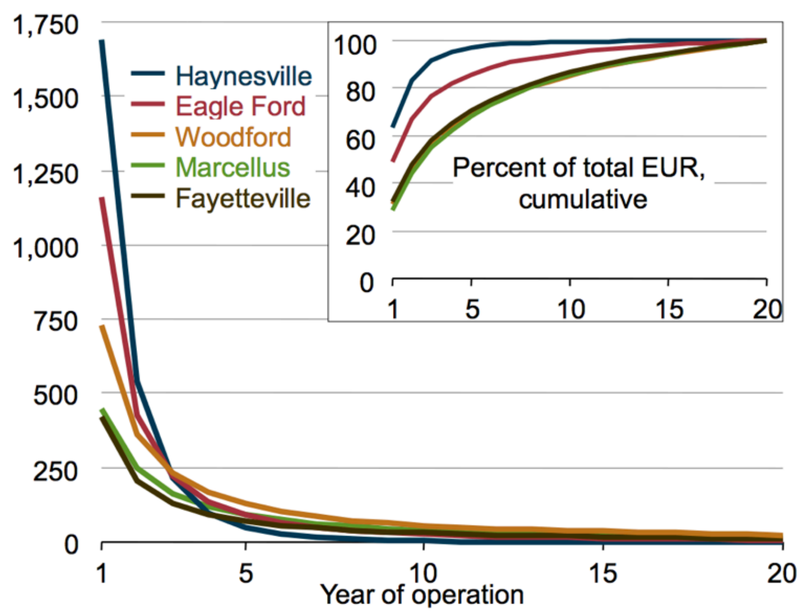


Figure 1: Average production profiles for shale gas wells in major U.S. shale plays (million cubic feet per year)[31]

1.3.1.2 Natural Gas Exports

As domestic natural gas production increased, supply flooded the market and wellhead prices dropped from close to $9/Mcf in 2008 to under $3/Mcf in 2012[18]. Because of higher natural gas prices in other regions of the world, producers are hoping to sell incremental quantities of natural gas to economically attractive Asian and European markets[19]. This has prompted a debate in the United States on the policy of liquefied natural gas (LNG) exports regarding whether additional LNG exports to non-FTA countries should be approved, and if so to what capacity.

While the majority of natural gas is transmitted via pipeline, over long distances it is economical to ship natural gas in the form of liquefied natural gas. Natural gas is liquefied by cooling it to -162°C at which point it condenses to a liquid with a volume of 1/600th of its volume in its gaseous state. LNG offers economic, flexibility, and security advantages over gas pipeline supply[32]. The processes of liquefying, shipping, and regasifying the LNG cost approximately 3.6 $/MMBtu[32] and result in about an 11% increase in emissions for electricity generation over non-LNG based electricity generation[33].

Globally, the LNG trade reached about 12 Tcf of natural gas in 2012[34], and based on the projects world-wide currently under construction or proposed, liquefaction capacity could increase by more than 100% within the next ten years[35]. The primary LNG importers are expected to be the rapidly developing countries in Asia and some European countries such as the UK, Spain, and France seeking to both supplement and diversify their natural gas supply[35].

The Natural Gas Act, as amended, requires the U.S. Department of Energy (DOE) to determine if LNG exports are in the "national interest" in order for approval. Additionally, the current regulatory framework requires exports to countries with which the U.S. has a free trade agreement (FTA) to be rapidly permitted, while applications to export to non-FTA countries undergo additional assessments[36]. The national interest determination involves consideration of economic, international, and environmental factors. To date, the DOE has approved thirty seven applications to FTA countries totaling 14 Tcf of natural gas exports annually, and has issued nine final and conditional approvals of applications to non-FTA countries, totaling almost 3.8 Tcf/year[36]. A recent regulatory change in 2014 streamlined the approval process of non-FTA

LNG export projects, and will likely lead to an increase in constructed export capacity in the near future[36]. The first non-FTA LNG export from the U.S. occurred in February 2016 from Cheniere Energy's Sabine Pass terminal in Louisiana.

1.3.1.2 Natural Gas as a Bridge Fuel

In addition to the economic motivation for natural gas development, interest in natural gas has in part been due to its potential to emit fewer emissions than coal on a life cycle basis[1,2]. In recognition of the role natural gas could play in transitioning to a low carbon energy future, it has often been called a 'bridge fuel'[37]. However, the viability of natural gas as a bridge fuel depends on assumptions about stabilization objectives, emissions, and technological change[3-6]. Some estimates suggest natural gas could reduce greenhouse gas (GHG) emissions over coal combustion[1, 33, 38], while other studies conclude the GHG footprint of natural gas could be as much as double that of coal[39] for electricity generation. The GHG benefits of natural gas in replacing coal for industrial heating and oil in the transportation sector are even less likely to result in GHG savings[33, 40]. These comparisons of emissions profiles between fuels are done using a method called life cycle assessment, which quantifies emissions at each stage in the supply chain from production to end use[41].

Understanding the life cycle GHG emissions of natural gas across its consumption pathways is important to long term decisions about power sector investments, industrial practices, regulatory policy, and sectoral applications[30]. GHG emissions are often quantified using a metric called global warming potential (GWP). GWP measures the radiative forcing of a GHG relative to CO_2. The GWP is reported by the IPCC for several different GHGs at two different time scales, the 100-year and 20-year time scale. The time scale is relevant for climate change due to the variation in half-life across gasses[42]. While the GWP is a useful metric because it can easily be applied across different GHGs to determine a CO_2 equivalent value, there are several shortcomings of the metric. One concern is that GWPs are highly uncertain, but the average value is frequently used deterministically. Another concern is that the GWP is a static value and does not account for dynamic changes in relative forcings over time as the concentration of gases in the atmosphere changes.

Beyond direct accounting of emissions, there are also second order impacts of natural gas adoption that could minimize the carbon benefits of natural gas. One factor is the energy-economy feedback effect. This effect describes the increase in global emissions that could arise due to a surge in economic activity resulting from low gas prices[43-45], because increased economic activity is associated with increase consumption of all forms of energy. A second factor is that natural gas could slow the development of even lower carbon renewable sources of energy[44]. Finally, besides reducing carbon emissions, if natural gas displaces coal electricity generation, SO_2 emissions will also decrease which will improve public health and reduce acid rain. However, SO_2 also plays a role in climate change effects of GHG emissions; it has a negative radiative forcing, which means it acts to help cool the atmosphere. With less SO_2 in the atmosphere, the net warming of CO_2 and other GHGs would increase[43].

1.3.1.2.1 Methane Leakage

Finally, in addition to the end use, the fuel it displaces, and potential consequential impacts of natural gas use, a key determinant in the climate change implications of natural gas consumption is the fugitive emissions rate. The fugitive emissions rate is the volume of natural gas that is leaked throughout the supply chain. After it has been processed, natural gas is composed of over 90% methane (CH_4)[46] which is a powerful greenhouse gas[42]. According to the IPCC fifth assessment report, the GWP of methane at a 100-year time scale has a mean of 36 with a standard deviation of 8.5, and the 20-year GWP has a mean of 87 with a standard deviation of 16[42].

The Natural gas system is expansive. Transmission, storage, and distribution of natural gas includes hundreds of thousands of miles of interstate and intrastate transmission pipeline, over 1400 compressor stations[47], and approximately 3.5 Tcf of underground storage throughout the U.S.[46]. Furthermore, there are thousands of abandoned wells and improperly completed exploration bore holes that are not currently active in the natural gas system but are important sources of methane emissions[21, 48]. Because of the scope of the system and the expense of monitoring equipment, the fugitive emissions rate is challenging to measure. The EPA estimates point source methane emissions in its GHG reporting inventory, which covers approximately 8,000 facilities and claims to capture 80-90% of total emissions. In 2014, the inventory reports methane emissions accounted for 10% of total U.S. GHG emissions; of this, the petroleum and natural gas sector accounted for a third[49]. A recent meta-analysis of twenty years of technical literature on methane leakage found the EPA estimate could be underestimating methane emissions by 50%[50]. The Environmental Defense Fund has been coordinating the largest ever compilation of sixteen complementary studies targeted at robustly estimating fugitive emissions from the natural gas sector[51-55]. They have found that the fugitive emissions from some portions of the supply chain are accurately captured by the EPA greenhouse gas-reporting inventory, while others underestimate it significantly. This is due to a combination of the GHG reporting rules, the GHG reporting threshold, and gaps (superemitters). Superemitters are a small number of highly emitting facilities that are responsible for the vast majority of fugitive emissions. A compiled estimate of fugitive emissions across the natural gas life cycle will be derived from these studies once they have been completed.

There are two general approaches to studying methane leakage. Bottom up studies, such as the EDF studies, estimate GHG emissions based on component-level emissions profiles and counts. In contrast, top down studies measure the methane in the atmosphere and then estimate how much of the measured methane can be attributed to the natural gas sector[30]. Oftentimes these two types of studies are compiled or compared using engineering-based modeling. Bottom up studies face many challenges; they are expensive and make conclusions about the natural gas system based on statistical sampling measures. Top down studies also are expensive and attribution of methane is difficult given the large number of potential methane sources, both anthropogenic and natural. Top down studies[56, 57, 58, 59] often estimate higher fugitive emissions rates than do bottom up studies and there is a large range of estimates in the literature, varying from 1% to 9%[60, 61]. The most common range tends to be between 2-4%[38, 50, 58, 62-64].

Although best management practices and effective leak detection could reduce methane emissions rates[65], until recently there was little regulation over the natural gas industry. The Climate Action Plan[66] is the first regulation that specifically identifies methane emissions from natural gas production and use as a priority and establishes an Interagency Methane Strategy to assess current emission data, address data gaps, identify technologies and best practices for reducing emissions, and identify existing authorities and incentive-based approaches to reduce methane emissions[30].

1.3.1.3 Social and Environmental Impact

Beyond the potential GHG benefits of natural gas consumption, there are several local environmental and social impacts associated with natural gas extraction that need to be addressed. Broadly speaking, environmental impacts include air quality, land use, soil contamination, water use, and species impacts. The specific effects of these potential impacts will depend on the hazard, exposure, and contamination pathway[67].

While the displacement of coal reduces SO_2, NO_x, and mercury near power plants[68, 69], local and regional air quality concerns include NO_x, particulate matter (PM), and volatile organic compounds (VOC) emitted from compressors, drill rigs, pumps, and other extraction infrastructure. Additionally, natural gas extraction results in dramatically increased regional trucking traffic, which also emits NO_x, PM, and VOCs[70]. VOCs react with nitrogen oxides in air and sunlight and form ground level ozone. Ozone is known to be a public health risk because it can exacerbate asthma and chronic obstructive pulmonary disease[71].

Soil contamination can occur due to the leakage of drilling waste, which increases the salinity and acidity of the soil[72]. Additionally, chemical pollutants in the soil can contaminate the surrounding environment if erosion or runoff occurs[73]. Animals can be impacted by pollutants as well. For example, a study by Latta et al. (2015) found metals associated with hydraulic fracturing activities in riparian songbirds[74].

1.3.1.3.1 Land Use and Fragmentation

The infrastructure required for drilling and hydraulically fracturing a well is extensive; primary infrastructure includes the well pad, access roads, and gathering line right of way clearings. In many regions, land needs to be screened for impacts on threatened, endangered, and special concern species and resources before hydraulic fracturing can be permitted to occur. However, this practice fails to consider the broad cumulative indirect impacts accrued from isolated land use changes within the core forest. The field of landscape ecology has identified that this cumulative impact, rather than the localized disturbance, is a source of risk to forest ecosystems.

The destruction of core forest habitat posses a threat to forest ecosystem health and biodiversity. While pure habitat loss alone can have an adverse effect on biodiversity, ecologists note that the pattern of habitat loss is often more important than the quantity of loss. In landscape ecology, a landscape is defined as a mosaic of habitat and non-habitat patches[75]. The spatial relationship between habitat patches within the mosaic influences the presence, movement, and persistence of species. Anthropogenic activities can compromise the structural and functional integrity of the landscape and impede ecological flows across the habitat[76,77]. When contiguous core habitats are fragmented into smaller patches, many sensitive species are unable or unwilling to cross non-habitat regions to reach these alternative habitat patches. While habitat loss can have an immediate impact on wildlife population, the ecological response to fragmentation is lagged, and affects different species at varying time scales[78].

In order to quantify changes induced by fragmentation, ecologists use landscape metrics. Generally, landscape metrics quantify specific spatial characteristics of patches, classes of patches, or entire landscape mosaics[77]. Structural metrics measure the physical composition or configuration of the patch mosaic without explicit reference to ecological processes[77]. Structural metrics are based on the theory of Island Biogeography, which interprets disjoint patches as analogous to oceanic islands embedded in an inhospitable or ecologically neutral background[79]. While this is an oversimplification of how species interact with the surrounding landscape[77], it provides a useful structure for tracking habitat pattern changes in a non-species specific manner and for interpreting general impacts of fragmentation. These metrics can then be interpreted by ecologists into a functional form regarding specific species needs and considerations.

1.3.1.3.2 Water Impacts

Natural gas development requires approximately 4-5 million gallons of water as input to the process[80]. While this water consumption is minimal across the natural gas life cycle, especially relative to other sources of electricity[30], the temporal aspect of the consumption is an important issue. When the water is rapidly withdrawn from small surface waters such as creeks or narrow rivers, it can cause localized environmental and social impacts. For example, low levels of water resulting from water withdrawals can potentially impact municipal water supplies, industrial water needs, recreational activities, and the health of aquatic life[81].

The process of hydraulic fracturing involves first mixing this freshwater with a fracturing fluid at the drilling site. The final mixture is composed of 84% water, 15% proppant, and 1% chemicals (by mass)[82]. Proppants are spherical white sand, ceramic, or man-made particles that keep fractures in the shale from collapsing. The chemicals serve many purposes such as preventing bacteria growth, keeping minerals from building up on the well walls, and ensuring the proppant is well mixed in the fluid[83]. More than 1,000 different chemicals have been used for fracking[71]. These chemicals range from benign to toxic. The precise mixture of chemicals required depends on the specific geology of the underlying shale. While the percentage of chemicals in the mixture is small, a single well requires a total of about 40,000 gallons of chemicals.

The most significant risk related to fracking fluid is the potential for the concentrated chemicals to be spilled during transportation or while being stored on site prior to being mixed with the water. An experimental release of hydraulic fracturing fluids in a forest in West Virginia to simulate such a spill demonstrated the environmental impact of the chemicals. In just 10 days, the exposure resulted in significant damage to ground vegetation. After two years, about half of the trees in the exposed region were dead and the sodium and chloride concentrations in the soil were fifty times higher than they had been prior to the release[84]. Another study seeking to identify the public health threat from fracking fluid identified 353 chemicals and found that 75% of them can affect the skin, respiratory system, and/or the gastrointestinal system. Furthermore, about half of these chemicals are known to have effects on the nervous system, immune system, and/or cardiovascular system[85]. Finally, another study of an accidental spill of fracking fluids into a creek in Kentucky found that the degradation of the water quality resulted in fish dying from gill lesions, and liver and spleen damage[86].

After hydraulic fracturing, wastewater flows to the surface of the well. Wastewater is the largest by-product by volume accompanying natural gas extraction[14]. Details about the volumes and management strategies of this wastewater, called produced water, are not well documented on a national scale across the thirty-one oil and gas producing states. A comprehensive study in 2015 estimated that the one million actively producing oil and gas wells in the United States generated over 21 billion barrels of produced water in 2012[87]. This volume is projected to increase to over 34 billion barrels by 2025[88].

The volume and quality of produced water is dependent upon the geographic location of the field, the geologic formation from where the water was produced, and the type of hydrocarbon being produced[14]. In the Marcellus region, each horizontal well will tap gas from approximately 83 acres, and less than half of the water will return to the surface[89]. Initially, the produced water has a chemical footprint similar to the injected water. However, after the first few weeks, the composition of the water changes. Most noticeable, its level of total dissolved solids (TDS) increases dramatically until it approaches steady state at about 10% NaCl equivalent. Most likely, this is a result of the injected water interacting with high salinity brine within the Marcellus formation[89, 90,91, 92]. Additionally, the depth of the hydrocarbons influences the salt and mineral levels present in the produced water; wastewater from shale extraction at 5,000-8,000 ft typically has twenty times the salt/mineral content of produced water from coal bed methane produced water[93]. This is important because while there are many contaminants in the produced water that need to be considered, the salinity, referred to as the total dissolved solids (TDS), is generally the defining characteristic that determines the economic and technical feasibility of various wastewater management options. Therefore, the way in which produced water is handled varies across the country and what is successful in one location may prove impractical in another region. In the Marcellus Shale region, wastewater management strategies include reuse, deep well injection, treatment at permitted centralized waste treatment facilities, or on-site treatment.

The two main risks of freshwater contamination from Marcellus wastewater are through the storage and disposal of produced water. Unfortunately there is little reliable information on the number and size of spills that occur during storage and disposal. The lack of comprehensive data demonstrates the absence of regulatory enforcement and is an impediment to characterizing the true risk from Marcellus development[94]. Wastewater storage poses a threat to the environment because containment ponds used by well developers for temporary storage can overflow during heavy rain, can leak as liners degrade, are accessible to wildlife, and are potential sources of air pollution as chemicals evaporate[95]. Pennsylvania is one of few states that maintain detailed records of spills and surface water releases. In 2013, the Pennsylvania Department of Environmental Protection (PADEP) detected a total of 523 violations during standard state inspections. The three most common violations were 1) failure to properly store, transport, process, or dispose of residual waste, 2) failure to adopt required or prescribed pollution prevention measures, and 3) failure to properly close a well upon abandonment. Spills were detected at 194 of the wells found in violation. One fifth of these spills contaminated land or surface water. Nine of the spills were large (over 3,500 liters). The time between the spill and reporting ranged from 1 hour to 6 weeks, and only 114 of the documented spills were reported by the liable drilling company itself. Failure to self-report a spill is in direct violation of Pennsylvania's reporting requirement and typically results in a fine.

The disposal of wastewater is the second key stage at which there is a risk for surface water contamination. Disposal is a complex issue for well developers. They must consider several factors, such as the volume of the produced water, the quality of the water, state and federal regulations, available disposal infrastructure, and characteristics of the specific shale play. Each disposal method results in different potential concentrations of contaminants being released through a specific pathway. This makes characterizing the risk of wastewater disposal challenging because the ultimate toxicity of surface water contamination is a function of concentration, pathway, and duration of exposure[95].

While there are few specific studies on the toxicology of produced water, there is anecdotal evidence of the public health risk posed by wastewater exposure. In particular, some farmers believe their cattle are at risk of exposure to fracking wastewater from leaks from storage pits or through unauthorized dumping of wastewater into creeks. One such example was when a wastewater storage container failed, resulting in the flow of produced water into a pasture and a pond used as a source of fresh water for the cattle. In total, 140 cows on the ranch were exposed to this wastewater while the remaining 60 cows were in a different pasture and did not drink from the pond. Of the exposed cattle, half died. In comparison, there were no reported health problems in the unexposed portion of the herd[96].

The composition of produced water varies depending on the specific geology of the formation. While concentration of specific components can vary by an order of magnitude from well to well, the components of all produced water are qualitatively similar[97]. The major categories of constituents are 1) salts (measured as salinity, conductivity, or total dissolved solids) including metal ions, 2) organic hydrocarbons (sometimes listed as oil or grease), 3) inorganic and organic compounds, and 4) naturally occurring radioactive material (NORM). In addition, produced water may contain diluted quantities of the chemicals used for fracturing. This review will focus on four of these major contaminants in produced water: total dissolved solids (TDS), organic compounds, bromides, and NORM. Each contaminant poses a unique environmental risk, public health threat, and challenge for wastewater treatment.

The first concerning contaminant in produced water is total dissolved solids (TDS). TDS is a measure of dissolved matter such as salts, minerals, and inorganic compounds. Examples include sodium, calcium, and chloride. The U.S. Environmental Protection Agency (EPA) set the recommended maximum level of TDS in drinking water at 500 mg/L. This limit was primarily established to avoid poor taste and to reduce the corrosive effects of the salt on plumbing fixtures rather than to address a direct public health concern[98]. However, the average TDS concentrations in shale gas wastewater ranges from 1.6 to 375 times the EPA recommended level[99]. Aquatic ecosystems are especially sensitive to changes in salinity. Even a release of high salinity water as small as 1 g/L can have an adverse effect on biodiversity. For example, in 2009 brackish (high salinity) wastewater from a coal mine spilled into Dunkard Creek, a tributary to the Monongahela River. The high TDS levels created conditions favorable to the creation of a bloom of toxic golden algae that devastated the aquatic ecosystem. The Pennsylvania Fish and Boat Commission (PFBC) estimated that over 42,000 fish, over 15,000 freshwater mussels and over 6,000 mudpuppies were killed[100]. As this example suggests, TDS is one of the primary concerns about accidental or unauthorized disposal of untreated wastewater into surface waters. One of the challenges of high TDS wastewater is that publicly owned treatment works (POTWs) and many waste treatment facilities do not have desalinization capabilities. In the early stages of Marcellus development, well developers shipped wastewater to these facilities. Because the facilities could not remove the salt, they discharged large volumes of high salinity water into surface waters. This led to an increase in the levels of chloride found in the water. As a result, the PADEP introduced discharge limits in 2010 to eliminate disposal of Marcellus wastewater to public wastewater treatment facilities[101].

Bromide poses another public health concerning related to high TDS levels in Marcellus produced water. On its own, bromide is not toxic. However, it is problematic during the drinking water treatment process. The bromide reacts with chloride disinfectants to form a disinfection byproduct (DBP) called brominated trihalomethanes (THMs). These volatile organic liquid compounds are carcinogenic and have been linked to birth defects. In 2010, inadequately treated produced water was discharged into Blacklick Creek in Indiana County, PA, with a measured mean bromide concentration of 1,070 ppm, which is 10,700 times the generally accepted bromide limit of 0.1 ppb[102]. When Marcellus drilling first began, produced water was sent to public treatment works for disposal. As a result, bromide levels in the rivers in Pennsylvania spiked and drinking water treatment facilities had trouble meeting the federal THM standard. Drinking water utilities in southwestern PA reported that 85-94% of the formed DBPs were brominated[103]. Once the PADEP banned disposal at public wastewater treatment facilities in 2010, THM levels generally returned to historical levels. In addition to Marcellus wastewater, there are other potential sources of bromide. These include new smokestack scrubbers at coal-fired power plants, where bromide is used to help keep mercury levels low. A third source of bromide is from salt used to de-ice roads in the winter. Some natural gas producers claimed that this road salt was the reason for the high observed bromide levels, but the spike first occurred in summer months, not during the winter road salting season. Additionally, there was no spike in chloride to accompany the spike in bromide, so road salt was likely not the culprit. To accommodate bromide in the water, water treatment facilities can change their processes to avoid the formation of THM, but it can be costly. For example, one facility changed its treatment method from chlorine to chloramines for $15,000[103]. However, the risk of switching disinfectant is unknown. Using chloramines instead of chlorine may increase the formation of other unregulated or undetected disinfection byproducts and can increase lead exposure[104].

A second category of concerning contaminants in Marcellus wastewater is called organics. One type of organic contaminant in particular, called volatile organic compounds (VOCs), may include compounds such as benzene, toluene, ethyl-benzene, and xylene. This grouping of compounds is commonly referred to as BTEX. BTEX has inherent toxicological risk; benzene in particular poses a threat to public health. The health effects most often associated with benzene include acute and chronic cancers, anemia and other blood disorders, and immunological effects[67]. One study reported that the quantity of benzene in fracking fluid is up to 13,000 times the legal limit for surface water discharge[105,106]. Another study found that the mean benzene concentration in produced water is over two times the drinking water standard, over six times the EPA consumption criteria, and 1.5 times the drinking water minimum risk level for chronic exposure for children[102]. In addition to posing a drinking water risk, VOCs can evaporate from wastewater stored in open pits. When evaporated, VOCs react with nitrogen oxides in air and sunlight and form ground level ozone. Ozone is known to be a public health risk because it can exacerbate asthma and chronic obstructive pulmonary disease[71]. Evaporated VOCs from water storage pits is a public health threat to workers as well as residents living close to the well site[107]. Finally, organic material can have a large environmental impact because some organics consume oxygen. Therefore, if released into a body of water, the organic material would keep aquatic species from obtaining the necessary oxygen they need to live.

A third contaminant category of concern is the presence of naturally occurring radioactive material (NORM). The NORM found in Marcellus produced water is generally radium derived from the radioactive decay of uranium and thorium present within the shale formation[103]. Disposal of wastewater containing NORM into freshwater streams or ponds over time leads to an accumulation of radium in the stream sediments. The specific ratios of radium isotopes can specifically identify the accumulated radium as having originated from a Marcellus well[108]. At one discharge site, the level of radioactivity was found to exceed the legal limit for a licensed radioactive waste disposal facility[109]. However, even if each discharge is within the legal limits, the radium can build up in the soil over time. As a result, toxic chemicals and radiation would slowly be released into the impacted regions, thereby posing a long term environmental and public health risk[108]. NORM levels have been found to have a median concentration of up to 90 times the legal limit for industrial discharge[110]. Although these levels may appear high, they are actually low relative to the quantity of radioactive material generated by other sources of energy. For example, electricity generation from coal can produce up to 50 times the amount of radioactive material as electricity generation from natural gas[111]. Finally, while bromide cannot be removed once it is in water, NORM can be removed from both soil[111] and water[112].

1.3.1.3.3 Social Impacts

Public opinion of unconventional natural gas development has been mixed. Recent surveys have found that the population in areas of high natural gas development such as Pennsylvania, Michigan, and Texas recognize and appreciate the positive economic influence of the industry, but are concerned over the environmental impacts[113, 114]. Estimates suggest in unconventional oil and natural gas development employed up to 1.7 million people in the U.S. in 2012. This is projected to increase to nearly 3 million jobs by 2020[115]. While there is likely to be significant local and regional economic value to natural gas development, some studies have been found to overstate such benefits[116]. Further economic value is generated to the public due to lower prices of natural gas for heating and electricity.

Beyond the economic benefits and potential public health impacts due to air pollution and water contamination associated with increased risk of morbidity and mortality[117-119], there are several non-chemical stressors that could negatively impact social systems, such as increased noise and higher accident rates due to increased trucking[119]. In particular, the exposures to these stressors are increasing because unconventional natural gas development occurs near population centers. These include stressors typical to boom industries, such as coal mining towns experienced in the 1900s. For example, the influx of workers can lead to increases in rental prices, crime rates, and the prevalence of substance abuse. These negative social externalities most frequently impact disadvantaged populations. The overall effect of these stressors on population health depends on several different factors and may vary from region to region[67].

1.3.2 Global Crude System

In 2014, global oil consumption reached over 33 billion bbls of crude[34], with approximately 49% of consumption by OECD countries and 51% by non-OECD countries. Crude consumption is disproportional to population, with the largest per capita consumption in North America and Europe. Between 2002 and 2013, Saudi Arabia and Russia were consistently the largest oil producers. United States oil production had been steadily declining from 1985 until about 2009 when production rapidly began to increase. Between 2010 and 2014, US production increased by 14%, and the United States currently accounts for approximately 13% of the world's crude supply, making it the largest global producer of oil[34]. This estimate includes conventional crude oil, tight oil, oil sands, and natural gas liquids. While historically oil production has been from conventional sources, it is projected that the percentage of crude supply from unconventional sources may increase substantially in the future depending on resource constraints, fuel prices, elasticity of demand, and potential climate policies[120].

1.3.2.1 International Trade

Of the over 32 billion bbls of oil produced each year, approximately 64% is exported from one country to another (21 billion bbls)[34], primarily by vessel movements (17 billion bbls[121], or 83% of exports). International trade poses a challenge for climate change mitigation. While traditional climate policies tend to focus on unilateral mitigation measures (actions taken by individual countries acting independently), international transport of goods and services is a global sector. As a result, the responsibility for reducing GHG emissions associated with shipping do not fall within the jurisdiction of any individual country[20].

International shipping can be monitored in a variety of ways. One widely used method is through the Automatic Identification System (AIS), which is a system through which vessels report their location in regular intervals throughout their voyages. Historically, only shore-based AIS receivers were available with a limited range of 50 miles. This low coverage led to extensive uncertainty about the shipping sector. However, since 2009, satellites have been used to collect AIS data, thereby greatly improving the quality of fleet operational data available[122].

Several studies have been done to quantify emissions from the shipping sector[122-125, 126]. According to a report by the International Maritime Organization (IMO), international shipping resulted in over 800 million tons CO_2-eq in 2012. Oil tankers had the third largest fuel consumption (almost 40 million tons fuel), surpassed only by container ships and bulk carriers. The Oil tanker fleet is comprised of over 7,000 vessels ranging in size from a few thousand dead weight tons (dwt) to over 200,000 dwt. Emissions from oil tankers alone were responsible for over 120 million metric tons of CO_2 emissions in 2012[122]. The IMO study was a bottom up assessment, in which vessels were categorized by fuel type, shipping category, and dwt to develop an emissions estimate. Shipping emissions can also be assessed using a top down approach whereby emissions are estimated based on total fuel consumed by the maritime industry. However, fuel consumption data often incomplete or unavailable[125].

1.3.2.2 Energy System Modeling

Because the global crude system is so crucial to social and economic stability, there is a long history of energy system modeling. Interest in studying the impact of energy policies on the economy began after the first oil shock in 1973[127]. Economic models are the most commonly used method of assessing energy systems[128]. Energy-economic modeling can be traced back to the first computable generalized equilibrium (CGE) model developed by Hudson and Jorgenson in 1974[129]. This subset of energy models typically requires many assumptions about international trade flow data, elasticity of substitution and transformation, and perfect substitution[130]. While the purpose of CGEs is to adequately represent global systems, the vast number of simplifying assumptions introduces substantial uncertainty. While other methods have been used to study energy trade patterns and emissions, such as input-output assessments[131] and network analysis[132, 133-136], CGEs[127, 137,138] remain the dominant energy modeling technique.

1.3.2.2.1 Application of CGE to Environmental Issues

In the early 1980s, CGEs shifted from models focusing specifically on energy and economics only to E3 models, or energy-economy-environment models[127]. Several studies have been done with CGE/climate model combinations such as TIAM-UCL model[139] and ROMEO model[120] among others[71, 140] to understand the implications of various policies on the global climate system. In particular, one commonly applied CGE used to study trade flows is called the Global Change Assessment Model (GCAM). GCAM is an integrated assessment model that includes representations of the economy, energy sector, land use and water[141]. Another widely used international trade CGE model is the Global Trade Analysis Project (GTAP). GTAP is a multi-regional, multi-sector CGE model based on input-output models describing bilateral trade patterns, production, consumption, and intermediate use of commodities and services.

1.3.2.2.2 Climate Policies

A particular interest of E3 CGE models is the market impact of climate policies. Through international climate discussions, such as at the Paris Climate Change Conference in 2015, policy makers have come to the agreement that the average global temperature rise caused by greenhouse gas emissions should not exceed 2 °C above the preindustrial average global temperature[142]. In order to limit the average global temperature rise, climate models have demonstrated a limit to the cumulative emissions that can be sustained by the climate system. This limit is called the global carbon budget. The Intergovernmental Panel on Climate Change (IPCC) reported that this budget is in the range of 870-1,240 Gt CO_2-eq between 2011-2050 in order to have a 56% chance of not exceeding this 2°C temperature increase[143]. A recent study by McGlade and Ekins (2015) used an integrated assessment model to explore the implications of this emissions limit for fossil fuel production[139]. In their scenario to keep average global surface temperature rise below 2°C for all years to 2200, they found 2050 GHG emissions must be constrained to 21 Gt CO_2-eq. This is compared to the 48 Gt CO_2-eq emitted in 2010, and a baseline 2050 projection of 71 Gt CO_2-eq given no emissions mitigation. This latter projection would result in less than 5°C global average temperature rise[139].

In addition to strict global caps on emissions, one specific impact of climate policies energy modelers have explored is carbon leakage[144]. Carbon leakage occurs when action taken by countries to reduce emissions is partially or wholly offset by increased emissions elsewhere in the world. Carbon leakage results due to asymmetries of unilateral climate mitigation strategies[138,145, 146, 147]. A second area of climate policy modeling is emissions trading schemes[148]. Emissions trading schemes are market mechanisms for transferring the right to emit from a country with a low cost of emissions abatement to a country with a high cost of emissions abatement. These trading schemes could be implemented at various geographic scales and are meant to provide a cost-effective, efficient pathway towards emissions reductions. Finally, a third important of climate policy modeling is carbon accounting schemes. Carbon accounting refers to the method by which emissions are attributed to countries. Some commonly discussed carbon accounting schemes include location-based, where countries are responsible for what is emitted within their borders, production-based, where the producer would be responsible for the full life cycle emissions of what they produce, and consumption-based, where countries are responsible for all life cycle emissions of what they consume[149]. Carbon accounting strategies could be used to assess a country's compliance with a given target, or they could be used to develop border cost adjustments (BCAs). BCAs are mechanisms to account for emission associated with traded goods, such as an import tax based on carbon content[138]. Such measures could be implemented to minimize carbon leakage. However, there is concern that such measures would cause the cost of emissions reduction to shift from developed to developing countries[146].

Modeling the impact of various climate policies on the energy system requires a comprehensive understanding of where emissions are produced in the system. This is important because internationally traded commodities are a large portion of global consumption. For example, Davis et al. (2011) found that 37% of global emissions are from fossil fuels traded internationally[150]. However, relatively little attention has been paid to the quantity of emissions associated with the consumption of goods and services in countries[131]. These emissions are called embodied carbon emissions. Sharing the responsibility for embodied emissions through BCAs or other measures could help facilitate international agreement on global climate policy[131].

1.3.3 Thesis Outline

The second and third chapters of this dissertation explore two important environmental concerns of natural gas development: ecological impact, addressed using principals from the field of landscape ecology, and management of wastewater byproducts. For each of these environmental concerns, this dissertation characterizes the ability for US to support energy consumption while simultaneously mitigating such environmental impacts. More specifically, the second chapter assesses the impact of infrastructure required for natural gas extraction on forest fragmentation. It then outlines strategies to effectively manage future infrastructure development to mitigate incremental fragmentation.

The third chapter quantifies the potential for surface water impacts arising both from the consumption of freshwater resources and through the generation of contaminated wastewater as a byproduct of natural gas extraction. This chapter then assesses the cost and greenhouse gas emissions of wastewater management strategies. It also compares the cost of the strategies to the cost of treating other significant water quality issues impacting the state of Pennsylvania including coal mine drainage and agricultural runoff. This comparison informs cost-effective policy decision-making regarding investments in water quality improvements.

The fourth chapter of this dissertation further explores the potential for natural gas to act as a bridge fuel by quantifying the impact of United States natural gas exports on global greenhouse gas emissions. The emissions are estimated under different end uses, global warming time scales, methane leakage rates, and fuel offset scenarios. In this chapter, the emissions are monetized using the social cost of carbon to identify the tradeoff between US social cost of extraction and global social savings resulting from displacing traditional fuels with US natural gas exports. Because of this tradeoff, location based carbon accounting strategies may unintentionally serve to de-incentivize global GHG mitigation efforts.

Finally, the fifth chapter of this dissertation continues to explore how different carbon accounting strategies could influence international trade. This is especially relevant if climate change policies continue to be unilateral rather than cooperative. In particular, asymmetric country-to-country mitigation measures present the potential for carbon leakage, where one country's emissions reductions are wholly or partially offset by emissions increases in another less carbon-constrained country. This chapter explores these concepts using the current global crude trade as a case study. It begins by characterizing the 2014 global crude system and then optimizes the system under various supply limits, crude quality (API gravity) targets, and climate policies such as a global carbon budget and various carbon accounting strategies.

Chapter 2. Assessment of Policies to Reduce Core Forest Fragmentation from Marcellus Shale Development in Pennsylvania[1]

2.1 Abstract

Marcellus Shale development is occurring rapidly and relatively unconstrained across Pennsylvania (PA). Through 2013, over 7,400 unconventional wells had been drilled in the Commonwealth. Well pads, access roads, and gathering lines fragment forestland, resulting in irreversible alterations to the forest ecosystem. Changes in forest quantity, composition, and structural pattern can result in increased predation, brood parasitism, altered light, wind, and noise intensity, and spread of invasive species. These fragmentation effects pose a risk to PA's rich biodiversity. This study projects the structure of future alternative pathways for Marcellus shale development and quantifies the potential ecological impact of future drilling using a core forest region of Bradford County, PA. Modeling presented here suggests that future development could cause the level of fragmentation to more than double throughout the lifetime of gas development. Specifically, gathering lines are responsible for approximately 94% of the incremental fragmentation in the core forest study region. However, by requiring gathering lines to follow pre-existing road routes in forested regions, shale resources can be exploited to their full potential, while essentially preventing any further fragmentation from occurring across the core forested landscape of Bradford County. In the study region, assuming an estimated ultimate recovery (EUR) of 1 to 3 billion cubic feet (Bcf) per well, this policy could be implemented for a minimal incremental economic investment of approximately \$0.005 to \$0.02 per Mcf of natural gas produced over the modeled traditional gathering line development.

2.2 Introduction

Pennsylvania (PA) has a rich history of oil and gas development. Over 350,000 wells have been drilled in the Commonwealth since the first successful oil well in 1859[25]. Prior to 2008, wells targeted conventional gas, coal bed methane, and PA grade crude oil. In December 2007, the oil

---

[1] The final version of this chapter is available as Abrahams, Leslie S., W. Michael Griffin, and H. Scott Matthews. "Assessment of policies to reduce core forest fragmentation from Marcellus shale development in Pennsylvania." *Ecological Indicators* 52 (2015): 153-160.

and gas production company Range Resources announced the use of horizontal drilling combined with hydraulic fracturing to successfully complete five unconventional wells[26]. This demonstrated the economic viability of using this combination of technologies to extract Marcellus Shale gas. In 2008, Marcellus Shale permits rose from 76 to 476, and between 2009 and 2010, permit applications again increased by 67%[26]. This general trend has continued, and through 2013 over 7,400 unconventional wells have been drilled in Pennsylvania[27].

The Marcellus is the most expansive shale gas play in the United States[10], covering 14-25 million hectares (ha) (35-62 million acres) at a depth of 600-3,000 m[11]. The economically viable region of the Marcellus crosses southern New York, much of Pennsylvania, northern West Virginia, and eastern Ohio. Because of the continuous nature and vast expanse of the shale deposit, developers have some flexibility in locating wells. Currently in PA the vast majority of well pads are drilled with little consideration for land use preservation. In 2008 alone, half of the wells were developed in PA forests[12]. This destruction of core forest habitat poses a significant risk to forest ecosystem health and biodiversity. Additionally, deforestation results in an increase in impervious surface, which is a threat to stream health, the integrity of headwater watersheds, and quality of drinking water[13].

Marcellus development has the potential to rapidly alter landscapes. The infrastructure required to drill and hydraulically fracture a well is extensive. Well pad construction involves removing topsoil, leveling and lining the area with geotextile fabric, and covering the pad with compact stone[152]. This development can span 1.2-2.8 ha (3-7 acres)[153]. Additional infrastructure such as access roads, water/wastewater storage, compressor stations, and gathering pipelines, is required to successfully develop the well and fully exploit the resource (Figure 2). These secondary components of the natural gas infrastructure on average require an additional 10 ha (25 acres) of land per well pad[153, 154]. The combined footprint of unconventional natural gas development can total more than 12 ha (30 acres) for a single well pad.

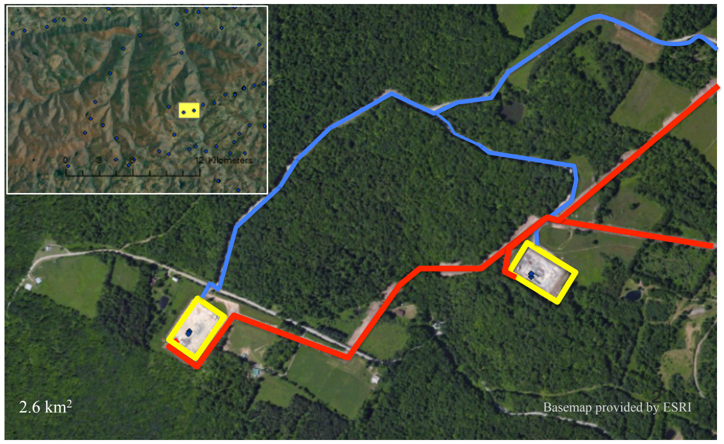


Figure 2: The three key components of Marcellus infrastructure include well pads (yellow), access roads (blue), and gathering lines (red)

When constructed in forests, infrastructure development significantly impacts the landscape. Current regulation requires Marcellus shale projects to be screened for impacts on threatened, endangered, and special concern species and resources using the Pennsylvania Natural Heritage Program's Pennsylvania Natural Diversity Inventory Environmental Review Tool[155]. While this protects the immediate habitat of specialized endangered species, it fails to consider the broad cumulative indirect impacts accrued from isolated land use changes within the core forest. This cumulative impact, rather than the localized disturbance, is a source of risk to PA's forest ecosystems.

Forest covers approximately 6.5 million ha (65%) of PA and is concentrated in the central and north central parts of the state[11]. Large contiguous patches of core forest in these regions maintain the majority of flora and fauna species richness and diversity in the state and are critical components of the global ecosystem (Figure 3). More than 71 species of birds, 43 species of mammals, and 48 species of reptiles and amphibians rely on PA forests for essential habitat[156]. In particular, PA core forests have rare bird populations that include the bald eagle, peregrine falcon, and interior-nesting warblers[153]. For some species PA is fundamental to their global survival; more than 19% of the global population of scarlet tanagers and 9% of the global population of wood thrush breed within PA forests[157].

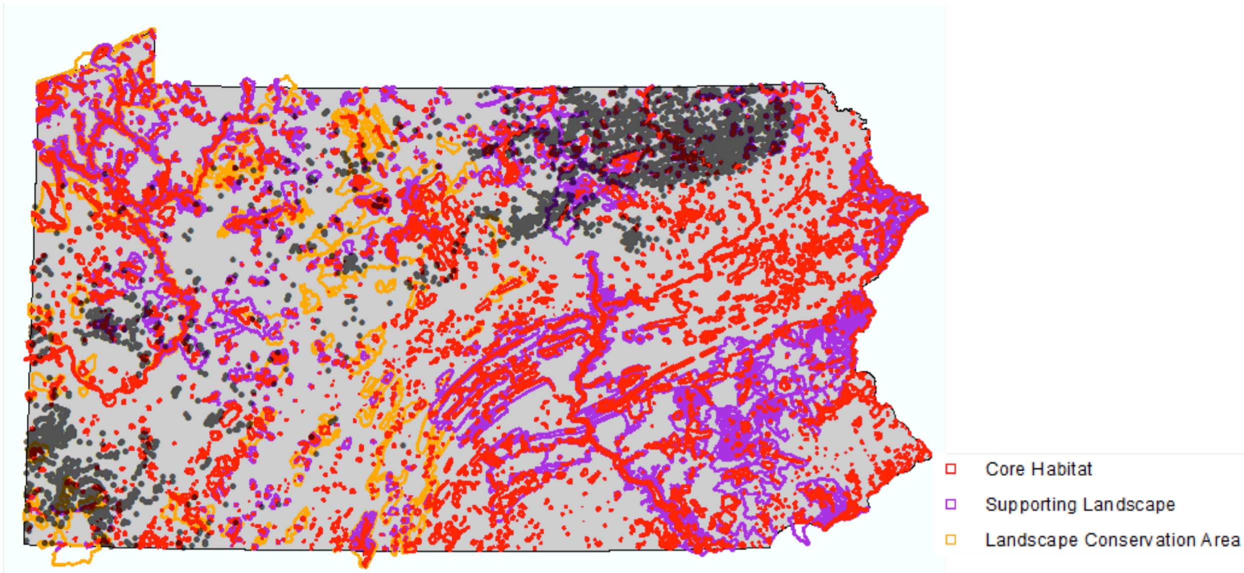


Figure 3: Important ecological regions in PA

While pure habitat loss alone can have an adverse effect on biodiversity, ecologists note that the pattern of habitat loss is often more important than the quantity of loss. In landscape ecology, a landscape is defined as a mosaic of habitat and non-habitat patches[75]. The spatial relationship between habitat patches within the mosaic influences the presence, movement, and persistence of species. Anthropogenic activities can compromise the structural and functional integrity of the landscape and impede ecological flows across the habitat[76,77]. When contiguous core habitats are fragmented into smaller patches, many sensitive species are unable or unwilling to cross non-habitat regions to reach these alternative habitat patches. While habitat loss can have an immediate impact on wildlife population, the ecological response to fragmentation is lagged, and affects different species at varying time scales[78].

A secondary impact of fragmentation is the creation of edges. Edges are generally defined as the 100m between core forest and non-forest habitat[3,4,153]. New edges affect the physical or biological conditions at the ecosystem boundary and within adjacent ecosystems[158]. Edge effects are believed to be detrimental by increasing predation, changing lighting and humidity, and increasing the presence of invasive species[153]. In particular, songbirds nesting near edges and openings are less likely to successfully raise young than individuals nesting in interior forest[157]. While 100m is commonly cited as the estimated depth of edge effect penetration[12, 153], the impact on specific species can be seen at much greater distances. In a meta-analysis of the effects of roads, power lines, and wind turbines on birds and mammals, bird populations were reduced as far away as 1 km and mammal populations at 5 km[159].

In order to quantify changes induced by fragmentation, ecologists use landscape metrics. Generally, landscape metrics quantify specific spatial characteristics of patches, classes of patches, or entire landscape mosaics[77]. Structural metrics measure the physical composition or configuration of the patch mosaic without explicit reference to ecological processes[77]. Structural metrics are based on the theory of Island Biogeography, which interprets disjoint patches as analogous to oceanic islands embedded in an inhospitable or ecologically neutral background[79]. While this is an oversimplification of how species interact with the surrounding landscape[77], it provides a useful structure for tracking habitat pattern changes in a non-species specific manner and for interpreting general impacts of fragmentation. These metrics can then be interpreted by ecologists into a functional form regarding specific species needs and considerations.

This study focuses on the following landscape metrics; the number of core patches (NCP), the largest patch index (LPI), and the percent habitat available in the landscape (PLAND) (Figure 6). NCP is a count of the number of contiguous forest regions larger than 4 ha that are greater than 100m from a non-forest opening. Core patch metrics are important measures of fragmentation because they integrate patch size, shape, and edge effect into a single measure[77]. The LPI is the percent of the total habitat that is made up by the single largest patch and is an indicator of a species' ease of movement around the landscape matrix (connectivity). PLAND is representative of the pure habitat loss associated with land use change. These metrics were chosen as proxy variables to fully describe the composition and the configuration of the landscape. The purpose of this study is to quantify changes in these metrics as a result of various future development pathways. In this paper, the metrics are used as proxies for disturbance and are considered a reflection of the ecosystem's overall health and stability. Impact on specific species or ecosystems will require more detailed studies.

The Nature Conservancy conducted an Energy Impact Assessment to quantify the potential impacts of natural gas development on habitats[153]. They first used aerial imagery to determine the spatial footprint of well pads, roads, and water storage facilities associated with 240 well pad development sites in north central and southwestern PA. They then used a machine learning tool to develop a 30m x 30m resolution probability map describing the likelihood of Marcellus development across PA (Figure 4). This modeling approach, called maximum entropy, was used to find relationships between 1,461 existing and permitted well pad locations and landscape variables such as Marcellus Shale depth, thickness and thermal maturity, percent slope, distance to pipelines, and distance to roads[153]. This was then re-sampled to different resolutions to reflect the separation distance (based on implied lateral length) of future alternative development scenarios. Well pads were placed in order of most probable pixels to least probable pixels until the appropriate number of well pads were located[153]. The number of well pads for each scenario represents an additional 60,000 wells across the state with varying number of wells per pad. While TNC's study presents a spatially explicit model of where well pads might be developed in the future and quantifies an overall estimate of pipeline and road development in core forest, it does not project the location of future secondary infrastructure. The primary infrastructure of natural gas development (the well pads) has an estimated average footprint of 3-7 acres[153]. The secondary infrastructure, including access roads, compressor stations, water storage, and gathering lines has been estimated to have an average footprint of approximately 25 acres per well pad[153, 154]. Therefore, the secondary infrastructure can have an impact of over 8 times that of the well pad itself (25 acres versus 3 acres). Because the secondary infrastructure has such a large footprint, the spatial distribution of this secondary infrastructure across the landscape is a key component of understanding the impact of development on ecosystem services.

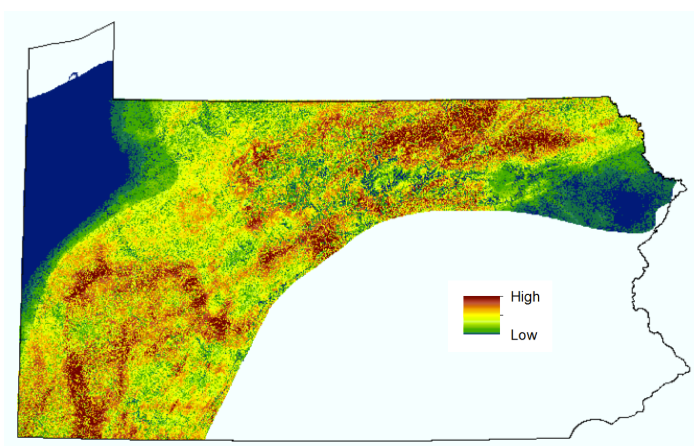


Figure 4: The probability surface raster depicting the likelihood of future Marcellus development throughout Pennsylvania[153]

This study expands on TNC projections for potential future development by including spatially explicit projections of secondary infrastructure in addition to the well pads. It also further explores the impact of decreasing well pad density on fragmentation by using TNC methods to project three additional future development scenarios. By modeling the structure of future alternative development pathways for Marcellus shale well pads, access roads, and gathering lines throughout the lifetime of gas development, the study quantifies the potential ecological impact of future drilling using a core forest expanse in the southwestern portion of Bradford County, PA (northeast PA) as a case study.

2.3 Methods

This study expands on previous work[12, 153, 156, 160] by modeling the pipeline and road construction that would accompany the projected well pad development. It then uses both the primary and secondary infrastructure projections to quantify overall deforestation and land use change under various potential regulatory scenarios geared towards preserving habitat while preserving ability to produce the shale resource. For this study, total lifetime Marcellus Shale production was held constant while varying well pad density. This represents alternative development pathways for achieving the total expected level of natural gas extraction throughout the lifetime of the play.

2.3.1 Study Area

The methods for determining the impact of natural gas development on forested land were developed and assessed by conducting a case study in Bradford County, PA, which overlies a highly productive portion of the Marcellus Shale play. Gas production in Bradford County was 870 billion cubic feet between 2010 and 2012[161]. The county was chosen because it has up to date records of pipeline development and is composed of about 56% forest. Specifically, the area modeled was a core forest region located in the southwestern corner of the county. The region is 35,000 ha of which 55% is public land and 91% is forest. Through 2012, 25 well pads had been developed in this region (Figure 5). By focusing on a minimally developed forested area, the potential influence of infrastructure development not related to the oil and gas industry is minimized.

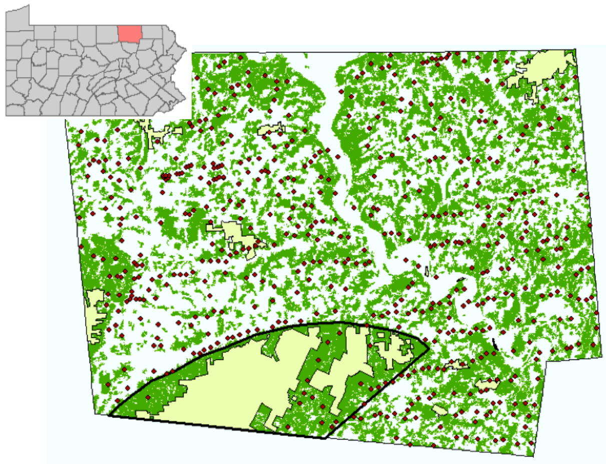


Figure 5: Bradford county forested land (green), state land (yellow), and well pads constructed through 2012 (black dots). The thick black line denotes the case study core forest region.

2.3.2 Land Change Model Development

Visualizing and analyzing land use change was accomplished using a spatially explicit model built in ArcGIS. This model was applied to both historical development to characterize past land change, and to forecasted gathering line and road routing, allowing for a complete picture of future fragmentation. This process for simulating land use change was modeled after the United States Geological Survey (USGS) method for updating land cover maps to account for new natural gas infrastructure[11].

Quantifying habitat impact was a four-step process. First, the infrastructure's geospatial position was determined. For well pads this position was a single point within the study region, and for access roads and gathering lines these were polylines. Second, these identified locations were enlarged (buffered) from points and polylines to the average footprint of the infrastructure they represent. For example, the points representing well pads were enlarged to an area of 3 acres, the average footprint of a well pad. Third, the forested regions that now contain natural gas infrastructure were re-categorized as non-forest on the land use map. Finally the chosen landscape metrics, as shown in Figure 6, were quantified.

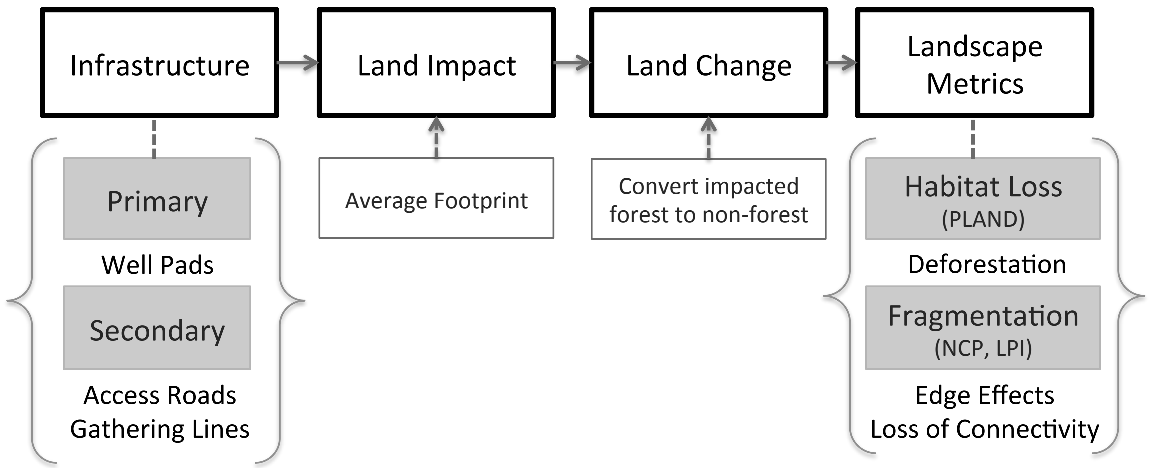


Figure 6: The general procedure for depicting land disturbance from Marcellus shale development

2.3.3 2000 Land Use Data

Year 2000 was chosen as the baseline year because 99.8% of wells exploiting unconventional gas development were drilled after this year. Land disturbance in the study region prior to 2000 was considered non-Marcellus shale related. The land use data raster file was obtained from Penn State University in the format of an ESRI raster grid, a file format compatible with ArcGIS[162]. Because the focus of this study is solely on forest ecosystems, this raster was reclassified into two categories of land use: forest (including coniferous, mixed, and deciduous forests) and non-forest. The original resolution of the raster was 30x30m. However, some components of Marcellus shale development such as roads are too narrow to be captured at this resolution. Therefore, the raster was resampled to a 10x10m resolution and the pixels underlying PA's year 2000 road network were re-categorized from forest to non-forest[163].

2.3.4 2012 Infrastructure

Year 2012, the most recent complete year of well location data available, was chosen to represent the current Marcellus shale development. Data on existing 2012 road network and the existing well pads were obtained from U.S. Census Bureau[163] and the PA Department of Environmental Protection (DEP) database[27], respectively. Roads connecting state and local roads to well pads are mostly private roads and are therefore missing from the road database. These roads were modeled as straight lines between the well pad and the nearest road in the road dataset. The lengths of the modeled roads were compared to lengths of roads constructed for Marcellus development as documented in the literature for validation (Figure 7)[164]. All roads were then buffered to 10 m to represent a typical footprint of a road used in Marcellus shale construction[165].

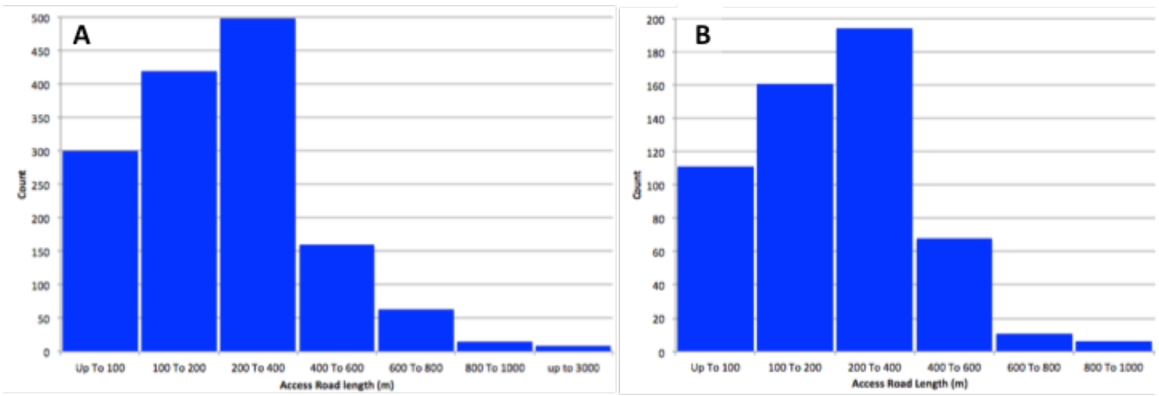


Figure 7: The distribution of access roads built for Marcellus shale development through 2012 (A) as built across all of PA, and (B) for Bradford County as modeled by the straight-line method.

In Bradford County, detailed locations of gathering lines are maintained by the county's planning office. These gathering line routes developed through 2012 were compared to a theoretical shortest length gathering line network developed by the model. This was done to (1) understand the potential for reducing surface disturbance from gathering lines, and (2) to identify the ways in which the modeled theoretical future gathering lines could vary from how the industry might realistically develop in the future without regulatory guidance. This theoretical network was created by first locating the two main transmission lines in the county from a map of PA's major gas pipelines[166]. The modeled gathering line network was constructed by adding the 2012 well pads one at a time in order of spud date and connecting each to the nearest previously existing main transmission line or gathering line. The resulting network was manually edited to reduce the number of hook up points to the main transmission lines to more closely resemble the observed number. The final modeled gathering line network had approximately 20 hook up points as compared to 16 hookup points identified by manually searching Google Maps along the two main transmission lines running through Bradford county. Marcellus gathering lines right of ways have documented widths ranging from 10-46m and main transmission lines have been measured at widths of up to 61m[154]. Therefore, in this study the gathering lines were buffered to

15 m and the main transmission lines were buffered to 46 m to represent conservative estimates of right of ways for the two installations. The buffered well pads, roads, and gathering lines were then overlaid on the land use raster and underlying forest was re-categorized as non-forest.

2.3.4 Infrastructure Impact

The areal extent of the well pad is often thought to be the dominant surface impact of natural gas development[167]. However, the principles of landscape ecology predict that the primary significant ecological impact will result from the linear gathering line corridors cutting through forest. To test this hypothesis, landscape metrics were calculated for disturbances from each of the three components of infrastructure individually and compared to landscape metrics resulting from the complete network of projected alternative development pathways for well pads, roads and gathering lines.

2.3.5 Policy Scenarios

The two regulatory measures considered here for reducing the impact of natural gas development are: (1) reducing well pad density, and (2) requiring gathering lines to follow the path of pre-existing roads in forested regions. Reducing well pad density can be accomplished by increasing the number of wells per pad, and/or by elongating the laterals[168].

In this study, six different well pad densities were considered to understand the impact of regulating the spatial distribution of well pads. Three of the scenarios were developed by TNC for their Energy Impact Assessment. An implied lateral length was calculated from the study's minimum separation distance. These three scenarios had 145, 88, and 58 well pads, with corresponding implied lateral lengths of 760 m, 880 m, and 1,100 m[153]. The remaining three scenarios were chosen to represent the current technological frontier in horizontal drilling (28 well pads with an implied lateral length of 2,100 m) and two scenarios representing technological advancement (8 and 5 well pads in the study region with implied lateral lengths of 4,300 m and 6,100 m respectively). These additional three scenarios were developed according to the TNC methods. First, the Nature Conservancy's probability surface was resampled to raster cells with diagonals that are twice the expected lateral length. Diagonals were used to designate minimum distance between well pads because the direction of the lateral is controlled by the minimum horizontal earth stress which is oriented NNW-SSE in PA[26]. Next, a point was placed in the center of each raster cell and the number of points equaling the expected number of well pads with the highest probability according to the resampled probability layer were chosen as the well pad locations. Finally, each well pad was then buffered to 1.2 ha circles (radius = 62m) to represent an average well pad footprint (Figure 8). These six scenarios correspond to well pad densities of 48, 32, 23, 15, 9.4, and 8.6 well pads/100km^2 (Table 1). The resampling process used to locate future development introduces variability across the scenarios in the probabilities used to identify well pad locations. This occurs because new probability values are interpolated from different cells of the original probability raster depending on the assumed lateral length for the specific scenario. To account for this uncertainty, a Monte Carlo analysis was conducted with randomized well pad placement within the probability raster cells.

Table 1: A summary of the implied lateral length, number of well pads, and well pad density of the six future development pathways

Scenario (#)	**Implied Lateral Length (m)**	**Number of Well Pads (#/Study Region)**	**Well Pad Density (#/100 km2)**
1	760	145	48
2	880	88	32
3	1,100	58	23
4	2,100	28	15
5	4,300	8	9.4
6	6,100	5	8.6

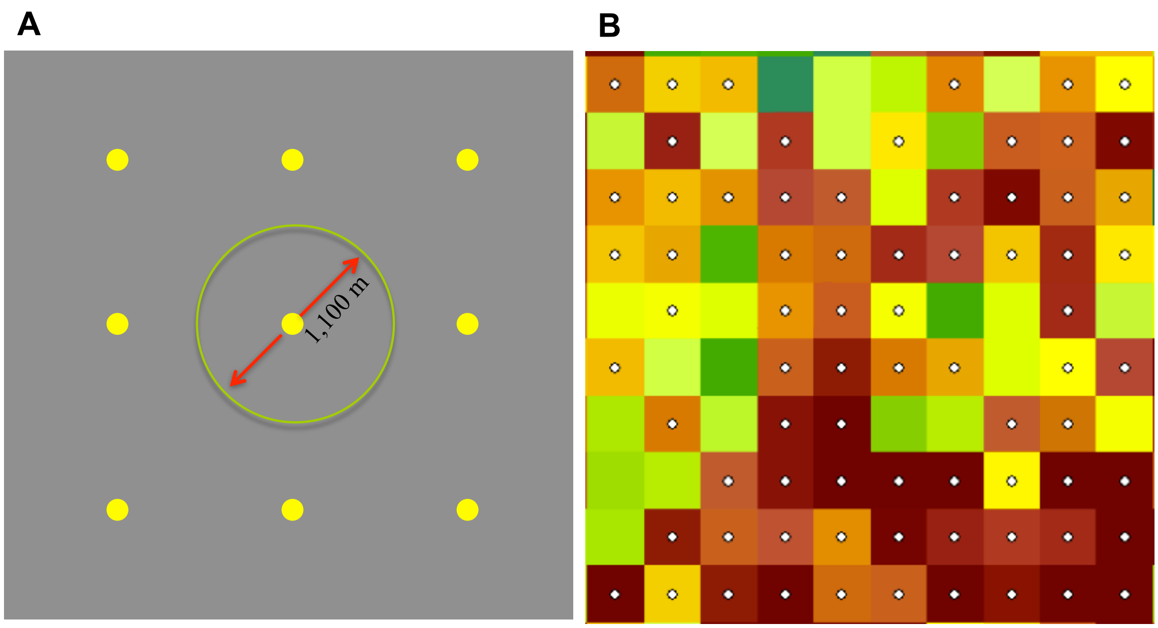


Figure 8: (A) The minimum separation distance diagonally was set equal to twice the length of the lateral in a given scenario, and then (B) the probability surface raster was resampled to reflect the minimum separation distance and well pads were randomly placed, filling the high probability sites first.

The roads for each scenario were modeled as straight lines between the well pads and the existing road network as of 2012 and then buffered to 10m. The gathering lines were modeled by connecting each well pad to the existing gathering line network one at a time in order of descending probability of future development based on the assumption that the likelihood of development is representative of a realistic construction order. These gathering lines were then buffered to 15 m. Finally, the buffered well pads, gathering lines, and roads for a given future scenario were overlaid on the derived 2012 land use raster. All raster cells underlying the infrastructure were classified as non-forest.

Policy scenario two, requiring gathering lines to follow the route of pre-existing roads, was modeled using the 32 and the 15 well pads/100km^2 scenarios. These scenarios were chosen because they represent both a typical lateral length and an approximate maximum length achievable with current horizontal drilling technology. The gathering lines were forced to follow the path of the existing roads by using the least cost path tool in ArcGIS where existing roads were given a resistance cost of 1 and the rest of the case study region was assigned a resistance cost of 100.

2.4 Results and Discussions

The impact of Marcellus Shale development on Pennsylvania's forests is of major concern to stakeholders interested in preserving the forest ecosystem and protecting Pennsylvania's endangered species that live in core forest habitats. The results of this study describe both the landscape impact that has already occurred as a result of historical natural gas development, as well as the potential future landscape disturbance due to unmanaged future development. Finally, the study results lead to potential policy measures that could mitigate the risk of additional forest fragmentation, while simultaneously allowing for natural gas extraction.

2.4.1 Historical Disturbance

Natural gas development over the past decade has contributed to fragmentation across all of Bradford County[12, 153]. Between 2000 and 2012, approximately 110 km of roads and 1,600 km of gathering lines have been constructed to support 1,080 wells. This has resulted in the loss of about 13,000 ha of core forest (including core forest lost to the creation of new edges) and an increase in the number of core patches in Bradford County from 900 to 1000 (Table 2).

Table 2: Historical land disturbance as built and as modeled

Scenario	NCP (#)	LPI (%)	Deforested (%)
2000	900	5	51
2012 as built	1000	4	51
2012 as modeled	970	4	51

This historical natural gas development has already impacted the core forest study region of Bradford County. Between 2000 and 2012, 25 well pads were developed in this area. Comparing the 2000 land use map to the 2012 land use map developed with the efficient theoretical gathering line route demonstrates that the number of core patches increased by 25% (from 65 to 81). Additionally, the single largest patch decreased from about 16% of the total habitat area to about 12% (Figure 9).

The model developed to simulate gathering line construction prior to 2013 was based on straight-line connections between well pads and the identified main transmission lines. This strategy resulted in a 2012 gathering line network that followed a similar pattern as the actual network of gathering lines constructed in Bradford County, although qualitatively, with fewer redundancies and extraneous branches. The modeled gathering lines (Figure 10B) totaled 900 km of pipeline across all of Bradford county versus the 1700 km of pipeline as built (Figure 10A)[169]. The modeled gathering lines serve as the baseline historical development that was used as the starting point for projections for future alternative development pathways. Thus, projected fragmentation modeled here is most likely an underestimation of the potential impact. Additionally, the drastic reduction in total pipeline length between the developed and modeled infrastructure shows that there are configurations that could connect the same number of wells at the same locations with reduced pipeline development. This implies that there could be an opportunity for concomitant economic and ecological benefits.

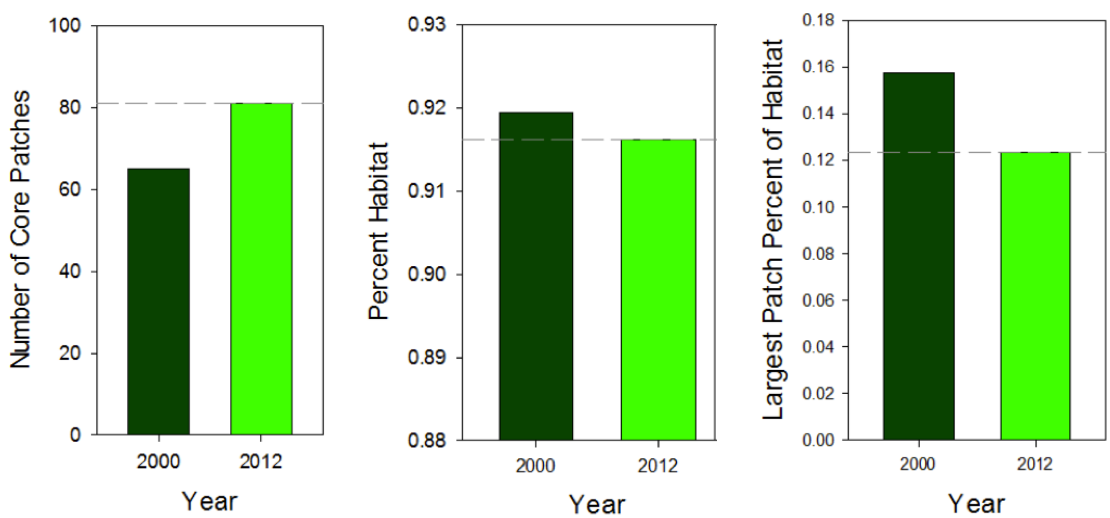


Figure 9: Modeled historical land disturbance in the study region of Bradford County

Upon inspection, Bradford County gathering line development shows that much of the development is in rural areas in relatively level terrain and thus did not appear to be impacted specifically by topographical concerns or land use issues. While further model refinement (e.g., addition of topography, land ownership, and additional land-use categories) could modify the modeled gathering line system to more closely resemble the shape of the current network, the model would still not follow the precise patterns of the developed gathering lines in Bradford

County, because the county was developed with little or no coordination by seven independent companies each trying to minimize their own costs.

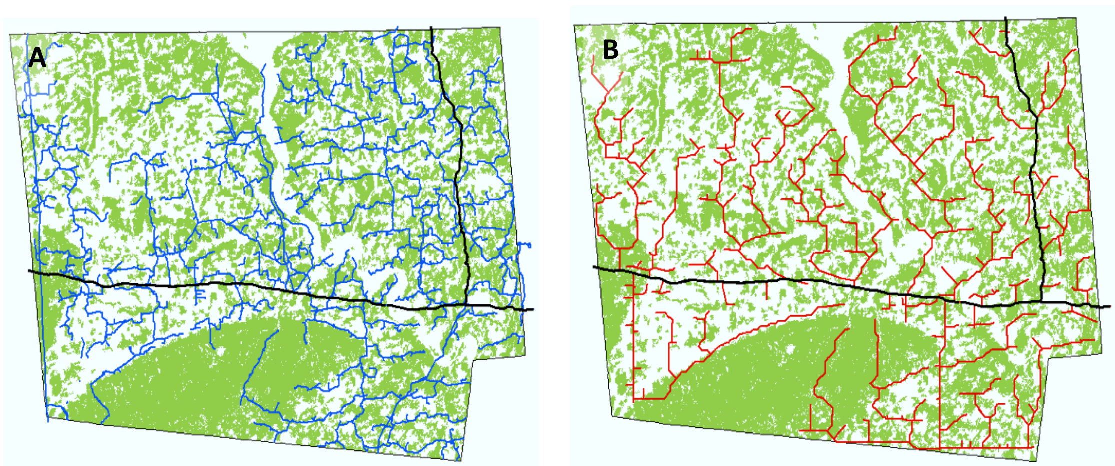


Figure 10: Main gas line (black) and gathering line network in Bradford County (A) as built through 2012 (blue) (B) as modeled by the straight-line method (red)

2.4.2 Infrastructure Impact Results

Directional drilling has been suggested as a solution to minimize natural gas-related deforestation. By drilling multiple wells per pad, developers claim they are reducing their surface disturbance and impact. However, by isolating the three key components of infrastructure and measuring their individual contributions to fragmentation, it is clear that well pads are responsible for little incremental land disturbance (Table 3) In the forest study region, the 2012 well pads alone were responsible for increasing the number of core patches by two. Likely this increase is due to the development of new roads between 2000 and 2012 rather than the development of the well pads. The access roads in isolation contributed to four additional core patches, and finally the gathering lines in isolation were responsible for a 20% increase in the number of core patches (14 additional core patches).

When the number of well pads increases beyond the 2012 existing development by 58 well pads in the study region, a similar trend is observed. The well pads and access roads in isolation cause minimal marginal impact, whereas the gathering lines are responsible for a prominent increase in core patches from 81 to 133. Thus, policies for reducing fragmentation from natural gas development should focus on regulating gathering line development.

Table 3: The isolated incremental impact of well pads, access roads, and gathering lines for 2012 and the 58 well pad scenario

Development Scenario	Number of Core Patches (#)	Largest Patch (%)
2000 (No Development)	***65***	***16***
2012 Well Pads	67	16
2012 Access Roads	69	16
2012 Modeled Gathering Line	79	13
Total 2012 Development	***81***	***12***
58 Projected Well Pads	81	12
Projected Access Roads	85	12
Projected Gathering Lines	133	10
Projected Total Future Impact	***151***	***9***

2.4.3 Policy Scenario Results

The two regulatory measures considered here are: (1) reducing well pad density, and (2) requiring gathering lines to follow the route of pre-existing roads. Reducing well pad density will decrease the number of well pads and reduce the number of required gathering lines. Requiring gathering lines to follow the route of roads would not reduce the quantity of gathering lines, and would likely increase the total length of the gathering lines but this would follow infrastructure that already has caused fragmentation. The important implication of this policy is that the location of the gathering lines would be regulated so as to not create new corridors across the core forest.

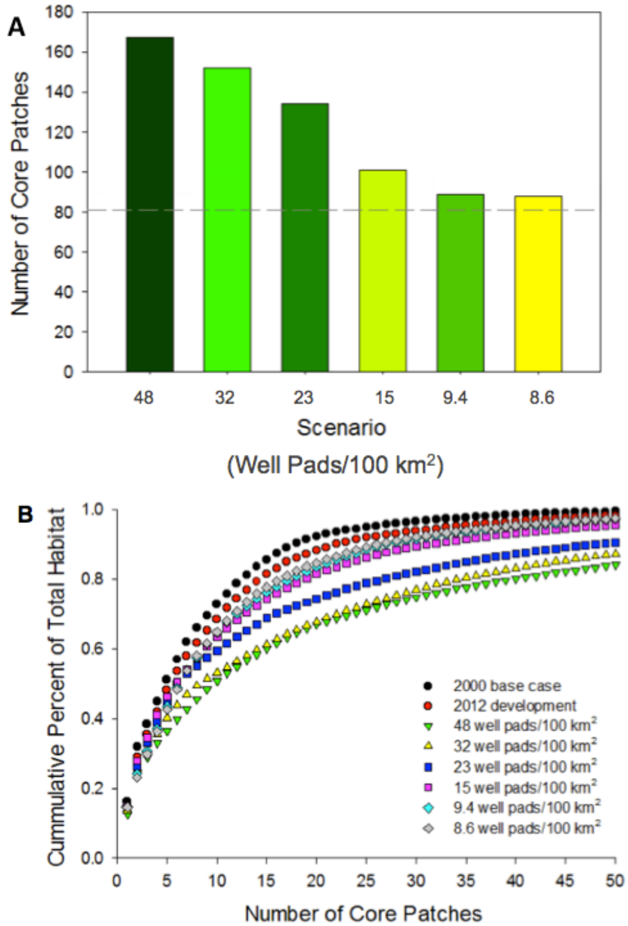


Figure 11: As the well pad density decreases (A) the total number of core patches decreases and (B) the area of each core patch remains larger

In the core forest study region, limiting well pad density results in the creation of fewer additional core patches. Furthermore, as well pad density decreases, the size of the core patches increases (Figure 11). Assuming the baseline year 2000, fragmentation from Marcellus Shale development through 2012 has increased the number of core patches by 25%. If left unchecked, future development could further increase the number of core patches from 81 in 2012 to 167 throughout the lifetime of the play. This would be a 100% increase above the 2012 level of fragmentation. Similarly, the LPI (largest patch index) and PLAND (total percent habitat) decrease as lateral lengths decreases. Because LPI and PLAND are indicators of fragmentation, the decreasing metrics signify that additional fragmentation is projected to occur (Figure 12).

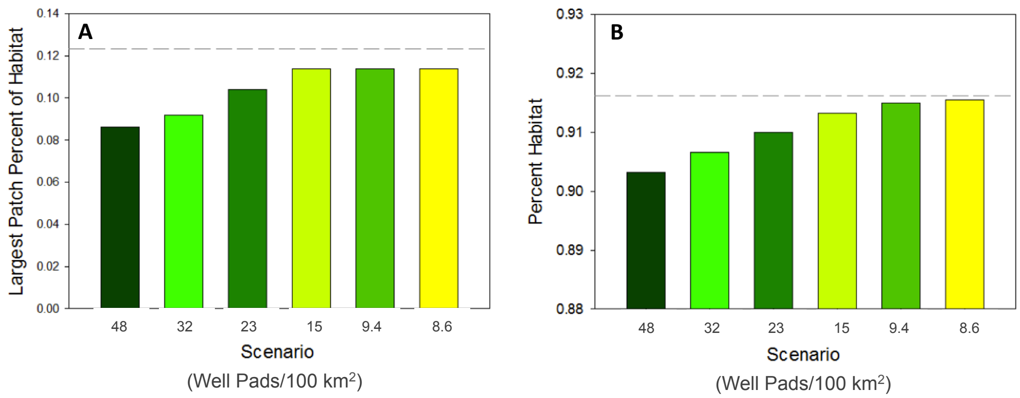


Figure 12: Changes in the (A) LPI and (B) PLAND for the six scenarios of varying well pad density

Horizontal drilling technology is currently capable of achieving laterals approximately 3,000 m long. Therefore, the minimum achievable density is about 15 well pads/100 km^2. This would still result in an increase in the number of core patches from 81 to 101 core patches, or a 25% increase above 2012 levels. The results show that even with a significant advancement in technology resulting in laterals over twice as long as currently feasible, 2012 levels of fragmentation cannot be maintained by decreasing well pad density alone.

The results of the policy alternative of requiring all gathering lines to follow the route of pre-existing roads (Figure 13) show that this policy essentially maintains fragmentation at the 2012 level regardless of well pad density. For example, in the 15 well pads/100 km^2 scenario, the model indicates that the number of core patches in the study region increases by one when the gathering lines follow the route of pre-existing roads, as opposed to the incremental 20 patches projected by the straight line gathering line model. Additionally, as shown in Figure 14, the distribution of core patch area is maintained at the 2012 level in addition to the number of core patches.

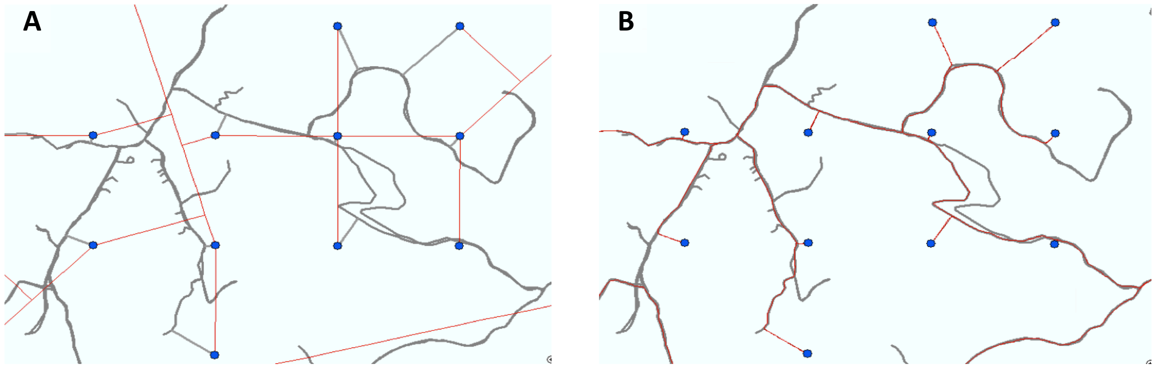


Figure 13: Gathering lines (red) can (A) be most efficiently linked to each other in a network independent from roads (gray) or (B) they may be routed so as to follow the path of existing roads

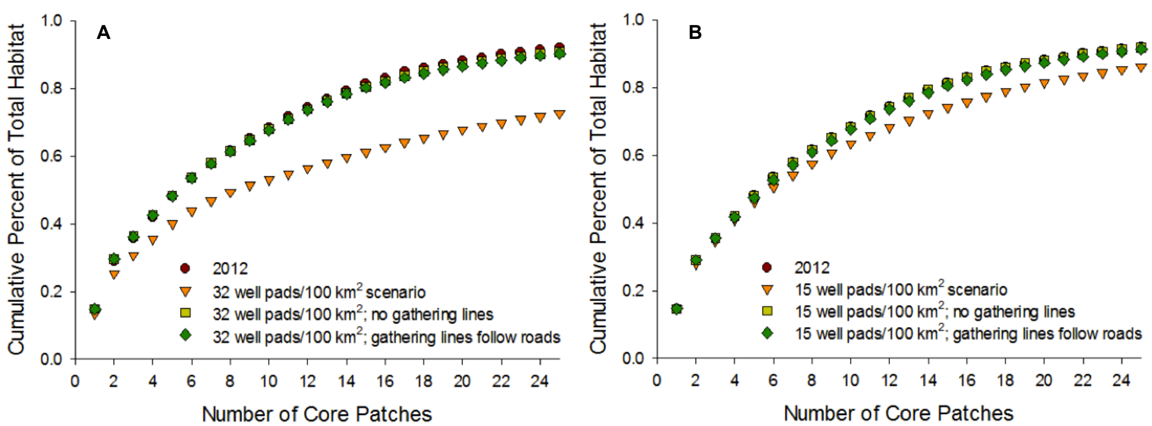


Figure 14: Policy scenario two compared to the (A) 32 well pads/100km^2 scenario and (B) 15 well pads/100km^2 scenario with and without gathering lines

Uncertainty analysis to determine the sensitivity of these policy scenario results to variations in projected well pad location within a given scenario was conducted using a spatially explicit Monte Carlo simulation. The original TNC method for well pad placement assumes that the well pads will be located in the center of the cells from the re-sampled probability surface map. This simulation was done to explore how the results change when well pads were instead allowed to be located randomly within the cell, given that the minimum separation distance is still observed. The uncertainty analysis was done using a spatially explicit Monte Carlo simulation that randomly placed the required number of well pads using code written in the statistical software program R and constructed the associated infrastructure and calculated the resulting landscape metrics for each iteration using a program written in ArcGIS 10.1. In this study, the Monte Carlo simulation was conducted for well pad densities of 32, 23 and 15 well pads/100km^2, which represent a realistic range of implied lateral lengths.

The inputs for the simulation were derived in R by using the rSSI tool in the Spatstat package[170]. The minimum acceptable distance between points was given as twice the desired lateral length. The maximum number of points used was the number of well pads falling within the forest case study region in the respective scenario above. For each iteration, the rSSI tool continued to add points until it was unsuccessful at locating an additional point given the minimum distance constraint 10,000 times. ArcGIS could not be used for generating these random points because the software limited the number of points that could be generated with a given minimum separation distance. The randomly chosen points were then assigned the probability value according to the resampled TNC probability raster used in deriving the scenarios as described above. These generated scenarios were then used as the input to the model, and the appropriate roads and gathering lines were drawn and landscape metrics were calculated for each.

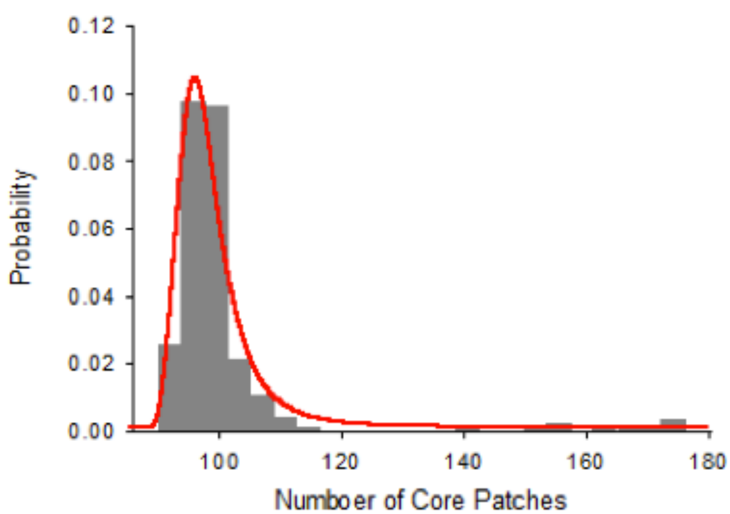


Figure 15: Simulation result and best-fit distribution (red line) for the number of core patches for the 28 well pads/study region density scenario

The distributions and 90% confidence intervals (Table 4) show that the location of the well pads results in varying levels of fragmentation even when the quantity of well pads is held constant (Figure 15). Despite this uncertainty, across the scenarios, the average number of core patches decrease from 140 to 100, and the average largest habitat patch increases from 8% to 11% as the well pad density decrease (Figure 16).

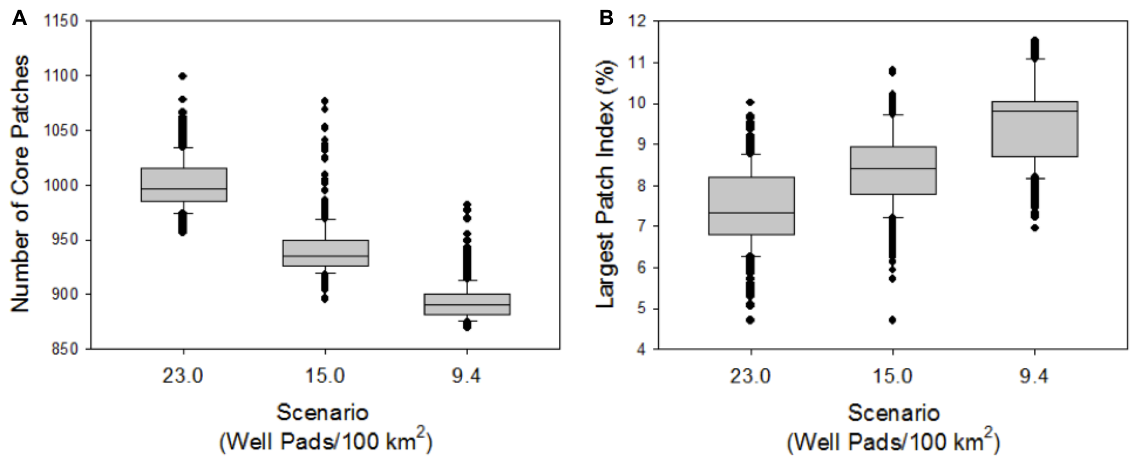


Figure 16: The simulation results for (A) NCP and (B) LPI for the 58, 28 and 8 well pads/study region scenarios

Table 4: Simulation results and best-fit distributions for the spatially explicit Monte Carlo simulation. As expected, there was variability in the landscape metrics depending on the location of the well pads.

Scenario	Metric	Number of Well Pads Sited	Distribution	Mean	90% CI	Number of Simulations
4	NCP	15-21	Log-logistic	100	[93,110]	500
4	LPI	15-21	ExtValMin	11%	[9%,12%]	500
3	NCP	49-58	BetaGeneral	130	[120,140]	250
3	LPI	49-58	ExtValMin	9.5%	[7%,11%]	250
2	NCP	76-88	Weibull	140	[130,150]	500
2	LPI	76-88	BetaGeneral	8%	[5%,10%]	500

While the metrics do vary depending on where the well pads are located, the results of the Monte Carlo simulation confirm that there is a distinct difference in the average level of landscape disturbance across the scenarios despite the uncertainty in exact well pad placement.

While fragmentation would increase beyond 2012 levels in all six scenarios, encouraging a decrease in well pad density via increased lateral length can have a small but positive impact on ecological conservation. Furthermore, increasing lateral length reduces the uncertainty in fragmentation impacts to the landscape. These results support the best practices as outlined in the literature [171] and as designated by the PA DNCR [172] that suggest careful consideration of well pad location prior to development can be a productive measure in reducing the impact of Marcellus shale surface disturbance.

This study demonstrates that Marcellus shale development has caused and will continue to cause further ecological disturbances in Bradford County's largest region of contiguous core forest if left unregulated. While the numerical results of this study are specific to the forested case study region within Bradford County, the observed trends and conclusions could be extended to the Commonwealth of Pennsylvania as a whole, across the United States, and globally using these techniques and approaches. Although there may be differences in the types of land cover and the degree of development, it is likely that the shale gas development will proceed rapidly and follow a spatial pattern dependent on the resources' potential and not constraints of the landscape, its cover, or ecosystem value[12].

While minimizing well pad density provides localized habitat conservation benefits, based on the analysis of the impact of each component of infrastructure in isolation, it is apparent that gathering lines are the single largest contributor to large-scale forest fragmentation from natural gas development. As a result, regulatory measures seeking to minimize the land use impact of Marcellus development on core forest should target gathering line routing practices. Specifically, using the core forest region in Bradford County as a case study, I find that requiring gathering lines to follow the route of pre-existing roads within the core forest region would successfully prevent additional fragmentation.

Successful implementation of such a gathering line siting regulation within a forest region would require both that the land be available for development and gathering line construction be coordinated among operators. These two constraints could be addressed through compulsory pooling and unitization laws. Compulsory pooling mandates that a landowner lease his land if a threshold percentage of neighboring land has been leased. Unitization requires a single operator to be responsible for coordinated development of gathering lines to all well pads in a given region. In addition to eliminating gathering line redundancies and reducing fragmentation, unitization would economically benefit developers; the 2012 modeled gathering lines in this study is representative of a unified development approach and demonstrates that the total length of gathering lines necessary to reach all well pads in Bradford County could have been reduced by almost half (from 1700km to 900km) had unitization been historically required. This reduction in gathering line length is significant, given the average cost of $1-$1.5M per mile of pipeline[155]. Furthermore, because developers can range from multinational corporations to small family owned operators, a significant challenge in regulating shale gas production has historically been the wide range in the abilities of actors in the industry to adhere to new regulatory requirements[173]. Unitization would overcome this challenge by forcing all developers to contribute proportionally to cooperative gathering line siting.

Compulsory pooling and unitization are already established in Pennsylvania through the Oil and Gas Conservation Law of 1961. However, as written, this law applies only to well bores that penetrate the Onondaga Horizon, thus Marcellus wells are currently exempt [26]. Because future natural gas development in Pennsylvania is likely to be primarily from shallower well bores targeted at the Marcellus Shale play, this law should be amended to include compulsory pooling and unitization for Marcellus wells in support of gathering line siting policies. Successful implementation of these policies would also require a clear plan for obtaining funding to enforce the rules and appropriately address violations to ensure remediation and dis-incentivize future violations[174].

In further support of compulsory pooling and unitization, Pennsylvania should adopt a requirement of comprehensive drilling plans (CDP) for the unit. CDPs are currently voluntary in Colorado and are being considered as a best management practice in Maryland. A CDP requires an operator to outline all aspects of any foreseeable future development (including resource protection, environmental monitoring, gas transmission plans, etc) in a given geographic region. This would allow developers to work together to create an integrated plan for efficient development[175]. By carefully mapping the “constraints” on gas development presented by a variety of environmental and socioeconomic factors and by identifying the foreseeable oil and gas activities in a defined geographic area upfront, energy companies working cooperatively with other stakeholders (including state natural resource agencies) can exploit the resource while minimizing impacts on local communities, ecosystems, and other natural resources.

Political economists believe that states might choose to minimize governmental interference out of fear of driving away developers, thereby pursuing short term economic gain over long term risk management[176]. Although gathering line siting regulations would increase the economic investment required by developers, over the lifetime of the play the incremental investment would be trivial. In the 15 well pads/100km^2 scenario modeled here, the total length of the gathering line network would increase by 4 km over the efficient 2012 gathering line network (from 96 to 100 km) within the core forest region. At an estimate of $1M to $1.5M per mile of gathering line[155], this regulation might cost an additional $2.4-$3.6M over non-restricted gathering line development in the core forest case study region. Given an estimated ultimate recovery (EUR) of 1-3 billion cubic feet (Bcf) per well[177,178,179], for this scenario of 28 additional well pads and assuming an average of 6 wells per pad, this translates into an investment of approximately $0.005 to $0.022 per Mcf of natural gas produced (Table 5). The minimal cost is more than offset by a more optimal development scenario and allows for preserving the delicate pattern and structure of contiguous core forest habitat while allowing private industry to engage in development and to exploit the resource.

Table 5: Assumptions and input ranges for the calculation of economic investment (over the lifetime of the wells) of constructing gathering lines that follow the route of pre-existing roads

Value	Parameter
4	km of incremental gathering lines
1.6	km/mile
1000	cf/Mcf
6	wells/well pad
28	well pads
4	additional gathering line length (km)
1-3	expected ultimate recovery (EUR) (bcf/well)[177,178,179]
1-1.5	cost of gathering line ($M/mile)[155]

Chapter 3. A systems level perspective on Marcellus wastewater management in Pennsylvania

3.1 Abstract

Contaminated wastewater, called produced water, is the largest byproduct of natural gas extraction. In Pennsylvania, prior to 2011, natural gas producers were able to truck wastewater to publicly owned treatment works for disposal. However, this practice was restricted after high levels of bromides were found in surface water discharge, which can lead to carcinogenic disinfection byproduct formation at drinking water treatment plants. The industry then began to reuse produced water to hydraulically fracture subsequent wells. This is a productive management strategy of produced water in the short term; in the long term, however, once the rate of well development slows all of this produced water will need to be either treated or disposed of. Based on current well development and treatment costs, this study estimates 67% of the time Class II disposal is the least cost option, 25% of the time CWT is the least cost option, and 8% of the time on-site treatment is the least cost option. The corresponding average costs are \$5.80/bbl (\$0.015/Mcf), \$7.80/bbl (\$0.020/Mcf), and \$8.40/bbl (\$0.021/Mcf), respectively. In addition to cost, however, there are several technical, ecological, regulatory, and logistical issues that also affect the relative feasibility of these three produced water management strategies. If regulators could capture producers' willingness to pay to dispose of water rather than treat the water, that money could be invested in treating other water quality issues in Pennsylvania such as coal mine drainage, which can be treated for \$0.064/bbl on average, or agricultural runoff, which could be mitigated at an average cost of \$0.08/bbl.

3.2 Introduction

Oil and gas extraction is an important component of the United States economy, accounting for 430 billion dollars in GDP in 2014[180]. Along with the production of valuable energy resources, waste streams are also generated that need to be managed in an economically and environmentally sound manner. The largest by-product by volume accompanying energy extraction is wastewater[14]. Details about the volumes and management strategies, called produced water, are not well documented on a national scale across the thirty-one oil and gas producing states. A comprehensive study in 2015 estimated that the one million actively producing oil and gas wells in the United States generated over 21 billion barrels of produced water in 2012[87]. This volume is projected to increase to over 34 billion barrels by 2025[88].

The volume and quality of produced water is dependent upon the geographic location of the field, the geologic formation from where the water was produced, and the type of hydrocarbon being produced[14]. In the Marcellus region, each horizontal well will tap gas from approximately 83 acres[89]. The total volume of produced water returning to the surface varies from well to well and has been estimated as 8-15%[95], 10-40%[181], 9-53%[9], and 30-70%[182]. Initially, the produced water has a chemical footprint similar to the injected water. However, after the first few weeks, the composition of the water changes. Most noticeable, its level of total dissolved solids (TDS) increases dramatically until it approaches steady state at about 10% NaCl equivalent. Most likely, this is a result of the injected water interacting with high salinity brine within the Marcellus formation[89, 90,91, 92]. Because the salinity derives from the brine within the shale, the depth of the hydrocarbons influences the salt and mineral levels present in the produced water due to differences in shale origins; wastewater from shale extraction at 5,000-8,000 ft typically has twenty times the salt/mineral content of produced water from coal bed methane produced water[93]. This is important because while there are many contaminants in the produced water that need to be considered, the salinity, referred to as the total dissolved solids (TDS), is generally the defining characteristic that determines the economic and technical feasibility of various wastewater management options. Therefore, the way in which produced water is handled varies across the country and what is successful in one location may prove impractical in another region.

Produced water management has become of particular concern over the last five years in Pennsylvania. While oil and gas operations have been active in Pennsylvania since the 1960s, historically, the wells were in shallow reservoirs with relatively low TDS levels. The common practice was to dilute the produced water with fresh water and discharge into surface waters[183]. However, when technological advances in hydraulic fracturing and directional drilling enabled the economic extraction of tight oil formations, e.g. shale gas, natural gas extraction in Pennsylvania increased dramatically (more than 1,400% since 2000) as a result of Marcellus Shale development[184]. The significant increase in produced water accompanying new Marcellus wells between 2008 and 2009 led to a seven-fold increase in the total volume of produced waters sent to surface-discharging treatment plants[15]. Additionally, the produced water from the Marcellus can have TDS levels up to 360,000 mg/L[16]. The combination of volume and quality of produced water makes managing the byproduct especially challenging in the region. Many produced water treatment options historically used by the industry such as reverse osmosis cannot be applied to such high salinity water. Additionally, the humidity and rainfall in PA limit the viability of evaporation in pits, which is a widely used produced water management option in other regions of the country.

Managing this wastewater is a complex process that reflects tradeoffs in economic objectives, system reliability, risk, liability, infrastructure availability, and technological capability of meeting treatment and disposal water quality targets. The primary wastewater management options available to natural gas producers in Pennsylvania are reuse for subsequent well development, disposal in underground injection wells (deep well injection), or treat to surface discharge quality via onsite treatment modules or centralized waste treatment facilities (CWTs)(Figure 17). In this analysis, I develop a decision support tool and use it to compare the economic and greenhouse gas tradeoffs associated with each of these management strategies on a systems level across the Marcellus region.

While there have been several publications identifying decision-making strategies for well developers[185] including integrated multi-criteria decision making tools, The Produced Water Management System (PWMS) developed for the DOE[186], the Water Decision Tree developed by the Petroleum Technology Alliance Canada[187], and multiple optimization models[141, 188, 189]. These models represent strategies best suited to decision making for a single developer or group of spatially clustered developers. Additionally, when considering well development on a small scale, resources such as fresh water and disposal capacity may be functionally constrained. This would not necessarily represent the real time circumstances of the ongoing industry operations, especially within the Marcellus region where water resources are not limited. Thus, these tools may not be well suited to inform policy targeted at large scale, statewide water quality conservation and wastewater management regulation.

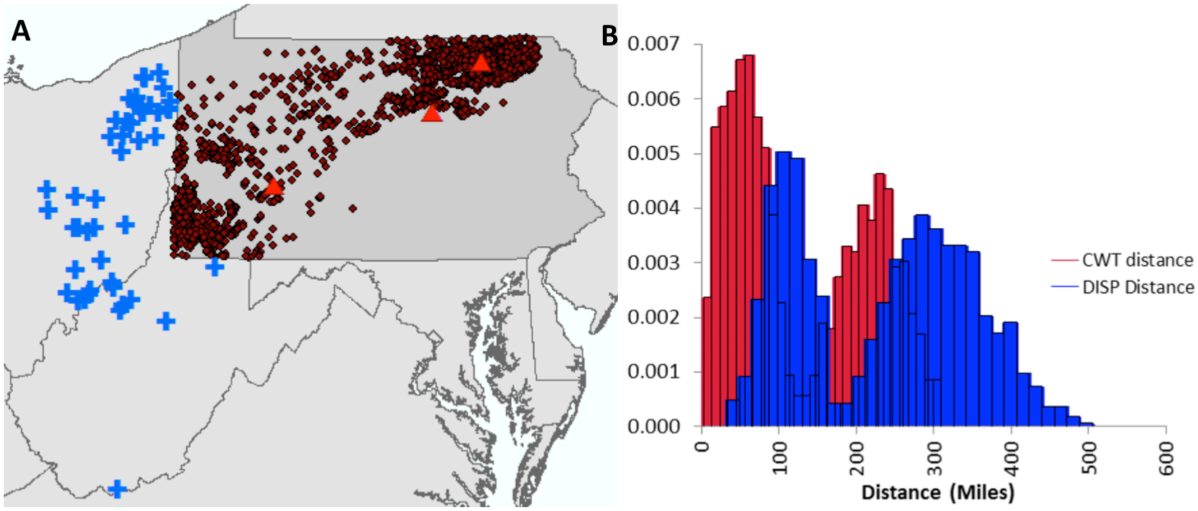


Figure 17: A) Map of Pennsylvania showing Marcellus horizontal wells (brown), CWT facilities (red), and Class II injection wells in Ohio and West Virginia (blue), and B) PDF of distances from all wells to all CWTs (red) and to all disposal wells in Ohio and West Virginia (blue). Distances include a 1.21 circuity factor.

3.3 Methods

3.3.1 Quantifying Marcellus Wastewater

In Pennsylvania, the Department of Environmental Protection (DEP) requires all natural gas developers to report the quantity of wastewater generated and the wastewater management method used (reuse, deep well injection, CWT, etc) on a per well per month basis. This data is then compiled bi-annually and made publicly available on the DEP Oil and Gas Reporting website[187]. In this study, I use the results from our own analysis of the DEP wastewater data validated with estimates from the other studies to develop a distribution to capture the likely range of wastewater generation for an average Marcellus unconventional well. I then use an estimate from The Nature Conservancy (TNC) of the total number of wells that could be developed in Pennsylvania throughout the lifetime of the play to quantify the potential total volume of wastewater generation from the natural gas industry in the Marcellus region.

3.3.2 Cost of Produced Water Management

Produced water has a high level of contaminants such as heavy metals and toxic chemicals, and a high level of salinity. In some cases, produced water can have up to six times the salinity of seawater. These TDS levels make treating the water to Environmental Protection Agency (EPA) established maximum contamination levels (MCLs)[190] energy intensive and costly. As previously discussed, CWTs with NPDES discharge permits must treat the water with primary, secondary, and tertiary treatment to remove dispersed oil and grease, remove soluble organics, remove bacteria, remove suspended solids, and desalinate the water[191-193]. Additionally, it may be required to undergo polishing and other miscellaneous treatments; for example if NORMS are present, it would be necessary to remove NORMs prior to discharging the treated water into surface water systems[182]. The primary cost driver of treatment is the tertiary treatment (desalination) stage. Most traditional desalination methods are not suited to treat the high TDS levels found in produced water[194].

To compare the produced water management alternatives of treating the water versus disposing of it, I developed a dynamic stochastic decision making tool. For a given well, water quality parameters and spatial characteristics are defined by the user. The objective is to identify the least cost management option by comparing the cost of treating the water to meet discharge water quality targets or disposing of the water in Class II disposal wells. For example, the user specifies contaminant concentrations of the current produced water stream and the distance to the nearest disposal well and CWT facility. The tool contains probability distributions representing costs of three different treatment trains for onsite treatment. Each train consists of a primary, secondary, and tertiary treatment process. The primary treatment filters the water to remove suspended solids or oil. The secondary treatment removes divalent ions such as calcium and magnesium. The tertiary treatment reduces TSS and can either be completed via Reverse Osmosis (RO), thermal distillation, or evaporation (Figure 18). Each of these tertiary treatments has design parameters specifying minimum input water quality contamination levels the secondary treatment must achieve prior to tertiary treatment. For example, RO cannot be applied to water with TDS over 50,000 mg/L due to membrane fouling limitations. These onsite treatment costs are then compared to injection disposal and CWT costs. Several studies[33, 188, 195] have estimated treatment costs based on published estimates in the literature[84, 93, 182, 196, 197]. I used these studies as the basis for treatment parameters and costs to develop most likely distributions (Table 6). Given these uncertainty distributions, the tool outputs cost estimates demonstrating the likelihood of economic tradeoffs between the alternatives. It is assumed that any flowback water that can be reused to hydraulically fracture a subsequent well would be designated for that purpose. Furthermore, even if produced water is reused to hydraulically fracture a subsequent well, eventually it will need to be disposed of or treated once supply of reusable water outpaces water demand for new well production.

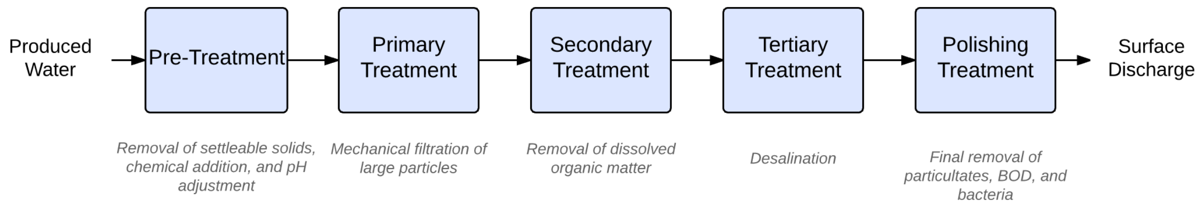


Figure 18: Sample process train for produced water treatment

In addition to characterizing treatment alternatives for an individual well, the tool can be applied more broadly across space and time to support produced water management policy making for the Marcellus Shale play as a whole. To do this, I developed distributions representing the distances from all wells to all disposal wells and CWTs using ArcGIS 10.1 based on spatial data of well locations included in the PA statewide waste reports between 2012-2014[161], the three CWT locations with high salinity produced water treatment facilities[187, 196, 198], and class II injection well locations used for Marcellus disposal[187, 199, 200]. I then applied a circuity factor of 1.21 to adjust straight-line distances to be representative of trucking distances via road and fit bimodal distributions to these data. These fitted distributions served as inputs to a Monte Carlo simulation to determine probabilistic estimates of wastewater treatment costs. The cost tradeoffs between treatment options are that CWTs benefit from economies of scale, however the developer incurs trucking costs (and liability risk of spillage) to transport the water for treatment versus the more costly onsite treatment that does not incur trucking costs. Injection disposal costs less per barrel of water disposed than does treatment, however because there are few local Class II wells in Pennsylvania, the water must be hauled to Ohio or West Virginia, which increases the total cost of injection disposal.

Finally, I estimated the present value of all future Marcellus water treatment to inform policy decisions based on the total cost of water management scenarios. Using the maximum, minimum, and most likely produced water generation estimate across the aggregated data, I developed a triangular distribution to estimate a likely volume of produced water generated per well. To estimate the total volume of water throughout the lifetime of the play, I used The Nature Conservancy's Pennsylvania Energy Impact Assessment projection of 60,000 wells[153].

3.3.3 Alternative Wastewater Sources and Treatment Options

To put these Marcellus produced water management options into context, it is important to understand the opportunity cost it poses by considering cost of treatment for other critical water quality threats in Pennsylvania. The two alternative sources of water pollution assessed in this study are coal mine drainage and agricultural runoff. A best management practice for CMD remediation is lime neutralization. To estimate the likely average CMD remediation cost, I derive distributions representing capital cost and operation and management (O&M) costs for low, medium, and high levels of acidity from existing estimates[201] across a range of treatment capacities. These estimates were validated against other reported values in the literature[202].

A best management practice to prevent agricultural runoff is to develop vegetative buffer strips (VBS) to act as a passive filter removing nutrients from runoff before it enters surface waters. In this study, I estimate cost of VBS implementation based on construction and maintenance data from Talbreth et al.[203] combined with land use opportunity cost values[14] and estimated corn, wheat, and oat farmland in PA derived from the 2012 agricultural census[204]. Typical annual runoff per acre was estimated using the NRCS curve number procedure. The curve number was assumed to be between 77 and 88[205], representing the runoff potential for row crops in soil hydrology group C, which is typical of PA soil[206].

3.4 Results and Discussion

The Marcellus is the most expansive shale gas play in the United States[10], covering 14–25 million hectares (ha) (35–62 million acres)[11]. It is estimated that 60 million gallons of wastewater will be generated each day in 2016[207], up from about 450 thousand gallons per day in 2007[14]. To hydraulically fracture a well, between 2-5 million gallons of water are mixed with proppant and fracturing chemicals and injected underground[208,82]. More than 1,000 different chemicals are known to have been used by various operators over time[71] ranging from benign to toxic to prevent bacteria growth, reduce mineral build up, and to ensure the proppant is well mixed in the fluid[83]. After hydraulic fracturing occurs, some of this injected water eventually returns to the surface of the well as produced water. In the Marcellus region, between 10-30% of the wastewater returns to the surface within the initial two weeks after the completion of the hydraulic fracturing process[191]. This initial stream of produced water is called flowback water.

Flowback water is defined in the Pennsylvania Department of Environmental Protection (PA DEP) reporting guide as the return flow of water recovered from the well bore of an oil or gas well within thirty days following hydraulic fracturing, or until the well is placed into production, whichever occurs first[209]. The water returning to the surface throughout the twenty to thirty year production lifetime of the well is referred to as produced water. However, due to the difficulty in distinguishing the two streams and inconsistent definitions used throughout industry and across the literature, both types of waste fluids are frequently referred to cumulatively as produced water.

Typically flowback water increases in salinity throughout the flowback period to over 100,000 ppm (w/v) as the injected fluid increasingly interacts with the underlying shale formation. Flowback water can be diluted and reused to hydraulically fracture another well. Historically, the flowback water required treatment prior to reuse, however companies have successfully revised the mixture of fracking chemicals to accommodate the chemical makeup of diluted, unprocessed flowback water after a simple, inexpensive filtration stage to remove suspended solids. Reusing flowback water has become a favorable strategy for developers in the Marcellus region. While this delays the need to address treatment options for this water, it does not solve the problem altogether; eventually the generation of flowback water will outpace new well development in the region and will need to either be disposed of or treated. In fact, this has been foreshadowed by the recent slow down in well development resulting from persistently low oil and gas prices[210, 211].

“Non-flowback” produced water (hereafter referred to as produced water or brine) poses a unique wastewater management challenge because of its chemical content, high salinity, and spatio-temporal granularity. In addition to the high TDS, produced water is contaminated with naturally occurring compounds within the shale formation, such as arsenic, naturally occurring radioactive material (NORM), and volatile organic compounds (VOCs)[208]. This brine continues to be generated at a rate of 400-4,000 gallons per day throughout the lifetime of the well[207], although the rate typically diminishes over time. Because the presence of TDS, barium, and other contaminants can interfere with the fracturing chemicals, produced water would have to undergo more extensive treatment in order to be feasibly recycled for subsequent hydraulic fracturing operations, which makes reuse with produced water more expensive than reuse of flowback water. In particular, the presence of sulfates in recycled produced water is a concern because they can cause scaling and block the hydraulic fracturing process[212]. Additionally, the rate at which produced water emerges decreases substantially over time, which would make produced water reuse logistically difficult as well as technically challenging.

Well producers have experimented with measures to reduce the volume of wastewater rising to the surface of the well as a means of reducing overall water management costs. Some of these measures include reducing the volume of water entering the well through mechanical blocking devices or water shut-off chemicals, or reducing the volume of water returning to the surface through dual completion wells or downhole oil/water separation[87]. However, many of these technologies are limited in application and/or not yet developed on a commercial scale. Another possibility might be to reduce the quantity of water required to hydraulically fracture a well, however I assume water sourcing costs and fracturing fluid costs have already incentivized producers to optimize the current water volume used in hydraulic fracturing. Alternatively, its been observed that well producers are hydraulically fracturing with excess recycled water in an effort to 'trap' the water within the well, thereby reducing the quantity of wastewater to be managed. While any safe measure to reduce the quantity of produced water to be managed should be encouraged, such measures would not fully eliminate the need for produced water management.

3.4.1 Characterization of Marcellus Wastewater

In this study, I conducted our own analysis of the DEP waste report, and used these results in conjunction with those of additional studies to estimate produced water trends for the average well in the Marcellus region. The DEP oil and gas waste reporting data have large inconsistencies across developers and across time. These challenges include: 1. wastewater for an entire well pad is sometimes reported under one well, 2. wells may not have produced water generation reported consistently across all reporting periods after the first date of production, 3. reporting requires listing only the first waste destination, it does not report the complete chain of custody of the waste, and is not updated if producers, for example, report waste as having been recycled when later it was instead decided to dispose of it instead, 4. some waste is reported as having been brought to CWTs under their recycle permit, when the facility in actuality has an NPDES permit and is discharging the treated water into surface waters. While the PA DEP is taking strides to be more stringent in reporting requirements to achieve greater accuracy, the relationship between gas production in the Marcellus region and wastewater generation is not well quantified. Despite these data limitations, some studies have attempted to work with the available data to identify reasonable estimates for flowback and produced water generation[95, 186, 195]. Additionally, I analyzed the data to validate these estimates. I did this by compiling the wastewater reports from the periods between 2008 and December 2015 (a total of 9,400 wells with spud dates in this period[27]) and matching them with the well production reports from the same periods. I made corrections for apparent averaged values across wells on the same pad and used weighted averages to adjust each well's reporting period to contain a full year or half year's number of reported producing days. I also removed wells that did not consistently report wastewater generation after the initial date of natural gas production. Many of these data analysis measures were consistent with those used in previous studies such as Lutz et al. (2013)[95], however I limited the wells to horizontal unconventional wells and had four additional years of data to include in the assessment. While it is not feasible to precisely address all reporting errors, I believe careful assessment of the data allowed us to characterize the potential range of produced water generation across the first six years of natural gas production of an average well.

Based on the compiled distribution estimates of produced water generation, I conducted a Monte Carlo simulation. The results of the simulation suggest an average of 36 thousand bbls (kbbls) of non-flowback produced water are generated per well (90% CI: 13-70 kbbls), primarily in the first two years of production (Figure 19). This includes waste reported under the category of "produced fluid," which should not include flowback water, however misreporting is possible. The remaining water returns to the surface in small volumes throughout the 20-40 year lifetime of the well. This temporal variation in water volumes and qualities suggests on-site treatment may not be technically or economically feasible beyond the first one to two years of production.

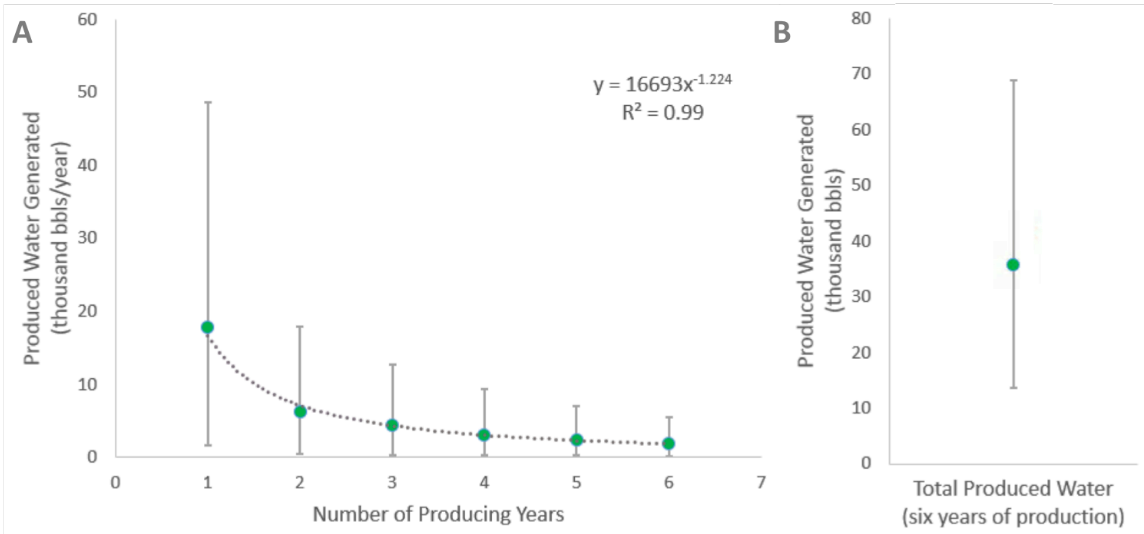


Figure 19: Estimates of produced water volumes A) generated annually per well, and B) summed across six years of production

3.4.2 Water Treatment Costs

There are potentially three management strategies for produced water, which is considered here: disposal via deep well injection, treatment at a centralized waste treatment facility, and onsite tertiary wastewater treatment. The water management tool (Figure 20) assumes the only zero discharge options use evaporation processes as a tertiary treatment option. This method is the only one feasible given the high level of TDS in the produced water. Using the compiled distributions of produced water treatment costs (Table 6) I used a Monte Carlo simulation to estimate the cost of each wastewater management option.

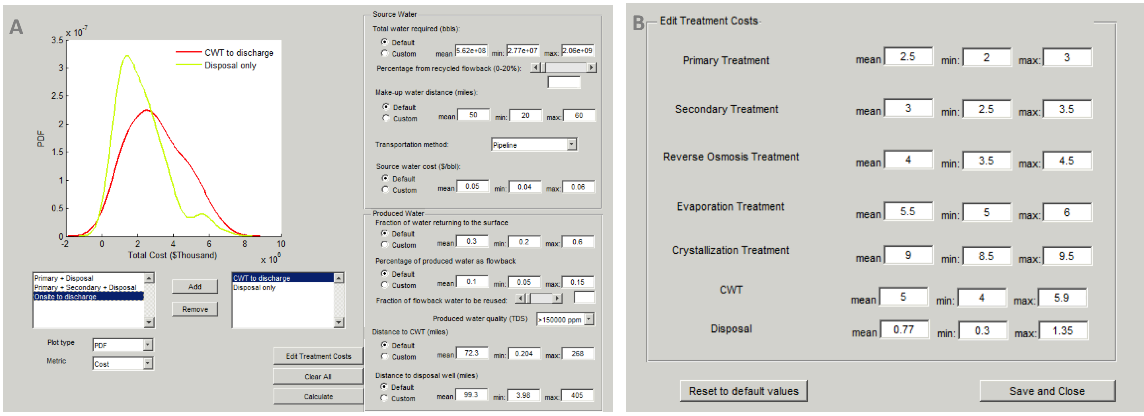


Figure 20: Screenshots of the produced water decision support tool A) user interface and B) treatment cost module. All inputs can be specified by the user or chosen to be default estimates

The results of the simulation suggest that estimates 67% of the time Class II disposal is the least cost option, 25% of the time CWT is the least cost option, and 8% of the time on-site treatment is the least cost option. The corresponding average costs are $5.80/bbl ($0.015/Mcf), $7.80/bbl ($0.020/Mcf), and $8.40/bbl ($0.021/Mcf), respectively (Figure 21). These results show that even with relatively long distance trucking, water to Ohio or West Virginia, deep well injection is economically preferable over other water treatment options. Assuming an average of 36 thousand bbls of produced water generated per well, on average deep well injection, CWT treatment, and on-site treatment would cost $210k, $280k, and $300k, respectively. Thus, deep well injection would on average save producers $70 thousand per well over CWT. While on-site treatment appears potentially economically feasible in this cost-analysis, no such facilities exist on a commercial scale. An estimate based on bottom up estimates of the unit processes required for pre-treatment, primary, secondary, and tertiary treatment plus a polishing step are over double this mean on-site treatment estimate. Additionally, the logistics of on-site treatment are particularly challenging given the temporal nature of produced water generation. Finally, the regulatory process for obtaining individual on-site disposal NPDES permits would be expensive and impractical for operators. Therefore, on-site treatment is not a likely wastewater management strategy, beyond pre-treatment of flowback water for reuse. In order to address the feasibility of disposal as a wastewater management strategy, a comprehensive analysis of wastewater disposal capacities, disposal rates, wastewater storage capacity, and temporal variation in wastewater generation across the entire play would need to be conducted.

Table 6: Water treatment cost parameters and assumptions

Assumption	Unit	Average	Range
On-site treatment cost[188, 213]	$/bbl	8.4	min = 6.8, max = 10
CWT cost[188, 195, 213]	$/bbl	5.10	min = 4.0, max = 5.9
Trucking cost[141, 213]	$/bbl-mile	0.02	min = 0.011, max = 0.03
Trucking distance to CWT	miles	140	5%= 20, 95% = 290
Trucking distance to disposal well*	miles	250	5% = 86, 95% = 430
Expected Ultimate Recovery[214]	10^6 m3	57	min = 14, max = 150
Deep well injection cost[188, 195,215, 216]	$/bbl	0.76	min = 0.17, max = 1.66

* distance includes 1.21 circuity factor

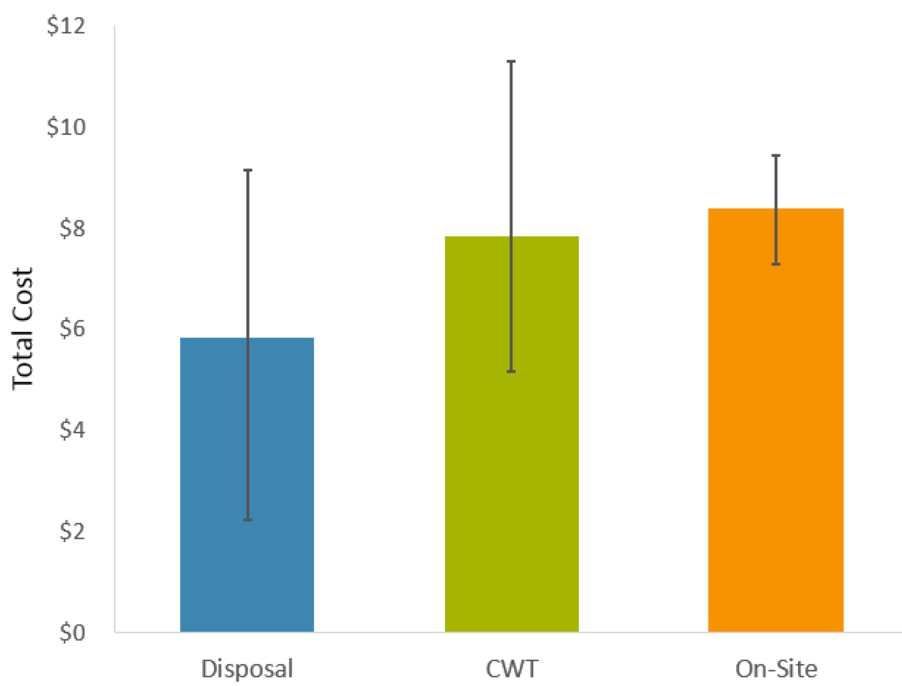


Figure 21: Cost of three waste management strategies for Marcellus Shale produced water

In Figure 21, the error bars indicate there are circumstances in which the combination of trucking distances, deep well injection cost, and CWT cost could lead to CWT being the preferred method. If instead of average costs, I consider the tradeoffs in these four parameters, I can map out this region of CWT preference. Figure 22 shows when CWT would be the least cost option as a function of distance to Class II well, distance to CWT, and cost of CWT for the minimum disposal cost (A) and the maximum disposal cost (B). Figure 23 displays which wells would have a lower cost of CWT over disposal, assuming an average disposal cost ($0.76/bbl) and average trucking cost ($0.02/bbl-mile). This analysis assumes no capacity constraints related to treating or deep well disposal.

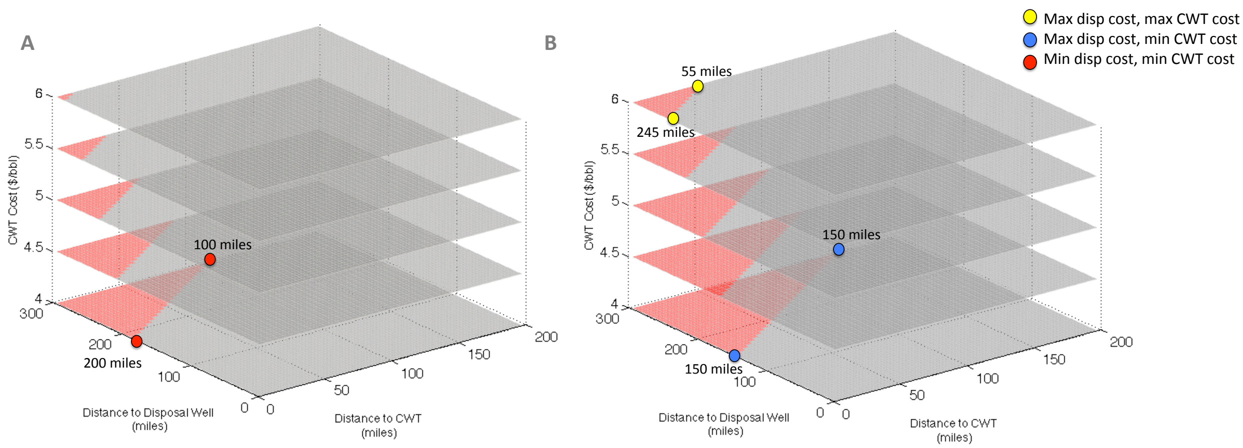


Figure 22: Produced water management decision as a function of distance to CWT, distance to disposal well, and CWT cost. Red indicates cost of treatment at a CWT is less than cost of disposal well for A) minimum disposal cost ($0.03/bbl) and B) maximum disposal cost ($1.20/bbl)

Table 7: Number of wells where CWT would be less expensive than disposal as a function of CWT and disposal costs. Total number of wells is 7,750 (through 2014)

CWT Cost ($/bbl)	**Number of wells choosing CWT vs. disposal**	**Percentage of wells choosing CWT vs. disp(%)**		
	At average disposal cost ($.76/bbl)	At average disposal cost ($.76/bbl)	At 95th percentile disp cost ($.97/bbl)	At 5th percentile disp cost ($.53/bbl)
6	1,660	21	24	0
5	2,560	33	40	30
4.25	3,940	51	53	47
3	4,360	56	57	56
2	4,670	60	62	58
0	6,600	85	92	78

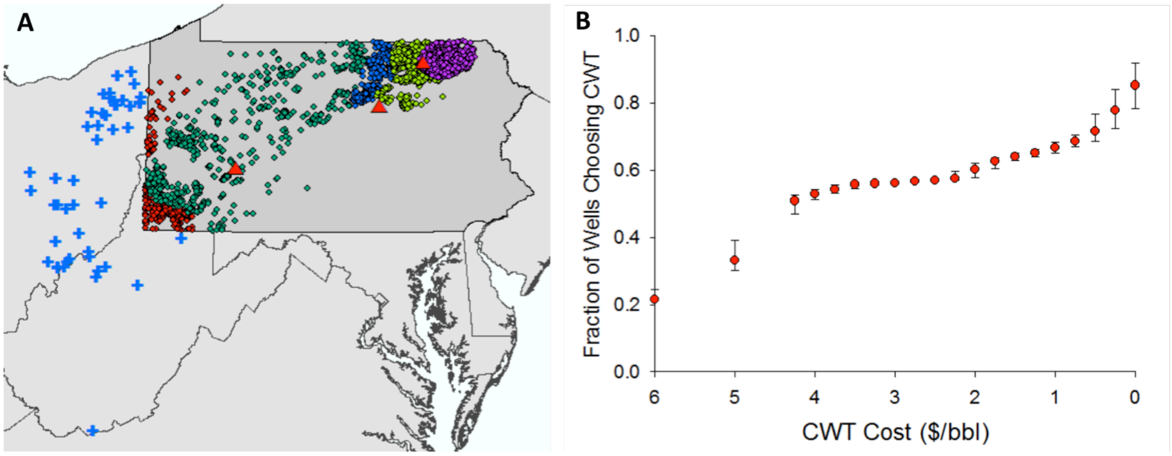


Figure 23: A) Wells where CWT would cost less than disposal for a CWT cost of $6/bbl (purple), $5/bbl (light green), $4.25/bbl (dark blue), or $0/bbl (green) given disposal cost of $0.76/bbl, and B) fraction of wells choosing treatment as a function of treatment cost.

Even when trucked to CWTs or processed using specialized on-site treatment trains, Marcellus wastewater is particularly challenging to treat to surface discharge MCL standards. Furthermore, even undetectable quantities of constituents can lead to public health threats later in the public water supply chain such as disinfection byproducts[140, 217]. Finally, even if the wastewater can be completely treated prior to discharge, there is evidence showing the public would still be averse to using it as potable water post treatment[218]. Because of the social perception, economic barriers, and risk/liability challenges associated with treating produced water, a cost effective compromise may be to dispose of the Marcellus water in injection wells where that is the least cost option for producers, and invest the money saved into other more viable water quality remediation measures critical to the health of PA's ecosystems.

Using this expanded produced water tool, I can extrapolate to estimate the investment in wastewater management throughout the lifetime of the play. To determine the net present value of the total treatment costs, I assume the distribution of well spud dates follow a gamma distribution, which describes a scenario where peak drilling will occur between 2015 and 2025 and then drilling tapers off after that. As there are currently no detailed production estimates in the literature, I conducted sensitivity analysis to test the robustness of this assumed distribution.

With an estimated 36 kbbls of non-flowback produced water generated per well (90% CI: 13-70 kbbls), and 60,000 future wells projected by TNC, the total value of the cost of disposal of over 2 M bbls of produced water (excluding time value of money) would be $13B (90% CI: $3B - $31B). The cost of treatment is $17B at CWTs (90% CI: $5.7B-$34B) and $18B (90% CI: $6.3 - $36B) on-site. By subtracting the totals, I calculate that on average the least cost treatment wastewater management option (CWT) costs $4.3B more that deep well injection disposal.

While wastewater re-use is consistently the least cost option, all produced water, whether recycled or not, must eventually be either treated or sent to disposal. Prior to 2011, Pennsylvania producers transferred this water to publicly owned treatment works (POTWs) or centralized waste treatment facilities (CWTs), neither of which had process trains designed to remove the high level of TDS. As a result, a significant volume of high salinity water was released into surface waters, which can lead to pipe corrosion, decreased efficiency of industrial boilers and heat exchangers, taste and odor problems in drinking waters, and can be toxic to aquatic ecosystems[15, 219]. Furthermore, the release of produced water-containing bromide (a component of TDS) can increase the rate of carcinogenic disinfection by-product (DPB) formation in downstream drinking water plants[15]. To address these public health, ecosystem, and infrastructure concerns, in 2010 the PA DEP adopted PA Code 95.10, which established effluent contaminant limitations for new and expanding treatment plants planning to discharge processed Marcellus produced water[183, 220]. These limitations include a monthly average limit of 500 mg/L of TDS. As a result of this code, CWTs can either obtain a WMGR123 permit, which enables them to handle the water and either return it to the production company for reuse or dispose if it appropriately. Despite any treatment conducted under this permit, the produced water maintains its status as residual waste. Alternatively, CWTs can apply for NPDES or WQM. NPDES permits regulate the discharge of produced waters into surface waters under the Clean Water Act. Under this permit, twenty parameters are regulated and monitored[192]. WQM permits allow disposal of treated effluent into groundwater or municipal sewer systems for further treatment. While this is a dramatic change for produced water treatment facilities, even the NPDES surface water discharge standards are still not as stringent as the quality targets necessary to “de-list” the water such that it would no longer be considered residual waste under the WMGR123 permit[192].

In order to reduce the TDS from over 100,000 ppm to 500 ppm, extensive treatment is required. While produced water from oil and gas operations in other regions with low TDS concentrations can be treated using evaporation ponds or reverse osmosis, Marcellus produced water treatment requires a treatment train consisting of crystallization and evaporation, followed by either distillation or reverse osmosis to remove residual organics and inorganics[183, 191, 192]. These tertiary desalination steps are in addition to the traditional produced water treatment train processes including pretreatment to remove solid materials, primary filtration to remove fine particulate matter, and ion-exchange process to remove hardness[182]. However, even with waste treatment facilities that have been specifically designed to handle produced water from shale gas development, radiological components, chemicals, and toxins have been released and later detected in freshwater sources[202].

As of January 2016, there are two CWTs in Pennsylvania with NPDES permits actively reporting effluent surface water discharge through the PA DEP's electronic discharge monitoring reports (eDMR) database[221]. These facilities include Eureka's Standing Stone facility (NPDES permit PA023235) and the Pennsylvania Brine Treatment – Josephine Facility (PA0095273). In 2014, according to the PA DEP waste production report data[222], approximately 6.8 Million gallons were sent to the Standing Stone facility, but the CWT only reportedly discharged approximately 3.2 M gallons[221]. This demonstrates the unwillingness of companies to pay for full produced water treatment. In the second half of 2015, however, when the volume of recyclable produced water outpaced the demand for new wells, ~10.4 M gallons of produced water was sent to the Standing Stone facility and essentially all of that water (10.3 M gallons) was reportedly treated and discharged. However, the total volume treated and discharged during this six-month period was 10.3 M gallons from Standing Stone (based on monthly average values) and 27 M gallons from Josephine Facility (based on monthly average values of the daily maximum effluent, resulting in upper bounds estimate). This accounts for only 5% of the total (~760 M gallons) of flowback and produced water reported by natural gas producers. Therefore, even at a time when there is a backlog of produced water, this demonstrated producers choose other, less costly produced water management options.

As demonstrated by the above discussion, the primary barriers to adopting produced water treatment are the cost of treatment and liability issues associated with the release of residual, undetected contaminants[194]. Although it has been estimated that that the produced water management market is worth more than $5 billion annually, and the treatment sector specifically is worth about $2.5 billion annually[194], few innovations have reached the market. This is in part due to the fact that water technology development requires time consuming and expensive field trials[88]. Because of the wide variation in produced water quality both spatially and temporally, a technology may work well in one region but fail to meet water quality standards in other regions. However, beyond the technical challenges, innovation has been hindered in large part due to regulatory uncertainty at both state and federal levels, and even variation across jurisdiction within states (i.e. state versus federally owned land). Another important source of regulatory uncertainty stems from the classification of oil and gas produced water, which determines treatment requirements[194].

There are currently two primary focuses of produced water treatment innovation. The first is considering the economics of mobile versus fixed treatment solutions. In some regions due to the geographically dispersed nature of production, it may be more economic to have mobile units. However, centralized treatment hubs are likely better able to handle variability in produced water quality and volumes by blending produced water from several producers[194]. The second focus of produced water treatment innovation is on sellable by-products. For example, Eureka Resource's business model suggests surface water discharge can only compete economically if they can extract products from the produced water. These by-products include 1. methanol, which can be sold as a fuel for oil and gas operations such as powering compressor stations, 2. salt, which can be used for de-icing and potentially as a water softening agent, and 3. calcium chloride brine, which can be used as an input in oil and gas drilling fluids[191]. However, while treatment opportunities may be increased by expanding by-product use outside of the oil and gas industry, this is limited by the lack of consistency in the quality and quantity of the water and by public perception[194].

Since the large-scale treatment and surface discharge of produced water is not currently economically competitive for all wells throughout the state, lacks the necessary infrastructure to accommodate produced water volumes, and may still result in environmental damages despite the extensive level of treatment, a viable alternative is to dispose of the produced water in Class II injection disposal wells. Class II injection wells are regulated by the EPA under the Safe Drinking Water Act[93]. Deep well injection of produced water has historically been the most common wastewater management method in the United States; in 2007, more than 98% of produced water from onshore wells was injected either in secondary recovery wells to increase the output of production wells (60%) or into nonproducing formations (40%)[14]. In 2012, the percentage of onshore produced water disposed of in wells decreased to 93%[87]. As evidence of its historical relevance to the oil and gas industry, traditional oil and gas producing states have many thousands of Class II wells. For example, Texas, California, Kansas, Oklahoma, and Wyoming have 52,000, 30,000, 16,700, 10,600, and 5,000 disposal wells, respectively[93].

Pennsylvania, in contrast, has around 1860 wells, and the vast majority of these are "Class IIR" wells for enhanced oil recovery, whereas only 11 wells are "Class IID" wells for brine disposal[223]. While it is often reported that Pennsylvania's geology does not support the development of Class IID wells, the large numbers of Class IIR wells demonstrates this is not the case, since all Class II wells have the same permitting requirements. Although the reservoirs might not be able to accommodate the disposal rates and pressures of those in other geologies, there are in fact opportunities to drill disposal wells. However, Class IID injection wells may cost millions of dollars to develop. In Pennsylvania, as previously discussed, producers were until recently able to very inexpensively dispose of their water at POTWs. Even after regulation eliminated this disposal option, reuse of produced water for hydraulic fracturing of new wells became a low cost management option. Furthermore, Ohio and West Virginia have many Class II disposal wells available for Pennsylvania producers. While trucking the water to these disposal wells is costly, it has traditionally been more economically favorable than treating the water at NPDES permitted CWTs or constructing a more conveniently located Class II injection well. In other words, the limited number of disposal wells in Pennsylvania demonstrates the lack of economic incentive to develop Class II wells as a produced water management option[223]. In fact, in 2012 there were only six active disposal wells in the state compared to the eleven currently active[93], and the EPA reports continued interest in new Class IID well development in Pennsylvania[223].

3.4.3 Risks Associated with Wastewater Management

Each produced water management option has risks associated with it[224]. Disposal via deep well injection in Class IID disposal wells is no exception. Historical evidence from regions with extensive oil and gas operations, such as the Western Canada Sedimentary Basin, suggest injection disposal can be a stable, low risk strategy[225]. However, in the United States there have been questions as to whether disposal via Class II injection wells may be responsible for induced seismic activity[197, 199, 226-229]. In particular, the central and eastern U.S. has experienced an increase in earthquake rates since 2009[229]. The largest of these was a magnitude 5.6 earthquake in central Oklahoma that destroyed 14 houses and injured two people[202]. The US EPA[230], USGS[227], National Research Council[226], and Congressional Research Service[231] have each developed reports to scope the risk and address these growing concerns. The consensus from these studies is that of the over 30,000 class II disposal wells in the U.S., very few can be associated with having produced seismic events with magnitudes greater than 4.0[226, 230]. However, the USGS suggests places in Oklahoma, Kansas, Colorado, New Mexico, Texas, and Arkansas may experience damage if the induced seismicity continues unabated[232]. When compared to injection of water for enhanced oil recovery, Class IID disposal wells do have a higher likelihood of inducing seismic events because the overall reservoir pressure increases, and is not relieved through the extraction of oil. In contrast, when compared to injection wells for carbon capture and sequestration (CCS), the likelihood of induced seismicity is lower, because CCS requires injection of large volumes of liquid over an extended period of time. While the EPA does not specifically address seismicity with specific regulatory requirements during the permitting process[230], most wastewater disposal wells typically involve injection at relatively low pressures into large porous aquifers that have high natural permeability and are specifically targeted to accommodate large volumes of fluid[226]. Furthermore, research suggests that monitoring injection rates and pressures can manage the risk of induced seismicity from wastewater disposal[186, 197, 227]. As an example of how this information can inform responsible wastewater disposal regulation, the Oklahoma Corporation Commission (OCC) has implemented a plan that will affect almost 600 wells in western and central Oklahoma. The objective of the OCC’s plan is to reduce injection volumes to 40% below the 2014 levels, which is a reduction of more than 300,000 bbls per day from the 2015 average injection volumes in an effort to address induced seismic activity concerns[233]. While it is challenging to attribute induced seismic activity to wastewater disposal, this is the first large scale regulatory experiment specifically targeted at managing geologic impacts from injection disposal wells. At a smaller scale, Ohio has also been proactively addressing seismic concerns from the underground injection program through Executive Order 2012-09K, which enables regulators to require additional tests or evaluations of brine disposal wells if deemed necessary[234]. These regulatory actions were a direct result of seismic activity occurring in Youngstown, Ohio where a series of over 100 small magnitude earthquakes (0.4-3.9M) occurred over a 14-month period where there were no known earthquakes in the past. These earthquakes were believed to be induced by the Northstar 1 disposal well[230] which had been drilled to a depth within the Precambrian basement rocks, and the earthquakes ceased when disposal activities were put on hold and the pressure dropped[199]

3.4.4 Alternative Wastewater Sources and Treatment Options

While traditionally, environmental impact from the natural gas, coal mining, and agricultural industries are monitored and regulated independently, I propose that a systems level perspective that merges these issues might generate more abatement, less overall pollution and can result in a more cost effective means of enhancing Pennsylvania's water conservation efforts and mitigating drinking water public health concerns.

To put the costs of Marcellus produced water treatment into context, it is important to understand the opportunity cost it poses by considering cost of treatment for other critical water quality threats in Pennsylvania. As a basis for such a comparison, I focus on two sectors that have historically negatively impacted Pennsylvania's water quality; coal mine drainage and agricultural runoff. The PA DEP estimates 200 million gallons of polluted water from coal mines flow into the state's surface waters each day[212], resulting in 4,000 miles of biologically dead rivers and streams[235]. Similarly, the agricultural industry pollutes Pennsylvania's surface waters when nitrogen and phosphorous from excess fertilizer runs off into rivers and streams, causing algae blooms that block oxygen, disrupting the aquatic ecosystem.

The first important surface water quality threat evaluated in this study is contamination from coal mine drainage (CMD). Coal mine drainage, also commonly referred to as acid mine drainage, is a toxic mixture of acid and dissolved metals. When it mixes with surface or ground waters, it destroys ecosystems, is toxic to aquatic organisms, corrodes infrastructure, and taints drinking water[236]. In 2015, the U.S. Geological Survey (USGS) reported CMD was the most extensive water pollution problem affecting the state[228].

CMD management options include either "source control" or "migration control"[226]. Source controls are techniques used to preclude the formation of CMD. However, this is often expensive and impractical. Therefore, in this analysis I focus on migration control, or remediation. Migration control can be further subdivided into passive and active treatment, and biological and abiotic treatment[226]. However, case studies have suggested that active, abiotic treatment systems are the most consistently effective treatment option[237]. Specifically, lime neutralization/hydrolysis is the most established and widely practiced treatment technology[183].

Based on a lime neutralization active treatment process, the average cost is $0.046/bbl (90% CI: $0.01 - $0.12/bbl). The cost primarily depends on the water's acidity, with more acidic water having a significantly higher treatment cost. Given the DEP estimate of 300 million GPD of CMD contamination, for the difference in cost of disposing of Marcellus wastewater rather versus treating it, Pennsylvania could pay to treat about 70 years worth of all CMD in the state.

The second critical source of water contamination in Pennsylvania is agricultural runoff. Non-point source pollution from agricultural runoff is a leading cause of hypoxic marine dead zones worldwide[203]. Much of the key productive regions of the Marcellus shale lie within the Susquehanna river basin, which is a part of the Chesapeake Bay watershed. Agriculture is the single largest source of nutrient emissions in the watershed; in 2010, only 18% of tidal waters met or exceeded guidelines for water clarity, only 38% of the Bay and its tidal tributaries met Clean Water Act standards for dissolved oxygen, and more than half of the stream health scores at monitoring sites were poor[238]. Federal and State initiatives have been targeted at improving the watershed quality since 1976, when congress directed the EPA to undertake a comprehensive study of the Bay's condition and measures by which it could be improved to its previous condition[239], however agricultural runoff remains an important environmental issue.

One best management practice (BMP) to address agricultural runoff is the establishment of vegetative buffer strips (VBS). VBS are grass or forested land planted to intercept runoff prior to it contaminating the surface water. The nitrate is mitigated through denitrification and nutrient uptake by the plants in the buffer region[240]. In the Chesapeake Bay watershed, VBS typically reduce 50-80% of the nutrients depending on the topography, hydrology, and climate of the specific location[192, 241].

Using the passive treatment technique of grass or forest vegetative buffer strip development, the average cost of agricultural runoff treatment is \$0.08/bbl (90% CI: \$0.05-\$0.11/bbl). This assumes an active buffer lifetime of 20-30 years, however it is likely the buffer would continue to be effective as long as it remains in place, thus this is an upper bound cost estimate. This is also likely an upper bound because it is based solely on rainwater runoff volumes from flat acreage; it does not consider subsurface leaching or runoff rates from steeper topology, which would both increase the water per acre, thereby decreasing the per bbl treatment cost.

In a recent study of agricultural pollutants in the Chesapeake Bay, the USGS classified cropland based on vulnerability, treatment need, and “enhanced targeting factors”. Vulnerability of fields to runoff is based on soil properties such as hydrologic group, slope, and erodability. Treatment need is based on whether other BMPs are being implemented on the cropland, such as limiting nitrogen fertilizer use. Finally, the “enhanced targeting” classification designates cropland that is spatially relevant to runoff risk. In the Susquahanna river basin, which covers a large portion of central PA, there are 655,000 acres of cropland with high runoff potential, 560,000 acres of critically under treated cropland, and 324,000 acres that are adjacent to surface waters. If the same investment made to treat Marcellus produced water rather than dispose of it was instead made to reduce agricultural runoff, the Susquahanna river basin could treat the equivalent of either 360% of its vulnerable (high runoff potential) acreage, 420% of all critically undertreated acreage, or 720% of the acreage adjacent to surface waters. It might also be considered that the Chesapeake Bay has a cost share assistance program in which approximately 75% of capital investment required to develop a VBS is subsidized. If this subsidy were to be factored in to the cost/bbl of treatment, the effective treatment cost would be \$0.02/bbl. This would increase the number of treated acreage to approximately 2.1 million. Additionally, beyond the benefits of reduced nitrogen, phosphorous, and sediment load in the Chesapeake Bay, developing buffer zones also has indirect value. For example, a study by Costanza et al. (2006) found a mean ecosystem services value of $3,850 per acre, derived from enhanced recreational services, aesthetics, and disturbance protection such as reduced risk of flooding[180]. Including the ancillary benefits of enhanced ecosystem services in these calculations would serve to further increase the cost-effectiveness of investing in VBS over produced water treatment, which would not promote such ecosystem services.

3.4.5 Policy Implications

Wastewater management is a complex process that reflects tradeoffs in economic objectives, system reliability, risk, liability, infrastructure availability, and technological capability of meeting treatment and disposal water quality targets. As previously outlined, the primary wastewater management options available to natural gas producers in Pennsylvania are to wastewater reuse for subsequent well development, dispose in underground injection wells, or to wastewater treatment to surface discharge quality via onsite treatment modules or centralized waste treatment facilities (CWTs). There have been several publications identifying decision-making strategies for well developers[185] including integrated multi-criteria decision making tools, The Produced Water Management System (PWMS) developed for the DOE[186], the Water Decision Tree developed by the Petroleum Technology Alliance Canada[187], and multiple optimization models[141, 188, 189]. However, these models represent strategies best suited to decision making for a single developer or group of spatially clustered developers. Additionally, when considering well development on a small scale, resources such as fresh water and disposal capacity may be functionally constrained. This would not necessarily represent the real time circumstances of the ongoing industry operations, especially within the Marcellus region where water resources are not limited. Thus, these tools may not be well suited to inform policy targeted at large scale, statewide water quality conservation and wastewater management regulation.

To successfully develop approaches that maximize the potential to positively impact on the environment while meeting the needs of the PA unconventional gas producers, it is important to consider the wastewater management issue from an industry wide perspective rather than from the standpoint of individual producers/decision-makers. Therefore, I have developed a tool that estimates the tradeoffs in economic wastewater management preferences of developers across the Marcellus region of Pennsylvania. I use those insights, combined with qualitative risk assessment to identify tradeoffs in the potential wastewater management pathways. I then compare these tradeoffs at the systems-level analysis to demonstrate how strategic wastewater management regulation in the natural gas industry designed to have maximum positive impact on water quality can contribute towards an overall risk reduction of contamination across all of Pennsylvania's surface water resources. This type of systems level analysis can lead to a cost-effective wastewater management policy that serves as a compromise across multiple stakeholders that otherwise often have competing objectives.

Wastewater from Marcellus natural gas production represents a nominal fraction of total industrial and public wastewater generated in Pennsylvania. However, due to the high level of contaminants in the water, it is important to develop a comprehensive strategy to address management of produced water throughout its life cycle, from generation through treatment or disposal. In the initial four years of Marcellus operations, producers were able to unrestrictedly dispose of their water at publically owned treatment works and other centralized waste treatment facilities at low cost, without TDS treatment. Even after these practices were restricted in 2011, producers were generally able to recycle a large portion of their wastewater for the hydraulic fracturing of subsequent wells. Both of these inexpensive disposal options delayed the development of environmentally sound wastewater treatment pathways. However, once the generation of produced water outpaces the production of new wells, there will be a backlog of contaminated water and no comprehensive management plan. This has already been observed in the Marcellus region where low natural gas prices have slowed the pace of new well development and left producers with large volumes of wastewater to address.

For Pennsylvania regulators, deciding whether to invest in Marcellus specific desalination technologies at a large scale to incentivize treatment or whether to rely on the historically preferred disposal method via Class II injection wells is an important next step in ensuring Marcellus Shale production proceeds in an environmentally responsible manner. This regulatory strategy should be based on a comparison of risks and economic feasibility. The work presented in this paper serves to inform this decision by first outlining environmental risks of incomplete treatment, feasibility of treatment technology, treatment capacity constraints, and risk of induced seismic activity from injection disposal. If it is demonstrated by the reduced injection rate regulation in Oklahoma, as well as by other seismic monitoring and reporting measures being taken across the country that induced seismicity can in fact be mitigated through moderated disposal rates, volumes, and pressures, from a risk standpoint disposal via injection wells may be the environmentally safer option than treatment and surface water discharge. As previously discussed, even facilities specifically designed to treat Marcellus wastewater may result in contamination of freshwater sources with toxic chemicals, bromides, and radioactive material which each pose a threat to aquatic ecosystems, infrastructure, and public health[99, 242, 243]. Furthermore, recent evidence suggests Marcellus wastewater may be more highly concentrated with bromine, chlorine, and NORMs than previously believed[244] making complete treatment even more challenging. Because of the potential for freshwater contamination post-treatment, regulatory liability issues could remain a barrier for large-scale adoption of treatment practices, should they become widely commercially available.

From a feasibility perspective, it is important to consider the capacities of the produced water management options to appreciate the capital investment required to develop the necessary infrastructure. Eureka Standing Stone facility is an example of a Marcellus wastewater treatment plant with surface water discharge ability, capable of treating up to 5,000 bbls per day[245]. Veil (2015) estimated Pennsylvania's oil and gas operations generated approximately 34 M bbls of wastewater (including both flowback and produced wastewater). Since produced water volumes are high immediately after hydraulic fracturing occurs and taper off throughout the lifetime of the well, this volume is likely to increase as Marcellus production of new wells increases and decrease in the future as new well production declines. However, using 2012 wastewater volume as a baseline, at a treatment rate of 5,000 bbls per day, Pennsylvania would need 19 CWT facilities of this size and technology to address the magnitude of wastewater generated annually throughout the play. This assumes that storage options are aligned such that capacity of each treatment facility is fully utilized each day. Due to the geographic clustering of Marcellus wells in the southwest and northern regions of the state, it is likely that many of these facilities, if constructed, would be co-located. While there are no known instances of surface water contamination from these CWTs currently discharging via NPDES permits, even contaminants under the MCLs could bioaccumulate. This co-location of CWTs would increase the environmental risk of such accumulation of contaminants in the surface waters, as multiple facilities would be discharging effluent to the same rivers and streams.

For comparison, Ohio currently has 215 active injection wells[246]. Given the total 2012 wastewater estimate of 34.1M bbls, each of these wells would only need to accept 450 bbls/day of wastewater for disposal. This is well within the typical Class II injection disposal rates and capacities. Additionally, more wells are currently being drilled or are in the permitting stage throughout Ohio and Pennsylvania[223, 246], thus capacity expansion continues to be underway. As a result, the receiving volume per well would continue to decrease. From an economic perspective, as more disposal wells become available, the price of disposal may also decrease due to the decreased competition for disposal capacity.

The second portion of this work assesses the economics of complete wastewater treatment versus disposal in Class II injection wells, assuming complete treatment is available at a large scale with no significant capacity constraints. I included both on-site treatment and centralized treatment as options, although given the complexities of varying input water rates and qualities, it is likely on-site treatment would not be technically feasible after the initial flowback period. Additionally, because costs reported for centralized waste treatment may have been misinterpreted in the literature as costs for complete treatment rather than non-TDS treatment for recycle, and because on-site treatment is not readily available on a commercial scale, I believe the costs presented here to underestimate the true cost of treatment.

Given current cost estimates, I determine treating Marcellus wastewater would cost an average of $80,000 per well more than disposing of it. Given The Nature Conservancy's estimate of 60,000 wells throughout the lifetime of the play[153], this additional cost of treatment could total $4.8B (without accounting for time value of money). While it may be determined by regulators that treating the water to return it to the hydrologic cycle merits such an investment, or that the risk of induced seismicity from disposal wells outweighs the risk of environmental impact of potential surface water contamination from incomplete treatment, it is important to put this investment in wastewater treatment into context. One means of doing this is to compare this investment to the cost of treatment for other key sources of water contamination in the state. If the ultimate goal is to protect freshwater resources, minimize public health impacts, and prevent ecosystem disruption, Pennsylvania should invest in the most cost effective wastewater to treat first. Two such important water quality concerns in Pennsylvania are coal mine drainage and agricultural runoff.

Coordination of resources and pollution abatement measures across the oil and gas, coal mining, and agricultural sectors might serve to better address the water quality concerns throughout the state. Because they are largely in the same watershed, on a regional scale they address the concerns of the same environmentalist and social stakeholders. I found that CMD water and agricultural runoff could be treated for $0.05/bbl, and $0.08/bbl, respectively using best management practices.

In order to avoid being regulated to treat their wastewater, oil and gas companies should have an average willingness to pay (WTP) of up to $2/bbl (the average differential between CWT and disposal costs, $7.80/bbl and $5.80/bbl, respectively). If a portion of this WTP could be captured through a regulatory measure such as a tax or bond and reallocated towards treating CMD or agricultural runoff, significant gains in water quality could be achieved throughout the state while simultaneously avoiding both the environmental risk of additional contamination to freshwater resources and the investment in costly, energy intensive centralized waste treatment infrastructure. This proposed reallocation of economic resources for water treatment is an example of how a systems-level approach to techno-economic policy issues can address multiple stakeholder concerns and provide for cost-effective regulatory solutions.

Chapter 4. Life Cycle Greenhouse Gas Emissions From U.S. Liquefied Natural Gas Exports: Implications for End Uses[2]

4.1 Abstract

This study analyzes how incremental U.S. liquefied natural gas (LNG) exports affect global greenhouse gas (GHG) emissions. Emissions of LNG exported from U.S. ports to Asian and European markets account for only 3.5-5.5% of pre-combustion life cycle emissions, hence shipping distance is not a major driver of GHGs. This study finds exported U.S. LNG has mean pre-combustion emissions of 37g CO_2-equiv/MJ when regasified in Europe and Asia. A scenario-based analysis addressing how potential end uses (electricity and industrial heating) and displacement of existing fuels (coal and Russian natural gas) affect GHG emissions shows the mean emissions for electricity generation using U.S. exported LNG were 655 g CO_2-equiv/kWh (with a 90% confidence interval of 562-770), an 11% increase over U.S. natural gas electricity generation. Mean emissions from industrial heating were 104 g CO_2-equiv/MJ (90% CI: 87-123). By displacing coal, LNG saves 550 g CO_2-equiv per kWh of electricity and 20 g per MJ of heat. LNG saves GHGs under upstream fugitive emissions rates up to 9% and 5% for electricity and heating, respectively. GHG reductions were found if Russian pipeline natural gas was displaced for electricity and heating use regardless of GWP, as long as U.S. fugitive emission rates remain below the estimated 5-7% rate of Russian gas. However, from a country specific carbon accounting perspective, there is an imbalance in accrued social costs and benefits. Assuming a mean social cost of carbon of $49/metric ton, mean global savings from U.S. LNG displacement of coal for electricity generation are $1.50 per thousand cubic feet (Mcf) of gaseous natural gas exported as LNG ($.027/kWh). Conversely, the U.S. carbon cost of exporting the LNG is $1.80/Mcf ($.013/kWh), or $0.50-$5.50/Mcf across the range of potential discount rates. This spatial shift in embodied carbon emissions is important to consider in national interest estimates for LNG exports.

---

[2] The final version of this chapter is available as Abrahams, Leslie S., et al. "Life Cycle Greenhouse Gas Emissions From US Liquefied Natural Gas Exports: Implications for End Uses." *Environmental science & technology* 49.5 (2015): 3237-3245.

4.2 Introduction

United States natural gas production is projected to reach 36 trillion cubic feet (Tcf) by 2040, representing an increase of 52% from 2012[31]. This abundance of domestic energy resources is seen by various stakeholders as a geopolitical advantage, an economic opportunity, and a pathway for increased environmental damages. Although use of natural gas results in greenhouse gas (GHG) emissions, life cycle emissions from the natural gas electricity generation supply are estimated to generally be lower than those from coal electricity generation[1,2]. Natural gas has often been discussed as a 'bridge fuel' that could be used as a coal replacement until the transition to large-scale reliable renewable energy sources. The viability of natural gas as a bridge fuel depends on assumptions about stabilization objectives, emissions, and technological change[3-6].

Prior to 2008, U.S. domestic natural gas production did not meet projected demand growth, and the national natural gas debate centered on LNG imports. As unconventional natural gas production (shale gas) became economically viable, U.S. technically recoverable natural gas reserves increased by 665 Tcf, which represents an increase in total U.S. natural gas resources by 38%[17]. As domestic natural gas production increased, supply flooded the market and wellhead prices in today's dollars in the U.S. dropped from close to $9/Mcf in 2008 to under $3/Mcf in 2012[18]. Because of higher natural gas prices in other regions, producers are hoping to sell incremental quantities of natural gas to economically attractive Asian and European markets[19]. This has prompted a debate in the United States on the policy of liquefied natural gas (LNG) exports regarding whether additional LNG exports to non-FTA countries should be approved, and if so to what capacity.

Globally, the LNG trade reached about 12 Tcf of natural gas in 2012[34], and based on the projects world-wide currently under construction or proposed, liquefaction capacity could increase by more than 100% within the next ten years[35]. The primary LNG importers are expected to be the rapidly developing countries in Asia and some European countries such as the UK, Spain, and France seeking to both supplement and diversify their natural gas supply[35].

The Natural Gas Act, as amended, requires the U.S. Department of Energy (DOE) to determine if LNG exports are in the "national interest" in order for approval. Additionally, the current regulatory framework requires exports to countries with which the U.S. has a free trade agreement (FTA) to be rapidly permitted, while applications to export to non-FTA countries undergo additional assessments[36]. The national interest determination involves consideration of economic, international, and environmental factors. To date, the DOE has approved thirty seven applications to FTA countries totaling 14 Tcf of natural gas exports annually, and has issued nine final and conditional approvals of applications to non-FTA countries, totaling almost 3.8 Tcf/year[36].

Historically, the DOE granted conditional approval prior to a full National Environmental Policy Act (NEPA) review. However, a regulatory change (effective August 2014) now requires LNG export applications from the lower 48 states to non-FTA countries pass the NEPA review process prior to the issuance of any export permits[36]. It is likely that this recent streamlining of the approval process of LNG export projects will lead to an increase in export capacity in the near future.

In light of the recent discussions regarding the non-FTA export approval process, the DOE released studies focusing on the upstream environmental impact of increased natural gas production[184] and the global emissions impact of increased LNG exports[247]. While the Federal Energy Regulatory Commission (FERC), the lead agency for environmental review of non-FTA export applications, only formally requires a report on direct, localized environmental impacts resulting from the construction and operation of the export facility[36], the DOE authorized these analyses as part of a broader effort to inform LNG export decisions[248]. Although the latter study models life cycle emissions of LNG exports, its specificity to the electricity sector limits its robustness as an analysis of net global changes in GHG emissions resulting from U.S. exports.

While the electric power sector accounts for a large portion of natural gas end use, industrial natural gas use generally accounts for a similar portion of natural gas end use. In these data, the industrial sector includes natural gas use for activities including processing and assembly, space conditioning, lighting, and as feedstocks for the production of non-energy products such as plastics and fertilizer. Additional descriptions of the sectors are available in the EIA's International Energy Outlook (2013)[249].

In general, in Table 8, the electric power and the industrial sectors are responsible for an approximately equivalent share of natural gas end use. This is especially true on a regional bases for OECD Europe, globally, and within the U.S. This supports the fact that the GHG emissions from heating should be considered in addition to the GHG emissions derived from natural gas combustion for electricity generation. One exception to the approximate equal share of end use between electric power and industrial sectors is China, where the industrial sector accounts for 46% of the end use while the electric power sector only accounts for 15%. This is partially due to the fact that China produces a large quantity of plastics and chemicals that use natural gas as a feedstock. On the other hand, in Japan and South Korea, the electric power sector's natural gas consumption far outweighs industrial natural gas consumption.

Table 8: Natural gas consumption by sector, total energy consumption, and LNG imports for relevant countries and regions of interest for this study[29-31.]

Country/Region	**Consumption by Sector in 2012 (%)[249]**					**Total Natural Gas Energy Consumption[249]**	**LNG imports[34]**
	Residential	Commercial	Industrial	Transportation	Electric Power Sector	(Quadrillion Btu)	Tcf
China	23.9	6.5	45.7	6.5	15.2	4.6	0.7
India	-	-	52.2	8.7	43.5	2.3	0.7
Japan	7.5	15.1	11.3	-	66.0	5.3	4.2
South Korea	23.5	11.8	17.6	-	47.1	1.7	1.8
OECD Europe	26.9	10.4	31.3	0.5	30.8	20.1	2.3*
U.S.	17.2	11.5	32.4	3.1	32.4	26.2	0.2
World	16.2	6.7	38.5	3.2	35.5	120.4	11.6

* OECD LNG import data obtained from the IEA gas medium-term market report[250]

In addition to the Skone et al. (2014) analysis, previous work has been done to quantify the GHG emissions from the LNG life cycle for electricity generation[1, 251-253], as a shipping fuel[254], and as a transportation fuel[255]. This study first expands upon the previous attributional life cycle analyses by considering additional uncertainties in the LNG life cycle such as GWP, fugitive emissions rate, percent methane of natural gas, shipping distances, and liquefaction emissions.

This study then seeks to further inform decisions about the potential environmental impact of LNG exports through a scenario-based first order consequential analysis in which the net change in GHG emissions resulting from natural gas displacement of coal or Russian natural gas is calculated on a per kWh basis. Finally, although all net savings in emissions benefits the global environment, this study considers tradeoffs in where along the supply chain the emissions occur from a country specific carbon accounting perspective. These tradeoffs are monetized using the emerging regulatory analysis metric, the social cost of carbon (SCC), as used by the U.S. federal government[256] in order to quantify country specific carbon accountability that would be relevant in the potential future scenario of an embodied carbon tax on exports. The social cost of carbon is the estimated global damages caused by an additional metric ton of CO_2 released into the atmosphere. Some of the monetized potential climate change damages considered in the SCC calculation include the impact of climate change on agriculture, water resources, air quality, human health, ecosystem services, and property damage from increased flood risk[257]. Recent work suggests the SCC as applied to CO_2-equivalent emissions could be an underestimate of the social cost of methane[258].

4.3 Methods

This work builds upon estimates from previous studies of upstream natural gas emissions[214,259] and LNG supply chain emissions[252,253] in order to develop an attributional life cycle assessment (LCA) of the GHG emissions of LNG exports from the United States. The assembled data was compiled into a Monte Carlo simulation to account for uncertainty in emissions from each stage in the LNG export life cycle. The results of the LCA simulation were then incorporated into a first order consequential life cycle assessment considering the impact of U.S. LNG exports displacing coal and Russian natural gas for electricity generation and industrial heating in Asia and Europe.

In this study, a “first order consequential” analysis is defined as a scenario-based comparison that serves to illustrate potential relevant impacts of a policy based on a qualitative description of conceivable market responses to the policy. For example, in this analysis, the policy being considered is an increase in U.S. LNG exports. Some potential market responses to this policy discussed in this analysis include decreased coal use for electricity and/or heat generation, decreased Russian natural gas use for electricity and/or heat generation, increased domestic natural gas production, and a shift towards increased domestic coal use. While a first order consequential analysis does not seek to quantify the degree to which these responses to a policy occur, the scenarios serve as a bounding analysis that can inform decision makers of non-intuitive potential market based consequences of a policy.

4.3.1 Global warming potential

For the upstream production and shipping stages of this LCA, both 100 and 20-year global warming potentials (GWP) for methane (fossil methane with climate carbon feedbacks) from the IPCC fifth assessment report (AR5) were used[42]. Uncertainty for both GWPs was quantified using a normal distribution based on the reported mean and 90% confidence interval. The distribution for the 100-year GWP has a mean of 36 and a standard deviation of 8.5, and the 20-year GWP has a mean of 87 and a standard deviation of 16. Given the wide range of uncertainty presented in AR5 and the continuing trend of increasing the GWP estimates with each assessment report, it is important to represent the complete distribution of possible values when simulating emissions. For example, with uncertainty it is possible that the 100-year GWP of methane could be double the AR4 estimate of 25. The exception to this use of the AR5 distribution in this model is for the liquefaction and regasification life cycle stages.

The GWP used to estimate the liquefaction and regasification emissions is the exception to the use of the AR5 distribution to quantify upstream and shipping CO_2–equivalent emissions. In the literature, these estimates have generally been reported as aggregate CO_2-equivalent values based on AR3, AR4, or AR5 GWPs, rather than as disaggregated CO_2 and CH_4 emissions. Therefore, the emissions cannot readily be adjusted based on the AR5 GWP distribution. Since the GWP has increased in AR5, the use of earlier GWP in the liquefaction and regasification stages of the LCA imply that the results presented in this study are likely lower bound estimates. This is especially true with respect to the 20-year GWP results, which have 100-year GWP embedded in the liquefaction and regasification estimates. However, because the majority of emissions from liquefaction and regasifaction stages derive from fuel combustion for energy rather than from methane leakage or venting[252], it is likely that these estimates would only nominally increase based on the AR5 adjustment of the methane 100-year or 20-year GWP. To test this assumption, the ratio of methane vented per MJ of natural gas during the liquefaction stage suggested by Tamura et al. (2001)[252] (.026 g/MJ) was applied to each of the other study estimates to adjust them to the AR5 GWP distribution. This did not change the overall life cycle emissions.

4.3.2 Upstream Emissions

In this study, the upstream natural gas process included well construction, well operation, natural gas processing, and pipeline transportation to a liquefaction facility at an export terminal. Weber and Clavin (2012) assessed five previous life cycle studies and used those estimates to develop 'best guess' distributions for each of these upstream stages for both shale and conventional sources of natural gas[214]. According to the Energy Information Administration (EIA), shale gas currently accounts for 40% of U.S. natural gas production[31]. The total upstream GHG emissions were therefore calculated as a weighted average of shale and conventional production emissions estimates. The inputs to the upstream portion of the Monte Carlo simulation, as well as a description of the validation of the upstream model are shown in Table 9.

Table 9: Production and transportation emissions parameters for upstream emissions simulation[214]

Parameter	min	mode	max
Well pad construction	0.05	0.13	0.3
Well drilling	0.1	0.2	0.4
Fracturing water management	0.04	0.23	0.5
Fracturing chemicals	0.04	0.07	0.1
Conv well completion	0.01	0.12	0.41
Unconv well completion: total vent/flare (mt CH4)	13.5	177	385
Well completion: flare rate (fraction)	0.15	0.41	1
Well completion: EUR (Bcf)	0.5	2	5.3
Flaring	0	0.43	1.3
Unconv Lease/Plant energy	2	3.3	4.1
Conv. Lease plant energy	2	3.3	4.3
CO_2 vent	0.2	0.7	2.8
Compression fuel	0.2	0.38	0.6
Leak percent[58]	2	3	4

The distributions from Table 9 were then used to calculate upstream emissions from both conventional and unconventional natural gas development (Table 10). These two estimates were then weighted by the percentage of their contribution to total natural gas development as projected by the Energy Information Association (EIA) in their Annual Energy Outlook.

The upstream model of the natural gas life cycle (extraction, production, and pipeline transmission) was validated by comparing this model's emissions estimates to the harmonized emissions estimates reported by Heath et al. (2014)[259]. To obtain upstream estimates from the harmonized life cycle emissions reported by Heath et al., the harmonized combustion emissions (360 g CO2-equiv/kWh) were subtracted from the total life cycle emissions. The resulting values were then converted to MJ and multiplied by the harmonized efficiency (51%) to obtain the emissions on a heat input basis (g CO2-equiv/MJ extracted). Finally, a weighted average of shale and conventional estimates was calculated using the assumption that 40% of the natural gas extracted in the U.S. is unconventional (Table 10).

Table 10: Calculations of upstream emissions

Parameter	Type	Unit	Value (min, avg, max)		
Energy input	Assumed parameter	MJ		1	
percent methane by volume	Triangular (min, avg, max)	%	0.83	0.93	0.95
Fugitive Emissions	Calculated (5th, avg, 95th)	g CO2-e/MJ	11.9	20.7	30.72
unconventional emissions	Calculated (5th, avg, 95th)	g CO2-e/MJ	18	27	37
conventional emissions	Calculated (5th, avg, 95th)	g CO2-e/MJ	19	28.5	39.5
% of gas mix from shale	Assumed parameter	%		0.4	
Total Upstream Emissions	Calculated (5th, avg, 95th)	g CO2-e/MJ	18.5	27.9	28.4

These harmonized upstream estimates adapted from Heath et al. are all based on the AR4 GWP value for methane of 25 and maintain unique assumptions of the fugitive emissions rate. To validate my model, each leakage rate from the harmonized studies in Heath et al. was inputted into my model, using the AR4 GWP. The results of the simulation with that leakage rate is reported in Table 11 (mean and 90% confidence interval). When compared to the harmonized upstream emissions using the AR4 GWP and the unique fugitive emissions estimates from each study, results in four of the harmonized estimates are within the 90% confidence interval. The model over estimates four of the studies and underestimates one study.

There is significant debate in the literature over the fugitive emissions rate for the upstream natural gas life cycle stages. As such, in this analysis the fugitive emissions rate is presented in three ways: 1) a 'most likely' range commonly cited in literature[38, 50, 58, 62-64]. This uncertainty is represented as a triangular distribution with a minimum of 2%, maximum of 4% and most likely value of 3%, 2) a sensitivity analysis showing the effects of fugitive emissions rates across a range encompassing most values discussed in the literature[60, 61], and 3) a discussion of the break-even fugitive emissions rate that would change the result of the analysis.

Table 11: A comparison of harmonized upstream emissions estimates (adapted from Heath et al. 2014) to the results from our model with the AR4 GWP and the reported leakage rate of each study as model inputs.

Comparison Study	**Harmonized Life Cycle Emissions[259]**		**harmonized upstream***	**leakage rate**	**Results from this study's model ($g\ CO_2$-e/MJ)**		
	Shale	Conventional	($g\ CO_2$-e/MJ)	%	mean	5%	95%
Howarth[39]	746	647	46.3	2.8	20.3	18.8	22
Howarth[39]	567	473	21.3	6.2	36.6	34.5	38.5
Jiang[38]/Venkatesh[1]	497	439	14.5	2.2	17.5	16	19
Skone[64]	438	439	11.1	3.9	25.6	23.8	27.4
Hultman[63]	438	438	11.1	2.8	20.3	18.8	22
Burnham[62]	517	557	25.6	2	16.5	15.1	18
Stephenson[260]	434	420	9.3	0.66	10	8.7	11.6
Heath[261]	459	450	13.3	1.3	13	11.8	14.8
Laurenzi[262]	470	450	13.9	1.4	13.6	12.2	15.2
This study	-	-	-	triang(2,3,4)	21.3	17.6	24.8

* Adapted from Heath et al. (2014)

4.3.3 Liquefaction Emissions

After the natural gas is produced, processed, and transported to the export location, it must be liquefied. Emissions from liquefaction derive from fuel combustion for electricity, natural gas venting, and fugitive methane leaks. There are several cooling technologies that may be used in a liquefaction terminal, each with unique energy requirements and energy efficiencies. Additionally, the capacity of the facility and the ambient temperature of the environment affect the efficiency and energy consumption. As a result, there is a wide range of estimated liquefaction GHG emissions in both peer-reviewed literature and in publicly available environmental impact assessments from the private sector. The literature is generally not transparent in the specific technology analyzed, either for proprietary reasons or because it was not considered important. To account for this variability, a distribution was fit to these point estimates (Table 12). Using the upper or lower bound did not significantly alter the overall life cycle emissions (Figure 24).

Table 12: Collected estimates from various studies on liquefaction stage emissions

Source	Estimate (g-CO2-equiv/MJ)
Hardisty 2012[251]	8.10
Artecini 2010[255]	6.50
Skone 2012[64]	7.60
Skone 2014[247]	8.24
Heede 2006[263]	6.15
Verbeek 2011[254]	5.90
LCFS[264]	7.30
Cohen 2013[265]	3.70
Biswas 2011[266]	7.70
Yoon 1999[267]	8.76
Okamura 2007[253]	8.40
Tamura 2001[252]	7.50
Yost 2003 [268]	3.80
Barnett 2012 [269]	2.40
Barnett 2012 [269]	5.20
Barnett 2012 [269]	3.80
Barnett 2012 [269]	4.00
Barnett 2012 [269]	4.00
Barnett 2012 [269]	3.40
Barnett 2012 [269]	6.80
Barnett 2012 [269]	8.10
Barnett 2012 [269]	3.80
Barnett 2012 [269]	5.90
Barnett 2012 [269]	4.20
Barnett 2012 [269]	4.90

The distribution derived from using the emissions estimates found in the literature and industry reports (Table 12) mostly lies between the maximum and minimum estimates, but primarily captures the lower values in the range (Figure 24). Because technology is becoming more efficient, it is likely that the emissions range of future liquefaction plants will fall within the uncertainty captured by the constructed distribution. Therefore, this distribution was used to represent the liquefaction stage GHG emissions in this study.

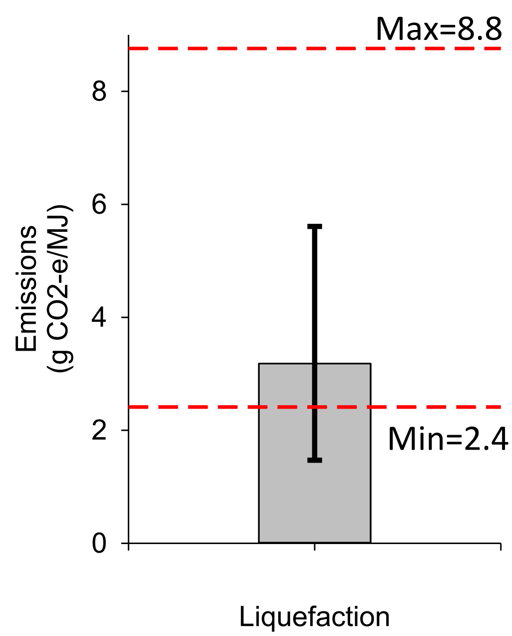


Figure 24: Liquefaction stage estimates using the distribution fit to the liquefaction stage estimates versus the maximum and minimum liquefaction stage estimates found in the literature

4.3.4 Shipping Emissions

LNG is transported on large ocean going vessels with capacities ranging from 75,000 to 265,000 m^3 of LNG[269]. Traditionally, LNG tankers run on steam engines powered by boil off gas (BOG). On average, BOG generated is approximately 0.15% by volume per day[263]. Tankers may have regasification facilities on board to supplement the BOG volume. However, in construction of new very large capacity tankers, there has been a transition to dual-powered diesel engines. These tankers are powered by diesel and re-liquefy the BOG onboard the vessel[32]. This technology is economically preferable for large cargos over long distances because the engines are more efficient and the full cargo of profitable LNG remains available for sale at the destination port. To quantify the impact the tanker technology has on the life cycle LNG emissions, the simulation was run under two different cases: a 140,000 m^3 capacity tanker powered by BOG generated steam supplemented with regasification technology, and a 265,000 m^3 capacity tanker powered by diesel, re-liquefying all of the BOG. These cases were chosen to bound the emissions because they represent the traditional and modern typical capacities and engine technologies. In both cases, port turn-around time was assumed to be three days[269], and the return trip was assumed to be of equal distance (returning to the port of origin) and fueled by diesel. In reality, a tanker fueled by BOG would likely retain enough LNG in its tanks to fuel the return voyage. Additionally, because there is a network of tankers, rather than being commissioned at its original port of origin, the tanker would likely be sent to the nearest port for its next LNG cargo. Therefore, these last two assumptions result in conservative estimates of the shipping emissions, which are likely to result in an upper bound shipping emissions estimate. For both cases the percentage of the journey spent in each engine mode was calculated according to Corbett (2008). Finally, steam and diesel engine efficiencies and associated combustion factors were represented by triangular distributions to capture the uncertainty (Table 13). In accordance with the assumption in the literature[252,269], this simulation presumes that there are no fugitive emissions released during shipping.

Table 13: Assumptions, parameters, and calculations used to estimate shipping emissions

Tanker Cargo Capacity (m^3 LNG)	260,000
Tanker Cargo Capacity (MJ NG)	5,839,688,400
Speed (knots)[263]	19.5
Natural BOG (%/day)	0.125
LNG boil off (m^3/day)	325
BOG NG equivalent (m^3/day)	195,002
BOG power (MW/day)	1,997
Distance (nm)	7,640
Number of hours of Journey (1 way)	392
Number of days of Journey (1 way)	16
Total BOG	32,600
Power (HP)[269]	40,000
Power (MW)	30

Number of engines	1		
Engine Type	STEAM	DIESEL	
Engine Efficiency	30%	50%	
MW input/hour	99	60	
Total MW input (full load)	38,953	23,372	
Engine Mode[124]	Trip (%)	Load factor (%)	
idle	1%	2%	
maneuvering	2%	8%	
precautionary zone operations	5%	12%	
slow cruise	7%	50%	
full cruise	85%	95%	
	STEAM	DIESEL	
percent of full load	0.9	0.9	
Total MW input	33116	19869	
Total MJ input	119,216,639	71,529,983	
	ALL BOG	ALL DIESEL	
natural BOG (MW)	32,600	32,600	
Re-gassified/re-liquefied (MW)	516	32,600	
% combusted to re-gassify/re-liquefy	3%	8%	** assumes NG is used in both cases
non propulsion combusted (MW)	15	2,608	
non propulsion combusted (MJ)	55,707	9,388,778	
return trip (MJ)	71,529,983	71,529,983	**assumes diesel in both cases
days at port[269]	3	3	
Engine Mode (diesel)	idle	idle	
port full load (MW)	4,295	4,295	
port at idle (MW)	86	86	
port at idle (MJ)	309,257	309,257	
Emission Factor (NG/diesel) g CO_2-equiv/MJ	48	72	
TOTAL EMISSIONS (g CO_2-equiv)	10,822,903,123	10,699,501,174	
Shipping Emissions (g CO_2-equiv/MJ)	1.9	1.8	

One parameter that could be expected to influence the environmental impact of LNG exports is the shipping distance. This shipping distance is determined as the most efficient trade route between the port of origin and the importing country. For this study, three three export terminals in the U.S., Sabine Pass, TX, Coos Bay, OR, and Cove Point, MD, were used. These three were chosen because they are either already approved to export LNG or have applied for DOE approval to export, and for their geographic diversity; the locations represent the maximum and minimum distance traveled from the U.S. to any one of the importing countries. For the import terminals, I selected ports in six countries: China, India, South Korea, Japan, UK, and the Netherlands. These countries were chosen because they are in the two key economically attractive markets (Asia and Europe) that either traditionally have imported large quantities of LNG or are expected to do so in the future. The distances traveled from each port of origin to destination were calculated using a port distance calculator[270], assuming that the Panama Canal upgrades were complete and therefore the canal was able to accommodate the large LNG tankers. The motivation behind this analysis is to understand if the origin and destination of the LNG impact life cycle emissions such that the DOE should consider contracted destinations of the LNG as part of the permitting approval process.

4.3.5 Regasification Emissions

The regasification stage of the LNG life cycle is the least discussed topic related to LNG in existing literature. There is a wide variation in energy required for regasification due to differences in ambient air temperatures and availability of resources such as seawater for heating, which can displace some of the energy requirements. Furthermore, regasification facilities can be co-located near power plants or other manufacturing facilities that require cooling. This co-location minimizes direct emissions from energy required to re-gasify the LNG.[32] For this study, a triangular distribution as described by Venkatesh (2011) was used to represent the uncertainty in emissions from the regasification component of the LNG process[1]. Total life cycle emissions through the regasification stage (including upstream, liquefaction, shipping, and regasification) are pre-combustion emissions. In this study, these emissions will be referred to as “landed emissions” since they represent the total GHGs emitted through the process of getting the natural gas on shore (i.e., to land) for use at the destination port. Table 14 provides estimates of landed LNG emissions for export origins and import destinations.

Table 14: Landed emissions (production, liquefaction, shipping, regasification) at each importing country

From/To	Japan			Korea			India		
gCO₂-e/MJ	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*
MD	39	*25*	*54*	39	*25*	*55*	38	*24*	*54*
OR	37	*23*	*53*	38	*23*	*53*	39	*24*	*54*
LA	39	*24*	*54*	39	*25*	*54*	39	*24*	*54*

From/To	China			UK			The Netherlands		
gCO_2-e/MJ	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*
MD	39	*25*	*55*	37	*23*	*53*	37	*23*	*53*
OR	38	*24*	*53*	38	*24*	*54*	38	*24*	*54*
LA	39	*25*	*54*	37	*23*	*53*	38	*23*	*53*

4.3.6 First Order Consequential Analysis

The results of the LCA were then used to conduct a first order consequential analysis of net GHG impacts resulting from U.S. exports, which considered the net change in emissions based on how the global market might respond to increased natural gas availability. As previous studies have discussed,[247] one possible outcome of U.S. LNG exports is that the natural gas could be used to replace existing fuel sources for electricity generation. For example, a country might choose to increase electricity production in natural gas combined cycle (NGCC) power plants, thereby reducing their reliance on coal power plants. Additionally, due to either price differentials or geopolitical reasons, a country may choose to use U.S. LNG as a replacement for natural gas previously imported from Russia or the Middle East. For this study, upstream coal life cycle emissions were obtained from Venkatesh et al. (2012)[271] adjusted to the AR5 GWP, and coal power plant combustion emissions were obtained from a distribution fit to the data from Steinmann et al. (2014) (Table 16)[272]. Emissions from Russian natural gas transported via pipeline were estimated using the same upstream and combustion emissions as the LNG exports. To account for the increase in pipeline transport distance associated with exporting natural gas from Russia, 3% was added to the U.S. fugitive emissions rate distribution. For example, where the average fugitive emission rate used to represent U.S. upstream production and transport was

3%, the average was assumed to be 6% for Russian upstream production and transport. This follows the Skone et al. (2014)[247] method to account for the extended pipeline transport distance required for Russian exports. This is likely to be a conservative estimate, as operational differences between U.S. and Russian natural gas production could further increase the Russian fugitive emissions rate.

Table 15: Parameters and assumptions used to estimate emissions from electricity generation from NGCC power plant

NGCC Power plant efficiency (min, most, max)	0.41	0.46	0.51
Emissions factor (min, max)[259] gCO_2-equiv/MJ	43	50	

Table 16: Parameters for the distributions used to represent upstream coal emissions (adapted from Venkatesh et al. 2012)[271] and coal fired power plant emissions (adapted from Steinmann et al. 2014)[272]

Coal production (g CO_2/MJ) triang(min,avg,max)	.4	.6	.7
Coal producgion – methane (CH_4/MJ)	.02	.15	.5
Coal transport (g CO_2/MJ)	.2	1.3	3.2
Coal Power Plant Emissions (kg/kWh)	log-logistic	mean = 1.09	std dev = .203

While electricity generation is the most common end use, natural gas is also regularly used as a source of thermal energy. Therefore, in order to depict broader representation of potential end use pathways, fossil fuel combustion for industrial heating was also included in this consequential analysis. For this study, a range of efficiencies and combustion factors were used for both natural gas and coal fired industrial boilers in order to capture the uncertainty in emissions (Table 17).

Table 17: Industrial heating efficiency and combustion emissions factor assumptions

Industrial heating efficiency (NG) (min, most, max)	0.7	0.8	0.94
Combustion emissions factor (NG) [min,max] g/MJ	43.0	50.0	
Industrial heating efficiency (Coal) (min, most, max)	0.75	0.85	0.89
Combustion emisions factor (Coal) [min,max] g/MJ	88	91	98

Another important component of a first order consequential analysis is to identify domestic opportunity costs of exporting natural gas rather than consuming it through domestic combustion. The end use and fuel displacement consequences described above all implicitly assume that U.S. demand would remain relatively flat, and in the absence of U.S. exports, there would be no additional natural gas produced for domestic use. While this study does not attempt to quantify the level of displacement through a global energy market model, it serves as a first step towards understanding potential consequences of displacement through bounding scenario-based assumptions. For example, this study explores the potential that given current U.S. natural gas electricity capacity, U.S. natural gas demand could increase such that all of the presumed export volume could be combusted domestically and displace U.S. coal baseload electricity generation. As a basis for this scenario, combustion emissions of U.S. coal power plants were compared to regional and global coal power plant average emissions. Data for this comparison were drawn from the EPA eGRID database[273] and from Steinmann et al (2014) that used regression models to predict coal power plant emissions factors.[272] Coal plants were limited to those with a nameplate capacity of over 100 MW with no combined heat and power generation. Furthermore, the plants were limited to those that were fueled by coal for over 95% of the electricity generated annually. It is important to note that this is a bounding analysis; in reality the electricity sector is complex and direct substitution occurring linearly in the short term based on cost differential is a simplifying assumption. Additionally, a decreased domestic demand for coal could increase the competitiveness of steam coal exports, which may either reduce some of the GHG benefits of increased domestic natural gas consumption[274] or contribute to further reducing global GHG emissions, such as in the case where U.S. PRB coal replaces other coal sources in new high efficiency coal-fired power plants in South Korea.[275]

4.4 Results

This study quantifies the GHG emissions from the production, liquefaction, shipping, and regasification phases of the supply chain, as well as the end use of the fuel in both the electricity and industrial heating sectors. These life cycle GHG emission results are discussed in the context of replacing alternative fuel sources, including locally produced coal and natural gas transported via pipeline from Russia.

4.4.1 Attributional Life Cycle Emissions

Mean landed (pre-combustion) life cycle GHGs for exported U.S. LNG after regasification at the importing country were found to be 37 g CO_2-equiv/MJ with a range of 27 to 50 (Figure 25). Of these landed emissions, the shipping stage of the life cycle contributes an average of 5%. Tanker capacity, rather than shipping distance or fuel type, is the most significant factor in determining shipping GHG emissions (Figure 26A). An analysis of the impact of origin and destination on shipping and landed life cycle emissions is shown in Figure 26. Both the average shipping and average landed life cycle emissions and 90% confidence intervals from each port of origin to each importing country can be found in Table 18 and Table 19, respectively.

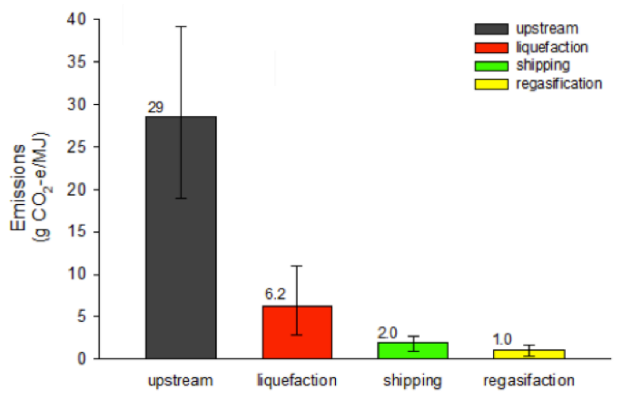


Figure 25: Landed (pre-combustion) life cycle emissions from US LNG exports

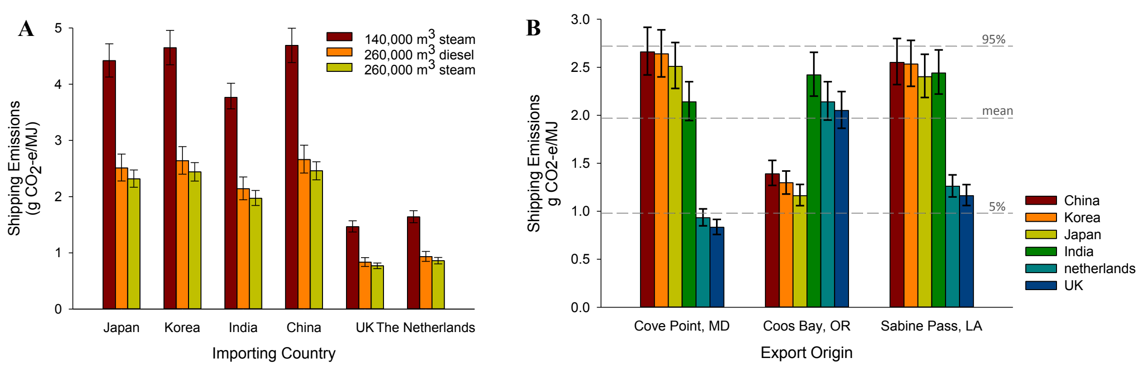


Figure 26: A) The shipping GHG emissions from Cove Point, MD to six import countries assuming a 140,000 m^3 steam tanker equipped with on board regasification (brown), a 260,000 m^3 diesel tanker equipped with reliquefaction (orange), or a 260,000 m^3 steam tanker equipped with on board regasification (green), and B) Shipping emissions from U.S. ports to six import countries, assuming 260,000 m^3 diesel vessel, and the mean and 90% confidence interval from the distribution fit to all eighteen voyage distances (dashed gray lines). Note: a 260,000 m^3 tanker equates to a capacity of about 5.5 Bcf in gaseous form and a 140,000 m^3 tanker equates to a capacity of about 3 Bcf in gaseous form.

Table 18: Estimated shipping emissions by origin and destination (100-yr GWP)

From/To	**Japan**			**Korea**			**India**		
gCO_2-e/MJ	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*
MD	2.5	*2.3*	*2.8*	2.6	*2.4*	*2.9*	2.1	*1.9*	*2.4*
OR	1.2	*1.1*	*1.3*	1.3	*1.2*	*1.4*	2.4	*2.2*	*2.7*
LA	2.4	*2.2*	*2.6*	2.5	*2.3*	*2.8*	2.4	*2.2*	*2.7*
From/To	China			UK			The Netherlands		
gCO_2-e/MJ	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*
MD	2.7	*2.4*	*2.9*	0.8	*0.8*	*0.9*	0.9	*0.8*	*1*
OR	1.4	*1.3*	*1.5*	2.1	*1.9*	*2.2*	2.1	*2*	*2.4*
LA	2.6	*2.3*	*2.8*	1.2	*1.1*	*1.3*	1.3	*1.2*	*1.4*

Table 19: Landed emissions (production, liquefaction, shipping, regasification) at each importing country

From/To	**Japan**			**Korea**			**India**		
gCO_2-e/MJ	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*
MD	39	*25*	*54*	39	*25*	*55*	38	*24*	*54*
OR	37	*23*	*53*	38	*23*	*53*	39	*24*	*54*
LA	39	*24*	*54*	39	*25*	*54*	39	*24*	*54*
From/To	China			UK			The Netherlands		
gCO_2-e/MJ	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*	*mean*	*5%*	*95%*
MD	39	*25*	*55*	37	*23*	*53*	37	*23*	*53*
OR	38	*24*	*53*	38	*24*	*54*	38	*24*	*54*
LA	39	*25*	*54*	37	*23*	*53*	38	*23*	*53*

Life cycle emissions from exported LNG were found on average to be 655 g CO_2-equiv/kWh for electricity generation and 104 g CO_2-equiv/MJ for thermal energy generation. These emissions primarily result from upstream production and downstream combustion. The liquefaction, shipping, and regasification components of the life cycle contribute an additional 72 g CO_2-equiv/kWh over the domestic natural gas electricity generation life cycle emissions from production and combustion (Figure 27). Therefore, exporting natural gas instead of combusting it domestically increases emissions from natural gas electricity generation by an average of 11%.

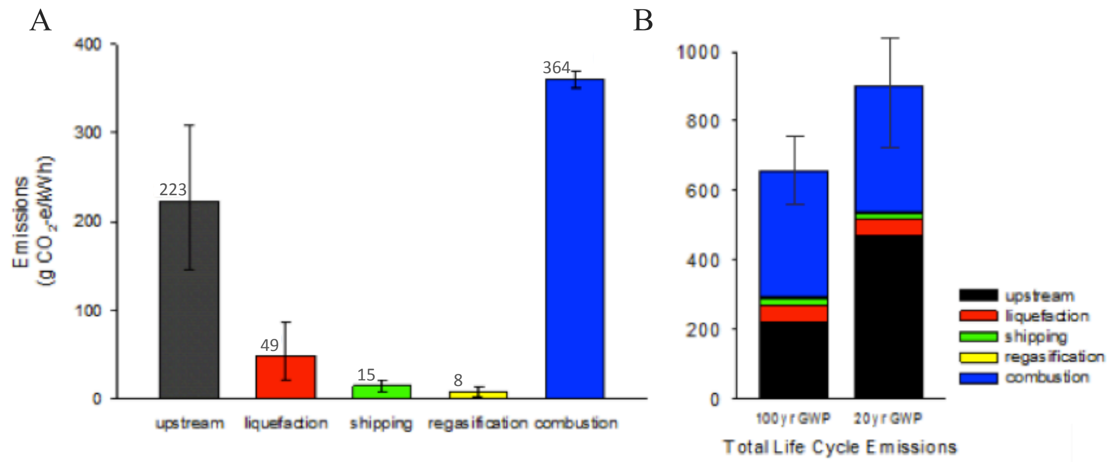


Figure 27: Comparison of emissions from U.S. LNG exports, Russian natural gas, and coal using a 100-year GWP for A) electricity generation and B) industrial heating

The life cycle emissions from LNG exports are most sensitive to the GWP and fugitive emissions rate (Figure 28). This implies that increases in efficiency in these processes, such as co-locating regasification facilities with power plants to use waste heat to regasify the LNG, would have a nominal impact on life cycle emissions. While still beneficial from an economic and absolute emissions perspective, the discussion of future efficiency increases related to the LNG process is not relevant to decisions based on life cycle emissions. A summary of the key parameters and their impact on the total life cycle emissions for electricity generation (based on the Spearman correlation coefficient) is outlined in Table 20.

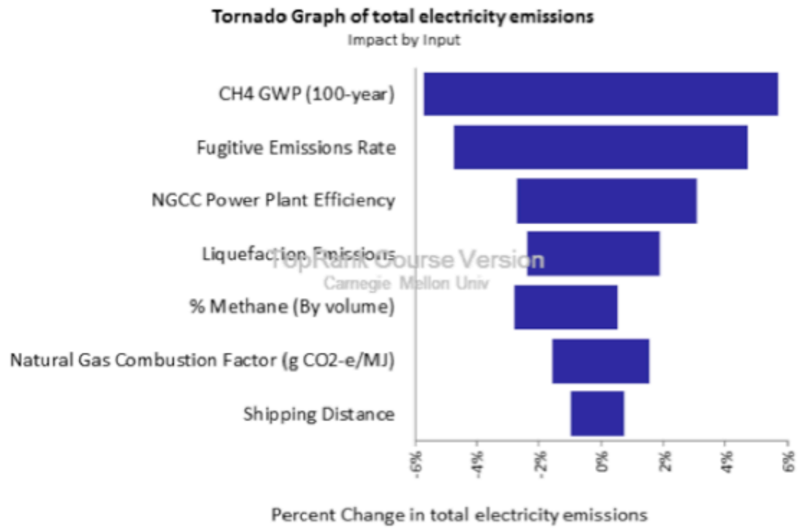


Figure 28: Tornado diagram representing the parameters and assumptions that have the largest impact on the results of the Monte Carlo simulation

Table 20: Summary of uncertainty parameters ranked by their Spearman correlation coefficient. Each parameter's distribution and associated units are also shown.

Rank	Name	Units	Description	Spearman Correlation Coefficient
#1	CH4 100-year GWP		RiskNormal(36,8.5)	0.73
#2	Fugitive Emissions Rate	%	RiskTriang(2,3,4)	0.40
#3	Liquefaction Emissions	g CO_2-e/MJ	RiskExtvalue(5.1,2.0)	0.35
#4	NGCC Power Plant Efficiency	%	RiskTriang(.41,.46,.51)	(0.20)
#5	Natural Gas Combustion Factor	g CO_2-e/MJ	RiskUniform(350,370)	0.14
#6	Natural Gas Percent Methane (by volume)	%	RiskTriang(.83,.93,.95)	0.09
#7	Unconv well completion: total vent/flare	Mt CH_4	RiskTriang(13.5,177,385)	0.07
#8	well completion: EUR	Bcf	RiskTriang(0.5,2,5.3)	(0.06)
#9	CO2 vent	g CO_2-e/MJ	RiskTriang(.2,.7,2.8)	0.06
#10	Flaring	g CO_2-e/MJ	RiskTriang(0,.43,1.3)	0.05
#11	Shipping Distance	nm	RiskTriang(1890.8,10514,10514)	0.05
#12	Regasification Emissions	g CO_2-e/MJ	RiskTriang(0,1,2)	0.04
#13	conv. Lease plant energy	g CO_2-e/MJ	RiskTriang(2,3.3,4.3)	0.04
#14	well completion: flare rate	%	RiskTriang(.15,.41,1)	0.03
#15	Well pad construction	g CO_2-e/MJ	RiskTriang(.05,.13,.3)	0.03

4.4.2 First Order Consequential Analysis

When considering the global benefits of LNG exports, the life cycle LNG emissions must be compared to emissions from alternative sources of fuel that U.S. exports would displace. The two most likely candidates for replacement would be coal and Russian natural gas.[247] The results of a Monte Carlo simulation show that the benefit of displacing either of these two sources of fuel depends on the GWP metric chosen and is highly sensitive to the upstream fugitive emissions rate from natural gas production and pipeline transport. When considering a 100-year GWP, mean life cycle exported U.S. LNG emissions are within the uncertainty bounds of Russian natural gas exports, and result in about 45% fewer emissions than coal electricity generation (Figure 29A). When considering a 20-year GWP, exported U.S. LNG would reduce emissions from electricity production via Russian gas by 27% and cut emissions from electricity production from coal by 32% (Figure 30B). The higher emissions from Russian natural gas are due to the higher leakage rate assumed to account for the longer pipeline transport distance. This further emphasizes the fact that emissions from liquefying, shipping, and regasifying natural gas are marginal relative to the production and combustion emissions, and that the life cycle emissions of natural gas production are highly dependent on the fugitive emissions rate.

In addition to electricity generation, natural gas is often used for industrial heating. Both coal and natural gas-fueled industrial boilers have higher efficiencies than power plants. As a result, coal use for industrial heating is more competitive with LNG on a life cycle GHG emissions basis. When considering a 100-yr GWP, mean GHG emissions from U.S. LNG exports would be 16% and 13% lower than industrial heating fueled by Russian natural gas exports and coal, respectively. However, when using a 20-year GWP, mean GHG emissions from U.S. exports would be 4% higher than coal (Figure 31B). Despite this increase in emissions, it is important to note that if LNG were to displace Russian natural gas, it would reduce emissions by 27% (Figure 31B). This is illustrative of the complexity of quantifying net impact of LNG exports; there are numerous consequential pathways influenced by the emergence of a U.S. natural gas export market, and the specific end use and resulting fuel displacement is outside the domain of control of U.S. policymakers who need to approve LNG projects.

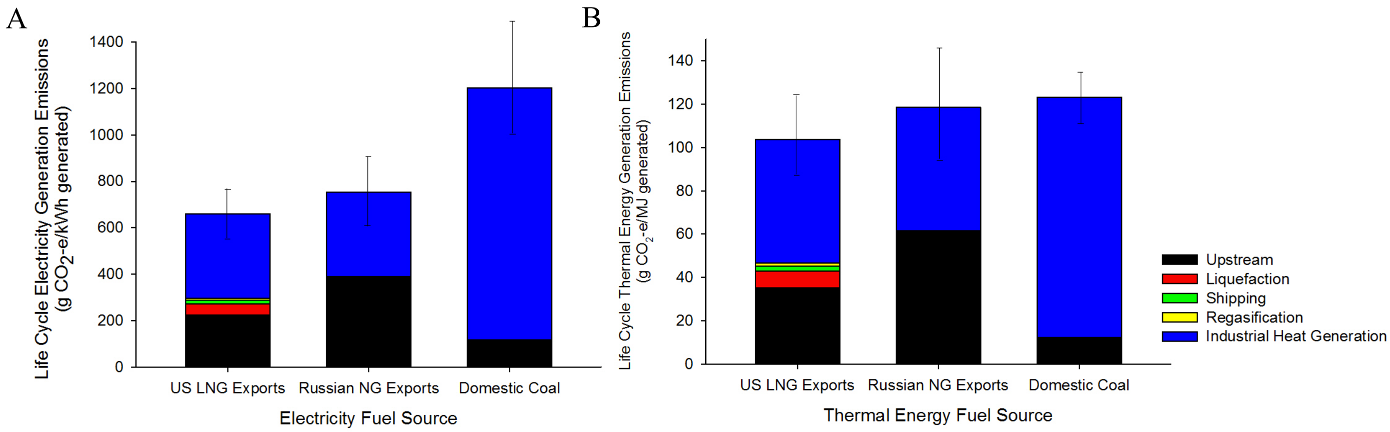


Figure 29: Comparison of emissions from U.S. LNG exports, Russian natural gas, and coal using a 100-year GWP for A) electricity generation and B) industrial heating.

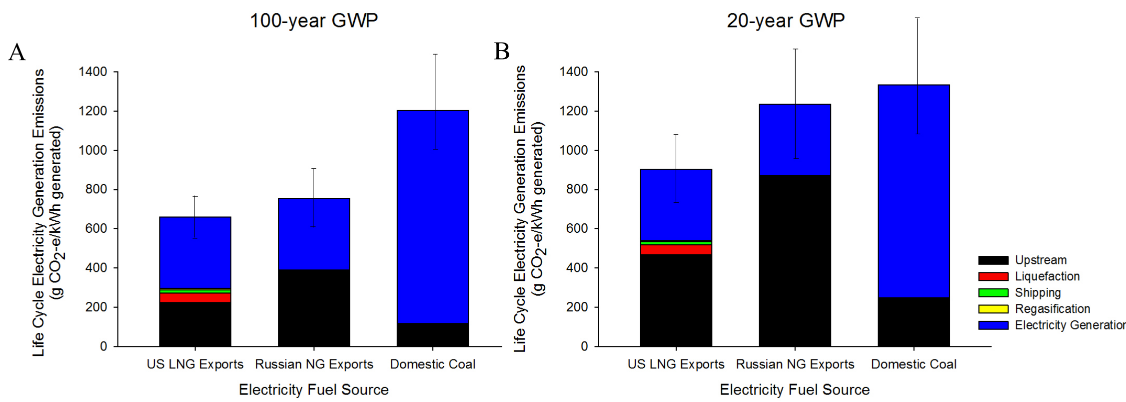


Figure 30: Comparison of life cycle greenhouse gas emissions from alternative fuel sources including U.S. LNG exports, Russian natural gas exports via pipeline, and coal for electricity generation

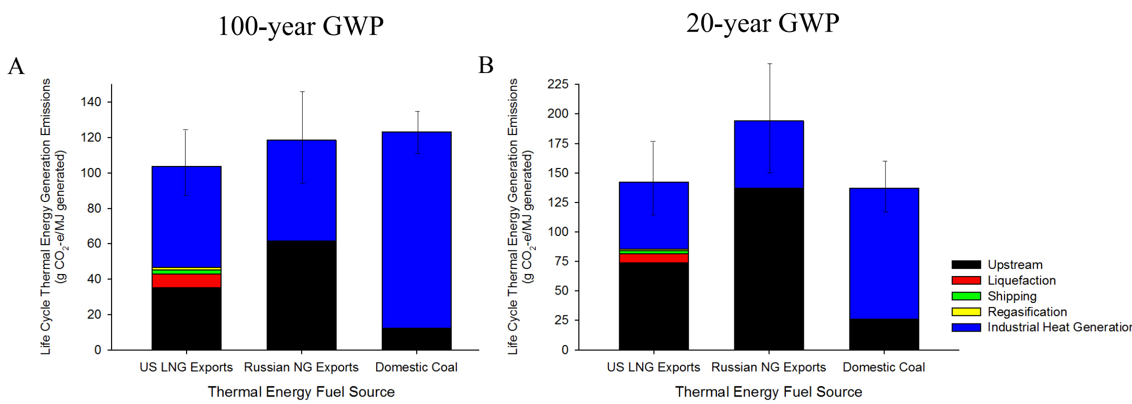


Figure 31: Comparison of life cycle emissions from LNG exports, Russian gas, and coal for industrial heating with A) a 100-year GWP and B) a 20-year GWP

4.4.3 Domestic Opportunity Cost

When comparing average coal power plant combustion emissions, taking into account such factors as coal type, plant age, and plant capacity, there is no significant regional variation (Figure 32)[272]. Therefore, all else being equal there are no marginal benefits of displacing coal emissions in Asia or the EU versus domestically in the United States. As a result, displacing U.S. coal generation by combusting U.S. natural gas is more efficient in reducing global GHGs than exporting it abroad; an equivalent reduction in combustion emissions can be obtained without the additional supply chain emissions required by the liquefaction, shipping, and regasification steps for export. Again, however, it is important to note that there is not a linear relationship between natural gas price and coal substitution[274], and therefore the potential for this domestic absorption of excess supply would likely be both market and policy driven. Additionally, there would be market changes resulting from this shift in U.S. natural gas consumption. For example, there could be an increase in U.S. coal exports which may serve to either increase or decrease net GHG emissions depending on where it is combusted and what fuel it displaces[275]. This wide range of possible market consequences makes the net impact of increased domestic natural gas use uncertain.

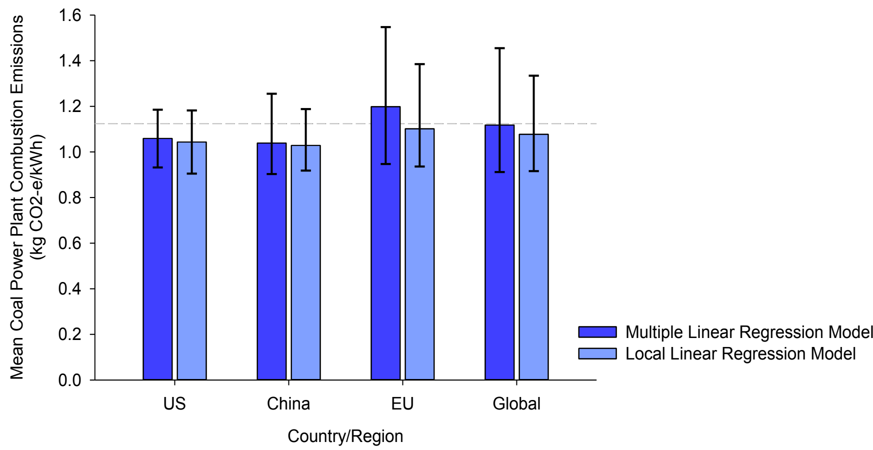


Figure 32: Comparison of coal power plant combustion emissions in the U.S., China, across the EU, and globally as modeled by Steinmann et al. (2014), and compared to EGRID emissions (dotted grey line). A description of the multiple linear regression model and local linear regression model is available in Steinmann et al.

4.5 Discussion

The current environmental component of the national interest calculation considers only localized environmental impacts directly related to the liquefaction project under review.[36] However, as the DOE and NEPA processes have begun to recognize[248, 276], it is also important to consider both the upstream emissions of increased natural gas production and the downstream life cycle emissions from these LNG export projects in order to ensure the U.S. is contributing to the minimization of global net GHG emissions. This analysis can then serve as a basis for determining the social cost of carbon embodied in these US exports, which affects both country-level GHG emissions inventories and the potential for future domestic GHG reductions.

This study found that the emissions from the liquefaction, shipping, and regasification segments of the LNG life cycle are fewer than 11% of the total life cycle emissions of LNG exports for electricity generation based on a 100-year GWP and 3% average fugitive emission rate. This percentage would continue to decrease as a result of increased methane leakage rates and/or a 20-year GWP assumption (Figure 33). Based on a sensitivity analysis of these results (Figure 28), the key model parameters that can have a significant impact on the LNG life cycle emissions are the end use efficiency, the GWP (both time horizon used, and value within a given time horizon probability distribution), and the fugitive emissions rate. Other uncertain parameters however, such as liquefaction plant efficiency, tanker capacity, tanker fuel, and shipping distance can vary widely without materially affecting the overall life cycle emissions.

From a first order consequential perspective, the mean global GHG savings from U.S. LNG exports are likely associated with coal displacement for electricity generation (Table 21). The break-even point for GHGs from U.S. LNG and coal electricity would be over a 9% fugitive emissions rate using a 100-year GWP, and a 6% fugitive emissions rate using a 20-year GWP (Figure 33). Additionally, GHG savings from U.S. LNG exports can be achieved through displacing coal for industrial heating. Using a 100-yr GWP, the break-even point for heating is a 5% leakage rate. However on a 20-year GWP basis, U.S. natural gas displacement of coal for

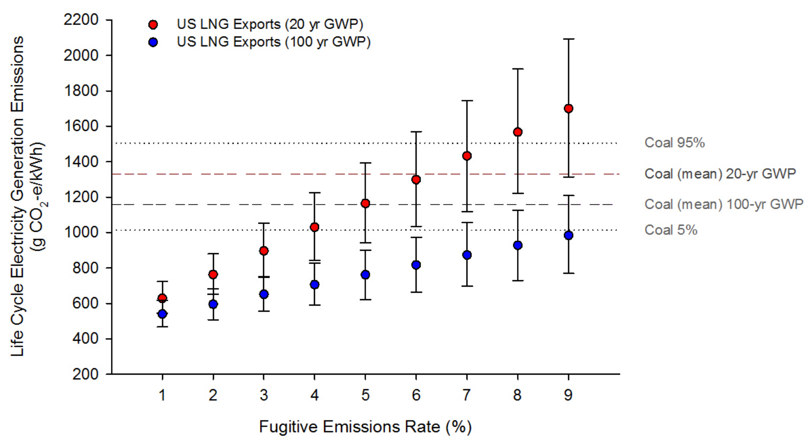


Figure 33: The sensitivity of life cycle emissions of LNG exports for electricity generation to fugitive emissions rates.

heating would be advantageous only up to about a 3% fugitive emissions rate (Figure 34). Finally, GHG savings are also associated with displacement of Russian pipeline natural gas for both electricity generation and industrial heating regardless of GWP, as long as the U.S. fugitive emission rate remains below the estimated 5-7% rate of Russian natural gas (Table 22).

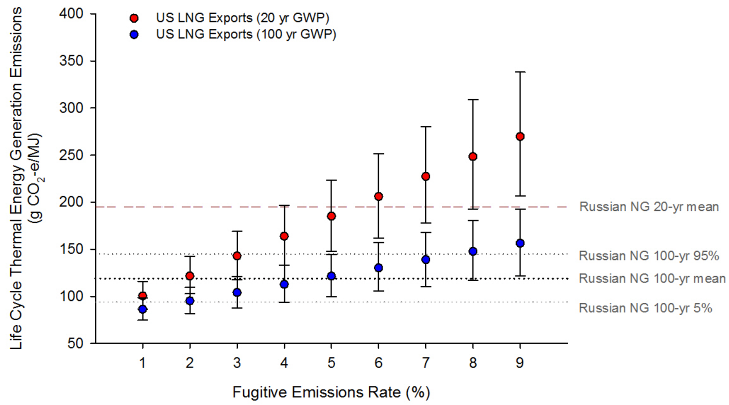


Figure 34: The sensitivity of life cycle emissions of LNG exports for electricity generation to fugitive emissions rates as compared to Russian natural gas with a constant fugitive emission rate of 5-7%. For the 100-yr GWP, natural gas results in fewer emissions than Russian natural gas up to 5% methane leakage rate. Using a 20-yr GWP, the break-even point is also around a 5% fugitive emissions rate.

Table 21: Life cycle emissions estimate results for LNG exports, Russian natural gas, and coal

100-yr GWP						
	mean	*5%*	*95%*	*mean*	*5%*	*95%*
LNG EXPORTS		g/kWh			g/MJ	
upstream	220	150	310	62	40	86
liquefaction	50	22	86	14	6	24
shipping	15	8	21	4	2	6
regasification	8	2	13	2	1	4
combustion	364	330	400	100	90	110
total	660	560	770	180	160	214
RUSSIAN NG:						
upstream	390	250	533	110	70	150
combustion	364	330	403	100	90	112
total	750	600	905	210	170	250
COAL:						
upstream	120	50	210	33	14	60
combustion	1,090	920	1,380	300	255	380
total	1,200	1,010	1,510	333	280	420

20-yr GWP			
	mean	*5%*	*95%*
LNG EXPORTS		g CO_2-equiv/kWh	
upstream	470	320	640
liquefaction	50	22	86
shipping	15	8	20
regasification	8	2	13
combustion	364	330	400
total	900	740	1,090
RUSSIAN NG:			
upstream	870	610	1,150
combustion	360	330	400
total	1,230	960	1,520
COAL:			
upstream	250	94	440
combustion	1,090	920	1,380
total	1,330	1,080	1,680

Table 22: Life cycle emissions for industrial heating using U.S. LNG, Russian natural gas, and coal

100-yr GWP			
	mean	*5%*	*95%*
LNG EXPORTS		g CO_2-equiv/MJ	
upstream	35	23	50
liquefaction	8	4	14
shipping	2	1	3
regasification	1	0	2
combustion	60	50	65
total	100	87	120
RUSSIAN NG:			
upstream	61	50	85
combustion	57	50	65
total	120	94	150
COAL:			
Upstream	12	5	22
combustion	110	105	130
total	124	113	140

20-yr GWP			
	mean	*5%*	*95%*
LNG EXPORTS	g CO_2-equiv/MJ		
upstream	74	50	100
liquefaction	8	4	14
shipping	2	1	3
regasification	1	0	2
combustion	57	50	65
total	140	115	174
RUSSIAN NG:			
upstream	140	96	180
combustion	57	50	65
total	200	150	240
COAL:			
upstream	26	10	49
combustion	110	105	120
total	140	120	160

While this study focused on electricity generation and industrial heating as two important end uses of natural gas, there are several additional end uses, including transportation, residential heating and cooking, and petrochemical production. The existence of these additional potential end uses further complicates the uncertainty in the emissions savings from U.S. LNG exports. Rather than outline all possible end use permutations and potential fuel displacement, the implication for the U.S. is that in order to ensure maximum GHG benefits of LNG exports, fugitive emissions rates must be reduced as much as possible. This is important because while the U.S. cannot designate a specific end use of the LNG, the U.S. fugitive emission rate is within U.S. regulatory domain. The government has recognized the importance of minimizing methane leakage as a step towards reducing the U.S. contribution to climate change.[277] As regulation progresses and the domestic fugitive emissions rate decreases, it will become more likely that LNG exports will result in global emissions savings.

An additional consideration in the evaluation of the U.S. national interest of LNG exports beyond the first order absolute net global emission savings is the GHGs embodied in trade.[278,279] The embodied CO_2 equivalent emissions in the exported LNG have implications for social impacts along the LNG supply chain that are not captured through a life cycle analysis. This affects both country-level GHG emissions inventories and the potential for future reductions. Because approximately 41% (58% using a 20-year GWP) of life cycle LNG export emissions would arise from domestic extraction, pipeline transport, and liquefaction, increased extraction of natural gas without the domestic benefits of reduced combustion emissions would likely not be advantageous for the U.S. from a country-based carbon accounting perspective. Our mean estimates are that each thousand cubic feet (Mcf) of natural gas loaded onto a ship in liquefied form at a U.S. port represents about 0.037 metric tons of GHGs (for 100-yr GWP from production, transmission and liquefaction). Using the 2020 social cost of carbon ($2014) from the U.S. Interagency Working Group[256] for a 3% discount rate of $49/metric ton GHG, this means each Mcf of natural gas converted to LNG and exported from the U.S. for electricity generation potentially could cost the U.S. about $1.80 of social cost for embodied GHGs, or $0.50 to $5.50/Mcf across the full range of estimates for the 2020 social costs of carbon. Assuming a natural gas price of $4/Mcf, exporting LNG could have a social cost of between 12.5% to 135% of the market price. Because climate change damages are global, potential GHG reductions in other countries also benefit the U.S. As an illustrative example, using $49/ton GHG as a social cost, 1 kWh of electricity generated by coal in the United Kingdom (UK) has a social cost of carbon of about $0.06. In contrast, 1 kWh of natural gas generation using LNG imported from the U.S. has a total social cost of about $0.032/kWh, with $0.013 of this cost comprised of U.S. production, transmission and liquefaction. Therefore, while U.S. LNG displacing coal in the UK results in a net global social cost savings of about $0.028/kWh ($0.06-$0.032), from a country-level accounting perspective, the U.S. incurs more costs than benefit; the U.S. incurs the aforementioned cost of $0.013 while the UK sees a cost savings of about $0.04 composed of the difference in social cost resulting from coal production and combustion versus the social cost attributed to LNG shipping, regasification, and combustion. Both the monetized global savings from LNG exports and the domestic cost of the embodied carbon are likely underestimates, as new evidence suggests the social cost of carbon may be several times larger than previously estimated.[280] This is an important consideration because increased international fossil fuel trade has prompted the discussion for new climate policies that recognize the responsibility of embodied carbon contributions at all points in the supply chain of goods and services, including mechanisms to account for carbon emissions at the point of fossil fuel extraction[281, 150]. The economic benefits of natural gas development explicitly for export needs to be analyzed against the monetized domestic social costs. These include environmental and public health risks, and potentially the expected increase in costs for domestic emissions reductions. This social cost differential may be important to consider as a component of the national interest determination, among other regional social costs of shale gas production such as air, water, and road impacts[7-9].

This study raises important policy implications for consideration in evaluating the national interest of LNG exports. From a global emissions perspective, this study has shown that exporting LNG can help to reduce life cycle GHG emissions from electricity generation and industrial heating. However, the extent to which this net reduction is realized depends on the end use of the fuel, the upstream methane leakage rate, the fuel displaced by the natural gas use, and the downstream consequences of the displaced fuel source. The downstream consequences of the fuel displacement, such as cheaper coal, can induce a rebound effect of additional fossil fuel consumption. While this may in fact increase net GHG emissions, it is important to also consider the economic benefits and accrued social benefits from the increased access to energy services. This demonstrates the complex interaction between environmental, social, and economic consequences that extend beyond what is captured through life cycle assessment and the current national interest determination. However, by quantifying both the direct GHG emissions from the LNG life cycle and the first order net impacts of fuel substitution and alternative end uses, the bounding scenarios in this study provide an important perspective in further informing the environmental component of the national interest discussion. In order to reduce the uncertainty of whether exports would result in a net global benefit, it would be most productive for U.S. policy to focus on incentivizing reductions in domestic fugitive emissions rates, including conducting more accurate and consistent leakage monitoring and indicting penalties for infractions, rather than prioritizing increased liquefaction, shipping, or regasification efficiency. Additionally, when discussing the national interest of LNG exports, it may be important for the U.S. to consider embodied carbon emissions in trade and identify the social cost accrued by the U.S. on behalf of global net GHG savings.

Chapter 5: Characterizing the Global Crude Trade and Quantifying the GHG Impact of Carbon Mitigation Policies

5.1 Abstract

In 2014, over 33 billion bbls of crude oil were produced globally. Of this total production, approximately 64% was traded internationally. This chapter begins by characterizing the 2014 crude system by identifying country trade partners and volumes, and developing a network in which supply, imports, exports, and consumption are balanced across 62 countries. In total, this system is estimated to cost ~$3T and was responsible for emitting ~16.5 Gt CO_2-eq, which equates to a third of 2010 global greenhouse gas emissions (49 +/-4.5 Gt CO_2-eq). This chapter then explores the relationship between cost savings and GHG reductions through several optimization scenarios regarding supply flexibility, refinery input crude quality (API) flexibility, a global carbon budget, carbon accounting strategies, and individual country emissions allocations. The results of the optimization show that given a severely constrained carbon cap and unilateral, country specific carbon allocations, there are dynamic shifts within the system depending on the carbon accounting strategy employed. In order to minimize crude specific emissions to 3 Gt CO_2-eq (the transportation portion, or ~14%, of suggested 21 Gt annual global cap), demand would need to decrease by about 80%. The cost-effectiveness of carbon mitigation under such a strict cap varies across accounting strategies. For example, under a consumer based accounting method whereby the consuming country is responsible for all embodied life cycle emissions, the carbon intensity is 4.1 mt CO_2/mt consumed versus 3.9 mt CO_2/mt consumed in the producer based and location based accounting scenarios. Furthermore, given no unilateral action (country specific carbon caps), total demand satisfied increases by over 40%. This is evidence of an important interaction between various climate policies, such as carbon accounting methods, a designated global carbon cap, and unilateral country specific emissions allocations that could inhibit cost-effective carbon mitigation if not considered from a systems perspective.

5.2 Introduction

The previous chapter demonstrated that GHG accounting practices could unintentionally inhibit global emissions reductions. In the case of LNG exports, displacing domestic coal with US natural gas resulted in a monetized global benefit of $28/MWh, while costing the United States $13/MWh. In this chapter, I explore how these carbon accounting practices influence trade on a global scale using the global crude oil trade as an example.

In 2014, global oil consumption reached over 33 billion bbls of crude[282], with approximately 49% of consumption by OECD countries and 51% by non-OECD countries. Crude consumption is disproportional to population, with the largest per capita consumption in North America and Europe (Figure 35). Between 2002 and 2013, Saudi Arabia and Russia were consistently the largest oil producers. United States oil production had been steadily declining from 1985 until about 2009 when production rapidly began to increase (Figure 36A). Between 2010 and 2014, US production increased by 14%, and the United States currently accounts for approximately 13% of the world's crude supply, making it the largest global producer of oil282 (Figure 36B). This estimate includes conventional crude oil, tight oil, oil sands, and natural gas liquids. While historically oil production has been from conventional sources, it is projected that the percentage of crude supply from unconventional sources may increase substantially in the future depending on resource constraints, fuel prices, elasticity of demand, and potential climate policies[120]. Of the over 32 billion bbls of oil produced each year, approximately 64% is exported from one country to another (21 billion bbls)[282], primarily by international shipping (17 billion bbls[121], or 83% of exports).

International trade poses a challenge for climate change mitigation. While traditional climate policies tend to focus on unilateral mitigation measures (actions taken by individual countries acting independently), international transport of goods and services is a global sector. As a result, the responsibility for reducing GHG emissions associated with shipping do not fall within the jurisdiction of any individual country[20]. The lack of responsibility for trade emissions, combined with the asymmetry of unilateral climate mitigation strategies, creates an opportunity for carbon leakage[138, 145-147]. Carbon leakage occurs when action taken by countries to reduce emissions is partially or wholly offset by increased emissions elsewhere in the world. In an effort to characterize current carbon leakage and identify the opportunity for leakage in the future, there

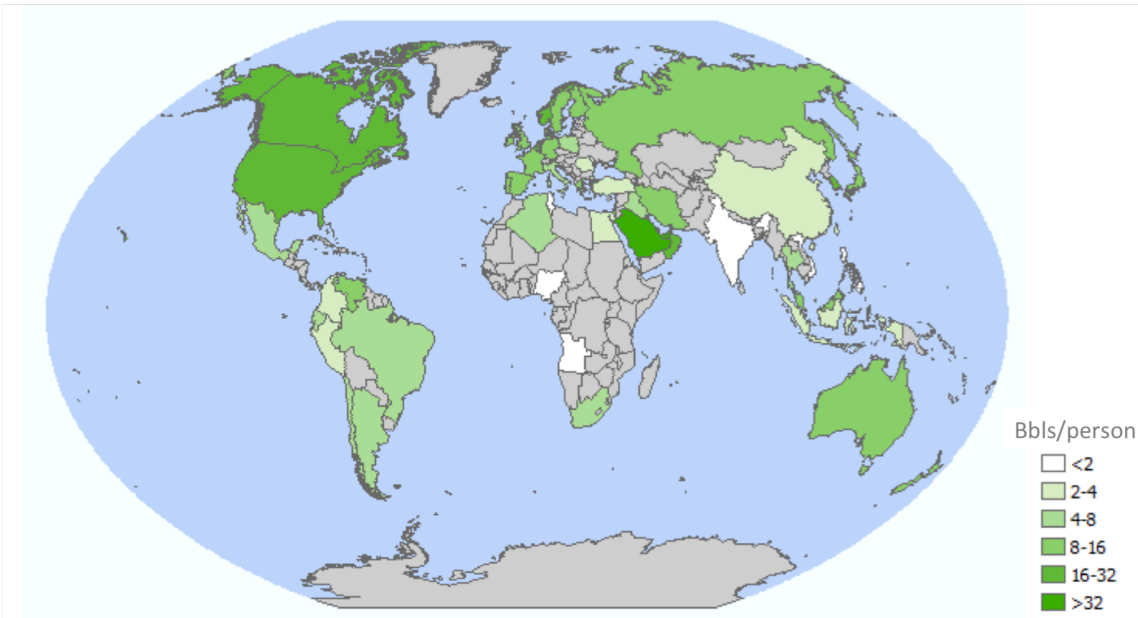


Figure 35: Crude consumption per capita[282]

has been increasing interest in quantifying embodied carbon in trade[131, 283, 284]. Additionally, strategies for mitigating carbon leakage have been theorized; one widely discussed set of strategies is called border control adjustments (BCAs)[285]. These are measures that would indirectly account for embodied carbon in traded goods, such as a border tax on imported goods. However, the implementation cost of such border adjustments would be high due to the complexity of regulation. Furthermore, there is concern that such measures would cause the cost of emissions reduction to shift from developed to developing countries[146].

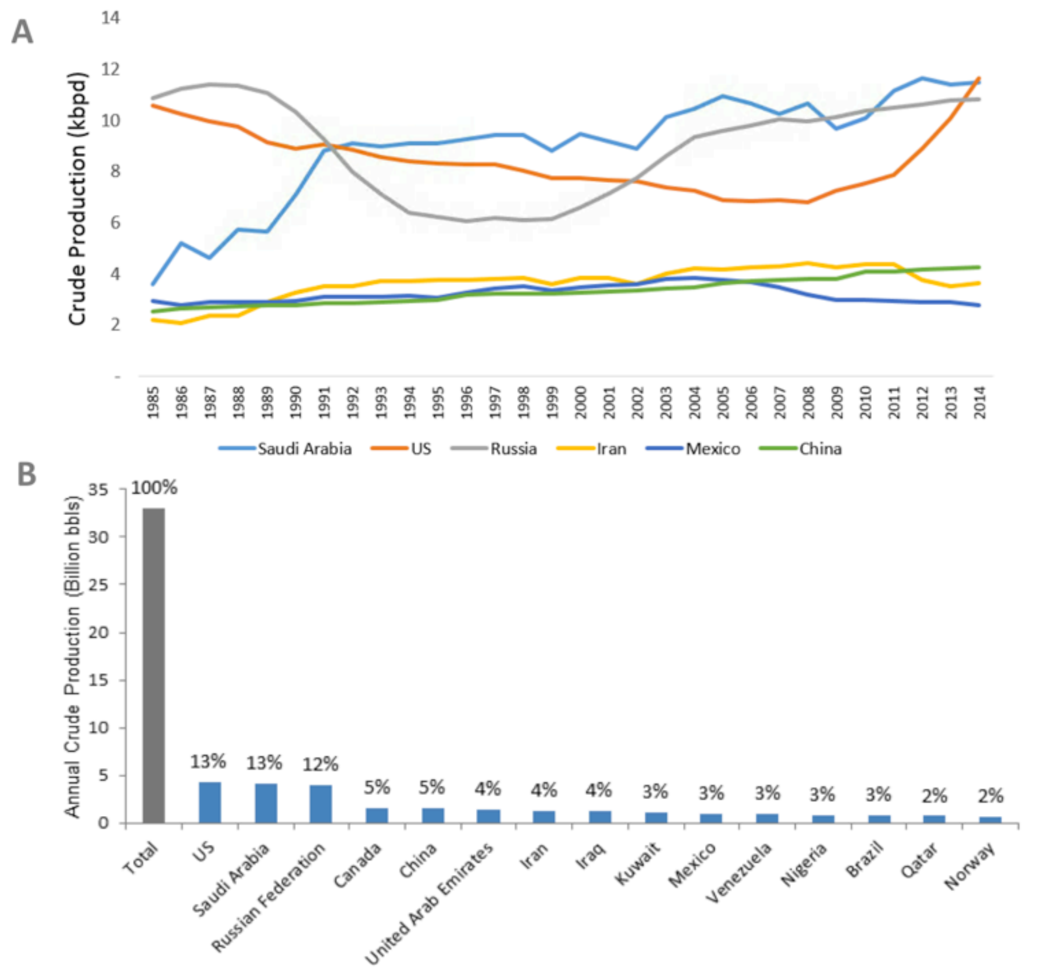


Figure 36: Crude production A) since 1985 by the 2014 top six producers, and B) by top 15 producers, accounting for 80% of total oil supply (2014)[282]

As climate policies continue to become more stringent, the need to address carbon leakage and account for embodied carbon emissions will become imperative. Through international climate discussions, such as at COP21 in 2015, policy makers have come to the agreement that the average global temperature rise caused by greenhouse gas emissions should not exceed 2 °C above the preindustrial average global temperature[143]. In order to limit the average global temperature rise, climate models have demonstrated a limit to the cumulative emissions that can be sustained by the climate system. This limit is often referred to as the global carbon budget. The Intergovernmental Panel on Climate Change (IPCC) reported that this budget is in the range of 870-1,240 Gt CO_2-eq between 2011-2050 in order to have a 56% chance of not exceeding this 2°C temperature increase[139]. A recent study by McGlade and Ekins (2015) used an integrated assessment model to explore the implications of this emissions limit for fossil fuel production[139]. In their scenario to keep average global surface temperature rise below 2°C for all years to 2200, they found 2050 GHG emissions must be constrained to 21 Gt CO_2-eq. This is compared to the 48 Gt CO_2-eq emitted in 2010, and a baseline 2050 projection of 71 Gt CO_2-eq given no emissions mitigation. This latter projection would result in less than 5°C global average temperature rise[139].

Metrics of global trade, such as cost and emissions, are often studied using environmentally extended economic input-output analysis[131, 286], network analysis[133-136, 287], or computable generalized equilibrium models (CGEs)[137]. The first two categories of trade analysis are characterization methods. They largely serve to describe the current state of interaction and detangle current emissions production to attribute embodied emissions. CGEs are linear optimization models based on economic interactions such as price elasticity of demand, trade elasticity, elasticity of substitution, welfare, etc. Economic models are the most commonly used method of assessing energy systems[128]. Energy-economic modeling can be traced back to the first computable generalized equilibrium (CGE) model developed by Hudson and Jorgenson in 1974[129]. This subset of energy models typically requires many assumptions about international trade flow data, elasticity of substitution and transformation, and perfect substitution[130]. While the purpose of CGEs is to adequately represent global systems, the vast number of simplifying assumptions introduces substantial uncertainty. While other methods have been used to study energy trade patterns and emissions, such as input-output assessments[131] and network analysis[133-136, 287], CGEs[137, 288] remain the dominant energy modeling technique.

In the early 1980s, CGEs shifted from models focusing specifically on energy and economics only to E3 models, or energy-economy-environment models[127]. Several studies have been done with CGE/climate model combinations such as TIAM-UCL model[139] and ROMEO model[120] among others[144, 289] to understand the implications of various policies on the global climate system. In particular, one commonly applied CGE used to study trade flows is called the Global Trade Analysis Project (GCAM). GCAM is an integrated assessment model that includes representations of the economy, energy sector, land use and water[290]. Another widely used international trade CGE model is the Global Trade Analysis Project (GTAP). GTAP is a multi-regional, multi-sector CGE model based on input-output models describing bilateral trade patterns, production, consumption, and intermediate use of commodities and services.

The objective of this chapter is to explore the opportunities for cost and greenhouse gas savings within international trade and to identify how climate policies such as carbon budgets and BCAs can influence the cost-effectiveness of carbon mitigation using the global crude trade as a case study. Similar to the concept proposed by Stromman et al. (2008), I explore how shifting trade patterns can be used as a tool for carbon mitigation[147]. By using a simple linear optimization model based on a mass balance of country specific production, exports, imports, and consumption, I explore the extreme bounds of cost and emissions reductions under various climate policy scenarios and investigate the influence of carbon accounting strategies on the crude trade network in a transparent manner. While I make many simplifying assumptions that would otherwise be included in a typical CGE, this allows us to understand the interactions between different carbon mitigation measures without them being confounded by other variables such as price, fuel substation, etc. Additionally, these simplifications enable country specific exploration, as opposed to aggregating on regional level as is required for computationally intensive CGE models. Similar to other simplified models, this is a single industry model that assumes perfect foresight[291].

5.3 Methods

Throughout this study, I consider international crude trade on a single year time scale. While CGE models characterize the progression of change over time as part of a feedback loop of price, supply, and demand, I use a single year time scale to understand how the global system would behave at its optimal configuration under various scenarios. The production and consumption parameters for the baseline model are from 2014. The global crude system I developed consists of 62 countries, accounting for 4,000 million metric tons of crude, or approximately 95% of total 2014 production. As a proxy for both price and greenhouse gas emissions, I used API gravity which is a measure of density and is often taken as an indicator of crude quality. All data used to develop the trade network was aggregated from 2014 data whenever possible. Exceptions to this are indicated throughout the methods section.

5.3.1 Current Trade Network

In order to quantify the potential for cost and emissions savings, I first needed to characterize the existing trade network. The volume of crude traded from each country to each other country are known and are compiled in existing, proprietary datasets; however, these datasets are not publicly available and are expensive to obtain[22]. Some of these data are published in aggregate form. For example, the BP Statistical Review contains trade data of totaled crude and petroleum products aggregated by region[282], while other datasets such as the JODI-Oil Database publish imports and exports aggregated by country rather than in trade partner format showing the volume of crude exported from a specific country to another[292]. Other datasets have a high resolution of detail, but are only available for limited countries. For example, the EIA maintains a detailed database of crude imports to the United States by country[293]. A frequently used publicly available data source for international trade analysis is the United Nations database[294]. While comprehensive, these data are self-reported and therefore may be subject to reporting errors, missing data, or inconsistencies from year to year. As an example of a shortcoming of this dataset, the flows are not balanced; total imports do not necessarily equal total exports; in 2014 petroleum imports totaled 14 billion bbls while imports totaled 8.7 billion bbls, a difference of ~40% (assuming a specific gravity of .88, or 140 kg per bbl of crude). A summary of the crude trade data sources used in this analysis is outlined in Table 23.

Table 23: Summary of relevant crude trade data used in this analysis

Data Source	Advantages	Disadvantages
BP Statistical Review[282]	Thorough, compiled from multiple sources, balanced	Aggregated by regions, combines crude and products in trade matrix, does not include trade within regions
UN Comtrade[294]	Reported by specific trade partner pairs	Self reported data, reporters are inconsistent across years, imports do not match exports
JODI database[292]	Country level detail	Imports and Exports are aggregated by country
IEA OECD matrix[295]	Crude only; country specific trade partners	Does not include non-OECD imports
Reuters vessel data[121]	Comprehensive, port to port data	Does not include crude traded by pipeline

The first objective of this study was to aggregate these various data sources in order to characterize the global crude system, including production, imports, exports, and domestic consumption. By taking advantage of the positive attributes of each dataset, I was able to realistically mirror global trade. The 62 countries I included were chosen based on the availability and consistency of data for these countries across all data sets. While there is no clear method for validating this matrix without access to proprietary data, I did my best to ensure the system balanced and that the trade ratios (i.e. net importers and net exporters) were accurate through an iterative smoothing process implemented using linear optimization.

To develop this global crude matrix, I first aggregated the UN Comtrade data with two other obtained data sources. From the Comtrade data, I included exports from 2014 reported under the HS commodity code 2709 (Petroleum oils, oils from bituminous minerals, crude). The second dataset is the IEA's trade balance of OECD countries. This includes import/export data from 2014 for 34 OECD countries, importing from 61 countries and exporting to 30 countries. The third dataset is of port-to-port oil tanker movements in 2015 obtained from Reuters[121]. This dataset includes approximately 25,000 vessel movements, with vessel capacities ranging from 56 thousand bbls to 2 million bbls. In aggregating these datasets, I found some trade partners were reported only in one of the datasets, while others were reported in multiple datasets. Where a trade partner pair was reported in one dataset only, I used the reported quantity as the trade value. Where a trade partner pair was reported in two or three of the datasets, I used the maximum of the reported values. This was to account for the fact that some of the data was self-reported and therefore might be biased towards underreporting trade values.

In addition to international trade, I added domestic consumption of a country's own production to the trade matrix. To do this, I obtained 2014 production and consumption data by country from the BP Statistical Review of World Energy[282], and filled in missing country production and consumption data from the JODI-Oil database[292]. I assumed a country's production minus its exports represented the domestic consumption of domestically produced oil. Therefore, the complete mass balance for any given country must be that annual production plus annual imports minus annual exports is equal to total consumption (refinery inputs). Because the compiled trade partner pair data described above did not balance with the production and consumption data, I followed an iterative process to "smooth" out the trade data, thereby producing a self-consistent production, trade, and consumption matrix. This smoothing was done using a linear optimization. The decision variables were the trade values between partner pair countries. The optimization was constrained such that the ratio of trade among trade partners should be maintained as closely as possible to the trade ratios from the original compiled trade matrix. The objective of the model was therefore to minimize the sum of the absolute value of the differences between the optimized trade ratios and the original trade ratios. Rather than minimize the differences evenly across the system, however, the differences were weighted by the country's contribution to supply and consumption such that larger consumers and producers were given higher importance in achieving correct trade ratios.

$$minimize\ \sum_{e=1}^{C}\left(\sum_{i=1}^{C}\left(abs\left(\underbrace{x_{ei}^{\mathrm{TR}}-\sum_{i=1}^{C}x_{ei}^{\mathrm{TR}}*r_{ei}^{\mathrm{EXP}}}_{\substack{Difference\ in\ trade\\ ratio\ across\ each\ country's\\ export\ trade\ partners}}\right)+abs\left(\underbrace{x_{ei}^{TR}-\sum_{e=1}^{C}x_{ei}^{\mathrm{TR}}*r_{ei}^{\mathrm{IMP}}}_{\substack{Difference\ in\ trade\\ ratio\ across\ each\ country's\\ import\ trade\ partners}}\right)\right)*w_e\right)\ (1)$$

with respect to

$$x_{ei}^{\mathrm{TR}} \in \mathbb{R}$$

$$\forall i \in \{1,\dots,C\}, e \in \{1,\dots,C\}, c \in \{1,\dots,C\}$$

subject to

$$p_c + \sum_{e=1}^{C} x_{ec}^{\mathrm{TR}} - \sum_{i=1}^{C} x_{ci}^{\mathrm{TR}} = u_c\ \forall c \quad (2)$$

$$p_c - \sum_{i=1}^{C} x_{ci}^{\mathrm{TR}} \geq 0\ \forall c \quad (3)$$

where x_{ei}^{TR} is the volume of crude traded from exporter e to importer i; r_{ei}^{EXP} is the designated proportion of exporter e's product that can be traded with partner country i; r_{ei}^{IMP} is the designated proportion of importer i's total imports that can be imported from partner country e; p_c is the designated production volume of a given country; u_c is the designated consumption volume of a given country; w_e is the weight of the exporting country's accuracy importance; C is the total number of countries in the trade network.

Equation (2) ensures each country's trade pattern balances such that its production plus its imports minus its exports is equal to its total consumption. Equation (3) ensures no country exports more crude than it produces. This assumes that no re-exports of imported product are allowed in the system (re-exports were < .06% of total exports in 2014 according to the UN Comtrade Database).

5.3.2 Cost and GHG Emissions Associated with Trade

5.3.2.1 API estimates

In this model, both cost and emissions are based on the API of crude as a proxy for crude quality. The API gravity is a measure of a crude's density relative to water, and can vary from <27° (heavy crude) to >50° (very light crude). Even within a given oil field, crude API can vary as a function of location and/or age of field production. Different crude qualities can be blended together to create a blended API. Refineries are able to accept a narrow range of blended crude APIs. Based on historical time series data showing average input blended APIs by PADD region for the United States between 1985 through 2015, this range tends to be ~3-7° (Figure 37).

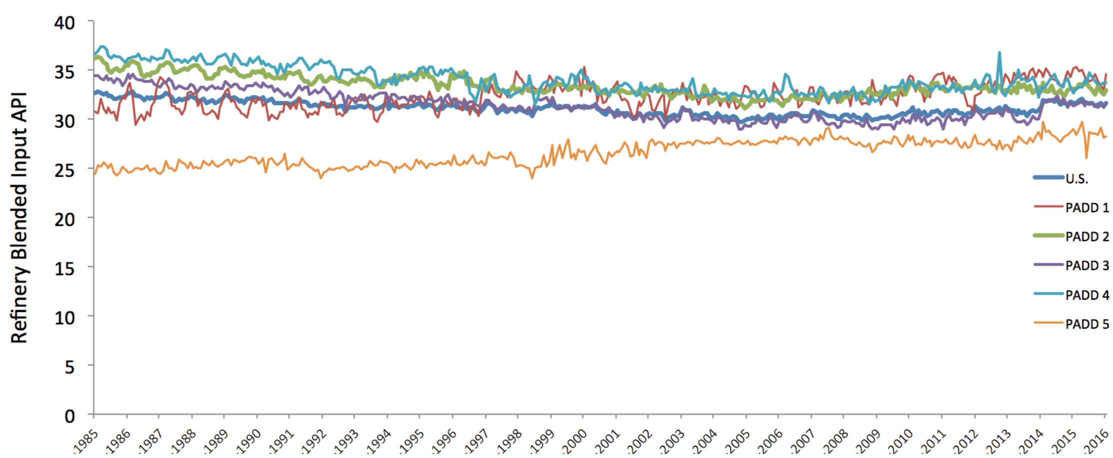


Figure 37: Blended refinery input API for the US and for PADD regions between 1985-2015

To address the importance of crude quality on price and emissions, I compiled a comprehensive database of crude APIs produced within each country. This data was aggregated from the Knovel Crude Oil Assay Database[296], a crude oil life cycle assessment conducted by the California Air Resources Board (CARB) which analyzed all crudes imported to California[297], a report by the

International Council on Clean Transportation which compiled data on all crudes combusted within the European Union[298], and estimates from the International Energy Agency (IEA)[299]. Each of these studies reported several API gravities of crudes from different countries. To determine the average API gravity of crude produced by each country as input to our model, I took the average of all crudes within a country in the aggregated dataset. Where a country was not represented within either of these three datasets, I used the average regional API estimated by the IEA[299].

5.3.2.2 Shipping Costs

To estimate the baseline 2014 total crude network cost, I compiled cost data from fourteen benchmark crudes, ranging in API from 13 to 38 from Deloitte's 2015 Oil and Gas Price Forecast report[300]. I use a line of best fit to represent the relationship between API and price. The line is fit as a differential between a crude's API/price and that of Brent crude such that any price can be inputted for Brent to determine crude price as a function of API for the other crudes. Since this study is based on 2014 data, the 2014 average price for Brent crude oil was used as the baseline.

In this model, shipping costs are estimating based on shipping distance and a oil tanker freight rate of $.004/ton-mile. This estimate is based on the Worldscale shipping rate (WS), which is widely used as the basis for shipping price negotiations between a ship owner and a charterer. The Worldscale rate is an annual reference rate based on the weighted average shipping distances, average bunker fuel price, port prices, etc for a standard vessel including four days of port time[301]. For example, in January-May of 2013, the WS varied between 36-43 $/ton for the route between West Africa and China[302]. Assuming an average WS of 40, and a distance of 10,600 miles[270], this equates to $.0037/ton-mile. The order of magnitude of this estimate is validated by several other sources[303, 304]. In reality, these costs would vary by ship size, fuel price, vessel demand, and load/discharge port. Shipping distances were modeled by the great circle distance based on a geographically representative subset of port locations from the World Port Index (WPI)[305] (Figure 38). A circuity factor of 1.3 was applied to account for variations from the great circle path[306]. In future work, all ports could be included in the analysis and maritime shipping distances could be used. However, for this exploratory analysis, given shipping emissions are approximately 3% of total cost and 2% of total emissions, these distance estimates are reasonable.

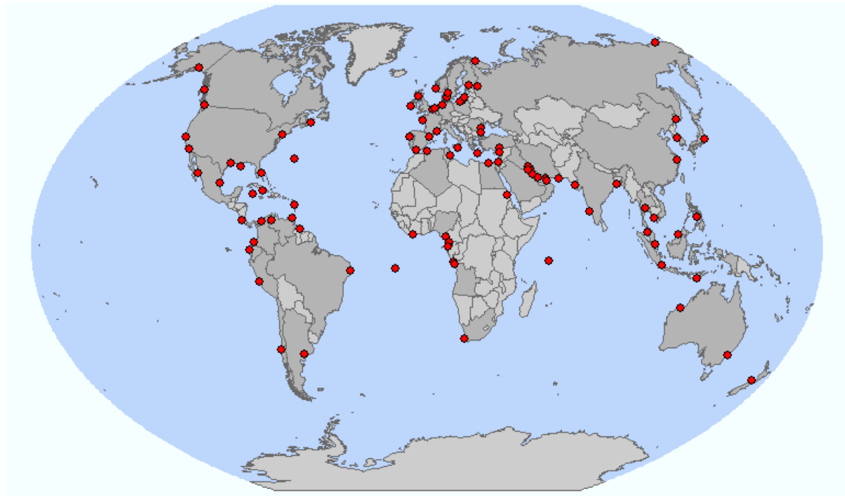


Figure 38: A) Selected ports from the World Port Index

5.3.2.3 Crude Life Cycle Emissions

Crude emissions can be quantified according to life cycle stages. In this model, I include upstream (extraction), shipping, midstream (refining), and downstream (combustion) emissions. The relationship between crude and the emissions from these life cycle stages is complex and driven by several different factors such as extraction technique, vent-to-flare ratio, fugitive emissions, refinery configuration, finished product mix, etc. These are explored in detail by the Oil-Climate Index report[307], which uses the Oil Production Greenhouse gas Emissions Estimator (OPGEE)[308] model to estimate oil field specific extraction emissions and the Petroleum Refinery Life Cycle Inventory Model (PRELIM)[309] to estimate midstream emissions. While API alone does not determine life cycle emissions of a given crude, as demonstrated by the Oil-Climate Index report[307], it is highly correlated with life cycle emissions. Therefore, in this study I use API as a proxy for crude-specific emissions from each life cycle stage, based on a linear fit of the results in the report. The exception to this is for upstream emissions, where the linear interpolation for points above the API range used to generate the line would result in negative emissions for very light crudes. To adjust for this, I assumed any crudes with API over 45 would have the same upstream emissions factor, and revised the best fit line accordingly. While the report is specific to United States default refinery configurations, product mix, and transportation end use, I take these emissions to be generally representative of global midstream and downstream emissions in absence of country specific additional data. Shipping emissions in this study are quantified based on the port-to-port distances described above and shipping emissions of .005 kg/t-km[122]. This emissions factor was a bottom up estimate based on crude vessel sizes and the portion of time spent idling at port or otherwise operating in inefficient modes. This value was validated by estimating a shipping emissions factor; assuming residual fuel oil is used with an energy intensity of 14 KJ/ton-km (from GREET[310]) and emissions factor of 77.4 t CO_2/TJ consistent with the assumption used by the IPCC[42], the calculated shipping emissions factor is .00108 kg CO2-e/t-km. This is reasonable because it is on the same order of magnitude, and slightly lower as expected due to operational factors and vessel capacity not being taken into account. While the emissions factor is treated deterministically in this model, future work could treat it stochastically to capture the uncertainty in emissions as well as the variability in ship size and fraction of the voyage spent in each engine operating mode.

5.3.3 Optimized Trade Network

The linear models representing the relationship between crude API and cost/emissions can be used both to characterize the existing global system, as well as to assess the changes in cost/emissions resulting from variations in the global crude system. Because cost and emissions are presented as a function of API and distance, any different pattern of trade can be analyzed as long as the average API blend being produced by each country is known.

After characterizing the existing global crude system, the next step is to explore the potential that exists within the network to reduce cost and greenhouse gas emissions. To do this I developed a second linear optimization model to compute optimal trade partner pairs while meeting global crude consumption. The basic model is described below:

$$\text{Minimize } \sum_{i=1}^{C}\left(\underbrace{\sum_{e=1}^{C} c_e^{\mathrm{CR}} x_{ei}^{\mathrm{TR}}}_{\text{cost of crude}} + \underbrace{\sum_{e=1}^{C} c^{\mathrm{SH}}\, d_{ei} x_{ei}^{\mathrm{TR}}}_{\text{cost of shipping}}\right) \quad (4)$$

with respect to

$x_{ei}^{\mathrm{TR}} \in \mathbb{R}$

$\forall i \in \{1, \dots, C\}, e \in \{1, \dots, C\}$

subject to

$\sum_{e=1}^{C} x_{ei}^{\mathrm{TR}} = u_i \ \ \forall i$

(5)

optional supply constraint:

(6)

$\sum_{i=1}^{C} x_{ei}^{\mathrm{TR}} = p_e \ \ \forall e$

optional API blend constraint:

$\sum_{i=e}^{C} API_e x_{ei}^{\mathrm{TR}} = \left(\sum_{e=1}^{C} x_{ei}^{\mathrm{TR}}\right) API_i \ \ \forall i$

(7)

where:

$c_e^{\mathrm{CR}} = \ (API_e * 0.87 \ + \ 66) * .0000073$ (\$/million mt)

$c^{\mathrm{SH}} = 1.8$ (\$B/(million mt – million km))

API_e is the average API produced in a given country

API_i is the target weighted average API consumed by a given country

d_{ei} is the distance between exporter e and importer i

This optimization model can be adjusted in several different ways to address a series of research questions: First, I look at a scenario where production and consumption are both constrained. This represents how trade partner pairs can be redistributed to reduce global GHG emissions and total system cost without any additional changes to the system. This supply and demand-constrained model is then compared to a model without supply constraints, which identifies the theoretical optimal crude producers to meet current crude demand. Both of these models additionally can be run requiring the weighted average crude mix consumed by each country to be equivalent to that of the current trade network, or allowing the crude mix to vary freely. If a country's crude API blend changes beyond what can currently be processed by that country's refinery portfolio, that country would need to invest in refinery updates to reconfigure the refinery to accept the new crude blend.

To evaluate the cost of such an investment, I used the Nelson Complexity Index (NCI)[311]. The NCI is a relative measure of the construction cost for a unit to upgrade a refinery compared to the cost of a distillation unit. While the complexity of a refinery does not specifically determine the API of the crude it can accept as input, the two parameters are highly correlated (Figure 45A). I use a comparison of NCI and average crude blend for refineries across the United States[312] to develop a linear model describing NCI as a function of API. Using this linear interpolation, I characterize the current average refinery by country based on the weighted average API blend consumed given the current trade matrix, and the optimal blend given the optimized trade matrix. Traditionally, a formula has been used to estimate the valuation of a refinery based on its complexity. This formula is a refinery's NCI multiplied by its capacity (bbls/day) times a valuation factor of $300 per bbl/day-complexity[313]. Therefore, to estimate the cost of investments required by each country to accept the optimal crude blend, I converted annual consumption to daily consumption, and multiplied it by the difference in NCI and the valuation factor. Valuation of a refinery is typically a third of the cost of constructing a new unit, but varies as a function of capacity. Therefore, this cost calculation is an underestimate of the actual refinery investments.

5.3.4 Climate Policy Scenarios

In addition to estimating the potential for cost and greenhouse gas savings within the 2014 crude system assumptions, I can use this optimization model to characterize the dynamics within the

system resulting from various climate policies. For example, I can implement a carbon tax across the system and set the objective to minimize total cost, consisting of crude cost, shipping cost, and GHG emissions times the carbon tax. This multi-objective optimization allows for the development of a Pareto frontier describing the tradeoffs in cost and emissions. As a starting point, I set the carbon tax at \$13/mt CO_2-e, which is consistent with the 2015 social cost of carbon (\$2015, 5% discount rate) as estimated by the Interagency Working Group on the Social Cost of Carbon[256] and vary it parametrically up to \$300/mt.
Additionally, I can set a global carbon cap across the system. To do this, I first determined each country's allocated portion of total emissions based on the estimated 2020 GHG targets as determined by each country's nationally determined contribution (NDC) resulting from the Conference of the Parties (COP) 21 in Paris in 2015[314]. If no 2020 or 2025 estimate was present in the data for a given country, I instead used 2012 emissions value reported by the World Resources Institute (WRI)[315]. These allocations indicate that if a carbon budget exists restricting emissions, each country in turn would have to restrict emissions based on its proportional 2020 emissions targets. For example, if a country's NDC target results in it emitting 10% of 2020 emissions, it would be allocated 10% of any future global carbon budget. Currently, the transportation sector contributes 14% of global emissions[316]. Assuming, this portion of total emissions approximately represents crude's contribution to global emissions, I attribute 14% of any global carbon budget as the carbon budget specific to the crude trade. For example, given a global carbon budget of 21 Gt CO_2-eq, the associated crude carbon budget would be 3 Gt CO_2-eq.

For both of these climate policy optimization scenarios, I can implement different carbon accounting strategies in order to explore the underlying dynamics of carbon leakage and embodied carbon. For example, in one carbon accounting strategy, the upstream producer was held accountable for upstream emissions, while the consumer was held responsible for the shipping, midstream, and downstream emissions. This assumes all refined products are used in that country, rather than being exported elsewhere. A second carbon accounting strategy was implemented in which the consumer was held responsible for all embodied emissions, including upstream extraction emissions. Finally, in a third carbon accounting strategy, upstream producers were held responsible for all embodied emissions throughout the crude life cycle. The climate policy scenario modifications to the optimization are as follows:

Optional carbon budget constraints:

$$\sum_i^C \sum_e^C x_{ie}^{\mathrm{TR}} * (GHG_e^{\mathrm{UP}} + GHG^{\mathrm{SH}} d_{ie} + GHG_e^{\mathrm{MID}} + GHG_e^{\mathrm{DWN}}) = GHG^{\mathrm{TB}} \quad (8)$$

- Scenario 1 (location based):

$$\sum_i^C GHG_c^{\mathrm{UP}} x_{ic}^{\mathrm{TR}} + \sum_e^C x_{ce}^{\mathrm{TR}} * (GHG^{\mathrm{SH}} d_{ce} + GHG_e^{\mathrm{MID}} + GHG_e^{\mathrm{DWN}}) = GHG_c^{\mathrm{B}} G * GHG^{\mathrm{TB}} \; \forall c \quad (9)$$

- Scenario 2 (producer based):

$$\sum_i^C x_{ic}^{\mathrm{TR}} * (GHG_c^{\mathrm{UP}} + GHG^{\mathrm{SH}} d_{ic} + GHG_c^{\mathrm{MID}} + GHG_c^{\mathrm{DWN}}) = GHG_c^{\mathrm{B}} G * GHG^{\mathrm{TB}} \; \forall c \quad (10)$$

- Scenario 3 (consumer based):

$$\sum_e^C x_{ce}^{\mathrm{TR}} * (GHG_e^{\mathrm{UP}} + GHG^{\mathrm{SH}} d_{ce} + GHG_e^{\mathrm{MID}} + GHG_e^{\mathrm{DWN}}) = GHG_c^{\mathrm{B}} G * GHG^{\mathrm{TB}} \; \forall c \quad (11)$$

where:

$GHG_e^{\mathrm{UP}} = (-3.0 * \mathrm{API}_e + 180) * (.0073)$ (million mt CO_2/million mt crude)

$GHG_e^{\mathrm{MID}} = (-1.4 * \mathrm{API}_e + 81) * (.0073)$ (million mt CO_2/million mt crude)

$GHG_e^{\mathrm{DWN}} = (-3.3 * API_e + 544) * (.0073)$ (million mt CO_2/million mt crude)

GHG^{TB} is the total global carbon budget

GHG_c^{B} is the fraction of total GHG emissions allocated to country c

$GHG^{\mathrm{SH}} = 16$ (million mt CO_2/(million mt – million km))

If the carbon constraints bound the problem such that meeting demand x_i^{CS} is infeasible, instead of minimizing total cost, I minimize the difference between the calculated consumption and the target consumption x_i^{CS}:

$$\text{Minimize } \sum_{i=1}^{C} \left(abs \left(\underbrace{\sum_{e=1}^{C} x_{ei}^{\mathrm{TR}}}_{\substack{\text{calculated} \\ \text{consumption}}} - \underbrace{x_i^{CS}}_{\substack{\text{target} \\ \text{consumption}}} \right) \right) \quad (12)$$

5.4 Results

5.4.1 Balanced 2014 Trade Matrix

The baseline 2014 global crude system developed in this analysis includes 62 countries representing 95% of total crude production. In this trade matrix, the sum of each country's production and imports equals its exports plus consumption (Figure 39). From these balances, it is clear which countries are net importers, net exporters, or largely domestic consumers of their own crude production.

The trade pathways showing the spatial distribution of trade can be seen in Figure 40. This figure was developed in ArcGIS using the estimated 2014 trade matrix. The figure shows imports/exports but does not indicate consumption of crude produced domestically. In the figure, the flow of the crude is represented by colored lines, indicating very light flows (green) to high volume flows (red).

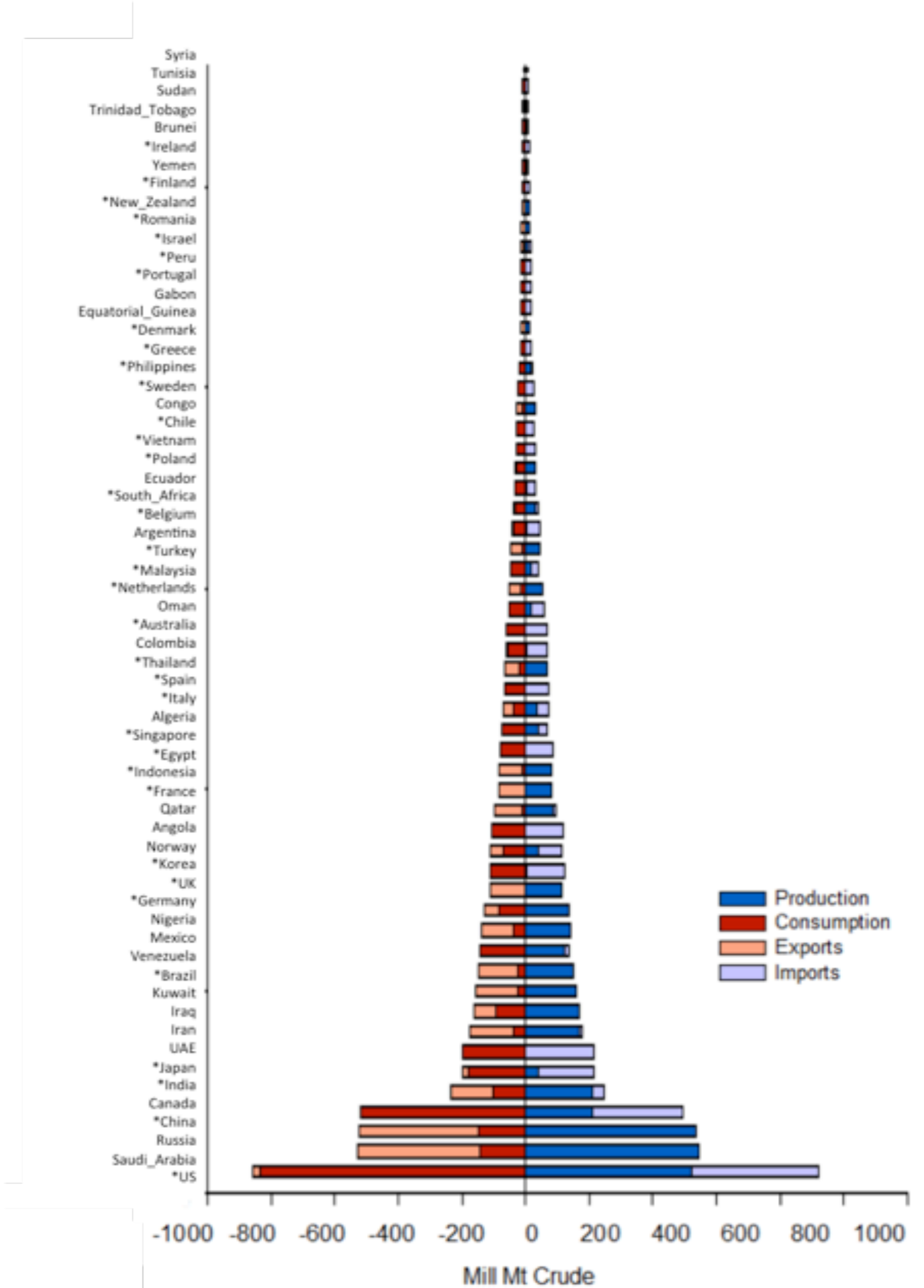


Figure 39: Production, consumption, and trade mass balance by country. Net importers are indicated with a *.

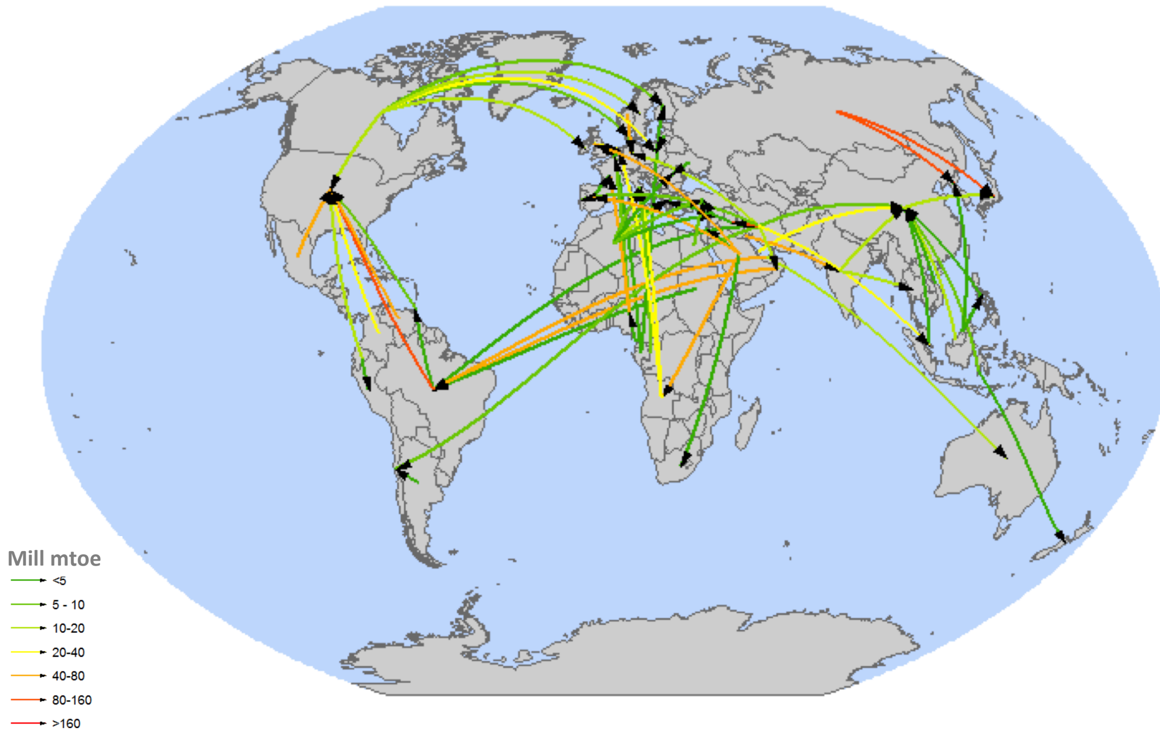


Figure 40: Global crude trade flows in 2014

5.4.2 Emissions Estimates

The upstream, midstream, and downstream emissions for a given crude is estimated based on the Oil-Climate Index (OCI) report[307]. While API alone does not determine life cycle emissions of a given crude, it is generally correlated with life cycle emissions. Therefore, in this study I use API as a proxy for crude-specific emissions from each life cycle stage, based on a linear fit of the results in the report. The upstream, midstream, and downstream emissions were estimated by linear fits to the results of the Oil-Climate Index report as shown in equations (13)-(15)[307].

$$GHG^{\mathrm{UP}} = -3.9 \times API + 186 \qquad (13)$$
$$GHG^{\mathrm{MID}} = -1.4 \times API + 81 \qquad (14)$$
$$GHG^{\mathrm{DWN}} = -4.4 \times API + 600 \qquad (15)$$

The emissions associated with these trade flows were calculated according to the three different carbon accounting strategies and can be seen in (Figure 41). Figure 41A shows the emissions by country given producers are responsible for extraction emissions and consumers are responsible for all other life cycle emissions. In contrast, Figure 41B shows embodied life cycle carbon emissions being attributed to the upstream producer, while Figure 41C shows embodied life cycle emissions being attributed to the downstream consumer. Figure 42 shows the relative impact of each of these carbon accounting methods for each country. In this figure, if the three accounting measures each have a third of the total (see Argentina for example), then that country's emissions are not sensitive to the accounting strategy used. If instead, however, one color dominates the bar, that country is highly sensitive to the carbon accounting method. For example, using a production-based carbon accounting strategy would greatly affect Yemen's emissions profile. Due to the asymmetric impact of carbon accounting methods on different countries, any global carbon accounting policy should carefully consider equity issues such as economic impact to developing countries.

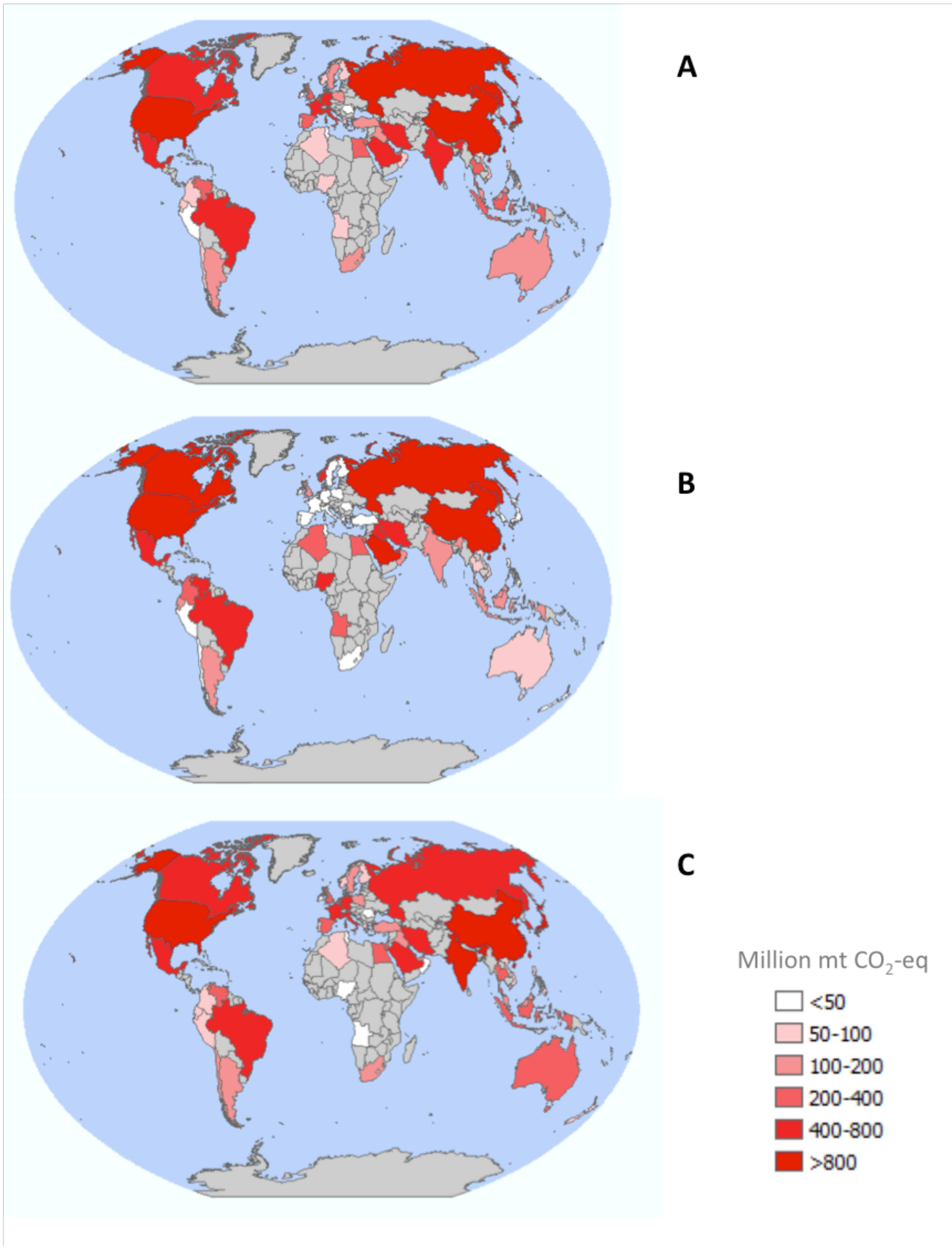


Figure 41: Greenhouse gas emissions from crude trade by country in 2014 assuming A) upstream emissions are attributed to producer, all other emissions to consumer, B) all life cycle emissions are attributed to the upstream producer, and C) all life cycle emissions are attributed to downstream consumer

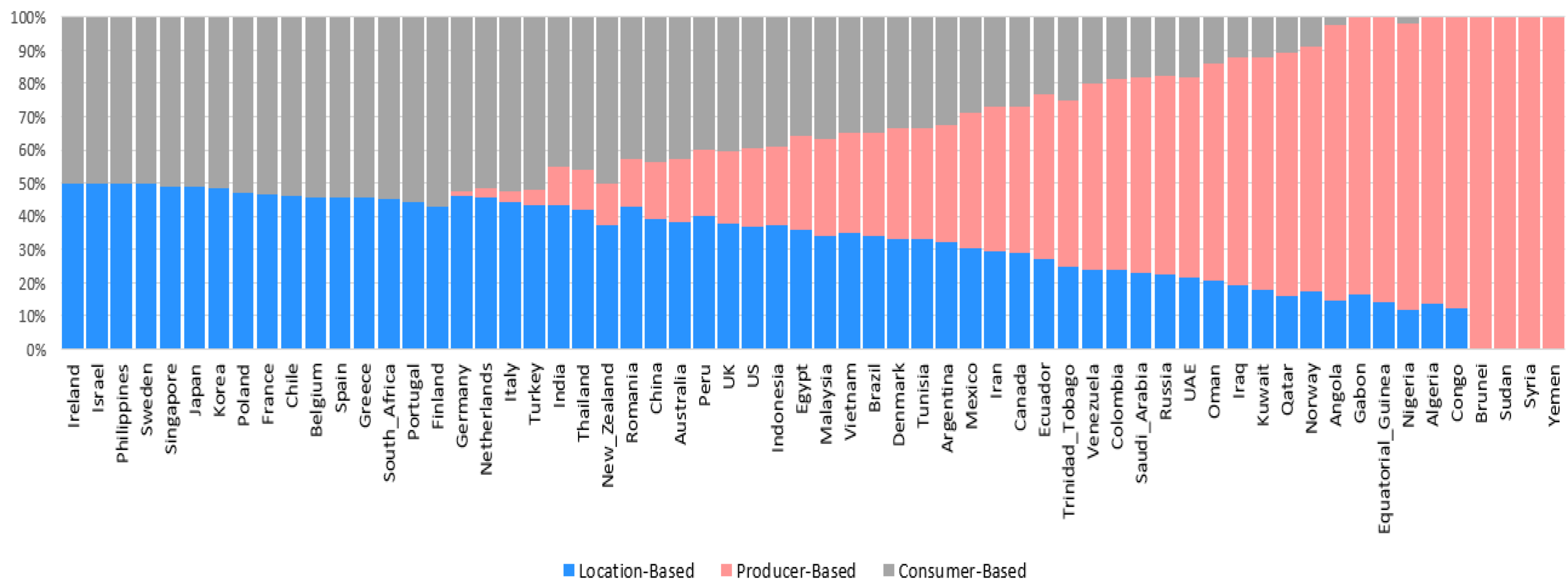
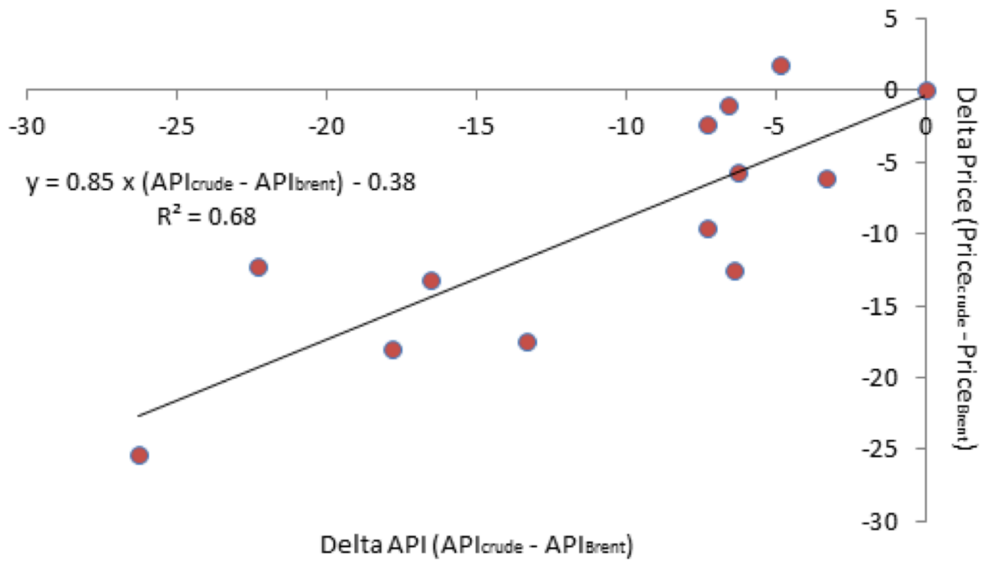


Figure 42: Relative impact of location, production, and consumption based carbon accounting methods on country emissions

5.4.3 Cost Estimates

The linear cost model representing crude cost as a function of crude API was developed from thirteen benchmark crudes (Figure 43). It is expressed as a differential from Brent (API of 38). In this analysis, because 2014 is the baseline year, I use the 2014 average price of Brent ($99/bbl)[300].



I use these costs, combined with the shipping cost as a function of distance and quantity shipped, to estimate cost of each country's consumption for the 2014 trade network. I then compare this to the optimized network (Table 24).

When minimizing by cost, the least cost scenario is to allow any country to produce an unconstrained volume of crude. This allows the cost to drop from $3T to $2.4T ($590/mt consumed), and also results in GHG increase of approximately 4.5 Gt CO_2. Allowing supply to vary freely but constraining the weighted average consumed API gravity by country to remain the same as in the baseline 2014 network also decreases costs by ~$230B across the system to $690/mt crude consumed.

Table 24: Summary of scenario results. Cost = cost minimized, demand constraints only; Cost_SL = cost minimized, supply and demand constrained; Cost_API = cost minimized, demand and API constrained; GHG = GHG minimized; GHG_SL = GHG minimized, supply and demand constrained; GHG_API = GHG minimized, demand and API constrained

		Baseline	Minimize Cost			Minimize Emissions (SCC)		
			Cost	*Cost_SL*	*Cost_API*	*GHG*	*GHG_SL*	*GHG_API*
Cost ($T)	Shipping	0.14	0.07	0.02	0.00	0.09	0.02	0.01
	Crude	2.8	2.3	2.73	2.77	3.35	2.73	2.86
	Total	3.0	2.4	2.75	2.77	3.46	2.75	2.86
GHG (Gt CO2-e)	Shipping	0.3	0.15	0.04	0.01	.19	0.04	0.01
	Crude	16.1	20.31	16.4	16.08	10.9	16.4	15.30
	Total	16.5	20.46	16.43	16.08	11.09	16.4	15.31
Consumption (mill mt)		4000	4000	4000	4000	4000	4000	4000
$/mt consumed		750	590	688	693	865	688	715
GHG/mt consumed		4.0	5.1	4.1	4.0	2.8	4.1	3.8

As seen in Figure 44, some countries realized cost savings as a result of the optimization, while some countries experienced loses (see Table 26 for additional detailed cost results). This is primarily a result of shifting crude consumption rather than shipping savings; shipping savings costs are nominal relative to crude savings for a given country. As an example of a country with a high potential for reducing cost, the United States, has the potential to save on the order of a hundred billion dollars. These savings arise from consuming domestically produced crude only in the optimized model, rather than importing crude.

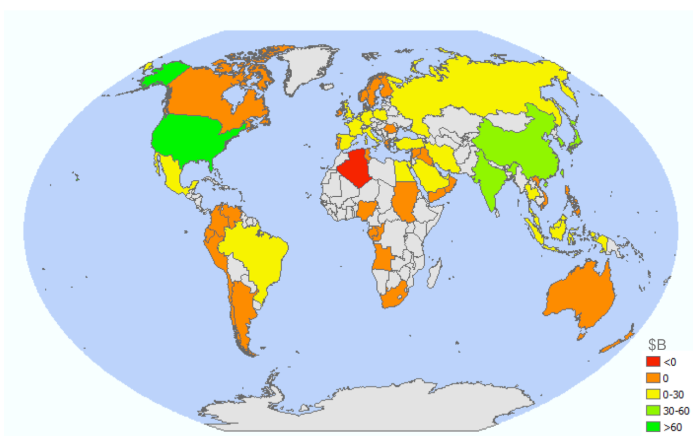


Figure 44: Cost savings from the optimized network over the 2014 network, where demand was constrained to 2014 levels, but supply by country and consumed blended API were allowed to vary freely

However, the savings from the optimized network would not be realized for some countries that might need to invest in refinery upgrades in order to accommodate heavier crudes. The degree to which countries must invest in their refinery infrastructure varies significantly, and depends on the weighted average blended API designated by the optimized network as compared to the

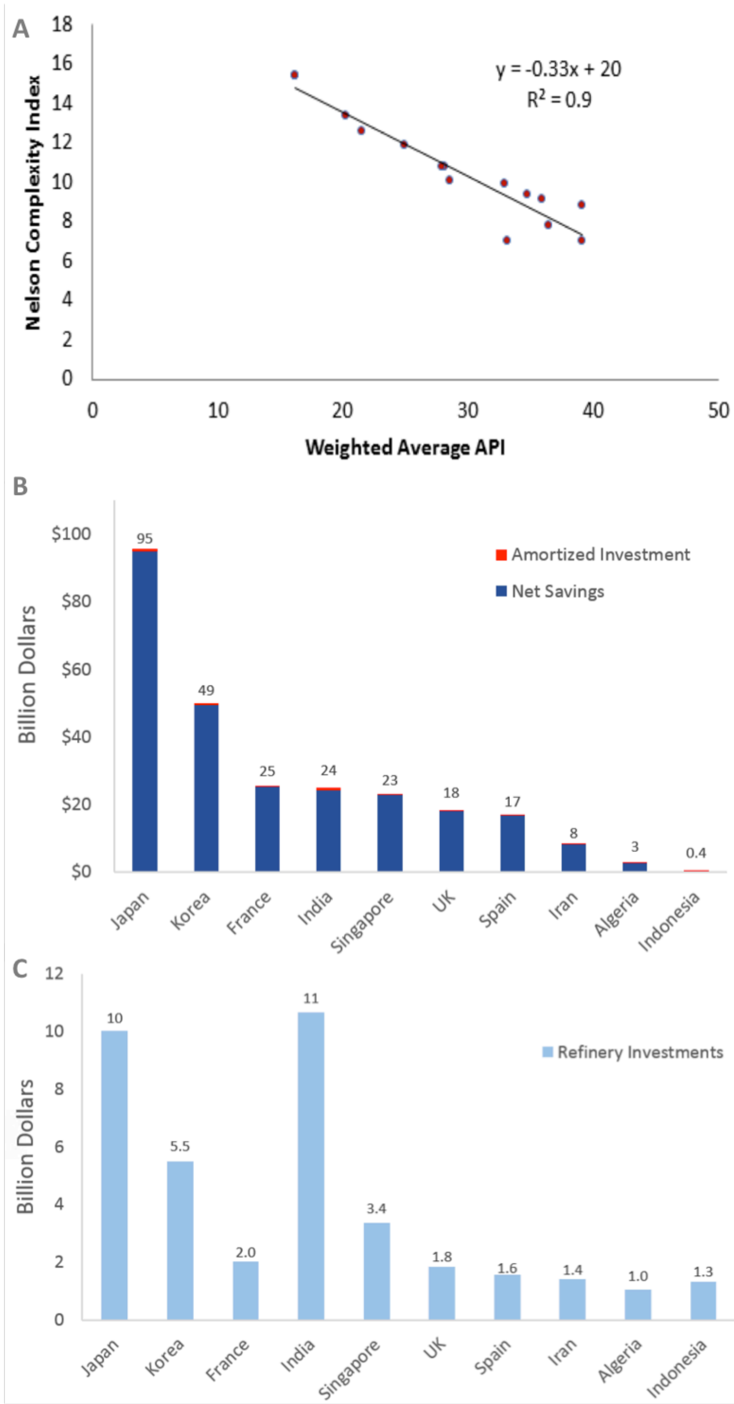


Figure 45: A) NCI as a function of API, B) top ten net cost saving countries with refinery investments at construction costs equal to the refinery valuation amortized over 20 years at 5% interest rate, and C) total refinery investment construction costs

baseline blended API (Figure 45A). Figure 45B shows the top ten cost saving countries, net of refinery investments amortized over 20 years at a 5% interest rate. This assumes the refinery valuation determined by the change in NCI is equal to the construction cost (Figure 45). If the construction cost is double or three times the cost of the valuation, some countries no longer realize a cost savings net of these investments.

When minimizing total GHG emissions across the system, there are again distinct differences in cost and emissions savings across the three scenarios. Allowing supply to diverge from 2014 production provides for global GHG savings of 6 Gt CO_2 for the same total system cost as the baseline. Using this model, in addition to minimizing based on the objectives of total cost or total greenhouse gas emissions, I also perform a multi-objective optimization to assess tradeoffs in carbon price versus emissions reductions. In this optimization, the objective function is the sum of monetized GHG emissions (weighted by the given social cost of carbon) and the total system cost (from crude purchases and shipping). Figure 46A shows a Pareto frontier of tradeoffs between total cost and GHG emissions. The extreme end points represent the cost minimized system (SCC = $0/mt) and the GHG minimized system, respectively. The points between these extremes illustrate the influence of parametrically varied SCC between $13/mt (2015 SCC, 5% discount rate) to $300/mt. This figure shows that there is a threshold between $110 and $120/mt where there is a significant emissions reduction and cost increase. Figure 47B shows the change in cost over the savings in GHG emissions. For lower SCCs that incentivize an emissions reduction, the GHG savings are expensive. For example, at a SCC of $66/mt, every Gt of CO_2 avoided costs $22T. However, as the SCC increases, the cost of avoided emissions decreases until it reaches the limit of the GHG minimized scenario ($8.7T/Gt).

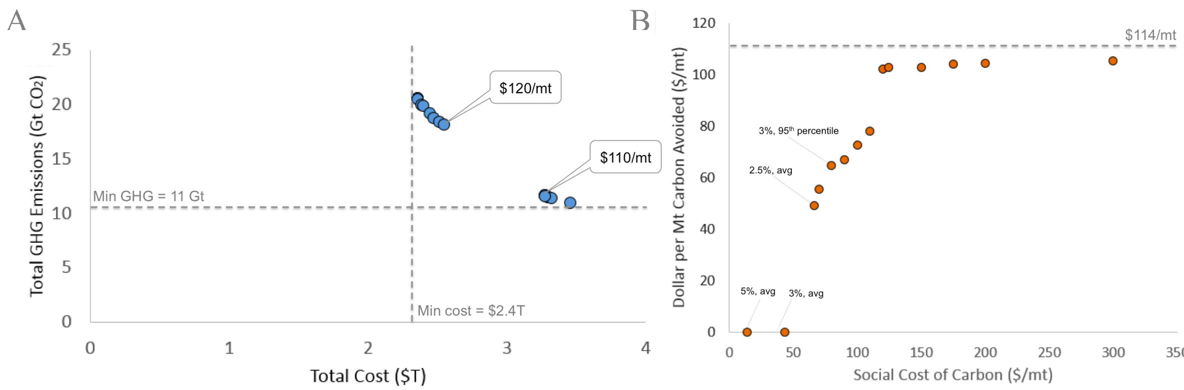


Figure 46: A) Pareto frontier between total cost and total GHG emissions, and B) dollars spent per mt CO_2 avoided, as a function of social cost of carbon

5.4.4 Climate Policy Scenarios

The optimized trade network under a global carbon budget is designed so that each country is allocated a portion of the global budget. This allocation is characterized by the NDCs resulting from COP21, suggesting that the ratio of emissions contributions will remain consistent with the ratio of committed 2020 COP21 targets in the future. The percentage of total carbon emissions allocated to each region is shown in Figure 47. These limits can either be set as absolute limits, or the model can be run such that a country pays a carbon tax for any emissions above its allocated budget. The latter formulation, with a global carbon budget and payments made to exceed individual country allocations, mimics a global cap and trade policy.

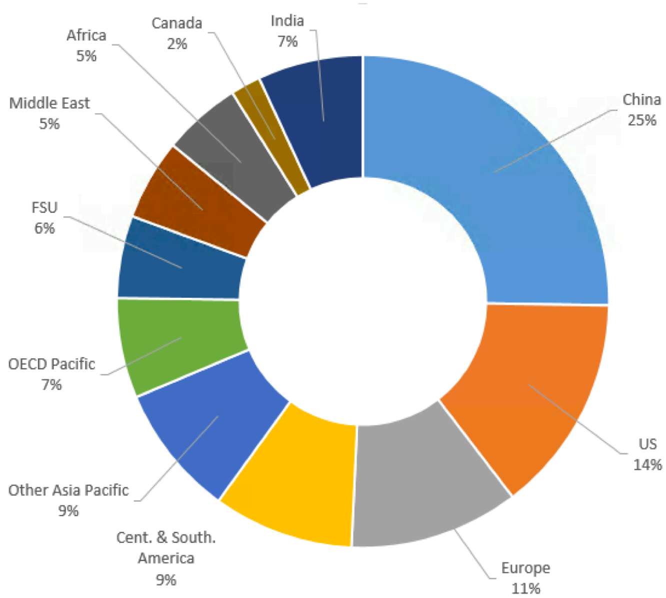


Figure 47: Proportion of carbon budget allocated by region according to NDCs from COP21

In addition to designating emissions allocations by country, the climate policy version of the model must also be specified with a carbon accounting strategy. As previously discussed, the three main accounting strategies are 1) location based, in which the upstream emissions are allocated to the producer and all other life cycle emissions are allocated to the consumer, 2) production based, in which all embodied life cycle emissions are allocated to the producer, and 3) consumption based, in which all allocated life cycle emissions are allocated to the consumer. In this model, each country tries to stay as close to its demand as possible while remaining within its carbon budget. The results of the optimization are summarized in Table 25, and detailed results can be found in Table 26 through Table 28. These results demonstrate that the trade matrix shifts dynamically depending on which accounting strategy is chosen. For example, Figure 48 shows how the crude producing countries shift under different scenarios. For reference, Figure 49 shows the average API produced by each country.

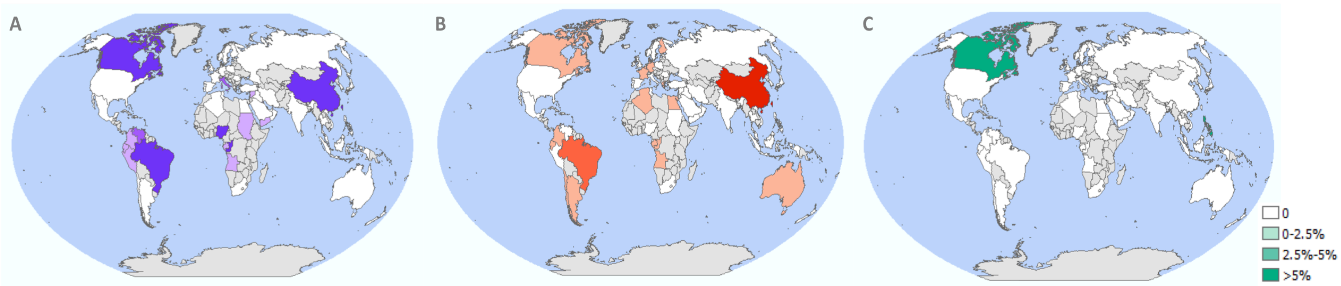


Figure 48: Crude producing countries by percentage of total production in A) location-based, B) producer-based, and C) consumer-based carbon accounting strategy under the 3GT global crude carbon cap.

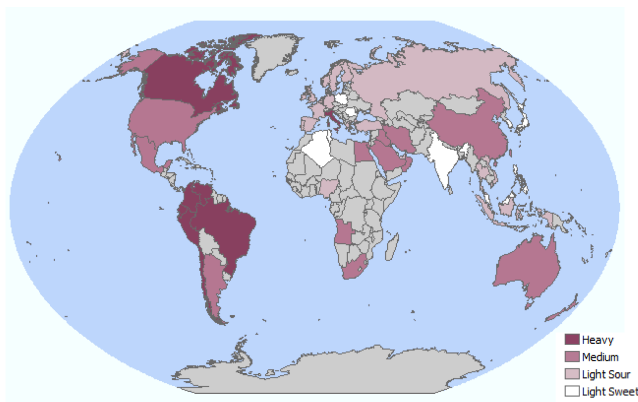


Figure 49: Produced crude average API gravity for each country

Table 25: Summary of optimization results under a 3 Gt global carbon budget. 3GT_L = location based carbon accounting; 3GT_P = production based carbon accounting; 3GT_C = consumption based carbon accounting

			Carbon Budget, Maximize Demand		
		Baseline	**3GT_L**	**3GT_P**	**3GT_C**
Cost ($T)	Shipping	0.14	0	0	0.02
	Crude	2.8	0.55	0.54	0.5
	Total	3.0	0.55	0.55	0.52
GHG (Gt CO2-e)	Shipping	0.3	0.01	0	0.05
	Crude	16.1	3.0	3	2.95
	Total	16.5	3.0	3	3
Consumption (mill mt)		4000	770	770	730
$/mt consumed		750	714	714	712
mt CO_2/mt consumed		4.0	3.9	3.9	4.1
$/mt CO_2		190	183	183	173

Regardless of the specific carbon accounting scenario, a strict 3 Gt carbon budget would constrain demand to at most 770 million mt per year (an 80% decrease over the 2014 baseline consumption) (Table 25). When considering specific accounting strategies, the different methods for carbon emissions attribution influences the total demand that can be satisfied while maintaining a global carbon budget of 3 Gt. For example, the location and production based models enable 770 million mt worth of demand to be met, while the consumer based methods only allows for 730 million mt. Because cost increases with API and emissions decrease with API, each country must balance its cost against its carbon budget. When consumers are only concerned with meeting demand and do not need to take shipping, midstream, and downstream emissions from their own consumption into account, they prioritize low cost crudes (high APIs) over lower emitting light crudes. Under a producer-based carbon accounting strategy, consuming countries are not incentivized to participate in mitigation efforts.

While consumption based accounting methods of embodied carbon are widely discussed as potential border carbon accounting measures, this model shows the ratio of total emissions to total consumed is 4.1 mt CO_2/mt consumed, which is higher than the 4.0 mt CO_2/mt ratio found for the baseline and 3.9 mt CO_2/mt for other two carbon accounting strategies. Therefore, consumption based accounting does not allow the global system to satisfy as much demand as a production based accounting method would. This strategy also results in a 10% higher carbon intensity.

All three of the scenarios discussed in the previous paragraph assume each country is allocated a specific fraction of the global carbon budget based on the ratio of COP21 NDC emissions targets for 2020. However, if this allocation constraint is relaxed, the total demand that can be satisfied increases to 1100 million mt for a cost of $930B ($850/mt consumed). Therefore, without unilateral carbon limits, the volume of demand satisfied increases by over 40%, for a lower cost per unit consumed than under the consumption based carbon accounting method with country specific carbon targets. This demonstrates there may be competing influences among the interactions between NDCs and carbon accounting strategies in the long run that could inhibit the cost effectiveness of climate change mitigation efforts. However, the results of the scenario without NDC based country specific carbon budgets shows the available crude is consumed by a limited number of countries; only ten countries receive crude, compared to 56 consuming product in the consumption based model. Therefore, while potentially resulting in a lower total volume of crude consumption across the system, unilateral carbon budgets acting within a global carbon budget could be used as a mechanism for promoting equity.

5.5 Model Limitations

The optimization model developed for this analysis is a single-industry model that assumes perfect foresight. As a result, it explores climate policy as though it were applied to a single industry only; it does not take into consideration the competition with other economic activities for factors of production nor does it consider substitution of crude for other products. Although substitution effects from direct crude consumption are minimal compared to substitution effects of coal versus natural gas for example, there are other industries that would be affected by such substitutions such as the petro-chemical sector can substitute petroleum-based inputs for natural gas based inputs. Additionally, because this is an optimization model, it represents the extremes of what is possible; it is not intended to replicate historical trade patterns nor predict future trade patterns.

Another limitation is that several simplifying assumptions were applied throughout the model. These assumptions include unlimited reserves by country, constant costs as a function of API only, constant shipping costs, and politically unrestrained trade interactions between countries. Additionally, the crude balance (production minus exports plus imports) does not take into account the trade of refined products, which would add an important degree of flexibility within the system. Although these assumptions serve as a departure from reality, they also enable the model to be straightforward, so the causal relationships between policy and shifting trade patterns can be readily interpreted. Finally, this model is based on deterministic estimates of life cycle crude emissions, shipping emissions factors, etc rather than stochastically considering the underlying uncertainty and variability intrinsic to these parameters.

5.6 Discussion

This exploratory analysis characterizes the current global crude system by aggregating multiple partial data sources and using linear optimization to ensure a mass balance across each country's production, imports, exports, and consumption. Using this network, I then quantify each country's emissions contributions according to three different carbon accounting strategies. These carbon accounting strategies could be used to issue border carbon adjustments (BCAs) or other policies targeted at avoiding carbon leakage. However, by quantifying emissions according to the three strategies (location based, producer based, and consumer based), different accounting practices impact countries differently. In particular, countries in Africa and Europe are most affected by the specified strategy because they tend to be largely net exporters and importers, respectively. Countries with a balance of crude exports and imports, such as the United States, China, Canada, and Russia do not experience meaningful shifts in emissions across the accounting scenarios. This suggests that the crude sector is unique from other trade sectors. Typically, the United States is characterized as an extreme net emissions importer, while China and Russia are net exporters[131]. However, in the case of global crude trade, emerging markets are not the primary exporters of embodied carbon. Therefore, carbon leakage from climate policies targeting crude based emissions may not be a significant threat to undermining carbon mitigation. This would suggest that BCAs may not be necessary, and in fact may asymmetrically harm developing countries in Africa depending on the accounting strategy implemented. There may be other sectors besides the petroleum sector where the traditional trend in emissions flows are not observed. Therefore, any BCA should not be applied unilaterally on imports or exports; instead it is important to first consider the unique network dynamics within any given sector to assess the potential for carbon leakage.

In addition to exploring the current petroleum trade network, in this analysis I developed an optimization to quantify the potential for cost savings and greenhouse gas mitigation within the framework of existing consumption patterns. From this optimization, some countries have the potential for large cost savings at the expense of other countries. However, in order to realize these savings, some countries would have to invest in refinery upgrades in order to accept heavier crude blends. These estimated investments can be up to $10B, although because the estimate is based on refinery valuation rather than construction cost, the real cost of investment could be up to three times higher. What is important to note, however, is that the trade network under the optimized 2014 scenario is not mirrored in the optimized trade network under a tight carbon budget. Therefore, private companies seeking to minimize their own costs in today's global crude system may be making infrastructure investments that pay off in the short term, but would be inefficient under a future carbon constrained system.

The future carbon constrained system depicted in this analysis is based on the carbon budget of 21 Gt CO_2-eq in 2050, which would likely maintain average global temperature rise below 2 °C temperature increase through 2020[139]. Assuming crude is associated with 14% of total annual emissions, this translates to a crude carbon budget of 3 Gt CO_2-eq. I found that to achieve this carbon budget, under an optimal trade network, consumption would have to decrease from about 33 billion bbls of crude to about 5.8 billion bbls of crude, a reduction of approximately 80%. Because this is an optimized system, and any price elasticity of demand, substitution effects, political trade preferences, etc would only serve to increase the percentage by which crude consumption must decrease to achieve this global carbon budget.

Finally, under such an extreme global carbon budget, the carbon accounting strategy used to track emissions would become influential in determining the structure of the trade network. In the case of crude trade, where emissions decrease as a function of API while price increases as a function of API, countries face tradeoffs in reducing their cost and reducing their emissions. Therefore, I find that a production based carbon accounting is inefficient; countries with spare allocation in their carbon budgets will produce heavy crude for other countries to buy cheaply, while countries with light, lower emissions crudes will not produce at all. Conversely, given a consumption based carbon accounting strategy, consumers will prioritize emissions over cost in order to squeeze more consumption out of their limited carbon budgets. While I found that in today’s global crude system carbon accounting practices might be largely ineffective, they may prove to be a powerful tool in avoiding carbon leakage and ensuring the most effective mitigation measures in a severely carbon constrained future.

5.7 Appendix to Chapter 5

The following tables present detailed model results across all scenarios. The baseline is the 2014 trade network.

Cost = cost minimized, demand constraints only
Cost_supply = cost minimized, supply and demand constrained
Cost_API = cost minimized, demand and API constrained
GHG = GHG minimized
GHG_supply = GHG minimized, supply and demand constrained
GHG_API = GHG minimized, demand and API constrained
3GT = 3 Gt carbon cap, no NDC constraints
3GT_Location = location based carbon accounting, 3 GT carbon cap
3GT_Producer= production based carbon accounting, 3 GT carbon cap
3GT_Consumer = consumption based carbon accounting, 3 GT carbon cap

Table 26: Detailed cost results by country for each scenario ($B)

	Baseline	Cost	Cost supply	Cost API	GHG	GHG supply	GHG API	3GT	3GT location	3GT producer	3GT consumer
Algeria	0	12	16	16	18	16	17	0	3	0	4
Angola	1	1	1	1	2	1	1	0	1	0	2
Argentina	19	17	19	18	25	19	19	0	5	5	6
Australia	28	25	28	27	35	28	27	0	11	0	7
Bangladesh	0	0	0	0	0	0	0	0	0	0	0
Belgium	26	19	23	23	29	24	24	0	2	14	2
Brazil	86	81	87	85	122	87	87	0	9	29	25
Brunei	0.2	0.1	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.0	0.1
Bulgaria	0	0	0	0	0	0	0	0	0	0	0
Canada	67	65	65	65	99	65	66	0	0	0	10
Chile	10	9	10	10	13	10	10	0	1	0	1
China	369	316	366	342	446	366	350	446	156	114	104
Colombia	10	9	10	10	14	10	10	0	0	0	3
Congo	0	0	0	0	0	0	0	0	0	0	0
Denmark	7	5	6	6	8	6	6	0	1	6	1
Ecuador	8	8	8	8	12	8	8	0	0	0	2
Egypt	34	26	30	29	37	30	30	0	4	0	4
Equatorial_ Guinea	0	0	0	0	0	0	0	0	0	0	0
Finland	8	6	7	7	8	7	7	0	1	7	1
France	65	50	64	60	74	64	62	0	7	0	7
Gabon	0	0	0	0	0	0	0	0	0	0	0
Germany	99	72	82	86	107	82	89	0	12	13	14
Greece	13	9	11	11	14	11	11	0	1	11	2
India	168	121	143	160	171	143	170	0	60	59	44
Indonesia	49	40	52	47	56	52	48	56	28	37	22

Iran	69	62	69	68	89	69	71	0	10	14	7
Iraq	19	16	19	18	23	19	19	0	3	0	3
Ireland	6	4	5	5	6	5	5	0	1	0	1
Israel	10	7	8	8	10	8	9	0	1	0	2
Italy	45	37	43	38	55	43	38	0	0	0	6
Japan	190	129	155	174	184	155	184	150	24	13	19
Korea	103	68	81	96	100	81	100	100	11	82	9
Kuwait	16	15	16	16	21	16	16	0	2	0	2
Lithuania	0	0	0	0	0	0	0	0	0	0	0
Malaysia	30	24	29	29	33	29	30	33	8	0	5
Mexico	63	55	63	62	83	63	63	0	9	0	11
Netherlands	35	26	31	31	38	31	32	0	3	32	3
New_Zealand	6	5	5	5	7	5	5	0	1	0	1
Nigeria	2	2	2	2	3	2	2	0	2	2	3
Norway	9	7	8	8	10	8	8	0	1	4	1
Oman	7	6	8	7	9	7	7	0	2	3	1
Pakistan	0	0	0	0	0	0	0	0	0	0	0
Peru	6	6	6	6	8	6	6	0	0	3	2
Philippines	14	10	11	13	13	11	13	13	3	0	2
Poland	25	15	18	20	23	18	21	0	7	0	7
Portugal	9	7	10	8	11	10	9	0	1	0	1
Qatar	8	7	8	8	10	8	8	0	1	8	1
Romania	9	6	7	8	9	7	8	0	2	0	2
Russia	117	95	114	115	139	114	118	0	33	0	36
Saudi_Arabia	109	94	109	108	136	109	111	0	7	28	6
Singapore	65	45	53	59	62	54	62	62	1	8	1
South_Africa	19	16	18	18	23	18	18	0	6	0	5
Spain	50	38	47	46	58	47	48	0	5	0	5
Sudan	0	0	0	0	0	0	0	0	0	0	0
Sweden	13	9	11	11	14	11	12	0	1	0	1
Syria	0	0	0	0	0	0	0	0	0	0	0
Thailand	44	35	42	41	49	42	42	49	6	0	4
Trinidad_Tobago	1	1	1	1	2	1	1	0	1	1	1
Tunisia	2	1	2	2	2	2	2	0	0	0	1
Turkey	23	18	21	21	26	21	22	0	6	23	7
Ukraine	0	0	0	0	0	0	0	0	0	0	0
UAE	31	26	31	30	37	31	31	0	3	0	2
UK	59	45	55	54	67	55	56	0	7	29	9
US	616	498	576	587	754	577	605	0	74	0	92
Venezuela	25	25	25	25	38	25	25	0	0	0	5
Vietnam	15	13	15	14	18	15	15	18	4	0	3
Yemen	0	0	0	0	0	0	0	0	0	0	0

Table 27: Production by country for each modeled scenario (million metric tons)

	Baseline	Cost	Cost supply	Cost_API	GHG	GHG supply	GHG API	3GT	3GT location	3GT producer	3GT consumer
Algeria	46	0	66	0	0	66	0	0	0	4	0
Angola	83	0	83	0	0	83	0	0	10	3	0
Argentina	29	0	29	0	0	29	0	0	0	7	0
Australia	19	0	19	36	0	19	38	0	0	11	0
Bangladesh	0	0	0	168	0	0	312	0	91	5	0
Belgium	0	0	0	0	0	0	0	0	0	2	0
Brazil	122	0	122	111	0	122	130	0	89	27	0
Brunei	6	0	6	0	0	6	0	0	1	0	0
Bulgaria	0	0	0	0	0	0	0	0	19	1	0
Canada	211	3997	210	224	0	206	113	0	59	12	411
Chile	0	0	0	13	0	0	14	0	0	2	0
China	211	0	211	471	0	211	499	0	47	174	0
Colombia	52	0	53	14	0	53	15	0	18	3	0
Congo	15	0	15	1987	0	15	1600	0	40	6	0
Denmark	8	0	8	0	0	8	0	0	0	1	0
Ecuador	29	0	30	9	0	30	13	0	11	2	0
Egypt	35	0	35	0	0	35	0	0	0	5	0
Equatorial_Guinea	13	0	13	0	0	13	0	0	3	0	0
Finland	0	0	0	0	0	0	0	0	0	1	0
France	1	0	1	0	0	1	0	0	0	8	0
Gabon	12	0	12	0	0	12	0	0	1	0	0
Germany	2	0	2	0	0	2	0	0	0	16	0
Greece	0	0	0	0	0	0	0	0	0	2	0
India	42	0	39	0	0	42	0	0	0	72	0
Indonesia	41	0	41	0	0	41	0	0	0	36	0
Iran	169	0	169	0	0	169	0	0	0	12	0
Iraq	161	0	161	0	0	161	0	0	0	4	0
Ireland	0	0	0	0	0	0	0	0	0	1	0
Israel	0	0	0	0	0	0	0	0	0	2	0
Italy	6	0	6	57	0	6	60	0	37	7	0
Japan	0	0	0	0	0	0	0	0	0	32	0
Korea	0	0	0	63	0	0	89	0	0	17	0
Kuwait	151	0	151	19	0	151	23	0	4	3	0
Lithuania	0	0	0	0	0	0	0	0	8	1	0
Malaysia	30	0	30	0	0	30	0	0	0	8	0
Mexico	138	0	138	85	0	138	88	0	0	12	0
Netherlands	2	0	2	0	0	2	0	0	0	4	0
New_Zealand	2	0	2	7	0	2	8	0	0	1	0
Nigeria	113	0	113	0	0	113	0	0	44	8	0
Norway	86	0	86	0	0	86	0	0	0	1	0
Oman	46	0	46	0	0	46	0	0	0	2	0
Pakistan	0	0	0	478	0	0	352	0	164	8	0

Peru	5	0	5	10	0	5	9	0	13	2	0
Philippines	1	0	0	117	3997	1	584	1094	0	4	318
Poland	1	0	1	0	0	1	0	0	0	8	0
Portugal	0	0	0	0	0	0	0	0	0	1	0
Qatar	83	0	83	0	0	83	0	0	0	2	0
Romania	4	0	4	0	0	4	0	0	0	3	0
Russia	534	0	535	0	0	535	0	0	0	42	0
Saudi_Arabia	544	0	544	0	0	544	0	0	0	9	0
Singapore	0	0	0	65	0	0	8	0	0	1	0
South_Africa	0	0	0	0	0	0	0	0	0	8	0
Spain	0	0	0	0	0	0	0	0	0	6	0
Sudan	5	0	5	0	0	5	0	0	18	3	0
Sweden	0	0	0	0	0	0	0	0	0	1	0
Syria	2	0	2	0	0	2	0	0	8	1	0
Thailand	16	0	16	0	0	16	0	0	0	7	0
Trinidad_Tobago	6	0	6	0	0	6	0	0	0	1	0
Tunisia	3	0	3	0	0	3	0	0	0	1	0
Turkey	2	0	2	0	0	2	0	0	0	8	0
Ukraine	0	0	0	0	0	0	0	0	54	7	0
UAE	167	0	167	0	0	167	0	0	0	4	0
UK	40	0	40	0	0	40	0	0	0	10	0
US	520	0	521	0	0	521	0	0	0	106	0
Venezuela	139	0	139	62	0	139	41	0	29	6	0
Vietnam	18	0	18	0	0	18	0	0	0	5	0
Yemen	7	0	7	0	0	7	0	0	5	1	0

Table 28: Detailed consumption results by country across all modeled scenarios (million mt)

	Baseline	Cost	Cost supply	Cost API	GHG	GHG supply	GHG API	3GT	3GT location	3GT producer	3GT consumer
Algeria	0	20	20	20	20	20	20	0	5	0	4
Angola	2	2	2	2	2	2	2	0	2	0	2
Argentina	27	27	27	27	27	27	27	0	8	7	7
Australia	41	41	41	41	41	41	41	0	13	0	11
Bangladesh	0	0	0	0	0	0	0	0	0	0	0
Belgium	33	33	33	33	33	33	33	0	3	17	2
Brazil	136	136	136	136	136	136	136	0	15	45	27
Brunei	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.0	0.2
Bulgaria	0	0	0	0	0	0	0	0	0	0	0
Canada	114	114	114	114	114	114	114	0	0	0	12
Chile	15	15	15	15	15	15	15	0	2	0	2
China	527	527	527	527	527	527	527	527	199	174	174
Colombia	16	16	16	16	16	16	16	0	0	0	3

Congo	0	0	0	0	0	0	0	0	0	0	0
Denmark	9	9	9	9	9	9	9	0	1	9	1
Ecuador	13	13	13	13	13	13	13	0	0	0	2
Egypt	43	43	43	43	43	43	43	0	6	0	5
Equatorial_Guinea	0	0	0	0	0	0	0	0	0	0	0
Finland	9	9	9	9	9	9	9	0	1	9	1
France	85	85	85	85	85	85	85	0	9	0	8
Gabon	0	0	0	0	0	0	0	0	0	0	0
Germany	122	122	122	122	122	122	122	0	18	19	16
Greece	16	16	16	16	16	16	16	0	2	16	2
India	199	199	199	199	199	199	199	0	75	72	72
Indonesia	66	66	66	66	66	66	66	66	41	52	36
Iran	102	102	102	102	102	102	102	0	14	20	12
Iraq	27	27	27	27	27	27	27	0	5	0	4
Ireland	7	7	7	7	7	7	7	0	1	0	1
Israel	11	11	11	11	11	11	11	0	2	0	2
Italy	62	62	62	62	62	62	62	0	0	0	7
Japan	217	217	217	217	217	217	217	176	34	16	32
Korea	119	119	119	119	119	119	119	119	17	119	17
Kuwait	24	24	24	24	24	24	24	0	3	0	3
Lithuania	0	0	0	0	0	0	0	0	0	0	0
Malaysia	39	39	39	39	39	39	39	39	10	0	8
Mexico	94	94	94	94	94	94	94	0	14	0	12
Netherlands	44	44	44	44	44	44	44	0	4	44	4
New_Zealand	8	8	8	8	8	8	8	0	1	0	1
Nigeria	3	3	3	3	3	3	3	0	3	3	3
Norway	11	11	11	11	11	11	11	0	1	6	1
Oman	10	10	10	10	10	10	10	0	2	5	2
Pakistan	0	0	0	0	0	0	0	0	0	0	0
Peru	9	9	9	9	9	9	9	0	0	4	2
Philippines	16	16	16	16	16	16	16	16	4	0	4
Poland	26	26	26	26	26	26	26	0	9	0	8
Portugal	12	12	12	12	12	12	12	0	1	0	1
Qatar	11	11	11	11	11	11	11	0	2	11	2
Romania	10	10	10	10	10	10	10	0	3	0	3
Russia	163	163	163	163	163	163	163	0	48	0	42
Saudi_Arabia	157	157	157	157	157	157	157	0	11	37	9
Singapore	73	73	73	73	73	73	73	73	1	11	1
South_Africa	26	26	26	26	26	26	26	0	9	0	8
Spain	65	65	65	65	65	65	65	0	7	0	6
Sudan	0	0	0	0	0	0	0	0	0	0	0
Sweden	16	16	16	16	16	16	16	0	1	0	1
Syria	0	0	0	0	0	0	0	0	0	0	0
Thailand	58	58	58	58	58	58	58	58	8	0	7
Trinidad_Tobago	2	2	2	2	2	2	2	0	1	2	1
Tunisia	2	2	2	2	2	2	2	0	1	0	1

Turkey	30	30	30	30	30	30	30	0	9	30	8
Ukraine	0	0	0	0	0	0	0	0	0	0	0
UAE	43	43	43	43	43	43	43	0	4	0	4
UK	76	76	76	76	76	76	76	0	11	41	10
US	867	867	867	867	867	867	867	0	123	0	106
Venezuela	42	42	42	42	42	42	42	0	0	0	6
Vietnam	21	21	21	21	21	21	21	21	5	0	5
Yemen	0	0	0	0	0	0	0	0	0	0	0

Chapter 6. Conclusions and Future Work

6.1 Research Questions Revisited

As a summary of the work presented in this dissertation, brief answers to the research questions presented in the first chapter are as follows.

Chapter 2: Assessment of policies to reduce core forest fragmentation from Marcellus Shale development in Pennsylvania

- *What is the historical impact of natural gas development on Pennsylvania's forest ecosystem based on the principals of landscape ecology?*
  Between 2000 and 2012, approximately 110 km of roads and 1,600 km of gathering lines have been constructed to support 1,080 wells. This has resulted in the loss of about 13,000 ha of core forest (including core forest lost to the creation of new edges) and an increase in the number of core patches in Bradford County from 900 to 1000. In the core forest study region, 25 well pads were developed between 2000 and 2012. Comparing the 2000 land use map to the 2012 land use map developed with the efficient theoretical gathering line route demonstrates that the number of core patches increased by 25% (from 65 to 81). Additionally, the single largest patch decreased from about 16% of the total habitat area to about 12%.

- *How is future development likely to impact PA's forest ecosystem?*
  If left unchecked, future development could further increase the number of core patches from 81 in 2012 to 167 throughout the lifetime of the play. This would be a 100% increase above the 2012 level of fragmentation. Similarly, the LPI (largest patch index) and PLAND (total percent habitat) decrease as lateral lengths decreases. Because LPI and PLAND are indicators of fragmentation, the decreasing metrics signify that additional fragmentation is projected to occur.

- *What development strategies could be used to mitigate the cumulative impact of natural gas development?*

Two development strategies are explored: increasing horizontal lateral length to decrease well pad density or construct gathering lines along the routes of existing infrastructure corridors. Horizontal drilling technology is currently capable of achieving laterals approximately 3,000 m long. Therefore, the minimum achievable density is about 15 well pads/100 km^2. This would still result in an increase in the number of core patches from 81 to 101 core patches, or a 25% increase above 2012 levels. The results show that even with a significant advancement in technology resulting in laterals over twice as long as currently feasible, 2012 levels of fragmentation cannot be maintained by decreasing well pad density alone.

The results of the policy alternative of requiring all gathering lines to follow the route of pre-existing roads show that this policy essentially maintains fragmentation at the 2012 level regardless of well pad density. For example, in the 15 well pads/100 km^2 scenario, the model indicates that the number of core patches in the study region increases by one when the gathering lines follow the route of pre-existing roads, as opposed to the incremental 20 patches projected by the straight line gathering line model. Additionally, as shown in Figure 4, the distribution of core patch area is maintained at the 2012 level in addition to the number of core patches.

Chapter 3: A systems level perspective on Marcellus wastewater management in PA

- *What volume of non-reusable wastewater will likely be generated throughout the lifetime of the Marcellus Shale play? Is Marcellus representative of other shale plays?*
  An average of 36 thousand bbls of non-flowback produced water are generated per well (90% CI: 13-70 kbbls), primarily in the first two years of production.

- *What are the wastewater management options available to Marcellus producers and what are the economic tradeoffs between them? What is the least cost option for the average Marcellus well?*
  There are potentially three management strategies for produced water, which is considered here: disposal via deep well injection, treatment at a centralized waste

treatment facility, and onsite tertiary wastewater treatment; 67% of the time Class II disposal is the least cost option, 25% of the time CWT is the least cost option, and 8% of the time on-site treatment is the least cost option. The corresponding average costs are \$5.80/bbl (\$0.015/Mcf), \$7.80/bbl (\$0.020/Mcf), and \$8.40/bbl (\$0.021/Mcf), respectively

- *What are other important water quality issues in Pennsylvania and how much would it cost to address these sources of pollution?*
  Coal mine drainage and agricultural runoff are other important water quality issues in Pennsylvania and would cost an average of \$0.064/bbl and \$0.08/bbl, respectively. For the difference in cost of disposing of Marcellus wastewater rather than treating it at a CWT, Pennsylvania could pay to treat about 70 years worth of all CMD in the state or approximately 360% of the high runoff potential acreage in the Susquehanna river basin.

Chapter 4: Life Cycle Greenhouse Gas Emissions from U.S. Liquefied Natural Gas Exports: Implications for End Uses

- *What are the greenhouse gas implications of the natural gas export landed life cycle, including upstream extraction and pipeline transmission, liquefaction, shipping, and regasification?*
  Mean landed (pre-combustion) life cycle GHGs for exported U.S. LNG after regasification at the importing country were found to be 37 g CO_2-equiv/MJ with a range of 27 to 50. Of these landed emissions, the shipping stage of the life cycle contributes an average of 5% and the variation in emissions across shipping routes is nominal.

- *What are the GHG cost or savings of displacing the traditional fuel sources of domestic coal (extracted in the importing country) or Russian natural gas transmitted via pipeline?*
  Life cycle emissions from exported LNG were found on average to be 655 g CO_2-equiv/kWh for electricity generation and 104 g CO_2-equiv/MJ for thermal energy generation. When considering a 100-year GWP, mean life cycle exported U.S. LNG emissions are within the uncertainty bounds of Russian natural gas exports, and result in about 45% fewer emissions than coal electricity generation. Given a 20-year GWP,

exported U.S. LNG would reduce emissions from electricity production via Russian gas by 27% and cut emissions from electricity production from coal by 32% When considering a 100-yr GWP, mean GHG emissions from U.S. LNG exports would be 16% and 13% lower than industrial heating fueled by Russian natural gas exports and coal, respectively. However, when using a 20-year GWP, mean GHG emissions from U.S. exports would be 4% higher than coal but 27% lower than Russian pipeline exports.

- *What are the environmental implications of LNG exports from the US perspective?* Because approximately 41% (58% using a 20-year GWP) of life cycle LNG export emissions would arise from domestic extraction, pipeline transport, and liquefaction, increased extraction of natural gas without the domestic benefits of reduced combustion emissions would likely not be advantageous for the U.S. from a country-based carbon accounting perspective. Our mean estimates are that each thousand cubic feet (Mcf) of natural gas loaded onto a ship in liquefied form at a U.S. port represents about 0.037 metric tons of GHGs (for 100-yr GWP from production, transmission and liquefaction). Using a SSC of $49/ metric ton GHG, this means each Mcf of natural gas converted to LNG and exported from the U.S. for electricity generation potentially could cost the U.S. about $1.80 of social cost for embodied GHGs, or $0.50 to $5.50/Mcf across the full range of estimates for the 2020 social costs of carbon. Assuming a natural gas price of $4/Mcf, exporting LNG could have a social cost of between 12.5% to 135% of the market price.

Chapter 5: Assessment of GHG mitigation opportunities in the global crude trade

- *What is the current global production, consumption, and trade of crude and what are the implications of different carbon accounting methods on characterizing country specific emissions burdens?*
  The current global crude system has a shipping and crude cost of $3T, of which $0.14T are from shipping and $2.8T are from purchasing crude. The total emissions are 16.5 Gt CO_2-eq, of which 16.1 Gt are from the crude upstream, midstream, and downstream life cycle stages and .3 Gt are from shipping. Total crude consumption is about 4,000 million mt.

- *What is the potential for cost and greenhouse gas savings in an optimized global crude system given current demand structure?*
  When minimizing by cost with supply and demand constrained to the 2014 production levels, total system cost is \$2.75T and total emissions are 16.4 Gt. When minimizing by cost with demand constrained to 2014 levels and API gravity constrained to target API, the total cost is \$2.8T and the total emissions are 16 Gt CO_2. When minimizing by cost with only demand constrained, allowing production by country and API to vary freely, the total cost is \$2.4T and total GHG emissions are 20.5 Gt CO_2.
  When minimizing by emissions with supply and demand constrained to the 2014 production levels, total system cost is \$2.75T and total emissions are 16.4 Gt. When minimizing by emissions with demand constrained to 2014 levels and API gravity constrained to target API, the total cost is \$2.9T and the total emissions are 15.3 Gt CO_2. When minimizing by emissions with only demand constrained, allowing production by country and API to vary freely, the total cost is \$3.5T and total GHG emissions are 11Gt CO_2.

- *What would be the impacts on global crude consumption given a severe carbon budget? How could carbon accounting strategies incentivize more effective mitigation behavior across the crude trade network?*
  With a 3Gt global cap on emissions from crude, and country specific carbon allocations based on COP21 NDCs, under a location based carbon accounting system production would be limited to 720 million mt. Under a producer based accounting system, production would be limited to 760 million mt. Finally, under a consumption based carbon accounting system production would be limited to 730 million mt. Alternatively, if countries did not have specific allocated portions of total emissions, total consumption could increase to over 1000 million mt. However, this scenario also results in equity concerns since demand is concentrated in a few countries rather than dispersed throughout the system. This demonstrates that the interaction between carbon accounting system and the country specific carbon limits should be developed in coordination, rather than independently in order to maximize the cost-effectiveness of carbon mitigation

measures. Finally, this shows that individual country carbon budgets within a larger global carbon budget could serve to promote equity across the system.

6.2 Discussion

As we seek opportunities to mitigate greenhouse gas emissions in an effort to address climate change, it is important to be cognizant of new environmental concerns that evolve as a result of these transitions. An example of this is the deployment of natural gas as a bridge fuel. Unconventional natural gas extraction has many social and environmental impacts that need to be understood and addressed by policy makers. In the second chapter of this dissertation, I projected future Marcellus Shale development in Pennsylvania and quantified the impact the infrastructure could have on the forest ecosystem. I then identified regulatory strategies that could mitigate core forest fragmentation while simultaneously enabling natural gas extraction to continue as projected. In chapter 3, I quantify the potential wastewater generation by Marcellus Shale development and compare the economic viability of wastewater management strategies. In this chapter, I suggest how reallocating financial resources from the natural gas sector to other water quality issues in Pennsylvania could both protect surface waters from additional risk of contamination and mitigate other important water quality issues from coal mine drainage and agricultural runoff throughout the state.

For the last two chapters, I shift the focus to greenhouse gas mitigation efforts, and the implications of carbon accounting methods on global emissions outcomes. In Chapter 4, I assess the potential for US natural gas exports to reduce global greenhouse gas emissions as a function of end use and displaced energy resources. Additionally, I monetize the emissions along the supply chain to demonstrate how the US might realize an increase in social cost for exporting LNG despite the benefits of global GHG reduction. This demonstrates the importance for careful consideration of how carbon accounting strategies might unintentionally de-incentivize globally beneficial mitigation measures. Finally, in Chapter 5 I further explore the effect of various carbon accounting strategies on international trade patterns using the global crude network as a case study. I find these measures to be especially asymmetric in their outcomes as the global carbon budget becomes more restrictive. Because different carbon accounting strategies and carbon budgets imply varying geospatial distribution of optimal supply, and because it takes several years for extraction operations and refinery infrastructure to shift, it is essential for policy makers to take a long term planning horizon now to ensure cost-effective carbon mitigation in the future. Additionally, from this chapter I see that there are important interactions between carbon accounting strategies and country specific carbon allocations. This demonstrates the importance for climate policies to be considered as a holistic, integrated system in order to ensure the cost-effectiveness of mitigation strategies.

6.3 Deliverables

The deliverables from this thesis work are peer-reviewed journal publications. Chapter 1 has been published in *Ecological Indicators* and Chapter 3 has been published in *Environmental Science & Technology*. Chapters 2 and 5 will be submitted for publication in the future. Additionally, a deliverable from this work includes a functional wastewater management decision support tool that can be run by any user with MATLAB access using either default or edited inputs. This tool will be made publically available through an online platform.

6.4 Research Contributions

This dissertation demonstrates how systems level thinking combined with quantitative modeling can serve both to effectively characterize impact and to identify regulatory solutions to environmental concerns. This work was the first of its kind to project the geospatial component of potential future natural gas infrastructure development as an indicator of potential ecological impacts. Additionally, this thesis is the first to address the issue of Marcellus wastewater management on a regional level and propose a framework through which Pennsylvania's water resources could be protected in the most cost-effective manner. A third contribution was the use of monetizing an LCA to inform policy makers of the national carbon accounting implications resulting from a spatial shift in embodied emissions. Finally, the fifth chapter contributes to the literature by demonstrating a method to smooth data to preserve relationships between trade partners while simultaneously balancing the system. Additionally, this work is the first to identify theoretical interactions between carbon accounting strategies for international trade and country specific carbon allocations under a global carbon cap for the crude sector.

6.5 Future Work

Proposed future work to expand on the chapters presented in this dissertation include:

Wastewater Management:

- Developing a temporally explicit wastewater generation model for the Marcellus region. This would enable the exploration of CWT and disposal well capacity constraints, and help identify the optimal locations and capacities of new treatment facilities. Additionally, by including the relationship between domestic natural gas prices and new well development and systematically varying natural gas prices over time, one could quantify the impact of natural gas prices on the need for wastewater management infrastructure development.
- Including calculations of water quality associated with partial water treatment in the decision support tool could enable consideration of other beneficial reuse options that may not require fully treated water, such as industrial processes.

Crude System Optimization Model

A number of additional analyses can be undertaken, including:

- Exploring the differences between a global cap and trade policy versus a global carbon tax to understand the implications of each policy on meeting climate objectives.
- Including production limits as a function of proven reserves by country to act as a resource constraint, thereby eliminating operational infeasible results such as a single country producing the entire global crude supply.
- Developing and incorporating a supply dispatch curve based on regional variations in extraction costs to account for suppliers' choices in producing or not producing crude reserves.
- Incorporating welfare functions for each country to improve equity in consumption reductions under a strict global carbon cap.
- Expanding to include sectors closely related to and influenced by the crude trade, such as trade of finished products or chemicals to understand how the cost/emissions reductions from shifting crude trade patterns might be enhanced or offset by trade of downstream products.

- Converting the model into a stochastic optimization to include distributions of crude prices, crude emissions, shipping costs, etc to capture the uncertainty and variability associated with these parameters and to determine how robust the results are.
- Conducting sensitivity analysis on country specific carbon budgets to determine how different sets of country carbon caps can incentivize different objectives such as global emissions reductions, equitable distribution of costs and consumption, and supply security. These optimal country carbon allocations could then be compared to the COP21 NDCs to propose modifications to more cost-effectively meet future climate objectives.

7. References

1. Venkatesh, A.; Jaramillo, P.; Griffin, W. M.; Matthews, H. S., Uncertainty in life cycle greenhouse gas emissions from United States natural gas end-uses and its effects on policy. *Environmental science & technology* **2011,** *45*, (19), 8182-9.
2. Logan, J.; Heath, G.; Macknick, J.; Paranhos, E.; Boyd, W.; Carison, K., Natural gas and the transformation of the US Energy Sector: Electricity. **2012**.
3. Hayhoe, K.; Kheshgi, H. S.; Jain, A. K.; Wuebbles, D. J., Substitution of natural gas for coal: climatic effects of utility sector emissions. *Climatic change* **2002,** *1-2*, (54), 107-139.
4. Wigley, T. M. L., Coal to gas: the influence of methane leakage. *Climatic Change* **2011,** *108*, (3), 601-608.
5. Zhang, X.; Myhrvold, N. P.; Caldeira, K., Key factors for assessing climate benefits of natural gas versus coal electricity generation. *Environmental Research Letters* **2014,** *9*, (11), 114022.
6. Levi, M., Climate consequences of natural gas as a bridge fuel. *Climatic change* **2013,** *118*, (3-4), 609-623.
7. Abramzon, S.; Samaras, C.; Curtright, A.; Litovitz, A.; Burger, N., Estimating the Consumptive Use Costs of Shale Natural Gas Extraction on Pennsylvania Roadways. *Journal of Infrastructure Systems* **2014,** *20*, (3), 06014001.
8. Litovitz, A.; Curtright, A.; Abramzon, S.; Burger, N.; Samaras, C., Estimation of regional air-quality damages from Marcellus Shale natural gas extraction in Pennsylvania. *Environmental Research Letters* **2013,** *8*, (1), 014017.
9. Vidic, R. D.; Brantley, S. L.; Vandenbossche, J. M.; Yoxtheimer, D.; Abad, J. D., Impact of shale gas development on regional water quality. *Science* **2013,** *340*, (6134), 1235009.
10. Kargbo, D. M.; Wilhelm, R. G.; Campbell, D. J., Natural Gas Plays in the Marcellus Shale: Challenges and Potential Opportunities. *Environmental science & technology* **2010,** *44*, (15).
11. Slonecker, E. T.; Milheim, L. E.; Roig-Silva, C. M.; Malizia, A. R.; Marr, D. A.; Fisher, G. B. *Landscape consequences of natural gas extraction in bradford and washington counties*; U.S. Department of the Interior; U.S. Geological Survey: 2012.
12. Drohan, P. J.; Brittingham, M.; Bishop, J.; Yoder, K., Early trends in landcover change and forest fragmentation due to shale-gas development in Pennsylvania: a potential outcome for the Northcentral Appalachians. *Environmental management* **2012,** *49*, (5), 1061-75.
13. Evans, J. S.; Kiesecker, J. M., Shale gas, wind and water: assessing the potential cumulative impacts of energy development on ecosystem services within the Marcellus play. *PloS one* **2014,** *9*, (2), e89210.
14. Clark, C. E.; Veil, J. A. *Produced Water Volumes and Management Practices in the United States*; Argonne National Laboratory, Environmental Science Division: 2009.
15. Wilson, J. M.; Wang, Y.; VanBriesen, J. M., Sources of high total dissolved solids to drinking water supply in southwestern Pennsylvania. *Journal of Environmental Engineering* **2014**.
16. Shaffer, D. L.; Arias Chavez, L. H.; Ben-Sasson, M.; Romero-Vargas Castrillon, S.; Yip, N. Y.; Elimelech, M., Desalination and reuse of high-salinity shale gas produced water:

drivers, technologies, and future directions. *Environmental science & technology* **2013,** *47*, (17), 9569-83.
17. Technically Recoverable Resources. http://www.eia.gov/todayinenergy/detail.cfm?id=14431 (10/26/14),
18. Natural Gas Wellhead Price. http://www.eia.gov/dnav/ng/hist/n9190us3A.htm (10/26/14),
19. Montgomery, W. D.; Baron, R.; Bernstein, P.; Tuladhar, S. D.; Xiong, S.; Yuan, M., Macroeconomic Impacts of LNG Exports from the United States. **2012**.
20. *International Transport, Climate Change and Trade: What are the Options for Regulating Emissions from Aviation and Shipping and what will be their Impact on Trade?*; International Centre for Trade and Sustainable Development (ICTSD): 2010.
21. Davies, R. J.; Almond, S.; Ward, R. S.; Jackson, R. B.; Adams, C.; Worrall, F.; Herringshaw, L. G.; Gluyas, J. G.; Whitehead, M. A., Oil and gas wells and their integrity: Implications for shale and unconventional resource exploitation. *Marine and Petroleum Geology* **2014,** *56*, 239-254.
22. World Energy Statistics and Balances online data service. In (IEA), I. E. A., Ed.
23. Kuuskraa, V.; Stevens, S. H.; Moodhe, K. D., Technically Recoverable Shale Oil and Shale Gas Resources: An Assessment of 137 Shale Formations in 41 Countries Outside the United States. **2013**.
24. Annual Energy Outlook with projections to 2040. **2015**.
25. PA DEP Oil and Gas Well Drilling and Production. https://thompson.house.gov/sites/thompson.house.gov/files/PA%20DEP%20Oil%20and%20Gas%20Well%20Drilling%20and%20Production.pdf
26. *Governor's Marcellus Shale Advisory Commission*; 2011.
27. Spud External Data. http://www.depreportingservices.state.pa.us/ReportServer/Pages/ReportViewer.aspx?/Oil_Gas/Spud_External_Data
28. Baihly, J. D.; Altman, R. M.; Malpani, R.; Luo, F., Shale Gas Production Decline Trend Comparison over TIme and Basinse. *Societ of Petroleum Engineers* **2010**.
29. Cluff, M. A.; Hartsock, A.; MacRae, J. D.; Carter, K.; Mouser, P. J., Temporal changes in microbial ecology and geochemistry in produced water from hydraulically fractured Marcellus shale gas wells. *Environmental science & technology* **2014,** *48*, (11), 6508-17.
30. Arent, D.; Logan, J.; Macknick, J.; Boyd, W.; Medlock III, K.; O'Sullivan, F.; Edmonds, J.; Clarke, L., A review of water and greenhouse gas impacts of unconventional natural gas development in the United States. *MRS Energy & Sustainability: A Review Journal* **2015**.
31. *Annual Energy Outlook with Projections to 2035*; U.S. Energy Information Administration: 2012.
32. Mokhatab, S.; John Mak; Valappil, J.; Wood, D. A., *The Handbook of Liquified Natural Gas*.
33. Abrahams, L. S.; Samaras, C.; Griffin, W. M.; Matthews, H. S., Life cycle greenhouse gas emissions from U.S. liquefied natural gas exports: implications for end uses. *Environmental science & technology* **2015,** *49*, (5), 3237-45.
34. BP *Statistical Review of World Energy*; British Petroleum (BP): U.K., 2013.
35. *An Overview of the World LNG Market and Canada's Potential for Exports of LNG*; Canadian Association of Petroleum Producers: 2014.

36. *Federal Approval Process for Liquefied Natural Gas Exports*; GAO-14-762; United States Government Accountability Office: 2014.
37. Schrag, D. P., Is Shale Gas Good for Climate Change? *American Academy of Arts & Sciences* **2012,** *141*, (2).
38. Jiang, M.; Griffin, W. M.; Hendrickson, C.; Jaramillo, P.; VanBriesen, J.; Venkatesh, A., Life cycle greenhouse gas emissions of Marcellus shale gas. *Environmental Research Letters* **2011,** *6*, (3), 034014.
39. Howarth, R.; Santoro, R.; Ingraffea, A., Methane and the greenhouse-gas foot- print of natural gas from shale formations. *Climate Change* **2011,** *106*, (4), 679-690.
40. Tong, F.; Jaramillo, P.; Azevedo, I. M., Comparison of life cycle greenhouse gases from natural gas pathways for medium and heavy-duty vehicles. *Environmental science & technology* **2015,** *49*, (12), 7123-33.
41. Matthews, H. S.; Hendrickson, C. T.; Matthews, D. H., *Life Cycle Assessment: QUantitative Approaches for Decisions That Matter*. 2015.
42. *Climate Change 2013 the physical science basis*; Intergovernmental Panel on Climate Change (IPCC): 2014.
43. Newell, R. G.; Raimi, D., Implications of shale gas development for climate change. *Environmental science & technology* **2014,** *48*, (15), 8360-8.
44. McJeon, H.; Edmonds, J.; Bauer, N.; Clarke, L.; Fisher, B.; Flannery, B. P.; Hilaire, J.; Krey, V.; Marangoni, G.; Mi, R.; Riahi, K.; Rogner, H.; Tavoni, M., Limited impact on decadal-scale climate change from increased use of natural gas. *Nature* **2014,** *514*, (7523), 482-5.
45. *Changing the game?: Emissions and market implications of new natural gas supplies*; Energy Modeling Forum: Stanford University, 2013.
46. Moore, C. W.; Zielinska, B.; Petron, G.; Jackson, R. B., Air Impacts of Increased Natural Gas Acquisition, Processing, and Use: A Critical Review. *Environmental science & technology* **2014,** *48*, 8349-8359.
47. About US Natural Gas Pipelines. https://http://www.eia.gov/pub/oil_gas/natural_gas/analysis_publications/ngpipeline/index.html
48. Kang, M.; Kanno, C. M.; Reid, M. C.; Zhang, X.; Mauzerall, D. L.; Celia, M. A.; Chen, Y.; Onstott, T. C., Direct measurements of methane emissions from abandoned oil and gas wells in Pennsylvania. *PNAS* **2014,** *111*, (51).
49. Greenhouse Gas Inventory. https://www3.epa.gov/climatechange/ghgemissions/inventoryexplorer/data_explorer_flight.html
50. Brandt, A. R.; Heath, G. A.; Kort, E. A.; O'Sullivan, F.; Petron, G.; Jordaan, S. M.; Tans, P.; Wilcox, J.; Gopstein, A. M.; Arent, D. J.; Wofsy, S.; Brown, N. J.; Bradley, R.; Stucky, G. D.; Eardley, D.; Harriss, R., Methane Leaks from North American Natural Gas Systems. *Science* **2014,** *343*, 733-735.
51. Allen, D. T.; Pacsi, A. P.; Sullivan, D. W.; Zavala-Araiza, D.; Harrison, M.; Keen, K.; Fraser, M. P.; Daniel Hill, A.; Sawyer, R. F.; Seinfeld, J. H., Methane emissions from process equipment at natural gas production sites in the United States: pneumatic controllers. *Environmental science & technology* **2015,** *49*, (1), 633-40.

52. Alley, B.; Beebe, A.; Rodgers, J., Jr.; Castle, J. W., Chemical and physical characterization of produced waters from conventional and unconventional fossil fuel resources. *Chemosphere* **2011,** *85*, (1), 74-82.
53. Brantley, H. L.; Thoma, E. D.; Squier, W. C.; Guven, B. B.; Lyon, D., Assessment of methane emissions from oil and gas production pads using mobile measurements. *Environmental science & technology* **2014,** *48*, (24), 14508-15.
54. Mitchell, A. L.; Tkacik, D. S.; Roscioli, J. R.; Herndon, S. C.; Yacovitch, T. I.; Martinez, D. M.; Vaughn, T. L.; Williams, L. L.; Sullivan, M. R.; Floerchinger, C.; Omara, M.; Subramanian, R.; Zimmerle, D.; Marchese, A. J.; Robinson, A. L., Measurements of methane emissions from natural gas gathering facilities and processing plants: measurement results. *Environmental science & technology* **2015,** *49*, (5), 3219-27.
55. Subramanian, R.; Williams, L. L.; Vaughn, T. L.; Zimmerle, D.; Roscioli, J. R.; Herndon, S. C.; Yacovitch, T. I.; Floerchinger, C.; Tkacik, D. S.; Mitchell, A. L.; Sullivan, M. R.; Dallmann, T. R.; Robinson, A. L., Methane emissions from natural gas compressor stations in the transmission and storage sector: measurements and comparisons with the EPA greenhouse gas reporting program protocol. *Environmental science & technology* **2015,** *49*, (5), 3252-61.
56. Petron, G.; Karion, A.; Sweeney, C.; Miller, B. R.; Montzka, S. A.; Frost, G. J.; Trainer, M.; Tans, P.; Andrews, A.; Kofler, J., A new look at methane and nonmethane hydrocarbonemissions from oil and natural gas operationsin the Colorado Denver-Julesburg Basin. *Journal of Geophysical Research: Atmospheres* **2014,** *119*, 6836-6852.
57. Karion, A.; Sweeney, C.; Kort, E. A.; Shepson, P. B.; Brewer, A.; Cambaliza, M.; Conley, S. A.; Davis, K.; Deng, A.; Hardesty, M.; Herndon, S. C.; Lauvaux, T.; Lavoie, T.; Lyon, D.; Newberger, T.; Petron, G.; Rella, C.; Smith, M.; Wolter, S.; Yacovitch, T. I.; Tans, P., Aircraft-Based Estimate of Total Methane Emissions from the Barnett Shale Region. *Environmental science & technology* **2015,** *49*, (13), 8124-31.
58. Schwietzke, S.; Griffin, W. M.; Matthews, H. S.; Bruhwiler, L. M., Natural gas fugitive emissions rates constrained by global atmospheric methane and ethane. *Environmental science & technology* **2014,** *48*, (14), 7714-22.
59. Peischl, J.; Ryerson, T. B.; de Gouw, J. A.; Gilman, J. B.; Holloway, J. S.; Lerner, B. M.; Nadkarni, R.; Neuman, J. A.; Nowak, J. B.; Trainer, M.; Warneke, C.; Parrish, D. D., Quantifying atmospheric methane emissions from the Haynesville, Fayetteville, and northeastern Marcellus shale gas production regions. *Journal of Geophysical Research: Atmospheres* **2015,** *120*, 2119-2139.
60. Karion, A. e. a., Methane emissions estimate from airborne measurements over a western United States natural gas field. *Geophys. Res. Lett.* **2013,** *40*, 4393-4397.
61. EPA *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2009*; US Environmental Protection Agency, Climate Change Division: Washington, DC, 2011.
62. Burnham, A.; Han, J.; Clark, C. E.; Wang, M.; Dunn, J. B.; Palou-Rivera, I., Life-cycle greenhouse gas emissions of shale gas, natural gas, coal, and petroleum. *Environmental science & technology* **2012,** *46*, (2), 619-627.
63. Hultman, N.; Rebois, D.; Scholten, M.; Ramig, C., The greenhouse impact of un-conventional gas for electricity generation. *Environmental Research Letters* **2011,** *6*, (4).
64. Skone, T.; Littlefield, J.; Eckard, R.; Cooney, G.; Marriott, J. *Role of Alternative Energy Sources: Natural Gas Technology Assessment*; National Energy Technology Laboratory (NETL): 2012.

65. Reduce your Methane Emissions. https://www3.epa.gov/gasstar/methaneemissions/index.html
66. The President's Climate Action Plan (Executive Office of the President, Washington DC).
67. Adgate, J. L.; Goldstein, B. D.; McKenzie, L. M., Potential public health hazards, exposures and health effects from unconventional natural gas development. *Environmental science & technology* **2014,** *48*, (15), 8307-20.
68. Jaramillo, P.; Griffin, W. M.; Matthews, H. S., Comparative Life-Cycle Air Emissions of Coal, Domestic Natural Gas, LNG, and SNG for Electricity Generation. *Environmental science & technology* **2007**.
69. Pacsi, A. P.; Alhajeri, N. S.; Zavala-Araiza, D.; Webster, M. D.; Allen, D. T., Regional Air Quality Impacts of Increased Natural Gas Production and Use in Texas. *Environmental science & technology* **2013**, 130311072134008.
70. Field, R. A.; Soltis, J.; Murphy, S., Air quality concerns of unconventional oil and natural gas production. *Enviornmental Science Processes & Impacts* **2014,** *16*, (954).
71. Goldstein, B. D.; Brooks, B. W.; Cohen, S. D.; Gates, A. E.; Honeycutt, M. E.; Morris, j. B.; Penning, T. M.; Orme-Zavaleta, J.; Snawder, J., The role of toxicological science in meeting the challenges and opportunities of hydraulic fracturing. *ToxSci* **2014**.
72. Annevelink, M. P. J. A.; Meesters, J. A. J.; Hendriks, A. J., Environmental contamination due to shale gas development. *Science of the Total Environment* **2016,** *550*, 431-438.
73. McBroom, M.; Thomas, T.; Zhang, Y., Soil Erosion and Surface Water Quality Impacts of Natural Gas Development in East Texas, USA. *Water* **2012,** *4*, (4), 944-958.
74. Latta, S. C.; Marshall, L. C.; Frantz, M. W.; Toms, J. D., Evidence from two shale regions that a riparian songbird accumulates metals associated with hydraulic fracturing. *Ecosphere* **2015,** *6*, (9).
75. Swift, T. L.; Hannon, S. J., Critical thresholds associated with habitat loss: a review of the concepts, evidence, and applications. *Biological reviews of the Cambridge Philosophical Society* **2010,** *85*, (1), 35-53.
76. Lele, N.; Joshi, P. K.; Agrawal, S. P., Assessing forest fragmentation in northeastern region (NER) of India using landscape matrices. *Ecological Indicators* **2008,** *8*, (5), 657-663.
77. McGarigal, K., Landscape Pattern Metrics. In *Encyclopedia of Environmetrics*, John Wiley & Sons, Ltd.: 2013.
78. Makki, T.; Fakheran, S.; Moradi, H.; Iravani, M.; Senn, J., Landscape-scale impacts of transportation infrastructure on spatial dynamics of two vulnerable ungulate species in Ghamishloo Wildlife Refuge, Iran. *Ecological Indicators* **2013,** *31*, 6-14.
79. MacArthur, R. H., *The theory of island biogeography*. Princeton University Press: 1967; Vol. 1.
80. Brantley, S. L.; Yoxtheimer, D.; Arjmand, S.; Grieve, P.; Vidic, R.; Pollak, J.; Llewellyn, G. T.; Abad, J.; Simon, C., Water resource impacts during unconventional shale gas development: The Pennsylvania experience. *International Journal of Coal Geology* **2014,** *126*, 140-156.
81. Mitchell, A. L.; Small, M.; Casman, E. A., Surface water withdrawals for Marcellus Shale gas development: performance of alternative regulatory approaches in the Upper Ohio River Basin. *Environmental science & technology* **2013,** *47*, (22), 12669-78.

82. Thiel, G. P.; Lienhard, J. H., Treating produced water from hydraulic fracturing: Composition effects on scale formation and desalination system selection. *Desalination* **2014,** *346*, 54-69.
83. Stringfellow, W. T.; Domen, J. K.; Camarillo, M. K.; Sandelin, W. L.; Borglin, S., Physical, chemical, and biological characteristics of compounds used in hydraulic fracturing. *Journal of hazardous materials* **2014,** *275*, 37-54.
84. Adams, M. B., Land application of hydrofracturing fluids damages a deciduous forest stand in West Virginia. *Journal of environmental quality* **2011,** *40*, (4), 1340-4.
85. Colbron, T.; Kwiatkowski, C.; Schultz, K.; Bachran, M., Natural Gas Operations from a Public Health Perspective. *Human and Ecological Risk Assessment: An International Journal* **2011,** *5*, (17), 1039-1056.
86. Papoulias, D.; Velasco, A., Hydraulic Fracturing Fluids Likely Harmed Threatened Kentucky Fish Species. *Southeastern Naturalist* **2013,** *12*, (Special Issue 4).
87. Veil, J. A. *Produced Water Volumes and Management Practices in 2012*; Veil Enviornmental, LLC: 2015.
88. Addressing Gaps In Water Treatment Technology. *Journal of Petroleum Technology* **2013**.
89. Engelder, T.; Cathles, L. M.; Bryndzia, L. T., The fate of residual treatment water in gas shale. *Journal of Unconventional Oil and Gas Resources* **2014,** *7*, 33-48.
90. Rowen, E. L.; Engle, M. A.; Kraemer, T. F.; Schroeder, K. T.; Hammack, R. W.; Doughten, M. W., Geochemical and isotopic evolution of water produced from Middle Devonian Marcellus shale gas wells, Appalachian basin, Pennsylvania. *AAPG Bulletin* **2015,** *99*, (02), 181-206.
91. Chapman, E. C.; Capo, R. C.; Stewart, B. W.; Kirby, C. S.; Hammack, R. W.; Schroeder, K. T.; Edenborn, H. M., Geochemical and strontium isotope characterization of produced waters from Marcellus Shale natural gas extraction. *Environmental science & technology* **2012,** *46*, (6), 3545-53.
92. Blauch, M. E.; Inc, S. W. S.; Myers, R. R.; Moore, T. R.; Lipinski, B. A.; Inc, E.-N. C. E.; Houston, N. A.; Inc., S. W. S., Marcellus Shale Post-Frac Flowback Waters – Where is All the Salt Coming From and What are the Implications? *Society of Petroleum Engineers* **2009**.
93. *Energy-Water Nexus: Information on the Quantity, Quality, and Management of Water Produced during Oil and Gas Production*; GAO-12-156; United States Government Accountability Office: Washington DC, 2012.
94. Souther, S.; Tingley, M. W.; Popescu, V. D.; Hayman, D. T. S.; Ryan, M. E.; Graves, T. A.; Hartl, B.; Terrell, K., Biotic impacts of energy development from shale: research priorities and knowledge gaps. *Frontiers in Ecology and the Environment* **2014,** *12*, (6), 330-338.
95. Lutz, B. D.; Lewis, A. N.; Doyle, M. W., Generation, transport, and disposal of wastewater associated with Marcellus Shale gas development. *Water Resources Research* **2013,** *49*, (2), 647-656.
96. Bamberger, M.; Oswald, R. E., Impacts of gas drilling on human and animal health. *New solutions : a journal of environmental and occupational health policy : NS* **2012,** *22*, (1), 51-77.

97. Fakhru'l-Razi, A.; Pendashteh, A.; Abdullah, L. C.; Biak, D. R.; Madaeni, S. S.; Abidin, Z. Z., Review of technologies for oil and gas produced water treatment. *Journal of hazardous materials* **2009,** *170*, (2-3), 530-51.
98. Wilson, J. M.; VanBriesen, J. M., Oil and gas produced water management and surface drinking water sources in pennsylvania. *Environmental Practice* **2012,** *14*, (4).
99. Olmstead, S. M.; Muehlenbachs, L. A.; Shih, J.-S.; Chu, Z.; Krupnick, A. J., Shale gas development impacts on surface water quality in Pennsylvania. *PNAS* **2013,** *110*, (13), 4962-4967.
100. Press release: PA Fish and Boat Commission sues Consol Energy over Dunkard Creek environmental disaster. In Pennsylvania Department of Environmental Protection (PADEP): 2011.
101. Wastewater Treatment Requirements, 25 PA Code 95. In Pennsylvania Department of Environmental Protection (PADEP): 2010.
102. Volz, C. D., Written Testimony: Before the Senate Committee on Environment and Public Works and its Subcommittee on Water and Wildlife, Joint Hearing "Natural Gas Drilling, Public Health and Environmental Impacts". In 2011.
103. Hammer, R.; VanBriesen, J. M.; Levine, L. *In Fracking's Wake New Rules are Needed to Protect Our Health and Environment from Contaminated Wastewater*; National Resources Defense Council: 2012.
104. Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources. In Development, U. S. E. P. A. O. o. R. a., Ed. United States Environmental Protection Agency: 2011.
105. Summary of the Technical Workshop on Wastewater Treatment and Related Modeling. In EPA), U. S. E. P. A. U., Ed. 2013.
106. Gordalla, B. C.; Ewers, U.; Frimmel, F. H., Hydraulic fracturing: a toxicological threat for groundwater and drinking-water? *Environmental Earth Sciences* **2013,** *70*, (8), 3875-3893.
107. *Assessment of risks from unconventional gas well development in the Marcellus Shale of Western Maryland*; Maryland Department of the Environment
Maryland Department of Natural Resources: 2014.
108. Vengosh, A.; Jackson, R. B.; Warner, N.; Darrah, T. H.; Kondash, A., A critical review of the risks to water resources from unconventional shale gas development and hydraulic fracturing in the United States. *Environmental science & technology* **2014,** *48*, (15), 8334-48.
109. Warner, N. R.; Christie, C. A.; Jackson, R. B.; Vengosh, A., Impacts of shale gas wastewater disposal on water quality in western Pennsylvania. *Environmental science & technology* **2013,** *47*, (20), 11849-57.
110. Zhang, T.; Gregory, K.; Hammack, R. W.; Vidic, R. D., Co-precipitation of radium with barium and strontium sulfate and its impact on the fate of radium during treatment of produced water from unconventional gas extraction. *Environmental science & technology* **2014,** *48*, (8), 4596-603.
111. Almond, S.; Clancy, S. A.; Davies, R. J.; Worrall, F., The flux of radionuclides in flowback fluid from shale gas exploitation. *Environmental science and pollution research international* **2014,** *21*, (21), 12316-24.
112. Gafvert, T.; C, E.; E, H., Removal of radionuclides at a waterworks. *Journal of Environmental Radioactivity* **2002,** *63*.

113. Brown, E.; Hartmna, K.; Borick, C.; Rabe, B. G.; Ivacko, T. *Public Opinion on Fracking: Perspectives from Michigan and Pennsylvania. Issues in Energy and Environmental Policy* Gerald R. Ford School of Public Policy, University of Michigan: 2013.
114. Krupnick, A. J.; Siikamaki, J. *Would You Pay to Reduce Risks from Shale Gas Development? Public Attitudes in Pennsylvania and Texas*; RFF Center for Energy Economics and Policy (CEEP): 2011.
115. *The Economic and Employment Contributions of Unconventional Gas Development in State Economies*; IHS Inc: Washington, DC, 2012.
116. Kinnaman, T. C., The economic impact of shale gas extraction: A review of existing studies. *Ecological Economics* **2011,** *70*, (7), 1243-1249.
117. Rabinowitz, P. M.; Slizovskiy, I. B.; Lamers, V.; Trufan, S. J.; Holford, T. R.; Dziura, J. D.; Peduzzi, P. N.; Kane, M. J.; Reif, J. S.; Weiss, T. R.; Stowe, M. H., Proximity to Natural Gas Wells and Reported Health Status: Results of a Household Survey in Washington County, Pennsylvania. *Environmental health perspectives* **2014**.
118. Shonkoff, S. B.; Hays, J.; Finkel, M. L., Environmental public health dimensions of shale and tight gas development. *Environmental health perspectives* **2014,** *122*, (8), 787-95.
119. Werner, A. K.; Vink, S.; Watt, K.; Jagals, P., Environmental health impacts of unconventional natural gas development: A review of the current strength of evidence. *Science of the Total Environment* **2015,** *505*, 1127-1141.
120. Brandt, A. R.; Plevin, R. J.; Farrell, A. E., Dynamics of the oil transition: Modeling capacity, depletion, and emissions. *Energy* **2010,** *35*, (7), 2852-2860.
121. Oil Flows Data. In Reuters, Ed. 2016.
122. *Third IMO Greenhouse Gas Study 2014*; International Maritime Organization: 2014.
123. Corbett, J. J.; Winebrake, J. J.; Green, E. H.; Kasibhatla, P.; Eyring, V.; Lauer, A., Mortality from Ship Emissions: A Global Assessment. *Environmental science & technology* **2007,** *41*, 8512-8518.
124. Corbett, J.; Winebrake, J., Emissions Tradeoffs among Alternative Marine Fuels: Total Fuel Cycle Analysis of Residual Oil, Marine Gas Oil, and Marine Diesel Oil. *Journal of the Air & Waste Management Association* **2008,** *58*, (4), 538-542.
125. Psaraftis, H. N.; Kontovas, C. A., CO2 Emission Statistics for the World Commercial Fleet. *World Maritime University Journal of Maritime Affiars* **2009**.
126. Endresen, O.; Sorgard, E.; Sundet, J. K.; Dalsoren, S. B.; Isaksen, I. S. A.; Berglen, T. F.; Gravir, G., Emissions from international sea transportation and environmental impact. *Jounral of Geophysical Research* **2003,** *108*, (D17).
127. Bhattacharyya, S., Applied general equilibrium models for energy studies: a survey. *Energy Economics* **1996,** *18*, 145-164.
128. D'Agostino, A. L.; Sovacool, B. K.; Trott, K.; Ramos, C. R.; Saleem, S.; Ong, Y., What's the state of energy studies research?: A content analysis of three leading journals from 1999 to 2008. *Energy* **2011,** *36*, (1), 508-519.
129. Hudson, E. A.; Jorgenson, D. W., US energy policy and economic growth, 1975-2000. *The Bell Journal of Economics and Management Science* **1974**.
130. Rutherford, T. F. *Climate-Linked Tariffs: Practical Issues*; The Graduate Institute Centre for Trade and Economic Integration.
131. Davis, S. J.; Caldeira, K., Consumption-based accounting of CO2 emissions. *Proceedings of the National Academy of Sciences of the United States of America* **2010,** *107*, (12), 5687-92.

132. An, H.; Zhong, W.; Chen, Y.; Li, H.; Gao, X., Features and evolution of international crude oil trade relationships: A trading-based network analysis. *Energy* **2014,** *74*, 254-259.
133. Gao, C.; Sun, M.; Shen, B., Features and evolution of international fossil energy trade relationships: A weighted multilayer network analysis. *Applied Energy* **2015,** *156*, 542-554.
134. Geng, J.-B.; Ji, Q.; Fan, Y., A dynamic analysis on global natural gas trade network. *Applied Energy* **2014,** *132*, 23-33.
135. Hao, X.; An, H.; Qi, H.; Gao, X., Evolution of the exergy flow network embodied in the global fossil energy trade: Based on complex network. *Applied Energy* **2016,** *162*, 1515-1522.
136. Ji, Q.; Zhang, H.-Y.; Fan, Y., Identification of global oil trade patterns: An empirical research based on complex network theory. *Energy Conversion and Management* **2014,** *85*, 856-865.
137. McGlade, C.; Ekins, P., Un-burnable oil: An examination of oil resource utilisation in a decarbonised energy system. *Energy Policy* **2014,** *64*, 102-112.
138. Bohringer, C.; Balistereri, E. J.; Rutherford, T. F. *The Role of Border Carbon Adjustment in Unilateral Climate Policy: Insights from a Model Comparison Study*; Harvard Kennedy School: 2012.
139. McGlade, C.; Ekins, P., The geographical distribution of fossil fuels unused when limiting global warming to 2 degrees C. *Nature* **2015,** *517*, (7533), 187-90.
140. Sun, M.; Lowry, G. V.; Gregory, K. B., Selective oxidation of bromide in wastewater brines from hydraulic fracturing. *Water research* **2013,** *47*, (11), 3723-31.
141. Yang, L.; Grossmann, I. E.; Manno, J., Optimization models for shale gas water management. *AIChE Journal* **2014,** *60*, (10), 3490-3501.
142. *Report of the Conference of the Parties on its twenty-first session, held in Paris from 30 November to 11 December 2015*; United Nations: 2016.
143. Clarke; al., e. *Climate Change 2014 Mitigation of Climate Change Working Group III Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*; 2014.
144. Felder, S.; Rutherford, T. F., Unilateral policies and impact on international trade. *Journal of Environmental Economics and Management* **1993**, (25), 162-176.
145. Wyckoff, A. W.; Roop, J. M., The embodiment of carbon in imports of manufactured products: Implications for international agreements on greenhouse gas emissions. *Energy Policy* **1994**.
146. Condon, M.; Ignaciuk, A., **2013**.
147. Strømman, A. H.; Hertwich, E. G.; Duchin, F., Shifting Trade Patterns as a Means of Reducing Global Carbon Dioxide Emissions. *Journal of Industrial Ecology* **2009,** *13*, (1), 38-57.
148. Xu, Z.; Tian-yu, Q.; Xun-min, O.; Xi-liang, Z., The role of multi-region integrated emissions trading scheme: A computable general equilibrium analysis. *Applied Energy* **2015**.
149. Steininger, K. W.; Lininger, C.; Meyer, L. H.; Muñoz, P.; Schinko, T., Multiple carbon accounting to support just and effective climate policies. *Nature Climate Change* **2015,** *6*, (1), 35-41.
150. Davis, S. J.; Peters, G. P.; Caldeira, K., The supply chain of CO2 emissions. *PNAS* **2011,** *108*, (45), 18554-18559.

151. Abrahams, L. S.; Griffin, W. M.; Matthews, H. S., Assessment of policies to reduce core forest fragmentation from Marcellus shale development in Pennsylvania. *Ecological Indicators* **2015,** *52*, 153-160.
152. Guidelines for administering oil and gas activity on state forest lands. In Forestry, B. o., Ed. Department of Conservation and Natural Resources (DCNR): 2011.
153. Johnson, N.; Gagnolet, T.; Ralls, R.; Zimmerman, E.; Eichelberger, B.; Tracey, C.; Kreitler, G.; Orndorff, S.; Tomlinson, J.; Bearer, S.; Sargent, S. *Pennsylvania Energy Impacts Assessment*; The Nature Conservancy: 2010.
154. Johnson, N.; Gagnolet, T.; Ralls, R.; Stevens, J. *Natural Gas Pipelines*; The Nature Conservancy: 2011.
155. Henderson, P., Report to General Assembly on Natural Gas Gathering Lines. In Commonwealth of Pennsylvania: Office of Governor Tom Corbett, 2012.
156. Bishop, J. Temporal Dynamics of Forest Patch Size Distribution and Fragmentation of Habitat Types in Pennsylvania. The Pennsylvania State University, 2008.
157. Brittingham, M., Habitat Fragmentation. In PennState School of Forest Resources: 2011.
158. Fischer, J.; Lindenmayer, D. B., Landscape Modification and Habitat Fragmentation: a synthesis. *Global Ecology and Biogeography* **2007,** *16*, 265-280.
159. Benítez-López, A.; Alkemade, R.; Verweij, P. A., The impacts of roads and other infrastructure on mammal and bird populations: A meta-analysis. *Biological Conservation* **2010,** *143*, (6), 1307-1316.
160. McGunegle, M. The Effects of Oil and Gas Development on Forest Fragmentation and Breeding Bird Populations in the Allegheny National Forest. The Pennsylvania State University, 2009.
161. PA DEP Oil & Gas Reporting Website. https://http://www.paoilandgasreporting.state.pa.us/publicreports/Modules/Production/ProductionHome.aspx (2014),
162. Pennsylvania Land Cover, 2000. http://www.pasda.psu.edu/uci/FullMetadataDisplay.aspx?file=palanduse2000.xml
163. Download Census 2000 TIGER/Line Data. http://arcdata.esri.com/data/tiger2000/tiger_county.cfm?sfips=42
164. Brittingham, M.; Drohan, P.; Bishop, J. In *Initial Landscape Changes Associated with Marcellus Shale Development; Implications for Forests and Wildlife*, 18th Central Hardwood Forest Conference, 2013; 2013.
165. *Reasonably Foreseeable Development Scenario for Oil and Gas for the George Washington National Forest Virginia and West Virginia*; U.S. Department of the Interior: 2011.
166. Marcellus Center for Outreach & Research (MCOR). http://www.marcellus.psu.edu/resources/maps.php
167. Soeder, D. J.; Sharma, S.; Pekney, N.; Hopkinson, L.; Dilmore, R.; Kutchko, B.; Stewart, B.; Carter, K.; Hakala, A.; Capo, R., An approach for assessing engineering risk from shale gas wells in the United States. *International Journal of Coal Geology* **2014,** *126*, 4-19.
168. Kiviat, E., Risks to biodiversity from hydraulic fracturing for natural gas in the Marcellus and Utica shales. *Annals of the New York Academy of Sciences* **2013,** *1286*, 1-14.
169. Molnar, S., Bradford County Gas Lines. In Bradford County Planning and Grants: 2013.
170. Baddeley, A. rSSI {spatstat}. http://www.inside-r.org/packages/cran/spatstat/docs/rSSI

171. Bearer, S.; Nicholas, E.; Gagnolet, T.; DePhilip, M.; Moberg, T.; Johnson, N. *Evaluating the Scientific Support of Conservation Best Management Practices for Shale Gas Extraction in the Appalachian Basin*; Environmental Practice, 2012; pp 308-319.
172. Shale Gas Development and State Forests. In Pennsylvania Department of Conservation and Natural Resources: 2013.
173. Konschnik, K. E.; Boling, M. K., Shale Gas Development: A Smart Regulation Framework. *Environmental science & technology* **2014,** *48*, (15), 8404-8416.
174. Wiseman, H., The Capacity of State Institutions to Govern Shale Gas Development Risks. *Environmental science & technology* **2014,** *48*, (15), 8376-8387.
175. Eshleman, K. N.; Elmore, A. *Recommended Best Management Practices for Marcellus Shale Gas Development in Maryland*; Appalachian Laboratory: 2013.
176. Rabe, B. G., Shale Play Politics: The Intergovernmental Odyssey of American Shale Governance. *Environmental science & technology* **2014,** *48*, (15), 8369-8375.
177. Laurenzi, I. J.; Jersey, G. R., Life cycle greenhouse gas emissions and freshwater consumption of Marcellus shale gas. *Environmental science & technology* **2013,** *47*, (9), 4896-903.
178. Jackson, R. B.; Vengosh, A.; Carey, J. W.; Davies, R. J.; Darrah, T. H.; O'Sullivan, F.; Pétron, G., The Environmental Costs and Benefits of Fracking. *Annual Review of Environment and Resources* **2014,** *39*, (1), 327-362.
179. Jiang, M.; Griffin, M. W.; Hendrickson, C.; Jaramillo, P.; VanBriesen, J.; Venkatesh, A., Life cycle greenhouse gas emissions of Marcellus shale gas. *Environmental Research Letters* **2011,** *6*, (3), 034014.
180. GDP by Industry. In Analysis, U. S. D. o. C. B. o. E., Ed. 2014.
181. Gregory, K.; Mohan, A. M., Current perspective on produced water management challenges during hydraulic fracturing for oil and gas recovery. *Environmental Chemistry* **2015,** *12*, (3), 261.
182. Arthur, D. J.; Langhus, B. G.; Patel, C., Technical summary of oil & gas produced water treatment technologies. *ALL Consulting, LLC* **2005**.
183. Protection, D. o. E., Wastewater Treatment Requirements. In 2010; Vol. Pa. Code, Chapter 95.
184. Assessment of the Potential Impacts of Hydraulic Fracturing for Oil and Gas on Drinking Water Resources. *U.S. Environmental Protection Agency* **2015**.
185. Mehta, N.; O'Sullivan, F., In *Food, Energy, and Water: The Chemistry Connection*, Ahuja, S., Ed. 2015.
186. Produced Water Management Technology Identification Module. http://www.netl.doe.gov/research/coal/crosscutting/pwmis/tim (1/18/2016),
187. FRACTURING FLUID FLOWBACK REUSE FEASABILITY STUDY AND DECISION TOOL. http://www.ptac.org/projects/43 (1/18/2016),
188. Gao, J.; You, F., Optimal design and operations of supply chain networks for water management in shale gas production: MILFP model and algorithms for the water-energy nexus. *AIChE Journal* **2014**, n/a-n/a.
189. Zhang, X.; Sun, A. Y.; Duncan, I. J., Shale gas wastewater management under uncertainty. *Journal of environmental management* **2016,** *165*, 188-98.
190. Small, M. J.; Stern, P. C.; Bomberg, E.; Christopherson, S. M.; Goldstein, B. D.; Israel, A. L.; Jackson, R. B.; Krupnick, A.; Mauter, M. S.; Nash, J.; North, D. W.; Olmstead, S. M.; Prakash, A.; Rabe, B.; Richardson, N.; Tierney, S.; Webler, T.; Wong-Parodi, G.;

Zielinska, B., Risks and risk governance in unconventional shale gas development. *Environmental science & technology* **2014,** *48*, (15), 8289-97.
191. McManus, K.; Ertel, D.; Bogdan, J.; LLC, E. R., A Sustainable Choice for Water Treatment:Recycling When Injection is Not an Option or Water Supply is Limited. *Society of Petroleum Engieners* **2015**.
192. Wunz, C. E. *Summary of Shale Gas Wastewater Treatment and Disposal In Pennsylvania*; Wunz Associates, LLC: 2015.
193. Hussain, A.; Minier-Matar, J.; Gharfeh, S.; Janson, A.; Adham, S., Advanced Technologies for Produced Water Treatment. **2014**.
194. Lyons, B. *Produced Water Asset or Waste?*; Atlantic Council Energy & Environment Program: Washington DC, 2014.
195. Jiang, M.; Hendrickson, C. T.; VanBriesen, J. M., Life cycle water consumption and wastewater generation impacts of a Marcellus shale gas well. *Environmental science & technology* **2014,** *48*, (3), 1911-20.
196. Veil, J. A., Water Management Technologies Used by Marcellus Shale Gas Producers. **2010**.
197. Dinske, C.; Shapiro, S. A., Seismotectonic state of reservoirs inferred from magnitude distributions of fluid-induced seismicity. *Journal of Seismology* **2012,** *17*, (1), 13-25.
198. Veil, J. *Water Management Technologies used by Marcellus Shale Gas Producers*; Argonne National Laboratory: 2010.
199. Kim, W.-Y., Induced seismicity associated with fluid injection into a deep well in Youngstown, Ohio. *Journal of Geophysical Research: Solid Earth* **2013,** *118*, (7), 3506-3518.
200. Getzinger, G. J.; O'Connor, M. P.; Hoelzer, K.; Drollette, B. D.; Karatum, O.; Deshusses, M. A.; Ferguson, P. L.; Elsner, M.; Plata, D. L., Natural Gas Residual Fluids: Sources, Endpoints, and Organic Chemical Composition after Centralized Waste Treatment in Pennsylvania. *Environmental science & technology* **2015,** *49*, (14), 8347-55.
201. Shoaei, F. Development of graphical user interface tools for optimal fluid management in shale oil and gas operations. Colorado State University, Fort Collins, CO, 2015.
202. Thorn, T. H., *Gas Shale Environmental Issues and Challenges*. John Wiley & Sons, Inc.: 2015.
203. Wilson, J. M.; Wang, Y.; VanBriesen, J. M., Sources of high total dissolved solids to drinking water supply in southwestern Pennsylvania. *Journal of Environmental Engineering* **2014,** *140*, (5).
204. Zammit, K. *Use of Produced Water in Recirculating Cooling Systems at Power Generating Facilities*.
205. Hansen, E.; Mulvaney, D.; Betcher, M. *Water Resource Reporting and Water Footprint from Marcellus Shale Development in West Virginia and Pennsylvania*; Earthworks Oil & Gas Accountability Project: 2013.
206. Mueller, D. K.; Helsel, D. R. Nutrients in the Nation's Waters; too much of a good thing? http://pubs.usgs.gov/circ/circ1136/circ1136.html
207. Keister, T., Conversion of marcellus production wastewater into salable products. **2014**.
208. Abualfaraj, N.; Gurian, P. L.; Olson, M. S., Characterization of Marcellus Shale Flowback Water. *Environmental Engineering Science* **2014**, 140716083132007.
209. *Oil and Gas Reporting Electronic Production and Waste Reporting Guide*; Pennsylvania Department of Environmental Protection, Office of Oil and Gas Management: 2016.

210. Drilling Slowdown – What to Do With Excess Produced Water? *Marcellus Drilling News* 2016.
211. Litvak, A., As rig counts decrease, wastewater floods the industry. *Pittsburgh Post-Gazette* 2016.
212. Curtright, A. E.; Giglio, K. *Coal Mine Drainage for Marcellus Shale Natural Gas Extraction*; The RAND Corporation: 2012.
213. Karapataki, C. Techno-Economic Analysis of Water Management Options for Unconventional Natural Gas Developments in the Marcellus Shale. MIT, 2012.
214. Weber, C. L.; Clavin, C., Life cycle carbon footprint of shale gas: review of evidence and implications. *Environmental science & technology* **2012,** *46*, (11), 5688-95.
215. Tiemann, M.; Folger, P.; Carter, N., Shale Energy Technology Assessment: Current and Emerging Water Practices. *Congressional Research Service* **2014**.
216. Veil, J. A.; Puder, M. G. *Offsite Commercial Disposal of Oil and Gas Exploration and Production Waste: Availability, Options and Costs*; US Department of Energy, Office of Fossil Energy and National Energy Technology Laboratory: 2006.
217. Parker, K. M.; Zeng, T.; Harkness, J.; Vengosh, A.; Mitch, W. A., Enhanced formation of disinfection byproducts in shale gas wastewater-impacted drinking water supplies. *Environmental science & technology* **2014,** *48*, (19), 11161-9.
218. Kitwadkar, A. H. GIS based location optimization for mobile produced water treatment facilities in shale gas operations. Colorado State University, Fort Collins, CO, 2014.
219. Farag, A. M.; Harper, D. D., A review of environmental impacts of salts from produced waters on aquatic resources. *International Journal of Coal Geology* **2014,** *126*, 157-161.
220. In *Code 95.10*, Pennsylvania, 2010.
221. PA DEP Electronic Discharge Monitoring Report Database.
222. PA DEP Oil and Gas Reporting Website.
223. Johnson, K., Underground Injection Control Program. In Chief Ground Water & Enforcement Branch, Philadelphia Regional office, EPA: PennState Extension, 2015.
224. Rozell, D. J.; Reaven, S. J., Water pollution risk associated with natural gas extraction from the Marcellus Shale. *Risk analysis : an official publication of the Society for Risk Analysis* **2012,** *32*, (8), 1382-93.
225. Ferguson, G., Deep Injection of Waste Water in the Western Canada Sedimentary Basin. *Groundwater* **2015,** *53*, (2), 187-194.
226. Induced Seismicity Potential in Energy Technologies. *National Research Council* **2013**.
227. Weingarten, M.; Ge, S.; Godt, J. W.; Bekins, B. A.; Rubinstein, J. L., High-rate injection is associated with the increase in U.S. mid-continent seismicity. *Science* **2015,** *348*, (6241).
228. Lee, J.-Y.; Weingarten, M.; Ge, S., Induced seismicity: the potential hazard from shale gas development and CO2 geologic storage. *Geosciences Journal* **2015,** *20*, (1), 137-148.
229. Ellsworth, W. L., Injection-Induced Earthquakes. *Science* **2013,** *341*, (6142).
230. *Minimizing and Managing Potential Impacts of Injection-Induced Seismicity from Class II Disposal Wells: Practical Approaches*; Underground Injection Control National Technical Workgroup, U.S. Environmental Protection Agency: Washington, DC, 2014.
231. Folger, P.; Tiemann, M. *Human-Induced Earthquakes from Deep-Well Injection: A Brief Overview*; Congressional Research Service: 2015.
232. Petersen, M. D.; Mueller, C. S.; Moschetti, M. P.; Hoover, S. M.; Llenos, A. L.; Ellsworth, W. L.; Michael, A. J.; Rubinstein, J. L.; McGarr, A. F.; Rukstales, K. S., 2016 One-Year

Seismic Hazard Forecast for the Central and Eastern United States from Induced and Natural Earthquakes. **2016**.
233. Regional Earthquake Response Plan for Central Oklahoma and Expansion of the Area of Interest. 2016.
234. Executive Order 2012-09K. In Governor, S. o. O., Ed. 2012.
235. McManus, K.; Ertel, D.; Bogdan, J., Recycling When Injection is Not an Option or Water Supply is Limited. *Society of Petroleum Engieners* **2015**.
236. Zammit, K. *Use of Produced Water in Recirculating Cooling Systems at Power Generating Facilities* EPRI: Palo Alto, CA, 2006.
237. Department of Environmental Protection, B. o. W. M. D. o. M. a. R. W., General Permit WMGR123 Processing and Beneficial Use of Oil and Gas Liquid Waste. In 2012.
238. Steinzor, N.; Baizel, B. *Wasting Away: Four states' failure to manage gas and oil feld waste from the Marcellus and Utica Shale April 2015*; EarthWorks: 2015.
239. Veil, J. A.; Pude, M. G.; Elcock, D.; Redweid, R. J. J. *A white paper describing produced water from oil, natural gas, and coal bed methane*; U.S. Department of Energy National Energy Technology Laboratory: 2004.
240. Wilson, J. M.; VanBriesen, J. M., Oil and Gas Produced Water Management and Surface Drinking Water Sources in Pennsylvania. *Evironmental Practice* **2012,** *14*, (4).
241. Brost, D. F. *Water Quality Monitoring at the Kern River Field*; ChevronTexaco: 2002.
242. Torres, L.; Yadav, O. P.; Khan, E., A review on risk assessment techniques for hydraulic fracturing water and produced water management implemented in onshore unconventional oil and gas production. *The Science of the total environment* **2016,** *539*, 478-93.
243. Harkness, J. S.; Dwyer, G. S.; Warner, N. R.; Parker, K. M.; Mitch, W. A.; Vengosh, A., Iodide, bromide, and ammonium in hydraulic fracturing and oil and gas wastewaters: environmental implications. *Environmental science & technology* **2015,** *49*, (3), 1955-63.
244. Shih, J. S.; Saiers, J. E.; Anisfeld, S. C.; Chu, Z.; Muehlenbachs, L. A.; Olmstead, S. M., Characterization and Analysis of Liquid Waste from Marcellus Shale Gas Development. *Environmental science & technology* **2015,** *49*, (16), 9557-65.
245. Standing Stone Facility. http://eureka-resources.com/facilities/standing-stone-plant/ (3/2016),
246. Class II Brine Injection Wells of Ohio. http://oilandgas.ohiodnr.gov/portals/oilgas/pdf/Class II Brine Injection Wells of Ohio 03102016.pdf
247. Skone, T.; Cooney, G.; Jamieson, M.; Littlefield, J.; Marriott, J. *Life Cycle Greenhouse Gas Perspective on Exporting Liquefied Natural Gas from the United States*; US Department of Energy; National Energy Technology Laboratory: 2014.
248. Life cycle greenhouse gas persepctive on exporting liquefied natural gas from the United States. http://energy.gov/fe/life-cycle-greenhouse-gas-perspective-exporting-liquefied-natural-gas-united-states (11/2014),
249. EIA *International Energy Outlook 2013*; U.S. Energy Information Administration: 2013.
250. IEA *Gas Medium-Term Market Report; Market trends and projections to 2018*; International Energy Agency: 2013.
251. Hardisty, P. E.; Clark, T. S.; Hynes, R. G., Life Cycle Greenhouse Gas Emissions from Electricity Generation: A Comparative Analysis of Australian Energy Sources. *Energies* **2012,** *5*, 872-897.

252. Tamura, I.; Tanaka, T.; Kagajo, T.; Kuwabara, S.; Yoshioka, T.; Nagata, T.; Kurahashi, K.; Ishitani, H., Life cycle CO2 analysis fo LNG and city gas. *Applied Energy* **2001**.
253. Okamura, T.; Furukawa, M.; Ishitani, H., Future forecast for life-cycle greenhouse gas emissions of LNG and city gas 13A. *Applied Energy* **2007,** *84*, (11), 1136-1149.
254. Verbeek, R.; Kadijk, G.; Van Mensch, P.; Wulffers, C.; Van Den Beemt, B.; Fraga, F. *Environmental and Economic aspects of using LNG as a fuel for shipping in The Netherlands*; TNO: 2011.
255. Arteconi, A.; Brandoni, C.; Evangelista, D.; Polonara, F., Life-cycle greenhouse gas analysis of LNG as a heavy vehicle fuel in Europe. *Applied Energy* **2010,** *87*, (6), 2005-2013.
256. *Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis*; Interagency Working Group on Social Cost of Carbon: 2014.
257. Tol, R. S. J., The Social Cost of Carbon. *Annual Review of Resource Economics* **2011,** *3*, (1), 419-443.
258. Marten, A. L.; Kopits, E. A.; Griffiths, C. W.; Newbold, S. C.; Wolverton, A., Incremental CH4 and N20 mitigation benefits consistent with the US Government's SC-CO2 estimates. *Climate Policy* **2015,** *15*, (2).
259. Heath, G. A.; O'Donoughue, P.; Arent, D. J.; Bazilian, M., Harmonization of initial estimates of shale gas life cycle greenhouse gas emissions for electric power generation. *Proceedings of the National Academy of Sciences of the United States of America* **2014,** *111*, (31), E3167-76.
260. Stephenson, T.; Valle, J.; Riera-Palou, X., Modeling the relative GHG emissions of conventional and shale gas production. *Environmental science & technology* **2011,** *45*, (24), 10757–10764.
261. Heath G, M. J., Fisher N *Chapter 1: Life cycle greenhouse gas emissions from Barnett shale gas used to generate electricity: Natural Gas and the Trans- formation of the U.S. Energy Sector: Electricity, eds Logan J, et al. (National Renew- able Energy Laboratory, Golden, CO), NREL/TP-6A50-55538.*; 2012.
262. Laurenzi, I.; Jersey, G., Life cycle greenhouse gas emissions and freshwater consumption of Marcellus shale gas. *Environmental science & technology* **2013,** *47*, (9), 4896–4903.
263. Heede, R., LNG Supply Chain Greenhouse Gas Emissions for the Cabrillo Deepwater Port- Natural Gas from Australia to California. **2006**.
264. *LCFS Life Cycle Fuel Pathway Report Method 2B Application: Topock LNG Plant to LNG and L-CNG Applied Natural Gas Fuels, Inc*; Applied Natural Gas Fuels, Inc.
265. Cohen, A. K. *The Shale Gas Paradox- Assessing the Impacts of the Shale Gas Revolution on Electricity Markets and Climate Change*; Harvard Kennedy School: 2013.
266. Biswas, W.; Engelbrecht, D.; Rosano, M. *A life cycle assessment of Western Australian LNG production and export to the Chinese market*; Western Australian Energy Research Alliance (WAERA): 2011.
267. Yoon, S. Y.; Yamada, T., Life-cycle inventory analysis on fossil energy in Japan. *Energy Economics* **1999,** *25*, (8), 22-48 [in Japanese].
268. Yost, C.; DiNapoli, R., Benchmarking Study Compares LNG Plant Costs. *Oil & Gas Journal* **2003,** *101*, (27).
269. Barnett, P. Life Cycle Assessment (LCA) of Liquefied Natural Gas (LNG) and its environmental impact as a low carbon energy source. University of Southern Queensland, 2010.

270. Port distance calculator. http://www.sea-distances.org/
271. Venkatesh, A.; Jaramillo, P.; Griffin, W. M.; Matthews, H. S., Uncertainty in life cycle greenhouse gas emissions from United States coal. *Energy & Fuels* **2012,** *26*, (8), 4917-4923.
272. Steinmann, Z. J.; Venkatesh, A.; Hauck, M.; Schipper, A. M.; Karuppiah, R.; Laurenzi, I. J.; Huijbregts, M. A., How to address data gaps in life cycle inventories: a case study on estimating CO2 emissions from coal-fired electricity plants on a global scale. *Environmental science & technology* **2014,** *48*, (9), 5282-9.
273. eGRID Database. (10/26/14),
274. Venkatesh, A.; Jaramillo, P.; Griffin, W. M.; Matthews, H. S., Implications of changing natural gas prices in the United States electricity sector for SO2, NOXand life cycle GHG emissions. *Environmental Research Letters* **2012,** *7*, (3), 034018.
275. Bohnengel, B.; Patiño-Echeverri, D.; Bergerson, J., Environmental Implications of United States Coal Exports: A Comparative Life Cycle Assessment of Future Power System Scenarios. *Environmental science & technology* **2014,** *48*, (16), 9908-9916.
276. CEQ;, W. H. Revised Draft Guidance for Greenhouse Gas Emissions and Climate Change Impacts. http://www.whitehouse.gov/administration/eop/ceq/initiatives/nepa/ghg-guidance
277. Office of the Press Secretary Administration Takes Steps Forward on Climate Action Plan by Announcing Actions to Cut Methane Emissions. http://www.whitehouse.gov/the-press-office/2015/01/14/fact-sheet-administration-takes-steps-forward-climate-action-plan-anno-1 (1/29/2015),
278. Pan, J.; Phillips, J.; Chen, Y., China's balance of emissions embodied in trade: approaches to measurement and allocating international responsibility. *Oxford Review of Economic Policy* **2008,** *24*, (2), 354-376.
279. Peters, G. P.; Hertwich, E. G., CO2 Embodied in International Trade with Implications for Global Climate Policy. *Environmental science & technology* **2008,** *42*, (5), 1401-1407.
280. Moore, F. C.; Diaz, D. B., Temperature impacts on economic growth warrant stringent mitigation policy. *Nature Climate Change* **2015,** *5*, 127-131.
281. Andrew, R. M.; Davis, S. J.; Peters, G. P., Climate policy and dependence on traded carbon. *Environmental Research Letters* **2013,** *8*, (3), 034011.
282. *BP Statistical Review of World Energy June 2015*; BP: 2015.
283. Kagawa, S.; Suh, S.; Hubacek, K.; Wiedmann, T.; Nansai, K.; Minx, J., CO2 emission clusters within global supply chain networks: Implications for climate change mitigation. *Global Environmental Change* **2015,** *35*, 486-496.
284. Kander, A.; Jiborn, M.; Moran, D. D.; Wiedmann, T. O., National greenhouse-gas accounting for effective climate policy on international trade. *Nature Climate Change* **2015,** *5*, (5), 431-435.
285. Sakai, M.; Barrett, J., Border carbon adjustments: Addressing emissions embodied in trade. *Energy Policy* **2016,** *92*, 102-110.
286. Larsen, H. N.; Solli, C.; Pettersena, J., Supply Chain Management – How can We Reduce our Energy/Climate Footprint? *Energy Procedia* **2012,** *20*, 354-363.
287. Giljum, S.; Dittrich, M.; Lieber, M.; Lutter, S., Global Patterns of Material Flows and their Socio-Economic and Environmental Implications: A MFA Study on All Countries World-Wide from 1980 to 2009. *Resources* **2014,** *3*, (1), 319-339.
288. Böhringer, C.; Löschel, A., Computable general equilibrium models for sustainability impact assessment: Status quo and prospects. *Ecological Economics* **2006,** *60*, (1), 49-64.

289. Manne, A. D.; Rutherford, T. F., International Trade in Oil, Gas and Carbon Emission Rights: An Intertemporal General Equilibrium Model. *The Energy Journal* **1994,** *15*, (1), 57-76.
290. Global Change Assessment Model (GCAM). http://www.globalchange.umd.edu/models/gcam/
291. Gielen, D.; Moriguchi, Y., CO2 in the iron and steel industry: An analysis of Japanese emission reduction potentials. *Energy Policy* **2002,** *30*, (10), 849-863.
292. JODI-Oil World Database. In.
293. Company Level Imports. http://www.eia.gov/petroleum/imports/companylevel/
294. UN Comtrade Database. In Nations, U., Ed. 2014.
295. Energy Balances of OECD Countries. In 2015.
296. Crude Oil Assay Database. In Speight, J. G., Ed. Knovel, 2015; Vol. 2016.
297. *LCFS Crude Oil Lifecycle Assessment*; California Environmental Protection Agency; Air Resources Board.
298. Malins, C.; Galarza, S.; Baral, A.; Kodjak, D.; Brandt, A. R.; El-Houjeiri, H.; Howorth, G.; Gabriel, T. *Upstream Emissions of Fossil Fuel Feedstocks for Transport Fuels Consumed in the European Union*; International Council on Clean Transportation to the European COmmission Directorate-General for Climate Action: Washington D.C., 2014.
299. *Monthly Crude Oil Imports*; International Energy Agency (IEA).
300. *Deloitte's OIl and Gas Price Forecast*; 2015.
301. Worldscale. https://http://www.worldscale.co.uk
302. *Review of Maritime Transport*; United Nations Conference on Trade and Development: New York and Geneva, 2013.
303. *Environmentally Conscious Transportation*. John Wiley & Sons, Inc.: 2008.
304. Devanney, J. *The Impact of Bunker Price on VLCC Spot Rates*; Martingale Inc: USA, 2009.
305. World Port Index. http://msi.nga.mil/NGAPortal/MSI.portal?_nfpb=true&_pageLabel=msi_portal_page_62&pubCode=0015
306. *A Guidebook for Forecasting Freight Transportation Demand*; Transportation Research Board; National Cooperative Highway Research Program: 1997.
307. Gordon, D.; Brandt, A. R.; Bergerson, J.; Koomey, J. *Know Your Oil: Creating a Global Oil-Climate Index*; Carnegie Endownment for International Peace: 2015.
308. OPGEE: the Oil Production Greenhouse gas Emissions Estimator. https://pangea.stanford.edu/researchgroups/eao/research/opgee-oil-production-greenhouse-gas-emissions-estimator
309. PRELIM: the Petroleum Refinery Life Cycle Inventory Model. http://www.ucalgary.ca/lcaost/prelim
310. Wang, M. *The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) Model;*; Argonne National Laboratory: 2009.
311. Nelson, W. L., Nelson complexity index. *Oil and Gas Journal* **1976**, (13), 20-27.
312. Elgowainy, A.; Han, J.; Cai, H.; Wang, M.; Forman, G. S.; DiVita, V. B., Energy efficiency and greenhouse gas emission intensity of petroleum products at U.S. refineries. *Environmental science & technology* **2014,** *48*, (13), 7612-24.
313. Johnston, D. C.; Johnston, D., *Introduction to Oil Company Financial Analysis*. PennWell Books: 2006.

314. Fenhann, J., Pledge Pipeline. In (UNEP), U. N. E. P., Ed. 2016.
315. CAIT Climate Data Explorer. http://cait.wri.org/
316. Global Greenhouse Gas Emissions Data. https://www3.epa.gov/climatechange/ghgemissions/global.html