Wells to Wheels: Environmental Implications of Natural Gas As A Transportation Fuel

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Abstract: We assessed freshwater consumption, greenhouse gas (GHG) emissions, and air emissions of using compressed and liquefied natural gas (NG) as transportation fuels by three heavy-duty NG vehicles (NGV) types from a wells-to-wheels (WTW) perspective. We analyzed freshwater consumption for NG production in major U.S. shale gas plays from recent reports and studies. We reviewed recent literature quantifying methane leakage from the NG supply chain and vehicle use to improve the estimates of NGV GHG emissions. Results show that NGVs

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could reduce freshwater consumption significantly and offer air emissions reduction benefits compared to their diesel counterparts. NGV WTW GHG emissions are largely driven by the vehicle fuel efficiency, as well as methane leakage rates of both the NG supply chain and vehicle end use: we estimate WTW GHG emissions of NGVs to be slightly higher than those of the diesel counterparts given the estimated WTW methane leakage. We found that the cost-effectiveness of NGVs is impacted by incremental cost of NG storage tanks and the price difference between NG and diesel fuels. These findings for NGVs shed light on their environmental and economic impacts from a WTW holistic point of view.

1. Introduction

In 2015, natural gas (NG) was the most produced energy source in the U.S., accounting for 32% of the total U.S. energy production (U.S. Energy Information Administration, 2016a). NG production in the U.S. has increased from 23.5 trillion cubic feet (TCF) in 2006 to 32.9 TCF in 2015 (U.S. Energy Information Administration, 2016b). This is largely due to production of NG from shale formations through the advent of horizontal drilling and high-volume hydraulic fracturing technologies over the past decade, which increased from 2.0 TCF in 2007 to 15.5 TCF in 2015, accounting for 47% of U.S. natural gas production currently (U.S. Energy Information Administration, 2016b). The shale gas boom is primarily a U.S. phenomenon as Canada and China are the only other countries that are producing commercial volumes of shale gas (U.S. Energy Information Administration, 2016c). A large portion of the U.S. shale gas production growth has occurred in the Marcellus play, where production reached an average of 5.7 TCF in 2015 (U.S. Energy Information Administration, 2016d), as shown in Fig. 1. Fig. B.1 shows the geographic distribution of major shale plays in the U.S. (U.S. Energy Information Administration, 2016e).

Abundant gas reserves and increased production provides an opportunity to expand the use of NG in various end-use sectors in the U.S. In 2015, NG is the second most consumed energy source (29%) in the U.S., of which 35% was consumed in the electric power sector, followed by the industrial sector (33%), residential and commercial sector (28%), and the transportation sector (3%). The transportation sector has been explored as a growth sector for NG, as it consumes predominantly petroleum-based fuels and accounts for 28% of the total primary energy consumption in the U.S. (U.S. Energy Information Administration, 2016f). As the NG cost has been consistently low lately, significant research and development has been undertaken to transform it to fuels, such as Fischer-Tropsch diesel, dimethyl ether, and hydrogen. In addition, significant efforts have been made to overcome technical and market barriers of its use as compressed natural gas (CNG) and liquefied natural gas (LNG). Research needs for impact analysis and assessment of expanding the use of NG fuels on energy consumption, emissions, energy security, and ownership costs have been highlighted (Wang et al., 2015a).

The objective of this study is to perform a wells-to-wheels (WTW) analysis of freshwater consumption, GHG emissions, air pollutant emissions, and ownership costs of CNG and LNG vehicles in the heavy-duty vehicle (HDV) sector. The WTW system boundary includes the wells-to-pump (WTP), which includes the entire NG supply chain that involves with production, processing, transmission, storage, and distribution of NG to CNG or LNG plants, as well as the pump-to-wheels (PTW) including refueling and use of CNG or LNG fuels by vehicles. This study provides important results on the environmental, technological, and economic impacts of introducing NGVs in the U.S. heavy-duty vehicle sector, which represents the second largest and fastest growing share of transportation energy demand (U.S. Energy Information Administration, 2016g). In this analysis, we examined three key NGV markets: refuse trucks, transit buses, and freight trucks (TIAX, 2013), and two types of NG fuels, i.e., CNG and LNG.

1.1 Freshwater consumption

The process of hydraulic fracturing (HF, also known as “fracking”)” uses large quantities of high pressure fluid to create fractures in a rock formation thousands of feet below the earth’s surface and facilitate the release of shale gas and liquid (EPA, 2016). More than 90% of the HF fluid volume is water, and the remaining portion usually consists of proppant (2%-10% of the total volume, mostly sand) and additive chemicals (U.S. Environmental Protection Agency, 2016a; U.S. Department of Energy, 2009). Although the amount of HF fluid could vary by the shale formation and the quality of local water, it has been estimated that a typical well may require 2-10 million gallons of water for HF completion (Harper, 2008; Ernstoff and Ellis, 2013; Yang et al., 2015). As the development of shale gas in the U.S. intensifies, concerns regarding its direct environmental impacts, especially on the consumption and contamination of the surface and ground water, have also increased.

The characteristics of various shale formations can lead to different estimated ultimate recovery (EUR) of shale gas. As the amount of HF water is closely associated with the production of shale gas and liquid, the water consumption factor (WCF) metric has been introduced to reflect the quantity of water directly consumed per energy unit of the final fuel products (Lampert et al., 2016; Clark et al., 2013). However, the EUR is an estimation of the quantity of fossil energy that is recoverable from a well, which could vary due to time, method, and technology of the exploration and estimation. For instance, U.S. Energy Information Administration (EIA) estimated the EUR of the Marcellus play at 0.39 billion cubic feet (BCF)/well (average of 1200 wells) in 2008, which increased to 5.78 BCF/well (average on 2191 wells) in 2012 (U.S. Energy Information Administration, 2016j; Staub, 2015). The US Geological Survey assessed the Jurassic and Cretaceous rocks of the US Gulf Coast back in 2010 and estimated a mean of 70.4 TCF of NG was contained in the Bossier and Haynesville Formations. In an updated 2017 USGS assessment, the same regions were estimated to contain a mean of 304.4 TCF of NG, in addition to 4.0 billion barrels of oil and 1.9 billion barrels of natural gas liquid (U.S. Geological Survey, 2017). Therefore, a sensitivity analysis of the WCFs affected by the EUR variations seems necessary for understanding the overall impact on water depletion by the shale gas pathways.

1.2 GHG emissions

Argonne National Laboratory has been evaluating the NG fuel pathways with a focus on quantifying the methane leakage of the NG supply chain and its impacts on WTW GHG emissions of NG fuel pathways and NGVs (Wang and Huang, 2000; Burnham et al., 2012; Burnham, 2016). The National Research Council discussed GHG emissions from CNG use primarily based on previous Argonne studies and the Greenhouse gases, Regulated Emissions, and Energy use in Transportation (GREET) model (National Research Council, 2013). Rose et al. (2013) found that significant reductions in life-cycle GHG and criteria air pollutant emissions were obtained by CNG refuse trucks compared to the diesel counterparts based on real-time operational data in British Columbia, Canada. WTW GHG emissions of light-duty, medium-duty, and heavy-duty vehicles powered by NG-derived fuels, including CNG, LNG, electricity, and hydrogen were examined and compared to their gasoline and diesel counterparts, and vehicle fuel efficiency and methane leakage rate of the NG supply chain were found to be major drivers to the relative GHG emission performances of NG-based vehicles (Tong et al., 2015).

Recently, new data covering previously overlooked emission sources of methane leakage associated with the NG supply chain, such as gas gathering plants, have been collected and provide a new opportunity to examine the GHG emission impacts of the NG industry (Zavala-Araiza et al., 2015; Littlewood et al., 2016). In an effort to reconcile divergent estimates of methane leakage from natural gas production through distribution obtained from top-down and bottom-up measurements, Zavala-Araiza et al. (2015) refined sampling techniques for top-down estimates and accounting methodologies for bottom-up estimates. Zavala-Araiza et al. (2015) found that estimates in the Barnett shale oil and gas play using both approaches converged with a 10% difference. While the methane leakage represented about 1.5% of natural gas production, a small proportion of high-emitters were found responsible for half of total methane emissions. Lately, a synthesis of methane emission data from a series of ground-based field measurements shows that 1.7% of the methane in natural gas is emitted from the production through distribution natural gas supply chain (Littlewood et al., 2016). Extending this natural gas supply chain to fuel production, refueling, and end use, recent findings on CNG and LNG refueling stations and vehicle methane emissions have identified a major information gap on the life-cycle performance of NGVs (Clark et al., 2016). The impact of these station and vehicle emissions is not well known and needs to be assessed on a WTW basis.

1.3 NO_x and PM emissions

One of the original drivers of NGV development has been its ability to have low air pollutant emissions. Diesel HDVs contribute substantially to nitrogen oxides (NO_x) and particulate matter (PM) emissions and to the resultant air quality effects of ground-level ozone formation driven by NO_x and the adverse health impacts of PM (Nelson et al., 2008; Peretz et al., 2008; Pope, 2004). In 2016, 211 U.S. counties with 119 million people, about 40% of the U.S. population, were in ozone nonattainment (i.e., not meeting the U.S. air quality standards) (U.S. Environmental Protection Agency, 2016b). Twelve counties in California's South Coast and San Joaquin air basins with a total population of 20 million were designated as "extreme" ozone nonattainment areas (U.S. Environmental Protection Agency, 2016b). These areas had ozone values 233% higher (175 ppb) than the previous standard of 75 ppb. In 2015, the U.S. Environmental Protection Agency (EPA) revised the ozone standard to 70 ppb, which results in 241 additional counties not meeting the standard based on 2012-2014 air quality data.

In 2016, 20 counties with 23 million people were in $PM_{2.5}$ (PM with aerodynamic diameters less than 2.5 micrometers) nonattainment areas with respect to the most recent 2012 annual standard. In 2016, additional areas were in non-attainment of previous $PM_{2.5}$ standards as 35 counties with 13 million people were in nonattainment of the 2006 24-hour $PM_{2.5}$ standard and 13 counties with 3 million people were in nonattainment of the 1997 annual $PM_{2.5}$ standard (U.S. Environmental Protection Agency, 2016b). In total 68 counties with 39 million people are in $PM_{2.5}$ non-attainment in 2016.

These situations have led to increasingly stringent standards for NO_x and PM emissions, with the EPA tightening the HDV engine standards for both pollutants by ~98% from 1988 to 2010. Currently, the EPA and California Air Resources Board (CARB) standards for NO_x and PM are 0.2 g/bhp-hr (break horsepower-hour) and 0.01 g/bhp-hr, respectively. Because of severe air quality concerns in California, CARB adopted optional low NO_x standards in 2014, with three levels to which engines can be certified: 0.10, 0.05, or 0.02 g/bhp-hr. In 2016, the South Coast Air Quality Management District along with 10 other state and local agencies, petitioned the EPA to revise the EPA HDV NO_x standard to be 0.02 g/bhp-hr (South Coast Air Quality Management District, 2016). The petition states that to meet the new 70 ppb ozone standard, these areas will need the lower HDV engine standard for NO_x emissions.

The first heavy-duty engines to meet both the EPA/CARB 2010 standards and the CARB optional 0.02 g/bhp-hr NO_x standard were natural gas powered 8.9-liter (L) engines, suited for refuse, transit, and freight (below 66,000 pounds gross vehicle weight) applications (Cummins Westport Inc, 2016). However, to meet these standards, NGVs rely on the integration of sophisticated engine controls and aftertreatment devices and not just the inherent qualities of the fuel. While all engines sold today meet the EPA/CARB 2010 standards, to fully understand the relative emissions of NGVs as compared to diesel vehicles, analysis of in-use performance is crucial (Nylund and Koponen, 2012; Miller et al., 2013; Yoon et al., 2013; Carder et al., 2014; Wang et al., 2015b; Quiros et al., 2016; Johnson et al., 2016; Anenberg et al., 2017; Sandhu et al., 2017).

1.4 Economics

The rapid increase in NG production from shale formations has led to significant interest in its use for transportation. NGVs typically have lower and more stable fuel costs than their conventional counterparts (U.S. Department of Energy, 2017a). However, due to the cost of NG storage tanks, NGVs cost significantly more than gasoline and diesel vehicles. In recent years, the focus on heavy-duty NGVs in lieu of light-duty vehicles has been driven by economics (Rood Werpy et al., 2010; Krupnick, 2011; Deal, 2012; Gao et al., 2013). Many heavy-duty vehicles are large fuel users that are able to pay back the incremental vehicle cost via the use of lower cost NG fuel (Johnson, 2010; Deal, 2012; National Petroleum Council 2012; Laughlin and Burnham, 2014; Jaffe et al. 2015). The literature has found that the major factors that drive the payback of NGVs versus diesels include incremental vehicle cost, fuel price differential, vehicle usage, and relative fuel efficiency.

2. Methodology and Data

We defined a WTW system boundary to estimate the GHG emissions, freshwater consumption, air pollutant emissions, and costs of NGVs, as shown in Fig. 2. The NGVs we analyzed include a Class 8 LNG combination short-haul truck, a Class 8 CNG transit bus, and a Class 8 CNG refuse truck, in comparison to their diesel counterparts. These NGVs represent the major niche markets that heavy-duty NGVs have entered in the U.S. (Cai et al., 2015b). The functional unit used is one ton of freight cargo moved for one -mile (ton-mile) by HDVs, which is a common metric used to evaluate energy and environmental performances of HDVs (NRC, 2010; Tong et al., 2015a). Table A.1 shows the engine type, vehicle application, fuel type, and fuel source considered in this analysis.

We employed two models to conduct the WTW analysis: 1) The GREET model (Argonne National Laboratory, 2016) for GHG emissions, freshwater consumption, and air emissions modelling, and 2) the Alternative Fuel Life-Cycle Environmental and Economic Transportation (AFLEET) model for costs analysis.

2.1 Freshwater consumption

WTW freshwater consumption of NGVs is estimated with GREET, which calculates the direct process water consumption and indirect water consumption associated with production, processing, transportation, and distribution of NG, CNG, and LNG fuel production and distribution. We analysed direct water consumption factors associated with shale NG production and used them to calculate the direct water consumption of the NG supply chain. We also used the water consumption factors characterized for various fuel production pathways, e.g. electricity, gasoline and diesel fuels (Lampert et al., 2016) to calculate indirect water consumption associated with the NG supply chain and the production of CNG/LNG.

Freshwater is consumed during the initial well development of conventional NG production and during horizontal drilling and HF for shale gas production. US shale gas production from HF increases the water intensity relative to conventional gas production (Lampert et al., 2016). To assess the water consumption of NG production, we analyzed the latest well-specific water consumption data available for shale gas production. For water consumption associated with the conventional NG production, we used the water consumption factors we determined earlier (Lampert et al., 2016).

Eight major regions of shale gas production in the U.S. were selected in this study: Marcellus, Utica, Barnett, Eagle Ford, Bakken, Fayetteville, Haynesville, and Woodford. States and counties within the boundary of each region were identified according to EIA's Drilling Productivity Report (U.S. Energy Information Administration, 2016d) and "Major tight oil and shale gas plays in lower 48 States" map (U.S. Energy Information Administration, 2016h). One-time HF water consumption (gal/well) for 6565 newly fractured wells in the eight shale gas plays in 2015 were extracted from FracFocus (FracFocus, 2016). Among the eight plays, Eagle Ford (2191), Bakken (1403), and Marcellus (1158) had drilled the most new wells.

Because FracFocus reporting was voluntary for many states and operators, duplicated entries, incomplete entries, and typos were first removed from all regions, which accounted for 34% of the total reported entries. Secondly, re-entries that might indicate re-fracturing activities in FracFocus were also identified (approximately 5% of the total reported entries) but eliminated because lack of clarity of their physical meanings in FracFocus. Moreover, even though the voluntary reports in FracFocus might not include all newly fractured shale wells in the U.S., the average HF water consumption of the wells available in each region was representative due to the normal distribution of reported wells and results from previous studies (Chen and Carter, 2016; Clark et al., 2013; Kondash and Vengosh, 2015; U.S. Environmental Protection Agency, 2015a, 2015b).

Each well co-produces gas and oil with varying fractions. Thus, shale gas-specific water consumption requires allocation of well-level water consumption between shale gas and shale oil production. Multiple co-product methods may be used for this purpose with its own strengths and limitations (Wang et al., 2011). We chose the energy-based allocation method because shale gas and shale oil are both energy products. Since the FracFocus data does not specify the production of gas and oil from each shale well, we applied the energy-based allocation method based on the energy outputs of total dry shale gas and unconventional oil products in each region where each well was located to calculate the HF water consumption associated with dry shale gas production only, as shown in Eq. (2):

$$V_{shale\ gas} = V_{overall} \times \frac{E_{shale\ gas}}{E_{overall}} \qquad \text{Eq. (2)}$$

where $V_{shale\ gas}$ is the well-specific HF water consumption that is allocated to dry shale gas only; $V_{overall}$ is the overall HF water consumption to produce both shale gas and oil; $E_{shale\ gas}$ is the energy output of the total shale gas; and $E_{overall}$ is the energy output of the overall shale gas and oil production (U.S. Energy Information Administration, 2016i).

We obtained the $E_{shale\ gas}$ and $E_{overall}$ in each region in 2015 from the EIA (U.S. Energy Information Administration, 2016i). EIA does not report either dry shale gas or shale oil production in Fayetteville or Barnett. For these plays, the allocation method was based on the ratio of shale gas wells out of all HF wells in the FracFocus report (U.S. Environmental Protection Agency, 2015b) and the share of dry shale gas production (U.S. Energy Information Administration, 2016j) of the gross shale gas withdrawal, as shown in Eq. (3):

$$V_{shale\ gas} = V_{overall} \times \frac{N_{shale\ gas\ well}}{N_{total\ well}} \times \frac{E_{dry\ shale\ gas}}{E_{gross\ withdrawal}} \quad \text{Eq. (3)}$$

where $\frac{N_{shale\ gas\ well}}{N_{total\ well}}$ is the fraction of shale gas well count among all HF wells reported in EPA's previous study on FracFocus (U.S. Environmental Protection Agency, 2015b) (data covered the period between January 2011 and February 2013); $E_{dry\ shale\ gas}$ is the U.S. average of the dry shale gas energy (U.S. Energy Information Administration, 2016i); and $E_{gross\ withdrawal}$ is the U.S. average of the gross withdrawn energy from shale gas wells (U.S. Energy Information Administration, 2016k).

The energy-based allocation factor for shale gas production ranges from about 13% for Bakken to about 100% in Haynesville and Fayetteville in 2015. After allocation, the plays that consumed the largest amount of HF water for shale gas production were Marcellus (9.17×10^6 gal/well), Utica (8.98×10^6 gal/well), and Haynesville (8.71×10^6 gal/well), which are all gas-dominant plays, while Bakken (0.52×10^6 gal/well), Barnett (2.08×10^6 gal/well), and Eagle Ford (2.84×10^6 gal/well), which are either shale oil-dominant plays (Bakken and Eagle Ford) or produced significantly more shale oil (Barnett) than other plays, consumed the smallest amount of HF water. This trend, as shown in Fig. B.2, agrees with what (Gallegos et al., 2015) found that HF of shale gas reservoirs often utilizes fluid with high water content (slickwater) whereas HF fluid for tight oil reservoirs contains more gel and less water. The variability of HF water consumption among plays are also attributable to other factors such as geologic characteristics, machinery technology, and vertical and lateral length of the well (Chen and Carter, 2016).

The volume of drilling water is considerably small compared to that of HF water (Clark et al., 2013; Horner et al., 2016), but it may vary by region due to the differences in geologic characteristics, drilling technology, etc. In this study, region specific drilling-to-HF water consumption ratios (Clark et al., 2013; Mantell, 2013, 2011), as shown in Table B.1, were adopted to calculate the average volume of drilling water use. Energy-based allocation of drilling water ranges from 0.02×10^6 gal (Bakken) to 0.52×10^6 gal (Haynesville) per well, with a production-weighted U.S. average at 0.20×10^6 gal per well. High drilling water usage in Haynesville has been previously reported (Clark et al., 2013; Mantell, 2013, 2011). As for Fayetteville, significantly shallower (4315 ft on average) and less dispersed true vertical depth of studied wells might have been one of the reasons of its relatively low water requirement for drilling.

The WCFs for drilling and HF represent the water consumed to produce a unit of energy output of such operations. We calculated the WCFs from the play-specific water consumption (gal/well, for both drilling and HF) divided by the average estimated ultimate recovery (EUR) of each shale gas play.

We reviewed the gas EURs in each play estimated in previous studies. Variation in estimation of gas EURs mostly reflected different methodologies and historical production data used to predict the EURs. We chose EIA's estimation when it is available (Staub, 2015), which generally aligns well with the range of other estimates in literature, as summarized in Table B2.

The total WCF for shale gas production in 2015 that includes drilling and HF water consumption ranges from 0.9 (Eagle Ford) to 3.4 (Utica) gal/mmBtu, with a production-weighted average at

1.7 gal/mmBtu, as shown in Fig. 3. Some play-specific WCFs derived from this study are higher than previously reported values (Bibby et al., 2013; Gallegos et al., 2015; Kondash and Vengosh, 2015; Mantell, 2013); for instance, Kondash and Vengosh (2015) reported 0.88 gal/mmBtu WCF for Marcellus and 0.90 gal/mmBtu for Haynesville, which were 0.61 and 0.59 gal/mmBtu, respectively, lower than those from this study. Differences as such are mainly caused by the increases in the average length of the horizontal drilling and its associated HF water demand: FracFocus reported a median of 5.79 million gallon HF water per well was required for Marcellus formation in 2013, which jumped to 8.87 million gal/well in 2015; similarly, HF in Haynesville formation consumed a median of 4.85 million gallon of water per well in 2013 and 8.27 million gal/well in 2015 (FracFocus, 2016; Gallegos et al., 2015). One major reason causing this increase is the increasing annual HF water use for newly fractured wells: all data in referenced studies reflected HF activities prior to early 2013; however, the annual HF water use per well increased significantly in many plays between 2013 and 2015 (14% in Woodford, 51% in Eagle Ford, 52% in Marcellus, 55% in Haynesville, 90% in Utica, and 92% in Bakken, after energy-based allocation). Factors such as well characteristics, number of fracturing stages, production strategy, and oil price might have caused different water consumption intensities among different shale gas plays and induced the change in water requirement for fracturing overtime (Bibby et al., 2013; Chen and Carter, 2016; Murray, 2013; Scanlon et al., 2014). Note that while these WCFs represent an up-to-date estimation of the major shale gas plays in the U.S., these WCFs are conservative estimates because we assumed that the water is freshwater entirely due to the limitation of the FracFocus database that does not distinguish the water sources, which might include recycled flowback or produced water, in addition to freshwater.

2.2 GHG emissions

GREET calculates the WTW GHG emissions of carbon dioxide (CO_2), nitrous oxide, and methane (CH_4) based on their 100-year Global Warming Potentials as suggested by the Intergovernmental Panel for Climate Change in its Fifth Assessment Report for various fuel production pathways and vehicle operations on a holistic platform. CO_2 emissions from fuel combustion are calculated on a carbon mass balance basis, and non-combustion CO_2 emissions such as fugitive CO_2 emissions from NG fields are taken into account separately in GREET. For the NGVs modeled in this study, we applied a fuel combustion technology-based approach to estimating the upstream process fuel production and WTW GHG emissions of each life-cycle stage on per mile of vehicle driven basis, using Eq. (B.1).

We analyzed the EPA's 2016 greenhouse gas inventory (GHGI) (U.S. Environmental Protection Agency, 2016c) along with other sources to estimate the methane leakage associated with NG production, processing, transmission, and distribution as seen in Table C.1. The EPA's GHGI is updated annually to improve its methodology and include the latest data found in other analyses. For example, the 2016 EPA GHGI incorporated the latest Greenhouse Gas Emissions Reporting Program data to update its emission factors and activity data for well completions and workovers. When analyzing methane leakage, both emissions and NG throughput are needed to develop emission factors. Using data from the U.S. EIA, we estimate the NG throughput for NG production, processing, transmission, and distribution, as shown in Table C.2.

Table 1 summarizes the methane emissions as a percentage of the volumetric NG throughput by stage for both conventional NG and shale NG supply chains. The overall NG supply chain leakage of a unit of NG throughput is calculated based on the leakage rate of each supply chain stage (Burnham et al., 2015), using Eq. (3). Compared to previous EPA GHGI-based updates, these new updates show that the methane leakage rates for completion and workover have increased significantly, and the well equipment emissions also increased dramatically due to inclusion of emissions from gathering plants that were not considered in previous EPA GHGIs. However, the leakage from NG transportation and NG distribution has decreased significantly. As a result, we estimated an overall methane leakage rate of 1.32% and 1.34%, respectively, for the conventional and shale gas supply chain, which represent a slight increase compared to the estimates on the basis of the 2015 EPA GHGI, as shown in Fig. C.1, which is mostly due to inclusion of leakage at gas gathering plants in the 2016 EPA GHGI.

$$L_{overall} = \frac{\left[\frac{1}{(1-L_1)\times(1-L_2)\times\ldots\times(1-L_n)} - 1\right]}{\frac{1}{(1-L_1)\times(1-L_2)\times\ldots\times(1-L_n)}}$$

$$= 1-(1-L_1)\times(1-L_2)\times\ldots\times(1-L_n) \qquad \text{Eq. (3)}$$

Where: L_1, L_2, and L_n are the methane leakage rates on a given NG throughput basis at supply chain stage 1, 2, and n, respectively; and $L_{overall}$ is the overall methane leakage rate on an NG throughput basis for the entire NG supply chain.

Our estimated methane leakage rate based on 2016 EPA GHGI is compared to previous bottom-up measurement. For example, recent measurement of methane leakage from U.S. gas fields, gathering facilities, and processing plants (Marchese et al., 2015), from the U.S. natural gas transmission and storage system (Zimmerle et al., 2015) and from U.S. natural gas local distribution systems (Lamb et al., 2015) showed that the methane leakage rate of the overall U.S. natural gas supply chain was about 0.99%. Our estimation is also on par with and the recent synthesis of measurements that suggested a 1.7% leakage rate between extraction and distribution (Littlefield et al., 2017). On the other hand, top-down estimates from satellite remote sensing and aircraft measurement suggested the methane leakage rates of specific NG production fields could be much higher. For example, methane emissions estimate from airborne measurements over the Uintah Basin ranged from 6.2% to 11.7% of the NG production (Karion et al., 2013). Remote sensing of fugitive methane emissions from shale oil and gas production Bakken and Eagle Ford showed that these emissions accounted for 2.8-17.4% and 2.9-15.3%, respectively, on a volumetric basis (Schneising et al., 2014).

Brandt et al. (2014) investigated the large discrepancy between the bottom-up measurement and top-down satellite observation of methane leakages. They found that national scale atmospheric measurements suggest EPA's total methane inventory undercounts emissions by 50% (+/- 25%), though they discuss the difficulties in trying to attribute the emissions to specific sectors, specific activities, and specific products. Those atmospheric measurements point to the NG sector for some of the unaccounted emissions and that a small fraction of "super-emitters" was likely an important reason why the estimates from airborne measurements were typically higher than inventories. Brandt et al. (2014) found that the top-down studies estimating high leakage rates were unlikely to be representative of the NG industry because if the leakage at those sites were commonplace, the emissions from the NG industry would exceed the unaccounted emissions from all fossil and biogenic sources. Most recently, super-emitters were found to cause additional emissions that could explain the gap between component-based methane emissions as characterized in the 2016 EPA GHGI and site-based measurement of methane emissions that were about 1/3 higher than the 2016 EPA GHGI (Marchese et al., 2015; U.S. Environmental Protection Agency, 2016c). Another finding on the role of super-emitters was revealed in an effort to reconcile divergent estimates (Zavala-Araiza et al., 2015). A potentially large but normally unaccounted source of methane emissions could be abandoned oil and gas wells, which could represent 5–8% of annual anthropogenic methane emissions in Pennsylvania (Kang et al., 2015; Kang et al., 2016).

Model-year 2014-2018 NGVs need to comply with the EPA/National Highway Traffic Safety Administration Phase 1 HDV fuel consumption and GHG emission standard. While NGVs use a lower carbon fuel than diesel vehicles, NGVs have significantly higher methane emissions from both tailpipe and crankcase (if it is not closed). In the proposal of the Phase 2 HDV fuel consumption and GHG standards standard, EPA stated that starting in 2021 NG engine crankcases needed to be closed. While EPA did not finalize that requirement, in 2016 Cummins Westport released their 8.9-L ISL G Near Zero engine with closed crankcase ventilation (CCV) and plans to have CCV on their 6.7-L ISB G and 11.9-L ISX G engine models by 2018 (Cummins Westport Inc, 2016). We considered both cases where new NGVs have open and closed crankcase in our emissions modeling.

NGVs with open crankcase ventilation allows methane emissions to escape to the atmosphere from the crankcase. Methane emissions from such NGVs have been a data gap due to limited emission testing until West Virginia University has recently conducted a comprehensive emission measurement study on 22 heavy-duty NGVs equipped with both the 8.9-L ISL G engine and the 11.9-L ISX G engine (Clark et al., 2016). In-use emissions were measured for eight CNG and six LNG refueling stations and the vehicles to provide complete PTW emission measurement. It was found that open crankcase methane emissions are the most dominant emission source, accounting for about 67% and 42% of the total PTW emissions for CNG refuse trucks and LNG short-haul trucks, respectively, while this sources contributes to about 16% of the PTW emissions for CNG transit buses (Clark et al., 2016). We used these new PTW emission factors, as shown in Table D.1, to update the previous assumptions adopted in the GREET model in this WTW analysis. We adopted the fuel economy assumptions for CNG and LNG vehicles that are adopted in GREET, as shown in Table D.2, which were based on a comprehensive literature review (Cai et al., 2015b). In this analysis, we evaluate the impact of different supply chain and vehicle CH_4 emission scenarios on the WTW GHG emissions of NGVs in a sensitivity analysis.

2.3 Air emissions

We used the EPA's MOVES (MOtor Vehicle Emission Simulator) 2014 model to estimate the in-use air pollutant emission factors of the diesel heavy-duty vehicles we considered. The emissions we simulated from MOVES2014 are: (1) volatile organic compounds (VOCs), (2) CO, (3) NO_x, (4) particulate matter less than 10 microns (PM_{10}), and (5) $PM_{2.5}$. In this analysis, we focus on NO_x and PM, as those are the air pollutant emissions that have received the most regulatory scrutiny for HDVs; for more details on other emissions see Cai et al. (2015b) and Table D.4. The MOVES2014 emission factors are model-year-specific and account for the impacts of different driving cycles.

However, MOVES2014 has several major weaknesses including not having in-use test data for HDVs meeting the EPA/CARB 2010 standards. The in-use testing of diesels did include model year (MY) 2007-2009 vehicles, which have diesel particulate filters to control PM, but did not include post-MY 2010 vehicles using selective catalytic reduction (SCR) to control NO_x. Therefore, EPA made assumptions on the performance of SCR-equipped diesels (U.S. Environmental Protection Agency, 2015c). New studies have found that in real-world operation diesel HDVs can emit NO_x at much higher rates than what their engines were certified to meet the EPA/CARB 2010 standards (Miller et al., 2013; Carder et al., 2014; Quiros et al., 2016; Anenberg et al. 2017; Sandhu et al., 2017). In a sensitivity analysis, we examine diesel results from these in-use studies as compared to MOVES2014 estimates.

In addition, MOVES2014 has limited data on alternative fuel vehicles, estimating current post-2007 stoichiometric heavy-duty NGVs by scaling in-use estimates of MY 1994-2004 lean-burn NGVs by their relative EPA certification data (U.S. Environmental Protection Agency, 2015c). Several studies have examined in-use emissions of stoichiometric NGVs meeting the EPA/CARB 2010 standard (Nylund and Koponen, 2012; Yoon et al., 2013; Carder et al., 2014; Wang et al., 2015b; Quiros et al., 2016). These engines use stoichiometric combustion with cooled exhaust gas recirculation and a three-way catalyst to meet the most recent standards, but do not need to use particulate filters or SCR. In October 2016, Cummins Westport began production of an

8.9-L stoichiometric engine with an improved three-way catalyst, branded the ISL G Near Zero, meeting CARB's optional NOx standard of 0.02 g/bhp-hr (Cummins Westport Inc, 2016). A recent study examined the in-use emissions of a NGV using this "near-zero" stoichiometric engine (Johnson et al., 2016). In this study we analyze the NOx and PM of NGVs meeting both EPA/CARB 2010 (0.2 g/bhp-hr), which we refer to as "current" and CARB 0.02 g/bhp-hr standards, which we refer to as "near-zero".

Several types of current NG and diesel HDVs, including refuse trucks, transit buses, and freight trucks, were tested on a chassis dynamometer over multiple duty-cycles to help understand how real-world driving conditions influence emissions (Carder et al. 2014). In a later study, a current NGV and several diesel freight trucks were tested on-road using portable emissions measurement systems (Quiros et al. 2016). As theses NGVs and diesel vehicles were tested on the same cycles, we use these studies along with MOVES2014 to understand how NGVs compare to their conventional diesel counterparts. We then supplement the other studies in Table D.4 to better understand variance of emissions for both current and near-zero NGVs.

Results from EPA/CARB 2010 compliant freight trucks showed that the heavy-duty NGVs had PM emissions ranging from 80% lower to 40% higher than diesels equipped with diesel particulate filters (DPFs), depending on the duty-cycle (Carder et al. 2014; Quiros et al. 2016). In absolute terms, the PM emissions from all the tested NGVs and diesels with DPFs were very low, ranging from about 1–10 mg/mi for all vehicles, which agrees well with MOVES2014 (Cai et al., 2015b).

The testing also showed that the current NG freight trucks had 42–95% lower NO_x emissions and that a NG refuse truck had NO_x emissions ranging from 70% lower to 116% higher than their diesel counterparts depending on the duty-cycle (Carder et al. 2014; Thiruvengadam et al., 2015; Quiros et al., 2016). However, using relative ratios can be misleading as all the current NGVs consistently had low NO_x ranging from 0.2–0.9 g/mi, while the diesel freight trucks NO_x ranged from 0.6–9.0 g/mi and diesel refuse ranged from 0.7–1.3 g/mi. In high-speed duty cycles, the SCR system performs well and the diesel trucks have relatively low NO_x emissions. The test results showed that diesel engine temperatures did not reach levels to support sustained SCR performance in duty-cycles with significant amounts of idling, low speeds, and low engine loads, resulting in very high NOx emissions. For example, in duty-cycles representing near-dock port and local drayage operations, the exhaust gas temperatures were below 250°C for more than 95% of the time, which limited SCR activity and resulted in NOx emissions of 9.0 g/mi and 5.4 g/mi, respectively (Carder et al., 2014; Thiruvengadam et al., 2015). The NGVs typically had much lower NO_x emissions in duty-cycles with low speeds and low engine loads, as diesel engine temperatures did not support sustained selective catalytic reduction aftertreatment performance in those operations.

A limitation of these studies is that they are not a consistent comparison of engine technologies. The natural gas and diesel engines tested were not always produced by the same manufacturer, have the same displacement, or from the same model-year. While the diesel trucks in Carder et al. (2014) were the first generation after 2010 standard implementation, the truck in Quiros et al. (2016) were newer (MY 2013–2014) and in the same duty cycles showed lower NO_x emissions. As the measurements and engines are not consistent between the two studies, it is unclear if the reduction in NO_x for the new diesel vehicles is due to original equipment manufacturers (OEMs) improving their SCR systems. Unfortunately, there is limited in-use emissions data on 2010-compliant NGVs versus diesel trucks, and thus it is difficult to draw conclusions from testing results.

While OEMs see a path to 0.1 g/bhp-hr with a small reduction in fuel efficiency for diesel vehicles, further NO_x reductions would likely cause larger efficiency penalties (Eckerle, 2015). Even with new diesel technologies, reaching 0.02 g/bhp-hr NO_x levels will be challenging (Eckerle, 2015). A CNG refuse truck equipped with a “near-zero” stoichiometric engine meeting that standard was tested on a chassis dynamometer over several different duty-cycles (Johnson et al., 2016). The results show similar PM emissions to the current NGVs, which is not surprising as the focus of the engine is to reduce NO_x. The “near-zero” NO_x emissions were extremely low ranging from 0.01–0.09 g/mi for hot start tests, ranging from 78–97% lower than the corresponding results for the current NGVs (Carder et al., 2014; Johnson et al., 2016). Limited cold start tests were done which showed slightly higher emissions 0.08–0.22 g/mi than hot starts. The impact of cold starts on in-use NOx for both diesel and NGVs should be analyzed further.

In this analysis, we assume that both the current- and “near-zero” NGVs have the same PM emissions as their diesel counterparts, as recent testing shows they have very similar emission rates. Furthermore, we assume that the refuse, transit, and freight NOx emissions are 0.9, 0.7, and 0.2 g/mi for current NGVs, and 0.09, 0.07, and 0.05 g/mi for “near-zero” NGVs, respectively.

2.4 Costs

The AFLEET Tool was developed for the U.S. Department of Energy's Clean Cities Program by Argonne National Laboratory to estimate both the economics and environmental performance of alternative fuel vehicles in various vocations (Burnham, 2016). We used the Simple Payback Calculator feature of the AFLEET Tool that calculates the mileage and time needed for operating savings to pay back the incremental acquisition cost of an NGV as compared to its conventional counterpart; see Appendix E for a discussion of payback versus net present value analysis. The annual operating costs included in this calculation are fuel, diesel emission control fluid (i.e., urea), and maintenance costs. The payback time is calculated by providing the expected annual mileage of the vehicle. The acceptable payback time will depend on each vehicle operator as municipal fleets may have a greater tolerance for longer paybacks than private fleets due to the potential environmental benefits of NGVs.

As retail diesel fuel prices can be quite volatile, we examined how the cost differential between diesel and both CNG and LNG impacts the economics of NGVs (U.S. Department of Energy, 2017a). We also examine different incremental costs of NGVs as the potential for lower cost storage tanks improves their economic viability.

2.5 Baseline diesel vehicles

We used GREET 2016 to model the GHG emissions, water consumption, and air pollutant emissions of MY 2015 diesel transit bus, diesel refuse truck, and diesel short-haul truck as baseline counterparts to the NGVs. The diesel fuel production pathway in GREET has recently been updated to address impacts of new crude sources including U.S. shale oil production in

Bakken (Brandt et al., 2016) and Eagle Ford (Yeh et al., 2017) and Canadian oil sands (Cai et al., 2015a), diesel fuel production (Elgowainy et al., 2014), as well as diesel vehicle tailpipe emissions (Cai et al., 2015b). Water consumption associated with petroleum diesel produced in U.S. refineries has recently been assessed and used in this study (Henderson, 2016). Tables D.2 and D.3 summarize the fuel economy and tailpipe NO_x, PM_{10}, and $PM_{2.5}$ emission factors of the three heavy-duty diesel vehicle types (Cai et al., 2015b).

3. Results and Discussion

3.1 WTW water consumption

WTW water consumption of NGVs provides an important angle to evaluate the environmental and sustainable impacts of utilizing NG as a feedstock to produce transportation fuels in a water resource constrained world. When the U.S. average NG mix that consists of 47% of shale gas and 53% of conventional NG is the feedstock for LNG production, the LNG combination short-haul truck has a WTW water consumption of about 0.0044 gallon per ton-mile (gal/ton-mi), assuming that freshwater-free air or seawater cooling technology is used for process cooling in LNG plants. This represents a reduction of WTW water by about 81% relative to that of the diesel counterpart, as shown in Fig. 4. Diesel has higher water consumption associated with on-shore crude oil production, as well as higher water consumption associated with the fuel production in the refinery than NG processing and NG fuel production (Lampert et al., 2016).

Water consumption associated with NG supply chain is the major contributor, accounting for about 78% of the WTW water consumption of the vehicle, while indirect water consumption associated with process energy consumption in the LNG plant and with transportation of LNG fuel to the refueling station by truck accounted for about 22% of the WTW water consumption. Shale gas production, of which higher water (1.7 gal/mmBtu) is consumed than conventional NG production (0.1 gal/mmBtu), contributes to about 34% of the WTW water consumption. NG processing that consumes mostly NG and a small amount of electricity, accounts for about 36% of the WTW water consumption. Thomas and Burlingame (2007) reported only 1 out of 24 LNG plants (3 out of 77 individual liquefaction trains) worldwide used freshwater cooling tower as the cooling method. In fact, their summary of the LNG plants in existence in 2007 showed the most common cooling method used by the LNG industry was seawater cooling (13 out of 24 plants, 54 out of 77 trains), followed by air cooling (10 out of 24 plants, 20 out of 77 trains).

On the other hand, if LNG plants use freshwater instead of air or saline water for process cooling, LNG production would be very water use intensive, and LNG combination short-haul truck could exhibit a WTW water consumption of as high as 0.68 gal/ton-mi with about 90 m^3 of cooling water required to produce one ton of LNG fuel (GEA Group, 2016). Because of the harmfully high water temperature and chlorination caused by seawater cooling system that have raised serious public concern on safety of the marine habitat and the fishery industry, in addition to the economic and/or operational reasons, the LNG industry nowadays has been gradually shifting its preference to the freshwater-free air-cooling technology (GE Oil & Gas, 2007; Gaucher et al., 2015; Thuncher, 2016).

For the CNG refuse truck and transit bus, water consumption associated with the NG supply chain, particularly shale gas recovery and NG processing, and with NG compression at CNG stations that consumes water-intensive electricity account for about 44% and 56%, respectively, of their WTW water consumption. On a WTW basis, these CNG vehicles provide a great opportunity to reduce water consumption by 62%, relative to their diesel counterparts. For all the three NGV types, increasing production and use of shale gas from the current level would increase their WTW water consumption. However, even if the LNG and CNG fuels are produced entirely from shale gas, the LNG short-haul combination truck, the CNG refuse truck and the CNG transit bus would still exhibit as much as 59%, 39%, and 39% lower WTW water consumption, respectively, than that of their respective diesel counterparts.

We conducted sensitivity analysis to examine the impact of variation in shale gas EUR of different plays as reported in literature on the WTW water consumption of the NGVs, as shown in the error bars in Fig. 4. Despite the resulting variation in WTW water consumption among different shale gas production regions in the U.S., all the NGVs still provide an opportunity to significantly reduce the WTW water consumption relative to their diesel counterparts. Our analysis shed new light on previous studies that have generally found that the water quantity impacts of shale gas production are, on average, not significantly worse than those of conventional NG production (Kuwayama et al., 2015).

3.2 WTW GHG emissions

In Fig. 5, WTW GHG emissions of the three NGVs with open crankcases versus their diesel counterpart are displayed. In our default case, the LNG combination short-haul truck, CNG refuse truck, and CNG transit but emit 1%, 6%, and 8% more WTW GHG emissions than their diesel counterparts, respectively. The WTW GHG emissions of these NGVs are strongly dependent on their vehicle fuel economy, as 71-75% of their WTW GHG emissions come from

CO_2 tailpipe combustion emissions. The NG supply chain contribute 13% of the WTW GHG emissions of the LNG combination short-haul truck. The emission contribution of the NG supply chain is even higher for the CNG refuse truck and transit bus, accounting for about 20% of their WTW GHG emissions.

The CNG vehicles have higher supply chain GHG emissions due to leakage from long distance NG transmission and local NG distribution pipelines delivering to CNG refueling stations. For LNG, liquefaction plants are typically built close to NG sources, with the NG is taken directly off the high-pressure transmission pipelines after a short distance. CNG compression, which consumes electricity with an energy efficiency of about 98%, is relatively a small emission source, accounting for about 4% of the total WTW GHG emissions. LNG production and distribution to the refueling station contributes about 11% of the total WTW GHG emissions because of significant energy requirement in the liquefaction process, which is about 91% efficient, (Brinkman et al., 2005) and CH_4 emissions due to LNG boil-off losses during distribution and storage.

Fig. 5 shows the WTW impact of eliminating crankcase CH_4 emissions from these NGVs. WTW GHG emissions would be reduced by 7%, 9%, and 7%, respectively, for the LNG combination short-haul truck, CNG refuse truck, and CNG transit bus. As a result, these NGVs would have about the same level GHG emissions as their diesel counterparts.

Two factors largely determine whether the NGVs provide WTW GHG emission reductions as compared to their diesel counterparts: NGV relative fuel economy and CH_4 emission leakage during the NG supply chain and vehicle use. In our sensitivity analysis, Fig. 6 shows the relationship between these factors for CNG and LNG vehicles. With the default fuel economy (15% reduction, 15% reduction, and 10% reduction) and WTW leakage rates (3.0%, 3.2%, and 2.9% for refuse, transit, and freight, respectively), these NGVs do not offer WTW GHG emission reductions as compared to diesels. The WTW leakage rates need to be less than 2.7% for CNG vehicles with a 15% fuel economy penalty, and less than 2.8% for LNG vehicles with a 10% penalty to breakeven with the diesel counterpart. Thus, if NGV manufacturers can improve both fuel economy and CH_4 leakage along the NG supply chain improves, NGVs can have better GHG emission performance. However, even as the NGV industry works to reduce vehicle CH_4 emissions, if supply chain CH_4 emissions prove to be much larger than currently estimated, significant fuel economy improvements will be needed to be on par with diesel vehicles.

3.3 WTW NO_x and PM emissions of NG vehicles

As shown in Fig. 7(a), WTW NO_x emissions of the diesel short-haul trucks are largely from tailpipe emissions, while the LNG short-haul trucks have much lower vehicle emissions and slightly higher upstream emissions. The LNG short-haul trucks show the largest relative NO_x benefit (57–64% reduction) as compared to the CNG refuse trucks (0–34% reduction) and transit buses (2% increase – 27% reduction) versus their diesel counterparts. NO_x emissions associated with feedstock activities, such as natural gas recovery, processing, and transportation are about 85% lower for LNG as compared to CNG, which accounts for most of the difference between these fuels. Because of the benefit from both upstream and tailpipe NO_x emissions, the LNG short-haul trucks offer significant WTW NO_x emission reductions, about 57% for current and 64% for near-zero, compared to their diesel counterparts.

For CNG vehicles, the upstream NO_x emissions are the major contributor to WTW emissions, mostly due to compressors used for long distance NG transmission pipelines to the refueling stations for CNG production (Dunn et al., 2013). Vehicle tailpipe emissions account for only about 28–39% of the WTW emissions for the current CNG transit bus and refuse truck, respectively. As a result, current CNG vehicles produce a similar amount of WTW NO_x emissions to their diesel counterparts. However, as transmission pipelines are not often in heavily populated areas, the benefit from lower vehicle emissions is likely more impactful to human health than the higher WTP emissions. The near-zero CNG vehicles reduce WTW NO_x emissions compared to diesel vehicles by about 27–34% for the transit bus and refuse truck, respectively.

As shown in Fig. 7(b), the NGVs have 14–34% lower WTW PM_{10} and $PM_{2.5}$ emissions than their diesel counterparts, due to their lower WTP emissions. We do not show separate PM results for current and near-zero NGVs, as they are assumed to have the same emissions. As mentioned previously WTP emission benefits in unpopulated areas have a smaller impact on human health than those in populated areas.

In our sensitivity analysis, we examine the impact of in-use diesel NO_x emissions higher than MOVES2014 estimates for a short-haul truck. In Fig. 8 and Fig.D.1, the current LNG, near-zero LNG, and diesel tailpipe and WTW NOx are shown, respectively, for various duty-cycles tested in Carder et al. (2014) and Sandhu et al. (2017). The figure shows that variability in diesel tailpipe emissions drives the absolute difference between diesels and NGVs. The absolute difference between current and near-zero short-haul NGVs is relatively small, though it is somewhat larger for refuse and transit applications. In duty-cycles where SCR performs poorly, diesel vehicles can have many times higher tailpipe emissions compared to NGVs (e.g. 2000% higher than current and 19000% higher than near-zero in near-dock cycle). Further research is needed to better estimate the in-use activity and duty-cycle NO_x emissions of all types of diesel HDVs to better inform where NGVs can provide the most air quality benefits.

3.4 Costs of NG vehicles

The economics of NGVs depends on having low enough operating costs to payback the incremental cost of storage tanks as compared to diesel vehicles. Retail natural gas prices have been consistently around $2.30 per diesel gallon equivalent (DGE) (or $2.00 per gasoline gallon equivalent) for more than decade (U.S. Department of Energy, 2017a). While diesel prices have ranged from about $2 to $5 per DGE in that same time, as seen in Fig. F.1 (U.S. Department of Energy, 2017a). Fuel prices can differ significantly by state and region, as areas such as California consistently have higher diesel prices. In addition, many fleets using NGVs build an on-site fueling station, which can lead to significantly lower fuel prices (Rood Werpy et al., 2010).

However, the challenge of building NGV refueling infrastructure is that a fast-fill refueling station, capable of filling near diesel station speeds, can cost $1 million or more, while time-fill stations, which fill overnight can approach $1 million for a large fleet (Smith and Gonzales, 2014). Short-haul NGVs can require refueling stations along their route, while return-to-base fleets, such refuse and transit, will fuel at a fleet depot. Currently, there are 938 public and 753 private CNG stations and 83 public and 57 private LNG stations (U.S. Department of Energy, 2017b). While CNG stations are found in both metro areas and highways, LNG stations are primarily found near highways.

Using the AFLEET 2016 tool and the NGV and diesel fuel economy data discussed above for each vehicle type, we analyzed the breakeven vehicle miles traveled (VMT) to payback the NGV incremental cost (Burnham, 2016). We analyzed a fuel price differential ranging from $0.00/DGE to $3.00/DGE and two scenarios of NGV incremental costs to examine the sensitivity of these two factors. The high incremental cost scenario uses default AFLEET 2016 data representing costs today ($50,000 for refuse and short-haul trucks and $60,000 for transit buses), while the low cost scenario assumes half of today's incremental cost representing an optimistic future scenario ($25,000 for refuse and short-haul trucks and $30,000 for transit buses). Following the sensitivity analysis, we examine recent CNG, LNG, and diesel prices to examine the current economic performance of today's NGVs.

As seen in Fig. 9(a), the high cost CNG refuse trucks are estimated to have a breakeven mileage of 660,000 miles at a $0.50 per DGE differential, while it drops to 137,000 miles at $1.00 per DGE and 77,000 miles at $1.50 per DGE. Refuse trucks are estimate drive about 23,400 miles/year on average, so for the above cases the payback 28.2 years, 5.9 years, and 3.3 years respectively. The payback curves are exponential in nature and for the high cost CNG refuse case, payback periods drop quickly beyond $0.75 per gallon.

The high cost CNG transit buses are estimated to have a breakeven mileage of 2.6 million miles at a $0.50 per DGE differential, while it drops to 413,000 miles at $1.00 per DGE and 225,000 miles at $1.50 per DGE. Transit buses are estimate drive about 35,000 miles/year on average, so for the above cases the payback 73.4 years, 11.8 years, and 6.4 years respectively. The high cost CNG transit case has payback periods that drop quickly beyond $1.25 per gallon.

The high cost LNG short-haul freight trucks have a similar curve to CNG transit buses. They are estimated to have a breakeven mileage of 1.3 million miles at a $0.50 per DGE differential, while it drops to 470,000 miles at $1.00 per DGE and 285,000 miles at $1.50 per DGE. Short-haul freight trucks are estimate drive about 170,000 miles/year on average, so for the above cases the payback 65.0 years, 7.6 years, and 2.7 years respectively.

When looking at the low vehicle cost scenarios in Fig. 9(b), where the increment costs are estimated to be half the high cost scenarios, the breakeven VMT is reduced in half as well. Changes in breakeven VMT are linearly dependent on the incremental cost.

From April 2011 to October 2014, national average retail diesel prices were consistently around $4.00 per DGE, resulting in a $1.65 per DGE price differential between retail CNG prices (U.S. Department of Energy, 2017a). During that same time private station CNG pricing was about $0.40 per DGE cheaper than retail CNG, and thus had more than a $2.00 differential with diesel, which does not have private station price benefit (U.S. Department of Energy, 2017a). In 2015, diesel prices dropped significantly and in the past two years have averaged about $2.65 per DGE. The fuel price differential for CNG versus diesel was $0.30 per DGE for retail and $0.70 per DGE for private stations. Historical LNG fuel price data is not available due to the limited number of LNG stations, but the most recent available data (July – October 2016) shows that retail LNG was priced at $0.10/DGE more than CNG.

We utilized private station pricing to examine the current economics of CNG refuse trucks and transit buses as we assumed they fuel at a fleet depot. At current vehicle costs and at the 2011-2014 fuel price differential of $2.05/DGE, CNG refuse trucks have a breakeven mileage of 51,000 miles and 2.2 years. At today's fuel price differential of $0.70 per DGE, CNG refuse trucks have a breakeven mileage of 261,000 miles and 11.2 years. At the 2011-2014 fuel price differential of $2.05/DGE, CNG transit buses have a breakeven mileage of 149,000 miles and 4.3 years. At today's fuel price differential of $0.70 per DGE, CNG refuse trucks have a breakeven mileage of 832,000 miles and 23.8 years.

We utilized retail pricing, with the assumption that LNG cost $0.10 more than CNG, for LNG short-haul trucks as we assumed they fuel at public stations. At current vehicle costs and at the 2011-2014 fuel price differential of $1.55/DGE, LNG freight trucks have a breakeven mileage of 273,000 miles and 1.6 years. At today's fuel price differential of $0.20 per DGE, LNG freight trucks do not have a breakeven mileage as the small fuel cost savings are cancelled out by the small increase in vehicle maintenance (Burnham, 2016). For both CNG and LNG vehicles, these large swings in diesel prices drastically influence their economics.

4. Conclusions

Environmental impacts of NGVs from the perspective of water consumption, GHG emissions, and NO_x and PM emissions were evaluated on a WTW basis. Significant reduction in water consumption is found for NGVs compared to their diesel counterparts, despite the variation in water consumption associated with shale gas production in various regions. This provides an opportunity to lessen the water stress caused by production and use of diesel in the HDV sector with the introduction and potential expansion of the use of CNG and LNG counterparts, especially in regions facing severe shortage of freshwater supply. Other water resource impacts, such as water quality, season-dependent stress on local water supply, and treatment of produced water in gas field, are not considered in this analysis and warrant further research despite on-going efforts.

WTW GHG emissions of NGVs are slightly higher than those of their diesel counterparts with the methane leakage rate of the natural gas supply chain estimated based on the EPA GHGI, the recent disclosed PTW methane emissions of NGVs, and their 10-15% fuel economy penalty. However, new NG engine with closed crankcase design that has reached the market has similar WTW GHG emissions. Other opportunities to reduce GHG emissions of NGVs include fuel economy improvement of NGVs and mitigation of methane leakage associated with the NG supply chain. However, without significant improvements in both methane leakage and fuel economy NGVs using fossil-based NG will not provide large GHG benefits.

Current NGVs that use stoichiometric engines equipped with a three-way catalyst can provide consistently low tailpipe PM and NO_x emissions across different duty-cycles, with new near-zero NGVs providing even lower NO_x emissions. However, the absolute NO_x benefits of NGVs, as compared to diesel counterparts, is largely driven by the performance of diesel vehicle's SCR aftertreatment systems, which can be highly duty-cycle dependent. In applications with long idle times, low speeds, and low loads, SCR performance can be poor, causing NO_x emissions to be several times higher than in applications where SCR performance is optimal. Further analysis is needed on in-use duty-cycles of various types of diesel HDVs and their impact on

NOx emissions. This information is necessary to update future versions of EPA's MOVES model and to better understand real-world NO_x emissions from the HDV sector. While NGVs provide lower tailpipe NO_x emissions, upstream emissions can be larger for CNG. Therefore, research on the location of WTP emissions is needed to fully understand the potential for NGVs to improve air quality and their impact on human populations.

NGVs offer the ability to reduce annual operating costs due to the low and stable cost of NG-based fuels. However, the challenges of NGVs are the high incremental costs of NG storage tanks and the potential high costs of NG fueling infrastructure. High fuel use heavy-duty vehicles, such as refuse trucks, transit buses, and short-haul freight trucks offer opportunities for NGVs to provide significant economic benefits. If fuel price differentials are more than $1.50 per DGE, these applications can provide fast paybacks. In such circumstances, NGVs may become cost effective and penetrate the market faster, which in turn helps overcome other technical and market barriers such as NG distribution and storage. An expansion of NGVs would bring about benefits of less water consumption and air emissions to the U.S. transportation sector.

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