Compatibility of fossil fuel energy system for UK climate targets

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Declaration

I declare that this thesis and the papers within it have been composed by myself and that no part of this thesis has been submitted for any other degree or qualification. The work described is my own unless stated otherwise.

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Abstract

The United Kingdom (UK) has an ambitious greenhouse gas (GHG) reduction target with legally binding commitment of 80% reduction by 2050 relative to 1990 levels. The Committee on Climate Change (CCC) sets carbon budgets to meet this goal, and suggested that electricity generation should be below 50 g CO_2e per $kWh_{(e)}$ by 2030. At the same time, the UK is renewing gas distribution pipeline systems to decrease leakages and increase efficiency of gas delivery, all while pursuing a domestic shale gas industry to meet continued demand as traditional gas production decreases. Competing in the same market, the United States (US) became a net exporter of natural gas at the end of 2017, largely due to increased production of shale gas, and holds contracts for distribution in the UK. It is clear that the UK is continuing reliance on natural gas in the near term, despite climate targets, and will need advanced mitigation strategies.

One such strategy is using Enhanced Oil Recovery (EOR) with CO_2 and coupling it with Carbon Capture and Storage (CCS). This strategy can create of a large commercial market for EOR offshore of the UK, and maintain a 50% chance of keeping temperature rise below +2°C throughout the 21st century. The market has the potential to accelerate CO_2 storage investment, and aid in meeting UK climate targets. A coupled CCS-EOR scenario might contribute to decarbonisation of UK grid electricity. Using UK data, progressive introduction from 2020 of 11 CCS-to-EOR gas-power plant projects is estimated to store 52 Mt CO2 yr^{-1} from 2030. These 11 projects also produce extra revenue of 1,100 MM bbls of taxable EOR-oil from 2020 to 2049. The total average electricity grid factor in the UK reduces from 490 to 90 – 142 kg CO_2e MWh^{-1}, with gas generating 132 TWh of clean electricity annually. This life cycle analysis (LCA) is unusual in linking oil production and combustion with CCS and gas fuelled electricity. With a full LCA, this aggressive CCS-EOR scenario provides a net carbon reduction, and progressively reduces net oil combustion emissions beyond 2040.

A second strategy could be needed if the projected domestic gas supply gap for power generation (without CCS) were to be met by UK shale gas with low fugitive emissions (0.08%). In this case an additional 20.4 Mt CO_2e would need to be accommodated during carbon budget periods 3 – 6. However, a modest fugitive emissions rate (1%) for UK shale gas would increase global emissions compared to importing an equal quantity of Qatari liquefied natural gas, and risk exceeding UK carbon budgets. Additionally, natural gas electricity generation would emit 420 – 466 Mt CO_2e (460 central estimate) during the same time period within the traded EU emissions cap.

In addition to electricity generation, shale gas supply chain emissions for heat are assessed. This thesis assesses the greenhouse gas emissions intensity of US and UK shale gas as determined by source, distribution and end use for heat. It assesses the merit order of shale gas imported to the UK from the US versus domestic production and use of shale gas in the US or UK, considering distribution network renewals and the total emissions intensity of shale gas used. The import and use of US-produced shale gas liquefied natural gas (LNG) in the UK would increase GHG emissions relative to domestic UK shale gas production and use by 178 Mt CO_2e, yet only increasing UK carbon budgets 3-6 by 14.2 Mt CO_2e (19.2%). It is found that losses in the distribution phase represent a highly uncertain, but potentially important component of shale gas GHG intensity. This thesis considers the implications for GHG emissions measurement and reporting, climate change mitigation via municipal pipeline renewal, and national carbon budgets to 2035.

Most importantly, under the current production-based greenhouse gas accounting system, the UK is incentivized to import natural gas rather than produce it domestically throughout each of the cases studied. This thesis gives policy recommendations to mitigate the impact of perverse incentives in new GHG regulations.

Lay summary

As part of a commitment to reduce impacts of climate change, the United Kingdom (UK) has an ambitious greenhouse gas (GHG) reduction target with legally binding commitment of 80% reduction by 2050 relative to 1990 levels. The Committee on Climate Change (CCC) sets carbon budgets to meet this goal, and suggested that electricity generation should be below 50 g CO_2e per $kWh_{(e)}$ by 2030. At the same time, the UK is renewing gas distribution pipeline systems to decrease leakages. The UK is pursuing a domestic shale gas industry to meet continued demand as traditional gas production decreases. The United States (US) became a net exporter of natural gas at the end of 2017, largely due to increased production of shale gas, and is now a net exporter of natural gas.

When combined with CCS, shale gas could support both the UK and US plans to meet reduced GHG electricity supply and maintain a 50% chance of keeping temperature rise below +2°C throughout the 21st century. However, there is a projected shortcoming in the UK's fourth carbon budget of 7.5%. This shortfall, and future carbon budget gaps, may be increased if the UK pursues a domestic shale gas industry to offset projected decreases in traditional gas supply. Additionally, importing foreign gas may aid in meeting carbon budgets while increasing total atmospheric GHG emissions for the same quantity of traded gas.

This thesis is structured as a collection of research papers discussing the GHG impacts of gas usage through different segments of the supply chain to end use. The first results chapter of the thesis considers the use of Enhanced Oil Recovery (EOR) as a funding mechanism for Carbon Capture and Storage (CCS) as a pathway to UK decarbonization targets. The second results chapter examines the implications to grid electricity targets for importing shale gas or producing it domestically. The third results chapter discusses the carbon budget implications of importing US shale liquified natural gas for used in the residential sector compared to the same quantity of UK shale gas. This thesis considers the implications for GHG emissions measurement and reporting, climate change mitigation via municipal pipeline renewal, and national carbon budgets to 2035.

Most importantly, under the current production-based greenhouse gas accounting system, the UK is incentivized to import natural gas rather than produce it domestically throughout each of the cases studied. Finally the thesis examines the carbon price and policy implications, of these pathways and suggest areas for policy changes.

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Finally, and above all, I would like thank my wife, Emer. Her support and love kept me going throughout this project, during which we were married, and moved to a new continent. I am excited to see what’s next.

Contents

Chapter 1 Introduction

1.1 Context

The introduction and growth of shale gas production in the US has increased over tenfold since 2007 (See Figure 1-1). The US energy market has decoupled natural gas prices from oil, and created a shift from coal to gas electricity generation. As a result, the US has significantly reduced US GHG emissions to meet US Kyoto Protocol targets, which were never ratified by the US (EIA, 2012).

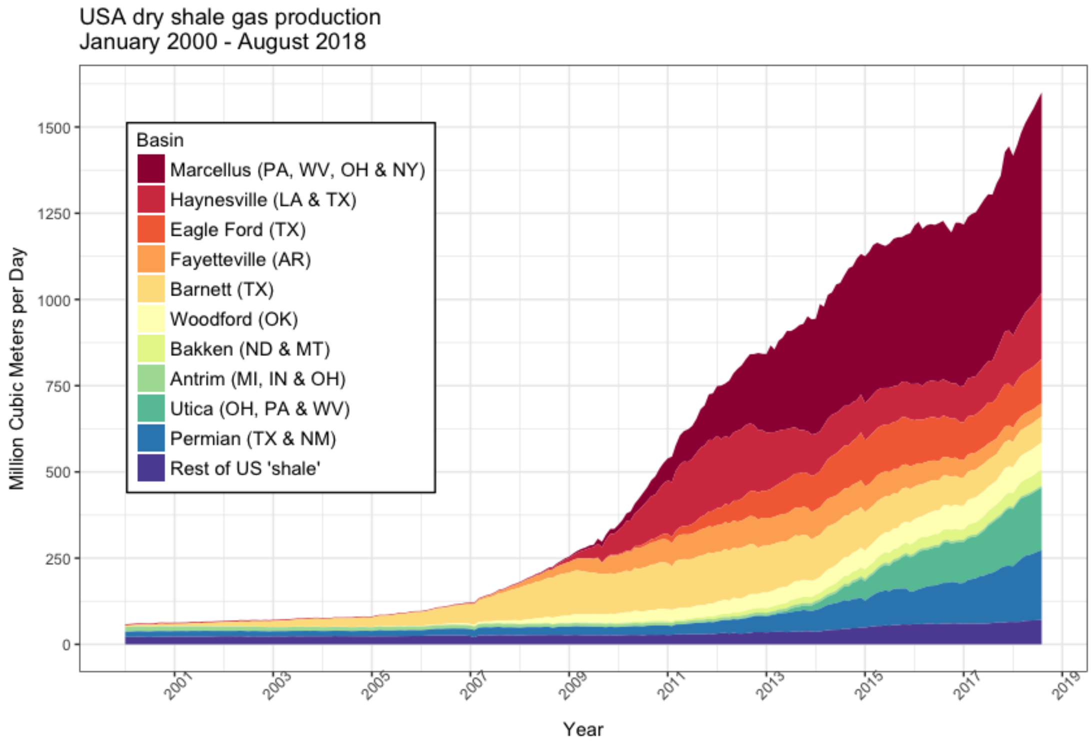


Figure 1-1 - US Shale Gas Production 2000 - 2018 in million cubic meters per day. Data adapted from US Energy Information Agency (EIA, 2018a).

In the US, hydraulic fracturing ("fracking") for shale gas was not initially regulated by federal laws specific to the new process. Instead, it was regulated under laws for traditional natural gas drilling, and drilling approved by individual state legislatures (Rahm, 2011). These factors highlight the need for clear regulation and measurement of shale gas developments to compare against a baseline of emissions.

Compatibility of fossil fuel energy system in the UK for climate targets. In the first paper on fugitive methane from shale gas, Howarth et al. (2011) mention that the sampling methods used for the US GHG inventory were shown to be from flawed methods (Harrison et al., 1996; Kirchgessner et al., 1997), due to selective and voluntary participation from industry partners rather than random sampling. Howarth et al. (2011) claim the Environmental Protection Agency's (EPA) baseline GHG emissions are therefore invalid. The paper called into question the ability for shale gas to reduce GHG emissions in the US, and set a break-even point of 2.4-3.2% of fugitive methane (CH_4) being the "break-even" point for gas being equal to coal for lifecycle CO_2e per MWh(e) of delivered electricity (Howarth et al., 2011; Howarth, 2014).

In the UK, electrical supply was roughly 500 kg CO_2e per $MWh_{(e)}$ of delivered electricity in 2013, including gas, coal, renewables (DECC, 2014a). The Committee on Climate Change (CCC) (2014) recommended reducing the electrical supply to 50 kg CO_2e per $MWh_{(e)}$ by 2030. When coal plants are removed from a national power inventory there is a reduction in the CO_2 intensity of power generation, because coal plants emit the most CO_2 of conventional power supply. In Figure 1-2 below, it is assumed that UK coal plants generate 1000 kg CO_2 per $MWh_{(e)}$ and gas plants generate 500 kg CO_2 per $MWh_{(e)}$. Removing coal plants and increasing renewable generation will gradually reduce the portfolio average towards the CCC (2014) recommended goal. If a larger quantity of coal is rapidly retired or removed from the portfolio, there will be an immediate reduction in emissions (Figure 1-2, point A). If this retired coal is replaced by new gas plants, it is assumed, in this simplified illustration, that the new gas plants will be used at least 20 years. The CO_2 emissions savings that was experienced by removing coal would eventually turn into a penalty (at point B) and extend the progress on reaching GHG reduction goals to point C.

Compatibility of fossil fuel energy system in the UK for climate targets.

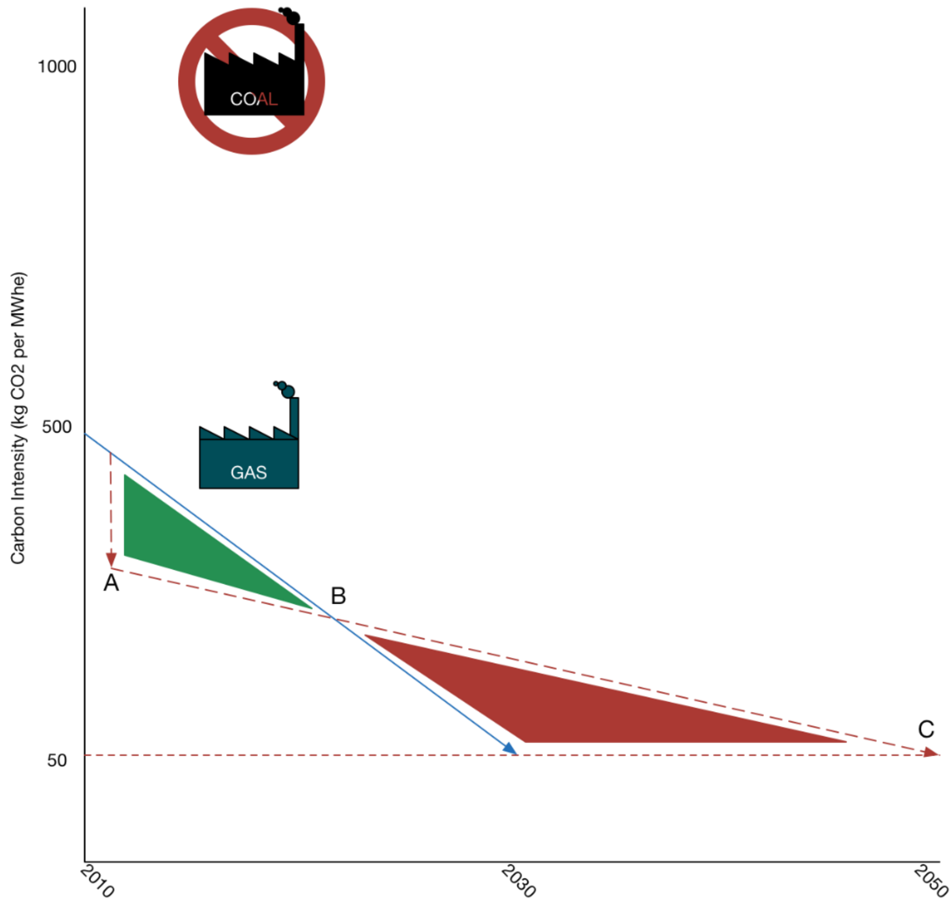


Figure 1-2 – Simplified fuel switching schematic and future carbon lock-in for UK power.

Regardless of these issues, the UK is moving forward with a shale gas agenda. There are six shale gas basins being explored in the UK for characterisation (Andrews, 2013)(See Figure 1-3). This thesis will aid in the understanding of the financial incentives, proper regulation, best practices, and climate forcing implications of a UK shale gas industry. It will look to discuss downrange of factors of climate impacts of the US shale gas boom, and model the GHG impacts of this boom in the UK. The thesis will examine UK shale gas development under the constraints of UK GHG reduction goals. The project will evaluate and discuss whether carbon capture and storage (CCS) and/or enhanced oil recovery (EOR) can lessen the climate impacts of shale gas while ensuring a secure electricity supply.

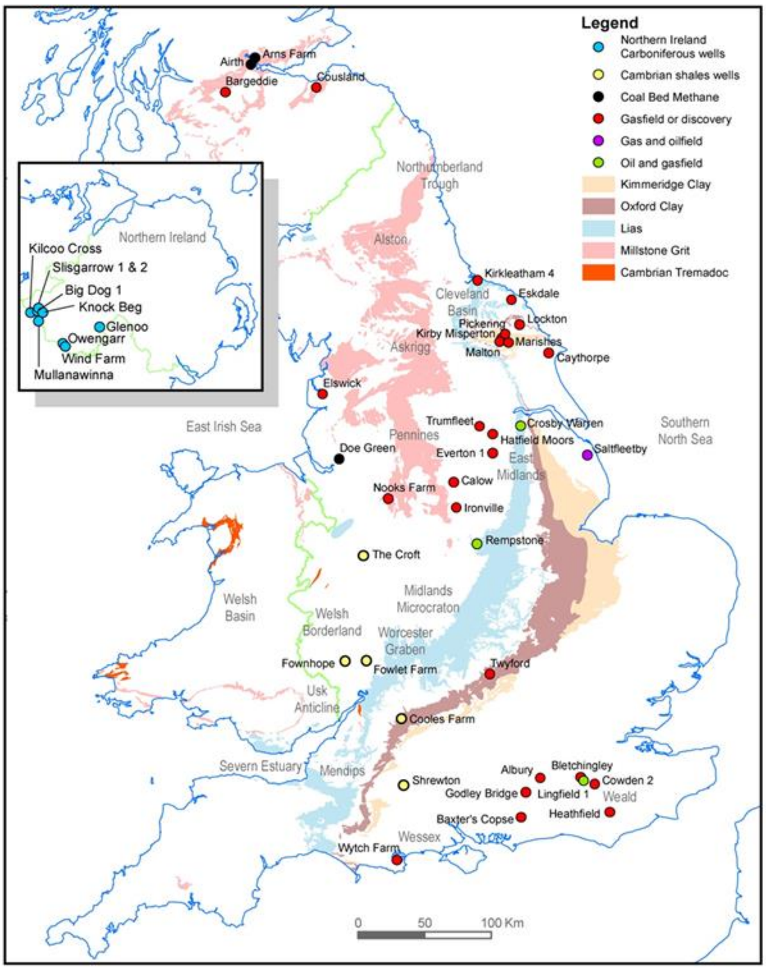


Figure 1-3 – UK shale gas basins identified by BGS in Andrews (2013). The UK may have upwards of 3.6 Tm^3 of shale gas across 6 shale basins, enough to supply energy at the current rate for over a generation.

1.1.1 Global Warming Potential and time horizons

An important consideration when calculating GHG inventories and the impact of methane is the Global Warming Potential of methane. The Global Warming Potential (GWP) is an index of the total energy added to the climate system by a gas relative to the same weight of that added by CO_2. The GWP is the default metric for converting and transferring values of GHGs on 20, 100, and 500-year time scales consistent with Houghton et al. (1990). Until 2006, national GHG inventories have reported CH_4 to have a GWP value of 21 on a 100-year time horizon, consistent with IPCC Second Assessment Report (1996). UNFCCC GHG reporting protocol was updated in 2006, but the same GWP value of 21 was used for weighted values of CH_4 to maintain consistency across previous reports and flexibility mechanisms (US EPA, 2014; Myhre et al., 2013). In 2015 GHG inventory reports, values for CH_4 were updated to be consistent with the IPCC Fourth Assessment Report (2007) value of 25 on a 100-year scale. It should be noted that there is no scientific basis for selecting a 100-year scale for GWPs in international reporting and flexibility mechanisms (Fuglestvedt et al., 2003; Shine, 2009).

The IPCC Fifth Assessment Report (AR5) updates the GWP for CH_4 to be 28 on a 100-year scale. The report includes indirect effects of CH_4 oxidation in the atmosphere (Boucher et al., 2009), which were not included in previous reports. Shindell et al. (2009) estimate that CH_4 has a substantial climate effect because of changes in the rate of oxidation of SO_2 to sulphate aerosols. When considering climate-carbon feedbacks, the 100-year GWP is increased by 20% to 34 due to a warming-induced release of CO_2 in the land biosphere and ocean (Gillett and Matthews, 2010). One notable shale gas study (Howarth et al., 2011) uses the 20-year GWP, because of the stated need for urgent action on mitigating the negative effects of climate change. These figures indicate a continuously evolving understanding of the effects of CH_4 on climate change, which are slow to be incorporated into carbon markets.

1.1.2 Fugitive methane emissions from shale gas in the US

In late 2010, the EPA (2010) concluded that fugitive emissions of methane from unconventional gas production may far exceed those of conventional gas. Furthermore, Howarth et al. (2011) stated that lifetime GHG emissions from shale gas wells may exceed the emissions from coal if fugitive emissions rates are greater than 2-3% of lifetime production. Extraction of shale gas has additional processes which are not used in traditional gas drilling. The hydraulic fracturing process involves large quantities of water and chemical lubricants (EPA, 2010), which aid in breaking shale rock layers, and releasing previously entrapped gas. As a result, the shale well is under greater pressure than a traditional gas well and incurs “flowback” during the well completion process, releasing pressure and gas / chemical mixtures. Based on the release of the flowback emissions, Howarth et al. (2011)

concluded that the total life-cycle fugitive emissions of shale gas wells are 3.6 – 7.9% of total production compared with 1.7 – 6.0% of conventional wells due to fugitive emission during extraction and processing (see Table 1-1).

Table 1-1 – Fugitive methane emissions by production process phase. Fugitive methane emissions associated with development of natural gas from conventional wells and from shale formations (expressed as the percentage of methane produced over the lifecycle of a well) (Howarth et al., 2011).

Process Phase	*Conventional Gas*	*Shale Gas*
Emissions during well completion	0.01%	1.9%
Routine venting and equipment leaks at well site	0.3 to 1.9%	0.3 to 1.9%
Emissions during liquid unloading	0 to 0.26%	0 to 0.26%
Emissions during gas processing	0 to 0.19%	0 to 0.19%
Emissions during transport, storage, and distribution	1.4 to 3.6%	1.4 to 3.6%
Total Emissions	*1.7 to 6.0%*	*3.6 to 7.9%*

Abrahams et al. (2015) indicate that the leakage rate is the most impactful figure on the upstream emissions in shale gas usage. However, the maximum leakage rate is that study is only 4% based on Heath et al. (2014) and Weber and Clavin (2012). There is discussion in the literature that fugitive emission rates have systemically reduced in the US (Schwietzke et al., 2016) since the beginning of the shale boom. However, there have been observations of leakage rates of 12% (Howarth, 2015; Schneising et al., 2014), and an overall increase in CH_4 emissions from gas production in US inventory data (EPA, 2017). Schneising et al. (2014) estimate 9.5% leakage rates at well-completion and an additional 2.5% from upstream distribution leakages based on satellite observations.

Heath et al. (2014) harmonized LCA estimates for shale gas and conventional gas power according to leakage rates at the time, and cite an emissions of intensity for upstream emissions of 21.3 g CO_2e MJ^{-1} HHV (17.6 – 24.8, 90% CI).[1]

In 2012, the IEA outlined "golden rules" (IEA, 2012) to be adopted to profitable reduce fugitive methane emissions in the shale gas industry. The most impactful is the usage of reduced emissions completions (RECs) or "green completions" which reduce flowback emissions and capture more commercial product. RECs have been compulsory in the US since 2015, and can reduce emissions from well completions by 75 – 99% (Oil and Natural Gas Sector: New Source Performance Standards, National Emission Standards for Hazardous Air Pollutants, and Control Techniques Guidelines).

Balcombe et al. (2017) surveyed the shale gas literature after the RECs rule change. They found that natural gas supply chain combined CO_2 and CH_4 emissions ranged from 3.6 to 42.4 g CO_2e MJ^{-1} HHV with a central estimate of 10.5. This is a significant decrease from previous findings by Heath et al. (2014). However, Balcombe et al. also found six estimates of fugitive emissions from shale gas production above 100 g CO_2e MJ^{-1} HHV (232, 285, 304, 618, 1910, 5250), illustrating a wide spread of fugitive emissions observations.

Following, the Howarth et al. (2011) paper, many studies have been carried out that have found underreporting of methane leakages, and characterize this spread of emissions rates. The studies follow two methodologies: "bottom-up" and "top-down" studies. Bottom-up studies are performed with on-the-ground measurements as close to a point source as possible. These point sources could be well pads, storage facilities, or pipelines. However, this sampling method could miss large single point sources of emissions.

[1] Note that Heath et al. (2014 and Abrahams et al. (2014) are based on methodology of Weber and Clavin (2012a) who cite guidance from EIA (2011) which recommends reporting higher heating value (HHV) rather than net or lower heaving value (LHV). The HHV is conventionally used in the energy accounting and conversion from chemical potential energy to delivered energy (e.g. electricity). The HHV exceed LHV by the latent heat of vaporization of water. Unless otherwise noted, it is assumed that HHV is used throughout this thesis.

Compatibility of fossil fuel energy system in the UK for climate targets. Top-down studies use tower, aerial, or satellite sensing to determine the total CH_4 fluxes in a studied area. This method can gather readings for large area, but may also miss individual point sources of methane outside a given study area.

1.1.1.1 Top-down leakage rates

Using aerial sensors aboard a single aircraft above the Marcellus Shale in SW Pennsylvania, Caulton et al. (2014) measured a regional methane flux of 2.0-14g CH_4 s^{-1} km^{-2} over a ~2,800-km^2 area. This measurement did not differ statistically from bottom-up calculations of 2.3-4.6 g CH_4 s^{-1} km^{-2}. However, large emissions (34 g CH_4 / per well) were detected from seven wells in the drilling phase of operations. This result is 2-3 orders of magnitude greater than the US EPA accounting for this phase of operations. These 7 wells represented ~1% of wells, and 4-30% of regional methane flux. This observation highlights the great uncertainty of methane leakage throughout the production chain. It is important to note that these figures were observed before the hydraulic fracturing took place, but after the drilling phase.

Other aerial studies have concluded similar under-reporting in the EPA baseline. Karion et al. (2013) used a mass balancing model approach with an aerial sensor over a natural gas field in Uintah County, Utah. They detected CH_4 emissions of 6.2 - 11.7% (1σ) of average hourly natural gas production. Kort et al. (2014) quantified CH_4 emissions over established oil and gas fields in the Four Corners region of the US using space-based remote sensing. They observed fugitive CH_4 totaling nearly 10% of EPA accounting estimates for all of the US. Miller et al. (2013) surveyed results from aerial and tower-based sensors around the US to adjust CH_4 inventories for ruminants and fossil fuels. They found that CH_4 from fossil fuels could be 4.9 ± 2.6 times larger than in the EPA database. They conclude that both animal husbandry and fossil fuel industries have larger GHG impacts than reporting in the EPA inventories.

1.1.1.2 Bottom-up leakage rates

Using bottom up sampling methods, Kang et al. (2014) measured direct methane leakage from retired oil and gas wells in Western Pennsylvania. The study concluded that the measured leaks from 19 sampled measured wells could be scaled to represent 4-7% of anthropogenic methane emissions in Pennsylvania. Furthermore, the EPA does not account for these methane leaks in current GHG inventories. The study area was expanded to an additional 88 wells from 163 measurements, and increase the estimates to 5-8% of annual anthropogenic methane emissions in Pennsylvania (Kang et al., 2016). The studies underscore the need for a reworked GHG baseline emissions study from the EPA in the US. In any further oil and gas development, it is imperative that old wells are properly surveyed and monitored, as the failure of concrete and steel casing over time can provide a pathway for subsurface migrating methane to be released (Ingraffea et al., 2014). This also demonstrates the need for monitoring wells after production has ceased.

1.1.1.3 Downstream distribution leakages

Phillips et al. (2013) used a cavity-ring-down mobile CH_4 analyser across 785 road miles in Boston to assess gas infrastructure leaks. They found 3356 CH_4 leaks exceeding 15 times the global background level of CH_4. Jackson et al. (2014) used the same sampling method as the Philips et al. in Washington DC and found CH_4 concentrations to be 45 times higher than background concentrations in the atmosphere. Both studies found a δ^{13} CH_4 signature consistent with fossil fuel CH_4, rather than biogenic sources.

Boothroyd et al. (2018) carried out a similar analysis in the UK and found leakage rates along the National Transmissions System (NTS) to be of similar magnitude and density to lower end of US distribution leaks (Chamberlain et al., 2016; Gallagher et al., 2015). However, Boothroyd et al. note that local distribution leak rates remain unclear.

Compatibility of fossil fuel energy system in the UK for climate targets. While these leakages are not directly related to shale gas extraction, the expansion of gas development in the US and UK would need to address infrastructure leaks to lessen impacts on climate.

1.1.1.4 Industry studies

A number of studies have been performed by industry operators in conjunction with partnership with industry lobbying groups such as America's Natural Gas Alliance (ANGA) and The American Petroleum Institute (API). Shires and Lev-On (2012) surveyed 91,000 wells operated by over 20 companies and found methane emissions to be 50% lower than EPA (2010) estimates, but only a small percentage were shale wells. The study claims Howarth et al. (2011) overestimate industry-wide leakage rates due to a small sample size and isolated incidents during well completion and liquid unloading. Specifically, the venting of methane into the atmosphere during liquid unloading is 86% lower than EPA estimates; 72% lower during well re-fracturing, and that re-fracturing rates are significantly lower than estimates. The survey did not randomly sample, and used selected industry wells for measurement.

Another industry study by Allen et al. (2013) measured methane emissions from 190 wells in Texas and found leakages between 0.01 Mg to 17 Mg per event. This was significantly lower than average of 81 Mg per event in the EPA inventory (2017), and corresponds to a fugitive emissions rate 0.42% of lifetime production. While these studies were not performed under best scientific practices, they do indicate that industry operators are capable of producing shale gas with minimal fugitive emissions if monitored, and best practices are enforced. Furthermore, Allen et al. (2013) states it is unlikely and uneconomical that 8% (Howarth et al. (2011) top-end leakage rate) of recovered shale gas is brought to the surface and not captured for industry use.

1.1.3 Shale gas growth

In 2011, the IEA predicted a 50% growth of natural gas by 2035, accounting for more than 25% of global energy demand (IEA, 2011). The IEA based these predictions on four main points: (1) increased gas usage in China, (2) slower growth of nuclear power development, (3) increased usage of natural gas in the transportation sector, and (4) low-cost shale gas.

The IEA (2012) outlines and calls for the adoption of twenty-two rules by governments and industry to minimize the environmental impacts of shale gas. One rule specifically pertinent to fugitive methane emissions requires measurement and disclosure of all air emissions from the supply chain.

The IEA predicts that the adoption of the best practices could increase the overall financial outlay of a shale gas well by 7% (IEA, 2012). However, operational efficiency savings at larger well fields could reduce these costs, and profitably reduce up to 80% of methane emissions (Harvey, 2012), indicating the possible competitiveness of shale gas on a global market, when managing fugitive CH_4.

1.1.4 UK resource estimates and recovery rates

Recent global shale gas and oil estimates have identified 1,013 Tm^3 of shale gas in place with 220 Tm^3 technically recoverable. In the UK, Kuuskraa et al. (2013) updated their 2011 global shale estimates and identified and 0.7 Billion barrels of shale oil and 736 Bm^3 of shale gas which are technically recoverable (see Figure 1-3). These estimates are part of a larger 3.8 Tm^3 of shale gas in-place and 17 billion barrels of shale oil in place. Stamford and Azapagic (2014) increased these figures to 3.6 Tm^3 of recoverable shale gas, based on industry samplings of 4 of the 6 shale basins in the UK. Undoubtedly, resource estimates will continue to climb in the UK, as operators are making the case for profitable shale gas on shore.

Kuuskraa et al. (2013, 2011) have indicated that roughly 20% of US shale gas in place is technically recoverable. However, due to less favourable geology, the estimate that a recovery rate for the UK would be roughly half of the US (Bickle et al., 2012).

Compatibility of fossil fuel energy system in the UK for climate targets. Life cycle emissions of MacKay and Stone (2013) estimate that extracting shale gas for electricity generation in the UK will be in line with the life-cycle emission of on-shore gas and imported LNG, due to the resource being largely onshore. In Scotland, methane emissions could further increase if development of well pads occurs on peat; but these emissions are avoided with best practices (Bond et al. 2014). Stamford and Azapagic (2014) estimate that UK shale gas could emit 412 – 1102 kg CO_2e per $MWh_{(e)}$[2] with a central estimate of 461 kg CO_2e. This central figure is comparable to current North Sea gas power of 401 kg CO_2e per $MWh_{(e)}$.

Westaway et al. (2015a) refute the findings of Stamford and Azapagic (2014) that shale gas would be as dirty as coal in the worst case scenario on the basis of kg CO_2e per MJ. Westaway et al. incorrectly cite MacKay and Stone (2013) figures on carbon intensity of electricity delivered rather than potential chemical energy before combustion. Stamford & Azapagic (2015) note this mistake in their response. This discussion highlights an area of confusion in the literature on the use SI units of chemical energy content and GWP of delivered electricity. Chemical energy (Q_T in g CO_2e MJ^{-1}) is the convention in engineering papers; GWP (Q_E in g CO_2e kWh^{-1}) is the convention in climate change-related papers. The choice of unit either embeds or ignores the efficiency and rate of converting chemical energy to electrical energy through combustion, adding further discussion of the merits of shale gas for achieving climate change targets.

1.1.5 Electricity Market Reform, Energy Performance Standard, and perverse incentives

The UK Energy Act 2013 established an ‘Emissions Performance Standard’ of 450 g CO_2e per $kWh_{\mathbf{(e)}}$ (Energy Act 2013, p.Part 2, Chapter 8) on new baseload power plants. In essence, new baseload power plants would need to be fitted with CCS technology to continue using coal. Old coal plants were exempted from this performance standard through 2015.

[2] Megawatt hours electrical energy.

Compatibility of fossil fuel energy system in the UK for climate targets. When considering the life cycle assessment (LCA) results (MacKay and Stone, 2013; Bond et al., 2014), shale gas appears to be an excellent new source of baseload power in accordance with The Energy Act (Energy Act 2013). In the time between passing The UK Energy Act 2013 and the expiration of the coal exemption two years later, the UK continued coal imports (see Figure 1-4), largely displaced from the US after shale gas made the coal unprofitable to use there. It is now the ambition of the UK energy sector to have shale gas displace some of these coal imports for domestic supply (BEIS, 2018b; Cuadrilla Resources, 2016).

Prior to the expiration of the coal exemption, there was a perverse incentive for UK coal operators to continue importing cheap coal from the US. US coal had become less economical in the US, due to the rise of shale gas. And although the US was claiming decreases in domestic GHG emissions because of shale gas, coal was still being extracted, but exported in greater quantities. As shown in Figure 1-4 below, coal was already in decline in the UK prior to the 2013 rule. However, the well-intentioned rule, gave a perverse incentive to use as much coal as possible during the window prior to expiration of the coal exemption. Although difficult to predict, future carbon-reduction rules should attempt to mitigate against perverse incentives and reduce program design failures which allow for GHG exemptions.

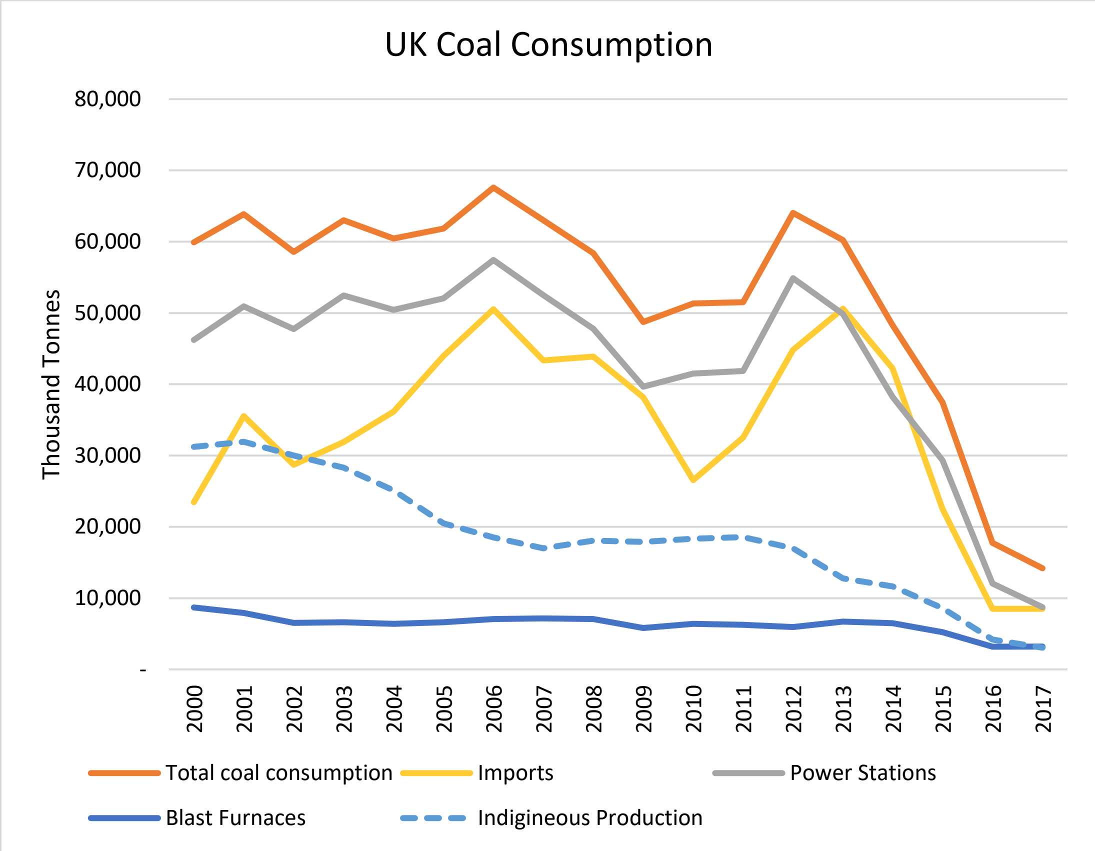


Figure 1-4 - UK Coal Imports, Production, and Consumption, annual data. UK Coal imports dropped from Q4 2008 until a steep climb in 2010 until the end of the exemption in The Energy Act 2013 (Energy Act 2013). Note that Coal exports are inconsequential compared to electricity generation, and imports/production. Data adapted from BEIS (2018a).

1.1.6 Carbon reduction goals

The UK has an ambitious GHG reduction target with legally binding commitment of 80% reduction by 2050 relative to 1990 levels. The Committee on Climate Change (CCC) sets carbon budgets to meet this goal, and suggest that electricity generation should be below 50 g CO_2e per $kWh_{(e)}$ by 2030 (CCC, 2014). The US has no such legally binding target.

Through “The Clean Power Plan,” President Obama proposed a 30% reduction in power plant emissions from 2005 levels by 2030, and a further 40-45% reduction on methane emissions from 2012 levels by 2025 (White House, 2013, 2015; Carbon Pollution Emission Guidelines for Existing Stationary Sources: Electric Utility Generating Units; Final Rule). President Obama also proposed an increase in natural gas capacity to 70%. The US

Compatibility of fossil fuel energy system in the UK for climate targets. GHG baseline from 2005 is 562 g CO_2e per $kWh_{(e)}$, and a 30% reduction would require electricity below 394 g CO_2e per $kWh_{(e)}$ (US EPA, 2014). It should be noted that these emissions figures do not include emissions associated with extraction or transportation of fossil fuels. If these emissions were included, US grid GHG emission factors would be increased to 655 g CO_2e per $kWh_{(e)}$ (Jaramillo et al., 2009).

The Trump Administration cancelled the “Clean Power Plan” prior to taking effect and recently proposed that individual states set their own carbon targets (US EPA, 2018) through the “Affordable Clean Energy Rule”. Under the proposed rule, states would set their own GHG reduction targets, and seek EPA approval. This process would theoretically allow for new coal plants to be built under a state-defined GHG target. The plan could slow down the reduction of coal plant usage for power, but not eliminate the growth of gas-fired power due to the favourable economics of gas compared to coal. The regulation proposal creates a loophole which allows for states to continue using coal power plants while relaxing regulations placed on coal plants (Rodriguez, 2018). It should be noted that this rule is not yet law, and the Clean Power Plan was not ratified either. The delays in regulating emissions in the US place emphasis on the economics of the energy market rather than the regulations.

US and UK plans indicated an increased use of natural gas as baseload power for domestic supply to meet GHG reduction goals. When combined with CCS, shale gas could support both the UK and US plans to meet reduced GHG electricity supply and maintain a 50% chance of keeping temperature rise below +2°C throughout the 21st century. However, the cumulative global carbon emissions between 2011 and 2050 must be limited to around 1,100 Gt CO_2e, while current global fossil fuel reserves represent three times this quantity of embodied CO_2e (McGlade et al., 2013; McGlade and Ekins, 2015; Ekins et al., 2013).

1.1.7 Science-based targets

One possible mechanism for to maintain the carbon budget and Paris Agreement ambition of staying below +2°C throughout the 21st century is the Science-Based Targets Initiative (SBTi). Through partnership with the CDP, UN Global Compact, World Resources Initiative (WRI), and World Wildlife Fund (WWF), companies declare actions and policies for decarbonization targets which evolve with the growing understanding of remaining carbon budgets. The SBTi encourages both absolute and intensity targets for companies, and purchasing of renewable energy under the GHG Protocol guidance (WRI, 2004). To be verified by the SBTi, targets must represent a 2.1% year-on-year decline in total emissions, in line with the RCP 2.6 pathway in IPCC AR5 (IPCC, 2013a), which is aligned with the +2°C budget.

The SBTi does not supersede national carbon budgets, rather seeks to be geographically ambivalent, and provide global guidance for corporate actions aligned with targets of the Paris Agreement. The initiative highlights global leaders through the CDP reporting mechanism, however, it is entirely voluntary.

It is not clear how the SBTi will adjust the GHG budget as the understanding of the GWP of GHGs evolves. As discussed above, the GWP for CH_4 has increased from 21 to 34, however carbon accounting and carbon markets accept values from AR4 and AR5. It is therefore possible, that ambitions and budgets could experience price shocks as GWPs are further characterized (see above), and the science-based target (SBTs) move accordingly.

1.1.8 Accounting and emissions transfers

Under current GHG accounting rules, emissions are attributed to the country where GHGs are emitted or produced, “production-based” (PB) accounting. For example, if shale gas extraction emissions occur in the US, but the final combustion of shale gas occurs in the UK, the extraction emissions are counted in the US and the combustion in the UK. An alternative method attributes all of the embodied emissions in a good to count against the budget where the final good is consumed, or “consumption-based” (CB) accounting. In the prior example, all emissions associated with the production, transportation and combustion emissions would count towards UK GHG budgets and targets. The PB system allows for countries to transfer, or offshore, the emissions associate with their goods. In the case of this thesis, the UK could potentially export the GHG responsibility of the extraction of natural gas to another country while consuming the same quantity of gas. If the imported gas displaces gas which would have otherwise been produced in the UK, the UK would see a net savings in GHG budgets, while atmospheric emissions may not see the same savings.

Peters et al. (2011) describe the above perverse incentive as “emissions transfers,” and they have grown from 4.3 Gt CO_2 in 1990 to 7.8 Gt CO_2 in 2008, representing an increase from 20% to 26% of global GHG emissions. This rise in CB emissions has occurred while developed countries (Kyoto Protocol Annex B countries) have stabilized their emissions and developing countries (Kyoto Protocol non-Annex B countries) have seen their emissions double (Peters et al., 2011; Fischer, 2011).

This perverse incentive could be decreased if countries counted their CB emissions and/or were signatories to the SBTi. However, the SBTi is currently being developed for corporate targets. Furthermore the CB emissions initiative would require new accounting protocols, and would necessitate rebalancing international carbon targets and budgets.

1.2 Thesis structure

This thesis is structured as a collection of research papers discussing the GHG impacts of gas usage through different segments of the supply chain to end use (See Figure 1-5 & Figure 1-6). The objectives of this work are to use shale gas extraction and trade as a series of case studies for examining GHG emissions through the natural gas supply chain, and point out gaps in the carbon accounting budgets for the UK under different gas production scenarios. Two papers, presented in Chapters 2 and 3, from this work have been published in peer reviewed journals. A third paper, presented in Chapter 4, is currently submitted and under review.

Chapter 2 outlines and discusses the potential for enhanced oil recovery to aid in the funding for carbon capture and storage with upstream gas sources from the UK and abroad.

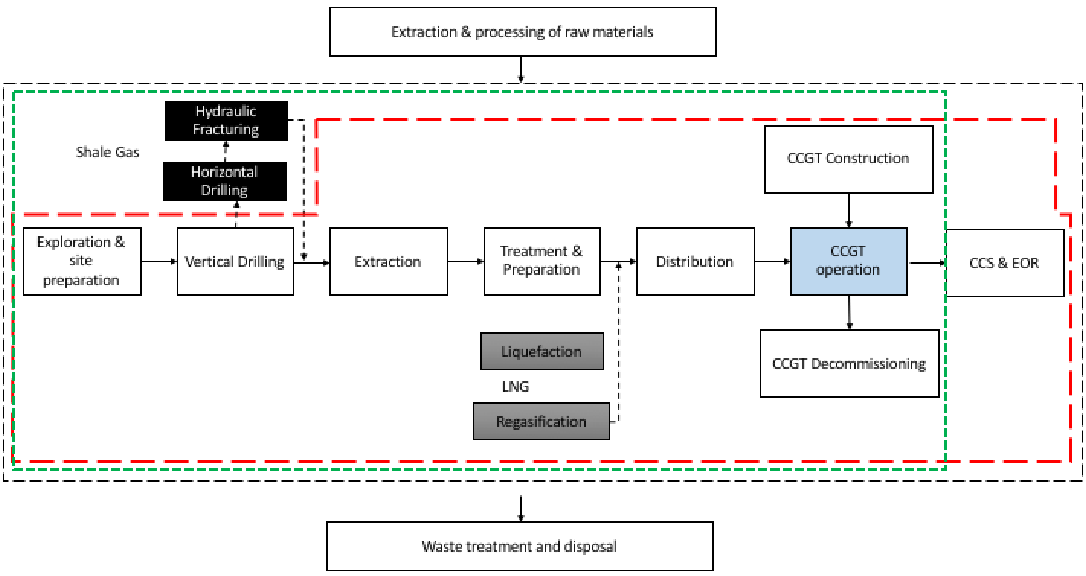


Figure 1-5 – Simplified shale gas schematic showing distinct phases in extraction, transportation, and usage. Chapter 2 discusses the usage of gas for electricity aided by CCS and EOR (red area). Chapter 3 discuss multiple sources of upstream gas, including shale gas in the UK power sector (green area). Figure adapted from (Stamford and Azapagic, 2014)

Chapter 3 expands the "gate" upstream to include UK shale gas for domestic power supply, and the usage of imported US shale gas for electricity production compared to the business as usual case. This chapter uses the well-characterized UK projections for gas demand as a basis for calculating emissions compared to UK climate targets and carbon budgets (BEIS, 2017e, 2016a).

Chapter 4 discusses the impact of importing US shale gas as LNG to the UK or extracting UK shale gas for domestic heat production in light of initiatives to improve pipeline distribution networks in the US and UK.

Chapter 5 summarizes the results of this work and makes recommendations for policy makers and areas of future study.

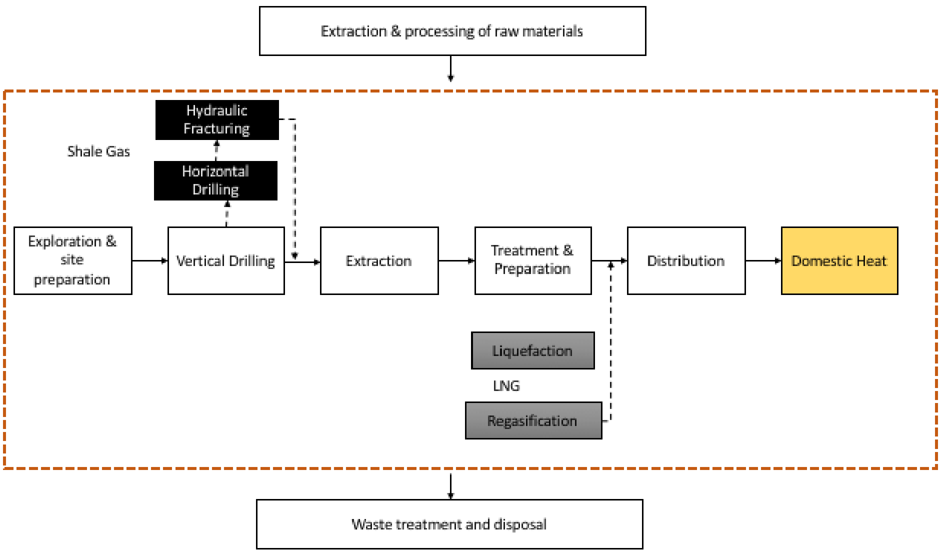


Figure 1-6 - Simplified shale gas schematic showing distinct phases in extraction, transportation, and usage. Chapter 4 discusses the usage of shale gas from US and UK in the distribution phases for domestic heat (red area). Figure adapted from (Stamford and Azapagic, 2014)

Chapter 2 UK grid electricity carbon intensity can be reduced by enhanced oil recovery with CO_2 sequestration

Summary

Enhanced Oil Recovery (EOR) using CO_2 coupled with Carbon Capture and Storage (CCS) can potentially accelerate CO_2 storage investment through creation of a large commercial market for EOR. This chapter assesses how coupled a CCS-EOR scenario might contribute to decarbonization of UK grid electricity. Progressive introduction of 11 CCS-to-EOR gas-power plant projects from 2020 is estimated to store 52 Mt CO_2 yr^{-1} from 2030. These 11 projects produce extra revenue of 1100 MM bbls of taxable EOR oil from 2020 to 2049. After each 20-year EOR project ceases, its infrastructure is paid for, and has many years of life. UK climate change targets would necessitate continued CO_2 storage at low cost. Considering all greenhouse gas emissions – from power generation, CCS-EOR operations, and oil production and combustion – this project suite emits an estimated 940–1068 Mt CO_2e from 2020 to 2049, while storing 1358 Mt CO_2. The total average electricity grid factor in the UK reduces to 90–142 kg CO_2e MWh^{-1}, with gas generating 132 TWh yr^{-1}. This life-cycle analysis (LCA) is unusual in linking oil production and combustion with CCS and gas-fueled electricity, yet provides a net carbon reduction, and progressively reduces net oil combustion emissions beyond 2040.

Work presented in this chapter is based on the manuscript published as: Turk, J. K., Reay, D. S., and Haszeldine R. S., UK grid electricity carbon intensity can be reduced by enhanced oil recovery with CO_2 sequestration, Carbon Management, vol. 9, pp 115-126, 2018, https://doi.org/10.1080/17583004.2018.1435959.

2.1 Introduction

The UK is legally bound to reduce greenhouse gas (GHG) emissions by 80% of 1990 levels by 2050. Within this goal, the Committee on Climate Change (CCC) recommends the GHG intensity of electricity generation fall to 50-100 kg CO_2 per megawatt hour (MWh) by 2030 (CCC, 2015a, 2010). Total UK electricity generation (including pumped storage) fell by 5.6% from 359 terawatt-hours (TWh) in 2013 to 339 in 2015; gas-fired generation increased from 96 TWh in 2013 to 100 TWh in 2015 representing 30% of total supply (DECC, 2015a; BEIS, 2016a). Coal-fired generation fell from 131 TWh in 2013 to 76 TWh in 2015, and is projected to continue to decline rapidly through the next decade (DECC, 2014b, 2015b; BEIS, 2017e) due to Electricity Market Reform (Energy Act 2013) and The Industrial Emissions Directive (Directive 2010/75/EU of the European Parliament and of the Council of 24 November 2010 on industrial emissions (integrated pollution prevention and control)). However, according to Government projections published in 2017, the UK is projected to need 96 gigawatts (GW) of new peak electricity generation capacity by 2035 to replace coal-fired generation, support renewables intermittency, and meet decarbonisation goals. Unabated gas power is projected to deliver 35 TWh by 2035, up from 28 TWh in the previous projections (BEIS, 2017e; DECC, 2015b). UK's Department for Business, Energy & Industrial Strategy (BEIS) indicates that this 20% increase in unabated gas electricity delivery will come from gas infrastructure with 20% reduced capacity in their latest projections (BEIS, 2017e; DECC, 2015b) for the years 2031-2035 compared to previous projections. This suggests that gas power will continue be used in very high load factors, being relied upon for peaking power, yet there will be less built capacity. This indicates a projected reliance on a high level of gas fossil fuels in power generation.

GHG emissions from power generation are traded as part of a UK EU-ETS capped market, and do not affect the UK's ability to meet its own carbon budgets. The continued use of fossil fuels by major power producers (MPPs), companies whose primary activity is electricity generation (DECC, 2012), maintains a domestic demand for fossil fuels which can create consequential emissions in other non-traded sectors of the economy (e.g. fugitive CH_4 emissions from coal and gas extraction). Despite cuts in coal-fired capacity (Directive 2010/75/EU of the European Parliament and of the Council of 24 November 2010 on industrial emissions (integrated pollution prevention and control); Energy Act 2013) and increased renewable capacity, BEIS indicates a gap in meeting Carbon Budgets 4 and 5 by 146 and 247 Mt CO_2e respectively (BEIS, 2017e). The projections include a 27% increase in interconnection capacity by 2035, which delivers electricity that does not affect the UK carbon targets, and is ostensibly zero carbon for the UK. Uncertainty in reaching grid decarbonisation goals through UK electricity generation places increased reliance on buying electricity through interconnection from elsewhere in Europe, and carbon offsets through the EU Emissions Trading Scheme (EU ETS). With the UK's place in the EU now in political negotiation, and hence EU-ETS will not be available in future, this chapter examines an independent multi-decadal plan to aid UK grid decarbonisation, and provide domestic oil production**.**

Carbon Capture and Storage (CCS) use in conjunction with large point sources of CO_2 emissions, such as power plants, is a key technology to support plans to meet UK decarbonisation goals (DECC, 2015b, 2014b), but investment and profitability issues have thus far limited CCS development in the UK (Scott et al., 2012; Carrington, 2015; BBC News, 2015; Oxburgh, 2016). The UK's Department of Environment and Climate Change (DECC) (2013b) concluded that electricity with CCS in the UK would cost between £64 – £128 MWh^{-1} while electricity with unabated gas costs between £63 – £109 MWh^{-1}. Similarly, Rubin and Zhai (2012) reported the cost of CCS as \$76 – 114 MWh^{-1} (mean \$93) and unabated gas as \$52 – 75 MWh^{-1} (mean \$63). The latter study also estimated a carbon tax of \$73 per t CO2 as being the breakeven point where CCS and unabated gas power generation reach equal cost. Welkenhuysen et al. (2017) probabilistically allocated Monte Carlo cycles to oil production uncertainty and selling price uncertainty and found that Enhanced Oil Recovery (EOR) in the North Sea has positive Net

Present Value with an oil price above €50 bbl^{-1}. In the absence of a direct CCS subsidy or a high carbon price, EOR coupled with CCS has the potential to bridge this gap in costs between unabated gas and CCS (Haszeldine, 2016; Welkenhuysen et al., 2017).

Here, this chapter examines the potential impact of such a transition to coupled EOR-CCS on grid intensity and greenhouse gas (GHG) emissions in the UK through to 2035 and beyond. This chapter assesses the size and number of EOR-supported CCS projects needed to satisfy projected CCS capacity projections by DECC (2015b), electricity decarbonisation targets (CCC, 2010, 2015a), and CO_2 storage goals by Element Energy (Durusut et al., 2013). This chapter provides the unique perspective of extending LCA estimates of grid electricity and downstream interventions to reduce grid emissions in the UK via EOR-CCS. Previous EOR studies excluded emissions associated with venting and flaring recycled CO_2 and CH_4 (Hertwich et al., 2008), or were too dissimilar in location and upstream fuel type (i.e. Canadian coal, Manuilova et al., 2014). Note that a modified amount of CCS electricity predicted in 2017 (BEIS, 2017e) for the UK is linked to the perception of CCS under current Government policies, it is not a target or pathway. In simple terms the amount of CCS electricity changed from 2016 (DECC, 2015b) to 2017 (BEIS, 2017e) has been replaced by a similar amount of “zero carbon” electricity imports, with no plan for delivery. In this analysis, this chapter offers scenarios for a cash-poor Government to obtain development of CCS, maintain electricity, keep high employment and efficient oil extraction from the UK offshore.

2.2 Methodology

In order to assess cradle-to-grave emissions of an EOR chain with natural gas combined cycle (NGCC) electricity, first the Life Cycle Analysis (LCA) of Stewart and Haszeldine (2015) is extended to include upstream phases (see Figure 2-1). Stewart and Haszeldine (2015) examined two EOR scenarios to assess GHG balances in a CO_2 storage-focused system compared to an oil recovery-focused system. Here a storage-focused system is used and the

Compatibility of fossil fuel energy system in the UK for climate targets. LCA scope extended upstream to include power capacity needs, coupled EOR-CCS development over the next 20 years, and further CO_2 storage via CCS beyond 2040. Throughout the estimations, it is assumed that CO_2 stored through EOR would have otherwise been emitted through unabated gas power. This unabated gas pathway in the UK is implied by DECC's 2015 projections of development of new-built gas capacity, used intensively to deliver electricity from unabated gas through to 2035 (DECC, 2015b; BEIS, 2017e). Stewart and Haszeldine (2015) focused on EOR emissions and present results in t CO_2 emitted or stored per barrel of incremental oil produced. This chapter shifts the focus upstream to examine implications of the modelled interventions on grid electricity targets. The EOR literature is split between including (Jaramillo et al., 2009; Condor et al., 2010) or excluding (Faltinson and Gunter, 2011) oil emissions in estimates. This chapter incorporates the CCS and EOR emissions, along with imported and produced oil emissions during EOR because they are a direct result of EOR interventions, and encompass the net GHG emissions more fully. To make comparisons with the business as usual (BAU) unabated gas pathway in kg CO_2e MWh^{-1}, the modelled sums of all emissions are divided by the delivered grid electricity to give LCA estimates in kg CO_2e MWh^{-1}, units normally used for grid factors not LCAs.

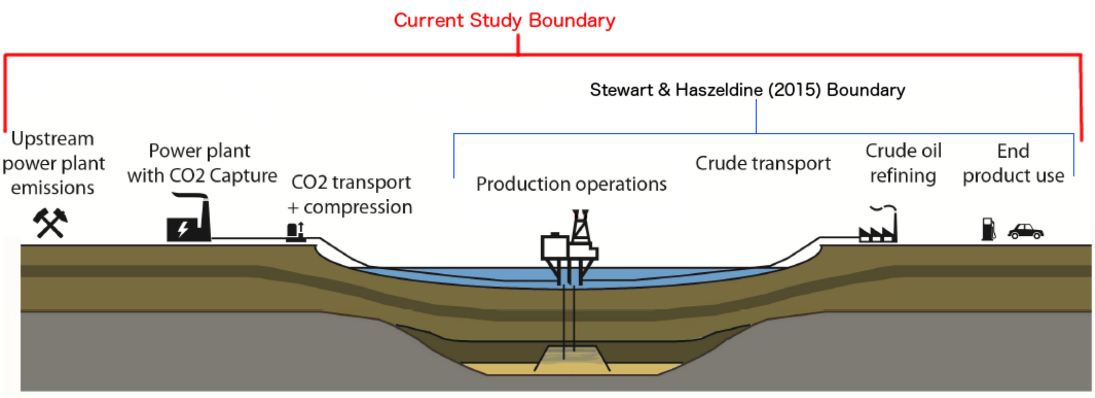


Figure 2-1 – In this chapter, the boundary of Stewart and Haszeldine (2015) is extended upstream to include fossil fuel and power plant emissions. Figure adapted from Stewart and Haszeldine (2015).

2.2.1 Natural gas supply and demand

In 2015, the UK consumed 741 TWh of gas from domestic and imported sources (DECC, 2015b; BEIS, 2016a). The total gas supplied was 861 TWh, of which 47.7% was met by UK sources. Imported pipeline gas (34.7%) came from Norway, Belgium, and The Netherlands via interconnections (BEIS, 2016b). An additional 152 TWh of liquefied natural gas (LNG) was supplied from Qatar (16.4%), Algeria (0.6%), Trinidad & Tobago (0.6%), and Nigeria (< 0.1%). UK net gas production was 410 TWh in 2015, (460 TWh total minus 50 TWh used by producers), and UK gas production is expected to fall by 5% yr^{-1} from 2020 until 2035, while demand is expected to remain steady (DECC, 2015b; BEIS, 2016b; OGA, 2017).

To satisfy the predicted requirement for electricity generation capacity in 2035 (DECC, 2015b) and CO_2 storage required to meet grid electricity GHG targets (Durusut et al., 2013), this chapter therefore develops a scenario based around the build of 11 new gas-fuelled electricity power plants, each of 1,930 MW capacity and producing 5.56 Mt CO_2 yr^{-1} per plant (see below). Then, the grid emissions intensity of this portfolio of unabated gas plants is compared to the same portfolio fitted with CCS and coupled to EOR operations offshore (North Sea). Under this scenario, additional oil produced through the coupled EOR-CCS system would displace imported oil production and would use CO_2 from each of the 11 power plants.

For the purposes of estimating upstream GHG emissions it is also assumed that the gas power plants are supplied with gas from the UK National Transmission System (NTS). When gas arrives in the UK via interconnection or LNG terminal, it enters the NTS making the geographic origin for the customer unknown. The net gas flows between the UK and Belgium and The Netherlands have represented +/- 1% of total supply since 2010 (BEIS, 2016b), and are therefore regarded as irrelevant for GHG estimation in this study. In the absence of nation-specific data, LNG imports from Nigeria and Trinidad & Tobago are assumed to have the same GHG emissions footprint as LNG from Algeria. Finally, imported North Sea gas is assumed to have the same emissions footprint as UK domestic gas.

To estimate UK gas supply, usage, and related GHG emissions to 2050 this chapter uses the projected gas demand from DECC (2015b), in the same geographic proportions (places of origin) as the current gas supply entering the NTS, as described above.

2.2.2 CCS and future UK electricity supply

The assumptions on UK electricity production and CCS roll-out are based on the DECC *reference scenario* (DECC, 2015b). Recent projections by BEIS (BEIS, 2017e) severely cut expected CCS capacity to just 963 MW in 2035, the final year of the projections. This modelled cut is in response to the UK Government’s withdrawal of £1 billion funding for CCS. These same BEIS projections also indicate a gap in Carbon Budgets 4 and 5, described above. The UK Government has since released a plan to invest up to £100 million in CCS development (BEIS, 2017d), which would still fall short of reaching mid-century carbon reduction goals (Hickman et al., 2017). Considering these recent funding cuts, and reiterated gaps in legally-binding carbon targets and pathways, this chapter considers the previous CCS projections from DECC (DECC, 2015b) and storage targets from Element Energy (Durusut et al., 2013) as benchmarks for grid decarbonisation goals and actions. The project output is multiplied by 11 to achieve these benchmarks, and explore the needed capacity (see below).

An estimated 365 TWh of electricity would be produced domestically in the UK in 2035 with nearly 40 TWh of this expected to be produced from *coal and natural gas CCS*. Unabated natural gas is projected to supply 44 TWh by 2035 (DECC, 2015b). In the shorter term it is assumed that – again, based on the DECC reference scenario - 638 MW of CCS capacity will be in place by 2019, increasing to 8,382 MW by 2035. DECC (2015b) projects that the electricity generated by CCS plants would then be utilised for base load power (85% load factor) by 2022, then reduce to less than 70% after 2030 (see Figure 2-2). A reduction in CCS load factor could have adverse effects on CO_2 capture rates and delivery to offshore injection sites. For simplicity, it is assumed that the electricity generated by the CCS plants would then be utilised for baseload power from 2022 onwards with a continuous 85% load factor.

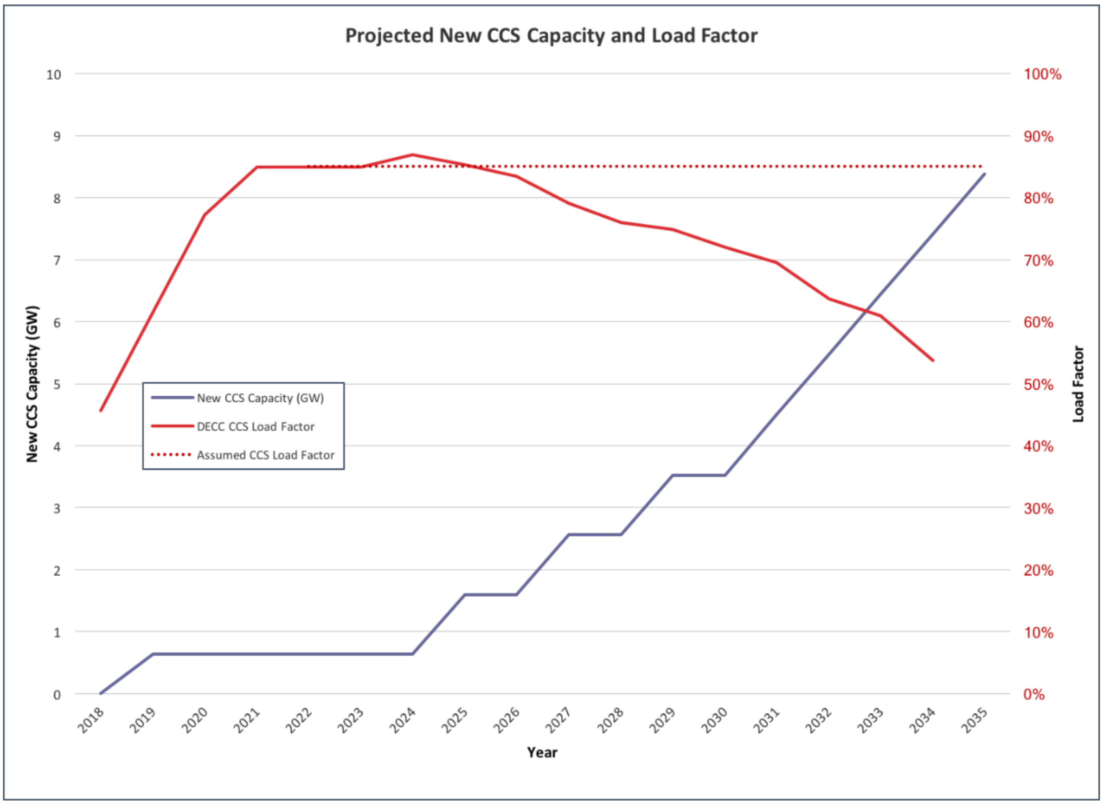


Figure 2-2 - Projected new CCS capacity from DECC (DECC, 2015b) reference scenario. Load factor is calculated from projected delivered electricity divided by capacity. DECC projections suggest using CCS plants for baseload power (85% load factor) early next decade before reducing to 70% after 2030. For simplicity, a continuous 85% load factor is assumed in the models.

To assess the GHG impact of natural gas electricity generation and infrastructure, a LCA was scaled from Stamford and Azapagic (2014). All phase emissions in the Stamford and Azapagic LCA are proportionally increased, and operational emissions equal to the onshore CO_2 required for Stewart and Haszeldine (2015). This assumes equal combustion (operational) emissions at a gas plant regardless of gas origin and so shifts CO_2 emissions variance for natural gas power plants to production and transportation emissions. The scaling method without CO_2 capture indicates that 96.4% of emissions associated with North Sea gas power are from combustion emissions, with a total output of 14.4 TWh yr^{-1} per plant before CO_2 capture. The combustion emissions intensity for all natural gas plants are assumed to be equal and from steady state combustion, at 386.3 kg CO_2e MWh^{-1}. This figure is from a proprietary dataset by Ecoinvent (Ecoinvent Centre, 2010; Weidema and Hischier, 2006), and falls within the calculated range of 365 – 415 kg CO_2e MWh^{-1} annually reported by DECC (2015a, 2014a, 2013a, 2012). As such, onshore emissions total 5.56 Mt CO_2 yr^{-1} for each of the 11 new unabated gas-fired power plants in the projections. This is considered the BAU case, projected to supply 44 TWh of unabated electricity (DECC, 2015b) by 2030.

For coupled EOR-CCS it is assumed that 90% of CO_2 operational emissions from the gas power plants (equivalent to 5.0 Mt CO_2 yr^{-1}) will be captured onshore and transferred offshore to EOR operations (see Figure 2-1). The carbon capture energy penalty was assumed to be 16% of total output - the mean value of recent findings ranging from 13% to 18% for natural gas combined cycle (NGCC) retrofits (Rubin et al., 2015). Note that Stamford and Azapagic (2014) assume a plant efficiency of 52.5% based on lower heating value (LHV), whereas DECC (2015a) lists the average for the UK as 47.0% based on LHV. Using the DECC figures would increase the combustion emissions relative to the 52.5% LHV value used in the projections.

The upstream emissions associated with CO_2 capture for EOR from Stewart and Haszeldine (2015) are identical to the CCS upstream emissions for each respective 20-year case study considered here. Both CCS and EOR models capture 90% of CO_2 from their respective power plants, and have identical grid electricity outputs for the fuel sources. The EOR models have additional operational emissions and reduced CO_2 storage associated with venting, and CO_2 recycling as described in Stewart and Haszeldine (2015). It is assumed that post-capture CO_2 or EOR does not need additional compression for, or during, transportation to offshore platforms, compared to the CCS case. It is assumed that the energy cost associated with compression and transportation of CO_2 to offshore platforms is minimal. For pipeline distances greater than 1,000 km, 6.5 kWh per tonne CO_2 is required for recompression (Jaramillo et al., 2009). This equates to less than 0.3% additional energy costs per project. When including coupled EOR-CCS operations, this figure is further diminished. In an offshore environment, these costs could increase, but would be mitigated by clustering of projects (Welkenhuysen et al., 2017), reuse of current pipeline infrastructure (Pershad and Slater, 2007), and shorter pipeline distances (Jaramillo et al., 2009). This is indeed an area in need of further study, but is not cost-prohibitive or significantly impactful on the full project GHG balances. Any energy and associated emissions required to compress or recompress CO_2 specifically for EOR are therefore excluded.

Offshore, CO_2 injection operations incur one-time fixed emissions demonstrated in EOR models (Stewart and Haszeldine, 2015). New well drilling, well workover, and steel construction are 45,816 t CO_2e. For coupled CCS-EOR, annual offshore operational emissions include fugitive CO_2, and additional offshore CO_2 compression for injection equal to 57,723 t CO_2e yr^{-1} (Stewart and Haszeldine, 2015). It is assumed that greater than 4 Gt CO_2 storage capacity exists in UK offshore oilfields with greater than 2,000 MM bbls recoverable EOR-oil (Pershad and Slater, 2007; Holloway et al., 2006). Liability and decommissioning costs for these systems remains uncertain (Pale Blue Dot Energy et al., 2016; Pershad and Slater, 2007). For this chapter zero liability is assumed, and decommissioning costs are excluded. Costs associated with extending the life of offshore platforms can be offset by proper investment of funds for decommissioning. Welkenhuysen et al. (2017) suggest that interest payments will create a positive cash flow, more than covering the cost of extending the life of offshore operational infrastructure. The storage liability would transfer at sale to an operator who can profit from production of EOR oil, but would likely transfer back to government if EU ETS CO_2 prices remain low. However, this is an area in need of future study.

2.2.3 Geographic origins of natural gas

To examine the sensitivity of modelled GHG emissions estimates to changing geographical origins of UK gas supply, this chapter also projects the supply of gas forward to 2050 based on DECC projections as discussed above. The presumed decline of UK North Sea gas generates three sub-models which is explored, specifically:

- **Fuel-Mix 1 (S1)**. The 2015 geographic distribution of natural gas supply extends through 2050 in the same distribution as 2015. The total gas demand fluctuates according to DECC and Oil & Gas Authority (OGA) (OGA, 2016, 2017; BEIS, 2016a) estimates to 2035, the 2035 value extends to 2050. This gives the baseline emissions figures.

- **Fuel-Mix 2 (S2)**. The total UK gas demand fluctuates identically to fuel-mix 1. As UK gas declines in production from 2020 through 2050 (DECC estimates end in 2035), North Sea gas from Norway increases to meet gas demand in place of falling UK production. Imported LNG supplies the same percentage of demand from 2015 forward. This demonstrates the shifting responsibility of extraction and transportation emission away from the UK, while total atmospheric emissions are equal.

- **Fuel-Mix 3 (S3)**. UK gas demand fluctuates as in fuel mix 1 and 2, UK gas declines in production from 2020 through 2050, Qatari LNG increases in supply to meet gas demand in place of falling UK production. This model also demonstrates a shift of responsibility of extraction and transportation away from the UK, and places more emphasis on GHG-intense LNG from Qatar.

2.3 Results and discussion

2.3.1 EOR emissions vs. CCS emissions

Without the use of EOR, a single CCS project can be deployed to nearly meet the DECC projection of 5 Mt CO_2 injected yr^{-1} by 2030 (DECC, 2014b, 2013a). Any of the CCS projects modelled here would inject 4.95 Mt CO_2 yr^{-1} starting in year 1 of injection, and continue for 20 years. The coupled EOR-CCS model projects 100 MM bbls of EOR-produced oil production over the course of the 20-year project. This oil contains an additional 44.83 Mt CO_2e in downstream emissions from oil refining (4.47 Mt CO_2e) and combustion (40.36 Mt CO_2e). It is assumed that the EOR-produced oil negates 100 MM bbls of oil being imported to the UK, thereby negating extraction and transportation emissions from oil produced elsewhere. In the case of Saudi oil, extraction emissions for 100 MM bbls are 3.93 Mt CO_2e and transporting this oil emits a further 0.62 Mt CO_2e (Mangmeechai, 2009). These values assume port-to-port emissions from Saudi Arabia to USA, transportation to the UK would likely be lower.

A key assumption of the EOR operation is that the 100 MM bbls of incremental oil will negate the same quantity of imported oil from another source. If imported oil is Saudi light oil, extraction, transportation, refining, and combustion of Saudi oil is 493.8 kg CO_2e bbl^{-1} (Mangmeechai, 2009) compared with 448.3 kg CO_2e bbl^{-1} for domestic EOR-produced oil described above. For the purposes of this study, the North Sea EOR-produced oil is assumed to have negligible transportation emissions; production emissions are counted in the EOR model.

2.3.2 EOR-to-CCS project

Under the coupled EOR-CCS scenario, after a 20-year EOR project is completed the complete infrastructure for further CCS has been financially subsidized by production of 100 MM bbls of EOR-oil. This would represent a very substantial saving for the UK Treasury as no CCS subsidy is needed. It would also represent a saving for the UK electricity customer, as infrastructure costs are paid via EOR revenue rather than rising electricity prices. A full economic analysis is beyond the scope of this study, but would be an important next step for this approach.

Compatibility of fossil fuel energy system in the UK for climate targets. After a typical 20 years of CO_2-EOR oil production lifespan, the pipeline and borehole infrastructure has 20 – 40 years of remaining life, and can be converted to operate as injection of pure CO_2 for storage into storage destinations well proven by CO_2-EOR injection (Pale Blue Dot Energy et al., 2016). If each EOR project transitions to CCS for an additional 20 years, annual injected CO_2 increases from 4.62 Mt CO_2 yr^{-1} to 4.95 Mt CO_2 yr^{-1} and annual net emissions reduction increases from -0.94 Mt CO_2 yr^{-1} (coupled EOR-CCS) to -1.66 Mt CO_2 yr^{-1} (CCS-only) when oil imports resume.

As such, a 40-year EOR-to-CCS project that switches to CCS-only in year 20 and uses only North Sea Gas, stores 192.7 Mt CO_2e, and emits 139.5 Mt CO_2e – a net carbon reduction of over 50 Mt CO_2e per project. These figures include 100 MM bbls of EOR oil in the first 20 years, and 100 MM bbls of Saudi Oil imported during the final 20 years. During the EOR phase, the project produces grid electricity while emitting 305 kg CO_2e MWh^{-1}, including the emissions associated with the EOR-produced oil. During the CCS phase, grid electricity is produced at 273 kg CO_2e MWh^{-1} including the emissions from imported Saudi oil.

If during the CCS phase, no oil is imported to replace the EOR-oil supply, grid electricity emissions intensity reduces to 68 kg CO_2e MWh^{-1} for a single CCS project. Of course, a project does not use gas from one geographic origin, rather what is supplied by the NTS as discussed above.

Stamford and Azapagic (2014, 2012) assumed a lifespan of 25 years for a gas plant. It is here assumed that, because construction emissions are less than 0.2% of total lifecycle emissions (Stamford and Azapagic, 2014), the extension of a service life of a gas plant to 40 years for the EOR-to-CCS model would have a negligible impact on overall project emissions savings.

2.4 EOR-to-CCS in the context of UK climate targets

DECC (2015b) projects that new CCS will contribute 638 MW of new power capacity in 2019, growing to 3,527 MW in 2030 and 8,382 MW in 2035. However, this capacity will contribute 2.55 TWh of electricity in 2019, growing to 23.14 TWh in 2030, and reaching 39.5 TWh in 2035. A single EOR-to-CCS project delivers 12.1 TWh to the grid from 1,930 MW of capacity, and meets the DECC CCS capacity and electricity projections for 2025.

The Committee on Climate Change's (CCC) Fourth Carbon Budget suggested that the carbon intensity of electricity in the UK would need to fall to 50 g CO_2e kWh^{-1} by 2030 with a mixture of renewables, nuclear, and CCS (CCC, 2010). In response to this, Element Energy (Durusut et al., 2013) modelled scenarios to fulfil the carbon intensity target, storing 52 Mt CO_2 in 2030 with coal CCS, gas CCS, and industrial sites. This is a highly ambitious scenario in this context. However, this scenario is included to illustrate the very large commitment required by the UK to reach the 50 Mt CO_2 yr^{-1} stored by 2030. This is of particular importance and relevance because of the statement made by Element Energy that, without CCS, reaching climate targets would double costs to all industries from a minimum £30 billion per year in 2050 (Durusut et al., 2015). This chapter calculates that reaching this storage goal is possible with only gas power through 11 EOR-to-CCS projects, starting one per year in 2020 (a scenario that is also in-line with DECC and BEIS projections of new unabated gas capacity in the UK through 2035 (BEIS, 2017e; DECC, 2015b)).

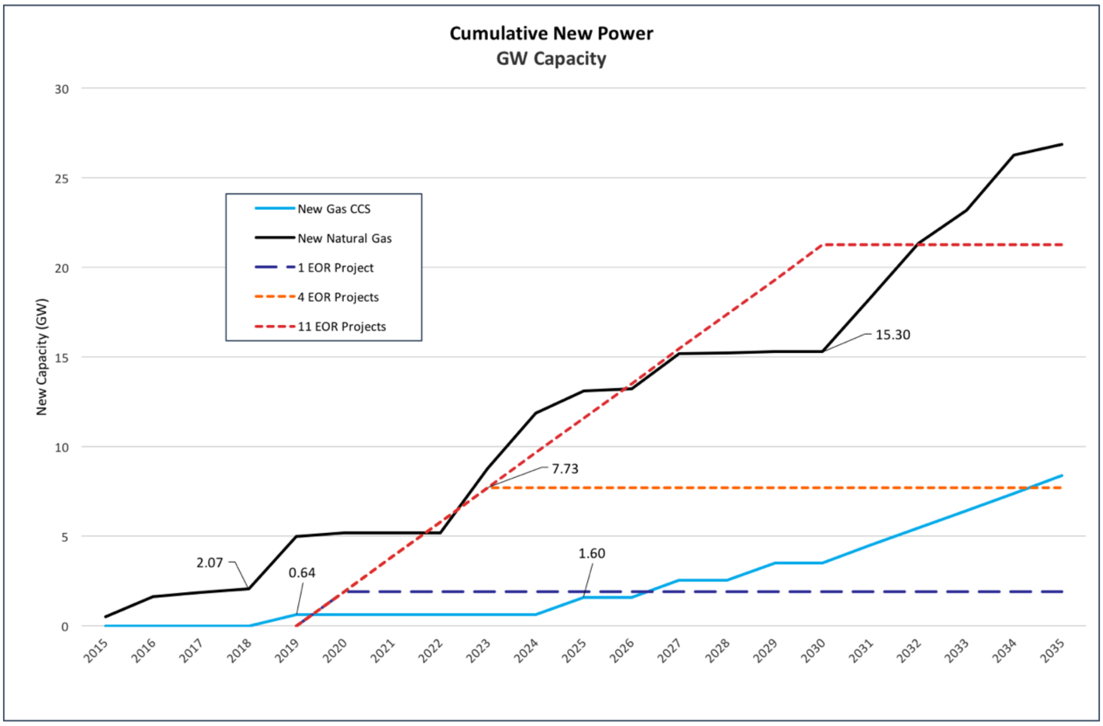


Figure 2-3 - Cumulative required new power capacity from 2015-2035 (DECC, 2015b). DECC projects that new CCS capacity will begin in 2019 with 640 MW (blue line) while 5 GW of new unabated gas capacity will be installed at the same time (black line). Unabated gas is projected to contribute over 27 GW of capacity by 2035. The projections of gas capacity with CO_2-EOR and CCS will contribute 1.9 GW (dashed blue), 7.7 GW (dashed orange), and 21.2 GW (dashed red) of capacity.

By deploying 1 EOR-to-CCS project per year from 2020 onwards, the suite of projects examined here also builds capacity at the same pace as DECC's projection (DECC, 2015b) for unabated gas power, while capturing CO_2, and lowering the grid intensity of UK electricity supply. As previously stated, deploying just one project meets the current projections for CCS capacity projections (see Figure 2-3).

If CCS alone were deployed to meet the Element Energy target, 54.5 Mt CO_2 would be injected by year 11 (2030), storing 1,416 Mt CO_2 by 2050. However, EOR does reduce the CO_2 injection rate after year 2. 11 EOR projects would inject 51.8 Mt CO_2 by 2030, meeting the Element Energy (Durusut et al., 2013) goal while providing the additional revenue required for expansion of CCS infrastructure. If these 11 EOR projects were mandated to continue to CCS after 20 years of EOR, 1,358 Mt CO_2 would be stored by 2050; that is a 56 Mt reduction in CO_2 emissions compared to using only CCS and no EOR, with oil imported from Saudi Arabia. Considering all emissions with EOR and CCS operations, including combustion of resumed Saudi oil imports, this suite of projects emits 1,076 – 1,204 Mt CO_2 from 2020 to 2049, while storing 1,358 Mt CO_2. If Saudi oil imports do not resume after EOR operations, total emissions are reduced to 940 – 1,068 Mt CO_2 from 2020 to 2049.

2.4.1 Coupled EOR-CCS and UK oil sources

The suite of coupled EOR-CCS projects considered here would provide 132 TWh of grid electricity by 2030 between 152 – 184 kg CO_2e MWh^{-1} without considering the EOR-produced oil. As each EOR project reaches its 20th year, and transitions to CCS only, the average grid intensity drops to 90 - 142 kg CO_2e MWh^{-1} by 2049, assuming oil is not imported or combusted to replace the EOR-oil – thus a net reduction of oil use.

Emissions from CCS offshore operations are 1.2 Mt CO_2 for 20 years and for EOR total around 13.5 Mt CO_2 for 20 years. 100 MM bbls of EOR-oil contains 44.83 Mt CO_2 (assuming normal combustion); 100 MM bbls of Saudi Oil (imports during CCS with combustion) contains 49.38 Mt CO_2. The sum of these differences indicates that a CCS project emits 7.75 Mt CO_2 less than an EOR project over 20 years when comparing CO_2 sourced from the same power plant. Therefore, if CCS is performed while importing oil with greater than 571.3 kg CO_2 bbl^{-1} embedded, EOR is actually more advantageous. This embodied carbon penalty of oil source requires low carbon production sources, and so would eliminate most shale oil and other synthetic crude from North America, as shown in Mangmeechai (2009).

Producing oil through CO_2-EOR helps to avoid oil transportation emissions, and incorporates extraction emissions into the EOR process. If the 100 MM bbls displaces Saudi (light) oil, there is an estimated additional saving of 4.55 Mt CO_2e from avoided extraction and shipping of Saudi oil (Mangmeechai, 2009). However, it should be noted that the production and most of the transport GHG emissions associated with oil from Saudi Arabia would fall outside of current UK GHG reporting boundaries for the United Nations Framework Convention on Climate Change (UNFCCC) (IPCC et al., 2006) as well as ISO 14064 carbon accounting frameworks (ISO, 2009a, 2009b, 2009c).

The key advantage of the EOR-to-CCS model envisaged in this chapter is that the EOR-oil helps to pay for CCS infrastructure. However, there is the issue of additionality (rather than substitution) of EOR-oil. There is an underlying assumption that introducing new EOR-oil would stop the same amount of Saudi Arabian oil from being produced.

2.4.2 Responsibility of upstream emissions

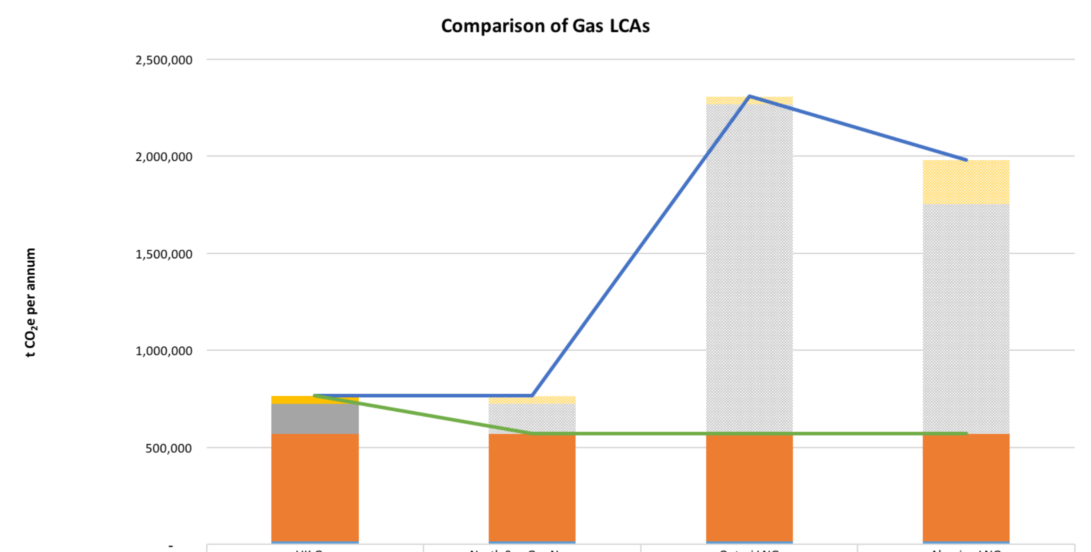
As UK gas production declines over the next decade, the projected gas demand remains constant (DECC, 2015b, 2014b). The 2015 gas mix is modelled along with gas decline under two *fuel-mix* scenarios as discussed above. When UK gas production declines, and other North Sea gas replaces the supply (S2), the responsibility – in terms of current reporting requirements - of gas production-related GHG emissions shifts away from the UK, while actual GHG emissions to the atmosphere stay constant. This is due to the equal LCA estimates of all North Sea gas, but shift in responsibility for reporting of extraction and transportation emissions.

For instance, if declining UK gas production is replaced with imported Qatari LNG, GHG emissions to the atmosphere from this source might increase, yet reported UK GHG emissions would decrease (see Table 2-1 and Figure 2-4). Under current accounting practices, only the combustion emissions are counted towards the grid intensity of electricity supply. Figure 2-5 demonstrates the grid intensity of three pathways to illustrate the differences in GHG emissions accounting compared to life cycle assessments.

When comparing all models and gas sources for CO_2e emissions per unit of power output, EOR from LNG is not as advantageous because of energy associated with liquefying and transporting gas for LNG (Stamford and Azapagic, 2012, 2014). North Sea gas appears the best option for CCS and EOR in this context due to the low production, transportation, and operational emissions. However, ‘UK-owned’ emissions would be lowest using Qatari LNG because the emissions associated with extraction and transportation are regarded as foreign emissions under current GHG accounting regimes (see Figure 2-4 and Figure 2-5).

Table 2-1 - Upstream emissions associated with gas power production for 9 model simulations to 2050. The figures do not include emissions associated with EOR/CCS operations, oil production, end-use of oil or stored CO_2. Emissions which occur in the UK, and impact UK GHG budgets, are lowest when imported gas is used in the fuel-mix. However, atmospheric emissions are highest in Fuel-Mix 3, which has greater dependence on Qatari LNG. This indicates a perverse incentive for UK GHG budgets to import rather than produce gas.

Upstream emissions through 2050 ($Mt\ CO_2e$)		Fuel-Mix 1 *UK Gas*	Fuel-Mix 2 *Norwegian Gas*	Fuel-Mix 3 *Qatari LNG*
1 EOR-to-CCS Project	Emissions to atmosphere	32.03	32.03	44.82
	UK emissions	20.54	18.92	18.92
	UK emissions share of project emissions	64.14%	59.07%	42.20%
4 EOR-to-CCS projects	Emissions to atmosphere	121.91	121.91	172.06
	UK emissions	78.20	71.83	71.83
	UK emissions share of project emissions	64.14%	58.92%	41.74%
11 EOR-to-CCS projects	Emissions to atmosphere	295.47	295.47	423.52
	UK emissions	189.53	173.26	173.26
	UK emissions share of project emissions	64.14%	58.64%	40.91%



	UK Gas	North Sea Gas Norway	Qatari LNG	Algerian LNG
Extraction	40,944	40,944	40,944	227,303
Transportation	154,940	154,940	1,697,003	1,182,942
Operation	555,556	555,556	555,556	555,556
Construction	13,781	13,781	13,781	13,781
Total Emissions	765,221	765,221	2,307,284	1,979,582
UK Emissions Ownership	765,221	569,337	569,337	569,337

Figure 2-4 – Comparison of onshore gas LCAs from Stamford and Azapagic (2014) for UK EOR and CCS. Under current emissions accounting practices, the UK would be responsible for emissions associated with construction and operation of plants in the UK, but not extraction and transportation outside of the UK (partially shaded). Using Qatari LNG would result in the highest GHG emissions to the atmosphere but the UK would have the lowest share of emissions ownership. The current accounting practices incentivizes the UK to pursue the cheapest gas, regardless of total GHG emissions.

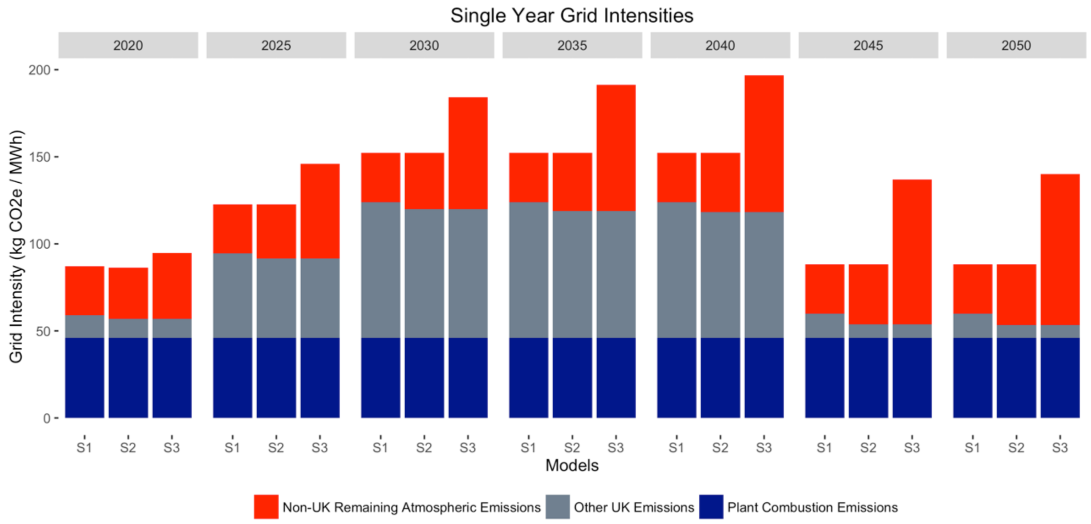


Figure 2-5 - Single year grid intensity for 3 EOR-to-CCS* fuel-mix *simulations. Plant combustion emissions are counted towards grid intensity under current GHG accounting practices. Other UK emissions are also counted towards the UK total emissions, but not counted towards grid intensity. These emissions include plant construction, UK fuel extraction and transportation, & EOR operational emissions. Non-UK Emissions include fuel extraction and transportation occurring outside the UK. Current accounting practices incentivize the UK to minimize domestic emissions (S2 or S3) for carbon budget targets and lower grid intensity.

2.4.3 Other upstream gas sources

DECC's projected demand for natural gas power would likely have other effects not examined in this chapter. Continued demand for natural gas, along with declining North Sea gas production, could incentivize new domestic gas sources, or other sources from abroad.

Stamford & Azapagic (2014) also estimate the GHG emissions of UK shale gas used for electrical power to be 412 – 1102 kg CO_2e MWh^{-1} with a central estimate of 462 kg CO_2e. Using the same methodology as above, utilizing this gas in CCS would correspond to a grid intensity 76.8 – 898.3 kg CO_2e MWh^{-1} (136 kg CO_2e mean), depending on limits to extraction emissions. This is an area in need of further study, as the UK plans to move forward with a shale gas agenda.

Increased UK gas demand could increase demand on the supply entering the NTS from The Netherlands and Belgium. This could increase gas supplied from Russia to the European Continental grid. This gas is of unknown LCA values, and would see an unknown change in atmospheric emissions.

Finally, the UK could also import gas from the USA. This gas would likely be shale gas transported as LNG and contains increased extraction emissions, and LNG processing & transportation emissions.

UK shale gas would be the only option of the above three which would add to the UK GHG emissions total. However, the global GHG footprint for each of these additional options is the subject of further study.

2.5 Conclusions and discussion

The goal of 50 Mt CO_2 stored can be achieved by 2030 with the use of 21.2 GW of gas capacity at baseload utilisation (constant 85% load factor) using coupled CCS-CO_2-EOR. However, this is achieved through the production of 1,100 MM bbls EOR-oil, and immediate development of these modelled projects (see Figure 2-6). Considering all greenhouse gas emissions - from power generation, EOR and CCS operations, and oil production and combustion - this project suite emits an estimated 940 – 1,068 Mt CO_2e from 2020 to 2049, while storing 1,358 Mt CO_2.

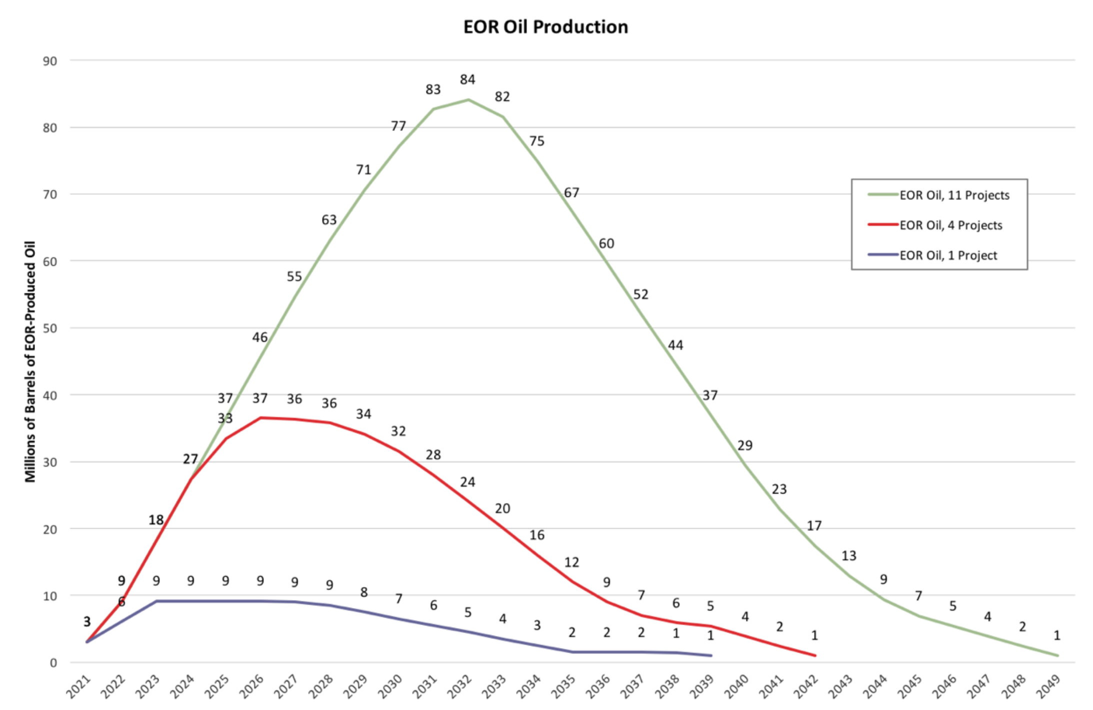


Figure 2-6 - EOR oil production per year from Stewart & Haszeldine (2015). A single EOR project (purple) produces 77.8 million barrels of EOR oil in the first ten years. The final five years produces 6.88 million barrels. A suite of 11 projects produces 405 million barrels in the first ten years.

Additional emission savings could occur if replacement oil imports do not continue after EOR operations cease. Note that there is greater variation emissions associated with EOR oil due to recycling of CO_2 in the production of EOR-oil (Stewart, 2014; Stewart and Haszeldine, 2015). The grid factor could be reduced from 273.9 – 388.1 kg CO_2e MWh^{-1} during EOR to 294.7 – 346.8 kg CO_2e MWh^{-1} during CCS. If oil is excluded from the CCS phase – or is not imported to replace EOR oil – the grid factor reduces again to 90.3 – 142.4 kg CO_2e MWh^{-1} (see Table 2-2). On an annualized basis, these projects emit 3.31 – 4.69 Mt CO_2e yr^{-1} compared to the current BAU case which emits 8.50 – 9.09 Mt CO_2e yr^{-1} (see Table 2-3). In any fuel-mix scenario, the projects modelled above are better than the BAU scenario of combusting unabated natural gas.

These savings could occur if EOR operators are contracted to continue CCS operations for an additional 20 years after the end of EOR operations. It is assumed that the financial incentive of producing 100 MM bbls per project will pay for the operational costs of 20 more years of CCS. In reality, policy action and/or carbon prices will be needed to force continued storage.

The above savings also rely upon decreases in oil use after EOR-oil is no longer produced. If, when EOR transitions to CCS and EOR-oil no longer is produced, oil imports continue, the advantage of CCS is decreased through the additionality of replacement oil. This assumes that the UK will decrease oil usage by 2040, when the first EOR-to-CCS project transitions to CCS. This also assumes that 1,100 MM bbls of EOR-oil from 2020 through 2049 does not exceed demand, and over-supply the UK and global market. Current DECC projections indicate that the demand for oil in the UK will not decrease below 500 MM bbls yr^{-1} before 2035 under any of the 7 modelled policy scenarios (DECC, 2015b). Annual EOR-oil production from 11 EOR-to-CCS projects will peak in year 2032 with 84 MM bbls, well below the 500 MM bbls yr^{-1} projected demand.

In the studied scenarios, it is assumed that deployment of these projects in 2020 in order to reach the goal of 50 Mt CO_2 yr^{-1} stored by 2030, however funding cuts have delayed the development of CCS. Recent funding pledges (BEIS, 2017d; Hickman et al., 2017) indicate a continued commitment to CCS. Considering these delays in combination with project development timelines, and legally binding carbon reduction commitments, the 2030 goal of 50 Mt CO_2 yr^{-1} stored is in jeopardy, which could cost UK industry £30 billion per year after 2050 (Durusut et al., 2015).

The key advantage of the EOR-to-CCS model envisaged in this chapter is that the EOR-oil helps to pay for CCS infrastructure to have long-term GHG savings. However, if produced EOR oil is not sufficient to cover costs for infrastructure, or there is disruption in the 40-year model, the advantage of CO_2 storage during the CCS phase could be lost. In this instance, the UK GHG budgets would experience the emissions associated with these projects without enjoying the GHG savings during the CCS phase. This could occur if the global oil prices are too volatile for dependable project funding, reputational risks depress the price of oil and prevent the UK or operational partner(s) from funding or proceeding with these projects, or public opinion will not tolerate a limited oil consumption to allow for more CO_2 storage.

Finally, it is assumed that the reduction of 1,100 MM bbls imported oil will aid in lowering UK GHG emissions, but also assume that the previously imported oil is no longer produced. The use of EOR-oil introduces new oil into the global system. If the previously imported oil is still produced, and EOR-oil is used, then there may be a small increase in oil-based GHG emissions globally. The 1,100 MM bbls of oil, additional or not, are trivial in comparison to the reduction realised in achieving CCS through the use of EOR. On a full life cycle analysis, with the unusual aspect of lining across the economy from electricity to oil, this aggressive CCS-EOR scenario provides a net carbon reduction. More carbon is stored, sooner, for less public funding, than any rival method.

Table 2-2 - Annual project emissions as grid factors for one EOR-to-CCS project compared to a BAU scenario. In all fuel-mix scenarios, the EOR-to-CCS project produces fewer emissions that the BAU scenario of the same mix. The BAU scenario compares the same power plant as in the EOR-to-CCS scenario, but delivers more power to the grid because no CO_2 is captured for EOR/CCS.

Grid emissions per project (kg CO_2e MWh^{-1})	Fuel-Mix 1 *UK gas*	Fuel-Mix 2 *Norwegian gas*	Fuel-Mix 3 *Qatari LNG*
EOR phase	247.5 – 254.5	247.5 – 254.5	257.9 – 291.3
including EOR oil	273.9 – 344.4	273.9 – 344.4	286.3 – 388.1
CCS Phase	90.3 – 98.6	90.3 – 98.6	102.7 – 142.4
including imported Saudi oil	294.7 – 303.0	294.7 – 303.0	307.0 – 346.8
BAU Emissions (no EOR/CCS) *including* imported Saudi oil	591.1 – 595.0	591.1 – 595.0	601.5 – 631.7
BAU emissions (no EOR/CCS) *excluding* oil emissions	419.5 – 423.3	419.5 – 423.3	429.9 – 460.1
BAU emissions (no EOR/CCS) *excluding* oil emissions, UK emissions only	393.7	388.6 – 394.8	377.6 – 408.8

Table 2-3 - Annual project emissions. In all fuel-mix scenarios, the EOR-to-CCS project produces fewer emissions that the BAU scenario of the same mix. The BAU scenario compares the same power plant as in the EOR-to-CCS scenario, but delivers 2.4 MWh more power to the grid because no CO_2 is captured for EOR/CCS.

Annual emissions per project (Mt CO_2e)	Fuel-Mix 1 *UK gas*	Fuel-Mix 2 *Norwegian gas*	Fuel-Mix 3 *Qatari LNG*
Upstream power plant with CO_2 capture	1.03 – 1.09	1.03– 1.09	1.18 - 1.62
EOR/CCS operations	0.03 – 0.83	0.03 – 0.83	0.03 – 0.83
CO_2 stored	(4.62 – 4.95)	(4.62 – 4.95)	(4.62 – 4.95)
Embedded oil emissions	2.24 – 2.47	2.24 – 2.47	2.24 – 2.47
Total annual project emissions	3.31 – 4.16	3.31 – 4.16	3.46 – 4.69
BAU power emissions	6.03 – 6.09	6.03 – 6.09	6.18 - 6.62
BAU oil emissions	2.47	2.47	2.47
BAU total emissions	8.50 – 8.56	8.50 – 8.56	8.65 – 9.09

Chapter 3 Gas-fired power in the UK: Bridging supply gaps and implications of domestic shale gas exploitation for UK climate change targets

Summary

The UK is legally bound to limit domestic GHG emissions, and reduce emissions associated with electricity generation. Currently, there is a projected shortcoming in the UK's fourth carbon budget of 7.5%, while projected demand for fossil fuels remains steady. This shortfall may be increased if the UK pursues a domestic shale gas industry to offset projected decreases in traditional gas supply. This chapter estimates that if the projected domestic gas supply gap for power generation were to be met by UK shale gas with low fugitive emissions (0.08%), an additional 20.4 Mt CO_2e would need to be accommodated during carbon budget periods 3 – 6. More importantly, it is found that a modest fugitive emissions rate (1%) for UK shale gas would increase global emissions compared to importing an equal quantity of Qatari liquefied natural gas. Additionally, it is estimated that natural gas electricity generation would emit 420 – 466 Mt CO_2e (460 central estimate) during the same time period within the traded EU emissions cap. It is concluded that domestic shale gas production with even a modest 1% fugitive emissions rate would risk exceedance of UK carbon budgets. Additionally, under the current production-based greenhouse gas accounting system, the UK is incentivized to import natural gas rather than produce it domestically.

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3.1 Introduction

The United Kingdom (UK) is legally bound by ‘The Climate Change Act’ to reduce greenhouse gas (GHG) emissions by 2050 to 80% below a 1990 baseline (Energy Act 2004). The Climate Change Act also empowers an independent body, The Committee on Climate Change (CCC), to advise the Government on progress towards the 2050 goal. The CCC recommends reduction targets in 5-year budget periods. The UK met its first carbon budget (CB1: 2008 – 2012) of 3,018 Mt CO_2e, and is likely to meet the second and third carbon budgets, covering the years 2013 – 2022 (DECC, 2015b, 2014b). However, the Department for Energy and Climate Change (DECC - now the Department for Business, Energy, and Industrial Strategy - BEIS) suggests that the UK may fail to meet its fourth carbon budget (CB4: 2023 – 2027) by as much as 146 Mt CO_2e (BEIS, 2017e; DECC, 2015b, 2014b). The DECC projections indicate a 7.5% overshoot in CB4, with an uncertainty range of 6-13%. This projected overshoot (DECC 2015a) increases into the fifth carbon budget (CB5: 2028-2032), which the UK is at risk of exceeding by more than 14% (BEIS, 2017e).

Major Power Producers (MPPs), companies whose primary activity is electricity generation (DECC, 2012), produce 94% of all electricity in the UK. In 2015 over half of this production was from the combustion of coal (120 Terawatt Hours - TWh) and natural gas (71 TWh) (DECC, 2015a). GHG emissions from MPPs are traded within the European Union Emissions Trading Scheme (EU ETS), and are capped by UK emissions allowances granted by EU ETS. These emissions do not affect the UK’s ability to meet carbon budgets directly, however, the continued use of fossil fuels by MPPs maintains a domestic demand for fossil fuels, which can create emissions in other non-traded sectors of the economy (e.g. coal and gas extraction). UK electricity production is projected to shift away from coal generation towards gas power, to supplement growing renewables and nuclear capacity (BEIS, 2017e, 2016a; Directive 2010/75/EU of the European Parliament and of the Council of 24 November 2010 on industrial emissions (integrated pollution prevention and control); Energy Act 2013). Yet, domestic production of natural gas will continue to fall (OGA, 2016). If the UK shifts the source of this gas from domestic North Sea gas to imported liquid natural gas (LNG), associated UK gas production emissions will fall while production-phase emissions elsewhere in the globe would increase. Conversely, a domestic shale gas industry could provide increased energy security for the UK, but would create a new source of domestic industrial emissions that is not yet fully accounted for within planned UK carbon budgets.

The CCC has stated that a UK shale gas industry is not compatible with UK climate change targets unless three test criteria are met: (1) production emissions are strictly limited; (2) UK gas consumption declines, remains in line with carbon budgets, and displaces imports; (3) production emissions are offset elsewhere in UK carbon budgets (CCC, 2016a, 2016b). The first test can be met by strict regulation of practices associated with shale gas production, many of which are agreed upon by industry (UKOOG, 2015). The second test may be met by maintaining or lowering UK gas consumption, while measuring and prioritizing the lowest carbon footprint gas to be consumed in the UK. The third test will require national coordination of sectoral GHG emissions to accommodate any additional emissions associated with domestic shale gas production.

In addition to the shale gas criteria, the CCC recommended a grid electricity emissions target of 50 g CO_2e kWh^{-1} by 2030 in the fourth carbon budget report (CCC, 2010), but have since relaxed that goal to below 100 g CO_2e kWh^{-1} by 2030 (CCC, 2015b). The CCC projects this goal will be met by a combination of renewables, natural gas, carbon capture and storage (CCS), and nuclear power. DECC / BEIS has projected several scenarios which illustrate pathways to a 100 g CO_2e kWh^{-1} target by 2030. This chapter aims to quantify the emissions associated with gas consumption in the UK for electricity generation under scenarios projected by DECC / BEIS, and identify ‘CCC-test’ limits on emissions for a domestic shale gas industry.

3.1.1 DECC & BEIS projections

DECC / BEIS produce annual projections for *Updated energy and emissions scenarios* (BEIS, 2017e; DECC, 2015b, 2014b) which incorporate GHG-reduction policies, fossil fuel prices, and economic growth projections. The *Reference Scenario* is based on central estimates of economic growth and fossil fuel prices - it is therefore treated as the central estimate in this study. It contains all agreed-upon policies and planned policies. DECC's *Low Growth, High Growth, Low Prices,* and *High Prices* projections assume the same policies as the *Reference Scenario* but incorporate variance on fossil fuel prices and economic growth. Their *Existing Policies* projection contains central estimates, but excludes planned policies; it is an assessment of the current state of policies projected forward. Finally, the DECC *Baseline Policies* projection contains only policies that existed before the Low Carbon Transition Plan of 2009 (Great Britain and HM Government, 2009), and is therefore excluded from this analysis.

3.1.2 Gas demand and production

In 2016, UK gas demand was 67.0 - 72.0 billion cubic meters (bcm) (BEIS, 2017b; OGA, 2017), which was supplied by domestic and imported sources. Total domestic production, before exports, was 37.0 bcm. Imported gas (30.9 bcm) came from Norway, Belgium, and The Netherlands via interconnections. An additional (13.9 bcm) of liquefied natural gas (LNG) was supplied from Qatar (12.9 bcm), Algeria (0.5 bcm), Trinidad & Tobago (0.5 bcm), with negligible amounts from Nigeria (44 mcm), and Norway (55 mcm). Exports through interconnection were 14.2 bcm with an additional 276 mcm exported as LNG. After subtracting exported gas, UK used around 27% of gas in electricity generation or 19.2 bcm. Of this, 16.9 bcm were used by MPPs, the remaining gas being used in autogenerators (BEIS, 2016a, 2016b; OGA, 2017).

The Oil and Gas Authority (OGA) forecasts that domestic oil and gas production will decline by 5% yr^{-1} from 2022 until 2035, with gas production dropping from 34.7 bcm in 2018 to 14.5 bcm in 2035. Demand across all UK sectors is projected to decline more slowly - from 74.1 bcm in 2018 to 61.5 bcm in 2035 - despite impetus from carbon budgets and further growth in renewable energy supply. Without a shale gas industry, falling domestic production could increase the UK's domestic supply gap to more than 796 bcm (OGA, 2017) (see Figure 3-1). This would require imports to increase from 53% of demand in 2018 to an estimated 76% of demand in 2035.

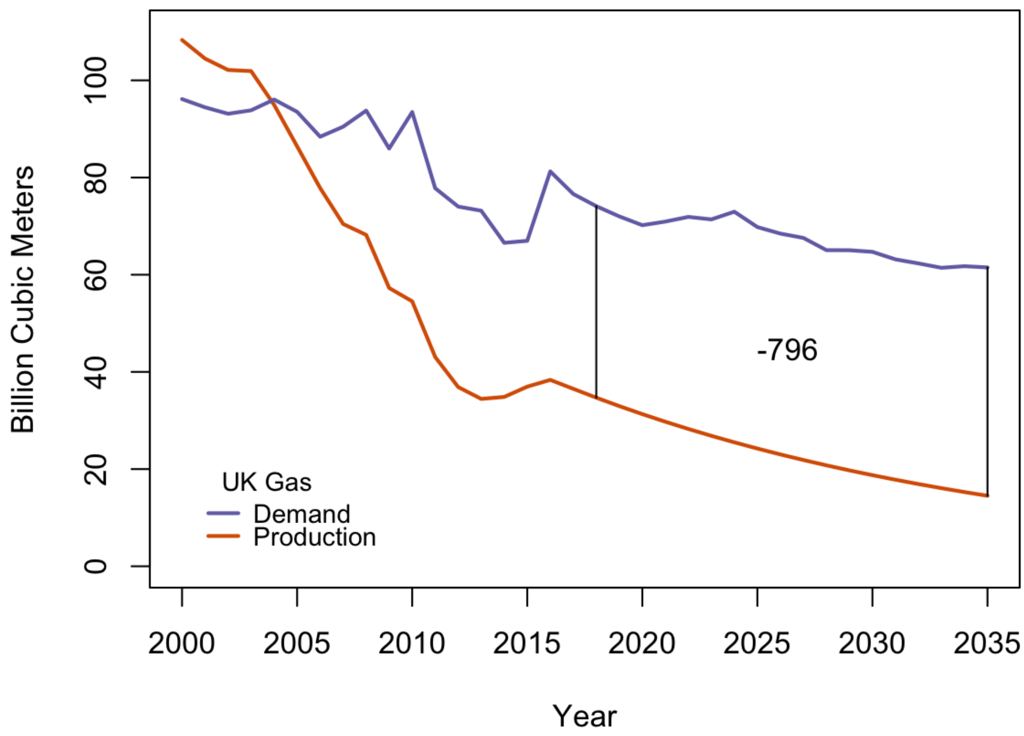

Figure 3-1 UK natural gas production and demand for all sectors projected to 2035. Declining domestic production will create an import demand of 796 billion cubic meters from 2018 to 2035. Figure generated from OGA projections (OGA, 2017).

3.1.2.1 UK shale gas resource estimates

The potential of the UK shale gas resource to fill the projected domestic gas supply gap is an area of substantial policy and commercial interest (Bradshaw et al., 2014). Advanced Resources International (ARI) estimate that the UK holds 3,783 bcm (133.4 trillion cubic feet, tcf) of *technically recoverable*– gas which may be accessed given geological knowledge, economic feasibility, and production history - from a larger 17,641 bcm (623 tcf) of total shale gas in place. Of the gas *in-place*, 728 bcm (25.7 tcf) is estimated to be economically recoverable (Kuuskraa et al., 2013). This corresponds to a 19.2% recovery rate of *technically recoverable resources*, similar to the mean estimates of around 20% used by Kuuskraa et al. (2013, 2011).

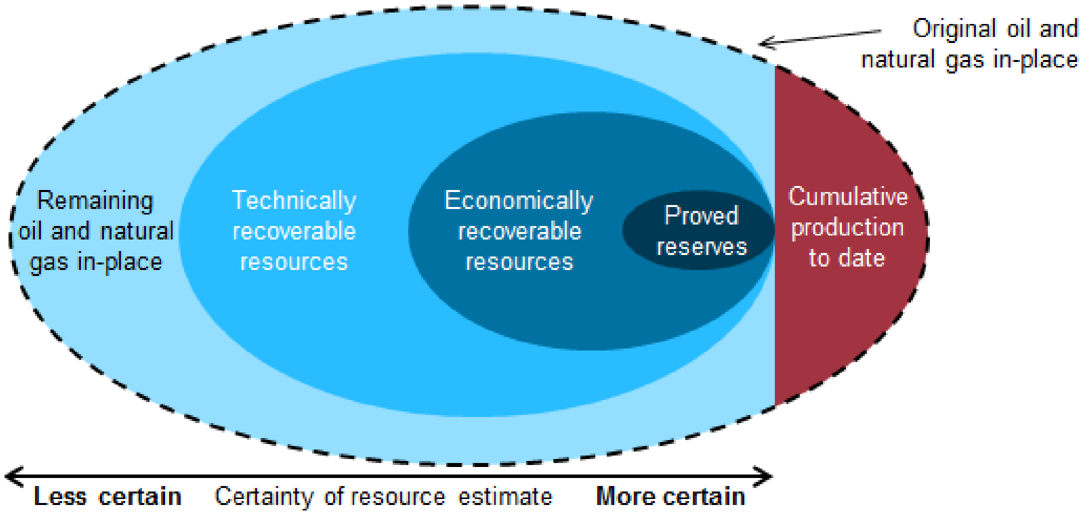

Figure 3-2 -Stylized representation of oil and natural gas resource categorizations. As geologic understanding, technical capabilities, and economic conditions allow, shale gas reserves can be designated from 'in-place' (identified) to 'proved reserves'. Schematic adapted from EIA (2014)

The British Geological Survey (BGS) estimate that the UK has a total of 39,900 bcm of shale gas in place, with a range of 24,700 – 68,400 bcm (Andrews, 2013; Monaghan, 2014). Though considerably more than the ARI estimate, BGS surveyed three basins (Carboniferous Midland Valley, Carboniferous Bowland-Hodder, Jurassic Weald) while the ARI estimate covers all Carboniferous shale basins grouped together along with the Jurassic Weald basin (it is likely that these estimates will be further refined as Cuadrilla begins shale gas exploration in Lancashire (Cuadrilla Resources, 2016).

BGS do not provide estimates of *technically recoverable resources*, or *economically recoverable resources*. If it is assumed that the ARI ratio of risked gas-in-place to total resource estimate of 21.4% is also applicable to the BGS resource estimates, the range of risked gas-in-place across the three BGS-surveyed basins is 5,297 – 14,668 bcm. Based on industry experiences in Poland, applying a conservative economic recovery rate of 10% (Inman, 2016; Kuuskraa et al., 2013) to these low-end gas *in-place* estimates (ARI and BGS-derived) therefore corresponds to 378 – 530 bcm of *economically recoverable* domestic shale gas. This compares to the UK's projected 796 bcm gas supply gap over the next two decades.

3.2 Methodology

This study examines two specific GHG emission scopes: 'grid emissions from natural gas power' and 'associated gas production emissions'. Here, 'grid emissions' are defined as the direct GHG emissions arising from combustion of fossil fuels for electricity generation divided by total electricity delivered. These GHG emissions are traded within the EU ETS, and count towards the grid emissions goal of 100 g CO_2e kWh^{-1} by 2030. 'Associated gas production GHG emissions' here include power plant construction, fuel extraction and processing, and fuel transportation.

When 'associated gas production emissions' occur within the UK, they are counted towards non-traded carbon budgets. 'Associated gas production emissions' occurring outside the UK do not affect carbon budgets, and would represent a saving to the UK carbon budgets, but break CCC's third test criteria (CCC, 2016a, 2016b). These emissions are also counted in a 'non-UK' category.

The sum of 'grid emissions' and all 'associated gas production emissions' provides a fuller assessment of the GHG impact of each fuel source used for UK electricity supply. The GHG footprint estimates from Stamford and Azapagic (2014) are used to calculate the impact of DECC / BEIS projections of future gas electricity against CCC's test criteria for UK shale gas exploitation and the grid emissions goal (100 g CO_2e kWh^{-1} by 2030). The Stamford and Azapagic (2014) study is used for all phase emissions for natural gas production and combustion for electricity generation. Other gas studies were examined but either did not compare geographic origins (Heath et al., 2014a; Laurenzi and Jersey, 2013; Dale et al., 2013) or embedded phase emissions (Bouman et al., 2015) which bundles the production, transportation, and combustion emissions.

'Associated gas production emissions' by UK carbon budget period for those occurring within the UK are also categorized. 'Associated gas production emissions' occurring outside the UK are also tallied based on origin of gas supply within each model (see 3.2.2 below).

3.2.1 Gas supply from NTS

Here it is assumed that all gas demand for electricity generation in the UK is supplied via the UK National Transmission System (NTS). Natural gas for combustion at each power plant is assumed to come from the NTS, combustion phase emissions are therefore assumed to be equal. When gas arrives in the UK via interconnection or LNG terminal, it enters the NTS making the geographic origin unknown to the end user. Gas flows between the UK and continental Europe are excluded from the calculations because they account for less than 1% of total consumption, fluctuate rapidly with market demand, and lack GHG footprint figures because of the unquantified sources of European continental gas (BEIS, 2017e; Stamford and Azapagic, 2014). In the absence of nation-specific data, LNG from Nigeria and Trinidad & Tobago are here assumed to have the same production phase emissions as LNG from Algeria. The imports from each of these countries represent less than 1% of gas supply, but are reported in annual UK gas supply figures (BEIS, 2016b). Finally, imported North Sea gas is assumed to have the same production phase emissions as UK domestic gas.

The projected gas demand from DECC / BEIS, and OGA (BEIS, 2017e; OGA, 2017) is used. Dependency on imports is assumed equal to OGA estimates of domestic demand minus domestic production. There are statistical differences between the net sum of all imports minus exports, and the import dependency. This is due to the constraints of monthly, quarterly, and annual measurements and reporting. It is assumed that the net import/exports are zero over the life of the calculations, and are rounded to zero by adjustments in pipeline imports. Zero bunkering of gas is assumed.

Upstream gas extraction emissions count towards the UK carbon budget, when occurring within the UK. Emissions from the transportation of fuel (natural gas) are reported in the *Mobile Combustion Process* category within the energy sector (IPCC, 2000), are pre-combustion, and so are not counted towards grid emissions. These emissions are counted under 'associated gas production emissions'. Furthermore, shipping of natural gas products as liquid natural gas (LNG) contains transportation emissions which are tallied in the carbon budget of the ship's home country.

The emissions intensity of imported gas represents the current estimates of global supply. Further expansion of fracking in nations from which gas is imported to the UK could further increase overseas emissions while leaving reported emission in the UK unaffected. To determine the potential impact of a shale gas industry versus increased gas imports on UK carbon budgets, four gas supply scenarios are modelled:

3.2.1.1 Scenario 1 – current gas supply projected forward

Here total UK gas demand fluctuates according to DECC / BEIS and OGA estimates from 2018 to 2035, while UK production declines -5% yr^{-1} from 2022 onwards (OGA, 2017). The 2016 global supply sources of natural gas to the UK (BEIS, 2017b) are assumed to remain the same through to 2035. The growing UK gas supply gap is assumed to be met by Norwegian North Sea imports, Qatari LNG, and other LNG imports. These three gas sources increase proportionately from 2016 onwards to meet projected demand, with a presumption that shale gas is not produced in the UK. This scenario represents the baseline emissions scenario.

3.2.1.2 Scenario 2 – increased gas from Norway

Here total UK production falls as projected by OGA estimates, as in scenario 1. As UK gas declines in production from 2022 onwards, North Sea gas supply from Norway increases to meet UK demand. Qatari LNG and other LNG imports continue in the same quantity as 2016 (10.1 bcm, and 0.91 bcm respectively). Again, this scenario assumes that shale gas is not produced in the UK. This scenario demonstrates shifting of associated gas production emissions from the UK to Norway.

3.2.1.3 Scenario 3 – increased LNGs from Qatar

Here UK gas demand fluctuates as in scenarios 1 & 2, UK gas production again declines as projected by OGA estimates. North Sea gas from Norway, and other LNG imports, continue to be imported in the same quantity as 2016 (19.6 bcm and 0.91 bcm respectively) and shale gas is not produced in the UK. Instead, Qatari LNG imports increase to meet the growing supply gap. This scenario demonstrates of shifting responsibility of associated gas production emissions from the UK, and places more emphasis on LNG from Qatar.

3.2.1.4 Scenario 4 – shale gas from UK

In this final scenario, UK gas demand fluctuates as in scenarios 1, 2, & 3, UK gas production declines falls as projected by OGA estimates. Imports from Norway, Qatar and other LNG are imported in the same quantities as in 2016 (19.6 bcm, 10.1 bcm, and 0.91 bcm respectively). Here, UK shale gas production is assumed to expand to meet the supply gap starting in 2018 with 8.9 bcm production. Shale gas reaches 27% of supply by 2035 with 16.4 bcm produced, totalling 246 bcm over the model period. This scenario illustrates the potential impact of UK shale gas exploitation on carbon budget periods due to increased domestic ‘associated gas production’ emissions.

3.2.2 Electricity source emissions and capacity

As discussed above, within the GHG footprint of electricity generation are combustion emissions and GHG emissions associated with power plant construction, fuel extraction, materials manufacturing, and decommissioning (Stamford and Azapagic, 2014). Combustion emissions divided by electricity delivery are counted toward the grid decarbonisation goal of 100 g CO_2e kWh^{-1}. Other 'associated emissions' count as non-traded industrial emissions and affect carbon budgets (CCC, 2016a, 2016b; IPCC, 2000). For the following electricity fuel sources, the 'combustion emissions' are multiplied by the DECC / BEIS projections for delivered electricity to measure the progress towards the grid emissions goal of 100 g CO_2e kWh^{-1}. 'Associated emissions' are emissions associated with the production and supply of gas that is then used in UK power generation.

3.2.2.1 Fossil electricity sources

Future GHG emissions from oil combustion are not considered in this analysis as oil has been phased out as a major grid electricity source in the UK since 2013, and generated just 0.60 TWh in 2015 (BEIS, 2017e). When incorporating past emissions, electricity from oil is assumed to have a 'combustion emissions' intensity of 750 g CO_2e kWh^{-1} (Weisser, 2007).

It is assumed that coal power in the UK will be phased out in the next decade (Directive 2010/75/EU of the European Parliament and of the Council of 24 November 2010 on industrial emissions (integrated pollution prevention and control)), but will play some role in UK power until 2026 particularly in the BEIS *High Prices Scenario* (BEIS, 2017e). It is assumed that coal has 'combustion emissions' factor of 941 g CO_2e kWh^{-1} (Stamford and Azapagic, 2014).

By 2035, it is projected that 25.6 GW of new gas capacity will have been built by MPPs. The total capacity will be 34.2 GW in 2035. Natural gas is expected to deliver 20.2 – 38.3 TWh in 2035, with 27.9 TWh as the central estimate. It is assumed that natural gas power in the UK has a 'combustion emissions' of 386 g CO_2e kWh^{-1} based on Stamford and Azapagic (2014). This is in agreement with the range of 365 – 415 g CO_2e kWh^{-1} annually reported by BEIS / DECC (BEIS, 2016a; DECC, 2015a, 2014a, 2013a, 2012). Likewise, GHG intensity figures from Stamford and Azapagic (2014) of 401 – 508 g CO_2e kWh^{-1} are assumed, including the shale gas central estimate of 462 g CO_2e kWh^{-1} (see Table 3-1). Variance in geographic gas supply changes the

GHG intensity figures for gas in each of the four fuel mix scenarios as described above.

Table 3-1 – GHG phase emissions of UK natural gas combined cycle power from Stamford and Azapagic (2014). All combustion emissions count towards the gird emissions target of 100 g CO_2e kWh^{-1}. The remaining 'associated emissions,' including plant construction, extraction and processing, and fuel transportation count UK carbon budgets when occuring within the UK. Stamford and Azapagic assume a small (0.08%) fugitive emissions rate for UK shale gas production during the extraction and processing phase.

Phase emissions (g CO_2e kWh^{-1})	UK North Sea gas	Norwegian North Sea gas	LNG Algeria	LNG Qatar	UK shale gas (central)
Plant construction	0.958	0.958	0.958	0.958	0.958
Extraction and processing	2.8	2.8	15.8	2.8	65.9
Fuel transport	10.8	10.8	82.2	118.0	8.9
Combustion	386.3	386.3	386.3	386.3	386.3
Total	400.8	400.8	485.3	508.1	462.0

3.2.2.2 Low GHG electricity sources

This chapter assumes 16.0 GW of new nuclear capacity by 2035, part of a projected 17.2 GW of total nuclear capacity producing 135 TWh or 38% of UK demand by 2035 based on BEIS projections (2017e). It is assumed the 'combustion emissions' for nuclear power are zero (Sovacool, 2008).

The installed UK renewable energy capacity projected by BEIS / DECC does not specify what percentage will be solar, wind, hydro, or biomass. To simplify, it is assumed that 174 – 177 TWh of renewable electricity will be generated from 63 GW of wind and solar capacity in 2035 based on BEIS projections (2017e). Previously, DECC (2015b) projected 152 TWh with a range of 142.8 – 159.0 TWh in the central scenarios. In 2016, solar and wind generated 57.8% of renewable electricity (12.4% and 45.3% respectively) up from 37% in 2009 (BEIS, 2017c). It is assumed that the 'combustion emissions' for these renewables are zero (Nugent and Sovacool, 2014). Biomass contributed 22% of renewable electricity in 2016 (BEIS, 2017c). However, there is very high variability in LCA estimates for biomass energy GHG intensity, including potentially negative emissions (Stephenson and MacKay, 2014). Therefore, biomass energy generation estimates are excluded from the scenarios and instead it is assumed that all UK renewable electricity is represented by an even split between solar and wind power.

3.2.3 Electricity generation GHG emissions

To examine the impact of a potential UK shale gas supply versus increased gas imports on the UK electricity intensity target of 100 g CO_2e kWh^{-1} (and UK carbon budgets), the BEIS / DECC projections (see 3.1.1 above) models were evaluated and annual grid intensity calculated. This assessment does not include microgeneration, systems with less than 50 kilowatts electricity generation capacity (Energy Act 2004), and instead focuses on major power producers from five main generation types (Renewables, Coal, Oil, Natural Gas, and Nuclear).

BEIS / DECC projects 7 versions of electricity emissions through to 2035, their *Reference Scenario* is here treated as the central estimate. The projected electricity generation type (TWh) is multiplied by the corresponding 'combustion emissions'. The annual sums of the electricity generation emissions are divided by the total electricity delivered in each model for that year. This gives the grid factor for each of the 4 fuel mix scenarios from the range of generation projected by BEIS / DECC through 2035.

The associated non-combustion emissions for natural gas power is separated from the combustion emissions. These emissions are categorized into carbon budget periods for UK emissions and non-UK emissions, to show the additional impact of domestic shale gas (scenario 4) on carbon budgets.

3.3 Results and discussion

3.3.1 Gas combustion emissions

In the calculations, variance in grid emissions, and progress towards 100 g CO_2e kWh^{-1}, is driven by the DECC / BEIS projections (2017e; 2015b, 2014b) for total gas-derived electricity. The geographic origin of the gas inputs to the NTS affects only the non-combustion associated emissions. For all 4 UK gas supply scenarios examined here, it is assumed that equal gas 'combustion emissions' (Stamford and Azapagic, 2014).

For the central BEIS projections, using the Stamford and Azapagic (2014) estimates of natural gas combustion intensity, 420 – 466 Mt CO_2e (460) will be emitted from natural gas power in the UK from 2018 – 2035, compared to 280 – 397 Mt CO_2e (397) in the 2015 projections (see Figure 3-3). The narrow range in the 2016 projections is due to increased proportions of renewables in the 2016 projections by BEIS (2017e). Natural gas for electricity generation is expected to be increasingly substituted by renewable power, yet will still be responsible for more than 460 Mt CO_2e over the next 2 decades from combustion alone. These emissions will require permits for trading through the EU ETS during carbon budget periods 3 - 6.

The use of UK shale gas does not affect the combustion emissions, as they are a function of electricity demand and total gas usage. Multiplying the DECC / BEIS projections (2017e; 2015b, 2014b) by grid intensities for electricity type suggests that the 100 g CO_2e kWh^{-1} goal will be reached by 2028 in the *Reference Scenario* for both 2015 and 2016 projections. However, this goal could be jeopardized by lack of progress on establishing more ambitious UK GHG reduction policy, illustrated by the *Existing Policies Scenario*. In this scenario, the grid intensity goal is not reached by 2035 (see Figure 3-4).

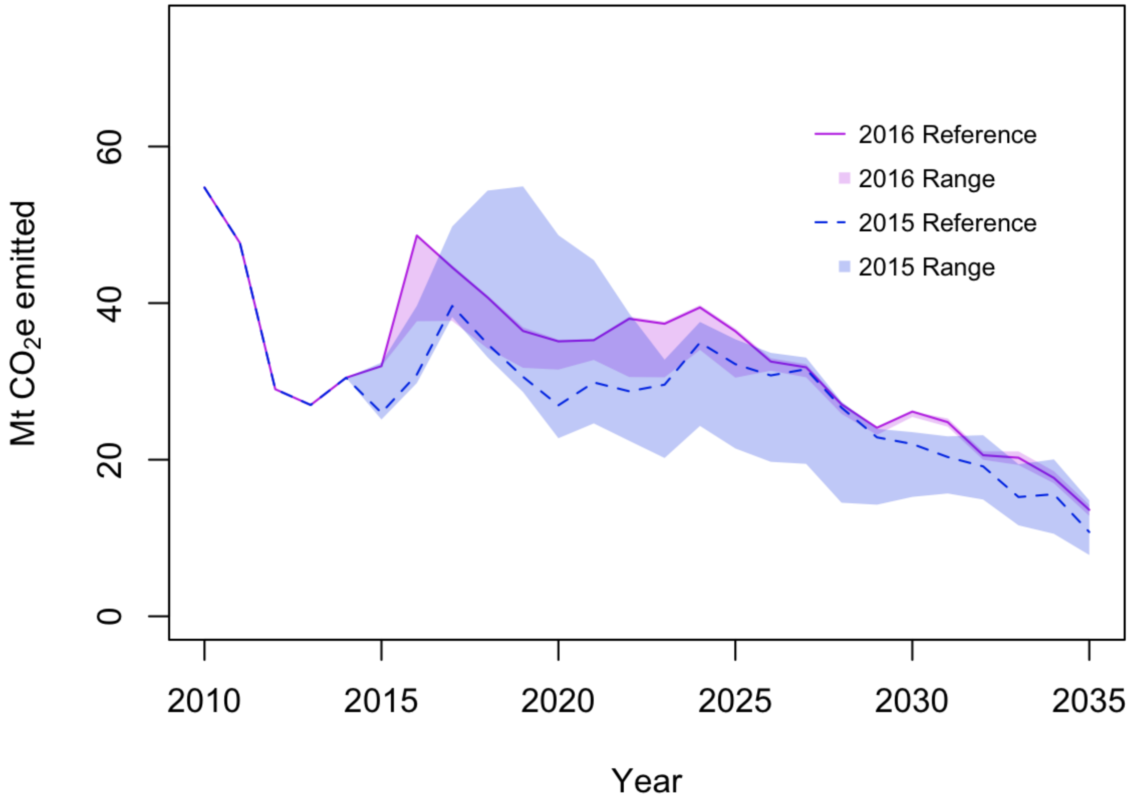

Figure 3-3 - CO_2e emissions from natural gas combustion for electricity supply in 2015 and 2016 BEIS / DECC projections (BEIS, 2017e; DECC, 2015b). Despite increased renewable capacity, the UK is projected to be reliant on natural gas power for electricity supply in all BEIS / DECC projections. It is assumed that combustion of gas is 386 g CO_2e kWh^{-1} (Stamford and Azapagic, 2014), and when multiplied by 2016 BEIS / DECC projections, the UK will emit 420 – 466 Mt CO_2e from 2018-2035 from natural gas electricity generation.

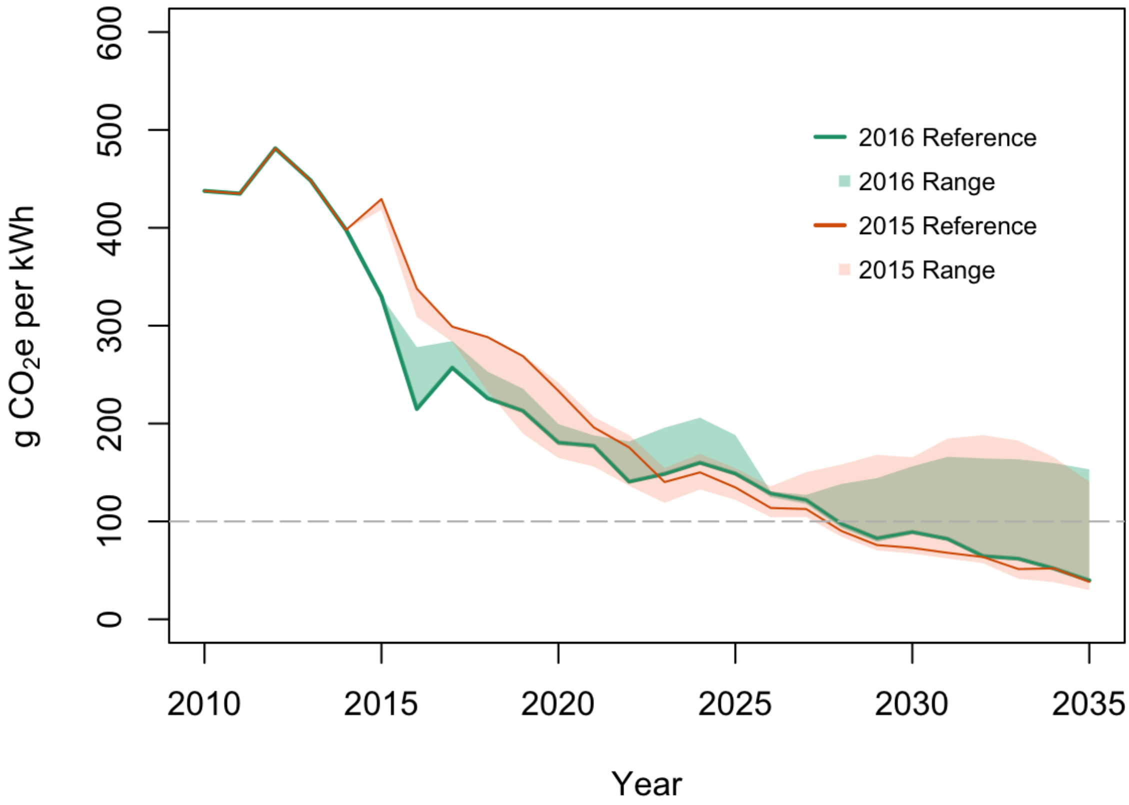

Figure 3-4 - Projections of progress toward UK grid intensity target of 100 g CO_2e kWh^{-1}. Current DECC / BEIS projections (BEIS, 2017e; DECC, 2015b) indicate the grid intensity target will be reached by 2028 in the* Reference Scenario. *However, the target may be in jeopardy if no new climate policies are enacted. The top of the range for 2015 & 2016 indicate that without further progress on climate legislation, the grid emissions target will not be met.

3.3.2 Associated gas production emissions

The associated gas production emissions are separated by carbon budget periods (CB3 – CB6) for UK emissions. Non-UK emissions are those from fuel production and transportation outside of the UK. In scenarios 1, 2, and 3, UK emissions are similar across all carbon budget periods (see Figure 3-5). In scenario 4 (domestic shale gas), the UK would emit 28.5 Mt CO_2e (25.8 – 28.8 range) from 2018-2035 (as reported under current guidelines from The United Nations Framework Convention on Climate Change - UNFCCC) with an additional 32.0 Mt CO_2e (29.0 – 32.4 range) being emitted outside of the UK. If UK shale gas was not produced and the UK relied more on Qatari LNG (scenario 3), the share of UK-associated emissions (as reported under UNFCCC guidelines) would reduce to 8.1 Mt CO_2e (7.2 – 8.1), but overall emissions to the atmosphere would actually increase to 64.9 Mt CO_2e (59.0 – 65.8), an 11.5 Mt CO_2e increase over scenario 4. These sums indicate that the shale scenario (4) globally saves 12.5 Mt CO_2e over the Qatari LNG scenario (3), but shifts emissions to the UK in terms of reporting requirements. Scenario 4, globally, leads to higher emissions than scenarios 1 and 2 (24.4 Mt CO_2e higher), and increases UK-reported emissions by 20.4 Mt CO_2e during carbon budget periods 3-6. This 20.4 Mt CO_2e of production-phase emissions is in addition to the 460 Mt CO_2e emitted due to combustion at UK gas power plants over the same period (see Figure 3-5).

3.3.3 Peaking gas power and renewable support

Stamford and Azapagic (2014) assumed 50% load factor for natural gas plants in their calculations based on the Ecoinvent database (Ecoinvent Centre, 2010; Weidema and Hischier, 2006). This chapter does not adjust for lower load factors (power generation divided by capacity) in the future scenarios. BEIS data (BEIS, 2017e) project a gas power load factor below 30% in 2028 and beyond, which are less efficient. Gas plants are more efficient when used for baseload power, assumed to be 85% in this thesis. If the UK commits to natural gas power for renewable support, higher load factors would be advantageous. A gas plant used at baseload power would emit more CO_2e per plant, but fewer plants would need to be built, and gas demand could be reduced and planned. This paradox between gas capacity and renewable support disagrees with the motivation for 100% renewable power, but would save construction costs and GHGs. This also assumes that storage and interconnection will not be able to provide 100% backup for renewable electricity.

If the UK imports power from continental Europe, there is an issue of GHG-intensity of this imported power. The 2016 *Updated Energy and Emissions*

Compatibility of fossil fuel energy system in the UK for climate targets. *Projections* (BEIS, 2017e) included storage in their calculations for electricity delivery the first time. The report also increased the supply of electricity from interconnection. These figures are outside of the boundaries of this study, but indicate a projected reliance on the EU continental grid for electricity supply.

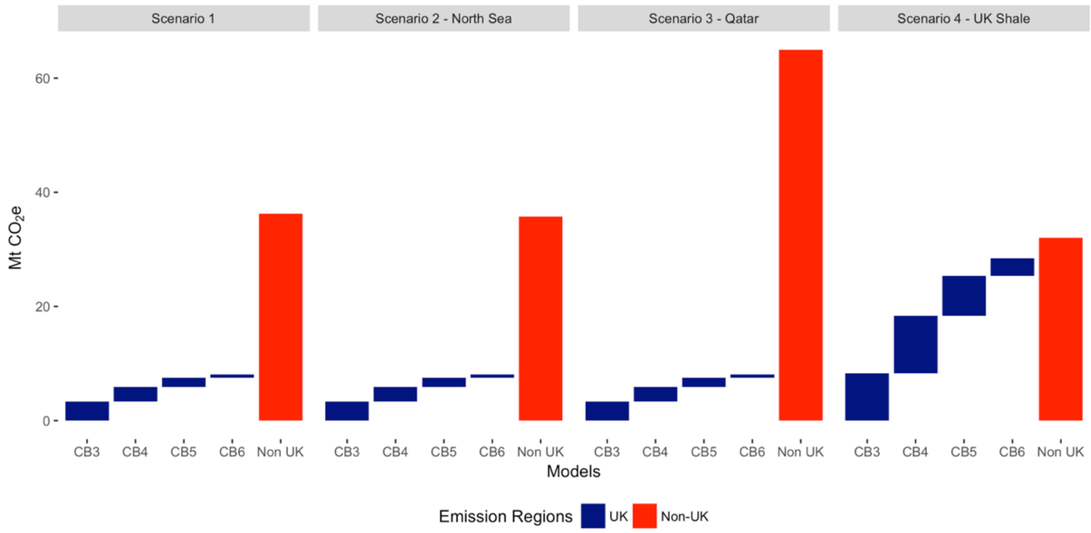

Figure 3-5 – Associated natural gas power production emissions by carbon budget period for 4 models in the central 2016 BEIS projection (BEIS, 2017e). Non-UK and total emissions are highest when using Qatari LNG. UK emissions are highest in scenario 4, when exploiting a domestic shale gas resource. The UK would be responsible for 28.5 These sums indicate that the shale scenario (4) globally saves 12.5 Mt CO_2e over the Qatari LNG scenario (3), but shifts emissions to the UK. Scenario 4 is globally worse than the scenarios 1 and 2 by 24.4 Mt CO_2e, and increases UK emissions by 20.4 Mt CO_2e during carbon budget periods 3-6.

3.3.4 Fugitive CH_4 emissions in shale gas production

A key assumption in the Stamford and Azapagic (2014) data is a low fugitive emissions rate in the central case during shale gas production. Stamford and Azapagic (2014) do not calculate fugitive emission percentages, rather state fugitive emissions as a set volume per meter drilled based on EPA and Ecoinvent figures (Ecoinvent Centre, 2010; US EPA National Center for Environmental, 2012; Weidema and Hischier, 2006). These values are 0 m^3 gas, 4.1 m^3 gas, and 54 m^3 gas per meter drilled for the best, central, and worse cases respectively. Assuming the well is 5773 m (vertical 2773 m, horizontal 3000 m), 0 m^3 gas, 23,669 m^3 gas, and 312,000 m^3 gas leaks in each respective case. Dividing these figures by the *estimated ultimate recovery per well* (EUR) of 84.95 Mm^3 (3 bcf), 28.32 Mm^3 (1 bcf), and 2.832

Mm^3 (0.1 bcf) respectively, suggests fugitive emissions rates of 0%, 0.08%, and 11.0% during well completion.

The composition of recovered gas is different in each case, shifting from sweet in the best case to sour in the worst case. The methane content of the gas in the best-case scenario contains 0.61 kg CH_4 m^{-3} compared to 0.555 kg CH_4 m^{-3} and 0.5 kg CH_4 m^{-3} in the central and worst cases respectively. The gas also contains 0.13 kg CO_2 m^{-3}, 0.115 kg CO_2 m^{-3}, and 0.1 kg CO_2 m^{-3} respectively. Assuming that 1 t CH_4 = 25 t CO_2e, each case leaks 0 t CO_2e, 331 t CO_2e, and 3,931 t CO_2e per well drilled during well completion.

Emissions from flaring and intentional venting are set as a function of gas recovered, therefore the worst-case scenario has the least amount of flared and intentionally vented gas. Stamford and Azapagic (2014) assume venting and flaring of 11.2 g CO_2 m^{-3} and 0.264 g CH_4 m^{-3} gas produced. It is assumed that the remaining CH_4 is vented and not flared. Assuming that 1 t CH_4 = 25 t CO_2e, 1,529 t CO_2e per well is vented and flared in the best case compared with 510 t CO_2e and 50.9 t CO_2e in the central and worst cases respectively. Considering all emissions in Stamford and Azapagic (2014), the shale gas scenarios represent increases of 2.8% (best-case), 15.3% (central-case), and 175% (worst-case) over the same quantity of UK North Sea gas.

Westaway et al. (2015a) criticize the high leakage used by Stamford and Azapagic (2014) in their worst-case scenario. They claim this is unlikely due to more strict UK oil and gas regulations, and the low EUR would make the well uneconomical. Stamford and Azapagic (2015) agreed with these assessments, but included this high estimate to illustrate the worst-case inferred in the USA by the literature at the time (Howarth et al., 2011). Howarth et al. (2011) conclude that 2-3% fugitive emissions rate would be the break-even point for conventional gas to be equal with coal in electricity generation GHG emissions. Experience in the US has shown fugitive emission rates as high as 12% in some cases (Howarth, 2015) with both systemic and accidental one-off events. These events are difficult to detect, measure, and quantify. Shale gas production practices and regulations will be more strict in the UK than the US (UKOOG, 2015).

The impact of 3% fugitive emissions is calculated. These emissions occur during well completion based on the Stamford and Azapagic (2014) central case. It is assumed that the EUR (28.32 Mm^3) and gas composition (0.555 kg CH_4 m^{-3} and 0.115 kg CO_2 m^{-3}) are the same, but the leakage rate is increased from 0.08% to 3%. This increases fugitive emissions during well completion from 331 t CO_2e to 11,886 t CO_2e while the planned venting and flaring is the same (510 t CO_2e). Repeating this process for 1% (EPA, 2017, 2014) and 2% (Howarth et al., 2011) leakage rates during well completion gives 4,472 t CO_2e and 8,434 t CO_2e leaked respectively. An additional 510 t CO_2e is emitted from flaring and venting. These changes in the leakage rates correspond to increases in the production phase emissions for the central case (Stamford and Azapagic, 2014) from 65.8 g CO_2e kWh^{-1} to 265 g CO_2e kWh^{-1}, 500 g CO_2e kWh^{-1}, and 735 g CO_2e kWh^{-1} in each case. This does not consider emissions for site preparation, or on-site diesel generators which are not quantified in Stamford and Azapagic (2014).

The fugitive emissions rate is adjusted to 1%, 2%, and 3%, and the process for calculating scenario 4 is repeated, to illustrate the increase in UK associated gas production emissions (see Figure 3-6 & Figure 3-7). When the fugitive emissions rate is increased to 1%, an additional 54.2 Mt CO_2e (49.5 – 55.0 Mt CO_2e range) are emitted during carbon budget periods 3 – 6. When the rate is increased to 3%, an additional 182.3 Mt CO_2e (166.0 – 184.7 Mt CO_2e range) are emitted compared to the low (0.08%) leakage rate in scenario 4. Comparing scenario 4 (UK shale gas) with 1% leakage rate to scenario 3 (increased Qatari LNG), an additional 74.6 Mt CO_2e are emitted in the UK carbon budget periods (68.8 – 75.6 Mt CO_2e range). This same adjustment increases global emissions by 41.7 Mt CO_2e (35.1 – 42.7 Mt CO_2e range) over the same period. This indicates that even a modest increase in fugitive emissions makes shale gas worse for carbon budgets, and global emissions, compared with increases in LNG imports to the UK.

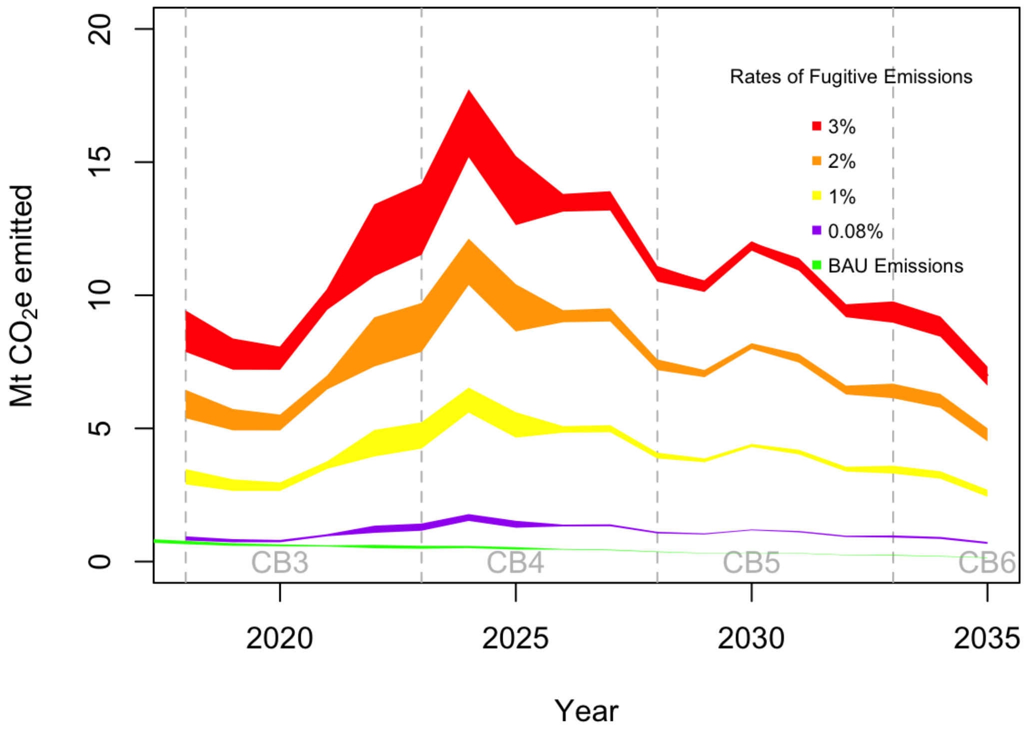

Figure 3-6 – Variance in greenhouse gas emissions from fugitive emissions, vented, and flared gas in the UK applied to scenario 4. Stamford and Azapagic (2014) assume a 0.08% leakage rate during well completion, which contribute to a small source of upstream shale gas emissions. Here leakage rates of 1%, 2%, and 3% are modelled and compared with Stamford and Azapagic, and business-as-usual (BAU) UK gas production emissions from DECC / BEIS (2017e). Variance in DECC / BEIS estimates of UK gas-derived electricity creates the range of each estimate. These emissions would need to be accommodated into UK carbon budget periods 3 – 6.

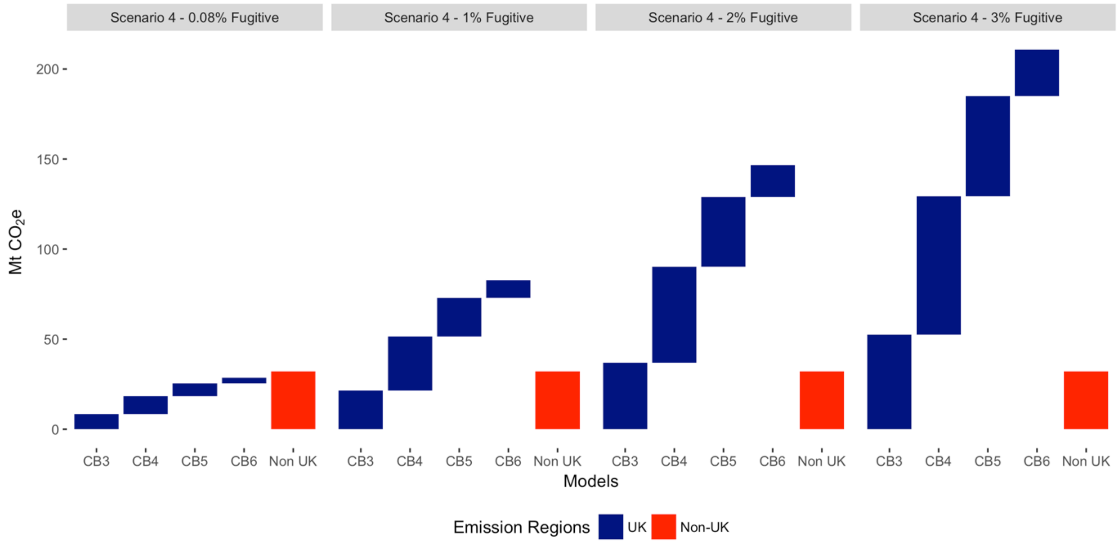

Figure 3-7 - Associated gas production emissions for natural gas power by carbon budget period for 4 models in the central 2016 BEIS projection (BEIS, 2017e), adjusted for higher fugitive shale gas emissions. Stamford & Azapagic (2014) assume 0.08% fugitive leakage in the UK. This chapter increases the leakage rate to 1%, 2%, and 3%, and find that up to 210 Mt CO_2e are emitted in gas production for natural gas power if leakage rates increase to equal observations in the US.

3.4 Conclusions

3.4.1 Impact on carbon targets and budgets

Under the current reporting regime (IPCC et al., 2006), the source of natural gas has no bearing on the UK's ability to reach the grid intensity target. The amount of projected natural gas in the electricity system, compared with lower GHG electricity sources, is the main factor in determining the grid intensity results. In response to UK Government funding cuts for carbon capture and storage (CCS), BEIS has all but eliminated CCS deployment from future projections, and increased the dependency on fossil fuel electricity to supplement the intermittency of renewables (BEIS, 2017e). This places more pressure on gas imports, or a domestic shale gas industry to meet the gas power supply gap. Imported gas would have a lower impact on UK carbon budgets under current reporting requirements, but UK shale gas may have lower overall emissions than imported LNG if shale gas production emissions were very low. However, domestic shale gas production with even a modest 1% fugitive emissions rate would risk exceedance of UK carbon budgets.

Regardless of these projections, this study indicates that the UK grid intensity target can be achieved in the second half of next decade in the BEIS *Reference Scenario* and *High Fuel Price Scenario* (BEIS, 2017e). If UK shale gas production were to go forward in 2018 (scenario 4), the UK would be responsible for all emissions industry-wide within carbon budgets under the current reporting regime. The BEIS projections (2017e) do not incorporate UK shale gas production estimates into their projections, however, using the BEIS *Existing Policies Scenario* gives the high estimates for grid intensity of gas electricity. Based on the current GHG accounting practices (see 5.2 below), current climate policies, and globalized market for LNG, the UK would likely use the lowest-priced gas regardless of origin. Without further progress on UK climate policies, cheaper LNG imports would place the grid intensity goal in jeopardy (see Figure 3-4).

3.4.2 Production and consumption-based accounting practices

This study illustrates potential gaps and unintended consequences in production-based GHG accounting practices. While production-based accounting is in line the UNFCCC reporting (IPCC et al., 2006), it allows for potential ‘offshoring’ of GHG emissions in the international trade of fossil fuels. As the four scenarios encompass a range of domestic and imported gas supplies, a scenario of increased imports could increase the emissions burden for the source nations. Even the highest import scenario considered here, the amount of gas imported to the UK from these nations is a small percentage of their total exports. The UK’s commitment to national carbon budgets based on production emission may create a perverse incentive to pursue fossil fuel imports and increase industrial emissions overseas that do not then appear in UK GHG accounts. Essentially, the UK would export responsibility of emissions to meet carbon budgets. Under these reporting systems, a domestic shale gas industry would bring some production emissions back within the UK budgets. As shown above, 20.1 Mt CO_2e in total would be emitted in the model with a low fugitive emissions rate (0.08%). These emissions would be acceptable in terms of net climate impact if they are less than the same quantity of emissions per unit of electricity from imported Qatari LNG. And, most importantly, the same quantity of LNG is not produced.

Under a consumption-based emissions accounting system, such offshoring of emissions might be avoided. For the UK to maintain a leadership position on GHG reduction policies, it is suggested that incentivization of use of the lowest GHG-intensity natural gas for power generation could be encouraged through such consumption-based accounting.

Chapter 4 Greenhouse gas emissions intensity of US and UK shale gas use for heat: impacts of source, distribution, and use

Summary

The United States became a net exporter of natural gas at the end of 2017, largely due to increased production of shale gas. The United Kingdom is pursuing a domestic shale gas industry to meet continued demand as traditional gas production decreases. At the same time, the UK is renewing gas distribution pipeline systems to decrease leakages. This chapter assesses the greenhouse gas emissions intensity of US and UK shale gas as determined by source, distribution and end use for heat. It assesses the merit order of shale gas imported to the UK from the US versus domestic production and use of shale gas in the US or UK, considering distribution network renewals and the total emissions intensity of shale gas used. It is concluded that the import and use of US-produced shale gas LNG in the UK would increase GHG emissions by 14.3 Mt CO_2e (19.2%) relative to the same quantity of domestic UK shale gas production and use. The chapter also find that losses in the distribution phase represent a highly uncertain, but potentially important component of shale gas GHG intensity. This chapter considers the implications for GHG emissions measurement and reporting, climate change mitigation via municipal pipeline renewal, and national carbon budgets to 2035.

Work presented in this chapter is based on a manuscript in review at *Science of the Total Environment.*

4.1 Introduction

In September 2016, the United Kingdom imported its first shipment of shale gas products from the United States (Davies, 2016) to Grangemouth, despite a Scottish moratorium on domestic shale gas. The shipment of ethane, a shale gas by-product, will be used by Ineos for plastics manufacturing. This shipment may herald significant imports of US shale gas to the UK in the coming years. According to US Energy Secretary, Rick Perry, the US has contracts to export 52 billion cubic meters (bm^3) yr^{-1} of US Liquefied Natural Gas (LNG) to Europe. The first US shale gas LNG shipments for Northern Europe arrived in Poland and The Netherlands in June 2017 (Perry, 2017), and LNG shipments arriving in the UK in March (EIA, 2018b)

In the US, traditional natural gas production growth plateaued in 1970 for decades (EIA, 2018c). However, beginning in 2007, the so-called ‘shale gas boom’ rejuvenated the industry such that the US became a net exporter of natural gas in 2017 (EIA, 2018b). The US Energy Information Agency (EIA) forecasts that the US will export over 210 bm^3 yr^{-1} by 2040 (EIA, 2017; Yen, 2016) in addition to continued domestic consumption. BP currently holds a contract in the UK for US shale gas LNG imports, and will begin importing this year (Perry, 2017).

The overall greenhouse gas (GHG) emissions intensity of shale gas use, either in the US or the UK, can be affected by a number of different factors along the gas supply chain. Much attention has been focused on the fugitive emissions of methane (CH_4) occurring during extraction of shale gas (e.g. Alvarez et al., 2012; Howarth, 2015), with the source of shale gas having a potentially very large impact on overall emissions intensity (Turk et al., 2018a). Likewise, processing, transport, distribution, and the end use of the gas, whether for electricity generation, heat or domestic supply, may significantly alter life cycle emissions (e.g. Stamford and Azapagic, 2014; Weber and Clavin, 2012a).

Of these key phases in the gas supply chain, the distribution phase is arguably the least understood in terms of its impact on overall emissions intensity (e.g. the extent of fugitive losses from municipal gas distribution networks, Jackson et al., 2014; Phillips et al., 2013; Zazzeri et al., 2017). The National Grid in the UK is in the process of renewing gas pipelines at the city level with polyethylene pipelines with 80 year lifespans (Dodds and McDowall, 2013; HSE, 2011), while in the US, some city-level CH_4 measurements have indicated high fugitive losses and have highlighted the potential mitigation benefits of pipeline renewal (Gallagher et al., 2015; Jackson et al., 2014; Phillips et al., 2013).

This chapter assesses the impact of pipeline renewal for US and UK natural gas transmission and distribution. This chapter is unique is extending the upstream gate of UK gas to include US shale gas, transported at LNG, and compares the entire life cycle emissions with domestically produced UK shale gas.

The multiple phases that determine overall emissions intensity of gas use, and the national and local variations in potential fugitive CH_4 losses in the US and UK shale gas supply chains, raise questions about what the lowest emissions actions for gas extraction, distribution methods, and end-use in the UK and US might be (see Figure 4-1). This chapter assess the GHG emissions intensity of imported US shale LNG to the UK compared to that of domestic shale gas production and use in the US and UK, considering production practices, transport, and scenarios of gas grid renewal. As such, this chapter explores a ‘climate change merit order’ for end-use of UK versus US-produced shale gas.

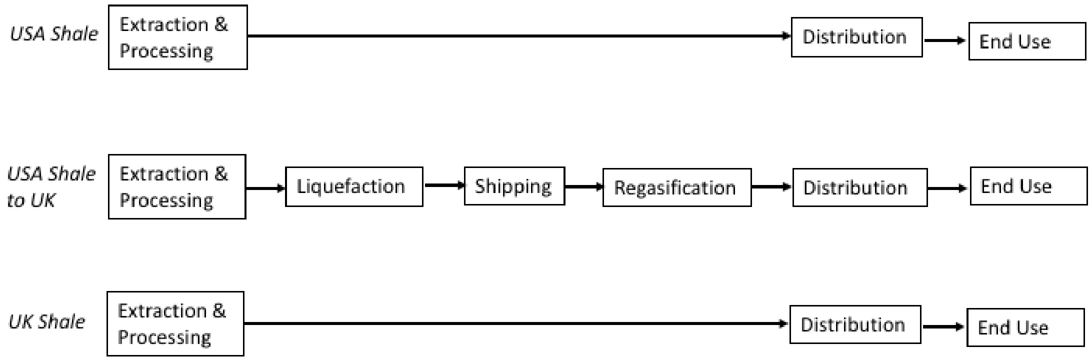

Figure 4-1 - Process diagram for shale gas use in the US & UK. Shale gas from the US is being imported to the UK via LNG. When the UK shale gas industry begins production, the liquefaction, shipping, and regasification processes are avoided. Differences in the efficiency of local distribution networks and end use will determine the least GHG-intense pathway.

4.1.1 LCA of shale gas

Abrahams et al. (2015) assessed life cycle greenhouse gas emissions from US LNG exports. They found that exported US shale LNG has mean pre-combustion emissions of 37 g CO_2e MJ^{-1} (27 – 50, 90% confidence interval - CI) when regasified in Europe or Asia. Shipping US LNG accounted for only 3.5 – 5.5 % of pre-combustion lifecycle emissions, demonstrating that shipping distance is not a major factor in total emissions. The bulk of pre-combustion emissions were found to be in upstream production, at 27.9 g CO_2e MJ^{-1} (18.5 – 28.4, 90% confidence interval). The combustion phase added a further 43-50 g CO_2e MJ^{-1}.

In a scenario of US shale gas LNG export for electricity generation in Europe, Abrahams et al. (2015) estimate that life cycle emissions are 655 g CO_2e kWh^{-1} with a confidence interval of 562 – 770, or 87 g CO_2e MJ^{-1}. This central estimate represents an 11% increase over the use of US-produced shale gas in US electricity generation (Abrahams et al., 2015).

Stamford and Azapagic (2014) estimate that traditional North Sea Gas has life cycle emissions of 401 g CO_2e kWh^{-1}, compared to UK-produced shale gas at 462 g CO_2e kWh^{-1} (with a range of 412 – 1108). Note that this study assumes an 11% leakage rate in the worst-case scenario for UK shale gas extraction, compared to the upper-bound of 4% in the Abrahams et al. (2015) study of US shale gas extraction (Turk et al., 2018a).

A criticism of the Stamford and Azapagic (2014) study is the assumed 52.5% efficiency for natural gas combined cycle (NGCC) power efficiency (Westaway et al., 2015a, 2015b). Westaway et al. (2015a) suggest using Stamford and Azapagic's own assumed efficiency to convert the 462 g CO_2e kWh^{-1} upstream to chemical energy to avoid this efficiency assumption. Converting the full life-cycle in SI units (where 1 kWh = 3.6 MJ), Stamford and Azapagic's (2014) estimates equate to 67 g CO_2e MJ^{-1} (60 – 161 g CO_2e MJ^{-1} range) when UK-produced shale gas is used for domestic electricity generation. This central estimate represents a 20 g CO_2e MJ^{-1} saving in GHG emissions intensity over the use of US shale gas LNG imports to Europe for electricity generation compared to the observations in Abrahams et al. (2015).

As US shale gas production will continue to exceed domestic demand, increasing exports to the UK (where domestic gas production is unlikely to meet demand) (OGA, 2017; Turk et al., 2018a) are likely. If these US shale gas imports represent an increase in GHG emissions intensity relative to UK-produced shale gas, the question then arises as to whether US shale gas leakage rates (i.e. upstream emissions) and supply chain emissions can be managed sufficiently to close the estimated 20 g CO_2e MJ^{-1} emissions intensity gap. In such a case, US shale gas LNG imports could allow for future UK gas demand gaps (OGA, 2017; Turk et al., 2018a) to be met while minimising impacts on overall emissions intensity for gas-fired UK electricity generation. For example, the Abrahams et al. (2015) study assumes that the equivalent of nearly 10 g CO_2e MJ^{-1} are lost due to fugitive emissions in US upstream production.

More recently, Balcombe et al. (2017) surveyed 454 papers on the natural gas supply chain largely focused on the US, and selected 250 for analysis. They found that natural gas supply chain combined CO_2 and CH_4 emissions ranged from 3.6 to 42.4 g CO_2e MJ^{-1} HHV with a central estimate of 10.5.

This is a significant decrease from previous findings by Abrahams et al. (2015). Balcombe et al. also found six estimates of fugitive emissions from shale gas production above 100 g CO_2e MJ^{-1} HHV (232, 285, 304, 618, 1910, 5250). These estimates illustrate the “fat tail” of emissions in shale gas production, where observations from “super-emitters” skew results upwards by 2-3 orders of magnitude during gas extraction and recovery.

4.1.1.1 Reduced emission completions (RECs)

A significant proportion of the literature on fugitive CH_4 leakages focuses on the flowback phase of well completion (Howarth, 2015; O’Sullivan and Paltsev, 2012). Flowback is the phase when large quantities of hydraulic fracturing fluid return to the surface and can bring with them large volumes of natural gas. Balcombe et al. (2017) note that these quantities are equivalent to 0-87 g CO_2e MJ^{-1} HHV per well completion. Three key differentiators separate the data on flowback emissions: First is whether the data are primary source measurements or secondary calculations. Second is whether the well is conventional or unconventional, as conventional wells do not utilize hydraulic fracturing and thus have no flowback emissions. Conventional well emissions equate to less than 0.1 g CO_2e MJ^{-1} HHV per well completion. Third is whether Reduced Emission Completions (RECs, also known as “green completions”) are utilized in the well completion process. RECs are a process where separate equipment are used to separate solids from liquids during the flowback period of well completion. RECs are used to capture gas during flowback for added production or flaring and have been compulsory in the US since 2015, and are required in the UK. They can reduce emissions from well completions by 75 – 99%. (Oil and Natural Gas Sector: New Source Performance Standards, National Emission Standards for Hazardous Air Pollutants, and Control Techniques Guidelines).

4.1.2 Transmission, storage, and distribution network

The UK natural gas pipeline distribution network consists of high-pressure national transmission network (NTS), medium-pressure distribution network (Local Transmission System, LTS), and low-pressure building connections.

The NTS was constructed in the 1960s from high-strength steel and is expected to stay in service for 80 years, or roughly until 2050 (Dodds and Demoullin, 2013; Dodds and McDowall, 2013). The UK is in a 30-year process of replacing cast-iron low-pressure distribution pipeline mains with polyethylene, the "Iron Mains Replacement Programme" (IMRP). This project began in 2002 and will conclude in 2032 (Dodds and McDowall, 2013; HSE, 2011).

4.1.3 UK gas demand

BEIS and OGA estimate that UK gas demand will fluctuate, but not decrease below 61.5 bm^3 yr^{-1} primary demand before 2035. Domestic production is projected to fall at -5% yr^{-1} from 2020 onwards (OGA, 2017; Turk et al., 2018a). Gas used for electricity generation is projected to fall, but still be used in large quantities through 2035. Additionally, 463 bm^3 gas will be used for industrial and domestic heating from 2020 to 2035 (446.7 – 487.7 range, see Figure 4-2) (BEIS, 2017e; OGA, 2017).

Currently the UK imports 45% of gas demand, and will rise steadily to 78% by 2035 (OGA, 2017). This means that the UK will depend upon increases in imported natural gas and/or new domestic sources of natural gas through 2035. This chapter examines the impacts of UK and/or US LNG shale gas supply meeting the UK gas demand.

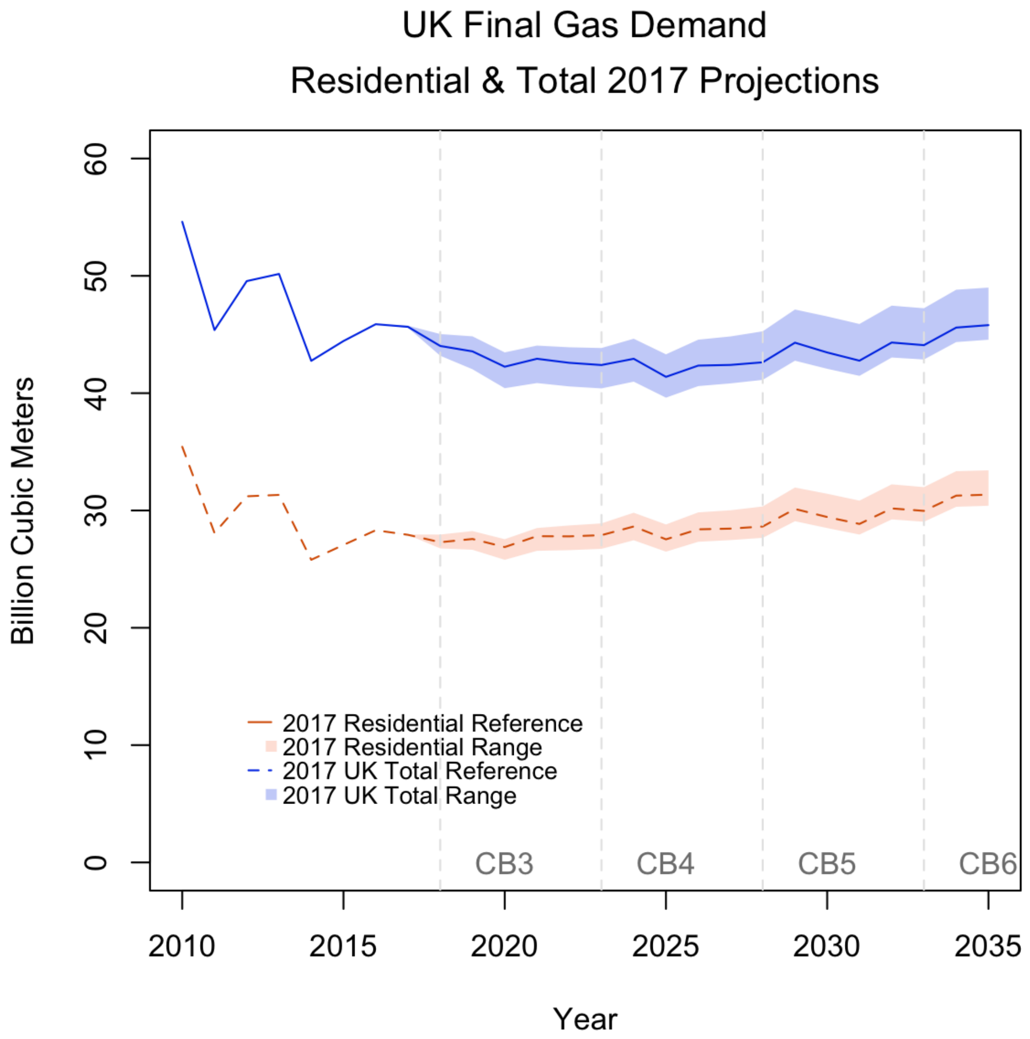

Figure 4-2 - UK final gas consumption calculated from BEIS (2017e). The UK is projected to have continued reliance on natural gas in residential sectors through 2035. Natural gas usage estimates in electricity generation are excluded in the total range.

4.2 Methodology

Table 4-1 – Natural gas phase emissions from Balcombe et al. (2017)

Phase Emissions (g CO_2e MJ^{-1} HHV)	Low	Median	95^{th} Percentile	Max
Pre-production	0.1356	0.59	1.8	8.48
Extraction	1.5	1.0	19.8	5465.6
Processing	0.86	4.1	10.5	13.6
Transmission, storage, and distribution	0.6	5.0	10.2	12.0
Estimated total	3.1	10.7	42.4	5499.7

Table 4-2 - Natural gas LNG phase emissions from Balcombe et al. (2017)

Phase Emissions (g CO_2e MJ^{-1} HHV)	Low	Median	95^{th} Percentile	Max
Pre-production	0.158	0.68	2.07	9.83
Extraction	1.8	1.11	22.8	6282.3
Processing	0.99	4.7	12.1	15.6
Transmission, storage, and distribution	0.69	5.7	11.8	13.8
Estimated sub-total	3.7	12.2	48.7	6321.53
LNG sub-total	3.9	8.9	15.9	20.3
Estimated total	7.6	21.1	64.6	6341.8

Table 4-3 - LNG phase emissions from Balcombe et al. (2017)

LNG Phase Emissions ($g\ CO_2e\ MJ^{-1}$ HHV)	Low	Median	95th Percentile	Max
Liquefaction	2.8	6.4	9.0	10.5
Transportation	0.86	1.98	4.8	7.3
Re-gasification	0.3	0.5	2.1	2.5
LNG sub-total	3.9	8.9	15.9	20.3

4.2.1 Extraction and processing emissions

The leakage rate is the most impactful figure on the upstream emissions in Abrahams et al. (2015). However, the maximum leakage rate is only 4% based on Heath et al. (2014) and Weber and Clavin (2012). There is discussion in the literature that fugitive emission rates have systemically reduced in the US (Schwietzke et al., 2016) since the beginning of the shale boom. However, there have been observations of leakage rates of 12% (Howarth, 2015; Schneising et al., 2014), and an overall increase in CH_4 emissions from gas production in US inventory data (EPA, 2017). Schneising et al. (2014) estimate 9.5% leakage rates at well-completion and an additional 2.5% from upstream distribution leakages based on satellite observations.

Heath et al. (2014) harmonized LCA estimates for shale gas and conventional gas power according to leakage rates at the time, and cite an emissions of intensity for upstream emissions of 21.3 $g\ CO_2e\ MJ^{-1}$ (17.6 – 24.8, 90% CI). Here, we use the same method to derive estimates from more recent observations (Howarth, 2015; Schneising et al., 2014).

A potential UK shale gas industry would, in theory, have a GHG emission intensity 20 $g\ CO_2e\ MJ^{-1}$ lower than that of imported US shale LNG. Up to 10 $g\ CO_2e\ MJ^{-1}$ of this difference is based on 1.3% of unconventional US shale gas released as fugitive emissions (Heath et al., 2014b), compared to near-zero UK fugitive emissions assumed by Stamford & Azapagic (2014; Turk et al., 2018a). Allen et al. (2014) demonstrated that leakage rates can be kept below 1% when best practices are observed in Texas. Systemic leakages observed and documented in the literature indicate that this low number is unlikely (Howard et al., 2015).

Considering the multitude of estimates, this chapter uses the data surveyed by Balcombe et al. (2017) in the calculations because the recentness of the study, and its incorporation of RECs within the analysis (see Tables 4-1,4-2, and 4-3). This chapter assumes that the median upstream extraction and processing emissions are 5.7 g CO_2e MJ^{-1} (2.5 – 32.1, 95% CI) and are 6.5 g CO_2e MJ^{-1} (3.0 – 36.97, 95% CI) when extracted for LNG. These estimates decrease the median from Heath et al. (2014), but demonstrate a more significate upper-end emissions intensity.

4.2.2 Liquefaction, shipping, and regasification emissions

Abrahams et al. (2015) found that liquefaction adds 6.2 g CO_2e MJ^{-1} (2.4 – 8.8, 90% CI) to natural gas carbon intensity, based on industry studies and academic literature (Arteconi et al., 2010; Barnett, 2010; Biswas et al., 2011; Cohen, 2013; Hardisty et al., 2012; Heede, 2006; LCFS, 2012; Okamura et al., 2007; Skone et al., 2014, 2012; Tamura et al., 2001; Verbeek et al., 2011; Yoon and Yamada, 1999; Yost and DiNapoli, 2003). The Monte Carlo method chosen by Abrahams et al. (2015) skews the results to the lower end of the range in the literature, and is justified by increased efficiency demonstrated in later studies. Balcombe et al. (2017) further refine this figure to a 6.4 g CO_2e MJ^{-1} median (2.8 - 10.5, 95% CI). Here this chapter uses figures from Balcombe et al. (2017) for the liquefaction phase in the overall emissions intensity calculations.

Shipping LNG from the Louisiana coast to UK and Netherlands has mean emissions of 1.2 – 1.3 g CO_2e MJ^{-1} according to Abrahams et al. (2015). These transport emissions nearly double if LNG originates on the Pacific coast in Coos Bay, Oregon. When normalizing for SI unit (where 3.6 kWh = 1

MJ), Stamford and Azapagic (2014) found shipping LNG from Algeria and Qatar to the UK emits 22.8 and 32.8 g CO_2e MJ^{-1} respectively. Stamford and Azapagic (2014) relied on LCA calculations from Ecoinvent (2010; Weidema and Hischier, 2006), whereas Abrahams et al. (2015) on more extensive literature (Corbett and Winebrake, 2008; Heede, 2006; Mokhatab et al., 2013; Skone et al., 2014; Tamura et al., 2001). It is unclear what assumptions are made in the Ecoinvent database for LNG shipping, including electricity loss in conversions. Balcombe et al. (2017) found LNG transportation emissions to vary widely depending on route, efficiency of tanker, and CO_2 combustion. They found total emissions from LNG transportation to be 0.9 – 7.3 g CO_2e MJ^{-1} HHV, and broadly in line with Abrahams et al. (2015) for similar routes from North American to Europe (2.0 g CO_2e MJ^{-1} HHV median, 0.9 – 3.1, 95% CI)(Balcombe et al., 2017). We use the range reported in Balcombe et al. (2017) in the calculations.

Regasification emits only 0.5 g CO_2e MJ^{-1} (0.26 – 2.53, 95% CI) mostly from CO_2 combustion for energy usage (Balcombe et al., 2017). This chapter uses this figure for this phase in the calculations.

4.2.3 Gas distribution and leakage

In Boston, Phillips et al. (2013) used mobile analysers to measure CH_4 leakages across the entire city distribution network. They found an isotopic signature indicating that the leakages were from anthropogenic sources, with concentrations as high as 15-times the global mean background level. McKain et al. (2015) estimate that 2.7% +/- 0.6% of lifetime natural gas production leaks from downstream natural gas components including transmissions, distribution, and end use. Emission inventories previously estimated 1.1% leakages. In Washington D.C., Jackson et al. (2014) measured city-wide methane concentrations 37% above 2012 global background concentrations observed at Mauna Loa. They used the same methods as Phillips et al. (2013), and confirmed the leaks as having anthropogenic sources through isotopic signatures. US cities which have undergone improvements in pipeline distribution have measured 90% - 96%

lower CH_4 concentrations (Gallagher et al., 2015; von Fischer et al., 2017). Von Fischer et al. (2017) suggest that the final leaks could be further reduced with upgrades to consumer appliances.

In the UK, data are often older and more scarce. In 1990, Mitchell et al. (1990) estimated that the distribution system leaked between 1.9% and 10.8% (5.3% mean). Lowry et al (2001), and Zazzeri et al. (2017, 2015) measured $\delta^{13}C\text{-}CH_4$ plumes in London, fingerprinting fossil fuel emissions within the distribution network. Zazzeri et al (2017) note that their mobile CH_4 measurements show 11 spikes of CH_4 density greater than 2.5 ppm above the background concentration in 155 miles driven. This is a lower ratio per mile compared to Phillips et al (2013) observations of 3356 leakages in Boston over 785 miles driven. Boothroyd et al. (2018) found leakage rates along the NTS to be of similar magnitude and density to lower end of US distribution leaks (Chamberlain et al., 2016; Gallagher et al., 2015), but note that local distribution leak rates remains unclear. These studies illustrate the need for further review of systemic leakage rates in UK and US cities. This chapter does not include these emissions rates in our calculations due to the low number of observations, however, as discussed below, quantification of these “super emitters” is an important area for future study and potential regulation.

Other studies estimate distribution leakage rates of 0.1 – 1.9% of produced CH_4 in the UK gas distribution network (Harrison et al., 1996; Lamb et al., 2015; Mitchell et al., 1990; Moore et al., 2014), equivalent to an emissions intensity for this phase of 0.3 – 7.2 g CO_2e MJ^{-1} (Balcombe et al., 2017). Here, this chapter assumes these same leakage rates for UK and US, along with the 90 – 96% savings potential of interventions described in Gallagher et al. (2015) and von Fischer et al. (2017). Chamberlain et al (2016) note that the prevalence of cast iron replacement programs significantly reduce CH_4 leakage rates, in agreement with Gallagher et al (2015). They note that these results suggest the prevalence of leak-prone pipe is the main driver of CH_4 distribution leakage, regardless of city size. This chapter assumes that the leakage rate for transmission, storage and distribution is 5.0 g CO_2e MJ^{-1} HHV (0.6 – 10.2 range) (Balcombe et al., 2017).

4.2.3.1 Pipeline renewal assumptions

We assume that the UK pipeline renewal project will be completed by 2032 (HSE, 2011) and will reduce fugitive emissions from distribution by 90 – 96% from a 2002 baseline (Balcombe et al., 2017; Chamberlain et al., 2016). For simplicity, it is assumed that a linear reduction from a baseline of 5.0 g CO_2e MJ^{-1} HHV (0.6 – 10.2 range) to 0.2 - 0.5 g CO_2e MJ^{-1} HHV. For simplicity, the improvements to transportation, storage, and distribution rates are applied to the total figures. Exact estimates of the portion of leakages (and therefore improvements) for only the local low pressure building connections is not possible to ascertain. It is possible that the interventions studied could skew high (more effective), but will give a view to the entire system being renewed.

The US does not have a coordinated or nationalize pipeline renewal program similar to the IMRP in the UK. Individual states set regulations and timelines for gas infrastructure improvement and repair (e.g. (New York State Electric and Gas Corporation (NYSEG), 2002; Pennsylvania Public Utility Commission (PUC), 2016; Reed, 2017). Engagement with regulations and repair programs is up to the individual operators, and can have disparate results (Chamberlain et al., 2016). In the US, two separate cases are assumed, where some locales have renewed pipeline (e.g. Ithaca, NY), while others use leaking cast iron distribution infrastructure of the same baseline rates as above (Chamberlain et al., 2016; Gallagher et al., 2015; von Fischer et al., 2017).

4.2.4 UK gas demand

It is assumed that future UK gas demand fluctuates according to government estimates from BEIS (2017e) and OGA (2017). These estimates project UK North Sea gas production declining -5% yr^{-1} from 2022 onwards. This chapter uses *scenario 4* from Chapter 3, Turk et al. (2018a), to model the quantity of shale gas required to meet UK demand. In this scenario, a fixed quantity of gas is imported to the UK from Norway (19.6 bm^3 yr^{-1}), Qatar (10.1 bm^3 yr^{-1}), and Algeria (0.91 bm^3 yr^{-1}). Additionally, 277.7 bm^3 of natural gas will be imported from the US as shale gas LNG or produced domestically as UK shale gas from 2020-2035. This gas accounts for 43.5% of demand in 2020 (11.0 bm^3) to 78.4% in 2035 (19.8 bm^3). BEIS projects that the UK will use 463 bm^3 natural gas (446.7 – 487.7 range) for residential heating in 2020 – 2035), compared to a national total final gas consumption of 692.2 bm^3 natural gas (666.6 – 729.9 range) over the same period (BEIS, 2017a).

It is assumed that this gas contains 39.6 MJ m^{-3} based on BEIS estimates for gas consumption in 2016 (BEIS, 2017a). If this gas were to remain in the US, it is assumed the same energy content used in heating and domestic use.

Abrahams et al. (2015) assume gas combustion for heating emits 57 g CO_2e MJ^{-1} (51 – 65, 90% CI) without accounting for leakages in the distribution network. The same figures for combustion for heat in the US and UK are assumed.

4.2.5 Cases examined

This chapter focuses on the GHG intensity of the 277.7 bm^3 supply of gas required to fill the UK production gap from 2020 - 2035. For simplicity, it is assumed that this quantity of gas is sourced either from US shale gas LNG imports or from UK domestic shale gas. This chapter first examines the GHG intensity of the supply of gas up to the point of combustion for residential heating in the following scenarios:

1. US shale gas used in the US
2. US shale gas exported to the UK as LNG
3. UK shale gas used domestically

For the UK cases, this chapter examines the impact on the progress of the IMRP over time (see **Error! Reference source not found.**). In the US the i mpact of completed renewals is compared with un-renewed cast iron distribution.

4.3 Results

4.3.1 Merit order for shale gas heat

Prior to interventions of distribution renewals, the GHG footprint of shale gas for heat emits 67.7 g CO_2e MJ^{-1} (52.9 – 107.3 range) in the UK compared to 78.1 g CO_2e MJ^{-1} (58.5 – 129.7 range) when using US shale gas transported as LNG (Figure 4-3). With intervention by completing the renewal of the pipeline distribution network, the GHG footprint of shale gas for heat emits 63.2 g CO_2e MJ^{-1} (53.6 – 98.12 range) in the UK compared to 73.6 g CO_2e MJ^{-1} (58.7 – 121.2) when using US shale gas transported as LNG (see Figure 4-4). If the US shale gas remains in the US for local heat 67.7 g CO_2e MJ^{-1} (54.1 – 107.3) are emitted.

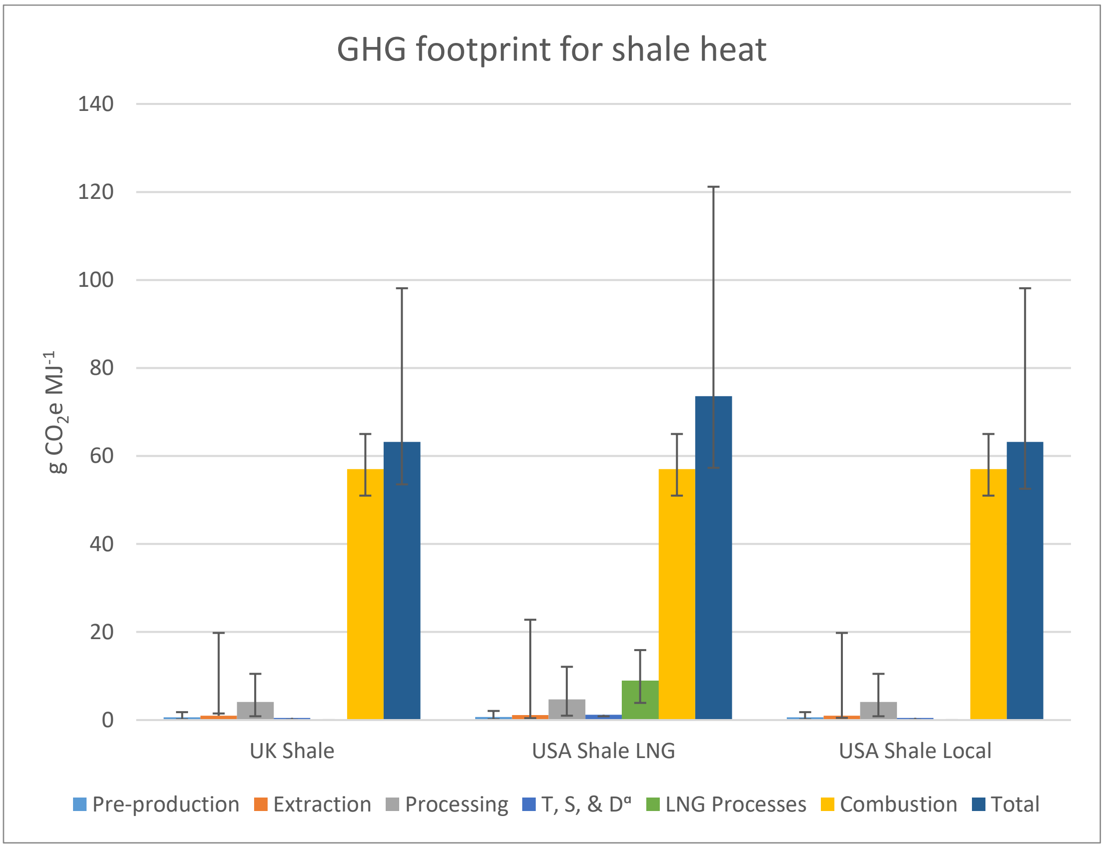

Figure 4-3 - Phase emissions for shale gas used for local heat in the UK and US. The range is expressed as the minimum observations to the 95th percentile observations by Balcombe et al. (2017). USA shale gas exported as LNG to the UK has the highest overall GHG profile due to additional emissions in the LNG process. αTransmission, Storage, & Distribution

Compatibility of fossil fuel energy system in the UK for climate targets. When considering the increases in demand for gas for residential heat in the UK, and the assumed energy content of gas (57 g CO_2e MJ^{-1}, 51 – 65, 90% CI), UK shale gas would emit 27.5 MT CO_2e (23.3 – 42.7 range) in 2020 compared to 32.0 (25.5 – 52.8 range) if US shale gas LNG were used in the same quantity. More importantly for the UK, 25.3 MT CO_2e would be released in the UK, and would save 2.2 MT CO_2e from carbon budgets in year 2020, while adding an additional 4.6 MT CO_2e to the atmosphere. Although there would be small savings in the UK carbon budget, the overall impact to global GHG balances would increase.

Projecting forward to the year 2035, as the domestic supply of conventional gas falls, and more shale gas is used, the same proportions of savings and emissions occurs. UK shale gas would emit 49.6 MT CO_2e if used for domestic heat (42.0 – 77.0 range), compared to 57.7 MT CO_2e if US shale LNG is imported in the same quantity (46.0 – 95.1 range). However, the UK's share of the LNG emissions would be 45.7 MT CO_2e (40.7 – 53.6 range).

These results suggest that the use of LNG is the least preferable option for atmospheric emissions due to the added processes for LNG, which increase emissions throughout the supply chain (See Supporting Tables 1, 2, & 3). The UK carbon budget would benefit from the imported gas as opposed to a new domestic industry, however, the total emissions to the atmosphere would increase.

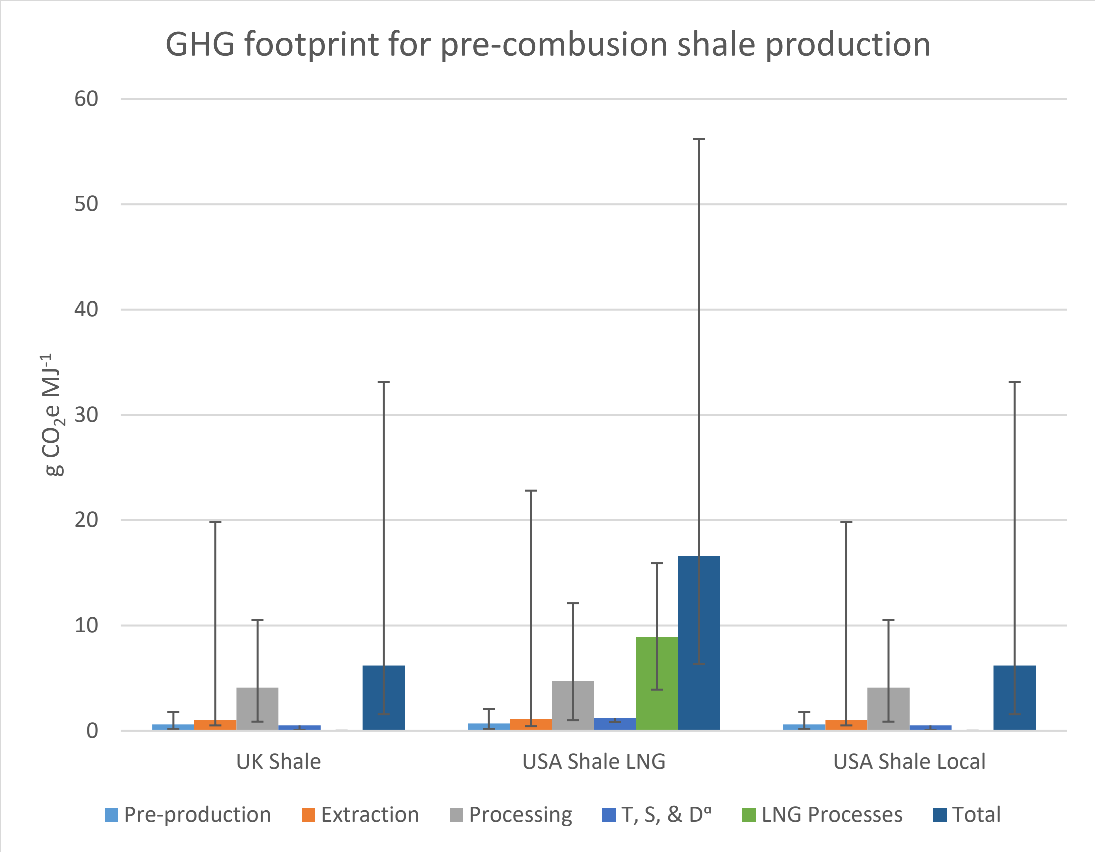

Figure 4-4 -GHG phase emissions for shale gas supply with renewed UK pipeline infrastructure from IMRP. The range is expressed as the minimum observations to the 95th percentile observations by Balcombe et al. (2017). αTransmission, Storage, & Distribution

4.3.2 Impacts of interventions

The impacts of pipeline renewals would save the 220 MT CO_2e in in the median scenario (see Figure 4-5) over the years 2002-2035 compared to a UK “business-as-usual” (BAU) scenario. The effectiveness of pipeline renewal is potentially greater than suggested in our results, as we examine only a portion of UK total gas demand. For simplicity, equal volumes of imported US shale gas LNG are compared with domestically produced UK shale gas. These quantities are a portion of the total gas used through the years 2020-2035, and would translate to greater losses prior to interventions, and greater savings as the IMRP progresses.

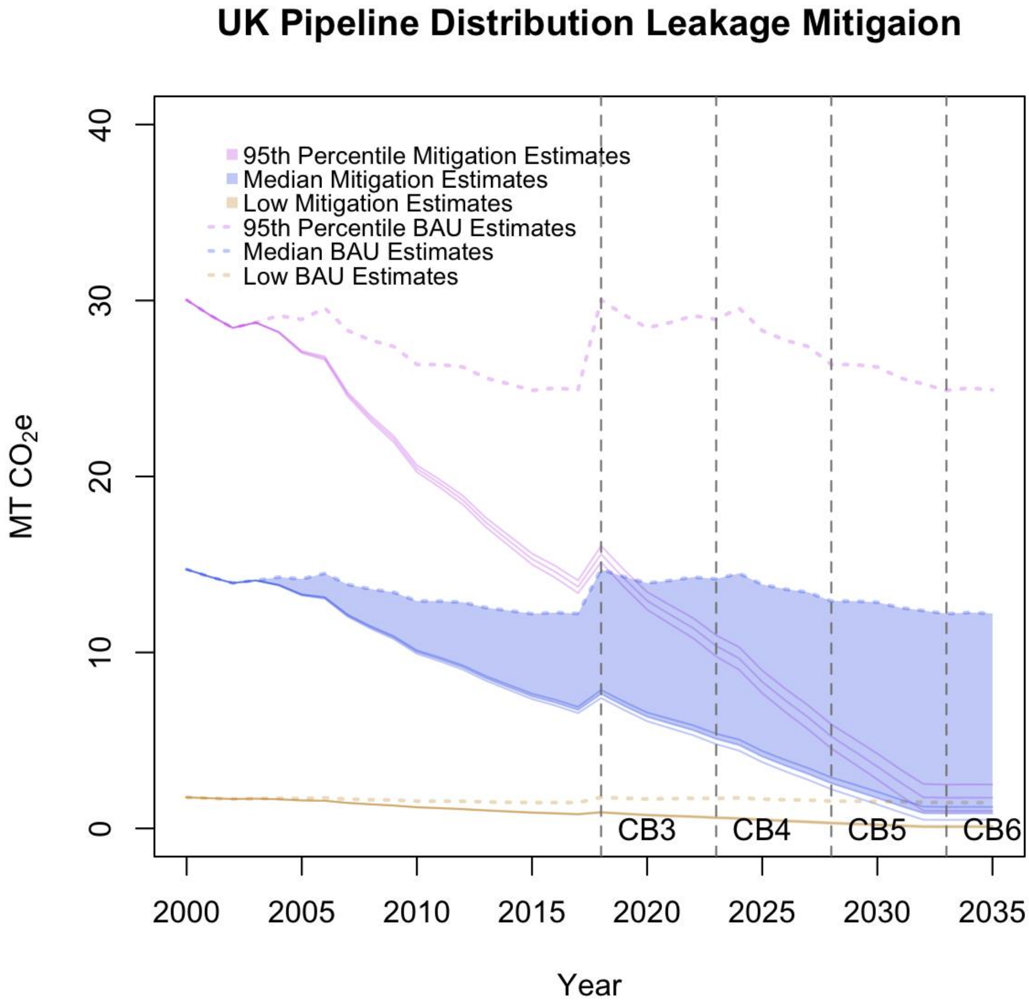

Figure 4-5 - UK gas pipeline distribution leakage rates projections based on emission estimates from Balcombe et al. (2017) and OGA (2017) gas demand, pipeline renewal estimates HSE (2011), Chamberian et al. (2016), and von Fisher et al. (2017). This chapter projects CO_2e savings in shale gas scenario with pipeline renewal projects across the UK will save 220 MT CO_2e (213 – 227 median range) from 2002 – 2035 from the BAU scenario compared to the same quantity flows of gas without interventions (blue wedge). The blue wedge represents the calculated mean UK national emissions saved during the 30 year IMRP.

When these quantities of gas are normalized for final heat delivery from 2020 – 2035, supply chain atmospheric emissions are highest when importing US shale LNG to the UK, compared to all other cases (see Figure 4-6). There is also an increase of 14.3 Mt CO_2e in UK emissions over carbon budgets 3-6 compared to a the same quantity of UK shale gas extracted from 2020 – 2035. This is due to the median estimates for regasification exceeding the median leakage rate for equal quantites of UK shale gas and US shale LNG (see Tables 4-1, 4-2, and 4-3). Importing this quantity of gas increases atmospheric emissions by an additional 163.77 Mt CO_2e from extraction, processing, gasification, and trasportation, during the same carbon budget periods. These US emissions would fall outside of UK boundaries, suggesting disadvantages for UK carbon budgets and for atmospheric emissions.

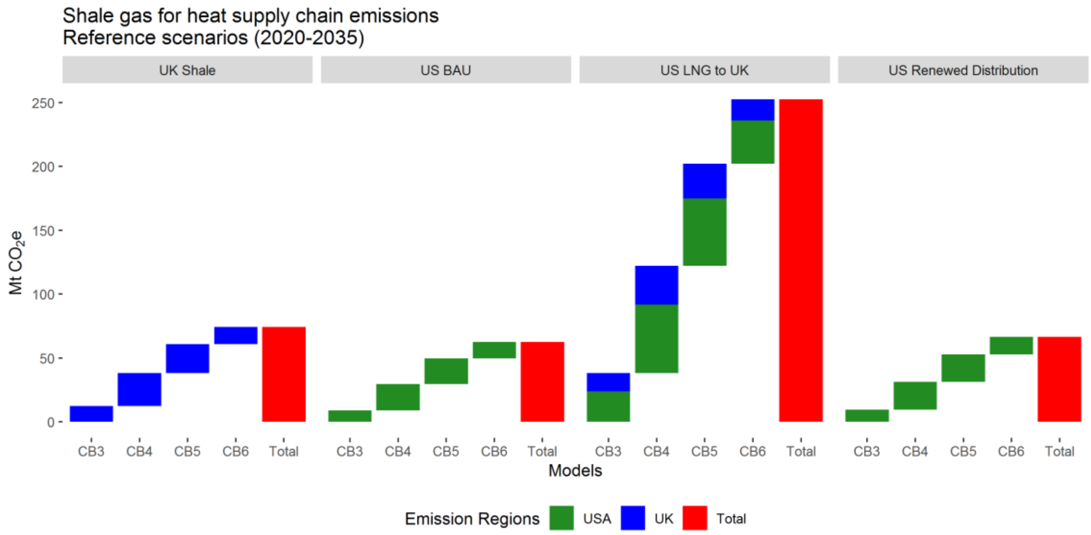

Figure 4-6 -Supply chain emissions for 277.7 bm^3 of gas required to fill the UK production gap from 2020 – 2035. When importing US shale LNG to the UK, there is an increase in UK emissions (blue) by 12.3 Mt CO_2e and total atmospheric emissions for the same quantity of gas supplied.

Super-emitting sites may also skew the effectiveness of the IMRP. As suggested by Lowry et al. (2001) and Zazzeri et al. (2015, 2017), a significantly large quantity of gas leaks in the London distribution network. However, the exact quantity of gas over time is not yet well known. It would be unrealistic to assume an exact figure, but the existence of these leaks suggests that the UK pipeline system has leaked at rates higher that the range used in this chapter. The IMRP would then be even more effective in the mitigation of GHGs in the UK. However, because a baseline has yet to be clearly established, the effect on carbon budgets is zero.

4.4 Discussion

Abrahams et al. (2015) compare US shale LNG for use in the UK and Netherlands to the “most likely” alternatives: Russian natural gas and locally produced coal. The UK has climate policies which prohibit the use of coal for electricity and heat without CCS (Directive 2010/75/EU of the European Parliament and of the Council of 24 November 2010 on industrial emissions (integrated pollution prevention and control)). This chapter therefore compares US domestic shale gas use with exported US shale LNG to the UK, together with estimates for UK domestic shale gas extraction and use based on the surveys of Balcombe et al. (2018, 2017) and Abrahams et al. (2015).

Abrahams et al. (2015) estimated pre-combustion emission from US shale gas extraction based on observations from Weber and Clavin (2012b), however these figures include production emissions prior to the US EPA rule change requiring RECs (Oil and Natural Gas Sector: New Source Performance Standards, National Emission Standards for Hazardous Air Pollutants, and Control Techniques Guidelines). Balcombe et al. (2017) reharmonize emissions including effects of RECs after the 2012 rule change.

There are several key differences between the Balcombe et al. (2018, 2017) and Abrahams et al. (2015) studies which should be noted:

First, Abrahams et al. (2015) use a 100-year GWP of 36 for CH_4, whereas Balcombe use 34. This small difference will increase the impact of leakages observed in Abrahams et al. (2015) calculations and magnify the effectiveness of any reduced emission interventions.

Second, Abrahams et al. (2015) surveyed estimates for each phase of shale gas LNG supply and performed a triangular Monte Carlo analysis. They then reported the results with a 90% confidence interval in units g CO_2e MJ^{-1}.

Compatibility of fossil fuel energy system in the UK for climate targets. Whereas, Balcombe et al. (2018, 2017) reported all emissions figures, excluded the top 5% of estimates which skew the mean because of the "fat tail" of super emitters, and reported the low, median, and 95th percentile estimates in g CO_2e MJ_{-1} HHV. Abrahams et al. is based on methodology of Weber and Clavin (2012a) who cite guidance from EIA (2011) which recommends reporting HHV rather than net or lower heaving value (LHV). Unless otherwise noted, it is assumed that Abrahams et al. and Balcombe et al. report results in the same units.

Third, Abrahams published results in 2015, around the same time the EPA rules (Oil and Natural Gas Sector: New Source Performance Standards, National Emission Standards for Hazardous Air Pollutants, and Control Techniques Guidelines) came into effect, whereas Balcombe captured more results after the requirement of RECs in the US.

4.5 Policy implications and further research analysis

4.5.1 Implications of leaking US distribution network

More direct measurements are needed to test the leakage rates across the distribution networks in the US and UK. Renewal, repair, or other methods of reducing losses in the gas distribution network has the added effect of reducing demand on supply. In other words, renewed gas distribution pipelines decrease losses and increase efficiency of delivery.

Balcombe et al. (2017) exclude the super-emitters from their modelling, as they skew results too high. The observations of pipeline leakages are observations on the super-emitters, and not clearly quantified. This chapter uses leakage rates of 0.1 – 1.9% of produced CH_4 (Harrison et al., 1996; Lamb et al., 2015; Mitchell et al., 1990; Moore et al., 2014), equivalent to 0.5 – 10.2 g CO_2e MJ^{-1}. However, the presence of the super-emitters would skew the data higher and make interventions more effective compared to abated CO_2e.

Lowry et al (2001) and Zazzeri et al. (2015, 2017) measured $\delta^{13}C\text{-}CH_4$ plumes in London, fingerprinting fossil fuel emissions within the distribution network, demonstrating super-emitters in London; agreeing with US studies cast-iron mains leakage. However, these leaks are difficult to quantify. This is an area that is in need of more study, and would have significant commercial implications. Mayfield et al. (2017) analyse policy options to reduce methane emissions from super-emitters by 83% for under 1% of annual operating costs. They find positive economic benefit from pipeline renewal, even without CO_2 tax/price.

4.5.2 Impacts of CH_4 GWP adjustments

Balcombe et al. (2017, 2018) expand upon the shale gas literature analysis of Abrahams et al. (2015), and update the GWP 100 of CH_4 to 34 from 36. This adjustment of the literature increases the impact of observed leakages, as well as the effectiveness of mitigation interventions. However, carbon markets and international policy count the GWP 100 of CH_4 as 25, based on IPCC AR4 (IPCC AR4 WG1 and WG1, 2007). For example, the 220 MT CO_2e (using a GWP of 34) projected to be saved in the IMRP would be decreased by 44% to 123 MT CO_2e, but would not alter the effectiveness of mitigation on the atmosphere.

4.6 Conclusion

This chapter finds that the import and use of US-produced shale gas LNG in the UK would increase GHG emissions by 14.3 Mt CO_2e (19.2%) relative to the same quantity of domestic UK shale gas production and use. As discussed above, the losses in the distribution phase represent a highly uncertain, but important component of shale gas GHG intensity and are in need of further study. The author is not aware of any other papers assessing a comparison of US shale gas, transported as LNG, with domestically produced UK shale gas. Additionally, the impact of the pipeline renewal process is quantified and shows a positive implications for GHG emissions reduction. This is an early conclusion and the measurement and reporting of gas distribution renewal is in need of continued study and monitoring before claims can be made national carbon budgets to 2035.

Chapter 5 Discussion and conclusions

The work of this thesis examines the impact of gas usage in three distinct usage phases: UK electricity with EOR & CCS, UK electricity with shale gas, UK domestic heat production with UK shale gas or US shale LNG (see Figure 5-1). The three preceding results chapters find a common theme, that the UK is incentivized by its carbon budgets and targets to offshore emissions associated with natural gas supply.

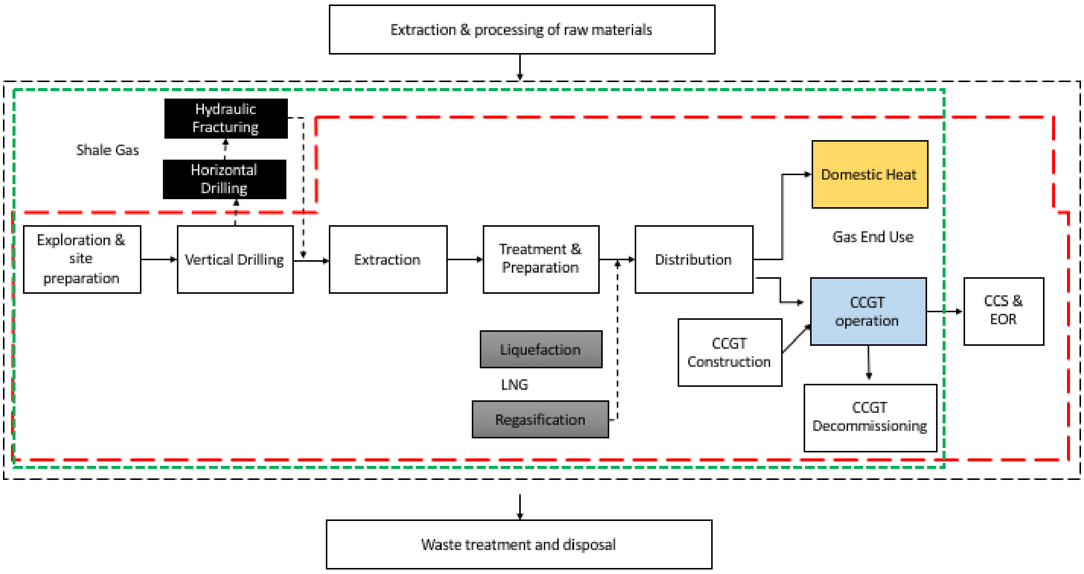

Figure 5-1 - Simplified shale gas schematic showing distinct phases in extraction, transportation, and end usage. Chapter 2 discusses the usage of gas for electricity aided by CCS & EOR. Chapter 3 discuss multiple sources of upstream gas, including shale gas in the UK power sector. Chapter 4 discusses the usage of shale gas from US and UK in the distribution phases for domestic heat. Figure adapted from (2014).

There are a multitude of choices for variables and inputs for each of the preceding chapters which were made in the review of the academic literature and creation of calculations in this thesis. Some were informed by government targets, such as the 100 g CO_2e kWh^{-1} electricity decarbonization goal (CCC, 2015b, 2014), or diminishing domestic gas supply (OGA, 2017, 2016; Turk et al., 2018a, 2018b). This is an area of research which is ever-evolving in public policy, industry, and academic literature. This chapter addresses some of the assumptions, and contextualizes the findings in policy recommendations, and suggests areas of future research.

5.1 A critical look at model assumptions

5.1.1 CO_2 supply and geographic uncertainty

One criticism of the assumptions in chapter 2 is the expected consistent CO_2 supply to gas power plants to store 50 Mt CO_2 yr^{-1} by 2030, without setting geographic sites or economic constraints. However, the approach of CO_2-EOR LCA carbon accounting is resilient in different scenarios of gas plant usages or CCS rollout. It doesn't actually matter if there are 1 or 11 gas plants with CCS, because the economics for 1 plant are strong with a modest oil price (EUR 50).

A second simplification of the modelling is the lack of extra CO_2 compression in the calculations. The modelling assumes the compressors would be needed anyway, for CO_2 to be transported to those sites for disposal in CCS. There could be a point, if CO_2 could be disposed in large tonnages into structures beneath the southern North Sea, that would use less input from compressors. This article discusses the additional CO_2 emissions incurred during CO_2-EOR operations offshore, which are not dependent on the offshore pipe.

For pipeline distances greater than 1,000 km, 6.5 kWh per tonne CO_2 is required for recompression (Jaramillo et al. 2009). This equates to 32,500 MWh per project year (with 5 Mt CO_2 yr^{-1} transported), or additional < 0.3% energy costs compared with 14,382,749 MWh. This energy cost would decrease with additional CO_2 -EOR of projects facilitating a network or pipelines (Welkenhuysen et al 2017), reuse of existing offshore pipelines (Pershad and Slater, 2007), and likely shorter pipeline distances (Jaramillo et al., 2009), further increasing the economic viability and geographic spread.

The precise geography of the storage sites is less impactful if these are on the east coast clustered or not - provided the CO_2 can get into a pipe network or shipping tanker to cross into the North Sea from Scotland, Teesside, or

Humber. Anywhere along the east coast from Hull northwards, the economics of CO_2-EOR cash and profit generation will be arithmetically powerful enough to attract CO_2 northwards, rather than paying for disposal in the Bunter Sandstone further south and east in the southern North Sea. These eastern sites should be given priority if the CO_2 originates from the UK (as opposed to Norway or Continental EU). As discussed above, the energy loss for recompression is minimal for distances less than 1,000 km, and would be further minimize with the economies of scale discussed in Chapter 2.

5.1.2 Carbon lock in

There is an inherent risk of carbon lockin and short-term emissions in the models presented.

The key advantage of the EOR-to-CCS model envisaged in Chapter 2 is that the EOR-oil helps to pay for CCS infrastructure to have long-term GHG savings. However, if produced EOR oil is not sufficient to cover costs for infrastructure, or there is disruption in the decades-long model, the advantage of CO_2 storage during the CCS phase could be lost. In this instance, the UK GHG budgets would experience the emissions associated with these projects without enjoying the GHG savings during the CCS phase.

In the shale gas scenarios examined, there is a risk that funding for new gas infrastructure would necessitate multiple decades to pay for financing. In this case, the short term emissions savings demonstrated the preceding models, could lock in long-term GHG emissions and jeopardize carbon budgets.

All of these scenarios are subject to global gas and oil prices volatility for dependable project funding. They also have exposure to reputational risks which could depress the price of oil and gas, and prevent the UK or operational partner(s) from funding or proceeding with these projects, or public opinion will not tolerate a limited gas or oil consumption.

5.1.3 Other gas sources

This thesis considers gas imports to the UK from several sources, however, does not consider gas from EU or Russia, as leakages rates are poorly characterized and difficult to quantify. In 2006, the IEA reported that 70 Bm^3 CH_4 leaked in the Russian gas system in 2004, roughly equivalent to one-third of their exports at the time (International Energy Agency, 2006). These leaks were estimated to be economically preventable for under \$10 t CO_2e^{-1} (Lechtenböhmer et al., 2007). More recently, the IEA reported that Russian methane emissions totalled 371.1 Mt CO_2e in 2011, 3.3% above 1990 levels (International Energy Agency, 2014). A future study on GHG emissions in the UK gas system should attempt to characterize the upstream emissions of gas from the EU and Russia.

5.1.4 Load factors and renewable support

The models in chapters 2 and 3 both assume high load factors for natural gas electricity supply. This is a simplistic view of the UK energy system and is based on past observations of natural gas power from DECC / BEIS (BEIS, 2017a, 2016a; DECC, 2015a). As the UK continues progress towards renewable energy generation (CCC, 2015b), it is possible that gas power will be used less for baseload, and more for renewable support and peaking power. The government data used in this thesis (BEIS, 2017e; DECC, 2015b) is reactive to past events, rather than predictive. Future iterations of the data will likely react to the increase in renewables, and update the load factors for UK gas power. This type of modelling is outside of this thesis, and an area in need of further study.

5.1.5 Reductions in possible UK shale recovery

The M4 model in Chapter 2 assumes a pervasive shale gas industry where all basins surveyed are exploited for shale gas. The ARI reports (Kuuskraa et al., 2013, 2011) represent the most ambitious estimates of recovery factors, ranging from 15 – 25% depending on clay content. Some basins are given a recovery factor of 10% with severe under pressure, or 30% when established performance is strong. Experience in Poland has shown that industry operators are not recovering shale gas with the same success as indicated in the ARI estimates (Inman, 2016). Industry-wide recovery rates are speculative at best for UK. As PEDL licenses are being sold and exploratory wells drilled, the estimates for basin size, recovery resources, and recovery rates are being refined (Andrews, 2013; Cuadrilla Resources, 2016; Kuuskraa et al., 2013).

ARI note that their 20% recovery rate is nearly twice as high as Cuadrilla's 2011 estimate of 10% recovery rate. However, this figure is incorrectly cited in the ARI report based on news releases (Bergin, 2011; Chazan, 2011) and preliminary resources estimates (Kuuskraa et al., 2011). The Institute of Directors suggested the recovery rate would be closer to 10% (Institute of Directors, 2013).

Regardless of the recovery rate, this thesis demonstrates that there is an pathway towards decarbonization which does not eliminate the UK's reliance on usage of natural gas in the next 20 years. However these pathways also export production emissions to other gas-producing countries in order to meet targets.

5.2 Mechanism for GHG audit trail in internationally traded gas

There is considerable discussion in the academic literature extolling the benefits of consumption-based (CB) accounting over traditional production-based (PB) accounting (Afionis et al., 2017; Bows and Barrett, 2010; Steininger et al., 2014, 2016). Such a mechanism would account for and close loopholes in GHG accounting across international borders.

5.2.1 Benefits of CB accounting

Afionis et al (2017) describe the threefold benefits of CB accounting described in the literature:

First, CB accounting will cover more global emissions by including the export sectors of developing countries of the global economy (Peters and Hertwich, 2008). After the Paris Agreement, this is a redundant argument in the literature, as all developing countries have already agreed to reduce emissions. However, CB accounting would provide an added measure of these emissions.

Second, CB accounting would incorporate aviation emission and bunkering of fuels, emissions which are not currently incorporated in PB accounting methodology. By assigning the embodied emissions to the consuming nation, the full impact of fuel consumption would be measured and accounted for.

Third, CB accounting would incorporate emissions in trade of most goods and off-shoring of production emissions. Traditional off-shoring occurs when a manufacturing base is moved to a developing area with less stringent environmental controls, while the consumption of the produced good remains the same. The net impact on GHG balances is worse, as developed countries are increasing their CB emissions faster than decreasing their PB emissions. This transfer has been occurring at an annual rate of 17% (Fischer, 2011; Peters et al., 2011).

In the case of shale gas, the US introduced a new sector with limited environmental controls (as discussed previously in Chapter 1). If the UK consumes US gas in substitute for domestic gas, the net effect is offshoring PB emissions from the UK to the US. CB accounting would place more emphasis on the impact of this transfer.

5.2.2 Challenges of CB accounting

Of course, a change in the GHG accounting system is not without challenges. While CB accounting would aid in closing gaps in GHG accounting, there are three main barriers which Afionis et al (2017) review in the literature:

First, the efficiency of a CB accounting market may be worse than that of PB accounting. Both styles of accounting have the same intention to reduce emissions, and provide transparency. PB accounting measures the GHGs in a given state, while CB accounting would require the import & export market to be more closely tracked. Jakob et al. (2014) posit that a switch in accounting will create a perverse incentive for less regulated states to export *more* goods and GHGs. If, for example, the US had zero intention of consuming shale gas, and relying on 100% renewables. The US could export as much shale gas as importers will buy and have zero emissions. A less severe example is the economic rebound effect which could occur when a more efficient system increases total consumption, albeit with lower intensities, rather than reducing consumption (Barrett et al., 2013)

Second, there are practical impediments to deploying a CB accounting system on a global scale. As Afionis et al (2017) point out, the PB accounting system was given favour by the UNFCCC not only for simplicity, but for the ability for countries to institute their own accounting compilations (Jakob et al., 2014). A CB system would require more complex modelling, most likely multi-regional input-output models (Turner et al., 2007). The UK government has tracked this method for some consumer goods and case studies (e.g. textiles), however it is not yet comprehensive of the UK economy (Barrett et al., 2013). This method would require practically all products to be registered with a central database tracking the carbon content. In effect, this would repeat the work of all carbon accounting to date.

Finally, the political incompatibility of CB accounting is perhaps the most difficult hurdle. Countries would be attributed with emissions that occur outside their borders. In examining the above two challenges, it is clear that pervasive data sharing would be needed along with cohesive action by all world governments in order to institute a comprehensive CB accounting system. This system has the impediment that a country does not have control of the emissions of a good it consumes (Afionis et al., 2017), and is beholden to the embodied emissions. This could be a political non-starter. However, this assumes that there is only one source for a particular good. This also has a straightforward solution; setting an appropriate global price on carbon with parallel disclosure of CB and PB emissions.

If a carbon price were appropriately applied it could have the co-benefit of more accurate measuring, reporting, and verification of emissions, although industries would need comprehensive regulation to comply. In the case of gas extraction and trade, suppliers could be perversely incentivized to hide emissions to avoid tax. If suppliers were required to report CB and PB emissions in tandem, the tax shelter could be exposed, or at least more clearly tracked and verified. There is also a timing issue, similar to the perverse incentive for UK coal from The Energy Act 2013 (see Chapter 1), that prior to the enacting a price on carbon, a dash for unregulated gas could occur. The UK could mitigate this impact by incorporating both CB and PB emissions into a cap, similar to a value chain assessment.

5.3 Applications of consumption-based accounting to shale gas emissions accounting gaps

As described in Chapters 2, 3, & 4, the UK will continue to consume natural gas through 2035 while relying in greater percentage on imports. And as described above, the current accounting systems do not discourage the UK from offshoring emissions to the US for gas. However, setting a price on GHGs could help create a global cap on emissions, and equitable consumption (Boyce, 2018).

Based on the analysis of each of the preceding results chapters, the GHG emissions from chapter 4 are most realistic in the long-term because they do not rely upon power generation policy changes, or CCS / EOR funding. Instead they are indicative of gas demand for UK residential use for the next 15 years (2020 – 2035), which will take considerable investment to change course (BEIS, 2017d). As discussed previously in chapter 4, this 220 Bm^3 could be produced in a new UK shale gas industry, or imported from the US as shale LNG.

5.3.1 Social costs of carbon and carbon taxes

One of the central inefficiencies of the carbon markets and the UNFCCC reporting regime is the delay from scientific findings of global warming potential until inclusion in carbon markets. For example the AR4 value of CH_4 – CO_2e is 25, and was established in 2007 and is still accepted in carbon markets, despite an update to 28 in AR5 (IPCC, 2013b; IPCC AR4 WG1 and WG1, 2007; IPCC TAR SYR and SYR, 2001). Howarth et al. (2011) advocated for 20-year CH_4 GWP of 86, due to the urgency of climate change. Other studies advocate for AR5 values of 34 or 36 depending on CH_4-climate feedback loops (Abrahams et al., 2015; Balcombe et al., 2017). Changes in these values could cause price shocks in the carbon markets, and delays in price updates don't reflect the true scientific understanding of the GWP of GHGs.

If the UK were to adopt a carbon price during the time when the energy system is importing or producing more gas, there would be significant carbon price implications. For example, the 220 Bm^3 discussed above and in Chapter 4 represents the projected supply gap for residential usage from 2020 – 2035. If it is assumed that this gas is 0.554 kg m^{-3}, it represents 12,188 t CH_4. Assuming that 1% of the gas leaks (and is not combusted to CO_2), it would emit 33,181 t CO_2 and leak and additional 121.9 t CH_4. Applying a GWP from 21 (AR3) to 86 (20-year GWP) creates a range of 35741 – 43,663 t CO_2e before adding a price.

The most recent quarterly high for the EU ETS was EUR 28, roughly $24 per t CO_2e. When applying a range of price from recent EU ETS trades ($23, $28), recommendations by Carbon Tracker ($32), up to a social cost of $233 recommended by Moore and Diaz (2015), the range of carbon costs are $822,000,000 – $9,737,000,000 (see Table 5-1) (Carbon Tracker, 2018; GmbH, 2018; Moore and Diaz, 2015). This is with just a modest 1% leakage rate across the entire gas supply chain. Increasing the leakage rate to 5% increases the range to $1,071,000,000 - $18,788,000,000 showing a disparate estimate of potential impacts depending on the GWP when including CH_4 in a carbon price.

5.3.2 Science-based GWPs in the market

While the 20-year CH_4 GWP is more closely linked to the short lifespan of atmospheric CH_4, carbon markets have resisted taking on this approach. One reason would be the immediate price shock could collapse the carbon markets and jeopardize ambition to decarbonize. As an intermediary solution, linking the GWP to the scientific literature on a more frequent basis would decrease the gap between market prices of GHGs (Current GWPs), and scientific impacts. On the other hand, the short-lived timespan of CH_4 in the atmosphere causes inherent transparency problems with converting to a 100 year GWP for carbon markets. The potentially high volatility of CH_4 GWP conversions (shown below in Table 5-1) illustrates the need for 2 metrics: CO_2e and a CH_4 budget. This transparency would limit the hiding of CH_4 emissions with older GWP values, and show the true budget of remaining GHGs to stay under +2°C.

Table 5-1 -Carbon prices of CH_4 combustion with 1% leakage rates for 220 Bm^{3}. Prices are based on (Carbon Tracker, 2018; GmbH, 2018; Moore and Diaz, 2015).

GWP	21	25	34	36	86
1,000's$ / t CO_2e	35741.31 t CO_2e	36228.83 t CO_2e	37325.75 t CO_2e	37569.51 t CO_2e	43663.51 t CO_2e
23	$822,050	$833,263	$858,492	$864,099	$1,004,261
28	$1,000,757	$1,014,407	$1,045,121	$1,051,946	$1,222,578
32	$1,143,722	$1,159,323	$1,194,424	$1,202,224	$1,397,232
83	$2,966,529	$3,006,993	$3,098,037	$3,118,269	$3,624,071
100	$3,574,131	$3,622,883	$3,732,575	$3,756,951	$4,366,351
223	$7,970,312	$8,079,029	$8,323,642	$8,378,001	$9,736,963

Compatibility of fossil fuel energy system in the UK for climate targets.

5.4 Key findings

- **20 years of CO_2 storage can be subsidized by EOR**, can support UK carbon targets while supplying up to 11% of projected UK oil demand at peak production. The average electrical output could supply nearly one-third of domestic electricity supply while lower current grid emissions and supporting UK grid decarbonization targets.
- **The UK can support a shale gas regime to decarbonize if the total fugitive emissions remains under 1%.** If the projected domestic gas supply gap for power generation (without CCS) were to be met by UK shale gas with low fugitive emissions (0.08%), an additional 20.4 Mt CO_2e would need to be accommodated during carbon budget periods 3 – 6. However, the UK carbon budgets would benefit from importing Qatari LNG rather than producing the same quantity domestically. With a modest increase in fugitive emissions would exceed UK carbon budgets.
- **A modest increase in UK fugitive gas emissions would exceed UK carbon budgets.** The same carbon budgets would not benefit from importing UK shale LNG for residential usage, rather than producing the same quantity of gas. However, a modest increase in UK fugitive emissions rates would also break exceed carbon budgets, along with adding significant GHG emissions outside the UK.

5.5 Recommendations

To facilitate improvements in emission reporting and compliance with the Paris Agreement, the following recommendations should be considered. These will allow for improved emissions reporting transparency, impact measurements, and expedited monitoring of GHG impacts of international gas trade. These recommendations are analogous to the switch to "science-based targets" (SBTs) in CDP reporting, where intentions and actions are directly reported against carbon budget implications.

As shown in Chapter 1, the phase out of coal created a brief perverse incentive to import and burn coal for 2 years prior to the expiration of the exemption of coal. If the UK carbon budgets are entirely focused on UK PB emissions, this incentivizes transfers of the emissions offshore where the UK has less control. Therefore, report CB emissions alongside PB emissions to flag perverse incentives and provide transparency in the origin of GHG emissions.

5.5.1 Report environmental flows alongside GHG accounting

It is recommended to report CB accounting figures alongside the traditional PB accounting figures accepted by UNFCCC. This reporting mechanism would inform border carbon adjustments and considering the dwindling global GHG budget, acceptance of the Paris agreement -including American Fortune 500 companies and cities – there is global consensus to limit emissions to +2°C. However, the reliance on PB accounting facilitates emissions leakages, transfers, and devaluations of GHG costs. As shown previously in Chapters 2 and 3, decarbonization intentions may not perfectly align with national interests for energy security. A CB reporting mechanism will put greater pressure on gas purchasers to consider lower-GHG fuels.

This is similar to Scope 3 emissions reporting in product footprinting, where a company is not necessarily responsible for a partner's Scope 1 and 2 emissions, but can put pressure on that partner to reduce their own GHG emissions.

As discussed above, CB accounting may be too difficult to establish as a replacement for PB accounting. However, there is already precedence in the UK government for CB accounting in some industries and goods (Barrett et al., 2013). This methodology should be extended to the entire UK economy.

There are political implications of CB accounting which may hinder the ability to fully measure environmental flows. The CB system would require the UK have regulation over foreign energy systems. A solution for this would be a *border carbon adjustment* (BCA) and an *internal price* on carbon and energy-intense goods. This would create the long-term incentive for lower carbon products.

5.5.2 Intermediary GWP market adjustments

It is recommended that GWPs are reviewed more frequently, and implemented to markets akin to Science-Based Targets (SBTs) advocated by CDP to alleviate lag between the reporting of CH_4 and N_2O, and CO_2e conversion factors and market acceptance. This delay allows for carbon leakages, both intentional and unintentional.

5.5.3 Report individual GHGs separately

The short-lived timespan of CH_4 in the atmosphere causes inherent transparency problems with converting to a 100 year GWP for carbon markets. The potentially high volatility of CH_4 GWP conversions (shown in Table 5-1) illustrates the need for 2 metrics: CO_2e and a CH_4 budget. This transparency would limit the hiding of CH_4 (and other GHG) emissions with older GWP values, expose organizations utilising unforeseen perverse incentives, and show the true budget of remaining GHGs to stay under +2°C.

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