The Greenhouse Gas Impact of Shale Gas Exploitation

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Master of Science by Research

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2016

Abstract

Shale gas is a hydrocarbon in impermeable shale rock up to 3.5km below the earth surface, requiring hydraulic fracturing, an 'unconventional' technique to stimulate production. This study sought to explore the accurate greenhouse gas impact of shale gas exploitation. Emissions from a producing conventional well pad are analogous to an unconventional well pad in production. Methane emissions were measured at three key sites; the KM5 conventional well pad, Knapton generation station and a rural crop farm control site, Cranford Farm. 288 hours of methane measurements were conducted using a Picarro Ring Down Spectrometer, which recorded methane concentration at 1 hertz. A Gull wind sonic was also used to record wind speed and direction. Therefore a flux could be calculated for two natural gas infrastructure sites for comparison. The control site had an average methane concentration of 1.93 ppm, which was higher than the 1.91 ppm CH_4 average at the KM5 well pad but lower than that of Knapton which was 2.01 ppm. The average calculated CH_4 flux at KM5 was 24.29 kg CH_4/year, 3% of the average flux from Knapton, which was calculated as 847.91 kg CH_4/year. These annual methane emissions are the equivalent of 3 sheep and 7 dairy cows respectively. Under highest case calculations, methane emissions from these sites equate to emission factors of 0.0077% for KM5 and 0.082% for Knapton. From literature analysis, pre-production emissions from shale gas exploitation will be higher from a) excess energy required for horizontal drilling and hydraulic fracturing and b) methane emissions from well completion. The extent to which it has a larger environmental impact than conventional gas is dependent on the use of REC (reduced emissions completions), which can reduce well completion methane emissions by over 90%. Over longer terms, the Climate Change Act (2008) requires an 80% reduction in GHG emissions from 1990 levels, which will require CCS implementation nationally regardless of shale gas exploitation.

Declaration:

I confirm that no part of the material presented in this thesis has previously been submitted by me or by any other person for a degree in this or any other university. In all cases material from the work of others has been acknowledged.

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Acknowledgements

This research was carried out as part of the ReFine (Researching Fracking in Europe) consortium led by Durham University and Newcastle University and has been funded by the Natural Environment Research Council (UK), Shell, Chevron, GDF Suez, Ineos, Centrica, and the environment agency.

I would like to thank Third Energy for permitting access to two of their sites; the KM5 well pad and to Knapton generation station for data collection. Also, thank you to the Houlston family of Ryedale, who permitted the use of their land as a control site for methane data collection, and also for their assistance when our vehicle was incapacitated.

I would like to thank Dr. Ian Boothroyd and Mr Sam Almond for accompanying me on data collection journeys to North Yorkshire.

A final thank you is to Professor Fred Worrall who as my thesis supervisor has provided a rigorous research design platform upon which my thesis was built and has provided highly intellectual analysis throughout this research project.

Contents

Figure List

Chapter 4:

Table List

Chapter 1:

Chapter 3:

Chapter 4:

Abbreviations

ALARP: As low as reasonably practical

BCM: Billion cubic metres

Bridge Fuel: Depicting a fuel which assists in the transition from carbon intensive energy production to a low carbon scenario.

BTU: British Thermal Unit

CCGS: Combined Cycle Gas system

CCS: Carbon Capture and Storage

CO_{2e}: Carbon dioxide equivalent- comparative unit for all greenhouse gas emissions

EGDAR: Emission Database for Global Atmospheric Research

EUR: Natural gas well production figure per year

EF: Emission factor- The % of total natural gas gross production which is emitted as leakage

GHG: Greenhouse Gas

Gt: Gigaton

GWP: Global warming potential

IPCC: Intergovernmental Panel on Climate Change

$kWh_{(e)}$: Electricity generation

LNG: Liquified Natural Gas

OECD: Organisation for Economic- Cooperation and Development

PPM: Parts per Million (measure of concentration)

REC: Reduced emissions completions

tWh: Terawatt hour

TWP: Technology warming potential

WTW: Well to wire

1.0: Introduction

1.1: Unconventional Hydrocarbon production:

Of the 121 million Gt of carbon stored at the near surface environment, approximately 70 % (78 million Gtc) is stored as fossil fuels and other sedimentary based rocks (Berner, 2002). Of the other 30%, 25% is stored in the deep oceans, and the remaining 5% exists as carbon in the upper oceans and on land as organic matter. Less than 1% of the carbon budget (750 Gtc) exists in the atmosphere as a gas (Berner, 2002). Terrestrial organic matter is produced by the transfer of energy from an inorganic form, sunlight, into an organic sugar using the photosynthetic pathway provided by the pigment chlorophyll (Berner, 2002). This material can be transported to the oceans where it combines with animals and plants from a variety of marine habitats, which following death, decompose and form part of the seabed (Berner, 2002). Over time, this layer of fine-grained sediment rich in organic material is compressed by the combined weight of the ocean and subsequent sediment (Mackay and Stone, 2013). Compression over geological timescales transforms the sediment into a layer of finely laminated shale rock, rich in organic material up to 3.5km thick (Hilton et al. 2011). Earth's highly variable topography with a wide stretching network of rivers has provided an abundance of organic matter and fine grained sediment for transport, which explains why shale makes up 35% of rocks at the Earth's surface (Stephenson, 2011). This makes shale the globally most abundant sedimentary rock. However, shale at the surface is exposed to oxygen and is weathered, so the majority of the organic matter has been degraded and potentially transported away (Stephenson, 2011).

The organic matter locked up in deep shale is in an anoxic environment, so further decomposition on a large scale is impossible. Instead, the increase in burial subjects it to more pressure and heat, with temperature increasing by 25 °C per 1km of depth (Stephenson, 2011). The organic material is heated (to variable extents) and forms both gas and oil in the shale. The temperature required to produce shale oil is generally between 60-90 °C, and gas is produced at 90-120°C (Stephenson, 2011). If there has not been an adequate burial of the shale, where oil and gas has not formed, the shale is known as immature, and in cases where it has undergone deep burial, excess temperatures destroy the oil and gas, in what is known as an over mature shale (Stephenson, 2011). In cases where this shale has been fractured naturally, the oil and gas produced can flow from this source rock up to the highly

permeable sandstone or limestone, where it is held in place by an impermeable 'cap' (Mackay and Stone, 2013). Where this migration has occurred it is known as the oil and gas reservoir and hydrocarbons can be extracted conventionally. By contrast the shale layer is impermeable, unlike sandstone, which means that large amounts of oil and gas can be locked in place in the shale, requiring stimulation by fracturing to access the resource (Allen et al. 2013). The exact definition of an unconventional resource is 'one which requires methods of extraction which are not conventional', but historically these shale layers were not productive and only regarded to be useful as the petroleum baring formations as the source rock of conventional reservoirs (Allen et al., 2014).

1.1.2: Unconventional resources

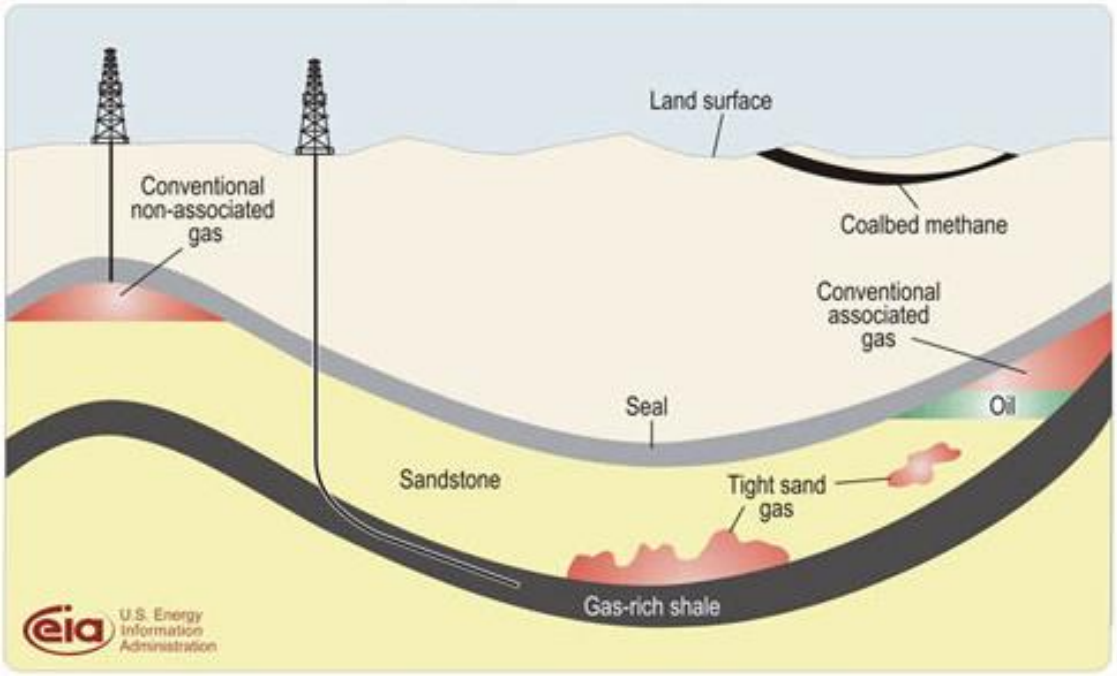

Figure 1.1: EIA Schematic of conventional and conventional hydrocarbon resources

There are also other unconventional sources of hydrocarbons which are of interest, but beyond the scope this study's research ability. The US EIA (2014) describe 3 other unconventional hydrocarbon sources (Figure 1.1);

1) Tight gas/oil

Tight gas and oil are unconventional sources of hydrocarbons stored predominantly in the reservoir rock sandstone, but can occur in limestone. The low permeability

and porosity of the lithology makes it an extremely similar to natural gas produced in shale layers.

2) Coal bed methane

Coal bed methane is an unconventional source of natural gas which has been adsorbed into the coal formation, and exists in a near liquid state in the coal matrix. It is typically called sweet gas due to H2S lack and contains few heavier hydrocarbons.

3) Shale Oil

Shale Oil is unconventional oil produced in the shale layer. Oil shale (kerogen) within the rock undergoes pyrolysis converting the organic material into oil. The kerogen can also be extracted and undergo thermal dissolution to produce what is preferred to be known as 'tight oil'. This just means crude oil refined from oil baring shales and was done to prevent confusion with crude produced from oil bearing shales. In the majority of cases, these hydrocarbon sources require hydraulic fracturing to stimulate the rock.

There can be large variability in the environmental impacts of hydraulically fracturing the shale layer or tight gas sands in comparison to stimulating the shallow coal bed methane arising from flow rates, well design and proximity to water aquifers (Allen, 2014).

1.2: Hydraulic Fracturing of Unconventional Hydrocarbons

Hydraulic fracturing, commonly known as fracking, is a method of natural gas extraction utilising high pressure water, sand and chemicals to stimulate the desired geology (Allen et al. 2013). This process was invented in 1947 by the Studolind energy company, in which they forced pressurised gelled gasoline and sand into the limestone formation of the Hugoton gas field, Kansas (Hubbert, 1962). This test was not particularly successful; however it was the first step to create the technology which exists today. Over the following 50 years, industry, in combination with large US government investment and research, has led to a surge in natural gas production from shale in the USA and it has awakened global interest in this previously disregarded technology and resource (Allen et al. 2013). An unconventional well is drilled in a similar process to a conventional well. On a well pad, the well is drilled to the geological depths of the desired source rock, such as

tight oil in sandstone, coal bed methane at the sub surface, or the deeper shale rock up to 3km below the surface (Howarth et al. 2011). The advent of horizontal drilling, and its combination with hydraulic fracturing have been the two most advantageous developments in unconventional gas production and is what separates conventional and unconventional production stages (US EIA 2010).

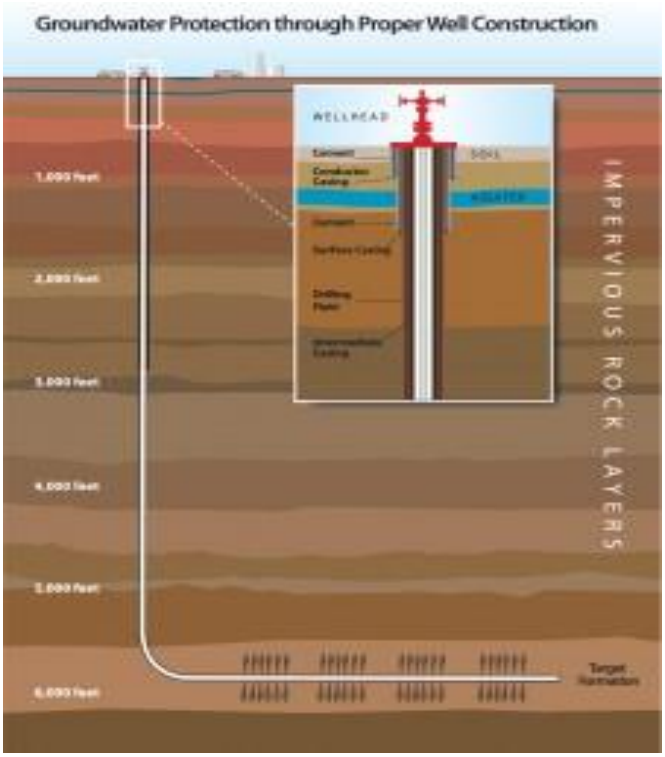

Figure 1.2: Schematic of Hydraulic fracturing of Shale from the New Mexico Oil and Gas Association.

Shale has wells are drilled up to 3500 metres deep and up to 2500 metres horizontally.

1.2.1: Life Cycle process of shale gas extraction

The production process is outlined below, adapted from Mackay and Stone, (2013)

A) Pre-production: 5-23 months

1**)Exploration**: This historically was conducted by trial and error drilling methods, however rapid technological developments have improved 3-D seismology analyses which through non-intrusive measurements assist in assessing economic viabilities of hydrocarbon reserves.

2) **Site Preparation**: The site will be removed of obstacles such as vegetation, followed by the construction of a well pad along with any other necessary infrastructure to alleviate transport and security issues.

3)**Drilling and Casing**: The well is drilled to depths of up to 3km to breach into the shale layer. At the well 'heel', the drill is tilted, ultimately running

parallel to the shale layer for distances of up to 5km. Casing is continually added in layers to prevent leakage from the well into shallower geology.

4) **Hydraulic Fracturing:** A perforation gun is moved systematically along the well, creating an interface between the well bore and shale rock. Sand and chemicals are mixed with 8,000 to 80,000 m^3 of water and pumped down at high pressure (70,000-140,000 KPa) to fracture the deep shale. Sand is used as a proppant to ensure the fractures remain open which allows the gas to flow continually. Otherwise the pressure exerted by gravitational mass of the shallower geology forces the fractures to close. The chemicals involved have a wide range of uses including as biocides and corrosion inhibitors, but they fundamentally are used to improve recovery rates (EUR). This is conducted in individual isolated segments of the horizontal section using isolation plugs to maximise the water pressure per area of shale exposed and hence increase fracture propagation, which can be as great as 568 m (Davies et al. 2012). Technological advancements in the past decade have resulted in the ability to drill up to 10 wells per pad, increasing the well pad production while minimising surface land use; however initial costs vary depending on conditions from geology to legal requirements (Figure 1.3) (US EIA).

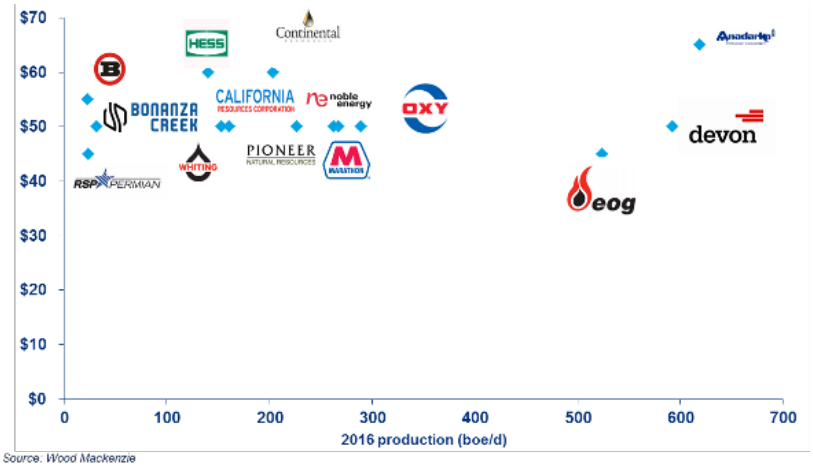

Figure 1.3: Price requirements for shale gas production to be profitable (Wood MacKenzie, 2014)

The recent decline in oil and gas prices has left some formations unprofitable

5) **Well completion and drill out:** Post fracturing, 20-50% of the fracturing fluid is returned to the surface over 3-10 days as produced water. During this time some methane can be released but green completions can reduce fugitive emission rates by over 90% (Howarth et al. 2011). The isolation plugs are then 'drilled out', and the well is depressurised to allow the flow of shale gas to the surface.

6) **Waste treatment:** Once the flowback fluid or produced water flows to the surface, it is transported to waste treatment centres for processing to remove hazardous chemicals and naturally occurring radioactive material, before being returned to the water course or disposed of appropriately.

B) Production phase: 5-40 years

The gas produced from US Shales varies in its composition. In the Haynesville and Fayetteville formations, they are 95% and 97% methane respectively and so require minimal if any treatment before being sold to market (Mackay and Stone, 2013). By contrast, the Antrim Shale formation is only 62% methane by dry bulk and the Barnett Shale, which is geologically most similar to the Bowland Shale is 85% methane, with 11% ethane and 3% as propane (Mackay and Stone, 2013). Natural gas produced typically requires processing before being separated and sold. Halliburton, a US oilfield service company, state that gas production from shale wells is initially very high, followed by a hyperbolic decline over short time periods and an extended exponential tail decline (Figure 1.4)

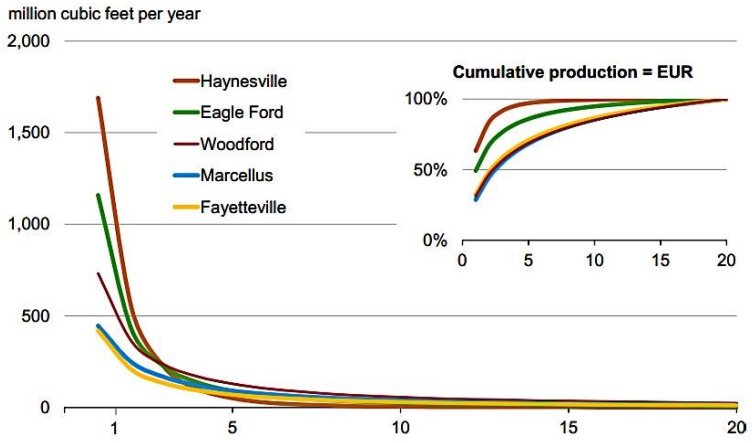

Figure 1.4: EIA-Shale gas well productivity: Reflects the steep EUR decline of US shale wells

The US Geological survey reports that ultimate recovery for basins ranges from 0.001 to 0.07 BCM (Billion m^3) per well. Collapse in gas prices has significantly impacted shale production in the US, and in the UK extraction costs are likely to be higher, hence making low EUR wells not viable (US EIA). This has focused attention on shale play zones where EUR could be in excess of 0.14 BCM per well pad.

As gas productivity decreases during the life of the well, workovers are conducted whereby the well is 'refractured' to re-stimulate the shale rock to increase flow rates (Mackay and Stone, 2013). Skone (2011) and Hultman et al. (2011) both assume that each well will be worked over at least once and would have emissions complementary to that of well completion, so emissions are therefore dependant again on practice standards.

C) Post-Production Phase: Once the well is deemed financially unviable the well can be plugged and or decommissioned (abandoned). This is done to prevent fugitive emissions from the well post production (Boothroyd et al. 2015). In the USA, wells have been temporarily plugged during this period of extremely low oil and gas prices with the aim of restarting production when prices exceed costs (Miller et al. 2013). The price at which companies can be profitable varies across the nation, as shown in figure 1.3 (IEA 2016)

1.2.2: The US Shale Revolution

The 'Shale Revolution' was the expansion of shale oil and gas production onto the US market. In 2001, shale gas represented less than 2% of US domestic production, and now represents 52% (EPA 2015). In 2000, less than 0.02 BCM per day was being produced from shale plays in the US, and by 2012 0.7 BCM was being produced onshore domestically per day (US EIA). Current US production equates to 396 BCM per year, which is 140% of peak North Sea gas production in 2001 (Mackay and Stone, 2013). It is and has been one of the largest energy revolutions in US history, providing jobs and has been publicised as a bridge fuel to allow the decrease of coal use and providing national energy security (Miller et al. 2013).

Figure 1.5: Map of USA Shale plays (EIA, 2012)

The largest shale formations in the USA are the Marcellus, Barnett, Haynesville and Bakken. The total recoverable resource is estimated at most to 17 627 BCM, more than all of Saudi Arabia (Miller et al 2013). Production has actually been so surprising that Cheniere Energy, a major US natural gas producer, converted its import LNG terminals in Louisiana, Texas and East Coast USA to become facilities ready to export gas produced from unconventional sources across America (Broderick et al. 2011).

Research into enhancing gas recovery has also been lucrative. In January 2007 across the Marcellus shale, average gas production per well pad was less than 0.03 BCM/day, and by January 2015 this had increased 10 fold to over 0.32 BCM/day (EPA, 2015).

The rationale for fracking's surge is incredibly complex and interconnected with global politics and economic goals. Oil embargos from the 1970's brought questions to the table as to the legitimacy of this finite product for energy production, not assisted by the 'peak oil' theory, although this was controversial at the time (Caulton et al. 2014). The nationalisation of major Middle Eastern petroleum industries meant then that oil and gas would feasibly be at the mercy of political relations between the importer and exporter. The necessity to secure energy security has been even more complicated as the energy cartel OPEC was agreeing production limits to set prices and Putin's flexing of muscles by shutting off eastern

and central Europe's gas supply in the late 1990's fed further paranoia (Finon, 2007).

The impact of shale gas exploitation has had both national and international implications. The rapid production of large volumes of natural gas decreased the Henry Hub price in the US by 60% from $5.50 in 2005 to $2.07 by 2016, and this domestic production decreased imports and reliance on OPEC (US EIA). The response from OPEC was to ramp up production to create a disincentive for the US shale producers and the end result has been a global oil glut causing a plummeting of oil prices to a 40 year low at $25.00 per barrel (BP statistical review, 2016). However technological innovation among other factors proved the fracking industry to be more resilient than expected, for example even though US drill rig numbers decreased, national production was able to remain high (EPA 2015).

1.2.3: UK Shale Industry

Natural gas usage in the U.K. is ~80 BCM / year (DECC, 2015). From 1975-2000, North Sea hydrocarbon extraction resulted in the U.K. having to import less than 20% of its petroleum products, however dwindling resources have resulted in UK oil and gas imports hitting a 32 year high in 2016 (OGA, 2015). Nevertheless, natural gas has remained a central section of the UK's energy supply, generating 55% of UK electricity and over 75% of residential heating, with the majority of imported gas directly piped from Norway and as LNG from the OPEC member gulf state, Qatar (Broderick et al. 2011). UK shale gas resources have been estimated to exceed 45 000 BCM in the Weald and Bowland basins, however as current technology maximises the estimated ultimate recovery (EUR) at 22% of resource, this majority of this is unlikely to be extracted under this technological standard of practice (BGS, 2012). The Weald basin in southern England hosts the upper Jurassic Kimmeridge shale, which is the source rock of the North Sea and has estimated onshore resources of 100 billion barrels (DECC, 2014). Increasing research into true reserve quantification and fracking technology can maximise the EUR (Recovery rate per well).

Hydraulic Fracturing for natural gas is currently not a technology being deployed in the UK. The only fractured well was at Preece Hall, Lancashire, drilled in 2011, and was decommissioned by 2014. The frack fluid lubricated a fault following fracking, which caused the fault to slip, inducing a seismic event and ultimately shut down the entire site (Green et al. 2012). There are therefore no opportunities for primary

data collection of greenhouse gas measurements from shale gas extraction sites in the UK. Regardless, even in the USA where shale gas produced from fracking is largest in the world, at 350 BCM in 2014, there is a lack of representative primary data available (Howarth et al. 2011).

The lack of primary data for the UK discombobulates policy decisions and does not allow for accurate greenhouse gas emission constraints for UK shale gas exploitation. This study has sought to measure on site methane emissions, as was conducted in North Yorkshire. This investigation is a UK example of greenhouse gas emissions measurements from an onshore well pad which supplies its gas directly to a local power generation station, allowing a comprehensive analysis of upstream fugitive emissions in a closed system.

1.3: Greenhouse Gas emissions from Shale Gas Exploitation:

There is now beyond overwhelming evidence that human induced emissions have accelerated and distorted natural climate change. The 'hockey stick' pattern shown in figure 1.6 presents that current CO_2 and CH_4 levels have increased at an exponential rate over the past 150 years and are the highest levels recorded over the past 200,000 years (IPCC, 2013).

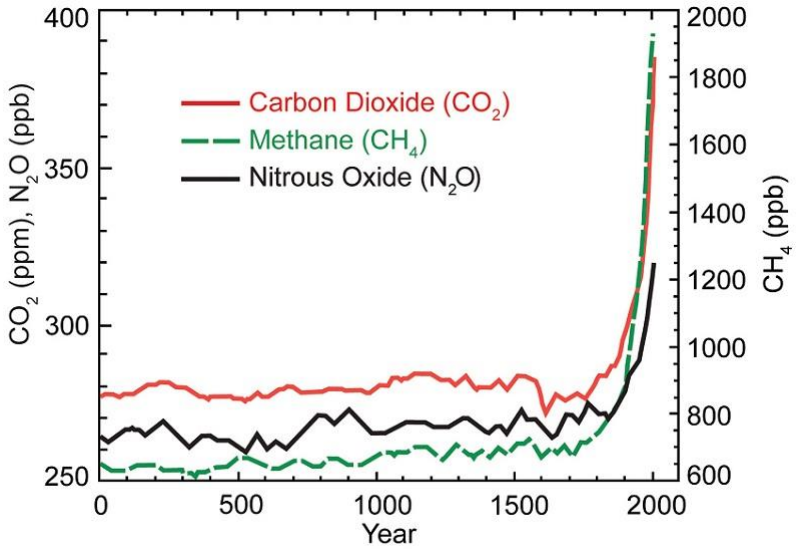

Figure 1.6: The change in atmospheric CO_2, CH_4 and N_2O emissions over 2000 years

Source: IPCC

Atmospheric greenhouse gas abundance is now the highest in human history

CO_2 is the dominant greenhouse gas in terms of warming potential. However, CH_4 is a more potent greenhouse gas over decadal scales, with a greenhouse gas warming potential (GWP) 25 times that of CO_2 over 100 year timescales, and up to 100 times that over 25 year timescales when the indirect effects of methane interaction in the atmosphere is taken into account (EPA 2013). Therefore, per cubic metre, this potent gas traps more heat than CO_2 however its residence time

in the atmosphere is shorter than CO_2 by up to 10 times (EPA 2013). In 1996, the EPA released a report on national GHG inventories where by its own admission 'potentially underestimated some emission sources' but noted that it was 'at a time when methane emissions were not of significant concern in the greenhouse gas debate'. As an upshot, the 2010 EPA report showed methane emissions to be higher from certain areas of the USA, and this has been attributed to the expansion of natural gas extraction and animal agriculture as well as improvements in measurement techniques.

1.3.1: The issue of greenhouse gas emissions

Using natural gas from conventional wells has been scientifically regarded as a much cleaner alternative to coal for 4 reasons outlined by de Gouw et al. (2014).

Per kWh, gas produces;

a) 44% less SO_2, which causes acid rain and is a risk to human health
b) 40% less NO_x which causes ground level smog, increasing the risk of chronic respiratory illnesses.
c) 80% less Particulate matter, which can be ½ the size of a human hair and is a globally recognised factor in chronic bronchitis and asthma.
d) 90% less Mercury, coal plants are responsible for more than half of anthropogenic Hg emissions.

Yet, with unconventional gas production, there have been investigations into reports of large methane leaks from natural gas systems which would negate the benefit of switching from coal to gas in terms of greenhouse gas emissions (Howarth et al, 2011). The extent to which methane leakage impacts its justification as a bridge fuel is one of the central concepts upon which this study will focus. Critically, it is understood that if the carbon stored as fossil fuels within the earth is combusted indefinitely without mitigation, the impact of catastrophic climate change is inevitable (IPCC 2013).

Methane emissions can come from upstream sources such as production emissions and leakage or downstream sources, such as pipelines or end user leakage. Up and until 2010, the upstream methane leakage rate for shale gas and conventional gas was regarded to be the same at 0.2 % and downstream emissions at 0.9%, giving an overall total of 1.1% (EPA 1996; EPA, 2011). The EPA (2009) used emission factors from 1996 in which shale gas was not mentioned as it had not been

implemented on a national scale. The EPA (2011) increased their estimates for conventional gas to 1.6% and introduced that shale gas would have a higher methane emission factor at 3% while downstream emissions remain at 0.9%, meaning leakage was dominated from upstream sources (EPA 2011). In 2013 the EPA reduced their emission factor for the upstream natural gas sector (both shale and conventional) to 0.88%, for a total of 1.78% (Howarth et al. 2014). The 2013 shale gas emission factor reduction was questioned by Karion et al. (2013) who claimed the justification for the reduction came from an industry report which argued that the methane emissions during flowback, well completion and refracturing were greatly overstated (Shires et al. 2012). As this suggests it is extremely difficult to measure, quantify and assess the provenance of natural gas emissions on a national scale with variable state regulation.

1.3.2: Direct emissions from Shale gas extraction and combustion

Diesel and or petrol generators are used on well pads as hydraulic pumps and to provide electricity to the site. The drilling rig is powered by a 'prime mover'. The size and power of the prime mover is dependent on the extent of drilling with shallow rigs only requiring 500 HP motors while 3,000 HP motors are required for depths of 6,000m (Natural gas.org). The emissions per site will therefore depend on the depth to which drilling takes place and the number of wells drilled per site. Estimated emissions from these are shown in table 1.1, with both authors concluding that site preparation emissions are negligible. Jiang et al. (2011) and Santoro et al. (2011) have suggested using gas produced by their shale plays to power gas fired generators to be used on site, but there is no evidence to suggest this has been implemented in the US.

Source	Emission Estimate (tCO_{2e} /well)	Note
Jiang et al. (2011)	**300-360**	Site Preparation, excluding drill rig transportation
Santoro et al. (2011)	**158**	Site Preparation, excluding drill rig transportation
NYSDEC (2011)	**15**	Transportation of drilling rig

Table 1.1: Stage 1 Pre-production direct emissions from Shale gas exploitation

The CO_{2e} unit can encompass all greenhouse gases. As the warming potential of CH_4, NO_x and CFC's is not complementary to CO_2, these other greenhouse gases are expressed in CO_2 equivalent to allow for direct comparison between regional, national and international emissions to constrain the cumulative cocktail of greenhouse gas emissions in the atmosphere.

The direct combustion of natural gas produces up to less than half the CO_2 of coal per BTU (British thermal unit). However, substantial methane leaks, as suggested by Howarth et al. (2011) and Sullivan et al. (2012) have complicated the benefits of this conversion.

Table 1.2 shown below presents best data assumptions of energy requirements and direct emissions from shale gas production. Estimates under Broderick et al. (2011) are lower as they only take into account additional energy use on top of the amount required for conventional production. The range of emissions from shale gas wells therefore is 656-1790 tCO_{2e} /well. The variation around this range, as described, is a function of the depth of desired geology and the length of time it takes to get from site preparation to having a producing well.

Source	Emission Estimate (tCO_{2e} /well)	Note
Jiang et al. (2011)	610 -1100	Drilling
	230-690	Fracking
	840-1790	**Total**
Santoro et al. (2011)	**1426**	Drilling and Fracking
Stephenson et al. (2011)	**711**	Drilling and Fracking
Broderick et al. (2011)	49-74	Drilling
	295	Fracking
	344-369	**Total**
NYSDEC (2011)	277	Drilling
	379	Fracking
	656	**Total**

Table 1.2: Stage 2 Pre-production emission estimates from shale gas exploitation

1.3.3: Fugitive emissions from Shale gas extraction:

Fugitive emissions are unintended leaks from natural gas infrastructure. In general, the higher the quality of the equipment the lower the leakage rate of methane, however regulation and standard of practice are becoming equally as important (US EIA, 2014). Apart from well completion (which will be discussed later), there are three main sources of fugitive emissions as described by Howarth et al (2011);

1) **Venting and equipment leaks**

   Once the well is producing and supplying gas, the 55-150 connections typical of any natural gas well will leak to some extent, including from heaters, dehydrators and compressors which are unavoidable, but can be limited with good practice (Cathles et al. 2011). Pneumatic pumps and dehydrators are suggested to be the largest undesired sources of fugitive emissions, while certain features are designed to purposefully vent gas such as pressure relief valves. Venting is being minimised and only justified in situations where the safety of workers and the site is at risk. GAO (2010) concluded that 0.3-1.9% of the gas produced is lost due to venting and equipment leaks. Other studies by Hayhoe et al. (2002) and Armendariz (2009) estimate this to be 0.5% or less while Shires et al. (2012) suggest it to be closer to 1%. Also, during the course of the life of a gas well, it may require liquid unloading, where liquid is removed from inside the well pipe to remove and minimise water intrusion from the surrounding geology as reservoir pressure drops. As not all wells require unloading, this can result in fugitive emissions of 0-0.26% of total production (EPA, 2010; Howarth et al. 2011; GAO 2010). Sub-surface leaks can occur from the wellbore unless zonal isolation is ensured, which can be best attained with self-healing cement and high frequency monitoring.

2) **Processing Losses**

   There are further leaks associated with the processing of the gas when it reaches the surface, and is largely dependent on the gaseous content of the natural gas produced. As discussed above this can vary hugely across different shale plays, however it can also vary within the same shale play. In North Eastern Pennsylvania, gas produced from the Marcellus shale is 'pipeline ready' while in South Western Pennsylvania the gas requires

processing (NYDEC 2009). This gas can contain large volumes of heavier hydrocarbons such as ethane and propane which can be processed in cracking facilities or can be rich in other impurities such as sour gas (H_2S), which can be sold or destroyed. In the Vale of Pickering, sour gas (H_2S) is produced along with methane and requires processing. Doing so can result in emissions of up to 0.45% of processed gas, however a study from Canadian processing plants suggest an emission factor of up to 4 times that suggested by Shires et al (2009) and by the EPA (2010).

3) Storage and Transport losses

Gas produced from shale layers that is pipeline ready can be transported, and or stored at specialised facilities. The 1996 EPA emission factor suggested a mean US leak rate of 0.53%, which is complementary with the work of Kirchgessner et al. (1997), while Russian emission factors estimates were averaged at 0.7%, however this is taken from the 1970's! (Howarth et al. 2011). Lelieveld et al. 2005 used the EPA (1996) data and their own inventory to suggest a range of 1-2.5% leakage rate from storage and transport, but Howarth et al. (2011) suggests the absolute minimum to be 1.4% considering the high uncertainties associated with the 1996 study. Taking into account all natural gas storage and transport infrastructure, Hayhoe et al. (2002) stated a large range of 0.2-10% emission rates! This is not useful for long term emission scenarios as each pole of the range gives two totally opposite scenarios. Howarth et al. (2011) gives a 'conservative estimate' of 1.4% to 3.6% fugitive emission rates for shale gas (excluding well completion), identical to that of conventional production.

1.3.4: Measurements of methane emissions:

The correct methods of greenhouse gas and specifically methane measurement have been a source of contention over the past decade as methane emissions have become a more prominent topic of investigation in the wake of accelerated global warming. There are two main methods;

1) Bottom Up: Where emissions are measured at ground level, such as in this investigation.
2) Top Down: Where methane measurements are taken at height such as from a plane or tower

Allen et al. (2013) used a bottom up approach to collect direct measurements from 190 onshore natural gas wells operated by 9 different companies, including 150 production sites, 27 well completion flowbacks, 9 well unloadings and 4 workovers. Their data suggested that well completion flowback methane emissions ranged from 0.01 Mg to 17 Mg (95% confidence bounds of 0.67-3.3Mg), which is at least factor of 5 less than the 2011 EPA national inventory estimate of 81Mg per event (Allen et al. 2013). However emission factors from pneumatic pumps, controllers and leaks were complementary to and in some cases higher than EPA estimates (Allen et al. 2013). Ultimately, if leaks from these upstream sources are representative of national emission standards, total annual emissions were calculated to be 957 Gg CH_4 (±200Gg), which combined with EPA inventories for other categories leads to a total of 2,300 Gg of methane emitted annually, which is 0.42% of gross gas production (Allen et al. 2013). This paper was criticised by Howarth (2014) who stated that the figure was similar to the best practice scenario calculation of 0.5%, and that considering the research was carried out with total reliance on the industry for sampling points, it proved no surprise that measurements were representative of best possible practice.

Conversely, Miller et al. (2013) adopted a top down approach in which 7710 plane observations and 4984 tower measurements were made between 2007 and 2008 across North America, combined with an inverse model to assess the total national methane budget. The results show that the EPA and the Emissions database for Global Atmospheric Research underestimate national factors by a factor of 1.5-1.7, with South Central US accounting for up to 25% of national emissions and estimates a methane source 2.7 times greater than most other inventories to date (Miller et al. 2013). Miller et al. (2013) suggest that regional methane emissions from fossil

fuels could be up to 5 times larger than in the Emissions database for Global Atmospheric Research (EDGAR) methane inventory. This detection of large methane emissions casts doubt on EPA estimates and the decision to downscale national methane emissions by 25-30% from 2011-2013 and proposes a leakage rate of 3.6% from 2007-2008 (Miller et al. 2013; Howarth, 2014). The Miller et al. (2013) has been criticised also for its lack of ability to conclusively attribute the atmospheric methane levels to the shale gas industry, rather than the entire natural gas industry. Howarth (2014) states that the increase is almost certainly due to the expansion of shale gas in the US, since Texas and Oklahoma were among the top 5 natural gas producing states between 2007 and 2008, and by August 2008 94% of wells drilled in the Barnett were horizontally drilled. Over these gas producing states, Miller et al. (2013) correlated higher atmospheric propane abundance (tracer of fossil hydrocarbons) with high methane concentrations (R^2=0.72) compared to other parts of North America where correlation is less (R_2=0.11 to 0.64), which is indicative of a fossil source. However the lack of isotopic analysis of atmospheric methane cannot therefore prove a causal source. Miller et al. (2013) discuss how methane emissions from ruminants and manure are up to twice the magnitude of existing inventories, which is not surprising considering Texas has the greatest number of cattle of any state, at 11.7 million and Oklahoma have the 5th largest cattle population, at 4.85 million (USDA 2009). Brandt et al. (2014) agree that 'official inventories consistently under estimate actual methane emissions', but claim that top down studies from Texas, Utah, Colorado and Oklahoma are unlikely to be representative of natural gas leakage rates. Brandt et al. (2014) agree that the methane emissions from the natural gas industry are likely greater than the 1.8% EPA estimate by 1.8-5.4%.

Caulton et al. (2014) also adopted a top down approach, using an instrumented aircraft platform to identify and quantify methane sources in a 2800km plot in SW Pennsylvania of June 2012. A flux of 2-14 g CH_4 s^{-1} km^{-2} was quantified for this region of the Marcellus shale, which is complementary to an averaged bottom up analysis of 2.3-4.6 g CH_4 s^{-1} km^{-2}. Although, they warn that the larger range of methane fluxes, especially considering the impact of scenarios at the upper limit of the flux warrants research, the data from which would not have been discovered under solely bottom up approaches (Caulton et al. 2014). A total of 7 well pads (1% of dataset) from the Marcellus area produced large methane emissions, averaged at 34 g CH_4/s per well, determined to have been measured during the *drilling phase* and accounted for between 4-30% of total observed regional flux (Caulton et al.

2014). This result is 2 to 3 orders of magnitude greater than the EPA estimate for this operational pre-fracking stage, during which methane emissions were considered to be transient and negligible (EPA 2011; Caulton et al. 2014). A possible explanation is 'gas kicks', whereby gas from surrounding lithology unintentionally enters the wellbore, despite the use of over-balanced drilling whereby a well is kept at a higher pressure to prevent this (EIA Energy Outlook, 2013). Alternatively, underbalanced drilling methods may have been adopted, whereby the well pressure is kept lower than the surrounding geology, forcing any gas into the wellbore and up to the surface during the drilling phase (ASMESSC, 2005). These wells were not drilled for coal bed methane but the entire region is underlain with coal deposits, making it impossible to avoid drilling through them, which is a previously unquantified methane emission factor from shale gas pre-production, unless the gas is contained or flared (Caulton et al. 2014).

There is no primary data from shale gas operations in the UK, and therefore reports are produced based on data from the US, such as the Tyndall Report (2013) and the MacKay and Stone (2013) report for the Department of Energy and Climate Change. Both of these papers use US data and attempt to extrapolate it to a UK scenario for comparisons of fuel type, and to assist quantification of national emission rates. Both reports state the trepidation of using US data for UK scenarios.

1.3.5: The Well Completion Conundrum:

The Howarth et al. (2011) paper concluded that 'compared to coal, the footprint of shale gas is at least 20% greater and even twice as great on the 20 year horizon'. Evidence to support the claim came from the study of 5 gas wells from across the US as shown in table 1.3 (Howarth et al. 2011).

	CH4 emitted during flowback/ 10^3 m3/year	Initial gas production	Lifetime production	% emitted as flowback
Haynesville (Shale)	**6800**	640	210	**3.2**
Barnett (Shale)	370	37	35	1.1
Piceance (Tight sand)	710	57	55	1.3
Uinta (Tight sand)	255	42	40	0.6
Den-Jules (Tight sand)	140	11	-	-

Table 1.3: Howarth et al. (2011) unconventional well source data

The fugitive emission factor from this stage is calculated by determining the amount of methane which leaks as a percentage of total production (EUR). Methane emitted during flowback in the Barnett, Piceance, Unita and Den-Jules plays are all in the same order of magnitude whereas the figure from the Haynesville shale is 6,800 x 10^3 m^3/year and is almost 10 times as large as the closest figure from the Piceance basin (710 x10^3 m^3/year) however Howarth et al. (2011) explained the good consensus between the initial production rates and methane emitted during flowback justify its inclusion and legitimacy. The Haynesville dataset led Howarth et al. (2011) to calculate an emission factor of 1.9% for well completion from shale gas, compared to well completion from conventional gas production which is 0.01%. The 1.9% is split into; 1.6% from flowback fluid and 0.33% from the drill out phase. The emissions from venting and equipment leaks were calculated to be complementary to conventional wells at up to 1.9% (Howarth et al. 2011). Ultimately, this meant that the difference in fugitive emissions between shale gas extraction (3.6-7.9%) and conventional extraction (1.7-6.0%) is a function of methane emitted during drillout and well completion. This was the basis on which Howarth suggests that shale gas has a higher emission factor than coal or conventional gas.

Although Howarth et al (2011) admit that the data is limited; it has still been met with severe criticism predominantly because of the declarative conclusions from the data available.

There are 4 main criticisms of this paper;

1) **Physics, safety and economics of gas production**

   Cathles et al. (2011) have claimed that Howarth et al. (2011) extrapolation of gas venting rates during completion and drill out to initial production is incompatible with the physics of shale gas production, and the safety of drilling operation. Cathles et al. (2011) state that high flow rates are not possible when the well bore is still loaded with frack fluid, meaning it therefore would only be able to flow in isolated pockets and would not flow at optimum rates before enough water has been removed to the surface. Therefore well completion flow rates could never exceed initial production rates. Cathles et al. (2011) also suggest that the 3.2% leakage rate from a shale gas well over a 10 day period suggested by Howarth et al. (2011) would represent $1,000,000 worth of gas at 2011 prices which would be unwarranted on economic grounds and a fire risk. Broderick et al. (2011) however state that although this analysis would require empirical validation, cold venting in a site with good ventilation and minimal sources of ignition would not necessarily be a fire risk. While the EPA report (2011b, pp.3-12) estimated that in 2010, 85% of flowback gas from unconventional wells was vented and less than 15% captured or flared, and stated that the low density of methane in air (58%) means its buoyancy when vented minimised fire risk. Limited primary data again limits the degree of confidence in adopting these estimates as standard industry practice (Broderick et al. 2011)

2) **Assumption of venting**

   The assumption that gas removed during liquid unloading, well completion and drill out would be vented has also been met with great scrutiny. The source of the Haynesville shale data from Howarth et al. (2011) is an IHS report, of which a co-author has stated that the report did not support the conclusion of high methane emissions during flowback and well completion (Barcella et al. 2012). Mr Philip Stark, the VP of IHS proclaimed that the use

of IHS data from the Haynesville had been 'seriously distorted', stating that the well in question produced 14 MMcf of natural gas per day while the operator was cleaning up the frac load, *none of which was vented to the atmosphere'* (IHS, 2014). Mr Stark confirmed that this data source for Howarth et al. (2011) presents no evidence to support their conclusions.

EPA data on release rates were 50% of that of the 3.2% estimate for the completion period, and this is likely because where flaring or capture of the gas is not required by law, methane is assumed to be vented by Howarth et al. (2011). This assumption is not made clear and is not typical of practice beyond 2014 (Cathles et al. 2012). The magnitude of difference in well completion emissions is shown below in figure 1.7.

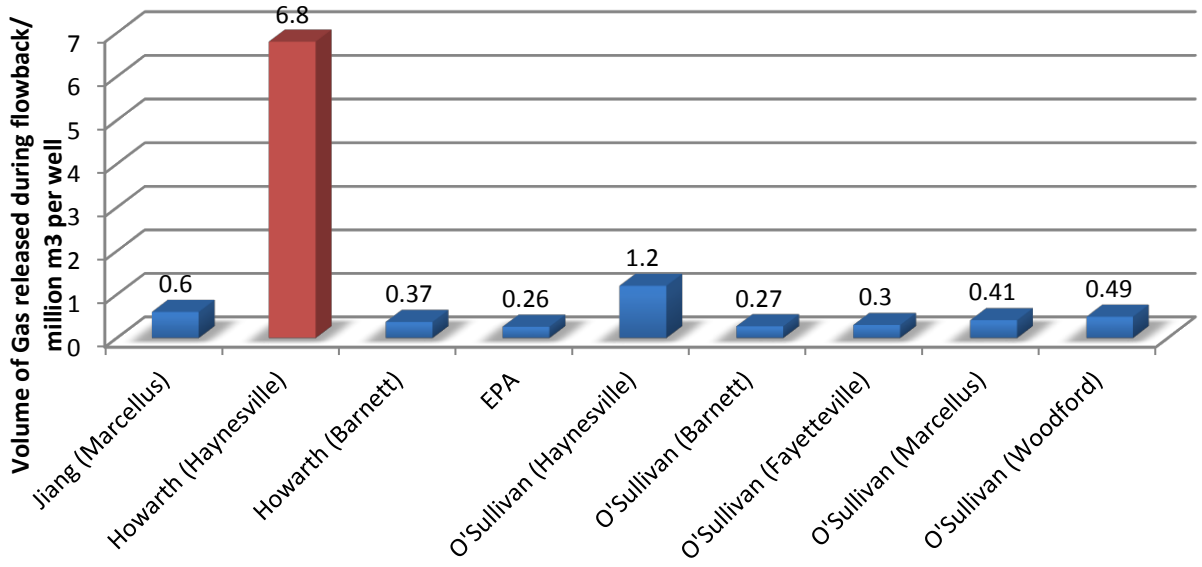

Figure 1.7: Comparison of well completion flowback emissions from different unconventional wells

The Howarth Haynesville is an outlier when compared to other peer reviewed results. The estimated emissions from Howarth's analysis are 14 times the average of the other emissions projections listed.

3) Under- recognition of green practices

Cathles et al. (2012) continue to suggest that throughout the Howarth et al. (2011) paper there is a persistent dismissal of technological improvements, such as green completions, which Howarth et al. (2011) state 'can reduce emissions by more than 90%' and have been implemented by the industry (EPA 2007, 2009). References cited in Howarth et al. (2011) including Fernandez et al. (2005) and GAO (2010) claim that under modern techniques, the 1.9% methane loss rate could be

at least 10 times lower. Howarth et al. (2011) also state that 'REC technologies require pipelines to the well are connected before completion', which is misleading in that it assumes that if the correct infrastructure is not in place it will be vented whereas it is more likely to be flared, minimising the greenhouse gas footprint.

4) Extrapolation to national scale

Perhaps the biggest criticism faced in Howarth et al. (2011) paper is the national extrapolation of only 5 wells to a national policy scale. The serious misuse of data for well completion, which was limited as admitted in the paper, promotes great uncertainty because of both quality and quantity and therefore is not appropriate as evidence to support extremely declarative conclusions.

Perhaps a more appropriate stance is that adopted by Brandt et al. (2014) in which they analysed 20 years of technical literature on US natural gas emissions and concluded that the official inventories were consistently underestimating methane emissions. The importance of 'super-emitters' was also highlighted, as they were identified in a study of 75,000 wells, of which 58% of methane emissions came from 0.06% of possible sources (EPA, 2006; Brandt et al. 2014). Therefore, datasets and conclusions which are not representative of the 1,000,000 wells in the US should be treated with great scepticism. However the uneven spatial pattern of methane emissions across the US, a function of both shale abundance and legal requirements in each state, further enhances the uncertainty in the scaling up of datasets (Miller et al. 2013). Brandt et al. (2014) conclude that if natural gas is to be a bridge 'it is one which must be traversed carefully' to ensure leakage rates are low enough to achieve sustainability goals. Brandt et al. (2014) statement is appropriate as it appreciates the difficult of using small, uneven datasets as evidence to justify national policy, but still warns that emissions are of great concern and threat, thus deserving detailed attention.

1.3.6: Emission Factors

An emission factor is the % leakage of the total natural gas production.

Howarth (2014) was the follow up review of the literature produced following the controversial Howarth et al. (2011) paper. This paper concludes with an even larger claim that 'over 20 year time periods both shale gas and conventional gas have a larger GHG impact than coal or oil regardless of end use'. This paper was not met with as much contention because many believed that by suggesting that both conventional and unconventional produced gas was 'a bridge to nowhere' nullified the reason to not use hydraulic fracturing technology as opposed to simpler conventional technology.

An analysis of the true emission factor of shale gas therefore requires an assimilation of all conventional and unconventional methane data for comparison to determine if the leakage rate exceeds the 3.2% threshold at which gas maintains a climatic benefit. This is shown in Table 1.4 on the following page

	Upstream Conventional Gas (%)	Upstream Unconventional Gas (%)	Downstream (%)	Total emission factor (%)
EPA 1996	0.2	-	0.9	**1.1**
Hayhoe et al	1.3	-	2.5	**3.8**
EPA 2010, US average	0.16	*	0.9	**1.1**
Jamarillo et al	0.2	-	0.9	**1.1**
Howarth et al	1.4	3.3	2.5	**4.7 / 5.8**
EPA 2011 (average)	1.6	3	0.9	**2.5 / 3.9**
Ventakesh et al.	1.8	-	0.4	**2.2**
Jiang et al.	-	2	0.4	**2.4**
Stephenson et al.	0.4	0.6	0.07	**0.47/ 0.67**
Hultman et al.	1.3	2.8	0.9	**2.2 / 3.7**
Burnham et al.	2	1.3	0.6	**1.9 / 2.9**
Cathles et al	0.9	0.9	0.7	**1.6**
EPA 2013, US average	0.88	*	0.9	**1.8**
Karion et al.	-	9	-	-
Allen et al. US average	0.42	*	-	1.5
Miller et al. US average	-	-	-	**>3.6%**
Brandt et al. US Average	-	-	-	**5.4**
Schneising et al	9.6	-	-	-
McKain et al.	-	-	2.7	-

*Table 1.4: Emission factors: Numbers in red represent total emission from unconventional gas drilling and the numbers in blue represent the conventional emission factor. Black final percentage numbers represent national emission factors for both conventional and unconventional sources. * =combined with conventional*

Upstream conventional gas emission rates range from 0.2 to 1.8%, while for shale gas it ranges from 0.6 up to 9%. All of these measurements have had a significant degree of uncertainty because of the poor dataset on methane emissions.

A large contributor to variation in the emission factor for unconventional wells sources is a function of lifetime well production (EUR) (Mackay and Stone, 2013). The upstream values especially are all scaled to production rates of each well, so the greater the production from a well, the lower the emission factor. Since shale gas expansion on this scale is a new phenomenon it has been difficult to assess the production rate, but recent studies have proven the productivity of shale wells is much less than conventional wells. The question is if academics and industry are overestimating the mean lifetime productivity of shale wells, this would increase the relative leakage percentage.

Downstream emissions are equally as uncertain. Almost 50% of the studies use the EPA 1996 emission factor for downstream emissions, which has remained unchanged since then. Hayhoe et al. (2002) note that this factor is uncertain and suggests that this is likely to be higher by a factor of 2-3. Half of the high pressure pipelines in the US are over 50 years old, which would imply a high rate of leakage, but studies attempting to analyse and quantify fluxes from downstream sources across the US have been unable to do so due to the inability to differentiate natural gas emissions with other sources (Howarth, 2015; Philips et al. 2013; Townsend Small et al. 2012). Of those studies from table 1.4 which did not use the EPA data, over 70% of the results were lower than the EPA estimate which is contradictory to suggestions that leakage rates are higher, but again the data scarcity and uncertainty justify further investigation.

The studies from Colorado tight gas field, Utah and Backen/Eagle ford shale gas fields were 4%, 9% and 9.6% respectively, and all three were collected with top down methods (Petron et al. 2012; Karion et al. 2013; Schneising et al. 2014). The Colorado basin overlaps with the high end scenario of upstream unconventional emissions by Howarth et al. (2011), but the other two are considerably higher and not representative of US emission factors as they focused on specific basins which have had recent declines in air quality, and thus would expect high methane concentrations (Howarth, 2014).

1.4: Background of expansion of research into Shale gas

It is very important when investigating the greenhouse gas impact of shale gas exploitation to understand why the U.S. has chosen to exploit such reserve, and why also it is important for the UK. Before 2007, Shale gas and Hydraulic Fracturing were scarcely mentioned in US politics and was not regarded by the EPA to be any different in emission factor to conventional hydrocarbon production. In 2016, shale gas production is one of the most contentious issues in the USA and UK. It has been a polar argument fuelled by contrasting opinions of environmental concern and economic prospects. In 2014 and 2015, federal and state investigations were launched in shale gas intensive areas such as Dimock, Pennsylvania and Dish, Texas where shale gas companies had been accused of contaminating water resources by hydraulic fracturing the Marcellus and Barnett Shale. In 2015, the EPA (2015) report on Shale gas in the US concluded that they did not find evidence that hydraulic fracturing has 'led to widespread, systemic impacts on drinking water resources'. Instead they stated that poor well pad leakage maintenance and spills were the dominant cause of water contamination, and in a small number of cases poor well construction led to methane migration into aquifers (EPA, 2015). They finally noted that the lack of pre-hydraulic fracturing data on aquifer contaminant concentration made it very difficult to prove any causal relationship (EPA, 2015). This mistake is not being repeated in the UK. Currently, the BGS and prospective UK shale gas companies are conducting seismic surveys and testing of water aquifers to create baseline data pre-hydraulic fracturing. It is also important to ascertain the ambient methane concentrations on actively producing conventional well pads, among other upstream and downstream natural gas infrastructure. By doing so, ambient methane concentrations and methane fluxes can be calculated for future comparison to shale gas if and when it is exploited in the UK.

In the USA shale gas exploitation is now the dominant producer of natural gas, providing 52% of total gas production in 2015 (US EIA 2016). Under current production scenarios, domestically produced shale gas reserves could be adequate to meet demand for 80-100 years. This has jump started global attention that the shale layer can be productive. The reason why the UK has taken such interest in shale gas is because from 2000, hydrocarbon production from the North Sea has been inadequate to meet demand and under current trajectories, by 2019, 70% of UK natural gas will have to be imported (DECC, 2015). As fossil fuels currently provide 86% of UK total energy usage this is an undesirable scenario, and one which

the UK government believes shale gas could alleviate (DECC, 2015). Natural gas alone provides 40% of UK power generation and 84% of UK homes are heated with gas, totalling 35% of final UK energy use (DECC, 2015).

Therefore, it is very important to understand how shale gas use would impact national and international emission rates. Considering fossil fuels are likely to form the bulk of global energy use for the next half century, it is important to research how it is possible to utilize such resources while minimizing the greenhouse gas footprint. It is also relevant with such a broad body of research into climate science relative to the shale gas expansion in the USA to utilize such research in order to provide solutions appropriate to mitigate against accelerating climate change.

1.5: Thesis Objectives

Overall, this thesis was designed to;

A) Measure the methane concentration of upstream hydrocarbon infrastructure in order to calculate ambient methane concentrations, relative to a control, and calculate a methane flux rate for comparative analysis.

Average methane concentration can be recorded to allow comparisons between sites with different functions. However, monitoring methane concentration of production sites is inadequate alone to calculate the amount of methane being emitted from the site per unit of time, i.e. the flux. Flux calculations from UK production sites will allow for direct comparison to other industries and other land uses. Flux measurements will quantify the site emissions, while monitoring of concentration will serve to identify anomalies.

B) Assimilate literature on pre-production emissions from shale gas exploitation to ascertain the emission factor of shale gas to determine whether or not it meets the 'bridge fuel' description.

There is currently no publicly available data on methane emissions from UK shale sites. This is because the UK shale industry is nascent. Literature and projections will have to be extrapolated from the US as it is where most development and data collection has taken place. Analysis of this literature will allow for lessons to be learned and to replicate best available techniques (BAT) for emission mitigation.

2.0 Methodology

2.1. Study areas

This study was conducted in 3 sites, 2 of which were part of a closed upstream natural gas infrastructure and the other was a control site. The approach of the data collection was to compare the atmospheric methane concentrations at sites which produce and combust natural gas, in comparison to an ambient control site. A flux was also calculated for the two natural gas infrastructure sites. The sample size allowed comparison of methane concentration fluxes over both space and time. These sites were chosen because they are producing and combusting gas produced from conventional methods of drilling in an area of North Yorkshire, which has the potential for shale gas exploitation.

Data collection took place over 12 months in four 2-3 week periods: October 2015 and January, April and July 2016. Conducting research throughout the year was done to take potential temporal and particularly seasonal variations in methane concentrations into account and or large wind speed variation, which would influence flux results. Overall, 248 hours of methane data collection took place at the KM5 wellpad, Knaption generation station and Cranford House farm, Ryedale.

Figure 2.1: map of KM5 (KM4) wellpad, KM8 site and Kirby Misperton

The KM5 and KM8 wellpad are less than 2 km from Kirby Misperton village

2.1.1: KM5 wellpad

The KM5 wellpad is a producing well pad near to the village of Kirby Misperton in the Vale of Pickering, North Yorkshire and is operated by Third Energy. Gas was first discovered in the Vale of Pickering in 1985 and production from the four main gas fields; Kirby Misperton, Pickering, Malton and Marshes began in 1995. Currently, Third Energy operates 6 gas pads which produce natural gas from the Permian age Kirkham Abbey Formation and from the deeper Namurian age reservoirs. KM5 was chosen for methane measurement as it provides a human conduit between a thermogenic gas source and the surface. In 2015, KM5 produced 1.968 million m^3 of natural gas. These fields are mature and have experienced significant declines in production over the past decade. This well pad is not due to undergo hydraulic fracturing, however the neighboring well pad KM8 is under consideration for the process.

Figure 2.2: KM5 wellpad monitoring of methane emissions
Measurements being conducted with the well head upwind of the intake device.

2.1.2: Knapton Generation Station

Knapton generation station is a site which combusts natural gas produced from four of the six gas fields. The gas is piped underground to the site to produce electricity which is sold to the national grid. KM4 is less than 2km from Knapton generation station. It has done so since 1995 and is the largest onshore generation station in the UK to use domestically produced onshore gas. The power generation involves a RLM 6000 EFT generation package with a LM6000 gas turbine. Since 1995,

Knapton generation station has processed 0.85 billion m^3 of gas and has generated over 2 million MWh of electricity. In 2015, Knapton generation station combusted 4.92 million m^3 of natural gas at 90% content CH_4.

Figure 2.3: Knapton generation station monitoring of methane emissions
Measurements were conducted downwind of the flare stack and surrounding infrastructure

2.1.3: Cranford House Farm, Ryedale:

Finally, atmospheric methane concentration was measured at Cranford House Farm in Ryedale, North Yorksire, less than 3km from both Knapton generation station and KM5. The measurement site was a single-track lane, which divided a 20 hectare vegetable farm which acted as a control site for ambient methane concentration as there were no known sources of methane which could have influenced results.

Figure 2.4: Cranford House farm monitoring of methane emissions

2.2: Gas measurement and analysis

A Picarro Surveyor P0021- S cavity ring down spectrometer (Pi-carro Inc., Santa Clara, CA- henceforward referred to as the Picarro) was used to measure CH_4 (precision 5ppb + 0.05% of reading ^{12}C) and $\delta^{13}C$- CH_4 (‰, Pee Dee Belemnite) whilst stationary at each site. A sample line was attached to the roof at the rear of the vehicle and the sample gas was measured at a frequency of 1Hz. Throughout the research the Picarro was re-calibrated three times to ensure continuous precision of measurements. A 2-D anemometer (WindSonic, Gill Instruments, Lymington, UK) was attached onto a tripod (2m above the ground surface) to measure wind speed (between 0 and 60 m/s ± 2% @ 12 m/s) and direction (0-359 ± 3°C). The wind speed data was used in combination with the methane concentration and Picarro software to map wind plumes and identify probable source areas. A GPS A21 (Hemisphere, Scottsdale, Arizona) attached to the roof of the vehicle to map the exact location of measurements. Instrument alignment was required to correct for minor disparities in collection times from the anemometer and the Picarro.

When measuring methane concentration at KM5 and Knapton, careful attention had to be paid to wind direction, and thus upon arrival at each site each day the location of the Picarro had to be downwind of any potential source. Therefore, the exact location of the equipment on each site relative to identified sources varied. In some circumstances where wind direction changed throughout the day, the Picarro was moved to account for this. Throughout the data collection, as well as measuring methane concentration, the isotopes of methane present in the local atmosphere were measured. The isotopic concentration of the methane sources at each site was determined using Keeling plots of $\delta^{13}C$- CH_4 against the inverse of the CH_4 concentration, with the intercept reflecting the source composition in per mille, ‰ (Pataki et al. 2003). Isoptopic concentrations about -50 ‰ are typical of ambient methane concentration, with isotopically lighter values (more negative than -55 to -60) suggestive of a biogenic source and isotopically heavier sources (more positive than -45 to -40) suggestive of a thermogenic source (Pataki et al. 2003).

2.3. Data Analysis:

Primarily methane concentration on each collection day was correlated to its respective time and presented in seasonal periods, autumn/winter and spring/summer. Presenting daily methane concentration was done to allow comparison on days and between days throughout each measurement period and

for comparison between sites. However, concentration is inadequate to quantify the extent of methane leakage from natural gas facilities and therefore a flux must be calculated. As there is no potential point source from Cranford House Farm, a flux from that site could not be calculated.

2.3.1 Flux calculation with 3-D Gaussian Plume Modelling

During the measurement of methane at KM5 and Knapton it was not possible to stay a fixed distance away from a particular point source such as the well head at KM4 or the flare stack at Knapton due to wind direction and speed. The distance away is important as with increasing distance from a potential source, methane concentration would decrease to ambient, and therefore any difference between these sites could be ascribed to the distance between the point source and the Picarro. This therefore required an analysis which took into account distance, and to do this the dynamic plume approach of Hensen and Scharff (2001) was adopted. A 3-D Gaussian plume model was applied to each measurement day from KM5 and Knapton. Data was collected as ambient methane concentration on the day (measured as ppm) and any methane measurements collected in the wrong half disk (i.e. methane measurements origination from the opposite side of the point source) were removed prior to any analysis. The concentration data were converted to mg/m^3 with assumptions of an air pressure of 101.4 Pa and an air temperature of 10°C. A 3-D Gaussian plume was then used to predict the flux from the source(s) at KM5 and Knapton.

Given the concentration of methane above ambient at a known distance away the 3-D model is;

$$\text{Conc.}(xyz) = \frac{Q}{2\pi u_x \sigma_y \sigma_z} e^{\frac{y^2}{(2\sigma_y)^2}} \left[e^{\frac{-(z-H)^2}{(2\sigma_z)^2}} + e^{\frac{(z+H)^2}{(2\sigma_z)^2}} \right]$$

Figure 2.5: 3-D Gaussian Dispersion equation

where: x= shortest distance from point of measurement to the fault (m); y = the perpendicular distance along the fault of the measurement (zero m in this study); z = the height of the detector above the ground surface (1.5 m); Q = the source strength (mg/s); u = the wind speed resolved along x (m/s); H = the height of the

source (m); and σ_y and σ_z = dispersion terms in the directions y and z. The dispersion terms are approximated as $\sigma_y = I_y x$, and $\sigma_z = I_z x$ and in near surface conditions the assumption is that there is no stable stratification and that therefore $I_z = I_y = 0.5$. The shortest distance to the potential point source from the Picarro was measured (x) and given the measurement of the wind speed and direction at height z then the wind speed could be resolved along the direction of the shortest distance to the potential source. As the natural gas would be under pressure, any potential crack or poorly sealed infrastructure from the well pad or from Knapton would be identifiable and quantifiable, hence making this research an investigation of performance. Ultimately, a result is calculated in mg/hour, which can be scaled up to kg/year for direct comparison with other natural gas facilities and other industries which produce methane emissions.

3.0 Results

Of the 80 hours of methane data collected at the Cranford House control site, the average methane concentration was 1.93 ppm. The 208 hours of data collection from the other sites had lower and higher average methane values by comparison, with KM5 and Knapton having average methane concentrations of 1.91 ppm and 2.01 ppm respectively. Average annual CH_4 flux at KM5 was 24.29 kg/year on average, with a measured maximum of 106.74 kg/year and the average flux at Knapton was over 35 times greater, calculated as 847.91 kg/year.

3.1: Cranford House Control site

Average methane concentration measured over 80 hours at Cranford House farm is shown in figure 3.1 below. The BGS (2012) state that ambient methane concentration in an area with no specific methane source should be in the range of 1.80-2.00 ppm. The wind rose (figure 3.1) displays no indication of a distinct methane source, with an even methane concentration distribution from all wind directions, about a concentration of 1.93 ppm CH_4. The complete time series of methane concentration for this site are shown in figures 3.2 and 3.3.

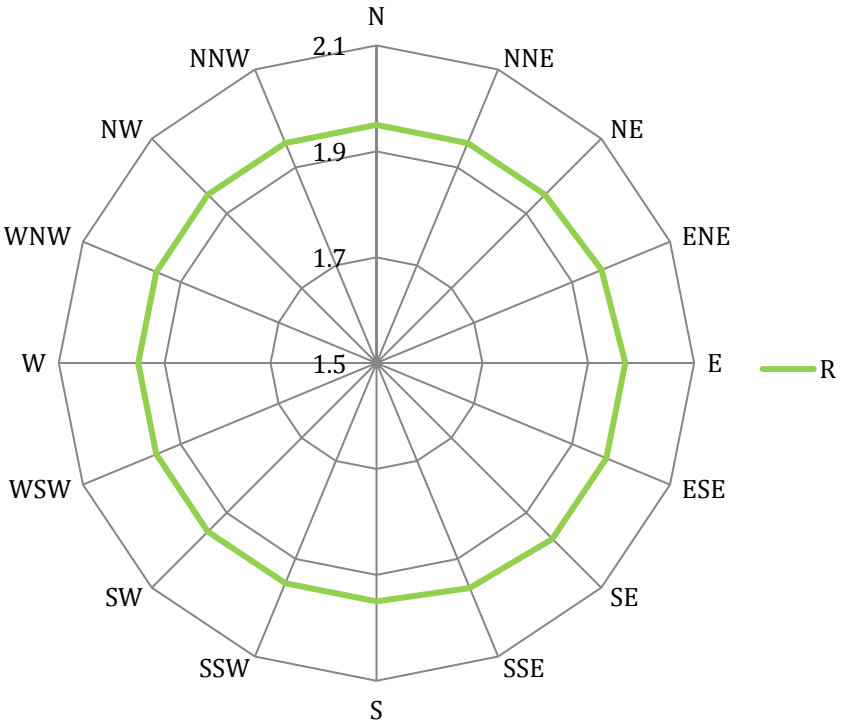

Figure 3.1: Cranford House (R) Methane wind rose
Even distribution reflects no distinct methane source

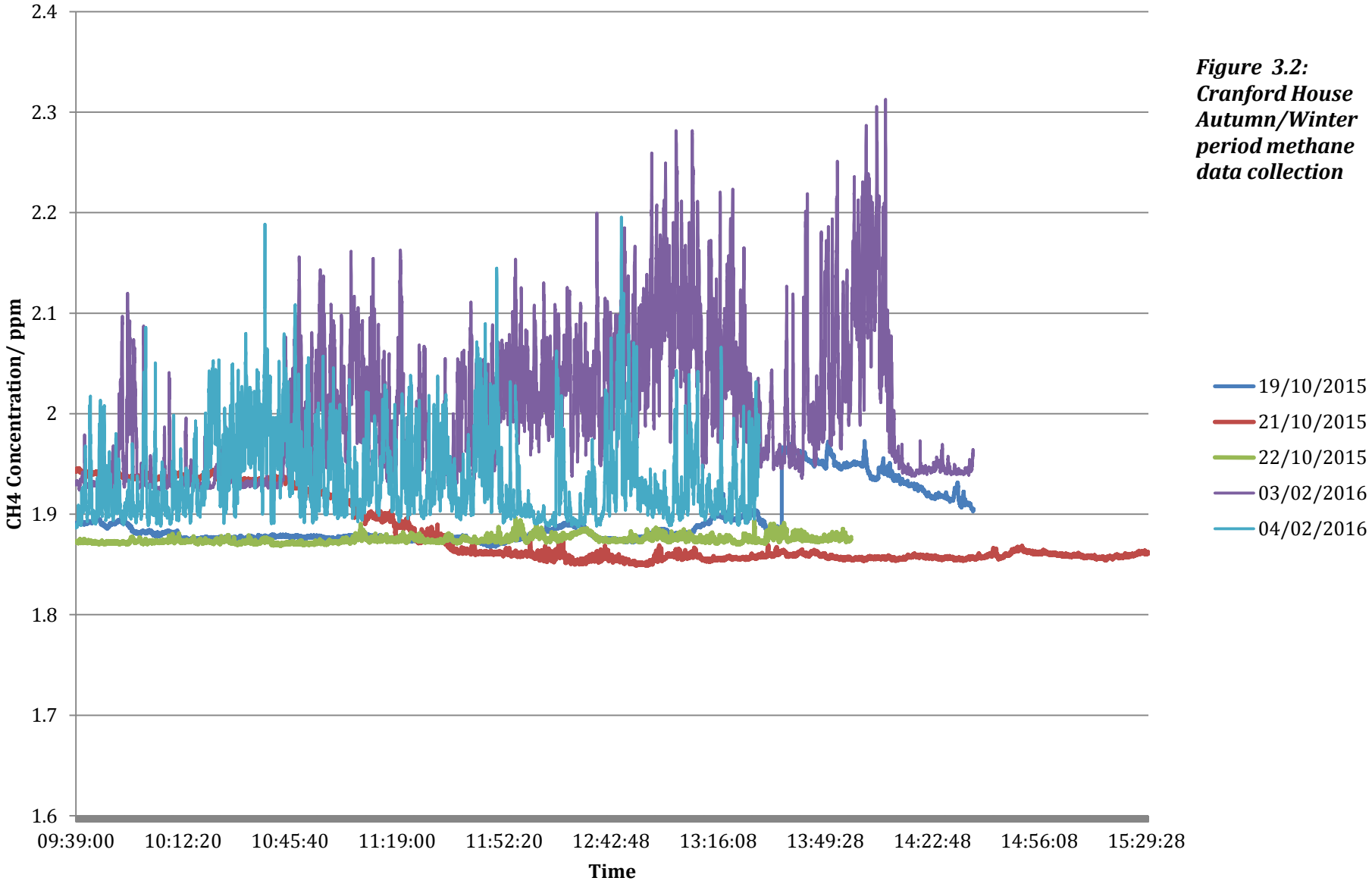

Figure 3.2: Cranford House Autumn/Winter period methane data collection

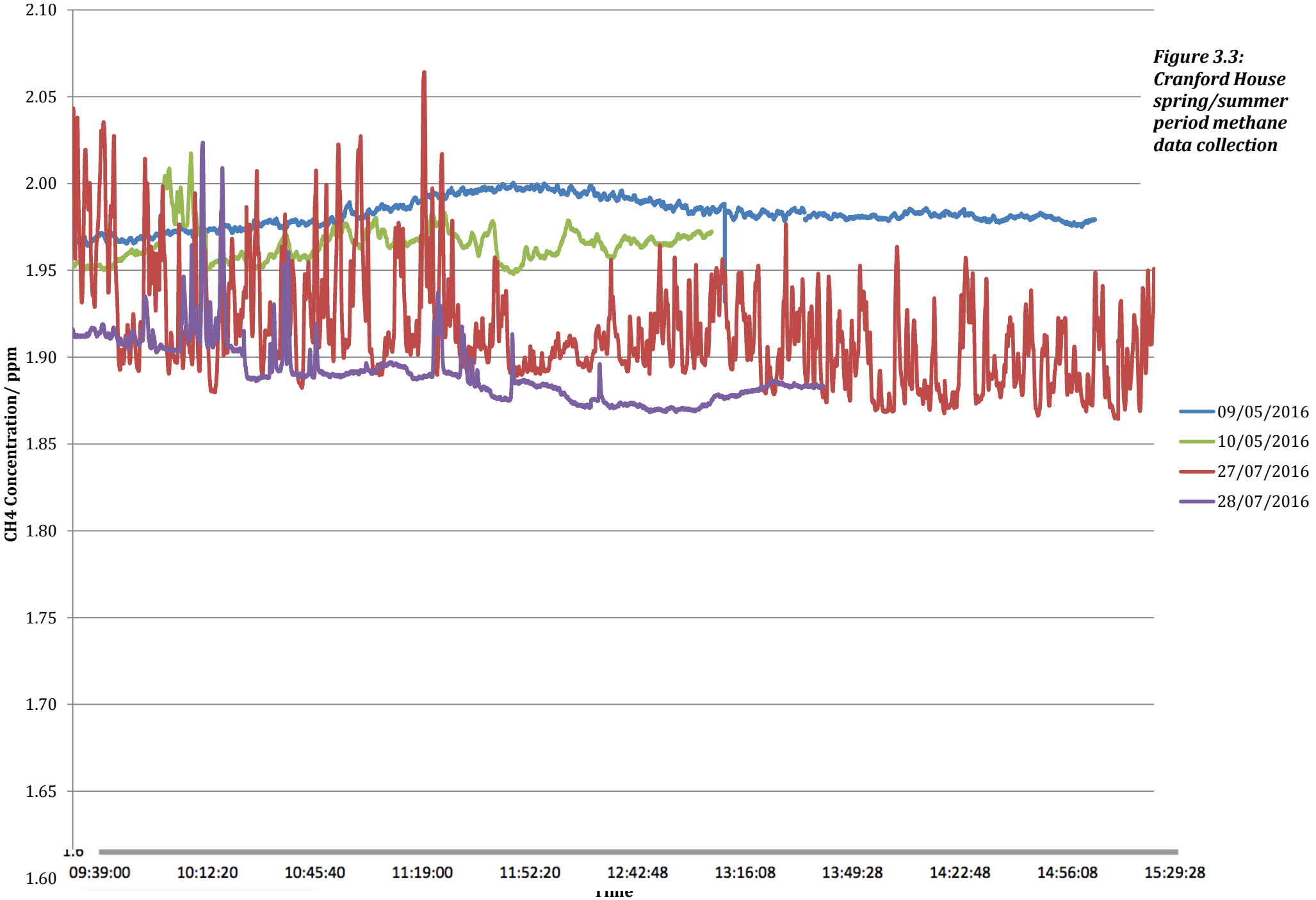

Figure 3.3: Cranford House spring/summer period methane data collection

The autumn/winter and spring/summer methane concentration results are shown in figures 3.2 and 3.3, where average CH_4 concentrations were 1.96 and 1.94 respectively. In the autumn/winter collection period, over 35% of the results were above 2 ppm CH_4, while in the spring/summer collection period, less than 10% of measurements exceeded 2ppm CH_4. However, across all Cranford House Farm measurements, CH_4 concentration did not decrease below 1.84 ppm. In the autumn/winter collection period in figure 3.2, there are regular fluctuations of CH_4 concentration from the February collection period when compared to samples collected in October, which remain relatively stable throughout the measured day. In figure 3.3, this pattern is replicated with CH_4 sampling during May showing less variability about BGS (2012) average ambient concentration of 1.90 ppm when compared to the July sampling period. It must be noted though that these fluctuations are very small, at ~0.2ppm. Data from the 9/5/16 sampling day had the highest average methane concentration of the spring/summer period at 1.97ppm, and recorded the lightest isotopic measurement at Cranford house of -61 ‰. Methane concentrations at Cranford House were typical of ambient concentrations suggested by the BGS (2012), but there was more variability in the autumn/winter period than during spring/summer data collection. Isotopic analysis of the methane measured revealed that over the year, Cranford house methane measurements had an isotopic composition of -52 ‰, which is typical of a mixed non-distinct methane source (BGS, 2012) and figure 3.4 below shows the isotopic composition from Cranford house. Variations in methane isotopic compositions at Cranford House are shown in the Keeling curves overleaf (figures 3.5 and 3.6). The lack of a distinct source at the control site meant it was not possible to give a flux estimate for the site.

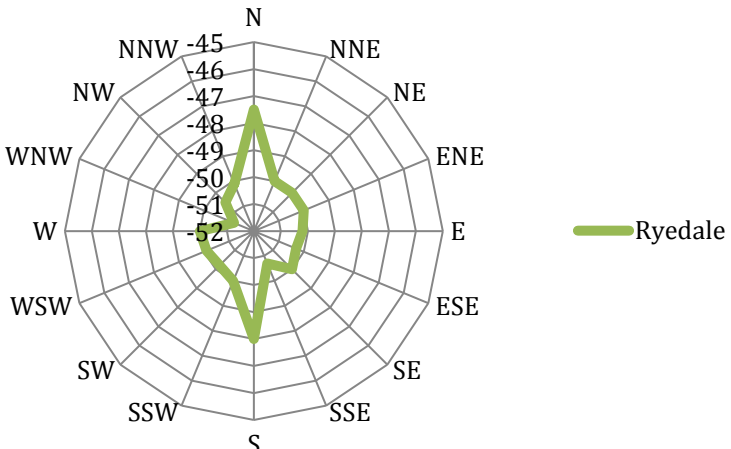

Figure 3.4: Cranford House (Ryedale) Isotopic compostion

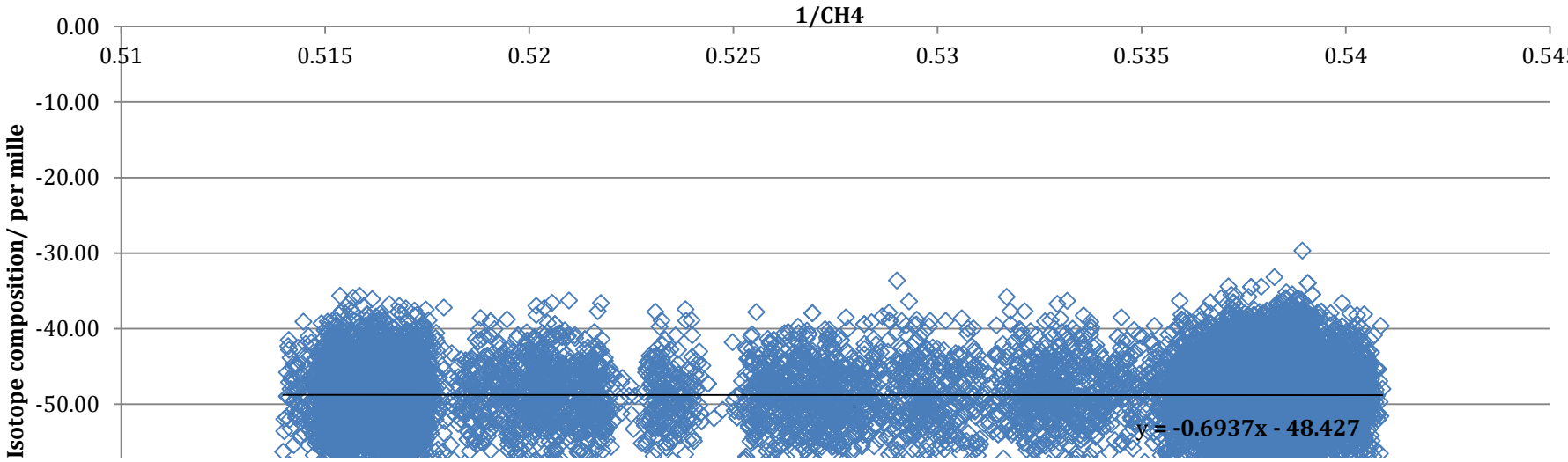

Figure 3.5: Methane isotopic composition from 21/10/16 at Cranford House

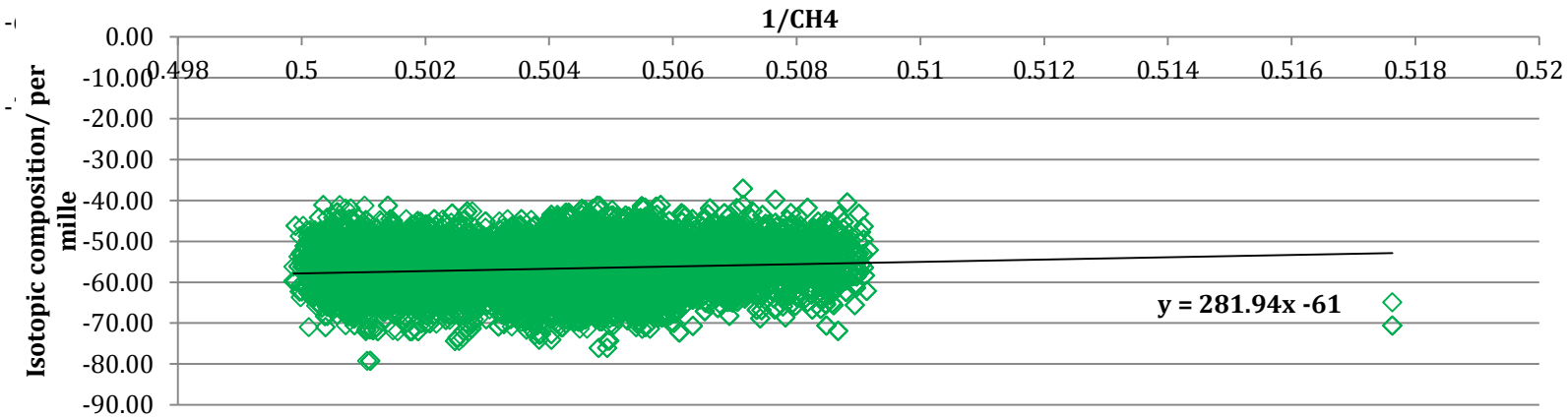

Figure 3.6: Methane isotopic composition from 9/5/16 at Cranford House

3.2: KM5 Results

The average methane concentration at KM5 of 1.91 ppm is lower than at the Cranford Farm control site (Figure 3.7). In the first round of fieldwork from the 13/10/15-16/10/16 (Figure 3.8), the Picarro was able to detect methane spikes above ambient concentration of 1.80-2.0 ppm (BGS, 2012). The Keeling Curve from this period reflects a detection of a thermogenic source (Figure 3.10); the site flux from this period was constrained to be 7-22 kg CH_4/year. On the sampling days in January 2016 (26^{th}-29^{th}); the methane concentration was consistently less than 2 ppm with the daily averages ranging from 1.89-1.92 (Figure 3.8); 75% of the sampling days had fluxes of < 1 kg CH_4/year. Data from the 28/1/2016 suggested a flux of 56 kg CH_4/year, but its average isotopic composition was -47 ‰, which does not reflect a distinctly thermogenic source.

In the spring/summer season measurements at KM5, almost 95% of the methane measurements are less than 2 ppm (Figure 3.9). In the morning to mid afternoon of the 20/7/16 however there are three distinct methane spike periods where the methane concentration reaches 4 ppm, and the calculated flux would equate to 22 kg/year. Isoptopic analysis of this sample revealed this methane spike to be biogenic in source, with an isotopic composition of -58 ‰ as shown in figure 3.12. Throughout the remainder of the spring/summer measuring period, there was no distinct thermogenic source detected, with the daily isotopic compositions ranging from -48 ‰ to -58 ‰.

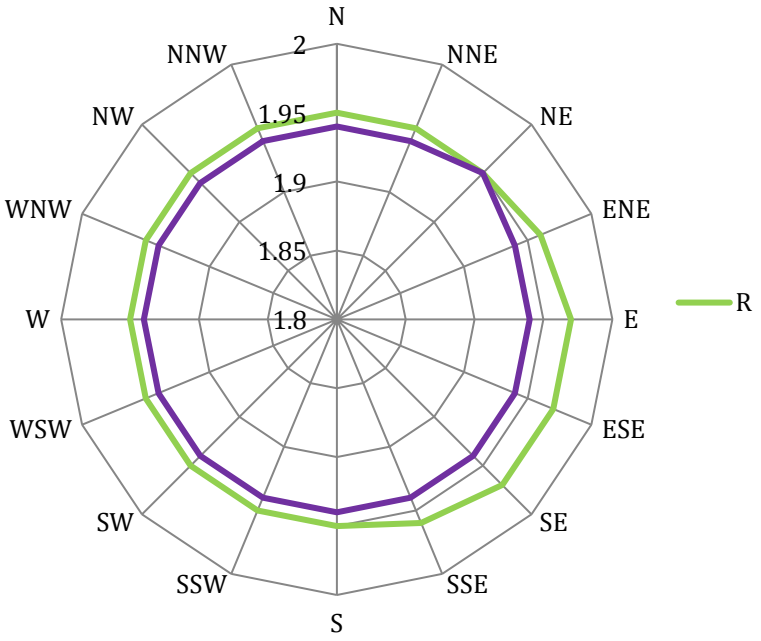

Figure 3.7: Average methane concentration at KM5 (purple) and Cranford House control site (R)

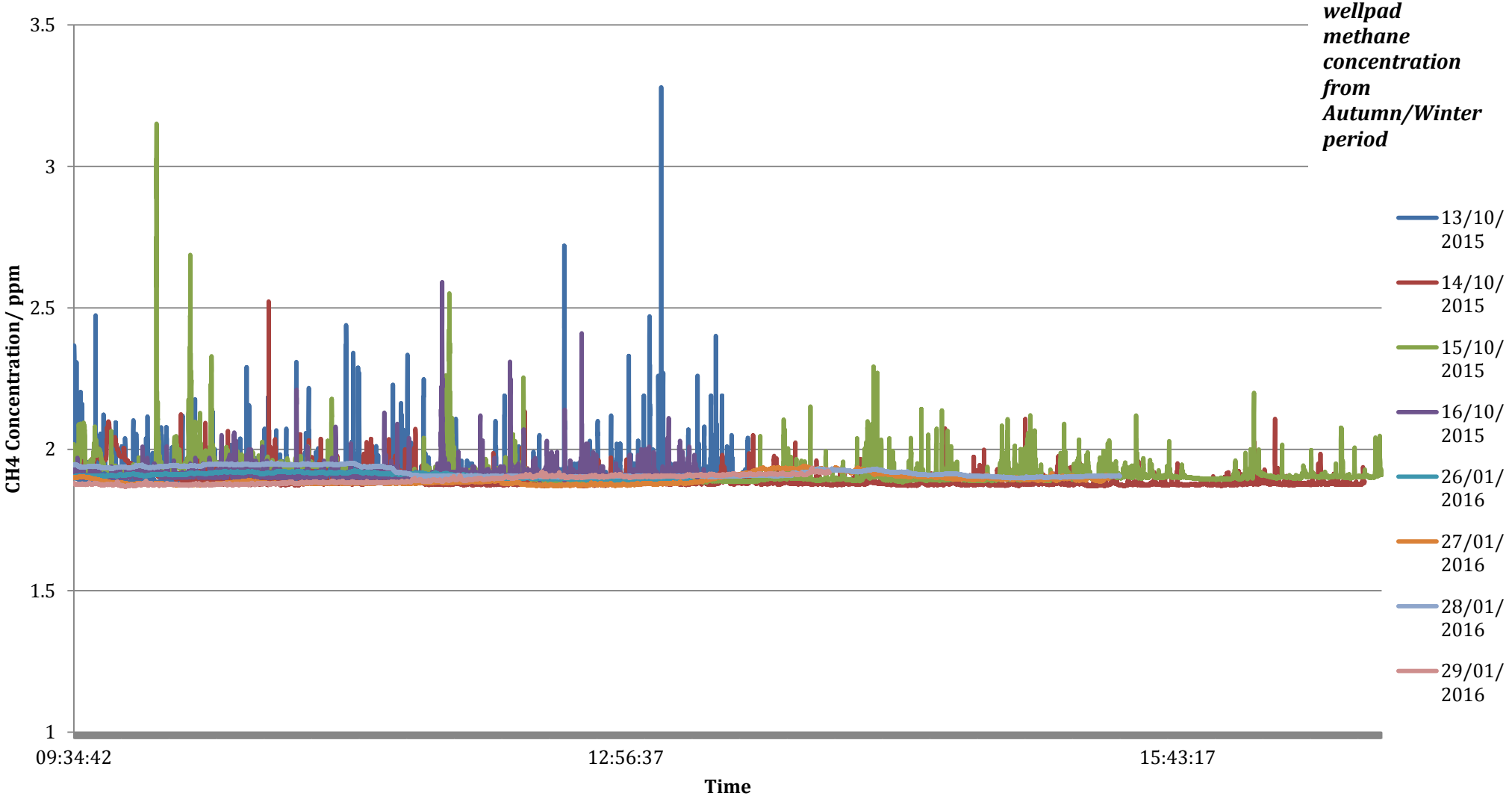

Figure 3.8: KM5 wellpad methane concentration from Autumn/Winter period

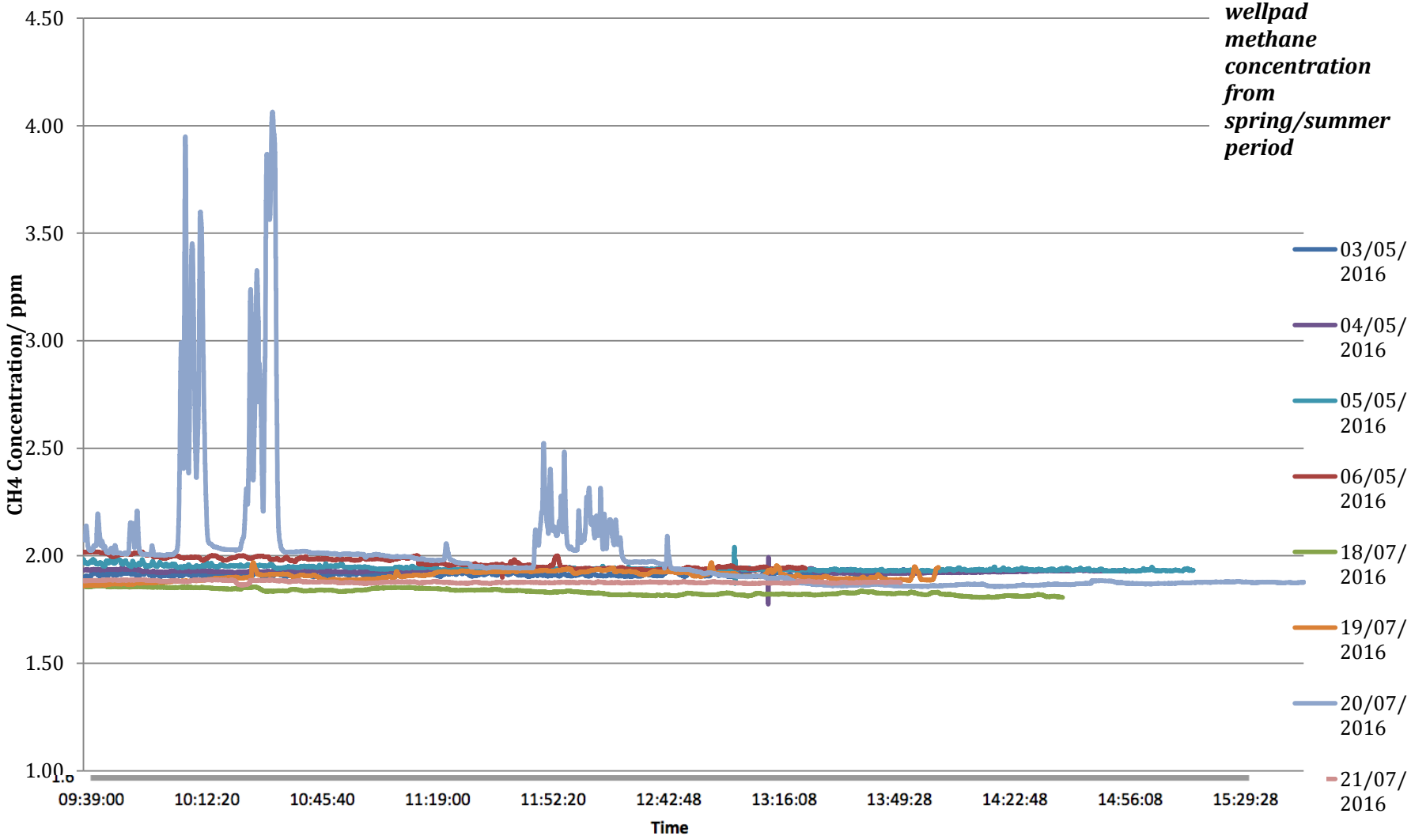

Figure 3.9: KM5 wellpad methane concentration from spring/summer period

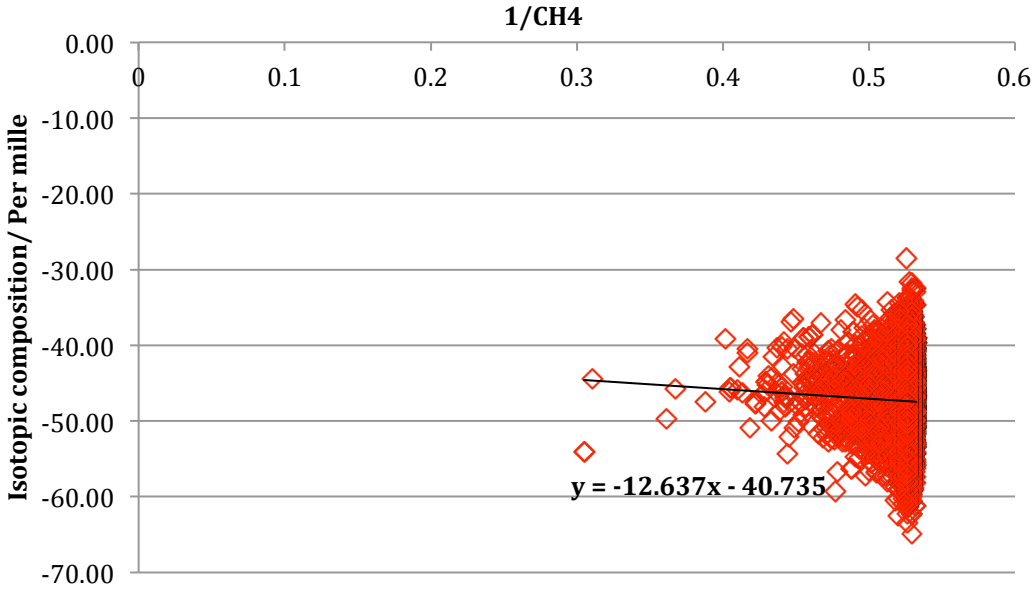

Figure 3.10: KM5 wellpad isotopic composition from 13/10/2015

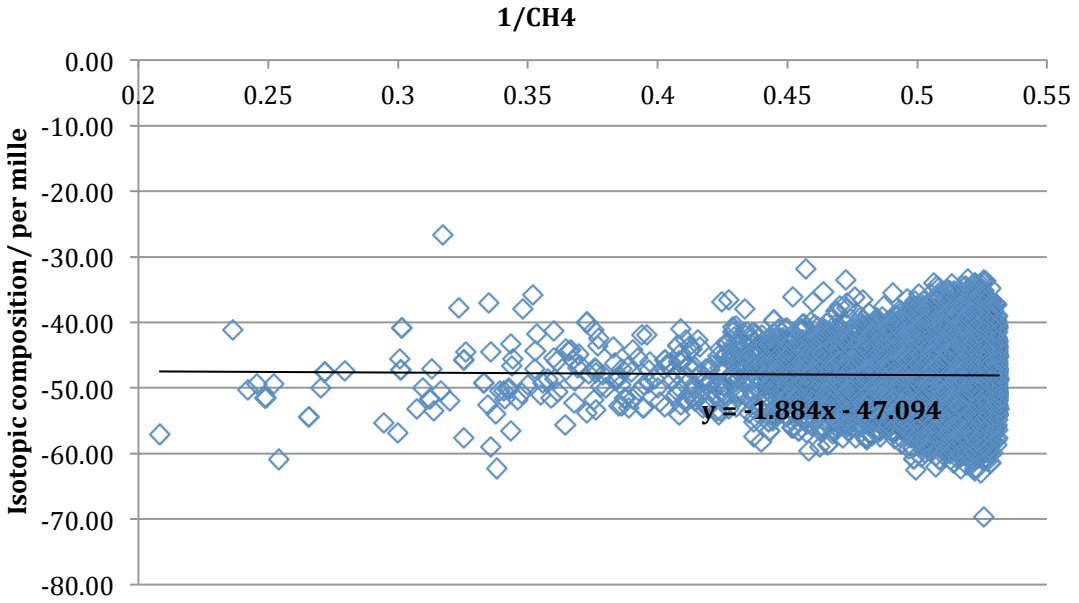

Figure 3.11: KM5 wellpad isotopic composition from 20/10/2015

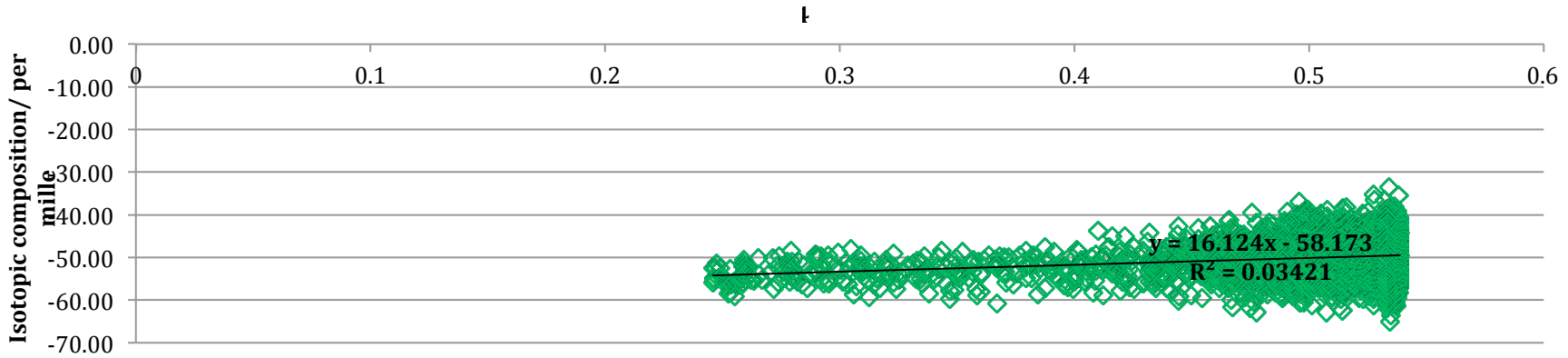

Figure 3.12: KM5 wellpad isotopic composition from 20/7/2016

Therefore when comparing KM5 methane concentration to the control, there is evidence of a weak thermogenic detection on one occasion at least (Figure 3.10) but methane emissions from the KM5 well are not large enough to detect a distinct point source. Nevertheless, when isotopic composition from each wind direction is compared to the Cranford Farm control site (Figure 3.13), methane emissions from the KM5 site were measured as be marginally isotopically heavier and hence more likely to have thermogenic provenance.

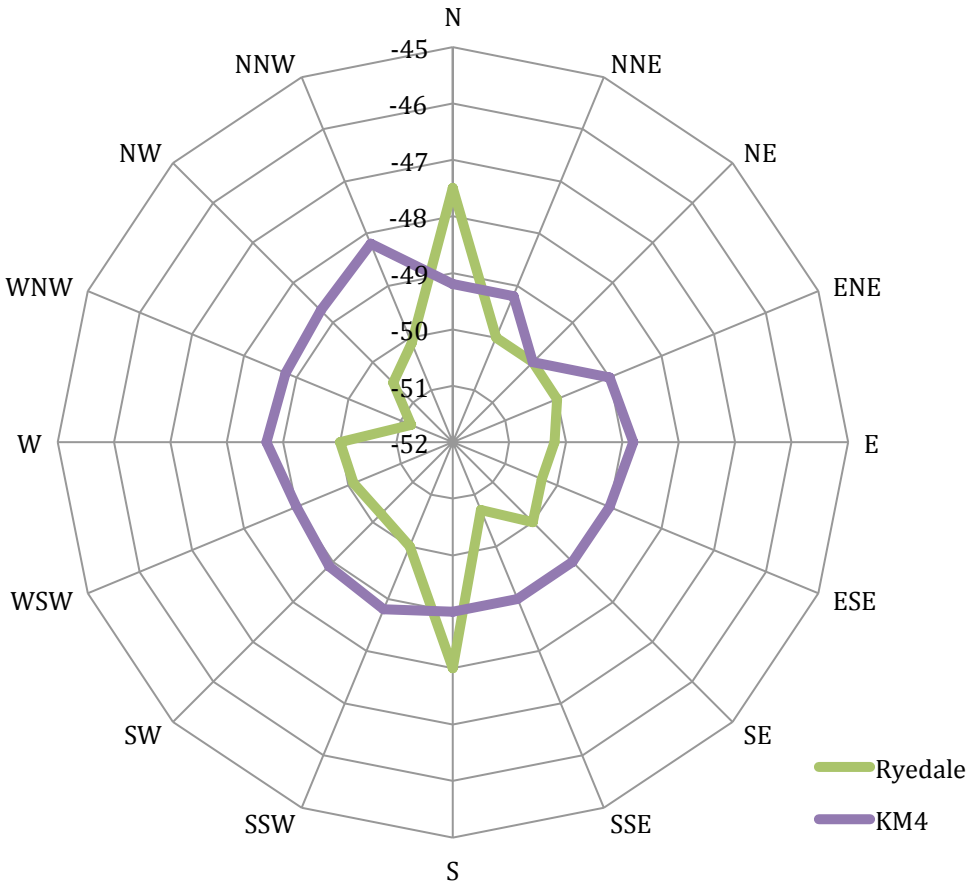

Figure 3.13: KM5 (KM4) and Cranford Farm (Ryedale) control isotopic composition

3.3: Knapton Results:

Knapton generation station had a higher methane concentration than KM5 and Cranford House (Figure 3.14). Methane measurements were taken from Knapton generation station in February, April and July 2016. The first two sampling days were distinctly different from each other, with the 1/2/16 sampling session not recording any methane above ambient concentration of 1.8-2.0 ppm. On the 2/2/16 the Picarro detected methane at much higher concentrations of up to 16.7 ppm CH_4, with an average of 7.5 ppm CH_4 over a 7 hour measurement (Figure 22). This above ambient methane concentration was detected as a thermogenic source (Figure 3.16), with an isotopic composition of -30 ‰, and when scaled up equates to a flux of 2 tonnes CH_4/year. Sampling from the 28/4/2016 also revealed a thermogenic source (-33 ‰), but its flux was less than 5% of the flux calculated from the 2/2/16. The 25/7/16 and 26/7/16 sampling days had average methane concentrations of 1.89 and 1.96 ppm, with the later neutral isotopic composition shown in Figure 3.18. The 29/4/16 data collection at Knapton detected an above ambient methane concentration with a distinctive thermogenic source (Figure 3.17), but only in the morning (0900-1200), when it was 2.76 ppm. Therefore, data from the 2/2/16, 28/4/16 and 29/4/16 were the only distinctive thermogenic sourced methane recordings at Knapton generation station, with both isotopic compositions at or isotopically heavier than -33‰. The variation in concentration detected is likely a function of site activity, which was not a variable within our control in data collection.

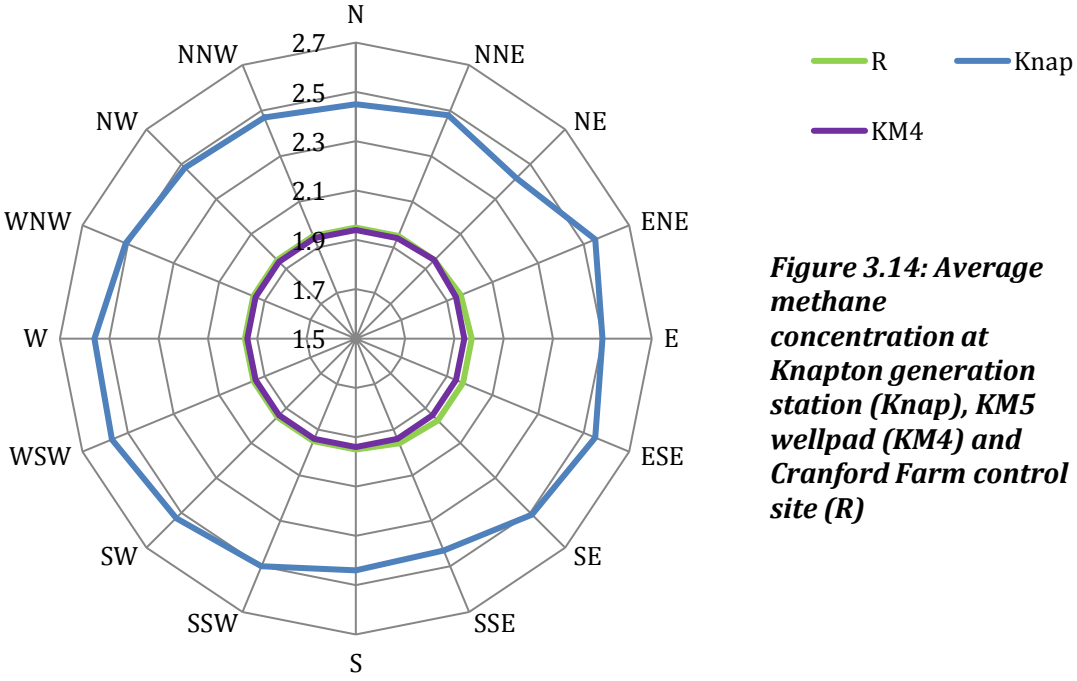

Figure 3.14: Average methane concentration at Knapton generation station (Knap), KM5 wellpad (KM4) and Cranford Farm control site (R)

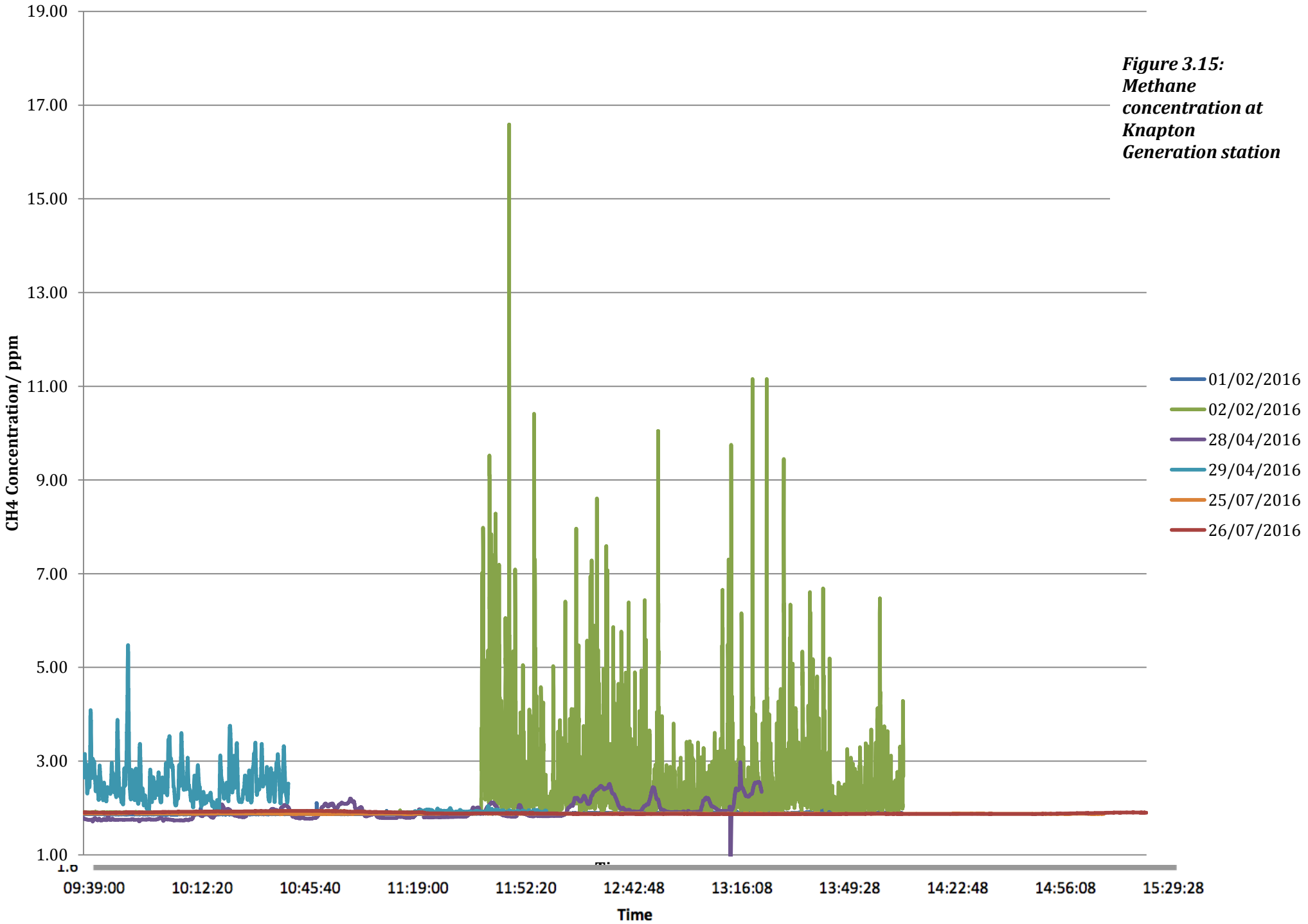

Figure 3.15: Methane concentration at Knapton Generation station

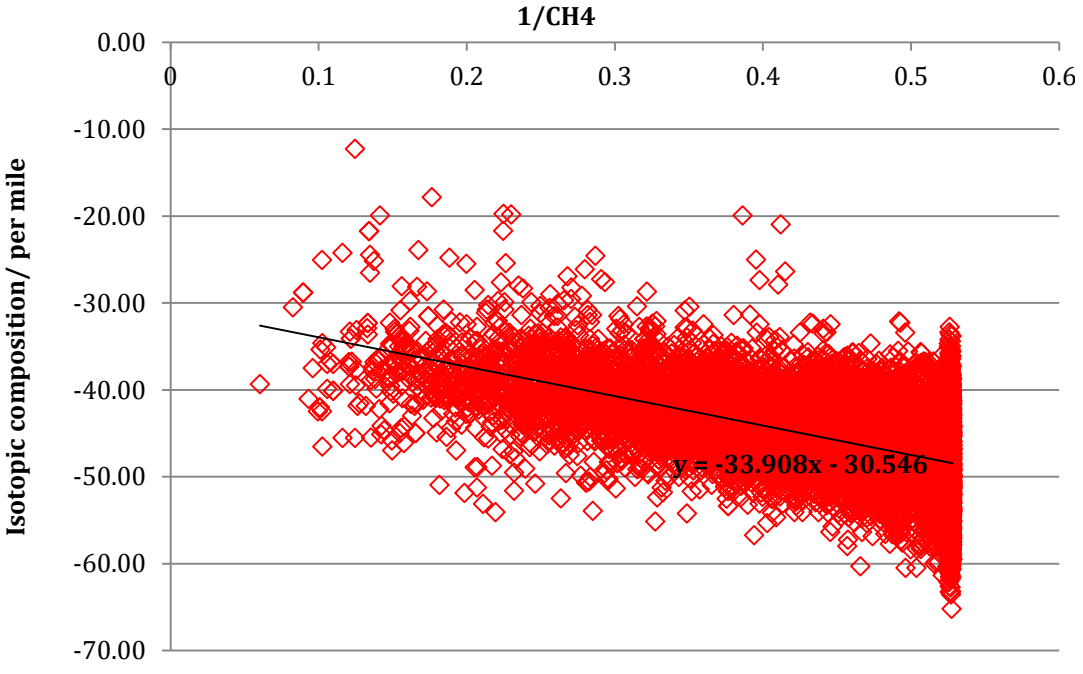

Figure 3.16: Isotopic composition of Knapton methane measurements from the 2/2/16

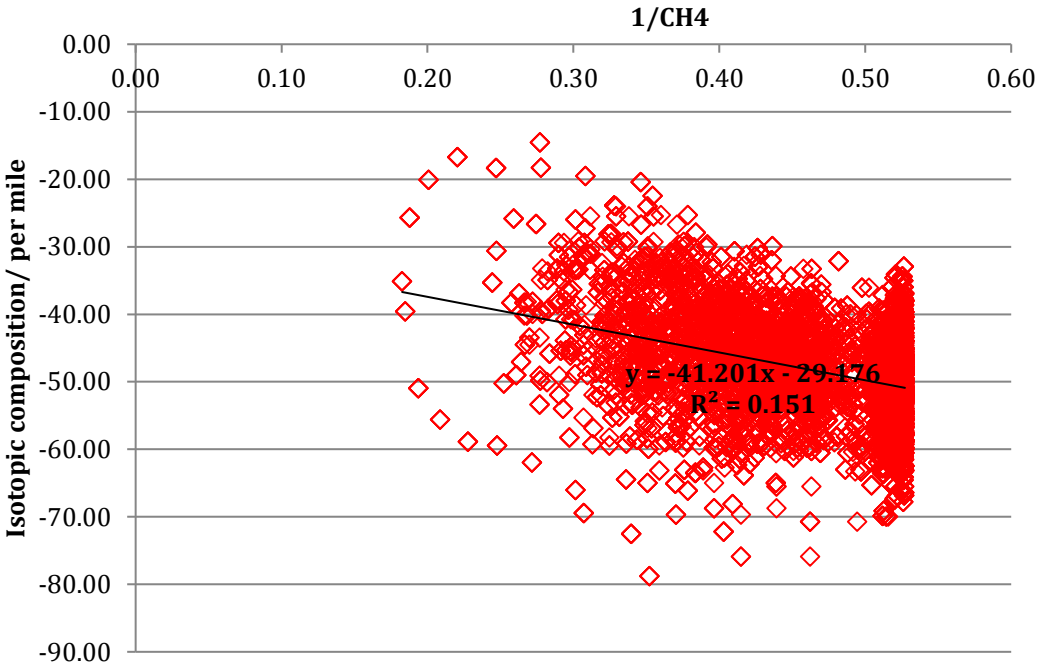

Figure 3.17: Isotopic composition of Knapton methane measurements from the 29/4/16

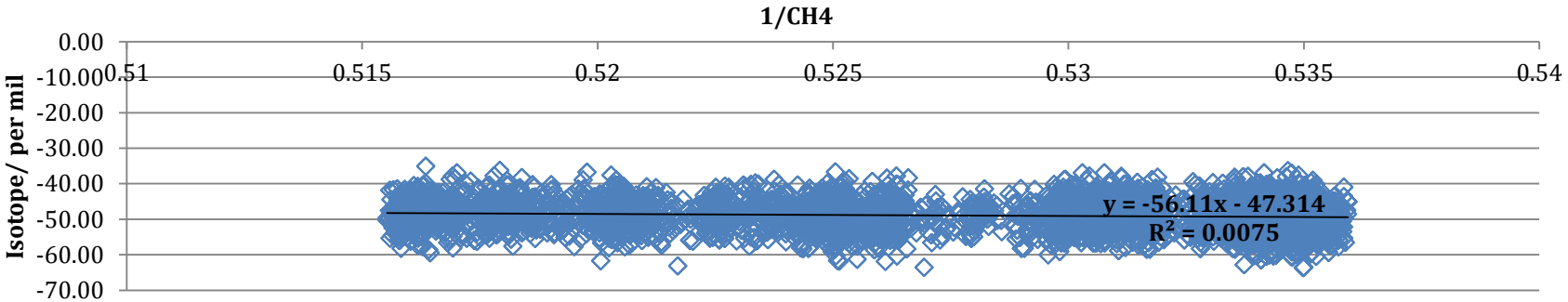

Figure 3.18: Isotopic composition of Knapton methane measurements from the 26/7/16

When comparing the average isotopic composition of each site, as shown (Figure 3.19), it is evident that Knapton generation station station site emits methane which is isotopically heavier than the Cranford House control site, and also of the KM4 wellpad. This is suggestive that the methane detected at Knapton has thermogenic provenance.

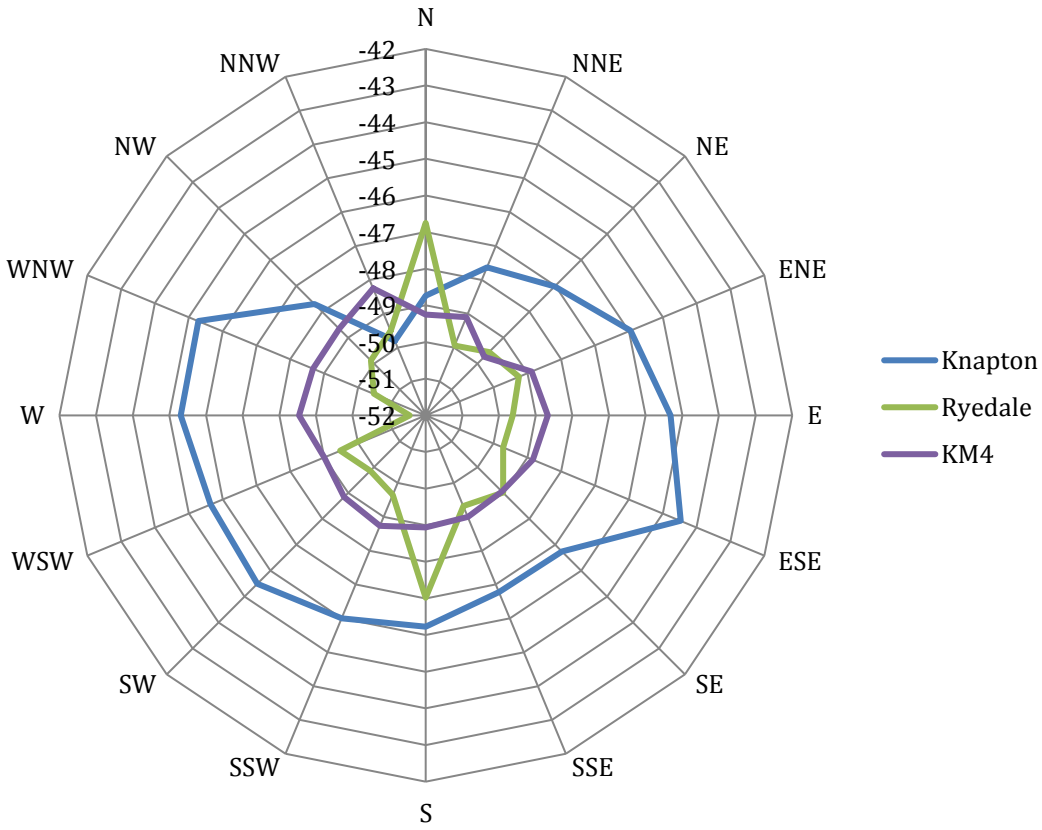

Figure 3.19: Isotopic composition of Knapton generation station (knapton), KM5 wellpad (KM4) and Cranford House control site (Ryedale)

3.4: Flux calculations:

A flux was calculated from the upstream natural gas infrastructure using a 3-D Gaussian plume model and the results are summarized in Table 3.1.

	Average concentration/CH_4 ppm	Average flux (kg/year of CH_4)	Flux range (kg/year of CH_4)
KM5	**1.9125**	**24.29862**	**0.25-106.74**
Knapton generation station	**2.005**	**847.914**	**3.86-2820.02**
Cranford House (control)	**1.93**	*	*

Table 3.1: KM5, Knapton generation station and Cranford House flux calculations

The results show that although all three sites have similar methane concentrations, but the fluxes between the two sites where a flux calculation was possible display noticeably different ranges. Knapton generation station was calculated to emit up to 2820.02 kg CH_4/year compared to the maximum calculated flux at KM5, which is 106.74 kg CH_4/year.

There is seasonal fluctuation in these flux numbers (see Appendices for individual methane concentration, flux calculation and isotopic composition), and the function of each site is different with larger amounts of gas passing through Knapton generation station than KM5. It is therefore important to calculate an emission factor per site, taking into account the fugitive methane emission made relative to volume of gas processed or produced per year.

3.5: Emission Factor

Operating under the highest emission calculation, KM5 would emit 106.74 kg CH_4/year as fugitive emissions and the KM5 well pad produces 1.968 million m^3/year. If the methane density of natural gas is 0.7 kg/m^3 then;

1 Kg CH_4 =1.428 cubic metres CH_4

106.75 Kg CH_4 = 152.50 cubic metres CH_4

152.50/1.968 million =

(0.00007749) * 100 = **0.007749% leakage rate for KM4**

For Knapton, again under the highest emission calculation, 2820 Kg CH_4/ year would be emitted from the site as fugitive emissions and Knapton generation station combusted almost 5 million m^3 of natural gas per year.

2820.02 kg CH_4 = 4028.6 m^3/year

(4028.6/4,921,468) * 100 = **0.0818% leakage rate per year for Knapton**

Calculated emission rates from Knapton are two orders of magnitude larger than that calculated for the KM5 emission rates, however both are well within one of the strictest leakage threshold of 3.6% which justifies the coal to gas shift (IEA, 2015).

4.1: Why is it important to control Methane emissions?

Methane is the second most important greenhouse gas after carbon dioxide, and in 2015 60% of global methane emissions had anthropogenic provenance (EPA, 2016). From 1990-2014, CO_2 contributed 72% of the total atmospheric warming potential, while methane and NO_x contribute 18% and 9% respectively (EPA, 2016). Methane is more effective at trapping heat than CO_2; however its short residence time means this effect is short lived, typically 8-12 years, before it is broken down in the atmosphere into water vapour and CO_2 (EPA, 2016). Ultimately, this means it has a GWP of 105 over 20 year and 33 over 100 year timescales (with an uncertainty of 23%) which has been recently upgraded from GWP factors of 100 and 25 respectively following the updated knowledge of methane interaction with atmospheric aerosols and ozone, which enhances its warming factor (Shindell et al. 2009). Methane therefore dominates the warming potential over 20 year timescales by a factor of 1.4-3 when compared to the impact of CO_2, and Nisbet et al. (2000) emphasise the importance of using the 20 year timeframe given the need to reduce global warming over decadal scales. However 20 year timescales are not appropriate when isolated as they do not take into account the short residence time of methane, and hence the natural reversibility of temperature increase over shorter periods (Cathles et al. 2012). The assumption however that releasing a tonne of methane is 25x worse than burning it and producing CO_2 is incorrect, as a tonne of methane combusted produces 2.75 tonnes of CO_2, so it is in fact 9 times worse (Joffe, 2013). Overall, a lack of definitive evidence of near term non-linearities makes the 100-year GWP more appropriate for policy decision (Allen et al. 2013; Broderick et al. 2011).

If the large estimates of leakage rates (up to 12.5%) from the natural gas industry are correct, the suggestion that natural gas use would nullify the benefits of switching to natural gas is possible considering the volume of global natural gas production and impacts of methane over short-term scenarios. However, these estimates have been heavily criticised for assuming worst practice and not taking into account 'green' applications, such as the 'NatGas Star Program' in the US (Cathles et al. 2012). Estimates for natural gas emission factors from upstream and downstream sources range from 0.45% to 12% of total production, and a suggested threshold rate of up to 3.6% justifies natural gas use as a bridge fuel, spanning the transition to a low carbon future (Allen et al. 2013). However, Muller & Muller (2015) from the centre of policy studies suggest that emission rates from natural

gas would have to exceed 12% in order to match the warming potential of coal use over 100 year timescales. Whatever the emission rate, it is at least clear this is an emission rate needing clarification (PA EPA, 2015).

Howarth et al. (2011), among others, stress the importance of minimising methane emissions from all sectors in the US. Turner et al. (2016) reported a 30% increase in atmospheric methane concentration between 2002 and 2014 in the US, although the paper did not attempt to assess methane provenance. Hausmann et al. (2016) estimated that 40% of the total growth of atmospheric methane between 2007 and 2014 could be attributed to the shale gas expansion. It has been suggested that this has been caused by fugitive emissions from leaking equipment, during well completion and through venting, but Wood et al. (2011) state that although emissions from these practices are likely, there is no reliable dataset to quantify the impact. The Pennsylvania Department of Environmental Protection admit it is likely the reported increase in atmospheric methane is partly due to shale gas development, but highlight that methane emissions from hydraulically fractured wells in the Marcellus shale have decreased by 73% since 2011 (PEPA, 2015). A 1068% increase in natural gas usage from 2008-2012 was accompanied with a 78% decrease in SO_2 emissions, a 46% reduction in particulate matter output and a 23% decrease in NO_x, and so fugitive emissions would not negate the shift to natural gas from coal. Nevertheless, natural gas and petroleum systems marginally represent the largest proportion of the US methane budget at 33%, closely followed by enteric fermentation and manure management at 30% (EPA, 2016).

Although historically methane emissions were not of great concern, the increase in CO_2 emissions over the past 3 centuries has put the global climate system in a more precarious state, relatively lowering the threshold capacity for the climate to respond to increased and sustained anthropogenic methane emissions (IPCC, 2014). The United Nations Environment Program (2016) state that under current GHG emission rates, without abatement, the increase in global mean temperature will reach 1.5°C by 2030 and the increase is likely to exceed 2°C before 2050. Initial increases in global temperature about 0.5°C have actually brought about increased CO_2 sequestration from vegetation as the temperature increased the rate of photosynthesis. However, recently the temperature increase has crossed the productivity threshold and has increased the frequency of drought, contributed to sea level rise and has been a causal factor in recent wildfires in Alberta, California and Alaska (Kirschbaum, 2004; IPCC, 2014).

A factor of great concern is that a global warming increase of 1.8°C is likely to be adequate to trigger a release of methane from artic permafrost, one of the largest carbon stores on earth. However, there is dispute over the potential rate of release of methane frozen within the ice clathrate under different warming scenarios (Hansen et al. 2005; Zimov et al. 2006; Hansen et al. 2007). The risk should not be ignored, as the IPCC (2014) report up to 1,400 gigatons equivalent of carbon is stored in the arctic as methane which if released would be the equivalent of almost 88 years of total CO_2 emissions from both China (9000 Mt/year) and the USA (7000 Mt/year) (Shakhova et al. 2008). Also, 5-10 % of arctic permafrost land is open, non-frozen land called taliks, and Shaknova et al. (2007) concluded that the abundance of these features provide a pathway for methane release of up to 50 Gt CH_4/ year, which would increase atmospheric abundance of methane by a factor of 12 and the high methane GWP would result in rapid, uncontrollable temperature rise. This is known as the clathrate gun hypothesis, and its severity is almost entirely dependent on the rate of release of methane, which is controlled by atmospheric temperature, hence is dependent at least partially on anthropogenic activities (Greinert, 2012). Such an event was said to have occurred in the Palaeocene-Eocene Thermal Maximum 55 million years ago where average global temperatures were over 4 °C warmer than currently, and was sustained for over 150,000 years (IPCC, 2014). The US department of Energy National Laboratory and the US Geological survey have identified methane hydrates on the shelf of the east arctic seas as a potential trigger and stated this hypothetical climate scenario to be one of the four most threatening climate risks of the 21st century. This has been of increasing urgency as a recent study by Rogelj et al. (2016) stated that the window for limiting global warming increase below 1.5 °C without temporarily exceeding it has closed, that 2/3 of the carbon budget to keep warming below 2 °C has already been emitted and that if global temperature increases are to be maintained below 2 °C as discussed in the Paris Agreement, the globe must have zero net CO_2 emissions by 2050 (Matthews et al. 2008; Meinshausen et al. 2009; IPCC. 2014; Knutti et al. 2015).

The UK emits less than 1.5% of global carbon emissions, constrained as 491 Million Tonnes of CO_2 per year (Mt CO_2e), which is a fraction of the emissions of China, USA and India (DECC, 2015). In terms of methane emissions, there is disparity compared to the US example, initially as methane emissions have been consistently falling since 1990 when emissions were 138 Mt CO_{2eq}/year, while in 2013 methane emissions were below 60 Mt $CO_{2eq,}$ a drop of over 56% (DECC, 2014). Unlike that of

the US, natural gas systems (onshore and offshore) do not dominate methane emissions in the UK, representing 14% of the annual total, compared to waste management which represents 37% and agriculture which accounts for 48% of UK methane emissions (Ricardo-AEA, 2015). This is shown in figure 4.1 below

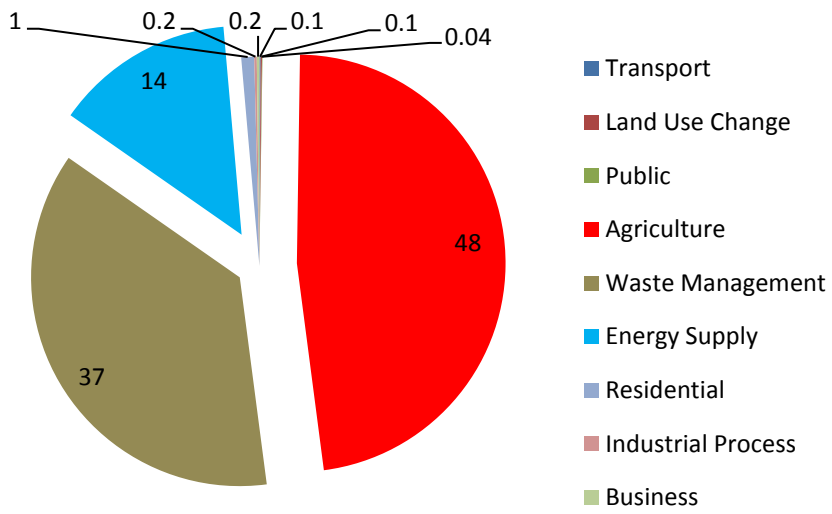

Figure 4.1: UK methane emissions by sector (DECC, 2015)

Since 1990, landfill CH_4 emissions have been reduced by 73% (57% of total reduction) and fugitive emissions from fuels have been cut by 75.6% (38.4% of total reduction), but emissions from agriculture have decreased by less than 18%, equivalent to 7% of total reductions since 1990 (UK GHG Inventory- DECC). At the very least, this must raise awareness that for the UK, waste management and animal agriculture sectors dominate the methane flux. Further, when taking into account the cradle to grave emissions of animal agriculture including the loss of forest land displaced for grazing animals or growing feed, the agriculture industry represents 51% of net global emissions, and even if fossil fuel production was to stop today, global CO_{2eq} emissions for 2030 will be exceeded purely from agriculture sourced anthropogenic emissions (Goodland et al. 2009; Oppenlander, 2013).

By no means does this suggest that lower methane emissions from natural gas sources justify the exploitation of shale gas, as unabated unregulated methane emissions would contribute to the emission factor. Natural gas usage would have to displace coal, a process already occurring in the UK, with natural gas usage increasing from 5% in 1970 to 47% in 2015, at the same time coal usage has

decreased from 70% to 16% (DECC, 2015). The overall UK CO_2 emission rates have fallen as a direct result of the transition from coal to gas, and at the same time methane emissions have been quartered. As long as fugitive methane emissions can be controlled, natural gas is appropriate from a UK and global climate perspective. The coal to gas transition has not been widespread however, with 95% of countries outside Europe and 1/3 of countries in Europe having greater 5 year coal consumption figures from 2010-2015 when compared to 1970-1975, with 2 EU countries (Greece and Portugal) having increased consumption by over 300%, likely due to their economic crises (Mc Glade et al. 2016). Appropriately, the natural gas industry is the subject of continuous assessment and critique in order to minimise methane emissions, but at current trajectories due to increased demand, global emissions from animal agriculture will rise by 80% by 2050 if the current regulation of agriculture methane emissions, or rather lack of, continues (Clark & Tilman, 2014).

Overall, at this heightened level of climate risk, large positive perturbations to the global methane budget pose a significant climate hazard and ideally any potential increase to the methane budget, such as the exploitation of unconventional hydrocarbons should be met with decrease from other sources. Critically, the benefits of methane reduction in the atmosphere would have very rapid impacts on temperature change over decadal timescales, unlike that of CO_2 (Howarth, 2015).

4.2: How is Methane measured and provenance assessed?

The concentration of methane in the atmosphere is currently ~1850ppm which is a 150% increase in concentration since the onset of the industrial era in 1750 and accounts at least for 17% of the increased atmospheric radiative forcing (Allen et al. 2013). Throughout the 800,000 years before 1800, proxy data shows that atmospheric methane concentration fluctuates between 400 and 800pm, but at no point exceeds 800 ppm (NOAA, 2015). From 1970-1990, Mauna Lowa station atmospheric methane concentration increased arithmetically at 1-2 % per annum, before a stagnation of concentration at ~1750 ppm throughout the 1990's (Dlugokencky, 2003; Turner et al. 2016) and finally followed by an increase in concentration throughout the 21st century to a contemporary concentration of 1850 ppm (Rigby et al. 2008; NOAA 2015). The recent upward trends in atmospheric methane abundance have been suggested to be caused by increased emissions from hydrocarbon exploitation across the USA (Wang et al. 2004; Franco et al. 2015) and microbial sources (Levin et al. 2012), while the accelerating rate of increase has been further suggested as an amplifying positive feedback process due to increasing methane emissions from warming wetlands (Bousquet et al. 2011; Pison et al. 2013). Contemporary global methane emissions are estimated at 550 ± 60 Tg/year, however the large population of diverse sources overlap spatially which promotes uncertainty in national measurements (Prather et al. 2012; Dlugokencky et al. 2011; Allen et al. 2013).

In 2012, President Obama pledged to reduce methane emissions from the oil and gas sector by 40% from 2012 levels by 2025, which increased and accelerated the scientific attention of this greenhouse gas (US EIA, 2016). It is therefore very important to quantify the methane emission factors; both anthropogenic and natural. As described previously, this can be conducted by broad-brush top down measurements or specific bottom up measurements, which have advantages and disadvantages as shown in the summary overleaf;

	Top Down	Bottom Up
Area measured	Large: Total shale plays	Small: well pad or transmission centre
Source	Difficult to attribute exact source	Can determine gas provenance and source
Time&Financial Cost	1 off expense: Plane trip etc	Multiple sites: high cost
Representation	Specific time period concentration	Continuous measurements at site
More likely to	Overestimate CH4 flux	Underestimate CH4 flux

Table 4.1: Top down vs bottom up means of methane measurement

Source: Broderick et al (2012), CCC (2016) and Allen et al. (2013)

The disparities in top down and bottom up approaches are of concern to this investigation as the conflicting claims about which method is more appropriate has implications for the estimation of the true methane emissions and therefore whether or not natural gas emissions negates the shift from coal.

In a recent EDF (Environmental Defense Fund) sponsored investigation of oil and gas CH_4 from the Barnett Shale, Texas, a coordinated investigation involved a range of universities (Houston/Dallas) and companies (Picarro) taking measurements at almost 600 production sites, compressor stations and processing plants (bottom up) and aircraft based near field and regional measurements (top down) (Araiza et al. 2015). The aim of the study was to construct a bottom up inventory which is reflective of the entire facility population and complementary to the top down results (Araiza et al. 2015). The Barnett was chosen as it is one of the most active oil and gas regions of the US, providing 7% of US gas production (Araiza et al. 2015). Data from previous studies suggested 79.5 % of Barnett Shale emissions originate from fossil sources using ethane abundance downwind of potential sources as an indicator, as ethane is the second most abundant hydrocarbon in natural gas, and has no known biogenic source (Smith et al. 2015; Karion et al. 2015; Araiza et al. 2015). Typically bottom up measurements rely on published emission factors due to high costs from measuring every source individually, but this EDF study did not use the recently decreased emission factor from the EPA (EPA 2015; Araiza et al. 2015). The results from the data collection is shown overleaf and expressed in kg CH_4/hour:

Source	Barnett Study	EPAModel facility	This Study
Production sites	1.76	0.47	**0.0028**
Compressor Stations	**64.2**	3.63 - 6.46	**36.75**
Processing Plants	195	N/A	**0.097**

Table 4.2: Methane emissions from Barnett, EPA and this research

The EPA model facility factors are estimates intended to be representative of emissions from the average facility, however this data shows that all measurements from the EDF investigation exceed EPA emission factors, but also that emission rates are dominant in compressor facilities and processing plants compared to the well pad. The results from data collection at sites in N. Yorkshire are shown also. The production site (KM4) measurement is over 2 orders of magnitude less than the EPA model facility and the compressor station average (combination of MM and TL) is over 10 times the lowest EPA average estimate but just under 50% of the EDF study from the Barnett shale EDF study.

Using mass balance, the study produced top down emissions from 7 flights on different days within the Barnett shale. There was excellent agreement between top down and bottom up emissions, with a difference in central estimate values of less than 0.1% for total methane emissions and 10% for fossil methane emissions (Araiza et al. 2015). The bottom up estimate for methane emissions from these sites was 59000 Kg/hour, equivalent to a loss rate of 1.5%, compared to model EPA upstream and processing losses of 0.6%, which is 60% lower than the EDF study (EPA 2015; Araiza et al. 2015).

Reasons are given to explain the higher emissions, and they highlight the issue of super-emitters as 2 % of natural gas facilities in the Barnett produced 50% of total emissions and 10% of sites were responsible for 90% of the emissions (Araiza et al. 2015). A non-normal emission distribution suggests that the bottom up collection was representative of the effect of skewed emission distributions by measuring high methane emitting sites. These super-emitters are described as 'temporally and geographically dynamic' with the high emitting sites changing over time, and are responsible for the fat tail log normal distribution of methane emissions from

oil and gas facilities (Allen et al. 2013; Araiza et al. 2015; O'Sullivan et al. 2012). If these facilities displayed a normal distribution, national extrapolation would be simple, but these super-emitters, which in this case were more likely to be to be a processing or compressing facility, can distort averages significantly and are hampering methane emission control policy (US EIA, 2015). The EDF study estimates that 30% of the production sites emit just over 1% of the gas entering the facility and these sites account for 70% of production site emissions (Araiza et al. 2015). The results from the EDF study are complementary to a sample of 115 wells from the Barnett shale, Texas in which it was reported the top 20% of sites accounted for over 60% of total methane emissions (Rella et al. 2015).

Another top down and bottom up Barnett Study conducted by Hariss et al. (2015) estimated methane emissions to be 50% higher than EPA estimates, but the largest contribution was from a greater number of compressor stations, which have been rapidly constructed alongside the shale gas revolution. The revised 2016 EPA emission statistics increased estimated compressor emissions in line with new evidence (Harriss et al 2015). Overall Harris et al. (2015) found shale oil and gas emissions account for 1.2% of production volume, and when oil sites were excluded the natural gas emission factor dropped to 1.1%. Similarly, in April 2015, Frankenberg et al. (2016) conducted airborne measurements of the Four Corners region of the US, which is the state border between Arizona, Utah, Colorado and New Mexico and is underlain by the San Juan Basin, which produced 36.8 billion cubic metres in 2014 from its 40,000 gas wells (EIA 2015). The large plume from this area was first identified in 2003 and has been suggested to account for 10% of annual USA methane fugitive emissions (Kort, 2014). A 2500- square mile plume with a total point source flux of ~0.3 Tg/year was detected in the Four Corners region, with the top 10% of emitters contributing up to 66% of this flux (Frankenberg et al. 2016). Of the sites measured, which include gas processing facilities, pipelines and well pads, the flux ranged from a low of 2 kg/hr up to 5,000kg CH_4/hour, and at highest flux was an order of magnitude less than the peak 60,000 Kg/hour CH_4 flux of the Aliso Canyon gas storage facility in California (Conley et al. 2016). Kort (2014) suggests that the plume is due to shallow coal bed methane extraction which is more prone to cause methane migration and emissions, and Levi (2012) suggests that both natural seepage emissions from Fruitland coal formation, and drilling through this layer is likely to further release more methane.

Omara et al. (2016) investigated super emitters from the Marcellus shale in the 3386 unconventional wells and 88,536 conventional wells across Pennsylvania and West Virginia. On average, 491 Gg CH_4 is emitted from all unconventional wells and 657 Gg CH_4 is emitted from the 26 times as many conventional wells in the same region per year (Omara et al. 2016). Interestingly, the CH_4 emissions of the top 15% of emitters from both conventional and unconventional revealed both represented over 60% of total methane emissions, highlighting the super-emitter to be an issue with all forms of natural gas production. Over the wells measured in the Marcellus, the combined unconventional and conventional emission factor was 1.4%, which is higher than Peischl et al (2015) estimate of up to 0.41% but much lower than Caulton et al (2014) analyses which suggest emissions are from 2.4% to 17.3%, however this study took into account emissions from oil well emissions. Also, although the much greater numbers of conventional wells makes it appear as if unconventional wells have high emission factors, in terms of gas production in the Marcellus region, conventional wells provided 5% of production compared to the 95% (4.8×10^9 MSCF) provided by unconventional wells. Therefore, total CH_4 emissions from conventional well pad sites were 57% of total methane production (660Gg) compared to the 490 Gg (43%) from unconventional wells.

The study conducted in North Yorkshire only took into account a single conventional well system, and was not able to quantify the impact of super-emitters, but they should be the subject of greater research. The lack of spatial pattern in US super-emitters therefore would require robust methane measurement portfolio at each site or at least the majority of sites in the UK.
If this study was to be repeated, it would be useful to measure methane emissions from a larger range of well pad sites to determine if any methane flux variation occurs nationally. The potential of these high emitting sites should be taken into great consideration for a UK perspective, as the US EPA adopts a process based approach which assumed a normal emission distribution for the life cycle emissions of shale gas production and use, which these studies have shown is not appropriate for methane emission measurements for the natural gas industry (Frankenberg et al. 2016).

Alvarez et al. (2012) used a Technology Warming Potential (TWP) framework to assist in quantification of radiative forcing from increased methane emissions, and this was adopted in the EDF investigation. The TWP is the ratio of total cumulative

radiative forcing from life cycle GHG emissions, including direct, indirect and fugitive emissions (Alvarez et al. 2012). They estimate that each percentage of raw methane gas lost to the atmosphere increases the radiative forcing on top of CO_2 emissions from combustion by ~30% over 20 year time scales or 10-15% over 100 year time scales (Alverez et al. 2012; Araiza et al. 2015). Under this model, CH_4 leakage from the Barnett Shale of 1.5% would increase the radiative forcing when compared to CO_2 emissions alone by ~50% over 20 year models or 20% over 100 year period, and this emission factor is low enough to support the shift to natural gas use compared to the radiative forcing from coal fired power plant emissions for electricity generation (Araiza et al. 2015). This study had good top down-bottom up consensus, however there can be significant variations in methane emission rates from other gas producing regions for reasons such as geology and practice, and the greatest variation will be from intermittent high emitting practices such as well completion and liquid unloading and the impact of extensive pipelines providing gas to regions far away from production sites (Howarth et al. 2011; Araiza et al. 2015; Jackson et al. 2014).

Shale rich areas across the US have been utilized for their former natural resources such as coal and conventional hydrocarbons for over 200 years. Thus, there can be other thermogenic sources of methane, such as abandoned wells, over 1,000,000 of which were drilled in the US before abandonment legislation came into action. Kang et al. (2014) also suggests that typical current and previous practice prioritises reserve conservation and groundwater protection over methane emissions. Subsequently, a bottom up study from Townsend- Small et al. (2016) into methane leaks from 138 abandoned US oil and gas wells revealed that; most wells do not emit CH_4, 6.5% of wells had measureable methane emissions, and 25% of wells that had not been plugged emitted more than 5 g CH_4 /hour. Isotopic analysis of the samples reveal that these decommissioned wells did emit thermogenic methane; however the contribution from these wells to the anthropogenic methane budget was less than 1% (Townsend-Small et al. 2016). Similarly, in a UK study conducted by Boothroyd et al. (2016), an analysis of 103 decommissioned UK onshore wells revealed that 30% had methane concentration above the control site and 39% of well sites had lower surface CH_4 concentration than the control. It was noted that where integrity failure occurred, it did so within 10 years of being abandoned, and the conclusive calculation revealed a flux of 364 +/- 677 kg CO_2 eq/ well/year, with the potential of being a net sink (Boothroyd et al. 2016). Post abandonment, the

methane emissions from these wells would be comparable to that of 2 sheep (Boothroyd et al. 2016). There has also been evidence of both methane in artesian wells from the 18th century in the US (Osborn et al. 2011).

In shale gas wells, Osborn et al. (2011) suggested that poor casing quality led to methane migration into groundwater, which has been extensively reported and can be emitted at the surface via conduits and ultimately contributing to the methane budget. However, Davies et al (2011) highlights Osborn et al (2011) did not take into account the impact of; 184,000 wells drilled before records were kept, natural seepage and the 8,000 wells which have not been plugged, all of which would likely contribute to contemporary methane concentration in groundwater sources and predate hydraulic fracturing. A large methane plume was recorded in the Marcellus during the drilling phase and was likely sourced from shallow coal methane or another source, biogenic methane, which is isotopically different to thermogenic methane. Biogenic methane produced from enteric fermentation of cattle produce at least 18% of global anthropogenic greenhouse gas emissions, which is more than the transport industry. Texas has over 13% of the US cattle population at almost 12 million, and a cow will produce 90-150kg of methane per year, meaning methane emissions of up to 1.8 million tones CH_4/year just from Texas, while methane emissions from the spatially resolved bottom up oil and gas measurements are 516,840 tones CH_4/ year, which is slightly less than 30% of methane produced purely from cattle in Texas (Drovers, 2015; Araiza et al., 2015). In other regions of the US, the proportion of biogenic methane production is significantly less, such as in Louisiana, North Dakota, Arkansas and Pennsylvania where cow population is less than 10% of national total, but these areas produced almost 80% of US shale gas in 2014 so it is more likely that larger atmospheric methane concentration increases in these areas could be attributed to the exploitation of onshore hydrocarbons (O'Sullivan et al., 2012; US EIA 2016).

Since 1990, Schaefer et al. (2016) have measured and collated global methane samples for isotopic concentration analysis. They concluded that the plateau of methane emissions from 1999-2006 was likely from diminishing thermogenic methane emissions and economic slowdown, but that the renewed CH_4 rise post 2006 was from predominantly biogenic sources, apart from in the arctic, where fossil methane has been released with global warming. With an average ^{13}C enrichment of -47‰ in the post 2006 methane increase, Schaefer et al. (2016) say

that either food production or climate sensitive natural emissions are the probable cause of the majority of the recent increase in methane emissions, which will require a more different and adaptive mitigation strategy to present.

The results from the data collection in North Yorkshire are a representation of methane emissions from a producing well pad and gas combustion site within a closed system, made relative to a rural control. In assessing what could have been done differently, it would have been extremely useful to have access to a UK shale gas well during the pre-production phase, as this would enable first hand bottom up measurement during drilling and well completion, as was conducted in the EDF study. An analysis of emissions from this process in the UK would enable more accurate projections of the greenhouse gas impact of shale gas extraction. A comprehensive constraint of methane emissions from pre-production well pads would enable primary data on UK shale gas wells to be analyzed to assess which REC technologies would be most applicable to each stage of drilling and hydraulic fracturing. As the UK does not currently have an extensive onshore hydrocarbon industry currently, with 120 sites producing up to 25000 barrels equivalent, it is reasonable to assume that emission factors of the other 119 sites would be similar by % methane loss, but it is not appropriate to scale up data from one site without further representative data collection (UKOOG, 2016). As the downstream pipelines system is the greatest emitter of methane in this study, further data collection of ageing and modern facilities would be useful to compare and ideally mitigate against. Flux data collected from other sites could be used in order to calibrate UK correct from future UK onshore shale gas sites.

4.3: How does Shale gas use compare to other energy sources?

Natural gas is being publicized as a bridge fuel, spanning the gap between coal, which is no longer appropriate if UK and global emission caps are to be met, and renewable energy, which is currently immature and incapable of meeting demand (Mackay and Stone, 2013). UK total energy usage including electricity, transport and heating has fluctuated about 200 MtO_e (million tonnes of oil equivalent) for the past 45 years as shown in Figure 4.2 below. In that time, coal use has decreased from 100 MtO_e to less than 30 MtO_e, while gas use increased from 5 MtO_e to over 70 MtO_e. Oil and gas have formed the bulk of UK energy sources since 1980 and in 2015 provided over 85% of energy usage, however the greatest shift from fossil fuel to renewable energy has occurred in the past 5 years (DECC, 2015). Nevertheless, wind, solar and hydro-electric energy provides less than 3% of total UK annual energy needs (DECC, 2015).

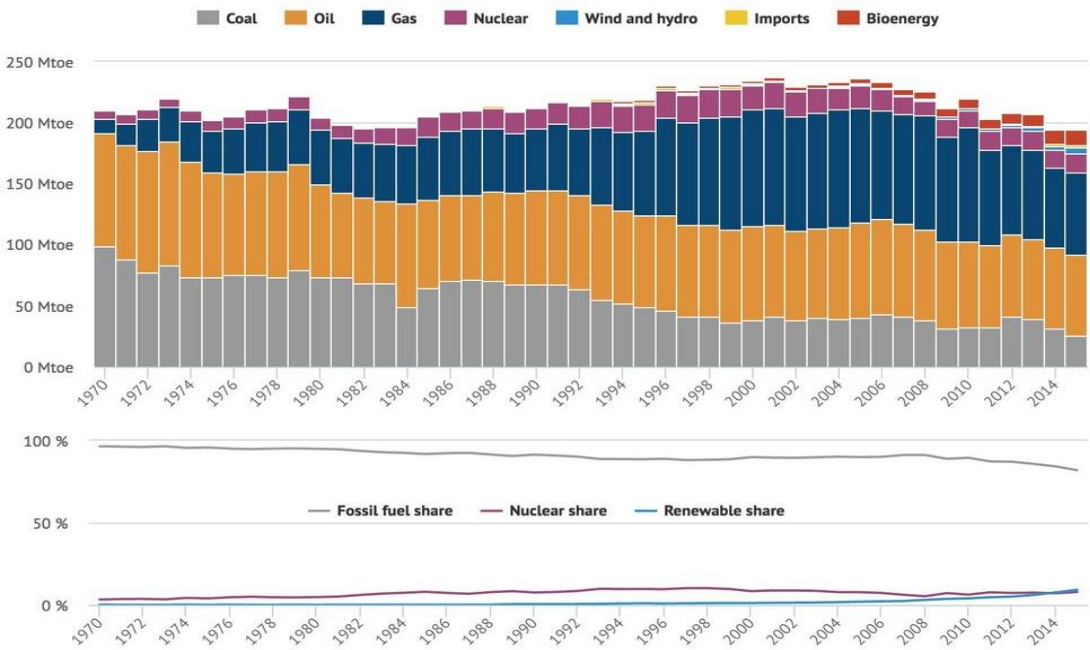

Figure 4.2: UK energy source by sector (Source: Carbon Brief)

The concept of energy and climate de-coupling has been the subject of great dispute, but as fossil fuels are certain to form the bulk of global energy generation by 2050 in a period of uncertain climatic shift, a comprehensive analysis of the emission factors of each fuel source is required in order inform policy.

Figure 4.3 below shows the life cycle emissions of each energy source when used to produce electricity, with the conversion factor $gCO_{2e}/kWh_{(e)}$ assuming a CH_4 GWP of 25 (Mackay and Stone, 2013).

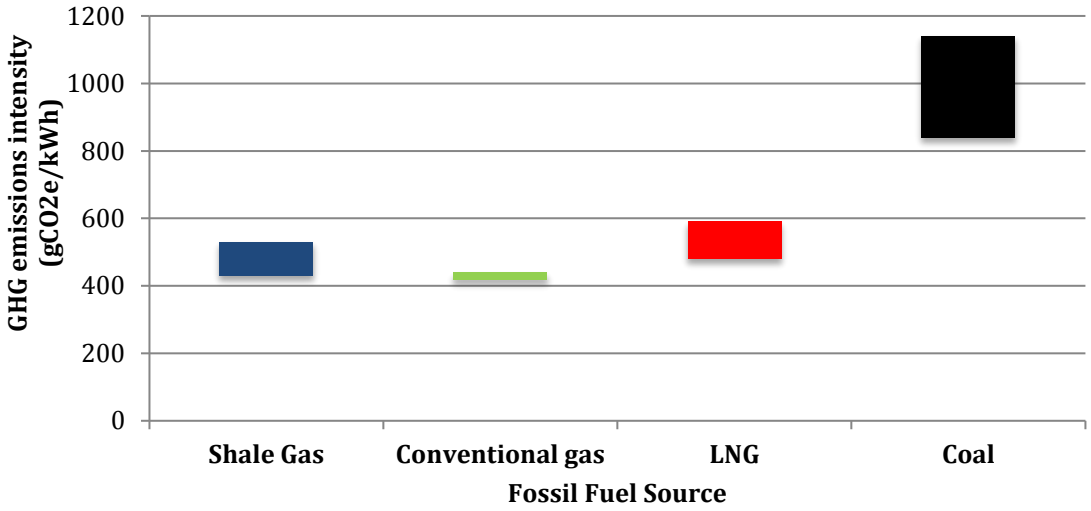

Figure 4.3: Emissions from fossil fuel sources (adapted from Mackay and Stone, 2013)

This shows that shale gas has an emission intensity of 430-530 $gCO_{2e}/kWh_{(e)}$, which has a higher upper limit than that of conventional gas at 440 $gCO_{2e}/kWh_{(e)}$ but lower than that of LNG (Liquified natural gas) which can have an emission intensity of up to 590 $gCO_{2e}/kWh_{(e)}$ (Mackay & Stone, 2013).

Gas that is transported by sea is almost always in the form of LNG in large tankers, this is done by liquefaction, lowering the temperature of the produced gas to -162°C, which takes up ~0.2% of the volume compared to CH_4 transported in its gaseous state (Honore, 2011). LNG is therefore more energy intensive due to the high energy requirements of cooling, storing and transporting the natural gas. The global production of LNG has increased gradually from 50 Mtpa (million tonnes/ year) in 1990 to 246 Mtpa in 2014, and Qatar, which has the world's third largest proven natural gas reserves, supplied 25% of global LNG in 2015 and 30% of UK natural gas imports (US EIA, 2015).

Henderson (2014) claims that the USA will likely expand LNG exports using gas from its shale plays, with the first ethane export transported to France in 2015, and an ethane shipment was delivered to the UK INEOS Grangemouth plant in Scotland in October, 2016 due to inadequate supply from North Sea production. US Shale gas

as LNG imported to the UK will have a life cycle emission intensity at least 10% greater than domestic production (Henderson, 2014).

The emission factor of coal is significantly higher than all other sources, with a emission intensity of up to 1140 $gCO_{2e}/kWh_{(e)}$, over double that of shale gas and this is dominated by the CO_2 output from direct coal combustion (Mackay and Stone, 2013). It is appropriate to compare gas to coal as the majority of US shale gas is being used to displace coal for electricity generation, and in the UK by 2025 all coal fired power stations are to be shut, which will require a substitute fuel which is likely to be gas (DECC, 2015).

The legitimacy of shale gas displacement of coal is therefore not an argument concerning emissions from its combustion, but Mackay & Stone (2013) state two key factors which could narrow the emission gap between shale gas and coal emissions;

1) **Well Productivity:** A lower EUR (annual well production) means the pre and post production emissions from shale gas development will a greater proportion of the total, with a 50% decrease in EUR doubling the specific emissions. Mackay and Stone divided this into three sections;
    - Low EUR: 2 bcf (57 million m^3),
    - Medium EUR: 3 bcf (85 million m^3) and
    - High EUR: 5 bcf (140 million m^3).

Among the concerns of shale gas exploitation, as the national expansion of this technology is occurring for the first time in the US, there is great uncertainty over resource scale. Newly drilled wells typically have very high initial production rates, followed by a rapid decline in production but as the shale resource depletion rate is not totally understood, it is difficult to draw definite conclusions concerning longer term gas production (Jacoby et al. 2012; Urbina, 2011; Lee et al 2011). The lack of long term knowledge of shale gas wells creates uncertainty in how much gas is produced over a well's lifetime, and therefore increases uncertainty in the emission factor (Mackay and Stone, 2013). Also the EIA and other groups and companies have been accused of over promoting longer term production rates to secure initial investment (Howarth, 2014). By contrast, there is low variation in

emission factors from Qatari gas wells as the average well productivity from the gulf state is 125 million m^3, so emissions from pre-production are proportionally limited (US EIA, 2015). In 2014, Pennsylvania conventional wells produced on average 2.6 million cubic feet per year, while production from a shale well was 1.5 billion cubic feet on average, but up to 6 wells per pad increase the relative EUR per pad (PA EPA 2015). Omara et al. (2016) reports that in the Marcellus shale, unconventional multi-well pads have a lower emission factor per unit area than conventional well pads, as the advent of horizontal drilling increased the amount of shale accessible from one site.

2) **Green Completion techniques:** Whether or not readily available REC (reduced emissions completions) technology is implemented can have large impacts on emission factors.

This investigation was able to accurately measure methane emissions from a conventional well pad in the production stage, which is analogous to a shale gas well in production. With an average CH_4 ppm of 1.91 and an average flux of 24.3 Kg CH_4/year, this is not a large emission factor and under national scales, Frankenberg et al. (2016) suggest any methane emissions from natural gas sites less than 1kg/hour will contribute very little to basin level and larger scale methane emissions. The KM5 well emits 0.0028 Kg CH_4/hour under the highest methane flux recorded from a sample day. This result suggests the standard of equipment on the KM5 well pad is of good rigor and no large leaks are present, and this is expected of all hydrocarbon wells. How therefore could shale gas extraction nullify the coal to gas shift? The problem section of this question lies in pre-production standards. As Howarth et al. (2011) states, up to 3.6% of total production can be lost during the well completion and liquid unloading phase, and although this percentage estimate has been heavily scrutinized, it at least can form an example of worst case scenario.

Methane can be emitted during re-fracturing treatments, which can occur up to 6 times in the lifetime of a well (Allen et al. 2013). The difference in emissions from regulated and unregulated shale gas scenarios is shown below in figure 4.4.

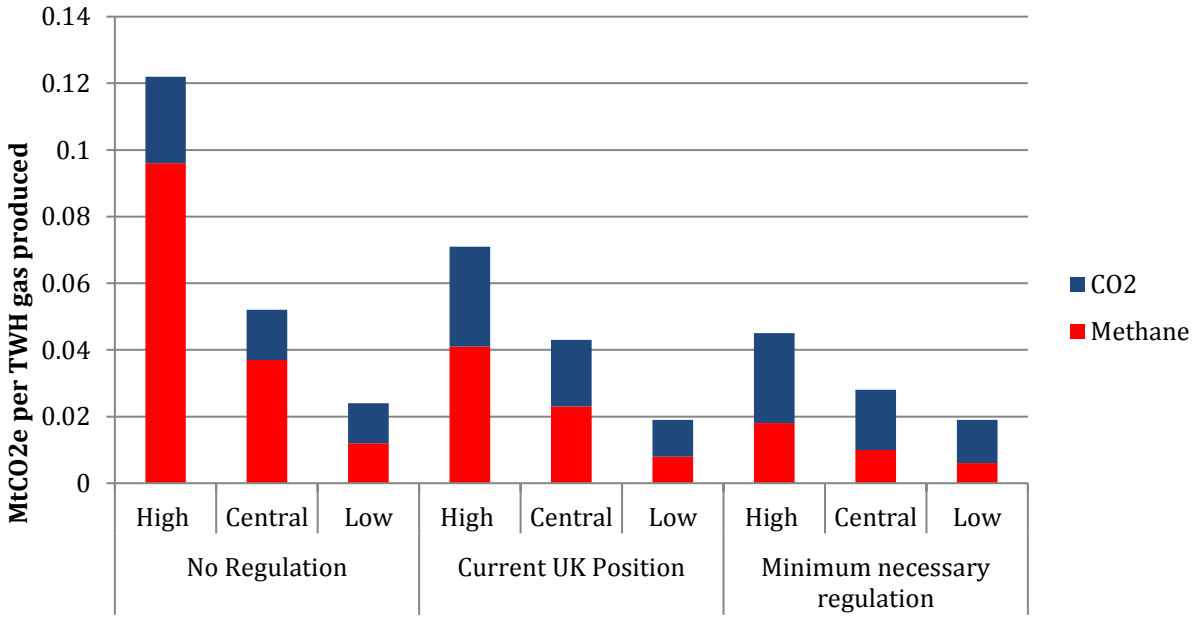

Figure 4.4: Emissions per tWh from shale gas exploitation under different regulation scenarios (Source: Committee on Climate Change, 2016)

Figure 4.4 shows the differentiation in greenhouse gas emissions from shale gas production as calculated by the Committee on climate change (CCC, 2016). Under no regulation, methane emissions could be up to 6 times higher than under UK practice, with regulation more capable of minimizing methane emissions than CO_2 (CCC, 2015). This report considers methane on a GWP_{100} basis, taking into account the short residence time of methane. In terms of how practice influences emissions, O'Sullivan et al. (2012) calculate that the GHG intensities for well pad practices of venting, flaring and REC are; 13.438 kg CO_2e, 1.714 Kg CO_2e and 1.344 Kg CO_2e under GWP_{100}. This demonstrates the vastly larger impact of venting practices compared to flaring and green completions. The final two numbers are similar as under REC, the gas is recovered and sold or combusted for whatever end use, however flaring on site is not 100% efficient, and therefore some (<3%) methane is likely to leak during flaring.

This CCC (2016) chart suggests that UK regulation could possibly not yet be robust enough, however Mackay and Stone (2013) suggest that emissions from shale gas exploitation should be lowered to ALARP (as low as reasonably practical) and then at greater economic efficiency reduce methane emissions in other UK sectors.

Howarth et al. (2011) insisted on the 20 year GWP timescale and in Howarth (2015) presented this data, which is made relative to analysis from Cathles et al. (2011)

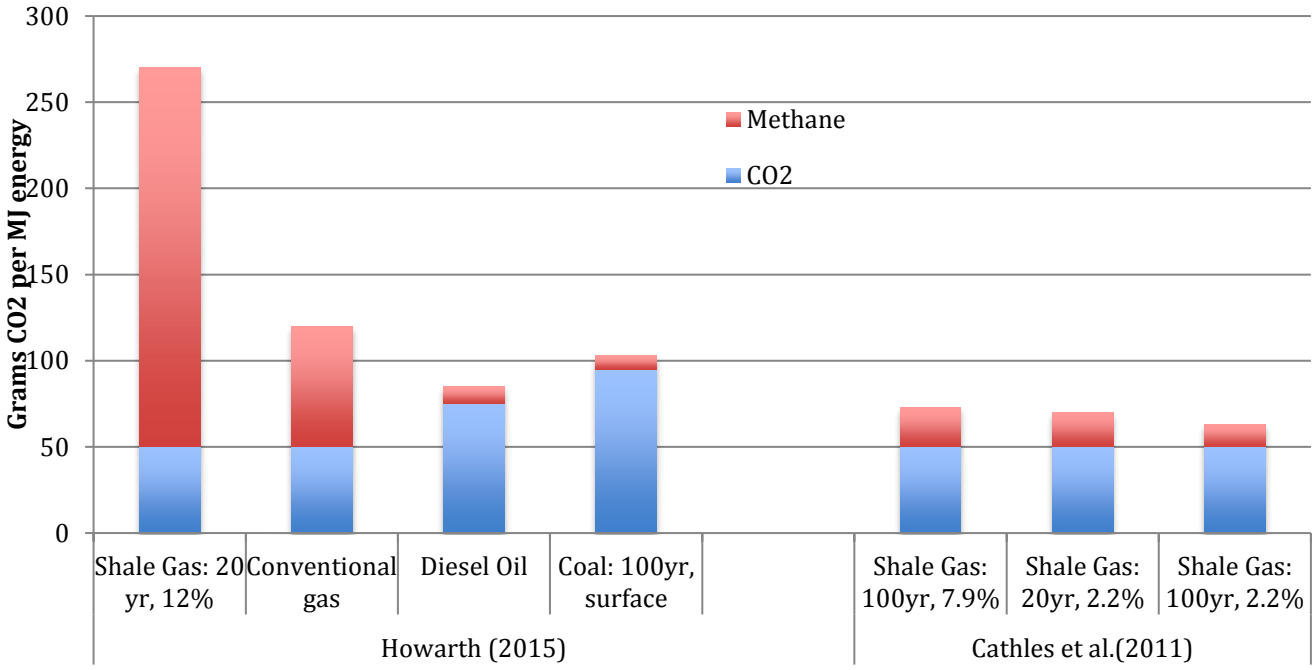

Figure 4.5 :The climatic effects and comparison of fuel source emissions under specific scenarios (Source: Howarth (2015) and Cathles et al. (2011))

Howarth (2015) adopts the 20 year GWP for methane of 86 and also assumes mean shale gas emissions of 12 % based on Schneising et al. (2014). This is the highest emission rate and the use of the 20 year GWP of methane is how Howarth (2015) justifies stating that the shift from coal to both shale and conventional gas is nullified by high methane leakage. The suggestion of high methane leakage from shale gas wells along with its application to conventional well emissions is heavily disputed and is regarded by many as a worst case scenario (i.e. no regulation) and or a large underestimation of reducing emissions techniques (Cathles et al. 2012). Had Howarth et al. (2011) used their lower end 3.6% emission rate for unconventional gas production with a 100 year GWP of CH_4, shale gas would have 47% of the impact of coal assuming energy conversion efficiencies of 60% for CCGS and 37% for coal (Cathles et al. 2012). Direct emissions from coal combustion are almost double the emissions of natural gas at 97g CO_2e/MJ, and up to 210 g CO_2e/MJ for life cycle emissions. A study by NETL (National Energy Technology Laboratory), a division of the US DEA, reported that new CCGS facilities have 53-58% less Well to Wire emissions when compared to a modern or old coal combustion facility. They state that the higher emissions associated with shale gas pre-production only account for a few percent of emissions over a well lifetime and

do not incorporate shale gas within the highest band of emissions, occupied by coal production (NETL, 2015).

Unconventional methods of drilling involve four key processes in the extraction section not typical of conventional drilling; horizontal drilling, hydraulic fracturing, well completion flowback, and the removal of flowback fluids (Broderick et al. 2011). In the UK, 10% of wells have undergone a form of hydraulic fracturing in order to stimulate less permeable reservoirs. (UKOOG, 2016).

In terms of literature stances on the GHG impact of unconventional vs conventional gas; Burnham et al (2011) suggests that shale gas exploitation has a lower greenhouse gas impact than conventional and Weber et al. (2012) suggests it is the same, stating that in fact separating segments of the natural gas industry is impractical and that the greater issues are downstream. The data from this study would agree with this statement, as upstream emissions from the well pad are negligible and the emissions from Knapton generation station are low and well within regulations of EPA (2016) best practice of a 0.45% leakage rate. Downstream flux data from this study are many orders of magnitude larger, with emissions of up to 70 Kg CH_4/hour, and this should be subject to great improvements. Stephenson et al. (2011) and Jiang et al. (2011) both state that the greenhouse gas emissions from shale gas exploitation are on average larger than from conventional wells, but all of these authors above conclude that using shale gas for electricity generation has a greenhouse gas footprint at most 2/3 of that of coal. Stephenson et al (2011) used a simulation model to determine the difference in emissions from shale gas compared to conventional wells. Using a WtW (well to wire) intensity measurement it was calculated that shale gas had an emission intensity 1.8-2.4% higher than conventional gas, with the majority of emissions from well completion (Stephenson et al. 2011). Under more extreme assumptions of leakage rates, WtW emissions would not be more than 15% higher than conventional gas (Stephenson et al. 2011).

It is also important to consider the different forms of unconventional sources of hydrocarbons. Fracking for shale and tight gas is relatively similar, and REC completions have applied to at least 65% of well drilled since 2010 (O'Sullivan et al. 2012). The Nat Gas star program (2012) states that up to 98% of emissions can be recovered, however this technology is only applicable for medium to high pressure wells, with initial well production pressures of 500 psia described as the absolute

minimum. The requirement of high initial production pressure suits shale gas exploration very well as initial production rates are typically higher than conventional wells (Howarth et al. 2011). In a US EPA (2015) survey of all unconventional wells, the depths of which varied from 150m to 3700m deep, there were only well pressure constraints with the shallow, coal bed methane wells. In these wells, the EPA has said 'experimental' compressors were used in attempts to increase well pressure, but this is very expensive and not very effective, so is seldom used (EPA, 2016). This combined with the risk of shallow drilling in terms of aquifer contamination and methane migration makes CBM the riskiest of all unconventional drilling (Jackson et al. 2012). The three largest CBM producing states in the US are Wyoming (7.5 BCM), New Mexico (10.6 BCM and Colorado (11.7 BCM) (US EIA, 2015). Venting is allowed but 'restricted' in Wyoming and New Mexico and is legal in Colorado with state approval. The restrictions have come under huge scrutiny as 94 % of Wyoming wells reported flaring and or venting gas and up to 1700 m^3 of gas per day can be vented or flared without a permit (Stephenson, 2011). Australia has seen a large expansion of CBM, which they call coal seam gas, however in some areas it has been possible to extract the methane from the coal lattice without hydraulic fracturing, and the lack of primary data has made contamination and methane emission attribution to unconventional wells difficult (Day et al. 2014). The Australian province of Victoria voted in September 2016 to ban coal seam gas exploration and fracking.

In Canada there is an unconventional source of fuel which is now one of the largest construction sites on earth, the Athabasca oil sands (Timoney & Lee, 2011). When comparing the efficiency of production for each fuel, large subterranean shale gas reserves are able to be extracted from a small area of land due to horizontal drilling, while tar sand oil production dominantly requires surface skimming of the sands, and treatment to separate the oil (Biello, 2013). Other methods of extraction have included blast steam treatment where large volumes of steam are pumped under the desired sand layer forcing the oil to the surface, but this is more expensive and energy intensive (Biello, 2013). Overall, this means oil sand extraction has a 'well to wheel' emission factor equivalent to 5 to 6 times that of conventional oil extraction. In terms of efficiency, the BP world energy report in 2016 reported that Saudi Aramco claim it takes approximately 1 barrel of oil worth of energy to extract 10 barrels of oil, whereas Canadian tar sands produce 2-3 barrels of oil for every barrel of oil used. Nevertheless, 2014 Canadian Tar sand reserves have been estimated at

188 billion barrels, the third largest oil reserves in the world behind Venezuela and Saudi Arabia and the Althabasca tar sands produce 47% of annual Canadian oil production (Annis, 2016).

It is important to determine if renewables could replace coal. If domestically produced shale gas was not to be used in the UK, gas would have to continue to be imported and or renewable energy would have to be expanded as coal use is diminished (DECC, 2015). In 2015, 24.6% (82 TWh) of UK electricity generation was generated from renewable energy sources, 45% of which came from wind power and 43% of which came from biomass (DECC, 2016). Scotland supplies 40% of total UK wind power, and in 2016 wind power provided 100% Scottish electricity demand for a day, but considering its 5.1 million population as a proportion of the UK's 70 million, this could not be replicated on a national scale without increasing renewable capacity by a factor of 14 (DECC, 2016). The UK had a total electricity usage of 312 TWh for 2015, which is the lowest level since 1996, but this is only 14% of the UK annual energy usage, which was 2,249 TWh, or 191.5 million tonnes of oil equivalent in 2014 (DECC, 2015). The scale of the difference between power and energy use is shown below in figures 4.6 and 4.7.

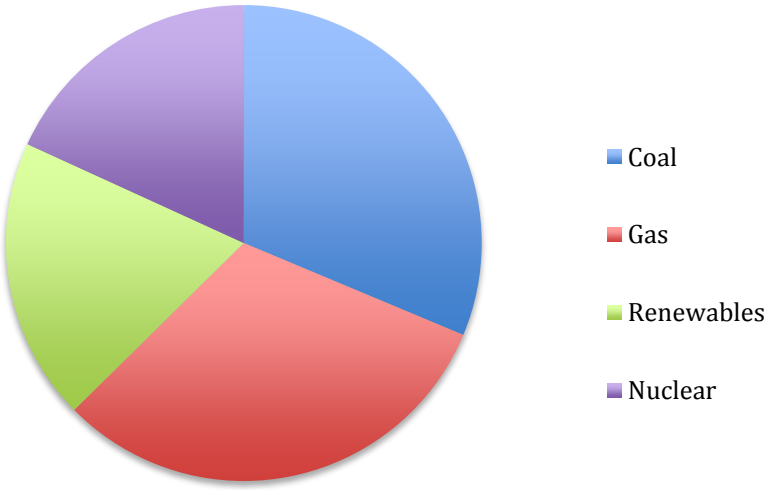

Figure 4.6: UK electricity use (312 tWh) by sector, 2014

Source: DECC (2015)

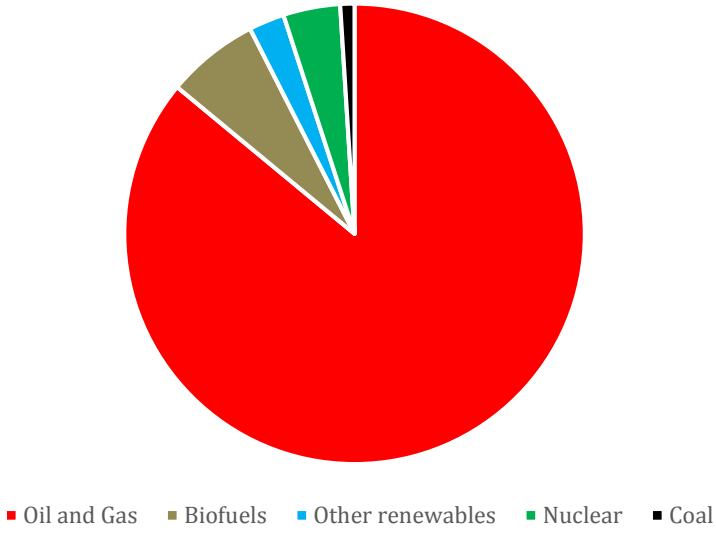

Figure 4.7: UK energy use (2249 tWh) by source, 2014 Source: DECC (2015)

The remainder of energy usage is split up between transport, heating and industrial energy usage. Ultimately, this means that renewable energy has supplied a historical maximum of 3.5% of annual UK energy usage. In 2014, natural gas supplied 34 % of total energy consumption in the UK, equivalent to 764 TWh, which is 2.5 times the total UK electricity generation (DECC, 2015). In terms of energy production per unit area, a study by Ernst & Young (2014) reported that over the 25 year lifetime of a shale gas well, 9.5 TWh would be produced from a site of 2 hectares. By comparison, assuming current efficiency rates in the UK, it would require a solar park of 925 hectares or a wind farm of 1450 hectares (EY 2014). In terms of land use this is between 460 and 725 times the area for the same amount of energy. On a global scale, electricity demand increased daily by 450 tWh from 1985 to 2011, and Bryce (2010) reports that in order to satisfy this incremental demand, 100 miles squared of wind turbines would have to have been built per day! This calculation also assumes a minimum average wind speed adequate to produce the electricity (Bryce, 2010). Faulkner (2015) have stated that the ability for shale gas to displace coal as an electricity producer has lowered US CO2 emissions by up to 10% over a 10 year time period, compared to the 1% drop in emissions from wind power generation over 25 years. Clearly there is a huge disparity, because (non-biofuel) renewables are not able to supply over 3% of UK energy demands but a constant reliance on fossil fuel use is continually increasing the risk of climate change, a good example of a hugely complex socio-environmental clash of priorities, the solution of which is likely to need to be just as complex.

4.4. How would Shale gas exploitation impact UK emission rates?

UK greenhouse gas emissions have been gradually falling over the past 25 years from over 800 million tonnes CO_2e in 1990, to 600 Mt CO_2e in 2008 and to 491 Mt CO_2e in 2015 (DECC, 2015). CO_2 accounts for 82% of total UK emissions and emissions are dominated by the power, business, transport and home sectors (DECC, 2015). The emissions over the past 2 decades are shown below;

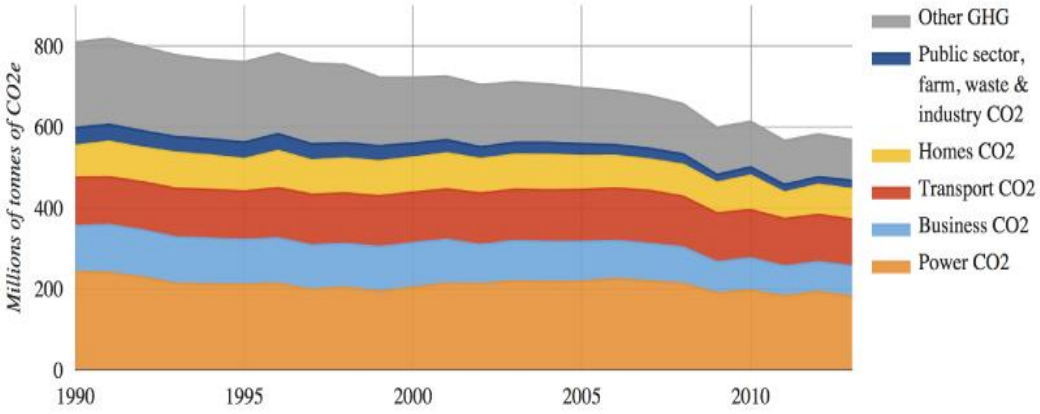

Figure 4.8: UK total greenhouse gas emissions per sector
Source: Carbon Brief

The Climate Change Act (2008) legislates for the UK to reduce annual CO_2e emissions to 456 Mt CO_2e by 2032 and by 80% from 1990 levels to 160 Mt CO_2e by 2050 (DECC, 2015). Although the UK has been meeting and exceeding targets set by the first, second and third carbon budgets, lowering the annual emissions to less than 200 Mt CO_2e will require large political intervention, and is impossible without the application of CCS, regardless of whether or not UK shale gas reserves are exploited (Mackay and Stone, 2013).

The IPCC (2014) indicates that greenhouse gas emissions can be reduced mainly by replacing current coal fired power plants with efficient CCGS power plants. The Committee on Climate Change report on shale gas (2016) stated that the switch to shale gas could be compatible with emission targets. The biggest unanswered question is how much gas is to be consumed by 2050 in the UK under current de-carbonisation pathways. It is likely that natural gas use will continue about current usage until a more applicable and reliable fuel source is found. Furthermore, if CCS is successful at an adequate scale, natural gas use could continue indefinitely, but at largely reduced net emissions (O' Sullivan et al. 2012).

Determining whether or not the use of shale gas is in keeping with UK climate targets is explained further in a Committee on Climate Change report (CCC, 2016) which states that there is great uncertainty over the amount of unconventional hydrocarbon resource which is recoverable and the size of the potential shale gas industry, so projections can be difficult.

There is a large disparity in the projected production values for UK shale gas. However, under high productivity, central emissions scenarios, where around 11 Mt CO_2e is emitted in 2030, domestic shale gas would satisfy carbon targets if three key recommendations the CCC report are adhered to;

A) Well emissions from pre-production to abandonment must be strictly limited and monitored;
B) Gas consumption remains in line with carbon budget requirements;
C) Shale gas emissions are accommodated by reductions in emissions from elsewhere in the UK economy.

If the UK is to adhere to limiting average warming by 2 ^{o}C, it will require total removal of coal use by 2025, as pledged by the government. However, considering the UK began traversing the coal to gas bridge in the 1970's, there is less of that bridge to cross. Hence, the potential for lowering CO_2 production from fuels is limited compared to large coal consumers, such as China which generates 69% of its electricity from coal (CCC, 2016). The proposed rate of emission decrease is shown overleaf:

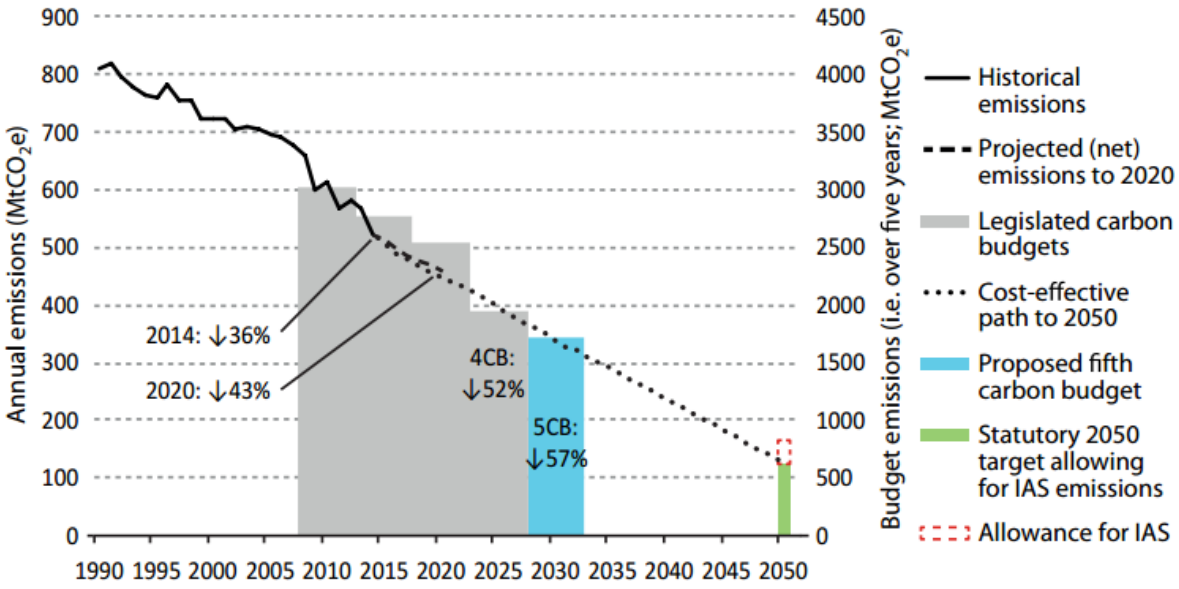

Figure 4.9: The UK decarbonisation pathway to 2050

Source: Committee on Climate Change

A large problem arises once gas use has totally replaced coal as a fossil fuel, as the only further solutions are to decrease carbon output as opposed to moving to less emission intensive fossil fuels. Therefore, the removal of fossil fuels or the implementation of CCS to achieve net lower emissions is essential, since gas is the cleanest fossil fuel when combusted (Stephenson et al., 2011). As fossil fuels will not be totally phased out until a reliable replacement is found, CCS is more likely to be the choice over the coming decades.

Schrag (2012) believes there is a large risk that the uptake of shale gas could displace renewable contribution to electricity generation. Also, with delays to Hinckley point pushing its construction completion to beyond 2035, there is scope for the expansion of a new generation of CCGS plants which could be built in relatively short time periods for a lower cost than coal fired plants. Verdolini et al. (2016) disagree that shale gas would displace renewables. From 2005 to 2016 shale gas production in Texas increased from 1% of state production to 63% at the same time as energy from renewables increased from 2% to 16%, while nuclear remained stagnant (US EIA, 2016). Verdolini et al. (2016) however do make it clear however that that renewables will have upper limits of production. Schrag (2012) states that under the UK's Copenhagen Accord commitments, 'shale gas offers no meaningful potential, even as a transition fuel' unless combined with CCS. However, in the UK

before 2025 there is a risk of not adopting shale gas. As US shale gas consumption increases, more coal produced in the US was exported for sale on the global market. This contributed to the increase in European coal consumption and CO_2 emissions in the 2009-2011 period while US CO_2 emissions decreased by 7% (Broderick and Anderson, 2012). From 2009-2012, UK electricity generation from coal increased from 27% to 39% (Mackay and Stone, 2013). This could be prevented in the future by adopting shale gas with a restricted maximum carbon cap and removing coal as a fuel option.

Mackay and Stone (2013) compiled a series of scenarios which include combined pre-production and production emissions using a GWP of 25 for CH_4, as shown below;

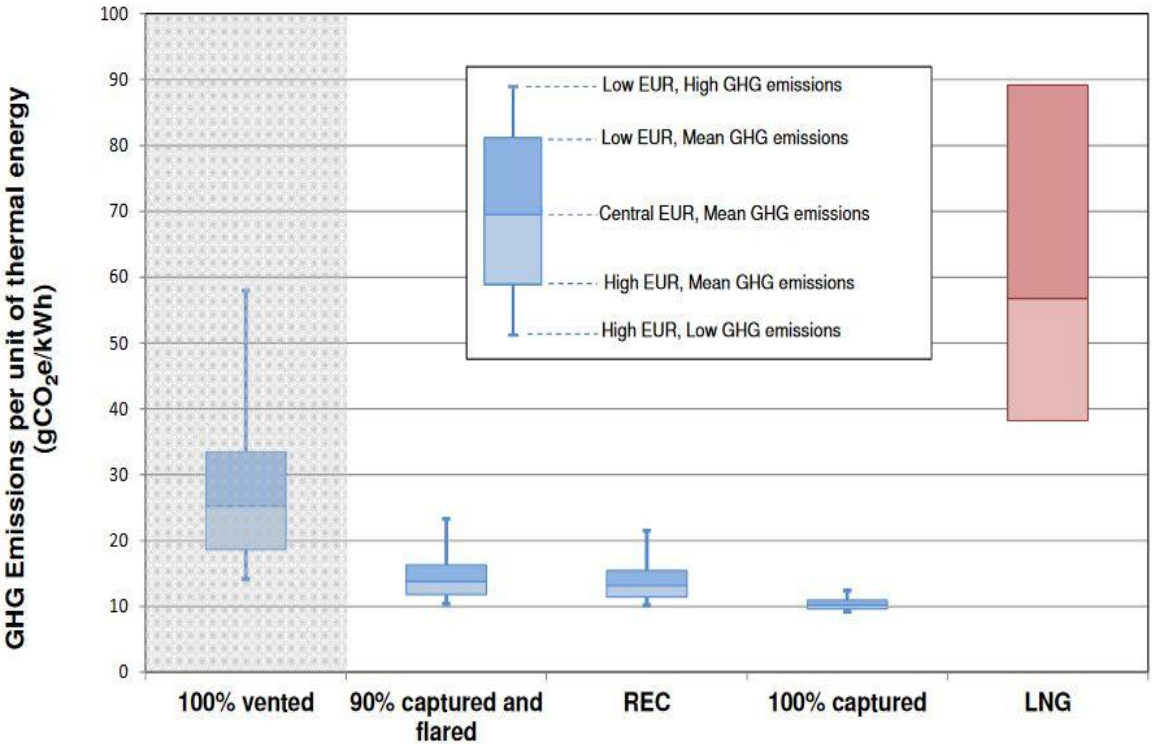

Figure 4.10: Emission intensity of shale gas under different scenarios, and LNG emissions Source: Mackay and Stone, GHG impact of Shale gas exploitation

The data was collected from 15 different studies which measured total emissions including those from drilling and well completion. Mackay and Stone (2013) did not include the Howarth et al. (2011) data in this graph stating it was likely an outlier and based significantly on the impact of venting methane during well completion. The Oil and Gas authority and the Environment Agency bans routine venting of gas in the UK (OGA, 2015). As there is limited data on UK shale gas composition, Mackay and Stone (2013) when calculating total emissions assumed that the shale gas is

86% methane, and 100% when processed, with a density of 0.76 g/m^3 and a calorific value of 52 MJ/Kg. So under regulated UK scenarios, REC applications on a well with high production should equate to GHG emissions of 11-13g CO_2e/kWh, which is 1/5 of the average GHG emissions per kWh of LNG, excluding combustion in both scenarios. When Howarth et al. (2011) data is included the average emission intensity for 90% captured and flared is 85% larger at 52 gCO_2e/kWh, but still less than LNG average of 58 gCO_2e/kWh. The emission factor for the combustion of the fuel must be included, which for methane is 190 g/kWh. There are two emission factors to include; the emissions from UK electricity generation (which is ~50% fueled by gas) and that from commercial and domestic heating (84% of UK homes heated with gas). Overall, the total thermal emission intensity from pre-production, production and combustion is likely to be 200-250 gCO_2e/kWh (Mackay and Stone, 2013). However, the emission intensity of electricity generation from natural gas is double the emission factor per kWh at 410-520 gCO_2e/kWh when compared to gas use for heating. This is simply because of system inefficiencies, and according to Mackay and Stone (2013) calculations, shale gas will still produce under 50% of total coal emissions per $kWh_{(e)}$.

In terms of potential for reducing emissions, it would seem therefore more sensible to remove gas from the electricity sector over the long term. However, this could prove difficult because since January 2016, CCGS has provided on average 45% and up to 60% of total daily electricity production (My Grid GB, 2016)

Over decadal time scales, methane emissions are more important than CO_2 emissions, and hence for the UK it is imperative to minimise anthropogenic methane emissions, which account for over 60% of the annual 3.7 Mt CH_4 emissions. Mackay and Stone (2013) do not include abandonment emissions but as discussed in Q2, Boothroyd et al. (2015) have shown that emissions from decommissioned wells can emit methane in volumes equivalent of up to 2 sheep.

Broderick et al. (2011) in the Tyndall report collated three UK total shale gas production estimates;

A) 150 bcm (DECC, 2010);
B) 566 bcm (US EIA, 2011);
C) 1,132 bcm (Cuadrilla).

All three of these scenarios have estimated peak gas production in 2040 at 6bcm/year, 24bcm/year and 48bcm/year for each scenario. Under Cuadrilla's scenario, of which the 1,132 bcm is an estimate based on an assumption of 20%

recovery rates on resources of 5,660 bcm. Broderick et al. (2011) note that this resource figure is generously optimistic, but if correct, considering Cuadrilla's exploration zone only covers 5% of the Bowland shale, the UK gas potential is up to over 10 times as large. It is important to project how each scenario would contribute to the national emission rates. Geny (2010) and Broderick et al. (2011) assume that the UK would have production rates proportional to the rate of US shale gas exploitation, and the Tyndall Centre estimation is shown below.

Scenario:	**Cumulative shale gas production (bcm)**		**Cumulative CO_2 emissions from shale gas ($MTCO_2$)**	**% of UK Domestic action budget**
	2030	***2050***	***2010-2050***	***2010-2050***
DECC 150 bcm	21	132	264	1.90%
EIA 566 bcm	79	499	1015	7.30%
Cuadrilla 1132 bcm	157	997	2029	14.50%

Table 4.3: Shale gas production and emission estimates (Broderick et al. 2011)

These emission rates are projections of the UK CO_2 budget under assumptions of best practice. Under all three scenarios, the majority of the gas (88%) would be combusted and between 264-2029 Mt CO_2 could be released by 2050. This would equate to between 1.9% and 14.5% of the total UK CO_2 budget (DECC, 2010). Broderick et al. (2011) state that shale gas exploitation in the UK could have neutral or beneficial emission factors only if it replaces LNG imports and UK coal use. CCS implementation could reduce this further. However it is as yet unproven at scale and the majority of CCS technology has focused on 'clean coal' (Mander et al. 2010).

The price of shale gas will also be an issue. In the USA gas prices have dropped to $2 per million BTU while UK, EU and Japanese gas prices have shown small fluctuations about the $5 per million BTU mark (US EIA, 2015). This can be useful for UK shale drillers as the higher costs from drilling and regulation will require a higher gas price to be economically viable. In the US, shale gas and oil exploitation

has been so successful that by producing so much they have caused a crash in the Henry Hub (US Gas Price) and the West Texas Intermediate (The US oil benchmark price) which made hydraulic fracturing not economically viable in many states and plays (Allen et al. 2013). It has not been thought that UK shale gas exploitation would drastically lower gas prices, as it is part of an integrated European market. This is not entirely desirable as lower costs would mean increased usage and therefore increased emissions. A Poyry report (2013) on the macroeconomic effects of European Shale gas production has estimated that shale gas extraction in the UK could drop energy costs by up to 4% by 2035 and the IEA golden age of gas report (2011) states that if European energy markets did not increase gas use, the cost of renewable energy and the carbon tax burdens on coal could see energy prices rise 35% by 2050.

Any increase in the carbon intensity of electricity generation would be undesirable. The CCC (2016) suggests it should be 50-100g/kWh by 2030, bearing in mind it is on average over 350g/kwh, and in the DECC (2012) gas generation strategy, a gas dominated electricity mix has a carbon intensity of 200g/kwh. Given that in the majority of low carbon transition plans, electricity generation is the sector most significantly decarbonized, any large increases in gas consumption without coal removal and CCS implementation would be tantamount to an abandonment of the 2050 goals of 80% emissions reduction (Ekins et al. 2014). Many UK studies including Broderick et al. (2011), Mackay and Stone (2011) and the CCC report (2016) state that great caution must be taken when extrapolating shale gas data from the USA for UK application, mostly because of scale issues. The UK is 250,000 km^2, which is 2.5% of the land mass of the USA (9.9 million km^2) and since 2005, 85,000 shale wells have been drilled in the US, which is almost 30 times the projected number of UK wells (DECC,2015; US EIA 2015). Under current BGS (2014) estimates, there could be up to 3700 BCM of natural gas reserves from shale reservoirs in the Bowland-Hodder and Weald basin, which is a large gas reserve proportion compared to remaining onshore conventional reserves of 1.98-5.7 BCM. By contrast, the US EIA (2015) updated estimate of US unconventional shale and tight gas reserves are almost 11,400 BCM, and in 2014 the US produced 350 BCM of shale gas, 10% of the UK total reserve!

Carbon capture and storage, essentially the sequestration of CO_2 from fossil fuel combustion will have to play a part in emission reduction regardless of shale gas

production. However, the UK CCS plan was abandoned in early 2016. Under modelling by McGlade et al. (2016) without CCS application, gas use would have to peak before 2030 and steadily decline from then on to prevent warming of above 2 ^{o}C. It is unlikely that the transition to a totally renewable energy economy would be ready by that point. Also, if the plan were to reduce gas consumption by 2030, it questions the wisdom of developing a UK shale gas industry for a limited period of time. In reality UK natural gas use is unlikely to fall by any more than 10% by 2040 (DECC, 2015).

Emissions should be controlled to as low as reasonably practical, but energy systems are not the largest contributor to UK methane emissions (DECC, 2015). Under IOD (2013) scenarios, a projection indicates that by 2050, 2800-3000 lateral wells will have been drilled in the UK at a rate of approximately 100 per year, assuming the first well is producing by 2020. Per well pad there are likely to be at least 4 laterals wells in order to target desirable shale 'sweet spots' which yield greater production values, which is economically and environmentally preferable (CCC, 2016). Under Howarth et al. (2011) 'worst-case' scenario from a highly productive Haynesville gas well, 4836 tons (6.8 million m^3) of CH_4 is emitted per well during flowback, so 100 worst case scenarios per year would mean 483,600 Tons CH_4/year is emitted to the atmosphere. To put this in context, the UK's 1.895 million dairy cows (which individually emit ~120 Kg/year CH_4), 9.7 million cattle (which emit ~60kg CH_4/year) and the 28 million sheep (which produce ~60Kg CH_4/year) produce methane emissions of at least 2,532,400 Tons CH_4/year (UK DOA, 2015). This is equivalent to almost 550 wells emitting under Howarth et al.(2011) worst case scenario well completion per year. If the theoretical 100 wells per year emitted methane under the worst-case scenario with well completion emissions similar to the Haynesville example, it would be the equivalent of 19% of annual cow, cattle and sheep methane emissions.

It is important to state the contribution of REC completions. Again under the worst case scenario, but instead assuming 98% of the methane is captured under REC and sold, 9672 Kg of methane would be emitted per year as leakage from well completion from 100 wells, or 967kg/well completion of methane per well, the annual equivalent of almost 7 cows (which is 0.08 of the average UK cow herd farm population) (Howarth et al. 2011;UK DEFRA 2015). Under this standard of practice, methane proportions equivalent to 0.38% of methane emissions from these three

animal methane sources would be emitted from fugitive emissions from shale gas wells during well completion. In 90% captured or flared scenarios, these 100 wells per year would emit 1.8% of annual cattle, cow and sheep methane sources.

Combined with the research from this study, with the calculations of emission factors from production stages of 0.0078% from KM5 wellpad and 0.082% from Knapton generation station under the largest plume scenarios, it does not appear that these measured and calculated upstream natural gas production emissions pose a larger threat in terms of methane emissions compared to conventional drilling in a UK context. Under poor regulation standards however there is the potential for fugitive methane emissions to increase the UK methane budget. Nevertheless, even under worst-case scenarios, this would be limited to less than 20% of current anthropogenic annual methane emissions in the UK purely on grounds of resource availability and rate of production (Joffe, 2015). Emissions from downstream sources are the same whether the gas comes from conventional or unconventional sources and are likely to pose larger environmental issues over longer terms given the increasingly stringent methane emission regulations for upstream natural gas production (Bower, 2015).

Overall, under good regulatory controls, ensuring capture of at least 90% of emissions, shale gas life cycle emissions can be lower than LNG life cycle emissions. However, there should not be a 'dash for gas' for the electricity sector, which could promote surplus use above carbon budgets and increase overall emissions (Joffe, 2015). Critically, the bridge between current unabated fossil fuel use to a low carbon economy and the timeframe over which emissions must be reduced is time-restricted, and if shale gas is to be exploited there must be sharper focus on ensuring methane emissions are limited.

4.5. How would Shale gas exploitation impact global emission rates?

Global CO_2e emissions have risen by 256% in the 50 years from 1960-2010, and the greatest rate of increase has been in the 21st century with emissions rising 35% to 2010 (Peters et al. 2012). Methane emissions have risen 47% in the 40 years since measurements began, and from 2000-2010 global methane emissions rose 20% (Peters et al. 2013) and currently in 2015, total global GHG emissions are 50 Gt CO2e/year (DECC, 2016). The annual CO_2 and CH_4 emission change over the past 40 years is shown below in figure 4.11.

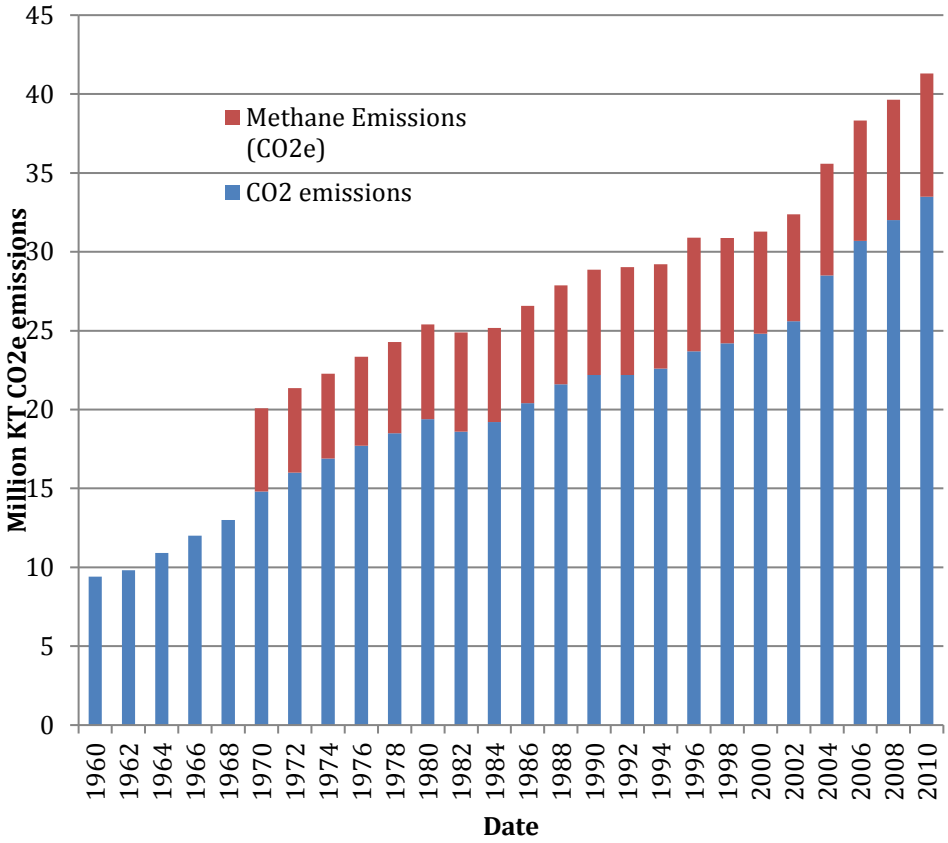

Figure 4.11: Global CO_2 and CH_4 emissions over 40 years
Source: EPA, 2016

There is discord in the literature on the overall effect shale gas exploitation is likely to have on global emissions and the rate of CO_2 production. One side of the argument states that shale gas can be the bridge fuel for countries, moving from high carbon fossil fuel usage to a low carbon economy by reducing combustion emissions (BP statistical review, 2016). Conversely, Broderick and Anderson (2012) argue that in a world so reliant on fossil fuels, it is more likely globally that shale gas will be used in addition to other conventional and unconventional

hydrocarbons, thus adding to the global carbon burden. The pathway which emissions rates take will be hugely dependent on global sensitivity, climate policies and price elasticities of coal and gas demand (Mackay and Stone, 2013).

Brown (2011) modelled 5 different scenarios with different natural gas usage in the US. The study accurately predicted that rapid and large increase in natural gas abundance would decrease natural gas prices and would displace coal as the fuel for electricity production. Without political intervention on carbon policies to limit emissions however, Brown (2011) projects that shale gas could boost net CO_2 emissions by 1% by 2030 from the displacement of nuclear and renewable fuels. Jacoby et al. (2012) also modelled US emission scenarios, and in this case used the MIT emissions predictor with 2 scenarios; 1) Ensuring renewable energy generation is at least 25% of electricity generation; and 2) Decreasing emissions by 50% by 2050 (from 1990 levels) using carbon pricing. Under the first scenario, without shale gas, US emissions would decrease 2% by 2050. With shale gas, it is projected to increase 13% by 2050 from 2005 levels, which Schrag (2012) also attributes to the likely reduction in investment in renewables (Jacoby et al. 2012). Under the second scenario, Jacoby et al. (2012) state, since emissions would be reduced 50%, the pace of technological advancement in CCS technology would be a function of the shale gas resource abundance, with greater gas use requiring a greater uptake of CCS. Jacoby et al. (2012) affirm however that this would result in less investment in coal CCS, and would practically remove the ability for renewables to penetrate the energy market further due to their inadequate ability to meet demand.

Although the USA's CO_2 emissions have declined over the past 7 years due to the coal to gas transition by up to 9%, the coal which shale gas is displacing is not staying in the ground to the same degree (Broderick and Anderson, 2012). From 2005-2010 up to 338 Mt CO_2e of the 645 Mt CO_2e saved from the gas to coal shift in the USA was emitted from exported coal combustion outside the US (Broderick and Anderson, 2012). In 2005, US gas price was $0.03/kwh while coal prices were $0.007/kwh, under ¼ of the price of gas, while in 2012 coal and gas prices were identical at $0.009/ kwh (US EIA, 2016). However, Plumer (2016) suggest that this emission surplus is a temporary glitch, and that the bankruptcy of Peabody Energy, the largest global private sector coal producer in April 2016 is an example of how the US market now dominated by natural gas imposed industrial and economic pressures, which the coal industry is incapable of resisting.

The greatest opportunity for emission reduction lies with China, the US and other large fossil fuel producers and consumers, specifically in reference to coal use. If China was able to reduce it's annual CO_2 emissions by 10.5%, this is the equivalent of removing the UK and Germany's annual CO_2 emissions from the global carbon budget (World Bank, 2015). China emits as much CO_2 per year (11 million Kt CO_2e) as the USA, the EU and India combined. China, the USA and the EU, which host 25% of the global population, emit 54% of global annual CO_2 emissions (US EIA, 2015). On a smaller scale, the UK has 1.1% of the global population, and emits 1.16% of the global greenhouse gas emissions, including 0.9% of global methane emissions (DECC, 2016).

High emissions from China result from their current production of 3,500 million tons of coal per year (US EIA, 2015). China consumed 3,753 million tons in 2011, which they seldom reduce in the future because it fuels economic growth (US EIA, 2015). There is however, ample opportunity for carbon reduction in the Chinese economy, since in 2014 73% of domestic electricity production was from coal, providing 21,709 tWh, a growth of 800% since 1995 (US EIA, 2015). Assuming an average emission intensity of 1050 $gCO_2e/kWh_{(e)}$ for coal combustion, this equates to 2.39 x 10^{16} g CO2/year. Shale gas combustion has an emission intensity of 450 $gCO_2e/kWh_{(e)}$ for electricity, which under the same electricity demand would be 1.025 x 10^{16} g CO2/year. From solely CO_2 combustion emissions, the switch from coal to gas would remove 1.365 million Kt CO_2/year from Chinese electricity production which is just under 4% of annual global CO_2 emissions or, more than the annual emissions of Japan, the world's 5th largest CO_2 emitter. This is achievable for China as they have the world's largest shale gas reserves with an estimated recoverable resource of 31600 BCM, double that of US reserves (US EIA 2015). However, the large freshwater demands for hydraulic fracturing of shale rock exceed the availability of water. Many scientists believe freshwater will have to be imported into China from Russia by 2030 (Gao, 2012).

In 2015, President Obama pledged to reduce annual US greenhouse gas emissions by 80% by 2050. A recent study by Heal (2016) suggests this would cost up to $176 billion/year or $5.28 trillion over 30 years, 75% of these costs would be forced onto utilities providers and energy companies, which would then impose capital costs on consumers. This equates to 1% GDP/year, and 70% of this cost is in energy storage, with an average stored electricity generation from a wind turbine per day costing

$7.8 million, double the cost of the turbine (Heal, 2016). Williams et al. (2014) conducted a similar study, determining costs at 0.7 % GDP/year, but this was under a shift from fossil fuel to nuclear rather than to other renewables. By their calculation, this is a less expensive route to decreasing emissions 80% by 2050. Heal (2016) concludes by saying that an 80% reduction in US emissions by 2050 is unlikely, but states a 50% reduction (from 2005 levels) is possible under the total replacement of coal by gas, a carbon cap and great investment in energy storage technology. Energy storage technology is currently immature but is the only alternative if intermittent renewable energy instead of constant nuclear power is decided to be the appropriate choice.

Globally, estimates of shale gas recoverable reserves vary between 187,535 bcm and 204,000 bcm, assuming recovery rates of 20-30% (US EIA; IEA GAOG). These estimates do not include that of Central Asia and Russia, the Middle East, central Africa or South East Asia because the conventional resources of these regions represent over 75% of global remaining reserves; there is a lack of available information on unconventional reserves (EIA. 2011).

Broderick et al. (2012) calculated the projections of global emissions based on this shale reserve information. They stress however, that the incomplete jigsaw of unconventional reserve data means real global emissions would be larger if these unaccounted regions decided to exploit their shale gas reserves. In their calculations they make three key assumptions:

- 50% of reserves would be exploited by 2050 and 100% of reserves by 2100
- These emission projections would not be inclusive of pre-production practices such as well completion, drill out or refracturing.
- Best practice, including REC completions, would be implemented strictly

Resource recovery rate	Amount of shale gas exploited by 2050 (bcm)	Cumulative shale gas emissions ($GtCO_2$) (2010-2050)	% of global emissions budget with 50% chance of < 2oC warming	Additional ppmv CO2 associated with shale gas (2010-2050)
10%	46 884	95	9.50%	5
20%	93 768	190	19.00%	11
30%	140 651	286	28.60%	16

Table 4.4 : Global Shale gas production and emissions estimates (Broderick et al. 2011)

Broderick et al. (2011) included 3 scenarios of recoverability from 10-30%, but a 20% recovery rate is representative of the average recovery from US shale to date, however there is sharp focus on improving recovery rates (Allen et al. 2013; EIA 2015). Under median estimates, the exploitation of these shale gas reserves by 2050 would emit 190 Gt CO_2 and occupy 19% of the global carbon budget with 50:50 chance of keeping warming below 2 ^{o}C (Anderson and Bows, 2011). This estimate does not include the emissions from coal, conventional gas or oil. Also, Rogelj et al. (2016) have claimed that 2/3 of the carbon budget to limit 2^{o}C warming had already been emitted. The combustion of the gas would raise the atmospheric CO_2 by 11 ppm, assuming 1 ppm is equivalent to 2.13 Gt of Carbon (Le Quere et al. 2009).

Broderick et al. (2011) highlight that it is almost certain that without strict global carbon limit agreements, these shale reserves will complement rather than replace current fossil fuels. This would enhance net CO_2 emissions and exacerbate already convoluted climate programs. For example, UK shale gas exploitation could promote an increase in the global carbon emissions. If LNG from Qatar and piped gas from the Netherlands and Norway were to be replaced by domestic UK shale gas, in this globalized world the gas could easily be re-directed to other countries considering the expected global energy demand is to increase by 48% by 2040. About 85% of this energy increase is predicted to be from non- OECD countries (US EIA 2016). It is highly unlikely these developing countries in Africa, S. America and Asia will opt for strict environmental controls and a low carbon economy at the cost

of economic growth, especially when developed nations have already done so in the past, and China is doing it currently (Broderick and Anderson, 2011). If the USA and Canada were unwilling to ratify the Kyoto protocol in 1997, it does not seem likely emerging economies will adhere to global carbon budgets if it is against their economic interest.

The greatest concern for the rise in long term global temperature is not the carbon emissions from each country, but the cumulative emissions over time from all countries, which is regarded as a more robust measurement for greenhouse gas and average temperature change association (Allen et al. 2009).

Overall, Jacoby et al. (2012), Broderick et al. (2011), Schrag (2012) and Mackay & Stone (2013) believe that in the absence of global climate policies to limit heavy carbon emissions, shale gas use could directly and indirectly increase global short term and long term emission rates. They all recommend that shale gas should be used solely to displace gas imports and as a replacement for heavier carbon fuels, and not for renewable energy generation, with the ultimate goal of expanding CCS technology and weaning off fossil fuels when alternative fuels are viable, affordable, and reliable.

4.6. What are the appropriate mitigation strategies for shale gas emissions and climate change?

Throughout the 1980's and 1990's, climate research accelerated with the creation of the IPCC in 1988 with the goal of presenting a global objective reconstruction of climate change and management techniques to minimize such rapid change. In 1989, the Montreal protocol was ratified which saw the regulation of fridges, aerosols and other sources to minimize CFC (Chloroflourocarbon) emissions which had proven to promote ozone depletion (Murdoch, 1997). Currently, CFC emissions are less than 10% compared to 1980 emissions, and the IPCC (2013) predicts their total removal from use in developed countries by 2020 and by 2030 in developing nations. The IPCC (2015) now proclaim that global greenhouse gas emissions must be reduced by 40-70% by 2050, and carbon neutrality must be achieved by 2100 if the world is to limit global warming to 2°C.

As regards shale gas, the IPCC (2013) stance is that hydraulic fracturing for shale and tight gas can be part of a positive transition towards a low carbon economy under appropriate regulation. They note this because coal use is currently the second largest energy source worldwide behind petroleum and is due to remain so until 2030, so there is large scope for reducing emissions (US EIA 2016). After 2030 coal is likely to fall to the third largest source of energy behind petroleum and natural gas, however coal global consumption will still increase at a rate of 0.6%/year from 153 quadrillion BTU in 2012 to 180 quadrillion BTU in 2040 (BP WER, 2016). In OECD countries, coal use is predicted to decrease with increased natural gas usage. In terms of energy intensity per kWh, the IPCC (2015) clearly state that this coal growth is much less desirable than the use of natural gas obtained by hydraulic fracturing. It is very important to ensure that shale gas will displace more carbon intensive fossil fuels such as coal, and that methane emissions are minimized, as currently 60% of total methane emissions are anthropogenic (DECC, 2016). These are two reasonable points of concern. There is little that can be done about the first problem from a national scale, as a global agreement on coal use limitation would have to be implemented, and with EIA growth numbers, it looks unlikely to materialize in non-OECD countries. On a national scale, shale gas could accelerate the demise of coal for electricity generation which would see positive climate reductions, as in the UK coal must be removed by 2025 at the latest (DECC, 2015). As regards controlling methane

emissions, the EPA Final Air Rules for the US Oil and Gas industry report required the compulsory use of REC technology on any new wells drilled from 2015 onwards (EPA, 2015).

Momentum is moving towards more appreciation of the human influence on the environment, with the new epoch the Anthropocene. Therefore, there needs to be a 2 phase mitigation strategy if shale gas expansion and other fossil fuels are to be in keeping with climate targets;

A) **Short and Long term methane emissions from all anthropogenic sectors must be nullified**
B) **Over longer terms, net CO_2 emissions must be decreased to as low as possible, ideally zero, to prevent further atmospheric CO_2 accumulation above the current 400ppmv.**

As an upshot, it is therefore desirable to minimize methane emissions from shale gas and other industries to prevent further rapid onset warming in this precarious climate condition, however it is important to understand current programs designed to do so. Firstly there is the US Federal Natural Gas star program which was established in 1993 to domestically share and trade knowledge of methane reduction techniques. Since its inception, 109 methane reducing parameters have been implemented which have removed 32.6 BCM of CH_4 emissions to date (NETL, 2011). On an international scale, the Global Methane Initiative (2016) was established in 2004 in the US and includes the co-operation of 43 other countries including Russia, the UK, Brazil and China. The goal of this group is to enhance the recovery of methane emissions from agriculture, coal mining, landfill and oil & gas production. Since 2005, 77.8 BCM of methane emissions have been successfully recovered that would otherwise have been emitted (US EPA, 2016). There are also voluntary internal US programs that have been set up in the advent of shale gas expansion such as the Centre for Sustainable Shale Development and the One Nation Energy Coalition which seek to implement REC technologies.

O'Sullivan et al. (2012) reports that REC implementation cost depends on the productivity of the well, with costs per well completion estimated to be from $39,000 for a Barnett Well to $166,000 for a Haynesville Well, bearing in mind some wells require up to 6 refracture treatments, liquid unloading and well completion

over its lifetime. Devon Energy (2008) estimates the average national cost of green completion to be $1000 per day, which at 2009 prices would have meant 95% of Barnett wells would still be profitable, and at $2000 a day 83% would remain profitable due to increased value of the well due to recovered potential fugitive emissions. In 2016, REC costs of $2000 would mean 65% of Barnett wells, some of the most productive in the country, would remain profitable (Mason et al. 2015). In these circumstances, REC application is a threat to the economic viability of the well. In the UK, this is likely to be different as the gas price in the UK has been 2.5 times the price of the US Henry Hub since 2012, which has recently tumbled over 70% to $2.60 since 2009 due to the mass expansion of hydraulic fracturing in the USA (EIA, 2016). REC completions will be required by law on UK shale sites as routine venting is illegal (DECC, 2016)

Mackay and Stone (2013) state that if shale gas expansion was to occur in the UK, economic viability is essential and thus emissions should be lowered to ALARP (as low as reasonably practical), dependent on the advancements made on limiting methane emissions from other sectors with larger emission rates. Centner (2016) states that emissions from shale gas can be drastically reduced under appropriate regulation of casing and cementing, handling of wastewater, venting and flaring and equipment rigor. This proposal would see methane limiting regulations complementary to current standards, but to include up to date REC technology developed in the US. By contrast, the CCC (2016) recommends that UK emissions should be lowered even further, as shown in figure 4.5.

As methane emissions from energy systems represent 12% of UK methane emissions, the relative cost-benefit of reducing emissions from one of the smallest contributors is poor (DECC, 2015). In the UK, fugitive emissions from natural gas industry have been decreased by 75.6% over 20 years, while over the same period methane emissions from agriculture have only decreased by 7.5% (DECC, 2015). A proposal is a ‘ Capital for Cows’ scheme whereby the capital generated from shale gas expansion under ALARP emission plans would be used to ideally nullify methane production by enteric fermentation, which is 48% of UK methane emissions (DECC, 2015). It is not practical, such as in the Global Methane Initiative, to capture emissions from animal agriculture as this diffuse source of methane would require either individual cow methane capture devices, or specially developed bio-gas facilities which are very expensive (Dach et al. 2014). Instead, it is more appropriate to nullify these emissions, as the incentive to re-use the methane as a fuel is

economically void since the concept of peak oil was discredited and more energy is available. A solution is to use a new type of feed with hops, honey and methanotrophic bacteria being developed at Cardiff University which can decrease ruminant methane production by over 90%, and also promotes energy retention which can increase animal growth rate, reduce bacterial infection risk and increase milk production (Blaxland- Pes Con, 2016). This would reduce direct methane emissions from animals, and could be more effective per £ spent when compared to natural gas reduction emissions, overall lowering UK methane emissions. Also the UK has the third greatest European animal agriculture methane emissions of 18,000 Kt CO_2e annually, while France and Germany emit 42,000 Kt CO_{2e} and 26,000 Kt CO_2e respectively per year (EU Climate report, 2015). This is the equivalent of 5.5% of annual German emissions and 13% of annual French emissions. So, at the very least, if there is a way in which UK can set a standard to reduce methane emissions, these two nations, among many others, will take great interest in any progress to reduce their own climate footprint.

The second part involves the control of CO_2 net emissions from shale gas production.

The first option is CCS (carbon capture and storage), which involves the burial of CO_2 gas produced from the combustion of fossil fuels to prevent further accumulation of greenhouse gases in the atmosphere. In the UK and around the world, the sectors longstanding economic and credibility issues have halted large programs such as the FutureGen CCS facility in the US and many across Europe. In the UK, £1 billion was pledged by the government to the White Rose CCS project, which sought to construct a oxy-fuel ultra-supercritical coal fired power plant in Selby, North Yorkshire and then pipe CO_2 emissions 165 miles offshore to the formerly hydrocarbon rich fields of the North Sea, such as the Goldeneye Oil field (DECC, 2015). The IPCC (2013) state that these formations could store CO_2 for millions of years, and they are likely to retain over 99% of the injected CO_2 over 1000 years. Up to 2 million tons of CO_2 per year as liquid or gas (90% of total emissions) would then be injected by 2020 into the depleted reservoir and plugged, with the impermeable red clap clay rock limiting leakage (DECC, 2015). The program, which involved stakeholders such as Shell, SSE and the national grid, was cancelled following the governmental removal of funds.

According to the Global CCS Institute, there are 13 carbon capture programs operational in the world, and a further nine under construction, which when

completed by 2025 should have the capacity to sequester 40 million tonnes of CO_2 /year (GCCSI, 2015). This is only 7% of UK CO_2 emissions, and in fact the only CCS projects operational in Europe are in Norway which sequester 1.6 million tons CO_2/year, which is at odds with the EU goal of a 220 million ton CO_2 burial capacity by 2030 (EU Climate report, 2015). The UK Energy and Climate change committee (2015) concluded that per 100bcm of gas extracted over 20 years in the UK, 3 CCS stations would be required to sequester the 200 Mt CO_2 produced. It was suggested that CCS could be expanded through technological support and through the use of carbon pricing to ease financial constraints. The largest CCS project in the world is the Alberta carbon trunk line which sequesters 14 million tons of CO_2 per year, and is being used for enhanced oil recovery of up to a billion barrels (Gaede, 2016). This is probably the best example of removal of CO_2 which would otherwise have been emitted, allowing the nation to meet near term emission regulations without threatening economic growth, however they state government financial assistance was imperative to meet the CAN$ 1.5 billion cost (Gaede, 2016).

Another alternative is the $10 million CarbFix Program in Iceland conducted by Icelandic and US researchers with the goal of developing a novel new technique of sequestering CO_2 as a solid and for training scientists on technological capabilities of CCS (Gislason et al. 2010). The technique of enhanced weathering involves reacting carbonated water (ratio of 1 ton CO_2:25 tons of water) at 400 °C with Calcium and Magnesium oxides present in 2000m deep geology, locking the CO_2 in a solid state with no dangerous byproducts (Gislason et al. 2010). In 2016, of the 250 tons of CO_2 injected over 2 years, 95% was sequestered as calcite (Kintisch, 2016). Benefits include its cost, which is less than 1/3 of conventional methods of CCS, however there are drawbacks as the ability to replicate this technology outside of Iceland is unknown, and in 2010 the World Geothermal congress reported that reinjection of liquid into these deep layers induced seismic activity (Gislason et al. 2010).

As far as the UK is concerned, the think-tank Energy and Climate Intelligence Unit (2016) state CCS is of central importance for UK emission rate minimization. They state although that current political and economic obstacles over the short term promotes stagnation in a technology which requires multi-decadal views of de-carbonization (ECIU, 2016).

There is large scope for shale gas wells to be reused post abandonment. Initially, as the shale wells are up to 3.5km below the surface they have the potential to be

used as geothermal energy sources for local communities, as the positive thermal gradient is ~25 ^{o}C per km. For example, Cuadrilla resources, a UK based shale Gas Company and GEL Ltd, a UK based geothermal company have signed a contract to share and allow the development of this technology in abandoned or existing wells. This will reduce the initial costs for geothermal energy production by 80% according to GEL ltd.

There is also scope for abandoned wells to be used for CCS. Shale gas wells have high initial production volumes, followed by a steep decline and a plateau of production caused by declining resources and the gravitational impact of the geology above the shale layer (Vidic et al. 2013). This forces induced fractures which were propped open by sand to be closed, decreasing shale basin permeability. This creates a large potential for the wells post decommissioning to be used for CCS, if nearby power stations can transport the produced CO_2 to depleted-reservoirs.

Using decommissioned wells and capital generated from shale gas production for CCS funding and application was recommended by the Committee on Climate change (2013), and encouraged further by the Task force on shale gas (2015). The IPCC (2013) state that installing CCS to existing power plants is not feasible as it would require total restructure, however it can readily be applied to a new constructed facility. The IPCC report on CCS application (2015) states that the cost of energy of the new generation of GTCC would be $0.03-0.05/kWh, and with capture and geological storage the cost is estimated to be $0.04-0.08/kWh. Using pulverized coal with CCS under IPCC (2015) calculations would cost up to $0.10/kWh as over double the amount of CO_2 is emitted per kWh of electricity produced when compared to gas. The IPCC (2015) estimate the cost of capture and geological storage of CO_2 from a gas fired power plant to be $20-40 per ton, and suggest that if CO_2 atmospheric concentrations are to be maintained between 450 and 750 ppm in a least cost portfolio, CCS has the potential to sequester 60-600 GtC cumulatively, representative of up to 55% of total global potential emissions by 2100. A large problem is initial investment requirements, which the IPCC (2015) estimate to be $1.5-3 billion between 2010 and 2050 and emphasise that this immature industry is by no means capable of providing this funding.

The main question is whether or not the rate of CCS global implementation will be adequate to minimize warming at current rates of emissions. Purely for the UK or

any other European nation to have net CO_2 emissions is inadequate to discontinue global rates of warming. As it is unlikely the world will see net zero emissions before 2100, that is another 80+ years of global emissions which are currently at the highest level in recorded history at 50 Gt CO_2e/year. Emissions are expected to rise further to 80 Gt CO_2e/year by 2050 driven by energy demand and economic growth in developing countries (US EIA, 2015). There is also scope for the atmospheric CO_2 concentration to increase at a greater rate solely from the decreased ability of the oceans and biosphere to sequester the CO_2 (McKinley, 2016). Under this scenario without very ambitious policies, atmospheric CO_2 concentration is likely to exceed 600ppm by 2050 (OECD Environmental Outlook), which increases climatic risk ever closer to an unknown threshold. Again a huge problem is that this heightened CO_2 concentration will remain as so for 100 or more years beyond net zero emissions given the longer residence time of atmospheric CO_2 (McKinley, 2016). So, technology which actively removes CO_2 from the atmosphere is desirable to prevent prolonged periods of heightened climatic risk, in which the earth would be more prone to ocean acidification sea level rise, storms and drought (IPCC, 2014).

This therefore requires negative emissions technologies, such as reforestation and direct air capture. Direct air capture technology is in its infancy, however current research at Columbia University is attempting to construct artificial trees which can sequester 1000 times more CO_2 than an actual tree, assuming it is of a similar size (Lackner et al. 2016). If reforestation was the option chosen, Mackay and Stone (2013) calculated that for every 100 bcm (130% of UK annual usage) of gas combusted, 5500km^2 of reforested land would be required to neutralize the emissions per year, which is equivalent to 3% of UK land area. Socolow & Tavoni (2013) state that for the most an understanding of the costs, benefits, applicability and technological capability of negative emissions technologies is limited. This is at odds with a Climate Change article by Gasser et al. (2015) which suggests that without the advancement and application of negative emissions techniques by 2050, fossil fuel use would have to be strictly limited to 20% of current usage if 2 °C warming is to be prevented. Lackner (2016) has stated that at current CO_2 accumulation rates of 2.2ppm and rising, the 2 °C warming threshold will be crossed at latest in 2032 and hence atmospheric carbon removal technologies are not just desirable but essential. Cao & Caldeira (2010) believe that the three biggest problems with Carbon removal are how little time there is to act, lack of research funding and no public acceptance of these technologies. Current emission targets

overall require carbon engineering in which net emissions are as low as possible and atmospheric concentrations can be maintained about a desirable concentration, but this concept is currently nothing more than a novelty at present.

The scale of the problem facing future climate policy is evidence in figure 4.12 shown below, which shows the necessary CO_2 reductions if the UK is to meet its requirements under the Climate Change Act (2008) (DECC, 2015).

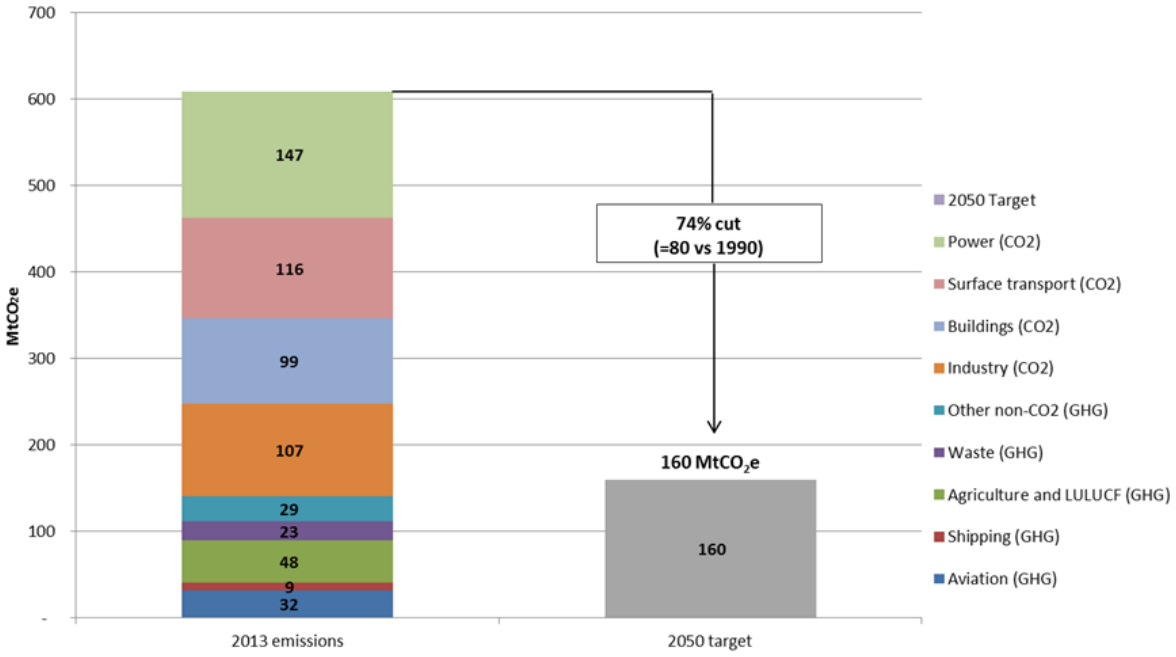

Figure 4.12: UK emission reduction requirements under Climate Change Act
Source: Committee on Climate Change

Hitting this climate target would require a 57% decrease in emissions by 2032 from 1990 levels, and a 74% decrease by 2050 from current levels, and previous UK legislation has seen a 35% decrease in annual emissions from 1990 levels (DECC, 2015). This decrease has been the easiest part of de-carbonisation through power generation, but further reduction methods will require more complex and expensive solutions. The Fifth Carbon Budget Progress Report (2015) shows that since 1990, UK emissions have fallen rapidly in the power sector due to the shift from coal to gas and renewables, but not in other industries. The removal of coal from power generation in favor of low carbon sources will only provide 40-50% of the reductions by 2032, and the CCC, (2016) states that a gap of 100 Mt CO_2e (47% of required reduction) exists between government plans and the required path to meet 2032 emission targets. The DECC and CCC both do not believe that this de-

carbonisaton program will be formed upon the basis of 1 solution, but a diverse web of complex solutions.

Pacala and Socolow (2004) suggested that global fossil fuel emissions could have been maintained at levels from 2004 (7 Gt C/year) using existing technologies such as fuel switching, reforestation, nuclear power, CCS and energy efficiency. They proposed that this would remove the business as usual emissions trajectories and by limiting annual fossil fuel emissions to 7 Gt C/year, rapid CO_2 accumulation in the atmosphere would be avoided. Instead of this, fossil fuels emissions have increased at a greater rate than predicted, with over 9 Gt C emitted from fossil fuel combustion in 2010 and in 2014 emissions were just under 10 Gt C/year, which is what Pacala and Socolow (2004) predicted business as usual emissions to be in 2025! They describe the biggest problem to be controlling emissions in a rapidly growing global population with an increasing energy demand of 40% by 2040. From 1812 to 1912, the population increased by 50% to 1.5 billion (UN DESA, 2015). From 1912 to 2012, the population increased almost 5 times to 7 billion, coinciding with the greatest period of atmospheric CO_2 accumulation in recorded history (UN DESA, 2015). The population is expected to grow to 9.7 billion by 2050, with 50% of the growth from Africa (UN DESA, 2015). Incremental energy demands from this growth will almost certainly be satisfied by fossil fuel use and will be at odds with global climate goals of ensuring global warming does not exceed the 2 °C threshold (BP WER, 2016).

A September 2016 report on the role of CCS in de-carbonisation stated that CCS has a central role to play in tackling climate change (Oxburgh, 2016). The report found that gas fired power stations with CCS could produce electricity at £85/mWh, less than the £92.50 EDF would charge for power generated from Hinkley Point (Oxburgh, 2016). This CCS cost does not include the initial infrastructure costs, which the report chair Lord Oxburgh states should be seen as a national infrastructure project like the Olympics and HS2 (Oxburgh, 2016). Government financing would be used to build the pipeline infrastructure for companies to utilize, with the remainder of the costs coming from private investment. The report recommends that if natural gas expansion in the UK takes place before the CCS infrastructure is operational, the new generation of power stations should be 'capture ready', i.e. with the capacity to switch to CO_2 burial when construction is completed (Oxburgh, 2016).

Renewable energy sources which do not emit CO_2 are useful to lower the global overall carbon burden. Global emissions are predicted to increase, so it is not only desirable to reach a state of net zero emissions, but also to remove CO_2 from the atmosphere to maintain optimal CO_2 atmospheric concentrations. Therefore a proposal, is, that emissions from shale gas wells should be lowered to as low as reasonably practical and profitable which includes REC technology. Capital tax generated from shale gas wells should be reinvested into CCS programs, and any means of reducing greenhouse gas emissions. Mackay and Stone (2013) suggest an ALARP (as low as reasonably practical) approach is required, with an adaptive shale gas regulatory framework dependent on the reduction capabilities of other sectors and how quickly CCS is commercially expanded. Also, as coal use has been replaced by gas over the past 40 years, with coal now supplying 20% of electricity demand compared to 55% in 1990, there is still scope for carbon removal from the UK power sector. Nevertheless, although shale gas appropriately can be described as the 'bridge fuel' from coal to gas, but there is a lot less of the bridge left to cross in the UK than there was 40 years ago and it is one which must be traversed carefully (Brandt et al. 2014).

5.0 Conclusion

5.1: Literature summary:

Pre-production emissions before the well is producing gas are almost certain to be higher from unconventional wells than conventional wells due to extra energy required for horizontal drilling and hydraulic fracturing. Pre-production emissions from shale gas exploitation are likely to be in the range of 656 up to 1790 tCO_2e/well (Jiang et al. 2011; Santoro et al. 2011; NYSEDC, 2011), which is up to 369 t CO_2e/well larger than emissions from the pre-production of a conventional well (Broderick et al. 2011). However, of central concern for understanding the greenhouse gas emissions from shale gas exploitation are the methane emissions from well completion and flowback fluids, just as combustion emissions have been the central concern for electricity production, which are 420-520 gCO_2e/kWh for natural gas compared to that of 840-1150 gCO_2e/kWh for coal (Mackay & Stone, 2013). In the USA, the shale gas revolution outpaced scientific research and policy regulation, so emissions were not strictly regulated from 2000-2010. From 2002-2014, US methane emissions increased by 30%, which has been suggested to be caused by shale gas exploitation among other sources (Tuner et al. 2016; Hausmann et al. 2016). However, the EPA (2014) improved regulation by requiring that from 2015 all new shale gas wells were fitted with Reduced Emission Completion (REC) technology to capture methane emissions which would otherwise have been vented or flared.

From a UK perspective, it is important to ensure methane emissions do not increase, and this should involve monitoring at onshore shale gas well pads to assist in super-emitter mitigation. By increasing the precision of methane measurements, it allows for better-informed policy decisions on how to minimise methane and to what extent from other sources if there are surplus methane emissions from shale gas exploitation. Although Howarth et al. (2011) state that shale gas has a larger greenhouse gas impact than coal, this declarative conclusion is supported by assumptions of widespread worst practice, which includes venting during well completion flowback and no REC implementation. Under appropriate regulation, shale gas would have a life cycle greenhouse gas impact of at most 60% of that of coal and as low as 75% of that of LNG (Mackay & Stone, 2013). At an estimated rate of 100 shale gas wells coming online per year in the UK, even under worst case scenarios, methane leakage from flowback and drill out could not exceed 20% of

annual enteric fermentation methane emissions from the UK dairy cow, sheep and cattle population. Overall therefore, shale gas will have an emission factor equivalent to, or slightly higher than conventional gas depending on pre-production emission control and well productivity (EUR). Methane emissions under surplus 'minimum necessary regulation' required by the Committee on Climate Change (2016) could not nullify the shift from coal to natural gas.

Over the short to medium term, shale gas can be a 'bridge fuel' in the UK if domestic production is used to replace natural gas imports and displace coal use. Over the longer term, net UK carbon emissions will have to decrease in line with the fifth carbon budget to reduce emissions by 80% from 1990 levels by 2050 (DECC, 2015). Meeting such a target will either involve the decreased used of fossil fuels or the implementation of CCS technology, with the latter being described by Oxburgh (2016) as the central focus for reducing annual emissions. At the very least, the next generation of efficient CCGS plants should be 'capture ready' if and when the appropriate CCS infrastructure is implemented by the government.

A global adoption of shale gas could reap huge climate benefits in nations that are largely dependent on coal, such as China, which generates over 70% of its electricity from coal (IEA, 2015). The US has seen a 9% reduction in CO_2 emissions from 2002-2012 of which a large reduction proportion has resulted from the switch to increased gas use, produced domestically from shale (EPA, 2016). It would be environmentally undesirable for shale gas to contribute surplus energy generation on top of current fossil fuel use or replace renewable energy production as this would increase the carbon burden and increase the risk of 2 °C warming.

It must be duly noted that even with efforts from OECD nations to decrease net emissions, global net emissions are projected to increase from current rates of 50 Gt CO_2e/year to 80 Gt CO_2e/year by 2050, with 85% of the growth from emerging economies (BP World Energy Report, 2016). This 'business as usual' emission growth is undesirable and will only contribute to the enhanced accumulation of CO_2 in the atmosphere. The UK currently emits ~1.5% of global greenhouse gas emissions, and even under the strictest possible climate regime, little contribution to global net emissions reduction would result (DECC, 2015). Therefore global initiatives such as the Paris Climate Agreement of 2016 are of great importance, and the UK should develop technology such as CCS to minimise emissions, and the technology can then be traded with developing nations who will be less likely to implement strict environmental controls. Lackner (2016) believes that CCS is

inadequate and that negative emissions technologies, such as direct air capture must be implemented because at current trajectories of CO_2 accumulation of 2.2ppmv annually, the 2 °C warming target will be passed in 2038, well before CCS could be globally effective.

To ensure shale gas does not exceed carbon budget limits, pre-production emissions must be minimised, shale gas must displace coal and replace gas imports, and any methane emissions from shale gas exploitation must be addressed by reduction from other methane sources, which Howarth et al (2011) describe as the 'low hanging fruit'. In terms of CO_2 emissions, the greatest potential for de-carbonisation is from the power sector, but the power sector only represents 13% of UK energy consumption and in terms of total energy usage, renewables are inadequate to meet national demand in the short and medium term without an energy revolution. Therefore, CO_2 and CH_4 emissions must be nullified and CCS is essential to ensure UK climate targets, with a wider variety of emission mitigating techniques being required over longer terms to prevent global warming above 2 °C regardless of shale gas exploitation.

5.2: Research Conclusions:

This investigation sought to quantify the methane flux from an onshore well pad and gas combustion site in the UK and explore the greenhouse gas emissions associated with shale gas exploitation documented in the literature. As there are no active shale gas wells in the UK, methane measurements from an unconventional well pad were not possible. Therefore, an investigation of methane emissions took place at the KM5 conventional well pad in production, which is analogous to a shale gas well in production. Other sites where methane emissions were measured include the Knapton generation station and, Cranford House farm in Ryedale. Results have been presented as ambient concentration and calculated as a flux using Gaussian plume modelling.

Overall results from data collection in North Yorkshire are as follows;

- Average ambient methane concentration at KM5 was 1.91 ppmv, which is lower than the Ryedale farm control at 1.93 ppmv.
- Knapton generation station average ambient methane concentration was 2.0 ppmv.
- Average daily fluxes ranged from the equivalent of 0.25 kg/year CH_4 from the KM5 well pad to 2820 kg/year CH_4 for Knapton generation station.
- The KM5 well pad had an average CH_4 flux of 24.3 kg/year and Knapton generation station had an average CH_4 flux of 847.9 kg/year.
- Ultimately, under worst-case emission scenarios, emission factors of 0.008% and 0.081% were calculated for the KM5 wellpad and Knapton station respectively.

5.2.1: Recommendations for future research:

The aims of this research have been met. Normal operation methane flux emissions from a producing onshore wellpad have been conducted successfully. It is unclear how representative the KM5 emission factor is of all wells in the UK, but it serves as a good dataset foundation for future work.

The literature assimilation has concluded that although there is a potential under poor regulation for large methane emissions to result. However, cost effective mitigation options exist which can ensure that methane emissions do not nullify the shift from coal to gas.

For future assessments of onshore conventional and unconventional well pads it would be very useful to have expansive bottom up means of assessing methane emissions. Doing so would likely involve more than one device per site, as the observed dynamism of wind speed and direction from this study would promote uncertainty in flux calculations. It would also be useful to conduct systematic top down assessments nationally to assess divergence between top down and bottom up methods of measuring methane emissions, as has been identified in the US (Allen et al. 2014; O'Sullivan et al. 2012). Doing so in any and all methane producing industries would promote greater transparency in methane budgets, enabling greater precision of flux calculations and therefore greater allowing more effective regulation of national emissions.

Data collection of methane fluxes would be valuable during pre-production, production and when decommissioned as these wells will have a legacy, and doing so will enable a comprehensive life cycle analysis to be conducted per well and nationally.

This research has shown that the greenhouse gas footprint of a producing conventional well, which is analogous to a producing unconventional well, will be the smallest proportion of its life cycle emission factor. Analysis of the Knapton gas processing facility in this study presents it to contribute a greater proportion of the total upstream GHG footprint, with an average flux over 30 times that of the well pad. The emissions from this kind of facility will be the same whether the gas originates from a conventional well pad or unconventional well pad. In terms of the local or national footprint, Knapton generation station is not a large methane source, and the worst-case scenario emission factor of 0.082% reflects this. Perhaps this rate of natural gas emissions could be seen as a standardised co-efficient of methane leakage per gas producing facility if this site is representative of national methane regulation from upstream gas facilities.

Research into UK emission factors would benefit from further analysis of downstream natural gas infrastructure, and it would be very useful to develop inexpensive comprehensive means of monitoring the extensive UK gas network for addition to upstream analysis conducted in this research.

The important questions raised in this research results include how methane emissions from upstream natural gas infrastructure should be further decreased and monitored cost effectively. Unless this research is a spurious representation of

such infrastructure, it does not represent a large proportion of the UK national methane budget. Nevertheless, as shale gas exploitation will be a relatively novel industry onshore in the UK, it is imperative to ensure a comprehensive understanding of the potential methane emissions, and mitigate accordingly with appropriate regulations to ensure future UK carbon budgets are not exceeded.

5.3: Emissions recommendations for the potential UK shale gas industry:

1) There should be monitoring of methane emissions during shale gas production to ascertain accurate methane fluxes from well pads. Measurements should particularly focus on periods with the greatest potential for large methane emissions. This monitoring will assist in policymaking and mitigation of the super emitters, such as identified in the USA.

2) Shale gas, if exploited, should be used solely to de-carbonise the power sector by displacing coal; and only to replace natural gas imports rather than add to current usage. Any surplus unabated gas consumption on top of current usage would be environmentally undesirable.

3) Methane emissions from shale gas exploitation should be as low as reasonably practical, and any surplus methane emissions from shale could be reduced from other industries where there is greater scope for methane emissions reduction at greater economic efficiency.

4) Ideally, the new fleet of CCGS power plants should be CCS 'capture ready' so as to minimise the future climate footprint in preparation for the 80% national emission reduction required under the Climate Change Act (2008).

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Tbl. 1

Date	Site	Average methane concentration (ppm)	Flux (Kg/year)	Average Isotopic signal (per mil)
13/10/2015	KM5	1.91	**7.22**	**-40**
14/10/2015	KM5	1.89	**9.81**	**-46**
15/10/2015	KM5	1.92	**20.41**	**-42**
16/10/2015	KM5	1.91	**9.28**	**-41**
19/10/2015	Ryedale Farm	1.89	*	**-54**
20/10/2015	Ryedale Farm	1.96	*	**-47**
21/10/2015	Ryedale Farm	1.88	*	**-48**
22/10/2015	Ryedale Farm	1.87	*	**-47**
26/01/2016	KM5	1.9	**0.52**	**-48**
27/01/2016	KM5	1.89	**0.94**	**-49**
28/01/2016	KM5	1.92	**56.61**	**-47**
29/01/2016	KM5	1.89	**0.00074**	**-50**
01/02/2016	Knapton	1.88	**3.86**	**-76**
02/02/2016	Knapton	2.26	**2,820.02**	**-31**
03/02/2016	Ryedale Farm	2	*	**-53**
04/02/2016	Ryedale Farm	1.94	*	**-51**
28/04/2016	Knapton	1.93	**194.57**	**-33**
29/04/2016	Knapton	2.19	**1,216.01**	**-29**
03/05/2016	KM5	1.91	**0.25**	**-58**
04/05/2016	KM5	1.93	**67.75**	**-58**
05/05/2016	KM5	1.94	**106.74**	**-48**
06/05/2016	KM5	1.97	**28.61**	**-52**
09/05/2016	Ryedale Farm	1.98	*	**-61**
10/05/2016	Ryedale Farm	1.97	*	**-58**
12/07/2016	Middleston Moor	10.97	**578,353.1**	**-38**
13/07/2016	Middleston Moor	11.33	**708,353.3**	**-38**
14/07/2016	Tow Law	1.9	**0.84**	**-47**
15/07/2016	Tow Law	1.97	**1030.17**	**-37**
18/07/2016	KM5	1.83	**N/A**	**-54**
19/07/2016	KM5	1.91	**9.60**	**-48**
20/07/2016	KM5	2	**22.44**	**-58**
21/07/2016	KM5	1.88	**N/A**	**-50**
25/07/2016	Knapton	1.88	**N/A**	**-56**
26/07/2016	Knapton	1.89	**5.11**	**-48**
27/07/2016	Ryedale Farm	1.92	*	**-54**
28/07/2016	Ryedale Farm	1.89	*	**-52**

Table 2

	Average concentration/ ppm	Average flux (kg/year)	Flux range
KM5	**1.9125**	**24.29862**	**0.25-106.74**
Knapton	**2.005**	**847.914**	**3.86-2820.02**
Ryedale	**1.93**	*	*
Middleston Moor	**11.1**	**643,353.20**	**578,353.1-708,353.3**
Tow Law	**1.94**	**515.505**	**0.84-1030.17**