METHANE AS VEHICLE FUEL – A WELL-TO-WHEEL ANALYSIS (METDRIV)

Report from a project within the collaborative research program *Renewable transportation fuels and systems*

September 2016

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THE SWEDISH KNOWLEDGE CENTRE
FOR RENEWABLE TRANSPORTATION FUELS

PREFACE

This project is financed and carried out within the f3 and Swedish Energy Agency collaborative research program *Renewable transportation fuels and systems* (Förnybara drivmedel och system).

f3 Swedish Knowledge Centre for Renewable Transportation Fuels is a networking organization which focuses on development of environmentally, economically and socially sustainable renewable fuels, and

- Provides a broad, scientifically based and trustworthy source of knowledge for industry, governments and public authorities
- Carries through system oriented research related to the entire renewable fuels value chain
- Acts as national platform stimulating interaction nationally and internationally.

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The project has been a collaboration between six f3 partners including both universities and companies as follows: Lund University, Environmental and Energy Systems Studies (Pål Börjesson, Mikael Lantz and Lovisa Björnsson) and Chemical Engineering (Christian Hulteberg and Helena Svensson); Bio4Energy/Division of Energy Science at Luleå University of Technology (Joakim Lundgren and Jim Andersson); E.ON (Björn Fredriksson Möller); Göteborg Energi (Erik Zinn); Scania (Eva Iverfeldt and Magnus Fröberg) and Volvo (Per Hanarp and Anders Röj).

This report should be cited as:

Börjesson *et al*. (2016) *Methane as vehicle fuel – A well-to-wheel analysis (MetDriv).* Report No 2016:06, f3 The Swedish Knowledge Centre for Renewable Transportation Fuels, Sweden, available at www.f3centre.se.

SUMMARY

The overall objective of this comparative systems study is to analyse and describe, from a well-to-wheel (WTW) perspective, the energy, greenhouse gas (GHG) and cost performance of existing and potential, new, methane-based vehicle systems solutions. Both thermal gasification (TG) systems using forest residues, anaerobic digestion (AD) systems using organic waste and residues, and natural gas (NG) systems are included, as well as different upgrading technologies and distribution systems including compressed and liquefied methane, gas grids and containers transported by trucks etc. The end-use technologies included are light-duty and heavy-duty vehicles using spark-ignited (SI) otto engines and dual-fuel (DF) diesel engines. The reference systems consist of gasoline-fuelled, light-duty vehicles and diesel-fuelled, heavy-duty vehicles. The GHG calculations are based on the methodology stated in the EU Renewable Energy Directive (RED), and the methodology recommended in the ISO standard of life cycle assessment.

The overall conclusion regarding the well-to-tank (WTT) GHG performance of the various renewable methane supply systems, is that these vary only to a limited degree. Thus, the selection of supply and distribution systems will have a minor impact on the WTW GHG performance. A similar conclusion can be drawn for the end-use technologies (tank-to-wheel, TTW), where the minor input of fossil diesel in DF trucks is partly compensated for by the higher energy conversion efficiency, compared with SI engine trucks. The WTW GHG reduction for the renewable methane systems analysed, compared with the reference gasoline and diesel systems, amounts to roughly 80% or more when the RED calculation methodology is applied. The corresponding reductions for NG-based systems are approximately 10%.

Applying the ISO calculation methodology will give similar reduction levels, but a somewhat changed interrelation between the TG and AD supply systems. Critical aspects regarding the WTW GHG performance are methane losses throughout the fuel chain. One example is methane boil-off emissions from on-board storage tanks of liquefied methane, which may occur if the trucks are not in operation for several days. The relative amount of diesel in DF trucks will also affect the GHG performance, which will be affected by driving patterns and transport operations, as well as the fuel consumption efficiency for SI engine trucks using compressed NG.

The WTW primary energy input is somewhat higher in methane-fuelled vehicle systems than in comparable gasoline- and diesel-fuelled vehicle systems, vaying from +3% up to +33% depending on the type of methane-based powertrain system. The WTW primary energy input in the systems using compressed methane in trucks with SI engines is in the range of 10-15% higher than systems using liquefied methane in DF trucks. If liquefied methane is used as energy carrier in methane-fuelled SI engine trucks, instead of compressed methane, the corresponding total primary energy input increases slightly. A critical aspect regarding the WTW energy efficiency for methane-fuelled SI engine trucks is the fuel consumption, since this may vary due to driving patterns and transport operations. It is assumed that the fuel efficiency of DF trucks and diesel trucks is similar.

The WTT costs of biogas (produced by AD) and bio-methane (produced by TG) vehicle fuel systems are estimated to be similar but these costs from smaller gasification systems are somewhat higher than the costs from the AD systems and larger TG systems. The costs of the different post-treatment and distribution systems of renewable methane are also comparable, and represent in the range of 20-40% of the total WTT costs. Thus, from an economic perspective, the selection of dif-

ferent production, post-treatment and distribution systems of renewable methane vehicle fuel systems are of minor importance. However, there are uncertainties in the WTT cost calculations performed, especially regarding the production costs of biogas and bio-methane.

The WTW costs of compressed methane-fuelled, light-duty vehicles are estimated to be in the range of 15-20% higher than the cost of gasoline-fuelled cars, indenpendently of renewable methane or NG. The WTW costs include the current market price of the fossil-based fuels, excluding VAT but including other relevant taxes, and the additional vehicle cost of methane-fuelled cars and trucks (thus not the complete cost of the vehicle). For light-duty vehicles, the additional vehicle cost is estimated to represent some 25% of the WTW cost. The WTW costs are sensitive to changes in the market price of fossil-based fuels, including changes in taxes for both fossil and renewable vehicle fuels.

Liquid biogas- and bio-methane-fuelled DF trucks have WTW costs similar to corresponding diesel trucks, whereas liquid NG-fuelled DF trucks have slightly lower WTW costs. Compressed methane-fuelled trucks are estimated to have roughly 15-20% higher WTW costs than diesel trucks. The additional TTW costs of methane-fuelled trucks are estimated to represent some 10% of the WTW costs, but this may vary from 5 to 15%. It is estimated that the additional vehicle cost for DF trucks and SI trucks are similar. The uncertainties in the production costs of biogas and bio-methane will have a significant impact on the WTW costs. The highest and lowest WTT costs included in the uncertainty analysis lead to 30-50% higher, and 25% lower WTW costs, respectively, for the renewable methane-fuelled trucks, compared with diesel-fuelled trucks.

The overall conclusions of this study are that the use of renewable methane vehicle fuel systems leads to significant WTW GHG benefits, compared with fossil-based vehicle fuel systems, that the WTW energy efficiency will be comparable or slightly lower than comparable gasoline- and diesel-fuelled vehicles, and that the WTW costs will be comparable or slightly higher, based on current market prices of fossil fuels. The selection of post-treatment and distribution system of renewable methane vehicle fuel systems will be of minor importance regarding the WTW GHG, energy efficiency and cost performance. Thus, there is an incentive to develop and commercially implement all of the various renewable methane systems assessed in this study.

SAMMANFATTNING

Den övergripande målsättningen med denna jämförande systemstudie är att analysera och beskriva energi-, växthusgas- och kostnadsprestanda för existerande och potentiella nya metanbaserade drivmedelssystem från ett well-to-wheel-perspektiv (WTW). Både förgasningssystem baserat på skogsbränslen och anaerob rötning baserat på organiskt avfall och restprodukter inkluderas, liksom naturgasbaserade (NG) system, samt olika uppgraderingstekniker och distributionssystem innefattande trycksatt och flytande metan, gasledning och containertransport med lastbil mm. Vid slutanvändning inkluderas lätta och tunga fordon med ottomotorer och dual-fuel dieselmotorer. Växthusgasberäkningarna baseras dels på den beräkningsmetod som är fastställd i EU’s Förnybarhetsdirektiv (RED), dels den som rekommenderas i ISO-standarden för LCA.

Den övergripande slutsatsen avseende växthusgasprestanda för de olika biobaserade metantillförselsystemen är att dessa varierar i liten omfattning utifrån ett well-to-tank-perspektiv (WTT). Val av produktions- och distributionssystem har således en relativt begränsad påverkan på den slutliga WTW-prestandan. Detsamma gäller val av fordonsteknologi (tank-to-wheel-perspektiv, TTW) då en begränsad inblandning av fossil diesel i dual-fuel motorer delvis kompenseras av den något högre motorverkningsgraden jämfört med ottomotorer som enbart drivs av biobaserad metan. Reduktionen av växthusgaser uppgår till normalt 80% eller mer ur ett WTW-perspektiv när biobaserade metandrivmedelssystem ersätter bensin och diesel och när RED’s beräkningsmetod används. Motsvarande reduktion för NG-baserade system är ungefär 10%.

När ISO-standardens beräkningsmetod används fås ungefär en motsvarande reduktion i växthusgaser men där det inbördes förhållandet mellan förgasningsbaserade system och rötningsbaserade system ändras. Kritiska aspekter för växthusgasprestanda är metanförluster längs hela produktionskedjan. Ett exempel är ”boil-off”-utsläpp från tankar med flytande metan när lastbilar står stilla ett flertal dagar. Andelen diesel som används i dual-fuel lastbilar påverkar också växthusgasprestandan liksom bränsleförbrukningen i lastbilar med ottomotorer som använde trycksatt naturgas. Dessa faktorer påverkas av körmönster och vilka transporttjänster som utförs.

Insatsen av primärenergi är ur ett WTW-perspektiv något högre för metan-baserade fordonssystem jämfört med bensin- och diesel-baserade system, mellan +3% upp till +33% beroende av system. Primärenergiinsatsen är cirka 10-15% högre i fordonssystem som använder trycksatt metan jämfört med flytande metan. När flytande metan används som energibärare i tunga fordon med ottomotor i stället för trycksatt metan ökar insatsen av primärenergi något ur ett WTW-perspektiv. En kritisk aspekt är bränsleförbrukningen av metan i tunga fordon med ottomotorer som påverkas av körmönster mm. Bränsleförbrukningen bedöms vara lika i en dual fuel-lastbil och en diesellastbil.

WTT-kostnaden för biogas från anaerob rötning och biometan från termisk förgasning bedöms vara ungefär lika men där kostnaden för mindre förgasningssystem beräknas vara något högre. Kostnaden för olika efterbehandlings- och distributionssystem för metan som drivmedel är också jämförbara och utgör 20-40% av den totala WTT-kostnaden. Från ett ekonomiskt perspektiv har således val av olika produktions-, efterbehandlings- och distributionssystem en begränsad betydelse. Det finns dock osäkerheter involverade i beräkningarna av WTT-kostnaderna, speciellt när det gäller produktionskostnaderna för biogas och biometan.

WTW-kostnaderna för personbilar som drivs av trycksatt metan beräknas vara 15-20% högre än för motsvarande bensinbilar. WTW-kostnaderna inkluderar nuvarande marknadspris för fossila driv-

medel, exklusive moms men inklusive andra relevanta skatter, samt den extrakostnad som tillkommer för fordon som drivs med metan (d v s hela fordonskostnaden är inte inkluderad). För lätta fordon uppskattas denna extrakostnad utgöra cirka 25% av WTW-kostnaden. WTW-kostnaderna är känsliga för förändringar i marknadspriserna för fossila drivmedel, inklusive förändrade skatter för fossila drivmedel och för förnybara.

Dual-fuel lastbilar som använder flytande förnybar metan beräknas ha jämförbara WTW-kostnader som dieseldrivna lastbilar, medan lastbilar som använder flytande NG beräknas ha något lägre kostnader. Lastbilar drivna av trycksatt metan beräknas ha 15-20% högre WTW-kostnader än dieseldrivna lastbilar. Extrakostnaden för metandrivna lastbilar beräknas motsvara cirka 10% av WTW-kostnaden man kan variera mellan 5-15%. Osäkerheterna i produktionskostnaderna för biogas och biometan har en stor påverkan på WTW-kostnaderna. De högsta WTT-kostnaderna för biogas och biometan som inkluderas i rapportens känslighetsanalyser beräknas medföra 30-50% högre WTW-kostnader för lastbilar drivna av förnybar metan, jämfört med dieseldrivna lastbilar. Å andra sidan medför det lägre WTT-kostnadsintervallet för biogas och biometan att WTW-kostnaderna för lastbilar drivna av förnybar metan blir 25% lägre än för dieseldrivna lastbilar.

De övergripande slutsatserna i denna studie är att fordonssystem baserat på förnybar metan leder till stora vinster ur växthusgassynpunkt jämfört med fossilbaserade fordonssystem, att energieffektiviteten är jämförbar eller något lägre jämfört med bensin- och dieseldrivna fordon samt att WTW-kostnaderna är jämförbara eller något högre när dessa baseras på marknadspriserna för fossila drivmedel. Val av efterbehandlings- och distributionssystem för förnybar metan som drivmedel har en mindre betydelse för fordonssystemens WTW-prestanda avseende växthusgasreduktion, energieffektivitet och kostnader. Därför är det motiverat att utveckla och kommersiellt implementera alla de system som inkluderats och analyserats i denna studie.

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1 INTRODUCTION

The interest in methane as a transportation fuel is growing quickly globally today for various reasons. One is the environmental benefits achieved in the form of reduced air pollution when diesel- and gasoline-fuelled vehicles are replaced, especially in densely populated areas where severe health problems occur due to air pollution (IEA, 2016). Methane may also lead to reduced emissions of greenhouse gases (GHG) when fossil liquid fuels are replaced, particularly when methane of renewable origin is utilised as vehicle fuel (JRC, 2014a,b). In Sweden, the major part of this methane is produced by anaerobic digestion (AD) of various organic materials. There is, however, one production plant in operation where renewable methane is produced by gasification of solid biomass (GoBiGas, 2016).

Another reason why the commercial interest in methane as a vehicle fuel is growing today is the increased extraction of fossil methane (natural gas) as shale-gas, for instance in North America (IEA, 2015). Thus, both the use of fossil and renewable methane has grown in the transport sector today, but the drivers behind this growth vary.

The use of methane as vehicle fuel is also growing in Sweden, albeit from a relatively low level. The current use is equivalent to approximately 2% (1.7 TWh per year) of the total fuel use in the road transport sector. This amount includes both renewable and fossil methane, with a market share of 1 TWh and 0.7 TWh, respectively (Energimyndigheten, 2015). Sweden is in the forefront globally regarding the use of renewable methane as vehicle fuel and the current systems are based on compressed methane, but there is an ongoing development of also liquefied methane as vehicle fuel. Thus, the commercial market for biogas is still under development and there exist several technical solutions regarding upgrading, distribution and final use in dedicated vehicles that are not yet fully commercialised.

1.1 AIM AND OBJECTIVES

In this comparative system study, the performance of existing and potential, new, methane-based vehicle fuel systems solutions is analysed and described from a well-to-wheel perspective. The parameters evaluated are energy efficiency, GHG reduction and costs. Important aspects are how different production systems, post-treatment technologies, distribution systems and end-use technologies will affect the performance of the various vehicle fuel systems. This systems knowledge is to a large extent lacking today, and is important for commercial actors, such as investors in gas handling systems, manufactures of gas vehicles etc. The comparative performance of the different systems is also important as input for public agencies and policy makers regarding prioritising R&D activities and in the development of various policy instruments.

2 METHODS AND GENERAL ASSUMPTIONS

2.1 NOMENCLATURE

The terminology for methane utilised as vehicle fuel in the gaseous or liquid form is not consistent in the literature. In addition, a wide range of different production/handling pathways is evaluated in the present study. The following denotations will thus be used throughout the study.

- Methane of fossil origin is denoted by the widely applied term natural gas, abbreviated NG.
- Methane of renewable origin is denoted as biogas if it has been produced by anaerobic digestion (AD) and bio-methane if produced by thermal gasification (TG).
- Compressed and liquefied methane of fossil origin will be denoted as compressed natural gas (CNG) and liquefied natural gas (LNG), respectively.
- Compressed and liquefied methane of renewable origin will be denoted as compressed biogas (CBG) and liquefied biogas (LBG), respectively. If a distinction is needed between methane produced by AD or TG this will be expressed as CBG-AD etc. If in addition the scale of production is relevant, this will be denoted as CBG-AD-30 GWh etc.

In the end-use of methane in vehicles, two engine types are evaluated, the spark-ignited otto engine denoted as SI and the dual-fuel methane-diesel engine denoted as DF.

Gas volumes given throughout the study are given as dry gas at 0 °C and 101 325 Pa. When methane production is given in energy units, the value given refers to the lower heating value. In the conversion of volume to energy units, the lower heating value of methane is used, where 1 m^3 corresponds to 35.8 MJ.

The terminology for the systems boundaries of the technical systems assessed follows the terminology utilised by the JRC (2013; 2014a,b). Thus, the complete system from feedstock to transport service is denoted as well-to-wheel, or WTW. The system including feedstock to production and distribution of vehicle fuel is denoted as well-to-tank, or WTT, whereas the system including only the end-use in vehicles is denoted as tank-to-wheel, or TTW.

2.2 SYSTEMS DESCRIPTION AND ASSESSMENT APPROACH

The assessment presented in this study includes the following four main parts: i) end-use, ii) filling station, iii) post treatment and distribution, and iv) production and upgrading. A more detailed description regarding the different technical solutions, and their combinations, included in this study is presented in Table 2.1. For comparison, distribution and utilisation of NG are also included, but are based on existing generic data. Thus, no explicitly new calcualtions have been made regarding CNG and LNG.

The assessment approach is thus from a wheel-to-well perspective as compared to the more normal well-to-wheel. The reason is that this study is primarily focused on the gate-to-wheel assessment and in particular, the final utilisation. However, production of methane by AD or TG is also included in order to present a complete well-to-wheel perspective.

Both light-duty vehicles, such as family cars, and heavy-duty vehicles, or long-haul trucks, are included in the study. Compressed methane (CBG and CNG) is used in light-duty vehicles, replacing gasoline, and both compressed and liquefied methane (CBG/LBG and CNG/LNG) are used in heavy-duty vehicles, replacing diesel. The transport and distribution of the methane involves gas grid systems, both low and high pressure systems, and containers (steel and composite) for compressed methane transported by truck. Liquefied methane is transported by truck to the filling station from the production plant. Liquefied NG is delivered by boat and then transported by truck.

The inventory and compilation of data is based on a combination of literature review and personal contacts with key actors within the specific technical fields included in the methane vehicle fuel systems (see Table 2.1). The aim is to utilise current data as far as possible, and no older than 2-3 years. The technology status throughout the complete WTW systems represents the best available technology (BAT) on the market today. In sensitivity analyses, critical parameters are tested, illustrating non-optimal conditions in the various steps in the methane vehicle fuel systems. Data for reference systems, or gasoline-, diesel- and NG-based WTW systems, represent generic data presented by JRC (2013; 2014a,b).

The production, distribution and final use of methane is assumed to take place in Sweden, except for the production of NG which is assumed to be imported from abroad. The results of the assessment regarding energy efficiency, GHG performance and costs are expressed per kilometre of transport service as well as per MJ fuel. A detailed description of the calculation methods utilised in the energy, GHG and cost assessments is given in Sections 2.3 and 2.4.

Table 2.1. Description of the technical systems of methane-based vehicle fuel solutions covered by the assessment.

	Biogas (from anaerobic digestion - AD)		Bio-methane (from thermal gasification - TG)	
I. End-use	Light-duty vehicles – methane SI (spark-ignited) engine			
	Heavy-duty vehicles – methane SI (spark-ignited) engine (Scania)			
	Heavy-duty vehicles – methane diesel DF (dual-fuel) engine (Volvo)			
II. Filling station	Container (gaseous)			
	Gas grid			
		Container – liquid & gaseous		
III. Distribution	200 bar – truck & steel container			
	250 bar – truck & composite container			
	Low pressure gas grid (existing)	High pressure gas grid (existing)		
		Liquefied – delivered by truck		
IV. Post treatment	Compression			
		Liquefaction		
IV. Upgrading	Water scrubber			
	Amine scrubber			
V. Annual production	30 GWh	100 GWh	520 GWh	1 600 GWh

2.3 ENVIRONMENTAL ASSESSMENT

The accounting method used is based on life cycle assessment (LCA), as described by the standards ISO14044 (ISO, 2006). The environmental impact is limited to emissions of greenhouse gases (GHG) since this is the focus in recent policy and regulation of biofuels in the EU (Directive 2015/1513/EC; Directive 2009/28/EC and Directive 2009/30/EC) (European Commission, 2009a,b; 2015), among others. The GHG performance is calculated using two different methods, the system expansion approach (according to ISO 14044, hereafter called “ISO-calculation”) and the energy allocation approach (according to the EU Renewable Energy Directive, RED, hereafter called “RED-calculation”) (see Figure 2.1). In addition, the energy efficiency performance of the various methane well-to-wheel systems is also calculated.

The system expansion method (ISO-calculation) includes the effects of potential by-products generated in the vehicle fuel production system, by calculating the indirect GHG effects of the by-products when they replace alternative reference products (substitute product). One example is digestate from anaerobic digestion utilised as fertiliser replacing mineral fertilisers and thereby reducing the need and production of mineral fertilisers. The system expansion method also includes potential indirect effects on soil carbon contents, affecting the GHG balance, when biomass feedstock is harvested. One example is when logging residues from forestry are harvested, leading to a somewhat reduced carbon content in the forest soil. The RED-calculation method does not include any indirect effects on soil carbon content due to biomass feedstock harvest. Furthermore, the GHG effects of potential by-products genereated in the vehicle fuel production system are handled by dividing (allocating) the total GHG eminssions from the production systems between the biofuel and the by-product, based on their lower heating value (LHV). Since, for example, digestate from anaerobic digestion normally has a water content above 90%, the LHV is assumed to be zero, leading to no allocation of GHG to the digestate and 100% allocation to the biofuel.

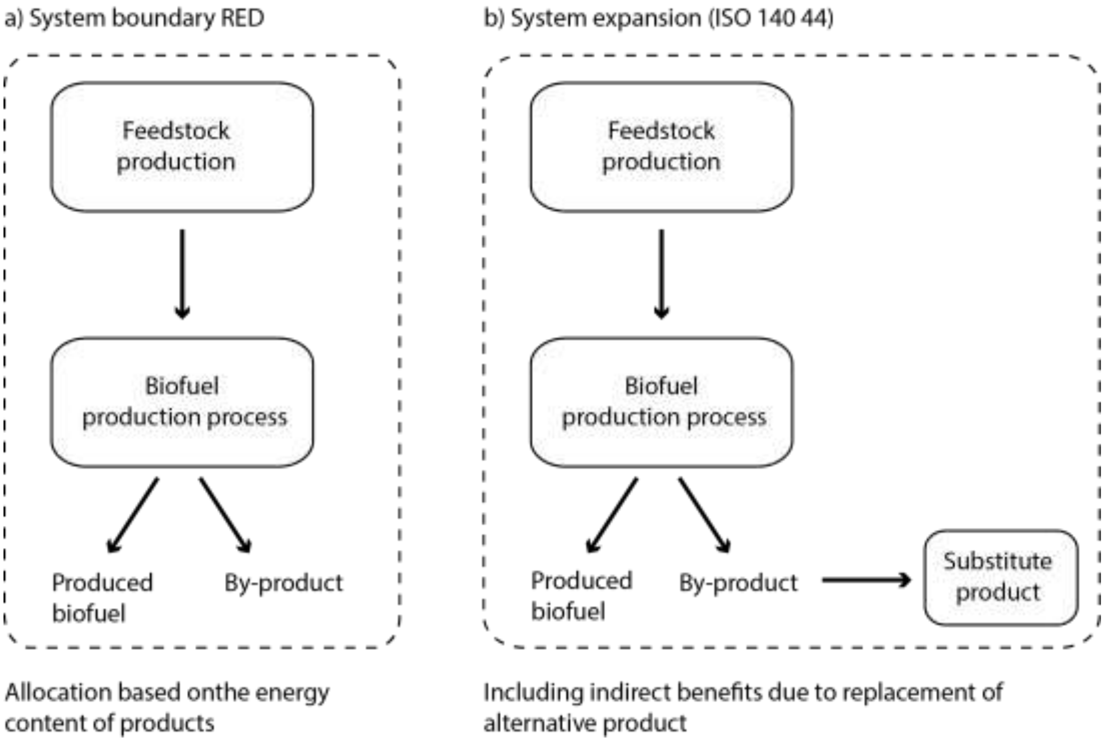

Figure 2.1. Greenhouse gas calculation methodologies utilised in this assessment (Börjesson et al, 2013).

Functional unit

In this study two different functional units (FU) are used; i) MJ methane at filling station, and ii) km transport service. In the latter FU, potential variations in fuel efficiency in different vehicle engine concepts are also included.

Data

The energy and GHG WTW data of renewable methane systems are based on current technical performance and current Swedish conditions. GHG calculations are based on data from manufactures and firms within the renewable methane sector and gas vehicle sector, in combination with data from complementary literature. Thus, the energy and GHG data represent today's best avalialable technology, BAT, both regarding renewable methane production systems and methane engine vehicles. The energy and GHG WTW data regarding the fossil fuel reference systems are based on the latest WTW reports from JRC (2013; 2014a,b).

GWP characterisation factors

The GHG emissions included in the assessment are fossil carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). For WTT calculations according to the ISO standard performed in this study, characterisation factors are based on the latest update by IPCC (2013), where 1 g CH_4 and

1 g N_2O are equivalent to 34 g and 298 g CO_2-equivalents, respectively. For calculations according to the RED, the corresponding factors are set to 23 g and 296 g CO_2-equivalents, respectively. Regarding TTW GHG emissions (not included in the RED), CH_4 and N_2O characterisation factors are taken to be 34 g and 298 g CO_2-equivalents, respectively. The fossil fuel reference systems (gasoline, diesel and NG), which are based on the latest reports from JRC (2013; 2014a,b,), utilise the characterisation factors 25 g and 298 g CO_2-equivalents, respectively. Thus, some variations in the WTW GHG performance between the different systems will arise from somewhat different characterisation factors, especially regarding methane, but sensitivity analyses show that these differences will have a minor impact on the overall results. All the characterisation factors above refer to the global warming potential (GWP) for a 100-year perspective. For short-lived gases, such as CH_4, the GWP value increases to 86 on a 20-year time horizon, whereas the GWP value for long-lived gases, such as N_2O, decreases to 268 (IPCC, 2013). Thus, depending on which GHG emission that is most critical for the overall GHG performance of the biofuel system, changes in time perspective will lead to different results. This is, however, not analysed further in this study where we use a 100-year perspective for the GWP in all calculations.

Energy balance

Energy balance calculations include all input of primary energy required for the production and distribution of renewable methane. The energy balance also includes all energy required to collect, transport and pre-treat raw materials such as organic waste and wood chips. Energy embedded in the raw material is not included in the energy balance. Wood chips used to produce process energy is, however, not considered a raw material but an energy carrier. In that case, energy embedded in the chips is also included.

2.4 ECONOMIC ASSESSMENT

The economic assessment performed in this study is based on a traditional investment analysis in which the annual cost of capital is calculated according to the annuity method.

The depreciation period for investments in biogas and upgrading plants as well as filling stations is set to 15 years which is also the estimated life span suggested by the Swedish EPA when calculating the need for investment subsidies within the current Swedish climate investment programme (Naturvårdsverket, 2016). For investments in district heating and other kinds of production facilities the Swedish EPA suggests 25 – 30 years. Based on discussions with industrial partners within this project the depreciation period for the gasification plants is thus set to 25 years. For additional investments in gas vehicles, as compared to conventional ones using liquid fossil fuels, the depreciation period is set to 10 years.

The weighted average cost of capital (WACC) is set to 6% using the same assumption as Nohlgren et al. (2014) to calculate the production cost of electricity from new facilities. In the sensitivity analysis, the effect of choosing a WACC of 10% is also evaluated.

The economic assessment also includes the cost of raw material, process energy, operation and maintenance as well as the sales of surplus electricity and heat produced in the production process.

As presented earlier in this chapter, the production cost of biogas is based on a literature review and is not calculated as part of this study.

3 INVENTORY

3.1 END-USE

Tank-to-wheel data regarding methane-fuelled heavy-duty vehicles are based on Euro VI engines, representing long haulage / regional trucks equivalent to a total gross weight of 30 tonnes. Two different types of trucks are included, one equipped with an internal combustion, spark-ignited (SI) engine manufactured by Scania (*methane otto engine*), and one with a prototype, dual-fuel (DF) methane diesel engine planned for serial production by Volvo. The methane otto engine is fuelled with CBG/CNG in the base case, but calculations are also made for LBG/LNG for storage on-board, which is then vaporised before injection. The dual-fuel methane diesel engine is fuelled with LBG/LNG mixed with a minor portion of diesel.

In Table 3.1, the TTW emission levels and fuel efficiency regarding heavy-duty methane vehicles (base-case) are shown. TTW emissions for diesel engines are also included as a reference. All vehicles meet the Euro VI standard. The measured emission levels of methane also include crankcase ventilation. Methane emissions from on-board storage tanks are estimated to be marginal in the base-case, both regarding storage of compressed and liquefied methane.

The emissions of CH_4 and N_2O are significantly higher for cold engines than for warm engines (see Willner & Danielsson, 2014), but since we assume long haul transports in our base-case, only emission data from warm engines are used in our calculations.

The TTW fuel consumption is estimated to be, on average, 18% higher per km for SI gas engines than for diesel engines (Fröberg, 2016). The fuel efficiency for DF engines and diesel engines is assumed to be similar (Hanarp, 2016). Compared with a traditional, diesel-fuelled, heavy-duty vehicle (Euro VI), the emissions of GHG are approximately 12% and 99% lower for gas engine vehicles fuelled with compressed natural gas and biogas, respectively. The corresponding reduction is estimated to be approximately 20% and 90% for dual-fuel engine vehicles fuelled with liquefied natural gas and biogas, respectively. The methane/diesel ratio is assumed to be 0.95/0.05 in the base-case (Alamia, 2015).

Table 3.1. TTW GHG emission levels and fuel efficiency for heavy-duty vehicles approved for emission standard Euro VI.[1]

Engine	GHG emissions								Fuel efficiency
	CH_4		N_2O		CO_2		GWP (CO_2-eq)		
	g/MJ	g/km	g/MJ	g/km	g/MJ	g/km	g/MJ	g/km	MJ/km
CNG (SI)	0.007	0.08	0	0	56.2	641	56.4	644	11.4
CBG (SI)	0.007	0.08	0	0	0	0	0.24	2.7	11.4
LNG (DF)[2]	0.056	0.54	0.006	0.058	57.1	552	60.8	588	9.68
LBG (DF)[2]	0.056	0.54	0.006	0.058	3.7	35	7.4	71.6	9.68
Diesel	0.002	0.019	0.006	0.058	73.2	709	75.0	727	9.68

[1] **Data from Willner & Danielsson (2014) regarding CH_4 and N_2O emission levels (representing a warm engine). Fuel efficiencies are based on data from Volvo Trucks (Harnarp, 2016) and Scania (Fröberg, 2016). Carbon dioxide emissions are based on JRC (2014a).**

[2] **5% diesel based on energy content (Alamia, 2015).**

The TTW fuel efficiency and GHG emission data regarding light-duty vehicles are based on the latest report from JRC (2014a). The time horizon is 2020+, which is assumed to be equivalent to Euro 6 engines. The powertrain regarding CNG/CBG is assumed to be port injection spark ignited engine (PISI), whereas the gasoline-fuelled reference powertrain is assumed to be direct injection spark ignited engines (DISI) (JRC, 2014a), which are assumed to be the most realistic and common technologies on the market during the coming years.

Input data regarding TTW GHG emission levels and fuel efficiencies for light-duty vehicles are shown in Table 3.2. The fuel consumption, expressed as MJ per km, is estimated to be 7% higher for CNG/CBG-fuelled vehicles than for gasoline-fuelled vehicles. The TTW GHG emissions are estimated to be approximately 22% lower for CNG vehicles, than for gasoline vehicles, and more than 98% lower for CBG vehicles. The methane emissions from CNG/CBG vehicles represent a methane leakage equivalent to approximately 0.2%. Previous estimations of methane leakage in CNG/CBG-fuelled, light-duty vehicles vary from 0.02 (Westerholm, 2008) up to 0.45 g/MJ (Uppenberg et al., 2001), where a best estimate has been 0.04 g/MJ (Börjesson et al., 2010). This is almost equivalent to the input data presented in Table 3.2. For comparison, the maximum level of methane emissions, according to the European emission legislation for gasoline engines, is 0.1 g CH_4 per km today.

Table 3.2. TTW GHG emission levels and fuel efficiency for light-duty vehicles equivalent to emission standard Euro 6.[1]

Engine	GHG emissions								Fuel efficiency
	CH_4		N_2O		CO_2		GWP (CO_2-eq)		
	g/MJ	g/km	g/MJ	g/km	g/MJ	g/km	g/MJ	g/km	MJ/km
CNG (SI)	0.042	0.064	0.001	0.002	56.2	85.4	57.9	88.0	1.52
CBG (SI)	0.042	0.064	0.001	0.002	0	0	1.73	2.68	1.52
Gasoline	0.010	0.014	0.001	0.002	73.4	104	74.0	105	1.42

[1] Data from JRC (2014a).

The sensitivity analysis regarding the TTW GHG emissions and fuel efficiency focuess on heavy-duty vehicles and includes the following assumptions (changes from the base case):

- 10% diesel consumption in DF methane diesel trucks (5% in base case)
- 1% and 3% methane losses from LBG/LNG on-board storage tanks by venting (insignificant losses in base case)
- 10% and 25% higher fuel consumption in CBG/CNG fuelled SI trucks compared with diesel trucks (18% higher in base case)

The increased diesel consumption in dual-fuel methane diesel trucks is assumed to reflect less than optimal conditions when the engine load varies to a larger extent, compared with normal long-distance transport situations. A rough estimation is that the diesel consumption may be doubled, from 5 to 10% of the total fuel use.

The time which the on-board methane tank can hold LNG/LBG without venting is called the “holding time”. Tanks are typically designed for holding times of five days (Delgado and Muncrief, 2015). However, because of atmospheric heat penetrating the tank, some of the LNG continuously evaporates. Once this “boil-off” gas reaches a certain pressure level, it is vented for safety reasons to the atmosphere through a pressure relief valve. Boil-off rates depend on several different parameters and are thus difficult to estimate. If the truck is left unused for several days with partly filled LNG/LBG tanks, there is a risk of methane losses through venting. Data of the magnitude of vented emissions is limited today but one estimation is that, on average, 2.6% of the initial amount of liquid methane in the tank is vented off per venting event (Delgado and Muncrief, 2015). Thus, a rough estimation in this assessment is that “boil-off” methane losses by venting may amount to 1 to 3% if the trucks are not utilised continuously. It should be noted that in normal continuous operations, LNG/LBG trucks should require minimal venting of methane, as assumed in the base case of this analysis.

The fuel consumption per kilometre transport distance is estimated to be somewhat higher for heavy-duty vehicles equipped with spark-ignition gas engines, compared with diesel and dual-fuel engines. There are many differing claims regarding the fuel consumption in CNG/CBG trucks. A rough estimation is therefore that the fuel consumption in CNG/CBG-fuelled trucks will be between 10 to 25% higher than the fuel consumption in diesel and dual-fuel trucks, depending on duty cycle.

Table 3.3 shows how the TTW GHG emissions and fuel efficiency performance are affected by the changes in input parameters in the sensitivity analysis, compared to the base case. An increase in diesel consumption in dual-fuel trucks, from 5% to 10%, results in approximately 6% higher GHG emissions regarding LNG-fuelled vehicles, whereas the GHG emissions increase by roughly 50% regarding LBG-fuelled vehicles. One and three percent boil-off methane emissions leads to approximately 11% and 34%, respectively, higher GHG emissions from LNG-fuelled trucks. Considering LBG-fuelled trucks, the corresponding increase in GHG emissions is doubled and almost by four times, respectively. The changes in fuel efficiency concerning CNG/CBG trucks will lead to a corresponding magnitude of change in GHG emissions.

Table 3.3. Changes in TTW GHG emission levels and fuel efficiencies, compared with the base case, when specific parameters for heavy-duty vehicles are varied.

Changed parameter		GHG emissions (CO_2-eq)				Fuel efficiency	
		Base case		Sensitivity analysis		Base case	Sensitivity analysis
		g/MJ	g/km	g/MJ	g/km	MJ/km	
10% diesel in DF	LNG	60.8	588	61.6	596	-	-
	LBG	7.4	71.6	11.1	107	-	-
1% boil-off methane losses / 3% boil-off methane losses[1]	LNG	60.8	588	67.6 / 81.2	654 / 786	9.68	9.78 / 9.97
	LBG	7.4	71.6	14.2 / 27.8	137 / 269	9.68	9.78 / 9.97
	LNG-SI (CNG)	56.4	644	63.2 / 76.8	720 / 876	11.4	11.5 / 11.7
	LBG-SI (CBG)	0.24	2.7	7.04 / 20.6	80.3 / 235	11.4	11.5 / 11.7
10% higher fuel consumption/ 25% higher fuel consumption	CNG	56.4	644	52.6 / 59.7	600 / 681	11.4	10.6 / 12.1
	CBG	0.24	2.7	0.22 / 0.26	2.5 / 2.9	11.4	10.6 / 12.1
Diesel		75.0	727	-	-	9.68	-

[1] 1% methane losses are equivalent to 0.2 g CH_4 per MJ.

An analysis of the price levels of light-duty bifuel vehicles, compared with corresponding gasoline-fuelled vehicles, shows that the selling price is often approximately 10% higher. The difference in price between bifuel and gasoline-fuelled cars varies with brand and model (see e.g. bifuel models of Audi, Fiat, Ford, Mercedes, Opel, Seat, Skoda, Subaru, Volkswagen and Volvo). A best estimate in this study is therefore that CNG/CBG light-duty vehicles are, on average, 25 000 SEK more expensive than corresponding gasoline-fuelled vehicles (with an average selling price of 250 000 SEK). Based on a life time of 10 years for a passenger car, and a total driving distance of 250 000 km, the increased cost per km is estimated to be, on average, 0.14–0.16 SEK at an interest of 6% (base case) and 10% (sensitivity analysis).

The increased costs of methane-fuelled, heavy-duty vehicles, compared to corresponding diesel-fuelled vehicles, are estimated to vary between 10 to 30% depending on model and whether compressed or liquefied methane is utilised as fuel. A traditional, diesel-fuelled truck is estimated to cost, on average, 1 million SEK, thus the estimated increase in costs for methane-fuelled trucks corresponds to roughly 100 000 to 300 000 SEK. The total driving distance for a long haulage/regional service truck is assumed to be, on average, 1 000 000 km, having a life time of approximately 10 years (Fröberg, 2016; Hanarp, 2016). The increased costs per km are then estimated to vary between 0.14 and 0.41 SEK, with an average of 0.28 SEK, compared with a corresponding traditional diesel truck, at an interest of 6% (base case). With an interest of 10%, the cost is 0.16–0.49 SEK/km, with an average of 33 SEK/km (sensitivity analysis). A rough assumption in the cost calculations is that the additional costs of methane-fuelled DF trucks and SI trucks are similar.

3.2 FILLING STATIONS

The assessment includes three different types of methane filling station. The first type is represented by a filling station where the methane is supplied by an existing, low pressure gas grid. The second alternative is a CBG/CNG filling station in which the compressed methane is supplied in containers transported by trucks. Both of these types of stations deliver compressed vehicle gas at a pressure equivalent of 200 bar (Energigas, 2016).

The third alternative represents a LBG/LNG filling station where the liquefied methane is supplied in containers. In order to avoid leakage of boil-off gas, it is assumed that the filling station has re-condensation capacity, or that the filling station has an alternative utilisation pathway for the boil-off gas, such as CBG. However, this alternative utilisation pathway involving compressed gas is not included in the following analyses regarding energy efficiency, GHG emissions and economy. Thus only the liquefied methane re-condensation alternative is further assessed.

In Sweden, there were 211 methane filling stations in operation in 2015, of which 42 were dedicated to busses (SCB, 2016). The methane sold the same year amounted to 1.6 TWh, with an average annual gas sale corresponding to almost 20 GWh for stations dedicated to busses and 4.5 GWh per station for the remaining filling stations.

To demonstrate the conditions for a somewhat more evolved vehicle gas market, it is assumed that CBG stations sell 10 GWh of vehicle gas annually and that LBG stations sell 30 GWh annually.

Inventory data on calculated electricity consumption and estimated investment costs, as well as operation and maintenance are presented in Table 3.4.

Table 3.4. Inventory data regarding costs and electricity consumption for different types of methane filling stations.

	CBG - grid	CBG – off grid	LBG
Vehicle gas (GWh/year)	10	10	30
Investment (SEK)	7 500 000[1]	7 500 000[1]	15 000 000[2]
Operation and maintenance (SEK/MWh)	50[3]	50[3]	50[4]
Electricity (kWh/m^3)	0.3[5]	0.07[5]	0.04[6]

[1] The investment cost varies considerably between different stations due to local conditions and choices made regarding capacity, equipment and redundancy etcetera. For a station with the capacity to fill both light duty and heavy duty vehicles the investment is estimated to be 7-8 million SEK (Emebrant, 2016).

[2] The investment cost for an LBG station also depends on local conditions and choices made regarding design and capacity etc. There are also few stations built today and an expanding market with more stations would probably affect the price. In this study, the investment cost is estimated to be 15 million SEK (Fredriksson Möller, 2016; Nilsson, 2016).

[3] The costs of operation and maintenance of filling stations are estimated to be 0.4-0.5 SEK/m^3 at an annual volume of approximately 10 GWh. Smaller volumes would probably lead to a higher cost per Nm^3 (Emebrant, 2016).

[4] Operation and maintenance costs for LBG stations have not been identified and is therefore estimated to be the same as for CBG stations.

[5] Based on Benjaminsson and Nilsson (2009)

[6] Electricity consumption varies over the year and somewhat between different filling stations. An annual average is estimated to be 0.04 kWh/Nm^3 (Zachrisson, 2016).

3.3 POST-TREATMENT AND DISTRIBUTION

Biogas and bio-methane can be distributed in gas grids or by truck. When distributed by truck the gas is compressed (CBG) or liquefied (LBG). In Sweden, 70% of the biogas utilised as vehicle fuel is distributed by truck as CBG and 30% is distributed via the natural gas grid or the vehicle gas grid in Stockholm (SEA, 2015).

In this study, several different distribution options are evaluated. Since transportation of CBG by truck is the most common solution, this is evaluated for all production plants, comparing different material of the containers (steel and composite), as well as different pressure (200 bar and 250 bar, respectively). Due to a larger production volume in thermal gasification systems and possibly more remote locations, and thereby potentially longer transportation distances, bio-methane is assumed to be transported only in trucks equipped with composite vessels.

As an alternative to compression, methane can be liquefied by lowering its temperature to between -155 ℃ to -125 ℃ (Hanarp, 2016). The liquefied methane is thereafter transported by truck to the filling station.

Regarding gas grid distribution, it is assumed that the smallest biogas plant (producing 30 GWh biogas per year) injects the biogas into the low pressure gas grid (4 bar). All other AD and TG plants inject the gas into the high pressure gas grid with a pressure up to 80 bar (Swedegas, 2016). In Sweden, the same heating value is applied for all natural gas distributed via the gas grid. Since biogas has a lower heating value, this is adjusted by adding propane. The amount required depends on the methane content in the biogas. A methane content of approximately 97% requires an addition of 8% propane on volume basis (EI, 2010). The trend is, however, that the Swedish gas grid will be supplied with gas from multiple sources and not only from Denmark which is the current situation (EI, 2016). Based on discussions with the industrial partners in this project, and given the forward-looking perspective of this study, we therefore choose not to include propane in this study. This is also motivated by the fact that the only TG plant in operation in Sweden does not add propane given the quality of the gas produced (Paradis, 2016).

Finally, all gas grid distribution takes place in the existing gas grid and potential investments in new gas grids are not considered.

Regarding natural gas, it is assumed to be transported via the gas grid or as LNG, imported by boat and thereafter distributed by truck.

Post-treatment included in this study is thus compression and liquefaction.

Compression

Biogas transported by truck is compressed to 200 bar (in steel containers) and 250 bar (in composite containers). When injected into the gas grid, biogas is compressed to 80 bar, representing the transmission grid, and 4 bar, representing the low pressure grid, respectively.

The amount of electricity required for each case is calculated using the ideal gas law with an assumed efficiency of 50% (Gode et al. 2011). The calculated electricity consumption for each compressor application is presented in Table 3.5 and the estimated investment cost is presented in Table 3.6. The annual cost for operation and maintenance, excluding electricity, is set to 3% of the investment (Lantz et al., 2013). Emissions of methane are assumed to be marginal in the compression step and are not accounted for.

Table 3.5. Calculated electricity consumption for compression of methane (kWh electricity/kWh methane).

	End pressure			
Start pressure	4 bar	80 bar	200 bar	250 bar
1 bar[1]	0.008	0.026	0.032	0.033
4 bar[2]	–	0.018	0.023	0.025

[1] Gas pressure from amine scrubber and TG.

[2] Gas pressure from water scrubber.

Table 3.6. Estimated investment cost (SEK) for the compression of methane regarding the various biogas and bio-methane supply systems.

	Annual production (GWh)			
	30	100	520	1 600
Local gas grid	1 000 000[1]			
Transmission grid		12 000 000[2]	18 000 000[3]	35 000 000[3]
CBG	7 000 000[4]	12 000 000[4]	35 000 000[5]	75 000 000[5]

[1] Our estimation for gas grid injection into the local low pressure grid.

[2] Based on Paradis (2016).

[3] Our estimation based on data from Paradis (2016) and Karlsson (2016).

[4] Estimated based on data from Karlsson (2016) and assumed civil works of 1 MSEK.

[5] Our estimation.

Liquefaction

An alternative to compressing the methane before transport is to liquefy the gas by reducing its temperature to approximately -155 ℃ to -125 ℃ (Hanarp, 2016). For natural gas applications, the electricity consumption is in the range of 0.2–0.4 kWh/Nm^3 for plants using the MRC process and 0.6 kWh/Nm^3 for plants using the N_2 expander (or Brayton) process (He and Ju, 2013). For comparison, JRC (2014) note an electricity consumption of 0.36 kWh/Nm^3 for LNG liquefaction.

However, these values are relevant for large-scale, natural gas applications. For small-scale liquefaction, power consumption is estimated to be in the range of 0.5–1.0 kWh/Nm^3 (Bauer et al., 2013). Wärtsilä (2016) states that the power consumption for small-scale liquefaction with their MRC process is 0.7 kWh/kg LNG, which corresponds to approximately 0.5 kWh/Nm^3. For comparison, a biogas liquefaction plant with an installed capacity of 60 GWh has reported an actual power consumption of 1.1 kWh/Nm^3 (Lidköping biogas, 2015). However, this power consumption also covers part of the production of the compressed biogas.

In this study, power consumption is set to 0.6 at an annual production of 100 GWh, assuming the use of the MRC process, and 0.5 and 0.4 kWh/Nm^3 with an annual production of 520 GWh and 1 600 GWh, respectively, assuming the use of the Brayton process.

Regarding methane slip from the liquefaction process, the only LBG production plant in Sweden states no such slip could be quantified in the control programme. For comparison, Schori (2012) set the methane slip to 0.05% for liquefaction and evaporation of natural gas. Since this study applies BAT, it is assumed that there is no methane slip from the liquefaction process as such.

According to Bauer et al. (2013), the investment cost for an LBG plant with a capacity of 60 GWh was approximately 80 million SEK. This was, however, an early installation and based on indicative numbers from Rahmaputro (2016), and it is assumed that this level of investment could be applied for a LBG plant with a capacity of 100 GWh as well.

For larger plants, no data on investment costs for biogas liquefaction are available. The investment cost for liquefaction of natural gas is, however, estimated to be 175 million SEK and 240 million

SEK, respectively, (Rahmaputro, 2016) with an annual capacity of approximately 500 and 1600 GWh.

For all liquefaction plants, the costs for operation and maintenance are set to 3% of the investment cost.

Distribution

In this study, five different distribution alternatives are evaluated; namely, injection into the low and high pressure gas grid, transport of compressed methane by truck in steel and composite vessels, and transport of liquid methane by truck. As presented in Chapter 2, all five distribution alternatives are, however, not included in all of the different production pathways.

When transporting compressed biogas by truck, the volume of gas that can be transported by each truck depends on the weight of the swap bodies and the pressure of the gas. In this study, it is assumed that swap bodies with steel vessels have an efficient capacity of 2 000 Nm^3 each (at a pressure of 200 bar) and that one truck can carry up to three swap bodies. Alternatively, compressed gas can be transported in composite vessels with an efficient capacity of 4 300 Nm^3 each (at a pressure of 250 bar) and that one truck can carry three such swap bodies (Emebrant, 2016).

When LBG is transported by truck, the loading capacity is approximately 18 – 30 tonnes depending on truck configuration (Benjaminsson and Nilsson, 2009; Nilsson, 2016; Hirsch, 2016). In this study, calculations are based on a capacity of 30 tonnes or 40 000 Nm^3.

Fuel consumption for truck transport is set to 0.47 dm^3 of diesel/km for CBG transport and 0.40 dm^3/km for LBG transport (Energigas, 2016). In addition to this fuel consumption, loading and unloading of swap bodies and LBG will also require some energy input. However, data regarding this energy consumption have not been found in the literature and are thus not included in the calculations presented here. Loading capacity and energy consumption for each distribution alternative are summarised in Table 3.7.

Although there might be some slip of methane during distribution, these emissions are assumed to be marginal using BAT (Göthe, 2013) and are not accounted for in this study.

Table 3.7. Loading capacity and energy consumption for distribution by truck.

	CBG - S	CBG - C	LBG
Nm^3 per swap body	2 000	4 300	42 000
Nm^3 per truck	6 000	12 900	42 000
Fuel consumption (kWh/km)	4.6	4.6	3.9

The cost of distribution of CBG consists of a capital cost for swap bodies as well as an hourly rate for the truck and its driver. The investment cost for swap bodies with steel or composite vessels are set to 0.9 and 2.1 million SEK, respectively (Emebrant, 2016). The hourly rate for truck and driver is estimated to be 1 000 SEK/h.

Based on discussions with the industrial partners in this project, the swap body capacity is set to 3 times the daily production of methane.

For large scale distribution of LBG, the distribution cost is set to 1 500 SEK/h including truck and driver (Hirsch, 2016).

The transportation distance is set to, on average, 200 km (including return transport) for AD and to 600 km for TG. In both cases, the average speed is set 60 km/h.

Regarding gas grid distribution, Vestman et al. (2014) report an average distribution cost of 0.08 SEK/kWh and a median cost of 0.06 SEK/kWh. Although not stated, these numbers are realistic for gas injected into the low pressure grid. According to Paradis (2016) there is no cost for distribution in the high pressure grid except the cost for OPEX and electricity. Thus, it is assumed that the grid distribution cost via the high pressure grid to customers at the low pressure grid is 0.08 SEK/kWh.

3.4 BIOGAS PRODUCTION AND UPGRADING

3.4.1 Biogas production

As mentioned in Chapter 2, biogas production as such is not the primary focus of this study. It is, however, included to give a complete picture of the biogas system.

The Swedish biogas production by anaerobic digestion is based on sewage sludge, food waste, manure and various types of industrial waste (Energimyndigheten, 2015a). The choice of feedstock has a high impact on energy balance, GHG emissions and production cost (Börjesson et al., 2010; Lantz, 2013).

Given this dependency on feedstock, data on energy balance and GHG emissions applied in this study are based on calculations presented in Appendix 2. Thereby, it is also possible to adapt the result depending on the availability of waste heat from the upgrading process which will differ depending on upgrading technology. It is also possible to apply updated GHG emission factors based on the IPCC (2013) as compared to the ones suggested by the IPCC (2007) which are used in most of the calculations found in the literature. Finally, it is also possible to compare calculations according to the ISO standard with calculations performed according to the RED.

Energy input and GHG emissions

In this study, the direct energy input for biogas production is calculated to be 0.12 – 0.13 MJ/MJ, using system expansion according to the ISO-standard and 0.16–0.17 MJ/MJ with system boundaries according to the RED. For more details, see Table 3.8. Background data and assumptions are presented in Appendix B.

GHG emissions not related to production and utilization of energy carriers are calculated to -9.5 g CO_2-eq./MJ using system expansion according to the ISO-standard and 0.1 g CO_2-eq./MJ with system boundaries according to the RED. For more details, see Table 3.9. Background data and assumptions are presented in Appendix B.

Table 3.8. Direct energy input for biogas production (MJ/MJ biogas).

	30 GWh		100 GWh	
	ISO	RED	ISO	RED
Food waste				
Transport (diesel)	0.036	0.036	0.036	0.015
Pre-treatment (electricity)	0.015	0.015	0.015	0.036
Transport				
Feedstock (diesel)	0.011	0.011	0.015	0.015
Digestate (diesel)	0.010		0.014	
Process Energy				
Electricity	0.023	0.23	0.023	0.023
Heat	0.080	0.080	0.08	0.08
Spreading of digestate (diesel)	0.007		0.007	
System expansion				
Avoided spreading of manure (diesel)	-0.003		-0.003	
Avoided production of mineral fertilizers	-0.061		-0.061	
Total Energy input	0.12	0.16	0.13	0.17

Table 3.9. GHG emissions not originating from energy carriers (g/MJ biogas)

	ISO	RED
Biogas production		
CH_4 slip	0.06	0.06
CH_4 from flare	0.02	0.02
Handling of digestate		
CH_4 from digestate storage	0.11	
N_2O from spreading of digestate	0.05	
System expansion		
CH_4 from storage of manure	-0.14	
N_2O from storage and spreading of manure	-0.06	
CO_2 from production of mineral fertilizer	-4.36	
CH_4 from production of mineral fertilizer	-0.01	
N_2O from production of mineral fertilizer	-0.01	
N_2O from spreading of mineral fertilizer	-0.02	
Changes in soil carbon	-5.17	
Total GHG emissions	-9.49	0.08

Economy

The production cost for biogas can vary considerably between different biogas plants depending on feedstock and scale etc. The production cost consists of the cost of capital, process energy and transport of feedstock and digestate. It is also affected by potential gate fees or cost for feedstock, as well as potential income from the digestate. Finally, the overall production cost is highly affected by the amount of biogas produced per tonne of feedstock and per m^3 of reactor.

According to an investigation presented by Vestman et al. (2014), the average and median production cost for seven biogas plants was 0.86 and 0.54 SEK/kWh, respectively. However, the authors stress the difficulty of gathering and presenting comparable data since different biogas plants are operated in different ways and in different contexts. For comparison, E.ON states that the production cost of biogas from MSOW and industrial waste varies from 0.4–0.8 SEK/kWh (Energimyndigheten, 2013).

In this study, the production cost for biogas is set to 0.6 SEK/kWh (170 SEK/GJ) in the base case, and a range between 0.4 and 0.8 SEK/kWh is evaluated in the sensitivity analysis.

3.4.2 Biogas – upgrading

Biogas produced from the feedstock mixture presented in Appendix B is estimated to have a methane content of 63%. In order to fulfill the Swedish standard for gas used as vehicle fuel the methane content must be increased to 95–99% and various contaminants must be removed. In Sweden, this upgrading is performed with various technologies, such as water scrubbers, amine scrubbers and pressure swing adsorption (PSA). In this study, calculations are based on the water scrubber which is the most common solution and the amine scrubber, which has the lowest methane slip (Bauer et al., 2013). A general overview of each upgrading system is presented in Figure 3.1 andFigure 3.2. A more detailed description of these and other upgrading technologies are presented in Bauer et al. (2013) and Hoyer et al. (2016), among others. Process energy requirements and methane losses are presented in Table 3.10.

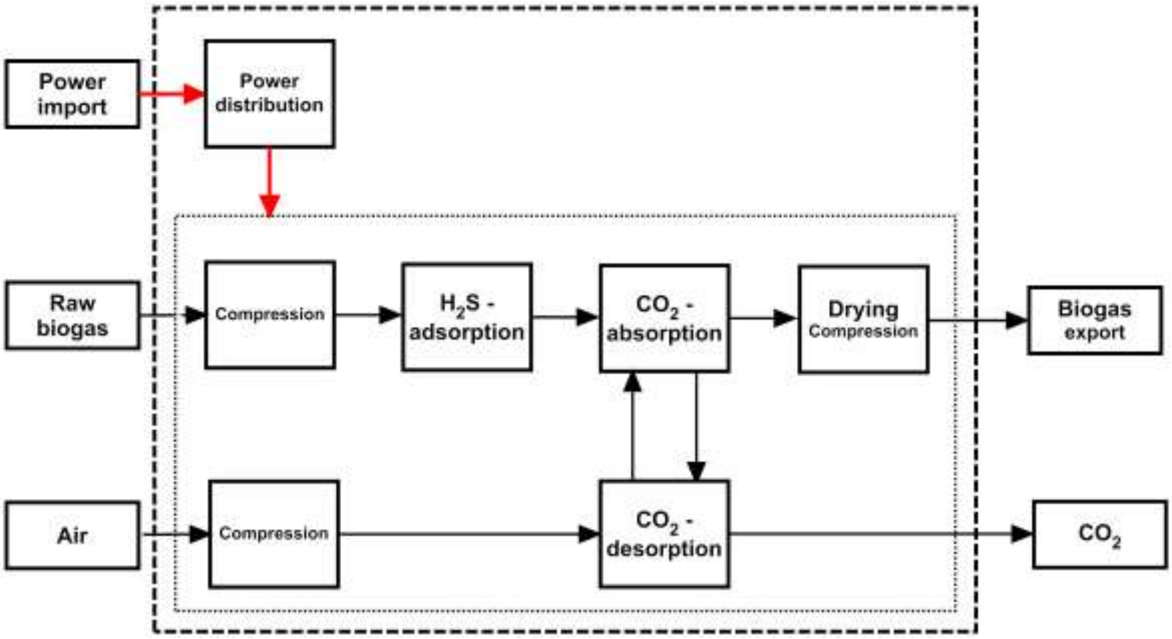

Figure 3.1. Biogas upgrading with water scrubbing.

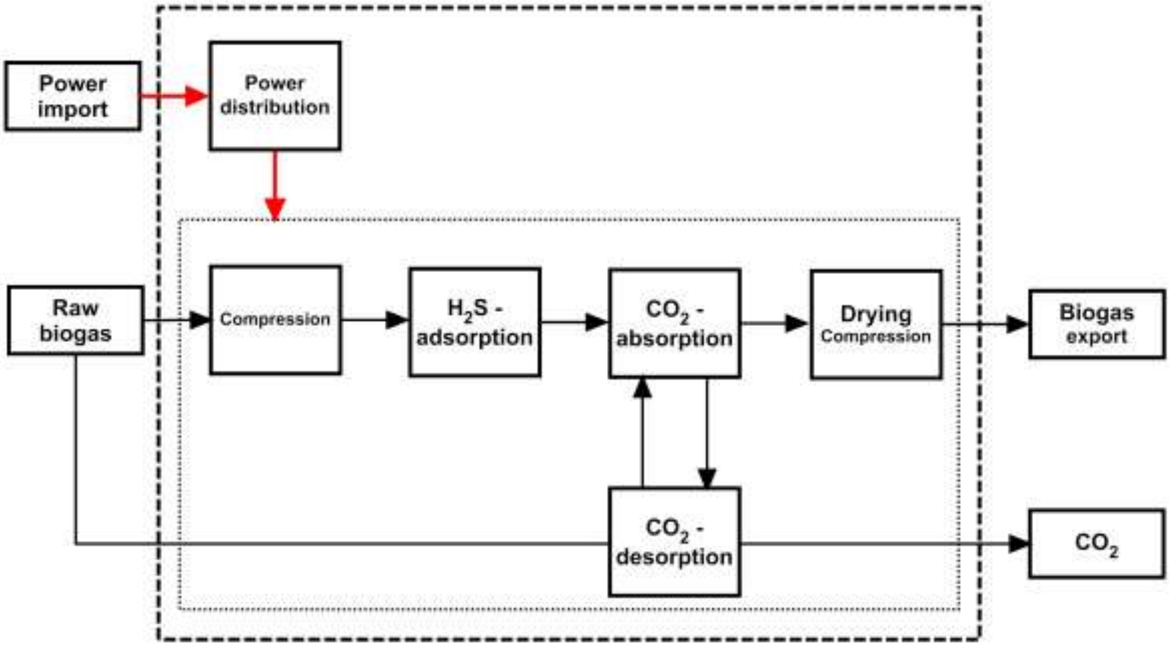

Figure 3.2. Biogas upgrading with amine scrubbing.

Table 3.10. Energy input and methane losses in the biogas upgrading process (Bauer et al., 2013; Hoyer et al., 2016).

	Water scrubber	Amine scrubber
Electricity (MJ/Nm^3)	0.9	0.4
Heat (MJ/Nm^3)	–	1.0
Methane losses (%)	0.5	0.1

The investment costs for the 30 GWh and 100 GWh amine scrubbers are estimated to be 17 and 21 million SEK, respectively, including a wood chips-fired boiler (Hulteberg, 2015). Thus, the investment cost might be somewhat overestimated since the heat is also used for the biogas process. The corresponding investment cost for the water scrubber is estimated to be 12 and 20 million SEK, respectively (Jeppsson, 2015). For both upgrading applications, the cost for operation and maintenance is set to 3% of the investment (Lantz, 2013).

3.4.3 Bio-methane – production

Gasification of biomass generates a gaseous product, mainly consisting of hydrogen (H_2), carbon monoxide (CO), carbon dioxide (CO_2), water vapour and also some methane (CH_4). Additionally, the gas contains various amounts of contaminants such as dust, tars, alkali, ammonia and sulphur that must be removed prior to the methanation. Different types of gasifiers provide different gas qualities and gas compositions and the choice of gasifier type has a strong influence on the design of the subsequent gas cleaning and conditioning as well as on the methane synthesis.

In this study, two types of biomass gasifier with different production capacities have been considered. The smaller unit, with an annual production of 520 GWh, is an indirect fluidised bed gasifier based on an up-scaled version of the GoBiGas plant in Gothenburg, see Figure 3.3. The larger gasifier is an oxygen-blown, circulating fluidised bed gasifier (CFB) with an annual production of 1 600 GWh, see Figure 3.4. Further information on the technical characteristics of the two gasifiers can be found in Fredriksson Möller et al. (2013) and in Larsson (2014).

Both gasifiers use dried wood-chips as fuel. The gas produced passes through various steps of gas cleaning and conditioning systems before the purified gas passes through the methanation process in which CO and H_2 are converted to CH_4 and CO_2. The bio-methane is then further upgraded to a quality suitable for transportation fuel.

Energy input and GHG emissions

The calculated energy balances for the two bio-methane production systems are summarised in Table 3.11. Data for the smaller unit, which is an indirect fluidised bed gasifier based on an up-scaled version of the GoBiGas plant in Gothenburg, is based on Larsson (2014). The corresponding data for the larger gasifier, which is an oxygen-blown, circulating fluidised bed gasifier (CFB), is based on Fredriksson Möller et al. (2013).

Information on greenhouse gas emissions and primary energy factors associated with the wood chips, electricity and RME used in the gasification process are presented in Appendix A. As mentioned in Chapter 2, the primary energy factor for wood chips does not include the energy embedded in the biomass when it is used as raw material, although it is included when used for process energy production. GHG emissions from wood chip production include losses of soil carbon when calculated according to the ISO standard, which is not the case when calculations are performed according to the RED, see also Chapter 2 and Appendix A.

In the case of excess heat, it is assumed that 62.5% (corresponding to 5000 h) could replace heat from wood chips. Regarding electricity, emissions and primary energy factors are based on the Swedish and Nordic electricity mix in the ISO and RED case, respectively. Methane emissions from the gasification plant are assumed to be insignificant.

Table 3.11. Energy balances of the two bio-methane production systems included in the study.

	Biomass input	Heat balance	Net electricity balance (own production-demand)	Others	Bio-methane production
Indirect gasification	100 MW	0 MW		2.5 MW RME used as tar solvent	65 MW
Circulating fluidised bed gasification	320 MW (of which 10-15 MW to power production)	10-30 MW	-8-10 MW		200 MW

Economy

The production costs are estimated to be, on average, 1 200 SEK/MWh methane with indirect gasification (520 GWh annually) and, on average, 758 SEK/MWh with CFB gasification (1 600 GWh annually), see Table 3.12.

The production cost is based on data from Fredriksson Möller et al. (2013) and Larsson (2014), but has to some extent been updated and revised where necessary. Due to the uncertainties involved in these cost estimations, the production cost is also calculated by changing the cost of biomass and capital by ± 30% in the sensitivity analysis presented in Chapter 4.

Table 3.12. Estimated production cost (SEK/MWh bio-methane).

	TG 520	TG 1 600
CAPEX[1]	542	220
Biomass[2]	308	320
Power[3]		60
RME, catalyst, personal and maintenance	131	108
Revenues (excess power and heat)[4]		-40
Total production cost	981	708

[1] Based on an investment of 3 600 and 4 500 million SEK respectively, a depreciation time of 25 years and 6% interest.

[2] Assuming a wood chip cost of 200 SEK/MWh, see Appendix A, and an efficiency of 65% and 62.5% respectively, see Table 3.11.

[3] Assuming 500 SEK/MWh, see Appendix A.

[4] Surplus electricity (16 MW) is sold for 400 SEK/MWh including electricity certificates, and surplus heat (10 MW) is sold for 250 SEK/MWh. Heat is, however, sold only 5 000 h annually of 8 000 h of production.

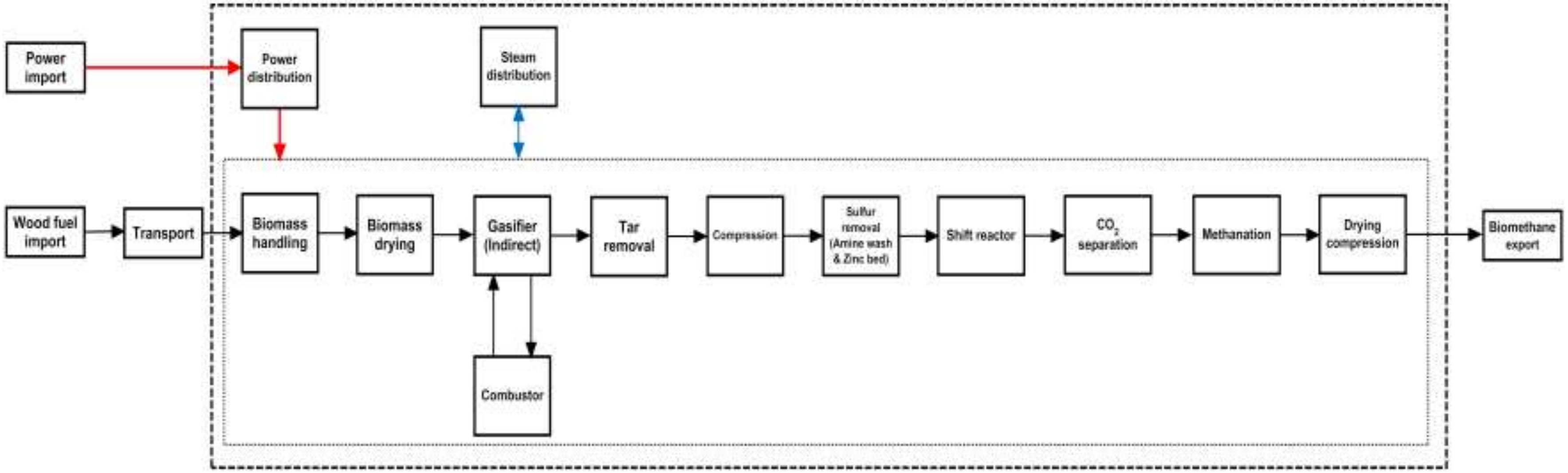

Figure 3.3. System description of the bio-methane production plant using indirect gasification.

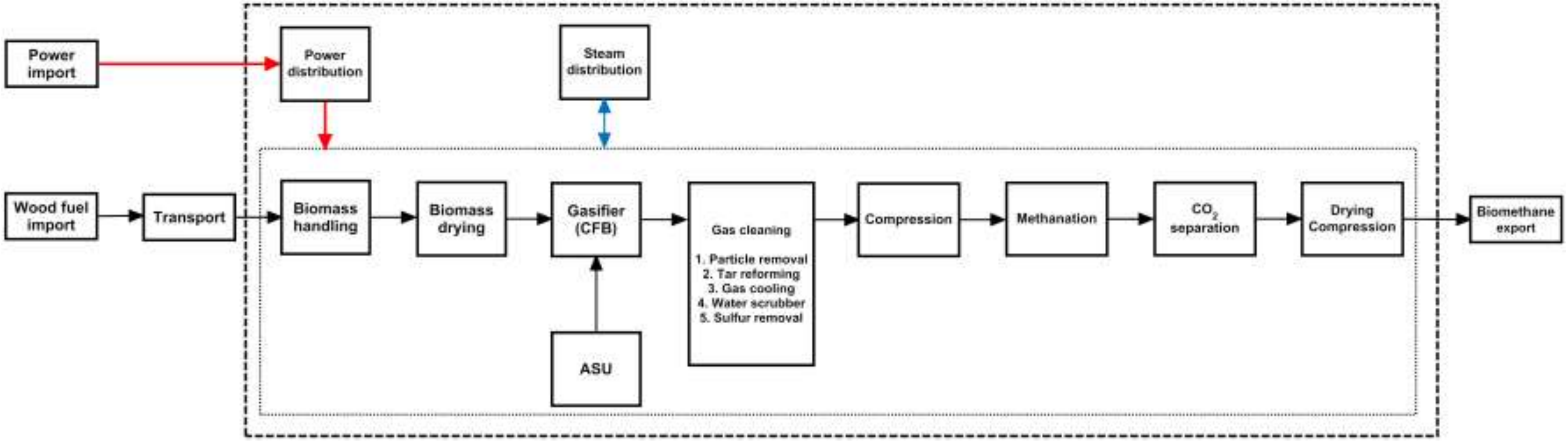

Figure 3.4. System description of the bio-methane production plant using a CFB-gasifier

3.5 FOSSIL REFERENCE

3.5.1 Natural gas

The natural gas used as vehicle fuel is assumed to be supplied by the natural gas grid in the case of compressed natural gas (CNG). Regarding liquefied natural gas (LNG), the vehicle fuel is assumed to be transported by boat. Data regarding GHG and energy efficiency performance of the NG-based vehicle production systems are based on JRC (2013; 2014a,b).

The CNG pathway includes production and conditioning at source, transport to markets by pipelines, distribution by the national NG grid and on-site compression. The LNG pathway includes production, conditioning and liquefaction at source, transport to markets by dedicated ships and distribution by trucks on land. The inventory data for natural gas supply systems are described in Table 3.12. The data are based on JRC (2014b), representing average European conditions.

Also other studies exist in which the WTW GHG performance of NG vehicle fuels is studied. For example Alamia (2015) presents somewhat lower WTT emissions for NG-based vehicle fuels than JRC (2014b). Here, the GHG emission from production and distribution of CNG and LNG is estimated to be approximately 7.2 g and 11.9 g CO_2-eq per MJ, respectively (Alamia, 2015). The WTT report by JRC (2014b) also presents uncertainty intervals regarding the GHG and energy balance performance of CNG and LNG supply systems. These uncertainty intervals, shown in Table 3.2, are utilised in a sensitivity analysis later in this report.

Table 3.13. WTT GHG emissions and energy balance in the production and distribution of CNG and LNG. Uncertainty intervals are shown within parentheses.[1]

Fuel	GHG emissions	Energy balance
	(g CO_2-eq./MJ $_{\text{final fuel}}$)	(MJ/MJ $_{\text{final fuel}}$)
CNG	13.0 *(11-15)*	0.17 *(0.14-0.19)*
LNG	19.0 *(17-21)*	0.26 *(0.21-0.25)*

[1] Based on data from JRC (2014b) referring to the European market today.

As presented in Appendix A, the average market price of CNG and LNG as vehicle fuel, excluding VAT but including other relevant taxes, is estimated to be 1000 and 900 SEK/MWh, respectively which corresponds to 278 and 250 SEK/GJ.

3.5.2 Crude oil-based fuels

The reference crude oil-based fuels in light- and heavy-duty vehicles are gasoline and diesel, respectively. The GHG and energy efficiency performance in the production and distribution of the reference fuels are described in Table 3.14. Data are based on JRC (2014b) and refer to average, conventional gasoline and diesel sold on the European market today, produced from the current mix of crude oil supply. Thus, the calculations do not include any “non-conventional” sources of crude oil. Also, uncertainty intervals regarding the WTT GHG emissions and energy balances for gasoline and diesel are shown in Table 3.14, based on JRC (2014b).

Table 3.14. TTW GHG emissions and energy balance in the production and distribution of crude oil-based gasoline and diesel. Uncertainty intervals are shown within parentheses.[1]

Fuel	GHG emissions	Energy balance
	(g CO_2-eq./MJ $_{\text{final fuel}}$)	(MJ/MJ $_{\text{final fuel}}$)
Gasoline	14.0 *(12-16)*	0.18 *(0.16-0.20)*
Diesel	15.5 *(14-17)*	0.21 *(0.18-0.24)*

[1] Based on data from JRC (2014b) referring to conventional crude oil-based fuels on the European market today.

The current market price of diesel and gasoline, excluding VAT but including other relevant taxes, is set to be, on average, 1100 SEK/MWh and 1200 SEK/MWh, respectively (see Appendix A). This is equivalent to 305 SEK/GJ and 333 SEK/GJ, respectively.

4 RESULTS AND SENSITIVITY ANALYSIS

The results regarding costs, GHG and energy efficiency performance of the different methane-fuelled vehicle systems included in the assessment are shown in the form of diagrams below. The results are shown both as well-to-tank (WTT) performance, per MJ fuel, and as well-to-wheel (WTW) performance, per km transportation service. Furthermore, the GHG performance is shown both according to the ISO calculation methodology and the RED calculation methodology.

4.1 COST PERFORMANCE

The costs of the different methane vehicle fuel systems are presented as WTT costs, expressed as SEK per GJ fuel, and as WTW costs, expressed as SEK per km. The WTW costs include the additional increase in vehicle costs of methane-fuelled vehicles, compared with conventional diesel- and gasoline-fuelled vehicles, but not the cost of the vehicle itself.

4.1.1 Well-to-tank

The production cost of methane from AD, including distribution and filling stations, is calculated to be 246–272 SEK/GJ of methane with an average of 255 SEK/GJ at an interest rate of 6%. The corresponding cost of methane from TG is calculated to be 234–354 SEK/GJ with an average of 296. The calculated production cost depends on production method (AD or TG), scale, upgrading technology and distribution technique, see also Figure 4.1.

Comparing the different pathways investigated here, it is clear that the difference between various post-treatment and distribution alternatives is minor, although there is a positive efficiency of scale, especially for liquefaction. Comparing the complete WTT system, all alternatives have a similar cost except the small gasification plant for which the cost is higher.

Instead, it should be noted that the cost of AD and TG represents approximately 60 – 80% of the total WTT cost. This cost is, however, relatively uncertain. The cost of AD has not been calculated in detail in this study but is based on literature data. Specific biogas plants with other kinds of feedstock mixtures and local conditions could lead to different production costs and thereby significantly affect the overall result. Regarding TG, the production cost has been calculated within this study but includes a relatively high uncertainty, for instance, concerning the investment cost as presented in Chapter 3.

For comparison, the WTT cost is also calculated at a 10% interest rate instead of 6%, see Figure 4.2. A higher interest rate will obviously increase the production cost, especially for gasification which has a relatively high cost of capital. It does, however, have a minor impact when comparing different pathways, the large TG plant has no longer the lowest production cost on this assumption. It should, however, be noted that the changed interest rate has not been applied for the AD cost since this is based on existing literature values.

For thermal gasification, Figure 4.3 shows the impact on the WTT cost when the investment and the cost of biomass is changed by ±30%. For the large scale TG plant it is clear that the cost of biomass has almost the same impact as the cost of capital. For the smaller plant, the cost of capital dominates, making the production cost more sensitive to the interest rate, depreciation time and potential investment subsidies. For the larger plants this is important as well but a cost efficient feedstock supply is of almost the same importance.

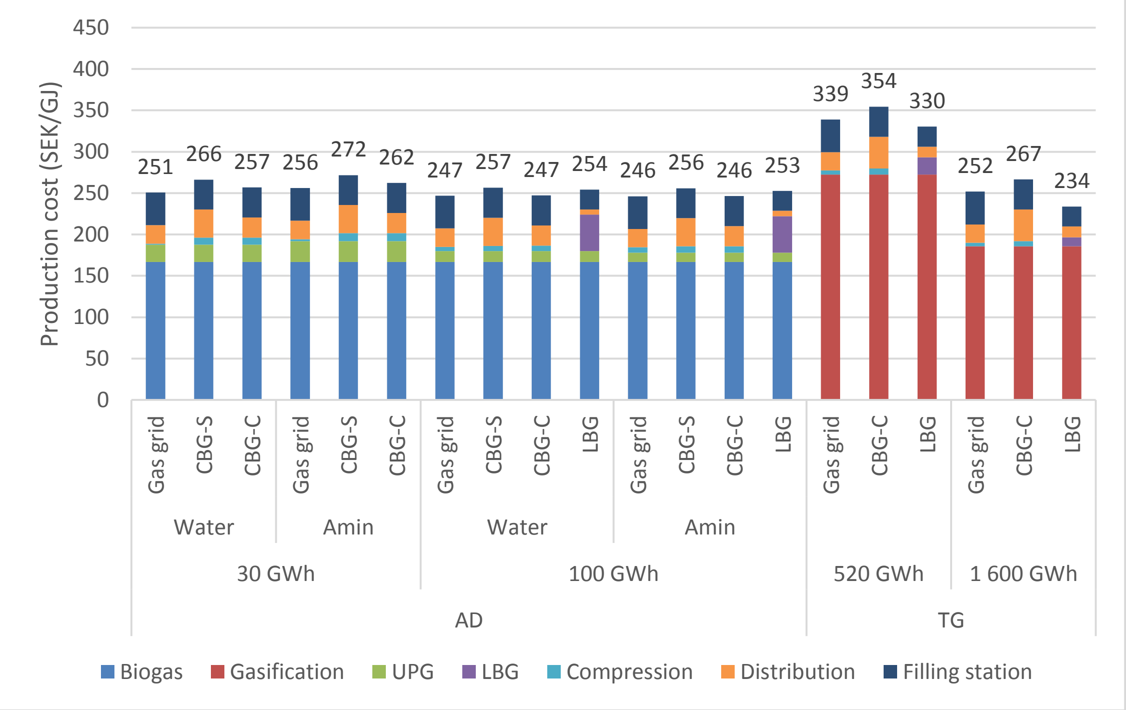

Figure 4.1. WTT cost of methane, including production, distribution and filling stations when CAPEX is calculated with a 6% interest rate.

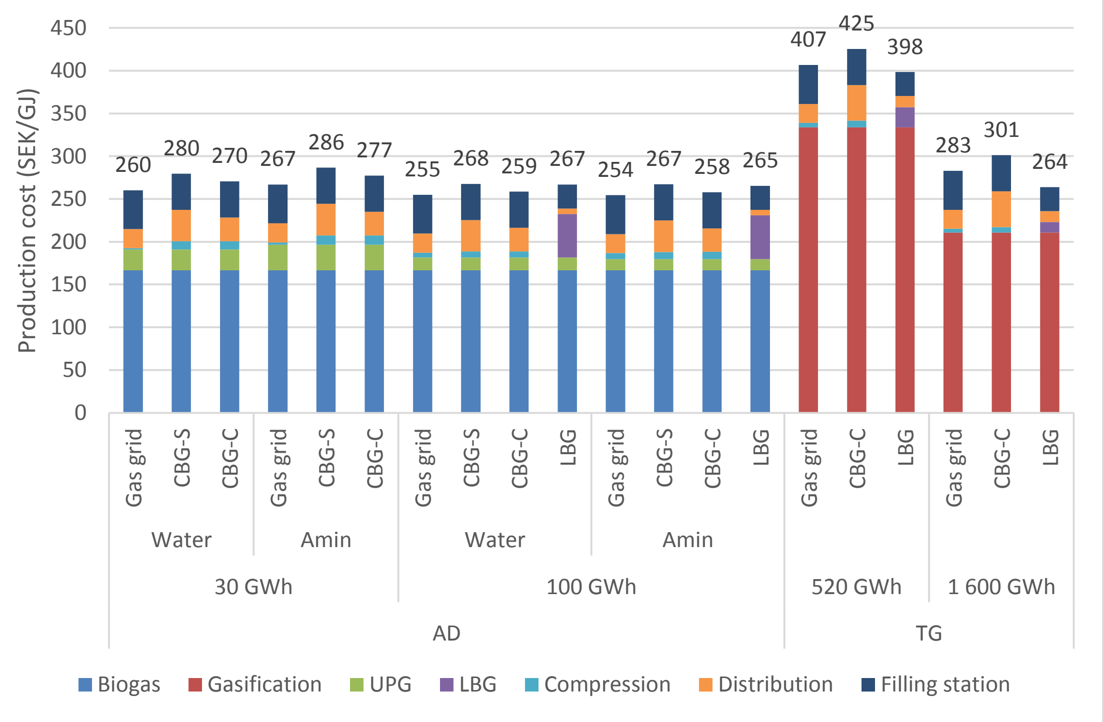

Figure 4.2. WTT cost of methane, including production, distribution and filling stations when CAPEX is calculated with a 10% interest rate.

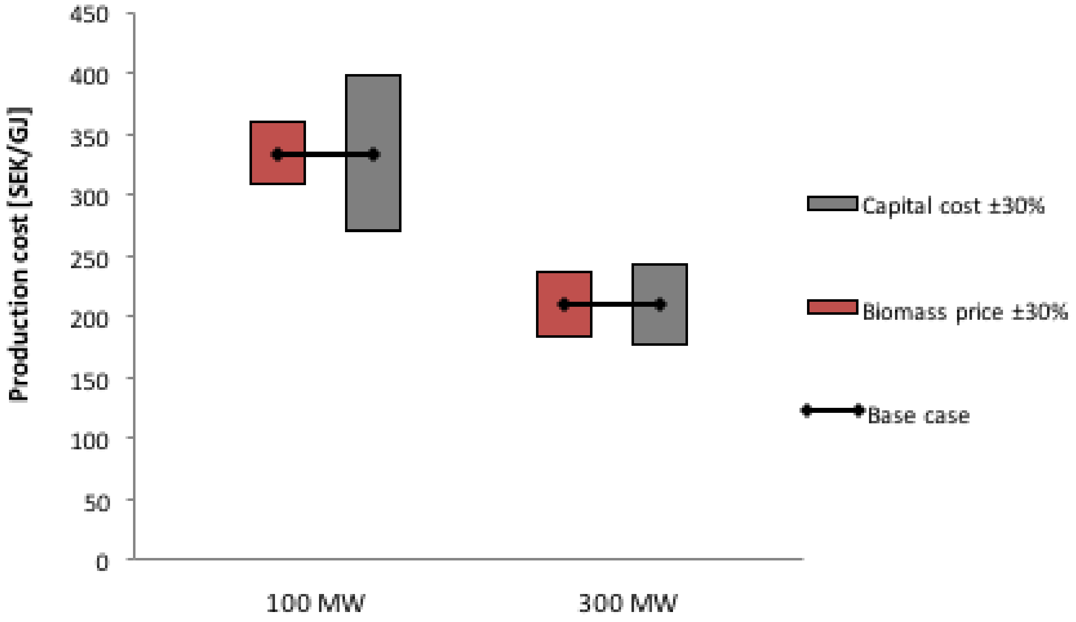

Figure 4.3. WTT cost of methane from TG when the biomass cost and CAPEX (with 10% interest) are changed by ± 30%.

4.1.2 Well-to-wheel

The total WTW cost performance for light-duty vehicles is shown in Figure 4.4, and for heavy-duty vehicles in Figure 4.5. The fuel production costs for CBG and LBG are estimated to be, on average, 275 SEK/GJ (see previous section). The corresponding fuel costs for CNG, LNG, gasoline and diesel are based on current market prices, excluding VAT.

The WTW cost of a CBG- and CNG-fuelled car is estimated to be approximately 18% higher than the WTW cost of a corresponding gasoline-fuelled car. The additional TTW cost for methane-fuelled cars represents roughly 25% of the total WTW cost. The WTW cost of a CBG- and CNG-fuelled heavy-duty vehicle is estimated to be approximately 16-17% higher than for a diesel-fuelled truck. The corresponding WTW cost of a LBG-fuelled DF truck is estimated to be similar as for a diesel-fuelled truck, whereas the WTW cost for a LNG-fuelled DF truck is estimated to be approximately 8% lower. The additional vehicle costs (TTW costs) of methane-fuelled trucks, compared with diesel-fuelled trucks, is estimated to represent around 10% of the total WTW costs.

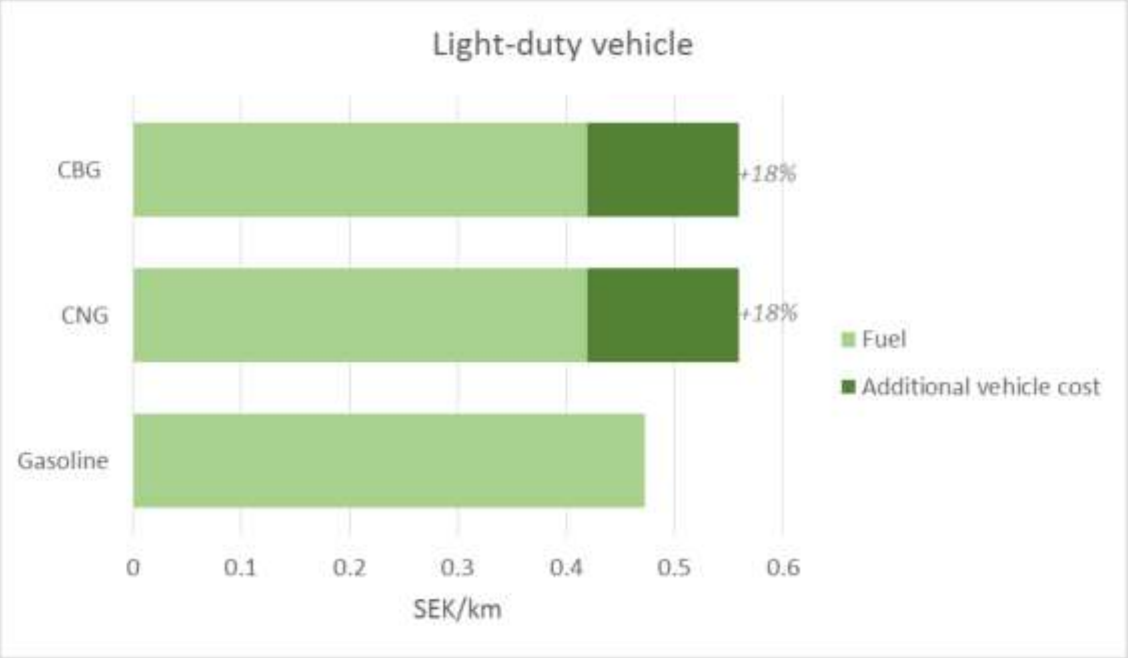

Figure 4.4. The WTW cost performance (SEK/km) for light-duty vehicles fuelled with compressed methane and gasoline, respectively. The costs of CNG and gasoline are based on current market prices, excluding VAT but including other relevant taxes.

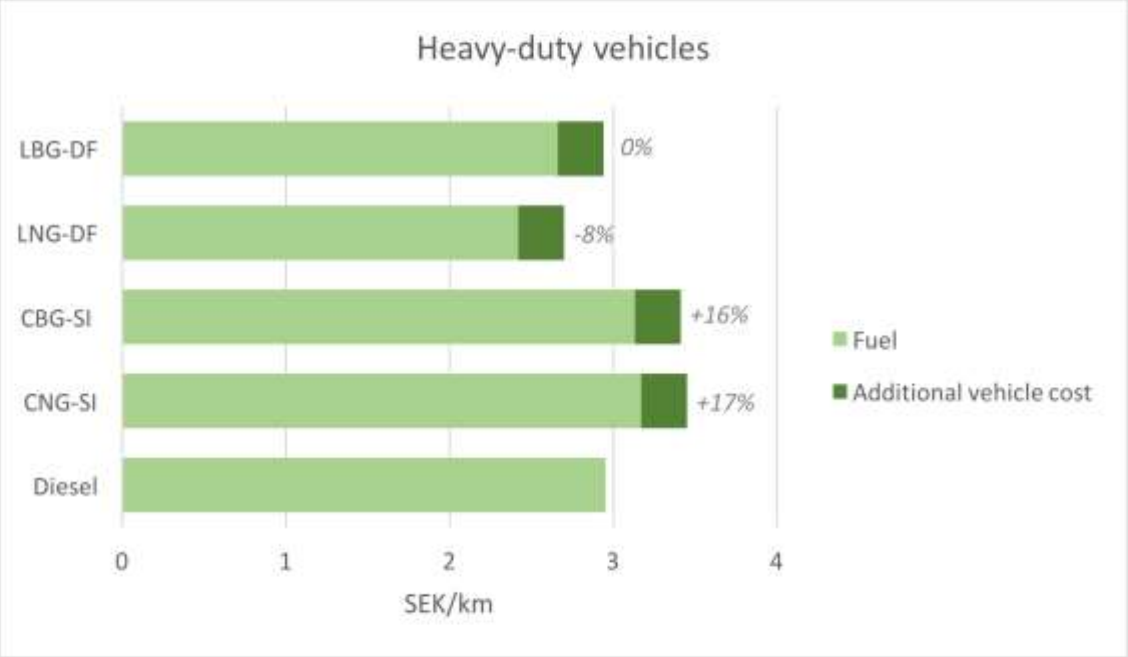

Figure 4.5. The WTW cost performance (SEK/km) for heavy-duty vehicles fuelled with compressed and liquefied methane, and diesel, respectively. The costs of CNG, LNG and diesel are based on current market prices, excluding VAT but including other relevant taxes.

A conclusion from the WTW cost performance presented above is that the costs for CBG- and CNG-fuelled light- and heavy-duty vehicles are 15-20% higher than for gasoline- and diesel-fuelled vehicles, wheras the costs for LBG- and LNG-fuelled vehicles are similar or somewhat lower. The production costs represent the major part of the WTT costs, often between 60–80%, whereas the costs of upgrading, compression, liquefaction, distribution and filling stations represent some 20–40% .

In Figure 4.6, the WTT costs of CBG and LBG are varied to cover the cost interval presented in the previous section of the calculated WTT costs for the different biogas and bio-methane systems, including changes in the CAPEX. The low and high WTT costs are equivalent to 200 and 400 SEK/GJ, respectively, compared with the base case costs equivalent to 275 SEK/GJ. As can be seen in Figure 4.6, an increased WTT cost equivalent to 400 SEK/GJ for LBG and CBG will result in WTW costs for LBG- and CBG-fuelled heavy-duty vehicles that are approximately 30% and 55% higher, respectively, than for a diesel-fuelled truck based on the current market price of diesel, excluding VAT. On the other hand, if the WTT cost is reduced to 200 SEK/GJ, then the WTW costs for CBG- and LBG-fuelled trucks will be approximately 13–25% lower.

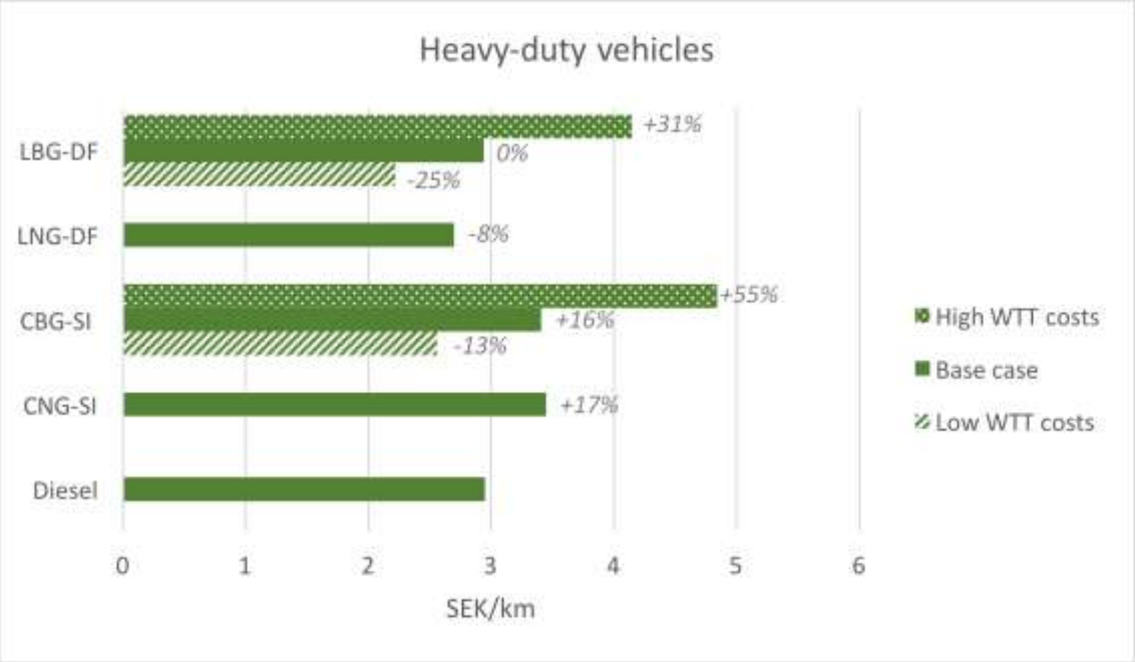

Figure 4.6. The WTW cost performance (SEK/km) for heavy-duty vehicles fuelled with compressed and liquefied methane, and diesel, respectively. The costs of CNG, LNG and diesel are based on current market prices, excluding VAT but including other relevant taxes. The High WTT cost of LBG and CBG is equivalent to 400 SEK/GJ, whereas the Low WTT cost is equivalent to 200 SEK/GJ (the WTT cost in the Base case is equivalent to 275 SEK/GJ).

Apart from uncertainties in the WTT costs of CBG and LBG production systems, also the TTW costs of the vehicles are uncertain. As described in the previous Section 3.1, the additional cost of a methane-fuelled, heavy-duty vehicle, compared with a diesel-fuelled vehicle, may vary from approximately 0.14–0.42 SEK/km (100 000–300 000 SEK per truck). As can be seen in Figure 4.7, this variation in additional TTW costs of methane-fuelled trucks results in changes in WTW costs equivalent to some +/- 5%.

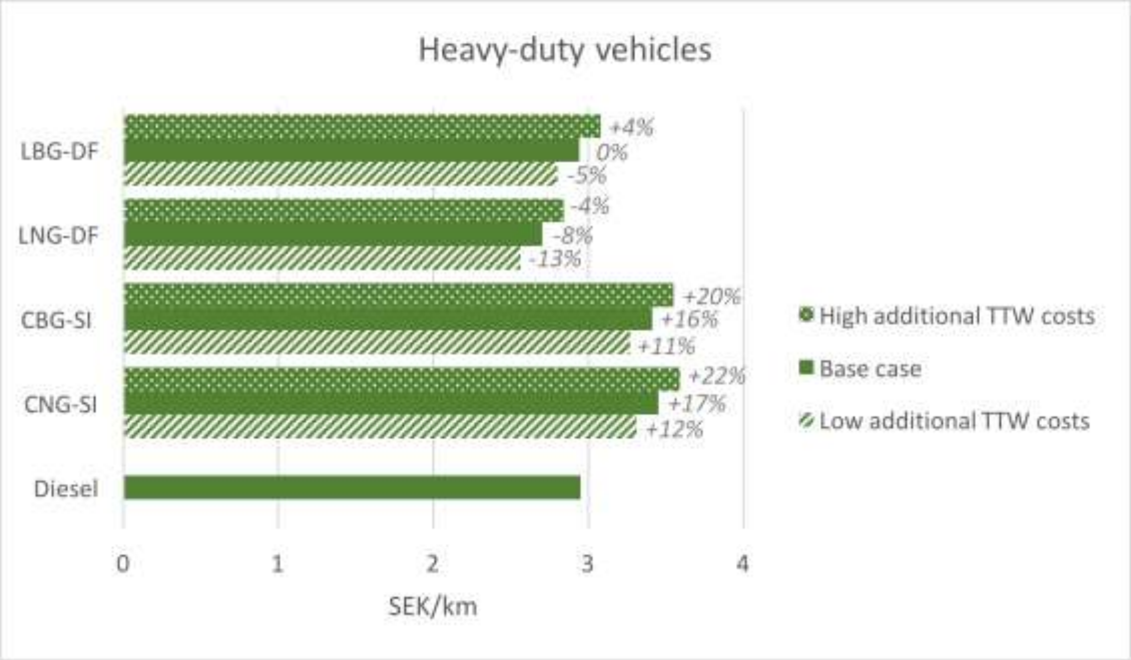

Figure 4.7. The WTW cost performance (SEK/km) for heavy-duty vehicles fuelled with compressed and liquefied methane, and diesel, respectively. The costs of CNG, LNG and diesel are based on current market prices, excluding VAT but including other relevant taxes. The High additional TTW cost of CNG, CBG, LNG and LBG vehicles, compared with diesel vehicles, is equivalent to 0.42 SEK/km, whereas the Low TTW cost is equivalent to 0.14 SEK/km (the additional TTW cost in the Base case is equivalent to 0.28 SEK/km).

4.2 GHG PERFORMANCE

The GHG performance for the various methane systems are initially presented from a WTT perspective, including more detailed information about the contribution from the various steps in the supply system. Thereafter, the results are presented from a complete WTW perspective, describing the size of the contribution from the WTT and from the TTW perspectives.

4.2.1 Well-to-tank

Based on the inventory data presented in Chapter 3, the GHG emissions from biogas production from AD are calculated to be -5 to 8 g CO_2-equivalents/MJ of biogas, according to the ISO and the RED calculation methodology, respectively, see Table 4.1. For upgrading and distribution of biogas, GHG emissions are calculated to be 1.5 to 5.5 g CO_2-eq/MJ of biogas according to the ISO methodology, see Figure 4.8. The corresponding GHG emissions, calculated according to the RED methodology, are 2.7 to 6.1 g CO_2-eq./MJ, see Figure 4.9.

In total, the WTT GHG emissions from biogas systems amount to -3.5 to 1.2 g CO_2-eq/MJ, with an average of -1.1 g, when the ISO calculation methodology is applied. The corresponding WTT GHG emissions amount to 10.2 to 14.0 g CO_2-eq/MJ, with an average of 12.3 g, when the RED calculation methodology is applied.

Table 4.1. GHG emissions from biogas production calculated according to ISO and RED.

	30 GWh		100 GWh	
	ISO	RED	ISO	RED
Food waste	2.7	3.0	2.7	3.0
Transport of feedstock	1.7	0.9	2.4	1.3
Process energy				
- Heat	1.1	1.1	1.1	1.1
- Electricity	0.3	0.8	0.3	0.8
Methane slip				
- Biogas	2.2	1.5	2.1	1.5
- Flare	0.6	0.4	0.6	0.4
Digestate				
- Storage	4.1		4.1	
- Spreading	14.2		14.2	
Soil Carbon	-5.2		-5.2	
Manure	-12.2		-12.2	
Mineral fertiliser	-14.4		-14.4	
Total emissions	-5.1	7.6	-4.3	8.0

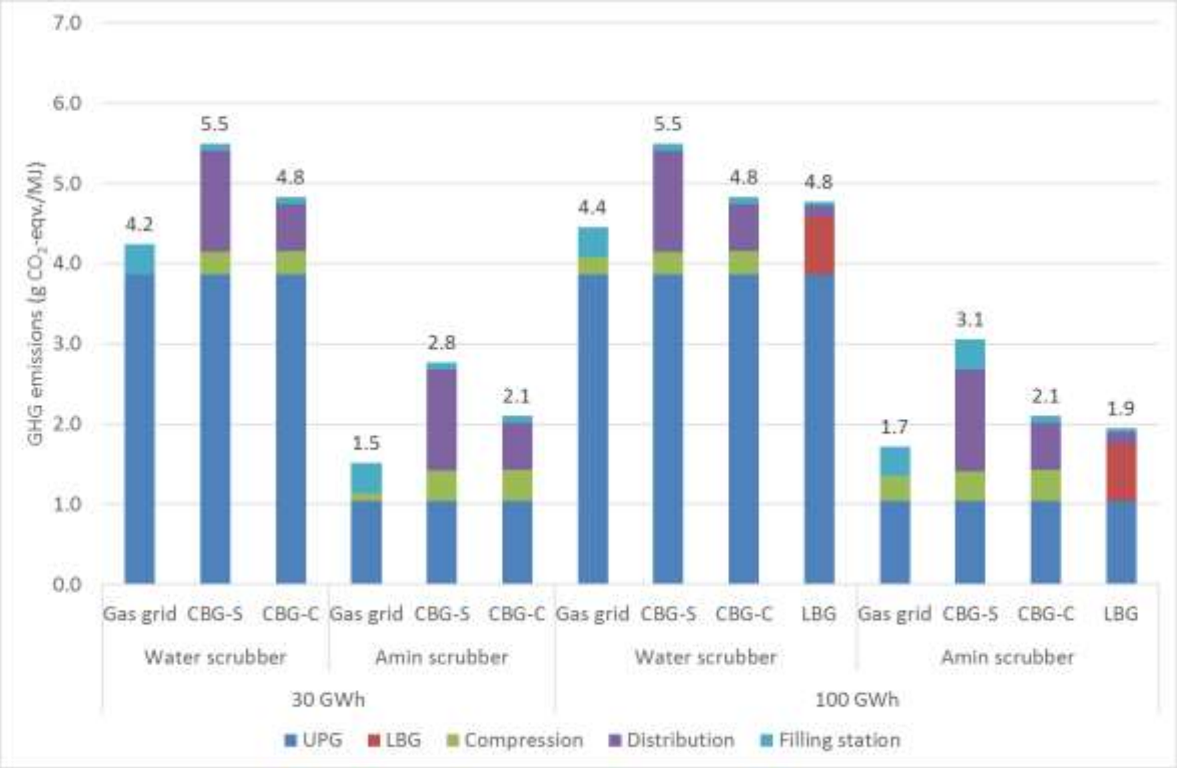

Figure 4.8. Emissions of GHG from upgrading and distribution of biogas according to ISO calculation methodology.

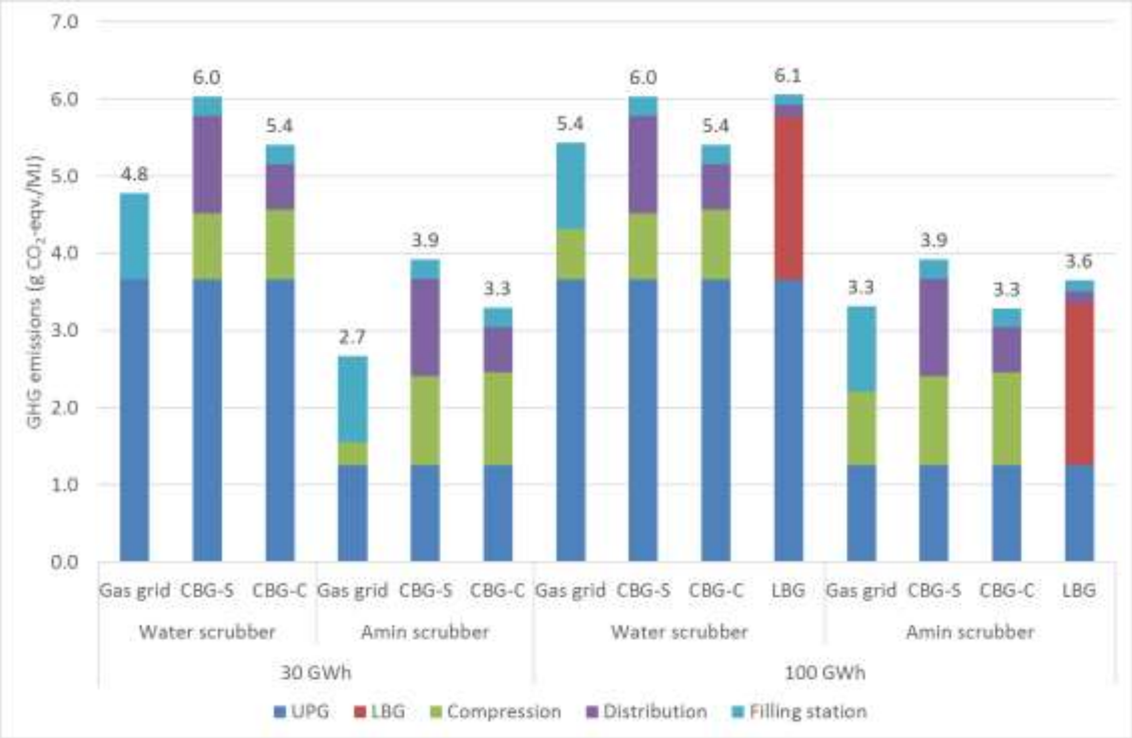

Figure 4.9. Emissions of GHG from upgrading and distribution of biogas according to the RED calculation methodology.

For bio-methane produced by gasification, the WTT GHG emissions are calculated to be 17.6 to 19.7 g CO_2-eq/MJ, with an average of 18.6, according to the ISO calculation methodology. According to the RED calculation methodology, the corresponding WTT GHG emissions are 5.4 to 7.4 g CO_2-eq./MJ, with an average of 6.3, see Figure 4.10.

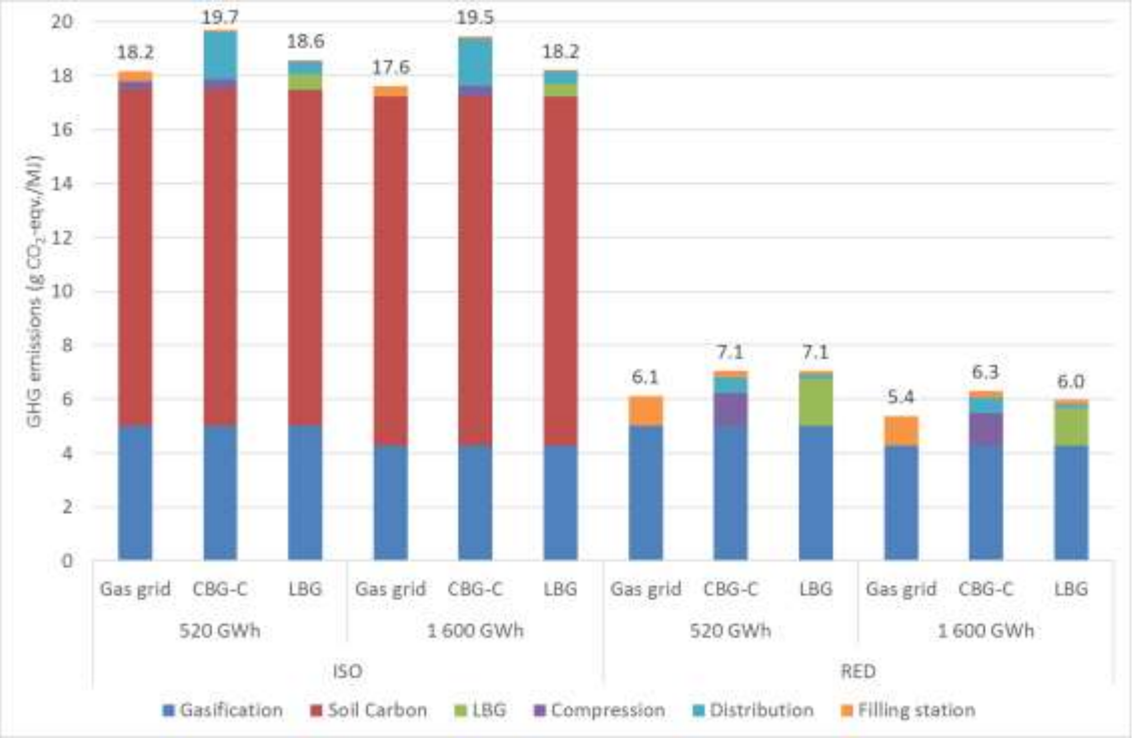

Figure 4.10. The WTT GHG performance regarding bio-methane supply systems, according to the ISO and RED calculation methodology, respectively.

A significant number of studies exist which point out the critical parameters in the biogas and bio-methane supply systems regarding their WTT GHG performance (see e.g. Börjesson et al., 2010; 2013 and Lantz, 2013). Examples of critical parameters are methane leakage from the anaerobic digestion plant, the upgrading facility, the handling and storage of digestate, etc. A general conclusion from these existing studies is that the methane losses in all steps of the production chain need to be kept to a minimum, less than a few per cent, to secure a high GHG performance. This knowledge has also been implemented in practice, leading to more or less legal demands on today's methane producers to minimise their methane losses and use efficient, existing technologies to minimise the risks of losses. In addition, the GWP characterisation factors for methane are continuoulsy revised by the IPCC and have increased from previous the 23 g CO_2-equivalents to todays 34 g (100 year perspective) (IPCC, 2013). This increase in methane's global warming potential reinforces its importance in the overall WTW GHG performance of methane-fuelled transportation systems. Furthermore, if the time perspective is changed from 100 years to 20 years, the GWP of methane will increase by 2.5 times.

4.2.2 Well-to-wheel

The WTW GHG performance regarding compressed methane in light-duty vehicles is shown in Figure 4.11, based on the RED calculation methodology, and in Figure 4.12, based on the ISO calculation methodology. The WTT GHG emissions presented in detail in the previous section are here expressed as an average of the different biogas and bio-methane production systems. For biogas production systems, the average well-to-tank GHG performance is 12.3 g and -1.1 g CO_2-eq/MJ, when the RED and ISO calculation methodologies are utilised, respectively (see the WTT results in the previous section). The corresponding GHG performance for bio-methane is calculated to be, on average, 6.3 g and 18.6 g CO_2-eq/MJ, respectively. The variations in the WTT GHG performance of the analysed biogas and bio-methane supply systems are tested later in sensitivity analyses.

Both systems based on biomass-based methane give high GHG reductions. The GHG emission reduction will be approximately 90% for bio-methane, compared with gasoline, when the RED calculation methodology is applied (Figure 4.11). The corresponding reduction for biogas is over 80%. When the ISO calculation methodology is applied, the results will be the opposite, where biogas leads to an almost 100% reduction and where bio-methane leads to a roughly 75% reduction (Figure 4.12). Natural gas fuels lead to roughly a 15% GHG reduction, compared with gasoline.

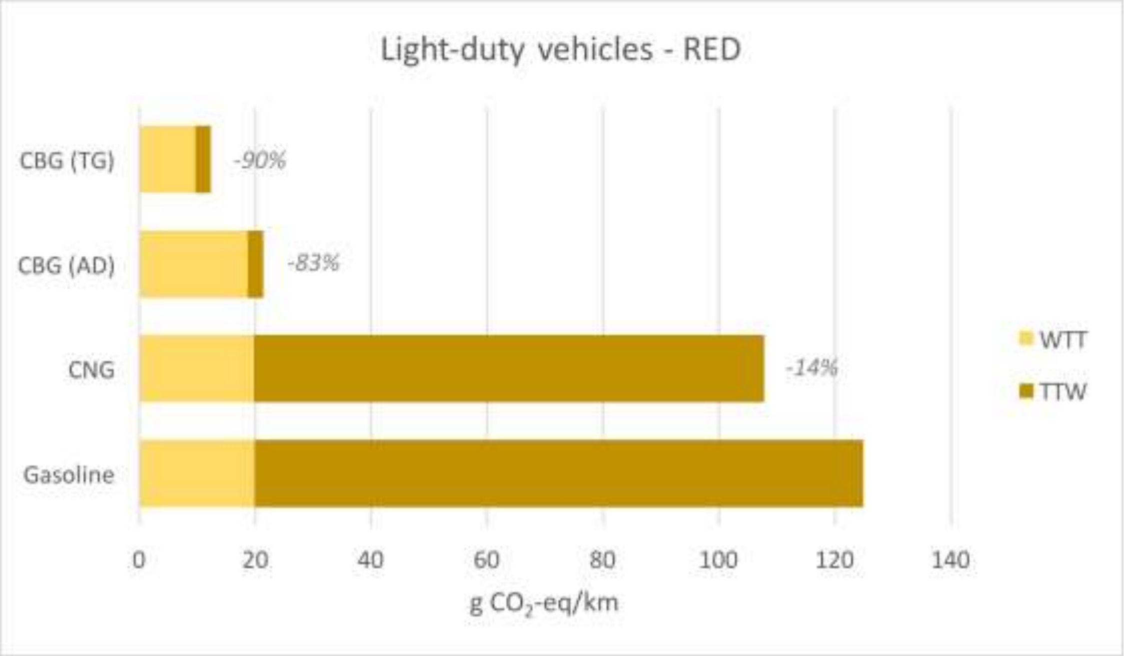

Figure 4.11. Well-to-wheel (WTW) GHG performance (g CO_2-eq/km) for compressed methane regarding light-duty vehicles and based on LCA calculation methodology according to the RED. For comparison, the GHG performance for gasoline is also shown and the corresponding GHG reduction (in %) for the methane vehicle fuels. The WTW GHG performance is divided between well-to-tank (WTT) emissions and tank-to-wheel (TTW) emissions.

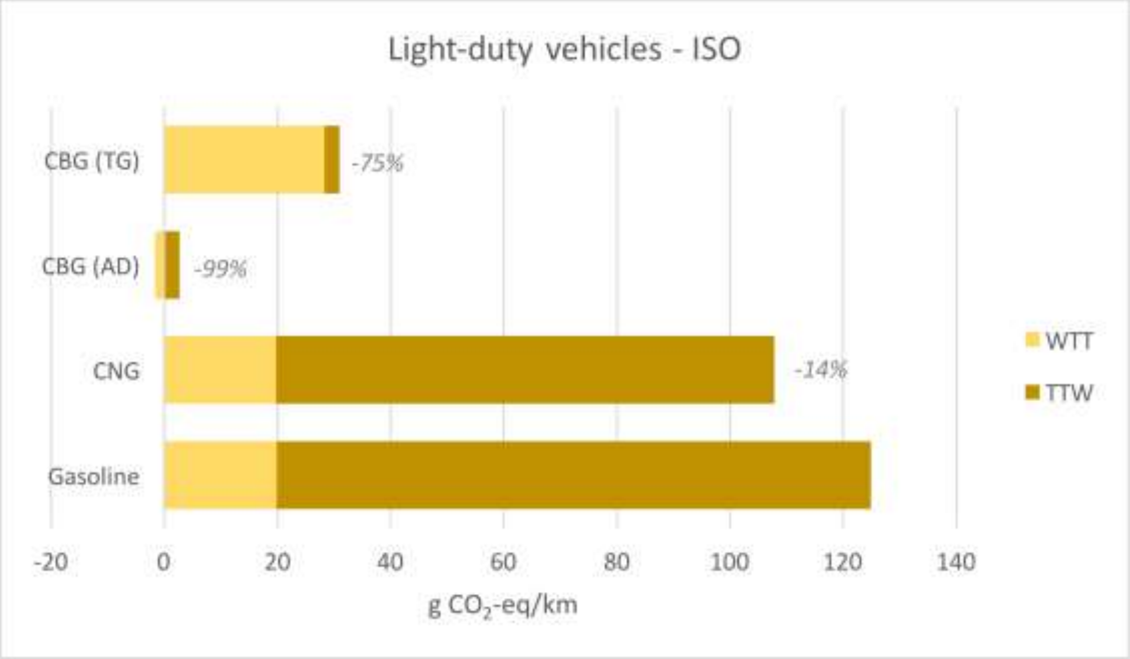

Figure 4.12. Well-to-wheel (WTW) GHG performance (g CO_2-eq/km) for compressed methane regarding light-duty vehicles and based on LCA calculation methodology according the ISO standard. For comparison, the GHG performance for gasoline is also shown and the corresponding GHG reduction (in %) for the methane vehicle fuels. The WTW GHG performance is divided between well-to-tank (WTT) emissions and tank-to-wheel (TTW) emissions.

The well-to-wheel GHG performance regarding compressed and liquefied methane in heavy-duty vehicles is shown in Figure 4.13, based on the RED calculation methodology, and in Figure 4.14, based on the ISO calculation methodology. The WTT GHG emissions, presented in detail in the previous section, is based on average emission levels (see above regarding light-duty vehicles).

The GHG emission reduction amounts to more than 80% and 90% when biogas and bio-methane is used as fuel in SI engine trucks, respectively, replacing diesel-fuelled trucks and when the RED calculation methodology is applied (Figure 4.13). There is an almost insignificant difference between SI engine trucks using compressed or liquefied methane. The GHG emission is somewhat higher for DF trucks (using approximately 5% diesel), but the GHG reduction is still very high compared with diesel trucks, almost 80%. Thus, the higher fuel efficiency in DF trucks does not completely compensate for the small share of diesel fuel used. Compressed and liquefied natural gas lead to roughly a 10% GHG reduction, compared with diesel, except when LNG is used in SI engine trucks which have roughly equivalent WTW GHG emissions as diesel-fuelled trucks. When the ISO calculation methodology is applied, the GHG emission reduction will increase for AD-based biogas systems, exceding a 100% reduction for SI engine trucks, whereas the gasification-based bio-methane system leads to a somewhat lower reduction, approximately 75% compared with diesel (Figure 4.14).

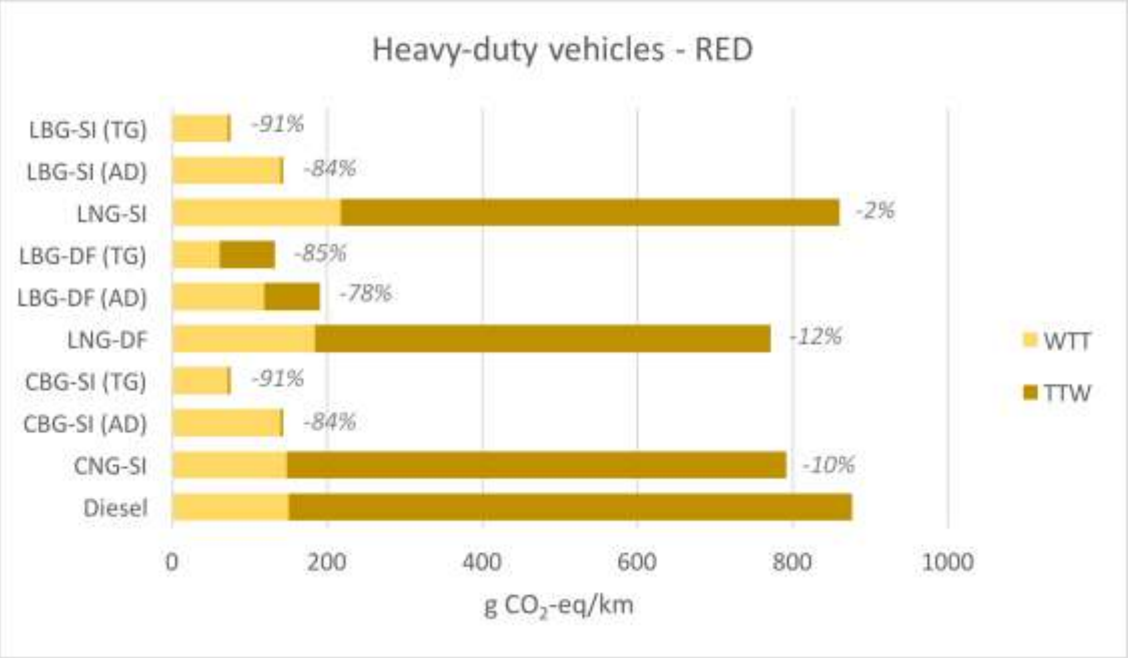

Figure 4.13. Well-to-wheel (WTW) GHG performance (g CO_2-eq/km) for compressed and liquefied methane regarding heavy-duty vehicles and based on LCA calculation methodology according to the RED. For comparison, the GHG performance for diesel is also shown and the corresponding GHG reduction (in %) for the methane vehicle fuels. The WTW GHG performance is divided between well-to-tank (WTT) emissions and tank-to-wheel (TTW) emissions.

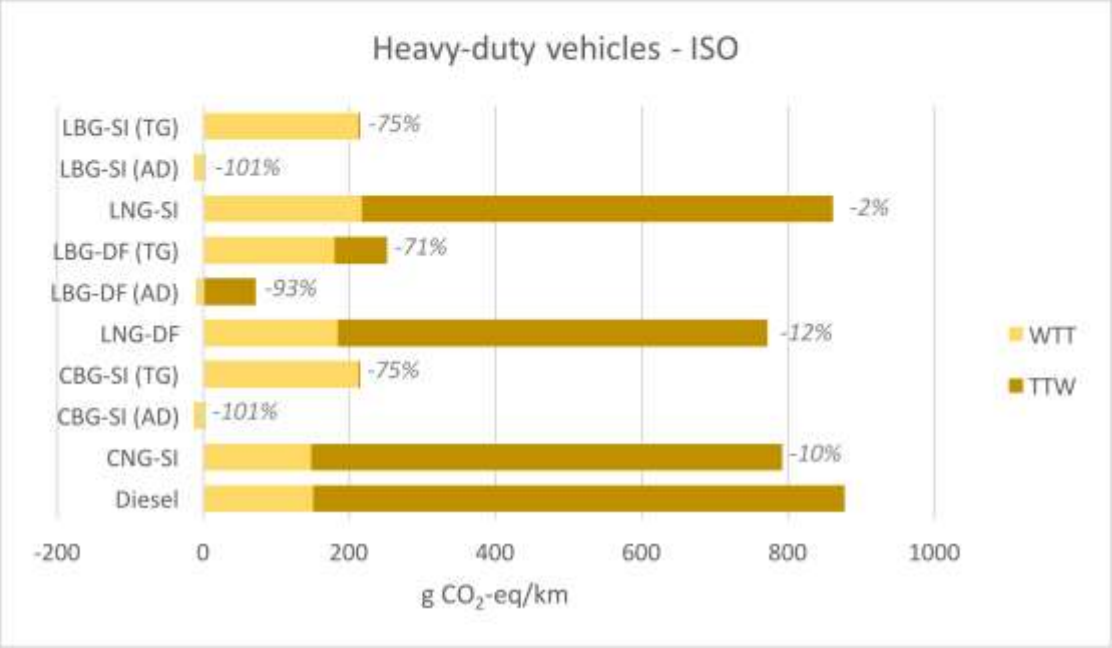

Figure 4.14. Well-to-wheel (WTW) GHG performance (g CO_2-eq/km) for compressed and liquefied methane regarding heavy-duty vehicles and based on LCA calculation methodology according to the ISO standard. For comparison, the GHG performance for diesel is also shown and the corresponding GHG reduction (in %) for the methane vehicle fuels. The WTW GHG performance is divided between well-to-tank (WTT) emissions and tank-to-wheel (TTW) emissions.

In this study, the WTT methane production systems are assumed to be based on the best available technology (BAT, see Section 2), meaning that the risk of methane losses is minimised. This, together with the significant existing knowledge regarding critical parameters for the GHG WTT performance, are the reasons why the sensitivity analysis in the study focuses primarily on the TTW emissions. However, the sensitivity analysis commences by summarising the variations in the GHG WTW performance for the alternative methane production chains based on the highest/lowest WTT GHG emissions calculated for the supply systems included in this study and presented in the WTT GHG diagrams previously in this Section 4.2.

For comparison, the variation in GHG WTW performance of CNG, LNG, gasoline and diesel is also included, based on the uncertainty interval regarding WTT GHG emissions presented by JRC (2014a). Here, the WTT GHG emissions are estimated to vary between 11–15 g CO_2-eq/MJ regarding CNG (13 g in base case), 17–21 g regarding LNG (19 g in base case), 12–16 g regarding gasoline (14 g in base case), and 14–17 g CO_2-eq/MJ regarding diesel (15.5 g in base case) (JRC, 2014a).

The results of the sensitivity analysis of the various WTT GHG emissions regarding light-duty vehicles are shown in Figure 4.15, and regarding heavy-duty vehicles in Figure 4.16. All calculations in this sensitivity analysis are based on the GHG calculation methodology according to the RED. A conclusion from the results presented in Figure 4.15 andFigure 4.16 is that the estimated variations in WTT GHG emissions will have a minor impact on the overall WTW GHG reduction when methane-based vehicle fuels are replacing gasoline and diesel. Depending on fuel supply systems, the changes in WTW GHG emission reduction for biogas- and bio-methane-based vehicle fuel systems, compared with gasoline and diesel, amount to approximately 6% or less. The corresponding changes in WTW GHG emission reduction for natural gas-based fuel systems are somewhat higher, namely, less than 8%.

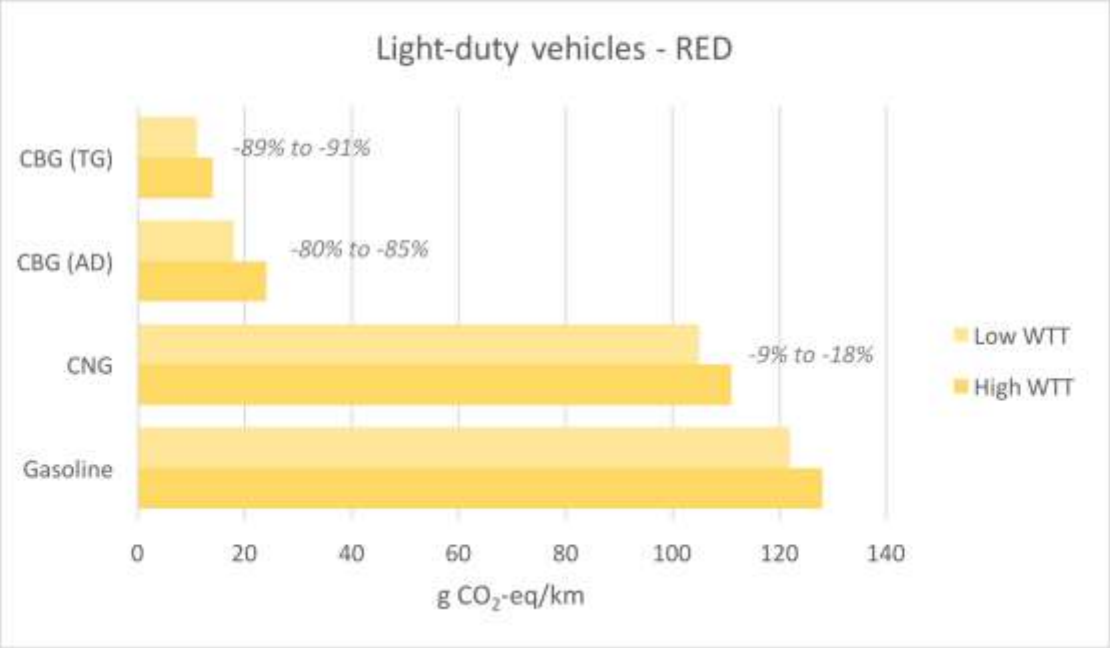

Figure 4.15. Variations in well-to-wheel (WTW) GHG performance (g CO_2-eq/km) for compressed methane fuel for light-duty vehicles, including the highest and lowest WTT GHG emissions calculated for the supply systems included in this study (and presented previously in this Section 4.2) and according to JRC (2014a) concerning CNG. The calculations are based on the RED calculation methodology. For comparison, the GHG performance for gasoline, including the variation in WTT GHG emissions according to JRC (2014a), is also shown, together with the corresponding GHG reduction (in %) for the methane vehicle fuels.

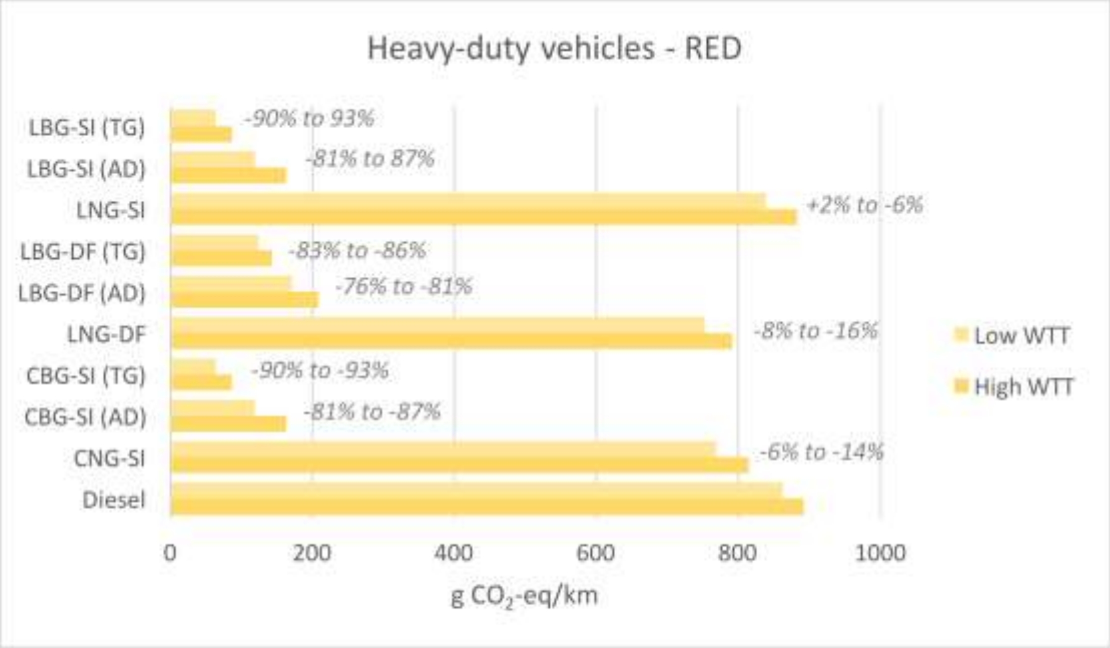

Figure 4.16. Variations in well-to-wheel (WTW) GHG performance (g CO_2-eq/km) for compressed and liquefied methane fuel for heavy-duty vehicles, including the highest and lowest WTT GHG emissions calculated for the supply systems included in this study (and presented previously in this Section 4.2) and according to JRC (2014a) concerning CNG and LNG. The calculations are based on the RED calculation methodology. For comparison, the GHG performance for diesel, including the variation in WTT GHG emissions according to JRC (2014a), is also shown, together with the corresponding GHG reduction (in %) for the methane vehicle fuels.

In the previous section, regarding end-use inventory data (Section 3.1), three critical aspects are discussed concerning the TTW GHG performance for heavy-duty vehicles. These are (i) the proportion of diesel used in DF LBG/LNG trucks, (ii) boil-off methane losses from the on-board tanks of liquefied methane, and (iii) the fuel consumption in CBG/CNG trucks with SI gas engines. These three critical aspects are tested in the following sensitivity analysis.

The proportion of diesel in the fuel of DF trucks are assumed to be equivalent to 5% in the base case, based on long haulage transportation services. However, depending on driving patterns, transportation services etc., the amount of diesel consumed in DF trucks may vary. In Figure 4.17, the WTW GHG emissions are therefore shown for heavy-duty vehicles when the proportion of diesel in the fuel of DF trucks is doubled, from 5 to 10%. The calculations are based on the RED calculation methodology. A conclusion from this sensitivity analysis is that the overall WTW GHG reduction is roughly 4% lower for liquid LBG-fuelled DF trucks, compared with diesel trucks, when the proportion of diesel in DF trucks is increased by 5%. The corresponding changes regarding LNG-fuelled DF trucks are, however, marginal.

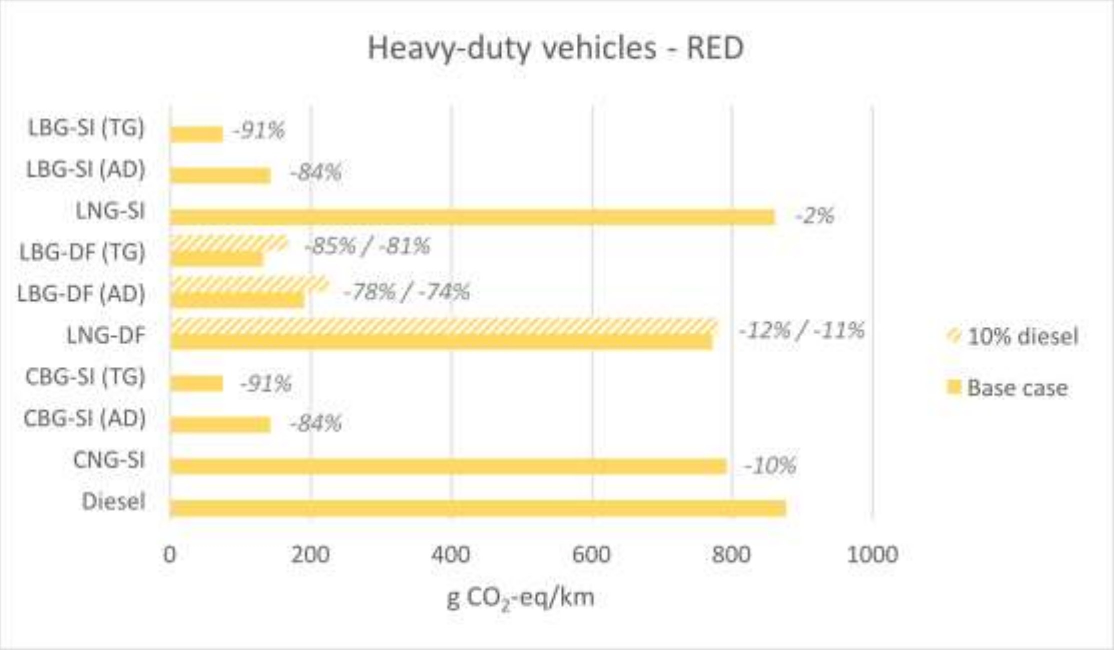

Figure 4.17. Variations in well-to-wheel (WTW) GHG performance (g CO_2-eq/km) for compressed and liquefied methane fuel for heavy-duty vehicles, including a change from 5% diesel consumption (base case) to 10% (sensitivity analysis) in dual-fuel (DF) LBG/LNG trucks. The calculations are based on the RED calculation methodology. For comparison, the GHG performance for diesel is also shown, together with the corresponding GHG reduction (in %) for the methane vehicle fuels.

As discussed in Section 3.1, a risk of methane leakage from the on-board storage of liquefied methane exists if the storage time exceeds the designed holding time without venting. On-board LNG/LBG storage tanks are typically designed for a holding time of approximately five days. Thus, if a LBG/LNG truck containing liquefied methane fuel in the tanks is left unused for several days, there is a risk of methane losses through venting. The impact on the WTW GHG performance of such boil-off methane emissions is tested in the following sensitivity analysis, assuming boil-off rates equivalent to 1% and 3% (see Section 3.1 regarding estimated boil-off rates), and according to the RED calculation methodology.

A conclusion from Figure 4.18 is that the boil-off methane losses may have a significant, negative impact on the WTW GHG performance for trucks using LNG/LBG, especially if the boil-off rates amount to several percent. Instead of having a WTW GHG reduction equivalent to 78–85%, compared with diesel trucks, the GHG reduction for LBG-DF trucks is reduced to approximately 56–62 % when the boil-off rate is equivalent to 3%. A similar deterioration of the GHG performance is seen for SI engine trucks using LBG. If the boil-off rate amounts to approximately 1.5%, then the WTW GHG performance will be similar for a LNG-DF truck as for a diesel truck. The corresponding boil-off rate for a LNG-SI engine truck is less than 0.5%.

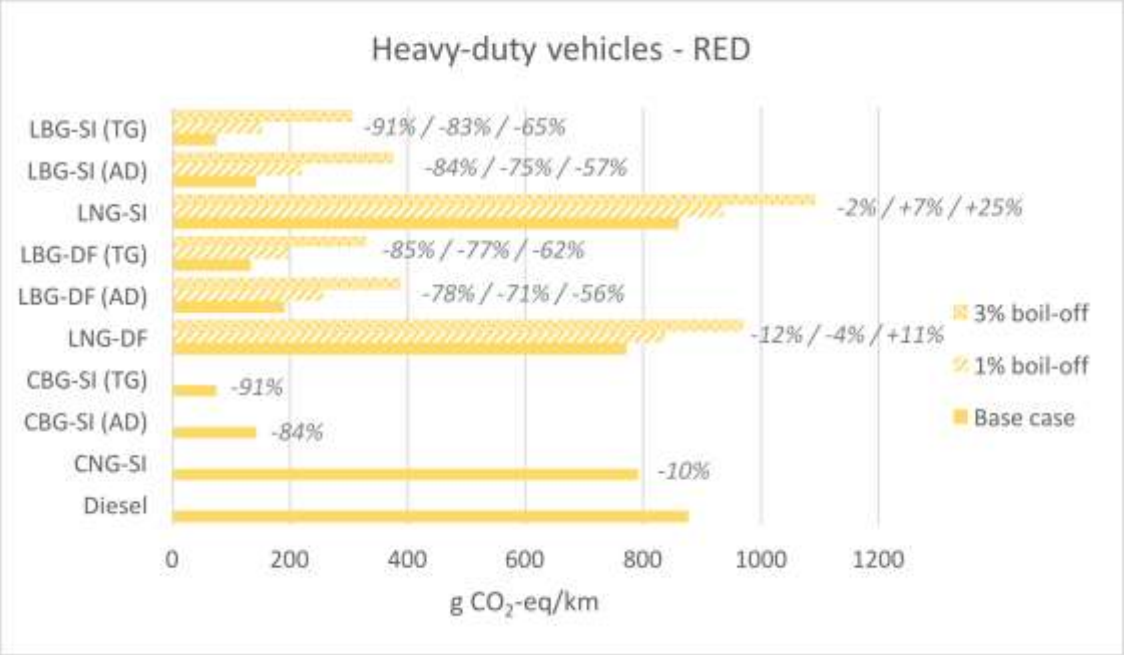

Figure 4.18. Variations in well-to-wheel (WTW) GHG performance (g CO$_2$-eq/km) for compressed and liquefied methane fuel for heavy-duty vehicles, including a change from 0% boil-off methane emissions (base case) to 1% and 3% (sensitivity analysis) in dual-fuel (DF) LBG/LNG trucks and spark-ignition (SI) trucks using LBG/LNG. The calculations are based on the RED calculation methodology. For comparison, the GHG performance for diesel is also shown, together with the corresponding GHG reduction (in %) for the methane vehicle fuels.

Similar to the variations in the proportion of diesel in the fuel of DF trucks, the fuel consumption may vary due to changes in driving patterns, transportation operations etc. In Figure 4.19, the effects on the WTW GHG performance regarding SI engine trucks are shown when the methane fuel consumption is increased, equivalent to 25% higher fuel consumption than in diesel trucks, or decreased, equivalent to 10% higher fuel consumption than in diesel trucks, compared with the base case (18% higher fuel consumption than diesel trucks). A conclusion from the results in Figure 4.19 is that these changes in methane fuel consumption will have a minor impact on the WTW GHG performance of biogas- and bio-methane-fuelled SI engine trucks, but a somewhat higher impact regarding NG-fuelled trucks.

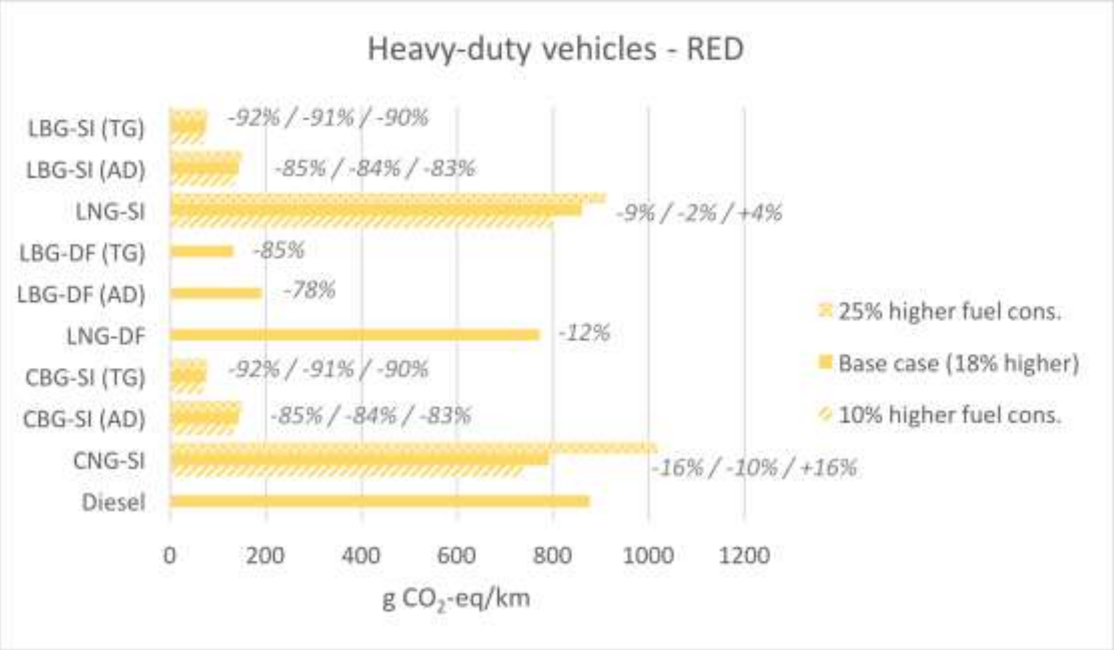

Figure 4.19. Variations in well-to-wheel (WTW) GHG performance (g CO_2-eq/km) for compressed and liquefied methane fuel for heavy-duty vehicles, including a change from 18% higher fuel consumption in spark-ignition (SI) trucks, compared with dual-fuel (DF) and diesel trucks (base case), to 10% and 25% higher fuel consumption (sensitivity analysis). The calculations are based on the RED calculation methodology. For comparison, the GHG performance for diesel is also shown, together with the corresponding GHG reduction (in %) for the methane vehicle fuels.

4.3 ENERGY EFFICIENCY PERFORMANCE

The energy efficiency performance of the different methane vehicle fuel systems is presented both from a well-to-tank perspective, including more detailed information about the contribution from the various steps, and from a complete well-to-wheel perspective.

4.3.1 Well-to-tank

The WTT energy efficiency performance for biogas and bio-methane is shown in Figure 4.20. For biogas supply systems, the primary energy input varies between approximately 0.3 and 0.4 MJ/MJ biogas, with an average of 0.32. The highest input of primary energy is related to the liquefaction process, which is not compensated for by a lower energy input in distribution and filling stations. The low energy input in the amine scrubber system is due to the possibility for waste heat utilisation, even though a somewhat higher energy input is needed in the compression step, compared with the water scrubber system. When comparing different scales of production, the small scale systems require somewhat less energy per MJ methane produced, but the differences are minor.

For bio-methane production, the primary energy input varies approximately between 0.2 to 0.3 MJ/MJ bio-methane, with an average of 0.24. Thus, the energy input is somewhat lower for the TG systems. However, it is clear that the design of the gasification process has an impact on the overall result. The larger plant gives roughly a 50% higher input of primary energy even though this is partly compensated for in the LBG system, where the energy demand is lower in the liquefaction process than in the smaller gasification system.

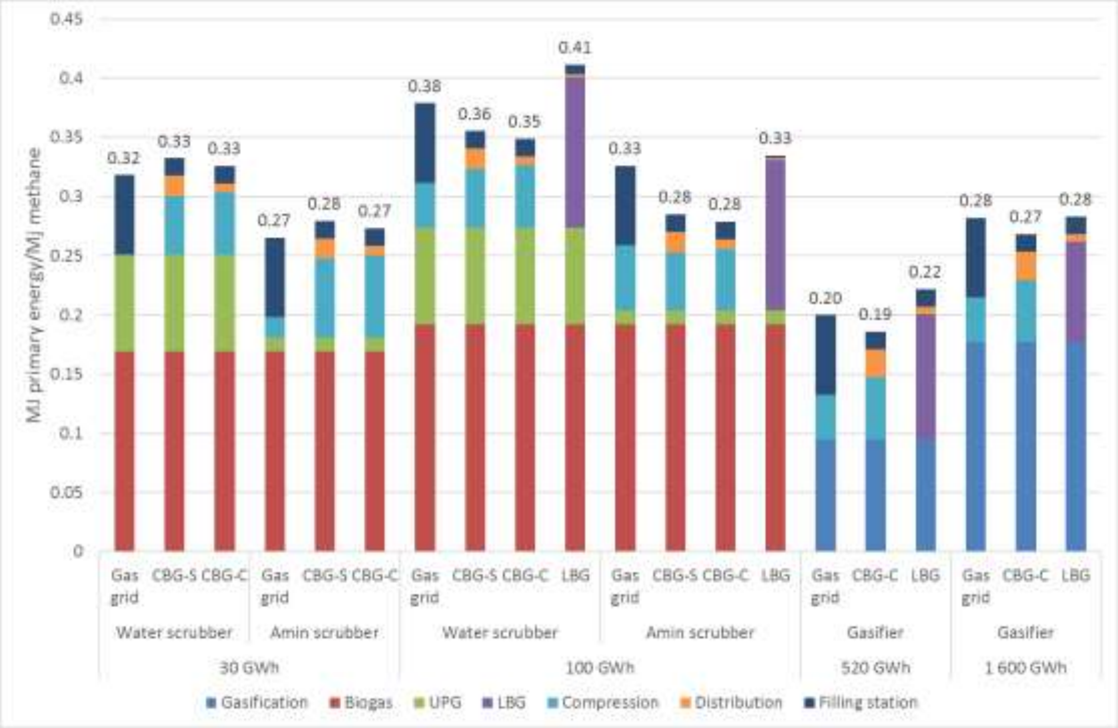

Figure 4.20. The WTT energy efficiency performance (MJ primary energy / MJ methane) for biogas and bio-methane supply systems.

4.3.2 Well-to-wheel

The WTW energy efficiency performance regarding compressed methane in light-duty vehicles is shown in Figure 4.21. In Figure 4.22, the corresponding energy efficiency performance regarding compressed and liquefied methane in heavy-duty vehicles is shown. For compressed biogas and bio-methane production systems, the WTT energy efficiency performance is, on average, 0.31 and 0.23 MJ primary energy/MJ methane, respectively (see the WTT results in the previous section). The corresponding energy efficiency performance for liquefied biogas and bio-methane is, on average, 0.37 and 0.25 MJ primary energy/MJ methane, respectively.

The WTW energy efficiency will be somewhat higher when compressed methane is utilised instead of gasoline in light-duty vehicles, varying from approximately 10% to 20% higher primary energy input in bio-methane and biogas systems, respectively (Figure 4.21). When CNG is utilised, the corresponding increase in WTW primary input is somewhat lower.

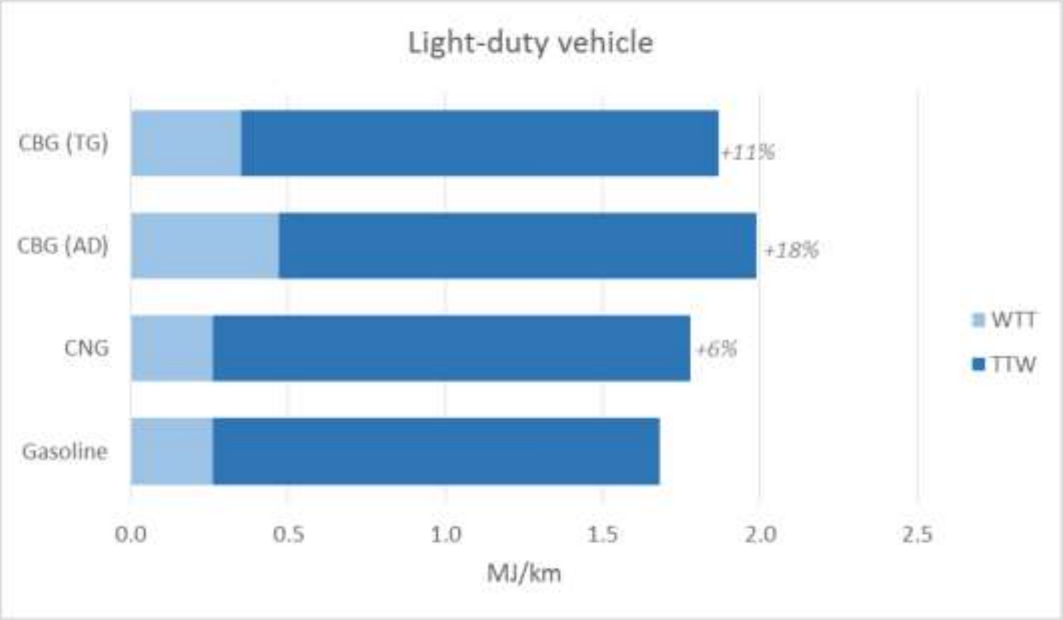

Figure 4.21. Well-to-wheel (WTW) energy efficiency performance (MJ/km) for compressed methane for light-duty vehicles. For comparison, the energy efficiency performance for gasoline is also shown and the corresponding changes in energy efficiency (in %) for the methane vehicle fuels. The WTW energy efficiency is divided between the energy input in well-to-tank (WTT) and tank-to-wheel (TTW), respectively.

For heavy-duty vehicles, the WTW energy efficiency will be similar, or somewhat higher, when liquefied methane is utilised in DF engine trucks, compared with diesel trucks (Figure 4.22). When compressed methane is used as fuel in SI engine trucks, the WTW primary energy input will be approximately 15% to 30% higher, compared with diesel-fuelled trucks. When liquefied methane is used as fuel in SI engine trucks, the WTW primary energy input will increase slightly more.

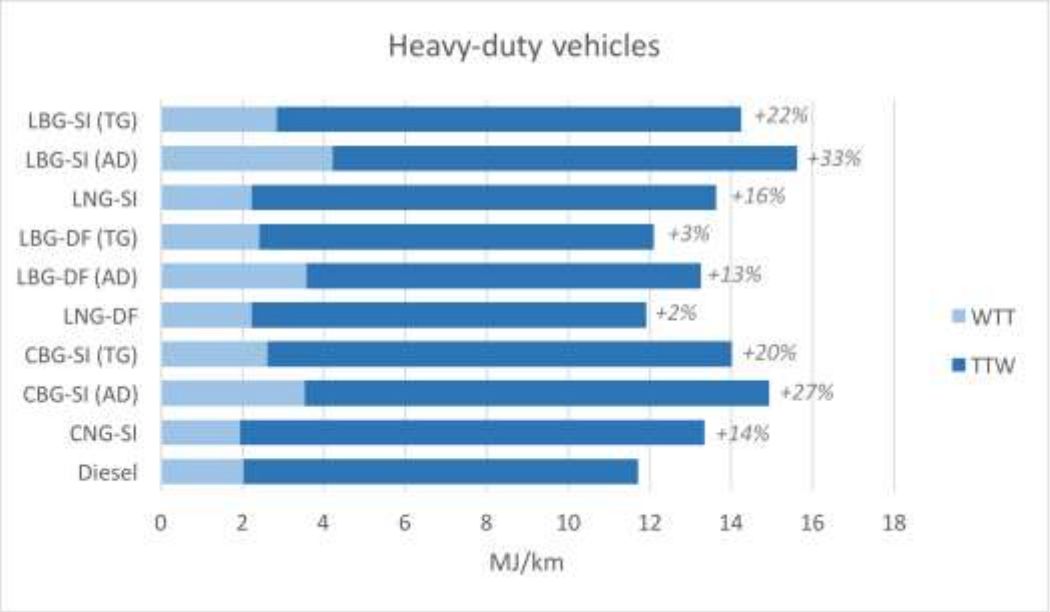

Figure 4.22. Well-to-wheel (WTW) energy efficiency performance (MJ/km) for compressed and liquefied methane for heavy-duty vehicles. For comparison, the energy efficiency performance for diesel is also shown and the corresponding changes in energy efficiency (in %) for the methane vehicle fuels. The WTW energy efficiency is divided between the energy input in well-to-tank (WTT) and tank-to-wheel (TTW), respectively.

In Figure 4.23, the effects on the WTW energy efficiency performance for SI engine trucks is shown when the methane fuel consumption is increased, equivalent to 25% higher fuel consumption than in diesel trucks, or decreased, equivalent to 10% higher fuel consumption than in diesel trucks, compared with the base case (18% higher fuel consumption than diesel trucks). Here, the higher/lower WTW energy efficiency will be equivalent in percent to the changes in the sensitivity analysis (Figure 4.23).

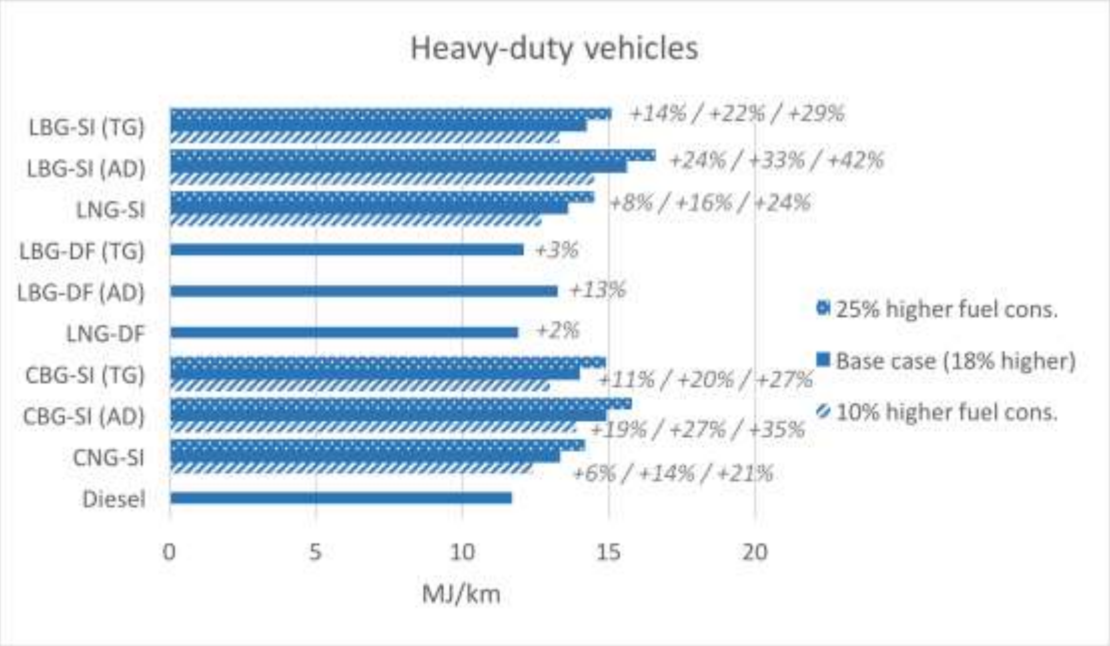

Figure 4.23. Variations in well-to-wheel (WTW) energy efficiency performance (MJ/km) for compressed and liquefied methane fuel for heavy-duty vehicles, including a change from 18% higher fuel consumption in spark-ignition (SI) trucks, compared with dual-fuel (DF) and diesel trucks (base case), to 10% and 25% higher fuel consumption (sensitivity analysis). For comparison, the energy efficiency performance for diesel trucks is also shown, together with the corresponding differences in WTW energy efficiency (in %) for the methane vehicle fuels.

5 DISCUSSION AND CONCLUSIONS

The overall conclusions from this study are that

- biomass-based methane vehicle fuel systems lead to significant WTW GHG benefits, compared with fossil-based vehicle fuel systems,
- the WTW energy efficiency for biomass-based methane is almost comparable, or slightly higher to that of fossil methane, gasoline and diesel,
- the WTW costs for biomass-based methane is comparable, or slightly higher than those for fossil methane, gasoline and diesel, based on their current market price, excluding VAT but including other relevant taxes,
- the selection of post-treatment and distribution system for biomass-based methane vehicle fuel systems is of minor importance regarding the WTW GHG, energy efficiency and cost performance,
- thus, these findings provide the justification for the further development and implementation on a commercial scale of all the various systems.

This study confirms that Swedish renewable methane well-to-wheel (WTW) systems, based on today’s best available technology, perform very well from a greenhouse gas (GHG) perspective. Independently of production systems (anaerobic digestion using current mix of organic waste and residues, or thermal gasification using forest residues), distribution systems (compressed or liquefied methane and gas grids or containers transported by trucks), and final use (spark-ignited otto engine vehicles using compressed methane or dual-fuel diesel engine vehicles using liquefied methane), the reduction of WTW GHG emissions typically exceeds 80% according to the calculation methodology in the EU Renewable Energy Directive (RED), compared with gasoline and diesel fuelled vehicles. This GHG reduction fulfils the requirement stated in the RED that new biofuel plants and systems after 2015 shall lead to a GHG reduction of 60% or more compared with gasoline and diesel (European Commission, 2015). Also when the ISO calculation methodology is applied, which includes the indirect GHG effects of by-products and soil carbon changes, the GHG emission reduction is roughly in the same order of magnitude, but where the performance is somewhat changed between the anaerobic digenstion (AD) and thermal gasification (TG) supply systems. Thus, the recommendation is that all the various renewable methane well-to-wheel systems included in this study should, from a GHG perspective, be furhter developed and implemented commercially.

The WTW systems based on natural gas (NG) will also lead to GHG benefits, compared with gasoline and diesel, but here the reduction is restricted to approximately 10%. A critical aspect regarding the GHG performance of methane-fuelled vehicle systems is methane losses throughout the complete WTW system. One example analysed in this study is the risk of boil-off methane losses from the on-board storage of liquefied methane. If such boil-off emission levels should amount to 3%, then the GHG reduction will be reduced to roughly 50–70% for renewable methane WTW systems, compared with diesel-based systems. This highlights the fact that special care must be taken so that liquid methane-fuelled vehicles are used in a proper, continuous manner and that long stops of the vehicles are avoided. Also in service and repair situations there is a need for liquid methane return systems to be in place. Other critical aspects regarding the WTW GHG performance of methane-fuelled vehicles are, for example, the proportion of diesel used in dual-fuel (DF) trucks, which may vary due to driving patterns and transport operations. Also, the fuel consumption efficiency in compressed methane-fuelled, spark-ignited (SI) engine trucks will affect the GHG performance.

The WTW primary energy input is higher in methane-fuelled vehicle systems than in comparable gasoline- and diesel-fuelled vehicle systems, vaying from +3% up to +33% depending on the methane-based powertrain system. Regarding powertrain systems based on SI engine, heavy vehicles fuelled with compressed methane, the WTW primary energy input is typically 10–15% higher than in systems based on DF engine, heavy vehicles fuelled with liquid methane. SI engine, heavy vehicles supplied with liquid methane, instead of compressed methane, will lead to only a marginally higher WTW primary energy input. The WTW primary energy efficiency is mainly affected by the fuel consumption, which may vary due to driving patterns and transport operations.

The well-to-tank (WTT) costs for biogas and bio-methane vehicle fuel systems are similar but the costs for smaller gasification systems are somewhat higher than the costs for the AD systems and the larger TG system, based on the assumptions made in this study. The costs for the different post-treatment and distribution systems of biomass-based methane are also comparable, and represent some 20–40% of the total WTT costs. Thus, from an economic perspective, the selection of different production, post-treatment and distribution systems of renewable methane vehicle fuel systems is of minor importance. However, there are uncertainties in the WTT cost calculations performed, especially regarding the production costs of biogas and bio-methane.

The WTW costs of CBG- and CNG-fuelled light-duty vehicles are somewhat higher than the cost of gasoline-fuelled cars, roughly 15 to 20% higher. The WTW costs include the current market price of gasoline, diesel and NG-based vehicle fuels, excluding VAT but including other relevant taxes, and the additional vehicle cost of methane-fuelled cars and trucks. For light-duty vehicles, the additional vehicle cost (tank-to-wheel cost, TTW) is estimated to represent some 25% of the WTW cost. The WTW costs are, however, sensitive to changes in the market prices of fossil-based vehicle fuels, including both changes in crude oil prices and in taxes, and if current tax exemption for renewable methane should be removed. Thus, the WTW cost comparisons between the different powertrain systems included in this study need to be updated if the economic conditions change.

Regarding heavy-duty vehicles, LBG-fuelled DF trucks have similar WTW costs as corresponding diesel trucks, whereas LNG-fuelled DF trucks have somewhat lower WTW costs. CBG- and CNG-fuelled trucks have roughly 15–20% higher WTW costs than diesel trucks. The additional TTW costs for methane-fuelled heavy-duty vehicles are estimated to represent some 10% of the WTW costs, but may vary from 5 to 15%. Apart from variations in the market price of fossil-based vehicle fuels, there are also uncertainties in the production costs of biogas and bio-methane, which will affect the WTW costs. For example, if the WTT costs of biogas and bio-methane increase by almost 50%, compared with the base case, representing the higher levels in the calculated cost interval, then the WTW costs of LBG- and CBG-fuelled trucks will be 30-50% higher than the WTW costs of comparable diesel-fuelled trucks. On the other hand, if the WTT cost is reduced by, say, 25%, representing the lower levels in the calculated cost interval, then the WTW costs of LBG- and CBG-fuelled trucks will be 15–25% lower than the WTW cost of diesel trucks.

6 ACKNOWLEDGEMENTS

This report is the result of a cooperation project within the Swedish Knowledge Centre for Renewable Transportation Fuels (f3). The f3 Centre is a nationwide centre, which through cooperation and a systems approach contributes to the development of sustainable fossil free fuels for transportation. The centre is financed by the Swedish Energy Agency, the Region Västra Götaland and the f3 Partners, including universities, research institutes, and industry (see www.f3centre.se). The report is also based on ongoing research projects financed by the Swedish Energy Agency, which we hereby acknowledge.

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APPENDIX A – ENERGY CARRIERS

In this study, energy carriers such as electricity, heat, wood chips and RME are used as process energy input or as a raw material for renewable methane. Some processes also result in waste heat or surplus electricity. In this appendix, assumptions made regarding GHG emissions, primary energy factors and market price are presented.

GHG EMISSIONS AND PRIMARY ENERGY FACTORS

Greenhouse gas emissions and energy balance for the energy carries included in this study are summarised in Table A.1.

Table A.1. Greenhouse gas emissions and primary energy factors applied in this study.

	GWP (g CO_2-eq/MJ)	Primary energy factor
Electricity (Swedish)[1]	11	2.1
Electricity (Nordic)[2]	35	1.7
Wood chips for process energy[3]	10.7	1.03
Wood chips as raw material[3]	10.7	0.03
Heat from wood chips[4]	13.4	1.3
RME[5]	26.4	1.27
Diesel[6]	80.6	1.09
Gasoline[6]	76.9	1.09

[1] Calculations according to the ISO standard are based on the average electricity mix in Sweden as presented in Gode et al. (2011).

[2] Calculations according to the ISO standard are based on the average electricity mix in the Nordic countries where GHG emissions are based on Martinsson (2012) and the primary energy factor on Gode et al. (2011) although the latter is calculated for the production of 2008 and the emission factor is an average for 2005–2009 and thus not completely correlating.

[3] Regarding wood chips, calculations are based on chips from logging residues including soil carbon changes after 2–3 rotations (Lindholm et al. 2011). When used as raw material for gasification, the primary energy factor includes collection and transport of wood chips. When used as fuel for process heat generation, the primary factor also includes the estimated boiler efficiency.

[4] Assuming the same wood chips as above and a boiler efficiency of 80% (Lindholm et al., 2011; Lantz and Börjesson, 2014).

[5] Gode et al. (2011)

[6] Diesel with 5% RME and gasoline with 5% ethanol (Gode et al., 2011).

ENERGY PRICES

Estimated market prices for the various energy carriers used in this study are presented in Table A.2.

Table A.2. Market prices for various energy carriers applied in this study.

	SEK/MWh
Electricity (process energy input)[1]	500
Electricity (surplus)[2]	400
Wood chips[3]	200
District heating (surplus heat)[4]	250
RME (process energy input)[5]	850
HVO[6]	1 100
Diesel[6]	1 100
Gasoline[6]	1 200
CBG/CNG[7]	1 000
LBG/LNG[8]	900

[1] The cost for electricity is set to 500 SEK/MWh based on the average electricity price in 2015 for industries with a consumption in the range of 2–20 GWh annually (SCB, 2016). The price includes transmission, electricity certificates and taxes. Potential differences between different biogas pathways due to geographic location, volumes or consumption patterns are thus not included.

[2] The value of surplus electricity is set to 400 SEK/MWh, corresponding to the average market price of electricity on Nordpool in 2015 and the average market price of electricity certificates during the same period (Nordpool, 2016; Energimyndigheten, 2016).

[3] The cost for wood chips is set to 200 SEK/MWh based on the average price paid by district heating plants in Sweden in 2012–2014 (Energimyndigheten, 2015b).

[4] The value of surplus heat is set to 250 SEK/MWh based on a wood chips cost of 200 SEK/MWh and an assumed boiler efficiency of 85%, plus approximately 15 SEK/MWh for operation and maintenance.

[5] The price for RME is set to 7.9 SEK/dm^3, given an estimated volume discount of approximately 10% compared to the price at filling stations and full tax reduction since it is not used as a biofuel (Preem, 2016a,b; Skatteverket, 2016). For comparison, the price of FAME in the ports of Amsterdam and Rotterdam was approximately 6.5 SEK/dm^3 in the second half of 2015 (Energimyndighetn, 2016).

[6] Fuel prices listed at Circle K (2016).

[7] Average market price in Sweden (Gasbilen, 2016).

[8] Market price at filling stations for LNG (Circle K, 2016) and a mix of LBG and LNG (Fordonsgas, 2016).

APPENDIX B – BIOGAS PRODUCTION

The Swedish biogas production by anaerobic digestion is based on a mix of food waste, manure, industrial waste and slaughter house waste. Since the performance of a specific biogas plant to a high degree depends on the composition of the feedstock, the energy input and GHG emissions in the present study are calculated for fictious biogas plants where the feedstock mix is based on the average mix in Swedish large-scale, co-digestion plants as in 2014 (Energimyndigheten, 2015a).

Calculations are performed for biogas plants with an annual production of 30 GWh and 100 GWh, respectively. Both plants are assumed to have the same feedstock mixture and process conditions, resulting in the same production of biogas and digestate per tonne of feedstock. Estimated feedstock composition is presented in Table B.1.

Table B.1. Annual input of feedstock and output of biogas and digestate in the two biogas plants included in the study.

	30 GWh	100 GWh	Maximal CH_4 production (m^3 CH_4/ton DM)	Practical CH_4 production (m^3 CH_4/ton DM)[3]
Feedstock[1] (tonne)	66 000	218 000		
- Food waste	14 000	47 000	453	408
- Manure	26 000	86 000	191	172
- Industrial waste	19 000	62 000	473	426
- Slaughterhouse waste	7 000	23 000	462	416
Digestate[4] (ton)	59 900	199 500		
Biogas[5] (m^3)	4 650 000	15 510 000		

[1] Estimation based on the feedstock used in 2014 in Swedish biogas plants (Energimyndigheten, 2015a).

[2] Calculated average values based on Carlsson and Uldal (2009) where DM= dry matter. Manure is based on a mixture of cattle and pig manure.

[3] Assuming 90% of the values given by Carlsson and Uldal (2009).

[4] Calculated according to the model presented in Lantz et al. (2013).

[5] Biogas production at a methane concentration of 63% before potential losses.

Table B.2. Estimated feedstock composition.

	DM (%)[1]	VS (%)[1]	C (% of DM)[2]	N-tot (kg/t)[3]	P (kg/t)[3]
Food waste	19	17	45	4.0	0.9
Manure	8.5	6.8	41	4.3	0.5
Industrial waste	13	12	45	4.3	0.5
Slaughterhouse waste	18	15	45	4.3	0.5

[1] Estimate based on Carlsson and Uldal (2009) where DM= dry matter and VS = volatile solids.

[2] Based on Huang et al. (2006) for pig manure and Rodhe et al. (2013) for cattle manure. For the various types of waste, the carbon fraction is based on an assumption.

[3] Estimate based on Ljung et al. (2013) for waste and Greppa Näringen (2011) for manure.

As presented in Chapter 2, calculations are performed according to the method presented in the ISO-standard as well as the RED. When using the method presented in the RED, calculations include direct energy input and methane slip from the biogas plant. Also, the electricity used is set to represent the average Nordic power production. Calculations according to the ISO-standard includes these parameters but also energy input and emissions from the handling of digestate. Since the system expansion approach is applied, calculations also include replacement of mineral fertilizers by digestate, including changes in soil organic matter, as well as avoided emissions from conventional storage and handling of manure. Finally, emissions originating from the use of electricity is based on the average Swedish power production. For a more detailed description of this systems expansion approach, see e.g. Tufvesson et al. (2013) and Börjesson et al. (2010).

DIRECT ENERGY INPUT AND GHG EMISSIONS

Data on energy input and related GHG emissions are based on a literature review, chosen to represent modern co-digestion plants (Table B.3). Although the two production plants differ in size, it is assumed that the same parameters can be applied in both cases. The only difference is that the transportation distance for feedstock and digestate is set to 40 km (including return transport) for the small biogas plant and 60 km for the larger one.

Calculations of GHG emissions and energy balance are based on the data and assumptions presented in Tables A.1 and A.2.

In addition to the direct energy input it is assumed that the methane slip from the biogas plant corresponds to 0.3%, representing the average slip at Swedish biogas plants (Avfall Sverige, 2016).

Also, it is assumed that 4% of the biogas is flared due to planned and unplanned stops in upgrading or distribution (Goeffeng, 2015). The methane slip from the flare is set to 2% (Lantz and Börjesson, 2014).

Table B.3. Energy input and methane emissions in the production of biogas.

Food waste - Collection and transport to pre-treatment[1] - Pre-treatment[2]	 308 MJ diesel/t 95 MJ electricity/t
Industrial waste, slaughter house waste and digestate - Loading and unloading[3] - Transport[3]	 1.8 MJ diesel/t 16 MJ diesel/km
Manure (incl. return transport of digestate) - Loading and unloading[3] - Transport[3]	 1.8 MJ diesel/t 18 MJ diesel/km
Process energy - Electricity[4] - Heat[4]	 36 MJ/t 126 MJ/t

[1] **Based on Börjesson et al. (2010). Transportation distance to pre-treatment facility is set to 20 km.**

[2] **Based on Avfall Sverige (2013)**

[3] **Based on Tufvesson et al. (2013).**

[4] **Based on Lantz and Björnsson (2016).**

DIGESTATE HANDLING

In addition to the energy input presented in Table B.3, storage and handling of digestate will also result in emissions of CH_4, as well as of direct and indirect emissions of N_2O (Tufvesson et al., 2013).

Digestate properties, presented in Table B.4., are calculated according to the method presented in Lantz et al. (2013) based on the assumed content of each type of feedstock as presented in Tables B.1 and B.2.

The digestate produced is stored under a roof and 1% of N-tot is assumed to be lost as NH_3-N (Karlsson och Rodhe, 2002). Based on IPCC (2006) it is assumed that there are no direct emissions of N_2O from the storage although 1% of the NH_3-N is indirectly transformed to N_2O.

Calculations of the losses of CH_4 from the digestate storage are based on Equation 1 which is modified from IPCC (2006), see also e.g. Tufvesson et al. (2013).

$$M_{CH4} = M_{VS} * B_0 * 0{,}72 * MCF \qquad \text{(Equation 1)}$$

M_{CH4} = emissions of methane (kg CH_4)

M_{VS} = organic material (VS) (kg)

B_0 = maximum methane production (m^3 CH_4/kg VS)

MCF = methane conversion factor (%)

0,72 = density of methane

The methane conversion factor is set to 3.5% which Naturvårdsverket (2013) also applies for liquid manure. The amount of VS and theoretical methane potential (B_0) are calculated to be 253 m^3 CH_4/tonne VS with the model presented in Lantz et al. (2013). When calculating the B_0 value of the digestate, the maximum CH_4 production given in Table A.1 is used as B_0 of each feedstock except manure. For the mixture of cattle and pig manure, the B_0 value is set to 254 m^3/ton DM (IPCC, 2006).

When spread on arable land, it is assumed that 10% of added NH_4-N is lost as NH_3-N (Karlsson and Rodhe, 2002). In addition, 1% of added N-tot will be lost directly as N_2O and 1% of the NH_3-N is indirectly transformed to N_2O (IPCC, 2006). Energy input for spreading of digestate is set to 10.8 MJ diesel/t (Tufvesson et al., 2013).

Table B.4. Digestate properties after storage losses.

	After storage
TS (%)	4.9
VS (% TS)	63
C (% TS)	41
N-tot (g/kg)	4.6
NH_4-N (kg/t)	3.3
P (kg/t)	0.7
K (kg/t)	2.2

SYSTEM EXPANSION

In addition to the direct energy input and emissions from the production of biogas and handling of digestate, the system expansion also includes the replacement of mineral fertiliser by digestate and the replacement of conventional handling of manure. The system expansion also includes changes in soil organic matter that occurs when digestate replaces manure and mineral fertilisers.

Conventional handling of manure

Conventional handling of manure results in emissions of greenhouse gases and losses of nitrogen from storage as well as from the spreading of manure. When manure is used as feedstock in a biogas system, these emissions are reduced, which is presented in Tufvesson et al. (2013) and Lantz and Björnsson (2016).

In this study, it is assumed that the manure used for biogas production consists of 50% liquid cattle manure and 50% liquid pig manure. It is assumed that all manure was previously stored under a floating crust and that 3.5% of N-tot was lost as NH_3-N (Naturvårdsverket, 2013). It is also assumed that 0.5% of N-tot is lost as N_2O and that 1% of NH_3-N is indirectly transformed to N_2O (IPCC, 2006).

Losses of CH_4 from storage of liquid manure are calculated using Equation 1 with a methane conversion factor of 3.5% Naturvårdsverket (2013). Manure composition and theoretical methane potential (B_0) are presented in Table B.2.

Finally, energy input and emissions from spreading of manure is based on the same emission factors as earlier presented for digestate.

Replacement of mineral fertilizer

As presented earlier, the digestate is spread as a fertiliser on arable land. In this study, the increased amount of NH_4-N and P that is spread compared to spreading of undigested manure is assumed to replace mineral fertilisers (Börjesson et al., 2009; Tufvesson et al. 2013). Thus, the biogas system benefits from avoided emissions from the production and spreading of mineral fertilisers.

GHG emissions from the production of mineral fertilisers are set to 6.7 and 3.2 kg CO_2-eqv. /kg of N and P respectively (Börjesson et al., 2010; Tufvesson et al., 2013). The primary energy factor is set to 48 and 18.7 MJ/kg N and P, respectively (Börjesson et al., 2010).

Spreading of mineral fertilisers also results in direct and indirect emissions of N_2O. Direct emissions are set to 1% of N-tot and indirect emissions are set to 1% of the nitrogen lost as NH_3-N. Based on Naturvårdsverket (2013), emissions of NH_3-N are set to 0.91% of N.

Soil organic carbon

Digestate as well as manure contain not only nutrients but also carbon of which some may increase the amount of carbon in the soil in a long term perspective, acting as a carbon sink. Here, it is assumed that 29% of the carbon in the digestate forms long-term, stable soil organic carbon. For manure, the corresponding factor is set to 23%. For more background and assumptions, see Björnsson et al. (2016) and Lantz and Björnsson (2016).

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