MS Thesis

Environment and Natural Resources

Extended Cost-Benefit Analysis of Maritime Fuel

Comparison of Heavy Fuel Oil and Methanol in Iceland

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May 2019

HÁSKÓLI ÍSLANDS

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Rafn Helgason

30 ECTS thesis submitted in partial fulfilment of a *Magister Scientiarium* Degree in Environment and Natural Resources

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Faculty of Economics

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Preface

This is a 30 ECTS thesis in partial fulfilment of a MS degree in Environment and Natural Resources submitted to the Faculty of Economics at the University of Iceland. The primary advisor for this thesis was Dr. Brynhildur Davíðsdóttir, and the secondary advisor was Dr. David Cook.

Abstract

The energy and emissions profile of the maritime industry needs significant adjustments to comply with future maritime regulation. Therefore, alternative fuels have been proposed to meet increasingly higher emission standards, proposed by the International Maritime Organization. Furthermore, the Icelandic government aims to introduce 10% renewable energy into the maritime sector before 2030 as well as eventually phase out heavy fuel oil. This thesis assesses the economic feasibility of introducing methanol into the maritime sector using the method of extended cost-benefit analysis, i.e., adding the economic implications of maritime emissions to the cost side of a cost-benefit analysis. The life-cycle analysis of conventional methanol, renewable methanol and heavy fuel oil are compared and evaluated via dynamic economic evaluation of estimated market prices between 2018 and 2050, as well as the economic implications of the environmental impacts caused by the fuels. Therefore, the purely economic and environmental trade-off between the fuels is shown. Methanol is assessed both in terms of a conventional production pathway but also a pathway that can be considered renewable, i.e., electrolysis and carbon capture and utilization, to assess the prospects of meeting the 10% renewable goal within Iceland's maritime sector, with methanol. In the purely economic sense, heavy fuel oil is the most feasible option. However, when the economic implications of the environmental impacts are added to the cost of the fuels, conventional methanol becomes cost-competitive in 2018 and becomes the most feasible option in 2050 in three fuel trajectories. Renewable methanol is the most expensive option in all trajectories. Relative cost increase in a transition from HFO to renewable methanol ranges from 215 to 375% of HFO's total cost in 2018, however, in the most favorable conditions for renewable methanol, in 2050, relative cost increase of said transition goes down to 109,5% of HFO's total cost, in HFO's least favorable conditions. Therefore, renewable methanol will unlikely become cost-competitive before 2050 and is unlikely, without financial instruments, to contribute to the 10% renewable energy goal in the maritime sector before 2030.

Ágrip

Eldsneytismál skipaiðnaðarins á Íslandi munu þurfa að breytast til muna á næstu árum til þess að mæta nýjum lögum og reglum, ákvæðum og markmiðum. Þess vegna hafa nýir orkugjafar verið nefndir, til þess að mæta hertri löggjöf um losun mengandi efna og magn þeirra frá skipum. Þar að auki hefur íslenska ríkið sett sér markmið um að hlutdeild endurnýjanlegrar orku í skipaiðnaðinum verði 10% fyrir árið 2030, ásamt því að hætta notkun svartolíu endanlega, á óskilgreindum tímapunkti. Í þessari ritgerð er efnahagsleg hagkvæmni metanóls og innleiðing þess metinn út frá útvíkkaðri kostnaðar og ábatagreiningu, þ.e., að bæta við kostnaðarhlið greiningarinnar efnahagslegum áhrifum mengunar. Umhverfisáhrif hefðbundins metanóls, endurnýjanlegs metanóls og svartolíu eru fengin frá lífsferilsgreiningum, síðan eru þau þýdd yfir í hægrænar stærðir og metin í samhengi við markaðsvirði eldsneytis frá 2018 til 2050. Með þessum samanburði er fórnarkostnaður settur í samhengi við umhverfisleg áhrif annarsvegar og hagræn áhrif hins vegar. Metanól er rannsakað út frá tveimur mismunandi framleiðsluaðferðum. Annars vegar hefðbundin framleiðsla frá jarðefnaeldsneyti og hins vegar er metanól framleitt úr fönguðum koltvísýringi frá jarðvarmavirkjun og með rafgreiningu. Tilgangur rannsóknarinnar er að leggja mat á hagkvæmni þess að mæta markmiði Íslendinga um að auka hlutdeild endurnýjanlegrar orku í 10% fyrir 2030, með metanóli frá íslenskum aðföngum. Niðurstöðurnar benda til þess að þegar þessar þrjár gerðir metanóls eru bornar saman, og umhverfisáhrif eru ekki tekin til greina í kostnaðar- og ábatagreingingunni er svartolía hagkvæmasti kosturinn. Hins vegar, þegar efnahagslegum áhrifum umhverfisáhrifa er bætt við kostnað eldsneytisins, er hefðbundið metanól þegar samkeppnishæft árið 2018, og verður enn hagkvæmasti kosturinn árið 2050. Endurnýjanlegt metanól er ætíð óhagkvæmasti kosturinn í öllum tilvikum. Aukinn kostnaður við að skipta úr svartolíu yfir í endurnýjanlegt metanól er á bilinu 215 til 375% af verði svartolíu árið 2018. Hins vegar, árið 2050, í því tilviki sem er hagstæðast fyrir endurnýjanlegt metanól en óhagstæðast svartolíu fer kostnaðaraukningin niður í 109,5% af verði svartolíu. Niðurstöður benda því til að það sé ólíklegt að endurnýjanlegt metanól verði samkeppnishæft fyrir árið 2030, þrátt fyrir að mögulegt sé að framleiða nóg metanól til að ná 10% markmiði íslenska ríkisins.

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Abbreviations and Acronyms

CAFE	Clean Air for Europe Program
CBA	Cost-benefit analysis
CCU	Carbon Capture and Utilization
CRI	Carbon Recycling International
CO_2eq	Carbon Dioxide Equivalents
DME	Dimethyl ether
EC	Elemental Carbon
ECA	Emission Control Area
ETL	Emission-to-liquid
EGCS	Exhaust Gas Cleaning Systems
EIA	U.S. Energy Information Administration
EU ETS	European Union's Emission Trading Scheme
GDP	Gross Domestic Product
GWP	Global Warming Potential
GHG	Greenhouse gases
HEATCO	Harmonized European Approaches for Transport Costing
HHV	Higher-Heating Value
HFO	Heavy Fuel Oil
ICE	Internal Combustion Engine
IEA	International Energy Agency
IMO	International Maritime Organization
IPA	Impact Pathway Analysis
LBG	Liquified Biogas
LCA	Life-cycle assessment
LH_2	Liquid Hydrogen

LHV	Lower-heating value
LNG	Liquified Natural Gas
LSMGO	Low Sulfur Marine Gas Oil
MARPOL	International Convention on the Prevention of Pollution from Ships
MDO	Marine Diesel Oil
MGO	Marine Gasoil
MEPC	Marine Environment Protection Committee
MeOH	Methanol
NEA	National Energy Authority
NG	Natural Gas
NMVOC	Non-Methane Volatile Organic Compounds
NO_x	Nitrogen Oxides
OCM	Organic Carbon Matter
PM	Particulate matter
SVE	Straight Vegetable Oil
SO_x	Sulfur Oxides
TTP	Tank to Propeller
UNFCCC	United Nations Framework Convention on Climate Change
VOC	Volatile Organic Compounds
WHO	World Health Organization
WTP	Well to Propeller
WTT	Well to Tank

1 Introduction

Increasing attention has been given, in recent years, to ameliorating harmful externalities of ship propulsion, namely eutrophication, climate change, ocean acidification and human health impacts (Brynolf, Fridell & Andersson, 2014; MEPC, 2008). This is facilitated via stricter environmental policies and regulations. These new emission regulations, set by the International Maritime Organization (IMO), address the quality of fuel, specifically the exhaust emissions produced by fuel combustion of ships. Several alternative maritime fuels have been proposed as substitutes for conventional fuels, namely liquified natural gas (LNG), liquified biogas (LBG), methanol (Brynolf, Fridell & Andersson, 2014), dimethyl ether (DME) (Goeppert, Olah, & Prakash, 2009), liquid hydrogen (LH_2), straight vegetable oil (SVE) or biodiesel (Gilbert, Walsh, Traut, Kesieme, Pazouki & Murphy, 2018). Alternatively, exhaust gas cleaning systems (EGCS) can be utilized, to meet the newest maritime emission regulations, set by the IMO, which are scheduled to take effect in various parts of the world in the next several years (Smith et al., 2015).

To facilitate a transition towards a more sustainable maritime sector, different technologies will have to be investigated, both alternative fuel types and EGCS. Knowledge and evaluation of all stages of the fuel supply chain, i.e., from feedstock to combustion properties, is a major step of this development and will offer support for decision- and policy makers, business owners and administrators (Ellis & Svanberg, 2018). Moreover, the economic implications of the environmental impacts of fuels, can influence a transition towards a new fuel paradigm in the maritime sector, if they are measured.

This thesis addresses the economic implications, including the cost of the environmental impacts, of three maritime fuels, heavy fuel oil (HFO) renewable methanol (CRI methanol) and conventional methanol (NG methanol). CRI methanol is a non-fossil fuel methanol production pathway and will be assessed in this thesis to offer input on the renewable energy initiative of the maritime sector in Iceland. This production pathway of methanol is based on electrolyzing water for hydrogen production, with renewable electricity and carbon capture and utilization from an effluent waste stream of a nearby geothermal powerplant. The company that owns this *emission-to-liquid* (ETL) methanol plant is called Carbon Recycling International (CRI) and is based in Svartsengi, Iceland.

The environmental impacts are found in a life-cycle analysis of the three fuel types, which are subsequently translated into economic cost values to extend a conventional cost-benefit analysis (CBA). Furthermore, these external costs are integrated into the CBA of all fuel types to represent both the purely economic and the environmental trade-off between the fuels. The fuel types are then compared in the context of energy security, production capacity and the regulatory and policy framework of Iceland, a nation highly dependent on its maritime resources.

Drawing inspiration from the aspirations of Arthur C. Pigou's *Economics of Welfare* from the 1920's and Rachael Carson's *Silent Spring* in 1962, this thesis seeks to add to the academic literature on the emergence of pollution externalities in fuel appraisal. Moreover, recognition of these costs is to ameliorate market failures, which are likely to lead to economically inefficient and sub-optimal outcomes, thus justifying the estimation of environmental costs and their inclusion in fuel appraisal.

The remainder of this section is devoted to background information, namely, the demarcation of Iceland‘s maritime fuel paradigm, it‘s environmental challenges, regulations and policies in a global and national context, along with a brief background of the economic theory utilized in this research and the aim of this study. Finally, an overview of the subsequent chapters of this thesis are outlined.

1.1 Regulatory framework

Globally, maritime energy utilization is predominantly fossil fuel based. The following paragraph is adopted from the *Third IMO Greenhouse Gas Study 2014* by (Smith et al., 2015). On average, the international shipping industry consumed approximately 201 million tonnes and 272 million tonnes, p.a., dependent on the measuring approach, i.e., consumption defined as the allocation of fuel to international voyages or fuel consumed by vessels in the context of international shipping (Smith et al., 2015). When all ships looked at by the IMO's third greenhouse gas study, maritime consumption amounted to between 247 and 325 million tonnes p.a., in the period 2007-2012 (Smith et al., 2015). Global trends suggest, that in the period 2012 to 2050, maritime carbon dioxide (CO_2)

emissions will increase by 50% to 250%, dependent on economic- and energy development. When estimating global fuel consumption and emissions of the international maritime sector, three main categories are assessed, fishing, international- and domestic navigation.

International marine bunkers emitted 682,35 million tons of CO_2 in 2016 or 2,1% of global anthropogenic CO_2 emission. The relative increase of emissions in this sector between 1990 and 2016, globally and in Iceland, is 83,6% and 85,6%, respectively (IEA, 2018a).

In 2016, the Icelandic maritime fleet consumed approximately 229,9 kilotons (kt) of fossil fuel, of which 50,6% was marine gasoil (MGO), 26,8% marine diesel oil (MDO) and 22,6% heavy fuel oil (HFO) (Baldursson, 2017; Hellsing et al., 2018). The sector is responsible for approximately 19% of the nation‘s total greenhouse gas emissions that can be counted towards the reduction targets of the EU’s effort sharing scheme. Therefore, the maritime sector, namely domestic navigation and the fishing fleet, are fields which policy makers need to consider in order to meet the nation‘s goal to reduce carbon emissions by 40% before 2030, compared to 2005 levels, and the Carbon Neutrality goal, set for 2040 (Ministry of Environment and Natural Resources, 2018).

In a parliamentary resolution, accepted the 31st of May in 2017, the Icelandic government set a sector-specific goal, to go from 0,1% to 10% utilization rate of renewable energy in sea-related industry, by 2030 (Parliamentary document no. 1002/2016-2017. Parliamentary resolution on Iceland‘s Energy Transition Action Plan; National Energy Authority, n.d.). This parliamentary resolution facilitated the production of Iceland‘s Climate Action Plan 2018 – 2030, which was published in its first version 10th of September, 2018. The 17th action of the Action Plan directly addresses the future of Iceland’s maritime sector by aiming to permanently phase out HFO in Iceland‘s territorial waters, with laws and/or regulations (Ministry of Environment and Natural Resources, 2018). This procedure is currently being developed by the Ministry of Environment and Natural Resources, the Ministry of Transport and Local Government and the Ministry of Industries and Innovation. Furthermore, the Icelandic maritime sector is directly influenced by laws set by the IMO. The revised MARPOL Annex VI of the IMO will reduce the global sulfur content cap in maritime fuel from 3,5% to 0,5%, effective from the 1st of January, 2020 (IMO, 2019). This will directly affect the Icelandic maritime sector as 22,6% of its fuel consumption, i.e., the HFO, has 1,9% sulfur content on average (Baldursson, 2017). Therefore, direct action is required in Iceland, in terms of maritime energy utilization, across five regulatory and policy aspects:

1. Before 2020, to meet the 0,5% global sulfur cap;
2. Eventually phase out heavy fuel oil in accordance with the nation's Climate Action Plan 2018 – 2030;
3. Before 2030, to meet the nation's carbon emission reduction target of 40%, compared to 1990 levels;
4. Before 2030, to meet the nation's 10% maritime renewable energy objective;
5. Before 2040, to meet the nation's overall carbon neutrality goal.

1.1.1 Global maritime emissions challenges and regulation

Reducing greenhouse gas (GHG) emission is one of the fundamental aim of the Paris Agreement, which aims to keep the average global temperature rise below 2°C above pre-industrial levels and preferably limiting said increase below 1,5°C (United Nations, 2015). The agreement was adopted in December 2015, for countries under the United Nations Framework Convention on Climate Change (UNFCCC) (United Nations, 2015). However, a substantial portion of the maritime sector, namely international shipping, is not directly included in the Paris Agreement. Between 1990 and 2016, CO_2 emissions, from international marine bunkers, grew by approximately 83% (IEA, 2018a), see figure 1.1.

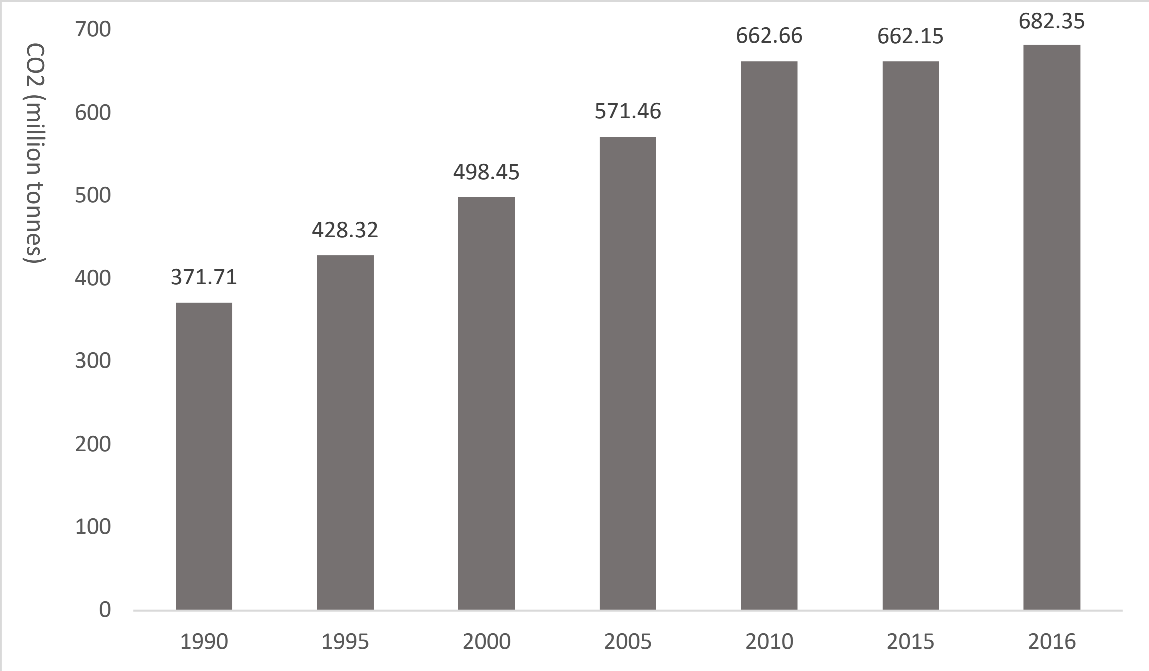


Figure 1-1 Emissions of CO_2 (million tons) from International Marine Bunkers between 1990 and 2016 (IEA, 2018a).

The challenge of mitigating the maritime sector‘s GHG emission is primarily in the hands of the IMO which is a United Nations agency, e.g., responsible for International Convention on the Prevention of Pollution from Ships, also known as MARPOL. This convention is partially responsible for addressing externalities related to growth in international shipping, which represents as much as 90% of all worldwide shipping, in terms of volume. For the period 2007-2012, global shipping accounted for approximately 2,8% of global annual greenhouse gases in terms of CO_2 equivalents (CO_2eq), based on a 100-year global warming potential (GWP) conversion from the Fifth Assessment Report (AR5). Moreover, the shipping industry, contributes 12% and 13% of global anthropogenic SO_x and NO_x emissions, respectively (Smith et al., 2015). Substantial concern has been raised due to the harmful effects of those pollutants to the health of humans and ecosystems. Specifically, emissions of sulfur- and nitrogen compounds are the precursors of acidifying compounds such as sulfuric and nitric acid. Therefore, they induce acidification as they reach soil or water which is one of the driving mechanisms behind ecosystem degradation, both in terrestrial and aquatic ecosystems. Moreover, NO_x is a precursor for ground-level ozone formation and is causally linked to eutrophication which has effect on species richness through the degradation of water quality. NO_x is also linked to respiratory illnesses in humans, in high concentration (OECD, 2013).

MARPOL, which is designed to reduce pollution from ships, both in standard operation and in the event of accidents, consists of six technical annexes. Most relevant to maritime fuel consumption and pollution is Annex VI, „*Regulations for the Prevention of Air Pollution from Ships*“, which first entered into force May 19th, 2005 (IMO, 2019). Annex VI imposes restrictions on the amount of main air pollutants in the exhaust gas of ships, including sulfur oxides (SO_x) and nitrogen oxides (NO_x). Since the time that MARPOL‘s Annex VI entered into force, it has been revised several times by the Marine Environment Protection Committee (MEPC), which at its 53rd sessions in July 2005, committed to revising the Annex by significantly strengthening the upper limits of emissions streams, especially with the emergence of technological improvements in the maritime sector. In 2008, at MEPC‘s 58th sessions, Annex VI was adopted in its revised version with the NO_x Technical Code 2008. The main changes were aimed at a progressive global reduction of three emission types, SO_x, NO_x and particulate matter (PM). Along with the revision, emission control areas (ECAs), were created to reduce said pollutant types even further based on significant marine traffic in sensitive areas, see figure 1.2. Furthermore, the EU has mandated, irrespective of IMOs decisions, to limit sulfur content in EU waters to 0,5% m/m from 2020, as can be seen in green in figure 1.2. The EU has their own directive regarding sulfur content of the maritime sector, Directive 2005/33/EC, which has limited sulfur content in harbor regions to 0,1% m/m since 2010 (Moirangthem & Baxter, 2016).

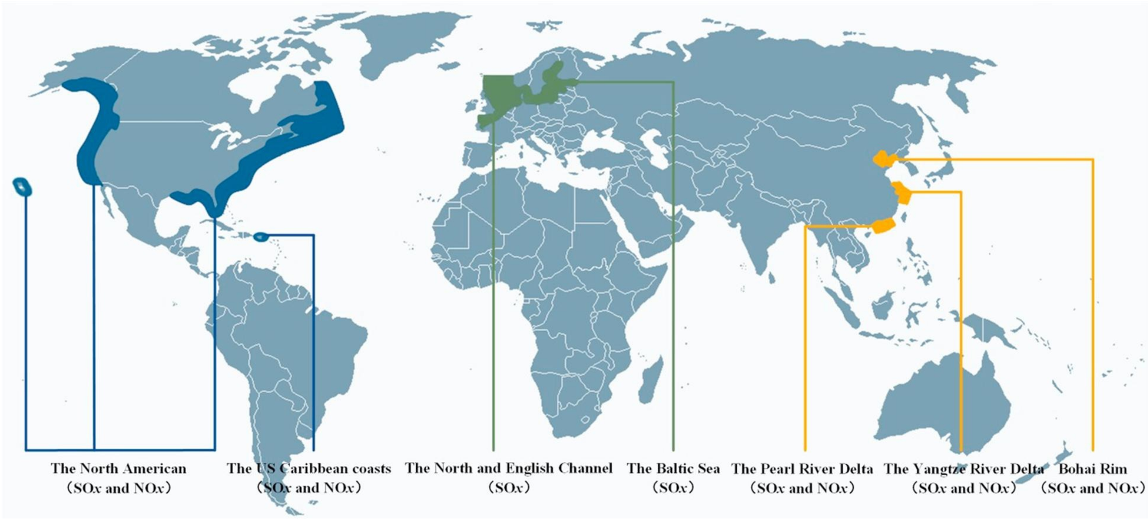


Figure 1-2 Map of emission control areas, pertaining to NOx and SOx emission from maritime vessels (Zhen, Li, Hu, Lv & Zhao, 2018).

In the 2008 revision, the global SO_x cap is reduced from 3,5% to 0,5% m/m (mass by mass), effective from 1 January 2020, while the cap in ECAs became 0,1% m/m, both for SO_x and PM. The cap for NO_x was imposed in three tiers, contingent to the construction year of the vessel, i.e., tier I, II and III for ships constructed prior to 2000, 2011 and 2016, respectively (IMO, 2019). The total weighted NO_x cycle emission limit, in terms of g/kWh, is relative to the engine's rated speed (rpm) (MEPC, 2008), see table 1.1.

Table 1-1 NO_x Technical Code control requirements of Annex VI, applicable to marine diesel engines of over 130 kW output power, excluding vessels utilized for emergency purposes. Tier III only applies to the specified vessels when operating in ECAs (MEPC, 2008).

Tier	Ship construction date on or after	Total weighted cycle emission limit (g/kWh) n = engine's rated speed (rpm)		
		n < 130	n = 130 - 1999	n ≥ 2000
I	1 January 2000	17.0	$45 \cdot n^{(-0.2)}$ e.g., 720 rpm – 12.1	9.8
II	1 January 2011	14.4	$44 \cdot n^{(-0.23)}$ e.g., 720 rpm – 9.7	7.7
III	1 January 2016	3.4	$9 \cdot n^{(-0.2)}$ e.g., 720 rpm – 2.4	2.0

1.1.2 Icelandic maritime emissions challenges

Some of the most relevant figures of Iceland‘s GHG accounting are from the *National Inventory Report Emissions of Greenhouse Gases 1990 – 2016* (NIR), published by The Environmental Agency of Iceland in 2018 and submitted under the United Nations Framework Convention on Climate Change and the Kyoto Protocol (Hellsing et al., 2018). According to the report, the maritime sector, is divided in to three categories, i.e., fishing (CRF 1A4ciii), international navigation (CRF 1D1b) and domestic navigation (CRF 1A3d). The report uses reported fuel use quantities based on sales of fuel from suppliers. When discerning between international and domestic navigation, the harbor to which the ship is sailing to is the determining factor, i.e., ships sailing to Icelandic harbors are counted as domestic and vice versa, regardless of country of origin. Maritime vessels that come to Icelandic harbors but do not buy fuel here, do not appear in the nation‘s emissions inventory. Moreover, CRF 1D1b, is not counted in the nation‘s inventory in the context of the nation‘s international commitments towards reduction targets. Aggregated data on fuel consumption and emissions levels, of all three maritime divisions, can be seen in figure 1.3.

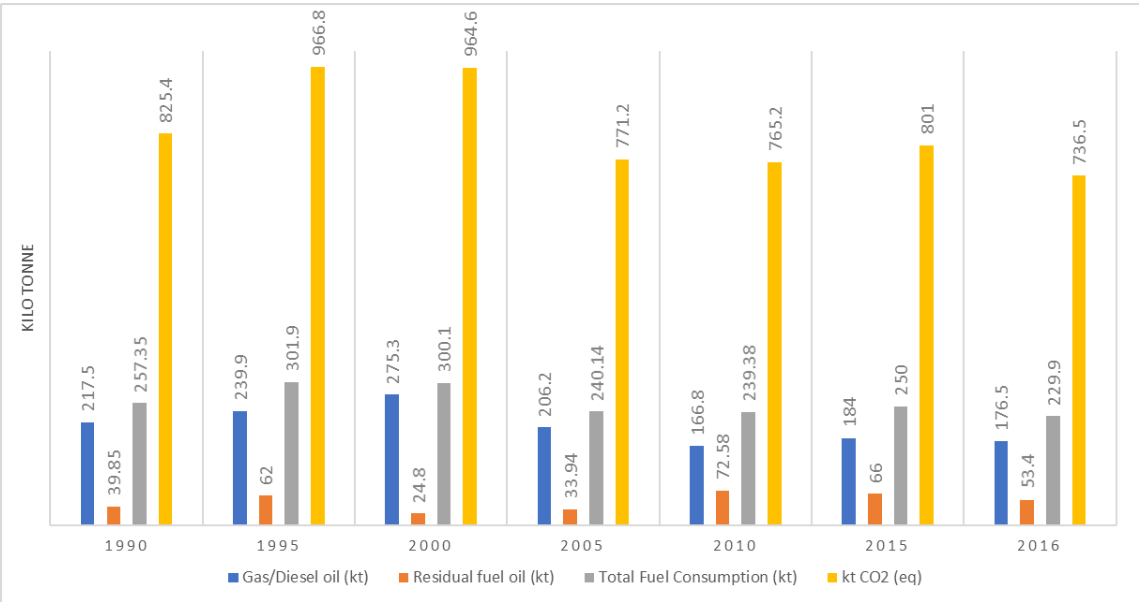


Figure 1-3 Aggregated fuel consumption and emissions of the Icelandic maritime sector (1D1b, 1A3d and 1A4ciii), 1990 – 2016, information from (Hellsing et al, 2018).

The most recent year, of GHG data of the maritime sector in the NIR, is 2016. That year the largest source of GHG emission, originating from the maritime industry was the fishing fleet, followed by international navigation and domestic navigation, 71%, 25% and 4%, respectively. Aggregated GHG CO_2eq emissions from the sector are approximately 736,5 kt, stemming from the utilization of approximately 229,9 kt of fuel (Hellsing et al., 2018), see table 1.2.

Table 1-2 Fuel use (kt) and resulting emissions (CO_2eq) of the Icelandic maritime sector in 2016, adopted from the National Inventory Report 2018 (Hellsing et al, 2018).

A	*Fishing Fleet (1A4ciii)*	
	Gas/Diesel Oil (kt)	133.6
	Residual Fuel Oil (kt)	29
	Emissions (kt CO2eq)	521.5
	Domestic Navigation (1A3d)	
	Gas/Diesel Oil (kt)	8.5
	Residual Fuel Oil (kt)	0.2
	Emissions (kt CO2eq)	28
	Total fuel (kt)	**171.3**
	Total Emissions (kt CO2eq)	**549.5**
B	*International Navigation (1D1b)*	
	Gas/Diesel Oil (kt)	34.4
	Residual Fuel Oil (kt)	24.2
	Emissions (kt CO2eq)	187
	Total fuel (kt)	**58.6**
	Total Emissions (kt CO2eq)	**187**
A+B	*Aggregate Maritime Fuel/Emissions*	
	Gas/Diesel Oil (kt)	176.5
	Residual Fuel Oil (kt)	53.4
	Emissions (kt CO2eq)	736.5
	Total fuel (kt)	229.9
	Total Emissions (kt CO2eq)	736.5

In the context of the nation‘s international commitments on GHG reduction, 1A3d and 1A4ciii, section A of the table above, will be the only measures that the nation can ameliorate to work towards those targets. In 2016, GHG emissions of those two sectors, i.e., domestic navigation and fishing, accounted for approximately 28 kt and 521,5 kt of CO_2eq, respectively (Hellsing et al., 2018). To put this into the context of the nation‘s targets, the total emissions, that can be counted towards the reduction targets, i.e., emissions that do not fall inside the European Union Emissions Trading Scheme (EU ETS) or Land Use, Land-use Change and Forestry (LULUCF), were 2.888 kt of CO_2eq, in 2016 (Ministry of Environment and Natural Resources, 2018). Therefore, the maritime emissions, that the Icelandic nation can count towards international carbon reduction targets, amounted to 19% of total reduceable emissions within the EU's effort sharing emission scheme. Subsequently, 1A3d and 1A4ciii are especially worthwhile avenues for change, if Iceland is to meet its carbon reduction targets, on time.

1.2 Economic evaluation background

The sustainability challenge at hand is the amelioration of maritime pollution and its effect on the environment and public health. According to the Lancet Commission on Pollution and Health, pollution is the greatest environmental cause of disease and premature death in the world and is said to have been responsible for 16% of premature deaths globally, in 2015 (The Lancet, 2018). Moreover, pollution-related disease is causally linked to up to 2% loss in gross domestic product (GDP), in low-income to middle-income countries. Welfare losses, induced by pollution, are estimated to be $4,6 trillion p.a. i.e., 6,2% of the global economic output and are expected to increase as additional links between pollution and disease are established (The Lancet, 2018). The impetus of this economic loss, imposed on the global society, is often neglected as market prices fail to communicate the economic cost of negative externalities. Moreover, this emphasizes that pollution control and economic prosperity are not as mutually exclusive as one might conclude. In this paper, the pollution of three maritime fuels is assessed in this context, i.e., comparison of HFO, MeOH CRI and MeOH NG via extended CBA which tries to effectively capture this aspect of the welfare paradigm, i.e., neglected pollution externalities.

Reformation of conventional economic theory can be feasible to facilitate an economic paradigm that is more aligned with the Sustainable Development Goals of the United Nations General Assembly 2030 Agenda. One such economic sub-discipline is environmental economics, which aims to incorporate the cost of the environment, in the price of goods, services and decision-support tools, based around the marginal social damage associated with its negative externalities (Nadeau, 2007; Bergh & Jeroen, 2000). An externality is an effect, imposed on a third party, which is outside the economic system but inside the natural and social system i.e., the production or consumption of one economic actor affects another who did not participate in said transaction and is, regardless of their non-involvement, affected by it (Nadeau, 2007). By effectively incorporating these externalities into appraisals we arrive closer to the socially optimal level of production (Centemeri, 2009), with what is referred to as *extended CBA,* hereafter.

In its theoretical form, CBA seeks to compare the aggregate benefits with aggregate costs, of a specific system, development project or policy initiative, across the estimated lifespan of the project. Therefore, this CBA will be based upon the market prices of the fuel types. However, this CBA will be extended, to capture the economic implications of the environmental externalities, associated with the whole lifecycle of the three fuels (Brynolf, Fridell & Andersson, 2014). In theory, this approach assumes that environmental goods and human-made goods are interchangeable and can be substituted for one another (Beder, 2011). Consequently, the cost of externalities will be added to the other costs of the CBA. For those externalities to be included in the CBA, their effect on the ecosystem and public health, must be converted into monetary values.

By extending the CBA, a more extensive perspective of welfare gains and losses is acquired. Therefore, environmental resources can be allocated more efficiently based on the externalities of the fuels. Moreover, the environment can be valued, where a monetary metric reveals the environment's inherent scarcity, in this case, a cleaner atmosphere and nature's ability to regulate human-induced degradation of the environment and the ability to regulate a healthy ecosystem. Therefore, the main aspect of the extension is to capture all costs and benefits of the fuel's lifecycle, including, but not limited to, degradation of ecosystem services and degradation of public-health. Subsequently, this extended CBA seeks to incorporate the economic costs of the environmental degradation into maritime fuel appraisal.

1.2.1 Application of environmental costing

Scientific studies have shown that climate change and atmospheric air pollutants can be detrimental for ecosystem vitality and public health. Consequently, assuming a price for these changes is of importance for the emergence of more accurate market signals. To do this, connections must be made between marginal increases in specific air pollutants and coinciding changes in environmental quality and public health damages, where there is a causal relationship. Particulate matter (PM) has been found to have adverse impact on public health (Schlesinger, Kunzli, Hidy, Gotschi & Jerrett, 2006). Certain chemical constituents of $PM_{2,5}$, namely elemental carbon (EC) and organic carbon matter (OCM) have been linked to increased mortality and morbidity through cardiovascular and respiratory diseases (Peng et al., 2009). Moreover, particulate matter has been found to be a risk factor for infant mortality (Woodruff, Darrow & Parker, 2008). These and commensurable effects can be converted into a monetary value through public health care cost via e.g., the statistical value of a human life and the value of a life years lost, see table 1.3.

Table 1-3 Summary of public-health valuation data (Maibach et al., 2008)

<table>
<tr><th>Mortality</th><th colspan="2">Based on median values (€)</th><th colspan="2">Based on mean values (€)</th></tr>
<tr><td>Infant mortality</td><td colspan="2">1,500,000/death</td><td colspan="2">4,000,000/death</td></tr>
<tr><td>Value of statistical life</td><td colspan="2">980,000/death</td><td colspan="2">2,000,000/death</td></tr>
<tr><td>Value of a life year</td><td colspan="2">52,000/death</td><td colspan="2">120,000/year</td></tr>
<tr><th>Morbidity</th><th>Low (€)</th><th colspan="2">Central (€)</th><th>High (€)</th></tr>
<tr><td>Chronic bronchitis</td><td>120,000/case</td><td colspan="2">190,000/case</td><td>250,000/case</td></tr>
<tr><td>Respiratory, cardiac hospital admission</td><td></td><td colspan="2">2,000/admission</td><td></td></tr>
<tr><td>Consultations with primary care physicians</td><td></td><td colspan="2">53/consultation</td><td></td></tr>
<tr><td>Restricted activity day (day when person needs to stay in bed</td><td></td><td colspan="2">130/day</td><td></td></tr>
<tr><td>Restricted activity day (adjusted)</td><td></td><td colspan="2">83/day</td><td></td></tr>
<tr><td>Minor restricted activity day</td><td></td><td colspan="2">38/day</td><td></td></tr>
<tr><td>Use of respiratory medication</td><td></td><td colspan="2">1/day</td><td></td></tr>
<tr><td>Symptom days</td><td></td><td colspan="2">38/day</td><td></td></tr>
</table>

Evaluation of systematic variables, i.e., discount rate, value of a human life and value of an ecosystem service are subject to change. Furthermore, adjustments to environmental cost calculations fluctuate based on the World Health Organization's (WHO) findings of potential damage of pollutants (Bruyn et al., 2018). Therefore, there is substantial uncertainty related to science and economics of environmental- and public health. One of the more well-known social cost calculation of an atmospheric air pollutant is The Interagency Working Group on the Social Cost of Greenhouse Gases' social cost of CO_2 (SC-CO_2). This metric is constructed to include, but not limited to, changes in human health, property damages, the value of ecosystem services and agricultural outputs (Interagency Working Group on the Social Cost of Greenhouse Gases, 2016). However extensive as this list may seem, the IPCC Fifth Assessment Report (AR5) pointed out that the models used to calculate SC-CO_2 omit certain impacts that are likely to increase the current price estimate of SC-CO_2 (EPA, 2017). Furthermore, some parameters of the effects of climate change are relatively straight-forward when it comes to the monetization process, e.g., cooling energy demand while climate sensitivity and agricultural productivity are subject to extensive research effort (Anthoff & Tol, 2013). Moreover, some parameters are more relevant in the short term while others are in the long term. Subsequently, the rate of time preference via discount rate plays a vital role in the economic evaluation of the social cost of air pollutants that affect climate change.

1.2.2 Accounting for externalities in maritime fuel appraisal

Extended CBA in maritime fuel appraisal can be used as a decision-support tool and may have the potential to clarify and discern between the economic feasibility of different fuel options including their environmental implications. Moreover, if emissions from maritime applications are not seen as costs within CBAs, there is little incentive for maritime firms to mitigate environmental externalities by investing in alternative energy carriers or EGCS.

In terms of maritime externalities, international ship traffic constitutes a major impact on human health. It has been estimated to induce external cost in Europe of 58,4 billion €/year in 2000 and is expected to increase to 64,1 billion €/year in 2020, constituting 7% and 12% of total health effects due to air pollutants in Europe (Brandt et al., 2011). A study by Tzannatos (2011) on maritime fuel externality cost in Greece, showed that in 2008, international and domestic shipping generated 7,4 million tonnes of CO_2, SO_2, NO_x, and PM of which 7 million tonnes were CO_2. The aggregated externality cost for those emissions was 2,95 billion €, that year, of which 155 million € was for the CO_2, i.e., 5% of total emission externality cost calculated in the study (Tzannatos, 2011).

Literature on pollution's externality cost is burgeoning. However, as it relates to maritime emissions, certain complications exist, namely the location of certain exhaust emissions relative to residential areas, i.e., emissions in port, emissions close to shore and offshore emissions (Holland & Watkiss, 2004). Furthermore, effects of pollutants have been researched less extensively, in the context of maritime emissions externality cost, compared to road transport. Moreover, the maritime emission externality publications are focused on the following pollutants for externality cost estimations: SO_2, $NO_{x,}$ PM, $PM_{2,5}$, VOCs and CO_2 (Holland & Watkiss, 2004; Maibach et al., 2008; Jiang, Kronbak & Christensen, 2010; Tzannatos, 2011; Kotowska, 2017; Holland, Pye, Watkiss, Franke & Bickel, 2005). In the abovementioned studies, cost estimates of pollutants are given in different contexts, contingent on spatial distribution of pollutants in relation to populated areas. Therefore, most of the studies differentiate between economic implications of pollutants when they are dispersed *at sea* and *in port.* Holland & Watkiss (2004) take it further by adding the dimension of *close to shore.*

In order to bring forth a representative picture of cost estimates for pollutants of the maritime sector in Iceland, the open sea estimates are likely to be more relevant as many of the port estimates are based upon the adjacency of a marine vessel in a port located in a city, populated with 100.000 people at minimum. Therefore, *in port* emission cost factors, would only apply when vessels are in port in Reykjavík, the capital of Iceland, the only city in Iceland with a population of over 100.000 people. Cost estimates for pollution factors in the maritime context can be seen in figure 1.4.

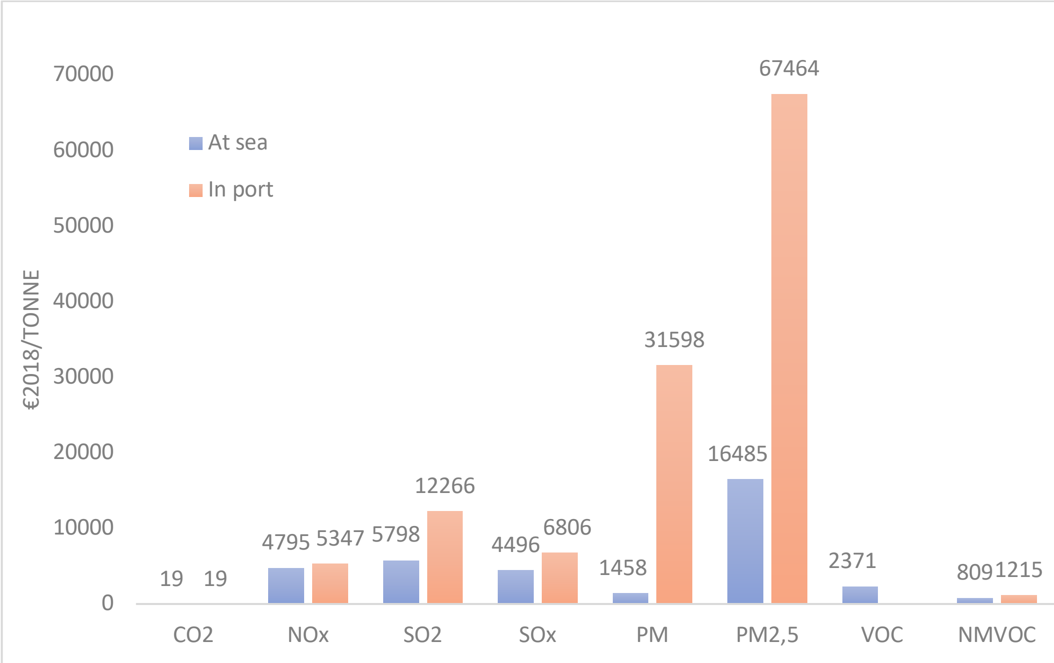


Figure 1-4 Average cost of maritime emissions (€2018/tonne) from Kotowska (2017), Holland & Watkiss (2004), Jiang, Kronbak & Christensen (2010), Maibach et al., (2008) & EU ETS, diffrentiated between emissions in port and emission at sea and adjusted via inflation rate to 2018 values.

The cost estimates in this table are differentiated in terms of pollutant factors and proximity to port cities in Europe, i.e., at sea or in port. These cost estimates are multiplied with the amounts of each pollutant, relative to the functional unit used in the methodology chapter. Not all cost estimates are relatable to the LCA (Brynolf, Fridell & Andersson, 2014), therefore, only the cost of pollutants that were both in the LCA and the maritime externality literature, will be used for a final cost estimate of externalities in the fuel appraisal.

1.3 Aim of the study

This thesis has three objectives;

- To assess the value of environmental externalities of HFO, CRI methanol and NG methanol in the maritime context;
- To compare, based on the value of environmental externalities and fuel prices, the economic feasibility of three fuel types;
- To facilitate further discussion and inform the debate in Iceland about the merits of different fuel types given the policy context demanding a transition from HFO to alternatives.

The primary research question of this thesis is:

How cost-competitive are CRI's methanol and NG methanol compared to HFO, when cost of externalities, associated with the life-cycle emissions of all fuels is incorporated into the fuel price?

To assess the cost-competitiveness, a secondary research question will be answered in this paper:

What is the aggregated cost of pollution, associated with the different fuel types, measured in €/ton-km (tkm), of all fuels?

To assess the base fuel prices, a third research question needs to be answered to fulfill both CBAs:

What are the approximate market prices of CRI's methanol, NG methanol and HFO?

To assess the production capacity of CRI's methanol in Iceland, a fourth research question needs to be answered to address physical constraints:

How much effluent CO_2 and electricity is needed to produce enough methanol to meet Iceland's 10% renewable energy goal in the maritime sector?

1.4 Thesis structure

The structure of the thesis is as follows: Section 2 provide background information on the fuels analyzed in this thesis. CRI methanol will be analyzed more extensively, namely in terms of production capacity, to facilitate a discussion on the merits of meeting the 10% renewable energy goal in Iceland's maritime sector with this fuel option. Section 3 outlines the methodology and data sources. First, the externality costs and their relationship to the LCA are assessed. Second, the price of fuels, both today's prices and prices towards 2050 are assessed. Section 4 outlines results in terms of three fuel price trajectories and three externality cost estimates. Section 5 contains a discussion on the main results as well as study limitations and practical- and policy implications. Section 6 provides a conclusion.

2 Fuel background

This chapter is devoted to background information of the fuels, heavy fuel oil (HFO) and methanol, in the maritime context as well as the two production pathways for methanol. Methanol will be assessed extensively, both in terms of the current production method in Iceland, i.e., CRI, and in terms of utilization as a maritime fuel. Moreover, the production capacity of methanol in Iceland, in the context of CRI's production methods, i.e., electricity demand and carbon capture potential, is assessed.

2.1 Heavy Fuel Oil

Relatively low cost and widely available, HFO gave rise to a vast period of extensive growth in international shipping. Maritime shipping is one of the most energy efficient freight options available, with a CO_2 footprint, per tonne-kilometer (tkm) of 10-15 CO_2. For reference, rail-, trucking- and aviation freight emit 19-41, 51-91 and 673-867 CO_2/tkm, respectively (Goeppert, Olah, & Prakash, 2009). Due to this energy efficiency and the possibility of using low-grade residual fuel, the dirtiest and cheapest produce of oil refineries, maritime shipping has an economic advantage, compared to other freight options. However, the low fuel cost does not reflect the degradation of human- and environmental health caused by its combustion as HFO produces larger amounts of harmful emissions compared to currently available alternatives, such as LNG and methanol (Brynolf, Fridell & Andersson, 2014).

HFO is a low-grade residual fuel which is one of the heaviest fractions obtained from the refinement of crude oil. Residual oil is sold to several sectors but most of it is utilized as marine vessel bunker fuel (EIA, 2019a). Since the 1960's, HFO has been the dominant type of marine fuel. HFO is a term that covers a range of different mixtures of fuel oil. Traditionally, maritime fuels are classified via the ISO 8217 standard, based on their kinematic viscosity at a set temperature, see table 2.1.

Table 2-1 Parameters of maritime fuel oil (ABS, 2018)

Fuel Types	ISO Category	Viscosity (cSt) (at 50°C for Residual and 40°C for Distillate Fuels)		Sulfur Content (%)
		Minimum	Maximum	
Heavy Fuel Oil (HFO)	Residual (RMA – RMK)*	10	700	1.0 – 3.5
Marine Diesel Oil (MDO)	Distillate (DMB)	2	11	0.10 – 1.5
Marine Gas Oil (MGO, Low Sulfur Distillate Fuel)	Distillate (DMA and DMZ)	2	4	0.10 – 1.0
0.10% Heavy Fuel Oil (HFO, ECA Fuel)	Not standardized	9	67	0.10
0.50% Heavy Fuel Oil (HFO, Global Fuel)	Not standardized	No requirements defined	No requirements defined	0.50

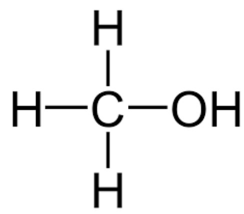
Other terms that are often used for HFO are bunker fuel oil, residual fuel or heavy diesel oil. In the context of maritime fuel in Iceland, three types of fossil fuels are utilized and differentiated as follows: HFO, MGO and MDO, see table 2.2. Moreover, MDO is a mixture of MGO and HFO.

Table 2-2 Total consumption of maritime fuel in Iceland in 2016 (Baldursson, 2017)

Fuel type	Quantity (litres at 15°C)	Sulphur content (%m/m)			Maximum national limit value (%m/m)
		Minimum	Maximum	Average	
Heavy fuel oils	58.418.075	1,77	2,00	1,90	3,5
Marine gasoil	130.667.794	0,01	0,1	0,08	0,1
Marine diesel oil	69.087.302	0,15	0,22	0,20	1,5

2.2 Methanol

The use of alcohols for the propulsion of engines dates as far back as the internal combustion engine (ICE), as some of the earliest ICE's were designed to run on alcohol. Methanol has been effectively tested, with promising outcomes, both on heavy- and light-duty vehicles on land, as well as being an interesting alternative for maritime engines (Ellis & Svanberg, 2018; Andersson & Salazar, 2015; Brynolf, Fridell & Andersson, 2014; Goeppert, Olah, & Prakash, 2009). Alcohols, such as methanol, are simple chemicals compared to HFO which is a mixture of many different hydrocarbons and additives (Goeppert, Olah, & Prakash, 2009).



*„Methanol, also called methyl alcohol or wood alcohol, is a colorless, water-soluble liquid with a mild alcoholic odor. It freezes at -97,6 °C, boils at 64,6 °C and has a density of 791 kg m^{-3} at 20 °C " (*Goeppert, Olah, & Prakash, 2009).

Methanol is a liquid fuel at ambient pressure and temperature, which is one of its main advantages in comparison to gaseous fuels. Methanol can be utilized in several prime movers, i.e. two-stroke and four-stroke diesel engines, Otto engines and fuel cells, which makes the fuel flexible (Brynolf, Fridell & Andersson, 2014). The boiling point of methanol is lower than diesel, which makes combustion easier as the mixture gas forms faster and more evenly. The combustion speed is faster than fuel oil which is associated with the reduction of particulate matter. However, the cetane rating of methanol is lower than that of fuel oil and the auto-ignition temperature is twofold compared to diesel and thus a pilot fuel is needed in most cases to start the ignition (Brynolf, Fridell & Andersson, 2014).

The energy density, i.e., lower-heating value (LHV) of methanol is 15,6 MJ/L, which is lower than conventional maritime fuels, such as HFO, which has a LHV of approximately 38,4 MJ/L (Kumar et al., 2011; Staffel, 2011). This means that, without efficiency improvements, commensurable engines will require approximately twice the volumetric content of fuel, when using methanol instead of HFO. However, efficiency improvements have been proposed in the context of engines built specifically for methanol, *dedicated methanol engines*, as opposed to retrofit engines. In the context of efficiency improvements, methanol has a higher-octane rating which allows engines to run on a higher compression ratio (Goeppert, Olah, & Prakash, 2009). Therefore, the mix of fuel and air can be compressed more before ignition which makes room for plausible efficiency improvements of methanol engines compared to conventional sparkplug engines running on fuel oil such as HFO, MDO or MGO. Moreover, methanol's latent heat of vaporization allows the fuel to subsume more heat when passing from liquid to gaseous state, thus allowing the engine to require less cooling (Goeppert, Olah, & Prakash, 2009.

Methanol has much lower kinematic viscosity than diesel, which might affect lubrication ability in injection pumps (Maritiem Kennis Centrum, 2017). However, this can be mitigated by applying sealing oil for pump lubrication. Lower kinematic viscosity also has impact on spray patterns for direct injection applications which requires manufacturers to redesign the injection pumps to facilitate methanol or to use viscosity improvers in addition to the fuel (Maritiem Kennis Centrum, 2017). In summary, a conventional maritime diesel engine cannot run on methanol without modifications, see figure 2.5 for a detailed summary of benefits and issues regarding methanol in vessels.

Methanol, to date, has only been used a few times as a commercial maritime fuel. The first application was the retrofit RoPax ferry *Stena Germanica*, which has operated on methanol, in at least one of its four engines, since 2015. As of 2017, all engines of *Stena Germanica* had been converted to dual-fuel methanol engines. Subsequently, seven *Waterfront Shipping* chemical tankers that were built to run on methanol, entered service in 2016 (Ellis & Svanberg, 2018). These chemical tankers, as well as *Stena Germanica*, have methanol / diesel dual-fuel engines. Therefore, they can choose situationally which fuel they operate on. The burgeoning interest in methanol as a maritime fuel became apparent after the IMO created ECAs, specifically for sulfur, i.e., Sulfur Emission Control Area (SECA), which came into effect in 2015 in the Baltic Sea. Methanol was less expensive than low sulfur marine gas oil (LSMGO) between 2011 and 2013. Therefore, methanol's prospects became attractive, both environmentally and economically. Furthermore, investment costs of converting large vessels to methanol was approximately equivalent to EGCS adaptations and lower than that of an LNG conversion (Ellis & Svanberg, 2018). Moreover, infrastructure expenses for methanol are modest compared to LNG. Installation cost of a small methanol bunkering system have been estimated to be €400.000, and a methanol bunker vessel conversion approximately €1,5 million. In comparison, infrastructure expenses related to LNG would be €50 million for terminal costs and €30 million for an LNG bunker barge (Methanol Institute, 2017).

Figure 2-1 Stena Germanica RoPax Ferry (Stena Line Freight, n.d.).

The IMO (2015) considered three ship types primarily for methanol i.e., working ships (e.g. offshore supply vessels), cargo ships (e.g. chemical tankers) and passenger vessels (e.g. cruise ships). In the report they calculated an approximation of the capital costs for a ro-ro vessel with 24 MW installed main engine power. For the additional cost associated with methanol for a newbuild, a total of 5,6 million US$ was estimated for methanol modifications; while for the cost of retrofitting a vessel with a methanol fuel system was 10,5 million US$ (IMO, 2015). The total project budget of Stena Germanica methanol conversion was 22 million € and the period of the project was between 2013 and 2015 (Haraldson, 2015).

2.2.1 Safety and handling

Methanol has a high dipole moment and dielectric constant which contributes to corrosion challenges due to its inherent ability to be a solvent for acids, bases, salts and some plastics (Goeppert, Olah, & Prakash, 2009). Therefore, in terms of methanol compatibility with storage, operation and handling, modifications are required for methanol systems. Previous modification in that context have demonstrated that they can be effectively designed to work with methanol. In terms of storage, several methanol-compatible storage tank materials exist, i.e., stainless steel, carbon steel or methanol-compatible fiberglass (Bromberg & Cheng, 2010). Storage tanks must be well grounded to avoid hazards related to static discharge (Methanol Institute, 2013). Methanol is widely used in the chemical industry and therefore there is a mature technology surrounding the transportation and storage of the substance (Maritiem Kennis Centrum, 2017). Storage facilities at docks and marine terminals are dedicated to methanol handling (Methanol Institute, 2013).

Spill-free nozzles, designed for methanol, are available for the service phase which can eliminate spills and concerns of fire-hazards and contact with humans (Bromberg & Cheng, 2010). In case of fire, methanol will burn with a non-luminescent flame and is therefore likely invisible to the naked eye in daylight. Furthermore, due to the lack of residual soot in methanol, there is no smoke. Therefore, responders should be equipped with infrared devices (Methanol Institute, 2013). However, methanol fires can be prevented by controlling vapor and eliminating ignition sources (Methanol Institute, 2017).

Methanol, unlike conventional fuels, is water soluble. In case of a spill, the substance is readily degraded through photooxidation and biodegradation processes (Goeppert, Olah, & Prakash, 2009). The half-life is between 1-7 days in soil, surface- and ground water. If a large methanol spill would occur, the environment is likely to recover quickly (Bromberg & Cheng, 2010).

Methanol, like most fuels, is a toxic substance and precautions need to be taken to avoid harmful exposure. While methanol is a naturally occurring substance and is produced in the metabolic process of humans, it is poisonous at higher concentrations. Methanol can cause blindness and other neurological effects if ingested and a lethal dose consist of 10-30 ML for adults (Methanol Institute, 2017). In comparison to gasoline, methanol is equally or less dangerous, however, certain stigma has surrounded the compound where toxicological concern is in the foreground. This is partly related to the fact that people have in some cases ingested methanol, thinking it would serve the same function as conventional alcohol. In the California test, where methanol vehicles were driven in the state of California over 200 million miles, there was not a sole case of accidental methanol poisoning (Bromberg & Cheng, 2010).

2.2.2 Methanol production

Methanol production capacity worldwide is estimated to be around 110 million tonnes. Global methanol consumption is estimated to be 100 million tonnes p.a. and growing fast, see figure 2.2. However, most of the methanol produced is not utilized as fuel. This is since methanol plays an important role in the chemical industry and is used as a precursor for the production of many different goods. Out of the global methanol production, 65% is used to produce acetic acid, methyl and vinyl acetates, methyl methacrylate, methylamines, metal-t-butil etere, fuel additives and other chemicals (Dalena, Senatore, Basile, Knani, Basile & Lulianelli, 2018).

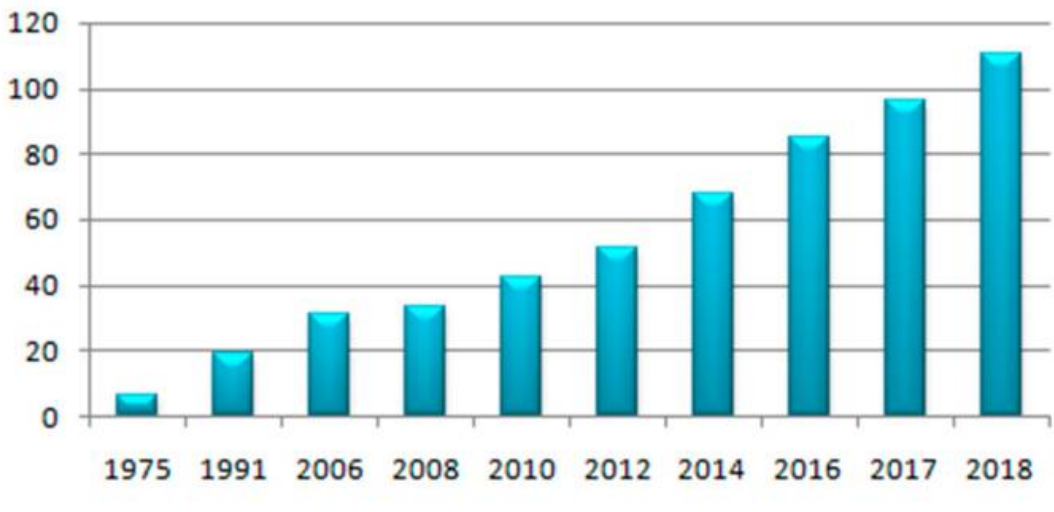


Figure 2-2 Global methanol production, expressed in million tonnes p.a. (Dalena et al., 2018).

Methanol can be produced from several feedstocks and production pathways (Edwards, Larivé, Rickeard & Weindorf, 2013), enabling enhanced energy security, even without access to fossil fuel reserves. Moreover, any feedstock that can be gasified into synthesis gas (syngas) is a potential feedstock for methanol production (Bromberg & Cheng, 2010). This makes potential production capacity of methanol high. Current production pathways and feedstocks range from reformation of natural gas (NG), gasification of coal, biomass and municipal waste and electrolysis of water combined with carbon capture and utilization (CCU), see figure 2.3.

Methanol is most commonly produced through fossil fuel pathways, including coal, but natural gas is currently the most common feedstock, amounting to 90% of methanol production, globally, (Dalena et al., 2018), via catalytic conversion of pressurized syngas. The fossil fuel pathways have been commercial processes for around 80 years (Fortes & Tzimas, 2016). Therefore, natural gas and coal to methanol technologies have matured and thus offers economic advantages compared to e.g., biomass or CCU and electrolyzed water to methanol.

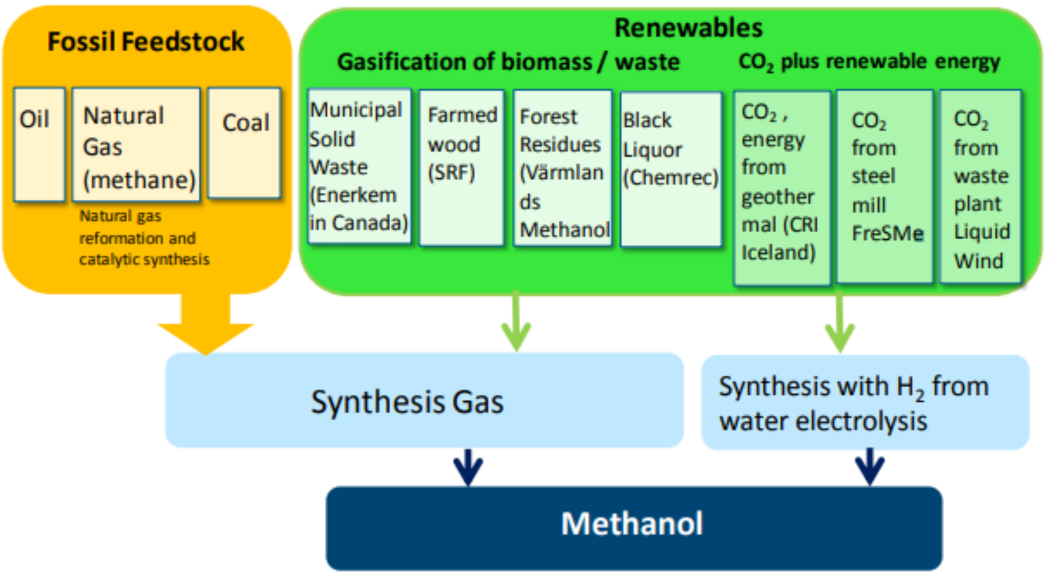


Figure 2-3 Production pathways and feedstocks that can be utilized for methanol production (Ellis & Svanberg, 2018)

2.2.3 Production capacity of renewable methanol in Iceland

The usage of CO_2 as a carbon feedstock for chemical storage of energy could impact the global carbon cycle positively. Supply of carbon feedstocks is abundant as they can come from geothermal plants, aluminum plants, biomass, fossil-fired power stations and fermentation processes. CRI utilizes carbon dioxide from a geothermal power plant, but other CO_2 flue-gas sources are available in Iceland e.g., from the aluminum industry. However, focusing on the geothermal waste streams is feasible as they are both abundant, in terms of needed CO_2 for this case study, and the process of geothermal CO_2 to methanol has been demonstrated successfully in CRI's pilot plant.

The total amount of geothermal CO_2 flue-gas, emitted in Iceland, in 2017, was 146 kt (NEA, 2018a), see table 2.3.

Table 2-3 Gas emission of geothermal power plants in Iceland from 1969 to 2017 (NEA, 2018a)

Virkjun eða svæði / Plant or area	Hnit / Coordinates		Heildarlosun gastegunda út í andrúmsloftið / Total gas release into the atmosphere						
	X	Y	Gastegund / Gas Type	2017	2016	2015	2014	2013	2012
Reykjanesvirkjun (Power Plant)	318910	374281	CO2 (tonne/annum)	16391	18.800	23.791	26.677	21.255	21.190
			H2S (tonne/annum)	546	581	865	941	779	800
			CH4 (tonne/annum)	3	2	3	3	2	2
Svartsengi (CHP Plant)	331640	379650	CO2 (tonne/annum)	55602	57.300	56.212	65.087	58.712	47.503
			H2S (tonne/annum)	1036	932	969	1.136	1.106	848
			CH4 (tonne/annum)	5	5	5	6	5	4
Hellisheiði (CHP Plant)	384646	395223	CO2 (tonne/annum)	23555	26.102	33.077	38.861	44.934	43.158
			H2S (tonne/annum)	2866	3.893	6.384	8.484	12.374	16.881
			CH4 (tonne/annum)	49	48	80	81	72	51
Nesjavellir (CHP Plant)	389713	402461	CO2 (tonne/annum)	14020	14.655	14.726	16.579	14.794	18.612
			H2S (tonne/annum)	6829	8076	8.350	9.275	8.709	11.349
			CH4 (tonne/annum)	31	44	54	55	46	28
Hveragerði (Heat Plant)	392903	390530	CO2 (tonne/annum)	160	160	160	160	160	160
			H2S (tonne/annum)	10	10	10	10	10	10
			CH4 (tonne/annum)	0	0	0	0	0	0
Bjarnarflag (CHP Plant)	599411	572753	CO2 (tonne/annum)	427	367	337	296	300	956
			H2S (tonne/annum)	551	521	477	447	539	826
			CH4 (tonne/annum)	4	4	4	4	7	11
Krafla (Power Plant)	602545	580265	CO2 (tonne/annum)	33397	30.802	32.483	34.395	32.307	38.604
			H2S (tonne/annum)	4780	4.689	4.483	4.520	4.392	5.151
			CH4 (tonne/annum)	11	11	11	11	7	14
Þeistareykir (Power Plant)	592992	599039	CO2 (tonne/annum)	2922	770	2.345	895	672	1.863
			H2S (tonne/annum)	787	150	971	391	136	246
			CH4 (tonne/annum)	2	0	1	1	0	1
Samtals / Total			CO2 (tonne/annum)	146.474	148.956	163.131	182.950	173.134	172.046
Samtals / Total			H2S (tonne/annum)	17.405	18.852	22.509	25.204	28.045	36.111
Samtals / Total			CH4 (tonne/annum)	104	114	158	161	139	111

CRI utilizes around 10% of Svartsengi's power plant CO_2 waste stream or 5500 tonnes p.a., from which they produce around 5000 m^3 of methanol (CRI, n.d.). Therefore, 1,1 tonne of CO_2 represents 1000L or 1m^3 of methanol or 15,6 GJ. Subsequently, if all geothermal waste streams in Iceland would be captured and utilized for methanol production, we would be looking at a production capacity of 133.158 m^3 of methanol i.e., 2077 TJ of energy. To put this into context, the Icelandic maritime industry consumed 258.173.170 L of oil in 2016, which is the equivalent of 9914 TJ, if the average energy content of HFO, MDO and MGO are 38.4 MJ/L. Subsequently, if all geothermal waste stream would be utilized for methanol production, Iceland could meet 21% of the energy demand of the maritime industry.

Current production capacity of CRI, 5.000.000 L amounts to 78 TJ or 0,8% of the maritime energy demand. Iceland aims to have 10% of the energy utilized by the maritime industry, from a renewable source, by 2030. To meet that goal, 50% of carbon streams from geothermal power plants in Iceland would need to be utilized. Svartsengi is currently producing 37,9% of geothermal CO_2 waste stream in Iceland (NEA, 2018) and thus only utilization of one more large geothermal power plant, e.g., Krafla or Hellisheiði, would be needed to capture ~50% of the geothermal CO_2 waste stream and meet the 10% renewable energy in the maritime sector in Iceland goal. This would entail building a new methanol plant at another geothermal power plant, as well as increasing production capacity at Svartsengi substantially.

The main feedstock for the electrolysis is electricity and water, both being abundant in Iceland. Moreover, Iceland is the largest producer of electricity per capita and has an abundance of renewable energy sources. Electricity production amounted to 19,2 TWh in 2017 and is expected to grow between 0,9 to 2,2% every year until 2050 (NEA, 2018b). In the context of the 10% renewable energy objective in the maritime sector, energy consumption of a renewable energy carrier would be 991,4 TJ, given that 100% of the energy consumption of the maritime fleet is 9914 TJ, currently. From the perspective of energy conversion efficiency in electrolysis methanol power plants of 51-68% (Mignard et al., 2003), 404 to 540 GWh of electricity would be required to produce 991,4 TJ of methanol i.e., 275 GWh of methanol.

3 Methodology and data

The purpose of this research thesis is to carry out an economic valuation of maritime fuels, where the negative externalities related to the combustion and production of HFO and methanol, are added to the cost side of a CBA in a fuel appraisal. Total cost ($TC_{€/MJ}$), of each fuel, is a sum of externality cost ($EC_{€/MJ}$) and fuel cost ($FC_{€/MJ}$), at a given point in time.

$$TC_{€/MJ} = FC_{€/MJ} * EC_{€/MJ} \tag{1}$$

This thesis assesses the annual cost of fuels, following a time trajectory from the present and towards 2050. Furthermore, two different production pathways of methanol are assessed to figure out how cost-effective a transition towards renewable methanol, compared to NG methanol, could be for the maritime sector in Iceland. Consequently, the price estimation of both fuels, given different timescales and production pathways, was estimated. Furthermore, the cost of externalities for different pollutants was assessed in terms of maritime externality literature based on an Impact Pathway Analysis (IPA), which was then extrapolated into an extended CBA, using an LCA conducted by Brynolf, Fridell & Andersson (2014), which compared HFO and methanol in a maritime engine. This step was done by multiplying the amount of each pollutant, relative to the functional unit (1tkm), with the cost of the same amount of the respective pollutant. Finally, with the combination of both steps, the total cost of the fuels is found, see figure 3.1. The following chapter will outline the methodologies and data sets that were utilized for these areas of the thesis.

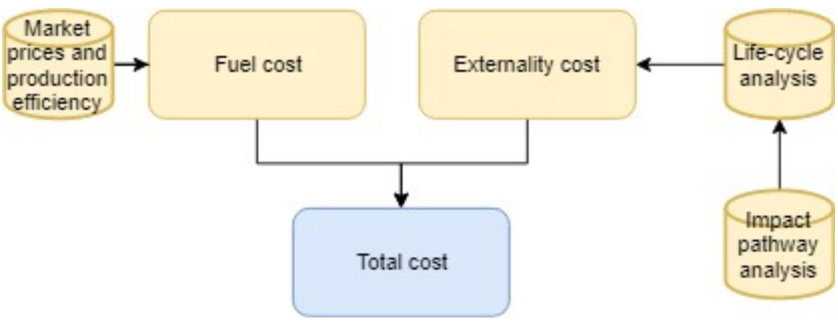


Figure 3-1 Estimation of total cost via the combination of external cost and fuel cost

3.1 Externality costs

This economic valuation, in the form of an extended CBA, is a fuel appraisal, where the combination of externality costs and fuel cost, are linked to form a market-based instrument that can set energy prices right. The externality cost, of all pollutant except CO_2, are estimated through an IPA. These cost estimates are adopted through several air pollutant externality studies in the maritime context (Kotowska, 2017; Holland & Watkiss, 2004; Jiang, Kronbak & Christensen, 2010; Maibach et al., 2008), which are all based on either the Clean Air for Europe program (CAFE) and/or Developing Harmonized European Approaches for Transport Costing and Project Assessment (HEATCO), see table 3.1 and Appendix 1.

Table 3-1 External cost of pollutants, from the at sea category, given in 2018€/tonne, i.e., cost-factors ($CF_{€/tonne}$)

Study	Country/Region	NOx	SO2	PM_{ud}	NMVOC	CO2
(Kotowska, 2017)	EU	3160		1458	608	
(Holland et al., 2004)	E-Atlantic	6542	6133			
-	Baltic Sea	2862	2181			
-	English Channel	7360	8041			
-	N-Mediterranean	8450	6406			
-	North Sea	4225	5860			
(Jiang et al., 2010)	North Sea	6790	9186			
(Maibach et al., 2008)	Baltic Sea	3544	5043		555	
-	Mediterranean	681	2726		333	
-	NE-Atlantic	2181	2998		444	
-	North Sea	6951	9404		2108	
(EU ETS, Average 2018)	EU					16
(EU ETS, Average 2019)	EU					22

The abovementioned studies provide cost estimates for NOx, SO_2, SO_x, $PM_{2,5}$, $PM_{undifferentiated}$ (PM_{ud}), NMVOC and VOC. However, the LCA (Brynolf, Fridell & Andersson, 2014) does not provide measurements for all the above-mentioned pollutants. Therefore, only the pollutants that are both in the maritime externality literature (Kotowska, 2017; Holland & Watkiss, 2004; Jiang, Kronbak & Christensen, 2010; Maibach et al., 2008) and in the LCA (Brynolf, Fridell & Andersson, 2014), are in table 3.1. The cost estimates are provided in 2000€, 2010€ and 2014€ values. Furthermore, for consistency, all cost estimates of externalities are converted into nominal 2018€ via cumulative inflation of the € i.e., 37,35%, 10,94% and 3,51% (Inflation Tool, n.d.) price increase from 2000, 2010 and 2014, respectively, and are expressed in €/tonne. Given a slight variation of cost estimates, between the underlying methodologies, i.e., CAFE and HEATCO, the average value of the nominal cost estimates is used as the basis of this part of the study. Furthermore, one standard deviation, σ, is added and subtracted from the mean, μ, to form a high and low case, while the average is the medium case. Therefore, three plausible trajectories are provided to facilitate a 68% confidence interval of externality cost. This is due to the data set being relatively small, 1 to 11 estimates for each pollutant. Therefore, a higher confidence interval, such as two standard deviations from the mean would have resulted in negative externality cost for certain pollutants. Estimates for undifferentiated PM were only found in one study (Kotowska, 2017) and will therefore not be differentiated between trajectories, i.e., static PM externality cost estimate of 1458 €/tonne.

$$\text{Low} = \mu - \sigma \tag{2}$$

$$\text{Medium} = \mu \tag{3}$$

$$\text{High} = \mu + \sigma \tag{4}$$

Table 3-2 Externality cost of pollutants, €/tonne, adopted from (Kotowska, 2017; Holland & Watkiss, 2004; Jiang, Kronbak & Christensen, 2010; Maibach et al., 2008)

	NOx	SO2	PM_{ud}	NMVOC	CO2
Low	2272	3196	1458	76	15
Medium	4795	5798	1458	809	19
High	7318	8400	1458	1543	23

All externality cost estimates in table 3.2 are from the *at sea* category of air pollutant externality estimations from the maritime sector. Consequently, *near port* and *in port*, externality estimation categories are omitted from this thesis on the basis that the maritime sector in Iceland is seldom in contact with as densely populated areas as the emission externality studies are based upon, i.e., HEATCO and CAFE, as they base their calculations around mainland European cities. The median population density of European countries is 106 persons/km^2 compared to Iceland's population density of 3,5 persons/km^2. However, the omittance of the in port category will most likely lead to low estimates of the impacts.

Climate change costs vary considerably between studies and are subject to e.g. discount rate and equity weights. Cost estimates for CO_2 are thus found in the EU ETS database, where a market value is accessible, and are based on the average price of an emission quota throughout 2018 and from 1 January to 30 April, 2019, i.e., 16 and 22 €/tonne, respectively, giving a value of 19 €/tonne. The reason for choosing the average values of 2018 and 2019 is that these are the most recent estimates of the market value of CO_2 quota in Europe. Moreover, this is to stay on the conservative side of externality cost assessments, as Maibach et al (2008) and Jiang, Kronbak & Christensen (2010), assume CO_2 cost to be 27€/tonne, in terms of the value of the € in 2018.

3.1.1 Life-cycle emissions

The amount of pollutant produced from fuels life-cycle is represented as $P_{g/MJ}$ in equation 1. This relates to emissions during well-to-propeller (WTP) phase, differentiated into WTT and TTP. The WTT emissions of methanol production pathways vary between 7,54 – 120 gCO2eq/MJ (Edwards, Larivé, Rickeard & Weindorf, 2013). The WTT emissions of CRI's methanol and natural gas is 7,54, see table 3.3, and approximately 20 gCO2eq/MJ (Brynolf, Fridell & Andersson, 2014), respectively.

Table 3-3 WTT gCO2eq/MJ of methanol production at CRI (Personal email with Benedikt Stefánsson Director of Business Development at CRI, 2018)

WTT emissions	gCO2eq/MJ
Emissions related to raw material	0
Emissions of electricity production	6.71
Emissions of steam production	0.22
Emissions of process specific inputs	0.21
Emissions of wastewater treatment	0.06
Plant to port	0.34
Total emissions WTT	7.54

The WTT emissions of HFO and methanol via the NG pathway, are taken from Brynolf, Fridell & Andersson (2014), i.e., 6,7 and 20gCO_2eq/MJ, respectively. However, that paper goes further than analyzing the CO_2eq of HFO and NG methanol production as there are more pollutants associated with their production than can be measured in terms of CO_2eq. The underlying reasoning for using only the CO_2eq is the fact that data on the production of methanol via the CRI pathway was only accessible in terms of CO_2eq.

TTP emission factors come from a study (Brynolf, Fridell & Andersson, 2014), which compared HFO and methanol in a commensurable engine and engine mode, i.e., engine size and operational mode, see table 3.4.

Table 3-4 TTP emissions (g/MJ) from HFO and Methanol in a 14,68 MW Ro-ro vessel, adopted from Brynolf, Fridell & Andersson (2014).

Tank-to-propeller emissions

	HFO	Methanol
CO_2 (g/MJ)	77	69
CH_4 (g/MJ)	0.00045	0
N_2O (g/MJ)	0.0035	0
NO_x (g/MJ)	1.6	0.28
SO_2 (g/MJ)	0.69	0
NH_3 (g/MJ)	0.0003	0
PM_{10} (g/MJ)	0.093	0.0043
NMVOC (g/MJ)	0.056	0
CO (g/MJ)	0.13	0

The results shown in table 3.4 come from the combustion of HFO and methanol in a 14,68 MW medium-speed four-stroke diesel engine for the HFO and a 14,68 MW dual-fuel engine, with 1% MGO as pilot fuel for the methanol. Engine efficiency was commensurable for both fuel tests i.e., 41%. However, cargo capacity of the vessel propelled with methanol decreased by 4%, from 7500 tonnes to 7200 tonnes. Reduced cargo capacity is due to larger storage requirements for the methanol relative to HFO given the respective lower-heating values of the fuels and the size of the vessel. Subsequently, vessel efficiency is reduced to 0,0591 kWh work/t km for the methanol engine, compared to 0,0568 kWh work/t km for the HFO engine. Therefore, fuel consumption is increased slightly in the methanol case i.e., 0,5189 MJ/t km compared to 0,4987 MJ/t km for HFO, due to less cargo capacity in the methanol vessel. Subsequently, pollution factors are adjusted according to increased fuel consumption, i.e., 4% increased energy consumption per t km see table 3.5. The functional unit in Brynolf, Fridell & Andersson (2014), was 1 tonne cargo transported 1 km with a ro-ro vessel, i.e., 1tkm, see table 3.6.

Table 3-5 TTP emissions (g/MJ) from HFO and Methanol in a 14,68 MW Ro-ro vessel, adopted from Brynolf, Fridell & Andersson (2014), adjusted for vessel efficiency.

Tank-to-propeller emissions

	HFO	Methanol
CO_2 (g/MJ)	77	72
CH_4 (g/MJ)	0.00045	0
N_2O (g/MJ)	0.0035	0
NO_x (g/MJ)	1.6	0.29
SO_2 (g/MJ)	0.69	0
NH_3 (g/MJ)	0.0003	0
PM_{10} (g/MJ)	0.093	0.0045
NMVOC (g/MJ)	0.056	0
CO (g/MJ)	0.13	0

Table 3-6 WTP emissions (g/t km) from HFO and Methanol (CRI) and Methanol (NG) i.e., adjustment for vessel efficiencies

Life-cycle emissions

		HFO	Methanol (CRI)	Methanol (NG)
WTT	CO_{2eq} (g/tkm)	3.34	3.91	10.38
TTP	CO_2 (g/tkm)	38.40	37.36	37.36
TTP	CH_4 (g/tkm)	0.000224	0	0
TTP	N_2O (g/tkm)	0.001745	0	0
TTP	NO_x (g/tkm)	0.797920	0.145292	0.145292
TTP	SO_2 (g/tkm)	0.344103	0	0
TTP	NH_3 (g/tkm)	0.000150	0	0
TTP	PM_{10} (g/tkm)	0.046379	0.002231	0.002231
TTP	NMVOC (g/tkm)	0.027927	0	0
TTP	CO (g/tkm)	0.064831	0	0

3.1.2 Total externality cost of fuel life cycles

In the maritime context, external costs can be generated as a result of the environmental and public-health impacts associated with harmful emissions, namely, the effects of SO_2, NO_x, PM, NMVOC and GHG. However, more pollutants could be included in the external cost of fuels but due to N_2O, N_2O, CH_4 and CO not being mentioned in the maritime externality literature, they are omitted from the results.

External costs are the sum of costs generated by pollutants during the fuel's lifecycle, measured in g/tkm. Therefore, external costs ($EC_{€/tkm}$) of fuels, from production (WTT) and combustion (TTP), is found via the respective cost factor ($CF_{€/g}$) of each pollutant, multiplied with the amount of said pollutant produced corresponding to the energy exerted in the main engine to propel the vessel and its cargo by 1tkm ($P_{g/tkm}$),see figure 3.2.

$$EC_{€/tkm} = CF_{€/g} * P_{g/tkm} \tag{5}$$

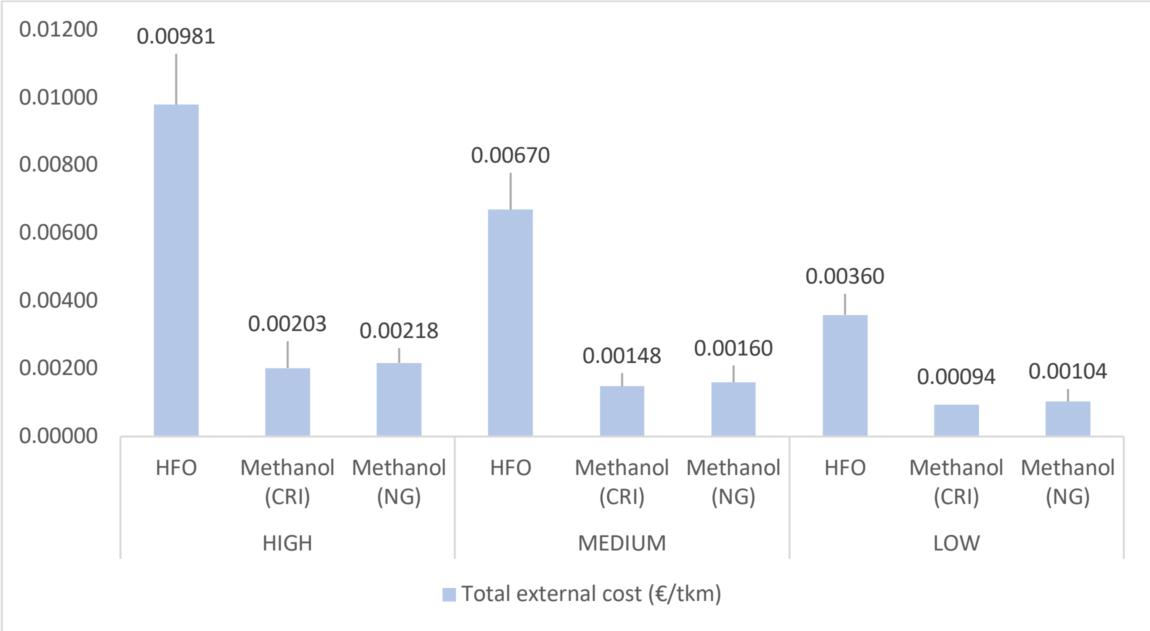


Figure 3-2 Total external cost relative to vessel traveling 1tkm, represented via high, medium and low estimation of externality costs.

3.2 Fuel cost

3.2.1 Price development of HFO

HFO price development is causally linked to international prices of crude oil. These are inherently volatile and subject to substantial amounts of variables, both from the consumer side and the production side. Production development of crude oil, to the year 2050, has been projected by the U.S. Energy information Administration (EIA), following several trajectories which are based on a range of assumptions, i.e., technological advancement of oil and gas resource extraction, economic growth, demographic- and policy assumptions. These factors are likely to affect the price development of crude oil and thus the production levels, see figure 2.1. One significant driver behind crude oil prices is growth of the global economy. According to OPEC, the global economy is expected to grow at a rate of between 3,1% and 3,7% in the medium-term, i.e., between 2016 and 2022. Similar trends are seen in the long-term, or an average 3,5% growth p.a. until 2040. Long-term growth rate of renewable energy is expected to peak at 6,8% p.a., however, their share in the overall energy mix, is only expected to reach 5,4% in 2040, due to their low initial base (OPEC, 2017).

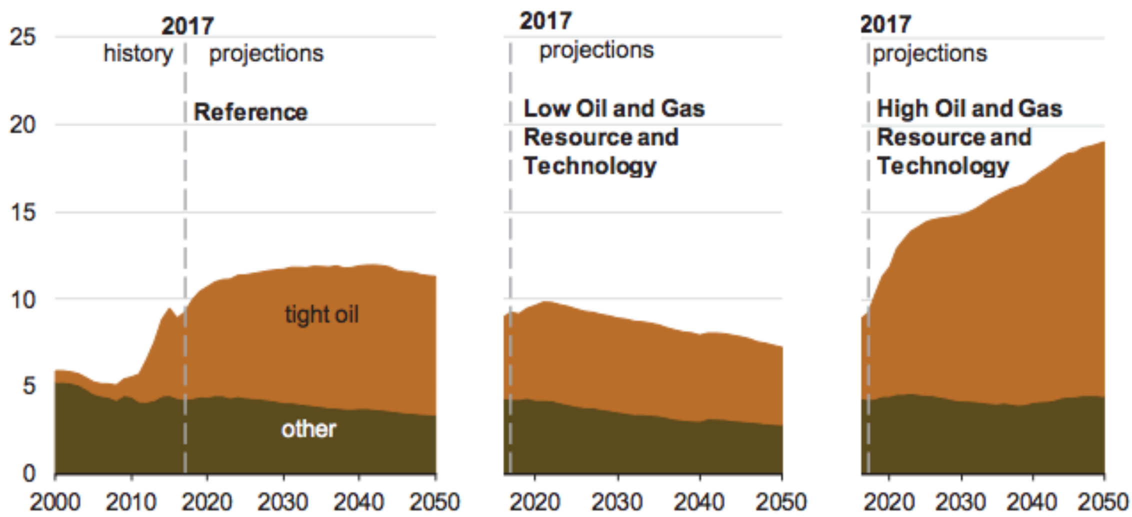


Figure 3-3 Crude oil production forecast in the U.S. (million barrels per day), historical reference and projection towards the year 2050 (EIA, 2019b).

Initial HFO price corresponds to price in June 2018, average Rotterdam spot price of 57 €/bbl (IEA, 2018b), where the conversion rate between $ to € is 1 to 0,89, respectively. Following scenario assumptions in crude oil demand and production, three cases of price trajectories, low, medium and high increase, are considered in terms of plausible development towards the year 2050 (Shafiei, Davidsdottir, Stefansson, Asgeirsson, Fazeli, Gestsson & Leaver, 2019). In the medium and high trajectories, price development is expected to increase linearly, following a constant growth rate, see figure 3.4.

- Low case: constant HFO price of 57€/bbl;
- Medium case: HFO price increase from 57€/bbl in 2018 to to 101€/bbl in 2050, average annual growth rate 1,8%;
- High case: HFO price increase from 57€/bbl in 2018 to 141€/bbl in 2050, average annual growth rate 2,9%.

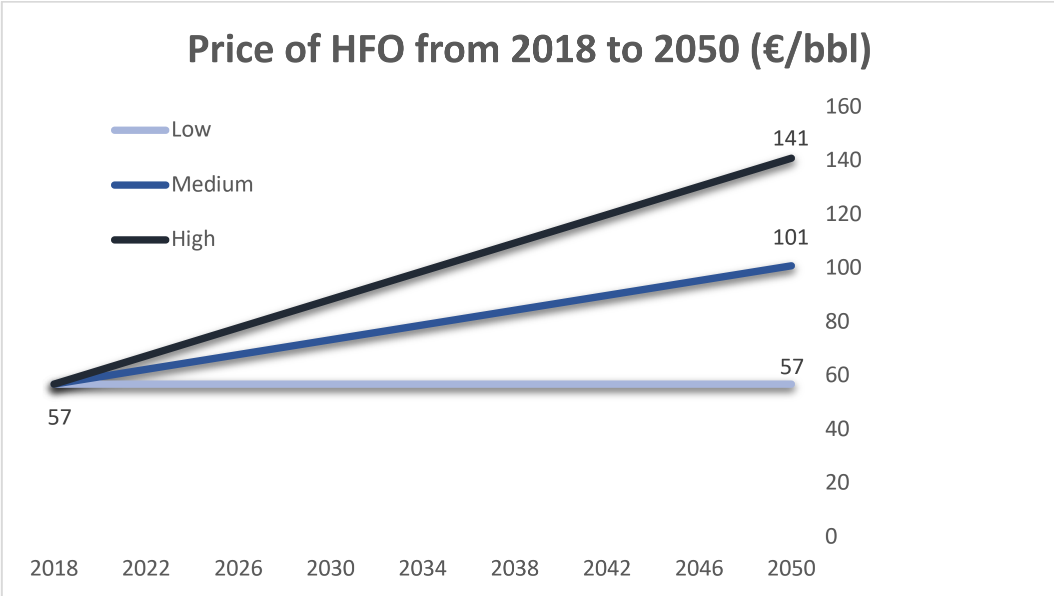


Figure 3-4 Three price trajectories of HFO from 2018 to 2050, following a low, medium and high case.

3.2.2 Price development of methanol

This thesis looks at methanol cost from the perspective of two different production pathways, i.e., conventional and renewable. Moreover, the renewable pathway in this study is explicitly methanol from electrolysis and CCU while NG methanol is from natural gas to methanol pathway. However, the price of the renewable pathway, CRI methanol, is not publicly available as the power plant is a pilot plant and it has not been possible to obtain exact price information from the company. Therefore, CRI methanol price is estimated from studies relating to the price difference between conventional and renewable pathways in methanol production (Goeppert, Olah & Prakash, 2009; Clifre & Badr, 2007; Mignard, Sahibzada, Duthie & Whittington, 2003) and base prices of NG methanol from Methanex, the largest producer and supplier of methanol, globally. Moreover, a price estimation is also conducted based on production efficiency of power plants similar to CRI's methanol plant, electricity prices and estimations of the proportion of electricity cost in total cost of production.

The company Carbon Recycling International (CRI) has been producing methanol in Iceland since 2012, where the feedstock of carbon is coming from a nearby geothermal powerplant in Svartsengi (CRI, n.d.). This is an exemplary model of industrial symbiosis as it utilizes a waste stream that would otherwise be released into the atmosphere and creates an economic product out of CO_2 as it turns waste emissions into a raw material for fuel production. CRI utilizes around 10% of Svartsengi's power plant CO_2 waste stream i.e., 5500 tonnes p.a., from which they produce approximately 5.000.000 L of methanol (CRI, n.d.). However, no techno-economic analysis has been publicly submitted through peer reviewed journals regarding the economic feasibility of their production. Consequently, methanol price will be assumed through studies of commensurable methanol plants.

CRI uses electro-chemical conversion by water electrolysis to produce hydrogen, utilizing electricity from the Icelandic energy grid, which is based on 99,9% renewable energy, i.e., hydroelectric, geothermal and wind (NEA, 2017). However, 96% of hydrogen production is derived from fossil fuels, globally, while electrolysis of water for hydrogen accounts for 4% (Fortes & Tzimas, 2016). Electrolysis for hydrogen production is estimated to be 3-4 more costly than conventional natural gas to hydrogen production (Goeppert, Olah & Prakash, 2009). Moreover, electricity price in a hydrogen electrolysis plants is estimated to be between 70-80% of total costs (Matzen, Alhajji, & Demirel, 2015). Alkaline electrolysis technology is the most commercial process in this context and is estimated to have around 57-75% energy efficiency, based on the higher-heating value (HHV) and 50-60% energy efficiency based on the LHV (Matzen, Alhajji, & Demirel, 2015).

According to an email from Ómar Freyr Sigurbjörnsson, director of sales and marketing at CRI, we can assume that the conversion efficiency of the total production process i.e., hydrogen production and the methanol synthesis, is between 50-60%. These conversion efficiencies are confirmed by Mignard et al., (2003) where the total electricity to methanol process efficiency, utilizing effluent CO_2 from a nearby power plant, is between 51-58% and can go up to 58-68% where waste heat is available (Mignard et al, 2003). Specific energy of methanol is 19,7 MJ/kg which equates to 5,47 MWh/tonne. Therefore, with theoretical 100% efficiency, we would need 5,47 MWh of electricity to produce 1 tonne of methanol. Given the efficiency parameters set by CRI and Mignard et al, (2003), we see that electricity required, is between 8,04 – 10,73 MWh for a total process efficiency spectrum of 68% and 51%, respectively. Moreover, Matzen, Alhajji & Demirel (2015), estimated that the proportion of electricity cost in a methanol plant, producing via the electrolysis and CCU pathway, was 23-65% of total cost. Therefore, if we assume an electricity price according to EIA's (2019c) LCOE estimation of hydroelectric power of 39,1 2018$/MWh i.e., 34,8 2018€/MWh, we can estimate that electricity cost in CCU/electrolysis methanol plant, with and efficiency between 68% and 51% to be from 280 to 373 €/tonne, respectively. Moreover, if we assume that electricity cost is between 23% and 65% of total cost, and the abovementioned parameters of power plant efficiency are valid, minimum and maximum price would be from 431 to 1622 €/tonne, respectively.

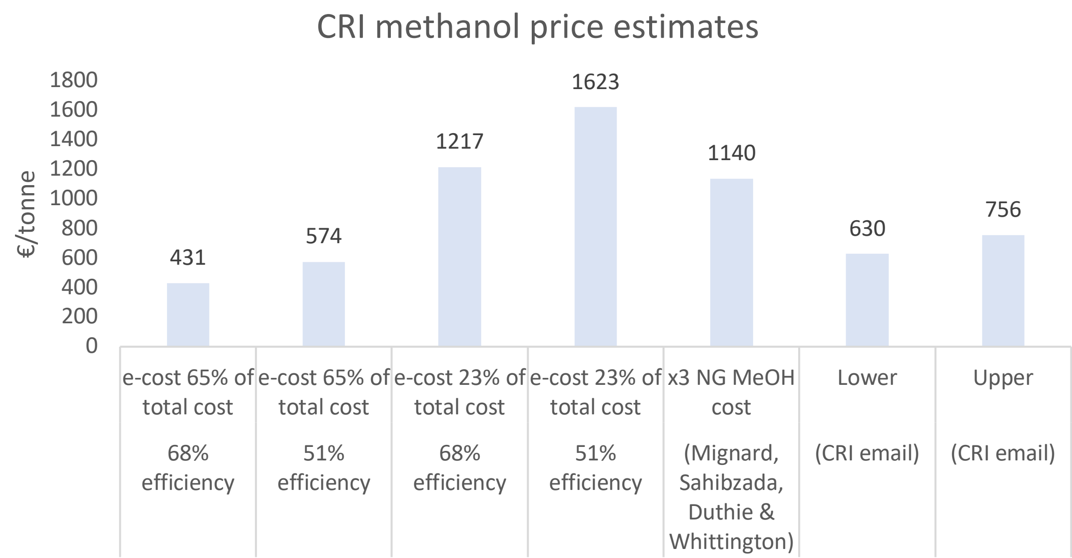


Figure 3-5 CCU/electrolysis methanol cost estimate from power plant efficiency, LCOE of hydroelectric power and proportion of electricity cost in total cost and Mignard, Sahibzada, Duthie & Whittington (2003) cost of CCU electrolysis methanol relative to conventional NG methanol.

Mignard, Sahibzada, Duthie & Whittington (2003), showed that the conversion between electricity and methanol, in a relatively commensurable process to CRI i.e., flue-gas CO_2 and renewable electricity, was possible with 59% efficiency. Furthermore, they estimated, through four cases of methanol power plants, utilizing electrolysis, the minimum selling price for methanol, in the context of both capital expenditure for the power plants and variable operating costs, to be between 0,272 to 0,422 2003£/liter, given that the power plant project would have a net present value (NPV) of zero after fifteen years of operations and a minimum acceptable rate of return (MARR) of 10% (Mignard, Sahibzada, Duthie & Whittington, 2003). Finally, they estimated that the price difference of fossil fuel methanol and renewable methanol was 1:3, respectively, before taxation and distribution (Mignard, Sahibzada, Duthie & Whittington, 2003).

Initial methanol price, for NG methanol, is retrieved from Methanex. This price will represent NG methanol, in the following comparison between fuels. Their posted contract price, in June 2018, was 380 €/tonne (Methanex, 2019). The time was chosen to correspond to the initial price chosen for the HFO price trajectories.

Price development towards 2050 of NG methanol and CRI methanol will follow price development of natural gas for NG methanol and electricity prices and learning curve estimations for CRI methanol, see figure 2.9.

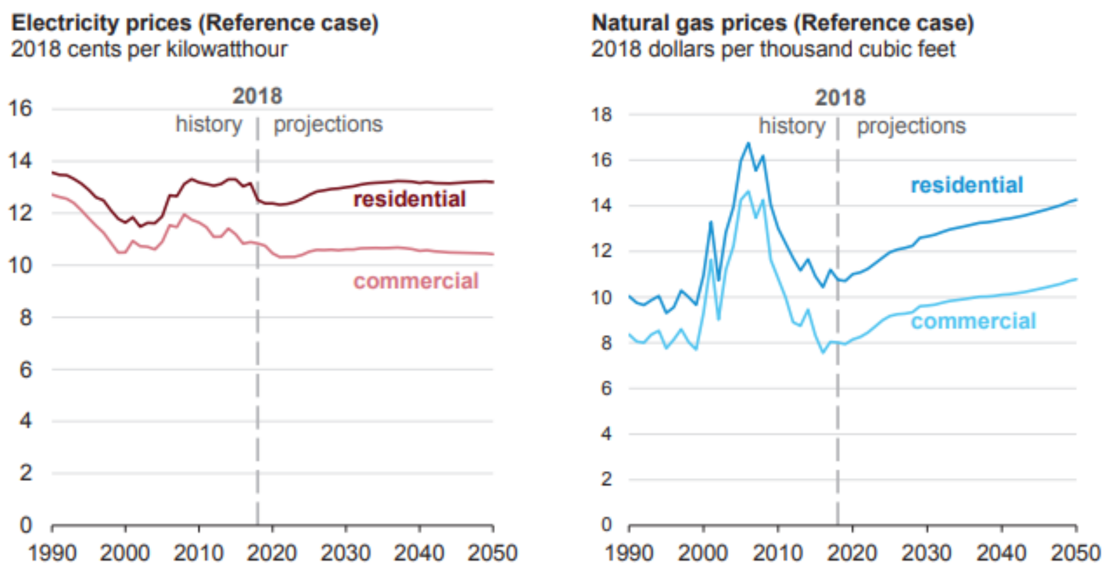


Figure 3-6 Electricity and natural gas price estimates towards 2050 (EIA, 2019b)

Learning curves of innovative technologies such as those used in pilot power plants, have shown that significant price reductions in capital costs are plausible with increased installed capacity and the positive externalities related to that e.g., efficiency improvement. The case of electrolysis and CCU in methanol production is technologically in its infancy. Therefore, using related technologies as a proxy for the learning curves is needed. In the case of solar photovoltaics, a cost reduction of 14%

was derived per doubling of cumulative installed capacity (Kersten, Doll, Kux & Huljic, 2011). Schmidt, Gmbhir, Staffel, Hawkes, Nelson & Few (2017) revealed from an expert opinion study that research & development (R&D) funding would decrease capital costs of solid oxide electrolysis cells by 0-24% and that the production cost would decrease between 17-30% per doubling of cumulative installed capacity, until 2030. Moreover, electricity prices are estimated to be relatively static throughout the study's time horizon, see figure 3.6.

Given the corresponding maturity of the CRI methanol pathways, a linear price reduction will be assumed for CCU and electrolysis methanol of 20% and 30% in 2050, relative to 2018 prices, to reflect cost change corresponding to technological advancements of CCU electrolysis methanol plants outside the boundaries of electricity cost. Given the high variation of price estimations of CCU and electrolysis methanol, initial price will be assumed to be the average of all seven estimates, from figure 3.5, i.e., 910 €/tonne. A slight price increase, is estimated for NG methanol towards 2050, following NG forecast of the IEA (2019) i.e., 20-25% price increase of NG methanol. In the medium and high trajectories, price development is expected to increase/decrease linearly, following a constant growth rate, see figure 3.7.

- Low case: constant NG methanol price of 380€/tonne;
- Medium case: NG methanol price increase from 380€/tonne in 2018 to 456€/tonne in 2050, average annual growth rate 0,6%;
- High case: NG methanol price increase from 380€/tonne in 2018 to 475€/tonne in 2050, average annual growth rate 0,7%.
- Low case: CCU/electrolysis Methanol price decrease from 910€/tonne in 2018 to to 637€/tonne in 2050, average annual decrease rate -1,1%;
- Medium case: CCU/electrolysis Methanol price decrease from 910€/tonne in 2018 to to 728€/tonne in 2050, average annual decrease rate -0,7%;
- High case: constant CCU/electrolysis methanol price of 910 €/tonne.

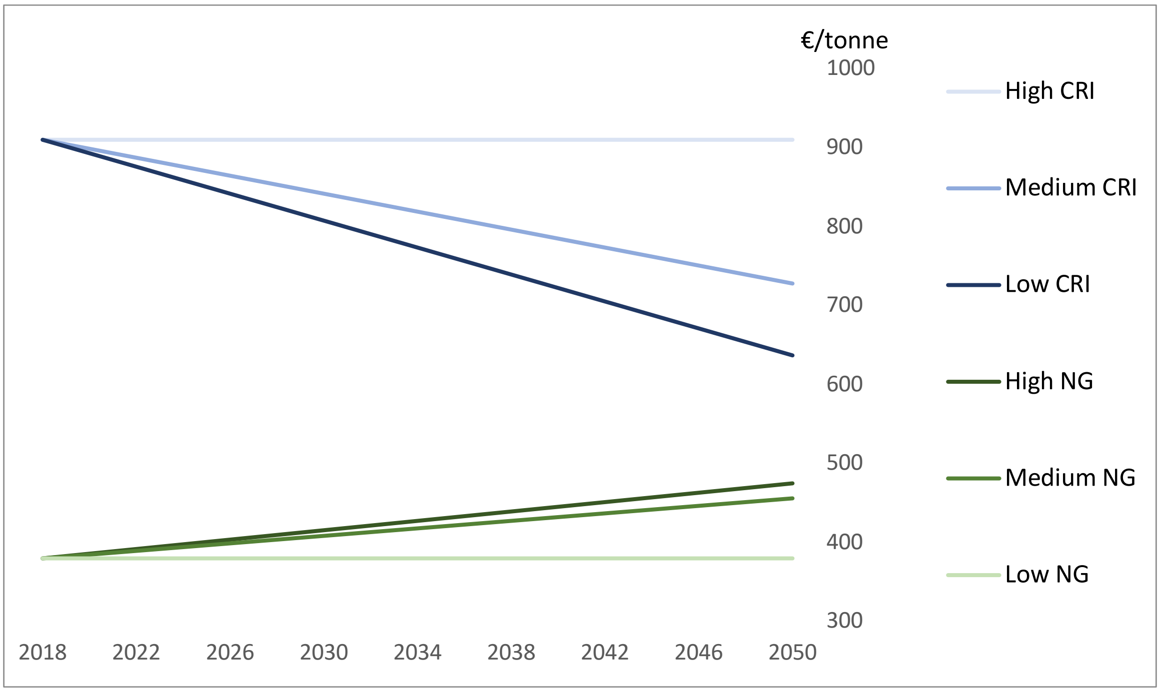


Figure 3-7 Three price trajectories of CRI and NG methanol, from 2018 to 2050, following a low, medium and high case.

4 Results

This section will outline the main results of the extended CBA. Firstly, the results of the fuel cost estimates are presented in terms of €/tkm. The cost estimates were initially presented in terms of conventional values, i.e., €/bbl and €/tonne, for HFO and methanol, respectively. They were then converted into €/MJ to align them with the measurements of pollutants from the LCA, i.e., g/MJ. Furthermore, they were converted into €/tkm, as different vessel efficiency, i.e., MJ/tkm, was associated with the combustion of different fuels. Secondly, the results of the externality cost are given. They are shown in terms of the functional unit, €/tkm. Thirdly, the external cost is incorporated into the fuel cost and outlined in three graphs, based on each external cost scenario. Therefore, this section will highlight fuel costs and externality costs separately in the beginning and then proceed to present the total cost data, where fuel cost and externality cost are merged.

4.1 Fuel cost

In order to evaluate the fuel economic feasibility, three trajectories are presented for each fuel. The trajectories are based on two core values, fuel cost and externality cost. Moreover, fuel cost is projected dynamically, where the cost of fuels is expected to change according to several assumptions as discussed in the methodology chapter. The fuel cost is investigated under three conditions: i) Low case, ii) Medium case, and iii) High case. The High case represents the highest fuel price and reflects discouraging conditions for the utilization of each fuel. Conversely, the Low case reflects the most optimistic case for each fuel. The Medium case is in between High and Low, and represents moderate changes in fuel price, see figure 4.1.

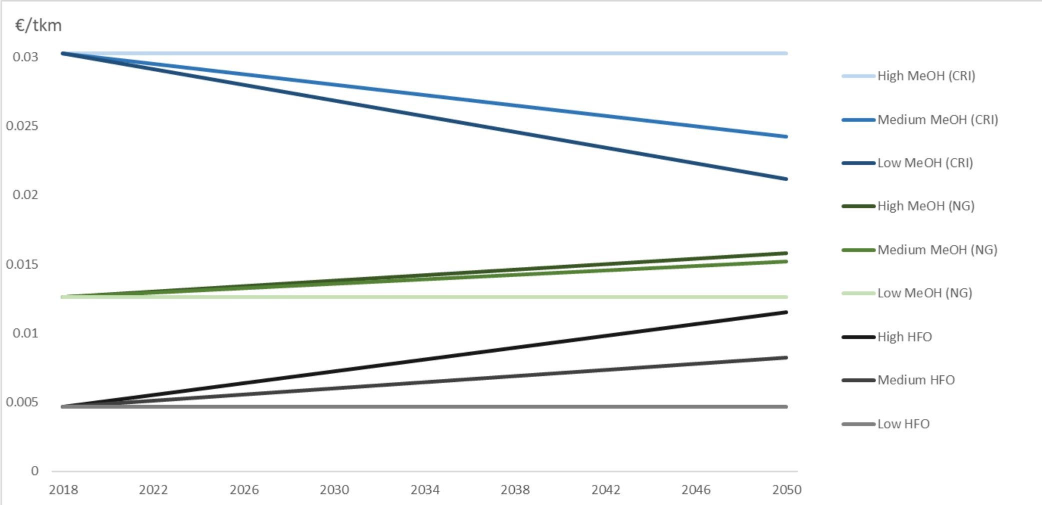


Figure 4-1 Fuel cost in terms of €/tkm in terms of low, medium and high case estimates of fuel price.

Fuel cost, according to all three conditions of fuel cost estimates, favors HFO, see figure 4.1. Furthermore, CRI methanol is not cost competitive, in terms of fuel cost alone. When comparing the Low fuel cost scenarios in 2018, cost of CRI methanol, NG methanol and HFO is 0,030, 0,012 and 0,004 €/tkm, respectively. However, in 2050, in the Low fuel cost scenario, cost of the same fuels is 0,021, 0,012 and 0,004 €/tkm, respectively. Therefore, the gap between NG methanol and CRI methanol is significantly lowered when looking at the Low fuel cost scenario in 2050. When comparing the Medium fuel cost scenarios in 2050, cost of CRI methanol, NG methanol and HFO is 0,024, 0,015 and 0,008 €/tkm, respectively. Furthermore, when comparing the High fuel cost scenarios in 2050, cost of CRI methanol, NG methanol and HFO is 0,030, 0,015 and 0,011 €/tkm, respectively. Moreover, cost of HFO ranges from 0,004 to 0,011 €/tkm, NG methanol ranges from 0,012 to 0,015 €/tkm and CRI methanol ranges from 0,021 to 0,030 €/tkm. Consequently, the trajectories never intersect.

4.2 Externality cost

Externality cost is assumed to be static during the time horizon, i.e., no change in externality cost over time. However, the results of externality cost, i.e., economic implications of the environmental and public-health impacts, were differentiated into three estimations, low, medium and high, where medium represented the mean cost estimation from the studies analyzed, and low and high represented the mean value ± 1 standard deviation. Figure 4.2 depicts the results

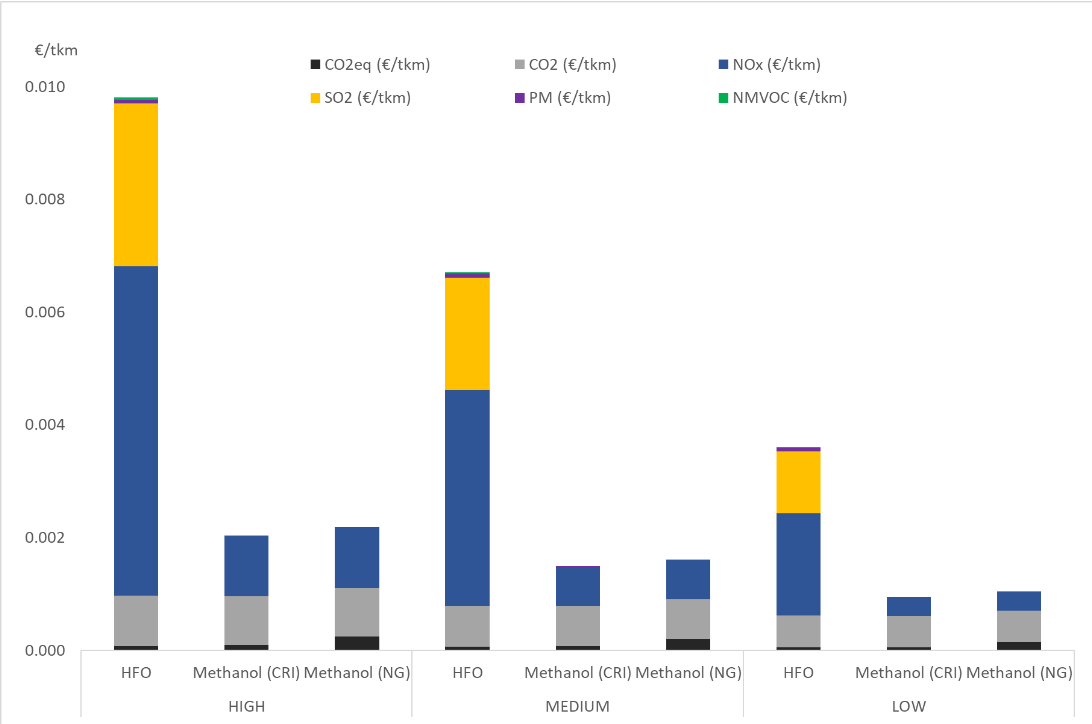


Figure 4-2 External costs of pollutants multiplied with the amount of pollutants during fuels lifecycle, presented in three dimensions, high, medium and low externality costs.

Firstly, the largest externality cost associated with the utilization of all fuels is NO_x. In the case of HFO, NO_x externality costs range from 50 to 60% of total external costs. In the case of both methanol pathways, NO_x constitutes from 32 to 52% of total externality cost of the fuels. The second largest externality cost is SO_2. The proportional cost of SO_2 is between 29 and 31% for HFO. However, no SO_2 emissions are in the methanol cases and therefore account for 0% of externality cost. Furthermore, NO_x and SO_2, contributed 81 to 89% of total external costs imposed by HFO utilization and are therefore the main contributors of externality cost in the case of HFO utilization.

In figure 4.2, CO_2 emissions are differentiated between CO_2 and CO_2eq, where the latter accounts for WTT emissions. In the case of methanol, CO_2 is the second largest contributor of externality cost. In the case of CRI methanol, CO_2 from fuel combustion constitutes between 43 to 59% of total externality cost, while CO_2eq constitutes between 4 to 6% of total externality cost. In the case of NG methanol, CO_2 from fuel combustion constitutes between 39 to 53% of total externality cost, while CO_2eq constitutes between 11 to 15% of total externality cost. This is due to the fact that CO_2eq emissions from NG methanol are significantly higher than that of CRI methanol, i.e., 4 and 10 g/tkm, respectively.

PM and NMVOC externality cost are low for all fuels. PM externality cost constitutes between 0,7 and 1,9% of total externality cost for HFO. For CRI methanol, PM constitutes between 0,2 and 0,3% of total externality cost and between 0,1 and 0,3% of total externality cost for NG methanol. NMVOC constitutes between 0,1 to 0,3% of total externality cost for HFO. Moreover, no NMVOC emissions are in methanol combustion and therefore constitute 0% of externality cost for methanol.

4.3 Total cost

The results of the fuel cost estimations, relying on a conventional CBA show that HFO is the least expensive fuel option, in all three fuel cost cases, looking at the whole-time horizon, i.e., from 2018 to 2050. However, as externality cost is added onto fuel cost, the total cost of fuels is estimated using an extended CBA. In this case trajectories start intersecting, which shows the point in time when methanol becomes cost competitive. Total cost of fuels is represented in three separate scenarios, each containing a different externality cost assumption, i.e., high, medium, low externality cost, see figures 4.3, 4.4, and 4.5.

Scenario 1: Externality cost is High, see figure 4.3. In this scenario, the high externality cost proportionally affects total cost the most. Therefore, CRI methanol proportionally benefits the most, and receives the lowest total cost out of all three scenarios of CRI methanol. This is due to the fact that emission costs are the lowest for CRI methanol. However, with the addition of High externality costs, the total cost of CRI methanol does not become lower than other total cost trajectories. In this scenario, NG methanol is the most cost competitive, with a lower total cost than HFO, in the High fuel cost case, after 2021 and onwards, and in the Medium fuel cost case, after the year 2028 and onwards, see figure 4.3.

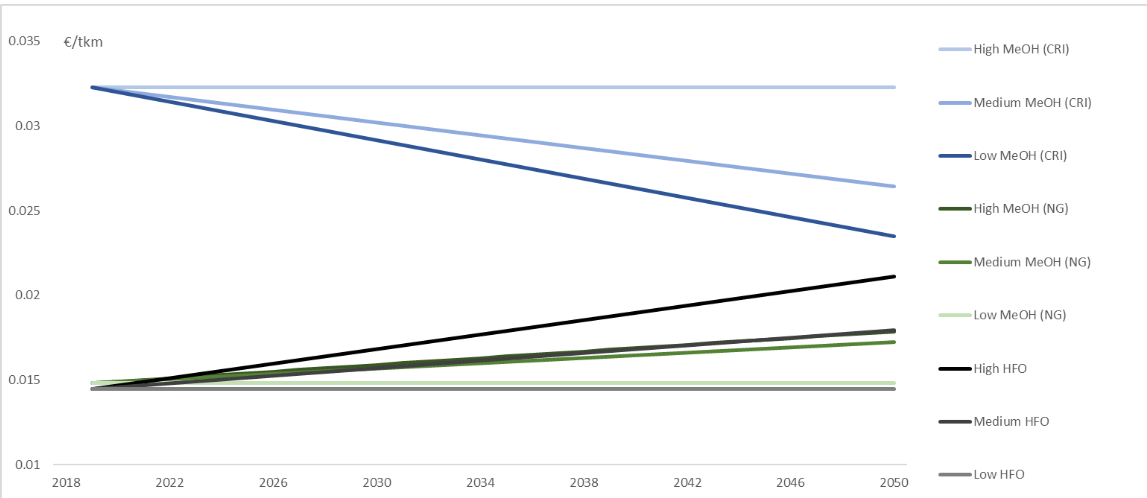


Figure 4-3 High external cost scenario: Total cost in terms of €/tkm, of low, medium and high case fuel cost estimates.

Scenario 2: Externality cost is Medium, see figure 4.4. In this scenario, the medium externality cost affects total cost less than in scenario 1. Moreover, NG methanol becomes cost competitive, in scenario 2, with a lower total cost than HFO in 2043 and onwards in the High fuel cost case. However, HFO is the most cost competitive in the Medium and Low fuel cost cases, throughout the study period.

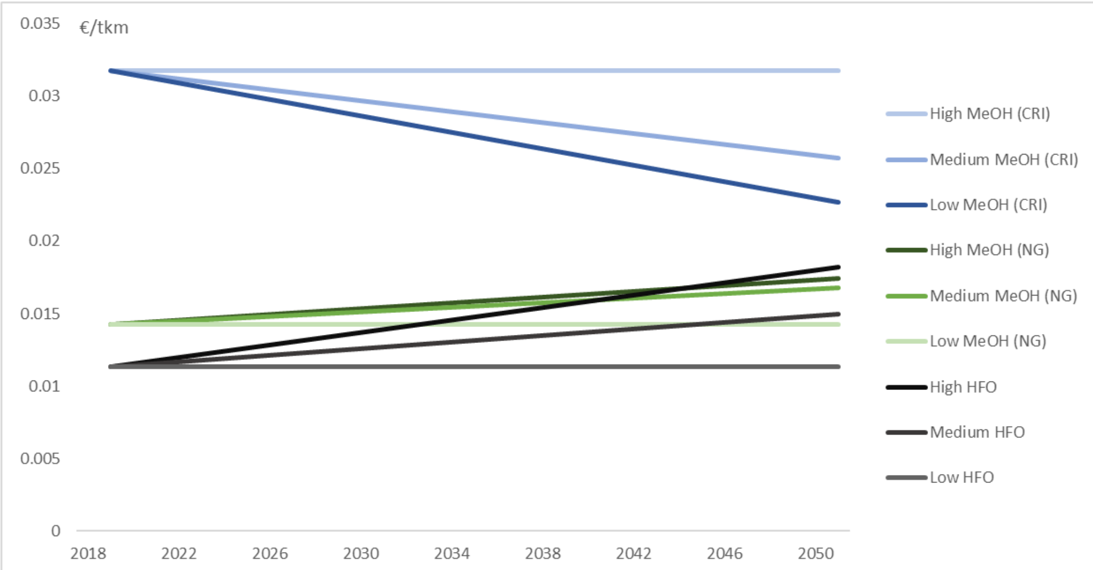


Figure 4-4 Medium external cost scenario: Total cost in terms of €/tkm, of low, medium and high case fuel cost estimates.

Scenario 3: Externality cost is Low, see figure 4.5. In this scenario, the low externality cost proportionally affects total cost the least. Therefore, HFO benefits the most. Moreover, HFO remains with the lowest total cost throughout the study horizon, see figure 4.5.

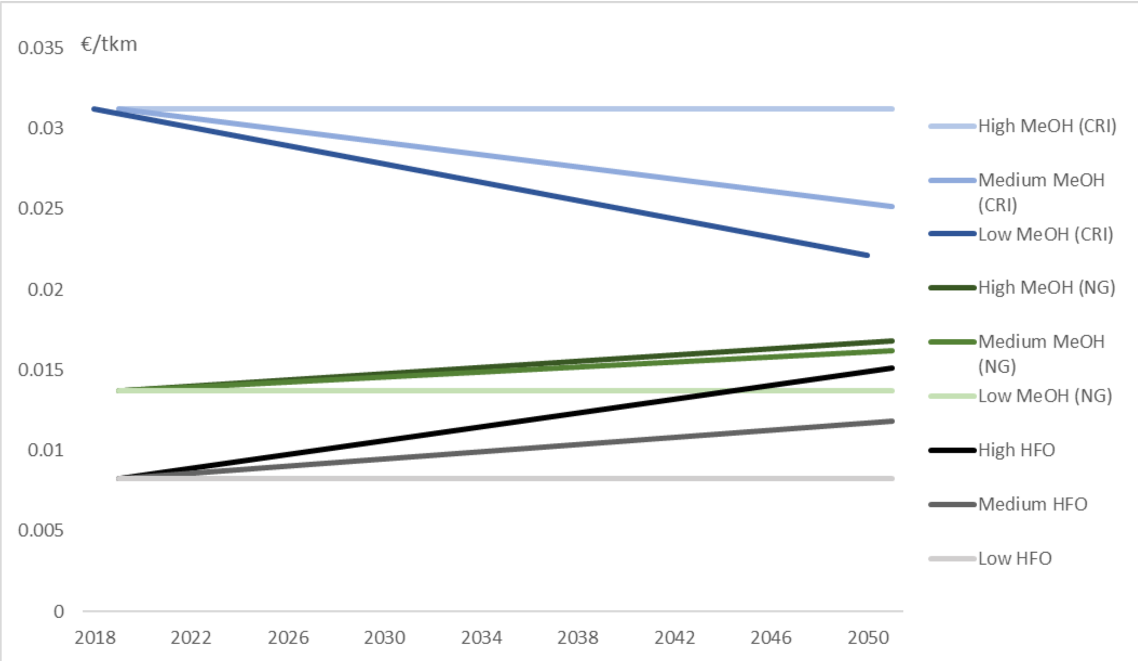


Figure 4-5 Low external cost scenario: Total cost in terms of €/tkm, of low, medium and high case fuel cost estimates.

In the case of High estimates of externality costs, NG methanol is nearly cost competitive in 2018, where total cost of NG methanol ranges between 102 to 165% of HFO total cost, see figures 4.3, 4.4, and 4.5. NG methanol's prospects are further improved towards 2050. Consequently, NG methanol becomes cost competitive in 2021, following the High change in fuel price and High externality cost trajectories, and in 2032, following the medium change in fuel price and high externality cost trajectories, see figure 4.3. However, HFO remains the most cost competitive option, throughout the study horizon, following the low change in fuel price and high externality cost trajectory, see figure 4.3. Moreover, High external cost scenario, figure

4.3, proposes the most favorable conditions for methanol, both NG and CRI. Therefore, HFO cost competitiveness is improved in figure 4.4. and figure 4.5, where lower externality costs affect the total cost. Moreover, NG methanol only becomes cost competitive in 2043 and onwards, following the Medium externality costs and High fuel cost change, see figure 4.4. Therefore, HFO has the lowest total cost throughout the study period in terms of Medium externality cost and Medium and Low fuel cost change, see figure 4.4. Furthermore, HFO has the lowest total cost in all trajectories, following the Low externality cost, see figure 4.5.

CRI methanol is always the least cost competitive options in all trajectories, i.e., low to high external costs and low to high change in fuel cost. This ranges from 115 to 275% cost increase, from HFO to CRI methanol, in 2018. However, in 2050, a switch from HFO to CRI methanol would only increase fuel cost by 9,5%, in CRI methanol's most favorable trajectory i.e., High fuel cost change and High externality cost and HFO's least favorable trajectory, High fuel cost change and High externality cost, see figure 4.3.

5 Discussion

5.1 Cost competitiveness

The objective of this thesis was to assess the value of environmental externalities of three maritime fuels in the context of fuel cost from 2018 to 2050. The primary research question of this thesis was:

How cost-competitive are CRI's methanol and conventional (NG) methanol compared to HFO, when cost of externalities, associated with the life-cycle emissions of all fuels is incorporated into the fuel price?

The approximate market prices for CRI methanol, NG methanol and HFO, were assessed in chapter 3.2. The economic implications of the environmental externalities, associated with the different fuel types, measured in €/tkm, was assessed in chapter 4.2. With the aggregation of these data points, economic valuation of fuel cost was linked explicitly to the externalities related to the combustion properties of the fuels. Finally, the production capacity of CRI methanol in Iceland, was assessed in chapter 2.2.3.

The results show that CRI methanol is not cost competitive, with or without the externalities of life-cycle emissions incorporated into its fuel price. However, NG methanol becomes cost competitive, in High and Medium fuel cost trajectories, in 2021 and 2028, respectively, following a High externality cost assumption. Furthermore, NG methanol is cost competitive in 2043, in the High fuel cost trajectory, following a Medium externality cost assumption.

The total cost of CRI methanol is due to the fact that this is a pilot plant, and by definition costs tends to be higher in such plants as "learning" has not taken place. However, from the perspective of reduced capital costs via learning rates, i.e., in the best-case scenario for CRI methanol, i.e., where High externality costs are assumed, the fuel cost increase of a transition from HFO to CRI methanol is only 9% in 2050, in the case of High price change of HFO and CRI methanol. However, the price increase of transitioning to CRI methanol from HFO in 2020, is 112%. Therefore, even with the expected price increase of a fuel switch within an ECA, i.e., 70-90% fuel cost increase, CRI methanol does not become cost competitive over the time-period analyzed.

5.2 Practical and policy implication

The results presented in this thesis are an addition to the ongoing debate (Bengtsson, 2011; MEPC 70, 2016) about transitioning to new energy carriers within the maritime sector. Utilizing methanol for use in maritime engines is technologically viable. The infrastructure for storage and distribution of methanol can be adapted easily relative to gaseous alternatives due to the liquid state of methanol under ambient conditions. A transition towards renewable methanol from HFO could be facilitated with NG methanol as a complement. However, this study illustrates that economic barriers are substantial for CRI methanol in Iceland, compared to NG methanol and HFO.

A transition to methanol in the maritime sector would be almost sulfur free and thus sufficient to meet IMO's Annex VI in 2020, both in terms of global sulfur requirements and in terms of ECAs. Furthermore, this criterion, of being sulfur free, will become increasingly valuable as more areas are being declared as ECAs. The cost increase related to the shift from a fuel with 1,5% to 0,5% sulfur content is likely to induce a price increase in bunker costs of 20 to 30% (ECSA, 2010). Moreover, a shift from 0,5 to 0,1% sulfur content fuel, is estimated to increase bunker costs by 50 to 60%. Therefore, the aggregate increase in bunker prices, when adjusting to the IMO's 2020 Annex VI on sulfur in ECAs, will amount to 70 to 90% (ECSA, 2010). Methanol is sulfur free, which means that vessels utilizing methanol, will be able to comply with both the global sulfur emissions regulations, as well as the ECAs. Furthermore, CRI methanol will become cost competitive in 2035 in ECAs if fuel cost will be 70% higher due to sulfur regulations, and in 2046 if fuel cost will be 90% higher. Moreover, NG methanol is cost competitive throughout the study horizon, in all externality cost scenarios and all fuel cost cases, if a 70% to 90% fuel cost increase is estimated. However, Icelandic waters are not defined as ECA but are subject to the IMO regulation Annex VI, of a global sulfur cap of 0,5%. Therefore, if bunker prices will increase by 20 to 30%, by meeting this regulation, NG methanol is cost competitive in all fuel cost cases in the High and Medium externality cost scenarios.

Iceland's Climate Action Plan 2018 – 2030, states their willingness to eventually phase out HFO. Therefore, a transition towards methanol would pave the way towards that goal. However, as was pointed out earlier in this thesis, Iceland aims to have 10% renewable energy in the maritime sector before 2030. In order to make a transition to CRI methanol more economically feasible for maritime operators, the utilization of financial support instruments would be required. This has been done before in the case of electric-vehicles (EV) in Iceland, where the value-added-tax was omitted from the cost of new EVs, in order to facilitate a faster transition towards EVs.

Alternatives most commonly considered by the maritime industry, in the short term, are low sulfur petroleum fuels and/or EGCS (CE Delft, 2016). These fuels and technologies can hardly be viable as a long-term solution given the advent of diminishing fossil fuel reserves and the regulations set by nations on utilizing renewable energy. According to the MEPC 70 report it is estimated that consumption of alternative marine fuels, such as LNG, methanol, biofuels, liquid propane gas (LPG) and dimethyl ether by 2020 will be negligible (MEPC 70, 2016). This represents in some sense a *technological lock-in*, which arises due to lack of infrastructure, development and general inertia in relation high investment cost in the beginning stages of an emerging technology. The shift towards methanol in the maritime sector is a long-term solution and should be considered in that context. Furthermore, the production capacity of CRI methanol is currently 78 TJ, which amounts to 0,8% of the maritime sector's total energy demand. In order to produce 10% of the energy content required by the maritime sector, more methanol plants would be required. However, the high availability of effluent carbon, water and renewable electricity, would make that project possible. The amount of geothermal effluent carbon needed for the production of 991 TJ of methanol, could be generated by two large geothermal power plants. Moreover, the need of electricity to produce 991 TJ of methanol would be approximately 3% of the total production capacity of renewable electricity in Iceland. However, more research and development of electrolysis for hydrogen production is needed as the economic efficiency of this process still favors conventional methods for hydrogen production i.e., from fossil fuels. Furthermore, carbon capture and utilization are still in its early stages of development. More research and efficiency improvements are required in order to make the CRI pathway of methanol production cost competitive.

The transition towards a new maritime fuel in Iceland is likely not necessarily to depend on CRI methanol. Currently, the main focus of maritime operators, is on LNG. However, if the debate on maritime fuel should be focused on renewable energy, methanol is an interesting option. Moreover, other renewable feedstocks can be utilized for methanol production, such as biomass, which is available in Iceland.

5.3 Study limitation

Some limitations in the methodology chapter of this thesis were identified. Firstly, there is a considerable degree of limitation associated with the lack of transferability of European emission cost estimates to the Icelandic context. This is likely to affect the externality cost estimates negatively and produce lower economic cost outputs for the environmental externalities associated with the fuels. Furthermore, the lack of research on the effects of externalities in the context of maritime emissions, is a source of limitation for this thesis and impacts the accuracy of the estimated cost values for environmental externalities.

The first step of an extended CBA of fuel in the maritime context is to analyze the amount of ship emissions, based on the fuel types and combustion methods. Secondly, the total external costs are calculated by multiplying the emission by their corresponding marginal external cost (Jiang, Kronbak & Christensen, 2010). In both steps, there is a degree of plausible issues regarding validity and reliability. In the first step, the combustion characteristics are straight forward. Furthermore, Brynolf, Fridell & Andersson's (2014) and Elliss & Svanberg's (2018) results on methanol's combustion characteristics, are commensurable, even though they were comparing different engine types i.e., 14 MW vs 1,3 MW. However, very few studies have outlined the combustion characteristics of methanol in a dual fuel compression ignition engine, and even fewer researcher groups have tested it, themselves. This is due to the fact that methanol has only been utilized and assessed as a maritime fuel for a few years. Furthermore, N_2O and CH_4, are not found in the maritime externality cost literature. However, they are greenhouse gases and could be monetized relative to CO_2, based on the global warming potential (GWP) of these emissions according to the IPCC's Fifth Assessment Report (AR5), i.e., 28-36 and 265-298, respectively. However, if they were incorporated into the external costs of fuels, via the multiplication of CO_2 cost with the respective GWP factor, they would play an insignificant role, i.e., <5% of total external cost of HFO. Moreover, the only emission factors that were included in the external cost analysis were the ones which both existed in the maritime externality literature and the life-cycle analysis of the fuels. Furthermore, the WTT analysis of CRI methanol only included CO_2 equivalents. Therefore, other pollutants associated with the production of CRI methanol were omitted from the external cost of CRI methanol, due to the fact that information was inaccessible, even though they were available for NG methanol and HFO.

The second step of the extended cost benefit analysis i.e., calculating the economic implications of the pollutants, has more potential of validity issues occurring. However, internalization of external costs is a critical issue in European transport research and policy development (Maibach et al., 2008). Furthermore, without policy intervention, external costs are not considered by fuel users leading to incorrect incentives and a distorted estimate of welfare gains and losses. Therefore, beginning to analyze these costs and incorporating them into fuel appraisal is important. Validity in this context refers to how well the study measures the economic implications of pollutants. The study would be valid of the economic damages attributed to each pollutant is caused by them. However, this can be hard to estimate, since the causality between pollutants and public-health or environmental degradation, is not always clear. Furthermore, little research has emerged on the topic of the economic implications of pollution Iceland, other than the effects of volcanic eruptions (Hlodversdottir et al., 2012) and the effect of hydrogen sulphate (H_2S) (Finnbjornsdottir et al., 2016). Therefore, future research is needed on the topic of environmental externalities and especially in terms of maritime emission, close to shore an in port.

Moreover, linear damage functions ignore the possibility of tipping points, which can involve exponential impacts to human health impacts at certain concentrations. However, the IPA approach of the HEATCO and CAFE projects, which was adopted in this thesis, was considered the best practice values (Jiang, Kronbak & Christensen, 2010). Furthermore, the limited number of studies, in the context of maritime externality literature, did play a role in the lack of validity of this thesis.

The maritime externality studies provide cost estimates in relation to the proximity of vessels to densely populated areas, differentiated into either two are three categories i.e., *in port*, *near port* and *at sea*, where near port and in port estimates are omitted from this thesis. Therefore, this study relies solely on *at sea* estimates. Moreover, a lack of transferability to smaller European nations, where population is less dense, is a considerable contributor to the lack of validity in this thesis. Furthermore, IPA estimates for less populated cities i.e., 50.000 – 100.000, and data on the amount of time spent by vessels in free sailing, maneuvering and berthing, would be required to safely add the near- and in port externality cost estimates into the study. Holland & Watkiss (2004) and Tzannatos (2011) evaluated the marginal external costs of air pollution in Europe and Greece, respectively. In their modelling they emphasize maritime pollution and externality cost relative to distance from impact areas e.g., densely populated areas and cropland. This method is extrapolated from the impact pathway approach, i.e., modelling emissions relative to dispersion of chemicals and exposure of sensitive receptors and impacts based on exposure response functions. The external effects that were analyzed by Holland & Watkiss (2004), were short-term acute effects of PM_{10}, SO_2, and ozone on mortality and morbidity and chronic long-term effects of PM_{10} on mortality and morbidity, to the extent that these short-term and long-term effects have been reported. Moreover, they included the effects of SO_2 and acidity on materials used in buildings and structures and effects of ozone on arable crop yield. Finally, the most relevant conclusion from their model is methodology to extrapolate effects from the urban setting over to the maritime context. In that aspect, they assume that emissions externalities in port are the equivalent of urban externality costs and that emissions close to shore are to be calculated based on a nation's expected rural externality cost (Holland & Watkiss, 2004). Furthermore, rural externality cost of NOx and VOCs is expected to be commensurable to urban externality cost for the same pollutants due to the fact that their quantified impacts are generated through the formation of secondary pollutants when those chemicals react to other chemicals in the atmosphere, i.e., ozone and nitrate aerosols, but their externality cost is not affected much by local variation in population density (Holland & Watkiss, 2004).

6 Conclusion

This extended cost benefit analysis outlines the economic feasibility of introducing an alternative fuel into the maritime sector in Iceland, i.e., methanol. Subsequently, the economic implications of the environmental impacts, were analyzed through maritime externality literature. Moreover, all fuel pathways were assessed in terms of market price or estimation thereof.

The results illustrate that the total cost of CRI methanol ranged from 0,031 to 0,032 €/tkm in 2018 and from 0,022 to 0,032 €/tm in 2050. The total cost of NG methanol ranged from 0,013 to 0,014 €/tkm in 2018 and from 0,013 to 0,018 €/tkm in 2050. The total cost of HFO ranged from 0,008 to 0,014 €/tkm in 2018 and from 0,008 to 0,021 €/tkm in 2050. Therefore, CRI methanol is assessed as not being cost competitive, seeing that the best-case scenario for CRI methanol and worst-case scenario for HFO, still favors HFO by 0,001 €/tkm. However, the results show that NG methanol is expected to become cost competitive in the early years of the next decade, as it yields a lower total cost than HFO, at the earliest in 2021.

Reducing maritime emissions is an urgent global priority. Assessing and regulating pollutants in terms of their spatial distribution and effects on ecosystems and public health are some of the analysis and mitigation strategies that can be utilized to facilitate pathways towards enhanced sustainability of the maritime sector. For a sustainable transition towards alternative maritime fuels, however fuel appraisal must be inclusive of cost values of externalities when comparing the economic competitiveness of different alternatives that can comply with various national and international policy objectives.

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Appendix 1

NOMINAL												
Study	Country/Region	Diffrentiation	Unit	Year	CO2	NOx	SO2	SOx	PM	PM2,5	VOC	NMVOC
(Kotowska, 2017)	EU	open sea	Euro/tonne	2018		3160		4496	1458			608
	EU	port	Euro/tonne	2018		5347		6806	31598			1215
(Holland & Watkiss, 2004)	Eastern Atlantic	open sea	Euro/tonne	2018		6542	6133			12402	2044	
	Baltic Sea	open sea	Euro/tonne	2018		2862	2181			3407	1363	
	English Channel	open sea	Euro/tonne	2018		7360	8041			16355	2590	
	Nothern Mediterranean	open sea	Euro/tonne	2018		8450	6406			13629	2317	
	North Sea	open sea	Euro/tonne	2018		4225	5860			13084	3544	
	City of 100.000 people	port	Euro/tonne	2018			8177			44976		
	City of 200.000 people	port	Euro/tonne	2018			16355			89951		
(Jiang, Kronbak, & Christensen, 2010)	North Sea	open sea	Euro/tonne	2018		6790	9186			37287		
(Maibach et al., 2008)	Baltic Sea	open sea	Euro/tonne	2018		3544	5043			16355		555
	Mediterranean Sea	open sea	Euro/tonne	2018		681	2726			7632		333
	North East Atlantic	open sea	Euro/tonne	2018		2181	2998			6542		444
	North Sea	open sea	Euro/tonne	2018		6951	9404			38161		2108
(EU ETS, Average 2018)			Euro/tonne	2018	16							
(EU ETS, Average 2019)			Euro/tonne	2018	22							
Average (open sea)			Euro/tonne	2018	**19**	**4795**	**5798**	**4496**	**1458**	**16485**	**2371**	**809**