Utrecht University

Greenhouse gas footprint of biodiesel production from used cooking oils

A Life-Cycle Assessment

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April 6th, 2018

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Abstract

The methodology used in the EU Renewable Energy Directive to assign default values to biofuel greenhouse gas emissions and reduction potentials has its shortcomings. These undermine its practical application in estimating actual values. This thesis proposes a more realistic greenhouse gas footprint of biodiesel based on used cooking oils. This greenhouse gas footprint of used cooking oil sourced hydrotreated vegetable oil has been calculated using an attributional life cycle assessment. Through a market analysis it is shown that hydrotreated vegetable oil production in the Netherlands is increasing due to its favourable attributes when compared to alternatives such as FAME. As a consequence of increasing policy pressure regarding waste based biofuels, used cooking oil has become a major feedstock for biodiesel production. Further analysis showed that used cooking oil demand in the Netherlands is higher than the available supply which leads to increased imports the feedstock from overseas origins. The main finding of this thesis is that the greenhouse gas footprint of used cooking oil based biodiesel depends on the origin of the feedstock and the subsequent transportation mode. Bulk shipping of used cooking oil using chemical tankers is a worse option than using flexitanks, with regards to greenhouse gas emissions. Likewise, used cooking oil imports from China show higher greenhouse gas emissions than US imports. But both options show inferior performance when compared to domestic and intra-EU used cooking oil collection, with the latter being the cleanest choice. The range of GHG emissions associated with UCO-transportation for hydrotreated vegetable oil production found in this thesis is 2.19 to 14.5 gCO_2-eq/MJ NExBTL.

Acknowledgement

I would first like to thank my thesis supervisor dr. Ric Hoefnagels of the Copernicus Institute of Sustainable Development, who offered me sustained encouragement and guidance throughout the course of this thesis. His enthusiasm towards the subject really pushed this thesis forward. Whether I had a pressing question or felt lost in the sea of used cooking oils, Ric promptly provided valuable advice.

I would also like to thank the experts who were able to provide be with crucial information regarding used cooking oil trade and transportation. In particular Lars Huizenga and Bram van Santen of Simadan who kindly gave me the opportunity to carry out an interview. My appreciation also to the other experts who answered my questions via e-mail, without their input this research would not have been possible.

I would also like to acknowledge dr. Li Shen of the Copernicus Institute of Sustainable Development as the second reader of this thesis. In addition, I thank Robert Haffner of Ecorys and prof. dr. Martin Junginger of the Copernicus Institute of Sustainable Development for piquing my curiosity in the field of (waste based) bioenergy.

Finally, I take this opportunity to express my gratitude towards my loving and supporting family and friends. In particular, my everloving parents.

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Nomenclature and Abbreviations

ALCA	Attributional Life Cycle Assessment	HS-code	Harmonised System code
BE	Belgium	HVO	Hydro treated Vegetable Oil
BSE	Bovine spongiform encephalopathy	IBC	Intermediate Bulk Container
BSFC	Brake Specific Fuel Consumption	ILUC	Indirect Land-use Change
CBS	Centraal Bureau voor de Statistiek	IMO	International Maritime Organisation
CCWG	Clean Cargo Working Group	ISCC	International Sustainability & Carbon Certification
CHP	Combined Heat and Power	ISO	International Organisation of Standardisation
CLCA	Consequential Life Cycle Assessment	JRC	Joint Research Centre
CN	China	LCA	Lifecycle Assessment
CN-code	Combined Nomenclature code	LHV	Lower Heating Value
COP21	2015 United Nations Climate Change Conference	LUC	Land-use Change
DE	Germany	MCR	Maximum Continuous Rating
dwt	Deadweight Tonnage	NL	Netherlands
EC	European Commission	NREAP	National Renewable Energy Action Plan
EF	Emission Factor	RED	Renewable Energy Directive 2009/28/ EC
ELCD	European Life Cycle Database	REDII	Renewable Energy Directive II proposal
EU	European Union	SFOC	Specific Fuel Oil Consumption
EURO1-5	European Emission Standards	SMR	Steam Methane Reforming
FAME	Fatty Acid Methyl Esters	TEU	Twenty Foot Equivalent Unit
FFA	Free Fatty Acids	TTW	Tank-To-Wheel
FQD	Fuel Quality Directive 2009/30/EC	UCO	Used Cooking Oil
GHG	Greenhouse Gas	US	United States
GWP	Global Warming Potential	WTT	Well-To-Tank
HFO	Heavy Fuel Oil	WTW	Well-To-Wheel

1 Introduction

To meet climate targets of maximum 2 °C temperature rise above pre-industrial levels, as agreed on at the COP21 in Paris in 2015, and improving energy supply security, the shift from fossil energy to renewable energy sources is increasingly urgent. (International) transportation of goods and people is a socio-economic phenomenon which is projected to grow in the coming years. Mainly aviation-, marine- and heavy road transportation are contributors to the growing globalisation. The finite nature of fossil fuels compels the international community to develop alternative ways of powering the world's growing number of motor vehicles. This notion is strengthened by increasing concerns over the negative environmental impact associated with the growing use of fossil fuel-related greenhouse gas emissions (Singh et al., 2015). Biodiesel, an alternative fuel in the transport sector, exhibits great potential in mitigating the requirements for fossil fuel resources (Demirbas, 2007). Biodiesel can be produced from many feedstocks, including over 350 types of oil-bearing crops (ibid.). However, the consumption of oil feedstocks for bioenergy including palm oil from Southeast Asia and soy oil from Latin America has raised sustainability concerns. These include food security, land conflicts and uneven environmental impacts that have emerged in a contemporary conflict surrounding biofuels (Raman & Mohr, 2014). The demand for alternative feedstocks which do not directly compete with food-security has grown rapidly as a feedstock for biodiesel production. An example is a use of biodegradable waste in the form of used cooking oils (UCO) (ibid.). However, one must take notice that using such feedstocks for biodiesel production does not automatically circumvent debates regarding their sustainability. Concerns about the passing of un-used oils as waste, market disturbance and lack of traceability and monitoring procedures have arisen (C. Goh et al., 2013).

In the European Union (EU), the Renewable Energy Directive (RED) requires 10 percent of all transport fuels to be delivered from renewable sources by 2020, of which more than 85 percent is expected to come from biofuels. Biodiesel accounted for 78.2 percent of the total final consumption (by energy) of biofuels in the European transport sector in 2013. Currently, the EU is the largest producer of biodiesel worldwide, with a share of around 40 percent of global biodiesel production (European Commission, 2015). Future energy demand in the transportation sector is projected to decrease according to the National Renewable Action Plans (NREAP), published in 2011, while the share of bio-based fuels will increase according to these targets. In other simulations total renewable biofuel demand in the transport sector are forecasted to increase as well (Hoefnagels, Resch, Junginger, & Faaij, 2014).

Several European policy- and legal instruments exert pressure on the usage of UCO as a feedstock for biodiesel production. The two main instruments are the Renewable Energy Directive 2009/28/EC and the Fuel Quality Directive (section 2.1). Likewise, an amendment has been proposed and entered into force on October 5, 2015, which aims to prevent indirect land use change (ILUC) resulting from increased biofuel production. This amendment also contributes to the increased UCO usage by including an element which enables double counting of the energy contribution of advanced biofuels towards the 10 percent blending target for 2020 for the member states. The introduction of this double counting mechanism in the RED, which enabled waste-based biofuels to be counted twice towards the annual obligation of renewable transport fuels, can be seen as one of the major drivers of European UCO usage.

As a result of the introduced double counting measure, Europe has seen a sharp increase in the consumption of biodiesel produced from UCO (Boutesteijn, Drabik, & Venus, 2016). In the Netherlands, this effect can be observed as well (C. S. Goh, Junginger, Mai-Moulin, & Junginger, 2016). A similar trend occurred in the UK, where in addition to the double counting measure all duty subsidies for biofuels were removed with the exception of UCO (C. Goh et al., 2013). This resulted in a dramatic increase in UCO usage and sequentially a price hike where UCO prices in some cases exceeded those of virgin oils (ibid.). If produced from certain renewable feedstocks, such as UCO, Fatty Acid Methyl Esters (FAME) biodiesel is qualified for the double counting mechanism. FAME consumption in the Netherlands rose from 3436 TJ in 2011 to 9671 TJ in 2015, with a majority of the market share of biofuel supply in 2015. Hydrotreated Vegetable Oil (HVO) consumption, also eligible for double counting if produced from selected feedstocks, rose from 1.65 TJ to 215 TJ in 2015. An increase of nearly 13,000 percent in four years (Dutch Emissions Authority, 2016). Note that these values exclude the double counting mechanism. Also note that FAME and HVO biodiesels have a variant which is not eligible for double counting on the market, but these variants have decreased in usage over the same period (Dutch Emissions Authority, 2016). In this research, HVO as a fast-growing alternative to FAME, will be investigated. Installed production capacity of HVO refineries in Europe has increased tenfold between 2010 and 2017 (Flach, Lieberz, Rondon, Williams, & Wilson, 2016).

To ensure a net positive impact of its biofuel policy, the EU has set mandatory sustainability criteria for liquid biofuels. The minimum greenhouse gas (GHG) emission saving performance is a key sustainability indicator (Schueler, Weddige, Beringer, Gamba, & Lamers, 2013). A useful tool to assess lifetime carbon reporting is a Life Cycle Assessment (LCA). However, while providing important information, such an assessment is accompanied by several caveats which have a large influence on the outcome and its interpretation. Moreover, the LCA method in the EU RED shows signs of misinterpretation and inconsistencies (Soimakallio & Koponen, 2011), which will be discussed in section 1.1.2.

1.1 Problem Definition

The problem, a possible weak GHG reduction performance of biodiesel, can be seen as a result of an increasing policy pressure on using UCO as a biofuel feedstock (1.1.1) based on assumptions which are not coherent (1.1.2).

1.1.1 European biofuels policy effects

As a result of the increased policy pressure, UCO as a feedstock for biodiesel production in the EU increased from 0.5 million tonnes in 2010 to an expected 2.3 million tonnes in 2017 (Flach et al., 2016), note that other estimates are more conservative as can be seen in Figure 1. However, this increase in

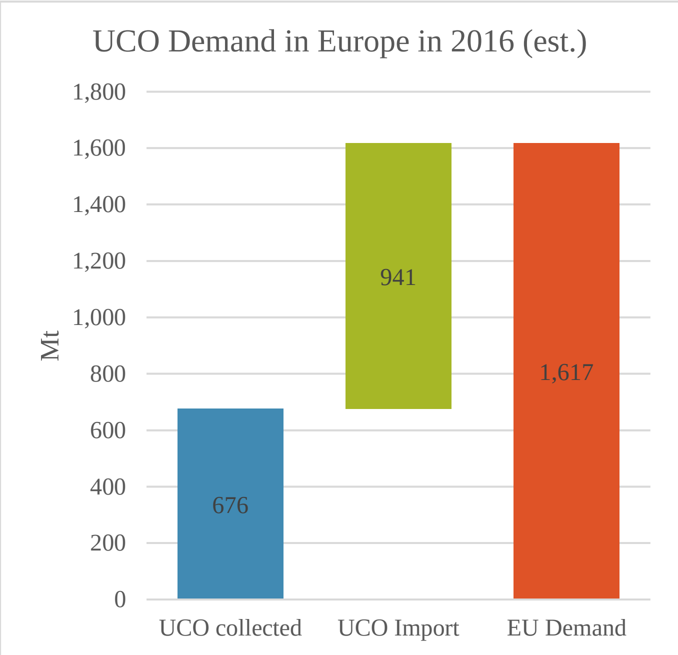


Figure 1 UCO Demand in Europe adapted from: (Hillairet et al., 2016)

demand for UCO seems to exceed the supply of UCO within the EU and a certain degree of feedstock imports is necessary to meet the RED blending target (Peters, Toop, van den Bos, & Spottle, 2013). This is illustrated in Figure 1. This trend can add to the transport emissions of the biofuel, which is an implication of the EU renewable energy policies. As UCO demand increased, a strong growth in UCO price can be observed as well. Stabilisation occurred in 2014 at around €530 per tonne, but in recent years the price has again increased to an average of €655, with December 2016 spiking at €810 per tonne (Hillairet, Allemandou, Golab, & Castaing, 2017).

Future policy effects can be attributed to the introduction of the RED II, which determines the policy course from 2020 onwards. Included in this directive is a cap on food-based biofuels which lies at 3.8 percent in 2030, and a -70 percent GHG reduction threshold for biofuels (Hillairet, Allemandou, & Golab, 2016). For UCO as a feedstock, a proposed target cap of 1.7 percent of the fuel mix in 2030 is the most significant (International Council on Clean Transportation, 2017). This cap could potentially hinder UCO biodiesel development past 2021.

1.1.2 Limitations of GHG-LCA calculations under the Renewable Energy Directive/ Fuel Quality Directive

The RED provides default values for GHG emission reductions of certain biofuels when compared to a reference fossil fuel. When comparing these values to values found in scientific research they tend to be on the lower side, especially when comparing values for FAME production from palm oil (Soimakallio & Koponen, 2011). Regarding the use of waste as a feedstock the RED is inconspicuous about the definition of 'waste or residues', this may have an indirect impact about the outcome as organic wastes and residues are not properly defined in the RED. Furthermore, avoiding generation of waste streams likely has a larger impact on GHG emission reduction then utilising waste stream (ibid.). This effect is enhanced when the price of waste exceeds that of the virgin product, which occurred with UCO (C. Goh et al., 2013).

An additional remark is that the allocation used in the RED methodology is largely done on the basis of energy contents, the lower heating value (LHV), of the (co-)products (Soimakallio & Koponen, 2011). This methodology may be flawed, as not all (co-)products are used for energy production purposes, and allocation on economic valuation could be the more suitable option for certain products (Guinée, Heijungs, & Huppes, 2004). Co-production of for example glycerol and fertilizer during FAME production (Talens Peiró, Lombardi, Villalba Méndez, & Gabarrell i Durany, 2010) likely influence the use, price and availability of competing products if sold. Note that this product substitution tends to have a decreasing impact on the total GHG emissions from biofuel production (Soimakallio & Koponen, 2011). A generic illustration of the spatial system boundary according to the RED methodology, which excludes indirect (market) effects, can be found in Appendix I.

Mortimer, Hatto, & Mwabonje (2015) identify several other shortcomings in the RED/FQD methodology, such as the lack of guidance for an emission factor of a hypothetical combined heat & power (CHP) plant used for energy during biofuel production, and a co-product allocation which is not applied universally. Also, it is unclear whether the emissions associated with the operations of blending depots and filling stations involved in delivering biofuels to final users should be included in calculations.

Additionally, it has been suggested that European UCO imports can contribute to an extent to ILUC with accompanied emissions (C. Hamelinck & Zabeti, 2016). If the UCO has to travel half a globe to be processed into biodiesel, how much impact does this have on the CO_2 mitigation potential? How does this relate to the double counting measure? Furthermore, what are alternative uses for UCO, and is it truly a waste product? These unknowns will be researched in this thesis.

1.2 Research Aim

If one looks at the pathway of waste oil to biodiesel in the RED, there is a proposed default GHG emission saving of 83 percent and a typical GHG emission saving of 88 percent (European Parliament, 2009). However, further investigation of the methodology used in Annex V in the directive indicates several deficiencies which are best described by the following quote.

> *…deficiencies which undermine its practical application in estimating actual values and its use in deriving existing and proposed default values. Fundamentally, these deficiencies are due to failures in applying the strict principles of life cycle assessment (LCA) during the development of the RED/FQD methodology"* (Mortimer, Hatto, & Mwabonje, 2015 pp. 7)

This thesis aims to present a more transparent and realistic GHG emission reduction potential for UCO based biodiesel.

There is literature on LCA's regarding biofuels produced from UCO, see Vinyes, Oliver-Solà, Ugaya, Rieradevall, & Gasol (2013) or Talens Peiró et al. (2010). However, these studies are not focussed on the EU and do not seem to include long-distance overseas import, but rather a local collection of used cooking oil.

This thesis will shed light on the sustainability of European biofuel production with UCO as a feedstock. Combining a market study with a well-defined LCA can give an estimate of the contribution of UCO based biodiesel towards the European renewable energy targets. This research will focus on UCO based HVO biodiesel solely, while other biofuels exist these are not relevant to the research question and will not be investigated to keep the scope of the study comprehensive. The research question that will be answered is: What is the actual greenhouse gas footprint of biodiesel production from used cooking oil and how does it compare to the default values proposed in the EU Renewable Energy Directive?

In order to answer this research question, several sub-questions and objectives need to be answered.

(i) What does the current Dutch UCO market look like, including international trade, and how did trends develop from 2005 onwards?

(ii) What is the role of UCO in the current feedstock mix for biofuel production now and how will this develop beyond 2020 after the implementation of the RED II?

(iii) What is the greenhouse gas footprint of UCO biofuel, using real flows and allocation in accordance with the calculation method of the RED (2009/28/ EC) Annex V framework?

(iv) What is the effect of sharing the environmental burden of virgin oils production with used cooking oil?

Objective (i) and (ii) are important to determine what the flows of UCO as a feedstock look like. Visualising these flows helps to understand the origin and destination of UCO, and additionally, the mass of UCO moved can be determined. This objective helps to answer objective (iii) and (iv). Objective (iii) and (iv) are important to determine what the actual GHG footprint of UCO biodiesel is for several scenarios. This variation is important to visualise the difference that a well-established LCA can make in the GHG reduction potential.

2 Background

2.1 Policy context

2.1.1 Fuel Quality Directive

The Directive 2009/30/EC, an amendment to Directive 98/70/EC on environmental quality standards for fuel, brings forward the intention of the EU to reduce negative aspects of bioenergy (Caputo, 2014). Directive 2009/30/EC, also called Fuel Quality Directive (FQD), includes a mechanism for reporting the potential reduction of life cycle GHG emissions from fuel. Also, alternative feedstocks for biofuels are promoted by establishing sustainability criteria which must be met to be counted towards the GHG intensity reduction obligation (European Union, 2009). This directive indirectly amplifies the favourability of alternative biofuel feedstocks with a high GHG emission reduction potential, UCO has a large potential as defined in the RED.

2.1.2 Renewable Energy Directive

The increase in FAME usage in the Netherlands can be seen as a direct result of the EU Renewable Energy Directive 2009/28/ EC, as this initiative aims to promote the use of waste and residues as an alternative to traditional feedstocks. Furthermore, in the directive, it is stated that "the contribution made by biofuels produced from wastes, residues, non-food cellulosic material, and lingo-cellulosic material shall be considered to be twice that made by other biofuels" (Art. 21) (European Parliament, 2009). This means that member states can achieve their Renewable Energy targets of blending 10 percent advanced biofuels for 2020 more easily (Flach et al., 2016). The most common method to produce biodiesel is through transesterification, a relatively simple method to produce FAME. UCO as a feedstock for FAME is considered a waste product and therefore eligible for double counting. This feedstock resembles 80.4 percent of double counting FAME consumed in the Netherlands in 2015 with the remainder being animal fat (Dutch Emissions Authority, 2016). For HVO, UCO resembled 83.2 percent of the used feedstocks and the remainder consists of used Fuller's earth (ibid.). The aforementioned double-counting measure has been implemented in 11 Member States, with the Netherlands, the United Kingdom and Germany being the largest producers of UCO based biodiesel (Flach et al., 2016). The default value of GHG reduction of waste vegetable oil biodiesel is 83 percent, as stated in Annex V of the RED (European Parliament, 2009).

2.1.3 ILUC Amendment

In 2015 new rules came into force in the EU which amend the FQD and the RED to reduce the risk of ILUC and to prepare the transition towards advanced biofuels. Specifically mentioned in this ILUC amendment is encouraging the production of "advanced" biofuels, produced from e.g. algae and waste. As these "provide high greenhouse gas savings with low risk of causing indirect land use change and do not compete directly for agricultural land for the food and feed markets" (European Commission, 2012). Furthermore, the production of UCO based biofuels is stimulated by an amendment of the GHG emission reporting, as "biofuels made from feedstocks that do not lead to additional demand for land, such as those from waste feedstocks, should be assigned a zero emissions factor" (European Union, 2015).

2.1.4 Renewable Energy Directive II

A proposal for a revision of the RED as described in section 2.1.2 has been pushed by the European Commission on 30/11/2016. This revision, from now on RED II, is to commence from 2021 to 2030 and will succeed the current directive. In this revision the double counting measure for UCO has been redacted and replaced with an approach (a blending sub-target of 3.6 percent) to stimulate the development of advanced biofuels coming from the feedstocks listed in RED Annex IX, Part A. UCO as a biofuel feedstock is listed in RED Annex IX Part B and is not considered an advanced biofuel feedstock. Furthermore, the contribution of UCO based fuels and other conventional low-carbon biofuels is limited to 1.7 of 6.8 percent. This would mean that UCO as a feedstock for renewable energy in transportation is capped post-2020 (International Council on Clean Transportation, 2017). This also holds true for other biofuels produced from organic wastes and residues. This limitation in RED II could mean a stagnation of the European UCO market, as other uses are limited (See section 5.3.1).

2.2 Fatty Acid Methyl Esters and Hydrotreated Vegetable Oils

FAME production using UCO as a feedstock is most prevalent in the Dutch UCO biodiesel market (Dutch Emissions Authority, 2016). HVO produced from UCO has grown in the last years and seems to be a suitable alternative for FAME (ibid.). FAME are the fatty acid esters that are the product of a transesterification process, where lipids react with the alcohol methanol (Dunn, 2010). This is a relatively easy manufacturing process, and FAME can be blended without engine adaptation to a certain extent. HVO is the product of hydrotreating vegetable oils. It offers several benefits over FAME such as better cold properties, lower NO_x emissions and increased storage stability (Aatola, Larmi, Sarjovaara, & Mikkonen, 2008). Commercial production of HVO in the Netherlands is carried out by Neste Oil, using the name NExBTL (Nikander, 2008). This plant has a capacity of 1,000,000 t/yr (Port of Rotterdam Authority, 2016). However, annual production capacity is closer to 800,000 t/yr (Neste Oil Corporation, 2011). In this study, the NExBTL HVO pathway will be investigated as it offers a more interesting view on future development and other benefits over FAME as mentioned earlier.

3 Method

3.1 Lifecycle Assessment

An LCA comprises of four phases: First one has to define the goal and scope of the assessment. Secondly, an Inventory Analysis has to be carried out. Thirdly, the Impact Assessment is performed. Each stage has to be interpreted as a fourth phase. These stages can be seen in Figure 2.

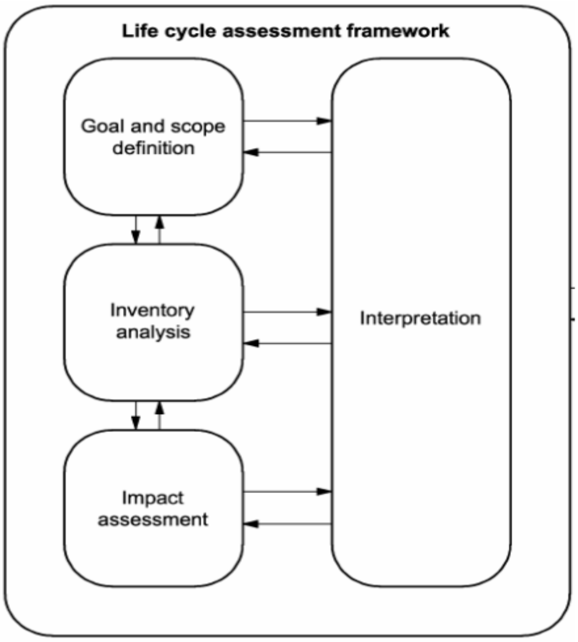


Figure 2 Stages of a LCA (SFS-EN ISO 14040:2006, page 8)

This would result in a clear overview of what part of UCO biofuel production has the largest impact on the sustainability and its relation to the RED methodology. As it is indicated that the EU UCO supply is insufficient to meet the demand (Peters et al., 2013) it would be beneficial to analyse the intra-EU market and compare it to the EU UCO imports.

It is important to distinguish the framework which will be investigated, whether one looks at UCO transport and subsequent biodiesel production, or one includes the origin of the UCO. The origin of the feedstock will be included in this research, keeping in mind that this can be a time-consuming process as UCO can have many sources. However, this will increase the quality of the research, as feedstock origin influences the price and availability of UCO in general. Meaning that in some cases the waste product can become more valuable than its source. In this analysis, no change in blending mandates is assumed. The LCA will be carried out using the SimaPro software, using inputs from the Ecoinvent 3 and ELCD databases as well as own input data.

3.2 Attributional versus Consequential Lifecycle Assessment

To ensure biofuels achieve GHG reductions relative to their fossil fuel counterpart, lifecycle carbon reporting is required is required in policies for supporting said biofuels. A Life Cycle Assessment (LCA) is a very useful tool in doing so, it allows the environmental attributes of a product/process to be better understood and therefore enables complex decisions made more critically (Menzies, Banfill, & Turan, 2007). However, the aforementioned policies tend not to distinguish between two kinds of LCA, namely attributional LCA (ALCA) and consequential LCA (CLCA). ALCA provides information about the (environmental) impact of the product lifecycle but does not include indirect effects resulting from changes in the output of a product. Whereas CLCA provides information about the consequences of changes in the output of a product, including effects outside the lifecycle of a product (Brander, Tipper, Hutchison, & Davis, 2008). The LCA method as conducted in the RED is mostly coherent with the ALCA type. However, one must note that for determining total GHG emissions, which is one of the main points of lifecycle carbon reporting related to biofuel policy, this may not be the most appropriate LCA method (ibid.). ALCA, useful for comparing products on a company scale, does not show real-world impacts on climate change. A CLCA is therefore superior in predicting policy effects and can help in decision making by policy makers (Plevin, Delucchi, & Creutzig, 2014). Performing and interpreting a CLCA should be done with caution, as a CLCA is to a large extent dependent on economic models regarding supply, demand, price elasticity, and other market effects. Furthermore, to draw conclusions from a well performed CLCA sufficient quality of input variables and market data is necessary.

3.3 Goal and Scope definition

The goal and scope of this study will be well-defined, to preclude misinterpretation of the results. The study will be carried out in line with ISO 14044:2006 standards for life cycle assessments. The goal of this study is twofold. Firstly, the aim of this study is to give a transparent analysis of the evidently opaque UCO market (See section 4.2). To support this strive towards transparency, only publicly available data has been used. As a consequence, more detailed but often restricted data regarding production processes, transport- and market information has been left out. The second goal of this study is to assign a range of GHG emissions for UCO biodiesel production. This research can be used as a reference tool for policymakers and academics, as well as an aid in comprehending the used cooking oil market and biodiesel production for the general public. Parameters to include in this analyses are discussed in the following sections.

3.3.1 Functional Unit

The choice of functional unit is important, very different conclusions can be drawn as the functional unit reflects the comparisons that are being made (Quek & Balasubramanian, 2014). 'MJ of fuel equivalent' is seen as a suitable functional unit when comparing different fuel production pathways, as these

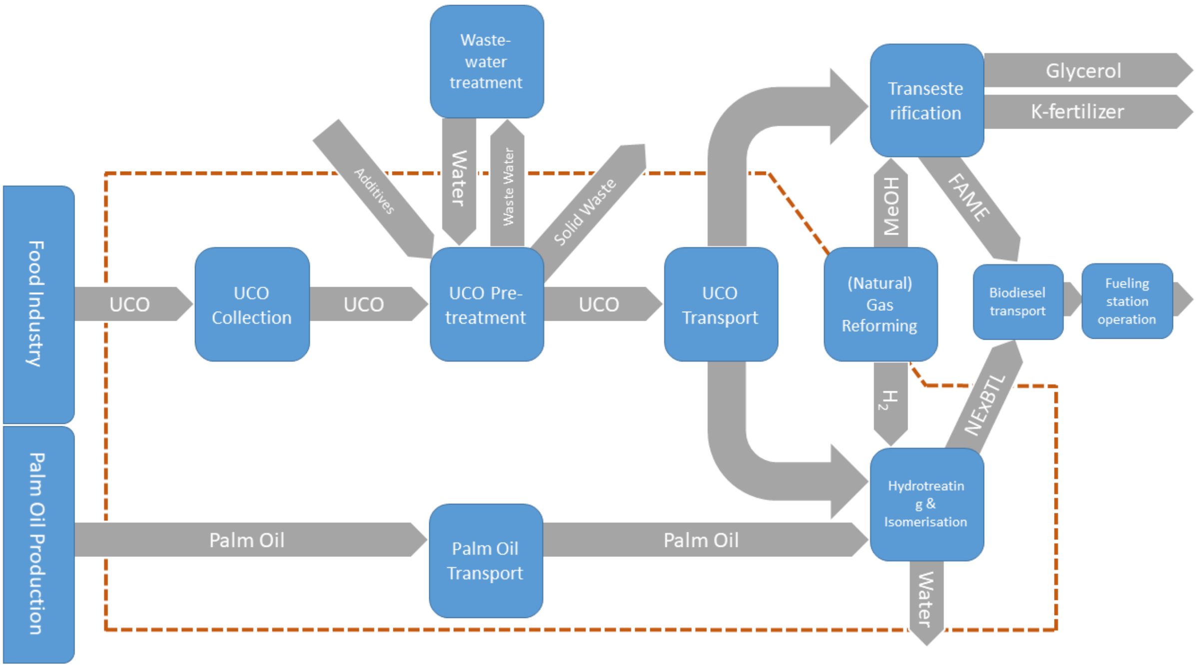


Figure 3 UCO pathway and system boundaries (adapted from: (Talens Peiró et al., 2010))

translate directly to anthropogenic energy requirements (Davis, Anderson-Teixeira, & DeLucia, 2009). This means that the functional unit of 'MJ of fuel equivalent' is suitable to compare fuels which share combustion characteristics such as HVO, a drop-in fuel, and fossil diesel. This functional is well paired with system boundaries that end at the factory gate as described in the next section.

3.3.2 System Boundaries

This analysis will focus on the current state of the UCO market, as the effects of the RED are already in place and there is limited future data available. Included will be, the transport of UCO to the refinery and emissions from esterification. A so-called cradle-to-factory gate approach. A schematic overview of the UCO biodiesel production system can be seen in Figure 3. The detailed FAME production process can be seen in Figure 4, while the HVO production process can be seen in Figure 5. Additional FAME and HVO production process chains can be found in Appendix I. In this study only the HVO production

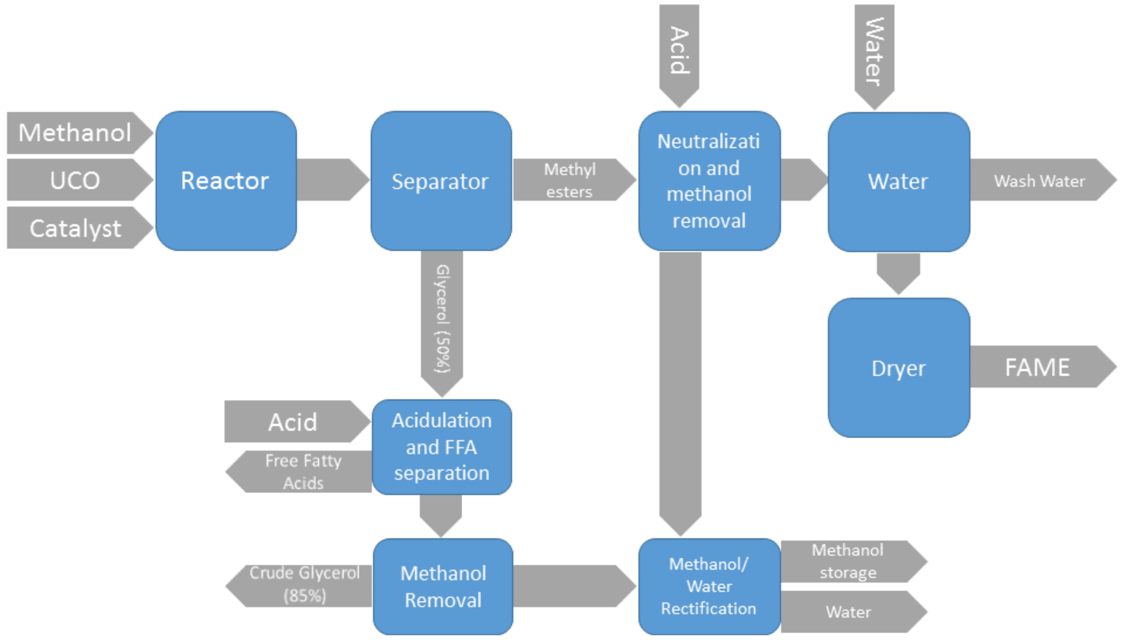


Figure 4 FAME production (adapted from: (Dunn, 2010))

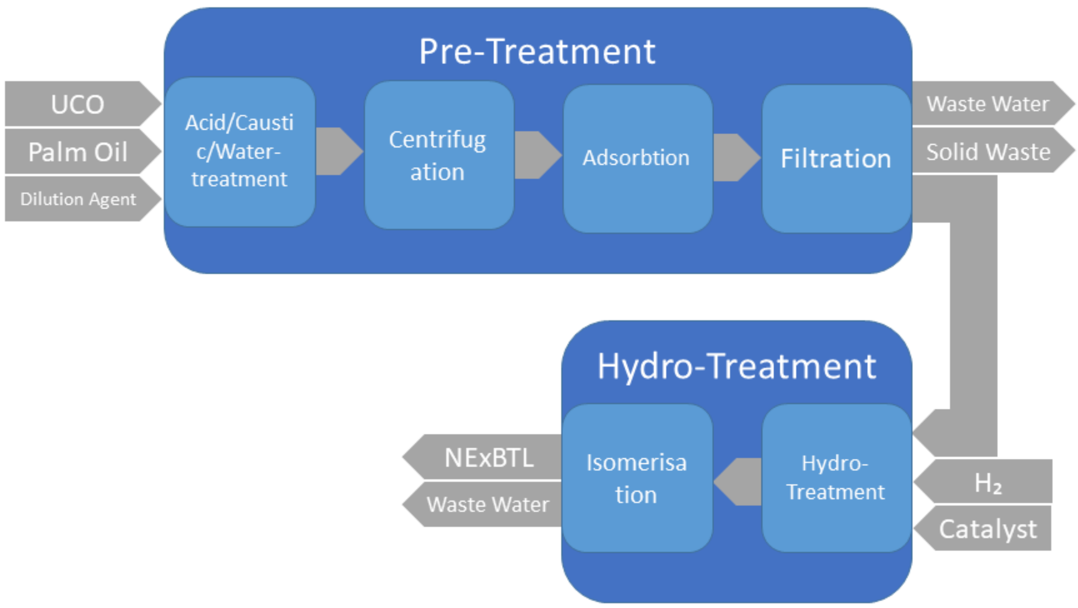


Figure 5 HVO production in the form of NExBTL (Adapted from: (Nikander, 2008)&(Laakkonen et al., 2013)

process will be investigated due to its growing prominence on the Dutch biofuel market, as discussed in section 2.2.

The main differences in GHG emissions submerge during the transport phase of HVO production when compared to the RED methodology. Therefore, the choice for a cradle-to-factory gate has been made.

Through a market analysis, the origin of the UCO in Dutch refineries will be determined and associated emissions for feedstock import and collection will be accounted for. This will be done on a mass basis as a share of input because imported UCO and domestic UCO share the same properties (Valente, Pasa, Belchior, & Sodré, 2011).

General Supply Chain

The supply chain is set up using various sources, such as a qualitative interview (see Appendix II) and a broad array of literature. A visualisation of the supply chain can be seen in Figure 6.

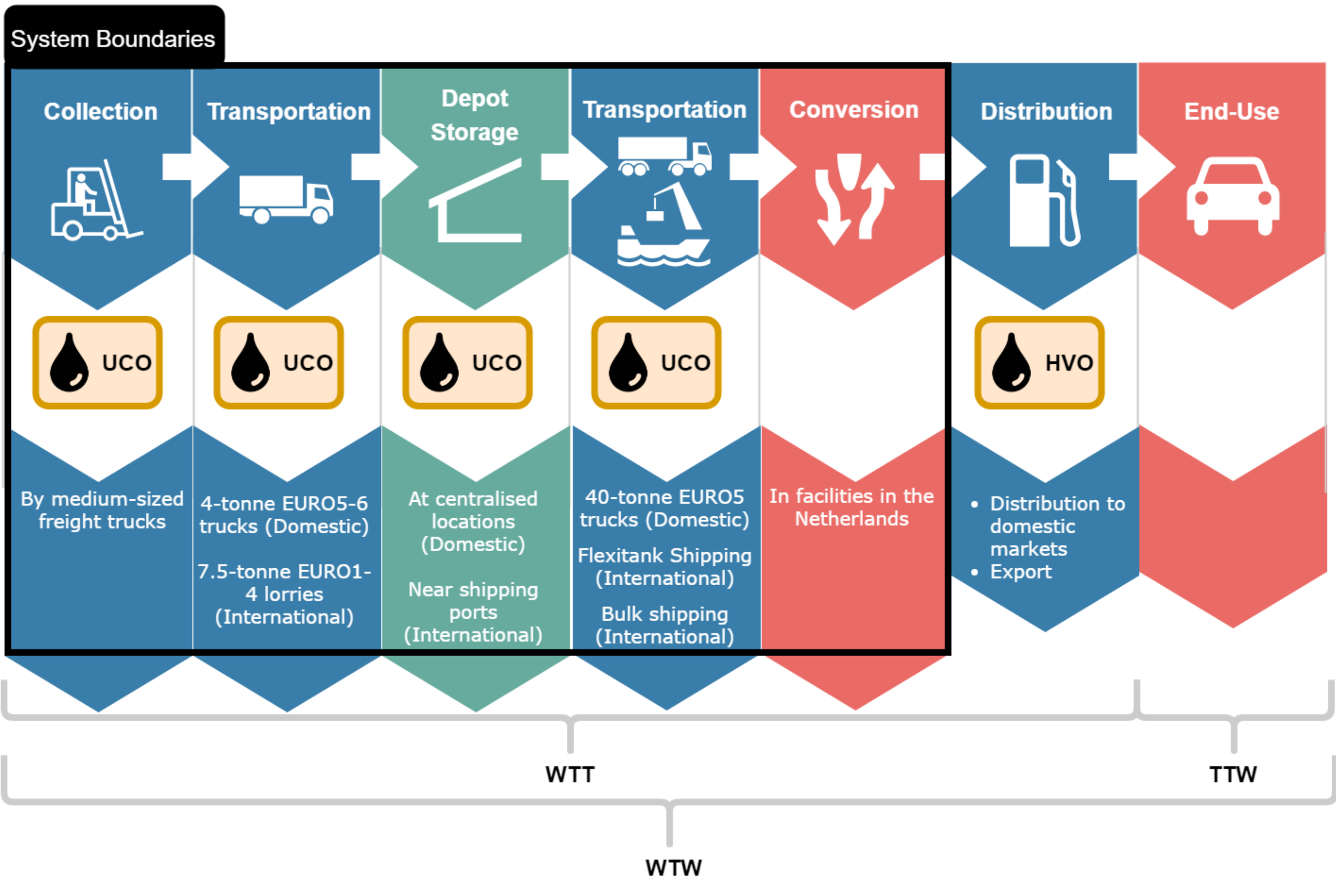


Figure 6 General Supply Chain of an UCO HVO biofuel

Disclosed in the supply chain are the system boundaries of this study, the Tank-To-Wheel (TTW), Wheel-To-Tank (WTT) and Well-To-Wheel (WTW) scopes of the JEC.

3.3.3 Treatment of co-products

The allocation method is dependent on the LCA that is applied. In this case, no allocation of co-products is necessary as during the ALCA of HVO no co-products are generated. If one would investigate the

FAME production process allocation would be needed due to the co-production of glycerol. Due to the changing nature of UCO as a 'waste' product, a form of burden-sharing will be investigated. This method of economic allocation can be done to allot a certain part of the environmental burden of virgin oil production to UCO. As the virgin oil (primary product) is technically different from UCO (reused product) a correction factor has to be implemented (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010). This correction factor will be a ratio based on the spot market value of both oils. A median of the market value of UCO and virgin oils over 2014-2016 has been taken. The system is seen as one with two outputs in different stages of a lifecycle, the first stage being a food-based frying oil and the second being a waste-based biodiesel feedstock. As these

$$Share\ allocated = \frac{P_{UCO}}{P_{UCO} + P_{SF}}$$

Where,
P_{UCO} = *Price of used cooking oil*
P_{SF} = *Price of sunflower oil*

Calculation 1 Burden sharing of environmental impacts of virgin oils

can be seen as two different products, with individual value, the price of UCO has to be divided by the sum of the price of UCO and the virgin oil to prevent allocation of the entire environmental burden. This allocation method is given in Calculation 1.

3.3.4 Impact Assessment

During impact assessment, the overall life-cycle impacts will be related to the results found during the inventory analysis. According to ISO 14044:2006 standards the inventory parameters are sorted and assigned to GHG emission. The impact is measured according to CO_2-eq.

Impact categories, indicators, and characterisation

The impact category that will be investigated is Global Warming, specifically the global warming potential (GWP) of all emissions during production. While there is no scientific basis for using a 100-year timeframe, this method will be applied during the research as it is a representative timeframe used in LCA studies in order to characterize the GHG impacts (Soimakallio, 2014). Used will be the GWP-100 factors as provided by the IPCC. Substances that are considered to be contributors to global warming, and will be included in this research, can be seen in box 1. The GWP of each GHG will be

Box 1: Greenhouse gases as specified by (Sánchez, Bhattacharya, & Mareckova, 2006)

- Carbon Dioxide (CO_2)
- Methane (CH_4)
- Nitrous Oxide (N_2O)
- Hydroflurocarbons (HFCs)
- Perfluorocarbons (PFCs)
- Sulphur Hexafluoride (SF_6)
- Nitrogen Trifluoride (NF_3)
- Trifluoromethyl Sulphur Pentafluoride (SF_5CF_3)
- Halogenated Ethers
- Other halocarbons not covered by the Montreal Protocol including CF_3I, CH_2Br_2, $CHCl_3$, CH_3Cl, CH_2Cl_2

expressed as CO_2-equivalents, (CO_2-eq.) i.e. the effects are expressed relative to the effect of CO_2. Note that availability of GHG emissions data depends on the dataset used. Important to notice is also that most carbon in UCO is biogenic and the percentage of fossil carbon in FAME is attributed to the origin of the methanol, which is mostly from fossil resources in the Netherlands (C. N. Hamelinck & Faaij, 2002). Acidification and other impact categories will not be investigated as they are not part of the research objective.

3.3.5 Baseline comparison

The results will be compared to the baseline emissions of cradle-to-factory gate emissions for fossil diesel and biodiesel on the basis of other vegetable oils. Estimates are available and will be compared,

$$Emission\ savings = \frac{E_f - E_b}{E_f}$$

$$where\ E_b = total\ emissions\ from\ the\ biofuel$$

$$and\ E_f = total\ emissions\ from\ the\ fossil\ comparator$$

Calculation 2 GHG emission reduction in the RED giving an average emissions intensity for the different fuels, as refined in the Netherlands. Furthermore, the reduction potential will be derived from a comparison with fossil diesel counterparts. In the RED this relative GHG reduction is defined as described in Calculation 2, this method will be applied to this research accordingly.

3.4 Inventory Analysis

During the inventory analysis, all the inputs and outputs of the UCO biodiesel lifecycle will be compiled and quantified. For each unit process, as shown in the system boundaries of Figure 6, data will be collected and subsequently related to the functional unit. Data collection consisted of several sub-categories as proposed by Kirchain (2006) which are given in box 2.

Box 2: Data collection categories and subcategories adapted from (Kirchain, 2006)

- Inflows
  - Materials
    - This will be UCO in several forms throughout the production process of HVO, palm oil is an additional input as well.
  - Energy
    - During most processes some form of energy is required, for simplicities sake, the Dutch energy mix will be consulted. If other forms of energy production, such as CHP are applicable these will be taken into consideration.
- Outflows
  - Primary product
    - HVO in the form of NExBTL
  - Co-products
    - No co-production is accounted for during HVO production
  - Releases to land, water and air
    - As described, the focus lies on GHG emissions
- Transport
  - Distance
    - Depending on the UCO origin, either intra-EU or international transport towards refineries in the Netherlands. Subsequent transport to filling stations will not be included.
  - Mode
    - Freight marine shipping and truck transportation are the main transport forms utilised.
- Qualitative
  - Description of activity under analysis
  - Geographic location
  - Timeframe

Data collection for the HVO production process will commence on the basis of the flow diagram as shown in Figure 5.

3.5 Calculations

3.5.1 Flexitank Shipping

The emissions per shipment are influenced by a large number of factors, a straightforward simplified calculation is shown below where only the utilisation factor is included. This calculation concerns container shipments.

$$flexitank\ emissions\ (g\ CO2/\ TEU\ km) = \frac{Emissionfactor\ flexitank\ (g\ CO_c/TEU\ km)}{Utilisation\ factor\ (\%)}$$

Calculation 3 CO_2 emissions per shipment (Adapted from: (Thibault, 2015))

A more detailed approach is given by Veidenheimer (2014) of which an adaptation can be found in Calculation 4.

$$Emissionfactor\ Flexitank\ (g\ CO2/TEU\ km) = \left(\frac{MCR * (AS/MS)^3 * AT * (BSFC * eHFO)}{TEU}/D\right) * 1000$$

Where,

MCR = Maximum Continuous Rating of the combustion engine in use (kW)
AS = Average Speed
MS = Max Speed
AT = Activity time (hours)
EF = Emission Factor for fuel used(kg/kW-hr)
BSFC = Brake Specific Fuel Consumption
eHFO = Emissionfactor of Heavy Fuel Oil
TEU = Size of the vessel in 20 foot equivalent unit (TEU)
D = Distance travelled

Calculation 4 CO_2 emissions per TEU (Adapted from: Veidenheimer (2014))

The load factor is comprised of the average speed and maximum speed. The emission factor is the brake specific fuel consumption (BSFC) multiplied by the emission factor of heavy fuel oil (HFO). To

$$CO_2\ emissions(kg) = \left(\left(\frac{Flexitank\ emissions\ (g\ CO2/\ TEU\ km)}{Flexitank\ volume\ (l) * UCO\ density(g/cm3)}\right) * Distance(km)\right)$$

Calculation 5 CO_2 emissions for flexitank usage

calculate the emissions per flexitank shipment Calculation 5 has been used. Included in the flexitank emissions is the utilisation factor as presented in Calculation 3.

3.5.2 Chemical Tanker Shipping

Calculation of emissions for tanker shipping can be done using a bottom-up and a top-down method. To calculate the emissions for tanker shipping using a bottom-up approach a reference vessel has been used.

$$STE = \frac{P * SFOC * h * eHFO}{(D * dwt)}$$

Where,

STE = specific tanker emissions in g/tkm

P = engine power in kW,

SFOC = specific fuel oil consumption in g/kWh

h = hours on engine power

eHFO = heavy fuel oil emissions in g CO_2/g

D = distance travelled in km

dwt = dead weight tonnage in tonnes.

Calculation 7 specific tanker emissions calculation for a small chemical tanker, bottom-up approach

This is a chemical tanker with a 3,500 dwt. This chemical tanker has a main engine installed with a power of 2040 kW (Nordic Tankers, 2015). An example of such an engine is the Wärtsilä 26 6L with a specific fuel oil consumption (SFOC) of 188,7 g/kWh according to ISO standards (Wärtsilä, 2016).

$$CO_2\ emissions\ (kg/t) = \left(\frac{STE\ (gCO2/tkm) * Distance\ (km)}{1000}\right)$$

Calculation 6 CO_2 emissions for bulk tanker usage

Using this data and travel time/distance according to https://sea-distances.org/ Calculation 7 has been established. Using the specific tanker emission of Calculation 7 and multiplying by the journey distance in Calculation 6 the CO_2 emissions per trip can be calculated.

A top-down approach can give a more realistic result as it is based on average fleet data. Both approaches are elaborated upon in section 5.1.1 (‘Chemical tanker Shipping’).

$$TE = \frac{((FOC * eHFO) * 10^6)}{\left(\frac{(v * t)}{1000}\right) * dwt}$$

Where,

TE = tanker emissions in g/tkm

FOC = fuel oil consumption in g/kWh,

eHFO = heavy fuel oil emissions in g CO2/g HFO,

v = average speed in m/s,

t = average time at sea in seconds

dwt = dead weight tonnage in tonnes.

Calculation 8 tanker emissions based on IMO data

Note that in Calculation 8 some conversions are already included (for speed and time at sea) and fuel oil consumption is based on the sum of the main-, auxiliary- and boiler engine.

To calculate emissions for tanker shipping either TE from Calculation 8 or STE from Calculation 7 will be divided by a utilisation factor in the same manner as described in Calculation 3, the main difference being is that the result is in gCO_2 per tonne instead of g CO_2 per TEU.

4 Market Analysis

4.1 Hydrotreated Vegetable Oils

With the number of biorefineries producing HVO increasing from 1 in 2010 to 13 in 2017 (Flach et al., 2016), the demand for HVO does not seem to be fully satisfied as production is still increasing. Competing with other biofuels, HVO is estimated to establish a 23 percent share on the European biodiesel market in 2020 (Hillairet et al., 2016) compared to a current 9 percent share (GREENEA, 2015). The price of HVO and biodiesel, in general, is mostly dependent on the underlying price development of feedstocks (see Appendix II). Therefore the market development of UCO will be described in the following sections.

4.2 The used cooking oil market

4.2.1 Pricing used cooking oil

When using UCO as a feedstock for biofuel production there is no standard feedstock quality available, the price is, therefore, dependent on quality. Specifications of the feedstock quality are based on geographical origin and processing technique (Greenea, 2014). Parameters which influence the UCO quality are the percentage of free fatty acids (FFA), moisture, impurities, iodine- and sulphur values. FFA, sensitive to alkali catalysts, displays soap formation during the transesterification process and are therefore desired to be a low fraction (Banerjee & Chakraborty, 2009). As the cooking oil is made from different feedstocks such as soybean oil in Latin-America, sunflower oil in Europe and palm oil in South-East Asia, quality differs per region where the UCO is collected. Further quality degradation takes place over usage time (Enweremadu & Mbarawa, 2009).

Figure 7 Local UCO collection

Transportation mode is another key factor which influences the pricing of UCO. Locally collected and delivered UCO, is commonly transported using small to medium sized freight (tank) trucks as can be seen in Figure 7. Demands by larger UCO FAME and HVO producers can be met by utilizing larger train and freight ship containers and buying in bulk. Note that this does not ensure a lower price.

Moreover, the feedstock price can increase as this kind of quantities are often solely provided by UCO traders who aggregate quantities from smaller collectors (Greenea, 2014).

Thirdly, certification of the feedstock trader matters as well in determining the UCO price. Certification schemes exist which can drive the UCO price up if applied. An example of this is the International Sustainability & Carbon Certification (ISCC) waste and residue certification, which verifies the type of raw material used and its status as a genuine waste or residue (International Sustainability & Carbon Certification, 2016).

The price development of UCO and comparable biofuel feedstock has been plotted in Figure 8.

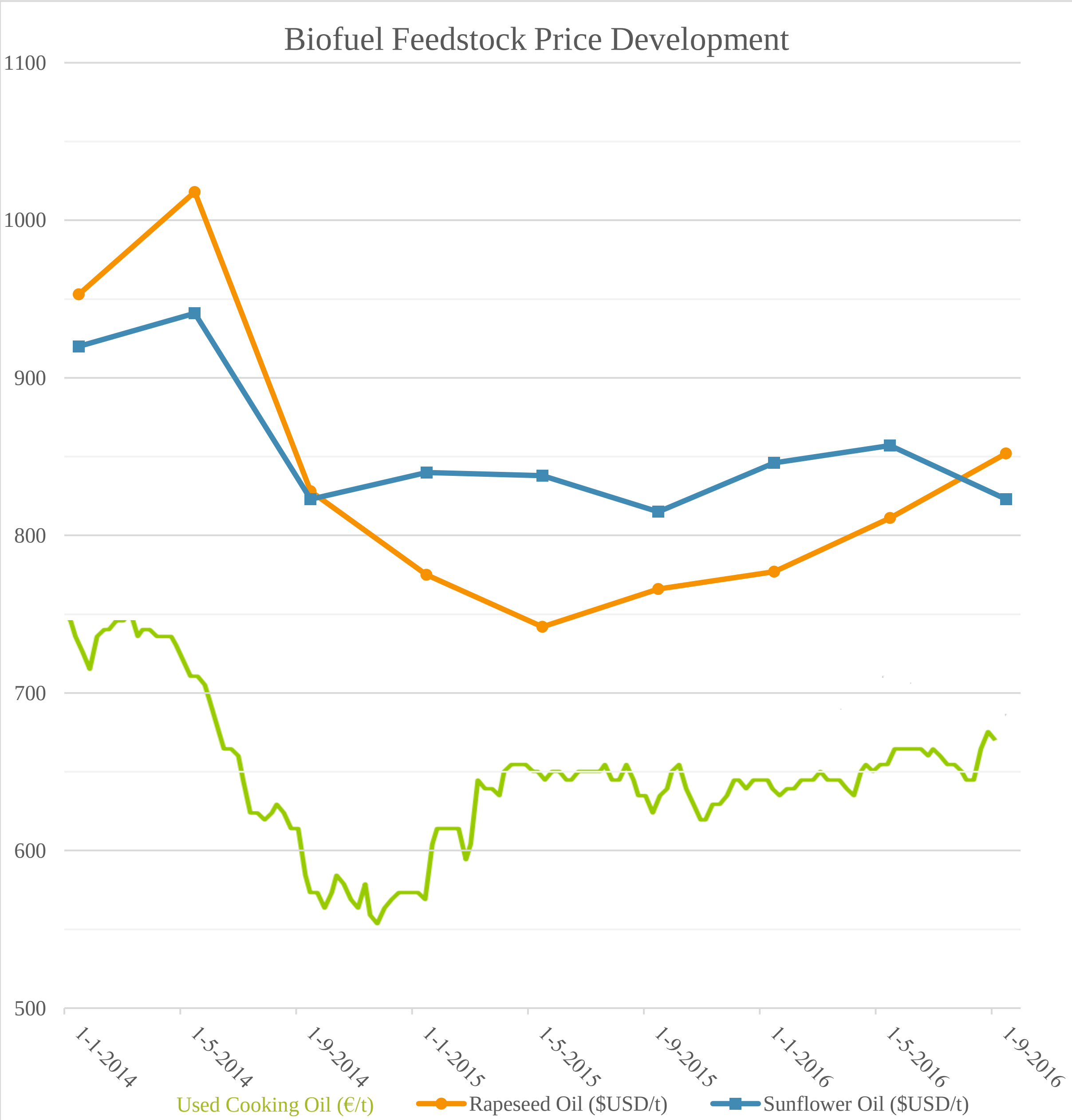


Figure 8 Biofuel feedstock price development 2014-2016 (Adapted from: (Hillairet et al., 2017; Thoenes, 2018))

Note that the price of the UCO is a European price development and the data of the remaining feedstocks is given in an international context, hence the difference in monetary units. While no direct conclusions can be drawn, the price development of each resource can be compared. All prices are decreasing near the end of 2014 with some recovery in early 2015. UCO seems to have a large upward impulse in 2015 but stabilises over the remainder of the year.

4.2.2 Used cooking oil availability

The Netherlands

According to the BioDieNet project (2007-2009), only a few years before 2009 the interest in estimating quantities of UCO generated and amounts available has grown in Europe. This change in interest is attributed to the Animal by-products legislation 1774/2002, which banned the use of UCO as an ingredient in animal feed due to concerns over food- and feed-borne crises related to, among other things, swine fever and Bovine Spongiform Encephalopathy (BSE) (BioDieNet, 2009). The countries represented in the BioDieNet consortium estimated a total UCO production of 1.921 billion litres in 2009. An overview of the amount of UCO collected can be seen in Table 1.

Country	UCO collected (m^3/yr)	UCO utilized (m^3/yr)
The Netherlands	67,000	67,000
Italy	60,000	60,000
Portugal	28,600	16,000
Spain	270,000	n/a
Germany	250,000	n/a
Hungary	5,500	5,000
Norway	1,000	300
UK	90,000	99,000

Table 1 UCO availability according to BioDieNet consortium

Note that the data in as provided in Table 1 is likely dated. According to a report by the Dutch branch organisation for oils and fats (MVO), 4000 tonnes of UCO have been collected in 2013. This is 40 percent of the 10,000 tonnes target in 2020, as set by the 2012 campaign ‘Recycle frying oils’ (MVO, 2014). Note that this number represents only the share of households, and the share of the catering industry is larger. An estimation for available UCO in the Netherlands in 2020 is given by Koppejan, Elbersen, Meeusen, & Bindraban (2009), who suggest a value of 120,000 tonnes derived from the catering industry and an additional 10,000 tonnes from the potato processing industry.

China

Cooking oil plays an important role in Asian cuisine. The size of the Chinese cooking-oil retail market of 2013 is estimated to be around 6.5 million tonnes sold. With the largest share being canola-, soy- and peanut-oil. It is expected that the market nearly doubles to 11.5 million tonnes in 2018 (Agri-Food and Agriculture Canada, 2014). While the potential availability in China is large, feedstock collection can be identified as a bottleneck. The collection is often carried out illegally due to attractive economic profits which may cause a shortage for legal collection of the feedstock (Liang, Liu, Xu, & Zhang, 2013). This lack of controlled collection is further discussed in section 5.3.1.

United States

An estimation of UCO availability in the US is given by Brorsen (2015), who argues that the UCO market in the US matured and not bound to increase. Note that in this estimation the term 'yellow grease' is used, which includes a majority of UCO but is also comprised of other discarded fats. Furthermore, it is argued that recovering used oils and fats from households is not profitable at current economies of size. Half of the available yellow grease is used for biodiesel production, a large portion for the production of animal feed, and some export. An overview can be found in Table 2.

	2014	Unit
Production	931,679	tonnes
Yellow Grease for FAME production	487,158	tonnes
Available for export and other uses	444,521	tonnes

Table 2 Yellow grease production in the US (Source: (Brorsen, 2015)*)*

A growth of 133,356 tonnes until 2019 can be expected (Brorsen, 2015).

4.2.3 Dutch Market Developments

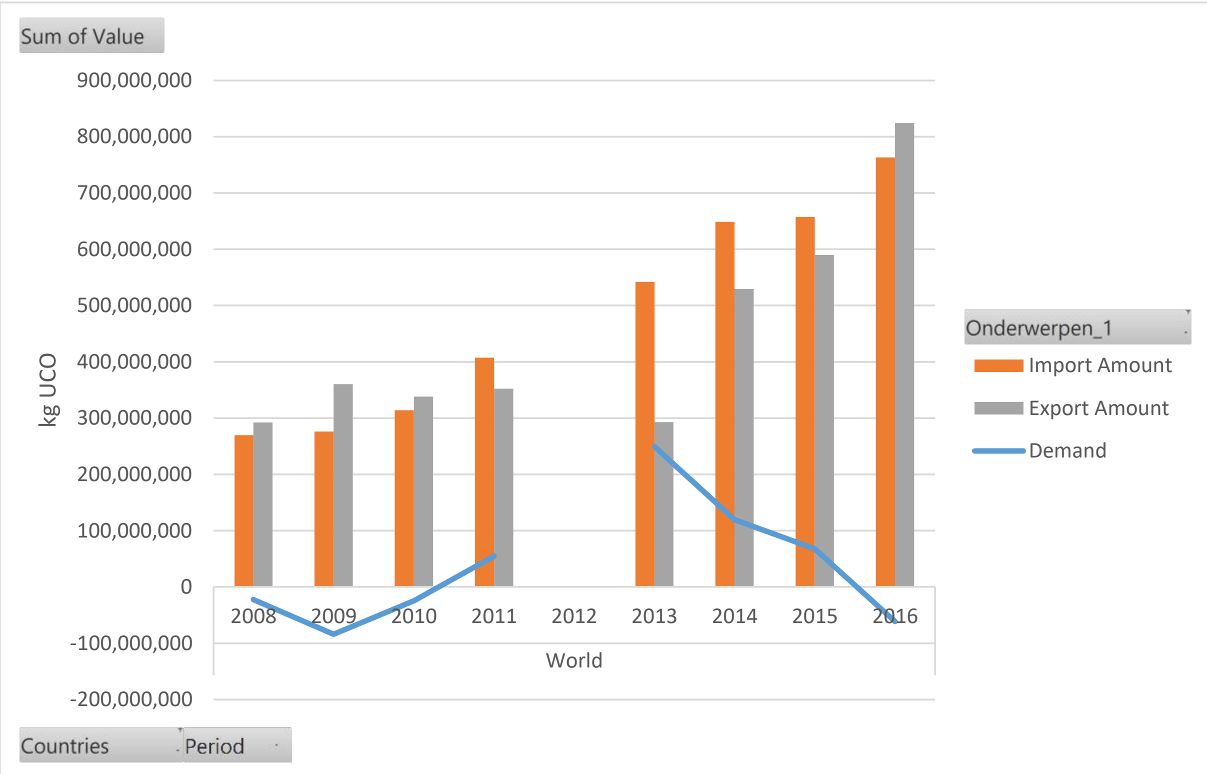


Figure 9 UCO Import and Demand in the Netherlands (Adapted from: (CBS, 2017)*)*

In Figure 9 the import and demand of UCO in the Netherlands are shown in a graph. Import is the total import of UCO under HS-code 15180095, as reported by the Dutch Bureau for Statistics 'Centraal

Bureau voor de Statistiek'(CBS). The gross domestic consumption is calculated by subtracting the import values by the export values. Note that data of 2012 is purposely left out due to data inconsistency.

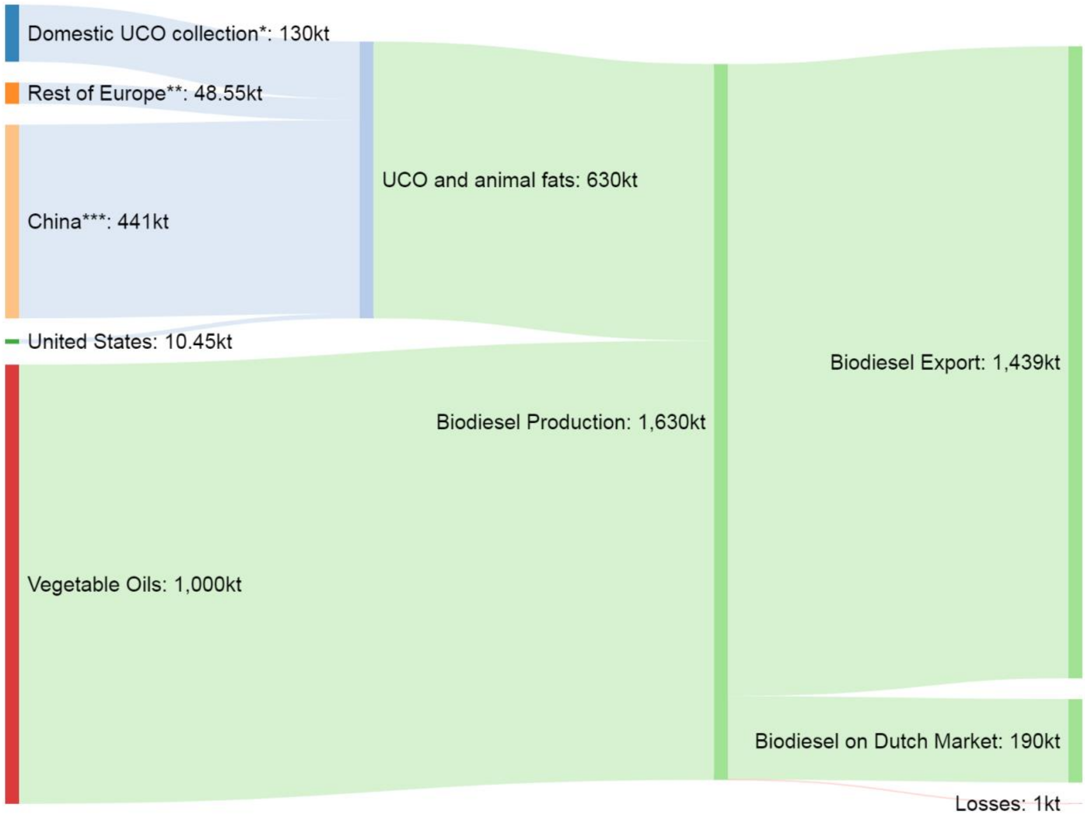


Figure 10 UCO flows and biodiesel production in the Netherlands in 2015, using a domestic UCO collection estimate for 2020 (Adapted from: (C. S. Goh et al., 2016) *& own calculations) [* see 4.2.2 'The Netherlands', ** actual numbers might be higher see Appendix II, *** and other Asian countries.]*

The origins of the UCO on the Dutch market are much differentiated and may change on a year-to-year basis, and even seasonal changes may occur (See Appendix II). According to the Dutch Emission Authority (2016), the main countries of origin of UCO in 2015 were the Netherlands, the United States and Germany with 18.5, 15.9 and 10.6 percent respectively. In 2016 the order has shifted with the US being the largest origin and Germany being replaced by Spain. Additionally, there has been an increased import from China and Taiwan with 6.5 percent and 8.8 percent respectively (Dutch Emissions Authority, 2017). Note that these percentages are based on the premise that only fuels for the Dutch transport market are considered. An inquiry with a Dutch UCO trader shows that the true origin might be more dependent on Asian countries such as China and Indonesia (See Appendix II). Figure 10 gives an overview of the Dutch UCO imports and biodiesel production, based on calculations of the total import, conversion and use of UCO based biodiesel in the Netherlands. For the calculations, it is assumed that 90 percent of the UCO has its origins in China, and the remaining 10 percent in the United States. Note that this is a simplification of the model and actual numbers might differ.

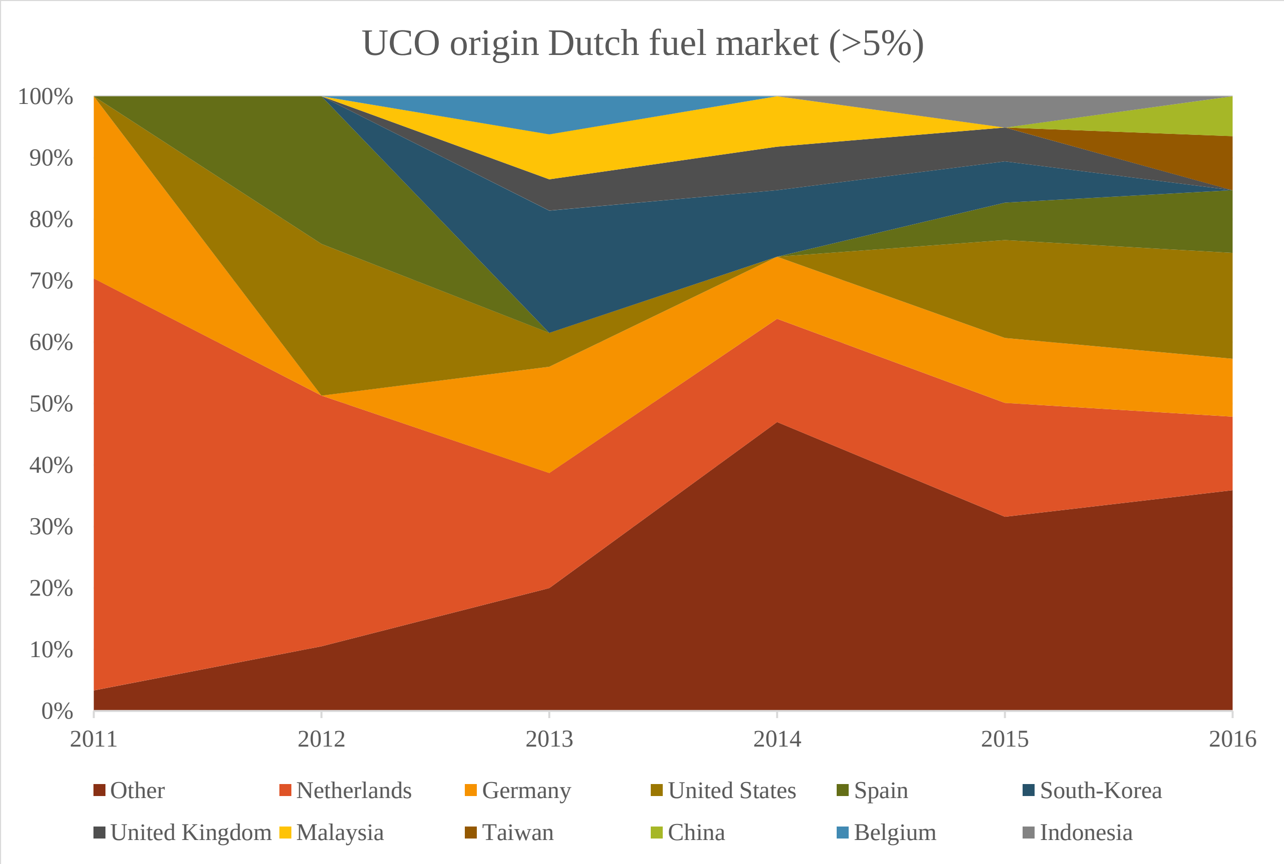


Figure 11 Countries of origin of UCO for biofuels supplied to the Dutch transport market (Adapted from: (Dutch Emissions Authority, 2012, 2013, 2014, 2015, 2016)*)*

The country of origin for UCO used for fuel on the Dutch market is depicted in Figure 11. Note, again, that these values do not take into account biofuel exports. This figure only accounts for values above 5 percent, other origins are merged in the 'Others' category. This figure shows that the Domestic UCO market is becoming less dominant over time and origins of UCO are very mixed and might change per year. Also, note that Chinese UCO imports might be under the 'Other' category pre-2016 or not destined for the Dutch biofuel market but rather used for export. A detailed table distinguishing each UCO source for the Dutch fuel market per year can be found in Appendix III (Table 19).

5 Life cycle Inventory

5.1 Inventory

The inventory of the analysed systems is drawn from primary and, to a greater extent, secondary sources. A general supply chain has been set up to visualise the journey of used cooking oil to become a biofuel this supply chain can be found in Figure 6.

5.1.1 Transport

Maritime Transportation

Flexitank Shipping

Shipping of UCO is often done in the form of 20-23 tonne flexitanks, which is also the most economically attractive packaging method for shipping edible oils (Elburg Global, 2017). Smaller quantities (<1000 L/0.8 t) are shipped using an Intermediate Bulk Container (IBC). Both methods utilise a standard 20-foot shipping container, which in turn is loaded onto a container ship. Each flexitank has a capacity of 23,000 litres per 20-foot container (ibid.). Data for CO_2 emissions for container shipping are given by the Clean Cargo Working Group (CCWG) and can be found in Table 3.

	2015	2014	2013
Asia to-from North Europe	33.7	37.9	43.8
North Europe to-from North America EC / Gulf	60.1	70.3	75

Table 3 CO_2 emissions by trade lane (grams of CO_2 per TEU kilometre) Adapted from: (CCWG, 2017)

These values represent one 20 foot equivalent unit (TEU) and are based on data for “dry” containers, meaning non-refrigerated cargo. A utilisation factor of 70 percent is adhered to. A more detailed explanation of the CCWG methodology can be found in the report by Thibault (2015).

However, the emission factors as presented in Table 3 are averages per trade lane and the emissions per TEU kilometre can have some variation. This variation is due to, amongst others, the difference in the utilisation factor of the vessel, vessel type and fuel used. For containerships, these variables are difficult to approximate due to a networking route design instead of clear start and endpoints of a trip (Christiansen, Fagerholt, Nygreen, & Ronen, 2013). To improve accuracy and transparency of the research a bottom-up approach of GHG emissions for flexitank shipping has been investigated.

Using Calculation 4 a reference vessel has been established for the most common type of containership according to the IMO. The most common type is 1,000-1,900 TEU (Smith et al., 2014). An average of this category has been taken. A table has been constructed showing input values for the calculations.

Size	1,508 TEU
Engine	6RT- flex50 D
MCR (kW)	10,470
BSFC (g/kWh)	168.8
Average Service Speed (knots)	18
Max Speed (knots)	18.8

Table 4 Specifications of a 1,500 TEU containership (Adapted from (Wärtsilä, 2015; Wingd, 2018)

Using Calculation 4, Table 4, a distance of 19,492.3 km and an activity time of 585 hours (for the Asia-> Europe trade route, US-Europe results are similar) a specific CO_2 emission of 96.13 g CO_2/TEU km can be calculated. This is then used in Calculation 3 accordingly with a 70 percent utilisation factor to give a flexitank emission of 137.3g CO_2/TEUkm.

Using the first part of Calculation 5 and assuming a capacity of 23,000 litres per TEU and a density for UCO of 0.91 g/cm^3 (Technical University of Crete, 2013) an emission of 6.56 g/tkm allocated to UCO can be calculated. Multiplying this by the distance per pathway gives the CO_2 emissions per pathway.

These are values for flexitanks, which account for 70-80 percent of global UCO shipping (See Appendix II). The remaining 20-30 percent is filled in with 1000-4000 dwt chemical tankers as these volumes are common for bulk vessel shipping of UCO[1].

Chemical Tanker Shipping

Psaraftis & Kontovas (2009) calculate an emission of 60.6 g CO_2/tkm for the category of chemical tankers under 4000 dwt, based on an average payload of 2,009 t and a capacity utilisation of 60 percent. Which is higher than the 18.5 g CO_2/tkm as calculated using Calculation 7. Note that the utilisation factor has not yet been included in Calculation 6. An utilisation factor percentage of 60 percent means that the ship spends more than half of its sea time full and the rest empty (on ballast). Note that these tankers are small sized and emissions per tkm decrease if the size of the tanker increases. This also indicates that this tanker category is mainly used over short distances as over longer hauls, generally speaking, a larger tanker size is preferred (Stopford, 2009).

For Calculation 8 the average fleet values of the Third Greenhouse Gas Study by the International Maritime Organisation (IMO) have been used, which can be found in Table 5.

Average days at sea	159
Average speed	9.8 knots
Average deadweight tonnage	2158 t
Average consumption main engine	800 t
Average consumption auxiliary engine	500 t
Average consumption boiler	600 t
HFO emissions	3.114 g CO2/g Fuel

Table 5 Input data for the average 0-4,999 dwt chemical tanker in 2012 (Smith et al., 2014)

Using this input data for Calculation 8 results in a 39.59 g CO_2 /tkm emission.

[1] This information has been obtained by personal communication with an expert in the UCO trade.

However, due to the generalising nature of the top-down approach, the bottom-up method as explained in section 3.5.2 will be adhered to. And included in the calculations will be an utilisation factor of 70 percent, similar to the flexitank calculations.

Road Transport

5.1.2 Feedstock Collection

The Netherlands

Several UCO collection systems are available, in the Netherlands, the most prominent collection method seems to be in the form of urban collection centres for private cooking oil consumers (Vinyes et al., 2013). Users bring their UCO in small containers to a collection centre where different municipal waste is collected, such as batteries, electronic devices and construction waste. From this centralised location, the UCO is transported via small to medium sized freight (tank) trucks as can be seen in Figure 7.

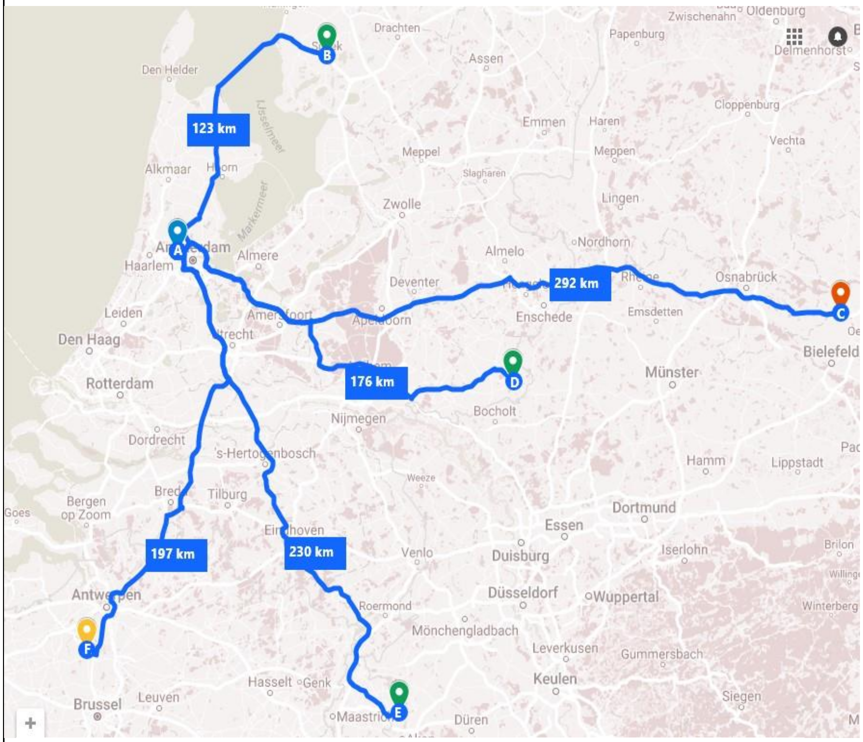


Figure 12 Locations of Rotie depots.

A)	Biodiesel Amsterdam B.V.	
B)	Depot in Sneek	123 km
C)	Depot in Bünden (DE)	292 km
D)	Depot in Winterswijk	176 km
E)	Depot in Kerkrade	230 km
F)	Puurs (BE)	197 km

Table 6 Rotie depots location and distance to the processing facility

Subsequently, larger trucks are used to transport larger quantities to the processing facility (See Appendix II). A decentralised solution is available for large consumers of cooking oil, who can make agreements with biodiesel producers for a door-to-door collection system (Technical University of Crete, 2013). Rotie, one of the largest UCO collectors in the Netherlands, collects from 35,000 addresses and has five centralised depots (Rotie B.V., 2017). These depots and their route to the processing facility can be found in Figure 12 and Table 6.

Note that two of the five depots are not located in the Netherlands, giving us a maximum distance of almost 300 km. A range of 123-292 km, with an average of 200 km, has been identified for transport by 40-tonne EURO5 trucks (See Appendix II). As NExBTL is produced in Rotterdam an additional distance of 110 km from Amsterdam to Rotterdam has to be accounted for.

Rotie collects approximately 50,000-60,000 tonnes of UCO annually (Mijnheer, 2016). Assumed is that half of the UCO is collected around the main location in Amsterdam as this area has the largest collection radius (see Appendix II). Enquiry with a Rotie truck driver showed that an approximate distance in the range of 200-400 kilometres is driven per trip for this depot. The remainder of the collected UCO is divided evenly per depot. The collection-radius of each remaining depot is assumed to be less than the Amsterdam depot, as this concerns only easily collected used cooking oil to be economically profitable. An arbitrary range of 100-200km with a 150 km average has been used.

China

Similar collection methods are applied in Asia. The UCO is collected at restaurants and catering facilities by garbage trucks, there are instances where the UCO is recovered from city sewage systems. Collection distance is, generally speaking, lower when compared to the Netherlands, as there is a high density of UCO collection points in big cities (See Appendix II). The main areas in China where UCO is collected and shipped to Europe are coastal regions like Guangdong, Sichuan, Hebei and Shandong. Alongside road transport, some transport by barges over rivers is possible. An overview of energy requirements for UCO collection in China and Japan can be found in Table 7.

	Amount	Unit	Country & Source
Energy used for UCO collection in the form of diesel (to collection points)	30	MJ/t UCO	China (Ou, Zhang, Chang, & Guo, 2009)
Energy used for UCO collection in the form of diesel (to the processing plant)	135	MJ/t UCO	
Diesel use when UCO collected as mixed waste	0.109	l/10^{3}l UCO	Japan (Yano, Aoki, Nakamura, Yamada, & Sakai, 2015)
Diesel use Source separation (households)	11.4	l/10^{3}l UCO	
Diesel use Source separation (businesses)	9.07	l/10^{3}l UCO	

Table 7 Energy requirements for UCO collection in China and Japan

While Ou et al. (2009) offer specific energy use for UCO transportation the energy use for collection without a collection point is lacking, as opposed to Yano et al. (2015). As seen in section 4.2.3 China is the largest exporter of UCO for the European market, therefore these numbers will be adhered to. Ou et al. (2009) give a collection radius of 35 kilometres. Subsequently, this value has been converted to 116 km for the transport to the processing plant. As a system process for the collection a mix of EURO1-4 standards 7.5-tonne lorries has been used, as this is the best representation of emission standards for diesel trucks in China (Huo et al., 2012). The transport to the processing plant is done by larger vehicles, thus a system process of mixed EURO1-4 standards for 40-tonne lorries is adhered to. Assumed is that the UCO is shipped from this plant to western Europe, a general route from Shanghai to Rotterdam of 19,492.3km (sea-distances.org, 2018) has been used.

United States

UCO in the US is gathered under the overarching term 'yellow grease' (see section 4.2.2 'United States'). Yellow grease collection in the United States is a well-established industry. UCO collection from hotels, restaurants and the catering industry is highly competitive and therefore it is not expected that collection rates will improve (Nelson & Searle, 2016). In 2015 28 percent of available yellow grease has been exported (Lane, 2016). The main export areas are near east coast ports, assumed is that the collected yellow grease is from this area. Note that the UCO collectors and processors in the US are reclusive about their operations and not very willing to share information regarding their collection methods, processing, storage and further data related to export and transportation. While several UCO collectors have been contacted, no response has been received. Thus no bottom-up approach could be applied in this case. Similar circumstances where found by Spöttle et al., (2013) who describe the situation as *"a general scepticism about giving information to Europeans in this highly competitive market".* Very concise answers regarding UCO collection in the US where given by an employee of the United States Department of Agriculture, which come down to a system where UCO is collected by truck, but covered distance is variable per location. Main export ports are located on the East Coast, and further transportation is commenced by bulk shipping. An arbitrary collection distance of 75 km has been used, which matches the Norfolk/Portsmouth urban area. An additional large lorry transport over a distance of 380 km has been added as well to account for longer distance UCO transports which coincides with the distance from Washington DC to the Virginia Port International Terminals.

5.1.3 Depot Storage and Transportation

China

To achieve results for the GHG emissions of the China-Europe UCO flexitank pathway Calculation 9, below, has been used. Included is the average utilisation factor of 70 percent, as described in section

$$102.32\ kg\ CO_2\ per\ 800\ kg\ UCO = \left(\left(\frac{137.33\ g\ CO2/TEU}{23000\ l * 0.91\ g/cm3}\right) * 19492.3\ km\right) * 80\%$$

Calculation 9 CO_2 emissions for flexitank usage for the entire China Europe pathway per share flexitank UCO (800kg)

5.1.1 (Maritime Transportation). Note that actual utilisation factors might be closer to 90 percent in 2015 for the Asia→North-Europe trade lane (Lloyd's list, 2016), as opposed to the average 70 percent and low estimates 50 percent mentioned earlier. Using Calculation 3 a GHG emission of 106.8 grams of CO_2 per TEU kilometre can be calculated for a 90 percent utilisation factor. Note that this value does not account for the North-Europe→Asia trade lane, for which the utilisation factor might be substantially lower. For final emissions as used in Calculation 5, this would mean that if the utilisation factor is low (50 percent) associated emissions rise to 143.24 kg per shipped tonne UCO. Conversely, if the utilisation factor is high (90 percent) associated emission decline to 79.58 kg per shipped tonne UCO.

To account for the difference in percentages for this pathway 80 percent of the distance (= 15,593.84 km) is assumed flexitank transport and 20 percent (= 3,898.46 km) is assumed tanker transport. Calculation 10 is used to calculate the emissions for chemical tanker usage. Included is a 70 percent utilisation factor.

$$103.03\ kg\ CO_2\ per\ 200\ kg\ UCO = \left(\frac{26.43\ g\ CO_2/tkm * 19492.3\ km}{1000}\right) * 20\%$$

Calculation 10 CO_2 emissions for chemical tanker usage for the entire China Europe pathway per share chemical tanker UCO (200kg)

United States

Similar to the methodology for Asia-Europe transportation, a North-America-Europe transportation chain has been set up. As mentioned in section 5.1.1, premier UCO export ports are located on the east coast of the United States. One of the main ports for UCO export to the Netherlands is the port of Virginia where a total cargo of close to 3 million metric tons has been exported to the Netherlands in 2016 (The Port of Virginia, 2016). 'Food Waste and Animal Feed' is the fifth largest export category with 1.5 million metric tons, which includes UCO (ibid.). The port of Virginia knows four locations, the two largest being Norfolk and Portsmouth. The shipping route from the port of Virginia in Norfolk to Rotterdam is approximately 6,569 km (Grunau, 2016). Using an emission of 96.06g/TEUkm, determined in section 3.5.1 and Calculation 5 a 6.56 g CO_2/tkm emission can be calculated for the UCO North America East Coast – Europe trade route for flexitank shipping. This is 102.32 kg CO_2 for the total route including the 80 percent flexitank usage, per tonne UCO. Using Calculation 6 a trade lane emission for tanker shipping of 103.03 kg CO_2 is determined.

5.1.4 Conversion Process

5.1.4.1 Used Cooking Oil Pre-Treatment

Not all UCO has the same quality. Often the UCO is filtered in the country of origin to improve its quality and reduce particle pollution. This is done by using regular gravity filtration, thus no energy is needed in this process (personal communication). The pre-treatment before the trans-esterification/hydrotreatment process differs per pathway and is therefore addressed separately.

5.1.4.2 Conversion process Fatty Acid Methyl Ester

The esterification process is described by Yano et al. (2015), included are input/output values for this process. Values can be found in Table 8. Note that this process is described in a Japanese facility and European conversion values might differ.

Output		
FAME	1	L
Glycerol	0.396	L
Input		
UCO	1.04	L
Methanol	0.131	kg
KOH	7.6	g
Paraffin	0.0274	L
Electricity	0.184	L

Table 8 FAME esterification process inventory, using UCO as a feedstock (Adapted from: (Yano et al., 2015)*)*

A more detailed overview of conversion values has been described by (Morais, Mata, Martins, Pinto, & Costa, 2010). Values can be seen in Table 9. Note that these values are the result of an alkali-catalysed process with FFA pre-treatment. Two other processes are described, namely an acid-catalysed process and a supercritical methanol process using propane as co-solvent. The alkali-catalysed process with FFA pre-treatment is the most commonly used FAME conversion process (Vyas, Verma, & Subrahmanyam, 2010). Included in Table 9 is therefore also the pre-treatment stage of the UCO.

Output		
FAME	1000	kg
Glycerol	106.37	kg
Input		
UCO	1042.25	kg
Methanol	126.8	kg
NaOH	9.8	kg
H_2SO_4	0.15	kg
H_3PO_4	7.95	kg
CaO	0.1	kg
Glycerol	0.05	kg
Medium-pressure steam (250°C)	935.3	kg
Low-pressure steam (100°C)	1750.81	kg
Electricity	1.01	kWh
Water (Process)	48.65	kg
Water (Cooling)	3143	kg
Waste		
Salts	16	kg
Hazardous liquid waste	37.92	kg

Table 9 FAME esterification process inventory, using UCO as a feedstock (Adapted from: (Morais et al., 2010)*)*

While no impact assessment of this process has been conducted it gives an interesting insight in the complexity and workings of biodiesel refining.

5.1.4.3 Conversion process Hydrotreated Vegetable Oil

The process of hydrotreating vegetable oil is currently used by major corporations to create renewable drop-in fuels under different names; e.g. NExBTL by Neste Oil, the Ecofining process by UOP and H-Bio by Petrobras. Other HVO licenses do exist but the production process is similar (Honig, Linhart, & Orsak, 2015). The difference being the different feedstocks, with NExBTL being the main production process in the Netherlands (Lehmus, VP R&D, & Neste Oil, 2014).

Hydrogen production

Hydrogen can be produced using several different methods such as thermolysis, electrolysis and steam methane reforming (SMR) (Ogden, 1999). The latter is the most common production method in the United States, with 95 percent of hydrogen being produced this way (Office of Energy Efficiency & Renewable Energy, 2016a). In the Netherlands, this percentage is similar, with an estimated 87.5 percent being produced by steam reforming of fossil hydrocarbons (Flux Energie, 2017). This method produces a syngas of H_2 and CO using CH_4 as a feedstock (Ogden, 1999). More sustainable production methods are being researched, including microbial biomass conversion, photoelectrochemical direct solar water splitting and electrolysis using intermittent renewable power (Office of Energy Efficiency & Renewable Energy, 2016b).

Pre-Treatment

The UCO is pre-treated using two distinct chemicals and water, Phosphoric Acid (H_3PO_4) as an acid catalyst and Sodium Hydroxide (NaOH) as a neutralizing agent (Nikander, 2008).

Conversion Process

In the case of HVO production using the NExBTL process, an input of 50 percent palm oil and 50 percent waste oils is used (Appleyard, 2014). Other sources indicate an additional input of rapeseed oil but do not disclose the proportions (Nylund et al., 2011). The hydrogenation process is described by Yano et al. (2015) in the earlier referenced study, values of this research are presented in Table 10.

Output		
HVO	0.5	L
High-boiling-point oil	0.04	L
Low-boiling-point oil	0.16	L
Offgas	0.13	m3
Wastewater	0.04	L
CH_4	0.0059	kg/m3 wastewater
Input		
UCO	1	L

Electricity	0.24	kWh
H_2	0.0926	m3
N_2	0.125	kg
Heat energy	9.07	MJ

Table 10 HVO hydrotreatment, using UCO as a feedstock (Values adapted from: (Yano et al., 2015)*)*

While the values presented in Table 10 are comprehensive, a seemingly more relevant process of the NExBTL process is given in the Well-to-Wheel study by Edwards et al. (2013) for the European Joint Research Centre. Combined with the inputs as presented by Nikander (2008) an inventory table has been constructed. These values can be found in Table 11.

Raw materials		
Total Oils	1214	kg
Utilities		
H_3PO_4	1.154897	kg
NaOH	1.8451025	kg
Cooling Water (Pre-Treatment)	70	kg
Process Water (Pre-Treatment)	28	kg
H_2	42	kg
Cooling Water (Hydrotreatment)	4	kg
Process water (Hydrotreatment)	25	kg
Energy consumption		
Steam (Pre-Treatment)	657	MJ
Electricity (Pre-Treatment	50	MJ
Steam (Hydrotreatment)	29	MJ
Electricity (Hydrotreatment)	107	MJ
Waste stream		
Water	113	kg
Biological CO_2	48	kg
Co-Products		
Biogasoline	25	kg
Propane	72	kg

Table 11 NExBTL production process inputs for 1 tonne of fuel (Adapted from (Nikander, 2008) *&* (Edwards et al., 2013)*)*

Note that the data in Table 11 does not specify the feedstock that is used. Additionally, the data is sourced from a study published in 2008 and might be deprecated.

Assumed is that the utilities are market sourced and little to no transportation is required due to the location of the facility in the Rotterdam harbour (See Appendix II) with the exception being water. Water is mainly used for steam production and is primarily sourced from the Maas river in the case of NExBTL production in the Neste Rotterdam plant. Additional cooling water is sourced from an outside supplier and wastewater is treated in an on-site treatment plant and disposed of in local waterways (Ha, 2016).

5.2 Overview standard HVO pathway

To summarise what a typical HVO production process, using UCO as a feedstock, in the Netherlands might look like Table 12 Variables for transportation of UCO based HVO has been made. This includes variables which have been described in the sections above and information given in Table 12.This table can be used in combination with Figure 6 to give an impression of what the journey of UCO might be. Note that not all mentioned variables are included.

Pathway China	Collection Radius	35	km
	Depot Transport	135	km
	Distance Shanghai- Rotterdam	19,492.3	km
Pathway US	US Collection Radius	75	km
	US Depot Transport	380	km
	Distance Virginia- Rotterdam	6,569	km
	Distance Shanghai- Rotterdam	19,492.3	km
Pathway Domestic	Amsterdam Collection Radius	150	km
	Radius Depot Collection	75	km
	Depot - Amsterdam	100	km
	Amsterdam Rotterdam	110	km

Table 12 Variables for transportation of UCO based HVO

Flexitank shipping is used for 80 percent of overseas shipping, the remaining 20 percent is filled with

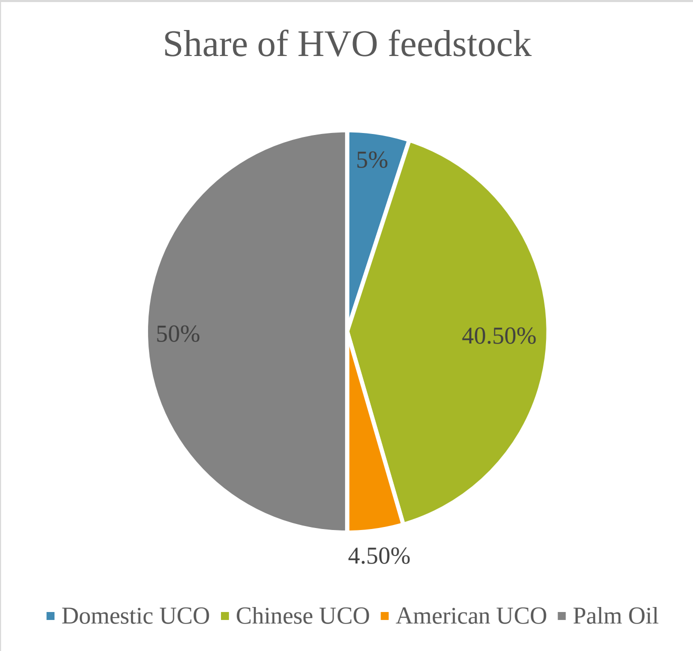


Figure 13 Feedstock input for the standard HVO pathway

bulk chemical tankers. Both modes of transportation have a utilisation factor of 70 percent. The input shares for the standard pathway are given in Figure 13.

5.3 Alternatives in the system

To determine the system boundaries of the CLCA, one has to seek out which processes see a change as a consequence of an increased demand of the functional unit. Furthermore, it has to be made clear if the input is the result of a multifunctional activity such as discarded cooking oil, whether the input is dependent on the activity or vice versa (Brandão, Martin, Cowie, Hamelin, & Zamagni, 2017). UCO is being regarded as a dependent product, as the output does not change due to an increased biofuel demand. But one can argue that this has changed in recent years due to the increased demand for UCO (see section 4.2.2). Also considered should be co-products during production and alternative feedstocks used for biodiesel production, which will be displaced.

Glycerol as a by-product of FAME production has a displacement effect on the production of regular glycerol. Petroleum-based glycerol production facilities have been closed down because of the market effects of the acceding FAME glycerol (Knothe, 2010). As described in section 5.1.4.3 ('Conversion Process') during HVO production there is no by-production of glycerol. This would mean that if the trend of increasing HVO production in the Netherlands (see section 4.1) will continue, a lack of glycerol as a by-product needs to be compensated for by petroleum-based glycerol (Knothe, 2010). Compared to the transesterification process the petrochemical route is more complex and energy consuming (ibid.), and associated GHG emissions would be higher.

5.3.1 Alternative uses UCO

Europe

In the European Waste Catalogue, UCO is classified under code 20 01 25/20 01 26 as a Municipal Waste (SEPA, 2015), and therefore sees no other function than being used for the production of biodiesel. In the past different uses for discarded cooking oil in Europe where apparent; before the implementation of EU Directive 2003/100/EC (European Commission (EC), 2003) following an incident as the result of feeding food wastes to animals, UCO has been widely used as an ingredient in animal feed (SEI, 2003). This practice is, as of 2017, still apparent in other countries such as Australia (SFMCA, 2016) and the United States (NRA, 2017), albeit strictly regulated.

China

Such strict regulations and policy instruments, as in place in Europe, do not seem to be in place throughout China, where UCO is often inconsiderately disposed of and further processing not regulated. In some cases, this leads to reuse of waste cooking oil for food applications which poses several health threats (Zhang, Wang, & Mortimer, 2012). This use alternative use is poorly tracked and data is generally unavailable, therefore it will be neglected. As this study focuses on the Netherlands and its import from Asia, no alternative uses for UCO have been identified.

United States

Yellow grease is generally stored at the food service establishment until it is transported to a rendering facility. Here a share of the collected yellow grease may be rendered into tallow which is used by the animal feed industry. This is permitted in the United States due to yellow grease being gathered from food preparation processes and not contaminated by wastewater (Motta, 2016).

5.3.2 Alternative feedstocks for biodiesel

As discussed in section 5.1.4.3 current production of NExBTL in the Rotterdam production facility involves, to some extent, oils of different origin than UCO. Included is palm oil which, depending on origin, can only meet sustainability standards if produced on degraded land as land use change is regarded as the main contributor to associated GHG emissions (Wicke, Dornburg, Junginger, & Faaij, 2008). In an earlier study, Wicke, Dornburg, Faaij, & Junginger (2007) identify a value of 50 g CO_2-eq/MJ for the crude palm oil chain which excludes LUC emissions. This is the best option as land use change is a main aspect of emissions and the choice of what land is planted with oil palm is significant (Wicke et al., 2007). Included in this research is transport to Rotterdam and production takes place in Malaysia. This value will be included in the results as it can influence the final GHG emissions for HVO production. Furthermore, it is interesting to pose a discussion on a change in feedstock percentages and will be expanded upon in section 7.1. For the standard supply chain a value of 50 g CO_2-eq/MJ has been used, with an energy density of 37 MJ/kg based on the JEC E3-database (version 31-7-2008) this becomes an emission of 1850 kg per tonne crude palm oil (CPO).

6 Life cycle Impact Assessment

6.1 UCO requirements

Due to the trends which have been observed the EU market should see an increase in demand for UCO based biofuels. The waste-based biodiesel consumption is expected to grow to more than 400,000 million tonnes in 2018 (Hillairet, Allemandou, Golab, & Limouzy, 2018). Note that this forecast is not restrained to UCO based biodiesel. This growth would mean an increased demand for UCO in the Netherlands, which will, highly likely, be filled with intra-EU and intercontinental imports. However, there are some factors which could reduce UCO demand in the Netherlands. Firstly, due to the proposed RED II, it will be more difficult to meet sustainability criteria for biofuels using UCO as a feedstock. This will be discussed more elaborately in section 7.5. Secondly, there will be an increased supply of UCO based biofuels originating from Asia. This region is growing production rapidly and developing to become a large producer of waste based biodiesel (Hillairet et al., 2018), therefore exports to Europe might decline for UCO as a feedstock. To summarise these expectations, the demand for UCO will grow, albeit it less exponential, and prices will rise for this feedstock.

6.2 Lifecycle Inventory

6.2.1 GHG emissions per pathway

When the pathways for HVO production in the Netherlands are compared with each other, UCO transport via chemical tankers seems to be the least favourable option in terms of GHG emissions. Shown in Figure 14 are the GWPs per pathway for 1 MJ of UCO based HVO, based on the assumption that 100 percent of the UCO originates from the indicated source. The conversion process emissions do not change in the comparison, this is because the process is identical for each UCO source. When analysing the results one can observe that chemical tanker shipping is the most prominent contributor to GHG emissions for UCO transportation and HVO production in general. As mentioned earlier this might be due to the polluting nature and low efficiency of smaller size vessels.

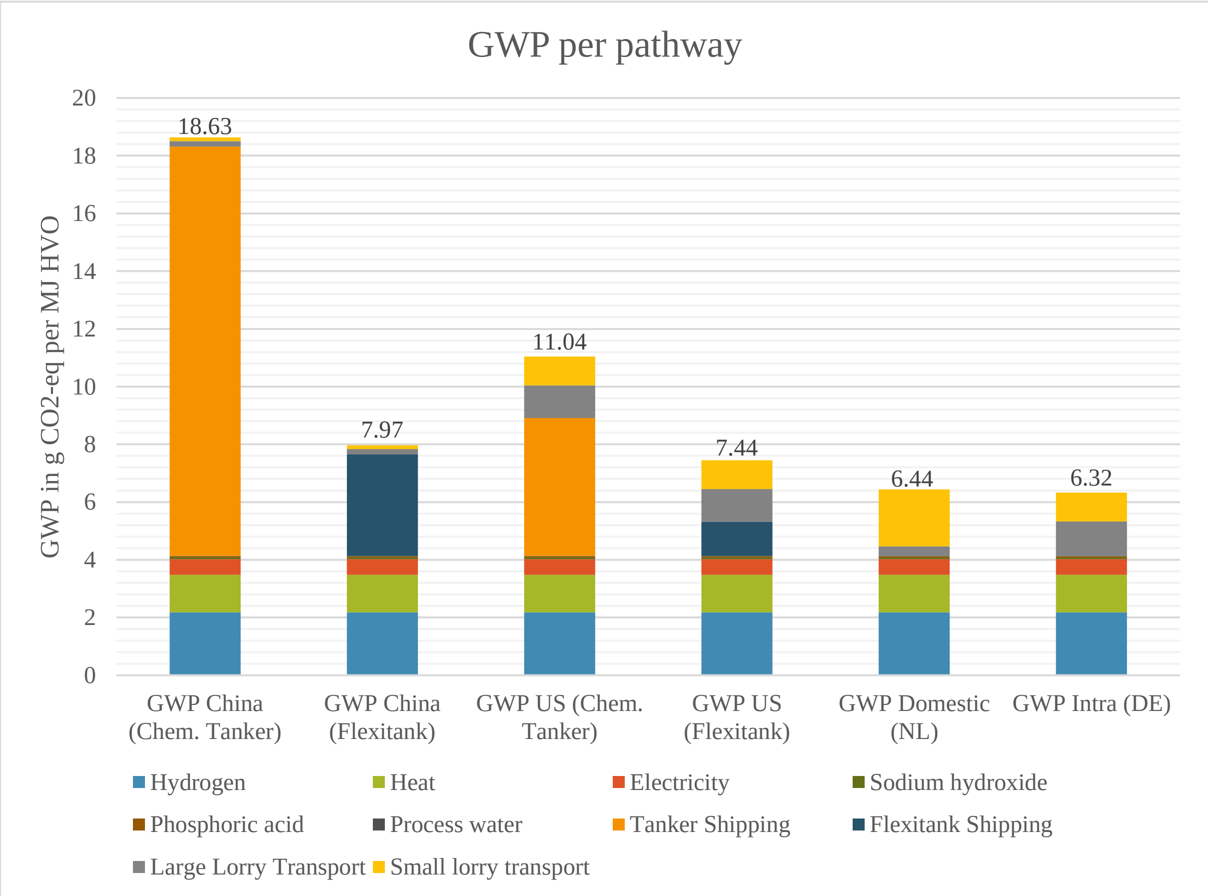


Figure 14 GWP of six different pathways per MJ of UCO based HVO

Comparing this with the default values for waste cooking oil HVO as given by the Joint Research Centre (JRC) (Edwards et al., 2016) is done in Figure 15, note that a direct comparison is inadvisable as there are different system boundaries adhered to.

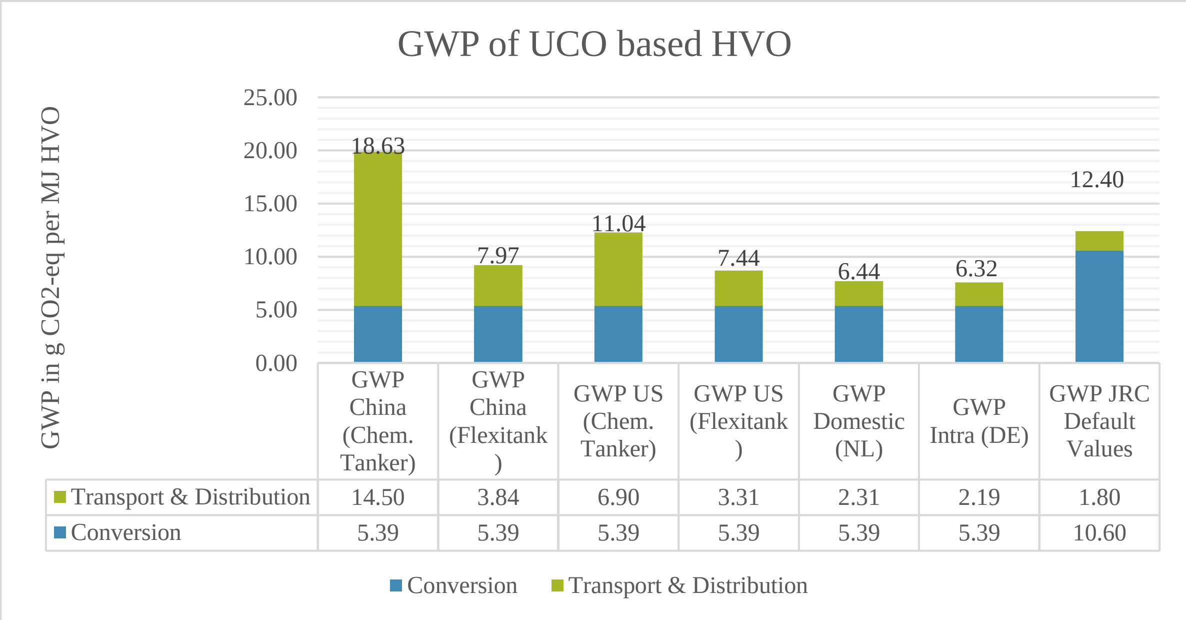


Figure 15 GWP of UCO based HVO pathways

Noteworthy is that the emissions of the conversion process as calculated by Edwards et al. (2016) are nearly twice as high as the results in this report. This may be contributed to a multiplication factor of 1.4 which is applied to the typical value in the JRC report. This results in higher conversion emissions. Note that there are differences in input variables which do not contribute as much to the lareger processing emissions. Electricity use, for example, is twice as low in the RED methodology 1.55 kJ/MJ$_{fuel}$ versus 3.56 kJ/MJ$_{fuel}$. H_3PO_4 input is also significantly lower, 0.00002 kg/MJ$_{fuel}$ compared to 1.15 kg/MJ$_{fuel}$. The latter can be explained as it is used as a cleaning chemical for UCO and this process is not considered in the JRC methodology (Edwards et al., 2016). The lower Transportation & Distribution are mainly contributed to the increased transportation distances and difference in transportation mode considered during this research.

6.3 LCA midpoint impact results

The following results have been calculated for the standard HVO pathway as described in section 5.2. Total GHG emissions are given in mg CO_2-eq as the difference between emissions is small, when carbon dioxide is not considered. Shown in Table 13 are the GHG emissions and the associated GWP for 1 MJ of biofuel. Note that these are accumulated values, a more detailed inventory table can be found in Appendix III (Table 20).

Total	*32.3766*	*g CO2-eq*
Carbon dioxide	31.5221	g CO2-eq per MJ HVO
Methane	0.8218	g CO2-eq per MJ HVO
Dinitrogen monoxide	0.0243	g CO2-eq per MJ HVO
Others	0.0085	g CO2-eq per MJ HVO

Table 13 Inventory of GHG emissions for the standard pathway of UCO NExBTL

These results show that the main emissions from HVO production from UCO are in the form of carbon dioxide.

6.4 Process contribution results

The results per pathway, solely for UCO transportation are given and illustrated in Table 14.

	GWP China (Chem. Tanker)	GWP China (Flexitank)	GWP US (Chem. Tanker)	GWP US (Flexitank)	GWP Domestic (NL)	GWP Intra (DE)
Tanker Shipping	14.1820	0	4.7794	0	0	0
Flexitank Shipping	0	3.5200	0	1.1863	0	0
Large Lorry Transport	0.1865	0.1865	1.1345	1.1345	0.3284	1.2001
Small lorry transport	0.1325	0.1325	0.9902	0.9902	1.9804	0.9902
Total	14.501	3.839	6.9041	3.311	2.3088	2.1903

Table 14 GWP of UCO transportation per pathway in gCO_2-eq per MJ HVO

It is clear that tanker chemical shipping from China is associated with the highest emissions. Road transportation of UCO has emissions ranging from .13 to nearly 2 gCO_2-eq. Noteworthy is that UCO when collected in the Netherlands by small trucks has higher associated emissions than flexitank shipping from the US. But the total pathway has a lower GHG footprint as there is also collection taking place in the US.

Shown in Table 15 is the contribution to the GWP in absolute numbers, for the standard UCO HVO pathway, again as described in section 5.2, measured in g CO_2-eq per MJ.

Total	32.37662	g CO2 eq per MJ HVO
Palm Oil	25.46377	g CO2 eq per MJ HVO
Hydrogen, liquid (EU)	2.176664	g CO2 eq per MJ HVO
Tanker Shipping	1.1917667	g CO2 eq per MJ HVO
Heat (EU)	1.2990434	g CO2 eq per MJ HVO
Electricity, medium voltage (NL)	0.54883942	g CO2 eq per MJ HVO
Flexitank shipping (CN)	1.1404935	g CO2 eq per MJ HVO
Small lorry transport (US&NL)	0.19308503	g CO2 eq per MJ HVO
Large lorry transport (US&NL)	0.082397146	g CO2 eq per MJ HVO
Lorry Transport (CN)	0.075543255	g CO2 eq per MJ HVO
Sodium hydroxide	0.054625702	g CO2 eq per MJ HVO
Small lorry transport (CN)	0.053665081	g CO2 eq per MJ HVO
Phosphoric acid	0.04609229	g CO2 eq per MJ HVO
Flexitank Shipping (US)	0.042705772	g CO2 eq per MJ HVO
Process water	0.007928567	g CO2 eq per MJ HVO

Table 15 Global Warming Potentials per process for the standard HVO pathway

These results show that a main contributor to the GWP of NExBTL HVO is palm oil. Other important aspects are hydrogen production, heat production and shipping. As the results might look skewed with

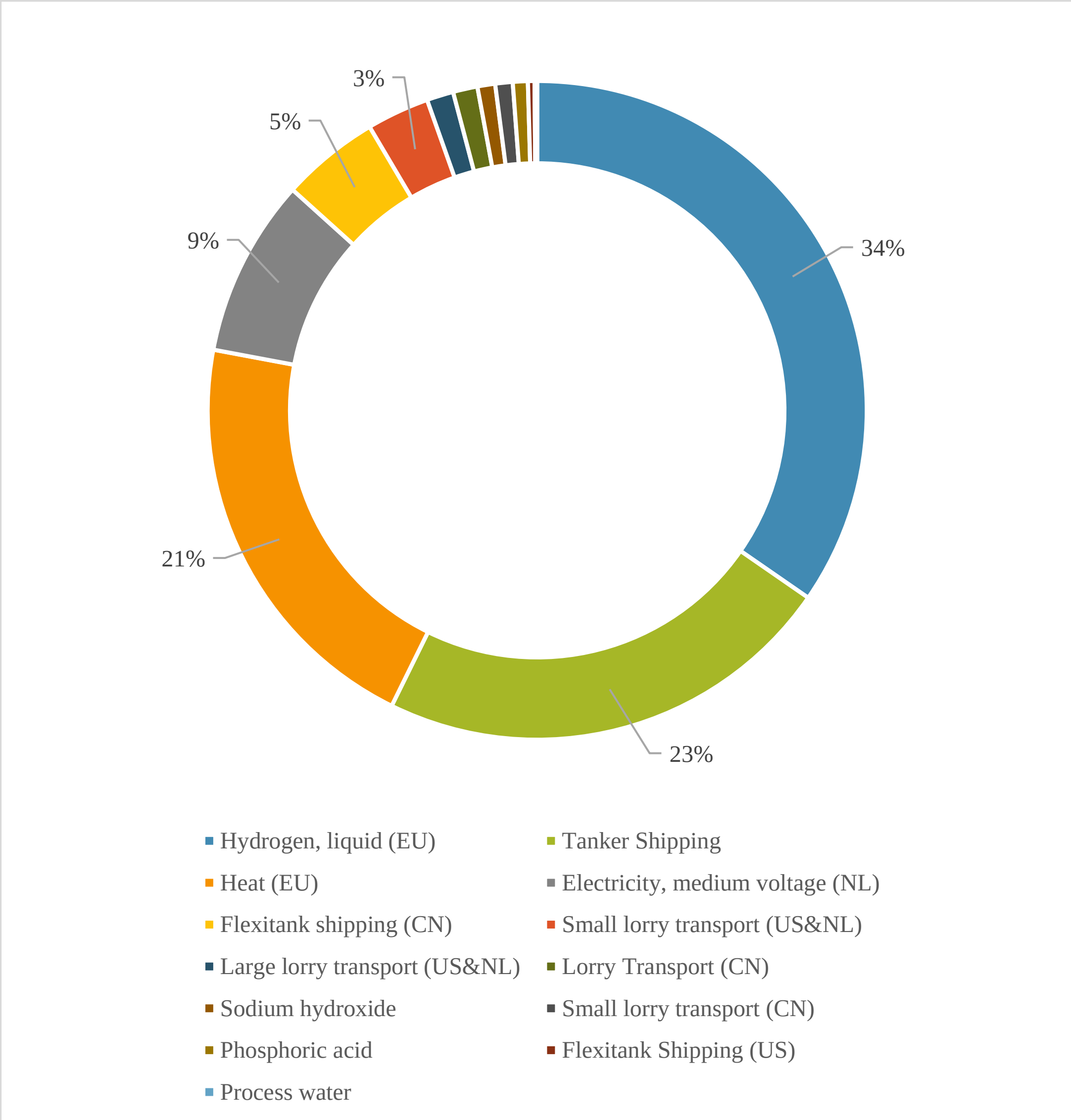


Figure 16 Process Contribution to emissions in HVO production from UCO minus palm oil

an input of palm oil, the contribution percentages are presented without it in Figure 16. This highlights the importance of some processes in the NExBTL HVO production process.

6.5 Sensitivity Analysis

Hydrogen production method

The base results for HVO conversion assume hydrogen production as a European mix. As discussed in section 5.1.4.3, the majority originates from SMR of natural gas but more sustainable methods of producing hydrogen are being researched. An overview of the GWP of different hydrogen production methods in Europe is given in Figure 17.

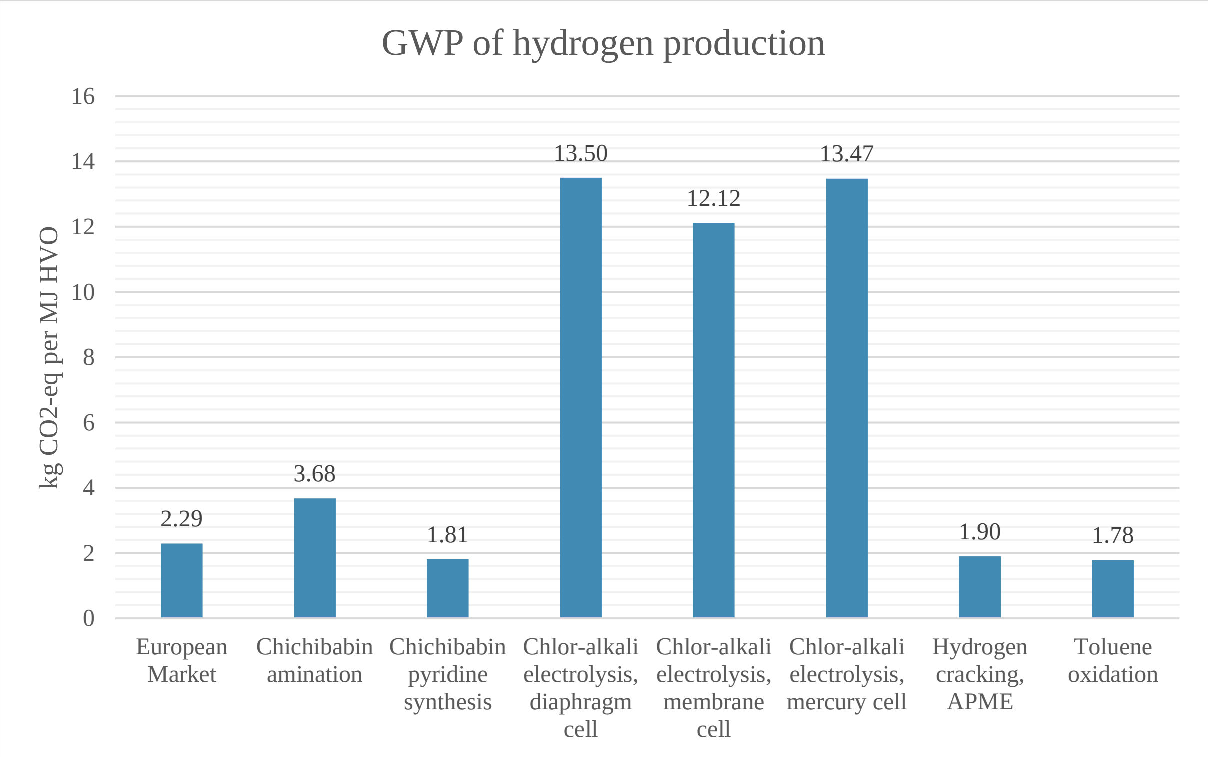


Figure 17 GWP of different hydrogen production methods in Europe

From this figure we can conclude that there is a large variation in environmental pressure of hydrogen production methods. When taking the outer bounds of pollution during hydrogen production and comparing this to the standard process the impact and sensitivity of the choice of hydrogen can be seen. This is shown in Figure 18.

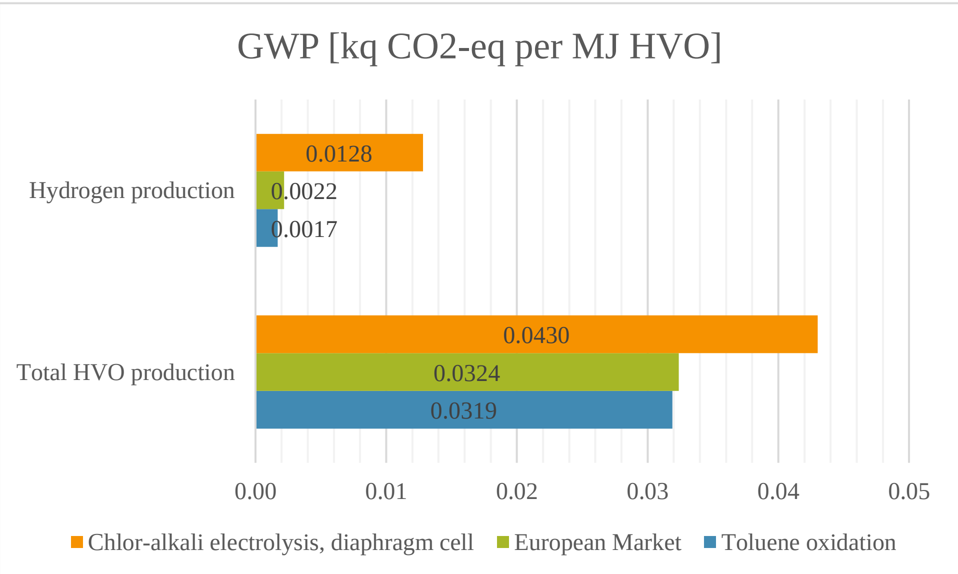


Figure 18 Bandwidth of GWP as a result of change in hydrogen production.

When choosing the least polluting option, in this case toluene oxidation, a decrease in GWP of hydrogen production of 22 percent is observed which results in a 1.5 percent decrease of the GWP of the standard HVO pathway. On the other hand, choosing chlor-alkali electrolysis by a diaphragm cell results in a 489 percent increase of GWP of hydrogen production and corresponding 33 percent increase of the standard pathway GWP. This shows that the GHG performance of HVO depends highly on the production method of hydrogen. Note that these production methods might not be suitable for industrial production.

Overseas UCO shipping

As shown in Figure 14 the choice between overseas UCO transport modes has a large impact on the GHG performance of UCO based HVO. When compared to the standard pathway it shows that the mode of overseas shipping does have a significant impact on both GWP of shipping and the GHG performance of the entire HVO production pathway, as shown in Table 16.

GWP Shipping	Total Pathway [kg CO2-eq per MJ HVO]	Difference	Shipping [kg CO2-eq per MJ HVO]	Difference
Standard Pathway (excl. Palm Oil)	0.0069	-	0.0024	-
Chem. Tanker (CN)	0.0186	170%	0.0142	497%
Flexitank (CN)	0.0080	15%	0.0035	48%
Chem. Tanker (US)	0.0110	60%	0.0048	101%
Flexitank (US)	0.0074	8%	0.0012	-50%

Table 16 GWP of overseas shipping in comparison with the standard HVO pathway

Note that the GWP of the palm oil has been excluded for this standard pathway to make an even comparison. One may conclude that chemical tanker shipping from China is the worst mode of overseas transportation regarding GHG emissions and flexitank shipping from the US is the most favourable. Only slight increases are seen in the total pathway GWP, especially when compared to the influence of the hydrogen production method. Another interesting perspective is given when the origin of the UCO is considered.

UCO Origin

When changing the origin of the UCO it becomes clear that, again, the GHG performance of UCO HVO is influenced. While hydrogen production methods and overseas shipping methods have a larger influence, the origin of UCO is still considerably important as can be seen in Table 17.

UCO Origin	Total Pathway [kg CO2-eq per MJ HVO]	Difference
Standard Pathway (excl. Palm Oil)	0.0069	-
Domestic (NL)	0.0064	-7%

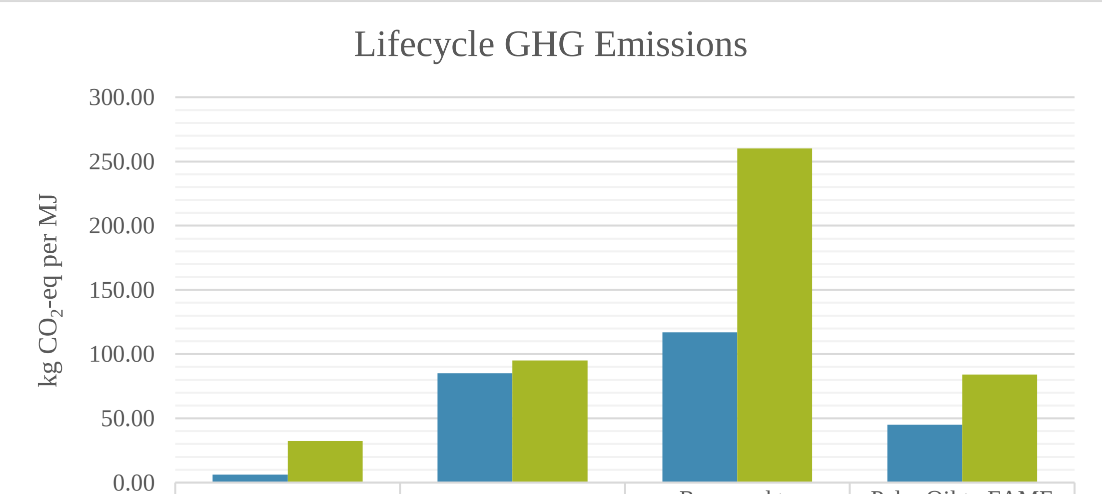
Intra (DE)	0.0063	-9%
Overseas (US)*	0.0082	18%
Overseas (CN)*	0.0101	46%

*Table 17 GWP of different UCO origins and standard pathway comparison *Assuming 80 percent flexitank and 20 percent chemical tanker shipping*

Note that palm oil is excluded in this analysis. The main conclusion from this analysis is that it is more beneficial to source UCO from within a near distance of the refinery. It is noteworthy that it does not mean that a smaller distance is always better as domestic UCO collection often commences over longer distances in smaller inefficient trucks when compared to their intra-EU counterparts. However, this reasoning is based on one case with various assumptions and in reality the results may differ. It is suggested to investigate these different pathways in follow-up research to give a deeper understanding of the UCO pathway and its ranges of uncertainty.

6.6 Comparison with other biofuels

To assess the true GHG reduction potential of UCO based HVO a comparison between different transport fuels has been made. This comparison can be found in Figure 19 (Lifecycle) GHG emissions



	NExBTL	Fossil Diesel	Rapeseed to FAME (EU)	Palm Oil to FAME (Indonesia)
Minimum	6.32	85.00	117.00	45.00
Maximum	32.38	95.00	260.00	84.00

Figure 19 (Lifecycle) GHG emissions of various transportation fuels (Adapted from: (Cherubini et al., 2009)

of various transportation fuels (Adapted from: (Cherubini et al., 2009) Note that for the values of NExBTL the best performing pathway is the Intra EU (DE) 100 percent UCO pathway. The worst performer is the standard pathway due to large palm oil emissions. The same holds true for other biodiesel alternatives where the impact of ILUC is included. When accounting for burden sharing of the GHG of virgin oils, as explained in section 7.4, an additional 5.2 gCO_2-eq will be added to the standard pathway and 10.4 gCO_2-eq if the HVO is wholly produced using UCO as a feedstock.

7 Interpretation

7.1 Feedstocks

An interesting perspective can be given if the outliers in the system are investigated, and what scenarios are possible. For example, as indicated there seems to be a trend towards increasing bulk tanker shipments (see Appendix II) with associated higher carbon emissions (see 5.1.3). Therefore the GHG footprint of UCO based biodiesel might increase in the future. Also, the collection distance of UCO is susceptible to variance, which might influence the GHG emissions to some extent. As well as the proportions in which palm oil and UCO are used in the HVO production process.

Currently, the Neste plant in Rotterdam is not solely using UCO as an input for the production of NExBTL. As indicated in 0 a percentage of 50 percent UCO and 50 percent palm oil is adhered to. With a capacity of 800,000 t/yr this means that annually 485,600 tonnes of UCO are utilised at this location. If there are ambitions to rely entirely on UCO for NExBTL production, 971,200 tonnes of UCO would be needed. This exceeds the collection in the Netherlands as indicated in Table 1 by almost 1,500 percent and the estimates for 2020, as proposed by Koppejan et al., (2009), by ~750 percent. This is, under certain circumstances, expected to be beneficial for the GHG footprint of NExBTL HVO as palm oil is a large contributor to the GWP and will be excluded from the mix. Note that the origin and transportation of UCO also play a large role.

7.2 Transportation

Transportation of UCO is a large contributor to the GHG footprint of NExBTL HVO. As mentioned in the sensitivity analysis it matters quite considerably where the UCO is sourced from and its mode of transportation. When the UCO transport is carried out using bulk transport as described in 5.1.3 instead of flexitank shipping, related CO_2 emission will increase from 6.56g CO_2-eq to 26.43g CO_2-eq per tkm. This increase of over 300 percent for bulk tanker shipments results in a potential increase of almost 134 percent of total GHG emissions for NExBTL fuel.

UCO transport emissions have a large range. Overseas shipping of UCO for biodiesel production has associated emissions ranging from 1.19 to 14.18 gCO_2-eq (see Table 14).The default GHG emissions as given by JRC for UCO shipping are .25 gCO_2-eq, which is considerably lower than the estimations found in this study. The GHG performance of UCO based biofuels can be altered significantly depending on the transportation mode. It is unlikely that new modes of UCO transportation will emerge, but the transport sector is incessantly evolving and more sustainable options will become available eventually (CCWG, 2017; Singh et al., 2015)

7.3 Conversion Process

The largest contribution to the GHG performance of HVO in the NExBTL conversion process is hydrogen usage, according to Figure 16. The changes in performance as a result of changing hydrogen production are discussed in section 6.5. To iterate: Several hydrogen production methods are available, ranging from polluting coal gasification to more sustainable renewable energy electrolysis(Singh et al., 2015). Of these production methods SMR is most commonly used for industrial processes but if alternatives such as renewable energy electrolysis become cheaper a large reduction in the GWP of NExBTL HVO is to be expected. The same holds true for the electricity used during the production process which, if renewably sourced, can reduce the GWP up to 10 percent according to Figure 16. The origin of heat and its impact on the sustainability of HVO NExBTL in the production process is prone to some assumptions. In this research an average heat mix in the chemical industry of Europe has been used. More sustainable options might be available if the heat is sourced from biomass from example, but this also comes with some discussion regarding its sustainability, availability and costs (Lamers, Hoefnagels, Junginger, Hamelinck, & Faaij, 2015).

7.4 Burden sharing

If one compares the price development of UCO as a biofuel feedstock to the price development of virgin vegetable oils, the value of the 'waste' UCO can sometimes approach or even surpass prices of virgin oils as can be seen in Figure 20. Note that UCO prices have been converted to a dollar per tonne price, and thus might be influenced by changing exchange rates.

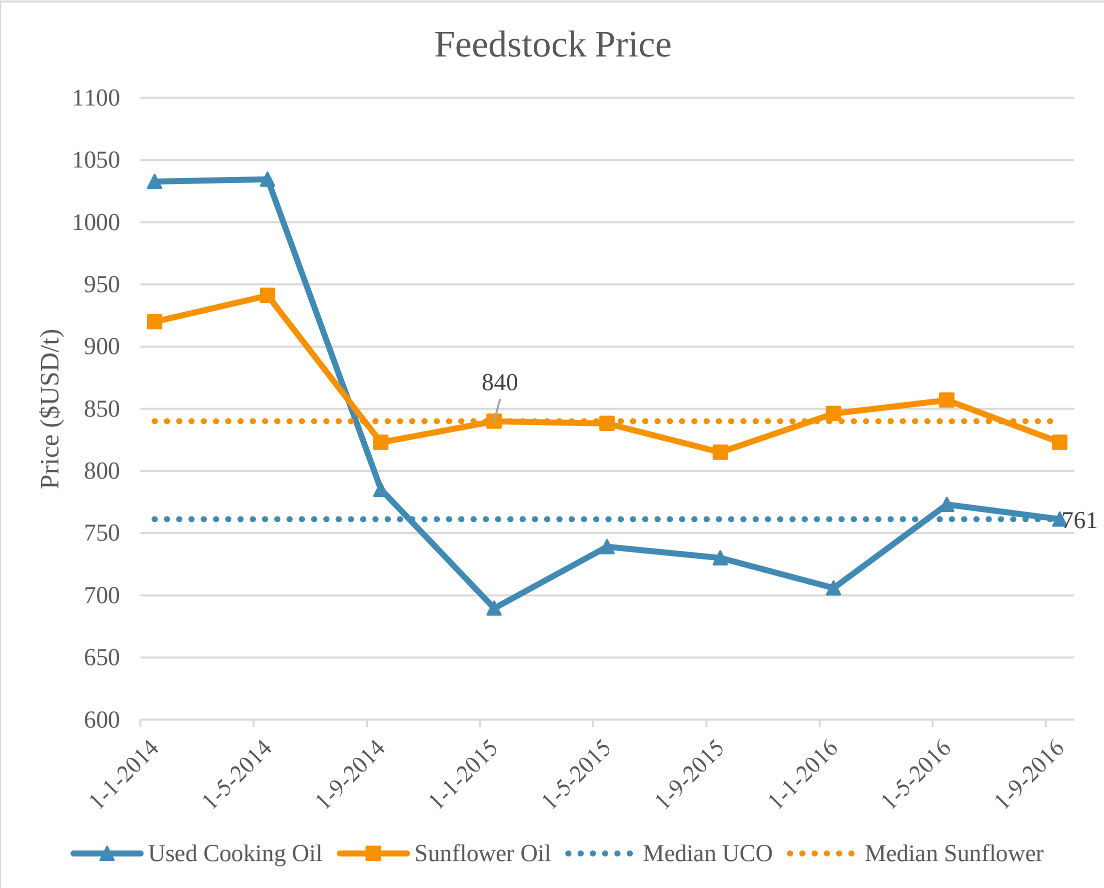


Figure 20 Virgin Sunflower oil and UCO price developments 2014-2016 (Adapted from: (Hillairet et al., 2017; Thoenes, 2018))

Thus one may argue that some of the GHG emissions resulting from virgin oil production have to be carried by the resulting UCO. A so-called GHG burden-sharing based on economic allocation can be calculated using Calculation 1 and the median prices of virgin sunflower oil and used cooking oils as presented in Figure 20 Virgin Sunflower oil and UCO price developments 2014-2016 (Adapted from: (Hillairet et al., 2017; Thoenes, 2018)). This method gives a value of 47.53 percent of sunflower oil production emissions which can be allocated to UCO. According to the Biograce GHG tool version 4d, the emissions of 1 kg sunflower oil are 795.62 g CO_2-eq. This value already includes an allocation to meal production and is in compliance with the RED calculation method based on JEC data (Marques, 2015). This means that 378.15 g CO_2-eq can be allocated to UCO, assuming that UCO consists of used sunflower oils. This means that for 1 MJ of NExBTL an additional 5.2 gCO_2-eq can be allocated to the standard pathway. Twice that amount for a 100 percent UCO pathway. Note that losses are not accounted for and 1 kg of virgin sunflower oil produces the equivalent amount of UCO.

7.5 GHG Reduction Potential

Using Calculation 2 GHG emission reduction in the RED and data from Figure 19 (Lifecycle) GHG emissions of various transportation fuels (Adapted from: (Cherubini et al., 2009), the GHG reduction potential according to RED methodology can be calculated, the results can be found in Table 18.

Reduction Potential per Pathway compared to fossil diesel (85-95 kg CO_2-eq/MJ)	GWP [gCO2-eq/MJ HVO]	Min	Max
Standard	32.38	61.9%	65.9%
Intra EU (DE) 100% UCO	6.32	92.6%	93.3%
Standard, Burden Sharing	37.58	55.8%	60.4%
Intra EU (DE) 100% UCO, Burden Sharing	16.72	80.3%	82.4%

Table 18 Reduction potential per UCO based HVO pathway

Unsurprisingly, the highest GHG reduction potential can be achieved with an Intra EU, 100% UCO based pathway. Contrarily, the lowest reduction potential is achieved when emissions from virgin oil production are accounted for and a share of palm oil is used during the production process. All pathways meet the current GHG savings requirements of 50 percent, stated in the RED (International Council on Clean Transportation, 2017). As the Neste NExBTL plant in Rotterdam started production in 2011 the stricter RED targets of 60 percent are not applicable, as this is only for plants after 2015. Considering that UCO based HVO is a biofuel listed in Annex IX of the newly proposed RED II it has to meet the requirement of 70 percent GHG savings from 2021-2030 (ibid.). According to Table 18 this is only possible if current operations change and either the origin of the UCO shifts towards intra-EU, UCO transportation modes change to less polluting alternatives or the share of palm oil decreases. Note that these statements are based on the methodology as proposed in this paper and might differ from values proposed in the RED II. As can be seen the default GHG emissions savings of 83 percent in the RED is, depending on the on the pathway, surpassed or never reached.

8 Discussion

The aim of this study was to assess the lifecycle GHG footprint of HVO produced from UCO taking into account the variability occurring from actual intra-EU and extra-EU feedstock supply chains. Additionally, a more transparent and realistic GHG emission reduction potential for UCO based biodiesel is presented and compared to the methodology as described in the EU RED. Namely, it is suggested that the applied methodology for default values in the RED has its shortcomings. To fulfil the aim of the study, data concerning recent developments in international UCO market has been gathered and subsequently an attributional LCA of NExBTL HVO in the Netherlands has been conducted.

The results show large variations between the GHG footprints of the assessed supply chains. This variation is to a great extent determined by the shipping method, followed by the origin of the UCO. Smaller variations are observed within the intra-EU supply chains where the impact of larger covered distances is compensated by more efficient transportation modes. These two aspects, the feedstock shipping method and origin are key factors in determining the sustainability of UCO as a sustainable biodiesel feedstock.

When comparing the results to the RED emission targets and default value the main findings are that current requirements are met but the stricter targets in the RED II can only be met if current operations change. Furthermore, it seems that the actual greenhouse gas footprint of UCO based NExBTL is higher than the default value as proposed in the EU Renewable Energy Directive 2009/28/ EC. This is an important finding considering that the production of this fuel is stimulated in said directive under the assumption that it is a sustainable biofuel. The difference can partly be explained due to different adhered system boundaries, differences in the production process and differing input-data .For example, a different value is used for overseas shipping of the feedstock. To overseas shipping of UCO the JRC adhered a value of 0.1892 tkm/ MJ_{UCO}. To compare this to values found in this study it has been converted using data given by the JRC. The result is an emission of 0.256 gCO_2-eq/ MJ NexBTL. This is substantially lower than the range of 1.19 to 14.18 gCO_2-eq (see Table 14) found in this thesis.

When comparing the results of this study to a system where the same production process is adhered to the results seem similar. Nikander (2008) gives a greenhouse gas footprint of 34.4 gCO_2-eq compared to 32.4 gCO_2-eq in this study. However, Nikander assumes a 100 percent UCO feedstock for this result while in this research a 50/50 mix of UCO/palm oil is assumed for the standard pathway. This would mean that the GHG footprint of this study is comparatively low as the majority of the emissions can be assigned to palm oil production. A pathway based solely on palm oil in this research would have, presumably, a larger GHG footprint. This is due to the comparably high palm oil emissions which are used during this research. Unsurprisingly, Nikander applies an emission factor of palm oil which is almost half as large (50 gCO_2-eq/MJ versus 26.3 gCO_2-eq).

When excluding palm oil from the production process and comparing the results of the 100 percent UCO pathways with a similar study, comparable results can be found. Johnson (2017) presents in his study a value of 9.3 gCO_2-eq/MJ for UCO based HVO compared to an average of 9.64 gCO_2-eq/MJ for the 6 100 percent UCO pathways in this study. However, this similarity can be misleading as the production method is different and economic allocation is applied by default in the referred study. When economic allocation in the form of burden sharing applied to the results found in this research, the final results are higher. Furthermore, comparing the average values with specific results can give a deceptive view. Thus, no direct conclusions can be drawn.

8.1 Limitations of the study

The carried out study is mostly based on secondary data and assumptions where data is lacking. A more resourceful conclusion can be drawn when first-hand reports of the NExBTL production process are provided. Assumptions include the type of tanker used for shipping UCO, as no specifics were found for this subject. This is an important aspect of the study which can have a large influence on the results as discussed during the sensitivity analysis. Further assumptions during the market analysis include the price of feedstock which is for the European market. Prices of UCO in China, for example, might differ and a global market price analysis is recommended for further research.

When researching biofuel pathway performance using LCA's one has to take into account different sources of variability and uncertainty. This variability can be induced by the model and method, for example system boundary selection, allocation method and quality of datasets, and might differ from actual uncertainty and variability in biofuel pathway performance (Kendall & Yuan, 2013). When examining the variability and uncertainty described by Kendall & Yuan (2013) it becomes clear that in this study not all sources have been covered. Although actual and method induced variability are indistinguishable (ibid.) it is suggested to repeat this study with a different methodology to reduce uncertainty. This can be a CLCA, expanding system boundaries to include distribution, use updated LCI datasets, include (indirect) land-use change, etc.

8.2 Future directions for the used cooking oil biofuel sector

During the research phase of this study it became clear that there are several barriers and obstacles that made the process of data collection demanding. In general, the used cooking oil industry can be considered opaque to the eyes of outsiders. Gathering detailed information is difficult due to the competitiveness within the industry. Additionally, as the industry of UCO based biofuels is still in rapid development there is inconsistency in definitions, distinctions and descriptions used. Moreover, these attributes are sometimes even lacking in literature and legislation. Therefore some critiques and suggestions for policymakers and industry professionals are given below to improve the consistency and transparency of the industry and legislation.

8.2.1 Uncertainties and inconsistencies in international statistics

The term UCO, as used in literature, may represent very different kinds of oils and fats. It can be difficult to make a characterisation between oils of animal and vegetable origin. While in Europe the term is mostly used for recovered vegetable cooking oil, in the US there seems to be no distinction being made. This results in the overarching term 'yellow grease' which includes animal fats derived from rendering, often of lower quality (Spöttle et al., 2013).

Further lack of consistency is found when investigating UCO trade, domestic and international. Use of Harmonised System (HS) nomenclature, which aims to be a standardised system to classify traded goods, is not applied consistently throughout the sector. The same holds true for Combined Nomenclature (CN) by the EU. This is due to the fact that there is no specific code for UCO. Eurostat suggests[2] the CN-code 1518 00 95 '*Inedible mixtures or preparations of animal or of animal and vegetable fats and oils and their fractions*' upon inquiry. Alternatively, the CBS suggests[3] code 1518 00

[2] Based on e-mail conversations with personnel of said institution
[3] Based on e-mail conversations with personnel of said institution

39 '*Fixed vegetable oils, fluid, mixed, inedible, n.e.s., for technical or industrial uses (excl. crude oils and for production of foodstuffs)*'.

The default values for the GHG reduction in the RED mention "waste vegetable or animal oil biodiesel", separately addressed are different HVO pathways; rapeseed, sunflower and palm oil. This would suggest that the pathway for UCO based HVO is not specifically described. This makes a comparison with the RED difficult.

These uncertainties and inconsistencies in international statistics are often perceived when studying international bioenergy trade data in general (Proskurina, Junginger, Heinimö, & Vakkilainen, 2017). International bioenergy trade statistics are sensitive to incoherent and insufficient data availability, which is currently low for China in particular (ibid.). Therefore, there should be international efforts made to attain a clearer insight in international bioenergy trade data statistics.

8.2.2 Transparency issues in the international used cooking oil market

The concerns about the passing of un-used oils as waste and market disturbance in the UCO market are briefly mentioned in the introduction. These can be traced back to an important issue, the lack of verification measures for wastes (C. Goh et al., 2013). While it is difficult to trace the origins of UCO it is an important aspect in the stipulation of whether UCO based HVO meets sustainability criteria or not. Therefore it is suggested to impose a certification scheme to determine the origin, contents and other details for the entire UCO based biodiesel production chain. Note that there are waste certification schemes available such as the ISCC waste and residue certification, but the scope of this scheme is biofuels in the EU and only certifies UCO as a feedstock. Meaning that if producers use different feedstocks alongside UCO the greenhouse gas footprint of the resulting biofuel is uncertain. Furthermore, it can be questioned if the sustainability can be guaranteed if the UCO is only traced back to the collection point, which is done in the ISCC. Fraud and swindle can occur much earlier in the UCO lifecycle. Therefore the scope of such certification schemes needs to be expanded to international standards, include biofuel production as a process and extend the pathway certification to much earlier in the UCO lifecycle.

When reporting the origin of feedstocks for biodiesel production it is not always clear where the source of the imported UCO lies. In Figure 11, for example, a large part of countries in the 'other' section is undisclosed. The origin of UCO imports for biofuel on the Dutch market is therefore untraceable. Whether this is a deliberate choice or not, is open for discussion.

In short, there is lack of transparency and insight into the actual UCO supply chains, including sources, collection, transportation, prices and composition.

8.3 Suggestions for further research

During this study only one type of biodiesel has been researched, to keep the study comprehensive. It is advisable to include a broader range of different fuel types during follow-up research, to give a complete picture of the current transportation fuel greenhouse gas footprint. The LCA as conducted in this research is an attributional LCA. As discussed in section 3.2, this method has its shortcoming and conducting a consequential LCA would give a better picture of the induced policy effects and outcomes. However, it is difficult to do so using the limited data that is available and careful continuation is advised. An interesting extension of this research would be to review the developments of the UCO/HVO market in the future, while some expectations are given in this study these are by no means certain. For example it would be interesting to look back on the effects of the implementation of the proposed RED II and which role UCO plays in the European transport market by then.

9 Conclusion

The greenhouse gas footprint of used cooking oil-based biodiesel is larger compared to the EU Renewable Energy Directive default values in cases where the feedstock is imported from overseas origins via polluting transportation modes. The transportation of UCO as a feedstock is an important aspect to consider when evaluating the GHG footprint of HVO. The range of GHG emissions associated with UCO-transportation for hydrotreated vegetable oil production found in this thesis is 2.19 to 14.5 gCO_2-eq/MJ NExBTL. Emissions for UCO transportation as adhered to in the RED are substantially lower at .25 gCO_2-eq/MJ. When reflecting upon academic literature it shows that the methodology used in the RED to assign default values to biofuel GHG emissions has its shortcomings. These undermine its practical application in estimating actual values. This thesis proposes a more realistic GHG footprint of biodiesel based on used cooking oils. The case study shows that it is difficult to conclude whether used cooking oil-based biodiesel can meet European sustainability criteria. A range of GHG emissions for UCO biodiesel production has been assigned to support this statement. This range of variances finds its limits above and below the RED default values for GHG emissions and associated GHG reduction potentials of UCO based biodiesel. Furthermore, when the denotation of used cooking oil shifts from being a waste product to a desired biofuel feedstock, the gap between proposed default values and true greenhouse gas emissions widens. This is shown by an economic allocation of the environmental burden carried by virgin oils to the resulting UCO. This questionable transition, as a result of an increasing policy pressure, has drastic consequences which seem less than desirable in an everlasting pursuit towards a sustainable society. Therefore, a system which guarantees the sustainability of biodiesel based on used cooking oil should be introduced when further policy incentives for waste-based biofuels are being considered. Stricter chain of custody requirements might be needed to ensure transparency and accountability over the whole supply chain, as the sustainability of used cooking oil-based biofuels depends on these two aspects.

In conclusion, the greenhouse gas footprint of used cooking oil based biodiesel depends on two key factors which are the overseas feedstock transportation procedure and the distance of the feedstock origin to the Netherlands. The sustainability of used cooking oil as a feedstock can only be assured if these two factors are taken into consideration within a transparent used cooking oil supply chain.

Literature

Aatola, H., Larmi, M., Sarjovaara, T., & Mikkonen, S. (2008). Hydrotreated Vegetable Oil (HVO) as a Renewable Diesel Fuel: Trade-off between NOx, Particulate Emission, and Fuel Consumption of a Heavy Duty Engine. SAE Technical Paper 2008-01-2500. *SAE Technical Papers*, (724), 12. https://doi.org/10.4271/2008-01-2500

Agri-Food and Agriculture Canada. (2014). Consumer Trends Cooking Oils in China, (January).

Appleyard, D. (2014). Green replacement fuels taking flight. *Renewable Energy Focus*, *15*(1), 38–39.

Banerjee, A., & Chakraborty, R. (2009). Parametric sensitivity in transesterification of waste cooking oil for biodiesel production-A review. *Resources, Conservation and Recycling*, *53*(9), 490–497. https://doi.org/10.1016/j.resconrec.2009.04.003

BioDieNet. (2009). *EL LIBRO, The Handbook for Local Initiatives for Biodiesel from Recycled Oil*. Retrieved from http://www.sec.bg/userfiles/file/BioDieNet/EL LIBRO.pdf

Boutesteijn, C., Drabik, D., & Venus, T. J. (2016). The interaction between EU biofuel policy and first- and second-generation biodiesel production. *Industrial Crops and Products*. https://doi.org/10.1016/j.indcrop.2016.09.067

Brandão, M., Martin, M., Cowie, A., Hamelin, L., & Zamagni, A. (2017). Consequential Life Cycle Assessment: What, How, and Why? In *Encyclopedia of Sustainable Technologies* (pp. 277–284). Elsevier.

Brander, M., Tipper, R., Hutchison, C., & Davis, G. (2008). Consequential and attributional approaches to LCA: a Guide to policy makers with specific reference to greenhouse gas LCA of biofuels. *Econometrica Press*, *44*(April), 1–14. Retrieved from http://onlinelibrary.wiley.com/doi/10.1002/cbdv.200490137/abstract%5Cnhttp://www.globalbioenergy.org/uploads/media/0804_Ecometrica_-_Consequential_and_attributional_approaches_to_LCA.pdf%5Cnhttp://d3u3pjcknor73l.cloudfront.net/assets/media/pdf/approachest

Brorsen, W. (2015). Projections of US production of Biodiesel Feedstock, (July 2015), 1–13.

Caputo, A. (2014). Trends in European Bioenergy Law : Problems , Perspectives and Risks, 1–2.

CBS. (2017). Goederensoorten naar land; natuur, voeding en tabak. Retrieved August 23, 2017, from https://opendata.cbs.nl/statline/#/CBS/nl/dataset/81267ned/table?dl=532F

CCWG. (2017). *Collaborative Progress; Clean Cargo Working Group (CCWG) 2016 Progress Report*.

Cherubini, F., Bird, N. D., Cowie, A., Jungmeier, G., Schlamadinger, B., & Woess-Gallasch, S. (2009). Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. *Resources, Conservation and Recycling*, *53*(8), 434–447. https://doi.org/10.1016/j.resconrec.2009.03.013

Christiansen, M., Fagerholt, K., Nygreen, B., & Ronen, D. (2013). Ship routing and scheduling in the new millennium. *European Journal of Operational Research*, *228*(3), 467–483. https://doi.org/10.1016/j.ejor.2012.12.002

Davis, S. C., Anderson-Teixeira, K. J., & DeLucia, E. H. (2009). Life-cycle analysis and the ecology of biofuels. *Trends in Plant Science*, *14*(3), 140–146. https://doi.org/10.1016/j.tplants.2008.12.006

Demirbas, A. (2007). Importance of biodiesel as transportation fuel. *Energy Policy*, *35*(9), 4661–4670. https://doi.org/10.1016/j.enpol.2007.04.003

Dunn, R. O. (2010). *The Biodiesel Handbook*. *The Biodiesel Handbook*. https://doi.org/10.1016/B978-1-893997-62-2.50015-2

Dutch Emissions Authority. (2012). Naleving jaarverplichting 2011 hernieuwbare energie vervoer en verplichting brandstoffen luchtverontreiniging.

Dutch Emissions Authority. (2013). Naleving jaarverplichting 2012 hernieuwbare energie vervoer en verplichting brandstoffen luchtverontreiniging Samenvatting.

Dutch Emissions Authority. (2014). *Rapportage hernieuwbare energie 2013*.

Dutch Emissions Authority. (2015). Rapportage hernieuwbare energie 2014.

Dutch Emissions Authority. (2016). *Rapportage Energie voor Vervoer in Nederland 2015*.

Dutch Emissions Authority. (2017). Rapportage Energie voor Vervoer in Nederland 2016, 1–55.

Edwards, R., Hass, H., Larivé, J.-F., Lonza, L., Mass, H., Rickeard, D., … Weindorf, W. (2013). Well-to-Wheels analysis of future automotive fuels and powertrains in the European context WELL-TO-TANK (WTT) Report. Version 4. *Joint Research Center of the EU (JRC): Ispra, Italy*, 1–133. https://doi.org/10.2790/95629

Edwards, R., Padella, M., Giuntoli, J., Koeble, R., O'Connell, A., Bulgheroni, C., & Marelli, L. (2016). *Biofuels pathways. Input values and GHG emissions.* https://doi.org/10.2790/658143

Elburg Global. (2017). Packaging Bulk refined oils. Retrieved August 16, 2017, from http://www.elburgglobal.nl/packaging/bulk/

Enweremadu, C. C., & Mbarawa, M. M. (2009). Technical aspects of production and analysis of biodiesel from used cooking oil-A review. *Renewable and Sustainable Energy Reviews*, *13*(9), 2205–2224. https://doi.org/10.1016/j.rser.2009.06.007

European Commission. (2012). Proposal for a Directive of the european parliament and of the council: amending Directive 2009/28/EC, *288*, 23. https://doi.org/10.1017/CBO9781107415324.004

European Commission. (2015). *The impact of biofuels on transport and the environment, and their connection with agricultural development in Europe*.

European Commission - Joint Research Centre - Institute for Environment and Sustainability. (2010). *International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance*. *Publications Office of the European Union* (1st ed.). https://doi.org/10.2788/38479

European Commission (EC). (2003). COMMISSION DIRECTIVE 2003/100/EC of 31 October 2003 amending Annex I to Directive 2002/32/EC of the European Parliament and of the Council on undesirable substances in animal feed. *Official Journal of the European Union*, *46*(L285), 33–37. https://doi.org/http://eur-lex.europa.eu/pri/en/oj/dat/2003/l_285/l_28520031101en00330037.pdf

European Parliament. (2009). Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009. *Official Journal of the European Union*, *140*(16), 16–62. https://doi.org/10.3000/17252555.L_2009.140.eng

European Union. (2009). Directive 2009/30/EC of the European Parliament and of the Council. *Official Journal of the European Union*, (April), L140/88-L140/113. https://doi.org/10.3000/17252555.L_2009.140.eng

European Union. (2015). Directive 2015/1513 of the European Parliament and of the Council of 9 September 2015 amending Directive 98/70/EC relating to the quality of petrol and diesel fuels and amending Directive 2009/28/EC on the promotion of the use of energy from renewabl. *Official Journal of the European Union*.

Flach, B., Lieberz, S., Rondon, M., Williams, B., & Wilson, C. (2016). *EU-28 Biofuels Annual EU Biofuels Annual 2016*. *Renewable Energy*.

Flux Energie. (2017). Nederland is al een grote producent van waterstof. Retrieved September 4, 2017, from https://www.fluxenergie.nl/nederland-is-al-grote-producent-waterstof/

Goh, C., Junginger, M., Joudrey, J., Chum, H., Pelkmans, L., Smith, C., … Goovaerts, L. (2013). Strategic Inter-Task Study : Monitoring Sustainability Certification of Bioenergy Task 3 : Impacts of sustainability certification on bioenergy markets and trade, 1–61.

Goh, C. S., Junginger, M., Mai-Moulin, T., & Junginger, M. (2016). *Sustainable biomass and bioenergy in the Netherlands: Report 2015*. Utrecht. Retrieved from http://english.rvo.nl/sites/default/files/2013/12/Sustainable biomass and bioenergy in the Netherlands - Report 2013.pdf

Greenea. (2014). *Pricing of UCO , Animal Fat and Waste-Based Biofuel : an Achilles Heel of this Market*.

GREENEA. (2015). Is HVO the Holy Grail of the world biodiesel market?, (September), 1–3.

Grunau, P. (2016). *A short introduction for loading, unloading and stowage of solid bulk cargoes including draught survey*.

Guinée, J. B., Heijungs, R., & Huppes, G. (2004). Economic allocation: Examples and derived decision tree. *The International Journal of Life Cycle Assessment*, *9*(1), 23–33. https://doi.org/10.1007/BF02978533

Ha, Q. (2016). Corporate Social Responsibility in Oil and Gas Industry in Finland : Performance of Neste Oil Corporation.

Hamelinck, C. N., & Faaij, A. P. C. (2002). Future prospects for production of methanol and hydrogen from biomass. *Journal of Power Sources*, *111*(1), 1–22. https://doi.org/10.1016/S0378-7753(02)00220-3

Hamelinck, C., & Zabeti, M. (2016). Low carbon biofuels for the UK. Retrieved from http://epure.org/media/1418/ecofys-2016-low-carbon-biofuels-for-the-uk.pdf

Hillairet, F., Allemandou, V., & Golab, K. (2016). *Waste-based feedstock and biofuels market in Europe*. Retrieved from https://www.greenea.com/wp-content/uploads/2016/11/Argus-2016.pdf

Hillairet, F., Allemandou, V., Golab, K., & Castaing, G. (2017). *Market Watch*. Retrieved from https://www.greenea.com/wp-content/uploads/2017/02/Greenea-Market-Watch-January-2017.pdf

Hillairet, F., Allemandou, V., Golab, K., & Limouzy, A. (2018). Looking back at the waste-based biodiesel market in 2017.

Hoefnagels, R., Resch, G., Junginger, M., & Faaij, A. (2014). International and domestic uses of solid biofuels under different renewable energy support scenarios in the European Union. *Applied Energy*, *131*(November), 139–157. https://doi.org/10.1016/j.apenergy.2014.05.065

Honig, V., Linhart, Z., & Orsak, M. (2015). Use of Blend of Hydrotreated Vegetable Oil With Biobutanol, 324–329.

Huo, H., Yao, Z., Zhang, Y., Shen, X., Zhang, Q., & He, K. (2012). On-board measurements of emissions from diesel trucks in five cities in China. *Atmospheric Environment*, *54*, 159–167. https://doi.org/10.1016/j.atmosenv.2012.01.068

International Council on Clean Transportation. (2017). January 2017 the European Commission’s Renewable Energy Proposal for 2030, (January).

International Sustainability & Carbon Certification. (2016). *ISCC 201-1 Waste and Residues*. Retrieved from https://www.iscc-system.org/wp-content/uploads/2017/02/ISCC_201-1_Waste_and_Residues_3.0.pdf

Johnson, E. (2017). A carbon footprint of HVO biopropane. *Biofuels, Bioproducts and Biorefining*.

Kendall, A., & Yuan, J. (2013). Comparing life cycle assessments of different biofuel options. *Current Opinion in Chemical Biology*, *17*(3), 439–443. https://doi.org/10.1016/j.cbpa.2013.02.020

Kirchain, R. (2006). Session 3 : Inventory Analysis LCA.

Knothe, G. (2010). Biodiesel and renewable diesel: A comparison. *Progress in Energy and Combustion Science*, *36*(3), 364–373. https://doi.org/10.1016/j.pecs.2009.11.004

Koppejan, J., Elbersen, W., Meeusen, M., & Bindraban, P. (2009). Beschikbaarheid van Nederlandse biomassa voor elektriciteit en warmte in 2020, (November), 99. Retrieved from http://edepot.wur.nl/51989

Laakkonen, M., Myllyoja, J., Toukoniitty, B., Hujanen, M., Saastamoinen, A., & Toivo, A. (2013). Process for manufacture of liquid fuel componenst from renewable sources.

Lamers, P., Hoefnagels, R., Junginger, M., Hamelinck, C., & Faaij, A. (2015). Global solid biomass trade for energy by 2020: An assessment of potential import streams and supply costs to North-West Europe under different sustainability constraints. *GCB Bioenergy*, *7*(4), 618–634. https://doi.org/10.1111/gcbb.12162

Lane, J. (2016). *Could renewable diesel's boom be cut short by feedstock access and availability?* Retrieved from http://www.biofuelsdigest.com/bdigest/2016/04/10/could-renewable-diesels-boom-be-cut-short-by-feedstock-access-and-availability/

Lehmus, P., VP R&D, & Neste Oil. (2014). Large scale chemical conversion of oils and residues in Rotterdam. *European Biofuels Technology Platform - 6th Stakeholder Plenary Meeting*, 26. Retrieved from http://www.biofuelstp.eu/spm6/docs/petri-lehmus.pdf

Liang, S., Liu, Z., Xu, M., & Zhang, T. (2013). Waste oil derived biofuels in China bring brightness for global GHG mitigation. *Bioresource Technology*, *131*, 139–145. https://doi.org/10.1016/j.biortech.2012.12.008

Lloyd's list. (2016). Vessel Utilisation in 2015 – Have YOU got the Load Factor?, *i*, 2014–2016.

Marques, P. A. (2015). Ref. Ares(2015)1741712 - 24/04/2015. https://doi.org/10.2903/j.efsa.2015.4031

Menzies, G. F., Banfill, P. F. G., & Turan, S. (2007). Life-cycle assessment and embodied energy: a review. *Proceedings of the ICE - Construction Materials*, *160*(4), 135–143. https://doi.org/10.1680/coma.2007.160.4.135

Mijnheer, D. (2016). Nederland, de vetput van de wereld. *Follow the Money*. Retrieved from https://www.ftm.nl/artikelen/nederland-de-vetput-van-de-wereld

Morais, S., Mata, T. M., Martins, A. A., Pinto, G. A., & Costa, C. A. V. (2010). Simulation and life cycle assessment of process design alternatives for biodiesel production from waste vegetable oils. *Journal of Cleaner Production*, *18*(13), 1251–1259. https://doi.org/10.1016/j.jclepro.2010.04.014

Mortimer, N. D., Hatto, C., & Mwabonje, O. (2015). *Review of the methodology contained in Annex V of the Renewable Energy Directive (2009/28/EC) and replicated in Annex IV of the Fuel Quality Directive: Deliverable 3 - Critical Comparison of Calculation Methodologies*. Sheffield UK.

Motta, J. (2016). Creating Renewable Energy from the Effective Management of Fats, Oils, and Grease (FOG), (June 2015).

MVO. (2014). *The Dutch Oils and Fats Industry; An International and Sustainable Chain*.

Nelson, B., & Searle, S. (2016). *Projected availability of fats, oils, and greases in the U.S. The International Council on Clean Transportation* (Vol. Working pa). Retrieved from http://www.theicct.org/sites/default/files/publications/Biodiesel

Availability_ICCT_20160707.pdf

Neste Oil Corporation. (2011). Neste Oil celebrates the grand opening of Europe’s largest renewable diesel refinery in Rotterdam. Retrieved from https://www.neste.com/en/neste-oil-celebrates-grand-opening-europes-largest-renewable-diesel-refinery-rotterdam

Nikander, S. (2008). Greenhouse gas and energy intensity of product chain: case transport biofuel. *Master of Science in*, 112. Retrieved from http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:GREENHOUSE+GAS+AND+ENERGY+INTENSITY+OF+PRODUCT+CHAIN+:+CASE+TRANSPORT+BIOFUEL#0

Nordic Tankers. (2015). *One Fleet Solution; Combining noon reports with automated data*.

NRA. (2017). FAQs. Retrieved August 9, 2017, from http://www.nationalrenderers.org/about/faqs/

Nylund, N. O., Erkkilä, K., Ahtiainen, M., Murtonen, T., Saikkonen, P., Amberla, A., & Aatola, H. (2011). *Optimized usage of NExBTL renewable diesel fuel OPTIBIO*. *VTT Tiedotteita - Valtion Teknillinen Tutkimuskeskus*.

Office of Energy Efficiency & Renewable Energy. (2016a). Hydrogen Production: Natural Gas Reforming. Retrieved September 4, 2017, from http://energy.gov/eere/fuelcells/hydrogen-production-natural-gas-reforming

Office of Energy Efficiency & Renewable Energy. (2016b). Hydrogen Production Processes. Retrieved January 1, 2017, from https://energy.gov/eere/fuelcells/hydrogen-production-processes

Ogden, J. M. (1999). Prospects for Building a Hydrogen Energy Infrastructure. *Annual Review of Energy and the Environment*, *24*(1), 227–279. https://doi.org/doi:10.1146/annurev.energy.24.1.227

Ou, X., Zhang, X., Chang, S., & Guo, Q. (2009). Energy consumption and GHG emissions of six biofuel pathways by LCA in China. *Applied Energy*, *86*(SUPPL. 1). https://doi.org/10.1016/j.apenergy.2009.04.045

Peters, D., Toop, G., van den Bos, A., & Spottle, M. (2013). Assessing the EC ILUC proposal Dutch National Impact Assessment. *ECOFYS Netherlands B.V.*

Plevin, R. J., Delucchi, M. A., & Creutzig, F. (2014). Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation Benefits Misleads Policy Makers. *Journal of Industrial Ecology*, *18*(1), 73–83. https://doi.org/10.1111/jiec.12074

Port of Rotterdam Authority. (2016). *Facts & Figures on the rotterdam Energy Port and Petrochemical Cluster*.

Proskurina, S., Junginger, M., Heinimö, J., & Vakkilainen, E. (2017). Global biomass trade for energy – Part 1: Statistical and methodological considerations. *Biofuels, Bioproducts and Biorefining*.

Psaraftis, H. N., & Kontovas, C. . (2009). Co2 Emissions Statistics for the World Commercial Fleet. *WMU Journal of Maritime Affairs*, *8*(1), 1–25.

Quek, A., & Balasubramanian, R. (2014). Life cycle assessment of energy and energy carriers from waste matter - A review. *Journal of Cleaner Production*, *79*, 18–31. https://doi.org/10.1016/j.jclepro.2014.05.082

Raman, S., & Mohr, A. (2014). Biofuels and the role of space in sustainable innovation journeys. *Journal of Cleaner Production*, *65*(January), 224–233. https://doi.org/10.1016/j.jclepro.2013.07.057

Rotie B.V. (2017). Frituurvet Ophalen. Retrieved November 1, 2017, from https://www.rotie.nl/nl/frituurvet-ophalen/

Sánchez, M. J. S., Bhattacharya, S., & Mareckova, K. (2006). Reporting Guidance and Tables. *2006*

IPCC Guidelines for National Greenhouse Gas Inventories. Volume 1: General Guidance and Reporting, 34. https://doi.org/10.1007/978-1-60761-476-0

Schueler, V., Weddige, U., Beringer, T., Gamba, L., & Lamers, P. (2013). Global biomass potentials under sustainability restrictions defined by the European Renewable Energy Directive 2009/28/EC. *GCB Bioenergy*, *5*(6), 652–663. https://doi.org/10.1111/gcbb.12036

sea-distances.org. (2018). Ports Distances. Retrieved from https://sea-distances.org/

SEI. (2003). *A Resource Study on Recovered Vegetable Oil and Animal Fats*.

SEPA. (2015). Guidance on using the European Catalogue (EWC) to code waste November 2015 Waste, (November).

SFMCA. (2016). *Safe use of fats and oils in stock feeds prepared for ruminants*.

Singh, S., Jain, S., Ps, V., Tiwari, A. K., Nouni, M. R., Pandey, J. K., & Goel, S. (2015). Hydrogen: A sustainable fuel for future of the transport sector. *Renewable and Sustainable Energy Reviews*, *51*(July), 623–633. https://doi.org/10.1016/j.rser.2015.06.040

Smith, T. W. P., Jalkanen, J. P., Anderson, B. A., Corbett, J. J., Faber, J., Hanayama, S., … Hoen, M., A. (2014). Third IMO Greenhouse Gas Study 2014. *International Maritime Organization (IMO)*, 327. https://doi.org/10.1007/s10584-013-0912-3

Soimakallio, S. (2014). Toward a more comprehensive greenhouse gas emissions assessment of biofuels: The case of forest-based fischer-tropsch diesel production in Finland. *Environmental Science and Technology*, *48*(5), 3031–3038. https://doi.org/10.1021/es405792j

Soimakallio, S., & Koponen, K. (2011). How to ensure greenhouse gas emission reductions by increasing the use of biofuels? - Suitability of the European Union sustainability criteria. *Biomass and Bioenergy*, *35*(8), 3504–3513. https://doi.org/10.1016/j.biombioe.2011.04.041

Spöttle, M., Alberici, S., Toop, G., Peters, D., Gamba, L., Ping, S., … Bellefleur, D. (2013). Low ILUC potential of wastes and residues for biofuels Low ILUC potential of wastes and residues for biofuels, 168.

Stopford, M. (2009). *Maritime Economics* (3rd ed.). Taylor & Francis.

Talens, L., Villalba, G., & Gabarrell, X. (2007). Exergy analysis applied to biodiesel production. *Resources, Conservation and Recycling*, *51*(2), 397–407. https://doi.org/10.1016/j.resconrec.2006.10.008

Talens Peiró, L., Lombardi, L., Villalba Méndez, G., & Gabarrell i Durany, X. (2010). Life cycle assessment (LCA) and exergetic life cycle assessment (ELCA) of the production of biodiesel from used cooking oil (UCO). *Energy*, *35*(2), 889–893. https://doi.org/10.1016/j.energy.2009.07.013

Technical University of Crete. (2013). Assessment of best practices in UCO processing and biodiesel distribution. *RECOIL The Power of Used Cooked Oil*, *3*, 37. Retrieved from https://www.google.co.za/interstitial?url=http://www.recoilproject.eu/index.php/en/publications/category/14-best-practices-in-processing-and-distribution%3Fdownload%3D42:european-and-national-technical-norms

The Port of Virginia. (2016). Hampton Roads Harbor, 2016 Trade Overview. Retrieved from http://www.portofvirginia.com/pdfs/about/POV2016_Trade_Overview.pdf

Thibault, A. F. (2015). *Clean Cargo Working Group Carbon Emissions Accounting Methodology*.

Thoenes, P. (2018). *Oilseeds, Oils & Meals; Monthly price and policy update*.

Valente, O. S., Pasa, V. M. D., Belchior, C. R. P., & Sodré, J. R. (2011). Physical-chemical properties of waste cooking oil biodiesel and castor oil biodiesel blends. *Fuel*, *90*(4), 1700–1702.

https://doi.org/10.1016/j.fuel.2010.10.045

Veidenheimer, K. Carbon Dioxide Emission in Maritime Container Transport and comparison of European deepwater ports: CO2 Calculation Approach, Analysis and CO2 (2014). Retrieved from https://books.google.com/books?hl=en&lr=&id=i3sNBAAAQBAJ&oi=fnd&pg=PA5&dq=%22 Port+Supply+Chain+Integration%22+and+%22Sustainability%22&ots=EmqQTnLZoh&sig=M WlqldF77_kbe4H-j81KrXwzdyg%5Cnhttps://books.google.com/books?hl=en&lr=&id=i3sNBAAAQBAJ&oi=fnd &pg=PA5

Vinyes, E., Oliver-Solà, J., Ugaya, C., Rieradevall, J., & Gasol, C. M. (2013). Application of LCSA to used cooking oil waste management. *International Journal of Life Cycle Assessment*, *18*(2), 445–455. https://doi.org/10.1007/s11367-012-0482-z

Vyas, A. P., Verma, J. L., & Subrahmanyam, N. (2010). A review on FAME production processes. *Fuel*, *89*(1), 1–9. https://doi.org/10.1016/j.fuel.2009.08.014

Wärtsilä. (2015). WSD80 1500 TEU Container Feeder.

Wärtsilä. (2016). W26 Guide.

Wicke, B., Dornburg, V., Faaij, A., & Junginger, M. (2007). A Greenhouse Gas Balance of Electricity Production from Co-Firing Palm Oil Product From Malaysia, 1–62.

Wicke, B., Dornburg, V., Junginger, M., & Faaij, A. (2008). Different palm oil production systems for energy purposes and their greenhouse gas implications. *Biomass and Bioenergy*, *32*(12), 1322–1337. https://doi.org/10.1016/j.biombioe.2008.04.001

Wingd. (2018). RT-flex50 specifications.

Yano, J., Aoki, T., Nakamura, K., Yamada, K., & Sakai, S. (2015). Life cycle assessment of hydrogenated biodiesel production from waste cooking oil using the catalytic cracking and hydrogenation method. *Waste Management*, *38*(1), 409–423. Retrieved from http://www.embase.com/search/results?subaction=viewrecord&from=export&id=L602077753% 5Cnhttp://dx.doi.org/10.1016/j.wasman.2015.01.014%5Cnhttp://sfx.library.uu.nl/utrecht?sid=E MBASE&issn=18792456&id=doi:10.1016/j.wasman.2015.01.014&atitle=Life+cycle+assess

Zhang, H., Wang, Q., & Mortimer, S. R. (2012). Waste cooking oil as an energy resource: Review of Chinese policies. *Renewable and Sustainable Energy Reviews*, *16*.

Appendix I

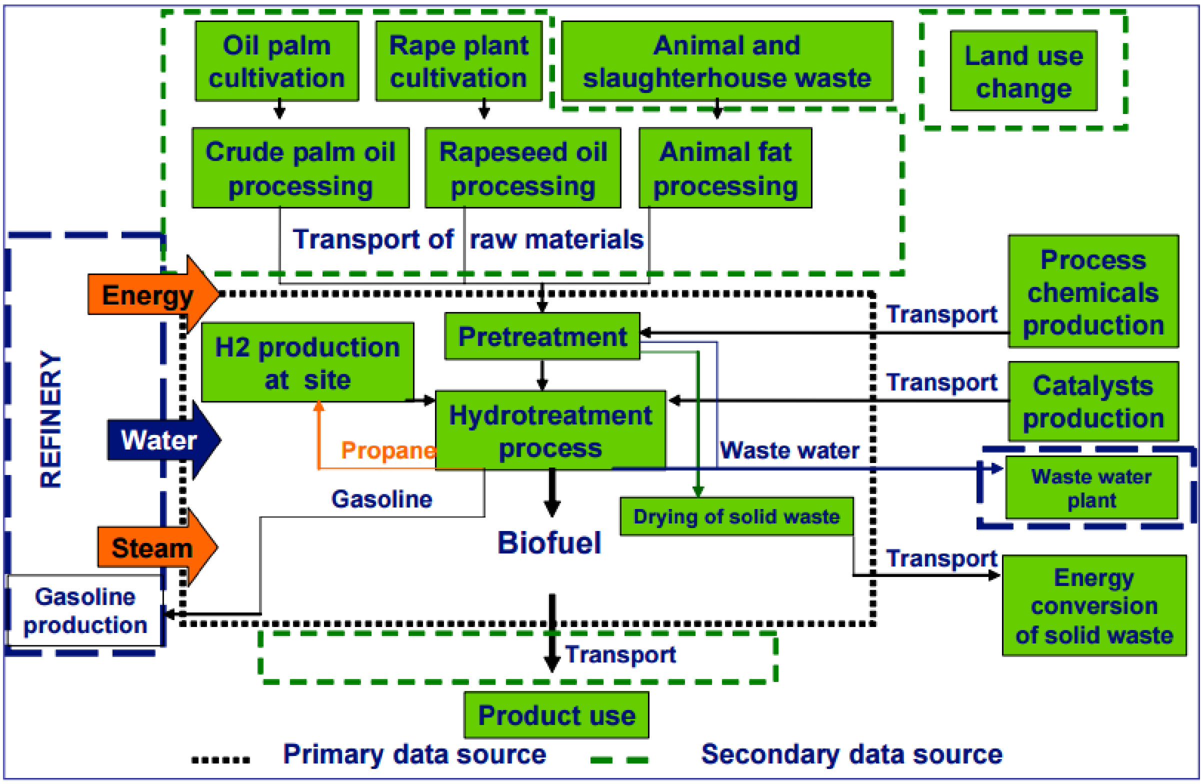


Figure 21 HVO production using the NExBTL procees (source: (Nikander, 2008)*)*

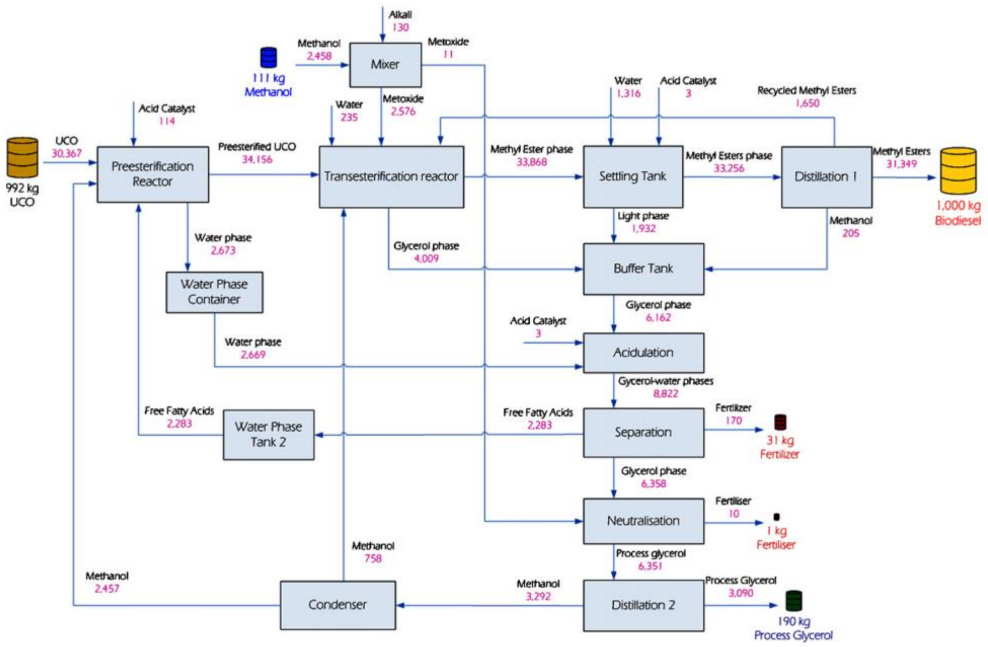


Figure 22 FAME production (Source: (Talens, Villalba, & Gabarrell, 2007))

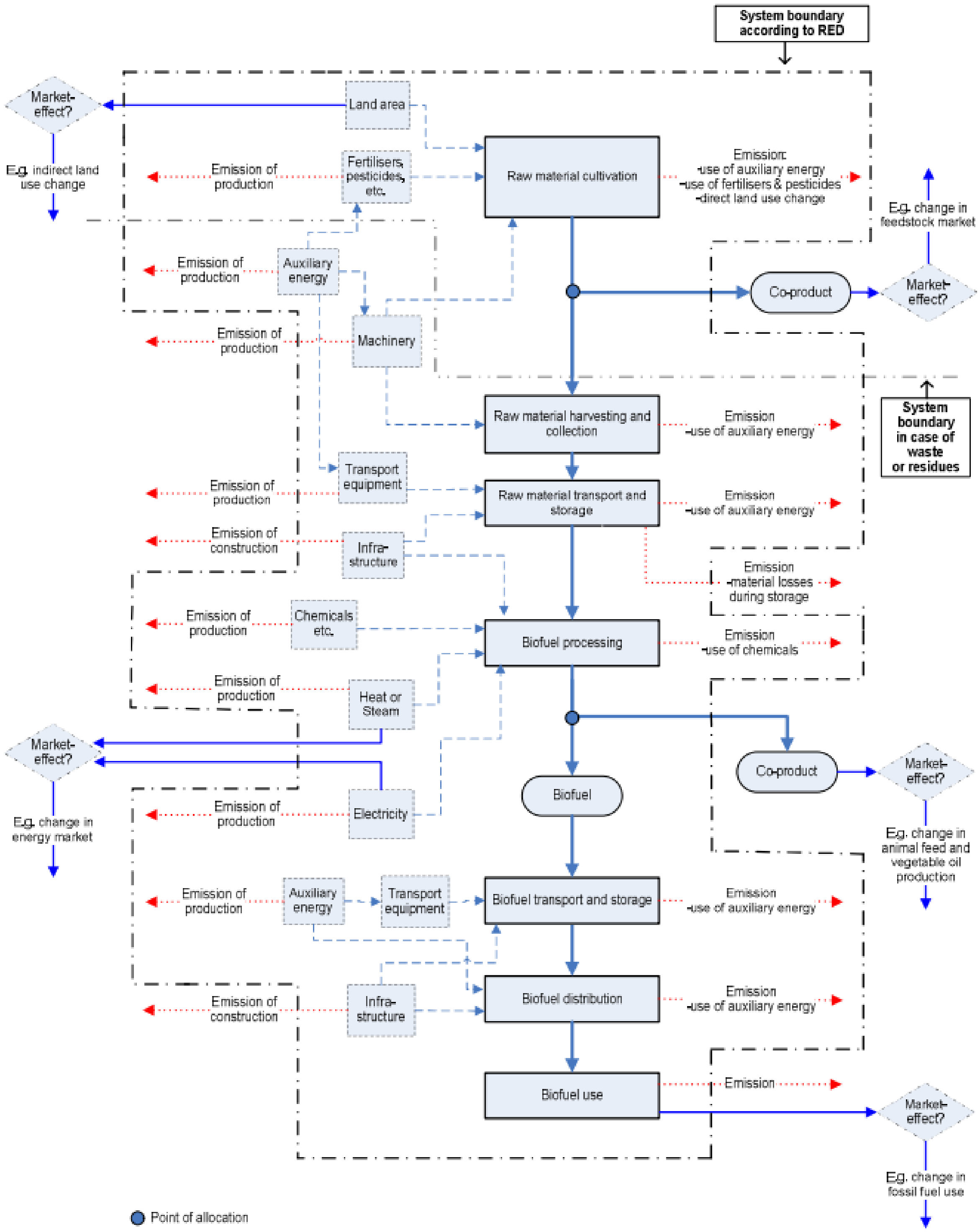


Figure 23 Generic illustration of the spatial system boundary according to the RED methodology with possible impacts outside the system (Source: (Soimakallio & Koponen, 2011)

Appendix II

Interview Lars Huizenga & Bram van Santen 23/10/2017

Asia, China and Indonesia are the main countries from where the UCO is originated. In the summer months around 80 percent is imported in the winter months around 60-70 percent. The rest comes from the Netherlands, Belgium, France, Germany, United Kingdom. Approximately from a 350 km radius around the Netherlands.

In the Netherlands the UCO is transported by trucks with a loading capacity of 4 tons. The distance the trucks cover is not exactly known, but the maximum would be 200 km. But there are also rides in neighbourhoods. It concerns EURO5 / EURO6 trucks. From the five depots of Rotie the UCO is transported to Amsterdam by larger trucks.

The transport from Asia occurs currently for 70 percent by flexi tank (FT), but increasingly bulk shipments from Asia. 4 to 10 thousand tonnes at a time. This is however going mainly to Spain or to the UK. Rotie does not really purchase bulk shipments.

The UCO that is received is already refined, there is already a filtration and settling and heating step. But this does not require much energy. Due to the warm nature of the countries of origin.

It is believed that the lorries have more impact on GHG's than the tankers, but Rotie sets the values by using RED's default values.

At the Biodiesel Amsterdam (BA) facility the main input is UCO, and additional animal fats. In 2009 the rate was 100 percent UCO, in 2010 after the double-counting measure Rotie became more and more an international market player. In 2012, UK, France and Germany joined, creating an increasingly European market for 2nd generation biodiesel. Then a switch to an increasing percentage of animal fats has been made, because higher FFAs can be processed here. All animal fats are from Europe, cat. 1 and 2 may not be imported from elsewhere. UCO is also used in the oleo chemistry for paint, cream and plastic for example. Largest part goes to the biodiesel market. An estimation on the upper side of 90-95 percent. It is also burned, for example, if the quality is poor. Poor quality is polluted and poorly filtered UCO, mainly a problem if the origin is from a warmer country. The price of UCO has not risen in recent years. Due to the falling oil price, the price of UCO also decreases. Biodiesel price is mainly determined by palm and rapeseed oil. From Q2 to Q4 the price has fallen € 150, -. At this time, the UCO price (per tonne red.) Is € 800, -. In the past, it was also sold for € 1000, -. The Netherlands is one of the main ports for UCO. Many oil traders are in Rotterdam, so it's a hub. The containers go straight from the ship to the truck to be transported further. All UCO arriving at Rotie first passes through the Rotiehal (refinery), will be sampled, unloaded, heated, settled and checked whether the quality is on-spec. Then the UCO goes into a delivery tank and subsequently goes to or BA or to another destination. Everything is on a field, so it will leave the site as UCO or Biodiesel. Glycerin, Potassium sulphate, filtercakes and heating oil by-products are all sold. Potassium sulphate goes to Germany, and glycerine to a biofermenter in Denmark and the UK. The rest is stored on site.

The collection chain in China is similar to here, the difference lies in collecting in big cities. A lot of people with a lot of fat live close together. Sometimes it is even picked up with a hand truck to be taken to a larger warehouse, loaded onto a truck and transported via a river to a port. In the Netherlands: Rotie has almost 40,000 unique retrieval locations, after which it goes to one of the five depots where the UCO goes to Amsterdam. Or it goes straight to Amsterdam (this location has the largest collection area). In China, it's the same, pick up> depot> holding tank> flexitanker> ship.

In the Netherlands, around 100-150 kt is being collected. But the processing capacity in the Netherlands is much higher. The Netherlands has become a kind of UCO hub. This has always been this way: Originally, lots of margarine was produced and many animal fats were available. Far before the biodiesel industry. Fats were/are incorporated into candles, oleo chemistry, gelatine, etc. Cat 3 is still used for this, cat 1 and 2 no longer after Creutzfeldt-Jakob. The Netherlands still has major renderers. Slowly, the UK starts to play a bigger role. Similarly in Spain, this also begins to develop. Import of foreign biodiesel based on UCO is still happening, everything goes through ARA (Amsterdam-Rotterdam-Antwerpen red.). In particular, biodiesel blending takes place in Rotterdam.

It is difficult to predict whether the trend continues (shift from flexi tanks to bulk). If the price drops, it pays off less to do bulk, flexi tankers are cheaper in transport. If the price rises, larger volumes will got to the competitors.

Biodiesel Amsterdam only allocates GHG's on heating oil (BHO), and does this on percentage output. 5.4% BHO at 100% output. 75% -80% of Rotie's UCO goes to Germany. Neste is estimated to use about 70% palmoil, 25% tallow and 5% UCO.

The UCO chain is becoming more and more professional, there is increasing chain responsibility. There is growing insight and transparency in the market. Also in the short term.

Appendix III

Countries of origin of UCO for biofuels supplied to the Dutch transport market						
	2011	2012	2013	2014	2015	2016
Australia			0.03%		0.10%	
Austria			1.40%			
Belgium	0.98%	1.00%	6.20%	3.00%	4.10%	3.60%
Canada			1.70%	1.00%		
Chili				3.70%		
China						6.50%
Czech Republic						0.20%
Denmark				0.10%	0.90%	
France	0.43%	3.10%	3.20%	0.90%	1.70%	3.60%
Germany	29.63%	1.60%	17.30%	10.10%	10.60%	9.40%
Indonesia		0.10%	0.20%	3.70%	5.10%	
Iraq			2.00%			
Ireland		0.10%	0.20%			
Japan		2.70%	4.50%	4.10%		3.00%
Malaysia			7.30%	8.20%		
Netherlands	67.03%	40.80%	18.70%	16.80%	18.50%	12.00%
Peru				1.00%		
Poland		0.10%	0.50%	1.40%	0.80%	0.70%
Romania					0.10%	2.20%
Russia				0.40%		
Saudi-Arabia					3.90%	
South-Korea			19.90%	10.80%	6.70%	
Spain		24.00%	2.20%	2.30%	6.10%	10.20%
Switzerland				4.70%	2.30%	
Taiwan						8.80%
Ukraine				0.20%		
United Arab Emirates				3.50%	3.30%	
United Kingdom		0.50%	5.10%	7.10%	5.50%	3.70%
United States	0.93%	24.70%	5.50%	3.70%	15.90%	17.20%
Other	1.40%	1.30%	3.70%	13.30%	14.40%	18.90%

Table 19 Countries of origin of UCO for biofuels supplied to the Dutch transport market

Total	32376.62	mg CO2-eq
Carbon dioxide	27970.62	mg CO2-eq
Carbon dioxide, fossil	3549.43	mg CO2-eq
Carbon dioxide, land transformation	2.06	mg CO2-eq
Chloroform	0.00	mg CO2-eq
Dinitrogen monoxide	24.30	mg CO2-eq
Ethane, 1,1,1,2-tetrafluoro-, HFC-134a	0.00	mg CO2-eq
Ethane, 1,1,1-trichloro-, HCFC-140	0.00	mg CO2-eq
Ethane, 1,1,2-trichloro-1,2,2-trifluoro-, CFC-113	1.74	mg CO2-eq
Ethane, 1,1-difluoro-, HFC-152a	0.05	mg CO2-eq
Ethane, 1,2-dichloro-	0.00	mg CO2-eq
Ethane, 1,2-dichloro-1,1,2,2-tetrafluoro-, CFC-114	0.38	mg CO2-eq
Ethane, 2-chloro-1,1,1,2-tetrafluoro-, HCFC-124	0.00	mg CO2-eq
Ethane, hexafluoro-, HFC-116	0.10	mg CO2-eq
Hydrocarbons, chlorinated	0.00	mg CO2-eq
Methane	4.56	mg CO2-eq
Methane, biogenic	4.13	mg CO2-eq
Methane, bromo-, Halon 1001	0.00	mg CO2-eq
Methane, bromochlorodifluoro-, Halon 1211	0.03	mg CO2-eq
Methane, bromotrifluoro-, Halon 1301	0.09	mg CO2-eq
Methane, chlorodifluoro-, HCFC-22	0.38	mg CO2-eq
Methane, chlorotrifluoro-, CFC-13	0.00	mg CO2-eq
Methane, dichloro-, HCC-30	0.00	mg CO2-eq
Methane, dichlorodifluoro-, CFC-12	0.11	mg CO2-eq
Methane, dichlorofluoro-, HCFC-21	0.00	mg CO2-eq
Methane, fossil	810.74	mg CO2-eq
Methane, land transformation	0.01	mg CO2-eq
Methane, monochloro-, R-40	0.00	mg CO2-eq
Methane, tetrachloro-, CFC-10	0.96	mg CO2-eq
Methane, tetrafluoro-, CFC-14	0.65	mg CO2-eq
Methane, trichlorofluoro-, CFC-11	0.00	mg CO2-eq
Methane, trifluoro-, HFC-23	0.10	mg CO2-eq
Nitrogen fluoride	0.00	mg CO2-eq
Sulfur hexafluoride	6.18	mg CO2-eq

Table 20 Inventory of GHG emissions for the standard pathway of UCO NExBTL 1 MJ

Appendix IV

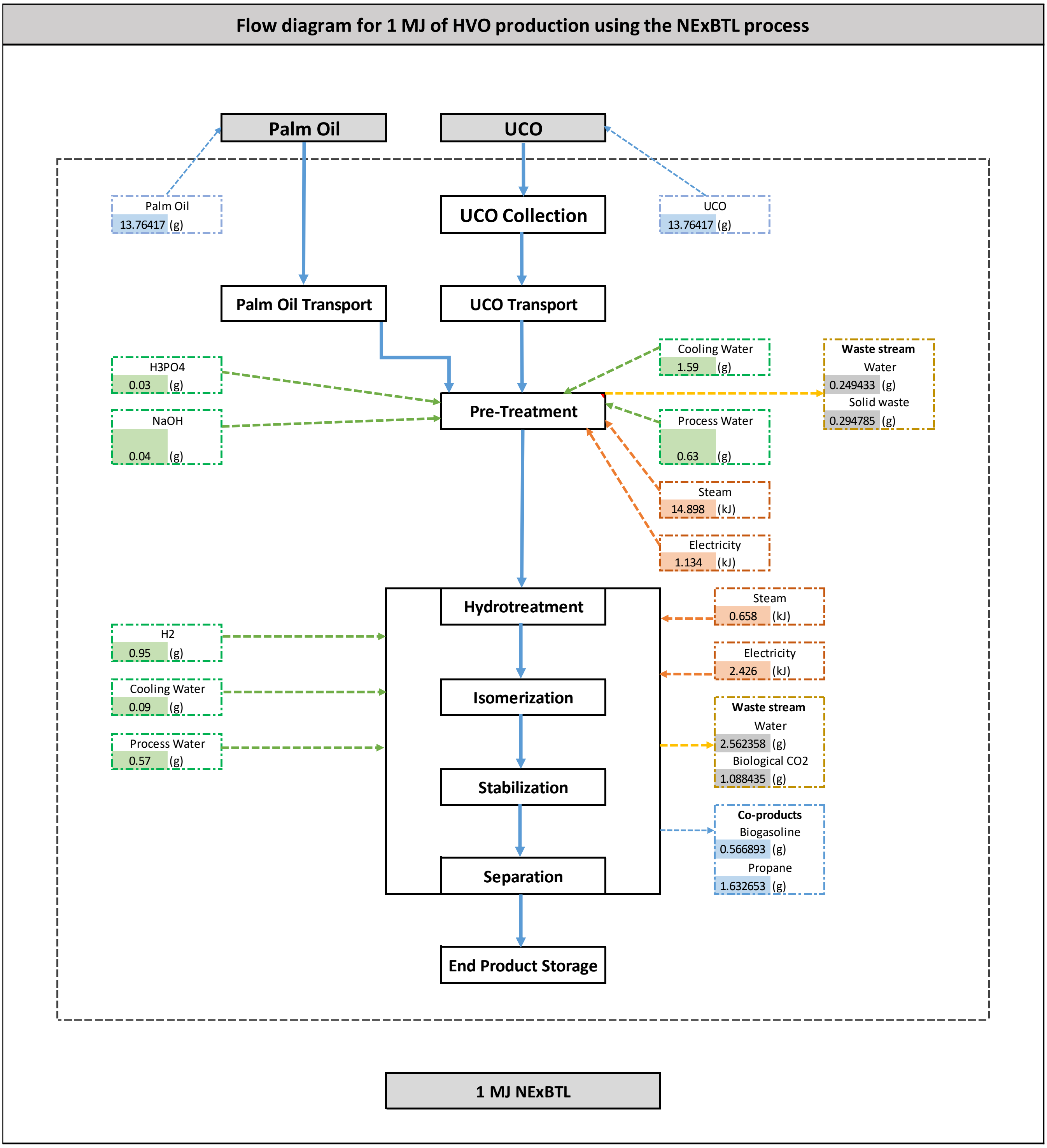


Figure 24 Flow diagram to produce 1 MJ of NExBTL