Perspective

Biofpr Biofuels Bioproducts & Biorefining

Review of life-cycle greenhouse-gas emissions assessments of hydroprocessed renewable fuel (HEFA) from oilseeds

Debrah Zemanek, Pascale Champagne, Department of Civil Engineering, Queen's University, Kingston, ON, Canada
Warren Mabee, Department of Geography and Planning, Queen's University, Kingston, ON, Canada

Received October 24 2019; Revised May 12 2020; Accepted May 19 2020;
View online June 25, 2020 at Wiley Online Library (wileyonlinelibrary.com);
DOI: 10.1002/bbb.2125; *Biofuels, Bioprod. Bioref*. 14:935–949 (2020)

Abstract: Renewable fuel from the hydroprocessed esters and fatty acids (HEFA) pathway represents a promising short-term option for reducing fossil fuel use in transportation. However, some life-cycle assessments (LCAs) have shown that HEFA diesel and jet fuel may have higher life-cycle greenhouse gas (GHG) emissions than the fossil fuels they replace. Many of these studies examined HEFA fuel derived from oilseed feedstocks. Here, results and methodology from 20 LCAs of HEFA fuel from oilseeds are reviewed in an effort to determine the sources of variability in the reported life-cycle GHG emissions of HEFA fuels. Although there was a 61–63% reduction in median life cycle GHG emissions of HEFA biojet and renewable diesel compared to conventional petroleum fuels, this review highlights the importance of standardized methodologies for life-cycle assessment (e.g., CORSIA, RSB) and indicates the need to prevent the conversion of forest land for biofuel production, as well as the potential opportunity for alternative oilseeds such as camelina and carinata as feedstocks to produce HEFA fuels with lower life-cycle GHG emissions. © 2020 Society of Chemical Industry and John Wiley & Sons, Ltd

Key words: life cycle assessment; LCA; hydroprocessed esters and fatty acids; HEFA; land-use change; biofuel; biojet; drop-in fuel

Introduction

In the past decade, there has been an increasing number of life-cycle assessments (LCAs) that focus on the life-cycle greenhouse gas (GHG) emissions of drop-in renewable fuels. Drop-in renewable fuels are alternatives to petroleum fuel that can be used in an existing engine without modification. Life-cycle assessments can be used to quantify the emissions intensity of the production and use of a drop-in fuel, in grams of carbon dioxide equivalent emissions (g CO_2eq) per megajoule of biofuel delivered, to provide a comparison to that of a petroleum fuel over an equivalent life cycle.

Hydroprocessed esters and fatty acids (HEFA) are one of the most promising short-term pathways for drop-in fuel production.[1] However, uncertainty in life-cycle scope and

Correspondence to: Warren Mabee, Department of Geography and Planning, Queen's University, Mackintosh-Corry Hall, 68 University Avenue K7L 3N6 Kingston, ON, Canada. E-mail: warren.mabee@queensu.ca

© 2020 Society of Chemical Industry and John Wiley & Sons, Ltd

methodology and in the values of life-cycle parameters has led to a large variability in the reported life-cycle GHG emissions of HEFA fuels, especially from rapeseed and canola.[2] Greater confidence in the life-cycle GHG emissions of HEFA fuel, along with a host of economic and political factors, may aid the adoption of HEFA fuel in Canada, one of the world's top producers of canola.[3] Between 2008 and 2016, there were at least 13 LCAs published on HEFA fuels derived from canola, camelina, or carinata.[4–16]

This review attempts to identify how variability in the reported emissions intensity of HEFA fuels can be attributed to four aspects of LCA methodology and scope: feedstock, the inclusion of emissions from land-use change, co-product allocation method, and refining technology.

In a previous review of biofuel LCAs, the inclusion of emissions from land-use change and co-product allocation methods were identified as dominant sources of variability in the reported life-cycle GHG emissions of biofuels.[2]

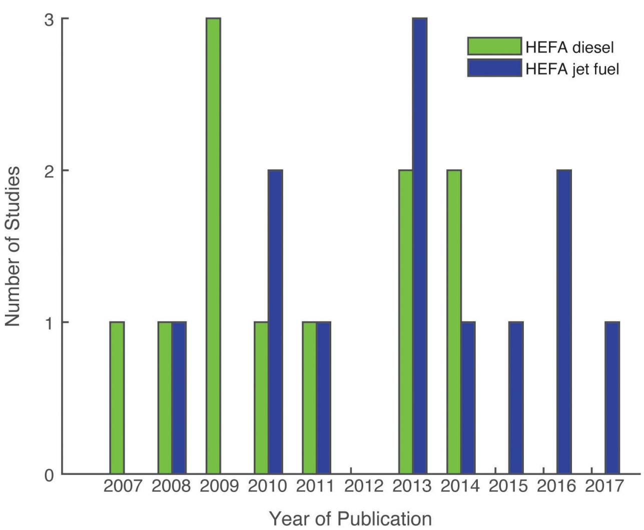

Figure 1. Number of LCAs of HEFA jet or diesel from crop-based oilseeds by year of publication.

The life-cycle GHG emissions of biofuels across different feedstocks and production pathway assumptions has also been reviewed,[17] but with a focus on biodiesel and ethanol. Another study[18] reviewed the methodological assumptions and data sources from some HEFA LCAs, but included a very limited number of publications. A literature search from 2000 to 2018 identified at least 15 LCAs of HEFA from oilseeds that have not been included in any HEFA LCA to date.

To address these gaps, this study reviews 20 LCAs of HEFA diesel and jet fuel from oilseeds, over half of which have been published since 2012 (Fig. 1). Life-cycle GHG emissions are reported and compared across choice of feedstock, co-product allocation method, inclusion of land-use change emissions, and refining technology. A better understanding of both intra- and inter-study variability across aspects of LCA methodology and scope will help guide further research on the LCA of biofuels from oilseeds, and may support better uptake of drop-in fuels in the future.

Review methodology

Scope

This review was restricted to LCAs of HEFA from crop-based oilseeds that can be grown in Canada: soybean, rapeseed / canola (*Brassica napus/rapa/juncea)*, camelina (*Camelina sativa*, sunflower (*Glycine max*), carinata (*Brassica carinata)*, and pennycress (*Thlaspi arvense*). All LCAs of HEFA fuels from these crops were analyzed, including studies where the geographic region of the study was outside of Canada. Canola and rapeseed are grouped together due to the chemical similarity of their oil,[19] although the range of yields reported for rapeseed is up to 50% higher than that of canola.[4,20]

All LCAs reviewed had a similar scope, beginning with the farming of raw feedstock and ending with the combustion of fuel. This scope, called Well-to-Wheels (WTW) for road fuels and Well-to-Wake (WTWa) for aviation fuels, is shown in Fig. 2. When combustion emissions are not

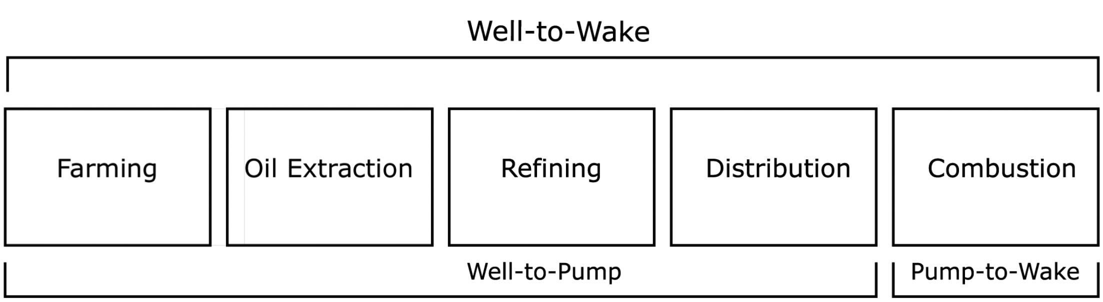

Figure 2. Well-to-wake scope and associated life-cycle stages of HEFA LCAs. The WTW scope can be subdivided into well-to-pump and pump-to-wake stages. Adapted from Elgowainy *et al*.[21]

© 2020 Society of Chemical Industry and John Wiley & Sons, Ltd | *Biofuels, Bioprod. Bioref.* **14**:935–949 (2020); DOI: 10.1002/bbb.2125

included in an LCA, the scope is referred to as well-to-tank (WTT). Life-cycle assessments with both WTT and WTW/WTWa scopes were included in this review because most WTW LCAs assume that GHG emissions at the combustion stage are negligible. Carbon dioxide makes up the majority of combustion emissions and is considered to be carbon neutral for biofuels, as this GHG is sequestered from the atmosphere during crop growth. However, some LCAs have considered the effect of non-CO_2 combustion emissions, namely NO_x and particulate matter, on radiative forcing.[13,22] Life-cycle assessments of both diesel and jet fuel were included because the potential sources of methodological variability affect both fuels equally, and the life cycles are very similar up to final use.

Analysis

To examine methodological variability, 76 scenarios were selected from the 20 studies reviewed. Selected scenarios were defined as sub-analyses within a study where one or more aspects of LCA methodology or scope were varied: feedstock, co-product allocation method, the inclusion of land-use change emissions, or HEFA refining technology.

A number of studies included sensitivity analyses where the value of life-cycle parameters such as nitrogen fertilization rate or crop yield were varied; only the baseline scenario was included in this analysis. A quantitative analysis of the relationship between these parameters and reported life-cycle GHG emissions was beyond the scope of this review, but the results of these scenarios are included in Table 1. The importance of certain parameters to life-cycle GHG emissions is discussed in the results and discussion section.

The distribution of life-cycle GHG emissions for HEFA fuels is reported herein (a) across all scenarios, grouped by HEFA product; and (b) across scenario analyses in the literature, grouped by publication and aspect of LCA methodology and scope. Where applicable, results are reported across the range of the other variables to remove possible confounding factors. For example, if a study applied four co-product allocation methods for both biojet and HEFA diesel, two sets of four data-points each are plotted, one each for biojet and HEFA diesel.

A statistical analysis of the difference between the mean life-cycle GHG emissions published for biojet and renewable diesel was performed. No statistical analysis was performed on results across other variables because the number of comparable scenarios was small, and because LCA is a relative approach where results are absolute calculations with inherent uncertainty and not samples of a larger population.[35] For a discussion of quantitative methods in LCA, see Beltran *et al*. and Grant *et al*.[36,37]

Reported life-cycle GHG emissions were compared in grams of CO_2 equivalent emissions per MJ of fuel. If results were reported in an alternative functional unit, such as distance (kg-km or passenger-km)[9] or mass (kg),[8] they were converted to MJ based on the energy density of the fuel or average fuel consumption of a vehicle. If numerical results were not reported for a scenario, results were estimated to the nearest half integer from available figures.

Aspects of LCA methodology and scope

Feedstock

Feedstock was analyzed as a potential source of variability in the LCA of HEFA fuels because the production system, and thus GHG emissions of every feedstock is different. However, all the feedstocks examined in this study were oilseeds, most of which share certain agronomic and physical characteristics. Seed oil content by weight ranges from 34 to 46% for all oilseeds except soy, which has a lower oil content (18–21%).[8,10,18] Inputs required in the farming stage are typically nitrogen, phosphorus, and potassium fertilizer, diesel (farming equipment), and occasionally pesticides; N_2O emissions from applied nitrogen fertilizer and from crop residues are also accounted for. Five studies compared life-cycle GHG emissions across different feedstocks.[4,7–10]

Land-use change

Land-use change emissions have long been identified as an important factor when considering the sustainable production of biofuels.[32,34] Land-use change emissions refer to the release of GHGs, namely CO_2, that can occur as vegetation and exposed soil naturally decompose when land is converted from one use to another. Land-use change emissions can be modeled as direct or indirect emissions. Direct emissions occur when land conversion takes place in the geographic region where a product system is located; indirect emissions occur when the production of a material within a product system results in land conversion elsewhere, and are related to economic signals.[32,34,38] Whether land-use change emissions are modeled as direct or indirect, the initial or reference land use is an important assumption when estimating land-use change emissions related to biofuel production, as some land uses have larger initial carbon stocks than others. There is a high level of uncertainty associated with land-use change estimates, as most accounting methodology relies on general assumptions that do not take into account soil carbon dynamics, soil texture, or crop type.[38] Five studies included an estimate of land-use change emissions.[4,6,7,16,33]

Table 1. Methdology and life-cycle greenhouse gas emissions of all studies reviewed (121 scenarios in 20 studies).

Study	Life cycle GHG emissions (g CO_2eq/MJ)	Feedstock	Co-product allocation method- meal	Co-product allocation method- other fuels	Land use change (LUC) included	Refining technology	HEFA product	Geographic location	Additional assumptions
Agusdinata *et al.*[11]	60	camelina	D	D	No	UOP	Jet	MT USA	
Arvidsson *et al.*[23]	29.6	rapeseed	D	D	No	Neste	Diesel	EUR	High seed yield 3.24 t/ha
Arvidsson *et al.*[23]	67.3	rapeseed	D	D	No	Neste	Diesel	EUR	Baseline seed yield 1.045 t/ha
Arvidsson *et al.*[23]	87.5	rapeseed	D	D	No	Neste	Diesel	EUR	3x N_2O EF- based on literature
de Jong *et al.*[24]	37	camelina	M	E	No	UOP	Jet	MT USA	
de Jong *et al.*[24]	44	camelina	D	E	No	UOP	Jet	MT USA	
deJong *et al.*[24]	47	camelina	E	E	No	UOP	Jet	MT USA	
deJong *et al.*[24]	47	camelina	$	E	No	UOP	Jet	MT USA	
Edwards *et al.*[9]	45.4	sunflower	D	D	No	Neste	Diesel	EUR	Modeled N_2O EF (GNOC calculator)
Edwards *et al.*[9]	55.8	soybean	D	D	No	Neste	Diesel	EUR	Modeled N_2O EF (GNOC calculator)
Edwards *et al.*[9]	57.1	rapeseed	D	D	No	Neste	Diesel	EUR	Modeled N_2O EF (GNOC calculator)
Edwards *et al.*[9]	57.7	rapeseed	D	D	No	UOP	Diesel	EUR	Modeled N_2O EF (GNOC calculator)
Fan *et al.*[25]	−18.3	pennycress	D	D	No	UOP	Jet	USA	
Fan *et al.*[25]	13.1	pennycress	D	D	No	UOP	Diesel	USA	
Fan *et al.*[25]	30.1	pennycress	E	E	No	UOP	Diesel	USA	
Fan *et al.*[25]	32.7	pennycress	E	E	No	UOP	Jet	USA	
Fan *et al.*[25]	40.7	pennycress	$	$	No	UOP	Diesel	USA	
Fan *et al.*[25]	44.9	pennycress	$	$	No	UOP	Jet	USA	
Han *et al.*[10]	30	soybean	M	E	No	UOP	Jet	EUR	
Han *et al.*[10]	31	soybean	E	D	No	UOP	Jet	EUR	
Han *et al.*[10]	39.5	soybean	E	E	No	UOP	Jet	EUR	
Han *et al.*[10]	39.5	soybean	E	M	No	UOP	Jet	EUR	
Han *et al.*[10]	41	soybean	$	E	No	UOP	Jet	EUR	
Han *et al.*[10]	41.5	soybean	E	$	No	UOP	Jet	EUR	
Han *et al.*[10]	43	soybean	D	E	No	UOP	Jet	EUR	
Han *et al.*[10]	47	camelina	E	E	No	UOP	Jet	MT USA	
Han *et al.*[10]	54	rapeseed	E	E	No	UOP	Jet	EUR	
Huo *et al.*[26]	−28.44	soybean	D	D	No	NRC	Diesel	USA	
Huo *et al.*[26]	24.65	soybean	E	E	No	UOP	Diesel	USA	
Huo *et al.*[26]	24.65	soybean	$	$	No	UOP	Diesel	USA	
Huo *et al.*[26]	30	soybean	E	$	No	NRC	Diesel	USA	
Huo *et al.*[26]	32.23	soybean	E	E	No	NRC	Diesel	USA	

© 2020 Society of Chemical Industry and John Wiley & Sons, Ltd | *Biofuels, Bioprod. Bioref.* **14**:935–949 (2020); DOI: 10.1002/bbb.2125

Table 1. (Continued).

Study	Life cycle GHG emissions (g CO_2eq/MJ)	Feedstock	Co-product allocation method- meal	Co-product allocation method- other fuels	Land use change (LUC) included	Refining technology	HEFA product	Geographic location	Additional assumptions
Huo *et al.*[26]	33.2	soybean	D	E	No	NRC	Diesel	USA	
Huo *et al.*[26]	34	soybean	E	$	No	UOP	Diesel	USA	
Huo *et al.*[26]	34.12	soybean	D	D	No	UOP	Diesel	USA	
Huo *et al.*[26]	36.02	soybean	$	$	No	NRC	Diesel	USA	
Huo *et al.*[26]	40.78	soybean	D	E	No	UOP	Diesel	USA	
Kalnes *et al.*[27]	12.7	soybean	M	M	No	UOP	Diesel	USA	H_2 from renewable fuels
Kalnes *et al.*[27]	14	soybean	M	M	No	UOP	Diesel	USA	H_2 from petroleum
Kalnes *et al.*[28]	17.3	soybean	M	M	No	UOP	Diesel	USA	Data from Sheehan *et al.*[29]
Kalnes *et al.*[28]	41	soybean	D	D	No	UOP	Diesel	USA	Data from Edwards *et al.*[30]
Kalnes *et al.*[28]	48.8	soybean	M	M	No	UOP	Diesel	USA	Data from Hill *et al.*[31]
Li and Mupondwa[12]	3	camelina	D	D	No	UOP	Jet	SK CAN	High yield, low N, urea ammoniaum nitrate, refining 1
Li and Mupondwa[12]	7	camelina	D	D	No	UOP	Jet	SK CAN	High yield, low N, urea ammoniaum nitrate, refining 2
Li and Mupondwa[12]	7.5	camelina	D	D	No	UOP	Jet	SK CAN	High yield, low N, urea, refining 1
Li and Mupondwa[12]	8.5	camelina	D	D	No	UOP	Jet	SK CAN	High yield, low N, urea, refining 2
Li and Mupondwa[12]	27	camelina	D	D	No	UOP	Jet	SK CAN	Low yield, high N, urea ammoniaum nitrate, refining 1
Li and Mupondwa[12]	28.5	camelina	D	D	No	UOP	Jet	SK CAN	Low yield, high N, urea ammoniaum nitrate, refining 2
Li and Mupondwa[12]	30.5	camelina	D	D	No	UOP	Jet	SK CAN	Low yield, high N, urea, refining 1
Li and Mupondwa[12]	31	camelina	D	D	No	UOP	Jet	SK CAN	Low yield, high N, urea, refining 2
Lokesh *et al.*[13]	31.4	camelina	E	E	No	UOP	Jet	MT USA	
Miller and Kumar[7]	58	canola	M	M	No	UOP	Diesel	AB CAN	2,2 N_2O EF
Miller and Kumar[7]	67	canola	M	M	No	UOP	Diesel	AB CAN	3,3 N_2O EF
Miller and Kumar[7]	75	canola	M	M	No	UOP	Diesel	AB CAN	4,4 N_2O EF
Miller and Kumar[7]	84	canola	M	M	No	UOP	Diesel	AB CAN	5,5 N_2O EF
Miller and Kumar[7]	33	canola	M	M	No	UOP	Diesel	AB CAN	Also, straw co-product
Miller and Kumar[7]	48	canola	M	M	No	UOP	Diesel	AB CAN	Baseline 0.76, 0.76- Alberta specific N_2O EF)

Table 1. (Continued).

Study	Life cycle GHG emissions (g CO_2eq/MJ)	Feedstock	Co-product allocation method- meal	Co-product allocation method- other fuels	Land use change (LUC) included	Refining technology	HEFA product	Geographic location	Additional assumptions
Miller and Kumar[7]	71	canola	$	$	No	UOP	Diesel	AB CAN	Baseline 0.76, 0.76- Alberta specific N_2O EF)
Miller and Kumar[7]	50	canola	M	M	No	UOP	Diesel	AB CAN	1,1 N_2O EF instead of 0.76, 0.76- Alberta specific)
Miller and Kumar[7]	94	canola	M	M	Grassland	UOP	Diesel	AB CAN	LUC 20 years
Miller and Kumar[7]	44	canola	M	M	No	UOP	Diesel	AB CAN	Maximum seed yield
Miller and Kumar[7]	55	canola	M	M	No	UOP	Diesel	AB CAN	Minimum seed yield
Miller and Kumar[7]	44	camelina	M	M	No	UOP	Diesel	AB CAN	2,2 N_2O EF
Miller and Kumar[7]	49	camelina	M	M	No	UOP	Diesel	AB CAN	3,3 N_2O EF
Miller and Kumar[7]	53	camelina	M	M	No	UOP	Diesel	AB CAN	4,4 N_2O EF
Miller and Kumar[7]	58	camelina	M	M	No	UOP	Diesel	AB CAN	5,5 N_2O EF
Miller and Kumar[7]	30	camelina	M	M	No	UOP	Diesel	AB CAN	Also, straw co-product
Miller and Kumar[7]	38	camelina	M	M	No	UOP	Diesel	AB CAN	Baseline 0.76, 0.76- Alberta specific N_2O EF)
Miller and Kumar[7]	57	camelina	$	$	No	UOP	Diesel	AB CAN	Baseline 0.76, 0.76- Alberta specific N_2O EF)
Miller and Kumar[7]	39	camelina	M	M	No	UOP	Diesel	AB CAN	Higher N_2O EF (1,1 instead of 0.76, 0.76)
Miller and Kumar[7]	33	camelina	M	M	No	UOP	Diesel	AB CAN	Maximum seed yield
Miller and Kumar[7]	43	camelina	M	M	No	UOP	Diesel	AB CAN	Minimum seed yield
Nikander[15]	33.4	rapeseed	M	M	No	UOP	Diesel	EUR	
Nikander[15]	59.3	rapeseed	D	D	No	UOP	Diesel	EUR	
Shonnard *et al.*[14]	–17.03	camelina	D	D	No	UOP	Jet	MT USA	
Shonnard *et al.*[14]	4.2	camelina	D	D	No	UOP	Diesel	MT USA	Baseline, 40% H_2 from SMR, urea
Shonnard *et al.*[14]	15.8	camelina	M	M	No	UOP	Diesel	MT USA	Baseline, 40% H_2 from SMR, urea
Shonnard *et al.*[14]	16.2	camelina	E	E	No	UOP	Diesel	MT USA	N from liquid NH_4
Shonnard *et al.*[14]	18.04	camelina	E	E	No	UOP	Diesel	MT USA	Baseline, 40% H_2 from SMR, urea
Shonnard *et al.*[14]	19	camelina	E	E	No	UOP	Diesel	MT USA	2x farm diesel
Shonnard *et al.*[14]	20.18	camelina	M	M	No	UOP	Jet	MT USA	
Shonnard *et al.*[14]	21	camelina	E	E	No	UOP	Diesel	MT USA	N from AN
Shonnard *et al.*[14]	22.36	camelina	E	E	No	UOP	Jet	MT USA	
Shonnard *et al.*[14]	23.1	camelina	E	E	No	UOP	Diesel	MT USA	2× N_2O EF

© 2020 Society of Chemical Industry and John Wiley & Sons, Ltd | *Biofuels, Bioprod. Bioref.* **14**:935–949 (2020); DOI: 10.1002/bbb.2125

Table 1. (Continued).

Study	Life cycle GHG emissions (g CO_2eq/MJ)	Feedstock	Co-product allocation method- meal	Co-product allocation method- other fuels	Land use change (LUC) included	Refining technology	HEFA product	Geographic location	Additional assumptions
Shonnard *et al.*[14]	23.4	camelina	E	E	No	UOP	Diesel	MT USA	100% H_2 from steam methane reforming
Shonnard *et al.*[14]	27	camelina	E	E	No	UOP	Diesel	MT USA	2× N fertilizer
Sieverding *et al.*[8]	30.8	camelina	E	E	No	UOP	Jet	ND/SD USA	Triangular input distribution ±4.5
Sieverding *et al.*[8]	34.5	carinata	E	E	No	UOP	Jet	ND/SD USA	Triangular input distribution ±4.6 GHG
Sieverding *et al.*[8]	37.9	sunflower	E	E	No	UOP	Jet	ND/SD USA	Triangular distribution ±4.8 GHG
Sieverding *et al.*[8]	44	canola	E	E	No	UOP	Jet	ND/SD USA	Triangular input distribution ±7.7 GHG
Stratton *et al.*[2]	54.9	rapeseed	$	E	No	UOP	Jet	ND/SD USA	
Stratton *et al.*[2]	97.9	rapeseed	$	E	Set aside grassland	UOP	Jet	EUR	30 years LUC (Fargione *et al.*[32])
Stratton *et al.*[2]	37	soybean	$	E	No	UOP	Jet	EUR	
Stratton *et al.*[2]	97.8	soybean	$	E	Set aside grassland	UOP	Jet	USA	30 years LUC (Fargione *et al.*[32])
Stratton *et al.*[2]	564.2	soybean	$	E	Rainforest	UOP	Jet	USA	30 years LUC (Fargione *et al.*[32])
Ukaew *et al.*[5]	42.7	canola	D	D	No	UOP	Jet	ND USA	Lowest regional N_2O EF (RSB methods)
Ukaew *et al.*[5]	43	canola	D	D	No	UOP	Jet	ND USA	Highest regional N_2O EF (RSB methods)
Ukaew *et al.*[5]	45.9	canola	D	D	No	UOP	Jet	ND USA	IPCC N_2O EF defaults (1.325,1.225)
Ukaew *et al.*[6]	−12	canola	D	D	Cropland	UOP	Jet	ND USA	30 years LUC; low seed price
Ukaew *et al.*[6]	21	canola	D	D	Cropland	UOP	Jet	ND USA	30 years LUC; high seed price
Ukaew *et al.*[6]	33	canola	$	$	Cropland	UOP	Jet	ND USA	30 years LUC; low seed price
Ukaew *et al.*[6]	36	canola	E	E	Cropland	UOP	Jet	ND USA	30 years LUC; low seed price
Ukaew *et al.*[6]	44	canola	$	$	Cropland	UOP	Jet	ND USA	30 years LUC; high seed price
Ukaew *et al.*[6]	49	canola	E	E	Cropland	UOP	Jet	ND USA	30 years LUC; high seed price
Uusitalo *et al.*[16]	257.3	rapeseed	E	E	Forest	Neste	Diesel	EUR	LUC 20 years
Uusitalo *et al.*[16]	419.8	rapeseed	None	E	Forest	Neste	Diesel	EUR	LUC 20 years
Uusitalo *et al.*[16]	433.2	rapeseed	D	D	Forest	Neste	Diesel	EUR	LUC 20 years
Uusitalo *et al.*[16]	46.9	rapeseed	E	E	Grassland	Neste	Diesel	EUR	LUC 20 years
Uusitalo *et al.*[16]	49.4	rapeseed	None	E	Grassland	Neste	Diesel	EUR	LUC 20 years
Uusitalo *et al.*[16]	70.4	rapeseed	D	D	Grassland	Neste	Diesel	EUR	LUC 20 years
Uusitalo *et al.*[16]	30.2	rapeseed	D	D	No	Neste	Diesel	EUR	
Uusitalo *et al.*[16]	36.9	rapeseed	D	D	No	Neste	Diesel	EUR	
Uusitalo *et al.*[16]	52.8	rapeseed	None	E	No	Neste	Diesel	EUR	

Table 1. (Continued).

Study	Life cycle GHG emissions (g CO_2eq/MJ)	Feedstock	Co-product allocation method- meal	Co-product allocation method- other fuels	Land use change (LUC) included	Refining technology	HEFA product	Geographic location	Additional assumptions
Wong[33]	21	soybean	M	E	No	UOP	Jet	USA	
Wong[33]	35.2	soybean	$	E	No	UOP	Jet	USA	
Wong[33]	37.3	soybean	D	E	No	UOP	Jet	USA	
Wong[33]	37.5	soybean	E	E	No	UOP	Jet	USA	
Wong[33]	38.5	soybean	D	E	Global average	UOP	Jet	USA	30 years LUC; Searchinger *et al.*[34]
Wong[33]	131.5	soybean	M	E	Global average	UOP	Jet	USA	30 years LUC; Searchinger *et al.*[34]
Wong[33]	289	soybean	$	E	Global average	UOP	Jet	USA	30 years LUC; Searchinger *et al.*[34]
Wong[33]	314.4	soybean	E	E	Global average	UOP	Jet	USA	30 years LUC; Searchinger *et al.*[34]

Co-product allocation

Co-product allocation is the method by which GHG emissions are allocated between multiple outputs of a product system. The allocation of GHG emissions between the main product and any co-products of a product system can occur based on an allocation ratio or a displacement credit. An allocation ratio is based on properties of the products, such as mass, energy content, or market value. A displacement credit subtracts GHG emissions from the life cycle of the main product, assuming that a co-product displaces a similar product in the global market (Fig. 3). The GHG emissions subtracted are equivalent to the life-cycle GHG emissions of the displaced product.

The two main HEFA life-cycle stages where co-products are generated are oil extraction and refining. The co-product from oil extraction is oilseed meal, which is typically sold as animal feed. Refining of HEFA co-produces other renewable fuels, the amounts of which vary depending on the refining technology and on whether the main fuel product is diesel or jet fuel. Although the feedstock production stage also co-produces straw, this is rarely treated as a co-product due to uncertainty surrounding its use.[16] Oilseed straw is typically generated in low volumes and is usually left on the field to decompose and replenish soil nutrients. Field studies have linked the removal of straw to direct land-use change GHG emissions through the loss of soil carbon.[39]

There are advantages and disadvantages to all co-product allocation methods, and different allocation methods can generate different estimates of life-cycle GHG emissions. To increase transparency, many LCAs perform a sensitivity analysis on the co-product allocation methods used. Ten studies reviewed herein compared the GHG emissions of HEFA fuels under different allocation methods.[7,10,14–16,26]

Refining technology

Four companies have patented HEFA refining technologies that have been licensed by commercial fuel producers: UOP Honeywell, Neste, Haldor Topsoe, and Axens. Natural Resources Canada (NRCan) also developed a process for HEFA diesel production called Super Cetane, but this process has yet to be commercialized. Super Cetane technology was modeled by one study in this review.[26] Three studies presented Neste process data,[9,15,16,23] and all other studies used UOP process data. UOP is the most widely licensed HEFA refining technology. Parameters that vary between hydroprocessing technologies include spread of final products (product slate), consumption of hydrogen gas, and consumption and source of energy (typically

 © 2020 Society of Chemical Industry and John Wiley & Sons, Ltd

natural gas). Two studies compared the GHG emissions of HEFA fuels from different refining technologies.[9,26]

Results and discussion

Distribution of results across all scenarios

The median life-cycle GHG emissions of HEFA fuel reported in the literature was 40.8 g CO_2eq/MJ from 66 scenarios for renewable diesel, and 37.5 g CO_2eq/MJ from 55 scenarios for biojet (Fig. 4). The range of reported results for both fuel products was large, between –28.4 and 433.2 g CO_2eq/MJ for diesel and between –18.3 and 564.2 g CO_2eq/MJ for biojet. Although there was no significant difference between the mean life-cycle GHG emissions reported for bio jet (μ = 55.8) and renewable diesel (μ = 55.5) in the literature (t = 0.02, $P = 0.98$), the median is a more appropriate characterization of the center of the two datasets as the inclusion of land-use change scenarios lends a right skew to the data. There is a 63% reduction in median life-cycle GHG emissions of biojet (37.5 g CO_2eq/MJ) compared with conventional low sulfur jet fuel (101.5 g CO_2eq/MJ),[40] and a 61% reduction in median life-cycle GHG emissions of renewable diesel (37.5 g CO_2eq/MJ) compared with conventional diesel (105.6 g CO_2eq/MJ).[40]

Feedstock

Five studies compared the life-cycle GHG emissions of HEFA fuel across different oilseed feedstocks. Results from inter-study feedstock analyses are compared in Fig. 5. Separate results from Miler and Kumar[7] are presented each for market-based allocation ($/$) and mass-based allocation (M/M), to

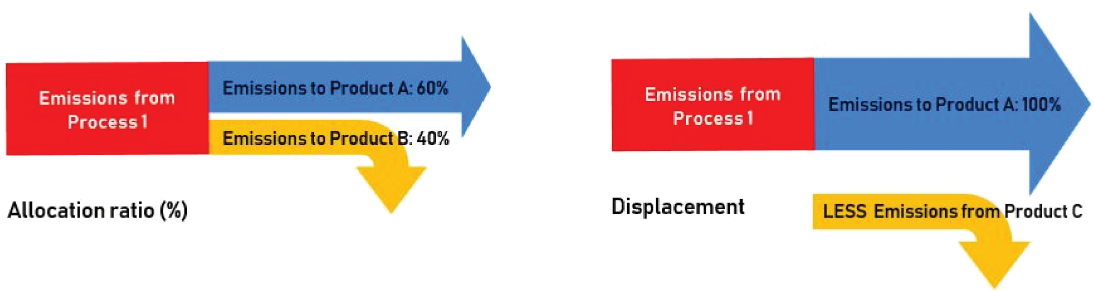

Figure 3. An example of the application of ratio allocation and displacement allocation to the flow of process GHG emissions between multiple outputs of a life cycle.

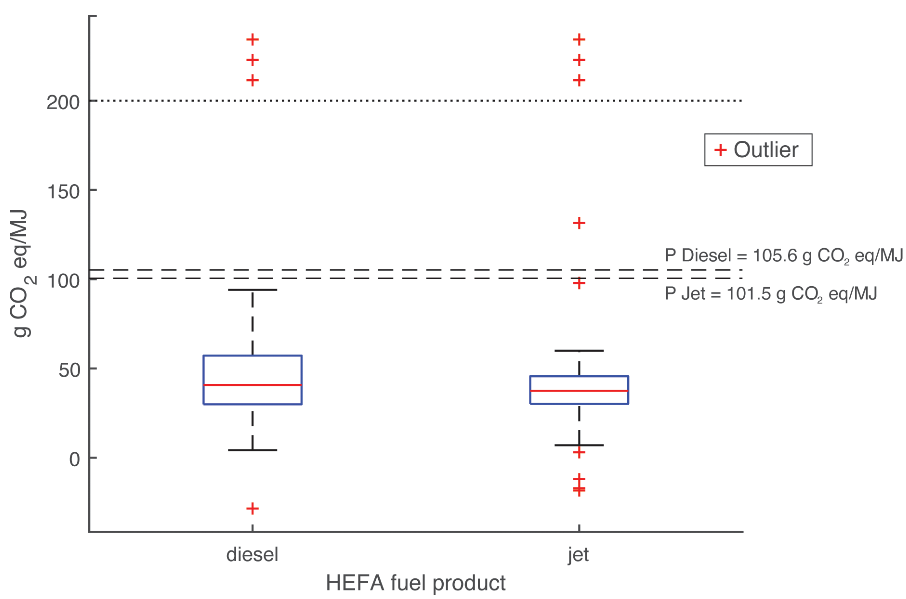

Figure 4. The distribution of reported life-cycle GHG emissions for HEFA diesel (n = 66) and HEFA biojet (n = 55) from scenarios that were reviewed within the literature. A data limit was set at 200 g CO_2 eq/MJ beyond which any outliers appear evenly distributed. Lines labeled P. Jet and P. Diesel provide a comparison with life-cycle GHG emissions of conventional petroleum fuels.

separate the effect of a co-product allocation analysis from the feedstock analysis. All studies that compared HEFA fuels derived from different oilseeds included canola, likely because it is one of the most commercially available oilseeds for HEFA production. In all cases, HEFA fuel from canola had higher GHG emissions than fuel from other feedstocks. No conclusive ranking of fuel from other feedstocks could be drawn, as no non-canola feedstock pairs were analyzed in more than one study.

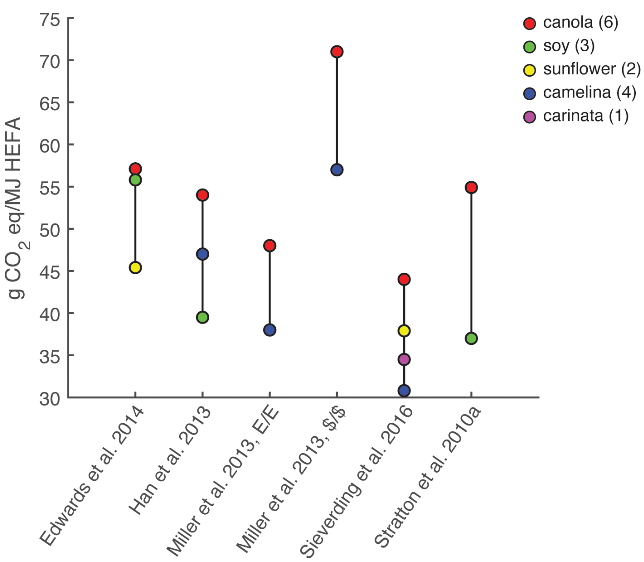

Figure 5. Studies comparing the life-cycle GHG emissions of HEFA fuels from different feedstocks. The number in parentheses indicates the number of studies that included a scenario with that feedstock.

The production and application of nitrogen fertilizer and the associated nitrous oxide (N_2O) emissions are major contributors to the life-cycle GHG emissions of fuel from oilseeds,[4,7–10] and may contribute to the higher GHG emissions reported for fuels from canola. Nitrogen-related emissions can represent as much as 40% of total life-cycle emissions.[7] Other parameters that differ between feedstocks include crop yield, oil yield, and other chemical inputs, N_2O is emitted naturally from soil through the microbial and abiotic oxidation of organic nitrogen compounds, but the application of nitrogen fertilizer results in increased N_2O emissions. The highest nitrogen application rates of any feedstock were reported for canola. Soybean requires little added nitrogen as legumes can convert atmospheric nitrogen gas (N_2) into plant-usable forms. Camelina, pennycress, and carinata all reportedly require less added nitrogen than canola, although the low production levels of these alternative oilseeds in North America means that the nitrogen application rates are subject to uncertainty.[14] Sensitivity analyses in the studies reviewed have shown that nitrogen application rate and crop yield are significant sources of variability in reported GHG emissions.[12,14]

Co-product allocation method

Fourteen different combinations of co-production allocation methods were applied in the studies reviewed. The co-products to which GHG emissions were allocated were

Table 2. The 14 different co-product allocation methods applied in the studies reviewed to each stage of the product system where a co-product is produced: farming (straw), oil extraction (meal), and refining (other fuels).

Co-product allocation method	Number of studies applied	Method applied to each co-product		
		Straw	Meal	Other fuels
$/$	6	None	Market	Market
$/E	3	None	Market	Energy
D/D	10	None	Displacement	Displacement
D/E	5	None	Displacement	Energy
E/$	3	None	Energy	Market
E/D	1	None	Energy	Displacement
E/E	12	None	Energy	Energy
E/M	1	None	Energy	Mass
M/E	3	None	Mass	Energy
M/M	7	None	Mass	Mass
None/$	1	None	None	Market
None/E	1	None	None	Energy
None/M	1	None	None	Mass
M/M/M	1	Mass	Mass	Mass

 © 2020 Society of Chemical Industry and John Wiley & Sons, Ltd

oilseed meal and other renewable fuels, although oilseed straw was also treated as a co-product in a scenario by Miller and Kumar.[7] All studies applied one of the following co-product allocation methods to each co-product: energy-based, mass-based, market-based, displacement, or no allocation (Table 2).

Half of the studies reviewed compared two or more co-product allocation methods. A total of 14 intra-study co-product allocation comparisons are shown in Fig. 6. Land-use change scenarios from two studies[16,33] were excluded from this analysis to enable readability, but are included in Fig. 7.

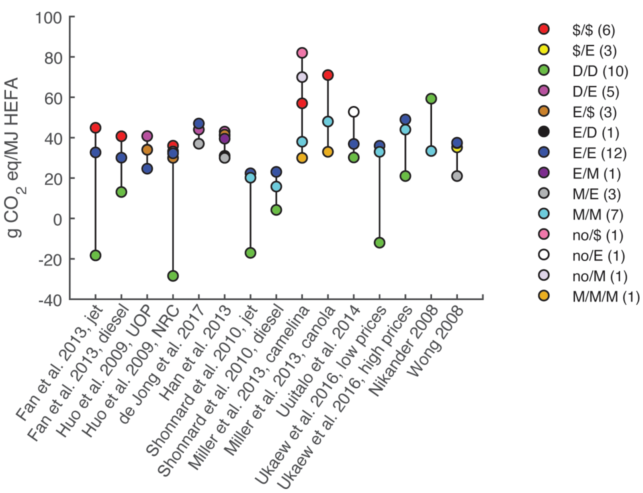

Figure 6. Studies comparing the life-cycle GHG emissions of HEFA fuels when different co-product allocation methods are applied. The number in parentheses indicates the number of studies that applied that co-product allocation method.

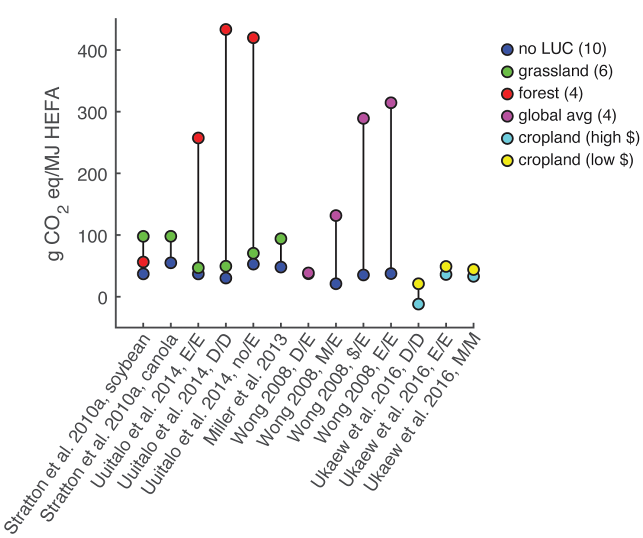

Figure 7. Studies comparing the life-cycle GHG emissions of HEFA fuels under different land-use change assumptions. The number in parentheses indicates the number of studies that used that assumption. For the cropland to cropland scenarios (Ukaew *et al.*[6]), high canola price (high $) and low canola price (low $) scenarios were analyzed.

In most studies, higher GHG emissions were reported for HEFA fuels when market-based ($/$) allocation was applied, compared to when displacement (D/D), mass-based (M/M), or energy-based (E/E) allocation was applied. Scenarios applying M/M or M/E allocation generated slightly lower GHG emissions than E/E scenarios. In all studies except Huo *et al.*,[26] D/D allocation resulted in the lowest GHG emissions for HEFA fuel when compared with other allocation methods. Across the literature reviewed, negative life-cycle GHG emissions were reported in 4/10 studies that applied displacement allocation to all co-products generated.[6,14,25,26]

The E/D and D/D allocation methods credit the HEFA fuel product with the production and combustion of displaced petroleum-derived fuels that have high life-cycle GHG emissions. When the yield of renewable fuel co-products is relatively high, the displacement credit can be very large. Hydroprocessed esters and fatty acids biojet fuel scenarios generated much lower GHG emissions than HEFA diesel scenarios when the displacement method was applied to the refining stage, as the co-product yield was much higher.[14,25] There was little difference observed in the reported GHG emissions of HEFA fuels for other refining allocation methods, as the mass, energy content, and market value ratios of diesel to jet fuel are similar.[10]

In the oil extraction stage, the choice of allocation method was more important. For example, canola seed is 44% oil by mass but 69% oil by energy content, so energy-based methods allocate more emissions to canola oil. The displacement credit applied to the meal co-product is also subject to more variation. Oilseed meal is typically used as animal feed and could replace corn, soy, or another grain in the global market. Stratton *et al.*[41] found that because displaced GHG emissions varied between animal feeds, the life-cycle GHG emissions of HEFA fuels was very sensitive to the choice of animal feed displaced by oilseed meal.[4] Soybean meal is the most common animal feed because of its high protein content, and it is often assumed to be the product displaced by oilseed meal production.[42] However, the life-cycle GHG emissions of the displaced soybean meal are subject to user assumptions and have varied by up to 500% in the studies reviewed.[4,25]

Land-use change

Five studies included estimates of GHG emissions from land-use change. All studies but Wong[33] explicitly considered GHG emissions from direct land-use change. In that study, existing literature values for global indirect land-use change emissions

© 2020 Society of Chemical Industry and John Wiley & Sons, Ltd

from an increase in corn ethanol cultivation in the USA[34] were adopted for soybean cultivation. All studies except Ukaew *et al.*[6] compared the GHG emissions of HEFA fuels with land-use change to a baseline scenario where emissions from land-use change were not included. In all scenarios, the inclusion of land-use change increased the GHG emissions of the final product compared to the baseline (Fig. 7). Where co-product allocation analyses were also performed, results were presented separately.[6,16,33]

Four direct land-use change scenarios and one indirect scenario were considered: forest to cropland, grassland to cropland, cropland to canola cropland under high and low canola prices, and a global average of indirect land-use change modeled by Searchinger *et al.*[34] The conversion of set-aside land to cropland was grouped under grassland conversion scenarios due to the two land uses having similar initial carbon stocks. Set-aside land is land that was previously cultivated, where some or all of the soil organic carbon (SOC) has been restored. Set-aside programs were put in place in Europe in the 1980s to cope with agricultural surpluses and restore the ecosystem benefits of grasslands.[43]

The life-cycle GHG emissions of HEFA fuel from LCAs assuming forest to cropland conversion was much higher than other land conversion scenarios, and more than triple that of baseline scenarios (Fig. 7). Both temperate and tropical forests tend to have higher carbon stocks than grasslands due to the larger stocks of above-ground biomass.[44] Global average conversion scenarios also generated high life-cycle GHG emissions, as the conversion of higher carbon stock land uses, including temperate and tropical forests, were included.[34]

Cropland to canola cropland conversion in Ukaew *et al.* generated the lowest life-cycle GHG emissions for HEFA fuel including land-use change reported in the literature, although there was no baseline provided for comparison.[6] An increase in canola prices resulted in an increase in GHG emissions as canola displaced crops with higher soil carbon inputs, such as wheat and sunflower.[6] Cropland has a low initial carbon stock, which limits the losses that can occur from land-use change. Grassland conversion scenarios also generated low life-cycle GHG emissions, as the initial carbon stocks are generally lower than those of other ecosystem types.[44] See MacWilliams *et al.* for a discussion of the environmental impact of canola production in Western Canada, including an accounting of emissions from land-use change.[45]

Two studies compared land-use change scenarios across different co-product allocation methods. In Uuitalo *et al.*, displacement allocation resulted in higher GHG emissions of HEFA fuel compared to other allocation methods.[16] In Wong, displacement allocation resulted in lower GHG emissions compared to other allocation methods.[33] The variability in LCA results can be partially attributed to underlying assumptions about the life-cycle GHG emissions of the displaced products. Oilseed meal was assumed to displace soybeans in both studies, but land-use change emissions for the displaced soybeans were only included in Uuitalo *et al.*[16] Uuitalo *et al.* also applied displacement to the renewable fuel co-product (D/D) whereas Wong applied an energy-based allocation (D/E).[16,33] Emissions allocated to the renewable fuel co-product increase proportionally when land-use change is included under an allocation ratio like energy-based allocation, but not under displacement, which could contribute to the lower life-cycle GHG emissions reported by Wong.[33]

HEFA technology

Different refining technologies were compared in two studies. In Edwards *et al.*, there was almost no difference in the GHG emissions of HEFA fuel from Neste and UOP refining processes (Fig. 8).[9] Displacement was applied to all co-products. However, when UOP and NRCan processes were compared across a range of co-product allocation methods in Huo *et al.*, there was a bigger difference in life-cycle GHG emissions.[26] The largest difference in GHG emisisons between technologies was observed when displacement was applied to the renewable fuel co-product (D/D). Displacement generated negative GHG emissions for fuel from NRCan, but not from UOP.[26] Process inputs of

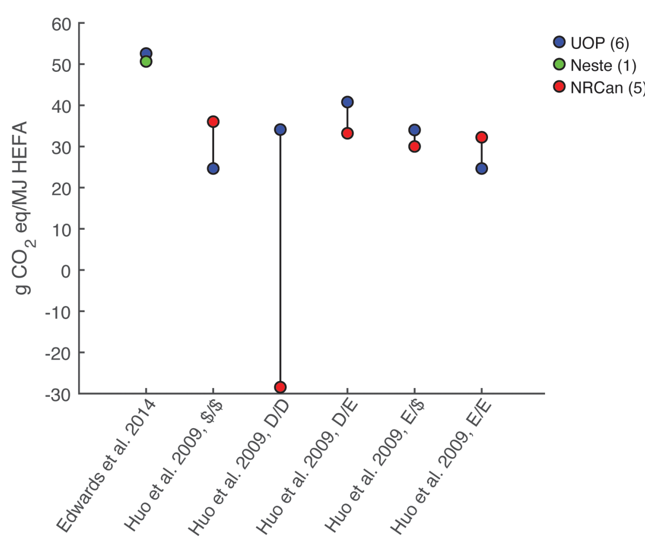

Figure 8. Studies comparing the life-cycle GHG emissions of HEFA fuels from different refining technologies. The number in parentheses indicates the number of studies that modeled that refining technology.

hydrogen, electricity, and natural gas were similar between the two refining technologies, but the co-product yield was much higher for the NRCan process. This resulted in lower life-cycle GHG emissions for fuel from the NRCan process under displacement allocation, but higher life-cycle GHG emissions when market allocation was applied, as the main fuel product had a higher market value than the co-products.

Conclusion and recommendations

Despite a 61–63% reduction in median life-cycle GHG emissions of HEFA biojet and renewable diesel compared with conventional petroleum fuels**,** the wide range in reported life-cycle GHG emissions for HEFA fuels is one of the barriers to the development of a HEFA drop-in fuel supply chain. The aim of this review was to analyze feedstock, co-product allocation method, the inclusion of GHG emissions from land-use change, and refining technology as potential sources of variability in the LCA of HEFA fuels such that these important aspects of LCA methodology and scope could be better understood. Twenty LCAs were reviewed and scenario analyses from selected studies were compared. Previous reviews have included very few of the LCAs on HEFA fuels reviewed herein.

When compared with fuel from other feedstocks, life-cycle GHG emissions were the highest for HEFA fuels from canola, most likely due to high nitrogen fertilizer requirements. Forest to cropland land-use change scenarios were associated with the highest life-cycle GHG emissions reported in the literature. Across a wide range of co-product allocation methods compared in the literature, life-cycle GHG emissions tended to be higher for market-based allocation than for mass- or energy-based allocation, and were the lowest and in some cases even negative when displacement allocation (D/D) was applied. The importance of refining technology was not widely compared in the literature, but the quantity of co-products produced strongly impacted life-cycle GHG emissions if displacement was applied as the co-product allocation method.

Feedstock, co-product allocation method and land-use change inclusion were confirmed as important sources of variability. 10/20 studies reviewed compared two or more co-product allocation methods, in accordance with recommendations by the International Standards Organization.[46] Aspects of LCA methodology, including the co-product allocation method, have been defined for renewable fuel accounting in standards published by the European RED and CORSIA, which use energy-based allocation methods; RSB, which uses market-based allocation; and US RFS2, which uses displacement.[47–50] All methodologies account for emissions from indirect land-use change, which were only included in one study reviewed.[33] Ultimately, LCA is a relative process and standardized methodology is the only way to enable decision making.

Despite variability in reported life-cycle GHG emissions, oilseeds are likely to be used for drop-in fuel production in Canada and elsewhere due to their availability and the commercial status of hydroprocessing technology.[1,51] The LCAs reviewed here also indicate both the need to mitigate land-use change concerns surrounding feedstock production for biofuels, and the advantages of oilseeds that require fewer agricultural inputs, such as camelina and carinata, as feedstocks for HEFA production.

Acknowledgements

The authors gratefully acknowledge funding from BioFuel Net and the Canadian Biojet Supply Chain Initiative (CBSCI). Additional funds were provided through the National Science and Engineering Council (NSERC) and the Canada Research Chairs Program.

References

1. de Jong S, Faaij A, Slade R, Mawhood R and Junginger M, The feasibility of short term production strategies for renewable jet fuels- a comprehensive techno-economic comparison. *Biofuels Bioprod Biorefin* **6**(3):246–256 (2015).
2. Stratton RW, Wong HM and Hileman JI, Quantifying variability in life cycle greenhouse gas inventories of alternative middle distillate transportation fuels. *Environ Sci Technol* **45**:4637–4644 (2011).
3. CBSCI Canadian Biojet Supply Chain Initiative. Policy Tools for Enabling Biojet (2019).
4. Stratton R, Wong H, Hileman J. *Life Cycle Greenhouse Gas Emissions from Alternative Jet Fuels v 1.2*. Partnership for Air Transportation Noise and Emissions Reduction, Cambridge, MA (2010).
5. Ukaew S, Beck E, Meki MN and Shonnard DR, Application of the roundtable on sustainable biofuels method to regional differences in nitrous oxide emissions for the rapeseed hydrotreated renewable jet life cycle. *J Clean Prod* **83**:220–227 (2014).
6. Ukaew S, Shi R, Lee JH, Archer DW, Pearlson M, Lewis KC *et al*., Full chain life cycle assessment of greenhouse gases and energy demand for canola-derived jet fuel in North Dakota, United States. *ACS Sustainable Chem Eng* **4**(5):2771–2779 (2016).
7. Miller P and Kumar A, Development of emission parameters and net energy ratio for renewable diesel from Canola and Camelina. *Energy* **58**:426–437 (2013).
8. Sieverding HL, Zhao X, Wei L and Stone JJ, Life-cycle assessment of oilseeds for biojet production using localized cold-press extraction. *J Environ Qual* **45**(3):967–976 (2016).
9. Edwards R, Hass H, Larivé JF, Lonza L, Mass H, Rickeard D. Well-to wheels analysis of future automotive fuels and powertrains in the European *context v 4.a*. European Commission Joint Research Centre (2014).

10. Han J, Elgowainy A, Cai H and Wang MQ, Life-cycle analysis of bio-based aviation fuels. *BioresourTechnol* **150**:447–456 (2013).
11. Agusdinata DB, Zhao F, Ileleji K and DeLaurentis D, Life cycle assessment of potential biojet fuel production in the United States. *Environ Sci Technol* **45**(21):9133–9143 (2011).
12. Li X and Mupondwa E, Life cycle assessment of camelina oil derived biodiesel and jet fuel in the Canadian Prairies. *Sci Total Environ* **481**(1):17–26 (2014).
13. Lokesh K, Sethi V, Nikolaidis T, Goodger E and Nalianda D, Life cycle greenhouse gas analysis of biojet fuels with a technical investigation into their impact onjet engine performance. *Biomass Bioenergy* **77**:26–44 (2015).
14. Shonnard DR, Williams L and Kalnes TN, Camelina-derived jet fuel and diesel: Sustainable advanced biofuels. *Environ Progress Sustain Energy* **29**:382–392 (2010).
15. Nikander S. Greenhouse gas and energy intensity of product chain: case transport biofuel [Masters thesis]. Helsinki University of Technology (2008).
16. Uusitalo V, Vaisanen S, Havukainen J, Havukainen M, Soukka R and Luoranen M, Carbon footprint of renewable diesel from palm oil, jatropha oil and rapeseed oil. *Renew Energy* **69**:103–113 (2014).
17. Shonnard DR, Klemetsrud B, Sacramento-Rivero J, Navarro-Pineda F, Hilbert J, Handler R *et al*., A review of environmental life cycle assessments of liquid transportation biofuels in the pan American region. *EnvironManage* **56**(6):1356–1376 (2015).
18. Fan J, Handler RM, Shonnard DR and Kalnes TN, A review of life cycle greenhouse gas emissions of hydroprocessed jet fuels from renewable oils and fats. *Int J Environ Sci Eng Res* **3**(3):114–138 (2012).
19. Canola Council of Canada. What is canola? (2020). Available: https://www.canolacouncil.org/oil-and-meal/what-is-canola/
20. Smith EG, Barbieri J. Summary of inputs and production practices used by canola growers in 2011. Technical report, Lethbridge Research Centre, Lethbridge (2012).
21. Elgowainy A, Han J, Wang M, Carter N, Stratton R and Hileman J, *Life Cycle Analysis of Alternative Aviation Fuels in GREET*. Energy Systems Division, Argonne National Laboratory, U.S. Department of Energy, Argonne, IL (2012).
22. Stratton RW, Wolfe PJ and Hileman JI, Impact of aviation non-CO2 combustion effects on the environmental feasibility of alternative jet fuels. *Environ Sci Technol* **45**(24):10736–10743 (2011).
23. Arvidsson R, Persson S, Fröling M and Svanström M, Life cycle assessment of hydrotreated vegetable oil from rape, oil palm and Jatropha. *J Clean Prod* **19**(2–3):129–137 (2011).
24. de Jong SD, Antonissen K, Hoefnagels R, Lonza L, Wang M, Faaij A *et al*., Lifecycle analysis of greenhouse gas emissions from renewable jet fuel production. *Biotechnol Biofuels* **10**(64):1–18 (2017).
25. Fan J, Shonnard DR, Kalnes TN, Johnsen PB and Rao S, A life cycle assessment of pennycress (Thlaspi arvense L.) -derived jet fuel and diesel. *Biomass Bioenergy* **55**:87–100 (2013).
26. Huo H, Wang M, Bloyd C and Putsche V, Life-cycle assessment of energy use and greenhouse gas emissions of soybean-derived biodiesel and renewable fuels. *Environ Sci Technol* **43**(3):750–756 (2009).
27. Kalnes T, Marker T and Shonnard DR, Green diesel : a second generation biofuel. *Int J Chem React Eng* **5**:1–8 (2007).
28. Kalnes TN, Koers KP, Marker T and Shonnard DR, A technoeconomic and environmental life cycle comparison of green diesel to biodiesel and syndiesel. *Environ Prog Sustain Energy* **28**(1):111–120 (2009).
29. Sheehan J, Camobreco V, Duffield J, Graboski M, Shapouri H. Life Cycle Inventory of Biodiesel and Petroleum Diesel for Use in an Urban Bus. Technical report, United States Department of Energy and United States Department of Agriculture Brussels, Belgium (1998).
30. Edwards R., Hass H, Larivé, JF, Lonza, L, Mass H, Rickeard, D. Well-to-wheels analysis of alternative fuels and powertrains in the European context v 1.0. Technical report, European Commission Joint Research Centre (2004).
31. Hill J, Nelson E, Tilman D, Polasky S, Tiffany D, Hill J *et al*., Environmental, economic , and energetic costs and benefits of biodiesel and ethanol biofuels. *Proc Natl Acad Sci U S A* **103**(30):11206–11210 (2006).
32. Fargione J, Hill J, Tilman D, Polasky S and Hawthorne P, Land clearing and the biofuel carbon debt. *Science* **319**:1235–1237 (2008).
33. Wong HM. Life-cycle assessment of greenhouse gas emissions from alternative jet fuels [Masters thesis]. Massachusetts Institute of Technology (2008).
34. Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, Fabiosa J *et al*., Use of U. S. Croplands for biofuels increases greenhouse gases through emissions from land use change. *Science* **319**:1238–1240 (2008).
35. Finkbeiner M, Inaba A, Tan RBH, Christiansen K and Kluppel HJ, The new international standards for life cycle assessment: ISO 14040 and ISO 14044. *Int J Life Cycle Ass* **11**(2):80–85 (2006).
36. Beltran AM, Prado V, Vivanco DF, Henriksson PJG, Guinee JB and Heijungs R, Quantified uncertainties in comparative life cycle assessment: what can be concluded? *Environ Sci Technol* **52**:2152–2161 (2018).
37. Grant A, Ries R and Thompson C, Quantitative approaches in life cycle assessment- part 1- descriptive statistics and factor analysis. *Int J Life Cycle Assess* **21**:903–911 (2016).
38. Goglio P, Smith WN, Grant BB, Desjardins RL, McConkey BG, Campbell CA *et al*., Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. *J Clean Prod* **104**:23–39 (2015).
39. Janzen HH, Campbell CA, Izaurralde RC, Ellert BH, Juma N, McGill WB *et al*., Management effects on soil C storage on the Canadian prairies. *Soil Tillage Res* **47**(3):181–195 (1998).
40. (S&T)2 Consultants Inc, *GHGenius Model 4.03 Volume 2: Data and Data Sources*. Natural Resources Canada, Ottawa, ON (2013).
41. Stratton RW. Life cycle assessment of greenhouse gas emissions and non-CO_2 combustion effects from alternative jet fuels [Masters thesis]. Massachusetts Institute of Technology (2010).
42. Weidema BP. System expansions to handle co-products of renewable materials. 7th LCA Case Studies Symposium SETAC-Europe. 1999; 45–48. [15 September 2019].
43. European Economic Community, *Commission Regulation (EEC) no 1272/88 of 29 April: Laying down detailed rules for applying the set-aside incentive scheme for arable land*. European Economic Community, Brussels (1988).
44. Aalde H, Gonzalez P, Gytarsky M, Krug T, Kurz WA, Lasco RD *et al*., Generic methodologies applicable to multiple land-use categories, in *2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 4: Agriculture, Forestry, and Other Land Use*, ed. by Eggleston HS, Buendia L, Miwa K, Ngara T and Tanabe K. Global Environmental Strategies (IGES) for the IPCC, Kanagawa, Japan pp. 2.1–2.59 (2006).

 © 2020 Society of Chemical Industry and John Wiley & Sons, Ltd

45. MacWilliams S, Sanscartier D, Lemke R, Wismer M and Baron V, Environmental benefits of canola production in 2010 compared to 1990: a life cycle perspective. *Agric Systems* **145**:106–115 (2016).
46. ISO. Environmental management- Life cycle assessment- Goal and scope definition and inventory analysis. ISO 14041:1998(E). International Organization for Standardization (1998).
47. CORSIA. *CORSIA* Supporting Document- CORSIA Eligible Fuels- Life Cycle Assessment Methodology (2019).
48. United States Environmental Protection Agency. EPA Lifecycle Analysis of Greenhouse Gas Emisisons from Renewable Fuels. EPA-420-F-09-024 (2009).
49. European Commission. Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. (No. 2009/28/EC).
50. Roundtable on Sustainable Biomaterials. RSB- Roundtable on Sustainable Biomaterials- RSB GHG Calculation Methodology v 2.3. RSB-STD-01-003-01.
51. Wei H, Liu W, Chen X, Yang Q, Li J and Chen H, Renewable bio-jet fuel production for aviation: a review. *Fuel* **254**:115599 (2019).

Debrah Zemanek

Debrah Zemanek holds an MASc in civil engineering from Queen's University in Kingston, Canada. Her research focuses on the life-cycle assessment of renew able transportation fuels and its application to GHG reductions from the transportation sector.

Pascale Champagne

Dr Pascale Champagne is a world-renowned researcher with expertise in wastewater treatment and bioresources engineering, a Tier II Canada Research Chair in Bioresources Engineering, and the inaugural Director of the Beaty Water Research Centre (BWRC) at Queen's University. She is a recipient of the 2019 Natural Science and Engineering Research Council (NSERC) Brockhouse Canada Prize for Interdisciplinary Research in Science and Engineering and the 2020 Professional Engineers Ontario (PEO) Engineering Medal in Research and Development. Her research program is interdisciplinary and is related to the development of new integrated bioresource management practices to create unique opportunities for the bioenergy and bioproducts sectors. Her achievements in pioneering advances in eco-engineered systems for sustainable and cost-effective wastewater treatment approaches have gained her global recognition (Peru, Mexico, South Africa, Italy, Sweden, the USA, China, Australia and Ecuador). Her long-term contribution to the improvement of wastewater treatment is significant, and has had far-reaching and substantial impact creating effective low-maintenance, and low-cost treatment technology for domestic wastewaters, agricultural runoff, landfill leachate, and acid mine drainage.

Warren Mabee

Dr Warren Mabee is associate dean and director of the School of Policy Studies at Queen's University. A full professor in the Department of Geography and Planning, he holds a Canada Research Chair (Tier II) in renewable energy development and implementation, and is also the Stauffer-Dunning Chair in Public Policy. He was part of the team that was awarded the 2018 SSHRC Impact Award, and was a recipient of the 2019 NSERC Brockhouse Canada Prize for Interdisciplinary Research in Science and Engineering. He is cross appointed to the School of Environmental Studies at Queen's. His international research program focuses on the interface between policy and technology in the area of renewable energy and fuels, addressing issues that bridge the gap between researchers and decision makers using tools such as life-cycle assessment, geographic information systems and agent-based logistical models. His past work experiences include stints at the University of British Columbia and the University of Toronto, as well as the Food and Agriculture Organization of the United Nations.