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Comparative techno-economic and life cycle analyses of synthetic "drop-in" fuel production from UK wet biomass

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ABSTRACT

Renewable synthetic hydrocarbon "drop-in" fuels can help mitigate greenhouse gas emissions from transport, particularly in hard-to-abate sectors like freight and aviation. However, no study has extensively addressed the concerns over biomass availability, cost viability, and CO_2 reduction feasibility that are associated with diverse production configurations and feedstocks. Here, we report detailed techno-economics and life cycle greenhouse gas emission assessments of drop-in fuel productions via hydrothermal liquefaction to assess their economic viabilities, CO_2 mitigation potentials, and prospects for scale-up specifically within the UK context. Our approach integrates key production factors which include regional availability of main feedstocks (digestates, food waste, biodegradable municipal waste, and sewage sludge), plant configurations (centralised vs decentralised) and hydrogen sources (grey, blue, green). We demonstrated the economic trade-off between economy-of-scale and feedstock transport distances in the centralised/decentralised configurations, and also the economic and emissions trade-offs associated with the use of different hydrogen sources. We find that co-processing of different waste feedstocks is an important strategy to minimise fuel selling price by enabling better economy of scale and feedstock transport, resulting in a fuel selling price of £14.76 – 20.30 per GJ. The corresponding greenhouse gas emissions from the co-processing case vary from 11.4 to 24.9 kg CO_2eq per GJ for 2021, based on the consequential life cycle assessment approach. Furthermore, we estimated that the utilisation of key UK wet feedstocks could only provide 4.5 % of current fuel consumptions and reduce emissions by 4.5 – 5.4 Mt CO_2eq/year, which translates to 3.4 – 4.0 % reduction in the UK's 2021 transport emissions.

1. Introduction

Addressing climate change by reducing greenhouse gas (GHG) emissions is one of the most important challenges faced by policymakers and industries globally. In the UK, the modern transport sector is responsible for a quarter of national GHG emissions [1]. Hence, the decarbonisation of transportation systems is considered essential, through strategies such as vehicle electrification and the development of other sustainable technologies. Although current policies across many countries promote vehicle electrification and by 2035 in the UK [2–4], the use of drop-in fuels is strongly recommended in the harder-to-decarbonise areas, such as freight, aviation, and to a lesser extent in the existing light-duty fleet which will be in operation beyond 2035. This is due to the challenges associated with vehicle electrification such as long-distance, aviation and heavy-goods transportation. The EU Renewable Energy Directive (RED) II has set a target of 14 % biofuel consumption in road and rail transport by 2030 [5]. However, currently, biofuels contribute only 5 % of the UK transport fuel consumption volumes [6]. Conventional biofuels such as biodiesel which are currently widely applicable in blended stocks with fossil fuels are limited in applications, due to engine compatibility [7]. In addition, conventional biofuels compete with food crops, and this has strongly intensified the food vs fuel debate, since the global food crisis in 2008 which was partly attributed to first-generation biofuels [8]. Therefore, the use of synthetic

Abbreviations: AD, Anaerobic digestion; BMW, Biogenic municipal solid waste; CO_2, Carbon dioxide; FCI, Fixed capital investment; GLE, Gasoline litre equivalent; GHG, Greenhouse gas; H2, Hydrogen gas; HTL, Hydrothermal liquefaction; LCA, Life cycle assessment; MFSP, Minimum fuel selling price; MSW, Municipal solid wastes; OPEX, Operating expenditure; RED, Renewable Energy Directive; TEA, Techno-economic assessment; USD, United States dollar.

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Table 1
UK wet biomass waste management processes in practical applications in 2021.

Waste feedstocks	Volumes, Mtpa	Key processing and disposal pathways							Based on data by
		Re.	AD	Incinerat.	Composting	Landfill	Land	Others	
Food wastes	11.1 – 13.1	16 %	38 %	13 %	4 %	14 %	7 %	4 %	[9,14–18]
Digestates	14.2[a]	–	n.a	3 %	n.a	2 %	95 %	–	[19,20]
Sewage sludge	30	–	n.a[b]	4 %	–	–	93 %	–	[21]
Landfilled BMW	3.9[c]	–	–	–	–	100 %	–	–	[22]
Manure	83	–	3 %	–	–	–	97 %	–	[19,23]

Key: Re. – recycling; AD – anaerobic digestion; Incinerat. – incineration; n.a – not applicable; BMW – biogenic municipal waste.
[a] Volume of digestates includes food waste treated in AD.
[b] Industrial AD treatment of sewage sludge was not accounted for in AD volumes but as dry sewage sludge in accordance with convention.
[c] Volumes of food wastes and digestates landfilled excluded.

Table 2
The summary of the various fuel production scenarios considered.

Feedstock	Production configuration	Hydrogen source
Digestates	Centralised	Grey hydrogen – steam methane reforming
Food waste	Decentralised	
BMW		Blue hydrogen – steam methane reforming with carbon capture
Sewage sludge		Green hydrogen – renewable electricity electrolysis
Co-processing		

liquid hydrocarbon "drop-in" fuels has been widely considered as a solution to current liquid fuels due to their compatibility with existing vehicle technologies and fuel distribution and storage systems, and non-competition with food crops. As a result, this work assesses the viability of drop-in fuel production from the UK wet waste biomass feedstocks, based on several production approaches, including different plant configurations and hydrogen sources. This study focuses on feedstock availability and utilisation in the UK, but general insights are applicable to other regions.

Biomass resources are generally considered sustainable, however, their utilisation can be limited due to the amount of biomass that can be sustainably removed and replenished. In the UK, the production of "fresh" biomass feedstocks (such as forestry) is limited, but waste biomass materials are generated in significant quantities. These waste biomass materials are often under-utilised or mismanaged, as significant quantities of waste, especially food waste and sewage sludge (SS), often end up on land and water bodies without proper recycling or energy recovery. Currently, 15 % of municipal solid waste (MSW) landfilled is food waste [9]. Landfilling of organic waste produces significant GHG emissions to the environment, with nearly 400 kg of CO_2eq/ton MSW released from landfills [10]. The current treatment method of sewage sludge is costly, about £200 per dry tonne of sludge [11]. Also, the use of dry sludge on land even after conventional treatment is subject to serious environmental and safety concerns like the contamination of water and the transmission of pathogens [12,13]. Hence, providing environmentally clean and cost-effective alternatives such as waste-to-energy could help improve wet biomass waste management. Generally, wastes are often managed by recycling, anaerobic digestion (AD), composting, incineration, land application and landfilling. The current wet waste generation volumes in the UK and the management techniques are shown in Table 1.

One of the processes in which wet waste biomass feedstocks can be managed is through hydrothermal liquefaction (HTL). HTL is a non-conventional way of treating and converting waste into energy in the form of synthetic liquid fuels. In HTL, sub-critical water is used to break down biomass feedstocks into three distinct products- biocrude, gas and char. Usually, combinations of high pressure of 5 – 30 MPa, medium temperature of 250 – 350 °C, and short reactor residence time of 5 – 90 min are employed on a wet biomass feedstock [24]. The use of wet biomass feedstocks in HTL avoids the high energy requirement of drying high moisture content biomass. A wide variety of biomass, such as sludge waste, algae and forest residue have been used in HTL, with some

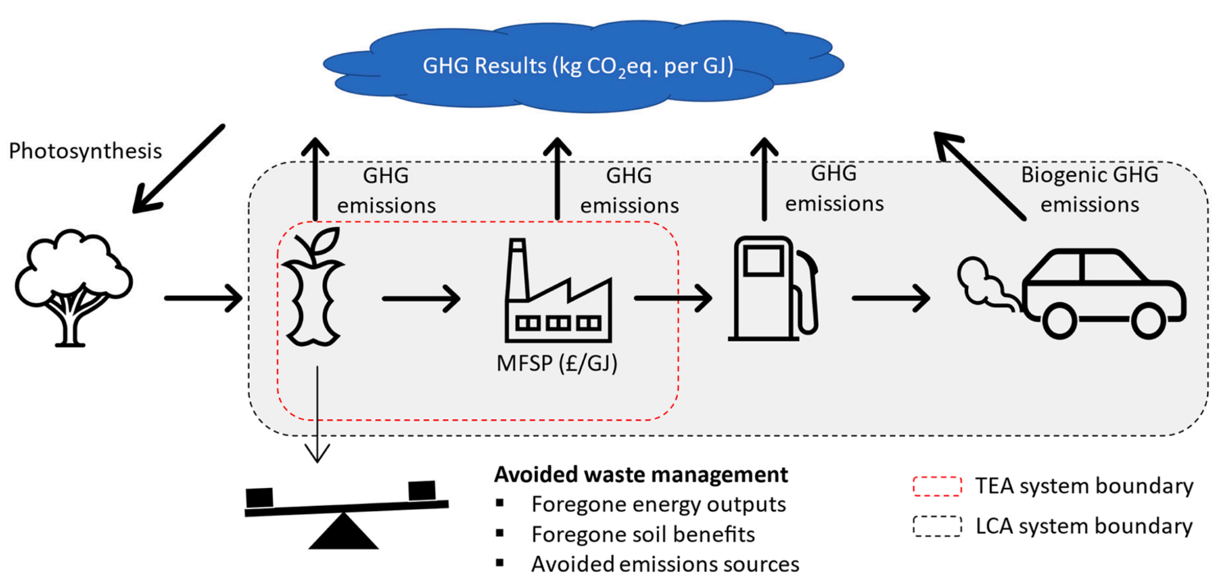

Fig. 1. System boundaries of the study for TEA and LCA.

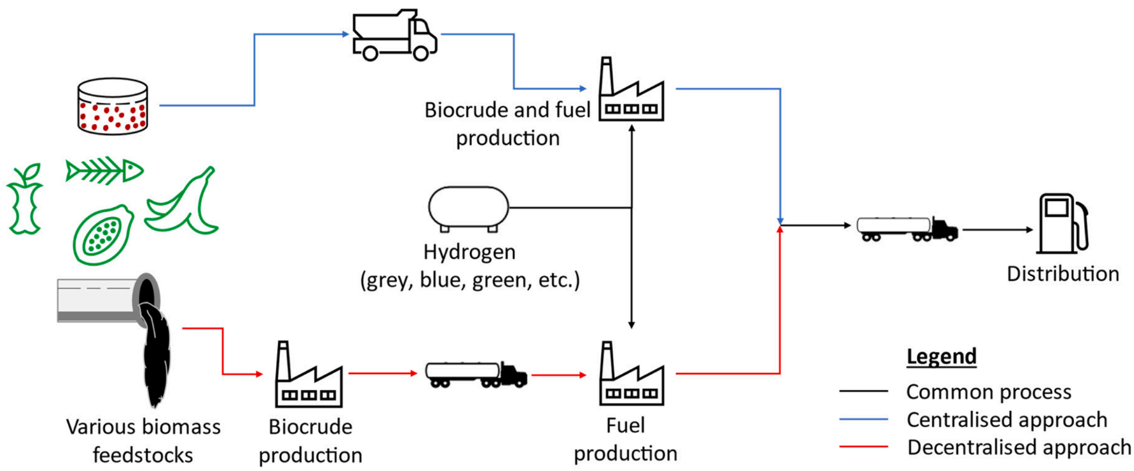

Fig. 2. Summary of HTL drop-in fuel production process.

Table 3
Properties and volumes throughput of waste feedstocks used in this analysis.

Wastes	Volume utilisation, Mtpa	Moisture content (%)	Energy content, dry basis (MJ/kg)	Feed cost (£/wet ton)	Biocrude yield (wt% per dry feed)	References
Digestates	10.7	90	18.0	−8 – 5 (0[a])	37.4	[20,27,58]
Food wastes	6.06	75	19.1	−15 – 15 (0[a])	39.6	[57,59,60]
Landfilled BMW	1.98	50	19.1	−25	39.6	[16,55,59,60]
Sewage sludge	30	95	15.7	−10	32.6	[11,27,61]
Co-processing	20.3	75	17.8	−6.6	37.1	This study

[a] Average value used in this analysis.
Note: (1) Negative feed prices represent the gate fees (excluding landfilled tax) charged by waste treatment plants.
(2) Co-processing case assumes utilisation of feedstocks based on competing use (75% digestates, 46% food waste, 30% BMW and 100% SS), see SI [Section 1.2.1] for additional details.
(3) Energy content on a dry basis obtained from mass and energy balance calculations using an energy efficiency based on Zhu et al. [42].

having moisture content of up to 96 % [25]. The wet treatment of biomass by HTL is reported to provide a significant advantage over other thermochemical processes like gasification and pyrolysis, due to the energy savings associated with the avoidance of drying [26]. Also, drop-in fuel via HTL can easily be integrated into existing infrastructures than other wet biomass technologies like AD [27,28]. However, like other thermochemical processes, it can be affected by issues such as tar formation and catalyst poison [29–32].

HTL technology is yet to be commercialised, as such the technical and economic parameters of the technology are yet to be fully understood with certainty. Previous techno-economic assessment (TEA) studies published employed various design and economic strategies including the location of the HTL biomass-to-liquid plant near the feedstock supply and the decentralisation of HTL of biomass feedstocks subject to a centralised conversion of biocrude to liquid fuels [27,33–39]. A study by Snowden-Swan et al. [27] claimed a high minimum fuel selling price (MFSP) of £30.16 per GJ [£0.96 per gasoline litre equivalent (GLE), 2021 cost basis] when a decentralised production approach was taken, where biocrude is produced at local sites with subsequent biocrude upgrading at a central processing facility. While this study aimed to determine the price of drop-in fuel from a decentralised production approach, the impacts of feedstock transportation over distances and of a centralised processing approach on the price of drop-in fuels were not considered. Transportation and production approaches can have a huge impact on drop-in economics, especially in the case of high moisture content feedstocks, which can lead to significant capital costs as well as transportation costs on the localised plant. As seen in studies by Aierzhati et al. [37], which evaluated the decentralised processing of food waste into biocrude, a high biocrude price with an MFSP of £22.16 per GJ was reported. This price is higher than the prices of drop-in fuels in the range of £13.54 – 19.08 per GJ from centralised approaches, as analysed by some authors such as Tews [40], Pedersen et al. [41] and Zhu et al. [42]. Zhu et al. [42] showed how the MFSP varied with the scale of the plant in a decentralised production operation, as MFSP increased when the scale of the plant at local sites decreased. For plant scale varying from 10 t/d to 2000 t/d, the MFSP varied from £65.24 to £16.00 per GJ, based on 2021 cost adjustment; however, the impact of feedstock transportation costs was not considered. Thus, it is not yet clear how the economy of scale will impact the MFSP of drop-in fuels using decentralised and centralised approaches as most TEA analyses study did not consider extensively the impact of transportation. Also, the availability of biomass feedstocks and where they will come from with regard to fuel production economics and sustainability have not been considered in most published TEAs. However, the sensitivity analysis of various TEA studies has shown that plant size, which can be strongly impacted by feedstock availability, impacts drop-in MFSP [27,33–39]. Additionally, some wet feedstocks like digestates and BMW have not been economically assessed for drop-in fuel production in previous literature. Furthermore, the use of green hydrogen and other hydrogen sources in the upgrading of biocrude to hydrocarbon fuels has not been evaluated in previously published TEA studies. Although current green or blue hydrogen production is negligible, in the future, green and blue hydrogen will likely make up a significant portion of the hydrogen market, and this could have a strong cost impact on fuel production using green hydrogen or other alternatives.

The GHG emissions and other environmental impacts of processes and products are evaluated, reviewed, and improved using life cycle

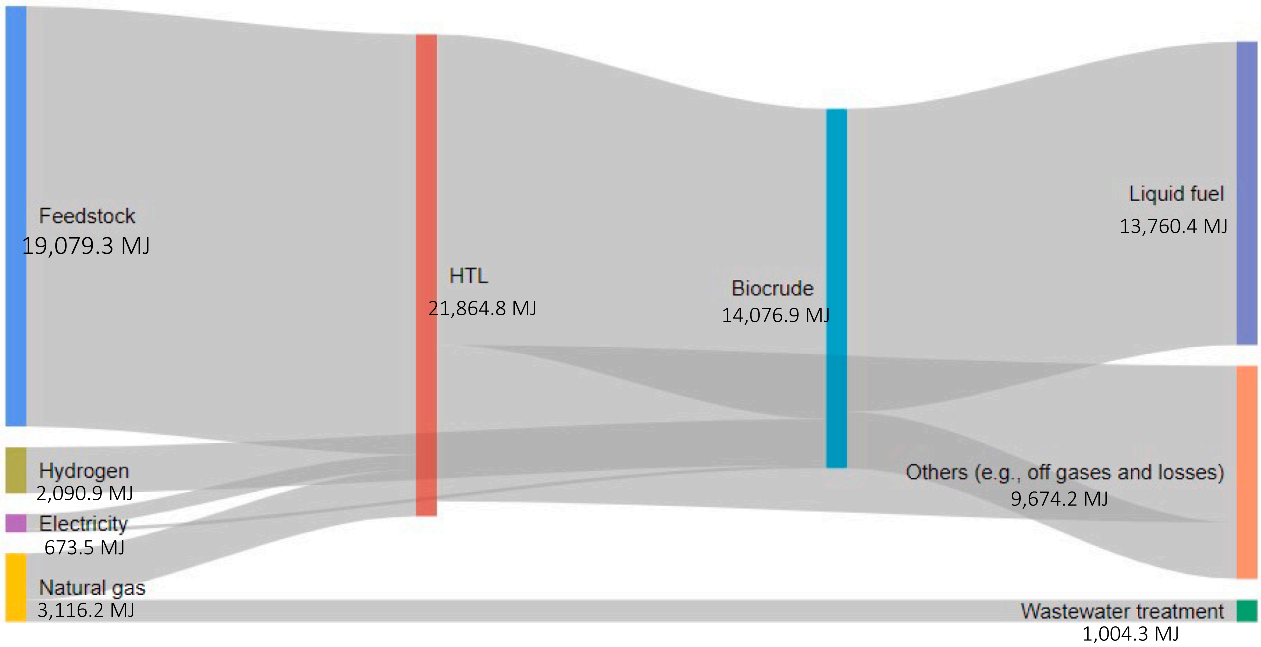

Fig. 3. Sankey diagram of the energy balance of the food waste HTL in drop-in fuel.

assessment (LCA). LCA, which is used to assess environmental impacts associated with new and emerging technologies, enables the comparison of drop-in fuel production approaches as well as drop-in fuels with conventional fuels. The production of drop-in fuels using various approaches, such as feedstock types, centralised vs decentralised production, and hydrogen sources, can have considerable impacts on the overall GHG emissions, as LCA is heavily dependent on energy and material requirements. Bennion et al. [43] revealed variations in the GHG emissions of producing drop-in fuels from microalgae using various production methods, with GHG emissions of −11.4 and 210 g CO_2eq per MJ for fuel production using HTL and pyrolysis, respectively. Many other studies have been published on the LCA of biofuels [44–50], and these analyses estimated the GHG footprint of drop-in biofuels in the range of –122 and 98 g CO_2eq per MJ. The upper band of the emissions numbers from some of the LCA studies are close to those of fossil fuels of 94 CO_2eq/MJ [51], as sustainability factors such as land use change [49] impact fuels' life cycle GHG emissions. Other factors which caused significant variations in the GHG emissions of these drop-in fuels are the fuel processing techniques [43,44,50], emission allocation methods (e.g., mass) [47], system boundaries [46], and LCA methodologies (attributional and consequential) [46,48], amongst others. The impact of different production approaches (centralised and decentralised) and hydrogen sources on GHG emissions have not been explored in current published literature, and so, it is not known to what extent the use of these approaches would affect drop-in fuel's GHG emissions. In addition, despite previous work evaluating the GHG emissions from various feedstocks, the HTL of various available wet feedstocks in the UK which are distributed nationally has received little to no attention, despite wet feedstocks such as food waste, digestates, and SS constituting a significant portion of UK biomass resources.

The development and deployment of drop-in fuels from biomass feedstocks are highly affected by the MFSP, the life cycle GHG emissions and the availability of biomass resources. This work aims to study the variations in factors such as feedstock types, plant configuration (decentralised vs centralised production approach) and hydrogen sources (grey, blue and green) on the cost of producing drop-in fuels. As a result, four wet feedstocks – digestates, food waste, landfilled biodegradable municipal waste (BMW) and SS – which are readily available in the UK are studied. Also, it estimates the amount of drop-in fuels which can be sustainably produced from these UK wet feedstocks. In addition, the GHG emissions associated with producing drop-in fuels, from wet feedstocks and the changes in plant operations, are estimated to determine the impact of diverting and using these feedstocks in drop-in fuel production.

2. Methodology

TEA and LCA methodologies are developed to evaluate drop-in fuel production from wet biomass feedstocks in the UK context under various production configurations (see Table 2). The TEA methodologies developed were based on a review of various important publications in the literature relating to HTL such as those from NREL HTL pilot plant models [27,39,42] and Aierzhati et al. pilot plant models [37]. Therefore, the TEA costing was done in Microsoft Excel to allow for the adoption of cost models related to HTL or other biomass-to-liquids rather than the generic costing model that is available in Aspen Plus or other costing models. Also, the LCA was done in Microsoft Excel with data obtained from various literature including Ecoinvent database. This allows data that is closely associated with the UK market to be considered in the analysis where available. The details of the specific methodology for the TEA and LCA analyses including their drawbacks are presented respectively in this section.

The main scenarios evaluated include the impact of various feedstocks, plant configurations and hydrogen sources. Four wet biomass feedstocks – digestates, food waste, biodegradable municipal waste (BMW) and SS – produced across the UK are considered, in addition to a co-processing scenario where all four feedstocks are processed to fuels at

Table 4
The cost and emission factors of hydrogen sources.

Hydrogen source	Cost ($/kg)	Emissions factor (g CO_2eq/MJ)	References
Grey hydrogen	0.7 – 1.6 (1.15*)	83.6	[65,66]
Blue hydrogen	1.2 – 2.1 (1.65*)	21.4	[65,66]
Green (renewable electricity) hydrogen	3.2 – 7.7 (5.45*)	0.1	[65,66]

* Average cost values, which were used in the base analysis

Table 5
Some key assumptions for the nth plant economic analysis, based on reports by Zhu [42], Snowden-Swan et al. [27], Jones et al. [39] and Aierzhati et al. [37].

Key assumptions	Value
Analysis type	nth
Based year	2021
Discount rate (r)	10 %
Inflation rate	2 %
Plant life	22 years
Income tax rate	35 %
Working capital cost	5 % of FCI
Capital spending	2/4 in 1st year, 1.6/4 in 2nd year, 0.4/4 in 3rd year
Depreciation schedule	10-years linear depreciation
Construction period	2.5 years
Post-construction start-up time	6 months
Plant salvage value	No value
On-stream factor	8000 h

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Table 6
Cost factors for direct and indirect cost analysis, based on Snowden-Swan et al. [27].

Direct costs	
Component	**% of total installed cost (TIC)**
Buildings	4 %
Site development	10 %
Additional piping	4.5 %
Total direct costs (TDC)	**18.5 %**
Indirect cost	**% of TDC**
Prorated expenses	10 %
Home office & construction fees	20 %
Field expenses	10 %
Project contingency	10 %
Startup and permits	10 %
Total indirect cost	**60 %**
Fixed capital investment (FCI)	**TDC + total indirect cost**
Working capital	**5 % of FCI**
TCI	**FCI + working capital**

the same site. The distribution of the feedstocks across various regions of the UK including East of England, East Midlands, London, North East, North West, N. Ireland, Scotland and Wales, is also considered in the analysis scenarios. Also, scenarios which are based on two production configurations – centralised and decentralised approaches – were evaluated. The centralised approach examines the processing of biomass into biocrude and subsequently into drop-in fuels within one central facility, while the decentralised approach considers the conversion of biomass feedstocks into biocrude at localised locations followed by the upgrading of biocrude into drop-in fuels at a central facility, as shown in Fig. 2. Lastly, scenarios considering different sources of hydrogen – grey, blue and green hydrogen – are also evaluated. Table 2 summarised the various scenarios considered in the analysis.

The key scenarios used to evaluate the various feedstocks are labelled: “Centralised, Grey H2” representing Centralised fuel production using grey hydrogen; “Decentralised, Grey H2” for Decentralised fuel production using grey hydrogen; “Centralised, Blue H2” for Centralised fuel production using blue hydrogen; “Decentralised, Blue H2” for Decentralised fuel production using blue hydrogen; “Centralised, Green H2” for Centralised fuel production using green hydrogen; and “Decentralised, Green H2” for Decentralised fuel production using green hydrogen. See Supplementary Information (SI) Section 1.1 for more details on scenario configurations.

2.1. Flow diagram

The process flow diagram of the fuel product is taken to start from the point of waste generation and ends in the fuel combustion stage. The system boundaries for the TEA and LCA analyses are shown in Fig. 1, see SI for more details on the individual process stages [Section 1]. A cradle-to-gate TEA approach and well-to-wheel LCA approach were taken for the study, as GHG emissions associated with the infrastructures (process plant and vehicles) were not considered. The mass and energy balances of the fuel’s production stages are used in the computation of cost and fuel production rate, while GHG emissions are computed from the key supply chain and fuel production approaches.

Table 7
Variable OPEX parameters.

Variable OPEX	Value	Reference
Feedstock, £/wet ton	−25–0	[11,20,55–57]
Feed transportation, £/km/ton waste (2021$)	0.26	[62]
Natural gas, £/100 scf UK prices (2021)	1.16	[67]
Electricity, £/kWh (2021)	0.13	[68]
Solid disposal, £/ton (2021)	16.66	[38]
Quicklime (CaO), £/ton biocrude input (2021$)	103.73	[69]
Process water, £/m^3 (2021)	0.44	[70]
Biocrude catalyst, £/kg feed input (2021)	0.0495	[27,71]
Hydrotreater catalyst, £/gal biocrude input (2021)	0.0087	[27]
Hydrocracking catalyst, £/gal biocrude input (2021)	0.0006	[27]
Cooling tower chemical, £/gal biocrude input (2021)	0.0003	[27]

2.2. Fuel production

In the fuel production process, which can be divided into biocrude production and upgrading, the feed is converted into biocrude and then subsequently upgraded into liquid fuels, as shown in Fig. 2. Biocrude is assumed to have a moisture content of 4 %, based on the model by Snowden-Swan et al. [27]. The fuel production model was based on the models by Zhu [42] and Snowden-Swan et al. [27]. In the first step, the feedstocks, which are the waste materials, are pumped and preheated in the pre-treatment unit of the plant before further heating in a fired heater using a hot oil system. The heated and pressurized feedstock is then fed to the HTL reactor where the HTL of the feedstock into biocrude is done. Some side products are produced in the HTL reactor which includes aqueous solution, solids and gas. The biocrude yields of the various feedstocks are shown in Table 3. The biocrude produced via HTL can be processed into liquid fuels- renewable gasoline and diesel- through hydroprocessing, in a way similar to that done in conventional petroleum refineries. Hydrogen from various sources can be used to break down the complex organic molecules of the biocrude during hydroprocessing [27,40,42,52]. In addition, sulphur, oxygen, nitrogen and metals are removed from the biocrude through this process. The liquid fuel yield is taken to be 77.9 wt% of biocrude, based on Snowden-Swan et al. [27]. A detailed description of the process model including

Table 8
Fixed OPEX parameters [72].

Fixed OPEX	Rate (£M/yr)	Number, plant size in dry tpa		
		Plant size < 0.5	0.5 < plant size < 5	Plant size > 5
Plant manager/ engineer	0.109	1	1	1
Plant engineer	0.052	0	1	1
Maintenance superv.	0.042	1	1	1
Lab manager	0.041	1	1	1
Shift supervisor	0.036	0	1	1
Lab technician	0.030	0	1	2
Maintenance tech.	0.030	0	1	2
Shift operators/ supervision	0.036	3	3	6
Clerks and secretaries	0.027	1	1	1
Overhead & maintenance	90 % of labour & supervision			
Maintenance capital	3 % of TIC			
Insurance & taxes	0.7 % of FCI			

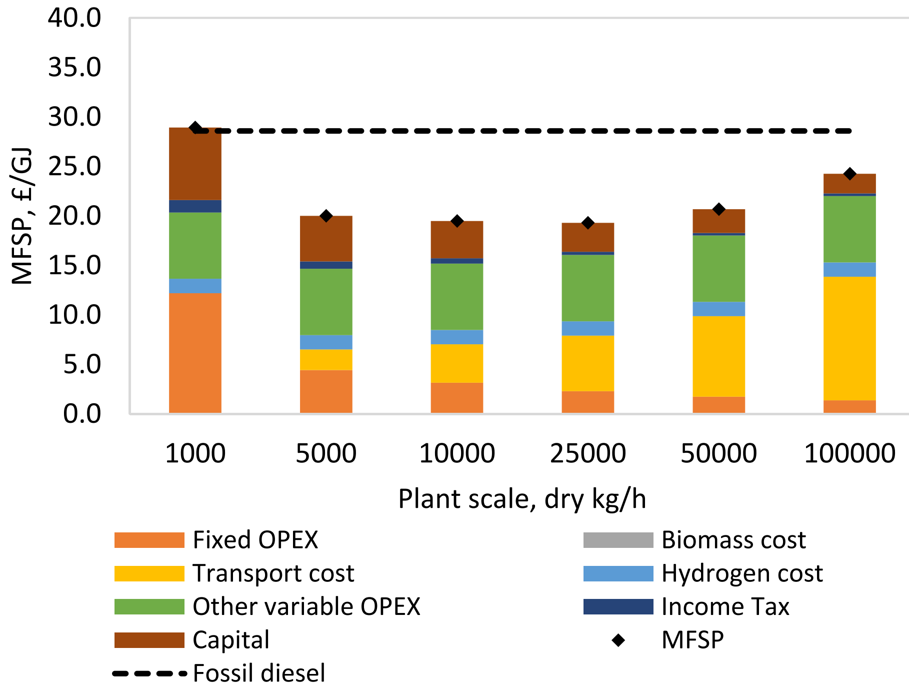


Fig. 4. Effects of plant scale with price breakdown for fuel production from food waste over England, based on Centralised, Grey H2 scenario and assuming the plant is located in Leicester, UK. The MFSPs are displayed in comparison with fossil diesel's price of £28.58 per GJ, based on 2021 wholesale prices of diesel published by RAC [76]. Prices in this study are driven by factors such as feedstock cost (which is zero or negative in some cases) and product yield. Royal Society [77] published biofuel prices from HTL of forest residues in the range of £20.2 – 29.1 per GJ.

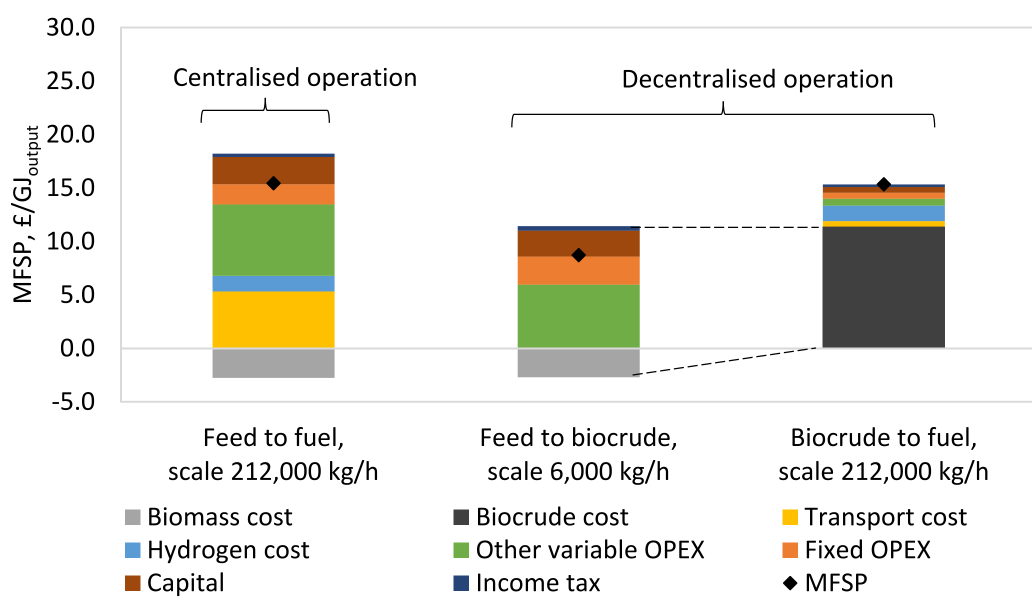


Fig. 5. Comparison between centralised and decentralised approaches for the co-processing of the feedstocks using grey hydrogen for upgrading. The graph shows the price from the conversion of biomass to fuel along various supply chains [units of MFSP in £/GJ_{fuel} for final fuel output and £/$GJ_{biocrude}$ for biocrude output].

biocrude production and upgrading and wastewater treatment can be found in SI [Section 1.2.4].

2.3. Material and energy balance

The material and energy balance of the fuel production processes were based on the biocrude yields shown in Table 3 and the process model by Snowden-Swan et al. [27]. HTL is an endothermic process which becomes slightly exothermic above certain temperatures (e.g., at 240 °C for cellulose, glucose, and wood HTL) [53,54]. As a result, the off-gases and char produced from the HTL of the feedstocks are used to provide the heating demand of the HTL. Additionally, natural gas is used to supply the remainder of the heating requirement of the process. Also, another place in which natural gas was consumed is in the thermal oxidation (THROX) unit for wastewater treatment. In the THROX unit, ammonia/air stream from the HTL aqueous stream is treated, as ammonia and organics are catalytically combusted to CO_2, nitrogen and water. No heating from natural gas is assumed to be used in the upgrading plant, as off-gases and heavy fuel oil from the upgrading unit are utilised, in line with the process model by Snowden-Swan et al. [27].

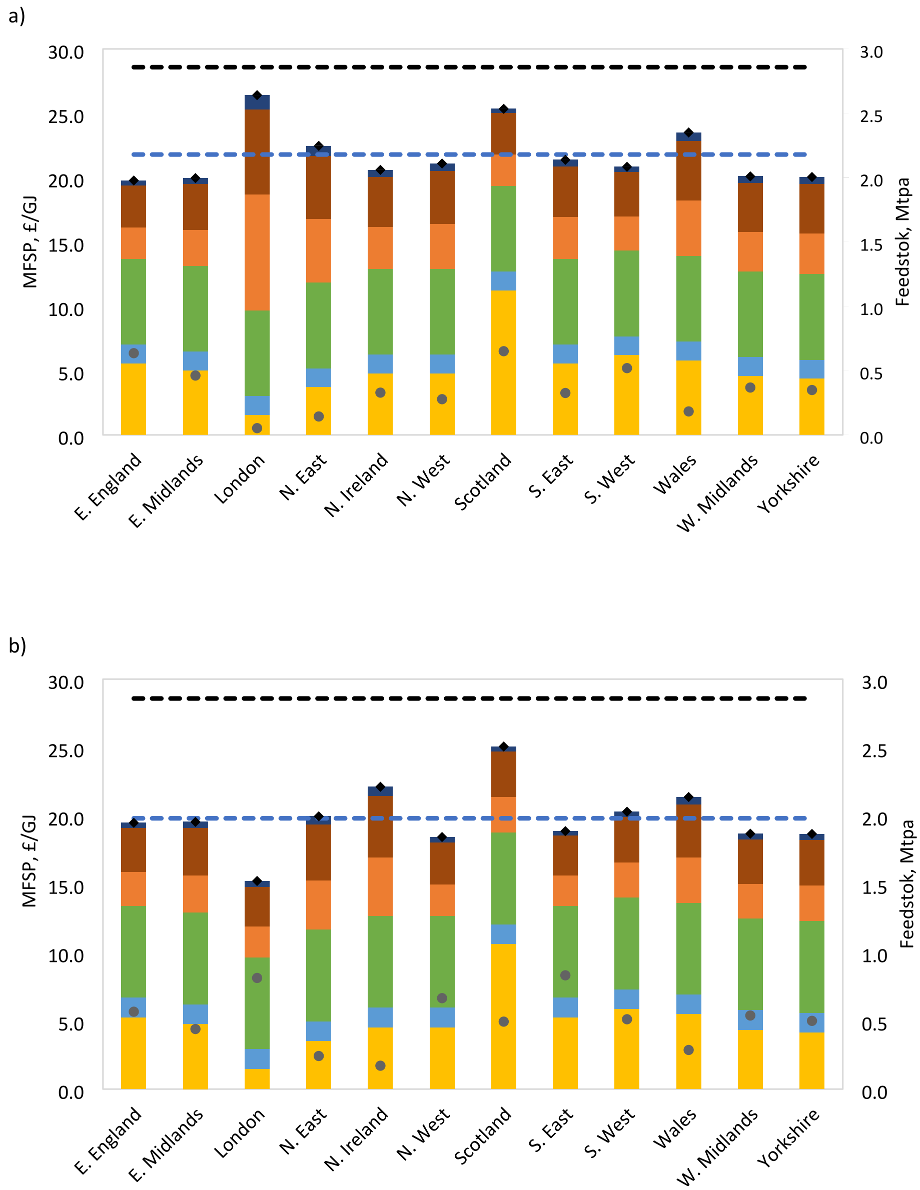


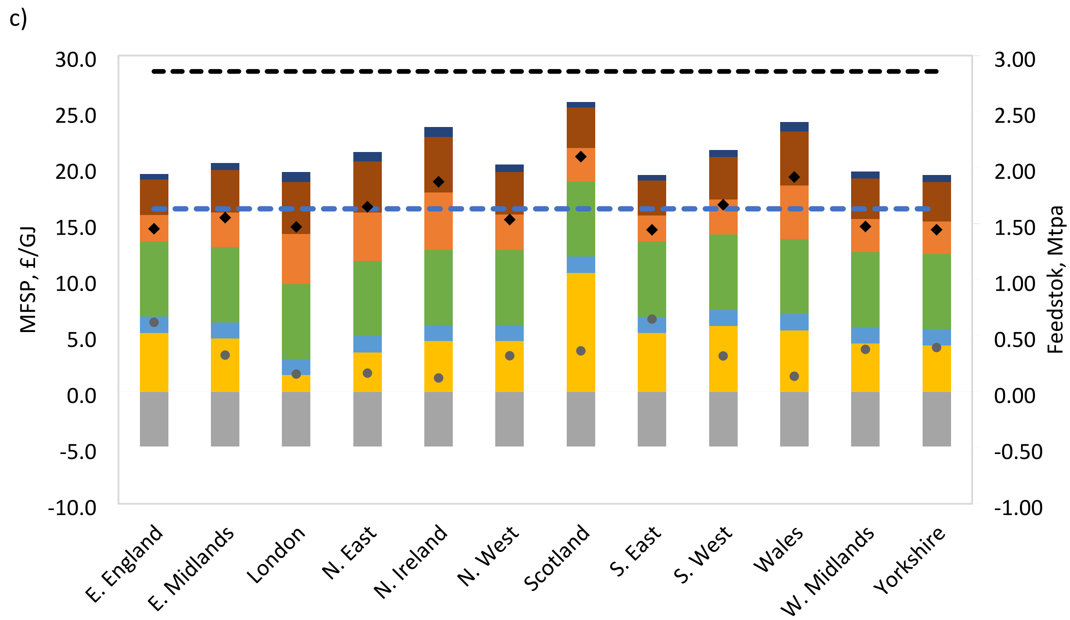
Fig. 6. UK regional distribution of MFSP for HTL processing of (a) digestates (b) food waste (c) BMW (d) SS and (e) co-processing the digestates, food waste, BMW and SS, based on the Centralised, Grey H2 scenario.

Fig. 3 shows the energy balance of food waste processing. More details of the mass and energy balance of the various feedstocks and processes including hydrogen and natural gas consumption are shown in SI [Section 1.2.4].

2.4. Feedstocks and current treatment

The feedstocks are assumed to be generated and collected in waste collection sites such as AD plants, councils' waste collection sites, and wastewater treatment facilities. Feedstocks which have high moisture contents, as shown in Table 3, are dewatered to 75 % moisture content at the waste collection sites, as this is the current practice with digestates and SS. They are then transported to the fuel production plant in the centralised approach or converted into biocrude at a local biocrude plant which is co-located with the waste generation or collection sites in the decentralised approach, see SI [Section 1] for details. Food waste and BMW do not need dewatering, due to their lower moisture content. However, it is assumed that BMW is diluted with water into 75 % moisture content at the biocrude production plant, to improve its pumpability prior to biocrude production due to pumping demands [27].

A key variable which has a strong impact on this economic analysis is the cost of the various waste biomass feedstocks, which varied from average values of -£25 to £0, based on data on feedstock cost/gate fees published in the literature [11,20,55–57]. Current feedstock cost/disposal gate fees are shown in Table 3, where a negative feedstock cost value indicates a payment would be received for accepting the feedstock. Because the prices of these feedstocks vary depending on factors such as season and location, there is a high level of uncertainty in the feedstock cost. For example, the price of food waste is reported to vary between -£15 and £15, based on the time of the year [57]. This variation leads to some uncertainty in the analysis. Also, it is not known what the



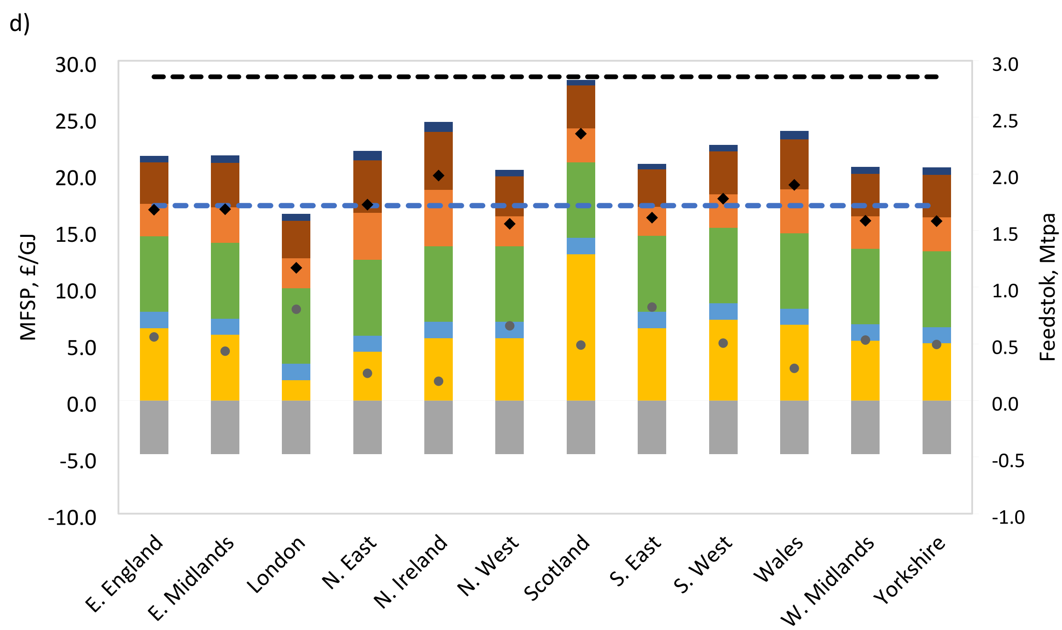


Fig. 6. (*continued*).

prices of these biomass feedstocks in the future will be, as other potential competing uses come into operation. Therefore, a sensitivity analysis based on the expected range of feedstock prices was done to determine the impact of changes in feedstock prices and presented in SI Section 4.4.

Current feedstock treatment methods are considered in the analysis of the GHG emissions, as utilisation of the wet feedstocks in fuel production can lead to direct and indirect impacts along the waste supply chain. Thus, several factors associated with using these feedstocks such as foregone electricity from current waste-to-energy technologies, foregone fertiliser credits from diverting wastes used as nutrients, and avoided treatment emissions from landfilled waste diversion were also considered, see SI [Section 3.1] for details. Hence, a consequential approach to the LCA is taken.

Because there is a trade-off between the economy of scale of facility size and feedstock transportation costs, it was assumed that the conversion of the waste feedstocks into fuel is done on a regional basis, with one central facility per region. The scale of the central facility in any region was based on the availability of the wet biomass feedstocks in the region. The trade-offs between facility capacity (economy of scale) and feedstock transportation cost were investigated to ensure this assumption is appropriate, see Section 3.1.1 for the impact of facility size and SI Section 4.4 for sensitivity analysis. It is important to note that this work does not propose sites where these facilities will be located nor consider the exact location where the waste sources are currently located. However, it attempts to estimate where these wastes will come from, based on the nature of the generation of these wastes and published national data.

In the centralised approach, the area of each region was used to determine the average transportation distance. Each region of the UK was assumed to be a circle, where the area-weighted average radius

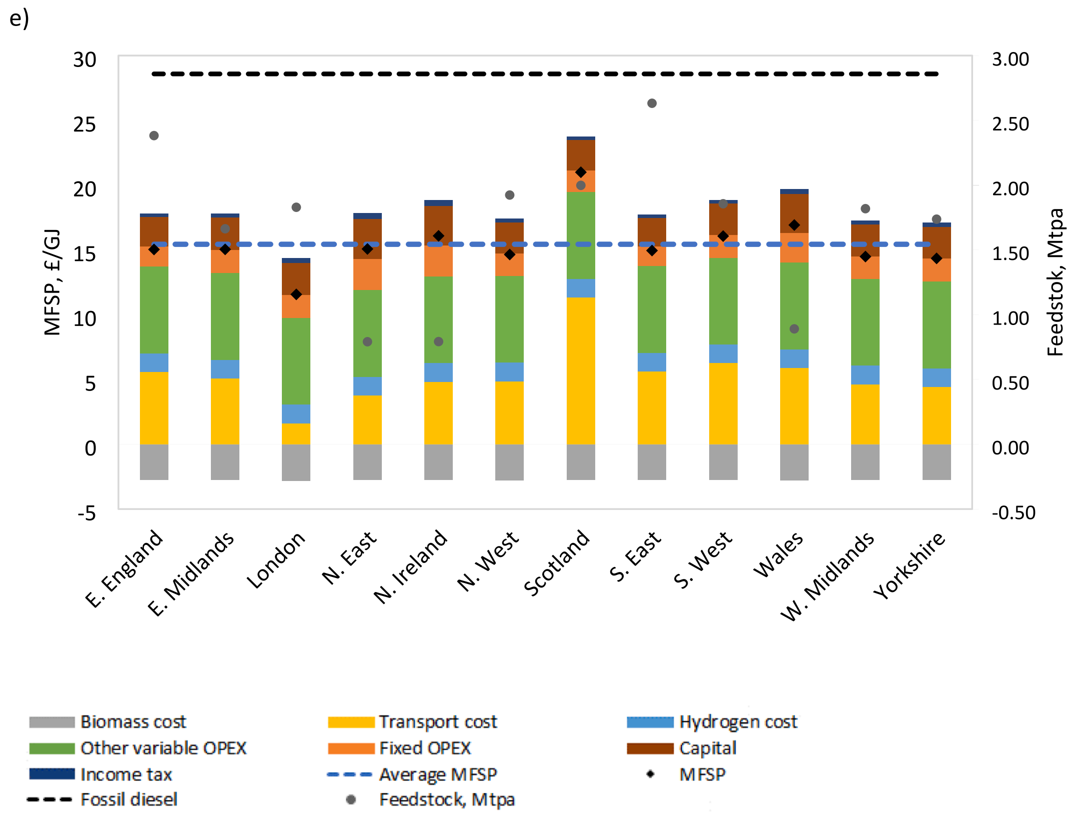


Fig. 6. (*continued*).

(two-thirds of the radius) of the circle is the average transport distance, see SI Section 1.2.2 for details. Since the various regions of the UK are not perfect circles, a sensitivity analysis is performed on the transportation distance to determine the impact of the uncertainty of the transportation distance on the MFSP, see SI Section 4.4 for details. Also, regional transportation cost is factored in the plant economics as this regional transportation of wet waste feedstocks may not be what is currently practised. The cost of transportation was estimated at £0.26 per km per ton of waste, based on reported data by Kocher [62].

In the decentralised approach, the transportation of biocrude produced from the feedstocks to a central drop-in fuel production plant is assumed to be part of the operating cost of the central plant and the distances involved are the same as in the central approach. However, in the decentralised processing of the feedstocks at local locations into biocrude, it was assumed that the local biocrude plants are co-located with the waste collection or treatment sites in local areas so that the transportation distance to the biocrude plant is negligible. Also, the assumption of a negligible impact of transportation of waste into biocrude in the local processing of wet feedstocks is supported by current applications, as currently waste gathering or collection is done via councils for food waste and the waste generator (including for digestates) pays for the transportation of the waste locally.

2.5. Hydrogen source

The hydrogen for hydroprocessing of biocrude can be obtained from a variety of sources such as natural gas and electricity. Grey hydrogen is the most widely available hydrogen currently, accounting for 95 – 99.5 % of the world's hydrogen supply [63,64]. The production of green and blue hydrogen is highly uncertain currently, as they make up only a negligible portion of world hydrogen production volumes, less than 1 %. However, scenarios considering the use of green and blue hydrogen are also analysed, assuming these will become available in the future, in line with current policies on hydrogen. The average of the cost values reported by IEA [65] and the emissions factors reported by BEIS [66], as shown in Table 4 were used in this study.

2.6. Economic analysis

The key approach adopted in the economic model for the estimation of the price of drop-in fuels is highlighted in this section. Data, such as equipment costs and costing methodology by authors such as Snowden-Swan et al. [27], Jones et al. [39] and Aierzhati et al. [37] were adopted for this analysis.

2.6.1. General assumptions and approach

The economic assessment was based on the nth plant costing analysis methodology, which represents a typical future costing, and this does not account for additional expenses such as equipment redundancies and longer start-up associated with designing and operating a first-of-a-kind plant [27,42], see Table 5 for details.

The cost analysis is performed based on the 2021 constant Pound Sterling (£) basis, therefore, equipment costs are updated to 2021, where necessary, using the chemical engineering plant cost indices. Conversions between £ and the US dollar ($) were made on the average 2021 exchange rate of £1 to $1.3496. The production costs determined are adjusted in line with inflation, and these are used to determine MFSP, based on a 25-year discounted cash flow rate of return, to assess discounted fuel selling cost and enable comparison with current fuel prices for potential financial viability.

2.6.2. Capital cost

Capital costs are usually based on estimates, as it is somewhat difficult and unrealistic to get the exact capital costs for a specific plant capacity, especially one which has not been commercially developed like HTL plants. In the estimation of the capital cost, original equipment costs are scaled to the current equipment/plant size, using a scaling factor (n) which is based on Equation (1), where n the scaling factor is

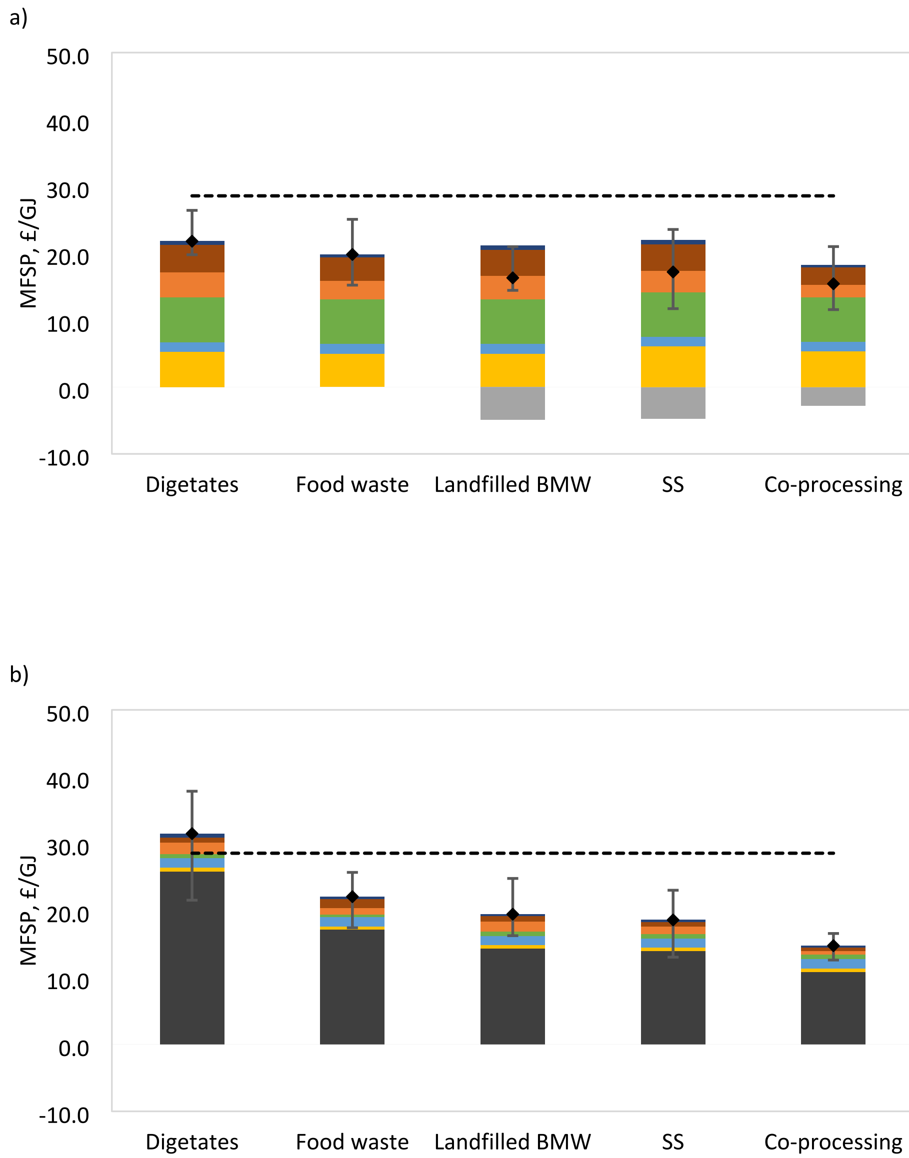


Fig. 7. Average MFSP distribution of various feedstocks using (a) Centralised approach based on grey hydrogen use (b) Decentralised approach based on grey hydrogen use (c) Centralised approach based on blue hydrogen use (d) Decentralised approach based on blue hydrogen use (e) Centralised approach based on green hydrogen use (f) Decentralised approach based on green hydrogen use. Error bars represent the variations in MFSPs across the various regions of the UK. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

typically between 0.6 and 0.7, based on reported values by Jones [39] and Snowden–Swan [27], see SI Section 2 for details. Adjustments were also made to account for the installation factors of various equipment.

$$\text{Scaled equipment cost} = \text{Cost at original scale} \times \left(\frac{\text{Scale up capacity}}{\text{Original capacity}}\right)^n \tag{1}$$

Furthermore, the capital costs of scaled equipment, which are typically in their respective cost years, are adjusted to the cost analysis base year of 2021 using the indices from the Chemical Engineering (CE) Index, by applying Equation (2)

$$\text{Cost in 2021 USD} = \text{Equipment cost in quote year} \times \frac{\text{2021 index}}{\text{Quote cost year index}} \tag{2}$$

The total installed cost (TIC) sums the various equipment cost, and this is used to calculate the total capital investment (TCI), which is the sum of the direct and indirect costs obtained using estimates based on the TIC.

Table 6 shows the various cost factors used for direct and indirect cost analysis.

2.6.3. Production costs

The production cost also referred to as operating cost can be divided into fixed operating expenditure (fixed OPEX) and variable operating expenditure (variable OPEX) and is calculated on an annual basis from the cost of operating the plant. The fixed OPEX consists mainly of the labour cost, while the variable OPEX consist mainly of the feedstock and transportation costs, raw materials costs, and utilities. These costs were adjusted each year to account for inflation. An inflation rate of 2 % per year was used in this analysis.

The variable and the fixed OPEX basis of this analysis are shown in Table 7 and Table 8, respectively. The variable OPEX calculations were based on the literature data on the consumption of raw materials, chemicals, wastes, and utilities. The costs of the feedstocks used are based on Table 3.

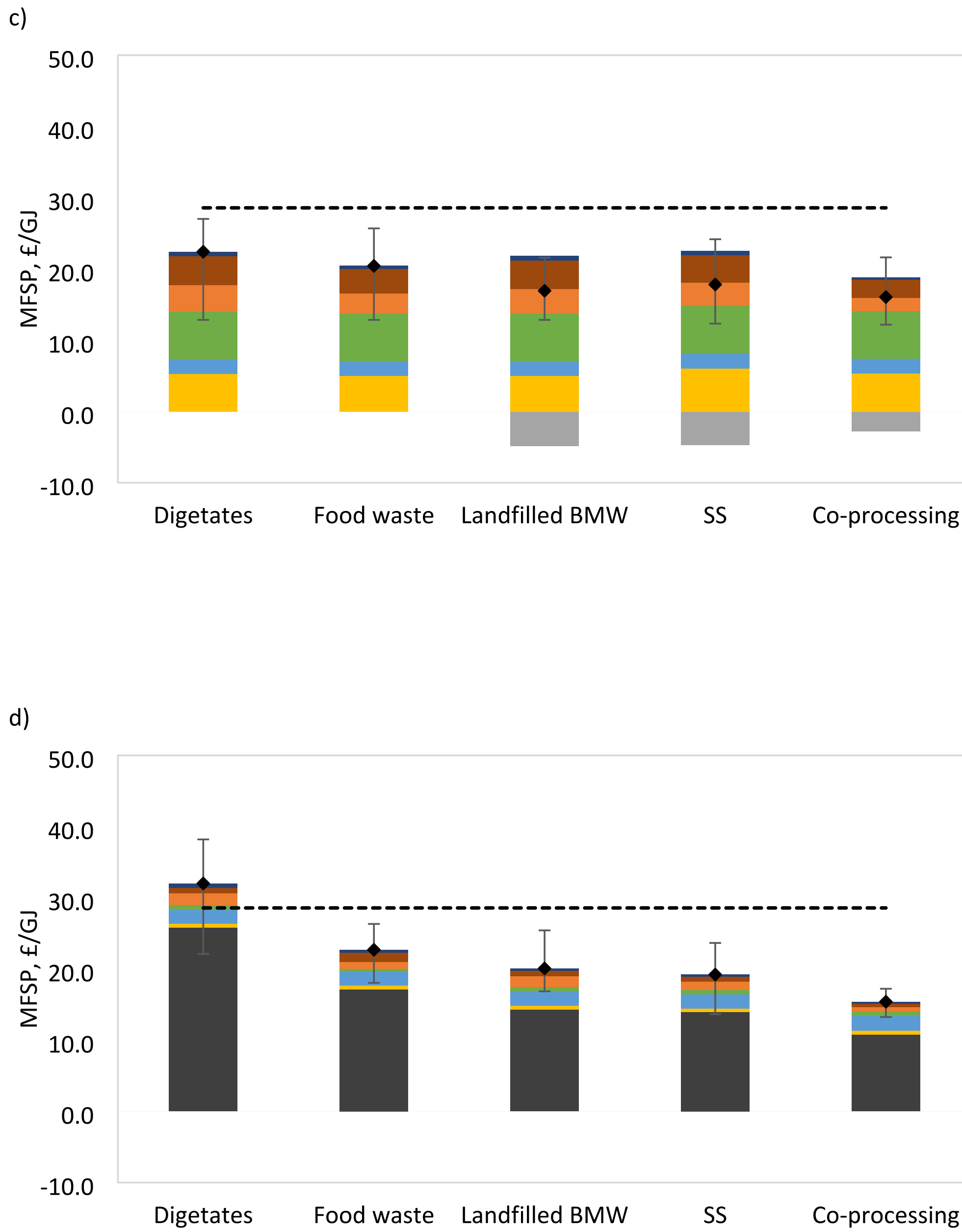


Fig. 7. (*continued*).

2.6.4. MFSP

A 25-year discounted cash flow rate of return, at a rate of 10 %, was used in the evaluation of the MFSP. The MFSP is the selling price of the fuel product at the factory gate that makes the net present value (NPV) of the project equal to zero, over the plant life. It was calculated based on Equation (3), where CF is the cash flow in any year (t) and r is the discount rate. In addition, economic cost factors that are found in Table 6 such as depreciation and tax were factored into the price analysis. The MFSP calculated is referenced to the factory gate as the sales point, hence, the MFSP here does not include other downstream costs factors such as the fuel distribution cost and retailer's margin.

$$NPV = \sum_{t=1}^{n} \frac{CF_t}{(1+r)^t} - CF_0 \quad (3)$$

2.7. LCA

The drop-in fuel life cycle, as shown in Fig. 1, consists of various stages such as the waste generation and the fuel production stages, and this defined the boundary conditions of the work. Various inventory data and emission factors from the literature were used to estimate the life cycle emissions involving the fuel production and utilisation processes, see SI Section 3 for details.

2.7.1. LCA goal and scope

This study aims to estimate the life cycle emissions of producing drop-in fuels using various UK wet feedstocks and fuel production scenarios. The analysis undertaken is similar to well-to-wheel LCA analysis where the emissions from the point the feedstocks become waste (raw materials) to the point the fuel is combusted are evaluated. Emissions

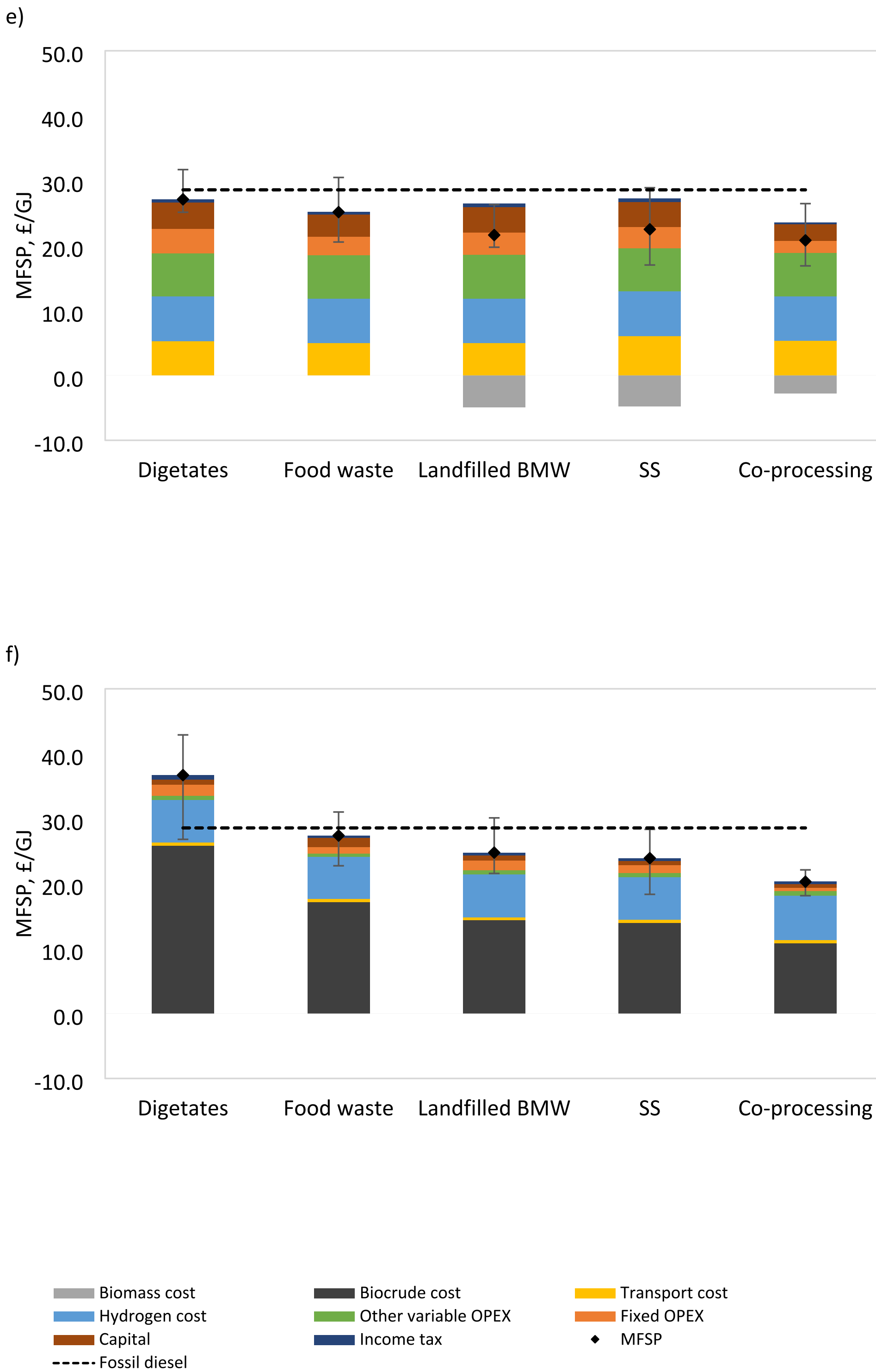


Fig. 7. (*continued*).

associated with building the infrastructures (process plant and vehicles) were not considered. Due to the nature of the various biomass feedstocks being waste materials, it is taken that the waste biomass materials carry a GHG emission rate of 0 g CO_2eq/kg at the point of collection. The functional unit of this study is 1 MJ of liquid (gasoline and diesel) fuel produced, while the GHG emissions are reported in kg CO_2eq per GJ fuel. The emissions in g CO_2eq per MJ of fuel can be converted to kg CO_2eq per GJ by using a factor of 1. The LCA approach is to determine both attributional and consequential GHG emissions of the drop-in fuels. Attributional LCA analyses the emissions impact directly associated with a process or product. While consequential LCA analyses the emissions impacts which may, directly and indirectly, arise from using a process or product. Generally, attributional LCA assigns an estimate of the share of global environmental burdens that belong to a process or product, while consequential LCA assigns an estimate of how a process or product affects the global environmental burdens [73]. In this study, emissions directly in the fuel production supply chain which were evaluated include emissions from feedstock transportation and fuel production, while emissions taken to come indirectly from the fuel including foregone electricity credits and foregone fertiliser credits were also

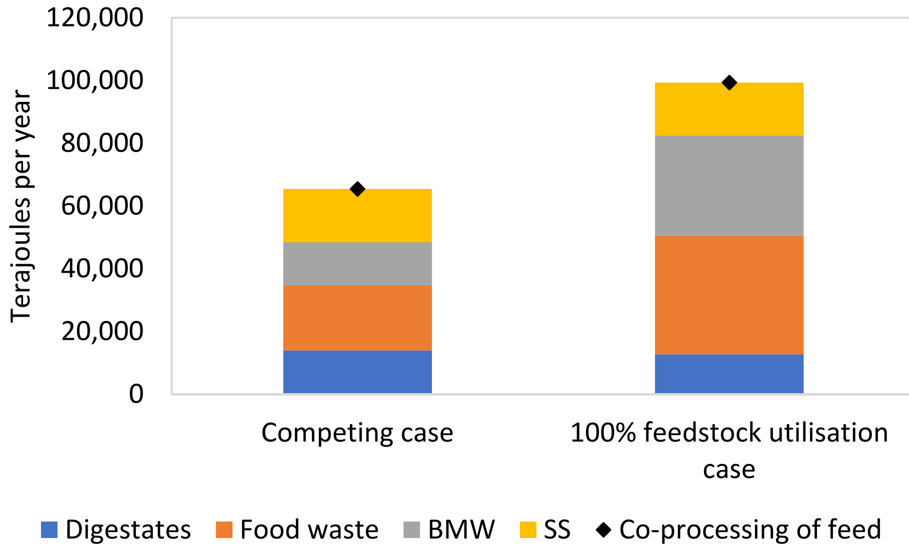


Fig. 8. The total drop-in fuel production volumes from the UK wet feedstock, based on the competing and 100% feedstock utilisation case.

evaluated. Other details on the assumptions about the LCA methodology can be found in the SI file [Section 3].

2.7.2. Avoided waste management

In the consequential LCA analysis, the indirect emissions associated with the diversion of the feedstocks from the existing waste management systems (shown in Table 1) were considered. The existing waste management systems' potential benefits and emissions that were analysed include electricity (for AD and incineration processes), soil benefits from fertiliser and carbon sequestration (for AD and composting processes), and avoided waste treatment emissions (for composting and landfilling processes). These benefits and emissions can be foregone when these feedstocks are used in fuel production. See Section 3.1 in the SI for more information on the indirect emissions factors of these processes.

2.7.3. Life cycle emissions outlook

Life cycle emissions outlook over the various fuel production approaches were analysed based on the BEIS [74] electricity emissions outlook for the UK. The BEIS outlook takes into account the energy policies in the UK in estimating future emissions. The electricity emission and heat emission factors, hydrogen emissions factors and sodium carbonate emissions factors were adjusted, based on BEIS publications [66,74], to develop the GHG emissions outlook in this study, with the year 2021 as the base year. Also, it was assumed that grid electricity instead of natural gas will be used to make up the heating demand of the fuel process from 2030, as heat generation shifts from natural gas to a lower carbon option. It is important to note that potential changes in competing uses in the future were not accounted for in the analysis.

2.7.4. Inventory data

The summary of the inventory data used in the LCA is outlined in SI Table 22. The data from the literature were collected on the various stages of fuel production ranging from feedstock collection and transportation to fuel transportation and distribution.

3. Results and discussions

3.1. Economic analysis

The results of the cost estimation for the various feedstocks, plant configurations and hydrogen sources are presented here. The impact of design and operating parameters such as plant scale and feed distribution are evaluated with respect to the MFSP. In addition, a sensitivity analysis of the key parameters impacting fuel production is also presented.

3.1.1. Plant scale

The effect of plant scale on the MFSP was evaluated for food waste processing as shown in Fig. 4. The MFSP decreases with an increase in plant scale as the economy of scale favours larger plant capacity until the transportation cost of the feedstocks becomes significantly dominant, then an increase in the MFSP is obtained with an increase in plant scale. At smaller production scales, the breakdown of the cost components of the MFSP, as shown in Fig. 4, is dominated by capital and fixed OPEX costs. However, as the production scale increases, the dominant cost components changed to transportation costs and other variable costs. As it is previously known, MFSP is highly dependent on the production scale, due to economy of scale, as reported by several authors such as Zhu et al. [42] and Snowden-Swan [75]. However, with wet biomass feedstocks being high in moisture content and consequently exhibiting very high transportation costs when long distances are involved, increasing the scale of the fuel production plant makes it less economical above certain plant capacity. Thus, a certain scale of operation is needed to reconcile the huge cost of transporting very high moisture content feedstocks. Fig. 4 suggests the optimum plant capacity for the centralised processing of the various wet biomass feedstocks is within 5,000 to 50,000 kg/h, as the MFSPs from these plant scales yield the best fuel prices. Furthermore, the scales of these drop-in fuel plants are expected to play a crucial role in the prices of fuel produced using either a centralised or decentralised production approach, as both approaches may vary with plant scale and subsequently in the MFSP. This can be seen in Fig. 5 when the plant scale for the decentralised conversion of the wet biomass to biocrude at localised locations is above a certain capacity, about 6000 kg/h. In this configuration, the decentralised case for the wet biomass feedstock becomes more economically favourable than the centralised approach. Thus, the benefit of using either a centralised or decentralised approach in wet biomass processing depends on the local availability of the feed and the distribution of the feedstock across locations.

3.1.2. Feedstocks

The MFSP of the drop-in fuels from any feedstock varied over different regions in the UK. As seen from Fig. 6a, for fuel production from food waste via Centralised, Grey H2, the MFSP varied from £15.23 per GJ in London to £25.07 per GJ in Scotland. The variation in the price of fuels over the various regions is highly driven by the

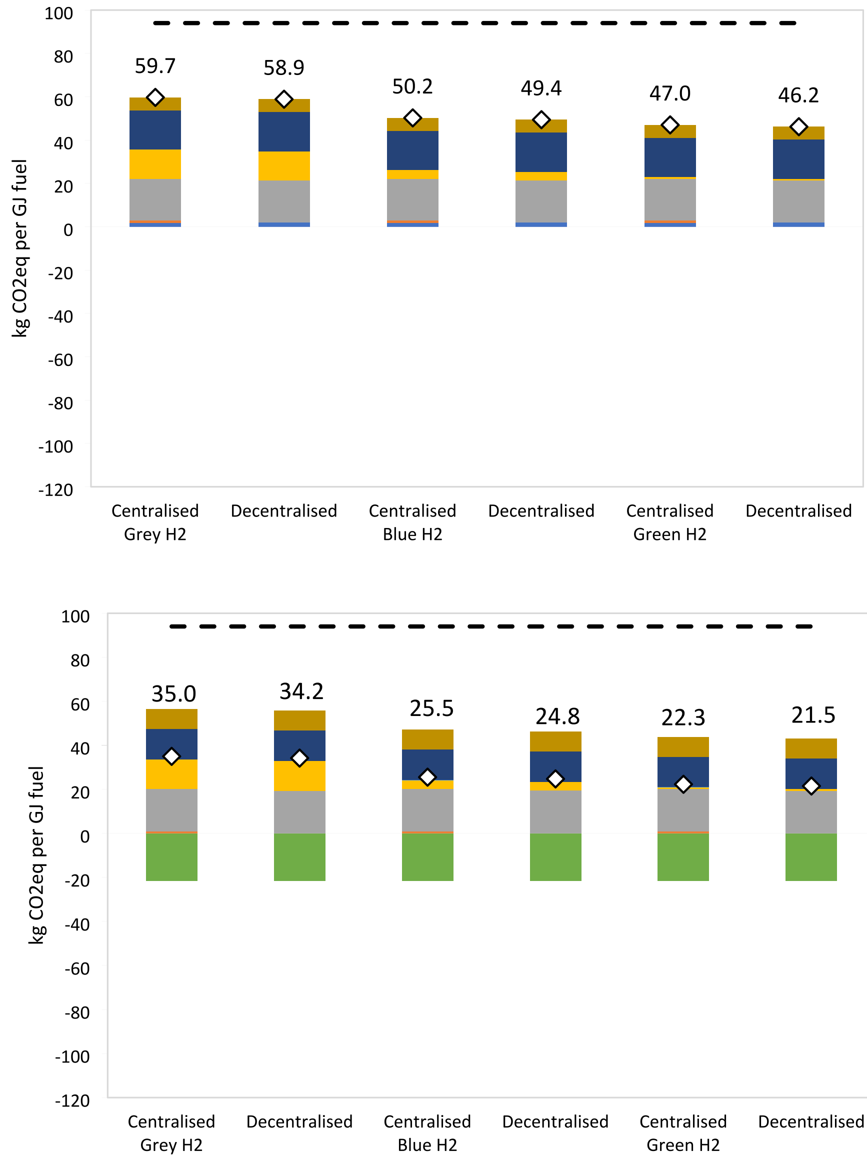


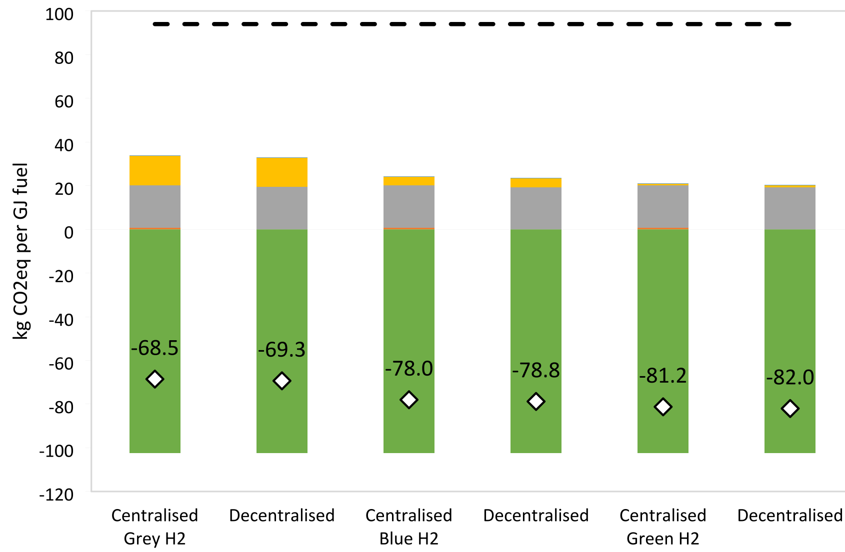
Fig. 9. Life cycle GHG emissions results of drop-in (gasoline and diesel) fuels from various production approaches for (a) Digestates (b) Food waste (c) BMW (d) SS and (e) co-processing of the feedstocks. The fossil diesel reference value by EU RED-II is 94 kg CO_2eq per GJ. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

availability and distribution of the feedstocks. London, which has the lowest MFSP for food waste processing, has one of the highest food waste availabilities. Also, the average transportation distance across London is significantly lower compared to the other regions. Conversely, Scotland which has a very high average transportation distance has the highest MFSP. Transportation distance constitutes a significant portion of the MFSP, up to 42 % of the MFSP for food waste processing, in the Centralised, Grey H2 scenario. Thus, the location of biofuel plants in any region can result in different MFSPs.

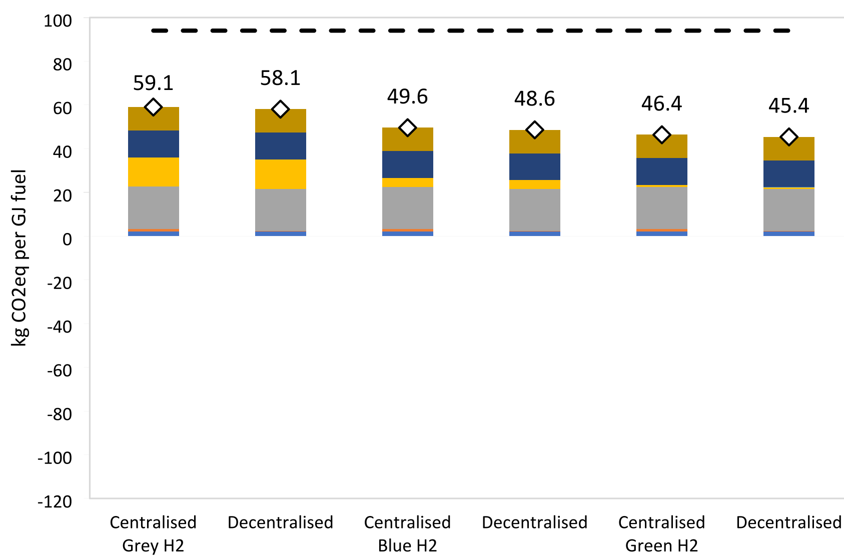
Also, the MFSP varied with the feedstock types, as seen in Fig. 6a–e, which showed the price distribution of fuel from the various feedstocks. The national average MFSP ranged from £14.76 to £36.71 per GJ (£0.47 – 1.17 per GLE) over different feedstocks and production approaches, see Fig. 7a–f and SI Section 4.3 for further details. All the feedstocks, except digestates in the decentralised cases, had prices which are competitive with 2021 wholesale fossil diesel's price of £28.58 per GJ. Therefore, excluding the decentralised scenarios for digestates, as digestates are very distributed, the average price is in the range of £14.76 to £27.11 per GJ (£0.47 – 0.86 per GLE). The significant variations in the MFSPs from the various feedstocks are due to factors such as feedstock cost/gate fees, product yield, feedstock availability and distribution over a location. The cost of feedstock/gate fee is expected to be different for the various feedstocks, as shown in Table 3. These feedstock costs are based on the current disposal methods, and there is no certainty about what the waste management market will look like in the future, considering the potential change that could come, such as in the use of new and emerging technologies. The uncertainty in the feedstock cost is further evaluated in the sensitivity analysis in SI Section 4.3.

3.1.3. Centralised vs decentralised operation

The distribution of the MFSP of fuel from the various feedstocks using centralised and decentralised approaches is shown in Fig. 7a–f. For the processing of the feedstocks, centralised production achieved the lower prices, while in the co-processing case, the decentralised approach achieved better prices. This is because the processing of digestates, food waste, landfilled BMW or SS into biocrude at localised areas prior to



c)



d)

Fig. 9. (*continued*).

conversion to fuel at a central site, requires many smaller scale HTL plants to be distributed over a much wider area, which is economically less favourable compared to the centralised approach. However, in the co-processing case, local conversion of the feedstocks is done at better plant scales since the decentralised case combines the various wet feedstocks (such as digestates, food waste and SS) that are available, thereby performing more optimally. As highlighted in Section 3.1.1, for localised processing of the feedstocks with volumes above 6000 kg/h, the decentralised approach performs better than a similar plant scale using the centralised approach. This shows how the economy of scale, distribution and transportation of feedstocks are key considerations when adopting production approaches in relation to wet biomass feedstocks. Where there is a low local availability of feedstocks, the centralised approach performs better but where there is ample local availability of feedstocks, the decentralised approach performs better. The average fuel price in the centralised approach for the grey hydrogen case is in the range of £15.44 – 21.79 per GJ compared to the decentralised approach which is in the range of £14.76 – 31.50 per GJ.

3.1.4. Hydrogen sources

The MFSP of drop-in fuels is highly dependent on the cost of hydrogen, with green hydrogen resulting in the highest fuel price, followed by blue hydrogen and grey hydrogen. As seen from Fig. 7a–f, the average prices of the fuel from the blue hydrogen scenarios for the various feedstocks are marginally higher (2 – 4 %) than those of grey hydrogen, while average prices of the fuel from the green hydrogen scenarios are significantly higher (17 – 38 %) than those of grey hydrogen. According to the IEA, the levelized cost of green hydrogen is about 3 – 5 times higher than the costs of blue and grey hydrogen, meanwhile, the incremental levelized cost of blue hydrogen is relatively smaller. While renewable hydrogen has a lower carbon intensity, it comes with a cost penalty. The price range of fuel from the grey hydrogen centralised scenarios of £14.76 – £31.5 per GJ (£0.47 – 1.00 per GLE) agrees with the prices of £15.39 – 29.1 per GJ (2021 prices) published by the Royal Society [77] and Jiang et al. [78], which used grey hydrogen for HTL of forest residues and SS. Also, the results of this study agree with Alamo et al. [79] drop-in fuel prices for SS HTL of

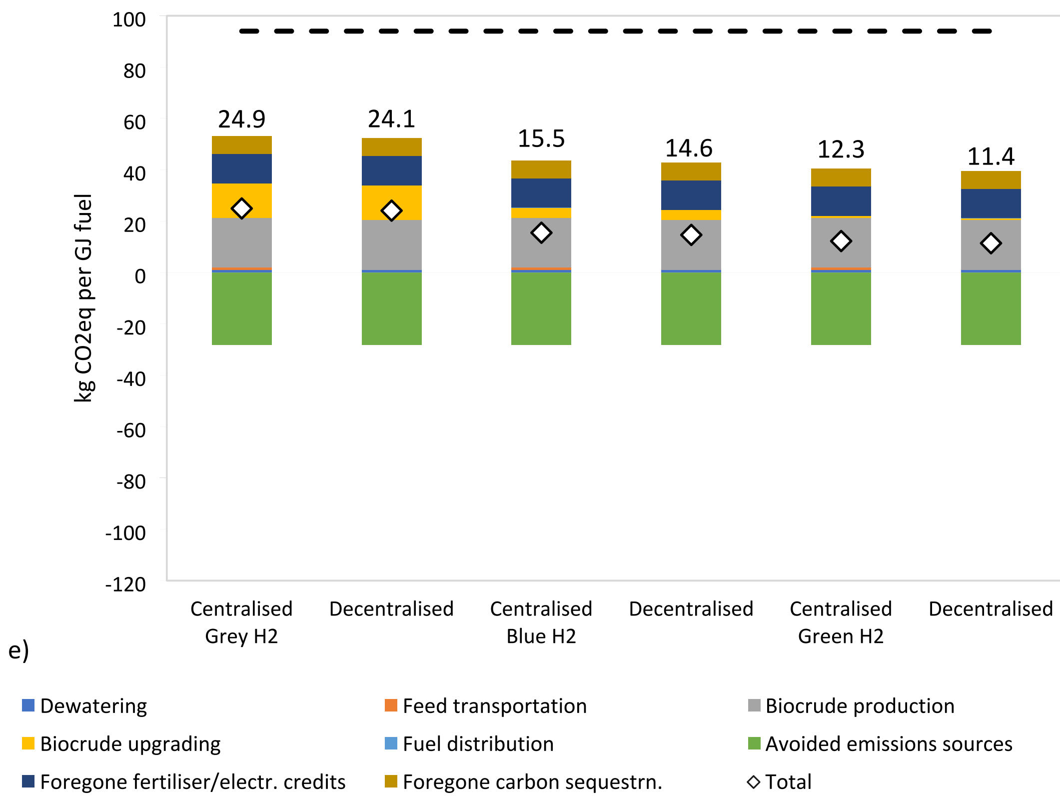


Fig. 9. (*continued*).

£18.48 – 30.10 per GJ over similar plant scales (100 – 400 tons/day). The prices of fuel from SS for this study using grey hydrogen centralised scenarios are in the range of £17.20 – 23.92 per GJ. Analyses by Jiang et al. [78] and Alamo et al. [79] also assumed negative feedstock prices for SS like the assumptions used here. However, these studies and many others in the literature do not consider the impact of using blue or green hydrogen in drop-in fuel production. Royal Society [77], Jiang et al. [78] and Alamo et al. [79] did not present a cost breakdown in their analysis but our study suggests that the fuel prices can increase by up to £5.40 per GJ when switching the hydrogen source from grey to green hydrogen. Additionally, the prices of drop-in fuel production via HTL are competitive with fossil diesel of £28.58 per GJ, based on 2021 wholesale prices of diesel published by RAC [76].

3.1.5. Total fuel quantities

Estimating the total volume of fuels that can be produced based on the UK wet feedstock is non-trivial, so here we assessed two bounding cases to serve as upper and lower limits. The competing case serves as the likely lower limit, reflecting the competitive use of biomass resources in line with current waste management techniques. Here, we assumed the use of 75 % of digestates (i.e., some digestates have low accessibility and poor quality), 46 % of food waste (i.e., some portion of the food waste is recycled or used in an AD), 30 % of landfilled BMW (i.e., due to food waste being landfilled and growth in competing uses), and 100 % of SS (i.e., new opportunity as current SS treatment is avoided due to high cost and safety concerns associated with sludge disposal). On the other hand, the upper limit assumed 100 % feedstock utilisation to reflect the theoretical maximum.

The volume of fuels which can be produced from the UK wet feedstocks is shown in Fig. 8. The competing case and 100 % feedstock utilization case resulted in a total potential volume of 1,865 ML/yr (65,395 TJ/yr) and 2,830 ML/yr (99,315 TJ/yr), respectively, at an MFSP of less than £20.80 per GJ (£0.66 per gasoline litre equivalent, see Fig. 7). This is approximately 4.5 % and 6.8 %, respectively, of the UK's total gasoline and diesel fuel consumption of 41,835 ML/yr in 2021 [80]. Importantly, these volumes are much lower than the 14 % target for biofuel consumption set by the EU RED-II [5,6] for 2030. While they can contribute to the current UK fuel mix, on top of the existing biofuel volumes of 5 % [6], this study highlights the need for complementary low-carbon fuel alternatives to close the supply gap in the mid and long-term. Here, the UK could consider low-carbon synthetic fuel often referred to as electro-fuel, produced from captured CO_2 and green hydrogen, as a complementary drop-in solution. However, the prices of electro-fuel of £72 – 95 per GJ estimated by the Royal Society [77] are significantly higher than those of drop-in biofuel from green hydrogen of £20.30 – 36.71 per GJ estimated in this study.

3.2. LCA

The life cycle GHG emissions of drop-in fuels from digestates, food waste, landfilled BMW and SS, based on the various scenarios analysed are represented in Fig. 9a–e. The impacts of plant configurations involving centralised and decentralised operations, and hydrogen sources were analysed, as shown in Fig. 9a–e. As seen from the figures, the GHG emissions were majorly driven by the biocrude production and upgrading stages, which require significant amounts of energy, chemicals, electricity and hydrogen. Also, depending on the waste types, other factors such as avoided emissions and foregone electricity credits, fertiliser credits and carbon sequestration significantly impact GHG emissions.

3.2.1. Feedstocks

All investigated fuels achieve lower emissions than conventional petroleum diesel, but the life cycle GHG emissions of the drop-in fuels are highly affected by the nature of the biomass feedstocks (Fig. 9). Diverting landfilled BMW to fuel production achieves significantly negative GHG emissions, due to avoided methane gas emissions associated with existing landfill management practices. Currently, about 17

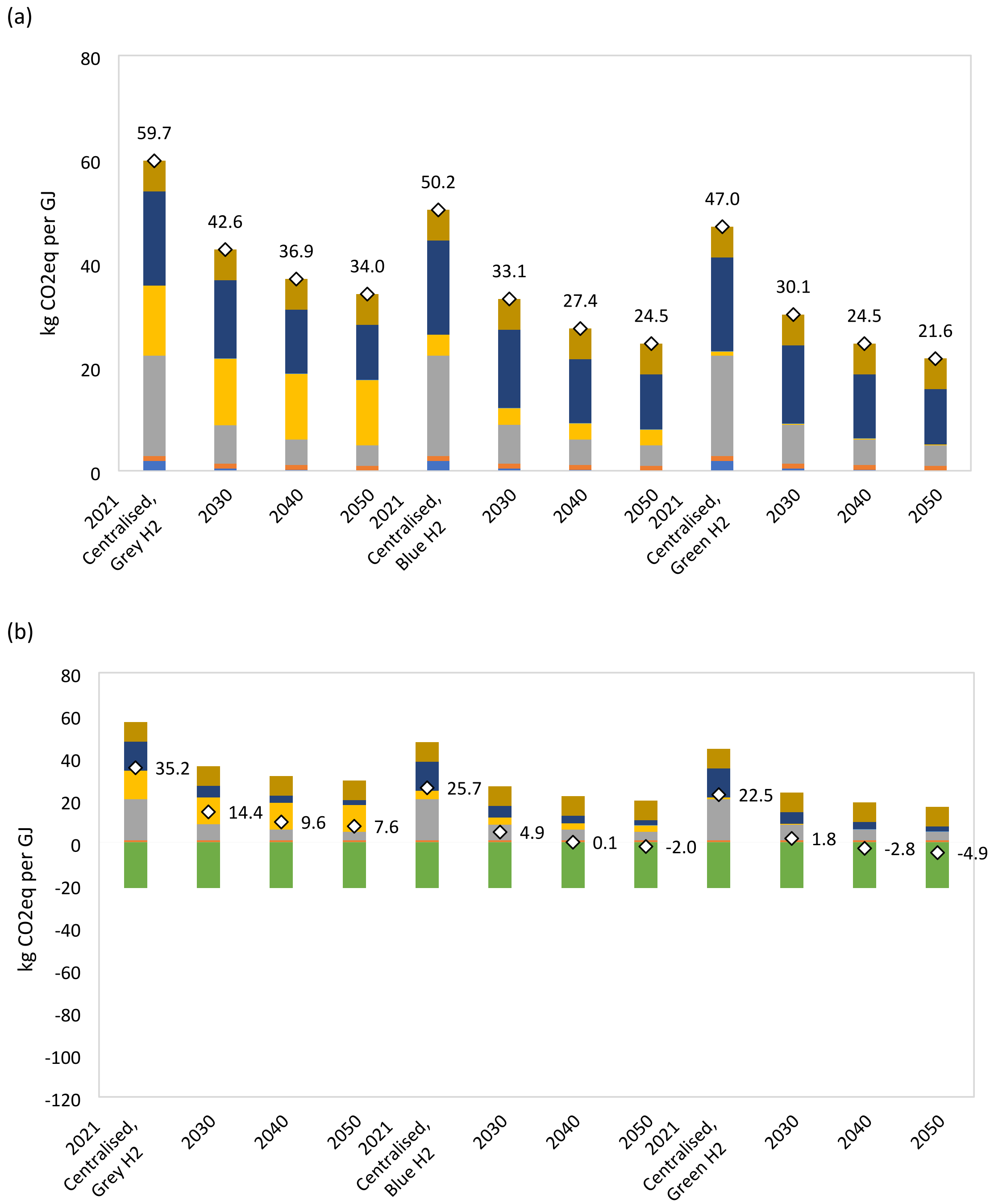
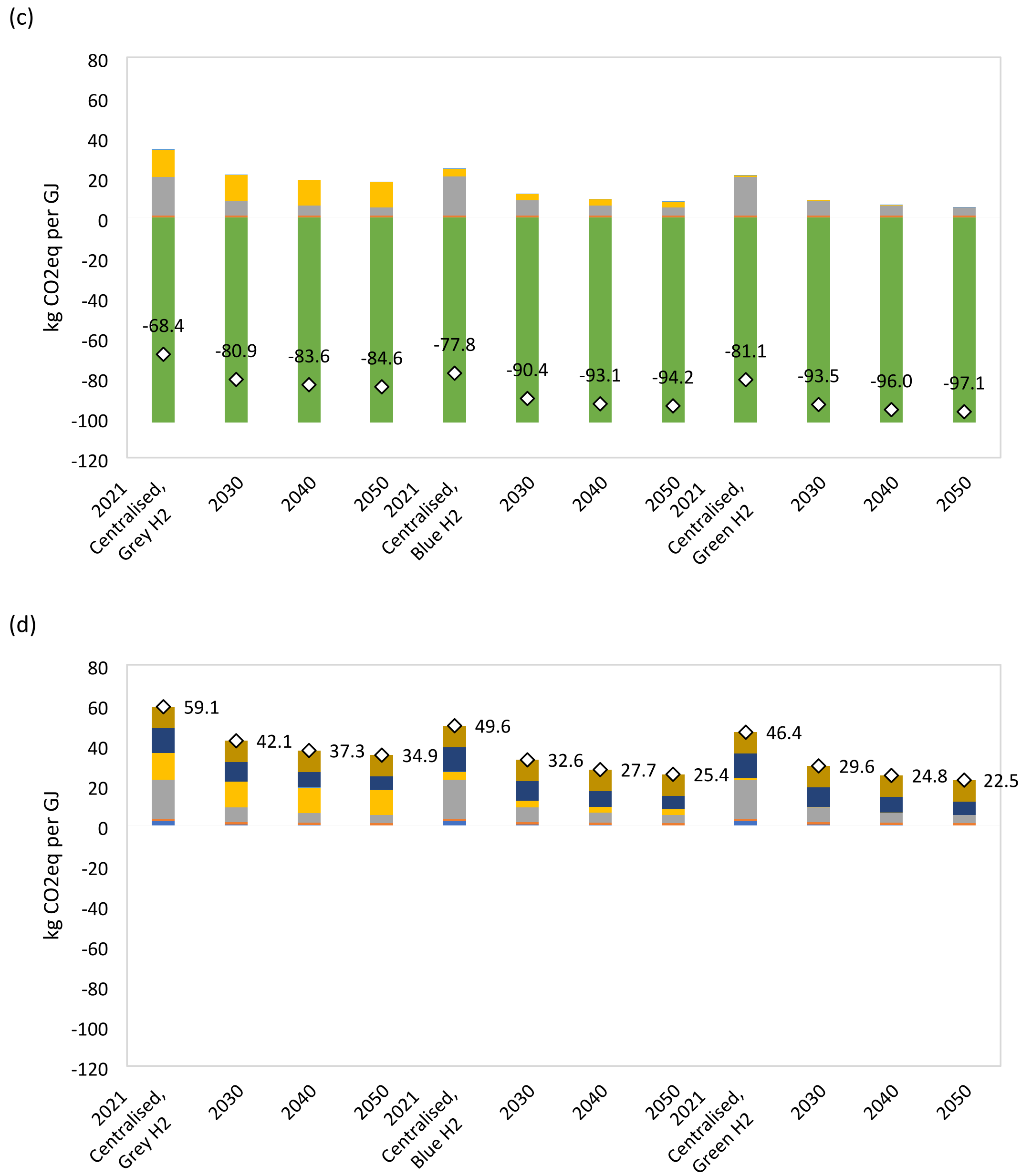


Fig. 10. Life cycle GHG emissions outlook scenarios for (a) digestates (b) food waste (c) Landfilled BMW (d) SS and (e) co-processing of the feedstocks, based on a consequential LCA approach.

% of landfilled MSW is food waste [9] and approximately 50 % of landfilled MSW is BMW [16]. A significant amount of methane is released from landfills despite some degree of methane recovery at these sites, and Nordahl et al. estimates about 400 g CO_2eq is released per tonne MSW [10]. Hence, −21.5 kg CO_2eq per GJ and −102.3 kg CO_2eq per GJ fuel contributes to the avoided emissions in the utilisation of landfilled food waste and BMW for fuel production, respectively. On the other hand, the much higher emissions for digestates and SS feedstocks are also driven by other consequential factors associated with the diversion from their existing uses as soil fertilizers to fuel use, foregoing fertilizer credit and carbon sequestration, and thus contributing about 11.1 – 17.9 kg CO_2eq per GJ fuel and 5.9 – 10.7 kg CO_2eq per GJ fuel, respectively. The total emissions from the digestates, food waste, landfilled BMW and SS feedstocks are 36 – 187 % lower than the 94 kg CO_2eq per GJ (94 g CO_2eq per MJ) fossil baseline under the EU RED-II [51]. This shows there is a strong benefit from diverting these waste materials into fuel.

However, it is important to note that the CO_2 accounting methodology under the EU RED-II is largely based on an attributional approach. Moreover, the RED-II requires that these advanced renewable fuels achieve at least 65 % (or 70 % if they are of non-biological origin) CO_2 reduction relative to conventional fuel for them to qualify under the regulation. Therefore, as part of the sensitivity analysis, we quantified the lifecycle emissions of these fuels by excluding the avoided emissions



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associated with current waste management practices. Without the consequential element of the analysis (i.e., the adoption of an attributional approach), the fuels' carbon intensities vary from a high range of 33.7 – 36.1 kg CO_2eq per GJ (for a centralised production with Grey H2) to a low range of 20.3 – 22.5 kg CO_2eq per GJ (for a decentralised production with Green H2). This translates to a CO_2 reduction potential of 62 – 78 % relative to conventional fuels (see SI Section 4.3 for detailed emissions for each route). Our overall lifecycle emissions are comparable to the literature values for similar HTL processes using woody and algae biomass as feedstocks that resulted in 27 kg CO_2eq per GJ and 35 kg CO_2eq per GJ, as reported by Fortier et al. [81] and Tews et al. [40], respectively.

3.2.2. *Centralised and decentralised approaches*

The use of a decentralised approach compared to a centralised approach is found to have only a marginal impact on the GHG emissions of the final fuel. As seen from Fig. 9a–e, the change in the mode of operation from centralised to decentralised resulted in about 1 kg CO_2eq per GJ difference, which is small compared to the overall life cycle GHG emissions of the fuel. The emissions associated with feedstock and fuel product transportation over distances are negligible compared to the overall fuel's life cycle emissions, hence, either a centralised or decentralised approach does not offer any strong advantage in GHG emission reduction. Moreover, in the future, the use of renewable fuel in feedstock and/or product transportation could lead to lower overall emissions.

3.2.3. *Hydrogen source*

Given that biocrude upgrading accounts for ca. 40 % of the total attributional lifecycle emissions, the use of a lower carbon intensity hydrogen in the hydroprocessing stage resulted in a lower overall lifecycle emission. As seen from Fig. 9a–e, the green hydrogen scenarios produced GHG emissions which are 18–37 % (12.7 kg CO_2eq per GJ) less than those produced using grey hydrogen, while the blue hydrogen scenarios produced 14–28 % (9.5 kg CO_2eq per GJ) less than those produced using grey hydrogen. As hydroprocessing requires a significant amount of hydrogen, about 4.4 kg hydrogen per 100 kg of biocrude, considerable emissions reductions can be achieved by switching to a less CO2-intensive hydrogen source during biocrude upgrading. However,

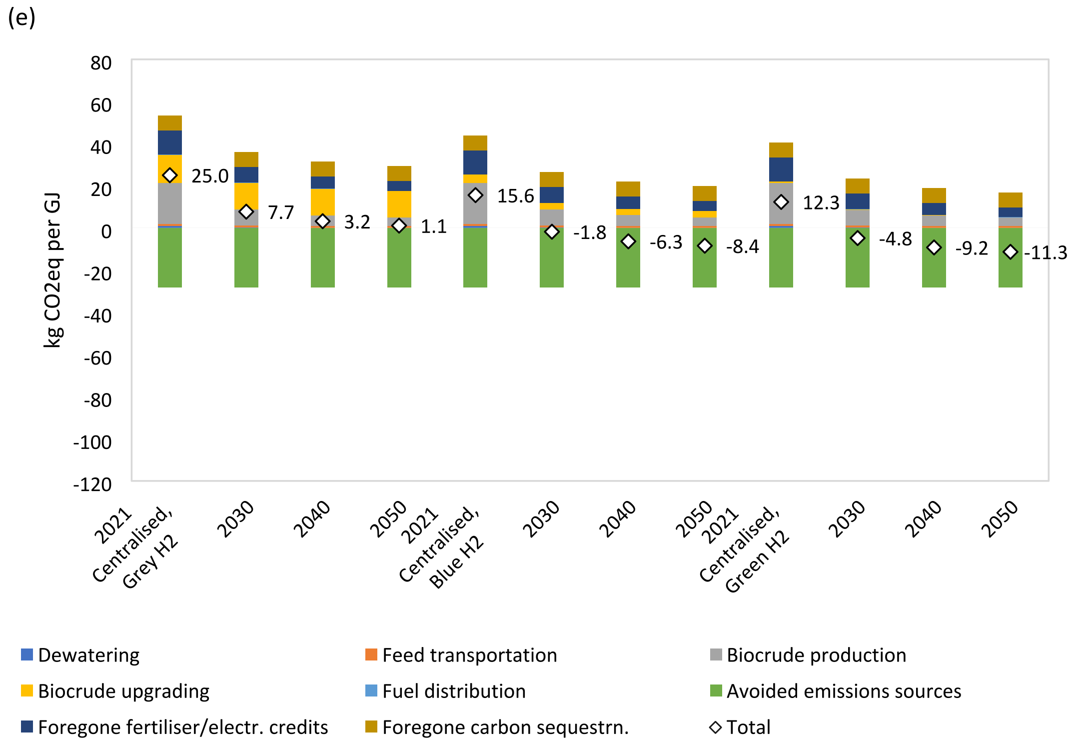

Fig. 10. (*continued*).

this comes with high production price implications, as seen previously in Section 3.1.4, which demonstrated the impact of hydrogen sources on MFSP.

3.2.4. Total emission quantities

The use of these drop-in fuels has a significant potential to reduce emissions from the UK's road transport sector, which currently emits about 133.7 Mt CO_2eq [1]. Substituting conventional fuels with these drop-in alternatives could reduce emissions by about 4.5 – 5.4 Mt CO_2eq per annum and 8.4 – 9.8 Mt CO_2eq per annum, based on the competing and 100 % feedstock cases, respectively. This is a 3.4 – 7.3 % reduction in the UK's total road transport emission attributable to the displacement of 4.5 % (competing case) and 6.8 % (100 % feedstock case) of conventional fuels by these alternatives. A further reduction will require larger quantities of drop-in fuels, however, our initial estimate suggests that there are limited feedstocks available in the UK, and therefore the UK would benefit from other low-carbon synthetic fuels that can be scaled up to facilitate decarbonisation of the transport sector.

3.2.5. Life cycle emissions outlook

The lifecycle GHG emissions for the fuels are expected to reduce over time as manufacturing processes shift to lower carbon electricity and feedstocks (Fig. 10). Based on the competing case, the drop-in fuels could achieve as low as 1.1 kg CO_2eq per GJ, −8.4 kg CO_2eq per GJ and –11.3 kg CO_2eq per GJ of fuels from a centralised co-processing configuration in 2050 using grey, blue and green hydrogen, respectively. Because the consequential LCA approach is highly dependent on both the direct emissions from the drop-in fuel and indirect emissions associated with other processes and products related to drop-in fuel's production and use, there is a high degree of uncertainty in the future emissions estimated using the consequential approach. Furthermore, as the global economy decarbonises in the future, the consequential factors (such as foregone fertiliser and electricity generation) would likely have a lower impact on the total lifecycle GHG emissions. As part of the sensitivity analysis, we repeated the analyses and adopted the attributional LCA approach, which considers only the emissions from processes directly related to the fuel production and use (Fig. 11). The emissions from the centralised processing of the feedstocks based on an attributional approach in 2050 are in the range of 17.6 – 17.7 kg CO_2eq per GJ, 8.0 – 8.2 kg CO_2eq per GJ and 5.1 – 5.3 kg CO_2eq per GJ for grey, blue and green hydrogen use, respectively. Interestingly, by 2050, the lifecycle GHG emissions of these drop-in fuels can be lower than the limits for low carbon hydrogen in the UK of 20 kg CO_2eq per GJ (LHV), and potentially achieving an 81 – 95 % GHG reduction relative to the EU RED-II baseline.

4. Conclusion

This study demonstrates the potential for economically viable production of drop-in fuels from wet waste feedstocks, capable of achieving low – and even negative – life cycle GHG emissions. The HTL of digestates, food waste, BMW, and SS into drop-in fuels using various production approaches, such as plant configurations and hydrogen sources, were studied using TEA and LCA methodologies.

Our analysis estimated the MFSP of the fuel to be in the range of £14.76 to £27.11 per GJ (£0.47 – 0.86 per GLE), which is lower than the current fuel prices in the UK. However, the MFSP metric should be interpreted with caution. The main intention of the metric is to provide a simple "rule of thumb" method for comparing different types of fuel production technologies, which may not consider other relevant issues, including the revenue stream and margin for the producers. From a CO_2 perspective, the fuels are estimated to have a carbon intensity in the range of −82.0 to 59.7 kg CO_2eq per GJ, which translates to a benefit in the range of 36 – 187 % relative to a conventional fuel baseline of 94 kg CO_2eq per GJ. However, the LCA approach adopted here is based on the consequential method, which covers a broader, market-mediated effect beyond the typical fuel's production boundary. By adopting an attributional LCA approach, the CO_2 reduction potential of the fuels are in the range of 62 – 78 % relative to a similar conventional fuel baseline. Furthermore, the lifecycle GHG emissions for the fuels are expected to reduce over time as manufacturing processes shift to lower carbon electricity and feedstocks.

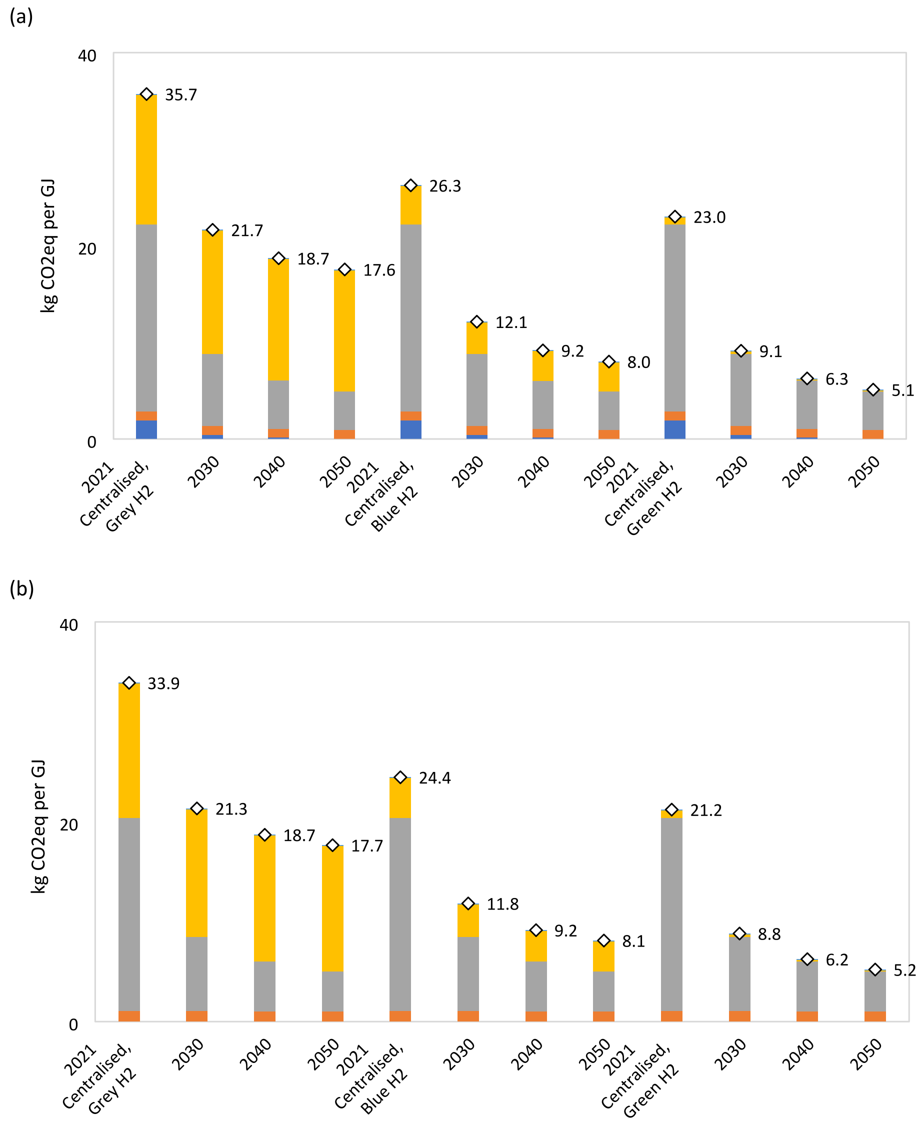

Fig. 11. Life cycle GHG emissions outlook for (a) digestates (b) food waste (c) Landfilled BMW (d) SS and (e) co-processing of the feedstocks, based on an attributional LCA approach.

Several key elements, such as plant configuration (e.g., centralised and decentralised operations) and feedstocks, as well as fuel production stages and supply chain factors, can have a significant influence on the MFSP and lifecycle GHG emissions of the fuels. The use of either a centralised or decentralised approach for the optimum processing of wet biomass is highly dependent on the availability, distribution and transportation of the wet biomass feedstocks, as both approaches can be beneficial. For the wet biomass feedstocks studied, when the local availability of feedstocks is less than 6000 kg/h of dry feed, the use of a centralised approach has significant cost advantages over the use of a decentralised approach. However, when the local availability of the feedstocks is above 6000 kg/h (dry), a decentralised approach provides better cost advantages over a centralised approach. The choice of centralised or decentralised configurations requires an optimal balancing between plant scaling and feed transportation distance, as both factors play important roles in the economics of the drop-in fuel production from wet biomass feedstocks in the UK. On the other hand, in terms of GHG emissions, the choice between a decentralised and a centralised approach has a marginal impact on the overall lifecycle emissions of the fuels. Similarly, the choice of hydrogen production methods can influence the cost and CO_2 impacts of the drop-in fuel, demonstrating the trade-off between the CO_2 benefit of utilizing a low-carbon hydrogen supply and its higher cost of production and supply. An additional production cost of up to £5.40 per GJ can be added due to the high costs of low-carbon hydrogen. However, a GHG emissions reduction of up to 40 % relative to the corresponding grey hydrogen scenarios can be achieved using low-carbon hydrogen.

There are significant opportunities for the production of drop-in fuels by diverting the UK's wet waste feedstocks from conventional waste treatments into transport fuels as a means to increase renewable energy penetration and mitigate transport emissions. However, our study estimated that the UK's wet biomass feedstocks could supply only about 4.5 – 6.8 % of the total gasoline and diesel consumption in 2021, potentially resulting in a 3.4 – 7.3 % reduction in the UK's total road transport emission. While they can contribute to the current UK fuel mix on top of the existing biofuel volumes of 5 %, this study also highlights

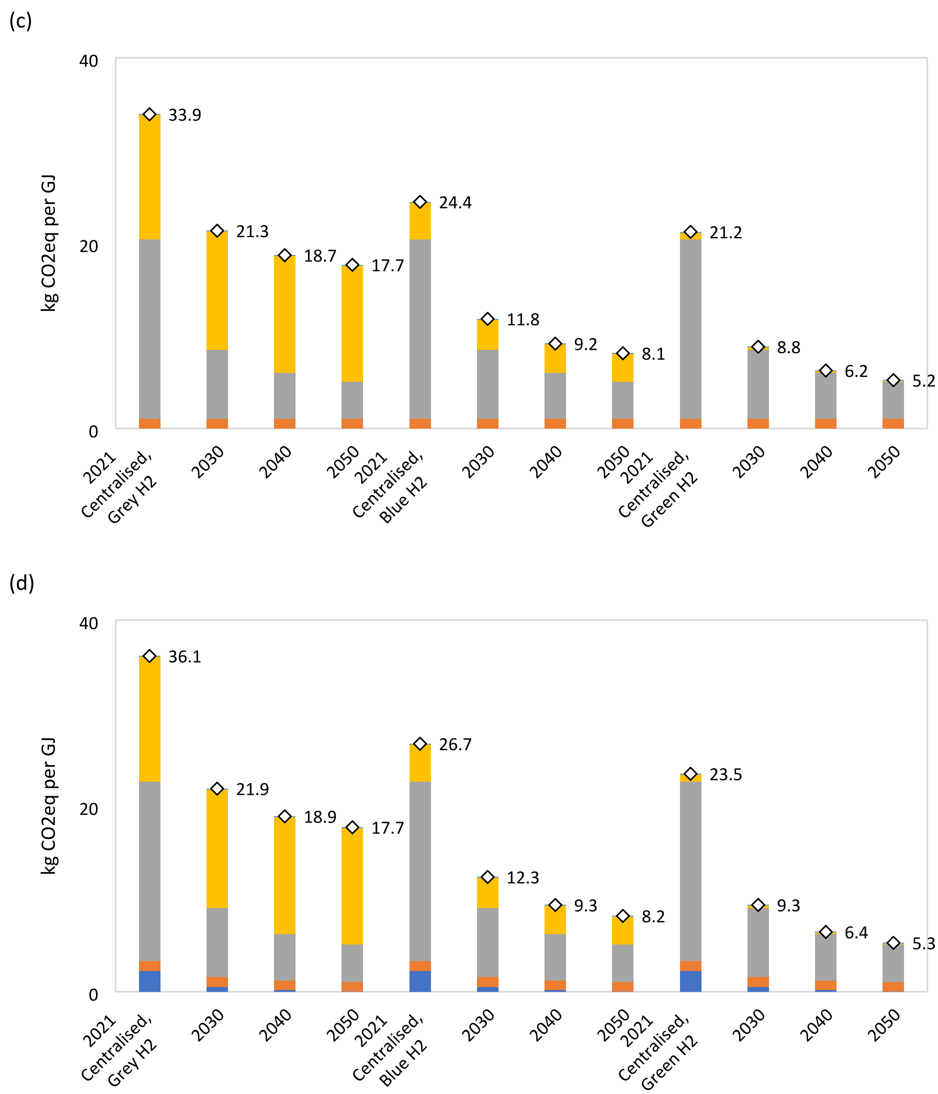

Fig. 11. (*continued*).

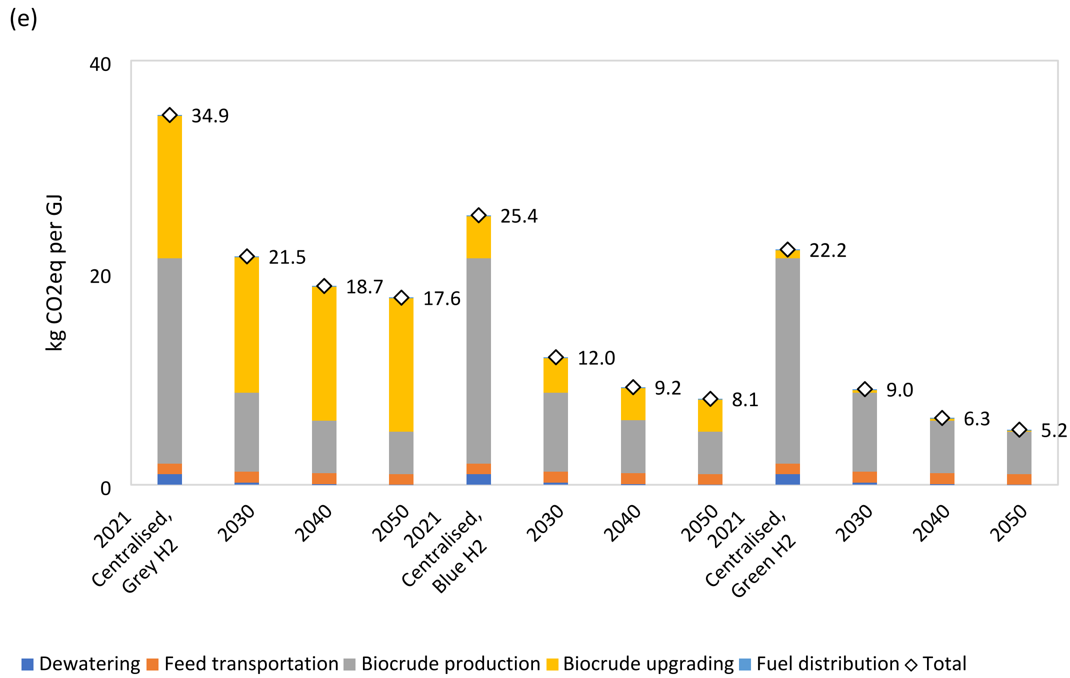

Fig. 11. (*continued*).

the need for other complementary low-carbon fuel alternatives to close the supply gap in the mid and long-term towards achieving a net-zero future. As a complementary measure, the UK could consider other synthetic fuel pathways, for example, the production of drop-in fuel from green hydrogen and CO_2 captured from existing, CO_2-intensive industries in the UK. This is an emerging technology that could potentially be scaled up. Synthetic electro-fuel has gained regulatory attention in the EU, which has recently allowed combustion engine vehicles to be sold after the 2035 ban as long as the vehicles operate exclusively on low-carbon, synthetic electro-fuel. This is a technology area that is worth considering for the UK and will require further investigation.

CRediT authorship contribution statement

Sylvanus Lilonfe: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Project administration, Software, Visualization, Writing – original draft, Writing – review & editing. **Ioanna Dimitriou:** Conceptualization, Methodology, Supervision, Validation, Writing – review & editing. **Ben Davies:** Conceptualization, Supervision, Writing – review & editing. **Amir F.N. Abdul-Manan:** Conceptualization, Funding acquisition, Writing – review & editing. **Jon McKechnie:** Conceptualization, Funding acquisition, Resources, Methodology, Project administration, Resources, Supervision, Validation, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

No data was used for the research described in the article.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.cej.2023.147516.

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