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Review Article

A review of techno-economic analyses and life cycle greenhouse gas emissions of biomass-to-hydrocarbon “drop-in” fuels

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ABSTRACT

Synthetic “drop-in” fuels are compatible with existing fuel and vehicle infrastructures and, when produced sustainably, they could play an important role in mitigating the emissions of greenhouse gases (GHG) from transportation, especially in the hard-to-decarbonise sectors like freight and aviation. However, there is a need to understand the availability of biomass resources for drop-in fuel applications and the potential challenges associated with using these feedstocks including the supply chain issues. Hence, this paper offers a critical review of non-food biomass and drop-in fuel production including the biomass availability in the UK, the production of drop-in fuels from biomass feedstocks via thermochemical routes, estimated fuel production prices and volumes, and life cycle GHG impacts. The paper explores several fuel production factors, including energy and hydrogen requirements, as well as supply-chain considerations, which were used to estimate the drop-in fuel potential in the UK economy. We estimate the availability of non-food biomass resources in the UK to be in the range of 167–205 Mtpa (wet) [839–1033 PJ per year], as more than 50 % of these volumes are from high moisture content feedstocks such as biogenic municipal waste and sewage sludge. Other biomass feedstocks that are produced in significant quantities include straw and wood waste. Also, it is estimated that the total UK drop-in fuel manufacturing potential is in the range of 269–563 PJ per year. When used to displace fossil fuels in road transportation, this could lead to a total GHG reduction of 18.7–64.4 Mt. CO_2eq per year which is 18.8–64.7 % relative to the UK’s overall road transport emissions from all fuels in 2021.

1. Introduction

One of the greatest challenges of the 21st century is the global transition to a low-carbon, sustainable energy system. Current energy systems are heavily dependent on emissions-intensive energy sources, resulting in the accumulation of greenhouse gas (GHG) emissions in the atmosphere and contributing to the changes in the global climate. According to the IEA (IEA, 2022c), about 36 gigatonnes of carbon dioxide (CO_2) were emitted in 2021 from combustion sources and industrial processes, and this was over 10 times more than the emissions at the start of the 20th century. About a third of the emissions in 2021 were from the transportation sector, where fossil fuels currently supply about 95 % of transport energy demand (Climate Watch, n.d.). However, concerns over energy security and environmental sustainability have led to a growing interest in the use of alternative technologies in the transport sector. Despite the strong interest in renewable energy such as

Abbreviations: AD, Anaerobic digestion; ADP, Abiotic depletion; AP, Acidification potential; BMW, Biogenic municipal waste; BTL, Biomass to liquids; CCS, Carbon capture and storage; CO_2, Carbon dioxide; EfW, Energy-from-waste; EP, Eutrophication potential; FTS, Fischer-Tropsch synthesis; GGE, Gasoline gallon equivalent; GHG, Greenhouse gas; GWP, Global warming potential; HTL, Hydrothermal liquefaction; HTP, Human toxicity potential; ICEVs, Internal combustion engine vehicles; iLUC, Indirect land use change; LCA, Life cycle assessment; LHV, Low heating value; LUC, Land use change; MFSP, Minimum fuel selling price; MSW, Municipal solid wastes; MTG, Methanol-To-Gasoline; RED, Renewable Energy Directive; SCWG, Supercritical water gasification; SI, Supplementary Information; SMR, Steam methane reforming; TEA, Techno-economics analysis; TIGAS, Topsoe Integrated Gasoline Synthesis; USD, United States dollar.

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biofuels, biofuels contribute only about 4 % of the UK and global transport fuel consumption volumes by energy in 2021, respectively (IEA, 2022a; DESNZ, 2023b). This contribution is significantly lower than the targets of roughly 15 % by 2030 set by numerous government mandates, such as the EU Renewable Energy Directive (RED II) (European Commission, 2019).

Conventional biofuels, often called first-generation biofuels such as bioethanol and biodiesel, which are used currently are limited in application due to concerns over their competition with food production, environmental impacts and compatibility limits with existing engines. The food vs fuel debate has strongly intensified since the global food crisis of 2008, which was partly attributed to first-generation biofuels (European Commission, 2011). Conventional biofuels are typically produced from food-based biomass through the fermentation of starch and sugar and the transesterification or hydrogenation of fats and oils. Also, the utilisation of biomass feedstocks for these biofuels can lead to considerable land and environmental impacts when biofuel crops displace the production of food or natural habitation. The land use change (LUC) and indirect land use change (iLUC) associated with using forests, food crops and grasslands can be significant (Searchinger et al., 2008; Plevin et al., 2010). According to Searchinger et al. (Searchinger et al., 2008), when accounting for the impact of LUC in corn cultivation, the GHG emissions of corn-based ethanol could double over 30 years instead of achieving the 20 % saving initially estimated without LUC. Although other studies have published a smaller estimate for the LUC effect of corn ethanol, the GHG consequence of LUC is generally non-trivial. Despite LUC and iLUC being the most uncertain components in the GHG emissions estimation of biofuels (Plevin et al., 2010), the availability and way in which biomass feedstocks are obtained play important roles in the sustainability of biofuels. As a result, current government documents such as the UK Biomass Strategy developed by DESNZ (DESNZ, 2023a) require that emissions associated with LUC and iLUC are accounted for in biomass applications. The strategy also required biofuels to have at least 60–65 % emissions savings relative to a fossil fuel comparator of 94 gCO_2eq/MJ in energy and transport applications. Additionally, bioethanol and biodiesel (transesterified) can only be used with petroleum fuel in blends up to a certain percentage in current engines without costly engine modifications. These limitations not only impact the production of first-generation biofuels but also their utilisation, and therefore, the development of alternative fuels such as renewable hydrocarbon fuels is considered a key innovation for addressing these issues.

Renewable hydrocarbon liquid fuels, such as renewable gasoline and renewable diesel, are designed as drop-in solutions so that they can be used as direct replacements for petroleum fuels, thus allowing for a broader application of renewable fuels in transportation. Drop-in fuels are chemically identical to petroleum gasoline, diesel, or jet fuel, and can be used in engines without the need for blending with petroleum fuels or costly engine modifications. Drop-in fuels are typically produced from second and third-generation biomass feedstocks such as agricultural waste (Anex et al., 2010) and algae (Jones et al., 2014). They can also be produced from waste CO_2 (d'Amore et al., 2023). Drop-in fuel production allows the utilisation of non-food biomass resources and can improve the sustainability of biofuels by addressing issues such as LUC, iLUC and the availability of biomass resources. Non-food biomass which is lignocellulosic biomass or food waste typically has LUCs of zero, especially for residues and wastes. Additionally, using non-food biomass resources such as agricultural residues, wood wastes and municipal wastes avoids iLUC and the competition of biofuel with food. However, the utilisation of non-food biomass for drop-in biofuel applications may come with other consequential indirect emissions which may arise from diverting these materials from existing streams (Ou et al., 2022; Lilonfe et al., 2024). These consequential emissions may be positive or negative depending on the foregone applications, for example, the diverting of food waste from landfilling or anaerobic digestion (AD). Furthermore, the availability of non-food biomass is generally substantial across many regions as almost every society generates wastes from biomass feedstocks such as food and forestry. Lignocellulosic biomass feedstocks are available on a renewable basis (Ho et al., 2011).

In 2019, the UK became the first major economy to pass a net zero emissions law which has legally binding climate targets across its economy (BEIS and Skidmore, 2019). As a result, there is a strong compelling drive towards mitigating GHG emissions by increasing the uptake of biofuels in the transport sector, particularly in hard-to-decarbonise areas. Consequently, many recent publications such as the Royal Society (Royal Society, 2023) and Geissler et al. (Geissler et al., 2024) estimated the potential volumes of biomass and drop-in fuels that can be produced in the UK and US, respectively. These studies estimated the prices, life cycle GHG emissions, potential volume availability and GHG emissions of drop-in fuel. However, they did not estimate the potential emissions savings of using drop-in fuels and the consequences of diverting biomass feedstocks from existing applications. Competing applications can have a significant impact on drop-in fuel production, both in terms of GHG emissions and cost. Hence, understanding the potential implications of diverting these feedstocks from the existing supply chain and corresponding GHG emissions savings from biofuels is particularly important in the sustainability of drop-in biofuels.

Therefore, in this report, we reviewed the availability of non-food biomass feedstocks and their existing applications across the UK market. Also, we reviewed thermochemical drop-in biofuel production technologies, their techno-economics and life cycle GHG assessments. Additionally, we assessed the potential volumes and GHG emissions impacts of using drop-in biofuels in road transport applications in a manner not captured in previous publications, by evaluating key features such as biomass supply chains, hydrogen demand, fuel potential, and attributional and consequential GHG emissions associated with these drop-in biofuels.

2. Methodology

The methodology used in the paper is divided into two parts: (1) The methods used for the critical literature review where the existing key literature on biomass and drop-in biofuel production including supply chain and techno-economics were analysed and: (2) The approaches that were used for estimating the potential drop-in volumes and GHG emissions impacts in the UK.

The methodology used for the literature review was focused on non-food biomass resources, drop-in fuel production, biomass and biofuel policies, and techno-economic and life cycle assessments of drop-in fuels. Therefore, search strings having these keywords were used to search for publicly available information in various databases including ScienceDirect, government websites, and reputable sources such as IEA, NREL, WRAP and NNFCC. Key materials retrieved and critically analysed included peer-reviewed publications, policies, regulations, theses, books, reports, and websites.

The estimation of the availability of UK non-food biomass feedstocks was done using two scenarios – the 100 % feedstock and the competing cases. For the 100 % feedstock case which estimates the whole availability of biomass, the estimation of the potential volumes of biomass available in the UK was done by reviewing existing literature. Also, in accounting for biomass availability in the competing use case, volumes of biomass from existing competing applications were used in estimating the availability of the biomass feedstocks. The impacts of existing biomass utilisation technologies, such as cost and environmental impacts, were used in determining the availability of the biomass feedstocks in the competing case. Additionally, due to the diverse range of biomass feedstocks, a case-by-case approach was also applied in some instances, see Sections 3.1.1 to 3.1.8 for details on specific methodologies.

The approach used for estimating the potential drop-in volumes and GHG emissions impacts in the UK was based on the material and energy balance data of pilots, demonstrations or experimental units of various drop-in fuel production technologies published in the literature. The GHG emissions estimates were based on well-to-wheel emissions data from important peer-reviewed literature such as those which provided detailed life cycle assessment (LCA) analysis. Where a significant variation in data exists for a parameter as reported by various authors, any or both of these approaches were adopted (a) Prioritisation of demonstration, pilot, and regional data (b) Use of averages of the most represented range in the literature. Also, because the UK biomass strategy required that biofuels should have at least 65 % emissions savings relative to a fossil fuel comparator of 94 gCO_2eq/MJ in transport applications, this criterion was prioritised in the selection of the fuel conversion routes. Furthermore, the UK Biomass Strategy 2023 considers mostly emissions directly associated with biomass processing or use, which may not represent other potential impacts that may arise from diverting biomass resources from existing applications. The indirect emissions it considers are only the iLUCs. Therefore, both attributional and consequential GHG emission impacts of using these biomass resources for drop-in fuel applications were evaluated. Consequential LCA analyses the direct and indirect emissions associated with the fuel. Direct emissions could include LUC, crop cultivation, fuel conversion, etc. Indirect emissions from the diversion of feedstocks for fuel production could include iLUC, foregone energy benefits, foregone fertiliser benefits, avoided landfill emissions, etc. Consequential emissions help in evaluating the potential impacts including iLUC that may arise from the diversion of the feedstocks in fuel production.

3. Findings and discussions

3.1. Non-food biomass feedstocks in the UK

The availability, distribution and characteristics of biomass resources are key in the production of biofuels. While biomass feedstocks are widely considered sustainable, they can be limited in availability (e. g., forest resources in the UK), and these could have considerable impacts on the environmental and economic benefits of biofuels, as well as biofuels' production volumes. In this section, the volumes and distribution, properties and competing applications of UK-origin, non-food biomass feedstocks, which can be obtained from agricultural residues, straw and other non-food biomass, are evaluated by reviewing existing literature.

3.1.1. Municipal solid waste (biogenic fraction)

Municipal solid waste (MSW) is comprised of wastes from households and other waste of similar nature and composition such as packaging waste, construction and demolition waste and commercial and industrial waste. As seen from Table 1, 34.6 Mtpa (wet) of biogenic municipal waste (BMW) is generated annually in the UK. E4tech (E4tech, 2017) and Royal Society (Royal Society, 2023) reported that 40.2 Mtpa and 40 Mtpa of BMW are generated in the UK, respectively. These are slightly higher than the estimate here. However, both publications failed to highlight the basis of their estimation, although the E4tech estimate was based on 2015 UK BMW generation rates.

The competing uses for BMW are strong, with many projects using or considering using BMW for energy or other purposes, see Supplementary Information (SI) Section 1.1 for more details on current BMW uses. However, a volume of 6.1 Mtpa of BMW was sent to landfill in the UK in 2020 (Defra, 2022b), see Fig. 1 for details. Excluding 1.9 Mtpa of food waste landfilled as part of BMW and including 1.8 Mtpa of BMW used in energy-from-waste (EfWs), the minimum availability of BMW in the competing case is estimated as 6.0 Mtpa. The generation outlook of BMW will likely remain constant over the next few years unless significant changes in production or consumer behaviour are initiated, as the generation rates of households and commercial & industrial sources have remained stable since 2015, see SI Section 1.1 for more details. Based on the capacity data on EfWs plants from Tolvik (Tolvik, 2022) and NNFCC (NNFCC, 2021), no BMW will be landfilled in 2030, as all BMWs are used majorly in energy applications.

BMW is suitable for bioenergy production via different pathways, both biochemical and thermochemical methods, due to its moisture content of roughly 50 % on average. Utilising BMW in thermochemical processes like gasification or hydrothermal liquefaction (HTL) may require pre-treatment before conversion to fuel. The lower heating value (LHV) of the biogenic component of MSW on a dry basis is estimated to be in the range of 13.1–17.5 MJ/kg (Moora et al., 2017; Fetene, 2021). The GHG emissions from current BMW usage are in the range of 0 kg of CO_2eq/t MSW for recycling to nearly 400 kg of CO_2eq/t MSW released from landfills (Nordahl et al., 2020). Energy and composting applications of BMWs have associated emissions which depend on the conversion method used. Hence, consequential emissions may be suitable for evaluating and comparing the emissions of potential biomass applications with existing ones. Additionally, gate fees are charged at EfW plants. The gate fee charged for processing BMW at EfW plants is around £90 to £95, while a fee of £111 to £116 is charged at landfills (WRAP, 2021). Gates fees can serve as a good source of attraction and benefits for drop-in fuels, making drop-in fuel production more competitive.

3.1.2. Straw

Straw is an agricultural waste whose cultivation in the UK is mainly derived from wheat, barley, oil seed rape and oats (Defra, 2019). Straw production in the UK is in the range of 11–12 Mtpa (NNFCC, 2014;

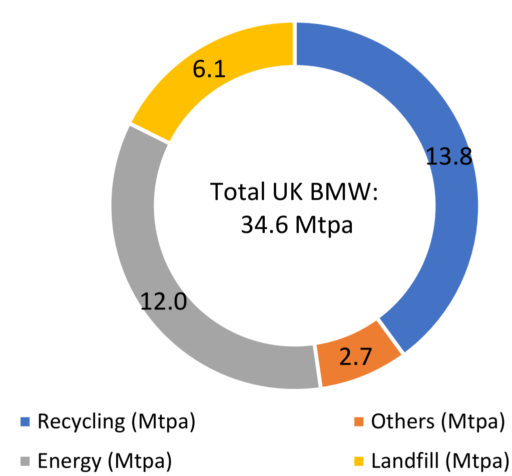

Fig. 1. UK BMW generation with competing uses in 2021.

Table 1
BMW streams in the UK.

Waste stream	Household waste stream, wet (Mtpa)	Commercial waste stream, wet (Mtpa)	References
Food waste	5.1	4.9	(Facchini et al., 2018; WRAP, 2019c, 2019b, 2020b; Jeswani et al., 2021; Defra, 2022b)
Garden & others	5.4	0.6	
Paper & cardboard	4.9	8.7	
Wood waste	1.0	4.0	
Total	**16.4**	**18.2**	

Defra, 2019). These volumes do not consider the straw uncollected during baling, which is up to 50 % of the total straw grown (total potential production). The amount of uncollected straw is highly driven by the markets, if the prices are uneconomic, straws are usually ploughed back into the soil (NNFCC, 2014; Defra, 2019; Tolvik, 2019). However, not all straws can be collected without significant iLUCs as some amount of straw is used to improve soil structure and nutrient status. Straw production in the UK is a regional market, with its main supply from the east of England (NNFCC, 2014).

Straw produced in the UK is typically used for animal bedding, animal feed, and power generation, as shown in Fig. 2. Assuming volumes of straw available in surplus and in power stations can be used for drop-in fuel production, the volume of straw available in the case of the competing application can be estimated as 2.9–3.6 Mtpa. Both the use of straw in power stations and drop-in fuel are in energy applications. Based on the data by Defra (Defra, 2019) and NNFCC (NNFCC, 2014), the total demand for and supply of straw is expected to remain stable up to 2030.

One of the key properties of straw is its low bulk density. The density of baled straw is in the range of 110–200 kg/m^3 while that of loose or chopped straw is 20–80 kg/m^3 (Satlewal et al., 2018). The low density of straw, even after balling, attracts a cost to the transportation of straw from farms. Straw has a moisture content of about 50 %, which reduces to less than 15 % after natural drying on fields (van Nguyen et al., 2016). The fixed carbon of straw is 11–19.5 % of dry mass, while volatiles is 35.0–72.4 % of dry mass (Montero et al., 2016; Satlewal et al., 2018). The LHV of straw is 14.3–15.6 MJ/kg on a dry basis (Teagasc, 2010; Bradna et al., 2016; Xia et al., 2019). Current bioenergy applications of straw in Europe are reported to have emissions varying from 16 to 55.8 g CO_2eq per MJ of electricity, based on a net electrical efficiency of 29 % (Giuntoli et al., 2013; Welfle et al., 2017). The cost of straw is reported in the range of £40–75 per tonne, depending on seasons and regions across the UK (AHDB, n.d.; Lewis Business Media, 2022).

3.1.3. Manure

Animal manure, made up mostly of animal faeces, is generated as a by-product of animal rearing including cattle, pigs, and poultry (NNFCC, 2019). The UK manure generation rates are estimated at 72 Mtpa to 90 Mtpa (Defra, 2009, 2011; Smith and Williams, 2016). The distribution of UK manure by sources is cattle (80 %), pigs (7 %), horses (5 %), poultry (5 %) and others (2 %), based on Smith and William (Smith and Williams, 2016).

Currently, almost all the manure produced is applied to agricultural land annually in the UK, with only a small amount used for AD, as shown in Fig. 3. Using manure for drop-in fuels instead of agriculture may attract significant emissions savings, up to -33.0 g CO_2eq/MJ fuel (-390 kg CO_2eq per dry tonne manure) as reported by Ou et al. (Ou et al., 2022) when consequential emissions are accounted for. However, a key challenge to accessing manure is its very low distribution. Hence, it can be assumed that 0–87.5 Mtpa of manure (excludes those used in AD) can be available in the competing case. Based on Defra's (Defra, 2022a) publication in 2022, there is no significant change in animal farming activities and policies recently that could impact animal faeces generation presently and in the future. Therefore, the total generation and utilisation outlook of animal manure is expected to remain flattened over the next few years, up to 2030.

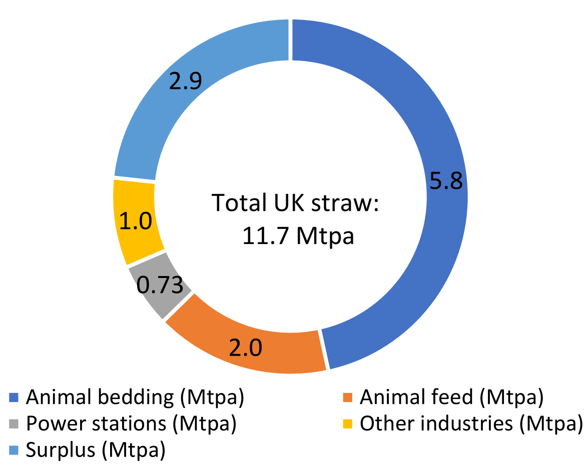

Fig. 2. UK straw production (Defra, 2019).

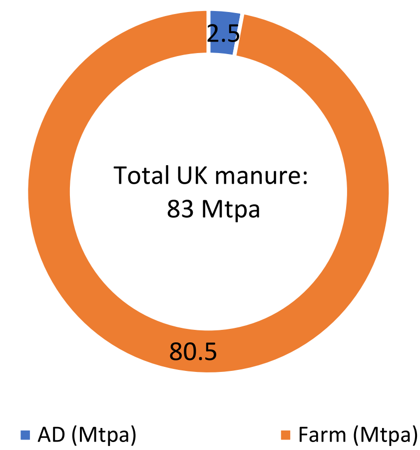

Fig. 3. UK manure availability in 2021 (NNFCC, 2021).

Manure is a very high moisture content feedstock, containing up to 96 % moisture (NNFCC, 2019), which makes it challenging for energy applications such as combustion and gasification. However, these have made it very suitable for wet biomass application processes like AD and HTL. Animal manure has a significantly high amount of volatiles and a low fixed carbon content on a dry basis, typically in the range of 62–74 % and 33–45 %, respectively (Cao et al., 2016; Tushar et al., 2016; Moghaddam et al., 2021). The LHV of animal manure on a dry basis is 14.4–16.8 MJ/kg, based on data from Sahu (Dávalos et al., 2002; Sahu et al., 2016). The GHG emissions for manure's land application stage range from 164 to 335 kg CO_2eq per dry tonne of manure, assuming a 10 % average manure moisture content (Aguirre-Villegas and Larson, 2017). Manure has a low-cost value and is estimated to have a value of up to £17 per tonne, based on the fertiliser replacement value of the nutrients it can contain (Alli Grundy, 2022).

3.1.4. Wood waste

According to Environmental Agency "wood, which is not virgin timber (that is, wood that has been used for any purpose) and associated residues such as off-cuts, shavings chippings and sawdust, either treated or not treated, is waste" (Environmental Agency, 2017). In 2019, wood waste generation was 5.0 Mtpa (Tolvik, 2019). Currently, the use of wood waste in the UK is significant, with only a small fraction left over as surplus, as shown in Fig. 4. Since both EfWs and drop-in fuel are energy applications, it can be assumed that the volume of wood waste available in the competing case is between 2.9 and 3.7 Mtpa. Wood waste outlook is expected to remain flattened over the next few years (Tolvik, 2019).

Wood waste has an average moisture content of 20 % and an LHV of 18.5–21.0 MJ/kg on a dry basis (Forestry Commission, n.d.; Huhtinen, 2005; Akhator et al., 2017). The GHG emissions of bioenergy (combined heat and power) from wood waste vary significantly depending on the grade of wood, ranging from 53.8 to 200 kg CO_2eq per MWh (14.9–55.6 g CO_2eq per MJ) for grade A and B wood waste and from 600.0 to 814.2 kg CO_2eq per MWh (166.7–226.2 g CO_2eq per MJ) for grade C and D. Grades C and D wood waste contains chemicals such as urea formaldehyde and urea melamine resin which contribute to their high GHG

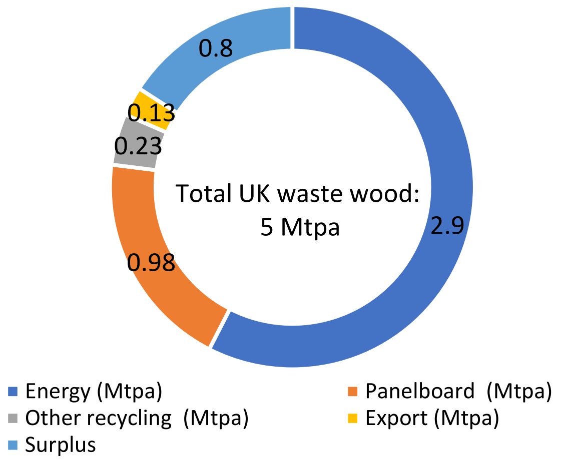

Fig. 4. UK waste wood availability and use (Tolvik, 2019; Defra, 2021).

emissions (Röder and Thornley, 2018). Also, the cost of wood waste is dependent on the grade of the wood waste, often attracting negative prices due to its waste disposal requirement. High-grade wood waste – which is clean and non-hazardous wood, majorly grade A to C wood waste such as softwood and process off-cuts from the manufacture of virgin/sawn timber – is reported to cost between -£5 to £20 in 2021 (letsrecycle.com, 2021b). Low-grade wood waste which includes treated wood costs between -£55 to -£5, as its disposal may be subject to hazardous waste regulations (letsrecycle.com, 2021b). In drop-in fuel application, using biomass, especially grade C and D of wood waste which contain chemicals, may cause additional operational complexity such as the increased risk of corrosion, catalyst deactivation and product contamination. Therefore, additional measures, for example, the use of corrosion-resistant alloys like nickel alloys, may be required to manage the varying nature of the feedstocks (Lee et al., 2016; Boukis and Stoll, 2021).

3.1.5. Sewage sludge

Sewage sludge is a by-product produced from the treatment of wastewater released from various sources, including homes, industries, medical facilities, street runoff and businesses. It has a significant amount of water compared to solids such as organics. Based on data by Nuamah et al. (Nuamah et al., 2012) and Ofwat (Ofwat, 2015), sewage sludge production in the UK was estimated at 1.4–1.5 Mtpa (dry), which is equivalent to 28–30 Mtpa (wet).

Sewage sludge disposal or recycling is typically done through spreading on agricultural land and incineration, as shown see Fig. 5. Several factors can impact sludge application on land (e.g., safety and

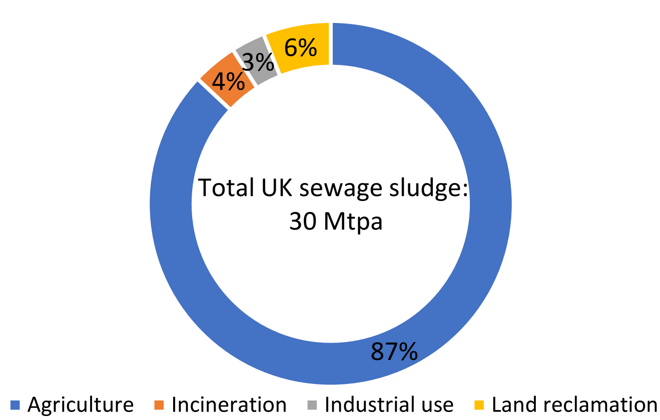

Fig. 5. UK sewage sludge generation and utilisation (Biosolids Assurance Scheme, n.d.).

cost implications). As sewage sludge can contain toxic elements (such as chromium) which are only safe if they're below set limits (Defra, 2018), sludge application on land can lead to the contamination of water and the transmission of pathogens via spreading sites or contaminated groundwater, drinking water wells, amongst others (Reilly, 2001). Furthermore, the cost implication of treating raw sewage is significantly high as it can cost up to £200 per dry tonne of sludge, only to be discharged almost free of cost (£1–2 per dry tonne) (Katsou, 2019). Therefore, using alternative treatment methods (e.g., HTL) can be key for optimum sewage sludge management. In HTL, digested sludge – which has been stripped of biogas and has not passed through high energy-consuming drying processes – can be used for drop-in fuel production. The forgone benefits such as fertiliser credits for using sewage sludge for drop-in fuel were estimated at 22.9 g CO_2eq per MJ (Lilonfe et al., 2024). The overall attributional and consequential GHG emissions of using sewage sludge for HTL in the UK were reported as 0–36 g CO_2eq per MJ and 59 g CO_2eq per MJ, respectively (Royal Society, 2023; Lilonfe et al., 2024). Hence, the competing case assumes all of sewage sludge can be available for drop-in fuel production.

Raw sewage sludge has a very high moisture content, with moisture content ranging from 84 to 96 % (Zhang et al., 2010; Chen et al., 2013; Marrone et al., 2018). On a dry basis, the average LHV of sewage sludge is 12–15 MJ/kg (Nuamah et al., 2012; Ostojski, 2018). Dry sewage sludge is often disposed of at almost free of cost, attracting selling prices between £1 to £2 per dry tonne (Katsou, 2019). Typical GHG emissions associated with the application of dry sludge on land can be assumed to be zero since almost all methane would have been stripped off from sewage sludge during its treatment in industrial AD and aeration plants.

3.1.6. Food waste

United Nations definition refers to food waste as food and related inedible portions removed from the human food supply chain sectors (United Nations, 2023). Based on the analyses by WRAP (WRAP, 2019a, 2020b), the total food waste generation in the UK is estimated at 11.1 Mtpa. Jeswani et al. (Jeswani et al., 2021) estimated UK food waste generation at 13.1 Mtpa, which is very close to the numbers by WRAP. Food waste generation in the UK is expected to continue to decrease, following the trend from 2007 and food conversation programs, which aim to reduce food waste by 50 % in 2030 compared to 2007 levels.

The competing use of food waste is high, with applications in areas such as biogas production, composting and recycling, see food waste SI Section 1.2 for details. However, a significant portion of food waste ends up in landfills. The volumes of food waste landfilled as part of BMW in 2020 can be estimated as 1.9 million tonnes, based on data from Defra for the fraction of food waste in MSW (Defra, 2017) and the volumes of MSW sent to landfills (Defra, 2022b). Drop-in fuel production from food waste which is landfilled in the UK can produce GHG emissions of -21.5 g CO_2eq per MJ by avoiding landfill. When the fuel production from food waste competes with existing applications like AD and composting, it is estimated to lead to indirect emissions of 22.8 g CO_2eq per MJ as food waste is diverted from existing applications (Lilonfe et al., 2024). The overall attributional and consequential GHG emissions of using food waste for drop-in fuel via HTL in the UK were reported as 33.7 g CO_2eq per MJ and 35.0 g CO_2eq per MJ, respectively (Lilonfe et al., 2024). Hence, considering existing energy and digestates applications of food waste, the volume of food waste available for drop-in fuel in the competing case can be estimated at 2–6 Mtpa.

Food waste has a very high moisture content, which ranges from 48 to 95 % (Amuzu-Sefordzi et al., 2014; Selvam et al., 2021). The LHV of food waste on a dry basis is 16.8–22.9 MJ/kg (Kadlimatti et al., 2019; Chae et al., 2020; Selvam et al., 2021). Like BMW, the GHG emissions from food waste usage are in the range of 0 kg of CO_2eq/t MSW for recycling to nearly 400 kg of CO_2eq/t MSW released from landfills (Nordahl et al., 2020). These emissions rates are dependent on the food waste management method used such as incineration, landfilling and composting. Food waste has a significantly high fraction of volatiles and a low fraction of fixed carbon, about 61–62 % and 6–23 %, respectively (Cao et al., 2016; Tushar et al., 2016; Moghaddam et al., 2021). The cost of food waste in AD is estimated to range from -£15 to £15 in 2021, according to letsrecycle.com (letsrecycle.com, 2021a). WRAP reported a similar average gate fee of £0 per tonne for food waste processing in AD plants (WRAP, 2022).

3.1.7. Digestates

Digestate, a by-product produced from AD, is a slurry waste consisting of fibre and slurry. It is rich in nutrients which makes it to be applied as fertiliser (NNFCC, 2016). The fibre contains a higher content of solids, making it more suitable for transportation, however, separation is not done on all AD sites (WRAP, 2020a). There was a total of 15.5 million tonnes of feedstock which were digested in 642 AD plants in the UK in 2021 (NNFCC, 2021). Using an average digestate yield of 92.5 %, an estimated 14.2 Mtpa of digestates were produced in the UK in 2021. There has been a significant increase in the number of AD sites, from 85 in 2013 to 642 in 2021. The number of AD plants is expected to continue to increase with the development of new sites.

Almost all the digestates produced in the UK are applied to soils, as shown in Fig. 6. WRAP (WRAP, 2020a) reported that 1.3 % of feedstocks are being removed during and after anaerobic digestion due to factors such as gross contamination and feedstock outside the input specification. Feedstocks including digestates from food waste can be contaminated by plastics, thereby, rendering their application on land inadequate (WRAP, 2020a). Digestates can find applications in drop-in fuel production due to their high moisture content. However, diverting digestates feedstocks from existing applications such as fertiliser and other energy recovery into drop-in production in the UK can produce consequential emissions of 24 CO_2eq per MJ (Lilonfe et al., 2024). The overall attributional and consequential GHG emissions of using digestates for drop-in fuel via HTL in the UK were reported as 0–36 g CO_2eq per MJ and 59 g CO_2eq per MJ, respectively (Royal Society, 2023; Lilonfe et al., 2024). Hence, in the competing case, it was taken that digestates can find applications in both fertiliser and drop-in fuel applications.

Digestate is a very high level of moisture waste, containing water in the range of 90–95 wt%. The LHV of digestates on a dry basis is 14.3–18.4 MJ/kg (Cao et al., 2019; Jurczyk et al., 2021; Taufer et al., 2021). Like dry sludge, typical GHG emissions associated with the application of digestates on land can be assumed to be zero since almost all methane would have been stripped off from digestates during AD treatment. Digestates have a high fraction of volatiles and low fixed carbon, about 79.8–85.1 % and 14.9–21.6 %, respectively (Cao et al., 2019; Jurczyk et al., 2021). The cost of digestates is reported by WRAP to be in the range of -£8 to £5 (WRAP, 2020a).

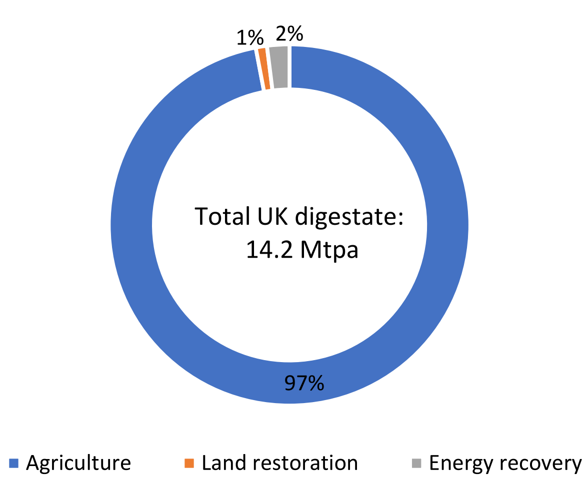

Fig. 6. UK digestates generation and utilisation (NNFCC, 2016).

3.1.8. Other feedstocks

Other biomass feedstocks produced small or in insufficient quantities in the UK, such as forest residues, miscanthus, short rotation coppice and crude glycerine, can be found in Table 2. Additional details on these feedstocks can be found in the SI. It is worthwhile to note that virgin wood which is produced in significant quantities in the UK is not sufficient to meet the UK national demand. The UK is strongly a net importer of virgin wood resources and imported a total of 20.6 Mt. of wood in 2020 (Forestry Commission, 2021).

3.1.9. Total volume of UK-origin non-food feedstocks

Table 2 summarises the UK-origin biomass feedstock availability. The total volume of UK-origin non-food biomass can be estimated as 839–1033 PJ per year (167–205 Mtpa) in 2021. This is significantly higher than the volumes published by DESNZ (DESNZ, 2023a) of 370 PJ per year for 2025 but agrees with the range of the volumes published by Supergen Bioenergy (Welfle et al., 2020) of 1014–2138 PJ per year for 2025. The volume reported by DESNZ (DESNZ, 2023a) is conservative and may likely represent the volume of biomass available after competing use. For example, in the breakdown of DESNZ (DESNZ, 2023a) volumes, the component of residual biogenic waste is 60 PJ per year. In this report, the total available potential volume of BMW is 330.4–382 PJ per year, with a minimum of 57.3 PJ per year available after competing use. The volumes presented by Supergen Bioenergy (Welfle et al., 2020) are for 2025 scenarios. Also, they considered some biomass resources that were not included in this report such as food crops and algae. This report considers only non-food biomass feedstocks that are of UK origin. Currently, algae production is negligible, but in the future, it may form a significant source of biomass in the UK.

3.2. Overview of drop-in fuel production

Drop-in fuels can be produced from non-food biomass feedstocks through two major routes, which can be classified into thermochemical and biochemical methods. The thermochemical methods usually involve a combination of thermal and chemical processes, such as gasification with Fischer–Tropsch synthesis (FTS) and pyrolysis with upgrading. Thermochemical routes often referred to as biomass to liquids (BTL), are fast and energy-intensive, occurring at moderate to high temperatures, see Fig. 7 and Table 3. Biochemical routes utilise microbes through processes such as AD and fermentation to break down biomass feedstocks into intermediates, which can be chemically converted to renewable hydrocarbons. Thermochemical methods have many advantages in fuel production over biochemical methods, such as (i) Fast reaction rate and plant start-up time, due to the short process completion time ranging from seconds to minutes, unlike biochemical methods which are relatively slow and often lead to the process plants having a very long start-up and plant stabilisation time of up to several weeks (Bridgwater, Meier and Radlein, 1999); (ii) Ability to handle a wide variety of feedstocks and a sudden change in feed composition without a significant process upset; (iii) Higher fuel conversion rates. However, unlike biochemical processes, thermochemical methods are usually characterised by (i) Higher energy demand due to the energy intensity of the processes, (ii) Undesirable tar and char formations which contribute to catalyst poisoning and corrosion, and (iii) Slag and salt depositions which can cause low heat transfer and corrosion (Kruse and Faquir, 2007; Kruse, 2008; Yakaboylu et al., 2015).

Energy consumption and integration of thermochemical processes can play an important role in fuel production yield and production price. Thermochemical processes with lower external thermal energy and lower external hydrogen demand would generally produce a lower product yield per biomass feed since some of the biomass feed and intermediaries would be consumed within the processes for hydrogen and heat generations. This can be seen from Fig. 7, which shows drop-in fuel production processes with higher energy consumption had higher product yield compared to processes with lower energy consumption.

Table 2
UK non-food biomass feedstocks availability.

Feedstock	Moisture content (%)	LHV on dry basis (MJ/kg)	Production, Mtpa (wet)	Competing case, Mtpa	Potential access for new competing uses	
					In 2021	In 2030 vs 2022
BMW	42–55	13.1–17.5	34.6–40 (19.6–24.6[a])	6–11	Good	Decreases
Straw	15	15–18.6	11–12	2.9–3.6	Good	Stable
Animal manure	78.5	14.4–16.8	72–90 (69.5–87.5[b])	0–87.5[b]	Good	Stable
Forest residues	50	15.8–21.4	2.7–4.1	~2.2	Good	Slight increase
Wood waste	20	18.5–21.0	4.5–5	0.8–3.7	Low	Decreases
Virgin wood	52	15.8–21.4	10.8	~0	Low	Stable
Sludge waste	95	12–15	28.2–30	28.2–30	Good	Stable
Food waste	58–95	16.8–22.9	11.1–13.1 (6.2–8.2[c])	2–6[c]	Good	Decreases
Digestates	90–95	14.3–19.6	14.2	0–14.2	Good	Increases
Miscanthus	16	17	0.08–0.13	~0	Low	Stable
Short rotation coppice	20	16.5–19	0.03–0.05	~0	Low	Stable
UCO	0	37	0.25–0.40	~0	Low	Stable
Black and brown liquor	82–85	9.0–13	0.28	~0	Low	Stable
Tallow	0	37.0–38.4	0.08–0.12	~0	Low	Stable
Crude glycerine	5–15	14.0–20	0.03	~0	Low	Stable

[a] Volumes of BMW excludes 10 Mt. MSW food waste and 5 Mtpa wood waste volumes to avoid double counting. See Sections 3.1.1 to 3.1.8 for details.
[b] Assumed volume of manure excludes 2.5 Mtpa used in AD to avoid double counting.
[c] Volume of food waste used in AD is removed from the volume of food waste available to avoid double counting.

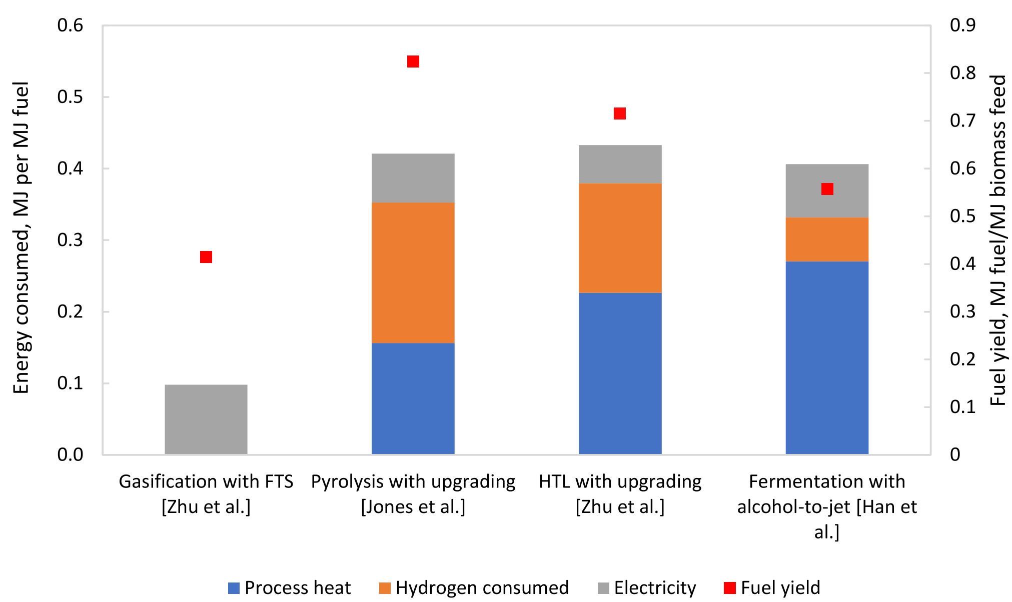

Fig. 7. Energy consumption of some thermochemical drop-in fuel production processes in comparison to biochemical-based processes, based on data reported by Zhu et al. (Zhu et al., 2011, 2014), Jones et al. (Jones et al., 2009), and Han et al. (Han et al., 2017). Process heat in these studies was supplied by natural gas, however, other low-carbon sources like biomass or green electricity can also be used for heating.

Table 3
Summary of key drop-in fuels thermochemical processing routes.

Process	Feed suitability	Conditions		Yield	Unit	Ref.
		Pressure	Temp.			
Gasification	Dry	Low	600–1300 °C	3–8.3 wt%(25–48.5 wt%)	H_2 (syngas)/kg dry feed	(Naik et al., 2010; Moghadam et al., 2014; Jaganathan et al., 2019)
Pyrolysis	Dry	Low	350–1000 °C	70–75 wt%	Bio-oil[a]/kg dry feed	(Bridgwater et al., 1999; Canabarro et al., 2013)
Supercritical water gasification (SCWG)[b]	Wet	22–35 MPa	350–800 °C	0.2–6 wt%	H_2/kg dry feed[c]	(Basu and Mettanant, 2009; Okolie et al., 2019; Yang et al., 2020)
HTL[a]	Wet	5–40 MPa	200–400 °C	20–80 wt%	Biocrude[b]/kg dry feed	(Zhang, 2016; Yang et al., 2020; Mishra et al., 2022)

[a] Bio-oil and biocrude which have similar properties to crude oil have energy contents of 13–18 MJ/kg and 16–19 MJ/kg, respectively.
[b] The operating boundaries of SCWG and HTL are different with SCWG using water at a supercritical state in contrast to the HTL operating with water at a sub-critical state. See Sections 3.2.3 and 3.2.4 for details.
[c] Methane is another valuable product produced in comparable quantity, 2–16 wt% CH_4/kg dry feed.

For biochemical processes like fermentation, the limitation is often driven by the reaction stoichiometry and subsequent biofuel upgrading pathway. As the energy demand of thermochemical processes can exceed 0.30 MJ/MJ of fuel produced, the energy penalty of thermochemical processes will maintain a very high influence on the fuel production yield and price. Therefore, heat integration can be critical to the success of drop-in fuel production. Heat integration of pyrolysis processes by Yang et al. (Yang et al., 2018) claimed up to 70 % reduction in energy consumption can be achieved in microwave-assisted pyrolysis. Also, Pedersen et al. (Pedersen et al., 2018) analysis showed similar benefits on energy demand following the heat integration of hydrothermal liquefaction of lignocellulosic biomass, as heating and cooling demand reductions of 55.2 % and 89.6 % respectively were achieved upon heat integration. Consequently, the impact of energy savings can have a significant reduction in the fuel production price, as more than a 20 % reduction in fuel production price was achieved by Yang et al. (Yang et al., 2018) in comparison to prices in the absence of heat integration.

This review considers key thermochemical methods for drop-in fuels from biomass, which can be sub-classified into four major pathways: gasification pathway, supercritical gasification pathway, pyrolysis pathway and HTL pathway. Table 3 shows some of the characteristics of the thermochemical drop-in fuel production methods.

3.2.1. *Gasification*

Gasification is a process which produces a gas mixture through the partial oxidation of carbon-based materials in the presence of oxidizing agents such as air, oxygen, and steam at high temperatures. The main product of gasification is synthesis gas, also known as syngas, which mainly consists of CO and H_2. The hydrogen gas yield is in the range of 30–83 (485) g H_2 (syngas)/kg feed, see Table 3. Gasification processes can be catalytic and non-catalytic. Catalytic gasification typically requires lower temperatures of around 900 °C compared to non-catalytic which requires as high as 1300 °C (Naik et al., 2010). Also, gasification is typically applied to low moisture content feedstocks due to the high energy requirements of drying higher moisture content feedstocks.

Drop-in fuel production through the gasification route involves the use of two or more key processes alongside, where the syngas from gasification is further upgraded into the final product. Processes used to upgrade syngas include FTS (Larson, Jin and Celik, 2009), Methanol-To-Gasoline (MTG) (Jones and Zhu, 2009), Topsoe Integrated Gasoline Synthesis (TIGAS) (Dimitriou, Goldingay and Bridgwater, 2018). As shown in Fig. 8, which summarises some of the various gasification pathways for drop-in fuel production, various liquid fuel products such as gasoline and diesel can be produced through this pathway.

3.2.2. *Pyrolysis*

Pyrolysis is a high-temperature process which decomposes carbon-based materials into smaller units in an oxygen-free environment, as shown in Table 3. Depending on the heating rate and temperature of the process, pyrolysis can be divided into fast and slow pyrolysis. Fast pyrolysis, which is usually employed in BTL, combines rapid heating of biomass, high temperature and short residence time of fewer than 2 s (Bridgwater et al., 1999). The high heating rate employed in fast pyrolysis is characterised by high heat transfer, making the use of fluidised bed reactors widely applicable in fast pyrolysis (Wang et al., 2005; Zhang et al., 2011). The product distribution is liquid (70–75 wt%), gas (10–15 wt%) and char (10–15 wt%). The liquid product often referred to as bio-oil is a highly viscous and complex mixture consisting of many organic compounds such as benzene, phenols and aldehydes (Mourant et al., 2013; Aysu and Küçük, 2014). It has a typical mass density and LHV of 1200–1300 kg/m^3 and 13–18 MJ/kg, respectively (Ringer et al., 2006). Like gasification, pyrolysis is suitable for low moisture content biomass feedstocks, due to the high energy demand associated with the drying of wet feedstocks before thermal degradation.

The production of drop-in fuels using the pyrolysis route is usually achieved through a combination of different processes, such as pyrolysis with upgrading (Anex et al., 2010) or pyrolysis with bio-oil gasification and upgrading (Li et al., 2015), where the bio-oil produced in pyrolysis is converted into liquid hydrocarbon fuels. Also, bio-oil can be co-processed (up to 20 %, by weight) with petroleum-intermediate feedstocks such as vacuum gas oil in existing refineries without triggering major plant modifications (Bhatt et al., 2020). Fig. 9 shows a schematic of the various drop-in fuel production pathways from the pyrolysis route.

3.2.3. *Supercritical water gasification*

Supercritical water gasification (SCWG) is a moderate temperature process which produces a fuel gas through the conversion of high moisture content biomass at high pressures. SCWG takes place in the presence of water, above its critical pressure of 221 bar and temperature of 374.29 °C, where the distinction of water between liquid and vapour phases does not exist (Heidenreich et al., 2016; Moghaddam et al., 2021). The water, by acting both as a reactant and catalyst, decomposes biomass feedstocks into a fuel gas which is mainly H_2 and CH_4. The hydrogen gas yield in SCWG is in the range of 1–30 mol H_2/kg dry feedstock (Basu and Mettanant, 2009; Okolie et al., 2019; Yang et al., 2020). In SCWG, biomass with a high moisture content that can exceed 90 wt% such as cattle manure and sewage sludge can be converted into drop-in fuels without employing the energy-consuming pre-treatment process of drying, which is required for processes such as gasification and pyrolysis (Heidenreich et al., 2016; Boukis et al., 2017). About 2.3 MJ/kg at atmospheric conditions is required to evaporate water in biomass feedstocks (Datt, 2011), and this can lead to negative net energy production in cases where the energy produced is less than the energy used for drying the biomass. Hence, SCWG is most suitable for high moisture-content feedstocks.

Like gasification, drop-in fuel production methods based on SCWG can employ processes such as SCWG with FTS (Campanario and Gutiérrez Ortiz, 2017), methanol synthesis with MTG, and gasification with TIGAS (Dimitriou et al., 2018). Fig. 10 shows a summary of the drop-in fuel pathways using SCWG.

3.2.4. *Hydrothermal liquefaction*

HTL is a process in which biomass is converted into a high-energy-density liquid product using pressurized hot water at sub-critical conditions (see Table 3). Typically, HTL is employed at high pressures of 5–40 MPa, moderate temperatures of 250–350 °C, and a short reactor residence time of 5–90 min. Like SCWG, HTL allows for the use of very high moisture feedstocks without the need for the energy-intensive

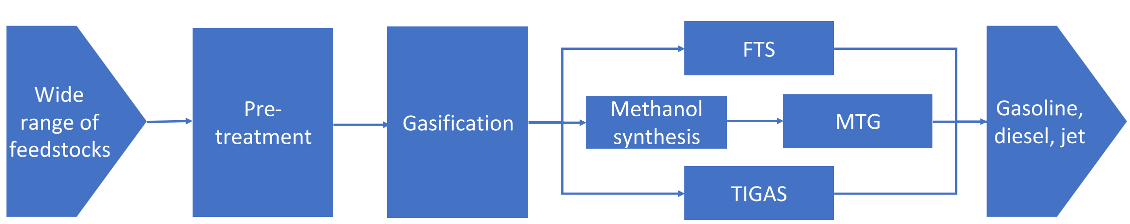

Fig. 8. Drop-in fuel production pathways via gasification pathway.

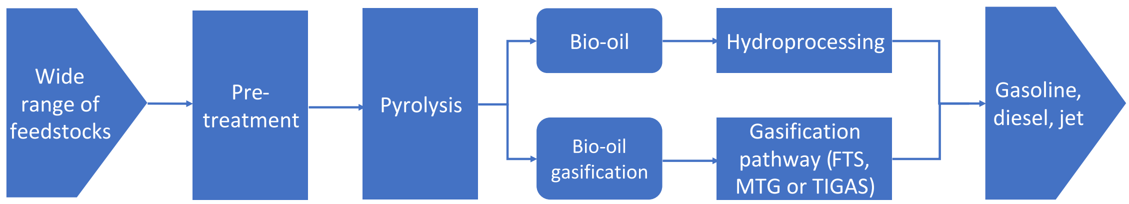

Fig. 9. Drop-in fuel production pathways via pyrolysis pathway.

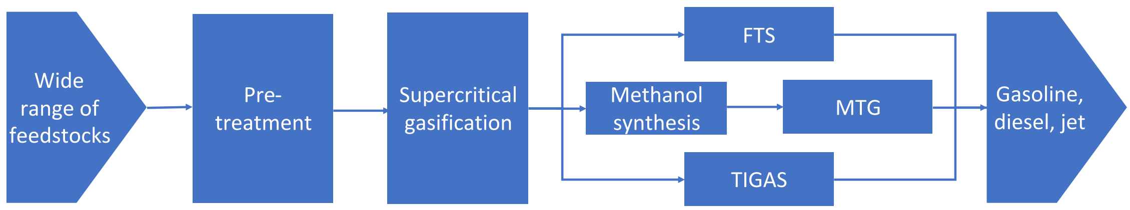

Fig. 10. Drop-in fuel production pathways via SCWG pathway.

process of drying. The liquid product, often referred to as biocrude, has a yield typically in the range of 20–80 wt% biocrude per dry mass of feed (Li et al., 2013; Mishra et al., 2022). Biocrude produced from HTL typically has a high mass density in the range of 910–1120 kg/m^3, which is similar to that of pyrolysis bio-oil. However, biocrude heating value of 30–40 MJ/kg is significantly better than those of pyrolysis bio-oil of 16–19 MJ/kg (Ringer et al., 2006; Yang et al., 2020). Like crude oil, the chemical composition of biocrude is complex, consisting of a mixture of many organic compounds ranging from fatty acids (myristic acid, palmitic acid, stearic acid, and oleic acid) to oxygenates (esters, ketones, alcohols, furans) (Yang et al., 2020).

Like pyrolysis, biocrude produced from HTL can be converted into drop-in fuel using hydroprocessing at dedicated refineries or by co-processing with petroleum in conventional refineries (Snowden-Swan et al., 2017; Bhatt et al., 2020). Co-processing of biocrude with petroleum intermediates in conventional refineries may be limited by technical requirements due to some unsuitable physio-chemical properties of biocrude like high nitrogen content. Also, biocrude can be gasified into syngas, where renewable hydrocarbon fuels can be obtained by further processing the syngas, although this process is not commonly practised mainly due to the high quality of biocrude and lower yield of gasification, see Fig. 11.

3.3. Hydrogen sources

In addition to thermal energy consumption, hydrogen is a key input in the thermochemical processes. The consumption of hydrogen is dependent on the drop-in fuel production routes, with pyrolysis bio-oil upgrading and HTL biocrude upgrading requiring 3.9–5.8 kg of hydrogen per 100 kg biocrude or bio-oil (Jones et al., 2014; Bennion et al., 2015; Jiang et al., 2019). Gasification routes may not directly consume hydrogen, as hydrogen is a key product in syngas, however, a water-gas-shift reaction is often used to improve the yield of hydrogen prior to fuel production using FTS. Hydrogen can be produced in situ or externally and imported into the plant for hydroprocessing. Generally, the production of hydrogen in situ in fuel production processes can cause serious impacts on fuel product yield and subsequently, the production price, especially when the biomass feeds are used in situ as sources of hydrogen. Based on Jones et al. (Jones et al., 2009), the use of biomass or bio-oil instead of natural gas for hydrogen production led to a 35 % reduction in the fuel product yield, as shown in Fig. 12. Also, approximately one-third of the biomass feedstock would be required for hydrogen generation in wood pyrolysis and upgrading, based on the 100 % renewable feedstock basis. Currently, other low-carbon sources of hydrogen are gaining market attention, such as green hydrogen produced from water electrolysis using renewables. This can improve the sustainability of drop-in fuel production.

The sensitivity analysis performed by many authors (Jones et al., 2014; Pedersen et al., 2018; Yang et al., 2018) indicated that the production price of drop-in fuels is highly dependent on the cost of

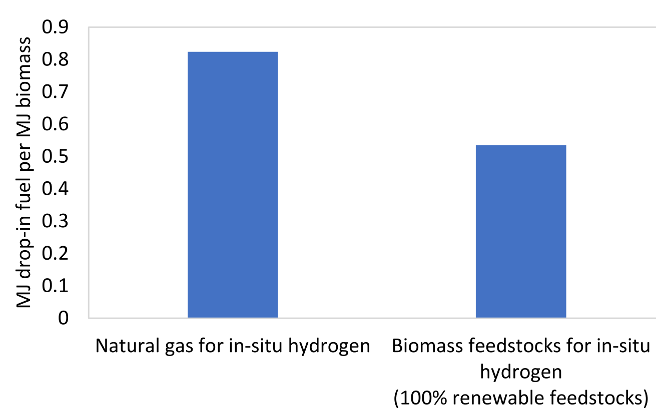

Fig. 12. Drop-in fuel product yield for different hydrogen sources in bio-oil upgrading facilities. Data from Jones et al. (Jones et al., 2009).

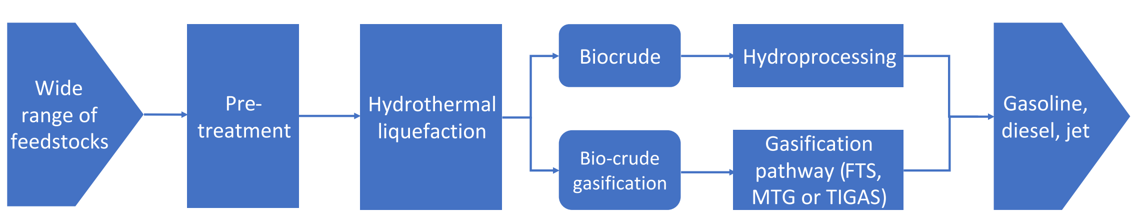

Fig. 11. Drop-in fuel production pathways via HTL pathway.

Table 4
Hydrogen sources, their cost, and emissions factors.

Hydrogen source	Cost, \$/kg		Emissions factors, g CO_2eq/MJ (LHV)		References
	2020	2050	2020	2050	
Natural Gas SMR (no carbon capture and storage [CCS])	0.7–1.6	–	83.6	82.0	(BEIS, E4tech and LBST, 2021; IEA, 2022b)
Natural Gas SMR with CCS	1.2–2.1	1.2–2.1	21.4	19.2	(BEIS, E4tech and LBST, 2021; IEA, 2022b)
Coal (no CCS)	1.9–2.5	–	203.3–229.2	203.3–229.2*	(Valente et al., 2021; IEA, 2022b)
Coal with CCS	2.1–2.6	2.2–2.5	64.2	64.2*	(Valente et al., 2021; IEA, 2022b)
Low-carbon electricity	3.2–7.7	1.3–3.3	0.1	0.1	(BEIS, E4tech and LBST, 2021; IEA, 2022b)

* Assumed negligible changed in emissions due to the fossil nature, similar to natural gas SMR.

hydrogen sources. Hydrogen from different sources has different prices and associated GHG emissions factors (see Table 4), and this may impact both the fuel prices and GHG emissions of the fuel since both fuel economics and GHG emissions are highly dependent on the material and energy inputs. Lilonfe et al. (Lilonfe et al., 2024) analysis of drop-in fuel showed that an additional fuel production cost of up to £5.40 per GJ can obtained from the use of low-carbon hydrogen. However, the GHG emissions benefits of using low-carbon hydrogen are better, with up to 40 % reduction in emissions relative to the corresponding grey hydrogen scenarios.

3.4. Production approaches: Centralised vs decentralised

The production of a product can be done in a single location as in the case of centralised production or in multiple locations where one form of processing occurs at a different location as in the case of decentralised production, see Fig. 13. A centralised production approach for drop-in fuels is achieved when both the feedstock conversion to intermediate products (such as syngas) and the conversion of the intermediates to drop-in fuels are done in a single facility. This approach traditionally offers a significant advantage in production cost because of the economy of scale, as often large plant scales are used in centralised fuel production. Also, it offers great opportunities for extensive heat integration and consequently potential reduction in energy cost, since off-gases produced from feedstock pre-treatment and conversion to intermediates can be used in the fuel conversion stage and vice-versa. However, the use of decentralised approaches in fuel production from biomass feedstocks has been suggested by some authors to offer better benefits in fuel production prices, especially in the reduction of biomass feedstocks transportation cost, which is typically significant due to the low bulk density and high moisture content of some biomass feedstocks such as straw and sewage sludge. Smaller plant scales are often used in a decentralised approach to meet the local demand and transformation of low-density and high moisture-content feedstocks into high-energy-density intermediates such as biocrude and syngas (Snowden-Swan et al., 2017). As a result, decentralised approaches are also suggested to offer better GHG emissions associated with the transportation of feedstocks, intermediates, and products. However, decentralised approaches leave little allowance for heat integration due to the nature of the segmented operations. Another key barrier in decentralised approaches is losing the economy of scale from the larger, centralised facility.

Despite the proposed benefits of centralised and decentralised approaches in drop-in fuel production prices and GHG emissions, not many extensive investigations have been made on the impacts of centralised and decentralised operations on fuel production cost. A study by Lilonfe et al. (Lilonfe et al., 2024) showed that the use of either a centralised or decentralised operation for optimising fuel production price is highly dependent on the availability and distribution of feedstocks in a location, as both the economy of scale and feedstock transportation plays a significant role in fuel economics. Based on the study, for biomass feedstocks with high moisture content, low availability and distribution, a centralised approach is more economical as the benefits of economy of scale outweigh the high feedstock transportation costs. However, above certain local availability and distribution of the feedstocks, 6000 kg/h of dry feed for wet biomass, the decentralised approach performs better than the centralised approach as the impact of transporting wet feedstocks becomes more dominant in the overall economics. Furthermore, the difference in GHG emissions from using a centralised versus decentralised approach in fuel production is negligible, as the emissions associated with the transportation of feedstocks make up only a small portion of the fuel's total GHG emissions.

3.5. Economic analysis

Drop-in fuels from biomass have attracted significant attention in recent years due to their environmental benefits and easy integration

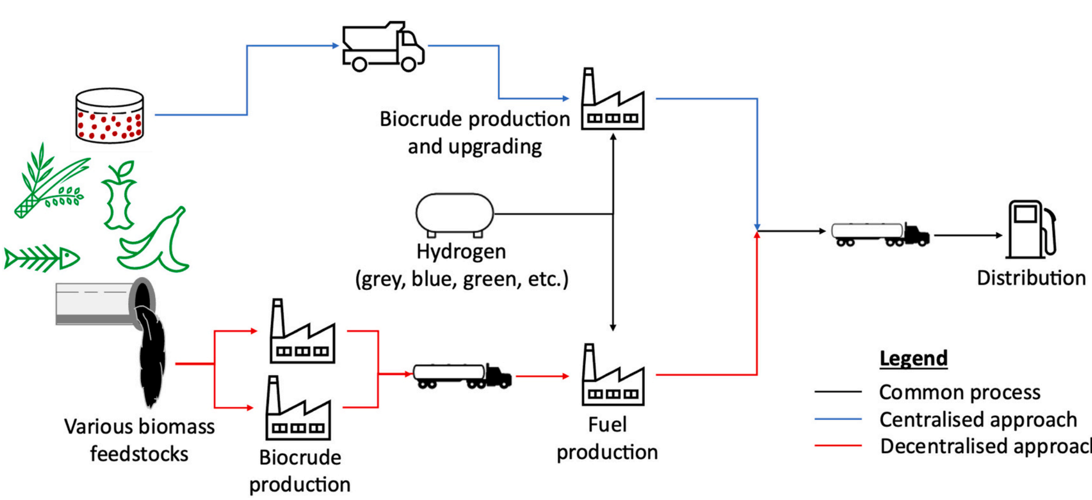

Fig. 13. Centralised and decentralised fuel production approaches in drop-in fuel production via HTL.

into existing vehicle engines and fuel supply chains. TEA is a tool for analysing the economic performance of industrial plants, processes, or products (Zimmermann et al., 2020). It is widely used in the estimation of the capital cost, operating cost, revenue, and product cost of drop-in fuels, based on technical and financial input parameters. These technical and economic data are particularly important and provide preliminary insights to inform plant design, construction, and operations of successful drop-in fuel plants. Numerous studies have been published on the TEA of BTL using various fuel production approaches, see Table 5. These studies reported minimum product prices for drop-in fuels in the range of $0.60–3.88 per gasoline litre equivalent (GLE) [$17.14–110.82 per GJ] when adjusted to the 2021 USD. Chemical engineering plant cost indices and 2021 exchange rate of £1 to $1.3496 were used where necessary, to adjust for changes in the original results, due to inflation and currency differences from the original analysis. The range of drop-in fuel prices, which are dependent on many factors, reflects the 2021 average price of conventional diesel fuels of $1.22 per GLE ($38.57 per GJ), based on 2021 wholesale prices of diesel published by RAC (RAC, n.d.). These fuel prices are highly impacted by factors such as the fuel conversion route, feedstock cost, transportation cost, tax rate, total capital investment, internal rate of return (IRR), product yield, plant scale, and hydrogen source. The optimal integration of these factors into the TEA is critical for the profitability and competitiveness of BTL.

Table 5
Summary of some important TEA studies on drop-in fuels via thermochemical processing routes.

Source	Technique	Product	Plant type	Capital cost (million USD_{2021})	MFSP (USD_{2021} per GLE)
(Michailos et al., 2017)	■ Feed input of 2200 t/d cane bagasse ■ Gasification with FTS	Drop-in fuels	No data	40–46	0.61
	■ Feed input of 2200 t/d cane bagasse ■ Fast pyrolysis with upgrading	Drop-in fuels	No data	33–50	0.66
(Li et al., 2015)	■ Feed input of 2000 t/d biomass/bio-oil ■ Fast pyrolysis, bio-oil gasification and FTS	Drop-in fuels	nth	685	1.66
(Zhu and Valkenburg, 2011)	■ Feed input of 2200 t/d wood chips ■ Gasification with FTS ■ Different gasifier heating scenarios considered	Drop-in fuels	nth	525–711	1.42–1.64
	■ Feed input of 2200 t/d wood chips ■ Gasification, methanol conversion and MTG ■ Different gasifier heating scenarios considered	Gasoline, LPG	nth	476–660	1.08–1.37
	■ Feed input of 2200 t/d wood chips ■ Fast pyrolysis with upgrading ■ Different gasifier heating scenarios considered	Drop-in fuels	nth	439	0.84
(Swanson et al., 2010)	■ Feed input of 2000 t/d corn stover ■ Gasification with FTS	Drop-in fuels	nth	725–870	1.56–1.72
	■ Two scenarios were considered- low and high temperature gasification, ■ Nth and first-of-its-kind plant	Drop-in fuels	first-of-a-kind	1594–2029	2.74–2.93
(Carrasco et al., 2017)	■ Feed input of 2000 t/d forest residues ■ Fast pyrolysis with upgrading ■ Laboratory demonstration and TEA of liquid fuels from forest residue via pyrolysis	Drop-in fuels	first-of-a-kind	584	1.85
(Tanzer et al., 2019)	■ Feed input of 500 t/d various agricultural residues ■ Gasification with FTS	Drop-in fuels	first-of-a-kind	281–358	1.90–3.88
	■ Feed input of 500 t/d various agricultural residues ■ Fast pyrolysis with upgrading	Drop-in fuels	first-of-a-kind	224–332	1.61–3.35
	■ Feed input of 500 t/d various agricultural residues ■ HTL with upgrading	Drop-in fuels	first-of-a-kind	268–370	1.50–4.27
(Dimitriou et al., 2018)	■ Feed input of 2000 t/d woody biomass ■ Gasification with FTS ■ Various BTL pathways	Drop-in fuels	No data	678–703	0.82–0.84
	■ Feed input of 2000 t/d woody biomass ■ Gasification, methanol conversion and MTG ■ Various BTL pathways	Drop-in fuels	No data	841–895	1.08–1.19
	■ Feed input of 2000 t/d woody biomass ■ Gasification, TIGAS ■ Various BTL pathways	Drop-in fuels	No data	661–711	0.90–0.92
(Rahbari et al., 2019)	■ Feed input of 16,000 t/y algae ■ SCWG with FTS ■ Solar-driven SCWG-reforming and FTS	Drop-in fuels	nth	172	2.69
(Campanario and Gutiérrez Ortiz, 2017)	■ Feed input of 60 t/h bio-oil aqueous phase ■ SCWG with FTS	Drop-in fuels	No data	85–189	1.08
(Zhu et al., 2014)	■ Feed input of 2000 t/d woody biomass ■ HTL with upgrading ■ Nth and first-of-its-kind plant	Drop-in fuels	first-of-a-kind	742	1.60
		Drop-in fuels	nth	678	0.90
(Pedersen et al., 2018)	■ Feed input of 1000 t/d organic matter ■ HTL with upgrading ■ Heat integration of model using pinch analysis	Drop-in fuels	No data	303	0.90–1.23
(Tews et al., 2014)	■ Feed input of 2000 t/d forest residue ■ HTL with upgrading ■ Comparative studies of fast pyrolysis and HTL	Drop-in fuels	nth	323	0.60
(Jones et al., 2014)	■ Feed input of 1349 t/d algae ■ HTL with upgrading	Diesel	nth	608	1.40

3.6. Life cycle assessment

Currently, many LCA studies have been done on the life cycle impacts of drop-in fuels (De Jong et al., 2017; Vienescu et al., 2018; Kolosz et al., 2020), however, these studies are heavily focused on the global warming potential (GWP) of fuels, neglecting other sustainability factors such as abiotic depletion (ADP) and human toxicity potential (HTP). GWP of fuels enables easy comparison of drop-in fuels with conventional petroleum fuels. The GHG emissions of drop-in fuels from various sources are estimated to be in the range of -122 and 98 g CO_2eq per MJ, in which the upper band is as high as those of fossil fuels (94 CO_2eq/MJ-UK and EU RED II reference numbers). The large variation is due to factors such as feedstocks and associated land use change impacts, fuel processing techniques, allocation methods employed to attribute impacts across multiple product outputs, system boundary definitions, and energy sources (Fig. 14 and Table 6). However, generally, when biomass sources are obtained sustainably with no or marginal LUC and iLUC impacts, the GHG emissions are usually significantly lower than those of petroleum fuels and, in some cases, can even be negative in value. It is important to note that current policies in the EU and UK require that advanced renewable fuels achieve at least 65 % GHG savings (or 70 % if it is renewable fuels of non-biological origins) relative to fossil fuels (DESNZ, 2023a). Fig. 14 presents the summary of various approaches adopted by researchers in LCA analysis of drop-in fuels from thermochemical pathways. Fuels produced via gasification typically have the lowest GHG emissions, followed by those produced via HTL and then pyrolysis, due to the different material and energy demands of the processes.

3.7. Demonstration, pilot and commercial plants

Several BTL plants are in their commercial, pilot or demonstration stages. These plants typically utilise dry feedstocks for drop-in fuel production, as shown in Table 7.

3.8. Estimating potential UK drop-in fuel and GHG emissions volumes from available feedstocks

As summarised previously in Table 2, the key available feedstocks in the UK are BMW, straw, animal manure, forest residues, wood waste, sludge waste, food waste, and digestates. A total of 156–195 Mtpa (745–939 PJ per year) of UK-origin biomass are produced from non-food crops, excluding virgin wood as the UK is a net importer of wood biomass. In this analysis, two scenarios were used to estimate the potential of drop-in fuel in the UK, namely: the 100 % feedstock case and the competing case. The 100 % feedstock case is a theoretical scenario that assumes that all the non-food biomass in the UK can be utilised in drop-in fuel production, except those already used in biofuel applications like UCO. The competing case considers the existing competition that may exist with the use of the feedstocks such as the potential cost and GHG emissions implications amongst the competing alternatives.

In estimating the fuel and GHG emissions potential, it is worthwhile to note that the use of either a centralised or decentralised approach in processing biomass feedstocks produces the same fuel volumes since the supply chain losses for both approaches were assumed to be zero. Also, the use of a centralised or decentralised approach offers a negligible difference in GHG emission, however, the fuel production prices from these approaches can vary significantly (Lilonfe et al., 2024).

The product yields of drop-in fuels from the available biomass feedstocks are shown in Table 8. These biomass feedstocks can be readily processed using various fuel production methods discussed in Section 3.2. The yield of drop-in fuels from these production methods varies by biomass and by the fuel production methods. The availability and distribution of the feedstocks, moisture content, and product yield are used to determine what processing techniques are suitable for the feedstocks. Generally, gasification and pyrolysis are more suitable for low moisture content feedstocks like straw, while HTL and SCWG are more suitable for high moisture content feedstock like manure. Also, priority was given to methods of producing fuels such as those through bio-oils or biocrude synthesis and upgrading, which typically have higher product yields compared to gasification with FTS. The biofuel production methods that are suitable for these feedstocks are evaluated in Table 9.

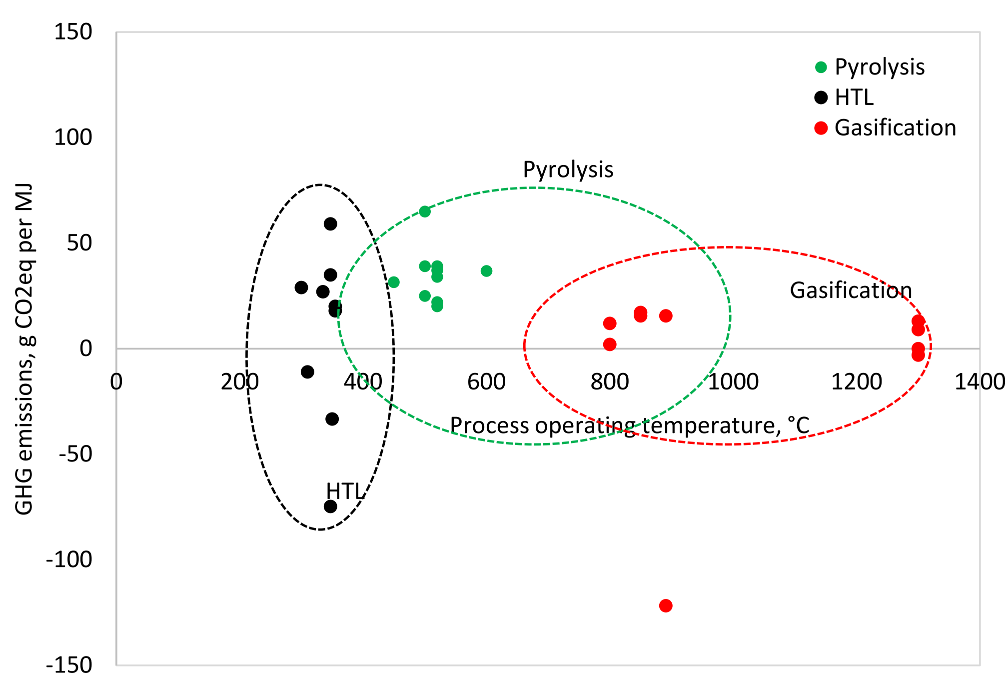

Fig. 14. The GHG emissions of drop-in fuels from key thermochemical processes. Negative net GHG emissions can be achieved incorporating CCS in the fuel production processes or avoiding severe consequential emissions sources.

Table 6
Summary of key LCA analysis on thermochemical fuel production routes.

Source	Techniques	LCA type	Metrics assessed	GHG Emissions (g CO_2eq/MJ)	Product	Other comments
(Bennion et al., 2015)	▪ Fuel path: HTL and pyrolysis of microalgae with hydroprocessing ▪ Boundaries: Well-to-Wheels analysis ▪ FU: 1 MJ of diesel ▪ Allocation method: Energy	Attributional	GWP	-11.4 and 210	Diesel	▪ Fuel emissions for HTL and pyrolysis routes respectively ▪ HTL had lower fuel emissions ▪ Major GHG emissions driver: Drying of microalgae feedstock
(Vienescu et al., 2018)	▪ Fuel path: Pyrolysis of corn stover and oil upgrading using various approach ▪ Boundaries: Well-to-Wheels analysis ▪ FU: 1 kg of biofuel ▪ Allocation method: Energy	Attributional	GWP	53.3 – 142.9	Liquid fuels	▪ Fuel emissions of bio-oil hydro-processing were the best, in comparison to other upgrading approaches ▪ Major GHG emissions driver: Electricity for energy demand
(Snowden-Swan and Male, 2012)	▪ Fuel path: Pyrolysis and hydroprocessing of woody biomass using various energy sources ▪ Boundaries: Well-to-Wheels analysis ▪ FU: 1 MJ of gasoline ▪ Allocation method: Energy	Attributional	GWP	31.5 – 36.8	Gasoline	▪ Fuel emissions from biomass power are lower than grid power ▪ Major GHG emissions driver: Power and natural gas usage
(Ou et al., 2022)	▪ Fuel path: HTL and hydroprocessing of swine manure ▪ Boundaries: Well-to-Wheels ▪ FU: 1 MJ of liquid fuels ▪ Allocation method: Energy	Consequential	GWP	-33.3	Liquid fuels	▪ Major GHG emissions driver: Avoided emissions from the current treatment method and fuel production stage
(Peters et al., 2015)	▪ Fuel path: Pyrolysis and hydroprocessing of hybrid poplar ▪ Boundaries: Well-to-Wheels analysis ▪ FU: 1 MJ of liquid fuels ▪ Allocation method: Energy and mass	Attributional	GWP, ADP, eutrophication potential (EP), acidification potential (AP)	39	Gasoline, diesel	▪ Major GHG emissions driver: Electricity consumption, especially from biomass dryer
(de Jong et al., 2017b)	▪ Fuel path: Jet fuel from various feedstocks via six fuel conversion technologies including Hydroprocessed Esters and Fatty Acids, FTS, HTL, pyrolysis, Alcohol-to-Jet ▪ Boundaries: Well-to-Wheels analysis ▪ FU: 1 MJ of jet fuel ▪ Allocation method: Energy and system expansion (displacement) methods	Attributional	GWP	-3 – 40	Jet fuel	▪ Thermochemical routes had the lowest fuel emissions. Emissions of gasification lower than HTL and HTL lower than pyrolysis processes ▪ Major GHG emissions driver: Feed cultivation and co-credit in the gasification route, while hydrogen use in HTL and pyrolysis
(Hsu, 2012)	▪ Fuel path: Pyrolysis and hydroprocessing of forest residues ▪ Boundaries: Well-to-Wheels ▪ FU: 1 km of travel by a light-duty passenger vehicle ▪ Allocation method: Energy	Attributional	GWP	25 – 39	Gasoline, diesel	▪ Fuel emissions from biomass-derived electricity lower ▪ Major GHG emissions driver: Fast pyrolysis stage
(Frank et al., 2013)	▪ Fuel path: HTL and hydroprocessing of algae ▪ Boundaries: Well-to-Wheels analysis ▪ FU: 1 MMBTU of diesel ▪ Allocation method: Energy	Attributional	GWP	29.4	Diesel	▪ Major GHG emissions driver: Hydrogen consumption in oil upgrading
(Tews et al., 2014)	▪ Fuel path: HTL and pyrolysis of woody feedstocks with hydroprocessing ▪ Boundaries: Well-to-Wheels ▪ FU: 1 MJ of liquid fuels ▪ Allocation method: Energy	Attributional	GWP	27 – 34	Gasoline, diesel	▪ Fuel via HTL had lower emissions than pyrolysis ▪ Major GHG emissions driver: Fuel production stage
(Masoumi and Dalai, 2021)	▪ Fuel path: HTL of algae with hydroprocessing, using biochar energy source ▪ Boundaries: Well-to-Wheels ▪ FU: 1 kg of fuel ▪ Allocation method: Energy	Attributional	GWP	-5.9 – -1.1	Liquid fuels	▪ Fuel emissions from the use of char for energy in HTL are lower than its use in the fuel catalysis ▪ Major GHG emissions driver: Methanol used as reactive organic co-solvent in HTL

(continued on next page)

Table 6 (*continued*)

Source	Techniques	LCA type	Metrics assessed	GHG Emissions (g CO_2eq/MJ)	Product	Other comments
(Tsalidis et al., 2017)	▪ Fuel path: Gasification with FTS of torrefied wood pellets, wood pellets and straw ▪ Boundaries: Well-to-Wheels ▪ FU: 1 km of distance travelled by a vehicle ▪ Allocation method: Energy	Attributional	GWP, EP, particulate matter potential, AP	15.6 – 17.2	Diesel	▪ Fuel emissions from torrefied wood pellets are lowest, followed by wood pellets and then straw ▪ Major GHG emissions driver: Gasification and gas cleaning
(Fernanda Rojas Michaga et al., 2022)	▪ Fuel path: Gasification of forest residues with FTS and CCS ▪ Boundaries: Well-to-Wheels ▪ FU: 1 MJ of jet fuel ▪ Allocation method: Energy and system expansion (displacement) methods	Attributional	GWP	-127.1 – 15.6		▪ Fuel production with CCS and allocation by system expansion methods had the lowest emissions ▪ Major GHG emissions driver: CCS and wood chips processes
(Sun et al., 2021)	▪ Fuel path: Fermentation to ethanol, pyrolysis to bio-oil, and gasification to jet fuel using corn stover ▪ Boundaries: Well-to-Wheels ▪ FU: 1 GJ biofuel consumed ▪ Allocation method: Energy and system expansion	Attributional	GWP, AP, EP, HTP and ozone layer depletion potential	19.9	Jet fuel	▪ Fuel emissions from fermentation were slightly lower than gasification, while pyrolysis had the highest emissions ▪ Major GHG emissions driver: Bio-oil upgrading in pyrolysis and fuel synthesis and refining in gasification

Table 7
List of some BTL plants.

Project, Location	Product	Key processes	Operation status	Type	Ref.
Sierra Biofuels Plant, USA	▪ Feed: MSW ▪ Capacity: 175,000 t/y of MSW ▪ Product: Drop-in fuels	Gasification, FTS	2022	Commercial	(Fulcrum BioEnergy, n.d.)
The COMSYN, Finland	▪ Feed: Various biomass ▪ Capacity: 800 t/y of biomass ▪ Product: Drop-in fuels	Gasification, FTS	2017	Demonstr. unit	(COMSYN, n.d.)
BTG Bioliquids, Netherlands	▪ Feed: Various biomass ▪ Capacity: 24,000 t/y of bio-oil ▪ Product: Bio-oil	Pyrolysis	2015	Commercial	(*Empyro Hengelo, NL, n.d.*)
TRI PDU, USA	▪ Feed: Various biomass ▪ Capacity: 4 t/y of biomass ▪ Product: Drop-in fuels	Gasification, FTS	2020	Demonstr. unit	(TRI, n.d.)

The potential GHG emissions associated with fuel production from the UK-origin biomass feedstocks, which are available in scalable quantities as shown in Table 9, were calculated using the emission factors in Table 10. Some of the wet biomass feedstocks like food waste may have GHG emissions of 35 gCO_2eq/MJ, producing emissions reductions of 63 % relative to a fossil fuel comparator of 94 gCO_2eq/MJ, which is just slightly missing the target mark of 65 % GHG emissions reduction relative to fossil fuel set by DESNZ (DESNZ, 2023a). Since the GHG emissions rates have some degree of uncertainties with the limited number of references available for these feedstocks, these numbers were nevertheless used in estimating the GHG emissions of fuel here. The fuel's GHG emissions from these feedstocks can be reduced by changing the supply chain or fuel production process. For example, by replacing the process heat supply from natural gas and grey hydrogen for hydroprocessing with renewables (Lilonfe et al., 2024). Using waste biomass feedstocks can have other additional impacts beyond GHG emissions as highlighted in Sections 3.1.1 to 3.1.7. Also, it is worthwhile for biofuel policies relating to the application of waste biomass feedstocks to consider the consequential impacts of utilising waste biomass feedstocks. Table 9 provides both attributional and consequential GHG emissions impacts of the non-food biomass feedstocks considered in this analysis.

3.9. *Drop-in fuel volumes*

There is a significant potential for the production of drop-in fuel from UK-origin biomass feedstocks. The volumes of drop-in liquid fuels, which can be produced from UK-origin feedstocks are estimated to be 563 PJ per year and 269 PJ per year, for the 100 % feedstocks and competing case scenarios, respectively, as shown in Fig. 15. Most of the potential fuel volumes originate from wet feedstocks, such as BMW, animal manure, food wastes and sludge wastes, which can provide more than 75 % of the total drop-in fuel potential. These wet feedstocks are produced by human activity in cities and due to agricultural practices across the different regions of the UK, and their utilisation for fuel production would not compete with food crops. Manure, which is the most available resource in the UK, produces the highest fuel volumes compared to the other feedstocks. Comparing these potential volumes with 2021 UK road transport fuel consumption, these volumes are 38.2 % and 18.3 % of the 2021 road transport fuel volume consumption, based on 100 % feedstocks and competing cases, respectively. DESNZ (DESNZ, 2023b) reported the UK road transport fuel consumption volumes of 1473 PJ (1557 PJ by gross heating value, equivalent to 42,094 million L) in 2021. This shows there is a significant potential for drop-in fuels production from non-food biomass feedstocks to meet a significant portion of the current UK demand for transport fuels. The target for renewable fuel volume in transport fuel consumption in 2030 is at least

Table 8
Product yield of various drop-in fuels production routes.

Feedstock	Gasification (MJ/dry kg feed)	Pyrolysis (MJ/dry kg feed)	HTL (MJ/dry kg feed)	Comment/reference
BMW	–	–	13.7	(Snowden-Swan et al., 2017; Chen et al., 2020; Lilonfe et al., 2024)
Straw	6.0–8.1 (8.1*)	–	–	(Larson et al., 2009; Anex et al., 2010)
Animal manure	–	–	11.9	(Snowden-Swan et al., 2017, 2020)
Forest residues	4.8–8.8	6.35–13.9	8.5–13.9 (13.9*)	(Jones et al., 2009; Zhu et al., 2011, 2014; Snowden-Swan and Male, 2012; Dimitriou, 2013; Tews et al., 2014)
Wood	4.8–8.8	6.35–13.9 (13.9*)	8.5–13.9	(Jones et al., 2009; Zhu et al., 2011, 2014; Snowden-Swan and Male, 2012; Dimitriou, 2013; Tews et al., 2014)
Sludge waste	–	–	11.3	(Snowden-Swan et al., 2017; Lilonfe et al., 2024)
Food waste	–	–	13.7	(Snowden-Swan et al., 2017; Chen et al., 2020; Lilonfe et al., 2024)
Digestates	–	–	12.9	(Snowden-Swan et al., 2017; Lilonfe et al., 2024)

* Value used in this analysis to estimate drop-in fuel potential.

14 %, based on RED II (European Commission, 2019). Our initial estimates indicate that, purely from a UK feedstock availability perspective, this can potentially be achieved using non-food biomass feedstocks.

3.10. Total drop-in fuel GHG emissions

There is a significant emissions reduction from the use of these non-biomass feedstocks in the UK. Based on attributional and consequential GHG emissions factors evaluation, GHG emissions savings of 39.2 and 64.4 Mt. CO_2eq per year can be achieved by using 563 PJ per year of these drop-in fuels in the 100 % feedstock cases, respectively, relative to fossil fuels. These emissions savings are equivalent to 39.4 % and 64.7 % of the UK's 2021 road transport GHG emissions of 99.5 Mt. CO_2eq (Tiseo, 2024), respectively. Also, in the competing cases, GHG emissions savings of 18.7 and 29.8 Mt. CO_2eq per year can be achieved by using 269 PJ per year of drop-in fuels, respectively. These emissions savings are equivalent to 18.8 % and 29.9 % of the UK's 2021 road transport GHG emissions. The various emissions associated with fuel produced from the biomass feedstocks and the use of conventional diesel to meet the remaining demand of UK 2021 road transport fuel consumption are shown in Fig. 16. While these drop-in fuels can supply up to 38.2 % of current UK road transport fuel demand, in addition to the current 4 % being supplied by first-generation biofuels, there is a need for supplementary fuels such as synthetic electro-fuels to balance the total fuel demand. In the future, alternative vehicle technologies such as electric vehicles may reduce the overall liquid fuel demand in the transportation sector, thereby, making it potentially possible to meet the UK transport fuel demand sustainably from biomass feedstocks.

3.11. Hydrogen demand

The demand for hydrogen is significant and critical for drop-in fuel applications. Total hydrogen demand from the production of drop-in fuels from UK non-food biomass is estimated at 38.5–75.6 PJ per year, which is roughly 13 % of the drop-in fuel production potential energy volumes, see Table 11.

4. Conclusion

Concerns over climate change, energy security, food price inflations, and forestry degradations have led to rising interest in the developments and use of advanced generation fuels, particularly for the hard-to-abate freight and aviation transport sectors. Sustainable drop-in fuels can be produced from a variety of sources, including second to fourth-generation biomass feedstocks, which leverage waste and residual biomass products with potentially lower environmental impacts. Thermochemical methods of producing these advanced biofuels such as gasification with FTS and pyrolysis with upgrading, can be integrated into existing refineries. Thermochemical methods enable the use of a wide variety of biomass feedstocks ranging from agricultural residues to high moisture content wastes like food wastes. The performance indicators of thermochemical processes, such as product yield and price,

Table 9
Selection of feedstocks and processing techniques.

Feedstock	Availability, Mtpa (wet)	Competing case, Mtpa	MC (%)	Suitable for			Available in scalable quantities	Selected for drop-in biofuel
				Gasification	Pyrolysis	HTL		
BMW	19.6–24.6	6.0–11	50	✓	✓	✓	✓	✓
Straw	11–12	2.9–3.6	15	✓	✓	×	✓	✓
Animal manure	69.5–87.5	0–87.5	79	×	×	✓	✓	✓
Forest residues	2.7–4.1	~2.2	50	✓	✓	✓	✓	✓
Wood waste	4.5–5	0.8–3.7	20	✓	✓	×	✓	✓
Virgin wood*	10.8	~0	52	✓	✓	✓	×	×
Sludge waste	28.2–30	28.2–30	95	×	×	✓	✓	✓
Food waste	6.2–8.2	2–6	75	×	×	✓	✓	✓
Digestates	14.2	0–14.2	90	×	×	✓	✓	✓
Miscanthus*	0.08–0.13	~0	16	✓	✓	✓	×	×
Short rotation coppice*	0.03–0.05	~0	20	✓	✓	✓	×	×
Imported forestry*	20.6	0	25	✓	✓	✓	×	×
UCO*	0.25–0.40	~0	0	–	–	–	×	×
Black and brown liquor*	0.28	~0	84	×	×	✓	×	×
Tallow*	0.08–0.12	~0	0	–	–	–	×	×
Crude glycerine*	0.03	~0	10	–	–	–	×	×

MC: Moisture content
* Biomass resource is not considered in drop-in biofuel applications due to its availability issues or existing biofuel application.

Table 10
Attributional, consequential GHG emissions and hydrogen demand factors used for the UK-origin feedstocks considered. GHG emission factors in g CO_2eq/MJ and hydrogen demand in MJ/MJ fuel.

Feedstock	Gasification		Pyrolysis		HTL		Selected route	External hydrogen demand	Reference
	Attributional	Consequential	Attributional	Consequential	Attributional	Consequential			
BMW	No data	No data	No data	No data	34	-69	HTL	0.15	(Lilonfe et al., 2024)
Straw	4	4*	25	25*	No data	No data	Gasification	0	(de Jong et al., 2017b; Tsalidis et al., 2017; Tanzer et al., 2019)
Animal manure	No data	No data	No data	No data	24	-33	HTL	0.15	(Ou et al., 2022; Royal Society, 2023; Lilonfe et al., 2024)
Forest residues	3	3*	31	31*	19	19*	HTL	0.15	(Snowden-Swan and Male, 2012; de Jong et al., 2017b; Tanzer et al., 2019)
Wood waste	3	3*	31	31*	19	19*	Pyrolysis	0.20	(Snowden-Swan and Male, 2012; de Jong et al., 2017b; Tanzer et al., 2019)
Sludge waste	No data	No data	No data	No data	18	59	HTL	0.15	(Royal Society, 2023; Lilonfe et al., 2024)
Food waste	No data	No data	No data	No data	34	35	HTL	0.15	(Royal Society, 2023; Lilonfe et al., 2024)
Digestates	No data	No data	No data	No data	18	60	HTL	0.15	(Royal Society, 2023; Lilonfe et al., 2024)

* For any selected fuel conversion route, consequential emission factor is used for attributional emission factor when no literature data is found on attributional emission, and vice-versa.

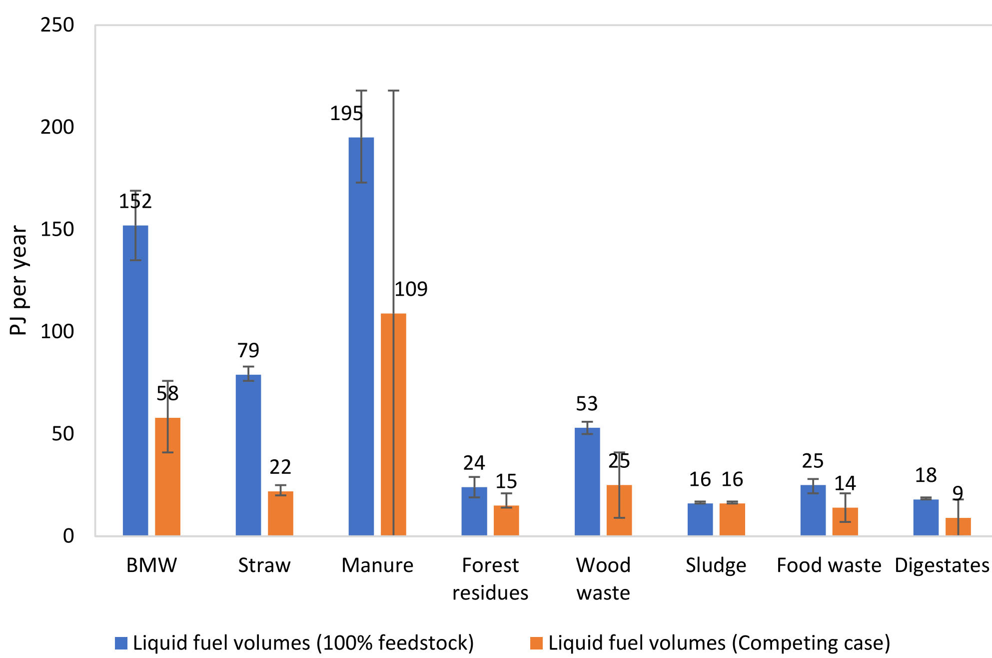

Fig. 15. UK potential liquid fuel supply. Error bars represent minimum and maximum fuel availability based on biomass resource availability range.

are highly dependent on energy and material consumption, as well as other process operation factors, such as production approach. Hence, several operational considerations, such as heat integration, hydrogen sources, and centralised and decentralised operations, are key to the profitability of thermochemical routes. Our review of the literature found the prices of drop-in fuels in the range of $0.60–3.88 per GLE [$17.14–110.82 per GJ]. The lower range of these prices showed that in some cases drop-in fuels can be competitive with the wholesale price of petroleum diesel of $1.22 per GLE ($38.57 per GJ).

Also, our review of LCA studies on drop-in fuels highlights the low-carbon potential of the pathways, and when produced sustainably, the GHG emissions of drop-in fuels can be much lower than those of petroleum fuels. The emissions from these fuels are in the range of -122 and 98 g CO_2eq per MJ. However, several key factors could impact the GHG emissions of these fuels, including the types of renewable feedstocks and energy sources, LUC and iLUC, LCA methodologies and foregone emissions.

The availability of feedstocks is an important factor for the successful operation of any process or production plant. Biomass is available in various regions in the UK, however, the availability is limited. The production of non-food UK-origin biomass feedstocks is estimated at 167–205 Mtpa (wet) [839–1033 PJ per year]. High moisture content feedstocks such as manure and sewage sludge, accounted for 140–173 Mtpa [523–650 PJ per year]. Other biomass feedstocks which are also

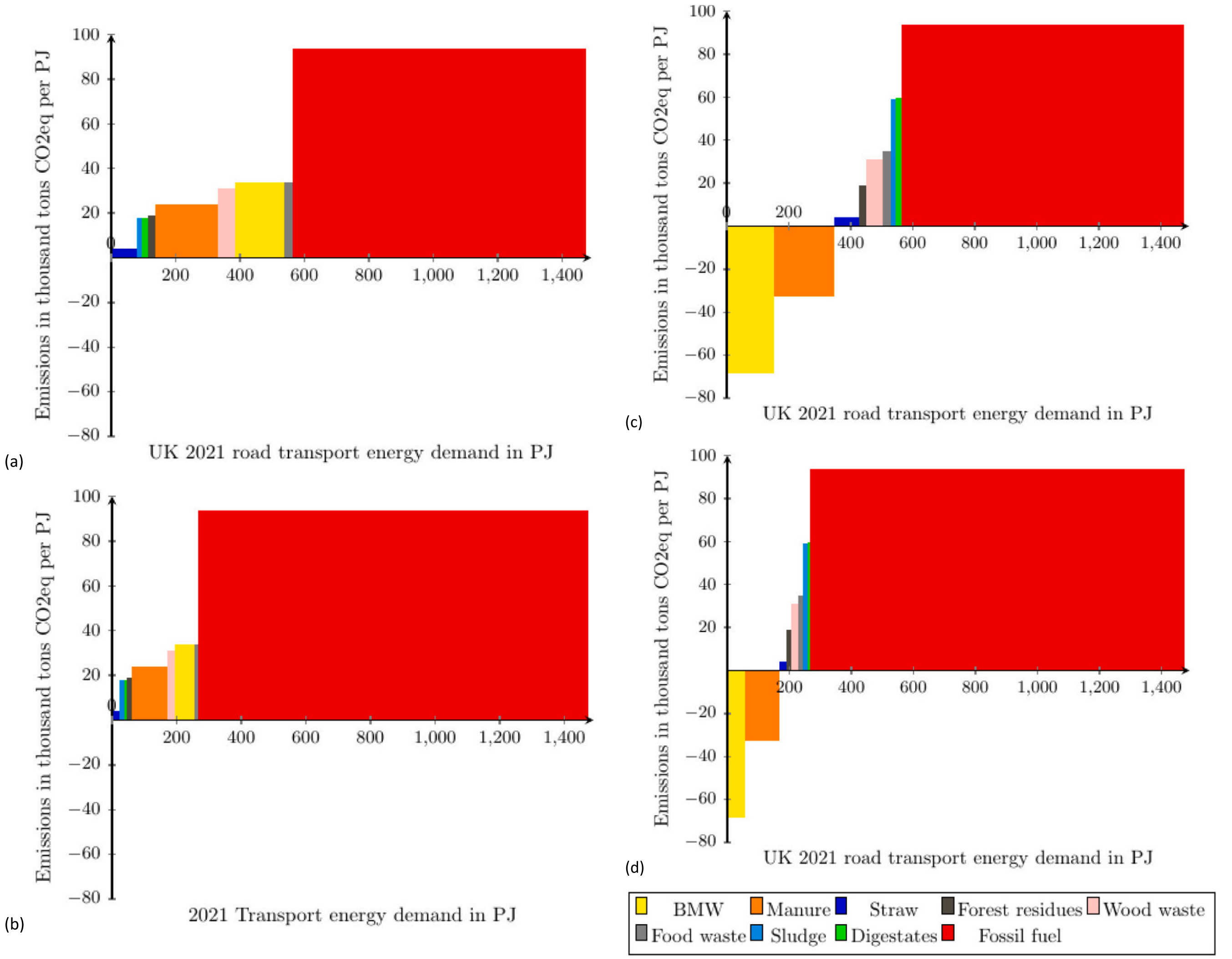

Fig. 16. a–d: Total GHG emissions associated with liquid fuel utilisation under (a) 100 % feedstocks utilisation, attributional (b) Competing case, attributional (c) 100 % feedstocks utilisation, consequential and (d) Competing case, consequential GHG emissions.

Table 11
Total UK drop-in fuel estimated potential production volumes and emissions.

	Fuel potential		Attributional emissions, Mt CO_2eq/year	Consequential emissions, Mt CO_2eq/ year	Hydrogen demand, PJ/year
	PJ/year	Millions L/year			
100 % feedstocks	563	16,076	13.7	-11.5	75.6
Competing case	269	7700	6.7	-4.4	38.5

produced in significant quantities include wood and straw. These feedstocks are subject to many competing uses such as bioenergy generation in biomass plants, panelboard, fertiliser applications and AD. However, wet feedstocks like BMW, food wastes, digestates are in excess, even after deducting for other competing uses, and are often disposed of without energy recovery.

Furthermore, it is estimated that a total of up to 563 PJ per year and up to 269 PJ per year can be produced from UK-origin biomass feedstocks, based on the 100 % feedstocks and competing cases, respectively. This fuel supply is dominated by fuel production from wet feedstocks such as manure and sewage sludge, which make up the majority of UK non-food biomass resources. Producing fuel from these waste resources could provide an alternative method for waste management, thus, partly addressing the challenges accompanying current waste management systems.

The total GHG emissions savings from the use of non-food biomass feedstocks as drop-in fuels in ICEVs can significantly reduce the UK transport sector's GHG emissions. Our initial estimate indicates that the emissions savings from the use of drop-in fuels in the UK could reach up to 64.7 % of the UK's total road transport GHG emissions in 2021. Thus, there is strong potential for the application of BTL in the UK.

Importantly, this study highlights the potential of sustainable drop-in fuels in general as an option for GHG mitigation for the UK's transport sector. More broadly, based on the insights of this review, we would recommend that policymakers also consider encouraging rapid developments and uptake of a wide range of advanced, low-carbon fuels as a complementary strategy to decarbonise the global transport sectors. LCA, and other comprehensive techno-environmental assessment tools, should be used to guide policy decisions towards achieving a decarbonised transport future.

CRediT authorship contribution statement

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

Declaration of competing interest

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Appendix A. Supplementary data

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