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Waste-to-energy and waste-to-hydrogen with CCS: Methodological assessment of pathways to carbon-negative waste treatment from an LCA perspective

Massimiliano Materazzi *, Suviti Chari, Alex Sebastiani, Paola Lettieri, Andrea Paulillo

Department of Chemical Engineering, University College London, London WC1E 7JE, UK

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ABSTRACT

A growing global population and rising living standards are producing ever greater quantities of waste, while at the same time driving ever-larger demand for energy, especially electricity, or new emerging markets, such as hydrogen in more industrialised countries. A key solution to these challenges of waste disposal, rising energy and hydrogen demand is BECCS (Bioenergy with Carbon Capture and Storage); the generation of bioenergy – in the form of electricity (WtE) or hydrogen (WtH$_2$), as well as heat – from the thermochemical processing of waste. The addition of carbon capture and storage (CCS) to WtE or WtH$_2$ has the potential to make waste a zero or even negative emissions energy source, thus contributing to the removal of greenhouse gases from the atmosphere. This work undertakes a pre-screening of different BECCS configurations based on state of the art technologies and then performed an assessment of representative cases in UK for WtE and WtH$_2$, necessary to understand if novel waste thermal treatment processes may become potential alternatives or improvements to current WtE plants when retrofitted with CCS. A systematic and comprehensive examination of different key Life Cycle Assessment methodological aspects reveals the importance of the functional unit and allocation approach in determining the preferred pathway in a specific context.

1. Introduction

According to estimates by the European Environmental Agency, emissions from waste management (excluding energy recovery) have decreased by 42% between 1995 and 2017 and the sector will continue to play an important part in the EU's ambition to achieve net zero by 2050.

The technologies for recovering energy from "residual waste" (i.e., remaining municipal solid waste left after the recycling and recovery operations and from source segregated collection) can play a critical role in mitigating the environmental issues associated to waste disposal (Di Maria et al., 2015). Aside from the valuable product, these technologies can result in a large decrease in the overall amounts of material requiring final disposal. This allows for simpler management in a controlled way while still adhering to pollution control regulations (Materazzi & Foscolo, 2019). A host of technologies are available for realizing the potential of residual waste as an energy source (as power or fuel), but the availability and general composition of waste affect the technologies that are suitable to deliver environmental benefits. For example, anaerobic digestion (AD) of crops, agricultural residues, and organic waste has grown strongly over the last years in Europe and worldwide (Scarlat et al., 2018). However, the potential of conventional AD is limited by the availability of suitable feedstock. New technologies that can process a wider range of materials, including lignocellulose and multi-component plastics, are required to deliver net zero and potentially contribute to decarbonise most carbon-intensive sectors, such as heat, power and heavy transport (heavy-good vehicles - HGV, shipping and aviation). To this end, thermochemical routes offer the most promising expansion in the coming years (Chen et al., 2022; Materazzi & Lettieri, 2017).

Thermochemical technologies have historically been used to produce heat and electricity (Waste-to-Energy, or WtE) via incineration of the waste feedstock, alone or together with other fuels (Makarichi et al., 2018). Electricity is generated from waste through direct combustion, with the heat used to produce steam to drive a turbine. Modern plants have an overall energy efficiency close to 20–30%, and the electricity generated is rapidly transmitted to the grid to meet the energy demand (Themelis, 2006). WtE plants have certainly a role to play in supplying

* Corresponding author.
E-mail address: massimiliano.materazzi.09@ucl.ac.uk (M. Materazzi).

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both heat and power, although many incineration plants in the world still do not operate in combined heat and power mode (CEWEP, 2022; Scarlat et al., 2019). Having more WtE plants to produce, in addition to electricity, heat for heat networks would substantially reduce their emissions by making use of the otherwise wasted heat to displace gas boiler heating. This will support a shift from using high-carbon gas generation to lower carbon generation in heat networks. Furthermore, discussion is ongoing to implement WtE with carbon capture and storage (CCS) systems connected to the stack, which not only offer a further reduction of CO_2 emissions but also, potentially, a way to sequestrate carbon from the atmosphere thanks to the high content of biogenic carbon in waste (Bisinella et al., 2021; Torvanger, 2021). For this reason, Bioenergy with Carbon Capture and Storage (BECCS) operated on residual waste has the potential to generate valuable renewable energy while delivering negative emissions, which will be critical to achieve Net Zero in the coming years (Almena et al., 2022).

At the same time, new technologies offer a valid alternative to WtE and post-combustion BECCS, in virtue of higher conversion efficiencies (i.e. valuable energy output in form of hydrogen or fuels over total energy input as feedstock) and the potential to step up in the waste hierarchy towards the more favourable recycling pathway. These advanced thermal technologies (ATT), such as gasification and pyrolysis, do not burn the waste, but instead they decompose it to its fundamental chemical blocks, CO, H_2 or hydrocarbons, from which new high-value materials and products can be generated (Arena & Ardolino, 2022; Chen et al., 2022). Due to the high-temperature nature of these processes (and the relatively high oxygen content in waste), CO_2 is still produced (either internally by partial combustion of the feedstock, or externally from the combustion of auxiliary fuels), making CCS a valid complementary technology for them too. In this case, CO_2 is removed from the syngas (usually after gas conditioning) before final utilization, delivering important energy savings to the system thanks to the relatively high concentration of CO_2 in the gas stream. In this sense, the maximum potential of process efficiency and carbon sequestration is achieved with the thermochemical treatment of waste for hydrogen production, which has attracted a lot of interest in the last few years (Amaya-Santos et al., 2021; Chari et al., 2023; Guo et al., 2022; Taipabu et al., 2022). Waste-to-hydrogen via gasification (WtH_2) can offer efficiencies in the range of 40 % to 60 % representing an effective means of hydrogen production, and leaving the entirety of the carbon content in waste available for capture and storage (Lui et al., 2020, 2022). As such, WtH_2 with CCS promises to be a very proficient and effective way to decarbonise carbon-intensive sectors (heating and gas-fired manufacturing, mostly) while removing substantial quantities of biogenic carbon from the atmosphere. Not surprisingly, H_2BECCS is seen by many governments as the most disruptive technology to achieve those negative emissions that will be necessary to offset hard-to-decarbonise sectors in the coming years in the UK (Mac Dowell et al., 2022).

However, ATT alternatives do not automatically guarantee the higher sustainability of the whole system (including energy utilization chain) (Ning et al., 2013; Wang et al., 2015). The stringent requirements of the catalysts for hydrogen production demand very extensive gas cleaning, which can add significantly to the energy, environmental and operational costs of the plant. Furthermore, the scale and configuration of different plant layouts, as well as the integration opportunities with the power, gas, and heat networks in the vicinity of the plant may also influence the overall environmental impact (Astrup et al., 2015).

A comprehensive assessment of different BECCS process configurations is necessary to understand if novel ATT based processes may become potential alternatives or improvements for the current WtE plants when retrofitted with CCS. However, the comparison between different routes is challenging, and needs to be contextualised within the geography of the plant, as well as the market and social conditions.

Guided by ISO standards (ISO 1997), life cycle assessment (LCA) is a powerful tool to quantify environmental impacts, which are not limited to climate change. It can also help identify the most critical steps in the whole life cycle of a product or service, and provide a benchmark for technologies comparison. LCA has provided reliable evaluation of MSW treatment technologies (Ardolino et al., 2023; Bianco et al., 2022; Cherubini et al., 2009; Morselli et al., 2008; Thushari et al., 2020), and BECCS (Almena et al., 2022; Fimbres Weihs et al., 2022; Hammar & Levihn, 2020). Dong et al. compared four operating plants (incineration, pyrolysis, gasification, and gasification-melting) from an LCA perspective (Dong et al., 2018). The study showed that the heterogeneity of MSW and syngas purification technologies are the most relevant impediments to the current ATT options for energy production. Their work also identified potential new developments that could revert the conclusions by incorporating into all process aspects to boost energy efficiency, improve incoming waste quality, and achieve efficient residue management. More recent studies have analysed the integration of WtE and CCS, albeit preliminarily, focussing on the main technical issues related to the application of carbon capture on WtE flue gases (target CO_2 removal efficiency, role of flue gas impurities, importance of process integration with the energy recovery and the flue gas treatment sections of the plant), also examining economic and societal issues (Dal Pozzo et al., 2023c; Magnanelli et al., 2021; Wienchol et al., 2020).

However, not many works consider alternative ATT and CCS configurations and, more importantly, compare them from an LCA perspective in the context of waste management and CO_2 removal. The first objective of this work is to review the most important technical challenges and latest developments of BECCS plants operated on residual waste (at commercial or close-to-commercial stage) to provide a common basis for comparison in relation to carbon negative technologies. Key aspects of the LCA methodology for BECCS evaluation are then discussed, including for example functional unit, system boundaries, and allocation between co-products. Finally, some of these aspects are further investigated using as exemplary models two different residual waste BECCS routes, one for hydrogen and one for electricity production in UK.

2. Technical appraisal and commercial considerations

2.1. Waste as a feedstock

The waste hierarchy principle states that when products do reach their end of life, recycling and reuse have priority over alternative methods. Waste that is not reused or recycled, including material that is too degraded or contaminated for these purposes, has historically been sent to landfill and grate incineration (including WtE) or transformed for energy recovery as refuse derived fuel (RDF) or solid recovered fuel (SRF) to be processed in different (often more efficient) thermal technologies, sometimes located in different countries (Fruergaard & Astrup, 2011).

SRF (a slightly more refined feedstock compared to RDF) is mixed solid waste that has been pre-treated (separated, dried and shredded) and it consists largely of combustible components such as unrecyclable plastic and biodegradable waste in fluff or densified pellet form depending upon fuel transport, storage, and feeding arrangements for a particular process (Nasrullah et al., 2014). Much of the ferrous and non-ferrous recyclable material in the original waste is removed and sent to be recycled as part of pre-treatment. This feedstock is much less variable than “black bag” residual waste and, as such, is much more suited for ATT reactors (most often employing fluidised beds), which are typically way more sophisticated (and, to some extent, less flexible) than mass burn incinerators.

The term “SRF” arises from work undertaken by the European Commission under CEN/343 to provide a systematic basis for the classification and standardization of fuels derived from non-hazardous wastes, to facilitate trade between producer and user, for informing process design, environmental permitting, etc. The use of a standardized approach to fuel specification is also a means by which uncertainties in the fuel supply chain can be addressed (Cozens and Manson-Whitton, 2019). The chemical composition of a typical solid recovered fuel is reported in the Supplementary Material, as well as the main physical and chemical parameters that would affect the LCA of any waste-based process. From an LCA perspective, the following points are deemed by the authors the most critical in determining the sustainability (environmental and financial, at least) of waste as a feedstock, especially for BECCS:

- Waste preparation site, whether this is at or near the ATT facility or imported ready for use from remote facilities. This is particularly relevant to plants fuelled with RDF/SRF feedstock.
- Waste heating value. It dictates the amount of energy (or valuable energy products, such as syngas) that can be extracted from the feedstock.
- Waste inorganic (ashes) content and composition. They control the amount of recoverable metals that can be separated from bottom ashes, as well as the quantity of residual materials that might need further treatment for safe disposal. The gas cleaning section is also highly affected by the presence of inorganic contaminants, such as sulphur, chlorine and heavy metals.
- Waste moisture level. It controls the amount of energy that is dissipated (i.e. cannot be recovered) during the thermochemical process. A significant consideration in this respect is that the thermochemical process itself will have an abundance of low-grade waste heat with which further drying could be accomplished.
- Waste biomass content. In BECCS it becomes necessary to understand the bioenergy content of the fuel to properly calculate the climate change impact and to estimate the potential of removing CO_2 from the atmosphere. This has also a significant impact on the operating cost of the plant because in many cases it is the bioenergy content alone that controls renewable energy incentives and carbon credits/taxation.

2.2. BECCS: Current status and technology development trends

Prior to the recent developments in EU and US for hydrogen and methanol production from waste, the most common perception of BECCS applied on waste management was that being simply the capture and storage of CO_2 recovered from the flue gas of a conventional incinerator, which is indeed still one of the prime focus of waste operators. For a BECCS-enabled WtE to remain commercially viable, carbon revenues and gate fees would need to become the principal source of income rather than power sales, due to the significant parasitic energy demand from the post-combustion capture of CO_2, coupled with the significant capital cost of post combustion facilities. Recent reports show that internal use of heat and electricity in WtE needed to sustain the CCS plant would take up around 40% of the energy in the waste fuel depending on the capture rate, halving the potential revenues from selling the power (Bisinella et al., 2021). As such, WtE with CCS facilities have an inherent economic exposure to gate price and other policy driven value streams fluctuations and it is unclear how these might evolve in the future.

Furthermore, in order to maximise BECCS' benefits to both society and the WtE plant operator, the facility would need to operate on base load, with lower prices for power output and therefore potentially lower profits. This is an operating condition that contrasts with one of the conceptual benefits of biomass/waste power – namely the ability to provide spinning reserve in a power market that is increasingly influenced by intermittent renewables (Yamaka et al., 2022).

By comparison, the potential future use of waste for the production of chemicals, biofuels or hydrogen coupled with BECCS offers some distinct advantages. First of all, sales prices for high value gas or liquid products (most often higher in cost per MJ of energy compared to electricity) are not impaired by operating in a BECCS mode. Most of ATT products are gases and liquids (e.g. methanol, SAF, hydrogen, etc.) that can be stored (long term or short term) to enable load following, and hence enhance energy sales revenues, whilst simultaneously allowing 24/7 base load operation to optimise BECCS performance. In fact, unlike electricity hydrogen or other chemicals do not need to be sold as soon as they are generated and the plant operator can exploit commercial advantage from arbitrage between the power generation and valuable product markets.

In the case of WtH_2 there is another great advantage: there is no apparent impairment of energy output (product hydrogen) resulting from operating in a BECCS mode. In fact, separation of the associated CO_2 occurs inevitably when producing clean hydrogen, and does not impose any additional parasitic energy demand on the host plant (the CO_2 is at all effects an incidental by-product to the hydrogen).

However, technical risks of emerging ATT technologies are still quite significant and this cascades to a slower progress on what is in theory a much better solution to waste management. The barriers to WtH_2 BECCS, despite the conceptual advantages described herein, are not only associated with market immaturity and near term lack of CCS and hydrogen offtake infrastructure; investors in the sector face risks connected with technological novelty of the thermal treatment process as well as the common entry barriers for CCS generally.

2.3. Thermal treatment processes

Post-combustion BECCS systems in a waste-fuelled WtE plant do not exist yet at industrial scale, although the technologies used for both waste incineration and for post combustion capture are mature and each at a state of development where they could be classed as commercially proven; that is to say at a Technology Readiness Level (TRL) of 9. Hence the technology risks associated with applying BECCS to WtE are moderately low.

By contrast the production of hydrogen via gasification of waste is not a mature technology. Examples of commercial scale waste gasification plants are available worldwide, but mostly for CHP applications via two-stage combustion processes (see Table 1). This category does therefore fall into the WtE concept, despite it has some major advantages against conventional grate incineration, including higher combustion efficiencies, recovery of metals in non-oxidized form, collection of ashes in inert-vitrified form and lower generation of some pollutants (Lombardi et al., 2015; Zhang et al., 2021). Most of these plants are in operation in Japan, licensed by Nippon Steel (as largest supplier), Kobelco-Eco, JFE, Hitachi Zosen, Ebara, Mitsui Engineering & Shipbuilding, and few others.

The state of technology development for biomass or waste gasification has recently been reviewed by the Department for Business, Energy & Industrial Strategy (BEIS) in UK (AECOM, 2021).

The key technological aspects of ATT operated on waste are summarised as follows:

- Most initial waste-fuelled gasifiers were developed as air-blown rather than oxygen blown. Air-aspirated gasifiers entrain large volumes of nitrogen in the syngas – the removal of nitrogen from the product (hydrogen, biomethane etc.) being expensive and difficult to accomplish (Materazzi et al., 2019).
- Waste gasification for chemicals production would be ideally undertaken at pressure above atmospheric, to avoid costly syngas compression downstream for catalytic applications. This introduces technical challenges for waste feeding and reactor operation.
- Compared to pure biomass, RDF and SRF introduce a greater concentration and diversity of contaminants, due to the high number and variability of sourcing points. This presents a major gas cleaning challenge, compounded by the fact that catalytic processes for H_2 and other chemicals production have very low tolerances.
- Waste feedstock is prone to production of significant amount of tars in conventional fluidised bed gasifiers, requiring additional thermal reforming (within the reactor or in a separate unit) before standard gas cleaning techniques (Materazzi, 2017). The capital and

Table 1
Advanced Thermal Technologies for waste treatment and final products. Highlighted cells indicate those assessed in this work (AECOM, 2021; Lombardi et al., 2015).

Primary technology Platform	Intermediate 1	Secondary technology platform	Intermediate 2	Tertiary technology platform	Final product	CO_2 capture type	TRL
Direct combustion (WtE)	Heat	Rankine cycle	None	None	Heat, Electricity	Post-combustion	9
Gasification	Syngas	Syngas combustion	Heat	Rankine cycle	Heat, Electricity	Post-combustion	9
		CCGT	None	None	Electricity	Post-or pre-combustion	9
		Allam cycle	None	Turbo-generator	Electricity	Post-or pre-combustion	9
		Catalysis	Hydrogen	CCGT	Electricity	Pre-combustion (intrinsic)	9
				Furnace, cookers, chemical plants	Heat, chemical feedstock	Pre-combustion (intrinsic)	7
				Fuel cells	Electricity	Pre-combustion (intrinsic)	6
			Methane (SNG)	Boiler, furnace	Gas fuel	Pre-combustion (partial)	8
				ICE	Heat, electricity	Pre-combustion (partial)	8
			MeOH	ICE	Liquid fuels	Pre-combustion (minor)	8
				Catalysis	Chemical feedstock	Pre-combustion (minor)	8
			FT liquids	ICE	Gasoline, Diesel, SAF	Pre-combustion (minor)	8
			FT gases	Catalysis	Olefins	Pre-combustion (minor)	7
Pyrolysis (high T)	Syngas, oils	Catalysis	Liquid hydrocarbons	ICE	Heat, Electricity	Pre-combustion (minor)	7
			Light hydrocarbons	Catalysis	Olefins	Pre-combustion (minor)	7

operational cost of reforming units can be of the same order of magnitude of the gasifier and share the same TRL.

- Compared to fossil feedstock used historically for hydrogen and chemicals production, waste (as well as biomass) contains large quantity of oxygen which makes control of C:H ratio in syngas for chemical synthesis more complicate.

Some novel developments have addressed (fully or in part) the above issues. Examples are given by companies like TRI, Enerkem, ABSL, Kew Energy and Repotec (AECOM, 2021). However, most of these solutions have been tested either at reduced scale (demonstration or semi-commercial), or for limited time (Materazzi et al., 2023). Up-scaling to a full commercial capacity is needed to prove satisfactory and sustainable performance. However, such a plant would be a first-of-a-kind facility and as such be seen by potential investors as presenting an enhanced technology risk, in comparison with technologies that had already accrued an operational track record and a TRL of 9. At the current time investors in WtH_2 projects are faced therefore with procurement of gasification technologies that are at around TRL 7 or 8 for clean syngas production, from which hydrogen, as well as other products such as methanol or SAF, would be manufactured. Technologies for production of valuable chemicals from syngas are well proven and at TRL 9, so the primary technological risk rests with the gasification and syngas polishing technologies.

2.4. CCS process and integration

Whilst carbon capture from the gas product of a thermochemical plant operating on waste is not yet a common practice, the technologies used for carbon capture are mature and each at a state of development where they could be classed as or near to commercially proven, especially for post-combustion configuration.

Table 2 reports a summary of CC technologies at TRL >/= 7, potentially suitable for both incineration and gasification-based plants (with some minor variation between the two), and their development stage.

Since most technologies are already at TRL 7 or higher, the technological risks associated with applying BECCS to WtE or WtH_2 are moderate. However, integration challenges cannot be ignored, especially when retrofitting an existing plant, and some process modifications or additions might be needed, some of which could add significant burdens from an LCA perspective. This is particularly relevant to WtE, since typical environmental regulations for air emissions are often less stringent than the contaminants limits imposed by catalytic applications in gasification plants (see Table 3).

Plant modifications could include changes in gas cleaning units or their operation, changes in chemicals (and effluents) handling, energy supply, the extent of downtime hours to interconnect the equipment and for maintenance, as well as the spatial area necessary to build and assemble the carbon capture section of the plant.

When an existing WtE plant is retrofitted with a CCS system, the flue-gas pre-treatment is a critical step. In fact, most liquid solvents used in the CCS may be affected by the flue gas composition. SOx and NOx can react with amine absorbents, forming heat-stable salts, which are difficult to regenerate and increase the solvent consumption for CO_2 capture (Porter et al., 2017; Wall et al., 2013). Particulate matter (PM) can cause

Table 2
Technologies for CCS and reference projects (Kearns et al., 2021).

Liquid solvents	TRL	Reference projects
Traditional amine	9	Widely used in fertilizer, soda ash, natural gas processing plants, e.g. Sleipner, Snøhvit, and used in Boundary Dam since 2014
Physical solvents (Selexol, Rectisol)	9	Widely used in natural gas processing, coal gasification plants, e.g. Val Verde, Shute Creek, Century Plant, Coffeyville Gasification, Great Plains Synfuels Plant, Lost Cabin Gas plant
Benfield (and similar)	8–9	Fertiliser plants, e.g. Enid Fertiliser. Demonstration plant for BioSNG/H2 production in Swindon (UK)
Sterically hindered amine	8–9	Demonstration to commercial plants depending on technology providers, e.g.Petra Nova carbon capture
Chilled ammonia	7	Pilot tests to demonstration plant feasibility studies
Solid sorbents		
Pressure Swing Adsorption / Vacuum Swing Adsorption	9	Air Products Port Arthur SMR CCS
Temperature Swing Adsorption (TSA)	7	Large pilot tests to FEED studies for commercial plants
Sorption enhanced water gas shift (SEWGS)	7	Large pilot tests to FEED studies for commercial plants (TNO)
Solids looping	7	Large pilot tests to FEED studies for commercial plants
Membranes		
Gas separation membranes for natural gas processing	9	Petrobras Santos Basin Pre-Salt Oil Field CCS
Polymeric membranes	7–8	Large pilot tests to FEED studies for commercial plants

Table 3
Contaminant thresholds for incineration and gasification plants and compatibility with CCS (Ardolino et al., 2020; IEAGHG, 2020; Zwart, 2009).

Contaminant	Emissions limit (2010/75/EU, IED)	WtE stack emissions (BAT)[1]	Syngas for Catalytic applications limits[2]	CCS limit[3]
Total Sulphur (SO_2, H_2S, COS)	50 mg/Nm^3	5–40 mg/Nm^3	<0.003 mg/Nm^3	25 mg/Nm^3
NOx (as NO_2)	200 mg/Nm^3	50–150 mg/Nm^3	n.a	35 mg/Nm^3
HCl	10 mg/Nm^3	2–8 mg/Nm^3	<0.001 mg/Nm^3	0.4 mg/Nm^3
HF	1 mg/Nm^3	0.5–0.9 mg/Nm^3	<0.001 mg/Nm^3	8 mg/Nm^3
Metals (Cu, Pb, As, Hg, Cd, Zn, Cr, etc.)	0.5 mg/Nm^3 total 0.05 mg/Nm^3 (Cd, Hg)	0.1–0.5 mg/Nm^3 total 0.005–0.02 mg/Nm^3 (Cd, Hg)	n.a	n.a
Total dust (PM1-10)	30 mg/Nm^3	5–10 mg/Nm^3	<10 mg/Nm^3	30 mg/Nm^3
VOC (as TOC)	10 mg/Nm^3	5–10 mg/Nm^3	<30 mg/Nm^3	n.a

[1] 50 MW_{th} WtE plant with 10% O_2 in flue gas.
[2] Copper based catalyst.
[3] Amine based solvent.

equipment blockage and foaming of the liquid absorbent. The equipment that are mostly subjected to changes are the deNOx and deSOx processes. As far as NOx emissions are concerned, the majority of existing WtE use the combination of SNCR with flue gas recirculation, which reduces NOx by ~70% (Dvořák et al., 2011). However, the low NOx concentration of inlet of CO_2 absorber can be reached only with a selective catalytic reformer (SCR). In WtE plant that are integrating CO_2 capture systems, as Alkmaar in Rotterdam (Netherlands) and Oslo Fortum (Norway), they have considered placing a SCR after the dust removal unit. For deSOx, in the carbon capture context, the requirement of very low sulphur concentration usually entails higher reagent consumption and waste-product generation, and sometimes mechanical modifications to the system to make it more efficient (Dal Pozzo et al., 2023b; Zhu et al., 2023). Other acid gases, such as hydrogen halides (mainly HCl and HF) are also present in the flue gas from waste incineration, often in quantities higher than sulphur. However, the release of the new European Commission reference document on the Best Available Techniques (BAT) for waste incineration has set already ambitious targets for the control of the emission of pollutants, and many modern plants are already compliant and below limits for CCS integration (see Table 3) (Ardolino et al., 2020; Dal Pozzo et al., 2023b). This improved performance of flue gas treatment systems in WtE facilities is, however, often associated to an increase of additional indirect environmental impacts related to the increased consumption of reactants and to the increased generation of process residues/wastewater in flue gas treatment (Dal Pozzo et al., 2023a).

Pre-combustion capture (called so with reference to IGCC plants were the syngas is eventually combusted) refers to removing CO_2 from syngas in reducing conditions, typically post water gas shift stage in a gasification or pyrolysis plant. The same concept applies to blue-hydrogen production plant, where syngas is generated from reforming of natural gas, so the technology risks are shared between the two low carbon hydrogen pathways. Since the syngas has already gone through extensive gas cleaning to preserve water gas shift catalysts, which are very sensitive to contaminants such as sulphur and chlorine, the plant does not typically require additional polishing for CCS integration. Compared to post-combustion technology, which removes dilute CO_2 (~5–15% CO_2 concentration) from flue gas streams at atmospheric pressure, the post-shift syngas stream is rich in CO_2 (30–60%) and often at higher pressure, which allows for easier removal (Antonini et al., 2021). Due to the more concentrated CO_2 (a direct consequence of steam-oxygen gasification), pre-combustion capture typically is more efficient, but the capital costs of the base waste gasification process and gas cleaning sections are often more expensive than traditional WtE plants.

3. LCA methodological aspects

3.1. Goal and functional unit

The definition of the goal of an LCA study is not only the first step of the methodology but arguably also the most important in that it affects all other steps (ISO, 2020), including the definition of the Functional Unit (discussed in this Section) and of the system boundaries (Section 3.2), and selection of allocation approach (Section 3.3), inventory data sources (Section 3.4) and comparative scenarios (Section 3.6).

With the exception of a few, out-dated technologies like landfill and incineration without energy recovery, most advanced technologies for waste management are multi-functional: they deliver additional functions to that of managing waste, including for example recovering or recycling materials for new uses, generating energy (electricity and/or heat) and producing valuable products, such as hydrogen; notably, the development of multi-functional waste management technologies follows the waste management hierarchy (European Commission, 2008) and circular economy principles (Morseletto, 2020) by aiming to reduce the environmental impacts of waste as well as the extraction of new resources.

The multi-functionality of waste management technologies entails that LCA practitioners may adopt different perspectives, focusing on the function of managing waste, of producing a product or both. The study's perspective affects the definition of the Functional Unit which is a quantified description of the function of the system analysed and is particularly important to guarantee the equivalency of alternative systems being compared (ISO, 2020). The choice of the perspective/functional unit must be aligned with the goal of the study; for example, if the LCA goal is to identify the most environmentally preferable technology for disposing a given type of waste in a given region, the perspective/functional unit need to focus on waste management. This means that WtE or WtH_2 technologies may be assessed on the basis of how efficiently they manage waste or produce energy or H_2, respectively. In addition to this, the function of capturing/sequestering CO_2 may also be relevant for waste management plants coupled with CCS when the goal of the study is to identify the most environmentally advantageous technology for permanent sequestration of carbon (for example, for comparison with Direct Air Capture and Storage, DACS).

Historically, LCA studies on WtE (but also landfill, compositing, anaerobic digestion) focused on the waste management function, with functional units typically set on a unitary basis (e.g. 1 tonne waste) or the total production of waste in a region/country (Iqbal et al., 2020; Laurent et al., 2014b). The historical perspective adopted is likely reflecting the main perceived function of WtE plants. However, recent developments have shifted the focus on the product with LCA studies on WtH_2 plants mostly focusing on the function of H_2 production (Amaya-Santos et al., 2021; Antonini et al., 2021; Khojasteh Salkuyeh et al., 2017) possibly reflecting the higher added-value of H_2 compared to electricity and thermal energy, and enabling comparison with other low-carbon hydrogen production routes (e.g. electrolysis).

3.2. System boundaries

The system boundaries, which define which processes in the life cycle are included (or excluded) in the analysis, must also align with the goal of the study, including depending on whether an attributional or consequential approach (e.g. see Finnveden et al., 2009) is adopted.

In attributional LCA studies on waste management, the system boundaries typically start either at the collection of the waste or at receipt of waste in the plant. Note that the activities in the life cycles producing the waste are not included because the waste is assumed to be burden-free (Gentil et al., 2010) and because it would be impractical to model all activities that generate a given type of waste. A small portion of studies do not include the waste collection phase on the basis of (i) perceived irrelevance and (ii) commonality in a comparative analysis (Iqbal et al., 2020; Laurent et al., 2014b). The system boundaries typically end at the production of new valuable products (e.g. electricity, heat, H_2) which are inputs to other life cycles and often replace equivalent products on the market. For waste management plants coupled with carbon capture, the system boundaries need to cover the capture phase as well as transportation, storage and injection when the capture CO_2 is assumed to be permanently sequestered. The system boundaries can be expanded for allocation (discussed in Section 3.3) or comparative (Section 3.6) purposes, particularly to include additional activities that make a product (e.g. H_2) fully equivalent (and thus substitutable) to another that is available on the market. The construction and decommissioning of waste management plants are rarely included in the system boundaries (Iqbal et al., 2020; Laurent et al., 2014b); this is also due to perceived irrelevance, but possibly also because most studies focus on the operational phase.

Consequential LCA studies need to expand the system boundaries to include effects of decisions, which include first order effects (i.e. direct substitution) as well as second and third order effects (Sandén & Karlström, 2007). The former are borrowed from neo-classical economics and deal with supply and demand, whilst the latter is borrowed from the theory of technical change and cover cumulative build-up of stocks and structures (e.g. such as physical structures, institutions, actors) that lead to altered availability and cost of technologies as well as to changes in preferences.

3.3. Inventory data

The use of high-quality primary data for the foreground system is key to accurately estimate the life-cycle environmental performance of any systems; but it is arguably more important for waste management technologies because the environmental performance is strictly linked to properties of waste and scale of the plant. There are specialised LCA software to model waste technologies, e.g. EASETECH (Gentil et al., 2010), but accurate inventory data can only be obtained via process simulations, for example via Aspen Plus (e.g. see Amaya-santos et al., 2021; Antonini et al., 2021).

The inventory data for the background system is typically based on commercial LCA databases like ecoinvent (Wernet et al., 2016) but the type of data depends on whether an attributional or consequential approach is adopted: the former relies on average data whilst the latter on marginal data (Finnveden et al., 2009). The underlying rationale is that only the technologies that are most likely to respond to changes in demand or supply that are consequences of decisions resulting from the study should be included in consequential LCAs (Weidema et al., 1999). Ekvall and colleagues show how taking into account the consequences of decisions for waste incineration significantly affect the LCA results. They argue that an increase in plastic waste incinerated in Sweden leads to lower waste exports from Europe resulting in either increased incineration or landfill in EU, because the waste incineration capacity in Sweden is constrained in the short term; their LCA results are significantly affected by the scenario considered (Ekvall et al., 2021).

3.4. Allocation approaches

Since waste management systems are typically multi-functional, their environmental impacts need to be allocated between their different functions when the intended application of the LCA study is to compare alternative systems for the delivery of a given function (ISO, 2020). However, when the focus of the study is on an individual system, allocation can be avoided by expanding the functional unit to cover all functions delivered by the system.

The ISO standards recommend that, when allocation cannot be avoided by subdividing the system into sub-systems that can be univocally linked to only one function, system expansions (including with crediting) shall be preferred over partitioning (ISO, 2020). System expansion with crediting of the avoided environmental impacts is the most widely applied allocation approach in waste management studies (Laurent et al., 2014b). However, the crediting approach remains controversial in the LCA community (Schaubroeck et al., 2022) because some Authors maintain that this approach reflects a consequential perspective and thus should only be used in consequential LCA studies, whilst others argue that the approach can also be used in attributional studies when average (rather than marginal) data is used (Finnveden et al., 2009; Paulillo et al., 2020). The partitioning of environmental impacts, which is used in a minority of studies on waste management, can be implemented using different rationales as basis for distributing impacts between different products (e.g. based on energy, mass or economic value; (ISO, 2020)) and different life cycles (Ekvall et al., 2020). The Circular Footprint Formula developed by the Joint Research Centre (JRC) of the European Commissions as part of the Environmental Footprint method reflects a crediting approach with pre-defined, material-specific factors for distributing environmental benefits between different life cycles (JRC, 2018). The formula can be used in both consequential and attributional studies, though the consequential application is limited to only first order effects (Schrijvers et al., 2021).

LCA results are significantly affected by the allocation approach (e.g. Ekvall and Finnveden, 2000; Tereshchenko and Nord, 2015). This is particularly relevant for WtH_2 systems because H_2 applications are not yet widespread and thus the choice of the substituted products/technologies is significantly uncertain; for example, H_2 for heat production can be assumed to substitute natural gas, or grey/blue H_2; this aspect is further discussed in Section 3.6. Allocation approaches can give the wrong incentives for waste management in particular circumstances. Ekvall and colleagues show that the circular footprint formula may incentivize energy recovery over recycling, which goes against both the waste management hierarchy and circular economy principles (Ekvall et al., 2021).

Allocation is a highly debated topic in LCA that still lacks international consensus. Different allocation approaches reflect different rationales; and the choice of the allocation approach is mostly arbitrary. This is why the ISO mandates sensitivity analysis when more allocation approaches are possible. For a similar reason, the allocation approach is typically normed in environmental product declarations like the International EPD system (Environdec, n.d), or the Production Environmental Footprint (JRC, 2018).

It must be noted that whilst credits are key for comparative purposes in LCA, they may be misleading when assessing the potential of a technology to remove carbon, so-called Carbon Dioxide Removal (CDR) or Negative Emission Technologies (NET) (Terlouw et al., 2021). This is because the application of credits (e.g. for energy production) may make a system have a negative climate change impact even if the system is not achieving net carbon removal. To assess the potential of a technology to remove CO_2 (and thus to qualify as CDR), the practitioner only needs considering (i) direct and indirect carbon emissions and (ii) biogenic carbon sequestration. When the resulting sum is negative the technology qualifies as CDR. In Section 4 we show practical examples of how the carbon removal potential is calculated.

3.5. Biogenic carbon and climate impacts

Solid waste contains biogenic carbon, which is defined as carbon that originate from biomass in contrast to carbon generated from fossil fuels (Masson-Delmotte et al., 2021). Sources of biogenic carbon in waste include food, wood, paper, textiles and bio-plastics. Biogenic carbon emissions are assumed to have lower climate change impacts than their fossil counterparts because the released carbon is (at least in part) balanced by its uptake by biomass during growth and prior to harvest (Cherubini et al., 2011). The proportion of biogenic carbon in waste is thus a key factor determining the environmental performance of waste management systems, particularly when they are coupled with CCS; this is because the sequestration of biogenic carbon reduces the amount of carbon in the atmosphere, thus mitigating climate alterations. Technologies that, during their life cycle, sequester more carbon than they release are described as "negative emission technologies" (CDR) (Masson-Delmotte et al., 2018); BECCS and Direct Air Capture and Storage (DACS) are two notable examples.

There are two valid approaches to model climate change impacts of biogenic carbon emissions in LCA (Christensen et al., 2009; Muñoz & Schmidt, 2016); these are summarised in Table 4 for CO_2 flows only. The approaches differ for two aspects: the GWP factors and whether biogenic carbon sequestration is considered in the LCA calculations.

One approach (named "biogenic CO_2 neutral") discriminates between biogenic and fossil carbon emissions; the GWP factors for biogenic emissions incorporate the amount of CO_2 uptake by biomass during growth, and are thus lower than their fossil counterpart. For example, the GWP for biogenic CH_4 emissions is 27.2 kg CO_2-eq./kg CH_4 compared to 29.8 kg CO_2-eq./kg CH_4 of fossil origin (Masson-Delmotte et al., 2021). The GWP factor for biogenic CO_2 emissions is 0 kg CO_2-eq./kg CO_2 because the amount released equals the biomass carbon uptake. This approach does not require accounting for carbon uptake by biomass (because it is already incorporated in the GWP factor) but it requires explicitly accounting for any carbon that is sequestered via CCS, otherwise the mass flows are not balanced and the climate benefits of CCS are not reflected in the analysis.

Table 4
Consistent Global Warming Potentials (in kg CO_2-eq.) for biogenic CO_2 flows for two approaches (after Christensen et al., 2009; Muñoz and Schmidt, 2016).

Biogenic CO_2 flows	Biogenic CO_2 neutral	Biogenic CO_2 not neutral
Uptake	0	−1
Emission	0	1
Sequestration	−1	0

The other approach (named "biogenic CO_2 not neutral") does not discriminate between biogenic and fossil carbon emissions; biogenic emissions are assigned the same GWP as fossil ones. In this case, LCA modelling requires considering explicitly the amount of CO_2 uptake by biomass, but not the successive sequestration obtained via CCS. The climate benefit of sequestering biogenic carbon are represented by the difference between uptake and release.

Both approaches are valid because they satisfy mass balances and therefore lead to equal LCA results (Christensen et al., 2009; Muñoz & Schmidt, 2016), but they differ in how individual activities contribute to the overall climate change impacts, for example in whether carbon uptake during biomass growth or carbon sequestration after capture results in negative climate change impacts. The former approach is used by the IPCC (Masson-Delmotte et al., 2021) and recommended for use in PEF studies (JRC, 2018), whilst the latter is recommended by the ISO carbon footprint standard (ISO, 2018). It is also important to note that both approaches are based on a key assumption, that biomass re-growth fully replaces harvest in the short-term (Wiloso et al., 2016). If this condition is not met, we suggest to use more detailed approaches, including dynamic ones (Brandão et al., 2013).

3.6. Choice of comparators

One of the most widespread applications of LCA is to benchmark the environmental performance of a system to identify environmental trade-offs and quantify the potential environmental benefits - for example looking at the implementation of CCS in WtE plants. In attributional studies, the benchmark is typically chosen to reflect the "status quo" or a "Business as Usual" scenario. Accordingly, the benchmark needs to represent the most common/conventional technology that at a given time and space fulfils the function that is being investigated, for example the management of municipal solid waste or the production of hydrogen. In alternative, the benchmark may also reflect an alternative scenario that is not representative of the "status quo", for example when comparing two emerging systems or when the goal of the study specifically identifies a scenario. The choice of the comparator – particularly when it reflects the "status quo" - is dependent on the time perspective adopted, i.e. retrospective vs prospective studies. Note that the approach to choose a comparator is akin to that of the "avoided" technology/product in the system expansion with crediting approach for attributional studies. In both cases, there should be full equivalency of the function of the product/technology that is being compared/substituted. Notably, quality matters when considering recycling: the lower the quality, the less valuable is the substitution (Vadenbo et al., 2017).

When the function analysed covers the management of municipal waste, the traditional comparators used include either the mix of technologies that are or will be available in a region/country, or a specific technology that is or will be relevant for a waste type, e.g. landfill or incineration for residual/non-recyclable waste. However, note that landfills are being phased out in high-income countries where incineration (with and without energy recovery) is more likely to represent the current Business-As-Usual scenario. LCA studies focusing on WtE plants can also investigate the environmental performance associated with the function of electricity and/or thermal energy production. The comparator for electricity production is typically represented by the current or future regional or national electric grid mix; for thermal energy, the comparator is most likely represented by natural gas-fired boilers - including domestic ones when thermal energy is used for district heating, and industrial ones, when thermal energy is exported from WtE in the form of high-quality energy (i.e. steam). Prospective LCA studies should consider emerging technologies that are projected to become prevalent in the future, for example heat pumps or hydrogen.

The choice of a comparator for hydrogen production is more challenging because H_2 is an emerging product, with a still immature market, but with numerous application opportunities in sectors like heating, transport and manufacturing. The most straightforward comparator includes H_2 produced via other technologies (e.g. blue/grey/green H_2), but it should be noted that this may not represent a Business-As-Usual scenario. For studies looking at H_2 used in transportation, the comparison should include the vehicle's construction and use phases to account for differences in drivetrain and overall efficiency (unless the comparator includes H_2). Suitable comparators include Internal Combustion Engine Vehicles using fossil fuels (i.e. petrol and diesel) for retrospective and short-term prospective studies, and electric or H_2-powered vehicles for long-term prospective studies. For many European countries, including the UK, a reasonable comparator for hydrogen in the short-medium term is natural gas for heating considering that the use of blended hydrogen in pre-existing natural gas lines is already being facilitated through several projects (Isaac, 2019; Mouli-Castillo et al., 2021).

3.7. Temporal/geographical aspects

Several aspects of a waste management system are specific to the geographical and temporal context they are set in. Waste compositions tend to vary greatly due to structure of waste management systems, as well as societal and cultural setting (Laurent et al., 2014a). For example, with improvements in source separation of recyclable plastics and reduction in organic food waste, compositions and the corresponding physio-chemical properties would change (CCC, 2020). With regards to LCA methodology, contextualising the region and temporal scope of a technology also influences the choice of comparators and crediting approach. This is more easily understood when considering a decarbonising energy system and its impact on the environmental assessment of a technology.

4. Case studies comparison and discussion

In this section we use two case studies to discuss the importance of different LCA methodological and modelling aspects, covering functional unit, waste type, approach to biogenic carbon accounting, allocation and temporal aspects. The case studies represent two "hypothetical" (i.e. not existing) commercial WtE and WtH_2 plants of similar scale (feedstock thermal input), both integrated with CCS and contextualised in the UK scenario. The system boundaries for the two cases are reported in Fig. 1, while full systems description are available in Supplementary Material.

The study collated data from a range of published literature sources and directly from UK companies across the supply chain. Mass and energy balances were created comprising typical combinations for various

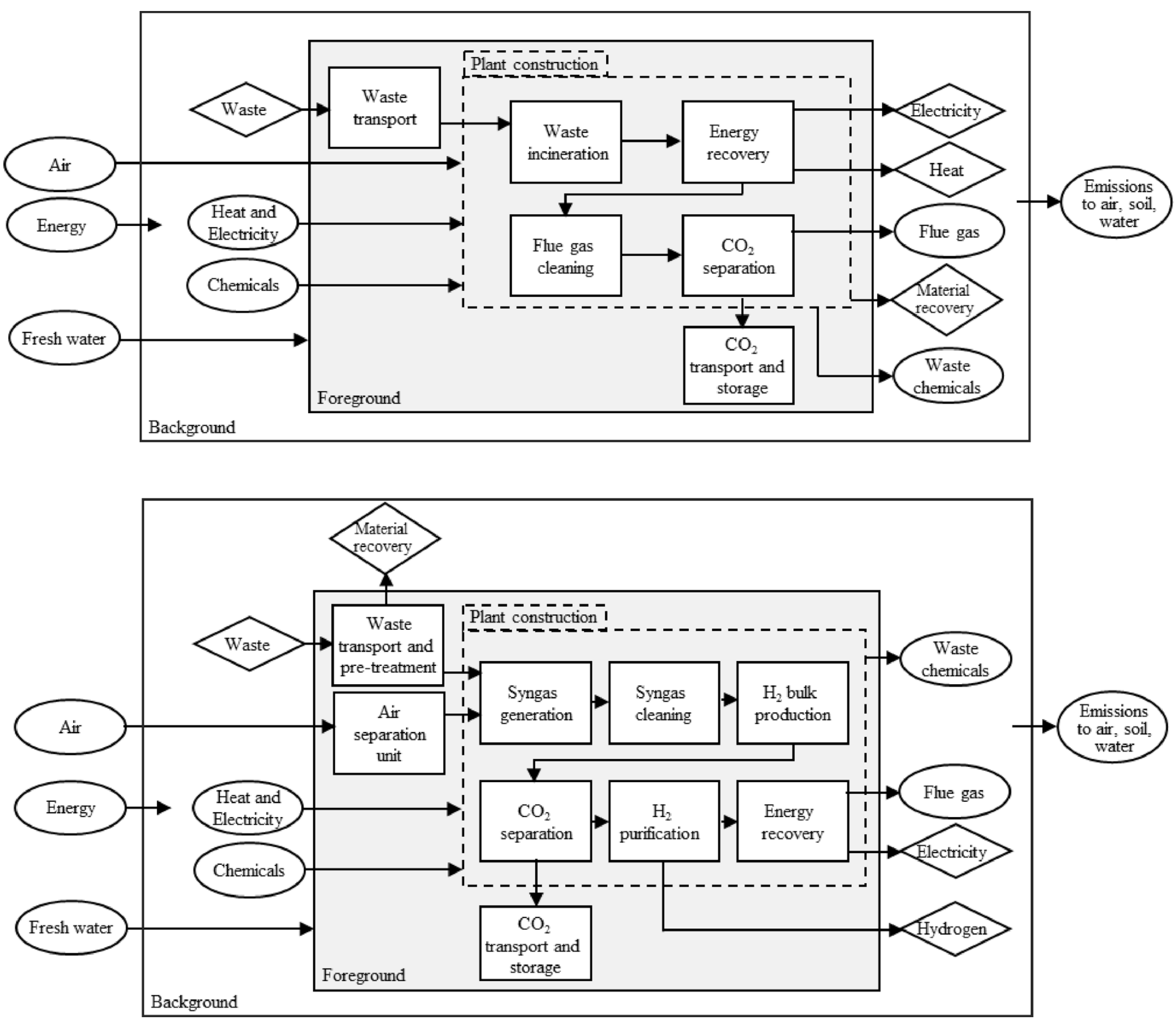

Fig. 1. System boundaries for examined WtE (top) and WtH_2 (bottom) with CCS.

production, distribution and dispensing pathways in UK. Low, central and high values were identified for a range of parameters, including plant and equipment energy consumption plus gas losses (e.g. H_2 and CO_2) across the supply chain. Central values were used to derive final emission and energy consumptions figures for the two case studies. Timelines for temporal LCA values were aligned with anticipated technology commercialisation and deployment, resulting in 2020, 2030 and 2050 being selected. Key inventory data are provided in Table 5 and further details on models can be found in the Supplementary Material.

Main assumptions of the plants models are:

- For the WtE case, both the reference and WtE with CCS scenarios are modelled in Aspen Plus and validated with data from a reference plant in UK (of similar scale), provided by the plant owner. The plant employs a moving grate incinerator fed directly with mixed MSW (HHV 9.7 MJ/kg, as received). The flue gas flow generated by the plant is described in its basic components, which are O_2, N_2, CO_2, H_2O, SO_2, NO_2, NO, and CO. Other species (PMs, VOCs, and volatile metals) are taken directly from plant data. The reference plant considers heat and power recovery, with an overall combined heat and power efficiency lower than the values representative of the overall European WtE fleet (CEWEP, 2022).
- For the WtH_2 case, the plant modelled in Aspen Plus refers to published data (Amaya-Santos et al., 2021), which were validated with data from a $5MW_{th}$ demo plant (TRL 7) in UK (Materazzi & Taylor, 2019). The plant employs a two stage gasification technology fed with RDF (HHV 15.1 MJ/kg, as received). The original MSW feedstock from which RDF is produced is modelled to be the same of the WtE plant. Hydrogen is produced at 98.6% purity level (suitable to injection in gas grid), and compression and transportation are included with production energy and GHG emissions evaluation.
- For the CO_2 capture system of both plants, we considered the design data of a commercial amine-based system (MEA 30%), modelled and optimised for operation in both thermal plants. The same carbon capture rate (CCR) of 90% is set for both systems to facilitate comparison of energy needs and chemicals consumption, which are critical from an LCA perspective. This is a very conservative assumption, as example of higher CCR are reported in the literature (IEAGHG, 2020). CO_2 compression, transportation and storage is also modelled in the same way for both technologies, with data obtained from the literature (Antonini et al., 2021) and readapted for the UK case, as described in (Chari et al., 2023).

The LCA models were developed in Gabi (now "LCA for Experts") using the databases from ecoinvent (version 3.9.1, cut-off) and Sphera for background processes. The environmental impacts were calculated using the EF3.0 method. Unless otherwise specified, we used system expansion with crediting to allocate impact to the chosen functional unit and the "biogenic CO_2 neutral" carbon accounting approach.

Table 5
Key inventory data for the modelled technologies namely WtH_2 with CCS, WtE with CCS and reference scenario, WtE without CCS.

	WtH_2 with CCS	WtE with CCS	WtE w/o CCS (Reference)
MSW in [tonne/h]	20.8	28.4	28.4
MSW in [MW_{HHV}]	56.1	76.5	76.5
H_2 produced (46 bar) [MW_{HHV}]	43.1	–	–
Net electricity [MW]	5.6_{input}	4.3_{output}	13.4_{output}
Net thermal energy [MW]	18.3_{input}	3.4_{output}	13.1_{output}
CHP efficiency [%]	–	10.2	34.7
CO_2 captured (120 bar) [tonne/h]	19.1	25.9	–
CO_2 released [tonne/h]	1.0	2.8	28.7
CO_2 recovery [%]	90%	90%	–
CC reboiler duty [MJ/kg CO_2]	2.76	3.84	–
Make-up solvent [kg/h]	49	68	–

4.1. Functional unit analysis

Advanced waste treatment plants are typically multi-functional. In Fig. 2 and Fig. 3 we report the climate change impacts of WtE and WtH_2 with CCS for two functions: waste treatment and carbon capture. The respective functional units correspond to 1 tonne of waste treated and 1 tonne of CO_2 captured. For both functional units, we provide comparison with reference scenarios represented by Waste-to-Energy (without CCS) for the functional unit based on waste treatment and Direct Air Capture and Storage (DACS) for biogenic carbon capture. We note that both technologies provide other functions beyond those investigated (including materials recovery, electricity and thermal energy generation, and hydrogen production) but that a direct comparison between WtH_2 and WtE is only possible for those functions that are in common, which include electricity generation and materials recovery. The reference scenario are modelled in Aspen Plus and validated with UK plant data for WtE and Terlouw et al. (2021) for DACS. For simplicity the LCA results for the other environmental categories are not discussed in detail below, but they are reported in the Supplementary Material.

For the function of waste treatment, Fig. 2 shows that both WtH_2 and WtE technologies when coupled with CCS perform markedly better than the reference scenario (i.e. WtE without CCS). The climate impacts (including credits) of WtH_2 and WtE equal ~−700 kg CO_2-eq. and ~−560 kg CO_2-eq. respectively, compared with ~120 kg CO_2-eq WtE (without CCS). The resulting difference between WtH_2 and WtE with CCs and the reference case thus exceeds ~650 kg CO_2-eq. per tonne of MSW. Note that the difference in direct/indirect carbon emissions between WtE with CCS and the reference scenario is driven by the capturing of carbon emissions. Climate change impact for WtE without CCS (reference scenario) varies in literature mainly due to composition of waste, implemented technology (e.g. thermal reactor, flue gas treatment), configuration (i.e. proportion of electricity and heat exported) and carbon intensity of the energy system. Bisinella et al. (2022) reported 149 kg CO_2-eq per tonne of MSW, Struthers et al. (2022) 97 kg CO_2-eq, and Pour et al. (2018) 217 kg CO_2-eq, for configurations that export both heath and electricity. For completeness, different cases for the reference scenarios, namely WtE with power export only and Incineration without any energy recovery are reported and discussed in the Supplementary Material.

The climate performance of WtH_2 and WtE is primarily driven by the sequestration of biogenic carbon, which amounts to ~−580 kg CO_2-eq. and is equal for WtE and WtH_2, assuming the same waste input and carbon capture rate. In addition, WtH_2 also benefits from significant credits from avoided burdens linked to H_2 production, which we assumed to be displacing natural gas in the gas grid (i.e. for heating). WtH_2 has higher direct and indirect emissions than WtE, but these are more than offset by the credits for H_2, thus making WtH_2 preferable to WtE from a climate perspective. The credits for electricity and materials recovery play a minor role. Struthers et al. (2022) reported −772 kg CO_2-eq. per tonne of waste treated for an average European energy and waste composition mix and with a similar biogenic carbon content. Pour et al. (2018) reported a similar result of −700 kg CO_2-eq. per tonne of MSW for the US, however based on input waste with a lower biogenic carbon content (~46%). In comparison, Bisinella reported a value of −670 kg CO_2-eq. per 1 tonne of waste for an electricity scenario comprising of 35/65 natural gas/renewables mix and a heat scenario of 27/73 biomass/electricity mix in Denmark (Bisinella et al., 2021). Adapting their model to the 2020 UK energy mix, which has a 50% fossil fuel share and thus high carbon intensity of the grid (~280 g CO_2/kWh), yields a climate change impact would of −1045 kg CO_2 eq. This highlights the importance of geographical context and also shows variation in WtE plant configuration and assumptions. Additional details on the comparison with literature data can be found in the Supplementary

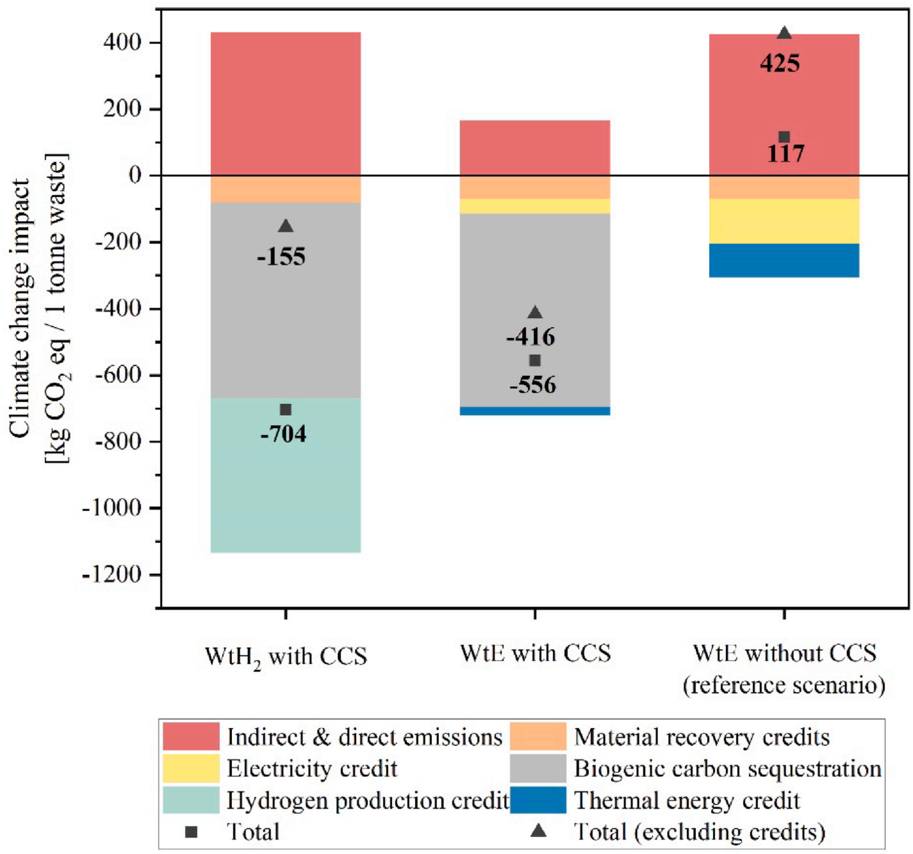

Fig. 2. Climate change impacts for WtH_2 and WtE with CCS for function of waste management and comparison with reference scenario (WtE without CCS).

Material.

The trend for the other environmental categories (reported in the Supplementary Material) is opposite to that of climate change indicating that WtE is preferable over WtH_2; this is mainly because biogenic sequestration credits do not impact other environmental categories. For most categories, differences between technologies can be attributed to the allocation of electricity credits in WtE compared to the allocation of hydrogen credits and electricity burdens in WtH_2.

Our analysis indicates that both WtE and WtH_2 with CCS qualify as Carbon Dioxide Removal (CDR) technologies because in both cases the benefits from sequestering biogenic carbon outweigh the climate impact from direct and indirect emissions. The net carbon removal potential (i.e., without credits) to −155 and −416 kg CO_2-eq. for WtH_2 and WtE respectively (note that, according to Section 3.4, we use the minus sign to indicate that carbon sequestration is larger than direct/indirect carbon emissions).

For the function of capturing biogenic carbon, Fig. 3 shows a similar pattern to Fig. 2 with both WtH_2 and WtE performing significantly better than the reference scenario which is represented by Direct Air Capture with Storage (DACS) (Terlouw et al., 2021). The climate change impact (including credits) for WtH_2 and WtE equal ∼−400 and ∼−157 kg CO_2-eq. per kg of biogenic carbon sequestered, compared to 381 kg CO_2-eq. for DACS. In this case, the discrepancy between WtE/WtH_2 and the reference exceeds 500 kg CO_2-eq. and is driven by the avoided burdens for WtE (without CCS) and H_2 generation (for WtH_2 only). Our analysis therefore suggests that from an LCA standpoint, multi-functionality makes WtE and WtH_2 significantly more advantageous than DACS for sequestering biogenic carbon; however, this advantage may diminish as the waste management and energy sectors decarbonise.

When looking at the potential for removing CO_2 from the atmosphere, the difference between DACS and WtE/WtH_2 with CCS is significantly reduced and DACS becomes a preferred option to WtH_2, and competitive with WtE. The net carbon removal potential can be estimated subtracting 1000 kg CO_2-eq. (which represents the amount of biogenic carbon sequestered) from the indirect/direct climate change impact; this yields −265 kg CO_2-eq. for WtH2, −715 kg CO_2-eq. for WtE and −620 kg CO_2-eq. for DACS.

4.2. *Waste type*

The composition of MSW, which changes both geographically and temporally, can significantly affect the performance of waste treatment technologies. In Fig. 4 we show the sensitivity of the climate performance to different waste types using WtH_2 with CCS as an example. Besides MSW, the chart includes two additional waste types: Waste Wood (WW) from construction industry and plastic-rich MSW (PR-MSW). Waste composition and ultimate and proximate analyses for the various feedstock are reported in the Supplementary Material and published elsewhere (Amaya-Santos et al., 2021; Chari et al., 2023).

The most notable difference between the waste feedstocks lies in their biogenic carbon content. WW is on one end of the spectrum with nearly 100% of carbon from biogenic origin, whilst PR-MSW is on the other end of the spectrum with ∼90% fossil carbon. The biogenic content of the modelled MSW is 64%. Other differences relate to the moisture and ash contents, which are known to affect direct and indirect emissions associated to the thermal processes. As expected, the chart indicates that the climate performance of WtH_2 with CCS is highly dependent on the biogenic carbon content: increasing the biogenic carbon content reduces climate change impact and enhances the carbon removal potential. At ∼−1350 kg CO_2-eq. per tonne of waste, the climate change impact for WW is significantly lower than MSW (−700 kg CO_2-eq), whilst that for PR-MSW is slightly higher at ∼−550 kg CO_2-eq. The biogenic carbon content affects the amount of biogenic carbon sequestered, which decreases when moving from WW to PR-MSW, and

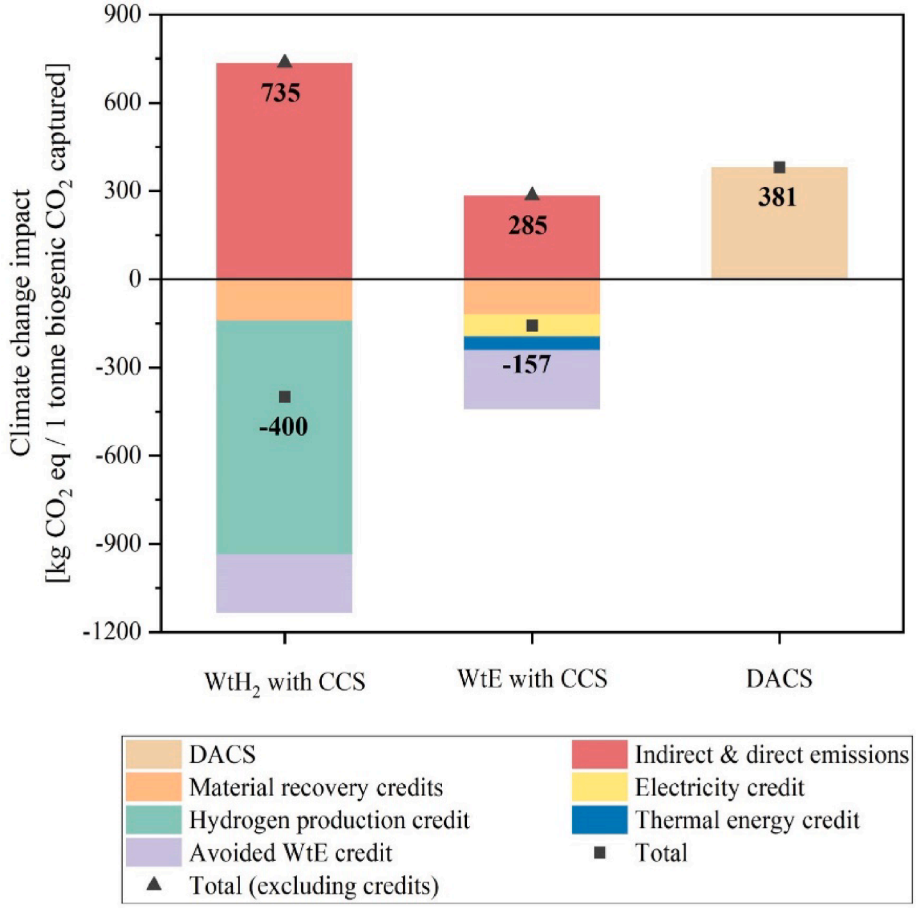

Fig. 3. Climate change impacts for WtH_2 and WtE with CCS for function of carbon capture, and comparison to DACCS.

the climate impact of indirect and indirect emissions, which increase from WW to PR-MSW. MSW is an exception to this trend due to presence of inerts and higher moisture (treated in a Materials Recovery Facility - MRF when producing RDF feedstock) that decrease the amount of carbon emissions per tonne of waste treated.

The composition of the waste feedstock also affects H_2 production. PR-MSW produces significantly more H_2 than WW and MSW due to higher hydrogen content in the waste. The resulting increase in credits, however, is more than offset by the increase in impact from direct fossil carbon emissions. As already discussed, the biogenic content of waste is key in determining the performance of WtH_2 to remove CO_2 from the atmosphere. WtH_2 with CCS fed with WW qualifies as CDR with net removal of ~−710 kg CO_2-eq. per tonne of waste. However, using PR-MSW makes the technology a net contributor to climate change (~1270 kg CO_2-eq.) because the amount of biogenic carbon sequestered is negligible compared to direct/indirect carbon emissions.

4.3. Biogenic carbon accounting approach

In Table 6 we show the climate change impacts for WtH_2 with CCS obtained using the two biogenic carbon accounting methods; these are named “Biogenic CO_2 neutral” and “Biogenic CO_2 not neutral” because the main difference is whether biogenic CO_2 emissions are or not considered climate neutral (Section 3.5). The two accounting methods produce the same overall results but differ in how life-cycle phases contribute to the climate change impact. In the “neutral” method the negative carbon impacts are driven by the sequestration of biogenic carbon. In the “not-neutral” method biogenic carbon emissions contribute to climate change impact whilst the CO_2 uptake during biomass growth provides negative climate impact. The difference between CO_2 uptake and biogenic emissions must equal the amount of biogenic CO_2 sequestered.

4.4. Allocation

In Fig. 5 we report the climate change impact for WtH_2 and WtE with CCS obtained using economic allocation. This approach entails partitioning the environmental impacts according to the revenues associated with the different products. The price values are based on gate fees at MRF and WtE plants and reported in the Supplementary Material (W.R.A.P, 2021). The resulting partitioning factors for the function of waste treatment equal 19% for WtH_2 and 52% for WtE.

The analysis shows that when using economic allocation (instead of system expansion with crediting, which is reported in Fig. 2), WtE becomes preferable over WtH_2, with the former having a climate impact of ~−217 kg CO_2-eq. and the latter of ~−29 kg CO_2-eq. There are two underlying reasons for this. First, the climate performance of WtH_2 obtained with the system expansion with crediting approach (Fig. 2) is heavily driven by the credits for H_2 production (assumed to displace thermal energy from natural gas, which is carbon intensive). The economic allocation approach does not use credits, therefore favouring WtE. Second, the partitioning factor for WtH_2 is nearly half that for WtE, therefore allocating less of the overall impact to the function of waste treatment for WtH_2. The rationale underlying economic allocation is that the reason for any economic activity is to generate profits, and therefore the products that contribute the most to the overall profits should also be attributed most of the environmental impacts (we note that revenues are typically used as a proxy for profits due to lack of data). For WtH_2, the revenues are roughly equally distributed between H_2 and gate fees; however, for WtE gate fees are dominant, thus

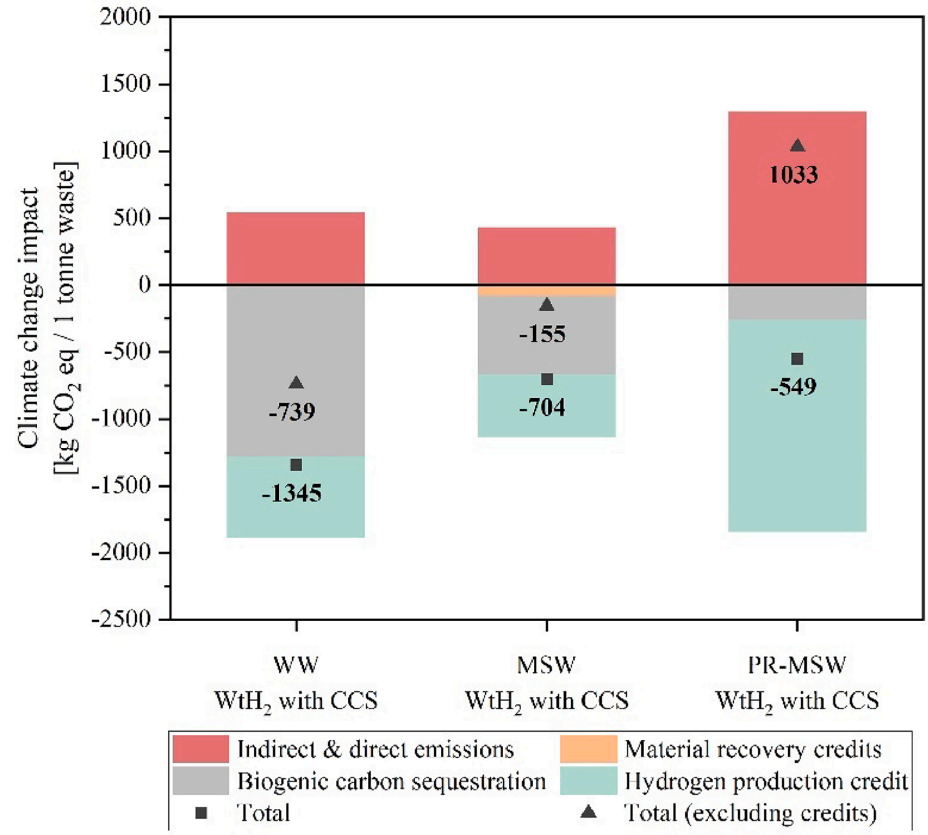

Fig. 4. Climate change impact of WtH_2 with CCS for three different waste types: Waste Wood (WW), MSW and plastic-rich MSW.

Table 6
Climate change impacts for H_2 w/ CCS for two carbon accounting approaches: "Biogenic CO_2 neutral" and "Biogenic CO_2 not neutral".

	Total	Indirect & direct emissions	Fossil CO_2, emitted	Biogenic CO_2 sequestered	Biogenic CO_2, emitted	CO_2 uptake
Biogenic CO_2 neutral	−704	−160	43	−586		
Biogenic CO_2 not neutral	−704	−160	43		59	−645

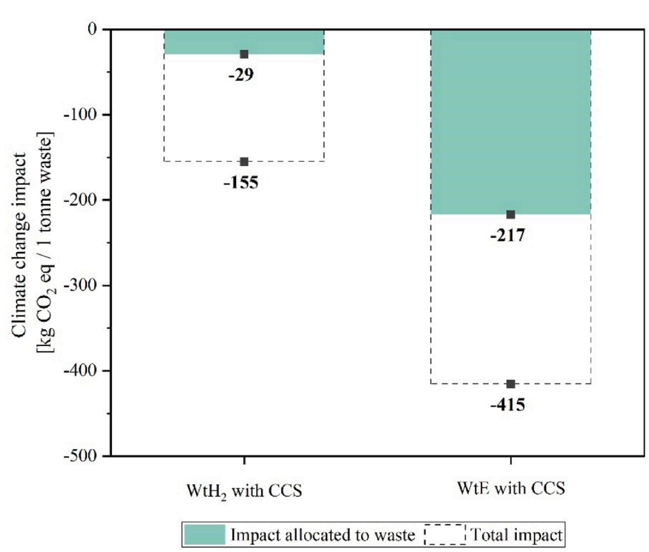

Fig. 5. Climate change impacts for WtH_2 with CCS for the function of managing waste using economic partitioning as allocation approach.

explaining why a larger proportion of impacts are allocated to the function of waste management. We note that both economic partitioning factors and credits are geographical and temporally dependent. We also note that this is a particular case where a higher allocation factor is favourable because the net impact is negative: typically, net environmental impacts are positive and therefore lower allocation factors are preferred.

4.5. Temporal aspects

In the previous sections we showed how the climate performance of WtH_2 and WtE is highly dependent (when using the relevant allocation approach) on the credits, in particular for H_2 production. Here we show how the performance changes temporally using energy scenarios for an optimistic decarbonisation pathway for the UK in 2030 and 2050 (National Grid, 2015). The technologies mix and the resulting carbon intensity for electricity grid and thermal energy production are also reported in the Supplementary Material.

Fig. 6 shows that as electricity and heat systems decarbonise, the climate change impacts (including credits) of both WtH_2 and WtE technologies increase. This is because the decrease in the credits for H_2 production (assumed to be used for thermal energy production) is larger than that for indirect emissions from consumption of electricity and thermal energy that are mainly associated with the carbon capture system. The only exception to this trend is found for WtH_2 for the 2030

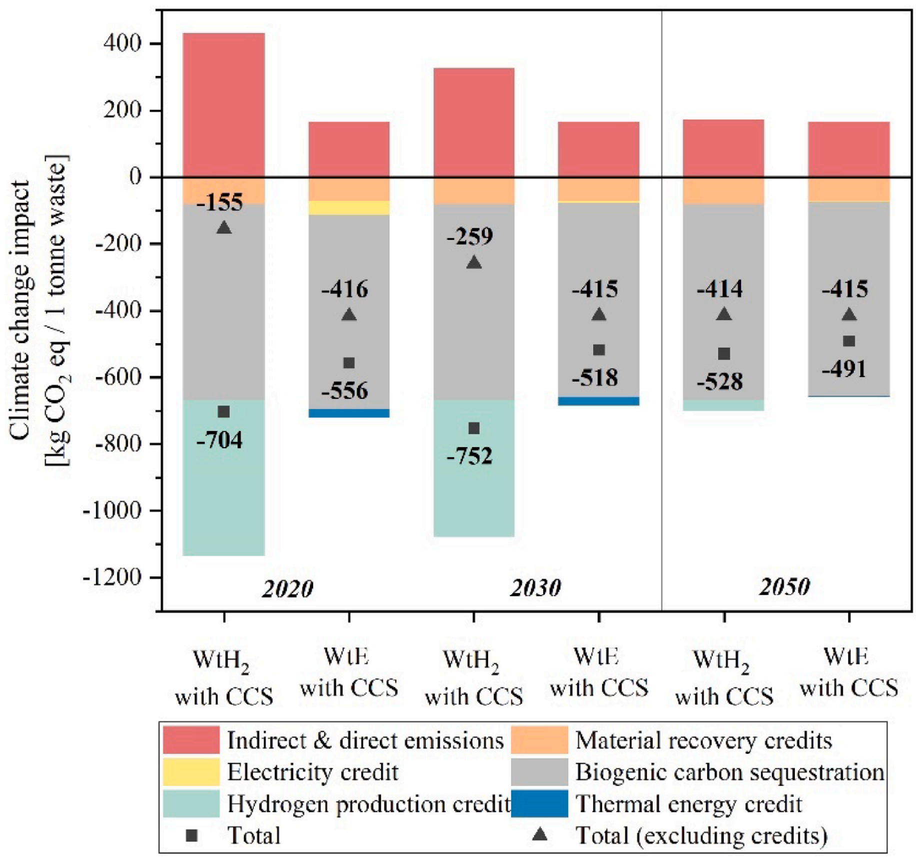

Fig. 6. Climate change impact of WtH_2 and WtE with CCS for the function of managing waste, for energy scenarios for 2030 and 2050.

scenario where the climate performance is slightly (~3%) better than that in 2020. The reason for this is that the UK energy scenarios forecast a much faster rate of decarbonisation for electricity than for heat, which in 2030 remains heavily reliant on natural gas (without CCS). This entails a larger reduction in indirect emissions from electricity consumption compared to the credits for H_2. In 2050, both heat and electricity are predicted to be largely decarbonised, thus reducing in a similar proportion the indirect emissions. All the above only applies to the case in which hydrogen is used as a substitute for natural gas for heating applications. By crediting the hydrogen instead for its use as a chemical, currently produced through water electrolysis or steam methane reforming, we can decouple hydrogen credits from decarbonising heat systems. A scenario analysis considering this crediting approach and the changing hydrogen production technology mix is detailed in the Supplementary Material.

Although the climate performance decreases due to lower credits, the net carbon sequestration potential of both technologies increases, in particular for WtH_2 for which it more than doubles from −155 to −414 kg CO_2-eq. Interestingly, by 2050 WtH_2 and WtE will sequester similar amounts of biogenic CO_2. The increase in carbon sequestration is driven by the reduction in direct/indirect carbon emissions.

Overall, the prospective energy scenarios indicate that WtH_2 is likely to be environmentally preferable until 2050 from an LCA perspective, but that the environmental benefits diminish, becoming negligible (and well within the uncertainty of our calculations) in 2050 projections. On the other hand, they also indicate that WtE is more advantageous in terms of net carbon sequestration, and that the benefits increase as the energy system is decarbonised, with similar performances achieved in 2050 by WtE and WtH_2. The scenarios are instrumental in showing how dependent are LCA results on temporal aspects, and that potential inversions in the ranking of technologies should be expected depending on future decarbonisation trends.

5. Conclusions

To date, the default presumption amongst many stakeholders has been that waste could be used in substitution of coal or natural gas as a boiler fuel or chemical feedstock, with electricity and hydrogen generation from the respective thermal treatment technology: the development of BECCS therefore being viewed as little more than the incorporation of a CO_2 capture plant in the flue gas/syngas system of such a thermochemical plant. By contrast, this work incorporated principles targeted at a more fundamental exploitation of the resource potential of waste feedstock in the emerging energy and recycling landscape.

To this end, a systematic review of most important technical and LCA methodological aspects related to thermochemical treatment of waste has been undertaken to highlight challenges and opportunities in the use of ATT technologies as a potential way to mitigate climate change. This study has strengthened the understanding of environmental performance of two major waste fuelled BECCS systems, based on Waste to H_2 (WtH_2) and Waste to Energy (WtE) technologies in the context of plants' characteristics and assumptions typical for UK, analysing various LCA methodological options. Major conclusions from this work are:

- Both WtE and WtH_2 with CCS (i.e. BECCS-enabled) have the potential to contribute to most governments' Net Zero strategies by: diverting waste from alternative, most polluting practices; through renewable energy/products production; by removing biogenic carbon from the atmosphere. We estimated that the WtE and WtH_2 with CCS are currently capable of net biogenic carbon sequestration of approximately −~420 and −~155 kg CO_2-eq. per tonne of MSW treated respectively
- Whilst post-combustion capture from the flue gas of a WtE plant is not yet a common practice, the technologies used for both power

generation and for post combustion capture are mature and each at a state of development where they could be classed as or nearly commercially proven (TRL 8–9). Hence the technology risks associated with applying BECCS to WtE are low.

By contrast the production of hydrogen via gasification of waste is not a mature technology (TRL 7). Thus, the barriers to WtH_2 with CCS are not only associated with market immaturity and near-term lack of CCS and hydrogen offtake infrastructure; despite the inherent advantages of gasification (higher conversion efficiencies and wider variety of renewable energy and chemical vectors), investors in the sector face risks connected with technological novelty as well as the common entry barriers for CCS generally.

- When considering multi-functionality of the systems we showed that although WtH_2 with CCS has higher direct and indirect emissions than WtE with CCS, these are more than offset by the credits for hydrogen production, thus making WtH_2 preferable to WtE from a climate perspective (−700 kg CO_2 -eq. per tonne of waste treated vie WtH_2 with CCS, against ∼−560 kg CO_2-eq. per tonne of waste treated in WtE with CCS). However, when using economic allocation (instead of system expansion with crediting), WtE becomes preferable over WtH_2, with the former having a climate impact of ∼−220 kg CO_2-eq. and the latter of ∼−30 kg CO_2-eq.
- The environmental performance of different BECCS technologies operated on waste is highly dependent on a number of factors; some of these factors are process-related and include thermal efficiencies, carbon capture rate, chemicals consumption, etc. Some factors are linked to geographical and temporal frameworks, which include waste composition, location of the plant and distribution and energy mix composition. Finally, LCA methodological aspects, such as those associated to functional unit, system boundaries and allocation methodology choice can significantly affect the narrative around the suitability of a technology in a specific context.

The study also identified areas where further work is required to improve GHG emissions and energy demand data fidelity, and confidence in technology performance. These include real world performance data of BECCS, especially for hydrogen production at scale, and long-term capture rates for CCS plants. In addition, it is currently assumed that the CO_2 captured and stored via CCS will remain captured indefinitely. This should be monitored as projects are deployed and real-world data becomes available.

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

Data will be made available on request.

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