Biohydrogen: a Life Cycle Assessment and comparison with alternative low-carbon production routes in UK

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Abstract

This study focuses on the production of hydrogen from municipal solid waste (MSW) for applications in transportation. A life cycle assessment (LCA) was conducted on a semi-commercial advanced gasification process for Biohydrogen (Bio-H_2) production from MSW to evaluate its environmental impact on five impact categories: Climate Change, Acidification, Eutrophication Fresh Water, Ecotoxicity Freshwater and Photochemical Ozone Formation (human health). The biogenic composition of waste and the effect of carbon sequestration were analysed for Bio-H_2, uncovering a net-negative carbon process. The counterfactual case of MSW incineration further bolsters the carbon savings associated to Bio-H_2. The production of Bio-H_2 from waste is proven to be competitive against alternative hydrogen productions routes, namely blue hydrogen (Blue-H_2) produced via steam methane reforming/autothermal reforming coupled with carbon capture and storage (CCS), and green hydrogen (Green-H_2) from solar and offshore wind, with respect to climate change. These climate change advantages are shown to carry forward in the context of decarbonisation of electricity grid mix, as analysed by scenarios taken for 2030 and 'net-zero' 2050.

1 Introduction

In recent years, low-carbon hydrogen has received increasing attention as a high efficiency energy vector that could be produced from both fossil and non-fossil sources, with low greenhouse-gas (GHG) emissions associated to production, and no emissions at the point of use. Globally, hydrogen is being promoted as an ideal energy/fuel source for heating and transport, particularly on bus, shipping and train routes that are not suitable for electrification (Staffell et al., 2019). For these reasons, hydrogen is recognised to have an important role in industrial transformation, and therefore has a major role in the UK's and European industrial strategies. The Committee on Climate Change (CCC) has recognised the important role that hydrogen plays in decarbonising the UK energy system in its recent net-zero report (Committee on Climate Change, 2020). For the UK to deliver a net-zero carbon energy system, it has

explicitly identified the requirement for 225 TWh/year of low carbon hydrogen production (Committee on Climate Change, 2020).

The development of a low-carbon hydrogen-based energy systems for industrial end use is mainly focused on three technology groups: methane reforming with Carbon Capture & Storage (CCS), water electrolysis, and biomass gasification with CCS. Amongst the number of technologies existing nowadays that produce hydrogen using different pathways and feedstock, the conversion of natural gas is the most widely deployed, as it offers the potential for bulk low carbon hydrogen production, at relatively low cost (Materazzi et al., 2019). Obviously, for the carbon benefit to be real, CO_2 must not be released to the atmosphere, i.e. the process would need to be coupled with a CCS system, to produce the so called Blue Hydrogen (Blue-H_2) (Muradov and Veziroğlu, 2005). Two principal technologies are available for Blue-H_2 production: Steam Methane Reforming (SMR) and Autothermal Reforming (ATR) (Holladay et al., 2009). SMR is a mature, commercial technology mostly employed for hydrogen production (Simpson and Lutz, 2007). ATR is a relatively less commercially available process for conversion of natural gas into H_2, however it has gained popularity as a Blue-H_2 alternative considering its favourable natural gas conversion efficiencies and carbon capture capabilities compared to SMR (Faheem et al., 2021). Pre-existing hydrogen plants can be retrofitted with CCS capabilities for both technologies. However, ATR is recognised as a more appropriate technology for low-carbon applications, since all the CO_2 ends up in the product stream at a high pressure and relatively high purity, making removal and storage much easier compared to SMR (Antonini et al., 2020).

Despite the large abundance of natural gas, the production of hydrogen via water electrolysis is also garnering attention. This technology, also known as Green-H_2 uses electricity to split water and produce hydrogen and oxygen (Holladay et al., 2009). The hydrogen produced is cooled, purified, compressed, and stored, whereas the oxygen can be either stored, or vented to the atmosphere. This technology has seen limited commercial scale usage for hydrogen production until recently, due to high capital costs (ranges between 3-15 €/kg H_2 produced, depending on the size of the electrolyser) and efficiencies, strictly related to the electrolysis system (Shiva Kumar and Himabindu, 2019; Ursúa et al., 2012). Alkaline Electrolyte (AE) and Polymer Electrolyte Membrane (PEM) electrolysers are commercially available (at low-scale production) for non-stationary applications, while Solid Oxide Electrolysers (SOEs) are currently at research and development scale, with several potential benefits to each (Singla et al., 2021). In the context of environmental performance, efficiencies are an important consideration. Commercial AE and PEM electrolysers manufacturers cite efficiencies between 47%–82% and 48%-65%, respectively (Ursúa et al., 2012; Wang et al., 2020). Solid Oxide Electrolyser (SOE), which operate at high temperatures, are expected to show higher efficiencies than AE and PEM (Brisse et al., 2008), but find little use in transportation. Electrolysis is a clean process, involving environmentally benign reactants and products. However, for the process to be considered 'green', the electricity input required must come from renewable resources such as solar and wind (Antonini et al., 2020; Bhandari et al., 2013).

The recent quest for systems that could potentially provide negative emissions and the requirement of a reliable and constant source of renewable energy have generated new interest towards new classes of low-carbon hydrogen, such as Biohydrogen (Bio-H_2) and in general Bioenergy associated to Carbon Capture and Storage (also known as BECCS). These can make a significant contribution to meeting 2050 climate change targets, as they support decarbonisation of problematic sectors, such as heating and transportation, while at the same time providing negative emissions, which is useful to compensate for other carbon intensive

sectors (Fajardy and Mac Dowell, 2017; García-Freites et al., 2021). The distributed nature of biomass means that it is unsuitable for flexible power production, but it does make it ideal for conversion to renewable gas, such as biomethane, bio-synthetic natural gas (bio-SNG) and Bio-H_2, which deliver excellent carbon savings, as well as the ability to store renewable energy in the gas networks (Balcombe et al., 2018). Bio-H_2 can be generated by thermochemical treatment (gasification or pyrolysis) of biomass, followed by a sequence of steps for gas conditioning and carbon dioxide removal (Arregi et al., 2018). An additional purification step is needed for transport-grade hydrogen, given the extreme sensitivity of commercial fuel cells to contaminants, such as CO (poison reversible <50 ppm per stack) and sulphur (poison at concentration 0.5 - 1.0 ppm) (NETL, 2004) . The process is most attractive when it uses waste as a feedstock, resulting in lower costs and added environmental benefits, contrary to the current standard yet polluting waste disposal alternatives (e.g. landfill and incineration). Although hydrogen is a clean fuel at the point of use, its production from waste and the operation phase of the plant can have both, positive and negative contributions to the environment: whilst waste is diverted from more polluting practises, such as landfill or incineration, the energy and material consumptions associated to the thermochemical plant can carry a significant environmental burden (Tagliaferri, et al., 2016). Therefore, it is imperative to carry out an exhaustive study to be able to understand the environmental impact of the production of Bio-H_2 from cradle-to-gate (e.g., from waste collection to final hydrogen production), and compare it with other alternative low-carbon hydrogen technologies. Life Cycle Assessment (LCA) is a powerful tool used to evaluate the environmental performance of a process or a product, by assessing the complete supply chain, thereby allowing the comparison of alternative ways of providing a service (Clift et al., 2000). LCA has been previously used to assess Waste-to-Energy processes for fuel production (Dastjerdi et al., 2021)

The current work aims at providing a life cycle overview of the most significant low-carbon hydrogen production routes, with specific focus on Bio-H_2 as a recent, available technology, and with the transport sector as the final beneficiary.

The main goals of this work are:

- To study the environmental burdens of a typical Bio-H_2 process and identify the hotspots.
- To compare the environmental impact of Bio-H_2 production in two different scenarios: with and without carbon sequestration.
- To quantify the impacts associated with the counterfactuals associated to municipal solid waste (MSW) disposal and content of biogenic carbon in MSW.
- To compare the contribution to climate change of the Bio-H_2 production process with alternative hydrogen production routes with low carbon emissions (i.e., Blue-H_2 and Green-H_2), in current and future electricity grid mix scenarios (Energy trends, 2020).

2 Technological Aspects of a Bio-H_2 Plant

Thermochemical treatment of biomass feedstock, and gasification in particular, is gaining strong traction in Europe giving the numerous opportunities associated to product flexibility and low environmental impact. Recent studies have proven that Bio-H_2 offers the largest potential in terms of GHG removal (Chai et al., 2021; Inayat et al., 2020; Tian et al., 2019). However, Bio-H_2 production should ideally rely on the use of second or third generation biomass as primary feedstock to avoid land use competition with food crops and intensification

of deforestation, habitat loss and loss of soil fertility (Mohr and Raman, 2015). The overall process examined in this work focuses on MSW as a source of biomass because of its large availability and low cost. Although it poses technical challenges for its use as chemical feedstock, its suitability has already been proven at pilot and demonstration scale (Materazzi et al., 2013, 2019). Furthermore, from a climate change perspective, the use of waste as feedstock not only ensures large and economical availability for consistent hydrogen supply, but also avoids use of current disposal technologies, which are known to contribute enormously to GHG emissions and water and land pollution (Sharma et al., 2020).

Compared to pure biomass, MSW introduces a greater concentration and diversity of contaminants, due to the high number and variability of sourcing points. This presents a major challenge, compounded by the fact that more sophisticated applications (including catalytic processes for Bio-H_2 production and fuel cells for transportations) have very low tolerances. A few pilot and demonstration scale examples are present in UK, and this work is specifically based on the plant scheme and mass & energy balance of one of these (see Supplementary Material) (Materazzi et al., 2019).

Bio-H_2 plant performances and environmental attributes are obviously strictly dependent on feedstock composition. Generally, the design point for the waste composition for a thermochemical facility is derived from several datasets for representative residual municipal, commercial and trade waste collected nationally as well as locally. This typically shows a substantial quantity of organic (biomass) content in the waste material, which is typically between 40% and 60% in weight (as received basis) (Larsen et al., 2013). The waste composition used in this study is indicated in Table 1. This is generated from averaging a number of datasets collected in UK (Tagliaferri and Lettieri, 2019), and used in this study to run the Bio-H_2 models and determine the environmental attributes.

Waste cannot be thermochemically treated in its original form when collected. The untreated municipal or commercial waste is first mechanically processed in a material recycling facility (MRF). This is done to homogenise the material and remove part of the moisture, recyclables (e.g., metals and dense plastics) and reject materials (e.g, oversize and inert). The material is then shredded using tearing motion to achieve a rough shred of waste residues, with a homogenous, predetermined particle size between 1-50 mm, depending on the gasification reactor requirements. The final feedstock is in the form of floc of refuse derived fuel (RDF), which is then further dried on-site using waste heat from the process. Typically, a 100,000 tonnes MSW feed produces an output of ca. 60,000 to 80,000 tonnes of RDF with a moisture content of 10–17 %, 10–20 % ash content and 15–25 MJ/kg calorific value (CV), as shown in Table 1 (refer to Supplementary Material for more details on RDF composition).

Table 1. Municipal Solid Waste composition analysis. Waste fractions expressed in weight percentage. (Tagliaferri and Lettieri, 2019)

Waste fractions [wt% as received]	MSW
Paper and cardboard	22.7
Wood	3.7
Metals	4.3
Glass	6.6
Textile	2.8
Waste Electrical and Electronic Equipment	2.2

Plastics	10
Inert/aggregates/solid	5.3
Organic fines	35.5
Miscellaneous	7.1
Carbon content[wt% (Dry-Ash free)]	**RDF**
Fossil Carbon	36.1
Biogenic Carbon	63.9
Energy content [MJ/kg Dry-Ash free]	**RDF**
Gross calorific value (GCV)	28.99
Net calorific value (NCV)	27.02

Fig. 1 schematically describes a typical Bio-H_2 process when using MSW as feedstock. The reported figures are normalised for the functional unit (FU) of this study, i.e., the production 1 MW_{HHV} (higher heating value basis) of transport-grade Bio-H_2 (>99.97% purity). The indicative values are scaled linearly from a reference plant treating around 100,000 tonnes per annum (tpa) of MSW, this being supplied from a reasonably sized UK town, accounting for residual domestic, commercial, and industrial waste arising. This is also similar in scale to small conventional energy from waste facilities (Defra, 2014). Bus fleets have been identified as the earliest likely adopters of hydrogen for transport. A typical bus will consume around 5 tpa (tonnes per annum) of hydrogen. A large depot will operate around 100 buses, i.e. 500 tpa or 20 GWh. This equates to around 5% of the Bio-H_2 plant scale identified.

The main stages of the Bio-H_2 process in Fig. 1 are as follows. Upon collection, MSW is dried, separated, and recyclable materials are recovered (Feedstock preparation). The obtained RDF is fed into a two-stage process for steam-oxygen gasification and tar-reforming (Syngas generation). Syngas stream is then cleaned using conventional cleaning technologies, including dry filters, and a combination of acid and alkaline scrubbers (Syngas cleaning) (Zwart, 2009). The hydrogen fraction of the clean syngas is then increased via a series of catalytic water gas shift reactors followed by an additional methanation step to reduce CO to ppm levels, suitable for fuel cell operation (H_2 bulk production) (Valdés-López et al., 2020). Carbon dioxide stream is chemically adsorbed with high CO_2 recovery (~99%) followed by liquefaction at 150 bar for storage and transportation (CO_2 separation; CO_2 liquefaction & storage) (Antonini et al., 2020). Pressure swing adsorption (PSA) is employed to achieve 99.97% purity required for H_2 to be used in low temperature fuel cells. The resultant transport-quality Bio-H_2 is further compressed to 300 bar for use in filling stations, while the tail gas is burnt in a gas engine to generate heat and electricity (Energy recovery). A detailed description of the Bio-H_2 process (Fig.1) is provided in Supplementary Material, as well as, reported in (Materazzi et al., 2019).
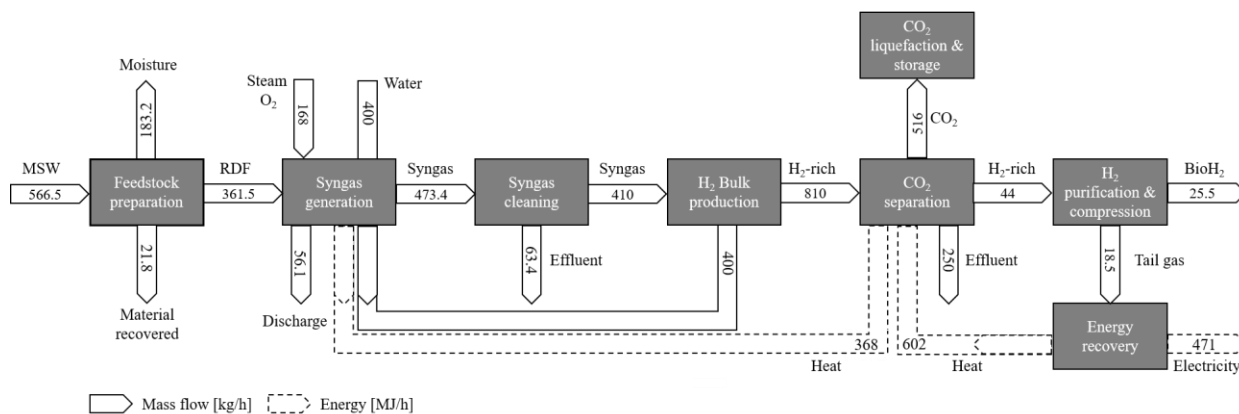
Fig. 1. Quantified mass flow diagram of the process for 1 MW_{HHV} of transport-grade Bio-H_2 production.

3 LCA Methodology

For the present study, the life cycle assessment (LCA) methodology is applied, following the guidelines of the internationally accepted standards ISO 14040 and ISO 14044, which set the fundamental principles of LCA and provide a general guideline to assess the impacts of different phases of the Bio-H_2 production process (ISO, 2006a, 2006b). According to these standards, the framework comprises four main phases, which are: the objective and scope definition, the life cycle inventory analysis (LCI), the evaluation of the impact of the life cycle (LCIA) and the interpretation of the results. The application of these steps leads to the quantification of the environmental impacts (i.e. climate change, toxicity, human health, etc.) for the system under examination.

In this work, a comprehensive LCA analysis has been performed to a hydrogen production plant using MSW as a primary feedstock, in order to study the environmental performance of the entire process, identify the operational units that have higher environmental impact (hotspot analysis), and understand the GHG removal potential when CCS is applied. For the construction of this LCA model, primary inventory data for a 50 MW Bio-H_2 plant have been collected from a UK-based waste gasification company.

Once quantified, the environmental burden of Bio-H_2 is compared to two other alternative routes of hydrogen production, i.e., Blue-H_2 and Green-H_2. To do this, a cradle-to-gate LCA model has been built for each of them, using bibliographic data. All models consider the same system boundary, from feedstock acquisition to transport-grade hydrogen production. The functional unit is the production of 1 MW_{HHV} of transport-grade H_2 (> 99.97% purity) produced at the plant according to ISO 14687 specifications (Mayer et al., 2019; ISO, 2019). The evaluation of the energetic and environmental impacts of the various hydrogen production routes has been performed using GaBi 10.0.0.71 software (Sphera, 2021). The Life Cycle Impact Assessment stage (LCIA) has been developed by using the methodology Environmental Footprint 3.0 (EF 3.0) (Zampori and Pant, 2019). The impact categories selected for the hotspot analysis are Climate Change (kg CO_2 eq.), Acidification (Mol H^+ eq.), Eutrophication freshwater (kg P eq.), Ecotoxicity Freshwater (CTUe) and Photochemical Ozone Formation - Human Health (kg NMVOC eq.). Further description of the impact categories and their units is shown in the Supplementary Material. Specific details for the construction of each LCA model are explained in the following sections. The impact categories which are considered

most significant for the purpose of the comparison of the three hydrogen production routes are Climate Change (kg CO_2 eq.) and Acidification (Mol H^+ eq.).

3.1 Bio-H$_2$ LCA model

Figure 2 shows the system boundary of the Bio-H$_2$ production process, in which the unit processes included within the LCA analysis are identified. In this study, a "cradle – to – gate" approach is adopted, which means that the system boundary ends when transport-grade hydrogen is produced (Valente et al., 2016). The boundary between the background and the foreground is also highlighted in Figure 2. The background system is composed by the processes that provide the foreground system with raw materials (e.g., minerals, crude oil, etc.) and primary energy (i.e. natural gas), whereas the foreground comprises the unit processes and flows that must be evaluated. Generally, primary data of actual processes are used to build the model of the process under study in the foreground, whereas secondary data from published databases can be used to describe the background system, when primary data is not available (Clift et al., 2000). The environment is the recipient of emissions into air, water and soil. The sourcing and collection of MSW, its transportation from the recovery facility, and burdens related to infrastructure build of the Bio-H$_2$ plant are not considered, in this work.

According to the LCA methodology applied in the present study, the production of Bio-H$_2$ from waste is considered a multifunctional process, defined as an activity that fulfils more than one function; in this case, the process produces H$_2$, recovers materials and also generates some net energy in the form of electricity (Clift et al., 2000; Hauschild et al., 2017). Following the ISO standards (ISO, 2006b, 2006a) the environmental benefits from recovered resources should be accounted for by expanding the system boundaries to include the avoided burdens of conventional production methods. Complying with the methodological approach of Clift et al. (2000), three different burden categories are considered and evaluated: direct burdens associated to the use phase of the process; indirect burdens due to upstream and downstream processes (e.g. MSW and energy provision); and avoided burdens associated to products or services supplied by the process, such as the electricity produced, and materials recovered.

The environmental burdens of Bio-H$_2$ production includes: the direct burdens allocated to all the operational units and elementary flows considered in the system boundaries; the indirect burdens allocated to the external supply of material and energy processes; and the avoided burdens allocated to the recovery of materials from waste during the RDF preparation stage (e.g. ferrous and non-ferrous metals), and the production of electricity from tail-gases.
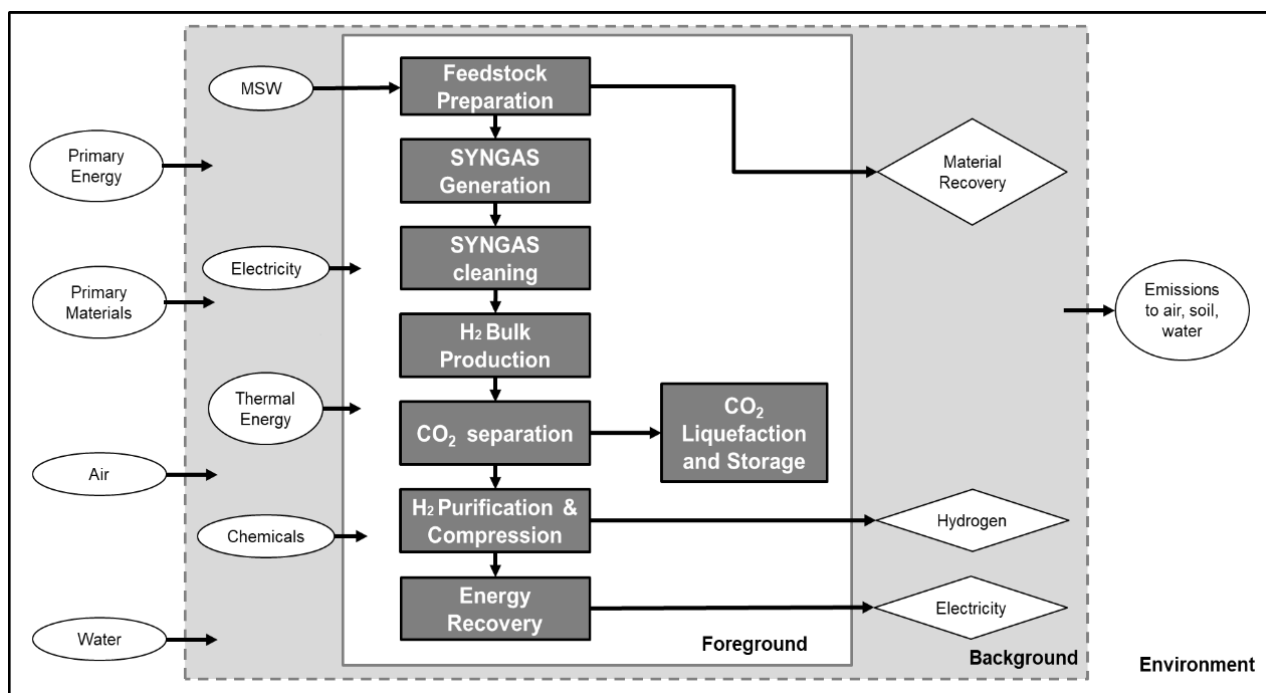
Fig. 2. System Boundary of the analysis where circles identify flows, dark squares identify the unit processes under assessment and products are identified as diamond-shaped.

A hotspot analysis is conducted on the Bio-H_2 production plant to reveal the contribution of each operational units, as depicted in Figure 2, to the following impact categories: Climate Change (kg CO_2 eq.), Acidification Terrestrial and Freshwater (Mol H^+ eq.), Eutrophication Fresh Water (kg P eq.), Ecotoxicity Freshwater (CTUe) and Photochemical Ozone Formation – human health (kg NMVOC eq.).

Two additional analyses of the environmental burdens of Bio-H_2 production process are performed to show differences between end-use of carbon. One scenario analysis in which all the CO_2 produced at the plant is liquefied and stored for permanent sequestration (CCS). Here, transportation and injection underground are not included; however, CCS systems are favourable when located close to source of emissions, or vice versa, and would take advantage of UK's existing built and natural assets (Donnison et al., 2020). Another scenario is presented, in which all the CO_2 separated from the hydrogen stream is released to the environment via direct emission or utilisation (i.e. no CCS). In both cases, impact from biogenic carbon emissions is considered as zero (or negative in case of CCS), while fossil-derived carbon emissions exhibit global warming potential (GWP) (Christensen et al., 2009; Liu, et al., 2017).

This approach ignores the issues associated with waste feedstock. In fact, if not treated in a Waste-to-Hydrogen facility, waste must be disposed of, generally by incineration or landfill, both responsible for substantial GHG emissions (Tagliaferri et al., 2016). Therefore, a third analysis is performed to evaluate the effects of diverting waste from conventional disposal methods. A conservative approach is adopted, taking the less polluting disposal method, i.e., incineration with energy and material recovery, as the counterfactual case in the current UK scenario, discounting all direct (fossil carbon) and indirect CO_2 emissions arising from MSW incineration in modern Waste-to-Energy (WtE) facilities.

The comparison analysis between Bio-H_2 and the other two alternative hydrogen production routes, Blue-H_2 and Green–H_2 has been carried out using the system expansion methodology (Hauschild, et al., 2018; Clift, et al., 2000). The multifunctionality associated to Bio-H_2 is addressed by assuming the electricity generated from tail gases to be used directly in the plant, and dismissing material recovery during feedstock preparation. To have equal functionality between the three routes, their environmental burdens are assessed and compared with a focus on transport-grade hydrogen as the only product.

An uncertainty analysis has been performed to probe the robustness of the LCA model. This was done by introducing key variations in the life cycle inventory data, including biomass content if the feedstock and changes in energy requirements according to different design choices in the Bio-H_2 process scheme. More information regarding uncertainty calculations can be found in the supplementary material.

3.2   Blue-H_2 LCA model

Two hydrogen production routes from natural gas reforming have been modelled: SMR and ATR. The SMR process model consists of the following unit processes: Natural Gas (NG) feedstock supply, SMR process, carbon dioxide liquefaction and hydrogen compression. A regionalised UK dataset from Sphera was used for the NG feedstock (see Supplementary Material). The SMR unit process, also sourced from Sphera, utilised data obtained from the Clean Urban Transport Europe (CUTE) project trialled across European cities (Binder et al., 2006). The dataset covers the steam reforming of natural gas into hydrogen including manufacture, maintenance, and end-of-life of the steam reformer. A hypothetical ATR case was also constructed using secondary data from literature and the SMR dataset as a proxy, in order to compare environmental performance at a high-level (van Cappellen et al., 2018). A minor limitation of this model is the lack of availability of primary data for ATR. Although SMR is generally used as the standard reference for comparative analysis in hydrogen energy systems (Valente et al., 2016), ATR with CCS generally shows improved techno-economic and LCA results and is thus developing further at industrial level (Salkuyeh et al., 2017). Future models will benefit from more robust and industrially validated ATR data. The commercialised reforming plants SMR and ATR processes share the same standard process, differing primarily in the reforming reactor.

3.3   Green-H_2 LCA model

The model for electrolytic hydrogen uses a well-researched liquid alkaline based electrolyser stack using a caustic solution of 30% KOH (Stolzenburg et al., 2009). The efficiency of the electrolyser is assumed at 62.5%, approximated from five electrolysers in the CUTE project supplied by Stuart Energy Systems and Norsk Hydro Electrolysers (Binder et al., 2006). Oxygen produced alongside hydrogen is released to the air, and credit for this by-product is not considered in this work. The main components of the electrolyser unit include transformer, rectifier, water purifier, lye handling system (cooling and pump), dryer, deoxidizer (Binder et al., 2006). Manufacture, maintenance and end-of-life of the electrolyser are also included (Sphera, 2021). Different renewable sources of electricity for the electrolyser and H_2 compression and its effect on environmental performance were investigated. Namely, an electricity supply derived of 100% from offshore wind, 100% from solar and a 74% offshore wind – 26% solar mix. The renewable mix is based on current share contributions of offshore wind and solar to the grid (BEIS, 2020a). Offshore wind dataset, sourced from Ecoinvent 3.6, is based on an average of offshore wind turbines (~2 MW) in the UK (Swiss Centre for Life

Cycle Inventories, 2014). Solar photovoltaics is based on a global average mix of technologies, sourced from Sphera (2021). Electrolysis was also compared to the baseline 2020 UK electricity grid mix case. Further details on these datasets can be found in the Supplementary Material.

3.4 Life cycle inventory

The inventory of the processes analysed are summarised below and further detailed in the Supplementary Material. Primary and secondary data used are regionalised, and specifically referred to the UK. Table 2 reports the key inventory data of the three hydrogen production technologies, summarising the total input and output flows per functional unit (1 MW_{HHV} of transport-grade H_2).

The key inventory data for the Bio-H_2 production process, considers the input flows for the total energy consumption, in terms of thermal energy and electricity, and the oxygen that are required for the process. The output flows quantify the internal electricity production at the plant and the material recovery from RDF preparation, namely ferrous and non-ferrous metals recovered at the feedstock preparation stage. Throughout the process, several operational units require steam. However, a large quantity of heat is recovered at various points (e.g. waste heat boiler and water gas shift reactors) to generate steam, which is in turn re-used within the process. A detailed heat and steam balance is shown in Supplementary Material, and only the external thermal energy necessary is referred in Table 2.

Table 2. Key inventory data of the three hydrogen production processes. Flow quantity is referred to functional unit (1 MW_{HHV} transport-grade H_2) and 1h as unit of time.

Key flows	Biohydrogen	Blue Hydrogen		Green Hydrogen
		SMR	ATR	
Input				
Feedstock type	MSW	Natural gas		Water
Feedstock [kg]	566.5	80.56	79.65	226.8
Oxygen [kg]	134	-	77.4	n.a.
Electricity [MJ]	800	217.13	296.91	4974
Thermal energy [MJ]	256	-	-	n.a.
Output				
Hydrogen [MJ]	3600	3600		3600
Materials recovered [kg]	21.8	-	-	-
CO_2 released [kg]	16.3	21.6	13.96	0
Sequestered CO_2 [kg]	516.7	194.4	265.5	n.a.

The Blue-H_2 LCA model for SMR was built with a 90% carbon capture rate with remaining CO_2 escaping to air. Examples from literature include reports by IEAGHG (2017) using MEA (monoethanolamine), Northern Gas Networks (2016) and Wood (2020), all citing a close to 90% carbon capture rate for SMR. The captured CO_2 is subsequently liquefied, requiring an electricity input of 45.4 MJ. The hydrogen exits the steam methane reformer at 10 bar and with a fuel-cell grade purity of 99.995%. It is then compressed to 300 bar for storage. The thermal

efficiency of the reformer is 79.8% and NG conversion efficiency to high purity H_2 of the process is 31.3% (Sphera, 2021). Similarly, a carbon capture rate of 95% was applied to the ATR model. Literature shows a tendency for higher carbon capture rates for ATR thanks to a more concentrated stream of direct CO_2 emitted directly from the PSA-tail gases (Antonini et al., 2020). Similar examples for ATR from UK's H21 and HyNet projects report a carbon capture rate of 94% and 93%, respectively (Cadent, 2018).

The inventory data for Green-H_2 considers freshwater entering the electrolyser as a primary feedstock. In this process, hydrogen produced exits the electrolyser at 10 bar with 99.995% purity and is subsequently compressed to 300 bar for storage.

4   Results and discussion

In this section, the elementary flows described in the life cycle inventory are translated into environmental impacts. The results show the main outcomes in line with the goal and scope of the study. All the environmental impact results are expressed with respect to the functional unit, 1MW$_{HHV}$ of transport-grade Bio-H_2.

4.1   Hotspot analysis for Bio-H_2 production

The hotspot analysis reveals the source of emissions at the different stages of the Bio-H_2 production chain (Figure 2) with the aim of identifying potential focus areas for process improvement (Nielsen and Wenzel, 2002). The results shown in Fig. 3, describe the contribution to five relevant impact categories (Climate Change, Acidification Terrestrial and Freshwater, Eutrophication Fresh Water, Ecotoxicity Freshwater and Photochemical Ozone Formation) of each of the operational unit described in Section 2. In the present analysis, no distinction has been made between the effects of biogenic and fossil derived carbon.
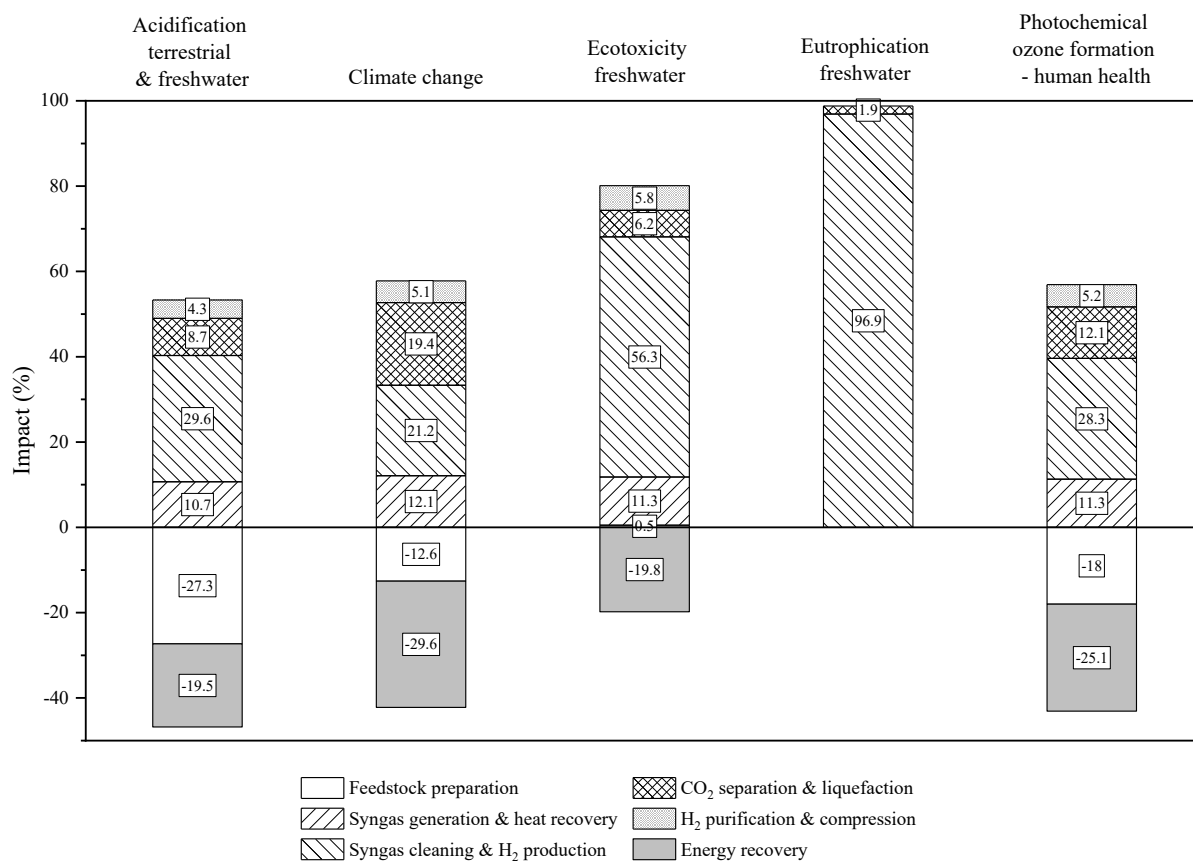
Fig. 3. Hotspot analysis of Bio-H$_2$

As observed in the hotspot analysis, energy recovery and feedstock preparation show a net negative contribution in all the environmental categories studied. The negative contributions resulting from the feedstock preparation stage (-27% to acidification, -13% to climate change and -18% to photochemical ozone formation) are mostly due to the avoided burdens associated to recovery and recycling of ferrous and non-ferrous metals. In fact, recycling enable to avoidance of those impacts associated to mining and production of virgin aluminium and steel, which generate leachate of polluting metals to soil and water. The benefits of material recovery during the feedstock preparation in MSW gasification processes was previously demonstrated by Evangelisti et al. (2015). When material recovery is not applied, the feedstock preparation stage shows a positive contribution to the environmental impact, due to energy inputs required for pretreatment of the feedstock, as also demonstrated by (Dong et al., 2018). The negative contributions associated to the energy recovery stage are due to avoided burdens for the recovery of thermal energy and steam, and electricity generation (acidification -20%, climate change -30%, ecotoxicity freshwater -20% and photochemical ozone formation -25%). Energy recovery from MSW gasification is also the largest contributor to negative emissions from WtE technologies included in the comparative study by Dong et al. (2019). Among the positive contributors to the environmental impacts, the gas cleaning section has the most significant burdens in all categories, especially in those related to water pollution (acidification 30%, climate change 21%, ecotoxicity 56%, eutrophication 97% and photochemical ozone formation 28%). This is due to the sole contribution of indirect burdens associated to the gas cleaning process, which include alkali, acid and dry filter scrubbers. The alkali scrubber is the standout

contributor in the gas cleaning units, owing to the production of sodium hypochlorite supplied to it, which involves chloride emissions, particularly to soil and water. An impact is also imparted by the activated carbon and sodium bicarbonate production used in the dry filter and scrubber, and sulphuric acid production used in the acid scrubber. The H_2 bulk production, on the other hand, relies on high energy consumption, both electrical and thermal. Although the energy and steam required by water gas shift reactors are internally sourced, the main burden is associated to the electricity consumption for syngas compression. Previous studies using a similar syngas refining process confirm what resulted from the present work (Evangelisti et al., 2015; Tagliaferri et al., 2016). Syngas generation (acidification 10%, climate change 12%, ecotoxicity freshwater 11% and photochemical ozone formation 11%) owes its impact values to the direct and indirect burdens associated to the supply of oxygen and electricity, for gasification and tar reforming, respectively. Although positive, the contributions derived from this operational unit account for a small fraction of the overall impact due to the coupling of the syngas generation unit with heat recovery, which translates into a reduced energy requirement. Similar results are shown for the CO_2 separation & liquefaction stage (9% acidification, 19% climate change, 6% ecotoxicity, 2% eutrophication and 12% photochemical ozone formation). The demand of thermal energy (as steam) required in this section is directly offset by the steam generated during other stages of the process. Thus, the steam cycle of the overall process is well balanced between steam generation and consumption. At the same time, direct burdens associated to CO_2 emissions are negligible due to the high efficiency in CO_2 recovery. When compared to similar processes, differences in the CO_2 separation technology can reveal a lower environmental performance. For example, the membrane adsorption system applied in (Susmozas et al., 2016) captures 70% of CO_2, resulting in higher CO_2 emissions in the flue gas. The sole indirect burden associated to H_2 purification and compression process units is the electricity consumption for pressurisation (4% acidification, 5% climate change, 6% ecotoxicity freshwater, 2% eutrophication and 5% photochemical ozone formation). No impact is associated to the purification of H_2 stream via the PSA (prior to compression), as the inlet stream is already at the required pressure.

Hotspot analysis is a useful tool to identify the environmental impact contributors of the process stages. In this study, the production of Bio-H_2 is shown as an efficient process, which requires a limited amount of external energy (as heat or electricity) for operation. From the analysis it is apparent that the areas that contribute the most to emissions are syngas generation and syngas cleaning, so these can be the identified as areas for future improvement of the process.

4.2 Bio-H_2 climate change contribution and counterfactual case

Different scenarios of the Bio-H_2 production process are presented with regards to climate change impact. These scenarios showcase the consequences of capturing point carbon emissions via carbon capture and storage (CCS) and of considering (thereby crediting) the biogenic carbon fraction of waste. A carbon capture rate of 99% is employed (Antonini et al., 2020). The baseline biogenic fraction of RDF used is 50%.

When accounting for the difference between biogenic and fossil carbon, biogenic carbon emissions to air are considered carbon neutral. Thus, in the scenario where CO_2 is not stored, only the impact from fossil source carbon emissions is considered (Christensen et al., 2009; Liu et al., 2017; U.S. Environmental Protection Agency, 2011). Corresponding scenarios produce a carbon negative impact when CCS is applied to the system, as carbon is effectively being removed from the natural carbon cycle. This translates to a total climate change impact

of the Bio-H_2 of 289 CO_2 eq./ MW_{HHV} of H_2 for scenario with no CCS, and -217 kg CO_2 eq./ MW_{HHV} of H_2 for scenario with permanent carbon sequestration. Another scenario with CCS that does not consider the biogenic fraction of waste is also presented. The climate change impact is net positive with 42 kg CO_2 eq./ MW_{HHV} of H_2 - in stark contrast to the scenarios where biogenic carbon is captured. The contribution of the biogenic carbon fraction to GWP becomes evident when results are compared to other BECCS system that uses 100% biomass, such as Susmozas et al. (2016). Their results showed a negative contribution to GWP (kg CO_2 eq.) of -14.63 per kg of H_2 produced, equivalent to -368.67 kg CO_2 eq. per MW_{HHV} of H_2. The environmental impact difference between biomass gasification and MSW gasification can be directly related to the biogenic carbon (i.e. biomass) content in the feedstock.

The uncertainty analysis has taken into account the variation of energy requirement and the efficiencies of the single technologies used, as well as a potential change in biomass content in the feedstock, ranging between of 40% and 60% of biogenic carbon (weight basis). This is one of the greatest sources of uncertainty, causing substantial variations in the carbon sequestration scenarios, and inevitably reflected in the GWP impact. In fact, when the nature of carbon in the feedstock is not considered, the contribution to impact due to technological uncertainties is very small, compared to the scenarios where biogenic carbon is considered. Waste composition, and biomass content therein, are among the most tested parameters in LCA waste management studies (Zhang et al., 2021). This is because the heterogeneity of this feedstock is a significant source of uncertainty, being subject to variability in the local and seasonal contexts (Laurent et al., 2014; Bisinella et al., 2016; Heijungs and Lenzen, 2014).
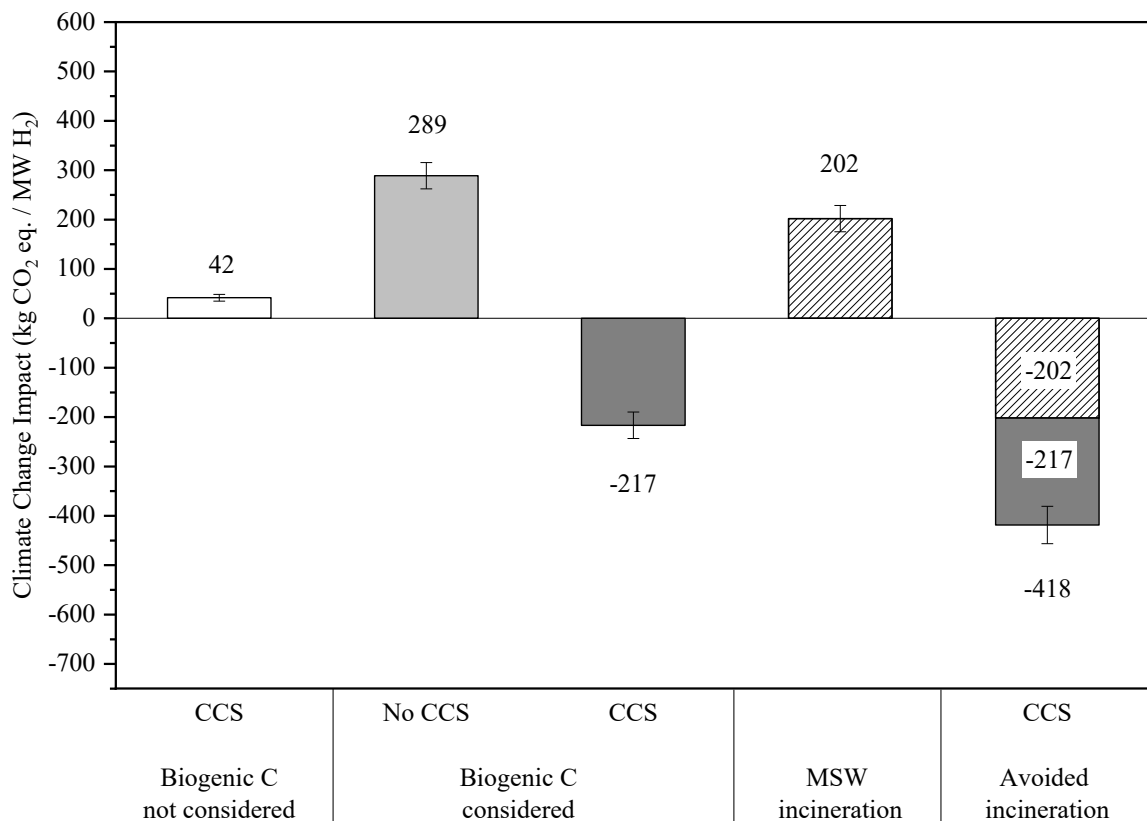
Fig. 4. Climate change impact (CO_2 eq. per FU) regarding carbon capture and storage and considering the biogenic fraction of the CO_2 stream. Uncertainties calculated based on waste composition (40-60% biogenic C variation) and technical variations in energy usage.

In Fig. 4, an additional analysis is presented, in which the avoided emissions associated to the MSW counterfactual are included. If not treated in Waste-to-Hydrogen facilities, current waste management practises call for disposal either through incineration or landfill. Incineration with energy recovery (WtE) represents the best current practice, and thus is here considered as the counterfactual (Kaza et al., 2018). Similarly, to previous cases, only emissions associated to the fossil carbon fraction of feedstock have been accounted for. To produce 1 MW_{HHV} of transport grade hydrogen, 566.5 kg of MSW are required (more details are reported in section S1 in the Supplementary Material). The same quantity was assigned as feedstock to a modern WtE (incineration with energy and materials recovery) model (Sphera, 2020). Although electricity and materials are recovered from the process, and thus credited on the final GHG output, the incineration option still shows a substantial climate change contribution of 202 kg CO_2 eq. per MW_{HHV} H_2. Therefore, by diverting waste from being incinerated, the benefit of Bio-H_2 on climate change can be further pronounced, with a negative contribution to climate change of -419 kg CO_2 eq. per MW_{HHV} of H_2 produced. Even higher benefits could be observed if considering as counterfactual other waste management practises, such as landfill or incineration with no energy recovery. There are currently several studies in literature on the production of hydrogen from first- and second- generation biomass as feedstock source (Bhatia et al., 2021; Cortés et al., 2019; Susmozas et al., 2016; Tian et al., 2019). However, the production of Bio-H_2, either from biomass or waste feedstock, would still need to be proven at a commercial scale to validate the model assumptions.

This section elucidates the environmental benefits associated to sequestration of direct CO_2 emissions from a thermochemical process converting mixed biogenic/fossil carbon feedstock, such as MSW, into transport-grade hydrogen. The choice of waste feedstock also has added benefits on consideration of the avoided burdens associated to its disposal. For all these reasons, biohydrogen can be a very useful energy vector to offset carbon emissions in the future energy scenarios.

4.3 Comparative analysis between Bio-H_2, Blue-H_2 and Green-H_2

In the present analysis, the environmental performance of the Bio-H_2 technology is compared to other two competitive low-carbon technologies, Blue- and Green-H_2, considering the three most relevant impact categories (Mayer et al., 2019): Climate Change (kg CO_2 eq.), and Acidification (mol H^+ eq.). The results are expressed per functional unit, 1 MW_{HHV} of transport grade hydrogen produced from all examined processes. The comparison of the environmental performance of the three routes has been performed taking into account the environmental burdens allocated solely to the production of hydrogen i.e. excluding system expansion methodology. Further research is directed towards an analysis that includes the additional facets of each process to expand system boundaries (for example, O_2 production in Green-H_2) and the use of harmonised LCA methodologies to synchronise comparative results from other studies (Valente et al., 2016).

4.3.1 Climate Change

To accurately compare Bio-H_2 with other technologies, credits associated to material recovery and the counterfactual effect of MSW incineration are not considered for analysis to ensure consistency in system boundaries between technologies. The emissions displayed for Bio-H_2 and Blue-H_2 are referred to processes that include CCS. The contributions to climate change are depicted in Figure 5.
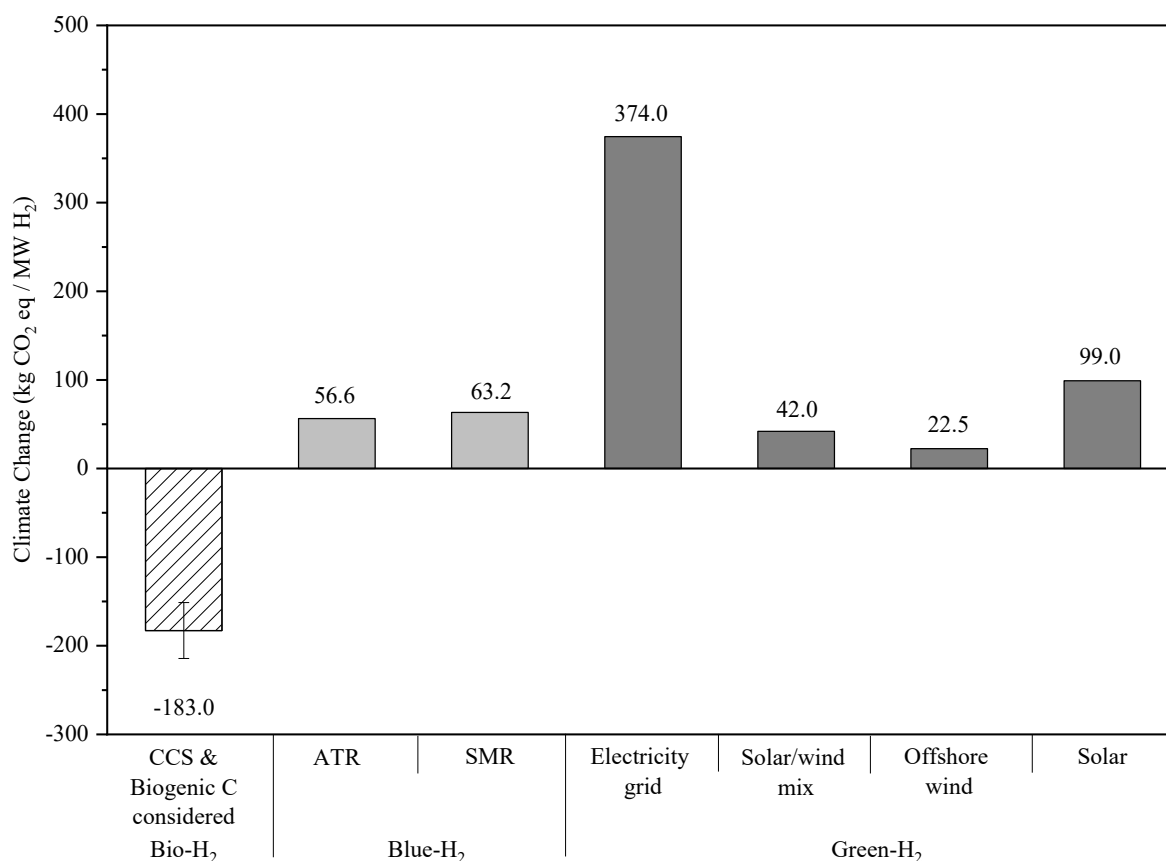
Fig.5. Climate Change contribution comparison of Bio-H_2, Blue-H_2 and Green-H_2 production technologies. Uncertainties calculated based on waste composition (40-60% biogenic C variation) and technical variations in energy usage.

Blue-H_2 produced via steam methane reforming process (SMR) with CCS process (carbon capture rate of 90%) produces 63.2 kg CO_2 eq. per MW_{HHV} transport-grade H_2. Approximately 40% of the impact derives from the embodied carbon of natural gas feedstock, rendering the process sensitive to changes in natural gas source. The upstream emissions are associated to its processing and, for imported NG, to its liquefaction and shipping (see Supplementary Material). An additional ~40% arises from the steam methane reforming process including 10% of the CO_2 process stream emitted to air. The remaining climate change impact is ascribed to the electricity required for CO_2 liquefaction and H_2 compression. The high-level analysis of ATR using literature and the SMR model as a proxy yielded a lower value of 56.6 kg CO_2 eq. for climate change, mainly as a result of differences in carbon capture rate (95% for ATR), and despite the higher electricity consumption (van Cappellen et al., 2018). These results are corroborated by literature wherein SMR with CCS produced 85 kg CO_2 eq./ MW_{HHV} H_2 (0.3 kg CO_2 eq./ Nm^3 H_2) and ATR with CCS emitted 68 kg CO_2 eq./ MW_{HHV} H_2 (0.245 kg CO_2 eq./ Nm^3 H_2) using the IPCC 2007 methodology for 'Global warming potential (GWP)' (Dufour et al., 2012, 2011). Similarly, Salkuyeh et al. (2017) recorded a GWP of 3.6 kg CO_2 eq./ kg of H_2 (equivalent to 90.7 kg CO_2 eq./ MW_{HHV} H_2) for SMR modelled with the same carbon capture rate of 90% in the Canadian context. It is important to note that environmental

impacts may vary in literature depending on geographical location, system boundary assumptions and methodology. Relevant to SMR and ATR, is the upstream emissions which vary significantly between regions like Canada and UK; for comparison, Climate Change impact for upstream emissions in Canada is 166% greater than in the UK (Sphera, 2020).

A competitive Green-H_2 route of production is limited by the high electricity demand of the electrolyser. This is evident when operating an electrolyser using the current UK electricity grid mix leading to a marked environmental underperformance, with a climate change impact of 374 kg of CO_2 eq. per MW_{HHV} H_2. This limitation can be overcome by using exclusively renewable sources as shown in Fig 5, where electricity produced using 100% solar, 100% wind or a 74:26 mix of both contribute to a total impact of 99 kg CO_2 eq. per MW_{HHV} H_2, 22.5 kg CO_2 eq. per MW_{HHV} H_2 and 42 kg CO_2 eq. per MW_{HHV} H_2, respectively. This study was limited to alkaline electrolysers; however, the electrolyser imparts negligible environmental impact (between 1-5% of total impact for sources analysed), and thus any advances in electrolyser technologies can be assumed inconsequential for this analysis. This conclusion is supported by Delpierre et al. (2021), wherein AE and PEM electrolysers were compared yielding very similar impacts. The majority of Green-H_2 emissions are associated instead to the energy intensive manufacture of mono- and multi- crystalline silicon solar cells for electricity production from solar, and to the materials required for electricity production from offshore wind turbines. It is also important to note that while no emissions are attributed to the feedstock, i.e., water, the copious amounts of water used may have an impact on other impact categories, particularly water scarcity (Mehmeti et al., 2018). Utgikar and Thiesen (2006) and Koroneos et al. (2004) reported a GWP of 140 kg CO_2 eq./ MW_{HHV} H_2 for solar electrolysis while Sadeghi et al. (2020) reported 78 kg CO_2 eq./ MW_{HHV} H_2 (GWP). In similar studies on wind powered electrolysis, Simons and Bauer (2011) and Spath and Mann (2004) reported 27.5 kg CO_2 eq./ MW_{HHV} H_2 (GWP) and 24.4 kg CO_2 eq./ MW_{HHV} H_2 (GWP) respectively. However, all studies mentioned focused on combined wind technologies, while this study is focused on 100% offshore wind to reflect the UK's planned large-scale deployment of offshore wind (BEIS, 2020b).

Bio-H_2 contribution to climate change equates to -183 kg CO_2 eq. per MW_{HHV} H_2 produced every hour, when CCS and biogenic carbon content are considered. A lower negative contribution to climate change is observed compared to the -217 kg CO_2 eq. per MW_{HHV} obtained in the previous section, as the negative impact from avoided burdens associated to material recovery is not credited to the system in this analysis. As the feedstock moves to greater plastic content, i.e., fossil carbon of 60%, Bio-H_2 remains competitive with an impact of -129.5 kg CO_2 eq./ MW_{HHV} H_2 (near the upper bounds of error). Another source of uncertainty is related to the various gasification and tar reforming technologies that are associated with different energy requirements available in the data sources. This generates an uncertainty that ultimately affects the environmental impact assessment (Ahmad et al., 2016).

Noteworthy, demand for waste as a feedstock for other biofuel and chemical applications is expected to increase in the future (Hofbauer and Materazzi, 2019; Paliwal and Chanakya, 2020), alongside an increased demand for low-carbon hydrogen as a transportation fuel (Hydrogen Council, 2021). This introduces a potential risk of limited feedstock availability for biohydrogen in the future. To ensure a steady ramp-up and stability of hydrogen supply, existing SMR and ATR plants can be retrofitted with CCS and large-scale electrolysers gradually deployed as the grid moves towards renewables (French, 2020). In this context, bio-H_2 is proposed as a complementary technology to aid the near- and medium- term transition as well as a long-term complement to other low-carbon production routes. These results show that

Bio-H_2 is not only an effective solution to waste disposal, but it is also appropriate to achieve the objectives proposed by the Net-Zero 2050 for it being a viable carbon-negative technology (Committee on Climate Change, 2020).

4.3.2   Acidification

Results for acidification terrestrial and freshwater (Mole of H^+ eq.) per functional unit are shown in Fig. 6.
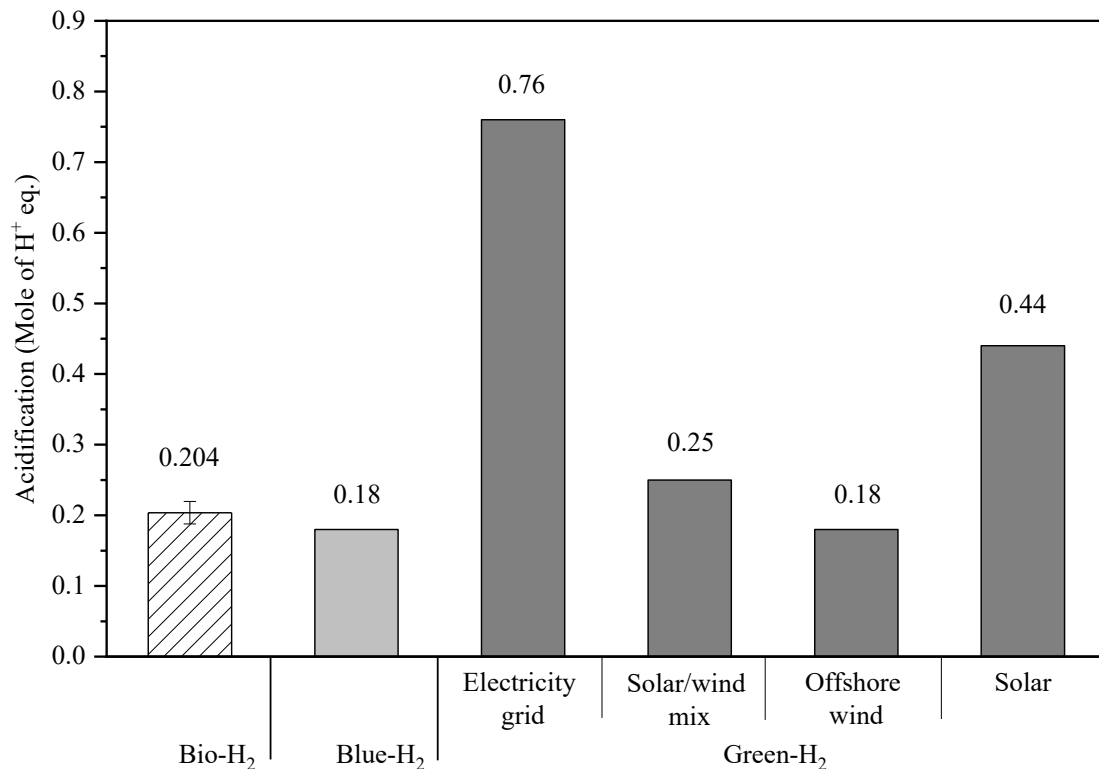
Fig. 6. Acidification contribution comparison of Bio-H_2, Blue-H_2 and Green-H_2 production technologies. Uncertainties calculated based on technical variations in energy usage.

The main source of acidification is air-borne emissions of SOx and NOx gases from combustion processes that release H^+ when they are degraded in the atmosphere or after deposition to soil, vegetation or water (Hauschild et al., 2017).

The higher contribution is presented by Green-H_2 with electricity from the 2020 UK grid as the energy source. Electrolysis using solar energy reveals an impact of 0.44 mol of H^+ eq./ MW_{HHV} H_2, relatively higher compared to offshore wind with an impact of 0.18 mol H^+ eq./ MW_{HHV} H_2. This can be explained by the higher proportion of SO_2 emission in silicon solar cell manufacture while materials used in construction of wind turbines emit a higher proportion of NO_2.(Swiss Centre for Life Cycle Inventories, 2007; Tawalbeh et al., 2021). Acidification potential is estimate on the basis of SO_2 and NO_2 emissions, among others, with SO_2 carrying

greater weight; the characterisation factor (CF) according to EF 3.0 methodology is 1.31 eq./kg H_2 for SO_2 and 0.74 eq./ kg H_2 for NO_2 (Fazio et al., 2018). This result is in line with those reviewed by Bhandari et al. (2013) where a number of studies show that solar photovoltaics present a higher contribution to acidification for the aforementioned reasons.

In parallel, Blue-H_2 via SMR presents an impact of 0.18 mol H^+ eq./ MW_{HHV} H_2 with 52% attributed to natural gas production and 30% to its conversion. Conventional SMR for hydrogen production, according to Susmozas et al. (2013), yielded a total acidification potential of 0.47 kg SO_2 eq. / MW_{HHV} H_2 (= 0.00844 kg SO_2 eq./ kg H_2) with only 18% attributed to natural gas feedstock and 77% of contribution concentrated in the reforming stage. This highlights the benefits of CCS, which are not limited to carbon dioxide footprint.

In the case of Bio-H_2, results showed a contribution to acidification of 0.21 mol H^+ eq./ MW_{HHV} H_2. Indirect burdens associated to the syngas refining and cleaning process are the major contributors to acidification, as also observed in other studies (Wulf and Kaltschmitt, 2013). The main source of uncertainty affecting the contribution to this impact category is associated to variation of efficiencies of the syngas generation and cleaning technologies. Overall, considering uncertainties, Bio-H_2 has an acidification impact comparable to the alternative routes, between 4 and 22% greater than Blue-H_2 and Green-H_2 via mixed solar/wind, and offshore wind.

Results show that indirect burdens dominate the contribution to acidification potential in the three hydrogen production routes compared. For Bio-H_2, this provides scope for finding alternative chemicals or alternative production methods of those chemicals that are supplied to the plant, so as to reduce acidification potential.

4.4 Scenario analysis of Bio-H_2, Blue-H_2 and Green-H_2: Climate Change

The evolving pertinence of these technologies within the energy transition landscape is an important consideration as LCA results have been reported to be strongly affected by the energy supply, particularly electricity (Kløverpris et al., 2008; Moora and Lahtvee, 2009). Hence, a scenario analysis was conducted to compare the environmental burden due to climate change of each hydrogen production route according to the electricity mix predicted for the UK in 2030 and 2050 (European Commision, 2016). The 2050 scenario analysed is the UK's target to achieve a carbon neutral, 'net-zero', economy by 2050, in accordance with the CCC's recommendation of keeping global warming well below 2ºC since the pre-industrial period (BEIS, 2020a; Committee on Climate Change, 2020). The environmental burdens of technology mix for these different energy scenarios, according to the data reported by BEIS (2020), have been modelled using GaBi database (Sphera, 2021). The net-zero scenario describes an electricity supply where hard coal, coal gases and heavy fuel oil are considered obsolete. Dependence on natural gas (9.7%) is still forecasted, however only if coupled with CCS (assumed here at 90% efficiency). Electricity from wind and photovoltaic are the largest contributors at 68% of the electricity grid mix (BEIS, 2020a).
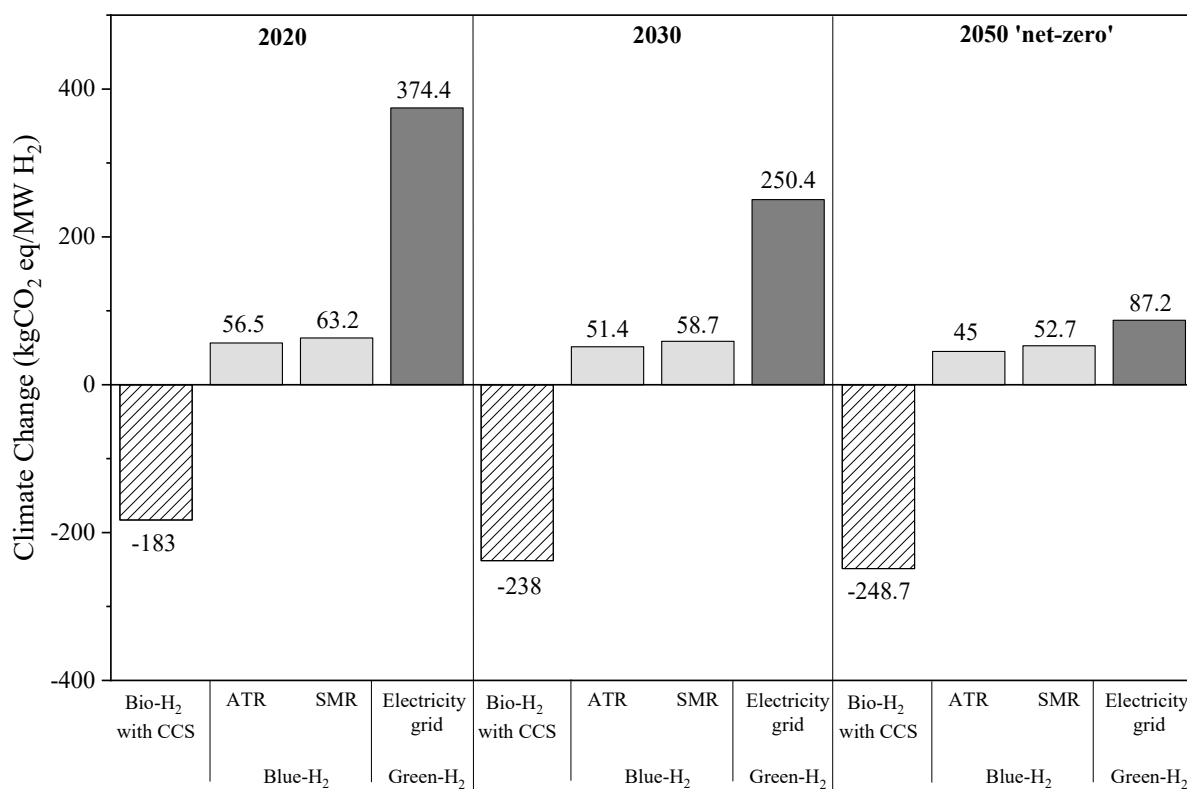
Fig. 7. Contribution to climate change for future electricity grid mix scenarios

As expected, the electricity supply strongly affects the contribution of each production method to climate change. Thus, any processes with a high electricity input will benefit from decarbonisation of the grid. Fig. 7 shows the contribution of climate change for the different hydrogen production routes per 1 MW_{HHV} of H_2 produced, as a function of the electricity demanded for each process.

Bio-H_2 production impact contribution decreases by 24% in 2030 to -284.16 kg CO_2 eq. / MW_{HHV} H_2 and by 56% in 2050 to -294.78 kg CO_2 eq. / MW_{HHV} H_2, relative to the present-day case. A limitation of this analysis is that it does not consider the changing composition of feedstock (MSW) over the next 30 years. This may show a pronounced sensitivity to the future performance of Bio-H_2. With increased recycling rates and improved waste management, fossil carbon content of MSW is expected to decrease, thereby further reducing the technology's environmental impact (Defra, 2018). Another consideration is the effect of technological improvements of Waste-to-Hydrogen technology on environmental impact, as it progresses towards commercialization. Further research into the future environmental effects of increasing scale and maturity of technology would be of interest.

In comparison, Blue-H_2 production via SMR has a smaller differentiation in climate change as the grid moves to decarbonisation; 7% decrease to 58.7 kg CO_2 eq./ MW_{HHV} H_2 by 2030 and 17% decrease to 52.7 kg CO_2 eq./ MW_{HHV} H_2 by 2050. This is due to its lower overall electricity consumption. Additionally, a large part of its contribution is dominated by upstream feedstock emissions and CO_2 process stream emissions and thus comparatively unaffected by changing electricity mix. In view of the similarity in models between the two Blue-H_2 technologies, the hypothetical ATR behaves closely to SMR in projections; 9% decrease by 2030 and 20% decrease by 2050 compared to the current scenario. The greater differentiation

in ATR is expected, owing to an electricity consumption that is 3 times that of SMR (van Cappellen et al., 2018).

The greatest decrease is presented by H_2 production via electrolysis with electricity supplied from the grid (not a Green-H_2 process) with a 33% reduction from 374.78 kg CO_2 eq./ MW_{HHV} H_2/ MW_{HHV} H_2 by 2030 and a 77% reduction to 87.17 kg CO_2 eq./ MW_{HHV} H_2 by 2050. This sensitivity to changes in grid carbon intensity is reflective of the large electric power necessary for operation and thus constitutes the main burden. The forecasted future efficiencies, as reported by Schmidt et al. (2017), in consultation with industry and academic experts, is expected to reach a low of ~53.6 kWh/kg of H_2 by 2030 (modelled efficiency is 54.0 kWh/kg of H_2). Thus, for standard alkaline water electrolysis, impacts in the future are unlikely to change drastically on account of improved technological improvements. A possible limitation of this analysis is that it does not consider the effect of improving electrolyser scale on environmental impacts. Delpierre et al. (2021) analysed this effect, with a large-scale wind-powered electrolyser in a 2050 scenario imparting a GWP impact of 0.739 kg CO_2 eq./ kg H_2 equating to 18.6 kg CO_2 eq./ MW_{HHV} H_2. These ex-ante results for large-scale electrolysers are, however, comparable to the results obtained for Green-H_2 via offshore wind, 22.5 kg CO_2 eq./ MW_{HHV} H_2 (Section 4.3.1). The true Green-H_2 route using solar and/or offshore wind are independent of changes in grid supply and are not projected onto future scenarios.

The scenario analysis demonstrates how environmental performances are foreseeably affected by sustainability policy measures to reduce GHG emissions. As a result of future reductions in grid carbon intensity, the environmental impact of Bio-H_2 is considerably diminished compared to Blue-H_2 technologies. This greater sensitivity to electricity grid changes further supports the claim that Bio-H_2 can be a suitable technology to supplement hydrogen production for transport in the long-term.

5   Conclusion

An attributional LCA methodology has been applied to evaluate the environmental performance of the production of low-carbon hydrogen for transport applications via a gasification process using MSW as feedstock, and associated CCS.

A hotspot analysis of the process and comparison to counterfactual case revealed the following:

- Material recovery during RDF preparation and net electricity production provided benefits associated to climate change and photochemical ozone formation. Burdens were attributed to the reforming processes, gas cleaning, liquefaction (CO_2) and compression (H_2).
- Contributions to acidification, eutrophication and ecotoxicity are dominated by the gas cleaning process.
- Consideration of avoided burdens from MSW incineration further bolsters climate change benefits by an additional ~90% reduction in climate change.

The Bio-H_2 process was compared to alternative low-carbon hydrogen production routes namely, Blue-H_2 (SMR and ATR) and Green-H_2 (offshore wind and/or solar photovoltaics) with current and future electricity grid supply of the UK. The comparative conclusions on the basis of contributions to climate change and acidification were as follows:

- Bio-H_2, with a climate change impact of -183 kg CO_2 eq. per MW_{HHV} H_2, outcompetes Blue- and Green-H_2 routes on the basis of processes efficiency, feedstock choice and sequestration of CO_2 from mixed waste feedstock and fares similar to other routes on the basis of acidification.
- Bio-H_2 imparts environmental burdens due to significant electricity requirements and is thus advantaged by future grid decarbonisation.
- The study identified as main sources of uncertainty the biomass content in the feedstock, followed by energy requirements attributed to different types of gasifiers and tar reformers available in the data sources. This uncertainty can be reduced by using primary data (for example from industrial operating plants) or more updated secondary data from literature.

Further research may look at expanding the LCA system boundaries during comparative analysis to include further functionalities. A life cycle costing assessment is also relevant to assess these technologies against economics (e.g. capital costs, operating costs and levelised cost of hydrogen) and ease of implementation. Future work should also look at employing harmonised LCA methodologies to allow more consistent comparisons with previous research in the hydrogen energy systems space.

This work showed that Bio-H_2 can be a competitive technology to aid the near- and medium-term transition to hydrogen economy, as well as a long-term complement to other low carbon hydrogen alternatives. Not only it is an effective solution to waste disposal, but it is also appropriate to achieve the objectives proposed by the Net-Zero 2050 for it being a viable carbon-negative technology.

Acknowledgments

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