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# **The Paris compatibility of green hydrogen and carbon capture and utilisation technologies**

Kiane de Kleijne

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# The Paris compatibility of green hydrogen and carbon capture and utilisation technologies

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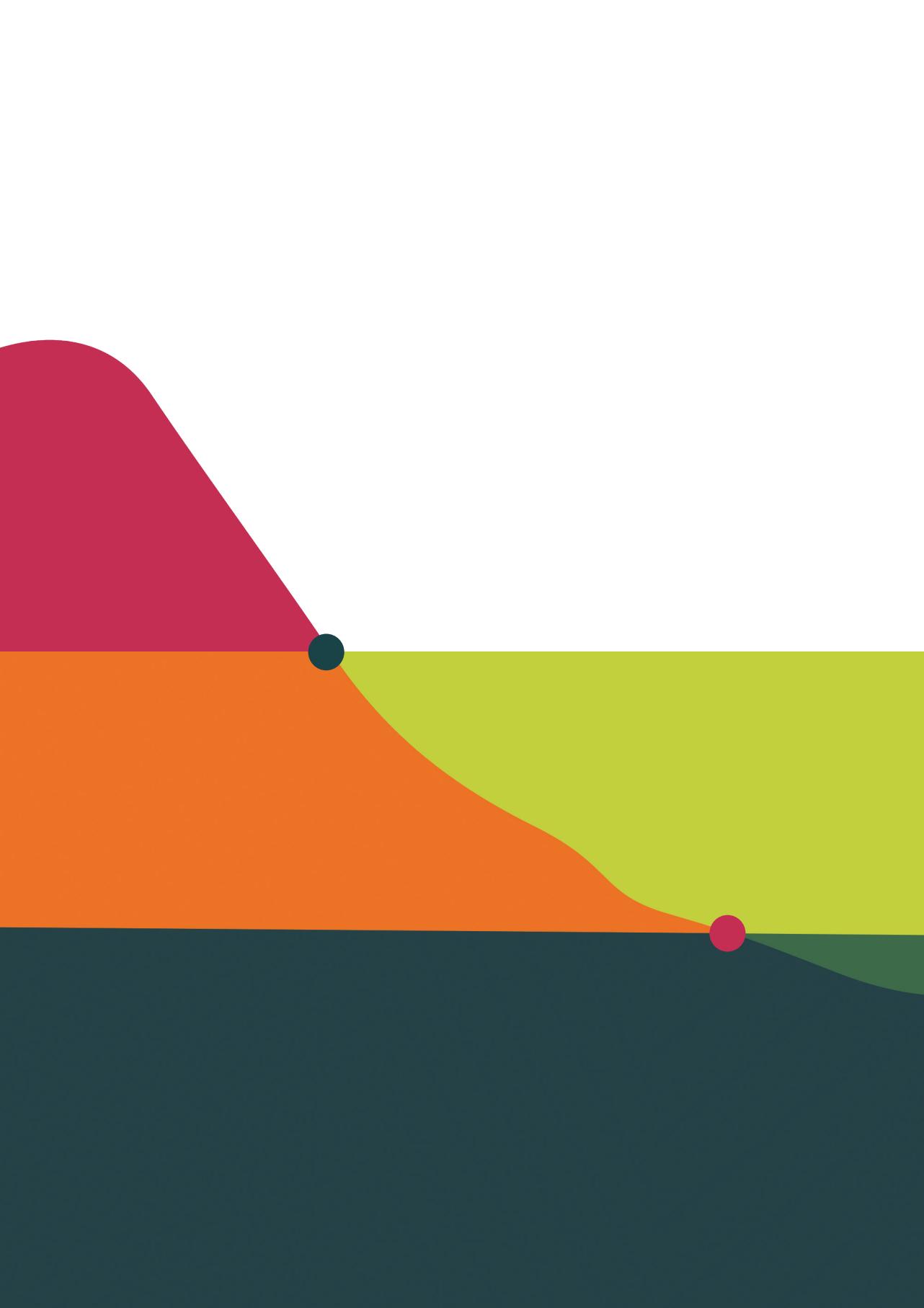
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# 1

## Introduction

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## 1.1 Climate change and the Paris Agreement

Human-induced global warming has reached 1.14°C over the decade 2013–2022 compared to pre-industrial levels (Forster et al. 2023), and is indisputably caused by the emission of greenhouse gases (GHG) from human activities (Gulev et al. 2021). As a result, weather and climate extreme events have increased in frequency and intensity in all regions of the world, and will continue to become more frequent and severe with further warming (Seneviratne et al. 2021). Since every tonne of carbon dioxide (CO<sub>2</sub>) emitted adds to global warming, net anthropogenic CO<sub>2</sub> emissions must be reduced to zero to halt global warming (Canadell et al. 2021).

### 1.1.1 The Paris Agreement

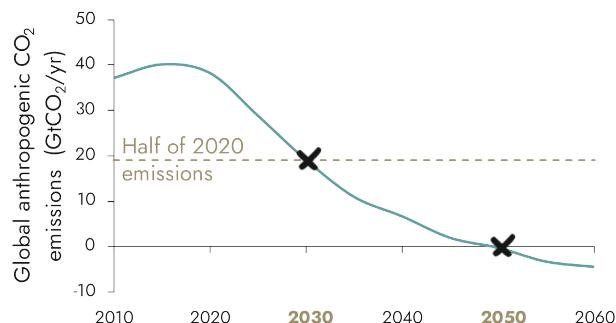
The global effort to address climate change is guided by the 2015 United Nations Framework Convention on Climate Change's (UNFCCC) Paris Agreement. Almost all of the world's nations agreed to collectively limit global warming to well below 2°C and to pursue efforts to limit global warming to 1.5°C above pre-industrial levels (Article 2.1) (UNFCCC 2016). This is known as the Paris Agreement's long-term temperature goal. Associated with a temperature goal is its remaining carbon budget, that is, the amount of CO<sub>2</sub> that can still be emitted to keep global temperature increase below the specified limit, with a given probability. The remaining carbon budget for limiting warming to 1.5°C with a 50% probability has decreased to 250 billion tonnes of CO<sub>2</sub> at the beginning of 2023 (Forster et al. 2023) – less than six years' worth of current emissions.

The need for emissions reductions and for reaching net zero GHG emissions is specified in Article 4.1 of the Paris Agreement: "*in order to achieve the long-term temperature goal set out in Article 2, Parties aim to reach global peaking of greenhouse gas emissions as soon as possible [...] and to undertake rapid reductions thereafter in accordance with best available science, so as to achieve a balance between anthropogenic emissions by sources and removals by sinks of greenhouse gases in the second half of this century, on the basis of equity, and in the context of sustainable development and efforts to eradicate poverty*" (UNFCCC 2016). Every nation is expected to communicate its intended emission reductions by 2030

through nationally determined contributions (NDCs), and encouraged to outline its strategy to achieve net zero GHG emissions “*by or around mid-century*” (UNFCCC 2022) in long-term low greenhouse gas emission development strategies (LT-LEDS). As of June 2023, 149 countries have brought out net-zero climate targets (Net Zero Tracker 2023), although for 90% of these targets the adoption of policies that would achieve them is lagging behind (Rogelj et al. 2023).

### 1.1.2 Paris-compatible emission reduction pathways

The interpretation of the Paris Agreement and the translation into what would be compatible emission reduction pathways has been the topic of scientific debate (Mace 2016; Schleussner et al. 2016), which has converged on the understanding that only the pathways with ‘no or low overshoot’ of 1.5°C as categorised in the IPCC Special Report on Global Warming of 1.5°C (Rogelj et al. 2018) are compatible with the Paris Agreement’s long-term temperature goal (Wachsmuth et al. 2018). Based on the IPCC 1.5°C Scenario Explorer and Data (Huppmann et al. 2018), these pathways are characterised by two key numbers: in 2030, global net anthropogenic CO<sub>2</sub> emissions are halved compared to 2020 emissions, and in 2050, net CO<sub>2</sub> emissions are zero (Figure 1.1). These two numbers have since been confirmed in an updated classification of Paris Agreement-compatible emission reduction pathways that also takes non-temperature criteria into account (Schleussner et al. 2022). In this thesis, a technology is classified as Paris-compatible when its deployment is in line with reaching the Paris Agreement’s long-term temperature goal.



**Figure 1.1. Paris-compatible emission reduction pathway as the mean of no and low-overshoot 1.5°C scenarios**, in which global net anthropogenic CO<sub>2</sub> emissions are halved in 2030 compared to 2020 emissions, and are zero in 2050 (Huppmann et al. 2018).

## 1.2 Net zero emissions in the industrial sector

The net zero CO<sub>2</sub> target implies that no sector can be exempted from deep emission reductions, including the industrial sector. While industrial GHG emissions have increased more since the year 2000 than in any other sector, achieving net zero CO<sub>2</sub> emissions by mid-century is still considered possible by a combination of: i) lowering emissions of primary production via increased energy efficiency, a switch to low or zero-GHG energy and feedstocks, and carbon capture and storage; and ii) reducing the amount of primary production via reduced demand and increased circularity, recycling and material efficiency (Bashmakov et al. 2022). Amongst the options for low to zero GHG energy carriers and feedstocks are carbon capture and utilisation (CCU) technologies and green hydrogen. The next paragraphs detail what these technologies entail.

### 1.2.1 Carbon capture and utilisation

Carbon capture and utilisation is a process in which CO<sub>2</sub> is technologically captured and subsequently used in or as a product. The CO<sub>2</sub> can be captured from the flue gas from the combustion of fossil fuels or biomass, industrial processes such as the calcination reaction in cement production, biomass fermentation, or directly from air. Depending on the composition of the stream from which the CO<sub>2</sub> is separated, different capture technologies are used, including solvents, sorbents, membranes, cryogenics and industrially cultivated organisms (Bui et al. 2018). After separation, CO<sub>2</sub> can be used directly or converted into a product through a variety of processes in different stages of technological development (Chauvy et al. 2019). CO<sub>2</sub> can be used directly in greenhouses, the food and drink industry or enhanced hydrocarbon recovery, or in the production of fuels and chemicals, mineral carbonates and construction materials (IEA 2019a). The CO<sub>2</sub>-based product can replace a fossil product, and depending on the product, the CO<sub>2</sub> is stored permanently or released after a certain period of time ranging from days to centuries, called its lifetime. For example, for fuels, the utilised CO<sub>2</sub> is emitted upon combustion, while in some construction materials CO<sub>2</sub> can be stored permanently.

### 1.2.2 Green hydrogen

Currently, most hydrogen is produced through steam methane reforming of natural gas, which is termed 'grey hydrogen'. Hydrogen can also be produced in an electrolyser, which uses electricity to split water into hydrogen and oxygen. When renewable electricity is used, this is called 'green hydrogen'. Different types of electrolyzers are available or under development. The most mature type of electrolyser is the alkaline electrolyser (ALK), followed by the proton exchange membrane or polymer electrolyte membrane (PEM) electrolyser, and the solid oxide electrolyser (SOEC). The electrolyser types differ in terms of efficiency, operating temperature and pressure, how quickly they can be turned off and on, material requirements and costs (Zhang et al. 2022; IEA 2022a). Which type of electrolyser is most suitable depends on the project and local context. After production, hydrogen can be stored in caverns, cylinders or tanks, and transported by pipeline or ship, as liquid hydrogen, ammonia or using liquid organic hydrogen carriers (IRENA 2022a). Alternative hydrogen production routes with potentially low emissions are production from fossil fuels combined with carbon capture and storage called 'blue hydrogen' (Bauer et al. 2022a), and biomass gasification (Antonini et al. 2021). Low-emission hydrogen can be implemented in existing applications of hydrogen in nitrogen fertiliser production and refining, as well as in new applications, for example as reducing agent in the steel industry, as feedstock in the chemical industry, in industrial processes requiring high temperatures, in the energy sector for long-term storage of energy and in the transport sector, either directly or as hydrogen-based fuel (Bashmakov et al. 2022).

## 1.3 Greenhouse gas footprints of CCU and green hydrogen

Reducing GHG emissions caused by a product or process requires a measure of what those emissions are. This measure, called the GHG footprint or climate footprint, can be used to compare technologies and to guide optimisation efforts towards those process steps responsible for the largest emissions. The GHG footprint and other environmental impacts of a product or process throughout its life cycle can be quantified with a method called life cycle assessment (LCA).

### 1.3.1 Quantification of GHG footprints

LCA requires the formulation of the goal and scope, which determines the geographical and temporal scale, functional unit (the unit in which all inputs and outputs are measured), and system boundaries (to specify which processes are included and excluded from the assessment). The choice of which life cycle stages are included depends on the goal of the LCA and are typically 'cradle-to-gate', including raw material extraction and manufacturing, or 'cradle-to-grave', additionally including distribution, product use and 'end-of-life', e.g., recycling or disposal. Resource extractions and emissions data from the relevant life cycle stages are compiled in the life cycle inventory, and converted into impact scores using an impact assessment method, for example ReCiPe2016 (Huijbregts et al. 2017). Midpoint indicators combine all contributions to a specific mode of change to the natural environment, such as acidification or global warming. For the latter, emissions of greenhouse gases throughout the entire life cycle are multiplied by their global warming potentials to arrive at one metric, kg of CO<sub>2</sub>-equivalents, reflecting the overall climate change impact of the product or process: the GHG footprint. Furthermore, endpoint indicators provide a measure of aggregated effects on areas of protection: human health, natural resources and ecosystem quality. These endpoint indicators may facilitate interpretation of the LCA results, but also increase uncertainty due to further aggregation (Hauschild and Huijbregts 2015).

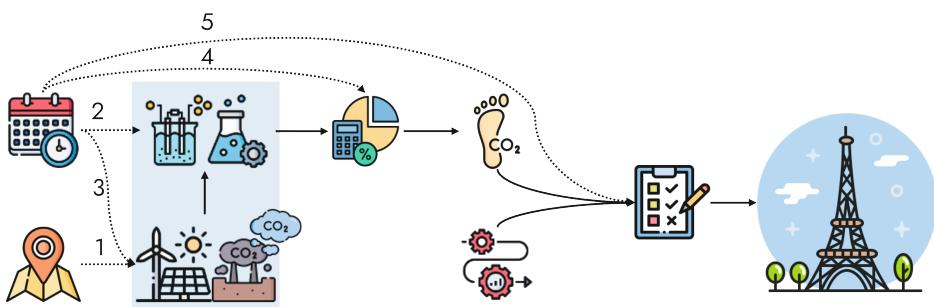
### 1.3.2 Variation in GHG footprints

Previous LCAs have identified factors that contribute to the variation in GHG footprints of green hydrogen and CCU. These include technological factors such as the GHG intensity of the electricity source used in the process (Kätelhön et al. 2019; Mac Dowell et al. 2021), the specific electrolyser technology (Zhang et al. 2017; Gerloff 2021), CO<sub>2</sub> capture technology (von der Assen et al. 2016), CO<sub>2</sub> conversion process (Cuéllar-Franca and Azapagic 2015; Thonemann and Pizzol 2019), and the status of technological development (Thonemann and Schulte 2019; Valente et al. 2020). Other factors relate to the methodological choices in quantifying the GHG footprint, such as the choice of system boundaries and how to approach multifunctionality. This is applicable to the electrolysis process that co-produces

hydrogen and oxygen (Bargiacchi et al. 2020), and the CCU production chain where an industrial facility or power plant produces a primary product as well as the CO<sub>2</sub> that is used as feedstock in the secondary CCU process (Fernández-Dacosta et al. 2019; Müller et al. 2020a). To evaluate whether or not emissions are sufficiently reduced, a comparison of the footprint to a benchmark is necessary. This can be the footprint of the conventional product, an alternative mitigation option, or an emission threshold. Depending on the selected benchmark, even the evaluation of the footprint can differ (Cuéllar-Franca et al. 2019; Mac Dowell et al. 2021).

### 1.3.3 Context-dependency of GHG footprints

The assessment of Paris compatibility of green hydrogen and CCU hinges upon their technological maturity and GHG footprints. To facilitate the assessment, the different factors contributing to variation in GHG footprints are grouped into technological, spatial and temporal context, and their interactions visualised in Figure 1.2.



**Figure 1.2. Interactions (dotted arrows) between temporal, spatial and technological context affect the GHG footprints of CCU and green hydrogen, and ultimately their Paris compatibility.**  
 1. Technological inputs such as grid mix composition and meteorological conditions affecting the GHG intensity of renewable electricity are location-specific; 2. The state of development of technologies such as the efficiency of electrolyzers, solar panels and CO<sub>2</sub> conversion technologies evolves over time; 3. The grid mix composition and availability of CO<sub>2</sub> sources evolves over time; 4. The suitability of methodological choices evolves over time; 5. The temporal context informs the selection of the benchmark.

#### *Technological context*

The specific technological circumstances, i.e., different types of inputs and technological processes in CCU and green hydrogen production are considered

under technological context. The inputs include various electricity sources and CO<sub>2</sub> sources, while technological processes include electrolyser types, hydrogen transport modes, and CO<sub>2</sub> capture and conversion processes.

### ***Spatial context***

The technology's location affects the GHG footprint through the availability of resources like CO<sub>2</sub> sources and types of renewable energy, and through transport distance if transport of the product is required. Furthermore, the composition of the electricity grid mix differs per region or country, and local meteorological conditions such as solar irradiation and wind speed affect the location-specific GHG intensity of renewable electricity (Bosmans et al. 2021; Dammeier et al. 2023).

### ***Temporal context***

While temporal change is unidirectional, it yields multidimensional effects. Firstly, the temporal and technological context interact through the state of development of technologies, and through the evolution of inputs such as CO<sub>2</sub> source availability and increased renewable energy in the electricity mix. Secondly, temporal context informs methodological choices in GHG footprint quantification, like system boundaries and approach to dealing with multifunctionality. ISO standards for LCA leave freedom to deal with multifunctionality (ISO 2006), but the applicability of the possible methods evolves with availability of, - and demand for involved products and sources. For example, reduced demand for co-produced oxygen would alter economic allocation factors, and in a net-zero CO<sub>2</sub> world, unabated fossil point sources would no longer be available to substitute. Lastly, the temporal context determines the benchmark, requiring increasingly steeper emission reductions as well as comparisons to alternative mitigation technologies, beyond the past's relative reductions compared to conventional production.

## **1.4 CCU and green hydrogen's appeal and expectations**

Although still under development and not yet implemented at large scale, CCU technologies and green hydrogen speak to the imagination. For CCU, it is the possibility of creating valuable products from a waste (Cuéllar-Franca and Azapagic

2015; IEA 2019b), contributing to circularity and sustainable waste processing (Lehtonen et al. 2019; Marchese et al. 2021), decoupling carbon demand from fossil resources (Kätelhön et al. 2019), and facilitating a less disruptive energy transition by making use of existing infrastructure (Ueckerdt et al. 2021; Mertens et al. 2023). For green hydrogen, it is the promise of a versatile and low-emission energy carrier that would help decarbonise a range of hard-to-abate sectors (IEA 2019c), facilitate the integration of renewables into the energy system (IEA 2019c), bring prosperity to developing countries with large renewable energy potentials through the introduction of a new trade commodity (Sadik-Zada 2021), increase energy access (Falcone et al. 2021), and increase water availability through desalination (FSR 2020).

#### 1.4.1 Support for the implementation of CCU and green hydrogen

The implementation of green hydrogen and CCU is supported by policies, strategies, funds and institutions worldwide. In Europe, support is provided for research, development, demonstration and scale-up of green hydrogen and CCU technologies via national funding programs, the European Union (EU) Innovation Fund, Horizon Europe and the proposed Net Zero Industry Act (Olfe-Kräutlein et al. 2021; European Commission 2022a, 2023a). The European Commission targets green hydrogen production of 10 million tonnes in the EU and 10 million tonnes import in 2030 (European Commission 2022b), established the European Hydrogen bank to bridge the investment gap (European Commission 2023b), and starting from 2025, aviation fuel suppliers at EU airports are required to mix at least 2% of synthetic aviation fuels in their blend, reaching 70% in 2050 (European Commission 2023c). Outside of Europe, the United States' inflation reduction act includes tax credits for CCU and green hydrogen (US House of Representatives 2022), Canada's federal budget includes an investment tax credit for equipment for CCU projects (Government of Canada 2021), China targets green hydrogen production of 100 to 200 thousand tonnes per year in 2025 (Reuters 2022) and Japan plans to invest 15 trillion yen (100 billion euros) in green hydrogen supply over the next 15 years (Reuters 2023).

### 1.4.2 High expectations on the role of CCU and green hydrogen in achieving climate-neutrality

Electrolysers and CCU technologies are referred to as 'net-zero technologies' in the Net Zero Industry Act (European Commission 2023a), and various other labels have been used in policy documents to emphasise the climate benefits of green hydrogen and CCU. Popular alternatives include 'carbon-neutral', 'climate-neutral', 'zero-emissions' and 'clean' (EPKS 2021; Government of Canada 2021; Cheng and Lee 2022; US House of Representatives 2022; G7 Ministers of Climate Energy and Environment 2023; G20 Energy Transitions Ministers 2023).

Furthermore, green hydrogen has been coined a "*silver bullet*" (Euractiv 2022), "*the driving force of our energy system of the future*" (Recharge 2022), and "*the gas of the future*" (Euronews 2022) by the energy secretary of the United Kingdom, the European Commissioner for climate action, and the German chancellor, respectively. The European Parliamentary Research Service opened its briefing on the EU hydrogen policy by stating that "*hydrogen is expected to play a key role in a future climate-neutral economy, enabling emission-free transport, heating and industrial processes as well as inter-seasonal energy storage*" (EPKS 2021). A similar role is set aside for CCU. CCU is considered a "*key technology to contribute to climate change mitigation and to meet the Paris Agreement*" by EU policymakers (Palm and Nikoleris 2021), and is connected to achieving climate-neutrality by 2050 in the EU (Thielges et al. 2022) and the 2050 net-zero GHG target in the US (US Department of Energy 2023a).

The high expectations of policymakers for CCU and green hydrogen as net-zero technologies for achieving climate-neutrality are not shared by everyone. Researchers have stressed the importance of avoiding negative externalities and risks (Cremonese et al. 2023), considering justice aspects (Dillman and Heinonen 2022; Swennenhuis et al. 2022), and assessing full life-cycle environmental impacts (Fernández-Dacosta et al. 2019; Osman et al. 2022), in particular the high energy and material requirements for CCU (Abanades et al. 2017; Mac Dowell et al. 2017; Kätelhön et al. 2019) and green hydrogen (Bareiß et al. 2019; Mac Dowell et al. 2021).

## 1.5 Problem setting

Previously calculated GHG footprints of green hydrogen and CCU range from zero or even negative, to larger than their fossil counterparts. This indicates that the implementation of green H<sub>2</sub> and CCU technologies does not inherently result in emission reductions. Despite this, policymakers have already started to rely on green hydrogen and CCU technologies to contribute to deep emissions reductions, as reflected in policies that incentivise the development and implementation of these technologies.

Policies and decision-making in the energy and sustainability transition have increasingly been informed by life-cycle thinking and LCA results (Sala et al. 2021; Hellweg et al. 2023). The interpretation of LCA studies has, however, not been guided by or interpreted in the context of the Paris Agreement. This is particularly relevant for national climate plans pledged under the Paris Agreement in the near term (NDCs) and for the mid-century (LT-LEDS), which are developed to contribute to achieving the Paris Agreement's long-term temperature goals (Rogelj et al. 2023). Article 4.1 of the Paris Agreement stresses the need to align emission reductions with scientific studies (UNFCCC 2016), but how can policymakers make sense of such diverging GHG footprints for green hydrogen and CCU, depending on a myriad of factors? A better understanding is needed of these footprints, their development over time, and the conditions under which green hydrogen and CCU can contribute to achieving the Paris Agreement's long term temperature goals, both in the near term and in the long term.

## 1.6 Aim and outline

Given the high expectations in the policy arena, the main aim of this thesis is to identify the conditions under which green hydrogen and CCU technologies are Paris-compatible.

Specifically, to identify these conditions, I assess how the GHG footprints of green hydrogen and CCU depend on temporal, spatial and technological context.

Different combinations of choices in contexts, choices in the calculation of the emissions and choices in benchmark are covered in Chapters 2 to 5, as summarised in Table 1.1.

Chapter 2 defines Paris compatibility and describes the framework for assessing Paris compatibility. It then evaluates the Paris compatibility of 74 CO<sub>2</sub> capture and utilisation processes in 2030 and 2050, covering different CO<sub>2</sub> sources and conversion routes. For 2030 and 2050, different multifunctionality approaches for the CO<sub>2</sub> sources are used, as well as different benchmarks and electricity mixes.

Chapter 3 focusses on the environmental benefits of a CO<sub>2</sub> capture and utilisation technology that produces urea from a residual gas in the steel industry, in comparison to conventional urea production and existing applications of the residual gas for five European countries.

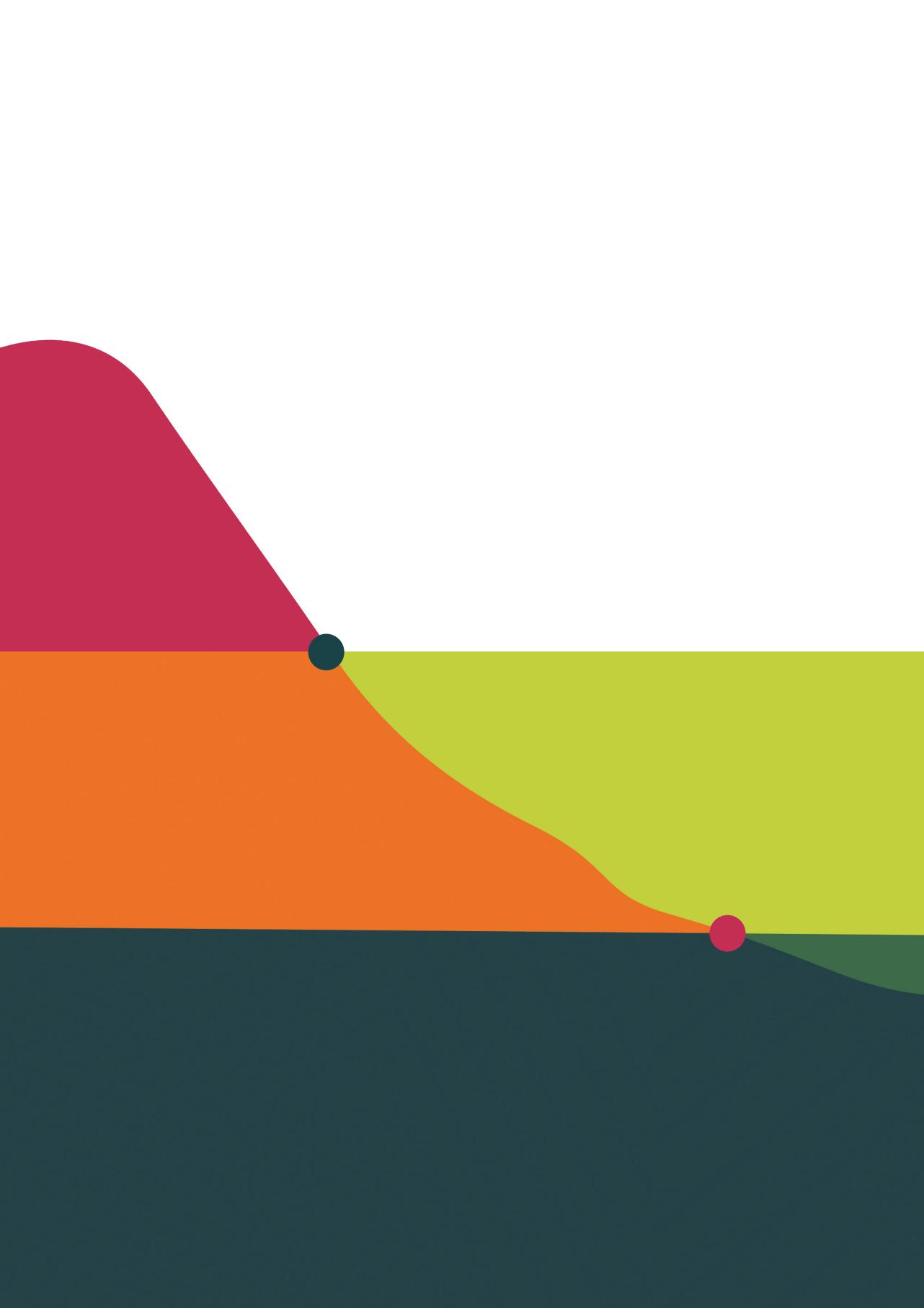
Chapter 4 assesses how the GHG footprint and emission reduction potential of green hydrogen depend on the current and future electricity source, accounting of co-produced oxygen and the grey or blue hydrogen benchmark against which the production emissions are compared.

Chapter 5 looks at the global supply chain of green hydrogen production for location-specific renewable electricity sources, different electrolyser technologies and transport options. The GHG footprints are compared to those of grey and blue hydrogen, and absolute, policy-based emission thresholds for hydrogen to be considered sustainable.

Chapter 6 provides a synthesis of the findings in the previous chapters and offers conclusions, reflections, implications and a research outlook.

**Table 1.1. Coverage of different choices in temporal, spatial and technological context, GHG footprint calculation method and benchmark for the evaluation of the GHG footprints of CCU technologies (Chapter 2 and Chapter 3) and green hydrogen (Chapter 4 and Chapter 5).**

	Chapter 2 Limits to Paris compatibility of CO <sub>2</sub> capture and utilisation	Chapter 3 Environmental benefits of urea production from basic oxygen furnace gas (BOFG)	Chapter 4 The many greenhouse gas footprints of green hydrogen	Chapter 5 Worldwide greenhouse gas emissions of green hydrogen production and transport
Spatial scale	Global	Europe	Europe	Global
Temporal scale	2030 2050	2020 2050	2020 2030	2030
Technological context	Electricity source: 1.5°C grid  Types of CO <sub>2</sub> sources: fossil, atmospheric, biogenic  44 CO <sub>2</sub> use processes	Electricity source: grid  BOFG uses: urea, heat, electricity, flaring  Fate of excess CO <sub>2</sub> : venting to air or storage	Electricity source: wind, solar, grid	Electricity source: wind, solar, hydro, nuclear, grid, excess  Electrolyser types: polymer electrolyte membrane, alkaline, solid oxide
				Transport modes: pipeline, shipping
Life cycle stages	Cradle-to-gate + end-of-life	Cradle-to-gate	Cradle-to-gate	Cradle-to-gate  Cradle-to-point of delivery
Functional unit	1 kg of CO <sub>2</sub> utilised	1 t of BOFG used	1 kg of H <sub>2</sub> produced	1 kg of H <sub>2</sub> produced  1 kg of H <sub>2</sub> produced and transported
Multi-functionality	Co-produced O <sub>2</sub> ; substitution of O <sub>2</sub> production  CO <sub>2</sub> source: - substitution in 2030 - no substitution in 2050 for fossil CO <sub>2</sub>	BOFG: all impacts allocated to main product (steel)  Waste heat: - all impacts allocated to main product (steel) - economic allocation	Co-produced O <sub>2</sub> : - all impacts allocated to main product (H <sub>2</sub> ) - economic allocation - substitution of O <sub>2</sub> production	Co-produced O <sub>2</sub> ; all impacts allocated to main product (H <sub>2</sub> )
Benchmark	50% of conventional product's emissions  Zero emissions	Conventional urea  Alternative uses of BOFG	Grey and blue hydrogen  Alternative uses of renewable electricity	Grey and blue hydrogen  Emission thresholds for sustainability



# Limits to Paris compatibility of CO<sub>2</sub> capture and utilisation

2

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## Abstract

The Paris Agreement's temperature goals require global CO<sub>2</sub> emissions to halve by 2030 and reach net zero by 2050. CO<sub>2</sub> capture and utilisation (CCU) technologies are considered promising to achieve the temperature goals. This paper investigates which CCU technologies – using atmospheric, biogenic or fossil CO<sub>2</sub> – are Paris-compatible, based on life-cycle emissions and technological maturity criteria. We systematically gathered and harmonised CCU-technology information for both criteria and found that CCU with technology readiness levels (TRLs) of 6 or higher can be Paris-compatible in 2030 for construction materials, enhanced oil recovery, horticulture industry, and some chemicals. For 2050, considering all TRLs, we showed that only products storing CO<sub>2</sub> permanently or produced from only zero-emissions energy can be Paris-compatible. Our findings imply that research and policy should focus on accelerating the development of those CCU technologies that may achieve (close-to-)zero net emissions, avoiding lock-in by CCU technologies with limited net emission reductions.

## 2.1 Introduction

In the 2015 Paris Agreement, almost all of the world's nations committed to collectively hold "*the increase in the global average temperature to well below 2°C above pre-industrial levels and pursuing efforts to limit the temperature increase to 1.5°C above pre-industrial levels*" (UNFCCC 2016). This is also known as the Paris Agreement's long-term temperature goal (LTTG). Climate change mitigation pathways with 'no or low overshoot' of 1.5°C (Rogelj et al. 2018) are compatible with the Paris Agreement's LTTG (Mace 2016; Schleussner et al. 2016; Wachsmuth et al. 2018; Geiges et al. 2020) and characterised by two key numbers: in 2030, global net anthropogenic carbon dioxide (CO<sub>2</sub>) emissions are halved compared to 2020 emissions, and in 2050, net CO<sub>2</sub> emissions are zero (Rogelj et al. 2018). This net zero CO<sub>2</sub> target implies that no sector can be excluded from deep emission reductions, some technologies that still have significant emissions need to be phased out, and a range of technologies is required to reach these emission reductions.

Carbon [dioxide] capture and utilisation (CCU) is among these options for potential emission reduction and is defined here as *a process in which CO<sub>2</sub> is technologically captured from CO<sub>2</sub> point sources or ambient air, and is subsequently used in or as a product*. The reason CCU could contribute to climate change mitigation is that it replaces fossil feedstocks, avoids upstream emissions, and temporarily keeps CO<sub>2</sub> out of the atmosphere until re-emitted in the use-phase of the product (Chauvy et al. 2019; Detz and van der Zwaan 2019; Hepburn et al. 2019; Kätelhön et al. 2019; Núñez-López et al. 2019; Thonemann and Pizzol 2019). CCU is distinguished from (permanent) carbon dioxide removal (CDR) of atmospheric CO<sub>2</sub>. The two terms only overlap if CO<sub>2</sub> in a CCU product has recently been removed from the atmosphere and is never re-emitted (IPCC 2018a). CCU appeals to policymakers and the general public as it is seen as part of the circular economy and a form of sustainable waste processing (Styring et al. 2011). It also appeals to industry as CCU creates value from waste through CO<sub>2</sub>-based products (Cuéllar-Franca and Azapagic 2015; IEA 2019b), while avoiding the storage costs and concerns of geological storage of captured CO<sub>2</sub>, known as carbon [dioxide] capture and storage (CCS) (Lilliestam et al. 2012).

Yet, the relevance of CCU in climate change mitigation is questioned in literature, based on several concerns: i) CCU products may not always substantially reduce emissions compared to their conventional counterparts that do not require the energy-intensive CO<sub>2</sub> capture and conversion steps (Bruhn et al. 2016; García-Herrero et al. 2016; Abanades et al. 2017; Mac Dowell et al. 2017); ii) Utilisation of captured CO<sub>2</sub>, rather than permanent geological storage, may result in a higher global warming impact, as utilised CO<sub>2</sub> is typically re-emitted when the CCU-product is used or disposed of (Cuéllar-Franca and Azapagic 2015; Abanades et al. 2017); iii) CCU may not be economically feasible because of the high financial costs associated with the energy-intensive CO<sub>2</sub> capture and conversion steps (Abanades et al. 2017; Fernández-Dacosta et al. 2019); and iv) CCU may form a political distraction from reducing CO<sub>2</sub> emissions, in particular when replacing CCS, as the scale at which CO<sub>2</sub> could be utilised is limited compared to the scale at which CO<sub>2</sub> could be stored geologically (Mac Dowell et al. 2017).

The goal of this review is to provide conceptual clarity on what CCU is, on what can be expected from different CCU technological routes, in particular in reaching the Paris Agreement's LTTG. We first describe the different process steps and varieties of CCU technologies. Next, we present a framework to assess 'Paris compatibility' in the context of CCU, using criteria based on technological maturity and greenhouse gas emissions reductions. We then show the results of a systematic review of the CCU literature following this framework. Lastly, we discuss our findings and provide a research and policy outlook for climate change mitigation through CCU. For an overview of acronyms used, see Supplementary Table 2.1.

## 2.2 What is CCU?

### 2.2.1 Scope of CCU

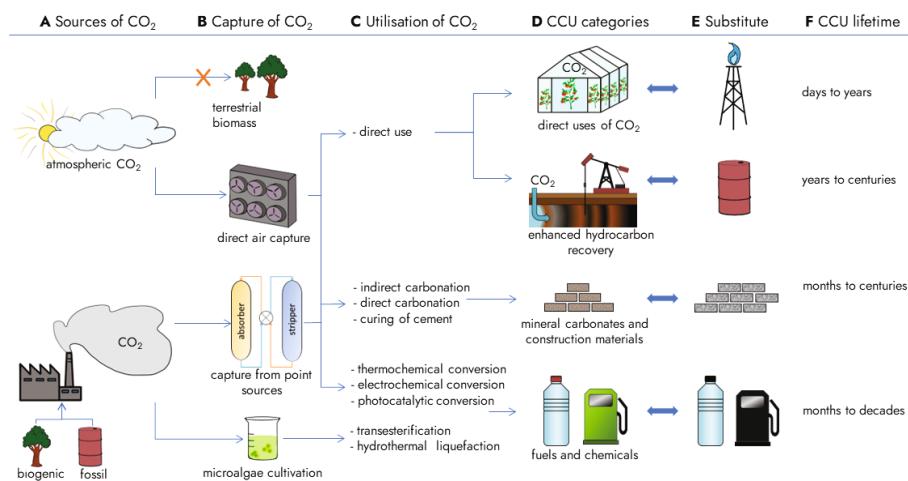
In line with our definition of CCU, we defined six key characteristics of CCU (Figure 2.1):

- A. **Sources of CO<sub>2</sub>.** CO<sub>2</sub> can originate from fossil fuel or biomass combustion in power plants or industrial plants, from industrial processes such as the

calcination reaction in cement production or biomass fermentation, or from the atmosphere directly using direct air capture (DAC).

- B. **Capture of CO<sub>2</sub>.** CO<sub>2</sub> is captured technologically on industrial scale, by separating CO<sub>2</sub> from a bulk gas stream or the atmosphere using a solvent or sorbent, membrane, cryogenics, or by industrially cultivated organisms, such as microalgae, to photosynthesise CO<sub>2</sub> into biomass.
- C. **Utilisation of CO<sub>2</sub>,** CO<sub>2</sub> is used directly, or indirectly by converting CO<sub>2</sub> into a range of products, often requiring electricity, heat, and/or catalysts.
- D. **CCU categories.** Resulting CCU products can be categorised as: direct uses; enhanced hydrocarbon recovery (EHR); mineral carbonates and construction materials; and fuels and chemicals.
- E. **Substitute.** A CCU product is assumed to *replace* a product in the conventional economy with the same chemical structure, composition or characteristics, typically produced from fossil fuels and referred to as the substitute (Zimmermann et al. 2020).
- F. **CCU lifetime.** CO<sub>2</sub> is, depending on the CCU product, either permanently stored or released to the atmosphere after a certain period of time called its lifetime, ranging from days to centuries. For example, for fuels, the utilised CO<sub>2</sub> is emitted to air upon combustion. For the purpose of this paper, we define storage as reaching permanency when it has a duration consistent with geological timescales: centuries or longer.

Since we consider CCU for climate change mitigation, we exclude the use of CO<sub>2</sub> from natural reservoirs, as this source of CO<sub>2</sub> does not reduce atmospheric CO<sub>2</sub> concentrations (NETL 2010). Our definition of CCU constrains CCU to processes that 'technologically capture CO<sub>2</sub>', thereby including industrial and engineered biological processes such as CO<sub>2</sub> capture from flue gases by microalgae, while excluding land-based CO<sub>2</sub> sequestration in biomass (in contrast to for example Detz & van der Zwaan (2019) and Hepburn et al. (2019)). The use of biomass for energy and materials is therefore also not in the scope of this review.



**Figure 2.1. Scope of CO<sub>2</sub> capture and utilisation.** This overview of CO<sub>2</sub> capture and utilisation includes sources of CO<sub>2</sub> (A), capture of CO<sub>2</sub> (B), examples of utilisation processes (C) leading to different CCU products in four categories (D), each with its substituted product in the conventional economy (E) and lifetime (F). CO<sub>2</sub> capture via terrestrial biomass (top process under "Capture of CO<sub>2</sub>") is not included in the this paper's CCU definition.

CCU is sometimes connected to CDR. CDR is a necessity to limit warming to 1.5°C (Rogelj et al. 2018) and has been defined as '*anthropogenic activities removing CO<sub>2</sub> from the atmosphere and durably storing it in geological, terrestrial, or ocean reservoirs, or in products*' (IPCC 2018a). CDR includes methods like bioenergy with carbon capture and storage (BECCS) and direct air carbon capture and storage (DACCs) (de Coninck et al. 2018). CCU can only be classified as CDR, following the criteria formulated by Tanzer and Ramírez (2019), when i) physical CO<sub>2</sub> is removed from atmosphere by capturing CO<sub>2</sub> directly from air or capturing CO<sub>2</sub> from a biogenic source; ii) the CO<sub>2</sub> is stored permanently and not re-emitted to the atmosphere at a later point; and iii) the net quantity of CO<sub>2</sub> stored permanently through CCU is greater than the quantity of CO<sub>2</sub> emitted over the product's full life cycle, including the use of the product and the emissions associated with the energy required for the CO<sub>2</sub> capture and conversion processes. When the criteria for CDR are not met, but the life-cycle emissions are lower than for the substituted product, CCU is considered a climate change mitigation measure. Avoided emissions compared to the substituted product may be presented as a negative number, but should not be confused with physical removal of CO<sub>2</sub> from the atmosphere (Tanzer and Ramírez 2019).

## 2.2.2 Capture of CO<sub>2</sub>

### *CO<sub>2</sub> capture from a point source*

CO<sub>2</sub> is typically captured from a point source such as flue gas streams at power plants or industrial plants, or from industrial processes where CO<sub>2</sub> is produced as a by-product. Capturing CO<sub>2</sub> from a bulk gas stream entails separating the CO<sub>2</sub> from the rest of the stream. Sometimes CO<sub>2</sub> separation is already required in the primary production process, for example in ammonia synthesis, natural gas processing and biogas upgrading to biomethane (IPCC 2005). This leads to a high-purity stream of CO<sub>2</sub> along with the primary product.

2

Depending on the concentration and components present in a gas stream, one of three main separation processes may be most suitable (Ramírez-Santos et al. 2018): i) absorption of CO<sub>2</sub> in a liquid solvent, ii) adsorption of CO<sub>2</sub> onto a solid, and iii) using a membrane to separate CO<sub>2</sub> through selective permeability (Leoncio 2018). In CO<sub>2</sub> absorption processes using amine-based solvents like monoethanolamine, CO<sub>2</sub> is chemically bound, after which the CO<sub>2</sub> is stripped to allow the solvent to absorb CO<sub>2</sub> again in a next cycle. This is called the regeneration of the solvent and requires heat. Besides single amine absorption, amine blends are in development where amines are combined for complimentary characteristics. An integrated CO<sub>2</sub> capture and utilisation process has also been proposed to reduce the overall energy demand, absorbing CO<sub>2</sub> from raw natural gas in methanol, after which conversion takes place (Jens et al. 2019). Other developments include ionic liquids, which are liquids composed entirely of ions with a melting point of less than 100 °C (Bui et al. 2018). Physical adsorption research has focussed on improving the adsorbents as well as the adsorption processes of regeneration (Bui et al. 2018). Alternatives like high temperature solid adsorption of CO<sub>2</sub> require less energy input for regeneration compared to low temperature liquid absorption (Ho et al. 2013; Ben-Mansour et al. 2016; Manzolini et al. 2020). The use of membranes for CO<sub>2</sub> separation is based on creating configurations specially for CO<sub>2</sub> selectivity from polymer or ceramic materials (Ben-Mansour et al. 2016).

Other emerging CO<sub>2</sub> capture technologies are high temperature solid looping systems: calcium carbonate looping in which calcium oxide is used as a sorbent that binds CO<sub>2</sub> to form limestone, and chemical looping combustion in which

a metal oxide is used to separate CO<sub>2</sub> from other components in the flue gas, foregoing the need for gas separation, followed by an exothermic metal oxide regeneration reaction (Abanades et al. 2015; Bui et al. 2018). Cryogenic systems are based on the differences in temperatures and pressure at which constituent gases in flue gas become liquid, removing CO<sub>2</sub> from the bulk stream by cooling and condensation (Ben-Mansour et al. 2016). CO<sub>2</sub> from a point source can also be captured photosynthetically or electro-synthetically into biomass by living organisms (Styring et al. 2011; Kassim and Meng 2017): microalgae are cultivated at CO<sub>2</sub> concentrations of 5-20%, making flue gas a suitable source, and have a conversion efficiency of solar energy into chemical energy higher than that of terrestrial plants (3-8% instead of 0.5%) (Wang 2019). Co-location of the facility at the site of the point source is important to avoid the need for transporting flue gas (Nilsson et al. 2020). Other options for biofixation include microorganisms such as acetogenic bacteria (Ramachandriya et al. 2013) or anaerobic CO<sub>2</sub>-sequestering bacteria (Wang 2019).

#### ***CO<sub>2</sub> capture from ambient air***

Direct air capture of CO<sub>2</sub> can be performed using a range of separation processes. Since atmospheric CO<sub>2</sub> concentrations (approximately 400 ppm) are 100-300 times lower than for point sources, energy requirements, and therefore costs, for DAC are substantially larger than for point source capture (Fasihi et al. 2019; de Jonge et al. 2019). Expert elicitation leads to expected cost declines to around 200 USD per tonne of CO<sub>2</sub> captured by 2050 (Shayegh et al. 2021), still higher than the 15-35 USD per tonne of CO<sub>2</sub> captured for high-purity point sources and 60-120 USD per tonne of CO<sub>2</sub> captured for lower-purity point sources, e.g. steel or cement production (Leeson et al. 2017; IEA 2019b). An exception would be when lower purity CO<sub>2</sub> streams are sufficient, such as for microalgae (Wilcox et al. 2017). The two main categories of DAC methods are based on: i) amine-functionalised solid sorbents, which require regeneration at low temperatures or via moisturising (Kulkarni and Sholl 2012); and ii) alkaline hydroxide capture solutions, which require high temperature solvent regeneration (Keith et al. 2018). Less energy-intensive regeneration processes are under development, including electrochemical regeneration (Shu et al. 2020) and bipolar membrane electrodialysis (Sabatino et al. 2020).

## 2.2.3 Utilisation of CO<sub>2</sub>

### *Direct uses of CO<sub>2</sub>*

CO<sub>2</sub> can be used directly, without conversion, in several sectors. In horticultural production, elevating CO<sub>2</sub> concentrations in greenhouses increases crop yields by approximately 50% (Oreggioni et al. 2019). This process is called CO<sub>2</sub> enrichment and is traditionally achieved by the combustion of fossil fuels such as diesel (Oreggioni et al. 2019) or natural gas (CE Delft 2018), which has the dual purpose of greenhouse heating. Since more CO<sub>2</sub> is required to reach the desired CO<sub>2</sub> concentration than is produced for heat, captured CO<sub>2</sub> can be used (Oreggioni et al. 2019). CO<sub>2</sub> can also be used directly as a refrigerant for supermarket applications, replacing hydrofluorocarbons with higher global warming potentials (McLinden et al. 2017), thereby reducing risks of leakage and associated global warming impacts (Gullo et al. 2018). Furthermore, CO<sub>2</sub> can be used as a carbonating agent in sugar production and soft drinks, as a solvent for the extraction of flavours, in the decaffeination process, as dry ice, in fire extinguishers, and in the pharmaceutical industry as respiratory stimulant (Cuéllar-Franca and Azapagic 2015; Duraccio et al. 2015; Mac Dowell et al. 2017)

### *Enhanced hydrocarbon recovery*

Enhanced oil recovery (EOR) is a method in which CO<sub>2</sub> is injected into the subsurface to recover oil from almost depleted reservoirs. EOR is a mature technology that has been practiced commercially for decades, starting in the early 1970s in North America (NETL 2010). Similarly, CO<sub>2</sub> can be used to recover natural gas from coalbeds (enhanced coalbed methane, ECBM), although there are currently no active ECBM projects (Global CCS Institute 2021). While the source of CO<sub>2</sub> for EHR is typically natural CO<sub>2</sub> reservoirs (not part of our definition of CCU), EHR can also be performed with CO<sub>2</sub> captured from point sources or DAC (Wilcox et al. 2017).

### *Mineral carbonates and construction materials*

Mineral carbonation is a process in which CO<sub>2</sub> reacts with magnesium or calcium oxides, -hydroxides or -silicates in an exothermic reaction, forming stable carbonate products (Sanna et al. 2014): examples of magnesium or calcium-rich materials are mined minerals (Khoo et al. 2011) or industrial residues such as slag from steel production (Xiao et al. 2014; Pan et al. 2016), red gypsum (Rahmani et al. 2014),

fly ash (Mo et al. 2017; Chen et al. 2020), or desalination brines (Galvez-Martos et al. 2016). These processes can be categorised into direct or indirect carbonation processes (Xiao et al. 2014): in the direct carbonation process, a gas-solid reaction takes place between CO<sub>2</sub> and the grinded minerals. The produced carbonates can be used in concrete, asphalt, and other construction practices (Global CO<sub>2</sub> Initiative 2016). In the indirect carbonation process, magnesium or calcium ions are first extracted in an alkaline solution, after which they react with CO<sub>2</sub> to form precipitated magnesium carbonate (PMC) (Galvez-Martos et al. 2016), precipitated calcium carbonate (PCC) (Xiao et al. 2014) or nano calcium carbonate (NCC) (Lee et al. 2020). These carbonates can be used as a pigment or filler material in the production of paper, plastics and pharmaceuticals (Mattila et al. 2014).

CO<sub>2</sub> can be used to cure concrete, absorbing CO<sub>2</sub> instead of steam in the hardening process. Steam-cured concrete normally re-absorbs roughly 30% of its production's CO<sub>2</sub> emissions during its lifetime (Xi et al. 2016; Cao et al. 2020), so emissions are not reduced by CO<sub>2</sub>-accelerated curing in itself. Instead, the reduction is achieved due to the lower steam requirement (El-Hassan and Shao 2014) and the improvements in mechanical properties, reducing the amount of cement required (Zhan et al. 2016; Monkman and MacDonald 2017; Liang et al. 2020), in a similar way as the addition of 'cementitious' materials like carbonated minerals reduces the need for cement in concrete (Mo et al. 2017; Ruan and Unluer 2017). Other options under development include the production of carbon nanomaterials like graphene (Liu et al. 2020b), nanofibers or nanotubes from CO<sub>2</sub> (Lau et al. 2016), which can be used in construction, reducing energy and material demand in the manufacturing process (Zhai et al. 2016).

### ***Fuels and chemicals***

Fuels and chemicals directly based on fossil fuels or fossil feedstocks are energy-dense products. Therefore, producing these chemicals or fuels from CO<sub>2</sub> often requires an energy-intensive conversion process at high pressure and/or increased temperatures, supported by catalysts, because CO<sub>2</sub> is an inert and thermodynamically stable molecule. Conversion processes include thermochemical-, electrochemical- and photocatalytic conversion.

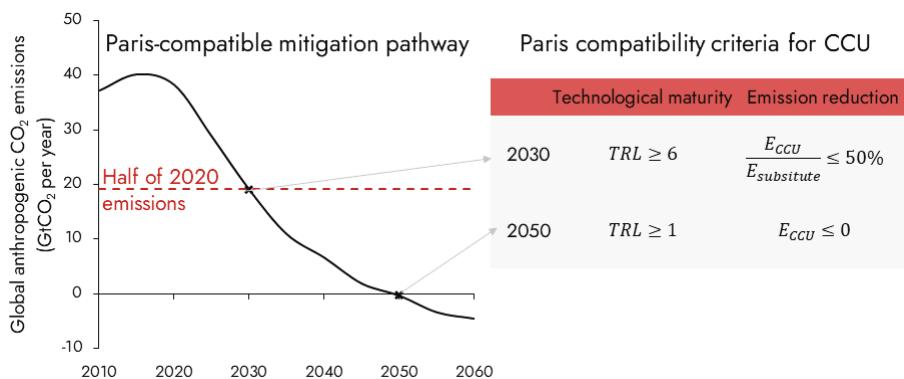
In thermochemical conversion (sometimes referred to as 'hydrogenation') CO<sub>2</sub> and H<sub>2</sub> are separately prepared and subsequently combined (Centi and Perathoner 2009). The hydrogen supplies part of the energy needed for the CO<sub>2</sub> conversion process. This process allows for the production of chemicals and fuels that would otherwise be of fossil origin, like methane or methanol (Thonemann and Pizzol 2019). Methanol can in turn serve as a feedstock for the production of other chemicals and fuels, such as ethylene (Ioannou et al. 2020), polyols (Hoppe et al. 2018) and dimethyl ether (DME) (Matzen and Demirel 2016). Methane production can be a method for long-term storage of renewable energy from intermittent electricity, producing H<sub>2</sub> with water electrolysis, followed by CO<sub>2</sub> hydrogenation to methane (Sabatier reaction), and methane combustion for power generation at a later time (Power-Methane-Power) (Wevers et al. 2020). Another key hydrogenation process is the synthesis of liquid hydrocarbon fuels, which consists of synthesis gas (syngas) production via the conversion of CO<sub>2</sub> to carbon monoxide (CO) in the reverse water-gas shift reaction (rWGS) and mixing it with H<sub>2</sub>; or via steam methane reforming (SMR); and finally using the Fischer-Tropsch process to create hydrocarbon chains (Cuéllar-Franca et al. 2019). The stoichiometric ratio of H<sub>2</sub> to CO in syngas can be adjusted to the targeted end-products, as currently occurs via the water-gas shift (WGS) reaction in conventional syngas-based processes (Artz et al. 2018).

In photocatalytic conversion, CO<sub>2</sub> is converted to for example methane or methanol using sunlight, water and dye-sensitised semiconductors (Trudewind et al. 2014). Electrochemical reduction, where CO<sub>2</sub> is reduced at atmospheric temperature using electricity (Aldaco et al. 2019), can be used to produce for example ethanol (Thonemann and Pizzol 2019), methanol (Qu et al. 2005; Pérez-Fortes et al. 2016), dimethyl carbonate (DMC) (Garcia-Herrero et al. 2016), formate (Zhao and Wang 2016) or formic acid (Aldaco et al. 2019). When CO<sub>2</sub> is captured by microalgae, the microalgae can be converted into fuels or chemicals, e.g. via transesterification or hydrogenation of algal oil (Soratana et al. 2013; Zaimes and Khanna 2013), or hydrothermal liquefaction (Somers and Quinn 2019). Microalgae may also be used as fish feed (Maiolo et al. 2020) or food (Schade et al. 2020).

## 2.3 Assessing Paris compatibility of CCU technologies

### 2.3.1 Paris compatibility criteria

In this paper, a CCU technology is classified as 'Paris-compatible' when its deployment is expected to be in line with reaching the Paris Agreement's LTTG and corresponding 1.5°C-mitigation pathway. This definition results in two questions to assess Paris compatibility: 1) is the CCU technology ready on time?; and 2) does the CCU technology sufficiently reduce CO<sub>2</sub> emissions? These questions result in the Paris compatibility criteria for 2030 and 2050 outlined in this section (Figure 2.2). The following two sections present a review of the literature underlying the criteria and explain how these criteria can be applied to CCU technologies.



**Figure 2.2. Paris compatibility criteria for CCU technologies.** Paris-compatible mitigation pathway using scenario data from Huppmann et al. (2018) as the mean of no and low-overshoot 1.5°C scenarios from which Paris compatibility criteria for CCU technologies for 2030 and 2050 were derived. TRL is the 2020 technology readiness level,  $E_{CCU}$  are the emissions associated with the CCU product (in kgCO<sub>2</sub>-eq./kg CO<sub>2</sub> utilised), and  $E_{substitute}$  are the emissions associated with the substituted product (in kgCO<sub>2</sub>-eq./kg CO<sub>2</sub> utilised).

#### Maturity criteria

Technological maturing, i.e., the process leading from research and development (R&D) to demonstration trials, early market formation and widespread diffusion, can span several decades and is ridden with financial risks and technological uncertainties (Grubler 2012). We include technological maturity as a criterion for

Paris compatibility because for a technology to be able to contribute to emission reductions in 2030 or 2050, it must be sufficiently developed to be ready for widespread diffusion.

The technological readiness level (TRL) of a technology is an indication of the maturity of a technology on a scale that summarises detailed information on technological maturity into a single value (Mankins 1995). The scale has nine levels, spanning from basic concept (TRL 1) to successful, real-life operation (TRL 9). TRL 6 presents a turning point in the technological development by requiring an operational system at a relevant scale. The time it takes to progress through the TRL scale differs per technology and is context-dependent, but for CCU technologies 10-15 years is typically assumed to be needed to progress from lab scale to full-scale implementation (Naims 2016; Chauvy et al. 2019; Chauvy and De Weireld 2020). In line with Chauvy et al. (2019), we therefore assume that for a CCU technology to be ready in 2030, it must be in at least TRL 6 in 2020. Given that it is possible to progress from the R&D phase (TRL 1-3) to real-life operation (TRL 9) in 20-30 years (Kramer and Haigh 2009; Chauvy et al. 2019), the TRL is no impediment for a CCU technology for 2050. This does not mean that it can be assumed that a technology in the R&D phase in 2020 will be ready in 2050; only that a technology cannot be excluded for Paris compatibility in 2050 based on its 2020 TRL.

### ***Emissions criteria***

1.5°C-pathways are characterised by two key numbers: in 2030, global net CO<sub>2</sub> emissions are halved compared to 2020 emissions, and in 2050, net CO<sub>2</sub> emissions are zero (Rogelj et al. 2018). For CO<sub>2</sub> emission reductions to be in line with 1.5°C-pathways, the CCU technology must halve CO<sub>2</sub> emissions associated with the production and use of the CCU product by 2030, as compared to the emissions associated with the current conventional, substituted product. In 2050, CO<sub>2</sub> emissions associated with the CCU product must be zero to be Paris-compatible. Although emission data are often provided for a basket of GHGs instead of CO<sub>2</sub> only, CCU is focussed on reducing CO<sub>2</sub> emissions, which is by far the most prominent contributor to total GHG emissions, directly as well as indirectly (e.g., van der Giesen et al. 2014; Cooney et al. 2015).

### 2.3.2 Maturity of CCU technologies

The TRL scale finds its origin at the American National Aeronautics and Space Administration in the 1970s and became more widely known once the US Department of Defense started using them to improve its technology R&D outcomes (Mankins 2009). It was first comprehensively described by Mankins (1995), and has since received recognition in policy-making, industry and academia (Buchner et al. 2019). For example, in 2010 the European Commission advised projects that received EU-funding to use TRLs for identification of technological maturity (Héder 2017). Its use has also been recommended specifically for CCU techno-economic assessments (Global CO<sub>2</sub> Initiative 2020; Zimmermann et al. 2020).

The European Commission (2014) defines TRLs in a generalised way, to allow for comparability of technologies in different fields, including of energy and climate technologies (Table 2.1). A downside of a generalised scale is the general description for each level, including criteria that may not be applicable, or not specific enough to achieve unambiguous TRL ratings (Buchner et al. 2019; Sick et al. 2020). Adaptations to the generalised scale were developed to better rate technologies in a certain field, for example in the chemical industry (Buchner et al. 2019), and even for CCU technologies (Chauvy et al. 2019) (Table 2.1). Despite the differences between descriptions of TRLs, the commonality is that TRL 6 represents a break from the foregoing TRLs by requiring an operational system at a relevant scale (pilot plant) to have been developed.

To assess whether a CCU technology fulfils the Paris compatibility maturity criteria, its TRL in 2020 must be determined from direct TRL specifications or by applying the milestones as defined by Buchner et al. (2019) (Table 2.1) to descriptions of the technology's state of development. Table 2.1 includes examples of CCU technologies for each TRL and Supplementary Table 2.2 provides a comprehensive overview of TRL ratings of CCU technologies. Some studies report ranges instead of a single TRL for a CCU technology. This can be done for three reasons: 1) The technology is a 'composed' technology, made up of multiple processes, each with its individual TRL. Hence, the range of these individual TRLs is reported; 2) More than one technology is in development to produce a CO<sub>2</sub>-based product and the full range

is reported for this product instead of for each production process separately; or 3) There is an uncertainty due to a lack of data. In this paper, we deal with these ranges in TRL in the following way: In case of a composed technology (reason 1), the lowest TRL is counted as the overall TRL because it is the weakest link in the chain in the process to commercialisation (Buchner et al. 2019); We specify a TRL for each CCU production process, rather than only per CCU product (reason 2); Uncertainty in the TRL is resolved by comparing reported process descriptions (e.g. proof of concept, bench-scale process, pilot plant) to the descriptions of Buchner et al. (2019) to find the best match (reason 3), as explained in Supplementary Table 2.2

**Table 2.1. Technology readiness levels for CCU technologies.** Descriptions of technology readiness levels (TRL) and examples of CCU technologies for each TRL. See Supplementary Table 2.2 for an overview of TRLs for all studied CCU technologies.

TRL	Description by European Commission (2014)	Description by Chauvy et al. (2019) for CCU evaluation	Description by Chauvy et al. (2019) for the chemical industry	CCU examples
1	Basic principles observed	Published research that identifies the principles that underlie the technology.	Idea. Opportunities identified, basic research translated into possible applications.	Methanol via photocatalytic conversion (Trudewind et al. 2014; Chauvy et al. 2019).
2	Technology concept formulated	Publications or other references that outline the application being considered, and that provide analysis to support the concept. The step up from TRL 1 to TRL 2 moves the idea from pure to applied research. A major part of the work is analytical or paper studies. Experimental work is designed to corroborate the basic scientific observations made during TRL 1 work.	Concept. Technology concept and/or application formulated, patent research conducted.	Ethanol (Thonemann and Pizzol 2019) and methanol (Rumayor et al. 2020) via electrochemical reduction (Roh et al. 2020).
3	Experimental proof of concept	Active research and development (R&D) has been initiated. At TRL 3, the work has moved beyond the publication phase to experimental work.	Proof of concept. Applied laboratory research started, functional principle/ reaction (mechanism) proven, predicted reaction observed (qualitatively).	Ethylene via electrochemical reduction (DECHEMA 2017), dimethyl ether via syngas (C2FUEL; Thonemann and Pizzol 2019).
4	Technology validated in lab	TRL 4–6 represents the bridge from scientific research to engineering. TRL 4 is the first step in determining whether the individual components will work together as a system.	Preliminary process development. Concept validated in laboratory environment, scale-up preparation started, short-cut process models found.	Precipitated calcium carbonate and precipitated magnesium carbonate (Martíla et al. 2014; Galvez-Martos et al. 2016; CCMUK 2020; Chauvy and De Weirld 2020).
5	Technology validated in relevant environment	The basic technological components are integrated so that the system configuration is similar to the final application in almost all respects.	Detailed process development. Process models found, property data analysed, simulation of process and pilot plant using bench scale information.	Formic acid via electrochemical reduction in aqueous environment (Thonemann and Schulte 2019); sodium bicarbonate using flue gas directly (Lee et al. 2018).

**Table 2.1. Technology readiness levels for CCU technologies.** Descriptions of technology readiness levels (TRL) and examples of CCU technologies for each TRL. See Supplementary Table 2.2 for an overview of TRLs for all studied CCU technologies. (continued)

TRL	Description by European Commission (2014)	Description by Chauvy et al. (2019) for CCU evaluation	Description by Buchner et al. (2019) for the chemical industry	CCU examples
6	Technology demonstrated in relevant environment	This represents a major step up in a technology's demonstrated readiness. TRL 6 begins true engineering development of the technology as an operational system.	Pilot trials. Pilot plant constructed and operated with low rate production, products approved in final application, detailed process models found.	Fischer-Tropsch fuels (Detz 2019; Chauvy and De Weirerd 2020) urea from steel gases (de Kleijne et al. 2020; STEPWise).
7	System prototype demonstration in operational environment	TRL 7 is a significant step beyond TRL 6, requiring an actual system prototype.	Demonstration and full-scale engineering. Parameter and performance of pilot plant optimised, (optional) demo plant constructed and operating, equipment speciation including components that are type conferrable to full-scale production.	CO and syngas via the reverse water gas shift reaction (Detz 2019; Thorenmann and Pizzol 2019); methane and methanol via hydrogenation of CO <sub>2</sub> (Chauvy et al. 2019; STORE&GO 2019; Recycling Portal 2021).
8	System complete and qualified	This TRL represents the end of true system development. The technology has been proven to work in its final form and under expected conditions.	Commissioning. Products and processes integrated in organisational structure (hardware and software), full-scale plant constructed.	Polyols (Chauvy et al. 2019; Econic; Covestro), construction materials from carbonated steel slag (CE Delft 2018; Hills et al. 2020; VITO).
9	Actual system proven in operational environment	The technology is in its final form and operates under the full range of operating mission conditions.	Production. Full-scale plant audited (site acceptance test), turn-key plant, production operated over full range of expected conditions in industrial scale and environment, performance guarantee enforceable.	CO <sub>2</sub> -enrichment in agricultural greenhouses (Patrício et al. 2017a); CO <sub>2</sub> -enhanced oil recovery (IEA 2020; Global CCS Institute 2021).

### 2.3.3 Emissions of CCU technologies

To assess Paris compatibility of CCU technologies, we determined the GHG emission intensity of CCU products ( $\text{kgCO}_2\text{-eq./kg CO}_2\text{ utilised}$ ) and of substituted conventional products (assuming 1:1 replacement) based on a literature search of life cycle assessment (LCA) studies on CCU. In this way, 1041 studies were identified (see Experimental Procedure section). After screening these papers for relevance and excluding reviews and meta-analyses without original data, 106 studies remained (see Supplementary Table 2.3). We then selected the most recent study, with the most complete life cycle inventory, for each available combination of CCU technology and type of  $\text{CO}_2$  source (fossil, biogenic or atmospheric). Not all combinations could be found in the literature. This resulted in 30 studies (1 on direct use, 6 on EHR, 8 on mineral carbonates and construction materials, and 15 on fuels and chemicals), together covering 44 unique CCU technologies and resulting in 74 CCU technology- $\text{CO}_2$  source combinations (see Supplementary Table 2.3, which also provides an overview of the specific  $\text{CO}_2$  sources and  $\text{CO}_2$  capture processes covered). These studies were harmonised with regards to: i) functional unit, ii) system boundaries, iii) electricity mix, iv) hydrogen production, v) dealing with multifunctionality, and vi) accounting of temporary carbon storage, as detailed below. For the substituted products, we followed the choices made in the original papers, assuming the CCU product either replaces a product with identical molecular structure, or, if this does not exist, a product with the same characteristics and function (Zimmermann et al. 2020).

#### *Functional unit*

The functional unit of CCU products can be end product-based (e.g., kg or MJ product) or input-based (e.g., kg  $\text{CO}_2$  utilised). Most CCU LCA studies use an end product-based functional unit, which allows estimating absolute emission reductions when switching to a CCU product. However, to be able to compare different CCU technologies, the functional unit should be the same, and thus input-based (Hanssen and Huijbregts 2019; Thonemann and Pizzol 2019). We therefore use 1 kg of  $\text{CO}_2$  utilised as functional unit in this paper. It is possible to go from an end product-based functional unit to the functional unit of 1 kg of  $\text{CO}_2$  utilised by determining the product's GHG intensity in  $\text{kgCO}_2\text{-eq./kg product}$  and multiplying it by  $\text{kg product/kg CO}_2$  utilised.

### ***System boundaries***

Most LCA studies quantify 'cradle-to-factory-gate' emissions, assuming that emissions beyond the gate are the same for CCU product and substitute. In our analysis, we determine the absolute CO<sub>2</sub> emissions of a CCU product and the percentual CO<sub>2</sub> emission reduction of a CCU product relative to its substitute. This requires a wider 'cradle-to-grave' approach that includes end-of-life emissions for both products. In practice, however, data on transport and use-phase emissions often lack in the reviewed LCA studies, except for the re-emission of CO<sub>2</sub> upon combustion or dissolution. So where cradle-to-grave data were not available, we used cradle-to-factory-gate emissions and added end-of-life emissions, as proposed by Fernández-Dacosta et al. (2019), assuming use-phase emissions were negligible. Combustion and dissolution were then included in the end-of-life emissions. We did not harmonise production infrastructure and transport emissions and follow the choices in the respective LCAs, because these emissions contribute negligibly to the total GHG intensity (Geisler et al. 2004; Hischier et al. 2005; Hoppe et al. 2016). For the substituted products emissions were also determined based on a cradle-to-grave basis.

### ***Electricity mix***

To improve inter-comparability of CCU products and take the expected decarbonisation of the electricity sector into account (van der Hulst et al. 2020), we harmonised the GHG intensity of electricity used in all foreground processes of the studies considered. This includes electricity use in the capture and conversion process, and hydrogen production via electrolysis. The harmonised GHG intensity of electricity was set to be in line with emissions pathways limiting global warming to 1.5°C (Rogelj et al. 2018): 0.17 kgCO<sub>2</sub>/kWh for 2030 and 0 kgCO<sub>2</sub>/kWh for 2050. For the zero-emissions electricity it is assumed that the (limited) emissions of renewable electricity production are compensated by carbon dioxide removal (e.g., from bioenergy with carbon dioxide capture and storage).

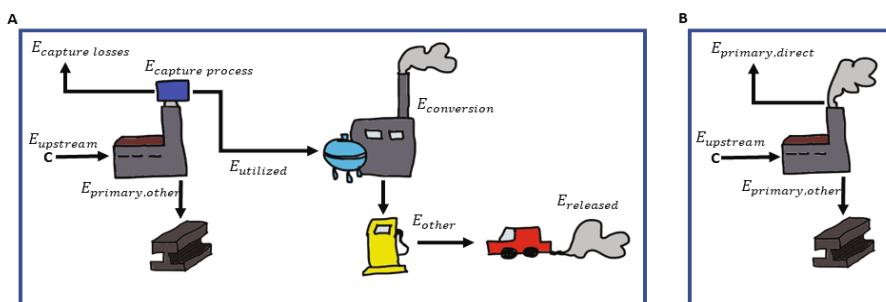
### ***Hydrogen***

Hydrogen is an important feedstock in the production of fuels and chemicals from CO<sub>2</sub> and is produced using fossil fuels with or without CCS, or water electrolysis. Since we assume hydrogen production to decarbonise in Paris-compatible pathways, we harmonise all studies towards electrolysis-based hydrogen following

their reported electricity requirements, unless  $H_2$  is co-produced in the conversion process. For an electricity requirement of 52 kWh/kg  $H_2$  and using aforementioned electricity carbon intensities for 2030 and 2050, electrolysis-based hydrogen results in 6.85 kgCO<sub>2</sub>-eq./kg  $H_2$  in 2030 and 0 kgCO<sub>2</sub>-eq./kg  $H_2$  in 2050. We use the substitution approach to deal with the co-produced O<sub>2</sub>, see the Experimental Procedure section.

### Multifunctionality

The CCU production chain is typically inherently multifunctional: an industrial facility or power plant produces a primary product (e.g., steel or electricity) as well as CO<sub>2</sub>, which is used as feedstock in the secondary CCU process. GHG emissions of this system must be divided between the primary product and the secondary CCU product. DAC-based CCU systems are not multifunctional because there is no primary product. We apply a system expansion via substitution approach to solve this multifunctionality problem following the LCA standard ISO 14044 and LCA guidelines for CCU (Global CO<sub>2</sub> Initiative 2020) (Figure 2.3).



**Figure 2.3. Determining the GHG intensity of CCU products through the substitution approach.**  
In the system expansion via substitution approach, the GHG intensity of the CCU product  $E_{CCU}$  is determined by the difference between the emissions of (A) the multifunctional system and (B) the marginal production system of the primary product.

System expansion via substitution assumes that a primary production plant *with* CCU directly substitutes an identical plant *without* CCU, to ultimately determine the emissions that can be associated with CCU itself. It is formalised as shown in equation 2.1 (for a full derivation see the Experimental Procedure section):

$$E_{CCU} = -E_{utilised} + E_{capture\ process} + E_{conversion} + E_{other} + E_{released} \quad (2.1)$$

Where E stands for GHG emissions (in kgCO<sub>2</sub>-eq./kg CO<sub>2</sub> utilised), specifically:

$E_{CCU}$  GHG emissions of the CCU product.

$E_{utilised}$  CO<sub>2</sub> utilised in production of the CCU product.

$E_{capture\ process}$  GHG emissions associated with capturing CO<sub>2</sub> and separating the CO<sub>2</sub> from the bulk gas stream.

$E_{conversion}$  GHG emissions of the conversion process of the CO<sub>2</sub> into the end-product.

$E_{other}$  Other (remaining) GHG emissions associated with the CCU product, such as GHG emissions at end-of-life other than the release of utilised CO<sub>2</sub>, for example from fossil feedstock added in the conversion process.

$E_{released}$  GHG emissions of utilised CO<sub>2</sub> to the atmosphere at the end-of-life of the CCU product.

In most cases  $E_{utilised}$  and  $E_{released}$  cancel each other out. However, is less than  $E_{utilised}$  when some of the utilised CO<sub>2</sub> is lost in the conversion process (i.e., included in  $E_{conversion}$ ); or when the CO<sub>2</sub> is stored permanently, then  $E_{released}$  is zero. A negative value for  $E_{CCU}$  can occur when CO<sub>2</sub> is stored permanently and the amount of CO<sub>2</sub> utilised is greater than the combined emissions of the CCU production, use and end-of-life processes.

The interpretation of a negative value for depends on the source of CO<sub>2</sub>. In case of CO<sub>2</sub> with a fossil origin, a negative means that the total GHG emissions of the system with CCU are reduced in comparison to the system without CCU, and that this emission reduction can be attributed to CCU. It does not mean that CCU is CDR, because CO<sub>2</sub> is not removed from atmosphere (Tanzer and Ramírez 2019; Müller et al. 2020a). In case of CO<sub>2</sub> that is removed from the atmosphere, either

directly (through DAC) or indirectly (through capture of CO<sub>2</sub> with biogenic origin), a negative value for  $E_{CCU}$  does mean CDR takes place. For biogenic CO<sub>2</sub>, CDR would also require sustainable sourcing of the biomass used.

The substitution approach is based on the assumption that a point source without capture is 1:1 substituted by the same plant with capture. In the near future (i.e., 2030) this assumption holds, as many plants without capture can still be substituted. In the long run, however, the reference system is expected to transform towards a zero-emissions economy. Therefore, in a Paris-compatible 2050 system, we can no longer assume that there are *unabated* fossil point sources to substitute and associated emissions to avoid. Since emissions are no longer unabated, considering CCU as an option to abate fossil CO<sub>2</sub> emissions, means that the responsibility for not emitting shifts to CCU. In terms of calculating the GHG intensity of CCU for a fossil CO<sub>2</sub> source in 2050, this means all emissions from the stack onwards are assigned to the CCU product (equation 2.2).

$$\begin{aligned} E_{CCU(\text{fossil in 2050})} = & E_{\text{capture losses}} + E_{\text{capture process}} \\ & + E_{\text{conversion}} + E_{\text{other}} + E_{\text{released}} \end{aligned} \quad (2.2)$$

Where E stands for GHG emissions (in kgCO<sub>2</sub>-eq./kg CO<sub>2</sub> utilised), specifically:

$E_{CCU(\text{fossil in 2050})}$       GHG emissions of the CCU product (for 2050 in a Paris-compatible scenario where substitution of unabated fossil CO<sub>2</sub> point sources cannot be assumed).

$E_{\text{capture losses}}$       CO<sub>2</sub> not captured in the capture process, assuming a capture rate of 95% (Bui et al. 2018; IEA 2020; Wevers et al. 2020).

In the results, we report  $E_{CCU}$ , but when this applies to fossil CO<sub>2</sub> sources in 2050, equation 2.2 is used for the calculation. For CCU processes based on biogenic or atmospheric CO<sub>2</sub> in 2050, equation 2.1 still applies, because availability of these sources in 2050 is in line with a Paris-compatible scenario.

### **Benefits of temporary storage of CO<sub>2</sub>**

The importance of a product's lifetime for CCU's contribution to climate change mitigation has been stressed in literature (Mazotti et al. 2005; de Coninck et al. 2018). Temporary storage of CO<sub>2</sub>, by lowering atmospheric CO<sub>2</sub> concentrations for the duration of the product's lifetime, decreases radiative forcing over this period of time (Brandão et al. 2013). We calculated the GHG intensity of CCU products without (base case) and with the benefit of temporary carbon storage (sensitivity analysis), using a time horizon of 100 years. Global warming potentials are used for this calculation, adjusted for temporary carbon storage (GWP<sub>storage</sub>). They describe the contribution to radiative forcing of delayed CO<sub>2</sub> emissions compared to direct CO<sub>2</sub> emissions over 100 years. We derived the GWP<sub>storage</sub>-100 factors using the method described by Guest et al. (2013), which is based on Clift and Brandao (2008), combined with the more recent atmospheric CO<sub>2</sub> decay curves described by Joos et al. (2013). The factors depend on the lifetime of the CCU product, as shown in Table 2.2. See the Experimental Procedure section for an overview of CCU lifetimes. To include the storage effect,  $E_{released}$  in equation 2.1 and equation 2.2 is multiplied by GWP<sub>storage</sub>-100. In this way, only captured CO<sub>2</sub> is affected by this temporary storage benefit, not fossil carbon added in some CCU routes as additional feedstock.

**Table 2.2. Global warming potential factors (GWP<sub>storage</sub>-100) for temporary storage of CO<sub>2</sub> in CCU products using a 100-year time horizon.**

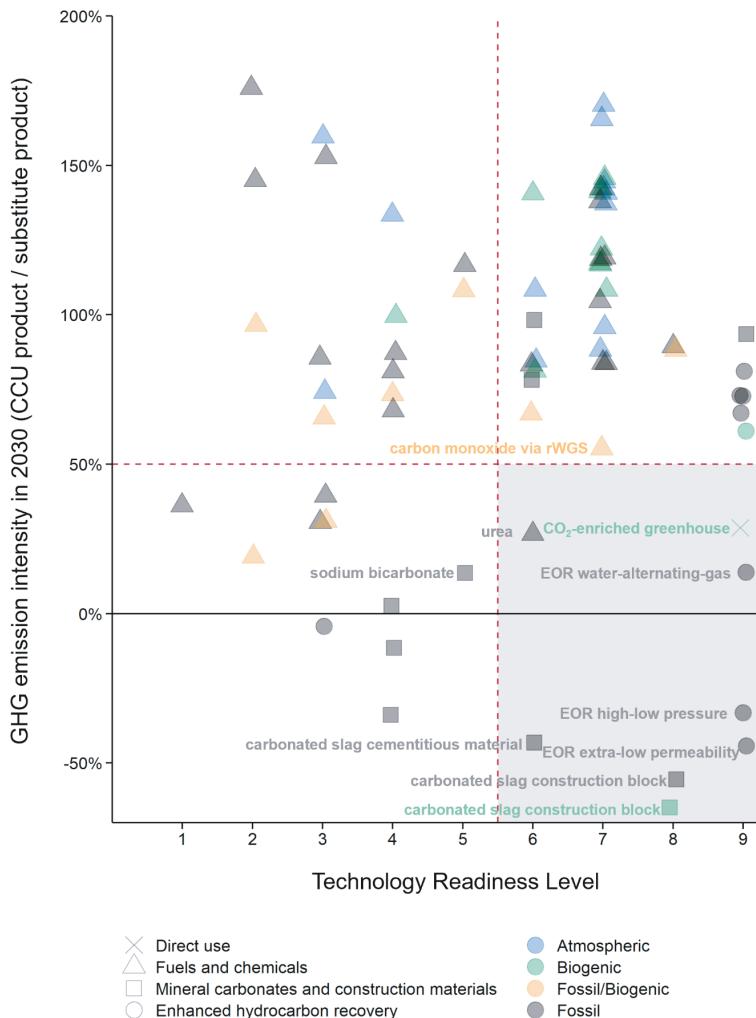
Lifetime	0–6 months	0.5–1 year	1 year	5 years	10 years	25 years	50 years	100+ years
GWP <sub>storage</sub> -100	1	0.99	0.98	0.92	0.85	0.67	0.42	0

## 2.4 Fulfilment of Paris compatibility criteria

### 2.4.1 Paris compatibility in 2030

Figure 2.4 shows the relative GHG intensity of CCU technologies in 2030 plotted against their 2020 TRL. As explained above, a negative GHG intensity ratio means that an emission reduction takes place for fossil CO<sub>2</sub> and CDR when the source of CO<sub>2</sub> is biogenic or atmospheric. CCU technologies fulfilling the 2030 emissions criterion can be characterised by one or more of the following attributes: (1)

Preventing (high) capture emissions; (2) Preventing (high) conversion emissions; (3) Preventing re-emission of CO<sub>2</sub>; and (4) Replacing an emission-intensive process.



**Figure 2.4. Paris compatibility of CCU technologies in 2030.** Technological maturity of CCU technologies in terms of their 2020 technology readiness level; and GHG emission intensity ratio of the CCU product in 2030 compared to its substitute. The four shapes represent the four CCU categories and the colours are used to differentiate between the different types of CO<sub>2</sub> sources. For a CCU technology to be Paris-compatible in 2030, it must currently have a TRL of 6 or higher and reduce emissions by at least 50% compared to its substitute: Technologies in the lower right grey area fulfil these Paris compatibility criteria. Full dataset including outliers not in this figure, and source data for this figure is provided in Data S1.

Paris-compatible CCU technologies in 2030 are: CO<sub>2</sub>-enrichment in the horticulture industry (Oreggioni et al. 2019) with CO<sub>2</sub> from a co-located biogas to biomethane upgrading unit, which is considered a by-product (i.e., is characterised by attribute 1) and can be used directly (attribute 2). In the carbonation process of steel slag to produce construction blocks, flue gas is used directly (Pan et al. 2016), omitting the capture step (1). CO<sub>2</sub> is stored permanently through an exothermic carbonation reaction (Di Maria et al. 2020) (2 and 3). In EOR, CO<sub>2</sub> is used directly (2), and can be Paris-compatible for several gas separation methods (Lacy et al. 2015; Liu et al. 2020c; Sminchak et al. 2020) as long as no more than 2 barrels of oil (bbl) are recovered per tonne CO<sub>2</sub> injected. This allows for a favourable balance of the CO<sub>2</sub> released upon the combustion of the recovered oil and the CO<sub>2</sub> stored permanently (3). Urea production from basic oxygen furnace gas (BOFG) uses waste heat from the steel plant for the capture and conversion processes (1 and 2) and replaces electricity production from BOFG (de Kleijne et al. 2020). Hence, the quantification of the GHG intensity of urea includes the replacement of electricity produced from BOFG by the 1.5°C-compatible electricity mix (4). Lastly, close to halving emissions is CO via rWGS with relatively low conversion emissions (2) (Thonemann and Pizzol 2019).

Low TRL CCU technologies which *only* fulfil the emissions criterion can be characterised by the same four attributes: The sodium bicarbonate process uses flue gas directly, avoiding the capture step (attribute 1) (Lee et al. 2018). Indirect mineral carbonation reactions producing PMC or PPC may have low emissions for a high efficiency of the alkaline absorption process, depending on process development. They store CO<sub>2</sub> permanently (3) (Galvez-Martos et al. 2016). Formic acid via hydrogenation reduces emissions sufficiently compared to the emission-intensive substitute (4) (Thonemann and Pizzol 2019). Electrochemical production of formic acid via supercritical CO<sub>2</sub> (Thonemann and Schulte 2019) and methane and methanol via photocatalytic conversion (Trudewind et al. 2014) have low conversion emissions (2) because they do not require electricity-intensive hydrogen.

High TRL CCU technologies *not* fulfilling the emissions criterion can also be characterised by the aforementioned attributes. However, here these are the reasons for *not* fulfilling the emissions criterion: CO<sub>2</sub>-curing of concrete (Monkman

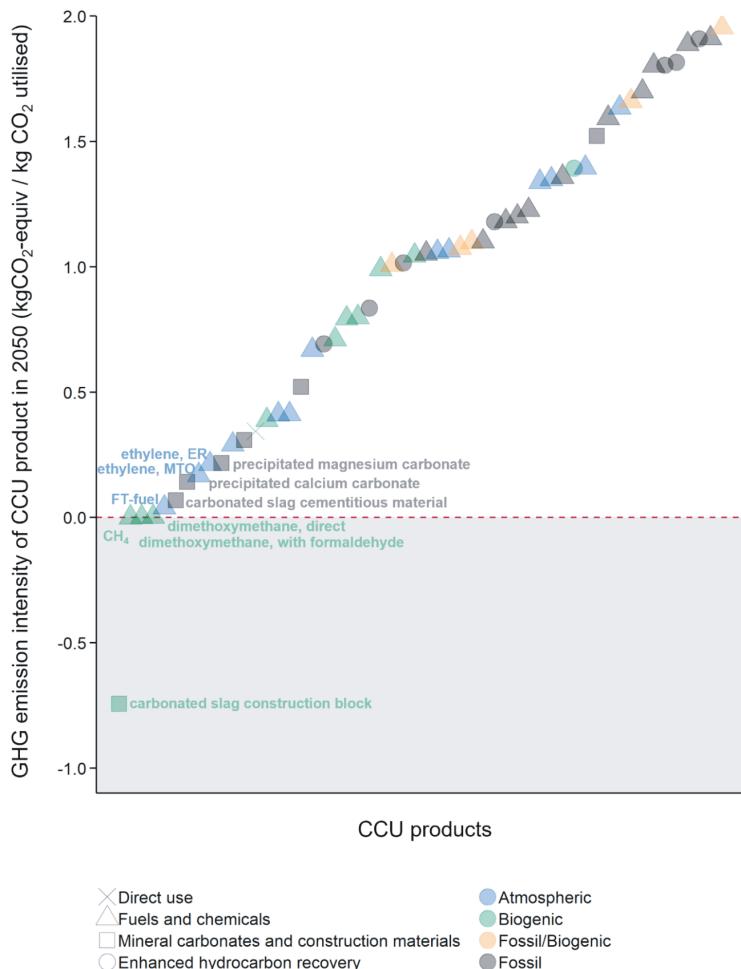
and MacDonald 2017; Huang et al. 2019) does not sufficiently reduce emissions due to the emissions associated with cement production (2). Fuels and chemicals based on thermochemical conversion typically have high conversion emissions due to electricity-intensive hydrogen production (2). Finally, EOR with a recovery ratio over 2 barrels of oil per tonne of CO<sub>2</sub> injected does not store a sufficient volume of CO<sub>2</sub> compared to combustion emissions (Hussain et al. 2013; Azzolina et al. 2016) (3).

### ***Benefits of temporary CO<sub>2</sub> storage***

Some products store CO<sub>2</sub> for several decades, i.e., polyethylene and polypropylene. Adding the temporary storage benefit based on a lifetime of 50 years leads to a reduced GHG intensity of roughly 25%. This, however, is not enough to lead to a difference in Paris compatibility. For long-lived polyols the reduction is only a few per cent because a large share of the embodied carbon is from fossil feedstock, to which the storage factor does not apply. The lifetime of the other chemicals and fuels is assumed to be six months, where the GWP<sub>storage</sub>-100 factor of 0.99 does not affect Paris compatibility. For EHR, construction materials and most mineral carbonates, permanent storage was already assumed in the base case.

#### **2.4.2 Paris compatibility in 2050**

Figure 2.5 shows the GHG intensity of CCU products in 2050 ( $E_{CCU}$ , in kgCO<sub>2</sub>-eq./kg CO<sub>2</sub> utilised) (see equations 1 and 2). In 2050 also low TRL technologies may be Paris-compatible. The characteristics of CCU technologies fulfilling the 2050 emissions criterion are simplified to: (1) Preventing re-emission of CO<sub>2</sub>; (2) Only using zero-emissions energy; and (3) Utilising CO<sub>2</sub> recently removed from the atmosphere (biogenic/atmospheric).



**Figure 2.5. Paris compatibility of CCU technologies in 2050.** GHG intensity of CCU products in 2050 (kgCO<sub>2</sub>-eq./kg CO<sub>2</sub> utilised). The four shapes represent the four CCU categories and the colours are used to differentiate between the different types of CO<sub>2</sub> sources. The TRL is not indicated as the 2020 technological maturity does not preclude technologies from fulfilling the Paris compatibility criteria by 2050: Technologies in the lower grey area fulfil these Paris compatibility criteria. Full dataset including outliers not in this figure, and source data for this figure is provided in Data S1.

Construction blocks from carbonation of steel slag using a purified stream of biogenic CO<sub>2</sub> (Di Maria et al. 2020) are Paris-compatible, resulting in CDR because the CO<sub>2</sub> is stored permanently (i.e., criterion 1 and 3 are fulfilled). In addition, steel slag carbonation using fossil flue gas directly to produce cementitious material (Pan et al. 2016) achieves close to zero emissions because no capture process is

required, the carbonation process is exothermic, and the CO<sub>2</sub> is stored permanently (fulfilling criterion 1 and 2).

Since CO<sub>2</sub> is not stored permanently in fuels or chemicals, these products can only be strictly Paris-compatible when the CO<sub>2</sub> is of biogenic or atmospheric origin, and zero emissions are associated with the capture and conversion processes (fulfilling criterion 2 and 3). This situation is approached when heat integration is applied, electric heat is used, or simply only electricity is required. Processes approaching zero emissions are: methane production from H<sub>2</sub> and biogenic CO<sub>2</sub> (Bargiacchi et al. 2020); DMM via the condensation with formaldehyde or via direct synthesis, both for a biogenic source (Deutz et al. 2018); and Fischer-Tropsch fuels from atmospheric CO<sub>2</sub>, based on a fully electric process for capture and conversion (Liu et al. 2020a). The assumption of zero-emission electricity is crucial, since the GHG intensity of these options using the 2030-electricity mix is between 1.21 and 1.39 kgCO<sub>2</sub>-eq./kg CO<sub>2</sub> utilised (see Supplementary Figure 2.1 for the absolute GHG intensity of CCU products in 2030).

#### ***Benefits of temporary CO<sub>2</sub> storage***

The inclusion of the benefit of temporary storage of 6 months is enough to obtain a negative absolute GHG intensity for DMM via the condensation with formaldehyde or via direct synthesis, both for a biogenic source (Deutz et al. 2018).

## 2.5 Discussion and research outlook

### ***Maturity***

We considered technological maturity in terms of TRLs of individual technologies, as data is available for individual technologies. Others have suggested 'System Readiness Levels' (Sauser et al. 2006; Knaggs et al. 2015) to reflect the technology's need to embed in a certain system. Due to this embedding, the assessment of technological readiness at a systemic level could lead to lower levels of maturity. This means that Paris compatibility of CCU could be overestimated. On the other hand, concerted, transdisciplinary action on innovation could speed up technological maturing of promising technologies. For example, this could apply to sodium bicarbonate which fulfils the emission reduction criterion but is in TRL

5. As a next step, 'Technological Innovation System' analyses (Hekkert et al. 2007; Bergek et al. 2008) can be performed to provide information on what actors can do to advance such technologies.

### ***Emissions***

Our harmonisation allowed for a comparison of CCU technologies, yet some uncertainty in future emissions remains because: i) LCAs of low TRL technologies may be based on idealised modelling data, underestimating the environmental impacts due to an information bias (Roh et al. 2020); ii) the environmental impact of technologies is expected to decrease as they mature due to energy efficiency improvements and/or the development of catalysts for more efficient conversion (Thonemann and Schulte 2019; van der Hulst et al. 2020); iii) some CCU processes could be adapted such that more CO<sub>2</sub> is utilised per product produced, for example for EOR (Núñez-López and Moskal 2019); and iv) if product lifetimes could be extended, they could, based on the temporary storage benefit, decrease their contribution to climate change. Future LCA studies could focus on CCU technologies that are developing quickly and for which no LCAs are available, for which only LCAs based on low TRL data are available, or on how CCU processes can be adapted to reduce overall emissions.

Even if the emissions criteria are not fulfilled, it is possible that there are no future alternative technological routes that provide the same product at a lower GHG intensity. Although a CCU technology is not strictly Paris-compatible in such cases, CCU could still play a role in the overall energy system transition. In addition, if other environmental impacts besides climate are included in the equation, CCU options may in certain cases be viewed more positively than remaining alternatives to our fossil fuel-based production system. For instance, CCU products do not have the land-use impacts of crop-based biomaterials (including biodiversity) (Hanssen et al. 2021). Further research could include alternative routes in the assessment and define emission reduction criteria per product group. Furthermore, the assessment of emission reduction could be extended beyond the technology level to the sector or even the system level. Determining the global climate change mitigation potential of CCU at scale requires that interactions with other mitigation options as well as broader changes in the economy are considered, which may be achieved using integrated assessment models. Further research could focus on including

CCU in these models and determining how much each CCU technology could contribute at scale to CO<sub>2</sub> emission reductions within and across different sectors.

### ***Multifunctionality***

The multifunctionality problem is one that needs to be solved carefully and transparently. Otherwise, it is possible to end up with statements that cannot be simultaneously true, such as: CCU can make traditionally CO<sub>2</sub>-emitting industries (e.g., fossil fuel electricity generation, chemicals, cement) carbon-neutral (Lau et al. 2016; Kätelhön et al. 2019) while producing carbon-neutral CCU products (Patrício et al. 2017b; Ioannou et al. 2020). The reality is that the eventual emission of CO<sub>2</sub> must be accounted for by at least one of the processes. And, as shown above, when moving to a future without unabated fossil emissions, this emission has to be allocated to CCU. The right year of this allocation shift is however unclear and may depend on the GHG reduction targets of a specific country, sector or company. While we consider the year 2050, based on Paris-compatible emission pathways in the IPCC at a global level, other countries or firms may have reasons to select a different point in time where fossil points sources can no longer go unabated.

### ***Decarbonisation of electricity***

The carbon intensity of electricity of 0.17 kgCO<sub>2</sub>/kWh that we used for 2030 is higher than the carbon intensity in many of the CCU LCA studies, for example when based on only renewables. This means that in these cases, we found a higher 2030 GHG intensity of CCU products compared to the original studies. Still, our 2030 carbon intensity of electricity may be an underestimation because it was based on emission pathways that only include use-phase emissions, not upstream emissions. Our 2030 results, however, are not sensitive to this increase in carbon intensity: doubling the carbon intensity (0.34 kgCO<sub>2</sub>/kWh) did not affect Paris compatibility of technologies in 2030. However, storing renewable electricity in a fuel or chemical results in a loss of primary energy (Sutter et al. 2019) and has a low energy return on energy invested (Mac Dowell et al. 2017). Hence, the use of low-carbon electricity for other mitigation options than CCU has been shown to achieve higher emission reductions per kWh, e.g. for e-mobility and heat pumps (Kätelhön et al. 2019) or DACCS (Daggash et al. 2018). Alternatively, CCU technologies have been proposed to use excess renewable electricity and aid the energy transition by balancing the peaks of renewable power production (Wohland

et al. 2018; Mikulčić et al. 2019). This suggestion is sobered by its limited economic attractiveness as a result of the low capacity factors (Wevers et al. 2020), but further research on the systemic effects of the electrification of the CCU process would be of value.

### ***Lock-in***

The only CCU technologies that are 2030 *and* 2050 Paris-compatible are construction materials based on carbonation of steel slag, either using fossil flue gas directly or using CO<sub>2</sub> captured from a biogenic source. Arguably, implementing CCU options that only reduce emissions sufficiently for 2030 but not for 2050 could lead to a carbon lock-in, which would occur when large investments are made in 2030, complicating phase-out and discouraging necessary transitions later on (Unruh 2000; Janipour et al. 2020). Hence, assuming a CCU production plant built in 2030 is still operational in 2050 would mean that the 2050 emissions criterion should be added to assess 2030 Paris compatibility. In that case, with the exception of some construction materials, none of the 2030-compatible options fulfil the 2050 emissions criterion. A lock-in in these CCU processes could be prevented by implementing an exit strategy for industries relying on CCU processes that are only 2030-compatible. CCU technologies that are currently used or considered and that are not 2030-compatible could lead to a lock-in within years. We recommend that these are critically evaluated on their potential for a short-term exit strategy.

### ***Sources of CO<sub>2</sub> and capture processes***

We found that energy requirements for CO<sub>2</sub> capture and their associated emissions differ between specific CO<sub>2</sub> sources and capture processes, in line with Müller et al. (2020a) and von der Assen et al. (2016). Whether or not a CCU process is Paris-compatible can thus depend on the capture process itself (e.g., a DAC process based on natural gas (Bargiacchi et al. 2020) rather than renewable electricity (Liu et al. 2020a) may render the entire CCU product non-Paris-compatible). Our research has focussed on available CO<sub>2</sub> source-CCU technology combinations in the LCA literature, and further research could harmonise CO<sub>2</sub> sources and capture processes to provide additional insights on remaining CO<sub>2</sub> source-CCU technologies combinations. This would allow for including more innovative CO<sub>2</sub> capture routes, and identifying more environmentally optimal combinations of capture processes and conversion processes for different CO<sub>2</sub> sources.

### **CCU vs CCS**

Cuéllar-Franca and Azapagic (2015) showed that per tonne of CO<sub>2</sub> captured, CCS results in lower emissions than CCU. While capture emissions are the same, emissions for compression and injection of CO<sub>2</sub> in geological formations are lower than most CCU technologies' emissions from conversion and ultimate release of CO<sub>2</sub>. Based on this, only CCU technologies with low conversion emissions and permanent storage could compete with CCS. Although the avoided emissions of the product that CCU replaces are not taken into account (Cuéllar-Franca and Azapagic 2015), including these could still lead to the same conclusion, as shown for methanol (Abanades et al. 2017). To deal with residual flows containing CO<sub>2</sub> from essential industries as long as they exist, further research could focus on the systematic comparison of CCS and CCU technologies in light of their product-specific substitutes (Hanssen and Huijbregts 2019) and TRLs.

## 2.6 Conclusions and policy recommendations

By combining the literature on technological maturity of CCU with the global emission reduction requirements consistent with the Paris Agreement, we conclude that only very few CCU options would be Paris-compatible. Moreover, some CCU options only meet the criteria in the short run and could lead to a lock-in towards 2050. We find that for a CCU technology to be Paris-compatible in 2030, it has to have low GHG emissions from CO<sub>2</sub> capture and conversion, replace a GHG-intensive substitute, and (in most cases) lead to permanent storage. For 2050, the criteria become more stringent, and Paris compatibility typically depends on a combination of no capture and conversion emissions (e.g., by using zero-emission electricity or waste heat), permanent storage, and the use biogenic or atmospheric CO<sub>2</sub> sources. Achieving CDR via CCU technologies is only possible when biogenic or atmospheric CO<sub>2</sub> is used and permanently stored.

While our conclusions are robust as a result of the harmonisation approach we adopted, we also showed that determining the CO<sub>2</sub> emission reduction associated with a CCU technology depends on a range of assumptions. When designing policies for the implementation of CCU technologies, clear guidelines for these assumptions must be in place. We recommend using a future electricity mix and

considering the full life cycle, including the possible re-emission of CO<sub>2</sub>. To clarify the emission reduction potential of CCU, it should also be made very clear against what CCU is compared. We distinguish three benchmarks: 1) Emission reduction of CCU compared to emitting CO<sub>2</sub> at the point source, 2) Emission reduction of CCU compared to the substituted original product, and 3) Emission reduction of CCU compared to other alternative technological routes replacing the original product. This study's assessment of the Paris compatibility of CCU was based on the first two, but the third benchmark could shed light on the potential trade-offs between CCU and other options for replacing fossil-based production, which may change the verdict on those CO<sub>2</sub> source-CCU options close to the Paris compatibility emission frontier.

Our findings have implications for climate and innovation policy. Research funding may currently be allocated to CCU options that are unlikely to be mature in time for the Paris temperature limits, or that are not able to reduce emissions sufficiently. To be in line with the objectives of the Paris Agreement, such funding would have to be redirected to mitigation options consistent with the Paris Agreement and unlikely to result in carbon lock-in. CCU encompasses a wide range of technologies with different conversion processes, product lifetimes and substituted products. Therefore, we recommend that decision-makers recognise this diversity in CCU, base their decisions on the share of emissions an individual CCU technology can reduce, and whether (close-to-)zero emissions or CDR can be achieved, rather than treating CCU as a homogenous technology. In addition, the technology's current level of technological maturity and when it is expected to be ready for diffusion should be considered. Such a focus could facilitate the creation of strategies that accelerate the development of technologies with low TRLs that may lead to (close-to-)zero emissions or CDR.

## 2.7 Experimental procedure

### ***Data and code availability***

For the identification of a Paris-compatible mitigation pathway we used data available from the IPCC IAMC 1.5°C Scenario Explorer and Data (Huppmann et al. 2018). Full dataset (Data S1) with harmonised emissions of CCU technologies

for 2030 and 2050, and source data for Figure 2.4, Figure 2.5 and Supplementary Figure 2.1 has been deposited at DANS EASY under DOI: 10.17026/dans-28h-n6zj.

### ***Identification of a Paris-compatible mitigation pathway***

We use all "no-overshoot" and "low-overshoot" 1.5°C pathways as defined in the IPCC IAMC 1.5°C Scenario Explorer and Data (Huppmann et al. 2018) (version 1.1). We take the mean of the global annual emissions in these pathways from 2010 to 2060 to arrive at the Paris-compatible mitigation pathway in Figure 2.2. The 'Half of 2020 emissions' in Figure 2 is found by multiplying the mean 2020 emissions in these pathways by 0.5.

### ***Selection of LCA studies to determine the GHG intensity of CCU products***

To assess Paris compatibility of CCU technologies, we determined the GHG emission intensity of CCU products (kgCO<sub>2</sub>-eq./kg CO<sub>2</sub> utilised) and of substituted conventional products (assuming 1:1 replacement) based on a literature search of life cycle assessment studies on CCU. The Web of Science search string that was used is:

TOPIC = CCU\* OR "carbon capture and utili\$ation" OR "carbon capture utili\$ation" OR "carbon capture and use" OR "carbon capture and re-use" OR "carbon dioxide capture and utili\$ation" OR "carbon dioxide capture utili\$ation" OR "carbon dioxide capture and use" OR "carbon dioxide capture and re-use" OR "carbon dioxide utili\$ation" OR "carbon dioxide use" OR "CO2 use" OR "CO2 utili\$ation" OR "CO2 re-use" OR "CO2-enhanced" OR "CO2-based" OR "CO2-activated" OR "CO2 capture and utili\$ation" OR "CO2 capture and use" OR "CO2 capture and re-use" OR "produc\* from CO2" OR "produc\* from carbon dioxide" OR "carbonat\* curing" OR "CO2 curing" OR "carbonat\* aggregate\*" OR (CO2 calcium carbonat\*) OR ("carbon dioxide" "calcium carbonat\*") OR (CO2 micro-alga\*) OR (CO2 microalga\*) OR ("carbon dioxide" microalga\*) OR ("carbon dioxide" micro-alga\*) OR (CO2 carbon nanotube\*) OR ("carbon dioxide" carbon nanotube\*) OR (CO2 carbon nanofib\*) OR ("carbon dioxide" carbon nanofib\*) OR (CO2 enhanced oil recovery) OR ("carbon dioxide" enhanced oil recovery) OR (CO2 Fischer-Tropsch synthesis) OR ("carbon dioxide" Fischer-Tropsch synthesis)

AND TOPIC = LCA OR life-cycle OR lifecycle OR "carbon footprint\*" OR "climate footprint\*" OR "environment\* impact\*" OR "climat\* impact\*" OR "GHG balance" OR "greenhouse gas balance" OR "carbon balance" OR "global warming potential" OR "global warming impact\*" OR "global warming footprint\*" OR "carbon-negative" OR "below zero".

On 30 November 2020 this search string led to 1041 results. After screening these papers for relevance (1) fulfilling our definition of CCU; 2) determining the global warming potential and not only LCA end-points; 3) including emissions from energy use) and excluding conference proceedings and reviews and meta-analyses without original data, 106 studies were remaining (Supplementary Table 2.3). Microalgae biofuel studies were excluded based on the agreement in the most recent reviews, meta-analyses and harmonised LCA studies that emissions were not consistently reduced compared to fossil fuels (Tu et al. 2018; Dasan et al. 2019; Valente et al. 2019) or on average even doubled (Garcia et al. 2020). For each individual combination of CCU technology and type of CO<sub>2</sub> source (fossil, biogenic or atmospheric), the most recent study with the most complete life cycle inventory was selected. This resulted in 30 studies (1 on direct use, 15 on fuels and chemicals, 8 on mineral carbonates and construction materials, and 6 on enhanced hydrocarbon recovery), together covering 44 unique CCU technologies and resulting in 74 combinations based on multiple CO<sub>2</sub> sources. If the same CCU process was presented with small changes in set-up or composition, the process with the lowest overall GHG emissions was used in this study. If several CO<sub>2</sub> sources of the same 'type' (fossil, biogenic, atmospheric) were given in the LCA, the source most likely to be deploying CO<sub>2</sub> capture in a 1.5°C pathway and to still exist in 2030 and 2050 was preferred (i.e. an industrial facility over a fossil fuel-based power plant; and a natural gas power plant over a coal-fired power plant). For the processes for which a pure biogenic source was not available but a source representing a mix of biogenic and fossil sources was available, then the mix was included in addition to the fossil and atmospheric sources. When the emissions for the substitute were not included in the study, the emissions reported in another LCA for the same product were used and referred to in Supplementary Table 2.4.

### ***Derivation of GHG emissions of CCU products in 2030***

We apply a system expansion via substitution approach to solve the multifunctionality problem following the LCA standard ISO 14044 and LCA guidelines for CCU (Global CO<sub>2</sub> Initiative 2020). This method assumes that a primary production plant *with CCU* directly substitutes an identical plant *without CCU*, to ultimately determine the emissions that can be associated with CCU itself. The GHG emissions of these two systems can be determined as shown in equations 2.3 and 2.4.

$$E_{\text{plant w/o CCU}} = E_{\text{upstream}} + E_{\text{primary, direct}} + E_{\text{primary, other}} \quad (2.3)$$

$$\begin{aligned} E_{\text{plant with CCU}} = & E_{\text{upstream}} + E_{\text{primary, other}} + E_{\text{capture losses}} \\ & + E_{\text{capture process}} + E_{\text{conversion}} + E_{\text{other}} + E_{\text{released}} \end{aligned} \quad (2.4)$$

Where: E stands for GHG emissions (in kgCO<sub>2</sub>-eq./kg CO<sub>2</sub> utilised): All emissions are scaled to the functional unit of 1 kg of CO<sub>2</sub> utilised. We specifically distinguish the following GHG emissions:

$E_{\text{upstream}}$  Upstream GHG emissions associated with the extraction of resources or production of the carbon feedstock feeding into the primary production process (as feedstock and/or fuel), including handling and transport to the plant.

$E_{\text{primary, direct}}$  Direct CO<sub>2</sub> emissions emitted at the point source in the primary production process.

$E_{\text{primary, other}}$  Other (remaining) GHG emissions associated with the primary production plant.

$E_{\text{capture losses}}$  Emissions of CO<sub>2</sub> that is not captured in the capture process (capture is not 100% efficient).

$E_{\text{capture process}}$  GHG emissions associated with capturing CO<sub>2</sub> and separating the CO<sub>2</sub> from the bulk gas stream (e.g., from additional electricity use).

$E_{conversion}$	GHG emissions of the conversion process of the CO <sub>2</sub> into the end-product.
$E_{released}$	Emissions of utilised CO <sub>2</sub> to the atmosphere at the end-of-life of the CCU product.
$E_{other}$	Other (remaining) GHG emissions associated with the CCU product, such as GHG emissions at end-of-life other than the release of utilised CO <sub>2</sub> .

Furthermore, the amount of CO<sub>2</sub> that is utilised in CCU can be determined as shown in equation 2.5.

$$E_{utilised} = E_{primary,direct} - E_{capture\ losses} \quad (2.5)$$

The GHG emissions that can be attributed to the CCU product via substitution can thus be determined via equation 2.6a-c.

$$E_{CCU} = E_{plant\ with\ CCU} - E_{plant\ w/o\ CCU} \quad (2.6a)$$

$$E_{CCU} = E_{upstream} + E_{primary,\ other} + E_{capture\ losses} + E_{capture\ process} \quad (2.6b)$$

$$+ E_{conversion} + E_{other} + E_{released} - E_{upstream} - E_{primary,direct} - E_{primary,\ other}$$

Solving and filling in equation 2.5 then results in equation 2.6c.

$$E_{CCU} = -E_{utilised} + E_{capture\ process} + E_{conversion} + E_{other} + E_{released} \quad (2.6c)$$

Note that in the case of DAC, there is no multi-functionality problem, but equation 2.6c can still be used to determine  $E_{CCU}$ , as  $E_{utilised}$  is simply the amount of CO<sub>2</sub> captured from air. When the CO<sub>2</sub> must already be separated in the primary process, CO<sub>2</sub> is a by-product and the capture emissions ( $E_{capture\ process}$ ) are zero.

Equation 2.6c is used in the main text as equation 2.1.

### ***Derivation of GHG emissions of CCU products in 2050***

The substitution approach is based on the assumption that a point source without capture is 1:1 substituted by the same plant with capture. In the near future (i.e., 2030) this assumption holds, as many plants without capture can still be substituted. In the long run, however, the reference system is expected to transform towards a zero-emissions economy. Therefore, in a Paris-compatible 2050 system, we can no longer assume that there are *unabated* fossil point sources to substitute and associated emissions to avoid. Since emissions are no longer unabated, considering CCU as an option to abate fossil CO<sub>2</sub> emissions, means that the responsibility for not emitting shifts to CCU. In terms of calculating the GHG intensity of CCU for a fossil CO<sub>2</sub> source in 2050, this means all emissions from the stack onwards are assigned to the CCU product (equation 2.7).

$$E_{CCU(\text{fossil in 2050})} = E_{\text{capture losses}} + E_{\text{capture process}} + E_{\text{conversion}} \\ + E_{\text{other}} + E_{\text{released}} \quad (2.7)$$

Equation 2.7 is used in the main text as equation 2.2.

To calculate  $E_{\text{capture losses}}$  in 2050, a high capture rate of 95% is assumed based on the finding that such high capture rates would be needed (Bui et al. 2018) and are expected to become dominant after 2040 (IEA 2020) under stringent decarbonisation targets. Furthermore, in achieving a net-zero CO<sub>2</sub> system, Wevers et al. (2020) found going beyond a 95% capture rate would be energetically unfavourable to deploying DAC for the remaining 5%.

In the results, we report  $E_{CCU}$ , but when this applies to fossil CO<sub>2</sub> sources in 2050, equation 2.7 is used for the calculation. For CCU processes based on biogenic or atmospheric CO<sub>2</sub> in 2050, equation 2.6c still applies, because availability of these sources in 2050 is in line with a Paris-compatible scenario.

### ***Assumptions to arrive at harmonised GHG emissions of CCU product and substitute***

To complement the 'Emissions of CCU technologies' section and Supplementary Table 2.4 which includes detailed assumptions per LCA study, this section provides details on overarching assumptions in the harmonisation of GHG emissions of CCU products.

For some CCU products, not all carbon in the end-product finds its origin in captured CO<sub>2</sub>. Instead, some processes use fossil methane or methanol as additional feedstock, for example in dry reforming of methane where (captured) CO<sub>2</sub> and methane are used to produce CO and H<sub>2</sub> (Sternberg et al. 2017). This results in the emission of additional CO<sub>2</sub> besides the  $E_{released}$  of 1 kgCO<sub>2</sub>/kg CO<sub>2</sub> utilised upon combustion of the chemical/fuel: these additional emissions are included in  $E_{other}$ . Conversely, in CO<sub>2</sub>-EOR the CO<sub>2</sub> utilised to produce the hydrocarbon does not end up in the product itself, but stays behind in the depleted oil field. The CO<sub>2</sub> is cycled and used several times in the process: the amount of CO<sub>2</sub> utilised per barrel of oil is taken to be the total amount of CO<sub>2</sub> injected in a well over the years of CO<sub>2</sub>-EOR operation and divided by the total number of barrels of oil produced. Given that the CO<sub>2</sub> utilised is defined as the CO<sub>2</sub> stored in the well,  $E_{released}$  is zero. The combustion of the produced hydrocarbon is included in  $E_{other}$ .

Several processes produce co-products with the CCU product or with the feedstock, leading to a multifunctionality problem. In ethylene production, propylene, butene and hydrogen are co-produced (Ioannou et al. 2020); in the electrochemical reduction process O<sub>2</sub> and H<sub>2</sub> are co-produced (Rumayor et al. 2020); and for hydrogen production via electrolysis 7.94 kg O<sub>2</sub>/kg H<sub>2</sub> is co-produced. We follow the LCA standard ISO 14044 guidelines and apply the system expansion via substitution approach. In particular, we assume that the production of 1 kg of H<sub>2</sub> via electrolysis avoids the production of 7.94 kg of O<sub>2</sub> via cryogenic air separation. We used the production process of O<sub>2</sub> using air separation in Ecoinvent v3.6 and adapted this process to ensure harmonisation of the electricity used in electrolysis and the substituted oxygen production process: we replaced the electricity source for the required 1.42 kWh/kg O<sub>2</sub> in the Ecoinvent process by the 2030 and 2050 electricity mix used in the harmonisation. In 2030, this results in an emission reduction factor of 22% for H<sub>2</sub> production due to substitution: we multiply the emissions for H<sub>2</sub> with a factor 0.78; in 2050 with electricity assumed to be emission-free H<sub>2</sub> production has (close-to-)zero emissions.

For cement-based substitutes and CCU-construction materials we include the CO<sub>2</sub> uptake during the use phase and end-of-life phase, which is not included in the analysed LCAs. Xi et al. (2016) show that from 1930 to 2013, 43% of the limestone calcination emissions in cement production were off-set due to atmospheric CO<sub>2</sub>

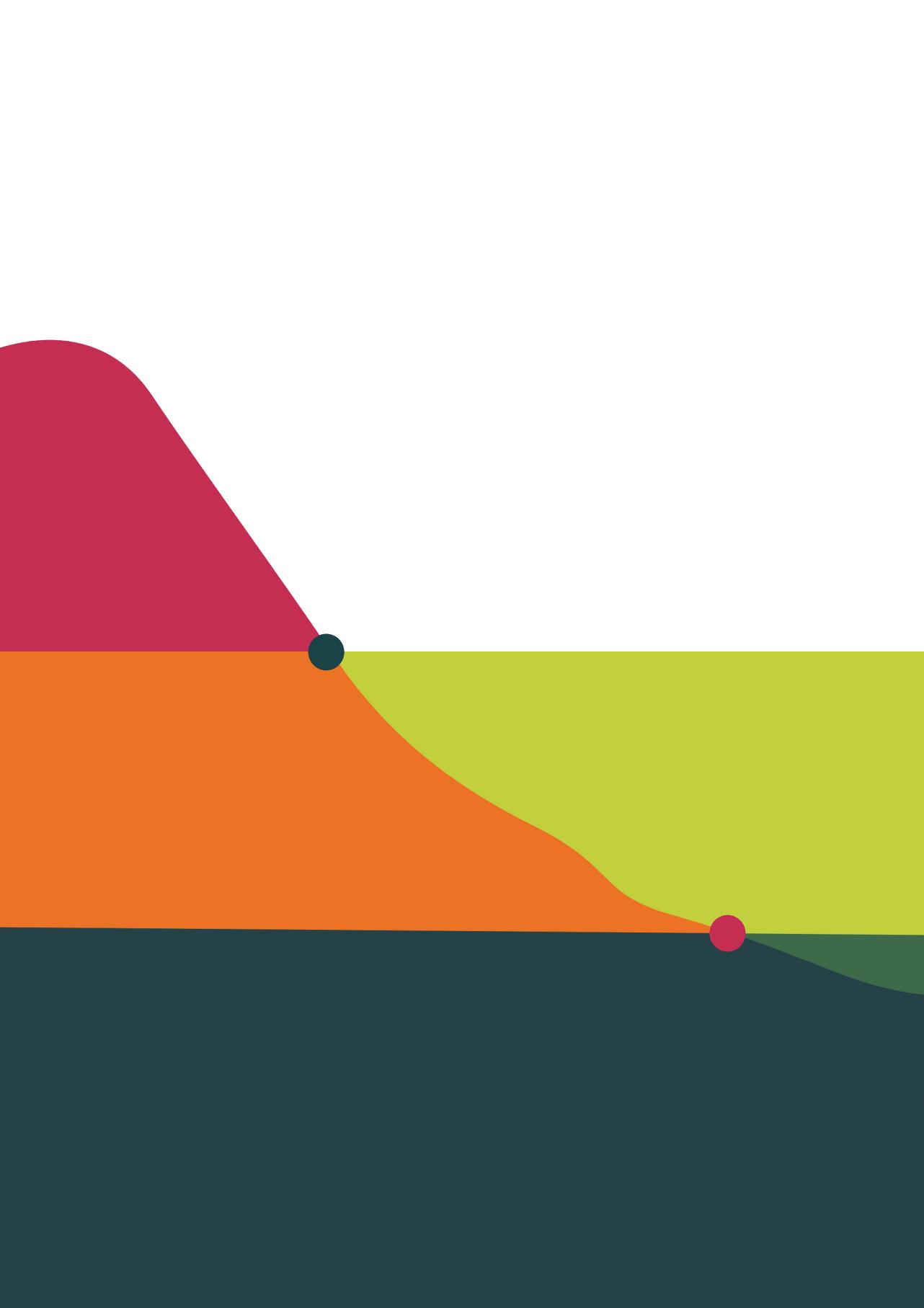
absorption in a natural cement carbonation process; Cao et al. (2020) project that from 2015 to 2100, 30% of cement production emissions are absorbed by cement carbonation, largely in line with Xi et al. (2016) considering that calcination of limestone currently accounts for 58.4% of the CO<sub>2</sub> emissions of cement production (Cao et al. 2020). To take this uptake of CO<sub>2</sub> into account, we multiply the emissions of cement production by a factor 0.7 to find the full life-cycle emissions including uptake.

In our analysis, we assume that industrial waste as feedstock is impact-free. Several processes in the mineral carbonates and construction materials category require the input of industrial wastes, e.g. steel slag, fly ash or desalination brines. Although some studies assume that CCU avoids landfilling or processing of these waste streams, thereby assigning avoided emissions to the CCU product, there is no consensus on how much these are (i.e., a factor four difference between avoided emissions from slag landfilling of -0.04 kgCO<sub>2</sub>-eq/kg CaCO<sub>3</sub> (Mattila et al. 2014) and -0.16 kgCO<sub>2</sub>-eq/kg CaCO<sub>3</sub> (Lee et al. 2020)). We follow the most regularly used and most conservative approach of assuming impact-free waste streams.

### *Lifetimes of CCU products*

We calculated the GHG intensity of CCU products without (base case) and with the benefit of temporary carbon storage (sensitivity analysis), using a time horizon of 100 years. To include the storage effect,  $E_{released}$  in equation 2.1 and equation 2.2 is multiplied by GWP<sub>storage</sub>-100, the value of which depends on the product-specific lifetime (see Table 2.2). For direct use of CO<sub>2</sub> in an agricultural greenhouse, a lifetime of less than 6 months is assumed, based on Mazotti et al. (2005). For enhanced hydrocarbon recovery, permanent storage (Mazotti et al. 2005) or at least millennia (Hepburn et al. 2019) is assumed. For mineral carbonates and construction materials, a lifetime of centuries to permanent is assumed, based on Hepburn et al. (2019), Pan et al. (2016) and Sanna et al. (2014), except for sodium bicarbonate which may release the CO<sub>2</sub> upon use, assuming a lifetime of six months. In the fuels and chemicals category, for more stable chemicals (polyethylene, polypropylene and polyols) a lifetime of 50 years is assumed, based on the lifetime of 'months to decades' for polymers (Hepburn et al. 2019) or 'decades to centuries' for polyurethanes (Mazotti et al. 2005). For the remainder of fuels and chemicals a lifetime of six months is assumed, based on the lifetime

of six months indicated for methanol and urea by Mazotti et al. (2005), and for methane, Fischer-Tropsch fuels and dimethylether by Hepburn et al. (2019).



# **Environmental benefits of urea production from basic oxygen furnace gas**

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**3**

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## Abstract

Basic oxygen furnace gas (BOFG) is a multi-component residual flow from integrated steel mills composed of CO, CO<sub>2</sub>, O<sub>2</sub>, N<sub>2</sub> and H<sub>2</sub>. In this study, we use the life cycle assessment method to quantify the environmental impacts of six applications of BOFG: heat for internal use in the steel mill, heat combined with CO<sub>2</sub> storage, electricity production, urea production, urea production combined with CO<sub>2</sub> storage and flaring. Urea can be produced from solely BOFG components, making use of the sorption enhanced water-gas shift technology. This application of BOFG is described here for the first time. The environmental impacts of these six applications were compared in light of the impacts of their conventional production ('counterfactual'). Using BOFG for the production of electricity and urea would result in net greenhouse gas emissions of 0.87 tCO<sub>2</sub>-eq/t<sub>BOFG</sub> and 0.69 tCO<sub>2</sub>-eq/t<sub>BOFG</sub>, respectively. When excess CO<sub>2</sub> is transported and stored, net emission savings could be achieved when producing urea (-0.11 tCO<sub>2</sub>-eq/t<sub>BOFG</sub>) and heat (-0.12 tCO<sub>2</sub>-eq/t<sub>BOFG</sub>) from BOFG, again as compared to their counterfactual of conventional production. Overall environmental impacts were slightly lower for electricity (5.2E-4 DALY/t<sub>BOFG</sub>, 1.9E-6 species.yr/t<sub>BOFG</sub>) compared to urea (6.2E-4 DALY/t<sub>BOFG</sub>, 1.9E-6 species.yr/t<sub>BOFG</sub>). Sensitivity analysis showed, however, that when replacing cleaner electricity (e.g. under future power decarbonisation), these relative environmental benefits of electricity production from BOFG are diminished. We conclude that urea production, rather than the current practice of electricity production, is the best investigated option to reduce environmental impacts of BOFG, which can lead to net environmental benefits when combined with CO<sub>2</sub> storage.

### 3.1 Introduction

To limit global warming to 1.5°C, global anthropogenic CO<sub>2</sub> emissions need to reach net zero by 2050 (IPCC 2018b). This means that deep emission reductions in all sectors are required. While this is a challenge in itself, decarbonising one sector may also imply extra challenges for another sector. For example, low-carbon energy production may increase metal demand (Kleijn et al. 2011). Such dependencies increase the challenge of decarbonising the industrial sector, which already accounts for roughly one-third of global greenhouse gas (GHG) emissions (IPCC 2014). In 2014, the iron and steel sector accounted for 23% of the global industrial final energy demand, and for 28% of its total direct CO<sub>2</sub> emissions, making it the leading CO<sub>2</sub> emitter in industry (IEA 2017a). Crude steel production is expected to keep growing according to international energy association (IEA) projections in baseline and climate mitigation (1.75°C) scenarios. At the same time, limiting global warming to 1.5°C requires CO<sub>2</sub> emissions from industry to be reduced by 65-90% in 2050 compared to 2010, or by 50-80% to reach 2°C (IPCC 2018b). For steel specifically, a 92% reduction in direct CO<sub>2</sub> emissions per tonne of steel produced in 2060 is required in the IEA 1.75°C scenario (IEA 2017a). There is no single technology that can achieve such emission reductions in this sector by itself. Instead, different technologies need to be combined and further developed, including carbon capture and storage (CCS) and carbon capture and utilisation (CCU) (de Coninck et al. 2018; Yan and Zhang 2019).

The life cycle assessment (LCA) method has previously been used to study the potential environmental benefits of different CCS and CCU technologies. It was found that while CCS and CCU technologies reduce the global warming potential (GWP) impacts compared to not using these technologies, the extent of the reduction depends on the specific technology used. Furthermore, other environmental impacts such as acidification and human toxicity increase (Cuéllar-Franca and Azapagic 2015). In the iron and steel sector, the success of CCS and CCU depends on the possibility for gas separation of the multicomponent steel mill off-gases (Ramírez-Santos et al. 2018). An important challenge for CO<sub>2</sub> capture is to find a family of chemicals that can replace the most commonly used post-conversion CO<sub>2</sub> capture method of absorption by monoethanolamine (MEA),

as its use and regeneration still result in significant CO<sub>2</sub> emissions that oppose further reduction of the GWP impacts (Cuéllar-Franca and Azapagic 2015). Recently, instead of focusing on low temperature liquid absorption, research efforts have focused on high temperature solid adsorption of CO<sub>2</sub> which requires less energy input for regeneration and is economically more attractive (Ho et al. 2013; Ben-Mansour et al. 2016). One such technology that i) allows for gas separation of multicomponent gases, and ii) is based on high temperature solid adsorption of CO<sub>2</sub>, is the sorption enhanced water-gas shift (SEWGS) technology. It uses a regenerative solid adsorbent, potassium-promoted hydrotalcite, for converting residual CO to CO<sub>2</sub>, while simultaneously adsorbing CO<sub>2</sub> (Boon et al. 2015; Van Dijk et al. 2017). A previous LCA on CO<sub>2</sub> capture on a steel mill found that the GWP impacts were reduced by an extra 33% for SEWGS compared to MEA CO<sub>2</sub> capture, while all other environmental impacts were reduced as well (Petrescu et al. 2019).

Historically, residual gases from a steel plant including Basic Oxygen Furnace gas (BOFG) are combusted in the reheating furnaces, flared, or sent to a power plant to generate power (Carpenter 2012). The sale of residual gases allowed the steel plant to import the necessary power at marginal cost and generate additional revenue steam to maintain profitable operation. However, in recent years two major changes have affected these dynamics. First, increasing penetration of renewable electricity combined with policy-based incentives allowed renewables to squeeze the conventional operators in such a way that these power plants operate less per year (Frontier Economics 2015). Second, the increase in carbon price in the EU Emissions Trading Scheme means the cost to the steel maker is increased (Boutabba and Lardic 2017). The combination of the two results in the revenue from the sale of residual gases to decrease and the costs due to the emissions to increase.

In the BOF2UREA project from which this paper is derived, a process is developed in which BOFG is utilised to produce urea, in collaboration with partners across the value chain including iron and steel producers, ammonia and urea industry and technology suppliers. BOFG consists of CO (60%), N<sub>2</sub> (24%), CO<sub>2</sub> (16%), O<sub>2</sub> (0.3%), H<sub>2</sub> (0.02%) and CH<sub>4</sub> (0.02%) with varying compositions (means are given, see Supplementary Information Section 3.1 for more information). The SEWGS technology can be used to capture CO<sub>2</sub> from BOFG, creating a pure stream of highly

concentrated CO<sub>2</sub> and a remaining stream containing N<sub>2</sub> and H<sub>2</sub> in ratios from which ammonia can be produced (Liu et al. 2002). After ammonia synthesis, part of the concentrated CO<sub>2</sub> stream can be combined with the ammonia to produce urea, leaving the rest of the CO<sub>2</sub> ready for storage. The resulting urea has a large potential market, at a current global market volume of 180 Mt per year (Chauvy et al. 2019). Its main use is as a nitrogen fertiliser, which is expected to increase by 50% towards 2050 to feed a growing global population (UN DESA 2017; FAO 2018), but can also be used in the production of chemicals, plastics, and dietary supplements for livestock. The environmental impacts of using this innovative technique to produce urea from BOFG have not been quantified, and a comparison with the impacts of other BOFG applications has not been made.

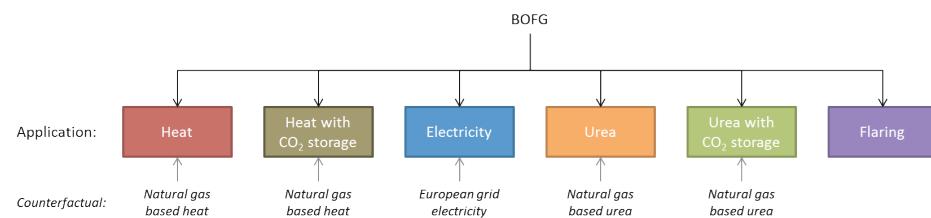
## 3

The aim of this study is to assess the environmental impacts of urea production from BOFG, with and without carbon capture and storage, heat production with carbon capture and storage, and three existing BOFG applications: internal use in the steel mill to meet heat demand, combustion in a power plant for electricity generation, and flaring. The environmental impacts of the applications were quantified using LCA and compared in light of the impacts associated with their conventional production, referred to as counterfactuals. The use of counterfactuals allows for a comparison of different energy and material applications of a single feedstock (Hanssen and Huijbregts 2019), because 1) environmental impacts are determined per feedstock utilised, and 2) the different nature of the applications is taken into account by making explicit what the applications replace (i.e., the counterfactual), and which emissions are thus avoided. Furthermore, this study aims to identify the process steps in urea production from BOFG with the largest environmental impact, allowing for targeted improvement of the process to reduce the overall impact.

## 3.2 Methodology

### 3.2.1 BOFG applications and counterfactuals

Figure 3.1 provides an overview of the six applications of BOFG from an integrated steel mill and their counterfactuals considered in this study. The assumed location of the steel mill is Europe.



**Figure 3.1. Schematic presentation of the applications of basic oxygen furnace gas (BOFG) and their respective counterfactuals (in *italics*) considered in this study.**

The first application of BOFG is combustion of BOFG to produce heat. BOFG, along with off-gases from the coke oven and blast furnace can currently be used internally to meet the heat demand for steel production. The counterfactual for heat production from BOFG is heat production from natural gas.

The production of heat through combustion can be decarbonised by processing BOFG with SEWGS. This results in a stream of H<sub>2</sub> and N<sub>2</sub>, as well as a concentrated stream of CO<sub>2</sub>. The H<sub>2</sub>/N<sub>2</sub> rich stream is then combusted for heat, while the stream of CO<sub>2</sub> can be transported in pipelines and injected for geological storage as the CO<sub>2</sub> purity is sufficiently high (Chisalita et al. 2019). The counterfactual is again heat production from natural gas.

Another current use of BOFG is to combust it in an on-site boiler to raise steam which drives a steam turbine to produce electricity. Production was based on the performance of average Western European captive power plant using steelworks off-gases as fuel with an efficiency of 32.1%, described in IEA GHG (IEAGHG 2013). The counterfactual for electricity production from BOFG is the current average European mix of electricity production technologies (see Supplementary Table 3.3).

Power generation combined with CO<sub>2</sub> storage is not considered because, as explained in the introduction, the revenue from the sale of BOFG is decreasing, which makes the production of electricity in the future an unlikely commercial avenue. Even if CO<sub>2</sub> capture can be implemented, the cost of electricity generation from renewables would be lower than power production from residual steel gases.

A new application of BOFG we investigate here is to use it to produce urea via the SEWGS technology. The resulting stream of H<sub>2</sub> and N<sub>2</sub>, as well as the concentrated stream of CO<sub>2</sub> are then used for urea production. The counterfactual for urea production from BOFG is conventional urea production based on steam reforming of natural gas. Both urea synthesis processes are described in the section 3.2.2 and in more detail in Supplementary Information Section 3.1.

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While the excess of CO<sub>2</sub>, i.e. the part of the concentrated stream of CO<sub>2</sub> from the SEWGS process that is not used in urea synthesis, is assumed to be vented in the base case of urea production from BOFG, it could instead be transported and injected for geological storage (Chisalita et al. 2019). The assumed counterfactual is again natural gas-based urea production, which, based on stoichiometry, causes no excess process CO<sub>2</sub> emissions.

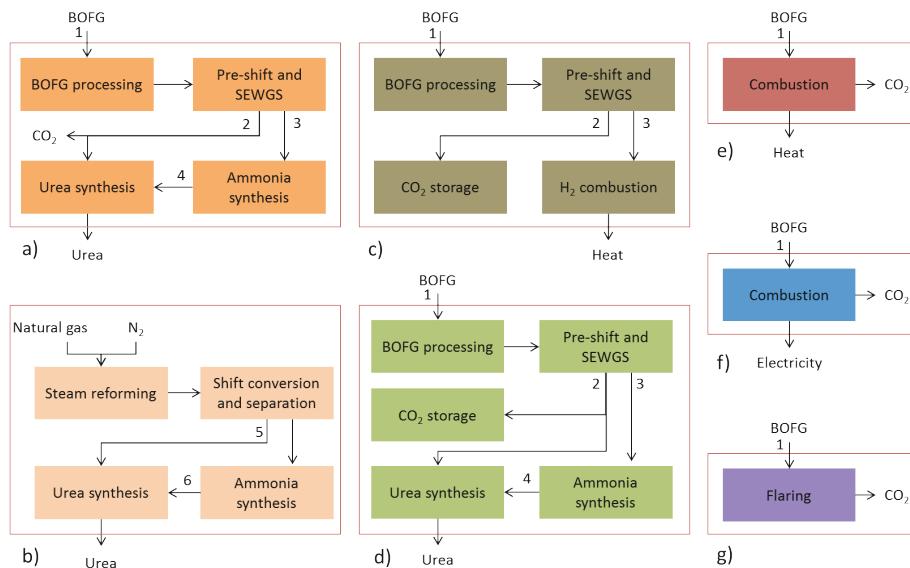
In some steel mills, the BOFG is currently not utilised at all, and is instead flared (Carpenter 2012). The 'worst-case' application of BOFG considered in this study is therefore flaring of BOFG. This application does not have a counterfactual as there is no material or energy to replace.

### 3.2.2 Description of the modelled process

#### *Urea production from BOFG and its counterfactual*

The urea synthesis process design using BOFG as feedstock was based on a production capacity of 50 kilo tonnes per annum (kta) and uses the SEWGS technology, which has reached technology readiness level (TRL) 6, i.e., the prototype stage (Van Dijk et al. 2017). The urea market is typically a regional market as transportation costs are high, hence localised production at the 50-200 kta scale is a commercially relevant scale. The design was assumed to be the same for production capacities up to at least 200 kta. For the process design, the plant

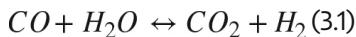
is subdivided into four sections, specifically; BOFG processing; pre-water gas shift (pre-shift) and SEWGS; ammonia synthesis and finally, urea synthesis. Each of these sections were modelled using ASPEN Plus for the units operating at steady state and Matlab for the SEWGS unit operating at cyclic steady state (for more details, see Boon et al. 2015, 2016). A simplified flowsheet of urea production from BOFG is shown in Figure 3.2a. Supplementary Information Section 3.1 contains more information on the process design, detailed process descriptions and compositions of the streams shown in Figure 3.2.



**Figure 3.2. Simplified flowsheets of the processes considered and system boundaries for the LCA as indicated by the red rectangles for:** a) urea production from BOFG using the SEWGS technology; b) conventional urea production from natural gas; c) heat production from BOFG with hydrogen combustion and CO<sub>2</sub> storage; d) urea production from BOFG in combination with CO<sub>2</sub> storage; e) heat from BOFG combustion; f) electricity from BOFG combustion and g) flaring of BOFG. The compositions of the flows indicated by the numbers 1 to 6 are given in Supplementary Table 3.1.

The average composition of BOFG (CO (60%), N<sub>2</sub> (24%), CO<sub>2</sub> (16%), O<sub>2</sub> (0.3%), H<sub>2</sub> (0.02%) and CH<sub>4</sub> (0.02%), see Supplementary Information Section 3.1) was based on data of the first quarter of 2018 for a typical Western European integrated steel mill with a production capacity of 6 million tonnes of steel per year. The BOFG processing steps consist of compression, heating and mixing the BOFG with

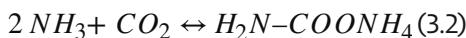
steam. In the pre-shift reactor, CO is reacted with H<sub>2</sub>O to form CO<sub>2</sub> and H<sub>2</sub>, through a moderately exothermic ( $\Delta H^0 = -41.2 \text{ kJ mol}^{-1}$ ) water-gas shift (WGS) reaction:



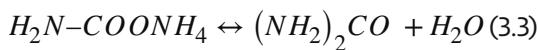
The N<sub>2</sub> does not participate in the WGS reaction. Further conversion of the CO is not possible in conventional WGS reactors due to the reaction mixture being at thermodynamic equilibrium (Lloyd et al. 1989). In order to maximise CO conversion and remove CO<sub>2</sub> at the same time, the SEWGS process is used. In this process, the WGS reaction is combined with CO<sub>2</sub> removal in a cyclic and continuous process with multiple columns filled with potassium-promoted hydrotalcite adsorbent (Hufton et al. 1999; Van Selow et al. 2009). This cyclic process is made up of adsorption, rinse, blowdown and purge steps and produces an H<sub>2</sub>/N<sub>2</sub> rich stream and a CO<sub>2</sub> rich stream (Yong et al. 2002; Boon et al. 2014, 2015). The steam demands for the rinse and purge steps are satisfied through heat integration with the rest of process and heat import from the iron and steel mill. Supplementary Information Section 3.1 contains a more detailed process description.

Before the resulting H<sub>2</sub>/N<sub>2</sub> rich stream can be used for ammonia synthesis, sulphur compounds are removed using a zinc oxide guard bed. Any remaining oxygen containing compounds are removed in a methanation step as these are poisons to magnetite catalysts used in the ammonia reactor. The resulting H<sub>2</sub>/N<sub>2</sub> mixture is compressed, heated and converted to ammonia in an ammonia synthesis loop following the state of the process for ammonia synthesis as described in (EFMA 2000a; Fiedler et al. 2003).

To synthesise urea, first ammonia and carbon dioxide react to form ammonium carbamate:



In a second reaction, ammonium carbamate decomposes into urea and water:



The process used to produce urea is a conventional process as described in (Fiedler et al. 2003).

Based on the BOFG composition and variation, the H<sub>2</sub>:N<sub>2</sub> ratio of the H<sub>2</sub>/N<sub>2</sub> stream of the SEWGS step is empirically found to be 2.4 (see Supplementary Information Section 3.1 for more details), which is a suitable ratio for ammonia synthesis (Liu et al. 2002). Based on this BOFG composition, there is an excess of CO<sub>2</sub>: 20.6% of the CO<sub>2</sub> rich stream from the SEWGS step is used for urea synthesis. The remaining 79.4% of CO<sub>2</sub> is vented or could be transported by pipeline for geological storage, based on Chisalita et al. (2019) and depicted in Figure 3.2d.

The counterfactual of urea production from BOFG is conventional urea production based on a state-of-the art natural gas reforming urea facility with the same small scale production capacity (up to 200 kta). This process is described in the Best Available Techniques (EFMA 2000b,a) and schematically depicted in Figure 3.2b. Natural gas is used as both fuel and feedstock, and the main process steps include sulphur removal, a primary and secondary reformer, high temperature and low temperature WGS reactions, amine based CO<sub>2</sub> removal, methanation and ammonia synthesis. The process CO<sub>2</sub> captured after the shift reactions is compressed and used for urea synthesis, while some CO<sub>2</sub> is emitted in a dilute form from the natural gas burner in the secondary reformer. The nitrogen required for the ammonia synthesis is introduced by firing the secondary reformer with air. For more details, see Supplementary Information Section 3.1.

### ***Heat and electricity production from BOFG***

For heat production, electricity production and flaring of BOFG, complete oxidation of BOFG is assumed (Figure 3.2e-g). When heat production is combined with CO<sub>2</sub> storage, the BOFG is processed in the same way as for urea production. The product streams of SEWGS are of same composition as in the urea case (see Figure 3.2c and Supplementary Table 3.1). The H<sub>2</sub>/N<sub>2</sub> stream is used for combustion in the reheating furnaces while the full CO<sub>2</sub> stream is compressed for storage. In the urea process, heat integration with the ammonia and urea production processes satisfied the steam demands for the rinse and purge steps of SEWGS. In this case, the additional heat demand is satisfied by waste heat from the reheating furnace

since only high temperature heat is used in the furnace (BCS Inc. 2008; Thakur et al. 2015) (see Supplementary Information Section 3.1 for more information).

### 3.2.3 Life cycle assessment

Life cycle assessment is a method to quantify the environmental impacts of a product or service, based on all the inputs from and outputs to the environment throughout its life cycle. An LCA consists of four main steps: goal and scope definition; compilation of Life Cycle Inventory (LCI) data with all inputs and outputs of the process; Life Cycle Impact Assessment (LCIA) where environmental impacts of the inputs and outputs are quantified; and interpretation of the results (ISO 2006).

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#### *Goal and scope*

The goal of this LCA was to compare the environmental impacts of various BOFG uses. The functional unit, which is the unit in which all inputs and outputs of the process are expressed, is the use of one tonne of BOFG. The geographical scope is Europe.

This study focused on alternative uses of a by-product of the integrated steel mill and not on the steel production itself, so the extraction and transport of resources, the steel production process itself, and construction and decommissioning of the integrated steel plant were not considered within the system boundaries of this LCA. BOFG was considered a residual stream feeding into the process, and here considered as the 'cradle' in the LCA. For all applications of BOFG, the steelmaking process and the treatment of BOFG in the steel mill were not affected.

The system boundaries for urea production from BOFG and from natural gas are indicated in Figure 3.2. Operation time was assumed to be 8400 hours per year. Packaging, transport and end-use of urea were not included, as the resulting impacts would be the same as for its counterfactual. This means the system boundary was drawn at the 'gate' of the factory, making this a 'cradle-to-gate' LCA. The LCA considered for all process steps within its system boundaries: the energy that was consumed in the sub-processes, the consumables in terms of catalysts and additional chemicals, all emissions to water, soil and air, and disposal

of waste from the processes. For the counterfactual of natural gas-based urea production, all upstream processes for natural gas production were considered, including extraction and transport.

This LCA did not consider the materials for construction or the transportation of catalysts. The materials for constructions are expected to be similar and precise information for the BOFG-based system was limited due to the early developmental stage of the concept. Furthermore, previous studies have shown the limited impact of both infrastructure and catalyst transportation on the overall outcomes (Koornneef et al. 2008; Petrescu et al. 2019). Although the transportation of catalysts was not included, the production of the catalysts was included. The composition of the catalysts was based on literature (Twigg 1989) and following the standard described in (van Kalkeren et al. 2013), we assumed a yield of 95% for the production process, at an energy use of 2.00 MJ of heat, and 0.333 kWh of electricity per kg of catalyst produced.

For combustion and flaring of BOFG, full oxidation was assumed based on the average composition of BOFG (see Supplementary Table 3.1). Following previous LCAs on steel production off-gases, clean burning of BOFG and emissions to stack was assumed, while materials for construction of the burners and other infrastructure were excluded (Chisalita et al. 2019; Petrescu et al. 2019). For the counterfactuals, all upstream processes for production of electricity and heat from natural gas were included in the system boundaries.

Following previous work (Chisalita et al. 2019), for storage of CO<sub>2</sub>, the transport, injection and pipeline losses (see Table 3.1) were considered within the system boundaries, while construction and decommissioning of the CO<sub>2</sub> transport pipelines were not included.

### ***Life cycle inventory***

The following data sources were used: industrial data; process simulations and modelling data; literature data from previous studies; Best Available Techniques reference documents; and background data from the EcoInvent database v3 (Wernet et al. 2016). As the geographical scope was Europe, the electricity market mix for the 'European Network of Transmission Systems Operators for Electricity'

(ENTSO-E) from the EcoInvent database was used. The market shares of electricity technologies in EcoInvent are based on IEA statistics from 2014 (IEA 2017b). See Table 3.1 for the life cycle inventory for the different applications and their counterfactuals.

**Table 3.1. Life cycle inventory for the six applications of BOFG and counterfactuals per functional unit of 1 tonne BOFG.** Power listings for applications 3, 4 and 5 correspond to the flow of BOFG in a 50 kta urea production system design.

Process inputs and outputs		Per tonne of BOFG utilised
<b>Application 1: Heat</b>		
Heat output BOFG combustion <sup>a</sup>	Combustion of BOFG with lower heating value of 7.51 MJ/Nm <sup>3</sup>	5.50 GJ
Emissions from BOFG combustion <sup>a</sup>	Assumed complete oxidation	1.09 t CO <sub>2</sub>
<b>Counterfactual Application 1: Heat from natural gas</b>		
Heat for internal use in steel mill <sup>b</sup>	From European market group (RER) for heat for industrial use	5.50 GJ
<b>Application 2: Electricity</b>		
Electricity output BOFG combustion <sup>c</sup>	Power plant efficiency 32.1%	0.49 MWh
Emissions from BOFG combustion <sup>a</sup>	Assumed complete oxidation	1.09 t CO <sub>2</sub>
<b>Counterfactual Application 2: Grid electricity</b>		
Grid electricity <sup>b</sup>	European electricity mix (ENTSO-E)	0.49 MWh
<b>Application 3: Urea</b>		
Water consumption <sup>d</sup>	Tap water	0.30 t
Condensate <sup>d</sup>	To waste water treatment	0.03 t
Electricity demand <sup>d</sup>	Compressors and pumps, total 4.66 MW	0.27 MWh
Hot utility <sup>d</sup>	4.1 MW; heat from combustion of purge gases from ammonia and urea section and impurities of NH <sub>3</sub> and CO <sub>2</sub> streams	0.02 t CO <sub>2</sub> 1.5E-4 t CO 1.6E-5 t H <sub>2</sub> 3.1E-22 t CH <sub>4</sub> 5.2E-11 t NH <sub>3</sub>
Steam from steel plant <sup>d</sup>	6 MW upstream ammonia section (0.62 t steam/t <sub>BOFG</sub> ) and 0.39 t steam/t <sub>BOFG</sub> for urea	1.01 t steam
Cold utility <sup>d</sup>	Cooling water in closed loops	0 t
Combustion impurities CO <sub>2</sub> excess <sup>a,d</sup>	Assumed complete oxidation	0.85 t CO <sub>2</sub>
Consumables <sup>d,e,f</sup>	Given as the sum of all	6.18E-4 t

**Table 3.1. Life cycle inventory for the six applications of BOFG and counterfactuals per functional unit of 1 tonne BOFG (continued)**

Process inputs and outputs		Per tonne of BOFG utilised
Waste <sup>g</sup>	To incinerator	6.18E-4 t
<b>Counterfactual Application 3: Natural gas-based urea</b>		
Feed <sup>d,h</sup>	Natural gas to reaction area of primary reformer	101.5 Nm <sup>3</sup>
Fuel <sup>d,h</sup>	Natural gas to burner area of primary reformer	35.1 Nm <sup>3</sup>
Emissions from burner <sup>d,h</sup>	Flue gas	0.08 t CO <sub>2</sub>
Water consumption <sup>d,h</sup>	For boiler feed water	0.25 t
Cooling water <sup>d,h</sup>	Closed loops	0 t
Condensate <sup>d,h</sup>	To waste water treatment	0.01 t
CO <sub>2</sub> extra requirement for urea <sup>d,i</sup>	Not all CO <sub>2</sub> can be supplied from CO <sub>2</sub> removed in previous process step, approximately 5% missing	0.01 t
CO <sub>2</sub> removed in NH <sub>3</sub> production <sup>d,i</sup>	All used in urea synthesis	0.21 t
Urea steam consumption <sub>d,i</sub>	17.3 bara, 205°C, imported from outside battery limits	0.76 GJ
Urea NH <sub>3</sub> stream impurities <sup>a,d</sup>	Impurities combusted and emitted to air	9.2E-4 t CO <sub>2</sub>
Urea CO <sub>2</sub> stream impurities <sup>a,d</sup>	Impurities combusted and emitted to air	9.1E-4 t H <sub>2</sub> O
Consumables <sup>d,e,f</sup>	Given as the sum of all	3.91E-5 t
Waste <sup>g</sup>	To incinerator	3.23E-5 t
Waste <sup>d,h,i</sup>	Emissions to air	6.73E-6 t
<b>Application 4: Urea with CO<sub>2</sub> storage. Changes with respect to Application 3</b>		
Electricity demand <sup>d</sup>	Compression requirement increased, total 6.41 MW	0.37 MWh
CO <sub>2</sub> injection mass <sup>a,d</sup>	CO <sub>2</sub> stream at 110 bar of which 98% is CO <sub>2</sub>	0.85 t CO <sub>2</sub> stream (0.84 t CO <sub>2</sub> )
CO <sub>2</sub> transport and injection loss <sup>j</sup>	Transport distance 800 km; pipeline loss 0.026%/1000 km; injection loss 0.1%	4.7E-7 t H <sub>2</sub> 6.1E-6 t CO 0.001 t CO <sub>2</sub>
CO <sub>2</sub> injection electricity <sup>j</sup>	7 kWh/t CO <sub>2</sub>	5.9 kWh
<b>Counterfactual Application 4: Same as Counterfactual Application 3</b>		
<b>Application 5: Heat with CO<sub>2</sub> storage</b>		
Water consumption <sup>d</sup>	Tap water	0.3 t
Condensate <sup>d</sup>	To waste water treatment	0.03 t

**Table 3.1. Life cycle inventory for the six applications of BOFG and counterfactuals per functional unit of 1 tonne BOFG (continued)**

Process inputs and outputs		Per tonne of BOFG utilised
Electricity demand <sup>d</sup>	Compression requirement increased, total 6.41 MW	0.37 MWh
Hot utility <sup>d</sup>	4.1 MW; met by waste heat from reheating furnace	
Steam from steel plant	6 MW for pre-shift and SEWGS sections	0.62 t steam
Cold utility <sup>d</sup>	Cooling water in closed loops	0 t
Combustion of H <sub>2</sub> stream for heat <sup>a,d</sup>	Complete oxidation	0.02 t CO <sub>2</sub>
Combustion of H <sub>2</sub> stream <sup>a,d</sup>	Energy output	4.63 GJ
Consumables <sup>d,e,f</sup>	Given as the sum of all	6.15E-4 t
Waste <sup>g</sup>	To incinerator	6.15E-4 t
CO <sub>2</sub> injection mass <sup>a,d</sup>	CO <sub>2</sub> stream at 110 bar of which 98% is CO <sub>2</sub>	1.06 t CO <sub>2</sub> stream (1.03 t CO <sub>2</sub> )
CO <sub>2</sub> transport and injection loss <sup>j</sup>	Transport distance 800 km; pipeline loss 0.026%/1000 km; injection loss 0.1%	5.9E-7 t H <sub>2</sub> 7.7E-6 t CO 0.001 t CO <sub>2</sub>
CO <sub>2</sub> injection electricity <sup>j</sup>	7 kWh/t CO <sub>2</sub>	7.4 kWh
<b>Counterfactual Application 5: Heat from natural gas</b>		
Heat for internal use in steel mill <sup>b</sup>	From European market group (RER) for heat for industrial use	4.63 GJ
<b>Application 6: Flaring</b>		
Emissions from flaring <sup>a</sup>	Assumed complete oxidation	1.09 t CO <sub>2</sub>
<b>Counterfactual Application 6: None</b>		

<sup>a</sup> see for composition Supplementary Table 3.1

<sup>b</sup> based on Ecoinvent database v3 (Wernet et al. 2016), based on OECD statistics for heat and electricity supply

<sup>c</sup> efficiency of power plants using steel off-gases as fuel based on (IEAGHG 2013)

<sup>d</sup> based on industrial data and process design modelled in ASPEN Plus and Matlab (Boon et al. 2015, 2016), and see Supplementary Information Section 3.1

<sup>e</sup> based on catalyst data (Twigg 1989)

<sup>f</sup> Supplementary Table 3.2 contains details on types of catalysts used for each of the process steps

<sup>g</sup> process in Ecoinvent database v3 (Wernet et al. 2016)

<sup>h</sup> based on Best Available Techniques for ammonia production (van Dijk et al. 2010)

<sup>i</sup> based on Best Available Techniques for urea production (EFMA 2000b)

<sup>j</sup> based on Chisalita et al. (2019)

### ***Life cycle impact assessment***

The ReCiPe2016 method (H) v1.1 was used at midpoint level to compare the GHG emissions associated with the different BOFG applications, and at endpoint level to quantify the environmental impacts of the BOFG applications (Huijbregts et al. 2017). ReCiPe2016 distinguishes three areas of protection, namely human health, ecosystems and resources. We focused on damage to human health and damage to ecosystems as we investigated the impacts from a residual flow, hence the direct damage to resource availability from BOFG is not relevant. Damage to human health is expressed in terms of disability-adjusted life years (DALY); damage to ecosystem quality in terms of species loss over time (species.yr) (Huijbregts et al. 2017). The software SimaPro v8.5 was used for all calculations (Pré Consultants 2017).

### ***Interpretation***

For each of the applications of BOFG and for each respective counterfactuals, the GHG emissions were calculated in tonnes of CO<sub>2</sub>-equivalents per tonne of BOFG. This was done for the process steps such that the steps with the largest contribution could be identified. A contribution analysis was performed to show the contributions to the environmental impacts of each application, and to identify the main contributing process steps. To assess the effect of assumptions of the input parameters on the results, sensitivity analyses were performed on parameters affecting the main contributing process steps.

*Electricity and natural gas mix.* Each application of BOFG and its counterfactual was assumed to be located in Europe. The average European electricity market mix (ENTSO-E) from EcoInvent was used for all electricity in the process steps as well as for the counterfactual electricity. The same applied to the natural gas that was used for urea production. To test how sensitive the results are to the choice of electricity mix and natural gas source, the GHG emissions for electricity production and urea production were quantified for five different countries in Europe (Netherlands, Spain, Belgium, France and Sweden), each with a different electricity mix in terms of shares of technologies and GHG intensity (see Supplementary Table 3.3). Likewise, the country-specific source of natural gas was used. This sensitivity analysis can also be regarded as a projection of BOFG use into the future when the overall

European electricity mix is expected to decarbonise, represented by the current Swedish mix.

*Captive power plant efficiency.* The effect of the assumed efficiency of the captive power plant was analysed. The assumed efficiency of 32.1% was based on the average Western European power plant using steelworks off-gases as fuel, described in an IEA GHG report on iron and steel (IEAGHG 2013). The efficiency of a power plant can vary between approximately 25% and 40% depending on when it was built. Therefore, the GHG emissions and environmental impacts for the electricity application were also calculated for an efficiency of 25% and 40%.

*Source of heat.* In the BOFG-based urea production process, waste heat from the steel mill was assumed to be available. In this sensitivity analysis we assess how the results change if instead the heat demand is met by combustion of excess blast furnace gas (BFG), another steel off-gas. The LCA again uses system expansion rules because the available BFG is normally used for electricity production. The net environmental impacts and GHG emissions of BFG-based heat can then be calculated by: i) taking the impacts and emissions of BFG-based heat production, ii) subtracting impacts and emissions of BFG-based electricity production, and iii) adding the impacts and emissions of the same amount of grid electricity. The emissions and impacts of combustion of BFG are again assumed to be the same for heat production or electricity production; hence effectively we only add the impacts and emissions of the grid electricity. This is calculated by assuming a power plant efficiency of 32.1%, as in section 3.2.1.

An efficiency of 88% is used for the conversion of heat from combusted BFG to steam, and due to the need for lower temperature steam, not the full lower heating value (LHV) of BFG of 3.49 MJ/Nm<sup>3</sup> can be used as not all the heat can be recovered. Hence for the 0.62 t steam/t<sub>BOFG</sub> demand for the processes upstream of the ammonia synthesis loop (see Table 3.1), there is a 6.8 MW BFG demand, equalling 0.48 kNm<sup>3</sup>/t<sub>BOFG</sub>. Furthermore, for the 0.39 t steam/t<sub>BOFG</sub> demand for the urea synthesis step, 0.32 kNm<sup>3</sup>/t<sub>BOFG</sub> of BFG is combusted. The associated electricity production from the total of 0.80 kNm<sup>3</sup>/t<sub>BOFG</sub> of BFG is 0.25 MWh/t<sub>BOFG</sub>. The European case and the Swedish case are used for the quantification of the difference in GHG emissions and impacts.

*Impurities associated with BOFG combustion.* In the applications where BOFG is combusted, full oxidation was assumed without any NO<sub>x</sub>, SO<sub>x</sub> and particulate matter impurities. Also for all other applications of BOFG these impurities were not considered. To test the effect of including NO<sub>x</sub>, SO<sub>x</sub> and particulate matter impurities on the total environmental impacts, IEA data on NO<sub>x</sub>, SO<sub>x</sub> and dust in flue gas from the Western European power plant using steelworks off-gases as fuel was used and added to the emissions from BOFG combustion for the energy application. These emissions were 60 mg/Nm<sup>3</sup> of NO<sub>x</sub>; 10 mg/Nm<sup>3</sup> of SO<sub>x</sub> and up to 5 mg/Nm<sup>3</sup> dust in the flue gas (IEAGHG 2013).

### 3.2.4 Plant-level environmental benefits

Different applications for BOFG discussed in section 3.2.1 can be used simultaneously in one steel mill, in varying shares. To investigate how urea production from BOFG in a steel plant could develop and to assess the environmental benefits of the choice for a certain application at the scale of a single plant, four different plant configurations with different shares of applications were composed (see Table 3.2).

**Table 3.2. Possible plant configurations and their respective shares of BOFG applications.**

BOFG application	'Current flared'	'Current modern'	'Medium term'	'Long term'
Heat		67%	67%	
Electricity		33%	8%	
Urea			25%	
Urea with CO <sub>2</sub> storage				100%
Flaring	100%			

The first configuration was the 'current flared' case where all BOFG is flared, which happens in some steel mills (Carpenter 2012). The second configuration was the 'current modern' case based on 2014 and 2017 data from a typical Western European steel mill, where approximately two-thirds of all BOFG produced in those years (4 PJ/yr or 7×10<sup>5</sup> t/yr) was used internally for heat while the excess of one third of all BOFG was sent to the power plant.

As the SEWGS technology is currently at TRL6 (Van Dijk et al. 2017) and needs to be scaled up to commercial use, the system design of section 3.2.2 was based on the smallest commercially viable capacity of urea production: 50 kta. This production capacity is considered to be within the range for commercial deployment of SEWGS in less than five years' time.

Based on the BOFG composition given in Supplementary Table 3.1, approximately 25% of all BOFG would be required for the production of 50 kta urea. Hence, this could be covered by the excess BOFG that would be sent to the power plant in the 'current modern' case and is not needed to meet the internal heat demand. The 'medium term' configuration was based on these shares, leaving a remainder of 8% of BOFG for electricity production.

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Ultimately, we are interested in the potential of using all BOFG from a steel mill for urea production in combination with storage of the excess CO<sub>2</sub>. This was therefore considered as the 'long term' configuration.

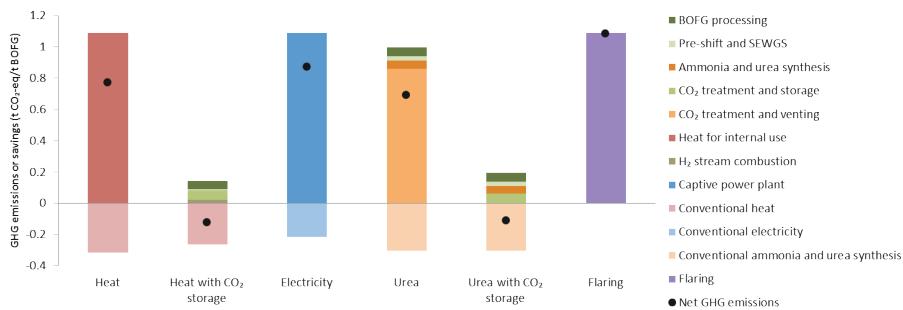
### 3.2.5 GHG emissions and environmental impacts per tonne of urea

Overall environmental impacts and GHG emissions of urea production from BOFG were also calculated per tonne of urea produced. This enabled explicit comparison with conventional, natural gas-based urea production. Upon first deployment, BOFG-based urea production will likely replace electricity production from BOFG. This BOFG-based electricity would have to be replaced by grid electricity. Following standard LCA system expansion rules, the net environmental impacts and GHG emissions of BOFG-based urea production can therefore be calculated by: i) taking the impacts and emissions of BOFG-based urea production, ii) subtracting impacts and emissions of BOFG-based electricity from those of BOFG-based urea production, and iii) adding the impacts and emissions of the same amount of grid electricity to those of BOFG-based urea.

## 3.3 Results

### 3.3.1 Greenhouse gas emissions

The GHG emissions and savings for the six different applications of BOFG are shown in Figure 3.3, in tonnes of CO<sub>2</sub>-eq per tonne of BOFG (tCO<sub>2</sub>-eq/t<sub>BOFG</sub>). The GHG emissions of the uses of 1 tonne of BOFG are shown as positive values while the GHG emissions of the corresponding counterfactuals are shown as negative values. The total GHG emissions are the sum of the emissions and savings associated with this application of BOFG.



**Figure 3.3. GHG emissions (positive values) or avoided emissions (negative values) for the use of 1 t BOFG for heat, heat with CO<sub>2</sub> storage, electricity production, urea production, urea production with CO<sub>2</sub> storage, and flaring. The black dots represent the net GHG emissions from the applications and the replaced counterfactuals.**

Out of the six BOFG applications and their counterfactuals, the GHG emissions are largest with 1.1 tCO<sub>2</sub>-eq/t<sub>BOFG</sub> when BOFG is flared as there is no counterfactual representing avoided GHG emissions. With 0.69 tCO<sub>2</sub>-eq/t<sub>BOFG</sub>, urea production has lower net GHG emissions than the energy applications of electricity with 0.87 tCO<sub>2</sub>-eq/t<sub>BOFG</sub> and heat with 0.77 tCO<sub>2</sub>-eq/t<sub>BOFG</sub>. The difference between the electricity and heat applications is due to a difference in counterfactual emissions; relatively larger emissions are associated with the replacement of heat by natural gas than the replacement of electricity by the European grid electricity, attributable to pipeline losses.

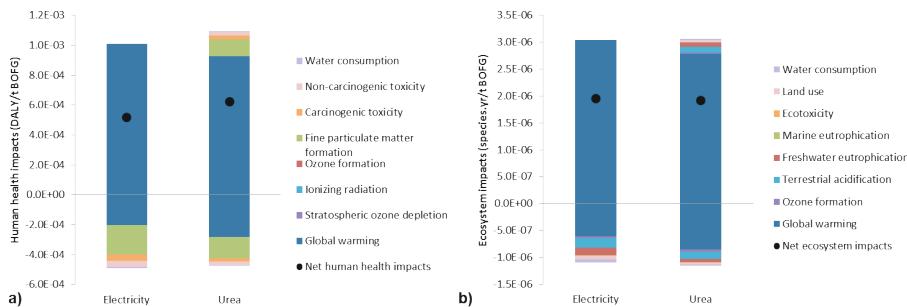
In the urea application, 20.6% of process CO<sub>2</sub> is used for urea production after separation from H<sub>2</sub> and N<sub>2</sub> from the BOFG stream with the SEWGS technology; the 79.4% excess CO<sub>2</sub> is vented, having a large contribution to the overall GHG emissions. When the excess CO<sub>2</sub> that cannot be used for urea production is transported and stored underground in the urea with CO<sub>2</sub> storage application, this results in an overall GHG benefit of -0.11 tCO<sub>2</sub>-eq/t<sub>BOFG</sub>. In the heat application combined with CO<sub>2</sub> storage, no CO<sub>2</sub> is utilised so the entire CO<sub>2</sub> stream after SEWGS is transported and stored. This results in a GHG benefit of -0.12 tCO<sub>2</sub>-eq/t<sub>BOFG</sub> compared to heat production from natural gas.

### 3.3.2 Environmental impacts

Figure 3.4 shows the contributions of the different impact categories to the total human health impacts and total ecosystem impacts that make up the electricity and urea production applications of BOFG respectively, expressed per tonne of BOFG. Avoided impacts by replacing the counterfactual are shown as negative impacts. We focus on these two applications as for the initial implementation scale of the SEWGS technology for urea production from BOFG, the situation will most likely be one in which BOFG that is currently used for electricity production would be substituted by urea production. Full results for heat, heat with CO<sub>2</sub> storage, urea with CO<sub>2</sub> storage and flaring are provided in Supplementary Figure 3.3 and Supplementary Figure 3.4.

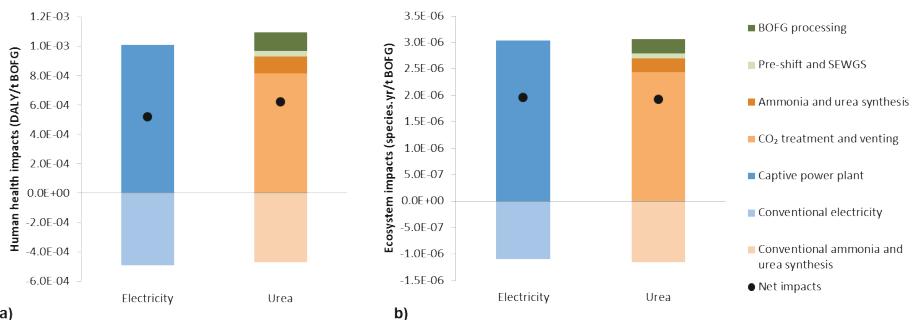
Figure 3.4 shows that the main impact category contributing to the overall human health impacts and ecosystem impacts is global warming for electricity as well as urea production from BOFG. The overall human health impacts are about 20% lower for electricity than urea, and the overall ecosystem impacts for the electricity and urea applications are comparable with less than 2% difference. Besides global warming, other impact categories do not contribute to human health impacts for the electricity application as it was assumed that when BOFG is combusted in the captive power plant, it is fully oxidised without emissions of impurities such as particulates. Similarly, no sulphur or nitrous oxide emissions are considered in all BOFG applications. For urea production, electricity from the grid is used in several of its processing steps, which causes impacts to fine particulate matter formation and non-carcinogenic and carcinogenic toxicity. As a result, the overall human

health impact of urea production is larger than that of electricity production. Important impact categories contributing to the overall ecosystem impacts are terrestrial acidification, freshwater eutrophication and land use impacts, also caused by the use of electricity from the grid.



**Figure 3.4. Human health impacts (a) and ecosystem impacts (b) for BOFG use for electricity and urea divided by contributing impact category.** Impacts associated with the BOFG application are shown as positive impacts, while impacts associated with the counterfactuals are shown as negative (avoided) impacts. The black dots represent the net impacts, the sum of the positive and negative impacts.

The contributions of the different process steps of electricity and urea production are shown in Figure 3.5. BOFG processing has no direct GHG emissions but has an electricity demand for the compressor that is used in this step. GHG emissions from BOFG combustion in the power plant and of the treatment, and venting of excess CO<sub>2</sub> in the electricity and urea applications respectively have the largest contribution to the overall damage to human health and ecosystems.

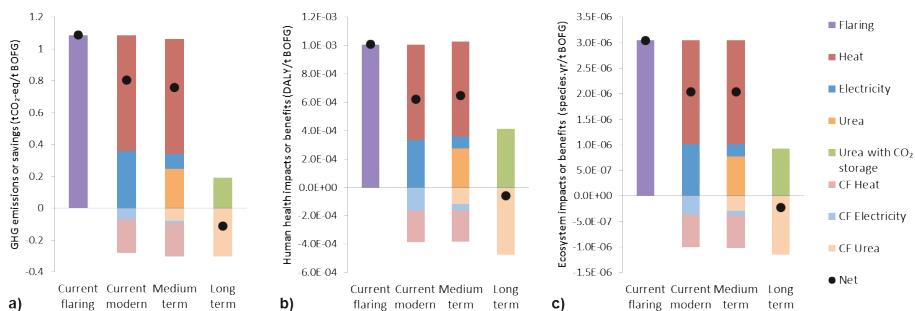


**Figure 3.5. Human health impacts (a) and ecosystem impacts (b) for BOFG use for electricity and urea divided by contributing process.** Impacts associated with the BOFG application are shown as positive impacts, while impacts associated with the counterfactuals are shown as negative (avoided) impacts. The black dots represent the net impacts, the sum of the positive and negative impacts.

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### 3.3.3 Plant-level environmental benefits

Figure 3.6 shows the GHG emissions, human health impacts and ecosystem impacts per tonne BOFG for the four plant configurations detailed in Table 3.2. The 'current flared' case where all BOFG is flared has the highest impact on all categories and the 'long term' case where all BOFG is used for urea production in combination with CO<sub>2</sub> storage results in avoided emissions or environmental benefits. The 'current modern' case has comparable impacts (0.81 tCO<sub>2</sub>-eq/t<sub>BOFG</sub>; 6.2E-4 DALY/t<sub>BOFG</sub>; 2.0E-6 species.yr/t<sub>BOFG</sub>), compared to the 'medium term' case (0.76 tCO<sub>2</sub>-eq/t<sub>BOFG</sub>; 6.5E-4 DALY/t<sub>BOFG</sub>; 2.0E-6 species.yr/t<sub>BOFG</sub>). There is a reduction of 6% for the 'medium term' case compared to the 'current modern' case in the GHG emissions category and an increase of 4% for overall human health impacts.

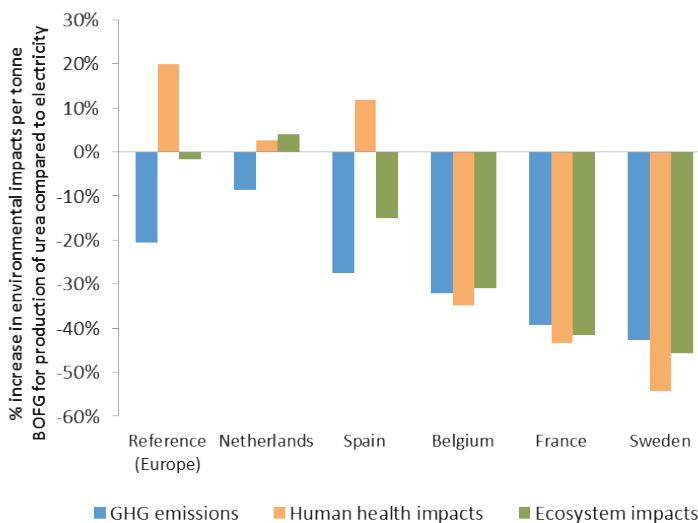


**Figure 3.6. GHG emissions (positive values) or avoided emissions (negative values) (a), human health impacts (positive values) or avoided impacts (negative values) (b) and ecosystem impacts (positive values) or avoided impacts (negative values) (c) for the use of 1 t BOFG for the four plant configurations detailed in Table 3.2.** The black dots represent the net emissions or impacts from the applications and the replaced counterfactuals (indicated as CF).

### 3.3.4 Sensitivity analyses

#### Electricity and natural gas mix

The GHG emissions for the electricity and urea applications were quantified for electricity mixes and natural gas sources of five different countries, i.e. the Netherlands, Spain, Belgium, France and Sweden. For the different locations, Figure 3.7 shows the percentage increase in GHG emissions of urea production compared to electricity production, determined by dividing the net GHG emissions per tonne of BOFG used for urea by the net GHG emissions per tonne of BOFG used for electricity. A negative value means that lower GHG emissions are associated with urea production from BOFG than electricity production: for the European reference case, the GHG emission reduction potential is 21%. The Dutch electricity mix has the largest GHG footprint and the Swedish electricity mix the smallest (see Supplementary Table 3.3). For Sweden, the GHG emission reduction potential is 43% for urea compared to electricity while for the Netherlands this is 9%, with the rest of the countries falling in between this range, in sequence of the size of the GHG footprint of their electricity. Irrespective of the GHG footprint of electricity in the country, GHG emissions for urea production from BOFG are lower than for electricity production.



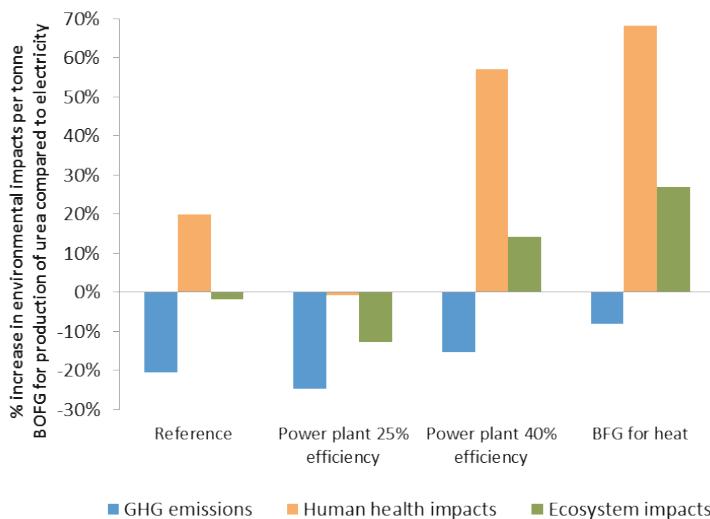
**Figure 3.7. Net greenhouse gas emissions, net human health impacts and net ecosystem impacts of urea production from 1t BOFG compared to electricity production from 1t BOFG for the Netherlands, Spain, Belgium, France, Sweden, and the reference case of Europe.**

Figure 3.7 also shows the human health and ecosystem impacts of urea compared to electricity for all locations. The increase in human health impacts for urea production compared to electricity production for Europe is larger than all other countries considered, followed by Spain and only then the Netherlands. Unlike the decrease in GHG emissions for urea compared to electricity production, this pattern cannot only be attributed to the electricity GHG footprint alone, being highest for the Netherlands, followed by Europe and then Spain (Supplementary Table 3.3). The cause of this is that particulate matter formation impacts associated with grid electricity are a factor three to four larger for Europe and Spain compared to the Netherlands. Particulate matter is associated with electricity from coal, and the difference in impacts results from an assumption on the coal origin in Ecoinvent. While for Netherlands, Belgium and France, the source of hard coal is 'Western Europe' (WEU), a process that assumes that 0% of coal is mined in opencast mine; for Spain and Sweden, the source of coal is 'Rest of the World' (RoW), where coal is also from opencast mines. A kg of coal from WEU causes only 30% of the RoW particulate matter emissions. For Sweden the share of coal is low (see Supplementary Table 3.3) and hence has little influence on the results. However, for Spain and the average European mix which is built up from mixes

from all European countries, including other countries for which the source of coal is RoW, this assumption results in net human health impacts being lower for electricity production from BOFG compared to urea production. Figure 3.7 shows that at locations with lower environmental impacts associated with their electricity production overall human health and ecosystem impacts are lower for urea production compared to electricity production (see Supplementary Table 3.3 for environmental impacts associated with the generation of 1 MWh of electricity in each country).

### ***Captive power plant efficiency***

To test the sensitivity of the results to the assumed efficiency of the captive power plant, the GHG emissions and environmental impacts were quantified for lower and higher efficiencies than the average efficiency of 32.1%: 25% and 40%. Only the amount of electricity that is produced and hence the amount of counterfactual-electricity changes. Lower power plant efficiency means smaller avoided GHG emissions from its counterfactual. GHG emissions, human health impacts and ecosystem impacts for BOFG used for urea production compared to electricity production decrease for an efficiency of 25% and increase for an efficiency of 40% compared to the reference case with 32.1%. For these different scenarios, Figure 3.8 shows the percentage increase in impacts of urea production compared to electricity production, determined by dividing the net impacts (GHG, human health and environmental) per tonne of BOFG used for urea by the net impacts per tonne of BOFG used for electricity. Figure 3.8 shows that for all assumed efficiencies, GHG emissions are lower for urea production than for electricity production. For an efficiency of 25%, urea outperforms electricity on all three categories while for an efficiency of 40%, the additional required counterfactual electricity results in a pronounced increase in overall human health and ecosystem impacts.



**Figure 3.8. Net greenhouse gas emissions, net human health impacts and net ecosystem impacts of urea production from 1t BOFG compared to electricity production from 1t BOFG** for a captive power plant efficiency of 25% and 40%, using blast furnace gas (BFG) as a source of heat, as well as the reference case using waste heat from the steel plant in urea production and a power plant efficiency of 32.1%.

### Sources of heat

When blast furnace gas (BFG) is combusted to meet the heat demand previously met by waste heat from the steel mill, the GHG emission reduction for urea production from BOFG compared to electricity production decreases from 21% to 8% (see Figure 3.8). Including the combustion of BFG results in overall environmental impacts being higher for urea production compared to electricity production (human health impacts are about 68% higher; ecosystems impacts are 27% higher). In the Swedish case (not shown in Figure 3.8), the reduction of GHG emissions, human health and ecosystem impacts decreases by only 2%-8% as the emissions associated with the electricity that is compensated are much lower.

### Impurities associated with BOFG combustion

While GHG emissions are not affected by including  $\text{NO}_x$ ,  $\text{SO}_x$  and particulate matter impurities from BOFG combustion, other impact categories are affected and show an increase of 1% to the overall human health and ecosystem impacts. For human health impacts the increase is caused by additional ozone and fine

particulate matter formation, while for ecosystem impacts the increase is caused by additional ozone formation and terrestrial acidification.

### 3.3.5 GHG emissions and environmental impacts per tonne of urea

For the functional unit of 1 tonne of urea produced instead of 1 tonne of BOFG utilised, we found that GHG emissions of urea production from BOFG were 0.42 tCO<sub>2</sub>-eq/t<sub>urea</sub>, accounting for the fact that urea would likely replace BOFG-based electricity. Conventional urea production in Europe causes emissions of 1.0 tCO<sub>2</sub>-eq/t<sub>urea</sub> based on our LCA, and falls within the range of reported values in the literature: For urea production in Europe, until the gate of the factory, Brentrup et al. (2016) reported a value of 0.89 tCO<sub>2</sub>-eq/t<sub>urea</sub>, as well as an up to threefold increase of this value for other world regions. Hasler et al. (2017) reported CO<sub>2</sub> emissions of 1.42 tCO<sub>2</sub>/t<sub>urea</sub> in Germany for urea production. The urea production process from the EcoInvent database causes 1.5 tCO<sub>2</sub>-eq/t<sub>urea</sub>, however as is noted by Brentrup et al. (2016), the process in the EcoInvent database is outdated due to improvements in process technologies and energy efficiency.

Assuming the value of 1.0 tCO<sub>2</sub>-eq/t<sub>urea</sub> for conventional urea production, using BOFG to produce urea instead of electricity would result in lower net GHG emissions than conventional urea production, reducing emissions by 0.61 tCO<sub>2</sub>-eq/t<sub>urea</sub>. The electricity mix affects the potential GHG emission reduction that can be achieved. Our results show that if instead of the European average, the Dutch grid electricity mix is used to compensate for electricity no longer produced from BOFG, the emission reduction would be 0.22 tCO<sub>2</sub>-eq/t<sub>urea</sub>; a reduction of 1.5 tCO<sub>2</sub>-eq/t<sub>urea</sub> could be achieved in the Swedish setting. This shows that the range of potential emission reductions from BOFG-urea is substantial and strongly depends on the electricity grid mix that is used to replace the electricity otherwise produced from BOFG. Looking at overall human health impacts and ecosystem impacts for urea production from BOFG in Europe (1.9E-3 DALY/t<sub>urea</sub>; 3.8E-6 species.yr/t<sub>urea</sub>) compared to conventional urea production (1.6E-3 DALY/t<sub>urea</sub>; 3.9E-6 species.yr/t<sub>urea</sub>), we see that the ecosystem impacts are comparable while human health impacts are higher for BOFG-based urea production. This is again primarily a result of venting the excess CO<sub>2</sub> after SEWGS.

## 3.4 Discussion

### 3.4.1 Interpretation of results

The application of BOFG for urea production compared to electricity production causes 21% lower net GHG emissions. The sensitivity analysis shows that these findings are robust: alternative locations (and associated grid mixes and natural gas sources), captive power plant efficiency, the accounting for impurities in BOFG and including BFG combustion to meet the heat demand influence GHG emissions, but not to the extent that the ranking of the investigated applications is affected.

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Urea production compared to electricity production from BOFG in Europe causes 2% lower ecosystem impacts, but 20% higher human health impacts due to an additional electricity demand for the process steps of urea production. The sensitivity analysis showed that these findings depend on the availability of waste heat for steam from the steel mill, as well as on the environmental impacts associated with grid electricity: for Belgium, France and Sweden the ecosystem impacts are between 31%-46% lower, while human health impacts are between 35%-54% lower for urea production compared to electricity production from BOFG. Particulate matter formation contributes for a large share to the human health impacts associated with the European grid mix resulting particularly from coal-fired power plants. The environmental benefits of producing urea rather than electricity from BOFG will become even larger when any replaced grid electricity is cleaner, as would be the case under future decarbonisation (Rogelj et al. 2018).

The observed strong influence of the electricity mix on the results is in line with current CCU discussions, where the attainable GHG emission reduction potential of replacing conventional chemical production by CCU was found to largely depend on the availability of low-emission electricity (Kätelhön et al. 2019). We found however, that BOFG-based urea caused the lowest GHG emissions of all investigated applications, regardless of the assumed electricity grid mix. This is because the amount of electricity required for urea synthesis is relatively limited, which can in turn be attributed to the fact that the use of carbon monoxide in the BOFG to urea process is thermodynamically favourable, as it is not a fully oxidised

feedstock. Including the counterfactual of each application enabled comparison of the different applications and a consequential estimate of the environmental impacts of the different applications in a way similar to previous studies (Pfau et al. 2019; Thonemann and Pizzol 2019).

On the level of a single plant, the results showed that in the near term, switching from a plant configuration that only produces heat and electricity from BOFG to a configuration that produces heat, 50 kta urea and less electricity does not result in the GHG emission reduction that would be required for an industrial transformation in line with the Paris targets. However, it does show potential for the long term, if all BOFG is ultimately used for urea production on a larger scale combined with CO<sub>2</sub> storage.

When urea is used as a fertiliser on agricultural land, the ammonia decomposes to supply nitrogen to the crop, while the CO<sub>2</sub> that was initially stored in the urea, is released. This release of CO<sub>2</sub> occurs irrespective of the path of urea production. The important contribution of this technology in limiting global warming is hence not that the CO<sub>2</sub> is stored permanently, but that a residual stream is used for urea production, preventing the extraction of natural gas as a feedstock, while also reducing emissions compared to using the residual stream for other applications.

### 3.4.2 Limitations and assumptions

Due to data limitations, various assumptions were made. First, it was assumed that per tonne of BOFG used in the urea application, the same amount of steam and electricity would be needed, regardless of whether urea was produced at the smallest commercial scale (50 kta) using only excess BOFG or at the production scale that would use all available BOFG from a typical steel mill (200 kta). This means that linear upscaling was assumed, which seems to hold if infrastructure is not considered, as the system design would be the same for both scales. When the production scales further increase and the system design changes, this assumption would no longer hold.

Second, under normal operation of the steel mill the quality and composition of the off-gases varies (see Supplementary Information Section 3.1 for a detailed

description of this process). Based on the 2018 first quarter BOFG composition data, the ratio of H<sub>2</sub>:N<sub>2</sub> was established to vary between 0.4 and 10, with an average of 2.4. Supplementary Figure 3.1 shows the variation in BOFG composition and resulting variation in H<sub>2</sub>:N<sub>2</sub> ratio for a one month time period. The system design was based on this observed, less than optimal H<sub>2</sub>:N<sub>2</sub> ratio of 2.4, which is both representative as well as conservative. The system has an inherent capacity to absorb limited variations of H<sub>2</sub>:N<sub>2</sub> without affecting the overall ammonia yield to a large extent (Liu et al. 2002). The larger changes that occur for short durations are expected to be absorbed due to the thermal inertia of the WGS and SEWGS systems, which work as buffers, minimising the propagation of these fluctuations to the ammonia reactor (van Dijk et al. 2010). Energy demand was also estimated based on the average BOFG composition. However, as long as the system has not yet been implemented on full scale it is not possible to evaluate to what extent these estimates correspond to actual use.

Third, data on plant infrastructure were limited. The environmental impacts of the plant infrastructure itself were therefore not included in the present LCA. Including infrastructure would lead to an increase of impacts for each of the applications, although not to the same extent for each of the applications. Based on a previous study (Koornneef et al. 2008) in which the contribution of infrastructure for CO<sub>2</sub> capture as well as transport and storage was quantified to be 0.3% of the total of GHG emissions for electricity production, the effect of infrastructure emissions on the results is expected not to change, especially since differences in GHG footprints of the various applications are larger than 0.3%.

Another assumption we made is that BOFG is a residual stream for which no upstream impacts are considered. This approach has been used before for considering alternative uses of the waste product of residual biomass from landscape management (Pfau et al. 2019) and in comparing alternative CO<sub>2</sub> conversion technologies starting with CO<sub>2</sub> from a flue gas or other source (Thonemann and Pizzol 2019). It could be argued that as BOFG has some (economic) value, economic allocation should be used to attribute part of the impacts from steel production to BOFG. Economic allocation suggests that for a production capacity of 50 kta urea, 1-2% of the impacts of the production of steel would be allocated to BOFG. Although this would mean a small increase in net

GHG emissions, the increase would be the same for all applications, not affecting the comparison of the different applications.

Similarly, it was assumed that steam from waste heat was available from the steel mill and no impacts from steel production were allocated to this steam. Based on economic allocation, the impacts of one tonne of steam were calculated to correspond to less than 0.1% of the impacts of 1 tonne of steel produced and were therefore not considered.

### 3.4.3 Recommendations

The recommendations following from this work can be separated into recommendations for future life cycle assessments and recommendations for further development of the technology to produce urea from BOFG.

To perform the LCA, we followed the guidelines prepared by the Global CO<sub>2</sub> Initiative (2018) as well as possible. However, we chose to use a different functional unit than advised in the guidelines and a different impact assessment method. We recommend that the guidelines prepared by the Global CO<sub>2</sub> Initiative are updated to include an option to define the functional unit in terms of a waste or residual stream used by a CCU technology. Currently, the guidelines advise the functional unit to be defined in terms of the CCU product (mass) or fuel (energy content). However, this does not allow for a comparison between different uses of a waste stream as this would result in different functional units. Defining the functional unit in terms of the waste stream (mass) accompanied by its counterfactual circumvents this problem. Furthermore, we chose to use one of the most recent LCIA methods for our assessment, instead of the recommended method that does not include the latest scientific developments. We recommend that the guidelines advise the use of an up to date impact assessment method, such as ReCiPe2016, which takes the latest insights into account (Dekker et al. 2019).

For further technology development, LCA can be a useful tool. Since preliminary LCA results have shown that electricity demand represented a substantive contribution to the overall impacts, an iterative step was used to adapt the design already in the process design of urea production from BOFG within the BOF2UREA

project. The electrode boiler that was used in the first design to raise the steam was replaced by further integration and utilisation of heat from off-gases from the ammonia and urea steps. This lowered electricity demand and therefore the impacts. Similarly, water demand was reduced by closing the cooling water loop, although it results in larger capital investment. With this, the use of LCA in early-stage design was shown during the project.

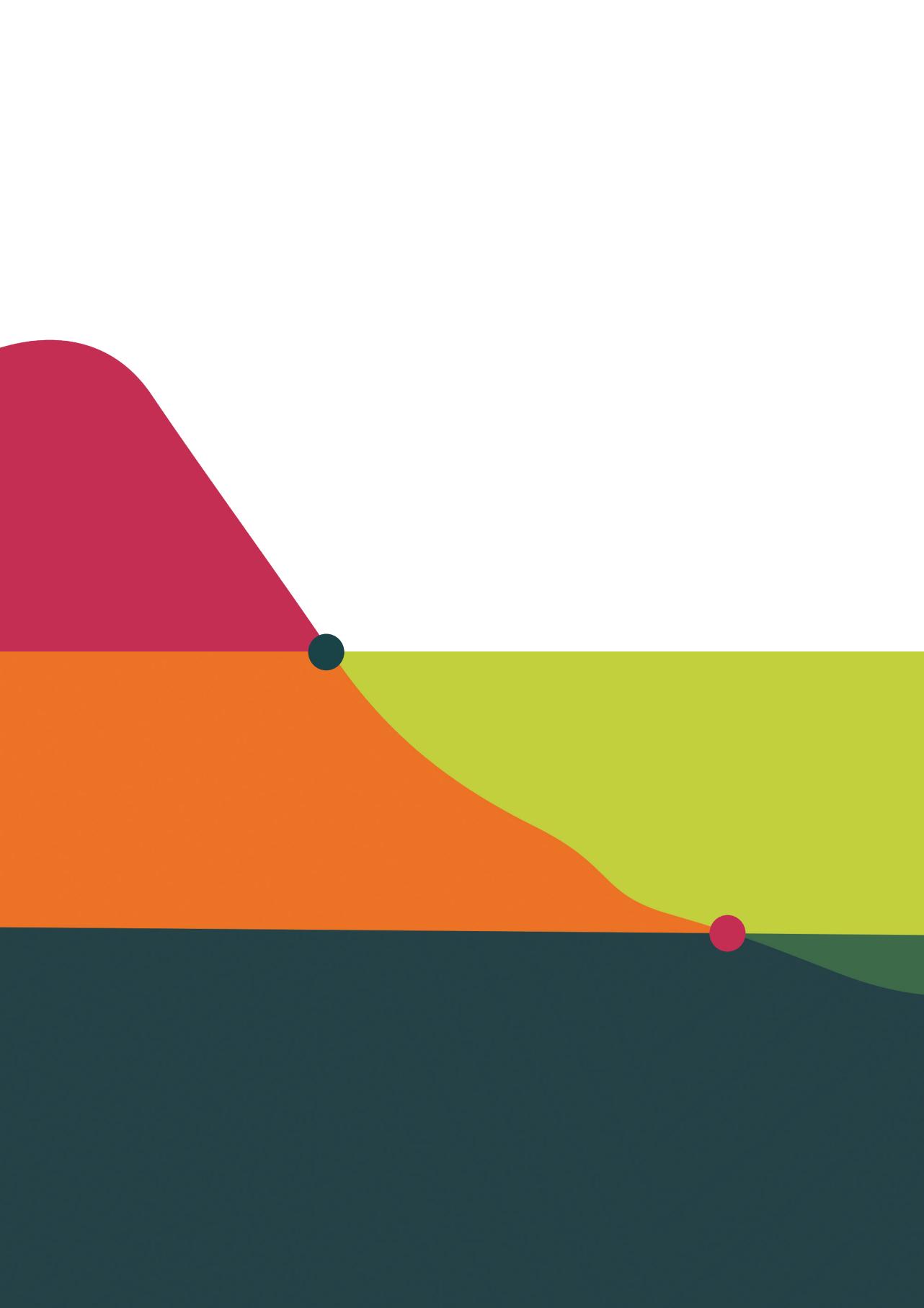
The process of using BOFG for urea production could be further improved by optimising the effective hydrogen utilisation. Currently, hydrogen coming into the plant is in the form of H<sub>2</sub> and CO that can be converted to H<sub>2</sub>. As the stream makes its way through the process, losses occur at various steps. Three major losses are from: incomplete conversion of CO to H<sub>2</sub>, consumption of H<sub>2</sub> in methanation and loss of H<sub>2</sub> in the ammonia loop purge. H<sub>2</sub> and CO lost in any of these steps is hydrogen potential that is not used for ammonia production. Optimising to reduce these losses would enable higher quantities of ammonia to be produced and allow for a larger share of CO<sub>2</sub> to be used instead of venting it, thereby reducing GHG emissions and environmental impacts. Furthermore, future improvements are foreseen due to technological learning and industrial scaling (Piccinno et al. 2016), as for example shown in (Caduff et al. 2012).

SEWGS proves an interesting technology for adsorbing CO<sub>2</sub> and producing a stream of N<sub>2</sub> and H<sub>2</sub> for ammonia production and urea production from a residual gas from the steel industry (Van Dijk et al. 2017), as BOFG-based urea production leads to lower GHG emissions than current BOFG uses and conventional urea production. We showed that venting the highly concentrated stream of excess CO<sub>2</sub> has a large impact on total GHG emissions and overall environmental impacts and that capturing and storing this CO<sub>2</sub> strongly reduces environmental impacts. An important focus for improving the BOFG to urea process would thus be to prevent venting and instead store the CO<sub>2</sub> underground, or utilising it in another way that prevents release to the atmosphere (Thonemann and Pizzol 2019).

## 3.5 Conclusions

We conclude the following: the use of a waste-utilisation focused functional unit (1 tonne of BOFG utilised) in combination with counterfactuals allows for the comparison of greenhouse gas balances and environmental impacts of different applications of a waste stream or residual flow. This comparison helps identify the best application of BOFG to reduce environmental impacts. Urea production from BOFG, making use of the sorption enhanced water-gas shift technology, avoids greenhouse gas emissions compared to other applications of BOFG, particularly when the excess of highly concentrated CO<sub>2</sub> is transported and stored. Compared to their counterfactual of conventional production, net emission savings can be achieved for urea and heat production when combined with CO<sub>2</sub> transport and storage. In the average European case, the overall environmental impacts are slightly lower for electricity production from BOFG than for urea production. However, sensitivity analysis showed that for locations with a cleaner electricity mix, overall environmental impacts are lower for urea production compared to electricity production. This implies that the relative environmental benefits of urea produced from BOFG are further enhanced under future power decarbonisation. Hence, out of the investigated options, urea production is the most promising use of BOFG to reduce environmental impacts, which when combined with CO<sub>2</sub> storage, can lead to net environmental benefits.





# The many greenhouse gas footprints of green hydrogen

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4

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## Abstract

Green hydrogen could contribute to climate change mitigation, but its greenhouse gas footprint varies with electricity source and allocation choices. Using life cycle assessment we conclude that if electricity comes from additional renewable capacity, green hydrogen outperforms fossil-based hydrogen. In the short run, alternative uses of renewable electricity likely achieve greater emission reductions.

## 4.1 Introduction

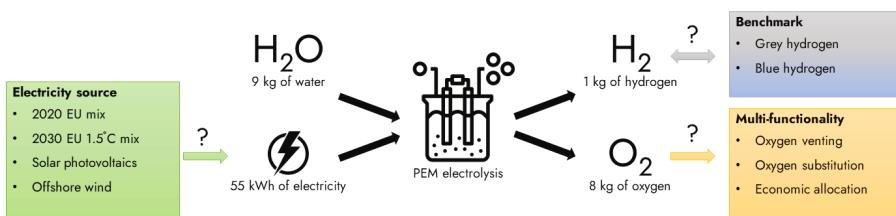
Hydrogen is widely considered both a promising industrial feedstock and a potentially important future energy carrier in the context of decarbonisation (IEA 2019c; European Commission 2020; Mac Dowell et al. 2021). The current global hydrogen demand of 75 million tonnes per year is predominantly met with so-called 'grey' hydrogen, produced from natural gas (IEA 2020). Alternative lower-emission hydrogen production methods have led to a colourful palette of hydrogen types, including 'blue' hydrogen produced from natural gas combined with carbon capture and storage, and 'green' hydrogen produced via water electrolysis using renewable electricity. In the IEA Sustainable Development Scenario (IEA 2020), these methods of hydrogen production are projected to scale-up towards 200 million tonnes of blue and 300 million tonnes of green hydrogen by 2070 (IEA 2020). Hydrogen can also be produced from biomass (Antonini et al. 2021), though this route is currently less technologically mature, depends on the availability of biomass against competing uses, and is more costly compared to green and blue hydrogen, which are therefore the focus for low-emission hydrogen production (IEA 2019c).

The estimated climate impacts of the various hydrogen production routes vary widely in the life cycle assessment (LCA) literature, which may confuse the best course of action on low-emission hydrogen for policymakers, investors and consumers. Blue hydrogen was recently found to reduce greenhouse gas (GHG) emissions compared to grey hydrogen by 5–36% (Howarth and Jacobson 2021), while a different set of assumptions for upstream methane leakage and carbon capture rates leads to a reduction of 26–75% compared to grey hydrogen (Bauer et al. 2022a). The source of electricity causes large variations in the GHG footprint of electrolytic hydrogen (Zhang et al. 2017; Bareiß et al. 2019; IEA 2019c; Valente et al. 2020; Mac Dowell et al. 2021; Bauer et al. 2022a) with up to 200% difference (i.e., the absolute difference divided by the average), as does the 'multi-functionality' question of how to allocate GHG emissions between hydrogen and co-produced oxygen (158% difference; Bargiacchi et al. 2020). GHG footprints of green hydrogen specifically, vary as a result of the use of different renewable electricity sources (wind or solar photovoltaics): 102–120% difference (Zhang et al. 2017), different

electrolysis technologies (alkaline electrolysis or polymer electrolyte membrane electrolysis: 16–40% difference (Zhang et al. 2017), and various assumptions on future improvements (increased efficiency and lifespan): 18% difference (Valente et al. 2020). The wide range in GHG footprints of green hydrogen warrants additional understanding of how these footprints are assessed, how they have come to diverge and what is required to lower them.

Of specific concern for green hydrogen is the principle of additionality (European Commission 2022c), which refers to producing green hydrogen using only newly installed, *additional*, renewable electricity capacity that matches the increased demand from electrolyzers (thereby preventing additional fossil-based electricity generation). The relevance of additionality is illustrated by the European Commission's 2020 hydrogen strategy that foresees a green hydrogen production of 10 million tonnes by 2030 (European Commission 2020), which would require 140% of the 394 TWh of electricity generated by all wind turbines in the European Union (EU) in 2020 (Eurostat 2021) at an electricity requirement of 55 kWh per kg of hydrogen (Bareiß et al. 2019). Realising these green hydrogen targets implies that renewable electricity generation needs to be increased or diverted from other uses.

Here, we evaluate how the GHG footprint of green hydrogen depends on three choices: i) the (future) electricity source; ii) the multi-functionality approach; and iii) the grey or blue hydrogen benchmark against which the emissions are compared (Figure 4.1). We focus on polymer electrolyte membrane (PEM) electrolyzers as the new generation, which is more efficient (Zhang et al. 2017) and less material-intensive (Bareiß et al. 2019) compared to the more mature alkaline electrolyzers. To illustrate how the combinations of different choices would result in different policy conclusions, we calculate the GHG footprint of electrolytic hydrogen produced in the EU for all combinations of these choices, based on the LCA ISO 14044 guideline. Furthermore, we assess how the use of additional renewable electricity for hydrogen compares to alternative uses of this electricity in terms of climate change benefits. Last, we suggest a way forward to comprehensively inform decisions of consumers, investors and policymakers based on ranges in GHG footprinting studies.



**Figure 4.1. Production scheme and choices for the electricity source, multi-functionality approach (i.e., method of accounting for co-products) and benchmark for electrolytic hydrogen serving as the basis for our LCA.** 1 kg of hydrogen output is associated with 8 kg of oxygen output, requires 55 kWh of electricity including compression, 9 kg of ultrapure water and infrastructure for the polymer electrolyte membrane (PEM) electrolyser stack and balance of plant (Bareiß et al. 2019).

## 4.2 Electricity source

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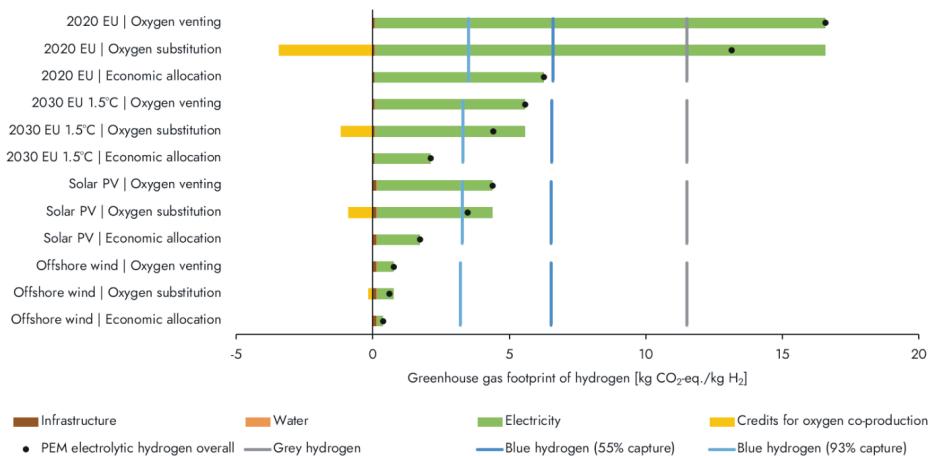
Green hydrogen is by definition produced through electrolysis using renewable electricity. Still, green hydrogen's GHG footprints vary depending on the exact renewable electricity source and on whether or not *additional* renewable capacity is used for hydrogen production. With regard to the latter, we argue that since electrolyzers cannot be considered to solely use the green component in the grid mix, they essentially run on the average grid mix unless additionality is guaranteed.

Figure 4.2 shows the GHG footprint of green hydrogen produced with different renewables (wind or solar) and current and future average grid electricity, calculated based on the life cycle inventory in Bareiß et al. (2019) (see Supplementary Information Section 4.1). Hydrogen powered by offshore wind (GHG footprints of 0.4–0.8 kgCO<sub>2</sub>-eq/kg H<sub>2</sub>) results in a twentyfold (93–97%) reduction in GHG footprint compared to grey hydrogen and a reduction of 76–94% compared to blue hydrogen. These results are comparable to the value of 0.7 kgCO<sub>2</sub>-eq/kg H<sub>2</sub> for wind-based hydrogen (with oxygen venting) reported by Mac Dowell et al. (2021), who included additional GHG emissions for hydrogen storage.

Using solar PV for hydrogen production leads to a GHG footprint of 1.7–4.4 kgCO<sub>2</sub>-eq/kg H<sub>2</sub> and equates to a 62–85% reduction compared to grey hydrogen and is in the same range as blue hydrogen (34% increase to 73% reduction in GHG emissions compared to blue hydrogen; see Figure 4.2). The GHG footprint of hydrogen

produced with solar PV electricity is approximately five times larger compared to offshore wind-based hydrogen, illustrating the large differences across different renewable electricity sources. The GHG intensity of renewable electricity is also location-dependent (Bosmans et al. 2021), which in turn affects hydrogen produced with it. Still, in absolute terms GHG footprints of green hydrogen are clearly low if based on renewables, and may further decrease in the future if higher electrolysis efficiencies can be obtained (Zhang et al. 2017; Valente et al. 2020).

Using the 2020 EU grid mix, for the case of non-additionality, electrolytic hydrogen has a GHG footprint of 6.3–16.6 kgCO<sub>2</sub>-eq/kg H<sub>2</sub>, which is in most cases higher than grey hydrogen (Figure 4.2). A cleaner 2030 grid mix (compatible with the EU targets for limiting warming to 1.5°C), results in a lower, but still sizable GHG footprint (2.1–5.6 kgCO<sub>2</sub>-eq/kg H<sub>2</sub>) in the range of blue hydrogen.

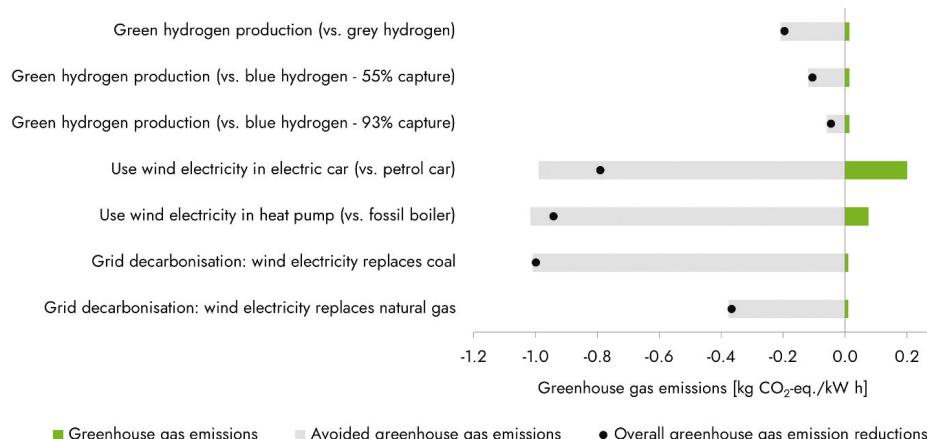


**Figure 4.2. The greenhouse gas footprint of PEM electrolytic hydrogen for different electricity sources and multi-functionality approaches in kg CO<sub>2</sub>-equivalents (kgCO<sub>2</sub>-eq) per kg H<sub>2</sub>.** The benchmarks grey and blue hydrogen (the latter with a CO<sub>2</sub> capture rate of 55% or 93%) are based on Bauer et al. (2022a) and were harmonised for each electricity source. The 2020 EU electricity mix and the 2030 mix for a 1.5°C-consistent emissions pathway are based on the integrated assessment model REMIND (Layritz et al. 2021); solar PV located in Europe was based on Bosmans et al. (2021); offshore wind was based on Bonou et al. (2016). Infrastructure emissions are based on 3000 full-load hours for the purely renewable sources while on 8000 full-load hours for the grid mixes. LCA inputs were based on Bareiß et al. (2019) and background life cycle inventory data on the Ecoinvent database version v3.7.1, using allocation at point of substitution. The ReCiPe2016 method (H) v1.05 was used at midpoint level to quantify the greenhouse gas footprints. Details on how we applied different methods to deal with multi-functionality can be found in Supplementary Information Section 4.1. The contribution of oxygen is shown as having a negative greenhouse gas footprint because it is a prevented emission; it does not signify carbon dioxide removal.

## 4.3 Alternative uses of new renewable electricity

While green hydrogen has low GHG footprints when based on additional renewables, the question remains whether green hydrogen production is the best way to use renewable electricity sources. From a climate perspective, it has been argued that priority should be given to technologies with the highest emission reduction potential per kWh of renewable electricity (Sternberg and Bardow 2015; Kätelhön et al. 2019). Figure 4.3 shows the emission reduction potentials for the use of 1 kWh of newly built offshore wind power for green hydrogen production replacing either grey or blue hydrogen, for electric cars replacing

petrol cars (Knobloch et al. 2020), heat pumps replacing fossil boilers (Knobloch et al. 2020), or directly decarbonising the existing grid mix by replacing coal or natural gas electricity (Hertwich et al. 2015). Here we see that green hydrogen production replacing grey hydrogen production results in approximately 2–5 times smaller emission reduction per kWh of renewable electricity compared to the competing uses. In the short run, while the grid can still be further decarbonised and electrification of heat and transport is still in progress, these applications of renewable electricity may take priority over green hydrogen production, if climate benefits are to be maximised. Only when (local) renewable electricity demand for the alternatives has been met, would green hydrogen production be effective in contributing to emission reductions. An important advantage of green hydrogen production from fluctuating renewable electricity compared to the alternatives is that it can be produced, when combined with hydrogen storage, when demand for renewable electricity elsewhere is low. This leaves opportunities for hydrogen production in areas with large renewable energy potentials, including renewable capacity without grid connection (Kätelhön et al. 2019).



**Figure 4.3. Greenhouse gas emissions and avoided emissions for different ways of using 1 kWh produced from newly built offshore wind capacity in kg CO<sub>2</sub>-equivalents (kgCO<sub>2</sub>-eq) per kWh.** The values for green and blue hydrogen are the same as in Figure 4.2, under the assumption that oxygen is vented. Emissions and avoided emissions for electric vehicles and heat pumps in the EU are calculated based on Knobloch et al. (2020), and for grid decarbonisation based on Hertwich et al. (2015). Details on how we calculated the greenhouse gas emissions and avoided emissions can be found in the Supplementary Information. Negative values are shown for avoided emissions and emission reductions, and do not signify carbon dioxide removal.

## 4.4 Multi-functionality

Oxygen that is co-produced in water electrolysis is typically vented to the air (Bareiß et al. 2019), meaning that all GHG emissions are allocated to the hydrogen produced. Alternatively, oxygen can be further purified for use in a subsequent process. Already in 2005, options to utilise the co-produced oxygen were explored as a way to bring down the relative economic costs of electrolytic hydrogen (Kato et al. 2005), and recently this has attracted renewed interest (Berenschot 2019). Analogously, the valorisation of co-produced oxygen has been included in footprinting studies (e.g, Thonemann and Pizzol 2019; Bargiacchi et al. 2020) in which part of the emissions of the hydrogen production process are allocated to the utilised oxygen. The ISO standards for LCA (ISO 14044) provide a hierarchy of methods to deal with this division, but leave freedom for the practitioner to choose an approach. When using a 'system expansion via substitution' approach it is assumed that co-produced oxygen replaces conventional oxygen production via air separation elsewhere (details can be found in Supplementary Information Section 4.1). As shown in Figure 4.2, this results in a considerably lower net GHG footprint for hydrogen compared to when oxygen is vented.

A second option to deal with multi-functionality is allocation of emissions based on economic value. Using recent market prices for hydrogen and oxygen to represent their economic value, the GHG footprint of electrolytic hydrogen under economic allocation is approximately halved compared to substitution (Figure 4.2). This leads to different conclusions on whether electrolytic hydrogen reduces emissions compared to benchmark blue or grey hydrogen.

When considering electrolysis at scale, large quantities of co-produced oxygen could lead to a market saturated for oxygen. Comparing the estimated European oxygen market of 17 million tonnes per year (Wernet et al. 2016) to the 80 million tonnes of oxygen that would be co-produced with the 10 million tonne green hydrogen target for 2030 (European Commission 2020), it is clear that there would likely be an oxygen surplus, even if the market for oxygen grows. This would make continued substitution of conventional oxygen production unrealistic and would diminish the share of emissions that can be economically allocated to oxygen.

Beyond oxygen market saturation, virtually all emissions can be assigned to hydrogen, leaving oxygen venting as both a conservative but also realistic default option.

## 4.5 Benchmark

The emission reduction potential of deploying electrolytic hydrogen depends on the benchmark against which it is assessed. Electrolytic hydrogen could first replace existing uses of grey hydrogen e.g. as feedstock in the chemical industry, in fertiliser production, as reducing agent in the steel industry, or as transport fuel (IEA 2019c). However, the benchmark could shift over time: grey hydrogen production facilities can be retrofitted to include carbon capture and storage (CCS) leading to blue hydrogen production, either capturing CO<sub>2</sub> from the steam methane reforming process only (55% capture rate), or also from the flue gas (93% capture rate). The GHG intensity of electricity used in green hydrogen production has to be sufficiently low to outperform this new benchmark of blue hydrogen; the cut-off is approximately 58 gCO<sub>2</sub>-eq/kWh (assuming oxygen is vented for green hydrogen and a 93% carbon capture rate for blue hydrogen), which is lower than the median value for solar PV electricity in Europe. The benchmark for (green) hydrogen application in industry and transport, i.e., what are realistically substituted products, could also shift over time, warranting careful interpretation of green hydrogen's long term emission reduction potential. A potential additional concern for all hydrogen pathways is leakage of hydrogen, which may increase its GHG footprint; research on this has only recently started (Ocko and Hamburg 2022).

In the context of green hydrogen's comparison against grey hydrogen, there is a technical, but critical concern that the often used Ecoinvent LCA database erroneously reports a fossil hydrogen GHG footprint that is five times lower (2.2 kgCO<sub>2</sub>-eq/kg H<sub>2</sub>) compared to the well-established footprint in literature of around 11.5 kgCO<sub>2</sub>-eq/kg H<sub>2</sub> for grey hydrogen (Bareiß et al. 2019; Bauer et al. 2022a) (details are provided in the Supplementary Information Section 4.4; the misrepresentation has been communicated to the database developers and a new process for grey hydrogen production will be added in the next version). The use of this Ecoinvent

value in LCA studies has led to incorrectly low GHG footprints (e.g., Thonemann and Pizzol 2019; Rumayor et al. 2019).

## 4.6 Implications

Our results illustrate that, depending on the choices in the calculation of the GHG footprint of green hydrogen (electricity source, multi-functionality and benchmark), green hydrogen production in the EU can achieve no, a small, or a large emission reduction. This dependency extends to many other products that are electricity-intensive or involve co-products, including the electrification of transport and the production of metals, as their footprints depend on the same choices. These accounting choices also interact: with a large electricity GHG footprint like the 2020 EU electricity mix, the allocation choice has a large effect on whether or not electrolytic hydrogen reduces emissions compared to the benchmark. Conversely, for a small electricity GHG footprint like for wind power, no matter which multi-functionality approach is used, the GHG footprint is lower than of the benchmarks.

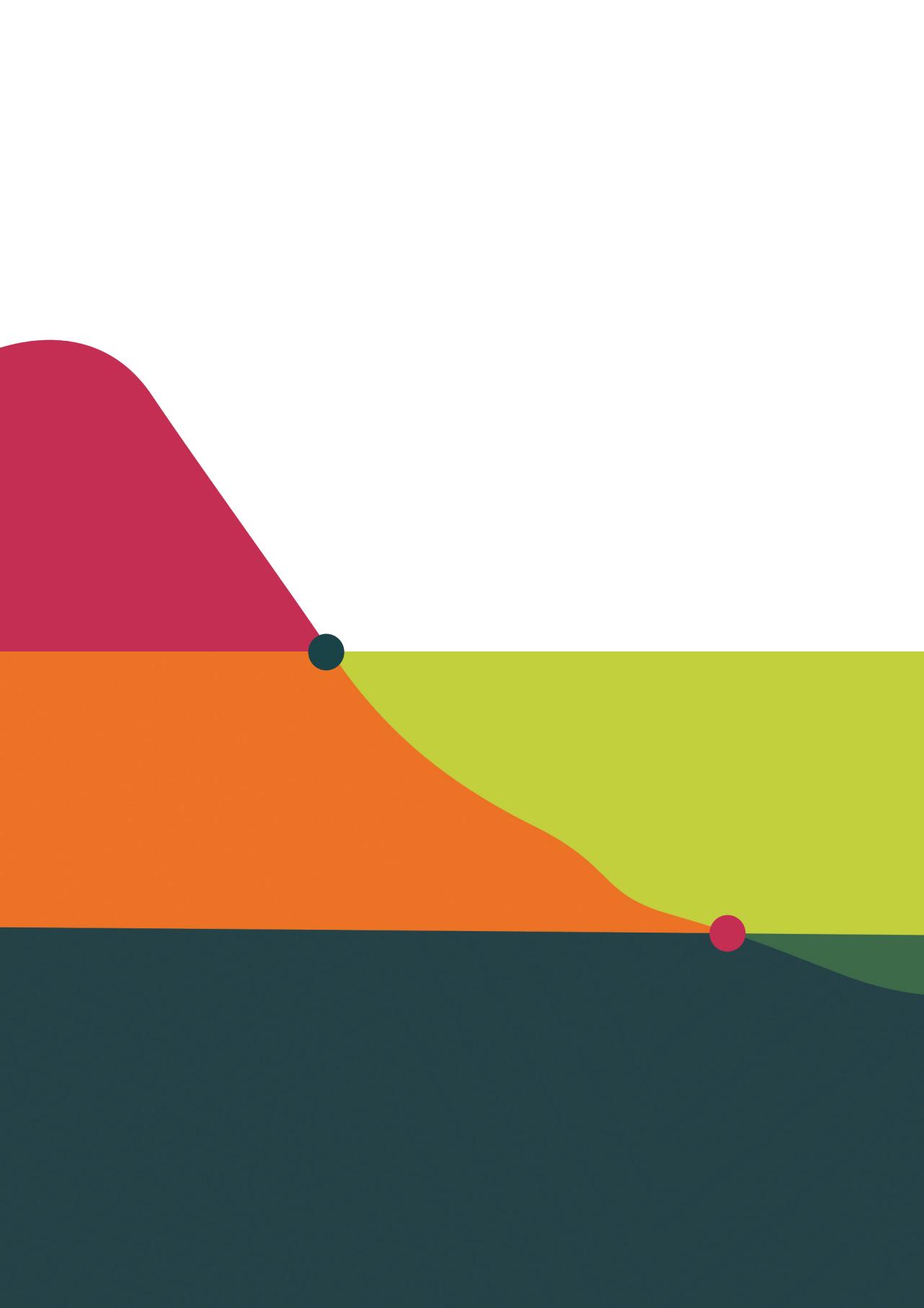
To indicate how much the choice of electricity source contributes to the calculated GHG footprint of green hydrogen, and how much the multi-functionality approach, we performed a variance decomposition analysis. The results show that the majority of variation in the GHG footprint is attributable to the GHG intensity of the electricity mix (92% of variance explained); the remaining 8% of the variation is attributable to the multi-functionality approach. From this we conclude that policies focusing on the GHG footprint of electricity used in electrolysis is most important in green hydrogen production.

Although green hydrogen could achieve the lowest GHG footprints in the long run, in the transition period until low-GHG electricity sources are readily available, deploying blue hydrogen may achieve higher emission reductions than green hydrogen. In addition, blue hydrogen may have lower environmental impacts than green, wind-based hydrogen when looking at impact categories beyond climate change, for example regarding mineral resource scarcity and freshwater ecotoxicity (Mac Dowell et al. 2021). This could indeed call for a "twin track" approach (Mac

Dowell et al. 2021), in which blue hydrogen and green hydrogen are deployed and scaled up simultaneously, taking into account regional preferences, environmental impacts, and economic and political interests.

We have shown here that the GHG footprint of green hydrogen strongly varies depending on a small number of choices. While footprints can inform and guide policymakers, investors and consumers, it is important to understand the context for which footprints are derived. Footprints ought to be viewed not just as a single number, but it needs to be considered how they change under different scenarios, choices and assumptions, over time, and how they compare to alternative options. This could steer actions that facilitate change in the desirable direction. The case of green hydrogen illustrates this: it is a no-regret for policymakers to reinforce decarbonisation of the electricity mix, to ensure that if electricity is used for hydrogen production, it has a low GHG footprint, and to ensure that if renewable power capacity is used for hydrogen production, it is not diverted from other uses that can achieve higher emission reductions. Only then, hydrogen is really 'green' and can fulfil its environmental promise as energy carrier and industrial feedstock.





# Worldwide greenhouse gas emissions of green hydrogen production and transport

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## Abstract

Green hydrogen is envisioned to play an important role in reducing emissions in industrial and energy systems. While current regulations view renewable electricity as zero emissions and do not account for component manufacturing and hydrogen leakage, we show that these are critical elements in the life-cycle greenhouse gas emissions of green hydrogen production and transport. Analysing 1,025 green hydrogen projects planned worldwide for 2030, we find a median production footprint of 3.0 kgCO<sub>2</sub>-eq/kg H<sub>2</sub> and overall emissions of 0.17 GtCO<sub>2</sub>-eq for a total hydrogen production volume of 65 million tonnes. Pipeline transport or liquid hydrogen shipping over 1000 km would increase emissions by 56%. Accordingly, local production generally has lower emissions than production in favourable far-away locations combined with long-distance transport. We conclude that not considering green hydrogen's full life-cycle emissions obscures large differences between renewable electricity sources and locations, and would make it deceptively simple to meet emission thresholds.

## 5.1 Main

Many actors expect hydrogen to become an important element in the transition towards a net-zero greenhouse gas (GHG) emissions society (IEA 2022a; IRENA 2022a; US House of Representatives 2022; BP 2023; European Commission 2023a; G20 Energy Transitions Ministers 2023). As a complement to electrification, hydrogen can contribute to the decarbonisation of industrial, transport and energy systems (Babiker et al. 2022). The current, fossil fuel-based hydrogen market of 100 million tonnes per year is projected to expand towards 300-800 Mt/yr of green hydrogen produced with electrolysis using renewable electricity by 2050, and up to 200 Mt/yr of blue hydrogen produced via steam methane reforming combined with carbon capture and storage (IRENA 2022b). Where renewable electricity and water are sufficiently abundant, green hydrogen might be produced, converted and exported to countries where production is more expensive or constrained, eventually turning hydrogen and its derivatives into internationally traded commodities (Van de Graaf et al. 2020). Domestic hydrogen strategies, such as those of the European Union and Japan, foresee a large and increasing role for hydrogen imports in reaching emission reduction targets, while other countries, such as Australia, Mauritania, and Chile, position themselves to become hydrogen exporters (IEA 2022a).

To quantify GHG emissions of hydrogen production, the International Energy Agency (IEA) and most proposed regulations and certification systems follow a methodology that assigns zero emissions to renewable electricity use, and excludes emissions from equipment manufacturing, yielding a footprint of 0 kgCO<sub>2</sub>-e/kg H<sub>2</sub> (IEA 2023). However, from a life-cycle perspective, hydrogen production emissions are far from zero and depend on the electricity generation technology (Bareiß et al. 2019; Mac Dowell et al. 2021), electrolyser technology (Gerloff 2021), and on the production system configuration (i.e., relative sizing of the different components and the type of grid integration) (Terlouw et al. 2022). Additional emissions arise from transporting hydrogen via pipeline (Tsiklios et al. 2022) or ship (Al-Breiki and Bicer 2020; Vilbergsson et al. 2023), and for conversion into derivatives that are more easily shipped, like ammonia (Ishimoto et al. 2020). As it is generally assumed that global climate targets are jeopardised if rapid investments in green hydrogen

supply chains are not realised (Velazquez Abad and Dodds 2020; Lebrouhi et al. 2022; Odenweller et al. 2022), clarity is urgently needed on how these different steps add up for different technological and geographical configurations, and what combination of factors would lead green hydrogen projects across the world to result in low emissions.

Here, we quantify GHG footprints of green hydrogen, considering the full production, conversion and transport chain, for different energy sources and transport modes. Since actual emissions will depend on how real-world hydrogen projects are planned, we analyse GHG footprints for 1,025 green hydrogen projects worldwide planned in 2030, covering different electrolyser technologies, capacities and electricity sources in 72 countries (2022b). This approach allows us to explore how much hydrogen might be available at what GHG footprint, and to what extent this potential supply would be consistent with emission thresholds such as the EU sustainability taxonomy or the US Clean H<sub>2</sub> standard. We also identify key parameters to reduce life-cycle GHG emissions of the introduction of green hydrogen on the global market.

## 5.2 GHG emissions of green hydrogen production

### 5.2.1 Scenario definition

We calculate project-specific hydrogen cradle-to-production gate emissions based on electrolyser-specific water, energy and material requirements, and electricity source documented in the *IEA Hydrogen Projects Database* (2022b). Since the majority of planned projects is estimated to be located in water-stressed regions (IRENA 2022b), we assume, for all projects, that water used for electrolysis is sea water desalinated in a reverse osmosis plant. GHG emissions of hydrogen production are determined by two main factors: The electricity generation technology (including its location), and the configuration of the project. We use country-specific renewable electricity GHG footprints based on meteorological conditions (Bosmans et al. 2021; Dammeier et al. 2023), facility type-specific hydropower footprints (Kadiyala et al. 2016) and national 2030 grid mixes modelled for a 2°C policy scenario (Knobloch et al. 2020). To account for the intermittency

of solar and wind energy, we consider three hydrogen production configurations (Table 5.1). Since electrolyzers' high investment costs prohibit excessive oversizing to accommodate peaks in renewable energy output, we use a fixed electrolyser utilisation factor of 0.5 (this means annual hydrogen production is half of continuous production at full capacity), in line with the cost-optimal size ratio of renewable, battery and electrolyser capacity as identified in previous studies (Palmer et al. 2021; Terlouw et al. 2022). In the first, *Off-grid: curtailment* scenario, there is no grid connection, renewable capacity is enlarged to double the size of the electrolyser capacity, and a lithium-ion battery is installed. Additional electricity generation beyond electrolyser and battery capacity is curtailed, and emissions associated with curtailed electricity are allocated to hydrogen. In the *Grid-connected: power export* scenario, additional electricity produced in the otherwise same configuration can be used by other consumers via a grid connection. This scenario could be applicable where renewable generation substitutes flexible fossil power, but may not be applicable in areas where renewable peak production regularly meets demand (Kolb et al. 2022). In the final *Grid-connected: power import* scenario, renewable peak capacity matches electrolyser capacity such that no batteries are needed and no surplus electricity can be produced. Instead, grid electricity is imported to top off intermittent renewables and to maintain the electrolyser utilisation factor of 0.5.

**Table 5.1. Green hydrogen production configuration scenarios based on intermittent renewables:**

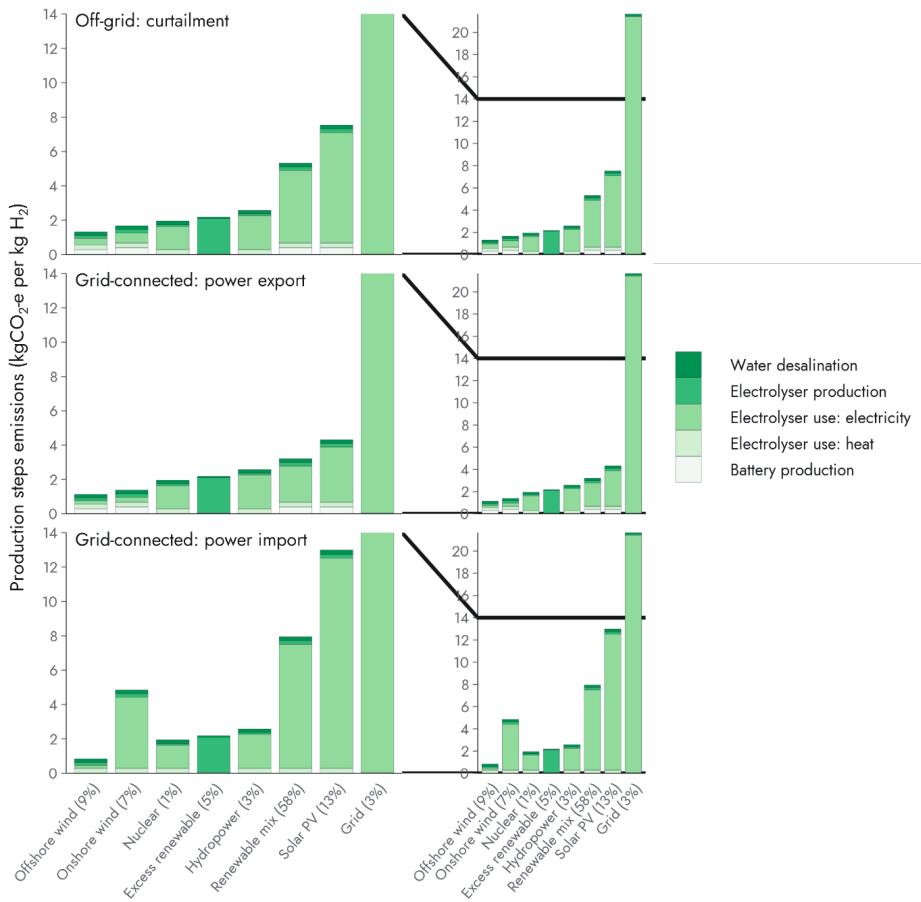
*Off-grid* with curtailment of surplus electricity, *Grid-connected* with a buyer of surplus electricity, and *Grid-connected* in a hybrid system where renewables are topped off with grid electricity, based on the configurations in Palmer et al. (2021) and Terlouw et al. (2022).

	1. Off-grid: curtailment	2. Grid-connected: power export	3. Grid-connected: power import
			
Electrolyser utilisation factor	0.5	0.5	0.5
Grid connection	No	Yes	Yes
Renewable peak capacity compared to electrolyser capacity	Double	Double	Equal
Battery	Yes	Yes	No
Curtailment factor	0.5	0	NA

### 5.2.2 The contribution of production inputs to the GHG emissions of green hydrogen

Life-cycle GHG emissions from green hydrogen production result from water desalination, electrolyser use (electricity and heat), and electrolyser and battery manufacture (Figure 5.1). Electricity generation to power the electrolyser is typically the largest contributor to production emissions, except when electricity from wind is used. Emissions of solar photovoltaic (PV) electricity exceed those of wind electricity due to higher emissions associated with manufacturing of solar panels compared to wind turbines. Electrolyser manufacturing emissions are attributed to the total amount of hydrogen produced over the electrolyser's lifetime, meaning manufacturing emissions for intermittent renewables-based production are about twice as large as for continuous production.

## Worldwide greenhouse gas emissions of green hydrogen production and transport



**Figure 5.1. Contribution of production inputs to the GHG footprint of green hydrogen produced from various electricity sources.** Contributions are medians across the 1,025 green hydrogen projects by electricity source presented per scenario. Percentages indicate what share of projects in the IEA Hydrogen Projects Database (2022b) use each electricity source.

### 5.2.3 The effect of electricity source and hydrogen production configuration on GHG emissions of green hydrogen

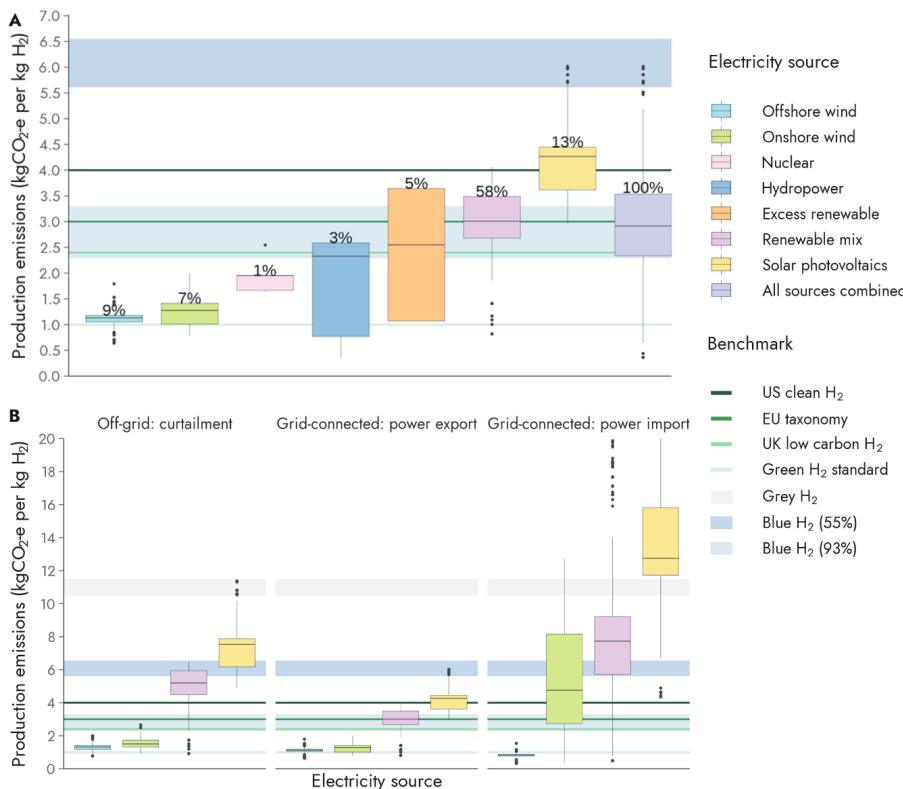
GHG emissions of green hydrogen production are between 0.4 and 30.5 kgCO<sub>2</sub>-e/kg H<sub>2</sub> produced across planned projects (interquartile range 2.4–3.6 kgCO<sub>2</sub>-e/kg H<sub>2</sub>; Power export scenario; Figure 5.2A). Median emissions across projects are 3.0 kgCO<sub>2</sub>-e/kg H<sub>2</sub>, which is similar to blue hydrogen with a 93% CO<sub>2</sub> capture rate and is exactly at the European Commission Taxonomy's threshold for sustainable

hydrogen production of 3.0 kgCO<sub>2</sub>-e/kg H<sub>2</sub> – between the US (4.0 kgCO<sub>2</sub>-e/kg H<sub>2</sub>) and UK (2.4 kgCO<sub>2</sub>-e/kg H<sub>2</sub>) thresholds for clean hydrogen. By electricity source, emissions are 1.0–1.4 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for onshore wind, 1.1–1.2 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for offshore wind, 3.6–4.4 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for solar PV, and 0.8–2.6 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for hydroelectricity (interquartile ranges across projects). Projects exceeding 6.3 kgCO<sub>2</sub>-e/kg H<sub>2</sub> are all powered by grid electricity and often exceed grey hydrogen production emissions as well (Supplementary Figure 5.1). These results are robust with regard to uncertainty in hydrogen production parameters, including equipment lifetime (<10% change in GHG emissions), electrolyser efficiency (<7%) and utilisation factor (<5%), battery capacity (<10%), site optimisation (<7%), and cooling water use cycles (20%; Supplementary Information Section 5.5).

In the *Off-grid: curtailment* scenario, power is overproduced and not valorised, resulting in higher production emissions of 1.3–1.7 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for onshore wind, 1.2–1.4 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for offshore wind, and 6.2–7.8 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for solar PV (interquartile ranges; Figure 5.2B). In the *Power import* scenario, production emissions for offshore wind are lower than in the *Power export* scenario, at 0.8–0.9 kgCO<sub>2</sub>-e/kg H<sub>2</sub>, mainly because less power production infrastructure is required while the import of grid electricity is limited due to offshore wind's high capacity factor (Figure 5.2B). Conversely, onshore wind and solar PV typically have lower capacity factors, leading to larger import of grid electricity and larger emissions: 2.7–8.2 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for onshore wind and 11.7–15.8 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for solar PV, exceeding grey hydrogen production emissions in 76% of solar projects.

Often it is proposed to use excess electricity for hydrogen production (Sternberg and Bardow 2015; Reuß et al. 2019). However, the available amount of excess electricity is likely limited in most grid systems in 2030 (Schill 2014; Daggash et al. 2018), which is reflected by a low electrolyser utilisation factor of 0.05 and results in relatively high emissions from electrolyser manufacture of 0.9–3.6 kgCO<sub>2</sub>-e per kg H<sub>2</sub> produced (depending on electrolyser technology; Figure 5.1). This means that even though emissions from electricity are considered zero when using excess electricity (Figure 5.1), the emissions of green hydrogen production from excess electricity still range 1.1–3.6 kgCO<sub>2</sub>-e/kg H<sub>2</sub> (Figure 5.2A). These emissions approximately double (2.5–6.2 kgCO<sub>2</sub>-e/kg H<sub>2</sub>) when excess electricity production emissions are attributed to hydrogen (Supplementary Information Section 5.5).

## Worldwide greenhouse gas emissions of green hydrogen production and transport

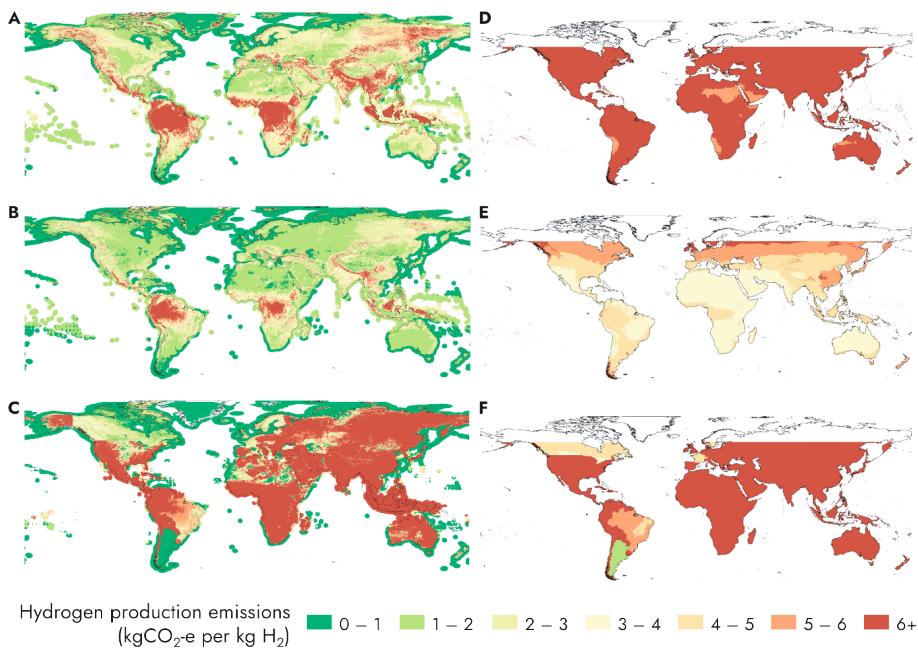


**Figure 5.2. Relation between green hydrogen production GHG emissions, and electricity source and renewable electricity configuration scenario.** Shaded areas indicate emission ranges for hydrogen production from natural gas (grey hydrogen) and from natural gas combined with CO<sub>2</sub> capture storage (blue hydrogen) with capture rates of 55% and 93%, with the range representing upstream methane leakage rates of 0.2% to 1.5% (Bauer et al. 2022a). Horizontal lines show cradle-to-gate emissions benchmarks for hydrogen production (IEA 2023). (A) Interquartile range and median of cradle-to-production gate emissions of hydrogen (kgCO<sub>2</sub>-e per kg H<sub>2</sub>) by electricity source in the *Power export* scenario, with outliers shown as black dots. Percentages indicate the share of projects in the IEA Hydrogen Projects Database (2022b) using each electricity source. Grid-based projects are shown in Supplementary Figure 5.1. (B) Cradle-to-production gate emissions for wind, solar photovoltaics (PV) and unknown renewable electricity represented by a mix of wind and solar ('Renewable mix') for three intermittent renewable-based hydrogen production configuration scenarios.

### 5.2.4 Spatial variation in green hydrogen production emissions

Variation across hydrogen projects with the same electricity source (Figure 5.2) results from differences in electrolyser type and location. Whereas the electrolyser type has only a small impact (Supplementary Figure 5.3), median solar-based

production emissions are 37% higher in Europe compared to South America due to differences in solar irradiation (Supplementary Figure 5.3): Emissions of renewable electricity are lowest in locations with high solar irradiation or wind speeds due to larger total electricity production over the energy facilities' lifetime, and therefore lower emissions per kWh (Figure 5.3). Although production emissions using wind electricity stay below 3.0 kgCO<sub>2</sub>-e/kg H<sub>2</sub> in the *Power export* scenario in most locations, except close to the equator, production emissions using solar electricity only stay below 3.0 kgCO<sub>2</sub>-e/kg H<sub>2</sub> very locally in South America (Figure 5.3). Emissions in the *Power import* scenario depend on the modelled GHG intensity of national grid mixes in 2030 (Knobloch et al. 2020), which generally result in higher emissions than in the other scenarios, unless the grid mix's GHG intensity is projected to be very low (Figure 5.3).



**Figure 5.3. Spatial variation in cradle-to-production gate GHG emissions of green hydrogen produced with wind and solar power.** Emissions are shown for hydrogen production using onshore and offshore (up to 200 nautical miles off the coast) wind power in the (A) *Off-grid: curtailment scenario*, (B) *Grid-connected: power export scenario*, and (C) *Grid-connected: power import scenario*. Solar photovoltaic-based hydrogen production emissions are also shown for the (D) *Off-grid: curtailment scenario*, (E) *Grid-connected: power export scenario*, and (F) *Grid-connected: power import scenario*. Emissions shown assume the use of a proton exchange membrane electrolyser and are based on regression model-derived GHG footprints for renewables (Bosmans et al. 2021, 2023; Dammeier et al. 2023) using local meteorological conditions (Davis et al. 2019; ESMAP 2020).

## 5.3 GHG emissions of hydrogen conversion and transport

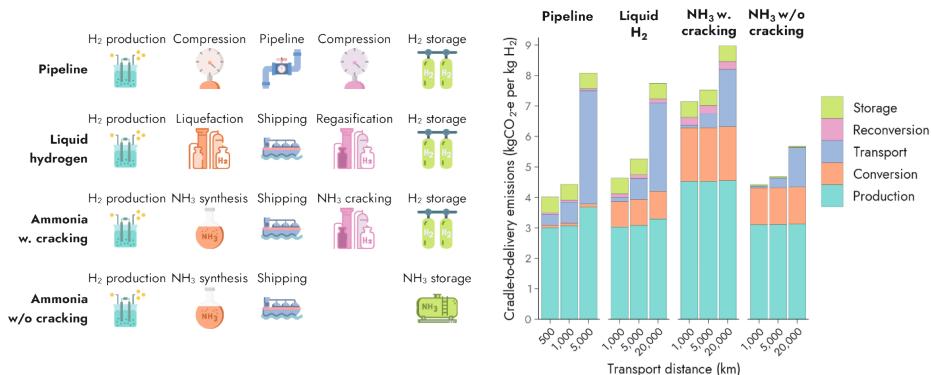
As green hydrogen is expected to become an internationally traded commodity, its life-cycle emissions should also include emissions from its conversion and transport. In our analysis of hydrogen transport, we include all documented projects (IEA 2022b), except those powered by grid electricity, and use the *Power export scenario*. We consider the most technologically mature and cost-efficient hydrogen transport options (IRENA 2022a): i) pipeline, ii) shipping of liquified hydrogen followed by regasification at the destination, iii) conversion to and

shipping as ammonia with reconversion (cracking) to hydrogen, or iv) shipping as ammonia without cracking (Figure 5.4). Pipelines are expected to be cheapest below 3000 km, liquid hydrogen up until 4000 km where pipelines are not possible, and ammonia shipping for remaining transport routes (IRENA 2022a).

Transport via pipelines leads to the lowest emissions across transport options on short distances, adding an extra 1.1 kgCO<sub>2</sub>-eq/kg H<sub>2</sub> for a distance of 500 km, and 1.5 kgCO<sub>2</sub>-eq/kg H<sub>2</sub> for a distance of 1000 km, as the median across projects. Conversely, for 5000 km transport, pipelines lead to the largest emission increase across transport options, adding 5.2 kgCO<sub>2</sub>-e/kg H<sub>2</sub>, as the median across projects (Figure 5.4). This steep increase results from the (re-)compression of gaseous hydrogen that needs to occur approximately every 125 km (Tsikrios et al. 2022) to maintain the pressure along the pipeline and the estimated 0.5% hydrogen loss per (re-)compression (Reuß et al. 2019; Tahan 2022). The hydrogen loss means (i) more hydrogen must be produced to arrive at 1 kg of hydrogen at destination, increasing production (emissions) by 3-26% for the 500-5000 km range (Figure 5.4; Supplementary Figure 5.4), and (ii) more hydrogen leaks into the atmosphere, which, based on an estimated 100 year-global warming potential (GWP) of 11 kgCO<sub>2</sub>-e/kg H<sub>2</sub> (Ocko and Hamburg 2022; Warwick et al. 2022), adds 0.3-2.3 kgCO<sub>2</sub>-e/kg H<sub>2</sub> delivered over a 500-5000 km distance (Supplementary Figure 5.5). If compressor losses can be reduced from 0.5% to 0.2% (Tahan 2022) and the interval between recompression stations extended to 500 km (Fraunhofer IFF 2023), while boil-off gas is reduced by 50%, then pipelines would have lowest emissions for distances up to 2000 km for wind-based production, and 5000 km for solar-based production (Supplementary Figure 5.5).

As for transport via ship, the compression and conversion of hydrogen to ammonia results in an extra emission of 0.8-1.0 kgCO<sub>2</sub>-e/kg H<sub>2</sub> input (interquartile range across projects). If ammonia is cracked at the destination, losses in conversion and reconversion (cracking) lead to increased hydrogen production (emissions) of 55-56% for the 500-20,000 km range. Ammonia transport without cracking results in relatively low emissions, but, crucially, requires ammonia to be demanded at the destination, not hydrogen. Shipping as liquid hydrogen causes less emissions than via ammonia with cracking (Figure 5.4), in line with earlier findings (Ishimoto et al. 2020), but are still substantial. Liquefaction itself is an electricity-intensive

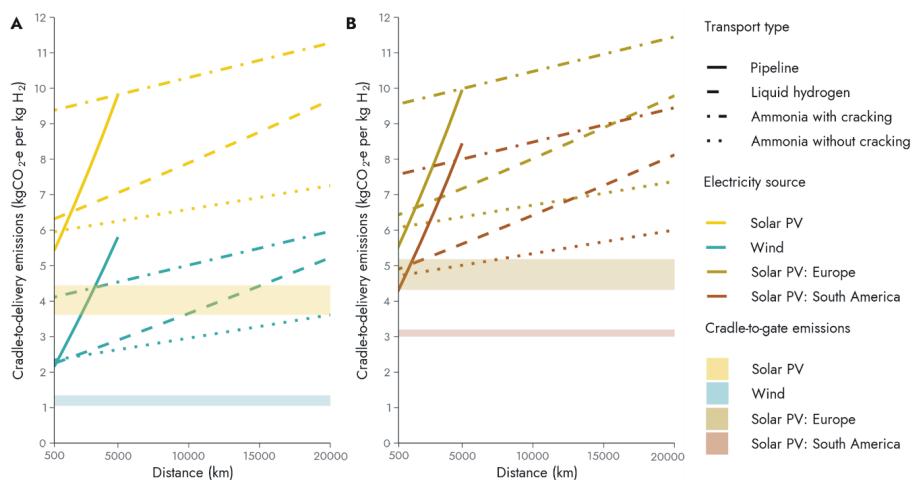
step, when using the same electricity source as the electrolyser it contributes 0.2–1.4 kgCO<sub>2</sub>-e/kg H<sub>2</sub> liquified, with a median across projects of 0.7 kgCO<sub>2</sub>-e/kg H<sub>2</sub>. Hydrogen boil-off losses resulting from cryogenic conditions during shipping (Al-Breiki and Bicer 2020) cause an increase in overall transport emissions (Figure 5.4), both from leaked hydrogen's direct role as GHG (0.4–1.3 kgCO<sub>2</sub>-e/kg H<sub>2</sub> delivered over 500–20,000 km) and additional hydrogen production (4–13% for the 500–20,000 km range). Furthermore, liquefied hydrogen's low volumetric density means more ships are needed and emissions from shipping itself (0.07–2.9 kgCO<sub>2</sub>-e/kg H<sub>2</sub> over a 500–20,000 km distance) are about twice as high as for ammonia (0.03–1.3 kgCO<sub>2</sub>-e/kg H<sub>2</sub> over a 500–20,000 km distance). For all transport options, storage is likely required. Storage infrastructure for 3 days' worth of hydrogen production adds 0.5 kgCO<sub>2</sub>-e/kg H<sub>2</sub> stored, or less than 0.05 kgCO<sub>2</sub>-e when stored as ammonia. Ultimately, which transport option results in lowest emissions depends on both transport distance and electricity source (Figure 5.5A), as transport losses compete with transport-specific emissions.



**Figure 5.4. Contributions of production, conversion, transport, reconversion and storage to cradle-to-point of delivery emissions of green hydrogen.** Transport options are transport by pipeline or shipping as liquid hydrogen and ammonia, with or without cracking. Median values across all projects (excluding grid electricity-based) in the Power export scenario are shown for transport distances of 500 km, 1000 km and 5000 km for pipelines, and 1000 km, 5000 km and 20,000 km for shipping.

A trade-off exists between hydrogen production at higher emissions in places with less solar irradiation that are close to demand, versus low-emission production in locations with high solar potential and then transporting it. Transport from the best solar locations would only compete with local production at very low

transport emissions, equivalent to a transport distance of 2000 km (Figure 5.5B). This means that production in Europe would for instance yield lower emissions than import from South America. Long-distance transport of wind-based hydrogen always leads to higher emissions than local wind-based production in our analysis, even if hydrogen is produced in the best wind locations and then transported to places with very low wind potential (Supplementary Figure 5.5), as the difference in production emissions between locations is too small to make up for transport emissions. Only if ammonia is demanded, long-distance transport from places with high renewable potential consistently results in lower emissions compared to local green ammonia production in places with low potentials: for wind-based production up to 7,000 km (Supplementary Figure 5.5), for solar-based production even for 20,000 km transport (Figure 5.5B).

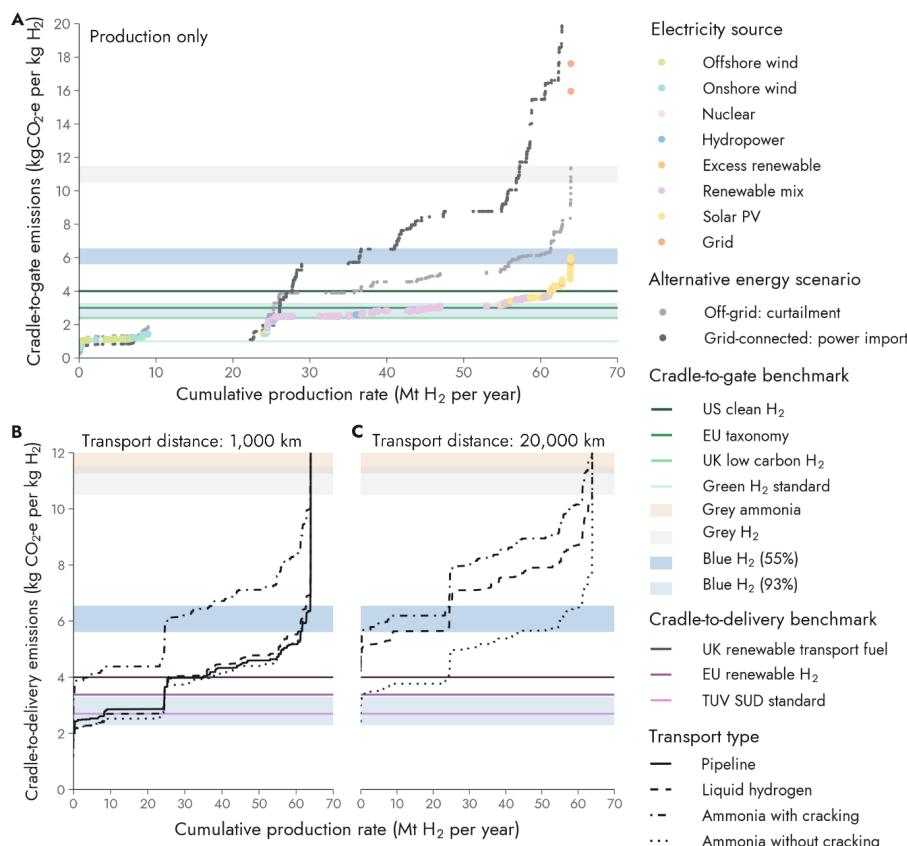


**Figure 5.5. GHG emissions of green hydrogen production and transport over increasing transport distances.** Median cradle-to-point of delivery emissions compared to the interquartile range of cradle-to-production gate emissions indicated by shaded areas (Power export scenario): (A) across all projects using wind electricity (onshore and offshore combined) or solar photovoltaic (PV) electricity; (B) across solar-based projects in Europe and solar-based projects in South America only. Cradle-to-point of delivery emissions for a transport distance of 0 km are higher than cradle-to-production gate emissions due to contributions after the production gate that do not scale with distance, including initial conversion and reconversion emissions, and storage.

## 5.4 Mitigation potential of planned green hydrogen projects

Using our GHG emission calculations and the 2030 hydrogen production rates of each project, we derive a global emission-supply curve for green hydrogen production (Figure 5.6A). In the *Power export* scenario, 53% of projects and 72% of cumulative production are below the EU taxonomy's threshold of 3.0 kgCO<sub>2</sub>-e/kg H<sub>2</sub> produced, that is based on an emission reduction of 73.4% compared to grey hydrogen (European Commission 2021). In the *Curtailment* and *Power import* scenarios 39% and 40% of cumulative production, and 25% and 20% of all projects, respectively, are below the EU taxonomy's threshold. At the more stringent Green Hydrogen Standard's threshold of 1.0 kgCO<sub>2</sub>-e/kg H<sub>2</sub>, only 0.6% of cumulative production and 4% of projects would meet the standard in the *Power export* scenario.

All planned hydrogen projects together would produce 65 million tonnes of hydrogen in 2030, emitting 0.17 billion tonnes of CO<sub>2</sub>-equivalents for production only (*Power export* scenario), equivalent to 2.5 kgCO<sub>2</sub>-e/kg H<sub>2</sub> on average. This value is lower than the median across projects, as larger projects in the database have, on average, lower production emissions, explained by the use electricity sources with lower emissions and planning in more favourable locations. Compared to producing the same amount of grey hydrogen, planned green hydrogen production would avoid up to 0.58 GtCO<sub>2</sub>-e/year (0.49 and 0.36 GtCO<sub>2</sub>-e in the *Curtailment* and *Power import* scenarios, respectively). Such replacement of existing applications of fossil-fuel based hydrogen is considered a priority (IEA 2021) and fits within the current grey hydrogen market of 100 Mt H<sub>2</sub> per year, but other uses of green hydrogen may result in different levels of avoided emissions. If 300 to 800 Mt of green hydrogen were to be produced annually as projected for 2050, assuming the same assumptions and distribution of production emissions as in the *Power export* scenario (but excluding grid based), annual emissions of green hydrogen production would range from 0.7 to 1.9 GtCO<sub>2</sub>-e. This is equivalent to about 1.3-3.5% of current global GHG emissions (Forster et al. 2023). Given the net zero targets for 2050 (Rogelj et al. 2023), this means emission reductions in the green hydrogen production process are necessary.



**Figure 5.6. GHG emission-supply curves for green hydrogen.** Emission-supply curves show the cumulative potential supply of hydrogen at increasing levels of GHG emission intensity. Horizontal lines and shaded areas indicate certification and alternative production technology benchmarks (Wang et al. 2021; Bauer et al. 2022a; IEA 2023). (A) Cradle-to-production gate emission-supply curve in colour for the *Power export* scenario, and grey and black the *Curtailment* and *Power import* scenarios, respectively; (B) Emission-supply curves for hydrogen production and 1000 km of transport and (C) for production and 20,000 km of transport, both for different transport modes in the *Power export* scenario.

When including 1000 km of transport in the emissions supply curves, total emissions become higher, at 0.26 and 0.27 GtCO<sub>2</sub>-eq for pipeline and liquid hydrogen transport, respectively. About 35 Mt of hydrogen per year could be produced and transported at emissions below the UK renewable fuel threshold (4.0 kgCO<sub>2</sub>-eq/kg H<sub>2</sub>), 25 Mt below the EU threshold (3.38 kgCO<sub>2</sub>-eq/kg H<sub>2</sub>) and 24 Mt below the TÜV SÜD threshold (2.7 kgCO<sub>2</sub>-eq/kg H<sub>2</sub>) – these thresholds include hydrogen distribution (Figure 5.6B). For a transport distance of 20,000 km, only

0.3 Mt of hydrogen-equivalent in the form of ammonia (without cracking) would stay below the EU threshold (Figure 5.6C). There are virtually no projects for which transport as ammonia combined with cracking would stay below any of the emissions thresholds, regardless of transport distance.

The European Commission's policy target of 10 Mt production in Europe by 2030 could be met based on the planned projects, where wind-based production (6.2 Mt in Europe) would stay below the EU taxonomy's threshold, whereas solar-based production (2.1 Mt) would not – for 10.9 Mt of planned European production the type of renewable is still unknown (Supplementary Figure 5.6). Entirely meeting the second target of 10 Mt of green hydrogen *import* while staying below the emission threshold would only be possible when ammonia is demanded. Further options for imports below the emissions threshold include wind-based production in North Africa with transport as liquid hydrogen or pipelines, or hydropower in North America with transport as liquid hydrogen, though these do not add up to 10 Mt.

## 5.5 Discussion and conclusions

Securing investment for export-oriented green hydrogen projects requires clarity on sustainability standards (Velazquez Abad and Dodds 2020), and it is critical to get these standards right for the hydrogen economy to lead to the envisioned emission reductions. Current regulations have a large blind spot: counting renewable electricity as zero emissions (e.g., EU Delegated Act on Article 27 of the RED II; European Commision 2023) and excluding emissions from component manufacturing and leakage (e.g., US Clean H2 Standard; US Department of Energy 2023b) would make it deceptively simple to stay below emissions thresholds for green hydrogen production, as opposed to when full life-cycle GHG emissions *are* included.

Including life-cycle GHG emissions shows that it is imperative that the supply of electricity has a low GHG intensity. Using electricity from the grid to produce hydrogen can result in emissions exceeding those of grey hydrogen production, questioning the climate mitigation value of such projects. 'Simply' claiming to

use intermittent renewable electricity obscures challenges around balancing operational hours of the electrolyser, the relative size of the renewable power generation facility, and choices regarding the inclusion of batteries, a grid connection and electricity curtailment. We show that the configuration of system components and securing buyers of surplus electricity may be critical to not exceed emission thresholds. In addition, it is problematic not to distinguish between types of renewables by assuming they all lead to zero emissions: whereas production using solar electricity would exceed emission thresholds in most locations, production based on hydropower or wind power would not. Furthermore, hydrogen transport may increase life cycle emissions significantly, depending on transport mode and distance, and on whether or not losses can be minimised. Since production closer to the user generally outperforms the benefit of producing at lower emissions in far-away locations with favourable weather conditions, including transport in the system boundaries of standards is required to understand the benefits and trade-offs of different locations. The inclusion of hydrogen's global warming potential plays a non-negligible role where transport losses occur and should therefore also be included. In this analysis, no losses in the production step are included, although they may be included in future analyses when more data become available.

Although average production emissions across projects are below the EU's threshold for sustainable hydrogen production, total emissions of projected hydrogen production are not near GHG neutrality. This raises the question whether the threshold is strict enough, even if additionality of renewable electricity would be guaranteed (de Kleijne et al. 2022). Our results indicate that efforts are needed to lower green hydrogen production emissions, including from the manufacture of electrolyzers, storage infrastructure and renewable energy technologies—which may, in fact, depend low-emission hydrogen availability for hydrogen-based steelmaking (Vogl et al. 2018). Furthermore, increasing electrolyser efficiency and operational hours, increasing hydrogen conversion and reconversion efficiencies, avoiding losses during transport and storage, or minimising transport altogether by co-locating industry with hydrogen production (Vartiainen et al. 2022; Devlin et al. 2023) are additional avenues to explore.

## 5.6 Methods

All analyses were performed in R (version 4.1.2), R Studio (version 2023.06.0) and Simapro (version 9.4.0.2), using Ecoinvent (version 3.8) and ReCiPe 2016 Midpoint (H) V1.07. The sections below detail the assumptions and data used for the analyses.

### 5.6.1 Selection of projects to include in analysis from the IEA database

Out of the 1477 hydrogen production projects in the IEA Hydrogen Projects database (2022b), 1025 were included in our analysis, based on selection criteria shown in Figure 5.7. The database includes information on project-specific electricity source to power electrolysis, electrolyser technology, country and project size (Figure 5.7).

A	Selection criteria	# projects remaining
	<b>Electrolysisbased hydrogen production</b>	1315
	<b>Expected to be operational in 2030</b>	1205
	<b>Size of project indicated</b>	1031
	<b>Location of project indicated</b>	1025

B	Electricity source	# projects	C	Electrolyser technology	# projects
	<b>Excess renewable</b>	51		<b>Alkaline (ALK)</b>	149
	<b>Grid</b>	27		<b>Polymer electrolyte membrane (PEM)</b>	234
	<b>Hydropower</b>	28		<b>Solid oxide (SOEC)</b>	25
	<b>Nuclear</b>	12		<b>Other</b>	617
	<b>Offshore wind</b>	96			
	<b>Onshore wind</b>	76			
	<b>Solar PV</b>	138			
	<b>Unknown</b>	597			

D	Continent	# projects	E	Project size (GW)	# projects	Cumulative capacity (GW)
	<b>Africa</b>	31		<b>0-0.001</b>	213	0.1
	<b>Asia</b>	123		<b>0.001-0.01</b>	235	1.2
	<b>Europe</b>	642		<b>0.01-0.1</b>	278	14
	<b>North America</b>	83		<b>0.1-1</b>	210	86
	<b>Oceania</b>	90		<b>1-10</b>	76	211
	<b>South America</b>	56		<b>&gt;10</b>	10	383

**Figure 5.7. Overview of hydrogen production projects. (A) Selection criteria based on which 1025 out of the 1477 projects in the IEA database (2022b) are selected.** Projects decommissioned before 2030 or starting after 2030 are excluded but if no start date is indicated, assumed starting date is 2030. Out of the 1025 selected projects, the number of projects is specified by (B) electricity source, (C) electrolyser technology, (D) continent and (E) project size.

## 5.6.2 Electrolyser energy and material requirements

The IEA database (2022b) distinguishes between alkaline (ALK), polymer electrolyte membrane or proton exchange membrane (PEM), solid oxide electrolysis cell (SOEC) electrolysers and 'Other Electrolysis', each with different energy and material requirements. We use the life cycle inventory for ALK, PEM and SOEC electrolysers by Gerloff (2021) as the basis of our analysis (Table 5.2), including construction of the electrolyser stacks and 'balance of plant' (Gerloff 2021). We adapted the electricity demand by using the expected 2030 values instead of the 2017 values from the same study (Smolinka et al. 2018). The SOEC has an output pressure of 1 bar (Vilbergsson et al. 2023), so its electricity requirement includes an additional compression step to reach the same output pressure as the ALK/PEM electrolysers (30 bar) (Palmer et al. 2021; Terlouw et al. 2022). We assume SOEC electrolyser heat demand is met using natural gas (Mehmeti et al. 2018) (Ecoinvent process 'market group for heat, district or industrial, natural gas {GLO}'), as this has lower emissions than the Ecoinvent process 'market group for heat, district or industrial, other than natural gas {GLO}' (Gerloff 2021). For 'Other Electrolysis', we calculate the average of the requirements of the ALK, PEM and SOEC electrolysers.

Since over 70% of planned projects are estimated to be located in water-stressed regions (IRENA 2022b), we assume that all water used for electrolysis (feed and cooling water) is sea water, desalinated in a reverse osmosis plant. After desalination, water is deionised, for which we adapt the relevant Ecoinvent processes based on Terlouw et al. (2022). The electricity requirement for desalination is around 3.8 kWh per m<sup>3</sup> of water (Palmer et al. 2021; Brigljević et al. 2022). Required cooling water (Gerloff 2021) is assumed to be recirculated five times to reduce water demand and thermal discharge into surface water (Hydrohub 2020).

**Table 5.2. Operating data for alkaline (ALK), polymer electrolyte membrane (PEM), and solid oxide (SOEC) electrolyzers.**

Operating data per kg H <sub>2</sub>	ALK	PEM	SOEC	Other Electrolysis	Reference
Water demand electrolysis (kg)	8.9	8.9	8.9	8.9	(Gerloff 2021)
Water demand cooling and compressor (5 circulation cycles) (kg)	17.6	17.6	128.9	54.7	(Hydrohub 2020; Gerloff 2021)
Electricity demand electrolysis (kWh)	49.7	53.7	40.6	48.0	(Smolinka et al. 2018)
Heat demand electrolysis (MJ)	-	1.0	18.9	6.6	(Gerloff 2021)
Potassium oxide (KOH) electrolyte (g)	3.7	-	-	1.2	(Palmer et al. 2021)
Additional electricity requirement for compression of H <sub>2</sub> to 30 bar (kWh)	0	0	2.1	0.7	Calculated based on equation 4.1 (Christensen 2020)

### 5.6.3 Green hydrogen production configurations

#### Continuous electrolytic hydrogen production

We assume continuous operation of the electrolyser for grid-connected systems (Terlouw et al. 2022), as well as for hydropower and nuclear electricity sources. The utilisation factor of the electrolyser ( $f_{utilisation}$ ) is equal to 1 for continuous operation. We use the documented (IEA 2022b) electrolyser capacity or annual hydrogen production rate to calculate the missing parameters using equations 5.1 and 5.2:

$$M_{H_2} = \frac{cap_{electrolyser} \cdot f_{utilisation} \cdot 8760 \text{ h}}{P_{electrolyser}} \quad (5.1)$$

$$cap_{electrolyser} = \frac{M_{H_2} \cdot P_{electrolyser}}{f_{utilisation} \cdot 8760 \text{ h}} \quad (5.2)$$

Where  $M_{H_2}$  is the annual hydrogen production rate in tonnes of H<sub>2</sub> per year,  $cap_{electrolyser}$  the installed electrolyser capacity in MW, and  $P_{electrolyser}$  the electricity requirement for electrolysis in kWh per kg H<sub>2</sub> (Table 5.2).

### ***Green hydrogen production configuration scenarios for intermittent renewables***

Hydrogen production systems operated by solar or wind electricity have a *futilisation* lower than 1 due to the intermittent character of these sources. Based on energy system optimisation, cost-optimal configurations for dedicated renewable systems have a between 0.5 and 0.6 (Terlouw et al. 2022), which means that over the course of the year, 0.5 to 0.6 times the hydrogen is produced of continuous production. In these configurations, electricity generation facilities are oversized in combination with battery storage and curtailment of electricity. The energy to power ratio of the battery is between 2 and 10 (mean: 6), which expresses how many hours' worth of electricity can be stored for full electrolyser operation. Despite the use of a battery that allows for production of hydrogen with a time lag from when the electricity is produced, curtailment of electricity is high, with a curtailment factor between 0.2 and 0.7 (mean: 0.5). An alternative configuration is characterised by a of 0.4, a battery that can store 3 hours' worth of electricity for full electrolyser operation and no curtailment (Palmer et al. 2021).

When electricity must be curtailed, emissions from manufacture of the full scale of the electricity facilities are allocated to hydrogen production. In this case, there is no grid connected or no user of this electricity (*Off-grid: curtailment scenario*). If a grid connection is in place and the surplus is used by other economic activities, these emissions need not be allocated to hydrogen (*Grid-connected: power export scenario*) (Table 5.1). Furthermore, we include a third scenario representing a grid-connected configuration where the renewable generation capacity equals the electrolyser capacity (Terlouw et al. 2022), and powers the electrolyser for the share of the year equal to the country-specific capacity factor (75<sup>th</sup> percentile). Based on the mean of the cost-optimal range for grid-connected electrolyzers being 4,500 hours (IEA 2019c), i.e., close to half the hours in a year, we use an electrolyser utilisation factor of 0.5. The difference between the renewable capacity factor and 0.5 would be supplied by the grid in this *Grid-connected: power import scenario* (Table 5.1).

Since the specific optimal configuration of energy generation oversizing, battery size and curtailment depends on hourly (Terlouw et al. 2022) or even minute-to-minute (Palmer et al. 2021) electricity generation profiles and local costs of the respective components, it is out of the scope of this paper to optimise this for each

specific project. Instead, we use hydrogen production configuration parameters based on these earlier studies (Palmer et al. 2021; Terlouw et al. 2022) (Table 5.3) and assess the choice of parameters in the sensitivity analysis.

### ***Green hydrogen production using excess renewable electricity***

When electrolysis is powered by excess renewable electricity, we assume that the electrolyser only operates when excess electricity is available. Only when intermittent renewables make up more than 50-60% of the electricity mix, the curtailment factor is higher than 0.025 (Daggash et al. 2018). For Germany, with relatively high renewables penetration, 471 surplus hours are foreseen in 2032 (Schill 2014), which means that would be 0.05, which we use for all excess-based projects.

**Table 5.3. Hydrogen production configuration by electricity source and intermittent renewable electricity scenario.**

	<b>Continuous (grid, hydro, nuclear)</b>	<b>Intermittent renewable</b>			<b>Excess renewable</b>
		Off-grid: curtailment	Grid-connected: power export	Grid-connected: power import	
Electrolyser utilisation factor	1	0.5	0.5	0.5	0.05
Battery energy to power ratio	NA	6	6	NA	NA
Curtailment factor	NA	0.5	NA	NA	NA
Electricity share passing through battery	NA	0.25	0.25	NA	NA
Battery round-trip efficiency	NA	0.9	0.9	NA	NA

### ***Batteries***

The battery is assumed to be a lithium-ion battery with a roundtrip efficiency of 90% (Palmer et al. 2021; Terlouw et al. 2022) (Table 5.3). Since the theoretical maximum of the share of electricity that passes through the battery before arriving at the electrolyser is 0.5, the gross electricity generation requirement is at most 5% higher than the net electricity requirement. Given that weather patterns are much more variable than required for this maximum, we assume that 0.25

of electricity passes through the battery. We use the GHG emissions of lithium nickel manganese cobalt oxide battery manufacture (including the battery system with pack, battery management system, energy management system, battery thermal management system, system container, inverter, and transport) (Schmidt et al. 2019), estimated at 68.6 kgCO<sub>2</sub>-eq/kWh of storage capacity in 2030 in a 2°C scenario (Bauer et al. 2022b).

#### 5.6.4 Emissions intensities of the electricity sources

In this section we describe how we estimate the GHG emissions intensity per project, based on the documented electricity source and location (IEA 2022b). For most electricity sources we use country-specific information, while for six projects where multiple countries are given as the location, their average is used.

##### *Excess renewable electricity*

Hydrogen production has been proposed as a power-to-X technology, using electricity when the supply of electricity temporarily exceeds demand. Some studies include the GHG intensity of renewable electricity generation for such electricity (Uusitalo et al. 2017; Koj et al. 2018; Rumayor et al. 2018; Bareschino et al. 2020; Briglijević et al. 2022), while other studies assume a zero GHG intensity for excess electricity arguing that the excess would have a zero cost or would otherwise have been wasted (Sternberg and Bardow 2015, 2016; Meylan et al. 2017; Biernacki et al. 2018; Daggash et al. 2018; Jens et al. 2019; Reuß et al. 2019). We assume a GHG intensity of 0 kgCO<sub>2</sub>-eq/kWh in the default calculations and the country-specific mean of solar and wind electricity in the sensitivity analysis (see Section on 'Unknown electricity source').

##### *Grid electricity*

To take the decarbonisation of grid mixes in the future into account, we use the GHG intensities of modelled 2030 grid mixes per country for a 2°C policy scenario (Knobloch et al. 2020). When a country is not modelled separately but as part of a region, we use the GHG intensity of the region it belongs to.

## **Hydroelectricity**

The GHG intensity of hydropower depends on the size of the facility and facility type, i.e., impoundment (reservoir, dam) or diversion (run-of-river) (Kadiyala et al. 2016). Although the IEA database (2022b) does not include information on the facility type for projects using hydropower, we used the project name and sources in the database to find the facility type and match it with the mean GHG intensity: 21.1 gCO<sub>2</sub>-e/kWh for small impoundment systems (0.1-30 MW); 40.6 gCO<sub>2</sub>-e/kWh for large impoundment systems (> 30 MW); 27.2 gCO<sub>2</sub>-e/kWh for small diversion systems (0.1-30 MW); and 3.5 gCO<sub>2</sub>-e/kWh for large diversion systems (> 30 MW) (Kadiyala et al. 2016). The higher emissions for large impoundment systems are a result of construction requirements and flooding of land.

## **Nuclear electricity**

Relatively large GHG ranges have been reported for nuclear electricity: 10-130 (Lenzen 2008) or 3.7-110 gCO<sub>2</sub>-eq/kWh (IPCC 2015), explained by different GHG intensities of the background economy, different uranium ore grades and different enrichment methods. We use the more recent value of 27.6 gCO<sub>2</sub>-eq/kWh found using a hybrid LCA approach (Pomponi and Hart 2021).

## **Offshore and onshore wind electricity**

We created a generalised linear model (equation 5.3) based on wind turbine data collected and analysed by Dammeier et al. (2023) to estimate the GHG intensity of wind electricity based on solely wind speed and onshore or offshore location:

$$\log_{10}(EF_{GHG}) = 2.2571 - 0.2025 \cdot v_{avg} + 0.3066 \cdot O \quad (5.3)$$

Where:  $EF_{GHG}$  is the emissions intensity of wind electricity (gCO<sub>2</sub>-e/kWh),  $v_{avg}$  is the 30-year average wind speed at 100 meter in m/s, and  $O$  is a categorical variable denoting onshore (0) or offshore (1) wind farms. The model's R<sub>2</sub> is 0.84 and end-of-life emissions of the wind farms are not included.

Wind speed is location-dependent. The GHG intensity of wind electricity is estimated based on onshore/offshore location and wind speed, using data from the GlobalWindAtlas (Davis et al. 2019), based on ERA5 data for the years 2008-2017. We use the exclusive economic zone to determine the offshore wind speed,

which is the area up to 200 nautical sea miles from the coast. We use the 75<sup>th</sup> percentile of wind speed at the country level or in the exclusive economic zone, assuming that the facility is built in a favourable location. In the sensitivity analysis, we use the median and 90<sup>th</sup> percentile of wind speed.

### **Solar photovoltaic electricity**

Based on a regression model requiring only panel type, yearly irradiation and age of a facility by Bosmans et al. (2021, 2023), the GHG intensity of solar electricity can be estimated using equation 5.4:

$$\begin{aligned}\log(EF_{GHG}) = & 64.3 - 0.031 \cdot Year + \beta_{type} \\ & - 0.00023 \cdot I_{mean} + \beta_{Y-type} \cdot Year\end{aligned}(5.4)$$

Where:  $EF_{GHG}$  is the emissions intensity of solar electricity ( $\text{gCO}_2\text{-e/kWh}$ ),  $Year$  is the absolute value and  $I_{mean}$  is irradiation in  $\text{kWh m}^{-2}\text{yr}^{-1}$ .  $\beta_{type}$  is -12.7 for CdTe, -49.4 for CI(G)S, 48.9 for mono-Si, 47.6 for poly-Si and 0 for a-Si;  $\beta_{Y-type}$  is 0.0062 for CdTe, 0.025 for CI(G)S, 0.024 for mono-Si, 0.024 for poly-Si and 0 for a-Si. This model has an  $R^2$  of 0.99, and end-of-life emissions of the solar PV facilities are not included. Further details on estimated panel type modelling is included in Supplementary Information Section 5.2.

Irradiation is location-dependent. The GHG intensity of solar electricity is estimated based on global horizontal irradiation at the country level, using data from the GlobalSolarAtlas (ESMAP 2020). The GlobalSolarAtlas is based on a SolarGIS model using several long term datasets, together providing a climatic mean (Solargis 2023), and covers latitudes from parallel 45°S to parallel 60°N. No hydrogen projects based on solar PV in the IEA database (2022b) are located outside this range. The country averages for countries passing through these parallels are based on the data that fall into the range (e.g., Sweden, Canada, Argentina, Chile). We take the 75<sup>th</sup> percentile of the area's irradiation, assuming that the facility is built in a climatically favourable location and can feasibly be connected to the grid (Bosmans et al. 2022). In the sensitivity analysis, we use the median and 90<sup>th</sup> percentile of solar irradiation.

### **Unknown electricity source**

When projects lack information on the electricity source used, we assume a shared contribution of solar PV (50%) and wind (50%)- half of it offshore and half onshore.

## **5.6.5 Green hydrogen processing and transport**

In our analysis of hydrogen transport, we include all documented projects (2022b), except those powered by grid electricity, and use the *Power export* scenario. We assume post-electrolysis processing of H<sub>2</sub> takes place at or near the H<sub>2</sub> production location. We assess the most technologically mature and cost-efficient transport modes (IRENA 2022a): pipelines, liquid H<sub>2</sub> shipping and ammonia shipping, with associated conversion and reconversion steps. For ammonia we include a scenario with reconversion to H<sub>2</sub>, and one without reconversion since ammonia may also be used directly to substitute grey ammonia. In the latter case, emissions are converted to H<sub>2</sub>-equivalents based on hydrogen content (kg H<sub>2</sub> per kg NH<sub>3</sub>).

5

The electricity requirement for the initial conversion steps at the production location are met by the same electricity source as electrolysis. We assume that all other electricity requirements during transport and reconversion are covered by a global average of solar and wind electricity, with a GHG intensity of 37 gCO<sub>2</sub>-e per kWh.

### **Pipeline**

Our analysis of hydrogen transport by pipeline is based on an LCA study that includes thermodynamic modelling of hydrogen transport via new steel pipelines with a diameter of 48 inch and operating at 80 bar (Tsiklios et al. 2022). As the pipeline network can be a storage solution (Tsiklios et al. 2022), hydrogen storage before transport is not needed. Since hydrogen is produced at lower pressure than required for the pipes, hydrogen is first compressed. Furthermore, the pressure drops along the pipeline and recompression is required every 125 km to reduce environmental impacts (Tsiklios et al. 2022) and costs (d'Amore-Domenech et al. 2023). The distance between re-compression stations depends on the hydrogen mass flow rate and pipeline diameter, where smaller flow rates and larger pipeline diameters correspond to larger distances between stations. We use the reference case of 125 km (Tsiklios et al. 2022) as the default, and test the maximum distance

of 500 km (Fraunhofer IFF 2023). We include leakage of hydrogen at the compressor station, which is estimated to be 0.5% (Reuß et al. 2019). New technologies may limit this loss to 0.2% (Tahan 2022), which we assess in an additional scenario. We assume that a final recompression step is needed before storage of gaseous hydrogen at the destination at 80 bar, and include the infrastructure requirement for three days' storage capacity (Palmer et al. 2021; Terlouw et al. 2022). Further details on pipeline transport are provided in Supplementary Table 5.4.

### ***Shipping***

We assume that the production plant is situated close to a port and include a storage capacity on land equal to the capacity of the tanker or to 21 days' worth of production capacity (Stolzenburg and Mubbala 2013). We further include loading and unloading, shipping and the evaporation of energy carriers: boil-off gas (BOG). When a liquid energy carrier is stored under cryogenic conditions, i.e., at a temperature below its boiling point, BOG is produced due to the difference in temperature between the liquid and the environment (Al-Breiki and Bicer 2020). While during storage the gas can be re-liquified and recirculated to storage (Ishimoto et al. 2020; Kolb et al. 2022), during loading/unloading and shipping the BOG is generally considered to be evaporated to the atmosphere and counted towards an energy/mass loss (Wijayanta et al. 2019; Al-Breiki and Bicer 2021; McKinlay et al. 2021; Kolb et al. 2022). Besides BOG generation, leakages may occur (Al-Breiki and Bicer 2021). Although in the future it might be possible to use BOG as part of the shipping fuel, this process still needs to be developed (Wijayanta et al. 2019; Al-Breiki and Bicer 2020; Ishimoto et al. 2020; McKinlay et al. 2021) and heavy fuel oil is typically assumed to be used as transport fuel for shipping liquid hydrogen and ammonia in 2030 (Al-Breiki and Bicer 2021; Kolb et al. 2022; Egerer et al. 2022) which is what we assume in our analysis.

BOG rates are time-dependent rates of loss:

- BOG during land storage in producing port dependent on production rate; the longer it takes to produce the tanker capacity, the more BOG is produced with a maximum of 21 days.

- BOG during shipping depends on duration of shipping, which is calculated from the shipping distance assuming a speed of 16 knots.
- BOG during land storage at receiving port set to fixed number of days of storage: 3 days.
- BOG during loading/unloading is a fixed percentage of load.

**Table 5.4. Characteristics of liquid hydrogen and ammonia.**

	Liquid H <sub>2</sub>	Ammonia	References
Storage temperature (°C)	-253	-34	(Al-Breiki and Bicer 2020)
Density (kg/m <sup>3</sup> )	71.1	682.8	(Al-Breiki and Bicer 2020)
Lower heating value (MJ/kg)	120	18.6	(Al-Breiki and Bicer 2020)
Tanker volume (m <sup>3</sup> )	160,000	160,000	(Wijayanta et al. 2019; Al-Breiki and Bicer 2020; Egerer et al. 2022)
Shipping mass per tanker (kg)	11,376,000	109,248,000	For 160,000 m <sup>3</sup> tanker
Shipping speed (knots)	16	16	(Kolb et al. 2022)

*Liquid hydrogen.* Gaseous hydrogen is liquified to increase the density for shipping. Industrial hydrogen liquefaction currently consumes between 12.5 and 15 kWh per kg H<sub>2</sub> (Wijayanta et al. 2019; Al-Breiki and Bicer 2021). For 2030, we use 12 kWh. We use a LCI of a 50 tH<sub>2</sub>/day liquefaction plant (Stolzenburg and Mubbala 2013; Kolb et al. 2022) and scale it to emissions per kg H<sub>2</sub> for each project. After liquefaction, liquid H<sub>2</sub> is stored at -253°C (Stolzenburg and Mubbala 2013; Al-Breiki and Bicer 2021), loaded, shipped, unloaded, re-gasified and compressed, and stored as gaseous hydrogen for three days. We adapt the Ecoinvent entry for liquified natural gas (LNG) sea transport to liquid H<sub>2</sub> shipping (Kolb et al. 2022), see Supplementary Information Section 5.3 and Supplementary Table 5.5 for further details on liquid hydrogen transport.

*Ammonia.* For (green) ammonia production, we consider a flexible, electricity-based Haber-Bosch ammonia production process (Armijo and Philibert 2020; Egerer et al. 2022), using N<sub>2</sub> from a cryogenic air separation unit for which we adapt the Ecoinvent process to harmonise the electricity source to be equal to that of electrolysis. In case of variable hydrogen production, i.e., based on intermittent

renewable electricity, the ammonia production process requires two days of hydrogen storage to buffer fluctuations (Armijo and Philibert 2020), based on the inventory for gaseous hydrogen storage vessels (Palmer et al. 2021; Terlouw et al. 2022). Ammonia is liquified and temporarily stored, for which we adapt the inventory for hydrogen storage vessels to the much higher volumetric density of liquid NH<sub>3</sub>. BOG is reliquefied and recirculated (Ishimoto et al. 2020). We adapt the Ecoinvent entry for LNG sea transport for ammonia shipping (Kolb et al. 2022). At the receiving port, ammonia is unloaded and temporarily stored, and BOG reliquefied (Ishimoto et al. 2020). If ammonia is delivered to the market as ammonia, three days of storage capacity are included, and one day if it is cracked to recover the hydrogen. Then, three days of gaseous hydrogen storage are additionally included. Cracking is an endothermic reaction requiring temperatures of 500°C or higher. The process uses electricity and part of the hydrogen as fuel, resulting in a H<sub>2</sub> recovery rate of 69.5% (Ishimoto et al. 2020). See Supplementary Information Section 5.3 and Supplementary Table 5.6 for further details on the ammonia transport modes.

### 5.6.6 Construction emissions and lifetime

The construction emissions of all technology components are scaled to the installed electrolyser capacity or annual hydrogen production rate for each project. Construction emissions are divided over the total amount of hydrogen produced in the project following equation 5.5:

$$EF_{construction} = \frac{EM_{construction}}{Lifetime \cdot M_{H_2}} \quad (5.5)$$

Where:  $EF_{construction}$  are the emissions from construction in kgCO<sub>2</sub>-e/kg H<sub>2</sub> produced,  $EM_{construction}$  are the total emissions from construction of the component in kgCO<sub>2</sub>-e,  $Lifetime$  is equipment lifetime in years, and  $M_{H_2}$  is the annual hydrogen production, as determined in equation 5.1, expressed here in kg H<sub>2</sub> per year. This means that depending on electrolyser utilisation factor and total amount of hydrogen produced, the manufacturing emission contribution varies between projects.

### 5.6.7 Calculation of spatial variation in production emissions (Figure 5.3)

The maps in Figure 5.3 show the spatial variation in cradle-to-production gate GHG emissions of green hydrogen production for wind and solar photovoltaics electricity, using a polymer electrolyte membrane electrolyser. For electrolyser-specific requirements see Methods Section 5.6.2. To calculate the local GHG intensities of wind and solar electricity, we combined the regression models of Dammeier et al. (2023) (using solely wind speed and on/offshore) and Bosmans et al. (2021, 2023) with wind speed and capacity factor maps from the GlobalWindAtlas ( $0.0025^\circ$  resolution) (Davis et al. 2019) and solar irradiation and capacity factor maps from the GlobalSolarAtlas ( $0.0025^\circ$  and  $0.0083^\circ$  resolution respectively) (ESMAP 2020). For offshore wind, we determine the footprint in the 'exclusive economic zone', which is the area up to 200 nautical sea miles from the coast. For solar electricity, we calculated the GHG footprint based on a solar PV facility for the year 2021 (Supplementary Information Section 5.2), which we use to represent the most recent market, and resampled the local capacity factor map for solar PV to a  $0.0025^\circ$  resolution. For both wind and solar, the GHG footprint of green hydrogen production was calculated for three scenarios: *Off-grid: curtailment*, *Grid-connected: power export*, and *Grid-connected: power import* (Table 5.1). For the *Grid-connected: power import* scenario we used additional information on the national 2030 grid mixes modelled for a  $2^\circ\text{C}$  policy scenario (Knobloch et al. 2020) when local capacity factors were below 50%. For visualisation purposes, all output maps were resampled to a  $0.025^\circ$  resolution by taking the mean value of all underlying cells. All analyses were performed in R (version 4.1.2), R Studio (version 2023.06.0) and ArcGIS Pro (version 3.1.2).

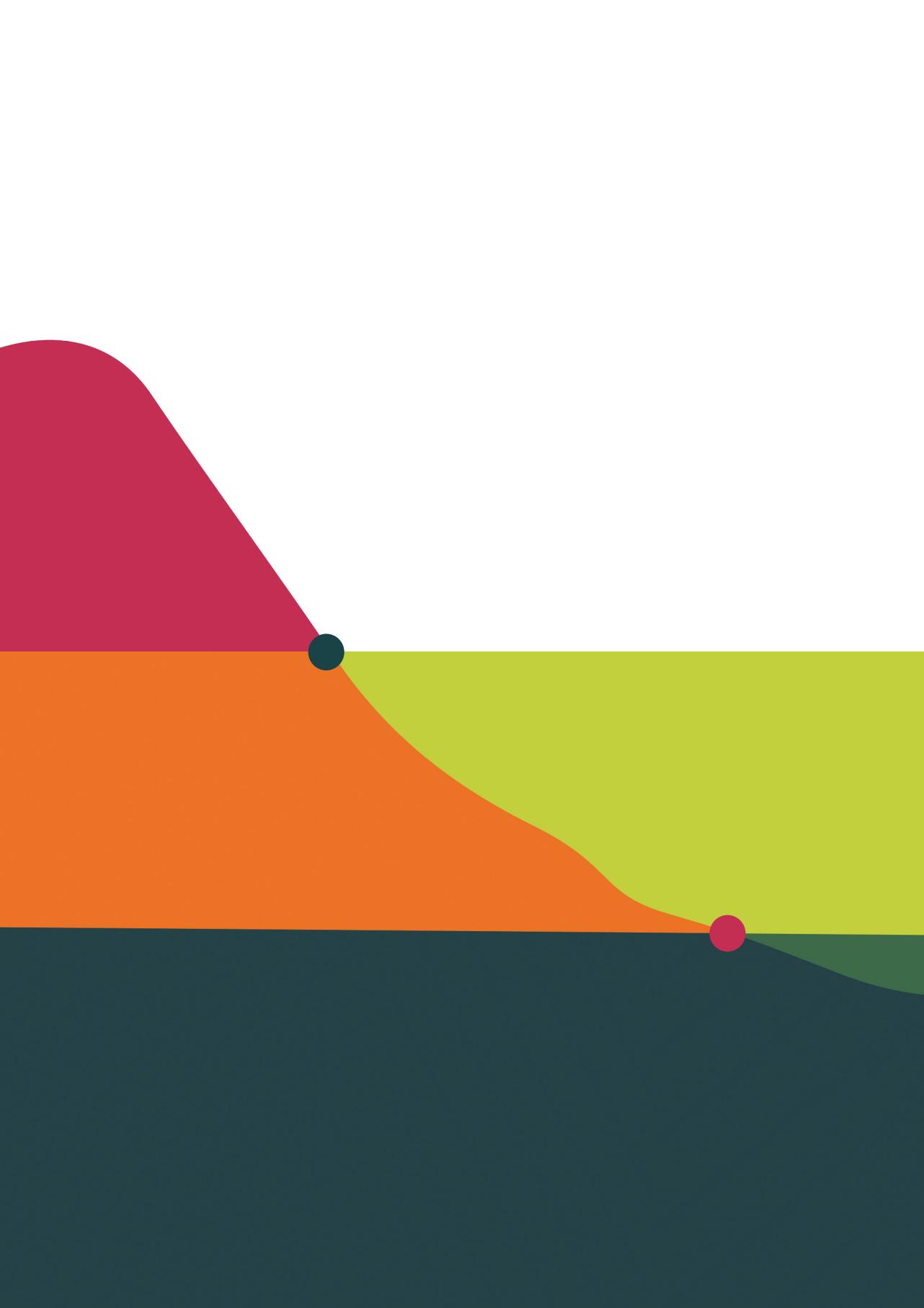
### 5.6.8 Benchmarks

- The GHG footprint of grey hydrogen depends on the upstream methane leakage rate, ranging from 10.5 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for a methane emission rate of 0.2% to 11.5 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for a rate of 1.5% (Bauer et al. 2022a).
- The GHG footprint of blue hydrogen additionally depends on the CO<sub>2</sub> capture rate, either only capturing high concentration process CO<sub>2</sub> or also capturing CO<sub>2</sub> from natural gas combustion in the reformer furnace (Bauer et al. 2022a). Resulting ranges are 5.6–6.6 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for a CO<sub>2</sub> capture rate of 55% and 2.3–3.3 kgCO<sub>2</sub>-e/kg H<sub>2</sub> for a CO<sub>2</sub> capture rate of 93%, calculated using electricity with a GHG intensity of 10 gCO<sub>2</sub>-e/kWh (de Kleijne et al. 2022).
- Grey ammonia is produced via steam reforming of natural gas, with a global average GHG footprint for an optimised process of 2.0–2.5 kgCO<sub>2</sub>-e/kg NH<sub>3</sub> (Wang et al. 2021) – equivalent to 11.3–14.1 kgCO<sub>2</sub>-e per kg H<sub>2</sub> based on hydrogen content.
- Policy-based GHG emissions thresholds are listed in Table 5.5.

**Table 5.5. Selected certification schemes, standards and regulatory frameworks for different system boundaries (IEA 2023).**

Name	Type	Spatial scope	System boundary	Emission threshold (kgCO <sub>2</sub> -e/kg H <sub>2</sub> )
Green Hydrogen Standard	Voluntary	Global	Well-to-gate	1.0
UK Low carbon hydrogen standard	Regulatory	UK	Well-to-gate	2.4
EU Taxonomy	Regulatory	EU	Well-to-gate	3.0
Clean Hydrogen Production Standard	Regulatory	US	Well-to-gate	2.5-4 2.5-1.5 1.5-0.45 <0.45
TÜV SÜD Standard	Voluntary	EU	Well-to-point of delivery	2.7-2.8
EU Renewable Energy Directive II	Regulatory	EU	Well-to-wheel	3.38
Renewable Transport Fuel Obligation for transport sector	Regulatory	UK	Well-to-point of delivery	4.0





# 6

## Synthesis

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## 6.1 Introduction

The main aim of this thesis was to identify the conditions under which green hydrogen and carbon capture and utilisation technologies are Paris-compatible. In Chapter 2 I have set out the criteria for Paris compatibility of technologies, that is, for their deployment to be in line with reaching the Paris Agreement's long-term temperature goal: (i) having technology readiness level (TRL) 6 in 2020 for Paris compatibility in 2030, while for 2050 no technologies are excluded based on their TRL, and (ii) reducing life-cycle GHG emissions of the product with at least 50% in 2030 compared to 2020, and to zero in 2050.

The GHG footprints of CCU products and green hydrogen depend on the temporal, spatial and technological context, as assessed in Chapters 2 to 5. The influence of the temporal context was analysed in Chapters 2 and 4 through the changes in GHG intensity of the grid mix over time, the time-dependent choices regarding allocation of GHG emissions in case of multifunctionality, and the change of the benchmark over time. How GHG footprints depend on spatial context was addressed in Chapters 3 and 5, based on the use of country-specific GHG intensities of the grid mix, spatially-explicit GHG footprints of solar photovoltaic and wind electricity, and transport distance ranges. Variation in GHG footprints due to the technological context was assessed in all chapters, including various types of (renewable) electricity generation technologies, three types of electrolyser technologies, three types of CO<sub>2</sub> sources and 44 CO<sub>2</sub> use options, and four hydrogen transport modes.

This chapter discusses the outcomes of the previous chapters in the context of the Paris compatibility framework, draws broader conclusions, and suggests how the results can be used by decision-makers and researchers. How interactions between contextual factors determine GHG footprints, and therefore Paris compatibility, is presented for CCU in Section 6.2 and for green hydrogen in Section 6.3. A discussion and reflections on the findings of this thesis are presented in Section 6.4, followed by conclusions in Section 6.5, policy implications in Section 6.6 and a research outlook in Section 6.7.

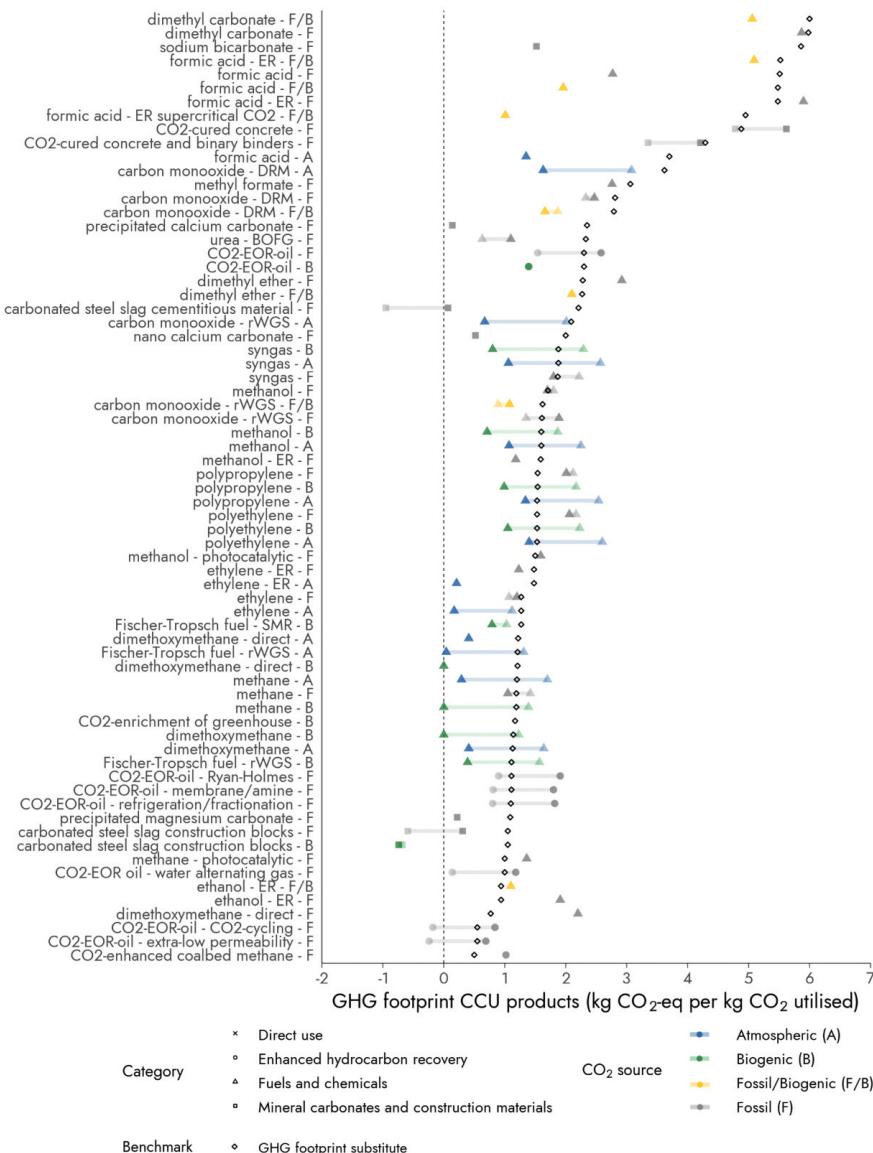
## 6.2 Paris compatibility of CCU

### ***Technological maturity***

Technological maturity of CCU technologies was assessed in Chapter 2. Of the 74 CCU processes considered, 47 employ technologies with a TRL of 6 or higher in 2020, covering capture from fossil and biogenic point sources and air, and subsequent use in enhanced oil recovery (EOR) and greenhouses, several thermochemical conversion processes to fuels and chemicals, CO<sub>2</sub>-curing of cement and some mineral carbonation processes. Additional CCU processes for 2050 are photocatalytic and electrochemical production of fuels and chemicals, and some other carbonation processes.

### ***GHG footprints of CCU in 2030 and 2050***

The GHG footprint of CCU products changes over time, reflecting the changing GHG intensity of electricity and approach to deal with CO<sub>2</sub>'s multifunctionality. Processes using biogenic and atmospheric CO<sub>2</sub> consistently have lower GHG footprints in 2050 (dark coloured shape in Figure 6.1) compared to 2030 (light coloured shape). For fossil CO<sub>2</sub>-based processes, a different CO<sub>2</sub>-allocation approach is applied in 2030 and 2050: in the 2030 substitution approach, a fossil point source without capture is assumed to be 1:1 substituted by the same plant with capture, effectively crediting the capturing process for avoiding the emission of CO<sub>2</sub>. In a 2050 Paris-compatible zero emissions system, *unabated* fossil point sources are no longer assumed to be available, and the fossil CO<sub>2</sub> capturing process is no longer credited for avoiding the emission of CO<sub>2</sub>. Whether or not the 2050 GHG footprint is lower than for 2030 therefore depends on the change in fossil CO<sub>2</sub>-allocation approach and on the reduction in electricity emissions. If at least 6 kWh per kg of CO<sub>2</sub> utilised is used in capture and conversion (including for hydrogen), fossil-CO<sub>2</sub>-based processes have a lower GHG footprint in 2050 compared to 2030. Examples are the production of methane, methanol, polyethylene and polypropylene (Figure 6.1).



**Figure 6.1. The development of the GHG footprint of CCU technologies from 2030 to 2050, based on data from Chapter 2 and Chapter 3.** Different shapes indicate to which CCU category a process belongs, where light shading of the shape indicates the 2030 GHG footprint and dark shading the 2050 GHG footprint. For technologies with a TRL of lower than 6 in 2020, only the 2050 GHG footprint is shown. The black diamond shapes indicate the 2020 GHG footprint of the substitute compared to which emissions must be reduced by 50% for 2030 Paris compatibility. Light grey shapes below zero do not indicate carbon dioxide removal, but emission reduction compared to emitting the CO<sub>2</sub>. Acronyms: ER = electrochemical reduction, DRM = dry reforming of methane, BOFG = basic oxygen furnace gas, EOR = enhanced oil recovery, rWGS = reverse water-gas shift, SMR = steam methane reforming.

### ***Paris compatibility of CCU in 2030***

Out of the 47 sufficiently mature processes, 8 resulted in an emissions reduction of at least 50% in 2030 compared to the 2020 benchmark, called the substitute. These processes are characterised by (i) low CO<sub>2</sub> capture and conversion emissions (e.g., by-product CO<sub>2</sub> from biogas separation or exothermic mineral carbonation), (ii) storing CO<sub>2</sub> permanently (e.g., construction materials, EOR) and/or (iii) replacing a GHG-intensive process (e.g., urea production from basic oxygen furnace gas (BOFG) replacing electricity production, saving emissions because other ways of electricity generation have lower emissions). In many CCU processes, capture and conversion process emissions are prohibitively high due to the amount of electricity (and heat) used, and the emissions associated with it, for example fuels and chemicals based on hydrogen: out of the 47 processes, 26 use hydrogen, of which only the process recovering hydrogen from BOFG is Paris-compatible, i.e., the 25 processes using electrolytic hydrogen are not.

Here, the Paris compatibility of CCU for several variations in context is explored, based on insights in Chapters 4 and 5 regarding the importance of the choice of electricity source and co-product accounting approach for the CCU processes using hydrogen. First, if additional renewable electricity capacity can be used, more CCU options (some fuels, chemicals and EOR processes) would be Paris-compatible in 2030, i.e., 18/47 for solar photovoltaic electricity and 25/47 for offshore wind electricity (Table 6.1). Even if offshore wind electricity is used, a very low-GHG electricity source, only about half of the options are Paris-compatible in 2030.

Second, for the 25 CCU processes in Chapter 2 using electrolytic hydrogen, co-produced oxygen is assumed to substitute oxygen production via cryogenic air separation. If oxygen cannot be valorised and is vented (Ch4), the GHG footprint of the CCU product would increase compared to when oxygen substitution is applied, because the emissions associated with oxygen production via air separation can no longer be subtracted. Whether or not this affects Paris compatibility in 2030 depends on the electricity source used. When using the 1.5°C grid mix, Paris compatibility (8/47) is not affected because already under the default assumption of oxygen substitution, no CCU processes based on electrolytic hydrogen were Paris-compatible. For offshore wind electricity, Paris compatibility (25/47) is not affected due to offshore wind electricity's very low GHG intensity, not leading to

the GHG footprint of the CCU product to change much. Only when electricity emissions are somewhere in between (solar PV), Paris compatibility in 2030 would be slightly lowered (17/47) (Table 6.1).

Third, the CCU processes using electrolytic hydrogen in Chapter 2 do not include hydrogen transport. Including emissions for hydrogen transport by pipeline over a distance of 1000 km (~1.5 kgCO<sub>2</sub>-e per kg H<sub>2</sub>, Ch5) does not lower Paris compatibility in 2030 when combined with the 1.5°C grid mix, but does when combined with wind electricity (17/47) and solar electricity (14/47), indicating that hydrogen transport may be an important life cycle stage to include in the analysis (Table 6.1).

Fourth, combining all 25 unique CO<sub>2</sub> use processes for 2030 with by-product CO<sub>2</sub> from biogas upgrading (Cuéllar-Franca et al. 2019; Oreggioni et al. 2019), and a direct air capture process (Deutz and Bardow 2021) led to some additional Paris-compatible processes (Table 6.1, Supplementary Information 6). For the combination with CO<sub>2</sub> from biogas upgrading, these include several EOR and chemicals processes. For the combination with fully electricity-based direct air capture, one process was added that initially relied on fossil-based heat in the capture step, but only when renewable electricity was used.

Additional insights are that, to realise the condition of low/zero emissions in capture and conversion: (i) using a low GHG electricity source is critical and Paris compatibility in 2030 depends on the type of renewable electricity source used; (ii) when hydrogen is used in the CCU process, Paris compatibility may depend on avoiding hydrogen transport, and only to a lesser extent on valorising co-produced oxygen; (iii) CO<sub>2</sub> capture without emissions (by-product CO<sub>2</sub>) or an electrified capture process using low GHG electricity could enhance Paris compatibility.

**Table 6.1. Paris compatibility of CCU based on Ch2,3 and considering alternative contexts.**

In bold the factor(s)/assumption(s) changed compared to the base case (first row) and resulting effect on number of Paris-compatible processes. Electricity GHG intensities are the based on the 1.5°C-compatible global grid mix (170 gCO<sub>2</sub> per kWh in 2030 and 0 gCO<sub>2</sub> per kWh in 2050; Ch2), and global median solar photovoltaic electricity and offshore wind electricity in 2030 and 2050 are used (Supplementary Information 6). In addition to the CO<sub>2</sub> sources in underlying studies ('Ref'), all unique CO<sub>2</sub> use processes are coupled to atmospheric CO<sub>2</sub> captured using a sorbent-based direct air capture process with a heat pump (DAC) (Deutz and Bardow 2021) and biogenic CO<sub>2</sub> from biogas upgrading (Bio) (Cuéllar-Franca et al. 2019; Oreggioni et al. 2019).

	Electricity source	CO <sub>2</sub> source	Oxygen co-product allocation*	Hydrogen transport*	GHG footprint substitute	2030 Paris-compatible	2050 Paris compatible
Base case	1.5°C grid	Ref	Substitution	Excluded	100%	8 out of 47	4 out of 74
Variations in context	<b>Solar</b>	Ref	Substitution	Excluded	100%	18	1
	<b>Wind</b>	Ref	Substitution	Excluded	100%	25	1
	1.5°C grid	Ref	<b>Venting</b>	Excluded	100%	8	4
	<b>Solar</b>	Ref	<b>Venting</b>	Excluded	100%	17	1
	<b>Wind</b>	Ref	<b>Venting</b>	Excluded	100%	25	1
	1.5°C grid	Ref	Substitution	<b>Included</b>	100%	8	1
	<b>Solar</b>	Ref	Substitution	<b>Included</b>	100%	14	1
	<b>Wind</b>	Ref	Substitution	<b>Included</b>	100%	17	1
	1.5°C grid	<b>Bio</b>	Substitution	Excluded	100%	9/25	9/42
	1.5°C grid	<b>DAC</b>	Substitution	Excluded	100%	6/25	9/42
	<b>Wind</b>	<b>Bio</b>	Substitution	Excluded	100%	16/25	9/42
	<b>Wind</b>	<b>DAC</b>	Substitution	Excluded	100%	15/25	9/42
	<b>Solar</b>	<b>Bio</b>	Substitution	Excluded	100%	15/25	9/42
	<b>Solar</b>	<b>DAC</b>	Substitution	Excluded	100%	13/25	8/42

\*applied to all CCU processes that use electrolytic hydrogen in conversion

### Paris compatibility of CCU in 2050

Out of the 74 CCU processes considered in Chapter 2, four were found to be Paris-compatible in 2050. Paris compatibility in 2050 requires meeting at least two out of three conditions: (i) a zero emissions capture and conversion process; (ii) using biogenic or atmospheric CO<sub>2</sub>, and (iii) permanent CO<sub>2</sub> storage. For CCU to be CO<sub>2</sub>-

removal, the last two conditions must be met, and capture and conversion must result in lower emissions than are stored permanently.

Chapter 2 uses a 1.5°C-compatible electricity mix with zero emissions in 2050, which implies that bioenergy with carbon capture and storage (BECCS; which can remove carbon dioxide from the atmosphere while producing electricity) would need to level out any remaining emissions of the other electricity sources. If renewable electricity is used instead, with a GHG intensity of larger than zero, or hydrogen transport is included when hydrogen is required, only processes with biogenic or atmospheric CO<sub>2</sub> stored permanently are still Paris-compatible, which is only one process out of the 74 processes assessed (Table 6.1).

Using atmospheric CO<sub>2</sub> or biogenic CO<sub>2</sub> in combination with the 42 unique CO<sub>2</sub> use processes for 2050 from Chapter 2 results in more processes to be Paris-compatible, including mineral carbonates and enhanced hydrocarbon recovery where atmospheric/biogenic CO<sub>2</sub> storage is higher than emissions from combustion of the recovered hydrocarbon, also when combined with renewable (non-zero GHG) electricity. While for CO<sub>2</sub> from biogas upgrading, in the case of zero-emission electricity, some fuels and chemicals are Paris-compatible (i.e., the first two conditions are fulfilled), this is not the case when renewable electricity is used.

Additional insights regarding the 2050 Paris compatibility conditions are that: (i) using biogenic or atmospheric CO<sub>2</sub> in products that store CO<sub>2</sub> permanently can be combined with solar and wind electricity; (ii) for the zero emissions capture and conversion condition renewable electricity does not suffice, and (iii) hydrogen transport must be avoided when hydrogen is needed in the CCU process.

## 6.3 Paris compatibility of green hydrogen

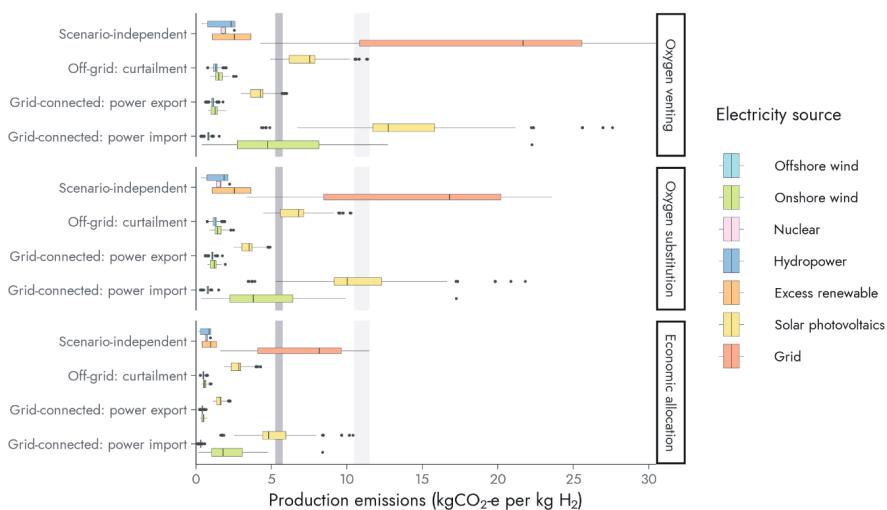
### *Technological maturity*

Electrolysers are in various stages of technological development. While alkaline (ALK) electrolysers reached TRL 9 before 2020, polymer electrolyte membrane or proton exchange membrane (PEM) electrolysers were in TRL 6-8 in 2020, and solid oxide electrolysers cells (SOEC) in TRL 5 (Pinsky et al. 2020; Khan et al. 2022)

– yet, SOECs reached TRL 7 in 2022 (IEA 2022a) and are therefore included in the rest of the analysis, although possibly requiring additional development to ensure readiness in 2030. Hydrogen liquefaction and conversion to ammonia, transport by pipeline, and shipping liquid hydrogen and ammonia are all in TRL 6 or higher (IEA 2022a; IRENA 2022a). Ammonia cracking is reported to be in TRL 4 by IEA (2022a) while in TRL 8 by IRENA (2022a): The former is low-temperature cracking, while the latter is high-temperature cracking, and the only option included in the analysis.

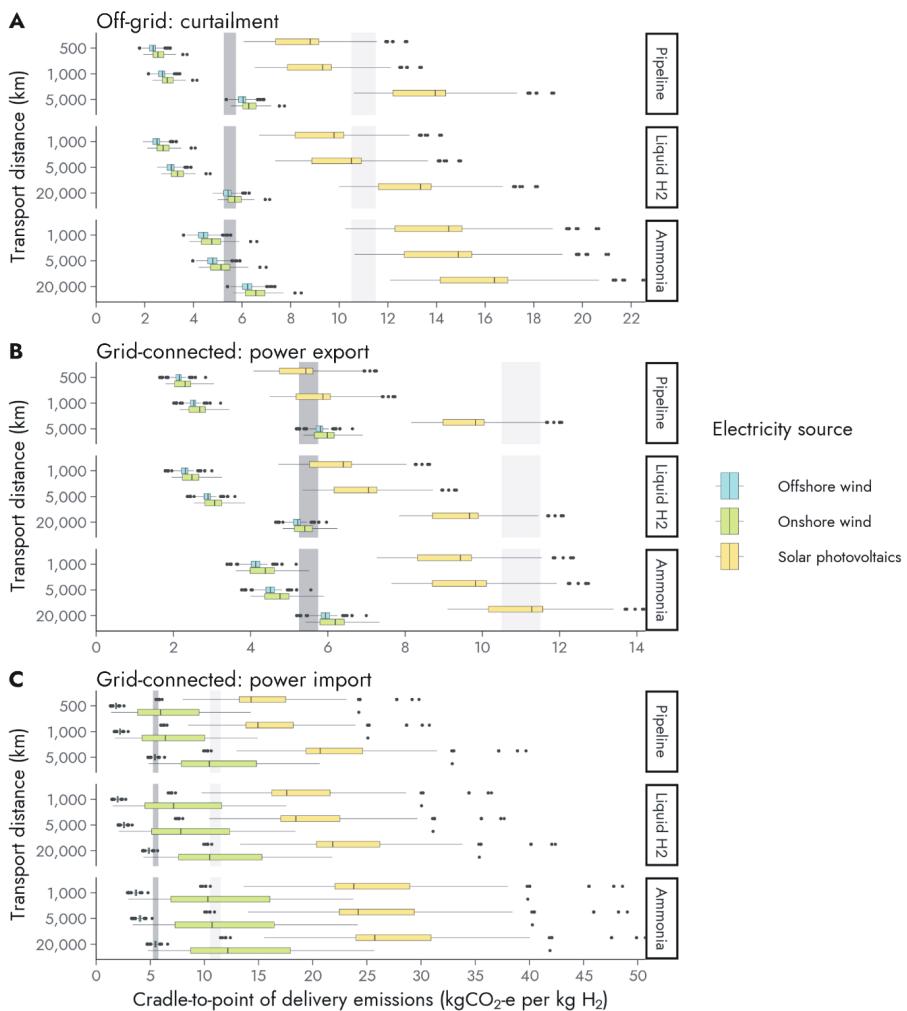
### ***GHG footprints of green hydrogen in 2030***

Cradle-to-production gate GHG emissions for electrolytic hydrogen are 0.2 – 30.5 kgCO<sub>2</sub>-e per kg H<sub>2</sub> (full range based on Chapters 4 and 5) (Figure 6.2). Production emissions depend on the allocation principle applied to co-produced oxygen (oxygen venting, oxygen substitution or economic allocation; Ch4, Figure 6.2), and on whether newly installed, additional renewable electricity capacity is used. Because if not, the electrolyser effectively runs on grid electricity (Ch4), typically resulting in emissions exceeding those of grey hydrogen production (Ch5). If additional renewable capacity is used, GHG footprints depend on the type and location of the renewable electricity technology (Ch4, 5), and on the configuration of the electrolyser and electricity facility (Ch5), modelled in three scenarios. In the *Off-grid: curtailment* scenario, the renewable electricity facility is oversized compared to the electrolyser and combined with a battery and electricity curtailment: emissions from manufacture of the full scale of the electricity facilities are allocated to hydrogen production. If a grid connection is in place and electricity surplus is used by other economic activities, these emissions need not be allocated to hydrogen (*Grid-connected: power export* scenario), resulting in lower GHG footprints than in the *Off-grid: curtailment* scenario. In the *Grid-connected: power import*, the renewable generation capacity equals the electrolyser capacity, and grid electricity is used to complement renewable electricity, which can again result in emissions exceeding those of grey hydrogen production (Figure 6.2).



**Figure 6.2. Green hydrogen cradle-to-production gate GHG emissions by electricity source and oxygen co-product accounting approach.** Interquartile ranges across projects by electricity source are presented per hydrogen production configuration scenario for intermittent renewables (Ch5), with whiskers extending to the smallest/largest value no further than  $1.5 \times$  lower/upper quartile. Hydropower, nuclear, grid and excess-based hydrogen production is independent of these hydrogen production configuration scenarios (scenario-independent). Oxygen co-product accounting approaches described in Chapter 4 were applied to the project-specific production emissions from Chapter 5. Benchmarks are grey hydrogen production emissions (light grey area), and 50% of the grey hydrogen production emissions (dark grey area), representing the threshold for 2030 Paris compatibility.

Cradle-to-point of delivery GHG emissions for electrolytic hydrogen are 1.3 – 50.7 kgCO<sub>2</sub>-e per kg H<sub>2</sub> (full range based on Chapter 5). These emissions depend on, in addition to the factors on which production emissions depend, transport mode (pipeline, liquid hydrogen shipping or ammonia shipping combined with cracking to recover the hydrogen) and transport distance (Figure 6.3). While transport as ammonia with cracking results in highest emissions, below 1000 km pipelines and liquid hydrogen shipping are approximately equal, and for transport distances above 1000 km liquid hydrogen shipping results in lowest emissions.



**Figure 6.3. Green hydrogen cradle-to-point of delivery GHG emissions by electricity source, presented per renewable electricity configuration scenario, transport mode and transport distance.** The figure is based on project-specific data from Chapter 5, for hydrogen transport via pipeline or ship as liquid hydrogen or ammonia, where ammonia is cracked at the destination. Interquartile ranges across projects are shown, and whiskers extend to the smallest/largest value no further than 1.5 \* lower/upper quartile. Co-produced oxygen is assumed to be vented. Benchmarks are grey hydrogen production emissions (light grey area), and 50% of the grey hydrogen production emissions (dark grey area), representing the threshold for 2030 Paris compatibility.

### ***Paris compatibility of green hydrogen in 2030***

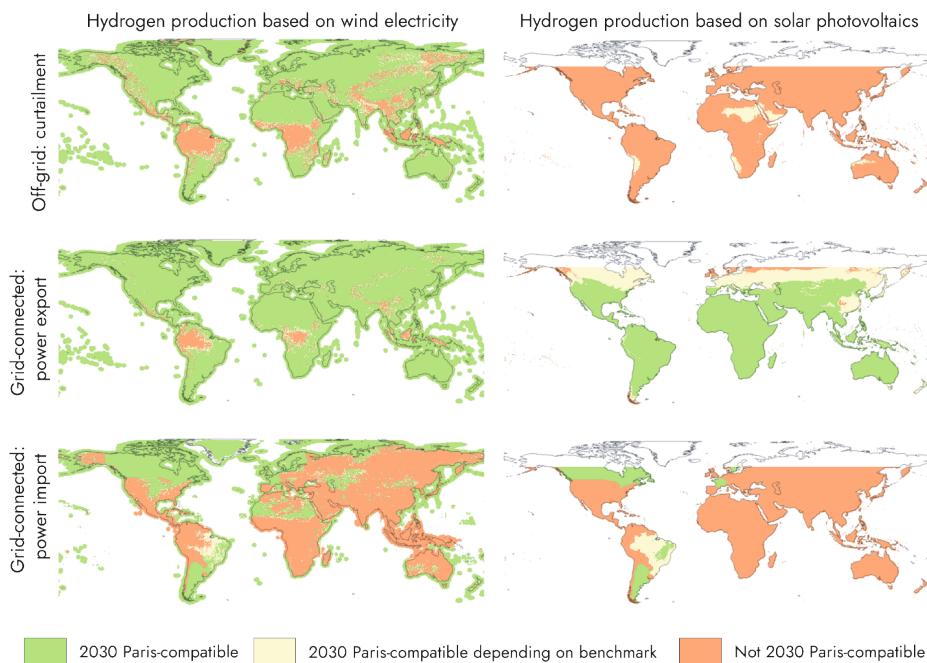
Green hydrogen can be used in many different applications, but substituting grey hydrogen is considered a near-term priority (IEA 2021), making this a good benchmark for determining 2030 Paris compatibility (Figure 6.2, Figure 6.3): Green hydrogen is considered Paris-compatible in 2030 if its GHG footprint is at least 50% lower than the GHG footprint of grey hydrogen.

The GHG intensity threshold of electricity powering electrolysis below which green hydrogen production would be 2030 Paris-compatible is 78 to 143 gCO<sub>2</sub>-e per kWh, and depends on the combination of four factors: (i) electrolyser technology, (ii) electrolyser operation (continuous or intermittent), (iii) oxygen co-product accounting approach, and (iv) the GHG footprint of the grey hydrogen benchmark (10.5 to 11.5 kgCO<sub>2</sub>-eq per kg H<sub>2</sub>, depending on upstream methane leakage rate; Bauer et al. 2022) (Table 6.2). Production using the 2030 global grid mix modelled for a 1.5°C scenario (170 gCO<sub>2</sub>-e/kWh; Ch2) would not be Paris-compatible regardless of electrolyser technology and co-product accounting as it exceeds the threshold for all combinations of these four factors. However, for the 2030 EU 1.5°C grid mix (100 gCO<sub>2</sub>-e/kWh, Ch4), green hydrogen production is Paris-compatible if oxygen is valorised, or if oxygen is vented while compared to the higher-emission grey hydrogen benchmark. Although this attractive feature of utilising and crediting oxygen is part of recent assessments of green hydrogen (Biernacki et al. 2018; DISER 2021; Brigljević et al. 2022), a reliable, long-term consumer of oxygen would then have to be found. In the case of an electrolyser operated in an intermittent fashion, i.e., powered by intermittent renewables, the Paris compatibility threshold per kWh of electricity is lower due to higher emissions per kg of hydrogen from electrolyser manufacturing and for the manufacture of a battery (Table 6.2).

**Table 6.2. Thresholds to GHG intensity of electricity used in electrolysis for 2030 Paris compatibility of green hydrogen production.** Electricity GHG intensity thresholds below which green hydrogen production is 2030 Paris-compatible depends on the combination of electrolyser technology and operation (continuous operation or an electrolyser utilisation factor of 0.5 when using intermittent renewables), grey hydrogen benchmark (with 0.2% or 1.5% upstream methane leakage), and oxygen co-product accounting approach (oxygen venting or substituting oxygen production elsewhere). Calculated based on data from Chapter 5, combined with co-product accounting approaches from Chapter 4.

Co-product accounting		$O_2$ venting		$O_2$ substitution	
GHG footprint grey H <sub>2</sub> benchmark (kgCO <sub>2</sub> -e/kg H <sub>2</sub> )		10.5	11.5	10.5	11.5
Operationality	Electrolyser technology	Electricity GHG limit 2030 (gCO <sub>2</sub> -e/kWh)		Electricity GHG limit 2030 (gCO <sub>2</sub> -e/kWh)	
<b>Continuous</b>	Alkaline	100	110	130	143
	Polymer electrolyte membrane	94	104	119	131
	Solid oxide	90	102	122	138
<b>Intermittent</b>	Alkaline	86	96	111	123
	Polymer electrolyte membrane	85	93	105	116
	Solid oxide	78	90	105	121

Whether or not the emissions associated with using the intermittent renewable electricity source stay below the 2030 Paris compatibility thresholds listed in Table 6.2 depends on: (i) the specific renewable energy generation technology used, (ii) the location-specific solar irradiation and wind speed, as these determine the local GHG footprints of solar and wind electricity, and (iii) on the configuration of the electrolyser and electricity facility. Figure 6.4 shows that using wind electricity results in Paris-compatible hydrogen production in most, but not all, locations, unless power is imported from the grid as a complement when its GHG intensity is higher than approximately 100 gCO<sub>2</sub>-e per kWh. Hydrogen from solar electricity is Paris-compatible in most locations compared to the higher grey hydrogen benchmark, as long as curtailment is prevented.



**Figure 6.4. 2030 Paris compatibility of green hydrogen production from wind (onshore and offshore) and solar photovoltaic electricity per hydrogen production configuration scenario.**

The location-specific cradle-to-production gate emissions of hydrogen production using a polymer electrolyte membrane electrolyser are based on Chapter 5, calculated using local wind speed and solar irradiation, national 2030 grid mixes modelled for a 2°C scenario when power is imported, and assuming co-produced oxygen is vented. Paris compatibility is determined based on whether or not an emission reduction of 50% is achieved compared to grey hydrogen production benchmarks with different upstream methane leakage rates.

Countries with lower renewable energy potentials and a high future green hydrogen demand are looking at importing from locations high renewable energy potentials. In this case, emissions from hydrogen conversion, transport, reconversion and storage must be included in the Paris compatibility assessment. Long-distance transport of liquefied solar-based hydrogen by ship from the locations with highest solar irradiance to places with low solar irradiance is Paris-compatible when compared to the higher grey hydrogen benchmark, for transport distances below 7,000 km (based on data from Chapter 5, *Grid-connected: power export* scenario, see also Figure 6.3). This would mean Europe is just within reach of South America. Pipeline transport of solar-based hydrogen would also be Paris-compatible for distances below 1,500 km, e.g., from North Africa to Europe (based on data from Chapter 5, Figure 6.3). Wind-based hydrogen produced at best wind

locations can be shipped as liquid hydrogen or ammonia for distances of 20,000 km and still be Paris-compatible due its low production emissions, or up to 5,000 km by pipeline (Figure 6.3).

In summary, for green hydrogen to be Paris-compatible in 2030:

- Electricity powering electrolysis must have a GHG intensity below 78-143 gCO<sub>2</sub>-e per kWh, depending on electrolyser technology, electrolyser operational hours, oxygen co-product accounting and grey hydrogen benchmark, when hydrogen is produced where it is used.
- When using additional renewable electricity, production must be at locations with high wind speeds or solar irradiation, grid import must be avoided unless the grid mix has a GHG intensity below approximately 100 gCO<sub>2</sub>-e per kWh, and avoiding curtailment is particularly critical for hydrogen production from solar electricity.
- Hydrogen can be imported from locations with high renewable energy potentials to locations with low renewable energy potentials for distances up to 7,000 – 20,000 km as liquid hydrogen or 1,500 – 5,000 km by pipeline, depending on whether solar or wind electricity is used.

### ***Paris compatibility of green hydrogen in 2050***

The GHG footprint of green hydrogen in 2050 depends on the extent to which emissions from the manufacture of solar photovoltaic panels (Louwen et al. 2016), wind turbines (Caduff et al. 2012), the electrolyser and battery, and any other components for conversion, transport and storage are reduced, or lower-emission alternatives are found: e.g., for storage (Gabrielli et al. 2020b), low-temperature cracking of ammonia (IEA 2022a), and using hydrogen or hydrogen-based fuels for shipping (Wijayanta et al. 2019; Al-Breiki and Bicer 2020; Ishimoto et al. 2020; McKinlay et al. 2021). Grid-based production could have lower emissions than solar or wind-based production in 2050 due to the introduction of BECCS, which may in some locations lead to a zero or negative GHG intensity of electricity (Rogelj et al. 2018; Hanssen et al. 2020). When renewable surpluses become more frequent in an energy system increasingly reliant on intermittent renewables, electrolyzers

running on surplus can increase operational hours, resulting in a lower contribution of electrolyser manufacturing emissions per kg of hydrogen.

Due to hydrogen's global warming potential (Sand et al. 2023), any losses and leakages during transport and storage already result in a GHG footprint of larger than zero. This means that hydrogen produced from only wind or solar power (with remaining life-cycle emissions) is not Paris-compatible in 2050 according to the Paris compatibility criteria. Green hydrogen can only be Paris-compatible in 2050 when using negative-GHG electricity to compensate for leakages and emissions from manufacturing and transport.

## 6.4 Discussion & reflections

Following the insights presented in Chapters 2-5, several questions have arisen in discussion with academics, industry representatives, journalists and policymakers. This section provides reflections on these issues based on existing literature and using and integrating the knowledge obtained throughout the research presented in this thesis. It furthermore discusses the contribution of this work to the existing literature base.

### *How does the Paris compatibility framework compare to other technology assessment frameworks?*

The Paris compatibility framework considers technology maturity and GHG emissions reductions for technologies in 2030 and 2050, which may lead to different evaluations of technologies in the near term and long term. It highlights the urgency of deploying currently available technologies to reduce emissions in line with limiting warming to 1.5°C in the near term, while further developing and scaling promising low-TRL technologies with potentially very low/zero GHG footprints (Ch2). Low-TRL technologies that do not have the potential for emissions reductions far beyond conventional technologies may not be Paris-compatible at all.

The Paris compatibility assessment of green hydrogen and CCU technologies uses emission reduction targets for 2030 and 2050 based on global CO<sub>2</sub> emission reduction pathways limiting warming to 1.5°C, because technologies in energy and industry mainly need to tackle CO<sub>2</sub> emissions (Bashmakov et al. 2022; Clarke et al. 2022). Yet, other GHGs also play a role, such as the emission of hydrogen contributing to CO<sub>2</sub>-equivalent emissions (Sand et al. 2023): therefore, I assessed Paris compatibility based on CO<sub>2</sub>-equivalent emissions, i.e., GHG footprints. This is in line with policy targets for industry (net-zero GHG in 2050; see Chapter 1), the hydrogen production criteria of the Climate Bond's Paris-aligned standard and certification scheme (Climate Bonds 2023), and Bataille et al.'s (2018) criteria for energy-intensive industry to be consistent with the Paris Agreement of reducing GHG emissions to very low, zero or negative levels. Regardless of the net-zero GHG year, the assessment allows for identifying priority technologies and emissions drivers to be tackled to reduce emissions further.

In my Paris compatibility framework, a technology is assessed individually, against a background system that is evolving (e.g., using a 1.5°C-compatible electricity mix for 2030 and 2050, Ch2), considering the full life cycle, without carbon offsetting or carbon dioxide removal. This, as opposed to for example strategies of oil majors to reach pledged net-zero emissions by 2050 (Trencher et al. 2023), transparently shows the remaining emissions. In addition to the individual technology assessment I followed in my thesis, a system-level perspective may be needed as well: To reduce emissions to net-zero by mid-century, either all technologies' emissions contributing to the industrial transition must go to zero individually, or some technologies would need to lead to carbon removal (e.g., combining pulp and paper with CO<sub>2</sub> capture and storage, or carbonation of alkaline wastes using biogenic or atmospheric CO<sub>2</sub>). Alternatively, remaining emissions would need to be compensated by carbon dioxide removal to reach net-zero CO<sub>2</sub> in the industrial sector (Gabrielli et al. 2023).

The Paris compatibility framework is relatively narrow with a focus on GHG emissions and technology maturity compared to the much broader 'feasibility assessment' framework aimed at identifying barriers and enablers of mitigation options across geophysical, environmental-ecological, technological, economic, sociocultural and institutional dimensions set out by Steg et al. (2022). It could

provide a valuable complementary perspective, since addressing a wide variety of barriers and enablers is likely a condition for successful implementation of Paris compatible technologies. The Paris compatibility framework dives much deeper into the indicator 'maturity and technology readiness' for technological feasibility, while GHG footprints are not used as indicator in the feasibility assessment framework (Steg et al. 2022), although this may in fact be a necessary additional indicator as it is not a given that mitigation options reduce emissions in all contexts. Another complementary perspective is offered through the planetary boundaries framework, including nine thresholds that define a safe operating space, of which climate change is only one. This framework shows which technologies are beneficial on multiple fronts (Meng et al. 2023), but also which trade-offs exist: e.g., the technology scoring best on GHG emissions reduction in the petrochemical industry transition (CCU based on electrolytic hydrogen powered by BECCS) surpasses the biosphere integrity threshold (Galán-Martín et al. 2021).

The Paris compatibility framework could be extended to include other environmental impacts to avoid burden shifting (Meys et al. 2021). Furthermore, a broader interpretation of Paris compatibility could inform the inclusion of indicators to reflect the final part of Article 4.1: "*on the basis of equity, and in the context of sustainable development and efforts to eradicate poverty*" (UNFCCC 2016). These could be equity and justice indicators (Dillman and Heinonen 2022; Swennenhuis et al. 2022), and attainment of sustainable development goals (Falcone et al. 2021; Ioannou et al. 2022). Alternatively, other frameworks could be developed along the principles of the Paris compatibility framework. For example, a 'Kunming-Montreal Global Biodiversity compatibility framework', with a focus on reducing the loss of areas of high biodiversity importance and high ecological integrity in 2030 to near zero, and halting human-induced extinction of threatened species and reducing the rate of extinction of all species tenfold by 2050 (UNCBD 2022).

The Paris compatibility framework could also distinguish more specific emission reduction targets depending on the local or technological context. Following the principle of 'common but differentiated responsibilities and respective capabilities, in the light of different national circumstances' (UNFCCC 2016), the

Paris compatibility framework could distinguish regional emission reductions targets. If regions would aim to reduce emissions quicker than the global average pathway, technology readiness levels would also need to be more critically assessed: TRL may need to be higher than 6 in 2020 for the 50% year, and TRL 1 insufficient for the zero-emissions year. Furthermore, achieving deep emissions reductions in all sectors relies on a range of mitigation technologies, and not every technology will contribute the same to reducing emissions at the same moment in time. The energy system transition could enable technologies in other sectors to reduce emissions, as has become clear in this thesis. This interdependency is not unique: the use of low-GHG solar or wind-based hydrogen could in turn help reduce emissions in other processes, for example steel production. Importantly, and counterintuitively, this means that technologies would need to be evaluated against different benchmarks in a given year. Green hydrogen could be a candidate for stricter emission reduction criteria in the near term, for example at least 70%, as used in the European Commission Taxonomy's threshold for sustainable hydrogen production (European Commission 2021).

### *How does the 'credit' for CO<sub>2</sub> capture depend on the CO<sub>2</sub> source and the moment in time?*

Energy requirements and emissions associated with CO<sub>2</sub> capture depend on the CO<sub>2</sub> source and purity (von der Assen et al. 2016; Müller et al. 2020a). Besides that, who is allocated the 'credit' for capturing CO<sub>2</sub> must be carefully considered to avoid double counting. CCU cannot simultaneously make traditionally CO<sub>2</sub>-emitting industries carbon-neutral (Kätelhön et al. 2019; Lau et al. 2016) while also producing carbon-neutral CCU products (Patrício et al. 2017b; Ioannou et al. 2020). The reality is that if CO<sub>2</sub> is eventually emitted, it must be accounted for by one of the processes or shared among them. Taking the example of urea production from residual gases in the steel industry of Chapter 3: steel production cannot claim to avoid emitting CO<sub>2</sub> used in urea production, when these avoided emissions are claimed by urea production as well. One solution could be to expand the system boundaries to include the entire system (Fernández-Dacosta et al. 2019), but this does not suffice when the GHG footprint of only the CCU product is the topic of interest.

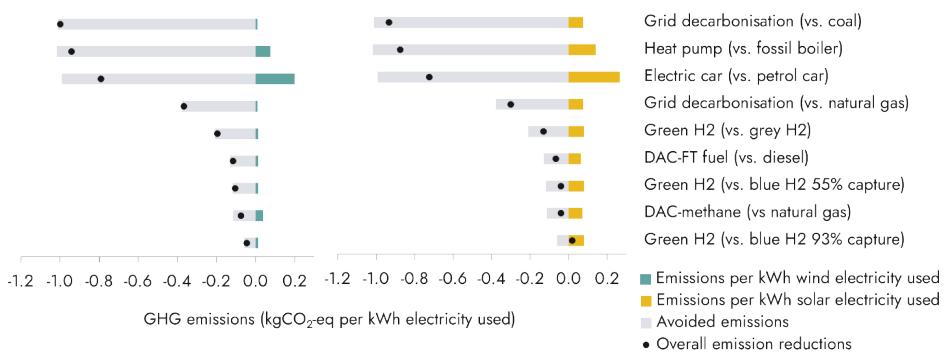
In the CCU LCA guidelines, the recommended approach is to allocate all emissions caused by changing the primary CO<sub>2</sub>-producing process to the CO<sub>2</sub> stream, including the emissions caused by the capture process and the avoided emission of CO<sub>2</sub> to air (Müller et al. 2020b). This is the approach followed in Chapter 2 for all processes in 2030. As noted by Moretti (2023), this recommended approach does not distinguish between fossil and biogenic sources of CO<sub>2</sub>: in both cases the CO<sub>2</sub> that is avoided from entering the atmosphere is credited to the CCU product, and the primary process may take credit for any CO<sub>2</sub> uptake in biomass when using the biogenic source.

However, fossil CO<sub>2</sub> cannot be used indefinitely to produce products in which the CO<sub>2</sub> is not stored permanently, as it prevents closing the carbon cycle (Daggash et al. 2018). This is also noted by the IEA: "*over time, and as fossil fuel use declines, the climate benefits associated with displacement will be reduced and the CO<sub>2</sub> used must increasingly be sourced from biomass or through direct air capture*" (IEA 2019a). In a 2050 Paris-compatible world, no unabated fossil CO<sub>2</sub> sources remain to be displaced (Ch2). Then, no such CO<sub>2</sub> avoidance credit is allocated to fossil CO<sub>2</sub> sources. This time-dependent distinction in accounting method is also followed in other assessments of CCU in the future (TNO and &flux 2022), and is in line with European Commission regulations, although the year in which the switch is made is earlier (2035 or 2040), depending on the sector of fossil CO<sub>2</sub> source: "*captured emissions from the combustion of non-sustainable fuels for the production of electricity should be considered avoided emissions up to 2035, as most should be abated by that date, while emissions from other uses of non-sustainable fuels should be considered avoided emissions up to 2040, as these emissions will remain longer*" (European Commission 2023d).

### *Is using scarce renewable electricity for green hydrogen and CCU beneficial for the climate?*

The use of renewable electricity may lead green hydrogen and CCU to be Paris-compatible, as has been shown in this thesis. However, green hydrogen and many CCU processes require large amounts of it. The high requirement of low-GHG electricity has been raised as a potential problem in previous research on hydrogen and hydrogen-based e-fuels (Ueckerdt et al. 2021), and the chemical industry: 18

PWh of low-emission electricity would be needed for production of chemicals in 2030 via CCU (55% of global electricity supply in 2030) (Kätelhön et al. 2019), and up to 32 PWh for global production of primary chemicals, mainly for hydrogen, in 2050 (Gabrielli et al. 2023). Low-GHG electricity is also a critical requirement for reducing emissions in other sectors: besides transitioning from fossil fuel-power, some examples are electrification of transport and household heating (Knobloch et al. 2020), aviation (Becattini et al. 2021; Sacchi et al. 2023) and for direct air CO<sub>2</sub> capture and storage (de Jonge et al. 2019; Deutz and Bardow 2021). Renewable electricity can therefore be considered a scarce resource, to which the question applies what its climate-optimal use is (Hanssen and Huijbregts 2019): Although the green hydrogen and CCU processes shown in Figure 6.5 reduce emissions compared to the 2020 conventional process, other use applications of renewable electricity achieve higher emissions reductions. This has been called the "opportunity cost" of renewable energy use for green hydrogen (Mac Dowell et al. 2021).



**Figure 6.5. Greenhouse gas emissions reductions for different applications of newly built renewable electricity capacity.** Based on Chapter 4, in addition including two processes from Chapter 2 that use atmospheric CO<sub>2</sub> to produce a fuel or methane. DAC = direct air capture, and FT = Fischer-Tropsch.

### Are green hydrogen and CCU then a bad idea?

Using scarce renewable electricity for green hydrogen and hydrogen/electricity-intensive CCU processes may not result in the largest climate benefit per kWh compared to other (replacement) options to use renewable electricity (Figure 6.5). However, what Figure 6.5 does not show is that hydrogen and CCU could utilise renewable electricity at locations and times of surplus, to produce products or energy carriers for later use (Sternberg and Bardow 2015; Mikulčić et al. 2019; Wang

et al. 2019), although using electrolyzers only for a small fraction of the year is currently still prohibited by high costs (Wevers et al. 2020).

In fulfilling the need for services and products in our economy, green hydrogen and CCU may be the best option compared to alternative mitigation options for the same services. Blue hydrogen can fulfil an important role in the near term until renewables are less scarce, but ultimately production in an optimised green hydrogen system allows for reaching lower emissions than blue hydrogen with high CO<sub>2</sub> capture (Ch4.5; Mac Dowell et al. 2021). Considering that e-fuels require 2-14 times more electricity than direct electrification to supply useful energy, Ueckerdt et al. (2021) advise to use to a merit order of applications of hydrogen and e-fuels to prioritise use cases that cannot be electrified.

CCU is one of the candidates of providing an alternative carbon feedstock for carbon-based products, conventionally directly produced from fossil fuels. Given that carbon-based products will be needed in the future, CCU must be compared to other carbon-feedstock options: biomass-based production and recycling of plastics. Biomass-based production of platform chemicals typically results in lower GHG emissions than CCU, unless BECCS is used for hydrogen production in the CCU process (Galán-Martín et al. 2021). However, land-use and biodiversity impacts are higher for the bio-based routes (Gabrielli et al. 2020a; Meys et al. 2021). Compared to plastic waste as carbon source for platform chemicals, CCU only results in lower emissions when the electricity's GHG intensity is lower than 39.5 gCO<sub>2</sub>-eq/kWh, and in a scenario additionally including biomass, CCU would even only be used when the GHG intensity is below 8.6 gCO<sub>2</sub>-eq/kWh (Meys et al. 2021). This value is below the current median GHG footprint of wind electricity (Dammeier et al. 2023). Introducing demand-side measures in addition to changing feedstocks may limit land and water scarcity through increased circularity and reduced production (Gabrielli et al. 2023). This points towards a strategy focused on demand measures, biomass and plastic waste, and evaluating what is needed in terms of CCU to close the gap, while ensuring that no fossil CO<sub>2</sub> enters the atmosphere after 2050 through preventing the use of fossil CO<sub>2</sub> in products with a non-permanent lifetime.

## 6.5 Conclusions

Based on the findings in this thesis, the following conclusions are drawn on the Paris compatibility of green hydrogen and CO<sub>2</sub> capture and utilisation.

- Green hydrogen and CCU technologies do not inherently reduce emissions. Emission reductions depend on the temporal, spatial and technological context in which these technologies are implemented. Ultimately, the combination of technology choices and requirements, electricity source, system configuration, CO<sub>2</sub> source, location and point in time determine their GHG footprints. Footprints of CCU and green hydrogen can be several times lower or several times higher than those of the (fossil) product that is intended to be replaced.
- Paris compatibility of CCU and green hydrogen requires several (but not necessarily all) of the following conditions to be fulfilled: (i) low energy requirements, (ii) low or zero GHG intensity of electricity, (iii) using biogenic or atmospheric CO<sub>2</sub>, (iv) permanent CO<sub>2</sub> storage in a product, (v) valorisation of co-products, and (vi) minimising or avoiding hydrogen transport. Low-TRL technologies that do not have the potential for deep emissions reductions are not be Paris-compatible at all.
- Renewable electricity use provides no guarantee for Paris compatibility of green hydrogen or CCU- not even when produced with newly-built additional capacity or, in the case of CCU, when combined with the use of atmospheric or biogenic CO<sub>2</sub>. Paris compatibility depends on the specific renewable electricity technology used, on the location-specific GHG emissions of renewable electricity, and on whether or not curtailment of surplus electricity can be avoided.
- Paris compatibility of CCU and hydrogen are interrelated: CO<sub>2</sub>-based fuels and chemicals preferably use Paris-compatible hydrogen, and hydrogen production is preferably co-located with industries with hydrogen demand to avoid transport emissions.

- Expectations of what 'technofixes' can achieve require recalibration. It is a big challenge for CCU and green hydrogen to live up to the expectations of being 'zero-emissions' or 'climate-neutral' technologies when full life-cycle emissions are taken into account. In the majority of the cases, zero emissions are far from achieved. Switching to these technologies alone is therefore not enough to reach zero emissions in industry.

## 6.6 Policy implications

- Standards and certification schemes could use Paris compatibility criteria to guide benchmarking and evaluation of GHG footprints, by including emission reductions to be achieved per year based on Paris-compatible emission reduction pathways.
- Research and policy should focus on accelerating the development of low-TRL technologies that may achieve (close-to-)zero emissions in the long term, and support the implementation of more mature technologies that can halve emissions in the near term.
- Technology assessments, for their footprint analysis, could follow these questions: (i) does the GHG footprint cover the full supply chain (including energy needs, manufacturing of components, losses, transport and use emissions)?; (ii) How close to zero emissions can the technology reduce emissions?; (iii) How does this technology compare to alternative technologies (under development) to fulfil in this need, now and in the future?; (iv) For which use cases is the technology uniquely situated, and for which not?; (v) Does the process require scarce resources such as renewable electricity, and is it the best use of these resources?
- Although renewable energy generation technologies may have zero *direct* emissions, their life-cycle GHG emissions are non-negligible. Treating them as zero-emissions technologies obscures the difference in GHG intensities between renewable energy technologies and between their GHG intensities at different locations. This is problematic because Paris compatibility of

green hydrogen and CCU depends on these differences. A life cycle approach is needed to identify combinations of technologies and locations that result in emissions below a given benchmark.

- The large amounts of low-GHG electricity required for green hydrogen and CCU will exert a high pressure on the energy system, especially since the demand for low-GHG intensity electricity increases in many sectors at the same time. If industries or countries intend to implement green hydrogen and CCU to reduce emissions, focus areas should be to (i) increase efficiency in production and transport; (ii) increase the supply of low-emission electricity and, (iii) reduce life-cycle emissions of renewable energy technologies.
- Capture of fossil CO<sub>2</sub> and biogenic CO<sub>2</sub> for utilisation is currently treated the same in policies, i.e., both are capturing CO<sub>2</sub> that would otherwise have entered the atmosphere. In the future, reaching zero emissions and closing the carbon cycle requires fossil CO<sub>2</sub> to be distinguished from biogenic and atmospheric CO<sub>2</sub>.
- From a life-cycle perspective, reaching zero emissions is very difficult. For green hydrogen it is practically impossible and for CCU only under strict conditions. This implies that reducing demand and increasing product lifetimes are critical pieces of the puzzle. Instead of labelling CCU and green hydrogen as zero-emissions solutions, national long-term strategies need to account for the contributions of these technologies (and hence these industries) to residual emissions. The requirement of carbon dioxide removal to compensate for these residual emissions to reach net-zero emissions at the system level could be included in the cost of the products of industry using such technologies, rather than spreading it out over society.

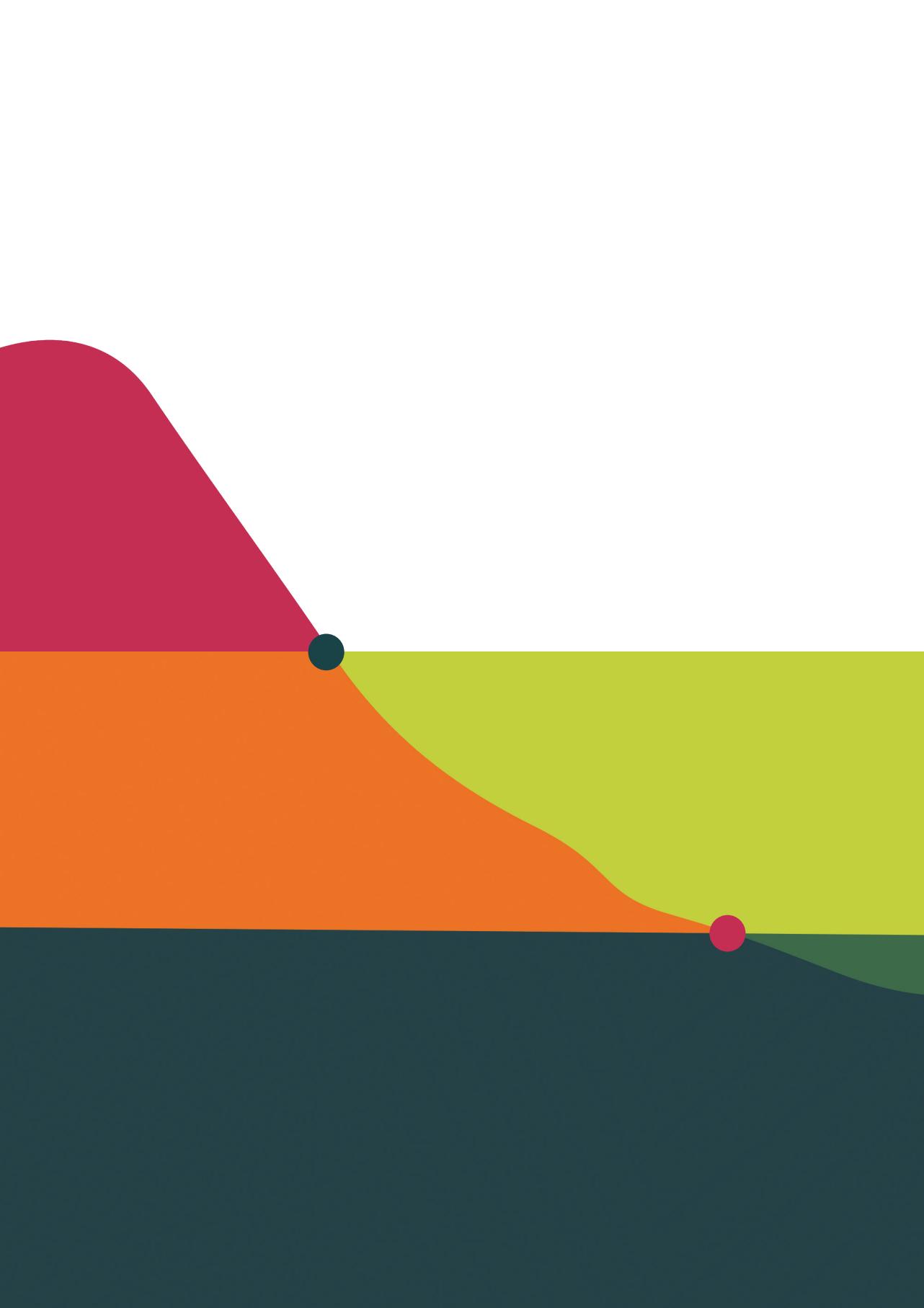
## 6.7 Research outlook

- Paris compatibility of technologies requires more than only reducing GHG emissions in time, so the framework could be extended to include, for example, other environmental impacts, and equity, justice or sustainable

development indicators. In addition, alternative frameworks could be developed along the principles of the Paris compatibility framework, such as a 'Kunming-Montreal Global Biodiversity compatibility framework'.

- The Paris compatibility of green hydrogen can be analysed for low-TRL technologies such as low-temperature cracking of ammonia, alternative hydrogen vectors and using boil-off gases as fuel for the hydrogen maritime vessel. For CCU, alternative heat sources can be included in the assessment, as most processes relied on fossil fuels for heat: e.g., 'waste' heat (while considering its limited supply and current use), electric heat pumps, hydrogen, or natural gas with CO<sub>2</sub> capture and storage.
- The assessment can also be extended to other mitigation technologies in the energy and industry sector, and to other sectors, such as transport, buildings or agriculture. As sectors move at a different pace towards GHG neutrality, and will have different shares of residual emissions at the year of net-zero at the system-level, sector-level emission reduction targets could be derived to guide benchmark setting for Paris compatibility assessments. Furthermore, region-specific emission reduction targets could be used.





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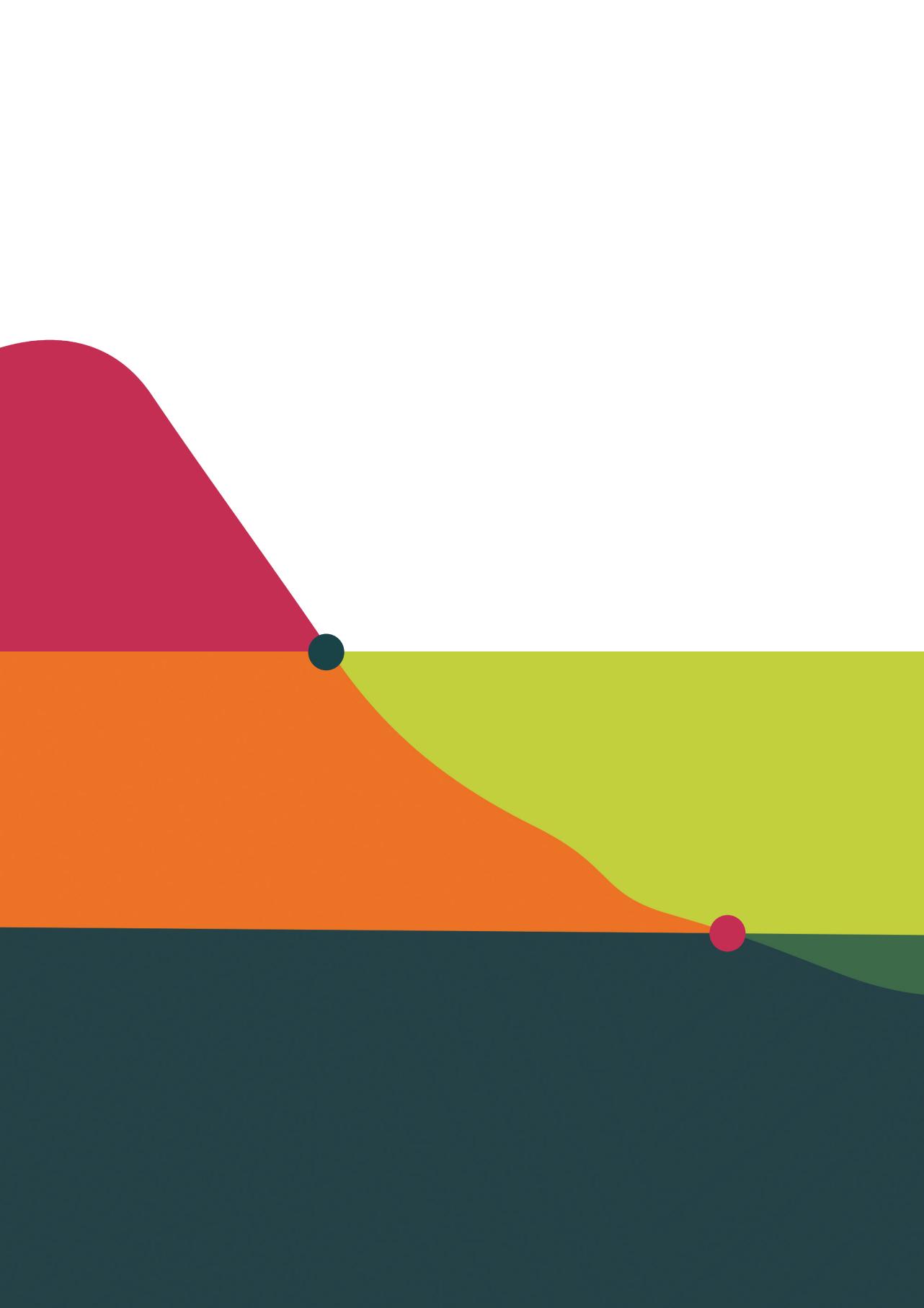
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# Appendices

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## Appendix 1. Research Data Management

The research in this thesis has been carried out under the RDM policy of the Radboud Institute for Biological and Environmental Sciences, version 5 April 2023. Data generated or used in this thesis can be accessed in the following way:

**Chapter 1** For the identification of a Paris-compatible mitigation pathway (Figure 1.1), we used data available from the Integrated Assessment Modelling Consortium (IAMC) 1.5°C Scenario Explorer and Data, hosted by IIASA: [data.ene.iiasa.ac.at/iamc-sr15-explorer](http://data.ene.iiasa.ac.at/iamc-sr15-explorer).

**Chapter 2** de Kleijne, K., S. V. Hanssen, L. van Dinteren, M. A. J. Huijbregts, R. van Zelm, and H. de Coninck, 2022: Limits to Paris compatibility of CO<sub>2</sub> capture and utilization. *One Earth*, **5**, 168–185, doi.org/10.1016/j.oneear.2022.01.006. For the identification of a Paris-compatible mitigation pathway we used data available from the IPCC IAMC 1.5°C Scenario Explorer and Data, hosted by IIASA: [data.ene.iiasa.ac.at/iamc-sr15-explorer](http://data.ene.iiasa.ac.at/iamc-sr15-explorer). Full dataset (Data S1) with harmonised emissions of CCU technologies for 2030 and 2050, and source data for Figure 2.4, Figure 2.5 and Supplementary Figure 2.1 has been deposited in the DANS EASY Archive: [Dataset] doi.org/10.17026/dans-28h-n6zj.

**Chapter 3** de Kleijne, K., J. James, S. V. Hanssen, and R. van Zelm, 2020: Environmental benefits of urea production from basic oxygen furnace gas. *Appl. Energy*, **270**, 115119, doi.org/10.1016/j.apenergy.2020.115119. All data were published in the article and supplementary information.

**Chapter 4** de Kleijne, K., H. de Coninck, R. van Zelm, M. A. J. Huijbregts, and S. V. Hanssen, 2022: The many greenhouse gas footprints of green hydrogen. *Sustain. Energy Fuels*, **6**, 4383–4387, doi.org/10.1039/d2se00444e. All data were published in the article and supplementary information.

**Chapter 5** de Kleijne, K., M. A. J. Huijbregts, F. Knobloch, R. van Zelm, J. P. Hilbers, H. de Coninck, and S. V. Hanssen. Worldwide greenhouse gas

emissions of green hydrogen production and transport. *In review.* We used publicly available data from the International Energy Agency (IEA) Hydrogen Projects Database (2022) for project-specific information on electricity source, electrolyser technology, location, and project size (accessible via: <https://www.iea.org/data-and-statistics/data-product/hydrogen-projects-database>). We used the greenhouse gas intensities of national 2030 grid mixes modelled for a 2°C policy scenario published by Knobloch, F. et al. (2020). Net emission reductions from electric cars and heat pumps in 59 world regions over time. *Nat Sustain* **3**, 437–447, <https://doi.org/10.1038/s41893-020-0488-7>. For the calculation of location-specific emissions of solar electricity, we used the solar irradiance map available from Global Solar Atlas 2.0, a free, web-based application developed and operated by the company Solargis s.r.o. on behalf of the World Bank Group, utilising Solargis data, with funding provided by the Energy Sector Management Assistance Program (ESMAP). For additional information: <https://globalsolaratlas.info>. For the calculation of location-specific emissions of wind electricity, we used wind speed maps available from the Global Wind Atlas 3.0, a free, web-based application developed, owned and operated by the Technical University of Denmark (DTU). The Global Wind Atlas 3.0 is released in partnership with the World Bank Group, utilising data provided by Vortex, using funding provided by the Energy Sector Management Assistance Program (ESMAP). For additional information: <https://globalwindatlas.info>.

- Chapter 6** Data additional to Chapters 2–5, underlying Table 6.1 and Figure 6.5 can be accessed in the DANS EASY Archive: [Dataset] doi.org/10.17026/dans-x8e-eaq2.

## Appendix 2. Supplementary Information to Chapters 2–6

The Supplementary Information to Chapters 2–6 can be obtained as a single PDF file from the Radboud Repository at <https://hdl.handle.net/2066/299507>.

# Summary | Samenvatting

## Summary

The global effort to address climate change is guided by the 2015 Paris Agreement. Most countries of the world committed to limiting global warming to well below 2°C and pursuing efforts to limit warming to 1.5°C above pre-industrial levels. The Paris Agreement's temperature goals require global carbon dioxide (CO<sub>2</sub>) emissions to halve by 2030 and reach net zero by 2050. No sector is exempted from reaching deep emission reductions, for which a range of mitigation strategies and technologies needs to be implemented. In this context, CO<sub>2</sub> capture and utilisation (CCU) technologies and green hydrogen are considered promising technologies to replace conventional processes based on fossil fuels, as reflected in policies that incentivise their development and implementation.

CCU is a process in which CO<sub>2</sub> is technologically captured from point sources or air and subsequently used, for example in agricultural greenhouses or enhance hydrocarbon recovery. Alternatively, CO<sub>2</sub> can be converted into fuels, chemicals or construction materials. Green hydrogen is produced via water electrolysis with renewable electricity, and has applications in transport, the chemical and steel industry, and as energy storage solution. The greenhouse gas (GHG) emissions associated with CCU and green hydrogen, called GHG footprints, range from negative to larger than their fossil counterparts. This indicates that implementation of these technologies does not inherently result in the emission reductions they intend to achieve.

The main aim of this thesis was to identify the conditions under which green hydrogen and CCU technologies are Paris-compatible. To identify these conditions, I assessed how the GHG footprints of green hydrogen and CCU depend on temporal, spatial and technological context, using the life cycle assessment method.

**Chapter 2** introduced the Paris compatibility framework, which evaluates technologies in the context of reaching the Paris Agreement's long-term temperature goal, based on technology maturity and emission reduction criteria.

The criteria require: (i) a technology readiness level (TRL) 6 or higher in 2020 for Paris compatibility in 2030, while for 2050 no technologies are excluded based on TRL, and (ii) reaching a GHG emission reduction of at least 50% in 2030 compared to the 2020 benchmark technology, and zero emissions in 2050. The Paris compatibility of 74 CCU processes was evaluated, covering fossil, biogenic and atmospheric CO<sub>2</sub> sources and 44 CO<sub>2</sub> use options, based on systematically gathered CCU-technology information and harmonised life cycle assessment studies. Paris-compatible CCU options in 2030 include construction materials, enhanced oil recovery, horticulture industry, and some chemicals. CCU is only Paris-compatible in 2050 when 2 out of 3 conditions are fulfilled: (i) capturing atmospheric or biogenic CO<sub>2</sub>, (ii) a zero-emissions capture and conversion process; and (iii) permanent CO<sub>2</sub> storage. The findings imply that research and policy should focus on accelerating the development of those CCU technologies that may achieve (close-to-)zero emissions, avoiding lock-in by CCU technologies with limited emission reductions.

**Chapter 3** assessed the environmental benefits of a CCU technology that produces urea, a nitrogen-based fertiliser, solely from basic oxygen furnace gas (BOFG). BOFG is a multi-component residual flow from integrated steel mills composed of CO, CO<sub>2</sub>, O<sub>2</sub>, N<sub>2</sub> and H<sub>2</sub>. Based on life cycle assessment, the environmental impacts of six applications of BOFG were quantified (heat for internal use in the steel mill, heat combined with CO<sub>2</sub> storage, electricity production, urea production, urea production combined with CO<sub>2</sub> storage, and flaring), and compared in light of the environmental impacts of the conventional production processes. Which application of BOFG results in the largest environmental benefits depends on the electricity source against which the use of BOFG for electricity is compared: for the average European case, overall environmental impacts are slightly lower for electricity production from BOFG than for urea production, while in locations with a cleaner electricity mix, overall environmental impacts are lower for urea production. Considering further decarbonisation of the electricity mix, urea production would have the lowest future environmental impacts, and when combined with storage of excess CO<sub>2</sub>, can lead to further environmental benefits.

**Chapter 4** analysed how green hydrogen's GHG footprint and emission reductions depend on (i) the current and future electricity source, (ii) time-dependent choices

regarding allocation of GHG emissions to co-product oxygen, and (iii) the grey or blue hydrogen benchmark against which green hydrogen's production emissions are compared. Depending on these three factors, green hydrogen production in Europe was demonstrated to achieve no, a small, or large emission reductions. Most of the variation in the GHG footprint is attributed to the GHG intensity of the electricity used in electrolysis. If electricity comes from additional renewable capacity that matches the increased demand from electrolyzers, green hydrogen outperforms fossil-based hydrogen. In the short term, however, alternative applications of the same renewable electricity likely achieve greater emission reductions. This implies that it is a no-regret for policymakers to reinforce the further development of renewable energy infrastructure.

**Chapter 5** provided a global and spatially-explicit analysis of GHG emissions of green hydrogen production, conversion, transport, reconversion and storage, based on location-specific renewable electricity GHG emissions and three different electrolyser technologies. For green hydrogen production emissions to stay below sustainability thresholds, low GHG electricity is required for electrolysis. Production using hydroelectricity and wind electricity would stay below these thresholds in most locations, while production using solar electricity would not. Hydrogen transport emissions depend on transport mode (pipeline, liquid hydrogen shipping or ammonia shipping), and increase with transport distance. A global emission-supply curve including 1,025 planned green hydrogen production projects in 2030 was derived, estimating total emissions and avoided emissions from projected hydrogen production, with or without transport, compared to grey hydrogen production. Based on these estimates, Chapter 5 concludes that efforts are needed to lower green hydrogen production and transport emissions.

**Chapter 6** discussed the findings of the previous chapters in the context of the Paris compatibility framework, reflected on this framework, discussed additional roles for CCU and green hydrogen, and drew overarching conclusions. Whether or not green hydrogen and CCU technologies reduce emissions depends on the temporal, spatial and technological context in which they are implemented. Conditions for Paris compatibility of CCU and green hydrogen are: (i) low energy requirements, (ii) low or zero GHG intensity of electricity, (iii) using biogenic or atmospheric CO<sub>2</sub>, (iv) permanent CO<sub>2</sub> storage in a product, (v) co-product valorisation, and

(vi) minimising hydrogen transport. Green hydrogen and CCU technologies have a role in the energy and industrial system transitions but often fall short of being truly 'zero-emissions' or 'climate-neutral' when considering life-cycle GHG emissions. Instead of labelling them as such, national long-term strategies should acknowledge their contributions to residual emissions, requiring carbon dioxide removal to achieve net-zero emissions at the system level.

## Samenvatting

Het klimaatakkoord van Parijs uit 2015 vormt de leidraad voor de wereldwijde inspanningen om klimaatverandering tegen te gaan. Het doel is om de opwarming van de aarde te beperken tot ruim onder de 2°C, strevend naar maximaal 1,5°C boven het pre-industriële niveau. Om dit te bereiken, moeten de wereldwijde koolstofdioxide-uitstoot tegen 2030 worden gehalveerd en tegen 2050 netto nul bedragen. Elke sector moet hiervoor diepgaande emissiereducties realiseren en diverse mitigatiestrategieën en -technologieën implementeren. CO<sub>2</sub>-afvang en hergebruik (CCU) en groene waterstof worden gezien als veelbelovende alternatieven voor processen op basis van fossiele brandstoffen, wat wordt weerspiegeld in beleidsmaatregelen ter bevordering en implementatie van deze technologieën.

CCU is een proces waarbij CO<sub>2</sub> technologisch wordt afgevangen bij energie- of industriële installaties, of uit de lucht, en vervolgens wordt gebruikt, bijvoorbeeld in kassen of voor de winning van koolwaterstoffen. Het kan ook worden omgezet tot brandstoffen, chemicaliën of bouwmateriaal. Groene waterstof wordt geproduceerd door elektrolyse van water met hernieuwbare elektriciteit en heeft diverse toepassingen, bijvoorbeeld in de transport sector, de chemische en staalindustrie, en als energieopslag. De emissie van broeikasgassen (BKG) die kan worden toegeschreven aan CCU en groene waterstof, genaamd de BKG-voetafdruk, varieert van negatief tot groter dan die van de fossiele tegenhanger. Dit geeft aan dat de implementatie van deze technologieën niet inherent leidt tot de beoogde emissiereducties.

Het hoofddoel van dit proefschrift was om de voorwaarden te identificeren waaronder groene waterstof- en CCU-technologieën Parijs-compatibel zijn. Ter identificatie van deze voorwaarden heb ik op basis van levenscyclusanalyses onderzocht hoe de BKG-voetafdrukken van groene waterstof en CCU afhankelijk zijn van temporele, ruimtelijke en technologische context.

**Hoofdstuk 2** introduceerde het Parijs-compatibiliteitskader met criteria om technologieën te evalueren in de context van het bereiken van de temperatuurdoelstelling van het Parijs-akkoord. Voor Parijs-compatibiliteit in

2030 zijn de criteria: een Technology Readiness Level (TRL) van 6 of hoger in 2020, en een broeikasgas-emissiereductie van minstens 50% in 2030 vergeleken met de benchmarktechnologie van 2020. Voor Parijs-compatibiliteit in 2050 zijn de criteria: een TRL van 1 of hoger in 2020, en nul emissies in 2050. Gebaseerd op systematisch verzamelde informatie over CCU-technologieën en geharmoniseerde levenscyclusanalyses heb ik de Parijs-compatibiliteit van 74 CCU-processen beoordeeld. Deze omvatten fossiele, biogene en atmosferische CO<sub>2</sub> bronnen en 44 CO<sub>2</sub>-hergebruiksopties. Parijs-compatibele CCU-producten in 2030 zijn o.a. bouwmaterialen, het gebruik van CO<sub>2</sub> in kassen, en sommige chemicaliën. CCU is alleen Parijs-compatibel in 2050 wanneer aan 2 van de volgende 3 voorwaarden is voldaan: (i) atmosferische of biogene CO<sub>2</sub> wordt afgevangen, (ii) het afvang- en conversieproces is emissievrij; en (iii) de CO<sub>2</sub> wordt permanent vastgelegd. De bevindingen impliceren dat onderzoek en beleid zich moeten richten op het versnellen van de ontwikkeling van die CCU-technologieën die (bijna) nul netto-emissies kunnen bereiken, en het vermijden van lock-ins door CCU-technologieën met beperkte netto-emissiereducties.

**Hoofdstuk 3** gaat over de milieuvoordelen van een CCU-technologie die ureum produceert, een stikstofhoudende meststof, uitsluitend uit gas (BOFG) dat ontstaat tijdens oxystaalproductie. BOFG bestaat uit CO, CO<sub>2</sub>, O<sub>2</sub>, N<sub>2</sub> en H<sub>2</sub>. Op basis van levenscyclusanalyse werden de milieueffecten van zes toepassingen van BOFG gekwantificeerd (warmte voor intern gebruik in de staalfabriek, warmte gecombineerd met CO<sub>2</sub>-opslag, elektriciteitsproductie, ureumproductie, ureumproductie gecombineerd met CO<sub>2</sub>-opslag, en affakkelen), en vergeleken met de milieueffecten van de conventionele productieprocessen. Welke toepassing van BOFG resulteert in de grootste milieuvoordelen hangt af van de elektriciteitsbron waarmee het gebruik van BOFG voor elektriciteit wordt vergeleken: voor het gemiddelde Europese geval zijn de algehele milieueffecten iets lager voor elektriciteitsproductie uit BOFG dan voor ureumproductie, terwijl op locaties met een schonere elektriciteitsmix de algehele milieueffecten lager zijn voor ureumproductie. Rekening houdend met verdere decarbonisatie van de elektriciteitsmix, zou de productie van ureum de laagste toekomstige milieueffecten hebben, en wanneer gecombineerd met opslag van overtollige CO<sub>2</sub>, kunnen verdere milieuvoordelen worden behaald.

**Hoofdstuk 4** analyseerde hoe de BKG-voetafdruk en emissiereducties door groene waterstof afhangen van drie factoren: (i) de huidige en toekomstige elektriciteitsbron, (ii) tijdsafhankelijke keuzes over de toewijzing van BKG-emissies aan co-product zuurstof, en (iii) de grijze of blauwe waterstofbenchmark waarmee de emissies van groene waterstofproductie worden vergeleken. De levenscyclusanalyse toonde aan dat de productie van groene waterstof in Europa, afhankelijk van deze factoren, zeer variabele emissiereducties kan behalen. De meeste variatie in de BKG-voetafdruk hing af van de BKG-intensiteit van de gebruikte elektriciteit bij elektrolyse. Hoewel groene waterstof gunstiger presteert wanneer elektriciteit afkomstig is van additionele hernieuwbare capaciteit die aan de toegenomen vraag van elektrolyse voldoet, kunnen alternatieve toepassingen van deze elektriciteit op korte termijn waarschijnlijk een grotere emissiereductie opleveren. Dit betekent dat het een 'no-regret' is voor beleidsmakers om de uitbreiding van hernieuwbare energie-infrastructuur verder te stimuleren.

**Hoofdstuk 5** biedt een wereldwijde en ruimtelijk expliciete analyse van BKG-emissies van groene waterstofproductie, conversie, transport, herconversie en opslag, gebaseerd op locatie-specifieke BKG-emissies van hernieuwbare elektriciteit en drie verschillende elektrolyser-technologieën. Elektriciteit met lage BKG-emissies is vereist om onder de gestelde emissiedempels te blijven. Waterkracht en windenergie voldoen hier over het algemeen aan, terwijl zonne-energie dat op de meeste locaties niet doet. De emissies van waterstoftransport variëren afhankelijk van de transportmodus en nemen toe naarmate de afstand groter wordt. Het hoofdstuk leidt een wereldwijde emissie-aanbodcurve af, op basis van 1.025 geplande groene waterstofproductieprojecten in 2030, en schat totale en vermeden emissies van geprojecteerde waterstofproductie, met of zonder transport, in vergelijking met grijze waterstofproductie. Op basis hiervan concludeert hoofdstuk 5 dat verdere inspanningen nodig zijn om de emissies van groene waterstofproductie en -transport te verminderen.

**Hoofdstuk 6** vormt de synthese waarin de bevindingen van de voorgaande hoofdstukken in de context van het Parijs-compatibiliteitskader worden besproken. Ik reflecteerde hier ook op het kader, op aanvullende rollen voor CCU en groene waterstof, en trok overkoepelende conclusies. Of groene waterstof- en CCU-technologieën emissies reduceren hangt af van de temporele, ruimtelijke

en technologische context van hun implementatie. Om Parijs-compatibel te zijn moeten CCU en groene waterstof voldoen aan specifieke voorwaarden, zoals een laag energieverbruik, minimale of geen BKG-emissies van gebruikte elektriciteit, gebruik van biogene of atmosferische CO<sub>2</sub>, permanente CO<sub>2</sub>-opslag in producten, co-productvalorisatie, en minimalisatie van waterstoftransport. Groene waterstof- en CCU-technologieën spelen een rol in de energie- en industriële systeemtransities, maar ze voldoen vaak niet aan de verwachtingen 'emissievrij' of 'klimaatneutraal' te zijn. In plaats van ze zo te noemen, zouden nationale langetermijnstrategieën hun bijdragen aan residuele emissies moeten erkennen, waarbij koolstofverwijdering nodig is om netto nul emissies op systeemniveau te bereiken.

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## About the author



I was born in Belém, Brazil, on February 2nd, 1993. I grew up in Groesbeek, The Netherlands, and attended the Stedelijk Gymnasium Nijmegen, graduating in 2010. I moved to Utrecht to study at University College Utrecht, an interdisciplinary honours college. I earned a BSc in Liberal Arts and Sciences (*magna cum laude*), focusing on Mathematics, Physics

and Cognitive Neuroscience. It was during this time that I first encountered a course on Energy and Sustainability, sparking a growing interest in environmental challenges. In 2013, I moved to Nijmegen to pursue my MSc at Radboud University. In 2017, this resulted in a degree in Physics with a specialisation in Science, Management and Innovation, centred on the theme of Climate and Energy (*bene meritum*). During the last year of my MSc, I began my work as a chapter scientist for the Intergovernmental Panel on Climate Change (IPCC) Special Report on Global Warming of 1.5°C, continuing this role post-graduation as a junior researcher at the Environmental Science department of Radboud University. This transitioned into another junior researcher position, focusing on the environmental impacts of a carbon capture and utilisation technology. In 2019, I started my PhD project within the same department, completing my thesis four years later. Throughout this period, I resumed my role as chapter scientist and contributing author for the IPCC Working Group III report on the Mitigation of Climate Change (2019–2022), and contributed to IPCC spin-off reports. My active engagement in the societal debate on the climate crisis led to the honour of receiving the 2022 Hermesdorf Talent Award, an acknowledgment granted to researchers who demonstrated courage and dared to take risks. Transitioning into 2024, I started a new phase in my career as a post-doctoral researcher at the Department of Industrial Engineering and Innovation Sciences at Eindhoven University of Technology.

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