

Abstract

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The Paris Agreement signalled global consensus to keep average temperature rise “well below” 2 °C by the end of the century. Results from integrated assessment models have made it increasingly evident that negative emissions (removing CO₂ from the atmosphere) are crucial to achieving this. Consequently, negative emissions technologies (NETs) have come to the forefront of mitigation discussions. NETs must however overcome challenges if they are to be realised at scale. Uncertainties around the large-scale biomass supply have fuelled a debate on whether negative emissions from bio-energy with carbon capture and storage (BECCS) are sustainably achievable, if at all. Reliable carbon accounting frameworks and policy incentives are needed to improve investment prospects. The direct extraction of CO₂ from air, or direct air capture (DAC), has since been demonstrated as a source of negative emissions. The large energy and economic costs associated with extracting CO₂ from air are proving prohibitive to achieving commercial viability of DAC technology. Without dedicated policy support for technological innovation, and further interdisciplinary research to constrain a variety of uncertainties, the world risks foregoing a portfolio of technologies that add much-needed flexibility in the mitigation toolbox. This chapter details the evidence for negative emissions, proposed means of achieving them and their barriers to commercial effectiveness.

CHAPTER 14

Negative Emissions Technologies

H. A. DAGGASH,^{a,b,c} M. FAJARDY^{b,c} AND N. MAC DOWELL^{*b,c}

^a Imperial College London, Grantham Institute – Climate Change and the Environment, Exhibition Road, SW7 2AZ, London, UK; ^b Imperial College London, Centre for Environmental Policy, 16–18 Princes Gardens, London SW7 1NE, UK; ^c Imperial College London, Centre for Process Systems Engineering, Exhibition Road, SW7 2AZ, London, UK

*Email: niall@imperial.ac.uk

14.1 The Need for Negative Emissions

The Paris Agreement signalled global commitment to limit average temperature rise to well below 2 °C (above pre-industrial levels) by the end of the century, and to pursue efforts to achieve 1.5 °C to minimise the risks and impacts of climate change. It already seems increasingly likely that concentrations of CO₂ and other greenhouse gases (GHGs) in the atmosphere will overshoot 450 ppm CO_{2eq}, widely seen as the upper limit of concentrations consistent with this 2 °C limit.¹ The Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report outlines many least-cost transformative pathways to achieve various concentration targets; the majority of the scenarios that result in 450 ppm CO_{2eq} or less involve temporary overshoot of the target and rely on the extensive deployment of negative emissions technologies (NETs) in the second half of the century to achieve climate stabilisation, specifically bio-energy with carbon capture and storage (BECCS). NETs are technologies that, directly or indirectly, result in the net removal of CO₂ from the atmosphere.

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Achieving pathways without overshoot will require more difficult, near-term mitigation efforts. Therefore, there is the need for more research and development to make NETs more feasible at scale. Achieving large-scale feasibility and implementation of NETs will have two key implications. First, abatement will be decoupled from emission sources which can compensate for emissions from technologies and sectors with more difficult and/or expensive abatement such as transport, thereby adding flexibility to the mitigation toolbox.² Secondly, NETs will allow faster declines in concentrations in the long-term and thus facilitate higher near-term emissions, effectively expanding the potential scope for overshoot. Hence, NETs have been identified as the most critical set of technologies in the context of the timing of emissions reductions.¹

Several studies, however, have shown that the relationship between cumulative emissions and peak warming is remarkably insensitive to the emission pathway (timing of emissions or peak emission rate).¹ So, policies designed to achieve cumulative emissions targets are likely to be subject to less uncertainty than those with emission-rate or concentration targets. This suggests that climate change risks and impacts can be mitigated by minimising historic emissions, a further motivator to explore the possibility of large-scale GHG removal from the atmosphere. Furthermore, studies have shown that stabilisation targets are more easily attainable with NETs like BECCS at a lower cost.^{3,4} The opportunities offered by the availability of negative emissions has brought NETs to the forefront of mitigation discussion seeking to reduce atmospheric GHG concentrations to safer levels.

14.2 Proposed Negative Emissions Technologies

Of the technologies that have been put forth in the past decades as having the capability to remove CO₂ from the atmosphere and therefore create net negative flows of carbon from the atmosphere, several have dominated discussion in the scientific literature: afforestation/reforestation, bio-energy with carbon capture and storage, direct air capture, soil carbon sequestration, ocean fertilisation and enhanced weathering of minerals. Although afforestation/reforestation only exploits natural phenomena of photosynthesis and the carbon cycle to create negative emissions, it is considered as *technology* as it provides the same service as the other stated technologies. Initial assessment of these prospective negative emissions technologies (NETs) has revealed a range of physical, technical, and socioeconomic barriers for each that must be overcome before commercial deployment is viable. An overview of each technology and its limitations is provided in the following sections.

14.2.1 Afforestation and Reforestation (AR)

Carbon is the basis of life on Earth – food, plants and animals are made up of it, and our economies have been built on carbon-derived energy.

The Earth's carbon flows are regulated by the carbon cycle, which maintains the stability of the global temperature, and, hence, the climate. Since the onset of industrialisation, anthropogenic GHG emissions, principally carbon dioxide (CO₂), have disrupted this cycle. The rate of release has outstripped the rate of absorption by natural carbon sinks (soil, ocean and plants), and this imbalance has led to the rising atmospheric GHG concentrations responsible for climate change. Plant biomass (crops, trees, *etc.*) comprises an important reservoir of carbon in the cycle; through photosynthesis, it absorbs and converts atmospheric CO₂ into glucose from which plant structures are built, thereby reducing its concentration in the atmosphere. Therefore, forests constitute an important carbon stock in our environment which should be maintained.

However, declining forest cover has been experienced globally in recent decades due to deforestation and forest degradation. Deforestation is the conversion of forestland to other purposes, while forest degradation is partial deforestation (reducing tree canopy cover to more 10–30%). Together, they are estimated to have contributed 12–20% of anthropogenic GHG emissions.⁵ Agriculture has been the main driver of this, with increasing demand for cropland and pasture leading to conversion of forests; the pulp and paper industry has also contributed through encouraging increased logging practices. Forests typically store more carbon than land used for other purposes (*e.g.* agriculture), hence the expansion of forestland (afforestation) and regeneration of depleted forests (reforestation) have often been proposed and implemented as a mitigation strategy.⁶ The possibility to grow forest on marginal agricultural land and riparian areas bolsters the prospects for mitigation *via* extensive reforestation.

Of the earlier stated NETs, afforestation/reforestation (AR) is the only one that has been implemented on a large-scale as a mitigation tool. Its mitigation potential however is often disputed, with a wide range of global forest biomass resource potentials cited in literature, mainly due to different assumptions on the constraints to sustainable forestland availability.¹ Furthermore, a range of technical, political and socioeconomic factors, such as difficulties in establishing measurement methodologies, non-permanence of carbon in forests, high opportunity costs of land-use, and the transaction costs generated by a weak and complex climate agreement in the Land Use Land Use Change and Forestry (LULUCF) sector have prevented further implementation of AR as a carbon sequestration technique.⁷

14.2.2 Biochar and Soil Carbon Sequestration (SCS)

When biomass is combusted in a low-oxygen environment, a process known as pyrolysis, volatiles and gases are released along with a carbon-rich co-product known as biochar. Biochar is highly resistive to biological and chemical degradation, making it stable. When added to soil, char creates a system that has greater abatement potential than typical bioenergy. Biochar

is doubly advantageous due to its indirect effects – biochar-enriched soil shows increased crop and biomass production, and decreased soil emissions of methane (CH₄) and nitrous oxide (N₂O) emissions, both potent greenhouse gases.⁸ Biochar was estimated to be able to avoid up to 6.6 GtCO_{2e} per year of CO₂, N₂O and CH₄ emissions (~18% of annual GHG emissions) and reduce total net emissions by 480 GtCO_{2e} over the course of a century. Woolf *et al.* estimate the maximum sustainable technical potential of biomass – that is biomass that can be utilised without endangering food security, habitat or soil conservation – to be 2.27 GtC per year from energy crops, residues and wastes.

While biochar has potential as a mitigation option, too few long-term field experiments have been carried out to substantiate meaningful soil carbon sequestration due to the soil application of biochar and claims of the universally beneficial impacts of its application on soil quality and the ability to support improved plant growth and yield are also not consistent with the available data.⁹ More technical questions remain to be answered, including: understanding the relationship between biomass feedstock, process conditions and biochar properties; effects of different biochar in different soils; effects of soil enrichment using biochar on other non-CO₂ greenhouse gases.² Furthermore, as with all biomass-dependent technologies, sustainable production and competition for land (for other economic activities) pose challenges to commercial biochar deployment.

14.2.3 Enhanced Weathering of Minerals

Despite being a very slow process, chemical weathering of rocks is an important process in the carbon cycle that maintains the CO₂ concentration in the atmosphere. Atmospheric carbon dissolves in the oceans to form carbonic acid, which is subsequently precipitated and dissolves rocks, releasing cations (Ca²⁺, Mg²⁺, Na⁺ and K⁺). The calcium and magnesium released during weathering are carried by rivers and deposited in oceans where they combine with carbonate to form limestones and dolomites, which are the major sink for CO₂.¹⁰ The second major sink is organic matter. After death, calcifying (shell-building) organisms such as corals, and some plankton are cemented under the action of heat and pressure to form sedimentary rocks, *e.g.* shale. Layers of organic carbon can also become fossil fuels (coal, oil, natural gas) instead of sedimentary rock.

As anthropogenic activity has increased the concentration of CO₂ in the atmosphere and possibly led to catastrophic events through climate change, increasing attention is being given to the possibility of accelerating this natural weathering to stabilise CO₂ concentrations at safer levels.

Several means to achieve this accelerated weathering have been proposed. Silicate rocks comprise 90% of the Earth's crust, hence availability is abundant, however their natural weathering rates are very slow. Increasing the surface area of the rocks has been proposed as a means to overcome this limitation and improve dissolution kinetics with atmospheric CO₂. Grinding

of the silicate rocks and spreading on the Earth's surface is a means to achieve this. The general reaction for this is given by the following:¹⁰ 1



There are, however, several challenges to overcome: 5

- (i) The silicate rocks must be mined, ground and transported before use, which is energy intensive. The total energy requirements for mining, comminution, transport and spreading have been estimated at 1.4–9.9 GJ per tCO₂ sequestered.¹¹ 10
- (ii) In order to minimise lifecycle emissions of the enhanced weathering process, carbon-free energy should be used to obtain and process rocks. Curtailed energy from renewables is unlikely to be sufficient, hence added capacity may be required. In addition, carbon-free transport for minerals (heavy goods) is also not widely available. 15
- (iii) In the absence of a credit for removing CO₂ (*e.g.* a carbon price), there are no incentives for deploying this technology, hence investment challenges are likely to be encountered. 15
- (iv) The accelerated release of cations will increase the pH of the waters exposed to them, which may have detrimental effects on their ecosystems. 20

While mostly mined rocks have been considered, there is the possibility of accelerated weathering by carbonation of calcium-rich industrial wastes, such as basic slag and demolition rubble.¹² Soil modified with these undergoes carbonation and accumulates CaCO₃, forming carbon sinks. Renforth *et al.* estimate the maximum global mitigation potential to be ~1 GtCO₂ per year. To overcome slow weathering rates, direct dissolution of basic silicate minerals such as Olivine (Mg₂SiO₄) in oceans has been proposed.^{13,14} This has the added advantage of minimising the need for geological storage, however, uncertainties exist about the effects on marine ecosystems and subsequent environmental impacts, hence further research is required. 25 30

For the UK, enhanced weathering could remove up to 430 GtCO₂ at a cost of \$24–578 per tCO₂, but will require 0.8–12.6 GJ per tCO₂.¹⁵ The energy requirements remain a limitation to the feasibility of the process, however energy costs are similar to those of other negative emissions technologies, *e.g.* direct air capture. 35

14.2.4 Ocean Fertilisation 40

Oceans sequester carbon partly as a result of biological productivity of marine microorganisms.¹⁶ Phytoplankton convert CO₂ absorbed from the ocean into particulate organic carbon (POC), most of which is eaten by other marine organisms that regenerate the CO₂ when they respire. Some, however, sinks to the deep ocean, thus reducing CO₂ in the surface layer and 45

elevating it in the deep sea.¹⁷ Consequently, the alkalinity of ocean surface water is reduced, allowing for increased uptake of CO₂ from the atmosphere. Phytoplankton productivity has been found to be limited by the availability of nutrients, including iron and nitrate. Geoengineering the ocean by adding nutrients to encourage blooms of phytoplankton, and hence accelerate their natural sequestration activity, has been proposed as a mitigation strategy to counter rising atmospheric CO₂ concentrations. This has been opposed for several reasons:

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- (i) High-nitrate and low-chlorophyll (HNLC) regions are the most important for ocean fertilisation as they suffer from iron deficiency. This restricts the suitable regions for stimulating phytoplankton growth to only ~30% of the Earth's oceans, mainly in parts of the Southern Ocean.⁸ ~~Simulations show that in the first 100 years, iron fertilisation could reduce~~
- (ii) Models suggest that pursuing this as a mitigation option will change the ecology of aquatic ecosystems. An oxygen-deficient environment (aquatic hypoxia or anoxia) that supports the growth of microbial populations that release potent greenhouse gases (*e.g.* methane, nitrous oxide) may result, thereby contradicting the objective of reduced atmospheric GHG concentrations.^{17–19}
- (iii) Sustained fertilization at scale will also be offset by CO₂ release during the acquisition, transportation and release of large volumes of iron in remote oceanic regions.⁸

The mitigation potential of ocean fertilisation remains largely speculative. Many have discredited it as a worthy field of research because of the uncertainties about environmental side effects (particularly to aquatic ecosystems) that may result and the impossibility of testing its viability without affecting ocean chemistry.^{8,17,20} Furthermore, the time and space requirements for experimentation, and a lack of a clear institutional framework for implementation gives it poor prospects as a near- or medium-term mitigation option.

14.3 Bio-energy with Carbon Capture and Storage (BECCS)

14.3.1 Introduction

Identified as one of the key technologies in IPCC mitigation pathways,¹ bio-energy with carbon capture and storage, or BECCS, has been receiving increasing attention in the past decade.^{21–23} By combining a carbon neutral feedstock – biomass – with the capture of the CO₂ emissions released upon biomass conversion, BECCS is theoretically a carbon negative technology. As a result, BECCS has been increasingly featured in Integrated Assessment Models (IAMs) as a way to offset inevitable CO₂ from hard to reach fields,

such as agriculture and transport, and enable to meet the well below 2-°C emission trajectory.^{3,24} If BECCS has been proven to deliver negative emissions on a case study basis,^{25–28} its deployment is tied with the deployment of two technologies which already face controversy: bioenergy, the carbon neutrality and resource efficiency of which has been questioned,^{29,30} and CO₂ capture and storage, which has not yet proven to be economical or fully reliable.

According to the IPCC 2DC emissions pathways, BECCS could be required at a scale of 8.5–16.5 Gt_{CO₂} per year by 2100,^{3,31,32} with mean scenarios predicting a deployment of 12 Gt_{CO₂} per year.³³ Considering a biomass heating value of between 18 to 21 GJ/t_{DM} and a carbon content between 46 and 52%,³⁴ 1 EJ of biomass represents approximately a carbon capture potential of between 80 and 106 Mt_{CO₂}. Assuming a capture efficiency of 90%, 72 to 95 Mt_{CO₂} could be removed from the atmosphere per EJ of primary bioenergy harvested. Removing 8.5 to 16.5 Gt_{CO₂} per year by 2100 with BECCS would thus require between 90 and 230 EJ per year of primary bioenergy. This approximate range is consistent with results from IAMS, forecasting the BECCS requirement to be in the range of 150–300 EJ per year.³

There has been a lot of predictions as to bioenergy potential deployment and how much of this deployment would be sustainable. As presented in Figure 14.1, the IEA indicates biomass availability ranges of as broad as 50–1500 EJ per year by 2100 based on literature, of which only 200 to 250 EJ per year could be sustainably sourced.³⁵

Based on these facts, providing carbon capture and storage (CCS) technology reaches large scale deployment, enough sustainable biomass could be sourced to meet our negative emission targets. However, as of 2017, there are only five operating BECCS projects in the world, with a removal capacity reaching 0.85 Mt_{CO₂} per year. With a total of sixteen projects in preparation, this capacity could reach 31 Mt_{CO₂} per year in the near future, which is still over 500 times less that what could be required by 2100. Figure 14.2 presents the location and advancement stage of all 20 projects in the world, with Table 14.1 giving details of these projects.²³

Whilst CCS gathers different capture technologies, post- and oxy-combustion capture, solvent absorption or membrane adsorption, bioenergy is itself a vast term that refers to different pathways. Figure 14.3 maps the different BECCS pathways in relation to the adequate feedstock.

Three fields stand out in the deployment of BECCS on a large scale: biomass to liquid or gaseous fuels with CCS,^{36,37} biomass to heat and power with CCS *via* combustion or gasification,^{26,38} and finally, CCS on biogenic sources from the industrial sector, such as the pulp and paper industry. Whilst the biomass-to-fuel pathway offers the advantage of yielding a very rich CO₂ stream, of between 85 and 95%,³⁶ which results in an overall cheaper process, the carbon removal potential of this technology is limited by the emissions released upon use of the biofuel. Unless captured and sequestered, these emissions inevitably decrease the system carbon efficiency, *i.e.* the fraction of the carbon fixed in the biomass which ends up in

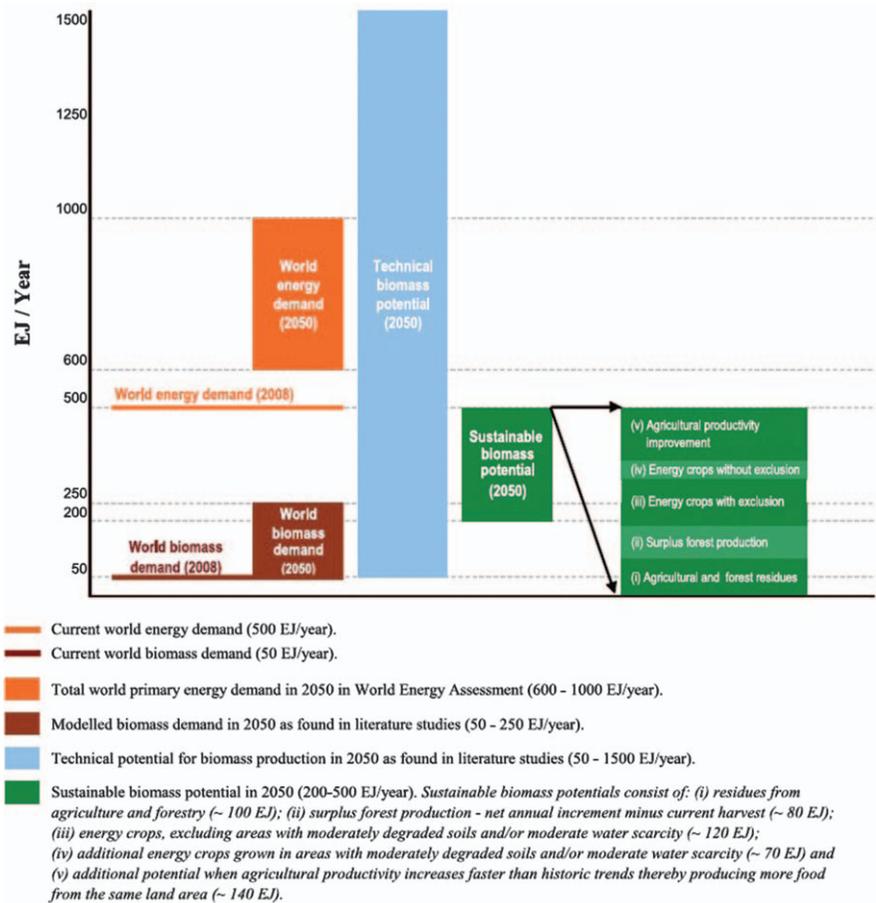


Figure 14.1 Bioenergy potential in EJ per year from the literature. While estimates for bioenergy availability ranges from 50–1500 EJ per year, the potential for sustainably sourced bioenergy ranges between 200–500 EJ per year.³⁵

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permanent geological storage. As all the emissions from biomass combustion and gasification can be theoretically captured, the biomass to heat and power pathway is superior in terms of carbon efficiency. However, the impact of the dedicated use of biomass in large-scale power plant facilities on power generation efficiency and logistics puts forward the question of the feasibility of deploying this technology on a large scale. Figure 14.4 presents a comparison of the theoretical and economical capture potential of biomass-to-power and biomass-to-fuels pathways to negative emissions.

Whilst biomass-to-power *via* combustion in circulated fluidized bed reactors could theoretically yield 10 Gt_{CO₂} per year by 2050, the negative

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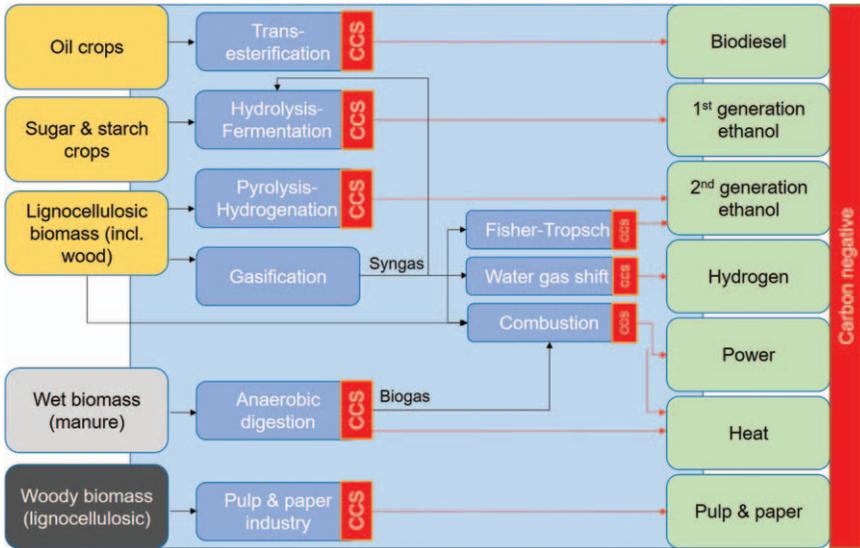


Figure 14.2 Number and locations of BECCS projects across the world. As of today, 0.85 MtCO₂ are removed per year, with a potential capacity of 31 MtCO₂ per year considering all the projects in preparation. This is still three orders of magnitude lower than what is required for BECCS by 2100.²³

emissions achieved *via* biomass-to-bioethanol do not reach 2 GtCO₂ per year.²³

The pulp and paper industry is an energy intensive and CO₂ emitting industry. As the predominant fraction of those emissions is biogenic, retrofitting or deploying pulp and paper facilities with CCS can deliver negative emissions.^{37,39}

This diversity of options for both bioenergy and CCS technologies results in a great number of BECCS different pathways, which requires case specific assessment in order to draw out relevant conclusions as to BECCS deployment potential.

14.3.2 Which Feedstock for BECCS?

Biomass feedstocks used in BECCS can differ at many levels: type – lignocellulosic woody biomass, lignocellulosic grass biomass, oil crop, sugar and starch crop, waste, origin – conventional agriculture with (main products and residues), energy dedicated agriculture, with perennial grasses and short rotation coppice, forest management and municipal wastes, and finally shape – chips, pellets, briquettes, bales, bulky biomass. To these supply options, processing steps can be added such as torrefaction, which can increase biomass mass and energy densities, enhance the fuel integrity in storage and transport, and improve the conversion performance.^{40,41} A summary of these feedstock options is outlined in Figure 14.5.

Table 14.1 Overview of BECCS projects across the world: size, state of development, BECCS pathway and location on the map (Figure 14.2). Adapted from ref. 23 with permission from Elsevier, Copyright 2015.

Project name	Location	Status	CO ₂ capacity (Mt _{CO₂} per year)	CO ₂ source	No. on map
IBDP and IL-ICCS project ^a	Decatur, IL, USA	Operating since 2011, IBDP completed	IBDP: 0.3 (1.0 in total) IL-ICCS: 1.0 (3.6 in total until 2015)	Archer Daniels Midland ethanol plant	1
Arkalon	Liberal, KS, to Booker, TX, USA	Operating since 2009/2010	0.29 (0.105 initially)	Conestoga's Arkalon ethanol plant	2
Bonanza	Garden City to Stuart Field, KS	Operating since 2011	0.15	Conestoga's Bonanza BioEnergy ethanol plant	3
RCI ^b /OCAP/ROAD	Rotterdam, NL	Operating since 2011	0.1 (Abengoa) 0.3 (Shell) (2.5 planned for 2015)	Shell's Pernis refinery, Abengoa's ethanol plant, Maasvlakte power plant, various others	4
Husky Energy	Lloydminster, SK, CA	Operating since 2012	0.1	Ethanol plant	5
Norcem	Brevik, NO	Testing since 2014, CO ₂ capture only	Small-scale	Cement plant, >30% biomass-fuelled	6
White Rose CCS Project	Selby, UK	Planned start in 2019	2.0	Drax power station, biomass (co)-firing	7
C.GEN North Killingholme Power Project	North Killingholme, UK	Evaluating, planned start in 2019	2.5	Biomass co-fired IGCC power plant	8
Södra	Värö, SE	Identifying and evaluating	0.8	Pulp and paper mill	9
Domsjö Fabriker	Domsjö, SE	Identifying and evaluating	0.26	Black liquor gasification pulp mill	10

Lantmännen Agroetanol	Norrköping, SE	Identifying and evaluating	0.17	Ethanol plant	11
CPER Artenay project ^c	Artenay and Toury, FR	Identifying and evaluating	0.045–0.2	Tereos ethanol plant	12
Sao Paulo	Sao Paulo state, BR	Identifying and evaluating	0.02	Ethanol plant	13
Biorecro/EERC ^d	ND, USA	Identifying and evaluating	0.001–0.005	Gasification plant	14
Skåne	Skåne, SE	Identifying and evaluating	0.0005–0.005	Biogas plant	15
Russel EOR research project ^e	Russel, KS, USA	Completed 2005	0.004 (0.007 in total)	Ethanol plant	16
Rufiji cluster	TZ	Cancelled	5.0–7.0	Sekab's ethanol plants	17
Greenville	Greenville, OH, USA	Cancelled in 2009 ^f	1.0	Ethanol plant	18
Wallula	Wallula, WA, USA	Cancelled ^g	0.75	Boise Inc's pulp mill	19
CO ₂ Sink	Ketzin, DE	Cancelled ^h	0.08		20

^aIllinois Basin Decatur Project (IBDP) and its successor, the Illinois Industrial CCS (IL-ICCS) project.
^bThe Bio-CCS activities are part of a larger cluster project led by the Rotterdam Climate Initiative (RCI), including the Organic Carbon Dioxide for Assimilation of Plants (OCAP) project and the Rotterdam Opslag en Afvang Demonstratieproject (ROAD).
^cProject partners are French Geological Survey (BRGM), Orleans Institute of Earth Sciences (ISTO) and Orleans Economics Laboratory (LÉO). Conducted in the context of the 2007–2013 State-Region Centre Plan Implementation Agreement (CPER).
^dThe project wants to use an existing well of the Plains CO₂ Reduction Partnership programme.
^eThe main objective of the project was to investigate EOR in mature, almost depleted fields. Due to weak results, the project never proceeded towards commercialisation. However, the Russel project was the first small-scale demonstration of Bio-CCS and delivered actual negative emissions.
^fThe project was cancelled in the very last stage due to local public opposition. It was part of the Midwest Regional Carbon Sequestration Partnership (MRCSP).
^gAfter the US DOE suspended funding.
^hDue to regulatory and technical difficulties, the Bio-CCS part of the project was cancelled. The project continued using CO₂ from a fossil source and completed injection in 2013.

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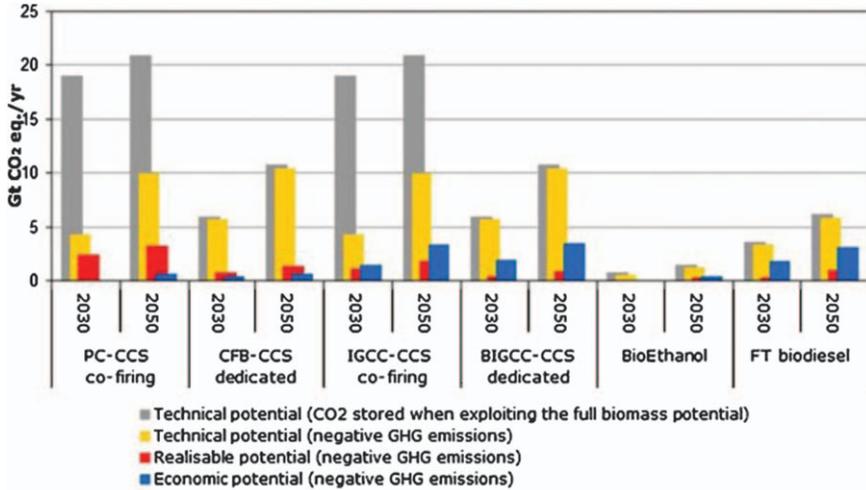


Figure 14.3 Diagram of BECCS different pathways to achieving negative emissions. Reproduced from ref. 175 with permission from IEAGHG.

Each combination of type, sector of origin and shape result in a different supply chain – production, processing and transport, and thus in a different energy use, carbon footprint, water footprint, land footprint and cost. Rather than studying a “generic biomass”, it is thus important to consider each case study specifically, as it will have a direct impact on BECCS sustainability. Among conventional crops, oil crops such as palm or corn, or sugar and starch crops such as sugar beet have been preferably used for biofuel production. To avoid competition with the food industry, lignocellulosic biomass from perennial grass crops such as switchgrass or agricultural residues, such as wheat straw or corn stover, have more recently been investigated for biofuel production.^{42,43}

Lignocellulosic biomass is also what is preferably used for bioelectricity production. Wood chips and wood pellets are the primary sources of cellulosic biomass for biomass power plants,^{44,45} but have also started to be supplemented with perennial grasses and residues.^{46–48}

There are inherent differences between these different feedstocks. Main agriculture crops used for bioenergy – oil, sugar and starch crops – require yearly land preparation and harvesting. These in-field operations typically involve seeding, tilling, packing of the land, herbicide spraying, fertiliser application (NPK), irrigation, harvesting and/or cutting and collection of the biomass. For agriculture residues, as by-products of a main crop, the difficulties lie in allocating life cycle CO₂ emissions and water and energy use between the main crop and residue production.^{49,50} Moreover, in certain cases, removing the residue from the field prevents the natural decomposition of the waste that supplies sufficient nutrients to the field. Additional nutrient input must therefore be accounted for, in addition to the residue collection, in the life cycle assessment of residues production and

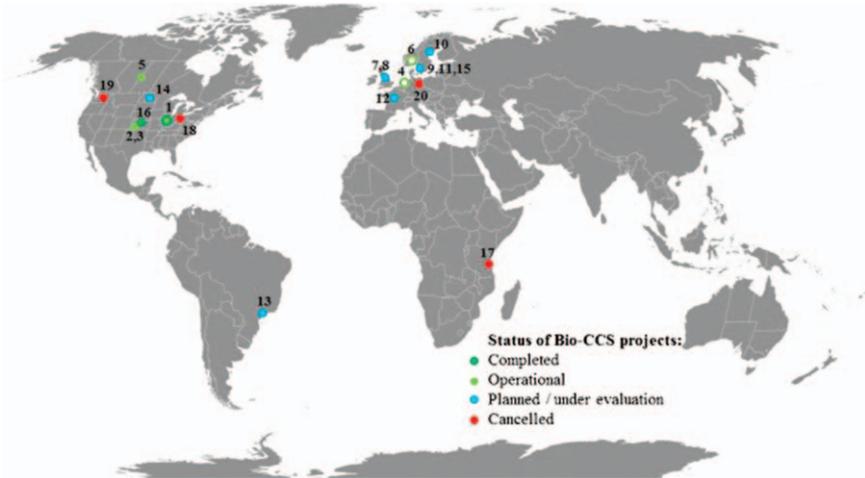


Figure 14.4 Technical and economic capture potential of BECCS pathways. While biomass-to-power reaches a negative emission potential of 4–10 GtCO₂ per year, biomass-to-fuel potential is lower than 2 GtCO₂ per year.²² Adapted from ref. 23 with permission from Elsevier, Copyright 2015.

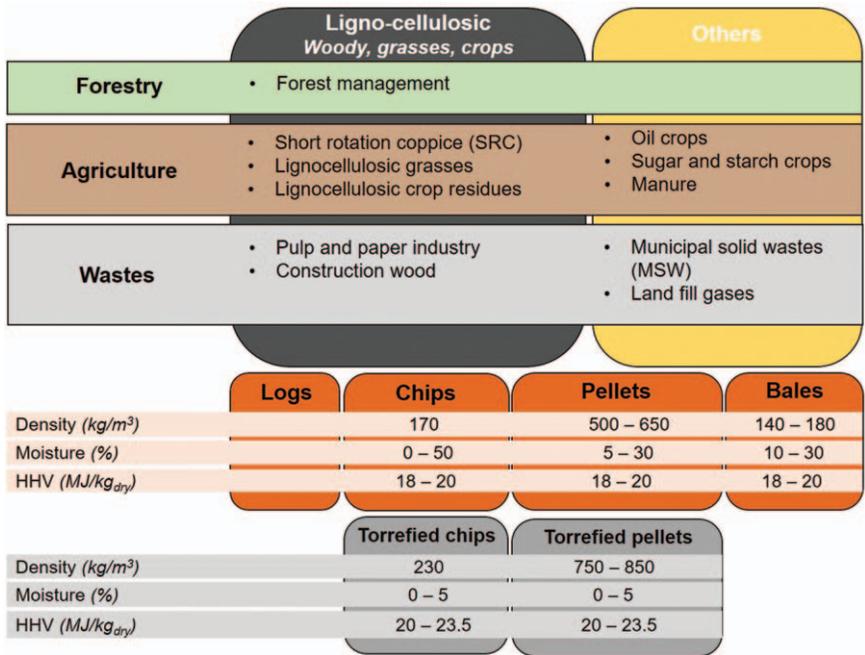


Figure 14.5 Bioenergy applications and biomass conditioning for different types of feedstocks.

supply.^{50,51} Energy dedicated crops usually differ from conventional crops in their high yield and the fact that they are perennial: although these crops also require annual water and nutrient input, the land needs to be prepared only once over the crop lifetime. For example, miscanthus can be harvested over 15–20 years, yielding approximately 15 to 40 t_{DM} ha⁻¹ per year.^{52–55} In comparison, wheat, as a one year rotation crop, has an annual yield in the 3–9 t_{DM} ha⁻¹ range.⁵⁶ Of all feedstocks types, woody wastes from the pulp and paper industry, or wastes such as municipal solid wastes (MSW) and landfill gases, overall require a less complex supply chain. However, the diversity in their quality, the potential toxic emissions upon their conversion and their low conversion performance are potential downsides that must also be considered.⁵⁷

14.3.3 Comparing BECCS Pathways to Negative Emissions

There are several options to achieve negative emissions *via* BECCS. This section focuses on the following options:

- (a) Bioelectricity and heat generation *via* direct combustion or gasification of biomass combined with CCS,
- (b) bioethanol production *via* biomass fermentation combined with CCS, and
- (c) CCS on biogenic emissions from the pulp and paper industry.

14.3.3.1 Carbon Negative Heat and Power

By co-firing biomass with coal in a pulverized fuel power plant, or gasifying biomass to use in gas fired power plants (bio-IGCC or BIGCC), a fraction of the emissions of the power plant can be biogenic, and subsequent capture and storage of these emissions can therefore lead to a carbon negative balance. In 2014, world bioelectricity installed capacity amounted to 90.2 GW, supplying 420 TWh to the electricity market, and is forecasted to supply 590 TWh per year by 2020.⁵⁸ Assuming a biomass conversion efficiency of approximately 30%,²⁵ this energy supply would correspond to a primary bioenergy supply of approximately 7 EJ per year, which is far from the 150 to 300 EJ per year required by 2100 to meet the 2-°C mitigation target.

14.3.3.1.1 Carbon Negative Potential. When biomass is used to generate heat and/or power through combustion or gasification, all of the biomass carbon is released as CO₂, and can then be captured with a 90 to 95% capture efficiency. This pathway is therefore the most carbon efficient option. Without considering the life cycle emissions of the BECCS value chain, between 70 and 95 g_{CO₂} can be captured per MJ of primary bioenergy. Assuming 30%_{HHV} power generation efficiency in a BECCS facility, carbon negative bioelectricity would thus result in a capture potential of between 233 and 316 g_{CO₂} per MJ of bioelectricity.

14.3.3.1.2 Biomass Co-firing with Coal. Biomass co-firing with coal has been intensively investigated as a way to improve coal power plant environmental performance, in terms of CO₂, NO_x and SO_x emissions.^{57,59} Depending on the coal quality and power generation efficiency, a coal power plant emits on average between 600 and 1400 kg_{CO₂} per MWh.⁶⁰ It is important to note that co-firing biomass would actually increase the emissions of a given power plant, as biomass is a lower heating value fuel, and must therefore be converted in a higher flowrate than that of coal to produce the same amount of power. The decrease in carbon intensity can only be true if biomass carbon neutrality is acknowledged: all the CO₂ released upon biomass combustion is captured by the biomass from the atmosphere and should therefore not be accounted for. Biomass sustainability is thus the key to considering that bioenergy leads to CO₂ emissions reduction.

Co-firing biomass and coal at high biomass proportions is however not straightforward. IEA Bioenergy communicated in a report that providing a finely grinded biomass, no modification of the boiler would be needed under a biomass proportion of 40% in energy, which corresponds to approximately 60% share in mass.⁶¹ Dedicated firing of biomass has been mainly developed at small scale, with capacities of between 1 to 100 MWe, a size that is mainly constrained by biomass availability and logistics costs.²⁵ As of April 2017, the 50 MW Mikawa power station has started operating on dedicated biomass in fluidised bed boilers. A 500 t_{CO₂} per day post-combustion capture plant is expecting to be running by 2020.⁶² In 2015, the UK DRAX power plant, formerly one of the largest coal plants in the UK, initiated a transition towards bioenergy by converting three of its six 660 MW ~~reactors~~ to dedicated biomass firing. At the time of writing (2019), four of the boilers are not operating with dedicated biomass, and 1 tCO₂ per day is being captured in a demonstration project in partnership with the CO₂ capture company C-capture. However, no option for CO₂ utilisation or storage has yet been identified. Furthermore, the technical implications to the boiler efficiency, combustion performance, material integrity, and fuel handling remain to be precisely evaluated. Immediate deployment of bio-electricity is thus more likely to occur on a small scale rather than on large scale, which according to a recent study by Sanchez *et al.*, is not the optimal scale for a BECCS system.⁶³ However, in order to be economically feasible, CCS is likely to be deployed on larger infrastructures. This difference between a bioenergy and CCS systems adequate scale could represent an obstacle to BECCS deployment.

AQ:5

14.3.3.1.3 Efficiency Penalty. A BECCS power generation facility suffers from a double efficiency penalty relative to unabated coal-fired power plants: first from the CO₂ capture system and second from the combustion of biomass. In a conventional solvent-based post combustion CO₂ capture process, steam is taken out of the power plant steam cycle to provide heat for solvent generation. As a result, a regular coal and

post-combustion capture facility operating at 90% capture with conventional solvents (*i.e.* alkanolamines) is 9% less efficient in efficiency points than a conventional power plant.⁶⁴ This efficiency could be reduced by decreasing the solvent regeneration heat duty, down to 2.3 GJ per t_{CO₂} with industrial solvents such as Cansolv, and down to 2 GJ per t_{CO₂} for advanced solvents.^{65,66} Options to further reduce this efficiency penalty have been investigated. In a study by Bui *et al.*, heat recovery from the combustion flue gas to supply the solvent regeneration heat duty was studied. The study shows that the adiabatic flame temperature and flowrate of flue gas could increase with biomass co-firing proportion, hence increasing the amount of recoverable heat for solvent regeneration. At 50% co-firing, 100% of the solvent regeneration heat duty was available from heat recovery, enabling the BECCS facility to reach an efficiency of 38%_{HHV}.⁶⁷

The second efficiency penalty is associated with the combustion or gasification of a lower quality fuel, mainly due to biomass lower energy density, lower O/C ratio, higher ash content and higher moisture content than that of coal. Table 14.2 gathers the average compositions of different coal (bituminous and lignite) and biomass types (switchgrass, poplar and wheat straw).

A higher moisture content and a lower O/C result in more energy loss through water evaporation, and more exergy losses.⁶⁸

Harder to quantify, another potential source for efficiency penalty comes from the potential higher slagging rate on the boiler walls, which could hinder heat transfer in the boiler, thus resulting in an overall lower boiler efficiency.^{69,70} Overall, efficiencies of as low as 17%_{LHV} and as high as 32%_{LHV} were observed with a BECCS power plant,²⁵ with the world average efficiency of coal power plants being approximately 30%_{LHV}.⁶⁰

Table 14.2 Elemental composition, higher heating value and moisture content of various biomass types (wheat straw, switchgrass and poplar) and coal (bituminous and lignite).

Composition	Switchgrass	Wheat straw	Poplar	Lignite coal	Bituminous coal
C (% _{DM}) ^a	46.9–47.8	46.0–48.3	48.5–51.6	63.5	72.5
H (% _{DM}) ^a	5.7–5.9	5.1–5.5	5.9–6.3	4.52	4.51
O (% _{DM}) ^a	35.1–43.0	38.3–41.4	41.5–43.7	28.5	9.04
N (% _{DM}) ^a	0.38–1.17	0.47–1.65	0–0.47	0.31	1.3
S (% _{DM}) ^a	0.06–0.11	0.07–0.10	0–0.05	0.70	0.7
Ash (% _{DM}) ^a	4.6–10.1	5.0–7.2	0.6–2.3	3.3	7.97
HHV (MJ per kg _{DM}) ^a	18.0–18.9	18.2–18.4	19.4–20.8	24.6	30.2
Moisture (% _{WM})	10–15 ^b	5–16 ^c	50–53 ^d	13.4 ^a	4.0 ^a

^aRef. 34.

^bRef. 43 and 109.

^cRef. 48, 51, 69, and 161.

^dRef. 105.

14.3.3.1.4 Biomass Handling and Logistics. On average, due to a lower O/C ratio as compared to coal, biomass higher heating value lies between 18 and 21 GJ per t_{DM} .³⁴ With a moisture content varying between 10–50% at harvest, the biomass heating value can thus drop down to 9–18 GJ per tonne, as compared to 30–38 GJ per tonne for coal. In a power plant, at a fixed heat rate, this directly impacts upon the fuel flowrate, which in turn impacts not only the size of the equipment, but also the logistics of handling a higher mass flow of fuel. According to Hetland *et al.*, over 500 of biomass must be provided per hour for a 40% efficient 800 MWe biopower plant operating at full capacity, which would correspond to seventeen 30 tonne lorries delivering biomass every hour.²⁵ Unless such a constant supply of biomass can be assured, such flowrates would likely require on-site storage of biomass. As such storage needs to be ventilated, or at least be in an inert atmosphere, to avoid biomass decomposition and CO build-up, these facilities would represent another efficiency penalty to the facility.

AQ:6

AQ:7

Biomass would also likely be sourced from different places. This means that the power plant would not have a constant biomass incoming stream in terms of physical properties. Whilst gasifiers are relatively flexible to the quality and properties of the biomass received, pulverised combustion boilers might not show this flexibility.⁶⁸ Instead of modifying conventional boilers, replacing them with more flexible devices, such as fluidized bed combustors, is another option.⁷¹

14.3.3.1.5 Biomass Pre-treatment at the Conversion Plant. Pre-treatment at the conversion plant is often required prior to biomass conversion. As moisture hinders the combustion performance, drying biomass to a moisture content of 10–15% is required.^{72,73} As a large scale power plant would require large amounts of biomass, it is likely that this biomass would have to ~~come ready dried and condensed as a pellet or briquette,~~ to facilitate long distance transport. For example, DRAX imports 65% of its biomass pellets supply from North America.⁷⁴ Grinding biomass prior to conversion is also compulsory. Though biomass maximal allowable particle size prior to conversion is higher than that of coal, it needs to be finely milled, to between 1 and 8 mm.⁷¹ This results in potential high grinding costs of approximately 350 MJ per t_{WM} , which further increases the BECCS efficiency penalty.⁷⁵

AQ:8

To improve the biomass density, heating value, combustion performance, grinding performance, resistance to moisture uptake and deterioration during storage, ~~as well uniformize the properties regardless of the biomass type,~~ the feedstock can also be torrefied prior to combustion.⁷⁶ Torrefaction is a mild temperature process at 200–300 °C, which occurs under an inert atmosphere, and which has been studied in the bioenergy industry to improve biomass properties for both combustion and gasification purposes.^{77,78} A drying–torrefaction process typically requires an energy input of 330 MJ per t_{DM} .⁴¹ However, it is important to note that drying typically accounts for 80% of the heat requirement.⁷⁹ In a scenario where drying down

to 10% moisture is already required, the torrefaction energy requirements would represent a marginal additional energy cost, likely to be compensated by the enhanced properties of the biomass. Indeed, in order to avoid moisture uptake and CO emissions from biomass storage, biomass needs to be stored in a ventilated and inert atmosphere. By increasing the hydrophobicity of biomass, torrefaction could enable outside storage, thereby decreasing the biomass storage cost. Furthermore, torrefied biomass grindability could be substantially lower than that of raw biomass, and possibly that of coal. In a study by Williams *et al.*, grinding energy costs were found to drop from 346–367 MJ per t_{WM} for raw pellets, to 14 MJ per t_{WM} for torrefied pellets, hence potentially five times lower than energy costs for coal.⁷⁵ In addition to lowering grinding and storage costs, torrefaction would also increase biomass combustion or gasification performance, hence increasing the biomass conversion efficiency.

14.3.3.2 Carbon Negative Biofuels

Another way to generate negative emissions *via* BECCS is through biofuel production with carbon capture and storage. In 2014, bioenergy supplied 4% of transport fuels, with a forecast to reach 4.3% in 2020.⁵⁸ Conventionally, oil crops have been used to produce biodiesel *via* esterification, and sugar and starch crops, bioethanol *via* fermentation. To avoid competition with food, the biofuel industry has been turning towards other feedstocks, such as lignocellulosic biomass from agriculture residues or energy dedicated crops. In a BECCS *via* biofuels pathway, CO₂ can be captured at the fermentation process and from the cogeneration boiler, usually fuelled with natural gas.³⁶

14.3.3.2.1 Efficiency Penalty. As the facilities and biomass supply chain networks already exist at large scale for biofuel production, biomass logistics are less of an issue when it comes to biofuel facilities. However, a bioethanol facility does not generate power, and power for the capture system must be supplied from the cogeneration boiler or from an extra power generation facility. In a study by Laude *et al.*, capturing CO₂ from the fermentation unit increased the non-renewable energy consumption by 40%, and capturing CO₂ from both the fermentation and the cogeneration units, by 60%. For the Decatur corn bioethanol facility, an additional unit of 100 MW will be required to provide the steam required for post combustion capture.⁸⁰ However, the fermentation process is particularly privileged as the purity of the CO₂ stream is very high, hence allowing capture and storage at lower costs.

Another efficiency penalty may be incurred by the progressive shift from 1st generation bioethanol to lignocellulosic biomass in 2nd generation bioethanol production. Due to their lower cellulose content, chemical (with ionic liquids for instance) or thermal pre-treatment (with steam explosion of the biomass) of the lignocellulosic feedstock is often mandatory, which in turn increases the cost of the process.⁸¹

14.3.3.2.2 Carbon Negative Potential. Another challenge to the deployment of this BECCS pathway lies in the CO₂ life cycle emissions of BECCS *via* biofuels. Because the CO₂ released upon the combustion of the biofuel is not captured, there are unabatable CO₂ emissions associated with the production of carbon negative biofuels. As a result, the carbon efficiency of BECCS *via* a biofuels pathway will be naturally lower than that of BECCS *via* bioelectricity. In a study on bioethanol from switchgrass fermentation with CCS, the negative potential of bioethanol was evaluated at 100 g_{CO₂} per MJ of bioethanol, including the lifecycle CO₂ emissions of the process.³² However, another study by Laude *et al.*, showed that the overall CO₂ balance of sugar beet bioethanol with CCS was still carbon positive when only fermentation process emissions were captured. Carbon negativity of the overall process was only achieved when both the cogeneration and fermentation units' emissions were captured. It is important to note that the total carbon balance also depends on the fuel used for cogeneration; when shifting natural gas to biomass residue, the carbon negative potential could further increase. At 100 g_{CO₂} per MJ of bioethanol, carbon negative fuels have a carbon removal potential 2.3 to 3.2 times lower than that of carbon negative bioelectricity. Delivering negative emissions *via* biofuels will likely have to be deployed in conjunction with other pathways, such as carbon negative bioelectricity, in order to meet the negative emissions targets.

AQ-9

14.3.3.3 Carbon Negative Industry: the Case of Pulp and Paper

Another way of deploying BECCS is through the deployment of CCS on industrial sources of biogenic emissions. Producing 1–3 t_{CO₂} per t_{DM} of pulp, of which 75–100% is biogenic, the pulp and paper industry has been investigated as a potential carbon sink for climate mitigation.^{37,39} In a typical pulp and paper mill, CO₂ is emitted from three sources: the recovery boiler, the multi-fuel boiler, and the lime kiln, which are fuelled with either fossil fuel or biomass. The feasibility of retrofitting amine-based post-combustion capture on these sources has been studied,^{39,83} and mainly relies on the feasibility of generating steam for the capture system, while minimising the energetic cost incurred to the system. Options such as the integration of a natural gas combined cycle, heat integration to use the excess steam for the process, and a biofuel-integrated gasification combined cycle have been evaluated in terms of energy penalty incurred on the system.⁸³ In Onarheim *et al.*, a case study of retrofitting 90% post-combustion capture to a modern pulp mill, and an integrated pulp and board mill showed that, providing steam turbine driven CO₂ compressors are used, the steam demand would increase from 0.7 GJ per t_{DM} to 7.8 GJ in the stand-alone mill, which could provide more steam from the process, to 13 GJ per t_{DM} for the integrated pulp and board mill.³⁹ In terms of negative emissions potential, the FAO reports that the total pulp and mill production amounted to 390 Mt_{DM} per year in 2015.⁵⁶ Considering the potential biogenic emissions emitted per ton

AQ-10

of dry product and 90% capture, 0.26–1.05 Gt_{CO₂} per year could theoretically be removed from the atmosphere *via* retrofitting pulp and paper plants with CCS. Though not negligible, this underlines the need for the deployment of other BECCS pathways, in addition to niche applications such as the pulp and paper industry, to meet our mitigation targets by 2100.

14.3.4 Sustainability Challenges of BECCS

Beyond technical feasibility considerations, there has been questions as to BECCS capacity to provide sustainable and resource efficient negative emissions, especially when it comes to water, land and energy use. In assessing BECCS sustainability, it is agreed that biomass supply chain in particular plays a determining role.^{84–86} Defining a water, energy, CO₂ and land accounting framework is ~~however~~ critical in order to design more efficient and sustainable supply chains.

14.3.4.1 Water Use

As of today, the agriculture and energy sectors account for 85% of the water consumption in the world.^{87,88} Throughout the BECCS value chain, water can be used during biomass production, biomass pre-treatment and biomass conversion, with the water footprint of each step depending on the biomass type and the conversion pathway studied. There is therefore a rising concern that BECCS deployment at the EJ scale could put additional pressure on an already scarce resource.⁸⁹

14.3.4.1.1 Biomass Water Footprint. Water consumption is defined as the amount of water evaporated, transpired, or otherwise removed from the water environment, as opposed to water withdrawal, which is defined as the amount of water removed from the ground or diverted from a water source for use, which is then returned to the source. This distinction is important as the choice of one metric or the other may potentially impact upon the order of magnitude of the BECCS water footprint.

If biomass is sourced from agriculture, water is consumed at the field level during crop growth, through evapotranspiration. As detailed in the FAO evapotranspiration guidelines, evapotranspiration is strongly dependent on the region climate where the biomass is produced, and is evaluated *via* the region reference evapotranspiration, and the crop water use efficiency during its growth cycle.⁹⁰ In order to distinguish between fresh water use and rain water use, the concepts of “green” and “blue” water footprints are used.⁹¹ While the green water footprint is the amount of rainwater absorbed by the crop, the blue water footprint can be defined as the marginal amount of fresh water required in addition to the green water footprint, in order to meet the crop evapotranspiration requirements. Depending on the precipitations in a given region, the share of the blue water footprint in total evapotranspiration may vary. Finally, the amount of polluted water resulting

Table 14.3 The green, blue, and grey water footprints of different crops – miscanthus, wheat, maize – in different regions – Brazil, Netherlands, and the world average. The biomass water footprints values are highly dependent on biomass type (yield and water efficiency) and climate (reference evapotranspiration and precipitation).

Crop water footprint (m ³ per ton)	Green water ¹⁶²	Blue water ¹⁶²	Water requirement = green + blue water ¹⁶³	Grey water ¹⁶²
Miscanthus – Brazil	—	—	828	—
Miscanthus – US	—	—	334	—
Wheat – Brazil	1989	1	—	135
Wheat – Netherlands	511	—	—	183
Wheat – World average	1277	342	—	207
Sugarcane – World average	139	57	—	13
Maize – World average	947	81	—	194

from the production of the biomass at the field level is defined as the grey water footprint, and is usually linked to nitrogen leaching from fertilizer input.⁹² Table 14.3 shows the values of the green, blue and grey water footprints for different crops.

As shown in Table 14.3, the share of the blue water footprint is dependent on the region climate (evapotranspiration and precipitation) and biomass type (water efficiency and yield). In bioenergy or BECCS water accounting, the assumptions behind the calculations can therefore yield very different results. In a study by Smith *et al.*, the BECCS water use at the field level was calculated by subtracting the crop evapotranspiration by that of a grassland, which was considered as the counterfactual scenario in their analysis. Assuming biomass evapotranspiration is around 1530 (1176–1822) m³ per year per ton of carbon captured, and subtracting by 1450 (900–2000) m³ per t_C per year, the evapotranspiration of a grassland, the BECCS water intensity at the field level was found to be around 80 m³ per t_C per year. It is worth noting that this value is also strongly dependent on the biomass capture potential in t_C per ha per year of the field, which is assumed to range between 4.7–8.6 t_C per ha per year for miscanthus in this analysis.³³ In work by Fajardy and Mac Dowell however, the blue, green and grey water footprints of four different bioenergy crops were computed. The biomass water footprints at the field level were found to be between 1100–3000 m³ per t_C, for a BECCS carbon capture potential of between 3.5–9 t_C per ha per year.⁸⁵

14.3.4.1.2 Biomass Pre-treatment and Conversion Biomass Water Footprint. In biomass conversion to bioethanol, water is used for both the fermentation process and the on-site power generation. In the case of cellulosic bioethanol, water consumption was evaluated within a 1.9–9.8 m³ L⁻¹ range. Moreover, adding CO₂ capture to a bioethanol facility will result in additional water consumption from steam generation.

In bio-power generation, electricity production and CO₂ capture incur a water cost that needs to be added to the overall balance, through the steam consumption at the process level. In electricity production, water

Table 14.4 Water consumption and withdrawal factors of different cooling technologies and power generation technologies. The choice of the cooling technology and factor – consumption and/or withdrawal – impacts the power plant water intensity by two orders of magnitude. Adapted from Macknick *et al.*, 2012.¹⁶⁴

Technology	Water consumption gal per MWh	Water withdrawal gal per MWh
Bio-power – Tower	480–965	500–1460
Bio-power – Once-through	300	20 000–50 000
Supercritical coal – Tower	458–594	582–609
Supercritical coal – Once-through	6–124	22 551–22 611
Supercritical Coal + CCS – Tower	942	1098–1148

consumption is strongly dependent on the cooling technology used in the process, but more predominantly, if it is the water withdrawal or water consumption that is considered in the balance. Water consumption and withdrawal from different technologies including bio-power, supercritical coal, supercritical+CCS, for both a cooling tower and a once-through cooling system are summarized in Table 14.4.

Whilst water consumption factors are similar from one technology to another, water withdrawal can be substantially different. For once-through systems for example, the water withdrawal factor is up to 350 times greater than the water consumption factor in the case of supercritical coal. It is thus important to specify the technology and definition adopted in the analysis when defining BECCS water use. For example, in a study by Smith *et al.*, BECCS water use at a capture plant represented over 60% of the BECCS water intensity.³³ However, in work by Fajardy and Mac Dowell, as the water consumption of a recirculating cooling tower was considered, the BECCS water use at the plant level was evaluated to be between 16–28 m³ per t_C. The weight of the power plant water footprint was thus found to be marginal compared to the biomass water footprint.⁸⁵ Table 14.5 summarizes different BECCS water footprint values, along with the assumptions of each evaluation.

14.3.4.2 Land Use

14.3.4.2.1 BECCS Land Footprint. The BECCS land footprint is strongly dependent on the feedstock type. If no or little land footprint can be attributed to municipal waste, the land footprint for agriculture and forestry biomass could be potentially great, and depends on two main parameters: 1) the crop yield per hectare of land cultivated and 2) the system carbon efficiency. A cereal crop such as wheat typically yields between 3 and 9 t_{DM} per ha per year,⁵⁶ whereas perennial grasses such as miscanthus can yield between 15–30 t_{DM} per ha per year on average, and up to 40–60 t_{DM} per ha per year in some experiments.^{52,53,93} This yield difference has a direct impact on the amount of CO₂ captured per ha

Table 14.5 BECCS water footprints across the literature, along with the models' assumptions. Depending on the water footprint perspective adopted at the field level (marginal evapotranspiration, total evapotranspiration, total evapotranspiration and grey water) and at the power plant level (water consumption or withdrawal, cooling technologies), the BECCS water footprint is found between 220 and 7400 m³ per t_C, *i.e.* 60 and 2020 m³ per t_{CO₂}.

	Field	Power plant	Total	Assumptions
Smith <i>et al.</i> (2016) ³³	80	—	720 km ³ per year to remove 3.3 Gt _C per year <i>i.e.</i> ≈ 220 m ³ per t _C	<ul style="list-style-type: none"> • Carbon capture of 4.7–8.6 t_C per ha per year • Evapotranspiration subtracted from grassland evapotranspiration: 1530–1450 m³ per t_C per year
Fajardy & Mac Dowell (2017) ⁸⁵	1100–2960	16–28	1120–2990 m ³ per t _C	<ul style="list-style-type: none"> • Miscanthus-based BECCS • Carbon capture of 3.5–9 t_C per ha per year • Water footprint at the field = green, blue and grey water • Water consumption of recirculating wet cooling tower in the power plant.
Smith <i>et al.</i> (2013) ¹¹³			1600–7400 m ³ per t _C	<ul style="list-style-type: none"> • Switchgrass-based BECCS • Carbon efficiency of 47%: out of the 2.11 t_{CO₂} stored in the biomass, 1 t_{CO₂} is stored geologically

of land. In Smith *et al.*, between 5.8 and 8.6 t_C per ha per year is available for the capture for miscanthus, but only 1.66–1.78 t_C per ha per year is available for capture for agricultural residues.³³ Table 14.6 summarizes the BECCS land footprints across the literature.

Again, these results are highly dependent on the feedstock and the supply chain considered, hence the variability. Considering this broad range, deploying BECCS at the required rate, 8.5–16.5 t_{CO₂} per year, could thus require the mobilisation of between 260–4320 Mha. Land classification usually distinguishes land by both economic activities and vegetation types: cropland, grassland, forests, inland waters, marginal land, *etc.* To put the BECCS land footprint values into perspective, the amount of land available for bioenergy production was assessed between 375–1114 Mha.^{94,95} Using high quality lands such as grassland or cropland for BECCS could thus result in a

Table 14.6 BECCS land footprints across the literature, along with the models' assumptions. The BECCS land footprint is found to be 1.019.0 t_C per ha per year, or 3.7–33.4 t_{CO₂} per ha per year.

	Average land intensity	Assumptions	
Smith <i>et al.</i> (2016) ³³	4.7–8.6 t _C per ha per year	<ul style="list-style-type: none"> • Carbon balance at the field level (NO₂ emissions, direct and indirect land use change) • Land intensity ranges for miscanthus and willow 	5
Fajardy and Mac Dowell (2017) ⁸⁵	3.5–9.0 t _C per ha per year	<ul style="list-style-type: none"> • Miscanthus-based BECCS • Carbon balance including field production (NO₂ emissions, in-field operations), processing, transport, conversion of the biomass, and CO₂ capture • Water footprint at the field = green, blue and grey water • Water consumption of recirculating wet cooling tower in the power plant. 	10 15
Smith <i>et al.</i> (2013) ¹¹³	220–990 Mha per Gt _C per year <i>i.e.</i> 1.01–4.5 t _C per ha per year	<ul style="list-style-type: none"> • Switchgrass-based BECCS • Carbon efficiency of 47%: out of the 2.11 t_{CO₂} stored in the biomass, 1 t_{CO₂} is stored geologically • Carbon balance including field production (NO₂ emissions, in-field operations), processing, transport, conversion of the biomass, and CO₂ capture 	20 25

strong tension with other markets such as food production. To avoid this competition, the use of marginal land – *i.e.* land that has not been assigned to any economic activity – for biomass has been investigated. However, marginal land is diverse in quality and type, which makes it complex to predict the biomass productivity response, as well as its availability.⁹⁶ Maximising the BECCS carbon efficiency by mobilising low carbon footprint supply chains of high yielding biomass will thus be crucial in BECCS sustainable deployment to meet climate targets.

14.3.4.2.2 Land Use Change. The conversion of land for bioenergy purposes also raises the question of direct and indirect CO₂ emissions associated with it, otherwise defined as direct (LUC) and indirect (ILUC) land use changes. Figure 14.6 provides a graphical explanation of potential emissions from land conversion.

Direct land use change is defined as the direct and indirect CO₂ emissions emitted upon conversion of land to bioenergy production, *i.e.* the change in the total carbon stock captured by ~~the vegetation~~. These values are very dependent on the vegetation types, and have been evaluated to be between

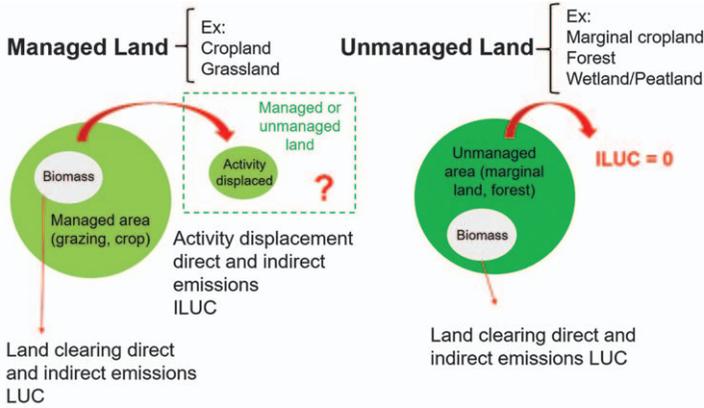


Figure 14.6 Diagram explaining the differences between direct (LUC) and indirect (ILUC) land use change mechanisms. Adapted from ref. 85, <http://dx.doi.org/10.1039/C7EE00465F>, under the terms of the CC BY 3.0 licence, <https://creativecommons.org/licenses/by/3.0/>.

6–69 kg_{CO₂} per ha for marginal land, to between 1 000 000–3 052 000 kg_{CO₂} per ha for a peatland forest. However, it was also found that when converting a low vegetation land, such as a marginal land or a cropland, to a perennial grass crop or a short rotation coppice, ~~that~~ the impact on the soil carbon pool could also be positive. For instance, in work by Smith *et al.*, a negative carbon change of –0.46 t_C per ha per year was considered when converting a land to miscanthus.³³ In a study by Cherubini *et al.* in 2009, a soil carbon sequestration potential of 65 kt_{CO₂} per year was also attributed to switch-grass production. Including this soil carbon sequestration, though not permanent, could thus temporarily improve biomass land use change effects.⁹⁷

AQ:11

~~However, to the conversion of the land must be added the CO₂ emitted resulting from the reallocation of the activity which has been supplanted by bioenergy, and which occurs when the land converted had a previous economic activity.~~ If a grassland was previously used for grazing, or a cropland for corn production, the CO₂ emitted during the reallocation of this activity, by converting a parcel of a forest for example, has proven to be substantially high in some cases, to the point of offsetting the carbon negativity of BECCS.⁸⁵ These conversions factors have been evaluated on specific case studies on biofuels, and depend strongly on the timeframe considered, biomass productivity, country and economic activity displaced. Figure 14.7 provides ranges of ILUC emissions factors across the literature.^{98–101}

For example, Plevin *et al.* evaluated these emission factors to range between 10–340 g_{CO₂} per MJ for biofuels, hence potentially higher than the 92 g_{CO₂} per MJ for gasoline life cycle emissions. Evaluating these effects with precision, as well as limiting their impact, is thus vital to ensure BECCS sustainability.

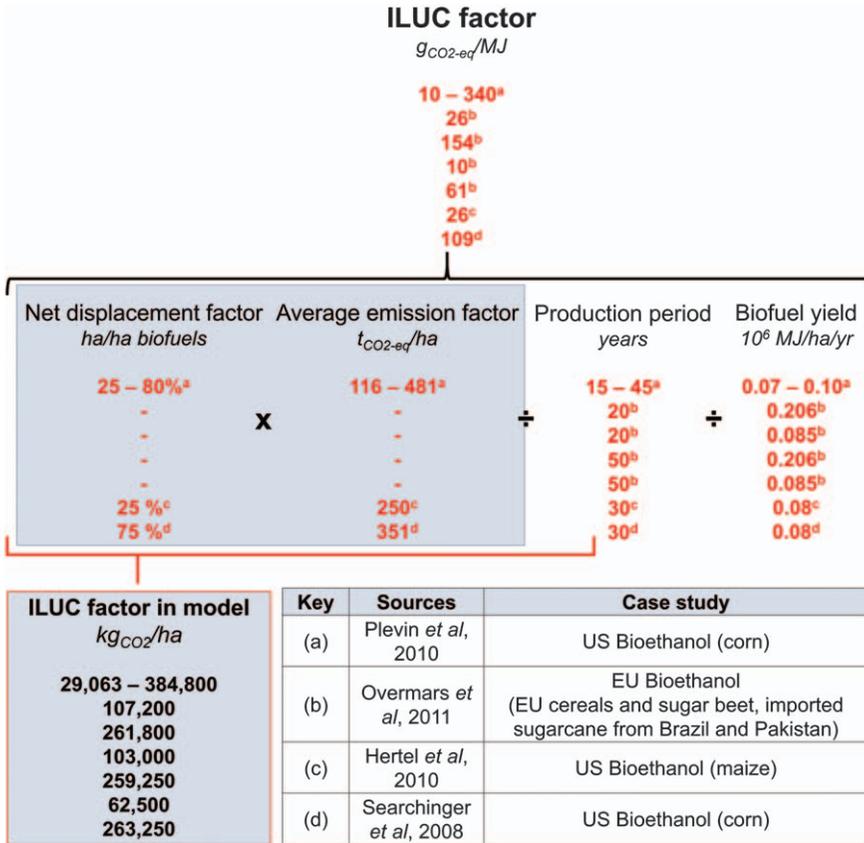


Figure 14.7 Indirect land use change emissions factors across the literature. Each value is highly case specific and depends mainly on the land emission factor, bioenergy yield and lifetime considered.

Adapted from ref. 85, <http://dx.doi.org/10.1039/C7EE00465F>, under the terms of the CC BY 3.0 licence, <https://creativecommons.org/licenses/by/3.0/>.

14.3.4.2.3 BECCS Carbon Breakeven Time. Theoretically, a BECCS or bioenergy facility ~~CO₂ removal capacity~~ is characterized by a CO₂ removal rate in t_{CO₂} per hour, its annual operation time in hours, and its lifetime in years. However, when considering land use change, *i.e.* the CO₂ emitted upon conversion of land at the beginning of a BECCS project, an initial “carbon debt” is incurred on the system, and will need to be amortized as a function of the lifetime of the BECCS project. Depending on the value of this initial carbon debt, and on BECCS CO₂ removal potential per year, the BECCS mitigation effect could be delayed. This is referred to as “carbon breakeven time”. Calculated in work by Withers *et al.* for biofuel production from forest management wood, it was defined as the time required for biofuels to start providing emissions reduction as compared to

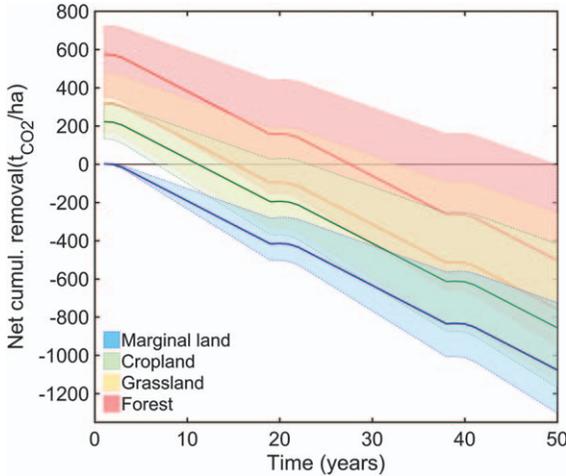


Figure 14.8 Cumulative CO₂ equivalent emissions over the lifetime of a BECCS project, as a function of the type of land used for the production of the biomass feedstock. When the production of biomass displaces a forest, cumulative emissions could become net negative after 50 years of operation of the BECCS plant. When biomass is grown on marginal land however (no direct nor indirect land use change), cumulative emissions in this case can become net negative in 1 to 2 years. This is the concept of carbon breakeven time.⁸⁵

a benchmark – gasoline.¹⁰² In this analysis, the carbon breakeven time of biofuels was found to be 59 years, considering a biofuel production over 30 years. In a study by Fajardy and Mac Dowell, the concept of carbon breakeven time was introduced for BECCS, and defined as the time required for the system to reach carbon negativity. In their analysis, depending on the fuel type, and conditions of the value chain deployment, the BECCS carbon breakeven time was found to reach values of greater than 50 years on grassland when considering land use change effects.⁸⁵ Figure 14.8 illustrates how different cumulative emissions over the lifetime of a BECCS plant, as a function of the land used for the production of the biomass feedstock, may lead to variable breakeven times.

Carefully evaluating the extent of the impact of land use changes is thus key in BECCS carbon accounting, both from a static and dynamic perspective.

14.3.4.3 Life Cycle Energy Use and CO₂ Emissions

Producing, processing and transporting biomass to a conversion facility can represent a significant amount of energy use and CO₂ emissions. This is referred to as biomass embodied energy and carbon footprint, and accounting for these values casts doubt on BECCS ability to efficiently remove CO₂ from the atmosphere. Furthermore, high biomass embodied energy

could result in BECCS being a net energy consumer, rather than a ~~consumer~~. Life cycle assessment of the biomass supply chain has been studied intensively,^{84,93,103–106} and the results strongly depend on the feedstock – dedicated agriculture, waste – and the boundaries of each case study. Tables 14.7 and 14.8 summarize ~~the~~ embodied energy and carbon footprint values across the literature.

As it can be observed, ranges vary a lot depending on the processes included in the models. A few key contributions to the biomass supply chain, such as fertilizer application, processing and transport, are described in this section.

14.3.4.3.1 Fertilizer Application. An important impact to consider is the application of nitrogen-based fertilisers. Wheat for example requires an average nitrogen fertilizer input of between 50–200 kg per ha per year.⁵⁶ Knowing that this chemical product has a carbon footprint of between 2.6–4.8 kg_{CO₂,eq} per kg_N,¹⁰⁴ and that wheat typically yields between 2–9 t_{DM} per ha per year of grain,⁵⁶ the resulting carbon footprint of wheat grain could be between 15–290 kg_{CO₂} per t_{DM}. To this can be added the potential nitrous oxide emissions from nitrogen soil decomposition. The decomposition mechanism is complex and depends on:

- the direct decomposition of nitrogen into N₂O–N;
- the amount of nitrogen that volatilizes into NH₃–N and NO_x–N,
- the amount of nitrogen that leaches,
- the amount of N₂O–N formed from the leached nitrogen,
- the amount of N₂O–N formed from the volatilized nitrogen.

Nitrogen decomposition mechanisms ~~will be~~ strongly dependent on feedstock and soil type, which makes these phenomena very complex to model. Guidelines to compute these emissions factors are available,¹⁰⁷ and based on values from the literature for each of these coefficients, emissions could potentially represent 5–37 g_{N₂O} per kg_N applied.^{108,109} As nitrous oxide has a global warming potential of 298 times that of CO₂, this could represent an emission factor of 1.5–11 kg_{CO₂,eq} per kg_N, hence potentially more than the nitrogen fertilizer carbon footprint.

For agricultural residues, nutrients used for the production of the main crop are not necessarily taken into account. However, the addition of nutrients to the crop resulting from the removal of residues must be accounted for. As explained in Paraluji *et al.*, a certain fraction of the straw nitrogen content is available to the crop. Assuming that 30% of the nitrogen in the straw is available to the crop, and a nitrogen content between 0.5–1.7%, removing straw from a wheat crop could result in an extra fertilizer use of 0.9–2.7 kg_N per t_{DM} of straw.⁵¹ Considering the nitrogen-based fertilizer carbon footprint, as well as N₂O emissions factor, the incurred direct and indirect CO₂ emissions would lie between 7–43 kg_{CO₂} per t_{DM} of straw, which is still much lower than that of conventional wheat.

Table 14.7 Biomass embodied energy values across the literature, along with the models' assumptions. Biomass embodied energy is found to be in the range of 0.21–13.3 G_e per t_{DM}. This broad range is influenced by the variability of the model assumptions (biomass yield, transportation distance) and the range of processes included (irrigation, grinding, drying).

Case study	Assumptions – model boundaries	Value	Source
Miscanthus, average	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation 	0.52 GJ per t _{DM}	Camargo <i>et al.</i> , 2013 ¹⁰⁴
Miscanthus, Germany	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation, • 100 k transport, • Chopping, milling, • Combustion 	1.25 GJ per t _{DM}	Lewandowski, 1995 ⁹³
Miscanthus, Ireland	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation, • Baling • Drying, pelleting, • 100 k transport 	0.28 GJ per GJ HHV <i>i.e.</i> ≈ 5–5.9 GJ per t _{DM}	Murphy <i>et al.</i> , 2013 ¹⁰⁶
Miscanthus, several	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation • Drying, pelleting • Transport to the UK. 	Brazil: 3.4–7.5 GJ per t _{DM} Europe: 2.5–5.4 GJ per t _{DM} US: 3.0–6.7 GJ per t _{DM}	Fajardy and Mac Dowell, 2017 ⁸⁵
Switchgrass, US	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation, • Baling or chopping. 	1.5–1.9 GJ per t _{DM}	Kalita, 2012 ¹⁶¹
Switchgrass, US	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) excl. N₂O emissions and excl. irrigation, • Baling. 	0.21–0.24 GJ per t _{DM}	Lu <i>et al.</i> , 2015 ⁴³
Switchgrass, average	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation. 	1.1 GJ per t _{DM}	Camargo <i>et al.</i> , 2013 ¹⁰⁴
Switchgrass, Texas US	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation, • Baling or chopping, • 40 k transport (pellets, bale, loose condensed). 	0.92–1.63 GJ per t _{DM}	Qin, 2006 ¹⁰⁹

Table 14.7 (Continued)

Case study	Assumptions – model boundaries	Value	Source
Miscanthus, several	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation • Drying, pelleting • Transport to the UK. 	Brazil: 2.5–7.3 GJ per t _{DM} Europe: 1.7–4.4 GJ per t _{DM} US: 2.4–5.3 GJ per t _{DM}	Fajardy and Mac Dowell, 2017 ⁸⁵
Willow, Sweden	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) excl. N₂O emissions and excl. irrigation, • 50 k transport. 	0.86 GJ per t _{DM}	Borjesson, 1996 ¹⁶⁵
Willow, average	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation. 	1.5 GJ per t _{DM}	Camargo <i>et al.</i> , 2013 ¹⁰⁴
Willow, Belgium	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) excl. N₂O emissions and excl. irrigation. 	3–16 GJ per ha per year	Djomo <i>et al.</i> , 2013 ¹⁶⁶
Wood pellets, Australia & Russia	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) excl. N₂O emissions and excl. irrigation, • Drying (50% of the mass), • Pelleting, • Transport from Australia and Russia to the US. 	1.13–7.5 GJ per t _{DM}	Ehrig, 2014 ¹⁰⁵
Willow, several	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation • Drying, pelleting • Transport to the UK. 	Europe: 7.5–5.3 GJ per t _{DM} US: 8.3–13.3 GJ per t _{DM}	Fajardy and Mac Dowell, 2017 ⁸⁵
Wheat straw, New Zealand	<ul style="list-style-type: none"> • Straw collection, • Baling, • 90 k transport, • Grinding, • Combustion 	0.24 GJ per GJe <i>i.e.</i> 1.3–1.5 GJ per t _{DM}	Forgie <i>et al.</i> , 2008 ¹⁶⁷
Wheat straw, UK	<ul style="list-style-type: none"> • Straw collection, • Baling, • 40k transport • Pre-processing, • Combustion. 	0.44 GJ per GJe <i>i.e.</i> 2.4–2.8 GJ per t _{DM}	Elsayed <i>et al.</i> , 2003 (Elsayed <i>et al.</i> , 2003)
Wheat straw, several	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation • Drying, pelleting • Transport to the UK. 	Brazil: 2.5–5.3 GJ per t _{DM} Europe: 1.7–3.0 GJ per t _{DM} US: 2.1–4.2 GJ per t _{DM}	Fajardy and Mac Dowell, 2017 ⁸⁵

Table 14.8 Biomass carbon footprint values across the literature, along with the models' assumptions. The biomass embodied energy is found to be 89–637 kg_{CO₂} per t_{DM}, without land use change, and potentially up to 3883 kg_{CO₂} per t_{DM} when considering land use change. This broad range is influenced by the variability of the model assumptions (biomass yield, transportation distance) and the range of processes included (irrigation, N₂O emissions, grinding, drying, direct and indirect land use change).

Case study	Assumptions – model boundaries	Value	Source
Miscanthus, Germany	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation, • 100 k transport, • Chopping, milling, 	111.8 kg _{CO₂} per t _{DM}	Lewandowski, 1995 ⁹³
Miscanthus, Ireland	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation, • Baling • Drying, pelleting, • 100 k transport 	20.6 kg _{CO₂} per GJ _{HHV} <i>i.e.</i> 371–433 kg _{CO₂} per t _{DM}	Murphy <i>et al.</i> , 2013 ¹⁰⁶
Miscanthus, several	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation and incl direct and indirect land use change (grassland, 50 years) • Drying, pelleting • Transport to the UK. 	Brazil: 327–791 kg _{CO₂} per t _{DM} Europe: 273–1463 kg _{CO₂} per t _{DM} US: 306–1283 kg _{CO₂} per t _{DM}	Fajardy and Mac Dowell, 2017 ⁸⁵
Switchgrass, US	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation, • Baling or chopping. 	191–204 kg _{CO₂} per t _{DM}	Kalita, 2012 ¹⁶¹
Switchgrass, US	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) excl. N₂O emissions and excl. irrigation, • Baling. 	145–147 kg _{CO₂} per t _{DM}	Lu <i>et al.</i> , 2015 ⁴³
Switchgrass, average	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation. 	172 kg _{CO₂} per t _{DM}	Camargo <i>et al.</i> , 2013 ¹⁰⁴
Switchgrass, Texas, US	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation, • Baling or chopping, • 40 k transport (pellets, bale, loose condensed). 	195–198 kg _{CO₂} per t _{DM}	Qin, 2006 ¹⁰⁹



Table 14.8 (Continued)

Case study	Assumptions – model boundaries	Value	Source
Switchgrass, several	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation and incl direct and indirect land use change (grassland, 50 years) • Drying, pelleting • Transport to the UK. 	Brazil: 265–3612 kg _{CO₂} per t _{DM} Europe: 199–2421 kg _{CO₂} per t _{DM} US: 199–2329 kg _{CO₂} per t _{DM}	Fajardy and Mac Dowell, 2017 ⁸⁵
Willow, UK-US	<ul style="list-style-type: none"> • Biomass production incl N₂O emissions, irrigation, and direct land use change (timberland, 40 years lifetime), • Processing, • 25–75 km transport to power plant. 	118–242 kg _{CO₂} per GJe <i>i.e.</i> 637–1525 kg _{CO₂} per t _{DM}	Stephenson <i>et al.</i> , 2014 (BEAC) ¹⁶⁸
Willow, average	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and excl. irrigation. 	89 kg _{CO₂} per t _{DM}	Camargo <i>et al.</i> , 2013 ¹⁰⁴
Wood pellets, Australia & Russia	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) excl. N₂O emissions and excl. irrigation, • Drying (50% of the mass), • Pelleting, • Transport from Australia and Russia to the US. 	143–594 kg _{CO₂} per t _{DM}	Ehrig, 2014 ¹⁰⁵
Willow, several	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation and incl direct and indirect land use change (grassland, 50 years) • Drying, pelleting • Transport to the UK. 	Europe: 571–2663 kg _{CO₂} per t _{DM} US: 646–3883 kg _{CO₂} per t _{DM}	Fajardy and Mac Dowell, 2017 ⁸⁵
Wheat straw, UK	<ul style="list-style-type: none"> • Straw collection, • Baling, • 40 k transport • Pre-processing prior to combustion. 	66 kg _{CO₂} per GJe <i>i.e.</i> 356–416 kg _{CO₂} per t _{DM}	Elsayed <i>et al.</i> , 2003 ¹⁰³
Wheat straw, several	<ul style="list-style-type: none"> • Biomass production (chemicals and in-field operations) incl. N₂O emissions and incl. irrigation, no direct and indirect land use change • Drying, pelleting • Transport to the UK. 	Brazil: 313–615 kg _{CO₂} per t _{DM} Europe: 248–492 kg _{CO₂} per t _{DM} US: 283–567 kg _{CO₂} per t _{DM}	Fajardy and Mac Dowell, 2017 ⁸⁵

Finally, an energy dedicated crop such as miscanthus requires a fertilizer input of 50–100 kg_N per ha per year, for an average yield of between 15–30 t_{DM} per ha per year. Under these assumptions, the carbon footprint associated with fertilizer application could thus reach levels in the range of 13–105 kg_{CO₂} per t_{DM} of miscanthus. Accounting for nitrogen-based fertilizer direct and indirect CO₂-equivalent emissions is thus strongly dependent on the feedstock type and yield, and can potentially have a great impact on the overall carbon balance.

14.3.4.3.2 Biomass Collection and Conditioning for Transport. The biomass carbon footprint also depends on the processing pathway used in the BECCS value chain, especially when it comes to transport.^{41,82} To increase the biomass density prior to transport, the feedstock may be baled, chipped or pelleted. As grinding and drying to 10% moisture are required prior to pelletisation, the cost is dependent on the biomass properties: moisture at harvest, particle size, and grindability.¹¹⁰ Woody biomass has a higher moisture content (50% at harvest) than that of perennial grasses (15–30%) or straw (8–15%). Depending on the dryer technology used, the energy requirement to remove 1 ton of water can be found to be within 1.3–4 GJ per t_{H₂O}.¹¹¹ hence there is a high embodied energy and carbon footprint associated with this stage.

14.3.4.3.3 Biomass Transport. Biomass transport naturally depends on the transport distance, transport mode and biomass density. Road transport of biomass in large capacity trucks can have an emission factor of up to 100 times higher than that of sea transport.^{103,112} Biomass density also impacts the final costs. In a study by Uslu *et al.*, the cost of shipping biomass from South America to Western Europe decreased from 0.9 GJ per t_{DM} for torrefied biomass to about 0.4 GJ per t_{DM} for torrefied and pelletised biomass.⁴¹ Although pelletisation adds substantially to biomass energy costs, there are clear trade-offs to consider between processing and transport costs.

14.3.4.3.4 Biomass Conversion and CCS. As mentioned in Section 14.2.5.3, biomass fine grinding at the power plant is required prior to conversion, which represents an extra CO₂ leakage to the carbon efficiency of the system.

In a CCS plant, 90% is often considered as the maximal capture efficiency, hence incurring a 10% carbon loss at this stage. As energy is used to capture, transport and inject the CO₂, further losses need to be considered. Depending on the carbon footprint of this power, a CO₂ loss of 6% was considered by Qin *et al.* in 2006, further decreasing the carbon efficiency of BECCS.¹⁰⁹ If BECCS is used on a power plant, the power can come from the power plant itself, incurring an efficiency penalty to the system. In a biofuel plant or pulp and paper factory, this electricity will have to be supplied to the system, and its carbon footprint will also need to be accounted for.

Based on tables, overall embodied energy values of as low as 0.21 GJ per t_{DM} and as high as 13 GJ per t_{DM} can be found across the literature. Assuming a biomass energy conversion efficiency of 30%_{HHV}, and a biomass HHV of between 18 and 21 GJ per t_{DM} , BECCS could be a net energy producer of 6 GJ per t_{DM} in the low embodied energy scenario, and a net energy consumer of 7.9 GJ per t_{DM} in the high embodied energy scenario. Similarly, carbon footprints of as low as 89 kg_{CO₂} per t_{DM} and as high as 337 kg_{CO₂} per t_{DM} are available in the literature, with values of up to 3883 kg_{CO₂} per t_{DM} upon adding direct and indirect land use changes. Assuming a capture potential of 1587–1716 kg_{CO₂}/ t_{DM} , BECCS could thus remove 1630 kg_{CO₂} per ton of dry solid biofuel, as well as emit 2300 kg_{CO₂} per ton of dry solid biofuel.

14.3.5 Converting Bioenergy into Negative Emissions: BECCS Carbon Efficiency

The conversion of exajoules of bioenergy into tons of CO₂ removed from the atmosphere is not straightforward. Depending on the mass loss along the biomass supply chain and the power plant CO₂ capture rate, all of the carbon fixed in the biomass by photosynthesis is not permanently stored in geological formations. Furthermore, BECCS life cycle CO₂ emissions can also significantly offset BECCS negative emissions. A useful metric to capture these sources of carbon leakage along the BECCS value chain is the carbon efficiency: out of the amount CO₂ captured biologically by the biomass, how much CO₂ is actually removed from the atmosphere? In work by Smith *et al.* (2015), ~~a case study on a switchgrass-based BECCS facility in the US evaluated the carbon efficiency of the system~~ at 47%: 2.11 t_{CO_2} needed to be biologically captured in order to securely remove 1 t_{CO_2} from the atmosphere.¹¹³ The carbon leakage along the BECCS value chain is represented in Figure 14.9.

In work by Fajardy and Mac Dowell (2017), it was found that this carbon efficiency was highly dependent on the choices made along the supply chain. Direct (LUC) and indirect (ILUC) in particular were found to have a significant impact on biomass sustainability. For example, a 90% capture BECCS facility in the UK using domestic switchgrass pellets from marginal land (no land use change) could reach a carbon efficiency of 65%, whereas the same facility operating with willow pellets from former grassland in the US would only be 3% efficient.⁸⁵ To put this into the context of bioenergy European policies, a sustainability criteria ensuring a minimum of 60% emissions reduction from the average European power carbon intensity was set for bioelectricity by the UK Bioenergy Strategy.¹¹⁴ This represents a maximum carbon intensity of 79 g_{CO₂} per MJ_e, which translates into a limit on the feedstock life cycle CO₂ of 28–36 g_{CO₂} per MJ_{HHV}, depending on the conversion efficiency of the power plant (30–45%_{HHV}). Considering a 90% capture rate, this maximum biomass carbon footprint would translate into a minimal carbon efficiency of 47–60%.

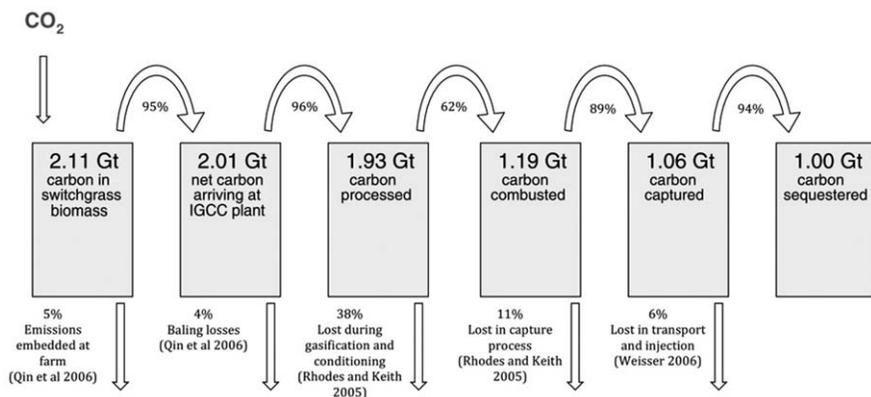


Figure 14.9 Carbon flux diagram of a BECCS system based on switchgrass-to-power. Because of carbon leakage along the BECCS value chain, out of 2.11 t_{CO₂} initially captured during biomass growth, 1 t_{CO₂} is stored in geological formations, and hence removed from the atmosphere. This is defined as the carbon efficiency of the system, which is 47% in this case.¹¹³ Reproduced from ref. 113, <https://doi.org/10.1007/s10584-012-0682-3>, under the terms of the CC BY 4.0 licence, <https://creativecommons.org/licenses/by/4.0/>.

It is worth noting that carbon efficiency also gives an indication as to the carbon negativity of the system: a system with a carbon efficiency of lower than 0% has a carbon positive balance. Knowing the system carbon efficiency, it is then more straightforward to convert the EJ of bioenergy into the t_{CO₂} captured. As seen in Section 14.2.5.1, 1 EJ of bioenergy could theoretically correspond to the capture of 72–95 Mt_{CO₂}. In the high efficiency scenario of 64%, this range decreases to 47–62 Mt_{CO₂}. The consequences of this are that more bioenergy is needed to meet the mitigation targets: removing 8.5–16.5 Gt_{CO₂} per year could require 140–350 EJ of bioenergy.

14.3.6 BECCS Cost

The economics of a BECCS project are strongly dependent on the conversion technology, the CO₂ capture cost, and the feedstock costs, but also on the conditions of the operation of the project: processing steps, co-firing proportion, capture efficiency. According to a review on BECCS economics by Kemper, the diversity in the feedstock–conversion–capture pathways results in very variable cost estimations for BECCS.²³

In an IPCC AR5 report, the cost of coal power generation + CCS was estimated at between 60 and 250 US \$ per t_{CO₂}.¹ The cost of the bio-power facility was evaluated to be within the same range, at between 70 and 250 \$ per t_{CO₂} by McLaren *et al.*¹¹⁵ However, this cost does not include the potential cost penalty related to biomass handling and efficiency loss. For example, the DRAX power plant communicated that 650 to 700 M£ was spent on converting three of the power plant 660 MW units to dedicated biomass.

This investment covered technical modifications to the boilers, biomass storage and feed systems, and two pelletization plants in the US producing 900 000 tons of pellets per year.¹¹⁶ Concerning the biomass-to-fuel pathway, Laude evaluated the cost of capturing CO₂ from the fermentation process at 56–86 € per t_{CO₂}, and 131–143 € per t_{CO₂} as the cost of capturing CO₂ from both the fermentation and the cogeneration processes. The process cost is also strongly dependent on the biomass feedstock cost, which itself depends on the biomass type and supply chain. In a study by Lu *et al.* for example, the cost of producing, collecting, pelletizing, transporting over 100 km and processing the biomass was evaluated at between 51 \$ per t_{DM} for wood chips and 105 \$ per t_{DM} for wood pellets.⁸² In Uslu *et al.*, the impact of the process was investigated, and similar ranges were found, with values of between 70 € per t_{DM} for pelletized biomass, to just under 110 € per t_{DM} for torrefied biomass.⁴¹ These evaluations for the BECCS costs are consistent with BECCS deployment models, which often consider that a carbon tax of 100£ per t_{CO₂} would be required to make a BECCS project economically viable.²³ Ranges from the literature review by Kemper *et al.* are presented in Figure 14.10, with values of between 10 and 300 € per t_{CO₂} via BECCS.²³

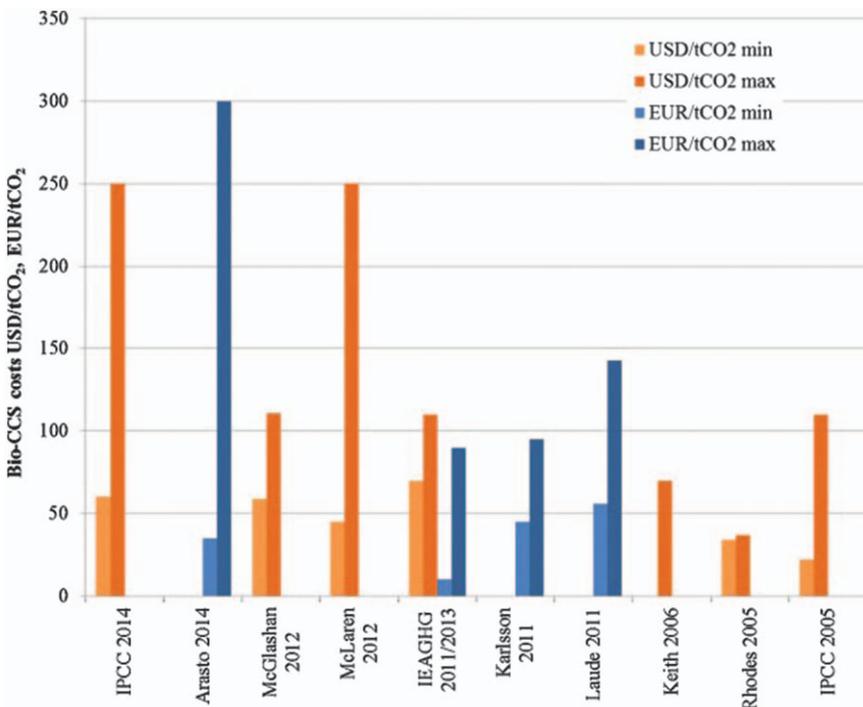


Figure 14.10 BECCS cost estimations across the literature. Values of as low as 10 € per t_{CO₂} and as high as 300 € per t_{CO₂} have been calculated for BECCS.²³ Reproduced from ref. 23 with permission from Elsevier, Copyright 2015.

Overall, BECCS is considered more competitive than other negative emission technologies such as DAC.¹¹⁷ However, these evaluations do not consider potential biomass sustainability and availability constraints. For example, Kriegler *et al.* evaluated that beyond a deployment of 12 Gt_{CO₂} per year, BECCS would no longer be economical because of the impact on the rising bioenergy demand on competition for land with other commodities.¹¹⁸ Furthermore, as biomass supply chain emissions become too great, and the carbon removal capacity of a BECCS power plant too low, the cost of removing CO₂ *via* BECCS can also increase significantly. In Mac Dowell *et al.*, the cost of removing CO₂ with a 45% efficient BECCS power plant increased from £640 per tCO₂ at a biomass carbon footprint of 270 kgCO₂ per MWh to £2500 per tCO₂ at 300 kgCO₂ per MWh, hence becoming less competitive than other negative emission technologies.¹¹⁹

14.4 Direct Air Capture (DAC)

450 ppm CO_{2eq} is widely regarded as the upper limit of atmospheric GHG concentration consistent with the 2-°C warming (above pre-industrial levels) limit set out at the 2015 Paris Agreement.¹ The IPCC Fifth Assessment Report outlined many least-cost transformative pathways to achieve this, the majority of which involved temporary overshoot of the concentration target and relied on the extensive deployment of negative emissions technologies (NETs) in the second half of the century to achieve climate stabilisation, specifically BECCS. The political momentum for the Paris Agreement is yet to translate into substantial mitigation action, with present day GHG concentrations at 407 ppm and overshoot seemingly increasingly inevitable.¹²⁰ At the time of writing, BECCS is the only technology considered as a source of crucial negative emissions in integrated assessment models (IAMs), and it has been shown to be greatly limited in its effectiveness by its associated biomass supply chain. The range of potential outcomes surrounding land-use and land-use change GHG emissions associated with large-scale biomass supply means that BECCS may take anywhere from 2 to over 50 years to provide *net* negative emissions, with the upper bound making the technology ineffective on the timescales in which significant mitigation needs to be achieved. The cost and financing of new technologies, opportunity cost of land-use (food security, *etc.*), and socio-institutional barriers to new technology acceptance all compound to the challenges faced.¹²¹ These have fuelled a debate on the eligibility of BECCS as a negative emissions technology (NET) and/or viable mitigation option. The poor prospects of BECCS and the other NETs discussed thus far forces the search for alternatives, one of which is the direct removal of CO₂ from the atmosphere, known as direct air capture (DAC).

14.4.1 Motivations for DAC

- **Delayed Mitigation Pathway:** Several studies have shown that the relationship between cumulative emissions and peak warming is

remarkably insensitive to the emission pathway (timing of emissions or peak emission rate), so policies designed to achieve cumulative emissions targets are likely to be subject to less uncertainty than those with emission-rate or concentration targets.¹²² This suggests that climate change stabilisation can be achieved by reducing historic emissions. This has garnered interest in the possibility of bulk removal of CO₂ from the atmosphere in later decades, allowing for faster declines in GHG concentrations in the long-term and thus facilitating higher near-term emissions, effectively expanding the potential for overshoot of the concentration target. DAC is seen as a valuable component of the mitigation toolbox because of its potential to offer this.

- **Decentralised Emissions Mitigation:** The transport sector, responsible for 14% of global GHG emissions, has proved a difficult sector to decarbonise using the current portfolio of mitigation technologies because of the mobile nature of the emitters.¹ The electrification of transport and/or increased reliance on biofuels are the most touted solutions to this, however, the former is considered currently unfeasible for aviation and marine transport, while the latter is hindered by issues associated with large-scale biomass supply and use, as discussed in Section 14.2.5. Aside from transport, billions of small GHG sources exist for which capture technologies such as CCS are not practical or financially viable. Large volumes of CO₂ removed from the atmosphere through extensive DAC deployment can compensate for decentralised emissions, allowing for continued use of existing transport infrastructure till carbon-neutral alternatives become competitive with current technology. More generally, DAC allows for a decoupling of abatement from emission sources, thereby adding much-needed flexibility to today's proposed mitigation strategies.
- **Lack of CO₂ Transport Infrastructure:** A major drawback to the large-scale deployment of CCS technology for climate change mitigation has been the absence of CO₂ pipeline transport infrastructure to suitable geological storage sites, mainly saline aquifers. The non-uniform distribution of these sites globally raises a myriad of possible physical, legal, socio-political and financial issues. CO₂ pipeline networks need to be extensive, as large point source emitters are often located great distances away from suitable aquifer sites. Apart from the increased likelihood of encountering geophysical barriers, extensive infrastructure may raise environmental concerns, face public scrutiny, and will be exposed to vandalism that may hike maintenance costs. Should the need to cross national borders to access sites arise, obtaining permits may be a complex legal process as disputes may arise if the entity that owns the storage site is different from the entity that is storing the carbon – who is to be allocated the credit for mitigation under the current carbon accounting framework is ambiguous. DAC potentially allows for the capture facility to be co-located with the storage site, eliminating the need for resource-intensive transport infrastructure development.

- **Synthetic Fuels:** Liquid hydrocarbon fuels, derived from fossil fuels, serve as the energy source for most of our daily domestic and economic activity. Efforts to avoid the devastating effects of climate change necessitate a transition to a low-carbon economy, and a move away from fossil fuels will play the central role in achieving this. A range of carbon-based fuels and chemicals (*e.g.* methanol, synthetic natural gas, dimethyl ether, *etc.*) can be synthesised using CO₂ as a starting raw material.¹²³ CO₂ isolated by a DAC plant can serve as a feedstock to the chemicals industry, thereby reducing the demand for fossil fuels and the need for their extraction.
- **Compensation for CO₂ Leakage from Storage Sites:** Leakage from well-managed geological reservoirs of CO₂ are considered unlikely¹²⁴ and it has been shown that leakage of up to a few percent a year does not affect the value of CCS.¹²⁵ Despite the small possibility of occurrence, leaks from storage sites are often cited as barriers to geological carbon sequestration development because of the complex accounting and legal issues that may arise. Leakages will reduce the avoided emissions and may become significant if continuous over time. DAC can serve as insurance to compensate for any leaks from sites.

14.4.2 CO₂ Separation Technologies

Air, by volume, is composed of 78% N₂, 21% O₂, 0.04% CO₂ and a range of other trace elements and water vapour. Removing CO₂ from air, therefore, is just a gas separation process, which is an established unit operation used in many industries to isolate desired products or for pollution control. The diluteness of CO₂ in air means its extraction is very energy intensive. Additionally, large volumes of air must be treated for a significant amount for CO₂ to be captured; this translates into larger equipment requirements and hence capital costs. Three main methods have been proposed for the separation of CO₂ from gaseous mixtures:

14.4.2.1 Separation with Solvents/Sorbents

Solid adsorbents such as zeolites, silica materials, metal-organic frameworks and activated carbons can be used to separate CO₂ from gas mixtures through pressure (PSA) or temperature swing adsorption (TSA).¹²⁶ PSA relies on the increased affinity of some gas molecules to solid surfaces at elevated pressures. Gas mixtures are made to flow through a packed adsorbent material at high pressure until the concentration of the desired gas approaches equilibrium, then the bed is regenerated by reducing the pressure. TSA works using a similar principle but cycles temperature instead of pressure. PSA and TSA are commercially practiced methods of gas separation, often used in the removal of CO₂ from natural gas or hydrogen, however adsorption is not yet considered attractive for large-scale separation of CO₂ from flue gas because the capacity and CO₂ selectivity of available

adsorbents is low.¹²⁷ However, its prospects may be improved if used in conjunction with another capture technology. The start-up *Climeworks* is currently developing DAC technology that employs an amine-functionalised sorbent for CO₂ capture.¹²⁸

Since the 1930s, amine scrubbing has been employed by the oil and gas, and chemical industries for the removal of acid gases, mainly hydrogen sulphide (H₂S) and CO₂, from natural gas and hydrogen. More recently, amines have been used in carbon capture and storage (CCS) technology to remove CO₂ from the flue gas effluents of power plants, often for the purpose of mitigation through geological sequestration or injection into oil fields for enhanced oil recovery. The lower concentrations of CO₂ in air compared to flue gases necessitate an increased energy cost of capture and render amine solvents unsuitable. Strong bases – potassium hydroxide (KOH) and sodium hydroxide (NaOH) – have been found to be better suited to the task of DAC. This is discussed in detail in Section 14.2.6.3.

14.4.2.2 Separation through Membranes

Membranes are selective barriers that allow different gas particles to move through them at varying rates. Membrane separation has the advantages of compactness, lower capital costs and energy requirements, however often membranes cannot achieve high degrees of separation, so multiple stages and/or recycling of one of the streams is necessary. This leads to increased complexity, energy consumption and costs.¹²⁷ A variety of polymeric and ceramic synthetic membranes have been identified as suitable for CO₂ separation due to high permeability to, and selectivity for CO₂. When investigated for post-combustion capture applications, the low CO₂ concentration and pressure of flue gas was found to greatly limit their effectiveness.¹²⁹ Ambient air conditions are in stark contrast with those desired by membranes, so it is unsurprising that no membrane-based DAC technology has been proposed in the literature.

14.4.2.3 Separation by Cryogenic Distillation

Cryogenic distillation separates the gaseous components of a mixture by cooling and condensation. Liquid CO₂ is obtained allowing for relative ease of handling and transport and eliminating the need for further compression. However the large energy requirements for refrigeration result in large capital and operation costs. Furthermore, cryogenic distillation is only suitable for streams with already high CO₂ concentrations (typically >90%), rendering the method unsuitable for CO₂ removal from air (Figure 14.11).

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14.4.3 Thermodynamics of Capturing CO₂ from Air

The energy cost of capture is a significant determinant of the feasibility of a CO₂ capture process, hence it is important to determine the lower bound

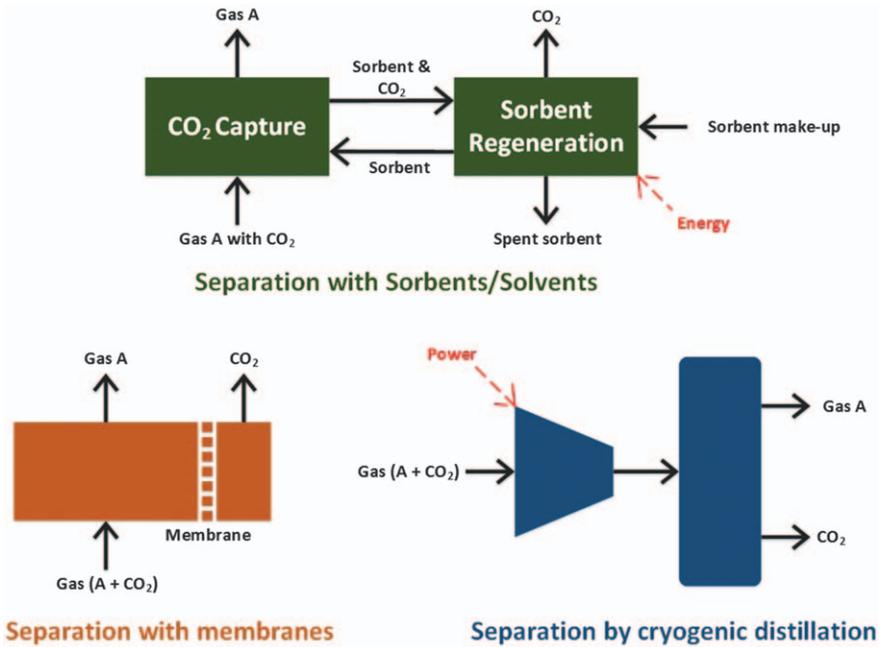


Figure 14.11 Basic methods for the separation of gases.

energy requirements for the capture process. The minimum work of separation has been derived from first principles below.

14.4.3.1 Minimum Work of Separation

Considering the open system illustrated in Figure 14.12, where a binary mixture of inert gaseous species 1 and 2, denoted by “12”, at temperature T_o and pressure p_o is separated into its pure components, also at T_o and p_o . The enthalpy (measured in Joules, J) and entropy (measured in JK^{-1}) of each stream is denoted by H_i and S_i , respectively, where $i = 1, 2$. The system has a work input, W , and can exchange heat, Q , with its surroundings at temperature, T . The material, energy and entropy balances are provided below can then be defined as below. Note that the dot above each variable reflects the rate at which that variable changes over time.

$$\text{Material balance: } \frac{dN_{i,\text{sys}}}{dt} = \dot{N}_{i,\text{in}} - \dot{N}_{i,\text{out}} \tag{14.1}$$

$$\text{Energy balance: } \frac{dE}{dt} = -\dot{Q}_{\text{out}} + \dot{W}_{\text{in}} + \dot{H}_{12} - \dot{H}_1 - \dot{H}_2 \tag{14.2}$$

$$\text{Entropy balance: } \frac{dS}{dt} = -\frac{\dot{Q}_{\text{out}}}{T_o} + \dot{W}_{\text{in}} + \dot{S}_{12} - \dot{S}_1 - \dot{S}_2 + \dot{S}_{\text{irr}} \tag{14.3}$$

where \dot{N} denotes molar flow rates and \dot{S}_{irr} is the increase in entropy associated with the irreversibilities in the system. ~~This term allows us to write as a balance even though entropy is not conserved.~~ Making a steady-state assumption, and eliminating the heat rate, \dot{Q} between (14.2) and (14.3) yields an expression for the work, \dot{W} , needed for separation:

$$\dot{W}_{\text{in}} = [(\dot{H}_1 + \dot{H}_2) - \dot{H}_{12}] - T_o[(\dot{S}_1 + \dot{S}_2) - \dot{S}_{12}] + T_o\dot{S}_{\text{irr}} \quad (14.4)$$

$$\dot{W}_{\text{in}} = -\dot{N}_{12}(\Delta h_{\text{mix}} - T_o\Delta s_{\text{mix}}) + T_o\dot{S}_{\text{irr}} \quad (14.5)$$

where $\Delta h_{\text{mix}} = (h_{12} - x_1h_1 - x_2h_2)$ and $\Delta s_{\text{mix}} = (s_{12} - x_1s_1 - x_2s_2)$, with x_1 and x_2 representing the molar fractions of 1 and 2 in the mixtures (N_1/N_{12} and N_2/N_{12} , respectively). The lowercase variables shown represent the specific molar quantities of the corresponding uppercase variables. From the second law of thermodynamics, we know that the bracketed term in (14.5) is the Gibbs Free Energy of Mixing, Δg_{mix} , at the so-called restricted dead state (T_o, p_o) at which the system and its environment are at mechanical, thermal and chemical equilibrium. Given that the enthalpy of mixing of an ideal mixture is zero, and that the minimum change in work potential occurs when a system undergoes a reversible isothermal, isobaric change, Δh_{mix} and \dot{S}_{irr} can be assumed to be zero.¹³⁰ The expression for the minimum rate of work per mole can then be obtained as:

$$w_{\text{min}} = \frac{\dot{W}_{\text{in}}}{N_{12}} = T_o\Delta s_{\text{mix}} \quad (14.6)$$

The entropy for mixing, Δs_{mix} , for non-interacting particles, as in the above system, can be calculated from a statistical interpretation of entropy and Boltzmann's entropy equation¹³¹ to obtain eqn (14.7).

$$w_{\text{min}} = -RT_o \sum_{i=1}^n x_i \ln x_i \quad (14.7)$$

Note that this is the general result for a mixture containing n constituents, where x_i is the mole fraction of the i th constituent and R is the universal gas constant, $8.314 \text{ J}(\text{mol}^{-1} \text{K}^{-1})$. Air consists of multiple components so to determine the minimum work needed for a DAC (extracting CO_2 only from the air, as opposed to a separation of all components into pure streams), we start by writing the extensive form of eqn (14.7), and then subtracting from it the work of separation for a mixture at the same mole fractions but with one less mole of CO_2 . One physical interpretation of this situation is that the mixture is very large and extracting a mole of CO_2 does not significantly change the molar concentrations of the original mixture.¹³²

$$w_{\text{min,CO}_2} = RT_o \ln \left(\frac{1}{x_{\text{CO}_2}} \right) \quad (14.8)$$

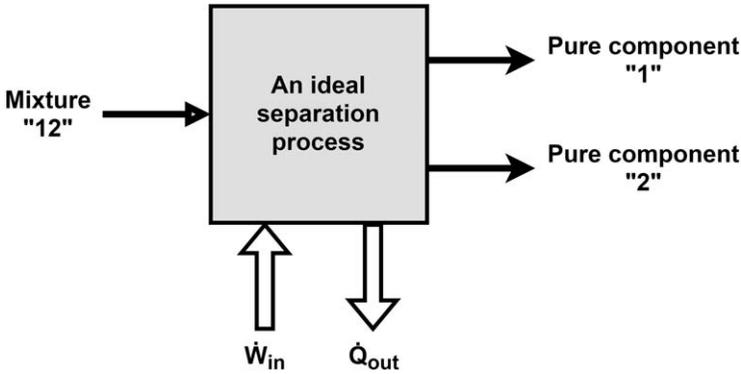


Figure 14.12 An ideal separation process separating a binary mixture, 12, of two inert (non-interacting) gaseous species into pure streams of each component.

The resulting expression given by (14.8), where x_{CO_2} is the mole fraction of CO_2 in air, *i.e.* 0.04, tells us that the thermodynamic minimum energy requirement for DAC is $\sim 19.1 \text{ kJ mol}^{-1}$ or 0.433 GJ per tonne of CO_2 . The minimum work required for CO_2 capture from other sources is also highlighted in Figure 14.13.

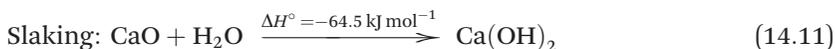
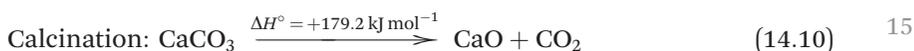
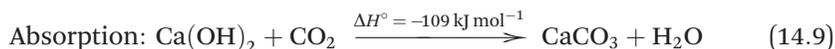
14.4.4 Current Technology

DAC is a relatively novel technology that has been brought to the forefront of climate change mitigation discussions by the increasing consensus on the importance of negative emissions in meeting the targets set out in the Paris Agreement. The DAC processes proposed in literature are mainly absorption-based using hydroxides as the capture solution, with the exception of Lackner’s Artificial Trees and Infinittree’s Carbon Sink, which employ moisture/humidity swing sorbents; both are discussed in detail below.¹³³ A number of quantitative and semi-quantitative analyses based on specific process designs proposed in literature or extrapolations from second law efficiencies of similar processes have yielded a wide range of energy costs – from 0.9–22.7 GJ per tCO_2 – for DAC.^{128,130,134–141} None of these have been realised on a commercial scale, however a few pilot projects are operational worldwide. The existing commercial ventures pursuing DAC technology are provided in Table 14.9. 

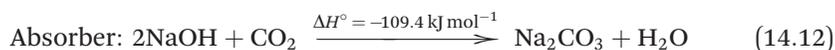
14.4.4.1 Direct Air Capture using Hydroxide Sorbents

The ultra-dilute nature of CO_2 in air necessitates the use of sorbents with strong CO_2 -binding affinities for capture. The alkali and alkali-earth hydroxides have been found to have high binding energies with CO_2 , with which they react to form carbonates, and this has led to their consideration as suitable chemical sorbents for DAC. Lackner in 1999 suggested the use of

calcium hydroxide ($\text{Ca}(\text{OH})_2$).¹⁴² $\text{Ca}(\text{OH})_2$ reacts with the CO_2 in air to form calcium carbonate (CaCO_3), which is then thermally decomposed to form quicklime (CaO) and CO_2 (14.10), a process known as calcination. The original $\text{Ca}(\text{OH})_2$ sorbent is then regenerated by hydrating the produced quicklime, a process known as slaking. The reactions are described by eqn (14.9)–(14.11). Calcination occurs at high temperatures of $>700^\circ\text{C}$ and requires 9 times the minimum work of separation of CO_2 from air (see Section 14.2.6.3), resulting in a highly energy intensive, and hence costly, regeneration process. Additionally, the low solubility of $\text{Ca}(\text{OH})_2$ in water limits the hydroxide concentration available to capture the CO_2 .¹⁴³ These drawbacks have driven the search for alternative sorbents.



Sodium hydroxide (NaOH) was proposed as this alternative. Its prospects as a capture solution for DAC are bolstered by the fact that it is already employed by the paper and pulp industry for the extraction of cellulose from wood *via* the Kraft process, which is partly based on similar sodium solution use and recycling. As shown by the similar ΔH° values, NaOH allows for similarly strong CO_2 -binding compared to $\text{Ca}(\text{OH})_2$ but has the added advantage of being highly soluble in water. Figure 14.14 shows a schematic of the NaOH -based CO_2 capture that was proposed by Baciocchi *et al.*,¹³⁵ this is the reference DAC process often cited in literature because of its use of existing technology and provision of material and energy balances that can be used for further cost analysis.



Here, the CO_2 in air reacts with NaOH solution, which stores it as sodium carbonate (Na_2CO_3). This reaction is exothermic at $109.4 \text{ kJ mol}^{-1}$, or 105 kJ mol^{-1} including solvation energy. Unlike CaCO_3 , Na_2CO_3 is highly soluble in water, so scaling within the absorber is avoided.¹³⁴ Traditional absorbers for gas separation employ a packed tower design, where the solution is contacted countercurrently with gas blown through the tower/column. The diluteness of CO_2 in air, and hence the low flow rate, requires a tower with a large cross-sectional area – perhaps hundreds of meters in diameter – through which large volumes of air can flow to allow for significant capture. A contactor of these dimensions would be very different from conventional packed towers and might look like a trickle-bed filter used in wastewater treatment plants: a wide cylindrical basin, drafted from underneath, with a rotating distributor arm required, which has further

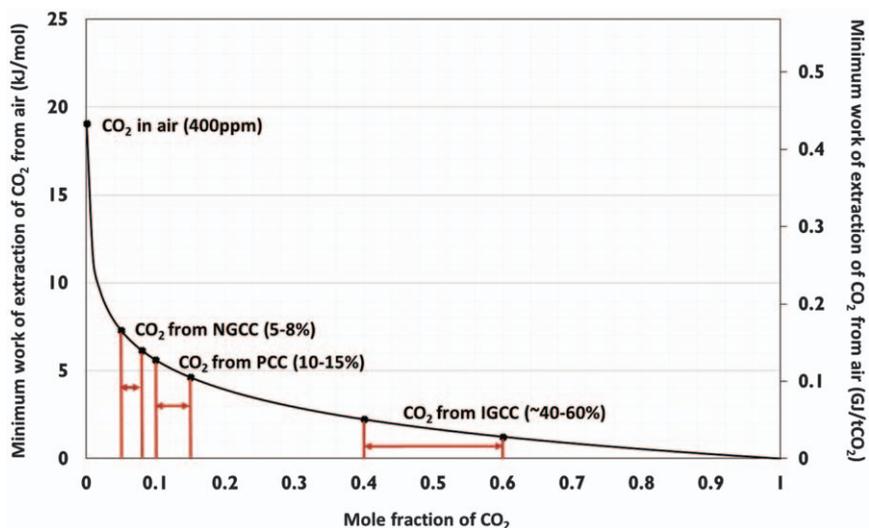


Figure 14.13 Thermodynamic minimum work of extraction of CO₂ (in kJ mol⁻¹ and GJ per tCO₂) at 20 °C as a function of the initial concentration in the gas stream from various sources. NGCC, PCC and IGCC represent the flue gases from a natural gas combined cycle power plant, a pulverised coal combustion power plant, and an integrated gasification combined cycle power plant, respectively.

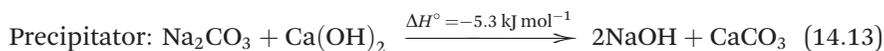
design implications and makes cost estimation difficult.¹¹⁷ To overcome this complication, Keith *et al.*¹¹⁷ instead proposed the use of a structure (110 m in height and 120 m in diameter) similar to a cooling tower in which NaOH solution is sprayed and fans are used to force air countercurrently; this more simplistic design would not incur the costs of a packed tower, and is sufficient for the assumed 50% capture efficiency and allows for maximising the contact area while maintaining a low pressure drop, however, creating the spray has its own associated energy losses.¹⁴³ Carbon Engineering (CE), an independent air capture company to which Keith is affiliated (see Table 14.1), have developed an intermittently-wetted air-contactor with crossflow slab geometry for their contactor, which is based on a similar induced draft cooling tower design.¹⁴⁴ In their 2011 report on DAC, however, the American Physical Society carried out a rigorous design analysis based on the process proposed by Baciocchi *et al.* that has a similar capture efficiency and found an array of 12 m in diameter towers to be the preferred choice for capturing 1 000 000 tonnes of CO₂ annually. Responses to the report by Holmes and Keith of CE have highlighted technical discrepancies with the APS' assumptions on pressure drop and capture efficiency, which are critical for cost estimation, and the corresponding industrial performance estimates for the hardware specified. Furthermore, the use of an array is not only costly (compared to a large integrated design as used by existing commercial air capture plants) but also suboptimal for feeding in

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Table 14.9 Companies operating or pursuing commercial-scale DAC technology worldwide.

Company	Type of Technology	Capture capacity (tCO ₂ per year)	Energy source	Final CO ₂ purity
Carbon Engineering ¹³⁹	Chemical absorption using KOH	365	Natural gas (6.7 GJ per tCO ₂)	
Climeworks ¹²⁸	Adsorption using porous granulates modified with amines	49 per collector	Low-grade/waste heat (~100 °C)	“Concentrated CO ₂ gas”
Global Thermostat ¹⁴⁰	Adsorption using dry amine-based chemical “sorbents” bonded to porous honeycomb ceramic “monoliths”	~40 000–50 000 per module	Low-temperature steam from residual heat Electricity	98%
Infinitree ¹⁴¹	Humidity swing absorption using ion exchange sorbent material		Plug-and-go systems need 120 V outlet only	Levels ideal for greenhouse applications

atmospheric air, as rejected air from the tower may be taken in by another that will significantly affect performance. It is important to note that KOH is a viable alternative to NaOH and can achieve similar capture performance, but as KOH is more expensive than NaOH, there are limited examples in the literature.¹⁴³ A study found that using a 2 m packed column, a 1.5 M KOH solution could absorb up to 45% of the CO₂ in ambient air. Increasing the concentration of the KOH to 2 M improved the capture efficiency to ~70%.¹⁴⁵ CO₂ is recovered from the potassium carbonate produced in the absorption column by acidifying the spent solution with sulphuric acid (or KHSO₄). The produced K₂SO₄ solution is then treated in an electro-dialytical unit with bipolar and cation exchange membranes, wherein the KOH scrubbing solution and the sulphuric acid are regenerated.



The high solubility of Na₂CO₃ in water means evaporating large quantities of water is necessary to precipitate it out of solution. This would be very energy-intensive, so instead it is converted to less-soluble CaCO₃ by reacting with Ca(OH)₂; where the NaOH capture solution is also recovered.

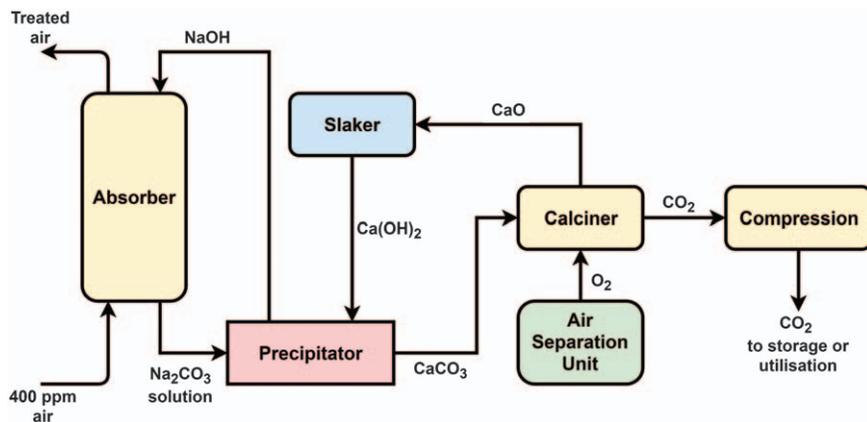
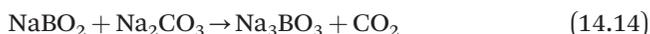


Figure 14.14 Schematic for the DAC plant adapted from that shown in the APS report, based on the air capture process proposed by Baciocchi *et al.*¹³⁵

Alternative methods for recovering NaOH, also known as causticization, have been proposed for the Kraft process used in the paper and pulp industry. Most notably, the use of sodium metaborate (NaBO_2) has generated interest. A trisodium borate salt is produced when NaBO_2 is reacted with Na_2CO_3 , this can then be dissolved in water to recover the initial metaborate and obtain NaOH, according to (14.14) and (14.15) below. The temperature in the recovery boiler ranges from $850\text{ }^\circ\text{C}$ to as high as $1250\text{ }^\circ\text{C}$, so the energy needs have limited the adoption of this method.¹⁴⁶



The reaction in the calciner is the same as in eqn (14.10). Calcination is a common process in the cement industry; it is carried out at temperatures of around $1500\text{ }^\circ\text{C}$ when clinker is desired, however, a temperature of $800\text{--}900\text{ }^\circ\text{C}$ is sufficient for the good quality quicklime (CaO) needed here.¹³⁵ Despite this, the process is endothermic at 179.4 kJ mol^{-1} and requires significant high-grade heat input, hence is the most energy-intensive unit in the DAC plant. The heat is provided by oxy-combustion of a suitable fuel, usually coal or natural gas, to obtain pure CO_2 (which is captured along with the CO_2 removed from air) and to avoid the need for a further separation step to remove nitrogen. The quicklime produced is subsequently hydrated, as in eqn (14.11), to obtain the Ca(OH)_2 required to precipitate the Na_2CO_3 out of solution. Ca(OH)_2 is reacted with K_2CO_3 used to regenerate the capture solution and the cycle is repeated.

The pure CO_2 obtained may be sent for pipeline transport then storage in geological reservoirs or use in CO_2 utilisation processes, but it must first be compressed/liquefied. At ambient temperature (taken as $20\text{ }^\circ\text{C}$), CO_2

liquefies at 57 bar. To determine the thermodynamic minimum energy required for its compression (*i.e.* for a reversible isothermal process), the ideal gas approximation is inappropriate, as CO₂ is near its critical point (31.1 °C and 73.8 bar). The work required for any isothermal compression is again given by:

$$\dot{W}_{\min}^{\text{compression}} = \Delta H - T\Delta S \quad (14.16)$$

where ΔH is the enthalpy (in J) and ΔS is the entropy change (in J K⁻¹). ΔH and ΔS were found to be -246.1 kJ per kgCO₂ and -1.1359 kJ per kgCO₂-K at $T = 293\text{K}$, respectively, using the NIST Reference Fluid Thermodynamic and Transport Properties Database.¹⁴⁷ Using these values, the minimum work for compression is 203.9 kJ per kgCO₂. The effects of efficiency losses on this number will be discussed in Section 14.3.2.

Several studies have cited that significant evaporative water losses will be experienced after ambient air is contacted with the NaOH solution, as a result of the difference between the moisture content of the ambient air entering the absorber and that of the air leaving the contactor.^{134,137,143,148}

Zeman reports this to be as high as 90 g of H₂O per gram of CO₂ captured. Let us consider a DAC process, as illustrated in Figure 14.14, with 50% capture efficiency taking in ambient air (400 ppm CO₂) and rejecting treated air with a CO₂ concentration of 200 ppm, all at 20 °C. The moisture content of air can be described by its relative humidity (*RH*), which is the ratio of the partial vapour pressure of water, p_w , to its saturation vapour pressure at that given temperature, p_v . At 20 °C, p_v is 2339.3 Pa.¹⁴⁹ While the *RH* of ambient air can be determined relatively easily, the following assumptions have to be made about the treated air leaving the absorber to determine its *RH*, denoted by *S*. They are: (i) treated air is at the same temperature as the incoming NaOH solution, (ii) the water vapour content of treated air is in thermodynamic equilibrium with the NaOH solution, and (iii) the presence of CO₂, carbonate and bicarbonate in the NaOH solution can be neglected.¹³⁴ *S*, which can also be defined as the degree of saturation of the air, is dependent on the conditions in the absorber and its operation. Figure 14.15 shows *S* as a function of the molar concentration of NaOH under ambient conditions. For a 2 M solution, as in the work by Baciocchi *et al.*, *S* is found to be 91%.

The change in p_w , denoted as Δp_w in eqn (14.17), can be used to determine the water loss in the absorber. The annual average relative humidity in London, UK, is 77%. Considering a DAC plant sited here and the determined degree of saturation, Δp_w is found to be 327.5 Pa.

$$\Delta p_w = p_{w,\text{out}} - p_{w,\text{in}} = p_v(S - RH_{\text{in}}) \quad (14.17)$$

A material balance yields the specific water loss as the ratio of the partial pressure differentials, as given in eqn (14.18). The difference in concentration of the inlet and outlet CO₂ streams is 200 pm, which is equivalent to a Δp_{CO_2} of 20.3 Pa. Therefore, a DAC plant sited in London would experience

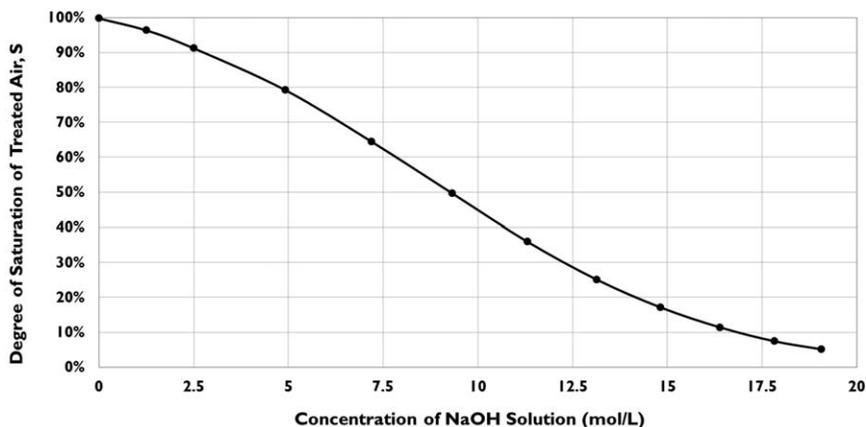


Figure 14.15 Variation in the degree of saturation, S , (equivalent to relative humidity) of air in equilibrium with sodium hydroxide (NaOH) solution with the concentration of NaOH under ambient conditions of 1 atm and 20 °C. Data from the APS Report.¹³⁴ S has been shown to be temperature independent in the interval considered.

water loss at a rate of about 7 tonnes H₂O per tonne of CO₂ captured, which would add to the already expensive costs of the process. A more arid location for the plant increases this loss.

$$\frac{\Delta p_w}{\Delta p_{CO_2}} = 16 \text{ mol H}_2\text{O per mol CO}_2 \text{ or } 6.6 \text{ g H}_2\text{O per g CO}_2 \quad (14.18)$$

14.4.4.2 Direct Air Capture via Moisture/Humidity Swing Sorption

As seen in the previous section, the energy required for the regeneration of chemical sorbents for DAC poses a great limitation on its commercial viability. Lackner *et al.* have developed a novel anionic exchange resin that captures CO₂ when dry under ambient conditions and releases it when wet (or exposed to high humidity conditions).¹³³ Lackner suggests that liquid water could be used as a low-cost source of energy to drive efficient moisture-swing sorption but a water-swing would require clean water and may incur significant water loss. Thermodynamic analysis has revealed that moisture swing behaviour is compatible with hydration energies changing with the activity of water on the resin surfaces, which is in turn set by the humidity.¹⁵⁰ The CO₂ produced is at low partial pressures, hence not suitable for geological sequestration, but may serve other CO₂ utilisation purposes, *e.g.* improving yield in greenhouses. Sufficient information is however not available to determine the technical and commercial viability of the technology.

14.4.4.3 Direct Air Capture using Physisorbents

Several commercial DAC ventures being pursued employ either hydroxides or solid organo-amine based chemisorbents, wherein amine functional groups are either physically or chemically anchored to the surface of cellulose, porous silicas or porous polymer networks.¹⁵¹ The regeneration of such sorbents is often energy-intensive, taking place at elevated temperatures (> 100 °C). As physical sorbents (physisorbents) bind CO_2 *via* weak intermolecular forces and do so reversibly, they often require much less energy to recycle. This has drawn attention to the possibility of using physisorbents to improve the commercial viability of CO_2 removal from the atmosphere by decreasing energy costs and, hence, process costs. Metal-organic frameworks (MOFs) have been extensively studied for the separation of gases, including CO_2 , *via* pressure and temperature swing adsorption, however, their high costs, limited stability and moderate selectivity in the presence of competing gases (such as water vapour) have deterred further interest in the field.¹⁵² Other metal-organic materials (MOMs) are however still showing promise. One of such is a class of porous solids known as hybrid ultramicroporous materials (HUMs), which have been found to exhibit exceptional selectivity towards CO_2 over non-polar gases like N_2 , because of their ability to combine two features that can enhance sorbent-sorbate interactions: the presence of inorganic anions that offer strong electrostatics and ultramicropores that offer a good fit for important sorbates such as CO_2 .^{153,154} Scott *et al.* have engineered HUMs to synthesise a subclass of physisorbents known as primitive cubic (pcu) networks that interselectively adsorb CO_2 from gas mixtures that simulate flue gas. Their performance in this context is among the highest for physisorbents measured to date and these materials are readily regenerated at 50 °C.¹⁵⁵ However, reviewed literature indicates that research direction in this field is to assess the suitability of the discussed materials for post-combustion capture of CO_2 from flue gases. The poor performance of MOMs at low CO_2 partial pressures, characterised by low CO_2 selectivity and uptake, has left chemisorbents as the focus of discussion and research for the DAC community.¹⁴³

14.4.5 Cost of Direct Air Capture

Cost is often the most important determinant of the adoption of a technology at commercial scale. To assess the viability of DAC as a mitigation option, its cost performance must be assessed relative to that of other technologies in the mitigation portfolio. It is however important to note that clean energy technologies (*i.e.* renewables) do not provide the same service as DAC, hence direct comparisons are inappropriate. This is somewhat reflected in the continued use of CCS costs as the benchmark for comparison. Parallels are drawn between them as both technologies are capital and energy intensive, and require consideration of CO_2 capture, transport, and

geological storage and monitoring. Furthermore, the commercial success of both is often cited as dependent on the implementation of policy incentives such as a significant carbon price, and/or increased demand for CO₂ for utilisation purposes, which add value to the otherwise waste emissions. Although widespread deployment has yet to be achieved, several megatonne-scale CCS facilities are operational worldwide. CCS technology costs are currently estimated to be in the range of \$48–114 per tCO₂ avoided (from coal plants) and \$74–114 (from natural gas plants).¹⁵⁶ Unlike CCS, DAC is still in the early stages of research and development, with relatively few pilot projects capturing CO₂ at the tonne scale. The lack of established process designs and engineering, compounded with the lack of accessible information from commercial entities developing DAC technology have led to a wide range of cost estimates in the literature.

AQ-18

Table 14.2 shows some of these estimates, many of which are based on proposed designs in literature that are yet to be tested, or extrapolations from minimum work and predicted second law efficiency calculations (Table 14.10).

AQ-19

The significantly varying costs of \$25–1000 have led to differing opinions on the prospects of DAC as a mitigation option, with the upper bound costs calling its commercial viability into question. With the exception of estimates by proponents of the technology, initial cost estimates are seen to be consistently higher than those for BECCS, for which preliminary analysis has produced ballpark costs of \$60–250 per tCO₂.¹ Taking these into account, even if technically possible at large scales, DAC proves to be the most cost-intensive source of negative emissions and further research and development is essential to drive down costs before it can be competitive with existing mitigation options. Additionally, research into DAC is important because: (i) it could provide an upper bound cost of mitigation should the need arise for “sudden” large-scale negative emissions to avoid catastrophic climate change; (ii) should the challenges to CCS deployment (*e.g.* investment, lack of transport infrastructure, *etc.*) and uncertainties associated with large-scale biomass supply and use persist and deter adoption of BECCS, DAC allows for another source of crucial negative emissions (iii) the possibility for negative emissions in general adds great flexibility to the mitigation toolbox.

14.4.6 Scalability and Roll-out Potential

Some CO₂ capture technologies have external geophysical and sociopolitical limitations to their large scale deployment because of uncertainties around their natural resource implications and policy structures needed to accommodate them, amongst other factors. DAC has been shown to have a relatively minimal land and water footprint, and does not require development of extensive transport infrastructure for widespread deployment as facilities can be sited at storage sites.^{134,157,158} However, issues around leakage and monitoring of geological reservoirs exist as with CCS technology.

Table 14.10 Cost estimates for DAC in the literature. The second law efficiencies have been calculated for the air capture units only relative to the $\sim 20 \text{ kJ mol}^{-1}$ minimum work calculated in Section 14.2.6.2, and excluding the energy required for compression from ambient to pipeline pressure. Energy required to compress a 5 kPa CO_2 stream to atmospheric pressure has been added to the minimum work of separation suggested by Lackner (2009), to ensure consistency with the remaining values.

Source	Year	DAC Process	Second law efficiency of capture (%)	Cost of capture (\$ per tCO ₂)
Zeman ¹⁶⁹	2003	Capture using NaOH solution with Ca-based regeneration cycle		~ 25–75
Keith <i>et al.</i> ¹¹⁷	2006	Capture using NaOH solution with Ca-based regeneration cycle	7.9 ¹³⁰	~ 136
Stolaroff <i>et al.</i> ¹³⁷	2008	Capture using NaOH spray	4.9–10.1	96 (excluding solution recovery and CO ₂ sequestration) 53–127 under alternate operating parameters and assumptions
Lackner ¹³⁸	2009	Moisture swing absorption using anionic exchange resin	49.8	200 (initial prototypes) 30 (long-term)
Nikulshyna ¹⁷⁰	2009	Solar-driven carbonation using Ca(OH) ₂ and Ca-based regeneration cycle	2.4 ¹³⁰	162–200 (solar calcination plant only)
APS ¹³⁴	2011	Capture using NaOH solution with Ca-based regeneration cycle	3.8	610–780 (lower bound is optimistic estimate)
House <i>et al.</i> ¹³⁰	2011	Unspecified – based on predicted second law efficiency	< 5	~ 1000
Simon <i>et al.</i> ¹⁵⁸	2011	Unspecified – “generic” air capture technology characterised only by its energy use, capture footprint, and water us	10	~ 293–863
Holmes and Keith ¹⁷¹	2012	Capture using NaOH solution with Ca-based regeneration cycle	Insufficient information	60 (base case, contactor only) 43–95
Kulkarni and Sholl ¹⁷²	2012	Temperature swing adsorption using an amino-modified silica adsorbent and a structured monolithic contactor unit, regeneration with steam.	6.4	95 (excluding capital costs)
Mazzotti <i>et al.</i> ¹⁷³	2013	Optimisation of process in APS report, which is a capture using NaOH solution and Ca-based regeneration	Insufficient information	518–568
Zeman ¹⁷⁴	2014	Optimisation of process in APS report, which is a capture using NaOH solution and Ca-based regeneration	Insufficient information	309

The commercial ventures seeking to employ DAC technology have mostly developed small, modular systems allowing for great flexibility in scaling up the process and likelihood to benefit from mass production and the economies of scale. However, the energy-intensive nature of this technology would require dedicated additional capacity as surplus renewable energy (from curtailment or other sources) is unlikely to be sufficient. While technical feasibility is not challenged, cost remains a key limitation to the adoption of DAC. Future research and development will reveal the opportunities for cost reductions, and if the technology can be made competitive with existing mitigation options.

14.5 Economic and Energetic Analysis of NETs

The early stages of research and development of NETs (except for afforestation/reforestation) indicates the need for significant investment in R&D before commercial viability is achieved. However, while cost is often the decisive metric in assessing commercial potential, the natural resource implications of developing a technology may hinder its adoption. In assessing the potential of NETs for climate change mitigation, a holistic approach must be taken to obtain the least-cost and least environmentally-impacting solutions. Table 14.11 highlights the investment needs and key resource implications of large-scale BECCS and DAC adoption.

To contextualise the resource needs of BECCS and DAC, 1 Mt_{CO₂} per year has been used as a benchmark capture target in Figure 14.16. Studies have revealed that BECCS deployment will have significant land and water footprint, however there is considerable disparity in the estimates. This is mainly due to uncertainties around sustainable biomass potential, farm-to-plant supply chains, and indirect land use and land-use change emissions. DAC on the contrary has negligible land use unless the associated power source is

Table 14.11 Comparison of the land footprint, water footprint, cost, energy consumption, carbon efficiency and carbon breakeven time of DAC and BECCS technologies. While BECCS deployment faces sustainability limits such as high land and water use, DAC faces technological limits such as high energy requirement and cost. Time wise, DAC can remove CO₂ immediately, as opposed to BECCS, the carbon breakeven time of which can be longer than the lifetime of a BECCS project.

Metrics	BECCS	DACS
Water	60–2020 m ³ per t _{CO₂}	11–300 m ³ per t _{CO₂}
Land	3.7–33 t _{CO₂} per ha per year	negligible
Energy	Potential net energy producer	0.9–18 GJ per t _{CO₂}
Cost	10–350 \$ per t _{CO₂}	100–1000 \$ per t _{CO₂}
Carbon efficiency	Strongly dependent on biomass supply chain and BECCS pathway:	Strongly dependent on the source of the electricity used in DACS process
Carbon breakeven time	Potentially over 50 years	Removes CO ₂ right away

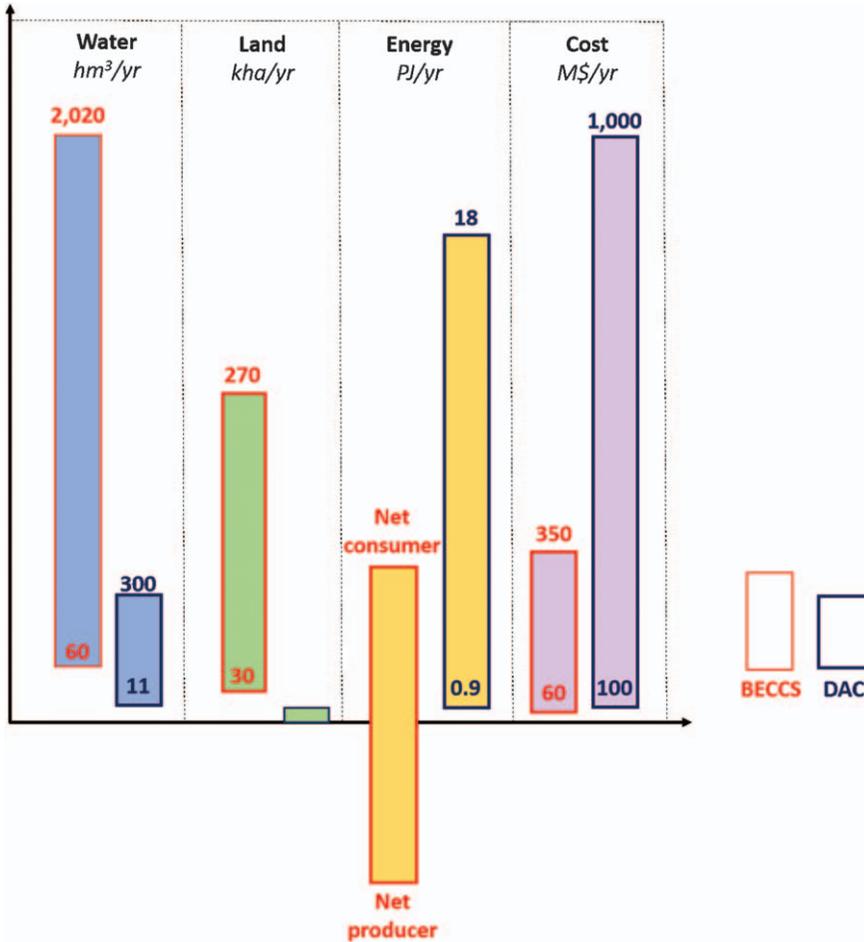


Figure 14.16 The resource implications of using BECCS and DAC to capture 1 MtCO₂ per year.

land intensive (e.g. solar PV). The DAC technology proposed in the APS report, however, is potentially water-intensive depending on the relative humidity of the region within which it is cited. Section 14.2.6.3 showed that DAC in humid regions will experience ~6 t_{H₂O} per t_{CO₂} losses, but this may be as high as 15 t_{H₂O} per t_{CO₂} in arid environments. Depending on the assumptions used for the calculations of BECCS water requirements, DAC and BECCS water uses could potentially be within the same range. One of the stated appeals of DAC is that there are fewer geographical limitations than BECCS, which needs reliable biomass supply to where it could be sited, provided utilisation, transport and/or storage infrastructure is available. This however raises questions about the suitability of DAC in some parts of the world that are arid and, likely, water-strained.

The \$10–350 per t_{CO_2} range of capture costs for BECCS indicates that it may be competitive with fossil fuels with CCS technologies that are being proposed today. A wide range of costs, however, has been cited for DAC technology, mainly because of the early stage of research and development of the technology – many, different process designs have been proposed but are yet to be engineered, even at demonstration scale, hence uncertainties about its technical feasibility persist. The upper bound of the estimates at \$1000 per t_{CO_2} indicates that DAC is the most cost-intensive negative emissions and emissions reduction option, and may provide an upper bound cost on climate change mitigation.

Providing carbon neutral power is used to supply energy for the DAC process, DAC carbon efficiency is by definition higher than that of BECCS. However, using carbon neutral power will ultimately increase the DAC costs, as well as land requirement, if the land footprint of the energy provided to the system is accounted for.

Ultimately, these technologies have different impacts on financial and natural resources and hence other economic sectors; should a preferred NET be sought, this will likely be dependent on geographical constraints and local resource availability.

In terms of the effectiveness of the technology in time, it was shown that wrong choices along the BECCS supply chain, such as the use of a low productivity biomass, high direct and indirect land use change, long road transport, high drying energy intensity, *etc.*, could lead to delayed carbon removal. There is thus a crucial difference between BECCS and DAC systems: providing the maturity of the technology, a DAC facility could actively start removing CO_2 from the atmosphere as soon as it starts operating, whereas BECCS net CO_2 removal from the atmosphere could be delayed by decades. Considering the time scale of climate change mitigation, with the need for negative emissions by mid-century, these CO_2 removal time efficiencies will likely play an important role when investing in a NET project.

14.6 Policy Implications

The proposed technologies to obtain negative emissions face different challenges to their development and their implementation. Relevant to all is the absence of an institutional framework addressing the legal and accounting implications of negative emissions. The carbon accounting framework for the land use, land-use change and forestry (LULUCF) sector has already received criticism because of the voluntary inclusion/exclusion of some emitting activities *e.g.* management of forest lands. An improvement of the existing system is necessary before it can be extended to apply to NETs. This is especially crucial for the biomass-dependent technologies.

In particular, BECCS faces two challenges that are specific to this technology: (1) How to deal with a diversity of stakeholders in the value chain? (2) How to set a quality standard for a BECCS project?^{85,159,160}

The BECCS value chain is diverse in terms of technologies and possibly stakeholders. Biomass production/collection and conditioning biomass conversion, CO₂ capture, transport, and finally injection and storage monitoring, are many steps that can likely be operated by different stakeholders. As biomass production, biomass conversion and CO₂ storage facilities are not all located at the same place, these stakeholders are likely to operate from different regions of the world. This poses two main challenges. Assuming BECCS would be deployed to answer a national specific carbon removal target, carbon negative crediting would thus have to account for the share of negative emissions achieved by each stakeholder. There are also sustainability and safety challenges associated with the large scale deployment of BECCS: to ensure actual and permanent CO₂ removal from the atmosphere, biomass has to be sustainably sourced, and CO₂ storage, carefully monitored. The liability as to the biomass supply chain sustainability and CO₂ storage monitoring will need to be established under a well-defined CO₂ accounting and monitoring policy framework. BECCS projects may also vary greatly in quality: carbon removal efficiency, resource efficiency, but also time efficiency, are many metrics that may change depending on the conditions of biomass sourcing, conversion, and CO₂ capture. Defining thresholds for each of these metrics, such as minimal water, land use, and minimal CO₂ efficiency could help to acknowledge these differences, and potentially weigh up the balance, along with cost, when deciding on the investment in a BECCS project. **AQ:20** Assuming a revenue is associated with the generation of negative emissions, BECCS carbon breakeven time will also play a determining role in the project financial viability, as delaying carbon negative revenues by 10 or 15 years could have a significant impact on the economic breakeven time of a BECCS project.

All of the NETs discussed face investment challenges because of a lack of incentives (such as a carbon credit) that improve the financial attractiveness of projects. Policy support is necessary in the early stages of development if commercial availability and deployment of these technologies is to be realised. However, the uncertainties around whether negative emissions can actually be achieved by any of the discussed means and continued debate discourage long-term policy commitment/support. The failure to act however risks a future without needed flexibility in the mitigation toolbox and possible lock-in.

14.7 Conclusion

There is a growing consensus that negative emissions technologies will play a critical role if the targets set-out at COP21 are to be met. With the exception of afforestation/reforestation, NETs are still in the early stages of research and development, and face a myriad of technical, bio-geophysical and socio-political barriers to their commercial viability. Preliminary studies have revealed that most NETs are not financially-viable without a financial incentive for the service they are to provide, *i.e.* a credit for CO₂ removal. Considerable

investment in research, development and demonstration, and policy that incentivises CO₂ removal projects are therefore needed to drive down technology costs, and attract investment by de-risking projects. The uncertainties surrounding indirect emissions resulting from the development of some NETs have raised questions about their ability to generate the net negative emissions necessary for climate change mitigation on the required time-scale. Developing coherent life-cycle assessment methodologies for emissions reduction technologies, and interdisciplinary research to observe the implications of NETs on other economic sectors will serve to constrain uncertainties. Furthermore, the Kyoto Protocol established a largely production-based carbon accounting framework that fails to account for leakages and allows for non-disclosure of emissions from some sectors. The current efforts to overhaul existing carbon accounting guidelines and methodologies (particularly in the LULUCF sector) provide an opportunity for the incorporation of NETs into mainstream climate change mitigation discussions. The inclusion of NETs in these mechanisms will also serve as a strong foundation for future policy integration. Failure to act may deprive the world of much-needed flexibility in mitigation solutions/pathways and may lead to lock-in to a future with more devastating climate change consequences or higher costs for mitigation.

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