Chapter **10**

Sabine Fuss (Potsdam Institute for Climate Impact Research, PIK)
Abby Lunstrum (University of Pennsylvania)
Ruben Prütz (Potsdam Institute for Climate Impact Research, PIK)
Friedemann Gruner (Potsdam Institute for Climate Impact Research, PIK)

Chapter scientist: *Friedemann Gruner (Potsdam Institute for Climate Impact Research, PIK)*

Cite as: *Fuss, S., Lunstrum, A., Prütz, R., Gruner, F. Chapter 10: Costs and Potentials, in* ***The State of Carbon Dioxide Removal 3rd Edition 2026*** *(eds. Edwards, M. R. et al.). DOI: https://doi.org/10.17605/OSF.IO/FBZ4X (2026)*

A farmer examines biochar made from rice paddy straw before its application during wheat crop sowing in Punjab.
By Aaran Patel

Chapter 10 | Costs and potentials

CDR methods vary widely in their estimated potentials, from less than 1 $GtCO_2$ per year to several tens of $GtCO_2$ per year. Removal via conventional methods is concentrated at the low-to-medium range, while some novel methods extend into the upper range. Cost estimates across all methods span a similarly wide range, from <US$10/$tCO_2$ to upper limits exceeding US$1,000/$tCO_2$. Substantial uncertainties underlie these estimates as well as the possible side effects of scaling each method.

Key insights

- Across and within CDR methods, potentials reported in the literature range widely. While some methods stand out in terms of lower (<1 $GtCO_2$ per year) or higher (tens of $GtCO_2$ per year) potentials, there is significant overlap. When considering higher-confidence estimates or more rigorous deployment assumptions, the low end of the ranges for almost all methods is near or below 1 $GtCO_2$ per year.

- Like potentials, cost estimates also range widely, with broad overlap across CDR methods. Most methods have upper limits well over US$200/$tCO_2$.

- Estimates of potentials are highly variable for many reasons, including low scientific understanding and data availability; different definitions of potentials; inconsistent assumptions about sustainability and durability; or inconsistent distinction between removals and avoided emissions.

- Cost estimates range widely for the same reasons. But other factors are also important, including different approaches within a method (e.g. there are several DACCS technologies); changing costs across level of deployment (e.g. for forestry); and different project boundaries (e.g. whether capital, operations and maintenance, or MRV costs are included and whether co-products are included in the valuation). As a result, cost estimates are often difficult to compare across methods, complicating design and policy decisions.

- Side effects can significantly shape a CDR method's removal potential and economic viability. Both positive and negative side effects are possible across all CDR methods. Importantly, side effects vary across deployment scenarios and scales.

- Limiting reliance on methods to well below their maximum potential accounts for the uncertainty in potential and preserves flexibility to implement CDR sustainably – maximizing positive side effects, limiting negative side effects and optimizing costs. This flexibility diminishes as reliance on CDR grows. Diversified portfolios of CDR methods can also help balance risks and mitigate resource constraints.

As the world remains on track to exceed the Paris Agreement's 1.5°C global warming limit – meaning that net-negative emissions will be required to reverse warming – questions arise around three dimensions of CDR:

1. How much CO_2 can be removed with each method?
2. What are the associated costs?
3. What trade-offs, risks or synergies may arise as the respective methods are rolled out?

Previous literature such as Fuss et al. (2018)[1] dates back almost a decade – a decade that has seen explosive growth in CDR knowledge[2] and the entrance of the topic into mainstream climate policy discussions.[3] In this chapter, we give an overview of the latest state of knowledge on estimated global potentials, costs and side effects of the CDR methods outlined in this report. Estimates draw on a targeted review of recent peer-reviewed research and grey literature (primarily 2018–2026, with older sources used where necessary), supplemented by expert review and input. We critically assess these heterogeneous estimates and highlight key uncertainties and knowledge gaps where evidence is limited or highly variable.

Unless otherwise indicated, potentials in this chapter are to be interpreted as technical potentials – defined, following the IPCC (2022),[4] as "constrained by biogeophysical limits as well as availability of technologies and practices. Quantification of technical potentials takes into account primarily technical considerations, but social, economic and/or environmental considerations are occasionally also included, if these represent strong barriers for the deployment of an option." Because they are based on biogeophysical limits, technical potentials are not additive, as each potential represents a resource use case maximizing the respective volumes of CDR available. Reported technical potentials for a given method may vary broadly across studies applying different assumptions on barriers to deployment. In the context of this chapter, we use "constrained" potential to indicate more rigorous assumptions on deployment conditions or higher agreement across studies. Potentials that are available at certain costs are called economic potentials.

Importantly, we do not interpret volumes of CDR deployed in scenarios of, for example, IAMs as potentials (see Chapter 8), even though these deployments also enter our assessment in comparison with the technical, bottom-up potentials. Both scenario deployments and bottom-up potentials depend on a plethora of factors such as the technological readiness level (TRL), permanence of storage and availability of resources. The treatment of these factors is not always consistent across studies; for example, estimates of the TRL for the same method may vary across different analyses. The aim of this chapter is to report estimates if they are documented in the peer-reviewed literature, along with a discussion of caveats and limitations.

Cost estimates are also heterogeneous, varying substantially depending on, for example, whether they contain capital costs or only operational costs, and whether they are assessed at the scale of a fully-mature market or a FOAK project – distinctions that are often impossible to disentangle. While detailed analyses assessing cost at scale are available for more mature technologies, many novel methods only have preliminary cost assessments.

In this chapter, we specify details about cost estimates (e.g. scope and scale) whenever possible. Ideally, a cost assessment across the different CDR methods would be based only on levelized costs of CO_2 removed (including capital and operational expenditures as well as the carbon removal efficiency), derived from LCA and techno-economic analysis to ensure comparability. However, very few such studies have been published, and those that are available review only a limited set of technologies, making a comprehensive and systematic review unproductive at this point (see Chapter 7). Where possible, we identify cost drivers and indicate whether costs are expected to increase or decrease in the future. These trends may be driven by multiple dynamics – on the one hand, increasing resource scarcities could drive up costs per unit CO_2 removed, while on the other hand, technological change, learning and economies of scale could simultaneously decrease costs.

Side effects arising from CDR deployment can affect both scalability and costs. Side effects generally refer to non-carbon outcomes of CDR deployment; these can either be positive (e.g. additional revenue streams from co-products like energy or construction materials, or ecosystem restoration that improves local microclimate or enhances habitat for endangered species) or negative (e.g. higher energy or food prices). Because the available information on side effects is too sparse for a systematic, quantitative review across methods, or across potential side effects domains,[5] we provide qualitative assessments in this chapter. We also identify this area as a significant knowledge gap that warrants further investigation, given growing demand not only from policymakers but also from buyers in the VCM, who increasingly seek information on the qualities of removal credits beyond carbon benefits (see Chapter 4). Another important aspect relevant in this context is potential competition with measures to directly reduce GHG. Significant resource bottlenecks may arise from competition among CDR methods and with other mitigation strategies, depending on scale, sustainability considerations and policy priorities.

Mitigation potentials and costs by CDR method

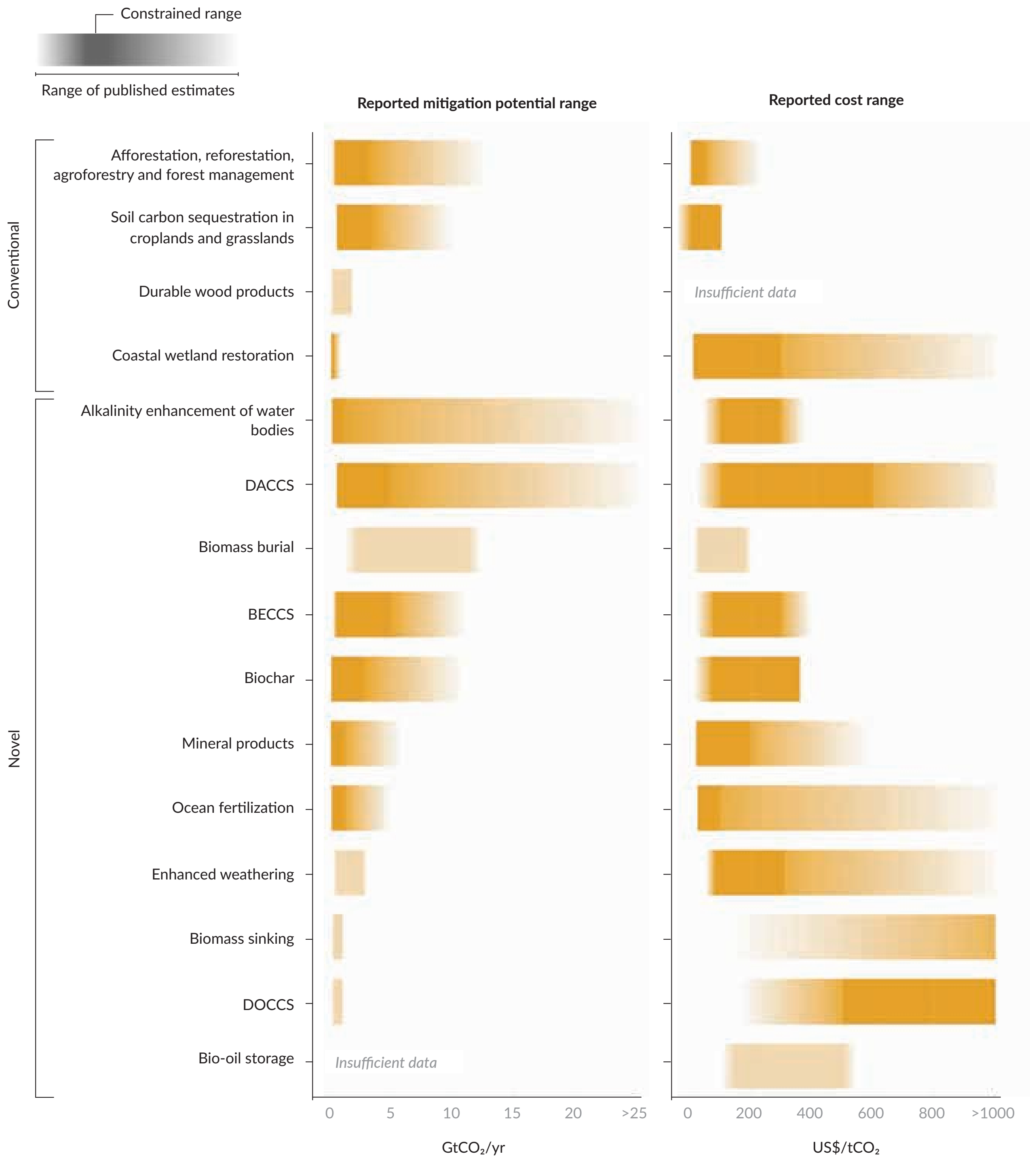

Figure 10.1 *Estimated ranges of mitigation potentials and costs of CDR methods. Presented ranges depict the spread of reported estimates in the literature. Darker shading indicates more constrained ranges, considering more rigorous deployment assumptions or higher agreement across reported estimates. Some estimates may also include avoided emissions, so we refer to mitigation potential instead of removal potential. Estimates of costs also vary across methods in terms of system scope and whether they are meant to represent current costs versus future costs at scale. For both potentials and costs, these different methodological approaches limit direct comparability across CDR methods. Further, note that estimates of potential are not additive due to overlapping resource demands and related constraints.*

10.1 CDR methods

Afforestation, reforestation, agroforestry and forest management

Afforestation, reforestation, agroforestry and forest management methods differ in implementation mode and potential side effects. They represent almost all current CDR (see Chapter 7) and are all characterized by a high level of technological readiness (TRL 8–9).[6] Nevertheless, estimates of sustainable and economic potentials vary widely: they are sensitive to the carbon price as well as uncertainties related to Earth system feedbacks.

Global potential: A rich body of literature exploring the removal potential of forest-based CDR methods has emerged in recent years. However, removal potential estimates vary substantially due to differences in the definition of forest, assumed land availability, sustainability and other constraints and feedbacks to CDR deployment.[7] The literature estimates place the technical removal potential of afforestation at up to 13 $GtCO_2$ per year.[8] Sustainable estimates – which account for unfavourable sustainability implications and Earth system feedbacks, and focus on reforestation rather than afforesting land that has historically not been covered by forest (often defined as >50 years) – suggest a considerably smaller removal potential of up to 2.2 $GtCO_2$ per year.[7,9] However, estimates of sustainable potential still vary considerably (<0.1–8.9 $GtCO_2$ per year), depending on the underlying assumptions.[7,10] In general, the assessed studies estimate CO_2 removal potential additional to current levels, which for afforestation and reforestation are approximately 2.2 $GtCO_2$ per year (see Chapter 7).

Climate-related Earth system feedbacks may emerge under warming levels above 1.5°C, possibly limiting removal potential by affecting sequestration capacity and permanence. These factors remain relatively underexplored.[11,12] The Highest Ambition scenarios (see Chapter 8) feature removals of 3.3–8.4 $GtCO_2$ per year via afforestation, reforestation and forest management by 2050, and up to 0.7 $GtCO_2$ per year via agroforestry. These scenario-based deployment rates are compatible with upper bound estimates of mitigation potential but exceed some of the more constrained estimates in the literature.

Costs: Evidence suggests removal costs at scale of around US\$5–US\$53 per tCO_2 removed,[1,6] while wider cost ranges (US\$0–US\$240/tCO_2) can be found in the literature.[13] Removal potential varies as a function of the underlying carbon price with up to 0.9 $GtCO_2$ per year at US\$0/tCO_2 and up to 2.1 $GtCO_2$ per year at US\$100/tCO_2 over a 30-year (non-discounted) period.[14] Note that these estimates are based on a small evidence base, narrowly focused on implementation and opportunity costs; considering additional factors, such as MRV or transaction costs, would increase the overall removal costs.[14]

Alkalinity enhancement of water bodies

There are several proposed methods for adding alkalinity to water bodies, ranging in complexity from coastal applications of enhanced weathering to electrochemical technologies. Because of the diversity of approaches, estimated TRL currently spans the complete range between fundamental research and operational deployment.[15,16] In most approaches, alkalinity is added to seawater, and the resulting uptake of atmospheric CO_2 occurs gradually over broad areas of the ocean surface, where quantification is achieved primarily through ocean modelling. Such approaches are typically referred to as ocean alkalinity enhancement (OAE). A small subset of technologies equilibrate CO_2 with alkalinity in reactors prior to discharge, facilitating MRV; these approaches use high concentrations of biogenic CO_2 from, for example, wastewater treatment plants or bioenergy facilities to facilitate faster reactions. Uncertainties remain about the efficiency of the method, with ongoing scientific study of the physical, biological and geochemical ocean processes that mediate CO_2 uptake.[17]

Global potential: The global technical potential of alkalinity enhancement is poorly constrained, estimated at <1 Gt to 100 $GtCO_2$ per year.[13,16] Potentials for specific applications in rivers or integration with existing industries (e.g. wastewater treatment plants) are estimated at 10s of $MtCO_2$ per year, and integration with the shipping industry may exceed 1 $GtCO_2$ per year.[18–20] Ocean applications have recently been considered in IAMs; in the Highest Ambition scenarios (see Chapter 8), deployment varies between <0.1 and 1.4 $GtCO_2$ per year by 2050. Economic models also anticipate slow deployment, not exceeding 1 $GtCO_2$ per year until 2100.[21] Apart from direct ocean applications, alkalinity enhancement of other water bodies has not yet been considered in IAMs.

Costs: Costs are currently estimated at US\$100–US\$150/tCO_2, although they may exceed US\$300/tCO_2 for some technologies and US\$600/tCO_2 for FOAK deployments.[16,22–24] While lower-tech options like direct placement of minerals on the seabed are estimated to be less expensive, their scientific uncertainties and MRV hurdles are higher.[1,25] Costs are expected to decrease with economies of scale, although the energy, minerals and infrastructure required may pose constraints.[26]

BECCS

BECCS is an umbrella term for a range of technologies that use biomass feedstocks for energy conversion, combined with carbon capture and geological storage. While feedstock sourcing and geological storage build on well-established practices[27] (see Box 10.1), the capture approaches occur at different levels of technological maturity. For example, bioethanol facilities with CCS are commercially deployed (TRL 9), and biomass combustion with CCS is at the demonstration stage (TRL 7), while gasification-based routes remain less mature.[28]

Global potential: Literature estimates of the global technical potential for BECCS by mid-century span 0.5–11 $GtCO_2$ per year.[13] Constrained potential, however, could be lower once sustainable biomass supply, land-use constraints and storage availability are considered. Earlier assessments suggested constrained potentials of 0.5–5 $GtCO_2$ per year by 2050.[1] A recent planetary-boundary analysis indicates a more limited sustainable potential of 0.1–0.9 $GtCO_2$ per year by mid-century from dedicated biomass plantations, assuming existing agricultural land remains reserved for food production.[29] Some additional sustainable potential may exist from agricultural and forestry residues and organic wastes, but these resource pools are region-specific and compete with other uses (see Box 10.2).

BECCS remains one of the main CDR methods deployed in IAM scenarios. In the Highest Ambition scenarios (see Chapter 8), modelled removals via BECCS reach 0.7–4.2 $GtCO_2$ per year by 2050. These scenario-based deployment levels exceed some of the lower estimates in the literature but are compatible with upper bound estimates of constrained potential.

Costs: Available cost estimates for BECCS remain sparse and divergent, reflecting the complexity of the value chain and differences in biomass feedstocks, conversion routes and system scale.[30] In addition, existing reviews often do not consistently report on the cost boundaries, for example which elements of the CCS value chain are included or whether revenues from BECCS energy products are accounted for in reported costs.

Considering this diversity of approaches, the IPCC reports a broad cost range of US$15–US$400/tCO_2,[13] while a review by Oh et al. (2025) spans US$13–US$288/tCO_2,[31] with biomass combustion routes at the upper end and bioethanol facilities at the lower end. For retrofitted bioenergy plants, modelled costs in 2030 are approximately US$140–US$260/tCO_2 for Fischer-Tropsch fuel plants, US$300–US$470/tCO_2 for energy-from-waste plants, and US$150–US$290/tCO_2 for biomass-fired power plants, where the cost estimates include forgone revenue due to the energy consumption of the CCS retrofit.[27] The IEA places current costs around US$75–US$300/tCO_2 and possible long-term costs around US$40–US$125/tCO_2.[32] While costs may fall with technological learning, limited biomass availability could also drive cost increases over time[30,31] (see also Box 10.2). This tension between learning effects and biomass constraints is also reflected in the wide range reported by the expert elicitation in Abegg et al. (2024), which focuses on biomass combustion routes and explicitly includes revenues from the energy produced, yielding estimates of US$65–US$325/tCO_2 by 2050.[30]

Biochar

While artisanal biochar production is not a novel process, readiness estimates for industrial use of biochar for carbon removal vary (TRL 7–9)[6,33] with lower readiness for biochar application cases other than soil amendment such as cement and concrete production.[27] Biochar is the largest contributor to current removals from novel CDR (see Chapter 7), and some estimates suggest high global potential; however substantial uncertainties in economic and sustainable potentials, as well as costs, remain.

Global potential: A recent review reports a wide range of estimates for biochar's technical potential, reaching up to 11 $GtCO_2$ per year.[34] But economic constraints may substantially limit deployment, with some studies suggesting an economic potential of <0.1 $GtCO_2$ per year.[34] This large range is rooted in different assessment scopes and assumptions about feedstock availability, with lower estimates when constraining feedstock sourcing to organic residues and waste materials. Recent evidence suggests a sustainable removal potential of up to 2.7 $GtCO_2$ per year, while total mitigation potential (including avoided and reduced emissions) could be up to 10.3 $GtCO_2$ per year.[34] In the Highest Ambition scenario (see Chapter 8), modelled removals from biochar reach between <0.1 and 1.1 $GtCO_2$ per year by 2050. These scenario-based deployment rates are compatible with the sustainable removal potential reported in the literature but exceed some estimates of economic potential.

Costs: A recent comprehensive cost analysis – considering biomass and biochar supply chains and pyrolysis costs as well as electricity-generation revenues – suggests future biochar costs of US\$70–US\$360/tCO_2,[35] while previous assessments of estimated costs by 2050 suggest lower costs of US\$10–US\$345/tCO_2.[1]

Biomass burial

While biomass burial is often described as readily available due to the simplicity of the proposed approach, a recent assessment suggests a TRL of 4–6.[6] The scientific literature on biomass burial is still small compared to other CDR methods.

Biomass slurry injection is a related method, wherein biomass feedstocks are injected as a slurry into deep geologic reservoirs. The technology is well-established for waste management applications and has been used for decades in the oil and gas industry.

Global potential: Several studies suggest that annual carbon removal from biomass burial could reach double-digit gigatonne levels.[36–38] Analyses of biomass slurry injection are similarly limited, with global potential estimated around 5 $GtCO_2$ per year.[39] More research is needed on both methods to assess the feasible removal potential given sustainability constraints and competing biomass uses.

Costs: Cost estimates for biomass burial at scale are on the order of US\$10–200/tCO_2,[37,40,41] while costs for MRV would further increase overall costs. Estimates for biomass slurry injection are <US\$100/tCO_2.[39]

Biomass sinking

Ocean biomass sinking can use either terrestrial (e.g. agricultural waste) or marine (e.g. macroalgal) biomass.[16,42,43] Each biomass type has different logistical and resource considerations, resulting in a range of estimated potentials, costs and side effects. Macroalgae cultivation is proposed in offshore locations – or in nearshore locations followed by transport of biomass to offshore locations, where it can be sunk to depths below 1,000 metres.[16,43] For terrestrial biomass, sinking is proposed in anoxic basins, either in inland water bodies or near-coastal areas, for logistical, legal and environmental reasons.[42] In addition, macroalgae sinking requires ocean modelling to verify uptake of atmospheric CO_2 into seawater, whereas terrestrial biomass captures CO_2 directly from the atmosphere (via photosynthesis on land), and does not require the same degree of ocean modelling. The TRL of both approaches is low.

Global potential: For farmed macroalgae, global CDR potential is estimated between 0.1–1 $GtCO_2$ per year.[16] Higher estimates have been put forth (e.g. up to 630 $GtCO_2$ cumulatively between 2020 and 2100); however, these derive from unrealistic simulations – with farms covering a significant fraction of the world's ocean and minimal nutrient limitation.[43] As with nutrient fertilization methods, nutrient robbing will reduce productivity in downstream ecosystems and may lower overall potential.[44,45] Uncertainties also remain about the durability of storage and impacts on ocean ecosystems, so more constrained estimates are unavailable. For sinking of terrestrial biomass, very little scholarship exists; estimates considering anoxic basin capacity propose a conceptual potential exceeding a range of 0.1–1 $GtCO_2e$ per year.[46] Realistic potential will be sharply limited by sustainable supply chains, logistics, competing biomass use and regulation.

Costs: For farmed macroalgae, production costs alone range from <US$100 to >US$10,000/tCO_2, driven by labour, transport and supplies.[47,48] Likely costs for CDR exceed US$1,000/$tCO_2$, and while economies of scale could reduce total cost to US$100/$tCO_2$, massive farms on the scale of millions of hectares would be required.[16,47,49] There are no cost estimates for terrestrial biomass sinking.

Bio-oil storage

Bio-oil storage relies on established technologies for fast pyrolysis (e.g. bio-oil production for chemicals) and underground storage (e.g. oilfield waste slurry injection), but the process of bio-oil injection for CO_2 storage is still novel and remains at an early stage of development (TRL 5).[28]

Global potential: Potentials for bio-oil storage have not yet been systematically assessed. No robust estimates of technical or constrained potential are available, and any feasible scale will depend strongly on sustainable biomass availability. Deployment in IAM scenarios remains limited: in the Highest Ambition scenario (see Chapter 8), modelled removals from bio-oil storage range from between <0.1 and 0.2 $GtCO_2$ per year by 2050.

Costs: Cost information is similarly sparse. A recent modelling study suggests that deploying many small, decentralized bio-oil facilities could achieve costs in the range of US$100–US$200/tCO_2.[50] Self-reported data of a removal company submitted to the Frontier advance market commitment (see Chapter 4) in 2024 indicated a price of around US$550/$tCO_2$.[15]

DACCS

DACCS technologies span a wide range of capture processes, each at different TRLs, while the storage component builds on well-established CO_2 geological storage practices (see Box 10.1). The most mature capture options, solid-sorbent systems and mineral looping, are already commercially deployed (TRL 9), while others remain at pilot scale, such as liquid solvent systems (TRL 6–7), or earlier development, such as electrochemical regeneration (TRL 4–6).[51] DACCS technologies are energy intensive, and performance is strongly shaped by energy demand and the carbon intensity of the energy supply.[51]

Global potential: While the literature generally treats the technical potential of DACCS as effectively unconstrained, assessments that account for limits in low-carbon energy supply and storage availability suggest more finite potential. Reported constrained estimates range from 5–40 $GtCO_2$ per year in IPCC AR6 (2022)[13] to 0.5–5 $GtCO_2$ per year by 2050 in Fuss et al. (2018).[1] Feasible CO_2 injection rates and the pace of scale-up over the coming decades could further constrain achievable annual deployment.[52]

Despite the large technical potential, IAMs often feature lower deployment levels of DACCS. In the Highest Ambition scenarios (see Chapter 8), modelled removals from DACCS range between <0.1 $GtCO_2$ per year and 1.8 $GtCO_2$ per year by 2050. Similarly, a recent systematic review by van der Spek et al. (2025) uses updated cost assumptions and finds little to no deployment across pathways remaining below 2°C or in 1.5°C pathways with limited overshoot, while higher deployment of around 3.6–9 $GtCO_2$ per year by 2100 emerges only in a 1.5°C pathway with high overshoot.[51]

Costs: The review by van der Spek et al. (2025) finds levelized costs of gross removal ranging from roughly US\$400–US\$2,500/tCO_2.[51] These wide ranges reflect differences in the technologies evaluated as well as the underlying assumptions used across studies, for example, regarding energy efficiency, plant configuration or solvent and sorbent choices. Many studies do not model the full DACCS value chain – notably CO_2 compression, transport and storage – and exclude additional cost components such as owner's cost or balance of plant items (i.e. the cost for supporting infrastructure). Furthermore, most studies focus on the United States, but location could also strongly influence costs. More comprehensive cost assessments are, therefore, needed. Looking ahead, the review by van der Spek et al. (2025) concludes that costs could fall to US\$100–US\$600/tCO_2 once deployment reaches gigatonne scale, largely through economies of scale and learning-by-doing (see Box 10.3). Variation in these projections is driven, in particular, by the assumed learning rates.

DOCCS

Most DOCCS technologies involve pumping seawater through electrochemical systems to extract dissolved CO_2. While this CO_2 extraction occurs at industrial facilities, the subsequent uptake of atmospheric CO_2 (into the CO_2-depleted seawater) occurs gradually over broad areas of the ocean surface. Current technologies are at lab-to-pilot scale (TRL 5).[16,28] Seawater mineralization is a technologically similar approach to electrochemical DOCCS, whereby seawater carbon is extracted in the form of solid carbonate minerals. The deployment, MRV and storage considerations of each method are distinct, however, and in this section we refer specifically to DOCCS.

Global potential: In theory, the technical potential of DOCCS is limited by the rate of CO_2 uptake by the ocean and geologic storage capacity, although realistic rates of seawater pumping are also likely limiting. Currently, only rough estimates of constrained potentials are available, estimated at 0.1–1 $GtCO_2$ per year, given realistic resource limits on critical metals for electrolysers and membranes, and energy demand.[16,22] Integrating DOCCS into coastal infrastructure like power plants and desalination facilities could offer the potential for approximately 50–60 $MtCO_2$ per year by 2050.[53] Given high costs, economically constrained potential remains very small (<0.1 $GtCO_2$ per year) by 2100.[21]

Costs: Although DOCCS is thermodynamically more efficient than DACCS, realistic operating conditions make the technology more expensive.[54] Estimates for the technology alone are as low as <US\$100/tCO_2; however, systems-level costs for seawater processing raise costs to US\$1,000–US\$2,000/tCO_2. Co-location with existing coastal infrastructure is often considered, reducing estimated costs to US\$500–US\$700/tCO_2.[53,55,56] These estimates do not include MRV or CO_2 storage.

Durable wood products

The TRL is rather high for sequestration of CO_2 in long-lived wood products,[57-58] but estimates for potentials and costs at the global level are inherently uncertain, contingent on changing trajectories regarding population, average floor space per capita and climate impacts on yields.

Global potential: Bottom-up estimates vary by what type of products are included and what assumptions are made about future developments. Estimates of the contribution of wood products to storing carbon in end uses, based on historical data extrapolated along Shared Socioeconomic Pathways (see Chapter 8), show a global pool of 0.3 $GtCO_2e$ per year in 2015, rising to 0.4 $GtCO_2e$ per year by 2030 under favourable socioeconomic conditions, and up to 0.6 $GtCO_2e$ per year by 2065.[57] Different scenarios over 30 years (2020–2050) for new urban buildings designed with timber reveal potential of <0.1–2.5 $GtCO_2$ per year, depending on scenario and average floor area per capita.[58] Modelling studies based on cost-optimization estimate a cumulative potential of 23–91 $GtCO_2$ over 50 years (2015–2065)[59] or 4.1–8.1 $GtCO_2$ over 80 years (2020–2100),[60] thus featuring much higher variation than bottom-up estimates. In the Highest Ambition scenario (see

Chapter 8), modelled removals from durable wood products contribute 0.2–0.3 $GtCO_2$ per year by 2050. These scenario-based deployment rates are consistent with ranges reported in the literature.

Costs: We abstain from estimating a cost range for carbon removal through wood products on account of the vast heterogeneity of methods and the absence of comparable cost estimates in the peer-reviewed literature. Sources of uncertainty include varying project boundaries and differences in lifecycle duration. For cases where wood is replacing another material, the relative differences in costs and emissions are also important for understanding cost and economic potential.

Enhanced weathering

Enhanced weathering takes place slowly in the environment – over months to centuries depending on rock properties and environmental conditions – and MRV of CO_2 uptake remains a challenge. Consequently, assessments of TRL vary broadly, with conservative estimates of 3–4 and supplier-provided estimates of 8–9.[13,15] Silicate rocks are typically considered in both research and commercial activities, although carbonate rocks can also be effective. Most applications use agricultural land, where acidic soils expedite reaction rates, and existing supply chains can be leveraged to distribute materials.

Global potential: Estimates of global technical potential typically focus on agricultural applications and range from 0.2–2 $GtCO_2$ per year, with cumulative potential over 50 years of 25–100 $GtCO_2$.[61,62] Sustainable potential, considering biophysical and economic limits, may be limited to 0.7 $GtCO_2$ per year.[63] Higher estimates, for example reaching 95 $GtCO_2$ per year, have been proposed for use of more reactive, but less common rocks; such estimates do not consider realistic geochemical feedbacks, however, and are likely unattainable.[64] All global estimates are somewhat uncertain because of knowledge gaps in biogeochemical cycling and Earth system processes. Global estimates typically extrapolate from laboratory data, whereas actual rates in the environment are both slower and decrease over time, and significant carbon losses may occur "downstream" in soils and groundwater.[65–67] The additionality of enhanced weathering has also not yet been comprehensively assessed, considering that application of agricultural lime is already commonplace in some areas.[68] Because of the large temporal and spatial scales needed to assess enhanced weathering, confirming net CDR with real-world monitoring is challenging and requires ongoing innovations in MRV. Economically-constrained assessments often do not integrate these uncertainties and are on the high end or even exceed technical estimates, reaching 1–10 $GtCO_2$ per year by 2100.[21,35] Enhanced weathering is increasingly considered in IAMs, and the Highest Ambition scenarios (see Chapter 8) model deployment reaching <0.1–1.5 $GtCO_2$ per year by 2050.

Costs: Current cost estimates for enhanced weathering often range between US$50 and >US$300/tCO_2, although more detailed estimates from lifecycle assessments can exceed US$1,000/$tCO_2$.[27,61,64,69] Costs are dominated by transportation and the crushing and grinding of rocks. Similar to estimates of global potential, however, the net CO_2 uptake in the environment is a critical consideration, and several uncertainties have not yet been integrated into cost estimates.

Mineral products

Mineral products can store CO_2 via reaction with naturally occurring calcium- or magnesium-rich silicate rocks, or industrial by-products like cement waste, steel slag and coal ash. In most applications, mineralization technologies use concentrated CO_2 to facilitate faster reaction rates. TRL for these approaches ranges from 3–9, depending on specific feedstocks and technologies.[70] In limited applications, highly alkaline wastes or reactive feedstock synthesized from natural rocks can react directly with atmospheric CO_2 to achieve both capture and storage. There is less literature available for these passive air capture approaches. For all approaches, the long-term fate of carbonated mineral products is important for assessing both durability and additionality, especially considering potential integration with existing processes in the mining and construction sectors.

Global potential: The potential for CO_2 storage in mineral products is limited by the availability of alkaline feedstocks. While geologic reserves of mafic and ultramafic rocks are essentially unlimited, current tailings volumes could sequester <200 $MtCO_2$ per year.[71] Industrial wastes currently offer a potential of 1 $GtCO_2$ per year, increasing to 2.3–3.3 $GtCO_2$ per year in 2050 and up to 5.9 $GtCO_2$ per year by 2100.[72] However, these wastes are produced within carbon-intensive industries, for example cement, so boundary definitions are important for defining suitability for CDR. For some industrial feedstocks, resource competition may occur between other CDR methods, including enhanced weathering and alkalinity enhancement of water bodies or with mitigation activities like displacement of cement in concrete mixtures.

Costs: The cost of carbon removal via mineral products will vary with feedstock and the technology used. Current estimates range from US$10s/$tCO_2$ for highly reactive industrial wastes to >US$500/$tCO_2$ for natural feedstocks that require more chemical processing.[73] These costs often encompass the mineralization process only and do not include the cost of CO_2 capture or concentration. Recovery of valuable co-products, such as cement additives or metals, may be possible for some feedstocks, helping to defray both costs and environmental impact.[74,75] The carbon-containing products are often more expensive than conventional alternatives used in the construction industry.[76] In optimized scenarios with limited scale, integrating CO_2 mineralization with construction industries could achieve costs around US$100/$tCO_2$.[77]

Ocean fertilization

While there is medium-to-high confidence, based on several field trials, that ocean fertilization enhances primary productivity, the net effect on atmospheric CO_2 uptake remains poorly constrained, with uncertainties about the depth and extent of biomass remineralization and ultimate export to the deep sea.[78] This leads to uncertainty about both the durability of stored carbon and the efficiency of the approach. TRL is consequently low (1–2).[13,16,79] Ultimately, the net uptake of atmospheric CO_2 occurs over broad areas of the ocean surface, requiring ocean modelling for MRV.[80]

Global potential: Most estimates of the technical potential of ocean fertilization range from 0.1–1 $GtCO_2$ per year, although some exceed several $GtCO_2$ per year.[16,81] Nutrient robbing will reduce productivity in downstream ecosystems, potentially reducing the overall CDR efficiency and causing unintended negative environmental and societal impacts.[16,79] Deployment is typically proposed via ships, and the Southern Ocean is widely considered the most promising location due to its abundance of preformed nutrients, expansive size and low iron concentration.

Costs: Estimated costs are highly sensitive to the assumed export efficiency, ranging from <US$10/$tCO_2$ for high assumed efficiency to >US$10,000/$tCO_2$ for low assumed efficiency.[82,83] Capital costs are also significant, but projections of total cost are relatively insensitive to learning rates, given persistent uncertainty around export and durability.[82]

Peatland and coastal wetland restoration

Peatland and coastal wetland restoration is a wide field with many different approaches. The TRL of these approaches is usually not assessed quantitatively but is generally considered to be at a medium-to-high level. For both peatland and coastal wetland restoration, reported mitigation potentials often combine emissions reductions and removals, along with anticipated future removals; these estimates are difficult to disentangle, so isolating a CDR potential is not straightforward.

Global potential of peatland restoration: The technical potentials for reduced emissions have been estimated at about 0.5–2.5 $GtCO_2e$ per year[84] in 2030 based on the extent of degraded peatlands from the Global Peatland Database. Applying a GIS approach with IPCC emissions factors results in an estimated range of 0.3–3.4 $GtCO_2e$ per year.[85] Those estimated ranges roughly align with the IPCC assessment of 0.5–1.3 $GtCO_2e$ per year.[86] Scenario analyses[87] running up to 2100 are well within this range – at about 1 $GtCO_22e$ per year. It is important to note, however, that potential removals will be contingent on the regrowth of the peat body in the longer term. While recent literature shows that some rewetted peatlands will eventually sequester carbon, they mostly remain potential sources of mainly methane during at least the first 20 years after restoration.[88] We follow the IPCC[86] in refraining from giving explicit removal potentials and costs for peatland restoration due to lack of evidence on:

- The performance under climate change, considering that a sink can revert to a source in a warmer or drier year;
- Potential interactions with diet change and food demand; and
- Geographical differences, which are not well-studied – for example, tropical peatlands are even less well-studied with respect to carbon cycling than northern peatlands, where carbon stocks are increasingly threatened by wildfire and permafrost melt.[89]

Costs of peatland restoration: Half of the mitigation potential of 0.5–1.3 $GtCO_2e$ per year is available at up to US$100/$tCO_2$.[90] Note that costs could be very low, depending on factors such as whether co-benefits are valued. Other cost uncertainties include methods used, local conditions and complexity of projects.

Global potential of coastal wetland restoration: Macreadie et al. (2022)[91] estimate a bottom-up technical potential of 0.6–1.1 $GtCO_2e$ per year by 2030. For 2050, the most recent IPCC assessment finds a smaller range of <0.1–0.8 $GtCO_2e$ per year, based on the peer-reviewed literature, of which <0.2 $GtCO_2e$ per year would be available at a price of up to US$100/$tCO_2$ (economic potential).[92] The grey literature offers a 2050 removal potential that is only marginally higher – up to 0.8 $GtCO_2e$ per year in these categories.[93] Lower potential ranges in 2050 than in 2030 may indicate increasing scarcity.

Costs of coastal wetland restoration: Cost estimates in the grey literature are more comparable across methods than estimates from the peer-reviewed literature, where it is often unclear whether variation is due to regional differences, technology or carbon sequestration. Ranging between US$11/$tCO_2$ and US$300/$tCO_2$,[94] with mangrove restoration at the lower end and seagrass meadows at the upper end, the grey literature not only offers more narrow cost estimates for 2050, but these estimates are also two to four times lower than those found in the peer-reviewed literature.[95] These differences may stem from different lifecycle emissions, inclusion of operational and capital costs, and other factors. Generally, costs are expected to decrease by more than one-third between 2030 and 2050 due to improvements in seeding and overcoming challenges in MRV.

Soil carbon sequestration in croplands and grasslands

Soil carbon sequestration comprises a variety of management practices aimed at increasing carbon uptake and storage in croplands and grasslands through organic fertilizer use, crop management or adjusted tillage practices, among others. It is characterized by a high level of technological readiness (TRL 7–9),[6,13,96] but the potential approaches to maintain and monitor the long-term effectiveness of soil carbon sequestration remain underexplored.

Global potential: Literature estimates of the technical removal potential of soil carbon sequestration in croplands and grasslands vary substantially, with annual double-digit gigatonne removal levels on the upper end of the spectrum.[97] Several studies find substantially lower technical removal potential,[1,98] with constrained estimates ranging between 0.5 $GtCO_2$ per year[8] and 3.4 $GtCO_2$ per year.[97] Saturation effects and the risk of climate-related reversibility of soil carbon are among the critical factors constraining potential.[1,99] Such factors require further investigation to increase the robustness of potential estimates. In the Highest Ambition scenario (see Chapter 8), modelled removals from soil carbon sequestration in croplands and grasslands range from <0.1 and 0.8 $GtCO_2$ per year via by 2050. These scenario-based deployment rates are compatible with upper bound estimates of constrained potential but exceed some of the lower bound estimates in the literature.

Costs: Removal costs are highly variable depending on the underlying soil carbon management practice and due to differences in the cost factors considered (e.g. labour costs), reaching US$105 per tCO_2e on the upper end,[6] with negative cost estimates in the best case (US$-45/tCO2) when accounting for crop yield increases potentially resulting from management practices.[1]

Box 10.1 Geological storage

Geological storage of CO_2 is a requisite component of several methods at scale, including DACCS, DOCCS and BECCS. For these methods, captured CO_2 is typically injected into deep saline aquifers or depleted oil and gas reservoirs, where it is physically and chemically trapped beneath impermeable rock formations. Both the transport (via pipeline) and injection of CO_2 are well-understood based on decades of experience in EOR, and leakage risks during storage are very low.[100] Alternatively, CO_2 can also be injected into mafic rock structures where it is mineralized into solid carbonates, resulting in highly secure and permanent storage; however, this method is at an earlier stage of development.[101] While bio-oil storage also requires suitable geologic storage, structural requirements are unique from CO_2 storage, and shallower reservoirs can be used.

Storage potential: Theoretical estimates of global, geologic CO_2 storage potential are on the order of 10,000 $GtCO_2$ or more, with a majority in deep saline formations.[100,101] This capacity is approximately evenly split between onshore and offshore locations.[102] However, only a fraction of the total capacity, perhaps closer to 1,000 $GtCO_2$, is considered usable in practice, due to geological, engineering and societal constraints.[100,102,103] While this (more limited) volume remains sufficient to accommodate storage needs in most IPCC scenarios, continued reliance on fossil CCS or the large-scale deployment of CDR could deplete the resource over the next century. This raises important questions about how to allocate and prioritize storage capacity across regions, sectors and time.[102]

Costs: The cost of geological CO_2 storage involves three main components: compression, transport and storage, which typically includes MRV. Compression is required to reduce the volume of captured CO_2, allowing it to be efficiently transported and injected into deep geological formations. This step typically costs US\$10–US\$30/tCO_2, depending on target pressure and the scale of the facility.[104,105]

Transport costs vary by distance, infrastructure type and scale. For high-capacity onshore pipelines (e.g. transporting more than 1 $MtCO_2$ per year), costs are typically only a few dollars per tonne per 100 km.[101,106] Offshore pipelines are generally 40%–70% more expensive than onshore systems.[101] In certain cases, ship transport may be more cost-effective, particularly for longer distances and modest volumes. For example, transporting 1 $MtCO_2$ per year over 1,000 km by ship is estimated to cost approximately US\$20–US\$30/tCO_2.[101]

Finally, storage costs are site-specific and depend on factors such as geological conditions, injection rates, site maturity and the regulatory environment. MRV costs are often included in storage cost estimates. Onshore storage is typically less expensive, especially in already explored geological sites, with costs ranging from a few dollars to less than US\$25/tCO_2.[104,105,107] By contrast, offshore storage is generally more expensive, with estimated costs ranging from approximately US\$5–55/tCO2.[104,105] However, costs for newly developed sites remain highly uncertain and are expected to increase, reflecting increasing prices for key plant and infrastructure components and limited near-term potential for technological cost reductions.[101,108]

10.2 Wide variation in CDR potentials

Across CDR methods, a wide range of technical potentials exists, from less than 1 $GtCO_2$ per year for some methods to several tens of $GtCO_2$ per year for others (see Figure 10.1). While some methods stand out in terms of lower or higher potentials, there is significant overlap, and almost all methods have constrained potentials – which apply more rigorous assumptions or show higher agreement across studies – on the low end near (or below) 1 $GtCO_2$ per year. Because of this large spread, methods cannot be easily ranked but instead can be grouped into illustrative bins according to maximum technical potential estimates. Several methods are currently thought to offer low potentials (<1 $GtCO_2$ per year), including peatland and coastal wetland restoration, biomass sinking, DOCCS and durable wood products. Methods with low-to-medium-potentials (<1 to 5 $GtCO_2$ per year) include ocean fertilization, enhanced weathering and storage in mineral products. Several biomass-based methods, including biochar, BECCS and biomass burial, could technically offer medium potentials (up to around 10 $GtCO_2$ per year), as do soil carbon sequestration and forestry-based methods. Finally, a few methods may theoretically offer high potentials (exceeding 10s of $GtCO_2$ per year), including DACCS and OAE.

The broad ranges within and across methods result, in part, from inconsistent scopes and constraints considered in technical potential assessments. Technical potentials primarily consider biogeophysical limits but can also include social, economic or sustainability considerations if these are thought to present significant barriers to deployment.[4] There is no required level of significance to include such considerations, preventing straightforward comparison across studies. For land- and biomass-intensive methods, sustainability assumptions are particularly important and differ widely across estimates. Even for mature, conventional methods like afforestation and reforestation, potentials span several orders of magnitude, a result of variable sustainability constraints such as land availability, biodiversity, albedo and food security. Some estimates in the literature also vary significantly in scope. For example, assessments of peatland and coastal wetland restoration often include avoided emissions and implications for fluxes of non-CO_2 greenhouse gases, which are difficult to disentangle from CDR, complicating assessments of potential.

Estimated potentials also vary because of scientific uncertainty, especially for methods that leverage natural processes within the environment to capture and store carbon. For enhanced weathering, the loss of sequestered carbon through biogeochemical cycling in soils, groundwater and rivers may be a significant limit to potential; however, these processes are poorly constrained, and expert opinions differ greatly on their magnitude.[66] Similarly, estimated potentials for ocean-based methods rely on ocean models that remain under active development and which have not yet been validated for CDR conditions.[109] As another example, the net climate benefit of coastal wetland restoration – considering, for example, non-CO_2 greenhouse gases and other carbon processes – is difficult to measure and poorly constrained across locations.[110] Research into relevant, fundamental Earth system processes for these methods is ongoing.

Potentials are also dynamic, and assumptions about durability and evolving capacities are not easily reflected in static estimates. These variable temporal dynamics make static estimates of potentials difficult to compare across methods. Climate change may reduce potentials for forestry-based methods over time, for example, by slowing CO_2 uptake rates and increasing the likelihood of carbon storage reversal (i.e. lowering permanence). Along these lines, the choice of underlying scenarios is critical to informing future potentials. For example, all methods that depend on CO_2 diffusion for sequestering carbon (e.g. diffusion in plant leaves for photosynthesis) will exhibit different potentials across scenarios with different, future atmospheric CO_2 levels. Similarly, the potential of wood products depends on assumed trends about population and demand. Durability across methods is a similar concern; soil carbon sequestration and ocean biomass-based methods, for example, may only offer durability at the decadal scale, yet this is poorly reflected in estimates of potentials. Some methods may see their potential fall over time as early deployments saturate the most feasible deployment possibilities – for example, in coastal wetlands. By contrast, methods such as peatland restoration have slow uptake, or even positive emissions, before achieving full potential.

Economic potentials are only available for the subset of methods with sufficient evidence and cost data. Because economic potentials rely on technical potentials and cost estimates as inputs, they inherit uncertainties from each. Economically optimized models naturally assign higher potentials to CDR methods with lower estimated costs, often aligning economic potential with the high end of technical potential ranges. Afforestation, reforestation and forest management is a clear example, with modelled removals in IAM mitigation scenarios (see Chapter 8) reaching up to 8.4 $GtCO_2$ per year by 2050. By contrast, methods with higher estimated costs are often modelled with lower economic potential, even if environmental and sustainability constraints may be less of a concern. DACCS, for example, remains at the low Mt-scale by 2050 in many scenarios.[51]

Box 10.2 Resource competition and interaction with other mitigation options

Removal potentials drawn from bottom-up analyses are not additive across methods. CDR methods compete for resources, both among themselves and with other mitigation efforts that reduce emissions. For example, removal methods relying on geological storage compete with storage demand from industry sectors using CCS. Methods that have high energy demands compete for carbon-free energy, along with electrification efforts (see Figure 10.2). Biomass-based methods – including BECCS, biochar, biomass burial and terrestrial biomass sinking – must compete with bioenergy (without CCS) and biofuel production. For example, Minx et al. (2018)[111] show that bioenergy deployment in 2°C scenarios unfolds at similar levels over the 21st century regardless of the availability of BECCS, indicating strong demand from other sectors. Biomass may also be an important source of carbon for the chemical industry.[112] Importantly, biomass and associated resources like land and water also face demands from agriculture to feed an increasing human population as well as livestock. The multiple demands for biomass illustrate that CDR potentials need to be understood in the context of the scenario considered. For example, a low-population scenario, or one with lower meat demand, may allow for higher biomass availability for CDR. Ultimately, more research into these interacting demands is needed for a holistic assessment of potentials.

While resource constraints may lead to competition between CDR methods or with other mitigation options, some CDR methods – especially those integrated with other industries – can also generate synergies by simultaneously removing CO_2 and avoiding emissions. In the construction sector, for example, durable wood products can substitute for emissions-intensive materials such as steel and cement, and mineral product technologies can produce supplementary material byproducts that partially displace cement. In agricultural applications, both enhanced weathering and biochar (when derived from residues or organic waste) do not compete for land resources with other biomass approaches and, by contrast, can even increase biomass yields. Improved soil fertility also lowers demand for emissions-intensive synthetic fertilizers. While CDR-coupled mitigation activities are likely beneficial from a climate perspective, all such co-benefits must be carefully assessed in lifecycle analyses. CDR and emissions reductions are each managed differently across policy and market contexts, so accurately attributing the climate benefit of each activity is critical to assessing the progress of both.

Examples of competition for resources between CDR and other mitigation activities

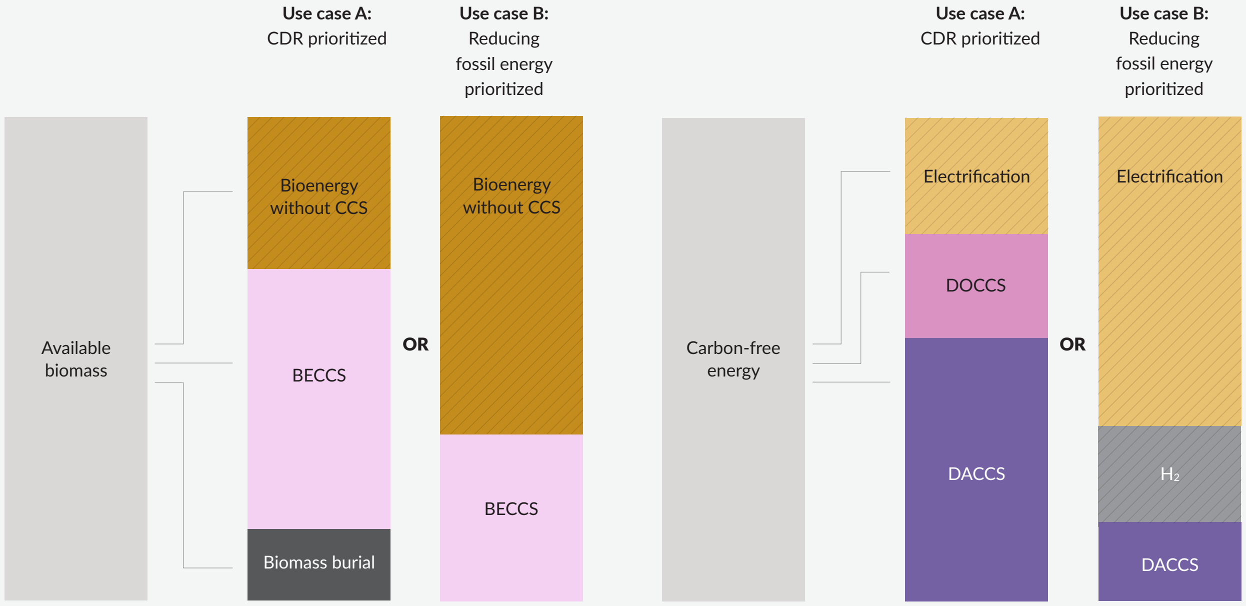

Figure 10.2 *Examples of how two limited resources may be apportioned between CDR and other mitigation activities. Hashed boxes represent non-CDR mitigation activities. For example, biomass can be used for several CDR methods, or for bioenergy without CCS. Similarly, carbon-free energy can be used for CDR pathways with high energy demands, or for activities that displace fossil energy like electrification or production of fuels like* H_2*. Land and water resources face similar trade-offs. Methods shown are illustrative and not comprehensive of competing approaches.*

10.3 Cost drivers and uncertainties

Cost estimates, like potentials, span wide ranges and exhibit significant overlap between methods (see Figure 10.1). For some methods, costs are relatively well-constrained below US\$100/tCO_2. Most methods, however, have broad cost windows: upper limits are well over US\$200/tCO_2, with some even exceeding US\$1,000/tCO_2. Considering variation in the types of cost presented across assessments, we group methods into illustrative bins of likely cost ranges. Forestry-based methods, soil carbon sequestration and biomass burial fall into the lowest range, with estimated costs often <US\$100/tCO_2. From a portfolio perspective, it is important to note that these CDR methods are typically associated with lower levels of permanence than methods employing geological storage, especially under ongoing climate change. Low-to-medium-cost methods – ranging from low US\$10s to several US\$100s/tCO_2 – include coastal wetlands, biochar, BECCS, mineral products and enhanced weathering. Medium-cost methods, ranging from US\$100 to US\$500/tCO_2, include alkalinity enhancement of water bodies and bio-oil storage. High-cost methods, with minimum costs ranging from several US\$100s to >US\$1,000/tCO_2, include DACCS and DOCCS. A few methods – biomass sinking and ocean fertilization – currently defy categorization due to extremely broad cost ranges, from <US$100 to >US$10,000/tCO_2. Similarly, cost estimates for durable wood products are heterogenous and currently not comparable to other methods. For most methods, current prices for CDR fall within these bottom-up cost ranges (see Chapter 4). Average prices for conventional methods fall on the low end of the cost ranges, whereas prices for novel methods are typically on the high end of (or even exceeding) their respective cost ranges. This trend is driven in part by catalytic funding to support novel methods, often in experimental deployments, but may also result from differences between FOAK costs of current deployments, versus Nth-of-a-kind (NOAK) costs represented in bottom-up analyses.

Costs are difficult to compare across methods for several reasons, including inconsistent approaches to cost analysis, variation in costs across different deployments, and scientific uncertainty. Cost analyses are often not advanced enough to distinguish current costs versus projected costs at scale or direct versus levelized costs over time. As well, they may omit information about the underlying dollar value reference year. These considerations are necessary to accurately compare estimates while simultaneously accounting for inflation. For some methods with more focused cost analyses, like DACCS, cost differentiation is possible; however, analyses for most methods do not offer this level of detail. In addition, project boundaries are not always clear in accounting considerations, for example, whether capital or operations and maintenance costs are included, whether co-benefits are included in the valuation, and whether MRV costs are considered. MRV, for example, is estimated to increase the cost of CDR by 10%–30% for most methods, although the variation in reported costs is large and some estimates exceed 50%.[15] Thus, costs presented here offer useful estimates but may not be comparable across methods. Better comparability can be expected in the future with detailed technoeconomic analyses, ideally enabling an assessment of levelized costs for all CDR methods.

Within a method, costs themselves may vary significantly because of different deployment approaches, or spatial and temporal variability. Forest restoration, for example, can be accomplished in different ways and at different costs. For resource-intensive methods like enhanced weathering, transportation distances significantly impact costs. Similarly, the distance from suitable geological formations influences the costs of methods requiring CO_2 storage. The costs of biomass-based, land-intensive methods in particular may vary across contexts, driven by differences in land opportunity costs and local wage levels. Some methods may achieve significantly lower costs by integrating with existing industries, for example DOCCS or OAE with desalination facilities to lower energy demands, and mineral storage with the construction industry to maximize co-product utilization. Costs also vary over time, which is not captured in static estimates. Highly engineered methods like DACCS expect significant cost declines with R&D and scale (see Box 10.3). Some conventional methods also expect costs to fall as MRV procedures mature; however, these same methods may also experience rising costs over time as research evolves scientific understanding or as cheaper locations are exhausted first. Thus, for many methods, the direction of cost trends remains uncertain. Considering current costs, conventional methods are more certain than novel methods but face growing uncertainty with climate change. While afforestation, for example, has been practised for a long time and its costs are relatively certain compared to more novel methods, a warmer climate could dramatically upend its cost effectiveness.

Finally, costs are also affected by scientific knowledge gaps, and uncertainty about the efficiency of some CDR methods. For novel, open-system methods – including enhanced weathering, alkalinity enhancement of water bodies, DOCCS, ocean fertilization and biomass sinking – the quantity of CO_2 durably removed per unit activity is poorly constrained, given ongoing scientific uncertainties related to Earth system processes. The few LCAs and techno-economic analyses available for these methods necessarily rely on assumptions for poorly constrained parameters. These uncertainties feed into additional costs related to MRV requirements.[15] Some conventional methods, like peatland restoration, face similar challenges resulting from possible overlap between CDR and emissions avoidance. LCA approaches to clearly delineate CDR from emissions reductions are still evolving for these methods.[113]

Box 10.3 Cost curves

Projecting future costs for CDR methods is subject to substantial uncertainty. Cost curves, which estimate cost changes over time, depend on assumptions about system boundaries, learning rates, overall system configurations (e.g. cost structure and merit order of the energy system) and regulatory frameworks.[27] Although learning curves are widely used to project future cost reductions, their application to CDR is challenging. They typically rely on historical cost and deployment data to establish a baseline for future learning, which is limited for CDR methods that remain at an early stage of development. They also assume smooth cost reductions as deployment increases, whereas actual cost trajectories may be uneven due to technological, policy or market developments. In addition, learning curves implicitly require a predictable long-run demand trajectory, which does not yet exist for CDR.[27] As a result, cost projections for CDR should be interpreted as indicative rather than predictive. We discuss examples for DACCS and BECCS to illustrate a range of potential cost changes over time.

Several studies have assessed the costs of FOAK and NOAK for DACCS plants, summarized in van der Spek et al. (2025).[51] FOAK plants face high capital costs, limited design optimization and a lack of operational experience, whereas NOAK plants benefit from technological learning and economies of scale. As a result, all DACCS technologies are expected to see significant cost reductions when moving from FOAK to NOAK, though the magnitude varies across technology types. The projections shown in Figure 10.3 depend on assumed capital expenditure (CAPEX) and operational expenditure (OPEX) learning rates, where OPEX learning rates remain especially uncertain given limited current deployment.

Future BECCS costs are shaped not only by technological learning but also by upstream and downstream factors such as feedstock prices and energy market conditions. While modelling studies often apply learning curves to BECCS capital costs, leading to projected cost reductions similar to DACCS,[27] recent expert elicitation shows divergent expectations. Some experts foresee this fall in costs associated with operational improvements; however, others anticipate rising costs driven by increasing feedstock prices and competition for waste materials.[30] Several experts also emphasize that uncertainty widens over time as multiple cost drivers evolve simultaneously, preventing even directional projections of cost evolution.

Published cost trajectories for DACCS technologies

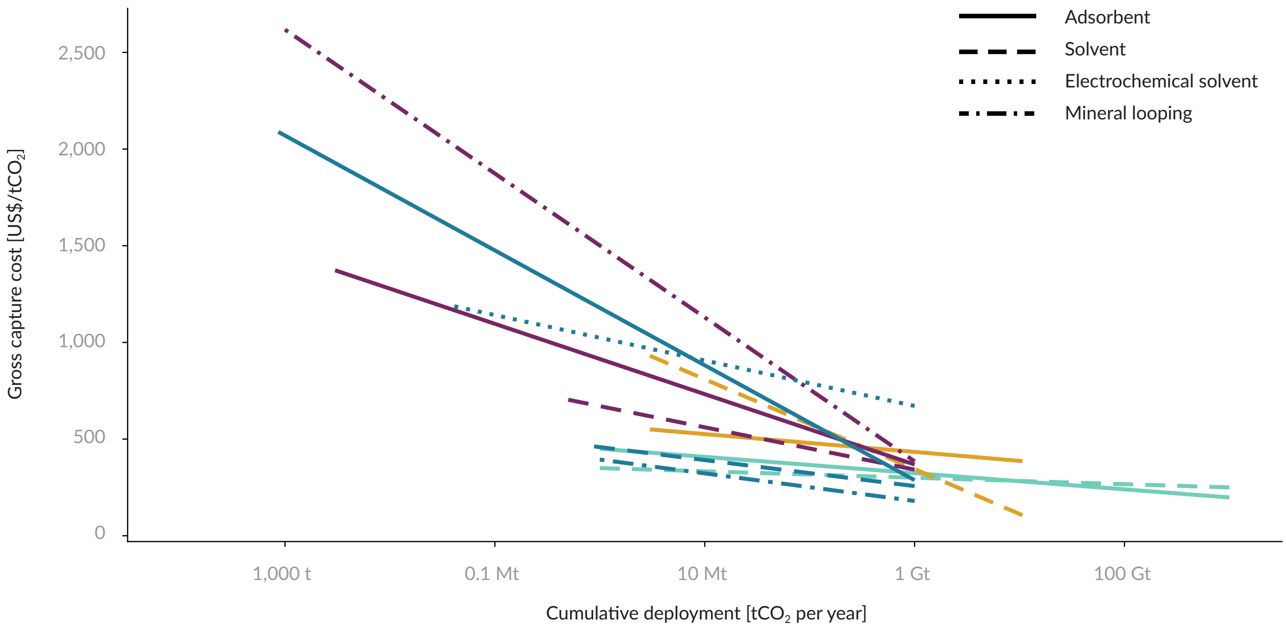

Figure 10.3 *Cost trajectories for DACCS technologies from different studies, harmonized to 2024 levelized cost of gross CO_2 captured. Not all estimates include the cost of CO_2 transport and storage. Figure adapted with permission (CC-BY) from van der Spek et al. (2025); line colors represent individual studies as cited in the original.*[51]

10.4 Consideration of side effects

Side effects refer to any deployment impact other than removing CO_2 from the atmosphere.[5] Side effects – both positive and negative – can significantly shape the scalable potential and economic viability of CDR. They influence where, how, how much and at what cost removal can occur sustainably. Specific side effects and their relative magnitude vary not only across methods but also across implementation modes and deployment contexts. For example, the sustainability of forest-based CDR depends on the suitability of land allocated for deployment; impacts on biodiversity or food production can be either positive or negative, depending on the deployment context.

Many possible negative side effects arise from extensive resource demands and resulting competition with other activities, including land, biomass, water, minerals and energy. Methods that require dedicated land use (e.g. forestry methods, durable wood products, and peatland and coastal wetland restoration) must balance CDR against other priorities. For example, competition with agriculture may lead to conflict about farmers' incomes or food security, thus limiting scalability.[84,114] However, new research also shows that, for example, cultivating crops on wet or rewetted peatlands can lessen economic and ecological trade-offs.[115] For forestry methods – next to the biogeophysical effects that may result from changes in temperature, precipitation, downstream water availability and flood mitigation[116] – there may also be competition with ecosystem conservation.[59,60,117] Biomass-based methods – including BECCS, biochar, bio-oil storage, biomass burial and sinking of terrestrial biomass – face similar challenges related to land-use prioritization and sustainable biomass sourcing.[4] Land-use conflict or competition for biomass (see Box 10.2) could lead to unsustainable sourcing and agricultural expansion, undermining climate benefits through increased fertilizer and water use, land-use change emissions, non-CO_2 GHG emissions and decreased albedo.[29,118] For methods requiring mineral feedstock – including enhanced weathering, mineral products and most alkalinity enhancement applications – expansion of mining also poses environmental concerns, unless waste products can be productively used. Finally, competition for energy with other mitigation activities is particularly challenging for energy-intensive methods like DACCS, DOCCS, some alkalinity enhancement methods and certain mineral storage methods. For example, deployment of DACCS at the one-gigatonne scale could increase global energy demand by around 5%–6%.[51] Across CDR methods, negative impacts related to resource use can be limited for well-designed deployments at limited scales. However, minimizing these side effects becomes more challenging as CDR scales.

Negative side effects may also arise from ecosystem impacts, especially for open-system methods distributed across the land or ocean. Land-based methods, including soil carbon sequestration and those using biomass, may affect water or nutrient cycling at scale.[5] Similarly, enhanced weathering could result in trace metal contamination of soil if feedstocks are not carefully monitored. For ocean-based methods, potential impacts on ocean ecology are a primary concern. Ocean fertilization and sinking of marine biomass must resolve concerns related to nutrient robbing, changes to ecological structure and deep ocean deoxygenation.[16,43,119] Nutrient robbing is of particular concern, as it may reduce the net CDR efficiency and amplify climate stressors on marine biota.[79] OAE and DOCCS may have different impacts on marine species, although these are expected to be more localized.[120,121]

At the same time, certain methods can yield positive impacts, or co-benefits, if carefully implemented. Reforestation, and peatland and coastal wetland restoration can provide multiple ecosystem services and support increased biodiversity, especially when using native and diverse plant species in places and where the historic land cover is compatible with a targeted intervention – such as reforestation in degraded ecosystems that were historically forest lands. Some methods can mitigate climate-related risks, making them valuable components of ambitious mitigation portfolios; for example, peatland and coastal wetland restoration may prevent flooding, reduce erosion and – in the case of peatlands – mitigate fire risk.[84,122] Soil-based methods like soil carbon sequestration, enhanced weathering and biochar can improve soil health, enhance crop productivity and reduce emissions of non-CO_2 greenhouse gases.[5] These methods may also offer indirect climate benefits by reducing fertilizer use. Such benefits are important for contextualizing CDR within broader environmental objectives and considering achievable potential deployment.

Ultimately, limiting reliance on CDR methods to well below their maximum potentials preserves flexibility to implement CDR sustainably, maximizing positive side effects and limiting negative side effects. This flexibility diminishes as reliance on CDR grows. Diversified portfolios of CDR methods also help balance potential trade-offs.

Box 10.4 Limitations and knowledge gaps

Knowledge gaps exist across methods, impacting estimates of both CDR potentials and costs. These gaps arise from unique challenges within each method (e.g. scientific uncertainties) as well as from systematic inconsistencies in scope and lifecycle accounting that span methods. While there has been much progress in recent years to reduce uncertainty across methods, assessments still necessarily rely on many assumptions and expert opinions to navigate remaining knowledge gaps. Improved harmonization across assessments and coordinated scientific effort targeting major uncertainties will help fill these gaps and increase confidence in CDR potentials and cost ranges. We list here the most important knowledge gaps distilled from this chapter's assessment:

- Different assessment methods and assumptions about sustainability, project boundaries, lifecycle duration and future trends (e.g. alternative biomass demands) contribute to uncertainties for both potentials and costs. Accurate comparisons will require more harmonized approaches.

- Across methods, MRV protocols are at early stages, and costs remain unclear. Comparable data on MRV costs are lacking, with only one reference, mainly based on developer estimates.[15] While the importance of this (often overlooked) cost component is recognized, there is a pressing need for more rigorous quantitative comparisons.

- Cost estimates for several CDR methods draw upon a small evidence base, narrowly focused on implementation and opportunity costs and with limited project boundaries. Comprehensive LCA is needed across technologies to enable consistent cost comparisons across methods. Future research should also engage in systematically analysing the sensitivity to different factors to better understand uncertainties, priorities in resolving them and implications for deployment strategies over time. In addition, some cost items are probably underestimated/neglected, such as those referring to insurance and finance. These and other costs may also vary geographically.

- Quantitative and comparable information on potential side effects remains scarce. The mode of implementation, local deployment context, deployment scale and counterfactual impacts of non-deployment are critical considerations when assessing the potential impact of both positive and negative side effects. As a next step, a systematic evaluation of the different approaches to estimating potentials, which was beyond the scope of this chapter, would be needed.

- For many open-system methods that capture or store carbon across broad areas in the terrestrial environment (e.g. peatland and wetland restoration, soil carbon sequestration and enhanced weathering), ongoing scientific questions contribute to uncertainty in CDR potentials. These include, for example, poorly understood impacts on biogeochemical cycling, non-CO_2 GHGs and Earth system processes that impact both carbon storage and scalability. Similarly, ocean-based methods face uncertainty related to impacts on marine ecology and biogeochemistry, as well as challenging MRV. Marine biomass-based methods (ocean fertilization and macroalgae sinking) must assess the impact of downstream nutrient robbing. All marine CDR methods, including OAE and DOCCS, must develop methods to accurately quantify atmospheric CO_2 uptake.

10.5 Outlook

Technical potentials range widely – both across and within CDR methods – from less than 1 $GtCO_2$ per year for some methods to several tens of $GtCO_2$ per year for others. Cost estimates also span wide ranges, from well below US$100/$tCO_2$ to over US$1,000/$tCO_2$. These results, based primarily on peer-reviewed literature, are broadly consistent with other recent CDR cost assessments, such as the recent CO_2RE report.[27] Accurately estimating potentials is difficult for many reasons, including low scientific understanding and data availability, different definitions of potential, inconsistent assumptions about sustainability and durability, or inconsistent distinction between removals and avoided emissions. While these factors also translate into uncertain cost estimates, uncertainty is further exacerbated by variable costs for different approaches within a method (e.g. restoration can be accomplished in different ways, biochar may use different feedstocks) and by different project boundaries. Costs may turn out to be much higher when capital, operations and maintenance, and MRV expenditures are all included. By contrast, some costs may turn out to be lower when co-benefits and co-products are included in the valuation. Because of these variations and uncertainties, a cross-method comparison of estimates for either costs or potentials is challenging. Side effects can significantly shape a CDR method's scalable potential and economic viability, but a consistent assessment is difficult here, as well, because side effects vary across deployment scenarios and scale. Limiting reliance on CDR methods to well below their maximum potentials preserves flexibility to implement CDR in ways that are environmentally responsible and socioeconomically sustainable. Diversified portfolios of CDR methods also help balance trade-offs.

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