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Bishop, G., Duffy, C., Berndes, G. et al (2026). Cascading wood use into bioenergy with carbon capture and storage ensures continuous and enduring temperature reduction. *Communications Earth and Environment*, 7(1).
<http://dx.doi.org/10.1038/s43247-026-03333-1>

N.B. When citing this work, cite the original published paper.

<https://doi.org/10.1038/s43247-026-03333-1>

Cascading wood use into bioenergy with carbon capture and storage ensures continuous and enduring temperature reduction



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Bioenergy with carbon capture and storage (BECCS) is a key component of pathways to net zero, yet potential interactions with forest carbon dynamics, cascading wood strategies, and progressive decarbonisation and CCS deployment are poorly represented in assessments. Here, using dynamic life cycle assessment, we explore these factors for sawmill residue-derived BECCS value chains over long, yet flexible, time-horizons. BECCS improves the climate performance of bioenergy and consistently delivers long-term global cooling, even in a fully decarbonised economy where substitution benefits cease, provided forest carbon stocks are maintained. Cascading wood use delivers greater near-term cooling via product substitutions compared to direct diversion to bioenergy, and provides temporary carbon storage complementing later deployment of permanent carbon storage via BECCS. Without cascading use, unharvested forests can deliver stronger near-term cooling than direct diversion to bioenergy, even with full BECCS deployment. However, the sink strength diminishes as forests mature, and sequestered carbon may be vulnerable to disturbances such as wildfire. Crossover points highlight the critical role of cascading wood use coupled with BECCS to ensure continuous and enduring cooling effects. Transferring biogenic carbon from forests to geological stores, via multiple uses, is likely to enhance the longevity and resilience of carbon dioxide removal in a rapidly warming world.

Climate change poses an existential threat, driven largely by unchecked anthropogenic greenhouse gas (GHG) emissions. To prevent the most catastrophic climate change impacts, the global community has agreed to the goal of limiting global warming to well below 2 °C, with efforts to pursue 1.5 °C¹. Achieving this ambitious 1.5 °C target will require global net zero carbon dioxide (CO₂) emissions by around 2050², necessitating both drastic emissions reductions and the deployment of carbon dioxide removal (CDR)

to extract and durably store CO₂ from the atmosphere²⁻⁴. CDR will be essential for offsetting residual 'hard-to-abate' emissions, as well as to facilitate a return from any temporary overshoot of 1.5 °C². The IPCC concludes that CDR is a key component of pathways that limit warming to 1.5 °C or below 2 °C (>67%), regardless of whether global emissions reach near-zero, net zero, or become net-negative².

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Bioenergy is the largest source of renewable energy globally, contributing nearly 55% of renewable energy and over 6% of the total global energy supply⁵. It provides a valuable opportunity to reduce fossil GHG emissions and dependence on fossil fuels, while complementing intermittent renewables like wind and solar. When combining bioenergy with carbon capture and storage (BECCS), it can also deliver CDR, though at the cost of reduced energy output⁶. However, as with other CDR options, BECCS is questioned as potentially delaying transitions away from fossil fuels and deployment faces challenges such as low public understanding, concerns over land-use conflicts and biodiversity impacts, the economic viability of adequate incentives, and ongoing debates over its overall effectiveness^{7–13}.

BECCS and other so-called biological CDR options differ from abiotic CDR options in that the ecosystems that maintain CO₂ removal (through photosynthesis) are themselves carbon pools. There may be a trade-off between the objectives to store carbon in ecosystems and the harvesting of biomass to produce bio-based products¹⁴. A reduction in ecosystem carbon stock can be considered equivalent to carbon emissions, which can compromise the climate benefits associated with BECCS and bio-based products. Conversely, increases in ecosystem carbon storage at the expense of biomass supply mean lost opportunities for substitution and carbon storage in the technosphere and geosphere. While contributions to the debate on the optimal use of forest resources sometimes favour *either* ecosystem carbon storage *or* biomass production^{15,16}, the two objectives are not mutually exclusive and ecosystem carbon storage can both increase and decrease if land management changes in response to increased demand for bio-based products and/or new income opportunities associated with CDR^{17,18}. In this context, the concept climate-smart forestry has been proposed, built around three main objectives: (i) increasing carbon sequestration; (ii) adapting forest management to build resilient forests; and (iii) sustainably increasing forest productivity and incomes¹⁹. Accordingly, research is increasingly focused on regionally appropriate strategies that balance ecosystem carbon storage with the production of biomass for energy and other wood-based products, delivering both climate and economic benefits^{20–24}.

Life cycle assessment (LCA) is a widely used methodology for evaluating the environmental performance of products or systems across their entire life cycle^{25,26}. In the context of BECCS, accurate climate change impact assessment must account for temporal dynamics, such as changes in carbon storage and the evolving performance of both wood-based and competing technologies. However, temporary carbon storage, introduced in forestry and harvested wood product (HWP) cascading strategies, where harvested wood is first utilised in products before eventual conversion to bioenergy, is poorly represented in traditional LCA. Conventional LCA methods typically apply a fixed 100-year time horizon at the impact assessment phase, limiting their ability to: (i) reflect the timing of emissions and removals, resulting in inconsistent temporal boundaries; (ii) capture the climate implications of temporary carbon storage; and (iii) incorporate the progressive deployment of new technologies^{27,28}. Dynamic LCA (dLCA) offers a more nuanced approach by integrating these temporal variations^{27,29}. Although past research has explored the environmental footprint of BECCS^{30,31}, a critical knowledge gap remains in understanding how forest carbon dynamics, cascading use of harvested wood, phased deployment of CCS technologies in bioenergy systems, and wider decarbonisation trajectories interact over time. Simultaneously, integrated assessment models (IAMs), which inform global mitigation pathways, often adopt a simplified representation of BECCS, assuming large-scale cultivation of dedicated energy crops to meet biomass demand³². Such approaches may overlook opportunities to utilise biogenic carbon already present in harvested wood products and residues, as well as the possibility that land management changes to increase the wood supply capacity can result in increased carbon storage in vegetation and soils.

Sawmill wood residues represent a considerable resource and biogenic carbon stock, existing at a global scale that makes their optimal management an important consideration for achieving net-zero targets. Commonly used for bioenergy or material applications, these wood residues are particularly

well-suited for BECCS due to their compatibility with existing bioenergy infrastructure and supply-chains. This study applies a dLCA framework to evaluate the climate impacts of using sawmill wood residues from Sitka spruce (*Picea sitchensis*) for BECCS, addressing key knowledge gaps. We examine: (i) how CCS deployment on bioenergy influences the environmental performance of sawmill wood residues used for either direct conversion to bioenergy or particleboard manufacturing before end-of-life cascading into bioenergy, (ii) the effects of a decarbonising economy on these pathways, and (iii) how the coupled effect of progressive decarbonisation and CCS deployment on bioenergy shapes optimal residue management.

While IAMs are crucial to understand the cumulative climate mitigation potential of BECCS deployment at scale, they inevitably lack detail on the timing and magnitude of emissions and removals for promising cascading value chains over successive harvests and technical transitions. Bottom-up prospective LCA studies can capture such value chain effects, yet are rarely linked to forest carbon dynamics. We address this gap by applying a multi-scale dLCA approach that links detailed time-explicit carbon accounting for multiple BECCS value chains to modelling of forest dynamics (Fig. 1). First, at the technosphere-level, we quantify the discrete climate effects of a single harvest flow of sawmill wood residues converted to bioenergy directly or cascaded via particleboard, with and without CCS, across multiple independent technosphere decarbonisation scenarios (Fig. 1). We then extend this analysis to simulate continuous annual residue flows over time, explicitly incorporating forest carbon dynamics, progressive CCS deployment, and progressive economy-wide decarbonisation (Fig. 1). Finally, we benchmark these BECCS scenarios against a counterfactual scenario in which forest harvesting ceases and ecosystem carbon storage is maximised. This bottom-up integrated, time-resolved approach provides a deeper understanding of how cascading wood use into BECCS can deliver continuous and enduring temperature reductions, whilst elucidating the influence of allocation method needed to link single wood flows to forest carbon dynamics (Fig. 1).

Results

Part One: single technosphere flow

Under the assumption of stable or increasing forest carbon (annual harvest \leq annual growth), our dLCAs reveal that all explored management pathways for a single flow of 1 tC in sawmill wood residues delivered net climate mitigation across a range of decarbonisation scenarios (Fig. 2). However, the magnitude of this mitigation varied considerably across system configurations, with the climate performance of sawmill residue-based bioenergy systems strongly influenced by the inclusion of CCS, the adoption of cascading wood strategies, and the level of decarbonisation.

Implementing CCS on bioenergy consistently and substantially enhanced climate change mitigation benefits. Across individual decarbonisation levels, scenarios with CCS (*b* and *d*) outperformed their non-CCS counterparts (*a* and *c*, respectively) (Fig. 2). For example, under the highest decarbonisation level, direct conversion to BECCS (scenario *7b*) achieved a cumulative impact of -4095 kg CO₂-eq after 100 years, compared with a smaller benefit of around -1013 kg CO₂-eq after 100 years from its non-CCS equivalent (*7a*). Indeed, in all scenarios incorporating CCS, 3300 kg CO₂ per tC in sawmill residues were transferred to geological storage in the year of energy conversion (Fig. 3). Notably, every direct conversion to BECCS scenario (*b*) outperformed all non-CCS bioenergy scenarios (*a*), regardless of the decarbonisation level (Fig. 2). This advantage was maintained despite lower electricity output in BECCS scenarios, which reduced avoided emissions from energy substitution, particularly at lower decarbonisation levels (Fig. 3). These substitution credits were included to reflect the emissions which could have happened in the absence of the studied system (see 'Methods').

The environmental impact of incorporating cascading wood use via particleboard production was highly sensitive to the level of decarbonisation. While particleboard converted to BECCS scenarios (scenario *d*) consistently had greater climate change mitigation than their non-CCS

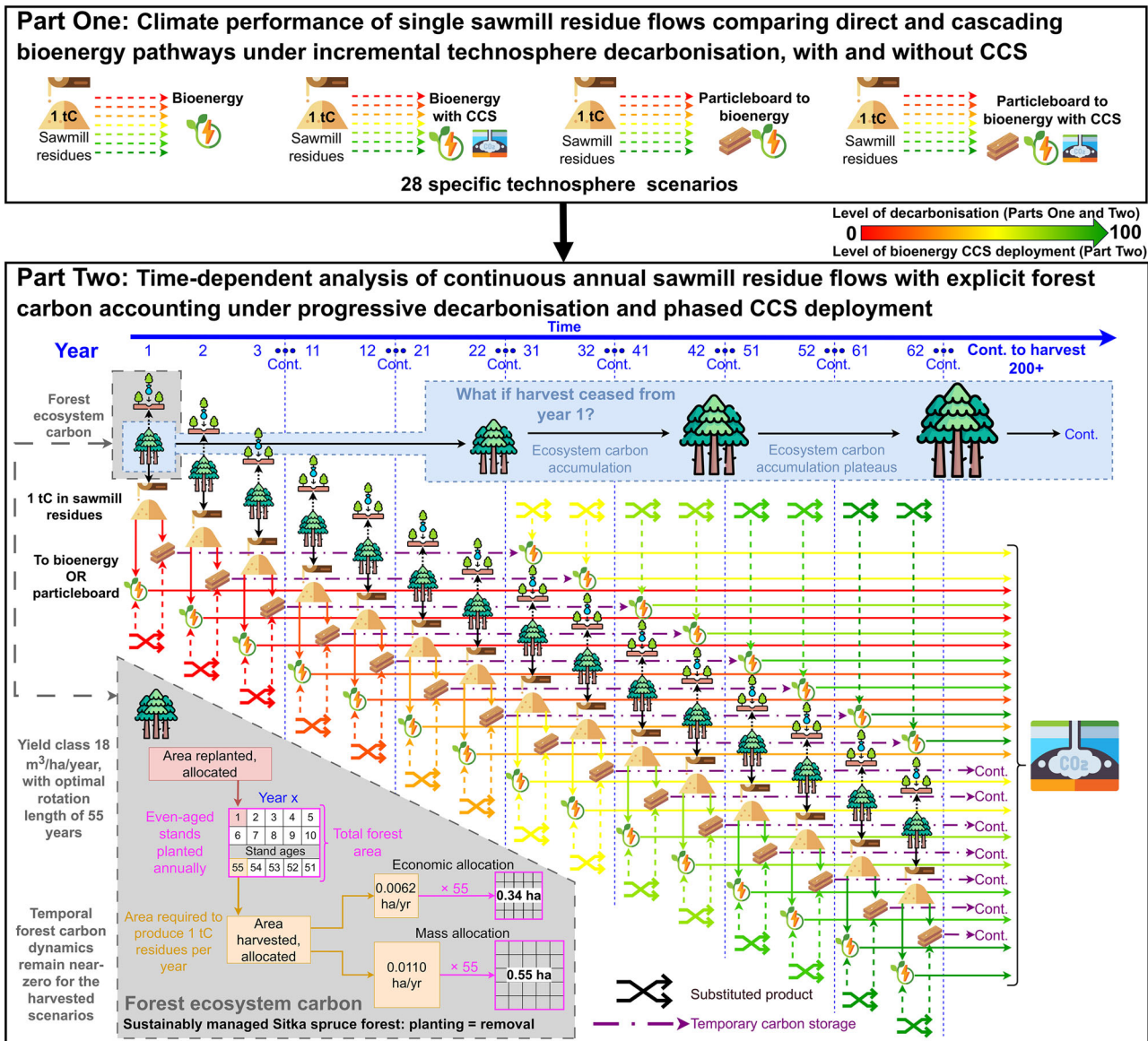
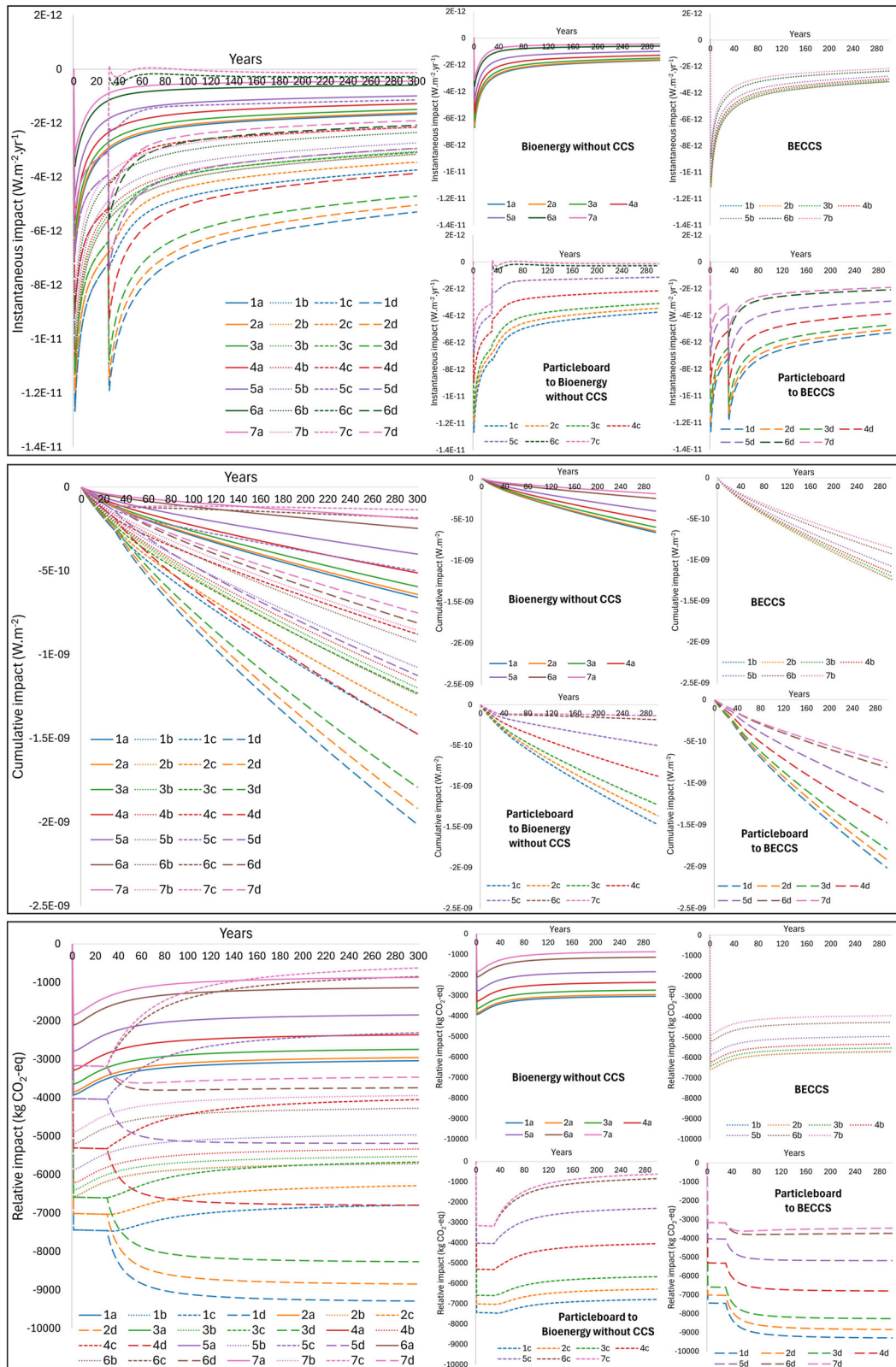


Fig. 1 | Overview of study design. Part One explores four pathways for sawmill residues: (i) direct conversion to bioenergy without CCS, (ii) direct conversion to bioenergy with CCS, (iii) use in particleboard with a 30-year service life followed by conversion to bioenergy without CCS, and (iv) the same cascading use followed by bioenergy with CCS. These produce 28 scenarios in total, represented by the red–yellow–green arrows indicating seven individual increasing levels of decarbonisation for each pathway. Part Two introduces a continuous time dimension, representing the annual flow of 1 tC in sawmill residues under progressive CCS deployment and decarbonisation. The two main utilisation routes modelled are (i) direct conversion of residues to bioenergy with progressive CCS deployment, and (ii) cascading use via particleboard, where carbon is temporarily stored for 30 years before conversion to bioenergy under higher CCS levels than the corresponding direct-use pathway. The forest supplying residues (grey box) was modelled in steady state, with annual removals equal to regrowth across 55 sequential areas of even-aged stands. At year 1, a decision point was introduced: either continue annual harvesting

(providing residues for bioenergy and cascading pathways) or cease harvesting entirely, allowing the allocated forest to mature unmanaged. Scenarios were simulated for over 200 years, tracking changes in above- and below-ground biomass, deadwood, litter, and soil organic carbon. The reference flow of 1 tC per year in sawmill residues was allocated to the harvested forest area using mass and economic allocation factors: 0.0100 ha per year (0.55 ha total) and 0.0062 ha per year (0.34 ha total), respectively. Net annual changes in forest ecosystem carbon were used to build the temporally explicit life cycle inventories for the dynamic LCAs. In harvested scenarios, forest carbon stocks remained near zero in equilibrium, while in unharvested scenarios they increased asymptotically toward a new steady state. Purple arrows indicate temporary carbon storage, the blue line represents time, the blue box denotes the counterfactual unharvested scenario, and the grey box illustrates the allocation approach (which only had an impact on the unharvested scenarios). In Part Two, the red–yellow–green arrows also indicate level of CCS deployment at the bioenergy plants. This figure has been designed using resources from Flaticon.com.

equivalents (scenario *c*) at the same decarbonisation level, the variation across the different levels of decarbonisation was far greater than observed in the direct conversion to bioenergy scenarios (Fig. 2). Under low-decarbonisation conditions, cascading use through particleboard production delivered particularly high climate benefits. For example, scenario *1d*, which represented the lowest decarbonisation level, achieved the largest net negative emissions across all scenarios (–9109 kg CO₂-eq per tC contained

in sawmill residues, after 100 years). As decarbonisation increased, however, the relative benefit of particleboard diminished due to reducing substitution credits as particleboard displaced less carbon-intensive materials (Fig. 3). For instance, under the lowest decarbonisation context, converting sawmill residues to particleboard prior to their use for bioenergy (scenarios *1c* and *1d*) yielded total substitution credits of approximately –7000 kg CO₂ per tC in sawmill residues in the year of particleboard production. In contrast,



under the highest decarbonisation level, these credits declined tenfold to around -700 kg CO₂ (scenarios 7c and 7d). As a result, scenario 7d achieved a lower cumulative impact of -3570 kg CO₂-eq per tC in sawmill residues, after 100 years.

Comparisons between converting sawmill residues directly to bioenergy (scenarios a and b) versus cascading by incorporating them into

particleboard (scenarios c and d) further underscored the importance of the decarbonisation context. Without CCS, cascading use via particleboard (scenario c) tended to outperform direct conversion to bioenergy (scenario a) across the same decarbonisation levels, reflecting the added value of material substitution (Fig. 3). However, when CCS was implemented, the relative advantage shifted. At lower decarbonisation levels, particleboard

Fig. 2 | Dynamic LCA results for the post-harvest technosphere perspective analyses of Part One. Results of the dynamic LCAs conducted using temporally differentiated GHG emissions and dynamic characterisation factors (DCF). *Top panel: Instantaneous Global Warming Impact* shows the radiative forcing in a given year, dynamically calculated using DCFs that track the radiative forcing contributions from all prior emissions while accounting for the decay of atmospheric GHGs over time. *Middle panel: Cumulative Global Warming Impact* sums the instantaneous global warming impact for all preceding years, reflecting the cumulative climate impact. *Bottom panel: dynamic LCA scores* express the cumulative global warming impact relative to the cumulative radiative forcing of a 1 kg CO₂ emission pulse at time zero over the assessment period. The four graphs to the right of each

panel mirror the graph to their left but are organised into grouped scenarios (*a–d*) for easier comparison, using the same axis scale. For scenarios *a* and *b*, a single flow of sawmill residues was processed into pellets and converted directly to bioenergy without and with CCS, respectively. For the cascading scenarios *c* and *d*, a single flow of residues was converted into particleboard with a service life of 30 years, before being converted to bioenergy without and with CCS, respectively. Scenarios 1 through 7 represent increasing levels of decarbonisation, considering factors such as electricity and heat sources, avoided emissions, particleboard substitution credits, and electrification of heavy goods vehicles (Table 2). Here, forest carbon dynamics were considered to be stable and are thus excluded.

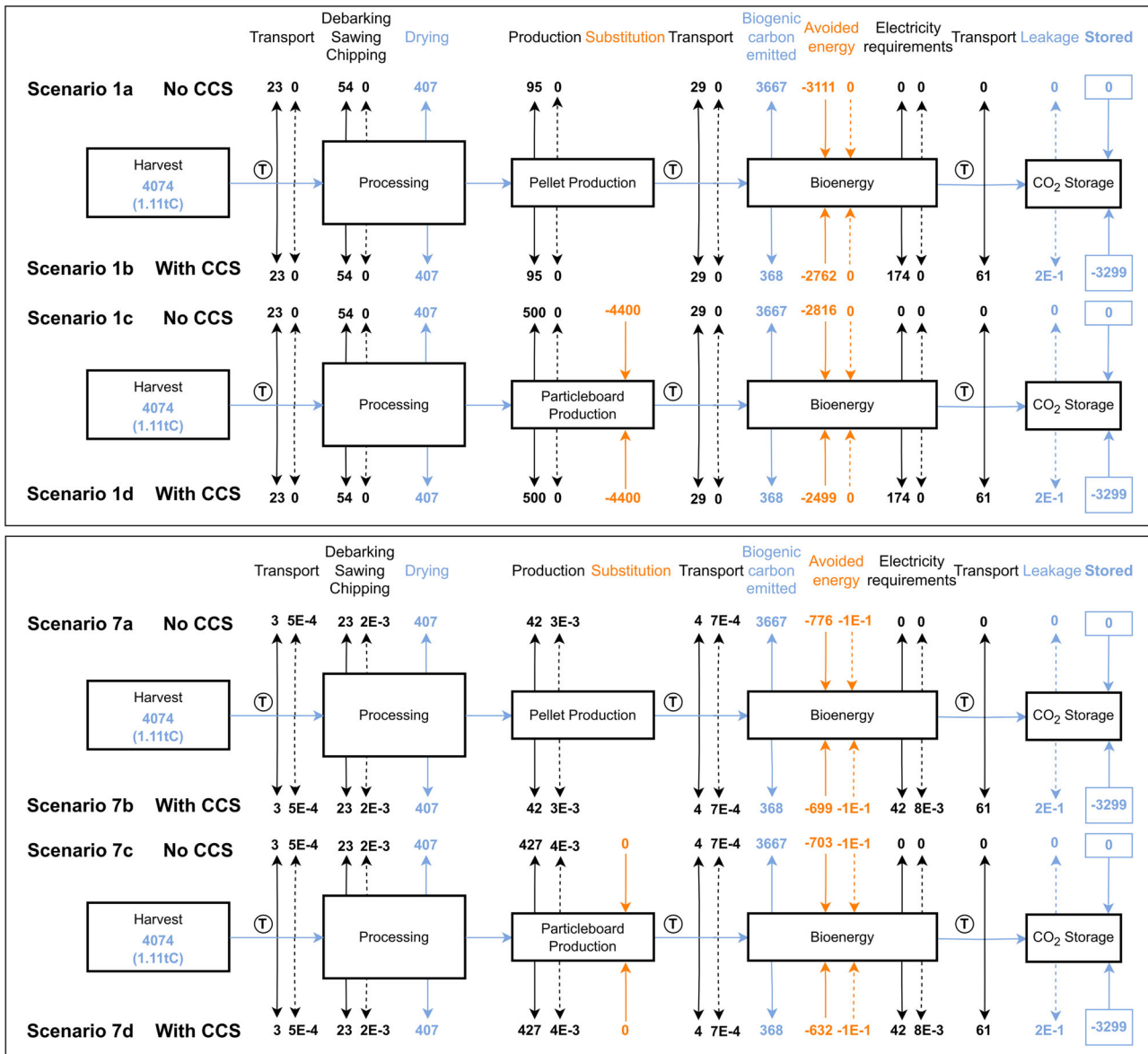


Fig. 3 | Carbon dioxide fluxes at each stage of the bioenergy and BECCS value chains studied for Part One: single technosphere flow. Values indicate carbon dioxide flows measured in kilograms, related to a reference flow of 1 tC contained in sawmill wood residues. Positive values represent emissions, negative values represent either sequestration of biogenic carbon dioxide or avoidance of fossil carbon dioxide emissions. Scenarios *a* and *b* involve processing sawmill residues into pellets for immediate conversion to bioenergy without and with CCS, respectively. In the cascading scenarios *c* and *d*, residues were first processed into particleboard with a service life of 30 years, before being used in bioenergy production without and with CCS, respectively. Scenarios 1 assumed no wider decarbonisation efforts, while

scenarios 7 reflect full decarbonisation, including electricity and heat sources and avoided emissions, particleboard substitution credits, and electrification of heavy goods vehicles (Table 2). Biogenic carbon dioxide flows are shown in blue, fossil carbon dioxide flows in black, and substitution credits in orange, reflecting the emissions that would have happened in the absence of the studied system. Arrows pointing away from the processes indicate emissions, while those pointing toward processes indicate substitution credits. Dotted arrows represent annual carbon storage losses. For this analysis forest carbon dynamics were considered to be stable and were thus excluded. Minor methane and nitrous oxide emissions were also calculated along the value chains but are not included here for clarity. T: transport.

with CCS (scenario *d*) offered far greater climate mitigation than direct BECCS (scenario *b*) (Fig. 3). However, at higher decarbonisation levels, the removal of these substitution credits reduced the advantage of cascading use, and direct BECCS (scenario *b*) became the slightly better performing option from a climate perspective (Fig. 2).

Increasing background decarbonisation consistently reduces the climate change mitigation effects of all value chains (Fig. 2). While this suggests that decarbonising the wider economy will significantly diminish the climate benefits of bioenergy, it is important to note that these scenarios do not account for the time 'bought' by cascading wood use via particleboard to develop and implement CCS in time before that wood is eventually combusted for bioenergy, nor the correlation between decarbonisation efforts and increased levels of CCS deployment. These considerations are accounted for in Part Two.

Part Two: continuous annual flows with progressive CCS deployment

In the second part of the study, we traced annual flows of 1 tC in sawmill wood residues derived from a sustainably managed Sitka spruce forest (yield class 18) (Figs. S4 and S6). We assessed time-dependent, progressive deployment of CCS on bioenergy, alongside decarbonisation trends and annual forest carbon fluxes. Two utilisation pathways were modelled: residues converted to bioenergy with progressive CCS deployment (*BE(CCS_{Prog})*) and residues converted to particleboard, followed by a cascaded end-of-life conversion to bioenergy with progressive CCS deployment (*Cascade(CCS_{Prog})*). Additionally, a scenario was considered in which harvesting ceases and the allocated forest area was left to mature unharvested from year 1 (*Unharvested*) (Figs. S5 and S6).

Although all scenarios resulted in positive environmental outcomes and will ultimately introduce a cooling effect, differences emerged in the comparative climate performances of the *BE(CCS_{Prog})* and *Cascade(CCS_{Prog})* pathways (Fig. 4). Cumulative radiative forcing was consistently more negative for *Cascade(CCS_{Prog})*, initially due to substantial substitution credits from displacing fossil-based materials with particleboard. Temporary carbon storage during the particleboard's 30-year service life also 'bought time' for greater CCS deployment at the time of bioenergy conversion, reducing carbon emissions relative to the same tonne of residues converted directly to bioenergy in the *BE(CCS_{Prog})* scenario (Figs. 5 and S7). As the economy approaches full decarbonisation, substitution credits from both particleboard and bioenergy production ends. Simultaneous (coupled) attainment of 100% CCS ensured sustained net-negative emissions and long-term biogenic carbon storage (Figs. 4 and 5).

The forest area required to supply the reference flow was allocated on both a mass and economic basis, with economic allocation requiring less area (as described in 'Methods'). This greatly influenced fluxes in the unharvested scenarios (Fig. S6). Consequently, comparisons between the environmental potential of leaving the forest unharvested versus the explored harvested value chains depended on the allocation method applied. *Cascade(CCS_{Prog})* consistently outperformed *Unharvested-E* and initially followed a similar trajectory to *Unharvested-M* over the first 40 years, with their relative rankings alternating before *Cascade(CCS_{Prog})* emerged as the clear best-performing scenario from year 43 onwards. In contrast, *BE(CCS_{Prog})* only overtook *Unharvested-E* as the climate-preferable scenario after 63 years and *Unharvested-M* after 137 years (Fig. 4). In terms of cumulative carbon storage, *Cascade(CCS_{Prog})* surpassed *Unharvested-E* and *Unharvested-M* in years 66 and 115, respectively, while *BE(CCS_{Prog})* achieved this after 102 and 147 years (Fig. 5). Notably, the choice of allocation method had negligible effect on the environmental performance of *BE(CCS_{Prog})* and *Cascade(CCS_{Prog})* (Fig. S8).

Additional scenarios explored the *BE(CCS_{Prog})* and *Cascade(CCS_{Prog})* value chains under futures where CCS on bioenergy was either never deployed (*CCS₀*) or fully implemented immediately (*CCS₁₀₀*) (Fig. 4). In these cases, *BE(CCS₀)* never performed better than any unharvested scenario in terms of cumulative radiative forcing, while *BE(CCS₁₀₀)*

Table 1 | Comparison of dynamic life cycle assessment (dLCA) and static life cycle assessment (sLCA) results across scenarios over 100 years

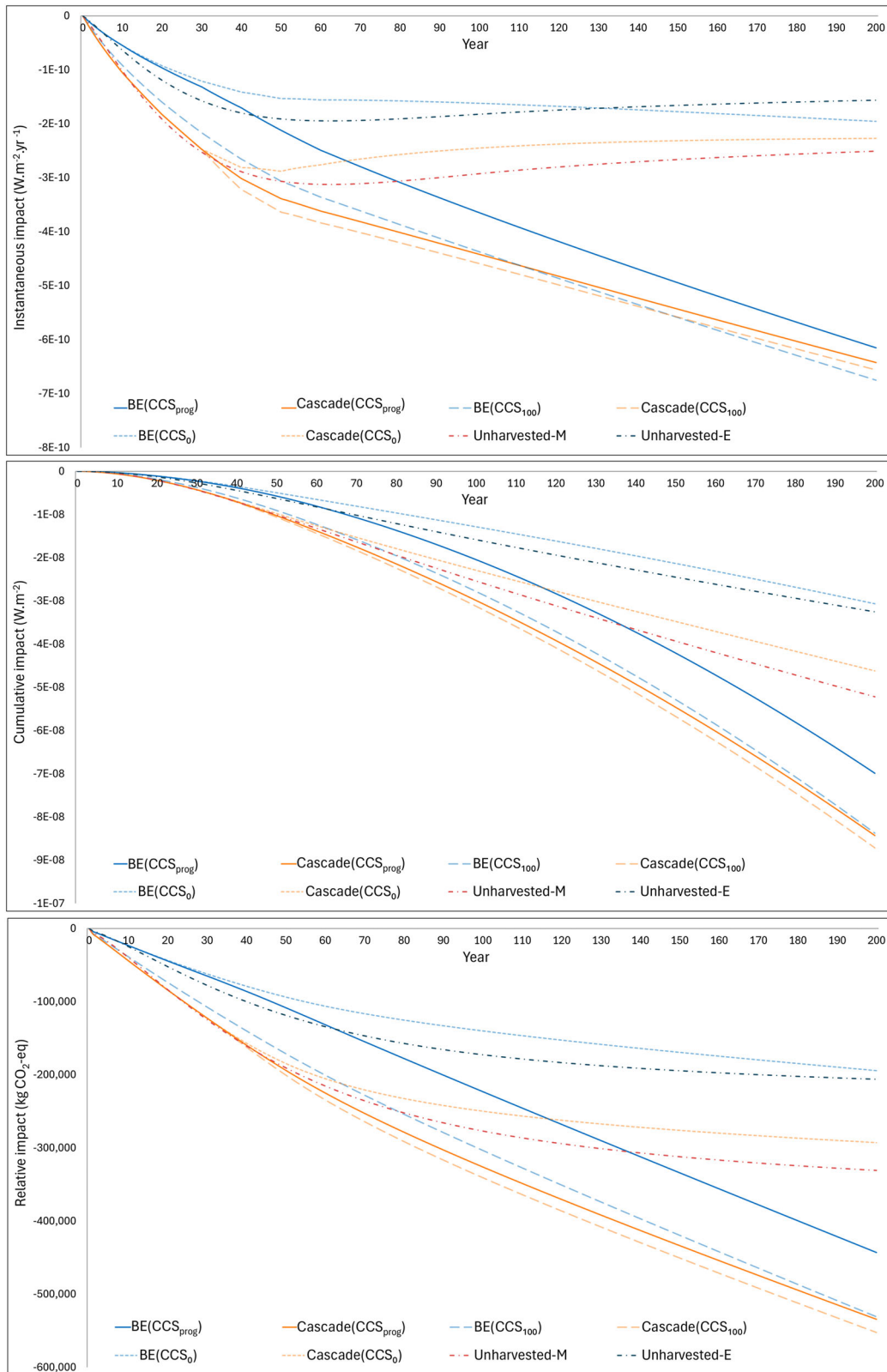
	Scenario	Relative impact (kg CO ₂ -eq.)	
		dLCA	sLCA
Part One	<i>1a</i>	-3176	-3167
	<i>1b</i>	-5861	-5845
	<i>1c</i>	-7056	-6840
	<i>1d</i>	-9109	-9557
	<i>7a</i>	-1013	-1008
	<i>7b</i>	-4095	-4083
	<i>7c</i>	-1237	-516
	<i>7d</i>	-3570	-3604
Part Two	<i>BE(CCS_{Prog})</i>	-223,110	-393,512
	<i>Cascade(CCS_{Prog})</i>	-326,102	-499,590

The dLCA results represent the cumulative global warming impact normalised to the cumulative radiative forcing of a 1 kg CO₂ emission pulse at time zero over the entire assessment period. The dLCA results at year 100 are displayed here. In contrast, the sLCA results use GWP₁₀₀ to quantify the climate change impact of emissions, expressed as the amount of CO₂ that would cause the same radiative forcing over a fixed 100-year time horizon. For Part One, in scenarios *a* and *b*, a single flow of 1 tC contained in sawmill residues was converted into pellets and used immediately in bioenergy without and with CCS, respectively. For the cascading scenarios, *c* and *d*, the same wood residues were converted into particleboard with a service life of 30 years, before being used in bioenergy production without and with CCS, respectively. Scenarios *1* through *7* represent increasing levels of decarbonisation. Zero forest ecosystem carbon flux was assumed. Part Two explored the impact of simultaneous carbon storage implementation and value chain decarbonisation of 1 tC contained in annual flows of sawmill wood residues. Two wood residue utilisation pathways were modelled: bioenergy with progressive CCS (*BE(CCS_{Prog})*) and particleboard use followed by bioenergy with progressive CCS after its 30-year service life (*Cascade(CCS_{Prog})*). Values are shown as kg CO₂-eq.

consistently outperformed *Unharvested-E* and surpassed *Unharvested-M* after 79 years. *Cascade(CCS₀)* always delivered better dLCA results through time than *Unharvested-E* but was never able to surpass *Unharvested-M*. However, *Cascade(CCS₁₀₀)* demonstrated the strongest overall environmental performance, maintaining the lowest cumulative radiative forcing of all scenarios throughout the assessment period (Fig. 4).

A final sensitivity analysis explored the effect of increasing forest yield class from 18 to 22. While this had little impact on ecosystem carbon fluxes in the harvested scenarios, it reduced the accumulated ecosystem carbon in the *Unharvested* scenarios, primarily due to the lower forest area required at higher yield classes (Fig. S9). Under the higher yield class, *BE(CCS_{Prog})* overtook *Unharvested-E* 3 years earlier (year 60) and *Unharvested-M* 16 years earlier (year 121) (Fig. S10). However, yield class had no effect on the crossover points for the *Cascade(CCS_{Prog})* scenario (Fig. S10). Overall, the effects of yield class differences were modest compared to the much larger influence of allocation method choice (Fig. S10).

When comparing dLCA and traditional static LCA (sLCA) results over a 100-year period (Table 1), minimal differences were found in Part One. This reflects the fact that most emissions, removals, and avoided emissions occurred within the first year, limiting the influence of temporal resolution. Minor variations appeared in scenarios *c* and *d*, reflecting a second pulse of emissions or removals occurring in year 31, associated with the bioenergy conversion of the particleboard (Fig. 3). In contrast, substantial differences emerged in Part Two, where emissions and removals occurred dynamically over time (Fig. 5). Here, sLCA produced a more optimistic climate change mitigation outcome compared to dLCA. This divergence arose because sLCA assumes a fixed 100-year time horizon for all emissions and removals, giving equal weight to emissions and removals in year 1 and year 99. dLCA, in contrast, explicitly accounts for the timing of these, assigning greater weight to earlier emissions or removals. As a result, removals occurring later in the assessment period exert a smaller cumulative climate benefit in dLCA than in sLCA.



Discussion

Our findings highlight the interplay between temporary and permanent carbon storage, the importance of cascading wood use strategies, and the resilience of BECCS as a climate mitigation option in a decarbonising world. They also show the strong influence of LCA methodological choices in shaping (perceived) BECCS outcomes. From a technosphere perspective,

BECCS value chains always achieve cumulative negative radiative forcing under a wide range of possible futures, even in a completely decarbonised economy, owing to the climate benefit of transferring most of biogenic carbon into long-term storage, i.e., negative emissions. For the same reason, BECCS always performs better than bioenergy without CCS. Thus, BECCS provides robust climate change mitigation, so long as it does not negatively

Fig. 4 | Dynamic LCA results for Part Two. The figure illustrates the results of the dynamic LCAs conducted using temporally differentiated GHG emissions and dynamic characterisation factors (DCFs). *Top panel: Instantaneous Global Warming Impact* shows the radiative forcing in a given year dynamically calculated using DCFs that track the radiative forcing contributions from all prior emissions while accounting for the decay of atmospheric GHGs over time. *Middle panel: Cumulative Global Warming Impact* sums the Instantaneous global warming impact for all preceding years, reflecting the cumulative climate impact. *Bottom panel: dynamic LCA scores* express the cumulative global warming impact relative to the cumulative radiative forcing of a 1 kg CO₂ emission pulse at time zero over the assessment period. Part Two of the study explored the impact of coupled progressive CCS deployment and wider decarbonisation on the climate change mitigation effects of sawmill wood residues converted to bioenergy, related to annual post-harvest flows of 1 tC in sawmill residues. Key factors considered included the simultaneous progression of CCS deployment to (i) bioenergy and (ii) substituted fossil fuel-based energy sources, electrification of HGVs, and progressive decarbonisation of the

manufacture of the materials that particleboard can substitute for. All these factors evolved in decadal increments (Table 3). The two main sawmill wood residue utilisation pathways modelled included conversion to bioenergy with progressive deployment of CCS ($BE(CCS_{prog})$) and particleboard use followed by a cascading conversion to bioenergy with progressive CCS after a 30-year service life ($Cascade(CCS_{prog})$). Additional scenarios explored the annual $BE(CCS_{prog})$ and $Cascade(CCS_{prog})$ value chains under futures where CCS on bioenergy was either never deployed (CCS_0) or fully implemented immediately (CCS_{100}): ($BE(CCS_0)$, $BE(CCS_{100})$, $Cascade(CCS_0)$, and $Cascade(CCS_{100})$). A scenario in which harvesting ceases and the allocated (on a mass (M) or economic (E) basis) forest was left to mature unharvested from year 1 ($Unharvested-M$, $Unharvested-E$). The temporal assessment in this study was continuous; here, we present results over a 200-year period. Because the dynamic LCA explicitly accounts for the timing of emissions and removals, the climate impacts can be flexibly evaluated at any time horizon (any point) within this period.

affect forest ecosystem carbon storage. This would be the case when, for example, wood is sourced from sustainably managed forests where annual increments are at least equal to harvest offtakes, aligning with recent research suggesting stable global forest carbon sinks, historically³³, and into the future¹⁸. However, realisation of these positive mitigation benefits will depend on proactive measures to guarantee the stability of forest carbon stocks amid escalating climate pressures and increasing biomass demand^{34–36}. Furthermore, carbon storage via CCS is effectively permanent, offering a more resilient and durable option than relying solely on forest-stored carbon, which remains vulnerable to disturbances such as wildfire, windthrow, disease, and insect outbreaks^{37,38}. These risks are expected to increase in frequency and severity with rising global temperatures³⁹, increasing the likelihood of abrupt carbon losses. This emphasises the importance of integrating forest management with long-term carbon storage strategies such as BECCS. In this context, deploying faster growing Sitka spruce (e.g., Figs. S9 and S10) could accelerate harvest rotations and carbon transfer to geological storage, reducing the risk of climate-related disturbance losses.

Considering the progressive deployment of CCS to bioenergy, coupled with wider decarbonisation through time, provides a more realistic and nuanced understanding of the potential contribution of BECCS to achieving mid-century climate goals¹. Our findings highlight the importance of integrating these temporal considerations when assessing the long-term climate implications of bioenergy systems, while also showcasing the importance of cascading value chains. Prioritising an initial use of the biomass in material products such as particleboard offers dual benefits: it temporarily stores biogenic carbon, delaying combustion until CCS infrastructure is more widely available, and generates immediate substitution credits by displacing emissions from alternative, more carbon-intensive materials. As a result, cascading value chains deliver earlier and stronger climate change mitigation than direct conversion to bioenergy pathways. These benefits could potentially be further enhanced through additional cascading uses, multiplying substitution benefits, extending temporary carbon storage⁴⁰, and buying more time for CCS deployment, or transitioning into a fully circular wood system⁴¹.

In this study, allocation was necessary to enable a direct comparison between harvested and unharvested forest scenarios. Allocating an area of forest to the production of sawmill wood residues provides a practical means of linking bioenergy pathways to a defined land area and carbon stock, allowing evaluation against the counterfactual of continued ecosystem carbon storage in an unharvested forest—recently proposed by some authors as providing greater climate mitigation¹⁶. This framing is deliberately conservative, testing bioenergy value chains against an economically unlikely scenario that maximises short-term carbon storage while assuming no indirect land use change, whether negative⁴² or positive⁴³, which could arise in an unharvested scenario if material demand persists. Doing this provides an indication of the worst-case crossover points in time when BECCS begins to outperform an unharvested forest in terms of climate

change mitigation. If the counterfactual were on-site residue decay rather than unharvested preservation, the relative climate performance of BECCS would be even more pronounced. Using both mass and economic allocation further strengthens the analysis by bounding the plausible range of results and explicitly revealing how methodological choices affect outcomes. While mass allocation provides a physically consistent basis for linking carbon flows to land area, economic allocation better reflects the financial drivers of forest harvest decisions. Importantly, the relative performance of the BECCS pathways remain robust to these choices: cascading use strategies match the short-term and exceed the long-term climate benefits of unharvested forest, whereas direct conversion to BECCS requires time to achieve parity. Nevertheless, our analyses indicate that BECCS supplied with wood residues from a sustainably managed forest can ultimately deliver greater cumulative carbon storage and mitigation than leaving the forest unharvested, especially under faster CCS deployment scenarios (Figs. 4 and 5). This points to the need for strategic, long-term policy frameworks and visioning, and immediate investment in expanding BECCS capacity and permanent CO₂ storage infrastructure.

Decarbonisation of the economy reduces the climate benefits of BECCS systems by lowering the emissions intensity of the fossil-based alternatives they displace (Fig. 2). However, it is possible that these substitution benefits may not decline at the same rate as decarbonisation in the economy as a whole, because the use of bioenergy reflects the character of the specific technological systems, which in turn reflects regional resource endowments and the historical build-up of technological infrastructure. In a future where the use of fossil resources is reduced over time, bio-based solutions are likely to be increasingly important where carbon-based solutions are difficult to eliminate. For example, bioenergy could displace residual, emissions-intensive sources (like dispatchable power generation to manage grid intermittency or industrial process heat) rather than substituting an average unit of emissions from the economy⁴⁴. This dynamic could enhance the long-term mitigation potential of BECCS by preserving its substitution benefits for longer than might otherwise be assumed within a deeply decarbonised economy. Nonetheless, an important uncertainty remains as to whether new bio-based production will genuinely displace alternative fossil-based production or instead induce additional energy or material demand. This issue is particularly relevant in the energy sector, where growing global electricity demand, driven by rising demand from data centres, industry, cooling of buildings, and electrification⁴⁵, may limit the extent to which additional low-carbon electricity generation, including BECCS, replaces fossil-based energy. In such cases, substitution benefits may be lower than assumed, though conversely they could be greater if BECCS continues to displace more carbon-intensive energy sources over a longer period as decarbonisation trajectories slow or extend. Further complicating this dynamic, emissions reductions in competing systems may increasingly rely on biomass themselves. For example, the climate advantage of using timber over concrete in construction may diminish as the cement industry adopts CCS and shifts from coal to bio-based fuels^{46,47}, while the

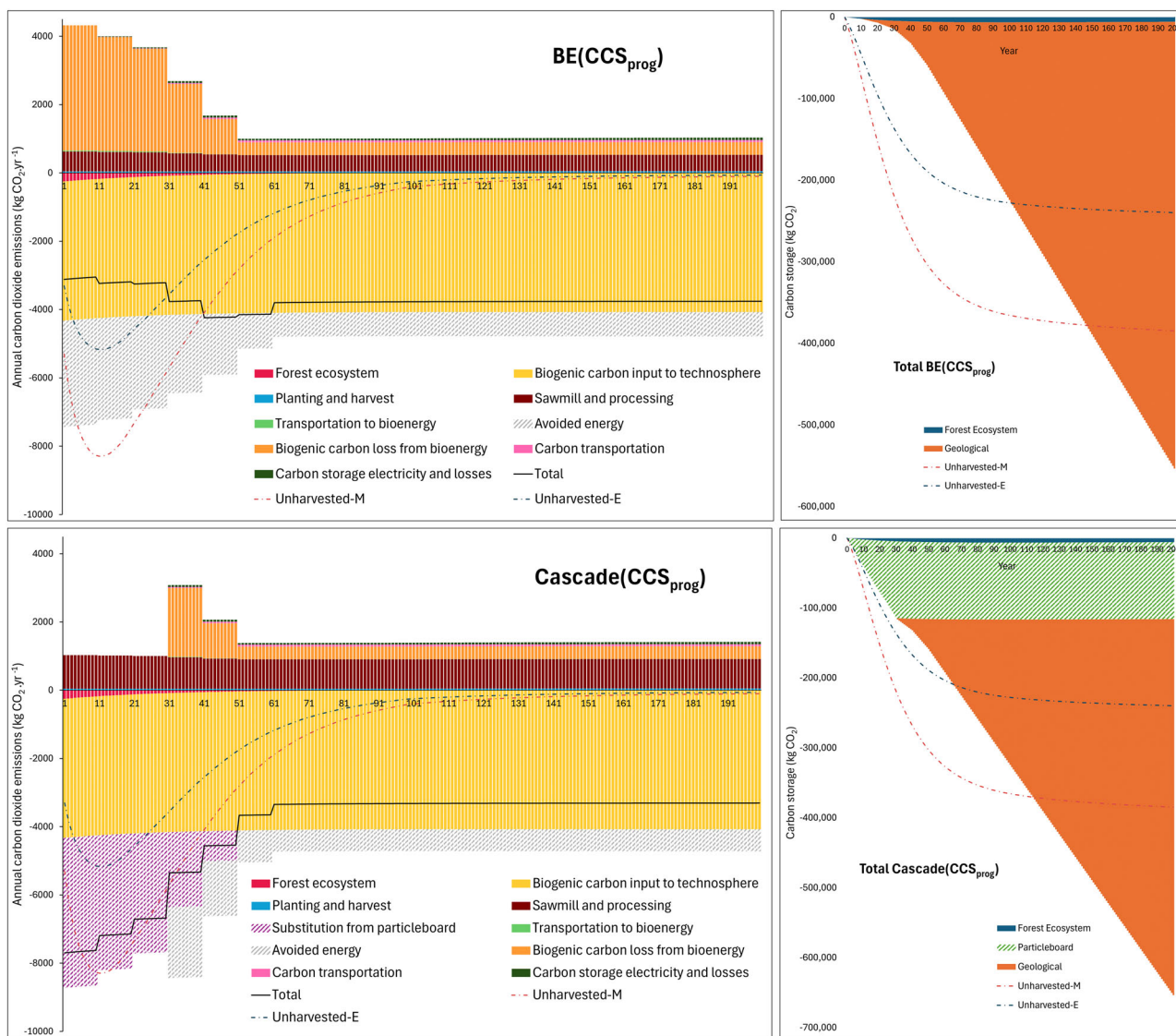


Fig. 5 | Annual CO₂ emissions and cumulative carbon storage for Part Two. Part Two of the study explored the impact of coupled progressive CCS deployment and wider decarbonisation on the climate change mitigation effects of sawmill wood residues converted to bioenergy, related to annual post-harvest flows of 1 t C in sawmill residues. Key factors considered included the simultaneous progression of CCS deployment to (i) bioenergy and (ii) substituted fossil fuel-based energy sources, electrification of HGVs, and progressive decarbonisation of the manufacture of the materials that particleboard can substitute for. All these factors evolved in decadal increments (Table 3). The sawmill wood residue utilisation pathways modelled included conversion to bioenergy with progressive deployment of CCS (*BE(CCS_{prog})*) and particleboard use followed by a cascading conversion to bioenergy with progressive CCS after a 30-year service life (*Cascade(CCS_{prog})*). The top row of the figure displays results for *BE(CCS_{prog})*, while the bottom row shows

Cascade(CCS_{prog}). Panels on the left illustrate annual CO₂ emissions, with coloured bars indicating contributions from individual processes. Striped bars denote substitution credits, representing emissions which could have happened in the absence of the studied system. The solid black line tracks total net annual CO₂ emissions. Minor methane and nitrous oxide emissions are also calculated along the value chains but are not included here, for clarity. Annual forest ecosystem carbon fluxes from scenarios in which harvesting ceases and the allocated forest area is left unharvested from year 1 (*Unharvested-M* and *Unharvested-E*, allocated on a mass or economic basis, respectively) are also shown. Panels on the right display cumulative carbon storage (kg CO₂) over time, partitioned between the forest ecosystem, temporary storage in particleboard (in *Cascade(CCS_{prog})*), and permanent geological storage from CCS. Cumulative ecosystem carbon storage for the *Unharvested-M* and *Unharvested-E* scenarios are included for reference.

incorporation of biochar as a filler material⁴⁸ further blurs substitution boundaries. Consequently, the approach we have taken here to assume diminishing substitution credits is likely pessimistic, and thus conservative with respect to results.

Although the static LCA approach remains closely aligned with UNFCCC GHG inventory reporting conventions and therefore retains direct relevance for climate policy and assessment of Nationally Determined Contributions⁴⁹, dynamic LCA proved critical to more accurately quantify climate forcing through time, particularly when dealing with emissions and removals over longer time horizons and temporary carbon storage. By explicitly accounting for these temporal dynamics, dLCA resolves temporal

inconsistencies that can arise when using static metrics such as GWP₁₀₀²⁷ and reveals how the timing of emissions and removals fundamentally alter the inferred climate benefits of BECCS. In this study, dLCA moderated the apparent climate benefits of BECCS when compared with static LCA results, emphasising the importance of representing temporal factors when evaluating bioenergy and CDR pathways. As discussed above, the allocation of ecosystem carbon storage effects across wood product portfolios can influence perceived climate impacts and is inherently normative. We therefore recommend adopting larger-scale whole forest-wood products system modelling approaches that incorporate fully representative product breakouts, which is more in line with country (national inventory), and

therefore policy, decisions. Nevertheless, by linking time-resolved carbon flows from cascading wood use into BECCS with forest dynamics and evolving decarbonisation scenarios, our study provides a robust framework that connects detailed bottom-up dynamic carbon accounting with the simplified biomass representations typical of IAMs. Incorporating these detailed value chain temporal dynamics into future IAMs would improve the modelling of BECCS and other bio-based mitigation pathways, capturing the effects of carbon storage timing, product cascades, and permanence on long-term climate outcomes.

The segregation of biogenic carbon flows and stores, fossil carbon flows, and displacement credits in this study provides clear insight into distinct drivers through which BECCS systems contribute to climate change mitigation (Figs. 5 and S7). Avoided emissions are often aggregated with emissions in conventional LCAs, obscuring their conceptual and temporal differences and increasing the risk of misinterpreting avoided emissions as genuine negative emissions⁵⁰. Such misunderstandings can lead to overestimating the long-term climate benefits of substitution effects, which are inherently uncertain and transient compared with the durability of carbon removals through BECCS. Recognising these distinctions is therefore critical for informed decision-making. As both the urgency of climate change mitigation and impacts of climate change increase, differentiating the permanence and sources of various carbon stores will be increasingly important for developing effective and sustainable bioenergy strategies.

Social cost of carbon discounting has been used in previous analyses¹⁶. In economics, a discount rate reduces the present value of future costs or benefits, meaning that in a climate context, emissions or removals occurring decades from now are considered less important than those happening today. This framing is ethically contentious⁵⁰, as it can undermine intergenerational equity by giving less weight to the welfare of future generations, and incur intragenerational equity by devaluing the welfare of the most vulnerable communities who bear the greatest climate burdens. Furthermore, discount rates reflect subjective judgments about how much less society values future welfare relative to the present. Nevertheless, as an additional sensitivity analysis, to explore the influence of time preference on our results we applied a 4% social discount rate to both annual emission and impact increments over 100 years. Discounting the annual dLCA results (CO₂-eq.) preserved the physical timing captured by the dynamic characterisation factors while introducing a societal weighting that emphasises near-term impacts. Applying the same procedure to raw annual CO₂ flux yielded a quantity-based metric analogous to a 'carbon rental charge'¹⁶ on additional atmospheric CO₂. We found that discounting reduced the climate benefits of all scenarios, particularly in Part Two where value chains' negative emissions ramp up over time and were thus valued less in present terms. Unharvested forest scenarios also reduced, but showed smaller relative changes as their benefits occur earlier and reduce as they mature over time (Fig. S11 and Table S1). In Part One, discounting notably penalised temporary carbon storage in particleboard followed by BECCS, as substitution credits realised in year 31 were discounted relative to early savings. Conversely, temporary storage without CCS performed slightly better, since its future emissions were weighted less heavily (Fig. S11 and Table S1). Fundamentally, dLCA already accounts for the timing of emissions through its biophysical characterisation of radiative forcing, whereas applying a social discount rate disregards these physical dynamics, undervaluing the long-term benefits of CDR technologies like BECCS and reducing the clear importance of temporary carbon storage, which our results show is significant.

In conclusion, we show that BECCS powered by sawmill residues sourced from sustainably harvested forests delivers robust long-term climate change mitigation across a wide range of decarbonisation futures. However, BECCS infrastructure will realistically take time-likely decades-to be deployed at scale. Cascading wood use via wood products before bioenergy strengthens early- and mid-century benefits by combining material substitution with temporary carbon storage that can "buy time" for the deployment of BECCS technology. At the same time, sustainably managed

forests regrow to replace harvested trees, continuing to draw CO₂ from the atmosphere, creating an enduring carbon removal pathway with a long-term cooling effect. Accounting for temporal dynamics revealed that the timing of emissions, removals, and CCS deployment critically shapes climate outcomes, underscoring the value of dLCA in evaluating bio-based mitigation pathways. Together, these findings reinforce the urgent need for governments to accelerate the deployment of permanent CO₂ storage infrastructure, and incentivise cascading wood use within bioeconomy policies, to deliver the durable and resilient carbon removal essential for climate stabilisation.

Methods

Scope of analysis

This study used dLCA to quantify the global warming impacts of converting sawmill wood residues for bioenergy, either directly or via an intermediate cascading use as particleboard. Scenarios were structured as a series of bio-based value chain pathways, from feedstock harvesting to final emission or geological storage of biogenic carbon. A reference flow of 1 tC in sawmill residues at the point of use served as the basis for comparison across scenarios.

The analysis was divided into two parts. Part One assessed the climate performance of a single flow of 1 tC in sawmill wood residues from a sustainably managed forest, directed to the above bioenergy pathways with or without CCS applied to the bioenergy plant, under discrete (increasing) levels of decarbonisation in the technosphere. Part Two expanded system boundaries to account for continuous annual flows of 1 tC in sawmill residues from a sustainably managed forest, incorporating forest carbon dynamics, progressive deployment of CCS on the bioenergy through time, and progressive decarbonisation of the wider economy, which reduces substitution credits through time. In both parts, time-explicit life cycle inventories and impact assessments captured the temporal profile of emissions, removals, and avoided emissions associated with each BECCS deployment pathway (Fig. 6). Full methodological details for both parts are provided below.

Part One scenarios overview: single technosphere flow

In Part One, the study evaluated the global warming impact of converting sawmill residues to bioenergy under discrete (increasing) levels of decarbonisation. Specifically, scenarios examined the use of sawmill residues for bioenergy production with 0% and 100% CCS, both with and without cascading use in particleboard for 30 years before being converted to bioenergy. Sitka spruce wood was assumed to be sourced from a sustainably managed forest in which carbon stock was constant, and there were no changes in ecosystem carbon attributed to wood residue value chains (details discussed under 'Methods: forest ecosystem carbon'). Part One scenarios are summarised in Table 2.

For all scenarios in Part One, the inventory began in year 1 with the transport of harvested wood from the forest to the sawmill. The sawmill residues were either: (i) converted to pellets for direct bioenergy production without CCS (scenario *a*) or with CCS (scenarios *b*), or (ii) processed into particleboard without and with CCS (scenarios *c* and *d*, respectively). Bioenergy production for scenarios *a* and *b* occurred in year 1, while for scenarios *c* and *d*, bioenergy was derived from the particleboard at the end of its 30-year service life, in year 31. Scenarios 1–7 reflect increasing levels of decarbonisation, encompassing electricity and heat sources and avoided emissions, particleboard substitution credits, and the electrification of heavy goods vehicles (HGV) (Table 2). These dynamic inventories represent 28 distinct scenario combinations, which are described in greater detail in subsequent sections.

In this study, we propose that the energy sources most likely to be substituted by bioenergy are carbon-based, dispatchable options such as natural gas. While natural gas remains a marginal source of dispatchable energy generation⁵¹, it is increasingly being replaced by renewable energy sources⁵². To reflect this, and reflecting the deployment of CCS on bioenergy, energy inputs for each process along the value chain and

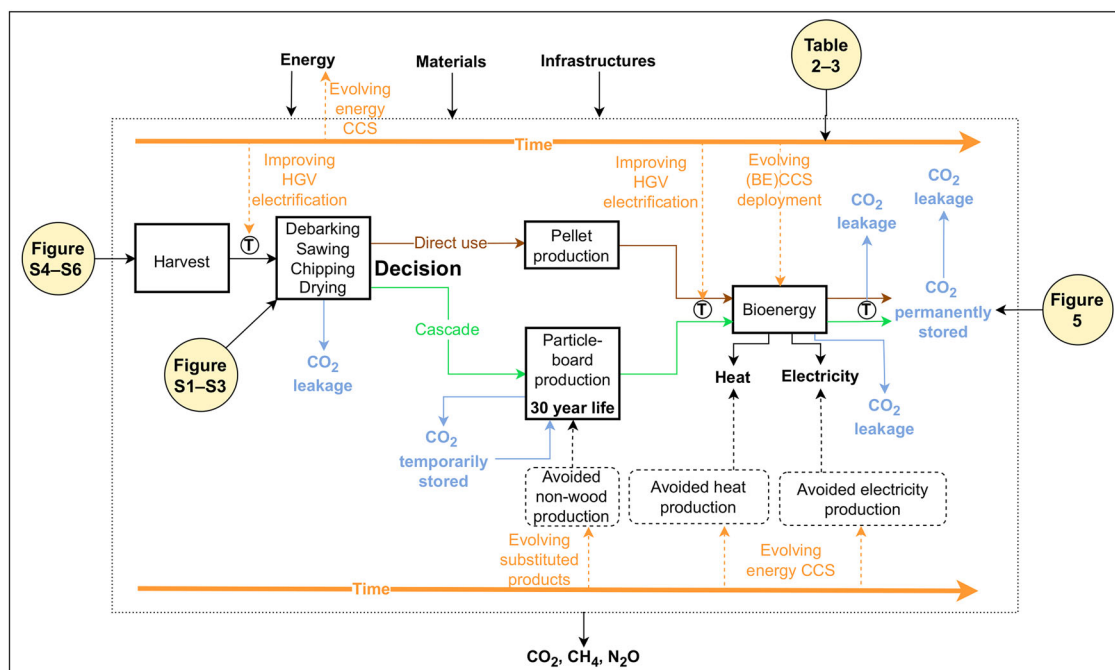


Fig. 6 | Sawmill wood residue conversion pathways to bioenergy: forest dynamics, direct versus cascading scenarios, and decarbonisation. The study framework evaluating the climate outcomes of sawmill wood residues converted to bioenergy, either directly via wood pellets (in brown) or following an intermediate cascading use as particleboard (30-year service life) (in green). Part One of the study considers a single flow of 1 tC in sawmill residues, whereas Part Two considers annual flows of 1 tC in sawmill residues. Both parts assume the wood was sourced from a sustainably managed Sitka spruce forest (yield class 18). Part One assumes zero annual forest ecosystem carbon flux, while Part Two explicitly allocates a corresponding forest area to the annual reference flow, simulating associated temporal forest carbon dynamics (which remain near-zero for the harvested scenarios). An alternative scenario in Part Two compares the climate effect of ceasing harvesting from year one leaving the allocated forest area to mature unmanaged. In Part One, the direct and cascading bioenergy conversion pathways are assessed with and without carbon capture and storage (CCS), across seven discrete scenarios of incremental

decarbonisation levels in the technosphere (28 scenarios in total; Table 2). Part Two conducts a continuous, time-dependent analysis of these pathways, incorporating progressive decarbonisation and phased CCS deployment on bioenergy plants (Table 3). Key decarbonisation variables in both parts (in orange) include reductions in substitution benefits of avoided fossil-based sources and increased electrification of heavy goods vehicles (HGV) used to transport biomass, modelled either as fixed levels (Part One, Table 2) or in decadal increments (Part Two, Table 3). Transportation burdens (T) are indicated within the value chain. In both parts, time-explicit life cycle inventories and dynamic impact assessments track the temporal profile of emissions, removals, and avoided emissions, including the fate of the 1 t of biogenic carbon (in blue), which was ultimately either released to the atmosphere (CO₂ leakage) or permanently stored in geological reservoirs via CCS. Dotted line boxes and arrows denote substitution credits, representing emissions which could have happened in the absence of the studied system. Full methodological details for both parts are provided below.

the marginal energy sources avoided by bioenergy production were modelled based on natural gas with an evolving rate of CCS application over time to represent a decarbonising future (Table 2). This approach prevents asymmetric bias in foreground (bio-based value chains) and background system (wider economy) rates of decarbonisation, and provides a robust proxy for multiple possible wider decarbonisation pathways. Further energy-related and other inventory considerations details are presented throughout the 'Methods'.

Part Two scenarios overview: continuous annual flows with progressive CCS deployment

Part Two expanded on Part One by modelling continuous annual flows of 1tC contained in sawmill wood residues, converted either directly to bioenergy via wood pellets ($BE(CCS_{prog})$) or via an intermediate cascading use in particleboard with a 30-year service life ($Cascade(CCS_{prog})$) (Fig. 6). Scenarios incorporated the progressive deployment of CCS on bioenergy plants and the gradual decarbonisation of the wider economy, both modelled in decadal increments (Table 3). Year 1 in these scenarios corresponds approximately to calendar year 2025.

Wood residues were sourced from a sustainably managed Sitka spruce forest (yield class 18 m³ ha⁻¹ year⁻¹), where annual harvest equalled annual growth. A forest area corresponding to the total sawmill input was calculated, with a proportion of this area allocated to the reference flow based on the ratio of sawmill wood residues to sawnwood output, using either mass (M) or economic (E) allocation. As this was a sustainably managed forest,

the net annual forest ecosystem carbon flux in harvested scenarios was near-zero (Figs. S4 and S6). For comparison, an additional scenario explored the effect of ceasing harvesting from year 1, e.g., due to a sawmill closure, allowing the allocated (formerly managed) forest area to mature unmanaged (*Unharvested-M*, *Unharvested-E*) (Figs. S5 and S6; see 'Methods: forest ecosystem carbon'). Allocations were also applied to some sawmill processes, but had negligible influence on environmental outcomes (Fig. S8; see 'Methods: allocation').

In addition to the primary scenarios, sensitivity analyses examined $BE(CCS_{prog})$ and $Cascade(CCS_{prog})$ pathways under alternative CCS futures: one with no CCS deployment (CCS_0) and one with immediate full CCS deployment from year 1 (CCS_{100}) [$BE(CCS_0)$, $BE(CCS_{100})$, $Cascade(CCS_0)$, $Cascade(CCS_{100})$]. All other parameters remained modelled as before (Table 3). A final set of sensitivity analyses assessed the effect of increasing forest yield class from 18 to 22 on the environmental performance of the primary scenarios (see 'Methods: forest ecosystem carbon').

Full methodological details, including temporally explicit emissions, removals, and avoided emissions for all processes, are provided throughout the 'Methods'.

Temporally differentiated inventory

In this section, we detail the annual calculations of emissions and removals for each GHG, ensuring a year-by-year accounting that reflects temporal variation in climate forcing throughout the study period. The life cycle phases considered in the main results are described below.

Table 2 | Key parameters relating to Part One scenarios

Scenario	Particleboard first?	CCS on bioenergy?	Marginal electricity	Marginal heat	Particleboard substitution value from non-wood products (kg C/kg C)	HGV electrification (%)
1a	N	N	Natural gas 0% CCS	Natural gas 0% CCS	-	0
1b	N	Y	Natural gas 0% CCS	Natural gas 0% CCS	-	0
1c	Y	N	Natural gas 0% CCS	Natural gas 0% CCS	1.20	0
1d	Y	Y	Natural gas 0% CCS	Natural gas 0% CCS	1.20	0
2a	N	N	Natural gas 10% CCS	Natural gas 0% CCS	-	20
2b	N	Y	Natural gas 10% CCS	Natural gas 0% CCS	-	20
2c	Y	N	Natural gas 10% CCS	Natural gas 0% CCS	1.08	20
2d	Y	Y	Natural gas 10% CCS	Natural gas 0% CCS	1.08	20
3a	N	N	Natural gas 20% CCS	Natural gas 10% CCS	-	40
3b	N	Y	Natural gas 20% CCS	Natural gas 10% CCS	-	40
3c	Y	N	Natural gas 20% CCS	Natural gas 10% CCS	0.96	40
3d	Y	Y	Natural gas 20% CCS	Natural gas 10% CCS	0.96	40
4a	N	N	Natural gas 50% CCS	Natural gas 20% CCS	-	80
4b	N	Y	Natural gas 50% CCS	Natural gas 20% CCS	-	80
4c	Y	N	Natural gas 50% CCS	Natural gas 20% CCS	0.60	80
4d	Y	Y	Natural gas 50% CCS	Natural gas 20% CCS	0.60	80
5a	N	N	Natural gas 80% CCS	Natural gas 40% CCS	-	100
5b	N	Y	Natural gas 80% CCS	Natural gas 40% CCS	-	100
5c	Y	N	Natural gas 80% CCS	Natural gas 40% CCS	0.24	100
5d	Y	Y	Natural gas 80% CCS	Natural gas 40% CCS	0.24	100
6a	N	N	Natural gas 100% CCS	Natural gas 80% CCS	-	100
6b	N	Y	Natural gas 100% CCS	Natural gas 80% CCS	-	100
6c	Y	N	Natural gas 100% CCS	Natural gas 80% CCS	0.00	100
6d	Y	Y	Natural gas 100% CCS	Natural gas 80% CCS	0.00	100
7a	N	N	Natural gas 100% CCS	Natural gas 100% CCS	-	100
7b	N	Y	Natural gas 100% CCS	Natural gas 100% CCS	-	100
7c	Y	N	Natural gas 100% CCS	Natural gas 100% CCS	0.00	100
7d	Y	Y	Natural gas 100% CCS	Natural gas 100% CCS	0.00	100

For scenarios a and b, a single flow 1C of sawmill residues were converted into pellets and used directly in bioenergy without and with CCS, respectively. For the cascading scenarios c and d, a single flow 1C of residues were converted into particleboard with a service life of 30 years, before being used in bioenergy production without and with CCS, respectively. Scenarios 1–7 relate to increasing levels of decarbonisation, including electricity and heat sources and avoided emissions, particleboard substitution credits, and heavy goods vehicles (HGV) electrification. For Part One a net-zero forest ecosystem carbon flux was assumed. Energy burdens and sources avoided by bioenergy production were modelled as natural gas, incorporating evolving CCS rates over time to represent a decarbonising future. Each kilogram of carbon contained in the particleboard was assumed to substitute an emission of 1.2 kg of carbon from non-wood products⁶³. To account for future decarbonisation trends in non-wood product sectors, substitution credits were dynamically adjusted using the same decarbonisation rate as electricity. Y: Yes; N: No.

Table 3 | Key dynamic parameters relating to the Part Two scenarios

Parameter	Time period						
	Years 1–10 2025–2034	Years 11–20 2035–2044	Years 21–30 2045–2054	Years 31–40 2055–2064	Years 41–50 2065–2074	Years 51–60 2075–2084	Years 61+ 2085+
Rate of bioenergy CCS deployment (%)	0	10	20	50	80	100	100
HGV electrification (%)	0	20	40	80	100	100	100
Marginal electricity	Natural gas with 0% CCS	Natural gas with 10% CCS	Natural gas with 20% CCS	Natural gas with 50% CCS	Natural gas with 80% CCS	Natural gas with 100% CCS	Natural gas with 100% CCS
Marginal heat	Natural gas with 0% CCS	Natural gas with 0% CCS	Natural gas with 10% CCS	Natural gas with 20% CCS	Natural gas with 40% CCS	Natural gas with 80% CCS	Natural gas with 100% CCS

Parameters develop through decadal time periods, capturing technological advancements and decarbonisation. The timing of bioenergy production (and thus bioenergy CCS deployment rate) depends on the use of the wood residues. In the BE(CCS_{Prod}) scenarios, bioenergy conversion of the annual 1 tC wood residues occurs in the same year as the harvest, while in the Cascade(CCS_{Prod}) scenarios, the annual flow of 1 tC wood residues is converted to particleboard with a 30-year service life before bioenergy conversion. Flows of material not sent to BECCS are sent to bioenergy without CCS, with a slightly greater electricity output. Energy burdens and sources avoided by bioenergy production were modelled as natural gas, incorporating evolving CCS rates over time to represent a decarbonising future. Each kilogram of carbon contained in the particleboard was assumed to substitute an emission of 1.2 kg of carbon from non-wood products⁶⁵. To account for future decarbonisation trends in non-wood product sectors, substitution credits were dynamically adjusted using the same decarbonisation rate as electricity.

Allocation

The reference flow for this study was defined as 1 tC contained in wood residues (specifically woodchips and sawdust) exiting the sawmill. As the analysis focused on a fraction of total sawmill output, allocation was required to attribute required forest area and some sawmill processing emissions to this reference flow. Two allocation methods were applied: mass allocation and economic allocation, based on the relative shares of residues and sawnwood in total sawmill output.

Mass allocation. Incoming wood was assumed to have a carbon content of 50% in its dry matter and a moisture content of 92% on a dry basis (48% wet basis)⁵³, meaning 3.84 t of wet biomass were required to deliver 1 tC of residues. Based on mass flow data from Forster et al.²⁰, woodchips and sawdust account for 42% of total sawmill output by mass, corresponding to 1.38 tC (5.29 t wet biomass) of sawnwood for 1 tC of residues produced. A 42% mass allocation factor was therefore applied. See Figs. S2 and S3 for full mass flows.

Economic allocation. Using the same mass flows, residues were valued at €46 per tonne⁵⁴ and sawnwood at €94 per tonne⁵⁴, generating €176.64 for residues and €497.06 for sawnwood per 1 tC of residues. Using this ratio resulted in a 26% economic allocation factor (Figs. S2 and S3).

While allocation method choice had a large influence on dLCA results in the *Unharvested* scenarios (by determining the comparable area of forest), its effect was negligible in harvested scenarios where it involved relatively minor emissions differences (Fig. S8).

Sawmill and processing

The processing stages included transportation to the sawmill, debarking, sawing, chipping of larger residues, drying of residues, and subsequent conversion into pellets or particleboard. No material losses were assumed at any stage, other than a fraction of incoming wood explicitly used for drying. Emissions for these activities were sourced from the ecoinvent 3.9.1 database⁵⁵ and adjusted accordingly. Electricity-related burdens were time- and scenario-dependent, reflecting progressive decarbonisation (Tables 2 and 3). Further details of processing, including process and mass flows, are provided in Figs. S1–S3.

Initial transport from forest to sawmill was modelled as 60 km by progressively electrified heavy goods vehicles (HGVs), incorporating both time-dependent energy emissions and reduced battery-related burdens⁵⁶ (Tables 2 and 3).

The incoming biomass comprised material destined for residues and sawnwood, but also some bark. The first processing stage of processing involved removing this bark, which later gets used as a fuel to dry the sawmill outputs. Bark accounted for 10% of the total mass input into processing²⁰. Back-calculating from our reference flow, total incoming bark equalled 1.01 t wet biomass (0.26 tC) for a total throughput of 10.14 t of wet biomass (2.64 tC) arriving from the forest (Figs. S2 and S3). Under mass allocation, 0.11 tC (0.43 t wet biomass) of bark was attributed to the reference flow, while economic allocation assigned 0.07 tC (0.27 t wet biomass). Debarking emissions⁵⁵ were distributed accordingly. Note, the mass of allocated bark was also relevant for the estimation of forest area requirements (see 'Methods: forest ecosystem carbon').

Following debarking, 9.13 t of wet debarked biomass (density: 529 kg m⁻³)⁵⁷ yielded 10.00 m³ of sawnwood (1.38 tC), with sawing emissions allocated to the reference flow according to the respective mass or economic ratios (Figs. S2 and S3). Simultaneously, 3.84 t of wet residues (1 tC) were produced, comprising woodchips, sawdust, and larger offcuts (slab/sidings). The larger residues (70% of total residues) were further processed in a chipper to prepare them for pellet or particleboard production (Figs. S2 and S3).

The bark recovered during debarking was used as fuel to dry the residues to 10% moisture (dry basis), as above. This bark was sufficient for the drying process, potentially supplemented with initial air drying to avoid

additional fuel requirements. Complete combustion of bark was assumed, releasing all carbon as CO₂ in the year of processing (Figs. S2 and S3).

In the direct conversion to bioenergy (*a* and *b*) scenarios from Part One (Table 2) and *BE(CCS_{prog})* scenarios in Part Two (Table 3), residues were converted into 2.2 t of pellets (1 tC). In the cascading use (*c* and *d*) scenarios used in Part One (Table 2) and *Cascade(CCS_{prog})* scenarios in Part Two (Table 3), all residues were processed into 2.2 t of particleboard (density: 700 kg m⁻³; volume 3.14 m³), incorporating burdens from formaldehyde resin addition⁵⁵. Both conversion processes were assumed to occur at the sawmill location, with electricity burdens reflecting time- and scenario-dependent decarbonisation (Tables 2 and 3).

Forest ecosystem carbon

The dynamic upstream ecosystem carbon flux attributable to the 1 tC reference flow and associated bark was quantified using two approaches: a post-harvest technosphere perspective assessment in Part One, and a continuous forest-level assessment in Part Two. Details of these methodologies are provided below.

Single flow technosphere—Part One. For Part One, the assessment decouples wood use from forest carbon dynamics, which aligns with the assumption that harvest offtakes were no greater than growth increments in any given year at a regional or national forest level. This approach provides insight into the overall climate change mitigation performance of BECCS systems when: (i) variations in carbon storage across individual forest stands offset each other at the whole-forest level, leading to a stable overall ecosystem carbon pool; (ii) forests are managed sustainably, whereby annual biomass (carbon) removals in harvest never exceed the forest's natural growth and regeneration/restocking capacity, thus maintaining at least a stable overall carbon stock; (iii) the existence of, and carbon dynamics within, commercial forests are pre-determined factors (e.g., because the forests are established on the basis of future revenue from harvests), so that a future-oriented perspective excludes changes in ecosystem carbon stocks from the scope. Thus, the assumption for the post-harvest technosphere perspective analysis was a zero upstream GHG flux for ecosystem management attributable to the reference flow. This means that the reference flow of carbon was assumed not to influence the overall carbon dynamics of the forest, providing a simplified yet robust framework for assessing downstream climate change mitigation of BECCS systems in the technosphere.

Continuous flow forest level—Part Two. In Part Two, a sustainably managed Sitka spruce forest was explicitly modelled to capture annual forest ecosystem carbon dynamics. Here, 'sustainably managed forest' refers to a forest managed such that long-term carbon stocks are maintained at equilibrium, consistent with established forestry nomenclature. This definition pertains specifically to carbon sustainability and does not necessarily imply the absence of other environmental trade-offs, such as biodiversity loss. The forest was assumed to have a yield class (YC) of 18 m³ ha⁻¹ year⁻¹, with growth curves generated using the Carbon Budget for the Canadian Forestry Sector CBM-CFS3 model⁵⁸ for a cool temperate moist climate. Simulations tracked cumulative gross and net annual changes in ecosystem carbon stocks, including aboveground and belowground biomass, deadwood, litter, and soil organic carbon (Figs. S4–S6).

The optimal rotation length for YC18 stands was 55 years. To represent a sustainably managed forest, we modelled a 55 ha system composed of 1 ha of even-aged stands established sequentially (annually) over 55 years. From year 56 onwards, 1 ha of 55-year-old stands were clearfelled and replanted each year, yielding an average harvest of 110.99 tC ha⁻¹ year⁻¹. This continuous rotation established a steady state forest in which annual carbon removals from harvest were balanced by new growth, resulting in near-zero net change in ecosystem carbon stocks (Figs. S4 and S5). The dynamic LCA simulations began in year 101 of the modelled forest, when the forest was in an equilibrium state. At this time, corresponding to year 1 of the study, a

decision point was introduced: either continue annual harvesting (supplying sawmill residues for bioenergy with or without cascading use), or cease harvesting altogether (e.g., due to sawmill closure), allowing the previously sustainably managed 55 ha forest to mature unmanaged (Figs. S4 and S5). Simulations were extended for a further 200 years, tracking net changes in ecosystem carbon stocks through time (Figs. S4–S6 and S9).

The total annual input of 2.64 tC of wood from the forest to the sawmill (for 1 tC residues) (Figs. S2 and S3) required 0.0238 ha to be harvested per year, equivalent to 1.309 ha cumulatively (×55 ha). Allocation of this forest area to the reference flow of 1 tC generated annually in sawmill residues used the above mass and economic allocation factors. Under mass allocation, this equated to 0.0100 ha harvested per year (0.55 ha total) for the annual reference flow; under economic allocation, 0.0062 ha harvested per year (0.34 ha total).

Net annual changes in forest ecosystem carbon stocks were modelled for each scenario to generate the temporally explicit inventory required for the dLCAs (Figs. S4–S6 and S9). In harvested scenarios, net changes in forest ecosystem carbon remained near zero in equilibrium. In *Unharvested* scenarios, forest ecosystem carbon stocks increased over time before reaching a new equilibrium following an asymptotic trajectory (Figs. S4–S6 and S9).

Biogenic carbon harvested from the forest was assumed to be temporarily stored in wood products in the year of harvest, then: (i) released immediately during drying, (ii) emitted during bioenergy conversion either in the year of harvest or after 30 years, (iii) released during minor leakage in CCS processes (see 'Methods: bioenergy'), or (iv) transferred to geological storage and never released (Figs. 5 and 6). Mass balance checks ensured conservation of carbon and prevented double counting.

Sensitivity of different yield class—Part Two. Sensitivity analysis examined the effect of increasing forest YC from 18 to 22 m³ ha⁻¹ year⁻¹, following the same approach described above. The key difference was the optimal rotation length, which decreased from 55 to 35 years for YC22 stands. As before, a sustainably managed forest was created by planting 1 ha of even-aged forest stands annually for 35 years, establishing a 35-ha forest. From year 36, 1 ha was harvested (and replanted) each year, removing 86.71 tC ha⁻¹ year⁻¹. This cycle continued until the year 100.

This sensitivity used the same reference flow of 1 tC contained in sawmill wood residues, maintaining identical system boundaries and substitution assumptions across scenarios. Increased forest productivity reduced only the forest area required to supply this reference flow, influencing the magnitude of associated ecosystem carbon dynamics. Thus, for the 2.64 tC annual input to processing (Fig. S2), the total harvested area was 0.0305 ha per year, equivalent to 1.07 ha cumulatively (×35 ha). Allocated to the annual 1 tC reference flow, this equated to 0.0128 ha per year (0.45 ha total area required) under mass allocation and 0.0080 ha per year (0.28 ha total area required) under economic allocation.

As with YC18, net annual ecosystem carbon changes in harvested scenarios remained near zero post-equilibrium. However, as discussed, the smaller areas required in the higher-yielding YC22 forest reduced the allocated ecosystem carbon accumulation in the *Unharvested* scenario (Fig. S9).

Forestry operations

In Part Two, planting and harvesting burdens encompassed several key activities: clearing brash and mounding (15 machine hours per hectare), seedling planting (at 2500 seedlings per hectare), pesticide application (1 round of 416 grams per hectare), harvester operation (32 machine hours per hectare), and forwarder operation (32 machine hours per hectare)²⁰. Emissions values for these activities were extracted from the ecoinvent 3.9.1 database and adjusted according to the required areas.

Bioenergy

The transportation of wood pellets or particleboard to the bioenergy plant was assumed to cover 150 km, resulting in a transportation burden of 330 t km for the reference flow. Emissions from HGV transport reflected progressive vehicle fleet electrification as detailed in Tables 2 and 3.

The level of BECCS deployment depended on the scenario under consideration (Part One, Table 2) or the timing of bioenergy deployment (Part Two, Table 3). Energy outputs from the bioenergy systems were calculated based on the lower heating value (LHV) of the wood pellets (or particleboard) (Eqs. 1–4)⁵⁹.

$$\text{Bioenergy Electricity (No CCS)} = \text{LHV} \times \eta \quad (1)$$

$$\text{Bioenergy Heat (No CCS)} = \text{LHV} \times \theta \quad (2)$$

$$\text{BECCS Electricity} = \text{LHV} \times \eta \times \eta_{\text{CCS}} \quad (3)$$

$$\text{BECCS Heat} = \text{LHV} \times \theta \times \theta_{\text{CCS}} \quad (4)$$

where the LHV of the wood pellets (at 10% moisture content) was 17.46 GJ tonne⁻¹. η was the share of electricity recovered from the combustion in the bioenergy plant, expressed as a ratio of the LHV value, and was assumed to be 0.33⁵⁹. θ was the share of heat recovered from the combustion in the bioenergy plant, expressed as a ratio of the LHV value, and was assumed to be 0.7⁵⁹. η_{CCS} was the ratio of electricity produced by the BECCS plant, compared with electricity production from the non-CCS bioenergy plant, and was assumed to be 0.39^{59,60}. This resulted in a total BECCS electricity efficiency ($\eta \times \eta_{\text{CCS}}$) of 0.13. θ_{CCS} was the ratio of heat produced by the BECCS plant, compared with heat production from the CHP plant, and was assumed to be 1.23^{59,60}. This gave a total BECCS heat efficiency ($\theta \times \theta_{\text{CCS}}$) of 0.86. The BECCS process allowed for greater overall heat capture than in non-CCS systems⁶⁰.

During BECCS, the carbon capture efficiency was assumed to be 90%. Non-captured carbon (10% from BECCS and 100% from bioenergy without CCS) was emitted as CO₂. Some losses of captured carbon during transportation and storage were accounted for, including pipeline losses of 0.032% of injected carbon in the year of injection⁶¹ and long-term storage losses of 0.005% annually⁶¹. The remaining captured carbon was considered to be sequestered beyond the temporal bounds included in this study.

Carbon captured through BECCS was modelled to be transported 2000 km by ocean, and electricity demands for each step of the CCS process was accounted for⁶¹, including CO₂ liquefaction (125 kWh tCO₂⁻¹), on-sight storage (11.5 kWh tCO₂⁻¹), loading CO₂ onto the ship (0.11 kWh tCO₂⁻¹), discharging CO₂ from the ship (0.12 kWh tCO₂⁻¹), and intermediate storage of CO₂ (8 kWh tCO₂⁻¹)⁶¹. Emissions for these processes were dependant on the scenario-specific or time-specific electricity sources (Tables 2 and 3).

Quantities of useful electricity and heat produced from bioenergy substituted for equivalent quantities of electricity and heat from marginal fossil-fuel sources, based on prevailing conversion efficiencies for the fossil fuels. An efficiency penalty of 7.5% was applied to CCS deployed on natural gas power generation, due to additional energy demands associated with the process⁶². The extent of CCS deployment for the (avoided) marginal electricity and heat production depended on the scenario or timing. To ensure consistent assumptions, marginal electricity CCS deployment was aligned with BECCS deployment, while CCS deployment for marginal heat proceeded at a slower rate²⁰ (Table 3).

Particleboard cascading use

Incorporating a cascading use of the low-value wood allowed for an additional material (and associated production emission) substitution, while delaying bioenergy deployment, thereby 'buying time' for higher levels of CCS implementation in Part Two. In this study we assumed that the intermediate cascading product was particleboard, modelled with a service life of 30 years. Particleboard is an engineered wood product made from wood chips and a synthetic resin, used for its versatility and cost-effectiveness for furniture, cabinetry, construction, and packaging. As a cascading use example of temporarily storing the carbon for an extra 30 years, a simplified assumption was made that the entire volume of the harvested wood product was conserved during service life and passed on to bioenergy.

Each kilogram of carbon contained in the particleboard was assumed to substitute an emission of 1.2 kg of carbon from non-wood products of equivalent utility, based on an average value determined from the review of 51 studies⁶³. Given the reference flow of 1 tC contained in the particleboard, this substituted 1.2 tC of emission. To account for future decarbonisation trends in manufacturing of non-wood products, substitution credits were dynamically adjusted using the CCS deployment rate for electricity (Tables 2 and 3) as a proxy for wider industrial decarbonisation. This approach resulted in substitution credits declining from 1.2 to 0 kg C per kg C, depending on the scenario.

After its 30-year service life, the particleboard was assumed to be cascaded to bioenergy production. Emissions and energy substitution values were calculated as described above. The LHV of particleboard (15.8 GJ tonne⁻¹) was lower than that for wood pellets⁶⁴. However, delayed bioenergy deployment allowed for greater CCS implementation in part two (Table 3).

Dynamic LCA

Dynamic LCA incorporates temporal variations in environmental impacts²⁷. It employs a dynamic inventory to track GHG emissions over time, using dynamic characterisation factors (DCF) to assess the impact of these emissions at each time step. Like global warming potentials (GWPs), the DCF is a radiative forcing-based metric, measured in watts of heat energy trapped per square metre of the Earth's surface per kg emission pulse (W m⁻² kg⁻¹), and is based on the equations and parameters established by the IPCC⁶⁵. Unlike GWPs, which calculate radiative forcing over fixed time horizons, the DCF dynamically estimates radiative forcing by accounting for changes in atmospheric GHG concentrations over time without a fixed time horizon. The DCFs were calculated for each GHG as the absolute GWP at each 1-year time step following an emission. They express the radiative forcing occurring between time $t-1$ and t caused by a pulse emission at time zero (Eq. 5).

$$\text{DCF}_i(t)_{\text{inst}} = \int_{t-1}^t a_i \cdot C_i(t) dt \quad (5)$$

where a_i is the instantaneous radiative forcing per unit mass increase of GHG i in the atmosphere, also called radiative efficiency (in W m⁻² kg⁻¹), and $C_i(t)$ is the atmospheric load of a GHG t years after an emission pulse (in kg kg⁻¹). For this study, DCFs were extracted from the DYNCO2 dynamic LCA software version 2.0⁶⁶. In the software, the residual atmospheric load following a pulse emission of carbon dioxide ($C_{\text{CO}_2}(t)$) was characterised by the Bern carbon cycle climate model⁶⁷, while the atmospheric decay of methane (CH₄) and nitrous oxide (N₂O) was calculated using first-order decay equations.

In simple terms, DCFs measure the amount of heat energy trapped by one kilogram of GHG emission, calculating its warming effect over time. DCFs calculate the impact for each year after the emission occurs, considering how long each GHG remains in the atmosphere. CH₄ and N₂O atmospheric loads eventually decrease to zero, while some CO₂ emissions persist long-term in the atmosphere.

These DCFs were used in combination with a temporally differentiated emissions inventory to calculate the instantaneous global warming impact in a given year t , $\text{GWI}_{\text{inst}}(t)$ [W m⁻² yr⁻¹]. $\text{GWI}_{\text{inst}}(t)$ represents the annual global warming effects in year t , defined as the radiative forcing in year t caused by the portion of all GHG emissions released from year 0 to year t that remain in the atmosphere. Each previous year's emissions contribution was determined by multiplying each of them by their respective DCF, which accounts for the period elapsed between their emission year j and the present year t (Eq. 6).

$$\text{GWI}_{\text{inst}}(t) = \sum_i \text{GWI}_i(t) = \sum_i \sum_{j=0}^t g_i(j) \cdot \text{DCF}_i(t-j) \quad (6)$$

where $g_i(j)$ denotes a GHG emission i in year j (in kg), and $DCF_i(t-j)$ is the DCF of $g_i(j)$ in year t after considering the decay of the GHG in the atmosphere, and $(t-j)$ denotes the time interval between year t and the emission year j .

To illustrate, consider the GWI_{inst} in year 20 ($t=20$). We start by calculating the radiative forcing occurring in year 20 arising from an emission produced in year 19 ($j=19$) using a DCF calculated over a 1-year period, denoted as $DCF(1)$. We then add the radiative forcing from emissions produced in year 18 using $DCF(2)$, and continue this process back to emissions from year 0 characterised by $DCF(20)$. GWI_{inst} in year 20 thus sums up the total radiative forcing occurring from all these emissions, for all GHGs, at year 20. The DCFs reflect the atmospheric reductions of emission i since the emission year j ; the higher the DCF number, the lower the value. A positive GWI_{inst} value indicates net radiative forcing attributable to life cycle GHG emissions, contributing to atmospheric warming in that year. Conversely, a negative value signifies a reduction in net radiative flux, implying a beneficial reduction of global warming in that year. Since GWI_{inst} dynamically tracks emissions' warming effects on a yearly basis, it effectively applies an infinite time horizon if the assessment period is extended indefinitely—a capability beyond the traditional use of GWP.

The cumulative global warming impact, $GWI_{cum}(t)$, represents the total sum of GWI_{inst} values calculated for all preceding years (Eq. 7). This metric reflects the overall radiative forcing caused by the studied life cycle GHG emissions over a given time period. Climate neutrality can be assumed to be achieved when the cumulative climate effects calculated using Eq. 7 decrease to zero. Though, the term 'climate neutrality' should be interpreted cautiously, as calculations fail to capture all climate effects, and the term is frequently misused to describe various net zero outcomes^{68,69}. However, in this context, the use of radiative forcing provides a more accurate measure of climate impact compared to a simple GHG balance. The terminology also aligns with that used in European policy frameworks.

$$GWI_{cum}(t) = \sum_{j=0}^t GWI_{inst}(j) \tag{7}$$

To make the dynamic LCA results comparable to those of traditional static LCAs, the results were converted into dynamic life cycle impact assessment scores. The relative impact, in terms of $kgCO_2\text{-eq}$, was the ratio of the life cycle GWI_{cum} over the cumulative impact of a 1 kg CO_2 pulse emission at time zero to the given time horizon TH (Eq. 8). The denominator was calculated from the cumulative forcing of Eq. 5. The TH can easily be adjusted for the dynamic LCA study.

$$LCA_{dyn} = \frac{GWI_{cum}(TH)}{\int_0^{TH} a_{CO_2} \cdot C_{CO_2}(t) dt} \tag{8}$$

Traditional LCA

To compare our dynamic LCA results with traditional static LCAs, Eq. 9 was applied. In static LCA the GWP over a 100-year time horizon (GWP_{100}) is used to assess the impact of GHG emissions. A static LCA sums the emissions of each GHG g_i over a period of 100 years, multiplied by their respective GWP_{100} values. This approach measures the total radiative forcing impact of emissions over a century.

$$LCA_{static} = \sum_i \sum_t g_i(t) \cdot GWP_{100}^i \tag{9}$$

where:

$$GWP_{100}^i = \frac{\int_0^{100} a_i \cdot C_i(t) dt}{\int_0^{100} a_{CO_2} \cdot C_{CO_2}(t) dt}$$

The GWP_{100} metric integrates the radiative efficiency a_i and atmospheric concentration $C_i(t)$ of each gas over 100 years (i.e., the cumulative radiative forcing caused by a pulse-emission for a given GHG over a 100-year time horizon) and compares it with the radiative forcing of CO_2 over the same period. GWP_{100} values considered were 1 for CO_2 , 29.8 or 27.0 for fossil or non-fossil CH_4 and 273 for N_2O ⁷⁰. GWP_{100} values effectively equate to the cumulative sum of the DCFs for i over 100 years divided by the cumulative sum of the DCFs for CO_2 over the same period, as they are both based on established IPCC equations and parameters⁶⁵. Temporary carbon storage of less than 100 years is typically disregarded.

Study limitations

There is inherent uncertainty regarding the extent to which BECCS-based bioenergy generation will replace low-emission (average) energy sources as the economy decarbonises. If it does not, the climate benefits of the bioenergy systems will be far greater than presented in Part Two, where substitution credits are assumed to decline though time. All scenarios illustrate the diminishing role of displacement effects as the global economy becomes increasingly decarbonised. In a future where global net emissions are low or net negative, and there are few GHG-intensive products to displace, the relative attractiveness of different bioenergy options depends on their ability to deliver CDR, rather than displacement effects (see Part One results). At this stage, performance in other environmental impact categories, beyond greenhouse gas emissions, may then be given greater consideration. For example, forest-based BECCS systems may also provide co-benefits such as biodiversity enhancement and other ecosystem services, which are not fully captured in this study.

The projected rates of CCS deployment and decarbonisation in the Part Two scenarios may appear pessimistic, particularly for a society which is (somewhat) actively pursuing net zero CO_2 emissions by 2050. However, Part One scenarios encompass a range of decarbonisation trajectories for the bio-based value chains. Further, in the progressive scenarios of Part Two, the rates of CCS deployment on bioenergy and economy-wide decarbonisation are coupled, reflecting a realistic interdependence: high levels of CCS deployment on bioenergy are unlikely without wider decarbonisation efforts, and vice versa. While the scenarios are conservative, they remain plausible within this context.

Finally, this assessment focuses solely on radiative forcing from greenhouse gas emissions and does not capture additional biophysical climate effects, such as changes in surface albedo. Albedo changes can influence the net climate impact of forest management and bioenergy systems^{71,72}. For example, maintaining unharvested forests and increasing aboveground biomass coverage could reduce surface reflectivity relative to harvested forests, decreasing radiation reflected back to the atmosphere, and potentially reducing long-term mitigation of global warming⁷². Although beyond the scope of this study, incorporating these biophysical effects into future assessments would provide a more comprehensive understanding of the climate implications of forest-based BECCS.

Data availability

The authors confirm the findings of this study are available within the article and its Supplementary materials. Net annual greenhouse gas fluxes for parts 1 and 2, dynamic LCA results for parts 1 and 2 and the carbon dioxide contribution from major processes for part 2 can be found in the Supplementary Data file.

Received: 10 July 2025; Accepted: 13 February 2026;

Published online: 18 March 2026

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Acknowledgements

This research was supported by Ireland’s Department of Agriculture, Food and the Marine through the INFORMBIO project (DAFM-2021-R423). We thank all members of IEA Bioenergy BECCUS Inter-task phase 2 for their insights in shaping this study and also Sustainable Energy Authority Ireland and Ireland’s Department of the Environment, Climate and Communications who supported engagement with IEA Bioenergy through the FORESIGHT project.

Author contributions

Bishop: conceptualisation, formal analysis, methodology, writing—original draft preparation. Duffy: data curation, validation, writing—review and editing. Berndes: conceptualisation, validation, writing—review and editing. Brandão: validation, writing—review and editing. Cowie: validation, writing—review and editing. Healey: validation, writing—review and editing. Hennig: validation, writing—review and editing. Koponen: validation, writing—review and editing. Gaffey: funding acquisition, validation, writing—review and editing. Styles: conceptualisation, validation, writing—review and editing.

Competing interests

The authors declare no competing interests.

Additional information

Supplementary information The online version contains Supplementary material available at <https://doi.org/10.1038/s43247-026-03333-1>.

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Peer review information *Communications Earth and Environment* thanks Jan Sandstad Næss, Tjerk Lap and the other anonymous reviewer(s) for their contribution to the peer review of this work. Primary handling editors: Martina Grecequet. [A peer review file is available.]

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