Member State specific pathway for NETP deployment

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Editors/Authors: Nixon Sunny, Solene Chiquier, Niall Mac Dowell

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V	Date	Beneficiary	Authors/ Reviewers
1.0	31.05.2023	ICL	Nixon Sunny (ICL), Solene Chiquier (ICL), Niall Mac Dowell (ICL)/ Kati Koponen (VTT), Conor Hickey (UOXF)

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Partners

VTT – VTT Technical Research Centre of Finland Ltd, Finland
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Executive Summary

The purpose of this document is to provide an assessment of the overall technical and commercial potential to deploy NETPs at the EU member state level, considering the more mature negative emissions technologies and practices (NETPs). This work expands upon previous deliverables 4.1 (NETP database), 4.2 (biogeophysics database), 4.3 (allocation of member state-specific CO_2 removal quotas), and 4.4 (development of the modelling prototype) to define the long-term CO_2 removal potential in the EU. This report presents a detailed methodology of the dynamic modelling and optimisation framework used to design the cost-optimal portfolio of NETPs. It discusses the key methodological assumptions, constraints (availability, technology, and biogeophysics), and the key decision variables (portfolio of different NETPs, scale, and location of deployment).

The analysis extends previous academic literature to define the indigenous potential for CO_2 removal, by also accounting for technologies such as biochar and enhanced weathering. Most notably, the findings suggest that the EU member states, and the UK have approximately 100 Gt $CO_{2,eq}$ potential offered by a combination of BECCS (67 Gt $CO_{2,eq}$ by 2100) and afforestation (33 Gt $CO_{2,eq}$ by 2100). The technical potential for BECCS is evaluated using a combination of dedicated energy crops such as miscanthus and willow from marginal agricultural land. Similarly, the overall afforestation potential is estimated based on areas with reforestation potential. Note that the deployment of BECCS using agricultural and forestry residues will increase the overall deployment potential and may hold the greatest potential in states such as Finland. However, it is worth noting that commercial deployment may be limited by forest expansion constraints and build rates of technologies, alongside CO_2 storage capacity.

Biochar does not offer a significant potential (4.08 Gt $CO_{2,eq}$ by 2100) for CO_2 removal as it is an inefficient use of biomass compared to more efficient technologies such as BECCS. The overall CO_2 removal potential of the biochar route improves with increasing use of the derivative bio-oils and gases in applications with CCS. On the other hand, enhanced weathering, although offers a very substantial potential (80 Gt $CO_{2,eq}$ by 2100 assuming the use of basalt with a weathered fraction of at least 5% over 80 years), is limited by the number of mining facilities and the weathering rates over time. In practice, a cost-effective deployment pathway will be reliant on technologies with a lower degree of risk (higher certainty on performance), and this is more likely to be expected in the case of engineered removals such as BECCS and DACCS.

The analysis suggests that the EU member states, and the UK have sufficient combined NETP potential to meet a cumulative CO_2 removal quota of approximately 81 Gt $CO_{2,eq}$ by 2100, apportioned based on a "responsible" share of the IPCC P3 pathway. The resulting cost optimal NETP portfolio is mostly comprised of BECCS (73%), afforestation (20%), biochar (5%), and enhanced weathering (2%). Note that this solution uses approximately 60% of the EU's overall CO_2 storage capacity, which leaves only 40% of available capacity for hard-to-abate mitigation activities. Thus, the capacity to deliver higher CO_2 removal quotas will be mainly constrained by CO_2 storage availability as opposed to technology supply or build rate constraints. This also emphasises the need to establish cross-border collaboration and the development of supportive policy frameworks to effectively implement these technologies. European regions such as Norway may offer additional CO_2 storage potential to support further deployment of engineered removals in the continent. It will also facilitate the trade of raw materials and feedstocks that are necessary for the implementation of these NETPs in member states.

Overall, this study presents a framework for undertaking thought experiments on the evolution of NETPs over a long-term time horizon to support the aims of the Paris Agreement. Ongoing and future work in WP 4 is aimed at expanding the analysis presented herein to include detailed scenario and uncertainty analysis.

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1. Introduction

The 2015 Paris Agreement binds nations to limit global warming to "well below" 2°C and encourages efforts to curb it to 1.5°C by reducing global greenhouse gas (GHG) emissions and reaching net-zero by mid-century. This translates into a remaining budget of 420 Gt $CO_{2,e}$ to stay within reasonable bounds of confidence[1]. Europe has committed itself to ambitious climate change targets that require net-zero GHG emissions by 2050[2]. Achieving net-zero emissions may be profoundly challenging or, in some instances, simply not possible in sectors such as aviation, heavy industry, agriculture, etc., without GHG removal[3]. Negative emissions technologies and practices (NETPs), play an important role in this vision by removing CO_2 from the atmosphere through various technologies. NETPs include, and are not limited to, afforestation, reforestation, enhanced weathering, biochar, soil carbon sequestration, bioenergy with carbon capture and storage (BECCS), and direct air capture and storage (DACCS).

GHGs, such as CO_2, remain in the atmosphere for long periods of time[4]. NETPs can improve climate resilience by reducing the atmospheric concentrations of GHGs, which can help mitigate peak warming, and contribute to the long-term stabilisation of the climate[5]. This is crucial to limit the impacts on soil, agriculture, water resources, and a range of different ecosystems. NETPs may support the conservation of biodiversity, provide ecosystem services, and improve agricultural output[6]. Nature-based solutions such as afforestation and reforestation, the use of biochar in soil, soil carbon sequestration, etc., all have the capacity to positively impact the ecosystem and promote sustainable land use practices[7].

Moreover, NETPs offer nations the unique ability to offset their historical contributions to the global pool of GHGs[8]. This can support the ongoing industrialisation and economic growth in developing countries around the world and ensure that the socio-economic burdens of the low-carbon transition are balanced responsibly across the states[8].

An early deployment of NETPs is crucial to ensure a smooth transition to a net-zero economy[9]. It will reduce atmospheric concentration of GHGs and limit the extent of future warming. By deploying NETPs, the available carbon budget can be managed to avoid the risk of overshooting both the emission and temperature targets, thereby reducing the need for more drastic measures in the future. Furthermore, early investments in NETPs will accelerate the pace of innovations in technology design and operation, enable cost reductions, and efficiency improvements through technology learning, thus making removals more cost-effective in the long run[10]. This can also enhance the economic competitiveness of a region through technology exports and local job creation[11].

Several NETPs are also strongly linked to GHG mitigation activities, and lessons from their commercialisation may be directly applicable to other GHG reduction efforts[12]. For example, the commercialisation of BECCS and DACCS will require a fully functional CO_2 transport and storage infrastructure, which is also necessary for industrial CCS. Thus, learnings from these projects can address the knowledge gaps in industrial CCS, and vice-versa[13]. This highlights the co-benefits and opportunities to develop skills through investments in such initiatives.

Studies have shown that Europe has contributed to 21% of the cumulative GHG emissions since the beginning of the industrial revolution[14]. In fact, Deliverable 4.3 of this project, analysed the responsibility for NETP deployment based on historical contributions and the capacity to influence investments. The findings showed that Europe may need to deliver around 21 – 47% of the cumulative NETP deployment quota. Additionally, Europe can influence the global economy by providing a strong example of leadership through NETPs. In particular, the implementation of NETPs within the member states can support local objectives, as well as encourage similar action by other countries around the world.

Despite scientific agreement on the role of NETPs in achieving net-zero, there has been limited progress in their deployment to date, except for afforestation[15]. Moreover, the European Union (EU) has lacked a clear vision on the scale of CO_2 removal that is needed over a long-term horizon[16]. The roadmap for NETP deployment in Europe, and the respective responsibilities of each member state in delivering it is unclear. Importantly, the economic implications of NETP deployment at scale is also poorly understood. This report addresses such questions by quantifying the technical limits to deploy NETPs at scale, using indigenous resources, and subsequently explores cost-effective portfolios of NETP deployment pathways until the year, 2100.

We focus on quantifying the total cost-effective potential to deploy NETPs within each EU member state. This informs the development of targeted policy interventions that can expedite investments. This assessment notes each region's capacity to contribute towards an overall NETP vision for Europe and identifies the value of regional collaboration in delivering the quota. A tailored approach is developed in this report, where each member state optimises their CO_2 removal based on their biogeophysical conditions and the availability of relevant resources.

1.1 Regional context

According to the UNFCCC accounting, the EU-27 emitted a total of 3.47 Gt $CO_{2,e}$ in 2021, which is a 30% reduction in emissions relative to 1990 levels[17].

Table 1: A breakdown of greenhouse gas emissions by member states in the EU based on reporting from the year, 2021. Countries are arranged in ascending order based on their overall emissions[17].

Country	Emissions (Mt/ yr)
Malta	2.1
Cyprus	8.7
Luxembourg	9.4
Latvia	10.7
Estonia	12.6
Slovenia	16.1
Lithuania	20.3
Croatia	24.4
Slovakia	41.3
Denmark	43.9
Sweden	47.8
Finland	47.9
Bulgaria	54.0
Portugal	56.5
Ireland	62.1
Hungary	64.2
Greece	77.5
Austria	77.5
Belgium	111.0
Romania	115.4
Czechia	119.0
Netherlands	167.7
Spain	288.8
Poland	399.9
France	414.8
Italy	417.6
Germany	760.4
Total	**3,471**

The EU aims to reduce greenhouse gas emissions by at least 55% by 2030 compared to 1990 levels and achieve climate neutrality by 2050 as part of the European Green Deal[2]. The EU's climate policies include various measures such as renewable energy targets, energy efficiency improvements, and other policy mechanisms to drive the transition to a lower carbon economy. Notwithstanding, member states within the EU have their own commitments on emissions reductions, and the uptake of renewable energy.

For this report, we include the 27 member states in Europe, in addition to the United Kingdom. Some of the key commitments from the member states are summarised as follows:

- Germany, as the largest economy and GHG emitter in 2021, aims to reduce their emissions by at least 55% by 2030 compared to 1990 levels, and achieve net-zero emissions by 2045[18].
- Italy, the second largest GHG emitter in 2021, aims to reduce GHG emissions by 33% by 2030 compared to 2005 levels[19].
- France, the third largest GHG emitter in 2021, aims to reduce emissions by 40% by 2030 compared to 1990 levels, and achieve net-zero emissions by 2050[20].
- Spain aims to reduce their non-ETS GHG emissions by 26% by 2030 compared to 2005 levels. This is combined with an ambitious target of 74% of renewables share in electricity generation and 100% by 2050[21].
- Sweden, one of the smallest emitters in the EU, aims to reduce their GHG emissions by 63% by 2030 compared to 1990 levels, and achieve net-zero emissions by 2045[22].
- Poland, as a large emitter in the EU, aims to reduce their GHG emissions by 30% by 2030 compared to 1990 levels. They also intend to displace a significant share of coal in the energy sector with renewable energy[23].

The examples above showcase a diverse array of ambitions on climate action in line with their respective national contexts. Similarly, the pace and uptake of NETPs are also likely to vary across member states in line with the capacity for deployment[24]. In contrast to renewable energy, some nations may be able to provide a greater commercial potential for NETPs than the amount they need to reach net-zero emissions or offset their historical contributions. This is because the capacity to deploy NETPs cost-effectively may vary based on the biogeophysical conditions of a region. For example, vast areas of land which had forests previously will likely hold the greatest reforestation potential, or areas with the greatest availability of basic or ultrabasic rock may offer the greatest potential for the supply of enhanced weathering technology. This will mean that effective policies need to be developed to recognise and acknowledge the supply of primary feedstocks for the deployment of NETPs.

It is important to note that while some EU member states explicitly incorporate NETPs in their climate change commitments, the level of detail and priority given to these technologies generally vary depending on the region. Countries such as France and Sweden have recognised the importance of technologies such as afforestation and soil carbon sequestration in helping them achieve their net-zero vision[25]. Finland and Germany have renewed their focus on sustainable forest management, whereas countries such as the UK are aiming to incorporate engineered CO_2 removals such as BECCS in their wider strategy on net-zero[25,26]. Nationally determined contributions (NDCs) to the Paris Agreement did not include NETPs within its remit, and this creates uncertainty on how countries intend to take responsibility for their deployment in the timeframe leading up to 2100.

EU member states face unique challenges in terms of NETP deployment, owing to their diverse geographical conditions, which directly influence the potential for negative emissions. Member states in Northern Europe may have more forested areas suitable for afforestation and reforestation[27], whereas countries in the south may enjoy more favourable conditions for renewable energy, which could favour engineered CO_2 removals such as DACCS. However, these regions may also face water shortages and competition over land. Moreover, there may be limited opportunities to use existing land for NETPs, due to the increasing need for agricultural lands[28]. There may be opportunities to improve on existing agricultural practices by reducing methane and nitrous oxide emissions from the use of fertilisers, whilst promoting the sequestration of CO_2 in the soil.

A key element of the problem relates to effective policy development and implementation. An effective policy instrument at the EU level will require coordination at multiple levels to ensure policy coherence across the different member states[29]. It will need to be cognisant of the need to work in collaboration with countries outside of the EU. Thus, policy measures must not be restrictive and needs to provide autonomy for the member states to develop their national climate policies and targets, but with adequate incentive to engage in international collaboration[14]. Policies will need to ensure that international standards align, encourage knowledge sharing, and establish governance mechanisms for the deployment of NETPs at scale.

There are numerous policy and regulatory gaps which will need to be addressed to enable the deployment of NETPs. A streamlined permitting process needs to be developed to encourage investments in new projects by providing investors and project developers with regulatory certainty. This may be combined with financial instruments that specifically promote projects that deliver negative emissions to the established standard[30]. This will help accelerate innovations in technology to improve their cost profile over time in line with the ambitions to scale up NETPs.

More importantly, however, an information gap, which compounds the uncertainties. A lack of a clear understanding of the overall technical potential to deliver NETPs in the EU using indigenous resources, and their sensitivities to starting assumptions in data is a key knowledge gap. The translation of the technical potential into a commercially realisable potential of NETPs is another key consideration. This dictates the extent to which the EU can operate in a self-sufficient manner with respect to its ambition on CO_2 removal, and the corresponding implications for international collaboration.

1.2 Relevant existing works

The National Academies of Sciences, Engineering, and Medicine published an agenda-setting report through the National Academies Press in 2019[31], wherein they explored questions related to the need for negative emissions at scale. They conclude that negative emissions technologies should be seen as part of the overall mitigation strategy rather than as a solution to reduce carbon dioxide levels after reaching emission avoidance. The key question is whether negative emissions are cost-effective and have few adverse impacts.

However, reducing emissions remains crucial, and the most efficient and least disruptive approach involves a portfolio of technologies with positive, near-zero, and negative emissions. It is worth noting that studies have identified technologies such as afforestation, reforestation, changes in forest management, soil carbon sequestration, and BECCS as technologies that are commercially ready for large-scale deployment[32]. These technologies offer ancillary benefits, for example, improvements in forest and agricultural productivity, biofuels, hydrogen, and electricity generation, etc[33]. The NAP notes that NETPs need to remove approximately 10 $GtCO_2$,eq/yr globally by 2050 and 20 $GtCO_2$,eq/yr by 2100[31].

Studies such as that by the Royal Society reference the need for a cumulative total of 810 $GtCO_2$,eq removal from 2018 until 2100[34]. These figures are broadly in line with estimates from the IPCC, showing a cumulative global NETP requirement between 348 and 1,218 Gt CO_2,eq by 2100 to limit warming to 1.5°C. According to the “middle-of-the-road” P3 scenario from the IPCC Special Report on Global Warming of 1.5°C, the global cumulative NETP requirement by 2100 is 687 Gt CO_2,eq.

If the aforementioned figures are apportioned based on historical responsibility, the EU would have to remove a cumulative total of 73 – 256 Gt CO_2,eq by 2100[24]. Based on a P3 scenario, this would result in a cumulative removal of 90 – 150 Gt CO_2,eq by 2100. To contextualise this based on an 80-year timeframe following the publication of the aforementioned figures, the EU would need to remove an average of 1.8 Gt CO_2,eq/yr, or 52% of its overall GHG emissions from 2021 to reach the upper figure. This is a profoundly challenging feat to achieve, and it is unclear if the EU can source such scales using indigenous capability, if at all. Moreover, given the EU's capability to influence investment in NETPs, the EU may eventually take a greater share of the cumulative NETP budget[24]. The technical potential of NETPs in Europe is unclear, which leads to policy uncertainty. Addressing these gaps is crucial as it can provide a solid foundation for supply-demand matching, thereby influencing the evolution of the carbon removal market.

Studies have shown that delaying the deployment of NETPs, specifically BECCS and DACCS in the EU could have significant economic and environmental implications. Galán-Martín found that each year of inaction in the EU translates into an additional cost of 0.12 - 0.19 trillion EUR 2015 to remove a cumulative total of 50 Gt CO_2[9]. This cost is comparable to the estimated annual investments required globally to limit global warming to 1.5°C. Moreover, postponing NETP deployment until after 2050 would restrict the maximum cumulative removal potential until 2100 to 56.4 Gt CO_2, which is equivalent to a decade's worth of emissions[9]. Additionally, reluctance from member states could lead to an uneven distribution of efforts, with some countries shouldering more of the burden than others. Their work is limited to BECCS and DACCS, but other technologies should also be considered, as they can also contribute in a timely manner to the cumulative deployment of NETPs.

Pozo et al. found that most member states, except Spain, France, and Romania would struggle to meet their "quota" for NETPs using indigenous potentials. Note that these quotas were derived using principles of burden sharing – responsibility to deliver NETPs, capability to finance deployment, and an equal right to be protected on a capita basis[24]. When considering per capita allocation, only 22 out of 27 member states can meet their quotas using indigenous potential. It is important to capture that the total domestic removal potentials are generated by assuming that BECCS would be applied to agricultural and forestry residues, and dedicated energy crops, together with reforestation of land. The corresponding total domestic potentials in the EU-27 and the UK amounts to 92 Gt CO_2[24]. This is contingent on the availability of natural resources and geological storage capacities to support the scale up of BECCS in comparison to other uses such as industrial CCS. However, this excludes the potential owing to technologies such as enhanced weathering, which is less constrained by the availability of natural resources.

In this context, the European Forest Institute assessed the capacity of the forestry sector to contribute towards the EU-27's climate neutrality target by 2050, as they currently offset about 10% of the region's greenhouse gas emissions. To meet the targets, the EU-27's Land Use, Land-Use Change, and Forestry (LULUCF) sector must remove an additional 170 $MtCO_2$,eq/yr by 2050[35]. Mitigation activities like avoiding deforestation, afforestation/reforestation, shifts in wood use, and increased efficiency can be combined to provide an additional mitigation of up to 72 $MtCO_2$,eq/yr by 2050 in the EU-27. Moreover, when combined with forest conservation activities, other active forest management, or decreasing forest harvest, this forest-based mitigation potential could increase to 125 $MtCO_2$,eq/yr, 138 $MtCO_2$,eq/yr, or 143 $MtCO_2$,eq/yr, respectively. Nonetheless, their findings suggest that forest-based mitigation activities alone will not be sufficient to meet the climate targets for the LULUCF sector by 2050.

1.3 Report structure

The remainder of this report is structured as follows:

- Section 2 describes the methodological approach used to evaluate the technical potential of NETPs in different member states across the EU. It further presents the mathematical formulation used to optimise the portfolio of negative emissions technologies across the EU.
- Section 3 quantifies the technical potential to remove CO_2 over a long-term horizon. It identifies key biogeophysical parameters that are responsible for the differences in indigenous removal potential in the EU member states. It presents a cost-optimal portfolio of negative emissions for each of the members states in the EU.
- Section 5 concludes with the key outcomes of this work, highlighting its limitations, and identifying other relevant research gaps to address as part of ongoing and future research.

2. Methodology

This section provides an overview of the key methodological relations used to calculate the net CO_2 removal potential of a NETP. The modelling framework uses a multidimensional approach, incorporating a wide array of data sources, sub-component models, and analytical methods. By considering both engineered and nature-based solutions, such as afforestation, reforestation, biochar, BECCS, DACCS, and EW, this methodology provides a comprehensive framework for estimating the net removal potential of CO_2. Note that all the key sources of data are described in more detail in deliverables 4.1 and 4.2 of the project. Readers who are primarily interested in the formulation of the modelling tool are encouraged to read deliverable 4.4 of the project.

2.1 Carbon intensity of the energy vectors

2.1.1 Liquid fuels

The incumbent fuels are replaced gradually by biofuels between 2040 and 2080 using an assumption of linear displacement, as shown in the equation below:

$$CI_{fuel}(t) = \%_{fossil\ fuel}(t) \times CI_{fossil\ fuel}(2020) + \%_{bio-fuel}(t) \times CI_{bio-fuel}(2020)$$

where: $CI_{fuel}(t)$ is the carbon intensity of fuels over time (kg CO_{2eq}/l), and t is time (yrs). $CI_{fossil\ fuel}(2020)$ and $CI_{bio-fuel}(2020)$ are the carbon intensities (direct and indirect emissions) of fossil-fuels and bio-fuels (kg CO_{2eq}/l), respectively, as shown in Table 2, and $\%_{fossil\ fuel}(t)$ and $\%_{bio-fuel}(t)$ are the shares of fossil fuels and bio-fuels over time (%), respectively. The sum of these % shares must equal to 100% as the total amount for a blended fuel feedstock.

Table 2: Direct and indirect carbon emissions for relevant fuels.

Type of fuel	Direct emissions (kg CO_{2eq}/l)	Indirect emissions (kg CO_{2eq}/l)
diesel	2.55	0.61
biodiesel	0.17	0.37

2.1.2 Electricity supply

Electricity generation is assumed to reach carbon-neutrality by 2050, using projections of the IPCC illustrative pathway P2, as shown in the equation below:

$$CI_{elec}(t) = CI_{elec}(2020) \times \%_{decarbonisation}(t)$$

where: $CI_{elec}(t)$ is the carbon intensity of electricity over time (g CO_{2eq}/kWh), and t is time (yrs). $CI_{elec}(2020)$ is the current carbon intensity of electricity (g CO_{2eq}/kWh), and $\%_{decarbonisation}(t)$ is the decarbonisation share over time (%).

2.1.3 Heat supply

Thermal energy is required to produce various NETPs. Current output thermal energy is provided by natural-gas, and gradually replaced by wood between 2040 and 2080, as shown in the equation below:

$$CI_{heat}(t) = \frac{\%_{NG}(t) \times CI_{NG}(2020) \times LHV_{NG}(2020) + \%_{Wood}(t) \times CI_{Wood}(2020) \times LHV_{Wood}(2020)}{\%_{NG}(t) \times LHV_{NG}(2020) + \%_{Wood}(t) \times LHV_{Wood}(2020)}$$

where: $CI_{heat}(t)$ is the carbon intensity of heat provision (kg CO_{2eq}/MJ), and t is time (yrs). $CI_{NG}(2020)$ and $CI_{Wood}(2020)$ are the current carbon intensities of natural gas, and wood (kg CO_{2eq}/MJ), respectively, and $LHV_{NG}(2020)$ and $LHV_{Wood}(2020)$ are the current lower heating values (LHV) of natural gas, and wood (MJ/kg), respectively as shown in Table 3. $\%_{NG}(t)$ and $\%_{Wood}(t)$ are the shares of natural gas and wood over time (%), respectively. The sum of these % shares must equal to 100% as the total amount for the heat supply.

Table 3: Estimate of carbon intensities for natural gas and wood chips.

Type of fuel	Direct emissions (g CO_{2eq}/MJ)	Indirect emissions (g CO_{2eq}/MJ)	LHV (MJ/kg)
Natural gas	56.59[36]	11.74 (10.8–15.6))[37,38]	44.76[36]
Wood chips	4.29[36]	2.24[36]	13.60[36]

2.2 Characterisation of space and time

To effectively model and analyse NETPs, a spatial granularity at the member state level is adopted (with the addition of the UK to the EU-27 states). This approach involves aggregating data based on the regional boundaries of the member states, considering factors such as the distribution of natural resources, geological storage capacity, and other relevant biogeophysical, and climate-related parameters. By focussing on each member state, the modelling accounts for the unique characteristics and resources available to each nation. This approach recognises the variations that are likely to result from regional differences in line with the discussions in the earlier sections.

Accounting for the region-specific biogeophysical condition is crucial in evaluating the potential for NETP deployment. Factors such as precipitation levels, crop yields, availability of land, availability of basic rock formations, and geological storage vary significantly across countries, thus influencing the viability of NETPs. By considering these variations, the model can more accurately assess the potentials at a member state level to support policy development and commercial deployment. This level of granularity is also necessary to further explore the value of NETP deployment on other dimensions of the natural environment, such as biodiversity impacts, although this is not explicitly evaluated in this work.

The model uses multiple sets for its characterisation of time. The first is a mathematical set covering each decade of investment from 2020 – 2030 until 2090 – 2100. This allows for long-term investment planning, which dictate the level of NETP deployment in each decade in line with maximum achievable technology diffusion speeds. Although the modelling framework presented in this project is deterministic, it is possible to expand upon this representation to consider stochastic formulations and address changes in the commercial landscape. The second time set covers all the operation periods across the entire operating horizon (2020 – 2100) with each period lasting a year. This second set enables a more granular evaluation of the dynamic response of NETPs over time to define phenomenon such as forest growth, mineralisation over time, biochar decay, etc.

2.3 Afforestation

Afforestation is modelled using five integrated sub-component models: 1) a forest growth model, 2) a forest management cycle model, 3) a biogenic carbon sequestration model, 4) an associated "fire-penalty" model, and 5) a forestry operations model. The remainder of this subsection presents the key methodological relations within each subcomponent model.

Forest growth depends on many parameters, such as climate, forest type or forestry practices[39]. The above-ground biomass stock of reference can be defined as a sigmoid curve, which is typical in even-aged stands in the absence of forest management (without human intervention).

In this model, the above-ground biomass stock of reference B_{Ref}^{AG} (tonnes$_{DM}$/ha) is characterised by ecological zones and parametrised with the IPCC default biomass stock B_{Ref} (tonnes$_{DM}$/ha) and net biomass growth rate G_{Ref} (tonnes$_{DM}$/ha/yr) of natural forests. B_{Ref}^{AG}, aboveground biomass stock, is calculated for each ecological zone *gez* and each year *yr* over the forest growth period as follows:

$$\forall\, gez, yr \le T_{Ref}(gez), \qquad B_{Ref}^{AG}(yr, gez) = \frac{L_{Ref}(gez)}{1 + \exp\left(-k_{Ref}(gez)\left(yr - x_{0,Ref}(gez)\right)\right)}$$

$$\forall\, gez, yr > T_{Ref}(gez), \qquad B_{Ref}^{AG}(yr, gez) = L_{Ref}(gez)$$

$$T_{Ref}(gez) = \frac{B_{Ref}(gez)}{G_{Ref}(gez)}$$

$$L_{Ref}(gez) = B_{Ref}(gez)$$

$$x_{0,Ref}(gez) = \frac{T_{Ref}(gez)}{2}$$

$$k_{Ref}(gez) = \frac{\ln\left(\frac{a_{Ref}}{100 - a_{Ref}}\right)}{x_{0,Ref}(gez)}$$

where:

- T_{Ref} is the growing period of reference (years),
- L_{Ref} is the maximum biomass stock of reference ($tonnes_{DM}.ha^{-1}$),
- $x_{0,Ref}$ is the mid-point of the reference sigmoid curve (years),
- a_{Ref} is the asymptotic coefficient of the reference sigmoid curve (-), whose default value is set to 99,
- and k_{Ref} is the slope coefficient of the reference sigmoid curve ($tonnes_{DM}.ha^{-}1.yr^{-1}$), such as:

$$\frac{d\ AGBio_{Ref}}{dt}\left(yr = x_{0,Ref}(gez), gez\right) = \frac{k_{Ref}(gez)}{4}$$

At the end of the growing period of reference, B_{Ref}^{AG} is assumed to have reached its maximal value (*i.e.,* L_{Ref}). The forest management cycle is modelled according to an adaptation of the mean annual increment and the maximum mean annual increment. This work uses the mean annual growth variable and the maximum mean annual growth variable, as introduced by Chiquier[40], to determine the forest growth phases.

The annual (above-ground) growth (AG) AG_{Ref}^{AG}, the mean annual growth MAG_{Ref}^{AG}, and the maximum mean annual growth $MMAG_{Ref}^{AG}$, are derived from the above-ground biomass stock of reference B_{Ref}^{AG}. They are calculated for each ecological zone, *gez* and each year *yr* as follows:

$$\forall\ gez, \qquad AG_{Ref}^{AG}(yr, gez) = \begin{cases} B_{Ref}^{AG}(yr = 1, gez), & if\ yr = 1 \\ B_{Ref}^{AG}(yr, gez) - B_{Ref}^{AG}(yr - 1, gez), & if\ yr > 1 \end{cases}$$

$$MAG_{Ref}^{AG}(yr, gez) = \frac{B_{Ref}^{AG}(yr, gez)}{yr}$$

$$MMAG_{Ref}^{AG}(gez) = \max_{yr} MAG_{Ref}^{AG}(yr, gez)$$

A schematic of the model-based decision-making process is captured in Figure 1. The forest management cycle is defined for each ecological zone *gez* and for each year *yr*, where:

- $yr_{E,END}$ is the last year of the establishment phase (years),
- $yr_{I,END}$ is the last year of the initial phase (years),
- MAG_{BT}^{AG} is the mean annual growth (MAG), before thinning ($tonnes_{DM}$/ha/yr),
- $MMAG_{Ref}^{AG}$ is the maximum mean annual growth ($tonnes_{DM}$/ha/yr), AT is the annual thinning stock ($tonnes_{DM}$/ha),
- $\%_T$ is the thinning share of the above-ground biomass stock (%),
- B_{BT}^{AG} is the above-ground biomass stock, before thinning ($tonnes_{DM}/h^1$),
- $yr_{last\ FVT}$ is the year during which the last thinning of the full-vigour phase occurred (years),
- and $yr_{last\ MT}$ is the year during which the last thinning of the mature phase occurred (years).
- $yr_{E,END}$ and $yr_{I,END}$ default values are respectively 5 and 15 years.

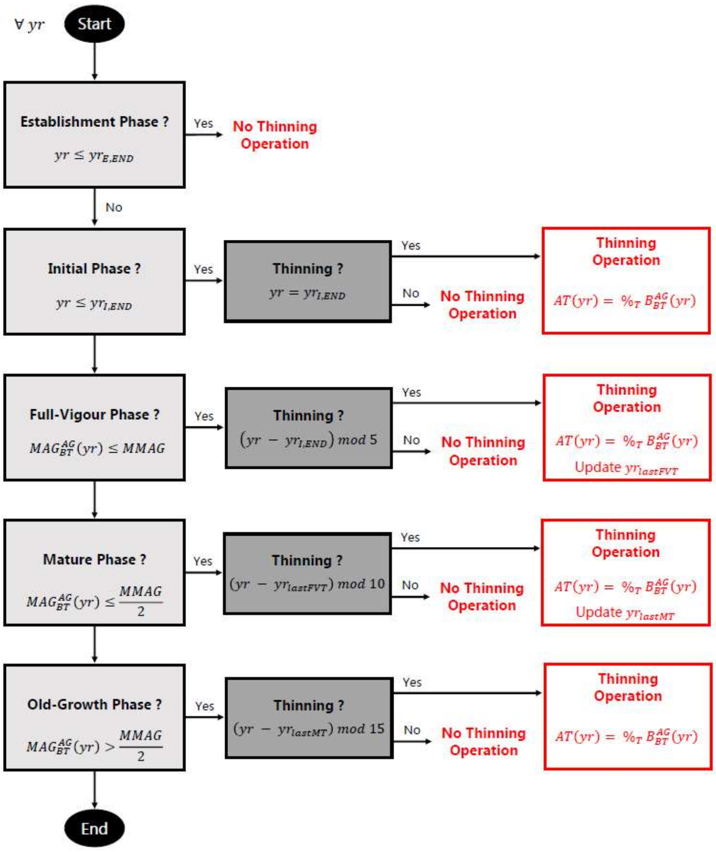

Figure 1: Model-based decision-making process schematic for forest management operations. Figure from Chiquier[40]

The below-ground biomass stock can be estimated from the above-ground biomass stock with the use of a "root-to-shoot" ratio. A "root-to-shoot" ratio usually depends on climate, tree species, soil type and declines with stand age and/or productivity. Extreme range values of 0.09 – 1.16 tonnes $root_{DM}$.(tonnes $shoot_{DM}$)$^{-1}$ have been reported in the literature, although average range values of 0.20 – 0.56 tonnes $root_{DM}$.(tonnes $shoot_{DM}$)$^{-1}$ might be more likely[39].

In this work, the "root-to-shoot" ratio R_{RS} evolves with the amount of above-ground biomass stock (before thinning) B_{BT}^{AG}. Specifically, R_{RS} is interpolated from the IPCC default values[39]. R_{RS} is characterised by ecological zones *gez*, tree species *sp* and by the amount of above-ground biomass stock aB_{BT}^{AG} , and calculated as follows:

$$\forall\, yr, gez, sp, \qquad R_{RS}\left(gez, sp, B_{BT}^{AG}(yr, gez)\right) = \begin{cases} R_1(gez, sp) \times \ln\left(R_2(gez, sp) \times B_{BT}^{AG}(yr, gez) + 1\right), & R_2(gez, sp) \neq 0 \\ R_1(gez, sp), & R_2(gez, sp) = 0 \end{cases}$$

where R_1 and R_2 coefficients are the coefficients interpolated[i] from the IPCC. The below-ground biomass stock B^{BG} derives from the managed above-ground biomass stock (after thinning) B_{AT}^{AG} and the "root-to-shoot" ratio R_{RS}, as follows:

$$\forall\, yr, gez, sp, \qquad B^{BG}(yr, gez, sp) = B_{AT}^{AG}(yr, gez) \times R_{RS}\left(gez, sp, B_{BT}^{AG}(yr, gez)\right)$$

Finally, the total biomass growth curve B^{Total} is defined for each ecological zone *gez*, each tree species *sp* and each year *yr*, and is calculated as follows:

$$\forall\, yr, gez, sp, \qquad B^{Total}(yr, gez, sp) = B_{AT}^{AG}(yr, gez) + B_{AT}^{BG}(yr, gez, sp)$$

The total biomass stock is combined with a factor C_f to estimate the overall carbon content of the biomass. The share of carbon in biomass C_f, depends on the climate, forest type, and tree characteristics. Average values have been reported within the range of 0.43 – 0.55 tonnes of carbon/ tonnes dry matter[39]. The total carbon content in the forest C_{AT}^{Total}, is defined for each ecological zone *gez*, each tree species *sp*, and each year *yr*, and parametrised with the IPCC default values for C_f[39], as illustrated in Table 4:

$$\forall\, yr, gez, sp, \qquad C_{AT}^{Total}(yr, gez, sp) = B_{AT}^{Total}(yr, gez, sp) \times C_f(gez, sp)$$

[i] $R_1$and R_1were obtained by solving a non-linear curve-fitting (data-fitting) problem in least-squares sense in Python 3.7 (function scipy.optimize.leastsq).

Table 4: IPCC default values for carbon content as a function of the type of forest and climate[39]

Climate domain	Carbon content (tonne C $tonne_{DM}^{-1}$)	
	Broadleaves	Conifers
Tropical	0.47	0.47
Subtropical	0.47	0.47
Temperate	0.48	0.51
Boreal	0.48	0.51

Thus, the total biomass CO_2 stock CO_{2AT}^{Total} is obtained by multiplying the C content of biomass in the forest with the molecular weight ratio of CO_2 over C:

$$\forall\ yr, gez, sp, \qquad CO_{2A}^{Total}(yr, gez, sp)\ = C_{AT}^{Total}(yr, gez, sp) \times \frac{44}{12}$$

This results in varying levels of maximum CO_2 sequestration potential for different forest types. Note that they are maximum removal potentials, owing to losses of biomass as dead organic matter, and as harvested wood products.

Table 5: Maximum CO_2 sequestration potential of above-ground, below-ground, and total biomass stocks, as characterised by the ecological zone. Data from Chiquier[40]

Ecological zone	Maximum above-ground biomass (tCO_2/ha)		Maximum below-ground biomass (tCO_2/ha)		Maximum total-ground biomass (tCO_2/ha)	
	Broadleaves	Conifers	Broadleaves	Conifers	Broadleaves	Conifers
Tropical rainforest	517	517	191	191	708	708
Tropical moist deciduous forest	310	310	71	71	381	381
Tropical dry forest	224	224	51	51	275	275
Tropical shrubland	121	121	48	48	169	169
Tropical mountain system	241	241	65	65	306	306
Subtropical humid forest	379	379	89	89	468	468
Subtropical dry forest	224	224	124	124	348	348
Subtropical steppe	121	121	39	39	159	159
Subtropical mountain system	241	241	65	65	306	306
Temperate oceanic forest	317	337	70	72	387	409
Temperate continental forest	211	224	57	58	268	283
Temperate mountain system	176	187	51	53	227	240
Boreal coniferous forest	88	94	31	33	119	127

Boreal tundra woodland	26	28	14	15	40	43
Boreal mountain system	53	56	23	24	76	80

Afforestation and reforestation, while effective in sequestering CO_2, are susceptible to disturbances that can reduce their long-term removal potential, both from natural and anthropogenic causes. The reliability of biogenic CO_2 sequestration is less certain compared to engineered removals such as BECCS or DACCS. Existing risk-accounting methods for natural disturbances like wildfires are limited in their scope and often specific to certain locations or regions[41,42]. Despite advancements in remote sensing and the availability of national and global datasets, such as those provided by programs like LandFire[43], these datasets are insufficient for a comprehensive risk evaluation. In this framework, the risk of wildfires is modelled using a penalty coefficient to assess their impact on CO_2 sequestration potential.

A risk-accounting methodology developed by Hurteau et al.[44] is used to define the wildfire-penalty coefficient R_{fire}, as a function of the ecological zones to be applicable to different regions. R_{fire} is built upon the severity of the fire − the potential biomass loss given a fire occurrence − and its periodicity − the probability of a fire event occurring during a specified time. This is written as follows:

$$\forall\ yr, gez, \qquad R_{fire}(yr, gez) = \begin{cases} 0, & yr < mFRI \\ VDep(gez) \times \left(1 - \dfrac{mFRI(gez)}{yr}\right), & yr \geq mFRI \end{cases}$$

where:

- $VDep$ is the vegetation departure index—ranging from 0% (zero potential biomass loss) to 100% (complete potential biomass loss) (%)
- and $mFRI$ is the mean fire return interval—ranging from 0 years (very frequent) to 1,000 years (very rare) (yrs). Both $VDep$ and $mFRI$ are climate-specific, depending on ecological zones *gez*.

The specific datasets used to define the penalty coefficients are highlighted in deliverable 4.1 and 4.2. It is worth noting that datasets at the US level is used to define the interval between wildfires and this is categorised based on the global ecological zones, which is subsequently extrapolated to the conditions in Europe. This will accrue uncertainty in the model representation and in the results. For more information, see Chiquier et al[14].

The risk-adjusted CO_2 sequestration potential CO_2^{Seq} is defined for each ecological zone *gez*, each tree species *sp* and each year *yr*, and is estimated with the use of R_{fire}:

$$\forall\, yr, gez, sp, \quad CO_2^{Seq}(yr, gez, sp) = (1 - R_{fire}(yr, gez)) \times \left(CO_{2A}^{Total}(yr, gez, sp) + CO_2^{Litter}(yr, gez, sp)\right)$$

Note that the total CO_2 sequestration potential can be further aggregated at the regional scale using national distributions of forest types, as follows:

$$\forall\, yr, sr, gez, \quad CCO_2\ Sequestration\ Potential_{ha}^{AR}(yr, sr, gez) = \sum\nolimits_{sp} CO_{2AT}^{Seq}(yr, gez, sp) \times \%_{sp}(sp, sr)$$

where:

- $CCO_2\ Sequestration\ Potential_{ha}^{AR}$ is AR's (cumulative) CO_2 sequestration potential per ha, in sub-region *sr* and ecological zone *gez*, and in year *yr* (tonnes CO_2 captured/ha),
- $\%_{sp}$ is the share of each forest type/tree species *sp* in sub-region *sr* (%). Note that $\sum_{sp,gez} \%_{sp}(sp, sr) = 1$.

Forestry operations include site establishment, forest road construction, ongoing maintenance, and annual forestry operations. The forest is established by land preparation and planting of new seedlings. For land preparation, mounding is carried out by an excavator[45–47], and herbicide and fertiliser are applied using a tractor[45,47]. Tree seedlings are prepared in nurseries[48] and planted by hand[45,47,49]. Forest roads are needed to access and maintain forests, and they have different classifications based on their frequency of use. Construction of type A and type B roads involves spreading blasted rock and applying a layer of crushed aggregate. Maintenance of type A roads includes annual re-grading or re-surfacing before thinning operations, while type B roads require re-grading and rolling of the aggregate layer before thinning operations. All these operations contribute GHG emissions, which reduce the overall removal potential of afforestation.

Harvesting involves felling trees with harvesters and removing them to the roadside with forwarders. Roundwood harvesting leaves branches and forest residues on the forest floor for ecological value, while the collected residues are compressed into bundles by forwarders. All extracted biomass is stored at the roadside for natural drying from 50% to 30% moisture content [45,50]. Dry matter losses are also observed along every step of the forestry operations − tree felling, harvesting, forwarding and storage − resulting in a total loss of 11.6% of organic matter. This value is consistent with the literature[39,50]. As a reference, the IPCC default projections for harvest loss are 10% for broadleaves and 8% for conifers[39]. Overall, the CO_2 removal potential of afforestation $CO_2\ Removal\ Potential^{AR}$ is the difference between the risk-adjusted CO_2 sequestration potential

$CO_2\ Sequestration\ Potential^{AR}$ and the total CO_2 emissions generated by all forestry operations $CO_2\ Emissions^{AR}$, resulting in a net balance. $CO_2\ Removal\ Potential^{AR}$ is calculated over time as follows:

$$\forall\ yr,$$
$$CO_2\ Removal\ Potential^{AR}(yr) = CO_2\ Sequestration\ Potential^{AR}(yr) - CO_2\ Emissions^{AR}(yr)$$
$$= CO_2\ Bio.Sequestration\ Potential^{AR}(yr)\ \times (1 - R_{fire}(yr)) - CO_2\ Emissions^{AR}(yr)$$

Similarly, the total costs of afforestation are evaluated using the data reported in deliverables 4.1 and 4.2.

2.4 BECCS

Modelling BECCS involves the characterisation of a biomass supply chain that covers indirect land use change (LUC), biomass cultivation, processing, and transport to the BECCS plant, where it is converted to the end-product, with CO_2 capture, transport, and storage[51]. The analysis considers various crops (miscanthus, switchgrass, short rotation coppice willow), residues (agricultural, forestry), lands (cropland, grassland, forest, and marginal land), and transport modes for biomass, including local, and imported sources. BECCS deployment involves converting land for biomass cultivation, resulting in direct land use change (changes in carbon stocks) and possible indirect land use change (additional carbon emissions due to displacement of agricultural land). Due to this carbon, it takes time for BECCS projects to achieve net negative emissions[51].

The carbon break-even time (CBT) of BECCS, i.e., the point at which a project begins to produce negative emissions, depends on the land type used for biomass. For example, converting cropland to biomass cultivation for BECCS results in 37.5 tCO2/ha of LUC[52] and 0.2 tCO2/ha of ILUC[53,54]. No LUC/ILUC is attributed to marginal land; medium LUC and high ILUC are attributed to cropland and grassland, as using these managed lands means an activity must be re-allocated elsewhere, and high LUC and no ILUC are attributed to forests. Modelling shows that BECCS using local energy-dedicated crops, such as miscanthus, on cropland has a CBT of 14 years, while for forests, grasslands, and marginal land (MAL), it is 20, 37, and less than a year, respectively.

Straw is a by-product of wheat production agricultural residues. Although there are several ways to assign energy consumption and greenhouse gas (GHG) emissions to crop production and agricultural residues, we do not attribute land conversion and farming contributions to the residue in our analysis, and instead assume that those impacts are associated with wheat.

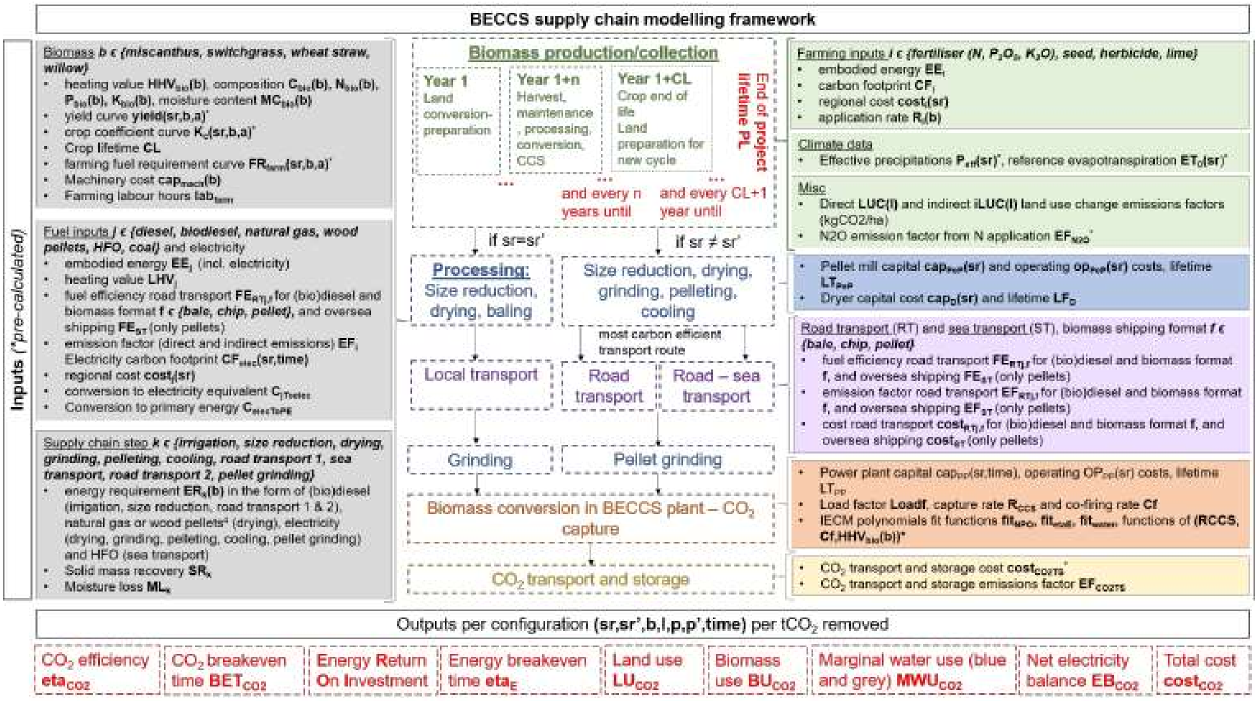

Figure 2: A schematic of the BECCS supply chain modelling framework. Biomass production, collection, processing, transport, conversion, and CO_2 capture, transport and storage are explicitly defined in the modelling. Figure from Fajardy[51].

We use regional yields, lifetime of crops, harvest rotation, and first and second harvest yield potential for each crop from the literature. To convert wheat grain yield into wheat straw yield, we consider a residue recovery factor. This factor can range from 0.6 to 2.0[55], but for energy crops, it is assumed to be 1. The removal of agricultural and forestry residues from the field leads to soil nutrient depletion, and it is necessary to compensate for this based on the amount of residue yield[56]. In the model, 30% of nitrogen content, 100% of phosphorus content, and 100% of potassium content are assumed to be available to the field[56]. Considering the composition of wheat straw, the amount of additional fertiliser input is calculated in the model. For energy crops, the fertiliser input is determined by the yield and the nutrient content of the biomass[57]. Diesel is assumed to be used for field operations such as harrowing, ploughing, seeding, packing, etc.

Biomass can be supplied to power plants in different forms, such as bales, chips, pellets, or briquettes. Typically, the transportation costs are lower for denser biomass feedstock. Pelleting biomass helps to minimise biomass loss in the supply chain. Since the optimal moisture content for biomass combustion is around 10-15 wt%[58], it is generally necessary to dry the biomass. Pelletizing biomass also requires a feedstock with approximately 10 wt% moisture[59].

To transform harvested biomass into a suitable fuel, several steps are involved: size reduction (chopping, chipping), drying to a maximum moisture content of 15%, further milling (grinding), and pelleting using a die extruder. These processing operations require thermal energy for drying and electrical power (for size reduction, drying, grinding, pelleting, and cooling).

By using the energy requirement (ER_k) of each process (k), and the biomass input $Bio_{proc,k-1}$ for each processing step, we calculate the contribution of each process to the embodied energy of biomass. The energy requirements for biomass processing are obtained from literature sources, which include experimental and industrial data, specifically for size reduction, grinding, and pelleting. Moisture loss, ML_k, and solid recovery, SR_k, factors are therefore considered for each process k, and links each process's biomass input, $Bio_{proc,k-1}(sr, sr', b, yr)$, and output $Bio_{proc,k}(sr, sr', b, yr)$:

$$Bio_{proc,k}(sr, sr', b, yr) = Bio_{proc,k-1}(sr, sr', b, yr) \times SR_k(sr, sr'), \qquad \forall\, stage\ k, sr \neq sr'$$

$$MC_{proc,k}(sr, sr', b, yr) = MC_{proc,k-1}(sr, sr', b, yr) - ML_k, \qquad \forall \text{ stage } k,\ sr \mathrel{/=} sr'$$

Where sr denotes a region in the model, b denotes the type of biomass, and yr denotes the time. For more information on modelling the processes in the biomass supply chain, refer to Fajardy[51].

The efficiency of a BECCS power plant depends on factors such as the type of steam cycle used, the performance of the CO_2 capture system, and the fuel composition. For analysis, this work uses a supercritical pulverized combustion power plant with post-combustion capture, using a 30% weight monoethanolamine solvent absorption system. The capture system requires 3.6 GJ per tonne of CO_2 recovered, although more advanced solvents could achieve solvent regeneration with 2 GJ of heat duty[60]. To examine the impact of large-scale BECCS deployment, this study uses a system size of 500 MW as the nameplate capacity. The Integrated Environment Control Model is used to assess the energy and water efficiencies of the base plant considering biomass co-firing, CO_2 capture rate, and the compression of CO_2 for transport at 110 bar. In the IECM framework, the efficiency of a coal plant without CO_2 capture is 38.9% HHV. The fuel flow rate is adjusted in the IECM model to achieve the 500 MW capacity based on the energy content of the input fuel.

The model uses a high-level characterisation of the CO_2 transport and storage element of the BECCS value chain. In this model, BECCS facilities and CO_2 storage sites are assumed to co-located within approximately 100 km. To account for energy use and emissions during compression and injection, a 6% leakage rate is assumed in the CO_2 transport and storage system[61]. The cost of transport is estimated to be \$15.4 per tonne of CO_2, based on the capital and operating costs of a 3 Mt CO_2 per year capacity pipeline with a lifetime of 20 years[62]. Additionally, a cost of \$5 per tonne of CO_2 is assumed for CO_2 injection. The water and energy balance for BECCS, although represented in the model, is not described here in detail for brevity.

The following methodology is applied to account for the GHG emissions along BECCS value chain:

- For each input i, the embodied carbon emissions are defined, $CF_i(b)$ for seeds, rhizome, and plantings.
- For each fuel j, an embodied emission factor term is defined for fossil fuels including natural gas, diesel, HFO, biodiesel, and wood pellets.
- Emission factors $EF_{m,f}$ are defined for each transport mode m, and biomass form f.
- Direct and indirect land use change emissions from land conversion, and direct N_2O emissions from fertiliser application are included in the assessment.
- The supply chain carbon footprint, $CF_{SC,ha}$ is defined as the total of all the emissions leading up to the power plant.
- The supply chain carbon emissions are divided by the total number of dry pellets delivered at the plants over the lifetime of the project, PL, to generate a supply chain carbon footprint estimation.

The carbon intensity of the BECCS plant $CI_P(b)$, is defined as a function of the capture rate R_{CCS}, the biomass co-firing rate C_f, and the biomass carbon content, $C_{bio}(b)$, and the net power output of the plant NPO:

$$CI_P(b) = \frac{F_{fuel}(b) \times ((1 - R_{CCS}) \times C_{fuel}(b) - C_f \times C_{bio}(b)) \times C_{C \rightarrow CO_2}}{NPO}$$

The net removal potential of a BECCS facility is thus calculated as the sum of the emissions generated in the supply chain and the negative emissions from the plant. The total cost of a BECCS project is defined as the sum of the cost across each element of the supply chain, with the costs defined as per the data in deliverable 4.1 and 4.2. For a full formulation of the BECCS supply chain used in the modelling framework, see Fajardy[51].

2.5 Biochar

Biochar is produced through the pyrolysis process, which involves heating organic materials in the absence of oxygen. It is a carbon-rich material that is typically derived from biomass, such as agricultural waste, wood chips, or plant residues. Biochar is a heterogenous material that consists of two distinct carbon pools with different degrees of persistence when applied to soil[63]. The persistent aromatic carbon (PAC) pool, which consists of larger clusters of aromatic carbon rings, generally with more than seven aromatic rings, is not susceptible to degradation. The PAC pool has a mean residence time exceeding 1000 years in soil[64,65], independent of common environmental factors such as soil humidity, temperature, freeze-thaw-cycles, biological activity, or agricultural practices like tillage. The semi-persistent carbon (SPC) pool, which contains aliphatic, small aromatic, and heteroaromatic carbon species, is more degradable in soil[66].

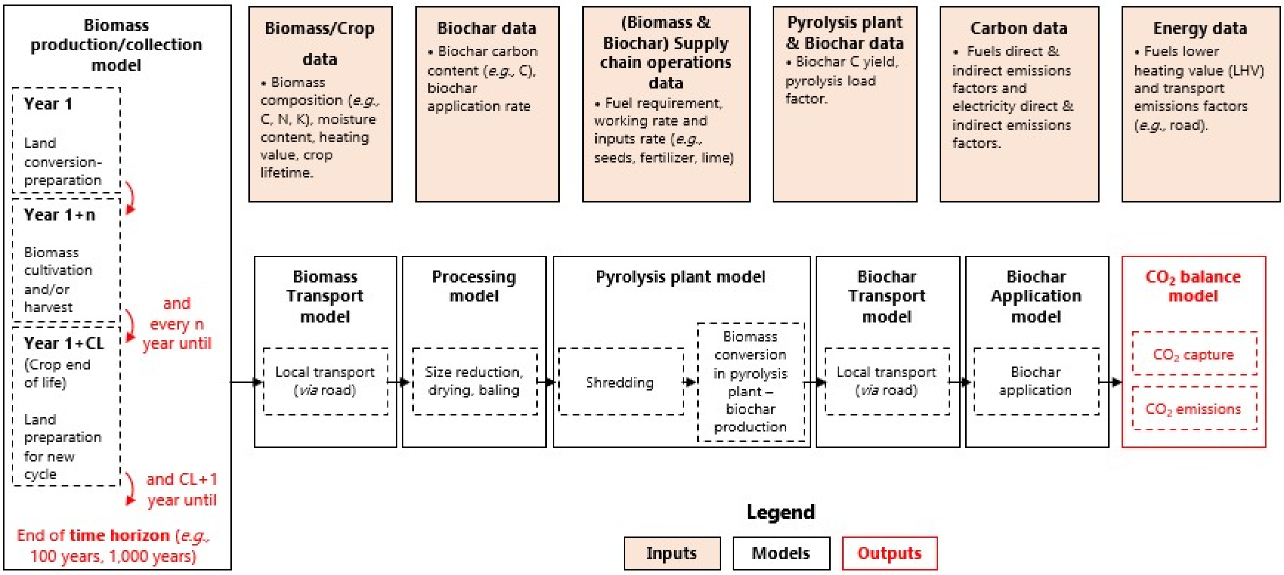

Figure 3: A schematic of the biochar supply chain processes as seen in Chiquier[40].

Modelling biochar as an NETP requires characterising the cultivation of biomass, processing, transport to the pyrolysis plant, and its subsequent application on soil. Different types of crops can be cultivated and analysed, as well as different pyrolysis processes (at different operating temperatures), although in this model, we limit the analysis to pyrolysis processes at approximate temperatures of 400°C and above, as these operating conditions are better suited to produce biochar with a higher fraction of PAC[67].

Three separate plant archetypes are defined to denote the possibility to operate at different scales, with a mix of co-products. They are as follows:

- Small scale facility, using 2,000 tDM/yr, dedicated to produce biochar alone as the product.
- Medium scale facility, using 16,000 tDM/yr, where the bio-oil resulting from the pyrolysis of the biomass may be sold for other applications, *e.g.* low grade heat or fossil fuel replacement.
- Large scale facility, using 185,000 tDM/yr, where the syngas being from the pyrolysis process may be converted to electricity to displace that from the power grid with a conversion efficiency of 38.

To model the longevity of biochar as an NETP, we begin with a function which represents the decay rates of both pools of carbonaceous materials. The decay rate (DR) of biochar in soil can be represented using a two-pool exponential DR [68–70] function as follows:

$$\forall yr, \qquad DR^{Bioch} \quad (yr) = L \times \exp\left(\frac{-ln(2)}{t_{1/2L}} \times yr\right) + \ R \times \exp\left(\frac{-ln(2)}{t_{1/2R}} \times yr\right)$$

where:

- L is the labile (or SPC) fraction of biochar (%).
- R is the recalcitrant (or PAC) fraction of biochar (%). Importantly, $L + R = 1$.
- $t_{1/2L}$ is the labile half-time (yrs).
- and $t_{1/2R}$ is the recalcitrant half-time (yrs).

The rate at which biochar decomposes depends on its properties, specifically how resistant the carbon compounds in the biochar are to both biological and non-biological degradation. This resistance is often measured by the molar ratio of hydrogen to organic carbon (H/C_{org}). The properties of biochar, including its H/C_{org} ratio, are influenced by the pyrolysis temperature and the type of biomass used as the feedstock for biochar production[68,70,71].

Note that there is limited data available to define the decay rates of the recalcitrant pool of biomass in soil[63]. The mechanisms under which they degrade may also be different, and thus, more research is needed to define the decay over time accurately. Woolf et al., 2021[70] presents data to define the exponential decay function to be applied to both pools of biomass. Recent studies have challenged the notion of applying an exponential decay function to the recalcitrant pool of biomass in the absence of sufficient empirical evidence[64]. They assert that such functions are not applicable, and the resulting share of biochar is assumed to be durable. Under this assumption, the decay rates generated by the exponential decay function may be too conservative and underestimate its overall potential to deliver negative emissions at scale.

The biogenic CO_2 sequestration potential of biochar $CO_2\ Bio.Sequestration\ Potential^{Biochar}$ is the initial amount of CO_2 captured biogenically via photosynthesis and sequestrated during the biomass growth. The CO_2 sequestration potential of biochar $CO_2\ Sequestration\ Potential^{Biochar}$ is the amount of CO_2 sequestrated into biochar via the pyrolysis process. However, owing to the decay of biochar in soil, the CO_2 sequestration potential of biochar is written as follows:

$$\begin{aligned}\forall yr, \qquad & CO_2\ Sequestration\ Potential^{\ Biochar}(yr) \\ & = Maximum\ CO_2\ Sequestration\ Potential^{\ Bioch} \quad \times DR^{Biochar}(yr)\end{aligned}$$

NEGEM

$$Maximum\ CO_2\ Sequestration\ Potential^{\ Biochar}$$
$$= CO_2\ Bio.Sequestration\ Potential^{Biochar} \times \eta^{Pyrolysis}_{Biochar}$$

where $\eta^{Pyrolysis}_{Biochar}$ is the biochar C yield (t C in biochar/t C in biomass) of the pyrolysis plant. Here, we assume that $\eta^{Pyrolysis}_{Biochar}$ = 38%. This means that for every tonne of CO_2 biogenically sequestrated within the biomass, only 0.38 tonne of CO_2 is subsequently sequestrated within the biochar, after the pyrolysis.

The CO_2 removal potential of biochar $CO_2\ Removal\ Potential^{Biochar}$ is the difference between $CO_2\ Bio.Sequestration\ Potential^{Biochar}$ and the total CO_2 emissions arising from all steps of the biomass supply chain, including the pyrolysis plant, the supply chain emissions $CO_2\ Emissions^{Biochar}$, and the non-stable C content of the biochar $Non\text{-}Stable\ C\ Emissions^{Biochar}$. Thus, the net removal potential is calculated as follows:

$$\forall\ yr,$$
$$CO_2 Removal\ Potential^{Biochar}(yr)$$
$$= CO_2\ Bio.Sequestration\ Potential^{Biochar} - CO_2\ Emissions^{Biochar}$$
$$- Non\text{-}Stable\ C\ Emissions^{Bioch} \quad (yr)$$

$$Non\text{-}Stable\ C\ Emissions^{Biochar}(yr)$$
$$= Maximum\ CO_2\ Sequestration\ Potential^{\ Biochar} \times \left(1 - DR^{biochar}(yr)\right)$$

$$CO_2\ Emissions^{Biochar}$$
$$= CO_2\ Emissions^{Biochar}_{Bio\ Supply\ Chain} + CO_2\ Emissions^{Biochar}_{Char\ Supply\ Chain}$$
$$+ CO_2\ Bio.Sequestration\ Potential^{Biochar} \times \left(1 - \eta^{Pyrolysis}_{Biochar}\right)$$

$$CO_2 Removal\ Potential^{Biochar}(yr) = CO_2\ Bio.Sequestration\ Potential^{Biochar}$$
$$- CO_2\ Emissions^{Biochar}_{Bio\ Supply\ Chain} - CO_2\ Emissions^{Biochar}_{Char\ Supply\ Chain}$$
$$- CO_2\ Bio.Sequestration\ Potential^{Biochar} \times \left(1 - \eta^{Pyrolysis}_{Biochar}\right)$$
$$- CO_2\ Bio.Sequestration\ Potential^{Biochar} \times \eta^{Pyrolysis}_{Biochar} \times \left(1 - DR^{biochar}(yr)\right)$$

The total GHG emissions from the supply chain $CO_2\ Emissions^{Bioch}$ include:

- direct CO_2 emissions from the combustion of diesel[36] for biomass cultivation, harvest, transport to the pyrolysis plant, subsequent transport and application on soil.
- emissions from natural gas or wood for drying biomass when relevant.
- indirect CO_2 emissions from the production of these fuels[36], and the generation of electricity[36,72] for biomass processing.
- indirect CO_2 emissions due to the manufacture of materials or equipment, aagrochemicals[73–76] (fertilizers, herbicides and lime)[73,74,76].
- direct CO_2 emissions from the pyrolysis plant.
- direct and indirect CO_2 emissions arising from land-use change, *i.e.* (I)LUC.
- direct N_2O emissions arising from the application of nitrogen-based fertiliser during the cultivation of biomass [56,77–79].

Similarly, the total costs of biochar production $Cost^{Bioch}$ includes:

- the cost of leasing land.
- the cost of feedstocks and materials, *i.e.* agrochemicals (fertilizers, herbicides and lime).
- the cost of energy needed for various operations along the supply chain.
- the cost of labour and machinery for the various operations along the supply chain.
- the CapEx and OpEx of the pyrolysis plant.

Overall, the equations and assumptions enable a comprehensive evaluation of the costs and emissions profile of biochar. However, there are significant uncertainties in various components of the supply chain performance of the technology, and this should be scrutinised further as part of sensitivity analysis and uncertainty analysis. Interested readers are encouraged to read Chiquier[40] for a full methodological description of all the variables in the model formulation.

2.6 DACCS

In this formulation, two DACCS archetypes are modelled. Solid sorbent and liquid solvent DACCS are used to capture CO_2 directly from the air. In the liquid solvent DACCS process, high-grade heat (900°C) is supplied by natural gas or hydrogen, and electricity is sourced from the power grid. CO_2 emissions resulting from natural gas combustion are assumed to be captured within the plant limits. In the solid sorbent DACCS process, heat and electricity are both obtained from the power grid, using an industrial heat pump (COP = 3) which converts electricity to low-grade heat (100°C).

The CO_2 sequestration potential of DACCS $CO_2\ Sequestration\ Potential^{DACCS}$, is amount of CO_2 captured at the plant, and subsequently stored into geological reservoirs. The overall stored amount reduces over time due to the very minimal CO_2 leakages associated with geological reservoirs. This means that the maximum CO_2 sequestration potential of DACCS is reached at the point of injection. This is written as follows:

$$\forall yr,$$

$$CO_2\ Sequestration\ Potential^{DACCS}(yr)$$
$$= Maximum\ CO_2\ Sequestration\ Potential^{DACCS} \times P_{geological\ reservoirs}(yr)$$

The total CO_2 and N_2O emissions associated with DACCS include:

- direct CO_2 emissions from the supply of heat, (*i.e.,* combustion of natural gas), at the capture plant.
- indirect CO_2 emissions from the supply of heat, (*i.e.,* production of natural gas), and power (*i.e.,* generation of electricity[36,72]) at the capture plant.
- indirect CO_2 emissions due to the use of electricity[36,72] for CO_2 compression, transport, and storage.

The CO_2 removal potential of DACCS $CO_2\ Removal\ Potential^{DACCS}$ is the difference between the CO_2 sequestrated into geological reservoirs $CO_2\ Sequestration\ Potential^{DACCS}$ and the total CO_2 emissions arising from the capture of CO_2 at the plant, and its subsequent transport and storage $CO_2\ Emissions^{DACCS}$, resulting in a net balance. $CO_2\ Removal\ Potential^{DACCS}$ is calculated over time as follows:

$$\forall yr,$$

$$CO_2\ Removal\ Potential^{DACCS}(yr) = CO_2\ Sequestration\ Potential^{DACCS}(yr) - CO_2\ Emissions^{DACCS}$$

The total costs of DACCS $Cost^{DACCS}$ include:

- the CAPEX and OPEX of the DAC plant, including labour, operating and maintenance costs[80,81].
- the cost of energy, *i.e.* heat[82–86] and power[82,85,87–89].
- the cost of CO_2 transport and storage[62].

For the liquid solvent archetype, the total cost of the system is derived from a conceptual process design published by Carbon Engineering[80] and an NAP report, in which the total cost of CO_2 capture of DACCS is disaggregated by energy cost, CapEx, and OpEx. Particularly, in Keith *et al.*[80], the total levelised cost of CO_2 capture of DACCS is evaluated around $113–168/t CO_2 captured across all configurations[ii]. As these estimates are aligned with the lower bound of DACCS cost reported in the literature[90,91] ($25–1,000/t CO_2), we assume a total cost of DACCS of $400 – 600/t CO_2 captured[80].

For the solid sorbent archetype, the total cost of the system has been reported by commercial providers to be around $1,200/t CO_2 removed[81]. Moreover, in contrast to the liquid solvent process, the solid sorbent archetype is a modular process and operated in a two-step approach, requiring more maintenance. We assume that, after excluding the energy cost, the levelised CapEx, and OpEx each account for 50% of the remaining cost of CO_2 capture.

2.7 Enhanced weathering

Enhanced weathering involves mining, crushing, and grinding basic or ultrabasic rocks to a fine powder which can subsequently be spread on agricultural land, or in a coastal environment to remove CO_2 from the atmosphere or soil. The total CO_2 removal capacity of enhanced weathering is a function of the rate of the weathering reaction, and plateaus once after all the rock has weathered. Smaller particle size distributions weather faster at optimal temperature ranges to capture CO_2 in a mineralised form such as carbonate[92]. However, basic rocks such as basalt have an approximate maximum CO2 sequestration capacity of 0.3 tCO_2/t rock[93], thus requiring significant quantities of mining and crushing rock to achieve a substantial amount of removal.

From Beerling et al., 2020[94], the chemical reactions involved with the rocks weathering process are the formation of bicarbonate ions (HCO_3^-), calcium (Ca^{2+}) or magnesium (Mg^{2+}) ions, from soil drainage waters to surface waters, and the precipitation of calcium carbonate ($CaCO_3$), resulting from the transport of ions HCO_3^- to the ocean, and their reactions with ions Ca^{2+}.

For example, forsterite (a silicate mineral) is dissolved trough the following reaction:

$$CaSiO_3 + 2CO_2 + 3H_2O \rightarrow Ca^{2+} + 2HCO_3^- + H4SiO_4$$

[ii] In Keith *et al.* [80], two DAC plant configurations are investigated. A first one, for which natural gas is used both for heat and power, and a second one, for which natural gas is replaced by electricity for power.

Wollastonite (another silicate mineral) is, instead, dissolved through the following reaction:

$$Mg_2SiO_4 + 4CO_2 + 4H_2O \rightarrow 2Mg^{2+} + 4HCO_3^- + H4SiO_4$$

Eventually, part of the ions HCO_3^- are transported to the ocean, where they are mineralised following the reaction below:

$$Ca^{2+} + 2HCO_3^- \rightarrow CaCO_3 + CO_2 + H_2O$$

Whilst in the first two reactions, 2 moles of CO_2 are sequestrated for 1 mole of Ca^{2+} or Mg^{2+}, 1 mol of CO_2 is emitted back, into the ocean, in the third reaction. Overall, because not all bicarbonate ions are transported to the ocean and then mineralised, it is conventionally assumed that, overall, 1.7 mol of CO_2 is sequestrated per mol divalent cation produced[93–95]. As such, the CO_2 sequestration potential is defined over time (t CO_2/t rock) as follows:

$$\forall\, yr,$$

$$\begin{aligned} CO_2\ Sequestration\ Potential\ (yr) &= \omega\ \times Carbonation\ Potential\ \times CR(yr) \\ &= Maximum\ CO_2\ Sequestration\ Potential\ \times CR(yr) \end{aligned}$$

where:

- $Carbonation\ potential$ is the maximum/theoretical carbonation potential of silicate rocks (t CO_2/t rock).
- ω is the "carbonation to CO_2 sequestration" conversion factor (%), which accounts for the additional drawdown from cation flux into the ocean. As explained above, $\omega = 1.7$.
- CR is the carbonation rate over time (%).

Typically, rock weathering rate WR is a function of soil characteristics, (i.e. temperature[96] and pH[97]), mineral composition, and the size of rock [92,95]. WR can be modelled as a function of the soil pH pH and temperature T and the mineral composition m, using generalized equations as shown in Beerling et al, 2020[94] and Taylor et al, 2016[98] and CR can then be modelled with a shrinking core model, as suggested in Renforth, 2012[93] .

The carbonation rate CR is expressed as follows:

$$CR(t, d_T) = \frac{d_T^3 - (d_T - 2 \times WR \times V_m t)^3}{d_T^3}$$

where:

- d_T is the target rock size after grinding (m),
- WR is the weathering rate of the rock (mol m^{-2} s^{-1}),
- V_m is the molar volume of the rock (m^3 mol^{-1}),
- t is the time (s).

The weathering rate WR is expressed as follows:

$$WR = \frac{\sum_m Mf_m \times Mm_m \times WR_m}{\sum_m Mf_m \times Mm_m}$$

where:

- Mf_m is the molar fraction of the mineral m within the rock (%$_g$),
- Mm_m is the molar mass/weight of the mineral m within the rock (g mol^{-1}),
- WR_m is the weathering rate of the mineral m (mol m^{-2} s^{-1}).

And the weathering rate of a mineral m WR_m, which is applied to a soil with a pH pH, and temperature T, is expressed as follows:

$$WR_m(pH, T) = k_{H^+} \times e^{-\frac{Ea_{H^+}}{R} \times \left(\frac{1}{T} - \frac{1}{298.15}\right)} \times 10^{-n_{H^+} \times pH} + k_{H_2O} \times e^{-\frac{Ea_{H_2O}}{R} \times \left(\frac{1}{T} - \frac{1}{298.15}\right)} + k_{HO^-} \times e^{-\frac{Ea_{HO^-}}{R} \times \left(\frac{1}{T} - \frac{1}{298.15}\right)} \times 10^{n_{HO^-} \times (pH - 14)}$$

where:

- k_i is the rate constant of the individual weathering agent, e.g. [H^+], [H_2O], or [HO^-] (mol m^{-2} s^{-1}),
- Ea_i is the apparent activation energy of the individual weathering agent (kJ mol^{-1}),
- R is the gas constant (kJ mol^{-1} K^{-1}),
- n_i is the reaction order of the individual weathering agent (-).

The carbonation rate of rocks increases as the size of the rocks decreases – it takes about 250 years for 10 µm fast-weathering basalts to be entirely weathered in near-perfect conditions in the EU, whereas it takes more than a millennium for 50 µm ones[40]. Owing to the type and composition of the rocks, dunite rocks weather faster than fast-weathering basalt. Moreover, the carbonation rate is also a function of the soil on which the rocks are applied (i.e., soil temperature and pH). Note that there is no clear consensus on the particle size distribution that is most appropriate for spreading on soil, especially as smaller sizes are favourable for fast kinetics, but they also pose respiratory risks, especially at PM 2.5 and PM 10 levels.

The CO_2 sequestration potential of enhanced weathering is the final amount of CO_2 sequestered as carbonate minerals. This amount increases over time in line with the weathering reaction, after the ground rock has been applied on soil. The CO_2 removal potential of enhanced weathering $CO_2\,Removal\,Potential^{EW}$, is the difference between the stored amount and the total CO_2 emissions arising from the system boundary of the process.

$$\forall yr,$$

$$CO_2\,Sequestration\,Potential(yr) = Maximum\,CO_2\,Sequestration\,Potential \times CR(yr)$$

$$CO_2\,Removal\,Potential\,(yr) = CO_2\,Sequestration\,Potential(yr) - CO_2\,Emissions$$

$$CO_2\,Removal\,Potential\,(yr) = Maximum\,CO_2\,Sequestration\,Potential \times CR(yr) - CO_2\,Emissions$$

The total costs of enhanced weathering include:

- the CapEx and OpEx of the extraction and processing facilities where rocks are excavated, crushed, and milled to size.
- the cost of materials and energy, *i.e.* diesel[99] for extraction, transport, application on soil, and electricity[82,85,87–89] for crushing and grinding.
- the cost of machinery and labour, *e.g.* trucks for the transport of rocks, or tractors for their application on soil.

More details on the costing parameters associated with the different operations that are necessary for enhanced weathering are described in deliverables 4.1 and 4.2.

2.8 Optimisation framework

The modelling architecture combines a dynamic model with an optimisation model. The dynamic modelling framework computes the technical potential to supply NETPs at scale. Following which, the optimisation dimension is used to distribute the supply as per the different national targets for negative emissions. Deliverable

4.3 details different approaches for allocating member state-specific targets for carbon dioxide removal. This work uses the findings from deliverable 4.3 to define the demand for negative emissions as a user input, to subsequently investigate the optimal supply routes.

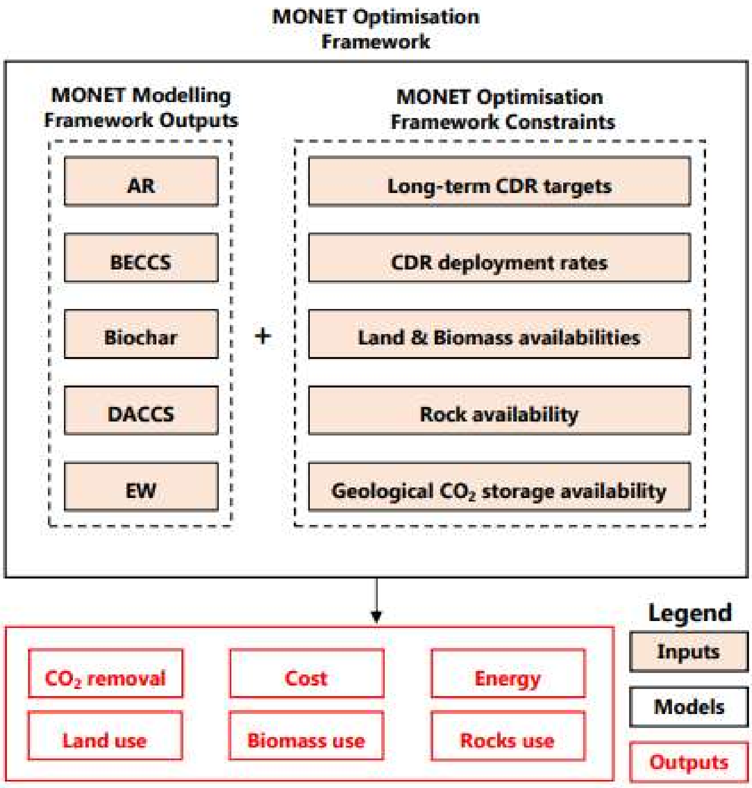

Figure 4: The optimisation framework components and key model inputs and outputs. Figure from Chiquier[40].

In summary, our reference scenarios use the P3 scenario from IPCC, which represents a middle-of-the-road outlook where societal and technological progress follows historical trends. The total NETP requirement is then apportioned based on different burden-sharing principles, as discussed in deliverable 4.3. In sensitivity analysis, we quantify the impact of higher and lower NETP requirements on the overall composition of the supply mix.

Burden-sharing principles based on responsibility, equity, and financial capacity have been explored in the context of carbon dioxide removal[24,100,101]. Under the Paris Agreement, national or regional NETP deployment targets are voluntarily set by the nations themselves, rather than being determined by deterministic analytical approaches. Article 6 of the Paris Agreement offers geopolitical mechanisms for trading emissions credits, including both positive and negative credits. These trades will likely be decided through international negotiations and financial considerations. Interested readers are encouraged to read Honegger et al.[102] and the references therein for a more extensive discussion on the subject.

The model is parameterised with information on the following NETPS:

- Afforestation as a function of the climate (e.g., tropical rainforest, subtropical humid forest, temperate oceanic forest, or boreal coniferous forest), and tree species (i.e., broadleaves and conifers).
- BECCS as a function of biomass type (e.g., miscanthus, switchgrass, willow, wheat straw, and forestry residues), by associated land type (marginal agricultural land, harvested wheat areas, and existing forests for residues).
- Biochar as a function of the same set of parameters as BECCS, but also including the size of the pyrolysis plant as a key decision variable.
- DACCS as a function of the process type (i.e., solid sorbent and liquid solvent CO_2 capture processes).
- Enhanced weathering as a function of the type of basic rock (i.e., basalt and dunite).

A key model constraint which defines the difference between the technical, and commercially viable potential for NETPs is the build rate constraint. In line with the assumptions used in the P3 scenario, technologies are assumed to be scaled at their historical deployment rates based on comparable process systems. Chiquier[40] discusses a maximum build rate constraint of BECCS facilities at 500 MW/yr at the member state level. Each plant offers a total removal capacity of 4.5 Mt CO_2/yr[iii]. The build rates for DACCS facilities are assumed to be identical to BECCS to simplify the assumptions, however, there is little academic evidence to support an evidence-based characterisation of the build rates associated with DACCS[103]. The deployment of biochar is assumed to be constrained by the availability of biomass, and the build rates are assumed to be equivalent to that of BECCS. We assume a maximum build rate of a single mining facility of 450 kton rocks/yr for enhanced weathering[40]. For afforestation, we use a maximum deployment rate based on forest area net change rates, assuming to be equal to 0.83%/yr in the EU based on the IPCC SR15.

In addition, certain limitations exist regarding the accessibility of dedicated land, rocks, CO_2 storage, and biomass. Other constraints define the amount of harvested wheat areas, forest areas, water stress areas, as well as the rates at which biochar and enhanced weathering can be applied to soil. The objective function of the optimisation model is the cumulative total cost of all the NETPs, defined as a function of time. Note that in the case of both BECCS and biochar, the revenues from a sale of electricity, or bio-derivates are factored into estimation of the total costs. The optimisation minimises the total costs incurred in meeting the cumulative NETP deployment targets by 2100. These constraints are discussed in more detail in chapter 3 of Chiquier[40].

[iii] The CO_2 capture rate of a BECCS plant is assumed to be 95%.

3. Quantification of removal potential

This section first presents deterministic analyses on the technical potential to deploy NETPs such as BECCS, biochar, afforestation, and enhanced weathering. The key parameters that influence the overall cumulative potential of different technologies are discussed in each of the respective sections. Characteristic examples are used to illustrate the key differences in technical potential offered by different member states in Europe.

It is crucial to emphasize that studies at the global level have examined this matter. For instance, Smith estimated that soil carbon sequestration and biochar application have a combined potential of 1 Gt C_{eq}/yr or 3.7 Gt $CO_{2,eq}$/yr each for negative emissions. On the other hand, BECCS and DACCS are projected to generate around 3.3 Gt Ceq/yr and between 3.0 Gt C_{eq}/yr to 3.5 Gt C_{eq}/yr, respectively[104]. The estimation with the highest uncertainty lies within afforestation, ranging from 1.0 Gt C_{eq}/yr to 3.4 Gt C_{eq}/yr. In comparison to the other NETPs, enhanced weathering is estimated to offer a net removal potential of 0.2 Gt C_{eq}/yr[105], with comparatively modest impacts on the water, land, and energy requirements. DACCS has the lowest land footprint per ton of carbon removed, which is at least two orders of magnitude larger than all the other NETPs combined[105]. However, DACCS deployment is expected to have the highest impact on the overall energy requirements. However, it remains unclear how these estimates can be translated into deployment quotas at the EU level, which are necessary for policy development.

Pozo et al.[24] assessed the misalignment between the technical potential to achieve negative emissions and the responsibility and financial capacity to implement them. Only 3 – 6 of the EU member states can meet the quotas for negative emissions that were allocated based on historical responsibility, financial capability, or equality. This assessment evaluated the domestic potential for negative emissions based on the availability of agricultural and forestry residues, and dedicated energy crops, the level of reforestation potential, and the amount of available CO_2 storage capacity. They excluded technologies such as biochar and enhanced weathering which may have an important role in delivering negative emissions in Europe. Moreover, they constrain the level of DACCS in the system using long-term geological storage capabilities and electricity grid expansion constraints. Overall, they estimate an NETP potential of 92 Gt $CO_{2,eq}$ across the entirety of the EU. Note that this figure does not align with their estimate of the EU's quota of 266 Gt $CO_{2,eq}$ of cumulative removal as per the responsibility principle, and 325 Gt $CO_{2,eq}$ of cumulative removal as per the financial capability principle. Thus, it is likely that the EU will need to depend on states outside the EU in a collaborative manner to achieve a negative emissions deployment that is line with its responsibility.

3.1 Technical potential of afforestation

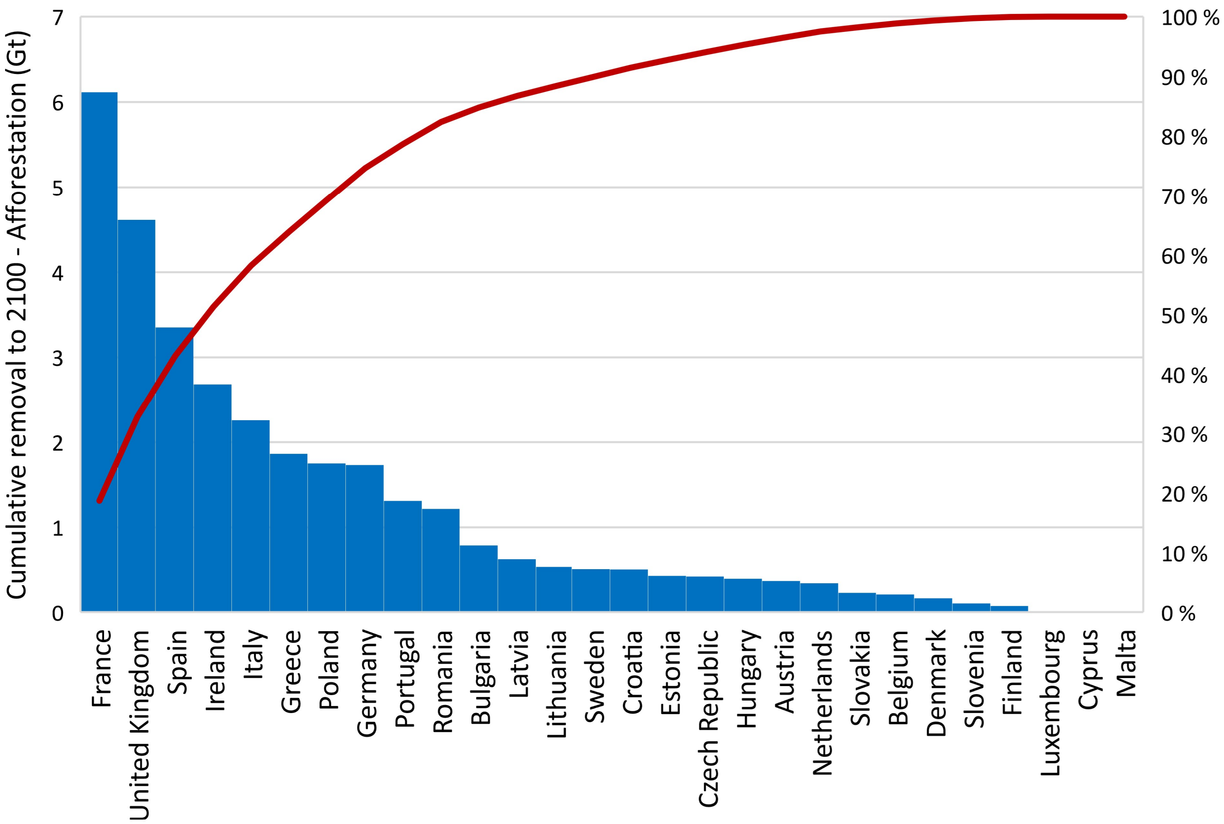

Figure 5: Maximum cumulative CO_2 removal potential to deploy afforestation across the member states in the EU + UK.

Figure 5 presents the technical potential to deploy afforestation at scale in member states of the EU, and in the United Kingdom. Note that 10 out of 28 states considered here are able to offer more than a cumulative removal of 1 Gt $CO_{2,eq}$ by 2100. The countries with the largest potential are France (6.11 Gt $CO_{2,eq}$), United Kingdom (4.62 Gt $CO_{2,eq}$), Spain (3.35 Gt $CO_{2,eq}$), Ireland (2.69 Gt $CO_{2,eq}$), and Italy (2.26 Gt $CO_{2,eq}$). Countries with the lowest potential for afforestation are those with the smallest available land areas for reforestation such as Slovenia, Finland, Luxembourg, Cyprus, and Malta. Some of these states are smaller than the other member states, thereby offering little potential for foresting its lands. Finland is a curious example as it is a country with substantial quantities of existing forests, and thus comparatively less land available for reforestation. Overall, the sum of domestic natural potential offered by afforestation under the base assumptions of this study is 32.7 Gt $CO_{2,eq}$.

3.2 Technical potential of BECCS

Figure 6 illustrates the technical potential to deploy BECCS in member states of the EU, and in the UK, using miscanthus as the primary feedstock. The key distinction between figures 5 and 6 is the considerably larger scale of deployment achievable via BECCS. In particular, owing to the favourable conditions for dedicated energy crops, Spain offers more than 20 Gt $CO_{2,eq}$ of cumulative removal by 2100. Interestingly, if Spain is removed from the list of member states, then the overall potential to deploy BECCS using miscanthus reduces greatly and appears to be comparable to the afforestation potential. This suggests that Spain has a critical role in delivering BECCS at scale, and the EU must develop strategies and incentives to unlock the very significant technical potential available to Spain.

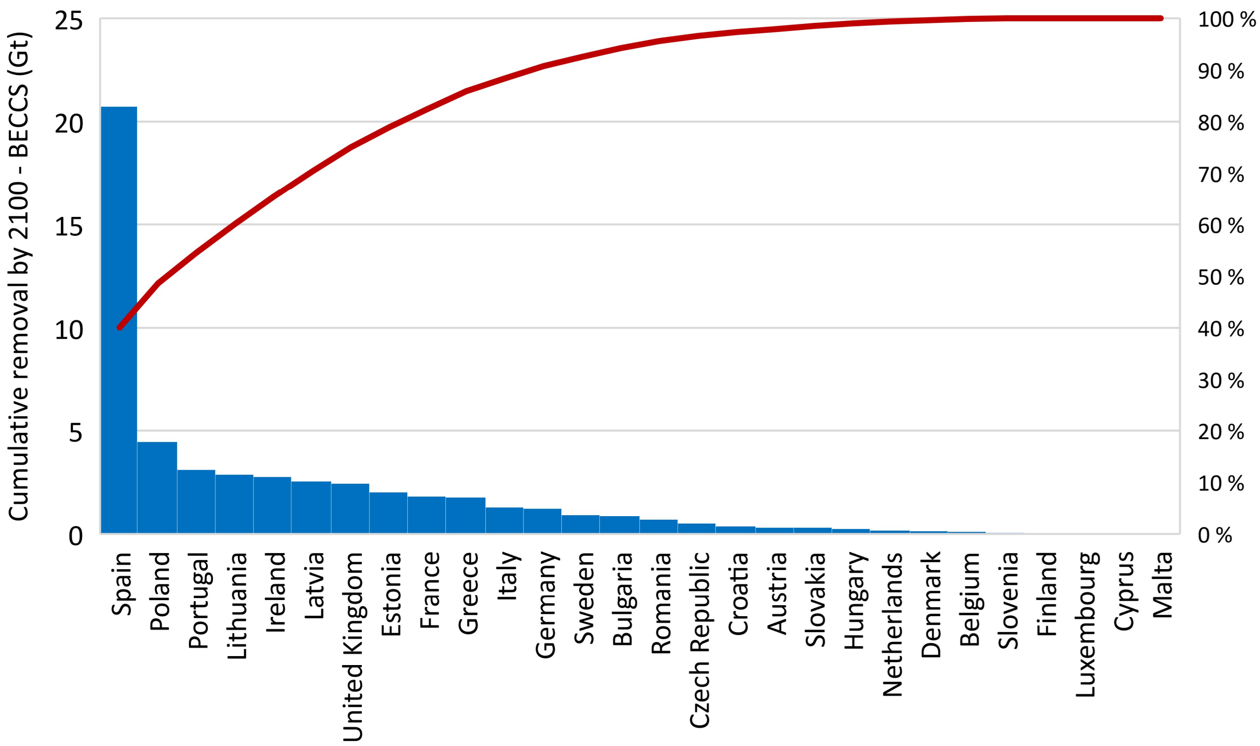

Figure 6: Maximum cumulative CO_2 removal potential to deploy BECCS using locally grown miscanthus in marginal agricultural land across the member states in the EU + UK.

Likewise, Figure 7 illustrates the member states that have the highest potential for BECCS when willow is utilised as the main feedstock. The countries with the largest potential for BECCS using willow are Spain (20.3 Gt $CO_{2,eq}$) offering 40% of the EU's overall potential, Poland (7.07 Gt $CO_{2,eq}$), Lithuania (5.38 Gt $CO_{2,eq}$), United Kingdom (5.00 Gt $CO_{2,eq}$), and Ireland (4.88 Gt $CO_{2,eq}$), and their relative importance changes substantially depending on the type of dedicated-energy crop. For example, Portugal can deliver 3.10 Gt $CO_{2,eq}$ via BECCS using miscanthus if it is grown as the primary energy crop, whereas it can only deliver 436 Mt $CO_{2,eq}$ in the event that willow is grown as the primary feedstock. This is largely due to the differences in the yield between both crops and their downstream potential for CO_2 removal. This suggests the need to undertake a strategic assessment of the NETP potential using different feedstocks for BECCS, while also accounting for the region-specific co-benefits of growing different feedstocks.

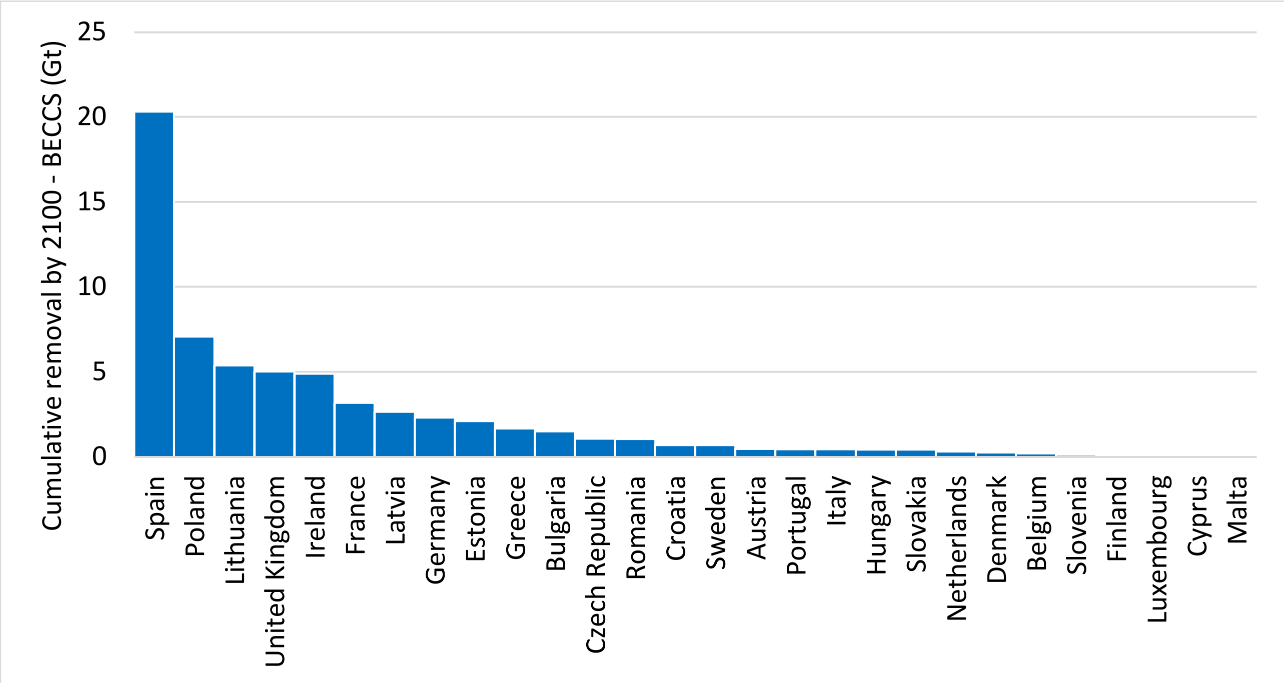

Figure 7: Maximum cumulative CO_2 removal potential to deploy BECCS using locally grown willow in marginal agricultural land across the member states in the EU + UK.

Countries with the lowest potential for BECCS are Slovenia, Finland, Luxembourg, Cyprus, and Malta. Overall, there is scope to deploy between 50 – 67 Gt $CO_{2,eq}$ via BECCS using marginal agricultural land. These figures can be increased by using agricultural and forestry residues, along with wheat straw from lands where wheat is planted. Note that while Spain has the highest potential for BECCS deployment due to its ample land availability and favourable crop growth conditions, it is possible for BECCS plants to be installed in other locations. In such cases, Spain and other leading countries would become suppliers of feedstock for the BECCS facility. Developing policies and incentives that encourage trade between nations with the highest and lowest potentials will be key to ensuring that the EU operates with the lowest level of price and compliance risk. Thus, member states with the greatest potential to supply feedstock may directly trade with a region that has lower potential with each profiting from the trade. Geopolitically, this provides a degree of insulation from the dynamics of the supply market outside of the EU, thereby offering some degree of security of supply.

3.3 Technical potential of biochar

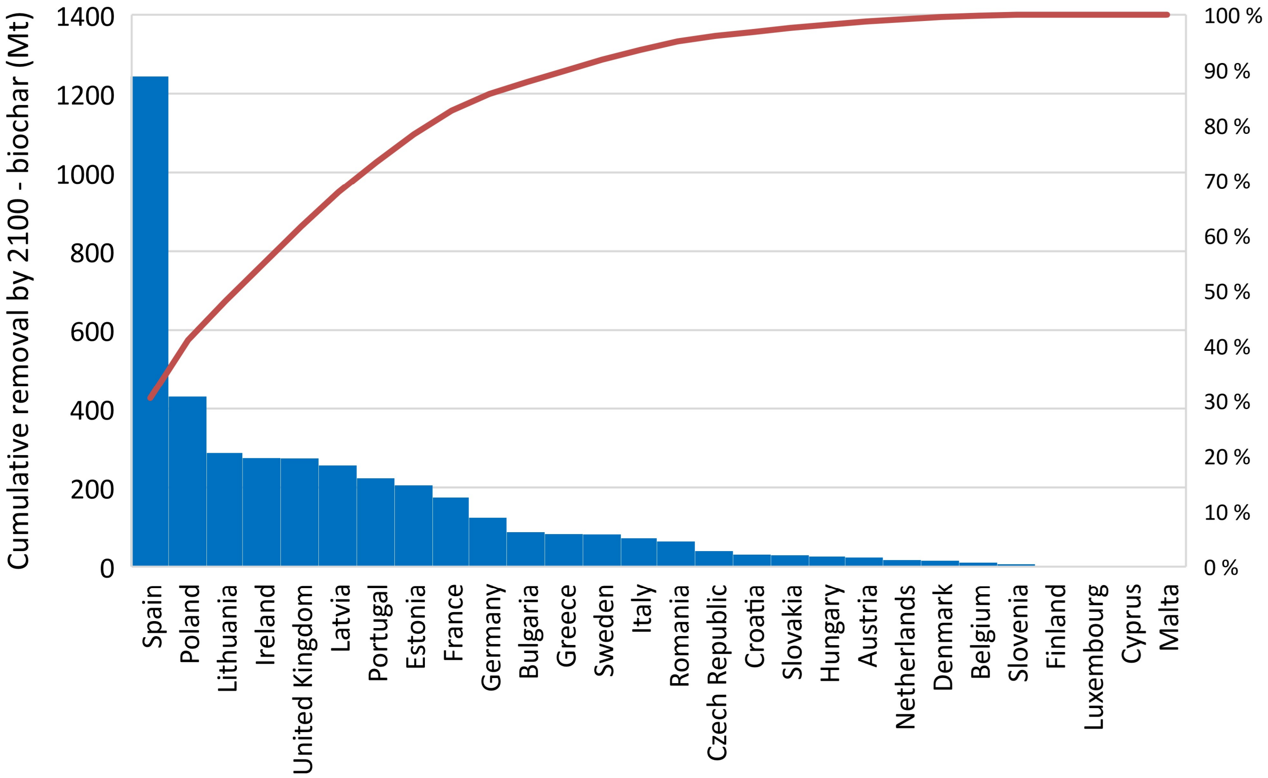

Figure 8: Maximum cumulative CO_2 removal potential to deploy biochar using locally-grown miscanthus in marginal agricultural land across the member states in the EU + UK.

Figure 8 highlights the overall potential to deploy biochar by using miscanthus in a pyrolysis process to remove CO_2 from the atmosphere. This assumes that all the miscanthus grown on marginal agricultural land is pyrolysed and spread on soil. Therefore, this is an alternative counterfactual to the scenario in which BECCS is used for CO_2 removal. Intuitively, the deployment of biochar does not translate into the greatest amount of CO_2 removal per unit of biomass consumed. The pyrolysis process produces biogases, oils, and other solid derivatives that are carbon-neutral along with the biochar fraction. In this instance, it is difficult to fully utilise the carbon content in the biomass for CO_2 removal without applying CCS to the end-uses which involve biogases, oils, and other derivatives. Here, it is worth noting that at most 40% of the original carbon content in biomass is retained in the biochar. However, all of this is not in a permanent state and will degrade over time. Therefore, the overall carbon removal efficiency of the char is comparatively lower than technologies such as BECCS. However, the overall value of this route will be contingent on the demand for the other products. In total, biochar offers the technical potential to achieve a cumulative removal of 4.08 Gt $CO_{2,eq}$ by 2100 using miscanthus grown in marginal agricultural land. Like BECCS, this estimate may increase depending on the availability of other land areas and feedstocks for pyrolysis.

3.4 Technical potential of enhanced weathering

In contrast to the aforementioned technologies, the overall technical potential for enhanced weathering is primarily a function of two parameters – the overall availability of the rock, and the weathering rate. There is a high uncertainty on the weathering rate and the cost of CO_2 removal may vary by several orders of magnitude as a result. The overall availability of different alkaline rocks is well understood and characterised by datasets as tabulated in deliverable 4.2. Figure 9 presents the cumulative removal of CO_2 using enhanced weathering by 2100. Note that this estimate is contingent on the soil and the environment conditions, both of which are difficult to predict over a long period of time. Although permanent, the level of certainty in performance for this technology is very low and it should be analysed with projections of climate evolution in future scenarios.

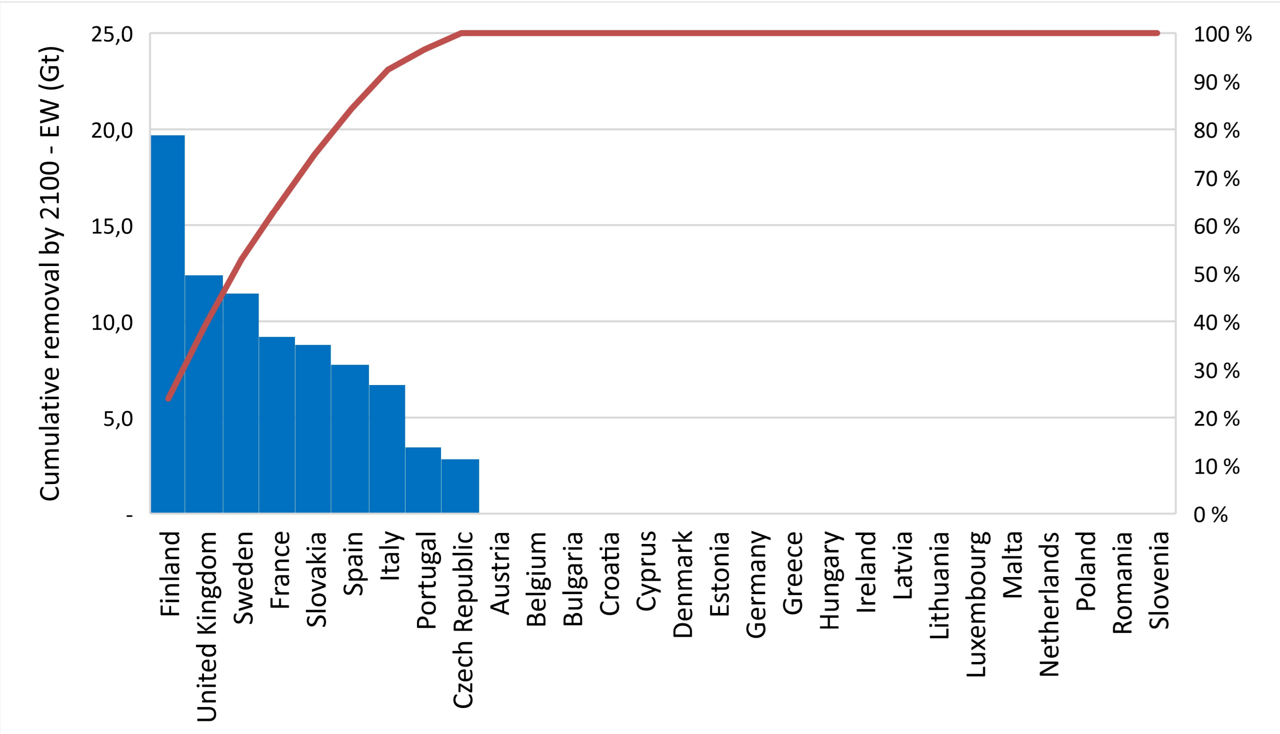

Figure 9: Maximum cumulative CO_2 removal potential to deploy enhanced weathering using locally-sourced basalt across the member states in the EU + UK. Note that constraints on land availability for application as per region has not been factored into the country-wide supply potentials. Thus, the overall technical potential is solely a function of the rock availability and the weathering rate. This result assumes that all of the rocks are crushed and ground and spread on soil, and at least 5% of the rock weathers over a period of 80 years. This assumption is likely to be conservative, however, the overall weathering rate is uncertain and will require field studies for further verification.

Given the distribution of rocks in the EU member states and the UK, 9 out of the 28 states considered in this study have the capacity to supply basalt to support a large-scale implementation. The countries with the largest rock availability are Finland, United Kingdom, Sweden, France, and Slovakia. Given that the most favourable weathering conditions may be obtained outside the region of supply, there is value in trading the rock with regions that are best suited for application. Importantly, the amount of negative emissions offered by this technology will depend on how much of the rock has weathered over time, and greater CO_2 removal is achieved when a higher fraction of the rock weathers over time.

3.5 Summary of the technical potential assessment

An overall portfolio of BECCS, afforestation, and some amounts of enhanced weathering are likely to offer the highest potential for NETPs in the EU, albeit with high uncertainty on enhanced weathering. Under the base assumptions of this assessment, this translates into a technical potential to deploy around 100 Gt $CO_{2,eq}$ by 2100, using afforestation and BECCS alone. If maximally utilised, this approach will use nearly 70% of the overall CO_2 storage available to the EU, thus only leaving 30% of the remaining storage capacity for mitigation activities involving industrial CCS, blue hydrogen, sustainable aviation fuel production, DACCS, etc. Thus, in practice, the overall quantities of BECCS in the EU are likely to be lower than the maximal technical potential figure of 67 Gt $CO_{2,eq}$ by 2100. However, it is worth noting that the overall potential to deploy BECCS has not included an assessment of agricultural or forestry residues. This will increase the overall potential to supply BECCS, but notably, the deployment of BECCS is more likely to be constrained by storage availability than technical supply constraints. While a lower range P3 target of 81 Gt $CO_{2,eq}$ by 2100 (21% of the total target) is achievable through BECCS and afforestation, a higher range P3 target of 192 Gt $CO_{2,eq}$ by 2100 (47% of the total target) will require significant deployment of DACCS, or investments outside the EU member states and the UK, with rigorous monitoring and verification in place for quality control.

3.6 Cost-optimal portfolio of NETPs in the EU member states and the UK

In this analysis, we examine a cost-effective portfolio of NETPs using the following technologies: afforestation, BECCS, biochar, DACCS, and enhanced weathering. The goal of the optimisation is to achieve the cumulative NETP targets from 2020 to 2100 that align with the 1.5°C ambition in the Paris Agreement. These cumulative CO_2 removal targets represent the remaining budget for the EU and the UK, as derived from the IPCC P3 pathway.

We apply a burden-sharing principle based on responsibility to determine these targets, as explained in deliverable 4.3 of the project. It is assumed that the regions collectively need to remove 81 Gt $CO_{2,eq}$ by 2100. Importantly, the problem formulation facilitates trade between member states of the EU and the UK, thus deployment of NETPS is not necessarily constrained by the supply in that region alone. The supply of feedstocks includes agricultural and forestry residues in addition to dedicated energy crops for BECCS. Both basalt and dunite are considered as feedstocks for the enhanced weathering process.

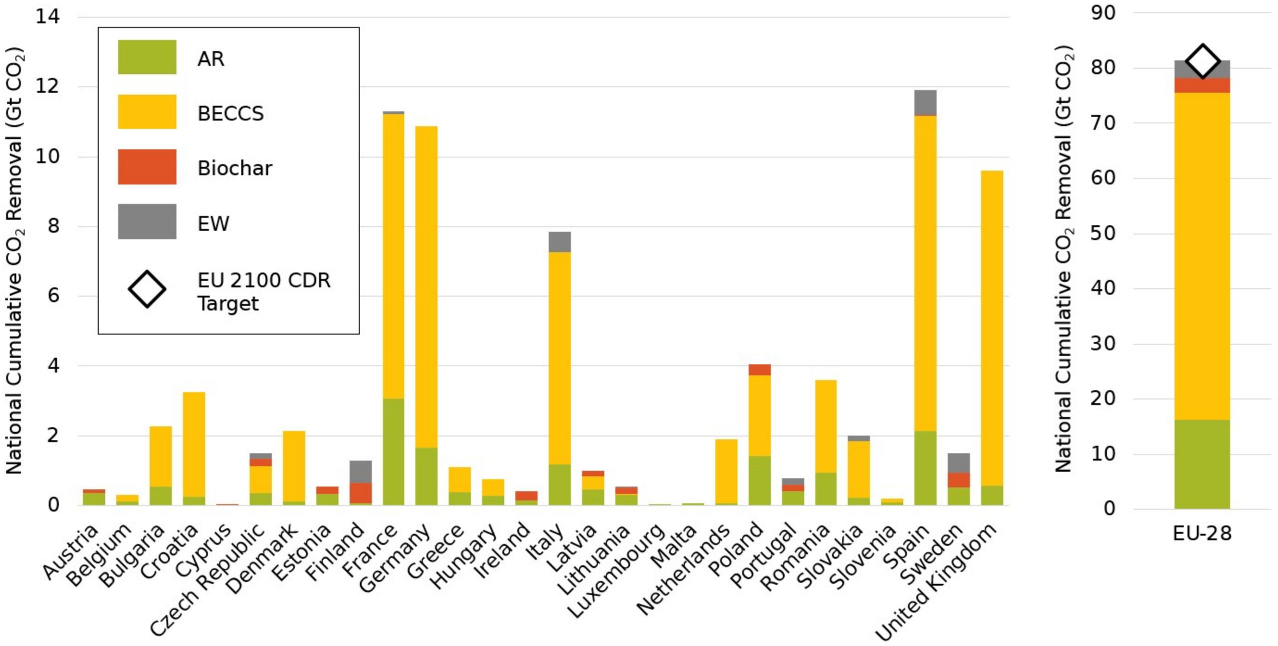

Figure 10: Cost-optimal cumulative CO_2 removal by 2100 for each EU Member State and the UK. Deterministic optimisation results as initially presented in deliverable 7.3 which has been confirmed with further analysis. Left: National breakdown of NETPs; Right: Aggregate portfolio of NETPs.

Overall, the cost-optimal portfolio of NETPs is comprised of BECCS (59 Gt $CO_{2,eq}$), afforestation (16 Gt $CO_{2,eq}$), biochar (4 Gt $CO_{2,eq}$), and enhanced weathering (2 Gt $CO_{2,eq}$). In this instance, DACCS is not deployed at all, as it is more expensive than the other technologies and the demand does not outstrip the technical potential in the EU member states.

By the end of the century, it is projected that BECCS will account for 73% of the total CO_2 removal achieved. This dominance can be attributed to its relatively high cost-efficiency. In other words, BECCS can achieve high CO_2 removal rates while using fewer resources compared to other technologies, all at a lower cost, in part owing to the revenue from the sale of electricity. The deployment of BECCS is primarily observed in France, Germany, Spain, and the UK. When it comes to selecting the feedstock for BECCS, energy-dedicated crops are the preferred choice. Miscanthus constitutes the majority at 55%, followed by forestry residues at 40%, and Willow at 5%. Afforestation is the second most deployed NETP, with 20% of the cumulative removal by 2100, mostly in temperate climates, that are favorable for CO_2 sequestration. However, the deployment of afforestation is limited by its national expansion rates that are already above historical afforestation rates.

Biochar accounts for only 5% of the total CO_2 removal in the EU member states and the UK. It leads to competition for land and biomass resources and serves as a substitute for BECCS in regions where CO_2 storage is limited. It is deployed in Sweden to address the barrier of insufficient CO_2 storage for BECCS deployment. It is important to note that the CO_2 transport is limited via pipelines for short distances up to 100km. Therefore, this study does not consider the possibility to transport CO_2 via ships to Norwegian CO_2 storage reservoirs, for example. The inclusion of other CO_2 transport options may increase the deployment potential for BECCS in the EU.

In some EU Member States, BECCS and biochar deployment is nationally limited by the availability of land, and subsequent biomass availability (*i.e.,* energy-dedicated crops, such as Miscanthus or Willow, or forestry residues). A lack of biomass availability is a key driver for the deployment of enhanced weathering (2% of the total NETP deployment), using basalt and dunite rocks, as seen in Italy or Spain. Moreover, the deployment of enhanced weathering is limited by maximum build rates (*i.e.* the number of mining facilities that can be built annually). Importantly, there are concerns about the toxicity impact of basalt and dunite rocks, when weathering in soil (see Deliverable 1.5 for more details). Therefore, the impacts of the deployment of enhanced weathering needs to be explored in more detail to prevent negative ecosystem impacts.

The lack of CO_2 storage availability as observed in Finland, coupled with a limited supply of energy-dedicated crops, and areas with afforestation potential would make Finland's 2100 quota infeasible with a limited portfolio of NETPs. However, the high availability of forestry residues in Finland can be used to deploy biochar cost-effectively and meet its 2100 quota. This example showcases why a portfolio approach to delivering negative emissions at scale will support an equitable deployment trajectory.

Overall, the EU member states, and the UK can deploy enough CO_2 removal by 2100 to meet the quota in the IPCC P3 pathway with overshoot. More analysis is needed to understand the impact of sensitivities and uncertainties on the cost profile of the NETPs, whilst simultaneously accounting for uncertainties in long-term prices of fuels, energy, materials, and equipment. This analysis is underway as part of an extended article on the cost-optimal deployment of NETPs in the EU, considering the key uncertainties in the design and operation of the NETPs following engagement with project partners and external stakeholders. This informs Monte Carlo analysis and stochastic optimisation model evaluations to define the uncertainties in the technical and commercial potential assessment on NETPs.

4. Conclusion and recommendations

This report has examined the cost-optimal deployment of negative emissions technologies across EU member states. The findings suggest that the EU member states have sufficient technical potential (largely using biogenic resources) to deploy CO_2 removal to meet a removal quota in line with an IPCC P3 trajectory. The technical potential using afforestation (on areas with reforestation potential) and BECCS (using dedicated energy crops on marginal agricultural land) amounts to approximately 100 Gt $CO_{2,eq}$ by 2100. Note that the use of BECCS with agricultural and forestry residues will increase this figure, however, in practice, the likely potential will be constrained by the amount of CO_2 storage capacity in the EU. Similarly, afforestation potential may be constrained by the forest expansion constraints. If the EU were to take a share of the overall CO_2 removal quota that is in line with its capability to finance investment, the overall NETP deployment quota by 2100 will increase to approximately 192 Gt $CO_{2,eq}$. Thus, requiring the deployment of technologies such as enhanced weathering, which is often associated with more speculative assumptions.

Higher removal quotas will likely necessitate the deployment of DACCS at a higher cost. Though, there is limited scope to deploy additional DACCS within the EU member states owing to CO_2 storage constraints. These findings underscore the significance of cross-border collaboration and the development of supportive policy frameworks to effectively implement these technologies. European regions such as Norway may offer additional CO_2 storage potential to support further deployment of engineered removals in the continent.

The technical potential assessment highlighted the vast NETP potential offered by several EU member states. However, it also highlights that not all states have enough indigenous resources to meet their own removal quotas. Cross-border collaboration can help address challenges such as the availability of CO_2 storage capacities, limited supply of energy-dedicated crops, and suitable areas for afforestation.

One key insight from the study is the critical importance of BECCS and afforestation in achieving cost-effective and scalable negative emissions within the EU member states. BECCS can significantly contribute to cost-optimal CO_2 removal while utilising fewer resources compared to other options, leading to an overall share of 73% of the NETP portfolio. Afforestation supplies 20% of the cost-effective NETP potential with additional benefits for biodiversity and the ecosystem, and it is mainly limited by the availability of areas with reforestation potential, and the expansion rates associated with natural forests. The remainder of the portfolio is made up of biochar and enhanced weathering, and the deployment of DACCS is limited to portfolios with greater ambitions on CO_2 removal in Europe.

Note that a key limitation in this assessment is the lack of a detailed assessment of non-financial objectives as a decision criterion in determining the portfolio of NETPs. In general, an appropriate portfolio needs to be developed at the EU level to have a minimal negative impact on the planetary boundaries, while simultaneously promoting sustainable development goals. Multi-objective optimisation using non-economic objectives may help address some of these limitations. Moreover, the capacity for the EU member states to meet both higher and lower shares for CO_2 removal as per the multiplicity of scenarios from the IPCC also needs further investigation.

Moreover, scenario-based analysis and uncertainty assessments are needed to appropriately contextualise these findings for effective policy development. This is part of ongoing research within WP 4 and 7 of the NEGEM project, where some of the limitations described earlier in this work is to be addressed via an extended article.

To prepare this report, the following deliverable has been taken into consideration:

D#	Deliverable title	Lead Beneficiary	Type	Dissemination level	Due date (in MM)
D 1.1	Justification of NETPs chosen for the NEGEM project	ETH	Report	CO	6
D 1.4	Comprehensive sustainability assessment of Bio-CCS NETPs	VTT	Report	PU	12
D 1.5	Comprehensive sustainability assessment of geoengineering and other NETPs	ICL	Report	PU	24
D 4.1	NETP database	ICL	Database	PU	4
D 4.2	Bio-geophysics database	ICL	Database	PU	12
D 4.3	Member State targets	ICL	Report	PU	15
D 8.1	Stocktaking of scenarios with negative emission technologies and	VTT	Report	PU	8

	practises				
D7.3	Link MONET-EU and JEDI	ICL	Report	PU	24
D 4.4	Software tool prototype	ICL	Report	PU	24

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