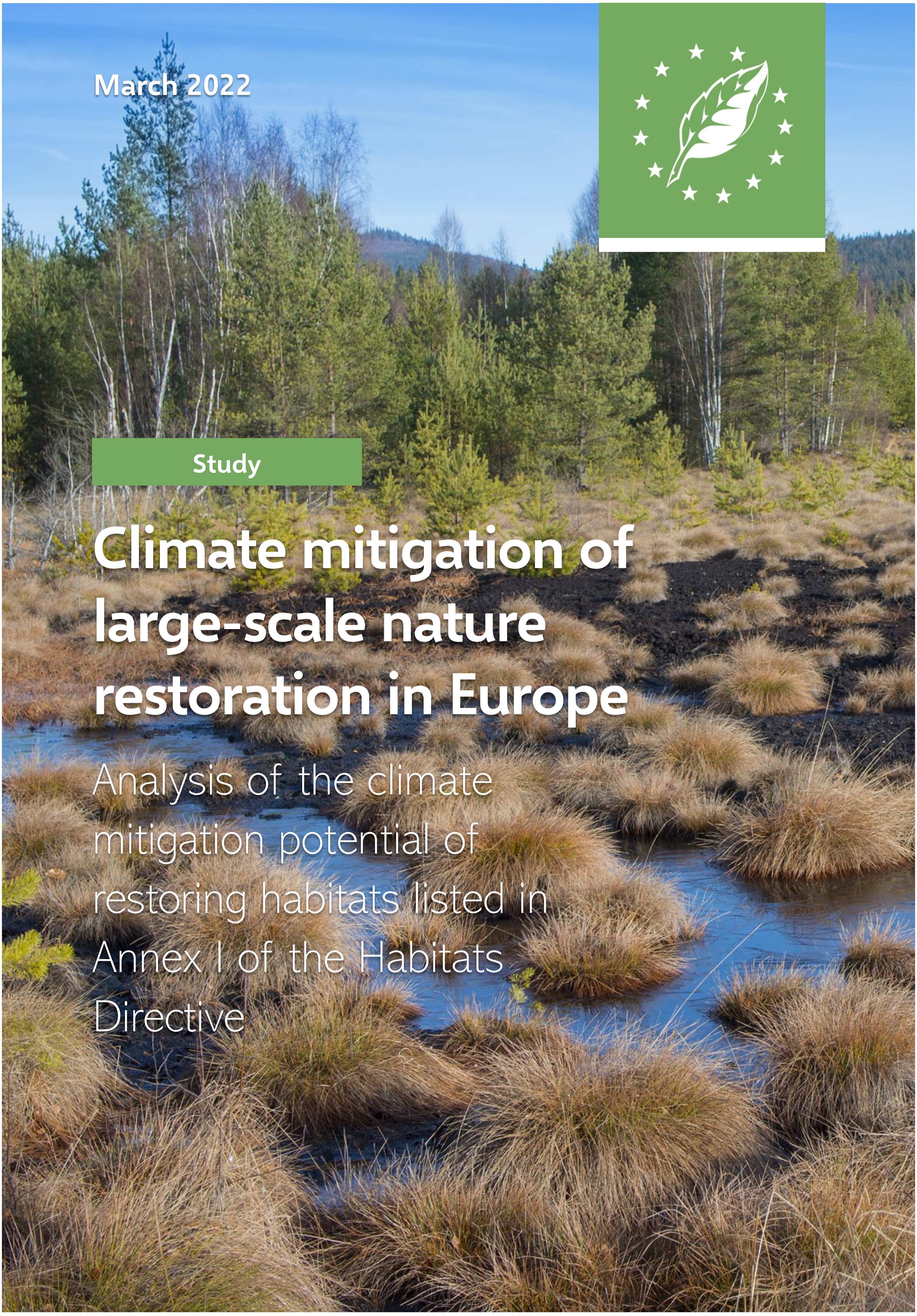

March 2022

Study

Climate mitigation of large-scale nature restoration in Europe

Analysis of the climate mitigation potential of restoring habitats listed in Annex I of the Habitats Directive

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CONTENTS

1. EXECUTIVE SUMMARY

The restoration of habitats listed under Annex I of the EU Habitats Directive has great potential to deliver climate mitigation benefits through increasing carbon sequestration and storage and avoiding emissions from degraded ecosystems. This explanatory note and the accompanying analysis explore the feasibility of **ranking Annex I habitats based on the climate mitigation potential of their restoration.**

The European Commission is expected to propose a set of **legally binding nature restoration targets** in early 2022. Alongside their primary objective to restore degraded ecosystems, the targets will address habitats with the greatest potential to capture and store carbon, thereby contributing to the achievement of both EU biodiversity and climate goals. This creates a great opportunity for Member States to design restoration plans which maximise synergies between climate mitigation and adaptation and biodiversity conservation, including restoration of habitats protected under Annex I of the Habitats Directive.

To compare the climate mitigation potential of Annex I habitats, **habitat types were ranked in terms of restoration priorities to maximise climate co-benefits** using (1) Member State data on Annex I habitats, and (2) information on the carbon storage and sequestration potential of those habitats (based on a review undertaken for the European Environment Agency (EEA) in Hendriks et al. (2020)). The results of this prioritisation exercise are presented in the form of a spreadsheet. This explanatory note provides the context for this exercise, discusses its value, and provides key considerations for the interpretation of the resulting figures.

The following **key factors must be considered**, prior to restoration, when prioritising Annex I habitats for restoration for carbon benefits

- Whether **protection of the existing habitat to reduce pressures is more efficient** for protecting carbon stocks and sequestration than active restoration of degraded habitats.
- The **potential trade-offs** between biodiversity conservation and natural carbon capture and storage. For Annex I habitats, restoring the biodiversity value of the habitat should be the primary objective.
- The **feasibility of restoration** considering the current state of the ecosystem, its context, and potential to recover.

It is important to assess the **factors affecting the success of restoration for carbon benefits,** including site conditions, the time needed to achieve restoration, the costs and benefits of restoration actions, the permanence of carbon gains, and how to monitor carbon flows.

The EU level ranking exercise developed and tested here is a useful starting point to help prioritise habitats for restoration and protection to maximise carbon co-benefits. However, the precise carbon sequestration and storage figures presented should be treated with caution. There are various **limitations and uncertainties** which must be considered. These include the availability and quality of the information on carbon storage and sequestration potential of each habitat; the local and temporal variations in carbon potential within habitats, the accuracy of habitat areas reported by Member States; and the assumptions surrounding sequestration performance of restored or recovering habitats.

Overall, this analysis shows that although it is not currently feasible to give precise numerical estimates of the carbon benefits of Annex I habitat restoration, there is good evidence for the strong carbon sequestration potential of restoring key habitats, namely wetland and forest ecosystems.

In addition, the following **key conclusions** are highlighted when considering the climate mitigation potential of Annex I habitat restoration more broadly:

- Restoration can deliver crucial ecosystem service benefits, including climate mitigation. If all Annex I habitat for which enough information is available (excluding sparsely vegetated and marine habitats) were restored, the restored area of 47.2 Mha could sequester around 80 MtC/ year. This covers 191 of the 233 Annex I habitats in 26 EU Member States (excluding Romania due to issues with reported area values). This is not a precise estimation, but an order of magnitude for the potential sink capacity of restored Annex I habitats without considering the current baseline scenario. It is based on currently available data for the theoretical maximum capacity after full restoration, which in some cases could take up to 100 years to achieve. It is important to also understand that this figure does not represent additional sequestration above a business-as-usual scenario as it does not take account of current losses and gains in carbon stocks from degraded ecosystems. The state of current knowledge is not enough at this aggregate level to determine whether this estimate is on the high side or low side, or how long it would take to achieve this level of sequestration.

- The restoration of the biodiversity value of the Annex I habitats should be the primary objective, in line with the aim of the EU Habitats Directive. Trade-offs between biodiversity conservation and increasing carbon storage and sequestration must be managed to ensure climate benefits do not come at the cost of the biodiversity and ecosystems that sequester and store carbon.

- Avoiding losses of carbon from habitats is more efficient in climate mitigation than restoring degraded habitats whilst others decline. Conservation of intact habitats is therefore key to ensure the protection and permanence of existing carbon stocks. Moreover, protection can be more efficient as restoration may not fully re-establish carbon sequestration and storage potential, and it may take decades to improve in condition. The 191 Annex I habitats for which enough information is available considered in this study hold an estimated 5564 MtC – 17807 MtC (over 87 Mha).

- Existing data sources can support Member States to identify broad restoration priorities to maximise biodiversity and carbon benefits but could be substantially improved with better more fine grained data and long-term monitoring. However, important limitations to estimating carbon restoration potential must be carefully considered.

- Restoration is complex and reflects the dynamic nature of natural systems. Key factors affecting the potential of restoration to deliver long-term carbon benefits must be assessed at the site level. Additionally, it is critical to ensure the permanence of carbon gains through long-term protection and management of pressures in restored sites. Climate change will increasingly modify the ability of restored habitats to reach the condition of undegraded habitats and will increase the risk of losing carbon gains in natural hazards such as forest fires, droughts, floods, and landslides.

2. AIM OF STUDY

The aim of this explorative study was to examine the feasibility of using Member States' reported data on habitat types protected by the EU Habitats Directive (Annex I habitats) and information on carbon removal potential of habitats to rank restoration priorities to maximise co-benefits in terms of carbon removals, whilst achieving biodiversity conservation objectives according to the EU nature directives. This explanatory note covers the most important considerations when prioritising restoration and decision-making. The note discusses the important decisions to take prior to restoration, the factors affecting the impacts of ecosystem restoration on carbon storage and sequestration, the limitations of the available data, and the outlook for restoration in terms of political developments and climate change risk. The annex provides more detail on restoration measures by habitat group and discusses the interpretation of the results of the ranking exercise.

We compiled a spreadsheet[1] with Annex I habitat information and carbon storage and sequestration information for most of the Annex I habitats, based on a literature review done for the European Environment Agency (EEA) in 2020 (Hendriks et al, 2020). We used this to carry out a prioritisation exercise to rank the habitats according to their carbon storage and sequestration potential, using the area to be restored as an input to calculate the maximum possible carbon removal potential. We included data on the proportion of habitat area that is degraded, and therefore potentially losing carbon, and the proportion outside the Natura 2000 network, to help identify where restoration actions outside protected areas can achieve additionality in terms of carbon. We compiled a literature log of the key studies that have informed the analysis, including literature covering biodiversity value, carbon stock and sequestration value, restoration success, timeframe to reach good habitat condition following restoration and adaptation to climate change at the ecosystem- and habitat level.

The spreadsheet can be used as a basis for prioritisation, and the calculations can be adjusted depending on the overall aim of the exercise. The tool is useful as a starting point for prioritising restoration efforts aiming to deliver both climate mitigation and biodiversity conservation as priority outcomes. There are however some uncertainties associated with the data, and the numbers should be treated with some caution. This note explains the key issues associated with the dataset.

[1] Prioritisation spreadsheet available on demand.

We recommend that the prioritisation exercise is critically examined in view of the challenges described in this explanatory note, the information available at Member State and local level, and possible trade-offs between biodiversity conservation priorities and carbon objectives. This explanatory note and the literature log could form the basis for a more in-depth paper on decision-making on restoration priorities. The literature log and an Excel spreadsheet with the data and analysis is available on request from IEEP.

3. EU POLICY OBJECTIVES FOR BIODIVERSITY AND CLIMATE

Ecosystem restoration and habitat protection play a central role in efforts to mitigate climate change, support climate adaptation and resilience, and halt biodiversity loss (Gregg et al, 2021). However, progress on landscape-level ecosystem restoration across the EU has thus far mostly been small-scale. The EU Biodiversity Strategy for 2030[2] sets ambitious targets to improve and widen the EU-wide network of protected areas (including Natura 2000 areas and green infrastructure or ecological corridors between protected areas) and to implement an EU nature restoration plan. The European Commission will put forward in early 2022 a proposal for legally binding EU nature restoration targets to restore degraded ecosystems, while requesting Member States to ensure no deterioration in the conservation status and trends of all protected habitats and species by 2030. Member States will also have to ensure that at least 30% of EU protected species and Annex I habitats not currently in favourable status are in that category or show a strong positive trend by 2030. In addition to the primary focus on nature conservation restoration priorities, the proposed nature restoration targets are expected to contribute to climate objectives by restoring ecosystems to capture and store carbon and to prevent and reduce the impacts of natural disasters and climate hazards.

There has also been significant progress in the climate agenda. The EU has committed to climate neutrality in 2050, and the Commission has proposed a target of 55% reduction in greenhouse gas (GHG) emissions by 2030 compared to 1990 levels in its 2030 Climate Target Plan[3]. The proposed target is a "net" target, meaning increases in the carbon sink are included in the target. Therefore, the agriculture and forest sectors have an important role to play, contributing through carbon sequestration in soil and vegetation to reach the targets and compensate for unavoidable GHG emissions from other sectors. Land-based action to restore and protect key habitats can contribute to meeting the global ambition set by the Paris Agreement of keeping global warming within a 1.5 °C limit. However, given the urgency to reduce GHG emissions and the possible time lag between restoration measures and the actual achievement of carbon removals, it is important that nature restoration action does not come 'in lieu of' the needed GHG emission reductions. In addition, failing to reduce GHG

[2] COM/2020/380 final

[3] COM/2020/562 final

emissions will lead to changes in the climate which will threaten the integrity of ecosystems and their ability to continue sequestering and storing carbon in the long-term. Therefore, restoration can play an important role in climate action only when it is achieved alongside rapid emissions reductions across all sectors (Portner et al., 2021).

These political developments provide significant room for Member States to be ambitious in their restoration plans and highlight where cost-effective restoration measures can achieve the maximum climate mitigation and adaptation potential while contributing to biodiversity conservation objectives.

4. KEY FACTORS TO CONSIDER WHEN PRIORITISING ANNEX I HABITATS FOR RESTORATION FOR CARBON BENEFITS

4.1 Focus on protection from pressures vs. active restoration

Prior to identifying habitats to restore and planning specific restoration measures, it is critical to determine the main pressures and threats facing the habitat and identify the most appropriate conservation tools to effectively address these, including protection of existing habitat from pressures. Protection of habitats from pressures is sometimes referred to as passive restoration. For example, continued and strict protection is necessary to secure the large carbon stocks held in old, undisturbed habitats, whereas restoration involving an alteration in management or plant communities can release carbon into the atmosphere. Nabuurs et al. (2017) showed that if 7% of EU forests were set-aside for strict protection by 2050, an additional carbon dioxide sequestration of ~64 Mt CO_2/year could be achieved on ~120,000 km^2. In the marine and coastal domains, marine protected areas (MPAs) that restrict certain human activities have been found to be a highly effective tool to secure marine carbon stocks, as well as protecting biodiversity and boosting fisheries yield (Sala et al. 2021). Well-managed MPAs can be highly effective in securing the organic carbon stores in inshore habitats like maerl beds and cold-water coral reefs, where the main threats are physical disturbance, moorings, coastal developments, and renewable energy (e.g., in Scotland blue carbon habitats in the inshore MPA network are estimated to store 248 000 t organic carbon per year) (Burrows et al. 2017).

Protection of existing intact habitats may be more efficient in terms of carbon storage potential because restored habitats may not fully re-establish in terms of their carbon sequestration and storage potential, and it may take over a decade to improve habitat condition and re-establish carbon cycling. For example, in rewetted wetlands, carbon storage 10-20 years after restoration has been found to still be lower than that of pristine wetlands and the speed of recovery has been found to vary greatly across wetland types and pedo-climatic conditions (Yu et al., 2017). Restored seagrass meadows accumulated carbon at a rate comparable to measured ranges in natural seagrass meadows 12 years post restoration (Greiner et al. 2013). Some restored saltmarshes take over 100 years to reach the carbon accumulation rates of natural counterparts (Burden et al. 2019; Purre et al. 2019; Yu et al. 2017).

Nevertheless, as a large proportion of the Annex I habitat areas in the EU are degraded, active restoration will play an important role in increasing the capacity of ecosystems to store and sequester carbon. For example, peatlands have been highlighted as valuable ecosystems, hosting unique biodiversity, buffering floods, retaining water in the landscape, filtering out nutrients and improving water quality, and storing large amounts of carbon in their peat layers. However, most natural peatlands in the EU have been lost, having been drained for centuries for agriculture, forestry, and peat extraction (Tanneberger et al, 2017). In the EU, around 50% of peatlands are degraded and, in some regions, degradations have been so drastic that most former peatlands have been lost. For example, only 5% of near-natural peatlands remain in Germany (BfN et al. 2020). While wetlands can be restored and carbon sequestration increased, restoration cannot compensate for the carbon accumulation in the original ecosystem before drainage within a period relevant for climate change mitigation (Hendriks et al. 2020). However, with the remaining peatlands highly degraded and a major source of GHG emissions (e.g. 38.98 t CO_2 e/ha/yr in the 2021 UK GHG Inventory), there is an urgent need to restore peatlands to maximize their ability to contribute to biodiversity and climate goals (Anderson, 2021).

4.2 Examine potential trade-offs between restoration for biodiversity and for carbon priorities

For Annex I habitats, the restoration of the biodiversity value of the habitat should be the primary objective, in line with the aim of the EU Habitats Directive to reach and maintain favourable conservation status of these habitats throughout their range in the EU.

In some cases, habitat restoration may involve a decision between managing land for carbon or enhancing biodiversity. For example, restoration of bog habitats often requires tree removal to restore hydrology and biodiversity (Gregg et al. 2021). In such cases, a strategic, integrated approach at the landscape level, combining Annex I habitat restoration with protection and actions elsewhere where appropriate to protect biodiversity, whilst protecting and increasing natural carbon sinks, is critical. A landscape level approach is also key to achieving important co-benefits, like water regulation, disaster risk reduction and flood risk mitigation.

Grassland restoration may also entail the removal of carbon stored in biomass. Many areas of Annex I grassland habitat and other open habitats such as sand dunes are affected by lack of management and are therefore becoming increasingly overgrown by woody vegetation, and eutrophication exacerbates this succession in some places[4]. Although this is contributing to the spontaneous growth of forest, particularly in the Mediterranean region, it is resulting in the loss of Annex I habitat and declining conservation status of these habitats. Restoration of these grasslands will reduce the above ground carbon sequestration, but it can also contribute to reducing forest fire risk, and therefore avoid large carbon losses from large scale fires. It is also important to note that for many Annex I grasslands, a certain proportion of scrub is a key part of the habitat structure, and trees and scrub form part of the habitat mosaic in which the habitats are found. At the landscape level, these can therefore still store a significant amount of carbon in the soil and above ground woody features once restored.

Restoration of highly degraded habitats may require active tree planting, but often natural regeneration is the more beneficial approach for biodiversity. It is important to grow the most adapted trees in the right places where both climate and soil are suitable for the selected species. Native tree species usually sequester carbon at rates comparable to, or higher than, non-native conifer plantations and support higher biodiversity. Planting trees on organic soils, particularly deep peat, can result in significant carbon emissions and can replace carbon in a stable, long-term store with carbon that can be quickly released into the atmosphere (Gregg et al. 2021).

In contrast, trees outside Annex I forest habitats, such as in hedgerows, within wood pastures and traditional orchards can play an important role in biodiversity conservation and contribute to carbon sequestration in managed landscapes. For example, the carbon sequestration potentials from agroforestry practices on European farmland range between 0.09 and 7.29 t C ha^{-1}, depending on the type of agroforestry (i.e., hedgerows, alley cropping, orchards with fruit trees and pollinator habitats) (Kay et al. 2019).

4.3 Examine feasibility of restoration – Habitat characteristics and carbon state

It is important to examine the feasibility of restoration actions and predict restoration success. Critical factors to consider include the present state of the ecosystem (habitat condition, level of degradation, current management), the quality of the surrounding landscape, regional/local climate change scenarios, and the potential of the ecosystem to recover without active restoration measures (Halme et al. 2013).

[4] Around half of the Annex I grassland habitat assessments in unfavourable condition were reported to be under pressures related to abandonment of grassland management (EEA, 2020)

It is important to be clear in the initial planning about the desired outcome and the baseline condition from which this outcome will be measured. For example, in forest restoration, it can be expensive for restoration practitioners to manage impacts on forests from fire and drought. Understanding tree migration biology and population dynamics and their relationship to climate change helps to develop restoration strategies that include the impacts of climate change, while incorporating knowledge from the past on the impacts of human activities, climate dynamics and forest response (Löf et al. 2019).

This information is also important to understand where an ecosystem is on the trajectory to any steady carbon state (habitat carbon stock equilibrium), which depends on management history and other disturbances like wildfires. Some ecosystems can take centuries to approach the assumed equilibrium, for example in temperate forests, but might be shorter in grasslands for example. This time scale is important alongside the planning and implementation of emissions reductions efforts (Anderson, 2021).

The history of land use, land ownership and use rights, economic feasibility, and funding availability also influence whether restoration will be achievable. Carrying out a detailed investigation into the habitat and its surrounding landscape, along with a thorough stakeholder mapping, can avoid future problems and support a successful outcome (i.e., restoration to good habitat condition, with long-term stakeholder engagement). Socio-economic and political barriers vary depending on what land users potentially face income losses or other costs, and whether the habitat is in a densely populated area or highly frequented by visitors. For example, plans to restore wetland hydrology can require considerable adjustments in farming systems and create resistance from local populations who fear flooded cellars.

5. KEY FACTORS INFLUENCING RESTORATION OUTCOMES

Restoration can have varying levels of impact on the carbon sequestration and storage potential of ecosystems, depending on the specific restoration measures or influences from the wider landscape. Furthermore, the time frame for ecosystems to sequester and store carbon at rates comparable to natural ecosystems, differs greatly.

5.1 Site conditions

Site conditions are an important factor determining the success of restoration measures to increase carbon stock and sequestration rates. For example, in Austria high elevation may support the accumulation of soil organic carbon as cooler conditions lower the productivity of forests and soil microbial activity (Jandl et al. 2021). For the restoration of grasslands on agriculturally degraded sites, annual carbon capture and storage rates can be accelerated by focusing measures on increasing plant diversity, especially of C4 grasses and legumes, which leads to higher aboveground production and root biomass (Yang et al. 2019; De Deyn et al. 2011).

The extent to which restoration of heath and scrub habitats can deliver carbon sequestration benefits can also be site-dependent and be affected by local temperature and rainfall conditions and on the original community being restored. For example, although increased scrub cover can sequester more carbon in these ecosystems, it can also drain carbon-rich soils due to high decomposition rates on deciduous vegetation, compared with evergreen heathland (Gregg et al. 2021). Additionally, planting trees in degraded upland heath can decrease carbon sequestration due to changes in organic matter depth and decomposition rates. There are also trade-offs between different restoration measures and carbon sequestration benefits from heath and scrub habitats. For example, studies have shown that grazing and burning can lead to increases in GHG emissions (van den Pol-van Dasselaar, 2018). Furthermore, although planting trees or allowing them to invade by secondary succession can store more carbon, this can lead to biodiversity losses, if tree cover is not carefully controlled. Nevertheless, restoration has proven to be successful for certain habitats. For UK upland heaths, the carbon benefit of restoring *Calluna*-dominated upland heath was 0.60 ± 0.70 t C ha^{-1} yr^{-1} since restoration (Quin et al. 2014). Restoration also increased the size of the recalcitrant carbon pool, which will increase carbon residence time and hence soil carbon accumulation in the longer term (Quin et al. 2014).

5.2 Time needed to achieve restoration

Time is another important factor determining the extent of carbon accumulation resulting from restoration. In saltmarshes, habitat creation and restoration of tidal flows will not necessarily result in ecosystems that behave or respond in the same way as natural systems, or in the reinstatement of pre-disturbance carbon stocks. However, rewetting measures and reconnecting former saltmarsh to the tide, through managed realignment of natural storm breaches, can eventually allow the return to conditions typical of a natural saltmarsh. Nevertheless, it can take many decades for plant communities in restored marshes to resemble those of natural marshes, which can affect carbon stocks and sequestration rates. Studies estimate that it can take approximately 65-100 years for restored sites to reach equivalency in carbon accumulation compared to natural saltmarshes (Burden et al. 2019). Nevertheless, saltmarsh restoration can lead to an initially rapid and subsequently sustained accumulation of carbon, in large part due to CO_2 uptake and carbon accumulation by saltmarsh plants growing on saturated soils, demonstrating that saltmarsh restoration can contribute to climate change mitigation (Burden et al. 2019).

The annex to this note provides a more detailed overview of the key restoration measures for ecosystems and indicates where possible the ones that can increase carbon sequestration and storage.

5.3 Costs and benefits

It is important to monitor the costs and benefits associated with ecosystem restoration throughout measure implementation and thereafter. Especially ensuring that the full range of benefits are captured and reported on is key to demonstrate the value of restoration and the far-reaching positive implications for society. For example, rewetting drained organic soils under agricultural use, which are currently losing carbon (both grassland and cropland i.e. 52 000 km^2) can lead to decreases in emissions of around 20 t CO_2eq ha^{-1} yr^{-1}, which would lead to 104 Mt of avoided CO_2 emissions per year (Glenk & Martin-Ortega, 2018). Using an estimated social cost of carbon of €100/t CO_2eq, this would result in potential benefits of up to €2000 per ha per year.

Alongside carbon sequestration and storage, restoration can deliver a wide range of ecosystem services, like provisioning services (food, feed, fibre), regulating services (soil stability and erosion control, wildfire prevention, water availability and quality), cultural services (use and non-use benefits) and important biodiversity benefits. For example, the restoration of seagrass beds can contribute to flood risk reduction and provide important fishery and nursery grounds of critical importance for local fisheries. The regulating and maintenance services of seagrass (*Cymodocea nodosa*) meadows in Gran Canaria have been valued at €95 per ha per year and the provisioning services at €866 per ha per year (Tuva et al., 2014). In the Mediterranean, the overall approximate benefits value of ecosystem services from seagrass (*Posidonia oceanica*) meadows is €284-514 per ha per year (Campagne et al. 2015).

Costs of restoration mainly involve those associated with the measures to halt further degradation, by effectively protecting and managing the habitat. These can vary greatly depending on location and the type of measure (e.g., mechanical vs. manual measures). The types of costs mostly relate to the costs of resources expended (like labour, materials, energy), recurrent management costs, administrative costs, and compensation payments (e.g., for income forgone, land purchases etc.).

5.4 Permanence of carbon gains and long-term monitoring

Ensuring permanence and long-term monitoring is an essential pre-requisite to secure the carbon benefits gained throughout restoration. For example, in wetland tidal restoration, if a tidal restriction were re-established 30 years post-restoration, emissions would resume. Similarly, other processes like fire or organic matter decomposition could release stored carbon back into the atmosphere (Kroeger et al. 2017). Therefore, it is critical that protection of the site is secured, degradation of the surrounding area is avoided and a concrete plan for the potential of impacts from climate change is developed.

6. LIMITATIONS OF DATA AND KNOWLEDGE FOR RESTORATION PRIORITISATION

There are various limitations and uncertainties that affect the reliability of carbon sequestration and storage information at the ecosystem and habitat level. These uncertainties must be considered when using the results of the EU level ranking exercise tested in this study.

6.1 Availability and quality of information

For terrestrial ecosystems, most studies focus on forests and wetlands, while information on carbon pools and carbon sequestration rates for other ecosystems like coastal, tundra and shrubs are relatively scarce. In the marine domain, carbon pools and sequestration rates have only been examined in detail for a small number of habitat types e.g., seagrass beds, and there is little information available on the large number of benthic habitats associated with different subtidal sediments (Hendriks et al. 2020). This means that while some estimates of carbon sequestration and storage at the Annex I habitat level are based on a number of studies, others rely on single studies or expert judgement. These differences in literature coverage affect the certainty of carbon estimates and are further outlined in Hendriks et al 2020. Furthermore, differences in the methodologies used to measure carbon results in a large range of values for carbon pools and carbon sequestration rates, which often cannot be fairly compared. Most studies do not investigate all components storing carbon in an ecosystem over time (i.e., living biomass carbon pools (stem, branches, leaves, roots), soil organic matter pools), meaning the nuances of carbon fluxes in the ecosystem cannot be fully captured, and the fluctuations that can occur, due to seasonal variability (climate), soil type and management are not considered. For example, studies that report the present carbon pool of peat soils at only one moment, are not considering the decomposition of soil organic matter, which can make the difference between the habitat acting as a carbon source or sink (Hendriks et al. 2020).

There is also a general lack of information on carbon stocks and flows in managed systems and the effects of management measures on carbon fluxes are not well documented (this can also be seen by the uncertainty of emissions estimates for the LULUCF Regulation reporting for agriculture, which is currently 45% and the general uncertainty of inventories for cropland is 48%; (Böttcher et al. 2019)). For example, in forests the cutting cycles, tree species and management intensity can affect carbon sequestration rates (Read et al. 2009).

Another complexity is that studies refer to different habitat conditions (i.e., pristine, degraded, managed) making it difficult to compare the carbon information across different sites as the habitat conditions are very specific. This also makes the calculation of additional carbon sequestration after restoration difficult as it can mask important differences in sequestration rates and carbon stocks associated with these different conditions. This is clearly seen in the case of wetlands where degraded wetlands can be important net emitters of carbon, while pristine wetlands have carbon flows that are close to zero, or act as small sinks (Evans et al, 2017). By taking an average sequestration rate in wetlands, the large potential avoided emissions of rewetting highly degraded wetlands are not captured. As an example, using the average data presented by Hendriks et al (2020), we calculated restoring all blanket bog area in the EU under a maximum restoration scenario would achieve as sequestration rate of around 0.1 Mt C /year. Emission factors for blanket bog in different conditions developed as part of the UK peatland code give intact blanket bog a sequestration rate of -0.1 t C/ ha/year and degraded blanket bog which has been drained or eroded, are given an emission rate of 1.27 t C /ha/year and 5.73 t C / ha/year respectively (Evans et al, 2017). Restoring blanket bogs through rewetting, does not only re-establish the carbon sequestration capacities of the habitat to give a -0.1 t C /ha/year rate, but also avoids the emissions from the degraded habitats giving an overall benefit of -1.37 t C / ha/year or – 8.56 t C /ha/year. If this value is extrapolated over the maximum blanket bog area to be restored in the EU, this would give an overall additional sequestration rate of around 0.2 Mt C/year to up to 1.14 Mt C/year, depending on the level of emissions of the degraded habitat. This illustrates that by not accounting for differences in carbon flows of degraded and restored habitats, some important changes in carbon storage might not be captured. The opposite scenario is likely in forest habitats where, even when degraded, some habitats are sequestering carbon. The average figure used in our exercise does not account for this baseline and, therefore, might overestimate the importance of restoring forest habitats solely for carbon sequestration.

Additionally, most information that can be found stems from site-level research or large metanalyses, meaning there is a lack of intermediate complexity i.e., studies at the regional, national or landscape level. Nevertheless, landscape-level restoration is most urgently needed, to improve the connectivity of green infrastructure and result in significant increases in natural carbon sinks. This is not to say that landscape level restoration is not occurring - an inspiring example is the Room for River project in the Netherlands (Nehren et al. 2014). However, there is currently a lack of long-term carbon information stemming from these large-scale projects, making it difficult to capture their value in terms of climate adaptation and mitigation and to use this as a template for further landscape-level restoration efforts.

6.2 Area and geographical location (information available at EU level)

When attempting to prioritise habitats for restoration based on their carbon potential, another important consideration is the effect of area and geographical location. Soil organic carbon stocks, for example, differ substantially depending on latitude and climatic regions, with the majority of carbon stock to be found at northern latitudes, particularly in the northern permafrost regions (Scharlemann et al. 2014). Therefore, although a habitat may cover a large area, carbon stock may be concentrated in particular locations, which affects the decision of where to focus restoration efforts.

The area information is based on Member State reporting data, which in some cases has been found to be vastly overestimated. Using this area information as the basis for carbon calculations and to prioritise restoration may lead to a general overestimation of the carbon potential of the restoration of habitats. For example, the area reported by Romania was removed from this analysis due to known inaccuracies in reporting as the total habitat areas reported exceeds the total area of the country. Areas reported by France under some habitats were also excluded as the sum of reported habitat area under good and not good condition exceeded the total area for the habitat. In some cases, reporting inaccuracies might also reflect underestimates of habitat area. For example, Portugal reported a total area of 0 for some habitats despite providing an estimate for the area of that habitat covered by the Natura 2000 network. There is also a certain risk of underestimates of habitat extent, but these are of a lower order of magnitude as they are most likely to occur for linear habitats such as rivers and in Member States where habitat areas are small and highly fragmented.

7. LIMITATIONS TO METHOD OF HABITAT RANKING TO MAXIMISE CARBON CO-BENEFITS

The prioritisation exercise carried out as part of this study is a useful starting point to prioritise habitats for restoration and protection to maximise carbon sequestration and storage co-benefits. However, the precise numbers should be treated with some caution since there are uncertainties surrounding the underlying carbon and area data due to limited availability and quality of information, particularly at the Annex I habitat level. Furthermore, it is worth highlighting some limitations in the methodology used to prioritise habitats based on this information.

For restoration prioritisation, average sequestration rates are used to calculate the annual carbon sequestration benefits if all habitat area which is not in good condition were to be restored. Several assumptions are used here including that carbon emissions and sequestrations from degraded habitats are in equilibrium, resulting in zero carbon sequestration and emission rates, so that the average sequestration rate of the restored habitat reflects additional carbon sequestration. Furthermore, we assume the average sequestration rates taken from the literature reflect that of non-degraded habitats. Both assumptions are not realistic but are considered reasonable for this exercise as they allow us to highlight what habitats will likely deliver the highest carbon benefits. For the prioritisation of habitats to protect, existing carbon stocks were used as a proxy for carbon benefits as, in most cases, degradation of a habitat can lead to losses in these stocks. This is not to say that the whole stock would be lost if the habitat were not protected, but rather that protecting habitats with the largest carbon stores can avoid the largest carbon losses. Another key limitation of analysing carbon data at the EU scale is that average carbon stock and sequestration rates do not capture important spatial and temporal differences.

Due to these limitations, this exercise should not be used as the sole tool to justify decision-making, and its limitations should be understood and clearly communicated. Instead, this work is valuable as an exploratory study outlining key considerations for prioritizing restoration to maximize carbon storage and synthesizing the current knowledge of carbon sequestration and storage at the Annex I habitat level.

For further discussion on the uncertainties related to the methodology used for the prioritisation exercise, please refer to Annex 2.

8. FUTURE OUTLOOK

With a rapidly evolving policy landscape, especially in the climate agenda, the importance of natural ecosystems for their carbon mitigation potential is being recognised across sectors, and being integrated into agricultural, forestry, climate, and biodiversity policies.

8.1 Revision of LULUCF regulation and improvements in data

The proposed amendments to the LULUCF Regulation[5] can have a significant impact on ecosystem restoration decisions. The Regulation sets a binding commitment for each Member State to ensure accounted emissions from land use are entirely compensated by an equivalent accounted removal of CO_2 from the atmosphere through action in the sector i.e., no-debit rule for the period 2021-2030. The revision has introduced an EU-level target for carbon removals in the LULUCF sector of 310 million tonnes of CO_2 eq by 2030 for the period 2026-2030. The scope of the regulation has been extended from only forests to include all land uses, including wetlands from 2026 onwards. This framework sets an incentive for Member States to increase natural carbon sinks, especially those with a large capacity to capture carbon in a natural (protected) or restored state.

Furthermore, the amendments to the LULUCF Regulation have increased the demand for tracking land use with high resolution and producing consistent maps of land use changes but also for more accurate emission factors at higher tier levels, to ultimately reduce overall uncertainties of GHG estimates (Böttcher et al. 2019). Over the next few years, this will likely improve the quality of emissions data and result in more consistent methodologies, leading to improved GHG accounts and greater accuracy of emission and removal estimates. This also has important consequences for measuring the carbon storage and sequestration rates resulting from restoration measures.

As soils are an important component of all terrestrial ecosystems' carbon cycles, the revised EU Soil Strategy contains actions to significantly improve our knowledge base on their carbon stocks and flows. The strategy aims to improve the monitoring of soil quality, including soil organic carbon. Some of these improvements will come from enhancements to the LUCAS soil survey, an EU-wide harmonized soil monitoring framework. Its sampling density for 2022 will be significantly increased to enhance its representativity, and various MS are testing ways to integrate it with national soil monitoring (Jones et al, 2022). This will

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include a link to the existing forest soil monitoring under the ICP Forests (International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests) and to the CAP Land Parcel Information System.

8.2 Uncertainties related to climate change

Although policy developments increasingly recognise the role of ecosystem restoration in achieving climate mitigation goals, climate change itself may affect restoration decisions and influence the success of restoration outcomes. Prior to developing restoration plans the potential uncertainties of future climate change scenarios should be carefully considered, which may pose a risk of non-permanence of restoration measures (and ultimately a release of CO_2). There is a need to foster overlap between ecosystem restoration, climate change adaptation and disaster risk reduction agendas to develop strategies at the national level and ensure local-level engagement of key actors (EEA, 2017).

Restoration and management measures are important to improve the resilience of habitats to climate change impacts. This can involve comprehensive planning, for example for adaptive forest management that considers the site-suitability of existing tree species at present and in the future, the possible long-term changes that can influence the forest (e.g., weather patterns, disturbance), diversification of tree species composition, stand structures and management approaches, proactive disturbance risk mitigation and continuous monitoring of the abiotic and biotic impacts of climate change and tree response (Forest Europe, 2020). For old-growth forests, restoration can re-establish forest structure, function and composition through both passive rewilding approaches and active restoration approaches aiming to restore characteristics which can accelerate stand development processes, and the establishment of late-successional biodiversity and ecosystem services such as carbon storage and flood resilience (Sabatini et al. 2020). The most appropriate approach will depend on the condition and context of the forest. In both cases, restoration can increase the resilience of forest ecosystems to future climate change impacts by increasing functional and genetic diversity and complexity.

For some Annex I habitats, achieving favourable conservation status will require habitat recreation to increase the overall area and range of the habitat to achieve the Favourable Reference Values identified by the Member States. It may also be necessary to increase the area of patches of habitat that are too small and/or isolated from other areas of the same habitat to be in good condition and support viable populations of key species. Whilst a high priority should be given to restoration of habitat within Natura 2000 sites, many areas of steppe and heathland habitat are now fragmented, and therefore re-creation that can increase the size, buffer or ecologically connect small and isolated sites is also particularly important in many regions, highlighting the importance of restoration and re-creation outside of the existing Natura 2000 network. At the larger scale, the creation of new habitats could help species respond to climate change by facilitating range expansion in fragmented landscapes (Hodgson et al. 2011). However, habitat recreation can be costly and difficult, for example if large areas of Annex I grassland habitat need to be re-created, then this will normally have to be carried out on fertile former arable land or species-poor grassland. In both cases the biotic and abiotic constraints can be severely challenging for the recreation of species rich grassland communities (see the annex for re-creation and restoration measures). Furthermore, it is critical to consider the trade-offs between the needs of different species and habitats and between short and long-term conservation priorities.

9. POTENTIAL OF ANNEX I HABITAT RESTORATION TO DELIVER CLIMATE MITIGATION

The restoration of habitats listed under Annex I of the EU Habitats Directive has great potential to deliver climate mitigation benefits through increasing carbon sequestration and storage and avoiding emissions from degraded land. Although it is not currently feasible to give precise numerical estimates of the carbon benefits of Annex I habitat restoration using existing data, there is good evidence for the strong carbon sequestration potential of restoring key habitats.

Forests and wetlands are particularly important ecosystems to restore in terms of carbon benefits. Habitats that have large spatial extent and a large potential area for restoration, alongside high carbon sequestration rates, have the highest potential for carbon gains. In the EU, these include beech forests, western Taiga, bog woodlands and Aapa mires. Coastal wetlands such as estuaries, salt marshes and coastal lagoons are also important carbon sinks as they can accumulate carbon over much longer periods of time than most terrestrial habitats (McLeod et al 2011). The same is true for other marine ecosystems which were excluded from this analysis as not enough information is available at a habitat level. For example, seagrass meadows are estimated to store carbon 30 times faster than forests (McLeod et al 2011). This incredible carbon sequestration capacity is weakened in degraded habitats meaning that seagrass restoration, despite some challenges, can deliver high climate benefits (Oreska et al 2020, Macreadie et al 2021). Although restoring other ecosystems such as grasslands and heathland delivers comparatively smaller carbon sequestration benefits, their restoration must not be overlooked. These ecosystems cover large areas of the EU, and their restoration can deliver many other important co-benefits.

Currently available data does not allow for the precise estimation of the additional carbon storage potential and sequestration of Annex I habitat restoration. This would require a better understanding of the carbon sequestration and stock under Annex I habitats considering their current level of degradation, as well as more studies quantifying the precise carbon benefits restoration can truly achieve. Keeping this in mind and while considering the limitations of the study outlined above, a rough approximation can be estimated using the figures from the literature review by Hendricks et al 2020. If all Annex I habitat in bad and unknown condition were to be restored, the restored area would sequester around 84 Mt C/ year (over 47.2 Mha). This could translate into very different additional sequestration rates depending on current carbon flows in different habitats. Where degraded ecosystems are currently emitting carbon, this would be an underestimate, while where they are sequestering, this would be an overestimate. Moreover, restoration cannot always re-establish the full carbon sequestration abilities of intact ecosystems. Despite this, the figure might be in the right order of magnitude. The UN estimated that restoration across ecosystem types could remove around 13-26 Gt of GHG over 350 Mha (UNEP 2019). This would be the equivalent of 175 Mt GHG, or 47.7 Mt C over 47.2 Mha[6] .

Currently available data on carbon stocks also clearly supports the protection of Annex I habitats. Annex I habitats for which enough information is available (excluding sparsely vegetated and marine habitats) hold around an estimated 5564 MtC – 17807 MtC over 87 Mha. These habitats are therefore hugely important carbon stocks which must be protected to ensure they are not depleted.

There is a huge potential for carbon benefits from restoration of land that is not Annex I habitat. The EU biodiversity strategy includes a target to bring back at least 10% of agricultural area under high-diversity landscape features, and these areas can also help enhance carbon sequestration and support climate adaptation. Member States will be responsible for using their CAP strategic plans to implement this 10% target at the national, regional, or individual farm scale and ensuring connectivity between the areas. The EU Biodiversity Strategy and the EU Forest Strategy for 2030[7] propose planting an additional 3 billion trees in the EU by 2030, in full respect of ecological principles, with a focus on urban trees and agroforestry. Sustainable agroforestry practices and restoration and creation of landscape features offer great potential to provide multiple benefits for biodiversity, people, and climate.

Conclusions

- Restoration is extremely important, resulting in significant benefits from ecosystem service delivery and enabling ecosystems to adapt to climate change.

[6] Calculated by extrapolating the figure of 13 Gt over 350 Mha to the Annex I habitat area of 47.2 Mha. GHG emissions in CO2 equivalents were converted to C by dividing by 3.667 (based on the molecular masses of both carbon dioxide and carbon). This is a very broad approximation to give a ballpark estimate of the order of magnitude of carbon sequestration that restoration could achieve. The UN study included ecosystems globally and aquatic environments which likely have different sequestration potentials to those of Annex I habitats.

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- The restoration of habitats listed under Annex I of the EU Habitats Directive has great potential to deliver climate mitigation benefits through increasing carbon sequestration and storage and avoiding emissions from degraded land. If all Annex I habitat in bad and unknown condition for which enough information is available (excluding sparsely vegetated and marine habitats) were to be restored, the restored area would sequester around 84 Mt C/ year (over 47.2 Mha). This number is not a precise estimation but an order of magnitude.

- Conservation of existing intact habitats is key to ensure the protection and permanence of important existing carbon stocks. Moreover, protection may be more efficient in terms of carbon storage potential because restored habitats may not fully re-establish in terms of their carbon sequestration and storage potential, and it may take over a decade to improve habitat condition and re-establish carbon cycling. Annex I habitats for which enough information is available (excluding sparsely vegetated and marine habitats) hold around an estimated 5564 MtC – 17807 MtC over 87 Mha. These habitats are therefore hugely important carbon stocks which must be protected to ensure they are not depleted.

- For Annex I habitats, the restoration of the biodiversity value of the habitat should be the primary objective, in line with the aim of the EU Habitats Directive to reach and maintain favourable conservation status of these habitats throughout their range in the EU. There are trade-offs to consider between biodiversity conservation and maximizing natural carbon sinks. Restoration should never come at the cost of biodiversity! It is important to consider protection alongside ecosystem restoration. Which is more suitable depends on various factors and a deep understanding of the pressures causing habitat degradation.

- Restoration is complex – there is a wide range of potential restoration measures, with abiotic and biotic effects on whether restoration is successful in restoring the carbon storage and sequestration potential to levels comparable to undegraded habitats. It can take over 100 years (for example in the case of some wetland habitats) for habitats to be restored to good condition.

- It is important to ensure that long-term monitoring is in place on restored sites and that pressures have been eliminated, to secure permanence of the carbon gains resulting from restoration measures.

- It is important to utilize the available data to support Member States in the process of identifying the most important habitats and locations to focus restoration efforts to maximise the co-benefits between biodiversity conservation and carbon storage and sequestration. However, there are many limitations to estimating carbon restoration potential, especially concerning the quality and quantity of carbon information, which can affect the reliability of carbon storage and sequestration data.

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10. ANNEX I: ANALYSIS OF HABITAT GROUPS

10.1 Agro ecosystems, natural and semi-natural grass

Restoration and recreation measures

The primary restoration goal for Annex I agricultural grasslands, heathlands, and dunes etc, is usually to re-establish appropriate grazing and/or mowing activities, and other farming practices that maintain High Nature Value (HNV) farm habitat systems. Therefore, the large-scale restoration of these habitats, and semi-natural habitats in general, is fundamentally dependent on maintaining the overall viability of the High Nature Value farming systems that underpin them. In some cases, grazing may need to be reduced or adapted to avoid or enable recovery from the impacts of over-grazing. In instances where degradation has resulted from under-grazing, or abandonment, scrub clearance may be needed, along with the repair or recreation of field boundaries and other features to allow livestock grazing. Where degradation has occurred from over-grazing, an adjustment of stocking densities and an alteration of grazing regimes to avoid over grazing in sensitive areas may be needed.

To achieve good condition to contribute to good condition of Annex I agricultural habitats, the following additional supporting restoration and re-creation measures are often needed, depending on the level of degradation:

- Hydrological restoration on previously drained wet grasslands.
- Removal of invasive plants.
- Measures for fire prevention and control.

In addition to this, measures to reduce deposition of nitrogen below critical levels will be required to achieve favourable conservation status.

If large areas of Annex I habitat need to be re-created, this will normally have to be carried out on former arable land or species-poor grassland, which is generally has a high nutrient level. In both cases the biotic and abiotic constraints can be severely challenging for the recreation of species rich grassland communities. To overcome such problems, the restoration/recreation of semi-natural grasslands must include a range of measures (Blakesley and Buckley, 2016) that:

- reverse vegetation successional processes on abandoned or under-grazed pastures (e.g. clear unwanted scrub from grasslands, or trees from heathland and scrubland);

- reduce high residual soil fertility resulting from previous agricultural practices and eutrophication from other sources;
- address topographical and hydrological constraints;
- restore appropriate management to encourage sward diversification, often by reinstating traditional practices, by reducing competition for space and resources with less desirable species, and providing micro-sites for the establishment of target species;
- provide sources of propagules, where lacking, to rebuild the desired target community; and
- reconnect and reintegrate restoration sites with remaining areas of species rich grassland sites in the wider landscape.

Carbon stock and flows

Grassland restoration measures that lead to enhanced carbon sequestration can be:

- Measures to restore existing grassland - management actions that reinstate grazing and/or mowing of abandoned grasslands or that improve soil conditions and thus increase grass growth and/or soil organic matter.
- Measures to create grassland - land use change that generates new grassland (e.g. the conversion of fallow areas and arable cropland to grassland).

The measures with the biggest potential for carbon sequestration are (Paulsen et al. 2020):

- rewetting of grassland on organic soils
- land conversion from arable to grassland
- prevention of conversion of grassland to arable land
- low intensity grazing management and biodiversity enhancement
- agroforestry (hedges, trees, shrubs, in field or around field boundaries)

Studies have quantified the carbon benefits of creating new grassland:

- A global meta-analysis identifies conversion from cultivation as the measure yielding the largest carbon benefits (0.87 Mg C ha-1 yr^{-1}) (Conant et al. 2016).

- Conversion of arable land to permanent grassland was estimated to increase carbon storage by 0.49 Mg C ha^{-1} yr^{-1} over 20 years (Conant et al. 2016).

- In the UK, the conversion of arable land to low input grassland under Countryside Stewardship was estimated to sequester 1.590 t CO2e ha^{-1} yr^{-1}, compared to carbon losses on the arable land (Conant et al. 2016).

- A study in the Balearic Islands and Galicia (Spain) found the conversion of arable land to grassland led to CO2 sequestration of 3.3-9.8 t of CO2 ha^{-1} y^{-1}. However, CH4 and N2O emissions increased (263 kt CO2/yr) due to grazing livestock (Conant et al. 2016).

Additional measures to restore grasslands and agroecosystems to halt the decline of target species:

- Maintenance and creation of boundary features and buffer strips / grass margins: The potential carbon sequestration benefits of hedgerows and non-forest woodlands have been estimated at 0.66-3.3 tCO2/ha/yr (Conant et al. 2016).

Prioritisation exercise using Annex I habitat data

Tables A.1.1. and A.1.2. show the top 5 ranked Annex I agroecosystem and grassland habitats to maximise carbon storage and sequestration benefits in the EU.

Grassland habitats naturally vary in their ability to capture and store carbon. These differences are mainly due to Soil Organic Carbon (SOC) pools as little carbon is stored in biomass (and if harvested it does not account towards the biomass carbon pool). Therefore, differences in potential carbon sequestration and stock mostly arise from differences in soil types, and other variables affecting SOC stocks such as climate and current and historic management practices. As Annex I habitats are not classified according to their soil types, ecosystems often have several soil types leading to variations in SOC within them. Similarly, management practices also vary within ecosystems.

In the grasslands prioritisation exercise, the top habitats for both protection and restoration are those with the highest spatial extent. However, this result must be critically assessed and judged by experts. For example, pseudo-steppes with grasses and annuals of the Thero Brachypodieta come out on top for both stocks and sequestration. This is due to the high spatial extent of this ecosystem. However, carbon pools in steppe habitats without trees are typically low due to their lower net primary productivity and dry climate which slows down sequestration rates. In fact, the range of carbon stock estimate for this habitat is

0-430.18 Mt C meaning that some studies found much lower estimates, and even estimates of 0Mt, for the carbon sequestration potential of this habitat.

In contrast, Dehesa with evergreen *Quercus* species, which is the grassland habitat with the highest area, are not in the top 5 habitats due to their relatively low stocks and sequestration rates. These habitats, found in the southwest of the Iberian peninsula, are agrosylvopastoral systems where crops, pasture land, or shrub are shaded by native oaks. Although dehesas are also characterised by semi-arid conditions and are subject to droughts and grazing, they are typically carbon sinks. Their SOC pools have high spatial and temporal variability and are influenced by many site-specific factors including management practices, topography and tree presence (Reyna-Bowen et al., 2018). Due to these natural variations in SOC and due to the small number of studies measuring carbon fluxes in this habitat, the true carbon sequestration and storage potential of dehesas is unknown (Andreu et al, 2021). This is the case for most grassland habitats.

When thinking about protection, it is also useful to consider how much carbon stock could be protected outside of Natura 2000 as this might represent additional protection which is not currently a priority. The ranking of top grassland habitats is similar when looking at max carbon stock outside Natura 2000.

Table A.1.1.: Top 5 agri-grassland habitats for restoration based on carbon sequestration rates

Rank	Habitat	Annex I habitat code	Max potential sequestration rate of restored area (Mt yr-1)	Range
1	Pseudo-steppe with grasses and annuals of the Thero-Brachypodietea	6220	1,49	0.24-1.49
2	Lowland hay meadows (*Alopecurus pratensis, Sanguisorba officinalis*)	6510	1.36	0.81-1.36
3	European dry heaths	4030	0.74	0.13-0.74
4	Endemic oro-Mediterranean heaths with gorse	4090	0.60	0.05-0.60

Table A.1.2.: Top 5 agri-grassland habitats for protection based on carbon stocks

Rank	Habitat	Annex I habitat code	Max carbon stock (Mt)j	Range
1	Lowland hay meadows (*Alopecurus pratensis, Sanguisorba officinalis*)	6520	440.48	220.24-440.48
2	Pseudo-steppe with grasses and annuals of the Thero-Brachypodietea	6220	430.18	0-430.18
3	Mountain hay meadows	6520	268.77	89.95-268.77
4	European dry heaths	4030	227.33	113.67-227.33
5	Semi-natural dry grasslands and scrubland facies on calcareous substrates (Festuco-Brometalia) (*important orchid sites)	6210	164.57	0-164.57

10.2 Heath and scrub

Restoration and recreation measures

Most heathlands and scrublands below the treeline are not climax ecosystems, but habitats that under natural conditions would often be transitory phases of natural succession unless this is checked by fire, grazing or other disturbances. Therefore, in the absence of such natural conditions, these habitats generally require active management to arrest the process of succession to woodland, to maintain their structure and desired composition of dwarf shrub communities. Key maintenance techniques for heathland are low intensity grazing management, mowing and scrub and tree management. Similarly, the maintenance of good condition of Mediterranean scrubland (sclerophyllous vegetation) almost always involves extensive grazing, often with shepherding, which needs to be adjusted to the conservation targets for that habitat type and local conditions. In some areas, especially in temperate heathlands, management burning (i.e. controlled rotational burning of patches of dwarf shrub vegetation) may complement grazing. However, as inappropriate burning can also cause significant damage, burning management needs to be carefully regulated and carried out according to burning plans.

As Alpine and Boreal Heaths are primarily natural climax communities, their restoration in the first instance normally involves removing the cause of habitat degradation (e.g. over-grazing, eutrophication from nitrogen deposition, hydrological changes, trampling by walkers). Restoration of Boreal and Alpine heaths often involves ensuring that any grazing by sheep, cattle and reindeer is sustainable, i.e. does not degrade the condition of the habitat. It must be highlighted that, in some cases, restoration activities necessary to re-establish biodiversity can lead to some losses in carbon stock (e.g. through tree clearance and soil disturbance). Therefore, some trade-offs might occur between carbon sequestration and restoration objectives and these should be carefully assessed. Although some restoration can result in carbon emissions, the biodiversity value of these habitats and delivery of other ecosystem and cultural services will in many cases justify possible carbon losses (Alonso et al, 2012).

Carbon stock and flows

Steppe, heath and shrub habitats capture carbon from the atmosphere through photosynthesis which is then stored underground in SOC pools. Although these habitats also have an important biomass later, vegetation is often removed by herbivores or management and is therefore not a permanent carbon store. Their carbon stocks and flows are hugely affected by ecological disturbances and management as well as ecological characteristics. The fraction of shrubs in a habitat determine the size of carbon stock. Sites with the largest fraction of woody evergreen shrubs have been found to have the biggest belowground biomass, while sites with a larger fraction of grasses and herbs have a smaller belowground biomass. In general, the dominant carbon pools in shrubland ecosystems are belowground, amounting to 10–30 times the carbon stored in the aboveground pools (Beier et al. 2009).

Replacement of grasslands by shrublands can lead to an increase in carbon sequestration and is therefore considered to be an important contribution to the carbon sink in the global carbon budget. Dwarf-scrub dominated vegetation has higher carbon stocks and can sequester more CO_2 than grass-dominated vegetation, which generally has higher respiration rates. The carbon sequestration in the soil is of particular importance as heather-dominated communities have been found to sequester more than double the carbon than grass vegetation (-3.45 ± 0.96 t C ha-1 y-1 vs -1.61 ± 0.57 t C ha^{-1} y^{-1}). These values are comparable to the average values of broadleaf and conifer woodland sequestration in the UK and therefore, restoring ericaceous vegetation in lowland and upland heathlands on mineral soils cover can increase carbon sequestration comparable to woodland (Gregg et al. 2020).

In scrub habitats, increases in woody vegetation can increase carbon accumulation in biomass and/or the soil.

Prioritisation exercise with Annex I habitat data

Tables A.1.3. and A.1.4. show the top 5 ranked Annex-1 heath and scrub habitats to maximise carbon storage and sequestration benefits in the EU.

The carbon sequestration and storage potential of steppe, heath and scrub habitats is mostly dependent on SOC as it is the most important carbon component of these habitats. Therefore, the carbon benefits of their protection and restoration depends on the site-specific factors that influence SOC pools including climate, soil type, geography, succession stage, vegetation and management practices. These variables vary within Annex I habitats meaning large variations are expected to exist within them making it difficult to give estimates of carbon flows and stocks at the habitat level. Clearly, habitats on organic soils will have the largest carbon stocks. Following the habitat classification in Romao (2020), these are included under the 'wetlands' ecosystem type. As outlined above, vegetation cover can also determine the size of carbon stocks in these habitats. However, no clear differences were found in the carbon pools of Calluna and Erica vegetation (Hendriks et al 2020).

The top 5 habitats for both protection and restoration broadly correspond to the steppe, heath and scrub habitats with the largest areas in the EU. The available carbon stock and sequestration data was not sufficiently to clearly differentiate whether differences in climate mitigation potential exist between different Annex I habitats. Some differences might exist as, for example, lowland and some coastal heaths which are exposed to drier conditions and typically have thinner soils might have lower carbon storage capacities than alpine and boreal heaths (Bartlett et al, 2020). However, as heathland carbon stores largely depend on soil characteristics, management, and other site-specific characteristics which vary within Annex I habitats, it may not be possible to prioritise them at this level for restoration for climate mitigation. A more site-level approach might be more appropriate to support any final restoration prioritisation decisions.

For additional maximum carbon stock to be protected outside Natura 2000, the ranking of priority habitats is broadly the same as for restoration and protection. It must be noted that some of the areas reported by MS within natura 2000 are larger than the total reported area. Therefore, these area figures are not accurately reported and should be further assessed.

Table A.1.3.: Top 5 steppe, heath and scrub habitats for restoration based on carbon sequestration rates

Rank	Habitat	Annex I habitat code	Max potential sequestration rate of restored area (Mt yr-1)	Range
1	Alpine and Boreal heaths	4060	1.08	0.16-1.08
2	Thermo-Mediterranean and pre-desert scrub	5330	0.49	0.14-0.49
3	Arborescent matorral with Juniperus spp.	5210	0.27	0.03-0.27
4	Stable xerothermophilous formations with Buxus sempervirens on rock slopes (Berberidion p.p.)	5110	0.19	0.03-0.19
5	Mountain *Cytisus purgans* formations	5120	0.11	0.03-0.11

Table A.1.4.: Top 5 steppe, heath and scrub habitats for protection based on carbon stocks

Rank	Habitat	Annex I habitat code	Max carbon stock (Mt)	Range
1	Alpine and Boreal heaths	4060	562.21	281.11-562.21
2	Thermo-Mediterranean and pre-desert scrub	5330	186.27	0-186.27
3	*Sarcopoterium spinosum* phryganas	5420	116.18	0-116.18
4	Arborescent matorral with *Juniperus* spp.	5210	109.51	54.76-109.51
5	Stable xerothermophilous formations with *Buxus sempervirens* on rock slopes (Berberidion p.p.)	5110	42.87	21.43-42.87

10.3 Forests

Restoration and recreation measures

It is widely recognised that there is a huge opportunity to enhance forests carbon mitigation potential through restoring degraded forests. The precise restoration measures needed must consider its structural, compositional, and functional characteristics. Some restoration measures have been specifically shown to increase SOC stocks in forests. These include avoiding the conversion of primary and old growth forests and avoiding the conversion of forests to cropland, avoiding clear-cut harvesting, avoiding the removal of harvest residues and litter, afforestation of cropland, reducing soil disturbance, species selection and management, avoiding drainage, fire management, and management of stand density and thinning (Mayer et al, 2020). However, the precise impact of these restoration measures on the potential carbon storage and sequestration potential of forest habitats also depends on site-specific factors such as soil type, forest age and species composition, climate and current and historic management.

Carbon stock and flows

Forest habitats have large potential carbon stocks and sequestration rates. The CO2 fixed by forest trees through photosynthesis plays a key role in carbon cycles and can be important for climate mitigation as carbon is persistently stored in forest carbon pools over long time frames (Barredo et al, 2012). These carbon stocks grow over decades to centuries before reaching a saturation point. In Europe, forests sequester around a tenth of gross CO2 emissions (Forest Europe, 2020). The key carbon pools in forests are found in forest soils, living vegetation (including stems, leaves, roots, branches etc), deadwood, and their litter layer. In forests, carbon sequestration is much larger than their stock. In the EU, over half of forest carbon stocks are found in soils, and around 35% is found in living woody biomass, 7% in below-ground biomass, and 8% in litter (Forest Europe, 2020). However, the exact distribution between carbon cycle components depends on climate, soil type, geography, and forest ecology. In addition, different studies cover different carbon cycle components making it difficult to compare carbon figures across them. Data is particularly lacking for SOC where trends at the EU level cannot be determined (Forest Europe, 2020). The carbon sequestration and storage potential of forests varies across forest types. Old growth and mature forests typically have higher carbon stocks per hectare than other forests in similar conditions. In addition, evidence shows these forests continue accumulating carbon well past reaching maturity meaning they can act as active carbon sinks for centuries (Barredo et al, 2021).

Prioritisation exercise using Annex I habitat data

Tables A.1.5. and A.1.6. show the top 5 ranked Annex I Forest habitats to maximise carbon storage and sequestration benefits in the EU.

Forest habitats are widely diverse with 80 distinct habitat types included in Annex I of the habitats directive. These can be broadly classified into boreal forests, temperate forests, Mediterranean and Macaronesian forests, and mountainous and coniferous forests. Alluvial forests and wooded meadows are included under different ecosystem types. The state of European forests report shows that the highest carbon stock per unit area can be found in forests of central-west and central-east Europe, with lower levels in the southern and northern regions of Europe. Studies also show lower carbon stocks for some Mediterranean forests and some boreal forests (Hendriks et al, 2020). These differences are broadly reflected in our prioritisation exercise. The top 5 habitats are those with the highest spatial extent in the EU, excluding some habitats which are found in the less carbon rich Mediterranean and other southern bioregions (e.g. Pannonian Balkanic turkey oak sessile oak forests, and Mediterranean pine forests with endemic Mesogean pines). However, on top of geographical region, other forest characteristics such as soil type, tree species, stand age and current and past management practices, have a huge influence on their carbon stock and storage potential (Hendriks et al 2020). These are sometimes not regionally distinct and can vary within Annex I habitats. Therefore, it is currently challenging to accurately measure the climate mitigation potential of forest habitats at the Annex I level.

Table A.1.5.: Top 5 forests habitats for restoration based on carbon sequestration rates

Rank	Habitat	Annex I habitat code	Max potential sequestration rate of restored area (Mt yr-1)	Range
1	Asperulo-Fagetum beech forests	9130	12.76	9.97-12.76
2	Western Taïga	9010	6.76	1.07-6.76
3	Nordic subalpine/subarctic forests with *Betula pubescens* ssp. *Czerepanovii*	9040	5.44	0.78-5.44
4	Atlantic acidophilous beech forests with *Ilex* and sometimes also *Taxus* in the shrub layer (Quercion robori-petraeae or Ilici-Fagenion)	9120	4.90	4.46-4.90
5	*Quercus ilex* and *Quercus rotundifolia* forests	9340	4.09	0.99-4.09

Table A.1.6.: Top 5 forests habitats for protection based on carbon stocks

Rank	Habitat	Annex I habitat code	Max carbon stock (Mt)	Range
1	Alpine and Boreal heaths	9130	1681.29	420.32-1681.29
2	Western Taïga	9010	817.09	272.36-817.09
3	Luzulo-Fagetum beech forests	9110	630.27	157.57-630.27
4	*Quercus ilex* and *Quercus rotundifolia* forests	9340	625.15	416.76-625.15
5	Pannonian-Balkanic turkey oak-sessile oak forests	91M0	517.18	172.39-517.18

10.4 Wetlands

Intact wetlands, and in particular peatlands, are important habitats in terms of carbon sequestration and storage. Although the carbon sequestration rates per hectare of most peatland habitats is smaller than that of forest habitats, carbon stocks are proportionally higher as they continuously accumulate carbon in growing peat layers. Wetlands have the highest carbon stocks of any terrestrial habitat making them a key ecosystem for carbon storage. They contain 30% of total organic soil carbon despite covering only around 5-8% of the world's area (Xu et al, 2018)[8]. Carbon storage potential varies across wetland types. Peatlands with thick peat layers and salt marshes contain the highest carbon stocks while wet heathlands and shallow peatlands have smaller stocks (Natural England, 2010). Despite variations in stock and sequestration rates across habitat types, if all Annex I wetland habitats were in good condition, the potential carbon sequestration rate would be around 12 Mt C yr-1 and the carbon storage would be between 1.7 Gt and 4.3 Gt of carbon. Quantitative information on carbon storage sequestration and storage in inland marshes appears to be generally lacking, even though reedbeds are known carbon sinks.

8

Restoration and recreation measures

The huge carbon sequestration and storage benefits of wetland habitats are tightly linked to their ecological condition. Therefore, restoration of degraded wetlands, particularly peatland, should be prioritised as a clear win-win option for climate mitigation and ecosystem restoration. The precise measures needed to restore degraded wetlands vary across sites as they depend on the purpose of restoration and site-specific factors including level of degradation, land-use, type of wetland, ecological conditions, and climate. Despite these differences, peatland restoration typically requires raising water tables (i.e. rewetting) and re-establishing wetland vegetation (Dinesen & Hahn, 2019). Rewetting alone can at least halt the loss of carbon from drained peatland and, in some cases, re-establish the carbon sequestration abilities of healthy peatlands. However, the restoration of true peat formation can take decades. Peatlands can start sequestering carbon in the first years following rewetting and carbon accumulation likely slows after that. However, this potential carbon sink is small compared to the carbon benefits from avoided emissions after rewetting. Moreover, carbon sequestration most likely only compensates from the increased methane emissions which naturally occurs after rewetting. Therefore, the largest climate benefits of rewetting are due to avoided emissions rather than increased carbon storage (Mrotzek et al, 2020).

Restoration activities can help biodiversity recover in degraded wetland sites as the ecological condition of the habitat is improved (Renou Wilson et al 2019). Despite some wetland habitats having relatively low species richness, peatland species are of very high biodiversity importance as their unique conditions are home to specialised plant and animal communities many of which are rare or threatened. The species abundance and composition of wetlands varies across wetland habitat types.

The majority of the EU’s mires are located in Northern Europe, in the boreal, continental and Atlantic biogeographic regions. It is estimated that around 50% of European mires are degraded, and around less than 17% are within protected areas. However, this varies across Europe with an increase in both degradation and protected area coverage from north to south (Tanneberger et al 2021).

Calculating the climate mitigation potential of restoring wetlands is difficult due to data gaps, unequal literature coverage and site-specific variabilities. Current data tends to focus on boreal systems and on peatland habitats (Grand Clement et al, 2015). Furthermore, large uncertainties exist around measuring the carbon benefits of wetland restoration. It is often difficult to compare studies as due to differences in methodologies, carbon cycle components included, and in some cases, the GHGs considered (Gregg et al 2021). In addition, the carbon fluxes and stocks of wetland habitats are hugely depended on local conditions including climate, soil type, wetland type, degradation, and present and past management. Moreover, carbon stocks and flows vary temporarily as well as spatially. This creates uncertainties when extrapolating average sequestration and stock rates across habitats as they might not reflect local conditions. Finally, the degree to which wetlands can be restored depends on the habitat's starting condition and several key barriers often prevent full restoration including habitat quality and species recolonisation problems. Furthermore, trade-offs can happen between different ecosystem services, including between carbon sequestration and biodiversity restoration. Therefore, these factors have to be carefully considered before restoration to ensure the optimal restoration measures are chosen (Lamers et al, 2014).

Carbon stocks and flows

Carbon flows in wetlands happen in the form of gaseous fluxes (including CO2 from respiration under aerobic conditions, CH4 from bacterial activity in anaerobic conditions, and N2O), waterborne fluxes (primarily dissolved organic carbon and particulate organic carbon), and changes in vegetation cover. Healthy peatlands which are currently forming peat, known as mires, are an important ecosystem in terms of carbon sequestration as they continuously capture and store carbon in accumulating organic matter due to slowed down decomposition under waterlogged conditions. Mires can be divided into bogs and fens according to their water source. When degraded or drained, peatlands can lose their ability to sequester carbon and turn into important sources of GHG emissions. In the EU, drained peatlands emit around 220 Mt CO2eq per year, roughly equivalent to 5% of total emissions (Bonn et al 2014). The importance of the biomass component of wetlands in terms of carbon sequestration varies across biogeographical regions and should be considered in some cases (Wilson, 2013).

Prioritisation exercise using Annex I habitat data

Tables A.1.7. and A.1.8. show the top 5 ranked Annex I wetland habitats to maximise carbon storage and sequestration benefits in the EU.

Wetlands encompass many different ecosystems which differ in their carbon cycling and, as a result, in the size of their carbon stocks and flows. In peatlands, stocks depend on the size of the peat layer. Within the peatland group, bogs in general have a higher carbon sequestration rates and larger stocks than fens. Here, bog woodland comes out as a priority habitat both in terms of its sequestration rate and its carbon stock. This is due the fact it is the wetland habitat with the largest spatial extent in the EU, and that it has high carbon stock per unit area. Although its sequestration rate per unit area is smaller relative to those found for coastal ecosystems, its large EU area in need for restoration makes it a priority. Similarly, Aapa mires, the second largest wetland habitat area in the EU, also rank highly due to their spatial extent and their high carbon stocks. This fen habitat mostly occurs in southern, middle and northern boreal regions as well as in some pre-alpine areas.

Intertidal habitats such as mud flats and salt marshes are important for both carbon storage and sequestration as they trap carbon in their sediment. These habitats are a net sink of carbon when exposed. However, less is known about their carbon exchanges when submerged and some studies suggest that when taking these into account , these habitats are net sources of CO_2 to the water column. In fact, studies estimating sequestration rates from mud flats are likely to be overestimates due to methodological issues in estimating the carbon fluxes of coastal habitats (Legge et al., 2020). Therefore, although these are undoubtedly important habitats in terms of carbon, particularly due to their large spatial extent, the high prioritization of habitat 1140 must be critically assessed. Salt marshes have the highest sequestration rate and carbon stock per unit area than other coastal wetland habitats. However, at the EU level they do not emerge as the top priority habitat due to their smaller area.

Coastal lagoons and estuaries also rank highly due to their relatively large spatial extent in the EU. A notable result of the prioritization for protection is the Peat grassland of Triodos, despite its very small recorded area in the EU. Its high prioritization value is due to a very high maximum carbon stock estimate, which should be critically analysed by assessing the literature on which the figure is based. The minimum stock estimate for the habitat is 0 Mt C yr^{-1}, which suggests the high maximum estimate might not be representative of the actual EU carbon stock for this habitat.

For carbon stocks, when looking at the maximum stock which would be protected by additionally protecting habitat areas outside of Natura 2000, the top 5 habitats which emerge are bog woodland, Aapa mires, transition mires and quaking bogs, alkaline fens and coastal lagoons. This is very similar to the priority habitats for protection ranked in terms of their carbon stock.

Table A.1.7.: Top 5 wetland habitats for restoration based on carbon sequestration rates

Rank	Habitat	Annex I habitat code	Max potential sequestration rate of restored area (Mt yr-1)	Range
1	Bog woodland	91DO	3.07	0.60-3.07
2	Mudflats and sandflats not covered by seawater at low tide	1140	2.02	0.26-2.02
3	Coastal lagoons	1150	1.96	0.56-1.96
4	Estuaries	1130	1.47	0.60-1.47
5	Aapa mires	7310	1.11	0.12-1.11

Table A.1.8.: Top 5 wetland habitats for protection based on carbon stocks

Rank	Habitat	Annex I habitat code	Max carbon stock (Mt)	Range
1	Bog woodland	91DO	1671.94	668.78-1671.94
2	Aapa mires	7310	1350.75	540.3-1350.57
3	Transition mires and quaking bogs	7140	712.67	237.56-712.67
4	Peat grasslands of Troodos	6460	116.18	0-116.18
5	Alkaline fens	7230	105.55	35.18-105.55

10.5 Marine

An in-depth review of the carbon storage and sequestration potential of marine ecosystem habitats in the EU is outside the scope of this study. As outlined in Hendriks et al (2020), currently available data on blue carbon is not sufficient to estimate carbon data at the Annex I level. However, some key differences between broad marine habitats can be seen which could help guide restoration activities. The literature log provides additional sources to understand the climate mitigation potential of marine ecosystem restoration. The summary below is mostly based on the literature review in Hendriks et al 2020.

Restoration and recreation measures

At the international level, habitat restoration is performed through active restoration (e.g., replanting seagrass) and/or passive restoration (leaving habitats undisturbed through protected areas, so that habitats recover naturally). Regardless of the restoration method, protected areas are needed to ensure the restored habitats do not degrade again.

Some marine restoration measures can enhance the climate mitigation potential of these habitats. These can be divided into actions which maintain carbon store and actions which enhance removal. Carbon stores can be maintained through protection and additional measures that ensure the integrity of habitats such as seagrass and maerl beds. Enhancing of removal can be achieved by increasing the area of habitats through their restoration and habitat recreation, as well as through measures such as decreasing pollutants, restoring hydrology and, in some cases, re-introducing some species. In addition, preventing harmful human activities such as bottom trawling can maintain important carbon stores (Hendriks et al 2020).

Carbon stocks and flows

Marine ecosystems contain the largest long-term carbon store on Earth, storing and cycling around 93% of all carbon. Most of this carbon is in the form of dissolved inorganic carbon, with a smaller proportion stored in the form of particulate organic matter and dissolved organic carbon. Some carbon is also stored in marine living biomass. All of these pools are constantly cycling with a small fraction, around 1% of carbon produced at the sea surface, stored in a more stable form in the deep ocean. The carbon sequestration and storage potential of marine habitats varies widely across different habitat types and geographical regions. The highest carbon concentrations can be found in the North Atlantic. In general, sequestration rates are highest in seagrass beds, brittlestar beds and maerl beds, storage rates highest for deep coral reefs and maerl beds. For a more in-depth analysis of the available data on the carbon stocks and flows of different marine ecosystems please see Hendriks et al 2020. In addition, marine carbon stocks and flows are affected by a variety of factors including seasonality, temperature, stratification, ocean currents, turbulence, climate change, sediment type (for subtidal sediments), species (for seagrass), and anthropogenic activities. The literature around blue carbon storage and sequestration is rapidly growing. However, estimates of carbon stocks and flows in marine habitats still have high uncertainties and some marine habitats are understudied.

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11. ANNEX II: DETAILS OF METHOD USED

Habitat classification: We grouped the Annex I habitats into ecosystem types according to a classification approach used by the EEA. This roughly follows the MAES typology, however for some habitats the ecosystem type chosen differs between MAES and the EEA approach. The excel spreadsheet allows a differentiation between the two.

The agricultural habitats and grasslands group includes a total of 35 Annex I habitat types. All grasslands, except alluvial meadows and a selection of habitats dependent on agricultural management from different types are included. The 'river, lake, alluvial and riparian habitats' group includes 32 Annex I habitat types. All rivers and lakes (codes 31xx and 32xx) and a selection of alluvial and riparian habitats from other MAES categories. The 'forests' group includes 69 Annex I habitat types, all forests with habitat code 9xxx, except wet, alluvial, riparian forests and wooded meadows, which were included in other groups i.e., in 'wetlands' or 'river, lake, alluvial and riparian'. For forest habitats the Habitats Directive definition[9] was used along with the Interpretation Manual of EU Habitats (EC, 2013) for additional criteria[10]. The 'steppe, heath and scrub' group includes 21 Annex I habitat types, all heath and scrubs, except wet heaths and those dependent on agricultural management, which were included in the 'wetlands' groups and the 'agricultural habitats and grasslands' group respectively. The 'wetlands' group includes 28 Annex I habitat types, all peatlands (71xx, 72xx, 73xx), several wetlands and halophytic (salt) habitats, wet heaths and wet forests. The 'marine' group includes six Annex I habitat types, while three other habitats normally considered marine are included in the 'wetlands' group i.e., 1130 estuaries, 1140 mudflats and sandflats not covered by sea water at low tide and 1150 coastal lagoons.

This leaves 48 Annex I habitats not covered in our ecosystem typology and therefore not included in further analyses described below. The main reasons are the lack of reliable carbon stock and sequestration rate information and the fact that many of these habitats, for example in the sparsely vegetated category have

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[9] (Sub) natural woodland vegetation comprising native species forming forests of tall trees, with typical undergrowth, and meeting the following criteria: rare or residual, and/or hosting species of Community interest.

[10] Forests of native species; forests with a high degree of naturalness; forests of tall trees and high forest; presence of old and dead trees; forests with a substantial area; forests having benefited from continuous sustainable management over a significant period.

relatively small total area coverage, justifying the preliminary focus on areas with more carbon information and greater area coverage. Most of the omitted habitats are sparsely vegetated (23) and heathland scrub (9) habitats according to the MAES classification system.

Areas: For the basis of the area calculations, we used the 2019 Habitats Directive Article 17 reporting data from Member States (reporting period 2012-2018), following the methodology from Romao (2021). Where area estimates seem unrealistic (for example information provided by France for habitats 4030, 6510 and 3270, 3280, 9230) the values were adjusted, according to Romao (2020) and in exceptional cases, we decided to omit the unrealistic[11] area values for these specific habitats to ensure a more accurate estimate of unknown area, described further below. Romania was excluded throughout, as these values are largely overestimated. We present the total habitat area which relates to the 'best estimate' provided in the Article 17 reporting. Where a best estimate was not available, we used the median between the minimum and maximum reported values. This approach was also used for the habitat condition information. Member States report the area in good, bad, and unknown condition for each habitat. To determine the areas to be restored (i.e., existing areas in need of improvement) we used the not good condition information to represent the minimum area to be restored. The maximum area to be restored was calculated by adding the area reported as not good condition and the unknown area i.e., assuming all unknown area is in bad condition. An 'estimate' area to be restored was also calculated by distributing the unknown area proportionally to the good and not good condition areas, as described in Romao (2020). When calculating habitat areas within the Natura 2000, also reported under Article 17 of the habitats directive, estimated Natura 2000 area reported by the 26 member states (EU-27 excluding Romania) were used. Member states whose reported area values were deemed unrealistic, were also excluded when calculating areas within Natura 2000.

Carbon potential: We used the information on the carbon potential at the Annex I habitat level from Hendriks et al. (2020) and associated EEA work. The authors carried out a literature review of the observed or modelled carbon stocks and carbon sequestration rates in the major carbon pools of terrestrial and marine ecosystems. These include above and belowground biomass, soil carbon, deadwood, and leaf litter. Expert knowledge and interpretation were used to match the descriptions of vegetation type, location, and specific conditions in the

[11] In some cases, estimates for area in good condition and area in not good condition exceeded the total area.

various studies to a specific habitat type. The carbon information includes both organic and inorganic carbon and the information is provided in metric tons (Mt). For the carbon stock a minimum and maximum value is provided for the total area of each habitat type. For the potential carbon sequestration rate a mean rate in metric tons per year (Mt y^{-1}) is provided for the total area for each habitat. To determine the potential carbon sequestration rate associated with ecosystem restoration for each habitat type, we carried out the following calculations:

*(Potential carbon sequestration rate (mean) (Mt y^{-1})/Total habitat area (km^2)) * (Area to be restored minimum (km^2)) = Potential carbon sequestration rate (mean) for the minimum restoration scenario (Mt y^{-1})*

*(Potential carbon sequestration rate (mean) (Mt y^{-1})/Total habitat area (km^2)) * (Area to be restored maximum (km^2)) = Potential carbon sequestration rate (mean) for the maximum restoration scenario (Mt y^{-1})*

*(Potential carbon sequestration rate (mean) (Mt y^{-1})/Total habitat area (km^2)) * (Area to be restored estimate (km^2)) = Potential carbon sequestration rate (mean) for the estimated restoration scenario (Mt y^{-1})*

This assumes a constant carbon sequestration rate, regardless of habitat condition, biogeographical region, seasonal variation, management, and restoration measures across time. Furthermore, we assume that prior to restoration, habitats are not sequestering or emitting carbon, having reached an equilibrium between carbon inputs and outputs. We recognize these omissions, and the fact that depending on the habitat and its condition (e.g., the level of degradation and the management measures) the resulting uncertainty may differ substantially. However, as we do not have consistent and reliable figures available for the carbon sequestration rates related to bad and good habitat condition and at the habitat level, we provide an estimate here, following the information collected by Hendriks et al. (2020). This is the best available summary of the carbon information of European habitats available to date. In the literature log, we provide further information describing how carbon sequestration rates can potentially differ with habitat condition, habitat management measures and over time following restoration.

Additionally, these calculations assume that all habitat area in not good condition will be restored.

Carbon stock is provided for the total habitat area. As in Hendriks et al. (2020), a minimum and maximum estimate is provided. To calculate the amount of this stock found outside Natura 2000 areas, the following calculation was carried out:

*(Max carbon stock for the whole habitat area/ habitat area) * (Total area – Natura2000 area)*

The figures calculated for carbon stocks also contains some caveats. For example, carbon stock can vary interannually and can be enhanced by soil and land use management practices, which is not considered in the numbers provided. Furthermore, biogeographical location can play a role, for example in some biogeographical regions, the aboveground biomass may be greater than on average (e.g., in agroecosystems of the Boreal region). For both carbon stock and potential carbon sequestration rates studies report large variations across and within habitats and their carbon pools. Furthermore, different methods in the studies themselves make it difficult to state reliable figures for individual habitats and reliable estimates of carbon across all components of the carbon cycle are lacking at habitat-level. Additionally, there is a lack of studies providing intermediate complexity, with most presenting results from site level monitoring or global metanalyses.

Prioritization/Ranking: There are various ways to use the data to carry out a ranking exercise. We chose to prioritize for restoration and protection. For restoration prioritization the mean potential carbon sequestration rate for the maximum restoration scenario was used to identify the top five habitats per ecosystem type (according to the EEA classification). For protection prioritization, the maximum carbon stock was used (which estimated the total carbon stock of each habitat type according to the total area) to identify the top five habitats per ecosystem type. This is purely a ranking and prioritization exercise and should not be used to justify decision-making, as explained in the previous section, the figures used, especially for the carbon sections are only estimates and omit various importance aspects, which can have significant effects on the carbon potential.

We added a column to reflect the time frame to return the habitat to carbon stock equivalent to natural counterparts. If this information was not available, we used time frame for the habitat to return to good condition, based on biodiversity data. This estimate indicates the overall likelihood of restoration increasing carbon potential, which can further feed into the ranking exercise, depending on what the goal of restoration is e.g., if there is a specific time frame within which restoration measures must be completed.

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