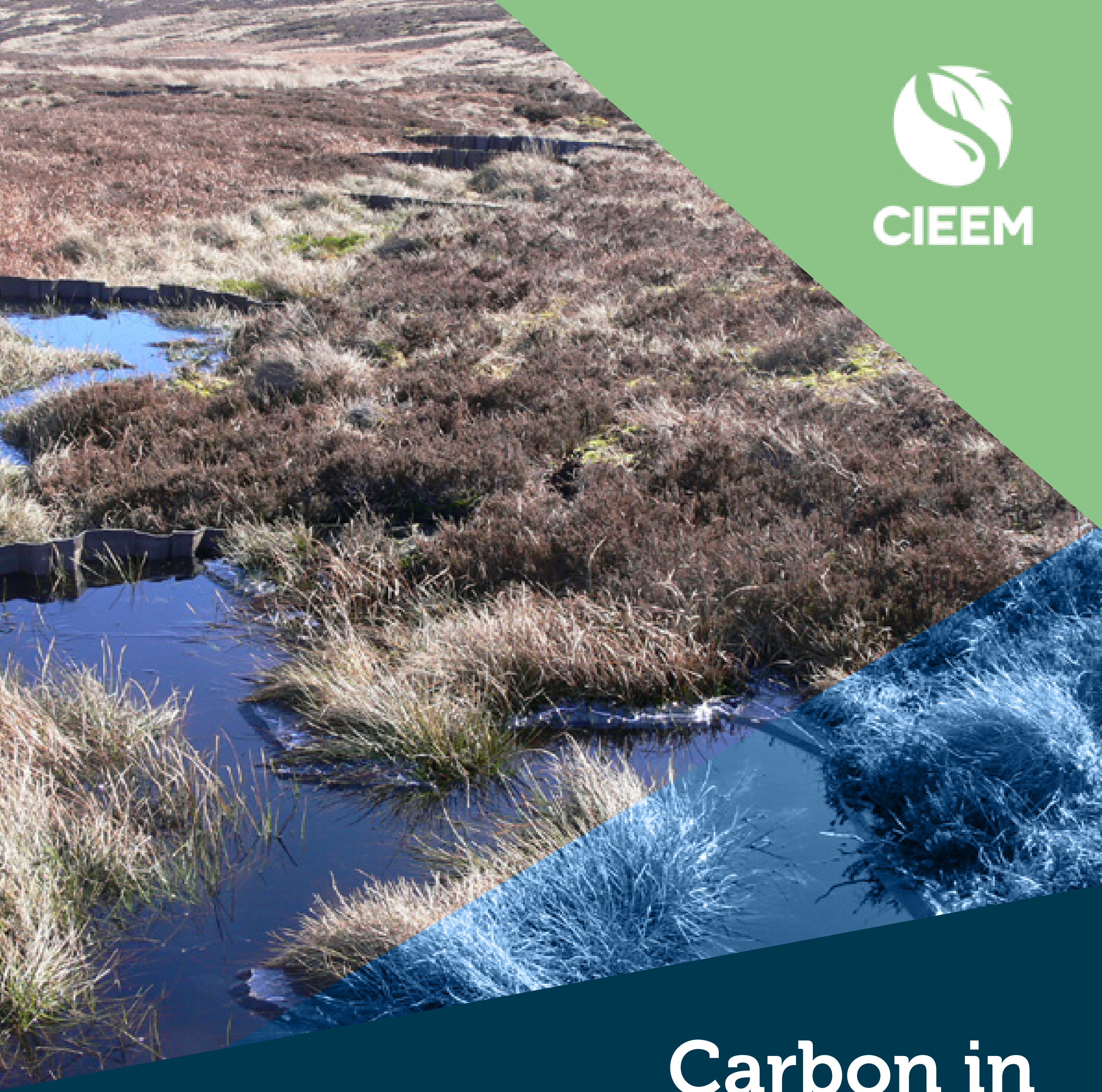

CIEEM

Carbon in Ecosystems

management, restoration & creation for carbon capture

Penny Anderson, CEcol, FCIEEM

March 2024 www.cieem.net

Declaration

I am a professional ecologist specialising in habitat creation, restoration and management. I first undertook this literature review to inform ecologists and environmental managers in 2021, and it was made available on CIEEM's website. I have now updated it with further research findings. I trust I have interpreted the literature correctly, but any comments or improvements would be welcomed.

Introduction

1. To reach the net zero carbon dioxide (CO_2) target requires removal of CO_2 from the atmosphere as well as significant reductions in CO_2 and other greenhouse gas (GHG) emissions. The latter include methane and nitrogen oxides, which are both more potent than CO_2. Methane is produced naturally mostly in wetland ecosystems and by ruminants' enteric fermentation and animal manures.

2. Nature-based solutions can make a significant contribution to reducing atmospheric carbon within a wider net zero strategy. This review focuses on carbon in ecosystems and the contribution these, especially in the UK, can make to capture more atmospheric carbon, whilst at the same time addressing the biodiversity crisis and incorporating other ecosystem services.

3. The priority must be to safeguard and enhance existing habitats to optimise their value for carbon capture and storage and for wildlife. Creating and restoring habitats without damaging those already of value is promoted.

4. Climate change has to be considered when planning new habitats and targets set that are achievable under future scenarios. Droughts, which are becoming more regular, as well as increasing temperatures and CO_2 levels are already affecting habitats, especially in lowland Britain.

Carbon in soils and vegetation (Chapter 2)

5. Most carbon is in soils - globally 3-5 times more is stored in soils than vegetation and 2-3 times more in soils than in the atmosphere. The creation and persistence of Soil Organic Carbon (SOC) is critical for carbon capture, which involves complex biological and biochemical interactions, depends on soil carbon:nitrogen (C:N) ratios and varies with environmental factors.

6. The soil microbes, fungi and soil fauna are key components of the carbon cycle in soils. Understanding their relationships is critical for identifying how to increase carbon capture, but the science is incomplete.

7. Soil organic matter is very complex in its chemical composition, molecular size and association with the mineral matrix, all of which are determined by vegetation type and climate as well as soil composition. Separation into particulate and mineral-associated organic matter helps partition carbon capture processes in soils.

8. The different and interactive roles of saprophytic and symbiotic fungi, (ericoid, ecto and arbuscular mycorrhiza) and their role in decomposition, including interactions with soil microbes, are important in determining carbon capture.

9. Different plant communities are able to transfer carbon to deeper soil layers and stabilise it in the mineral layer e.g. broadleaved trees, more diverse grassland with deeper rooted species and communities dominated by slow-growing, conservative rather than acquisitive species and via bioturbation from soil animals, e.g. earthworms.

10. 40-60% of soil carbon is in the soil's top 20-30 cm, although this varies with vegetation and environmental features. Many studies fail to measure deeper soil carbon stores, which can be significant, especially in floodplains and peat.

11. Peat contains more carbon than any other soils, often to great depths. Organic-rich soils have many times the level of carbon of other soils, followed by gleyed soils, stagnogleys and podzols. Brown calcareous earths and rendzinas have lower carbon contents owing to a more rapid breakdown of biomass and often low productivity. In mineral soils, more carbon can be stored in clay-rich soils than sandy ones and in soils in wetter, cooler climates, even within the UK.

12. In general, soil carbon stocks are highest in peatland, followed by heathland, closely followed by soils under woodland, then different kinds of grasslands with arable

Soil profile with high OM at surface typical of a Podzol.
(Greensides, near Buxton)

soils at the bottom of the list.

13. Carbon stocks in vegetation are generally much lower than that in soils. Only woody plants – trees/shrubs, dwarf-shrub heathland and saltmarsh hold significant carbon stocks.

14. There is believed to be a carbon saturation equilibrium in some mineral-based habitats such as grasslands, although there is some contrary evidence for ancient semi-natural woodlands, active floodplains and marine environments. However, it can take 100 years or more to reach any new equilibrium, especially from very degraded states.

15. Total carbon stocks in the UK depend on the extent of each habitat, thus that in pasture may be disproportionately greater than that in woodland owing to its widespread scale rather than its intensity of carbon.

16. Carbon is lost when soils are damaged or disturbed and habitats lost. Higher losses than any sequestration rates occur from heavily drained peatlands, but losses occur when any habitat is degraded or lost, whether this is during land management operations or when disturbed or removed for development. Ploughed arable land mostly loses carbon over the annual cycle.

Carbon and different habitats (Chapter 3)


Ancient woodland and limestone grassland in Manifold Valley, Staffordshire

17. Carbon sequestration can be restored and increased. Although over time new woodland can accumulate more carbon than most other habitats, it usually takes at least 20-30 years to reach a positive carbon budget and decades for carbon stocks to accumulate. Other habitats can show near equivalent carbon sequestration rates and sometimes more rapidly than can woodland.

18. There is sufficient evidence to advocate for a mixture of new habitats rather than a dependency on planting trees, although there is inadequate research for the best approach in all habitats.

    - Restoring **peatlands** is imperative to stop or significantly reduce the current carbon losses that contribute significantly to climate change. Active peat (i.e. net accumulation) is more difficult to achieve and sequestration levels depend on high water tables, but managed to minimise methane emissions. Full restoration of peatlands can take many decades or more depending on the level of previous damage. Paludiculture has the potential to cut carbon emissions on agricultural peats.

    - Ancient and old growth **woodlands** can sequester carbon over hundreds of years. Restoring and maintaining existing woodlands with minimum management will optimise the carbon budget over time. Creating new woods can lose more carbon than they sequester, at least for some decades, due to ground disturbance in establishment, loss of previous carbon, the time needed for canopy development and litter fall and for establishment of woodland soil microbial and fungal functioning.

    - Natural colonisation, preferably on clay soils that are already disturbed, with supplementary planting of ‘missing’ species, minimal soil and habitat disturbance or damage, using a range of native broadleaved species and minimising management would all result in the largest carbon store over the longest period.

    - Organic-rich soils (generally more than 15% organic matter) and peat should **not** be planted (or drained or disturbed for planting) as these then lose more carbon through organic matter decomposition than they can accumulate.

    - Carbon cycle estimates for trees are derived mostly from forestry research linked to **plantations** rather than semi-natural woodland. The gains in carbon depend on tree density, thinning, the harvesting cycle, degree of ground disturbance for harvesting and re-planting and future use of the harvested wood. High carbon sequestration rates occur in the faster growth periods for trees but are not sustained. Conifer plantations have a threshold beyond which further carbon is not accumulated.

- **Floodplain meadows** with full plant diversity and regular flooding can be high value carbon accumulators continuing for centuries.
- **Heathland** sequestration rates compare well with other habitats when heather is in its building phase, but then declines, although the contribution of mosses has not been fully explored. Heathland carbon sequestration rates can be double that of acidic grasslands on similar soils.
- **Neutral Grasslands** restored to high diversity with red clover (or other deep-rooted legumes) added can sequester as much carbon as many other habitats, but there is considerable variation. Low/intermediate grazing levels and no artificial fertilisation are essential for high carbon capture.
- **Acid grasslands** can hold high carbon stocks, but sequestration levels are only modest. Little information exists on how to optimise carbon capture in other grassland types.
- Small well vegetated **ponds** and small **lakes** can have high carbon capture rates, but methane emissions need to be minimised or these water bodies can be GHG emitters.
- Optimal conditions in **rivers** for trapping carbon are those with low gradients, with high channel complexity and plenty of dead wood, but carbon sequestration levels are not available and generally are probably low.
- There is little information on other **wetland habitats**, although reed beds could be valuable carbon stores. Fens managed for nature conservation hold more carbon than damaged or degraded sites.
- **Marine and coastal** habitats hold more carbon than terrestrial ones in a smaller area. The most important are saltmarshes, seagrass beds and estuarine muds, but others like maerl beds and biogenic reefs are important where they occur.
- **Saltmarsh** sequestration rates can be high and the habitat created on a large scale. Deposits can build up too over time but might be squeezed by sea level rises. Managed retreat has a significant role in increasing saltmarsh habitat.
- **Intertidal and subtidal sediments** and shallow sea basins can be important sediment and carbon sinks but are vulnerable to disturbance.
- **Seagrass beds** can capture carbon rapidly in sediments and vegetation but have suffered large-scale losses and are vulnerable to damage and disturbance. Restoration is possible and new techniques are being developed.
- **Urban** environments have the potential to contribute to carbon sequestration and stocks through urban trees, other habitats, green roofs and gardens. Although these cannot significantly reduce urban CO_2 levels, they can make a valuable contribution directly and indirectly to achieving net zero.

Saltmarsh in Dyfi Estuary, Wales

19. **Peatland, woodland and saltmarsh carbon codes** have been or are being developed.

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Units and abbreviations

- tonnes carbon per hectare (tC ha^{-1}) or per year yr^{-1}.
- CO_2e = tonnes of carbon dioxide equivalent, thus including other GHGs, although these are not always measured, so may not be representative.
- Divide by 3.667 to convert CO_2e ha^{-1} yr^{-1} to tC ha^{-1} yr^{-1}.
- 1000kgC km^{-2} = 0.01tC ha^{-1}

GHG – Greenhouse Gas

GHGs – Greenhouse Gases

SOM – Soil Organic Matter

SOC – Soil Organic Carbon

OC – Organic Carbon

TOC – Total Organic Carbon

ECM – Ectomycorrhizal fungi

ERM – Ericoid mycorrhizal fungi

AM - Arbuscular mycorrhiza

ICW- Integrated Constructed Wetlands

1

Chapter 1

Introduction

2022 was a year of unmitigated international disasters from floods and wildfires to heat waves and droughts, with severe effects on people, food production and ecosystems. The frequency and severity of these calamities now, and the predictions for the future, are underlined by the stark warnings spelt out in the most recent IPCC report (2022), which point overwhelmingly to the urgent need for drastic cuts in carbon emissions. This is a wake-up call to us all to increase our endeavours urgently to reduce greenhouse gases (GHGs). This review sets out the evidence base on which you can make these decisions in relation to habitat management, restoration and creation.

The approach in the UK is manifold, from Government level to individual choices. The Committee on Climate Change (CCC), an independent, statutory body established under the Climate Change Act 2008, advises the UK and devolved governments on emissions targets and reports to Parliament on progress made in reducing GHG emissions and preparing for and adapting to the impacts of climate change. In 2019, the UK Government committed to net zero target, as the CCC recommended, by 2050. The Scottish Government, however, has set a more challenging net zero target by 2045 with a 75% reduction by 2030 (Oyesiku-Blakemore & Dondini, 2022), while Ireland's Climate Change Action Plan commits to a legally binding net zero target by 2050 and a 51% reduction by 2030. Net zero describes a comprehensive commitment to decarbonisation and climate change action and is a state when carbon emissions into and removal from the atmosphere are balanced. The longer it takes to reach this goal, the more carbon and other GHGs there will be in the atmosphere and the greater impact they will have on global climate and therefore also on people and ecosystems. At the same time, therefore, it is essential to reduce the current GHGs in the atmosphere to a lower, safer level.

The most relevant of the CCC (2020) recommendations to this review is one of the five investment priorities that includes a landscape-scale change to increase tree cover to 17% of the land surface by 2050, (currently at 13%, with 3.21 million ha - the new target - needing an additional 30,000 ha per year by 2050), further lowland and upland peatland restoration and an increased quantity and quality of urban greenspaces, including green roofs, tree planting, park restoration and sustainable drainage schemes. Pledges have already been made by the Scottish Government in 2022 (up to £95 million towards woodland creation targets and over £12 million for peatland restoration to meet 2025/6 targets), the Welsh Government (£17 million over the next two years) and for England (£750 million Nature for Climate Fund 2020-2025 primarily for peatland restoration and tree planting), although only limited assessments of the land-use change needed and climate change mitigation achievable at this scale have been undertaken (Bradfer-Lawrence *et al*., 2021).

UK GHG emissions in 2018 were 44% below those of 1990, mostly due to reductions in electricity generation, and in the waste and industrial sectors (CCC, 2020). We were on track to meet the carbon budgets up to 2022 but not subsequent ones. The CCC considers we need to achieve an annual emissions target reduction of 15.5 $MtCO_2e$ per year for the next 30 years to meet this target.

As well as government targets and policies, many Local Authorities, Institutions, organisations and businesses have also developed carbon reduction or net zero carbon plans and targets. Individuals are also making a contribution through personal decisions on energy source and use and other carbon-saving measures. Many efforts depend on nature-based solutions (NbS) supported by the evidence-base presented here, plus that by Gregg *et al.* (2021) and Stafford *et al*. (2021).

Tree planting, farmland, Cheshire edge of Peak District

The biodiversity crisis is the other critical challenge that has to be addressed simultaneously and is often an integral part of the climate emergency. At the same time, there are other ecosystem services that can be enhanced or restored. For the most part, NbS provide multiple benefits, contributing to carbon capture and reversing the biodiversity crisis, whilst simultaneously reducing flooding, improving water quality, increasing pollinator services and enhancing wellbeing for local people as part of their ecosystem services (Stafford *et al.,* 2021, Bradfer-Lawrence *et al.,* 2021). NbS are also cost-effective solutions ameliorating several issues simultaneously compared with hard engineering. These are strong and persuasive arguments for applying NbS as widely as possible.

However, the overall capacity of habitats to contribute to climate change targets needs to be put into perspective. Currently, habitats overall in the UK are estimated to be an emission source of 11.5 $MtCO_2e$ if losses from peat are included, with more than 80% of the losses coming from lowland agriculture on peat (https://post.parliament.uk/research-briefings/post-pn-0668/). Excluding these losses, all other habitats are sequestrating around 9.8 $MtCO_2e$ per year (Cornelius *et al*., 2020). With a reduction of 15.5 $MtCO_2e$ per year needed (CCC, 2020), NbS alone cannot generate the significant emission reductions essential to reach net zero. Indeed, Bradfer-Lawrence *et al*. (2021) point out that the cumulative total carbon capture of the most ambitious CCC programme through peatland restoration and woodland expansion targets with saltmarsh restoration added into the mix is equivalent to only about three years' worth of the UK's total annual net emissions (estimated to be 424.5 $MtCO_2$ in 2021, O'Sullivan, 2022).

The conclusion is that habitats have a vital part to play in reaching net zero as part of a wider strategy to decrease emissions and reduce the effects of climate change. Indeed, there is an international 4 per 1000 initiative to increase soil carbon annually by 0.4%, particularly on agricultural land, that was introduced in 2015 at COP21 as part of the Lima-Paris Plan of Action, https://sdgs.un.org/partnerships/4-1000-initiative-and-its-implementation. This worthy goal has now been signed by more than 550 members and partners as part of the Global Agenda for Action which is partnered by the FAO's Global Soil Partnership and the UN Decade on Ecosystem Restoration.

The opportunities are immense for our profession as environmental managers and ecologists to think outside our ecological boxes and to embrace the wider ecosystem needs. We should be at the forefront of habitat creation and restoration and be thinking large scale and in an integrated fashion to maximise the benefits for society and wildlife through NbS that also contribute significantly to other ecosystem benefits. We should be stepping up to contribute to policy making and reviews too. This all means both working more in collaboration with other specialists, especially soil scientists, to ensure these opportunities and benefits are welcomed, understood and embraced but also ensuring more and new training to understand carbon in habitats better.

This review focuses on carbon and biodiversity and what the ecological profession can do, but many of the solutions also contribute significantly to other ecosystem benefits. The detailed scope is not always easy to quantify. There has been much research across the world, but finding definitive numbers for a particular habitat in a specific location is still difficult. Habitats, species composition, geographical contexts, soils and history of management or disturbance all vary on a small scale. We do not have time to wait for all the detail to be available but need to act now based on the available information. Care therefore needs to be taken in interpreting, adopting and adapting the data presented here and in other sources. Limitations on data need to be recognised and their application undertaken with care (e.g. for peatlands, see Heinemeyer and Ashby, 2023).

2

Chapter 2

Carbon and ecosystems

Life on Earth is carbon based with the carbon cycle taking carbon dioxide (CO_2) from the atmosphere and oceans into organisms and back out again. Ecosystems can store the carbon in vegetation, soils, sediments and shells over decades or millennia, (Gregg *et al*., 2021). These carbon stocks are not necessarily static and change over time with habitat development or degradation. The rate at which carbon is released back into the environment is critical in determining its accumulation in ecosystems. At present just over half of emissions arising from human activity are absorbed by land or oceans. The rest remains in the atmosphere as CO_2, which, along with other GHGs, gives rise to the consequences of global warming (Deng *et al*., 2016).

GHGs include methane and nitrous oxide as well as CO_2. These are (although different authorities give different measures) respectively 28 and 298 times more potent as GHGs than CO_2 over 100 years. In ecosystems, methane is a product mostly of wetlands and of enteric fermentation by ruminants and animal manures, whilst nitrogen is lost from fertiliser and manure application, soils, sediments and water bodies (Zhu-Barker & Steenwerth, 2018). The highest emissions from different land uses are from cropland and intensively managed grassland (16.28 and 7.39 kg N^2O-N ha^{-1} yr^{-1} respectively, Evans *et al*., 2023). Added to these, of course, are industrial applications and transport emissions. For example, industrial nitrogen fixation for agricultural inorganic fertilisers contributes 1.8% of global CO_2 emissions (https://royalsociety.org/-/media/policy/projects/green-ammonia/green-ammonia-policy-briefing.pdf.) but this in combination with losses of nitrogen after deployment accounts for some 5% of GHG emissions (Jain, 2023). Nitrogen oxides are also derived from the combustion of fossil fuels, with nitric oxide the dominant portion. According to DEFRA statistics, totals have reduced since 1990 and in 2021 were as follows for different sectors:

Sources of UK annual emissions of nitrogen oxides	2022 million tonnes
Energy industries	0.122
Industrial Combustion	0.093
Non-road transport	0.098
Road transport	0.195
Other	0.135

Source https://assets.publishing.service.gov.uk/media/65c5f3f69c5b7f0012951bad/nitrogen_oxides_key_emission_sources_2022.csv/preview

Cumulative and short-lived climate pollutants impact in different ways. CO_2 is an accumulative pollutant causing radiative forcing in line with its total stock, whilst that of methane, for example, scales with its emission rate. Allen *et al.* (2018) suggest that methane's short life needs to be given more consideration in the calculation of global warming potential (GWP), relabelled as GWP*, which tracks the sustained level of methane emissions. They argue that this remodelling allows emissions to be considered in a common cumulative framework and provides a more accurate measure of progress towards climate stabilisation. Any changes in accounting could have significant effects on how methane emissions are treated. The IPCC's 4th Assessment values, for example, were based on the 100 year GWP for methane and nitrogen oxides.

2.1 Carbon in soils

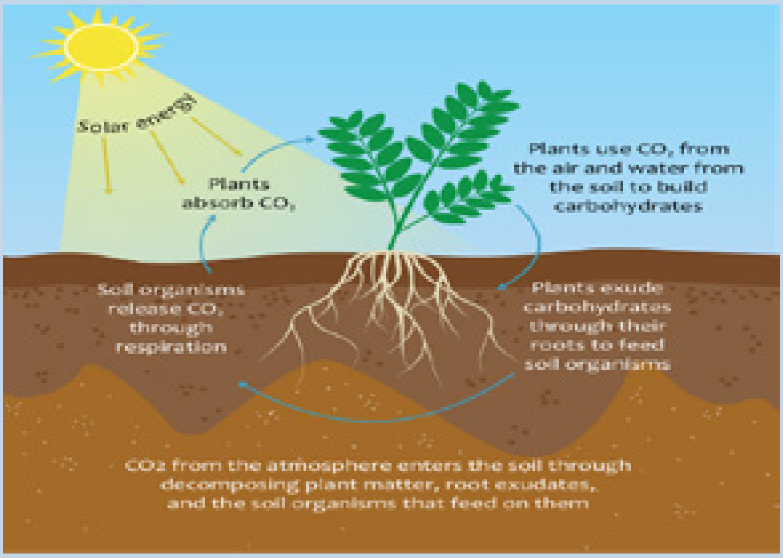
Soils are fundamental to the carbon cycle (**Figure 1**). Globally there is 3-5 times more carbon stored in soils than vegetation and 2-3 times more than in the atmosphere, although the amount of soil carbon stored under any habitat depends on climate, soil texture, site management and history (British Society of Soil Science, 2021). Soil Organic Matter (SOM), which normally only constitutes some 1-5% of the soil mass (except in organic soils), is the fundamental basis of the terrestrial carbon cycle, mediating the flow of carbon, nitrogen and other nutrients as they cycle between the atmosphere and the land (Sokol *et al.*, 2022). SOM is very complex in its chemical composition, molecular size and association with the mineral matrix, all of which are determined by the type of vegetation and climate as well as the soil composition. Our understanding of the processes affecting soil carbon turnover places increasing importance on the role of substances produced by plant roots and soil microorganisms and particularly how these interact with soil minerals to form aggregates.

1

Figure

The Carbon Cycle

(from Lavallee, https://theconversation.com/soil-carbon-is-a-valuable-resource-but-all-soil-carbon-is-not-created-equal-129175).



The decomposition cascade

The critical components in soils that play a major role in carbon cycling are the plant litter quality, bioturbation (principally through earthworms), soil microbes (principally bacteria and archaea) and fungi. An understanding of how these interact with SOM and determine carbon levels in soils helps to focus efforts to capture more carbon. The carbon sources in the soil are plant (mostly) and animal debris, which form the surface litter, with dead roots (small roots often turnover rapidly) and plant exudates in the rhizosphere: the small space around roots. Plant litter is more complex physically and chemically than root exudates.

Litter breakdown occurs through fragmentation into smaller sizes, leaching of soluble compounds into the soil and catabolism by decomposer organisms. Saprophytic fungi are the primary decomposers in the litter layer, using this as their carbohydrate source, whilst mycorrhizal fungi are responsible for much of the breakdown of partly decomposed material below the litter level (Bödeker *et al.*, 2016) as they do not have the range of enzymes needed to decompose cell walls fully. Many soil invertebrates also feed on plant litter, together with the fungi, producing a nutrient cycling cascade of resource quality (Read *et al.,* 2004). Burrowing earthworms are particularly important as structural engineers and in their bioturbation role, affecting air and water relations in soils, feeding on and breaking down litter, mixing material far down the soil profile. They are also important chemical engineers owing to their involvement in nutrient cycling and interactions with microbial activity (Le Bayon *et al*., 2017).

Roots contribute the majority of carbon input into soils, both as litter and via root and root-derived carbon exudates, thus deeper rooting plants may have the potential to produce more carbon at depth than shallow rooted species (Detheridge *et al*., 2014). Exudates comprise organic acids, amino acids, protein, sugars, phenolics and other secondary metabolites, which are more labile carbon compounds compared with material in plant litter. 30-50% or more of the plant's fixed carbon is exuded by roots, depending on the species. Plants can even adjust their root exudates to stimulate specific microbial activity, especially during times of stress, so that they produce various metabolites that increase, for example, the resistance of roots to drought or pathogens, (for further information see https://soilguide.co.uk/members/page16.html).

Carbon is lost through respiration, including from the soil via microbial respiration. This also includes stimulating microbial decomposition via root-derived carbon inputs (sometimes referred to as soil carbon priming), whilst a proportion of the original carbon is retained in soils. SOM can be seen as a continuum of progressively decomposing organic compounds (Lehmann & Kleber, 2022). Microbes use humic fractions of organic matter and build bio-pores throughout the soil profile. These are critical in providing habitat for root growth and microbes and act as channels and stores for water and air owing to their highly charged nature. Carbon built into these bio-pores can remain stable within the soil for hundreds or more years. Humus also attaches to and stabilises clay particles and thus has an important role in water movement, retention and nutrition.

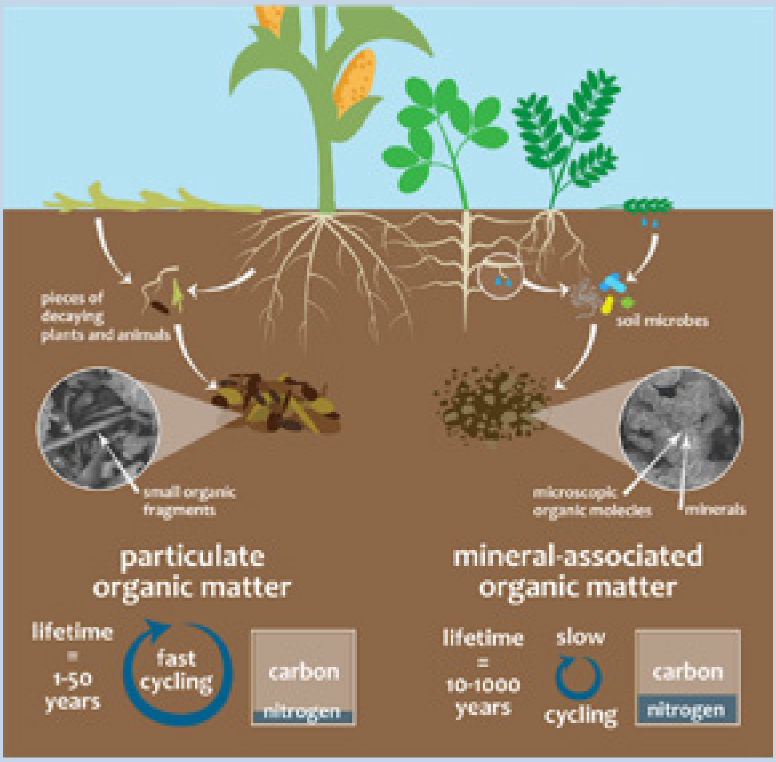
A soil's clay content also determines the amount of Soil Organic Carbon (SOC) stored in mineral soils as particles of organic matter become absorbed onto clay surfaces, coated with clay particles or buried inside small pores or aggregates – all of which protect them from microbial attack. In sandy soils, in contrast, microorganisms are able to access organic carbon more easily, which results in greater decomposition. Climate also interacts through regulation of plant productivity, so the potential SOC will be lower for similar soils in drylands compared with wetter climes.

The creation and persistence of SOM is dependent on complex interactions between soil biota (stressed by Filser *et al.*, 2016), the chemistry and physics of the mineral soil, temperature and precipitation (Hunt *et al.*, 2020) and is often divided into Particulate and Mineral Associated Organic Matter (POM and MAOM) dependent on their physical or chemical characters (**Figure 2**).

2

Figure

The formation of particulate and mineral-associated organic matter form and function (Lavallee, https://theconversation.com/soil-carbon-is-a-valuable-resource-but-all-soil-carbon-is-not-created-equal-129175).



Lavallee *et al.* (2020) summarise their general properties (**Table 1**).

Table 1.

Summary of the properties of Particulate Organic and Mineral Associated Organic Matter (Lavallee et al., 2020).

Property	POM	MAOM
Residence time	<10 years (depending on soil type and circumstances)	Decades or centuries
Main formation route	Fragmentation, depolymerisation	Transformation or modification of low molecular weight compounds
Subject to saturation	No	Yes?
Main constituents	Plant derived e.g. phenols, celluloses, fungal derived e.g. chitin	Low molecular weight compounds of microbial e.g. polysaccharides, amino acids, and plant origin
C:N ratio	10-40	8-13

Nutritional role	Not assimilable by plants nor mostly by microbes, more complex compounds	More simple compounds, more assimilable for microbes & plants
Protection	None	Mineral associations – occlusion in aggregates, organo-mineral clusters, micropores, sorption onto mineral surfaces

POM is the product of litter and rhizosphere through fragmentation and is only partially processed by soil biota meaning that it is more structurally complex and lightweight (it actually floats in water when separating POM from bulk soil). POM dominates in organic-rich soils such as peat (Sokol *et al*., 2022). MAOM is a diverse pool that can form in different ways but encompasses more microbial-derived compounds (more so in grasslands) or materials that have leached from plant material (more so in temperate forests) and tend to be small biopolymers and monomers such as proteins, lipids and polysaccharides. These compounds are of a heavier density, a small size and are more nutrient-dense owing to the lower C:N ratio making them easier for plants and microbes to access. Litter with lower C:N ratios typically forms MAOM more efficiently, such as from legumes or those plants with a lower lignin content. Fine root biomass, length and surface area (as in many grasslands) and deeper roots (as in deciduous trees and diverse grassland) are also positively associated with greatest MAOM formation (Sokol *et al*., 2022). Our understanding of the increased importance of the role of plant exudates and microbial substances in forming MAOM compares with the view only 20 years ago when litter quality was considered to be the main biotic driver.

This differentiation of POM and MAOM and research into the processes and environmental factors in determining rates of accumulation in different climates and soils plays an important role in the development of soil carbon models leading to improved global-scale predictions of soil carbon stocks (e.g Abramoff *et al*., 2021).

MAOM can persist for much longer in soils, at least in temperate regions, owing to limitations of the interactions between decomposers and organic matter by the complex three-dimensional architecture of the mineral mix as well as soil physiochemical properties which promote sorption. Soussana *et al*. (2010) quote figures of millennia and 2-10,000 years in undisturbed, deep soils in America and France respectively using ^{14}C dating methods. There is, though, a subset of MAOM which is highly dynamic (Sokol *et al*., 2022). MAOM is more important for nitrogen sources for plants and microbiota, whilst it is the POM that is much more vulnerable to loss or alteration due to soil disturbance such as ploughing. MAOM formation can also be altered or diminished by physical disturbance or use of agro-chemicals that affect the soil micro-organisms (Lavallee *et al*., 2020, Sokol *et al*., 2022).

Retention and storage of persistent MAOM is key to increasing long-term C stocks in soils. Indeed, globally, MAOM accounts for 34-51% of the total organic C in the terrestrial biosphere and 65% of the carbon in SOM, although this varies widely across ecosystems, (Sokol *et al*., 2022). Across the range of soils in one study, grasslands had consistently more MAOM and less POM than forests, especially coniferous forests (Cotrufo *et al*., 2019). Although POM can in theory continue to accumulate indefinitely, mineral soils are thought to have a carbon saturation capacity (although there are exceptions, described later) which depends on the vegetation, climate and management (Gregg *et al.,* 2021).

If soils are disturbed or changed, a new equilibrium will be reached in the altered ecosystem, but there is still considerable disagreement on the direction and magnitude of change in soil carbon stocks with some landuse changes, making broad generalisations difficult (Deng *et al*., 2016). Severely degraded soils (ploughed, drained or stripped for development for example) can take many decades or more to re-build high carbon levels of both POM and MAOM.

There are also differences amongst the mycorrhizal fungi that determine carbon capture. Mycorrhizal fungi form symbiotic associations with plants, in particular with some 80% of angiosperms and all gymnosperms, assisting in obtaining water, phosphorus and other micronutrients from the soil, which benefit the plants, receiving carbon-rich substances in return. Ectomycorrhizal (ECM[1]) and ericoid mycorrhizal fungi (ERM) produce extracellular enzymes (more so in ERM than ECM) giving them greater access to organic nitrogen sources than arbuscular mycorrhiza (AM), (Cotrufo *et al*., 2019). Ericaceous species usually occupy soils low in nitrogen. ERM fungi can access chitin (a major component of fungi) as well as the very acidic and lignified plant litter to release nitrogen which is then available to the plants (Read *et*

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[1] Ectomycorrhizal fungi are those that wrap round plant roots rather than invade their cells, ericoid mycorrhiza are associated specifically with heathland shrubs in the Ericaceae.

al., 2004). These ERM are essentially ecosystem engineers of the heathland systems and partly account for the high carbon levels in heathland soils, along with the litter quality, pH and bedrock.

In contrast, AM fungi are dependent on saprotrophic organic matter decay and mineral nitrogen production and do not produce extracellular nitrogen-degrading enzymes (and thus nitrogen depletion in the SOM and a greater C:N ratio). As a result, Averil *et al*. (2014) show that ecosystems dominated by ECM fungi contain 70% more carbon per unit nitrogen than soils in ecosystems dominated by AM–associated plants. Similarly, Cotrufo *et al.* (2019) found more carbon stocks in ECM compared with AM-dominated broadleaved and mixed forests across Europe.

ECM fungi (with over 20,000 species of basidiomycetes, ascomycetes and zygomycetes) can account for 30% of the non-plant biomass in forest soils. They characteristically colonise lateral roots of host trees forming an interlacing structure that penetrates between and around root epidermal cells to form a "Hartig net". This is connected to a mantle that envelops host roots and extends into the soil for resource exchange, thus providing an extended surface area that is the site of nutrient exchange (Stuart and Plett, 2020). ECM fungi form symbiotic relationships with 80-90% of all temperate and boreal forest trees[2]. They collectively drive forest soil processes e.g. SOM decomposition, nutrient cycling and carbon sequestration, although there is variation between species, and are believed to supress soil respiration and thus increase soil carbon storage, although this may depend on the specific fungal species and soil conditions (Stuart and Plett, 2020).

Russula ectomycorrhizal fungi in conifer plantation, Sperrins, N. Ireland

AM fungi in the phylum Glomeromycota (with only some 100s of morphotypes) penetrate into cells, forming tree-like branch structures (arbuscules) and have a dense hyphal network that enmeshes and crosslinks soil particles, helping in water retention and soil structure, plus providing a carbon sink (2-5% of SOC), which can last up to decades. The AM fungi are dependent on saprotrophic organic matter decay and mineral nitrogen production (and thus nitrogen depletion in the soil organic matter and a greater C:N ratio) rather than producing nitrogen-degrading enzymes as ECM fungi do. They help improve nutrition and stress resistance and are associated in particular with herbaceous plants.

2.2 Carbon in the soil profile

Many studies measure soil carbon to only 15 or 30 cm, but Cotrufo *et al*. (2019) note a soil carbon average of only 40-60% in the top 20 cm, similar to Salome *et al*.'s (2010) estimate of 50% of the soil carbon below 20-30 cm. This average varies with habitat and soil type though. Total carbon stocks in grassland soils up to 1 m depth is more than three times that estimated for only 15 cm depth (Ward *et al*., 2016), whereas in forests, some 50% of soil carbon lies below this depth (Cotrufo *et al*., 2019). Functioning floodplain grasslands, with regular deposition of sediment during flooding, accumulate carbon in buried profiles over time mostly below 1 m depth resulting in 34% more carbon/ha being stored in these soils in the 0-3 m depth rather than 0-0.3 m depth (D'Elia *et al.,* 2017). The dearth of comprehensive depth measurements for a range of soils and habitat types necessitates careful interpretation and comparison of the available data and exposes one of the knowledge gaps in measuring carbon stocks comprehensively.

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[2] See https://mycorrhizae.com/wp-content/uploads/2017/03/Mycorrhizal-Status-of-Families-and-Genera-v1.6.pdf for a list of mycorrhizal types in a wide range of plants.

Effect of plant community on SOC stocks

The plant community also affects the amount of carbon that reaches the mineral layers. For example, broadleaved trees have a greater capacity than conifers to transfer carbon to deeper soil layers and stabilise it in the mineral layer. This could be related to the higher root mass of broadleaved trees and their exudates (Crane, 2020). Henneron *et al*. (2019) show that in grasslands at least, more competitive, acquisitive plants such as rosebay willowherb (*Chamaenerion angustifolium*), rye grass (*Lolium* spp) or nettles (*Urtica dioica*) for example, grow faster than slow-growing, conservative species, resulting in a larger and faster turnover of SOC through rhizosphere deposition. Communities dominated by slow-growing, conservative species (Grime's stress tolerators, Grime, 2001) like sheep's fescue (*Festuca ovina*), quaking grass (*Briza media*), or harebell (*Campanula rotundifolia*) are associated with slow decomposition promoting high SOC sequestration. The mechanisms for this are not fully understood but might relate to fast-growing species triggering SOC loss by enhancing soil microbial activity and hence SOC decomposition, thus releasing nutrients which in turn enhance plant growth.

Organic-rich mor humus over loess in the Peak District uplands

Non-mineral soils such as peats differ in that the organic matter does not decompose owing to waterlogging, low pH (except in some fen peat) and low temperatures, all of which slow decomposition (Detheridge *et al.*, 2014). Compared with mineral soils where earthworms are key, as pH decreases, enchytraeids take over, but are not the ecosystem engineers that earthworms are, so there is far less mixing, resulting in a layered humus which accumulates and locks away more of the SOM, eventually leading to peat formation in the right conditions.

Stored carbon is greatest in this mostly undecomposed material at 259/576 tC ha^{-1} to 0.5/1 m depth respectively, (Alonso *et al*., 2012, Cantarello, 2011), and much greater for deeper peat – 5248 tC ha^{-1} on Dartmoor for peat 6.63 m deep (Fyfe *et al.,* 2014). Milne and Brown (1997) give soil carbon for the Avery Soil groups for England and Wales (depth not specified) in which peat and earthy peat soils stand out at over 2-3 times the levels in other soils (1122 and 766.7 tC ha^{-1} respectively), with gleyed soils, stagnogleys and podzols in the 200-440 tC ha^{-1} range, although these figures have been questioned based on issues with soil densities and have since been updated to provide estimates of total carbon stocks under different landuses and in the different GB countries (Bradley *et al.*, 2005). As podzols are extensive, they hold about 10% of all UK soil carbon at 175-211 tC ha^{-1} (Alonso *et al*., 2012, but topsoil only measured). This emphasises the importance of these soils, but does not account for the extent and depth of peat as a vital carbon store as suggested in **Table 2,** which shows a ranking of average carbon content found in soils but only to 1 m depth beneath different habitats in Scotland. Peatland soils stand out, with the organic-rich heath and moorland soils not far behind. Woodland fills the intermediate position, with grasslands and then arable soils comparatively low in carbon stocks.

Table 2

Carbon content in some Scottish soils (Lilly & Baggaley, 2020).

Soils under different habitats	Carbon tC ha^{-1} to 1 m
Arable	115
Improved grasslands	138
Semi-natural grasslands	185
Woodland	267
Heathland/Moorland	290
Peat	528

More detail of carbon stocks is given in **Table 3**, but mostly only to 30 cm depth, which shows humic-alluvial gley soils, of the mineral types, hold important carbon stocks and the potential importance of flood meadows for soil carbon. The list also highlights the importance of salt marsh and woodland soils and the generally low levels found in more base-rich conditions.

Table 3

Examples of soil carbon stocks.

Soils or habitat	Carbon stocks in soil (tC ha^{-1})
Humic-alluvial gley soils	438
Peatland	259 to 0.5 m, 354-619 to 1 m, 5,248 to 6.3 m, raised bog 1,620 to 3.8 m
Floodplain grasslands	286-354 (1 to 3 m)
Podzols under heath	175-211
Seagrass	6.65-194
Saltmarsh	29-93 0.1-0.3 m,
Broadleaved mixed wood	108-173, 255-354 to 1 m
Acid grassland	87
Heath lowland and upland	81-103
Bracken	55-77
Fen, marsh, swamp	76, 810-2530 on 3.8 m peat
Conifer plantation	73-120
National average all wood types	62-66
Calcareous grassland	51-69 to 0.15 m
Neutral grassland	60 to 0.15 m
Agriculturally improved grassland	59-61, 72-204 to 1 m
Arable	43-64
Hedges untrimmed for 3 years	98.7
Ponds	16-28
Rivers	0.2-4.8

NB soil depths of measurements differ between projects, most are only to 0.3 m unless specified. Data from Alonso *et al.*, 2012, Gregg *et al.*, 2021, Heinemeyer *et al*., 2023, Axe *et al*., 2017 & Milne & Brown, 1997 for soils and see text. Note that there is considerable variation between sites and authors, sometimes based on different assumptions or inbuilt parameters. Use the data as a guide.

Some SOC stocks will also vary greatly over time or between locations according to the above ground vegetation, such as under conifers where soil carbon varies with age, density and species of tree. This is exemplified in Ireland by Tuohy *et al.* (2021), whose study of ten dairy farms on heavy textured soils down to the 'C' horizon (this varied from 112 to 160 cm overall sample depth) revealed both within and between field differences as well as variation with soil type from 172 tC ha^{-1} in humic podzols to 319 tC ha^{-1} in stagnic luvisols (rich in clay). The soils that also stood out (apart from oligotrophic peat) were Brown podzolic, Brown earths and Alluvial soils with levels around 300 tC ha^{-1}. These figures were mostly more than double the official large-scale estimates for Irish soils, but no information was gathered on whether they were stable, increasing or losing carbon. Rather, an estimate of the potential capacity for increasing the carbon stock was offered which was quite significant, if achievable.

2.3 Carbon in vegetation

Table 4 shows how little carbon vegetation holds compared with that in soils but highlights woody vegetation and saltmarshes as exceptions. Carbon is stored as trees and shrubs grow, accumulating more rapidly when growth is stronger in the early part of their lives, but slowing down as they mature. About 50% of the total tree carbon is in the harvestable trunk, but the amount can vary by a factor of 2, with broadleaved species being higher than conifers. Carbon in the branches and foliage can contribute 30-70% of the above ground biomass, but roots may be 20-35% of the total carbon stock, with more in broadleaved trees than conifers (Morison *et al*., 2012). Of the whole wooded environment though, around 72% of the woodland carbon lies in the soils, 17% in the leaves and wood, 1% in dead wood and 6% in the roots (although this varies with species and management). In contrast, there is little carbon in herbaceous vegetation, although this will vary with its composition. Although that in arable crops varies it will be lost annually with harvesting.

Table 4

Carbon stock in vegetation.

Habitat	Carbon in vegetation (tC/ha)
Broadleaved mixed wood	81-251 depending on age
Conifer plantation	59-94, average 75, depends on age, density, yield and climate
National average all wood types	62-66
Hedges untrimmed for 3 years	45.8
Saltmarsh	8.32
Heath lowland and upland	2-9
Floodplain grasslands	Not available
Peatland	c. 2
Seagrass	0.5-2.52
Acid grassland	1
Neutral grassland	1
Agriculturally improved grassland	1
Arable	1-2.36

Source Alonso et al., 2012, Gregg et al., 2021, and see text

2.4 National total carbon stocks

The total carbon stock on a UK scale relates to the area of each habitat, thus pasture carbon, for example, may be disproportionately greater than woodlands, merely because of the much larger area involved. This is demonstrated by Field *et al*. (2020) for High Value Conservation habitats (i.e. not including agriculturally improved or arable land) taken from Land Cover mapping, but which does not sample the full soil profile (**Table 5**). This store represents 30% of the UK terrestrial carbon but on 20% of the land area. If the full soil and peat depth were taken, the totals would be much higher and the relative importance of each could change. Owing to topography, climate and scale of semi-natural habitats, a larger proportion of this stored carbon is in Scotland and least in Northern Ireland and Wales. The largest store shown lies in heathland that is also extensive, especially in Scotland. Note not all habitats are listed and that carbon will also be lost from some. It is the soils that will be most important.

Woodland with multiple aged trees and shrubs

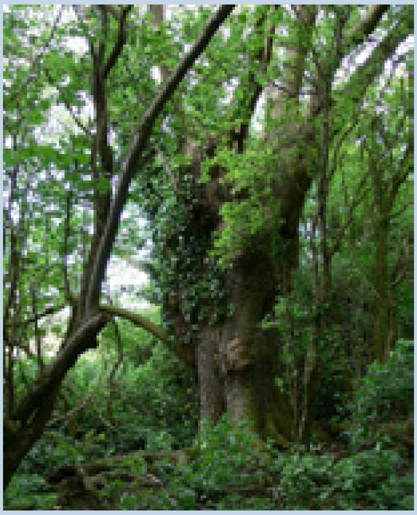


Table 5

Total area and carbon store in vegetation and soils to 30 cm in High Value Habitats in the UK.

High Value Habitat in UK	Area (thousands ha)	Carbon store Gt
Bog (blanket and raised)	957	0.1
Fen	17	0.002
Heath	2,441	0.24
Littoral mud	164	0.02
Saltmarsh	80	0.01
Semi-natural grassland	941	0.09
Woodland	422	0.01
Total	5,022	**0.55**

Source: Field *et al*. 2020. (Gt= Gigatonne)

2.5 Carbon losses

Damaged and disturbed soils lose carbon. Drainage in particular will dry out wet soils and increase decay rates which will result in loss of stored carbon. For example, Tuohy *et al*. (2021) quote levels of 3.14 tC ha^{-1} yr^{-1} lost from carbon-rich organic or humic soils where subject to drainage. The highest losses occur where drainage is large scale and severe as in peatlands that are drained and used for arable agriculture (e.g. Cambridgeshire Fens and Lancashire Mosses). Updated figures from Evans *et al*. (2023), for example, reveal up to 37 tCO$_2$e ha^{-1} yr^{-1} loss at the most deeply drained lowland arable sites, a figure that is elevated to 38.98 tCO$_2$e ha^{-1} yr^{-1} in the 2021 UK GHG Inventory (quoted by Gregg *et al.,* 2021). Intensively and extensively managed grassland on deep drained peat loses 22.00 and 15.88 tCO$_2$e ha^{-1} yr^{-1} respectively, similar to that from a drained, eroding, modified blanket bog, although there will be considerable variation between the types of degradation in different situations (Evans *et al*., 2023). Evans *et al*. (2016) noted that for every 10 cm lowering of water table, CO$_2$ emissions increased by around 4 tCO$_2$e ha^{-1} yr^{-1}.

Many of these figures are significantly higher than the sequestration rates of any habitat (**Table 7**) and account for the key role peatland restoration has as promoted by the CCC (2020) and supported by different governments.

Cultivating drained, peat soils in the Cambridgeshire Fens

Loss of carbon in peat translates into subsidence of 0.38 to 0.86 cm yr^{-1}, and a restricted lifetime for some peatland soils (exacerbated by losses to wind erosion particularly from arable soils). Considering that at least 80% of the UK's peatlands are damaged and degraded, then the loss of carbon is large scale and serious, contributing the equivalent of approximately half of the amount released through the UK's agricultural sector (Dunn *et al*., 2021) and 5-10% of global annual anthropogenic CO_2 emissions (Loisel & Gallego-Sala, 2022). On the other hand, lowland peat that is re-flooded after extraction (possibly with too high a water table or managed to attract bird life) might be losing >10 $tCO_2e\ ha^{-1}\ yr^{-1}$ at the most waterlogged sites owing to methane emissions (Evans *et al*., 2016).

On mineral-based soils, ploughing, drainage, habitat destruction or removal and redistribution of soils as part of developments results in carbon loss (total or partial), as SOM is physically broken down and decays. When on a large scale, as for example when converting long-established grassland to arable or reseeding grassland or for large-scale development projects, carbon losses can be severe and rapid. The annual cycle on arable soils when ploughed results in release of more carbon than is stored in the order of 0.14 $tCO_2e\ ha^{-1}\ yr^{-1}$(0.51 $tCO_2e\ ha^{-1}\ yr^{-1}$), thus contributing to increased CO_2 levels in the atmosphere (excluding the additional carbon lost during machine use). Moreover, the intensification of agriculture has resulted in an average loss of its carbon to the tune of 0.6%/yr since the 1940s, (https://soilguide.co.uk/members/page16.html). It follows that other disturbance or damage can also result in loss of carbon from a habitat. Clearance of scrub and trees as part of management will all have negative impacts, some measures of which are provided in **Table 6**. Scrub invasion of grassland, on the other hand, can be positive or negative, with gains of up to 13.1 tC ha^{-1} to losses of 56.1 tC ha^{-1} from dry or wet sites respectively (Gregg *et al*., 2021).

Table 6.

Examples of carbon loss after habitat damage or change

Change or damage to habitats	Carbon loss $tCO_2e\ ha^{-1}\ yr^{-1}$
Grassland to arable	3.58-6.23
Upland heath to improved grass	3.3-4.03
Scrub removed from lowland heath	2.56 (average over 5 years)
Restoration lowland heath – burning, grazing, scrub clearance	4.46 (average over 5 years)
Lowland raised bog partly cut for peat, heather dominant	2.18-2.60
Gullied blanket peat	12.17
Flooded lowland peat after extraction	>10
Lowland raised bog converted to arable and deeply drained e.g. Cambridgeshire fens	38.98

Sources –Alonso *et al*, 2012, Gregg *et al*., 2021 & see text

Overall carbon losses are significant nationally and cumulative across activities related to habitat removal or degradation. Habitats are lost to development or to landuse changes as from agricultural conversion to arable or improved grassland. Overgrazing (which reduces litter and root production); peat extraction, gullying, drainage, decay and erosion; soil erosion; habitat removal; ploughing or other soil disturbances all contribute to soil carbon losses. Janssens *et al*. (2005) calculated a loss of 6.3 gC m^2 (0.06 tC ha^{-1} yr^{-1}) in total across grasslands, forests, cropland and peatland in the UK in the 1990s, with the negative effects of peatland and cropland losses counterbalancing the positive gains in the other habitats. In a comparison across European countries, those with more forest and less peat or agricultural land were sequestrating more carbon than they were losing, whilst countries like Denmark, Estonia, Portugal and Poland were losing higher levels than the UK (Janssens *et al*., 2005).

Drain installed 2015 in wet pasture

3

Chapter 3

Carbon sequestration - where best to focus

3.1 Introduction

As carbon loss needs to be avoided and mature habitats can contain large quantities, the focus must be on safeguarding and maintaining existing sites (Gregg *et al.,* 2021). Where these habitats are not in the best condition for biodiversity or carbon sequestration, restoration and enhancement are the first priority. Field *et al.* (2020), for example, estimate a near doubling of the potential carbon sequestration just in existing peatland and heathland habitats if they were in good condition, equivalent to nearly 32% of the annual emissions from agriculture. **Table 7** provides some comparative carbon sequestration rates for existing habitats, although there is considerable variation in the literature based on different soils, climates, vegetation, age of habitat and the methods deployed.

Good condition raised bog,
Clara Bog Ireland

Table 7.

Some estimates of the range of carbon sequestration in different habitats.

Habitat: soils and vegetation	Carbon exchange, tCO$_2$ ha^{-1} yr^{-1}	tC captured ha^{-1} yr^{-1}	Comments/sources
Reed bed	18.34-73.34	5-20*	On worked out peat Brown 2009
Alder carr	18.34-36.67	5-10*	Possibly on worked out peat Brown 2009
Ancient/old growth woodland	4.77-17.97	1.3-4.9*	Thomas *et al.* 2011 Britain & Europe
Broad-leaved wood	9.17, 2-13* 100 yr old mixed broadleaved	2.5	Cannell 1999 2-7 tC ha^{-1} yr^{-1} average across rotation, Gregg *et al.* 2021
Conifer plantation	11.01-22.0	3-5.6*	Dewar & Cannell, 1992. Sitka spruce, averaged over cycle, see Table 9
Saltmarsh	2.35-23.83	0.64-6.5*	Burrows *et al.* 2014, Beaumont *et al.* 2014, average 1.2-1.5 tC ha^{-1} yr^{-1}
Heathland	3.34-12.65	0.91-3.45*	Alonso *et al.* 2012, Quin *et al.* 2015
Floodplain	1.83-10.63	0.5-2.9*	Walling *et al.* 2006, Sutfin *et al.* 2016
Estuaries in intertidal and subtidal mud	0.59-2.35	0.16-0.64*	Alonso *et al.* 2012, Hutchings *et al.* 2020
Sand dunes	2.13-2.68	0.58-0.73*	Dry – wet slacks, Beaumont *et al.* 2014
Peatland in good condition	+0.32 to -3.7*	+0.09 to -1.01	Artz *et al.* 2013, Evans *et al.*, 2023

* measures given in research papers. Figures are converted to tC ha^{-1} yr^{-1} or to tCO$_2$ ha^{-1} yr^{-1} for comparison, but may not include all GHG. NB some peat habitats in good condition can be net emitters of CO$_2$

However, habitat restoration alone is insufficient to reduce CO$_2$ levels to a safe level nor reverse the biodiversity crisis. Thus, new habitats are also essential on a large, interconnected scale. Rebuilding nature on the scale envisaged in the Nature Recovery Areas and beyond is not only critical for biodiversity but also for carbon sequestration.

Current recommendations are for peatland restoration and afforestation along with urban greening to help absorb more atmospheric carbon (CCC, 2020). But are these the best or only worthwhile approach? Cotrufo *et al*. (2019) stress how implementing soil carbon sequestration strategies needs clear science-based guidelines that include consideration of site-specific soil and ecosystem properties, including the relative distribution of SOM into its POM and MAOM components, and an understanding of saturation points in different systems. Additionally, Brown (2020) urges more systematic monitoring, collation and interpretation of data from diverse land uses, soils, climate zones and management regimes, particularly because land use change can produce outcomes differing from initial assumptions.

Moreover, the urgency of the need to reduce emissions of GHGs dictates sequestration projects with the most immediate, largest and long-lasting effects. All these requirements in one ecosystem are difficult to achieve, thus multiple projects with habitat mixtures designed and located to support biodiversity restoration and other ecosystem services simultaneously will provide more benefits overall if achieved on a large enough scale. This immediacy is actually compromised by focusing on woodland planting since these are mostly not instant carbon capture machines and generally take at least 10 to 30 or more years to become a significant positive carbon sink. Thus, although they are regarded as potentially the largest carbon sinks in the UK and can provide good opportunities for removing atmospheric carbon over time (depending on the conditions and management – as explained in the woodland section below), this is not realised for some decades while canopies, root mats, woodland microbial and fungal functioning establish, and compensation for the sometimes high carbon establishment costs are achieved (Gregg *et al.,* 2021). Additionally, woodland largely precludes agriculture and is small scale compared with the magnitude of new habitats needed for both carbon sequestration and biodiversity rebuilding. It is essential therefore to consider soils, ecological landscapes, climate and timescales, and to select those habitats or mix of habitats that could make the greatest difference for both climate change and biodiversity rebuilding on all timescales. **Table 8** provides some indicative potential sequestration rates for restored or new habitats which are embellished in the separate habitat sections below.

Table 8.

Indicative carbon sequestration rates for new ecosystems.

Habitat: soils and vegetation	Carbon exchange, $tCO_2\ ha^{-1}\ yr^{-1}$	tC captured $ha^{-1}\ yr^{-1}$	Comments/sources
Restoring actively eroding bog to modified bog	21.3*	NA	Gregg *et al*. 2021. The total is mostly preventing further losses, not sequestration
Restore acid grassland to heather heathland	12.65	3.45*	Quin *et al.* 2014 heather in building phase
Adding red clover to semi-improved grassland	11.62	3.17*	De Deyn *et al.* 2011, no fertilisers + additional diversification
Arable to wetland	8.07-16.87*	2.2-4.6	Alonso *et al*. 2012
Natural woodland generation on former arable soils	7.33-14.3	2-3.9*	Poulton *et al*. 2003 average over c.120 yrs
Constructed wetlands	8.03-9.79	2.19-2.67*	Mitsch *et al.* 2013
Create wood pasture from pasture	4.8-5.7*	1.3-1.55	Alonso *et al.* 2012, 10% tree cover
Hedge restoration / adding trees	3.67-5.87	1-1.6*	Gregg *et al*. 2021, increased biomass, more trees
Small ponds, well vegetated	0.92-77.8	0.25-21.22*	Taylor *et al*. 2019, Gilbert *et al*. 2014, Anderson *et al.* 2013, 2020, Downing *et al*. 2008

Creation of reedbed from arable or grass	4.00*	1.09	Alonso *et al*. 2012, higher if from arable
Restore flower-rich grassland from improved sward or arable	3.8-6.96*	1.04-1.89	Alonso *et al*. 2012, (but includes whole C cycle),
Creation of intertidal/ saline habitat from arable or grassland	3.8/2.9*	1.03/0.79	Alonso *et al*. 2012
Arable to heathland, 1-100 yrs	3.32-7.3*	0.91-1.99	Alonso *et al*. 2012, (but 3.32 = a whole C cycle estimate), higher where wet heath restored, Gregg *et al*. 2021
Grassland to wetland	2.39-14.30*	0.65-3.9	Alonso *et al*. 2012
Restoring modified bog to near natural bog	1.46*	0.4	Gregg *et al.* 2021
Create fen from arable or grass	0.9*	0.25	Alonso *et al*. 2012
Restore peatlands	0.8-6.93*	0.24-1.89	Svenson, *et al*. 2019, lowland raised bog, Artz *et al.* 2013 blanket bog

* measures given in research paper/s. and converted to either tC ha^{-1} yr^{-1} or tCO_2 ha^{-1} yr^{-1} for comparison, but may not include all GHG. NA – not appropriate to convert without details on methane.

Critical to the discussion is the idea that carbon sequestration reaches a likely near equilibrium state whereby the amount absorbed becomes equal to that emitted from an ecosystem (Gregg *et al*., 2021). Angst *et al*. (2023) challenge this concept and promote a more systems approach to carbon capture in soils, suggesting that the carbon-saturation concept is based on the assumption that the amount of silt and clay-sized minerals determine the overall capacity of the carbon store which focuses on MAOM as the major soil pool. They posit that while MAOM does tend to saturate, the overall capacity of the soil to store more carbon is not reached at that point. Formation of MAOM, they suggest, may be less efficient, but will not be zero and that POM inputs can continue to accumulate as labile or stabilised material provided conditions are suitable and there are no significant perturbations. Total carbon capture rates might be much reduced but still positive in these situations.

Provided a site is properly functioning under a suitable climate, carbon can accumulate over time in organic soils for much longer than in some mineral soils. The same principle applies where sediments rich in carbon, as in marine environments like mudflats, seagrass beds and saltmarshes can continue trapping sediment. Functional floodplains, where repeated flooding brings in more sediment, could also accumulate carbon for centuries. In contrast, models for carbon in planted forest are predicated on the system reaching a near equilibrium point whereby sequestration equals absorption (Cannell & Milne, 1995, Dewar & Cannell, 1992). There is some evidence for this from field measurements for conifer plantations, but it is challenged for old growth and ancient woodland by Xiong *et al.* (2020), who found carbon accumulation persisted in a broadleaved forest unmanaged for over 400 years, whilst it ceased in an adjacent old conifer plantation.

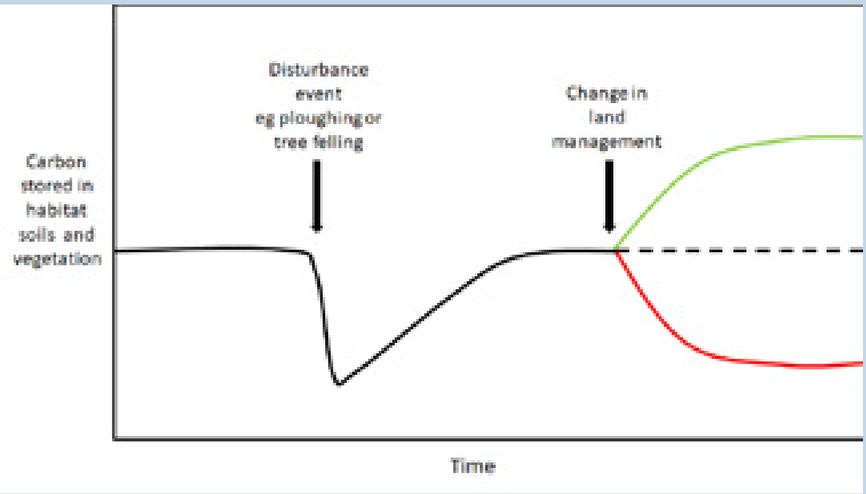
Kielder Forest

Gregg *et al.* (2021) present a conceptual model of habitat carbon stock equilibrium disturbed by land-use change (**Figure 3**) and emphasise the importance of knowing where an ecosystem is on its trajectory to any steady carbon state. This is dependent on its management history as well as other factors related to climate or disturbances events like wildfire. Some ecosystems can take many centuries to approach the assumed equilibrium, for example in temperature forests, but might be shorter for grasslands for example. This time scale is important as creating or restoring ecosystems to capture carbon on a large enough scale 'buys time' in which longer term solutions are devised and implemented to reduce GHG emissions (Cannell, 1999).

3

Figure

Conceptual model of habitat carbon stock equilibrium and land use change (Gregg et al., 2021)



Some advice on enhancing soil carbon is presented before the evidence for each major habitat type is explored in terms of potential carbon stock and sequestration rates over time to gauge the advantages or disadvantages of different approaches. It should be noted that there are considerable variations in the data related to sampling, habitat ranges, climates and soils, whether data are modelled or measured and the factors included in the measures given. Different methods of measurement also make comparisons more difficult and it is not always clear what has been measured, such as methane and nitrous oxides as GHGs. Some studies are only first steps in exploring a particular effect, so warrant further investigation, while others represent a single point rather than changes over time. Repeated measures in some studies at specific soil depths rather than accommodating changes in soil horizons (which can increase with more organic matter), might affect results and conclusions (Benham *et al*., 2012). Some variations in measures are to be expected based on differences in soils and climate as well as vegetation as outlined above, even within the UK. The figures shown, therefore, should be treated as indicative and part of a range.

3.2 Soils

Rather than focusing on increasing carbon stocks through the longer lived MAOM carbon, Angst *et al*. (2023) suggest more measured targets based on a systems approach that recognises soils as complex systems in which the POM and MAOM are intertwined parts dependent on the local environmental factors. The advice by Angst *et al*. (2023) is:

- Average carbon saturation deficits are roughly 50% in surface soils globally, so improved MAOM-carbon through management will add organic matter with low C:N and lignin:N ratios and which have abundant reactive minerals – i.e. soils rich in silts and clays. This would include optimising grazing regimes and intensities, multi-trophic rewilding and restoration of plant diversity, all of which can alter the quantity of rhizodeposits and quantity and quality of POM in grasslands, boosting formation of MAOM-carbon and POM as a precursor pool. In agricultural situations, total SOM can be increased through high-quality cover crops (legumes), cultivars with deeper root-derived inputs, more use of perennials, retention of crop residues, reduced tillage and increased organic amendments.

- Where soils may be already at their MAOM carbon saturation point, promoting plants with high quality litter or greater root exudation could diminish the POM pools through more complete decomposition of plant litter and mineralisation of plant-derived carbon. A focus here on more POM through higher amounts of structural and recalcitrant plant inputs would be a better approach, for example retaining more biomass residues (leaves and branches) in timber production sites or through increased litter inputs and no tillage to avoid damaging existing POM in agricultural soils.

- Where soils have a low MAOM capacity, increased POM could be an important pathway for carbon sequestration focusing on low-quality plant inputs, low soil nitrogen availability and pH. These conditions are typical of many coniferous and some broad-leaved forests. Plants with high C:N rations such as those with high tannin, wax and lignin contents can reduce microbial metabolization, further exacerbated by the low pH, while ectomycorrhizal fungi may also create conditions unfavourable for efficient bacterial conversion of plant litter. In these conditions, a thick POM layer can make a significant contribution to total SOM pools.
- Similarly, POM in heathlands is likely to persist due to high contents of lipids, aliphatic compounds and sterols originating mostly from heathland vegetation and the sustained high inputs from organic matter.
- In many forests, soil carbon stocks can be maximised through increasing tree diversity with both higher and lower litter quality and retention of harvest residues.
- In wetlands, reduced oxygen availability favours accumulation of POM, but can be lost through drying and oxidation, thus pointing towards the need to restore and rewet wetland habitats to increase POM carbon storage.

3.3 Wetlands

Ponds and lakes

Lough Corrib, Ireland

Lakes, reservoirs and ponds can capture CO_2 through plant photosynthesis and emit it through mineralisation[3] and de-gassing, they can bury carbon in sediments but also have the potential to emit methane produced in anaerobic sediments (Peacock *et al.,* 2021). The balance between these determines whether wetlands can be a net GHG store rather than emitter.

Wetlands can capture significant amounts of OC and accumulate it in bottom sediments where it is stored rather than subjected to decay and loss through respiration as occurs in terrestrial systems. Studies describing this depend though on measuring OC accumulation and not all take into consideration any offsetting losses of carbon in its formation nor of methane production, which can be high from wetland surfaces. Taylor *et al*. (2019) showed that small ponds are particularly effective at trapping OC, with levels as high as 247 gOC m^{-2} yr^{-1} (equivalent to 2.47 tC ha^{-1} yr^{-1}), but averaging 142 gOC m^{-2} yr^{-1} across 20-year-old small ponds on a restored coal mining site in Northumberland. Equivalent levels were measured in a series of small, natural ponds in the same county (Gilbert *et al.*, 2014), at 1.49 tC ha^{-1} yr^{-1}. Taylor *et al.* (2014) found OC started to accumulate rapidly after three years, so the average takes this early colonising period into consideration. Greater accumulation resulted from vigorous aquatic plant growth. Methane losses from the ponds were limited: 2.3g CO_2e m^{-2} yr^{-1}, which was equivalent to 1.7% of the mature pond's burial rate. Taylor *et al*. (2019) noted that other studies suggest a range of methane levels, with rates equivalent to between 0.7 and 19.7% of the burial rates found.

However, recent studies of a range of lake and pond sizes show high GHG concentrations in ponds rather than large lakes (quoted in Peacock *et al.,* 2021). This could be related to the greater interaction with the edge and often greater plant productivity, frequent water mixing and shallow depths. Baron *et al.,* (2021) note greater production of methane in shallow water bodies where ebullitive methane release (bubbling) was studied but found no clear relationship with other factors. Peacock *et al*. (2021) also found artificial ponds and ditches are higher GHG emitters than natural ones across a range of landscapes and climates, although this depends on the catchment habitat around natural ponds. The authors also confirmed, in a study of 96 ponds and 64 ditches, that nutrient concentrations (nitrogen and phosphorus) stimulated concentrations and fluxes of GHGs and at times the levels of emissions counteracted the carbon capture of their associated habitats. An interaction between temperature and phosphorus concentrations was also noted in ponds in Quebec (DelSontro *et al.,* 2016).

It is very important that ponds do not contribute to GHG emissions, although they can be excellent wildlife habitats. The key

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[4] Mineralisation is the process of decomposition of organic matter by micro-organisms

is to minimise eutrophication, for example by careful siting if in an agricultural setting, providing a marsh or other vegetation as a buffer and filter around the pond or fencing ponds from livestock, (Malerba *et al*., 2022), all of which also improve water quality and therefore wildlife value.

Ponds with high organic loads and low oxygen levels favour methane emissions and are likely to be net emitters (Peacock *et al.,* 2021). DelSontro *et al*. (2016) confirm this for American beaver ponds where methane bubbling was significant. Beaver activity affects SOC storage positively and negatively, resulting in its re-distribution with the development of new wetlands. More research is needed but global estimates of beaver ponds suggest a range that varies from being sink to a source of GHGs (Nummi *et al*., 2018). This could have significant implications for widespread beaver introduction in the UK.

The role of ponds to support carbon sequestration is dependent therefore on vegetation type, management, nutrient control and sediment carbon accumulation. Further research is needed, but clusters of ponds of different ages and other larger wetlands in mixtures of habitats would be ideal within a wider landscape, but with potential GHG emissions minimised.

Restoring old ponds risks losing the carbon contained in bottom silts. Creating new ponds is better but should not be so shallow as to dry out annually as this precipitates loss of the stored carbon (Gilbert *et al*., 2014). At the same time, new ponds should not be constructed in habitats that are carbon-rich that would be lost on disturbance. Ponds can be good for carbon capture - they are small but can be numerous within the landscape context, although groups of them can support more biodiversity, contribute to clean water and mitigate floods when streamside.

A well vegetated pond can capture more organic carbon

Turning to lakes, good rates of OC capture were found in lake sediments in Minnesota, with averages of 25-70 gOC m^{-2} yr^{-1} (0.25-0.7 tOC ha^{-1} yr^{-1}), (Anderson *et al.,* 2013), and these match average carbon entrapment in European lakes at 0.5 tC ha^{-1} yr^{-1} Double these levels are attained in eutrophic lakes. Gregg *et al*. (2021) separate out the English lakes from a later study by Anderson *et al*. (2020), which were all classified as eutrophic and noted burial rates of 0.82-2.97 tC ha^{-1} yr^{-1}, those with the highest burial rates being in the Norfolk Broads and Shropshire-Cheshire meres.

The nature of the surrounding landuse has the greatest impact, with higher levels in agricultural settings with greater nutrient runoff. Anderson *et al*. (2013) consider that lakes are long-term sinks for OC, especially compared with most terrestrial situations. Downing *et al*. (2008) confirmed these findings with much higher levels of OC in small ponds/ impoundments or lakes, mostly in agricultural contexts, compared with larger ones and in those with smaller catchments reflecting the level of sediment entrapment and eutrophication levels. Carbon was deemed to be derived from sediments running into the impoundments from the catchment and from carbon fixation in the water. Average rates as high as 21.22 tOC ha^{-1} yr^{-1} were found across a wide variety of lakes and impoundments in Iowa, but with an anomalously massive 173 tOC ha^{-1} yr^{-1} in one small farm pond (sizes not given).

These lake accumulation rates need to be balanced by the loss of methane in particular, although there is a general dearth of relevant UK research on this. Gregg *et al*. (2021) cite a study in North West England where methane emission levels of 0.7 t ha^{-1} yr^{-1} were shown. DelSontro *et al*. (2016) suggest ebullitive methane occurs more in water bodies under 3 m deep and can reach 18-23% of total methane emissions depending on the area of the lake that is in the shallow littoral zone, although bubble concentrations of methane vary, making measurement difficult. The authors also confirm the relationship between phosphorus and methane production, suggesting eutrophic lakes would be producing more methane, even though they are also accumulating OC, but warn that climate change could change these balances in favour of greater emissions.

One study comparing newly created with long established wetlands showed that newly created wetlands (3-8 years old) compared with long-established sites in Ohio (Hossler and Bouchard, 2010) revealed lower levels of plant biomass, SOC, mineralisable soil carbon and macroaggregates, plus higher bulk densities and silt-clay fractions in the new sites. SOC was correlated positively with macroaggregate quantity but negatively with higher microaggregate and silt-clay fractions. Carbon stocks were low in the new ponds: under 50 gC kg^{-1} in the new wetlands compared with 100-225 gC kg^{-1} in the mature wetlands. (The nature of the wetlands beyond being excavated hollows was not described). The authors estimated that it could take 300 years to match the SOC content of the mature wetlands (although 30 and 400 year time periods were quoted from the literature), assuming the trajectory was real. The new ponds mitigated losses from landuse changes and illustrate issues related to creating replacement habitats and biodiversity net gain calculations. The time needed to be functionally equivalent was principally derived from excavating new wetlands into subsoils with higher microaggregate fractions caused by compaction and subsoil exposure. Performance was much better in one new wetland where the former surface organic material was placed into the bottom. The authors modelled the replacement ratios required from 2.2 to 5.1:1 in order to mitigate fully for the losses. This emphasises the importance of protecting existing wetlands and demonstrates the potential timeline (there are significant differences in this depending on habitat details) to recapture lost carbon. There was no mention though of methane emissions in this study.

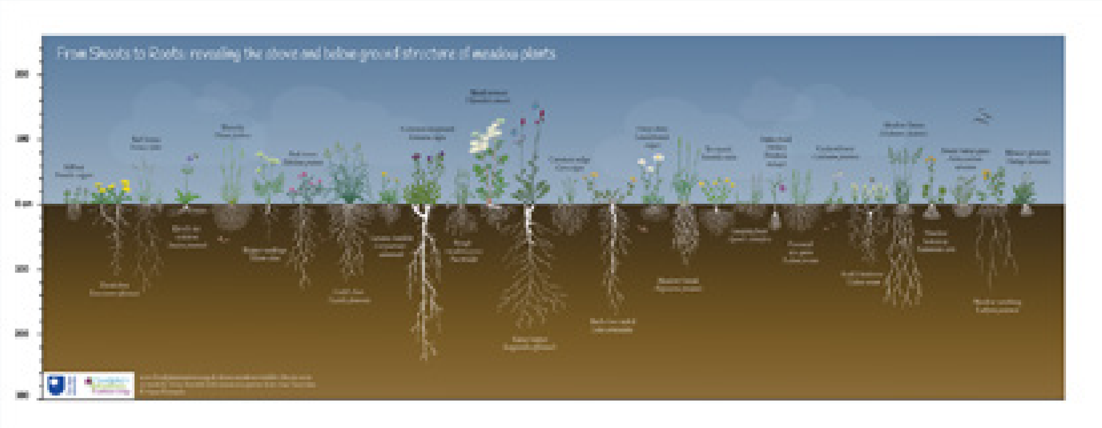
Floodplains

Floodplain carbon stocks come from both flooding events (trapped from sediments and prevented from passing down the river system to the sea) and from sequestration and at a much larger scale than for other wetlands. Analysis of several South and South West England rivers differing in their geology and landuse catchments, showed similar levels averaging around 0.92 tC ha^{-1} yr^{-1}, with some variation between rivers and within catchments, ranging between nearly 2.0 tC ha^{-1} yr^{-1} and less than 0.5 tC ha^{-1} yr^{-1} (Walling *et al*., 2006). Similar levels have been found in Austrian studies (Zehetner *et al*., 2009). There was no analysis of vegetation differences between sample sites in the English study, but the Floodplain Meadows Partnership suggests that a diverse meadow flora with variable rooting depths would sequestrate more than a monoculture rye-grass sward or arable field on flooded floodplain (unpublished data, 2020 newsletter, **Figure 4**). This would be consistent with evidence for the superior quality of diverse grasslands being able to sequester more carbon than monoculture grassland and the deeper rooting plants would pass exudates much further down the soil profile. The Floodplain Meadows Partnership analyses show very high levels of carbon stored, (including levels of 109.4 tC soil carbon stock in the top 10 cm under Cricklade NNR floodplain - quoted by Gregg *et al.* 2021) and significant carbon storage down to 1 m of restored floodplain grasslands; much higher than in other habitats.

4

Figure

The above and below ground structure of flood-plain meadow plants (https://floodplainmeadows.org.uk/discover/learn/biodiversity/shoots-to-roots, Bowskill and Tatarenko, 2021).



This experience is mirrored in America where carbon stocks were present to great depths in buried horizons (to 3 m) produced by regular flooding and larger than in forests, pasture or wetlands at 286 tC ha^{-1} (D'Elia *et al*., 2017), but Cierjacks *et al.* (2010) found much higher levels of 354 tC ha^{-1} in just the upper 1 m on an Austrian floodplain. There can be, however, a high variability in the extent and depth of buried horizons reflecting equal variation in flooding events (D'Elia *et al*., 2017) with carbon trapping both long-term and rapid in its accumulation depending on flooding magnitudes and regularity, but also on the floodplain habitat type, which can be very varied. This variability is reflected in the measurements available – 2.9 tC ha^{-1} yr^{-1}along the Danube (Sutfin *et al*, 2016) to 0.5 to 2 tC ha^{-1} yr^{-1} in South/South West England (Walling *et al*., 2006).

These data provide a sound basis for floodplain restoration, supported by investigations into montane meadow restoration in the Sierra Nevada (California), where restoration of their flood-plain function, (blocking of erosion gullies plus revegetation) resulted in 2.33 tC ha^{-1} yr^{-1} accumulating over a 22 year chronosequence of sample sites. This was tightly coupled with increases in nitrogen concentration, although the soil C:N ratio did not change over time, and followed a significant increase in biomass, particularly below ground (Reed *et al.,* 2022).

Thus, floodplains that accommodate regular sediment input will accumulate significantly more carbon than those that are divorced from their rivers and can continue to do so as it is effectively trapped in lower layers, whilst those that are also rich in plant species with a wide range of rooting depths would also be sequestrating more carbon than other swards. Zehetner *et al*. (2009) suggest a timeline of some 100 years of accumulation rates of 1 tC ha^{-1} yr^{-1}, reaching levels exceeding those in forests, followed by a levelling off over 300-500 years subsequently to 0.08-0.18 tC ha^{-1} yr^{-1} for sampled sites in Austria. Regular flooding serves to reset soil formation continually back to an early phase, thus rejuvenating the carbon capturing cycle but having buried the previous store, although much could be lost in sediment in the floods themselves. This research also showed that cultivation can annihilate this carbon sequestration potential.

Beechener *et al*. (2021) note that floodplains are the UK's most widespread freshwater system, but 42% are separated from their rivers and around 65% has been extensively altered for agriculture. Typical floodplain habitats like fens, marshes and bogs have been reduced to only 0.5% of the English floodplain area, giving very significant restoration opportunities with concomitant ecosystem service benefits, particularly for downstream flooding and biodiversity. Where flooding overtops river-side flood-defence bunds, the risk of more emissions from stagnant trapped water is greater, as Gregg *et al.* (2021) notes, warning of possible increased fluxes of methane and nitrous oxides in such situations.

Rivers

There is little research on rivers and carbon, but riparian systems with the most structural diversity (multiple channels, backwaters, channel complexity, sinuosity, variability in substrates including logjams and fallen wood) and lined with trees will maintain more biological hotspots that facilitate breakdown of OM and filter excess nutrients and Dissolved Organic Carbon (DOC) from surface and subsurface waters (Sutfin *et al*., 2016). The carbon store lies in the riparian biomass, wood held in the water, sediment including OM, litter and humus on or beneath the channel and instream biomass; the latter accounting for relatively little of the portion of carbon stored in the other elements of river systems listed (the store is around 0.2-4.8 tC ha^{-1}). The amount in fallen wood can be significant, particularly in small rivers in damp temperate regions where wood decay rates are slow (10-100 years depending on species compared with <10 years in the tropics for example). Optimal conditions for OC retention are low gradient river systems in broad unconfined valleys with high levels of channel complexity and plenty of dead wood.

Stream with woody debris, Northumberland

Owing to the very diverse geologies and other catchment characteristics, rivers and streams in the UK are highly diverse, thus displaying high regional variability in their carbon and GHG fluxes. Some can also be sources of methane where there is dense vegetation as in some chalk streams or settling nutrient-rich sediment (Gregg *et al.,* 2021). Many are conduits for redepositing eroded soils and their carbon down to their estuaries. Indeed, loss of CO_2 from streams could be the main flux, at levels averaging 0.01-0.03 tC ha^{-1} yr^{-1} (summarised in Gregg *et al.,* 2021).

Consideration of beaver introductions is relevant to the river habitat. The high methane generation in small ponds is described above, and if waters entering a beaver dam are nutrient and sediment-rich, the potential for methane loss is high. Nummi *et al*. (2018), for example, show that beaver ponds can decrease or increase carbon storage, often resulting in redistribution owing to the regular disturbance. The carbon can be released into the atmosphere, transported as DOC downstream, or settle into bottom sediments. CO_2 emissions in a sample of ponds were found to vary from 0.14 to 11.2 gCO_2 m^{-2} day^{-1}, with an average of 4.9, whilst methane emissions varied from 27 to 919 mg m^{-2} day^{-1}, averaging 222. Ecke *et al.* (2017), in a meta-analysis of beaver effects across Europe and North America, also found higher methane levels associated with beaver impoundments. Beaver introduction programmes will need to consider the effect of negative changes in the carbon cycle as part of their environmental impact assessment.

Other wetland habitats

Reedbeds are known carbon sinks (Gregg *et al*., 2021), although ditches within the system can release methane (see Ponds and Lakes above). There is little evidence available on carbon sequestration rates, although Gregg *et al*., (2021) quote a Somerset County Council report of between 5 and 20 tC ha^{-1} yr^{-1} being possible in the Somerset Levels, but the details were unavailable. Common reed (*Phragmites australis*) is a cosmopolitan species and research in Australia suggests wetter reedbeds, flooded annually, can store 167 tC ha^{-1} to one metre, slightly more than in drier, less frequently flooded reedbeds (116-138 tC ha^{-1}) where drying was considered to be responsible for carbon loss (Whitaker *et al.,* 2015). The regularly flooded site with dominant common reed sequestered only 0.05 tC ha^{-1} yr^{-1} but within a semi-arid environment. Whitaker *et al.* (2015) noted higher levels quoted for a Danish reedbed of 5.04 tC ha^{-1} yr^{-1}, but methane releases (mostly through internal transport through the plants) reduce this potential significantly. Up to 15% of the net C fixed in a reedbed wetland can be released as methane (Brix *et al*., 2001), which can balance out owing to methane's short life in the atmosphere, leaving the wetland as a carbon sink in the long-term.

Other wetlands like fens are covered in 3.3 Peatlands below. There is little information on marshes. Constructed wetlands are included in the Urban Habitats Section (3.8).

3.4 Peatlands

Peatland restoration stands out as the most effective means of stopping the loss of carbon (much of which has accumulated since the Atlantic Period, 7,500 years ago) in DOC, POC or direct to the atmosphere, but can result in new carbon capture as well. The approaches have similarities on lowland, blanket and other mire ecosystems, although there are differences in detail and results. Peatland restoration's importance relates to both its spatial extent and depth and to the high current rates of loss. It is especially important on modified bogs as they are extensive, and the principle benefits are from stopping the very high carbon loss from drained agricultural peats (**Table 6**). New types of paludiculture are being investigated, seeking productive ways of re-wetting agricultural peats whilst producing commercially useful crops, such as in the Water Works project in the Great Fen (Fitton *et al*., 2021). Beechener *et al*. (2021) suggest a reduction in carbon loss could be derived from intermediate water tables and conversion to semi-natural fen meadows or similar as a compromise approach where it is impossible to restore the hydrological conditions needed for peatland generation.

Restoring blanket and raised mire peatlands in general is dependent on controlling or reversing the factors responsible for peat loss. These could be drainage, bared peat from wildfires, gullies and erosion, possibly exacerbated by overgrazing or, in the past, air pollution. There is extensive experience in many different projects now on the best techniques and measures (see for example Moors for the Future and Yorkshire Peat Partnership websites). However, vegetation also plays a role. Cottongrass (*Eriophorum*) dominance is correlated with an increased methane flux, especially in wetter ground, whilst a high *Sphagnum* coverage can show lower levels owing to a symbiotic relationship with methanotrophic bacteria (Larmola *et al.,* 2010). A high water-table produces anaerobic conditions which suppresses decomposition of OM and can encourage more *Sphagnum* cover.

Gully blocking on blanket bog raising the water table

Net CO_2 sequestration rates of peatlands vary from 0.2-0.71 $tCO_2e\ ha^{-1}\ yr^{-1}$ on Moor House (modified blanket peat), to 3.7 $tCO_2e\ ha^{-1}\ yr^{-1}$ on a relatively undamaged raised bog at Auchencorth Moss in Scotland. Artz *et al.* (2013) calculated a net sequestration rate of 0.76 $tCO_2e\ ha^{-1}\ yr^{-1}$ on ombrotrophic peatland inclusive of methane and carbon fluxes. Gregg *et al*. (2021) give overall sequestration rates of 0.02 $tCO_2e\ ha^{-1}\ yr^{-1}$ for rewetted, semi-natural modified bog and for near natural bog in some updated figures which take losses of DOC and methane into account. Measurements show that carbon capture is not necessarily constant and can represent a loss in particularly dry years. However, at Glencar in relatively good condition, measurements showed carbon sequestration in 4 out of 6 years (Artz *et al*., 2013).

Only communities with abundant *Sphagnum* were capturing net new carbon on restored lowland raised mires (that had been previously been drained and partly cut over) in Ireland at 0.32 or 0.78 $tC\ ha^{-1}\ yr^{-1}$ on different sites, both within a short 5-7 year period following restoration (and taking methane emissions into account). Heather (*Calluna vulgaris*) and marginal drier areas were still a carbon source (2.39 or 1.57 $tC\ ha^{-1}\ yr^{-1}$) owing to lower water tables despite restoration measures being implemented on the bog as a whole (Svenson *et al*., 2019, Wilson *et al*., 2022). These levels and conditions are within the range found elsewhere on restored raised mires (Wilson *et al*., 2022). Rewetting strongly inhibited ecosystem respiration, which, combined with increased plant productivity, switched the restored area from a major carbon source to a sink within year one, which increased further in years 3 to 5 (with no droughts occurring that could reduce the raised water table effects), (Wilson *et al.,* 2022).

Conversion from a carbon source to a sink is dependent on the water table being about 0.1 m below the surface, with a low slope and no marginal drainage. Achieving this on severely degraded bog consistently is challenging and ensuring carbon is not lost and adding to atmospheric levels might be the key achievement in places, with only patches sequestrating carbon again. The degree and time scale of restoration depends on the condition of the peatland prior to restoration. Evans *et al*. (2023) consider that an effective water table is the overriding factor in determining CO_2 emissions and recommend depths <20 and not > 12 to obtain the best results. These depths will determine the vegetation type with more *Sphagnum* and less heather where water tables are high.

Coir rolls trapping bare peat allowing cottongrasses to re-establish, Peak District

All restoration targeted at reducing carbon loss from damaged peatlands will result in carbon savings provided methane generation does not exceed the carbon-saved benefits, but the time to achieve this could be two years to more than a decade if the peat is severely damaged. Carbon gains could be expected once the peat is functional again, which could take much longer for the whole site or develop in patches over time as water tables are raised and stabilised and erosion channels blocked. Gregg *et al.* (2021) give more details on net GHG gains as adopted by the different condition categories in the Peatland Code, with gains (accounting for methane and other carbon pathways) varying from 1.46 $tCO_2e\ ha^{-1}\ yr^{-1}$ for restoring modified to near natural bog, to 21.30 $tCO_2e\ ha^{-1}\ yr^{-1}$ for restoring actively eroding bog to a modified condition. The figures given do not separate raised bogs from blanket bogs and depend largely on non-British sites, even though there are significant differences in their bulk density, hydrology and management. This warrants further research.

Despite several decades of peatland restoration and many more of peatland research, the ecology of the functioning and processes in peatlands are complex and many aspects remain little known, all of which are affected by different restoration methods. Methane emission is a critical example for climate change adaption and mitigation. The active role of plants in reducing methane emissions is not fully known. Stępniewska *et al*. (2018), for example, have demonstrated fluxes of methane from different plant species (*Sphagnum*, common sedge *Carex nigra,* cranberry *Vaccinium oxycoccus* and hare's-tail cottongrass *Eriophorum vaginatum*) in a peatland in Poland, fluctuate with seasons, condition of the plants and plant composition with different methanotrophs associated with different plants in varying quantities. Moreover, Larmola *et al*. (2010) showed that the water table is the key factor regulating methanotrophy in *Sphagnum*, with more in wetter conditions, implying a loose symbiosis between *Sphagnum* species and methanotrophic bacteria that accounts for potentially 10-30% of *Sphagnum* carbon. Such variation over a mire surface with season, vegetation and water table, let alone geographical differences, all complicate measurements related to restoration. These examples demonstrate the complexity of a natural system and therefore the inevitable smoothing and averaging of outcomes used to measure success; measures that are then used in policy and codes as exemplified below.

The Peatland Code

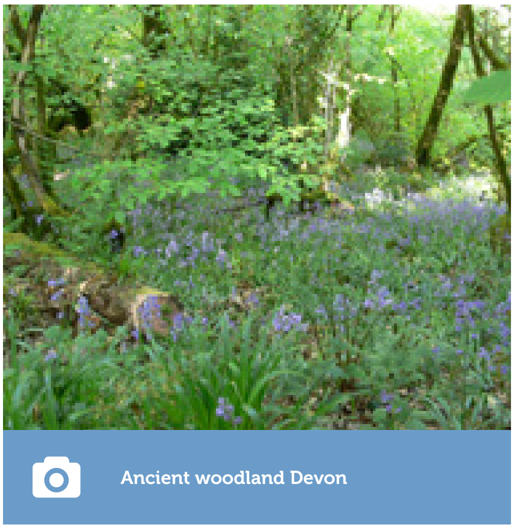
A Peatland Code to sell Carbon credits has been developed by the IUCN UK Peatland programme (https://www.iucn-uk-peatlandprogramme.org/peatland-code-0). This was one of the first examples of Natural Capital financing to provide assurance and clarity for business and other investors. This code is tied to the condition of the peatland prior to restoration and the expected reduction in emissions more than future sequestration after a period of years. The code sets out a series of best practice requirements with subsequent independent validation ensuring that the benefits will be regularly measured and monitored over a minimum of a 30-year lifetime of any project.

Fen peats

Fen habitats on peat are widespread but limited in extent and are less well researched in terms of carbon stocks or sequestration levels, (Gregg *et al.* 2021). Evans *et al*. (2023) consider the water table to be the main control on CO_2 emissions and Gregg *et al*., (2021) collate a range of figures on carbon fluxes in different fen habitats ranging from carbon sequestration at 10.31 $tCO_2e\ ha^{-1}\ yr^{-1}$ in the Norfolk Broads to a loss of 4.88 $tCO_2e\ ha^{-1}\ yr^{-1}$ ha on a fen converted to grassland from arable use. Conservation-managed fens are shown to be amongst the most effective carbon sinks. Methane was not detected on sites where the water table was below 25cm (Gregg *et al*. 2021). Evans *et al*. (2023) give emission totals of -0.36 $tCO_2e\ ha^{-1}\ yr^{-1}$ for undrained near-natural fens and 3.31 $tCO_2e\ ha^{-1}\ yr^{-1}$ for rewetted fen where methane loss as a proportion of emissions is greater.

3.5 Woody habitats

Key issues



Ancient woodland Devon

Determining the best solutions for woodland and carbon is difficult. Most of the carbon modelling has focused on commercial plantations rather than semi-natural or old growth forest, so understoreys and ground cover are largely omitted in the calculations and models, which is critical in any ecological consideration, and measures are based on closely planted trees, either conifers or a small selection of hardwoods (Morison *et al.*, 2012). These measures are not representative of semi-natural woodlands with a range of tree ages as well as shrubs, ground flora layers and epiphytes.

Several studies (e.g. Luyssaert *et al*., 2008) show that new woodland mostly does not become carbon positive for at least 10 to 30 years after establishment, leading to delays in expected benefits. This means that capturing carbon more immediately in other habitats in the short term to help reduce atmospheric carbon now is critical at the same time as establishing more woodland. Whether woodland creation is the best option on any site also depends on many factors including tree establishment methods, tree type and density and management and the carbon budgets of any existing habitat and soils, which might be better than woodland, at least in the first few decades or more.

There is also some debate over whether there is a finite carbon carrying capacity in woodlands. Luyssaert *et al*. (2008) argue that as individual trees are lost in old growth forests, there is generally new recruitment and a second canopy layer waiting to take over as the older trees fail or retrench. Decomposition of the fallen wood can take decades, whilst the new growth is more rapid, thus maintaining or continuing to accrue carbon in the wood, albeit slowly, and possibly mostly in the POC (Angst *et al*., 2022). Luyssaert *et al*. (2008) estimated average annual carbon sequestration rates of 2.4+/-0.8 in temperate and boreal forests 200 plus years old that reduce to 0.46+/- 0.1 tC ha^{-1} yr^{-1} in old growth sites based on a review of 519 plot studies. Although these figures are challenged by Gundersen *et al*. (2021), who suggest 30% less carbon sequestration than this in 200-year-old forests, both author groups maintain that continued carbon sequestration is still plausible in older natural or semi-natural woodlands. This is not straightforward as it is also known that trees with sufficient nutrition allocate more carbon to aboveground plant parts, whilst those with low-nutrient availability convert less of their photosynthates to biomass, similarly reducing their carbon allocation, possibly particularly to root symbionts (Vicca *et al*., 2012). Soil fungi and microbes will have a big part to play in this. It is clear that there are outstanding research questions still to pursue regarding long-lived and little disturbed woodland and carbon sequestration sites under different climates, soils and conditions.

It can be concluded that provided change in these established and old woods is driven by small-scale and localised events over time, they should maintain or continue to add overall biomass for centuries. Xiong *et al.* (2020) showed that carbon continues to accumulate in the soils in particular in ancient forests undisturbed for at least 600 years. Therefore, there seem to be key differences between plantation forestry and semi-natural woodland, the latter being the superior habitat accumulating larger carbon stocks over time, but also continuing to sequestrate carbon long into the future.

Sequestration rates in more natural woodlands

The carbon sequestration rates of ancient or long-established broadleaved woodland in broadleaved old forests across temperate Europe and America vary mostly between about 1.3 and 4.9 tC ha^{-1} yr^{-1}, with ancient beech (*Fagus sylvatica*) forests at the high end of the range (collated in Thomas *et al*., 2011). There have been too few measurements from old growth or ancient woodland communities rather than individual trees to be clear whether there is variation in their continued ability to sequester more carbon between climate, soil or woodland types.

Measurements taken in Wytham Woods, Oxfordshire over two years show significant variation in overall respiration across the yearly cycle using eddy covariance (which provides measurement of CO_2 fluxes), with elevated soil respiration figures in one year corresponding to periods of higher temperatures without moisture deficits (Thomas *et al.,* 2011). Greater carbon capture occurs with lower respiration rates (up to 3.6 tC ha^{-1} yr^{-1}, which is a very favourable rate and, most importantly, over the long-term). These measures are within the range provided for plantation forests, although these will be for a shorter time period, thus demonstrating that semi-natural woodland can be more useful for carbon sequestration in the long-term, whilst simultaneously producing a superior wildlife habitat and potentially capturing and storing carbon for much longer.

Comparison with other studies in broad-leaved woodland with similarly little management but using the same eddy covariance methodology suggests rates of 3.8 tC ha^{-1} yr^{-1} at Alice Holt (but this broadleaved plot would have been managed previously), 4.9 tC ha^{-1} yr^{-1} in a 250-year-old beech-dominated deciduous wood in Germany and 2 tC ha^{-1} yr^{-1} in a 450 year old diverse temperate forest in China (Xiong *et al*., 2020). These figures also show that woodland and commercial plantation sequestration rates are comparable with some other habitats, not always exceeding them, as shown in **Table 7**.

Woodland establishment

The imperative must be to retain and restore existing semi-natural broadleaved woodlands for carbon and wildlife (Gregg *et al.,* 2021). However, we also need more woodland for carbon capture as well as biodiversity and other functions. But tree establishment that is poorly planned and executed can increase CO_2 emissions and also have long-term deleterious effects on biodiversity and landscapes (Di Sacco *et al*., 2021). Crane (2020) provides a useful review of the alternatives and opportunities, summarised as the right tree in the right place, which is echoed by others, especially the Woodland Trust (Bavin, 2021).

Numerous factors need to be considered, especially soil type, pre-existing vegetation, woodland design, tree type and density, future management and future timber use. The timespan over which carbon is stored in wood products is also important. All these factors will determine how rapidly the new site can develop a positive carbon capture spreadsheet and have to be balanced against the optimum solutions for biodiversity and/or other ecosystem services as well.

Avoiding tree planting on organic-rich soils

Deep drainage in blanket bog on peat for plantation forestry, Sperrins, N Ireland

As far as soils are concerned, new woodlands should avoid organic-rich soils, especially peat (Crane, 2020, Gregg *et al*., 2021, Stafford *et al*., 2021, Bavin, 2021) as more carbon is lost from these soils than the woodland can replace as a result of disturbance, drainage, aerobic peat decay and loss of DOC in runoff. Indeed, under the UK Forestry Standard, areas of peatland and other low productivity land are considered inappropriate for tree planting (Fletcher *et al*., 2021). The guideline, though, is for organo-mineral soils with less than 50 cm of OM, which has recently been reduced to a 30 cm threshold in England (Bavin, 2021).

The carbon effects of planting trees into organic rich soils is illustrated by Warner *et al.* (2021) in a comparison of afforested (average 20 years old) and unafforested plots in Scotland, who found topsoil carbon in the upper 10 cm was lower (188 tC ha^{-1}) in the former and much higher (298 tC/ha) in the untreated heathland. Again, a comparison of hairy birch (*Betula pubescens*) and Scots pine (*Pinus sylvestris*) stands after 12 and 39 years and nearby heather communities in Scotland showed, despite the increased woody biomass, a net loss of carbon in four of the birch stands and no net gain in the others, (Friggens *et al*., 2020). This was explained by the altered mycorrhizal communities and autotrophic carbon inputs which led to a positive priming of the SOM leading to loss of SOC. This matches Miles' (1981) conclusion that birch colonising heathland can change a mor to a mull humus in about 20 years owing to deeper rooted trees bringing more minerals to the surface and to more readily decaying leaf litter. The soil fauna changes too with increases for example in earthworms. These all speed-up decay and therefore the soil respiration rates. Replacing heathland with trees can thus result in a negative carbon outcome for at least decades. Planting on peat, which usually involved drainage and ploughing, is even more damaging to the carbon stock, which explains the drive to remove plantations and restore damaged peat in, for example, the flow country (Anderson, 2010).

Woodland establishment on mineral soils on ex arable land

Positive alternatives lie in selecting more suitable soils for woodland establishment. Bare arable fields or already degraded sites should be the best starting point since the carbon content here would be low at the outset. Additionally, natural colonisation would also logically be the best method of establishment as it minimises ground disturbance and can result in a more diverse woodland structure. Support for this comes from two naturally colonised former arable sites at Rothamsted Experimental Farm on silty clay-loam, fenced for up to 120 years, one acidic now dominated by pedunculate oak (*Quercus robur*) and one formerly limed and supporting an ash (*Fraxinus excelsior*) / sycamore (*Acer pseudoplatanus*) woodland. These show carbon net accumulation rates of 2 tC ha^{-1} yr^{-1} and 3.39 tC ha^{-1} yr^{-1}over 118 or 120 years respectively. Significantly, both sites were still rapidly accumulating carbon 120 years after reversion, although carbon storage in the first 25 years was relatively slow as soil nitrogen was limiting in the early stages of growth (Poulton *et al*., 2003). Crane (2020) quotes Russian studies of natural regeneration on former arable land that found increased SOC across all regions, with high initial rates of carbon accumulation, possibly as the land had been so depleted previously, but the rate declined over 20-50 years and seemed to reach a new equilibrium after 60-80 years.

A net gain in carbon from the outset on former arable land it not always guaranteed however. Newly established plantation forests can lose SOC in their early years, as Laganière *et al*. (2010) found for plantations developed on former arable soils under 10 years old which averaged 5.6% loss of carbon, gaining 6.1% in 10–30-year-old stands and 18.6% in plantations over 30 years old. Vesterdal *et al*. (2002) found no SOC increase in new woodland composed of Norway Spruce (*Picea abies*) and pedunculate oak on former arable soils within 30 years, possibly owing to the low litter production in the early years and the slow development of a root rhizosphere, which might be constrained in an arable soil (here a nutrient-rich and moist sandy loam over a calcareous till deposit). The same had been found under poplar, aspen and willow (*Populus* species, *Populus tremula* and *Salix* species) in Germany over nine years. However, the carbon in the woody biomass needs to be added to the SOC for a full comparison, although the importance of SOC as part of the total needs to be noted. In contrast, Ashwood *et al*. (2019) recorded SOC stocks in secondary woodland 50-110 years old established mostly on arable land on clay-rich soils that were equivalent to those in ancient woodland within the National Forest.

Woodland establishment on high clay-content soils is more likely to produce positive carbon capture benefits more quickly where the fine clay particles contribute to the formation of stable organo-mineral complexes that protects the carbon against decomposition (Bavin, 2021, Laganière *et al*., 2010). Fast-growing trees on fertile soils may result in rapid carbon accumulation but this is matched by a faster turnover of litter with less entering the soils, thus not benefiting the carbon equation. Relatively infertile, clay soils would therefore be the most useful location for new woodland.

Planting woodland into grassland

The site vegetation and its relationship with the soils prior to woodland establishment is also critical. Woodland establishment into permanent pasture or other well or long-established vegetation could result in significant losses of carbon and take 30 or more years to be a positive carbon repository, although this is not always the case. Beckert *et al*. (2016) compared 24-year-old plots of hybrid larch (*Larix* x *eurolepis*), Scots pine and sycamore planted as wood pasture or woodland, with adjacent permanent pasture in Aberdeenshire. The soils (sampled to 50 cm depth) were freely-drained humus-iron podzols and brown earths with low clay contents of 2-8%. The wood pasture plots were planted at 100, 200 and 400 stems/ha or the woodland with 2,500 trees/ha. After 24 years the pasture (rye-grass, *Lolium perenne*, dominated) held the lowest total carbon store, whilst the woodland held most, although the total soil carbon did not differ significantly between treatments and the pasture. Even though the biomass per tree was greater in the silvo-pasture treatment for all the species, there was still more carbon in the woodland on a per hectare basis.

New planting plus ground flora along a walled boundary to add habitat and carbon

A meta-analysis of data on changes in SOC concluded that it declines if land use changes from pasture to plantation (losing 10% on average), although the magnitude was affected by tree type and precipitation and was greater for conifers than broadleaved trees (Guo & Gifford, 2002). In areas where rainfall was low (<1200 mm yr^{-1}), the conversion had little effect, but was greater (minus 23% soil carbon) in high rainfall areas (>1500 mm yr^{-1}). At the same time, natural regeneration, although still reducing soil carbon, had no significant impact compared with tree planting, (with concomitant site preparation disturbances). The authors regard the overall conclusions as indicative owing to the limitations of the analysis. Several reasons might contribute to these findings. Jobbagy and Jackson (2000), in a review of SOC and soil depth, found that more occurs below 20 cm in grasslands in general than in woodlands, so litter inputs could increase the surface SOC under trees. Grass roots, being shorter-lived and more fibrous, contribute more to SOC, whilst the annual turnover of tree roots is smaller. Woody plants may be less effective than some perennial grasses in some environments at storing carbon in soil. Higher rainfall and a cooler climate is also thought to be associated with a larger SOC pool and greater leaching of carbon to the deeper profiles.

Planting in soils to suit woodland communities

Fletcher *et al*. (2021) have teased out the carbon sequestration potential for native Scottish woodland based on natural regeneration but producing a wide variety of woodland types and densities based on their soil preferences from a classic upland oak woodland with birch and bluebells (W11 in the NVC) to some of the scrub and scattered tree habitats. They used an average carbon sequestration of 0.84 tC ha^{-1} yr^{-1}over 100 years compared with the 1.35 tC ha^{-1} average developed for UK woodlands to allow for low canopy cover in some communities, and assumed it takes 100 years for a woodland to mature over this period. Modelling the area of each woodland or scrub type hypothetically available across Scotland, the highest carbon contribution would be derived from W11, followed by W18 (Scots pine with heather) and W18/W17 (sessile oak-birch woodland with bilberry) woodland types. This is a very useful approach, based on assumptions of canopy cover and tree density (which although very little in some of the open habitats, is potentially over a large area), which can guide the best output for woodland wildlife and carbon in Scotland, without relying on plantation forestry.

In this context, Warner *et al.* (2021) investigated carbon levels in some naturally colonising ~20 year old Scottish Caledonian forest plots and compared these with unforested and mature forest areas that were both grazed and ungrazed. There was no effect of grazing on carbon, but there was more nitrogen in the unforested plots. The mature forest topsoil carbon (only to 10 cm) held 313 tC ha^{-1} (which is high), compared with only 14 tC ha^{-1} in the re-forested plots (20 years old on average) and 298 tC ha^{-1} in the unforested area (also high). The low level in the newly wooded areas may be obscured by the shallow sampling but could also relate to mycorrhizal fungal changes. For example, Ward *et al.'s* (2021) study of American forests with an undershrub layer of ericoid species beneath a mixed canopy found that the cover of ericoid species enhanced the soil (only measured to 7 cm) carbon and nitrogen concentrations which was attributed to ERM fungi which can scavenge nitrogen more efficiently in this environment than can ECM fungi. The biomass and necromass of ERM fungi contain high levels of melanin and polyphenolic compounds (which can bind with organic nitrogen to create protein-tannin complexes), both of which slow decomposition and increase SOM accumulation. ECM tree dominance was associated with lower carbon and nitrogen concentrations in comparison. This suggests that open woodland with a productive ericoid understory could hold more carbon than closed plantations and also provides a mechanism for the loss of carbon in heathland soils after tree planting, irrespective of the level of disturbance. Whether the total carbon can be higher in such woodland compared to heathland needs exploration within the UK context.

Broadleaved woodland versus conifer plantations

Well structured wood, N Wales

Irrespective of the existing habitat prior to woodland creation, a woodland for wildlife should have a varied structure with understory and canopy trees, a representative ground flora and be varied in age and light patterns to provide the optimum opportunities for a range of other plants, animals and fungi. This is far from dense plantations. There is a place for the latter to provide timber, but they are very inferior for wildlife (Crane, 2020) and carbon. Generally, conifers produce more stem volume more quickly than broadleaf trees, but the latter have more branchwood and their wood is at a higher density (by a factor of nearly 2 typically, Morison *et al*., 2012). Broadleaves also produce more root material and at a greater depth and contribute more to deeper, stabilised carbon in the soil profile and for longer than do conifers (Bavin, 2021). There are also concerns that developing large areas of coniferous plantations under climate change mitigation could lead to a notable decrease in SOC and turn these forests into carbon sources instead of sinks over time as crops mature and are harvested (Crane, 2020, Jonard *et al*., 2016).

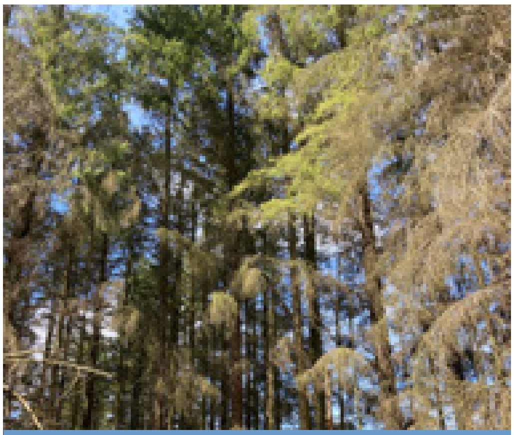
The advice from Crane (2020) and others is to plant fast-growing trees for high levels of timber and biomass if the objectives are to capture as much carbon as possible in a short time and to produce timber, but as long-term carbon stores are required over 100-year timescales, then establishing mixed native broadleaved woodland has greater carbon benefits and also supports more wildlife. Cannell (1999) supports this and advises that more carbon is sequestered in forest biomass by establishing amenity woodlands that are left to reach full maturity than in plantations grown for timber, although this equation depends on the how any plantation timber is used and survives in products. Bavin (2021) advocates a diversity of broadleaved trees rather than single species stands, species with deeper roots to contribute more to deep soil carbon stocks, those with higher chemical recalcitrant compounds like suberin and those with higher mycorrhizal associations. This links to Xiong *et al*.'s (2020) studies which found that the chemical character of the leaf litter was more important than its quantity for carbon sequestration. Trees like sessile oak (*Quercus petraea*) have been found to be associated with higher topsoil carbon than for example sycamore, but little is known about such differences and therefore which semi-natural woodland types are likely to hold more carbon over time (Bavin, 2021).

Managing new woodlands

Planted woodland, unless carried out progressively over a prolonged period, is uniform in age and, in order to match the old growth forests, will need to be diversified through management over time to achieve the age and structural variation, varied light conditions and natural regeneration to maintain carbon sequestration. Moreover, a woodland ground flora typical of long-established sites is slow to colonise or will be absent, so a diverse ground flora should be established as the woodland develops with varied rooting depths to contribute to carbon stocks. However, there is little evidence base for how best to achieve these objectives. This presents multiple opportunities for basic research into site, climate and soil variability in relation to how much carbon can be sequestered with different tree, shrub and ground flora communities.

Carbon in new woodlands

The carbon budgets for alternatives approaches are difficult to tease out and depend on many variables, not all of which are adequately measured, and many are based on models with a wide range of assumptions. This has been emphasized recently by Calders *et al.* (2022) who compared a new three-dimensional laser technique for measuring woody content of part of Wytham Woods (Oxfordshire) with the normal allometric models. They showed that the above ground woody biomass was 1.77 times more than previous models (based on very limited data over 50 years previously) had predicted. This has significant implications for calculating carbon stores in woody vegetation and, therefore, the amount that new woodlands composed largely of native broadleaved species with a good ground cover are predicted to accumulate under different scenarios in the future.



Conifer plantation Kielder
Northumberland

Dewar and Cannell (1992) give modelled uptake levels (**Table 9**) using time-averaged storage of carbon at equilibrium, based on a balance between carbon gains and losses, showing the carrying capacity for carbon capture of some plantation trees used in commercial forestry. The table demonstrates that in this scenario uptake and storage depend on planting density, yield classes (growth of wood per year), harvesting age and whether thinned or not, but all within a forestry context. These figures exclude ground cover (minimal in dense plantations but may be dominated by bryophytes which may play an important role in carbon budgets), any losses of carbon in establishment or from disturbance to the preceding habitat and soils, any losses of DOC or POC or emissions of methane or nitrous oxides (all of which will vary with woodland type and could be significant) and are averaged over the forestry rotation length. Total carbon includes that in trees, wood products, litter and soils, but based on several assumptions that need further research. Within these limitations, the figures suggest not dissimilar carbon totals over the whole rotation cycle for different species.

Table 9.

Some plantation carbon capture figures, based on single species plots (adapted from Dewar and Cannell, 1992), averaged over the rotation cycle - Sitka spruce (*Picea sitchensis*) 55 years, beech and southern beech (*Fagus sylvatica, Northofagus* species) 92 years.

Species	Planting spacing (m)	Average carbon sequestration rates tC ha^{-1} yr^{-1}	Total carbon tC ha^{-1} time averaged at equilibrium
Sitka spruce Yield class 24 unthinned	2	5.6	254
Sitka spruce Yield class 16 unthinned	2	4.5	229
Sitka spruce Yield class 12 unthinned	2	3.7	198
Sitka spruce Yield class 8 unthinned	2	2.9	169
Sitka spruce Yield class 24 thinned	2	4.4	211
Sitka spruce Yield class 16 thinned	2	3.6	192
Sitka spruce Yield class 12 thinned	2	3.0	167
Populus Yield class 12 unthinned, 26 yr rotation	2.7	7.3	212
Southern beech Yield class 16 thinned	1.7	4.6	179
Sitka spruce Yield class 12 thinned	3	3.0	170
Scots pine Yield class 10 thinned	1.8	2.7	178

Lodgepole pine Yield class 8 thinned	1.8	2.5	155
Beech Yield class 6	1.2	2.4	200
Oak Yield class 4	1.2	1.8	154

The total equilibrium value is the maximum amount of carbon that can be stored when the system has reached equilibrium, averaged over time

The sequestration figures compare well with those given by Thomas *et al.* (2011) for ancient woodland, although these are measurements of carbon fluxes rather than averaged models based on growth rates which are not necessarily directly comparable. Carbon sequestration is unlikely to continue for centuries in plantations, although there may be exceptions, possibly more for broadleaved trees. Meyer *et al*. (2021) for example found substantial increases in woody biomass (including dead wood) in 142-year-old pure beech and 113-year-old mixed oak stands in Germany, unmanaged for 50 years, suggesting continuing carbon capture in the woody biomass at least. In contrast, Xiong *et al*. (2020) found that carbon accumulation had stopped in an undisturbed conifer plantation in sub-tropical conditions planted in the 1930s compared with an adjacent old growth forest that was still sequestering carbon in its soils after at least 400 years of no management. A similar pattern was attributed to changes in fine roots down to 90 cm that were shown to peak at 30-36 years, plateau and then decline without compensation from mycorrhizal spread in another pine species in Canada (Waslyliw & Karst, 2020). Xiong *et al*. (2020) consider that the differences relate to litter quality and C:N ratios since the type of litter and its decay characteristics rather than its biomass is correlated with SOC.

The peak uptake rates of CO_2 for commercial forests are generally 5-20 tCO_2 ha^{-1} yr^{-1} (1.36-5.45 tC ha^{-1} yr^{-1}) for tree crops, excluding any field or shrub layer, which are favourable levels compared with other habitats in **Table 7** (Morison *et al*., 2012), but only occur during peak timber increment in the period of canopy closure and are lower before and thereafter, although carbon stocks continue to increase during the growth cycle. To maintain these peak levels requires rapid tree establishment at high densities, management and harvesting at appropriate ages. A typical rotation length of Sitka spruce in the UK is now 35-45 years, but this pre-dates its peak period of sequestration, thus losing potential benefits. These requirements are not compatible with a good wildlife habitat.

However, management to achieve peak CO_2 uptake is not the same as maximising tree carbon stock and Crane (2020) points to a number of issues. First, soil carbon continues to accumulate well beyond the typical harvesting age in a conifer forest up to 80 or 90 years old and continues for many broadleaved species and in ancient and old growth forests. The formation of stable soil carbon stocks (which is less labile and less easily lost after disturbance) takes time and is more effective under deciduous than conifer trees. Thinning reduces the carbon store by about 15%, reducing Sitka spruce stand sequestration to an average of 4.3 tC ha^{-1} yr^{-1} instead of 5.6 for unthinned stands. The difference is greater in higher yield class stands (**Table 9**, Dewar and Cannell, 1992).

Carbon and harvested timber

The longer-term effect of planting commercial crops has to account for the carbon effects of wood use. If replacing carbon-intensive materials like steel and concrete or made into goods like furniture with a long life, a proportion of the harvested carbon is preserved, but if used for short term use (such as paper or cardboard) and then ending in landfill or burnt, the carbon has a much shorter life span (Crane, 2020). The carbon budget also depends on how much of the tree is harvested (usually about 50% of the tree's carbon is in the merchantable stem – Morison *et al*., 2012) and the fuel costs of this; how far it is transported; how much is left on site or lost in its processing; the rate of the subsequent branch, litter and root decay and their release of carbon; the extent of site disturbance in terms of ploughing, drainage and fertilisation for planting the next crop; and the rate of its growth subsequently. There are few direct measurements for some of these parameters although they have been estimated and modelled for several yield classes for commercial plantations (Morison *et al*., 2012). Examples given for a 200-year time span show a stand of oak at Yield Class 6 planted at 6,750 stems/ha (which is far higher than in a semi-natural woodland) with thinning and felling after 150 years and Sitka spruce at Yield Class 12 (at 2,500 stems/ha) with two cycles in 160 years average overall sequestration at 5.7 tCO_2e ha^{-1} yr^{-1} (1.55 tC ha^{-1} yr^{-1}) and 4.1 tCO_2e ha^{-1} yr^{-1} (1.12 tC ha^{-1} yr^{-1}) respectively. They are all assumed to have been established in heather or upland grassland, which will affect the outcomes. These figures update those of Dewar and Cannell shown in **Table 9** by including carbon effects of establishment. Overall, these are not high figures over a long period of time and relate to the losses from the thinning and harvesting cycles and from the organic-rich soils at the start. In general, the shorter the harvesting cycle, the lower the contribution to carbon stocks and sequestration, particularly if harvesting pre-dates the maximum carbon sequestration period.

Conclusions on new woodland creation

The outcome of these considerations strongly supports conservation of existing ancient and semi-natural woodland – they should still be sequestering carbon. Additional new woodlands should facilitate natural regeneration as a priority on clay soils that have been arable or are bare/degraded. Natural regeneration (recommended by Bavin, 2022) might need to be supplemented with planting those species less likely to arrive naturally to assure a mixed, structurally diverse woodland in the future for wildlife (Gregg *et al*., 2021). Several authors recommend planting a diverse range of broadleaved species where they are native, in order to minimise disease issues, provide a range of rooting depths and litter quality, and to ensure a range of complementary carbon sequestration rates (e.g. Lewis *et al.,* 2019, Bavin, 2022, Laganière *et al*., 2010, Crane, 2020). Over time this would deliver the largest carbon stocks for longer, although at a slower rate at times than in the most productive short-lived plantations. Cotrufo *et al*. (2019) also recommend the use of trees that have ECM-associated fungi, which excludes sycamore and ash[4], to maximise carbon accrual.

The optimum solution is also for minimum intervention of these new woods provided there is a good range of structure, new understory species waiting to take over from larger ones and plenty of undercover and ground flora. Coppicing would reduce the carbon stock in the same way as short-term forestry (Crane, 2020), but the rootstock remains and then regrows strongly. Coppicing could be highly selective and help diversify new woodland structure with even-aged trees and could be important for other reasons. Natural events like gales and disease could well deliver structural variation over time in any case.

The research to date shows too that establishing woodlands for carbon objectives is not to be recommended on organic-rich soils and that heathlands and acid grasslands would be better habitats than woodland for carbon capture (Bevan, 2021). Any new planting needs to minimise soil and ground disturbance, with no drainage, ploughing, artificial fertilisation or herbiciding (both very carbon intensive in their production) and minimal access routes into the site (Bavin, 2022). Adding some organic matter to depleted soils could be a useful source of nutrients though where essential. Bavin (2022) also considers the carbon implications of different woodland management but does not provide detailed guidance into different woodland types, including other vegetation (shrubs and a diverse ground flora for example), in order to optimise carbon capture.

A Woodland Carbon Code has been developed (https://woodlandcarboncode.org.uk/), which is advertised as a quality assurance standard for woodland creation projects in the UK. Establishment can be through natural regeneration or planting and under various management regimes from minimum intervention to regular clear-felling (also see Anderson & Morris, 2021).

Wood pastures

There are many woody habitats other than woodland which could contribute to carbon sequestration (see Fletcher *et al.,* 2021 for the scope in Scotland), although there is a general dearth of relevant research about them. Wood pastures with many veteran trees and unimproved grassland may already be important carbon stores (Gregg *et al*., 2021). Tree growth in open conditions is greater than in more closely spaced woodland, so creating new wood pastures combined with diverse grassland could be valuable for carbon capture whilst maintaining agricultural use (Gregg *et al*., 2021), but optimum density or numbers of trees and shrubs is not clear.

Wood pasture – hawthorn and birch with bluebells

Hedges

Similarly, hedges with trees and shrubs have been little researched for their carbon content and sequestration capabilities in relation to management and diversity. Unlike other woody habitats, hedges are usually regularly managed, thus removing some of

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5. See https://mycorrhizae.com/wp-content/uploads/2017/03/Mycorrhizal-Status-of-Families-and-Genera-v1.6.pdf for a list of mycorrhizal types in different trees and shrubs.

the biomass through trimming or laying. The best carbon stores will be in hedges that are wide and tall (Axe *et al.,* 2017). Hedges untrimmed for three years at 3.5 m high and 2.6-4.2 m wide can hold 42 tC ha^{-1} in their above ground biomass, whilst those minimally managed store 45.8 tC ha^{-1}. This compares well with the 47 tC ha^{-1} average found in a meta-study by Drexler *et al.* (2021). The total can reduce to 40.6 and 32.2 tC ha^{-1} if trimmed to 2.7 m and 1.9 m respectively (Axe *et al.*, 2017). Data are sparse to compare species-rich with species-poor hedges, although bramble (*Rubus fruticosus*) invasion is suggested to increase carbon content and one study shows significantly more SOC under mixed compared with species-poor hedges at 175.9 tC ha^{-1} to 1 m depth (in Gregg *et al.*, 2021).

SOC under hedges can be much greater than that on adjacent arable land owing to the range of root depths and lack of ground disturbance. Figures vary depending on the hedge type and measurement system, but Axe (2020) reports stocks to 30 cm of 98.7 tC ha^{-1}, which is higher than the 44 tC ha^{-1} Drexler *et al.* (2021) found in another case. There is also some evidence (although not consistently across different situations) that hedgerows exert an influence beyond their footprint of some 2 m, although soil carbon levels reduce rapidly after 2.2-3.4 m (in Gregg, *et al.*, 2021).

Varied carbon fluxes of hedges are reported - 1 tC ha^{-1} yr^{-1} sequestration for shrubby hedges established on arable land and 1.6 tC ha^{-1} for the addition of hedgerow trees in the Environmental Stewardship Scheme is suggested and Drexler *et al*. (2021) note a range of 2.1 to 5.2 tC ha^{-1} yr^{-1} for different age periods; comparable to many forest studies, but other figures vary from 0.46 tC ha^{-1} yr^{-1} to 12.19 tC ha^{-1} yr^{-1} from different sites with variation linked to woody species type and situations which are sometimes not strictly hedges (Gregg *et al.,* 2021). Carbon sequestration rates will vary with hedge age, density, number and age of trees, overall size and management. It will also vary with climate. In wet years on seasonally wet soils, positive sequestration rates modelled at 6-10 CO_2e ha^{-1} yr^{-1} were found to switch to a net source under droughted conditions to 5.8 tCO_2e ha^{-1} yr^{-1} (-1.6 to -2.73 tC ha^{-1} yr^{-1} changing to +1.58 tC ha^{-1} yr^{-1}) (Ford *et al*., 2021).

Establishing more or restoring existing poor-quality hedges have great potential to help balance carbon losses from agriculture and increase overall sequestration as well as provide good habitat. Hedgerows contouring across arable land can have the added benefit of trapping eroding sediment holding carbon that would otherwise be lost (Gregg *et al.*, 2021) as well as improving water quality and help mitigating flood events.

Scrub

Scrub (excluding dwarf shrubs and young or regrowing trees) is an important habitat in different situations and can be invasive or slow to spread. It can be a valuable carbon store provided it is not encouraged on already carbon-rich habitats like some grasslands or heathland. There is, though, little research specifically on its contribution to carbon budgets in British conditions (Gregg *et al.,* 2021). Some European studies after farmland abandonment in mountainous areas cited in Gregg *et al*. (2021) suggest scrub invasion can result in lower carbon stores in soils that were previously grassland, whilst other authors found the opposite, which could be attributed to differences in soils and climate. There is also a suggestion that more carbon is sequestered when scrub develops on dry soils or in drier climates than on wetter soils or in wetter climates. Scrub development will result in carbon trapped in the woody vegetation, but overall outcomes depend on the amount lost or gained in the soil. More research is needed to provide better guidance.

Mixed native scrub, Monk's Dale NNR, Peak District

3.6 Heathlands

The total carbon stock in heathlands consists of about 98% in belowground reserves and the rest in above ground vegetation. There is a general average of about 90 tC ha^{-1} in heathland soils, although higher levels of 103 tC ha^{-1} have been recorded in South West England with 7.11 tC ha^{-1} in the vegetation (Cantarello *et al*., 2011). Heathlands can be on soils with variable depths of organic matter, so differences in carbon stocks are expected. These soils can be significant carbon sinks, operating over 100s or 1000s of years owing to the high moisture levels and vegetation rich in recalcitrant compounds such as heather litter, which limits decomposition rates and results in the accumulation of SOM (Field *et al.*, 2017).

Carbon in the vegetation changes over time as heather matures, with the carbon stock increasing significantly in the

11-18-year-old community, but little subsequent change in the 18-27 year age bracket (Kopittke *et al*., 2013). This matches the maximum carbon sequestration being during the building and into the mature growth phases. The rates will vary according to the abundance of heather or other heathland shrubs and of mosses (Kopittke *et al*., 2013, show these can be significant in the early growth phases and are usually ignored in carbon calculations but may be important) and the site's management. Carbon levels in the vegetation have not been measured through the whole heather growth cycle for different areas and under different grazing regimes, nor with varying amounts of companion species like bilberry (*Vaccinium myrtillus*).

Good quality coastal heathland Jersey

Field *et al*. (2017) show that low nitrogen additions (as from atmospheric deposition) can increase sequestration rates significantly, but that this shortens the rapid growth phase and ages heather more quickly, in turn affecting heathland carbon dynamics. These findings suggest that a proportion of dwarf shrubs managed to optimise carbon capture should be in their higher growth periods of the heather cycle, although cutting or burning will destroy some vegetation carbon stock (which is low compared with the soils beneath) (Field *et al*., 2017). Farage *et al*. (2009), demonstrate that a burning cycle of 15-20 years in the Yorkshire Dales resulted in the loss of <10% of the total carbon from the system, which needs to be seen in the context of the amount of carbon captured in the underlying soils. Gregg *et al*. (2021) recommend longer burning cycles with smaller proportions burnt, cool burning to avoid damage to the soil surface and burning only in appropriate conditions to minimise carbon losses from soils. Burning or cutting are also management tools that help prevent more damaging wildfires on upland and lowland heathland by managing the amount of old woody dwarf-shrub cover. Charcoal derived from burning has an effect on carbon budgets as it has been shown to increase bulk density and possibly have a negative effect on microbial activity and hence lower decomposition (Heinemeyer *et al.,* 2019), at least on peat.

Heathland sequestration rates in heather's building phase can be as high as 3.45 tC ha^{-1} yr^{-1} (Aberdeen) – not only comparable with many woodland figures (Quin *et al*., 2014), but also attaining these levels earlier than much tree planting. As a habitat of cultural importance with biodiversity and specialist species, creating more heathland to replace lost areas also makes an important contribution to carbon capture targets. Alonso *et al*. (2012) suggest a figure of 3.32 tCO$_2$e ha^{-1} yr^{-1} (0.91 tC ha^{-1} yr^{-1}) averaged for 1-100 years for restoring heathland from arable, but this includes all the carbon losses and gains from the landuse change.

Tree planting on heathlands, particularly on wetter sites or with deeper organic-rich soils, is likely to result in more carbon losses than gains or no net benefits for carbon. Morrison *et al.* (2012), for example, calculate that afforestation of the East Anglian heaths resulted in soil carbon losses of around 0.6 tCO$_2$e ha^{-1} yr^{-1} for 21 years, similar to Friggens *et al*. (2020) findings for plantations up to 39 years old in Scotland.

Where soils are suitable, restoring or re-creating heathland would capture more carbon than many other habitats. Furthermore, restoring acid grassland to heathland (provided there are no other constraints like waxcap interest), can capture twice the carbon levels of acid grassland (Quin *et al*., 2015). These changes could take up to about 10 years depending on the initial conditions. Alonso *et al.* (2021) recommend gradual removal of trees for heathland restoration with minimum soil disturbance so that not all the carbon is lost rapidly. Grazing levels need to be low to maintain the heathland and avoid significant loss of biomass (and carbon) but burning or cutting management may be needed to optimise heather growth phases and safeguard from devastating wildfires. If wet heath is an option, then restoring the hydrology and *Sphagnum* cover are priorities to maximise soil carbon capture and minimise methane production. In general, there may be necessary trade-offs between achieving nature conservation objectives and maximising heathland carbon in order to prevent natural

Heather recolonising acid grassland to restore heathland in dorset

succession to woodland and the loss of specialist and valued species (Gregg *et al*., 2021).

Bracken is a common invader of heathland soils in the uplands and lowlands, but there is a dearth of information on its contribution to carbon stocks. Gregg *et al*. (2021) note figures of 13-119 tC ha^{-1} across 49 sites, and the average in **Table 3** is taken from carbon guidance for Lake District land managers (Hagon *et al*., 2013). Since much tree planting in the uplands is currently taking place within bracken beds, research on the effects of this on carbon stocks is urgent.

3.7 Grasslands

Introduction

Grassland carbon sequestration potential is too often ignored in favour of tree planting yet has the potential to play a vital role in capturing and storing carbon – more so in many cases than can afforestation, since it can simultaneously maintain some agricultural output, work at a much larger scale and respond to change more rapidly, provided there is not a lag effect from past damaging land use change. Agricultural emissions account for some 10% of the UK's carbon emissions (Wentworth & Plumpton, 2019): a similar level to that across the world. It also contributes 33% of methane emissions from stock and is the largest source of nitrous oxide emissions both of which have been increasing recently (IPCC, 2019). Reducing agricultural emissions is another driver for more carbon-sensitive grassland management/restoration as in the 4 in 1000 and the regenerative agriculture movements.

Grasslands are a very diverse group of communities, ranging across the wetness and acidity spectrum, varying with soil type, climate and management. Add to these variables the prevalence of dunging and the depth to which soils are sampled and it is not surprising that the carbon measurements of stock and sequestration are equally variable (Soussana *et al*., 2010) and that, as in woodlands, there are insufficient data across this variation, which is not always taken into account in cross-regional studies. Making recommendations for the best grasslands to capture and store carbon has, therefore, some degree of uncertainty; a reason perhaps for not being promoted more by the CCC. None-the-less, that grasslands could play an important role in climate change mitigation is widely acknowledged (e.g. Soussana *et al*., 2010).


Flower-rich grassland established in Essex on ex-arable soils

All grasslands in the UK have to be managed through grazing (not necessarily with stock) or cutting, so it is more important to incorporate the implications of this for grassland compared with other habitats. Unfortunately, there is a general dearth of data on the whole field carbon cycle (including methane and nitrogen emissions from stock and soil as well as other inputs and outputs) for different grassland communities, but the available research is summarised here.

As in other habitats, most carbon in grasslands is in the soils owing to active rhizodeposition and to earthworm activity which promotes macroaggregate formation. Root turnover creates the largest carbon store in grasslands and favours soil carbon storage owing to the high levels of lignin and polyphenols that tend to lead to recalcitrant rather than labile root litter. More varied and deeper roots thus add to this potential significantly. The SOM is also richer in aromatic compounds under grassland compared with arable cultivation, which also adds to its greater ability to resist degradation (Soussana *et al*., 2004).

Carbon stocks in grasslands

The Countryside Survey 1978-2007 found carbon stocks varying from an average of 60 tC ha^{-1} in neutral to 87 tC ha^{-1} in acid grasslands to 30 cm (Emmett *et al*., 2010), but this does not account for up to 60% of carbon below the topsoil horizon. Ward *et al*. (2016) registered a much higher 403 to 446 tC ha^{-1} of total carbon stock (i.e. organic and inorganic) in a range of soils to 1 m depth, whilst heavy soils managed for dairy cows in Ireland averaged around 300 tC ha^{-1} for the best sites for the whole soil profile (Tuohy *et al.,* 2021). This contrasts with levels of 58 to 100 tC ha^{-1} in Northern British upland meadows on soils 15 cm deep, (Eze *et al*., 2018). Examples taken from Switzerland, France, Ireland and UK show variation from 68 to 125 tC ha^{-1} at 20 cm depth and 93 to 149 tC ha^{-1} at 30 cm depth, all in intensively managed permanent grassland with some clover (Jebari *et al.,* 2021). These figures are often higher than stocks found in soils under other habitats, including broadleaved woodland (although samples at the same depth need to be compared), emphasising the value grassland soils can develop. There is little in the above ground biomass though to add to these stocks compared with that in woodlands.

Grassland management intensity affects SOC, with totals decreasing with increasing intensity of management, mostly in the upper 7.5 cm of soil. However, effects can be still significant at 40 cm depth and more weakly at 60 cm (where soils are sufficiently deep) and 60% of grassland carbon lies below 30 cm depth (Ward *et al*., 2016). High grazing pressure resulting in short vegetation and compaction can become an issue, especially on wet soils, resulting in up to 27% less SOC than on lightly grazed grassland from low litter production levels (Eze *et al*., 2018). It is the quality of the litter and its C:N ratio and abundance of other compounds like phenols rather than its quantity (just as in woodlands) that is more relevant to carbon capture as these help dictate decay rates. Sünnemann *et al.* (2021) show how grasslands managed at low intensity promote microbial activity, metabolic quotient (basal respiration rate per unit of microbial biomass) and microbial biomass as well as stimulating fungal growth, leading to higher carbon storage capacity.

In contrast, Ward *et al.* (2016) found that intermediate levels of management were linked to the highest total carbon stored in grassland soils, with extensive management next and intensive management lowest. The differences on average led to a significant 10.1 tC ha^{-1} in surface soils and 13.7 tC ha^{-1} from 0.3 to 1.0 m more carbon under intermediate compared with intensive levels of management (but intermediate consisted of a single hay crop, low levels of inorganic fertiliser and moderately diverse swards). Moreover, a reduced grazing level and reduced fertiliser input will also lead indirectly to lower methane and nitrous oxide production from stock, depending on its type. Soussana *et al.* (2004) also found that a moderate fertiliser use increased the net OC input, but that intensive fertiliser use not only induced a rise in production but also accelerated mineralisation and enhanced decomposition, hence reducing carbon stocks. As clovers can contribute 100 kg/N/ha/yr or more, then replacing inorganic fertilisers (which also have high embedded carbon) would be more favourable ecologically.

Similarly, liming grasslands can also affect SOC. Abdulla *et al*. (2022), in a global review of the effects of liming, found increased biomass production with liming, leading to greater OM inputs, but increased mineralisation accelerated OM turnover, thus resulting in higher CO_2 emissions and lower SOC stock. Furthermore, liming had no effect on methane (although this is usually low from grassland, Soussana *et al*., 2004) or nitrous oxide emissions from the swards, but as these have greater warming potential, they tended to counteract the effects on biomass and respiration rates.

Overall, Soussana *et al.'s* (2010) review showed higher carbon capture levels with:

- light rather than intensive grazing;
- reduced nitrogen inputs in highly intensive leys;
- grass-legume mixtures rather than short-term leys;
- permanent grasslands rather than leys;
- moderate intensification of nutrient-poor grasslands;
- grazing management rather than cut and grazed or cut only.

Critically, these carbon stocks need to be increasing to support climate change mitigation. This is measured by examining carbon soil stock changes over time or indirectly by eddy covariance flux measurements that can show changes annually. Changes over time tend to be slow, whilst flux measurements provide a carbon balance calculation for the sampling period. Jebari *et al*. (2021), for example, noted positive but small changes over time on several, but not all, sites based on repeated carbon stock measurements under fairly intensive agricultural management, whilst the Countryside Survey data in GB shows increases in all grassland soil types between 1978 and 1998, but a reduction by 2007 (Emmett *et al*., 2010).

However, the true sequestration rates in grasslands need to take the full range of GHGs into consideration, not just carbon capture. Few studies have measured the full gas fluxes from grasslands and results also vary with the methods used and parameters incorporated. However, the indications are that grasslands are still operating as a carbon sink most of the time, even when nitrous oxide and methane emissions are included, along with animal respiration, meat offtake, cutting, manuring and fertilisation (although not counting the operational carbon as in transport and manufacture). Soussana *et al*.

(2004) noted an average 0.5 tC ha^{-1} yr^{-1}overall fixation for intensely grazed grasslands by comparing SOC but only 0.22 tC yr^{-1} capture using flux measurements, while Jones *et al*. (2016) calculated an average GHG balance of 3.66 tCO$_2$e ha^{-1} yr^{-1} (equivalent to about 1 tC ha^{-1} yr^{-1}sequestered) for a South East Scottish field. The overall carbon sequestration total was moderated by a 54% reduction when methane and nitrous oxide emissions were included in the calculations. Rates within the general range were also achieved on the North Wykes Farm Platform for high sugar reseeded grasses plus white clover (*Trifolium repens*) at 2.57 tC ha^{-1} yr^{-1} compared with 0.94 tC ha^{-1} yr^{-1} for permanent pasture (Carswell *et al*., 2019). Two years of flux measurements across the European GREENGRASS nine sites showed positive carbon capture, but only at 0.85 tCO$_2$e ha^{-1} yr^{-1}, which was not significantly different from zero (Soussana *et al.,* 2007). There were very high variations between the two years of the measurements and most of the grasslands involved were intensively managed.

Grassland restoration


Scarified grassland in order to restore species diversity will lose carbon in topsoil

Just restoring grassland on arable land where SOC is likely to be low has significant benefits. A global review by Deng *et al*. (2016) showed that soil stocks significantly increased on average across all land use changes examined after conversions from arable farmland to grassland (0.3 tC ha^{-1} yr^{-1}), although Warner *et al*. (2020) taking a lifecycle assessment approach, suggest a higher level of 1.59 tCO$_2$e ha^{-1} yr^{-1}(0.45 tC ha^{-1} yr^{-1}) for arable reversion to low-input grassland. It is important to note that conversion of grassland to arable results in a rapid loss of carbon, whilst carbon accrual through restoration to permanent grassland is much slower. In one example, about half what had been lost was recaptured over 20 years (Soussana *et al*., 2004).

The evidence so far suggests a more legume-rich, permanent grassland, managed largely through grazing, is likely to provide higher levels of carbon sequestration than a more homogenous grassland with few species, but other research points to the greater potential in more diverse swards that would also meet biodiversity objectives. A global review by Conant *et al.* (2017) found most gains from conversion from arable to grassland (0.87 tC ha^{-1} yr^{-1}), addition of legumes (0.66 tC ha^{-1} yr^{-1}) and moderate fertilisation (0.57 tC ha^{-1} yr^{-1}), although higher rates were generally found in shorter studies than long-term ones. A key requirement is for any enhanced carbon sequestration to continue over time.

Norton *et al.* (2022) found that semi-improved grasslands contain around 15-20% more soil carbon than improved grasslands just in the top 15 cm of soil. This is supported by long-term experiments at Colt Park meadows (Ingleborough NNR) on moderately fertile brown earths over limestone, which showed elevated sequestration rates after ceasing inorganic fertiliser application. This increased rates of carbon and nitrogen accumulation despite an associated reduction in biomass above and below ground, which accompanied an increase in plant diversity (De Deyn *et al.*, 2011). Further increases in plant diversity did not alone increase carbon and nitrogen accumulation, but adding red clover (*Trifolium pratense*) at high seeding levels (5.2 g m^{-2}) to plots already diversified with other species and no fertilisation had a significantly larger effect and importantly, within two years of the treatments, with 3.17 tC ha^{-1} yr^{-1} recorded. The red clover cover increased only from 0.4 to 1.6% after seeding, which is much lower than that found in some MG5 grasslands. Further research is needed to explore the importance of alternative deeper rooting legumes such as bird's-foot trefoil (*Lotus corniculatus*) and changes over time.

Grassland creation with high red clover cover

This carbon capture rate is more than five times the average recorded in some European grasslands but is comparable with conversion of degraded arable cropland to species-rich grassland (De Deyn *et al*., 2011) but lower than the 18.22 $tCO_2e\ ha^{-1}\ yr^{-1}$ (on a 25 year old upland hay meadow) or 27.58 $tCO_2e\ ha^{-1}\ yr^{-1}$ on a 150 year old upland hay meadow (4.97 to 7.52 tC $ha^{-1}\ yr^{-1}$) Eze *et al*. (2018) recorded owing to the low soil respiration rates in upland hay meadows, even with some inorganic fertiliser addition. However, these figures are for measured net ecosystem exchange and exclude carbon exports from cutting or grazing and are therefore not fully comparable. De Deyn's red clover plot results also contrast with their other plots with continuing inorganic fertiliser application and no red clover or other species additions, which overall lost 0.08 tC $ha^{-1}\ yr^{-1}$ owing to faster soil respiration rates and rapid breakdown of litter. Moreover, the rate of carbon accumulation related positively to the cover of red clover. SOM also increased and soil ecosystem respiration reduced with red clover addition. The long-term species addition (that preceded this experiment) also had a low soil ecosystem respiration rate.

Care is needed not to increase nitrogen too much and hence reduce the C:N ratio to the point that more nitrogen would be released as emissions. Guenet *et al*. (2021) warn that higher nitrogen availability using fertilisers can lower the allocation of photosynthates to the root system, thus potentially increasing biomass above ground without an equivalent enhancement of SOC, although the net inputs of nitrogen to soils by legumes have been shown to correlate with a net accumulation of SOC by providing the nitrogen required to stabilise additional SOC in the soils. Using dense nitrogen-fixing cover crops could lead to a nitrogen surplus of about 0.04 tN $ha^{-1}\ yr^{-1}$ compared to the use of non-leguminous plants for example, although legumes also reduce nitrogen leaching and thus would improve water quality (Guenet *et al.,* 2021). There is no specific advice on optimum legume cover but working within semi-natural grassland community averages would seem sensible.

The red clover experiments demonstrate that diversifying grasslands to trap more carbon depends on specific species or traits, not just on the number or total plant biomass. Support for the importance of legumes comes from some American restoration from arable to prairie grassland on sandy soils (Yang *et al.,* 2019). Plots with 1, 2, 4, 8 or 16 species were developed and recorded over 22 years. Those with the higher species number accumulated some 178% more carbon than the monocultures over this time period that was strongly related to two legumes and the abundance of slow-growing prairie grasses.

Lange *et al*. (2015) also found a clear relationship between high plant diversity and high carbon sequestration in plots restored from arable to species-rich grassland in Germany (the Jenna long-term experiment), but without the legume enhancement effect, although they considered this was related to a reduced level of fine root density in the plots with more legumes. They also showed that elevated carbon was a direct function of the soil microbial community activity primed by the increased rhizosphere carbon inputs from the greater plant diversity. Steinbeiss *et al*. (2008), working on the same Jenna project, noted the importance not only of higher species richness but in particular, of tall herbs with their strongest effect at depth in the soils in sequestering carbon and avoiding its loss during establishment. Cong *et al*. (2014) also support the idea of more carbon with greater diversity but without legumes in some Dutch experiments, related to increased soil carbon input and nitrogen retention enhancing plant productivity without fertilisers. Brown knapweed (*Centaurea jacea*) particularly enhanced biomass and carbon soil stocks, but these were not entirely dependent on this species and increasingly positive effects were detected over time (the experiment was established in 2000) on plant productivity. This soil carbon stock is derived essentially from the root biomass and exudates as the above ground material was cut annually.

Grasslands are usually considered to have a carrying capacity for carbon sequestration, but this might take many years to reach with Soussana *et al*. (2010) suggesting several centuries. Yang *et al*. (2019) considered that it would take over 100 years for North American prairie restoration from arable soils, possibly as their specialist grasses colonised slowly, and suggested consistent annual increments for at least 70 years. Their measures showed that there was 90% more carbon in the soils in the 13-22 year period than in the 1-13 year period in the more diverse plots, and that the annual storage rates were 88 and 253% greater respectively in the 13-22 year period in the 0-60 and 0-20 cm profiles than in the first thirteen years. These averaged at 0.54 tC ha^{-1} yr^{-1} for years 1-13 and 0.71 tC ha^{-1} yr^{-1} for the highest diversity treatment in the 13-22 year period. These rates are markedly lower than the Colt Park red clover plots achieved (De Deyn *et al*., 2011), but show useful changes over time.

Given that carbon sequestration rates differ between soil types as well as climate and plant species, then the general principles that higher plant diversity, more stress-tolerators rather than high acquisitive plants, with a good range of potential rooting depth and levels of deep-rooted legumes like red clover (rather than white clover which roots shallowly) provide a positive way forward whilst still maintaining agricultural activity. The research suggests benefits in carbon capture can commence within two years and continue for perhaps 100 years, depending on the condition of the soil at the outset. This makes grassland restoration a worthy alternative or addition to woodland creation on a larger scale and more quickly.

However, care is needed in grassland restoration where this might need cultivation and preparation of a seed-bed. Preparing ground after development or from arable land where all habitats and carbon have been already lost or are very low would not result in much further loss. However, disturbing existing grassland would destroy its carbon stock and the time taken to replenish this will depend on the level at the outset. Compromises will need to be made between seed bed preparation and preserving carbon stocks, perhaps with greater dependency on over-sowing with hay meadow seed or material after cutting and removal of the vegetation. Soil inversion would also result in some carbon loss, although topsoil carbon would be buried. Research is needed on the impact of this practice on carbon as it is used to bury nutrient-rich top-soils for a variety of habitat creation projects.

Low intensity grazing may be the optimum for carbon capture, with lower stock numbers reducing the amount of biomass consumed (therefore leaving more litter to enter the soil) and reducing methane emissions with fewer animals. However, on many restored grassland locations, cutting may be the only management possible outside an agricultural context. Cutting as late as possible (autumn - unless the material can be used for hay) to allow more organic material to reach the soil and for invertebrates to complete their life cycle is one option. If the material is composted and respread in the area (not onto the grassland though), then the carbon is at least retained locally.

In conclusion, the potential for grasslands to contribute to climate mitigation seems enormous. With the stress farmers are facing with increased fertiliser and fuel costs, one approach would be to restore flower-rich grassland to match our best types on the most appropriate sites ideally managed though extensive grazing where possible, or cutting with or without grazing. But then seek to diversify a much larger area of higher fertility sites with a more limited range of species capable of coping with the nutrient levels. These should include a mixture of root depths and red clover and other legumes depending on the soils and sites. This could result in both significant improvements in grassland wildlife but also carbon sequestration and reduced costs for farmers.

Other grassland types

Gregg *et al.* (2021) suggest that purple moor-grass (*Molinia caerulea*) dominated swards could be losing 0.85 tCO_2e ha^{-1} yr^{-1} with high grazing levels or be sequestering carbon at between 0.5 and 0.53 tCO_2e ha^{-1} yr^{-1} with low or no grazing. This will also be affected by saturation levels and the degree of anaerobic conditions associated with methane production as well as the variety and abundance of companion plants like *Sphagnum*. In contrast, there is very little information on calcareous grasslands which tend to be on shallow soils. Estimates given by Gregg *et al*. (2021) are between 51 and 69 tC ha^{-1} in soils up to 15 cm deep, but sequestration measurements are scarce, with Dawson & Smith (2006) giving 1.2 tC ha^{-1} yr^{-1} for cut species-rich limestone grassland with no other information. There is no separate information on representative wet or marshy grasslands, although many of the principles already described as well as of methane generation and water tables might be applicable.

3.8 Urban environment

Many of the habitats already described occur in urban environments as habitat islands of varying size, encapsulated by inhospitable urban infrastructure. The accounts provided will be equally applicable in these locations where the habitats are large enough to display natural functioning and processes. However, with the increased emphasis on anthropogenic activities and effects, some additional information is warranted. Not all the habitats set out above have been studied in the urban context and carbon budgets associated with urban specialities such as colonised waste materials on brownfield sites have not been studied. In addition, there is greater potential for habitats and individual trees for example, to help counteract climate change directly through carbon sequestration and storage and indirectly by moderating temperatures, insulating surfaces or shading buildings and urban infrastructure as well as providing other crucial ecosystem services.

The size of carbon reservoirs within urban areas can be substantial, although it is small when compared with carbon emissions per unit area (Nowak & Crane, 2002). Most of the earlier measures focused on trees rather than all habitats and vegetation (including gardens) within the urban boundary. Recognising this, Davies *et al*. (2011) surveyed the whole of Leicester City (approximately 73 km^2, population 300,000) in 2009 and estimated 231,521 tC were stored just in the above ground vegetation, with 97.3% of this carbon pool in trees rather than herbaceous or other woody vegetation. Domestic gardens stored just 0.76 kgC m^{-2} (7.6 tC ha^{-1}), with the greatest above-ground carbon density (288.6 tC ha^{-1}) associated with tree cover on publicly open spaces. None of these figures include the substantially greater carbon stored in the soils nor the indirect impacts on energy use from temperature modulation. Further information is available on some of these urban habitats.

Urban Ponds

Information on ponds and waste-water treatment ponds, although sometimes also relevant to a more rural environment, is exemplified by recent research by Peacock *et al*. (2019) into the carbon dynamics of artificial urban ponds in Sweden. They found that methane concentrations tended to be greatest in high-nutrient ponds, but that there were many other, as yet unidentified, factors that might affect emissions such as sediment properties and associated microbial communities, water depth and water source (carbon-rich ground water for example). There were no differences however, between ponds of differing size or whether they were ornamental or for water regulation, but this might be related to their urban location and the overriding anthropogenic influences. Across 40 sample sites, annual methane emissions averaged 8.3 g m^{-2} yr^{-1} (0.083 tC ha^{-1} yr^{-1}) and all ponds emitted this gas. Twenty-eight of the ponds were net emitters, whilst 12 were net consumers of CO_2. The average loss of CO_2 was 2.06 tC ha^{-1} yr^{-1}, (although the authors considered this to be an underestimate). Similar results in Denmark (Audet *et al.,* 2020) showed urban ponds also generally acting as a GHG source, with tree cover, water temperature and nitrate concentration identified as the main drivers of CO_2, methane and nitrous oxide concentrations respectively. Minimising eutrophication, as for rural ponds, will assist in reducing methane losses and has to be balanced against the cooling effect of water within an urban heat island as well as its biodiversity value and the pleasure that can be brought to many people.

Constructed Wetlands

The development and use of constructed wetlands for wastewater treatment during the 20th century focussed on the biodegradation of organic pollutants using subsurface and vertical flow wetland systems that in many ways can encourage the release of CO_2 through mineralisation and oxidation of the waste stream. However, over the past 30 years the use of overland flow wetlands and, in particular, the development of Integrated Constructed Wetlands (ICWs), which make use of natural and beneficial biochemical and biophysical processes in the design and management of the wetlands, presents an opportunity to enhance carbon sequestration through design and management of new wetland systems. The concept behind ICWs is to create wetlands which are multifunctional, with the primary aims of managing water quality, water quantity and biodiversity within an appropriate landscape context (Harrington & McInnes 2009). Constructed and natural wetlands can sequester carbon through high rates of organic matter

Constructed wetland in urban setting '- Peter Worrall input and reduced rates of decomposition (Pant *et al*., 2003). Mitsch *et al*. (2013) demonstrated that in created temperate wetlands in Ohio after 15 years of development, carbon storage ranged between 219 and 267 gC $m^{-2} yr^{-1}$ (equivalent to 2.19 to 2.67 tC ha^{-1} yr^{-1}). This compared to flow-through temperate wetlands which had much shorter hydraulic retention times and carbon sequestration rates of 124 to 160 gC $m^{-2} yr^{-1}$ (1.23-1.60 tC ha^{-1} yr^{-1}).

In the design of ICWs and overland flow wetlands, a key parameter is the hydraulic retention time of the wetland (Wu *et al*., 2015). Initially, the need for long retention times and permanent water cover within these wetlands was to facilitate the retention of phosphorus whilst at the same time creating diverse and robust wetland ecosystems. The link between carbon and the retention of phosphorus is likely to be associated with the capacity to sequester carbon because an increase in humic acid availability in constructed as well as natural wetlands increases the capacity of the wetland systems to retain phosphorus sustainably (pers. com. Rory Harrington, 2021). Such integrated processes of carbon sequestration and phosphorus retention have implications on how wetland systems are managed with hydraulic retention times and permanence of surface waters becoming key design parameters. As Beechener *et al.* (2021) point out, longer established constructed wetlands could be sinks or sources of GHG and their future as a useful carbon trap depend on a better understanding and design of the most beneficial wetland systems to sequester carbon.

Urban trees

Trees in town centre providing shade and habitat – John Box

Trees are a ubiquitous part of urban landscapes, covering from 10 to 67% of urban and community areas in the USA and 20% of Greater London (quoted in Edmondson *et al.*, 2014). There is no doubt that urban trees hold significant carbon stocks, but few studies have investigated the detail. Davies *et al*. (2011), highlighted the value of urban trees as above ground carbon stores in a typical English City – Leicester, holding 225,217 tC (95% confidence level) in nearly 900,000 trees. A further 1744 tC is stored in other woody vegetation. Of the trees, the four most common species accounted for 40% of all trees – hawthorn (*Crataegus monogyna*), field maple (*Acer campestre*), ash (*Fraxinus excelsior*) and cherry (*Prunus avium*). This dramatic division of above-ground carbon was not repeated below ground (mostly clay-rich soils). In a comparison of soil carbon to 1 m depth under ash, pedunculate oak and field maple in the same City study, Edmondson *et al.* (2014) found more modest differences between tree and grassland carbon stocks, which were highest under ash (and of concern owing to ash dieback) and then maple (260 and 190 tC ha^{-1} respectively). Stocks under oak, mixed woodland and adjacent grassland were not significantly different (140-150 tC ha^{-1}). Perhaps surprisingly, there was no significant difference in soil carbon stocks under trees of different ages (from some 10 years to several centuries).

Since trees generally do not regenerate naturally within an urban environment when isolated in streets or pavements, a continual re-planting programme will be needed over time to maintain or enhance carbon stocks and sequestration rates and species need to be chosen that can reach their productive life span but at the same time tolerate future climate change. Davies *et al*. (2011) note that urban tress already suffer stress and reduced growth rates owing to atmospheric pollutants and heat island effects of increasing temperatures as well as more droughts exacerbated by compaction and impervious surfaces. New tree diseases are an additional threat. Not only should a wider range of trees be established to counteract these stresses and ensure survival into the future, but these also need to be selected to thrive in the face of climate change. Predictive provenancing studies have been undertaken in the United States (Arnold *et al*., 2012), for example, and Roloff *et al*. (2009) review 250 urban woody species in Central European parks and gardens for their fitness to climate change. Sjöman *et al*. (2012) identified tree species and genotypes adapted to inner city environments in northern parts of central Europe and found only four were regularly used, meaning that a further 23 species could increase diversity and resilience in the urban environment.

Urban grasslands

Few studies focus on urban grasslands but Schittko *et al.* (2022) show in Berlin, where urban dry grasslands (semi-natural or highly modified and thus varied in plant diversity) were found to hold on average 83.5 tC ha^{-1} in the top 30 cm of soil, but more where plant species richness was highest (up to 140 tC ha^{-1}). This is at the high end of the rural grassland range, so highly beneficial in an urban setting. Soil multifunctionality measured through plant richness, soil faunal and earthworm diversity were all positively correlated with higher levels of SOC in this German investigation.

Green Roofs

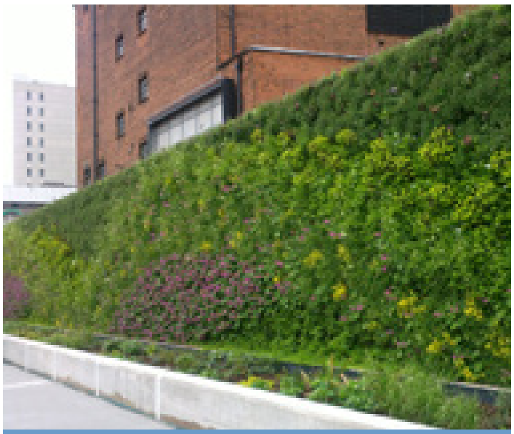
Green roofs can help reduce carbon emissions in urban areas directly and indirectly. The roof habitat can capture and store carbon, whilst it can also reduce building energy consumption through reflecting, absorbing and emitting heat. The ground-breaking IGNITION project in Greater Manchester set out to increase the city's resilience to the extremes of climate change using NbS including rain gardens, street trees, green roofs and walls[5]. Their website includes a summary of the research to support the application of these solutions and their Green Roof Benefits Calculator provides a useful high level decision making tool (https://ignitiongreenroofbenefitscalculator.greatermanchester-ca.gov.uk/default.cshtml).

Vegetation and soil properties are the key factors affecting the performance of building energy consumption and CO$_2$ sequestration (Shafique *et al*., 2020) but there is no relevant research evaluating different substrates and vegetation to optimise carbon capture on green roofs. Some modelling of their potential value has been attempted in North America with between 0.07 and 0.09 tCO$_2$ ha^{-1} yr^{-1} potentially removed by green roofs (quoted in Shafique *et al.,* 2020), but the models used were not devised for green roofs. Improving the substrate performance by adding biochar or sewage sludge in China gave an average carbon storage capacity of 9.3 or 7.9 gC m^{-2} (equivalent to 93 and 79 tC ha^{-1}) respectively (quoted in Shafique *et al*.'s review, 2020), which is very respectable compared with terrestrial habitats (see **Tables 3 & 4**). London is reported to have around 29 ha of green roofs, which is around 40% of UK green roofs (Roofing Times, May 2019). This now substantial area can make a significant contribution to carbon sequestration targets.

In order to optimise biodiversity benefits as well as carbon sequestration, Shafique *et al*. (2020) recommend using *Sedum* species plus local grassland annuals, based on some comparative studies, rather than just *Sedum* species that were not considered the best for wildlife (although the studies were not of British species, nor did they test other groups without *Sedum*).

Gardens

As allotments cover more than 8,000 ha in the UK (Edmondson *et al.,* 2014) and gardens over 433,000 ha there is potential for contributing seriously to carbon capture. Moreover, there are about 30 million trees in gardens, making up about a quarter of the national tree resource outside woodland, covering over 47,000 ha (only just less than the entire New Forest National Park). About 16% of gardens contain ponds –in the order of 3.5 million ponds or 350 ha of pond habitat (equivalent to a quarter of Lake Windermere!), (Wildlife Gardening Forum).



Green wall, Birmingham New Street Station

The best approach to carbon in gardens is to minimise use of carbon-intensive materials and to optimise the application of measures to increase carbon capture and storage. These have been expounded in a CIEEM blog (https://cieem.net/how-to-get-more-wildlife-into-your-garden-and-absorb-more-carbon-by-penny-anderson/) and in a Royal Horticulture Society publication (Nex, 2021). Advice focuses on developing soils, (creating and adding compost; minimising disturbance like cultivation, not using peat); managing lawns to increase height, plant diversity and flowering; minimising use of chemicals; and planting trees and shrubs). Reducing energy use in buildings is encouraged by growing plants on walls, establishing trees safely around buildings to provide shade and adding green rooves.

Gardeners are encouraged to manage their plots for wildlife through a number of media channels, but there is less emphasis on management for carbon, although many of the measures will benefit both. Few gardeners will know, for example, that there is more carbon in soils than their plants and that soil disturbance will reduce or destroy soil carbon. There is an opportunity therefore to raise awareness and encourage wider thinking. The British Association of Soil Science has a leaflet on carbon and soils[6], for example, but such examples need wider dissemination and advertising to make a

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6. Their website includes an evidence base for these NbS and a carbon calculator to help make decisions (https://www.greatermanchester-ca.gov.uk/what-we-do/environment/natural-environment/ignition/)

7. https://soils.org.uk/grounded/

difference. CIEEM has the opportunity to support and initiate further initiatives.

3.9 Marine and coastal habitats

Marine habitats have the advantage of generally not releasing methane as the sulphide in the sediments inhibits the bacteria responsible, although Beaumont *et al*. (2014) quote recent evidence that locally high emissions are possible from saltmarshes, especially if grazed, and Gregg *et al*. (2021) suggest it can be released in the water column. Marine habitats are also often large scale and therefore have the potential to be much more effective at capturing carbon than many terrestrial habitats restricted in extent. Altogether, marine and intertidal habitats hold more carbon than terrestrial ones in a much smaller area. Moreover, many accrete vertically and can continue accumulating carbon for centuries if not disturbed. Carbon trapping rates will vary as they depend on the hydro-periodicity, salinity, nutrient status and sediment supply. However, many of the marine habitats have been largely destroyed or disturbed by, for example, bottom trawling or nutrient enrichment, and their carbon stores are easily disturbed.

Intertidal and subtidal sediments are also potentially important carbon stores, with fine sediments storing more than sandy ones, giving carbon hotspots in places (Gregg *et al*., 2021). There is generally inadequate information on the full range of marine and coastal habitats, but the potential to support carbon sequestration efforts are considerable. Key habitats for sequestration are saltmarshes, sand dunes, machair, seagrass (*Zostera*) beds and muds. Carbon sequestration will be greatest where soil/sediment accumulation is most rapid and where growth rates are high such as in saltmarshes and seagrass beds.

Saltmarshes

Porlock beach and salt marsh, Somerset

Saltmarshes can reach 6.5 tC ha^{-1} yr^{-1} accumulation or more in ideal conditions, (McCleod *et al*., 2011 quote figures of 0.18 to 17.13 tC ha^{-1} yr^{-1} internationally), with rates of 0.6 to 2.2 tC ha^{-1} yr^{-1} from different UK studies (quoted in Armstrong *et al*., 2020), higher by some 30% in non-sandy material. Mason *et al.* (2022) calculated an average sequestration rate of 8.2 tCO_2e ha^{-1} yr^{-1} in natural British and Northern European saltmarshes of similar type compared with 13.3 tCO_2e ha^{-1} yr^{-1} for restored marshes (although from a small sample), mostly re-established in the last 20 years, although there was considerable variation. Gregg *et al*. (2021) also show variation in stocks associated with different plant communities.

Saltmarshes are reputed to have the highest carbon burial rate per unit area compared to other blue carbon habitats (quoted in Armstrong *et al*., 2020), and higher than most terrestrial habitats. Gao *et al*. (2022) confirms these generalisations in a global review, revealing the potential for the highest carbon capture density of 12.26 tC ha^{-1} yr^{-1} (including sediment contributions) and sequestration intensity (2.24 tC ha^{-1} yr^{-1}) in saltmarshes plants. Saltmarshes can have very deep deposits (over 10 m – Beaumont *et al*., 2014) and thus can accumulate large carbon stores provided these are not disturbed over a long time period. However, with sea level rise, saltmarshes can be squeezed out and drowned or they can steadily accumulate in the rising water levels. Gao *et al,* (2022) also stress the high rate of loss of saltmarshes globally (53% since 1900).

Restoring saltmarsh habitat, as in several managed retreat schemes, combined with transition habitats to brackish and freshwater conditions, with large reed beds for example, has the potential to contribute significantly to carbon capture. However, Gregg *et al*. (2021) suggest restoration of high saltmarsh can be slow in accumulating carbon, whilst the accumulated carbon in low restored shore sites were not significantly different from the natural ones after 15 years. Mason *et al.* (2022) also list sediment type, vegetation community, pH, salinity, sulphate levels, temperature and elevation as potential drivers of SOC pools in saltmarshes. Other studies found sequestration rates were high (averaging 1.04 tC ha^{-1} yr^{-1} in the first 20 years after restoration) before dropping to around 0.65 tC ha^{-1} yr^{-1} thereafter (Burden *et al.,* 2019). There is a suggestion that it could take around 100 years for a restored saltmarsh to gain the equivalent carbon stocks to a natural site.

A pilot Saltmarsh Carbon Code is being developed by UKCEH based on a rigorous and scientifically based voluntary certification standard, enabling saltmarsh carbon to be marketed and purchased by private investors. This should provide an income stream for restoration projects and supporting the achievement of national net zero goals. (See https://www.ceh.ac.uk/our-science/projects/uk-saltmarsh-code).

Tidal sediments

Intertidal and subtidal sediments can be equally important as a carbon sink from accumulated sediment from both the marine and terrestrial environment. 0.16 tC ha^{-1} yr^{-1} for example has been calculated for the Humber estuary (Alonso *et al*., 2012), and 1.12-1.98 tCO_2e ha^{-1} yr^{-1} (0.31-0.54 tC ha^{-1} yr^{-1}) has been given for intertidal and subtidal sediments (Gregg *et al*., 2021). Estuaries and mud banks that are accreting sediment will also be adding to their carbon stores, Hutchings *et al*. (2020) estimate that some 40% of deposited material was buried long-term in sediments with rates up to 0.64 tC ha^{-1} yr^{-1} depending on the characteristics of the estuary and adjacent coastline. This is higher than the average 0.11 to 0.37 tC ha^{-1} yr^{-1} used by Armstrong *et al.* (2020) in their estimate of the carbon value of Welsh marine habitats. Natural mud flats in the Blackwater Estuary, Essex, were found to have a higher sequestration rate than managed re-alignment sites (0.94 tC ha^{-1} yr^{-1} compared with 0.73 tC ha^{-1} yr^{-1}), which at a large scale could be significant.

Sea

Similarly, deep or shallow seas store biologically derived sediment mostly from land via rivers, estuaries and sea lochs. This accumulates faster closer to land and the sediment source. Much also comes from the phytoplankton and breakdown of shells. This carbon can survive in deep sediments for hundreds of years (Burrows *et al*., 2014).

Seagrass meadows

Seagrass meadows can capture carbon rapidly and have the added benefits of protecting beaches from erosion and providing good habitat, but they have suffered large-scale losses around British estuaries estimated at an 85% since the 1920s with little natural recovery (Unsworth *et al*., 2019). Gao *et al.* (2022) note that the decline in Europe has slowed since the 1980s and reversed in the 2000s, with density metrics improving or remaining stable. Farrant (2023), point to the additional damage that recreational moorings cause and illustrate advanced mooring systems that can reduce this impact. Seagrass beds can only grow in shallow waters, mostly below 20 m, but occasionally up to 90 m deep (Gao *et al*., 2022). Poor water quality driven primarily by excess nutrients is one of the largest threats (Farrant, 2023). The anoxic nature of marine seagrass sediments, the low sediment hydraulic conductivity and slow microbial decomposition rates all assist carbon burial leading to carbon preservation in seagrass sediments for potentially thousands of years (Armstrong *et al*., 2020).

Seagrass restoration has been limited until recently but has enormous potential benefits. Burrows *et al*. (2014) quote sequestration rates of 0.2 to 2 tC ha^{-1} yr^{-1}, also corroborated by McCleod *et al*. (2011), who give an average of 1.38 tC ha^{-1} yr^{-1}: a high level, through trapping sediment rich in carbon and capturing it through photosynthesis. Gao *et al*. (2022) support this with a global average of 1.17 tC ha^{-1} yr^{-1}, although all measures are varied. This is higher than the 0.83 tC ha^{-1} yr^{-1} averaged by Armstrong *et al.* (2020) and the 0.27 tC ha^{-1} yr^{-1} they apply to evaluate the Welsh seagrass beds. Greiner *et al*. (2013) reported 0.37 tC ha^{-1} yr^{-1} carbon sequestration averaging over 10 years in newly established seagrass beds of eelgrass (*Zostera marina*) in Virginia coastal bays and with higher annual rates in 10-year old meadows compared with four-year-old ones and bare mud. They calculated that new beds should be trapping as much carbon as mature ones within 12 years of restoration. Watanabe & Kuwae (2021) found similar levels of 0.27-0.28 tC ha^{-1} yr^{-1} in an unplanned seagrass meadow in Japan that developed in a bay after its mouth was widened to give better marine water access and found the carbon accrual rate continued for more than 20 years at least. Unsworth *et al*. (2019) give details of some experimental work showing the importance of scale and the selection of appropriate site conditions plus novel methods of seeding for establishing seagrass beds off South Wales whilst Farrant (2023) describes recent restoration methods and achievements by the Ocean Conservation Trust in Southern England.

Macroalgae

These occur along coasts of all continents, from the intertidal zone to considerable depths receiving little light, particularly on rocky shores. They also play an essential role in coastal ecosystems as refugia, nursery habitat and food sources (Gao *et al*., 2022). Wild macroalgae, like kelp beds, have the potential to sequester some 0.62 tC ha^{-1} yr^{-1}, but have been declining extensively caused by warming, eutrophication, pollution and harvesting. As a result, more large-scale macro-algae are being cultivated in some areas using bamboo poles or floating buoys. These have the potential to sequester in the order of 2.38 tC ha^{-1} yr^{-1} (Gao *et al*., 2022), which is high, and also trap and utilise nutrients in the sea, thus reducing ambient levels derived from agricultural runoff and sewage outflows. The overall long-term value depends on how the seaweed is used when harvested (fertiliser, food, toothpastes, shampoos etc) and the overall carbon costs of this and its transport.

Coralline algae and biogenic reefs

There are other important marine habitats for carbon capture, many of which are priority habitats, as described in Burrows *et al.* (2014). Maerl beds, which depend on species of coralline algae, provide a long-term store as calcium carbonate, but their rates of accretion are slow – typically 0.23 mm yr^{-1}, although they are extensive around Scotland and the Southern and Western coasts of the British Isles, with a large area and volume. Deposits can be 60 cm deep with dead material even deeper. Possible sequestration rates can be about 0.74 tC ha^{-1} yr^{-1} of calcium carbonate and 1.7 of organic carbon. Biogenic reefs are solid structures created by accumulations of organisms usually rising from the seabed or at least clearly forming a substantial discrete community or habitat which is very different from the surrounding seabed. The structure consists of reef-building organisms and their tubes or shells with sediment, stones and shells bound together by the organisms. Corals, mussels, tubeworms and flame shells are all involved. Horse mussel reefs are the richest of all and are a Special Area of Conservation feature. Their net carbon accumulation is between 0.4 and 4.2 tC ha^{-1} yr^{-1}, again which is potentially very high (Burrows *et al*., 2014).

Sand dunes

Sand dunes are important habitats, but little is known about the best way to manage or restore them for carbon sequestration. Beaumont *et al.* (2014) give 0.58-0.73 tC ha^{-1} yr^{-1}, average sequestration rates, which is very similar to the 0.58 tC ha^{-1} yr^{-1} for dry dunes and 0.73 tC ha^{-1} yr^{-1} for wet dune slack habitats given by Jones *et al.* (2008). Methane fluxes are likely to be very low on sand dunes owing to generally low moisture levels (Gregg *et al.,* 2021).

High value sand dunes Jersey

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Chapter 4

Impacts of climate change

One last and critical factor is climate change itself. It is important to climate-proof projects. Climate change is affecting carbon sequestration and stocks, so will have an impact on decisions we make to create or restore different habitats. The research reflects either measurements taken over several years with links to climate variables, or to imposed drought or temperature increases on experimental plots covering a range of habitats and countries. The key factors are drought, changes in precipitation and increases in CO_2.

In general terms, Wang *et al*. (2022) estimated from a global review that SOC could decline by 6%, 4.8% and 1.3% respectively from 0-0.3, 0.3-1 and 1-2 m depths with 1°C warming, plus an additional 4.2, 2.2 and 1.4% decline for every additional 1°C, thus affecting topsoils more than subsoils, which would be more protected. Most might be lost in boreal rather than other types of forests, but temperate forests were estimated to lose 10% of SOC with 1°C and 34% with 5°C warming, with only 10% loss in the tropics and subtropics. These changes are heavily influenced by precipitation and temperature changes, but less so by soil type (although these will differ in their responses to drought) or landform. In general, Wang *et al*. (2022) predict greater losses where SOC are high at the outset.

This is particularly important in peatlands. Bioclimatic envelope modelling shows that higher peat decomposition and lower carbon sequestration potential is likely with lower rainfall, more evapotranspiration and water tables below 12 cm under climate change scenarios (Heinemeyer et al., 2019, Evans et al., 2022, Gallego-Sala & Prentice, 2012). Many blanket bog regions are thus at risk of progressive peat erosion and vegetation changes as a direct consequence of climate change (Gallego-Sala & Prentice, 2012). Since degraded peatlands are already responsible globally for 5-10% of anthropogenic $C0_2$ emissions (Loisel & Gallego-Sala, 2012), this is a major threat.

Earlier work offered mechanisms for potential carbon losses (Wang *et al*., 2021). Drought was found to reduce photosynthate production and therefore also translocation below ground to the rhizosphere. Soil available phosphate also decreased with drought, unless more nitrogen was added, suggesting plants were relying more on mycorrhizal symbiosis to take this up. More of the photosynthate was also partitioned into roots and less into soils under drought conditions, possibly to enhance access to water. In these experiments, the reaction to drought seemed to be more dependent on nitrogen availability and fungal growth, which was also lowered in drought conditions. These experiments demonstrate that climate change effects can be very complex and show that mycorrhizal associations can strongly influence plant carbon allocation, collaborating with free-living microbes for nutrient foraging and contributing to carbon allocation into roots and root respiration.

Parched limestone grassland, Peak District, Biggin Dale 2022

Although there are several limitations and uncertainties in Wang *et al*.'s (2022) wider assessment, the results are in line with individual studies. Soussana *et al*. (2010) describe the severe heat wave and drought across Europe in 2003 when total primary productivity was reduced by 30%, resulting in a strong anomalous net source of atmospheric CO_2 reversing the effect of four years of net ecosystem carbon sequestration in the grasslands studied. These authors were concerned that future droughts could turn temperate grasslands into carbon sources rather than sinks, noting that increased soil temperature is likely to speed decomposition and result in reduced carbon stocks, even if increased net primary productivity could slow this loss. Moreover, projected changes in temperature and precipitation are also predicted to reduce overall plant diversity, which could impact carbon sequestration as well.

Grossiord *et al*. (2022), by imposing drought and reduced moisture experimentally on beech and pubescent oak (*Quercus pubescens*) in France, found that year-round warming advanced leaf flushing but did not affect senescence, hence increasing the growing season. It also increased leaf photosynthetic activity. Reduced soil moisture had little impact on phenological events but resulted in reduced photosynthesis as stomata closed, which would also reduce exudate production in the rhizosphere.

There are other implications associated with elevated CO_2 levels in the atmosphere. Walker *et al*. (2020) considered that CO_2 could be in surplus and not affect photosynthesis, with other factors such as nitrogen and other nutrients being limiting. Overall, however, Walker *et al*. (2020) considered with medium confidence that the collated evidence showed increased terrestrial gross primary productivity with increasing CO_2 and a similar increase in wood biomass productivity but with low confidence on the magnitude, which is possibly related to nutrient availability. The same authors also show that there has been an increase in tree mortality, possibly related more to droughts, which increase transpiration from the larger leaf surface area, thus exacerbating mortality risks. Brienen *et al*. (2020) confirm, this, showing that increasing CO_2, together with rising temperatures and nitrogen deposition are increasing tree growth across the world. This increases stem mortality and biomass recycling resulting in a 23% reduction in tree life-span but provides no long-term increase in biomass stocks. Increased growth rates also affect wood density, with less durability affecting future uses and thus their value (Crane, 2020).

More specifically related to SOC, Walker *et al.*'s (2020) review supports the theory that increasing CO_2 increases SOM decomposition rates as a result of increases in microbial biomass, rhizosphere priming, mycorrhizal association and increases in soil water content. There is medium confidence in the increase in SOM decomposition but less so for its magnitude. This is reflected in the overall sequestration figures in Wytham Woods which were halved after periods of higher temperatures (with no water deficits) when net soil respiration rates were elevated (Thomas *et al*., 2011).

Barraclough *et al*. (2015) consider that climate change may influence soil carbon indirectly through changing vegetation cover and litter quality. Neutral grasslands have been shown to increase their resistance to drought when diversity is restored, possibly owing to reduced water demand, greater rooting depth and lower productivity of a more diverse sward (Gregg *et al.,* 2021).

On heathlands, a significant 60% of carbon in wet heathland soils was lost in just two months of induced drought as soils became aerobic (Gorissen *et al*., 2004). Similar losses could be predicted from peatlands when these are droughted, adding to the current high level of losses. Thaysen *et al*. (2017) explored the detail of such changes on wavy-hair grass (*Avenella flexuosa*)/ heather heathland in factorial experiments combining CO_2, drought and warming. They found warming (0.3 and 0.4°C elevation in different soil horizons) significantly decreased soil carbon and nitrogen stocks in the A2 horizon and, when combined with CO_2 elevation, further reduced the soil carbon stock from 17.68 to 13.55 tC ha^{-1} in this thin horizon. Combining all three treatments resulted in major declines in the carbon stock in the same horizon along with a 40% reduction in nitrogen in the light litter fraction. This horizon was more affected than elsewhere in the system, possibly responding to greater changes in root rather than above ground plant productivity, although carbon turnover was higher in the A1 horizon. It was the light fraction (the decomposing litter mostly) that was affected rather than the MAOM (the heavy fraction), suggesting increased soil respiration in a warming world. Thaysen *et al*. (2020) warn that these results will be highly significant in countries with extensive dwarf-shrub cover.

As droughts of varying lengths are becoming more regular, and as temperatures and CO_2 levels rise, these will affect many different habitats, more particularly in lowland Britain where the combinations are expected to be more severe. These factors will need to be taken into consideration when creating habitats for carbon sequestration.

Dorset heath on fire – an increasing risk with climate change

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Chapter 5

Conclusions

This review explores the relationship between carbon and habitats to inform ecologists and environmental managers working to counteract both the biodiversity and the climate change crises in combination, while also enhancing other ecosystem services. The key conclusions are that the restoration of peatlands is the highest priority, as identified by the CCC (2020) and Gregg *et al*. (2021).

However, the seemingly universal mantra to plant trees to capture carbon would be better replaced by a range of alternative approaches that would suit the local landscape ecology, topography, soils and climates. This view does not accord with the CCC (2020) recommendations, which focus on new tree planting (as well as peatland restoration). It is shown here that new woodlands (rather than plantations) are important ecologically but may take up to 30 years before their carbon budgets are positive. This demonstrates that alternative or additional approaches are needed simultaneously to capture carbon more immediately and at large scales. Gregg *et al*. (2021) come to similar conclusions about tree planting.

Milled raised peat bog, Shannon, Ireland, cessation of extraction and restoration of such sites urgent

The evidence points to habitat protection, restoration and enhancement as the priority to maintain existing carbon and wildlife sites. In addition, habitat creation is required not only to mop up more carbon, but to implement Lawton's principles of bigger, better, more and more joined up habitats (Lawton *et al*., 2010) and reverse the huge losses and fragmentation of high value habitats over the last century or more. A range of habitats can be employed to fulfil these roles, not just planting trees. Indeed, positive carbon sequestration balances can be more rapidly achieved in non-wooded habitats in many cases, providing opportunities for focusing on large scale habitat mosaics. Soils, microbiology, fungi and sediments are the key and an understanding of soil functioning and the opportunities presented is critical in designing the best new habitats to capture carbon most effectively.

Although this review shows that the relationship between carbon and habitats is complex and not fully understood in terms of the variability and effects of different climates, soils, vegetation communities and habitat management, sufficient is known to make strong arguments for a wider approach to carbon capture in different habitats. Marine and intertidal habitat restoration and creation on a large scale together with reconnecting rivers to their floodplains converted into flower-rich flood meadows would both provide major benefits and simultaneously enhance other ecosystem services. Furthermore, the creation of other wetlands from ponds to reed beds would also be productive and fit into the rest of the mosaic, provided methane generation is minimised. New heathland creation and diversification of neutral grasslands plus adding red clover and possibly other deep-rooted legumes would do much to stem the losses over the last century or more where soils are suitable and would provide massive wildlife and other benefits. New woodlands (not plantations) with mixed trees and shrubs, managed for amenity and wildlife, preferably established through natural generation (plus planted where necessary) on the right soils in the right place would provide better long-term carbon storage than short-term commercial plantations. Grassland diversification combined with wood pasture at local or landscape scales would maintain different suites of animals and fungi.

The importance of this review is in showing that there are options for capture carbon; that carbon capture may not be immediate in some circumstances but could take time, and that there is huge potential for capturing carbon whilst at the same time enhancing and rebuilding biodiversity and supporting other ecosystem services. How this could be achieved in terms of finance and other resources is partly explored in Beechener *et al.* (2021).

Critical to the overall effectiveness of habitat restoration and creation is the scale at which it is implemented. Gregg *et al*. (2021) conclude that the most carbon uptake could be in more woodlands, plantations and saltmarsh. Restoring peatland is highlighted as critical for reducing carbon loss. They also advocate protecting and restoring natural coastal processes to allow habitats to maintain themselves and re-establish inland as sea levels rise. To meet the scale of change required, more hedgerow planting, good soil management and innovative agricultural approaches such as paludiculture are also needed.

Rye-grass dominated hay meadow restored to species-rich with much red clover

However, it is the scale of any of these actions that will be critical. Marine and brackish habitats could be restored at a far larger scale than much woodland establishment for example. Furthermore, restoring only 10% of our agricultural grasslands (which cover some 40% of the UK including semi-natural grasslands (Gregg *et al*., 2021) at 96,998 km^2), to more species-rich pastures or hay meadows incorporating red clover, whilst retaining agricultural use could be capturing similar amounts of carbon. This would also accumulate carbon more rapidly compared with the current goal for woodland/plantation establishment (30,000 ha/yr until 2050). Other habitats, created in mosaics with hotspots within a wider more biodiverse suite of habitats suited to the local edaphic and climate conditions could produce similar widescale benefits. This is exemplified by Cantarello *et al*. (2011) in a study in South West England which compared different land use strategies and their potential to create larger carbon stores. They found that greater carbon stocks would be amassed with a Rebuilding Biodiversity strategy covering 824,244 ha compared with a Forest Strategy comprising 16,000 ha of new woodlands, thus creating more benefits over a wider area for biodiversity and other ecosystem services.

Any amount of habitat creation and restoration cannot substitute the need to stop releasing GHGs to the atmosphere. Nor can it reduce the existing levels of atmospheric GHGs to safer ones. Sound policies and their rapid implementation are still imperative, but ecosystems can support and assist – and we must harness these now if we are to have a safer future.

Acknowledgements

My thanks to John Box, David Parker, Gigi Hennessy and Andreas Heinemeyer for many constructive comments and to Peter Worrall for the section on constructed wetlands.

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Appendix 1

Measuring Carbon

Detailed measurements

In order to make decisions on habitat creation, restoration and the effects of different measures, it is often useful to understand better the carbon content of soils in particular. However, soil carbon content is not easily measured (Smith *et al*., 2020, BSSS, 2021), yet a measurement/monitoring, reporting and verification system is essential both for national reporting and emissions trading.

Direct measures of SOC depend on quantification of fine and coarse mineral fractions, OC concentration (%) for example using dry combustion methods of the fine earth fraction and soil bulk density. Sampling has to take into account the large spatial variability of SOC and the variation in depth of soil profiles.
This process is costly, particularly when carried out at scale (Smith *et al*., 2019). A combination at plot scale measurements and modelling reduces these costs.

An alternative is to construct a full carbon budget by measuring net fluxes into and out of the system, this presents a point in time budget and involves numerous assumptions in the data processing, although modern equipment has improved the estimates significantly (Smith *et al.,* (2019).

New spectral methods for measuring SOC concentration and stocks are developing rapidly for direct point field measurements and for larger scale patterns and regions. The methods rely on reflectance of light on soil in the infrared region using known sample results to interpret the analyses. However, soil bulk density, needed to measure carbon stocks, cannot be readily determined using spectral methods. Smith *et al*. (2019) explore some of the developing methods using spectral, satellite and remote sensing.


IRGA analysis for soil CO_2 efflux
- Mark Nason

Non-academic measurements

Assuming lack of access to more academic and accurate equipment, there are cheap and cheerful alternatives, although the results will be indicators rather than accurate measures. The basis for these measurements is the use of loss on ignition, which involves taking a soil sample to a given depth (or several over different depths) and measuring the loss on ignition – which is a normal part of soil sample analysis. This involves measuring the fresh weight, drying the sample thoroughly and reweighing to measure the water content. The sample then needs to be combusted at about 360-450°C to burn off the organic matter, revealing the total organic content. Different sources suggest the carbon content is between 55 and 58% of the organic matter (soil guide). The Field Studies Council describes the method (https://www.field-studies-council.org/resources/16-18-geography/water-and-carbon/carbon-cycle/method/).

Improved accuracy rather than using a conversion factor would be to divide the dried sample into two and treat one half with an acid (if the soil's pH is above 7.2) and reweigh as this removes the inorganic carbon. The other half would be dry combusted using an elemental analyser which heat the sample to around 900°C and measure the CO_2 gas as a combustion by-product. The results are expressed as % of carbon. The bulk density of the soil also needs to be measured so that the final carbon stock is the product of the carbon content multiplied by the bulk density.

Some soil testing laboratories are offering soil carbon testing, giving total carbon, total OC, OM, total nitrogen, C:N ratio and total carbon stock. Internet searches show costs of around £40/sample, which would be reasonable for a small-scale project.

It is possible to convert %C in a sample to tC ha^{-1} for comparison with other data. For a hectare, the calculation is 10,000 m^2 x soil depth x bulk density x % SOC= tC ha^{-1}.

Future measurements may be revolutionised by new equipment. In the US, for example, soil probes equipped with sensors that measure carbon concentration and bulk density non-destructively and cost-effectively, are being developed (https://www.fwi.co.uk/arable/land-preparation/soils/how-to-accurately-measure-the-organic-carbon-content-of-soil).

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Chartered Institute of Ecology and Environmental Management

CIEEM
Grosvenor Court
Ampfield Hill, Ampfield
Romsey, SO51 9BD