Carbon and ecosystems: restoration and creation to capture carbon

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Declaration

I am a professional ecologist specialising in habitat creation, restoration and management. I have undertaken this literature review to inform like-minded ecologists. I trust I have interpreted the literature correctly, but any comments or improvements would be welcomed.

Summary

- This paper focuses on carbon (C) and ecosystems, identifying the contribution those especially in the UK can play in the race to capture more carbon from the atmosphere whilst at the same time addressing the biodiversity crisis and incorporating other ecosystem services.
- To reach a 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.
- C in soils is fundamental and 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 C capture, which involves complex biological and biochemical interactions, depends on the C:N ratios and varies with environmental factors.
- 40-60% of soil C is in the top 20-30cm, but this varies with vegetation and environmental features. Many studies have ignored deeper soil C stores which can be very significant.
- Organic-rich soils have over 2-3 times the level of C of other soils, followed by gleyed soils, stagnogleys and podzols. Brown calcareous earths and rendzinas have lower C contents owing to a more rapid breakdown of biomass and often low productivity. More C can be stored in clay-rich soils than sandy ones and in those in wetter, cooler climates.
- Soils supporting more stress-tolerator plants on generally infertile soils accumulate more C than do fertile soils, with rapid turnover species.
- There is believed to be a carbon saturation equilibrium in some mineral-based habitats such as grasslands, although there is contrary evidence for ancient semi-natural woodlands, active flood plains and marine environments. However, it can take a 100 years or more to reach any new equilibrium, especially from very degraded states.
- There is mostly little C in vegetation except for woody species and some marine habitats.
- Total C stocks in UK habitats depends on the extent of each, thus that in pasture may be disproportionately greater than that in woodland owing to its widespread scale rather than the intensity of C stores. 30% of the UK terrestrial C is in High Value Conservation habitats on only 20% of the land area, with more in Scotland where peat and podzolic soils are more extensive.
- C 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. Ploughed arable land also overall loses C annually.
- Carbon sequestration can be restored and increased. Although over time new woodland can accumulate more carbon than most other habitats, it usually takes 10-30 years to reach a positive carbon budget and decades for C stocks to accumulate. Other habitats show near equivalent C sequestration rates and sometimes more rapidly than can woodland. A mixture of habitats is therefore needed in the most appropriate conditions, rather than a dependency on planting woodland:
    - Small well vegetated **ponds** and small **lakes** can have high C capture rates, equivalent to some woodland;

  - **Flood-plain meadows** with full plant diversity and regular flooding can be high value C accumulators continuing for centuries;
  - Optimal conditions in **rivers** for trapping C are those with low gradients, with high channel complexity and plenty of dead wood, but C sequestration levels are not available and generally probably low;
  - Restoring **peatlands** is imperative to stop/reduce the current C losses that contribute significantly to climate change. Active peat is more difficult to achieve and sequestration levels depend on high *Sphagnum* cover and high water tables, balanced with methane production. Full restoration can take decades depending on the level of previous damage. Paludiculture has the potential to cut C emissions on agricultural peats.
  - Ancient and old growth **woodlands** can sequestrate carbon over hundreds of years with no obvious limit. Restoring and maintaining existing woodlands with minimum management will optimise the C budget over time. Creating new woods can lose more carbon than they sequester, at least for some decades. Natural colonisation, preferably on clay soils which are arable or disturbed already, with supplementary planting of 'missing' species, minimising soil and habitat disturbance or damage, using a range of native broadleaved species and minimising management would all result in the best C store over the longest period. Organic-rich soils should not be planted as these then lose more C than they accumulate.
  - **Heathland** sequestration rates can be as high as some other habitats when heather is in its building phase, but then declines, although the contribution of mosses has not been fully explored. If on organic-rich soils, these are very important for C. C sequestration rates can be double that of acidic grasslands.
  - **Neutral Grasslands** restored to high diversity and red clover (or other deep-rooted legumes) cover have been found to sequester as much C as many other habitats, but there is considerable variation. Low/intermediate grazing levels and no artificial fertilisation are essential.
  - **Marine** habitats hold more C than terrestrial ones in a smaller area. The most important are salt marshes, seagrass beds and estuarine muds, but others like maerl beds and biogenic reefs are important where they occur.
- The review encourages engagement in habitat restoration and creation for carbon and wildlife across many different habitats and situations. Climate change effects on these habitats and on their future potential for carbon sequestration need to be taken into consideration.

Introduction

The Committee on Climate Change (CCC) is an independent, statutory body established under the Climate Change Act 2008 to advise the UK and devolved governments on emissions targets and to report to Parliament on progress made in reducing greenhouse gas emissions and preparing for and adapting to the impacts of climate change. In 2019, the UK Government and the devolved administrations committed to net zero target as the CCC recommended by 2050. Net zero is when carbon emissions into and removal from the atmosphere are balanced. The longer it takes to reach this goal, the more carbon and other greenhouse gases there will be in the atmosphere and the greater impact they will have on global climate and therefore also on ecosystems and people. At the same time, therefore, it is essential to reduce the current greenhouse gases (GHG) in the atmosphere to a lower, safer level.

Of the CCC (2020) recommendations, one is that Ministers should seize opportunities from the Covid-19 crisis to build a resilient recovery that supports the transition to a net-zero economy and improve the UK's resilience to climate change impacts. One of the five investment priorities includes a landscape-scale change to increase tree cover to 17% of the land surface by 2050, (currently it is at 13%, with 3.21million ha). This target requires an additional 30,000 ha/year to be tree covered 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.

CCC reports that UK emissions in 2018 were 44% below those of 1990, mostly due to reductions in the electricity generation, waste and industrial sectors. It considers that we are on track to meet the carbon budgets up to 2022 but not subsequent ones, which in any case need to be updated from the former 80% reduction target by 2050. CCC considers we need to achieve an annual emissions target reduction of 15.5MtCO$_2$e/year for the next 30 years to meet this target.

There are, however, other challenges and opportunities that can be addressed simultaneously with multiple benefits and which, themselves, are part of the climate change challenge. The International Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) in its Global Assessment Report (2019) state that biodiversity and ecosystem functions and services are deteriorating worldwide; that direct and indirect drivers of change have accelerated during the past 50 years giving rise to a rate of change unprecedented in human history; that goals for conserving and sustainably using nature would only be achieved through transformative changes across economic, social, political and technological factors; and concluding that nature can be conserved, restored and used sustainably while other global societal goals are met only though urgent and concerted efforts to foster transformative change.

The State of Nature reports (Hayhow *et al.,* 2019) for the UK and overseas territories provide little comfort. The latest 2019 report demonstrates that, on average, the abundance and distribution of the UK's species have declined and continue to decline since 1970: a point when they had already been severely affected by centuries of persecution, pollution, habitat loss and degradation. The latest assessment suggests that although progress is being made at reversing some negative trends, the UK will not meet most of the Convention on Biological Diversity's 2020 Aichi targets. Moreover, climate change is predicted to have significant effects on biodiversity, so it is in everyone's interest to enhance the resilience of habitats, increasing their extent and quality whilst simultaneously engineering capture and storage of carbon as a contribution to reducing the impacts of climate change.

At the same time, there is increasing interest in the wider ecosystem services that habitats can provide whether related to flood control, cleaner water provision, health and well-being of those that can see or visit, or pollinators for crops, as emphasised in the most recent British Ecological Society report by Stafford *et al*, (2021). The opportunities for our profession to think outside our ecological boxes and to embrace the wider ecosystem needs are immense. We should be at the forefront of habitat creation and restoration to service all the many environmental needs and to be thinking large scale and in an integrated fashion to maximise the benefits. These are nature-based solutions (NBS) for multiple benefits. 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.

Carbon and ecosystems

Life on earth is carbon based with the carbon cycle taking carbon from the atmosphere and oceans into organisms and back out again. Ecosystems can store the carbon dioxide (CO_2) 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 rate 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 greenhouse gases, gives rise to the consequences of global warming (Deng *et al*., 2016).

Greenhouse gases include methane and nitrous oxide as well as CO_2. These are respectively 28 (although different authorities give different measures) and 298 times more potent as GHG than CO_2. Methane is a product mostly of wetland ecosystems 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 issue of the life-span of methane has recently been raised as Global Warming Potential (GWP) is usually calculated on a 100-year time frame, but methane only persists in the atmosphere for some 12 years as it degrades to water and CO_2. Allen *et al.* (2018) have suggested that this short life needs to be given more consideration in the calculation of GWP, relabelled as GWP*, which tracks the rate of change of methane emissions rather than the levels of 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.

Carbon in soils

Soils are fundamental to the carbon story. Globally there is 3-5 times more carbon stored in soils than vegetation and 2-3 times more than in the atmosphere. There are considerable differences in the amount of soil carbon stored, which depends on climate, soil texture, site management and history. Climate affects carbon accumulation through biotic processes associated with productivity of the vegetation and decomposition rates of organic matter (Deng *et al*., 2016). The soil organic matter (SOM) complex consists of plant and animal residues, in varying states of decay, and microbes, living and dead, that have fed on the residues. Roots contribute the majority of carbon in soils, thus deeper rooting plants may have the potential to produce more carbon at depth than shallow rooted species (Detheridge *et al*., 2014). Some carbon is lost in microbe respiration, whilst a proportion of the original carbon is retained in soils.

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). A stable SOM mostly consists of carbon, but also nitrogen, phosphorus and sulphur in predictable ratios that reflect the microbial origin. These are essential for microbial and plant growth and function. In general, 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 can take many decades or more to re-build high carbon levels.

Internal soil features also affect carbon accumulation. Soils with a higher biomass of ectomycorrhizal and ericoid mycorrhizal fungi (EEM[1]) produce nitrogen-degrading enzymes giving them greater access to organic nitrogen sources than arbuscular mycorrhiza (AM). This is due to their differential ability to degrade organic matter and release mineral N. (Cotrufo *et al*., 2019). The AM fungi are dependent on saprotrophic organic matter decay and mineral N production rather than producing nitrogen-degrading enzymes as EEM fungi do. As a result, Averil *et al*. (2014) show that ecosystems dominated by EEM fungi contain 70% more carbon per unit nitrogen than soils in ecosystems dominated by AM–associated plants. This is one factor determining the higher carbon levels in acidic heathy soils. Similarly, Cotrufo *et al.* (2019) found more C stocks in EEM compared with AM-dominated broadleaved and mixed forests across Europe.

Ectomycorrhizal fungi, which are very diverse (at least 6000 species of basidiomycetes, ascomycetes and zygomycetes), can account for 30% of the microbial biomass in forest soils. They characteristically form a mantle that envelops host roots and extends into the soil for resource exchange, contributing as much as 200m of hyphae/gm of dry soil. They are associated largely with woody species (shrubs and trees) forming symbiotic relationships with 80-90% of all temperate and boreal forest trees. They drive forest processes e.g. SOM decomposition, nutrient cycling and C sequestration and are believed to supress soil respiration and thus increase soil C storage.

AM fungi in the phylum Glomeromycota (which includes only some 100s of morphotypes) penetrate into the cells, forming arbuscules and have a dense hyphal network that enmeshes and crosslinks soil particles, helping in water retention and soil structure, plus providing a C sink (2-5% of soil organic carbon - SOC), which can last up to decades. They help improve nutrition and stress resistance and are associated in particular with crop plants, eg cereals, vegetables and fruit trees (Chen *et al*., 2018).

In general terms, there is a considerable amount of carbon at depth in soils, related to higher soil densities at depth. Many studies measure soil carbon to only 15 or 30cm, but total stocks of carbon in grassland soils up to 1m depth is more than three times that estimated for only 15cm depth and more than double that estimated down to 30cm (Ward *et al*., 2016). Salome *et al*., (2010), for example, give an estimate of 50% of the soil carbon in subsoils below 20-30cm and suggest that it persists out of range of decomposers rather than in a more intractable form, although physical protection of more labile fractions can also occur (Detheridge *et al*., 2014). This matches the average of 40-60% of soil C being in the top 20cm found by Cotrufo *et al*. (2019), but this varies with vegetation type with more than 50% in forests whilst the top 20cm in grasslands only accounted for some 42%. Functioning flood plain grasslands, with regular deposition of sediment in flood conditions, accumulate carbon in buried profiles over time mostly below 1m depth resulting in 34% more carbon/ha being stored in these soils in the 0-3m depth rather than 0-0.3m depth, (D'Elia *et al.,* 2017).

The amount of SOC stored in mineral soils is higher with increasing clay carbon content as particles of organic matter can become absorbed onto clay surfaces, coated with clay particles or buried inside small pores or aggregates – all of which protect it from microbial attack. In sandy soils, in contrast, microorganisms are able to access organic carbon more easily, which results in greater decomposition, although acidic conditions, especially in higher rainfall, cooler areas, also reduce microbial activity and results in deeper humic organic layers. Climate also interacts with these factors since it regulates plant production, so in drylands the potential SOC will be lower than the

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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.

same soils in wetter climates. The mineral-associated OM is stuck to soil particles and tends to remain so for long periods contributing to a 10-1,000 year lifetime slow cycle. Particulate OM (POM) is part of a faster cycle of 1-50 years (Cortrufo *et al*., 2019).

The plant community also affects the amount of carbon that reaches the mineral layers. For example, broadleaved trees have a greater capacity to transfer carbon to deeper soil layers and stabilise it in the mineral layer than conifers. This could be related to the higher root mass of broadleaved trees as root litter contributes as much or more carbon to the soil than foliar litterfall, (Crane, 2020). Henneron *et al*., (2019) show that in grasslands at least, more competitive, acquisitive plants 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) are associated with slow decomposition promoting high SOC sequestration. Thus temperate grasslands dominated by acquisitive (i.e. competitive) species are associated with lower SOC storage than those dominated by conservative species (i.e. on infertile soils or stressed growing conditions). The mechanisms for this are not fully understood but might be related 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.

Improving SOM through C sequestration also has the potential to improve soil health significantly in terms of nutrient availability and water-holding capacity. The allocation of photosynthate-C to soil via living roots through rhizodeposition and associated rhizosphere processes is now widely recognised as major components of terrestrial ecosystem C cycling (Henneron *et al*., 2019). Root exudation and death lead to formation of new root-derived SOC and root-derived carbon is generally considered to make the largest overall contribution to total soil aggregate-associated carbon (Rees *et al.,* 2018).

Organic C in soils is stored in many different chemical compounds, many of which contain nitrogen (N) and/or are formed through microbial activity that needs N. Indeed, SOM requires more N per unit of C than plant biomass does. C accumulation, persistence and response to N availability is better considered by separating particulate and mineral-associated organic matter (POM and MAOM). The POM is largely of plant origin and contains structural C compounds with a low N content that persist in soils. MAOM is essentially made of microbial products richer in N and persist in soils through chemical bonding to minerals and physical protection in small aggregates. This makes MAOM less vulnerable to disturbance with slower cycling (Cotrufo *et al*., 2019).

Organic-rich mor humus over loess in the uplands

Storage of C in MAOM dominates in soils with relatively low C content, but MAOM can saturate, reducing or stopping further accumulation of C. Additional C stock is then dependent on more POM accrual, which is greater under a forest than grassland cover. 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).

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). Stored carbon is greatest in this mostly undecomposed material at 259/576tC/ha (Alonso *et al*., 2012, Cantorello, 2001), to 0.5m/1m depth respectively, and much greater for deeper peat - 5248tC/ha on Dartmoor for peat 6.63m 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 (112.2 and 76.67kt/km^2 respectively), with gleyed soils, stagnogleys and podzols in the 20-44 kt/km^2 range. As podzols are extensive, they hold about 10% of all UK soil carbon at 175-211tC/ha (Alonso *et al*., 2012).

A comparison of carbon content found in soils (mostly only to 30cm depth) beneath different UK type habitats (Table 1) illustrates these differences. Although humic-alluvial gley soils top the list, the carbon stock will depend on the nature of their vegetation and its management and is likely to be lower than peat soils where these are deep and mire-covered. The potential importance of flood meadows for soil carbon is clear as it lies in the top three but the levels will depend on active flood management, where sediment is deposited and in the presence of good quality flood plain vegetation. The list also highlights the importance of heathland, salt marsh and woodland soils, but many levels will be much higher when the full profile is taken into account. Some will 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. Scores for brown calcareous earths and rendzinas are quite low owing to the more rapid breakdown of biomass, shallower soils and often low productivity.

Table 1 Carbon content in some soils and vegetation

Soils or habitat	**Carbon in soil (tC/ha)**	**Carbon in vegetation (tC/ha)**
Humic-alluvial gley soils	438	Not appropriate
Peatland	259 to 0.5m, 576 to 1m, 5,248 to 6.3m, raised bog 1,620 to 3.8m	2
Floodplain grasslands	286-354 (1 to 3m)	Not available
Podsols under heath	175-211	2
Seagrass	6.65-194	0.5-2.52
Saltmarsh	29-93 0.1-0.3m,	8.32
Broadleaved mixed wood	108-173, 255-354 to 1m	81-251 depending on age
Acid grassland	87	1
Heath lowland and upland	81-103	2-9
Bracken	55-77	Not available
Fen, marsh, swamp	76, 810-2530 on 3.8m peat	Not available
Conifer plantation	73-120	59-94, average 75, depends on age, density, yield and climate
National average all wood types	62-66	57
Calcareous grassland	51-69 to 0.15m	Not available
Neutral grassland	60 to 0.15m	1

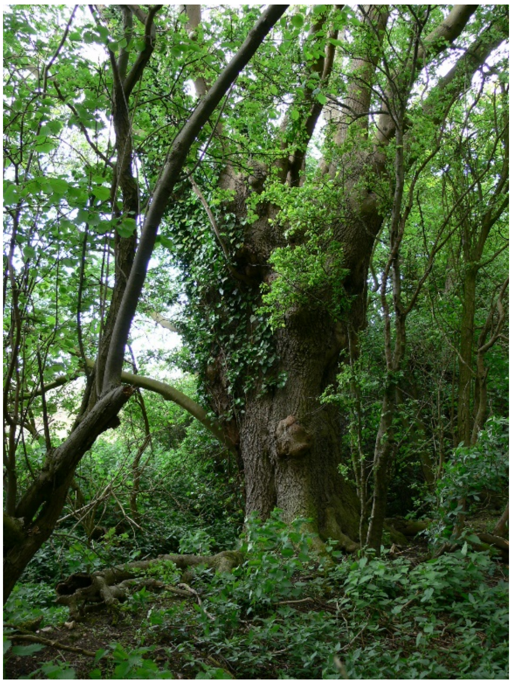
Agriculturally improved grassland	59-61, 72-204 to 1m	1
Arable	43-64	1-2.36
Hedges untrimmed for 3 years	98.7	45.8
Ponds	16-28	Not available
Rivers	0.2-4.8	Not available

NB soil depths of measurements differ between projects, most are only to 0.3m. Data from Alonso *et al.,* 2012, Gregg *et al*., 2021, Axe *et al.,* 2017 & Milne & Brown, 1997 for soils and see text.

However, the importance of the carbon stock lies in the ecosystem's ability to retain and increase it. Some of the carbon stocks in soils listed in Table 1 are being lost or are very vulnerable to loss as explained below.

Carbon in vegetation

Table 1 shows how little carbon is in most vegetation 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 20-70% 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, 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 will vary with species and management).



Woodland with multiple aged trees and shrubs

National total carbon stocks

The total carbon stock on a UK scale relates to the area of each habitat, thus pasture, 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 2). 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, itself a high-density carbon ecosystem that is also extensive, especially in Scotland. Note not all habitats are listed and that carbon will be lost from some as well annually. It is the soils that will be most important.

Table 2 Total area and carbon store in vegetation and soils to 30cm in High Value Habitats in the UK

High Value Habitat in UK	Total 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)

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. Where this is large scale and severe as in many of the UK peatlands, then losses can be equally severe. Peatlands that are drained and used for arable agriculture as in the Cambridgeshire Fens and Lancashire Mosses lose the most carbon. Evans *et al*., (2016), for example, found up to 30tCO_2e/ha/yr lost at the most deeply drained sites; a figure that is elevated to 38.98tCO_2e/ha/yr in the 2021 UK GHG Inventory (Gregg *et al.,* 2021). Evans *et al*. (2016) noted that for every 10cm lowering of water table, CO_2 emissions increased by around 4tCO_2e/ha/yr. The scale of these figures is supported by Artz *et al*., (2013) for peatland drained for forestry (20tCO_2-e/ha/yr carbon loss) and losses from intensive grassland grown on drained and dried out peat are similar in scale at 27.54tCO_2e/ha/yr (Gregg *et al.* 2021, who also give some updated details on losses under different land management on peat). Drained modified blanket bog loses 3.54 and eroding, modified bog 12.17tCO_2e/ha/yr in comparison respectively (Gregg *et al*., 2021). Many of these figures are significantly higher than the sequestration rates of any habitat (Table 4).

Cultivating peat soils in the Cambridgeshire fens

Loss of carbon in peat translates into subsidence of 0.38 to 0.86cm/yr, and a restricted life-time for some peatland soils (exacerbated by losses to wind erosion 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). 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 >10tCO_2e/ha/yr at the most waterlogged sites owing to methane emissions.

On mineral-based arable land, ploughing or other disturbance results in C loss, 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, then carbon losses can be severe. The annual cycle on arable soils when ploughed results in release of more carbon than is stored in the order of 0.14tC/ha/year (0.51tCO_2e/ha/yr), thus contributing to increased CO_2 levels in the atmosphere – and this is not counting the carbon embedded in arable agriculture in machinery use and agri-chemicals.

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 3.

One last and critical factor is climate change itself. Drought can result in increased carbon emissions. 60% of carbon in wet heathland soils were lost in just two months of induced drought as soils became aerobic (Gorissen *et al*., 2004) and overall carbon sequestration figures were lower after periods of higher temperatures (with no water deficits) when net soil respiration rates were elevated in Wytham Woods (Thomas *et al*., 2011). Barraclough *et al*., (2015) consider that climate change may influence soil C indirectly through changing vegetation cover and litter quality. In contrast, neutral grasslands increased their resistance to drought through restoring diversity, possibly owing to reduced water demand and lower productivity of a more diverse sward (Gregg *et al.,* 2021). As droughts of varying lengths are becoming more regular, and as temperatures rise, these will affect many different habitats, more particularly in lowland Britain.

Table 3. Examples of carbon loss after habitat damage or change

Change or damage to habitats	**Carbon loss $tCO_2e/ha/yr$**
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, derived from habitat loss to arable or improved grassland as well as development, overgrazing (which reduces litter and root production), peat decay and erosion, soil erosion, ploughing or other soil disturbance. Soil carbon losses of 0.6-2% per year in England and Wales have been calculated (Bellamy *et al*., 2005) although this is questioned as it uses derived rather than measured bulk density parameters, the variability of which can have significant effects on carbon density, (Detheridge *et al*., 2014).

Restoring carbon, carbon sequestration - where best to focus

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 approach everywhere? Cotrufo *et al*. (2019) stress how implementing soil C 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) urged 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 GHG in the atmosphere dictates sequestration projects that have the most immediate, largest and the long-lasting effects. All these requirements in one ecosystem will be 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. This immediacy is compromised by focusing on woodland planting since these are mostly not instant carbon capture machines and generally take 10 to 30 years to become a significant positive carbon sink. Thus, although they are regarded as potentially the largest carbon sinks and stores in the UK and can provide the best opportunities for removing GHG over time, this is not realised for some decades while they develop their canopy and root mat and compensate for the sometimes high levels of carbon lost in their establishment, (Gregg *et al.,* 2021). Moreover, 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.

Critical to the discussion is the idea already set out that carbon sequestration reaches a balance or equilibrium whereby the amount of carbon absorbed equals that emitted from an ecosystem (Gregg *et al*., 2021). This does not apply to organic soils and, provided the site is functioning properly and the climate remains suitable, carbon can accumulate over time in these environments for millennia. The same principle applies where sediments rich in C also accumulate, as in marine environments like mudflats and salt marshes where continual sediment trapping is possible. Functional flood plains, where repeated flooding brings in more sediment, could also accumulate carbon for centuries. In comparison, most other habitats on mineral soils are deemed to have a threshold beyond which more carbon is not added significantly to the ecosystem. Measures for carbon in planted forests, for example, are predicated on the system reaching an 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.

Gregg *et al.* (2021) present a conceptual model of habitat carbon stock equilibrium disturbed by land-use change 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 disturbances related to climate or 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 and store atmospheric carbon, (Cannell, 1999).

The evidence for each major habitat type is now explored in terms of potential carbon stock and sequestration rates over time so that the advantages or disadvantages of different approaches can be gauged. Agricultural practice is generally not covered here, including livestock impacts on the carbon cycle, but Gregg *et al*. (2021) provide some information on these aspects. It should be noted that there are considerable variations in the data in terms of number of samples, range of habitat, climates and soils and the fact that some measures are modelled, 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 nitrogen dioxide as GHG. Some figures represent the first steps in exploring a particular treatment or habitat creation measure, so warrant further investigation, while others represent a single point rather than changes over time. Repeated measures in some studies relate to specific soil depths rather than any changes in soil horizons (which can increase with more organic matter), which could also 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 should be treated as indicative and part of a range.

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) estimate a near doubling of the potential carbon sequestration just in peatlands and heathlands if they were in good condition, equivalent to nearly 32% of the annual emissions from agriculture. However, habitat restoration alone would not be enough to reduce CO_2 levels to a safe level nor reverse the biodiversity crisis. Thus, new habitats are also needed on a large, interconnected scale. Table 4 provides comparative carbon sequestration rates for a selection of habitats or habitat change scenarios extracted from the sources given.

Table 4 Indicative carbon sequestration rates for new and existing ecosystems.

Habitat: soils and vegetation	**Carbon exchange, tCO_2ha/year**	**tC captured/ ha/year**	**Comments/sources**
Restored or created habitats			
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.,* 2010, 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 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	NE*,* 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	NE*,* 2012, higher if from arable
Restore flower-rich grassland from	3.8-6.96*	1.04-1.89	Alonso *et al*., 2012, (but includes whole C cycle), NE, 2012

improved sward or arable			
Creation of intertidal/saline habitat from arable or grassland	3.8/2.9*	1.03/0.79	NE, 2012
Arable to heathland, 1-100yrs	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
Create successional scrub on pasture	1.8*	0.5	NE, 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	NE, 2012
Restore peatlands	0.88-6.93*	0.24-1.89	Svenson, Artz *et al.,*
Existing habitats			
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	Cannel 1999 2-7tC/ha/yr 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 5
Saltmarsh	2.35-23.83	0.64-6.5*	Burrows *et al.,* 2014, Beaumont *et al.,* 2014, average 1.2-1.5tC/ha/yr
Heathland	3.34-12.65	0.91-3.45*	Alonso et al, 2012, Quin et al, 2015
Flood plain	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	0.2-3.7*	0.05-1.01	Artz *et al.*, 2013

*Author's measures. Figures are converted to tC/ha/yr or to tCO_2/ha/yr for comparison, but may not include all GHG. NA – not appropriate to convert without details on methane.

Wetlands

Ponds and lakes

Wetlands can capture significant amounts of OC and accumulate it in bottom sediments where it is stored rather than subjected to decay and loss in respiration as in terrestrial systems. The papers 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) have shown that small ponds are particularly effective at trapping OC, with levels as high as 247gm OC/m^2/yr (equivalent to 2.47tC/ha/yr), but averaging 142gm OC/m2/yr 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 1.49tC/ha/yr. Taylor *et al.* (2014) found OC started to accumulate after 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: 2.3gm CO_2-e/m^2/yr, 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. This still demonstrates a net capture of OC in the sediments.


A well vegetated pond captures more organic carbon

Similarly good rates of OC capture were found in lake sediments in Minnesota, with averages of 25-70gm OC/m^2/yr (0.25-0.7tOC/ha/yr), (Anderson *et al.,* 2013), and these match average carbon entrapment in European lakes at 0.5tC/ha/yr. Double these levels are attained in eutrophicated lakes. Gregg *et al*. (2021) separate out the English lakes from a later study by Anderson *et al*. (2020), which were all classified as eutrophicated and noted burial rates of 0.82-2.97tC/ha/yr, 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 C fixation in the water. Average rates as high as 2,122gm OC/m^2/yr (21.22tOC/ha/yr) were found across a wide variety of lakes and impoundments in Iowa, but with an anomalously massive 17,392gmOC/m^2/yr (173tOC/ha/yr) in one small farm pond (sizes not given).

Creating new ponds rather than restoring old ones avoids losing the existing carbon content and should not be so shallow as to dry out annually as this precipitates loss of the stored carbon (Gilbert *et al*., 2016). Although ponds punch are good for carbon capture, they are small within the landscape context, although groups of them can support more biodiversity and contribute to clean water and to flood control when streamside. Ponds with high organic load and low oxygen levels favour methane emissions and could be net emitters not absorbers of carbon (Gregg, *et al*., 2021). 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.

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 C through high rates of organic matter 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 was found to range between 219 to 267gC/m^2/year (equivalent to 2.19 to 2.67tC/ha/yr). This compared to flow-through temperate wetlands which had much shorter hydraulic retention times and C sequestration rates of 124 to 160gC/m^2/yr (1.23-1.60tC/ha/yr).

In the design of ICWs and overland flow wetlands, a key design 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 C and the retention of phosphorus is likely to be associated with the capacity to sequestrate C 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 C 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 sequestrate carbon.

Flood plains

Flood plain 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 0.92tC/ha/yr, with some variation between rivers and within catchments, ranging between nearly 2.0tC/ha/yr and less than 0.5tC/ha/yr, (Walling *et al*., 2006). Similar levels have been found in other studies in Austria (Zehetner *et al*., 2009). There was no analysis of the differences in vegetation between sample sites in the English study, but the Flood Plain 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 flood plain (unpublished data, 2020 newsletter). This would be consistent with the evidence for the superior quality of diverse grasslands being able to sequester more carbon than homogenous grassland. Their analyses show very high levels of carbon stored in the top 10cm of soil, (including levels of 109.4tC stock in the top 10cm of soils under Cricklade NNR floodplain - quoted by Gregg *et al.* 2021) and significant carbon store down to 1m depth of restored flood plain grasslands; much higher than in other habitats.

This experience is mirrored in America by D'Elia *et al*. (2017), who show carbon stocks to be present to great depths in buried horizons (to 3m) produced by regular flooding, which were larger than in forests, pasture or wetlands at 286tC/ha, but Cierjacks *et al.* (2010) found much higher levels of 354tC/ha in just the upper 1m on an Austrian floodplain. There was, 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 flood plain habitat type, which can be very varied. This variability is reflected in the measurements available – 2.9tC/ha/yr along the Danube (Sutfin *et al*, 2016) to 0.5 to 2tC/ha/yr in South/South-west England (Walling *et al*., 2006).

Thus flood plains that accommodate regular sediment input from bank overtopping 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 would also be sequestrating more carbon than other swards. Zehetner *et al*. (2009) suggest a timeline of some 100 years of accumulation rates of 1tC/ha/yr, reaching levels exceeding those in forests, followed by a levelling off over 300-500 years subsequently to 0.08-0.18tC/ha/yr 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. This research also showed that cultivation can annihilate this carbon sequestration potential.

Beechener *et al*. (2021) note that flood plains 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 flood plain habitats like fens, marshes and bogs have been reduced to only 0.5% of the English flood plain area, giving very significant opportunities for restoration with the 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 (EA pers com.), 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 in terms of 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 DOC from surface and subsurface waters (Sutfin *et al.*, 2016). The C 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 C stored in the other elements of river systems listed (the store is around 0.2-4.8tC/ha). 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.

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 0.01-0.03tC/ha/yr (summarised in Gregg *et al.,* 2021).

Other wetland habitats

Reedbeds are known carbon sinks (Gregg *et al*., 2021), although ditches within the system can release methane. There is little evidence available on carbon sequestration rates, although Gregg *et al*., (2021) quote a Somerset County Council report of between 5 and 20tC/ha/yr being possible in the Somerset Levels, but the details were unavailable.

Peatlands

Peatland restoration stands out as the most effective means of stopping the loss of C (much of which has accumulated since the Atlantic Period, 7,500 years ago) in dissolved organic carbon, particulate organic carbon 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. Peatland restoration is especially important on modified bogs as they are extensive, but major benefits could be gained from stopping the very high carbon loss from drained agricultural peats, (Table 3). New forms 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 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, and 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. If dominated by cottongrasses (*Eriophorum*), these are correlated with an increased methane flux, whilst a high *Sphagnum* coverage often shows lower levels owing to a symbiotic relationship with methanotrophic bacteria. A high water-table both encourages more *Sphagnum* cover and produces anaerobic conditions which suppresses decomposition of OM.

Gully blocking on blanket bog raising the water table

Net CO_2 sink rates of peatlands vary from 0.2-0.71t CO_2e/ha/yr on Moor House (modified blanket peat), to 3.7t CO_2-e/ha/yr on a relatively undamaged raised bog at Auchencorth Moss in Scotland. Artz *et al.* (2013) calculate a net sequestration rate of 0.76tCO_2e/ha/yr on ombrotrophic peatland inclusive of methane and carbon fluxes. Gregg *et al*. (2021) give overall sequestration rates of 0.02tCO_2e/ha/yr for rewetted, semi-natural modified bog and for near natural bog in some updated figures which take losses in dissolved organic carbon and methane into account. Measurements show that carbon capture is not necessarily constant and can represent a loss in particularly dry year. However at Glencar in relatively good condition, measurements showed carbon sequestration in 4 out of 6 years (Artz *et al*., 2013).

The only communities that were capturing new carbon on a restored lowland raised mire in Ireland were those where *Sphagnum* was abundant, whilst *Calluna* and marginal drier areas were still a carbon source rather than sink owing to low water tables despite restoration measures being implemented (Svenson *et al*., 2018). Conversion from a carbon source to a sink is dependent on the water table being about 0.1m from the surface, with a low slope and no marginal drainage.

Achieving this on severely degraded and gullied blanket bog 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. Quicker and more effective results can be obtained from restoring less damaged peat than severely eroded sites. Evans *et al*., (2016) found that by maintaining water tables to 0-0.1m below the peat surface on raised bogs, overall GHG emissions (including methane and CO_2 but not nitrogen) could be neutral.

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 1.46tCO_2e/ha/yr for restoring modified to near natural bog, to 21.30tCO_2e/ha/yr for restoring actively eroding bog to a modified condition. The figures given do not separate raised bogs from blanket bogs even though there are significant differences in their bulk density, hydrology and management. This warrants further research.

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*. (2016) 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 10.31tCO_2e/ha/yr in the Norfolk Broads to a loss of 4.88 tCO_2e/ha/yr on a fen converted to grassland from arable use. Methane was not detected on sites where the water table was below 25cm. Conservation-managed fens are shown to be amongst the most effective carbon sinks.

Woodland

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 undergrowth 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 necessarily representative of semi-natural woodlands with a range of tree ages and shrub and ground flora layers.

Several studies show that new woodland mostly does not become carbon positive for 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 GHG 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 as suggested at the beginning of this section. 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 continue to accrue carbon in the wood.

Therefore, provided change in these 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 ancient forests in the soils in particular.

The carbon sequestration rates of ancient or long-established broadleaved woodland vary mostly between about 1.3 and 4.9tC/ha/yr in a range of broadleaved temperate old forests across temperate Europe and America, 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 3.6tC/ha/yr). These measures are within the range provided for plantation forests, thus demonstrating that semi-natural woodland can be as useful for carbon sequestration whilst simultaneously producing a superior wildlife habitat and potentially capturing carbon for much longer.

Carbon sequestration is less likely to continue for centuries in some plantation woodland. Xiong *et al*. (2020) found, for example, 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. This was related to the differences in litter quality and C:N ratios. The type of litter and its decay characteristics rather than its biomass is correlated with SOC. Dewar & Cannell (1992) use time-averaged storage of carbon at equilibrium in their model, based on a balance between carbon gains and losses, showing the carrying capacity for carbon capture of plantation trees.

There are other implications associated with elevated CO_2 levels in the atmosphere. Brienen *et al*. (2020) demonstrate that these, 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). These factors will need to be taken into consideration when creating woodland for carbon sequestration.

It is clear that the priority 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.

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.

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) 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 dissolved organic carbon in runoff. Furthermore, even on heathland soils, replacing heathland with trees can result in a negative carbon outcome for at least decades. Comparison of hairy birch (*Betula pubescens*) and Scots pine (*Pinus sylvestris*) stands after 12 and 39 years and nearby heather (*Calluna vulgaris*) 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 Mile's (1981) conclusion that birch colonising heathland can change a mor to a mull humus in about 20 years owing to deeper rooting 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.

Woodland establishment on high clay-content soils are 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 (Laganière *et al*., 2010). In contrast, 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.

The site vegetation and its relationship with the soils prior to woodland establishment is also critical. Natural regeneration on bare arable fields or already degraded sites should be the best starting point since the carbon content here would be low at the outset and less would be lost on woodland colonisation. 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 English oak (*Quercus robur*) and one limed and forming an ash (*Fraxinus excelsior*) / sycamore (*Acer pseudoplatanus*) woodland, showed carbon net accumulation rates of 2tC/ha/yr and 3.39tC/ha/yr 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 is 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. 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 averaged 5.6% loss of carbon, gaining 6.1% in 10–30-year-old stands and 18.6% in plantations over 30 years old. 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. There are variations though. Vesterdal *et al*. (2002) found no SOC increase in new woodland composed of Norway Spruce *Picea abies* and English 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 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.

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, compared with adjacent permanent pasture in Aberdeenshire. The soils (sampled to 50cm 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. They found that 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 silvopasture treatment for all the species, there was still more C in the woodland on a per hectare basis.

A meta-analysis of data on changes in soil carbon stock concluded that it declines if landuse 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 (<1200mm/yr), the conversion had little effect, but was greater (minus 23% soil carbon) in high rainfall areas (>1500mm/yr). 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 20cm 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.

If natural regeneration is unsuitable, selecting appropriate species and woodland design are the next choices to be made. A woodland for wildlife should have a varied structure, with understory and canopy trees, a representative ground flora and be varied in age and patterns to provide the optimum opportunities for a range of other plants, animals and fungi. There is a place for dense plantations and economic forestry to provide timber, but commercial plantations are very inferior for wildlife (Crane, 2020). In the UK, Sitka spruce accounts for around 51% of the conifer area, followed by Scots pine and larches (reported in Crane, 2020). There are pros and cons for planting conifer plantations for carbon too. 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. There are concerns that maintaining large areas of coniferous plantations under climate change 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).

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 and to produce timber, but as long-term carbon stores are required over 100-year timescales, then planting 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 growing 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.

The carbon budgets for these alternatives 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. Dewar and Cannell (1992) give modelled uptake levels for a range of plantation species, (Table 5), which demonstrates that uptake and storage depend on planting density, yield classes (growth of wood per year), harvesting age and whether thinned or not. These figures exclude ground cover (which will be minimal in dense plantations) and any losses of carbon in establishment or from disturbance to the preceding habitat and soils and are averaged over the rotation length. Total carbon includes that in trees, wood products, litter and soils, but based on several assumptions that need further research. The figures suggest not dissimilar carbon totals over the whole rotation cycle for different species.

Table 5 Some plantation carbon capture figures, based on single species plots (adapted from Dewar and Cannell, 1992), averaged over the rotation cycle - Sitka spruce 55 years, beech 92 years.

Species	**Planting spacing**	**Average carbon sequestration rates tC/ha/yr**	**Total carbon tC/ha time averaged at equilibrium**
Sitka spruce Yield class 24 unthinned	2m	5.6	254
Sitka spruce Yield class 16 unthinned	2m	4.5	229
Sitka spruce Yield class 12 unthinned	2m	3.7	198
Sitka spruce Yield class 8 unthinned	2m	2.9	169
Sitka spruce Yield class 24 thinned	2m	4.4	211
Sitka spruce Yield class 16 thinned	2m	3.6	192
Sitka spruce Yield class 12 thinned	2m	3.0	167
Populus Yield class 12 unthinned, 26yr rotation	2.7m	7.3	212
Nothofagus Yield class 16 thinned	1.7m	4.6	179
Sitka spruce Yield class 12 thinned	3m	3.0	170
Scots pine Yield class 10 thinned	1.8m	2.7	178
Lodgepole pine Yield class 8 thinned	1.8m	2.5	155
Beech Yield class 6	1.2m	2.4	200
Oak Yield class 4	1.2m	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 figures compare well with those given by Thomas *et al.* (2011) for ancient woodland, although these vary between years with prevailing weather conditions and are measurements of carbon fluxes rather than averaged models based on growth rates. Comparison with other studies in broad-leaved woodland with similarly little management but using the same eddy covariance methodology suggests rates of 3.8tC/ha/yr at Alice Holt (but this broadleaved plot would have been managed previously), 4.9tC/ha/yr in a 250 year-old beech-dominated deciduous wood in Germany and 2tC/ha/year in a 450 year old diverse temperate forest in China. 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 4.

The peak uptake rates of CO_2 for commercial forests are generally 5-20tCO_2/ha/yr (1.36-5.45tC/ha/yr) for tree crops, excluding any field or shrub layer, which are favourable levels compared with other habitats on Table 4 (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.

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 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 4.3tC/ha/yr instead of 5.6 for unthinned stands. The difference is greater in higher yield class stands (Table 5, Dewar and Cannell, 1992).

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 its release of carbon, the extent of site disturbance in terms of ploughing, drainage and fertilisation for planting the next crop and the rate if its growth subsequently. There are few direct measurements for some of these parameters although they have been estimated and modelled for a number of 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) 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 5.7tCO_2e/ha/yr (1.55tC/ha/yr) and 4.1tCO_2e/ha/yr (1.12tC/ha/yr) 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 4 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.

The outcome of these considerations strongly supports conservation of existing ancient woodland – they should still be sequestering carbon at a good rate. The next best action is to facilitate natural regeneration of woodlands on clay soils as a priority that have been arable or are bare/degraded at the start of the scheme. Natural regeneration may be limited in terms of local desirable species that can colonise any site, so 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 and to ensure a range of complementary carbon sequestration rates (e.g. Lewis *et al.,* 2019, Laganière *et al*., 2010, Crane, 2020). Over time this would deliver the largest carbon stocks for longer, although at a slower rate than in the most productive short-lived plantations. Cotrofu *et al*., (2019) also recommends the use of trees that have EEM-associated fungi, which excludes sycamore, ash and lime, to maximise carbon accrual. However, consideration of the time to become a carbon sink overall rather than a carbon source is needed for planting new woodland as this defers its potential role, possibly for decades, in mitigating atmospheric CO_2 levels.

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), although it could be highly selective and only partial in any coupe and could be important for other reasons. It could also be a way of diversifying new woodlands with even aged trees. Natural events like gales and disease could well deliver this over time in any case.

The research to date shows too that establishing woodlands for carbon objectives is not to be recommended on organic soils and that heathlands and acid grasslands on these would be better habitats than woodland for carbon capture. Any new planting needs to minimise soil and ground disturbance, with no drainage, ploughing, fertilisation or herbiciding and minimal access routes into the site. Overall, though, more research is needed to provide detailed guidance into different methods of establishment, different woodland types (excluding commercial plantations) and including other vegetation (shrubs and a diverse ground flora for example) in order to optimise carbon capture.

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

Wood pastures

There are many woody habitats other than woodland which could contribute to C sequestration, 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 e*t al*., (2021), but optimum density or numbers of trees and shrubs is not clear.

Hedges

Similarly, hedges with trees and shrubs have been little researched for their carbon content and capture capability in relation to management and diversity. Unlike other woody habitats, hedges are usually regularly managed, thus removing some of the biomass through trimming or laying. The best carbon stores will be in hedges that are wide and tall. Hedges untrimmed for three years at 3.5m high and 2.6-4.2m wide can hold 42tC/ha in their above ground biomass, whilst those minimally managed stored 45.8tC/ha. This reduces to 40.6 and 32.2tC/ha if trimmed to 2.7m and 1.9m respectively (Axe, 2015). There is little data to compare species-rich with species-poor hedges, although bramble (*Rubus fruticosus*) invasion is suggested to increase C content (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) report stocks to 30cm of 98.7tC/ha. One study shows significantly more SOC under mixed compared with species-poor hedges at 175.9tC/ha to 1m depth (in Gregg *et al*., 2021). There is also some evidence (although not consistently across different situations) that hedgerows exert an influence beyond their footprint of some 2m, although soil carbon levels reduce rapidly after 2.2-3.4m (in Gregg, *et al*., 2021).

Varied carbon fluxes of hedges are reported. 1tC/ha/yr sequestration for shrubby hedges established on arable land and 1.6tC/ha for the addition of hedgerow trees in the Environmental Stewardship Scheme is suggested but other figures varying from 0.46tC/ha/yr to 12.19tC/ha/yr from different studies show 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 as shown by Ford *et al*., (2021) who show in wet years on seasonally wet soils, positive sequestration rates modelled at 6-10CO$_2$e/ha/yr switched to a net source under droughted conditions to 5.8tCO$_2$e/ha/yr (-1.6 to -2.73tC/ha/yr changing to +1.58tC/ha/yr).

Hedgerows in arable land can have the added benefit of trapping eroding sediment holding carbon which would otherwise be lost. Thus hedges contouring across arable landscapes can have an important carbon trapping function (Gregg *et al.*, 2021).

Scrub

Scrub (excluding dwarf shrubs and young or regrowing trees) is an important habitat and can be invasive or slow to spread. Scrub can be valuable for C store and accumulation (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 C 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 C 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 scrub developing on dry soils or in drier climates sequestrate more C than those on wetter soils or in wetter climates. Scrub development will result in C trapped in the woody vegetation, but overall outcomes would depend on amounts lost or gained in the soils. More research is needed to provide better guidance.

Heathlands

The total carbon stock in heathlands consists of about 98% in the belowground reserves and the rest in the above ground vegetation (Table 1 shows a general average of about 90tC/ha, although higher levels of 103tC/ha have been recorded in South West England with 7.11tc/ha 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. 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. Field *et al*. (2017) also show that low nitrogen additions (as from air pollution) 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*. (2008), 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.

Heathland sequestration rates in the building phase can be as high as 3.45tC/ha/yr (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 makes an important contribution to carbon capture targets. Alonso *et al*. (2012) suggest a figure of 3.32tCO_2e/ha/yr (0.91tC/ha/yr) averaged for 1-100 years for restoring heathland from arable, but this includes all the carbon losses and gained 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, calculates that afforestation of the East Anglian heaths resulted in soil carbon losses of around 0.6tCO_2e/ha/yr 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 C levels of acid grassland (Quin *et al.*, 2015). These changes could take up to about 10 years depending on the starting 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. 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 some nature conservation objectives and maximising heathland carbon in order to prevent natural succession to woodland and the loss of specialist and valued species (Gregg *et al*., 2021).

Heather recolonising acid grassland to restore heathland in Dorset

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 13-119tC/ha across 49 sites, and the average in Table 1 is taken from carbon guidance for Lake District land managers, (source not given).

Grasslands

As in other habitats, most carbon in grasslands is in the soils. Most of the research focuses on neutral grasslands and there has been little comparison between grasslands on different soil types, such as those that are clay-rich or not. A wide range of carbon stocks might be expected owing to this variation. The Countryside Survey 1978-2007 found carbon stocks varying from an average of 60tC/ha in neutral to 87tC/ha in acid grasslands, as quoted by Gregg *et al*. (2021), 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 446tC/ha of total carbon stock (ie organic and inorganic) in a range of soils to 1m depth. This contrasts with levels or 58 to 100tC/ha in Northern upland meadows on soils 0.15cm deep, (Eze et al., 2018).

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 maintain some agricultural output simultaneously, work at a larger scale and respond to change more rapidly, provided there is not a lag effect from past damaging land use change such as drainage. A global review by Deng *et al*. (2016), for example, showed that soil stocks significantly increased on average across all land use changes examined after conversions from arable farmland to grassland (0.3tC/ha/yr) and forest to grassland (0.68tC/ha/yr), although Warner *et al*. (2020) taking a lifecycle assessment approach, suggests a higher level of 1.59tCO_2e/ha/yr (0.45tC/ha/yr) for arable revision to low-input grassland.

Neutral grasslands

On neutral grasslands, a diverse sward with good red clover (*Trifolium pratense*) cover was found to offer the greatest carbon capture potential, equivalent to many woodland figures. However, there are also a range of lesser alternatives. A grass monoculture will capture more carbon which is more stable than a regularly ploughed arable field and this accumulates if the grassland is permanent rather than a ley. Grasslands treated with farmyard manure will be higher in carbon than those receiving inorganic fertilisers, (the production of which is also carbon-intensive). Rushy pastures may not be desirable for carbon sequestration as rushes produce more root exudate that decays rapidly and loses carbon (Alonso *et al*., 2012). Restoring diverse wet grassland communities would have better carbon-capture characteristics.

Flower-rich grassland is significantly superior to species-poor swards for carbon capture. Semi-improved grasslands contain around 15-20% more soil carbon than improved grasslands just in the top 15cm of soil (Norton *et al.,* 2021). Long-term experiments at Colt Park meadows (Ingleborough NNR) on moderately fertile brown earths over limestone showed elevated sequestration rates after ceasing inorganic fertiliser application, which increased rates of carbon and nitrogen accumulation despite an associated reduction in biomass above and below ground (De Deyn *et al.*, 2010). This accompanied an increase in plant diversity. Further increases in plant diversity did not alone increase carbon and nitrogen accumulation, but adding red clover at high seeding levels (5.2gm/m^2) to plots with added other species and no fertilisation had a significantly larger effect and importantly, within two years of the treatments, with 3.17tC/ha/yr 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 and changes over time.

Grassland creation with high red clover cover

This C 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*., 2010) but lower than the 18.22tCO$_2$e/ha/yr (on a 25 year old upland hay meadow) or 27.58tCO$_2$e/ha/yr on a 150 year old upland hay meadow (4.97 to 7.52tC/ha/yr) 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 the Colt Park plots with continuing inorganic fertiliser application and no red clover or other species additions, which overall lost 0.08tC/ha/yr owing to faster soil respiration rates and rapid breakdown of litter. Moreover, the rate of C 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.

These experiments demonstrate that diversifying grasslands to trap more C depend 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. 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,

Grassland management intensity affects soil carbon levels, with total soil carbon decreasing with increasing intensity of management, mostly in the upper 7.5cm of soil. However, effects are still significant at 40cm depth and more weakly at 60cm (where soils are sufficiently deep) and 60% of grassland carbon lies below 30cm 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, not its quantity (just as in woodlands) that is more relevant to carbon capture as these help dictate decay rates. Ward *et al.* (2016) found that intermediate levels of management involving a single hay cut and intermediate grazing levels were linked to the highest total carbon stored in grassland soils, with extensive management next and intensive management lowest. The differences on average lead to a significant 10.1tC/ha in surface soils and 13.7tC/ha from 0.3 to 1.0m more carbon under intermediate compared with intensive levels of management. 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.

Grasslands are usually considered to have a carrying capacity for carbon sequestration, but this might take many years to reach. Yang *et al*. (2019) considered that it would take over 100 years for prairie restoration from arable soils, possibly as their specialist grasses took many years to colonise, 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 0-20cm profiles than in the first thirteen years. These averaged at 0.54tC/ha/yr for years 1-13 and 0.71tC/ha/yr for the highest diversity treatment in the 13-22 year period. These rates are markedly lower than the red clover plots at Colt Park achieved (De Deyn *et al*., 2014), 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 of stress-tolerators rather than high acquisitive plants, and with higher levels of legumes like red clover provide a useful 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. This makes grassland restoration a worthy alternative or addition to woodland creation on a larger scale, more quickly.

Other grassland types

Gregg *et al.* (2021) suggest that *Molinia caerulea* (purple moor-grass) dominated swards could be losing 0.85tCO_2e/ha/yr with high grazing levels or be sequestering carbon at between 0.5 and 0.53tCO_2e/ha/yr with low or no grazing. 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 69tC/ha in soils up to 15cm deep, but sequestration measurements are scarce, with Dawson & Smith (2006) giving 1.2tC/ha/yr for cut species-rich limestone grassland with no other information.

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 C 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 salt marshes, sand dunes, machair, seagrass (*Zostera* species) beds and muds. C sequestration will be greatest where soil/sediment accumulation is most rapid and where growth rates are high such as in salt marshes and seagrass beds.

Salt marshes

Salt marshes can reach 6.5tC/ha/yr accumulation or more in ideal conditions, (McCleod *et al*., 2011 quote figures of 0.18 to 17.13tC/ha/yr internationally), with rates of 0.6 to 2.2tC/ha/yr from different UK studies (quoted in Armstrong *et al*., 2020), higher by some 30% in non-sandy material. 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. They can have very deep deposits (over 10m – Beaumont *et al*., 2014) and thus accumulate large carbon stores provided these are not disturbed. However, with sea level rise, salt marshes can be squeezed out and drowned or they can steadily accumulate in the rising water levels.

Restoring salt marsh habitat, as in several managed retreat schemes, has the potential to contribute significantly to carbon capture. However, Gregg *et al*. (2021) suggest restoration of high salt marsh 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. Other studies found sequestration rates were high (averaging 1.04tC/ha/yr in the first 20 years after restoration) before dropping to around 0.65tC/ha/yr thereafter. There is a suggestion that it could take around 100 years for a restored salt marsh to gain the equivalent carbon stocks to a natural site.

Tidal sediments

Intertidal and subtidal sediments can be equally important as a carbon sink from accumulated sediment from both the marine and terrestrial environment. 0.16tC/ha/yr for example has been calculated for the Humber estuary (Alonso *et al*., 2012), and 1.12-1.98 tCO_2e/ha/yr (0.31-0.54tC/ha/yr) 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 0.64tC/ha/yr depending on the characteristics of the estuary and adjacent coastline. This is higher than the average 0.11 to 0.37tC/ha/yr 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 (0.94tC/ha/yr compared with 0.73tC/ha/yr), 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 fish 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). Poor water quality driven primarily by excess nutrients is one of the largest threats. 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) quotes sequestration rates of 0.2 to 2tC/ha/yr, also corroborated by McCleod *et al*. (2011) who gives an average of 1.38tC/ha/yr: a high level, through trapping sediment rich in carbon and capturing it through photosynthesis. This is higher than the 0.83tC/ha/yr averaged by Armstrong *et al.* (2020) and the 0.27tC/ha/yr they apply to evaluate the Welsh seagrass beds. Greiner *et al.* (2013) reported 0.37tC/ha/yr carbon sequestration averaging over 10 years in newly established seagrass beds of *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. 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 *Zostera marina* beds off South Wales.

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 0.23mm/yr, 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 60cm deep with dead material even deeper. Possible sequestration rates can be about 0.74tC/ha/yr of calcium carbonate and 1.7 for 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. Their net carbon accumulation is between 0.4 and 4.2tC/ha/yr, again which is very high potentially, (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 0.58-0.73tC/ha/yr, average sequestration rates, which is very similar to the 0.58tC/ha/yr for dry dunes and 0.73tC/ha/yr for wet dune slack habitats produced 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).

Conclusions

This review explores the relationship between carbon and habitats to inform ecologists working to counteract both the biodiversity and climate change crises, with opportunities to combine this with enhancing other ecosystem services. The key conclusions are that the restoration of peatlands is of the highest propriety, as identified by the CCC (2020) and Gregg *et al*., (2021), but that the seemingly universal mantra to plant trees to capture carbon would be better replaced by a range of alternative approaches that would best suit the landscape ecology, topography, soils and climates. This does not reflect the CCC (2020) recommendations which focus on new tree planting. This paper shows that new woodlands (rather than plantations) are important ecologically but may take up to 30 years before their carbon budgets are positive, demonstrating that alternative or additional approaches are needed simultaneously to capture carbon more immediately and at large scales to compensate for this. Gregg *et al.* (2021) come to similar conclusions.

The evidence point to habitat protection, restoration and enhancement as a priority to maintain existing carbon and wildlife oases and habitat creation not only to mop up more carbon, but to fulfil Lawton's principles (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 tree planting. Indeed, positive carbon sequestration balances can be more rapidly achieved in non-wooded habitats in many cases. Soil and sediments are the key and an understanding of soil properties and the opportunities they present is critical.

The review has demonstrated that there are strong arguments to focus on marine and intertidal habitat restoration and creation on a large scale and to reconnect rivers to their flood plains, converting floodlands into flower-rich flood meadows again, both providing major benefits for other ecosystem services. The creation of other wetlands from ponds to reed beds would also be productive and fit into the rest of the mosaic. 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 (rather than plantations) with mixed trees and shrubs, managed for amenity and wildlife, preferably established through natural generation, but planted where necessary too in the right place and on the right soils would provide a better long-term carbon stock and store than short-term commercial plantations (although these would be needed as well for commercial reasons), and a case can be made too for both grassland diversification combined with wood pasture at different scales to maintain some agricultural output and maintain different suites of animals and fungi.

The review also helps show that the relationship between carbon and habitats is complex and not fully understood in terms of the variability between different climates, soils and vegetation communities or the effects of climate change on them. Management also has a significant effect on carbon sequestration rates and budgets.

The importance of this review is the realisation that there are many options, that carbon capture may not be immediate in most, but will take time depending on the preceding conditions, and that there is a huge potential for capturing carbon at various scales whilst at the same time benefiting 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 give the net carbon uptake or loss of different scenarios in their Figure 7.2 (although some habitats are omitted), which shows the most carbon uptake in woodlands, plantations and saltmarsh, but 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. Similarly, restoring only 10% of our agricultural grasslands (which cover some 40% of the UK including semi-natural grasslands (Gregg *et al*., 2021) at 96,998km^2), to more species-rich pastures or hay meadows incorporating red clover, whilst retaining agricultural use could be capturing similar amounts of carbon and more quickly as the current goal for woodland/plantation establishment (30,000ha/yr until 2050). This is just one example. Other habitats, created in mosaics with hotspots like ponds 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 824,244ha compared with a Forest Strategy comprising 16,000ha of new woodlands, thus creating more benefits over a wider area for biodiversity and other ecosystem services.

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Units

tonnes carbon per hectare (tC/ha) or per year /yr.

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_2-e/ha/yr to Ct/ha/yr

1000kgC/km^2 = 0.01tC/ha

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