Natural England Research Report NERR094

Carbon storage and sequestration by habitat: a review of the evidence (second edition)

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14 October 2021 V2.1

ISBN 978-1-78354-732-6

Project details

This report should be cited as:

R Gregg, J. L. Elias, I Alonso, I.E. Crosher and P Muto and M.D. Morecroft (2021) Carbon storage and sequestration by habitat: a review of the evidence (second edition) Natural England Research Report NERR094. Natural England, York.

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Acknowledgements

We are grateful to the following external experts who reviewed and provided comments on the report, or specific chapters within it: Dr. Rob Field (RSPB), Dr. Doug Warner (University of Hertfordshire), Prof. Fred Worrall (Durham University), Dr. Mark Broadmeadow (Forestry Commission), Dr. James Morison (Forest Research), Dr. Jo-Anne Pitt, Dr. Mark Diamond, Dr. Glenn Maas, Dr. Judy England, Alice Hiley, Dr Mark Bourn (Environment Agency), Dr. Ruth Parker and Lisa Benson (Cefas), Prof. Chris Evans (UKCEH). We are also grateful to Samantha Broadmeadow (Forest Research) who produced Figures 2.4 and 2.5; and Dr Amanda Thomson and Dr Hannah Clilverd for their assistance regarding the 2021 update to the UK Peatland GHG Emissions Inventory.

We are grateful to the following (current and former) Natural England staff who provided input, comments and feedback on report drafts: Nigel Pilling, Emma Goldberg, Marion Bryant, Steve Peel, Andy Cole, Richard Jefferson, David Whiting, Alistair Crowle, Deborah Land, Dave Glaves, Iain Diack, Chris Mainstone, Ruth Hall, Simon Duffield, Ginny Swaile, Margaret Street, Katrin Bohn, Sue Rees, Hazel Selley, Victoria Copley, Alexandra Fawcett, Theresa Redding, Jakub Olewski and Sarah Taylor.

Record of report edits and corrections

A record of minor edits and corrections to NERR094 since first publication in April 2021 is stored in Appendix 3, page 222.

Executive Summary

Report background

Achieving 'net zero' greenhouse gas (GHG) emissions by 2050 is a statutory requirement for the UK and England. It will require major changes in the way we manage the land, coast, and sea, alongside decarbonisation of the energy, transport and other sectors. The natural environment can play a vital role in tackling the climate crisis as healthy ecosystems take up and store a significant amount of carbon in soils, sediments and vegetation. Alongside many other negative impacts, the destruction and degradation of natural habitats has resulted in the direct loss of carbon stored within them. Restoring natural systems can start to reverse this damage at the same time as supporting and enhancing biodiversity, alongside delivering co-benefits for climate change adaptation, soil health, water management and society. This Natural England Research Report is designed to clearly set out the evidence for how restoration and good management of habitats can contribute to climate change mitigation.

In this report, we review the scientific evidence base relating to carbon storage and sequestration by semi-natural habitats, in relation to their condition and/or management. This new report updates and expands previous work by Natural England on 'Carbon storage by habitat' published in 2012[1]. We cover terrestrial, coastal and marine habitats, and the freshwater systems that connect them, in order to quantify their relative benefits for carbon management.

We set out to:

- Review the available evidence and summarise the carbon storage and sequestration rates of different semi-natural habitats with an indication of the range of values and the degree of confidence we can place in them.
- Facilitate the comparison of carbon storage and sequestration rates between semi-natural habitats.
- Apply evidence to England. Our main focus has been on evidence gathered on British ecosystems, but we have also included studies from other regions, particularly north west Europe, where they are relevant and helpful.
- Identify key evidence gaps in order to highlight where there is need for future research to support land use and land management decisions for carbon.
- Provide those working in land management, conservation and policy with relevant information required to underpin decisions relating carbon in semi-natural habitats.

[1] Carbon storage by habitat: Review of the evidence of the impacts of management decisions and condition of carbon stores and sources (NERR043)

Nature-based solutions

Climate change and biodiversity loss are inextricably linked and need to be addressed in an integrated way. Nature-based solutions (NbS) is a broad concept which describes how protecting, restoring and managing natural systems can solve societal problems. A widely used definition is that of the International Union for the Conservation of Nature (IUCN): nature-based solutions are "actions to protect, sustainably manage, and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits".

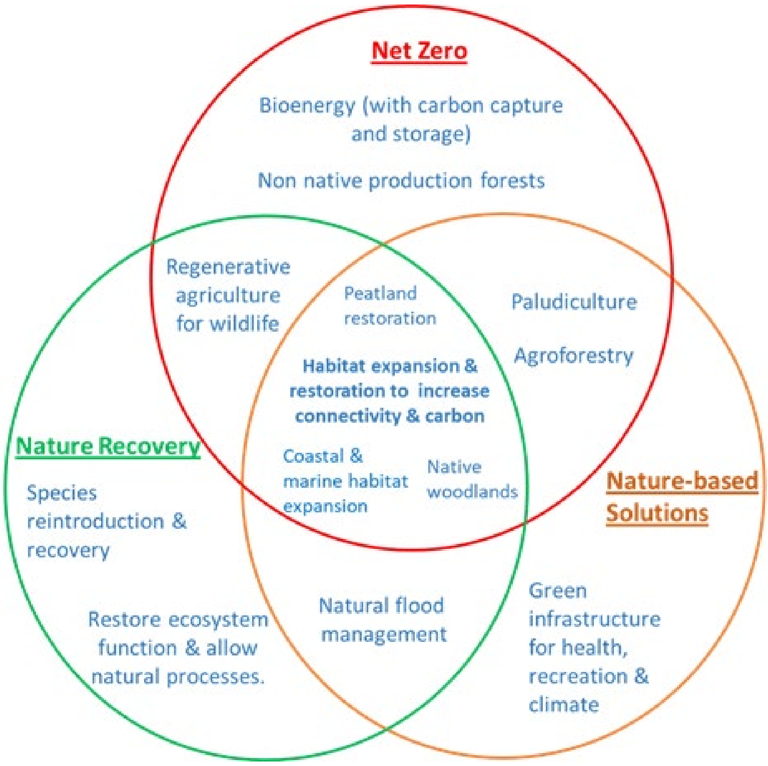

Figure 1 Examples of relationships between nature-based solutions, nature recovery and net zero

The most effective NbS for climate change mitigation are often those based on habitat restoration and creation (figure 1), as land use change from a degraded habitat to a functioning, resilient one offers the greatest potential in capturing carbon dioxide from the atmosphere. The protection of existing habitats is also vital, as their biodiversity and carbon stocks may have taken centuries to millennia to become established and are quickly lost if disturbed.

Carbon storage and sequestration by habitat – key messages from this report

Since the last Natural England report on carbon storage and sequestration by habitats in 2012, both science and policy have increasingly recognised the importance of natural ecosystems in climate change mitigation and their wider benefits for society. Comparing across habitats is complicated, as there is no standard protocol for collecting carbon and flux data, and habitats and land use are often looked at in isolation. Broad habitats in the carbon reporting literature are typically aggregated together, but in reality comprise a range of different systems, the attributes of which may affect carbon cycling (for example fens include a variety of types in different hydrological conditions and heath includes both dry and wet heath). As a result, reviews of this kind are rarely undertaken but are necessary to understand how best to target and prioritise habitat creation, restoration and land management to mitigate climate change. To facilitate this approach, we conducted in-depth literature reviews across a range of habitats relevant to conservation in England, set out in the following habitat chapters.

Figures 2 and 3 present a high-level overview of carbon storage and sequestration by different habitats. Key messages are identified but it should be noted that a huge range of different information underlies this overview, which is set out in the chapters.

Chapter 2: Woodland, trees and scrub – The largest carbon sequestration rates amongst semi-natural habitats are in woodlands. Native broadleaved woodlands are reliable carbon sinks that continue to take up carbon over centuries with benefits for biodiversity and other ecosystem services, although the rate varies greatly with tree species and age and is strongly influenced by soils and climate. Sequestration rates decline over time, but old woodlands are substantial and important carbon stores. Although woodland management may be important for a range of reasons, it is not essential to maintain carbon sequestration. Native woodland managed with a minimum intervention approach can be an effective climate change mitigation measure. Timber production can have benefits for climate change mitigation where wood products store carbon for the long-term, or replace more fossil fuel intensive materials and fuels; and can be produced in ways that support biodiversity, such as using native tree species and management of rides and forest edges. However, non-native species of tree generally support lower levels of biodiversity and plantations on peatlands have led both to the loss of biodiversity and carbon. Hedgerows, orchards and other trees outside woodland can also sequester and store carbon as well as providing other benefits within an agricultural and biodiversity context.

Chapter 3: Open habitats and farmland – Open habitats such heathlands and semi-natural grasslands sequester and store more carbon than modern agricultural landscapes but typically store less carbon than peatlands, saltmarsh and established woodlands. They also sequester less carbon than woodlands, as they do not accumulate woody matter. Vegetation may also be managed by grazing or cutting, representing a loss of carbon from the system. Agricultural land use on peat soils gives rise to extremely high carbon emissions. Carbon is almost entirely stored in the soils of these habitats and stores are variable depending on climate, soil and management history, but can be significant. Protection of old, established habitats is important for biodiversity, as well the carbon stocks they hold, as both may have taken centuries to accumulate.

Chapter 4: Blanket bogs, raised bogs and fens – peatland habitats hold the largest carbon stores of all habitats. When in healthy condition they sequester carbon slowly but are unique in that they can go on doing so indefinitely. Peatlands in England have long been subjected to damaging land use, resulting in them becoming a large source of greenhouse gas emissions, releasing carbon previously stored for millennia. Restoration interventions in many cases will reduce these emissions, allow biodiversity to recover, increase peatlands resilience in the face of a changing climate and provide a range of benefits for people and society. Restoring the carbon sink function of peatlands is possible though may take decades depending on the initial level of damage to a site. Restoration actions include blocking drains, stopping burning and removing forest plantations.

Chapter 5: Rivers, lakes and wetland habitats – Rivers, lakes and wetlands form an important element of carbon cycling by habitats due to their interactions at a catchment scale. This includes the transport of dissolved and particulate organic carbon and the impact of natural and artificial drainage patterns on the hydrology of other habitats, particularly wetlands. River systems are hard to characterise on an area basis in the same way as other habitats, but evidence suggests that they are mostly sources of carbon due to nutrient inputs and physical modifications preventing them from acting as sinks. Standing waters, such as lakes, can act as carbon sinks, storing carbon within the sediments long-term, but nutrient inputs from neighbouring land can tip these systems from sinks to sources. Fluvial systems also have an important role as a conduit between the terrestrial and marine environments.

Chapter 6: Marine and coastal habitats – Saltmarshes are large carbon stores, although they are subject to erosion and accretion through natural coastal processes and are affected by rising sea levels. Sea grass meadows also have the potential to store large quantities of carbon within the sediments if undisturbed. Their vegetation can also sequester significant amounts of carbon *in situ*, as well as acting to trap and store carbon released from elsewhere. There are significant evidence gaps in our understanding of carbon cycling for many marine and coastal habitats. However, the protection and re-establishment of coastal habitats will also provide climate change adaptation benefits in addition to those for mitigation.

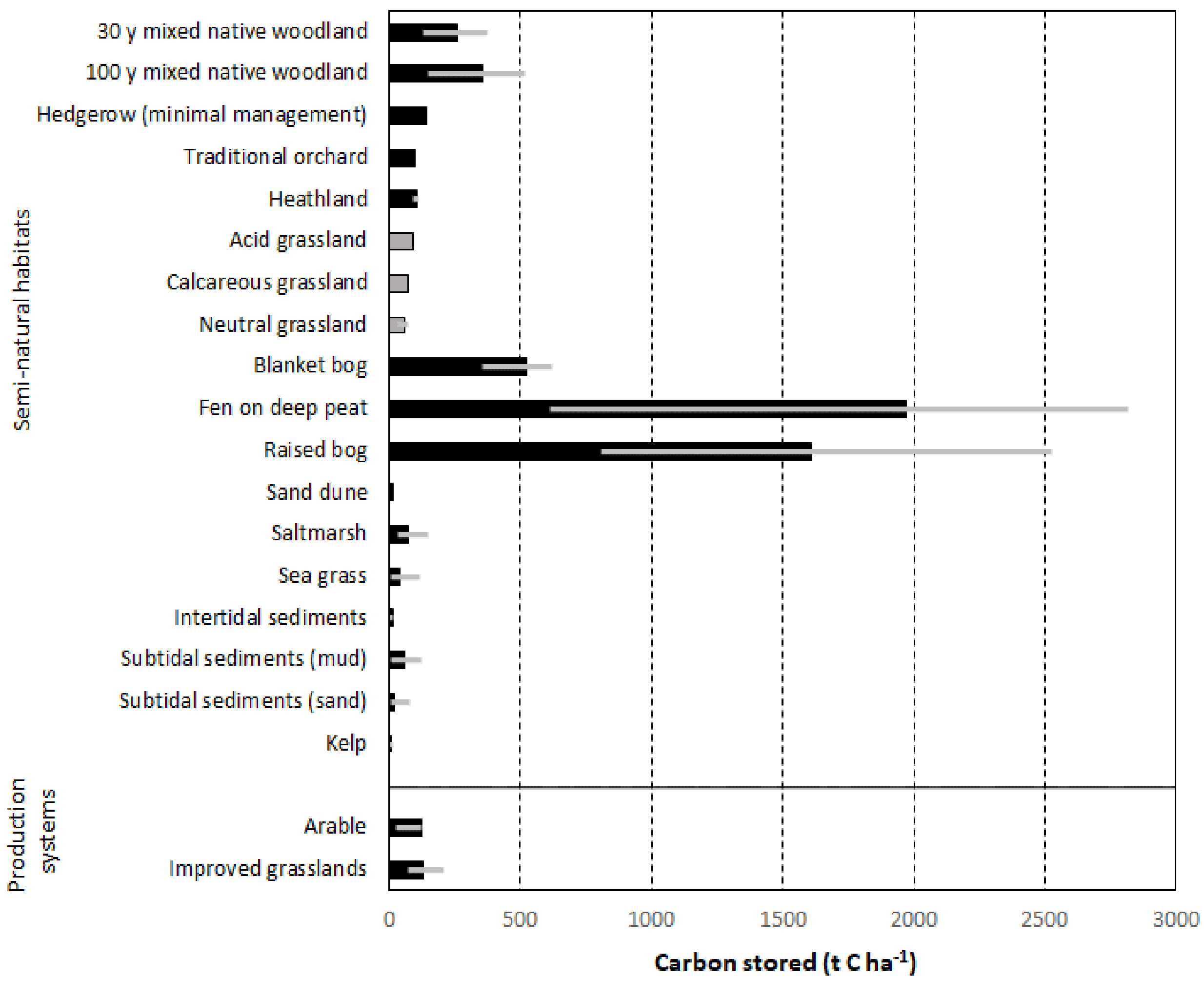

Figure 2 Carbon storage in contrasting habitats and land managements, using the best available data. Note that the semi-natural grasslands data are from the top 15 cm of soil only are shown in grey. Other habitats (shown in black) vary in their depths from 15 cm to 380 cm. Blanket bog carbon stocks are based on catchment scale estimates – see section 4.3. Fen data here are restricted to deep semi-natural fens; there are a range of other types – see Section 4.5. Numerical data and soil depths are provided within the review chapters and Appendix A

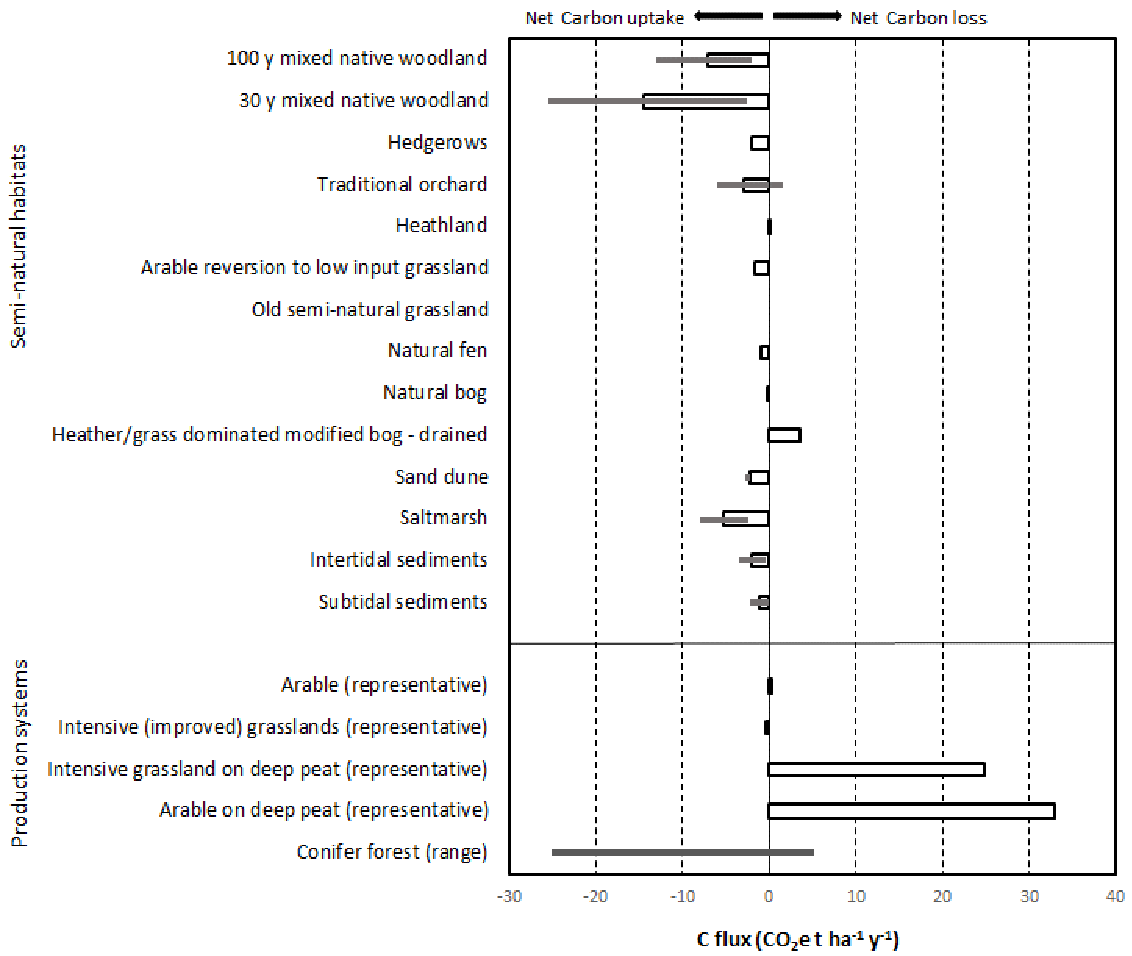

Figure 3 Carbon flux in contrasting habitats and land managements, using representative data. Best available data have been used and includes data from a wide range of different sources, modelled and field data. A negative value indicates sequestration, positive values are emissions. The grey bars indicate the likely range of values across sites where this is available. Habitats with no suitable data are not included and we refer the reader to the chapters where this is reviewed and discussed. Numerical data and soil depths are provided within the review chapters and Appendix B

Opportunities for NbS to deliver for climate change mitigation

NbS is a key concept for tackling the climate and biodiversity crises. A joined-up approach that addresses both climate change and biodiversity decline together is the only realistic way of meeting the multiple demands on our environment. Within this report we identify where creation, restoration and improved management of natural habitats can contribute to delivery of the net zero target. Nevertheless, there is a need to be realistic: it is not possible to offset anything close to current UK emissions across the different sectors of the economy through better environmental management alone. Deep cuts in emissions across all sectors is required with NbS playing an important role in mitigating the residual, hardto-eliminate emissions.

The success of NbS for climate is dependent on location. It is important to be rigorous in assessing how much difference any change in land use or management will make to biodiversity and climate in a particular place. If NbS is to make a significant contribution to achieving net zero by 2050, implementation needs to increase significantly and immediately. For many habitats there is a lag between instigating creation and restoration approaches and seeing the benefits in terms of carbon.

Nature-based solutions can deliver for climate adaptation, as well as mitigation, and future change must be considered. Nature-based solutions should be designed, managed and evaluated to ensure that they will continue to be effective in a future which is warmer and subject to changes in rainfall, including more extreme events such as droughts and floods.

By looking across a range of habitats, and considering biodiversity alongside climate change mitigation, we identify the following **key principles** from our assessment:

1. **Protect and restore peatlands.** Peatlands are our largest natural carbon stores and it is important to slow and eventually halt greenhouse gas emissions, including through raising water tables, stopping burning and removing planted trees.
2. **Create new native broadleaved woodlands.** Native woodland is an effective carbon sink and over much of England can deliver comparable carbon uptake to non-native species and provide more benefits for biodiversity. Growing the right trees in the right place is however critical to maximise these benefits.
3. **Protect and restore natural coastal processes.** This allows habitats, such as saltmarsh, to maintain themselves and re-establish inland as the sea level rises, and to sequester and store carbon. It is also an important and urgent aspect of climate change adaptation. Active intervention will be necessary to restore some habitats, such as sea grass.
4. **Protect existing semi-natural habitats.** Most of England has been intensively managed for a long time and semi-natural habitats, of all types, are rare fragments containing many of our native species that are not found elsewhere. Many of these, including grasslands and heathlands, also store appreciable amounts of carbon in their vegetation, undisturbed soils and sediments.
5. **Target incentives for NbS to places where they can have most benefit.** Different approaches work better in different places and it is important to maximise synergies and minimise trade-offs if we are to deliver net zero ambitions at the same time as restoring biodiversity and meeting the needs of people. Decisions about NbS need to consider the wider context of land use and management and the need to maintain, and where possible increase, domestic food and timber production in ways which do not lead to increased emissions either in the UK or overseas.
6. **Integrate NbS for climate into landscapes which are primarily devoted to agriculture or production forestry.** To meet the scale of change required in greenhouse gas emissions, there is a need to take land out of agriculture, particularly for woodland creation and peatland restoration. Actions such as hedgerow planting, good soil management and innovative agricultural approaches, such as paludiculture, can also contribute whilst enabling agricultural production to continue. Within production forest biodiversity can be supported by including broadleaved trees and appropriate management of forest rides and edges.
7 **Carry out research and monitoring to fill evidence gaps.** There are still large knowledge gaps for many habitats. For example, there is significant potential to increase carbon stocks for coastal and marine habitats, but we lack evidence in the English or UK context. Across all habitats, the carbon content of soils, sediments and vegetation, and ecosystem carbon fluxes are rarely measured. Even the depth of soil is rarely monitored. The role that freshwater habitats can play in climate change mitigation is also an understudied area.
8. **Ensure mitigation and adaptation to climate change are planned together.** This is important to ensure the durability of solutions for carbon sequestration and storage and to promote synergies rather than conflicts between objectives. We should look for multifunctional and integrated opportunities when planning our responses to the climate and biodiversity crises.

Contents

Appendices

Acronyms and abbreviations

AG – Acid Grassland
BW – Broadleaf Woodland
DSH – Dwarf Shrub Heath (broad habitat type)
CW – Coniferous Woodland
LH – Lowland Heathland
UH – Upland Heathland
PAWS – Plantations on Ancient Woodland Sites

C – Carbon
N – Nitrogen
P – Phosphorus
CO_2 – Carbon dioxide
CO_2e – Carbon dioxide equivalent
CH_4 – Methane
N_2O – Nitrous oxide
GHG – Greenhouse Gas
GWP – Global Warming Potential

OM – Organic Matter
OC – Organic Carbon
SOM – Soil organic matter
SOC – Soil Organic Carbon
DOC – Dissolved Organic Carbon
POC – Particulate Organic Carbon
PIC – Particulate Inorganic Carbon
POM – Particulate Organic Matter
BD – Bulk Density

g – gram
kg – kilogram
t – tonnes
kt – kilotonnes (10^3 tonnes)
Mt – Megatonne (10^6 tonnes)
Gt – Gigatonnes (10^9 tonnes)
Tg – teragrams (10^{12} grams)
µg – microgram
mol – moles
km^2 – square kilometre
ha – hectare
cm – centimetre
m – metre
m^3 – cubic metre
y – year

< – less than
> – greater than

UN – United Nations
UNFCC – United Nations Framework Convention on Climate Change
IPCC – Intergovernmental Panel on Climate Change
BEIS – Department of Business, Energy and Industrial Strategy
CCC – Climate Change Commitee
ONS – Office for National Statistics
CS – Countryside Survey
JNCC – Joint Nature Conservation Committee
CEFAS – Centre for Environment, Fisheries and Aquaculture Science
FC – Forestry Commission
FR - Forest Research
IUCN – The International Union for Conservation of Nature

HCV – High Conservation Value
BAP – Biodiversity action plan
LULUCF – land use, land use change and forestry
BECCS – bioenergy with carbon capture and storage
NNR –National Nature Reserve
WCC – Woodland Carbon Code

NPP – Net Primary Productivity
NEE – Net Ecosystem Exchange
NECB – Net Ecosystem Carbon Balance

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1 Introduction

1.1 The carbon cycle

Carbon is cycled dynamically between the atmosphere, land and ocean due to the processes of photosynthesis, respiration, decomposition and combustion (figure 1.1). Ecosystems can capture this carbon, in the form of carbon dioxide (CO_2), storing it in their vegetation, soils and sediments over years or even millennia; and accumulate vast stocks in the process. Carbon stocks held in ecosystems are not static, they change naturally as habitats progress through successional changes or degrade (Ostle and others 2009). When carbon inputs are greater than emissions, then the ecosystem is a net sink, and when emissions outweigh inputs then the ecosystem is a net source of carbon.

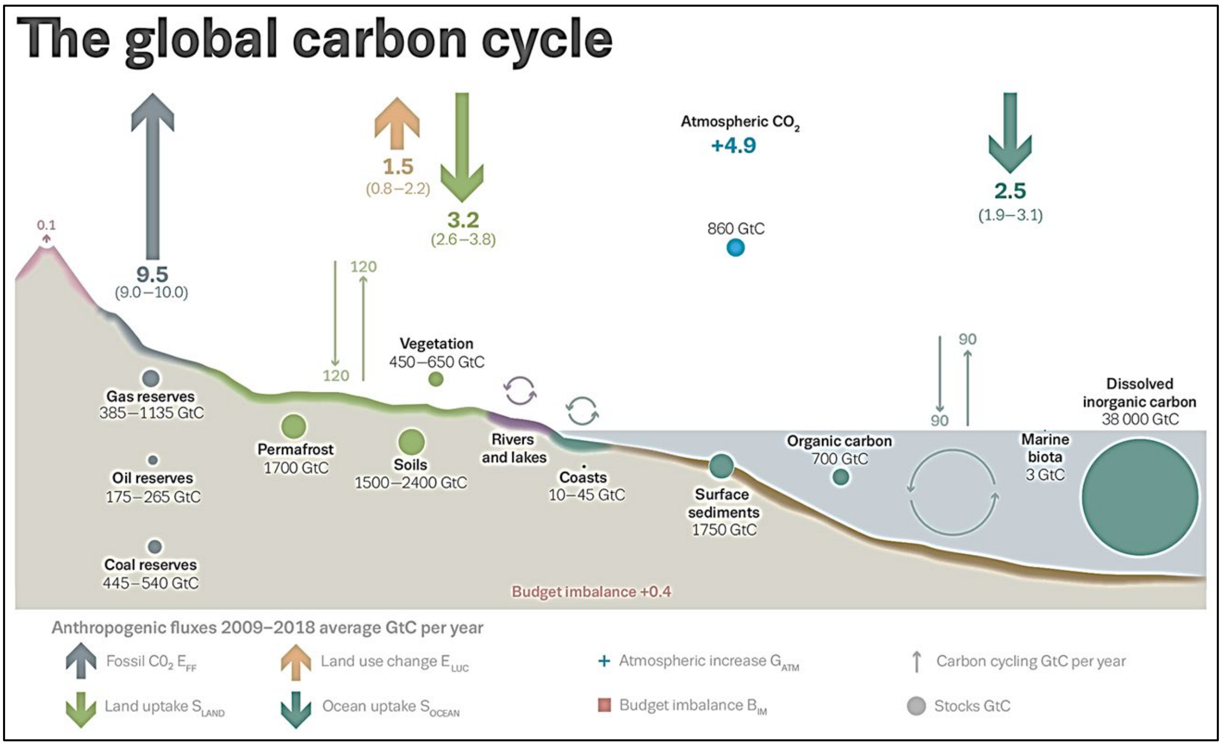


Figure 1.1 The global carbon cycle, from Friedlingstein and others (2019)

Ecosystems that have been undisturbed for long periods of time are typically assumed to be at an equilibrium, where carbon inputs are equal to emissions, and the carbon stock size is therefore constant (Sozanska-Stanton and others 2016). The IPCC methodology[2], as used by the UK greenhouse gas (GHG) emissions inventory, states that management-induced changes to carbon stock change will typically be manifested over a period of several years to

[2] IPCC Guidelines for National Greenhouse Gas Inventories

decades, and habitats at equilibrium and under stable management will not be a source or sink of carbon.

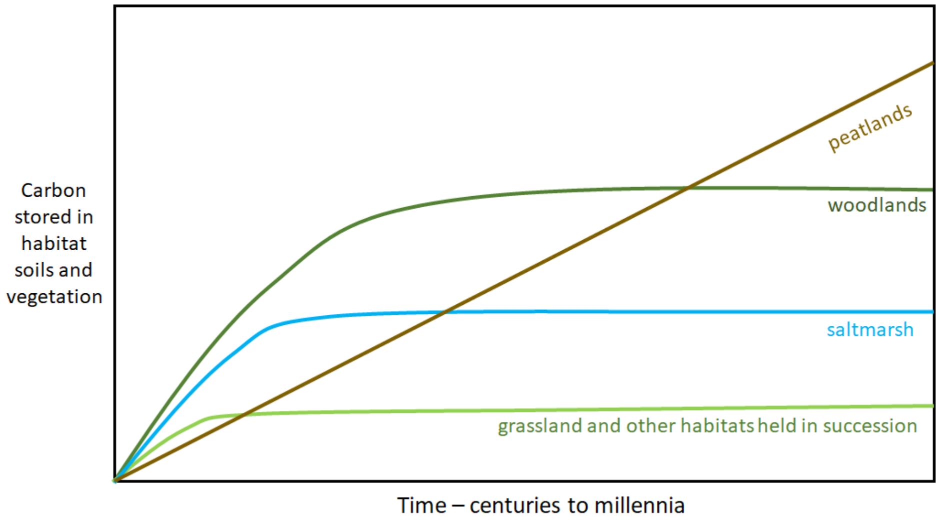


Figure 1.2 Conceptual model of habitat trajectory towards carbon stock equilibrium. The rate of sequestration and capacity to store carbon is different for different habitats, with every site having an equilibrium specific to its management, climate and soils. The exception to this is peatlands, which can continue sequestering carbon for many millennia. Note – this figure is conceptual, axis are for illustration and are not to scale. Trajectories assume no disturbance within the habitat

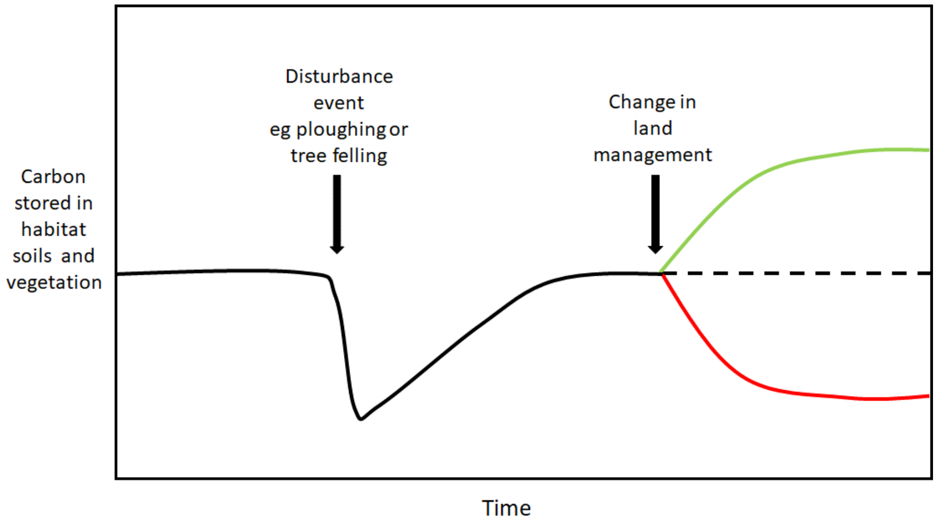


Figure 1.3 Conceptual model of habitat carbon stock equilibrium and land use change. Once a habitat achieves equilibrium carbon stocks will maintain a steady state until disturbed or a new land management intervention is imposed. The green line indicates a change in land use to a habitat with a higher equilibrium (eg modified grassland conversion to woodland), and the red line a change to a lower equilibrium (eg semi-natural grassland to arable). The dashed line indicates a continuation in the established land management. Note – this figure is conceptual, axis are for illustration and are not to scale

Knowing where a habitat is on its trajectory to achieving a steady carbon state is dependent on its management history, as well as the influence of other factors such as soil and climate variability (figures 1.2 and 1.3). Some ecosystems, including temperate forests, can take many centuries to approach equilibrium and peatlands can continue to sequester carbon over millenia. Having a clear understanding of the factors that underpin this balance, or carbon flux, is essential when considering the role that semi-natural habitats can play in mitigating climate change.

Land use and management can have direct and indirect effects on these carbon inputs and losses through their influence over vegetation cover and soil or sediment disturbance (figure 1.3). Human activities on land have a significant impact, with the agriculture, forestry and other land use sector (AFOLU) responsible for 23 per cent of global greenhouse gas emissions largely due to land use change practices such as deforestation and agriculture (IPCC 2019). However, about 29 per cent of all anthropogenic GHG emissions are absorbed by plants and soils in terrestrial ecosystems (Friedlingstein and others, 2019; Fig. 1.1). Land is an essential resource in our attempts to mitigate climate change, with potential to enhance ecosystem sinks alongside reducing existing emissions from land use (Griscom and others 2017). Coastal and marine habitats, often referred to as 'blue carbon' habitats, also play an important role in reducing GHG emissions but are comparatively understudied. Globally approximately 22 per cent of anthropogenic GHG emissions are taken up by the ocean (Fig. 1.1). In the UK, soils represent the largest terrestrial carbon stock, holding approximately 95 per cent of land carbon (Bradley and other, 2005; Ostle and others 2009). Therefore, land use changes that disturb the stability or function of soils pose the biggest threat to the UK's carbon stores (Field and others 2020). On the other hand, positive land use change by restoring ecosystems and their functional processes could deliver a long-term carbon gain.

1.2 Net zero and emissions reporting

Reducing GHG emissions is an urgent priority to avoid dangerous climate change. This is reflected in the 2015 Paris Agreement of the UN Framework Convention on Climate Change[3] (UNFCCC) which commits nations to keeping global temperature rise to well below 2 ^{o}C and pursuing efforts to limit it to 1.5 ^{o}C. Analysis has shown that this requires emissions to reduce global CO_2 emissions by around 45 per cent by 2030, and reach net zero by 2050 (IPCC, 2018). Net zero means that whilst there may still be some emissions of GHGs they are balanced by removals from the atmosphere, for example via photosynthesis by vegetation. Protection, expansion and improved management of natural areas is therefore essential to preventing dangerous climate change, alongside other actions such as stopping fossil fuel burning and improving energy efficiency.

The UK has committed to achieving net zero by 2050; this is a statutory requirement for the UK and England under the Climate Change Act 2008[4] and will require major changes in the way we manage the natural environment, alongside decarbonisation in the energy,

[3] 2015 Paris Agreement of the UN Framework Convention on Climate Change
[4] Climate Change Act 2008

transport and other sectors. In 2019, UK net emissions were 455 Mt CO_2e, approximately 44 per cent lower than in 1990 (BEIS, 2021 a). Carbon dioxide equivalent ($CO2_e$) is a metric measure used to report the global warming potential (GWP) of different GHGs and allows consistent comparisons across sectors and gases[5]. National emissions have been falling steadily in recent years because of a shift away from burning fossil fuels, particularly coal, for energy. In addition, much of our manufacturing and carbon intensive industry has moved production abroad so removing it from our national emission reporting. However, at 5.3 t CO_2 the UK's per-capita CO_2 emissions are still above the global average (4.8 t CO_2 in 2018) (Carbon Brief 2020) and reaching net zero will require major changes.

Reducing emissions from degraded ecosystems and promoting carbon uptake, for example through restoring peatlands and planting trees, are important elements of achieving net zero. In 2019, the UK GHG emissions from the agriculture sector and land use, land use change and forestry (LULUCF) sector, were 46.3 Mt CO_2e and 5.9 Mt CO_2e respectively (BEIS 2021a). Together the two sectors account for 12 per cent of all UK emissions. The 2019 emissions reporting was the first year that a major methodology change has been incorporated to better represent emissions from peatlands. Previously a net GHG sink, this change to more fully include peatlands has resulted in the LULUCF sector becoming a net source of emissions due to the high emissions associated with agricultural land use on peatlands soils (BEIS 2021b). Overall, the inclusion of degraded peatlands represents an increase of 16 Mt CO_2e (3.5 per cent) to national emissions. Forestry is the dominant carbon sink represented in the LULUCF sector, reporting an uptake of 18.2 Mt CO_2e in 2018 (Brown and others 2020b). However, nationally the ageing profile of woodlands and decline in planting rates is weakening the strength of forests to sequester CO_2 (Climate Change Committee 2020a). The importance of blue carbon habitats in GHG mitigation is also being increasingly recognised, although historically they have not been included in UK and international emissions inventories and considerable evidence gaps remain.

1.3 Nature-based solutions and climate change mitigation

Climate change and biodiversity loss are closely linked problems and need to be addressed in an integrated way (Roberts and others 2020). Nature-based solutions (NbS) are actions which support biodiversity and provide for people, including health and wellbeing, at the same time (figure 1.4). A widely used definition is that of the IUCN (2020):

"Nature-based Solutions are actions to protect, sustainably manage, and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits."

[5] Anthropogenic and Natural Radiative Forcing. In: Climate Change 2013: The Physical Science Basis

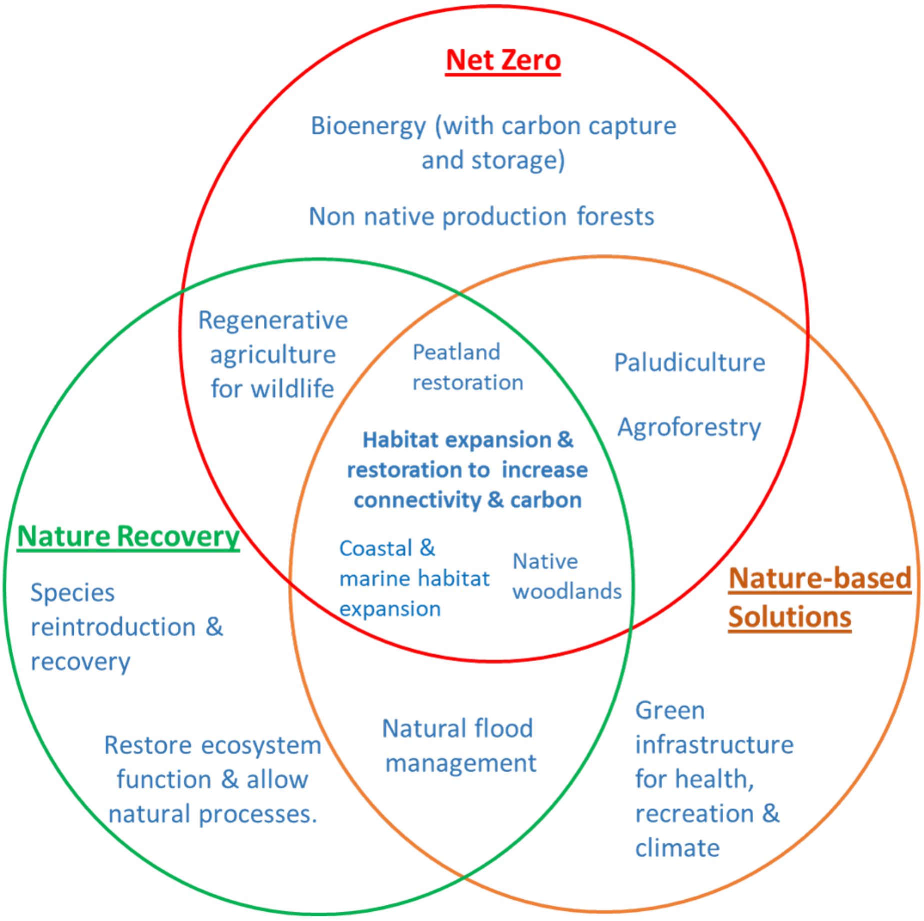


Figure 1.4 Examples of relationships between nature-based solutions, nature recovery and net zero

The natural environment can play a vital role in tackling the climate crisis as healthy functioning ecosystems take up and store a significant amount of carbon in soils, sediments and vegetation. The destruction and degradation of natural habitats has resulted in a direct loss of carbon stored within them, and improved land stewardship currently the most mature and cost-effective carbon dioxide removal method (Griscom and others 2017). The conservation and restoration of natural systems can reduce net emissions at the same time as supporting and restoring biodiversity. Recent initiatives for habitat creation and restoration in England, including the Nature Recovery Network[6] and the commitment to 500,000 ha of habitat expansion in the 25 Year Environment Plan[7], have the potential to make an important contribution to achieving net zero.

To reach the UK's net zero target will require changes in how we use our land and coast. The Climate Change Committee (CCC) (2020b) estimated that 20 per cent of agricultural land will need to be released before 2050 to deliver actions that reduce and sequester carbon. Such a significant shift in land use change, however, will need to look beyond a single goal of carbon capturing and identify opportunities that deliver wider co-benefits for climate

[6] Nature Recovery Network
[7] A Green Future: Our 25 Year Plan to Improve the Environment

change adaptation, biodiversity, water management and soil health. For example, woodland planting can be sited to promote water infiltration and protect communities from flooding. Coastal vegetated habitats also capture and bury terrestrial carbon, as well as carbon from their own habitat, whilst providing coastal protection and habitat for economically important species (Roberts and others 2020).

Nature-based mitigation approaches must also consider potential trade-offs with other ecosystem services as there is a risk for these to conflict with one another, with impacts for land use and emission targets for decades or even centuries to come. Historic examples of this can be seen in the legacy of converting and draining peat soils for coniferous forestry plantations and intensive agriculture, and the significant GHG emissions that have resulted (BEIS 2021b). By taking a strategic, integrated approach to land use change it is possible to reduce land based GHG emissions and sequester carbon whilst contributing to other priorities such as food production, climate change adaptation, for both people and nature, and biodiversity enhancement (CCC 2018).

The reliance of ecosystem services, such as climate regulation, on biodiversity means that, for nature to substantially contribute to mitigation approaches will require an expansion in land cover of healthy, semi-natural habitats (Roberts and others 2020). The large carbon stocks held in old, undisturbed habitats require continued protection, as alteration in management or plant communities can release significant quantities of carbon back to the atmosphere (Sozanska-Stanton and others 2016). Protection of established habitats also provides the most benefits for biodiversity. The carbon stores potentially held in offshore sediments is also a point for further exploration and consideration for future protection. However, many of our ecosystems are degraded, which means that habitat protection must be paired with large scale habitat restoration and recreation (Roberts and others 2020). This has the potential to significantly increase their capacity to store carbon (Field and others 2020). But in some cases, habitat restoration may mean land managers must decide between managing land for carbon or enhancing biodiversity (for example, in tree removal from bog habitats to restore hydrology or holding succession to maintain open grassland).

Habitat protection and restoration must play a central role in efforts to mitigate climate change and support societal adaptation to change. Since the publication of Natural England's last review highlighting the important role semi-natural habitats play in the storage and sequestration of carbon (Alonso and others 2012[8]) both the scientific evidence and policy has increasingly recognised the importance of natural ecosystems in climate change mitigation and their wider benefits for society. Terrestrial habitats, such as woodlands and peatlands have received much of the attention regarding their ability to sequester and store carbon, but the potential role of 'blue carbon' in marine and coastal systems to deliver contributions to climate change mitigation is being increasingly recognised (Macreadie and others 2017). Likewise, the need for inland waters such rivers, streams and lakes to be embedded into mitigation strategies is important due to the role these systems play in the transport of carbon, as well as its storage and mineralisation in

[8] Carbon storage by habitat: Review of the evidence of the impacts of management decisions and condition of carbon stores and sources (NERR043)

their banks and sediments (Battin and others 2009). Collectively considering the terrestrial, coastal and marine habitats, and the fluvial and other freshwater systems that connect them, will support a catchment scale approach to carbon management and broaden the scope beyond individual sites.

This report aims to bring these habitats together, to quantitatively review contemporary evidence relating to carbon storage and sequestration by semi-natural habitats, and provide those working in land management, conservation and policy with the relevant information required to underpin decisions.

1.4 Project Rationale

The aim of this review of the carbon storage and sequestration by habitat was to:

- Review the available evidence and summarise the carbon storage and sequestration rates of different semi-natural habitats, in relation to their condition and/or management, with an indication of the range of values and the degree of confidence we can place in them.
- Facilitate the comparison of carbon storage and sequestration rates between semi-natural habitats.
- Summarise carbon storage and sequestration in agricultural settings as a baseline for land use change to habitat creation and restoration.
- Identify key evidence gaps in order to highlight where there is need for future site-based research required to underpin land use and land management decisions for carbon management.

1.5 Methodology

- This review takes a narrative approach, using current scientific literature to provide a detailed overview of carbon sequestration cycling and storage in each of the habitats covered and to facilitate comparisons between them.
- The findings of the report have received extensive peer review by external academics, conservation stakeholders at the Royal Society for the Protection of Birds and the Wildlife Trust, habitat specialists within Natural England and colleagues from the Forestry Commission, Forest Research, Environment Agency and the Centre for Environment Fisheries and Aquaculture Science (CEFAS).
- We assigned confidence levels to the data that we considered representative of each habitat. This is in the form of a RAG assessment, with low - medium - high scores used to represent as red - amber - green indicating the amount of evidence and the agreement of evidence. This was carried out collectively by the authors using their professional judgement.

1.6 Scope of the report

- We have reviewed evidence that we deemed relevant to the English situation, but particular attention has been paid to work from England, UK and North West Europe as these will be most directly applicable. Locations of source data are indicated in the report text.

- This review focuses on the carbon storage and flux of soils and vegetation of the identified habitat, including when possible, carbon dioxide, methane, and dissolved and particulate organic carbon. Emissions from management techniques through livestock and machinery use are not included due to their complex and variable interactions but are referred to where relevant.
- The report is not intended to be a systematic review but is a collation of the relevant evidence and data to reflect the current understanding in January 2021.
- This is not a manual for carbon offsetting. The carbon storage and flux values reported here are the best available in the scientific literature using author judgement. However, there are additional important considerations, for example about verification, how to deal with low confidence in some estimates, geographic variation, permanence of carbon storage in semi-natural habitats and the variability of flux rates in space and time.

1.7 Audience

This review will support the following groups with a clear and concise evidence base for nature-based climate change mitigation approaches:

- Land managers and advisers who need to consider, identify and quantify the role land management and interventions can play in contributing to climate change mitigation.
- Habitat specialists, to inform the co-benefits of their work areas to deliver for climate change mitigation and biodiversity, as well as identification of future evidence needs.
- Policy leads, to support the embedding of climate change mitigation and nature-based solutions into development of environmental and land management policy.

2 Woodlands, trees and scrub

2.1 Chapter summary and key messages

Woodland covers 10 per cent of England's area but would have once been much more extensive. The planting of new woodlands and trees, as well as the improved management of existing woodlands will play an integral role if the UK is to meet net zero targets by 2050 Given the focus of this report on semi-natural habitats, we focus on native broadleaved woodlands which are important for biodiversity climate change mitigation and a wide range of other ecosystem services. Although not the focus of the report, non-native conifer plantations are included for comparison where appropriate. The potential of trees outside woodland settings are also considered with reviews of hedgerows, orchards and scrub.

The key messages are:

- New native woodlands can sequester carbon at a higher rate than other semi-natural habitats, with the right combination of soil type and tree species. The sequestration rates fall with time but are typically higher than other habitats, even after 100 years or more without management. Due to the many variables that influence carbon sequestration in woodlands, we assign a medium confidence to our assessment.
- As woodlands become older, they become significant carbon stores, both above and below ground. Due to the significant body of scientific literature and the availability of Woodland Carbon Code data we have assigned medium-high confidence to the data reported here.
- There are large variations between sites in carbon storage and sequestration potential for new woodlands. Maximising carbon benefits depends on growing trees in the right places where both climate and soil are suitable for the species.
- Climate change adaptation is essential to build the resilience of woodlands to climate change. This includes ensuring woodlands have a diversity of species and that selected species are likely to grow well under future as well as present climates.
- Natural colonisation and natural regeneration of woodland offer potential advantages for climate change adaptation and mitigation although more evidence is needed to test and quantify this.
- Over much of England, native tree species can sequester carbon at rates comparable to, or in some cases higher than, non-native conifer plantations and support higher biodiversity. The highest carbon sequestration rates are likely to be achieved in conifer plantations growing in wetter climates, but only in suitable soil conditions.
- Significant carbon emissions can result from planting trees on organic soils and particularly deep peat, where emissions from drained peat can exceed carbon sequestration rates by trees. It also replaces carbon in a stable long-term store with carbon that may be relatively quickly released to the atmosphere.
- Comparing the carbon sequestration and storage benefits of woodlands managed primarily for timber production with those primarily managed for nature conservation is not straightforward due to the range of uses to which timber is put,

whether carbon is stored for the longer term and/or substitutes for more fossil fuel intensive materials or fuels.

- Trees outside woodland, including in hedgerows, within wood pasture systems and traditional orchards as well as scrub, can contribute to carbon sequestration and storage at the same time as supporting important aspects of biodiversity. They can also provide other benefits within agricultural systems, including reducing soil erosion and providing shelter for livestock. However, the literature is relatively limited compared to that for woodlands, and as a result we assign low confidence in our reported values.
- Tables 2.1 and 2.2 summarise the carbon storage and flux values identified as representative for habitats reported in this chapter.

Table 2.1 Summary of carbon storage values for woodland, tree and scrub habitats, using typical values derived from Woodland Carbon Code data and scientific literature

Habitat Description	Soil Carbon (t C ha^{-1})	Soil Depth (cm)	Vegetation Carbon (t C ha^{-1})	Soil + Veg. Carbon (t C ha^{-1})	Confidence [High, Medium, Low]	References
Woodland						
100-year Mixed native broadleaved woodland on mineral soil (to 1m)	151[bcd] [108 to 173]	*100cm*	203[abc] [41 to 344]	**354** **[149 to 517]**	**Medium/ high** Confidence is medium-high see Note at bottom of table.	[a]Woodland Carbon Code (2021) [b]Poulton and others (2003) [c]Butt and others (2009) [d]Vanguelova and others (2013)
100-year Mixed native broadleaved woodland (to 15cm soil depth)	55[b] [50 to 59]	*15cm*	203[abc] [41 to 344]	**258** **[91-403]**	**Medium/ high**	[a]Woodland Carbon Code (2021) [b]Vanguelova and others (2013)
30-year mixed broadleaved native woodland on mineral soil (to 1m)	151[b] [108 to 173]	*100cm*	114[a] [22 to 204]	**255** **[130-377]**	**Medium**	[a]Woodland Carbon Code (2021) Vanguelova and others (2013)
30-year mixed broadleaved native woodland (to 15cm soil depth)	55[b] [50 to 59]	*15cm*	114[a] [22 to 204]	**169** [72-263]	**Medium**	[a]Woodland Carbon Code (2021) [b]Vanguelova and others (2013)
Hedgerow						
Minimal/ Un manged Hedgerows	98.7 [66.52 to 111.93]		45.8	**144.5**	**Low**	Axe, 2015, reported in Axe 2020
Orchards						
Traditional Orchards	73.75 [47 to 111]	*30cm*	21.4 [8.6 to 230.4]	**95.15**	**Low**	Robertson and others (2012) – Top 30 cm of soils & above and below ground biomass
NOTE: Woodland confidence is medium-high rather than high because the relationship between soil and tree carbon is not clear – maximum growth is probably on sites with medium soil carbon						

Table 2.2 Summary of carbon flux values reported in the literature for woodland, tree and scrub habitats

Habitat Description	C flux ($t\ CO_2e\ ha^{-1}\ y^{-1}$)	Range (if possible)	Confidence	References
Woodland				
Mixed native broadleaved woodland (100 year)	-7	-2 to -13	Medium	Woodland Carbon Code (2021) Thomas and others (2011) Poulton and others (2003) Ashwood and others (2019) Rates averaged over 100 years
Mixed native broadleaved woodland (30 years)	-14.5	-2.5 to -25.5	Medium	Woodland Carbon Code (2021) Ashwood and others (2019) Rates averaged over 30 years
Hedgerow				
Hedgerows	-1.99[a]	-3.67[b] to -1.67[a]	Low	[a] Robertson and others (2012) [b] Falloon and others (2004)
Orchards				
Traditional orchard with low intensity management	-2.89	-5.89 to +1.65	Low	Robertson and others 2012
Intensive orchard	-5.99	-7.77 to -4.21	Low	Robertson and others 2012

2.2 Woodlands

England was once largely covered by forest, the ‘wildwood’. Within the forest there would have been more open areas and, with mosaics of scrub and grassland as well as closed canopy forest, although the extent of these is debated. Most of the tree cover was cleared before the medieval period. Today woodland covers 10 per cent of England (compared to 13 per cent for the whole UK). Of this, only 12 per cent (1.2 per cent land area) has a continuity of forest cover through recorded history and is termed ancient semi-natural woodland. Ancient woodland is not untouched natural forest - it has been subject to varying degrees of management over millennia - but it retains native tree and ground flora species with a high diversity of other species, and soil that is relatively undisturbed. Other woodlands have been planted or regrown naturally on land which had formerly been cleared for agriculture and extensive areas of formerly open peatlands have also been drained and planted. Some woodlands were converted from ancient semi-natural broadleaved stands to plantations for timber production in the past 100 years and are referred to as Plantations on Ancient Woodland Sites (PAWS). These typically retain remnant biodiversity and relatively intact forest soils. Overall, 74 per cent of English woodland is broadleaved, most of which are native species, compared to 51 per cent for Great Britain as a whole (Forest Research 2020). Much of the 26 per cent of woodland that is conifer is in plantations, such as the extensive areas of Kielder and Thetford forests, although best practice for forestry now encourages diversification of species and the support of biodiversity. The main tree species in English forests and the area of land which they cover are shown in table 2.3.

This chapter focuses on native broadleaved woodlands which are important for biodiversity and offer the potential to sequester and store significant amounts carbon. Non-native conifer production forests are included for comparison; more detailed information on timber production systems can be found in Forestry Commission and Forest Research publications (eg Morison and others 2012).

Table 2.3 Area covered by the major trees species in England and Great Britain (GB) (Forest Research 2020). Note, these data do not include trees outside of woodland or young trees.

Species or group of species	**Broadleaf (B) or conifer (C)**	**Native to Great Britain?**	**Area in England 1000 ha**	**Area in GB 1000 ha**
Oaks	B	Yes (sessile and pedunculate oak)	167	219
Ash	B	Yes	123	157
Birch	B	Yes	96	236
Sitka spruce	C	No	80	665
Sycamore	B	No, but present in northern France; long established and naturalised.	75	106
Beech	B	Yes (at least in South and East)	72	94
Hazel	B	Yes	65	87
Scots pine	C	Yes (at least in the Scottish Highlands); long established and naturalised across the whole of Britain)	61	218
Hawthorn	B	Yes	57	73
Alder	B	Yes	31	58
Willows	B	Yes	41	65
Corsican pine	C	No	40	46
Larches	C	No	40	126
Sweet chestnut	B	No, native to southern Europe but long established and naturalised in the south of Britain.	28	29
Norway spruce	C	No	27	61
Douglas fir	C	No	25	46
Lodgepole pine	C	No	8	100
Other broadleaves			146	212
Other conifers			24	40
All broadleaves			**902**	**1 337**
All conifers			**307**	**1 308**

From a conservation perspective all native woodland in England is included in the 'lowland mixed deciduous woodland' broad habitat. There are also a number of priority habitats (table 2.4). Carbon storage and sequestration have been studied in very few semi-natural woodlands in England (a notable exception being Wytham Woods, Oxfordshire) but data are available for the main species of trees, often collected within a forestry context. We have therefore taken a species-based approach rather than looking at plant communities explicitly, but this can be approximately mapped onto habitat classifications as shown in table 2.4.

Table 2.4 Broadleaved woodland habitats in England and their principal trees

Broad habitat	**Principal tree species**
Lowland mixed deciduous woodland	Oak, birch, ash, beech, hazel, hawthorn, alder, willows, holly, yew
Priority habitats	
Beech and yew woodland	Beech
Upland oak woodland	Oak
Upland mixed ash woodland	Ash
Wet woodland	Alder or willows
Wood pasture and parkland	Oak (and others)
Traditional orchards	

Broadleaved woodland is often managed for multiple objectives including nature conservation and amenity as well as timber production. Many, particularly, small, woodlands are not actively managed, although small scale production of firewood has increased in recent years. In some cases, a conscious decision may be made to take a minimum intervention approach to management, for either conservation or research purposes.

2.2.1 Overview of carbon cycling in woodlands

Woodland is critical to achieving net zero because of its potential to take carbon out of the atmosphere, particularly when creating new woodland. In all ecosystems, carbon is taken up by plants through photosynthesis and released by plants, animals and microorganisms during respiration (figure 2.1). Woodlands are unique in that a significant proportion of the carbon that is taken up is stored above ground in wood and this builds up over time as photosynthesis (primary productivity) exceeds total respiration. In a deciduous woodland carbon in leaves returns to the soil annually, and much of it returns to the atmosphere as a result of respiration by decomposer organisms within a period of a few months, although some is integrated into the soil and dead wood can take years or even decades to break down. In a new woodland, primary productivity exceeds ecosystem respiration by a large margin and the woodland acts as a substantial carbon sink. Over time, as organic matter builds up, total respiration increases and after a long period (centuries) an equilibrium is approached as net sequestration falls.

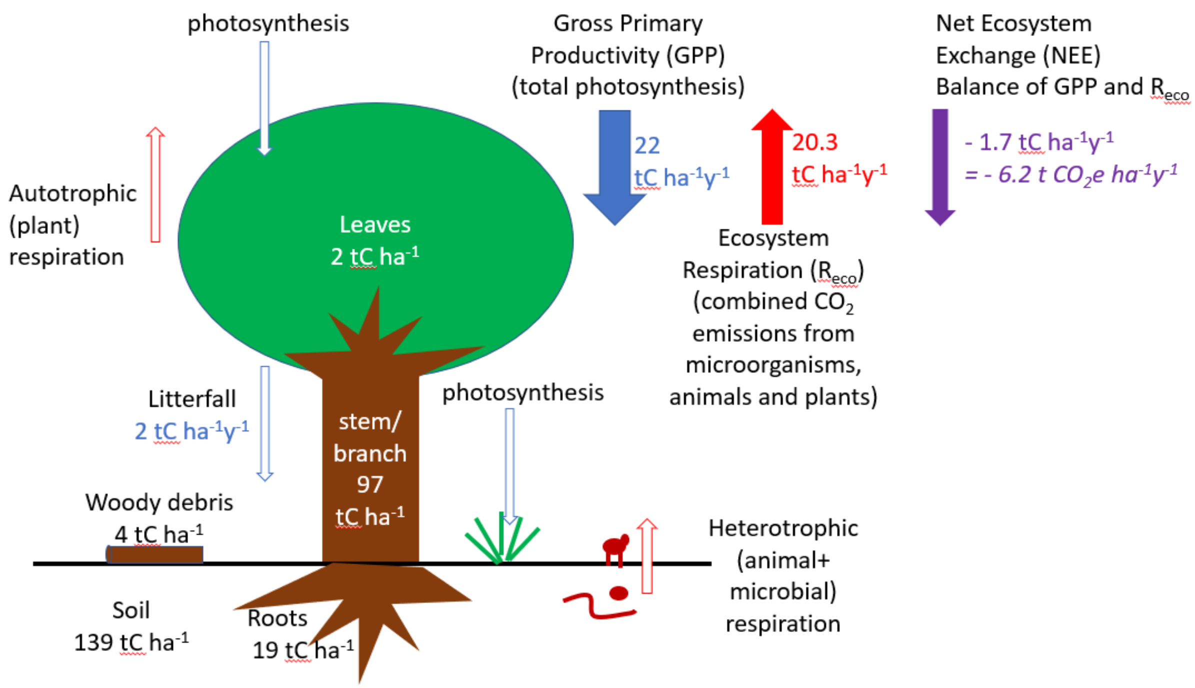


Figure 2.1 Simplified woodland carbon cycle. Blue arrows represent carbon uptake through photosynthesis, red arrows represent carbon releases to the atmosphere from respiration. Measured values of stocks and fluxes of carbon in Wytham Woods from data in Butt and others (2009) and Fenn and others (2015)

2.2.2 Carbon storage and sequestration in woodlands

The total carbon stock stored within UK forests is about 4,000 Mt CO_2e (Forest Research 2020). Assessments for international reporting indicate that it has increased since 1990 (table 2.5). The carbon stored in forest soils accounts for around 70 per cent of total forest carbon stock (data reported to 1 m depth), although much of this reflects forests which have been planted on drained peatlands rather than natural forest systems and is decreasing as a result of emissions to the atmosphere and in watercourses (Evans and others 2017).

Table 2.5 Forest carbon stock for the UK and England (Mt C). Data from Forest Research (2020) (original data presented at Mt CO_2e)

	UK					**England**
	1990	**2000**	**2010**	**2015**	**2020**	**2020**
Carbon in above-ground biomass	102	131	160	172	184	92
Carbon in below-ground biomass	37	47	57	62	66	33
Carbon in dead wood	35	38	39	40	41	17
Carbon in litter	45	48	50	51	52	22
Soil carbon to 1 m depth	645	690	716	743	752	235
Total forest carbon	864	954	1022	1068	1094	399

Vanguelova and others (2013) reported results of a survey of soil carbon across 167 woodland sites across Great Britain (72 in England) in which samples were taken to 0.8 m depth and interpolated to 1 m. This is the most thorough assessment of soil carbon in woodlands. The average carbon stock down to 80 cm depth for seven main soil types ranged between 108 and 448 t C ha^{-1} with maximum values from 511 to 927 t C ha^{-1}. Based on these data, national estimates of total forest soil carbon stocks for England, Wales and Scotland were 163, 46 and 337 Mt C, respectively, with an additional 17, 4 and 21 Mt C within surface organic layers (litter and fermentation horizons). Field and others (2020) estimated that carbon stock within High Conservation Value woodland was 0.08 Gt C (80 Mt C) across the UK for subset of woodlands identified as being of high value for nature conservation but this only considered soil to 30 cm depth.

Carbon contents for different soil types across the UK are summarised in Table 2.6 to 1 m and 15 cm depth. For comparison the Countryside Survey 2007 (CS2007) which has a wider sample of different vegetation types but is not specifically focused on woodlands, found that in the top 15 cm of soil, there was 72.9 t C ha^{-1} for broadleaved, mixed and yew woodlands and 81.4 t C ha^{-1} for coniferous woodland across different soil types (Emmett and others 2010).

Table 2.6 Carbon stock (t C ha^{-1}) in different soil types under woodland (Vanguelova 2013) to 15 cm and 1 m depth

Soil type	Carbon stock to 1 m depth (t C ha^{-1})	Carbon stock to 15 cm depth
Rankers and rendzinas	108	58
Brown earths	152	52
Podzols and Ironpans	154	50
Surface-water gleys	167	59
Groundwater gleys	173	58
Peaty gleys/podzols	362	107
Deep peats	539	97

UK forests removed an estimated 18.2 Mt CO_2e in 2018, the last year for which full data are available (Brown 2020b). This represents around 4 per cent of total UK greenhouse gas emissions of 451.5 Mt CO_2e in 2018. The Committee on Climate Change (CCC 2020) in a recent report on climate change and land use suggested increasing UK forest cover from 13 per cent to at least 17 per cent by 2050 by establishing around 30,000 ha of new woodland each year. Together with improved woodland management they estimated that this would deliver an additional annual sequestration of 14 Mt CO_2e by 2050 with an additional 14 Mt CO_2e from using harvested materials as replacements for fuel and materials.

Forests are the largest carbon sinks in the UK and offer the potential for offsetting emissions from other sources. It should however be noted that even the most ambitious afforestation plans will only be able to offset a small proportion of current emissions, so large cuts in other sectors will still be necessary to achieve net zero GHG emissions. Furthermore, woodlands generally take ten to thirty years to become significant sinks, so only those planted in the near future will contribute markedly to the 2050 net zero target. The potential for new woodland to store carbon has been well-studied in the context of forestry and the Woodland Carbon Code[9] provides look up tables of carbon sequestration by different species, different timber yield classes, tree spacing and whether thinned or not. These data are based on long-term measurements of tree growth for timber production. There have also been a number of detailed studies of carbon fluxes and storage within a small number of woodland sites.

From a starting point of agricultural land, establishing woodland of all types will normally create a carbon sink, except for sites on peat and some organo-mineral soils, where soil disturbance leads to large greenhouse gas emissions (see below). Left to natural processes, woodlands will go on taking up carbon for centuries, although the net rate of uptake declines (Figure 2.2). Even after trees reach maturity, they continue to take up carbon, new trees fill gaps and organic matter builds up in the soil and dead wood. Eventually the net rate of uptake starts to approach zero as rates of respiration from trees, decomposers and other organisms increase. Old woodlands are however very valuable both as significant carbon stores and important sites for biodiversity. Large old trees, such as oaks in conservation sites store a large amount of carbon for the long-term and provide unique habitats for specialist species.

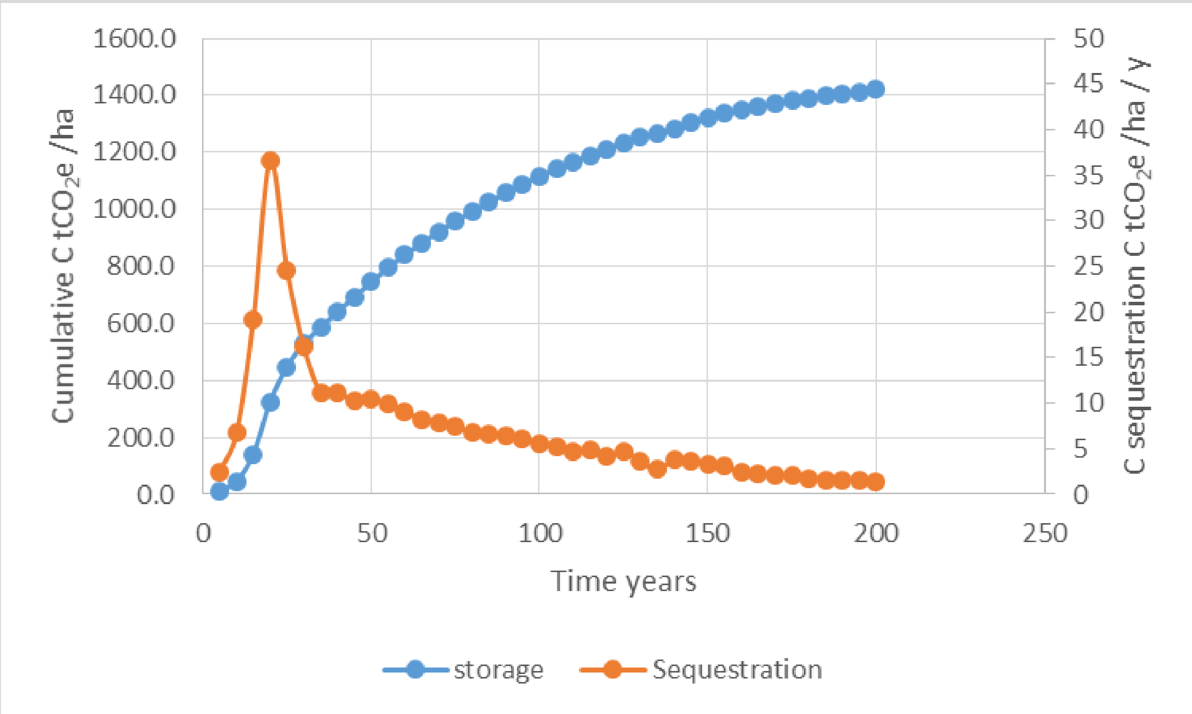
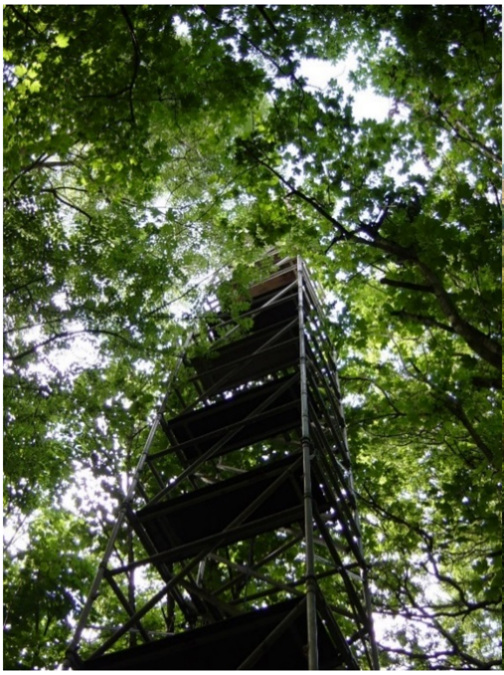


Figure 2.2 Illustration of how carbon sequestration of new woodlands peaks after a few decades, whereas carbon storage increases towards an equilibrium. (Based on Woodland Carbon Code data for un-thinned Yield Class 8 Oak in 5-year time intervals on a mineral soil with minimal soil emissions. Note: the modelling of early growth is limited by a lack of data so the timing and height of the early peak should only be treated as illustrative.)

[9] https://www.woodlandcarboncode.org.uk/

2.2.3 Field measurements of carbon fluxes and storage in native woodlands

Field measurements of CO_2 fluxes have been made at a small number of woodland sites in the UK using a variety of techniques including eddy covariance systems on flux towers which measure net gas exchange above large areas of woodland. The longest running flux tower in broadleaved woodland is in an oak plantation, with shrub understorey, the Straits Inclosure, planted in the 1930s, at Alice Holt Forest, Hampshire. The mean annual Net Ecosystem Exchange (NEE: the overall balance between photosynthesis and respiration, so the net sequestration rate) over 12 years was -18 t CO_2e ha^{-1} y^{-1} (Wilkinson and others 2012; Morison and others 2012). Another flux tower has operated for shorter periods of time at Wytham Woods, Oxfordshire over an area of unmanaged ancient semi-natural woodland, with a range of species, including older oak, sycamore and hazel (photo 2.1 and 2.2). Over two years, NEE was much lower at Wytham compared to Alice Holt: -4.4 t CO_2e ha^{-1} y^{-1} (Thomas and others 2011; Fenn and others 2015); this is however still a significant carbon sink compared to most semi-natural habitats. The rate of photosynthesis (Gross Primary Productivity) was similar in both cases, but there was a higher level of ecosystem respiration at Wytham, particularly from the trees themselves (Fenn and others 2015), which may reflect the older trees at this site and the build-up of decaying deadwood. This is ancient woodland, with some trees in excess of 200 years old and much of the area around the flux tower had not had timber removed for several decades or more (Thomas and others, 2011). Another study from Wytham Woods (Fenn and others, 2015) derived NEE from component fluxes and found a value of 6.2 t CO_2e ha^{-1} y^{-1}. In practice both approaches are subject to errors and fluxes vary year to year: in this context there is a good degree of consistency between the estimates.




Photos 2.1 and 2.2 Wytham Woods eddy covariance flux tower (left) and view over the canopy from the tower (right) © Natural England / Mike Morecroft

A unique study recorded carbon storage in soil and trees over 118 years at Rothamsted Experimental Station, Hertforshire, following cessation of arable agriculture and natural colonisation by trees. This showed that one site, Geescroft, on which oak dominated woodland established, accumulated carbon at an average rate of 2.0 tC ha^{-1} (7.3 tCO_2e ha^{-1}) and that another, Broadbalk, with a higher pHsoil, in which a mixed species ash – sycamore woodlands established, accumulated carbon at an average rate of 3.39 tC ha^{-1} (12.4 tCO_2e ha^{-1}) (Poulton and others, 2003). Both showed relatively slow initial carbon accumulation followed by increasing rates, which Poulton and others attributed to increasing nitrogen availability over time.

2.2.4 Carbon sequestration in tree biomass

Different tree species grow at different rates and allocate different amounts of carbon to different parts of the tree, for example between branches, stems and wood. Broadleaves typically produce a smaller timber volume compared to conifers, because they have a higher proportion of the wood in branches and roots than in the stem, and they sequester more carbon per unit of volume because of their greater wood density (Morison and others 2012). It is therefore important not to directly equate the production of timber volume with carbon sequestration when comparing between species, particularly between conifers and broadleaves. Rates of timber production and carbon sequestration within the same species vary greatly between sites (figure 2.3); planting the right tree in the right place is essential to maximise carbon sequestration.

Timescale has a significant impact on which species sequester most carbon. The fastest growing species in the 30-year time interval for meeting the net zero commitment, are not necessarily the same as those that will store most carbon in the longer term. Amongst broadleaves, light demanding species such as birch and sycamore, grow quickly on open site, whereas shade-tolerant species such as beech, tend to grow slower at first but may sequester more carbon over 100–200 years as the canopy closes. Shade tolerant species are also better able to regenerate under the canopy of an existing woodland and so maintain tree cover in an unmanaged wood, or with continuous cover forestry.

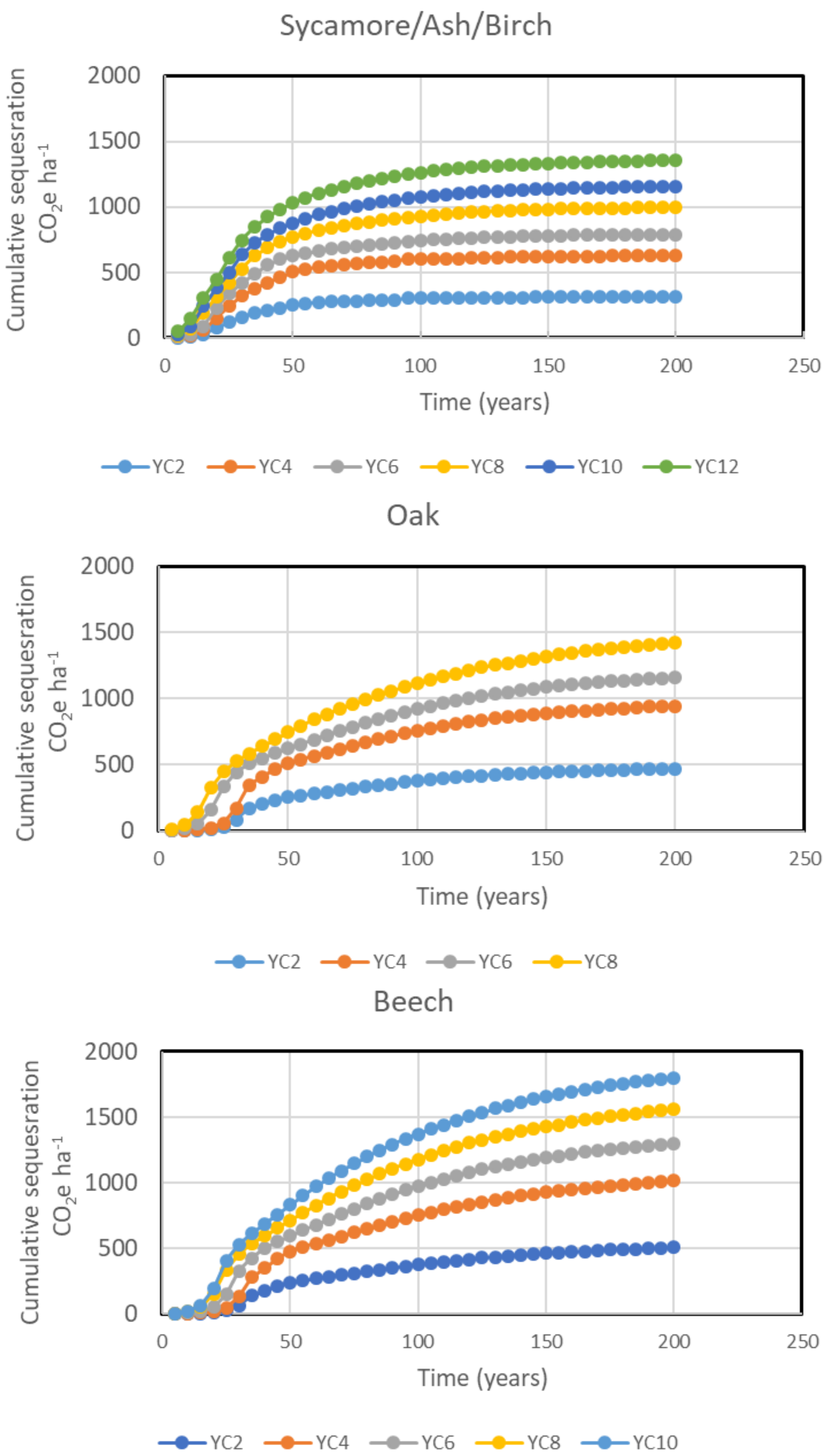


Figure 2.3 Cumulative sequestration of carbon (t CO_2e ha^{-1}) by different trees species at a range of Yield Classes (YC), using Woodland Carbon Code data (Woodland Carbon Code, 2021). Yield class is an index of the potential productivity of even-aged stands of trees. It is based on the maximum mean annual increment of cumulative timber volume in units of cubic metres per hectare per year. The range of variation in yield class within species is larger than the differences between species

In much of England, native broadleaved species can sequester similar or higher amounts of carbon compared to conifers (figure 2.4). In the wetter areas in the north and west of England, which also typically have more acidic soil, non-native conifers are often capable of sequestering more carbon than broadleaves. Which species has the greatest potential to sequester carbon in a particular place depends therefore on site-specific soil and climatic conditions (figure 2.5). To optimise carbon uptake, decisions on which species to plant need to be made on a site by site basis, taking account also of other objectives in creating woodland, including whether or not timber production is an objective (see section 2.2.6 below). Planting for carbon alone could create risks for biodiversity, hydrology or ecosystem services for people. It is also important to ensure a diversity of species are planted to increase resilience to climate change and other threats (see 2.2.9 below). The Ecological Site Classification[10] tool can help to identify which tree species are likely to grow best in different places and also allows an assessment of changing suitability with climate change. Optimising climate change mitigation potential and other benefits also needs to take account of silvicultural techniques and management.

[10] Ecological Site Classification Tool

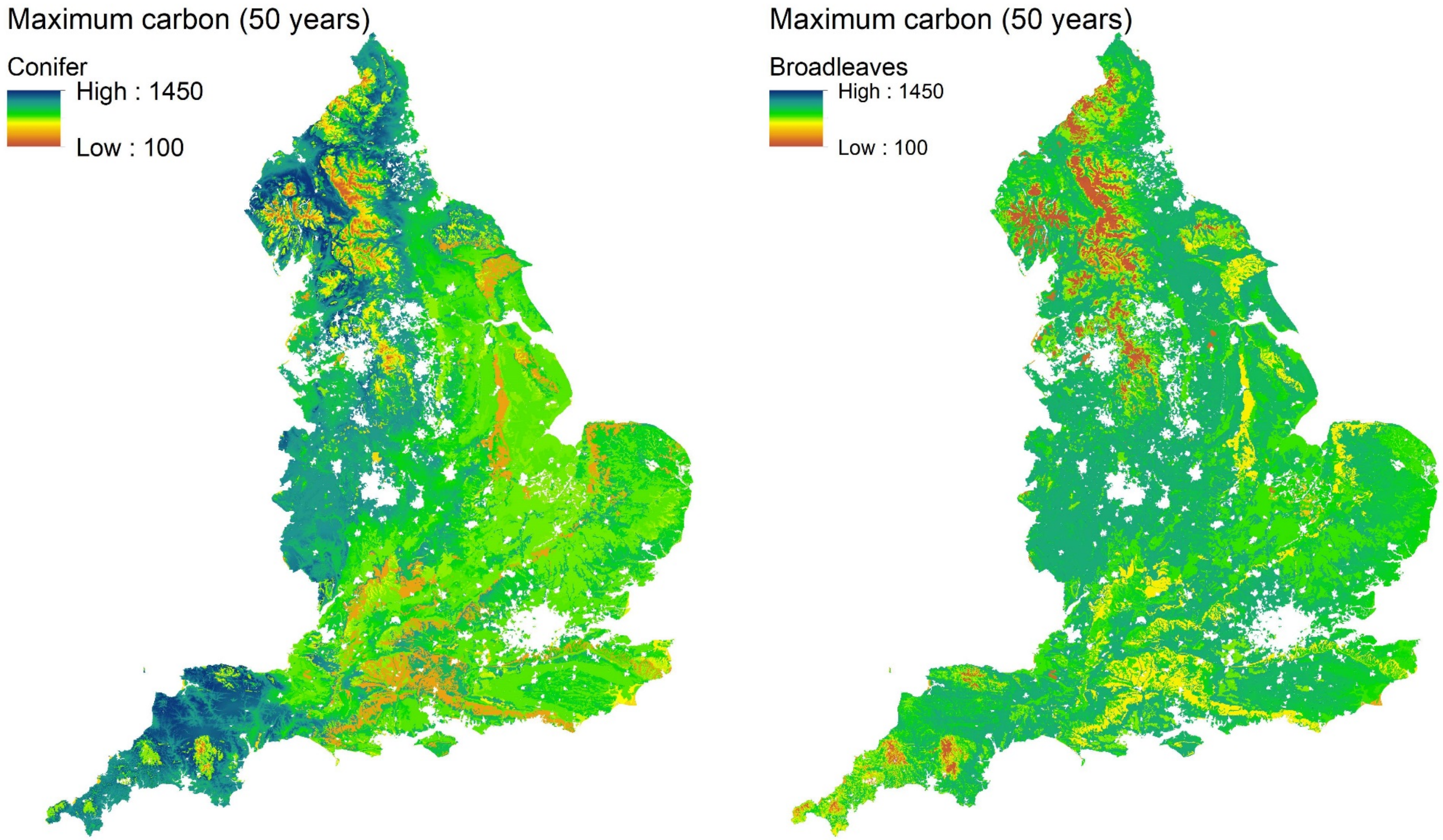


Figure 2.4 Maximum potential CO_2 uptake of a) conifer and b) broadleaved species in different locations across the UK over 50 years from woodland planting. Map is derived from Ecological Site Classification data to identify potential yield of different species from climate and soil information and Woodland Carbon Code to derive carbon sequestration from potential yield. Units: t CO_2 ha^{-1}. Source: Forestry Commission/Forest Research analysis

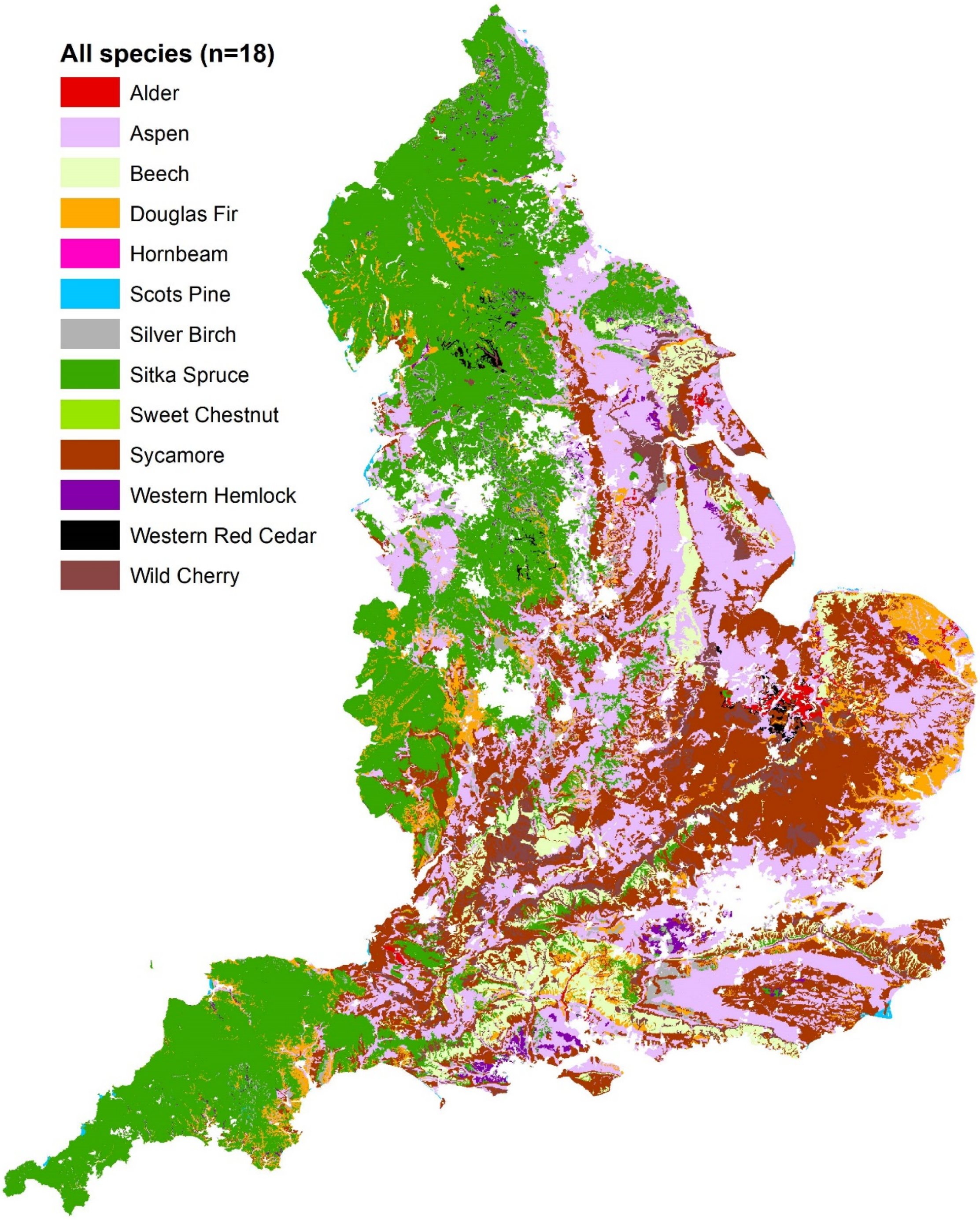


Figure 2.5 Species with highest indicative potential sequestration of carbon in different parts of the UK over 50 years, based on Ecological Site Classification and Woodland Carbon Code, together with national soil mapping (it should not be taken as a guide to site specific-decision making) Source: Forestry Commission/Forest Research analysis

2.2.5 Impacts of soil type and management interventions on woodland carbon

Soil type and management is an important aspect of the carbon balance of woodlands, as well as a major determinant of yield class and species suitability.

Soil carbon typically builds up over time in new woodland on arable soils. Ashwood and others (2019) showed that soil carbon under secondary woodland on ex-arable land on gley soils in the Midlands accumulates at about 0.5 t C ha^{-1} y^{-1} (1.8 t CO_2e ha^{-1} y^{-1}) and after 50–100 years soil carbon stock was not significantly different from ancient woodland. This rate of increase in carbon stock in new woodland was comparable to the build-up of soil carbon in the woodland succession study at Rothamsted Research Station (Poulton, 2003). Whilst Ashwood and others (2019) showed that soil carbon stock was significantly higher in woodland compared to arable land, it was comparable to that of long-term pasture on equivalent soils (although the total carbon stock, including carbon in the trees themselves, will be higher in the woodland than the pasture). Poulton and others (2003) however showed higher rates of soil carbon under woodland than grassland.

A similar process of soil carbon accumulation occurs with rotational management of forests for timber production. Benham and others (2012) reported that soil carbon stock increased with age of forest stands in a chronosequence in oak woodland on clay soils at Alice Holt Forest, Hampshire. However, a site which had been clear felled and ditched to promote the establishment of new trees had 7 t C ha^{-1} less carbon in the soil than a mature stand. Drainage, soil preparation and tree planting disturb the soil and can lead to significant releases of carbon through oxidation. Losses in establishing new woodland can be particularly large in organic soils with high carbon stocks and where the disturbance is large, as in the case of ploughing, it can also continue over the long-term. The Woodland Carbon Code provides an approach to estimating soil emissions from tree planting with different cultivation methods (West 2011), based on measured soil carbon stocks where available and typical figures for different existing land uses where site specific data are not available. The UK Forestry Standard[11] (UKFS) also requires that soil disturbance through cultivation should be minimised, while still achieving successful establishment and guidance for application in England to support this has recently been published.

Large scale surveys of topsoil carbon contents across all land uses have found contradictory messages for changes in soil carbon in woodlands over time. Bellamy and others (2005) found small declines in the concentration of organic carbon in the top 15 cm of soil in both deciduous and coniferous woodland in data from the National Soil Inventory of England and Wales between 1978 and 2003. Results from the Countryside Survey (Reynolds and others 2013) showed increases between 1978 and 1998, followed by decrease to 2007. Kirby and others (2005) in a survey targeted on woodland sites, found no overall change in soil organic matter across 103 woodland sites between 1971 and 2001, with significant increases on 15 sites and declines on 8. Differences between surveys reflect a combination of differences in

[11] The UK Forestry Standard

methodologies, site differences and year to year changes with management and climate. Results from topsoil surveys of fixed depth need to be interpreted carefully as Benham and others (2012) show there can be significant differences in both the depth and soil carbon content of different soil horizons over time.

Woodland can establish naturally without planting in most parts of Britain and this is a potentially important mechanism for woodland creation that could eliminate the carbon losses from soil disturbance which are associated with planting. It does however require the right conditions including a local seed sources from existing woodland and protection from excessive grazing animals, particularly deer. In the early stages competition from tall herbaceous species can also create problems for seedling establishment (Hutchings and others, in press) and establishment may be slower and less even compared to planting.

Woodlands on peat soils

Planting trees on peat soils, which in most cases in the UK would naturally have been open habitats, will lead to greenhouse gas emissions which can reduce or negate the benefits of the carbon sequestration by the trees (Evans and others, 2017; Figure 2.6). Carbon can also be lost in water draining from afforested peatlands as particulate organic carbon (POC) or dissolved organic carbon (DOC) (Campbell and others 2019). Peat contains large amounts of carbon in partially decomposed organic matter which accumulates in waterlogged conditions; tree planting often requires cultivation and draining of the peat, leading to oxidation and the emission of CO_2. Although a high yielding plantation might sequester more carbon than emitted from the peat, a low yielding one would not. The most recent estimates for international reporting (2021 update to the Emissions Inventory for UK Peatlands – to be published in April 2021 in the 2021 UK GHG Inventory[12,13]) suggest typical net carbon fluxes range from emissions of 4.15 t CO_2e ha^{-1} to a small sequestration of -0.16 t CO_2e ha^{-1} (emissions of 5.46 to 1.15 t CO_2e ha^{-1} if nitrous oxide emissions are included). Even a high-yielding forest on peat will effectively be replacing carbon in a form which has remained stable over millennia, for one that cycles much more rapidly and may not represent long-term storage. Forestry plantations on drained peat may also affect the hydrology of surrounding areas and lead to degradation of a larger peat body. The UK Forestry Standard excludes planting on deep peat soils (over 50 cm) but recent evidence shows that net carbon sequestration on shallow peats and many carbon-rich organo-mineral soils can be low (Brown 2020a; Matthews and others 2020).

Although most peatlands (see Chapter 4) in the UK were largely treeless (in contrast to tropical peatlands), wet woodland habitats on peats can form naturally in valley bottoms with impeded drainage. The dominant alder and willow species in these habitats can withstand waterlogging. This is now a rare woodland type that has been drained and lost from our landscapes on a large scale. It has a double benefit for carbon sequestration and

[12] https://unfccc.int/ghg-inventories-annex-i-parties/2021
[13] NAEI website: https://naei.beis.gov.uk/reports/

storage in that peat can form in the waterlogged conditions as well as supporting tree growth. In the right locations, wet woodland has the potential to provide multiple benefits including natural flood management when established on floodplains and water pathways by slowing and reducing run off. There is some uncertainty around the likely methane emissions from wet woodland at different stages of development and the potential for nitrous oxide emissions from alder, which is a nitrogen-fixing species: more research is needed on the likely GHG balance for this understudied habitat.

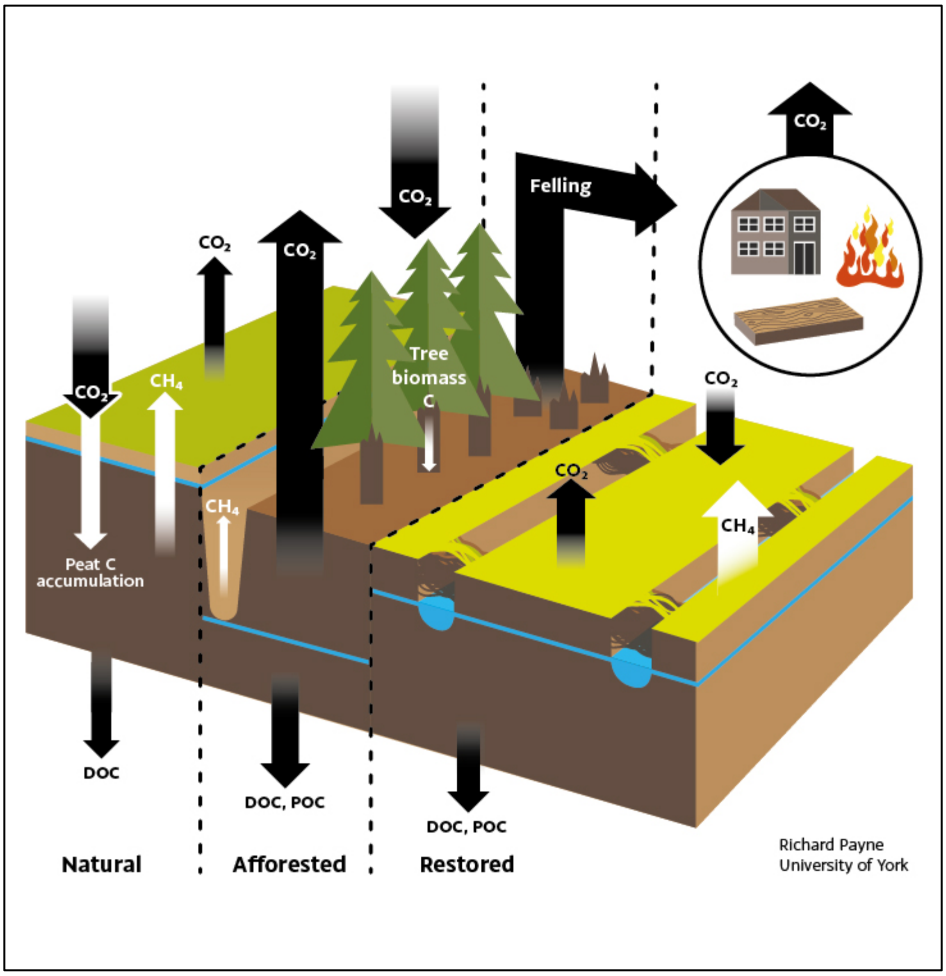


Figure 2.6 Conceptual diagram of key carbon cycle pathways and changes with peatland afforestation and restoration (Payne & Jessop 2018)

2.2.6 Timber production

Comparing woodlands which are managed for timber production with those that are primarily managed for biodiversity conservation is not straightforward. The long-term benefit for GHG emissions depends on the use to which the timber taken from managed forests is put. Carbon can be stored for the long-term, if, for example, it is used in construction. At the other extreme it will be released quickly if used, for example, for newspaper or fuel (Harmon 2019). Where wood products do not provide a long-term carbon store, but woodlands are replanted after harvest, there is still a net benefit for climate change mitigation from the conversion of agricultural land to forestry, given that at any one time, most areas under active forestry will be growing trees and the average carbon stocks on the land will have increased. In addition, if wood is used to replace the use of a fossil fuel or a material that has high CO_2 emissions associated with its production, this will also contribute to climate change mitigation. It is important to note however that as other renewable energy sources such as wind and solar increase or if nuclear energy capacity is increased, the mitigation benefits of biomass energy will become less important. If bioenergy with carbon capture and storage (BECCS) is introduced, this will provide long-term storage carbon as well as energy substitution. There a wide range of issues around BECCS which are beyond the scope of this report but it should be noted that the risks and opportunities will depend on the scale of deployment, the location of bioenergy crops and the nature of the feedstock. Small-scale use of waste materials from forest harvesting might have very little environmental impact, whereas large scale planting of non-native trees could threaten biodiversity, food production and a wide range of ecosystem services.

Where timber is harvested, higher overall rates of carbon uptake can be maintained over time. Fast growing, non-native conifer species are the mainstay of the UK timber industry and in the right conditions, species like Sitka Spruce can produce high sequestration rates (Morison and others 2012). The measured rate of net sequestration at a flux tower in a Sitka spruce plantation with a typical yield class (14–16 m^{-3} ha^{-1} y^{-1}) on a podzolised brown earth soil at Griffin Forest in Perthshire was 24 tCO_2e ha^{-1} y^{-1} (Clement, Moncrieff & Jarvis 2003; Jarvis and others 2009). However, soil disturbance during establishment needs to be considered, particularly on soils with a high organic matter content. This is particularly problematic on peatland (see above) but studies in Scotland have shown that on other highly organic soils, this can also be sufficient to negate the sequestration of the trees over large areas of land (Brown 2020a; Matthews and others 2020). Woodland management techniques, such as continuous cover forestry, can decrease these losses, as can minimising cultivation.

The biodiversity benefits of conifer plantations are typically less than for native woodlands (Burton and others 2018; Humphrey and others 2003), although with appropriate management they can support a range of species and the UK Forestry Standard (Forestry Commission 2017) expects forests to be managed in a way that conserves or enhances biodiversity. There are numerous intermediate strategies for removing timber whilst still maintaining a woodland with high conservation and amenity value and woodland can be managed conservation objectives. Native woodland stands supporting biodiversity can also be planted alongside those primarily intended for timber production. Decisions at a site level need to be made on a case by case basis, taking account of the full range of objectives for a woodland, the soil and climate. It is however important to note that in large parts of England a native broadleaved woodland will sequester as much carbon as non-native conifers. It is also the case that a woodland managed with a minimum intervention approach can go on sequestering relatively large amounts of carbon for many decades if not centuries and is a realistic option if carbon and biodiversity are the main aims for a site.

The uncertainty around whether and how forests will be managed, and the use of any harvested wood, complicates the quantification of long-term carbon sequestration and storage by new woodlands. Over the 30-year timeframe in which the UK is seeking to achieve net zero GHG emissions, carbon sequestered in new woodlands will largely remain unharvested, but it is important to plan forestry with a longer-term perspective. In making a full assessment of climate change mitigation potential, it is also important to ensure decisions in the UK do not lead to increasing emissions in countries, for example by importing timber that is unsustainably produced.

2.2.7 Quantitative assessment of carbon storage and sequestration by native woodland woodlands

The Woodland Carbon Code provides a way of assessing the potential to sequester and store carbon in woodlands across the whole of England (and the rest of GB), based on the CSORT model calibrated with long-term data on timber production. This provides a way of exploring the likely range of woodland carbon storage and sequestration in woodland across England. Table 2.7 summarises indicative values of carbon storage and sequestration for native broadleaved woodlands based on data taken from the Woodland Carbon Code Look-Up Table (Woodland Carbon Code, 2021). This provides typical data for net carbon sequestration, according to species, age, spacing and thinning regime. At present species-specific data for broadleaved species are restricted to beech and oak, but there is also a generic ‘mixed broadleaved’ dataset, based on data for sycamore, ash and birch. The data have been combined with data on soils derived from Vanguelova and others (2012) and Ashwood and others (2019).

On this basis, a representative carbon sequestration rate for mixed broadleaved woodland of 100 years old, which could be taken as a typical value for existing woodland of high conservation value, is approximately uptake of 7 t CO_2e ha^{-1} y^{-1} (averaged over the 100 year period). Single species beech and oak stands of the same age may be more productive than mixed broadleaves, although are likely to be less resilient to climate change). This is consistent with the field measurements reported in 2.2.3 above, accepting that there is a wide variation between sites.

At 30 years the rate of uptake averaged over the whole period will be higher; approximately 14.5 t CO_2e ha^{-1} y^{-1} because of the high sequestration rates seen in the early decades of growth, after establishment. This is important in assessing the potential to meet net zero greenhouse gas emissions by 2030, although subject to more uncertainty than estimates over longer timescales.

Table 2.7 Carbon storage and sequestration rates for native broadleaved woodland: representative values and ranges. Data based on Woodland Carbon Code (WCC) using representative yield classes in the middle of the range and maximum and minimum values from highest and lowest quoted yield classes. Values assume un-thinned stands. Soil carbon storage is based on data from Vanguelova and others (2013); data are average values for all soil types except deep peat and peaty gleys/peaty podzols; ranges represent the variation between different soil types, not site-to-site variability. Soil carbon accumulation rates of 0.5 t CO_2 ha^{-1} y^{-1} are assumed (from Ashwood and others 2019). Broadleaves are assumed to be planted on mineral soils and with minimum soil disturbance or as a result of natural colonisation, so no allowances are made for emissions in planting. Tree spacings were 1.5 m for mixed broadleaves and 1.2 m for beech and oak. Combined soil and tree carbon ranges are derived by adding respective minima and maxima together; they should only be treated as an approximate guide. 30 y = 30 years; 100 y = 100 years

Habitat	Tree biomass or soil	Carbon storage t C ha^{-1}		Carbon sequestration averaged over time period t CO_2e ha^{-1} y^{-1}		Notes
		Range	Representative value	Range	Representative value	
Broadleaved mixed woodland (SAB) 30 y	trees	22–204	114	3–25	14	Representative: YC6 Range YC2–12
	soil	108–173 (50-59 to 15 cm depth)	151 (55 to 15 cm depth)	0.5	0.5	
	total		**265**	**3.5–25.5**	**14.5**	
Mixed broad-leaved woodland (SAB) 100 y	trees	41–344	203	2–13	7	Representative: YC6 Range YC2–12
	soil	108–173 (50-59 to 15cm depth)	151 (55 to 15 cm depth)	0	0	
	total	**149 to 517**	**254**	**2–13**	**7**	

Table 2.7 Continued

Habitat	Tree biomass or soil	Carbon storage t C ha^{-1}		Carbon sequestration averaged over time period t CO_2e ha^{-1} y^{-1}		Notes
		Range	Representative value	Range	Representative value	
Beech **30 y**	trees	9–144	89	1–18	11	Representative: YC6 Range: YC2–10
	soil	108–173 (50-59 to 15cm depth)	151 (55 to 15 cm depth)	0.5	0.5	
	total		**240**	**1.5–15.5**	**11.5**	
Beech **100 + y**	trees	51–374	266	2–14	10	Representative: YC6 Range: YC2–10
	soil	108–173 (50-59 to 15cm depth)	151 (55 to 15 cm depth)	0	0	
	total		**417**	**2–14**	**10**	
Oak **30 y**	trees	12–144	121	1–18	15	Representative: YC6 Range: YC2–8
	soil	108–173 (50-59 to 15cm depth)	151 (55 to 15 cm depth)	0.5	0.5	
	total		**272**	**1.5-18.5**	**15.5**	
Oak **100 + y**	trees	51–304	251	2–11	9	Representative: YC6 Range: YC2–8
	soil	108–173 (50-59 to 15cm depth)	151	0	0	
	total		**402**	**2–11**	**9**	

2.2.8 Wood pasture

Wood pasture sites are a form of wooded habitat where the key features are the presence of grazing animals, open-grown trees, scrub, open habitat and relatively large volumes of dead wood and wood decay. The Vera model (Vera 2000) suggests that an open woodland structure or mosaic, maintained by the grazing of large herbivores, similar to wood pasture, would have been a significant component of natural temperate woodlands. Wood pastures contains many of England's oldest trees, which create a unique habitat including significant amounts of deadwood within standing trees (Harding & Rose 1986). The habitat is especially important for many rare and specialised species associated with wood decay and ancient trees, particularly lichens, fungi and invertebrates. Wood pasture may overlap with a wide variety of other priority habitat types, including grassland, heathland and scrub and it grades into both woodland and traditional parkland. Sites may originate as a result of traditional land management originating in (or before) the medieval period (wooded commons, hunting forests, chases and parkland) or as designed landscapes incorporating earlier agricultural landscapes. In recent times there has been renewed interest in silvo-pastoral systems that combine trees and grazing.

The Wood Pasture and Parkland Inventory records 278,050 ha of the habitat in England but the recorded distribution is patchy, with the habitat more extensive in some areas compared to others. It is likely that the habitat is not well captured in several areas of the country, particularly in the uplands.

We have not found any studies specifically focused on the carbon balance of wood pasture or parkland. It is however possible to draw some qualitative conclusions from the nature of the habitats. Old wood pasture trees often contain a large timber volume so are significant stores of carbon, as well as being important biodiversity and landscape features and are important to protect. Within a closed canopy ancient semi-natural woodland, carbon is very unevenly distributed between trees and it is possible that wood pasture may store similar amounts of carbon in some circumstances. For example, in an 18 ha plot at Wytham Woods, Butt and others (2009) estimated that approximately 29 per cent of the carbon was in oak (*Quercus robur*) trees, which are mostly large old trees, and made up less than 2 per cent of the individual stems (photo 2.3). Wood pasture soils are often relatively undisturbed and may also be a valuable carbon store, depending on soil type and management history.

Large, old trees typically grow slowly and carbon losses from respiration, including from decaying wood, will eventually approach the point where they balance carbon uptake. However, wood pasture can play a role in enhancing carbon sequestration through planting new trees or allowing natural regeneration and re-wilding approaches. Trees growing in an open location with more access to light can grow faster compared to those in a closed canopy woodland. A study by Upson (2016) found that a 14 year old silvo-pastoral system with a combination of trees and grassland stored about 5 per cent more carbon (a combination of above ground and soil carbon) than the equivalent separate areas of woodland and pasture, putting this down, at least in part, to the greater size of the silvo-pastoral trees. How long this difference would continue as the trees continue to grow is not clear however; as reported above, carbon sequestration in new woodlands increases sharply over the first 20 years or so.

Photo 2.3 Oaks in wood pasture, Savernake Forest © Natural England / Mike Morecroft

2.2.9 Climate change impacts and other interactions

Tree photosynthesis and growth vary with climatic conditions and changes on a year to year basis depending on water supply and temperature (eg Morecroft 2008; Butt and others 2014). Soil respiration rates are also sensitive to temperature and water content (Yamulki & Morison 2017; Fenn and others 2010) which leads to seasonal and inter-annual differences in loss of carbon from soils.

To be an effective carbon store it is important to create woodlands that are resilient to climate change and other pressures, including novel pests and diseases. A key element is to plant mixed species stands which reduces the risk of the whole stand being impacted or even dying. Other factors that can promote resilience include mixed age structures and ensuring natural regeneration through controlling herbivore pressure. Natural regeneration and natural colonisation of trees increases genetic diversity compared to planting and builds resilience. Planting more southerly provenances in new woodland may also play a role in enhancing resilience, given likely warmer and drier summers, particularly where timber production is an important objective (Forestry Commission, 2020) but is normally avoided in conservation sites. The Natural England & RSPB (2020) Adaptation Manual provides information on adaptation and resilience building in semi-natural woodlands. In particular, it is worth noting that beech woodland, whilst an effective carbon sink at present is vulnerable to drought which is an increasing risk with climate change, particularly on the thin soils that it has generally been planted on in the south and east of England. Adaptation measures include ensuring that new beech woodlands include a range of other species and accepting beech as a native species in the north and west of England, which it did not naturally colonise following the last glaciation. There are a range of other native species, such as hornbeam and small leaved lime which are currently uncommon but are well adapted to likely future climates and would have been more widespread in the past.

Woodland in the right place can also provide a wide range of important benefits, including nature-based solutions for climate change adaptation, such as providing shade, shelter and natural flood management.

2.2.10 Woodland evidence gaps and future needs

Woodland Carbon Code Data are currently only available for 3 types of broadleaved trees – beech, oak and a generic dataset for sycamore, ash and birch, which are used as surrogates for other species. Most tree growth data are from plantations managed for timber production and may not reflect the situation in more natural woodlands with a diversity of species, tree ages and canopy structure. More information is needed for semi-natural woodland systems on the carbon balances of different combinations of species and soil types and their interactions with management. This includes an evidence need to understand the carbon balance of naturally colonised or regenerated woodlands in comparison to planted ones, which has not yet been fully evaluated.

The largest site-based surveys of soil carbon are only for topsoil to a fixed depth, usually 15 cm. They do not take account of changes in soil depth or changing patterns of carbon storage at different depths. Woodland soils are often strongly stratified with a thick litter layer and highly organic top horizon. The litter layer has often not been measured.

Evidence, regarding the impacts of different soil preparation techniques on soil greenhouse gas emissions, including natural process-based techniques (mob grazing, pigs or wild boar), have only been partially evaluated. Soil carbon emissions under a range of different management techniques, soil types, climates and weather conditions have not been fully evaluated. The extent of inter-annual variation in soil carbon fluxes is poorly understood.

Whole stand net carbon flux measurements have been made at a very small number of forest sites, covering a very limited age range, and may not be typical of the full range of forest types and ages.

Increased understanding of the synergies and trade-offs between carbon storage and sequestration, biodiversity and the wide range of other services that woodlands provide is an important evidence requirement.

Carbon stock and flux of wood pasture and parkland hasn’t been measured. Similarly, the impacts of different regeneration techniques in wood pasture.

2.3 Hedgerows

Hedgerows are human-created linear features in the farmed landscape, with woody tree and shrub species planted to keep livestock contained or to designate a field boundary. Hedges often indicate ancient field patterns, making them valued for their cultural record and contribution to landscape quality, as well as their value to biodiversity and connectivity between other habitats (Rackham 1986). They are found across the UK, and can be associated with arable or pastoral systems, in upland and lowland settings.

Hedgerow management begins initially with the shrubs first being laid, and sometimes being twisted into shape to improve their role as a windbreak or barrier. They are cut regularly to limit their growth outwards, with a short period trimming cycle every 1–3 years, and then structurally restored after approximately 40 years (Axe and others 2017). Hedges may be 1–5 metres wide (Holden and others 2019), can be dense or gappy in their structure, and encompass the ground and vegetation beneath them, including associated ditches or earth banks (Wolton and others 2014). They can be made up of a diverse range of plant species, but in England and Wales the most frequently occurring woody hedge species are hawthorn *Crataegus monogyna* and blackthorn *Prunus spinosa* (Axe and others 2017).

In the UK there is approximately 456,000 km of actively managed hedgerow, with a further 200,000 km in very poor or fragmented state (Carey and others 2007; Holden and others 2019). However, in 1945 this extent was much greater with about 1.4 million km of hedgerow in 1945 in England and Wales alone (O'Connell and others 2004). Post-war agricultural policy increased mechanisation of farming and led to the loss of thousands of kilometres of hedges in order to boost production and food security. Although the Hedgerow Regulations were introduced in 1997 to limit their removal and agri-environment schemes aim to provide incentives to create and restore them, hedgerows have continued to see declines in their length and structural condition due to neglect and lack of rejuvenation management (Staley and others 2015). Hedgerows are now recognised as an important part of the UK's approach to meeting the UK's net zero targets with a 40 per cent increase in length included in the land use change scenarios (CCC, 2020).

2.3.1 Carbon storage and sequestration in hedgerows

Hedgerow vegetation

Hedgerows have the potential to sequester and store carbon in their biomass (table 2.8). Despite this, little research has been carried out in the UK regarding the role of hedgerows with published evidence typically based on modelling approaches and underpinned by data taken from agricultural or woodland settings (Axe and others 2017). Hedge management differs from that of woodlands, with hedges routinely cut and traditionally laid to promote growth of dense woody biomass to provide a barrier. Without regular laying, after around 40 years hedges lose this density and become gappy as vegetation progresses to taller trees. Structural restoration via laying or coppicing is then required (Axe and others 2017) with hedges losing biomass in the process.

The width and height of hedgerows directly influence the amount of carbon they can store. A case study of hawthorn and blackthorn flailed hedges reported increased carbon storage with increases in hedge width and height. Hedgerows untrimmed for 3 years (height 3.5 m, width 2.6–4.2 m) contained a carbon stock of 42 ± 3.78 t C ha^{-1} in their above ground biomass, and this reduced to 40.6 ± 4.47 t C ha^{-1} and 32.2 ± 2.76 t C ha^{-1} when trimmed to a height of 2.7 and 1.9m high respectively. Minimally managed hedges at the same site were reported to store on average higher amounts of carbon in their vegetation at 45.8 ± 12.26 t C ha^{-1} (Axe 2015).

No other empirical studies considering storage of carbon in hedgerow biomass are reported in the UK. Crossland (2015) does measure hedgerow biomass as part of a scoping exercise into the carbon sequestration potential of coppiced hedgerows for woodfuel management and is a useful comparison to the findings by Axe and others (2017). Unmanaged hedgerows consisting of blackthorn, hawthorn or hazel (3.5–6m width) contained 45.08–131.5t C ha^{-1} and 1 year after coppicing (0.55–1.5 m width) contained 34.35–25.65 t C ha^{-1}. Woodland data based on figures from 118-year-old woodland at Rothamstead (UK) (Poulton and others 2003) was used by Robertson and others (2012) as a proxy for hedgerow carbon stocks. Hawthorn and hazel represented a gappy hedge and used to estimate that, depending on their height, hedges will store 11.25–45 t C ha^{-1}.

There is little evidence regarding the impact of plant species biodiversity has on hedgerow carbon stock. The presence of bramble was reported to contribute an additional 3.8 ± 1.46 t C ha^{-1} by Axe and others (2017), suggesting that plants other than the woody tree and shrub species may be important. Thiel and others (2015) report that hedgerows planted for biodiversity in British Columbia, Canada, had similar aboveground carbon stocks to less diverse remnant hedgerows, but a direct comparison is difficult due to the highly variable stocks reported, and differences in age of hedgerows. However, more diverse hedgerows were positively related to soil organic carbon stocks and is discussed in more detail below.

Hedgerow management means that a proportion of their biomass is removed on a regular basis, representing a carbon loss from the habitat and making it difficult to provide an annual sequestration rate. Robertson and others (2012) calculate that hedgerows could sequester 0.13–0.51 t C ha^{-1} y^{-1} (0.47–1.87 t CO_2e ha^{-1} y^{-1}) depending on their height. As this is based on woodland understory data, which are less dense than hedgerows, it may be an underestimate. Taylor and others (2010) based their sequestration rates for non-flailed hedgerows from a Welsh pastoral setting on short rotation poplar, suggesting higher rates than Robertson and others with a mid-range estimate of 6.37 t C ha^{-1} y^{-1} (23.36 t CO_2e ha^{-1} y^{-1}). Using data from long-term experiments and literature, Falloon and others (2004) estimate that land use conversion from arable to shrubby hedgerow has the potential to sequester 1.0 t C ha^{-1} y^{-1} (3.67 t CO_2e ha^{-1} y^{-1}) in vegetation and soils.

For hedgerows to represent a longer-term accumulation of carbon they should be managed to be taller and wider. Raising height from 2.0 m to 2.7 m (with widths ranging from 2.8–4.3 m) would represent an increase in size to 70 per cent of currently managed hedgerows across England and Wales, with a potential to sequester an additional 2.0 Mt carbon in hedge biomass in the farmed landscape (Axe and others 2017). Allowing trees to become established along a hedgerow's length may provide further opportunities to store carbon but needs to be balanced with the hedgerow's ability to remain a dense, vegetated barrier. Taking a modelling approach, Warner and others (2011) estimate the application of the agri-environment option EC23 'Establishment of hedgerow trees' in the Environmental Stewardship scheme would have sequestered an additional 1.6 t CO_2e ha^{-1} y^{-1} in increased biomass by promoting the growth of two trees per 100 m of hedge length.

Table 2.8 Carbon stored in hedgerow biomass, as reported in the literature

Hedge description	**Hedge Width (m)**	**Hedge Height (m)**	**Biomass Carbon Stock (t C ha^{-1})**	**Reference**
Hawthorn and blackthorn flailed hedges	1.9–3.5	2.6–4.2	32.2–42.0	Axe and others 2017
Minimally managed hedge	4.1 ± 0.21	3.9 ± 0.45	45.8 ± 12.3	Axe 2015
Unmanaged hedges (blackthorn, hawthorn or hazel)	3.5–6.0	Not reported	45.08–131.50	Crossland 2015
Coppiced hedges (blackthorn, hawthorn or hazel) – 1-year post coppice	0.55–1.50	Not reported	25.65–34.35	Crossland 2015
Planted hedgerows (British Columbia, Canada)	4	Not reported	76 ± 32 (including below ground biomass)	Thiel and others 2015
Remnant hedgerows – mean age of 38 years established (British Columbia, Canada)	7.5	Not reported	124 ± 21 (including below ground biomass)	Thiel and others 2015

Hedgerow Soils

Hedgerows can influence storage of SOC through the dominance of deeper rooting woody species and greater return of recalcitrant litter inputs than adjacent field systems. The lack of ground disturbance from tillage and other field operations directly below hedgerows can also increase the residence time of carbon in the soil (Follain and others 2007; Ford and others 2019).

The SOC associated with hedgerows is related to the land use of the field they boundary. At a commercial mixed arable and pasture farm in Northern England the SOC at a 2–7 cm depth was found to be greatest under hedgerows and pasture soils, with SOC in arable soils reported to be only 40 per cent of that under the hedgerow boundary (Holden and others 2019). Thiel and others (2015), in British Columbia, Canada report a difference of 40 per cent between hedgerows planted for biodiversity value and neighbouring production land planted with annual crops. No significant difference was reported by Holden and others (2019) for SOC between hedgerow soils and those in permanent pasture. On grazed permanent pasture in North Wales, no significant difference in SOC was observed between pasture bounded by hedgerows or stonewalls with both measured around 7 kg C m^{-2} (70 t C ha^{-1}) to a depth of 15 cm (Ford and others 2019). The difference between arable and pasture soils is likely due to the SOC under hedgerows being representative of original conditions, with carbon loss being at greater rates under agricultural practices that disturb the soil (Holden and others 2019) and/or the increased and more diverse organic matter inputs from hedgerows when compared to arable soils (Thiel and others 2015).

Hedgerows may influence SOC beyond their footprint. Ford and others (2019) also report that SOC stock was greatest closer to hedgerows, with values 15 per cent greater within 2 m of the hedgerow boundary and decreasing markedly between 2.2 and 3.4 m from the hedgerow base. A similar relationship was observed by Follain and others (2007) in north-west France who measured a median SOC stock of 16.6 kg C m^{-2} (166 t C ha^{-1}) in the vicinity of hedges (to a depth of 55 cm), compared to SOC stock of 13.3 kg C m^{-2} (133 t C ha-^{1}) in the wider landscape. This suggests that hedgerows can have a narrow influence on the SOC in an adjacent pasture and such variability should be considered when assessing carbon stocks of farmed landscapes.

Hedgerows could hold a significant proportion of their biomass carbon below ground. Axe and others (2017) report this as almost half (35.8 t C ha^{-1} for roots, 2.4 t C ha^{-1} sub-surface woody debris) and is suggested to be an underestimate due to root laterals growing beyond the study area. The SOC values from the same site, managed as a mixed farming system in Gloucestershire, England, are also reported to hold significant stocks to a depth of 30 cm with 98.7 t C ha^{-1} reported (Axe 2015 cited in Axe 2020). Crossland (2015) reports SOC stocks in a similar range. Values from 74–112 t C ha^{-1} for an unmanaged stretch and between 67–95 t C ha^{-1} under cutting management, suggesting soil carbon storage can exhibit a wide range within the same hedgerow. The diversity of the hedgerow species may play a role in this variation. Thiel and others (2015) demonstrate that hedgerows planted for biodiversity store significantly more SOC than remnant hedgerows with values to a 1 m depth reported at 175.9 ± 13.2 t C ha^{-1} and 132.7 ± 7.3 t C ha^{-1} respectively. The authors suggest differences may be due to differences in root structure and exudates, litter, or greater interception by the denser planted hedgerows of eroded soil from adjacent production fields. Carbon stocks in soils below hedgerows are summarised in table 2.9.

Table 2.9 Carbon stored in hedgerow soils, as reported in the literature

Hedge description	Hedge Width (m)	Hedge Height (m)	Carbon Stock (t C ha^{-1})	Reference
Crataegus monogyna, *Prunus spinosa* and *Corylus avellana*	2–3	Not reported	68.2 ± 1.1	Ford and others 2019
Castanea sativa, *Quercus robur* hedges planted on earth banks	Not reported	Not reported	166	Follain and others 2007
Hawthorn and hawthorn/blackthorn mix planted on pelocalcaric gley soil or lithomorphic brown rendzina	2.6–4.2	1.9–3.5	98.7	Axe 2015 reported in Axe 2020

Few studies have considered soil carbon fluxes rates below hedgerows. Crossland (2015) is one of the few in the UK to suggest modelled sequestration rates based on hedgerow data and consider soils with unmanaged hedgerows a net sink between 2.74–12.19 t C ha^{-1} y^{-1} (10.05–44.70 t CO_2e ha^{-1} y^{-1}), though the underpinning data is based on a small number of field observations and is very high, comparable to that of forestry plantations. Robertson and others (2012), again basing their estimates on Rothamstead woodland understory data, suggest hedgerows have the potential for long-term accumulation of SOC, taking at least 766 years to reach equilibrium. The age of the hedgerow will determine the sequestration rate, with new and old hedgerows estimated to sequester 0.54 and 0.46 t C ha^{-1} y^{1} (1.99 and 1.67 t CO_2e ha^{-1} y^{-1}) respectively, rates much smaller than those suggested by Crossland (2015). Soil type, and its interaction with seasonal weather events may also influence the source-sink status of soils under hedgerows. Ford and others (2020) modelled drought and drought exclusion scenarios for pasture adjacent hedgerows on seasonally wet and free draining soils. Under drought exclusion hedgerows on both soil types were net carbon sinks (storing 6–10 t CO_2e ha^{-1} y^{-1}), but the hedgerows on seasonally wet soils switched to a net source under a drought event (5.8 ± 0.8 t CO_2e ha^{-1} y^{-1}), with the change driven by a sudden spike in soil CO_2 emissions. Further investigations are needed to identify appropriate locations to plant hedgerows to maximise their carbon benefits.

Hedgerows also play an important role in preventing erosional loss of carbon from the landscape, particularly on hill slopes (photo 2.4). The anti-erosion effect of contour hedgerows is well known, with eroded topsoil accumulating upslope of hedgerows and the land downslope a source of erosional loss (Follain and others 2007; Ford and others 2019). Walter and others (2003) observed a gradual thickening of the soil A horizon from the top of hills to hedges downslope, with a greater stock of SOC associated with this. This raises the need to consider the influence hedges exert over the wider landscape, not just in their immediate vicinity, with the authors reporting an order of magnitude difference in SOC stock depending on hedge density. Landscapes with a high-density hedge network (200 m ha^{-1}) the mean total SOC reached 117 t C ha^{-1}, with a hedge effect of 38 per cent. Under low density hedge networks (50 m ha^{-1}) the mean total SOC stock was 84 t C ha^{-1}, with the contribution of hedges at 13 per cent.

Photo 2.4 Mature hedgerow boundaries indicating traditional field structures in North Devon © Natural England / Peter Roworth

2.3.2 Climate Change impacts and other interactions

Hedgerows are at low risk from the impacts of climate change (Natural England & RSPB 2020). However, drought, flooding and longer growing season may exert stress to hedgerow species, increasing their susceptibility to pests and disease. Resilience to future climate change should be considered when planting, restocking or filling gaps, using diverse hedgerow species that are adapted to a range of climatic conditions.

The main risk to hedgerows is mismanagement, with neglect causing hedgerows to develop into a line of trees and over-cutting causing them to lose structure and become gappy (People's Trust for Endangered Species, accessed 2020). Hedgerows also suffer from damage by stock and wild animals from browsing, trampling, dunging, burrowing. While hedgerows are protected from removal under Hedgerow Regulations in England and Wales, reducing their rate of loss in recent times, under half (48 per cent) of managed hedgerows in Britain were classified as being in good structural condition in 2007 (Carey and others 2008; Firbank and others 2011) and hedgerows continue to decline in length and structural condition (Staley and others 2015).

As linear landscape features hedgerows are at risk from edge effects and the intensification in management of adjacent agricultural land. This may lead to nutrient enrichment or damage to hedge structure, particularly if buffer strips and margins are lost or reduced in size (Natural England 2020). Management operations in-field may damage the tree and shrub roots that extend beyond the hedge footprint leading to decline in hedge condition or premature death.

2.3.3 Evidence gaps and future needs

Data available regarding carbon storage and sequestration in hedgerows does not represent the diversity of hedgerows found in the English landscape. The only UK based empirical data cited in this review comes from a single study site (Axe and others 2017, Axe 2015). To aid a greater understanding of the contribution hedgerows make to carbon stocks further research is required to investigate the influence that vegetation management, different tree and shrub species, soil type and depth have on their ability to accumulate and store carbon both above and belowground.

Data modelling in the past has used woodland species as a proxy for hedgerows. Due to hedgerows being more densely planted this may lead to an underestimation of their carbon benefits. Research cited here from Canada suggests greater carbon stocks under more diverse hedgerows. Further investigation in the UK is needed to effectively inform the need to boost both biodiversity and climate change mitigation potential of hedgerows, including quantifying the carbon benefits of allowing hedgerow trees to become established.

However, as they consist of managed trees and shrubs it is a fair assumption that hedgerows need to be in place long-term for them to deliver benefits for climate change mitigation (Wolten and others 2014). Understanding the duration it takes for hedgerows to reach a carbon equilibrium is required to inform their management, as well as how susceptible the soil carbon is to loss under restocking. As a managed habitat, cut regularly and rejuvenated every 40 years or so, further information is required regarding how carbon storage in soils and biomass is offset by greenhouse gas emissions from trimming, flailing, disposal, laying, coppicing and cultivation methods, as well as the role hedgerows can play in producing biomass for wood fuel, replacing fossil fuel emissions.

2.4 Traditional and intensively managed orchards

Orchards fall into two broad categories, traditional and intensive. Traditional orchards typically consist of low intensity management of widely spaced fruit or nut trees, on permanent grassland that may be grazed by livestock or cut for hay. They are found throughout the lowlands of England as well as in localised hotspots in sheltered upland valley settings, such as the Lyth Valley in Cumbria. Intensive orchards are more densely planted, often using a short-lived dwarf or bush type (espalier) rather than standard trees. Their floors maybe mown or kept bare and managed with herbicides (Robertson and others 2012). For both categories the production of crops is maintained via practices such as grafting and pruning, and understory management is integral to keep the trees open.

Orchards differ from other wooded habitats in that they are defined by their management and structure, rather than vegetation type and not necessarily aligned to a particular topography or soil type. Orchards may also be associated with scrub, either as hedgerows along their boundary or amongst the trees in unmanaged sites.

2.4.1 Carbon storage and sequestration in orchards

Orchard vegetation

Evidence is scarce in the literature regarding the carbon storage and sequestration potential of orchards but estimates of their contribution is important as they are often concentrated within landscapes and cover significant areas (Demestihas and others 2017). The planting structure of orchards is a key determinant in the amount of carbon stored in orchard biomass. Higher carbon stocks were reported in traditional orchards where trees are planted at lower density but grow larger and accumulate a high amount of woody biomass, while in intensive orchards trees are grown at higher densities but are managed to promote fruit production rather than woody biomass (Robertson and others 2012; Anthony 2013). Robertson and others (2012) estimate that traditional orchards accumulate 0.116–0.710 t C ha^{-1} y^{-1} (0.425–2.603 t CO_2e ha^{-1} y^{-1}) in their biomass while intensively managed orchards ranged 0.877–1.260 t C ha^{-1} y^{-1} (3.215–4.42 t CO_2e ha^{-1} y^{-1}) (table 2.10).

While orchards planted in the traditional style hold greater carbon stocks in their biomass, intensively managed orchards may sequester carbon at greater rates due to management approaches such as pruning promoting photosynthesis, their younger age and their dense planting structure (Robertson and others 2012; Demestihas and others 2017). A significant amount of carbon is removed from the system each year, particularly via the fruit harvest which may be between 40–70 per cent (Robertson and others 2012). The potential of orchards to store carbon long-term is further limited by the removal and disposal of biomass via pruning, and, in the case of intensive orchards, the relatively short lifespan of the trees which may only last 15–30 years, before they are replaced to support crop production (Anthony 2013; Demestihas and others 2017).

Considering a carbon balance approach Robertson and others (2012) report that intensive orchards are net sequesters of carbon over their productive lifetime (1.15–2.12 t C ha^{-1} y^{-1};

4.21–7.77 t CO_2e h^{-1} y^{-1}), driven by the density of planting. Traditional sites were more variable, and in some cases are net emitters (-1.59–+0.45 - t C ha^{-1} y^{-1}; -5.83–+1.65 t CO_2e h^{-1} y^{-1}) due to reduced net primary productivity with age. However, the study did not consider the role traditional orchards play in protecting existing, potentially large carbon stocks, or the method used to dispose of trees when they are restocked. Trees are often burnt, returning much of the stored carbon to the atmosphere. Finding alternative uses for the biomass, such as wood fuel, could help to mitigate some of this loss of carbon.

In order to stagger the role of orchard in storing and sequestering carbon, Robertson and others (2012) suggests taking a landscape scale approach, balancing the increased rates of uptake of new orchards with the long-term storage importance of old orchards.

Table 2.10 Carbon stored and sequestered in orchard biomass, as reported in the literature

Orchard management	C Storage in above and belowground biomass (t C ha^{-1})	Sequestration (t CO_2e ha^{-1} y^{-1})	Reference
Traditional cider orchard	33.31–230.0	-	Anthony 2013
Traditional	8.6–33.2	-0.425 to -2.603	Robertson and others 2012
Intensive/commercial apple orchard	> 15 years old: 12.21 < 15 years old: 9.57	-	Anthony 2013
Intensive	13.9–18.4	-3.215 to -4.42	Robertson and others 2012

Orchard soils

Traditional sites that have remained in consistent management over decades to centuries will likely have suffered relatively little ground disturbance and will hold greater stocks of soil carbon than intensively managed orchards, which are characterised by removal and replanting of trees every few years (Robertson and others 2012) (table 2.11). Older sites tend to be managed in the traditional manner, with stand-alone trees and the presence of ground vegetation. Anthony (2013) reports significant increases in soil carbon between age categories across 26 commercial orchards ranging from 4–100 years old. The maintenance of bare ground and use of herbicides in intensive sites reduce the organic matter return to the soil, limiting further what can be stored long-term in the soils.

Though older sites tend to be of importance for their soil carbon stocks, younger, more intensive sites may accumulate carbon at greater rates (Robertson and others 2012). Such rates are dependent on the dynamic relationship between carbon pools as they move towards an equilibrium; disturbed sites will be accumulating carbon as they recover, whilst older sites may already have reached equilibrium after 50–100 years. Intensively managed orchards in the case study reported by Robertson and others (2012) tended to demonstrate this, averaging soil carbon accumulation at 0.34 t C ha^{-1} y^{-1} (1.25 t CO_2e ha^{-1} y^{-1}), compared to 0.17 t C ha^{-1} y^{-1} (0.62 t CO_2e ha^{-1} y^{-1}) at traditional sites. However, one traditional orchard had much greater soil carbon accumulation rates than all the other sites at 0.52 t C ha^{-1} y^{-1} (1.91 t CO_2e ha^{-1} y^{-1}), considered due to high leaf inputs and past arable management. The limited data from the case study is not enough to conduct statistical analysis, but it does demonstrate the influence past and current land management may play in an orchard setting, and the variable outcomes regarding carbon.

Table 2.11 Carbon storage and sequestration in orchard soils, as reported in the literature

Orchard management	Soil depth (cm)	Carbon Storage (t C ha^{-1})	Sequestration (t CO_2e ha^{-1} y^{-1})	Reference
Traditional	30	47–111	0.11–1.91	Robertson and others 2012
Intensive	30	31–47	1.17–1.32	Robertson and others 2012
Commercial orchards (including traditional sites)	5	< 15 years old: 7.02 > 15 years old: 12.63 > 35 years: 16.21	Not reported	Anthony 2013

2.4.2 Climate change impacts and other interactions

Traditional orchards are considered to have low sensitivity to climate change (Natural England & RSPB 2020). While fruit trees maybe sensitive to drought, rising temperatures, and increased storm frequency, the heavily managed nature of this habitat means such risks can be mitigated by irrigation, new management approaches and appropriate replacement and replanting of the tree stock.

Changes to agricultural policy maybe more of a risk to traditional orchards and the carbon stocks they hold due to a move away from small-scale fruit farming in the UK in the last 70 years (Woodland Trust 2014). Current agri-environment approaches favours support for traditionally managed orchards on historic or existing sites, or where new planting may link or buffer existing sites. Looking beyond these conditions would support the expansion of orchards range, whilst maintaining a productive land use.

2.4.3 Evidence gaps and future needs

There continues to be limited studies into the storage and sequestration of carbon in orchards, but the work by Robertson and others (2012) and Anthony (2013) demonstrates their significant potential. More information is required on the management of orchards, such as pruning and livestock management, and the impact this has on carbon stored in the biomass and soils. Intensive management approaches, such as the application of herbicides and pesticides, has been shown to impact the soil biodiversity of orchards (Montanaro and others 2017). Further work is necessary to understand how this may affect soil functional processes and the cycling of carbon.

Recognising that orchard trees have a finite life, and grubbing out can negate the carbon gains made over the orchard lifetime, traditional management practices where individual trees are replaced as required should be encouraged and uses should be found for the woody material, such as a substitute for fossil fuels. However, this substitution should not be considered carbon neutral (Booth 2018)

Finally, orchards are typically concentrated in the landscape and may be at different stages of carbon sequestration. Management plans should work at a landscape scale, rather than site scale, to maximise benefits for climate change mitigation through staggering orchard age and development. Fruit production could also be incorporated into agroforestry systems. There is significant potential for the creation of new orchards, particularly taking a traditional approach, to deliver biodiversity and carbon benefits on farms. However, a negative impact could occur if they replaced mature trees or scrub.

2.5 Scrub habitats

The ecological value of scrub hasn't been recognised or appreciated until relatively recently. In 2000 JNCC published the report 'The nature conservation value of scrub in Britain' (Mortimer and others 2000) and that work stimulated further research into this habitat. That report defined scrub as 'all stages from scattered bushes to closed canopy vegetation, dominated by locally native or non-native shrubs and tree saplings, usually less than 5 m tall, occasionally with a few scattered trees. This definition excludes dwarf shrub heaths, planted stands of young trees and coppice stump regrowth less than 5 m high. The Countryside Survey 1990 gave an area of scrub of 900 ± 200 km^2 in Britain, of which 600 ± 100 km^2 is in England (Mortimer and others 2000). Importantly, scrub can be an intermediate and important stage towards 'natural forest regrowth', defined as 'the recovery of forest cover on cleared lands through spontaneous regrowth after cessation of previous disturbance or land use' (Cook-Patten and others 2020).

Scrub is an important habitat for a variety of species, including some Red Data Book and BAP species (eg dormice *Muscardinus avellanarius*, breeding warblers, Duke of Burgundy butterfly *Hamearis lucina* among many others). There are agri-environment options to create or manage it, eg WD8: Creation of successional areas and scrub or WD7 Management of successional areas and scrub in the current Countryside Stewardship. Its management is influenced by both, the conservation value of the scrub and that of the habitats in which it occurs, as well as the requirement to balance the two. Some scrub types can expand quite rapidly, whereas others are slower and require minimal intervention and management (Day and others 2003).

Shrub species can encroach into open habitats due to climate change or due to land use changes (Guidi and others 2014; Urbina and others 2020), such as abandonment of management or due to changes in the fire regime. There have been studies on the direction of secondary succession after abandonment or neglect of priority habitats but not many which have focused on the carbon implications or either allowing scrub to colonise other habitats or the management to contain it. There seems to be a specific evidence gap in the UK, with most studies on the carbon implications of scrub development in Europe from Mediterranean or Nordic regions.

2.5.1 Carbon storage and sequestration in scrub

Scrub vegetation

Increases in woody vegetation leads to increased carbon accumulating in biomass, soil or both (Ferlan and others 2016 and cited within Guidi and others 2014). More woody material results in increased carbon density in the aboveground biomass (Urbina and others 2020), about 2 t C ha^{-1} y^{-1} (7.33 t C ha^{-1} y^{-1}) for birch woodland development (Uri and others 2012); or 1.69 t C ha^{-1} y^{-1} (6.20 t C ha^{-1} y^{-1}) for mixed ash *Fraxinus excelsior* and sycamore *Acer pseudoplatanus* (Alberti and others 2008). However, the contributions of the understory vegetation are modest and depend on the shading effect (Uri and others 2012). In the first stages of scrub development most of the carbon is in the soil but, at least for birch scrub-woodland development, it changes from about 20–30 years old (Uri and others 2012) when more carbon accumulates in the aboveground parts of trees.

Soils under scrub

Secondary succession and scrub development result in environmental changes, including in carbon fluxes. Li and others (2016) found in a meta-analysis of global sites that the SOC content due to shrub development could vary between -50 per cent to +300 per cent. The increases in SOC were higher in semi-arid and humid regions and when the encroaching shrubs were legumes. There were also differences by soil type, with sandy soils showing higher increases than clay and silty soils. Soil organic carbon was also negatively correlated with soil total nitrogen and pH (Hunziker and others 2017). Cook-Patton and others (2020) also found that carbon sequestration in aboveground vegetation was higher in warmer and wetter biomes, though there were not significant differences in soil accumulation rates. Similar results by Alberti and others (2011) showed that dry sites undergoing secondary succession gained soil carbon, whereas wet sites lose carbon, probably related to nitrogen leaching.

Studies of the changes in SOC at depths down to greater than 30 cm after woody encroachment of abandoned grasslands in the Pyrenees and the Alps found that SOC was higher in the grasslands and lowest in the developing shrublands (Nadal-Romero and others 2018). Total SOC increased again in young and old forests (Guidi and others 2014; Hunziker and others 2017; Nadal-Romero and others 2018). These authors report similar results in other sites in Europe for grasslands but acknowledge that there is high uncertainty and opposite results in the literature (as also reported by Li and others 2016). This may be due to the different type of soils studied (eg mineral vs organic), the differences in root systems and even the impacts of the changing soil fauna with succession (Guidi and others 2014).

Increased scrub cover may lead to slower geochemical cycles or no increase in SOC in older stands (Uri and others 2012; Urbina and others 2020) or even emissions from soil stocks from habitats such as grasslands and heathlands that had a significant soil carbon stock before being scrubbed-up. Guidi and others (2014) found that, in the medium-term (< 120 y), scrub invasion of grasslands which have significant soil carbon stores can result in decreases in SOC. Only after a significant period of tree cover carbon stores are replaced again, which has implications for meeting net zero targets in the next 30 years. A summary of carbon stocks at different successional stages is collated in table 2.12.

Table 2.12 Carbon stocks in scrub under different successional stages (t C ha^{-1}) as reported in the literature. Carbon stocks are disaggregated into soil and vegetation where possible

Location	Grassland	Scrub	Forest	Reference
Pyrenees, Spain	57.9	70 y old *Genista scorpius, Juniperus communis, Rosa gr. Canina* & *Buxus sempervirens* 48.4	67.4	Nadal-Romero and others 2018 (Ecosystem carbon reported)
Pyrenees, Spain	Nardion 147.9	*J. communis* & *Silybum marianum* 91.7	50-70 y old Coniferous 121.3-101.3	Nadal-Romero and others 2018 (Ecosystem carbon reported)
Alps, Switzerland	~100 y old Poaceae SOIL= 100	15–40 y old *Adenostylion* assoc. *Rhododendron-Vaccinion* assoc. *Poion alpinae* assoc. SOIL= 81	90 y old *Alnus viridis* SOIL= 174	Hunziker and others 2017
S Alps, Italy	100 y old Poaceae SOIL= 86 VEG= ~1* TOTAL= ~87*	10–35 y old *Corylis avellana, Juniperus communis* *Betula pendula* SOIL= 72-51 VEG= ~3-60* TOTAL= ~75–110*	>150 y old *Fagus sylvatica* *Pinus sylvestris* *Picea abies* SOIL= ~70* VEG= ~90* TOTAL= ~160*	Guidi and others 2014
Germany & Italy (Alps)	Meadows VEG (stems)= 0 SOIL= 75	*Picea abies* 30-60 yrs VEG ~200 SOIL= 55.5 (74% of meadow)	*Picea abies* >60 yrs VEG= 350 SOIL= 62 (83% of meadow)	Thuille & Schulze 2006

In a review of 2,700 SOC profiles of sites invaded by scrub or woodland, Jackson and others (2002) found that grasslands SOC was indistinguishable from woodland with 200 mm y^{-1} precipitation but woodland had 43 per cent less SOC than grasslands at 1000 mm y^{-1}. Also, the same authors, looking at paired sites grassland vs shrub/wood found from gains of 13.1 t C ha^{-1} (dry) to losses of 56.1 t C ha^{-1} (wet). Land abandonment also occurs in croplands and revegetation can happen by secondary succession (taking around 100 years to achieve the forest stage) or quicker through artificial afforestation (Nadal-Romero and others 2016). However, these authors found that the changes in carbon and other soil properties were not significantly different between natural succession and afforestation, ie soil recovery is slow, even under a plantation regime.

2.5.2 Climate change impacts and other interactions

Scrub encroachment and woodland expansion are often seen as positive in terms of GHG balance, but this is not universally true and depends on the characteristics of the original ecosystem, its soils, the encroaching species and climatic conditions (Li and others 2016). In some cases, tree growth can lead to increased carbon stocks only after many decades; Thuille and Schulze (2006) calculated that it took about 80 years to regain former stock levels in grassland soils planted or succeeded to spruce. In other cases, succession can lead to carbon losses (Jackson and others 2002). Increased scrub and tree cover also may have negative impacts on water resources due to evapotranspiration, on biodiversity, with the loss of species of open niches, and result in cultural losses (Ferlan and others 2016).

Some studies have shown an important impact of periods of drought or high humidity on the carbon fluxes. Ferlan and others (2016) observed carbon sequestration on a karst grassland succeeding to shrubland and trees on a short period of time (four-year study). However, there was a high seasonal variation, with low carbon sequestration during period of drought, which increased with rainy episodes. The periods of rain and drought determined which type of vegetation replaced the grassland and therefore had an impact on the soil chemical characteristics. Jackson and others (2002) also found that drier grassland sites invaded by woody plants gained, but wetter sites lost SOC. The losses in the wetter sites was enough to offset the gain in biomass carbon.

Guidi and others (2014) conclude that in the long-term and for climate change mitigation, forests can accumulate more carbon (80 t ha^{-1}) in the whole system than grasslands or transitional scrub stages, but the accumulated carbon in the woody biomass is less stable and more susceptible to external factors, such as management, harvest and other environmental modifications, like fires.

2.5.3 Evidence gaps and future needs

There is not a large body of evidence about the contribution of scrub habitats to total ecosystem carbon stocks. However, the existing evidence consistently points towards; a) an increase in the above ground biomass carbon as a result of the development of woody species on previous grassland or cultivated land; b) a negative impact on soil carbon stores in the short to medium term, dependent on soil type; c) a recovery of soil carbon stores in the long-term, 80–100 years. This will have implications on decisions on the future management of land brought out of production or left to regenerate naturally.

There are very few studies on the changes on vegetation and soil carbon stocks in the UK. Most research seems to be from countries which have suffered from depopulation of rural areas and abandonment of farming practices in the most difficult and marginal areas, such as the mountainous regions of Spain and Italy. Given the more Atlantic climatic conditions and the fact that both, soil types and climate, have a significant impact on carbon implications, UK based research is required. Caution is required in using the figures cited above in a UK context.

3 Open habitats and farmland

3.1 Chapter summary and key messages

Open habitats and farmland dominate land cover in England - agricultural land use alone represents around 70 per cent of England's land area. Habitats such as heathlands and semi-natural grasslands require traditional low intensity management, such as grazing or cutting, in order to halt succession to scrub or wooded habitats. Farmland land is highly modified from its original habitat cover in order to promote production of biomass or livestock. These habitats and land uses are grouped due to the need for human management interventions to maintain them, and that agricultural land often forms the baseline of change when considering restoration or creation of semi-natural habitats. Though these habitats and farmland typically sequester and store carbon at a lower density than peatlands, saltmarsh and woodlands, their land area means they play a significant role in England's carbon stocks.

The key messages:

- Heathlands store significant carbon stocks, mostly in their soils, comparable to that of peatlands in the top 15 cm. There is medium confidence in the data due to the relative consistency in reported carbon stocks.
- Heathlands require management to prevent succession to woodland or scrub in order to maintain their heathland biodiversity interest, with the resulting vegetation loss representing a net loss from the system. Adopting a gradual felling cycle, as opposed to clear felling can support carbon stocks to be maintained (Broadmeadow & Matthews 2003).
- Established semi-natural grasslands represent an important carbon stock due to their undisturbed soils. Most studies to date focus on the carbon stock of topsoils, but approximately 60 per cent of a grassland's total carbon stock may be below this depth (Ward and others 2016). The 2007 Countryside Survey remains the most comprehensive overview of soil organic carbon stocks in the UK.
- The capacity of grasslands to uptake further carbon becomes negligible after around a century post establishment. Reversion to extensive, low input semi-natural grassland from a lower carbon land use, such as arable farmland, represents an opportunity to create a carbon sink alongside biodiversity benefits. There is limited evidence regarding carbon flux data for semi-natural grasslands in England or the UK, most notably for calcareous grasslands.
- Grazing or cutting management is often used as an intervention in the management of open habitats such as heathlands and grasslands to halt succession to woodland or scrub. Gains in soil and vegetation carbon from habitat restoration and creation need to be balanced with emissions that result from the grazing of ruminant animals and machinery use and are outside the scope of this review.

- Arable and improved grassland systems are extremely difficult to characterise with regards to 'typical' carbon storage and sequestration, as these systems can be located on a wide range of soil types, each having their own properties with regards to carbon dynamics. As such we assign these a low confidence.[14]
- They are considerably modified from their natural state through drainage, liming, cultivation, fertilisation and other 'improvements' in order to produce a crop. Such 'improvements' can lead to soil degradation and carbon loss, as evidenced by the relatively low carbon stocks held in agricultural topsoils in comparison with semi-natural habitats (Emmett and others 2010).
- Peatlands drained and modified for agricultural use represent the largest emitters of greenhouse gases in the land use sector. Farming with raised water levels, or paludiculture, could mitigate some of these emissions (though not to the extent of habitat restoration – see peatland chapter) but there are currently significant practical, economic and societal challenges for its widespread adoption (Mulholland and others 2020).
- Overall, due to the variable and lack of field measurements for the habitats within this chapter mean we assign low confidence to the carbon flux values cited as representative.
- Tables 3.1 and 3.2 summarise the carbon storage and flux values identified as representative for habitats reported in this chapter.

[14] Grassland management varies from intensively managed agriculturally improved sites, through semi-improved grasslands, that have been subject to a diverse past regime of ploughing, seeding and inputs, to semi-natural grasslands (as covered in section 3.3). To facilitate comparison and reflect the evidence base this section focuses on the intensively managed agricultural grasslands, however further evidence is needed to characterise carbon stocks and cycling in semi-improved grasslands which makes up a significant amount of England's grassland area.

Table 3.1 Summary of carbon storage values as reported in the literature for open habitats and agricultural land uses. NB range of depths are not standardised in the literature.

Habitat Description	**Soil Carbon ($t C ha^{-1}$)**	**Soil Depth (cm)**	**Vegetation Carbon ($t C ha^{-1}$)**	**Soil + Veg. Carbon ($t C ha^{-1}$)**	**Confidence** [High, Medium, Low]	**References**
Heathlands						
Upland & lowland Heathland	94[a] [88[b] to 103[c]]	15[a,b] to 30[c] cm	6* [2[b] to 9[a]]	100 [90 to 112]	Medium	[a]Van Paassen and others. (2020) [b] Ostle and others (2009), based on CS2007 [c] Cantarello, Newton and Hill (2011)
Semi-natural grasslands						
Acid grassland (without vegetation)	87	15 cm	No Data	–	Medium	Emmett and others (2010)
Calcareous grassland	69	15 cm	No Data	–	Low	Emmett and others (2010)
Neutral grassland	60[a] [33.31[b] to 68.74[c]]	15 cm	No Data	–	Medium	[a] Emmett and others (2010) [b] Fornara and others (2013) [c] Eze and others (2018a); Eze and others (2018c)

Table 3.1 Continued

Habitat Description	Soil Carbon (t C ha^{-1})	Soil Depth (cm)	Vegetation Carbon (t C ha^{-1})	Soil + Veg. Carbon (t C ha^{-1})	Confidence [High, Medium, Low]	References
Farmland						
Arable / cultivated land	120[a] Range for 30cm [27.5 to 88.2[b]]	100 cm 30 cm	No vegetation stocks are given – as management (grazing and cutting) removes biomass annually	–	Low	[a] Moxley and others (2014) [b] Cantarello and others 2011
Improved grassland	130[c] Range for 30cm [72 to 204[d]]	100 cm 30 cm		–	Low	[c] Moxley and others (2014) [d] Cantarello and others (2011)
Intensive grassland on deep peat soils	1980	200 cm		–	Low	Evans and others (2016)
Arable on deep peat soils	Range [1290 to 3880]	75 to 200 cm		–	Low	Evans and others (2016)

Table 3.2 Summary of carbon flux values as reported in the literature for open habitats and agricultural land uses. Note for agricultural land use on peat fluxes have been adjusted to represent carbon only fluxes, and the influence of nitrous oxide removed, to be consistent with other habitats in this report. The full peatland GHG table is reported in section 4.2.2.

Habitat Description	C flux ($t\ CO_2e\ ha^{-1}\ y^{-1}$)	Range (if possible)	Confidence [High, Medium, Low]	References
Heathlands				
Lowland heathland & Upland heathlands	+0.054	–	Low	Warner and others (2020)
Semi-natural grasslands				
Arable reversion to low input grassland	-1.590	–	Low	Warner and others (2020)
Undisturbed semi-natural grassland under long-term management	Negligible, equilibrium reached.	–	Low	Sozanska-Stanton and others (2016)
Farmland				
Arable land use	+0.29	-	Low	Muhammed and others (2018)
Improved grasslands	-0.36	-1.28 to +0.92	Low	Soussana and others (2010)
Intensive grassland on deep peat soils	+24.87	-	Medium	2021 update to the Emissions Inventory for UK Peatlands – to be published in April 2021 in the 2021 UK GHG Inventory[15,16]
Arable on deep peat soils	+32.89	-	Medium	2021 update to the Emissions Inventory for UK Peatlands – to be published in April 2021 in the 2021 UK GHG Inventory

[15] https://unfccc.int/ghg-inventories-annex-i-parties/2021
[16] NAEI website: https://naei.beis.gov.uk/reports/

3.2 Heathlands

The broad habitat dwarf shrub heaths include vegetation types distributed across the uplands and lowlands of England. It is characterised by species of the Ericaceae family, typically heathers *Calluna vulgaris* and *Erica* species (Gimingham 1972; Webb 1986), with over 25 per cent in cover (JNCC 2009a); or more when in association with other characteristic species in the uplands (JNCC 2009b). This habitat is found mostly in the Atlantic region of Europe and shows great geographical variation, both in latitude and altitude and degrees of wetness. It appears mainly on acidic and oligotrophic soils and in transitions with other habitats, such as acid grasslands, woodlands and peatlands. Locally there can be variations, known as "lichen heath" or "grass heath", but the research on the carbon impacts usually doesn't go into that level of detail for each type, so they are not considered here.

Most heathlands are the result of centuries, if not millennia, of use and exploitation. This was mainly for grazing domestic livestock, sometimes after burning, and cutting vegetation for building or fire-making materials (Webb 1986); though there would have been open areas with heathland vegetation in the original wildwood. Sands, gravels and rocks have also been extracted over time. Only in very exposed or cold areas, such as coastal cliffs or the tops of mountains, can they be the climax vegetation. Some heaths may have been cultivated for short periods, and then abandoned, due to poor fertility.

Heathlands in good condition show a heterogeneous structure at different scales, with a combination of the following elements: dwarf shrub vegetation in various growth stages, grassy and herbaceous patches, some bare ground, bryophytes and lichens and a limited cover of trees, bracken and other species (JNCC 2009a, b). Favourable condition requires active management, especially in lower lying altitudes.

Heathlands have declined in extent and condition over the last century (Van der Wal and others 2011). Only about 20 per cent of lowland heathland is left in England from the 19th century extent, and there have been more than 27 per cent losses of heather moorland in England and Wales since 1945 (JNCC 2008). Current estimates[17] are about 1.2–1.3 million hectares (ha) in the UK, of which over 220,000 ha is in England (CS2007; JNCC 2019). Most of the remaining patches in the lowlands are small and managed for conservation, whereas in the uplands, although they are more extensive, they are still intensively managed, in most cases for game sports or grazing.

This review includes evidence for both upland and lowland heathlands, which ecologically can be considered a continuum, but are managed differently. Few references separate dry and wet heaths. In some cases, papers may refer to heathland on deep peat, which the authors consider degraded peatland, which should be restored if possible (Dixon and others 2015), and it is dealt with in Chapter 4 Peatlands. A detailed classification of the different heathland vegetation types can be found in Rodwell (1991).

[17] Habitats Directive, Article 17 report 2019, types H4010, H4020, H4030, H4040, H4060.

3.2.1 Carbon storage and sequestration in heathlands

Heathland soils

Ostle and others (2009) estimated that dwarf shrub heathlands in the UK store 130 Mt C belowground (7 per cent of the national amount up to 50 cm depth). Field and others (2020) provided an even larger figure: 240 Mt C within "high conservation value (HCV)" dwarf shrub heath in the UK. However, they suspect that some of these areas currently classed as heath are degraded blanket bog on deep peat.

Most of the carbon on heathland ecosystems is in the soil (table 3.3), on average assumed to be 98 per cent of the carbon stocks in soils and 2 per cent in the vegetation biomass (ONS, 2020) There are differences between the stocks of thin mineral soils and those of deeper organic ones, some of which could be degraded peatland with some ericaceous cover or transitions to them (Cantarello and others 2011; Alonso and others 2012; Sozanska-Stanton and others 2016; Bartlett and others 2020). The Countryside Survey 2007 reports that dwarf shrub heaths contain approximately 88 t C ha^{-1} in the soil and 2 t C ha^{-1} in the vegetation, in England. Cantarello, Newton and Hill (2011) report higher values from a study in the South West England, 103 t C ha^{-1} and 7.11 t C ha^{-1} respectively. Brown (2020) points out that organo-mineral soils (eg gleysols and podsols) with a surface organic horizon are important carbon stores that are not usually included in the core peatland resource.

Table 3.3 Summary table of carbon stocks (t C ha^{-1}) reported for dwarf shrub heaths

	Soil (t C ha^{-1})	Vegetation (t C ha^{-1})	References
Dwarf Shrub Heath	88	2	Ostle and others (2009); CS2007
	103 [50.7 - 196]	7.11 [2 - 17.5]	Cantarello, Newton & Hill (2011)
	94	9	Van Paassen and others (2020)
	-	49	Morison and others (2012)
	98		Field and others (2020)

In any case, soil conservation seems to be one of the best mitigation tools against increasing GHG emissions. However, halting natural succession to maintain heathlands typically involves some disturbance, like removing vegetation and creating bare ground (Photo 3.1). In most sites soils haven't been previously cultivated (ie deeply ploughed), so they are also a repository for historic environment interest and are examples of undisturbed soil types. Any management which involves soil disturbance should be carried out with caution (Hawley and others 2008) and the potential trade-offs between managing for biodiversity, heritage and carbon recognised and assessed.

Photo 3.1 Bare ground creation on dry heath, Ash Ranges, Surrey. © Natural England / Des Sussex

Heathland Vegetation

Sozanska-Stanton and others (2016) reported background GHG emissions for upland and lowland heathlands in Special Areas of Conservation (SAC) all over the UK between 0.013 and 0.2 t CO_2e ha^{-1} y^{-1} (except upland heathland in Sunart (Scotland) which had a sequestration rate of 5.6 t CO_2e ha^{-1} y^{-1}). These figures were similar to other semi-natural habitats except fens and bogs which had background emissions of around 2.3 t CO_2e ha^{-1} y^{-1}.

Morison and others (2012) reported that unmanaged, fenced upland heathland vegetation could potentially stock in excess of 180 t CO_2 ha^{-1} (49 t C ha^{-1}). Efforts to restore the habitat and increase its extent, rather than cultivate it or further disturb the soils with successive forestry plantation cycles should help to lock carbon in the vegetation and soils. In fact, at high altitudes, most of the carbon stores are associated with low-stature vegetation rather than with forest (Hartley and others 2012). Changing the land use or the type of vegetation cover can reduce soil and hence ecosystem carbon storage and even result in carbon losses due to changes in the belowground microbial communities (Friggens and others 2020).

Warner and others (2020) calculated that the greenhouse gas emissions for managing lowland heathland under the Countryside Stewardship option LH1 (Management of Lowland Heathland) were 0.059 t CO_2e ha^{-1} y^{-1} (0.054 t CO_2e ha^{-1} y^{-1} if emissions from livestock and machinery are excluded).

3.2.2 Carbon cycling and management interventions in heathlands

Heathland restoration from grassland

Grasslands can replace heathlands under increased nutrient loads, inappropriate burning regimes or excessive grazing (Alonso, Hartley & Thurlow 2001; Condliffe 2009) and converting upland heathland to improved grasslands results in net emissions (Dawson & Smith 2007). Dwarf shrub dominated vegetation has been reported to have higher carbon stocks and sequester more CO_2 than grass-dominated vegetation, which has higher respiration rates (Quin and others 2014, 2015; Thomas and others 2020; Urbina and others 2020). Sørensen and others (2018) also found that the litter carbon pool for heathland was significantly larger than for grasslands, as it decomposes, and therefore releases carbon into the atmosphere, more slowly.

Degraded (grass-dominated) areas had a higher carbon pool in the vegetation, but lower in the soil carbon and the total carbon stock was also lower than those with dwarf shrubs, either those restored or the target communities (Quin and others 2014) (Table 3.4). The heather-dominated community sequestered more than double the carbon than the grass vegetation (-3.45 ± 0.96 t C ha^{-1} y^{-1} vs -1.61 ± 0.57 t C ha^{-1} y^{-1}). This was also observed in subalpine grasslands being encroached by heathland (and other shrub) species (Urbina and others 2020). These values are in the middle of the estimated sequestration by UK woodlands: broad leaf woodland in UK sequester -1.8 to -7.3 t C ha^{-1} y^{-1}; conifers -2.1 to -6.5 t C ha^{-1} y^{-1} (Broadmeadow & Matthews, 2003 and others cited in Quin and others 2015). Therefore, restoring the ericaceous vegetation in lowland and upland heathlands on mineral soils cover can increase carbon sequestration in a way comparable to woodland.

Table 3.4 Summary table of the impacts of potential changes to heathland on total C stocks (t C ha^{-1}) as a result of degradation or restoration. UH = Upland Heathland; AG = Acid Grassland

Management	**Soil (t C ha^{-1})**	**Vegetation (t C ha^{-1})**	**References**
UH converted to AG	88.21 ± 4.53	14.3 ± 1.5	Quin and others (2014) 15cm soil depth
UH converted from AG	100.16 ± 5.66	12.1 ± 0.8	
Target UH, to compare (Lightly grazed, not burnt)	102.01 ± 4.10	12.0 ± 0.6	

Organic carbon stocks were modelled to a 1m depth by Aitkenhead & Coull (2016) in terms of broad soil types and broad habitat types in Scotland. According to these authors, heathland has 433.9 t C ha^{-1}, whereas deciduous woodland has 376.8 t C ha^{-1} and coniferous woodland has 345.9 t C ha^{-1}; in comparison the peatland stock is 631.2 t C ha^{-1}. However, these values are much larger than site-based data reported by Quin and others (2014).

Deforestation and afforestation of heathlands

Restoration of heathland from neglect (eg scrub removal) and from forestry can result in carbon emissions (Warner 2008; Sozanska-Stanton and others 2016) but it is likely to increase the biodiversity of the priority habitats, by providing the required open habitat niches (Webb, Drewitt & Measures 2010). Where biodiversity is the main driver of land use change, small changes to ecosystem carbon storage may have to be accepted and balanced with other potential benefits. However, the way the restoration is carried out could have important impacts of the carbon loss rates: eg rapid tree clear felling can lead to significant carbon losses, whereas carbon stocks could be maintained with a more gradual felling cycle (Broadmeadow & Matthews 2003).

On the other hand, there is mixed evidence from tundra studies that shrub expansion can cause carbon sequestration, release or be neutral, depending on the community it is replacing (Sørensen and others 2018). Heathland succession to woodland can increase C sequestration in the long-term by about 3 t CO_2e ha^{-1} y^{-1} (Sozanska-Stanton and others 2016), but can negatively affect the current soil C content and the biodiversity of a heathland site for no significant gain (Bartlett and others 2020; Lee and others 2019) or even lead to C losses (Table 3.5). Sørensen and others (2018) concluded that although increased scrub / shrub cover sequestered more C in tundra ecosystems, it also drained the carbon-rich soils because of the higher rate of decomposition of deciduous vegetation, compared with evergreen heathland.

Planting heathlands with trees may not render the expected carbon sequestration benefits, as most of the C is in the soil (Bartlett and others 2020; Friggens and others 2020). Planting on wetter soils further constrains or negates net carbon gains (Brown 2020). Planting trees in degraded upland heath can result in a decrease in C sequestration due to changes in organic matter depth and rate of decomposition (Mitchell and others 2007). The substantial ground disturbance associated with commercial tree planting negatively impacts soil C by exposing organic layers increasing its mineralisation.

In fact, tree planting in East Anglian heaths reduced soil C by approximately 0.6 t CO_2 ha^{-1} y^{-1} in 21 years (Morison and others 2012). Friggens and others (2020) reported significant losses in SOC due to increased soil respiration when birch or pine were planted on some heathlands in a period of 12–39 years, despite the increase in wood biomass and C. They reported no net ecosystem (below + above ground) C sequestration in that period. Lee and others (2019), also found that, in Norway, planting trees on coastal heath could result in 0.4 Tg C absorbed in 50 years, but the heaths already have 0.24 Tg C in the soil.

Brown’s (2020) analysis in Scotland of actual locations for recent afforestation showed that new woodland primarily occurred on land that was marginal for agriculture, usually on wetter uncultivated land, often on organic soils, which constrains net carbon gains. Thomas and others (2020) looked at the impact of land use change on SOC and found that restoration of heaths and bogs led to increases in SOC, both from “moorland grassland” and from “upland wooded” areas.

Friggens and others (2020) recommend evaluating SOC stocks before considering planting trees, rather than just looking at whether the peat is above or below 50 cm deep. Brown (2020) concurs, adding that there is a 'need for 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.

Table 3.5 Summary table of the impacts of changing scrub or tree cover on C fluxes (t CO_2 ha^{-1} y^{-1}). DSH = Dwarf Shrub Heath (broad habitat); UH = Upland Heathland; LH = Lowland heathland; (+) emissions; (-) sequestration

Habitat	Vegetation change	Carbon Flux	References
DSH	Natural afforestation	-3.0	Sozanska-Stanton and others (2016)
UH	Conversion to mix-plantation	+0.6	Morison and others (2012)
LH	Scrub/Trees removed	+2.56 to +4.46	Warner (2008)

Burning on heathlands

Management or controlled burning is used either to create structural diversity in the vegetation, mostly in the lowlands; or to create the best conditions for gamebird rearing, mostly in the uplands. This type of management, carried out in winter, is regulated by the Heather and Grass Burning regulations 2007 with good practice set out within the Heather & Grass Burning Code[18]. Upland and lowland heathlands can also burn in an uncontrolled way at any time due to arson or accidents in relation to recreational activities (Glaves and others 2020).

Controlled fires that do not damage the organic soil layer could be considered carbon neutral (Clay & Worrall 2008 cited in Legg & Davies 2009), as the vegetation recovers quickly, though this depends on intensity, severity and rotation length (Glaves and others 2013). Warner and others (2020) take a lifecycle assessment approach, considering that vegetation regeneration replaces carbon lost as burnt biomass, but calculate that the emission of N_2O and CH_4 equates to 0.072 t CO_2e ha^{-1} where 10 per cent of the area is burnt, or 0.006 t CO_2e ha^{-1} where this area is reduced to 0.9 per cent. However, Grau-Andrés and others (2019) found that heathlands change from C sinks to sources when burnt due to reduced photosynthesis, though to a lesser extent than raised bogs. Methane emissions from heathlands are negligible, according to the same authors. Sozanska-Stanton and others (2016) estimated that GHG emissions could be reduced by 7 t CO_2e ha^{-1} y^{-1} by stopping prescribed burning on heathlands (and moor grass), based on IPCC methods.

[18] Heather and grass burning code

Hot, uncontrolled fires are more likely to result in carbon release (Forgeard & Frenot 1996) via biomass loss and damage to soil profile, though not all wildfires are necessarily more intense or severe than controlled burns (Clay, Worrall & Rose 2010). Carey and others (2016) estimated GHG emissions from burning *Calluna* biomass following the legal codes to be +0.45 to +1.42 t CO_2e ha^{-1} y^{-1} (table 3.6). Longer burning cycles and smaller proportions burnt annually could help to reduce that figure. To reduce fire severity and C emissions, controlled fires should only be carried out when the Fuel Moisture Content (FMC) of the moss and litter layer is > 150 per cent and the soil > 200 - 300 per cent (Grau-Andres and others 2018).

Reverting drainage on heathlands
Carey and others (2016) and Sozanska-Stanton and others (2016) report high levels of GHG emissions from draining either upland or lowland heathlands. On the other hand, restoring wet habitats, including wet heaths, could potentially reduce those emissions by 0.353–0.821 t CO_2e ha^{-1} y^{-1} (table 3.6).

Grazing heathlands
Sozanska-Stanton and others (2016) indicate that grazing of heathlands can increase slightly GHG emissions (Table 3.6), and report potential reductions in emissions by 0.1 t CO_2e $ha^{-1}y^{-1}$ with a seasonal reduction in grazing. However, they used modelled results based on farmed animals, which also considered the disposal of manure and provision of feedstuff. When livestock grazing heathland systems are not supplementary fed, and the grazing pressure doesn’t lead to soil erosion or disturbance, those emissions should be even smaller than those modelled, and soil C stocks would be protected (Bartlett and others 2020).

Grazing animals can also affect the carbon stocks by reducing the amount of invading scrub and trees resulting in increased vegetation carbon; on the other hand, an adequate grazing pressure can conserve vegetation cover and soil carbon (Bartlett and others 2020).

Table 3.6 Summary table of the net effect of typical heathland interventions on C fluxes (t CO_2e ha^{-1} y^{-1}). UH = Upland Heathland; LH = Lowland Heathland; DSH = Dwarf Shrub Heath (broad habitat); (+) emissions; (-) sequestration

Habitat	Management	Net effect (t CO_2e ha^{-1} y^{-1})	References
UH	Conversion to grassland	+6.75 [(a)] +3.30 to +4.03 [(b)]	[a]Quin and others (2015) [b]Dawson & Smith 2007 [c]Sozanska-Stanton and others (2016) [d]Carey and others (2016) [e]Warner and others (2020)
	Grazing	+0.23 to +1.4 [(c,4)] (+0.06 in mountain heaths)	
	Burning	+0.47 to +0.5 [(c,d)] (+0.017 in mountain heaths)	
	Draining	+9 to +9.5 [(c,d)] (0 in mountain heaths)	
LH	Grazing	+1.4 to +1.56 [(c,d)]	
	Burning	+0.072[(a,d)] +0.43 to +1.42 [(c,d)]	
	Draining	0 to +17 [(c,d)]	
	Machinery use	+0.003[(e)]	
DSH	Reduction in grazing (per animal)	-0.1 [(c)]	

3.2.3 Climate change impacts and other interactions

Heathlands have a ’medium’ sensitivity to climate change (Natural England & RSPB 2020), particularly in relation to alterations in hydrology, as a result of changes in rainfall patterns and frequency of droughts. There is also a potential increase in the extent and frequency of fires due to raised temperatures. These factors coupled with increased nutrient availability (eg through atmospheric nitrogen deposition) could result in unpredicted and unwanted changes in vegetation composition and structure, such as increased biomass (Britton and others 2001; Field and others 2017) which could affect the current biodiversity of the habitat and carbon cycles. Increased nitrogen deposition leads to increased carbon sequestration in the litter and organic horizons until a point of saturation (Field and others 2017) and keeping the heather at a ‘building stage’ maximises the carbon sequestration.

There may be a trade-off between achieving the conservation objectives of heathland habitats and species and achieving climate mitigation objectives based on planting trees (Sozanska-Stanton and others 2016) or allowing trees to invade by secondary succession (Cordingley and others 2015). Planting or encroaching trees on heathlands will, with time (many decades), store more carbon than heathland vegetation, but this won’t necessarily lead to gains in C stocks in the whole ecosystem (Sozanska-Stanton and others 2016, Thomas and others 2020) and it is very likely to lead to biodiversity losses. There is therefore a need for careful planning of tree planting (NCC 2020). However, we have the opportunity to address both the biodiversity and the climate crisis, and there is evidence that in the past, projects that focused on carbon storage resulted in further biodiversity losses, whereas if the objectives were integrated both could be achieved in similar measure (Field and others 2020). Tree cover, particularly broadleaved (eg birch *Betula* spp or oak *Quercus robur*) could increase in some areas, to provide shading and reduce wildfire risk. However, to reduce the loss of heathland species and maintain favourable condition, tree cover should be kept below 15 per cent in lowland heathlands and below 20 per cent (scattered native trees and scrub) in upland heathlands.

Drought can result in increased CO_2 emissions in heaths due to increased soil respiration rates (Sowerby and others 2008; Carey and others 2016). Mesic sites are less likely to be affected or may recover quicker during the winter, but wet heathland soils did not recover from experimentally induced drought becoming net sources of CO_2 (Carey and others 2016). On the other hand, some characteristic species could find suitable habitat further north if their climate envelope increases (Dunford & Berry 2012).

Increased atmospheric nitrogen deposition is another significant global driver of change. There is inconclusive evidence of its effect on carbon sequestration on heathlands, as nitrogen can either increase carbon sequestration on heathlands (De Vries and others 2009) or result in emissions due to increased decomposition rates (Van Paasen and others 2020), depending on vegetation compositions and other variables.

Heathland management, including controlled burning, cutting and grazing by livestock may help to maintain the openness of the heathlands for the priority species and help to prevent or reduce the severity of wildfires (JNCC 2009a,b; Grau-Andrés and others 2018; Glaves and others 2020). It may be possible to increase the resilience of heathland sites by controlling pressures like heavy access, nitrogen deposition and continuing appropriate management (NE & RSPB, 2020).

Heathlands are subject to other global impacts, such as increased nitrogen deposition from the atmosphere (Fagúndez 2013) and climate change, which interact with carbon cycles. These factors could result in unpredicted and unwanted changes in the vegetation composition and structure (eg increased biomass (Britton and others 2001; Field and others 2017) which could affect the current biodiversity of the habitat.

3.2.4 Evidence gaps and future needs

The carbon stocks in wet and dry heathland soils respond differently to management interventions. More experimental research on the impacts on different types of heathlands would help when providing tailored advice and management.

There may be higher public and politic pressure to increase tree cover on heathlands, among other open habitats. However, the impacts of higher scrub and tree cover on carbon fluxes and particularly trade-offs with specialist species which require open niches are not clear.

The carbon impacts of using traditional management on heathlands (cutting, burning, ploughing) in order to maintain their biodiversity require more refined quantification, as well as an assessment of whether they can be mitigated in order to better support delivery for both biodiversity and climate change mitigation, eg by working on smaller areas at any given time or better use of arisings.

3.3 Semi-natural Grasslands

Grasslands form the most widespread land cover in the UK with pasture and semi-natural grassland making up approximately 40 per cent of land cover (ONS, 2015). Most grasslands in the UK would rapidly turn into scrub and woodland without grazing and cutting management, but some have persisted for centuries or millennia. The extent of grassland cover before forest clearance is unclear and disputed, but there would have been at least some areas of open woodland and glades and these may have been extensive. Grasslands represent a diverse range of habitats, from those which have been modified for productive purposes and consist of a generic and species-poor composition of agricultural grasses and clover, to species-rich semi-natural grasslands with distinctive plant communities which reflect local climate, soil, geology and management methods (Bullock and others 2011). This chapter covers semi-natural grasslands, with modified grasslands considered within the 'Arable Land and Agricultural Grasslands' chapter of this report.

Semi-natural grasslands provide provisioning, regulatory and cultural ecosystem services and as such are considered to have a high conservation value (Bullock and others 2011; Bengtsson and others 2019). They are typically created from low-intensity, traditional land management and have not been agriculturally improved using synthetic fertilisers, herbicides, re-sowing or cultivation (Ridding and others 2015). Despite their importance, semi-natural grasslands underwent significant declines in the 20th century with 97 per cent lost between 1930 and 1983 (Fuller 1987). These declines continue into modern times with a decrease in area of 47 per cent between 1960 and 2013, with losses being attributed to grassland 'improvement' or arable cultivation (Ridding and others 2015). The 2015 land cover map (Rowland and others 2017) indicates that 4.7 per cent of England is semi-natural grassland, compared to 32.8 per cent under improved grass. However, the bulk of this (3.6 per cent) is classified as acid grassland and may include degraded bog or heath habitats. Blackstock and others (1999) estimate semi-natural grasslands represent only 1–2 per cent of permanent grassland in lowland England and Wales.

3.3.1 Carbon storage and sequestration in semi-natural grasslands

Grassland soils

Carbon in grassland ecosystems is mostly stored belowground in soils rather than in the aboveground vegetation. The carbon storage and sequestration of these systems is particularly complex as they vary due to a wide range of factors, including soil, climate and management (Soussana and others 2004). The National Ecosystem Assessment (Bullock and others 2011) reported that grasslands hold the highest terrestrial carbon stock of any UK broad habitat due to their large area coverage (including improved grasslands). Due to their typically more extensive management and less disturbed soils semi-natural grasslands play a disproportionately important role in climate change mitigation. The most comprehensive survey of carbon storage in the UK's grassland habitats remains the Countryside Survey, a national survey of vegetation and soils spanning 1978–2007. The survey reports grassland soil carbon stocks in semi-natural grasslands range between 60 t C ha^{-1} and 87 t C ha^{-1} for English neutral and acid grasslands respectively (Emmett and others 2010).

When quantifying carbon stored in grasslands, estimates may be uncertain due to the potential for underestimating both the national extent of semi-natural grassland and the amount of carbon held in deeper soils. The specific importance of semi-natural grasslands regarding their carbon storage was reported by Field and others (2020) in their study of the value of habitats of conservation importance to climate change mitigation in the UK. Here the authors reported carbon stocks of 0.09 Gt C held nationally in the top 30cm of semi-natural grassland soils, behind only dwarf shrub (0.24 Gt C) and bogs (0.10 Gt C). However, the authors acknowledge this is likely to be a significant underestimate due to the uncertainty regarding the extent of non-intensive grasslands in the UK. Also, most grassland studies focus on the topsoil, the zone where management actions interact with the soils and vegetation. Considerable carbon stocks are believed to be held below 30cm depth. Ward and others (2016) estimate it could be around 60 per cent of total carbon, and in their study of a diverse range of grasslands across England estimated total soil carbon stocks to 1 metre to be 2097 Tg C.

3.3.2 Carbon cycling and management interventions in semi-natural grasslands

Sink equilibrium in grasslands
Soil type and management approaches are the major factors determining SOC stocks and along with pressures from climate change, pollution and other factors can make grasslands either a sink or source of carbon (Kühnel and others 2019, Thomas and others 2020). Not only does the current management of a site influence its flux status, but so does its historic management, which can exert a legacy effect many decades after a land management or land use change. Grasslands cannot be a perpetual sink for carbon, they instead reach an equilibrium following a change in management, which in some cases may be a century later (Smith 2014). Therefore, Semi-natural grasslands that are managed extensively over a long time period are not considered a significant sink of greenhouse gases, though empirical studies in undisturbed habitats are limited (Sozanska-Stanton and others 2016). Where the management change is positive for carbon sequestration, rates will be at higher rates initially before tailing off, making it important to understand the long-term management and any previous land use. Warner and others (2020), taking a lifecycle assessment approach, report that reversion of arable land to low input grassland under Countryside Stewardship will sequester 1.590 t CO_2e ha^{-1} y^{-1}. Carbon stocks are not permanent, and a reversion back to a land use with a lower equilibrium will result in a loss of the carbon gains.

Grazing
Grasslands are typically grazed or mown as a way of providing biomass to support livestock production and as such can result in the removal of a large proportion of the annual above ground net primary production. However, grazing is also an important conservation management tool to maintain sward composition and structure and prevent the build-up of nutrients. Grazing on a grassland may stimulate plant and root growth, taking up carbon from the atmosphere, while over-grazing will damage the sward, leading to bare ground,

compaction and physical degradation of the soil, actively depleting the soil of carbon (Eze and others 2018b).

Photo 3.2 Conservation grazing using longhorn cattle © Natural England / Peter Roworth

The impact of grazing is highly variable, site specific and claims of significant carbon gains from adopting new grazing practices is lacking in empirical evidence with the greatest potential on already degraded sites (Garnett and others 2017). A move to more extensive management, including reduced livestock numbers have been attributed to enhanced soil carbon sequestration by European grasslands, including those in the British Isles (Chang and others 2016). However, a move to more extensive management may also mean an associated reduction in ploughing and reseeding, the contribution of which is not disaggregated by Chang and others (2016). Any potential gains in soil carbon sequestration also need to be balanced with emissions of methane and nitrous oxide that result from the grazing of ruminant animals.

Grazing is just one element within many other complex factors which interact to influence SOC stock, including fertiliser regime, liming, soil properties and climate. Despite the many factors, a meta-analysis of the impact of grazing intensity on extensive grassland SOC in moist cool climatic zones (relevant to the UK climate) found that all grazing resulted in a decrease of SOC (Abdalla and others 2018). In a global meta-analysis grazing was found to decrease SOC stock by 15 per cent; heavy grazing intensity resulted in a loss of 27 per cent, double that of lighter grazing regime (Eze and others 2018b). This was attributed to removal of vegetation and less organic matter return to the soil in the form of litter, with greatest reductions reported in the tropics and the least in temperate zones. Liming and addition of fertilisers increased the SOC stock, but only partially mitigated the loss from grazing. Such results suggest a potential trade-off between management of specialist grassland species for biodiversity and for carbon. Further UK specific work is required to understand the interactive nature of grassland management techniques and their impact on carbon stores of semi-natural grasslands. Grazing levels are often used as a proxy for the intensity of management, but associated management interventions also need to be considered, particularly any reduction in grassland cultivation that may be associated with a move to a more extensive farming system.

Relative to grazing, the impact mowing has on grasslands has been rarely looked at despite it being an important part of management, particularly for grasslands such as hay meadows that are of conservation importance. No data was found in the UK grassland context but a study from permanent semi-arid pasture under traditional extensive management in the Kiskunság National Park, Hungary, observed 20 per cent higher soil respiration rates under mowing treatments than grazed treatment, with the 17 per cent greater aboveground biomass under mowed grass being put forward as a possible explanation (Koncz and others 2015). However, further studies are required before the impact of mowing management practices on grassland carbon can be fully understood.

Management intensity

Drivers of carbon storage and sequestration are complex, interactive and as such, studies of grasslands under long-term management are important in order to understand them further. A study of English grasslands by Ward and others (2016) demonstrated that total carbon (the study did not separate inorganic and SOC) stored in grassland soils is vulnerable to a site's management legacy, which exerts an influence to a significant depth. Three broad categories of management intensity were used; intensive, extensive and intermediate, and are defined in table 3.7. Carbon stocks (total C and accounting for differences in bulk density) were found to be greatest under intermediate management, followed by extensive management (mapped to semi-natural broad habitat types), with the lowest stocks reported under intensive management.

This makes the difference in total carbon stock at 1 m depth 10.7 per cent greater under intermediate management relative to intensive, equating to a difference of 10.1 t C ha^{-1} in surface soils and 13.7 t C ha^{-1} from 30 cm to 100 cm. The authors suggest that the balance between organic inputs, including addition of organic manure and limited grazing pressure was conducive to the greater carbon stocks observed. The study did not consider the grassland age or history of regular cultivation and reseeding required to maintain intensively managed grasslands, which may have influenced carbon stocks of these sites. Kuhnel and others (2019) also reported the importance of the return of organic matter via manure inputs in maintaining or increasing SOC in their monitoring of grasslands in Bavaria, Germany, over 27 years. However, the benefits to soil carbon stocks need to be balanced with the GHG emissions associated with manure and fertiliser regimes (Garnett and others 2017).

Table 3.7 Total carbon storage in grasslands, disaggregated by management intensity (Ward and others 2016)

Management Intensity	**Total C Stock - kg C m^{-2} (C t ha^{-1})**	**Reference**	**Notes**
Extensive – relatively high plant diversity and conservation status, typically receives less than 25 kg N ha^{-1} y^{-1}, and have been managed in traditional, low intensity manner for many decades	41.38 (413.8)	Ward and others 2016	Total carbon, to 1 m depth
Intermediate – typical inputs of 25–50 kg N ha^{-1} y^{-1}, and intermediate levels of plant diversity, grazing and cutting	44.62 (446.2)		
Intensive – low plant diversity of mainly MG6 and MG7 NVC communities, typically receive > 100 kg N ha^{-1} y^{-1}. Have been under higher grazing pressure and more frequent cutting for silage since the 1950's	40.30 (403.0)		

Similar findings to Ward and others (2016) are reported by Allard and others (2007), working on French upland sites, who observed that while a decrease in fertiliser input and grazing pressure strongly reduced both CH_4 and N_2O emissions. Such approaches gradually reduce capacity of the grassland to store carbon, due to the reduced nitrogen status of the extensive grassland. This suggests a balance; semi-natural grasslands are of importance for their high nature value, potentially large belowground carbon stocks, and the reduced emissions associated with extensive management, but their role in sequestering carbon long-term may be more modest (Sozanska-Stanton and others 2016).

Grassland broad habitat types

Grasslands vary considerably, both in plant communities and how the land is utilised. Three main categories exist for semi-natural grasslands in England; acid, calcareous and neutral. Here, evidence specific to each is reviewed, regarding how condition, land use and management interventions influence semi-natural grasslands in storing and sequestering carbon.

3.3.3 Acid grasslands

Acid grasslands are a broad habitat type occurring on soils with a pH below 5.5. In dry conditions, on brown podzolic soils or rankers, they typically consist of fine grasses, such as common bent *Agrostis capillaris*, sheeps fescue *Festuca ovina*, and sweet vernal grass *Anthoxanthum odoratum*. On organo-mineral soils moorland grasses, such as matgrass *Nardus stricta* and purple moorgrass *Molinia caerulea*, dominate. Acid grasslands tend to be botanically less diverse than the other grassland broad habitat types. The CS2007 reports that the carbon stock (0–15 cm depth) of acid grassland soils in England was 87 t C ha^{-1}. This was the second largest carbon stock that the CS2007 reported, next to dwarf shrub, for broad habitats in England, but both occur together in habitat mosaics.

The relatively low productivity of acid grassland and low pH of their soils means these grasslands are characterised by extensive management, typically grazed by low densities of sheep and cattle. The bulk of acid grassland occurs in the uplands, although there are lowland acid grasslands of significant conservation importance. Some acid grassland, especially in the uplands, may be degraded heath or peatland habitats, in which case carbon gains may be made by restoring such sites back to mire or heath vegetation (Quin and others 2014, 2015; Field and others 2020). Balancing grazing management is vital to the maintenance of these grasslands, as their soils and vegetation can be easily damaged by over grazing, but under grazing can allow the establishment of scrub and damage their value for nature conservation.

Above ground biomass is a potentially dense store of carbon in these habitats and grazing directly impacts the grassland vegetation structure as well as the plant matter return to the soil. Carbon stocks in the plant biomass of purple moorgrass swards in Scottish acid grasslands increased with a reduction in grazing intensity (table 3.8) (Smith and others 2014). Using the Rothamsted Carbon Model to ascertain the impact of the grazing regimes over a hundred year time frame on SOC, commercial grazing was predicted to lead to a loss of 23.11 t C ha^{-1}, while increase in SOC were predicted in the grazing exclusion (14.36 t C ha^{-1}) and low grazing (13.62 t C ha^{-1}) regimes (Smith and others 2014). This long-term modelling approach differs from the empirical observations by (Medina-Roldán and others 2012) who reported that a seven year grazing exclusion in upland acid grassland soils resulted in increasing the proportion of dwarf shrubs at the expense of graminoids, slowed down soil nutrient cycling processes and increased litter return and storage. However, it had no detectable impacts on the total carbon and nitrogen stocks in the surface soils, though seven years may not be a long enough timeframe during which to detect change in soils with a large starting carbon stock.

Warner and others (2020) investigated the impact the Environmental Stewardship agri-environment scheme had on two case study agricultural estates in England. They reported that, despite the implementation of conservation management grazing across the site, options sited on rough permanent grasslands showed the largest declines in SOC over the ten-year agreement period (loss of 2.69 - 4.55 t C; 9.86-16.68 t CO_2e ha^{-1} y^{-1}). This loss is potentially linked to habitat degradation by former management approaches (including drainage) and suggests the 10-year Environmental Stewardship agreement length is not long enough for restoration of soil functional processes to be restored. SOC is typically lost at a faster rate after a change in management than it is gained under restoration, and, if managing for carbon a permanent approach is needed to effect change.

Bracken *Pteridium aquilinum* may colonise and form a dense cover on acid grassland. Bracken has become increasingly widespread in recent years and is considered a threat to priority habitats, archaeology, and agricultural production, and as such is often viewed as needing to be managed. While it may support some woodland associated ground flora, in general it is considered floristically poor (Pakeman and Marrs 1992). Despite being highly productive, there is very little in the literature to provide an understanding of its contribution to carbon stocks. Using supporting data provided by Rowe and others (2016) in their study of productivity and soil macronutrients under bracken stands in upland areas of England and Wales, we calculate a mean soil carbon stock of 55 t C ha^{-1} under bracken. However, stocks ranged between 13–119 t C ha^{-1} across the 49 reported sites. Hagon and others (2013) in their 'Managing land for carbon' guidance for land managers in the Lake District, cite 77.1 t C ha^{-1} in the top 15 cm of soil under bracken, and 2 t C ha^{-1} in the vegetation, but do not provide the original source of the data.

Table 3.8 Summarising the carbon stock and flux associated with acid grasslands. For fluxes, negative numbers indicate net gain of carbon

Land use / Management	**Carbon Stock** (t C ha^{-1})	**Carbon flux** (t CO_2 ha^{-1} y^{-1})	**Reference / notes**
CS 2007 Soil Survey	Soil: 87	-	Emmett and others 2010 (0-15cm)
Molinia caerulea swards under commercial grazing (2.7 ewes ha y)	Biomass: 3.83	+0.85	Smith and others 2014 (adjusted to annual flux by report authors, rather than over 100 years as originally reported. See text.)
Molinia caerulea swards under low level grazing (0.9 ewes ha^{-1} y^{-1})	Biomass: 5.01	- 0.50	Smith and others 2014 (adjusted to annual flux by report authors, rather than over 100 years as originally reported. See text.)
Molinia caerulea swards – ungrazed	Biomass: 6.85	- 0.53	Smith and others 2014 (adjusted to annual flux by report authors, rather than over 100 years as originally reported. See text.)
Grazing exclusion –7 years	Soil: 58.89	-	Medina-Roldán and others 2012 (total carbon reported)
Grazed – sheep	Soil: 62.38	-	Medina-Roldán and others 2012 (total carbon reported)
Agri-environment scheme on degraded grassland	-	+9.86 to +16.68	Warner and others 2020 (Rough permanent grassland where historic drainage still influences water levels)
Bracken	Soil: 55	-	Rowe and others 2016
Bracken	77.1	-	Reported in Hagon and others 2013

3.3.4 Calcareous grassland

Calcareous grasslands are found on base rich soils with a pH higher than 6.5 overlying substrates such as limestone or chalk. Due to the free draining nature of the soils these grasslands tend to be dry habitats and prone to desiccation during summer months. The soil structure and texture coupled with the highly alkaline conditions means these habitats have distinctive plant communities and are the most diverse of the semi-natural grasslands in terms of wildflowers and grasses (Wilson and others 2012).

Photo 3.3 Wildflowers on a chalk grassland, Hampshire © Natural England / Chris Gomersall

Calcareous grasslands are a relatively rare habitat with less than 41,000 ha remaining in the lowlands, and less than 25,000 ha of upland sites. Their coverage has declined and become fragmented as a result of agricultural intensification, especially on flatter sites. Sites that have avoided this fate are often situated on steep slopes and have rocky outcrops making them unsuitable for modern machinery. As a result, they tend to be managed as extensive grazing pasture and are at risk from abandonment leading to encroachment by rank grasses and scrub.

Regarding reporting of carbon stocks and fluxes, calcareous grasslands are underrepresented in literature with many studies tending to focus upon their biodiversity and conservation value (Table 3.9). The Countryside Survey (2007) was unable to report on soil property trends from calcareous grasslands due to an insufficient number of samples but estimate their typical soil carbon stock to be approximately 69 t C ha^{-1} to a 15cm depth (Emmett and others 2010). These grasslands typically occur on shallow rendzina soils, which will further constrain the amount of organic carbon they can potentially store but may include large amounts of carbon in a mineralised form in their subsoils and bedrock.

Studies of alpine calcareous pastures in continental Europe offer some reference regarding carbon stocks. Thuille & Schulze (2006) report that total ecosystem carbon stock at the calcareous meadow stage prior to afforestation was 75 t C ha^{-1} to maximum depth of 70 cm, but such depths may not be representative of UK calcareous grasslands which tend to be shallower (R. Jefferson, pers. comms.). Niklaus and others (2001) observed total stocks of approximately 51 t C ha^{-1} to depth of 10 cm, with 44 t C ha^{-1} associated with litter and belowground stocks. Further work is necessary in order to align their importance for climate change mitigation with their recognised role in supporting biodiversity.

Table 3.9 Summarising the carbon stocks reported for calcareous grasslands

Land use / Management	Carbon Stock (t C ha^{-1})	Reference / notes
CS 2007 Soil Survey	69	Emmett and others 2010 0-15 cm depth, SOC.
Calcareous alpine meadow	75	Thuille and Schulze, 2006 0 – 70 cm depth, total C reported.
Upland grazing pasture	51	Niklaus and others 2001 0 – 10 cm depth, estimates include vegetation, litter and soils, total C reported.

3.3.5 Neutral grassland

Neutral grasslands occur in the lowlands and uplands. Also termed mesotrophic grasslands, this grassland type occurs on soils that are neither strongly acid nor alkaline, with a pH range of 5.5–6.5. The neutral grassland broad habitat includes grasslands on neutral soils that range from unimproved to semi-improved, usually with moderate to high species richness and a low cover of agriculturally favoured species such as perennial ryegrass. They may be managed as pasture, hay meadow or for silage or, outside of agricultural holdings, by episodic cutting such as on road verges. The CS2007 reports that the mean carbon stock (0–15cm depth) of neutral grasslands in England is 60t C ha^{-1} (Emmett and others 2010).

A study of upland neutral grasslands under traditional hay and silage in Northern England found these sites stored significant amounts of carbon in their soils (0–15cm), ranging from 58.93 ± 3.50 to 100.69 ± 8.64 t C ha^{-1} (Eze and others 2018a). Significantly greater stocks were reported under the silage pasture, which received inorganic nitrogen additions which the authors hypothesised increased litter return and belowground root biomass, both of which could positively influence carbon stocks. Organic matter return via manure additions and excretal return was relatively similar across the sites. Despite these differences the grasslands were net carbon sinks irrespective of their management and fertiliser regimes, with the sites removing 1822–2758 g CO_2e m^{-2} y^{-1} (18.22–27.58 t CO_2e ha^{-1} y^{-1}) (Eze and others 2018c). Sites subject to N fertilization were greater annual sinks due to available nitrogen influencing gross primary productivity. The carbon sink reported by Eze and others (2018c) is very high, larger than those reported in European grassland sites due to low ecosystem respiration. As the reported values represent net ecosystem exchange (NEE) only, and do not include carbon exports from site via cutting or livestock grazing, we do not present them as representative for neutral grasslands.

Fornara and others (2013) report increased carbon sequestration in lowland mesotrophic grasslands after 19 years of nitrogen only additions. Relative to the 'no input' hay meadow control sites the fertilised sites stored on average 11 t C ha^{-1} more carbon in their soils. Again, the authors linked this finding to greater plant detritus in the topsoil. These findings suggest that neutral grasslands have the capacity to store more carbon, but it needs to be balanced with possible impacts to other ecosystem services such as biodiversity or water quality via the transport of nitrogen and phosphorus. Any gains in soil carbon sequestration must be balanced with emissions of methane and nitrous oxide that may result due to the storage and application of fertilisers

Restoration of plant diversity on species rich neutral grasslands has been demonstrated to have carbon benefits. De Deyn and others (2011) observed that long-term biodiversity restoration approaches of an upland meadow increased soil carbon and nitrogen storage, with the greatest rates in treatments that were combined with promotion of the nitrogen fixing legume red clover *Trifolium pratense*. Sequestering 317 g C m^{-2} y^{-1} (11.62 t CO_2e ha^{-1}) in the most successful management treatment, this is over 5 times greater than the average C sink estimated for European grasslands (Janssens and others 2005, as reported by De Deyn and others 2011). Further understanding is required regarding how restoration of plant communities can impact on soil properties, with such studies advocating for a combined above-below ground approach when setting in place land management for biodiversity. Table 3.10 summarises the carbon stocks and fluxes reported in literature for neutral grasslands.

Table 3.10 Summarising the carbon stock and flux associated with neutral grasslands. For fluxes, negative values indicate net carbon gain

Landuse / Management	Carbon Stock (t C ha^{-1})	Carbon flux (t CO_2e ha^{-1})	Reference / notes
Neutral grassland (CS 2007 Soil Survey)	60	-	Emmett and others 2010 0 – 15 cm depth
Upland hay meadow – established for 150 years	58.93 ± 3.50	-26.73 ± 1.34	Eze and others 2018a; Eze and others 2018c 0 – 15 cm depth
Upland hay meadow – established for 25 years	68.74 ± 3.15	-18.22 ± 0.85	Eze and others 2018a; Eze and others 2018c 0 – 15 cm depth
Upland permanent pasture – grazed for 25 years, no inorganic fertiliser additions	64.58 ± 6.29	-22.69 ± 0.61	Eze and others 2018a; Eze and others 2018c 0 – 15 cm depth
Neutral grassland – no nutrient additions	33.31	-	Fornara and others 2013 Carbon values cited for control plots
Upland meadow	46.3 ± 0.08	-11.62	De Deyn and others 2011 C rates reflect treatment under seed addition, cessation of fertiliser, and addition of *T.pratense.*

3.3.6 Climate change impacts and other interactions

Grasslands have a low–medium sensitivity to climate change, varying according to the priority grassland type, with pressures of management change and the influence of farm economics expected to exert greater pressure on semi-natural sites (Natural England and RSPB 2020). At greater risk are wetter sites such as flood meadows, and grasslands on organic soils, where availability of water and seasonal variation in precipitation will impact on the species composition of grasslands, and upland sites which may experience increased competition from lowland species.

Grassland productivity is directly related to climate variables such as annual temperature and precipitation and therefore their carbon stocks and balance are sensitive to future changes in climate. While grazing management may result in a decline of organic carbon held in grassland soils, it has been shown that this may be mitigated by rises in mean annual temperature and precipitation that extend the growing season, enhance plant growth and promote the return of carbon to the soil (Eze and others 2018b). While the understanding of climate change – management interactions in temperate grasslands is incomplete, this does indicate temperate grasslands are potential carbon sinks in the face of climate change.

Extended periods of drought however may negate these benefits, resulting in net carbon loss from the soil (Lei and others 2016). Restoration of grasslands to restore their species diversity has been demonstrated to have benefits for carbon accumulation (De Deyn and others 2011). Results from manipulated grassland plant diversity experiments (Isbell and others 2015) suggest that diverse grassland communities are more resistant in the face of extreme climate events, such as drought or flooding, with biodiversity playing a role in stabilising ecosystem productivity and productivity dependent ecosystem services. Experimental neutral grassland plots at Colt Park, Northern England, have demonstrated that grassland biodiversity restoration can increase the resistance of carbon fluxes to summer droughts, possibly due to the species rich grassland having an lower productivity baseline and reduced water demand (Cole and others 2019). Therefore, management decisions aimed to confer grassland resistance to future extreme weather events may have to consider trade-offs with agricultural yields.

There is also a need to understand how responses vary at a site level and the influence of seasonal variation in local climate. Long-term monitoring of Bavarian semi-natural grasslands demonstrated that seasonal climate variables explained the highest variability of SOC stock changes in established grassland sites. Increasing autumn precipitation led to decreased SOC stocks, whereas increasing spring and summer precipitation led to increased SOC stocks on sites at high elevations with low slopes (Kühnel and others 2019). Future responses to climate change will not be uniform but instead driven by complex interactions of locally specific factors.

Increased nitrogen deposition has been shown to influence species diversity in grasslands, with a clear negative relationship between deposition and species richness reported for acid grasslands (Stevens and others 2004) with a more variable response observed in neutral and calcareous grasslands (Stevens and others 2016). Nitrogen deposition driven changes to grasslands is expected to continue in the coming decades with changes to species composition, reduced occurrence of terricolous lichens and reduced species richness, especially in acid grasslands projected to be evident by 2030 (Stevens and others 2016). Nitrogen deposition may impact in carbon cycling, through fertilisation of nitrogen limited habitats leading to increases in net primary productivity and greater return of organic matter in the form of plant biomass to the soil. Tipping and others (2017) report this may have resulted in an additional 38 Mt of SOC accumulated in the UK between 1750 and 2010. However, this additional carbon stock is sensitive to future deposition rates and changes to management. Much of the carbon is held in the 20 year carbon pool, and with anthropogenic N deposition expected to decline, could result in increased CO_2 emissions from semi-natural grassland soils in the coming decades and within the timeframe of the UK's target to meet net zero GHG emissions by 2050.

3.3.7 Evidence gaps and future needs

There remain significant evidence gaps with regards to semi-natural grasslands and their carbon stocks and sequestration. The most comprehensive assessments of semi-natural grassland soil carbon in the UK are the Countryside Survey, which last reported in 2007 and Ward and others (2016) who reported findings based on management intensity rather than grassland type. Greater distinction is required between semi-natural grassland types and their location in the landscape, for example upland vs lowland systems, and their interaction within semi-natural habitat mosaics. Evidence is especially lacking for calcareous grasslands.

There are few carbon flux datasets from grasslands and very few which assess changes with different management practice or grassland restoration from arable sites. Taking a carbon stock change approach (as in Ward and other 2016) by comparing a comprehensive range of carbon stock values from grassland sites under contrasting management could be an approach to fill this evidence gap.

Restoration of semi-natural grasslands remain a priority for their biodiversity importance, but evidence is required to aid understanding between the trade-offs and synergies between the specific management interventions required to manage these habitats and their potential to store and sequester carbon. This should include consideration of soil type, and management approaches such as cutting, liming, and burning, where traditionally practiced. While studies have shown that restoration of grassland vegetation can be positive for carbon storage (De Deyn and others 2011) further work is necessary to improve the understanding of the impact aboveground biodiversity conservation approaches may have on soil properties and mitigation of climate change.

Grasslands hold most of their carbon in their soils, but studies typically sample only the topsoils, often limited to 15 cm depth. Studies have shown that significant carbon stocks are held below this depth. Further work is required to quantify carbon stores at depth and their interaction and sensitivity to grassland interventions, as well as understanding how the legacy of past management can continue to influence a grassland's carbon storage potential.

3.4 Arable land and agricultural grassland

Natural habitats are a product of a specific set of factors, relating to soil and climate, that play a key role in determining the distinctive plant and soil-based communities that constitute a habitat. Agricultural systems, on the other hand, exist on a broad range of soil types, often considerably modified from their natural state through drainage, liming, cultivation, fertilisation and other 'improvements' in order to produce a crop. These agricultural improvements and practices can lead to the loss of SOC which can ultimately lead to soil degradation and associated environmental impacts.

Agricultural systems such as annual cropland and non-wooded grassland lack long-lived woody biomass. Their plant biomass stocks are relatively small and seasonal due to annual harvesting and grazing. The only large and persistent organic carbon stock is therefore in the soil (Paustian and others 2019a). Arable and improved grassland systems, however, are extremely difficult to characterise with regards to 'typical' carbon storage and sequestration, as these systems can be under a broad range of management regarding fertiliser inputs and disturbance as well as located on a wide range of soil types, each having their own properties with regards to carbon dynamics. Agricultural grasslands typically have higher levels of SOC compared to arable land due to their vegetation cover, increased inputs to the soil via residues and root exudates, and reduced disturbance (Buckingham and others 2013).

Intensification of arable systems has led to increased crop yields but has also resulted in negative environmental impacts such as soil erosion, compaction and loss of associated farmland biodiversity (Westhoek and others 2013). Loss of organic matter in UK soils is estimated to cost in excess of £500 million annually (Environment Agency 2019). These simplified arable systems depend on fossil fuel agro-chemical inputs such as fertilisers and pesticides, which have negative environmental impacts on multiple scales, from the immediate soil environment to catchment-scale water quality impacts to regional alterations in atmospheric nitrogen deposition to global shifts in greenhouse gas concentrations (Jenkinson 2001). However, while these additional emissions will be considered, this report will focus on the carbon stored and sequestered in the soils and vegetation of agricultural land.

Agricultural land currently covers 70 per cent of the terrestrial area of England. The UK target to meet net zero is dependent on making changes to the way we use and manage our land, with agricultural land often forming the baseline of land use change. The Committee on Climate Change (2020) recommend that around one-fifth of agricultural land will need to be released before 2050 for actions that reduce emissions and sequester carbon. The large area of agricultural land in England, and its management, means it plays a significant role in England's carbon balance. The sector contributed 10 per cent in total national GHG emissions in 2019 (BEIS 2021).

3.4.1 Carbon storage and sequestration in arable land and horticultural land

The calculation of soil carbon stocks requires a measurement of soil organic carbon (SOC) concentration, soil bulk density (BD), stone content, and soil depth all of which are spatially variable and prone to measurement errors (Schrumpf and others 2011). Arable and horticultural soils have the lowest carbon stocks, 43 t C ha to 15 cm depth, of all the broad habitats reported by the CS2007 (Emmett and others 2010). However, due to the extent of land cover of this land use, this equates to 11.9 per cent of the belowground carbon stock of Great Britain, an amount comparable to that of woodland (11.4 per cent) (Ostle and others 2009). Following a systematic literature review, Cantarello and others (2011) assigned mean, minimum and maximum vegetation and soil carbon stocks to 11 land use types in South West England, including non-irrigated arable land. Their results demonstrate the variability of these systems, with soil carbon stocks ranging between 27.5–88.2 t C ha^{-1}, with a mean average of 63.9 t C ha^{-1} to 30cm depth.

The focus on the carbon stock of agricultural topsoils will miss a considerable amount of carbon held at depth (table 3.11). A soil carbon and land use database was created by Bradley and others (2005) based on interpolated data on soil types and land use from across the UK to estimate stocks of soil carbon at depths of 0–30 cm and 30–100 cm. Reporting SOC stocks under arable land of 69.8 t C ha^{-1} and 119.9 t C ha^{-1} at depths 30cm and 100cm respectively, the values are similar to those reported by Cantarello and others (2011). Moxley and others (2014) also use the work of Bradley and others (2003, 2005) to underpin reference SOC stocks in their work investigating the impact cropland and grassland management has on soil carbon in the UK's LULUCF inventory, citing stocks at equilibrium to be 120 t C ha^{-1} (1 m depth) for English arable soils.

Table 3.11 Carbon stocks in soils and vegetation under arable land use, as reported in the literature

Agricultural land use	Soil carbon stock (t C ha^{-1})	Vegetation carbon stock (t C ha^{-1})	Reference / notes
Non-irrigated arable land	27.5 – 88.2	1.56 – 4.47	Cantarello and others 2011 Soil depth: 30 cm
Cultivated land – average UK	70 (120)	-	Bradley and others (2005) Soil depths: 30cm (1m)
Arable – horticultural soils	43.02	1	Ostle and others 2009 with soils data taken from CS2007 Soil depth: 15cm
Arable	120	-	Moxley and others 2014, based on work from Bradley and others (2003, 2005) Soil depth to 1m

While carbon trends have been the subject of debate over the last two decades, arable land management is generally considered to be a net source of carbon emissions (Buckingham and others 2013). Data from the Countryside Survey 2007 report a consistent loss of SOC from arable and horticultural since 1978, the only land use category to do so (Emmett and others 2010). Also using CS2007 data, Chamberlain and others (2010) report that the total stock of topsoil carbon in arable soils decreased by 45 million Mt C during the period 1998 to 2007. Applying the Roth-CNP model, Muhammed and others (2018) show this decline has occurred over the long-term, estimating that arable land in the UK has lost 0.18, 0.25 and 0.08 t C ha^{-1} y^{-1} during the time periods 1800–1950, 1950–1970 and 1970–2010 respectively. This long-term loss is because carbon losses from arable land were higher than the carbon inputs, from plant residues for example, but may be underestimated due to the difficulty in capturing loss represented by extreme, high intensity, short duration events such as soil cultivation and erosion.

How arable land is managed will impact its ability to sequester carbon. Most carbon in arable systems is stored in soils and only released as a result of natural (such as erosion) and anthropogenic processes (such as field cultivations), physically degrading the soil resource. Erosion is a natural process but becomes a problem when the rate outstrips soil formation, leading to carbon loss via water, wind and soil removal during harvest. Seventeen per cent of arable soils in England and Wales show signs of erosion, and 40 per cent are thought to be at risk (Environment Agency 2019). Slope aspect is an important determinant of soil erosion, with Borelli and others (2016) estimating carbon through soil erosion on Italian arable soils is between 0.05 t C ha^{-1} y^{-1} (0.18 t CO_2e ha^{-1} y^{-1}) for negligible slopes to 0.1–0.3 t C ha^{-1} y^{-1} (0.37–1.10 t CO_2e ha^{-1} y^{-1}) on steep slopes. Warner and others (2020), looking at the potential impact Countryside Stewardship options deliver for climate change mitigation suggest a baseline loss of 0.7 t CO_2e ha^{-1} y^{-1} for English agricultural soils. Providing year-round vegetative cover and having barriers in the landscape are considered the most effective measures to mitigate carbon loss from erosion (Tautges and others 2019).

Tillage

Around 3.9 million hectares of agricultural land in England and Wales is at risk from soil compaction, often as a result of heavy machinery use and working the land for extended periods and in wet conditions. Compaction can lead to increased nitrous oxide emissions, risk soil loss via sheet erosion, increase energy use for cultivation and reduce vegetation organic matter return to the soil. Topsoil compaction has been estimated to reduce biomass accumulation and organic matter return to the soil by up to 13 per cent (Louwagie and others 2008).

Tillage is a management practice often applied to arable land to reduce the build-up of weeds, pests and soil compaction. Zero- and minimum-till have previously been advocated as approaches to promote carbon sequestration in arable soils (Alonso and others 2012) but this may not be the case. Rather than increasing SOC, reduced tillage practices are instead believed to redistribute SOC in the soil (Moxley and others 2014; Powlson and others 2014). Crop residues accumulate on the soils surface rather than being incorporated at depth causing an increase in topsoil carbon and decreasing at depth. The interaction of tillage practices with depth is important to understand as not quantifying impacts below the plough line may have led to overestimates of the benefits of zero-till. Reducing tillage intensity may result in reduced soil bulk density, as a result of reduced disturbance of the soil profile. Monitoring carbon stock changes using fixed depths, and not considering this influence of bulk density via use of the equivalent soil mass approach, has been demonstrated to lead to overestimate of soil carbon in surface layers, but underestimates at depth (Xiao and others 2020).

Table 3.12 Carbon sequestration rates under arable land practices, as reported in the literature (+ indicates loss)

Agricultural land use	Sequestration rate (t CO_2e ha^{-1} y^{-1})	Reference
Land under arable	+0.29	Muhamed and others 2018
Arable - erosion	+0.7	Warner and others 2020
Zero and min till	0	Moxley and others 2014

The stabilisation of carbon in arable soils can have differential effects according to soil depth. Ignoring the dynamics of carbon sequestration in deeper layers of soil may lead to false conclusions about the impact of management practices on C sequestration. Changes to tillage practices have impacts beyond the carbon status of arable soils. The impact of reduced tillage on emissions of N_2O are of interest due to its powerful global warming potential and increases could offset any gains in SOC. However, findings in the literature have been highly variable and influenced by the crop, soil type and local climate, making a generalised conclusion difficult to make (Abdalla and others 2013; Guenet and others 2020). Undertaking approaches that ensure more efficient application and uptake of fertilisers will also have GHG benefits through decreasing emissions from their use and manufacture, which are significant. Moves to minimum and zero-tillage may also result in reduced operational greenhouse gas emissions; management changes from deep and shallow tillage have been demonstrated to reduce fuel use by 60 and 47 per cent respectively (Godwin, 2014).

Photo 3.4 bare soil after ploughing ©Natural England / Paul Muto

Trade-offs with nitrogen inputs

The total GHG emissions of cropping systems do need to be considered so we can fully understand their potential to mitigate climate change. Soil carbon sequestration can benefit soil fertility, though there are trade-offs that may negate any mitigation potential. In addition, targets such as the '4 per 1000'[19] initiative call for increases in SOM in order to deliver soil carbon benefits, particularly to degraded soils. The sequestering of carbon increases the demand for nitrogen and other nutrients, potentially via inorganic fertilisers or symbiotic N_2 fixation, which is often omitted from carbon balance calculations (van Groenigen and others 2017).

The incorporation of residues from N-fixing cover crops has been found to increase soil carbon sequestration, however, there was also an increase in N_2O emissions (Lugato and others 2018). As N_2O is a potent greenhouse gas, this outweighed any gains due to the sequestration of soil C. This demonstrates the importance of accounting for total GHG emissions associated with each cropping system. Hijbeek and others (2019) concluded that while soil carbon sequestration can play a role in mitigating climate change, it cannot compensate for total agricultural GHG emissions. Nonetheless, sequestering carbon can be beneficial for climate change mitigation in soils which can stabilise carbon and beneficial for improving soil fertility in soils where carbon retention is minimal. It is important to remember that SOC sequestration is a reversible process and changes in agricultural practices need to be maintained in order to contribute to an improved GHG budget (Andrén and Kätterer 2001).

[19] https://www.4p1000.org/

3.4.2 Leys and cover crops in arable rotations

Intensification of arable systems has led to increased crop yields, but has also resulted in negative environmental impacts such as soil erosion, compaction and loss of associated farmland biodiversity (Westhoek and others 2013). Loss of organic matter in UK soils is estimated to cost in excess of £500 million annually (Environment Agency, 2019). These simplified arable systems depend on fossil fuel agro-chemical inputs such as fertilisers and pesticides, which have negative environmental impacts on multiple scales, from the immediate soil environment to catchment scale water quality impacts to regional alterations in atmospheric nitrogen deposition to global shifts in greenhouse gas concentrations (Jenkinson 2001).

Ley crops

Much of the negative impact of continuous arable systems is due to the decrease in SOM. As SOM is maintained through the input of root biomass and exudates combined with aboveground plant residues, there is an inherent dilemma in that crops selected for maximum photosynthetic transfer to the harvestable crop will deliver the lowest return of organic residues to the soil. In addition, the de-coupling of arable from livestock production systems has also resulted in the reduction of organic fertiliser such as manure and slurry application to cropped land, thus further decreasing the total amount of organic matter inputs.

Traditionally, SOM levels were restored through the inclusion of forage and nutrient-building crops in the arable rotation. These ‘temporary grasslands’ are known as a ’ley’. A ley is a mixture of grasses, legumes, and/or other forbs grown from several months to several years before being replaced again by an annual crop. Their incorporation into arable rotations is receiving renewed attention as an approach to creating sustainable cropping systems (Jarvis and others 2017). Ley cropping usually implies the integration of ruminant livestock into the agricultural system in order to utilise the forage crop, however, stockless approaches are possible as well (Prade and others 2017). The integration of livestock, known as Integrated crop-livestock (ICL) systems or mixed farming, benefit both crop and livestock production while providing economic and biological diversification for the agro-ecosystem (Brewer and Gaudin 2020).

Both the ley crop and the return of livestock manures results in a net gain of SOC due to the quality and quantity of SOM entering the soil and the reduction of tillage (Paustian and others 2019b). The rate of gain in organic carbon, however, will vary depending on the initial levels in the soil, the total input of organic matter, the rate of decomposition, soil physical and chemical characteristics and duration of the ley (Johnston and others 2008; Bacq-Labreuil and others 2018). Many of the factors that determine the rate of change are under the control of the land manager, such as the species composition of the ley (ie grasses, legumes, and/or forbs) (Rutledge and others 2017; McNally and others 2015), and whether the ley pasture is grazed or cut (Nüsse and others 2018; Whitehead and others 2018; Skinner 2008).

Johnston and others (2017) show that for a sandy soil in south-east England this accumulation is equivalent to an annual increase rate of 9 per cent where the leys were grazed; 5 per cent with stockless leys and 3 per cent where farmyard manure was applied 1 in 5 years in the all crop rotation for the first 28 years after the change was implemented (table 3.13). Differences have been found between grazing and mowing management of leys with regards to SOC sequestration. Higher and more diverse carbon input under grazing compared to mowing appears to favour SOC sequestration (Gilmullina and others 2020).

The maximum annual rates of C sequestration for soil in ley-arable rotations can range from 0.26–0.36 t C ha^{-1} y^{-1} (Johnston and others 2017) to 1.1 t C ha^{-1} y^{-1} (Christensen and others 2009) depending on the duration of the grass ley in the arable rotation with half of the C accumulation potential being reached after around 6 years (range 4–11 years). Changes in organic carbon, however, depended on the species composition of the ley, with some crops, specifically lucerne and sainfoin, resulting in no increase in organic carbon.

Increases in SOC concentrations of 0.36 and 0.59 t C ha^{-1} y^{-1} were found in a ley dominated rotation compared with a cereal monoculture rotation after 35 years of management. Soil texture had a significant impact, with increases found in both topsoil and subsoil in a loam soil but only in the topsoil in a clay soil (Börjesson and others 2018). Lack of rooting into the subsoil in fine textured soils is likely reason for the lack of SOC increase, caused by high bulk-density, anoxic conditions or lack of soil phosphorus in the subsoil.

The integration of leys into all-arable rotations will support sustained/improved productivity, lead to increases in SOM over a 20 – 40 year period as the soil reaches its new equilibrium organic matter content (Knight and others 2019). Although it is possible to achieve a consistent increase in soil carbon by the introduction of leys, the rate of increase would eventually reach an equilibrium, and due to the impact of periodic cultivation, the SOM accumulation in ley-arable soils is less than in permanent grasslands (Recous and others 1997; Johnston and others 2008). Other factors that influence the rate of C sequestration include nitrogen inputs, soil cultivation practices and soil texture (Recous and others 1997; Johnston and others 2017). These gains are also reversible if there is a return to continuous arable production.

Table 3.13: Carbon sequestration in leys, as reported in Johnston and others (2017)

Arable practice	Change in rate of accumulation of carbon, as compared to the baseline scenario	Reference
3-year grazed ley + 2-year arable	+9 per cent	Johnston and others 2017
3-year lucerne ley followed by 2-year arable	+3 per cent	Johnston and others 2017
8-year grass/clover ley + 2-year arable	+7.2 per cent	Johnston and others 2017

Cover crops
The inclusion of cover crops, such as brassicas or legumes, within arable rotations can also help to improve SOC stocks. Their design, management and net impact on SOC follow the same principles as ley crops, however due to shortened growing season their contribution to SOM is typically much less. According to their purpose in the rotation, they can also be referred to as catch crops or green manure crops. They can be established sequentially with arable crops or overlapping as either an under-sown or nurse crop. Regardless of the terminology, there are multiple benefits of cover crops including the reduction of nutrient loss, improved weed management, protection from erosion and provision for pollinators and farmland birds. Some consideration needs to be paid to ensure that they don’t act as a bridge for pests and disease.

Cover cropping works by increasing net primary productivity (NPP) and the quantity plant residues available for integration into SOM. The magnitude of soil carbon stock change is, however, dependent on local environmental conditions and ongoing management practices. In a global meta-analysis, McClelland and others (2020) found that the inclusion of cover crops increased soil carbon stocks by 12 per cent, averaging 1.11 t C ha^{-1} compared to a no cover crop control. The factors that determined SOC response were duration of cover crop growth, total biomass production and soil clay content. In other studies, Poeplau and Don (2015) reported an annual change rate of 0.32 ± 0.08 t ha^{-1} y^{-1} (1.17 t CO_2e ha^{-1} y^{-1}) at a mean soil depth of 22 cm during the observed period of up to 54 years where cover crops were incorporated as a green manure and compared to bare fallow.

Cover crops have been reported to increase soil carbon not only in the topsoil, but also within the subsoil with gains of between 0.09 and 0.32 t C ha^{-1} y^{-1} compared with bare fallow (Olson and others 2014). These results, however, are from spring and summer cropped arable systems in the US with a bare winter fallow. In the UK, where winter cereals predominate, the gains in soil carbon would likely be considerably less. If spring cropping becomes more prevalent in the UK, the dynamics of carbon gains/losses due to cover cropping will need further investigation. As with other approaches to soil carbon sequestration, gains can be reversed due to subsequent alterations to land management.

3.4.3 Carbon storage and sequestration in intensive grassland

Intensive grassland has a high potential to sequester carbon due to its extensive and diverse root system and high turnover of aboveground growth. Soil carbon accumulation, particularly in the first years after land use change is time-limited and needs to be considered in any analysis involving carbon balance (Godde and others 2020). This is covered in more detail in the semi-natural grassland chapter within the ‘sink equilibrium in grasslands’ section.

Though modified via nutrient inputs, reseeding, and grazing or cutting, intensive grasslands tend to have higher carbon stocks than arable systems, though lower than their semi-natural grassland counterparts (Bradley and others 2005). However, there is also a great deal of variability due to different management practices (sward composition, fertiliser inputs, grazing management, frequency of renovation) and soil types (den Pol and others

2018). The CS2007 survey report that improved grassland soils in England hold 64.6 t C ha^{-1} to a 15 cm depth (Emmett and others 2010). Moxley and others (2014), using the work of Bradley and others (2003, 2005) to underpin reference SOC stocks in the UK's LULUCF inventory, cite at equilibrium carbon stocks under pasture to be 130 t C ha^{-1} (1 m depth). Ward and others (2016) use management intensity to categorise total carbon stocks, finding in their study that, accounting for bulk density grasslands under intensive management had the lowest total carbon stock at 403 t C ha^{-1} (to 1 m depth, note this figure includes both inorganic and organic forms of carbon). Cantarello and others (2011), using data obtained via a systematic literature review, group pasture with natural grasslands and suggest a C stock range between 72–204 t C ha^{-1}, with a mean of 121 t C ha^{-1}.

The relationship between ecosystem net primary productivity and soil carbon stocks is complex. The use of fertilisation inputs in more intensive grassland systems can enhance carbon storage due to greater plant productivity, residue returns and root inputs to soil (Moxley and others 2014). However, there is a trade-off between the amounts of biomass produced and its decomposition, with slower growing plants in nutrient-poor environments producing litter with a high C:N ratio, thus slowing the decomposition rate. Much of the extra productivity will also be removed as silage or animal biomass. Likewise, improved forage cultivars with higher yield, produce leaf litter that is higher in simple carbohydrates which may increase its rate of decomposition by soil organisms resulting in greater loss as CO_2 compared with more recalcitrant tissues (Humphreys and O'Donovan 2014).

Effect of fertilisers

Productive, modified grasslands are usually maintained with high fertiliser inputs of nitrogen, phosphorus, potassium, sulphur and lime, or with the regular application of manures and slurries. Nitrogen is applied to grassland in both organic (ie farmyard manure, slurry etc.) and inorganic/synthetic forms or by biological nitrogen fixation through legumes, which are maintained as a component of the grassland. Fertiliser use contributes to soil carbon sequestration by increasing biomass production and by improving the carbon to nitrogen (C:N) ratios of residues returned to the field. Both mineral nitrogen additions and nitrogen from biological fixation have been found to increase SOC storage, however, the impact can be limited if other nutrients are a limiting factor (Macdonald and others 2018). Soil phosphorus is an important factor in nitrogen-fixing systems, where phosphorus demand is high and it also plays a role in the stabilisation of SOM due to the stoichiometry of plants and microorganisms that determine photosynthetic and decomposition processes (Hartman and Richardson, 2013).

Nutrient cycling within grasslands can also impact SOC stocks. The addition of nutrients in the form of manures or slurries can favour SOM sequestration compared with synthetic forms of nitrogen due to the more diverse carbon inputs and more efficient microbial functioning (Gilmullina and others 2020) and may deliver wider ecosystem service benefits overall (Ward and others 2016). Increasing nutrient inputs to grassland, however, may lead to trade-offs with GHG emissions such as N_2O and carbon emissions due to the manufacture of synthetic nitrogen (Hijbeek and others 2019). There are also trade-offs with biodiversity, both in plants and invertebrates, that need to be considered when determining grassland management outcomes. In a study of agri-environment measures in Switzerland, extensive management, especially in pastures, favours all ecosystem services with the exception of forage provision (Le Clec'h and others 2019). Intensive management may favour carbon sequestration and fodder provision, but other ecosystems services may be reduced. The challenge is to design forage production systems based on species combinations that provide stable, productive and resource efficient plant communities that can also enhance ecosystem services.

Plant diversity effects
Improved grasslands have a greatly reduced level of plant diversity compared to their semi-natural grassland counterparts, often consisting of one to several sown species and a few naturalised species or weeds. Nonetheless, the presence of additional species in the sward has been demonstrated to influence soil carbon sequestration. Grass species, in general, have a high root density which is beneficial for both SOC accumulation and for soil aggregation, which aids in the protection of accumulated SOM. The addition of clover into swards has been found beneficial to SOC. This has been attributed to the differences in the C:N ratio of the root biomass (De Deyn and others 2010). The addition of other species, particularly forage herbs with deep roots such as chicory *Chicorium intybus* L. and plantain *Plantago lanceolata* L. have also been shown to increase SOC, particularly at depth (McNally and others 2015). In addition to the impact on carbon sequestration, plant secondary metabolites from diverse pastures have been shown to influence soil processes, both directly and through manure deposition, potentially mitigating nitrogen loss by slowing N mineralisation (Clemensen and others 2020). These effects also influence the fate of nitrogen compounds within the ruminant, shifting excretion from urine to faecal pathways, thus stabilising N in organic compounds resulting in fewer losses to the environment through leaching and de-nitrification (Distel and others 2020). There is strong evidence that replacing single-species forage diets with diverse mixtures will provide multiple benefits for both ruminant health and environmental sustainability.

Effect of grazing
Grazed grassland sequesters more carbon than mown grassland due to the greater return of organic matter and nutrients. In addition, grazing alters the soil microbial community which enhances the availability of substrate which favours SOC sequestration (Gilmullina and others., 2020). Grazing management strategies will affect the exchange of greenhouse gases between the soil and the atmosphere but most research focuses on CO_2, whereas measurements of N_2O and CH_4 flux are scarce. In Central Europe, grazed grassland was a net sink of CO_2 of between 0.24 to 4.9 t C ha^{-1} y^{-1} but only 21 per cent of this was offset by N_2O and CH_4 emissions (Hörtnagl and others 2018).

Frequency of renovation
Improved grassland which becomes degraded through the loss of sown species or through damage to soil may be renovated. This is accomplished through varying degrees of soil disturbance, from complete destruction of the existing sward with tillage to less disruptive approaches using over-sowing with minimal disturbance to the soil. As the degree of soil disturbance increases, so will soil organic C and N emissions to the environment also increase. Ideally, swards should be retained as long as possible or left intact by ensuring that good management practices (maintenance of fertility and avoidance of overgrazing, poaching and compaction) are adopted (Kayser and others 2018).

3.4.4 Carbon storage and sequestration in peatlands under agricultural use

Over the last 400 years, intensive agriculture has had a considerable impact on lowland deep peat soils, draining and modifying them from diverse natural habitats to arable and horticultural cropping and species-poor intensive grassland. In this report we have made the distinction between intensively managed peatlands modified for agricultural production and those under extensive management or conservation management (see Chapter 4).

In England, 240,000 ha of lowland peat is farmed, around 183,000 ha of which is managed as cropland (ONS 2019). Due to its contribution to food production this land is considered of high agricultural value. For instance, the Fenland in the east of England where the deep peats account for approximately 10 per cent of area nationally producing potatoes, sugar beet and vegetables (Graves and Morris 2013). Other large peatland areas drained for agriculture include the Humberhead Levels, Somerset Levels and Lancashire Mosslands. However, their value is based on agricultural productivity and not the other important ecosystem services that healthy, functioning peatlands underpin. When considering the combined agricultural and environmental effects of continuing agricultural production on fen peats Morris and others (2010) estimate a net annual cost of between -£200 and -£500 ha^{-1} y^{-1}.

Currently, the agricultural value of these peatlands depends on their continued artificial drainage and modification, practices that cause severe degradation of the peat deposits beneath the crops (photo 3.5). The carbon loss under arable management can be especially rapid due to oxidation caused by lowered water tables, extended periods of bare ground exposure and regular disturbance via ploughing and other cultivations. Arable farming on peats is estimated to lead to annual peat wastage of 10–30 mm, and under a 'business as usual' approach, the majority of the remaining peats will become wasted over the next 30 to 100 years, depending on their current peat depth and management (Graves and Morris 2013). Evans and others (2016) suggest at current rates of loss some arable peats may lose all their organic matter over the next 200 years, and if loss via wind erosion is accounted for this could be within a century.

Photo 3.5 Drainage channels associated with cereal production on peat © Natural England / Peter Wakely

Agricultural peatlands are one of the largest emitters of greenhouse gases in the land use sector. Arable and horticultural cropland covers 24 per cent of the England's peat area, but is the source of 64 per cent of peatland GHG emissions (though there is some uncertainty regarding emissions from wasted peats). Intensive grassland occupies a further 9.6 per cent of the England's peat area and emit 20 per cent of total England's peat emissions (Evans and others 2017). In their study of lowland peatland systems in England and Wales, Evans and others (2016) report all study sites under agricultural management were net carbon emitters during the study (table 3.14) with emissions increasing with land use intensity. Carbon stocks reported to 50 cm depth were highly variable, in part due to the mix of organic and mineral material in cores at some sites. Peatlands used for food production are also subject to regular application of fertiliser, causing nitrous oxide emissions to contribute an additional 20–50 per cent to total GHG emissions of the intensively managed sites reported in table 3.14, while at unfertilised peatland sites N_2O emissions are considered to be negligible (Evans and others 2016). The legacy impact of fertiliser applications on GHG emissions from sites post-restoration is unknown.

The site-based values reported by Evans and others (2016) (table 3.14) are slightly lower than those reported in the 2021 update to the for UK emissions inventory for UK Peatlands – expected to be published in April 2021 in the 2021 UK GHG Inventory[20,21] (see section 4.2.2 for more information). The inventory update provides emission factors of 13.03 t CO_2e ha^{-1} y^{-1} for extensive grassland (combined bog and fen); 27.54 t CO_2e ha^{-1} y^{-1} for intensive grassland; and 38.98 t CO_2e ha^{-1} y^{-1} for arable cropland. Note these figures include the GWP of direct N_2O emissions.

Table 3.14 Summary of carbon stocks and flux on English peatlands under agricultural use, as reported by Evans and others (2016)

Land use	Carbon Stock (t C ha^{-1})	Reference
Extensive grassland	610 - 1650	Evans and others 2016 (to 50cm depth)
Intensive grassland	1980	
Arable	1290 – 3880	
Carbon flux / balance (t CO_2e ha^{-1} y^{-1})		
Extensive grassland	5.59 - 12.36	Evans and others 2016 (inc CH_4 and CO_2)
Intensive grassland	16.71	
Arable	23.38 - 28.45	

[20] https://unfccc.int/ghg-inventories-annex-i-parties/2021
[21] NAEI website: https://naei.beis.gov.uk/reports/

As shown by Evans and others (2016) the main control on carbon emissions from lowland peat sites is water levels. A 10 cm lowering in the mean water table will increase CO_2 emissions by 3.7 t CO_2 ha^{-1} y^{-1}. Methane displays the reverse relationship, at water table depths greater than 25 cm CH_4 fluxes are consistently near zero, but with each 1 cm rise above this they increase by 0.21 t CO_2e ha^{-1} y^{-1}. Raising water levels to within around 5 cm below the surface could balance GHG emissions from lowland peats to near zero values (Evans and others 2016) and would therefore deliver considerable benefits for climate change mitigation.

The carbon benefits of restoring these peats-forming habitats on this land is covered in Chapter 4 which focuses on semi-natural peatlands, their management and restoration. While agricultural peats represent a very large carbon source and protecting its remaining carbon stock should be a land use priority, given its likely continued importance for food production it is unlikely that the entire extent will be returned to peat-forming habitats. Acknowledging this, attention has turned to more sustainable management approaches, such as farming with raised water levels and paludiculture, which offers potential to reduce GHG emissions both by rewetting peat soils and replacing fossil fuels with renewable biomass alternatives (Wichtmann, Schroeder & Joosten, eds 2016).

While still at the experimental stage with further development necessary, raising water levels to control GHG emissions whilst continuing with production has been investigated in England (Mulholland and others 2020). Wen and others (2019) demonstrated that raising water levels outside of the cropping season decreased CO_2 loss from cultivated peat soils, while the use of cover crops lowered the potential of N_2O emissions associated with wet, fertilised soils. Higher water tables may not necessarily impact food production should the right crops be targeted for this approach. Musarika and others (2017) raised water tables under radish crops from a depth of 50 cm to 30 cm, reducing CO_2 emissions whilst increasing the crop yield. Taft and others (2018) in their study of a range of GHG mitigation approaches on horticultural peat report that raising the water level to surface level was the only approach that consistently reduced GHG emissions, and approaches were most effective if applied during the growing season. The authors note the difficulties for land managers in co-locating mitigation approaches with ongoing management activities.

The studies cited are based on core and mesocosm experiments with further investigations at scale and under field conditions an evidence need. Paludiculture approaches also extend beyond food production. The propagation of *Sphagnum*, as a growing medium to replace peat or for donor material for restoration projects, and the cultivation of biomass for bioenergy are possible alternative land uses to conventional cropping reliant on artificial drainage on peat soils that may deliver climate change mitigation benefits (Mulholland and others 2020).

3.4.5 Climate change and other interactions

Scenarios to achieve the UK's greenhouse gas target of net zero by 2050 require a significant area of land use change from high carbon to low carbon practices. In their in-depth advice on UK agricultural and land use policies the Committee on Climate Change (CCC) (2020) state one fifth of land will need to be released from traditional agricultural production and subject to management that benefits long-term carbon sequestration. In some cases, this will be a move to the creation or restoration of habitats, such as peatland and woodland, which continued agricultural use will struggle to match in terms of carbon sequestration potential. However, the use of agroforestry, biofuels and buffer approaches such as hedgerows could also support continued productive use of agricultural land whilst also delivering benefits for climate change mitigation. Agricultural policy now reflects a 'public money for public good' approach (Defra 2019) where, post-Common Agricultural Policy, the aim is to reward land managers for delivery of wider environmental benefits, including climate change mitigation.

The agricultural sector is also a significant emitter of greenhouse gases, 10 per cent of national emissions in 2019 (BEIS 2021). Agricultural land uses differ significantly from other habitats covered in this report in that emissions are dominated by nitrous oxide and methane. As covered previously in this chapter, management to enhance carbon may have trade-offs with these other gases, and these wider interactions within agricultural systems should be identified when capturing their potential to mitigate climate change. Where the potential to sequester carbon on agricultural land is low the reduction of emissions of methane and nitrous oxide, and retention of nitrogen, should be prioritised (van Groenigen and others 2017). A high uptake of low carbon farming practices that reduce greenhouse gas emissions from soils, ruminant livestock and manure management and a use of low carbon fuels in buildings and farm operations will be needed to achieve net zero, and could deliver up to 10 Mt CO_2e emissions savings by 2050 (CCC 2020).

3.4.6 Evidence gaps and future needs

In soils with particularly high background levels of SOC, it is difficult to assess short (1–5 years) to medium-term (> 5–10 years) changes. It is therefore necessary to monitor changes in SOC due to different management practices over periods greater than 10 years (Kätterer and others 2012). This clearly demonstrates the importance of both long-term experiments and long-term monitoring networks. Long-term field experiments have also demonstrated that changes in SOC stocks are most rapid immediately after a modification in management but the rate of changes levels out with time as it reaches a new equilibrium value (Powlson and others 2012).

In order to determine the impact of climate change policy decisions for agricultural land, accurate estimates of SOC stock changes are required. The United Nations Framework Convention on Climate Change (UNFCCC) requires that C sources and sinks in managed lands are reported, and that the data be used to monitor emission trends and GHG mitigation efforts (UNFCCC, 1998). Wesemael and others (2011) examined the planned soil monitoring efforts of 10 countries and concluded that most are inadequate to report SOC changes over a period of 5–10 years. A complete review of the methodology used by various countries to estimate changes in soil C stocks for agricultural land use on mineral soils is available in Smith and others (2019).

Changes in SOC stocks due to agricultural management practices are measured primarily in the arable layer (0–30cm) but the impact of management on subsoil C is a major evidence gap (Lorenz & Lal 2005). It is estimated that as much as 50 per cent of SOC in farming systems is located at depths greater than 30 cm (Jobbágy & Jackson 2000). Ward and others (2016) determined that approximately 60 per cent of grassland total carbon was below 30 cm depth and that this carbon was sensitive to management with the highest levels under intermediate rather than intensive management. As changes to soil carbon content can affect the bulk density of the soil, it is therefore necessary to use an 'equivalent mass basis' approach when comparing SOC stocks across land uses and different management regimes (Smith and others 2019). Deeper soil sampling for monitoring purposes to 100 cm is recommended by the Food and Agriculture Organization, but this can be costly and requires specific machinery (Amanullah 2019).

Although biomass removal occurs to some extent in natural habitats, it is a characteristic feature of many agricultural systems. In arable systems, biomass removal occurs through the removal of the primary crop, usually the seed biomass in the case of grains and oilseeds, but to an even larger degree with the removal of straw and other remaining plant materials. In grassland systems, biomass removal as conserved forage is the main mechanism of carbon loss from the system but can be largely replaced if manures and slurries are returned to the field. Grazed grasslands have a much lower loss of carbon as much of it is immediately recycled through manure deposition. Because of the impact that this has on net carbon flux, it is necessary to account for both biomass removals and additions in order to model the impact on SOC stocks and overall GHG balance of the agricultural system.

4 Blanket bogs, raised bogs and fens

4.1 Chapter summary and key messages

Peatlands, defined here as blanket bogs, raised bogs and fens, represent England's largest terrestrial carbon stores. Healthy, functioning peatlands have a net cooling effect on the climate, locking up carbon and playing an important role in climate regulation. Such peatlands are also vitally important in the provision of other ecosystem services such as biodiversity conservation, water regulation and cultural heritage. However, England's peatlands have been severely degraded by management interventions such as drainage, burning and agricultural use, and now represent a net source of carbon and a warming effect. Their restoration is now recognised as a priority for reducing greenhouse gas emissions from land use in the UK. Reviews on forestry on peatlands are in 'Chapter 2 - Woodland, trees and scrub', agricultural land use on peats in 'Chapter 3 - Open habitats and farmland', and peatland streams in 'Chapter 5 - Rivers, lakes and wetland habitats'.

The key messages are:

- England's peatlands have been severely degraded by management interventions such as drainage, burning and agricultural use, and now represent a net source of carbon and have a warming effect on the climate. Their restoration is now recognised as a priority for reducing greenhouse gas emissions from land use in the UK.
- Peatlands represent the largest terrestrial carbon store by habitat. Due to their extent, blanket bogs hold the greatest stock, but the deep peats in raised bogs store the most carbon on a density by area basis.
- Fens are extremely diverse, and this is shown in the wide variability in their depths. The deep peats and associated carbon stocks reported in this review demonstrate their capacity to store large quantities of carbon. However, these sites may not be representative of fen habitats as a whole, which may typically have shallower depths.
- Undrained, semi-natural fen habitats are net carbon sinks. Deep-drained fen and lowland bog habitats converted to arable agriculture are the largest net carbon emitters. Intensive and extensive fen grasslands on drained fen peats and bog grasslands are also net carbon sources to the atmosphere.
- There is increasing consensus in the scientific evidence that burning on blanket bog is damaging and having a detrimental impact on carbon stocks and sequestration rates. Raising the water level and restoring peat forming vegetation is a more effective way of managing these sites to reduce fire risk (Granath and others 2016), as well as delivering significant benefits for climate change mitigation, biodiversity and water quality.
- Consideration of vegetation establishment and associated water levels is an important factor when restoring all peatland habitats. For example, *Sphagnum*

mosses are a critical component in restoring function and maintenance of the carbon store in bogs and acidic fens and require a permanently high-water table to survive. Other species favoured by high-water tables, however, such as cotton-grass species *Eriophorum* spp. and other graminoids, can be associated with higher methane fluxes.

- Raising waters in drained peatlands has mixed effects on carbon flux depending on habitat. Raising water tables in raised bogs can have an immediate effect on carbon fluxes. Blanket bogs are less responsive to drainage and rewetting alone, but it can be beneficial when coupled with peatland stabilisation and re-establishment of vegetation cover.

- Overall, our confidence assessment is low–medium for carbon stocks and medium for carbon flux, due to the inclusion of peatlands in the UK GHG Inventory. While some full-scale assessments have been carried out, large spatial variability has been shown and studies have often been carried out at the same sites or regions. Further work to quantify the carbon stocks held in vegetation is required across all peatland habitats.

- Tables 4.1 and 4.2 summarise the carbon storage and flux values identified as representative for habitats reported in this chapter.

Table 4.1 Summary of carbon storage in semi-natural peatland habitats, as based on the review of literature. * indicates the report authors have taken the midpoint of the range as representative. # indicates where total stocks are reported without vegetation data.

Habitat Description	Soil Carbon ($t\ C\ ha^{-1}$)	Soil Depth (cm)	Vegetation Carbon ($t\ C\ ha^{-1}$)	Soil + Veg. Carbon ($t\ C\ ha^{-1}$)	Confidence [High, Medium, Low]	References
Blanket Bog	525* Range [354 to 619 [a]] 259[b]	Full peat profile [a] 50cm [b]	See Chapter	525[a,#]	**Medium**	[a] Heinemeyer and others 2020 (catchment scale estimates) [b] Ostle and others (2009) (0.5m depth)
Raised Bog	1610[a] Range [810 to 2530[b]]	Ave depth for Habitat (100cm to 380 cm)	See Chapter	1610[a, #]	**Low**	[a]Natural England, 2010 [b]Evans and others 2016
Fens (on Deep Peat)	1971* Range [610 to 2,820]	40–380 cm	No data	1971[#]	**Low**	Evans and others 2016

Table 4.2 Emission factors (EF) (t CO_2e ha^{-1} y^{-1}) for peat condition types taken from the 2021 update to the Emissions Inventory for UK Peatlands. Note fluxes have been adjusted to represent carbon only fluxes, and the influence of nitrous oxide removed, to be consistent with other habitats in this report. The full peatland GHG table is reported in section 4.2.2.

Habitat Description	Carbon Flux			References
	Annual Carbon Gain / loss for the habitat			
	t CO_2e ha^{-1} Y^{-1}	Range (if possible)	Confidence [High, Medium, Low]	
Near Natural Fen (undrained)	-0.93	-	Medium	All emission factors taken from the 2021 update to the Emissions Inventory for UK Peatlands – to be published in April 2021 in the 2021 UK GHG Inventory[22,23]
Near Natural Bog (undrained)	-0.02	-	Medium	
Rewetted Bog	3.87	-	Medium	
Rewetted Fen	8.05	-	Medium	
Rewetted Modified (Semi-natural) Bog	-0.02	-	Medium	
Modified Bog (semi-natural Heather + Grass dominated - Drained	3.48	-	Medium	
Modified Bog (semi-natural Heather + Grass dominated - Undrained	2.25	-	Medium	
Eroding Modified Bog (bare peat) - Drained	13.14	-	Medium	
Eroding Modified Bog (bare peat) - Undrained	12.03	-	Medium	
Extracted Domestic (drained)	13.23	-	Medium	
Extracted Industrial (drained)	13.14	-	Medium	
Cropland	32.89	-	Medium	
Intensive Grassland	24.87	-	Medium	
Extensive Grassland (combined bog/fen)	11.02	-	Medium	

[22] https://unfccc.int/ghg-inventories-annex-i-parties/2021
[23] NAEI website: https://naei.beis.gov.uk/reports/

4.2 Peatlands

Peatlands are wetland ecosystems, where waterlogging has prevented the decomposition of plant matter leading to a net accumulation of organic matter known as peat. Carbon is stored in the dead plant remains which build up as a thick organic layer over thousands of years, making peatlands a vitally important carbon store. Peatlands develop in diverse environments, including those experiencing high precipitation and low evapotranspiration, topographical basins, and those where inflows of water from groundwaters or fluvial systems exceed the rate at which they can leave. At an ecosystem level a general distinction is made between peatlands whose surfaces are fed by precipitation or meteoric water which are known as bogs, and fens, whose surface vegetation is fed by minerotrophic water from groundwater or surface water, as well as precipitation (McBride and others 2011; Lindsay 2018). In the UK, mosses, mainly *Sphagnum* species, are the main contributors to peat formation in bogs, while sedges and other graminoids, brown mosses, and in some cases wood, make up a higher proportion of fen peats.

Peatlands make up an estimated 11 per cent of England's land area. Deep peats (defined as peat soils with a depth greater than 40 cm, though 30 cm is often used as an ecological definition) cover 495,829 ha, with an additional 186,372 ha considered 'wasted' – heavily degraded by drainage and cultivation for agriculture (Evans and others 2017). Less than 1.3 per cent of England's peatlands remain in a near-natural state with the rest having been affected by peat extraction, grazing, fire and drainage for agriculture and forestry planting (Evans and others 2017). Being dependent on high-water tables and vegetation cover to function properly, peatlands that are drained or have bare peat exposed, continue to dry out, deteriorate and erode unless remedial action is taken to rewet them.

Peatlands are vitally important in the provision of other ecosystem services such biodiversity conservation, water regulation and cultural heritage (Bonn and others 2016). The management induced degradation and loss of the ecosystem function of peatlands has led to the reduction of these wider benefits, placed additional costs on society, as well as them becoming a major source of greenhouse gas emissions in the UK (IUCN 2018: Evans and others 2017). The changing climate exacerbates these problems, placing more stress on these less resilient, managed peatlands (Natural England & RSPB 2020), which may lead to more rapid loss of carbon (Worrall and others 2010). In total, the UK's peatlands GHG emissions are estimated at 23.1 Mt CO_2e y^{-1}, 11.1 million of which is from England (Evans and others 2017), though there is uncertainty relating to extent and emissions from wasted peats (C. Evans, pers. comms). Peatlands have been represented within biodiversity strategies over the last 3 decades, but their restoration is now a priority in climate change mitigation scenarios as an effective approach to reduce emissions (CCC 2020). Over 22 per cent of England's peatlands are under restoration management, particularly blanket and raised bogs, with evidence that this is reducing greenhouse gas emissions and protecting carbon stocks (Trenbirth & Dutton 2019).

4.2.1 Carbon cycling in peatlands

Peatlands are distinct from other ecosystems in their capacity for long-term carbon accumulation, which under the right conditions has the potential to continue for millennia. Their longevity and deep soils mean peatlands hold the largest carbon stocks of England's habitats (Bain and others 2011). Peat accumulates slowly, around 0.3–2 mm of peat each year due to low nutrient status, anaerobic conditions, the resistant nature of *Sphagnum* tissues to decomposition and the presence of *Sphagnum* vegetation cover slowing further decomposition (Lindsay and others 2014). The rate of peat accumulation varies considerably between different peatland habitats; peat accumulation in blanket bogs can be half that of raised bog due to warmer climatic conditions (Lindsay 2010). Furthermore, hydrological condition, vegetation, chemistry and nutrient status all influence peat accumulation and therefore carbon storage and sequestration, with fen habitats in particular characterised by high plant diversity and high peat decomposition rates (Loisel and Bunsen 2020).

Picture 4.1 Sphagnum and other mosses on blanket bog ©Natural England/Ruth Gregg

As wetland habitats, the hydrology of peatland and fen habitats exerts a considerable influence on peatland greenhouse gas emissions. The depth of the water table from the peatland surface determines the boundary of oxic - anoxic conditions and the redox level within the peat soils, which in turn controls the balance of emissions of CO_2 and CH_4. Utilisation of peatlands for productive purposes has typically involved lowering the water table, enhancing decomposition processes and allowing the peat body to be oxidised, and releasing CO_2 to the atmosphere as a result (Moomaw and others 2018). Conversely, peat accumulation depends on a high-water table close to the surface, which in turn creates the anaerobic conditions required for methanogenesis (the microbial formation of CH_4) and elevated emissions of CH_4 (Abdalla and others 2016). While increased emissions of CH_4 will somewhat offset the carbon gains from peat accumulation, raised water levels generally result in a lower emissions overall than drained sites as the increased sequestration of CO_2 outweighs loss via CH_4 (Zhong and others 2020). Günther and others (2020) demonstrate that due to the relatively short atmospheric lifetime of CH_4, compared to CO_2, delaying rewetting due to concerns of CH_4 emissions acts to increase the long-term warming effect of drained bogs through continued CO_2 emissions.

The sink-source status of a peatland is dependent on the balance between biomass production and decomposition (Joosten and others 2016). Peatland vegetation reflects the long-term water levels and has specialised traits which tolerate and support the continued development of low nutrient, waterlogged conditions (De Deyn and others 2008). The vegetation component of the peatland ecosystem contributes to regulation of GHG exchange and carbon cycling via organic inputs in the form of litter and root exudates, influencing water levels through transpiration, and by providing a bypass for CH_4 formed at depth via aerenchyma (air channel within the stem and leaves) in plants such as cotton grass *Eriophorum spp* (Couwenburg and others 2011). While the presence of peat is indication of the importance of peatlands as long-term carbon sinks, their status can vary from year to year. Modification of peatlands via human activity has disrupted these self-regulatory functions, and damage such as draining, burning, over-grazing, cutting as well as their use for agriculture, peat extraction and plantation forestry (see section 2.2.5) can shift peatlands to being a net greenhouse gas source (IUCN, 2018).

4.2.2 Carbon storage and sequestration in peatlands

In this section we begin by covering an overview of the evidence relating to carbon cycling in peatland ecosystems, before reviewing the literature relevant to specific peatland habitats (blanket bog, raised bog and fens) in the sections that follow. The peatland carbon cycle is illustrated in figure 4.1.

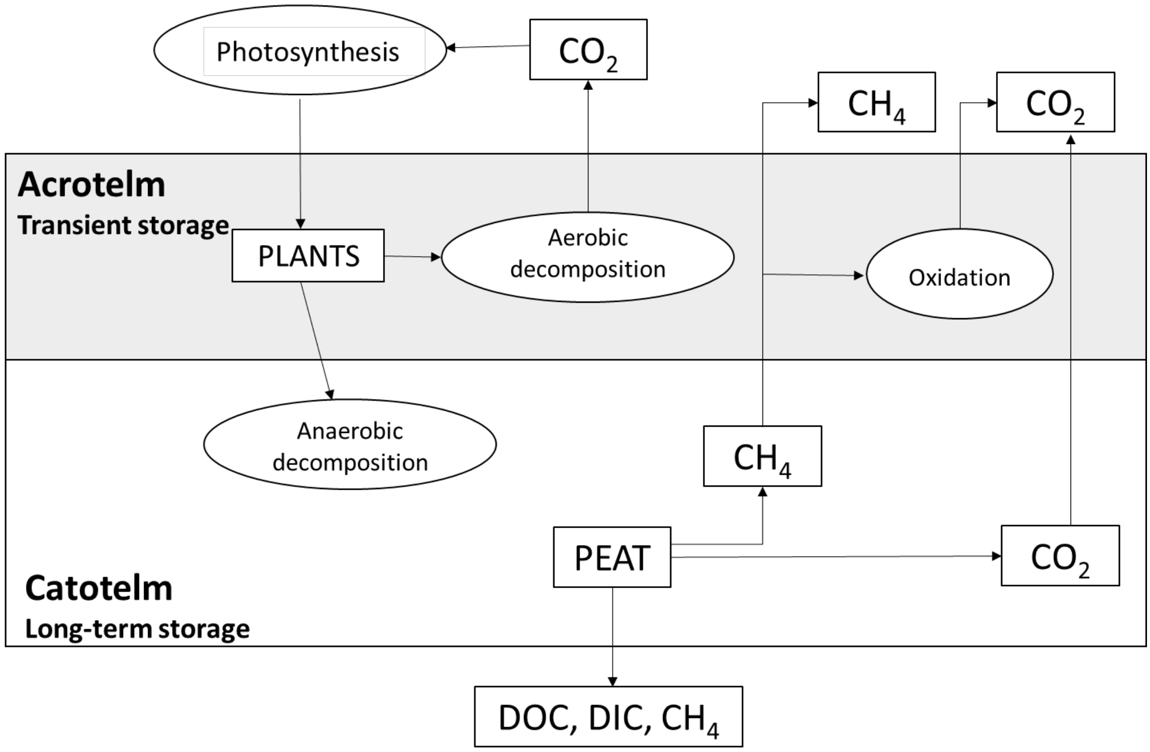


Figure 4.1 The peatland carbon cycle (adapted from McLaughlin 2004). 'Transient storage' refers to the acrotelm being an active zone where fresh plant material is added and most decay takes place. As the peat body develops this is transferred to deeper depths where 'long-term storage' occurs, typically below the water table

Carbon stocks

Estimates of peat carbon stocks in the UK and England are highly uncertain due to the considerable variation in the depth of peat soils. Peat depth is not uniform and varies over short distances due to the underlying topography (Parry and others 2014a). Under blanket bog, peat thickness is typically 0.4–6 m; it can be up to ten metres and often more in raised bogs, and in fens is 0.4–5 m. However, average depth estimates are typically very generalised, and the UK lacks a systematic survey approach to determine the extent and depth of peats (Lindsay 2010) that is needed to generate an accurate estimate of national peat stocks. The carbon density is typically greater in the deeper anaerobic catotelm layer than in the acrotelm located at the peat-vegetation interface (figure 4.2).

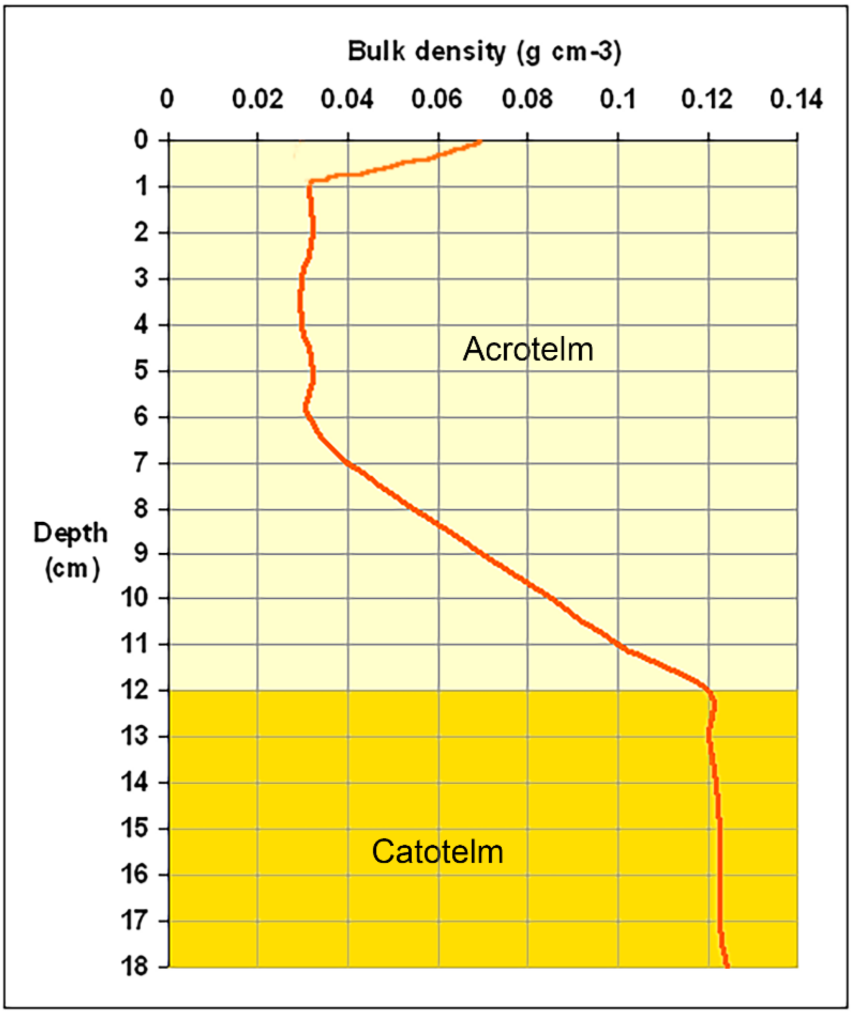


Figure 4.2 Dry bulk density values from the bog surface to a depth of 18 cm (used with permission from Lindsay, 2010, who adapted from Clymo, 1992).

Despite the uncertainty regarding depth, attempts have been made to estimate peat carbon stocks in UK and English peatlands with values and geographic focus varying greatly across studies. Smith and others (2007a, 2007b) taking a modelling approach estimated UK carbon stocks of 5.1 billion tonnes, with the vast majority (4.5 billion tonnes), situated in Scottish peat soils. The Countryside Survey 2007 reported across broad habitat types, sampling the top 15cm of soil and estimated that 'fen, marsh and swamp' and 'bog' hold stocks of 76 t C ha^{-1} and 74 t C ha^{-1} respectively (Emmett and others 2008). In Great Britain this equates to store 259 ± 8 t C ha^{-1} when extrapolated to 50 cm depth, and a peatland carbon store of 550 Mt C (Ostle and others 2009). Natural England (2010), using data gathered from literature, soil survey data and field measurements, estimate that England's deep and shallow peatland soils store 584 Mt C. However, the authors highlight that estimates are constrained by limited peat depth data. Worrall and others (2010) state that based on current data it is impossible to give a definitive estimate of the amount of carbon stored in UK peatlands but using evidenced assumptions report a UK peatland carbon stock of 3200 ± 300 Mt C. Field and others (2020) estimate 102 Mt C is held in high conservation value bogs and fens in the UK, though this is recognised to be conservative as their figures do not capture stocks stored below 30 cm or situated outside semi-natural peatland habitats. While these estimated stock values may vary, there is a wide consensus that peatland habitats represent our most significant belowground carbon stocks. Ongoing work in Natural England to update the England Peat Map will contribute to increased accuracy in depth estimates in the near future.

Often overlooked in peatlands is the carbon stored in the vegetation layer as it is estimated to be a much smaller carbon store than within the peat below. But when wanting to make a comparison to other habitats, as in this report, it is a key consideration. Both Ostle and others (2009) and Field and others (2020) use the figure of 2 t C ha originally reported by Milne and Brown (1997). However, it is unclear if this value is referring to aboveground biomass alone, or that also found at depth as part of the acrotelm. Carbon stocks held in vascular plant cover across blanket bog at Moor House, North Pennines, is estimated to be in the region of 3 t C ha (Garnett and others 2001; Ward and others 2007; Lindsay 2010).

During this review authors did not identify any field-based studies that quantify carbon held in the living *Sphagnum* layer. *Sphagnum* mosses are vitally important in their role as peat formers due to their slow decomposition rates and ability to grow indefinitely (Fenner and others 2007). Lindsay (2010) attempts to further understand the contribution of the *Sphagnum* acrotelm, the 'active' zone where fresh plant material is added and most decay takes place. Using a standardised estimate of bulk density and carbon content it is anticipated that an acrotelm of 15 cm depth may hold stocks of 45.5 t C ha^{-1}. The carbon stocks reported from vascular peatland vegetation and the high estimates made by Lindsay (2010) suggest that the 2 t C ha^{-1} suggested by Milne and Brown (1997) may be a considerable underestimate, and the living layer of sphagnum may hold substantial amounts of carbon. More work on carbon stored in peatland vegetation is required to increase the accuracy of stock estimates for peatland ecosystems.

Carbon fluxes

Carbon budgets of peatlands have been estimated through two methods; the dating of historical peat accumulation via peat cores or by the direct measurement of carbon fluxes from peat (Ratcliffe and others 2018). Accumulation studies have a long temporal reach, but only capture the carbon that is accumulated and then retained in the peat. They cannot distinguish between short periods when the peatland may switch between a sink and source, such as during fire or drought events, and may be unrepresentative of current conditions. Flux measurements distinguish and quantify the carbon pathways from a peatland in the present and are required when assessing the net ecosystem carbon budget (NECB) of a site. However, many approaches overlook some aspects of the carbon budget, such as CH_4 or fluvial pathways.

Several attempts have been made to quantify peatland emissions on a national scale. Based on field and modelling evidence, Worrall and others (2011) estimated that net emissions for UK peatland on deep peat soils across a range of land uses was at 3.72 Mt $CO_2e\ y^{-1}$. Evans and others (2017), in their work providing UK specific 'tier 2' emission factors, estimate that UK peatlands have moved from being a net sink of 0.25 Mt $CO_2e\ y^{-1}$ pre-anthropogenic influence to a large emissions source of over 23 Mt $CO_2e\ y^{-1}$. This represents a greater than tenfold increase in emissions than previously reported under 'tier 1', due to changes in the IPCC reporting methodology and more detailed underpinning peat data, and is enough to convert the UK 'Land use, Land use change and Forestry' sector from its unique status nationally as a net GHG sink to a net source of GHGs. The work by Evans and others (2017) and it's review in 2021[24,25] (based on an updated literature review and meta-analysis, and following the same criteria as in Evans and others, 2017) represents the most comprehensive and up to date overview of emissions from UK peatlands that is available. The emissions inventory table is reproduced below (table 4.3). However, it does not disaggregate based on peatland habitat type but instead provides emission factors based on peatland land use and condition categories.

The reporting categories in table 4.3 reflect the availability of data relevant to UK peatlands. It should be acknowledged that these figures represent reporting at a national level and some caution is needed when applying at a site level. Impacts from specific management interventions such as burning, and drainage cannot be differentiated and are considered under the 'modified bog categories'. Most studies have focussed on heather-dominated modified bogs, and the lack of evidence from graminoid-dominated systems means these two categories cannot yet be assigned separate emission factors.

[24] https://unfccc.int/ghg-inventories-annex-i-parties/2021
[25] NAEI website: https://naei.beis.gov.uk/reports/

Table 4.3 Emission factors (EF) (t CO_2e ha^{-1} y^{-1}) for peat condition types taken from the 2021 update to Evans and others (2017). Note that the emission factors include CO_2, CH_4 and N_2O. A positive EF indicates net GHG emission, and a negative EF indicates net GHG removal. [a] Tier 1 default EF (IPCC 2014); [b] Tier 2 EF (updated literature analysis in 2019 incorporating data from Evans and others 2017); [c] Tier 3 Forest Research CARBINE model implied EF for 1990 to 2019. (reproduced with permission from UKCEH)

Peat Condition	Drainage status	Direct CO_2	CO_2 from DOC	CO_2 from POC	Direct CH_4	CH_4 from Ditches	Direct N_2O	Total
Forest	Drained	2.52 to -1.79[c]	1.14[a]	0.3[b]	0.06[a]	0.14[a]	1.31[a]	**5.46 to 1.15**
Cropland	Drained	28.60[b]	1.14[a]	0.3[b]	0.02[b]	1.46[a]	6.09[a]	**37.61**
Eroding Modified Bog (bare peat)	Drained	6.18[b]	1.14[a]	5.0[b]	0.14[a]	0.68[a]	0.14[a]	**13.28**
	Undrained	6.18[b]	0.69[a]	5.0[b]	0.15[a]	0[a]	0.14[a]	**12.17**
Modified Bog (semi-natural Heather + Grass dominated)	Drained	0.13[b]	1.14[a]	0.3[b]	1.26[b]	0.66[a]	0.06[b]	**3.54**
	Undrained	0.13[b]	0.69[a]	0.1[b]	1.33[b]	0[a]	0.06[b]	**2.31**
Extensive Grassland (combined bog/fen)	Drained	6.96[b]	1.14[a]	0.3[b]	1.96[b]	0.66[a]	2.01[a]	**13.03**
Intensive Grassland	Drained	21.31[b]	1.14[a]	0.3[b]	0.68[b]	1.46[a]	2.67[b]	**27.54**
Rewetted Bog	Rewetted	-0.69[b]	0.88[a]	0.1[b]	3.59[b]	0[a]	0.04[b]	**3.91**
Rewetted Fen	Rewetted	4.27 [b]	0.88[a]	0.1[b]	2.81[b]	0[a]	0[a]	**8.05**
Rewetted Modified (Semi-natural) Bog	Rewetted	-3.54[b]	0.69[a]	0[b]	2.83[b]	0[a]	0[a]	**-0.02**
Near Natural Bog	Undrained	-3.54[b]	0.69[a]	0[b]	2.83[b]	0[a]	0[a]	**-0.02**
Near Natural Fen	Undrained	-5.41[b]	0.69[a]	0[b]	3.79[b]	0[a]	0[a]	**-0.93**
Extracted Domestic	Drained	10.27[a]	1.14[a]	1.01[b]	0.14[a]	0.68[a]	0.14[a]	**13.37**
Extracted Industrial	Drained	6.18[b]	1.14[a]	5.0[b]	0.14[a]	0.68[a]	0.14[a]	**13.28**
Settlement	Drained	0.07[b]	0.57[a]	0.15[b]	0.63 [b]	0.16[a]	0.03[b]	**1.61**

Despite these caveats the 2021 update to the peatland emission factors offers important updates. The Eroded Modified Bog EF (2017) was previously believed to underestimate the loss of carbon via the particulate organic carbon pathway and the category is now split into its component parts (Evans and others 2017). Eroded Modified Bog EF now represents bare peat and is applied to actively eroding areas, while the Modified Bog represents areas dominated by heather and grass. The increase from the 2017 data, from 3.55 to 12.17 t CO_2e ha^{-1} y^{-1} and 4.85 to 13.28 t CO_2e ha^{-1} y^{-1} for undrained and drained eroded bogs respectively, brings these EFs closer to the large estimates by Smyth and others (2015), in their generation of EFs for The Peatland Code (discussed below). Smyth and others (2015) assign 'actively eroding bog' its own category, with an EF of 23.84 t CO_2e ha^{-1} y^{-1}.

Other significant changes are the generation of an specific EF for Rewetted Modified Bog which aims to reflect the net emission savings that can be achieved by restoring the hydrology of semi-natural peatlands (ie modified bog dominated by heather and grass). The negative EF (-0.02 t CO_2e ha^{-1} y^{-1}) indicating a net GHG sink, reflects the effectiveness of rewetting peatlands at a relatively less degraded baseline, for example ones that have existing cover of semi-natural vegetation and are considered to be functioning in a near healthy state. The Near Natural Fen and Bog categories have also both been adjusted, with EFs for both now reported as net sinks for GHGs.

As concern for peatland carbon stocks and GHG emission has grown, the focus on restoring our degraded bogs and fens to a near natural, lower emissions state has increased. The ONS estimate that the monetary benefits, in terms of GHG emissions, of achieving the CCC's scenario of having 55 per cent of peatland in good status, are in the order of £45 to £51 billion over the next century. This is considered conservative as it does not include the degraded peats under agriculture (Trenbirth and Dutton 2019). Developments in restoration techniques over the last 20 years means that degraded peatland areas can undergo rehabilitation management and can be significantly improved for carbon storage and biodiversity in the longer term.

The Peatland Carbon Code is a voluntary certification standard for UK peatland restoration projects to access environmental finance. The emission factors set out in the code represent verifiable and validated figures and it is the only approved scheme for purchasing and reporting carbon credits in UK peatlands. The Peatland Code differs from its woodland equivalent, the Woodland Carbon Code, in that emission estimates are based on the condition of the peatland before and after restoration and represent avoided loss or reduced emissions, rather than active sequestration. Emission factors are based on a combination of data reported in the UK relevant literature and IPCC default values (Birnie and Smyth 2013; Smyth and others 2015) but are being updated in 2021 to reflect the work of Evans and others (2017). The Peatland Code provides estimates of the net effect in GHGs that moving from one condition factor to another may have (table 4.4).

Table 4.4 Net effect in changes to GHG emissions in moving between Peatland Code condition categories (Smyth and others 2015)

Condition Category Change	Net Effect ($t\ CO_2e\ ha^{-1}\ y^{-1}$)
Restoring from Modified to Near Natural	Saves 1.46
Restoring from Drained to Near Natural	Saves 3.46
Restoring from Drained to Modified	Saves 2.00
Restoring Actively Eroding to Modified	Saves 21.30
Restoring Actively Eroding to Drained	Saves 19.30
Allowing Drained to develop into Actively Eroding	Loses 19.30

Standardised approaches to assigning emission factors are considered evidenced approximations, and do not replace the need for long-term peatland monitoring to aid understanding of in-situ greenhouse gas fluxes or pick up the heterogeneity in fluxes within sites. Due to the lack of data neither Smyth and others (2015) or Evans and others (2017) separate ombrotrophic bogs into separate blanket and raised bog categories. Though blanket peats have denser peats than raised bog peats (Chapman and others 2015), the two habitats are considered to function in a similar way (Smyth and others 2015). This is uncertain, as blanket bogs and lowland raised bogs do have significant differences in their use, topography, nutrient status, climate and hydrology (Evans and others 2017). Further evidence to support the separation of these two peat categories is urgently required.

The restoration of peatland is not instantaneous with interventions taking at least 5 years or more for ecosystem changes to stabilise, when the end point is dependent on achieving an established peatland vegetation community. In terms of carbon saving from restoration, this will be dependent on the starting condition prior to restoration with severely degraded sites taking longer to achieve emissions reduction than less affected peatlands (Artz and others 2012). In terms of restoring the carbon sink function of a peatland, Lindsay (2010) suggests a timeframe of around four decades before restoration to a fully functional bog can achieve net carbon gain, although emissions reduction will occur much earlier.

4.3 Blanket Bog

Blanket bog is a climate derived ecosystem formed under high rainfall and cloud cover, and as such is typically found in the uplands of the north and west UK where conditions have historically favoured peat formation. English blanket bogs occur at relatively high altitude, with a minimum of 160 days of rain per year and a relatively high annual rainfall of at least 1200mm (Rodwell 1991). These peatlands are so-called as they form a blanket of peat across the landscape where land is flat or gently sloping. Blanket bog initially fills in wet basins in the topography, their saturated conditions creates a microclimate which aids their further expansion over many millennia. As they develop, the peat layers form a buffer between the vegetation and underlying geology and as such, are acidic and low nutrient habitats, covered by plants such as *Sphagnum* moss, heathers *Calluna vulgaris* and other *Ericaceous* species and cotton grasses *Eriophorum spp*. In England this habitat is categorised at 40 cm peat depth with shallower organo-mineral soils being considered a wet heath habitat type, often forming a complex and hydrologically linked mosaic with blanket bog.

Blanket bog is by far the most extensive peatland habitat in England with around 355,000 ha of blanket bog soils and over 230,000 ha still classified as Priority Habitat with semi-natural vegetation still persisting (Natural England 2010). Much of the blanket bog habitat now has cover of heather, grassland or forestry and substantial areas of eroded, bare peat, which impacts it ability to sequester and store carbon. While blanket bog predominates, and where conditions allow, it does so in a mosaic and may be linked with other habitats such as minerotrophic fen, flushes, acid grassland and wet heath.

4.3.1 Carbon storage and sequestration in blanket bog

Estimates of peatland carbon stocks are dependent on accurate estimates of peat depth. Peat depth can be highly variable due to the belowground topography, with blanket bogs extending over large areas that can be relatively shallow compared to the deeper basin peats. Many landscape and national scale estimations of peatland carbon stocks often only consider the surface peats, basing estimations on depths of 1 m or less, missing significant stocks at depth. In addition to this, standardised estimates of carbon content and bulk density are typically applied when estimating carbon stocks which can also vary depending on site conditions (Natural England 2010).

As discussed previously in this chapter, Ostle and others (2009) use CS2007 data to estimate that bog habitats store 259 ± 8 t C ha-1 when extrapolated to 50 cm depth. Ward and others (2007) working across control, burn and grazed plots in the North Pennines report an average total carbon density of 500 ± 5 t C ha-1 in blanket peat to 1 m depth. Heinemeyer and others (2020) used ground penetrating radar (GPR) depth surveys coupled with site based organic carbon and bulk density data to calculate area weighted mean soil carbon stocks for the full peat profile of three upland blanket bog catchments in northern England, as well as manual measurements for study plots located on flatter areas of blanket bog. The GPR survey reported carbon stocks ranging 354 – 619 t C ha^{-1} and captures the inherent variability in blanket bog topography at a catchment scale, while plot based stocks estimates ranged between 653 – 954 t C ha^{-1}, reflecting deeper peat depths under flatter areas.

The carbon stored in English peatlands below 1 m depth is estimated at approximately 42 per cent of that stored in the surface peats and is typically not accounted for (Billett and others 2010; Fyfe and others 2014). Working in blanket peats in Dartmoor Fyfe and others (2014) report a variability in total carbon stocks at an order of magnitude, largely controlled by the depth of peat. Blanket peat to a depth of 1 m store 565 t C ha^{-1}, similar to studies cited above. The deepest peats however, at depths of 6.63m hold 5248 t C ha^{-1}. Overall, Fyfe and others (2014) report the mean total carbon storage in blanket peats on Dartmoor is 1581 ± 1056 t C ha^{-1}. The issue of depth and its importance to the calculation of blanket bog carbon stock is evident in the literature when attempting to define typical carbon stocks in this habitat.

Table 4.5 Summary of peat carbon stocks, as reported in the literature

Peat depth (m)	Carbon Stock (t C ha^{-1})	Reference
0.5	259 ± 8 (organic carbon)	Ostle and others (2009)
1.0	500.10 ± 4.96 (total carbon)	Ward and others (2007)
2.0	653 – 944 (organic carbon)	Heinemeyer and others (2020)
1.01 6.63	565 (total carbon) 5248 (total carbon)	Fyfe and others (2014)

Few studies have focussed on capturing the emissions from 'pristine' or intact blanket bogs, with most work in the UK investigating the impact from management interventions. In their comprehensive review of UK peatlands Billett and others (2010) report historic rates of sequestration of -0.35 to -2.09 t C ha^{-1}, with contemporary figures at -0.56 to -0.72 t C ha^{-1}. Artz and others (2014) argue that converting these figures to CO_2 equivalent does not adequately capture the impact of more potent greenhouse gases such as CH_4 and suggest that UK peatlands with minimal management can be compared with other boreal and northern temperate peatlands. Artz and others (2014) suggest that ombrotrophic peatland sequestration rates, inclusive of CO_2 equivalents of CH_4 and other carbon losses, would be in the region of -0.76 ± 0.39 t CO_2e ha^{-1} y^{-1} (net sequestration).

4.3.2 Carbon cycling and management interventions in blanket bog

Water level management

Due to their low agricultural value blanket bogs in the UK have long been subject to modifications to improve their productivity in managing the land for livestock, grouse shooting and forestry plantations. Drainage via grips (open, shallow drains) dug into the peat was extensive through the 20th century, peaking in 1960 and 70s as a result of government subsidies (Wilson and others 2011), with the aim of replacing peatland plant species with an increased grass coverage to support sheep grazing. As discussed previously in this chapter, drainage can lead to enhanced loss of carbon by increasing the volume of oxidised peat, promoting decomposition and high rates of CO_2 emissions.

Raising water levels via drain or 'grip' blocking has been used as a restoration method on peatlands since the 1980s to restore hydrological function and create the conditions for re-establishment of peat forming vegetation (Green and others 2018) (Picture 4.2). Despite the link between water level management and the potential benefits for climate change mitigation, there is limited empirical data in the context of blanket bog rewetting. Rowson and others (2010) completed a carbon budget of a drained peatland catchment post-ditch blocking in the North Pennines, reporting that the catchments was a net source of all forms of carbon at between 63.8 and 106.8 Mg C km^{-2} y^{-1}, (0.638 - 1.068 t C ha^{-1} y^{-1}; 2.341 - 3.920 t

$CO_2e\ y^{-1}$) with carbon export via waterborne pathways greater than that to the atmosphere. However, as there was no control and the study only monitored one year following ditch blocking, only the short-term response was captured and cannot be used to infer the long-term impact of raising water levels. Green and others (2016) compared open ditches, dammed ditches and re-profiled ditches over four years at Migneint blanket bog in the upper Conwy catchment in North Wales and reported no clear GWP benefits from ditch blocking. For the initial 3 years of the study, all areas had a positive GWP (radiative warming), including restored treatments. In the fourth year GWP was negative (radiative cooling), though explained by the authors as a result of changes in meteorological conditions between the years of study rather than hydrological restoration. Working in shallow, *Molinia* dominated blanket peats at Exmoor, Gatis and others (2020) also report no significant differences in carbon flux between restored and control plots even 6 years post rewetting.

Picture 4.2 Grip blocking techniques use to raise water levels. Plastic has been used to block the drains or grips, but now natural materials, such as bales, brash, wood and peat is preferred. Left photo © Natural England/David Glaves; right photo © Natural England/Peter Wakley

Several reasons have been proposed regarding the lack of response of blanket bog peats to ditch-blocking and rewetting. Blanket bog peats have been reported as less responsive to drainage and rewetting than other peat types due to their inherent low hydraulic conductivity (Evans and others 2014). Long-term drainage in the uplands has led to subsidence in peat soils, lowering the peat surface to that of the drained water table level (Williamson and others 2017). Oxidation of the subsided peat may have led to carbon loss of 3 Mg C ha^{-1} (3 t C ha^{-1}) and the legacy of this loss and decreased peat surface may mean restoration by rewetting alone may not be as effective as hoped in delivering reduced carbon emissions (Williamson and others 2017). Finally, Gatis and others (2020) suggest that the regular drying in drained peat leads to a subsurface pipe network, leading to rapid throughflow when water levels are raised and potentially negating the benefits.

While little impact of ditch blocking on carbon fluxes have been observed on the blanket peat body, due to the ground disturbance and water level rises an effect could be expected in the drain itself. Evans and others (2018) working at the same site as Green and others (2016) report that ditch blocking had no impact on fluvial carbon export but note the drains under study were not observed to be actively eroding. The infilling of ditches has led to the ditches themselves being reported as a CH_4 hotspot, driven by the colonisation of the bare peat of *Eriphorum vaginatum*, which transports CH_4 via its aerenchyma from the anaerobic peat depth to the atmosphere. Cooper and others (2014) report that such ditches generate almost half the total CH_4 emissions from a rewetted blanket bog from less than 10 per cent of the site surface in the two years post blocking with hay bales, and that this may be an underestimation as CH_4 loss from ebullition was not measured. In contrast, Green and others (2018) did not see such an effect, commenting that restored ditches do not necessarily have to be a carbon source, even in the immediate post-blocking period. The difference in observations between the studies maybe due to high variability in fluxes observed (Green and others 2018), differences in blocking techniques (hay bales vs peat), or presence of *Eriophorum spp*. (Cooper and others 2014).

While the response to raised water levels in blanket peat may be considered unexpected, it should not deter those working in their restoration from seeking to restore hydrological function. Instead, practitioners should seek to identify where peatlands have suffered subsidence or have adjusted to past drainage regimes in order to prioritise where water table recovery is possible (Holden and others 2016). Raising water levels should be considered a 'no-regrets' option, in that benefits may be realised over longer periods as the peatland ecosystem adapts to wetter conditions (Evans and others 2018). Raised water levels are required in order to facilitate the establishment of peat-forming vegetation such as *Sphagnum* mosses, prevent further subsidence of the peat and increase resilience of blanket bogs to future climate change (Natural England & RSPB 2020).

Vegetation management

The unclear relationship between raised water levels and carbon cycling could be due to a time lag in the response of other peatland functions, such as shifts in vegetation composition. Green and others (2017) did not identify any treatment effects of ditch blocking on blanket bog vegetation, though the abundance of sedges and *Sphagnum* did vary significantly between years of the study. Furthermore, *Sphagnum* species increased in abundance with time over the 5-year study, though not limited to gripped sites. Green and others (2017) comment that the lack of response to ditch blocking maybe due to the legacy of past degradation on the blanket bog and that water table levels may not be the primary driver in determining peatland vegetation communities. Bellamy and others (2011) working on blanket bog in the Flow Country observed ditch blocking increased the cover of plant species indicative of wet soil conditions, in particular active peat-forming vegetation. However, results were variable and mainly limited to the site on where drains had been blocked for the longest period (11 years), possibly due to the slow growth rates of *Sphagnum* species.

The composition of restored vegetation communities warrants consideration due to plant-soil interactions influencing the carbon cycle. As discussed above (Cooper and others 2014) areas of *Eriophorum spp.* can lead to hotspots of CH_4 emissions, an effect that was also noted by McNamara and others (2008) in eroded peatland gullies. Armstrong and others (2010) observed that catchments dominated by *Calluna* tend to have higher DOC concentrations. The dominant plants in a peatland community can exert significant control of GHG fluxes. Vegetation associated with degraded sites such as rushes and grasses can stimulate carbon losses from the peatland system via labile litter and root exudates. Whilst the ericaceous shrubs and sphagnum species present in an ecologically functioning bog are associated with reduced carbon fluxes and enhanced sink capacity (De Deyn and others 2008; Ward and others 2013; Dunn and others 2016).

The revegetation of peatland plant communities on bare peat areas should be a priority in peatland restoration, as without carbon inputs from photosynthesis exposed peat will always be a net emitter of gaseous carbon (Parry and others 2014b), with carbon losses as high as 5.2 t C ha^{-1} y^{-1} put forward by Worrall and others (2011). Revegetating sites can act to both physically stabilise the carbon stored in the peat as well as sequester it in plant biomass. We have found no UK based studies that have quantified the impact of targeted revegetation with peatland species, though there is evidence based on the common approach of using a grass nurse crop to revegetate peat surfaces, with heather brash and jute netting to reduce erosional loss. Such sites have unsurprisingly been reported to have greater rates of gross photosynthesis than bare peat, and when revegetation and stabilisation approaches have been combined productivity at the site is greater than that of least disturbed control plots over a 5 year monitoring period (Dixon and others 2013). Working at the same site Qassim and others (2014) report increasing DOC concentrations in soil porewater post revegetation, but state this would in fact be reduced if restoring vegetation was coupled with raising of water levels. These studies suggest that a suite of restoration approaches is most effective at restoring a peatlands ecological functions and its status as a carbon sink, though further work should prioritise the impact of successional changes toward peatland vegetation communities on carbon cycling. This can take 5–10 years or more (Lindsay 2010), a time period longer than the duration of many of the studies cited in this review.

Restoration of afforested peatlands

The impact of woodland on carbon cycling in peat soils has been discussed section 2.2.5. Afforestration on peatlands typically involves drainage, to lower water levels, and disturbance of the deep, anaerobic catotelm during ground preparation and planting (Robson and others 2019). As described in section 4.2.1 ‘Carbon cycling in peatlands’, such actions lead to loss of carbon via oxidation of the previously water-logged soils, and physical disturbance and erosion. Over the long-term, water table drawdown due to transpiration by the trees exposes a greater depth of peat to oxidation (Lindsay and others 2014), and carbon loss via CH_4 emitted from ditch networks may be important (Baird and others 2009). While large amounts of carbon will be sequestered in the tree biomass, it may not be enough to negate the losses from oxidation of peat soils, especially when considered over the long-term eg centuries (Evans and others 2017).

Data regarding the restoration of plantations to semi-natural peatland is lacking in the evidence base, with few UK based studies from which to quantify its benefits relating to carbon cycling. Hambley and others (2019) used eddy-covariance flux measurements to compare carbon fluxes from a site 10 years post restoration (0.80 t C ha^{-1} y^{-1} / 2.93 t CO_2e y^{-1}) and 16 years post restoration (-0.71 t C ha^{-1} y^{-1}/ -2.60 t CO_2e y^{-1}). Differences between the sites were due to carbon loss by respiration, rather than carbon uptake by photosynthesis. Hermans and others (2019) demonstrate in a core incubation study that forestry on peat alters peat quality and nutrient availability, an impact that persists following restoration intervention and continues to influence rates of CO_2 emissions. Hermans (2018), working in the Flow Country across a forest to bog chronosequence, observed CO_2 and CH_4 fluxes are gradually returning to values similar to undamaged blanket bog. These studies suggest that following restoration, peatlands previously under forestry management can return to a functioning net carbon sink. Though carbon data is lacking for this area of land management, a comprehensive overview of forest to bog restoration techniques and example case studies are discussed in Robson and others (2019).

Burning

Burning has been used as a tool in the English uplands for hundreds of years as a way of diversifying the age and structure of shrubs for game management and creating new growth for livestock grazing. Burning is also cited as a conservation approach, where it has been carried out to manage pest outbreaks, create firebreaks and create structural diversity in the vegetation community (Glaves and others 2013). Thirty per cent of blanket bog in England has been subjected to burn management, and despite its extensive and long-term use research into its effect on carbon cycling in peatlands is not definitive, with variable responses reported (Heinemeyer and others 2020; Glaves and others; 2013; Natural England 2010).

Many studies use the long-term Hard Hill plots at Moor House National Nature Reserves (NNR) in the North Pennines to explore the relationship between burning and carbon cycling (picture 4.3). Garnett and others (2000) report significantly less carbon stored in the peat after 30 years under the 10 year burn regime compared to the unburnt plots, but it is not known if this is due to loss in the peat carbon store from burning or a reduction in the accumulation rate. Working on the same plots, Marrs and others (2019) report contrasting results. Observing that though burning reduced accumulation rates, and these impacts intensified with burn frequency, burning did not prevent accumulation of carbon in surface peats which continued to increase even after 6 burns. Also working on the Hard Hill plots, Ward and others (2007) found burnt treatment plots had reduced carbon stocks in vegetation and surface peats when compared to control plots, with the total loss of carbon due to burning estimated to be 25 g C m^{-2} y^{-1} (0.25 t C ha^{-1} y^{-1}). Changes in vegetation as a result of long-term burning led to an increase in the gross CO_2 fluxes of respiration and photosynthesis.

Picture 4.3 Hard Hill burn plots, Moor House National Nature Reserve © Natural England/David Glaves

Clay and others (2010) used data from the Hard Hill plots and data from the Environmental Change Network to create a catchment scale carbon budget. The results differ from the previous studies as all treatments are reported as carbon source, and this increases with time since burn (5.75 t CO_2e y^{-1} at unburned plots; 4.62 g CO_2e y^{-1} at 20 year burn interval; 4.02 t CO_2e y^{-1} at 20 year burn interval). A similar relationship was observed by Clay and others (2015) using a 10-year burning chronosequence at a different site in Northumberland, where more recent areas of prescribed burning were smaller sources of carbon than older plots. Whilst burning reduces the net carbon emissions, driven by increased photosynthetic uptake of carbon in younger plants, burning management on these *Calluna* dominated peatlands does not result in a net sink and instead the results represent an 'avoided loss'. The authors note for net storage to occur the vegetation needs to convert to a wetter, peat forming *Sphagnum* dominated community.

In recent years the limitations of using single sites have been recognised and research into the impact of burning in the uplands has been commissioned on multi-catchment scales. The EMBER (Effects of Moorland Burning on the Ecohydrology of River basins) study was undertaken across 10 catchments split equally between burnt and unburnt in the Pennines. While the study did not look directly at the impact of burning on peatland carbon cycling, it did report that burning caused multiple negative environmental impacts that could be associated with the reduction in a peatlands potential to store or sequester carbon; such as lowered water tables, greater soil surface temperature extremes, loss of bog vegetation and peat-forming mosses and increases in water runoff (Brown and others 2014).

Defra commissioned a 5 year project on *'Restoration of blanket bog vegetation for biodiversity, carbon sequestration and water regulation'* (Heinemeyer and others 2020) to test management interventions, including burning, to reduce dominance of *Calluna*, and to monitor the effects of the techniques on vegetation, carbon and water. The reported carbon values show that prior to the experimental management interventions all management groups (burnt, mowed, no intervention) acted as net carbon sources (though all had previously been subject to rotational burning). In the four years post-management, the carbon budget values indicate that burnt and mown treatments were net carbon sources and the non-intervention sites plots either a small net source or sink depending on if whether the mean or median flux components were used to calculate the NECB. Overall, NECB varied considerably across the sites and years.

Much of the findings reported above are from sites where bog vegetation is dominated by heather, *Calluna vulgaris*, an ericaceous shrub indicative of dry, degraded conditions on peatlands. Prescribed burning can be recommended as a technique to reduce *Calluna* dominance and mitigate the risk of hotter, more intense wildfires (Marrs and others 2019). Whilst burning can reduce fuel load and hence wildfire hazard, Glaves and others (2020) found only limited evidence of its direct effect on wildfire ignition, behaviour, severity and extent, or in reducing wider negative impacts. Moreover, Glaves and others (2020) suggest that where appropriate, such as on dry heath, grouse-moor-type habitat, small patch burning on moderate rotations 'might not necessarily provide the most effective spatial pattern, frequency or approach specifically for reducing wildfire risk, occurrence and impact. Where this is an objective, a more strategic approach targeted at high risk locations such as access hotspots/routes, probably with more frequent and varied treatments, might be more effective and efficient, and potentially also result in a smaller total area being burnt.'

Responding to Marrs and others (2019), who recommend the use of controlled burning to mitigate wildfire risk, Baird and others (2019) note that degraded blanket bog sites are inherently a fire risk due to their combustible vegetation cover, while ecologically functioning peatlands with high-water tables and *Sphagnum* moss cover reduce the risk of deep burning by increasing the energy required to ignite peat and thereby restrict the burn depth. Though burning management on peat remains a contentious issue, there is increasing consensus in the scientific evidence that burning on blanket bog is damaging. Raising water levels and restoring peat-forming vegetation is a more effective way of managing these sites to reduce fire risk (Granath and others 2016; Glaves and others 2020), as well as delivering significant benefits for climate change mitigation, biodiversity and water quality.

4.3.3 Climate change and other interactions

Blanket bog is considered to have a 'medium' sensitivity to impacts under projected climate change (Natural England & RSPB 2020), with an increased risk associated with interaction with the damaging land uses described in the sections above. Though blanket bog is one of England's most extensive semi-natural habitats it is restricted to areas of relatively high altitude and rainfall. Climate change will influence carbon cycling on bog habitats primarily through changes to precipitation patterns, but also via temperature. This could be positive, for example by lengthening the growing season of *Sphagnum* mosses, or negative, by increasing evapotranspiration and rates of microbial decomposition (Lunt and others 2019).

Future climatic changes have been projected to result in the contraction of the distribution of active blanket bog in the UK towards the north and west, with the main driver being increasing summer temperatures (Gallego-Sala and others 2010). There is uncertainty with regards to the fate of peat outside this bio-climatic envelope. It may cease active growth and no longer be an active carbon sink, but it is suggested that increased resilience may be provided, and rapid carbon loss avoided if *Sphagnum* cover is maintained (Gallego-Sala & Prentice 2013). However, peatlands have withstood millennia of climate fluctuation in the past and persist in modern times. Degraded peatland systems are more vulnerable to climate change as they are already under pressure, therefore their restoration should be considered a priority.

Atmospheric deposition of sulphur and nitrogen may impact carbon cycling in blanket bogs through impacts to peatland vegetation and chemistry (Evans and others 2014). Blanket bogs close to urban centres, such as in the South Pennines, were subject to sulphur deposition from the burning of fossil fuels resulting in the severe acidification of these already acidic habitats. Though sulphur deposition peaked during the 1960s and 70s, and has now significantly declined, it continues to have a legacy impact leading to the die back of *Sphagnum* mosses and other sensitive peatland species leading to bare peat and erosional loss. However, sulphur deposition is related to reduced carbon emissions via DOC (Monteith and others 2007) and suppressed CH_4 emissions (Gauci and others 2004). As habitat recovery from acidification progresses, losses via these pathways may be exacerbated. Nitrogen deposition, associated with emissions from agriculture and transport, is a continuing problem resulting in the enrichment of these nutrient limited habitats. Low levels of nitrogen deposition may increase the carbon sink on peatlands, through stimulating vegetation growth. However, this displacement of keystone species such as

Sphagnum mosses by vascular plants with high nutrient demands will reduce the sink strength in the longer term, potentially turning the peat into a carbon source (Evans and others 2014).

4.4 Raised Bogs

Raised bogs are peatland ecosystems which develop primarily in lowland areas such as the head of estuaries, along river floodplains and in topographic depressions. In such locations drainage has been impeded by a high groundwater table, or by low permeability of the underlying rock or sediment. The resulting high-water table gives rise to open water or fen habitats, the surface of which, through succession and accumulation of peat become increasingly distant from the groundwater table and the influence of minerotrophic conditions. This promotes the establishment of *Sphagnum* mosses, which then further acidify their environment. The term 'raised' bog is derived from this long-term accrual of peat that over time elevates the bog surface forming a gently curved dome raised above the surrounding landscape (Lindsay 2016) (figure 4.3). Peat depths under raised bogs are variable but can exceed 12 m, and therefore represent the deepest peat deposits in the UK (Artz and others 2012).

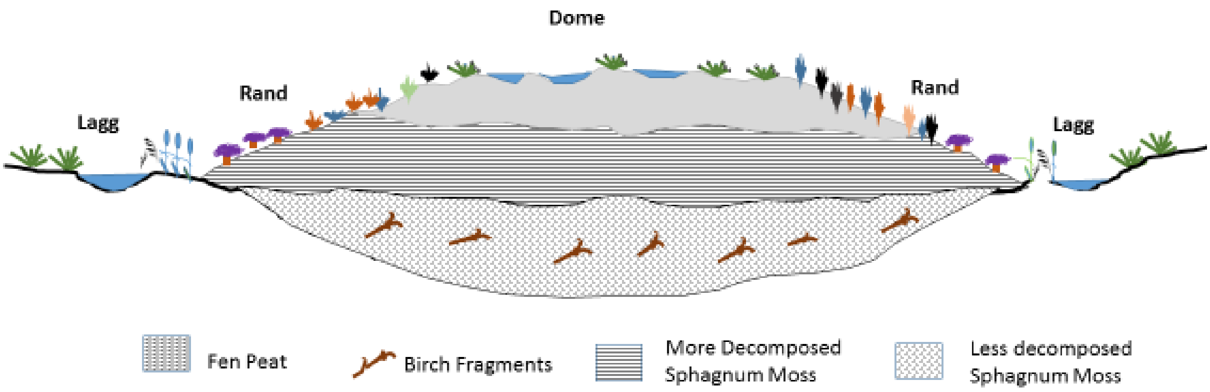


Figure 4.3 A cross section of a raised bog, showing its structure and layers of development (adapted from Aalen and others 1997)

Raised bogs are ombrotrophic, supporting a specialised vegetation community dominated by *Sphagnum* mosses, ericaceous shrubs, and scarce plants such as bog rosemary *Andromeda polifolia* and sundew *Drosera spp* (Biodiversity Reporting and Information Group 2008) (picture 4.4). While this plant assemblage is similar as that on blanket bog, raised bogs are distinguished by their location, depth, altitude, extent and often the presence of fringing 'lagg' fen vegetation (figure 4.3) where acid water draining from the bog meets water draining towards the bog from the surrounding landscape (Shepherd and others 2013). The lagg plays a major role in the overall hydrology and functioning of the bog. Its loss can have substantial long-term impacts on the bog in terms of hydrology, peat morphology, biodiversity and therefore the carbon storage within the bog (Howie & Tromp-van Meerveld 2011; Lindsay 2016a).

Picture 4.4 Raised bog vegetation at Roudsea Woods and Mosses NNR. ©Natural England/Ruth Gregg

4.4.1 Carbon storage and sequestration in raised bogs

Though raised bogs have a restricted range, covering around 35,000 ha in England (Natural England 2010) and national stocks estimated at 57.5 Mt C, on an area basis they hold the largest carbon store of peatlands due to their very deep soils (Natural England 2010). Despite their importance few studies report carbon stocks based on the full depth profile. Working on a range of lowland peatland sites Evans and others (2016) report variable carbon stocks from four raised bog sites. An arable site on deep peat in the Manchester Mosses holds 1,290 t C ha^{-1}, whilst a rewetted semi-natural site at the same location reports stocks of 2,530 t C ha^{-1}. The lowest stocks are reported at the rewetted former peat extraction sites at the Manchester Mosses and Thorne Moors, storing 810 and 940 t C ha^{-1} respectively. These figures are substantially higher than those reported in national surveys such as the Countryside Survey (the bog category estimates 259 ± 8 t C ha^{-1} when extrapolated to 50 cm depth) and illustrate the significant and often unaccounted carbon stocks held at depth.

Whilst raised bogs are found in upland environments such as the Border Mires, they are typically found more in a lowland setting in England. Lowland peatlands have been subject to greater pressures than their upland counterparts due to their proximity to towns and cities, more intensive land use in agriculture (see section 3.4.4) and through peat extraction, both industrial and via hand-cutting for fuel. Drainage to facilitate their use occurred earlier than in the uplands, with lowland peats being drained to improve them for agricultural use from the 17th century (Holden and others 2004). The impact of lowering the water table, and the subsequent oxidation of surface peat is described previously in this chapter. However, due to differences in the hydraulic conductivity, raised bogs are more sensitive to drainage management than blanket bogs (Evans and others 2014). There is evidence for the higher hydraulic conductivity of raised bogs from the wider spacing of drainage channels on raised bogs, with ditches influencing an increased area before the next ditch needs to be cut.

The area of raised bog in the UK that retains a largely intact surface is estimated to have diminished by around 94 per cent, from an original extent of around 95,000 ha to around 6,000 ha today (Biodiversity Reporting and Information Group 2008). As a result, there is a lack of long-term studies assessing the carbon budget of undamaged raised bogs. Auchencorth Moss, in central Scotland, represents one of the longest lowland peatland carbon flux studies to date and has been the subject of several studies quantifying its carbon budget. Dinsmore and others (2010) worked at the Auchencorth catchment for two years reporting that the site was a net sink for both carbon (0.69 t C ha^{-1} y^{-1}) and GHGs (3.52 t CO_2e ha^{-1} y^{-1}). Helfter and others (2015), using 11 years of continuous eddy-covariance data report Auchencorth Moss is a consistent yet highly variable CO_2 sink (0.64 ± 0.34 t CO_2 ha-1 y-1), but weaker than that reported by Dinsmore and others (2010). The importance of carbon export by fluvial routes at the site was demonstrated by both Billet and others (2004), who report this pathway is of a similar magnitude to the net CO_2 flux, and Dinsmore and others (2010) who report aquatic fluxes represent 41 per cent of NEE carbon. However, Auchencorth Moss has been subject to drainage and peat extraction in the past, and is grazed, thus could be considered more like a transitional peatland than raised bog. Therefore, England or UK based raised bogs flux measurements remains a significant evidence need.

4.4.2 Carbon cycling and management interventions in raised bogs

Peat extraction, industrial and domestic cutting

Peat extraction has an extremely damaging impact on raised bogs as the process strips away the living layer of the bog to expose large milling flats, where the peat can dry before being removed, stacked and dried further before being bagged for horticultural use (Lindsay and others 2014). It has been carried out in the UK for centuries, first for domestic fuel use via hand cutting and then on a larger industrial scale for the horticultural and growing sector. While peat extraction in the UK is on the decline, and the UK Government has set a target to cease using peat in horticultural products by 2030 in its 25 year plan for the environment (2019), large volumes continue to be removed with 800,000 m^3 extracted in 2015 (Trenbirth and Dutton 2019).

Peat extraction requires the raised bog to be extensively drained, with total removal of the acrotelm, leading to oxidation of the surface peats and enhanced emissions of CO_2. The milled surfaces are also kept bare, removing the plant mediated uptake of CO_2 from the atmosphere that in other degraded peat environments offsets the net emissions. Bare peat is also susceptible to desiccation and loss via wind erosion as it is not protected by vegetation cover, but this is yet to be quantified in terms of carbon loss. Working in extracted peatland sites in the UK and Ireland, Wilson and others (2015) give a mean emission factor of 1.70 (± 0.47) t CO_2e ha^{-1} y^{-1} for industrial sites. Evans and others (2017) included two raised bog peat extraction sites on in their work, one was an active extraction site in the Manchester Mosses and the other, at Thorne Moors, Humberside, was abandoned, though the site was still drained and without vegetation cover. The emissions from the sites were similar to each other at 7.36 and 6.27 t CO_2e ha^{-1} y^{-1}, suggesting that until restoration is started, active and abandoned extraction sites behave similarly. Methane emissions were low, and a net sink at the Thorne Moors site, due to low water levels and constant disturbance of the peat surface. However, neither study monitored CH_4 emissions from the drainage ditches which could be significant.

The emissions reported above reflect *in situ* peat and do not cover emissions from peat removed from site for use in horticulture or energy. Evans and others (2016), using figures from Wilson and others (2015), calculates this may result in indirect, off-site emissions from harvested peat of around 40 t CO_2-C ha^{-1} y^{-1}.

The impacts of past hand cutting for domestic fuel can typically be seen around the edges of raised bogs. While this activity has now declined, its cultural legacy has meant that many bogs still have ‘turbary rights’ for communities to cut peat. In some instances, this has now resulted in land ownership that is split up from a central point into multiple segments of the bog. Though hand cutting is on a smaller scale than industrial peat extraction its effect on carbon emissions is similar as drainage at the peat margins also affects water tables within the peat body. On sites where industrial extraction has not occurred this has still led to subsidence of the peat dome characteristic of raised bogs and indicating degradation of the peat extends across a wider area than the bogs edges. As a result, Wilson and others (2015) report similar emission factors for domestic extraction sites as they do for industrial extraction sites at 1.64 (±0.44) t CO_2e ha^{-1} y^{-1}.

Restoration of raised bogs

Due to the recognition of the importance of raised bogs to both climate change mitigation and biodiversity, increased funding and effort has gone into their restoration in recent years. Peatland restoration aims to re-establish the cover of peat-forming vegetation but the type of intervention to do this is dependent on the level of modification, but typically consists of reworking the peat surface to form cells, raising water levels to counteract drainage and undertaking vegetation management such as scrub removal or revegetation of bare peat surfaces (picture 4.5).

Picture 4.5 Earthworks during peatland restoration at South Solway Mosses, Cumbria. © Natural England / Ian Crosher

On peatland sites that have been modified by artificial draining, raising water levels and rewetting the peat is essential to restore hydrological functioning and limit further peat loss from oxidation. The objective of hydrological restoration is to raise the water level to within 10 cm or so of the peat surface to maintain saturated surface conditions conducive to growth of bog species, particularly *Sphagnum* mosses, and consequently the re-establishment of the acrotelm. As with blanket bog this is normally achieved by blocking all artificial drainage and sealing pipes and cracks in the damaged peat that provide water with escape routes from the bog. However, raised bogs differ from blanket bogs in that they have a low surface slope. In recent years, techniques involving creation of small-scale ‘cells’ contained by bunds of impermeable peat sunk below the surface and protruding for 10–15 cm above the surface have proved effective in rewetting previously very dry and damaged bog surfaces (Natural England 2016). The carbon fluxes from restored sites can be similar to intact peatlands but may be more variable as site specific factors such as time since rewetting, vegetation composition and previous land use will influence carbon cycling (Renou-Wilson and others 2019).

Two rewetted raised bog sites were included in the Lowland Peatland study Evans and others (2016) with both reported as net emitters of carbon. Astley Moss, previously an extraction site that has been rewetted incrementally using bunds has a NECB of 29.8 g C m^{-2} y^{-1} (1.09 t CO_2e ha^{-1} y^{-1}). The site was a net sink of CO_2, driven by high rates of uptake by *Molinia* vegetation that had been dominant pre-rewetting and had persisted. However, this net sink of CO_2 was offset by large fluxes of CH_4 and DOC which was attributed to deep pools and water levels remaining continuously at or above the peat surface. The second site, Thorne Moors, is also a rewetted extraction site that has been undergoing restoration for over a decade and now dominated by cotton grass *Eriophorum* in large bunded cells. Water levels are at or above the surface, with flooding often occurring in some parts of the site during winter, but some drawdown occurs during the drier summer months. While CH_4 fluxes are positive, they are lower than those reported at Astley Moor. The net CO_2 exchange oscillates either side of zero during the growing season but the site becomes a net CO_2 source during hot summer periods with an annual NEE reported at a 223 g C m^{-2} y^{-1} (8.54 t CO_2e ha^{-1} y^{-1}), and an overall NECB of 255.7 C m^{-2} y^{-1} (9.38 t CO_2e ha^{-1} y^{-1}). These results suggest that rewetting alone is not enough to restore the carbon sink function in raised bogs and further interventions are necessary.

Working in a restored and 'cutover' (add and explanation?) raised bog in Ireland, Swenson and others (2019) argue that the interaction between vegetation and water levels require more consideration when assessing the climate change mitigation potential that restoration may have on degraded peatlands. On the cut area of the bog that had received no active restoration interventions, the *Calluna* cutover ecotype was a considerable carbon source, the *Eriophorum* cutover ecotype was approximately carbon neutral, and the *Sphagnum* cutover ecotype was on average a moderate carbon sink. The *Eriophorum* dominated areas were also reported to produce the highest CH_4 emissions. On the restored bog and despite similar hydrology, *Sphagnum* dominated areas (mean average 98 per cent *Sphagnum* cover), on average, were a carbon sink while lower quality areas (mean average 57 per cent *Sphagnum* cover) were, on average, a moderate carbon source. Such results indicate the importance of restoring *Sphagnum* cover when aiming to re-establish a carbon and GWP sink on raised bogs, and the need to restrict the establishment of CH_4 transporting species such as *Eriophorum*.

4.4.3 Climate change and other interactions

As self-controlling systems, intact and fully functioning raised bogs are considered relatively resilient to the impacts of climate change. However, as described previously, all raised bogs in England have been severely degraded from land use pressure, and this interacting with climate change means they are assigned as at medium risk (Natural England and RSPB 2020). Warmer, drier conditions in summer in combination with drainage impacts means that raised bogs are at risk from drought impacts, drying out the peat and increasing peat loss via oxidation and erosion. Lower water tables enable establishment of scrub and trees, such as *rhododendron* spp., birch and conifers, further exacerbating the degradation of these sites. Tree and scrub removal are necessary to restore the hydrology of raised bog sites but will represent a loss of carbon from the ecosystem (Natural England 2016). The trade-offs associated with removal of vegetation from habitats with predominately organic soils is discussed in more detail in section 3.2, 'Heathlands'.

The margins of raised bogs have very often been damaged by peat-cutting and drainage to facilitate various activities including forestry and agriculture. This has hydrological impacts across the entire bog habitat, limiting the effectiveness of restoration interventions in central areas of the bog, and preventing the full restoration of the natural hydrological processes and full range of biological diversity. As a result of this damage to the margins, there are few remaining examples in England of the transitional lagg habitat at the interface of the ombrotrophic and minerotrophic environments, and nearly all that remains is modified (I. Diack pers. comms). As with blanket bog, atmospheric nitrogen deposition from agricultural fertilisers and storage of slurries can enrich these nutrient- poor habitats, potentially leading to a shift in the relative abundance of different *Sphagnum* species, to favouring vascular plants over bryophytes, with impacts to carbon cycling (Evans and others 2014).

Despite their known importance to climate change mitigation and biodiversity, raised bogs are still subject to extraction to supply demand for horticultural peats. Priority should be given to protecting existing stocks. With industrial extraction being phased out and peat reserves being exhausted at many locations, such sites will become available for restoration in the coming years and decades. Post extraction, sites tend to be restored to a nature conservation or public amenity land use. This typically constitutes open water and a habitat mosaic of fen, woodland and sometimes grassland. Sites have rarely been restored to peat forming habitat, and those that do have tended to be owned by conservation organisations. In recent years previously extracted sites such as Bolton Fell Moss[26] and Thorne Moors[27], managed as National Nature Reserves by Natural England, have demonstrated large scale restoration with restoring peat forming conditions a central aim, as part of the EU Life+ programme. Paludiculture, or farming under raised water levels, is another potential use for previously cutover bogs. As most peat used in horticulture is *Sphagnum* peat, approaches being explored include the farming of *Sphagnum* for harvest as a growing medium whilst delivering co-benefits for the environment and society (Mulholland and others 2020).

4.4.4 Evidence gaps and future needs for blanket and raised bogs

We have assessed that blanket bogs and raised bogs have similar evidence gaps and futures needs so cover them together in this section for conciseness.

For all peatlands, their carbon stock is directly related to their depth. Improved accuracy in peat depth mapping continues to be an evidence need in order to increase accuracy in peat carbon stock estimates.

For emissions from ombrotrophic bogs there is a lack of data to differentiate between blanket bog and raised bog, with assumptions made that the two habitats function in a similar fashion (Smyth and others 2015). Within these categories, evidence regarding the impact of dominant vegetation type and burning management on carbon and greenhouse gas emissions is not sufficient to generate specific emission factors (Evans and others 2017). Further site-based studies in these areas would support quantification of the potential benefit for carbon and greenhouse gas emissions that restoration interventions may have on peatland habitats.

Most studies investigating peatlands tend to measure only one or two components of the carbon cycle, which is not sufficient to construct a full carbon or greenhouse gas balance for a site. The use of data generated by Evans and others (2016) in the generation of a method for reporting greenhouse gas emissions from peatlands in the UK's emissions inventory (Evans and others 2017) demonstrates how important comprehensive, site based studies are in aiding our understanding of the roles peatlands play in climate change mitigation. Further to this, field studies typically span 5 years or less making it difficult to separate the impact of land management and restoration, from other drivers such as climate or

[26] Cumbria BogLIFE+
[27] Humberhead Peatlands restoration LIFE+ project

atmospheric impact. This review found no studies that monitored the changes in peatland carbon cycling through the restoration trajectory.

Possibly due to their extent, blanket bogs make up a significant proportion of the studies investigating carbon and peatlands. However, despite this, the understanding of responses to impacts from gully blocking, peat stabilisation, burning and grazing are still variable and often based on studies from a limited number of sites. The need to revegetate bare peats with peat forming vegetation such as *Sphagnum* mosses to protect carbon stocks and reinstate sequestration is often cited, but the benefits have not yet been quantified, with the few studies that are available focussing on the established practice of using a grass crop to stabilise degraded peats.

In recent years raised bogs in England have been subject to intensive restoration projects, benefiting from investment from the EU Life+ funding programme. Their role as a carbon sink is a prominent reason for their restoration, though raised bogs are poorly represented in the scientific literature and no long-term UK based studies monitoring their recovery have been identified. Existing studies tend to focus on heavily degraded or modified sites, and there is a lack of 'intact' reference sites. Further research is required in this area to quantify the co-benefits of raised bog restoration for biodiversity and climate change mitigation.

4.5 Fens

Fen habitats are comprised of wetlands that can occur on either peat or mineral soils. Unlike ombrotrophic peatlands, fens receive water and nutrients from both groundwater and surface water, creating mineral rich growing conditions that support a more diverse range of flora and fauna compared to ombrotrophic peatlands (McBride and others 2011).

Fen habitats are widely distributed across the UK and can be found from sea level up to mountainous regions. They are highly diverse habitats, forming mosaics with other associated habitats including wet woodland, lowland heathlands and meadow (Natural England & RSPB 2020), and range in size from a few metres to hundreds of hectares. In the UK, lowland peatlands comprise approximately 325,000 ha, 194,100 ha (60 per cent of the total stock) of which is designated as cropland (Mulholland and others 2020; ONS 2019). A large majority of the lowland peat habitat designated for cropland is situated in England; approximately 182,700 ha, with the largest area of cropland situated in the East Anglian Fens (Mulholland and others 2020). Semi-natural fen and reedbed habitats now cover only around 24,000 ha in England. Within this section we review fen habitats. Peatlands under agricultural land use are discussed in section 3.4.4.

4.5.1 Carbon storage in fen peats

Fen habitats are extremely diverse and occur across the entire spectrum of wetland conditions from highly acidic and nutrient-poor through to base-rich and nutrient-rich, and in combinations of all these base and nutrient states. There appears to have been little study of the carbon storage and sequestration in these different fen types, however, despite the potentially large variation in carbon cycling between them.

Carbon stored in fen soils is comparatively less well studied than other peatland habitats in England. A report by Natural England (2010) estimated that lowland fens in which deep deposits have been maintained, store approximately 144 Mt C, compared to ‘wasted’ lowland fen, which were estimated to store 186 Mt C. It should be noted though that the extent of wasted peat is not well quantified, and this figure comes with high uncertainty. Evans and others (2016), including data from Peacock and others (2019), found a range of carbon stocks between 610 and 2,820 t C ha^{-1} in fens of contrasting land use across England and Wales, standardised to the top 50 cm of the peat, with reported depths ranging from less than 50 cm to over 380 cm (table 4.6).

There is great uncertainty regarding peat depth, as observed in a study based in the Fenlands of East Anglia (Holman 2009). Burton & Hodgson (1987) estimate that over half of the peat soils in the Fenlands are under 1 metre in depth, with deeper soils tending to be found in nature reserves or under grass in flood relief washlands. Using data from Burton & Hodgson (1987), Holman (2009) estimates the average depth of arable fen peat is 71 cm, and 79 cm under the nature reserves and washlands (figure 4.4). The large carbon stocks in the much deeper peats reported by Evans and others (2016) therefore may not be representative of fen habitats as whole.

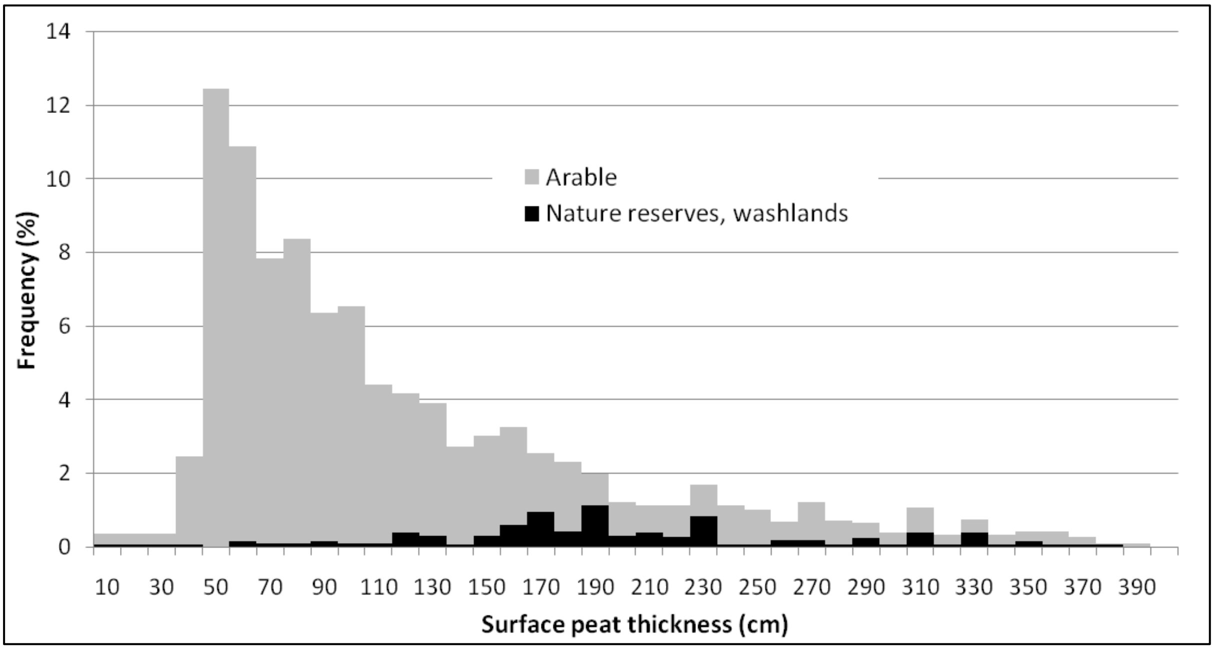


Figure 4.4 Frequency of surface peat thickness in Fenland, East Anglia (reported in Holman (2009) from Burton & Hodgson (1987)

Despite the lack of evidence on carbon stocks and cycling in fen habitats, an understanding of the geomorphological and ecological processes can be used to make predictions. For example, areas that have shallow peat and short vegetation are likely to store less carbon, compared with much deeper peats and tall vegetation which will have higher carbon storage. Higher nutrient status is also usually associated with greater vegetation height, especially when vegetation is above chest and head height (McBride and others 2011). Vegetation height will have an influence on carbon dynamics, as more CO_2 is removed from the atmosphere through photosynthesis and is subsequently stored within the plant (Evans and others 2016).

4.5.2 Carbon cycling and management interventions in fens

Semi-natural fen habitat

Fen habitats have been significantly damaged by human activities, all of which impact on their ability to store and sequester carbon. One of the largest factors affecting the ability of a fen habitat to function as a carbon sink is its water supply. Therefore, extensive drainage of surrounding landscapes, within the habitat itself and of surrounding freshwater ecosystems all impact on the natural functioning of a fen habitat. Much of England's natural fenland has been claimed for agriculture and industry throughout the last century, with most of the peatland soils used for intensive grassland or arable cropping. Furthermore, remaining fen habitats are now surrounded by agricultural land and are exposed to nutrient enriched runoff from fertiliser and manure. Fen peats not used for agriculture were cut and extracted for use in industry, although this was relatively small scale in comparison to lowland raised bogs.

The most extensive review of lowland peat systems across England and Wales is by Evans and others (2016). Full carbon budgets were measured on ten fen sites, including five considered semi-natural (as opposed to being under agricultural management). Of those, four were shown to be functioning as net carbon sinks over two annual measurement cycles, all of which were classified as semi-natural, with semi-permanent fen vegetation (table 4.6). Other sites in the study included fen peats under arable and grassland, all of which were net carbon sources (see section 3.4.4 for further discussion).

Similarities between the carbon budgets for Wicken Fen NNR and the high-nutrient Anglesey Fen site, suggested that vegetation community may be a better indicator of CO_2 balance than nutrient status. Tall vegetation such as common reed *Phragmites australis* and Great fen-sedge *Cladium mariscus* were both present at Wicken Fen and the high-nutrient Anglesey Fen site, whereas the low-nutrient Anglesey Fen site was more characteristic of short fen vegetation species. The three semi-natural sites were subject to minimal management over the study period. However, it was also noted that a period of drought experienced during the summer of 2013 caused severe water table drawdown at the Wicken Fen site, causing the site to switch from a sink to a temporary source of CO_2. Overall, Evans and others (2016) stated that the main control on CO_2 emissions is mean water table depth, estimating that every 10 cm increase in water table depth resulted in an average increase of CO_2 emissions by around 4 t CO_2e ha^{-1} y^{-1}. This also highlights the importance of site conditions in determining whether a system may be performing as a source or a sink of CO_2; the same fen type or vegetation with different water tables may differ in their carbon cycling throughout the year.

Table 4.6 Carbon stocks and sequestration rates for fen habitats in England and Wales, taken from Evans and others 2016)[1] and Peacock and others (2019)[2]

Site	Land-use	Depth (cm)	C storage (t C ha^{-1})	C Flux (t CO2e ha^{-1} y^{-1})
Wicken Fen, East Anglia[1]	Low-nutrient, semi-natural	380 cm	2,820	-3.29
Cors Erddreiniog, Anglesey[1]	High-nutrient, semi-natural	315 cm	2,170	-3.66
Cors Erddreiniog, Anglesey[1]	Low-nutrient, semi-natural	280 cm	1,370	-1.75
Norfolk Broads[1]	High-nutrient, semi-natural	> 300 cm	1,830	-10.31
Norfolk Broads[1]	Low-nutrient, semi-natural	> 300 cm	1,900	+1.46
Wicken Fen, East Anglia[2] (data used in Evans and others 2016)	Semi-natural, conservation managed (data used in Evans and others 2016)	380	2,820	-3.81
Baker's Fen (within Wicken Fen), East Anglia[2] (data used in Evans and others 2016)	Converted grassland from cropland	40	610	+4.88

Water table depth also impacted on CH_4 fluxes. Methane emissions were not detected at sites where water table depth was below 25 cm. However, every 1 cm the water table rose above the 25 cm threshold resulted in an estimated 0.2 t CO_2e ha^{-1} y^{-1}. This increase continued where water tables rose above the soil surface, meaning that water-logged sites are potentially important sources of CH_4. The two sites with the largest CH_4 emissions were semi-natural, low and high nutrient sites in East Anglia both of which had very shallow water tables and were frequently water-logged. Despite some offsetting by CH_4 emissions, conservation managed fens were still shown to be among the most effective carbon sinks.

More recently, Peacock and others (2019) reported full carbon budgets for a conservation managed fen on deep peat and a site formally used for arable crops and subsequently converted to grazed grassland, on shallow peat (40 cm). Both sites were again situated within the Wicken Fen NNR in East Anglia. The conservation managed fen was found to be a net carbon sink, sequestering 104 g C m^{-2} y^{-1} (3.81 t CO_2e ha^{-1} y^{-1}). In contrast, the converted grassland site was found to be a net carbon source, emitting 133 g C m^{-2} y^{-1} (4.88 t CO_2e ha^{-1} y^{-1}), with the largest loss occurring during periods of low water table depth leading to exposed soil, primarily in summer. Methane fluxes were apportioned by the dominant vegetation type. The conservation managed fen was dominated by common reed *Phragmites australis* and *Cladium mariscus* and was considered a net CH_4 source. The converted grassland site showed vegetation effects on CH_4 flux, with creeping bentgrass *Agrostis stolonifera* a net CH_4 sink and hard rush *Juncus inflexus* a net source. Assuming equal cover of vegetation across the site, the overall CH_4 flux was near-zero*.*

Figure 4.6 Fen vegetation at Woodwalton National Nature Reserve. ©Natural England/Peter Wakely

Methane emissions reported for fens are likely to be underestimated; most studies only report emissions from vegetation and not ebullition fluxes (bubbles of CH_4 released from the soil or sediment). A recent study by Stanley and others (2019) measured CH_4 emissions from two sites; Sutton Fen and Strumpshaw Fen in the Norfolk Broads, both sites under conservation management. Both sites are deep peat fens dominated by *Phragmites australis* but have contrasting nutrient status; Sutton Fen was classed as low nutrient and Strumpshaw Fen high nutrient. Fluxes were comparable across both sites, but ebullition flux accounted for over 38 per cent of total CH_4 emissions over spring and summer. Soil temperature was found to be the primary control of CH_4 ebullition flux, but water level was also a factor, with increases in ebullition flux when the water level was within 10 cm of the peat surface. In contrast, decreases in ebullition flux were associated with increasing plant cover. The high nutrient Strumpshaw Fen site had significantly higher aboveground plant biomass, resulting in enhanced CH_4 oxidation in the peat. Management of water levels and vegetation in floodplain fens could therefore have the potential to alter the total CH_4 flux to the atmosphere. Furthermore, reed cutting practices could reduce plant-derived CH_4 fluxes but could increase ebullition fluxes, as reduced biomass would lead to reduced CH_4 oxidation belowground by rhizomes.

Reedbeds

Reedbeds are found across the UK but are most common in the lowland fen areas of East Anglia. Most of the UK reedbed habitat is dominated by the common reed species *Phragmites australis* and are an especially important habitat for rare bird species, some of which are dependent solely on this habitat. Reedbeds cover around 5,000 ha of land in the UK, although relatively few sites are greater than 20 ha (Natural England & RSPB 2020). As well as providing vital habitat, reedbeds are of commercial importance and research on their potential use as a biofuel is on-going. Reedbeds are a transitional fen habitat, without management they can dry out through the build-up of litter and develop into lowland fen and eventually wet woodland.

Reed beds are known carbon sinks and reed habitats in good condition have high standing biomass values and therefore high annual primary productivity (Brix and others 2001). However, there is limited evidence available to enable quantification of their contribution to the overall wetland carbon sink in the UK. The presence of reedbeds within fen landscapes and surrounding ditch networks is frequently reported (Evans and others 2016; Peacock and others 2019) but their standing carbon stocks are not commonly specified. A 2009 report commissioned by Somerset County Council on carbon storage and sequestration in the Somerset Levels, England (Brown 2009), quoted an organic carbon flux of between 5 and 20 t C ha^{-1} y^{-1}. However, it is unclear of the origin of these values as the original publication could not be found. Some evidence suggests that tall fens, including reedbeds, may have greater climate mitigation benefits than wetter short fens, potentially acting as net GHG sinks under optimal management (Evans and others 2016). Evidence on the carbon stocks and sequestration rates of reedbed habitats outside of the Fens was not found.

Drainage ditches

Fens are associated with networks of drainage ditches, deep channels dug to move water away from arable farmland. Drainage ditches are generally considered sources of carbon, especially CH_4 and to a lesser extent, CO_2, although emissions data are scarce and show high spatial and temporal variability (Peacock and others 2017). Some ditches containing emergent vegetation have been observed as small CO_2 fixers (Peacock and others 2017; Vermaat and others 2011). Ditches with standing or slow flowing water allow for the growth of aquatic plants, which trap and allow settling of sediment. This could contribute to a small pool of carbon, although high nutrient concentrations and low dissolved oxygen promotes CH_4 production. Regular removal of sediment through dredging is carried out to maintain drainage, meaning any sedimentary carbon stored at the bottom of the ditch is likely to short-lived and released back into the atmosphere upon disturbance.

Not all surveys include ditch emissions in their reporting. However, both greenhouse gas budgets reported by Evans and others (2016) and Peacock and others (2019) (as described above) included ditch emissions in their measurements. Evans and others (2016) found that CH_4 emissions from ditches were highly variable in time and space, ranging from 0–12.3 g m^{-2} y^{-1} (0–0.51 t C ha^{-1} y^{-1}). Higher fluxes were often recorded during summer and ditch characteristics, including whether they were incised on mineral soil and contained vegetation, tended to be associated with smaller fluxes. A significant contribution to overall emissions were made at sites with larger areas of open water. Fluxes of CO_2 were even more variable, ranging from a sink to a source (-0.12–1.34 t CO_2 ha^{-1} y^{-1}).

Peacock and others (2019) reported higher ditch CO_2 and CH_4 fluxes at the conservation managed site compared to the converted grassland. Ditch emissions for the converted grassland site were low and are thought to be linked to low water tables and low organic carbon content. However, measurements were made using only two gas flux chambers per site. More intensive sampling was carried out in an earlier study by Peacock and others (2017). The three sites chosen represented different management intensities including semi-natural fen, cropland and cropland restored to low-intensity grassland (the same site used in Peacock and others 2019). All three fen sites emitted greenhouse gases to the atmosphere but showed extensive variation both seasonally and within site. Annual CH_4 fluxes from the three ditches were 0.39, 0.18 and 0.27 t CH_4 ha^{-1} y^{-1} for semi-natural, restored grassland and cropland respectively. For CO_2, annual fluxes were 11, 1.7 and 14.4 t CO_2 ha^{-1} y^{-1} for semi-natural, grassland and cropland respectively. No significant differences were found between sites for CH_4 fluxes, but CO_2 fluxes were significantly higher at the cropland compared to the grassland and semi-natural sites. Measured CH_4 fluxes from fen ditches are much higher than those reported for upland blanket bog (Cooper and others 2014, see blanket bog section), highlighting the effects of nutrient status on ditch emissions.

4.5.3 Climate change and other interactions

Semi-natural lowland Fens are highly sensitive to climate change and reedbeds have medium sensitivity (Natural England and RSPB 2020). Given their dependence upon local hydrological conditions, lowland fens are particularly sensitive to any changes in the quality and quantity of water supply, which is likely to change significantly with climate change. Reedbeds in particular, need above or near surface water tables year-round. However, damaging modifications such as drainage will have a more immediate impact than climate change. Longer dry spells, particularly during the summer risks fen habitats drying out, potentially leading to a loss of vegetation species as well as promoting soil oxidation and release of CO_2 to the atmosphere. Changes in the frequency and intensity of storms may lead to more frequent inundation in the lowlands, which may have links to elevated CO_2 and CH_4 emissions; Evans and others (2016) observed higher CH_4 fluxes to the atmosphere in water-logged sites. Increased mean temperatures are likely to increase the growing season, requiring altered management, especially for cutting regimes and livestock density.

Increased nutrient loads to drainage ditches, for example through storm inputs, could lead to eutrophication and increased emissions of CO_2 and CH_4 as productivity increases. Drainage ditches could also dry out during longer periods of drought, exposing stored sediments to the atmosphere and increasing CO_2 emissions. Future management of ditches should prioritise reduction of the application of fertilisers and manure, as well as preventing drying out by allowing the colonisation of fringing plants that provide shade, and wetland plant species that retain moisture through dry spells. However, given the lack of evidence available, mitigation approaches for drainage ditches are speculative and further research is necessary.

Furthermore, future demand on resources is likely to increase; including for both water usage and food production. This makes fen habitats particularly vulnerable and may lead to further loss of habitat through land-use change and diffuse water pollution from neighbouring farmland. Fens are typically phosphorus limited, but nitrogen enrichment, predominantly from fertilisers could be a contributing factor to the further loss of carbon from lowland fens (McBride and others 2011). Therefore, minimising adverse impacts from the management of adjacent habitats is likely to be the most important factor in their ability to function as a carbon sink.

4.5.4 Evidence gaps and future needs

Most of the literature available is concentrated around lowland sites within the East Anglian fens, which contains the largest contiguous area of fen peatland in the UK (Morrison and others 2013; Baird and others 2009). Even within the studies reported, high spatial and temporal variability is evident and carbon dynamics in one site may not reflect that in another. Therefore, further work is necessary to create an evidence base which is more representative of the diversity of fen habitats across England and the rest of the UK.

Current knowledge on the extent of fen habitats is also lacking, especially in the uplands. In the literature, fen habitats tend to be aggregated into a single category as real extents are not available for all of the country. This makes it difficult to accurately quantify the maximum carbon storage and sequestration potential of fen habitats on the whole. Some studies are beginning to document the scale and richness of these habitats (eg Callaghan 2012; Tratt and others 2012; Jerram 2015) but more research is needed to assess their true climate change mitigation potential.

Conservation managed fen habitats are considerable sinks of carbon, but the balance between climate change mitigation, biodiversity and land use pressures needs to be met and the trade-offs between different management approaches needs to be further understood. Quantification of the carbon balance should also include associated drainage ditches, which have the potential to offset any sink achieved by the fen habitat itself.

5 Rivers, lakes and wetland habitats

5.1 Chapter summary and key messages

Freshwater habitats are extremely diverse, ranging from peatland streams, peatland pools, large rivers, lakes, ponds and floodplains**.** Freshwaters cover around 5444 km^2 of the total UK area, with floodplains covering 16,000 km^2. Rivers and streams are important drivers of the carbon cycle, transporting and depositing carbon rich material downstream, depositing on floodplains, in estuaries and out to sea. Standing waters, such as lakes, have the capacity to bury considerable amounts of carbon long-term. When functioning naturally, freshwater habitats mostly function as carbon sinks, but centuries of human modification, such as drainage, construction and pollution can cause these systems to function as atmospheric sources of CO_2.

Key messages

- Floodplains are important carbon sinks and have the potential to store significant amounts of carbon, but have been subject to intensive management. However, there is a lack of evidence in the English / UK context, resulting in a low confidence assessment (Table 5.1 & 5.2).
- Streams draining degraded peatlands have higher particulate organic carbon fluxes than pristine or restored peatlands. In addition, gas flux measurements of headwater streams draining peatlands show that they are small sources of carbon dioxide and methane to the atmosphere.
- Lowland streams, mostly impacted by agriculture, are atmospheric sources of carbon dioxide and methane. Fine, organic matter rich sediments enhance greenhouse gas production and are expected to increase due to land use change and increased weathering rates.
- Chalk bed streams seem to be more sensitive to organic matter inputs, potentially causing hotspots of carbon dioxide and methane in areas that are generally considered to have good ecological status.
- Ponds, if well managed, could be carbon sinks and could have high carbon burial rates. Creating new ponds and restoring those neglected could help sequester more carbon and be readily integrated with other land uses. However, ponds prone to drying out can switch from carbon sinks to carbon sources and ponds prone to high nutrient input and low oxygen can result in higher methane emissions. The use of vegetation, such as *Sphagnum* mosses, can help protect carbon rich sediments from exposure to the atmosphere.
- Very little information exists for upland standing waters such as lakes, tarns and peatland pools. There is a need for further information on these systems to understand the impacts of land management on their carbon cycling.

- Lowland lakes are the most productive and can act as net carbon sinks. Higher nutrient inputs can reverse these systems, making them net sources of carbon to the atmosphere.
- Freshwater habitats are intimately linked to their surrounding catchments. Land management therefore directly impacts the source-sink dynamics of freshwaters. Consideration of the mitigation potential of freshwater habitats should therefore be included in land-based restoration efforts.
- Tables 5.1 and 5.2 summarise the carbon storage and flux values identified as representative for habitats reported in this chapter.

Table 5.1 Summary of carbon storage in river, lake and wetland habitats, as based on the review of literature. Only floodplains are reported; no other data on carbon stocks for the other habitats could be found.

Habitat Description	**Total Habitat Carbon Storage**			**Notes** *Inc. References used*
	t C/ha	**Sediment depth (cm)**	**Confidence**	
			[High, Medium, Low]	
Floodplains	109.4	10	Low	Unpublished survey at North Meadow Cricklade NNR

Table 5.2 Summary of carbon flux in river, lake and wetland habitats, as based on the review of literature. DOC stands for dissolved organic carbon and POC stands for particulate organic carbon

Habitat Description	**Annual carbon accumulation / loss for the habitat**									**References**
	t CO_2e ha^{-1} Y^{-1}	**Range (if possible)**	**t CO_2e ha^{-1} Y^{-1}**	**Range (if possible)**	**t CO_2e ha^{-1} Y^{-1}**	**Range (if possible)**	**t CO_2e ha^{-1} Y^{-1}**	**Range (if possible)**	**Confidence**	***Inc. notes***
	Land – freshwater flux		**Freshwater flux (in stream transport)**		**Freshwater – atmosphere flux**		**Carbon accumulation (burial rate)**		[High, Medium, Low]	
Streams draining peat	No data	No data	+2.32[ab]	+0.73[a] to +3.92[b]	+141.9[c-e]	+0.52[c] to +424.34[d]	N/A	N/A	Low	[a]Pawson and others (2007); [b]Worrall and others (2003); [c]Hope and others (2001); [d]Billett & Harvey (2013); [e]Billett and others (2015)
Lowland Rivers & Streams	+1728	No data	+531	No data	+947	No data	N/A	N/A	Low	Worrall and others (2016). Note: Values presented here are totals, no per hectare values were given.
Chalk bed streams	No data	No data	No data	No data	+8.9	+7.15 to +10.63	No data	No data	Low	Romeijn and others (2019)
Floodplains	No data	No data	No data	No data	No data	No data	-3.365	-2.54 to -4.19	Low	Walling and others (2006)
Lakes	No data	No data	No data	No data	No data	No data	-7.1[a-c]	-0.46 to -23.6	Low	[a]Anderson and others (2014); [b]Casper and others (2000); [c]Scott (2014)
Reservoirs	+1021.17 (POC) +227.33 (DOC)	+432.67 to +1609.67 (POC) +128.33 to +326.33 (DOC)	No data	No data	+386.835	+66 to +707.67	-636.165	-407.00 to-865.33	Low	Stimson and others (2017) Note: Values presented here are totals, no per hectare values were given.
Peatland pools	No data	No data	No data	No data	No data	No data	No data	No data	No data	No data
Ponds	No data	No data	No data	No data	+6.69[a]	-17.53[a] to +30.91[a]	-16.12[b-d]	-2.90[b] to -29.33[c]	Low	[a]Gilbert and others (2016); [b]Taylor and others (2019); [b]Ockenden and others (2014); [d]Gilbert and others (2014);

5.2 Rivers, lakes and wetland habitats

The movement of water through our landscapes drives the carbon cycle, providing a continuum of transport between terrestrial and coastal / marine environments. This makes freshwater habitats particularly important in the effort to reduce greenhouse gas emissions both on land and at sea, as well as maintaining the provision of ecosystem services. However, the loss of organic carbon to the atmosphere through inland waters is not well understood, in part due to the highly dynamic nature of freshwater. Global estimates suggest that around 2.7 billion tonnes of terrestrial carbon enter freshwaters each year, 50 per cent of which returns to the atmosphere as CO_2 (Biddanda 2017).

The UK is home to a diverse range of freshwaters, from headwater streams, through to lowland rivers, large lakes and small ponds (Mainstone and others 2016). Freshwater habitats have been subject to centuries of human modification. For example, river flows have been engineered to benefit food production, leading to drainage of connected wetlands for crop production and land development. This disruption of natural habitats limits their ability to function as effective carbon sinks and restoring natural ecosystem function could co-benefit both biodiversity (Addy and others 2016) and climate change mitigation. However, there is limited evidence on the extent that nature-based solutions can play for climate change mitigation in freshwater habitats.

5.2.1 Carbon cycling in freshwaters

Naturally functioning freshwaters generate their own sources of carbon through the net primary production of aquatic plants and algae. In addition, freshwater ecosystems receive carbon from land and naturally functioning catchments, contributing to biodiversity and carbon storage / sequestration. Rivers carry multiple forms of terrestrial carbon, but the dominant forms are particulate organic carbon (POC) and dissolved organic carbon (DOC) occur through erosion and overland flow, and direct input from trees and other riparian vegetation. Inorganic forms of carbon also enter river systems from the weathering of bedrock and mineral precipitation.

Rivers can also deposit large amounts of sediment in floodplain soils that then act as a carbon sink. Their carbon storage potential is dependent on geologic, hydrologic and geomorphic characteristics of the associated river system (Swinnen and others 2020). This, when coupled with local variables in climate, topography, vegetation cover and land use leads to high variability between sites. The dynamic nature of these processes means that deposited carbon is at risk from loss from erosion and disturbance and may be lost via respiration during fluvial transport or reach the ocean where again it may be buried and stored. The cool temperate climate of the UK can favour high organic carbon concentrations in floodplain sediments, particularly in catchments fed by peatland headwaters.

In shallow standing waters, most respired CO_2 will occur within the surface mixed layer, allowing gas exchange back to the atmosphere. This also occurs in deeper, stratified lakes, but CO_2 in the hypolimnion (the lower layer of water) may remain within the water column for long periods of time. In all lakes, the burial of particulate organic carbon is an important carbon sink, which may be stored for centuries to millennia. Upwelling and disturbance can re-mobilise available carbon back into the water column where it can be mineralised and respired. Productivity rates in standing waters vary widely, due to regional and climatic differences.

Impacts of human modification on the freshwater carbon cycle

Organic matter inputs from soils into rivers and streams was previously thought to be a net carbon sink, as eroded soil organic carbon lost from land is eventually replaced, while eroded material is stored by downstream burial in the estuary or transported to shelf seas (Worrall and others 2016). More recent evidence has shown that carbon transported through rivers to the oceans is only a small fraction of that entering rivers from land (Aufdenkampe and others 2011). Therefore, rivers are now considered a net source of CO_2, as most of the terrestrially derived organic matter is mineralised during transport. Emissions of CO_2 from rivers to the atmosphere are therefore likely to change with changing land use, and the flux of carbon from land to freshwater ecosystems has already increased worldwide due to human pressures on land (Butman and others 2015).

Physical modifications of rivers and streams further exacerbates their ability to function as carbon sinks. Weirs, dams and other-in channel structures create a build up of fine, nutrient rich sediments, increasing productivity and generating sediment anoxia (low oxygen), leading to hotspots of CO_2 and CH_4. This is likely to cause river and stream greenhouse gas emissions to be higher than they would be if functioning naturally. Furthermore, human modifications on floodplains, such as drainage, flood defences and stream diversion stop inundation, thereby eliminating the ability of the floodplain to function as a natural carbon sink. However, there is a general lack of evidence of 'pristine' systems to use as a baseline reference.

Standing waters enriched with nutrients are actually considered to be the most productive, resulting in generally higher carbon sequestration rates. This is because nitrogen and phosphorus inputs from agriculture and sewage effluent increase primary productivity. This, to a certain point, can lead to a decrease in the atmospheric carbon flux and actually increase carbon sequestration. However, the negative consequence of increased productivity is the depletion of oxygen supply. When high nutrient inputs occur, eutrophication depletes oxygen supply and increases anaerobic activity, resulting in an increase in methane (CH_4) emissions. Some of this will be oxidised and released as CO_2 and some will reach the atmosphere as CH_4. Lakes have been shown to contribute approximately 70 per cent of freshwater CH_4 emissions globally, which is disproportionately large compared to their extent. A recent study by Sanches and others (2019) showed that the diffusive flux of CH_4 in lakes with total phosphorus concentrations higher than 0.9 mol L^{-1} was five times higher than in lakes with total phosphorus concentration of less than 0.9 mol L^{-1}. Therefore, elevated nutrient concentrations in lowland standing waters are likely to be an important driver of annual freshwater greenhouse gas emissions.

5.3 Rivers and streams

Rivers and streams cover approximately 0.86 per cent of the total land area of the UK (approximately 1,940 km^2). Rivers and streams in the UK are highly diverse, owing to the nature of the catchment geology and land use (Photo 5.1). Therefore, high regional variability in carbon and greenhouse gas fluxes is to be expected. Naturally occurring features such as changes in size, flow, geology, pH, groundwater and biodiversity will all affect carbon cycling. High gradient streams with low baseflow and higher peak flows have lower potential for carbon storage, but high capacity for particulate and dissolved organic carbon transport during high flow events. In contrast, low-gradient streams can accumulate carbon through the settling of sediments during periods of low or slow flow. However, this is thought to be a relatively short-term carbon store due to seasonal changes in flow and storm events causing re-suspension. Some fluvial systems, such as chalk streams (see section 5.2.1) can also be sources of methane, although the flux is generally thought to be smaller than the CO_2 flux. Where nutrient rich sediments can settle, such as in areas with dense vegetation, hotspots of methane may also occur.

Photo 5.1 Examples of rivers and streams found across England: Clapper bridge over River Teign. Dartmoor National Park © Natural England / Paul Glendell (Left); River Beult – Kent © Natural England / Peter Wakely (Centre); River Clun © Natural England / Jenny Wheeldon (Right).

5.3.1 Carbon storage and sequestration in Rivers and streams

Streams draining peat

Streams draining peat have been shown to be consistently supersaturated with CO_2 across the northern hemisphere and have the potential to act as sources of atmospheric CO_2 and CH_4 (Billet & Harvey 2013). Systems prone to low flows and stagnation in the summer months are also likely to have elevated CO_2 production. However, direct measurements of greenhouse gas fluxes to the atmosphere are lacking, in part due to the difficulty in measuring fluxes from rivers with changing topography and discharge (Billet & Harvey 2013).

An alternative way to determine emissions to the atmosphere is to measure the difference between aquatic CO_2 and CH_4 concentrations at the source and the outlet of the stream. Hope and others (2001) at Brocky Burn, Scotland, and Dawson and others (2002) at Afon Hafren, Wales showed that higher concentrations are directly related to peat distribution. Lower concentrations occurred downstream due to outgassing and a change from peat soils to humic podzols, which contain significantly lower soil CO_2 concentrations. Hope and others (2001) also estimated a total flux of carbon to the atmosphere as 14.1 g C m^{-2} y^{-1} (0.141 t C ha^{-1} y^{-1}). A later study by Billet & Harvey (2013) measured atmospheric CO_2 and CH_4 fluxes in headwater streams draining six UK peatlands. Mean CO_2 and CH_4 evasion (degassing to the atmosphere) rates were 367 µg CO_2 m^{-2} S^{-1} (115.73 t CO_2 ha^{-1} y^{-1}) and 1.45 µg CH_4 m^{-2} S^{-1} (0.46 t CH_4 ha^{-1} y^{-1}) respectively. A later study by the same authors scaled up evasion rates from the same peatland streams, resulting in an average catchment scale CO_2 flux of 23.3 ± 6.9 g C m^{-1} y^{-1} (0.23 t C ha^{-1} y^{-1}), comparable to the downstream DOC flux of 29.1 ± 12.9 g C m^{-2} y^{-1} (0.291 t C ha^{-1} y^{-1}), with methane emissions undetectable at the outlet (Billet and others 2015). Upscaled to a nationwide estimate of CO_2 flux from peatland streams resulted in an emission source of 0.57 Mt C y^{-1} to the atmosphere.

During intense storm events in upland areas, sediment loads will increase, potentially increasing CO_2 fluxes downstream. Pawson and others (2007) found that DOC and POC increased with discharge during storm events in a heavily eroded peat catchment in the southern Pennines. Total organic carbon export was 40.8 t C, or 107 g C m^{-2} y^{-1} (1.07 t C ha^{-1} y^{-1}), apportioned as 82 per cent POC and 18 per cent DOC. Moorhouse National Nature Reserve (NNR) is another example of a degraded system, but recent revegetation has seen some reductions in erosion rates. Worrall and others (2003) still found POC to be the dominant flux at 68 per cent of the total fluvial carbon flux (19.9 g C m^{-2} y^{-1} or 0.199 t C ha^{-1} y^{-1}), highlighting the need for upland restoration to prevent terrestrial losses to freshwater systems and subsequent losses to the atmosphere.

Dissolved organic carbon dynamics have been extensively studied and can have implications on downstream CO_2 emissions (Evans and others 2005). Worrall and others (2020) found that river DOC concentrations have been increasing countrywide, using data collected over a 46 year period (1974−2019). The largest concentrations were recorded in northern areas of the UK, in areas draining upland peat with little urbanisation. In contrast, large declines in DOC concentration were seen in the south and east of the UK, where improvements in wastewater treatment may have contributed to reduced DOC export in urban dominated catchments. A new study on the controls of riverine DOC export in the UK (Williamson and others 2021) showed that the main factor influencing DOC export is upland conifer plantation forestry, which is estimated to have raised the overall DOC export by 0.168 Tg C y^{-1} (168,000 Mt C y^{-1}), further highlighting the need to integrate land management and freshwaters in the context of carbon storage and sequestration.

Lowland Rivers and Streams

The largest dataset of greenhouse gas emissions in UK rivers is that of Worrall and others (2016). Here, fluxes of particulate organic matter from 80 catchments were compiled, with land uses ranging from arable to grassland and urban. Erosion of soil organic matter into rivers was 1,728 kt y^{-1}, of which 947 kt CO_2e y^{-1} was emitted to the atmosphere (Worrall and others 2016; Figure 3). In estuaries, flux to the atmosphere was 1,107 kt CO_2e y^{-1}. Overall, when the individual greenhouse gases were considered, CO_2 represented 74 per cent of the greenhouse gas warming potential and CH_4 represented 3.8 per cent. When all greenhouse gas fluxes shown in Figure 3 of Worrall and others (2016) are considered, one tonne of POC entering rivers gives a median emission factor of 5.5 t CO_2e y^{-1}. Therefore, erosion of particulate organic matter from the surrounding catchment represents a greenhouse gas source to the atmosphere, through the fluvial network. The authors state that erosion could only ever represent a carbon sink if 'replacement' (re-accumulation from atmosphere to land) was high and erosion rate low.

Chalk bed streams

Studies on greenhouse gas production from stream sediments indicate that chalk streams may be more sensitive to nutrient inputs and warming. Romeijn and others (2019) measured CO_2 and CH_4 production from sediments in two agriculturally impacted streams (River Lambourn and River Tern). Chalk sediment with the highest organic matter (OM) content had the highest CO_2 production, with a mean of 33.40 ± 10.55 mg CO_2 m^{-2} hr^{-1} (2.9 t CO_2 ha^{-1} y^{-1}). This was around 48 per cent higher than the high OM sandstone sediment, which produced 22.53 ± 7.72 mg CO_2 m^{-2} hr^{-1} (1.95 t CO_2 ha^{-1} y^{-1}). The mean CO_2 production rate for all sediments combined was 13.40 ± 12.72 mg CO_2 m^{-2} hr^{-1} (1.17 t CO_2 ha^{-1} y^{-1}), which equals an annual rate of 117.36 mg CO_2 m^{-2} y^{-1} (10.18 t CO_2 ha^{-1} y^{-1}). Scaled up to the rest of the UK, Romeijn and others (2019) estimated a total CO_2 production from streambed sediments as 1.02 t C km^{-2} y^{-1} (0.01 t C ha^{-1} y^{-1}) and suggested that CO_2 fluxes from the streambed could account for between 1.4 per cent and 86 per cent of total CO_2 fluxes from UK streams and rivers, depending on OM content. The estimated UK CO_2 flux from Romeijn and others (2019) is similar to previous estimates of between of between 1.4 and 2.9 t C km^{-2} y^{-1} (0.01–0.03 t C ha^{-1} y^{-1}) (Worrall & Lancaster 2005).

The highest CH_4 production was also found in the chalk sediment with the highest organic matter content, at 0.4778 mg CH_4 m^{-2} hr^{-1} (0.04 t CH_4 ha^{-1} y^{-1}) (Romeijn and others 2019). This was 656 per cent more than the sandstone sediment with the highest organic matter content, which produced only 0.0632 mg CH_4 m^{-2} hr^{-1} (0.005 t CH_4 ha^{-1} y^{-1}). Both the high OM chalk and sediment samples accounted for 95 per cent and 100 per cent of the total CH_4 production respectively. When CH_4 production for all sediments was combined, the mean flux was between 0 and 0.48 mg CH_4 m^{-2} hr^{-1} (0 - 0.046 t CH_4 ha^{-1} y^{-1}) scaled up to a mean annual flux of 0.83 g CH_4 m^{-2} y^{-1} (0.072 t CH_4 ha^{-1} y^{-1}). Using the mean flux from the high OM chalk sediment, Romeijn and others (2019) estimated a total maximum CH_4 flux of 3.0×10^{-5} Tg CH_4 y^{-1} (30 t CH_4 y^{-1}).

An earlier study by Sanders and others (2007) estimated the total methane flux from UK chalk streams as 3.2×10^{-6} Tg CH_4 y^{-1} (3.2 t CH_4 y^{-1}) using in situ measurements; around one order of magnitude smaller than estimates made by Romeijn and others (2019). The total methane flux estimated by Sanders and others (2007) had a strong seasonal variability linked to the growth of *ranunculus* spp water crowfoot in spring and summer. Estimates were based in the River Frome in Dorset, where > 90 per cent of the CH_4 flux was dominated by transport through *ranunculus* spp stems. No CH_4 emissions from the streambed were detected in winter. In the winter months, eroded material is likely to remain mobile and potentially mineralised to CO_2 or transported and deposited into estuarine and coastal areas.

5.4 Floodplains

Floodplains lie adjacent to river channels that occupy valley bottoms (Photo 5.2). Soils are typically alluvium or silt deposited by floods that lie over coarser deposits such as gravel. Floodplains can be very variable systems due to having both terrestrial and freshwater components (Lawson and others 2018) and form a complex and dynamic mosaic of habitats including fen, bog, swamp, wet woodlands and drier habitats distributed according to natural hydrological pathways and floodplain microtopography. Floodplains may also contain peat soils, particularly around upwelling areas and floodplain fringes receiving water from valley spring lines. These areas have higher water tables than non-peat soils through the summer months (Rothero and others 2016).

Photo 5.2 Cuckmere Valley - East Sussex © Natural England / Peter Wakely.

Due to the seasonal deposition of nutrients floodplains have historically been valued as an important part of agricultural systems, frequently being managed and grazed as hay meadows. Wetland habitats such as fens, wet woodland and swamps would have been found in areas where high-water levels persisted into the summer months and may have been utilised for their biomass products such as willow, reed and straw (Lawson and others 2018). Flood alleviation and drainage has meant more intensive land management approaches, and their biodiversity and cultural value has been replaced with more intensive agriculture and urban development. However, floodplains and their restoration are again receiving considerable attention as a method to regulate water flow and reduce the impact of flooding, as well as restore their ecological function (Environment Agency, 2020).

In England and Wales, floodplains cover an area over 1.6 million hectares (Maltby and others 2011). Almost 70 per cent is under intensive agricultural land use such arable crops, horticulture and intensive grassland. Only 11 per cent support semi–natural habitats, with land coverage of species rich grassland and alluvial forest and bog woodland covering 3,000 ha and 8,750 ha respectively (Lawson and others 2018). This move to modified land uses means at least 42 per cent of floodplains in the UK are no longer connected to river systems (though this varies between rivers) and no longer function hydrologically as a floodplain. The situation in England is particularly severe, with intensive agriculture coverage of floodplains increasing from 38 per cent to 64 per cent between 1990 and 2015, and a near ubiquitous loss of natural floodplain functioning (Entwistle and others 2019).

5.4.1 Carbon storage and sequestration in floodplains

Floodplains cannot be defined by a single habitat and naturally consist of complex mosaics that have been replaced by a mixture of different land covers ranging from intensive agriculture and urban development to semi–natural vegetation. Alluvial soils form from a succession of sedimentation and erosional processes, as well as soil formation in situ, and are highly variable in both space and time (Bullinger-Weber and others 2014). While the role of other wetlands, such as bogs and fens, are recognised for their importance regarding carbon storage the role of flood plains has not been as comprehensively quantified. As a result, it is difficult to define a representative value for their role in carbon storage and sequestration and more information is needed in both natural and human modified floodplain systems.

The natural system of deposition and erosion on riverine floodplains means soil formation is continually reset to early phases. Organic carbon accumulates at initially very high rates and high carbon sequestration rates relative to other habitats are sustained over long periods. Zehetner and others (2009) working on floodplains along the Danube, Austria, report rapid carbon accumulation of up to 100 g m^{-2} y^{-1} (1 t C ha^{-1} y^{-1}) over the first century of soil formation, declining to 8–18 g C m^{-2} y^{-1} (0.08–0.18 t C ha^{-1} y^{-1}) at 300–600 years. While this decrease was exponential, the long–term rates reported are still greater than many other terrestrial habitats. Mean carbon storage across 78 transects in the mountain headwater floodplain of the River Dee, Scotland are reported as 323.27 ± 12.58 t C ha^{-1}, with higher stocks reported in lower energy systems (Swinnen and others 2020). Cierjacks and others (2011) also report that highly dynamic locations on Danube floodplains, indicated by higher stem numbers, greater understory vegetation cover, lower mean stem diameter and lower canopy cover had significantly lower concentrations of soil organic C and lower total organic C stocks.

In their study of six river floodplains in Southern England, Walling and others (2006) observed carbon sequestration rates ranging between 69.2 and 114.3 g m^{-2} y^{-1} (0.692 – 1.143 t C ha^{-1} y^{-1}), concluding that floodplains of British rivers are important carbon sinks but could not explain the variability observed in the data. A small survey of floodplain soils at Cricklade National Nature Reserve report carbon stocks of 109.4 t C ha^{-1} in the top 10 cm, indicating the potential of floodplains to hold greater carbon stocks than other similar land cover, such as semi-natural grassland (Floodplain Meadows Partnership 2018; unpublished). Whilst floodplains may capture depositional carbon it is important to be aware of other greenhouse gas pathways. Flooding of intensively managed grasslands may increase fluxes from the potent greenhouse gases nitrous oxide and methane from agricultural land. Nitrous oxide emissions may peak during the wet-dry cycle of a flood event due to high nitrogen availability and as nitrous oxide emissions are exacerbated by moderately anaerobic conditions. Standing water can lead to increased methane emissions due to the lack of oxygen in saturated soils (Sánchez-Rodríguez and others 2019).

The land use and its manipulation of floodplain dynamics is a strong determinant of carbon storage potential. Cropland on alluvial soils along the Danube was reported as having significantly lower organic carbon and microbial biomass compared with grassland and forest sites, with the authors suggesting that cultivation of floodplain soils may ‘annihilate’ their high carbon sequestration potential (Zehetner and others 2009). Land use was found to have a stronger influence on carbon stocks than the age of the soil. The response of floodplains to restoration has been observed to be variable, and dependent on the initial land use. A study of Swiss floodplain soils found restoration from embanked mature forest to pioneer vegetation (herbaceous and bush communities) resulted in a decrease in carbon stock (82.7 to 20.7 t C ha^{-1}), whilst restoration from embanked grassland to pioneer vegetation had no significant change (10.5 to 17.1 t C ha^{-1}) (Bullinger-Weber and others 2014).

5.5 Standing Waters

Inland waters cover approximately 3,504 km^2 (approximately 1.4 per cent) of the UK, 675 km^2 of which are in England (JNCC 2008; Scott 2014). In the uplands, freshwater ecosystems are generally characterised by oligotrophic lakes that have low nutrient inputs, low vegetation content and are therefore less productive. Dystrophic lakes are also oligotrophic but are generally smaller in size and contain higher levels of humic substances from surrounding peat areas. Catchments containing these ecosystems mostly occur on hard, acid rocks, characteristic of northern and western parts of England and are situated away from enclosed agricultural land, occupying predominantly moorland or heathland areas. Eutrophic lakes are most common in the lowlands of England due to high nutrient inputs. Ponds are also widespread across the lowlands of England and play an important role in buffering human derived fluxes of atmospheric greenhouse gas emissions (Taylor and others 2019) (Photo 5.3).

Land use pressures can increase sediment accumulation in standing waters. Evidence suggests that drainage, afforestation and deforestation of short rotation forestry, heather burning and changes in livestock grazing can all accelerate soil inputs to freshwaters. A study from Northern Ireland showed that catchments with greater than 50 per cent cover of plantation forestry can alter algal communities within upland lakes (Stevenson and others 2015). This in turn could alter lake productivity, carbon and greenhouse gas emissions. Studies in England and Scotland have also shown that streams draining large areas of mature or second phase forestry were more acidified than small forested areas or moorland catchments, limiting their capacity to store and sequester carbon (Curtis and others 2014).

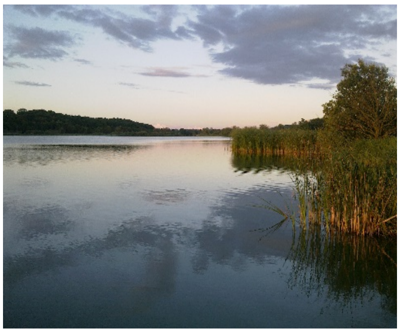
  

Photo 5.3 Examples of standing waters found across England: Aqualate Mere, Staffordshire © Natural England / Mel Brown (Left); Harland Moor site of special scientific interest – Dorset © Natural England / Peter Wakely (Centre). Ladybower Reservoir, High Peak, Derbyshire © Natural England / Peter Wakely (Right).

Land management in the lowlands also has the potential to affect standing waters in the uplands, and vice versa. For example, evidence suggests that land cover change, land use intensification and exposure of bare agricultural soils in lowland areas of the catchment could promote the atmospheric transfer of nutrient rich dusts, contributing to nutrient loading in remote lakes (Anderson and others 2020). Therefore, while most upland standing waters have low nutrient inputs, they are still vulnerable to eutrophication. Although vulnerable, the extent of eutrophication from nitrogen deposition in upland lakes is not fully known. Analysis of sediment cores from upland tarns in the Cumbrian Lake District suggest that these areas have been nitrogen enriched for around 100 years (Aquatic restoration partnership; accessed October 2020).

5.5.1 Carbon storage and sequestration in standing waters

Lakes

To the best of our knowledge, there are no published data on carbon storage, sequestration and emissions from natural, upland lakes (such as tarns) in the UK. However, long-term monitoring of dissolved organic carbon (DOC) suggests that concentrations are increasing. The Acid Waters Monitoring Network (AWMN) measured 22 upland waters, showing an average increase of 91 per cent over 15 years. This may be attributed to a combination of declining acid deposition from industry (acid rain) and rising temperatures related to climate change (Evans and others 2005). As pre-industrial conditions for UK surface waters are not well known, observed DOC increases in upland waters cannot be confidently linked to a single precursor. Increases in DOC content may be part of the natural recovery process following human induced acidification, natural fluctuation or human induced nutrient inputs. However, some of the dissolved organic matter released from upland areas will be transported into estuaries and oceans, which may impact on energy, nutrient and light regimes in these regions and ultimately impact atmospheric CO_2 (Evans and others 2005).

In the review of lake carbon sequestration across Europe, Anderson and others (2014) compiled data from 'culturally impacted' lakes and estimated an average burial rate of around 50 g C m^{-2} y^{-1} (0.5 t C ha^{-1} y^{-1}) throughout the last century. Burial rates for eutrophic lakes (categorised by P concentrations greater than 60 ug L^{-1}) were in the region of 100 g C m^{-2} y^{-1} (1 t C ha^{-1} y^{-1}) which is roughly a four-fold increase in carbon burial rates over the last 100 years. Of the nine English lakes included in the study, four were considered eutrophic and all had higher burial rates, ranging from 82.4 to 296.6 g C m^{-2} y^{-1} (0.824–2.966 t C ha^{-1} y^{-1}) compared to a range of 12.5–54.5 g C m^{-2} y^{-1} (0.125–0.545 t C ha^{-1} y^{-1}) for the non-eutrophic lakes (less than 60 ug P L^{-1}). Lakes with the highest C burial rates included shallow lakes of the Norfolk Broads and Shropshire–Cheshire Meres regions, and Marsworth Reservoir, Buckinghamshire (see 'reservoirs' section for more information). Lakes with lower C burial rates and lower total P were all situated within the Lake District National Park.

During a two year intensive survey of two lakes in the Shropshire–Cheshire meres region, Scott (2014) quantified carbon fixation and sequestration, as well as scaling up to the rest of the UK. Surveys of Rostherne Mere and Tatton Mere show that they fix on average 121 g C m^{-2} y^{-1} (1.21 t C ha^{-1} y^{-1}) and sequester (bury) 68 g C m^{-2} y^{-1} (0.68 t C ha^{-1} y^{-1}), giving a burial efficiency of 60 per cent. Scaled up to the Shropshire–Cheshire meres region (60 lakes in total), annual carbon accumulation was estimated at 506 t C y^{-1}. From this, it was estimated that eutrophic waters in the UK could be sequestering a combined value of 0.12 Mt C y^{-1}. However, caution is needed when scaling up to the whole of the UK due to high variability.

Two studies have quantified CO_2 and CH_4 fluxes in natural lowland lakes in England. A literature analysis by Sanches and others (2018) cited one study from the UK, which recorded a total CH_4 flux of 12 mmol m^{-2} d^{-1} (0.7 t CH_4 ha^{-1} y^{-1}) and CO_2 flux of 40 mmol m^{-2} d^{-1} (6.43 t CO_2 ha^{-1} y^{-1}) from a productive natural lake in the North West of England (Priest Pot) (Casper and others 2000). Furthermore, an analysis of 20 lakes in the Lake District National Park estimated CO_2 efflux (emissions) at between ~0.01 and 0.7 Gg C y^{-1} (10-700 t C y^{-1}) and showed that CO_2 flux increases with increasing net primary productivity (Maberly and others 2012).

Reservoirs

Carbon burial rates have been shown to be higher in human-made systems such as reservoirs, due to catchment instability and high erosion rates (Anderson and others 2020) but can also be large sources of CO_2 and CH_4 where sediment builds up behind dammed areas (Deemer and others 2016). Recent evidence has shown that reservoirs situated in peat dominated catchments are particular 'hotspots' for freshwater carbon cycling. Budgets for DOC, POC and dissolved CO_2 were determined for the Kinder reservoir in the Peak District National Park, which lies in a heavily eroded peat catchment (Stimson and others 2017). Particulate organic carbon represented the greatest input at between 171–265 t C y^{-1} in year 1 and 118–439 t C y^{-1} in year 2 of monitoring. This is almost double the DOC input, which was 89 t C y^{-1} in year 1 and 35 t C y^{-1} in year 2. Dissolved organic carbon was in turn at least double the fluvial CO_2 flux at 11 t C y^{-1} in year 1 and 17 t C y^{-1} in year 2. Measurements of inputs and outputs of the three carbon pools showed that the Kinder reservoir was a DOC sink in 2012 and a DOC source in 2013, while the opposite occurred for fluvial CO_2, suggesting annual variability linked to rainfall and temperature. Modelled carbon burial rates varied between 111 t C y^{-1} and 236 t C y^{-1}, whilst atmospheric C export varied between

18 t C y^{-1} and 193 t C y^{-1}. This is the first (to the authors knowledge) carbon budget for upland reservoirs in the UK, but studies of reservoirs in other temperate regions show a similar range (Barros and others 2011; Huttunen and others 2011, Teodoru and others 2011). Methane was not studied for the Kinder reservoir, but global averages are approximately one to two orders or magnitude lower than CO_2 (Stimson and others 2017; Teodoru and others 2011).

Peatland pools

Peatland pools are can form following peatland restoration, as drains are blocked and water tables rise. Pools also form naturally on functional blanket and raised bogs. In temperate and boreal ecosystems, peatland pools have been shown to be sources of both CO_2 and CH_4, though they are often not included in ecosystem scale greenhouse gas budgets (Pelletier and others 2014; Turner and others 2016). Pools act as recipients for dissolved and particulate organic carbon fed either through or over the surrounding peat. Regional variability is to be expected as the source and composition of organic carbon entering the pools will depend on local conditions, which will influence fluxes of CO_2 and CH_4 (Turner and others 2016). Information on sedimentary carbon storage and burial in peatland pools is lacking.

In a study of 66 peatland pools across the UK (Turner and others 2016), mean pool water dissolved CO_2 and CH_4 concentration ranged from 0.15–2.01 mg L^{-1} and 0.19–93.0 µg L^{-1} respectively; comparable to concentrations reported in peatland streams in Scotland (1.05–6.83 mg CO_2 L^{-1} and 4.81–28.88 µg CH_4 L^{-1} (Dinsmore and others 2013; Hope and others 2001). Shallow pools with greater vegetation cover had lower dissolved CH_4 concentrations, which was linked to the presence of *Sphagnum* mosses, and lower DOC concentrations overall. In contrast, the presence of *Eriophorum* Cotton grass within pools have been shown to generate larger methane fluxes (Peacock and others 2013).

Fluxes of CH_4 and CO_2 are lacking for the UK, but a study of Canadian peatland pools by McEnroe and others (2009) showed that the average CO_2 and CH_4 fluxes can be up to five times higher in small, shallow pools (< 1000 m^2, < 45 cm depth) at 0.35 ± 0.47 g C m^{-2} d^{-1} and 0.31 ± 0.69 g C m^{-2} d^{-1} (1.28–1.32 t C ha^{-1} y^{-1}), respectively compared to larger deeper pools (> 1000 m^2, > 70 cm). This was attributed to increased decomposition in shallow pools, but the study makes no links to vegetative cover, suggesting that older, larger pools may approach equilibrium (McEnroe and others 2009, Turner and others 2016).

The study of Peacock and others (2013) further recommended that peatland pools should be deeper than 0.5 m to give the greatest carbon benefit. This is because *Eriophorum* cotton grass cover decreases with pool depth, as the roots cannot readily establish, whereas *Sphagnum* mosses increase with depth. However, *Sphagnum* coverage varied widely between pools, suggesting that deeper pools formed behind constructed dams result in less vegetation cover, as lower light penetration slows the rate of vegetation establishment (Peacock and others 2013). Deeper pools could also provide a longer upward travel time for methane, encouraging further oxidation, although this will vary with region (Peacock and others 2013). Shallower pool depths may also promote better mixing of water and therefore promote aeration, allowing for greater oxidation of CH_4 within the water column (Turner and others 2016). Therefore, optimal pool construction for greenhouse gas mitigation will depend on individual site conditions, how readily vegetation can establish and whether the pools are situated behind constructed dams.

Ponds

Small ponds are common in agricultural settings. They can be either natural or artificial (i.e. dug into the landscape). They provide an important habitat for wetland wildlife and can act as a buffer against agricultural pollutants. Ponds may also come under the definition of a constructed wetland, commonly found in agricultural settings to prevent sediments entering nearby rivers and streams. Under appropriate management, ponds have the capacity to function as carbon sinks (Photo 5.4).

Photo 5.4 Marl Pond – Hampshire. © Natural England / Peter Wakely.

Small bodies of water are highly variable, making their contributions to C storage and sequestration difficult to assess. Gilbert and others (2014) measured a series of small, natural ponds in the north east of England (Druridge Bay, Northumberland) with four distinct characteristics (permanent naturally vegetated, arable field, grass pasture field, and dune slack) and estimated a burial rate of around 149 g C m^2 y^{-1} (1.49 t C ha^{-1} y^{-1}). Sediment organic carbon content was also found to be highest in extensively vegetated, uncompacted sediments from permanent ponds (7.68–12.86 per cent OC). The lowest per cent OC contents (mean 3.72 per cent OC) were from temporary ponds in arable fields, which often dry out and expose bare mud, are not well vegetated and are regularly disturbed by ploughing, making them relatively inefficient at trapping carbon. Furthermore, Taylor and others (2019) estimated carbon sequestration rates of ponds at the same site as 142 g m^2 y^{-1} (1.42 t C ha^{-1} y^{-1}) on average, with a range of 79–247 g m^{-2} y^{-1} (0.79 – 2.47 t C ha^{-1} y^{-1}) depending on the ponds' vegetation. Furthermore, Ockenden and others (2014) measured carbon accumulation rates in ten ponds (referred to as constructed wetland) across Cumbria and Leicestershire at between 4–8 t C ha^{-1} y^{-1} but no greenhouse gas measurements were made.

As temporary ponds dry out, CO_2 fluxes can quickly change from net sinks to net sources. Gilbert and others (2016) measured mean CO_2 shifts from an uptake of 641 mg m^2 d^{-1} (4.78 t ha^{-1} y^{-1}) to emissions of 3,792 mg m^{-2} d^{-1} (8.43 t ha^{-1} y^{-1}) during the transition from holding water to drying out. In this case CH_4 fluxes were negligible. Globally, agriculturally eutrophic ponds may bury carbon at an average rate of 2,122 g m^2 y^{-1} (21.22 t C ha^{-1} y^{-1}), exceeding other natural standing water ecosystems (Gilbert and others 2014; Downing 2010).

Ponds with high organic matter load and low oxygen levels favour higher methane emissions. Therefore, reducing nutrient loads from the landscape is likely to be the most effective in minimizing carbon emissions from these systems, given the strong relationship between CO_2, CH_4 and phosphorus and nitrogen enrichment. Reducing direct livestock access to farm waterbodies will also improve water quality by reducing nutrient inputs (Leavitt and others 2019). Recent evidence by Taylor and others (2019) has highlighted the role of vegetation in sequestering carbon and the role of moss carpets in protecting sediments exposed during dry conditions, compared to the limited role of 'bare' ponds in sequestering carbon. However, Leavitt and others (2019) also found that either phytoplankton or submerged plants actually supported greater CH_4 production in farm waterbodies. Other studies internationally have demonstrated high methane emissions from small artificial water bodies (Grinham and others 2018). Therefore, the role of farm ponds in negative climate forcing is heavily dependent upon management, vegetation type and extent, nutrient enrichment and sedimentary carbon accumulation rates.

Given the range of beneficial ecosystem services that ponds can provide, restoring ponds lost to land reclamation could help sequester more carbon and can be readily integrated alongside other land uses (Taylor and others 2019). However, this relies on careful management, including consideration of the implications of soil disturbance upon restoration. A combination of factors should therefore be considered to optimize the reduction of carbon emissions. The main factors, as identified by Webb and others (2019) include increasing the water residence time by creating deeper ponds, considering the role of groundwater in reducing methane production, facilitating methane oxidation through the water column, and reducing terrestrial inputs of organic matter. The construction of deeper ponds, coupled with vegetation, should also help prevent drying out in the summer months and in future periods of drought.

5.6 Climate change impacts and other interactions in rivers, lakes and wetland habitats

All freshwater ecosystems in the UK show high sensitivity to climate change (Natural England & RSPB 2020). The natural variability of the UK climate makes exact changes harder to predict (Watts and others 2015), but changes in regional climate have already been observed. Long-term analyses of weather station data for the UK indicates that upland areas are experiencing greater climatic change than lowland areas, with greater increases in winter minimum temperature and precipitation (Curtis and others 2014). Changes in rainfall patterns are a significant threat to the mitigation potential of freshwater systems, particularly the increase in frequency and intensity of storms and the projected increased frequency of drought / heatwaves.

In the future, pressure on all freshwater systems is predicted to increase, due to the combined effects of increasing human demands and a changing climate. In the UK, river and lake water quality may further decline as water temperatures increase and enhance algal blooms. Lower flow regimes in rivers through drier summer periods may also exacerbate algal blooms and contribute to elevated CO_2 and CH_4 emissions, as well as warmer, wetter periods in the winter (Watts and others 2015). In addition, increased water abstraction due to population demand is likely to reduce river flows further and increase the concentration of effluents from point sources such as sewage treatment works (Bussi and others 2016). Reduced river water inputs into lakes are also likely to increase residence times and potentially cause shallow lakes to dry out, exposing silts to the atmosphere unless sufficiently vegetated.

Natural and artificial small ponds are particularly vulnerable to drought conditions and quickly switch from sinks to sources of greenhouse gases when dry. For major rivers in the north of England, longer drier spells could lead to increased residence times of nutrients, leading to elevated greenhouse gas emissions and unfavourable conditions for biodiversity. The south of the England may be more resilient to lower flows as groundwater recharge partly regulates river water levels. However, increased groundwater abstraction may cause more extreme low flows. Therefore, current pressures on freshwater ecosystems, in conjunction with projected increases in demand make the future management of these ecosystems extremely complex.

Land management and land use will further impact freshwater ecosystems through changes in terrestrial nutrient cycles. An example is the River Thames, where it was found that increasing the agricultural fraction of the catchment is likely to trigger an increase of phosphorus due to diffuse inputs, which will increase the likelihood of algal blooms and elevated greenhouse gas emissions (Bussi and others 2016). Efforts to reduce point and diffuse pollution sources from rivers and lakes is readily practiced, though eutrophic events are still a problem countrywide. Increased particulate organic matter from soil erosion could also have detrimental impacts on lakes and the beds of slow flowing rivers, as oxygen levels deplete, reducing aerobic respiration and enhancing methane production.

In the uplands, the reduction of atmospheric nitrogen deposition from agriculture and fossil fuel combustion will gradually reduce eutrophication events in upland lakes and reservoirs, although areas with nitrogen rich soils may see a more gradual decrease (Curtis and others 2014). However, evidence suggests that there will be, and have already been, stronger increases in winter minimum temperatures and increasing winter precipitation in the uplands and there is little prospect of controlling lake temperature and water level in relation to climate change (Curtis and others 2014). It will therefore become increasingly important to reduce erosion rates from the uplands and protect upland freshwaters from high sediment loads and nutrient enrichment.

5.7 Evidence gaps and future needs of rivers, lakes and wetland habitats

Our understanding of the role of freshwaters in the carbon cycle is still evolving and significant knowledge gaps remain. To increase our confidence in the evidence base, we make the following recommendations for priority areas:

Better representation of key freshwater habitats. Areas earmarked as potentially significant carbon stores in the wider literature, such as floodplains, do not have good representation in the UK literature and further research in these areas is needed to assess their role in climate change mitigation. Furthermore, information on carbon storage and sequestration is particularly lacking in upland standing waters in comparison to lowland lakes and rivers.

Consideration of the linkages between terrestrial nature-based solutions and freshwaters. Freshwater systems are intimately linked to their surrounding catchment and any efforts on land will impact nearby freshwaters. An example of this is upland conifer plantation forestry, which was estimated to be the main factor influencing the spatial distribution of DOC export in the UK (Williamson and others 2021). An area which is better quantified is in degraded peat catchments, where elevated inputs of organic matter have been demonstrated (see section 5.2.1). Furthermore, reducing erosion and diffuse pollution could provide a considerable benefit to climate mitigation and biodiversity combined. Therefore, nature-based solutions for climate change mitigation carried out on land, should also consider the mitigation impacts of nearby freshwater systems.

Assessment of the carbon benefit of restoring human-modified habitats back to their natural state. Evidence suggests that human modifications can inhibit freshwater systems from functioning as a natural carbon sink. An example of this is the build up of sediments behind dammed or vegetated areas (see section 5.5.1) causing hotspots of CO_2 and CH_4. Floodplains have also been subject to intensive land uses for agriculture but have the capacity to store significant amounts of carbon if they are naturally functioning. Restoring these habitats to enable them to function as naturally as possible, could be beneficial for enhancing carbon storage and reducing greenhouse gas emissions, notwithstanding the positive impacts on biodiversity. However, we lack evidence in ‘pristine’ systems and more evidence is needed before, during, and after restoration to fully understand the carbon mitigation potential.

6 Marine and coastal habitats

6.1 Chapter summary and key messages

Blue carbon is the carbon dioxide removed from the atmosphere by coastal and marine habitats. Vegetated coastal habitats, particularly saltmarsh and seagrass, have the capacity to store and sequester considerable amounts of carbon through photosynthesis and subsequent burial in soils and sediments, as well as trapping and storing carbon transported from terrestrial and other marine habitats. When undisturbed, some marine and coastal habitats have the capacity to store carbon long-term. Key blue carbon habitats along our coastlines have undergone widespread habitat decline, largely due to physical disturbance, physical modification, pollution and disease and these habitats also show high sensitivity to climate change. There is considerable potential to enhance carbon storage and sequestration in marine and coastal habitats through restoration and protection, although large evidence gaps remain.

Key Messages

- Based on the available data, saltmarsh and seagrass represent the largest sedimentary carbon store of the coastal and marine habitats. When undisturbed, both habitats have the potential to store carbon long-term.
- Intertidal and subtidal sediments can also store large quantities of carbon. Evidence suggests that muddy sediments store more carbon than sandy sediments, resulting in ‘hotspots’ of carbon rich areas where muddy sediments accumulate.
- Kelp forests seem to contain the highest biomass carbon stocks of the vegetated coastal-marine habitats. However, it is important to bear in mind that evidence for all coastal-marine biomass carbon stocks is lacking, making it difficult to provide an accurate comparison.
- There is also a growing body of evidence in the wider literature highlighting the contribution of kelp-derived carbon to other blue carbon habitats or receiver / sink sites; a process which may considerably increase their importance in the coastal and marine carbon cycle and warrants further investigation.
- There is good agreement that saltmarsh restoration provides a sustained, albeit modest, sink for atmospheric CO_2 (Burden and others 2013). However, it must be noted that, while managed realignment does provide some carbon benefit, there are varying levels of success with regard to biodiversity (Mieszkowska 2020) and restoration may not provide the same ecosystem services as a natural saltmarsh system.
- Overall, our confidence assessments for most of the coastal and marine habitats are low (Table 6.1 & 6.2). For some habitats, no values could be found that had been measured in the English / UK context.
- We have medium confidence for saltmarsh habitats due to a more robust body of evidence, comparisons between different habitat conditions and analysis of carbon stocks and sequestration throughout restoration.

- Particularly for seagrass, some estimates of UK carbon stocks and fluxes rely on data from European and global studies, reflecting different habitat and climatic conditions, species composition and abundance. Care should be taken when interpreting predicted UK stocks derived from figures obtained outside of the UK context.
- Geographical spread of the available evidence is also an issue. Most of the data available for sand dunes, saltmarsh, seagrass and kelp are all focused to specific sites on the west and south-west coast. A better geographical spread is needed to address regional scale variabilities and increase our confidence in the evidence base available.
- There is considerable potential to protect, manage and increase carbon stocks and sequestration rates in coastal and marine habitats. However, we need a better understanding of current habitat extents, condition, regional variability and their carbon cycling, to fully grasp their role in climate change mitigation.
- Tables 6.1 and 6.2 summarise the carbon storage and flux values identified as representative for habitats reported in this chapter.

Table 6.1 Summary of carbon stocks in marine and coastal habitats, as based on the review of literature.[28]

Habitat Description	Sediment Carbon (t C ha^{-1})	Sediment Depth (cm)	Vegetation Carbon (t C ha^{-1})	Confidence [High, Medium, Low]	References
Coastal Habitats					
Sand dunes	9.5 [4 to 15]	15	5 (n=3) [1.6 to 8]	Low	Beaumont and others 2014; estimates presented here are for England only. Per hectare value has been calculated using the predicted extent of English sand dunes by Beaumont and others 2014.
Saltmarsh	59[a-e] [[a]20 to [e]134]	10-30	13[a]	Medium	[a]Beaumont and others 2014 [b]Ford and others 2012; [c]Burden and others 2013; [d]Ford and others 2019; [e]Burden and others 2019
Intertidal sediments (sandflats and mudflats)	12[a-c] [0.13[a] to 1.72[a]] English [5.5[b] to 18.4[b]] Welsh [20[c] to 89[c]] Scottish	20[a] English 10[b] Welsh 50[c] Scottish	N/A	Low	[a]Trimmer and others 1998; [b]Armstrong and others 2020; [c]Potouroglou 2017
Seagrass	39[ab] [6.7[b] to 114.2[a]]	30	0.3[b] [0.07[b] to 0.5[b]]	Low	[a]Green and others 2018; [b]Lima and others 2020
Kelp	N/A	N/A	6.7[a-c] [1.37[a] to 11.987[a]]	Low	[a]Pessarrodona and others 2018; [b]Smale and others 2016; [c]Gevaert and others 2008
Biogenic reefs	No data	No data	N/A	-	No data
Subtidal sediment	55 (mud) [6 to 123] 18 (sand) [4 to 76]	100	N/A	Medium	Parker and others 2020. Based on c.1000 measurements on sediments in Secretary of State (SoS) seas.

[28] Note: Carbon stock values for sand dunes and saltmarsh updated in October 2021 to correct a misprint in the the source material.

Table 6.2a Summary of carbon flux in marine and coastal habitats, as based on the review of literature. Seagrass is not included as no data could be found.

Habitat Description	Annual Carbon burial rate / loss for the habitat			References
	$CO_2e\ ha^{-1}\ y^{-1}$	Range (if possible)	Confidence [High, Medium, Low]	
Sand dune	-2.18	-2.13 to -2.68	Low	Jones and others (2008); measurements were made in Anglesey, Wales. No data are available for England.
Saltmarsh	-5.19	-2.35 to -8.03	Low	Beaumont and others 2014. based on previous assessments by Cannell and others 1999; Chmura and others (2003) and Adams and others (2012). Estimates are for the whole of the UK
Intertidal sediments	-1.98[ab]	-0.40[a] to -3.45[b]	Low	Armstrong and others (2020); estimated values for Wales Adams and others (2012) measured values for the Ouse estuary, England.
Subtidal sediment	-1.12[ab]	-0.07[a] to -2.16[a]	Low	Queirós and others 2019; measured values from the English Channel. De Haas and others (1997) estimated value for the North Sea.

Table 6.2b Summary of carbon flux in kelp beds, as based on the review of literature

Habitat Description	**Annual Carbon burial rate / loss for the habitat**					**References**
	CO_2e ha^{-1} y^{-1}	**Range (if possible)**	**Co_2e ha^{-1} y^{-1}**	**Range (if possible)**	**Confidence** [High, Medium, Low]	
	Donor flux		**Receiver flux**			
Kelp	+11.63[ab]	+7.42[a] to +15.84[a]	-0.33[b]	No data	Low	The donor flux is the flux of kelp detritus moving away from the kelp bed. The receiver flux is how much of that donor flux reaches the sublittoral sediment. '+' values in the donor flux column refer to the kelp being moved away from that habitat as a source to other habitats and not necessarily a source to the atmosphere. Pessarrodona and others 2018; based on measurements in warm and cold sites in England, Scotland and Wales. Smale and others 2016; measured for sites off the coast of Plymouth.

6.2 Marine and coastal habitats

Coastal and marine habitats are receiving increasing attention due to their potential to store and sequester large quantities of carbon. The coastal and marine contribution to the global anthropogenic carbon dioxide budget is now recognised and reported by the Intergovernmental Panel on Climate Change (IPCC) and other international bodies, highlighting the increasing importance of these ecosystems in mitigating against climate change (Bauer and others 2013; Luisetti and others 2019). However, in comparison to terrestrial systems, the role of coastal and marine habitats as a source and sink of greenhouse gases is comparatively under-studied.

The term ‘blue carbon’ refers to the carbon dioxide removed from the atmosphere by vegetated coastal and marine habitats, through photosynthesis and subsequent burial in soils and sediments. These areas are recognised as potential hotspots for carbon storage and sequestration, leading to the development of blue carbon strategies to mitigate and adapt to climate change through restoration and protection (Krause-Jensen and others 2018). In the UK context, the main blue carbon habitats are saltmarsh and seagrass meadows. Macroalgae such as kelp beds could also provide a potential carbon stock to other habitats, referred to as a ‘donor’ habitat (see table 6.2b for definition), but their inclusion in blue carbon strategies remains open for debate (Krause-Jensen and others 2018). Unvegetated Intertidal sediments (mud and sand flats) and subtidal sediments are not currently considered a blue carbon habitat, but are known to contain large carbon stores and bury carbon from both terrestrial and other marine habitats.

Along UK coastlines, sand dune habitats, saltmarshes and machair dune grassland (particularly in Scotland) make up around 93 per cent of the coastal margin habitat. The other 7 per cent consists of coastal vegetated shingle, shingle beaches, saline lagoons, maritime cliffs and slopes and small islands. Very little is known about carbon stocks and sequestration in the other 7 per cent of habitats (Beaumont and others 2014) and are not included in our review here. Small carbon stocks may exist in sheltered saline lagoons where sediment is able to build. Any carbon stocks in these areas are likely to be relatively short lived due to coastal shift and tidal recharge, making it difficult to effectively measure or manage these areas. However, there is a need for more work to fully understand these habitats and their contribution to the coastal-marine carbon cycle (Beaumont and others 2014).

6.2.1 Carbon cycling in coastal and marine habitats

A simplified conceptual diagram of the marine carbon cycle is shown in figure 6.1. In the marine environment, primary production is dominated by phytoplankton. Here, organic carbon is produced from carbon dioxide and oxygen in the water column through photosynthesis. Much of this is recycled in the water column, but some particulate organic carbon will reach the seabed, where it is processed by living organisms. This constant cycle of carbon dioxide uptake, sinking of detritus and decomposition at depth is known as the biological carbon pump. The movement of particulate inorganic carbon, mainly from the shells of marine organisms, also contributes to the sedimentary carbon store through the physical carbon pump (Burrows and others 2014). The uptake of CO_2 in the shelf seas may have increased over the last two decades as the difference in pCO_2 (partial pressure of carbon dioxide) between the air and the sea has increased (Legge and others 2020), which could have implications on the marine carbon pumps. However, there are uncertainties surrounding benthic carbon stock assessments, due to large variations in carbon stocks across the sea floor.

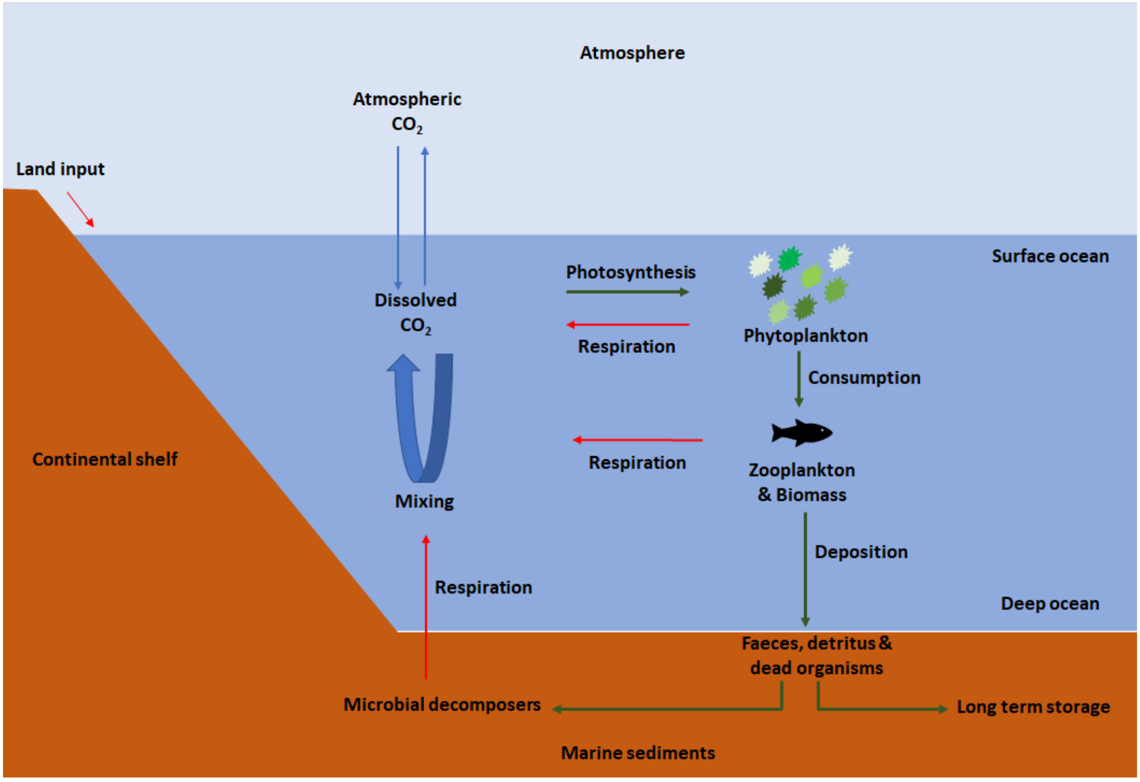


Figure 6.1 Conceptual diagram of the marine carbon cycle

Along our coastlines, intertidal and subtidal muddy sandflats are colonised by angiosperms such as seagrass or by saltmarsh plants, which develop on the extreme upper levels of sheltered fine sediment shores (JNCC 2015). Where these plants occur, they can fix atmospheric carbon directly through photosynthesis where it is stored in the vegetation, roots and rhizomes of the plants in relatively short timescales. This is subsequently stored in the sediments for hundreds to thousands of years. Saltmarsh and seagrass habitats also effectively trap sediments from other terrestrial and marine habitats, making them a potentially significant carbon store. However, carbon stored in these systems is particularly vulnerable to disturbance and other environmental drivers such as nutrient inputs. In these cases, carbon may be transported to other habitats, or mineralised and released to the atmosphere as CO_2 and to a lesser extent, methane (CH_4). Continual disturbance, for example through anchoring, mooring, dredging etc will further impact on the ability of these habitats to accumulate carbon in the future. Management of these carbon rich areas is therefore a key factor in their ability to play a role in climate change mitigation.

Less is known about CH_4 stores and fluxes in coastal and marine habitats. Methane is mostly generated microbially in anoxic sediments, which is released into the water column where some of it will be oxidised, reducing the amount of CH_4 released to the atmosphere. Most of the CH_4 released to the atmosphere occurs around near-shore areas where methane released from sediments can be released before oxidation in the water column can occur (Weber and others 2019).

6.3 Sand dunes

Coastal sand dunes are formed from sand particles blown inland from beaches. Dunes are therefore constantly forming, leading to a succession of ridges, increasing in age and stability the further inland they are (Burden and others 2020) (Photo 6.1). Low-lying areas are dominated by dune wetlands, whereas areas higher up the coastline are a mixture of fixed and mobile dune grassland, dune and scrub slacks. They are crucial habitats for a high diversity of rare insect species (Beaumont and others 2014; Burden and others 2020).

Photo 6.1 Ainsdale Dunes NNR (left) Duddon Estuary – Marram grass and sand dune (right) © Natural England / Neil Pike (left) / Paul Glendell (right)

Sand dunes are not considered a traditional blue carbon habitat, but still play a role in storing and sequestering carbon on the English coast. Dune habitats have the capacity to store and sequester carbon through the establishment of plant and scrub species through time, which contribute to the soil organic matter stock through the production of leaf litter and root detritus. Therefore, sand dunes increase their carbon storage capacity with age. This is likely to be highly regionally dependent, as coastal processes, morphology and vegetation type will all influence the capacity of these habitats to store and sequester carbon. Current estimates for total area of sand dunes in the UK is approximately 71,000 ha, around 11,778 ha of which is in England (Beaumont and others 2014). Dune systems in the UK have undergone long-term habitat changes due to human activities. In total there has been approximately a 30 per cent loss of dune slacks in some protected areas in England between 1990-2012 (Stratford 2014). Historically, areas have been forested with pine trees and managed for agriculture. More recently, dune areas have been claimed for urban expansion, tourism and leisure (Beaumont, Jones and others 2014).

6.3.1 Carbon storage and sequestration in sand dunes

The most comprehensive estimation of carbon stocks in dune habitats is that from Beaumont and others (2014). Soil carbon stocks (to 15 cm depth) were estimated at 178.7 kt C for dune grasslands, 46 kt C for dune slack and 'negligible' for mobile and semi-fixed dunes. Vegetation carbon stocks (including belowground root biomass) were 93.7 kt C for dune grasslands, 19 kt C for dune slack and 67.5 kt C for mobile and semi-fixed dunes. This totals 405 kt C for dune habitats in England. Carbon stocks on a per hectare basis were not given. To aid comparison, carbon stocks per hectare were calculated using the extent values presented by Beaumont and others (2014) and are shown in table 6.1. No other studies to date have quantified an inventory of carbon stocks for dune habitats in England or the rest of the UK. Therefore, the mitigation potential of dune habitats remains uncertain.

Jones and others (2008) calculated mean carbon sequestration in a long-term study at Newborough Warren in Anglesey, North Wales. Sequestration rates in dry dune grasslands were 58.2 g C m^{-2} y^{-1} (0.582 t C ha^{-1} y^{-1}), and 73.0 g C m^{-2} y^{-1} (0.73 t C ha^{-1} y^{-1}) in wet dune slack habitats, giving an average carbon sequestration rate for dune habitats of 59.5 g C m^{-2} y^{-1} (0.595 t C ha^{-1} y^{-1}). Sequestration values in CO_2 equivalents are given in table 6.2. Methane fluxes in dune habitats are unknown, but are likely to be negligible due to their low moisture contents (Beaumont and others 2014).

6.4 Saltmarsh

Saltmarsh habitats form on intertidal sand and mudflats that have been raised above the tidal level and receive inputs from both terrestrial and marine carbon sources (Picture 6.2). Saltmarshes are often characterised by zonation, with high carbon turnover rates in lower elevation marshes which are subject to natural coastal processes, such as erosion and accretion (growth). Natural high marshes are botanically diverse, with higher soil carbon contents and slower turnover rates. Overall, saltmarsh habitats are net carbon sinks and the restoration of these habitats is receiving increasing attention due to their carbon sequestration potential (Burden and others 2013).

Picture 6.2 Low elevation saltmarsh, Lymington, Keyhaven Marshes, Solent Maritime SAC / Solent and Southampton Water SPA © Natural England / Peter Wakely

The most recent estimates for saltmarsh extent in England is between 32,162 ha (Beaumont and others 2014; Burden and others 2020) and 37,953 ha (Natural England 2020 unpublished), with the largest saltmarsh areas situated within estuaries of Hampshire, north Kent, Essex, Norfolk, Lincolnshire, and Lancashire (May & Hansom 2003; Burden and others 2020). The five largest saltmarsh sites in England (Wash, Inner Solway, Morecambe Bay, Burry estuary, Dee estuary) account for about one third of the UK total extent (Burden and others 2020). Current saltmarsh extent around the UK is less than past coverage and continues to decline, largely due to drainage for agriculture and industrial development, and tidal separation by human-made sea defences (Burden and others 2020). Saltmarsh habitats on the west coast of the UK are typically characterised by shallow, organic rich clay and are commonly grazed, whereas marsh habitats on the south and east of the UK are typically un-grazed and are characterised by a deep organic rich clay substrate (Beaumont and others 2014; Armstrong and others 2020). Thus, high regional variability in carbon stocks and sequestration is to be expected.

6.4.1 Carbon storage and sequestration in Saltmarshes

Beaumont and others (2014) estimated saltmarsh sediment carbon stocks as 4,324.7 kt C for England and 5,413.2 kt C for the UK. More recently, Ford and others (2019) measured 23 marshes along the coast of Wales. Vegetation and soil characteristics were measured up to a depth of 10 cm and were used as a predictor of soil organic carbon stocks across all Welsh saltmarsh habitats. 44 per cent of the variation in surface soil carbon stocks were attributed to plant community and soil type, with higher soil carbon stocks found under *Juncus gerardii* and *J. maritimus* plant communities (40–60 t C ha^{-1}) than in the *Atriplex* and *Puccinellia* communities (20–50 t C ha^{-1}). Predictions of sediment organic carbon stock based on soil type, indicated that sandy soils store less carbon (29 t C ha^{-1}) than non-sandy soils (43 t C ha^{-1}) (to 10 cm depth).

Biomass carbon stocks in saltmarsh habitats are generally around an order or magnitude lower than soil organic carbon stocks. Beaumont and others (2014) estimated the saltmarsh vegetation stock as 419.6 kt C for England, and 584.6 kt C for the UK, calculated from biomass and loss on ignition data from sites on the west coast of England, Wales and the south east of England (Ford and others 2012; Burden and others 2013). Saltmarsh biomass stock was estimated to be 127.5 g C m^{-2} (1.275 t C ha^{-1}) for Scotland’s inshore marine protected area (MPA) network (Burrows and others 2014).

The most recent estimates of carbon sequestration rates in UK saltmarshes range from 64 to 219 g C m^{-2} y^{-1} (0.64–2.19 t C ha^{-1} y^{-1}; equivalent to 2.35–8.04 t CO_2e ha^{-1} y^{-1}), with typical figures in the range of 120–150 g C m^{-2} y^{-1} (1.2–1.5 t C ha^{-1}; equivalent to 2.35–8.04 t CO_2e ha^{-1} y^{-1}) (Beaumont and others 2014). The carbon sequestration range given by Beaumont and others (2014) is based on previous assessments of national carbon source and sink inventories by Cannell and others (1999), the global review of Chmura and others (2003) and measurements of carbon accumulation in the Blackwater Estuary, south east England by Adams and others (2012). The estimates given in Beaumont and others (2014) fall broadly in line with the global estimate of 151 g C m^{-2} y^{-1} (1.51 t C ha^{-1} y^{-1}) from Duarte and others (2005). The evidence base for carbon sequestration in UK saltmarsh habitats is more limited than the information available for saltmarsh soil carbon stocks. However, more recent studies suggest that regional variability of carbon sequestration rates are a function of time elapsed since restoration, grazing and other management practices (see below).

Impacts of restoration on saltmarsh carbon dynamics

Natural, high elevation marsh sites were shown to have the highest soil carbon stocks (93 t C ha^{-1} originally reported as 31.1 kg C m^{-3} sampled to a depth of 30 cm), and the slowest carbon turnover rate in a study by Burden and others (2013) at Tollesbury marsh, Essex. The high elevation marsh site which had undergone managed realignment 15 years prior, stored 66 t C ha^{-1} (reported as 22.1 kg m^{-3}) (Table 6.3), but had approximately twice as much aboveground plant biomass than the natural high elevation marsh due to a monoculture of *Puccinellia maritima*. However, this did not lead to higher soil organic matter concentrations than the natural high marsh, which was significantly higher than all other sites (21.8 per cent OC) due to species rich vegetation consisting of woody perennials. The restored high marsh site was much more similar to the area of marsh claimed for agriculture, which was 62 t C ha^{-1} (originally reported as 20.7 kg m^{-3}), indicating that the recovery of the restored high marsh is slow. Carbon storage in the natural and managed low shore sites were not significantly different from each other, with carbon stocks of 41 t C ha^{-1} and 33 t C ha^{-1} (originally reported as 13.7 and 10.9 kg m^{-3}) respectively, due to the constant replenishment of carbon rich sediments.

Burden and others (2019) investigated the effect of restoration on saltmarsh carbon, at nine sites in Eastern England representing a time–series from 16 to 114 years since restoration. A natural site and an un–restored, agricultural site was also measured. There was no clear relationship between age and soil carbon stock (to 30 cm), with the youngest sites having the highest carbon stocks and the intermediate (58-66 years post restoration) having the lowest (Table 6.3). In terms of carbon sequestration, rapid sequestration rates were observed in the first 20 years of restoration, at an average of 1.04 t C ha^{-1} y^{-1}, slowing to 0.65 t C ha^{-1} y^{-1} thereafter. The study confirmed that it would take approximately 100 years for restored sites to attain equivalent soil carbon stocks to natural salt marshes, agreeing with previous estimates (Burden and others 2013).

An earlier study by Adams and others (2012) in the Blackwater estuary, Essex, showed that both natural saltmarshes and managed realignment areas were small methane sources, at 0.10–0.40 g CH_4 m^{-2} y^{-1}. Areas earmarked for managed realignment in the Blackwater estuary (a 53.2 km^2 area of saltmarsh and intertidal mudflat) could therefore sequester an extra 4,174 t C y^{-1} (15,294 t CO_2e y^{-1}), when offset against methane and N_2O sources, while sites in the Humber estuary (74.95 km^2) could sequester (9,132 t CO_2e y^{-1}) (Andrews and others 2006; Adams and others 2012).

Table 6.3 Carbon stocks listed by time elapsed since restoration and saltmarsh condition, as reported in Burden and others ([a]2019; [b]2013). The un–restored agricultural site is referred to as 'field' and the un-managed site is referred to as 'natural'. 'High' refers to marsh above 1.75 m elevation and 'low' refers to marsh below 1.75m elevation.

Saltmarsh condition and restoration status	C stocks t ha^{-1}
[a]Field	59
[a]16-20 years	75
[a]58-66 years	56
[a]114 years	68
[a]Natural	69
[b]Natural high	93
[b]Managed high	66
[b]Natural low	41
[b]Managed low	33
[b]Agricultural	62

Impacts of grazing on saltmarsh carbon

At Crossens Marsh in the north-west of England, sediment carbon stocks were measured at two sites with contrasting management types; grazed and un-grazed. Soil carbon stocks were higher in the grazed site at 4.74 kg C m^{-2} (47.4 t C ha^{-1}) compared to the un-grazed site at 3.69 kg C m^{-2} (36.9 t C ha^{-1}) measured to 15 cm depth. In contrast, biomass carbon stocks were greater in the un-grazed site (Ford and others 2012). Furthermore, Harvey and others (2019) found that grazing management on saltmarsh habitats in the UK had no significant negative impact on belowground carbon storage, regardless of stock density, based on 22 salt marshes along the west coast of the UK. There is an evidence gap concerning how grazing may impact on carbon sequestration rates, although high stock densities reduce plant biomass and root growth, which could in turn reduce carbon sequestration rates in comparison to natural saltmarshes or sites with light to moderate grazing regimes (Harvey and others 2019).

Methane stocks and fluxes in saltmarsh ecosystems are near zero (Adams and others 2012). However, 'hotspots' of belowground methane have been measured on grazed saltmarshes due to soil compaction and higher soil moisture, with methane fluxes in a grazed saltmarsh peaking at 9.82 mg CH_4 m^{-2} hr^{-1} (86 g CH_4 m^{-2} y^{-1} or 0.86 t CH_4 ha^{-1} y^{-1}), compared to un-grazed (0.28 mg m^{-2} hr^{-1}) (2.5 g CH_4 m^{-2} y^{-1} or 0.025 t C ha^{-1} y^{-1}) at Crossens Marsh in the Ribble estuary, north-west England (Ford and others 2012; Beaumont and others 2014).

6.5 Intertidal sediments (sandflats and mudflats)

Intertidal mudflats and sandflats (picture 6.3) account for the unvegetated areas in shallow tidal and estuarine zones. Intertidal areas receive carbon inputs from both the terrestrial and marine environments, and due to their anoxic nature, are generally a carbon sink. At low tide, carbon is exchanged directly with the atmosphere in intertidal habitats (Legge and others 2020).


Picture 6.3 Duddon Estuary Site of Special Scientific Interest © Natural England / Peter Wakely (left). Intertidal mud in The Wash & North Norfolk Coast SAC, Norfolk © Natural England / Philip Ray (Right)

Particle size in intertidal areas is determined primarily by processes such as exposure to wave action and the slope of the intertidal area. Different particle sizes give rise to characteristic habitats including shorelines and estuaries and are generally composed of coarse- and fine-grained sediments including shingle (mobile cobbles and pebbles), gravel, sand or mud. Habitat extents predicted by Natural England suggest that intertidal mud covers 70,729 ha and intertidal sand and 'muddy sand' covers 96,598 ha, totalling 167,327 ha (Natural England 2020 unpublished). Sediment type and particle size are likely to influence carbon storage potential, with fine silty sediments storing more carbon compared to coarse gravel and sandy sediments. Due to the dynamic nature of intertidal habitats, sediment composition is likely to change with prevailing coastal processes.

6.5.1 Carbon storage and sequestration in intertidal sediments

Wood and others (2015) quantified the sedimentary composition of intertidal mud and sand flats in Essex and around Morecambe Bay, north-west England. Per cent organic carbon ranged from 0–7.5 per cent across all of the sites. Sediments collected from Essex had a higher average organic carbon content, compared to Morecambe Bay, which is more characteristic of sandy sediments and thus less capacity to store carbon.

Only one study reporting carbon stocks for English intertidal sediments could be found. Trimmer and others (1998) found carbon stocks (to 20 cm depth) of between (13.33 g C m^{-2}–171.75 g C m^{-2} or 0.13 t C ha^{-1}–1.72 t C ha^{-1}) (originally reported as 1.11–14.30 mol m^{-2}) for sites in the lower Ouse estuary. A review of the carbon sink potential of the Welsh marine environment (Armstrong and others 2020) estimated the sedimentary standing stock of Welsh intertidal sediments as 0.55–1.84 kg C m^{-2} (5.5–18.4 t C ha^{-1}) in the top 10 cm. However, this estimate was derived from a review of organic carbon standing stock in subtidal sediments of the north-west European continental shelf (Diesing and others 2017). Values presented in Diesing and others (2017) were multiplied by a factor of two, on the premise that nearshore sediments have the capacity to store more carbon. A PhD thesis by Potouroglou (2017) quantified carbon stocks of Scottish mudflats (unvegetated) of between 20 and 89 t C ha^{-1} for the top 50 cm of sediment.

The review of carbon in Welsh marine habitats (Armstrong and others 2020) estimated a sequestration rate of 0.011–0.037 kg C m^{-2} y^{-1} (0.11–0.37 t C ha^{-1} y^{-1}). Carbon sequestration rates were previously measured in the Blackwater Estuary, Essex, by Adams and others (2012). Natural mudflats had higher carbon sequestration rates compared to managed realignment sites, at 93.7 g C m^{-2} y^{-1} (0.94 t C ha^{-1} y^{-1}) and 73.3 g C m^{-2} y^{-1} (0.73 t C ha^{-1} y^{-1}) respectively. Globally, Duarte and others (2005) estimated the total carbon burial in unvegetated intertidal sediments as 126 Tg C y^{-1} (1.26 x 10^8 t C y^{-1}), highlighting their significant contribution to coastal and marine carbon burial.

Information on methane fluxes in intertidal sediments in the UK is lacking, but evidence from elsewhere shows that near-shore areas can be hotspots for the release of methane to the atmosphere (Weber and others 2019). Upstill-Goddard & Barnes (2016) reported methane fluxes from six intertidal areas across the UK (Humber, Forth, Tamar, Tyne, Tees and Tay). All six areas were atmospheric CH_4 sources, with annual emissions ranging from 3.1 x 10^7 (31 t CH_4 y^{-1}) to 10.9 x 10^7 g CH_4 y^{-1} (109 t CH_4 y^{-1}) resulting in a UK wide flux of 5.8 x 10^9 g CH_4 y^{-1} (5800 t CH_4 y^{-1}).

6.6 Seagrass

Seagrass meadows provide nursery grounds for economically important fish species, improve water quality and provide coastal protection (Nordlund and others 2018) (Picture 6.4). Seagrass meadows slow the flow of water, sequestering dissolved CO_2 and contributing to the long-term sedimentary carbon sink. Dense seagrass are also effective in trapping detritus, resulting in a sediment accumulation capability which has been recorded to be greater than that of saltmarsh plants (Couto and others 2013; Garrard & Beaumont 2014), storing carbon in the sediments from centuries to millennia. However, physical disturbance of seagrass sediments can cause re-suspension of long-term stored carbon, resulting in a potential release of CO_2 and CH_4 and a reduction in future carbon storage potential (Macreadie and others 2015).

Picture 6.4 Seagrass meadow in the Isles of Scilly © Natural England / Emily Priestly

Records of seagrass extents vary. In England, estimates are between 4,500 ha (England and Wales combined) (Luisetti and others 2019) and 11,600 ha (Natural England 2020 unpublished). Green and others (2021) documented 8,493 ha of recently mapped seagrass across the UK since 1998, resulting in an estimated 0.9 Mt C. This study estimated that at least 44 per cent of UK seagrass has been lost since 1936 and 39 per cent since the 1980's. Based on these estimates, historical seagrass meadows could have stored up to 11.5 Mt C. *Zostera marina* and *Zostera noltii* are the most abundant species, with mapped *Z. marina* beds making up around half of the total UK seagrass area (Luisetti and others 2019). However, seagrass wasting disease, nutrient enrichment (Hughes and others 2018) and physical damage through human activities are the main causes of decline, such that sea-grass meadows are considered scarce in UK waters (Garrard & Beaumont 2014). Overall, evidence suggests that UK seagrass meadows are mostly in poor condition relative to global averages, although areas situated away from human populations are perceived to be healthy (Jones & Unsworth 2016).

6.6.1 Carbon storage and sequestration in seagrass habitats

Green and others (2018) was the first study to measure seagrass sedimentary carbon stocks in England, using 13 sites all situated in the western English Channel. Measured sedimentary stocks (to 30 cm) ranged from 29.4 t C ha^{-1} to 114.02 t C ha^{-1} .Carbon stocks for the study sites were extrapolated to 100 cm depth, giving an average of 66,337 t C and an estimated UK wide stock of between 108,427 t C and 221,870 t C. This is substantially higher than previous estimates by Garrard & Beaumont (2014) of between 8050 t C and 16,100 t C for European sedimentary seagrass stocks. In Scotland, Potouroglou (2017) measured carbon stocks stored in the upper 50 cm of sediment under Z. *marina* and Z. *noltii* at between 22.7 t C ha^{-1} and 107.9 t C ha^{-1} with a mean of 57 t C ha^{-1} across seven sites. Scaled up to the rest of Scotland, the total estimated carbon stock in seagrass sediment is 91,200 t C. The global average sedimentary carbon stock for seagrass ecosystems is 194.2 t C ha^{-1}, compared to 2.52 t C ha^{-1} stored in living biomass, two-thirds of which is found within the roots and rhizomes of the plant (Fourqurean and others 2012; Garrard & Beaumont, 2014).

As the largest store of carbon in seagrass systems is in the sediment, the studies presented above do not consider living biomass in their assessment of seagrass carbon stocks. However, a recent study by Lima and others (2020) recorded vegetation seagrass stocks between 0.07 t C ha^{-1} and 0.5 t C ha^{-1} for 6 locations within the Solent, southern England. Significant variation was also found in the sedimentary carbon stocks (to 30 cm depth), ranging between 6.65 t C ha^{-1} and 51.13 t C ha^{-1}. Such large variations were attributed to sediment characteristics and could be an important indicator for carbon storage potential in seagrass meadows used for climate change mitigation. The authors further highlight that individual seagrass meadows might not be representative of the whole ecosystem and care needs to be taken when extrapolating across different regions.

To the best of our knowledge, no measured data exist for carbon sequestration for English / UK seagrass ecosystems. Carbon accumulation rates of UK seagrass meadows was recently estimated to be 2,500 t C y^{-1} (Luisetti and others 2019). Similar estimates have been made for the carbon sequestration capacity for Scotland (1,321 t C y^{-1}) (Burrows and others 2014). However, both estimates relied on values of carbon sequestration for seagrass meadows of varying species, from the north-east Atlantic (Fourquerean and others 2012) and a limited dataset from the Mediterranean (Duarte and others 2005).

6.7 Kelp

Macroalgal beds dominate shallow rocky coastlines in temperate and subpolar regions. In the UK, the most common species is the kelp *Laminaria hyperborea*, which covers over 80 per cent of all macroalgal species in the UK (Smale and others 2016) (picture 6.5) and is a dominant net primary producer, averaging 340 g C m^{-2} (across four UK sites) (Smale and others 2020). Kelp coverage in the UK is unclear and most estimates are based on areas where kelp are theoretically most likely to be abundant. Yesson and others (2015) predicted that the area of habitat in the UK suitable for *L. hyperborea* to establish is 15,984 km^2 (1,598,400 ha). Smale and others 2016 predicted that UK kelp forests are currently likely to cover around 51 per cent of the area proposed by Yesson and others (2015), around 8,151 km^2 (815,100 ha).

Picture 6.5 Kelp forest understory communities, South West England © Natural England / Angela Gall

6.7.1 Carbon storage and sequestration in kelp habitats

Kelp forests are not thought to store carbon in soft sediments (Alongi 2018). The most recent estimate of UK carbon standing stock (biomass) in the dominant kelp species *L. hyperborea* ranged from 137.4 to 1198.7 g C m^{-2} (1.37–11.987 t C ha^{-1}) (Pessarrodona and others 2018) at four sites; two 'cold' water sites in Scotland (northern and south-west), one 'warm' water site in south Wales and another 'warm' water site in south-west England. Averages for the warm and cold sites were 980 g C m^{-2} (9.8 t C ha^{-1}) and 310 g C m^{-2} (3.1 t C ha^{-1}) respectively. Previous estimates using the same four sites ranged from 355–1,146 g C m^{-2} (3.55 t C ha^{-1}–11.46 t C ha^{-1}), resulting in a study-wide average of 721 g C m^{-2} (7.21 t C ha^{-1}) (Smale and others 2016). Both studies found significantly higher standing stocks of carbon in the cold sites, both in Scotland. Carbon stocks measured in the Scottish sites were higher than previous estimates of 94–187 g C m^{-2} (0.94–1.87 t C ha^{-1}) for areas of abundant (> 20 per cent coverage) in Scottish seas (Burrows and others 2014). Given that these studies all focused on a single dominant species, total carbon stocks in the UK are likely to be higher than estimated here. For example, another kelp species relevant to the English coastline is *L. digitata*, which was found to have a standing stock of around 403.2 g C m^{-2} (4.03 t C ha^{-1}) in the eastern English channel (Gevaert and others 2008).

Carbon transport to other ecosystems

There has been increasing interest in the transport of macroalgae to other habitats and its contribution to long-term carbon storage. Recent international research suggests that plant debris can be transported offshore, enhancing organic carbon sequestration in the deep sea (Kokubu and others 2019; Ortega and others 2019), as well as in seagrass beds and saltmarsh habitats (Smale and others 2016; Hill and others 2015). Other studies highlight the high percentage (around 80 per cent) of kelp detritus consumed and transported by grazers (Filbee-Dexter and others 2020; Hill and others 2015). A recent study on the kelp species *L. hyperborea* reported an annual detritus production of 478 g C m^{-2} (4.78 t C ha^{-1}), based on 10 sites in northern Norway (Pedersen and others 2020). In the UK, Pessarrodona and others (2018) reported the carbon flux of particulate detritus to other ecosystems as 432.1 g C m^{-2} y^{-1} (4.32 t C ha^{-1} y^{-1}) for the cold sites in Scotland, and 202.4 g C m^{-2} y^{-1} (2.024 t C ha^{-1} y^{-1}) for the warm sites in south Wales and south west England, totalling 634.5 g C m^{-2} y^{-1} (6.35 t C ha^{-1} y^{-1}) for the sites studied.

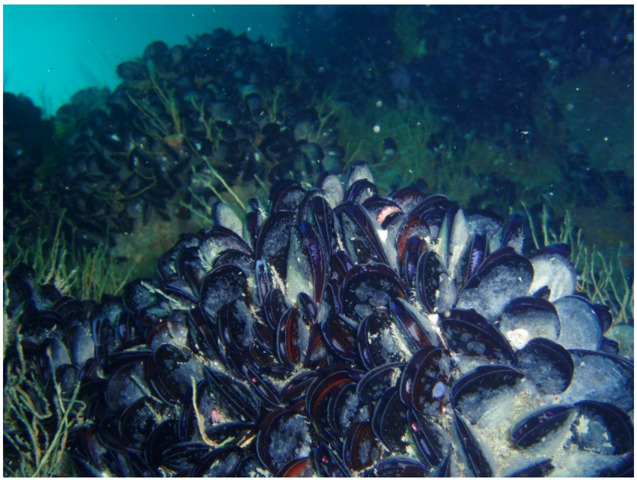
The proportion of kelp macroalgal detritus reaching shelf seas and accumulating there is less clear, and some studies assume that macroalgal derived organic matter would decompose too quickly for long-range export and burial to occur (Howard and others 2017). A recent study by Queirós and others (2019) recorded a macroalgae derived sequestration rate of 8.77 ± 9.85 g C m^2 y^{-1} (0.09 t C ha^{-1} y^{-1}), in sediments off the coast of Plymouth. Scaled up to the rest of the UK, this would give a net flux of 0.70 T g C y^{-1} (700,000 t C y^{-1}) in similar sedimentary habitats. This is predicted to equal around 4–9 per cent of macroalgae detritus released annually, becoming sequestered in coastal sediments. The estimated UK macroalgae flux presented in Queirós and others (2019) is largely in agreement with UK estimates made in Pessarrodona and others (2018) (~0.57 Tg C y^{-1} or 570,000 t C y^{-1}) and are comparable to the total annual carbon accumulated in European saltmarsh habitats (0.72 Tg C y^{-1} or 720,000 t C y^{-1}) (Ouyang & Lee 2014). The proportion of kelp contribution to the long-term carbon store is likely to be a function of shelf conditions adjacent to the kelp bed, sea-bed characteristics, current and wave driven hydrodynamics and macroalgal species composition. Therefore, high regional variability should be expected.

The studies presented above represent kelp particulate organic carbon; less is known about fluxes of kelp derived dissolved organic carbon and their contribution to the carbon sink. Kelp derived dissolved organic carbon has been estimated to represent up to a quarter of the total organic carbon assimilated and released by *L. hyperborea* along the west coast of Norway (Abdullah & Fredriksen, 2004), though large uncertainties remain regarding the fate of dissolved organic carbon from kelp species (Pedersen and others 2020).

6.8 Biogenic reefs

Biogenic reefs are known to contribute to climate change adaptation, by providing coastal protection and contributing to the build-up of sediments (Lovelock & Duarte 2019). However, biogenic reefs are not currently thought to contribute to greenhouse gas mitigation, as the process of calcification during shell development releases CO_2, making these habitats potential net CO_2 sources rather than CO_2 sinks (Lovelock & Duarte 2019). Quantitative information on biogenic reefs is scarce, and future research on the potential role of calcifying organisms in carbon sequestration is needed.

In the UK, invertebrate species that form biogenic reefs include blue or horse mussels (picture 6.6), oysters, and honeycomb or ross tubeworms. However, most of the studies on carbon storage and sequestration have focused on American oyster reef species.



Picture 6.6 Mussel bed (*Mytilus edulis*) off the north coast of Cornwall © Natural England / Angela Gall

6.8.1 Carbon storage and sequestration in biogenic reefs

A review by Fodrie and others (2017) found that reefs on intertidal sandflats were net sources of CO_2 (7.1 ± 1.2 t CO_2 ha^{-1} y^{-1}) resulting from predominantly carbonate deposition. Shallow subtidal reefs and saltmarsh fringing reefs (where oyster reefs are present at the edge of a saltmarsh) were small net sinks (-1.0 ± 0.4 t C ha^{-1} y^{-1} and -1.3 ± 0.4 t C ha^{-1} y^{-1} respectively) due to the presence of organic carbon rich sediments. The study highlighted that climate change mitigation is not a service that should be expected of reefs constructed over sandflats.

Biogenic reefs may facilitate carbon sequestration in other habitats, thus providing an indirect mitigation potential. Observations from a reserve in the United States (Fodrie and others 2017) showed that saltmarsh fringing reefs have facilitated the sea-ward migration of saltmarshes, increasing their carbon storage capacity. To date, no existing climate mitigation initiatives consider the role of shellfish reefs in carbon burial, neither are there standardised methodologies for assessing how shellfish reefs influence coastal and marine carbon cycling (Fodrie and others 2017).

6.9 Subtidal sediments

Subtidal sediments extend to the depth at which there is no effect from waves; typically around 50–70 m, which covers a large part of English waters. Their extent covers around 1,560,708 ha (mud) and 12,152,508 ha (sand) (Natural England 2020 unpublished). Subtidal sediments play a vital part in the storage of carbon in marine ecosystems (Ravaglioli and others 2019) and are considered the primary marine store of biologically derived carbon (Legge and others 2020; Burrows and others 2014). Organic carbon may be sequestered into subtidal sediments through the deposition of organic matter present in the water column, which has the potential to be stored in sediments for decades to centuries. However, sediment characteristics are driven by physical processes such as particulate movement and bed migration due to storms, as well as disturbance from human activities such as bottom trawling (Diesing and others 2017; Luisetti and others 2019).

Subtidal sediments can act as a sink for terrestrial carbon (Luisetti and others 2020). However, particulate inorganic carbon is thought to be the dominant source of carbon in subtidal sediments. Smeaton and others (2021) mapped sedimentary carbon stocks (to 10 cm depth) across the UK Exclusive Economic Zone (EEZ). PIC was estimated at 2582 ± 168 Mt IC, whereas organic carbon was estimated at 524 ± 68 Mt. Spatial mapping highlighted hotspots of organic matter accumulation, indicating that muddy sediments store the greatest quantity of organic carbon. Any future management of the seafloor could therefore prioritise these hotspots of high organic matter stocks. In contrast though, Diesing and others (2017) argued that muddy sediments contributed less to the overall carbon stock due to their limited extents. Coarser sand and gravel contributed around 71 per cent of the overall particulate organic carbon stock due to ‘widespread occurrence in the study area’. Shelf sediments that are dominated by sandy, highly permeable sediments tend to be even richer in PIC compared to organic carbon. The residence time of inorganic carbon in sediments is also thought to be much longer than organic carbon, with residence times of several centuries.

6.9.1 Carbon storage and sequestration in subtidal sediments

The Centre for Environment, Fisheries and Aquaculture Science (Cefas) collected ~1000 carbon concentration measurements in subtidal sediments around the UK, which were used to derive the overall carbon stocks, integrating to a depth of 1 m. When apportioned by sediment type, organic carbon stocks for muddy sediment types ranged between 0.6–12.3

kg C m^{-2} (6–123 t C ha^{-1}) (average 5.5 kg C m^{-2} or 55 t C ha^{-1}) and sandy sediments ranged between 0.4–7.6 kg C m^{-2} (4–76 t C ha^{-1}) (average 1.8 kg C m^{-2} or 18 t C ha^{-1}) (Parker and others 2020; Diesing and others 2017).

The available evidence for carbon sequestration rates in UK subtidal sediments is lacking. The most recent assessment of carbon sequestration is that by Queirós and others 2019 in a 13-month study of deep (up to ~45 m) coastal sedimentary sites in the English Channel and subtidal regions around Plymouth. Here, the average net sequestration of particulate organic carbon in sediments was estimated as 58.74 g C m^{-2} y^{-1} (0.59 t C ha^{-1} y^{-1}). An earlier study by Dde Haas and others (1997) derived an organic carbon sequestration rate as 0.2 g C m^{-2} y^{-1} (0.002 t C ha^{-1} y^{-1}) as an average value for the North Sea.

Marine habitats are known to be a minor source of methane to the atmosphere, but data are limited for English and UK waters. Borges and others (2016) reported the average methane flux of near-shore (up to 3 m depth) southern regions of the North Sea (Belgian coastal zone) as 761.1 g CH_4 m^{-2} y^{-1} (or 7.61 t CH_4 ha^{-1} y^{-1}; originally reported as 130 µmol m^{-2} d^{-1}), one order of magnitude higher than typical values for the continental shelf (175.6 g CH_4 m^{-2} y^{-1} or 1.76 t CH_4 ha^{-1} y^{-1}; originally reported as 30 µmol m^{-2} d^{-1}) and three orders of magnitude higher than the open ocean (2.3 g CH_4 m^{-2} y^{-1} or 0.023 t CH_4 ha^{-1} y^{-1} reported as ~0.4 µmol m^{-2} d^{-1}). High methane fluxes were related to the shallow and well-mixed water column that facilitates transfer of methane from the seafloor to the surface. The study further suggests that methane emissions in near-shore areas could increase in response to regional warming of surface waters.

6.10 Climate change and other interactions in coastal and marine habitats

Climate change

Most coastal and marine habitats, including saltmarsh, seagrass meadows and kelp beds are highly sensitive to climate change, while sand dunes are classed as having a medium sensitivity (Natural England & RSPB 2020). Coastal and marine habitats are likely to be impacted by both the direct (air and sea surface temperatures, rainfall) and indirect (coastal erosion and sea level rise) consequences of a changing climate (Burden and others 2020). The Marine Climate Change Impacts Partnership (MCCIP) gives a high confidence that air and sea temperature increase, and sea level rise are already impacting UK shores. Overall, there is medium confidence that the effects of climate change will continue to impact coastal and marine environments in the future, which will also impact on the biological and physical processes necessary to store and sequester carbon (MCCIP 2020). Further pressures of a growing population and changing management practices, both on land and at sea, are expected to exert even greater risks on mitigation potential of these habitats and their resilience to future climate change.

Increasing air and sea temperature are particularly concerning for biomass carbon stocks, including seagrass, macroalgae and saltmarsh. For example, laboratory experiments have suggested that the seagrass Z. *marina* stops growing at > 20 ^{0}C and starts dying at > 25 ^{0}C, with dieback during summer heat waves already observed in northern latitudes (Winters and others 2011; Robins and others 2016). Yesson and others (2015) also showed reduced biomass, height and age of kelp forests in warmer seas on the south coast of the UK, indicating that rising sea temperatures may impact on kelp structure and therefore their capacity to store carbon and provide ecosystem services. Recent surveys of English dune slacks suggest that these areas are drying out due to increasing temperatures and dry periods, especially in the south and west of England. UK modelling suggests that dune water tables in some areas may drop by 1 m by 2080 (Clarke & Ayutthaya 2010; Burden and others 2020). In contrast, dunes in Scotland were found to be largely unaffected by climate change, highlighting a strong regional variability (Stratford and others 2014; Pakeman and others 2015).

Furthermore, sea level rise can result in 'coastal squeeze' which refers to a restriction of natural landward migration of intertidal and coastal habitats when they become restricted by coastal developments such as sea defences or coastal infrastructure. In intertidal areas, changes in the frequency and intensity of storms may affect sediment transport and cause remobilisation in intertidal areas, influencing the carbon storage capacity. Coastal erosion is also predicted to reduce UK saltmarsh areas by 8 per cent by 2060 (Beaumont and others 2014). Sea level rise and increasing frequency and intensity of storms, particularly during winter, may reduce the ability of vegetated habitats to recover, potentially leading to a reduction in areas where restored habitats can thrive.

While ocean acidification will negatively impact biodiversity and other ecosystem services, it may positively impact seagrass and kelp populations. Under current conditions, seagrass habitats in particular are carbon-limited with respect to photosynthesis (Harley and others 2006). Ocean acidification could therefore result in a significant increase in seagrass standing stock, increasing the carbon sequestration potential. One study on seagrass response to ocean acidification estimated an increase in UK standing stock of *Z. marina* by 82 per cent, based on current seagrass coverage (Garrard & Beaumont 2014). However, whether the benefit of increased productivity will offset other climate change impacts on these habitats is largely unknown.

Coastal development

Continuing development and population demand will further affect the mitigation potential of coastal and marine habitats. In particular, elevated nitrogen and phosphorus concentrations from sewage, fertiliser and agricultural runoff leads to eutrophication, anoxia (low oxygen levels) and disease and have the potential to affect all coastal and marine habitats. In subtidal sediments, infauna (animals living in the sediments) play a key part in the storage of carbon in marine ecosystems by burying accumulated organic matter originating from the water column (Ravaglioli and others 2019). Low oxygen, combined with increases in seawater carbon dioxide concentrations through ocean acidification, and rises in sea temperature, may negatively impact on infauna and their ability to maintain carbon fluxes at the sediment–water interface, impacting on their role in carbon storage and sequestration (Ravaglioli and others 2019). However, research on the impacts of low oxygen on carbon storage potential of subtidal sediments is still at an early stage.

There is a need to understand how coastal development, offshore industry and climate change will impact on the carbon storage potential of coastal and marine environments in the future. Already there have been declines in the quality and extent of sand dunes, saltmarsh, intertidal sediments and seagrass habitats due to urban expansion, agriculture, port and harbour expansion, fishing practices and sea defences. In subtidal sediments, bottom trawling and aggregate dredging causes the remineralisation of re-suspended organic carbon and mixing of sedimentary particulate organic carbon, affecting carbon storage (Luisetti and others 2019) and disturbing living biomass, which controls carbon remineralisation. This disturbance also changes the depth and rate of organic carbon burial and the seabed communities that are important for carbon sequestration. For example, fishing with bottom towed gear is known to be the cause of major physical disturbance for benthic ecosystems. This type of gear passes over the seabed or interacts with the top layers of sediment, resulting in 20–50 per cent of the biota being removed or damaged (Kaiser and others 2006). Little is known about the trawling impacts on carbon, but a predicted 50 per cent reduction in organic carbon storage in the top 10 cm of marine sediments was reported by Luisetti and others (2019), as a result of resuspension linked to fishing with bottom towed gear. The most frequently trawled sediments can be particularly carbon rich due to high levels of prey species biomass found there. Further research is needed to make more informed decisions on the future management of these ecosystems for climate change mitigation.

6.11 Evidence gaps and future needs

There are significant gaps in the evidence available for the carbon stocks and sequestration rates of coastal and marine habitats. Most studies focus on a very small area of the UK, for example along the south coast. Considerable regional variability is clear from the limited evidence available, making UK-wide predictions of carbon stocks and sequestration rates uncertain. To increase our confidence in the evidence base in the context of climate change mitigation, we make the following recommendations for priority areas:

Better spatial distribution of carbon stocks and sequestration measurements for marine and coastal habitats is needed. Overall, values for soil and sedimentary carbon stocks are more abundant than for sequestration rates. However, the evidence base for both stocks and sequestration across all coastal and marine habitats should be improved, to give a better understanding of regional variability, and to ensure that any country-wide estimates are representative. This is especially the case for seagrass beds, kelp forests and intertidal sediments, which are mostly focused along the south coast of England and the west coast of Scotland.

There are an abundance of reviews outlining the impacts of climate change and human activities on habitat condition. However, quantification of the effects of habitat condition on carbon storage and sequestration must be priority to better understand this interaction and its future resilience to climate change. This includes the impact of industries on habitat condition, as well management and climate change. This will allow a better assessment of the benefits of protecting existing carbon stocks and the carbon benefit of habitat restoration. In doing so, measuring carbon stocks and sequestration rates before, during and after restoration and protection should be carried out, including timescales of recovery and change. The only habitat where this has been carried out is saltmarsh, where carbon sequestration measurements were made along a time-series gradient of restoration, enabling predictions of future carbon stocks and comparisons to natural, unmanaged sites (Burden and others 2019).

The linkages between terrestrial, coastal and marine habitats need more consideration in the context of carbon cycling. It is well known that nutrient rich contaminants from erosion, fertiliser runoff and sewage effluent threaten coastal and marine habitats through eutrophication, depleted oxygen and disease. However, the carbon impacts of these is largely unknown. As terrestrial, coastal and marine environments are intrinsically linked, it is important to understand how delivering nature-based solutions on land will affect these habitats. For example, reducing terrestrial nutrient inputs may also provide a carbon benefit along the coast by improving habitat condition and therefore carbon stocks. If this is the case, it is important to know by how much carbon stocks are improved and could be improved in the future.

Combining all the above recommendations would ensure more robust predictions of future habitat carbon stocks and sequestration rates. The current evidence base used to predict carbon stocks and sequestration rates rely heavily on data from other regions around the world and a combination of different species not always relevant to the UK context.

7 Conclusions and Opportunities for Nature-based Solutions

7.1 Conclusions

Within the habitat chapters of this report we have demonstrated the importance of ecosystems for carbon storage and sequestration. Protecting, restoring and creating natural and semi-natural habitats is important both to prevent emissions and remove carbon dioxide from the atmosphere.

We have summarised the best available information for carbon storage and sequestration, in the English context, in figures 7.1 and 7.2 respectively. Comparing between habitats is complicated because the data have often been collected in different ways, for example with different depths of soils and sediments sampled, and we have had to use a mixture of field survey and modelled data. Carbon stocks and sequestration rates in marine habitats are also calculated differently to terrestrial habitats and include carbon originating from different sources. This makes direct numerical comparisons across habitats problematic, but necessary when attempting to understand how best to target habitat creation, restoration and management to mitigate climate change. The underlying data, which are derived from the information in the habitat chapters are available in Appendix A.

The numbers presented in figures. 7.1 and 7.2 are our best assessment of representative carbon storage and flux values. Because of the variability in data, some are means, some are medians, some are values associated with typical locations and habitat conditions. All are the result of expert judgement of the authors and have been reviewed by habitat experts in Natural England and external reviewers from the academic and conservation sectors. Where suitable data are not available to make this judgement, we have not included a habitat. In all cases there is significant variation between sites; the habitat specific chapters give more detail about this, where it is available, as well as setting out the evidence that has underpinned our decisions. In some cases, we have been able to give an indication of the range of variation between sites, where the data support this, in other cases this has not been possible.

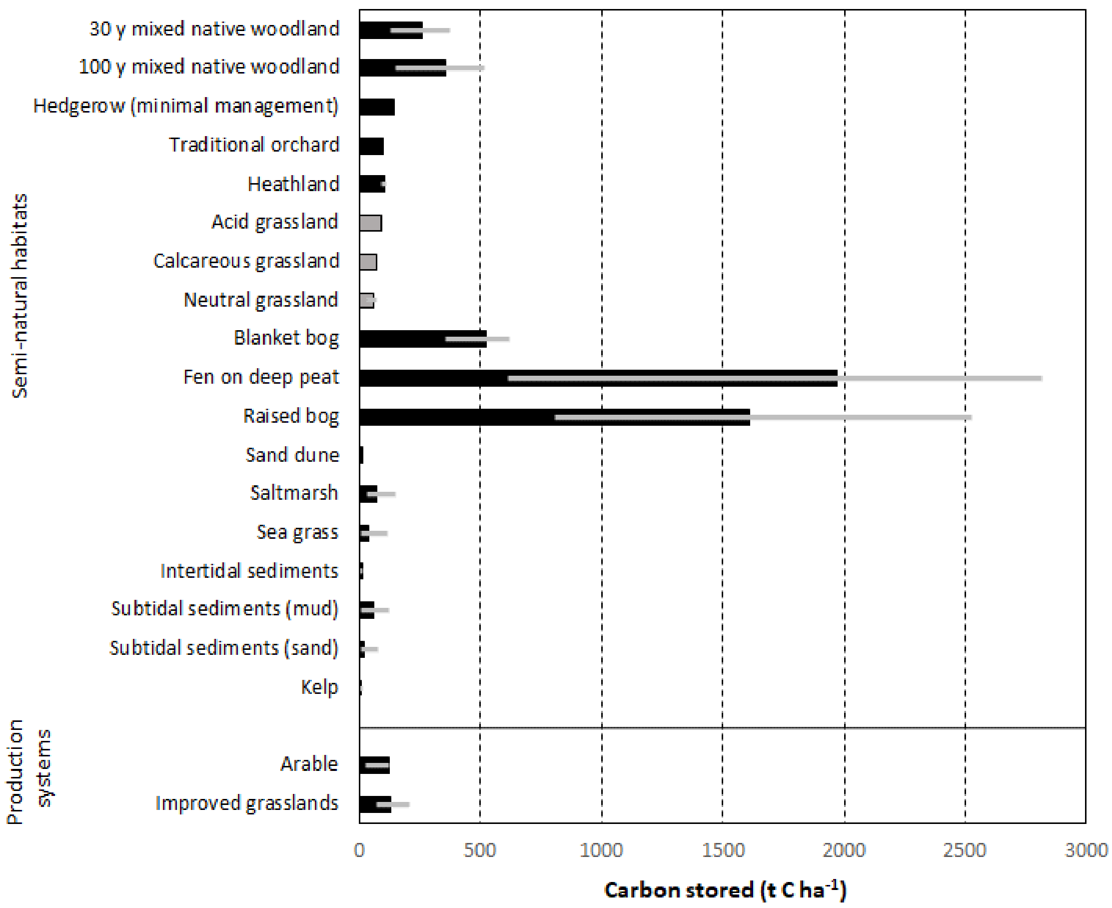


Figure 7.1 Carbon storage in contrasting habitats and land managements, using the best available data. Note that the semi-natural grasslands data are from the top 15 cm of soil only are shown in grey. Other habitats (shown in black) vary in their depths from 15 cm to 380 cm. Blanket bog carbon stocks are based on catchment scale estimates – see section 4.3. Fen data here are restricted to deep semi-natural fens; there are a range of other types – see Section 4.5. Numerical data and soil depths are provided within the review chapters and Appendix A

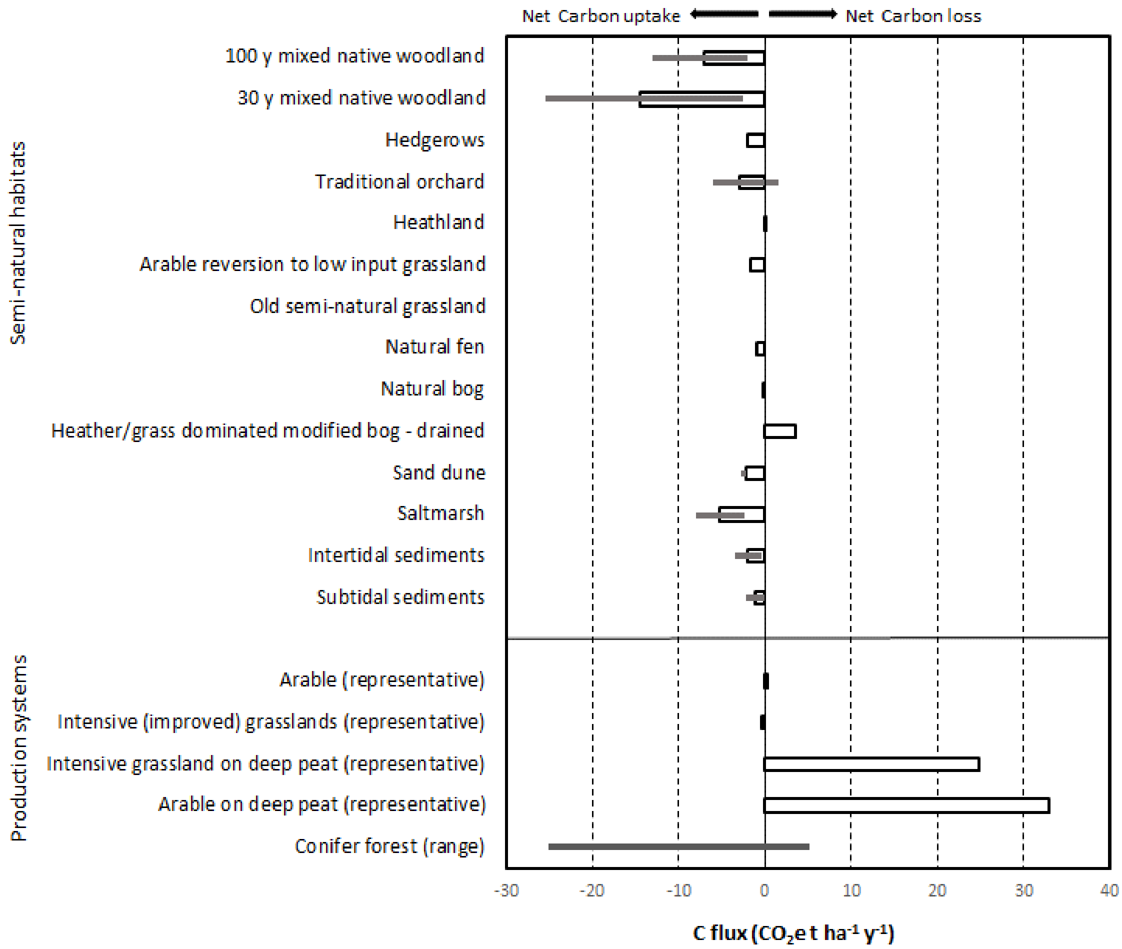


Figure 7.2 Carbon flux in contrasting habitats and land managements, using representative data. Best available data have been used and includes data from a wide range of different sources, modelled and field data. A negative value indicates sequestration, positive values are emissions. The grey bars indicate the likely range of values across sites where this is available. Habitats with no suitable data are not included and we refer the reader to the chapters where this is reviewed and discussed. Numerical data and soil depths are provided within the review chapters and Appendix B

Despite these caveats there are some clear qualitative patterns and unambiguous differences between habitats.

The largest carbon stores are in **peatlands**: when in healthy condition they sequester carbon slowly but are unique in that they can go on doing so indefinitely. Fens are a varied group of habitats, including some with very deep deposits. Raised bogs can also have peat many metres deep. Blanket bogs are important as the most extensive peatland, covering large areas of the uplands. However, most peatlands in England have been damaged by drainage, conversion to agriculture or forestry, burning, air pollution and over-grazing; resulting in them becoming a large source of greenhouse gas emissions, releasing carbon stored for centuries. In particular, the rates of emissions from peatlands under intensive agricultural management are extremely high. Restoration interventions will in many cases reduce these emissions in a few years although restoring the carbon sink function of peatlands may take decades.

The largest carbon sequestration rates are in **woodlands**. Native woodlands are reliable carbon sinks that continue to take up carbon over centuries. The net sequestration rate is slow over the first few years but can then increase quickly. Over time, sequestration declines in an unmanaged woodland, but remains significant over century timescales with old woodlands becoming substantial carbon stores. The rates of sequestration vary greatly with species, soil type and climate; and planting species in places where they can grow well, both now in future climates is essential to maximise carbon sequestration. Forest management by removing timber can maintain higher sequestration rates, but will also increase emissions by variable amounts, depending on what the timber is used for. Native woodland managed with a minimum intervention approach is likely to be a very effective climate change mitigation measure in many circumstances, with benefits for biodiversity and other ecosystem services.

Hedgerows, orchards and other **trees outside woodland** can also sequester and store carbon as well as providing other benefits within an agricultural and biodiversity context. It should however be remembered that the area covered is often relatively small compared to continuous woodland; also, in some cases, particularly hedgerows, they may be cut at regular intervals and any carbon gained would be lost again.

Some **coastal and marine habitats** are large carbon stores, although large evidence gaps remain for many. **Saltmarsh** in particular, can store carbon in similar amounts to many peatlands, although they are subject to erosion and accretion with natural coastal processes and are affected by changing sea levels. They may sequester significant amounts of carbon in situ, but they also trap and store carbon sequestered elsewhere. **Sea grass meadows** also have the potential to store large quantities of carbon within the sediments if undisturbed, although many have been lost.

Heathlands and **semi-natural grasslands** have been managed for centuries and require grazing or cutting to maintain them. They typically sequester and store more carbon than modern agricultural landscapes but store less carbon than peatlands, saltmarsh and old woodlands; and have relatively low net sequestration rates, although the underlying data is limited. Carbon is almost entirely stored in the soils of these habitats and stores are variable depending on climate, soil and management history, but can be significant. Protection of these areas, many of which have been lost over the last century is important for biodiversity and also prevents emissions.

River systems are important elements of ecosystem carbon cycling. They are hard to characterise on an area basis in the same way as other habitats, but evidence suggests that they are mostly atmospheric CO_2 sources. Their importance comes from the wider impact at a catchment scale, including the transport of dissolved and particulate organic carbon and the impact of natural and artificial drainage patterns on the hydrology of other habitats, particularly wetlands. Fluvial systems also act as a conduit between the terrestrial and marine environments. **Standing waters**, such as lakes and ponds can act as carbon sinks, storing carbon within the sediments long-term. However, too much terrestrial input can tip these systems from sinks to sources and research has shown that most standing waters are already supersaturated with CO_2 due to terrestrial nutrient input.

Much of England is managed for agricultural production. Arable land and intensively managed grassland typically have lower carbon storage and sequestration rates than semi-natural habitats. However, there is a large variation in carbon storage with soil type and carbon fluxes vary with management. Agricultural land can also be a significant source of emissions. The agricultural sector as a whole; including emissions from land, as well operational and livestock carbon emissions, which we have not addressed, is responsible for about 10 per cent of UK greenhouse gas emissions. However, it is possible to reduce net emissions by appropriate management of soils and inputs to increase soil carbon. It is also possible to increase carbon sequestration and storage at a landscape scale, by tree and hedge planting and creation of field margins and buffer strips.

Production forestry with non-native conifer species can sequester large amounts of carbon. Timber production is an important ecosystem service which can also help to reduce overall emissions by reducing reliance on fossil fuel intensive materials or energy sources. However, many forest products are not long-term carbon stores. Furthermore, high emissions of greenhouse gases are possible, especially on afforested peats, and net sequestration can be much reduced on other afforested organic soils. In the right places, native woodlands managed less intensively or by minimum intervention can be a good way to sequester carbon at the same time as protecting and enhancing biodiversity and providing other ecosystem services. To optimise climate change mitigation and other benefits it is important to ensure that the different types of woodland are created in places where they can have most benefit.

7.2 Opportunities – nature-based solutions for climate change mitigation

There is considerable interest in the concept of nature-based Solutions (NbS) for climate change mitigation and adaptation, including achieving the objective of net zero greenhouse gas emissions by 2050 (see introduction).

NbS is a key concept for tackling the climate and biodiversity crises. A joined-up approach that addresses both climate change and biodiversity decline together is the only realistic way of meeting the multiple demands on our environment. However, expectations of what

NbS can do need to be realistic. It will not be possible to offset anything close to current UK emissions across the different sectors of the economy through better environmental management alone. Tree growth is the most straightforward and best evidenced way to take carbon out of the atmosphere, but at the moment UK forests don't even offset the emissions of agriculture and other land uses. Forests removed about 4 per cent (18.2 Mt CO_2e) of UK greenhouse gas emissions in 2018 (Brown 2020b). The Committee on Climate Change (2020) presented a scenario for meeting net zero in which forest creation could sequester an additional 14 Mt CO_2e $ha^{-1} y^{-1}$ (excluding any storage or substitution benefits from harvested products) and peatland restoration could prevent 5 Mt CO_2e $ha^{-1}y^{-1}$ of emissions. Deep cuts in emissions in all sectors are therefore clearly required to achieve net zero, with offsetting reserved for a small residual amount of hard-to-eliminate emissions. We also have to be realistic that there is no easy solution that works everywhere; different approaches will be appropriate in different places. It is important to be rigorous in assessing how much difference any particular change in land use or management will make to biodiversity and climate in a particular place, and good spatial targeting will be essential to maximise benefits for both biodiversity and carbon without compromising food or timber production at a national scale.

Climate change adaptation to reduce the impacts of unavoidable climate change on people and nature is increasingly important; and new NbS should be evaluated to ensure that they will continue to be effective in a future which is warmer and subject to changes in rainfall, including more extreme events such as droughts and floods. There are NbS solutions to reduce these risks to people, such as natural flood management, and it may be possible to integrate these with carbon sequestration and storage in many cases. It is also important to consider the wider benefits of the natural environment to people, including to their health and wellbeing. We therefore increasingly need to identify solutions that provide a range of benefits. For example, a new native broadleaved woodland strategically placed in a catchment can sequester carbon, provide a habitat for many species, contribute to reducing flood risk and, reduce erosion into rivers, lakes and estuaries, and provide a place for people to exercise and enjoy being outdoors.

Key Principles

A number of key principles for climate change mitigation, in ways that maximise benefits for biodiversity, come out of our assessment:

1. **Protect and restore peatlands.** Peatlands are our largest natural carbon stores and it is important to slow and eventually halt greenhouse gas emissions, including through raising water tables, stopping burning and removing planted trees.

2. **Create new native broadleaved woodlands.** Native woodland is an effective carbon sink and over much of England can deliver comparable carbon uptake to non-native species and provide more benefits for biodiversity. Growing the right trees in the right place is however critical to maximise these benefits.

3. **Protect and restore natural coastal processes.** This allows habitats, such as saltmarsh, to maintain themselves and re-establish inland as the sea level rises, and

to sequester and store carbon. It is also an important and urgent aspect of climate change adaptation. Active intervention will be necessary to restore some habitats, such as seagrass.

4. **Protect existing semi-natural habitats.** Most of England has been intensively managed for a long time and semi-natural habitats, of all types, are rare fragments containing many of our native species that are not found elsewhere. Many of these, including grasslands and heathlands, also store appreciable amounts of carbon in their vegetation, undisturbed soils and sediments.

5. **Target incentives for NbS to places where they can have most benefit.** Different approaches work better in different places and it is important to maximise synergies and minimise trade-offs if we are to deliver net zero ambitions at the same time as restoring biodiversity and meeting the needs of people. Decisions about NbS need to consider the wider context of land use and management and the need to maintain, and where possible increase, domestic food and timber production in ways which do not lead to increased emissions either in the UK or overseas.

6. **Integrate NbS for climate into landscapes which are primarily devoted to agriculture or production forestry.** To meet the scale of change required in greenhouse gas emissions, there is a need to take land out of agriculture, particularly for woodland creation and peatland restoration. Actions such as hedgerow planting, good soil management and innovative agricultural approaches, such as paludiculture, can also contribute whilst enabling agricultural production to continue. Within production forest biodiversity can be supported by including broadleaved trees and appropriate management of forest rides and edges.

7. **Carry out research and monitoring to fill evidence gaps.** There are still large knowledge gaps for many habitats. For example, there is significant potential to increase carbon stocks for coastal and marine habitats, but we lack evidence in the English or UK context. Across all habitats, the carbon content of soils, sediments and vegetation, and ecosystem carbon fluxes are rarely measured. Even the depth of soil is rarely monitored. The role that freshwater habitats can play in climate change mitigation is also an understudied area.

8. **Ensure mitigation and adaptation to climate change are planned together.** This is important to ensure the durability of solutions for carbon sequestration and storage and to promote synergies rather than conflicts between objectives. We should look for multifunctional and integrated opportunities when planning our responses to the climate and biodiversity crises.

References

Introduction

Alonso, I. and others. 2012 Carbon storage by habitat - Review of the evidence of the impacts of management decisions and condition on carbon stores and sources. Natural England Research Reports, Number NERR043. Natural England.

Battin, T. and others. 2009. The boundless carbon cycle. *Nature Geoscience*. 2. 598–600.

Bradley, R. I. and others. 2005. A soil carbon and land use database for the United Kingdom. *Soil Use and Management.* 21: 363-369.

Brown, P. C. and others. 2020b. UK Greenhouse Gas Inventory, 1990 to 2018. Annual Report for Submission under the Framework Convention on Climate Change.

Carbon Brief. 2020. Analysis: UK's CO2 emissions have fallen 29% over the past decade. Web Page: https://www.carbonbrief.org/analysis-uks-co2-emissions-have-fallen-29-per-cent-over-the-past-decade Accessed: 1 March 2021.

Committee on Climate Change, 2018. Land use: Reducing emissions and preparing for climate change. Climate Change Committee, London.

Committee on Climate Change, 2020 a. The Sixth Carbon Budget: Agriculture & land use, land use change and forestry (LULUCF). Climate Change Committee, London.

Committee on Climate Change, 2020 b. Land Use: Policies for a Net Zero UK. Climate Change Committee, London.

Department for Business, Energy and Industrial Strategy. 2021 a. 2019 UK Greenhouse Gas Emissions, Final Figures. Web address: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/957887/2019_Final_greenhouse_gas_emissions_statistical_release.pdf Accessed: 1 March 2021

Department for Business, Energy and Industrial Strategy. 2021 b. Planned methodology changes for UK greenhouse gas emissions statistics 1990-2019 Web address: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/956525/ghg-national-statistics-methodology-changes-1990-2019.pdf Accessed: 1 March 2021

Field, R.H. and others. 2020. The value of habitats of conservation importance to climate change mitigation in the UK. *Biological Conservation*, 248, 108619.

Friedlingstein and others. 2020. Global Carbon Budget 2020. *Earth System Science Data.* 12. 3269-3340.

Griscom, B.W. and others. 2017. Natural Climate Solutions. *Proceedings of the National Academy of Sciences.* 114(44). 11645-11650.

IPCC, 2018. Global Warming of 1.5° C: An IPCC Special Report on the Impacts of Global Warming of 1.5° C Above Pre-industrial Levels and Related Global Greenhouse Gas Emission Pathways, in the Context of Strengthening the Global Response to the Threat of Climate Change, Sustainable Development, and Efforts to Eradicate Poverty. Intergovernmental Panel on Climate Change.

IPCC, 2019. Climate change and land. In: An IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse Gas Fluxes in Terrestrial Ecosystems. Intergovernmental Panel on

Climate Change. https://www.ipcc.ch/srccl/IUCN (International Union for Conservation of Nature).
IUCN Global Standard for Nature-Based Solutions: A User-Friendly Framework for the Verification, Design and Scaling Up of NbS. First edition. Gland, Switzerland: IUCN. https://doi.org/10.2305/IUCN. CH.2020.08.en.
Le Quéré C. and others. 2015. Global carbon budget 2014. *Earth System Science Data*. 7:47–85.
Macreadie, P.I. and others. 2017. Can we manage coastal ecosystems to sequester more blue carbon? *Frontiers in Ecology and the Environment.* 15: 206-213.
Ostle, N.J. and others. 2009. UK land use and soil carbon sequestration. Land Use Policy, 26S:274–83
Roberts, C.M., O'Leary B.C. & Hawkins JP. 2020. Climate change mitigation and nature conservation both require higher protected area targets. *Philosophical Transactions of the Royal Society*. 375. 20190121.
Sozanska-Stanton, M. and others. 2016. Balancing conservation and climate change – a methodology using existing data demonstrated for twelve UK priority habitats. *Journal for Nature Conservation.* 30. 76–89.

Woodlands, trees and scrub

Woodlands

Ashwood, F. and others. 2019. Woodland restoration on agricultural land: long-term impacts on soil quality. *Restoration Ecology,* 27**,** 1381-1392.

Bellamy, P.H. and others. 2005. Carbon losses from all soils across England and Wales 1978–2003. *Nature,* 437**,** 245-248.
Benham, S.E., Vanguelova, E.I. & Pitman, R.M., 2012. Short and long term changes in carbon, nitrogen and acidity in the forest soils under oak at the Alice Holt Environmental Change Network site. *Science of the total environment*, *421*, pp.82-93.
Brown, I. 2020a. Challenges in delivering climate change policy through land use targets for afforestation and peatland restoration. *Environmental Science & Policy,* 107**,** 36-45.
Brown, P. C. and others. 2020b. UK Greenhouse Gas Inventory, 1990 to 2018. Annual Report for Submission under the Framework Convention on Climate Change.
Butt, N. and others. 2009. Initial results from establishment of a long-term broadleaf monitoring plot at Wytham Woods, Oxford, UK. *University of Oxford Report*.
Butt, N. and others. 2014. Relationships between tree growth and weather extremes: spatial and interspecific comparisons in a temperate broadleaf forest. *Forest Ecology and Management,* 334**,** 209-216.
Campbell, D. and others. 2019. Peatlands and forestry. IUCN Peatland Programme
Committee on Climate Change. 2020. Land use: Policies for a Net Zero UK. CCC London.
Clement, R., Moncrieff, J.B. and Jarvis, P.G. 2003. Net carbon productivity of Sitka spruce forest in Scotland. *Scottish Forestry*. 57, 5-10.

Emmett, B.A. and others. 2007. *Countryside survey: soils report from 2007.* Centre for Ecology and Hydrology, Wallingford, UK

Evans, C. and others. 2017. Implementation of an Emissions Inventory for UK Peatlands. *Centre for Ecology and Hydrology. 1-88*.

Fenn, K. and others. 2015. The Carbon Cycle of a Maritime Ancient Temperate Broadleaved Woodland at Seasonal and Annual Scales. *Ecosystems***,** 1-15.

Fenn, K. M., Malhi, Y. & Morecroft, M. D. 2010. Soil CO2 efflux in a temperate deciduous forest: Environmental drivers and component contributions. *Soil Biology & Biochemistry,* 42**,** 1685-1693.

Field, R.H. and others. 2020. The value of habitats of conservation importance to climate change mitigation in the UK. *Biological Conservation*, 248, 108619.

Forestry Commission. 2017. The UK Forestry Standard. Forestry Commission, Edinburgh

Forestry Commission. 2020. Managing England's woodlands in a climate emergency A guide to help foresters and agents implement adaptation actions. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/872285/Climate_Change_Full_Guide.pdf

Forest Research. 2020. *Forestry statistics 2020*. Forest Research, Wrecclesham, UK.

Harding, P. T. & Rose, F. 1986. *Pasture woodlands in Lowland Britain: A review of their importance for wildlife conservation*. Natural Environment Research Council, Institute of Terrestrial Ecology.

Harmon, M. E. 2019. Have product substitution carbon benefits been overestimated? A sensitivity analysis of key assumptions. *Environmental Research Letters*, 14(6), 065008. Payne and Jessop.

Humphrey, J. W., Ferris, R., and Quine, C. 2003. Biodiversity in Britain's planted forests. *Forestry Commission, Edinburgh*, 118.

Jenkinson, D. S. 1990. The turnover of organic carbon and nitrogen in soil. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences,* 329**,** 361-368.

Kirbyk, K. and others. 2005. Long term ecological change in British woodland (1971–2001). *English Nature research reports,* 653**,** 1-139.

Matthews, K. and others. 2020. Not seeing the carbon for the trees? Why area-based targets for establishing new woodlands can limit or underplay their climate change mitigation benefits. *Land use policy,* 97**,** 104690.

Morecroft, M. D. and others. 2008. Effects of climate and management history on the distribution and growth of sycamore (Acer pseudoplatanus L.) in a southern British woodland in comparison to native competitors. *Forestry*, *81*(1), 59-74.

Morison, J. and others. 2012. Understanding the carbon and greenhouse gas balance of forests in Britain. *Research Report-Forestry Commission, UK*.

Natural England and RSPB. 2020. Climate Change Adaptation Manual (NE751) – Evidence to support nature conservation in a changing climate. http://publications.naturalengland.org.uk/publication/5679197848862720. Accessed Jan 2021.

Poulton, P. R., Pye, E., Hargreaves, P. R., & Jenkinson, D. S. (2003). Accumulation of carbon and nitrogen by old arable land reverting to woodland. *Global Change Biology*, *9*(6), 942-955.

Reynolds, B. and others. 2013. Countryside Survey: National “Soil Change” 1978–2007 for Topsoils in Great Britain—acidity, carbon, and total nitrogen status. *Vadose Zone Journal,* 12**,** 1-15.

Thomas, M. V. and others. 2011. Carbon dioxide fluxes over an ancient broadleaved deciduous woodland in southern England. *Biogeosciences*, 8(6), 1595-1613.

Upson, M.A., Burgess, P.J. & Morison, J.I.L., 2016. Soil carbon changes after establishing woodland and agroforestry trees in a grazed pasture. *Geoderma*, *283*, 10-20.

Vanguelova, E. I. and others. 2013. A new evaluation of carbon stocks in British forest soils. *Soil Use and Management,* 29**,** 169-181.

Vera, F.W.M. 2000. *Grazing ecology and forest history*. CABI publishing.

West, V. 2011. *Soil Carbon and the Woodland Carbon Code* [Online]. Woodland Carbon Code. Available: https://www.woodlandcarboncode.org.uk/images/PDFs/SoilCarbonandtheWoodlandCarbonCode_FINAL_14July2011.pdf [Accessed March 2021].

Wilkinson, M. and others. 2012. Inter-annual variation of carbon uptake by a plantation oak woodland in south-eastern England. *Biogeosciences,* 9**,** 5373-5389.

Woodland Carbon Code. 2021. WCC Carbon Calculation Spreadsheet V2.4. https://www.woodlandcarboncode.org.uk/images/Spreadsheets/WCC_CarbonCalculationSpreadsheet_Version2.4_March2021_bsml.xlsb. [Accessed March 2021].

Yamulkiy, S. & Morison, J. I. 2017. Annual greenhouse gas fluxes from a temperate deciduous oak forest floor. *Forestry: An International Journal of Forest Research,* 90**,** 541-552.

Hedgerows

Axe, M. 2015. Carbon measurement, prediction and enhancement of the agricultural hedgerow ecotone. PhD Thesis. Royal Agricultural University.

Axe, M. S., Grange, I. D. & Conway, J. S. 2017. Carbon storage in hedge biomass—A case study of actively managed hedges in England. *Agriculture, ecosystems & environment*, 250, 81-88.

Axe, M. 2020. Calculation of hedgerow carbon sequestration potential in Wales. Unpublished report for NRW.

Carey, P.D. and others 2008 Countryside Survey: UK Results From 2007. NERC/Centre for Ecology & Hydrology, CEH Project Number: C03259, 105.

Countryside Survey: England Results from 2007 (published September 2009). NERC/Centre for Ecology & Hydrology, Department for Environment, Food and Rural Affairs, Natural England, 119pp. (CEH Project Number: C03259).

Crossland, M. 2015. The carbon sequestration potential of hedges managed for woodfuel. https://www.organicresearchcentre.com/manage/authincludes/article_uploads/project_outputs/TWECOM%20ORC%20Carbon%20report%20v1.0.pdf [Accessed March 2021]

Falloon P., Powlson, D. & Smith, P. 2004. Managing field margins for biodiversity and carbon sequestration: a Great Britain case study. *Soil Use and Management*, 20, 240-247.

Firbank, L. & Bradbury, R. (2011). Enclosed farmland. Chapter 7. In: The UK National EcosystemAssessment Technical Report. UK National Ecosystem Assessment, UNEP-WCMC, Cambridge.

Follain, S. and others. 2007. Induced effects of hedgerow networks on soil organic carbon storage within an agricultural landscape. *Geoderma*, 142, 80-95.
Ford, H. and others. 2019. How do hedgerows influence soil organic carbon stock in livestock-grazed pasture? *Soil Use and Management*, 35, 576-584.
Ford, H. and others. 2021. Hedgerow effects on CO2 emissions are regulated by soil type and season: Implications for carbon flux dynamics in livestock-grazed pasture. *Geoderma*. 382 https://doi.org/10.1016/j.geoderma.2020.114697
Holden, J. and others. (2019). The role of hedgerows in soil functioning within agricultural landscapes. *Agriculture, Ecosystems & Environment*, 273, 1-12.
Natural England & RSPB. 2020. Climate Change Adaptation Manual, 2nd Edition.
O'Connell, P.E. and others. 2004. Review of Impacts of Rural Land Use and Management on Flood Generation. Impact Study Report, R&D Technical Report FD2114/TR Department for Environment, Food and Rural Affairs, London.
Rackham. 1986. The History of the Countryside. Dent Paperpacks. ISBN 0460125524
Robertson, H. and others. 2012. Economic, biodiversity, resource protection and social values of orchards: a study of six orchards by the Herefordshire Orchards Community Evaluation Project. Natural England Commissioned Report NECR090
Staley, J. T. and others. 2015. Re-structuring hedges: Rejuvenation management can improve the long-term quality of hedgerow habitats for wildlife in the UK. *Biological Conservation*. 186:187–196.
Thiel, B. and others. 2015. Using hedgerow biodiversity to enhance the carbon storage of farmland in the Fraser River delta of British Columbia. *Journal of Soil and Water Conservation*, 70(4). 247-256.
Warner, D.J., Tzilivakis, J. & Lewis, K.A. (2011) A revisit to previous research into the current and potential climate change mitigation effects of environmental stewardship. Final Report for Project BD5007 for Department for Environment, Food and Rural Affairs (Defra).
Wolton, R.J. and others. 2014. Regulatory services delivered by hedges: the evidence base. Report of Defra project LM0106. 99.

Traditional and intensively managed orchards

Anthony, R. 2013. Carbon Storage in Orchards. PhD thesis. Bangor University.
UK National Ecosystem Assessment (2011) The UK National Ecosystem Assessment Technical Report. UNEP-WCMC, Cambridge.
Countryside Survey: England Results from 2007 (published September 2009). NERC/Centre for Ecology & Hydrology, Department for Environment, Food and Rural Affairs, Natural England, 119pp. (CEH Project Number: C03259).
Demestihas, C., and others. 2017. Ecosystem services in orchards. A review. Agronomy for *Sustainable Development*. 37. 12. 10.1007/s13593-017-0422-1.
Wolton, R.J. and others. 2014. Regulatory services delivered by hedges: the evidence base. Report of Defra project LM0106. 99.
O'Connell and others. 2004. Review of Impacts of Rural Land Use and Management on Flood Generation. Impact Study Report, R&D Technical Report FD2114/TR Department for Environment, Food and Rural Affairs, London (2004)
Carey, P.D and others. 2008. Countryside Survey: UK Results From 2007. NERC/Centre for Ecology & Hydrology, CEH Project Number: C03259 p. 105

Montanaro, G. and others. 2017. Orchard management, soil organic carbon and ecosystem services in Mediterranean fruit tree crops, *Scientia Horticulturae*. 217, 2017, Pages 92-101,

Natural England and RSPB. 2020. Hedgerows in Climate Change Adaptation Manual (NE751) Accessed September 2020 http://publications.naturalengland.org.uk/publication/5679197848862720

Firbank, L. & Bradbury, R. (2011). Enclosed farmland. Chapter 7. In: The UK National Ecosystem Assessment Technical Report. UK National Ecosystem Assessment, UNEP-WCMC, Cambridge

Robertson, H. and others. 2012. Economic, biodiversity, resource protection and social values of orchards: a study of six orchards by the Herefordshire Orchards Community Evaluation Project. Natural England Commissioned Report NECR090

Woodland Trust. 2014. Hedgerows and Hedgerow Trees in Woodwise - Woodland Conservation News Summer 2014. https://www.woodlandtrust.org.uk/publications/2014/06/wood-wise-hedgerows-and-hedgerow-trees/ Accessed September 2020

Scrub habitats

Alberti, G. and others. 2011. Impact of woody encroachment on soil organic carbon and nitrogen in abandoned agricultural lands along a rainfall gradient in Italy. *Regional Environmental Change*. 11: 917-924.

Alberti, G. and others. 2008. Forest ecosystem carbon accumulation during a secondary succession in the Eastern Prealps of Italy. *Forestry*. 81(1).

Cook-Patton, S.C. and others. 2020. Mapping carbon accumulation potential from global natural forest regrowth. *Nature.* 585: 445-500 +extras.

Ferlan, M. and others. 2016. Carbon and water flux patterns of a drought-prone mid-succession ecosystem developed on abandoned karst grassland. *Agriculture, Ecosystems and Environment*. 220. 152-163.

Day, J., Symes, N. and Robertson, P. 2003 The Scrub Management Handbook: Guidance on the management of scrub on nature conservation sites. FACT/English Nature.

Guidi, C. and others. 2014 Changes in soil organic carbon and nitrogen following forest expansion on grassland in the Southern Alps. *Forest Ecology and Management.* 328: 103- 1 16.

Hunziker, M., Caviezel, C., Kuhn, N. 2017. Shrub encroachment by green alder on subalpine pastures: changes in mineral soil organic carbon characteristics. *Catena.* 157: 35-46.

Jackson, R.B. and others. 2002. Ecosystem carbon loss with woody plant invasion of grasslands. *Nature.* 418: 623-626.

Li, H. and others. 2016. Effects of shrub encroachment on soil organic carbon in global grasslands. *Scientific Reports.* 6:28974.

Lo, Y.H., Blanco, J.A., Canals, R.M., Gonzalez de Andres, E., San Emeterio, L., Bosco Imbert, J., Castillo, F.J. 2015. Land use change effects on carbon and nitrogen stocks in the Pyrenees during the last 150 years: a modelling approach. *Ecological Modelling*. 312: 322-334.

Mortimer, S.R. and others. 2000. The nature conservation value of scrub in Britain, August 2000, JNCC Report No. 308, JNCC, Peterborough, ISSN 0963-8091.

Nadal-Romero, E. and others. 2016. Effects of secondary succession and afforestation practices on soil properties after cropland abandonment in humid Mediterranean mountain areas. *Agriculture, Ecosystems and Environment*. 228: 91-100.
Nadal-Romero, E. and others. 2018. Woody encroachment and soil carbon stocks in subalpine areas in the Central Spanish Pyrenees. *Science of the Total Environment*. 636, pp. 727-736.
Thuille, A., Schulze, E.D. 2006. Carbon dynamics in successional and afforested spruce stands in Thuringia and the Alps. *Global Change Biology*. 12:325-342.
Uri V. and others. 2012. Biomass production and carbon sequestration in a fertile silver birch (Betula pendula Roth) forest chronosequence. *Forest Ecology and Management*. 267: 117-126.

Open habitats and farmland

Heathland

Aitkenhead, M.J. & Coull, M.C. (2016) Mapping soil carbon stocks across Scotland using a neural network model. *Geoderma*, 262: 187–198.
Alonso, I., Hartley, S.E. & Thurlow, M. (2001) Competition between heather and grasses on Scottish moorlands: interacting effects of nutrient enrichment and grazing regime. *Journal of Vegetation Science*, 12:249–60.
Alonso, I. and others. 2012 Carbon storage by habitat - Review of the evidence of the impacts of management decisions and condition on carbon stores and sources. Natural England Research Reports, Number NERR043. Natural England.
Bartlett, J. and others. 2020. Carbon storage in Norwegian ecosystems. NINA Report 1774. Norwegian Institute for Nature Research. Trondheim.
Britton, A. J. and others 2001, Impacts of climate, management and nitrogen deposition on the dynamics of lowland heathland. *Journal of Vegetation Science*, 12: 797–806.
Broadmeadow, M. & Matthews, R. (2003) Forest, carbon and climate change: the UK contribution. *Forestry Commission Information Note 48*.
Brown, I. 2020. Challenges in delivering climate change policy through land use targets for afforestation and peatland restoration. *Environmental Science and Policy*, 107: 36–45.
Cantarello, E., Newton, A.C. & Hill, R.A. 2011. Potential effects of future land-use change on regional carbon stocks in the UK. *Environmental Science & Policy*, 14: 40-52.
Carey, P.D. and others. 2016. Priority Habitats, Protected Sites and Climate Change: Three Investigations to Inform Policy and Management for Adaptation and Mitigation. DEFRA CR0439.
Clay, G.D. and others. 2010. Kinder-Grindsbrook wildfire 2008 - a case study. *Report to Moors for the Future*. Edale: Moors for the Future.
Clay, G. D., Worrall, F. & Rose, R. 2010. Carbon budgets of an upland blanket bog managed by prescribed fire. *Journal of Geophysical Research-Biogeosciences*, 115:G04037.
Condliffe, I. 2009. Policy change in the uplands. In: Bonn, A., Allott, T., Hubaceck, K. and Stewart, J. (Eds) *Drivers of environmental change in uplands*, pp 59–89. London: Routledge.

Cordingley, J.E. and others. 2015. Habitat Fragmentation Intensifies Trade-Offs between Biodiversity and Ecosystem Services in a Heathland Ecosystem in Southern England. *PLoS ONE.* 10(6): e0130004.
CS2007 - Countryside Survey: England Results from 2007 (published September 2009). NERC/Centre for Ecology & Hydrology, Department for Environment, Food and Rural Affairs, Natural England, 119pp. (CEH Project Number: C03259).
Dawson, J.J.C. & Smith, P. (2007) Carbon losses from soil and its consequences for land-use management. *Science of the Total Environment*: 382, 165-190.
De Vos, B. and others. 2015. Benchmark values for forest soil carbon stocks in Europe: Results from a large scale forest soil survey. *Geoderma* 251-252: 33-46.
de Vries, W. and others. 2009. The impact of nitrogen deposition on carbon sequestration by European forests and heathlands. *Forest Ecology and Management*. 258: 1814–1823. Link.
Defra 2007. The Heather and Grass Burning Code.
Dixon, S.D. and others. 2015. *Calluna vulgaris* canopy height and blanket peat CO_2 flux: Implications for management. *Ecological Engineering* 75, 497-505.
Douglas, D.J.T. and others. 2015. Vegetation burning for game management in the UK uplands is increasing and overlaps spatially with soil carbon and protected areas. *Biological Conservation*, 191: 243-250.
Dunford, R.W. & Berry, P.M. 2012. Climate change modelling of English amphibians and reptiles. Report to Amphibian and Reptile Conservation Trust (ARC-Trust).
Fagúndez, J. 2013. Heathlands confronting global change: drivers of biodiversity loss from past to future scenarios. *Annals of Botany*, 111(2): 151-172.
Field, C.D. and others. 2017. Long-term nitrogen deposition increases heathland carbon sequestration. *Science of the Total Environment* 592 426–435.
Field, R.H. and others. 2020. The value of habitats of conservation importance to climate change mitigation in the UK. *Biological Conservation.* 248.
Forgeard, F. & Frenot, Y. 1996. Effects of burning on heathland soil chemical properties: An experimental study on the effect of heating and ash deposits. *Journal of Applied Ecology*, 33, 803-811.
Friggens, N.L. and others. 2020. Tree planting in organic soils does not result in net carbon sequestration on decadal timescales. *Global Change Biology*.
Gimingham, C.H. 1972. Ecology of heathlands. London: Chapman and Hall.
Glaves, D.J. and others. 2020. The causes and prevention of wildfire on heathlands and peatlands in England. Natural England Evidence Review NEER014. Peterborough: Natural England.
Glaves, D.J. and others. 2013. The effects of managed burning on upland peatland biodiversity, carbon and water. Natural England Evidence Review, NEER004. Sheffield: Natural England.
Grau-Andrés, R. and others. 2018. Fire severity is more sensitive to low fuel moisture content on *Calluna* heathlands than on peat bogs. *Science of the Total Environment*, 616-617: 1261-1269.
Grau-Andrés, R., Gray, A., Davies, G.M., Scott, E.M. and Waldron, S. (2019) Burning increases post-fire carbon emissions in a heathland and a raised bog, but experimental manipulation of fire severity has no effect. *Journal of Environmental Management*, 233: 321-328.

Hartley, I.P. and others. 2012. A potential loss of carbon associated with greater plant growth in the European Arctic. *Nature Climate Change*, 2(12), 875–879.

Hawley, G. and others. 2008. Impact of heathland restoration and re-creation techniques on soil characteristics and the historic environment. Natural England Research Reports, Number NERR010. Natural England.

JNCC. 2008. Biodiversity: The UK Steering Group Report - Volume II: Action Plans (December 1995, Tranche 1, Vol 2).

JNCC. 2009a. Common Standards Monitoring Guidance for Lowland Heathland, Version February 2009, JNCC, Peterborough, ISSN 1743-8160.

JNCC. 2009b. Common Standards Monitoring Guidance for Upland Habitats, Version July 2009, JNCC, Peterborough, ISSN 1743-8160.

JNCC. 2019. Article 17 Habitats Directive Report 2019: Habitat Conservation Status Assessments.

Lee, H. and others. 2019. Soil carbon storage of coastal heathlands in Norway under different management practices. *16th European Heathland Workshop*.

Legg, C. & Davies, M. 2009. What determines fire occurrence, fire behaviour and fire effects in heathlands? In: Alonso, I. (ed.) Proceedings of the 10th National Heathland Conference – Managing Heathlands in the Face of Climate Change, 9-11 September 2008, York, UK. Natural England Commissioned Report 014. pp. 45-55.

Marrs, R.H. and others. 2019. Experimental evidence for sustained carbon sequestration in fire-managed, peat moorlands. Nature Geoscience, 12:108-112.

Mitchell, R.J. and others. 2007. The cascading effects of birch on heather moorland: a test for the top-down control of an ecosystem engineer. *Journal of Ecology*, 95, 540-554.

Morison, J. and others. 2012. Understanding the carbon and greenhouse gas balance of forests in Britain. Forestry Commission Research Report, Edinburgh.

Natural Capital Committee (NCC). 2020. Advice on using nature based interventions to reach net zero greenhouse gas emissions by 2050. Defra, London.

Natural England and RSPB. 2020. Climate Change Adaptation Manual (NE751) – Evidence to support nature conservation in a changing climate. http://publications.naturalengland.org.uk/publication/5679197848862720. Accessed Jan 2021.

Office for National Statistics (ONS). 2020. UK Natural Capital: Experimental carbon stock accounts, preliminary estimates.

Ostle, N.J. and others. 2009. UK land use and soil carbon sequestration. Land Use Policy, 26S:274–83.

Quin, S.L.O. and others. 2014. Restoration of upland heath from a graminoid- to a *Calluna vulgaris*-dominated community provides a carbon benefit. *Agriculture, Ecosystems & Environment*, 185: 133-143.

Quin, S.L.O. and others. 2015. *Calluna vulgaris*-dominated upland heathland sequesters more CO_2 annually than grass-dominated upland heathland. *Science of the Total Environment*, 505: 740–747.

Rodwell, J.S. (ed.). 1991. British Plant Communities 2: Mires and heaths. Cambridge: Cambridge University Press.

Sørensen, M.V. and others. 2018. Draining the Pool? Carbon Storage and Fluxes in Three Alpine Plant Communities. *Ecosystems*, 21:316-330.

Sozanska-Stanton, M. and others. 2016. Balancing conservation and climate change – a methodology using existing data demonstrated for twelve UK priority habitats. *Journal for Nature Conservation,* 30: 76–89. Link.
Sowerby, A. and others. 2008. Contrasting effects of repeated drought on soil carbon efflux in hydric and mesic heathland soils. *Global Change Biology*, 14, 2388–2404.
Thomas, A. and others. 2020. Patterns and trends of topsoil carbon in the UK: Complex interactions of land use change, climate and pollution. *Science of the Total Environment*, 729: 138330.
Urbina, I. and others. 2020. Encroachment of shrubs into subalpine grasslands in the Pyrenees changes the plant-soil stoichiometry spectrum. *Plant and Soil*, 448(1-2): 37-53.
Van der Wal, R. and others. 2011. Chapter 5: Mountains, Moorlands and Heaths, in: Anonymous UK National Ecosystem Assessment: Technical Report. UNEP-WCMC, Cambridge.
Van Paassen, J.G. and others. 2020. Legacy effects of nitrogen and phosphorus additions on vegetation and carbon stocks of upland heaths. *New Phytologist*. 228. 226-237.
Warner, D. 2008. Research into the current and potential climate change mitigation impacts of environmental stewardship. Report to Defra.
Warner, D.J. and others. 2020. Establishing a field-based evidence base for the impact of agri-environment options on soil carbon and climate change mitigation – phase 1. Final Report. Work package number: ECM50416. Evidence Programme Reference number: RP04176. Natural England.
Webb, J.R., Drewitt, A.L. & Measures, G.H. 2010. Managing for species: integrating the needs of England’s priority species into habitat management. Natural England Research Reports, Number NERR024, Natural England.
Webb, N. 1986. Heathlands. A natural history of Britain's lowland heaths. Pp 223, Collins, London.

Semi-natural grassland

Abdalla, M. and others. 2018. Critical review of the impacts of grazing intensity on soil organic carbon storage and other soil quality indicators in extensively managed grasslands. *Agriculture, ecosystems & environment*. 253, 62-81.
Allard, V. and others. 2007. The role of grazing management for the net biome productivity and greenhouse gas budget (CO2, N2O and CH4*) of semi-natural grassland. Agriculture, Ecosystems & Environment*, 121: 47-58.
Bengtsson, J. and others. 2019. Grasslands—more important for ecosystem services than you might think. Ecosphere, 10, e02582.
Blackstock, T.H. and others. 1999. The Extent of Semi-natural Grassland Communities In Lowland England and Wales: a Review of Conservation Surveys 1978-96. *Grass and forage science*, v. 54,.1 pp. 1-18.
Bullock, J.M. and others. 2011. Semi-natural grasslands. The UK National Ecosystem Assessment Technical Report. 161-196.
Chang, J. and others. 2016. Effect of climate change, CO2 trends, nitrogen addition, and land-cover and management intensity changes on the carbon balance of European grasslands. *Global Change Biology*, 22: 338-350.

Cole, A. J. and others. 2019. Grassland biodiversity restoration increases resistance of carbon fluxes to drought. *Journal of Applied Ecology*. 56, 1806-1816.

De Deyn, G. B. and others. 2011. Additional carbon sequestration benefits of grassland diversity restoration. *Journal of Applied Ecology.* 48, 600-608.

Emmett, B. and others. 2010. Countryside survey: soils report from 2007.

Eze, S. and others. 2018a. Soil organic carbon stock and fractional distribution in upland grasslands. *Geoderma*. 314, 175-183.

Eze, S., Palmer, S. M. & Chapman, P. J. 2018b. Soil organic carbon stock in grasslands: Effects of inorganic fertilizers, liming and grazing in different climate settings. *Journal of environmental management*. 223, 74-84.

Eze, S., Palmer, S. M. & Chapman, P. J. 2018c. Upland grasslands in Northern England were atmospheric carbon sinks regardless of management regimes. *Agricultural and Forest Meteorology*. 256, 231-241.

Field, R. and others. 2020. The value of habitats of conservation importance to climate change mitigation in the UK. *Biological Conservation.* 248. 108619.

Fornara, D. A., Banin, L. & Crawley, M. J. 2013. Multi-nutrient vs. nitrogen-only effects on carbon sequestration in grassland soils. *Global Change Biology.* 19, 3848-3857.

Fuller, R. M. 1987. The changing extent and conservation interest of lowland grasslands in England and Wales: a review of grassland surveys 1930–1984. *Biological conservation*. 40, 281-300.

Garnett, T. and others. 2017. Grazed and confused? Ruminating on cattle, grazing systems, methane, nitrous oxide, the soil carbon sequestration question – and what it all means for greenhouse gas emissions. Food Climate Research Network, University of Oxford.

Hagon, S. and others. 2013. Managing land for carbon: a guide for farmers, land managers and advisors. Lake District National Park Authority.

Isbell, F. and others. 2015. Biodiversity increases the resistance of ecosystem productivity to climate extremes. *Nature*. 526, 574-577.

Koncz, P. and others. 2015. Higher soil respiration under mowing than under grazing explained by biomass differences. *Nutrient Cycling in Agroecosystems*. 103, 201-215.

Kühnel, A. and others. 2019. Controlling factors of carbon dynamics in grassland soils of Bavaria between 1989 and 2016. *Agriculture, Ecosystems & Environment*. 280, 118-128.

Lei, T. and others. 2016. Drought and carbon cycling of grassland ecosystems under global change: A review. *Water.* 8, 460.

Medina-Roldán, E., Paz-Ferreiro, J. & Bardgett, R. D. 2012. Grazing exclusion affects soil and plant communities, but has no impact on soil carbon storage in an upland grassland. *Agriculture, Ecosystems & Environment.* 149, 118-123.

Natural England and RSPB. 2020. Climate Change Adaptation Manual (NE751) – Evidence to support nature conservation in a changing climate. http://publications.naturalengland.org.uk/publication/5679197848862720. Accessed Jan 2021.

Niklaus, P. A. and others. 2001. Effects of six years atmospheric CO2 enrichment on plant, soil, and soil microbial C of a calcareous grassland. *Plant and Soil.* 233, 189-202.

Office for National Statistics. 2015. UK Natural Capital Land Cover in the UK. http://www.ons.gov.uk/ons/rel/environmental/uk-natural-capital/index.html

Ridding, L. E., Redhead, J. W. & Pywell, R. F. 2015. Fate of semi-natural grassland in England between 1960 and 2013: A test of national conservation policy. *Global Ecology and Conservation*. 4, 516-525.
Pakeman, R.J. & Marrs, R.H. 1992 The conservation value of bracken Pteridium aquilinum (L.) Kuhn-dominated communities in the UK, and an assessment of the ecological impact of bracken expansion or its removal. *Biological Conservation.* 62, 101-114
Quin, S.L.O. and others. 2014. Restoration of upland heath from a graminoid- to a Calluna vulgaris-dominated community provides a carbon benefit. *Agriculture, Ecosystems & Environment*. 185: 133-143.
Quin, S.L.O. and others. 2015. Calluna vulgaris-dominated upland heathland sequesters more CO2 annually than grass-dominated upland heathland. *Science of the Total Environment.* 505: 740–747.
Rowe, E. C. and others. 2016. Productivity in a dominant herbaceous species is largely unrelated to soil macronutrient stocks. *Science of The Total Environment.* 572, 1636-1644.
Rowland, C.S. and others. 2017. Land Cover Map 2015 (25m raster, GB). NERC Environmental Information Data Centre.
Smith, P. 2014. Do grasslands act as a perpetual sink for carbon? *Global Change Biology*. 20, 2708-2711.
Soussana, J. F. and others. 2004. Carbon cycling and sequestration opportunities in temperate grasslands. *Soil use and management.* 20, 219-230.
Sozanska-Stanton, M. and others. 2016. Balancing conservation and climate change – a methodology using existing data demonstrated for twelve UK priority habitats. *Journal for Nature Conservation.* 30. 76-89.
Stevens, C. J. and others. 2004. Impact of nitrogen deposition on the species richness of grasslands. Science, 303, 1876-1879.
Stevens, C. J. and others. 2016. How will the semi-natural vegetation of the UK have changed by 2030 given likely changes in nitrogen deposition? *Environmental Pollution.* 208, 879-889.
Thomas, A. and others. 2020. Patterns and trends of topsoil carbon in the UK: Complex interactions of land use change, climate and pollution. *Science of The Total Environment.* 729, 138330.
Thuille, A. & Schulze, E.-D. 2006. Carbon dynamics in successional and afforested spruce stands in Thuringia and the Alps. *Global Change Biology.* 12, 325-342.
Tipping, E. and others. 2017. Long-term increases in soil carbon due to ecosystem fertilization by atmospheric nitrogen deposition demonstrated by regional-scale modelling and observations. *Scientific Reports.* 7, 1890.
Ward, S. E. and others. 2016. Legacy effects of grassland management on soil carbon to depth. *Global Change Biology.* 22, 2929-2938.
Warner, D.J. and others. 2020. Establishing a field-based evidence base for the impact of agri-environment options on soil carbon and climate change mitigation – phase 2. Final Report. Work package number: ECM50416. Evidence Programme Reference number: RP04176. Natural England.
Wilson, J.B. and others. 2012. Plant species richness: the world records. *Journal of Vegetation. Science.* 23, 796–802.

Arable land and agricultural grassland

Abdalla, M. and others. 2013. Conservation tillage systems: a review of its consequences for greenhouse gas emissions. *Soil Use and Management*. 29: 199-209.

Alonso, I. and others. 2012 Carbon storage by habitat - Review of the evidence of the impacts of management decisions and condition on carbon stores and sources. Natural England Research Reports, Number NERR043. Natural England.

Amanullah, D. R. 2019. Measuring and modelling soil carbon stocks and stock changes in livestock production systems –Guidelines for assessment. Food and Agriculture Organization of the United Nations. http://www.fao.org/3/CA2934EN/ca2934en.pdf.

Andrén, O. & Kätterer, T. (2001). Basic principles for soil carbon sequestration and calculating dynamic country-level balances including future scenarios. *Assessment Methods for Soil Carbon*: 495-511.

Bacq-Labreuil, A. and others. 2018. Effects of cropping systems upon the three-dimensional architecture of soil systems are modulated by texture. *Geoderma* 332.

Börjesson, G., and others. 2018. Organic carbon stocks in topsoil and subsoil in long-term ley and cereal monoculture rotations. *Biology and Fertility of Soils*. 54. 549-558.

Bradley, R. I. and others. 2005. A soil carbon and land use database for the United Kingdom. *Soil Use and Management.* 21: 363-369.

Brewer, K. & Gaudin, A. 2020. Potential of crop-livestock integration to enhance carbon sequestration and agroecosystem functioning in semi-arid croplands. *Soil Biology and Biochemistry*. 107936.

Cantarello, E., Newton, A. & Hill, R. (2011). Potential Effects of Future Land-Use Change on Regional Carbon Stocks in the UK. *Environmental Science and Policy.* 14: 40-52.

Clemensen, A. and others. 2020. Do plant secondary metabolite-containing forages influence soil processes in pasture systems? *Agronomy journal*. 112(5). 3744-3757.

Cowie, A., Smith, P. & Johnson, D. (2006). Does Soil Carbon Loss in Biomass Production Systems Negate the Greenhouse Benefits of Bioenergy? *Mitigation and Adaptation Strategies for Global Change*. 11: 979-1002.

van den Pol, A. and others. 2018. Grazing for carbon. In Sustainable meat and milk production from grasslands: Proceedings of the 27th General Meeting of the European Grassland Federation Cork, Ireland 17-21 June 2018 (pp. 682-684). (Grassland Science in Europe; Vol. 23). European Grassland Federation EGF.

Deyn, G. B., and others. 2010. Additional carbon sequestration benefits of grassland diversity restoration. *Journal of Applied Ecology*. 48: 600-608.

Distel, R. and others. 2020. Designing Diverse Agricultural Pastures for Improving Ruminant Production Systems. *Frontiers in Sustainable Food Systems*. 4: 596869.

Gilmullina, A. and others. 2020. Management of grasslands by mowing versus grazing – impacts on soil organic matter quality and microbial functioning. *Applied Soil Ecology*. 156: 103701.

Guenet, B. and others. 2020. Can N2O emissions offset the benefits from soil organic carbon storage? *Global Change Biology*. 2020; 00: 237-256.

Godde, C. and others. 2020. Soil carbon sequestration in grazing systems: managing expectations. *Climatic Change.* 161. 385-391.

Godwin, R. J. 2015. Potential of "no-till" systems for arable farming. Report to the Worshipful Company of Farmers, 44 pp. Newport, Shropshire, U.K.: Soil and Water

Management Centre, Harper Adams University. Retrieved from: https://cdn.harper-adams.ac.uk/document/project/Potential-of-No-till-Systems-for-Arable-Farming-Report.pdf. Accessed 02 December 2020.
Hartman, W. & Richardson, C. 2013. Differential Nutrient Limitation of Soil Microbial Biomass and Metabolic Quotients (qCO2): Is There a Biological Stoichiometry of Soil Microbes? *PLoS ONE*. 8: e57127.
Hijbeek R. and others. 2019. Fertiliser use and soil carbon sequestration: opportunities and trade-offs. CCAFS Working Paper No. 264. Wageningen, the Netherlands: CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS).
Hörtnagl, L. and others. 2018. Greenhouse gas fluxes over managed grasslands in Central Europe. *Global Change Biology.* 24. 1843-1872.
Humphreys, M. & O'Donovan, G. (2014). Comparing synthetic and natural grasslands for agricultural production and ecosystem service. *Grassland Science in Europe.* 19 :215-229
Jarvis, N. and others. 2017. Long-term effects of grass-clover leys on the structure of a silt loam soil in a cold climate. *Agriculture, Ecosystems & Environment.* 247: 319-328.
Jenkinson, D. 2001. The impact of humans on the nitrogen cycle, with focus on temperate arable agriculture. *Plant and Soil.* 228: 3-15.
Jobbágy, E. & Jackson, R. 2000. The Vertical Distribution of Soil Organic Carbon and Its Relation to Climate and Vegetation. *Ecological Applications.* 10: 423-436.
Johnston, A., Poulton, P. & Coleman, K. (2008). Chapter 1 Soil Organic Matter: Its Importance in Sustainable Agriculture and Carbon Dioxide Fluxes. *Advances in Agronomy.* 101: 1-57.
Johnston, A. and others. 2017. Changes in soil organic matter over 70 years in continuous arable and ley-arable rotations on a sandy loam soil in England: Carbon sequestration and losses over 70 years. European Journal of Soil Science 68. https://doi.org/10.1111/ejss.12415
Kätterer, T. and others. 2012. Strategies for carbon sequestration in agricultural soils in Northern Europe. *Acta Agriculturae Scandinavica*. 62:4, 181-198.
Kayser, M., Müller, J. & Isselstein, J. 2018. Grassland renovation has important consequences for C and N cycling and losses. *Food and Energy Security*. 7(4): e00146.
Knight, S. and others. 2019. Achieving sustainable intensification by integrating livestock into arable systems – opportunities and impacts. Defra Technical report
Le Clec'h, S. and others. 2019. Assessment of spatial variability of multiple ecosystem services in grasslands of different intensities. *Journal of Environmental Management*. 251 https://doi.org/10.1016/j.jenvman.2019.109372
Lorenz, K. & Lal, R. 2005. The Depth Distribution of Soil Organic Carbon in Relation to Land Use and Management and the Potential of Carbon Sequestration in Subsoil Horizons. *Advances in Agronomy.* 88: 35-66.
Lugato, Emanuele & Leip, Adrian & Jones, Arwyn. 201). Mitigation potential of soil carbon management overestimated by neglecting N2O emissions. *Nature Climate Change*. 8. 10.1038/s41558-018-0087-z.
Macdonald, C. and others. 2018. Soil Nutrients and Soil Carbon Storage. *Modulators, Mechanisms and Modelling*. 167-205.

McClelland, S., Paustian, K. & Schipanski, M. 2020. Management of cover crops in temperate climates influences soil organic carbon stocks: a meta-analysis. *Ecological Applications*. doi: 10.1002/eap.2278.
McNally, S. and others. 2015. Root carbon inputs under moderately diverse sward and conventional ryegrass-clover pasture: implications for soil carbon sequestration. *Plant and Soil*. 392.
Medhn, B. and others. 2020. Effects of long-term straw return on soil organic carbon storage and sequestration rate in North China upland crops: A meta-analysis. *Global Change Biology*. 26. https://doi.org/10.1111/gcb.15018
Morris J. and others. 2010. Restoration of Lowland Peatland in England and Impacts on Food Production and Security. Report to Natural England. Cranfield University, Bedford.
Moxley, J. and others. 2014. Capturing Cropland and Grassland Management Impacts on Soil Carbon in the UK LULUCF Inventory. Contract report prepared for the Department for Environment, Food and Rural Affairs. SP1113
Muhammed, S.E. and others. 2018. Impact of two centuries of intensive agriculture on soil carbon, nitrogen and phosphorus cycling in the UK. *Science of the Total Environment.* 634. 1486-1504.
Mulholland, B. and others. 2020. An assessment of the potential for paludiculture in England and Wales. Report to Defra for Project SP1218, 98 pp.
Musarika, S. and others. 2017. Effect of water table management and elevated CO_2 on radish productivity and on CH_4 and CO_2 fluxes from peatlands converted to agriculture. *Science of the Total Environment.* 584-585. 665-672.
Nüsse, A. and others. 2018. Effect of grassland harvesting frequency and N-fertilization on stocks and dynamics of soil organic matter in the temperate climate. *Archives of Agronomy and Soil Science*. 64:14. 1925-1931.
Olson, K. Ebelhar, S. & Lang, J. 2014. Long-Term Effects of Cover Crops on Crop Yields, Soil Organic Carbon Stocks and Sequestration. *Open Journal of Soil Science*. 04: 284-292.
Ostle, N. J. and others. 2009. UK land use and soil carbon sequestration. *Land Use Policy*. 26S: S274-S283.
Paustian, K. and others. 2019a. Quantifying carbon for agricultural soil management: from the current status toward a global soil information system. *Carbon Management.* 10: 1-21.
Paustian, K., Collins, H. & Paul, E. 2019b. Management Controls on Soil Carbon. In: Soil Organic Matter in Temperate Agroecosystems. CRC Press. 15-49.
Poeplau, C. & Don, A. 2015. Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis. *Agriculture Ecosystems & Environment*. 200: 33-41.
Powlson, D. S. and others. 2012. The potential to increase soil carbon stocks through reduced tillage or organic material additions in England and Wales: A case study. *Agriculture Ecosystems & Environment*. 146. Issue 1: Pages 23-33
Prade, T., Kätterer, T. & Björnsson, L. (2017). Including a one-year grass ley increases soil organic carbon and decreases greenhouse gas emissions from cereal-dominated rotations – A Swedish farm case study. *Biosystems Engineering*. 164: 200-212.
Rutledge, S. and others. 2017. The carbon balance of temperate grasslands part I: The impact of increased species diversity. *Agriculture, Ecosystems & Environment*. 239: 310-323.

Saffih-Hdadi, K. & Mary, B. 2008. Modeling consequences of straw residues export on soil organic carbon. *Soil Biology and Biochemistry*. 40: 594-607.
Schrumpf, M., E.-D, S., Kaiser, K. & Schumacher, J. 2011. How accurately can soil organic carbon stocks and stock changes be quantified by soil inventories? *Biogeosciences*. 8: 1193-1212.
Skinner, H. 2008. High Biomass Removal Limits Carbon Sequestration Potential of Mature Temperate Pastures. *Journal of Environmental Quality*. 37: 1319-1326.
Smith, P. and others. 2019. How to measure, report and verify soil carbon change to realise the potential of soil carbon sequestration for atmospheric greenhouse gas removal. *Global Change Biology* 26(1). 219-241.
Tautges, N.E. and others. 2019. Deep soil inventories reveal that impacts of cover crops and compost on soil carbon sequestration differ in surface and subsurface soils. *Global Change Biology*. 25: 3753-3766.
Tadt, H., Cross, P. & Jones, D.L. 2018. Efficacy of mitigation measures for reducing greenhouse gas emissions from intensively cultivated peatlands. *Soil Biology and Biogeochemistry*. 127. 10-21.
UNFCC. 1998. Kyoto protocol to the united nations framework. Review of European Comparative and International Environmental Law. 7: 214-217.
Ward, S. and others. 2016. Legacy effects of grassland management on soil carbon to depth. *Global Change Biology*. 22. https://doi.org/10.1111/gcb.13246
Wesemael, B. and others. 2011. How can soil monitoring networks be used to improve predictions of organic carbon pool dynamics and CO2 fluxes in agricultural soils? *Plant Soil*. 338: 247-259.
Westhoek, H., Overmars, K. & Zeijts, H. 2013. The provision of public goods by agriculture: Critical questions for effective and efficient policy making. *Environmental Science & Policy*. 32: 5–13.
Whitehead, D. and others. 2018. Management practices to reduce losses or increase soil carbon stocks in temperate grazed grasslands: New Zealand as a case study. *Agriculture, Ecosystems and Environment*. 265: 432-443.
Environment Agency. 2019. State of the environment – Soil. 2019. Accessed 8/10/20 https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/805926/State_of_the_environment_soil_report.pdf
Borrelli, P. and others. 2016. Effect of good agricultural and environmental conditions on erosion and soil organic carbon balance: a national case study. *Land Use Policy*. 50, 408-421.
Warner, D.J. and others. 2020. Establishing a field-based evidence base for the impact of agri-environment options on soil carbon and climate change mitigation – phase 1. Final Report. Work package number: ECM50416. Evidence Programme Reference number: RP04176. Natural England.
Wen, Z. and others. 2019. Tradeoffs among root morphology, exudation and mycorrhizal symbioses for phosphorus-acquisition strategies of 16 crop species. New Phytologist. 223: 882-895.
Louwagie, G., Gay, S. H. and Burrell, A. 2008. Sustainable Agriculture and Soil Conservation. Final report. EUR 23820 EN – 2009. DG Agriculture and Rural Development, European Commission.

Powlson, D.S. and others. 2014. Limited potential of no-till agriculture for climate change mitigation. *Nature Climate Change*. 4(8), 678.
van Groenigen, J.W. and others. 2017. Sequestering soil organic carbon: a nitrogen dilemma. *Environmental Science and Technology*. 51(9). 4738–4739.
Xiao, L. and others. 2020. Evaluating soil organic carbon stock changes induced by no-tillage based on fixed depth and equivalent soil mass approaches. *Agriculture, Ecosystems & Environment*. 300, 106982.

Blanket bogs, raised bogs and fens

Blanket and Raised Bogs

Aalen, F., K. Whelan, and M. Stout, eds. 1997. Atlas of the Irish Rural Landscape. Cork, Ireland: Cork University Press.
Abdalla, M.A. and others. 2016. Emissions of Methane from Northern Peatlands: A Review of Management Impacts and Implications for Future Management Options. *Ecology and Evolution.* 6 (19): 7080–7102.
Armstrong, A. and others. 2010. The impact of peatland drain-blocking on dissolved organic carbon loss and discolouration of water: results from a national survey. *Journal of Hydrology*. 381, 112e120.
Bain, C. and others. 2011. IUCN UK commission of inquiry on peatlands, IUCN UK Peatland Programme.
Baird, A.J., Holden, J. and Chapman, P. (2009) A literature review of evidence on emissions of methane in peatlands, Defra research project SP0574 final report. Defra.
Baird A.J. and others. 2019. Validity of managing peatlands with fire. *Nature Geoscience*. 12. 884-885.
Billett, M.F. and others. 2010. Carbon balance of UK peatlands: current state of knowledge and future research challenges. *Climate Research.* 45, 13-29.
Biodiversity Reporting and Information Group. 2008. Lowland Raised Bog from UK Biodiversity Action Plan Ed. A Maddock. JNCC. https://data.jncc.gov.uk/data/aadfff3d-9a67-467a-ac65-45285e123607/UKBAP-BAPHabitats-31-LowlandRaisedBog.pdf Accessed Jan 2021
Bonn, A. and others. 2016. Peatland Restoration and Ecosystem Services: Science, Policy and Practice. Cambridge University Press, Cambridge.
Brown, L.E. and others. 2014. Effects of moorland burning on the ecohydrology of river basins. Key findings from the EMBER project. University of Leeds.
Chapman S.J., Artz, R.R.E. & Poggio, L. 2015. Determination of organic carbon stocks in blanket peat soils in different condition - assessment of peat condition. Final Project Report submitted to: Scottish Environment Protection Agency https://www.sepa.org.uk/media/162691/sepa_carbonstocks_in_blanket-bog_final_report_a.pdf. Accessed Jan 2021.
Clay, G.D., Worrall, F. & Rose, R. 2010. Carbon budgets of an upland blanket bog managed by prescribed fire. *Journal of Geophysical Research-Biogeosciences.* 115, 14.
Clay, G.D. & Worrall, F. 2015 Carbon stocks and carbon fluxes from a 10-year prescribed burning chronosequence on a UK blanket peat. *Soil use and management.* 31(1). 39-51.

Climate Change Committee. 2020. Land use: Policies for a Net Zero UK. https://www.theccc.org.uk/publication/land-use-policies-for-a-net-zero-uk/ Accessed Jan 2021.
Cooper, M.D.A. and others. 2014. Infilled ditches are hotspots of landscape methane flux following peatland re-wetting. *Ecosystems.* 17(7). 1227-1241.
Couwenberg, J. and others. 2011. Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. *Hydrobiologia.* 674(1), 67-89.
De Deyn, G.B., Cornelissen, J.H.C. & Bardgett, R.D. 2008. Plant functional traits and soil carbon sequestration in contrasting biomes. *Ecological Letters.* 11, 516–531.
Dixon, S.D. and others. 2014. Restoration effects on water table depths and CO2 fluxes from climatically marginal blanket bogs. *Biogeochemistry.* 118. 159-176.
Dunn, C. and others. 2016. Plant Species Effects on the Carbon Storage Capabilities of a Blanket bog Complex. *Wetlands*. 36:47–58.
Evans, C. and others. 2016. Lowland peatland systems in England and Wales - evaluating greenhouse gas fluxes and carbon balances. Final report to Defra on Project SP1210, Centre for Ecology and Hydrology, Bangor. http://randd.defra.gov.uk/Document.aspx?Document=14106_Report_FINAL.pdf. Accessed December 2020.
Evans, C. and others. 2017. Implementation of an Emissions Inventory for UK Peatlands. Centre for Ecology and Hydrology.
Evans, C.D. and others. 2014. Contrasting vulnerability of drained tropical and high-latitude peatlands to fluvial loss of stored carbon. *Global Biogeochemical Cycles.* 28, 1215e1234
Evans C.D. and others. 2018. The impact of ditch blocking on fluvial carbon export from a UK blanket bog. *Hydrological Processes.* 32(13), 2141-2154.
Emmett, B.A. and others. 2008. Countryside Survey. Soils Manual. NERC/Centre for Ecology & Hydrology, 180 pp. (Annex 4:CS Technical Report No. 3/07).
Fenner, N. and others. 2007. Elevated CO2 effects on peatland plant community carbon dynamics and DOC production. *Ecosystems.* 10(4), 635-647.
Field, R.H. and others. 2020.The value of habitats of conservation importance to climate change mitigation in the UK. *Biological Conservation.* 248. 108619.
Fyfe, R.M. and others. 2014. The importance of sub-peat carbon storage as shown by data from Dartmoor, UK. *Soil Use and Management.* 30: 23-31.
Garnett, M.H., Ineson, P. & Stevenson, A.C. 2000. Effects of burning and grazing on carbon sequestration in a Pennine blanket bog, UK. *Holocene*. 10. 729-736.
Gatis, N. and others. 2020. Drain blocking has limited short-term effects on greenhouse gas fluxes in a Molinia caerulea dominated shallow peatland. *Ecological Engineering.* 158. 106079.
Glaves, D.J. and others. 2013. Natural England Review of Upland Evidence 2012 - The effects of managed burning on upland peatland biodiversity, carbon and water. Natural England Evidence Review, Number 004.
Granath, G. and others. 2016. Mitigating wildfire carbon loss in managed northern peatlands through restoration. *Scientific Reports.* 6. 28498.
Green, S.M. and others. 2016. The effect of drainage ditch blocking on the CO2 and CH4 budgets of blanket peatland. Appendix D in: Green, S.M., Baird, A.J., Evans, C., Ostle, N., Holden, J., Chapman, P.J., and McNamara, N. Investigation of Peatland

Restoration (Grip Blocking) Techniques to Achieve Best Outcomes for Methane and Greenhouse Gas Emissions/Balance: Field Trials and Process Experiments Final Report. Defra Project SP1202, University of Leeds, Leeds.

Green, S.M. and others. 2017. An experimental study on the response of blanket bog vegetation and water tables to ditch blocking. *Wetlands Ecology and Management.* 25(6), 703-716.

Green, S.M. and others. 2018. Methane and carbon dioxide fluxes from open and blocked ditches in a blanket bog. *Plant and Soil*. 424(1-2). 619-638.

Günther, A., Barthelmes, A., Huth, V. and others. 2020. Prompt rewetting of drained peatlands reduces climate warming despite methane emissions. *Nature Communications.* 11, 1644.

Heinemeyer, A. and others. 2020. Restoration of heather-dominated blanket bog vegetation on grouse moors for biodiversity, carbon storage, greenhouse gas emissions and water regulation: comparing burning to alternative mowing and uncut management. Final Report to Defra on Project BD5104, Stockholm Environment Institute at the University of York, York, UK.

Hambley, G., Andersen, R., Levy, P., Saunders, M., Cowie, N.R., Teh Y.A. & Hill, T.C. (2019): Net ecosystem exchange from two formerly afforested peatlands undergoing restoration in the Flow Country of northern Scotland. *Mires and Peat*. 23(05). 1-14.

Helfter, C., Campbell, C., Dinsmore, K. J., Drewer, J., Coyle, M., Anderson, M., Skiba, U., Nemitz, E., Billett, M. F., and Sutton, M. A. 2015. Drivers of long-term variability in CO2 net ecosystem exchange in a temperate peatland. *Biogeosciences.* 12, 1799–1811.

Hermans R, Zahn N, Andersen R, Teh YA, Cowie N, Subke J-A (2019) An incubation study of GHG flux responses to a changing water table linked to biochemical parameters across a peatland restoration chronosequence. *Mires and Peat*. 23. 8.

Hermans, R. (2018) Impact of forest-to-bog restoration on greenhouse gas fluxes. PhD thesis, University of Stirling. https://dspace.stir.ac.uk/bitstream/1893/27319/1/Thesis_final%2bcorrections.pdf [Accessed March 2021]

Holden, J., Chapman, P. J., & Labadz, J. C. (2004). Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration. *Progress in Physical Geography.* 28(1), 95-123.

Holden, J. and others. 2016. The impact of ditch blocking on the hydrological functioning of blanket peatlands. *Hydrological Processes.* 31. 525–539.

IUCN. 2018. UK Peatland Strategy 2018 – 2040. https://portals.iucn.org/library/sites/library/files/documents/2018-015-En.pdf. Accessed December 2020.

Joosten, H. and others. 2016. The role of peatlands in climate regulation. In A. Bonn, T. Allott, M. Evans, H. Joosten, & R. Stoneman (Eds.), Peatland Restoration and Ecosystem Services: Science, Policy and Practice (Ecological Reviews, pp. 63-76). Cambridge: Cambridge University Press.

Lindsay, R. 2010. Peatbogs and carbon: a critical synthesis to inform policy development in oceanic peat bog conservation and restoration in the context of climate change. RSPB Scotland, 315.

Lindsay, Richard. (2018). Peatland Classification. 10.1007/978-90-481-9659-3_341.

Lindsay, R., Birnie, R. & Clough, J. 2014. Peat Bog Ecosystems: Key Definitions. IUCN UK Committee Peatland Programme Briefing Note No. 1, https://www.iucn-uk-peatlandprogramme.org/sites/default/files/2019-05/1%20Definitions%20final%20-%205th%20November%202014.pdf Accessed Jan 2021

Lindsay, R., Birnie, R. & Clough, J. 2014. Commercial Peat Extraction". IUCN UK Committee Peatland Programme Briefing Note No. 6, http://www.iucn-uk-peatlandprogramme.org/sites/www.iucn-uk-peatlandprogramme.org/files/6%20Commercial%20peat%20extraction%20-%205th%20November%202014.pdf. Accessed Jan 2021

Lindsay, R. 2016. Peatland Classification. in: Finlayson, C. Max, Everard, Mark, Irvine, Kenneth, McInnes, and others (ed.) The Wetland Book: I: Structure and Function, Management, and Methods Springer Netherlands.

Loisel, J. & Bunsen, M. 2020. Abrupt Fen-Bog Transition Across Southern Patagonia: Timing, Causes, and Impacts on Carbon Sequestration. *Frontiers in Ecology and Evolution*. 8. 273.

Marrs, R. H. and others. 2018. Experimental evidence for sustained carbon sequestration in fire-managed, peat moorlands. *Nature Geoscience*. 12, 108–112.

McBride, A. and others. 2011. The Fen Management Handbook. Scottish Natural Heritage, Perth.

McLaughlin, J. 2004. Carbon Assessment of Boreal Wetlands of Ontario. Report number: FRIP 158 Ontario Ministry of Natural Resources and Forestry. Web page: https://www.researchgate.net/publication/310438143_Carbon_Assessment_in_Boreal_Wetlands_of_Ontario Accessed: March 2021

McNamara, N.P., and others. 2008. Gully hotspot contribution to landscape methane (CH4) and carbon dioxide (CO2) fluxes in a northern peatland. *Science of The Total Environment*. 404(2-3). 354-360.

Milne, R. & Brown, T.A. 1997. Carbon in the Vegetation and Soils of Great Britain. *Journal of Environmental Management.* 49(4). 413-433.

Moomaw, W. R. and others. 2018. Wetlands in a Changing Climate Science, Policy and Management. *Wetlands*. 38(2). 183–205.

Natural England. 2010. England's peatlands: carbon storage and greenhouse gases (NE257). http://publications.naturalengland.org.uk/publication/30021

Natural England. 2016. BogLIFE – Bringing Lowland Raised Bogs to Life. Issue 2. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/569985/boglife-newsletter-issue2.pdf Accessed March 2021

Natural England and RSPB. 2020. Climate Change Adaptation Manual (NE751) – Evidence to support nature conservation in a changing climate. http://publications.naturalengland.org.uk/publication/5679197848862720. Accessed Jan 2021.

Ostle, N.J. and others. 2009. UK land use and soil carbon sequestration. Land Use Policy 26S (2009) S274–S283.

Parry, L. E. and others. 2014a. Evaluating approaches for estimating peat depth. *Journal of Geophysical Research; Biogeosciences.* 119. 567– 576.

Parry, L.E., Holden, J.E. & Chapman, P.J. 2014b. Restoration of blanket peatlands. *Journal of Environmental Management.* 133. 193-205.

Qassim, S.M. and others. 2014. A 5-year study of the impact of peatland revegetation upon DOC concentrations. *Journal of Hydrology*. 519. 3578–3590.

Ratcliffe, J. and others. 2018. Contemporary carbon fluxes do not reflect the long-term carbon balance for an Atlantic blanket bog. *The Holocene*. 28, 140-149.

Renou-Wilson, F. and others. 2018. Rewetting degraded peatlands for climate and biodiversity benefits: Results from two raised bogs. *Ecological Engineering*. 127. 547–560.

Robson and others (2019) Commission of Inquiry on Peatlands Update: Peatlands and Forestry https://www.iucn-uk-peatlandprogramme.org/sites/default/files/2020-01/CoI%20Forestry%20and%20Peatlands_reduced%20size.pdf Accessed March 2021

Rodwell, J.S. (ed.). 1991. British Plant Communities 2: Mires and heaths. Cambridge: Cambridge University Press.

Rowson, J.G. and others. 2010. The complete carbon budget of a drained peat catchment. *Soil Use and Management*. 26. 261-273.

Shepherd, M. J. and others. 2013. Natural England review of upland evidence - Restoration of Degraded Blanket Bog. Natural England Evidence Review, Number 003.

Smith, P. and others. 2007a. ECOSSE: Estimating Carbon in Organic Soils—Sequestration and Emissions. Final Report. SEERAD Report. ISBN 978 0 7559 1498 2. 166.

Smith, P. and others. 2007b. Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978–2003. *Global Change Biology.* 13. 2605-2609.

Swenson, M.M. and others. 2019. Carbon balance of a restored and cutover raised bog: implications for restoration and comparison to global trends. *Biogeosciences.* 16. 713–731.

Trenbirth, H. & Dutton, A. 2019. UK natural capital: peatlands. https://www.ons.gov.uk/economy/environmentalaccounts/bulletins/uknaturalcapitalforpeatlands/naturalcapitalaccounts. Accessed Jan 2021.

Trumbore, S. E. and others. 1999. Carbon cycling in boreal wetlands: A comparison of three approaches. *Journal of Geophysical Research: Atmospheres.* 104. 27673-27682.

Tolonen, K. & Turunen, J. 1996. Accumulation rate of carbon in mires in Finland and implications for climate change. *The Holocene.* 6. 171-178.

Ward, S.E. and others. 2007. Long term consequences of grazing and burning on northern peatland carbon dynamics. *Ecosystems.* 10. 1069e1083.

Ward, S.E. and others. 2013. Warming effects on greenhouse gas fluxes in peatlands are modulated by vegetation composition. *Ecology Letters*. 16. 1285–1293.

Williamson, J. and others. 2017. Historical peat loss explains limited short-term response of drained blanket bogs to rewetting. *Journal of Environmental Management*. 188. 278–286.

Wilson, L., Wilson, J.M & Johnstone, I. 2011. The effect of blanket bog drainage on habitat condition and on sheep grazing, evidence from a Welsh upland bog. *Biological Conservation.* 144(1). 193 – 201.

Wilson, D. and others. 2015. Derivation of greenhouse gas emission factors for peatlands managed for extraction in the Republic of Ireland and the United Kingdom. *Biogeosciences.* 12(18). 5291.

Wilson, D. and others. 2016. Greenhouse gas emission factors associated with rewetting of organic soils [in special issue: Greenhouse gas fluxes in degraded and restored peatlands: global perspectives]. *Mires and Peat.* 17(4). 28.
Worrall, F. and others. 2010. Peatlands and climate change. Report to IUCN UK Peatland Programme, Edinburgh.
Worrall, F. and others. 2011. Carbon fluxes from eroding peatlands—the carbon benefit of revegetation following wildfire. Earth Surface Processes and Landforms. 36(11). 1487–1498.
Worrall, F. and others. 2011. A review of current evidence on carbon fluxes and greenhouse gas emissions from UK peatland. JNCC Report, No. 442.
Zhong, Y., Jiang, M., & Middleton, B.A. 2020. Effects of water level alteration on carbon cycling in peatlands. *Ecosystem Health and Sustainability*. 6(1). 1806113.

Fens

Brown. 2009. Carbon storage and Sequestration in the Somerset Levels, UK. A Desk based Assessment and Report. Commissioned by Somerset County Council (Project reference:2009/1). McBride, A., Diack, I., Droy, N., Hamill, B., Jones, P., Schutten, J., Skinner A. and Street, M. (2011) The Fen Management Handbook. Scottish Natural Heritage, Perth.
Cooper, M.D.A. and others. 2014. Infilled ditches are hotspots of landscape methane flux following peatland re-wetting. Ecosystems, 17 (7). 1227-1241. 10.1007/s10021-014-9791-3
Evans, C. and others. 2016. Lowland peatland systems in England and Wales - evaluating greenhouse gas fluxes and carbon balances. Final report to Defra on Project SP1210, Centre for Ecology and Hydrology, Bangor. http://randd.defra.gov.uk/Document.aspx?Document=14106_Report_FINAL.pdf
Callaghan, D., (2012). A Survey of flushes on the Long Mynd, Shropshire. Report to Natural England.
Holman, IP. 2009. An estimate of peat reserves and loss in the East Anglian Fens, Commissioned by the RSPB. Cranfield University. https://www.rspb.org.uk/globalassets/downloads/documents/positions/agriculture/reports/an-estimate-of-peatreserves-and-loss-in-the-east-anglian-fens-.pdf
Mulholland, B. and others. 2020. An assessment of the potential for paludiculture in England and Wales. Report to Defra for Project SP1218, 98 pp.
Natural England (2010) England's peatlands: carbon storage and greenhouse gases (NE257).
ONS. 2019 UK natural capital: peatlands. Office for National Statistics. Available from: http://tiny.cc/ONSPeatNC. Accessed 08/01/2021.
Peacock, M. and others. 2017. Management effects on greenhouse gas dynamics in fen ditches. *Science of the Total Environment*. 578, 601-612. https://doi.org/10.1016/j.scitotenv.2016.11.005
Peacock, M. and others. 2019. The full carbon balance of a rewetted cropland fen and a conservation-managed fen. *Agriculture, Ecosystems & Environment*. (269) 1-12.
Stanley, K. M. and others. 2019. The importance of CH_4 ebullition in floodplain fens. *Journal of Geophysical Research: Biogeosciences*. 124, 1750– 1763.

Tratt, R., Eades, P., & Shaw, S.C. 2012. Alkaline Fen & Transition Mire Survey of the North York Moors National Park & Bishop Monkton Ings. Report to Natural England, Telford.
Telford. J.R. 2015. Survey of Alkaline Fens in the western Lake District. Report to Natural England, Telford
Vermaat, J.E. and others. 2011. Greenhouse gas fluxes from Dutch peatland water bodies: importance of the surrounding landscape. *Wetlands*, 31 493-498. DOI 10.1007/s13157-011-0170-y

Rivers, Lakes and Wetlands

Addy S. and others (2016) River Restoration and Biodiversity: Nature-based solutions for restoring rivers in the UK and Republic of Ireland. CREW reference: CRW2014/10.
Alin, S. R., & Johnson, T. C. 2007, Carbon cycling in large lakes of the world: A synthesis of production, burial, and lake-atmosphere exchange estimates, *Global Biogeochem. Cycles*, 21, GB3002.
Anderson N.J., and others. 2020. Anthropogenic alteration of nutrient supply increases the global freshwater carbon sink. *Science Advances*. 6(6): 2375-2548.
Aquatic restoration partnership: http://aquaticrestorationpartnership.org.uk/upland-lakes/ [Accessed October 2020].
Aufdenkampe, A.K. and others. 2011, Riverine coupling of biogeochemical cycles between land, oceans, and atmosphere. *Frontiers in Ecology and the Environment*, 9: 53-60.
Biddanda, B. A. 2017. Global significance of the changing freshwater carbon cycle. Eos 98.
Billett, M.F., Garnett, M.H. & Dinsmore, K.J. 2015. Should aquatic CO2 evasion be included in contemporary carbon budgets for peatland ecosystems? Ecosystems, 18(3), pp. 471-480.
Billett, M.F. & Harvey, F.H. 2013. Measurements of CO_2 and CH_4 evasion from UK peatland headwater streams. *Biogeochemistry* 114**,** 165–181.
Bullinger-Weber, G. and others. 2014. Carbon storage and soil organic matter stabilisation in near-natural, restored and embanked Swill floodplains. *Geoderma.* 228-229:122-131.
Bussi, G. and others. 2016. Modelling the future impacts of climate and land-use change on suspended sediment transport in the River Thames (UK). *Journal of Hydrology*. 542: 357-372.
Butman, D. and others. 2015. Increased mobilization of aged carbon to rivers by human disturbance. *Nature Geoscience.* 8, 112–116.
Cierjacks, A. and others. 2011. Organic matter distribution in floodplains can be predicted using spatial and vegetation structure data. *River Research and Applications*. 27. 1048-1057.
Curtis, C.J. and others. 2014. The future of upland water ecosystems of the UK in the 21st century: A Synthesis. *Ecological Indicators.* 37: 412-430.
Dawson, J.J. and others. 2002. A comparison of particulate, dissolved and gaseous carbon in two contrasting upland streams in the UK. *Journal of Hydrology*. 257: 1-4: 226-246.
Deemer, B.R. and others. 2016. Greenhouse Gas Emissions from Reservoir Water Surfaces: A New Global Synthesis, *BioScience*. 66(11). 949–964.

Dinsmore, K.J., Billett, M.F. & Dyson, K.E. 2013. Temperature and precipitation drive temporal variability in aquatic carbon and GHG concentrations and fluxes in a peatland catchment. Global Change Biology, 19: 2133-2148.
Downing JA. 2010. Emerging global role of small lakes and ponds: little things mean a lot. *Limnetica.* 29:9–24.
Entwistle N.S. and others (2019) Recent changes to floodplain character and functionality in England. *Catena.* 174. 490-498.
Environment Agency, 2020. National Flood and Coastal Erosion Risk Management Strategy for England. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/920944/023_15482_Environment_agency_digitalAW_Strategy.pdf Accessed: December 2020.
Evans, C.D., Monteith, D.T. & Cooper D.M. 2005. Long-term increases in surface water dissolved organic carbon: Observations, possible causes and environmental impacts. *Environmental Pollution*. 137(1) 55-71.
Floodplain Meadows partnership 2018. Soil carbon stocks in UK grasslands. A brief summary of the literature, 2018. http://www.floodplainmeadows.org.uk/sites/www.floodplainmeadows.org.uk/files/Soil%20Carbon%20stocks%20summary%20130619.pdf Accessed: March 2021.
Gilbert PJ and others. 2014. Variations in sediment organic carbon among different types of small natural ponds along Druridge Bay, Northumberland, UK. *Inland Waters* 4: 57– 64.
Grinham, A. and others (2018) The importance of small artificial water bodies as sources of methane emissions in Queensland, Australia. Hydrology and Earth System Sciences. 22. 5281-5298.
Hope, D. and others. 2001, Carbon dioxide and methane evasion from a temperate peatland stream. *Limnology and Oceanography*, 46(4). 847-857.
Huttunen, J.T. and others. 2002. Fluxes of CH4, CO2, and N2O in hydroelectric reservoirs Lokka and Porttipahta in the northern boreal zone in Finland. *Global Biogeochemical Cycles* 16(3). 3-17.
Joint Nature Conservation Committee. 2008. UK Biodiversity Action Plan Priority Habitat Descriptions: Eutrophic Standing Waters. Retrieved Jan 2021, from http://jncc.defra.gov.uk/page-5706.
Lawson, C. and others. 2018. The natural capital of floodplains: management, protection and restoration to deliver greater benefits. Valuing Nature Natural Capital Synthesis Report VNP09.
Maberly, S. and others. 2013. Catchment productivity controls CO_2 emissions from lakes. *Nature Climate Change.* 3. 391–394.
Maltby, E. & Acreman, M.C. 2011. Ecosystem services of wetlands: pathfinder for a new paradigm. *Hydrological Sciences Journal*. 56:8. 1341-1359.
McEnroe, N. A. and others. 2009. Do pool surface area and depth control CO2 and CH4 fluxes from an ombrotrophic raised bog, James Bay, Canada? *Journal of Geophysical Research*: *Biogeosciences*. 114. G01001.
Mainstone, C.P., Hall, R. and Diack, I. (2016) A narrative for conserving freshwater and wetland habitats in England. Natural England Research Reports, Number 064.

Natural England and RSPB. 2020. Climate Change Adaptation Manual (NE751) – Evidence to support nature conservation in a changing climate. http://publications.naturalengland.org.uk/publication/5679197848862720. Accessed Jan 2021.

Romeijn, P. and others. 2019. Streambed organic matter controls on carbon dioxide and methane emissions from streams. *Environmental Science & Technology. 53*(5), 2364-2374.

Pawson, R.R. and others. 2008. Fluvial organic carbon flux from an eroding peatland catchment, southern Pennines, UK. *Hydrology and Earth System Sciences Discussions, European Geosciences Union*. 12(2). 625-634.

Peacock, M., and others. 2013. Natural revegetation of bog pools after peatland restoration involving ditch blocking – the influence of pool depth and implications for carbon cycling. *Ecological Engineering.* 57. 297-301.

Pelletier, L. and others. 2014. Carbon release from boreal peatland open water pools: implication for the contemporary C exchange. *Geophysical Research Biogeosciences*. 2013JG002423.

Rothero, E., Lake, S. & Gowing, D. (eds) (2016). Floodplain Meadows – Beauty and Utility. A Technical Handbook. Milton Keynes, Floodplain Meadows Partnership.

Sanches, L.F. and others. 2019. Global regulation of methane emission from natural lakes. *Scientific Reports.* 9. 255.

Sánchez-Rodríguez, A.R. and others. 2019. Extreme flood events at higher temperatures exacerbate the loss of soil functionality and trace gas emissions in grassland. *Soil Biology and Biochemistry*. 130. 227-236.

Sanders, I.A. and others. 2007. Emission of methane from chalk streams has potential implications for agricultural practices. *Freshwater Biology*, 52. 1176-1186.

Scott, D.R. 2014. Carbon fixation, flux and burial efficiency in two contrasting eutrophic lakes in the UK (Rostherne Mere & Tatton Mere). Loughborough University. Thesis. https://hdl.handle.net/2134/16949. Accessed December 2020.

Stimson A.G. and others. 2017. Reservoirs as hotspots of fluvial carbon cycling in peatland catchments. *Science of the Total Environment* 580: 398-411.

Stevenson, M.A. and others. 2016. Impacts of forestry planting on primary production in upland lakes from north-west Ireland. *Global Change Biology*, 22: 1490-1504.

Swinnen, W. and others. 2020. Geomorphic controls on floodplain sediment and soil organic carbon storage in a Scottish mountain river. *Earth Surface Processes and Landforms*, 45: 207– 223.

Taylor, S.T. and others. 2019. High carbon burial rates by small ponds in the landscape. *Frontiers in Ecology and the Environment.* 17(1). 25– 31.

Teodoru, C.R. and others. 2011. Spatial heterogeneity of surface CO2 fluxes in a newly created Eastmain-1 reservoir in Northern Quebec, Canada. *Ecosystems*. 14. 28–46.

Turner, T.E. and others. 2016. Regional variation in the biogeochemical and physical characteristics of natural peatland pools. *Science of the Total Environment*. 545-546, 84-94.

Walling, D.E. and others. 2006. Sediment Dynamics and the Hydromorphology of Fluvial Systems. (Proceedings of a symposium held in Dundee, UK, July 2006). IAHS Publ. 30.

Watts, G. and others. 2015. Climate change and water in the UK – past changes and future prospects. *Progress in Physical Geography: Earth and Environment*. 39(1). 6-28.
Webb, J.R. and others. 2019. Regulation of carbon dioxide and methane in small agricultural reservoirs: optimizing potential for greenhouse gas uptake. *Biogeosciences*. 16, 4211–4227.
Williamson J.L and others (2021) Landscape controls on riverine export of dissolved organic carbon from Great Britain. *Biogeochemistry.*
Worrall, F. and others. 2003. Carbon budget for a British upland peat catchment, *Science of the Total Environment.* 312, 133–146.
Worrall, F. & Lancaster, A. 2005. The Release of CO_2 from River waters – the Contribution of Excess CO_2 from Groundwater. *Biogeochemistry* 76, 299–317.
Worrall, F. Burt, T.P and Howden, N.J.K. (2016) The fluvial flux of particulate organic matter from the UK: the emission factor of soil erosion. *Earth surface processes and Landforms*. 41. 61-71.
Worrall, F. and others. 2020. The dissolved organic carbon flux from the UK – A new Bayesian approach to flux calculation. *Journal of Hydrology*, 590: 125511.
Zehetner, F. and others. 2009. Rapid carbon accretion and organic matter pool stabilization in riverine floodplain soils, *Global Biogeochemical Cycles*, 23, GB4004.
WWF (2018) Saving the Earth: A sustainable future for soils and water. https://www.wwf.org.uk/sites/default/files/2018-04/WWF_Saving_The_Earth_Report_HiRes_DPS_0.pdf. Accessed Jan 2021.

Coastal and marine

Abdullah, M. and others. 2004. Production, respiration and exudation of dissolved organic matter by the kelp Laminaria hyperborea along the west coast of Norway. *Journal of the Marine Biological Association of the United Kingdom.* 84(5), 887-894.
Adams, C. A., Andrews, J.E. & Jickells, T. 2012. Nitrous oxide and methane fluxes vs. carbon, nitrogen and phosphorous burial in new intertidal and saltmarsh sediments. *Science of the Total Environment*. 434. 240-251.
Alongi, D. M. 2018. Kelp Forests. Blue Carbon: Coastal Sequestration for Climate Change Mitigation. D. M. Alongi. Cham, Springer International Publishing. 53-57.
Andrews, J. E. and others. 2006. Biogeochemical value of managed realignment, Humber estuary, UK. *Science of the Total Environment*. 371(1-3). 19-30.
Armstrong, S. and others. 2020. Estimating the Carbon Sink Potential of the Welsh Marine Environment. NRW, Cardiff, 74p.
Bauer, J. E. and others. 2013. The changing carbon cycle of the coastal ocean. *Nature*. 504(7478). 61-70.
Beaumont, N. J. and others. 2014. The value of carbon sequestration and storage in coastal habitats. *Estuarine, Coastal and Shelf Science.* 137. 32-40.
Borges, A. V., Delille, B. & Frankignoulle, M. 2005. Budgeting sinks and sources of CO_2 in the coastal ocean: Diversity of ecosystems counts. *Geophysical Research Letters*. 32(14).
Burden, A., Garbutt, A. and Evans, C.D. 2019. Effect of restoration on saltmarsh carbon accumulation in Eastern England. Biology Letters. 15(1). 20180773.
Burden, A. and others. 2013. Carbon sequestration and biogeochemical cycling in a saltmarsh subject to coastal managed realignment. *Estuarine, Coastal and Shelf Science*. 120. 12-20.

Burden, A. and others. 2020. Impacts of climate change on coastal habitats, relevant to the coastal and marine environment around the UK. http://www.mccip.org.uk/impacts-report-cards/full-report-cards/2020. Accessed Feb 2021.

Burrows, M. T. and others. 2014. Assessment of carbon budgets and potential blue carbon stores in Scotland's coastal and marine environment. Project Report. Scottish Natural Heritage Commissioned Report No. 761. 7, 90 pp.

Cannell, M. G. R. and others. 1999. National inventories of terrestrial carbon sources and sinks: The U.K. Experience. *Climatic Change*. *42*(3). 505–530.

Clarke, D. & Ayutthaya, A. 2010. Predicted effects of climate change, vegetation and tree cover on dune slack habitats at Ainsdale on the Sefton Coast, UK. *Journal of Coastal Conservation*. 14. 115-125.

Couto, T. and others. 2013. Salt marsh plants carbon storage in a temperate Atlantic estuary illustrated by a stable isotopic analysis based approach. *Ecological Indicators*. 32. 305-311.

De Haas, H., Boer, W. & Van Weering, T.C.E. 1997. Recent sedimentation and organic carbon burial in a shelf sea: The North Sea. *Marine Geology*, *144*(1–3)

Diesing, M. and others. 2017. Predicting the standing stock of organic carbon in surface sediments of the North–West European continental shelf. *Biogeochemistry*, *135*(1–2).

Duarte, C.M., Middelburg, J. J. and Caraco, N. 2005. Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences*. *1*. 1–8.

Filbee-Dexter, K. & Wernberg, T. 2020. Substantial blue carbon in overlooked Australian kelp forests. *Scientific reports.* 10. 12341.

Fodrie, F. J. and others. 2017. Oyster reefs as carbon sources and sinks. Proceedings of the Royal Society. *Biological Sciences*. 284(1859). 20170891.

Fourqurean, J. W. and others. 2012. Seagrass ecosystems as a globally significant carbon stock. *Nature Geoscience*. 5(7). 505-509.

Ford, H. and others. 2012. Methane, carbon dioxide and nitrous oxide fluxes from a temperate salt marsh: Grazing management does not alter Global Warming Potential. *Estuarine, Coastal and Shelf Science*. *113*. 182–191.

Ford, H. and others. 2019. Large-scale predictions of salt-marsh carbon stock based on simple observations of plant community and soil type. *Biogeosciences*. *16*(2). 425–436.

Garrard, S. L. & Beaumont, N.J. (2014). The effect of ocean acidification on carbon storage and sequestration in seagrass beds; a global and UK context. Marine Pollution Bulletin. 86(1-2). 138-146.

Gevaert, F., Janquin, M.A. and Davoult, D. (2008). Biometrics in Laminaria digitata: A useful tool to assess biomass, carbon and nitrogen contents. *Journal of Sea Research.* 60(3). 215-219.

Green, A., Chadwick, M.A. and Jones, P.J.S. 2018. Variability of UK seagrass sediment carbon: Implications for blue carbon estimates and marine conservation management. *PLoS One*. 13(9). e0204431.

Green, A. and others (2021) Historical analysis exposes catastrophic seagrass loss for the united kingdom. *Frontiers in Plant Science.* 12:629962

Harley, C. D. and others. 2006. The impacts of climate change in coastal marine systems. *Ecology Letters* 9(2). 228-241.

Harvey, R. J. and others. 2019. No Detectable Broad-Scale Effect of Livestock Grazing on Soil Blue-Carbon Stock in Salt Marshes. *Frontiers in Ecology and Evolution*. 7(151).

Hill, R. and others. 2015. Can macroalgae contribute to blue carbon? An Australian perspective. *Limnology and Oceanography*. 60. 1689-1706.

Howard, J. and others. 2017. Clarifying the role of coastal and marine systems in climate mitigation. *Frontiers in Ecology and the Environment.* 15(1). 42-50.

Hughes, R. G. and others. 2018. Seagrass wasting disease: Nitrate enrichment and exposure to a herbicide (Diuron) increases susceptibility of Zostera marina to infection. Marine Pollution Bulletin. 134. 94-98.

JNCC (2015) The Marine Habitat Classification for Britain and Ireland Version 15.03. [Accessed March 2021]. Available from: **https://mhc.jncc.gov.uk/**

Jones, B.L. and Unsworth, R.K. 2016. The perilous state of seagrass in the British Isles. *Royal Society Open Science.* 3(1). 150596.

Jones, M.L.M., and others. 2008. Factors controlling soil development in sand dunes: evidence from a coastal dune soil chronosequence. *Plant and Soil*. 307(1). 219-234.

Kaiser, M.J. and others. 2006. Global analysis of response and recovery of benthic biota to fishing. *Marine Ecology Progress Series*. 311. 1-14.

Krause-Jensen, D. and others. 2018. Sequestration of macroalgal carbon: the elephant in the Blue Carbon room. *Biology Letters*. 14. 20180236.

Kokubu, Y. and others. 2019. Revealing the deposition of macrophytes transported offshore: Evidence of their long-distance dispersal and seasonal aggregation to the deep sea. *Scientific Reports.* 9. 4331.

Legge, O. and others. 2020. Carbon on the Northwest European Shelf: Contemporary Budget and Future Influences. *Frontiers in Marine Science* 7: 143.

Lima, M.A.C., Ward, R.D. & Joyce, C.B. 2020. Environmental drivers of sediment carbon storage in temperate seagrass meadows. *Hydrobiologia*. 847. 1773-1792.

Lovelock, C.E. & Duarte, C.M. 2019 Dimensions of Blue Carbon and emerging perspectives. *Biology. Letters.* 15. 20180781.

Luisetti, T. and others. 2019. Quantifying and valuing carbon flows and stores in coastal and shelf ecosystems in the UK. *Ecosystem Services*. 35. 67-76.

Macreadie, P.I. and others. 2015. Losses and recovery of organic carbon from a seagrass ecosystem following disturbance. *Proceedings of the Royal Society Biological Sciences*. 282: 20151537

May, V. and Hansom, J.D. 2003. Coastal Geomorphology of Great Britain, Geological Conservation Review Series No. 28, Peterborough. Joint Nature Conservation Committee.: 33-128.

MCCIP. 2018. Climate Change and Marine Conservation: Saltmarsh. C. Ladd, Skov, M., Lewis, H. and Leegwater, E. (eds), MCCIP, Lowestoft: 8.

MCCIP. 2020. Report Card. http://www.mccip.org.uk/media/1999/mccip-report-card-2020_webversion.pdf. Accessed March 2021.

Mieszkowska, N., Burrows, M. & Sugden, H. 2020. Impacts of climate change on intertidal habitats relevant to the coastal and marine environment around the UK. MCCIP Science Review: 256–271.

Natural England and RSPB. 2020. Climate Change Adaptation Manual (NE751) – Evidence to support nature conservation in a changing climate.

http://publications.naturalengland.org.uk/publication/5679197848862720. Accessed Jan 2021.
Natural England 2020 (unpubilshed). Figures derived from analysis carried out on the Natural England marine evidence base (last updated November 2020).
Nordlund, L. M. and others. 2018. Seagrass ecosystem services - What's next? Marine Pollution Bulletin. 134. 145-151.
Ortega, A. and others. 2019. Important contribution of macroalgae to oceanic carbon sequestration. *Nature Geoscience*. 12. 748-754.
Ouyang, X. & Lee, S.Y. 2014. Updated estimates of carbon accumulation rates in coastal marsh sediments. *Biogeosciences*. 11(18). 5057-5071.
Pakeman, R. J. and others. 2015. Species composition of coastal dune vegetation in Scotland has proved resistant to climate change over a third of a century. Global Change Biology. 21(10). 3738-3747.
Parker, R. and others. 2020. Carbon stocks and accumulation analysis for Secretary of State (SoS) region. Cefas project report for Defra. 40 pp.
Pedersen, M.F. and others. 2020. Detrital carbon production and export in high latitude kelp forests. *Oecologia*. 192. 227-239.
Pessarrodona, A. and others. 2018. Carbon assimilation and transfer through kelp forests in the NE Atlantic is diminished under a warmer ocean climate. Global Change Biology. 24(9). 4386-4398.
Potouroglou, M. 2017. Assessing the Role of Intertidal Seagrasses as Coastal Carbon Sinks in Scotland. PhD Thesis. School of Applied Sciences, Edinburgh Napier University.
Queirós, A. M. and others. 2019. Connected macroalgal-sediment systems: blue carbon and food webs in the deep coastal ocean. *Ecological Monographs*. 89(3). e01366.
Ravaglioli, C. and others. 2019. Ocean acidification and hypoxia alter organic carbon fluxes in marine soft sediments. *Global Change Biology*. 25(12). 4165-4178.
Robins, P. E. and others. 2016. Impact of climate change on UK estuaries: A review of past trends and potential projections. *Estuarine, Coastal and Shelf Science*. 169. 119-135.
Smale, D.A. and others. 2016. Linking environmental variables with regional-scale variability in ecological structure and standing stock of carbon within UK kelp forests. *Marine Ecology Progress Series.* 542: 79-95.
Smeaton, C. and others. 2021. Sediment type and surficial sedimentary carbon stocks across the United Kingdom's Exclusive Economic Zone and territorial waters of the Isle of Man and the Channel Islands. *Frontiers in Earth Science. 9:593324.*
Stratford, C. J. and others. 2014. Survey and analysis of vegetation and hydrological change in English dune slack habitats. Natural England Commissioned Report NECR153: 1-98.
Trimmer, M. and others. 1998. Nitrogen fluxes through the lower estuary of the river Great Ouse, England: The role of the bottom sediments. *Marine Ecology Progress Series*. *163*. 109–124.
Upstill-Goddard, R.C. and Barnes, J. 2016. Methane emissions from UK estuaries: Re-evaluating the estuarine source of tropospheric methane from Europe. *Marine Chemistry*. 180. 14-23.
Weber, T and others. 2019. Global ocean methane emissions dominated by shallow coastal waters. Nature Communications 10. 4584

Winters, G. and others. 2011. Effects of a simulated heat wave on photophysiology and gene expression of high- and low-latitude populations of Zostera marina. *Marine Ecology Progress Series*. 435. 83-95

Wood, C.L. and others. 2015. Coastal Biodiversity and Ecosystem Service Sustainability (CBESS) total organic carbon in mudflat and saltmarsh habitats. NERC Environmental Information Data Centre https://doi.org/10.5285/d4e9f0f7-637a-4aa4-b9df-2a4ca5bfaded [last accessed February 2020]

Yesson, C. and others. 2015. The distribution and environmental requirements of large brown seaweeds in the British Isles. *Journal of the Marine Biological Association of the United Kingdom.* 95(4). 669-680.

Appendix 1 – Carbon storage by habitat – chapter tables

Reproduced from habitat chapters – see for evidence. # in Totals Column indicates only soil carbon stock used due to unavailable of data for vegetation. * Authors representative value or mean of range

Habitat Description	Soil Carbon (t C ha^{-1})	Soil Depth (cm)	Vegetation Carbon (t C ha^{-1})	Soil + Veg. Carbon (t C ha^{-1})	Confidence [High, Medium, Low]	References
Woodland						
100 year Mixed native broadleaved woodland on mineral soil (to 1m)	151[b,cd] [108 to 173]	*100cm*	203[a,b,c] [41 to 344]	**354 [149 to 517]**	**Medium/ high**	[a]Woodland Carbon Code (2021) [b]Poulton and others (2003) [c]Butt and others (2009) [d]Vanguelova and others (2013)
100 year Mixed native broadleaved woodland (to 15cm soil depth)	55[b] [50 to 59]	*15cm*	203[a,b] [41 to 344]	**258 [91-403]**	**Medium/ high**	[a]Woodland Carbon Code (2021) [b]Vanguelova and others (2013)
30 year mixed broadleaved native woodland on mineral soil (to 1m)	151[b] [108 to 173]	*100cm*	114[a] [22 to 204]	**255 [130-377]**	**Medium**	[a]Woodland Carbon Code (2021) [b]Vanguelova and others (2013)
30 year mixed broadleaved native woodland (to 15cm soil depth)	55[b] [50 to 59]	*15cm*	114[a] [22 to 204]	**169** [72-263]	**Medium**	[a]Woodland Carbon Code (2021) [b]Vanguelova and others (2013)
Hedgerow						
Minimal/ Un manged Hedgerows	98.7* [66.52 to 111.93]		45.8	**144.5**	**Low**	Axe, 2015, reported in Axe 2020
Orchards						
Traditional Orchards	73.75* [47 to 111]	*30cm*	21.4* [8.6 to 230.4]	**95.15**	**Low**	Robertson and others (2012) – Top 30cm of soils & above and below ground biomass

Habitat Description	Soil Carbon (t C ha^{-1})	Soil Depth (cm)	Vegetation Carbon (t C ha^{-1})	Soil + Veg. Carbon (t C ha^{-1})	Confidence [High, Medium, Low]	References
Heathlands						
Upland & lowland Heathland	94[a] [88[b] to 103[c]]	15[a,b] to 30[c] cm	6* [2[b] to 9[a]]	100 [90 to 112]	Medium	[a]Van Paassen and others. (2020) [b] Ostle and others (2009), based on CS2007 [c] Cantarello, Newton and Hill (2011)
Semi-natural grasslands						
Acid grassland (without vegetation)	87	15 cm	No Data	–	Medium	Emmett and others (2010)
Calcareous grassland	69	15 cm	No Data	–	Low	Emmett and others (2010)
Neutral grassland	60[a] [33.31[b] to 68.74[c]]	15 cm	No Data	–	Medium	[a] Emmett and others (2010) [b] Fornara and others (2013) [c] Eze and others (2018a); Eze and others (2018c)
Farmland						
Arable / cultivated land	120[a] Range for 30cm [27.5 to 88.2[b]]	100 cm 30 cm	No vegetation stocks are given – as management (grazing and cutting) removes biomass annually	–	Low	[a] Moxley and others (2014) [b] Cantarello and others 2011
Improved grassland	130[c] [72 to 204[d]]	100 cm		–	Low	[c] Moxley and others (2014) [d] Cantarello and others (2011)
Intensive grassland on deep peat soils	1980	200 cm		–	Low	Evans and others (2016)
Arable on deep peat soils	Range [1290 to 3880]	75 to 200 cm		–	Low	Evans and others (2016)

Habitat Description	Soil Carbon (t C ha^{-1})	Soil Depth (cm)	Vegetation Carbon (t C ha^{-1})	Soil + Veg. Carbon (t C ha^{-1})	Confidence [High, Medium, Low]	References
Peatland						
Blanket Bog	525* Range [354 to 619 [a]] 259[b]	Full peat profile [a] 50cm [b]	See Chapter	525[a,#]	**Medium**	[a] Heinemeyer and others 2019 (Catchment scale estimates) [b] Ostle and others (2009) (0.5m depth)
Raised Bog	1610[a] Range [810 to 2530[b]]	Ave depth for Habitat (100cm to 380 cm)	See Chapter	1610[a, #]	**Low**	[a]Natural England, 2010 [b]Evans and others 2016
Fens (on Deep Peat)	1971* Range [610 to 2,820]	40–380 cm	No data	1971[#]	**Low**	Evans and others 2016

Chapter 5. Rivers, lakes and wetlands

No Table – As don't store carbon in a comparable way to other habitats, see chapter for summary tables and explanation.

Habitat Description	Total Habitat Carbon Storage			References
	t C ha^{-1}	Sediment depth (cm)	Confidence	
			[High, Medium, Low]	
Floodplains	109.4	10	Low	Unpublished survey at North Meadow Cricklade NNR

Chapter 6. Marine and coastal Habitats

Habitat Description	Sediment Carbon (t C ha^{-1})	Sediment Depth (cm)	Vegetation Carbon (t C ha^{-1})	Confidence [High, Medium, Low]	References
Coastal Habitats					
Sand dunes	9.5 [4 to 15]	15	5 (n=3) [1.6 to 8]	Low	Beaumont and others 2014; estimates presented here are for England only. Per hectare value has been calculated using the predicted extent of English sand dunes by Beaumont and others 2014.
Saltmarsh	59[a-e] [20[a] to 134[e]]	10-30	13[a]	Medium	[a]Beaumont and others 2014 [b]Ford and others 2012; [c]Burden and others 2013; [d]Ford and others 2019; [e]Burden and others 2019
Marine Habitats					
Intertidal sediments (sandflats and mudflats)	12[a-c] [0.13[a] to 1.72[a]] English [5.5[b] to 18.4[b]] Welsh [20[c] to 89[c]] Scottish	20[a] English 10[b] Welsh 50[c] Scotish	N/A	Low	[a]Trimmer and others 1998; [b]Armstrong and others 2020; [c]Potouroglou 2017
Seagrass	39[ab] [6.7[b] to 114.2[a]]	30	0.3[b] [0.07[b] to 0.5[b]]	Low	[a]Green and others 2018; [b]Lima and others 2020
Kelp	N/A	N/A	6.7[a-c] [1.37[a] to 11.987[a]]	Low	[a]Pessarrodona and others 2018; [b]Smale and others 2016; [c]Gevaert and others 2008
Biogenic reefs	No data	No data	N/A	-	No data
Subtidal sediment	55 (mud) [6 to 123] 18 (sand) [4 to 76]	100	N/A	Medium	Parker and others 2020. Based on c.1000 measurements on sediments in secretary of state (SoS) seas.

Habitat Description	Annual Carbon burial rate / loss for the habitat					Notes *Inc. References used*
	t CO_2e ha^{-1} y^{-1}	**Range (if possible)**	**t CO_2e ha^{-1} y^{-1}**	**Range (if possible)**	**Confidence** [High, Medium, Low]	
	Donor flux		**Receiver flux**			
Kelp	+11.63[ab]	+7.42[a] to +15.84[a]	-0.33[b]	No data	Low	The donor flux is the flux of kelp detritus moving away from the kelp bed. The receiver flux is how much of that donor flux reaches the sublittoral sediment. '+' values in the donor flux column refer to the kelp being moved away from that habitat as a source to other habitats and not necessarily a source to the atmosphere. Pessarrodona and others 2018; based on measurements in warm and cold sites in England, Scotland and Wales. Smale and others 2016; measured for sites off the coast of Plymouth.

Appendix 2 - Carbon fluxes by habitat – chapter tables

Reproduced from habitat chapters – see for supporting evidence.

+ve figures are emissions to the atmosphere

-ve figures are sequestration from the atmosphere back into the vegetation or soil by the ecosystem.

Chapter 2. Woodland, trees and scrub

Habitat Description	Annual carbon gain / loss for the habitat			References
	t CO_2e ha^{-1} y^{-1}	Range (if possible)	Confidence [High, Medium, Low]	
Woodland				
Mixed native broadleaved woodland (100 year)	-7	-2 to -13	Medium	Woodland Carbon Code 2021 Thomas and others 2011 Poulton and others 2003 Ashwood and others 2019 Rates averaged over 100 years
Mixed native broadleaved woodland (30 years)	-14.5	-2.5 to -25.5	Medium	Woodland Carbon Code (2021) Ashwood and others (2019) Rates averaged over 30 years
Hedgerow				
Hedgerows	-1.99[a]	-3.67[b] to -1.67[a]	Low	[a] Robertson and others 2012 [b] Falloon and others 2004
Orchards				
Traditional orchard with low intensity management	-2.89	-5.89 to +1.65	Low	Robertson and others 2012
Intensive orchard	-5.99	-7.77 to -4.21	Low	Robertson and others 2012

Chapter 3. Open habitats and farmland

Habitat Description	Annual Carbon Gain / loss for the habitat			References
	$t\ CO_2e\ ha^{-1}\ y^{-1}$	Range (if possible)	Confidence [High, Medium, Low]	
Heathlands				
Lowland heathland & Upland heathlands	+0.054	–	Low	Warner and others (2020)
Semi-natural grasslands				
Arable reversion to low input grassland	-1.590	–	Low	Warner and others (2020)
Undisturbed semi-natural grassland under long-term management	Negligible, equilibrium reached.	–	Low	Sozanska-Stanton and others (2016)
Farmland				
Arable land use	+0.29	-	Low	Muhammed and others (2018)
Improved grasslands	-0.36	-1.28 to +0.92	Low	Soussana and others (2010)
Intensive grassland on deep peat soils	+24.87	-	Medium	2021 update to the Emissions Inventory for UK Peatlands – to be published in April 2021 in the 2021 UK GHG Inventory[29,30] The full table is reported in section 4.2.2.
Arable on deep peat soils	+32.89	-	Medium	

[29] https://unfccc.int/ghg-inventories-annex-i-parties/2021
[30] NAEI website: https://naei.beis.gov.uk/reports/

Chapter 4. Blanket bogs, raised bogs and fens

Habitat Description	Annual Carbon Gain / loss for the habitat			References
	t CO_2e ha^{-1} y^{-1}	Range (if possible)	Confidence [High, Medium, Low]	
Peatlands States				
Near Natural Fen (undrained)	-0.93	-	Medium	All emission factors taken from the 2021 update to the Emissions Inventory for UK Peatlands – to be published in April 2021 in the 2021 UK GHG Inventory. Note fluxes have been adjusted to represent carbon only fluxes, and the influence of nitrous oxide removed, to be consistent with other habitats in this report. The full table is reported in section 4.2.2.
Near Natural Bog (undrained)	-0.02	-	Medium	
Rewetted Bog	3.87	-	Medium	
Rewetted Fen	8.05	-	Medium	
Rewetted Modified (Semi-natural) Bog	-0.02	-	Medium	
Modified Bog (semi-natural Heather + Grass dominated – Drained)	3.48	-	Medium	
Modified Bog (semi-natural Heather + Grass dominated – Undrained)	2.25	-	Medium	
Eroding Modified Bog (bare peat) - Drained	13.14	-	Medium	
Eroding Modified Bog (bare peat) - Undrained	12.03	-	Medium	
Extracted Domestic (drained)	13.23	-	Medium	
Extracted Industrial (drained)	13.14	-	Medium	
Cropland	32.89	-	Medium	
Intensive Grassland	24.87	-	Medium	
Extensive Grassland (combined bog/fen)	11.02	-	Medium	

Chapter 4. Rivers Lakes and wetland habitats

See next page, in landscape

Habitat Description	Annual Carbon accumulation / loss for the habitat									References *Inc. notes*
	t CO_2e ha^{-1} Y^{-1}	Range (if possible)	t CO_2e ha^{-1} Y^{-1}	Range (if possible)	t CO_2e ha^{-1} Y^{-1}	Range (if possible)	t CO_2e ha^{-1} Y^{-1}	Range (if possible)	Confidence	
	Land – freshwater flux		Freshwater flux (in stream transport)		Freshwater – atmosphere flux		Carbon accumulation (burial rate)		[High, Medium, Low]	
Streams draining peat	No data	No data	+2.32[ab]	+0.73[a] to +3.92[b]	+141.9[c-e]	+0.52[c] to +424.34[d]	N/A	N/A	Low	[a]Pawson and others (2007); [b]Worrall and others (2003); [c]Hope and others (2001); [d]Billett & Harvey (2013); [e]Billett and others (2015)
Lowland Rivers & Streams	1728	No data	531	No data	947	No data	N/A	N/A	Low	Worrall and others (2016). Note: Values presented here are totals, no per hectare values were given.
Chalk bed streams	No data	No data	No data	No data	+8.9	+7.15 to +10.63	No data	No data	Low	Romeijn and others (2019)
Floodplains	No data	No data	No data	No data	No data	No data	-3.365	-2.54 to -4.19	Low	Walling and others (2006)
Lakes	No data	No data	No data	No data	No data	No data	-7.1[a-c]	-0.46 to -23.6	Low	[a]Anderson and others (2014); [b]Casper and others (2000); [c]Scott (2014)
Reservoirs	+1021.17 (POC) +227.33 (DOC)	+432.67 to +1609.67 (POC) +128.33 to +326.33 (DOC)	No data	No data	+386.835	+66 to +707.67	-636.165	-407.00 to-865.33	Low	Stimson and others (2017) Note: Values presented here are totals, no per hectare values were given.
Peatland pools	No data	No data	No data	No data	No data	No data	No data	No data	No data	No data
Ponds	No data	No data	No data	No data	+6.69[a]	-17.53[a] to +30.91[a]	-16.12[b-d]	-2.90[b] to -29.33[c]	Low	[a]Gilbert and others (2016); [b]Taylor and others (2019); [b]Ockenden and others (2014); [d]Gilbert and others (2014);

Chapter 5. Marine and coastal habitats

Habitat Description	Annual Carbon burial rate / loss for the habitat			References Inc. notes
	$t\ CO_2e\ ha^{-1}\ y^{-1}$	**Range** (if possible)	**Confidence** [High, Medium, Low]	
Sand dune	-2.18	-2.13 to -2.68	Low	Jones and others (2008); measurements were made in Anglesey, Wales. No data are available for England.
Saltmarsh	-5.19	-2.35 to -8.03	Low	Beaumont and others 2014. based on previous assessments by Cannell and others 1999; Chmura and others (2003) and Adams and others (2012). Estimates are for the whole of the UK
Intertidal sediments	-1.98[ab]	-0.40[a] to -3.45[b]	Low	Armstrong and others (2020); estimated values for Wales Adams and others (2012) measured values for the Ouse estuary, England.
Subtidal sediment	-1.12[ab]	-0.07[a] to -2.16[a]	Low	Queirós and others 2019; measured values from the English Channel. De Haas and others (1997) estimated value for the North Sea.

Habitat Description	Annual Carbon burial rate / loss for the habitat					Notes *Inc. References used*
	CO_2e ha^{-1} y^{-1}	Range (if possible)	CO_2e ha^{-1} y^{-1}	Range (if possible)	Confidence [High, Medium, Low]	
	Donor flux		**Receiver flux**			
Kelp	+11.63[ab]	+7.42[a] to +15.84[a]	-0.33[b]	No data	Low	The donor flux is the flux of kelp detritus moving away from the kelp bed. The receiver flux is how much of that donor flux reaches the sublittoral sediment. '+' values in the donor flux column refer to the kelp being moved away from that habitat as a source to other habitats and not necessarily a source to the atmosphere. Pessarrodona and others 2018; based on measurements in warm and cold sites in England, Scotland and Wales. Smale and others 2016; measured for sites off the coast of Plymouth.

Appendix 3 – Record of report edits and corrections

Chapter	Sub-Chapter / Figure / Table	Page	Description of edit	Edited text (where relevant, new text added in bold)	Date of edits
Executive Summary	Figure 2: Carbon storage in contrasting habitats and land managements, using the best available data.	vi	Figure and caption have been updated to reflect changes to blanket bog and marine data reporting in the text – see below for details.	**Blanket bog carbon stocks are based on catchment scale estimates – see section 4.3.**	October 2021
Acronyms and abbreviations		xiii	Corrections of gigatonnes and megaton definitions Acronyms added for gram, kilogram, kilotonne. List has been reordered to incorporate changes.		October 2021
3 Open Habitats and Farmland	3.1 Chapter summary and key messages	53	A foot note has been added to section 3.4.3, to distinguish intensive grasslands from semi-improved grasslands	**Grassland management varies from intensively managed agriculturally improved sites, through semi-improved grasslands, that have been subject to a diverse past regime of ploughing, seeding and inputs, to semi-natural grasslands (as covered in section 3.3). To facilitate comparison and reflect the evidence base this section focuses on the intensively managed agricultural grasslands, however further evidence is needed to characterise carbon stocks and cycling in semi-improved grasslands which makes up a significant amount of England's grassland area.**	October 2021
3 Open Habitats and Farmland	3.4 Arable land and agricultural grassland	80	Text added to cover variation in agricultural management as well as soils, both of which will	Arable and improved grassland systems, however, are extremely difficult to characterise with regards to 'typical'	October 2021

			influence carbon dynamics:	carbon storage and sequestration, **as these systems can be under a broad range of management regarding fertiliser inputs and disturbance as well as** located on a wide range of soil types, each having their own properties with regards to carbon dynamics.’	
3 Open Habitats and Farmland	3.4 Arable land and agricultural grassland	80	The repeat of acronym SOC has been replaced with the term soil organic carbon	The calculation of soil carbon stocks requires a measurement of **soil organic carbon** (SOC) concentration, soil bulk density (BD), stone content, and soil depth all of which are spatially variable and prone to measurement errors (Schrumpf and others 2011).	October 2021
4 Blanket bogs, raised bogs and fens	Table 4.1 Summary of carbon storage in semi-natural peatland habitats, as based on the review of literature.	97	The blanket bog representative values have been changed to reflect the edits in the report text on page 108 (see below).		October 2021
4 Blanket bogs, raised bogs and fens	4.3.1 Carbon storage and sequestration in blanket bog	108	Descriptive text and additional data added to reflect catchment scale peat carbon stock estimates.	**Heinemeyer and others (2020) used ground penetrating radar (GPR) depth surveys coupled with site based organic carbon and bulk density data to calculate area weighted mean soil carbon stocks for the full peat profile of three upland blanket bog catchments in northern England, as well as manual measurements for study plots located on flatter areas of blanket bog. The GPR survey reported carbon stocks ranging 354 – 619 t C ha^{-1} and captures the inherent variability in**	October 2021

				blanket bog topography at a catchment scale, while plot based stocks estimates ranged between 653 – 954 t C ha^{-1}, reflecting deeper peat depths under flatter areas.	
6 Marine and coastal habitats	Table 6.1 Summary of carbon stocks in marine and coastal habitats, as based on the review of literature	151	The data table has been amended and a foot note added to reflect an error in data reporting in the source material.	**Note: Carbon stock values for sand dunes and saltmarsh updated in October 2021 to correct a misprint in the source material, Beaumont and others 2014.**	October 2021
6 Marine and coastal habitats	6.3.1 Carbon storage and sequestration in sand dunes	157	National sand dune carbon stocks are now reported as kilotonnes to correct the misprint in the source material.		October 2021
6 Marine and coastal habitats	6.4.1 Carbon storage and sequestration in Saltmarshes	158	National saltmarsh carbon stocks are now reported as kilotonnes to correct the misprint in the source material.		October 2021
7 Conclusions and Opportunities for Nature-based Solutions	Figure 7.1 Carbon storage in contrasting habitats and land managements, using the best available data.	173	Figure has been updated to reflect changes to blanket bog and marine data reporting in the text.	**Blanket bog carbon stocks are based on catchment scale estimates – see section 4.3.**	October 2021

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