Achieving net zero

A review of the evidence behind potential carbon offsetting approaches

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Contents

Foreword

Scientific research and analysis underpins everything the Environment Agency does. It helps us to understand and manage the environment effectively. Our own experts work with leading scientific organisations, universities and other parts of the Defra group to bring the best knowledge to bear on the environmental problems that we face now and in the future. Our scientific work is published as summaries and reports, freely available to all.

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Professor Doug Wilson
Director, Research, Analysis and Evaluation

Executive summary

Introduction

The UK has set out in law the target of achieving net zero by 2050. To achieve this the annual rate of Green House Gas (GHG) emissions will need to be cut by over 260 million tonnes (Mt) CO_2e (carbon dioxide equivalent) from 2019 levels to less than 90 MtCO_2e in 2050 (Committee on Climate Change, 2019a). The Environment Agency's own net zero target, set for 2030, includes reducing emissions by 45%, and addressing remaining emissions using best practice carbon offsetting techniques.

There is currently debate in the UK, and internationally, about the most appropriate approaches to carbon offsetting. The evidence base is still emerging and there are many factors to consider in developing carbon offsetting systems in the UK.

This report has reviewed the evidence behind a wide range of approaches which could be used for carbon offsetting in the UK. This evidence base will be used to inform the development of the Environment Agency's own carbon offsetting strategy.

What we did

We reviewed the evidence behind the following 17 potential carbon offsetting approaches:

- upland peat restoration
- lowland peat restoration
- woodland creation
- grassland management
- freshwater wetlands - floodplain restoration
- freshwater wetlands - constructed wetlands management
- saltmarsh restoration
- seagrass restoration
- kelp restoration
- agricultural soil management practices - arable land
- agricultural soil management practices - pasture grassland
- hedges and trees outside woodland
- enhanced weathering
- biochar
- household insulation
- household low carbon heating
- other built environment measures (for example, renewable electricity consumption, reducing water consumption, building with timber and low carbon transport)

Each offsetting approach was assessed to determine the following factors:

- readiness for implementation
- speed and scale of potential impacts
- permanence, leakage and additionality (reductions or removals of GHGs that would not have happened otherwise)
- co-benefits
- confidence in the science

- measuring impact
- risks and barriers
- costs

The findings of this assessment form the main body of this report. Using this evidence, we scored each approach against each of the factors listed above using a red-amber-green rating (RAG) to determine the relative strengths and weaknesses of potential offsetting approaches.

What we found

- All the approaches reviewed have strengths and weaknesses with regard to their potential to be used for offsetting residual carbon emissions. No silver bullet offset solution was found.
- Some of the approaches reviewed remove GHG emissions from the atmosphere, others reduce the rate of GHG emissions to the atmosphere, and some progress from reductions to removals over time.
- Different offsetting approaches remove GHGs from the atmosphere at different rates. So, the speed at which an approach becomes effective is a critical consideration when developing an offsetting strategy.
- It is likely that most organisations will need to adopt multiple approaches as part of their offsetting strategy. This would integrate the strengths of several approaches to maximise the likelihood of meeting climate targets.
- Only carbon offsetting projects that remove GHGs will be compatible with true net zero emissions – this is where GHGs emitted into the atmosphere are balanced by removing equal amounts of GHG emissions from the atmosphere. However, organisations may still find value in accelerating reductions elsewhere through carbon offsetting. This is especially the case in the shorter term, where the potential for removing GHGs is more limited.
- At present, there are only 2 accredited carbon offsetting standards in the UK – the Woodland Carbon Code and the Peatland Code. This means that for the other approaches reviewed in this report, there are no accredited and verified carbon offsetting schemes currently available in the UK. More research and development will be needed in the future to expand the number of nature-based and built environment offsetting schemes that are available.

Outputs and next steps

The main outputs from this evidence review include a report which summarises the review findings, together with an infographic that provides a short summary of this literature review, and gives a high-level overview of each of the offsetting approaches reviewed.

The Environment Agency will use the outputs from this research to shape the development of its own organisational carbon offsetting strategy.

Chapter 1. Introduction

1.1 Background context

The UK has set out in law the target of achieving net zero by 2050. To achieve this, annual emission rates will need to be cut by over 260 $MtCO_2e$ from 2019 levels to less than 90 $MtCO_2e$ in 2050 (Committee on Climate Change, 2019a). The Environment Agency's own net zero target, set for 2030, includes reducing emissions by 45%, and addressing remaining emissions using best practice carbon offsetting techniques.

While approaches to reducing greenhouse gas (GHG) emissions in the UK are relatively well documented and understood, those which enable offsetting of residual emissions have been less thoroughly explored. The purpose of this report is to provide a review of the evidence behind 17 different offsetting approaches, to understand their potential offset residual GHG emissions. It focuses on reviewing the scientific evidence base relating to a variety of carbon reduction and removal approaches, which could be implemented across the UK as part of a carbon offsetting project.

This report will be used by the Environment Agency to inform the development of its corporate offsetting strategy. As part of the development of this strategy the Environment Agency is also:

- considering how it could use the existing international carbon market
- assessing the ethical and political considerations related to offsetting
- modelling various offsetting scenarios to assess which approaches may be the most suitable to deliver its net zero target

This work sits alongside the substantial efforts presently underway to directly reduce the organisation's emissions. Find out more about the Environment Agency's carbon footprint and net zero target.

1.2 Defining carbon offsetting

Carbon offsetting

Carbon offsetting is practiced by many businesses, public sector organisations and governments, but there is no unifying definition which explains what it means. For the purposes of this report it is defined as:

> the practice of reducing or removing greenhouse gas emissions to balance ongoing greenhouse gas emissions, in order to achieve claims such as climate neutrality or net zero

There are 3 main components of this definition:

1. To balance ongoing greenhouse gas (GHG) emissions that have not yet been eliminated.

2. By balancing ongoing GHG emissions, to claim a status such as climate neutrality or net zero
3. The mechanism used to carbon offset can include:
   - reducing the rate of GHG emissions into the atmosphere
   - removing GHG emissions from the atmosphere

This final point, regarding the difference between **reducing and removing** GHG emissions, is especially important in the context of meeting the Paris ambition of limiting warming to no more than 1.5 degrees.

Net zero

Net zero, as it is widely interpreted, means that for an organisation, region or country, total GHG emissions to the atmosphere are equal to, or less than, emissions removed from the atmosphere. To achieve net zero it is a priority to reduce GHG emissions as much as possible. Should any emissions remain (after all efforts have been made to reduce emissions) to claim net zero it will be necessary to remove these remaining GHG emissions from the atmosphere at the same rate they are produced.

There is significant discussion over which carbon offsetting mechanisms are compatible with achieving net zero. The 2 types of mechanism are[1]:

- **Emissions reductions** - This mechanism uses projects to help other parties reduce their own emissions. This reduces the amount of GHG emissions that are going into the atmosphere but does not remove any GHG from the atmosphere, so emissions continue to build up, but at a slower rate. An example of such a mechanism is an energy efficiency measure, which reduces the need for fuel consumption in buildings.
- **Emissions removals** - This mechanism develops projects that remove GHGs from the atmosphere, often using nature based solutions, such as growing trees.

A consensus is forming that carbon offsetting approaches which **reduce** emissions elsewhere will not be scientifically compatible with true ‘net zero’ in the long term (Allen and others, 2020; Science Based Targets, 2020). If reduction type offsetting projects are used, there will always be net GHG emissions into the atmosphere, which is not compatible with meeting the Paris ambition to limit global temperature rise to no more than 1.5 degrees. Carbon offsetting that is compatible with achieving the aims of the Paris agreement will therefore involve projects that **remove** GHG emissions from the atmosphere. Net zero can only be achieved when ongoing GHG emissions are balanced with removals of equal GHG emissions from the atmosphere.

This does not mean that carbon offsetting projects which reduce emissions should be entirely discounted. There are 2 reasons for this:

[1] Other definitions and terminology to be used in the context of carbon offsetting have been proposed, for example, by The Science Based Target Initiative. These may be widely adopted in future. Within the context of this report the simple use of reductions and removals has been opted for.

1. Accelerating the rate of GHG emission reductions has an immediate climate benefit.
2. Technologies and land-use change solutions that remove GHG emissions from the atmosphere are still in early stages of development and will take time to be scalable (such as bioenergy carbon capture and storage (BECCS)).

In the short term, there may therefore still be a useful role for carbon offsetting projects that reduce emissions. For organisations, such as the Environment Agency, with net zero goals in 2030, the extent to which residual emissions can be offset with purely GHG removal projects, is uncertain. In some instances, there may be a need to use a combination of carbon offsetting projects that both reduce and remove emissions to meet these short-term targets. This portfolio approach to offsetting would ultimately have to transition towards purely carbon removal projects to achieve true net zero.

The Science Based Targets Initiative

The Science Based Targets Initiative (SBTi), a leading organisation which facilitates science-based target setting in line with a 1.5°C emissions reduction pathway, whilst promoting the importance of offsetting using GHG removal projects, has acknowledged the continued role that can be played by carbon offsetting projects which reduce emissions. SBTi has suggested that if an organisation uses carbon offsetting projects that reduce GHG emissions, this should be associated with the term ‘climate positive’, as opposed to net zero (Science Based Targets, 2020). Discussions about these definitions and mechanisms of carbon offsetting are evolving at the time of writing, and, therefore, it is not possible to say for certain what will become common practice for carbon offsetting in the UK.

In light of the potential importance of both carbon offsetting mechanisms – those that reduce GHG emissions, and those that remove GHG emissions – this report includes approaches in both categories (and some that change between reductions and removals over time). The language of **reductions and removals** is used throughout the report to make it clear the way in which each potential approach benefits the climate. This report does not intend to make recommendations regarding which approaches should be actually implemented as carbon offsetting projects and is restricted to presenting the scientific evidence base.

1.3 Scope of this review

The focus of this review is on GHG emission reduction and removal approaches that may be suitable for implementation as carbon offsetting projects in the UK. The implementation of carbon offsetting projects in the UK is less widespread than in other countries around the world. This is the result of an international carbon offsetting system that has historically prioritised carbon offsetting activities in countries across Africa, Asia, South America and North America. However, the implementation of the Paris Climate Agreement and potential changes to rules on international carbon trading, are likely to create new opportunities to implement carbon offsetting projects in the UK. This potential change to international carbon trading rules, combined with growing numbers of UK-based organisations with net zero targets, has prompted increasing interest in the potential to carbon offset in the UK.

Two carbon offsetting project certification bodies currently operate in the UK: the Woodland Carbon Code, and the Peatland Code. In addition to considering these existing carbon offsetting approaches, an important aim of this report is to assess the feasibility of other potential carbon offsetting approaches, and support innovation in the UK carbon market.

There are numerous GHG emission reduction and removal approaches that could, in principle, be considered as potential carbon offsetting approaches. At the start of this research project, the project's steering group used a shortlisting process to identify approaches most relevant to the Environment Agency. The shortlist process is described in Appendix 1.The conclusions of this process resulted in the 17 approaches listed in Table 1-1 being selected for review.

Table 1-1 Approaches reviewed in this report

Approach	GHG emission reduction or removal?*
Upland peat restoration	Reduction and potentially removal
Lowland peat restoration	Reduction and potentially removal
Woodland creation	Removal
Grassland management	Removal
Freshwater wetlands – flood plain restoration	Removal
Freshwater wetlands – constructed wetlands	Removal
Saltmarsh restoration	Removal
Seagrass restoration	Removal
Kelp restoration	Removal
Agricultural soil management practices - arable land	Removal and reduction
Agricultural soil management practices - pasture land	Removal and reduction
Hedges and trees outside woodland	Removal
Enhanced weathering	Removal
Biochar	Removal
Household insulation	Reduction

Approach	GHG emission reduction or removal?*
Household low carbon heating	Reduction
Other built environment measures such as: renewable electricity consumption; reducing water consumption: building with timber and low carbon transport	Reduction

* This classification can vary depending on specific project contexts and the boundary of the system being considered.

For each of the offsetting approaches listed in Table 1-1 our review considered the following factors:

- readiness for implementation
- speed and scale
- permanence, leakage and additionality
- co-benefits
- confidence in the science
- measuring impact
- risks and barriers
- costs

These factors are described in more detail in chapter 2.

In chapters 3 to 6 this report discusses each of the selected carbon offsetting approaches listed in Table 1-1 against these characteristics in order to build an understanding of their potential viability for carbon offsetting in the UK. Each approach has been evaluated by reviewing relevant academic and grey literature. Authoritative evidence in the public realm was sought out and supported with input from academics with expertise in the relevant habitat types. The collected evidence is intended to inform the development of potential UK carbon offsetting approaches.

Other carbon removal technologies, such as bioenergy carbon capture and storage (BECCS) and direct air carbon capture and storage (DACCS), were considered in the preliminary stages of this report but excluded in the shortlisting stage (see Appendix 1) because they are unlikely to be available for widescale application by 2030, the Environment Agency's net zero target date.

Natural England have recently published a report which describes in detail the carbon storage and sequestration potential of different semi-natural habitats in relation to their condition and/or management (Gregg and others, 2021). Natural England's review covers additional habitats not included in this report.

Given that many of the approaches evaluated in this report are relatively novel in the context of carbon offsetting, the value of this review sits as much in the questions and evidence gaps it raises, as the evidenced information it conveys. Implementing many of the approaches covered in this report as carbon offsetting projects will require filling these evidence gaps and overcoming challenges. Assessing areas where evidence is strong and those where evidence is weak is therefore an important first step in expanding the number of carbon offsetting project types in the UK.

1.4 Report structure

This chapter has provided an introduction to this carbon offsetting literature review.

The report is structured into 7 chapters which includes:

- chapter 2 – reviewing the evidence
- chapter 3 – land-based approaches
- chapter 4 – river and coastal approaches
- chapter 5 – agricultural approaches
- chapter 6 – built environment approaches
- chapter 7 - conclusions

Alongside this review, we have also developed an infographic which provides a short summary of this literature review, and a high-level overview of each of the offsetting approaches reviewed.

Chapter 2. Reviewing the evidence

2.1 Introduction

Each of the carbon offsetting approaches reviewed in chapters 3 to 6 were assessed in relation to the following 8 characteristics:

- approach overview
- readiness for implementation
- speed and scale
- permanence, leakage and additionality
- confidence in the science
- measuring impact
- risks and barriers
- costs

This assessment has enabled us to present the current status of the evidence and viability of the various reduction and removal approaches to be implemented for carbon offsetting. This chapter provides some background context before describing the 8 characteristics listed above.

2.2 Accounting for greenhouse emissions and the carbon cycle

Accounting for greenhouse gas emissions

Ultimately, the benefits of an offsetting approach will depend on how much GHG it removes from the atmosphere or avoids being emitted in the first place. Therefore, the determination of GHG benefits must be comparable across different measures.

The most commonly referenced GHG is carbon dioxide (CO_2), as this is the most common GHG in our atmosphere. There are, however, many other GHGs, a number of which are relevant to measures assessed in this report. To compare these GHGs, a value called the Global Warming Potential (GWP) is widely used to compare how much global warming a gas creates for each additional tonne in the atmosphere. For simplicity, the GWP of CO_2 is set at the value of 1, with all other gases measured relative to CO_2. For example, methane (CH_4) is set at a much more powerful 28 (one tonne of methane has the same impact as 28 tonnes of CO_2), and nitrous oxide (N_2O) at 265.[2]

Over time, most GHGs change into different compounds while they are in the atmosphere. However, this happens at different rates for different gases. For example, methane quickly oxidises to produce CO_2, with a 'half-life' (the time after which 50% of the gas remains) that

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[2] GWP values based on the IPCC Fifth Assessment Report (AR5).

is dramatically shorter than that of CO_2. To achieve comparability, it is therefore necessary to compare gases over a specific period of time. The usual period applied is 100 years.

The problem with the 100-year time frame is that the current imperative of reducing atmospheric GHGs means that short-term impacts are very important. While methane has a GWP of 28 over the 100-year timeframe, a 20-year timeframe yields a GWP of methane of around 84. This is critical when considering the benefits of different carbon offsetting measures because the benefits of avoided methane will be achieved more intensely over a shorter time period.

Carbon dioxide equivalents (CO_2e) allow the greenhouse effect of all relevant GHGs to be combined into a single measure (the sum of the amount of each GHG emitted multiplied by its 100 year GWP). They depend on the period over which measurements are made and therefore are not precisely equivalent. Nonetheless, this is a well-established approach that allows comparison, and so this report uses CO_2e as the unit of mass for GHG emissions, while noting that for certain gases such as methane there is a significant additional benefit associated with avoiding them in the short-term which is not captured in the CO_2e figures. Should IPCC (Intergovernmental Panel on Climate Change) guidelines for calculating the impact of different GHGs change in the future, it is recommended that these changes should be reflected in calculations of GHG impacts at the project level.

The carbon cycle

The natural environment measures featured in this report involve the transfer of carbon between the atmosphere and organic materials. Research relating to storage of carbon in organic materials usually gives values for carbon (C) rather than CO_2e since the carbon is not stored as a gas but in a compound. This report therefore also refers to stored carbon (C) as well as CO_2 and CO_2e where relevant.

As carbon dioxide and methane are absorbed and released, the 'stock' of C in the material increases or decreases. In certain cases where carbon dioxide is the only relevant GHG, values given in the literature in terms of the mass of C have been converted into carbon dioxide equivalents[3]; this is indicated throughout the report using this symbol: "†".

2.2 Approach overview

Chapters 3 to 6 describe a range of different potential approaches to offsetting. Each 'Approach overview' section introduces the approach, discusses the technical complexities involved in its implementation, and provides general considerations regarding its appropriateness as an offsetting method.

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[3] This conversion is achieved by multiplying by 3.67 (44/12), the ratio of the mass of a carbon dioxide molecule and the mass of a carbon atom.

2.3 Readiness for implementation

Not all carbon reduction and removal approaches may be ready immediately to be implemented for carbon offsetting. Implementation timescales depend on considerations like technological readiness, the availability of carbon offset certification standards (and associated monitoring, reporting and verification methods), as well as the political landscape. For example, an approach may not yet be considered a valid offsetting measure within the UK and it will take time to establish these accounting and certification methods. This section of each chapter sets out the relevant considerations which will have a bearing on how quickly the approach could be implemented for carbon offsetting.

2.4 Speed and scale

The total GHG emissions that can be reduced or removed through a carbon offsetting approach can be affected by a range of factors, such as:

- the availability of land to implement a particular approach (for example, conversion to woodland)
- the capacity of a technology to reduce emissions
- any ‘competition’ from other technologies

The impact of each approach in terms of the quantity of GHG emissions reduced or removed over time usually falls into one of 3 broad categories:

1. Many nature-based removal measures have very little impact initially, but their ability to sequester (capture and store) carbon dioxide grows as the measure matures (Figure 2-1). As the project ages, its ability to sequester falls as storage becomes saturated.
2. Other natural environment measures are reduction measures which become removal measures over time (Figure 2-2). Offsetting approaches here can have a high initial impact, reducing emissions from that source. As the project matures, it turns from a carbon source to a net sink, sequestering carbon. The total carbon ‘savings’ here (the net difference between the emissions after a project is implemented and the emissions in the baseline scenario) directly depend on the baseline, which can be hard to establish.
3. Most reduction approaches have a high immediate impact, which reduces over time because the baseline level of emissions from that source also falls over time (Figure 2-3). As a result, the emissions saving that can be attributed to the measure reduces with time.

Within each speed and scale section, the characteristics of the reduction or removal profile of each approach are explored to understand the extent of reductions or removals that could be achieved.

Figure 2-1 Illustrative GHG 'savings' from removal measures with a peak capture rate. They take some time to reach peak capture rate, sequester for some time, and then diminish back to zero as carbon saturation is reached.

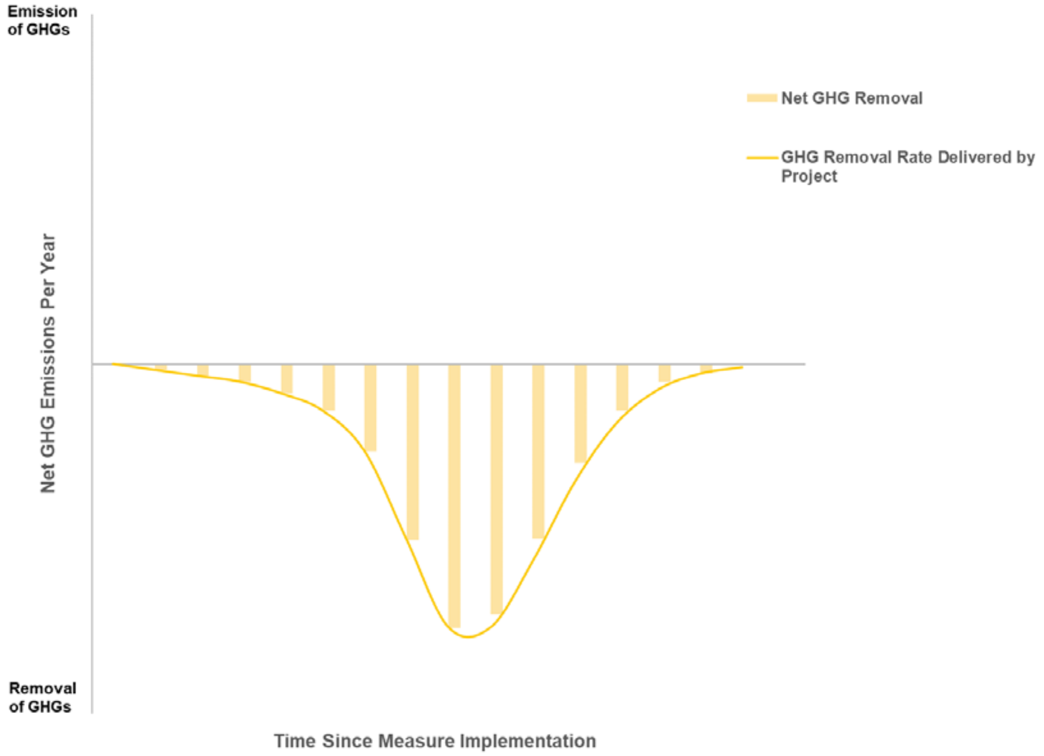


Figure 2-2 Illustrative GHG 'savings' from measures where carbon sources are converted to sinks. The overall savings depend on the baseline scenario, which can be difficult to assess into the future. This graph shows 2 possible scenarios: one in which emissions are tackled progressively (orange), and one where climate change causes natural emissions to 'rebound' upwards (purple).

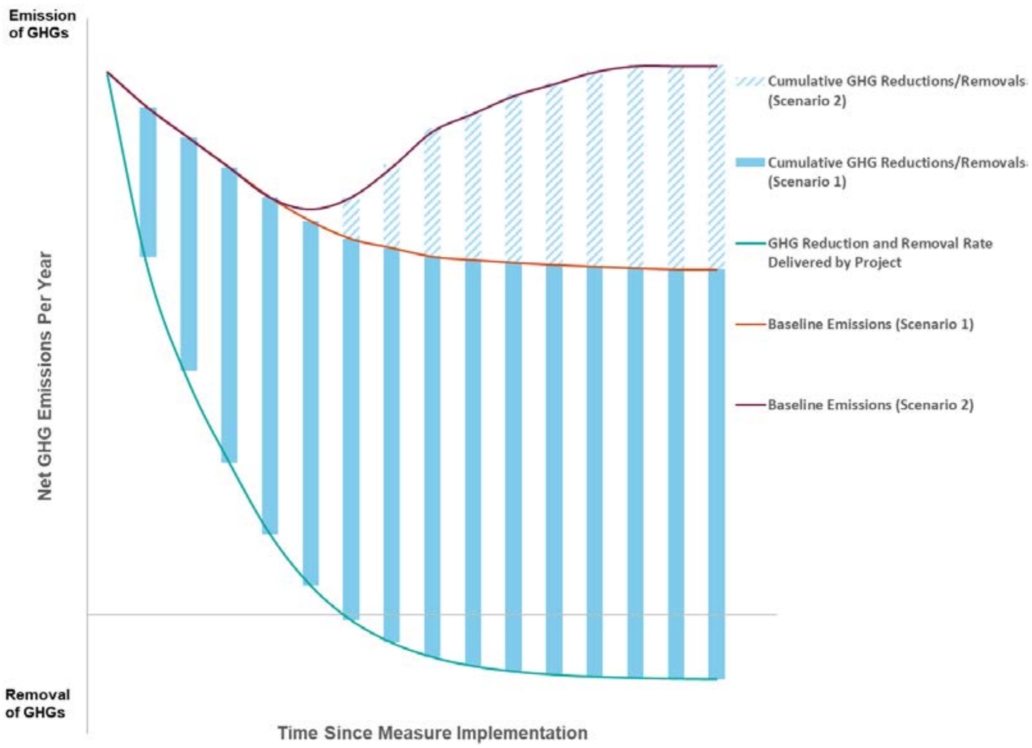
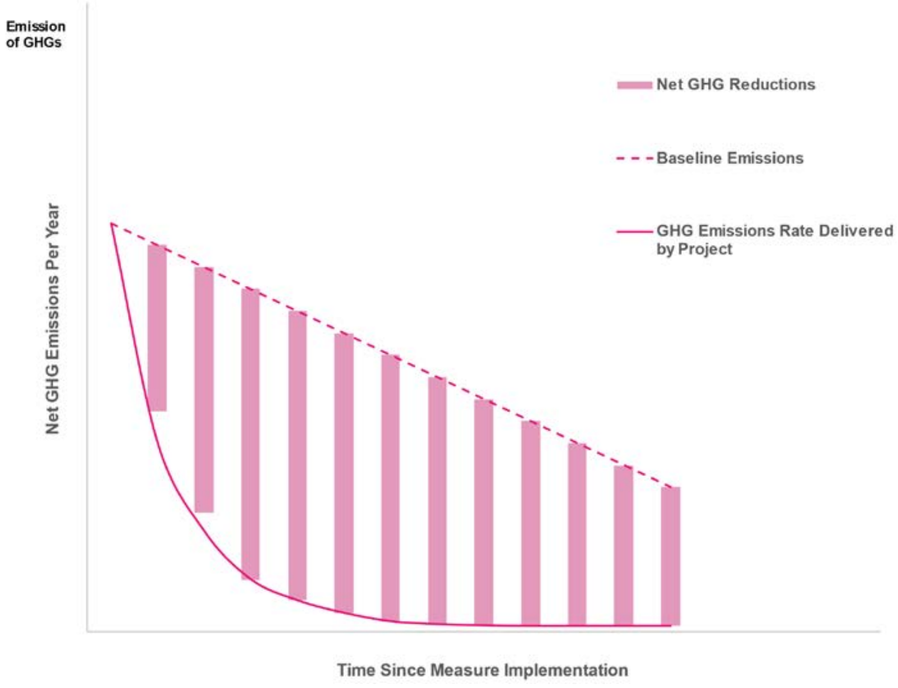


Figure 2-3 Illustrative net GHG emissions 'savings' of a typical reduction measure



2.4.1 Baseline carbon emissions

Strongly related to the speed and scale of an approach, is the baseline scenario. A baseline scenario, sometimes called ‘business as usual’ (BAU) or the counterfactual scenario, is ‘what would have happened without intervention’. It is the starting position against which any emissions removals or reductions are compared. The baseline scenario is likely to be unique to a project and needs careful consideration to ensure calculations of emission reductions and removals are accurate. A baseline scenario could show ongoing GHG emissions (for example, ongoing use of fossil fuels, or degrading peatland), an equilibrium state (such as a grassland that removes and emits equivalent GHG emissions), or a removal state (such as a healthy saltmarsh – although in this instance, the implementation of a project would have to increase the rate of carbon removal).

Baseline emissions trajectories can be hard to forecast because they depend on information about the future, which is inherently uncertain. How government policy might change, the costs of future technology and energy, and whether new financing mechanisms may emerge, are difficult to predict in the future. For example, decarbonisation policy scenarios provided to the government by the Committee on Climate Change show that heat pumps should be installed rapidly between now and 2050, but it is not guaranteed that this will be achieved. In this case, it is difficult to establish the number of heat pumps likely to be installed in UK homes in 2030 in a baseline scenario. Equally, the need to address GHG emissions from degraded peatland is well known, but exactly how and when appropriate measures will be implemented is unclear. In a hypothetical future where new finance is made available for peatland restoration, and rates of degradation and emissions are reduced, the relative benefit of securing offsetting funding, relative to this baseline scenario, is lessened.

These examples illustrate the importance of transparency regarding assumptions built into baseline scenarios, and regularly reviewing the baseline assumptions to ensure they remain fair and justifiable.

2.4.2 Theoretical maximum national abatement potential

For each of the approaches described, a category is included for the theoretical maximum national abatement potential. Categories are deliberately large given the multiple assumptions necessary for any such estimation. They are only provided to give a general overview of how the potential of approaches compare with each other in terms of their potential scale in the UK. The numerical boundaries to each category, which again should be approached with caution, are shown in Table 2-1.

Table 2-1 UK theoretical maximum reduction or removal potential of the approaches

UK national abatement potential ($MtCO_2e$)	Corresponding category	Approaches
n/a or uncertain	n/a	• Constructed wetlands • Grasslands • Kelp
<1	Low	• Saltmarsh restoration • Seagrass restoration
1-5	Moderate	• Agricultural soil management practices - arable land • Freshwater wetlands - floodplain restoration
5-10	High	• Biochar • Hedges and trees outside woodland • Lowland peat restoration
>10	Very high	• Upland peat restoration • Enhanced weathering • Woodland creation • Agricultural soil management practices - pasture grassland • Household insulation • Household low carbon heating

To ascribe each of the approaches to a category, the maximum theoretical area each approach could be applied to was estimated. For example, agricultural soil management practices on grasslands could in theory be applied to all grasslands in the UK. We then used the per unit abatement potential (usually a range) and this area to obtain the category. More details on how a range for the theoretical maximum implementation area in the UK was obtained is included in the chapters, at the end of the 'Speed and scale' sections.

2.6 Permanence, leakage and additionality

Permanence, leakage and additionality are important criteria that are used to determine whether carbon offsetting projects are achieving real benefits.

2.6.1 Permanence

Permanence can be explained in several different ways, for example:

- It can refer to whether GHG emissions removals can be reversed. For example, a tree plantation that sequesters carbon must be protected indefinitely. If it is cut down and burnt, the CO_2 that it absorbed is released. In this case, CO_2 sequestered was 'non-permanent.'[4]

- Avoided GHG emissions (in other words, GHG emission reductions) will never be released into the environment, and so these are permanent reductions. For example, a tonne of CO_2 emissions that is avoided after insulation is installed has been avoided permanently. Even if the insulation is later removed and emissions rise again, the emissions avoided while the insulation was in place have still been avoided. As a result of this dynamic, reduction measures are highly likely to be permanent. In contrast, most removals approaches carry some vulnerability to lack of permanence, meaning that it must be ensured throughout the project that emissions sequestered through removal measures are not reversed.

- It can also refer to an intervention's long-term ability to reduce or remove GHG emissions. For example, this can include:
  - the potential of a built environment offsetting project, such as household insulation, to reduce GHG emissions in the long term, given reductions would be expected in the baseline scenario due to national policy incentives
  - the potential of natural environment removal approaches to continue removing GHG emissions over time, given that most landscapes reach a 'saturation' point and can no longer sequester further CO_2

To confidently implement an approach for carbon offsetting, the permanence of reductions or removals achieved must be understood, and regularly monitored. Relevant considerations to permanence are addressed in each of these chapter sections. In the 'best case', GHG emissions removed from the atmosphere will be indefinitely stored. However, we must

[4] It is possible to allow harvesting of timber within projects certified by the Woodland Carbon Code, so long as the carbon associated with any harvested timber is discounted from the total claimed carbon removals.

recognise there are practical challenges to achieving this, and it is essential that any project used for offset clearly documents potential permanence risks, and how these are mitigated.

2.6.2 Leakage

If a project demonstrates ‘leakage’, then implementing a project to reduce or remove GHG emissions leads to more emissions elsewhere. For example, changing management of a particular habitat may cause the current land management practices to move elsewhere, undermining some of the carbon benefit of the original land use change. Or, during project implementation, GHG emissions could be released through construction works, which reduce the overall GHG emissions savings achieved. It is important that any carbon measurement approaches applied in a carbon offsetting project account for the leakage of emissions. Each of these report sections discusses possible GHG emissions sources or unintended consequences of project implementation that could be considered leakage.

2.6.3 Additionality

‘Additionality’ means that funding a carbon offsetting project leads to reductions or removals of GHGs that would not have happened otherwise. For example, if a habitat creation project was going ahead anyway irrespective of it being considered as a carbon offsetting project, it would not be considered ‘additional’. The established carbon offsetting market requires projects to pass a set of additionality tests, which are used to demonstrate whether a project is additional.

This report highlights potential challenges which may undermine project additionality. Such challenges include future finance sources that could become available or the outcomes of future policy requirements.

In reality, a project’s emissions baseline, permanence and additionality are highly interconnected. If developing an offset project, particularly if not using established certification standards such as the Woodland Carbon Code and Peatland Code, robust rules must be in place to address these considerations.

2.7 Co-benefits

A carbon offsetting project that achieves ‘co-benefits’ results in additional, secondary positive impacts that may or may not have been part of the main aims of the project. For example, a co-benefit of restoring woodland may be an increase in biodiversity and green space for visitors to enjoy. These report sections discuss the range of co-benefits that could be provided by each approach.

While the focus of this report is on the climate impacts of the various approaches, in all instances the reduction or removal potential of an approach should not be considered in isolation to other environmental and social impacts. In addition, as demand for investment in ‘public goods’ increases, it is also foreseeable that other sources of finance may become available to support landscape revitalisation, on the basis of achieving benefits such as biodiversity enhancement. These potential market developments strengthen the case for understanding the multiple benefits that could be achieved alongside GHG emission reductions or removals.

2.8 Confidence in the science

All scientific research has a degree of uncertainty associated with it. Uncertainty is a measure of how confident we are in the results of the research. A low level of scientific uncertainty in a measure's ability to sequester CO_2 means we have high confidence in how much CO_2 it will remove. Understanding the extent of uncertainty regarding quantities of GHG emissions and removals enables decisions regarding carbon offsetting strategy to be made with confidence.

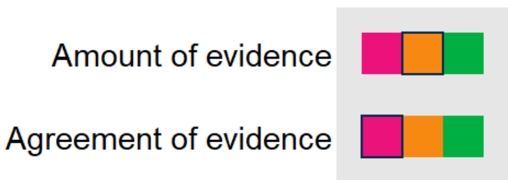
This report reviews the level of confidence in the evidence base for each of the approaches covered in chapters 3 to 6. A confidence assessment is carried out for each of the 8 characteristics described here. This confidence assessment focuses on 2 areas:

- the amount of evidence relating to each topic
- the extent to which the results in the literature are in agreement

The scoring system is described in Figure 2-4. Throughout the report, this analysis is represented by the graphic depicted in Figure 2-5.

Figure 2-4 Explanation of confidence categories

Figure 2-5 The confidence infographic used throughout the report (selected values highlighted in dark blue)



The literature review carried out as part of this project focused on identifying literature sources where evaluations of various carbon reduction and removal approaches had already been carried out. The evaluations of confidence in the evidence made by researchers for this report is therefore qualitative and relies on professional judgement. It follows that the confidence in the evaluations included within this report should not be interpreted as resulting from any analysis of the numbers of published articles or systematic comparisons of the results generated by various academic sources. The confidence in the evidence assessments are intended to be indicative and used to compare the relative carbon reduction and removal approaches within the context of this report.

Grasslands, kelp and biochar were not given a red-amber-green (RAG) rating as depicted in Figure 2-5 because these offsetting approaches were not reviewed as extensively as the other approaches covered in this report. This was because less science was available (grassland and kelp) and because it was included at a later stage in the review (biochar).

2.9 Measuring impact

Once an approach has been implemented, it is important to be able to quantitatively measure the GHG emission reductions or removals achieved. This is to ensure that the carbon offsetting project is having the anticipated effect. For natural environment approaches, it is possible to monitor their effect by testing biological material, but this can be both expensive and technically challenging. This means therefore that project implementers often rely on values taken from the scientific literature as a proxy. For built environment approaches, it is generally not possible to physically measure changes in carbon emissions, which means proxy values must be used instead (for example, measured reduction in gas consumption after insulation has been installed).

2.10 Risks and barriers

The risks and barriers section of each chapter draws together the main challenges to implementing an approach for the purpose of carbon offsetting. It will therefore include some repetition from other sections, but may also include considerations that do not neatly fit elsewhere in the discussion. Risks and barriers identified may relate to:

- long-term additionality or permanence being uncertain
- emissions leakage
- uncertainty in the scientific evidence base
- potential for negative environmental or social impacts
- the risks that climate change itself poses to this approach (for example, where increased temperatures may lead to failure of a land use change because the habitat it is based upon is no longer viable)
- the compatibility of an approach with definitions of net zero

2.11 Costs

The costs associated with each approach will have a significant effect on whether it can be implemented of not. The costs associated with project implementation include:

- **capital costs** - the amount of funding that is needed upfront to implement an approach
- **operational costs** - the amount of money needed to ensure that an approach continues to reduce or remove GHG emissions. For example, certain forests need to be ‘thinned’ to ensure the health of the trees and to maximise sequestration rates,

which requires ongoing funding. In addition, it is likely that costs will be associated with undertaking project monitoring and achieving carbon credits
- **opportunity costs** – the foregone income that would have been earned had an offsetting project not been implemented. This would, for example, include the lost income of a farmer who converts productive land to forest

These costs can be presented as a **cost of carbon abatemen**t. This is the total amount of money that is spent per tonne of CO_2e reduced or removed. This captures both capital and ongoing costs. This is sometimes captured as:

- the ‘marginal abatement cost’, which attempts to calculate the cost of cutting one more tonne of CO_2e. As ‘easy’ emissions are reduced, decarbonising the next tonne of CO_2e becomes progressively harder and more costly
- the ‘lifetime cost of carbon abatement’, the cost of an individual project divided by the lifetime carbon sequestration or reduction

There is often a high level of uncertainty in these figures because they depend on a variety of assumptions. These include the lifetime of a project (which can vary greatly), the project’s continuing ability to reduce emissions (particularly pertinent for built environment measures), the costs that should be included in the calculation, as well as the forecasts of these costs. Where available in the literature, marginal abatement costs are given.

Chapter 3. Land-based approaches

3.1 Upland peat restoration

The following 2 chapters cover the restoration of upland and lowland peatlands.

In climate change terms, the greatest impact will be the initial reduction of carbon emissions resulting from halting the loss of carbon from the degraded peat. Once the peatland is restored, it may then revert to being a carbon sink. Peatland restoration is therefore a reduction strategy rather than a removal strategy.

Table 3-1: Summary results for 'Upland peat restoration'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Reduction then removal	10-100	2-20	Very high[5]	Ready	Ready	>10	Long term	Medium	High

3.1.1 Approach overview

The UK is one of the top 10 countries in the world in terms of total peatland area. Although most of it is located in Scotland, England has 670,000 hectares (ha) of deep peat soil. Just over half of these deep peats, 355,000 ha, are located in the uplands (RSPB, 2014). Wales has some 70,000 ha of upland blanket peat soils (Welsh Assembly Government, 2011).

The most common type of peat in the uplands is blanket bog. Blanket bogs are semi-natural habitats in which water is sourced solely from rainfall, mist and snow, and which accumulate with their own 'perched' water table on upland plateaux. In England, upland bogs are mainly found on Dartmoor, the Lake District, the Peak District and the Pennines, but also to a lesser extent on Exmoor and the North York Moors.

[5] The categories for national abatement potential correspond to the following ranges: 'Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

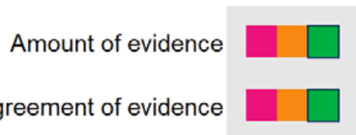
These upland deep peats provide several regulating services, including climate regulation through carbon storage, with an estimated 140 million (Tonnes of carbon (tC) stored in England's upland deep peats. Other services include water quality regulation and water quantity regulation (valuable from both a water resource and flood risk management perspective). Upland peats are of high value for conservation and biodiversity.

Land uses of upland peats include extensive livestock grazing, plantation forestry and grouse shooting. Plantation forestry and grazing tend to be of low productivity (although this is not always the case), and establishing tree cover on upland peats is considered to be ecologically undesirable. Indeed, much of the upland peat in England and Wales has been degraded by drainage, overgrazing, plantation forestry, rotational burning for grouse shooting and pollution (both historical pollution in the form of acid rain and ongoing nitrogen deposition from vehicles and intensive livestock farming) (Ramchunder and others, 2013), (Evans and others, 2016). As a consequence, the majority of upland peat is physically degraded and nearly half of it has been cleared of peat forming vegetation. Today, only 4% of the deep peat is in good enough condition to still be actively forming peat, and the majority of upland peat is losing carbon to the atmosphere and into water bodies. These degradations are all the more concerning as they increase the vulnerability of upland peat to climate change, which could result in a three-fold increase in the rate of carbon lost over the next few decades (Committee on Climate Change, 2013).

Restoration is therefore crucial to prevent future emissions. Upland peatland restoration is a reduction rather than a removal measure. It aims not to sequester carbon but to prevent future emissions. With this in mind, the low agricultural productivity of grazing and plantation forestry in upland peat means opportunity costs of restoration are much lower than for lowland peat. Therefore, although emissions savings may be slightly lower for upland peat restoration, the majority of peatland restoration efforts are taking place in the uplands. Efforts already in place include blocking drains, revegetating areas of bare peat, and changing fire management. While such restoration does not focus on reducing carbon emissions specifically, these efforts will increase the resilience of upland peatlands to nitrogen deposition and enhance their potential recovery from past pollution (Evans and others, 2016).

However, these efforts only cover one third of mapped upland peatland, with substantial areas of unrestored degraded peat remaining, both mapped and unmapped. The latter include shallower peats that are also at risk of degradation and loss. There is therefore significant potential for a wider uptake of upland peat restoration. This additional uptake could yield large net benefits, even under conservative assumptions about emission savings, allowing for high costs. Furthermore, increasing restoration efforts is necessary before climate change makes degradations irreparable (Committee on Climate Change, 2013).

3.1.2 Readiness for implementation



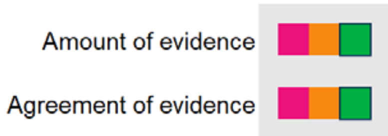
As people have been managing wetlands and peatlands for many years, a great deal is known about restoration. The restoration of degraded upland peats is therefore ecologically and technically feasible and is ready to be scaled.

There are 3 main types of peatland restoration measure. First, revegetation involves using dwarf shrubs and seeds to re-establish vegetation over bare peatlands. This has been shown to reduce particulate organic carbon (POC) release. Second, rewetting drained peatland involves blocking drainage ditches and gullies and reduces sediment loss. Third, vegetation management for rotationally burnt or overgrazed peatland consists of reducing or ceasing burning and reducing the number of grazing animals. This will reduce dissolved organic carbon concentration (Committee on Climate Change, 2013).

Partnerships to restore upland peatland already exist and are often motivated by improving the water supply. They involve the water industry (for example, South West Water and Yorkshire Water), National Park Authorities, environmental bodies (such as Natural England or the Environment Agency) and non-governmental organisations such as the National Trust or RSPB (Committee on Climate Change, 2013).

Different mechanisms already exist to pay for these peatland restoration projects, some of which are funded by the UK government. For example, Defra invested £10 million in peatland restoration in 2017 to 2018 and the Welsh government is funding restoration through EU LIFE[6] funded projects. Private funding can also contribute to restoration projects for their carbon benefits via the Peatland Code. Developed by the International Union for Conservation of Nature (IUCN) and launched in 2015, the Peatland Code is a certification standard for UK peatland projects wishing to market the climate benefits of restoration. The Code creates a standard and builds market confidence towards funding restoration projects for these climate benefits.

3.1.3 Speed and scale



A near natural bog can remove 3.54 tCO_2/ha/year. However, peatlands are also a source of nitrous oxide and methane due to atmospheric nitrogen deposition and their waterlogged nature. When these GHGs are included, near natural bogs are close to climate neutral (Evans and others, 2017).

However, the majority of England and Wales' upland peats are not in near natural state. Rather, they are degraded. This turns them into carbon sources, both of carbon dioxide and of methane, the latter of which are especially linked to gullies (McNamara and others, 2008). Overall, upland (and lowland) peatlands today are a large net source of emissions. The Peatland Code estimates that GHG emissions of degraded blanket bogs range from 2.54 tCO_2e/ha/year for modified sites through to 23.84 tCO_2e/ha/year for actively eroding bare peat (Smyth and others, 2015; Committee on Climate Change, 2019b).

Peatland restoration reduces these emissions over time. In the Peatland Code, the carbon benefits of restoration are estimated based on the GHG emissions emanating from different levels of degraded peatlands (as described above). They are calculated as the difference

[6] The LIFE Programme is the EU's funding instrument for the environment and climate action created in 1992.

between emissions from peat prior to and after restoration (see Table 3-2). For example, the value 'saves 21.30 tCO_2e/ha/year', describes the reduction in emissions when peat is restored from an actively eroding state to a drained state, and is calculated as follows:

$$\textit{Emission factor for actively eroding peat} - \textit{Emissions factor for modified peat}$$

$$= \mathbf{23.84 - 2.54 = 21.30}$$

Table 3-2 Net GHG effect from upland peat under different management approaches. The emission factors used were calculated based on a literature review of carbon emissions from degraded peat (Smyth and others, 2015)

Condition category change	Net effect (tCO_2e/ha/year)
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 from actively eroding to modified	Saves 21.30
Restoring from actively eroding to drained	Saves 19.30
Allowing drained to develop into actively eroding	Loses 19.30

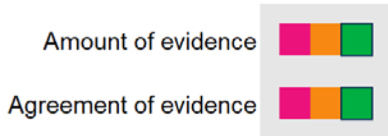
Scaling up restoration is a national ambition, with the Committee on Climate Change advising the restoration of around 20,000 ha each year by 2050 (Committee on Climate Change, 2019c). In 2014, the Royal Society for the Protection of Birds (RSPB) advocated restoring 200,000 ha of England's damaged upland peatland (RSPB, 2014). This call to action shows that efforts are already in place to incentivise and regulate peatland restoration efforts, mainly to reduce carbon emissions (Natural England, 2010). According to a Natural England report, if all of the blanket bog peatlands in England were restored, and assuming a 40-year time frame where restoration emission rates occur for a decade followed by 30 years of emissions typical of undamaged blanket bog, the estimated emissions reductions would be 0.86 Mt CO_2-e yr^{-1}.

The time it takes for a restored site to revert to an undamaged state varies between sites. Re-establishing peat-forming vegetation takes 5 to 10 years, but changes in greenhouse gas balances are harder to discern, and it may take decades for peatland functionality to recover (Committee on Climate Change, 2013). In turn, the time it takes for a restored site to revert to an undamaged state varies between sites. The speed of emission reductions varies based on (1) how long ago restoration occurred and (2) the initial state of the peatland. First, there may be high carbon benefits in the initial years after restoration because of rapid vegetation re-establishment and growth, which leads to an uptake of carbon from the atmosphere. Carbon sequestration then decreases as the restored peatland reaches a more stable phase. Second, restoration of less damaged sites can revert to sequestering small amounts of carbon within a <10-year time frame. In contrast, severely damaged sites may show only modest results in terms of reverting to carbon sequestration even after 50 years (Artz and others, 2012). Overall, restoration of severely damaged peat rapidly reduces emissions but it can be decades before these actively sequester carbon. In contrast, restoring less damaged sites does not reduce emissions as much, but can achieve stable near natural conditions and accompanying small scale sequestration within a decade. This sequestration will not necessarily be as high as for natural sites. Indeed, carbon storage and accumulation of soil organic matter can remain lower in restored lowland peatland sites compared to natural sites even after 50 to 100 years (Moreno-Mateos and others, 2012).

Theoretical maximum abatement potential for this measure in the UK

The theoretical maximum abatement potential for upland peatland restoration in the UK has been categorised as very high. To obtain this category, the approximate per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. For this measure, the theoretical maximum area was derived from the total area of upland peat in the UK and the estimate that only 4% of this is in good condition. This area stands between 2 million and 3 million hectares (Artz and others, 2019).

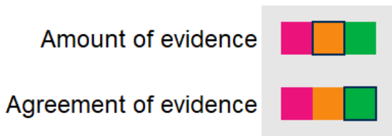
3.1.4 Co-benefits



There are multiple co-benefits associated with restoring upland peat. These include benefits for water supplies in England and Wales. Peatlands are the headwaters for some of the countries' major water supply catchments and supply reservoirs in several regions. For example, peatlands in the Peak District provide water for the 4 million people living in the areas of Sheffield and Manchester (RSPB, 2014). Water derived from restored or functioning peatlands is naturally of high quality because of low weathering rates and low human impacts (Committee on Climate Change, 2013). In contrast, degraded peatland can cause discolouration of water, which is expensive for water companies to remove.

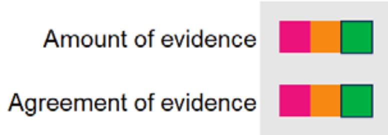
Co-benefits also exist for biodiversity. Nearly 40% of upland peats in England are designated as Sites of Special Scientific Interest and blanket bogs make up one-fifth of all Special Areas of Conservation (SAC) in England. Blanket bogs are important nesting and feeding habitats for upland breeding bird species and are home to several rare invertebrate species, including dragonflies. Some sphagnum mosses typical of bogs are priorities for conservation (Committee on Climate Change, 2013). Restoration will increase populations of these valuable species.

3.1.5 Confidence in the science



Studies which measure the carbon losses resulting from degradation exist (Clay and others, 2010), as do studies estimating the impact of different climate change scenarios on future upland peat degradation (House and others, 2011). The long-term impact of bog restoration on preventing these losses is less well known and quantified, mainly because, as of yet, there are few long-term studies of the effect of peatland restoration on carbon emissions. Close long-term monitoring of ongoing restoration efforts will provide more insights. In the meantime, estimates are provided in the Peatland Code.

3.1.6 Measuring impact

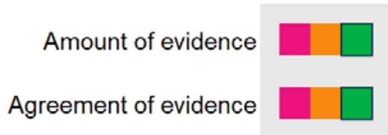


Empirically, the carbon budgets of peatlands are complex to calculate. However, calculation remains possible and is based on fluxes of carbon through different pathways. It involves measuring or estimating CO_2 exchange, dissolved CO_2, methane (CH_4) emissions, dissolved inorganic carbon export and input from weathering of underlying strata (Clay and others, 2010). Assumptions can be made regarding the links between some of these variables, temperature and water table depth (see Clay and others, 2010).

Within the Peatland Code, removals are estimated using default values based on the type of restoration practices, on the type of peatland and on its state prior to restoration. The baseline scenario is established using different categories, which describe how deteriorated the peat is, for example, whether or not it is actively eroding.

The Peatland Code specifies that validation is required before implementing the restoration plan. Validation determines if it conforms to agreed requirements and if its implementation will result in specific GHG benefits. During implementation, regular verifications evaluate whether the project is complying with agreed requirements. Both consist of reviews of documentation and of site visits.

3.1.7 Risks and barriers



Restoration may have social impacts. Livestock grazing is a common land use for upland peats. Restoration may involve reducing the intensity of grazing. Restoration efforts should include provisions so that these reductions do not harm farmers in the long term. Reducing or ceasing ecologically-damaging rotational burning will also have an impact on red grouse shooting (Heinemeyer and Vallack, 2019). Grouse shooting contributes to the upland economy and is a traditional leisure activity. Alternative peat management which does not involve burning will affect this.

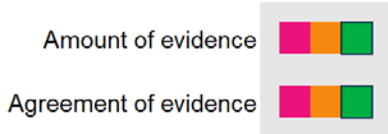
Functioning peatlands may be susceptible to climate change, but there is evidence that they may be able to self-adapt and be resilient (Committee on Climate Change, 2019b). In contrast, degraded peatlands are very vulnerable to climate change, as is their potential for carbon storage (House and others, 2011).

Therefore, rather than being harmed by climate change, restoration of upland peat will reduce the vulnerability of the habitat to climate change.

Post-Brexit and with the abolishment of the Common Agricultural Policy (CAP), Defra’s intention to focus on ‘public money for public goods’ is conducive to a focus on wider ecosystem services. Likewise, loss of generic support from the CAP in upland areas may lead to a reduced focus on agricultural production and greater focus on the wider ecosystem perspective promoted by Defra (Committee on Climate Change, 2019b). This contrasts with lowland areas, where reliance on subsidies is lower and market returns are the main agricultural income. Adopting a wider ecosystem perspective – one that is compatible with restoration – is therefore less costly and more likely in upland areas (Committee on Climate Change, 2019b).

Although government policies and funding are prioritising peatland restoration, one risk is that they outcompete and disincentivise private investment via the Peatland Code. Public and private funding need to be integrated to optimise overall funding available to scale up restoration (IUCN, 2020).

3.1.8 Costs



Capital costs include upfront costs like erosion control nets to stabilise bare peat, dams for blocking drainage gullies or fencing from stock control. They vary widely, ranging from £150 per hectare to £7,000 per hectare (see Table 3-2).

Ongoing costs include management and monitoring costs such as replacement of dams. They may also involve opportunity costs if restoration displaces current land use activities like grouse shooting or farming. The opportunity costs for restoring upland peats range from £25 to £200/ha/year (see Table 3-3). They are comparatively lower than opportunity costs in lowland area. This is because predominant land uses (livestock grazing and plantations) in upland areas are unproductive relative to agriculturally valuable lowland peat. The value of displaced production is, therefore, lower in upland areas.

Overall, the Adaptation Sub-Committee of the Governmental Committee on Climate Change found that predicted benefits of restoration of upland peats were higher than these costs. These benefits also increase with the magnitude of climate change (Committee on Climate Change, 2013). This increases the economic rationale for restoration.

The Peatland Code provides a mechanism to cover the costs of restoration. It is a ‘place-based’ Payment for Ecosystem Services (PES) scheme (Reed and others, 2017). This means it looks at the multiple ecosystem services provided by peat in specific locations, and establishes relations between specific service users and providers (Reed and others 2017). Because of this, prices for restoration vary depending on what is agreed by buyers (of carbon credits) and sellers (those undertaking restoration efforts). In the Peak District, for example, the price agreed on was £54 and £107/tCO_2e equivalent removed for revegetation and ditch blocking respectively. The prices vary based on the costs of restoration and the perceived risk for stakeholders (Reed and others, 2017). This same study identified a price per tonne of £11.18/tCO_2e at a restoration site near Dumfries in Scotland.

Table 3-3 Estimated costs for 5 restoration options (Committee on Climate Change, 2013)

Restoration option	Capital costs (£/ha)		Ongoing costs (including opportunity costs) (£/ha/year)	
	Low	**High**	**Low**	**High**
Revegetation of bare peat	200	7,000	25	100
Grip blocking	150	600	25	200
Gully blocking	1,000	4,000	25	100
Reduced burning	0	300	25	200
Reduced livestock intensity	0	3,000	25	150

3.2 Lowland peat restoration

Table 3-4 Summary results for 'Lowland peat restoration'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	**Per unit (tCO_2e/ha/year)**	**National abatement potential**	**Technology**	**Certification method**	**Years**		**Evidence volume**	**Evidence agreement**
Reduction then removal	Uncertain	5-20	High [7]	Ready	Not ready	>10	Long term	Medium	High

3.2.1 Approach overview

Lowland peatlands, which comprise fens and raised bogs, are defined as peat soils under 200m altitude which are formed under waterlogged conditions. Fens are relatively extensive areas of low-lying wetland based on peat soils which receive water from various sources, including groundwater and surface run-off. Raised bogs are localised domes of peat fed mainly by water from precipitation, covering just 3,300 ha in England, Wales and Northern

[7] The categories for national abatement potential correspond to the following ranges: 'Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

Ireland (JNCC, 2008). Due to the significantly larger area of fens, and therefore considerably higher potential carbon sequestration, they are the main focus in this chapter.

In England, lowland peatlands cover around 325,000 hectares. Currently, around 74% - 240,000ha – are used for agriculture (Morris and others, 2010). In the UK as a whole, lowland peatlands cover 465,000 hectares and one source reports that 90% of this has been drained for agricultural use (CEH, 2016). These drained areas now comprise some of the highest-value arable land in the UK (Evans and others, 2017a).

In England, the fens in East Anglia comprise the largest and most intensively modified area of lowland peat. Almost all of this has been drained and only 800 ha exist as undrained fen, protected in nature reserves (Peacock and others, 2019).

Drained peatlands have higher rates of respiration and, therefore, emissions of CO_2 than intact peatlands (Haddaway and others, 2014). Drainage significantly alters the dynamics of the peat soils, leading to a significant increase in emissions of greenhouse gases as peats are oxidised, including CO_2 and, in areas where the rewetted peat was previously fertilised, nitrous oxide (Evans and others, 2017a; Peacock and others, 2019). Having been a carbon sink when intact, lowland peatlands are consequently now one of the largest sources of GHGs from land use in the UK (Evans and others, 2017a).

Globally, it is estimated that draining and burning peat soils accounts for 5% of human greenhouse gas emissions. The most recent IPCC report emphasises the importance of cultivated peatlands as a significant source of GHG emissions (Peacock and others, 2019). Some peatlands have also been degraded by extracting peat at extraction sites. However, our focus here is on drained peatland used for agriculture.

Restoration of lowland agricultural peatlands can halt the GHG emissions caused by the oxidation of drained and cultivated peat soils and, in some cases, allow them to revert to being a carbon sink (Peacock and others, 2019). Lowland peatlands managed under conservation management appear to be among the most effective carbon sinks per unit area in England and Wales (Evans and others, 2017a).

Restoration of peatland involves ‘rewetting’, in which channels which were created to drain the peatlands are purposely blocked in order to restore water levels. A higher water table is the main driver of lower CO_2 emissions. However, waterlogging also re-establishes methane emissions. Despite this, the reduction in CO_2 emissions generally offsets increased methane emissions (Günther and others, 2020). An optimum water table depth in terms of GHG balance is around 0 to 10cm (Evans and others, 2017a). This is high enough that it allows a return of wetland vegetation, but it is not so high that waterlogging would drastically increase methane emissions.

Restoration of peatland may also involve revegetation where peat has become bare. This may be as straightforward as excluding grazing animals and allowing natural regeneration of vegetation, but may also involve establishing nurse crops and manually introducing seeds. Vegetation management such as removing scrub or woodland that has developed may also be necessary (Lunt and others, 2010). Sites may need to be subject to ongoing extensive or ‘conservation’ management. These management actions and the technical expertise required for them are well established, but they do involve significant earthworks.

Successful restoration of lowland peat on cultivated land can be challenging due to significant losses of peat soils, compaction of remaining peat, loss of seed banks, the legacy of nutrient enrichment through addition of fertilisers and heavy modification of drainage systems. In some cases, complete restoration to original lowland peat systems is impossible because hydrological characteristics have been irreversibly changed by historical modifications like drainage infrastructure (Klimkowska and others, 2010). In these cases, the best that can be done is to convert the land to semi-natural fen meadows, whereby some intermediate level of water level and vegetation management persists (Peacock and others, 2019). Introducing management of agricultural peatlands, whereby measures to reduce drainage-related GHG emissions are implemented, for example, by shifting from deep-drained to shallow-drained cropland or grassland, are an alternative (C Evans and others, 2017, p. 1210). Both of these options still bring benefits in terms of reduced rates of carbon loss compared to conventional agriculture on peatland, but do not achieve the carbon reductions or habitat benefits of fully restored peatland, so the latter should be prioritised wherever possible (Evans and others, 2017a; Peacock and others, 2019).

3.2.2 Readiness for implementation

Amount of evidence

Agreement of evidence

Methods of restoring lowland peatlands on agricultural land are well established in the UK and there are several existing projects which could be a source of information and expertise. These include the Wicken Fen Vision and Great Fen Project, which aim to restore 9,000 ha of wetland in East Anglia, primarily by taking agricultural land out of production (Peacock and others, 2019).

Implementing restoration of lowland peats will require partnership and collaboration with farmers and landowners who currently manage the land for arable agriculture.

3.2.3 Speed and scale

Amount of evidence

Agreement of evidence

In England alone, lowland peatland areas currently account for greenhouse gas emissions of more than 10 $MtCO_2e$ per year. About 60% of this is from lowland peatlands which have been drained for intensive arable agriculture and around 27% from areas converted to agricultural grasslands (Centre for Ecology and Hydrology, 2019). The conflict with agricultural use means it is unlikely that this can all be reduced, however restoring lowland peats in the UK could contribute significantly to reducing the UK’s emissions.

With an emission factor of -0.61 tCO_2e/ha/year, near natural fens are modest carbon sinks (Evans and others, 2017b). Their relatively low emission factor partly results from methane emissions. Due to their waterlogged nature, near natural fens emit CH_4, which counterbalances some, though not all, of the CO_2 they sequester.

However, the carbon benefits of lowland peat restoration mainly centre around halting carbon emissions from degraded peatland rather than reverting them to sinks. Indeed, the priority of restoration efforts is to reduce current high emissions, rather than halting them altogether. Although negative emissions from peat formation are beneficial, they are not the priority of restoration.

Confidence in the emission factors for fens is lower than confidence in emission factors for acidic bog habitats (be they upland or lowland). Because of the often-fragmented placement in the landscape, fens have not been studied cohesively across the UK. As a result, standards like the Peatland Code (described in section 3.1) do not apply to fens (although they can apply to lowland bogs).

Despite this, the few studies providing comprehensive GHG balances for lowland peat show that carbon emissions from degraded peatland are high. Research for Defra published in 2017 provides some of the most complete data, which found the following GHG balances for a number of lowland peatland sites under different management (see Table 3-5), where positive balances indicate net emissions and negative balances indicate sequestration).

Table 3-5 GHG balances for lowland peatland sites under various management practices. A positive value indicates GHG emission to the atmosphere, while negative values indicate sequestration (Evans and others, 2017). The GHG balance includes CO_2 and methane but excludes nitrous oxide.

Land cover	GHG balance (tCO_2e/ha/year)† [8]
Intensive arable cultivation	+23.4 to +28.5
Extensive grassland management	+5.6 to +12.4
Fenland with semi-natural vegetation	-3.6 to +7.7
Extraction site	+6.3 to +7.4
Rewetted former extraction site	+11.3 to +12.9

These measurements show that GHG emissions are highest for peatland sites used for intensive arable cultivation. The remaining ‘lifetime’ for such peat soils if drainage and agricultural practices continue is only around 100 years (Chris Evans and others, 2017). Extensive or conservation management is better in terms of GHG balance, with a lower (although still positive) balance.

Sites that comprise peatland with semi-natural vegetation, that is not managed as grassland or for agricultural production, showed the lowest emissions and, in some cases, these sites

[8] N_2O fluxes not included.

can be overall GHG sinks. However, the results showed that previous peat extraction sites, which have been restored by rewetting, generally show positive GHG balances and, in fact, in the sites presented here, emissions that are higher than un-rewetted sites. This appears to be mainly due to increases in short-lived methane emissions from the waterlogged soils and, although inundation may be necessary to re-establish peatland vegetation, the optimum water level for GHG balance in the longer term would be just below the peat surface (Evans and others, 2017a).

It should be noted that the data presented here reflect just 2 rewetted sites and measurements over only 3 years. Values may be affected by how soon after rewetting the study was carried out, and the significant potential inter-annual variability of emissions (Evans and others, 2017a). Other studies suggest that, in the long-term, rewetting of peatland sites tends to benefit the total GHG balance (Günther and others, 2020).

Methane emissions are important to consider in relation to restoring peatlands. They tend to vary depending on the vegetation species, the depth of the soil and the water levels or inundation. Methane emissions are generally higher in sites undergoing restoration compared to intact fens. Although methane makes up a very minor component of the GHG fluxes over peatland, it is important to consider due to its higher GWP value than carbon (Peacock and others, 2019). Water-logged soils can also create the conditions for denitrification, which leads to the release of N_2O, but emissions are generally lower for restored peatlands compared to agricultural land or grassland (Evans and others, 2017b).

The success of restoring lowland peatlands to being carbon sinks varies depending on the starting state of the land. Some restored fens remain sources of carbon decades after the restoration begins. For example, an area of Wicken Fen in East Anglia, which was restored from arable land, was found to still be a source of carbon – at around 123 g C/m^2/year – 20 years after the introduction of conservation management. This was due to very high losses of peat soils, which meant that the remaining soils were very shallow, and a water table that was too low to allow recovery of wetland fen vegetation (Peacock and others, 2019). However, compared to carbon balances of croplands, which were found to be between 693 and 773 g C/m^2/year for sites in East Anglia, even partially successful rewetting and restoration of fenland can reduce carbon losses by 80%. Fluxes of CO_2 from abandoned agricultural peatlands, which are able to partially recover towards their natural state, are also estimated to be lower than arable peatlands (Peacock and others, 2019).

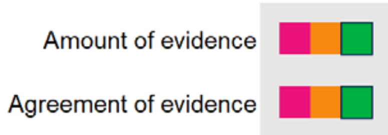
Moving from intensive arable cultivation to fenland with semi-natural vegetation could result in a reduction in emissions of 20.8 tCO_2e/ha/year. Moving from extensive grassland management to fenland with semi-natural vegetation could result in a reduction in emissions of 4.7 tCO_2e/ha/year. Both these assumptions involve moving from the highest rate of emissions in one category to the highest rate of emissions in another. Further evidence is required to understand potential reduction rates in detail following land use change.

Rewetting lowland peatlands drained for agriculture could provide GHG emissions benefits on a nationally-significant level (Evans and others, 2017a). To date, emissions from peatland have not been comprehensively included in the UK's greenhouse gas emissions inventory reporting, but they are due to be included by 2022. Once the full emissions are accounted for, restoring peatland habitats could become an increased priority (Committee on Climate Change, 2020). Over the next decade, restoring less damaged sites could provide relatively significant GHG benefits in the short term initially, by preventing further emissions from the degraded landscape and, in the longer term, by acting as potential sinks. Despite this, it must be noted that carbon storage and accumulation of soil organic matter can remain lower in restored lowland peatland sites compared to natural sites even after 50 to 100 years.

Theoretical maximum abatement potential for this measure in the UK

The theoretical maximum abatement potential for lowland peatland restoration in the UK has been categorised as high. To obtain this category, the approximate per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. For this measure, the theoretical maximum land area was derived from the total area of lowland peat in the UK and the estimate that 90% of this is drained. This area stands between 100,000 and 500,000 hectares (Artz and others, 2019).

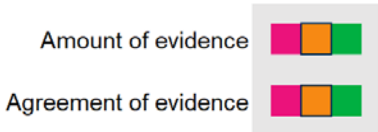
3.2.4 Co-benefits



Healthy lowland peatlands in the UK support a number of rare or threatened plant, animal and bird species and, therefore, have high value for biodiversity conservation. Some typical peatland plant groups, for example, are better represented in the UK than anywhere else in the world, and many are considered to be of European importance (Littlewood and others, 2010).

They also have significant cultural value, and sites like Wicken Fen in East Anglia attract significant numbers of visitors.

3.2.5 Confidence in the science

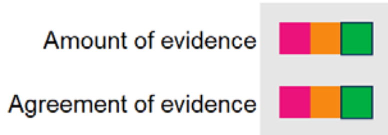


Compared to upland peats, there are relatively few studies providing complete carbon and GHG budgets for lowland peats in the UK. A review for Defra published in 2017 provides the most comprehensive data (Evans and others, 2017a).

There is also evidence to suggest trade-offs between the emissions of CO_2 and other GHGs, specifically methane. This seems to be strongly dependent on the water table, with a higher water level generally reducing CO_2 emissions but, in some cases, increasing methane emissions.

There are still knowledge gaps around the impacts of agricultural fen restoration on carbon cycling and how restored fens compare to those that have never been under agricultural management and have been managed for conservation (Peacock and others, 2019). Factors affecting emissions beyond changes in the water table level are not fully understood (Peacock and others, 2019).

There is also substantial uncertainty around the inter-annual variability in emissions from peatland. Carbon fluxes from peatland can vary significant between years as a result of variation in climatic conditions. It has been shown that even peatland in good condition can be a carbon sink one year and source the following. In the absence of long-term experimental data on the impacts of inter-annual variability for degraded peatland, it can only be assumed that emission factors are similarly uncertain. Indeed, even though most degraded peatlands have been shown to be carbon sources, drained grassland has, in one instance, proven to be a carbon sink (Peacock and others, 2019). This suggests more research is needed on the specific causes of emissions and on the effects of inter-annual variability for long-term sequestration.



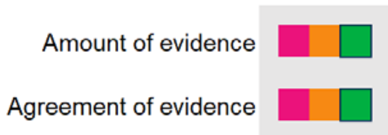
3.2.6 Measuring Impact

In addition to gaseous CO_2 emissions, GHG emissions from lowland peatlands involve significant fluxes of methane. Significant amounts of carbon can also be lost as dissolved organic carbon (DOC) in water. All of these should be measured and accounted for when considering the carbon removal potential of lowland peatlands (Roulet and others, 2007).

Carbon balances in fenlands can be quantified through a number of direct measures. These include measurement of elemental carbon content in soil cores and direct measurement of GHG fluxes using gas chambers and CO_2 eddy covariance measurements (Peacock and others, 2019). Emissions of carbon as (DOC) in water can also be measured directly (Evans and others, 2017a). Assumptions can be made regarding the links between some of these variables, temperature and water table depth (see (Clay and others, 2010)).

Within the Peatland Code, removals are estimated using default values based on the type of restoration practices, on the type of peatland and on its state prior to restoration. However, default values are not currently provided for lowland peat.

3.2.7 Risks and barriers



There is a risk that constraints in water availability lead to changes in vegetation, which may limit the sustained success of restoration (Peacock and others, 2019).

Lowland peatland is vulnerable to the potential changes to the hydrologic regime predicted to result from climate change. For example, GHG emissions can increase when water levels are low due to drought or abstraction (Peacock and others, 2019). Climate change will increase the occurrence of such events. Estimates suggest that emissions from degraded peats may increase by 30% for each degree increase in temperature (Morris and others, 2010). However, compared to upland peatlands, there is little research on whether restoration decreases vulnerability to climate change impacts in the UK.

Short-term changes in management can also significantly impact the carbon balance of restored peatlands. For example, one study found that a rewetted peat grassland was a net sink for CO_2 one year (-147g C/m^2/year) but a source the next year (+88 g C/m^2/year), apparently due to mowing the grass in the second year (Beetz and others, 2013). Reversibility of removals is, therefore, high and long-term sequestration relies on sustained appropriate management.

The greatest CO_2 removals seem to be achieved on more productive fen sites with tall vegetation such as managed reed beds, compared to wetter short fens. However, there is a trade-off to be made with lower plant biodiversity due to the dominance of tall vegetation species (Evans and others, 2017a).

Taking land out of agricultural production leads to a risk of ‘leakage’ in which land elsewhere is then converted to agriculture in order to compensate for the loss in production, potentially resulting in no overall net change in carbon removals (Powlson and others, 2020).

Lastly, unlike bog restoration, fen restoration is not yet accounted for in the Peatland Code because of uncertainty concerning emission factors. The IUCN is working to address the climate, biodiversity and water impacts of degraded lowland peats so it can include fen peatland into its eligibility criteria.

3.2.8 Costs

Amount of evidence

Agreement of evidence

Drained lowland peatlands comprise some of the highest-value arable land in the UK (Evans and others, 2017a), so the costs of displaced agricultural production as well as the direct costs of purchasing land or paying compensation will be high.

However, when considering the balance of agricultural earnings against the economic cost of environmental degradation, specifically, erosion and loss of soil, from continuing to farm on lowland peatlands, it is estimated there will be net losses of -£200 to -£500 per hectare per year (Morris and others, 2010).

Estimates of the mean costs of peatland restoration are around £1,080 to £1,200 per hectare, although this data is for Scotland where most peatlands are upland areas. Costs are likely to be higher on sites where peat is very eroded, and are estimated to be almost twice as high where restoration involves removing trees or scrub (although this applies less to peatlands currently used for agricultural production, as the majority of lowland peatland areas are) (Glenk and others, 2020; Artz and others, 2018).

Cost estimates that include a ‘price per tonne’ for lowland peat restoration could not be identified. Given the lack of evidence, it does not seem suitable to make an estimate based on information not necessarily applicable to the lowland peat context. Further research in this area is, therefore, recommended. Future estimates should take into consideration ongoing management and maintenance costs, in addition to any requirement to compensate landowners for reduced agricultural productivity.

3.3 Woodland creation

Table 3-6 Summary results for 'Woodland creation'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Removal	20-25	11	Very high[9]	Ready	Ready	~10	Long term	High	High

3.3.1 Approach overview

This chapter focuses on afforestation, the conversion of non-wooded areas to woodland by planting trees.

Afforestation can involve broadleaved or deciduous trees (for example, oak, ash or beech, often collectively called 'hardwoods') and coniferous species (for example, Sitka spruce, Scots pine or Douglas fir, often collectively called 'softwoods'). Most tree species native to the UK are broadleaved, deciduous species, but the vast majority of commercial afforestation in the 20th century in the UK involved plantations of non-native coniferous trees.

Conifer plantations are faster growing and, therefore, often sequester carbon at a faster rate than deciduous forests. However, although the rate of sequestration may often be faster for conifers, the final total carbon sequestered per unit area is often greater for deciduous trees as the wood is denser and the biomass of branches is greater (Morison and others, 2012).

This chapter focuses on afforestation on improved grasslands (Morison and others, 2012), since afforestation on arable land is more economically challenging, carried out less often, and has higher risks of 'leakage' due to indirect land use change. The afforestation considered here is distinct from the creation of agroforestry systems, defined as the various combinations of crops and livestock production with trees and other woody perennials.

Non-native conifer plantations were historically established in the UK on these ancient woodland sites as a source of fast-growing timber (Forestry Commission, 2016). Their restoration to broadleaved species is an ongoing policy objective to enhance biodiversity, restore the cultural heritage value of UK woodlands and enhance recreational opportunities (Forestry Commission, 2016). However, although carbon **storage** is frequently higher in

[9] The categories for national abatement potential correspond to the following ranges: 'Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

broadleaved forests (Maclennan, 2019), the rate of carbon **sequestration** is higher in coniferous plantations (Morison and others, 2012). Therefore, although it has multiple other benefits, the restoration of previously ancient woodland sites does not represent a clear or consistent case for increasing carbon sequestration and it is, therefore, not a focus in this report.

Trees absorb carbon dioxide from the atmosphere during photosynthesis, and the resulting biomass is incorporated in the tree as it grows. Major stocks of carbon are in the trunks and roots of trees and in the soils in forests, where it accumulates through the deposition of leaves and other plant matter (in UK woods and forests, up to 75% of the total carbon stock in forests is in soils (Morison and others, 2012)). Establishing woodland can, therefore, contribute significantly to carbon sequestration and to achieving net zero.

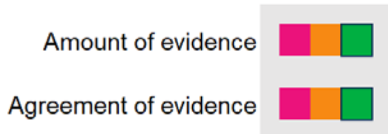
With appropriate weeding and protection from browsing, trees can be established on grassland sites through natural regeneration (if suitable seeds persist in the seed bank or surrounding habitats), direct seeding (uncommon) or planting of young saplings (Morison and others, 2012).

As they grow, trees and forest provide much greater carbon sequestration and a larger stock of carbon in terms of biomass carbon storage than grassland sites, although the impacts are less certain for soil carbon storage (Morison and others, 2012), which is explored in section 3.3.3 Speed and scale.

It is important to note that on organic soils such as peatland soils, afforestation can result in a significant amount of greenhouse gases being released. Therefore, trees should not be planted on organic soils. Areas of species-rich grasslands should also be avoided as they have significant habitat value and the establishment of forest is not necessarily preferable from a biodiversity or soil carbon perspective.

Woodland creation is one of the most established and well known ‘negative emissions’ strategies, reflected by the creation of the Woodland Carbon Code. This is an independently verified code for calculating the amount of carbon dioxide sequestration produced by woodland creation projects in the UK (Woodland Carbon Code, 2020). If done in the right places, woodland creation also provides a range of co-benefits for biodiversity, flood risk mitigation and recreation.

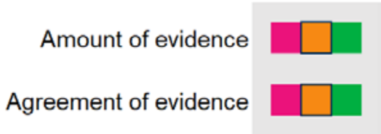
3.3.2 Readiness for implementation



Planting woodland is a well-established practice. In particular, promotion of native broadleaved woodland over conifer plantations has been a policy priority in the UK since the late 1980s (Woodland Carbon Code, 2020). Expertise and examples of woodland establishment in the UK are, therefore, readily available and guidance is available from sources such as the Forestry Commission (Forestry Commission, 2004). The Woodland Carbon Code and Woodland Carbon Guarantee provide established methods for calculating carbon sequestration from woodland establishment and pathways for certified carbon offsets (Woodland Carbon Code, 2020).

If chosen to be a part of an organisation's offsetting strategy to reach net zero by 2030, tree planting should begin as soon as possible as the establishment of trees and the early growth needed to reach the point of peak carbon sequestration takes several years (around 10 to 30 years) (Morison and others, 2012).

3.3.3 Speed and scale



In the UK, the maximum rate of carbon accumulation of broadleaved woodland during the full vigour phase of fast-growing stands is about 10 tC/ha/year, although a realistic average over a full commercial rotation may be no more than 3 tC/ha/year (Broadmeadow and Matthews, 2003). This is equivalent to 11 to 36.6 tCO_2/hectare/year. On a UK-wide basis, the Committee on Climate Change estimates that 8 to 18 $MtCO_2e$/year could be sequestered through afforestation projects by 2050 (it should be noted that this estimate includes both broadleaf and conifer plantations) (Committee on Climate Change, 2018).

The amount of carbon removal and storage varies depending on the baseline condition of the planting area, including soil type, as well as tree species, growth rate and management. Carbon sequestration rates increase at an exponential rate for approximately 30 years following tree planting, with peak carbon uptake occurring during the period of maximum timber development after canopy closure. After 50 to 60 years, the rates of sequestration start to level off and the rates of CO_2 uptake then decline as the stand matures, although the total stock of carbon held in the standing trees continues to increase (Morison and others, 2012)(Maclennan, 2019).

The net carbon removal potential of afforestation depends on the carbon balance of the starting land cover.

Most notably, afforestation affects soil carbon, and its impacts vary greatly depending on the underlying soil type. On organic peat soils, for example, conversion to forest results in significant losses of soil carbon (Morison and others 2012). Trees should, therefore, not be planted in areas with organic soils.

The impact of afforestation on soil carbon in areas previously covered by grassland is unclear. There is some evidence to suggest that grassland soil carbon stocks are higher than those found in forests. However, other studies suggest that soil carbon accumulation increases after afforestation on grasslands and that when trees are established on soils which have been used for long-term grassland cultivation, soil carbon stocks can rapidly increase (Morison and others, 2012). One estimate is that afforestation on these grasslands could increase net soil carbon by 0.37 tCO_2e/ha/year (0.1 tC/ha/year), but could also result in no change to soil carbon (Ostle and others, 2009). However, compared to improved or cultivated grasslands, semi-natural grasslands may have higher soil carbon stocks. Additionally, the biodiversity value of grasslands should be taken into account before trees are planted; semi-natural grasslands with high nature value, for example, should not be considered as areas for tree planting.

Overall, carbon sequestration and storage in broadleaved woodland is generally higher than in improved grasslands. Stocks of carbon in plant biomass of permanent grasslands are typically around 30 tCO_2e/ha, while the average carbon stock in forests is approximately 209 tonnes CO_2 per hectare (Morison and others, 2012). Although biomass carbon stocks in grassland can vary depending on factors including management practices, this is generally exceeded even by young stands of trees (Morison and others, 2012).

3.2.4 Co-benefits

Amount of evidence

Agreement of evidence

Broadleaved woodland is the natural climax of the ecological community in much of the UK and establishing mixed broadleaved woodland is likely to significantly enhance biodiversity compared to permanent pasture or other grasslands (Maclennan, 2019). Broadleaved woodland also offers recreation opportunities and research suggests that the British public prefers the aesthetics of a diverse woodland with varied tree sizes and spacing and some large, mature trees; features that are characteristic of mixed broadleaved woodland (Forestry Commission, 2016).

Establishing woodland can also contribute to nature-based management of water quality, water resource management and flood risk. In to urban settings and on the fringes of towns and cities woodland can provide a range of benefits – such as improving air quality, reducing noise pollution and providing resilience to weather extremes such as heavy rain and high summer temperatures.

Theoretical maximum abatement potential for this measure in the UK

The theoretical maximum abatement potential for woodland creation in the UK has been categorised as very high. To obtain this category, the approximate per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. For this measure, the theoretical available area is taken from the Committee on Climate Change's (CCC's) estimates of the number of hectares of additional woodland required by 2050 if the UK is to meet its carbon neutral target, an area which stands between one and 2 million hectares.

3.3.5 Confidence in the science

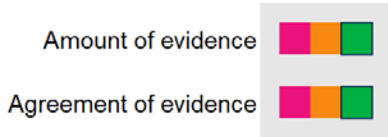
Amount of evidence

Agreement of evidence

There is general consensus about the carbon removal and sequestration potential of trees and forests. Through strategies like the Woodland Carbon Guarantee, the UK government is incentivising tree planting for carbon sequestration (Forestry Commission, 2020).

Nevertheless, there is a lack of studies providing direct quantification of CO_2 uptake or comprehensive greenhouse gas balances for forests in the UK. Variability in the impact of afforestation and reforestation on soil carbon – where a significant proportion of carbon is stored in most habitats – means that it is critical that this is taken into account when planning woodland establishment (Morison and others, 2012).

3.3.6 Measuring impact



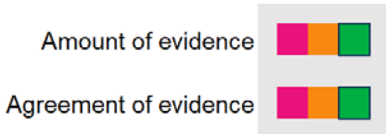
Methods for quantifying carbon stored in forest and trees are well established. They are often based on measurements of biomass focusing on the volume of the tree trunk (‘stem volume’) combined with estimates of carbon sequestration using Forestry Commission standards (Woodland Carbon Code, 2020). However, while the relationship between stem biomass and total tree carbon stock is well known for commercial conifer species, it remains ‘largely unknown’ for many broadleaved species (Morison and others, 2012).

The Woodland Carbon Code provides a publicly available tool and detailed guidance for calculating likely carbon sequestration by trees or forests of different species and under different management regimes (Woodland Carbon Code, 2019a). This has previously been used in Devon and Cornwall by the Environment Agency to estimate carbon removal scenarios based on 2 different tree planting regimes (Maclennan, 2019).

Methods for directly measuring carbon stocks in the soil and litter layers of forests are well established. There are also established methods for directly measuring carbon dioxide fluxes in the air around tree stands, including eddy covariance (Morison and others, 2012).

However, there have been very few studies providing direct measurement of CO_2 uptake or calculating complete carbon balances for UK woodland. There are just 2 long-term sets of measurements – one conifer and one broadleaved – plus some shorter records, mostly over conifer plantations (Morison and others, 2012).

3.3.7 Risks and barriers



There will be a time lag before carbon sequestration from establishing or restoring woodland reaches its maximum; sequestration generally increases for around 10 years after planting and peaks at around 30 years. Therefore, tree planting should start soon to contribute to the 2030 net zero target. Additionally, appreciable sequestration – on the scale to contribute substantially to the Environment Agency’s net zero target – will require very large numbers of trees. Nevertheless, trees planted today will be very valuable for carbon removals beyond 2030 because of the significant sequestration beyond the 10 years following planting (Maclennan, 2019).

Carbon sequestration and storage is dependent on woodland management once trees are established. Here, there can be trade-offs between sequestration and storage. For example, the maximum sequestration rate is higher for thinned than for unthinned stands, because it encourages more vigorous tree growth in the available space. However, the total carbon stock in the trees would be greater in an unthinned stand due to a greater density of standing trees (Morison and others, 2012). Similar trade-offs occur in terms of harvesting. As peak carbon sequestration occurs before tree maturity, maximum long-term removals would be achieved better by a continual process of harvesting and re-establishing trees rather than letting them develop into mature woodlands.

Permanence also depends on the end-use of any harvested trees. If used for fuel, the carbon sequestered during growth will be released during combustion. If the harvested tree is used instead as a substitute for other more energy-intensive materials, for example, as timber in construction instead of concrete, the carbon remains ‘locked’ in the wood and emissions from use of more energy-intensive materials are avoided (Morison and others, 2012). On the other hand, longer standing forests (which are not harvested) represent a stable carbon stock (Morison and others, 2012). Long-term management practices and any potential uses or markets for harvested wood need to be considered from the outset of woodland creation efforts to account for these trade-offs.

Once woodlands are established the permanence of carbon storage carbon in forests is relatively long-term, although it can be reversed by deforestation or by natural disturbance to forests such as fires, disease or droughts (Environment Agency, 2020a).

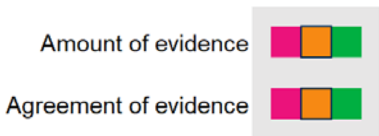
Soil type must be determined before establishing woodland as, on some soils, planting trees can cause carbon losses. This may particularly be the case on some soils previously used for permanent pasture (Valentin, 2019). Under the Woodland Carbon Code, organic soils – those with a deep organic layer – are not eligible for woodland creation for carbon removals due to the fact that carbon losses through disturbance of these soils would most likely outweigh removals by tree growth in the long-term (Woodland Carbon Code, 2019b). One study found that on a site with deep peat soils that were ploughed and afforested, the soil became a net source of carbon for around 2 years after planting, and switched to become a sink after 8 to 9 years of planting. Other studies have found other dynamics, showing that this is an area of uncertainty (Morison and others, 2012).

The natural value of the existing habitat type(s) on areas of land considered for trees planting must also be considered. There have been a number of recent cases of trees being planted on high priority habitats such as peatlands. Grasslands with high nature value (that is, species rich, semi-natural or unimproved grasslands) should also be avoided when considering locations for planting trees (BBC, 2020).

It is also critical to acknowledge that planting trees on grassland used as pasture could lead to displacement of grazing onto other land and, therefore, incurs the risk of leakage (Veldman and others, 2015).

Policies for establishing deciduous trees are often already pursued for biodiversity and conservation objectives (Forestry Commission, 2016). Most tree planting schemes would also be expected to attract funding and investment beyond funds for carbon. As with all potential measures, investment for carbon purposes will need to be scrutinised from an additionality point of view. The Woodland Carbon Code requires projects to source at least 15% of lifetime costs from the sale of carbon offsets, in order to be considered ‘additional’.

3.3.8 Costs



A study on the cost-effectiveness of forestry for climate change mitigation estimates that the costs of afforestation (planting and fencing) in England is around £5,000 per hectare alongside government administration costs of around £640 per hectare (Eory and others, 2015).

In addition, on agricultural land, there are opportunity costs involved due to the displacement of agricultural production. These are estimated at around £220 per hectare per year, but will vary significantly depending on the productivity of the agricultural land (Valentin, 2019). On lower productivity agricultural land, there is some evidence that income from planting woodland may be low or negative without subsidies (Eory and others, 2015).

The costs of establishing broadleaved woodland are higher than for conifer woodland (Valentin, 2019).

Carbon offsetting through woodland projects is one of the most well-developed potential options for carbon offsetting in the UK, especially with the introduction of the Woodland Carbon Code, and, therefore, price per tonne information is more readily available. Recent auctions through the Woodland Carbon Guarantee have resulted in prices between £20 and £25 per tCO_2e. It should be anticipated this price will rise in future as market demand for carbon offsetting solutions increases. It is also important to note that these price per tonne costs only cover a small proportion of total implementation costs, and should not be interpreted as marginal abatement costs.

3.4 Grassland management

In this section on grasslands we have not carried out a RAG rating for each of the different criteria. This is because this approach was not reviewed as extensively as the other approaches covered in this chapter, partly because less science was available and also because it was included at a later stage in the review.

Table 3-7 Summary results for grassland management on road verges

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence Agreement
Removals	n/a	2	n/a	Ready	Not ready	<10	Short/med	Low	Low

3.4.1 Approach overview

This chapter focuses on **road verges**, because that is where much of the relevant research and experience exists. However, much of the information can also be applied to other areas of mown grassland such as embankments along waterways and areas around building assets, which are of more direct operational interest to the Environment Agency.

Landscape features, including road verges and waterway embankments cover a significant area of land in England and Wales. For example, the area of vegetated land on road verges in the UK is estimated to be 2,400km^2 , amounting to 1% of the UK's land area (Phillips and others, 2020; Jorat and others, 2017). This is an area equivalent to the remaining area of lowland species-rich grassland (Bromley and others, 2019).

Where these features support vegetation and are managed, the habitat commonly comprises regularly mown short grass with relatively low species diversity (O'Sullivan and others, 2017). In the case of road verges, this is because they are mainly managed for safety; vegetation is kept short to improve visibility for road users (Phillips and others, 2020). Similarly, on embankments along waterways that need regular inspection, vegetation is kept short to enable easy access and visibility.

Changes in management towards practices aimed at enhancing environmental outcomes could increase the ecosystem services provided by road verges, including carbon sequestration (Phillips and others, 2020). There is evidence that road verges can act as a carbon sink. For example, one study in the US found that road verges had similar carbon sequestration to other grassland areas (Phillips and others, 2020). Additionally, suitably managed road verges can provide biodiversity, aesthetic value and a visual and sound barrier between roads or other built features and the surrounding area.

Management measures to enhance these services commonly include reducing mowing frequency to once or twice a year after plants have had the chance to flower and set seed, planting species-rich grass and herb seed mixes, and establishing shrubs and trees (Bromley and others, 2019; O'Sullivan and others, 2017).

Reduced mowing frequency can lead to increased below-ground biomass development, as well as increased plant abundance and, in some cases, increased species diversity. Reduced mowing also results in reduced carbon emissions from the grassland management regime (Johnston, 2015).

There is evidence that more diverse meadow-like grasslands with a greater diversity of species and abundance of herbs store higher total carbon than low diversity grassland areas, through higher below ground biomass which comprises higher soil organic carbon (Norton and others, 2019; Chen and others, 2018; Cong and others, 2014). Planting seed mixes which specifically contain deep-rooting grass species such as tall fescue and herb species such as plantain and red clover can help to immobilise more carbon at depth in soils (Ostle and others, 2009).

Larger, woody plant species will tend to store more carbon in standing biomass than grasses and herbs. Therefore, areas with a combination of species-rich grassland and some shrubs and trees will tend to have higher carbon sequestration than areas with only grassland. On road verges, vegetation closer to the road could be maintained as grassland to allow maximum visibility, while taller shrubs or trees could be included as a hedgerow along the back edge of the verge (Bromley and others, 2019).

3.4.2 Readiness for implementation

The relevant management techniques for maintaining species-rich grassland are well established. However, road verges and similar features will vary greatly in soil type, slope, existing species composition and micro-climatic conditions. Therefore, designing a suitable management regime can be complex. Management should be tailored to the conditions in a given location, which could be facilitated by input from local environmental experts (Bromley and others, 2019).

3.4.3 Speed and scale

There is relatively limited research on carbon sequestration by habitats associated with road verges or embankments, the impacts of different grassland mowing regimes, or comparing carbon sequestration between regularly mown species-poor grassland and less frequently mown species-rich grassland.

One estimate for the carbon storage potential of vegetated road verges suggests that, globally, road verges cover around 270,000 km^2 and store 0.015 gigatonnes of carbon per year. This is equivalent to 0.56 tonnes carbon/ha/year which, if all converted to CO_2, is equivalent to 2 tonnes CO_2/hectare/year (Phillips and others, 2020).

Carbon sequestration by grasslands on road verges is estimated in many studies to be similar to the average value for other areas of grassland habitat (Phillips and others, 2020).

If shrubs, and especially trees, are established or allowed to develop, the carbon sequestration could be higher (Phillips and others, 2020). Large, fast-growing trees will tend to sequester the greatest amount of carbon, particularly where maintenance or management activities, which contribute to emissions, are minimal (O'Sullivan and others, 2017). However, this has to be weighed against road safety (visibility and tree safety) implications. Or, in the case of embankments, risks from roots to the physical integrity of engineering structures, and the potential for branches to interfere with water flows, site inspections and so on would need to be considered.

3.4.4 Co-benefits

There are numerous potential co-benefits of managing road verges and waterway embankments that allow taller and more diverse vegetation to develop. In particular, there has been increasing attention recently on the potential value of road verges for biodiversity.

The experience of road verges is that they often support higher diversity of plant species than adjacent habitats. This is due to the fact that they can act as dispersal routes for seeds and, therefore, receive inputs of a greater variety of seeds from different plant types. They are also often found to support similar or higher diversity of insects than comparable habitats such as grasslands (Phillips and others, 2020).

Road verges can also provide other benefits, including air quality improvement, local climate regulation, water filtration, flood abatement, erosion control, noise reduction, and aesthetic improvements (Phillips and others, 2020; O'Sullivan and others, 2017).

Providing many of these outcomes may be enhanced by management regimes that encourage the development of more structurally diverse vegetation and an abundance of flowering plants. Reducing mowing to twice a year, using alternating cutting regimes, and spatially varied regimes can all improve the abundance and diversity of flowering plant species and pollinators (Phillips and others, 2020; Garbuzov and others, 2015). Reduced mowing frequencies have the added benefit of savings in fuel, machinery and staffing costs (O'Sullivan and others, 2017).

3.4.5 Confidence in the science

To date, most research on amenity grassland management, most of which has focused on road verges, has generally focused on the potential to provide biodiversity benefits and air and water filtration. There has been significantly less research on carbon sequestration potential.

In general, there are high uncertainties in determining carbon gains or losses associated with changes in grassland management due to the multitude of possible scenarios. These include soil characteristics, slope, existing plant species composition, previous management regime and specifics of the proposed new management regime (Dawson and Smith, 2007).

Impacts of different mowing regimes on carbon are also subject to uncertainties. For example, one study on urban lawns found that the impact of reduced mowing on carbon sequestration is unclear (Watson and others, 2020).

Determining an optimum mowing regime is, therefore, complex. Reduced mowing tends to lead to an increase in above-ground vegetation mass, whereas increased mowing may lead to an increase in below-ground biomass. Intermediate levels of mowing will tend to lead to intermediate levels of both above and below-ground biomass (Dickinson and Polwart, 1982). Therefore, a balanced management regime, which is tailored to the local conditions, will need to be carefully devised.

3.4.6 Measuring impact

The impacts of changing management regimes on amenity grasslands on road verges can be measured in similar ways to those described in section 5.2.6.

3.4.7 Risks and barriers

The main barrier for both road verge and embankment management is that protecting the main function of the asset will and should take priority over inevitably secondary considerations, such as carbon. Additionally, these operational priorities are often locked into long-term management arrangements and contracts (O'Sullivan and others, 2017). For example, the main priority in road verge management is road safety and maximised visibility for road users. This will tend to promote shorter vegetation, which presents less obstruction. Similarly, management on waterway embankments or other sites where visibility, structural integrity and regular management access are necessary will need to be suited to the safety requirements in the specific location.

3.4.7 Costs

There is limited publicly available information comparing the costs of different management regimes for vegetated road verges.

It is likely that reduced frequency of mowing will reduce machinery maintenance, staff and fuel costs. On the other hand, seed mixes, which may be used to establish more species-rich grassland can be costly (Bromley and others, 2019).

There is ongoing interest and investment in better meeting environmental objectives along roads in the UK. For example, Highways England recently designated £300 million for environmental projects around roads (Phillips and others, 2020).

Chapter 4. River and coastal approaches

4.1 Freshwater wetlands – flood plain restoration

Freshwater wetlands comprise areas of water-saturated soils and open areas of water (peatlands, while also sometimes categorised as freshwater wetland, are covered in separate chapters). The following 2 chapters focus on 2 types of wetlands and the potential impacts of their restoration or management for carbon sequestration:

- river flood plain restoration
- management of constructed wetlands

Flood plains are areas of lowland adjacent to river channels. Although the body of rivers themselves act as important channels for moving carbon through ecosystems, their role for carbon sequestration and storage is limited. Flood plains offer the major carbon storage potential in river systems (Sutfin and others, 2016). Here, we focus on river flood plains and specifically on the impacts of restoring more natural river flood plain dynamics. The principles highlighted are also relevant to washlands.

Table 4-1 Summary results for 'Freshwater wetlands – flood plain restoration'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Removal	>1,000	10	Moderate[10]	Ready	Not ready	>10	Med/long	Low	Low

4.1.1 Approach overview

Flood plains are areas of land adjacent to rivers which are periodically inundated when the river floods. Periodic flooding leads to the deposition of alluvial sediments which can create deep soils with high organic matter (OM) and carbon content (European Environment Agency, 2019). This deposition of carbon-rich sediments does not constitute removal of carbon (as it is simply relocated from another system), but flood plain soils can support the growth of diverse vegetation communities which capture atmospheric carbon through

[10] The categories for national abatement potential correspond to the following ranges: 'Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

photosynthesis. This is then incorporated in the flood plains' soils when the plants die and decompose.

There is no one 'flood plain' ecosystem and they can support a range of different habitats, from wet meadows and grassland to marsh, to riparian or flood plain woodland. In natural flood plains, these habitats often exist as a diverse mosaic.

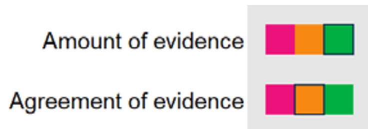
Flood plains are the most widespread freshwater system in the UK, covering 963,700 ha. In the UK, canalisation (straightening) of rivers and construction of embankments for flood protection have separated two-fifths (42%) of the flood plain area from their rivers. Drainage of flood plain land to allow farming or construction has also led to destruction of flood plain wetland habitats. Around 65% of the total flood plain area in England has been extensively altered by agriculture, with the average coverage of any given flood plain by agriculture increasing from between 15 and 60% in 1990 to between 80 and 95% in 2015. Habitats typical of more natural flood plains, like fens, marshes and bogs have been reduced to just 0.5% of the flood plain area in England (Entwistle and others, 2019). This means that there is a large area theoretically available for flood plain restoration (Maltby and others, 2011).

Flood plain restoration involves different measures to restore the hydrological connection between rivers and adjacent low-lying land, restoring 'functional flood plains'. This may involve lowering or removing flood barriers, modifying the river channel and flow, changes to land management and reintroducing native vegetation.

The restoration of more natural flood plain habitats and the recovery of semi-natural standing vegetation can lead to increased carbon uptake by flood plain vegetation relative to that of agricultural land uses. This carbon is then stored in flood plain soils when the vegetation dies and decomposes. Evidence collected at Cricklade nature reserve (Wiltshire) has indicated carbon stocks of 109.4 t C ha^{-1} in the top 10 cm which demonstrates the potential of floodplains to store carbon (Floodplain Meadows Partnership 2018).

The dynamic and interconnected nature of river flood plain systems means that the process involves significant planning and cooperation by large numbers of stakeholders, complex earthworks and disruption to potentially large areas. Additionally, recovery of flood plain habitats will only occur in cases where some 'ecological integrity' of the flood plain habitat still exists. In situations where the land has been entirely cultivated, the flood plains may be 'functionally extinct' and recovery of more natural flood plain vegetation may not occur even if river flood plain connectivity is restored (Entwistle and others, 2019). So, while there is considerable expertise and experience around flood plain restoration techniques, deployment in practice is complex, and there are limited examples at a large scale in the UK.

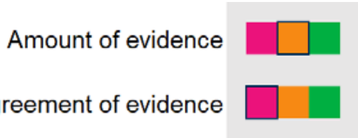
4.1.2 Readiness for implementation



Flood plain restoration techniques are well established and there are many examples of successful flood plain restoration projects throughout Europe (European Environment Agency, 2019), such as the Skjern River Valley in Denmark (Bregnballe and others, 2014). These projects have generally been carried out for purposes other than carbon removal such as wider catchment management and habitat restoration.

Although there is growing interest in flood plain restoration as a measure for carbon sequestration, and some indications of a potentially significant scale of removals, there has been relatively limited research on this to date. It is likely that any future use for carbon removals would (and should) need to integrate and be driven by other landscape management outcomes, and other sources of funding; carbon mitigation may be an added benefit.

4.1.3 Speed and scale



There are 3 major stores of carbon on flood plains; in standing vegetation, in soils and litter and, in some systems, in fallen dead wood.

Standing vegetation

The nature of the vegetation that develops on a restored flood plain has a significant influence on potential carbon removals. Depending on the dominant vegetation species, structure and age or successional stage, estimates of carbon storage range from 7 tC/ha to 360 tC/ha (Sutfin and others, 2016). The highest values tend to be in riparian forests and the lowest in herbaceous meadows or willow scrub, due to the respective amount of plant biomass in the different vegetation types (Sutfin and others, 2016).

Mature forests on flood plains of the Danube River in Austria have been found to hold 160 to 280 tC/ha in the biomass of trees and shrubs, while areas of reforestation comprising younger, recently replanted trees contained 35 tC/ha (Cierjacks and others, 2010). The standing carbon stock in younger stands will be lower than for mature forest, but the rates of accumulation will be higher. Reforestation of a restored flood plain is, therefore, likely to increase carbon accumulation rates, at least in the initial few decades.

In contrast, herbaceous flood plain meadows have much lower biomass carbon stocks. One study found stocks could be around 7 to 21.5 tC/ha (Sutfin and others, 2016).

The vegetation type will also determine the longevity of the carbon storage. Where trees are allowed to mature and stand for long periods, sequestration rates eventually plateau and then decrease slightly as trees age, but carbon stored in the trees lasts for centuries. Stocks of carbon in herbaceous plant and grass biomass will be much shorter lived before they are transferred to other parts of the ecosystem.

Changes in vegetation cover on flood plains, particularly from artificial or hard covering but also from short-mown amenity grassland, towards more natural tall grass, shrublands and woodland will significantly increase the amount of biomass carbon sequestration on flood plains (Natural Water Retention Measures, 2013) although this depends on the tree species and planting density (Morison and others, 2012).

Soils and litter

The alluvial soils deposited onto flood plains often contain high levels of organic matter and, therefore, organic carbon. The decomposition of leaf litter and dead flood plain vegetation also adds carbon to the soil.

The impact of flood plain restoration on the soil carbon balance relative to the current balance on the flood plain isolated from its river will depend on the existing land use on the flood plain and whether this is a source or sink for carbon already. Once hydrological connection is restored, the rate of sediment deposition due to flooding will be the biggest determinant of soil carbon sequestration. Over time, as the flood plain recovers some of its more natural functions, the level of soil saturation and biological influences of developing vegetation will have important impacts over the soil GHG balance.

A study in a section of the Danube River with semi-natural flood plain river dynamics in Austria found accumulation rates of 2.9 tC/ha/year (Sutfin and others, 2016; Tockner and others, 1999), equivalent to over 10 tCO_2/ha/year[†]. Organic carbon content of soil in natural flood plains was been found to be 154 to 212 tC/ha to a depth of 1m (Sutfin and others, 2016).

One study investigating soils up to 1m deep suggests that overlying vegetation does not have a significant impact on soil carbon stocks in flood plains, with soils containing 176 tC/ha under young replanted forest, and an average of 174 tC/ha under different types of mature forest. Soil carbon content was found to be 212 tC/ha under meadow and reed vegetation, the slightly higher value being due to a deeper organic soil layer (Cierjacks and others, 2010). The impact of reforestation also depends on the method of planting. If it involves deep cultivation as part of the land preparation phase, this incurs a larger loss of soil organic carbon.

The maximum potential for carbon sequestration in a river flood plain system depends on a number of characteristics. Some are these will be inherent to location and not possible to change; cooler climates and wider, flatter valleys will tend to favour higher organic carbon sequestration. Other factors may be altered by restoration efforts. For example, carbon retention is likely to be favoured by the creation of more complex and braided channels. However, such restoration represents a very large scale and costly intervention, with the need to convert large areas of land from their current land use to semi-natural and periodically inundated areas (Bregnballe and others, 2014). Carbon removal is also favoured by conditions which allow flood plains to stay wet and management of the river channel and flood plains to encourage accumulation of soils rather than erosion which causes losses or organic carbon (Sutfin and others, 2016).

Theoretical maximum abatement potential for the measure in the UK

The theoretical maximum abatement potential for flood plain restoration in the UK has been categorised as moderate. To obtain this category, the approximate per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. For this measure, this total land area was derived from the additional area which would be needed for all river courses to be naturalised, an area which stands between 100,000 and 500,000 hectares.

4.1.4 Co-benefits

Amount of evidence

Agreement of evidence

Restored flood plain systems can support a variety of habitats and very high biodiversity (Ward and Stanford, 1995; Lawson and others, 2018). They also provide ecosystem services such as natural water retention and purification, flood protection, particularly in the case of flood plain woodland (European Environment Agency, 2019), and recreational opportunities. The ecosystem services provided by fully preserved or restored flood plains are generally greater than on flood plains, optimised only to provide services such as water purification, fish nurseries, arable farming or grazing (European Environment Agency, 2019).

The disconnection of flood plains from river channels, as well as increasing the risk of bank erosion and loss of organic carbon, has, in many cases, led to worse flooding. Indeed, the straightening of the river allows water to flow more quickly, leading to ‘flashy’ responses to high rainfall events, and increased flooding downstream. Restoration is valuable for natural water retention as it helps store and slow run-off and rainwater, reduces erosion and increases infiltration.

The flood, drought and carbon storage benefits provided by floodplain meadows may also help human society adapt to predicted climatic extremes (Rothero and others, 2016).

Projects by the Environment Agency to restore river flood plains would have high potential co-benefits and could contribute to the currently limited evidence about the implications of flood plain restoration for carbon removal potential.

4.1.5 Confidence in the science

Amount of evidence

Agreement of evidence

Compared to other land ecosystems, there is relatively little data available on the carbon stocks and storage potential of flood plains. Among wetlands, river flood plain systems are one of the least researched landscapes in terms of potential carbon storage, with landscapes like peatlands receiving much more attention (Sutfin and others, 2016).

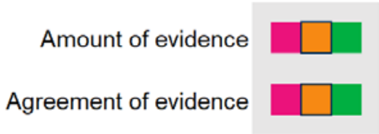
River flood plain restoration projects are usually not implemented with carbon removals as the main objective and consequently, studies of the impacts of restoration do not focus on carbon.

The research that does exist suggests that the carbon removal potential of restored river flood plain networks could be considerable, but that uncertainty remains. The highly dynamic nature of restored river flood plain systems means there is also the possibility for carbon stored in vegetation and soils to be disturbed and re-released relatively rapidly, for example, in the event of an inundation event which remobilises the dissolved organic carbon stored in sediments. The long-term storage of soil organic carbon depends on the residence time of the flood plain sediment, the soil type which will influence the rate of drainage, and on microbial activity and pH.

Total restoration of flood plains involves a complete change in the landscape from a relatively straight channel, isolated surrounding habitat features and relatively predictable flow and flood dynamics to a complex and braided channel with a mosaic of interconnected habitats which may flood unpredictably. Partial restoration may, therefore, be preferable, but the degree of restoration needed to achieve potential benefits of restored flood plains is currently not known (European Environment Agency, 2019). There is also a need for further research into whether or to what extent flood plain restoration can mimic natural processes and, therefore, achieve comparable levels of carbon retention (Sutfin and others, 2016).

Flood plains can comprise a variety of soil types, dominant vegetation types and existing and historical land uses. They are also subject to different river dynamics, including the volume, magnitude and sediment loading of the flow as well as the frequency and scale of flooding. The topography and elevation of the flood plains will also impact deposition. Consequently, their carbon removal and storage potential vary hugely and any interventions will need to be tailored to the conditions at a specific site.

4.1.6 Measuring impact



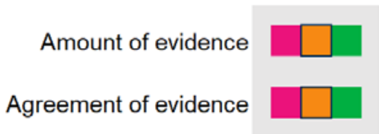
Quantifying removals in flood plains is complicated by the diverse and dynamic land cover patterns associated with a 're-naturalising' river system – all occurring at a large, landscape scale.

In common with other carbon removal scenarios, different components of the flood plain system could have their carbon estimated using field-based sampling and lab-based analysis. These include soil samples and lab testing with carbon-nitrogen analysis to determine total organic carbon (Donovan, 2013). Also, field-based measures of biomass carbon, that is carbon contained within vegetation. For example, for trees, measures of average tree height and width can be taken and biomass calculated using empirical equations. Carbon content can then be estimated using commonly accepted values for the average carbon content of the relevant plant species (Cierjacks and others, 2010).

Within a flood plain system these measures, and sampling designed for them, would need to be stratified across different components of the system, and taken at intervals over time.

More recently, different remote sensing techniques have been applied to assessing flood plain biomass and carbon stocks. Some of these appear to be promising, and estimates of carbon stocks using high-resolution remote sensing data have been found to be comparable to field-based quantifications (Cierjacks and others, 2010; Suchenwirth and others, 2012). These remote methods can be 'ground-truthed' by in-field measurements at a smaller number of sites.

4.1.6 Risks and barriers



The main risks and barriers relate to:

- the complexity, scale and cost of using flood plain restoration as a technique
- variability and uncertainty over carbon sequestration outcomes, both short and long-term because research usually focuses on flood mitigation potential rather than carbon sequestration, leading to limited data sets

- technical barriers in terms of verification and validation of any carbon benefits that may be gained

A major challenge to flood plain restoration is that many currently support settlements, fertile agricultural land and other economically important land uses. Restoration will also tend to involve significant earthworks and potentially large-scale disruption to the immediate river basin as well as upstream and downstream. Resulting challenges include displacement of existing land uses, acceptability to the public of the potentially controversial removal of structural flood defences, and the need for agreement and collaboration between a broad range of stakeholders (European Environment Agency, 2019). While these risks are all significant, they would rarely be weighed against carbon benefits alone. Indeed, the cost-benefit analysis for most potential projects would generally be driven by wider strategic flood plain management considerations.

From a carbon accounting point of view, the works themselves will result in potentially considerable greenhouse gas emissions, which will need to be taken into account when considering the likely greenhouse gas balance of restoration actions.

Flood events in restored flood plains will need to be large enough to ensure deposition of sediment and accumulation of carbon. Moderate flooding levels with limited inundation of flood plains can instead increase the movement of carbon through the system without allowing deposition on the flood plain (Sutfin and others, 2016).

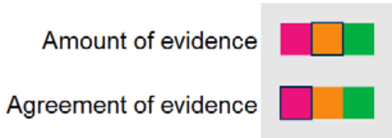
As for some other systems covered here (for example, lowland peats), the effectiveness of flood plains as a carbon sink depends on soil water level. When water levels are low and soils are moist, the activity of microbes that metabolise soil organic carbon (SOC) and release CO_2 can be high. Alternatively, saturated soils can result in lower microbe activity and increased CO_2 storage (Sutfin and others, 2016), although emissions of methane may be higher from waterlogged soils (Evans and others, 2017a). It also depends on soil depth and temperatures, especially for methane emissions, which are temperature dependent and higher during summer flooding events.

There will be a balance to be found between rates of flood plain sediment deposition that are optimal for carbon accumulation and which achieve other objectives. For example, some evidence finds that soil carbon sequestration rates increase with upstream deforestation, but this clashes with environmental objectives which aim to avoid forest clearance. Additionally, it represents the relocation of carbon lost from upstream rather than an overall net removal. Other studies find that degraded flood plain ecosystems sequester less carbon and that restoration is therefore beneficial (Sutfin and others, 2016).

It is estimated that higher temperatures caused by climate change could lead to increased decomposition of organic matter and therefore, release of CO_2 to the atmosphere.

The attention on the carbon removal potential of flood plains is growing, but they are not yet recognised by carbon reporting bodies. This reflects the complexity and uncertainty outlined above. Furthermore, flood plain restoration is already pursued for a number of different environmental objectives and is likely in the future to be driven by these other objectives at least as much, if not more than, carbon agendas. Additionality may, therefore, be hard to demonstrate.

4.1.7 Costs



The costs of flood plain restoration vary greatly because they depend on multiple factors. In terms of restoration, costs depend on soil and hydro morphological characteristics (which have an effect on construction costs and the type of project needed). They also depend on the geographical location of the river basin (for example, if it is more upstream or downstream) and on local price levels, with capital and labour costs. Where restoration involves land acquisition and compensation, the cost of this will depend on factors such as the degree of urbanisation or the degree to which flood plains are cultivated.

For example, the acquisition of land for the restoration of the River Scheldt in Flanders and in the Netherlands cost between 10,000 and 700,000 €/ha depending on whether the land served agricultural or restoration purposes. Construction and rehabilitation for this same river cost 136,542 €/ha, and operation and maintenance is costing 1,226 €/ha/year (European Environment Agency, 2017).

Net costs for carbon would also need to consider the potential value of co-benefits provided by a project. The full cost benefit analysis will, of course, vary depending on multiple factors, but in one example it was found that creating an extra 50 ha of flood plain (in the Norfolk Broads) provides £1 million of carbon sequestration benefits and £27 million of recreational value over 100 years (Tinch, 2011).

4.2 Freshwater wetlands - constructed wetlands management

Table 4-2 Summary results for 'Freshwater wetlands - constructed wetlands'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Removal	Uncertain	Uncertain	Uncertain	Ready	Not ready	Uncertain	Uncertain	Medium	Low

Within this approach we focus on the carbon balance of constructed wetlands. Evidence suggests that constructed wetlands do not necessarily result in net carbon (GHG) removals (Brix and others, 2001). Therefore, we focus mainly on research investigating how different types of constructed wetlands and different management of constructed wetlands may reduce emissions and increase carbon removals from wetlands. The choice to include this approach is based on the prevalence of constructed wetlands in the Environment Agency's operational functions. Although it is currently unclear how to use constructed wetlands for carbon removals, further research on this in this areas could help inform both management and construction decisions in order to minimise emissions and increase removals.

4.2.1 Approach overview

Constructed wetlands are man-made systems which mimic natural wetlands, often using shallow water volumes, aquatic vegetation and non-soil substrates. They are most commonly constructed to treat wastewater through sedimentation, achieve uptake of nutrients by plants and reductions of pathogens through exposure to sunlight and rewet landscapes following historical drainage. They can also be effective as part of strategies to reduce pluvial (extreme rainfall) and fluvial (river) flooding. These processes function irrespective of where constructed wetlands are located, either in rural or urban areas where they are sometimes referred to as sustainable urban drainage systems (SuDS). The UK counts over 1,200 constructed wetland systems (Cooper, 2008), though what area these cover has not been published.

The increasing prevalence of constructed wetlands, coupled with concerns about climate change, has led to growing interest in their GHG balances. constructed wetlands absorb carbon dioxide from the atmosphere through photosynthesis of their vegetation and soil formation, contributing to above and below ground biomass (Barbera and others, 2015). Constructed wetlands also emit methane (CH_4) as a result of organic matter fermentation in their anaerobic soils (Brix and others, 2001) and nitrous oxide (N_2O) as a result of nitrification and denitrification, both of which depend on vegetation and nitrate run-off and deposition.

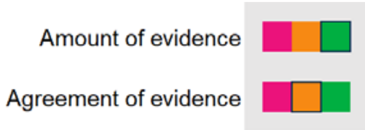
The overall GHG balance depends on the size and scale of the constructed wetland, which range from simple ponds to complex multi-stage wetland systems. It also depends on where and how water flows (Mander and others, 2014), with more detail in the ‘potential scale’ section below:

- free water surface flow: shallow and low flow wetlands with areas of open water and floating plants. These have lower CO_2 emissions but higher methane emissions
- subsurface flow, which can be horizontal, in which case water flows from an inlet to an outlet through a porous medium under the surface of a planted bed. These have higher CO_2 and higher methane emissions
- vertical, in which case a flat porous bed is planted with vegetation and fed intermittently with large batches of water. These have higher CO_2 but much lower methane emissions

Lastly, GHG balances depend on management actions and timescales involved (de Klein and van der Werf, 2014). Therefore, the GHG balances of constructed wetlands are complex, dynamic and multifactorial. Constructed wetlands can, as a result, be sinks or sources of CO_2 and other GHGs depending on their type, management and on the timescales involved as explored in the carbon removal potential section.

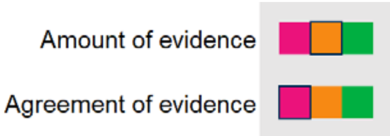
Because of this complexity, it is difficult to recommend constructing wetlands for carbon sequestration purposes. However, it remains valuable to look into ongoing research on the carbon balances for different types of constructed wetlands and how these are affected by specific management practices. This summary, therefore, considers carbon balances for constructed wetlands and evidence on how management practices, such as supporting the development of vegetation, can affect these budgets. The multifactorial nature of carbon fluxes in constructed wetlands means there is no single recommendation for how to manage them as carbon sinks; case by case assessment of sites is necessary to determine what management will be suitable in a given constructed wetland so that they are carbon sinks rather than sources.

4.2.2 Readiness for implementation



Methods for constructing and managing constructed wetlands are well known (McKenzie and McIlwraith, 2015). Methods to measure carbon and nitrogen emission factors for wetlands are also established and have been applied to constructed wetlands on a case-by-case basis, but have not yet been applied to determine general trends in carbon balances of constructed wetlands.

4.2.3 Speed and scale



There is some empirical data on carbon budgets for different types of constructed wetlands, which show that they are both carbon sources and sinks. For example, a recent study of a multi-functional constructed wetland dominated by emergent phragmite vegetation in the Netherlands found it to be a net sink of GHGs (2.7 to 24 tCO_2e/hectare/year) (de Klein and van der Werf, 2014). In contrast, another study of 4 constructed wetlands in Northern Europe with varying vegetation types and hydrology (horizontal and vertical subsurface flow, free surface water, and overland and groundwater flow (OGF) found them to be GHG sources, ranging from 0.057 to 0.26 tCO_2e/hectare/day in the summer and 0.0083 to 0.051tCO_2e/hectare/day in the winter (note that these values are per day rather than per year) (S∅vik and others, 2006).

These budgets are valuable for understanding the processes of carbon and nitrogen fluxes, however, they are site specific and findings cannot be transferred to other constructed wetlands this is because emissions depend on multiple variables such as: wetland type, water flow, climatic conditions, temperature, the presence and composition of vascular plant species and plant management (Barbera and others, 2015).

There is, therefore, no general consensus concerning whether the 1,200+ constructed wetland systems in the UK are carbon sinks or sources. Rather, the consensus is that this is highly dependent on the type of wetland and on its management, and that we do not have general principles and models for carbon balancing yet. This means that recommending a specific measure for carbon removal by constructed wetlands is complicated. What follows is, therefore, an overview of the research on how different characteristics of constructed wetlands have variously been found to result in reductions of emissions and removal of carbon.

First, whether constructed wetlands are viewed as sources or sinks depends on the timescales involved because of the different half-lives of methane and CO_2 in the atmosphere. For example, a study of common reed wetlands concluded that these are sources of GHGs over decades because the stronger warming effect of methane overrides CO_2 sequestration. However, they can be sinks over longer timescales because the shorter half-life of methane (methane has a stronger warming potential but lasts significantly less time on average than CO_2) (Brix and others, 2001).

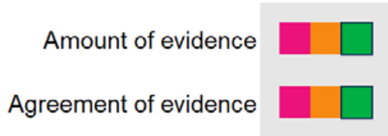
Emissions of these different GHGs depend on the type of water flow (Mander and others, 2014). No significant differences in N_2O emissions were found for different types of flow. However, carbon dioxide emissions in free water surface constructed wetlands have a median value of 95.8, lower than the median emissions from horizontal and vertical subsurface flow constructed wetlands of 137.0 $mgC/m^2/hour$. In contrast, vertical subsurface flow constructed wetlands have lower methane emissions than free water surface and also horizontal subsurface flow constructed wetlands. As detailed above, the carbon impact of emissions, therefore, depends on the timescales involved. Free water surface constructed wetlands have higher carbon impacts in the short term (because of higher methane emissions), but lower ones in the long term because of lower CO_2 emissions. Whereas vertical subsurface flow have lower carbon impacts in the short term (because of low methane emissions), but higher ones in the long term (because of higher CO_2 emissions).

The removal of carbon and nitrogen in constructed wetlands also depends on hydrological conditions. Longer hydraulic retention times (the average length of time that water remains in the constructed wetland) increase the removal of carbon and nitrogen by increasing sedimentation and the duration of contact between nutrients and the vegetation of constructed wetlands (Jahangir and others, 2016). It has also been suggested that fluctuating water levels (sometimes referred to as 'pulsing hydrology' because water is present intermittently) reduce methane emissions in free water surface flow constructed wetlands and horizontal sub-surface flow constructed wetlands (Mander and others, 2014). But they can also increase CO_2 emissions and N_2O emissions if water-filled pore space increases above a certain level, resulting in a shift from nitrification to denitrification, which increases N_2O emissions (Maucieri and others, 2017).

The removal of carbon and nitrogen in constructed wetlands also depends on the extent and type of vegetation in the constructed wetland. Plants influence CO_2 and methane emissions and uptake in constructed wetlands through their root systems (with oxygen release and by increasing the activity of microbes and bacteria), as the main source of carbon for micro-organisms, and by regulating constructed wetland hydrology and temperature (Barbera and others, 2015). Overall, extensive aquatic plant cover seems to suppress methane and N_2O emissions on free water surface constructed wetlands (Mander and others, 2014). However, this depends on temperature and light (de Klein and van der Werf, 2014), as well as on the species and type of aquatic plant. For example, the common reed (P. australis) and willow have been shown to reduce methane emissions, whereas hare's-tail cotton grass (Eriophorum vaginatum) or bulrush (Typha latifolia) lead to methane emissions. It has also been found that mixed species remove more carbon and nitrogen pollutants than monocultures because they increase microbial biomass and diversity, but empirical evidence in this respect is scarce (Jahangir and others, 2016).

Over the next decade, further research on the variations in GHG emissions from constructed wetlands over time and in different locations could help uncover clear models which account for such variations by type of constructed wetland. These models could, in turn, help inform management and construction decisions on a case-by-case basis so as to minimise emissions. For example, further research can help determine which vegetation should be used to minimise emissions in which conditions, or which type of wetland is most beneficial for carbon sequestration. As such, partnerships with researchers and universities are key to establishing specific management practices as carbon offsetting approaches. As an owner and manager of many constructed wetlands, the Environmental Agency is well situated to support such research.

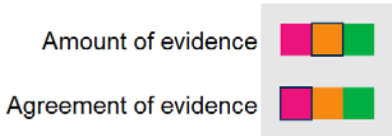
4.2.4 Co-benefits



The benefits of using constructed wetlands rather than conventional wastewater treatment facilities are well known and documented. Their main benefit is that they treat wastewater reliably and continuously, usually in a cost effective and environmentally friendly way.

Yet constructed wetlands are multifunctional and their use and management has multiple co-benefits. Depending on how they are managed, they can provide reclaimed water for the irrigation of crops and can produce biomass which can be harvested and used for energy generation (Barbera and others, 2015). Although in this case removals could not be used for carbon offsetting as they would not be permanent. If they are correctly managed and depending on their type, constructed wetlands (similarly to natural wetlands) also provide a range of ecosystem services with respect to flood mitigation, regulation of river flows and habitat for plants, mammals, amphibians, fish birds and invertebrates (McKenzie and McIlwraith, 2015).

4.2.5 Confidence in the science



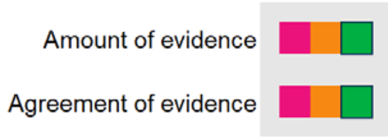
The existing studies on the emissions of specific constructed wetlands show that there are few generic rules for the carbon balances of constructed wetlands. The complexity of emissions and sequestration mean that constructed wetland carbon balances are site specific (de Klein and van der Werf, 2014) and findings from one site cannot currently be taken and applied elsewhere (Jahangir and others, 2016).

Even then, there are gaps in our knowledge, such as what are the effects of changing water levels on carbon and nitrogen removal. Although a fluctuating water table decreases methane emissions, it has also been found to both increase and decrease N_2O emissions; the controlling mechanisms for this are unclear (Jahangir and others, 2016).

Currently the potential for constructed wetlands to reduce or remove GHG emissions is not well understood (Jahangir and others, 2016).

More generally, the efficiency of removing carbon and nitrogen in constructed wetlands is inconsistent and the processes behind it, including pollution swapping (the increase in one pollutant as a result of a measure introduced to reduce another pollutant), have not been thoroughly studied and are not always accounted for in analyses (Jahangir and others, 2016).

4.2.6 Measuring impact

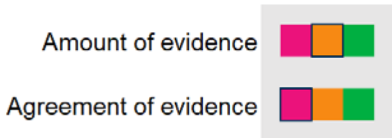


GHG budgets for constructed wetlands are determined experimentally by in-field measurements of carbon sequestration and N_2O and CH_4 emissions.

Soil carbon sequestration can be measured by taking soil samples and analysing them in a lab. Biomass carbon sequestration can be studied by measuring root, rhizome and aboveground biomass production. N_2O emissions can be derived from Intergovernmental Panel on Climate Change (IPCC) default emission factors for the proportion of nitrogen which is released as N_2O. A more time-consuming approach is to use flux meters and sample inflow and outflow of surface water, atmospheric deposition, nitrogen accumulation in the soil, and nitrogen fixation in roots, rhizomes and aboveground biomass (de Klein and van der Werf, 2014). Methane emissions can be determined using a closed cylindrical chamber placed in the sediment of the wetland. However, the closed chamber method for measuring GHG has large uncertainties because of daily variations in emissions. An alternative method is to use bubble traps to measure ebullition and diffusion – how many gas bubbles come up to the surface of the water and are released into the air. This is challenging in constructed wetlands covered by vegetation because of difficulties in estimating how much gas is transferred and how fast (Jahangir and others, 2016).

Once emission rates are obtained for each GHG, they can be expanded and expressed as CO_2 equivalents from which net sequestration is derived.

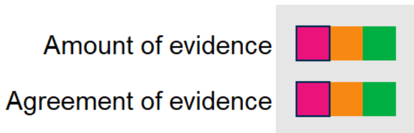
4.2.7 Risks and barriers



Because the GHG emissions for different types of constructed wetlands remain uncertain, it is not yet possible to recommend specific management or construction methods for constructed wetlands in terms of carbon sequestration. It is, therefore, hard to prioritise carbon sequestration in management decisions. As a consequence, long-term additionality cannot be guaranteed yet.

General limitations to using constructed farm wetlands include large land requirements, seasonal variability in the rate of pollutant removal and the need for higher retention times.

4.2.8 Costs



It is not yet appropriate to cost carbon removal for constructed wetlands for reasons detailed above, namely that it is not yet clear how different management decisions affect the source or sink status of constructed wetlands. Moreover, the maintenance costs of constructed wetlands vary widely. Also, the net cost from a carbon procurement point of view, taking into account the value and business case for other outcomes, could vary significantly, and could call into question additionality in many cases.

An overview of capital costs for constructing wetlands can be found in Table 4-3. This again shows great variation and is included for indicative purposes even though constructing wetlands is not the main focus of this chapter.

Table 4-3 The cost of constructing different wetland options (adapted from Wetlands for Life, 2015)

Wetland type	Capital cost (£/m^2)
Swale	10-15
In-ditch field wetland	895[11]
Sediment traps or ponds	5-100
Constructed wetlands (low to moderate strength waste)	4-25
Constructed wetlands (moderate to high strength waste)	5-100

4.3 Saltmarsh restoration

Table 4-4 Summary results for 'Saltmarsh restoration'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidence Volume	Evidence agreement
Removal	Uncertain	2-8	Low[12] [13]	Ready	Not ready	<10	Long term	High	Med

[11] From one case study: Environment Agency and LEAF, 2010.

[12] The categories for national abatement potential correspond to the following ranges: Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

[13] (Beaumont and others, 2014; Burden and others, 2020).

4.3.1 Approach overview

This approach focuses on the restoration of saltmarshes through managed realignment, in which areas of coastal land which were previously or have the potential to develop into saltmarsh are reverted to saltmarsh by removing coastal barriers to restore tidal inundation.

Marine sediments are a significant store of organic carbon as the predominantly anaerobic environment prevents oxidation of carbon to carbon dioxide (Paranjoti, 2016). Coastal habitats including tidal saltmarshes are increasingly recognised as highly efficient carbon sinks (Mcleod and others, 2011).

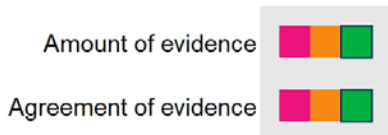
Saltmarshes are the upper, vegetated portion of inter-tidal mudflats occurring in shallow, sheltered coastal areas. They comprise sandy and muddy sediments that are periodically covered by high tides and support characteristic plant communities (JNCC, 2008). Photosynthesis carried out by these plants sequesters carbon in the plant biomass and when the plants die this accumulates in the sediments. Inundation by the tide inhibits microbial activity that breaks down the carbon in the sediments, meaning saltmarshes act as a carbon sink (National Academies of Sciences, Engineering, and Medicine, 2017). They also trap carbon in sediments transported from inland terrestrial habitats via waterways or overland run-off and along the coast by the tide (however, these constitute a loss of carbon from elsewhere in the system and so do not count towards additional carbon sequestration).

Estimates of the current area of saltmarsh around the UK are around 46,000 ha, with approximately 34,000 ha in England and 7,000 ha in Wales. The most extensive areas occur in estuaries in Hampshire, Kent, Essex, Norfolk, Lincolnshire and Lancashire. The area of saltmarsh has declined significantly due to drainage and conversion to other land uses (Burden and others, 2020). It is estimated that over 15% of saltmarshes have been lost since 1945 (Beaumont and others, 2014).

Today, the extent of many existing or potential saltmarshes in the UK is constrained by sea defences which have been constructed to protect ‘reclaimed’ coastal land, initially for grazing, but often now used for arable agriculture. Restoration of saltmarshes involves ‘managed realignment’ of the coastline by removing or breaching these defences to allow tidal flooding of the land (Paranjoti, 2016; JNCC, 2008). In some cases, a new defence is built further inland, which the restored saltmarsh protects against erosion (Boorman and Hazelden, 2017).

In the literature, saltmarsh restoration most commonly refers to the re-establishment of saltmarsh through managed coastal realignment in areas where it was previously. That is, therefore, the main focus here.

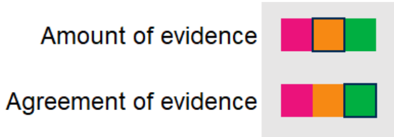
4.3.2 Readiness for implementation



Techniques for saltmarsh restoration through managed realignment are well established; there are 88 managed realignment sites in Europe, with 41 in the UK (Boorman and Hazelden, 2017). Details of many realignment projects are available from ABPmer (ABPmer, 2020).

Implementing managed realignment by the Environment Agency typically involves collaborating with other partners involved in managing coastal assets and with landowners on relevant coastal agricultural land.

4.3.3 Speed and scale



The estimated rates of carbon sequestration by saltmarshes are considerably higher than for many land systems. Therefore, despite their limited extent, saltmarshes could represent significant carbon removal potential (Mcleod and others, 2011).

Studies investigating the impacts of saltmarsh restoration through managed realignment on carbon accumulation rates found that carbon removal by restored systems was initially very rapid – on average 3.8 tCO_2/ha/year† for the first 20 years – and then slowed to a fairly constant rate of around 2.4 tCO_2/ha/year† thereafter (Burden and others, 2019). As a specific example, the estimated rate of carbon sequestration in a saltmarsh in Essex, UK restored by managed realignment was found to be around 3.4 tCO_2e/ha/year (Burden and others, 2013). Recent evidence from the Steart Marshes restoration in the Severn Estuary, however, indicates much higher carbon burial rates of 25 – 32 tC/ha/year (91.8 – 117.5 tCO_2e/ha/year†) (Wildfowl and Wetland Trust, 2020). As this evidence base evolves, the overall GHG removal estimate presented at the top of this chapter (2-8 tCO_2e/ha/year) may prove to be conservative, and require updating.

For comparison, the average estimated sequestration potential of natural saltmarshes in the UK range from 2.8 to 6.9 tCO_2/ha/year (Ouyang and Lee, 2014). Typical figures in another study are given to be around 2.4 to 8.0 tCO_2/ha/year (Beaumont and others, 2014). A recent review of carbon removal potential of saltmarsh in the Solent assumed a very wide range of 0.7 to 62.8 tCO_2/ha/year – with a median of 5.1 tCO_2/ha/year and a mean of 7.7 tCO_2/ha/year (Watson and others, 2020). The relative carbon removal potential of a saltmarsh varies depending on factors such as the dominant vegetation community and the tidal range they are subjected to (Ouyang and Lee, 2014).

In one study comparing saltmarsh restored through realignment to a natural saltmarsh, the carbon stocks held in restored saltmarshes were found to reach approximately the same levels of carbon storage as those found in a natural saltmarsh after 100 years; 74 tC/ha in a restored saltmarsh compared to 69 tC/ha in a natural saltmarsh (Burden and others, 2019).

There is less research quantifying the impacts of saltmarsh restoration on the emissions of other GHGs like methane and nitrous oxides, and there remains some uncertainty around measuring and analysing these balances. However, there is some evidence that salinity is an important determinant of methane emissions, and that restoration of tidal inundation of wetlands including saltmarshes, to restore salinity, can significantly reduce methane emissions (Kroeger and others, 2017).

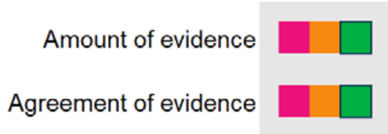
Nitrous oxide emissions vary considerably depending on factors such as soil texture, structure and compaction. Both natural saltmarsh and those restored by coastal realignment can emit small amounts of methane and nitrous oxide, but generally only enough to offset carbon removals to a limited extent, thereby still resulting in net removals of greenhouse gases (Watts and others, 2018).

There are currently no existing systems of accrediting carbon balances for saltmarshes equivalent to the Woodland Carbon Code and Peatland Code.

Theoretical maximum abatement potential for this measure in the UK

The theoretical maximum abatement potential for saltmarsh restoration in the UK has been categorised as low. To obtain this category, the approximate per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. For this measure, the theoretical available area was derived from the current area of saltmarsh around the UK and an estimate of how much of this was lost since 1945. This maximum theoretical area stands below 10,000 hectares (Garrard and Beaumont, 2014a).

4.3.4 Co-benefits



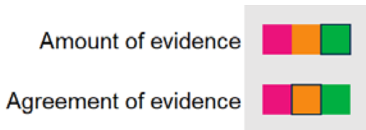
In addition to significant carbon sequestration potential, saltmarshes also provide protection against coastal erosion and flooding by dissipating the energy of incoming waves (Environment Agency, 2011).

Restoring saltmarshes through managed realignment also removes the need for coastal flood defence structures and, therefore, removes construction and ongoing maintenance costs (Paranjoti, 2016).

Saltmarshes have also been shown to filter out nutrients and pollutants, preventing them from entering the sea, which benefits seagrass communities and other marine habitats (Boorman and Hazelden, 2017).

Saltmarsh habitats have significant value for biodiversity. In the UK, several rare and protected bird species, such as the British redshank, breed on saltmarshes. Upper saltmarshes also provide habitat for significant populations of small mammals and a diversity of invertebrates. Saltmarsh plant communities are unique, and 2 types of saltmarsh habitat are listed for protection in Annex I of the EU Habitats Directive. Saltmarshes and related coastal habitats can also act as fish nurseries and offer potential recreational opportunities, including nature watching.

4.3.5 Confidence in the science



There is increasing interest in the carbon sequestration potential of marine habitats, referred to as ‘blue carbon’, and therefore interest in research to test their potential. The evidence base for saltmarshes is better than for some other coastal and marine habitats, although there is still a wide range of values found for sequestration potential and, as yet, no verified mechanism to quantify these (for example, equivalent to the Woodland Carbon Code).

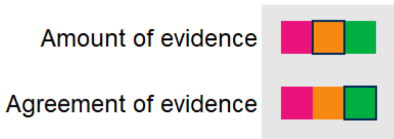
There is generally agreement that saltmarsh restoration can provide a sustained CO_2 sink, although there is significant variation in the rate of carbon sequestration by saltmarshes in different conditions and under different management regimes (for example, see Ford and others, 2012).

There is also some evidence that restored saltmarshes are not the same as natural systems in terms of dynamics such as vegetation composition, amounts of organic carbon available in sediments and populations of benthic fauna such as crabs (Staszak and Armitage, 2013; Rezek and others, 2017). It will take several decades to measure how natural and restored systems compare in terms of carbon sequestration potential.

Sequestration rates for natural and restored saltmarshes have often been estimated from carbon content of existing saltmarsh sediments combined with observed rates of new sedimentation, rather than more direct quantification (Beaumont and others, 2014). Therefore, there is still some uncertainty around the potential for increased removals through saltmarsh restoration in the UK (Burden and others, 2019).

There are fewer studies available which quantify the carbon sequestration impacts of rehabilitating degraded existing saltmarsh habitats, although there is evidence that draining land and overstocking grazing animals can result in rapid releases of GHGs (Johnson and others, 2016).

4.3.6 Measuring impact



In saltmarshes the majority of carbon will be stored in sediment rather than plant biomass (Johnson and others, 2016).

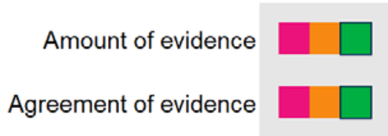
Methods for measuring carbon fluxes in saltmarshes are relatively well developed. For example, there are methods for directly measuring soil carbon stock and assessing changes in this stock. Gas fluxes over the saltmarsh can also be measured directly, for example, using gas sampling chambers (Burden and others, 2013). These measures can by multiplied by the area of the habitat to calculate the overall stock and fluxes (National Academies of Sciences, Engineering, and Medicine, 2017).

Methods for measuring, assessing and analysing carbon fluxes from coastal habitats including saltmarshes are detailed in a manual published by the Blue Carbon Initiative (Howard and others, 2014).

Accurate measurements of other greenhouse gases like methane and nitrous oxide are less well developed. Salinity is often used as a proxy for methane emissions due to the challenges of directly measuring emissions (lower salinity appears to be linked with higher methane emissions, although the relationship is not reliably linear) (National Academies of Sciences, Engineering, and Medicine, 2017). Nitrous oxide is difficult to quantify as it varies considerably over space due to a number of interacting factors, including soil structure and compaction. Where nitrates are present in the soil, inundation could increase nitrous oxide production. However, this should be counteracted where vegetation growth increases under restoration through assimilation of nitrogen within the plant biomass. Nevertheless, potential

N_2O release during transitions from one vegetation community to another, for example, from grassland to saltmarsh species, should be considered.

4.3.7 Risks and barriers



Managed coastal realignment is already part of the Environment Agency's Regional Habitat Compensation Programme (RHCP). So, while saltmarsh restoration is complementary to ongoing work and has carbon benefits, there may be challenges around additionality (Eastern Solent Coastal Partnership, 2020).

Risks and barriers to managed realignment can include negative perceptions of the approach among the public and local stakeholders, including landowners, that will be directly affected by the flooding of coastal land. The decision to flood otherwise productive agricultural land could potentially be viewed negatively by the wider public.

Since 2013, saltmarshes can be included in national GHG accounting and national reporting from all countries to the United Nations Framework Convention on Climate Change (UNFCCC) (Johnson and others, 2016), which makes it likely that funding and focus on their restoration will increase.

There are uncertainties about the fate of the carbon stored in saltmarsh sediments if they revert to open water (for example, due to sea level rise) or if they are eroded. More research is needed to develop hydro-biogeochemical models which could help understand the long-term fate of carbon in these systems (National Academies of Sciences, Engineering, and Medicine, 2017).

Restoration of the biogeochemical functioning of saltmarshes, such as the cycling of nutrients and accumulation of carbon, can be slow, and can take around 100 years. Until this point, the soil carbon stock and below-ground biomass may remain more similar to the un-restored site than to the conditions found in a natural saltmarsh, with implications for the carbon removal and storage potential (Burden and others, 2013).

There are long-term plans to realign 10% of the coastline in England by 2030, which will require a five-fold increase in the current rate (Committee on Climate Change, 2013). These long-term plans will help restore salt marshes and this will have a carbon sequestration benefit. However, in order to count this sequestration benefit and say it has contributed to an organisation's offsetting targets is challenging, careful consideration will be needed to ensure that it is truly additional and not an activity which would have happened anyway.

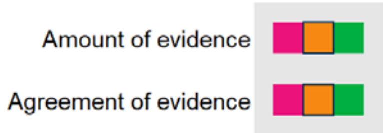
Saltmarshes face several threats. Climate change and related sea level rise cause 'coastal squeeze'. This is where saltmarshes are forced to migrate inland due to rising tide levels but constrained by erosion defences. Some projections are for potential losses of saltmarsh from sea level rise to be 4.5% over 20 years (Beaumont and others, 2014). However, there is some evidence that, where saltmarshes are not constrained by hard boundaries and are able to extend laterally, rising sea levels can, in fact, lead to greater accumulation of saltmarsh sediments and higher concentrations of sediment carbon than if there was no sea level rise (Rogers and others, 2019).

Coastal realignment by removing erosion defences helps to counteract this ‘coastal squeeze’ and loss of saltmarshes by providing scope for the habitat to extend inland. However, the effectiveness of such measures and the longevity of any restored saltmarsh depends on the rate and height of projected sea level rise. In addition, higher water levels, combined with increasing frequency and severity of storms, threatens to damage saltmarshes and interrupt sediment flows. This has implications for the permanence of carbon sequestration in saltmarshes.

Other competing demands for uses of coastal land and estuary areas, including economic activities such as port developments, pose potential barriers to saltmarsh restoration efforts. The recovery of saltmarsh is also hindered by dredging, which is done to improve shipping access but reduces the natural sediment supply to the saltmarsh systems (Committee on Climate Change, 2013).

There has been relatively little research to date into the balances of other greenhouse gases in restored saltmarshes. Evidence suggests emissions of methane can be high locally, particularly in grazed systems, coinciding with spots of high soil moisture and the occurrence of soft rush (Juncus effusus), which has been shown to act as a conduit of methane from the soil to the atmosphere. Evidence also suggests that nitrous oxide may also be important, but is largely unstudied (Beaumont and others 2014).

4.3.8 Costs



Estimated costs of coastal managed realignment to (re)create saltmarsh areas were estimated to be around £10 to £15,000 per hectare (2006 prices) but can vary significantly (Environment Agency, 2015). However, managed realignment case studies found in Burgess-Gamble and others (2017) show costs ranging from £12,000 per hectare to approx. £20,000.

Managed realignment is significantly cheaper than other methods of marsh creation, such as sediment recharge, which involves the redistribution of large quantities of sediment (Boorman and Hazelden, 2017)[14].

Managed realignment will entail land purchase costs which can represent a high proportion of the total cost of restoration (80 to 85%) (Environment Agency, 2015). The price of land also varies according to geography.

[14] Opportunities may exist to use materials created by other projects, for example, the managed realignment at Wallasea Island used inert sediment created by the Crossrail link tunneling (Wright and others, 2010).

4.4 Seagrass restoration

Table 4-5 Summary results for 'Seagrass restoration'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Removal	Uncertain	1.6	Low[15]	Not ready	Not ready	>10	Uncertain	Low	Low

4.4.1 Approach overview

This approach focuses on the restoration of seagrass habitats, by planting and seeding seagrass in areas where it was previously found around the UK coastline.

Seagrasses are marine flowering plants which form large meadows in shallow coastal waters. In the UK, seagrass meadows are dominated by one species, eelgrass (Zostera marina). These plants take up carbon during photosynthesis. When the plants die, the carbon in the dead plant material is trapped in seabed sediments where it can remain buried under relatively anaerobic conditions for centuries (Twigg, 2017). Globally, the seagrass carbon pool has been estimated to lie between 4.2 and 8.4 GtC (Fourqurean and others, 2012). The carbon stock in the UK's seagrass meadows is one of the largest in Europe (Green and others, 2018).

However, seagrasses are threatened by human activity, including habitat loss from coastal development, eutrophication and climate change. Seagrass meadows in the UK have been described as being in a "perilous state" and are declining at unprecedented rates (Jones and Unsworth, 2016). Their erosion has accelerated oxidation and remineralisation of sediment organic carbon.

Seagrass restoration has been shown to have considerable potential to transfer organic carbon back to the sediment in the United States (Oreska and others, 2020). Future research in the UK context could confirm that this is also the case for the UK's seagrasses. The methods of restoration for small-scale projects are relatively well understood. They involve re-establishing vegetation on the sea bed using natural or purposeful dispersal of seeds, transplantation of seedlings grown in aquaria or of mature plants taken from healthy

[15] The categories for national abatement potential correspond to the following ranges: 'Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

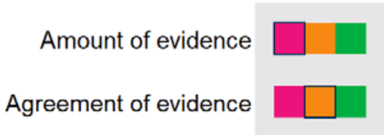
beds. Some advocate re-establishing suitable conditions in study areas (for example, by improving water quality) assuming that seagrass will naturally recolonise them (IPCC, 2013), although this is a much slower process. Although these methods are relatively well understood, they can be costly and further research is needed on efficiently scaling up restoration methods in the UK before restoration can be increased substantially.

Already underway for conservation purposes, restoration could be a valuable avenue for offsetting. However, the quantity of carbon stored varies hugely between and within meadows and there are multiple interacting factors in this variation, including species present, depth or temperature. This, as well as difficulties associated with quantifying changes in carbon stocks resulting from restoration, makes it difficult to estimate the carbon removal potential of seagrass restoration.

Despite recent excitement about the carbon removal potential of seagrasses, it remains very uncertain and rests on assumptions which are hard to verify. However, the potential scale of restoration projects in the UK means that such projects, if they were shown to sequester carbon, could have high returns. The Environment Agency could play a key role in promoting and facilitating further research on the carbon benefits of restoration and on its potential scale.

In 2015, the Verified Carbon Standard Program (VCS) published the first seagrass offset-credit accounting framework (VM0033 - Methodology for Tidal Wetland and Seagrass Restoration). This has been used by countries wishing to incorporate seagrass systems into their national GHG inventories, but individual projects have not claimed credits yet.

4.4.2 Readiness for implementation



Seagrass restoration may help transfer organic carbon back to marine sediments. Several studies have measured organic carbon stock in seagrass sediment.

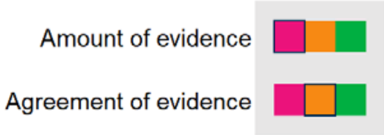
However, identifying the net greenhouse gas offset benefit of seagrass restoration means isolating the organic carbon sequestered by seagrass specifically and understanding 'community metabolism', for example, the presence of seagrass might increase emissions of different greenhouse gases such as methane or nitrous oxide. There is a lack of representative data on these fluxes (Macreadie and others, 2019a).

The first study to empirically calculate the net carbon benefit of restoration is very recent (Oreska and others, 2020) and is based in the North American context. Overall, therefore, the carbon offset potential of seagrass restoration is not yet widely understood nor quantified. Currently, there are also no systems of carbon balances and credits in the UK for seagrasses equivalent to the Woodland Carbon Code and Peatland Code.

Despite this, one seagrass restoration project, the Seagrass Ocean Rescue project, is already underway in Wales. Run by Sky Ocean Rescue, WWF, Cardiff University, Swansea University and the Pembrokeshire Coastal forum, it is restoring seagrass in a small experimental 2-hectare area. If the project works and additional funding is found, the aim is to scale the approach to other areas of the UK, including the Stour, Orwell and Humber estuaries.

This suggests partnerships with NGOs and universities may be key to developing the groundwork on which future seagrass restoration projects might develop in the UK.

4.4.3 Speed and scale



The IPCC's national GHG inventory sequestration rate for seagrass systems is 0.43 tC/ha/year (equivalent to around 1.6 tCO2/ha/year†) (IPCC, 2013). However, this sequestration rate was established by measuring the carbon stock in the sediments of a meadow populated by one specific species (Posidonia oceanica) in the Mediterranean. This species is known to have a high content of organic carbon. Moreover, this figure is based on a study which only looked at sediment carbon but did not consider GHGs emitted by the seagrass bed nor biomass. Therefore, we need to be cautious if using this as a general sequestration rate.

Only one study has empirically quantified the net GHG removal from the atmosphere deriving from a restoration project (Oreska and others, 2020). Researchers studying an eelgrass meadow in Virginia (USA) used the most reliable method of estimating carbon removal: measuring sediment organic carbon stock repeatedly over time. They found that the meadow sequestered 0.42 tCO_2e/ha/year.

Beyond these averages, it is important to remember that the carbon removal potential of seagrass meadows is thought to vary hugely and runs the risk of being overestimated if based solely on stock data.

Storage and sequestration rates depend on the age of the restored meadow, and it can take a decade or more for the plant biomass and sediment carbon sequestration rates to be equivalent to natural meadows (Oreska and others, 2020). It is not yet clear whether seagrass carbon benefits continue to accumulate indefinitely (Oreska and others, 2020). It is possible that, as with other biomass, carbon sequestration will fall to zero eventually.

Moreover, although leakage risks seem to be minimal, there are risks to permanence. For example, one study found that a marine heatwave on the Australian coast considerably damaged a large seagrass meadow and led to large losses in its carbon stock. (Arias-Ortiz and others, 2018). Other studies found that the carbon sequestration potential of seagrass is vulnerable to exposure to oxygen which increases microbial respiration (a process where microbes transform oxygen into CO_2)(Macreadie and others, 2019b).

The storage and sequestration potential of seagrass habitats is highly variable, with one survey of 17 Australian seagrass habitats in different climate zones and with different species measuring an 18-fold difference in carbon stored (Lavery and others, 2013).

This variability is due to plant density, water depth, nutrient availability, meadow size, wave exposure, and substrate type. There can be up to a tenfold difference in carbon storage potential between different species of seagrasses (Bedulli and others, 2020). Sediment characteristics such as dry bulk density, percentage of mud, pore water acidity and concentration of nutrients also affect carbon storage, as has recently been shown in UK meadows (Lima and others, 2019).

Beyond this, many factors affecting the sequestration potential of seagrass restoration, such as the drivers of the variability in methane and N_2O fluxes, are not yet understood (Oreska and others, 2020).

Moreover, complex interactions between environmental factors such as temperature or hydrology further complicate estimates. For example, different studies have yielded contradictory results with respect to the influence of nutrient availability (Macreadie and others, 2019a).

All of these sources of variability make it difficult to estimate carbon stocks and sequestration potential of specific sites.

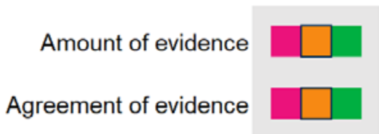
Given the high rates of loss of seagrass meadows in the UK, the potential for restoration is huge. Indeed, although assessments of the overall environmental conditions of seagrass meadows in the UK are sparse, the sequestration potential of seagrasses in the UK has been estimated at between 9,615 and 19,231 tCO_2/year[†] (Garrard and Beaumont, 2014b). However, future climate change scenarios, including rising temperatures and storms could limit the potential of seagrass restoration. At high temperatures, eelgrass beds have been found to stop growing and die (Winters and others, 2011), which could, with anticipated warming, affect restored beds in the south of the country. Similarly, seagrasses are sensitive to nutrient input (Jones and Unsworth, 2016), and coastal population increases could also limit the scope of restoration.

The Environment Agency could have a role in contributing to our knowledge about carbon removal in terms of measurement methods, speed, total potential, timescales and controlling factors. It could also help to identify, on a large scale, which British coasts should be targets for restoration.

Theoretical maximum abatement potential for this measure in the UK

The theoretical maximum abatement potential for seagrass restoration in the UK has been categorised as low. To obtain this category, the approximate per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. For this measure, the theoretical available area was derived from the current area of seagrass around the UK and an estimate of how much of this was lost over the last 35 years. This maximum theoretical area stands between 10,000 and 30,000 hectares (Green and others, 2018).

4.4.4 Co-benefits



Restoring seagrass systems can provide many benefits above and beyond carbon storage and sequestration.

Seagrass is an important nursery and foraging habitat for specific fish, shellfish and wildfowl species. As foundational species, seagrasses enhance ecosystem biodiversity and are home to many vertebrate and invertebrate species, including intrinsically valuable species like the seahorse. Their value for biodiversity is recognised by their importance in the UK Biodiversity Action Plan. They are also thought to stabilise sediments and shorelines as well as protect from erosion.

Economically, they provide fisheries support, for example, as nursing grounds for common fishing species such as cod or herring.

4.4.5 Confidence in the science

Amount of evidence

Agreement of evidence

There are multiple studies quantifying the carbon stocks of seagrass habitats. Although useful, these are insufficient for understanding their carbon sequestration potential and, therefore, for estimating the blue carbon potential of specific meadows and of seagrasses more generally.

Gross organic carbon stocks fail to account for background organic carbon sequestration that would occur without seagrasses, nor do they account for increases in GHG fluxes resulting from meadow restoration (Oreska and others, 2020). There is a data gap concerning the effect of restoration on methane and N_2O fluxes. More research could show that increases in seagrass population lead to decreases in N_2O emissions as nitrogen is assimilated into plant tissue.

Organic storage of carbon in seagrass ecosystems is not directly linked with the removal of atmospheric CO_2 because water separates the atmosphere from the bottom of the sea (Macreadie and others, 2019a). Science gaps concerning the inorganic and organic biogeochemical processes occurring in this water column complicate our understanding of the potential removal of atmospheric carbon through seagrass restoration. Research in this respect is complicated by difficulties in accessing samples on a wide scale.

Moreover, the factors influencing the variability in carbon sequestration potential of seagrass systems are complex and poorly understood. All of these factors mean that reliable assessment methods are still being developed.

All of these uncertainties suggest research remains a priority and something which the Environment Agency could contribute to.

4.4.6 Measuring impacts

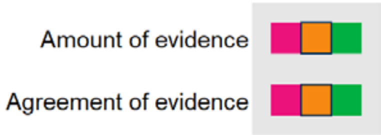
Amount of evidence

Agreement of evidence

The offset-credit methodology VM0033 advises measuring the sediment organic carbon stock repeatedly over time to quantify its increases. This approach is deemed preferable to measuring the organic carbon stock at a single location or estimating the accumulation of organic carbon from burial rates. These approaches risk overestimating the net CO_2 removal from the atmosphere because of uncertainties with dating techniques and confusions in the measurement of burial fluxes (Johannessen and Macdonald, 2016). They also include carbon fixed outside of the aquatic system under review (allochthonous carbon) which must be excluded from offset methodologies.

It can be hard to quantify how much CO2 seagrass can remove because seagrass beds are located in subtidal areas which can make collecting samples challenging (Twigg, 2017). If this is the case, projects seeking credits under the Verified Carbon Standard can cite default values of CO_2 emission reductions. These values are meant to conservatively underestimate project benefits. There is also growing potential to use remote sensing methods for seagrass research.

4.4.7 Risks and barriers



Seagrass restoration is incompatible with dredging activities and extensive coastal development. It is also incompatible with poor water quality, manifested through turbidity, eutrophication or low light. Indeed, one of the forms of human pressures on the marine environment is nutrient loading, for example, with nitrogen run-off from farming (Jones and Unsworth, 2016). Excessive nutrients are incompatible with seagrass restoration, which could have indirect social impacts in terms of restricting specific polluting practices.

Because they occupy intertidal and shallow water environments, future carbon sequestration of seagrasses will be influenced by sea level rise. Sea level rise can result in sequestration gains by increasing the area available for restoration and space available to accumulate sediment. Conversely, it can lead to remineralisation of stored organic matter, intense storms and marine heatwaves, all of which threaten carbon sequestration and carbon storage.

However, there is a lack of local scale descriptors of potential changes in exposure of seagrass systems. For example, storm associated waves are central to determining the persistence and growth of seagrass habitats, but local assessments of these waves are often unavailable. This makes it hard to predict the effects of climate change on specific restored habitats.

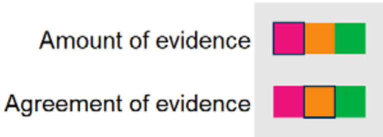
Climate change may also indirectly affect the persistence of seagrasses. First, seagrass systems are vulnerable to the responses of adjacent ecosystems to climate change. Second, the interaction between climate change and direct human disturbances may exacerbate threats to seagrasses. For example, one study found that deterioration in water quality may exacerbate the impacts of sea level rise on seagrasses (Saunders and others, 2013).

Conversely, one study suggests that ocean acidification (the fall in ocean pH resulting from absorbed atmospheric CO_2 and deposition of other gases (Hagens and others, 2014)) could increase the carbon storage and sequestration potential of seagrass meadows because it would increase above and below ground biomass (Garrard and Beaumont, 2014b). This potential storage capacity depends on healthy seagrass meadows, which is another reason for restoration.

Recently, Natural England granted £2.5 million of funding to help protect and restore seagrass meadows. The project (ReMEDIES) is partly funded by the EU.

As explained above, there is an offset accounting framework for seagrass restoration (VM0033) which has been used by countries wishing to incorporate seagrass systems into their national GHG inventories but not by individual projects as of yet.

4.4.8 Costs



The 7km^2 South Bay restoration project in Virginia (Oreska and others, 2020) cost US$800,000. In this case, monetised carbon offsets (sequestration from restoration costed using VM003) could have helped pay for ~10% of the restoration. However, this project reflects costs in the United States only. Moreover, the cost of restoration of this project was very low (US$1,200/ha) compared to other projects, whose costs vary hugely between US$1,900 and US$4,000,000/ha (Oreska and others, 2020). Average costs are one to two times higher than the Virginia project.

There is very little published data on the costs of restoration in the UK where seagrass restoration efforts are just beginning. The £2.5 million of the ReMEDIES project will, in part, serve to restore 4 hectares of lost seagrass meadows in Plymouth. However, the grant is also being used for education and research purposes, and these numbers cannot be translated to a per-hectare basis.

Given the absence of data, it is not yet possible to give a general figure for the cost of widespread restoration in the UK. As things stand, indications are that restoration costs, and therefore associated carbon costs per tonne, are very high. As a general rule, however, costs depend on the prior state of the meadow, the location of the restoration project (exposed open coast versus sheltered estuaries), the methods used and how proactive and equipment-intensive they are, as well as the size of the project. If seagrass restoration is scaled up, costs are likely to fall, although they will continue to vary substantially.

4.5 Kelp restoration

In this section on kelp we have not carried out a RAG rating for each of the different criteria. This is because this approach was not reviewed as extensively as the other approaches covered in this chapter, partly because less science was available and also because it was included at a later stage in the review.

Table 4-6 Summary results for kelp restoration

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Removal	n/a	2.15	n/a	Not ready	Not ready	Unclear	Unclear	Low	Low

4.5.1 Approach overview

'Kelp' refers to several species of large brown seaweed. These marine macro-algae differ from seagrasses, which are marine angiosperms (flowering plants). The structure of kelp looks similar to that of a plant, with a branching holdfast, a thin hard stalk and fronds spreading out like leaves. In suitable conditions, kelp form dense underwater 'forests', through which they are one of the dominant primary producers of coastal zones.

Because of their productivity, kelp have an important role in the planet's carbon cycle. Despite this, they were previously considered to play a secondary role in coastal carbon storage because macroalgal-derived matter was assumed to decompose too quickly for burial. This is because kelp mainly grows on rocks where in situ burial of organic carbon is impossible, or because we do not have consistent reliable estimates of the amount of carbon released by macroalgae (Pessarrodona and others, 2018). However, recent studies are increasingly suggesting that macro-algae derived carbon can be transported to depositional areas and to the deep sea and buried. There is therefore a rapidly growing interest in kelp as 'blue carbon' (Krause-Jensen and Duarte, 2016).

The UK has an extensive kelp population, especially along the wave-exposed south, west and north coasts. Scotland alone has been estimated to have 290,000 hectares of kelp beds off its shores (Burrows and others, 2014). This population is more diverse than that of any other country in Europe, with 7 different species co-existing (Smale and others, 2013). However, the UK kelp population is also markedly under studied, inshore mapping data is lacking and little is known of the effects of direct human pressures on kelp populations in the UK (Burrows and others, 2014).

What we do know is that the UK's kelp population has been declining in subtidal and intertidal areas. This is partly due to trawling, which tears kelp from the sea floor, preventing natural regeneration and removes top of the chain predators (such as lobsters and fish), and overfishing, which leads to overgrazing by kelp predator populations (such as sea urchins) (Williams and Davies, 2019; Smale and others, 2013). It also results from pollution and urbanisation which can lead to excess nutrients and turbidity (Burrows and others, 2014). Lastly, ocean warming is a major threat to kelp forests, which are sensitive to temperature, acidification and extreme storms.

In light of this, kelp restoration has been receiving growing attention. To restore kelp beds one of the first activities needed involves identifying the reasons for their decline and then removing any destructive activities. This can involve reducing trawling, as has been discussed by the Sussex Inshore Fisheries and Conservation Authority (Williams and Davies, 2019). Minimising disturbance to kelp can also involve management of coastal nutrient supply (Queirós and others, 2019). This 'assisted recovery' can facilitate kelp recruitment and re-establishment, with results of varying success (Layton and others, 2020). Secondly, 'active restoration' consists of transplanting adult and juvenile kelp from donor sites or from a lab to a restoration site. These methods have not been tried in Britain yet and are still experimental.

4.5.2 Readiness for implementation

Kelp restoration is ready to be implemented experimentally and on a small scale in the UK. However, it is not ready to be scaled. This is, in part, because appropriate and reliable methods of restoration in the UK need to be identified and experimented upon to assess their success. It is also because the carbon assimilation and transfer linked to healthy kelp beds is still poorly understood. Most notably, the proportion of the carbon fixed by growing kelp that is effectively removed from the atmosphere over decades or centuries and incorporated into longer term carbon stores is uncertain (Smale and others, 2013).

Therefore, kelp restoration is not a proven carbon removal approach yet. Further research and experimental restoration of kelp beds off the UK coast, in partnership with universities and local authorities, are needed before we can scale up restoration as a carbon removal method.

4.5.3 Speed and scale

Kelp forests have a large role in the carbon cycle of our planet. They have very high net primary production (NPP) rates, ranging around 1,000gC/m^2/yr in the Atlantic (Burrows and others, 2014).

However, most of the carbon taken up by this productivity is not sequestered in situ because kelp does not grow in habitats that are considered to accumulate large stocks of organic carbon. Rather, part of it is exported to depositional areas or to the deep ocean thanks to its ability to drift (Krause-Jensen and Duarte, 2016). Kelp is, therefore, a ‘carbon donor’ (Pessarrodona and others, 2018) or a source of ‘allochthonous carbon’, that is, carbon that comes from outside the system from which it is effectively removed.

The question then becomes how to quantitatively link net primary productivity and carbon sequestration. According to a recent global analysis (Krause-Jensen and Duarte, 2016), macroalgae export about 43% of their production as particulate organic carbon (POC) and dissolved organic carbon (DOC). Some, though not all, of this carbon can then reach depositional areas and be sequestered in sediments or reach the deep sea where it cannot exchange with the atmosphere.

The exact quantity which is effectively sequestered is uncertain. One study in the English Channel reported an average net sequestration of particulate organic carbon of sediments of 58.74 g C/m^2/yr (2.15 tCO_2/ha/yr) (Queirós and others, 2019). This same study estimated the net POC sequestered through kelp forests in the UK to be 4.70 Tg C/yr. These values must be treated cautiously and cannot be applied uniformly to all kelp habitats in the UK.

It can be assumed that restoration would increase sequestration to values approaching those above, but these assumptions remain theoretical. Like the sequestration rates themselves, the impact of restoration on sequestration is unclear.

4.5.4 Co-benefits

Kelp restoration has clear benefits for biodiversity. Kelp forests are complex and three-dimensional habitats supporting diverse communities of flora and fauna, and act as the foundation of coastal food webs by producing food for grazers, detritivores (animals that feed on dead organic material) and microbes. Restoration therefore contributes to marine biodiversity. In one restoration initiative in Tasmania, transplantation of adult kelp was shown to encourage the recruitment of invertebrate and fish which were ecologically and economically valuable (Layton and others, 2020).

The high biomass and fast growth rates of kelp also yield co-benefits in terms of coastal nutrient cycling and bioremediation (the degradation of pollutants by micro-organisms) (Smale and others, 2013).

Lastly, larger kelp forests provide coastal defence. Indeed, they contribute to buffering against strong waves and, in doing so, reduce coastal erosion. With predicted increases in more severe weather linked to climate change, restoring kelp forests is also therefore gaining in importance (Smale and others, 2013).

4.5.5 Confidence in the science

Kelp is only just emerging as a potential avenue for ‘blue carbon’. As such, it is early stages for the science on its carbon removal potential. There are very few in situ measurements estimating the fraction of carbon sequestered by kelp in the long term. Transport mechanisms and burial rates are also uncertain. Moreover, the evidence required to estimate the contribution of kelp has been distributed under different research fields and integration of these fields is needed (Krause-Jensen and Duarte, 2016).

All of this uncertainty concerns kelp itself. There is even more uncertainty concerning kelp restoration and its impacts on carbon sequestration.

4.5.6 Measuring impact

Studies investigating kelp and carbon use seabed samples obtained with corers, often sampling multiple sites because of variability. Markers of macroalgae such as stable carbon isotopes coupled with lipids, sterols and carotenoids can be used to determine the contribution of macroalgae to sediments (Krause-Jensen and Duarte, 2016).

One of the few studies quantifying how much of the carbon cycling through kelp forests is effectively sequestered analysed the samples by extracting environmental DNA (eDNA) and by using stable isotope mixing modelling (SIMM) to determine where carbon came from (Queirós and others, 2019).

4.5.7 Risks and barriers

Reducing the extent of trawling may have an impact on fleets that depend on it as a form of fishing. However, restoring kelp habitats may also see the return of populations of fish and crustaceans, for example, lobsters, which are valuable for fishermen.

Although it would help protect coasts from climate change, kelp restoration and associated carbon removal, is also threatened by climate change because kelp are cool water species. The structure, distribution and species of kelp forests in the Atlantic is already changing as a result of climate related stressors (Smale and others, 2013). Productivity will also be affected by climate change: kelp forests in warmer climates have been found to assimilate and donate less carbon (Pessarrodona and others, 2018).

Kelp restoration is not permitted by carbon reporting bodies as of yet, and further research is needed before standardised sequestration values can be agreed upon.

4.5.8 Costs

There are no estimates which can be provided currently on the cost of kelp restoration in the UK. For ‘assisted recovery’, costs will include the costs of foregoing damaging activities like trawling. For active restoration, one initiative in Australia estimated a cost of ~$570 per square metre of restored beds, covering a 4-person team, boat, scuba tank fills. This cost does not include the science necessary to decide on donor sites and sizes of restoration patches.

Chapter 5. Agricultural approaches

5.1 Agricultural soil management practices - arable land

Within this approach, we consider management techniques designed to increase carbon removal and storage in agricultural soils. We consider:

- arable land, used to grow crops
- pasture land used to graze animals

Table 5-1 Summary results for 'Agricultural soil management practices - arable land'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Removals and reductions	100-1000+	0.5-1	Moderate[16]	Ready	Not ready	<10	Med/long	Medium	Medium

5.1.1 Approach overview

The use of soil carbon practices on agricultural soils, often included as components of 'regenerative agriculture', 'conservation agriculture' and 'agroecology' (Lampkin and others, 2015), refers to management measures to increase carbon sequestration on working agricultural soils (as opposed to conversion of the land to semi-natural habitats). It has been receiving increasing interest over recent years for its potential to contribute to meeting climate change targets. For example, the 4 per 1,000 (4p1000) initiative introduced at the Paris Climate Conference (COP21) in 2015, estimates that increasing soil carbon storage by 4% a year would contribute significantly to achieving global climate change targets (4p1000, 2018), although there are important biological limits to consider such as adequate availability of nitrogen (van Groenigen and others, 2017).

Regenerative agriculture measures on arable land are intended to mimic more natural ecosystems which retain higher levels of carbon than cultivated systems, by:

[16] The categories for national abatement potential correspond to the following ranges: 'Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

- maintaining more continuous vegetation cover on the soil
- minimising soil disturbance
- increasing the amount and diversity of organic matter retained in the soil
- maximising nutrient and water use efficiency by plants

On arable land, key measures include (Paustian and others, 2020):

- reduced tillage or no tillage: methods of tillage or ploughing where disturbance to soil is minimised
- integration of crop residues to soil
- cover or catch crops
- intercropping
- crop rotations
- addition of manure or other organic matter

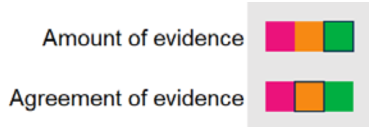
These measures increase soil carbon stocks by (1) increasing the extent to which organic matter from plant growth is returned to the soil, and (2) increasing the extent to which organic matter is retained in the soil. They also reduce carbon lost through soil respiration and erosion and through the reduced use of inputs like fertilisers and pesticides.

Carbon removal by arable land already under some sort of soil management in the UK, comprising inputs of fertiliser, manure and/or crop residues, was calculated as -0.17Mt CO_2-C in 2017 (Clilverd, 2019). This is relatively minor relative to the removals calculated using the same methodology for land converted to forest or remaining under forest, which were -4.9Mt CO_2-C. However, given that arable land without soil management measures were calculated to be sources of CO_2 emissions (Clilverd, 2019), using these methods could convert arable areas from sources of carbon to sinks.

The carbon sequestration potential of soil management will vary significantly depending on local factors, particularly the starting state of the soil. Additionally, the sequestration potential is not infinite. After management is introduced, rates of carbon accumulation may increase sharply over periods of around 20 to 50 years, but after this time this will plateau as soils reach a carbon saturation point (Lugato and others, 2014).

It should be noted that some of the most productive areas of arable land in the UK are on organic or peat soils that have been drained to support agricultural production and these are the source of significant greenhouse gas emissions. From a carbon perspective, these areas should be considered for restoration.

5.1.2 Readiness for implementation



The proposed measures are well-established agricultural practices and do not involve any novel techniques or technologies. They are, therefore, relatively straightforward to implement.

There is general consensus that the implementation of measures which reduce disturbance and increase vegetation cover on farmland, mimicking a more natural landscape, enhance carbon sequestration in agricultural land (Smith and others, 2008). However, there is some debate as to the scale of the impact of these measures and evidence that effects will vary depending on a range of environmental circumstances.

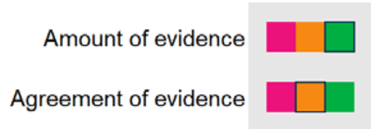
The potential for these practices to be scaled up to have significant climate change mitigation impacts has not yet been demonstrated in practice, and there are uncertainties over the compatibility with current farm business models. Adopting the measures could be supported by a shift to paying farmers for providing public goods, which is intended to be a feature of the UK Agricultural Bill. This will support landowners who implement measures aimed at climate change mitigation (Defra, 2019a; 2019b), and is a feature of emerging private sector markets for ecosystem services.

There may be ‘opportunity costs’ in the form of lower yields associated with implementing these sorts of farming practices. In the view of some researchers, this reduces the ‘efficiency’ of the land in terms of producing enough food to meet growing demand (Searchinger and others, 2018; Smith and others, 2008). Others argue that practices which increase soil organic matter may increase the long-term resilience of crop production, maintaining fertility and moderating soil moisture regimes.

Implementing these soil carbon measures is likely to require a degree of specialist farm advice/extension services.

There are live discussions about establishing a Soil Carbon Code, similar to the Woodland Carbon Code, but nothing has been established yet.

5.1.3 Speed and scale



Managing agricultural soils for enhanced carbon sequestration may involve implementing just one of the above practices, or a combination, with different associated carbon removal potentials. Some estimates are shown in (Table 5-2).

One estimate of the potential impact of no tillage compared to conventional management on carbon stocks finds a 16% increase in soil organic carbon (Ogle and others, 2005). However, there is conflicting evidence over whether reduced or no tillage consistently leads to increased soil carbon stocks (Smith and others, 2008). It appears to vary due to different climatic conditions and vegetation characteristics. A review of studies of reduced or no tillage in the UK found no significant impact on carbon sequestration compared to conventional management (Powlson and others, 2012). Specifically, carbon accumulation is found to be limited to the upper soil layers and can be relatively easily re-released due to the breakdown of organic matter by soil biota (Lampkin and others, 2019). There is also uncertainty about the net impact of reduced or no tillage for N_2O, with some studies indicating that reduced tillage may increase N_2O emissions due to increased soil moisture levels (Lampkin and others, 2019; Powlson and others, 2012). But, other evidence suggests it results in lower emissions compared to conventional tillage (Abdalla and others, 2019).

Cover crops have been found to significantly increase soil organic carbon (SOC) without significantly increasing emissions of N_2O, resulting in an overall reduction in GHG emissions of 2.06 ± 2.10 tCO_2e/ha/year compared to no cover crops (Abdalla and others, 2019).

The addition of manure, when imported from off-farm sources, is not considered to result in net carbon removals as it represents organic material moved from another system and, therefore, constitutes ‘leakage’.

Carbon accumulation in soils under soil carbon management regimes generally happens over decades (Ostle and others, 2009).

Due to the significant area covered by arable land in the UK, the total carbon sequestration and storage potential of agricultural soils could be considerable, but it will depend on how widely measures are implemented. Existing arable land is estimated to account for 198 $MtCO_2e$, or 12% of the UK’s total top-soil carbon stock, and the use of carbon sequestration practices could enhance this (Ostle and others, 2009). However, barriers to implementation, including costs of implementation and ongoing monitoring, and willingness of farmers, are estimated to limit greenhouse gas removal through agricultural management to less than 30% of the total biophysical potential (Smith and others, 2007).

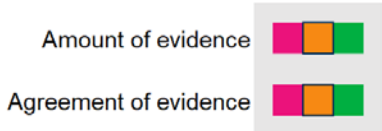
Table 5-2 GHG sequestration potential associated with soil carbon measures on arable land in a cool-moist climate (Smith and others, 2008)

Management measure	Mean estimated CO_2 sequestration (tCO_2/ha/year)	Mean estimated sequestration of all GHG (including methane and N_2O) (tCO_2e/ha/year)
Agronomy, including cover crops and crop rotations	0.88	0.98
Reduced tillage and retention of crop residues	0.51	0.53

Theoretical maximum abatement potential for the measure in the UK

The theoretical maximum abatement potential for implementing agricultural soil management practices on arable land in the UK has been categorised as moderate. To obtain this category, the approximate per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. Total land area was derived from the total potential cropping area in the UK in 2019, an area which stands between 5 and 10 million hectares (Defra and others, 2020).

5.1.4 Co-benefits



Some researchers state that the existing evidence indicates that the carbon removal potential of regenerative agricultural practices is limited, but when considered in combination with co-benefits for soil health and climate resilience, it is still beneficial (Powlson and others, 2020).

Increasing the organic carbon content of agricultural soils is almost always beneficial for decreasing soil erosion and improving conditions for the growth of crop roots. It increases the resilience of agricultural systems to adverse weather conditions, including the impacts of climate change (Powlson and others, 2020).

In addition, many measures for increasing soil carbon come with their own co-benefits. For example, cover crops can reduce water pollution and sedimentation from run-off, and may benefit yields in some cases (Snapp and others, 2005). Intercropping and rotations can reduce the need for pesticides and fertilisers, with benefits for both biodiversity and greenhouse gas emissions (the latter in particular associated with the production and application of artificial N fertilisers).

There is some evidence to suggest that in the long term, soil carbon practices could improve the stability of yields (Oldfield and others, 2019).

5.1.5 Confidence in the science

Amount of evidence

Agreement of evidence

The effect of soil management interventions for carbon sequestration has received increasing attention in recent years. It has been relatively well studied globally, although there is an apparent lack of studies applicable to UK soils and climatic conditions (Powlson and others, 2012).

The scale of the carbon sequestration potential of the management of agricultural soil is debated (Paustian and others, 2020; Powlson and others, 2012). Many of the experiments evaluating carbon removal potentials of agricultural soils are relatively short term and the values cited for removal potential tend to be those measured in the early years after a management change. As with most landscape carbon sequestration interventions, over time the rate of carbon accumulation declines and reaches a steady state (Powlson and others, 2020). The timescales for these changes in rates vary considerably depending on site characteristics but are in the order of decades (Mayer and others, 2018).

The carbon removals achieved using these soil management measures vary in different climate conditions, soil types and management systems. However, understanding the variables involved is relatively good, allowing location-appropriate regenerative agriculture practices to be designed.

5.1.6 Measuring impact

Amount of evidence

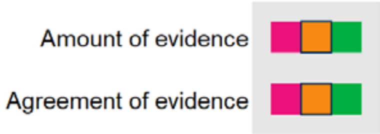
Agreement of evidence

As with the monitoring and measurement of the impact of most land management practices, a distinction needs to be made between methods used for collating evidence as part of a scientific study, and methods used for estimating performance in practice. The latter, especially at a farm level, will, for cost-effectiveness reasons, often rely on proxy measures, which are backed up by scientific studies.

Carbon removals through soil management can be directly quantified in several ways. One method is to quantify changes in soil organic carbon (SOC). This is commonly done using soil cores in which the SOC content is compared to a baseline measurement taken before the intervention was implemented. Cores are commonly taken to a depth of 15cm, but there is consensus that there is also a need for deeper cores in order to understand the impacts of above-ground management on soil carbon storage at deeper soil depths (Olson and Al-Kaisi, 2015).

Taking measurements at regular intervals allows the rate of sequestration to be calculated, although this will be resource intensive if done over a wide area. Other methods of quantifying emissions include using flux chambers which measure air composition and fluxes of GHGs over areas of land under different management (Carolan and Fornara, 2016). The fact that several soil carbon measures are used together makes attributing the effect of each difficult.

5.1.7 Risks and barriers



Many of the interventions to store carbon in agricultural soils are easily reversed. For example, most, if not all, of the carbon potentially stored in soils under no tillage can be rapidly released if the soil is ploughed. This is a risk in the UK as many farmers currently practice rotational tillage (Powlson and others, 2012). In addition, where reduced tillage is used, there is some indication that it can increase emissions of N_2O, partly counteracting carbon removals.

Research so far indicates that the impacts on farm productivity are mixed. Some studies have found that soil carbon practices can increase yields, while others have found they may reduce production. A reduction in yields creates the risk that more land elsewhere will be converted to agricultural production to compensate for the shortfall. This leads to ‘leakage’, in which the net effect on carbon uptake is minimal (Smith and others, 2019).

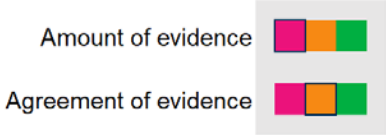
Effective carbon sequestration in agricultural soils relies on enough nitrogen being available and there have been some suggestions that this will entail additional input of fertilisers (van Groenigen and others, 2017). However, in many farming systems in the UK there is an excess of nitrogen which is often leached to waterways. Stabilising this nitrogen by using cover crops would be a co-benefit of soil carbon measures (Paustian and others, 2020).

There are some practical barriers to farmers adopting soil carbon measures. Cover crops have been unpopular with farmers in the UK due to issues such as difficulty with establishing crops and rigid rules about when and how cover crops should be grown to qualify for financial support under the Basic Payment Scheme (Storr and others, 2019).

The affordability of adopting regenerative agricultural practices is due to be supported by the upcoming UK Agricultural Bill, which will include a focus on paying farmers for providing ‘public goods’, including climate change mitigation (Defra, 2019a, 2019b). This means that support for agricultural soil management measures for carbon removal are complementary to prevailing policy conditions, but it may make it challenging to prove additionality for carbon offsetting. If any particular land management practices are mandated by law, this would rule out additionality.

There is some evidence to suggest that increasing temperatures as a result of climate change could enhance microbial activity in the soil. This could lead to a release of carbon dioxide, representing an increased risk to permanence of the carbon removal potential of these measures (Amundson and Biardeau, 2018).

5.1.8 Costs



Costs involved in this approach include a potential reduction in agricultural productivity and the related costs of financial support to farmers to implement and maintain the management options. This financial support will be an ongoing cost to support farmers to sustain the measures in the long term.

Quantitative cost estimates for arable soil carbon management options are sparse. One estimate from America shows that measures including cover crops and crop rotations, could cost as little as £15/ha, which equates to £15/tCO_2e/year. Tillage and residue management are estimated to cost around £4/ha or £7/tCO_2e/year. The cost per unit of CO_2e removal achieved will vary depending on climatic, soil and vegetation starting conditions (Smith and others, 2008).

Some estimated costs for implementing various management options for carbon mitigation on cropland in Scotland are shown in Table 5-3. Costs of implementing measures, when assessed on a purely economic basis, were found to outweigh the cost savings in many cases. However, these measures may have co-benefits, for example, for biodiversity, which are not captured in these values. No cost information is provided for reduced/no tillage separately, but it is included as one of the measures in 'conservation agriculture'. The costs presented in Table 5-3 are noticeably higher than the £15 per tCO_2e/year described above, demonstrating the uncertainty associated with cost estimates in these areas and the need to investigate this factor in detail.

Table 5-3 Costs of soil carbon management measures on cropland in Scotland (Lampkin and others, 2015)

	Cost	Cereal	General cropping	Cost/benefit ratio (Increase in farm income £000/ktCO_2e)[17]
Conservation agriculture (comprising crop rotations, use of catch-crops and reduced or no tillage)	**Increased costs (£/farm)**	4,013	8,035	-9.3
	Cost savings (£/farm)	3,335	3,928	
	Net increase in farm income (£/farm)	-678	-4,108	

[17] This value is across a range of farm types, not just arable, including mixed farms, dairy, lowland sheep/cattle and upland grazing on Less Favoured Areas.

	Cost	Cereal	General cropping	Cost/benefit ratio (Increase in farm income £000/ktCO$_2$e)[17]
Better organic nitrogen planning (improved use/application of manure)	**Increased costs (£/farm)**	559	668	-3.4
	Cost savings (£/farm)	305	366	
	Net increase in farm income (£/farm)	-253	-301	
Legumes in crop rotations	**Increased costs (£/farm)**	1,121	4,631	-21.9
	Cost savings (£/farm)	104	124	
	Net increase in farm income (£/farm)	-1,017	-4,507	

5.2 Agricultural soil management practices – pasture grassland

Table 5-4 Summary results for 'Agricultural soil management practices - pasture grasslands'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO$_2$e	**Per unit (tCO$_2$e/ha/year)**	**National abatement potential**	**Technology**	**Certification method**	**Years**		**Evidence volume**	**Evidence agreement**
Removals and reductions	10-1000+	0.2-4	Very high[18]	Ready	Not ready	<10	Med/long	Medium	Low

[18] The categories for national abatement potential correspond to the following ranges: 'Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

5.2.1 Approach overview

Agricultural soil carbon practices on pasture grassland are intended to mimic more natural ecosystems, which tend to retain higher levels of carbon than intensively managed agricultural systems. Here, we consider modified or improved grasslands; those actively used as agricultural pasture.

Grasslands account for around 70% of UK agricultural land (Defra, 2020), so the potential area over which carbon sequestration and storage could occur is considerable. Existing improved grasslands are estimated to account for 274 $MtCO_2e$ or 33% of the UK's total top-soil carbon stock, second only to peat bogs, which have much higher carbon sequestration and storage per hectare but cover a much smaller area than agricultural land. The total carbon stock of agricultural grasslands is also higher than that of arable land (Ostle and others, 2009).

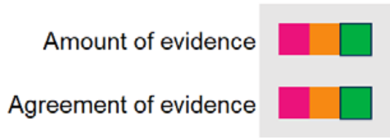
Various management measures are proposed to enhance the carbon sequestration and storage potential of agricultural grasslands. These commonly include (Conant and others, 2017; Smith and others, 2008; Garnett and others, 2017):

- improved grazing: changes to the timing and intensity of grazing to optimise carbon accumulation in soils. This can be done by altering the stocking density (the number of animals grazing a given area over a given time period), which impacts the level of disturbance to grasses and the soil structure, thereby impacting where and how carbon is stored within the pasture system. Changing the timing of grazing, and specifically using rotational grazing in which animals are grazed on the land at high intensity for short periods, can sequester carbon by allowing a controlled amount of trampling of soils. This is thought to increase the incorporation of carbon, as well as providing an input of carbon to the soil in the form of manure. However, in common with conventionally grazed systems, there will also be GHG emissions in the form of methane from the digestive processes from animals, as well as nitrous oxide from manure (Conant and others, 2017)
- fertilisation: for example, by adding manure, which constitutes a direct input of organic carbon as well as boosting plant growth and thereby the returns of plant litter to soil
- sowing legumes: adds carbon to the soil in the form of the legume root stock and by stimulating grass growth through increased availability of soil nitrogen
- sowing grasses: including deep-rooted varieties to enhance soil organic material

Any increase in carbon accumulation is not indefinite and will stabilise over time; it is estimated that after around 80 to 100 years of continuous management, the rate of change in soil organic carbon will be around zero (Yeluripati and others, 2019).

Consideration of this measure should take into account a backdrop of potentially significant shifts in strategic influences on livestock farming in the UK, including markets and subsidy regimes. In unpredictable ways, these may drive as much or more change in grazing management as interventions driven by carbon considerations, in effect shifting the baseline against which carbon impacts could legitimately be attributed.

5.2.2 Readiness for implementation

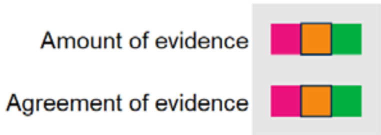


The measures are established agricultural practices and do not require special technologies or expertise to implement, meaning they are ready to implement. However, there are uncertainties about the practicalities of applying them at scale and whether this is compatible with profitability within the current agricultural system. They will also constitute a significant change in many farmers' practices, which will require specific expertise and inputs as well as a willingness to adopt new practices.

Implementation will involve collaborating with agricultural land managers. Reversibility of many of the measures is high and effective long-term implementation will require some enabling conditions. For example, longer-term contracting and tenure arrangements with land managers and guaranteed ongoing financial incentives will encourage longer term commitment. Creating an auditable standard for measures that have been verified to enhance carbon accumulation would also help encourage and track implementation of measures.

There is ongoing discussion about the potential for a grassland carbon code similar to the Woodland Carbon Code, but there is debate about whether this is achievable.

5.2.3 Speed and scale



Overall, reviews of peer-reviewed studies on the impact of the introduction of pasture management practices comprising the measures above find an average sequestration potential of around 0.47 to 0.50 tC/ha/year (around 1.8 tCO_2e/ha/year) (Conant and others, 2017; Garnett and others, 2017). However, one review found that values varied greatly from 0.18 to 3.81 tCO_2e/ha/year for different practices and across different studies (Garnett and others, 2017). Sequestration potential varies between measures. As measures are often implemented together, it is also often difficult to work out the impacts of single measures.

The increase in accumulation rates is most rapid over the first 5 to 10 years after the change in management. Soils are generally estimated to approach a new equilibrium after around 30 to 70 years of continued management (Garnett and others, 2017). This dynamic is similar to changes in carbon sequestration following conversion of land to grassland from cropland. In the short term, increases in soil carbon are rapid, but this begins to stabilise after around 70 years, after which there is limited further increase in soil carbon content (Garnett and others, 2017; Smith, 2014).

Table 5-5 Estimated increase in soil C stocks on grasslands under different management practices (Conant and others, 2017).

Measure	Estimated increase in soil C stocks (tC/ha/year)
Sowing legumes	0.66

Measure	Estimated increase in soil C stocks (tC/ha/year)
Fertilisation – organic	0.82
Fertilisation – inorganic	0.54
Improved grazing (for example, lower stocking rates, rotational or seasonal grazing)	0.28

Little evidence could be identified regarding the UK-wide potential for sequestration through pasture management practices.

Theoretical maximum abatement potential for the measure in the UK

The theoretical maximum abatement potential for implementation of agricultural soil management practices on pasture grasslands in the UK has been categorised as very high. To obtain this category, the approximate per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. For this measure, potential total land availability was derived from the total grassland area in the UK in 2019, an area which stands between 10 and 15 million hectares (Defra and others, 2020).

Sowing legumes

Sowing legumes stimulates carbon sequestration in soils both through the growth of the legume roots and by stimulating the growth of grasses. This is as a result of nitrogen fixed by the legume acting as a fertiliser to other plants when organic material from the legume breaks down in the soil. Carbon sequestration can be around 2 tCO_2e/ha/year. However, in some cases, particularly in waterlogged soils, legumes can contribute to the release of N_2O which outweighs some of the carbon sequestration effect (Garnett and others, 2017).

Some estimates suggest planting legumes has greater carbon sequestration potential than grazing regime management (Henderson and others, 2015).

Grazing

One study estimates that the soil carbon sequestration potential for pasture lands which undergo a change in grazing management aimed at improving carbon sequestration is 0.05 tCO_2e/ha/year (Henderson and others, 2015). However, the effects of changes in grazing management are very difficult to generalise due to the significant variation in grazing practices and the underlying starting conditions in terms of plant species, soils and (micro)climates (Smith and others, 2008).

Some studies show a decline in soil carbon concentrations following a change in management, although one review of evidence found an overall average 10% increase in soil C concentration following targeted changes in grazing management specifically aimed at improving C sequestration (Conant and others, 2017).

In general, good grazing management can help maintain soil carbon stocks and, in some contexts, can help sequester more carbon than would be achieved without any grazing. This is mostly in places where soils are degraded and below their carbon saturation capacity (Garnett and others, 2017).

The exact intensity of grazing – the stocking rate – that will favour soil carbon sequestration varies significantly between locations. However, the general rule for an appropriate stocking level is that there is just enough disturbance to stimulate plant growth but not so much that it is inhibited (Garnett and others, 2017).

It should be noted that any farming practice that involves grazing will include emissions from the livestock animals in the form of methane from enteric digestion and both methane and nitrous oxide from manure. The balance and attribution of these greenhouse gases relative to any benefits to soil carbon sequestration need to be accounted for when considering pasture management for the purposes of climate change and carbon removal targets.

Combined management

Pasture management often involves implementing several measures together. Advocates of the value of grazing systems for carbon sequestration and climate mitigation emphasise the need for holistic systems of management to realise carbon sequestration benefits.

One study estimates that pasture management that combines improved grazing management and fertilisation could result in carbon sequestration equivalent to an average of 0.81 tCO_2e/ha/year (Smith and others, 2008).

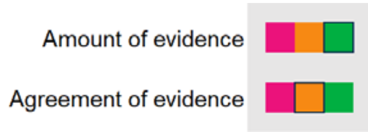
Another study comparing intensive, extensive or intermediate levels of management involving a combination of fertilisation, soil disturbance and inputs of plant matter to soils showed that intermediate levels of management led to the highest soil carbon content, including in deep soils at up to 1 metre below the surface (Ward and others, 2016).

Overall, pasture soil carbon practices must be carefully tailored to the specific agroecological, soil, vegetation and climatic conditions of a site, as well as its management history to ensure the implemented measures are optimised for carbon sequestration.

In general, carbon sequestration potential is much greater for soils in poorer condition which currently do not store significant amounts of carbon because the shortfall to be made up is much larger. Healthier soils which are already near carbon saturation point offer less potential sequestration.

Using these sorts of measures carbon removal takes several decades. Following a change in management, it takes up to 20 to 40 years for the rate of change in soil organic carbon (SOC) to stabilise (Yeluripati and others, 2019).

5.2.4 Co-benefits

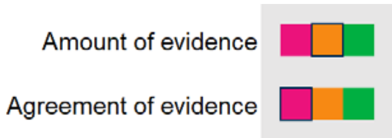


Soils that are managed to be rich in carbon generally have the additional benefit of improved soil fertility, with potential associated benefits for agricultural productivity.

Protecting soils for carbon sequestration purposes can produce benefits in the form of reduced erosion and loss of nutrient-rich top-soils, which, in turn, reduces the need for inputs of nutrients which have costs and embodied GHG emissions. It can also reduce soil compaction and waterlogging. Healthier soils generally support higher soil biodiversity, including species like earthworms which contribute to soil aeration and nutrient cycling, with benefits for plant growth (CPRE, 2018). The related benefits to land managers include more resilient crop yields as a result of enhanced soil fertility and moisture retention, and reduced nutrient loss through run-off.

On-farm benefits can lead to important downstream benefits such as water quality (nutrient and sedimentation) and water quantity (flood risk and water resource) management.

5.2.5 Confidence in the science



The range of estimates for the carbon sequestration potential of management on pasture lands is very large, reflecting considerable variation in the types and combinations of management practices and the impact of the location, agro-ecological conditions and management history of the study sites (Carolan and Fornara, 2016).

There are multiple factors to consider in managing pasture for carbon sequestration. The timing and intensity of grazing, for example, must be carefully planned; some level of trampling may help incorporate organic carbon into the soil, whereas too much trampling can lead to soil erosion and a decline in grass growth, leading to increased carbon losses (Garnett and others, 2017).

Grassland systems are incredibly diverse and appropriate management in one location may be different from what is appropriate or possible in another system. This means it is not possible to advise on a single ‘correct’ stocking rate. However, it is clear that overgrazing, which damages vegetation and soil, will lead to carbon losses and, in these cases, a reduction in stocking rate will often lead to increased carbon sequestration.

The evidence for the impacts of grazing management on soil carbon sequestration, particularly in relation to adjusted timing of grazing, is limited and often contradictory. Isolating the impacts of one measure on carbon sequestration in pasture management is often challenging and some argue that an overall, holistic farm management approach, rather than individual measures, will result in improved carbon sequestration. However, the tailored nature of such a holistic approach means that it is difficult to generalise impacts or management recommendations (Garnett and others, 2017; Nordborg, 2016).

Studies measuring soil carbon content in grasslands rarely measure soils below depths of 30cm, but some evidence suggests that management can influence soil carbon at depths of up to 1 metre. There is currently a gap in the evidence around the dynamics of this (Ward and others, 2016).

5.2.6 Measuring impact

Amount of evidence

Agreement of evidence

Soil carbon stocks and carbon sequestration rates can be quantified by directly measuring the soil organic composition. This is commonly done by extracting soil cores and analysing the concentration of carbon and nitrogen in the soil using commercially available equipment (Garnett and others, 2017). Soil cores are commonly taken to depths of between 0 to 30cm, but evidence indicates that soil carbon stocks at depths of up to 1 metre can be considerable and impacted by grassland management practices (Ward and others, 2016). Deeper soil cores should, therefore, be used wherever possible.

Such measurements need to be done over a large scale and a significant time period (decades) in order to pick up statistically significant changes in soil carbon.

Emissions of CO_2 and other greenhouse gases from an area of land can also be quantified directly using gas exchange chambers.

Levels recorded after the introduction of an intervention can be compared to baseline values recorded in the same way before the intervention. Regular measurements of carbon stocks can quantify the sequestration rate.

These methods are well established and commonly used in scientific research on environmental carbon fluxes.

Rapid and repeatable field sampling methods will need to be established to track changes in estimated carbon storage in response to different management practices. These methods and tools should ideally be suitable for use by land managers themselves.

Alternatively, quantifications can use models such as the Global Livestock Environmental Assessment Model, particularly when emissions are being quantified at a scale larger than the field level (FAO, 2020).

5.2.7 Risks and barriers

Amount of evidence

Agreement of evidence

The measures for grassland soil carbon removal have a high degree of reversibility. Improvements to carbon sequestration will only persist as long as the measures are in place. Most can be stopped or removed at short notice, with potentially immediate effects on carbon sequestration rates. For example, grazing animal stocking levels and applications of fertiliser can be altered from one day to the next, while the sowing of legumes or grasses can be reversed by ploughing, which is likely to re-release a significant amount of the C accumulated in the soil (Carolan and Fornara, 2016).

Successful long-term sequestration of carbon therefore relies on land managers committing to maintain the management measures in the long term. This may be challenging to achieve when a large proportion of farmers in the UK manage land on relatively short-term tenancy agreements, and, therefore, cannot necessarily guarantee long-term continuity of management systems.

There is also some uncertainty about the compatibility of soil carbon measures with the profitability of agriculture. A farming practice designed to optimise livestock productivity is not necessarily optimal for carbon sequestration and vice versa (Garnett and others, 2017). The economics of managing farmland for carbon sequestration may be helped by proposed payments to farmers for providing public goods under the upcoming UK Agricultural Bill, and by emerging private markets for carbon and other ecosystem functions. Although project additionality would have to be considered in the presence of these new funding sources. It should also be noted that the economic status quo for many livestock production systems, especially in upland areas, is loss-making. Therefore, the bar for comparing alternative systems is very low.

Any system that involves livestock will have impacts on GHGs other than carbon dioxide, namely methane (produced from animals' digestion and manure storage systems) and nitrous oxide (released from manure) (Conant and others, 2017). Additions of inorganic fertilisers also lead to significant releases of N_2O. Management must, therefore, be carefully designed and tailored to local conditions in order to ensure an overall net benefit in terms of GHG sequestration.

Although good grazing management has the potential to maintain soil carbon stocks, and, in areas where pasture is degraded or there is currently poor grazing management, to increase sequestration, there are multiple agroecological factors. This means that sequestration can often be negligible and/or very reversible. This is particularly true on soils that are already in good condition (Garnett and others, 2017).

There is a risk that increasing temperatures associated with climate change may lead to the release of carbon from certain grassland soils, although this could result in an increase in soil carbon in other situations (Jones and Donnelly, 2004).

The impacts of a change in pasture management will vary considerably depending on many different factors, including soil and livestock type and local climate. Additionally, the starting state of the land and, specifically, the legacy impacts of any previous land management practices, which can persist for decades or centuries, will have a considerable influence on the relative impact of any management system on carbon sequestration.

5.2.8 Costs

Amount of evidence

Agreement of evidence

The cost of soil management on pasture will equate to the level of financial support required to incentivise land managers to adopt the measures. This will be influenced by the potential financial impact of measures on other aspects of the farm system (in particular, impacts on saleable livestock or forage production).

One report provides costs associated with carbon removal measures on various farm types in Scotland as well as cost/benefit estimations (Lampkin and others, 2015). Values for some selected carbon management measures from this report are shown below.

Table 5-6 Costs associated with some soil carbon measures on pasture-based farms (Lampkin and others, 2019)

Measure/costs	Farm type (pasture-based systems)		Cost/benefit ratio (Increase in farm income £000/ktCO_2e)[19]
	Dairy	Lowland sheep/cattle	
Improved fertiliser application			
Increased costs (£/farm)	5,473	3,737	-13.5
Cost savings (£/farm)	844	444	
Net increase in farm income (£/farm)	-4,629	-3,293	
Reduced soil compaction			
Increased costs (£/ha)	317	238	+2.2
Cost savings (£/ha)	476	357	
Net increase in farm income (£/ha)	+159	+119	
Legumes in grassland			
Increased costs (£/ha)	10,788	2,765	-2.6
Cost savings (£/ha)	2,815	1,854	
Net increase in farm income (£/ha)	-7,973	-911	
Improved grazing management			
Increased costs (£/ha)	5,114	3,367	-4.5
Cost savings (£/ha)	13,351	3,104	
Net increase in farm income (£/ha)	+8,237	-263	

[19] This value is across a range of farm types, not just arable, including mixed farms, dairy, lowland sheep/cattle and upland grazing on Less Favoured Areas.

Overall, there are relatively few estimates of the costs of introducing such measures. An alternative broad indication of costs may be interpreted from a study in 2015 which estimates that the marginal abatement cost (that is, the cost of reducing an environmental negative such as GHG emission, per unit) associated with reducing emissions through better soil management was £51/tCO_2e at 2009 prices (Graves and others, 2015). However, this figure is considered a significant underestimate and relates to general soil management rather than being specific to measures on pasture land (CPRE, 2018).

5.3 Hedges and trees outside of woodlands

Table 5-7 Summary results for 'Hedges and trees outside of woodlands'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/ year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Removal	15-30	2-7	High[20]	Ready	Not ready	>10	Long term	Medium	High

5.3.1 Approach overview

In this section, we investigate the carbon storage and sequestration potential of 2 types of features: (1) hedges and (2) trees outside woodlands in agricultural landscapes (as opposed to on roadsides or in cities). Like trees in woodlands, trees outside woodlands and in hedges absorb carbon dioxide from the atmosphere during photosynthesis and incorporate it in their biomass as they grow. In this section, we focus on this process. We do not look at the additional potential storage of sequestered carbon associated with the lifecycle of wood products/arisings.

Hedges

Hedges are a prevalent feature of the British countryside, with 465,000 km of hedge in England and Wales alone (Carey and others, 2008). The most common species is hawthorn, but blackthorn and hazel are also frequent. Hedges provide multiple ecosystem services, including carbon sequestration in above and below ground biomass. Recent studies are starting to quantify how much carbon is stored and sequestered by hedges, and show that

[20] The categories for national abatement potential correspond to the following ranges: 'Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

this depends very much on how they are managed. For example, increasing the width of hedges increases biomass carbon and soil organic carbon pools, suggesting a potential use for uncultivated field margins (Axe and others, 2018).

Trees outside woodlands

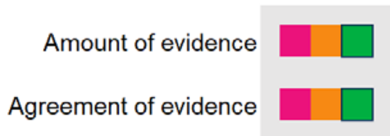
There are 565,000 hectares of tree cover outside woodland in England and 93,000 hectares in Wales (Forestry Commission, 2017). This includes trees on roadsides and in urban areas, as well as in agricultural areas. Although roadside trees and urban trees provide multiple ecosystem services, our focus here is on increasing the number of trees outside woodlands on pastures and arable land to sequester carbon. This approach differs from woodland creation in that its aim is not to convert agricultural land into forests, but rather to increase tree cover while, at the same time, maintaining agricultural production. The resulting density of trees varies but remains lower than in woodlands.

Because they are grown at wider spacing and are managed differently, the shape and size of trees outside woodlands differs from that of woodland trees. Their central trunk is less dominating and they have larger crowns and heavier branches (Williams, P.A., Gordon, A.M., Garret, H.E. and Buck, L., 1997). Carbon balances for increased trees outside woodlands depend on tree biomass (in leaves, branches, trunk, root system and litterfall) but also on soil carbon.

Both hedges and trees outside woodlands can be integrated into agricultural landscapes. The different patterns of tree inclusion are situated on a wide spectrum. At one end of the spectrum, there are small increases in trees and hedges within conventional agricultural landscapes. These increases are relatively easy to implement. On arable land, they involve increasing woody vegetation along field margins. In pastures, trees can be more readily planted on fields and at higher densities. At the other end of the spectrum is the creation and development of ‘agroforestry’, using trees in arable and pastoral systems (Mohan Kumar and Ramachandran Nair, 2011). These are a lot more integrated and require major changes in farming practices. In ‘agroforestry’ systems, the trees can be fruit producing, and, in some cases, trees outside woodlands could cover systems that may be described as traditional style orchards (though not modern conventional orchards, which are much more densely and uniformly stocked, and are outside our definitions).

Irrespective of their density, planting trees outside woodlands and hedges to sequester carbon has many advantages, including not requiring drastic land use nor risk leakage, and that methods for increasing woody vegetation are well known.

5.3.2 Readiness for implementation

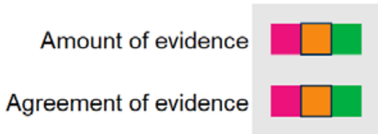


Methods for increasing tree cover on agricultural landscapes are not recent and there are many historical examples. They involve partnerships between landowners, businesses, governmental agencies and conservation charities like the Woodland Trust.

The mechanisms by which different species of woody vegetation store and sequester carbon are also well known. However, in agricultural systems, potential sequestration depends not only on the trees and hedges themselves but also on the type of soil, where they are being planted, their density and how they are managed. Empirically, carbon balances have been calculated for both hedges (Axe, 2018) and individual trees (Beckert and others, 2016).

However, as yet, the Woodland Carbon Code does not account for the carbon benefits of increased woody vegetation cover outside of woods, nor is there a centralised and unified method to estimate the carbon sequestration of increased tree cover on agricultural landscapes. Therefore, further policy developments are needed to formalise the carbon sequestration value of hedges and trees outside woodlands.

5.3.3 Speed and scale



Hedges

The carbon stored by hedges is the sum of above ground biomass, below ground biomass and soil organic carbon. Values depend on the species and on how hedges are managed. For example, hedges which are not managed by cyclical cutting, and are allowed to ‘grow out’ may sequester greater amounts of carbon (Crossland, 2015).

Table 5-8 displays specific values for carbon removal potential. Another study, which has not yet been replicated, found that unmanaged hawthorn hedges sequestered 44.7 tCO_2/ha/year, compared to 33.7 tCO_2/ha/year for unmanaged blackthorn and 10.05 tCO_2/ha/year for unmanaged hazel. This shows variations depending on species, and whether hedges are managed or unmanaged (Crossland, 2015). When managed (pruned) the accumulation of above ground woody biomass is limited to the cut dimensions of the hedge, beyond which the majority of the sequestration occurs in below ground carbon. This distinction between managed and unmanaged hedges appears to result in substantial variation to carbon sequestration rates.

Table 5-8 Carbon storage and sequestration of hedges (Axe, 2015; Crossland, 2015)

Hedge type	Carbon storage (tCO_2/hectare)
Hawthorn hedges	605.7
Nearby field margin	336.2
Unmanaged hawthorn hedge	44.7
Unmanaged blackthorn hedge	33.7
Unmanaged hazel	10.0

Hedge type	Carbon storage (tCO_2/hectare)
Managed hawthorn hedge	2.31
Managed blackthorn hedge	1.94
Managed hazel hedge	7.48

As of yet, studies on the impact of hedges on soil organic carbon (SOC) have been inconclusive (Ford and others, 2019). There are also no empirical studies focusing on changes to sequestration rates over time. However, there are models for biomass carbon storage which predict that rates fluctuate over time and are dependent on the species as well as on hedge management. If hedges are unmanaged, biomass above and below ground increases linearly over 100 years as hedges turn into lines of trees. If hedges are managed by coppicing, below ground biomass increases linearly (at a lower rate than for unmanaged hedges). Above ground biomass increases and then is reduced, first during trimming and then to zero with each coppicing cycle, with impacts for carbon which depend on what is done with the removed biomass. However, these trends do not account for soil carbon changes, and remain models rather than empirical evidence. Further empirical studies sampling changes in soil carbon and above and below ground storage are needed (Crossland, 2015), as are replications of the above-mentioned studies.

Trees outside woodlands

The rate of carbon sequestration by trees within a system depends on the species (for example, via growth rate (different species have different growth rates), age (mature trees store carbon but do not sequester at any notable rate), tree density (with more carbon sequestered per tree at lower densities) and tree management. Factors that contribute to higher carbon sequestration in agroforestry systems include greater above and below-ground diversity in the vegetation cover, increased litterfall inputs to the soil, extensive root exploration and production of higher amounts of biomass (Mohan Kumar and Ramachandran Nair, 2011).

Net sequestration values have been estimated for roadside trees in the south of England and stand at 6.5 kg C/tree/year or at 21.7 tCO_2/hectare/year. These values could be assumed to be quite similar to those of trees outside woodlands (Rouquette and Holt, 2017).

It must, however, be kept in mind that planting trees in agricultural landscapes potentially affects soil differently than in the case of roadside trees. Quantifying changes in tree and soil carbon pools helps account for the potential loss in soil carbon following the implementation of trees on pastures or arable land. One study looked at soil and tree carbon changes following the establishment of woodland and agroforestry trees on a grazed pasture (Upson and others, 2016). Agroforestry trees were established on 4% of the pasture. Fourteen years after planting the trees, the total carbon biomass for the silvopasture system was 232.5 tCO_2/ha compared to 218.5 tCO_2/ha for pasture and 301.0 tCO_2/ha for woodland. IN this case, silvopastoral agroforestry stored more carbon than equivalent areas of separate woodland and pastures. The amount of carbon stored by the agroforestry system was greater than a hypothetical scenario in which 4% of the land was woodland and 96% was pasture.[21] This suggests the interaction of trees and pasture has storage outcomes which are more than additive – combining two vegetation types increases the capture of resources like solar radiation and water.

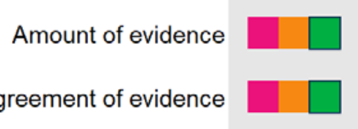
More generally, a common figure used for carbon sequestration in European agroforestry systems is 2tC/ha/year or 7.34 tCO_2/ha/year[†] (Lampkin and others, 2019). However, it must be held in mind that, as stated in the description of trees outside woodland, planting density can be carried out at a range of scales, which will, of course, influence the rate of sequestration (for example, sporadic planting at field edges versus more substantive agroforestry systems). Further evidence on how sequestration rates vary based on these planting densities would be recommended. A report by the Committee on Climate Change suggests nationwide sequestration potential of approximately 1 $MtCO_2e$/year if 1% of the grassland and arable land area were converted (Eory, 2015). This concerns agroforestry specifically and does not include hedges.

Variation in sequestration rates over time are likely to follow those of afforestation, as covered in section 3.3 Woodland creation, with low sequestration in the decade following tree plantation and then rapid increases in sequestration until the trees reach maturity. Beyond this, sequestration depends on how trees are managed, including whether or not they are harvested. Empirical estimates of the evolution of carbon sequestration over time are scarce and complex, partly because the residence times of the carbon stored in the biomass and soil pools differ (Upson and others, 2016).

Theoretical maximum abatement potential for these measures in the UK

The theoretical maximum abatement potential for hedges and trees outside woodlands in the UK has been categorised as high. To obtain this category, the approximate per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. For this measure, it was assumed the coverage of trees outside woodlands and hedges increased to match levels right after the Second World War (roughly, if their extent doubled). This stands between one million and 2 million hectares (Forestry Commission, 2017).

5.3.4 Co-benefits



There are multiple potentially tradeable co-benefits associated with increasing woody vegetation cover on agricultural landscapes. These include providing windbreaks as well as shade and shelter for livestock (reducing heat stress). Properly

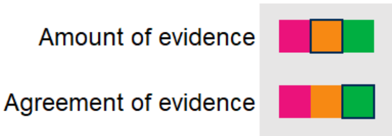
[21] In a hypothetical scenario where 4% of the land was woodland and 96% was pasture, total biomass would be equal to $0.04 \times 82.1 + 0.96 \times 59.6 = 60.5 \frac{\mathrm{tC}}{\mathrm{ha}}$.
We therefore see that: $63.4\ \mathrm{tC/ha} > 60.5\ \mathrm{tC/ha} \therefore$
$$\text{silvopastoral agroforestry storage} > \text{separated equivalent areas of habitats}$$

located woody vegetation can also contribute to water conservation and retention. It helps reduce soil and water movement by increasing water infiltration rates and slowing down the flow of transported sediments. Furthermore, deeper rooting trees contribute to soil stability and organic matter from litter and debris structure soils. All of this can limit run-off and reduce soil erosion both below the trees and hedges and in fields (Holden and others, 2019). Trees outside woodlands and hedges can also improve water quality by trapping pollutants, including phosphates and nitrates (Van Vooren and others, 2017), and by retaining nutrients (Burgess and Rosati, 2018), as well as increasing air quality.

From a farmer and/or landowner's perspective, increasing woody vegetation also involves less disruption to business than creating woodland, and less risk of reducing the value of their land. Well-sited hedges and trees outside woodlands also contribute to landscape quality and cultural heritage.

For conservation purposes, woody vegetation provides overwinter refuges, nesting sites, pollen and nectar feeding sources for pollinator populations throughout the year. Shelterbelts are 'highways' for the movement of various vertebrates and invertebrates and are, as such, highly valuable for biodiversity. Hedges and trees outside woods also increase soil biodiversity, for example, of earthworms and fungi (Holden and others, 2019).

5.3.5 Confidence in the science



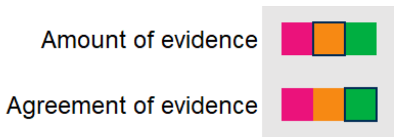
Hedges

Although some studies empirically quantify carbon storage by hedges at a point in time, more long-term chronological studies of sequestration processes specific to hedgerows are needed. Collecting further empirical data on the carbon sequestration potential of hedgerows will validate existing estimates and models and inform decisions not only at the farm management level but also for wider policy (Crossland, 2015). Replication of existing studies is required for additional certainty.

Trees outside woodlands

Our understanding of carbon storage and sequestration for trees outside woodlands is reasonably high, though less comprehensive than for woodland. Further models and data collection will help build common carbon budgets for different species planted in different agricultural systems in the UK.

5.3.6 Measuring impact



Hedges

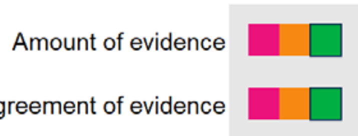
Empirically, studies quantify carbon removals by hedges using stratified random sampling of above ground biomass (with quadrats and sections) and of below ground biomass (with soil samples and excavation) (Axe, 2015). However, most studies limit soil sampling depths to 20 to 30cm. This depth is thought to correspond to the cultivated layer of arable land that is responsive to changes in management practices (Crossland, 2015). Yet it is now known that planting hedges and trees on arable land may increase soil organic carbon in upper layers of soil, but also decrease it in deeper layers. Insufficient sampling depth can lead to an overestimation of carbon stocks and sequestration rates for hedges and trees outside woodlands. Some studies also overlook the variations in carbon stocks over time, relying on a single sampling event. It is preferable to use multiple sites of different ages, however, this requires more research resources (Crossland, 2015).

Removals can also be quantified using models, but it should be noted that in the absence of measured data for hedgerows, these models often take values from average carbon stocks for other vegetation types. Resulting assumptions do not account for the effects of carbon cycling processes unique to hedgerows. Other assumptions, for example concerning hedge structure or vegetation type, also limit the accuracy of models (Crossland, 2015).

Trees outside woodlands

Carbon sequestration by an individual tree can be estimated using ‘allometric’ equations. These are based on changes in the tree diameter at breast heights and on the species. Soil organic carbon and root biomass are obtained by measuring coarse and fine root distributions, as well as by sampling soil (Upson and Burgess, 2013). However, high coefficients of variation when assessing changes in soil organic carbon complicate estimates of sequestration (Upson and others, 2016).

5.3.7 Risks and barriers

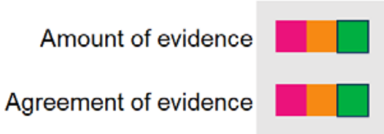


For farmers, there may be practical barriers to increasing woody vegetation cover. Trees and hedges need to be protected and may complicate moving machinery. There is some concern about tree roots (for example, of poplars or willows) damaging field drains (Raskin and Osborne, 2019). Trees also effectively reduce the area available for crops and livestock and require both knowledge of farm forestry and complex long-term investments. These barriers need to be balanced with the benefits (economic and otherwise) of increasing tree cover.

Up until now, planting trees and hedges on agricultural land could also conflict with Common Agricultural Policy (CAP) grants. If the cover of woody vegetation was high, then farmers risked not being ‘cross-compliant’ with respect to the requirement that land be in ‘good agricultural and environmental condition’. This could disqualify them from obtaining Basic Payment from the CAP.

Common Agricultural Policy is being replaced by the Environmental Land Management (ELM) scheme. It may be that ELM’s focus on environmental services beyond agricultural production will increase compatibility between woody vegetation cover and agricultural policy. In order to anticipate future climate change, it may be advisable to avoid planting trees which are vulnerable to warmer temperatures or droughts.

5.3.8 Costs



The costs of establishing hedges and trees on agricultural landscapes depends on species and location, and vary widely. Minimum costs stand around £5 per tree or plant, including protection and weeding. This is more expensive than planting trees in woodlands. There are also short-term costs associated with foregoing arable land (in terms of reduced yields), as well as further maintenance costs down the line. However, NGOs like the Woodland Trust have programmes to provide financial support and advice.

Maintenance costs for hedges can be quite high because they require regular trimming and periodic laying and relaying (Hinsley and Bellamy, 2000). The costs of these management processes depend on the type of hedge. Trimming costs around £1 per metre and laying £10 per metre of hedge. These values are provided to give a general idea of the costs of hedge maintenance but should not be used as references.

Establishment and maintenance costs for agroforestry systems also depend on the proportion of land converted. In the UK, if 1% of 75,000 ha of grassland is converted, establishment has been estimated to cost £11 million and maintenance £5 million a year (Eory and others, 2015).

In terms of cost per tonne of GHG removal, in the above case, assuming a removal rate of 7.34 tCO_2e/ha/year, conversion of grassland to agroforestry represented a cost effectiveness of £30/tCO_2e. Using a similar reasoning, the cost effectiveness of agroforestry on arable land for 1% of land converted is £15/tCO2e (Eory and others, 2015).

5.4 Enhanced weathering

Table 5-9 Summary results for 'Enhanced weathering'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/year)	National abatement potential	Technology	Certification method	Years		Evidenc e volume	Evidence agreement
Removal	40-360	6	Very high[22]	Ready	Not ready	>10	Med/long	Low	Low

[22] The categories for national abatement potential correspond to the following ranges: 'Low' corresponds to 0-1 Mt CO_2, 'Moderate' corresponds to 1-5 Mt CO_2, 'High' corresponds to 5-10 Mt CO_2 and 'Very high' corresponds to more than 10 Mt CO_2.

5.4.1 Approach overview

Within this approach, we focus on enhanced weathering through the addition of silicate rock materials specifically to arable soils.

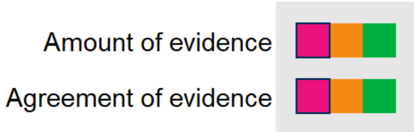
In the inorganic carbon cycle, silicate rocks naturally break down and are transformed to carbonate rocks through a process known as ‘chemical weathering’. This chemical reaction transforms CO_2 from the atmosphere into bicarbonate ions. Most of these ions eventually wash into the ocean where carbon becomes locked in as marine carbonates. Some dissolved inorganic carbon can also be sequestered on land through the formation of soil carbonate minerals (The Royal Society, 2018).

The process is a known climate control mechanism which happens over geological time scales. Enhanced rock weathering (ERW) is a CO_2 removal technique which aims to scale the process up by pulverising silicate rocks (obtained by mining or potentially by using silicate waste from mining, cement, or ash). This increases their reactive surface area and the rate of mineral dissolution. The rock dust can then be spread onto land, including on agricultural land which is already managed and altered. Here, plant roots and microbes accelerate the chemical reaction (Beerling and others, 2018).

Enhanced rock weathering could be used for offsetting purposes in the UK. Unlike other sequestration approaches, it does not compete with land used in agriculture (rock dust is spread on existing fields, reducing the risk of production displacement), nor for fresh water. Additionally, research has shown that there are potential co-benefits for soil improvement, crop yields and countering ocean acidification (The Royal Society, 2018).

Despite this theoretical promise, enhanced weathering is a long way from being ready to be implemented or scaled. The evidence base for its costs and benefits is as yet insufficient, as is empirical evidence concerning its carbon removal potential. Sourcing rocks remains an issue and may require mining if recycling waste materials is insufficient or unsuitable. Pilot projects and programmes could, within one or two decades, diminish uncertainty and help assess its suitability for carbon offsetting (The Royal Society, 2018).

5.4.2 Readiness for implementation



Technically, enhanced weathering could be used immediately. Some of the logistical infrastructure is in place. Many farmers already apply ground limestone onto fields to buffer soil pH (with an emission factor provided by the IPCC) (Hartmann and others, 2013). This means existing farm equipment could be used to apply fine-grained silicate rock (The Royal Society, 2018). The technology for crushing rocks and mining them (if needed) is also available, as are road networks for distribution onto farmland.

Despite this, the sequestration potential of ERW remains uncertain and estimates vary widely depending on what assumptions are made. Its impacts and costs are also uncertain. Thorough monitoring of negative environmental and social impacts is required, as is the development of mechanisms and incentives that would get farmers on board. Experimental and small-scale evaluations of ERW's efficacy and permanency as an offsetting method are also required (The Royal Society, 2018). These will help determine whether enhanced weathering can be used in the future as part of a portfolio of CO_2 removal techniques.

Researching the potential to implement ERW will, therefore, require close collaboration with researchers and universities who have been at the forefront of testing this approach (see the research output of Newcastle University's SUCCESS project as an example) (Newcastle University, n.d.). It will also mean building trust and engagement with the farmers who will be at the forefront of rolling out the approach.

5.4.3 Speed and scale

Amount of evidence

Agreement of evidence

The carbon removal potential of ERW depends on climate, with warm and seasonally wet climates conducive to high silicate rock weathering efficiency. Although countries in the tropics have consequently received more attention as candidates for ERW, the approach could also be used in the UK (Hartmann and others, 2013).

The carbon removal potential of enhanced weathering also depends on particle size and surface area-to-volume ratio of rock dust, on its application rates and on the carbon sequestration potential of the type rock used. The Royal Society Report on Greenhouse Gas Removal uses a simplified model to estimate the total potential of enhanced weathering if rock dust was applied to all arable land in the UK. It found that with moderate application rates (10 to 20 tonnes of rock dust/ha/year), the theoretical maximum CO_2 captured potential would be 12 to 21 $MtCO_2e$/year. If application rates were high (30 tonnes/ha/year), the maximum CO_2 captured potential would be 19 to 27 $MtCO_2$/year (The Royal Society, 2018). However, it is unclear which numbers, areas and rock types were used to obtain these estimates. At a per hectare level, the theoretical maximum gross carbon capture potential is 0.3 tCO_2e per tonne of basic rock (Renforth, 2012).

However, silicate dissolution can vary depending on pH, temperature and saturation (The Royal Society, 2018). The estimates above do not account for these uncertainties and variations. Moreover, other social and environmental limits of ERW will likely impose capacity limits below the maximum storage potentials cited above (Renforth, 2012).

Once rock dust is applied, it has been reported that carbonate forms in one to two decades, although here again the research is sparse. Thereafter, because it is assumed application is annual and dissolution rates are calculated on an annual basis, the sequestration rate is presented as stable, though this derives from simplified models with assumptions which are hard to verify.

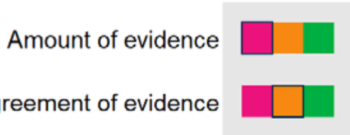
Ultimately, most of the CO_2 captured from the atmosphere by enhanced weathering will be stored in the ocean as dissolved inorganic carbon (The Royal Society, 2018). The storage capacity of the ocean is stable and large and carbon can stay concentrated in the Deep Ocean for the time it takes the surface and deep ocean to mix (1,000 years on average). Some of the carbon would also be stored as carbon minerals in soils or elsewhere on land, although it is unclear how long this storage would last.

Overall, the potential and permanency of storage are uncertain and depend on multiple factors. This makes the efficacy of CO_2 removal uncertain.

Theoretical maximum abatement potential for the measure in the UK

The theoretical maximum abatement potential for enhanced terrestrial weathering in the UK has been categorised as very high. To obtain this category, the per unit abatement potential found in the literature was multiplied by an estimate of the theoretical maximum area available for implementation in the UK. For this measure, the total available land area was derived from the total crop-bearing land area in the UK in 2019, an area which stands between 5 and 10 million hectares (Defra and others, 2020).

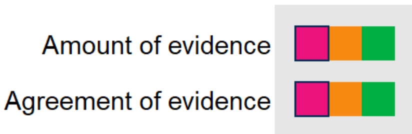
5.4.4 Co-benefits



ERW has potential co-benefits for agriculture and for oceans. Depending on the type of rock used, it can make water more alkaline and could, therefore, help counter ocean acidification which is considered a major threat for marine ecosystems (Bach and others, 2019).

The small-scale use of silicate rich slag as fertiliser in farming dates back to the 19th century. It is known to improve soils by increasing nutrient availability, carbon soil stocks and pH, and by reducing acidification. Supplying silica back into soils can also protect crops from pests and diseases which could reduce the need for pesticides, although this is largely unproven. It also potentially means increased crop productivity and yields. One study of sugarcanes in Mauritius found that adding basalt dust to soils increased yields by 30% over 5 successive harvests (D'Hotman De Villiers, 1961). However, effects depend on the rock material – different rocks have different compositions and can, therefore, potentially have different impacts on soil and crop growth.

5.4.5 Confidence in the science



Research on ERW is still in its infancy. As mentioned above, more research is needed to understand how much CO_2 the approach would capture. This research can help quantify saturation rates (key to whether rocks dissolve completely) and weathering rates (key to how quickly they dissolve) at a field scale.

Further research is needed to determine what happens to carbonate minerals and heavy metals in soils and their impact on soil biodiversity and food safety. Evaluations are also required to assess the impact of rock particles on rivers and oceans. These could decrease water clarity and lead to pH changes, with potential impacts on freshwater and marine biodiversity.

If silicate waste (from mining, cement, ash or slag) is used, research is needed to determine its suitability for cropland. The risk of toxicity is not well known and there may be a release of metals and persistent organic compounds. The impact of these compounds depends on the composition of rock and the type of crop grown. For example, sodium contained in plagioclase feldspar (a type of rock) is toxic to potatoes but beneficial to sugar beets.

5.4.6 Measuring impact

Amount of evidence

Agreement of evidence

To date, the potential for CO_2 removal from enhanced weathering has largely been calculated using the theoretical predictions of modelling studies and laboratory experiments. The carbon capture potential of specific rocks is calculated in laboratories and used to estimate how much carbon would be removed per tonne of rock disseminated (Renforth, 2012).

Assumptions, for example, that all silicate dissolves, are often left unchecked in these calculations. More generally, long-term experimental evidence from field sites is required, meaning ERW is not a viable option for sequestration in the near future. Evidence from the field and from laboratories will help determine the effectiveness and speed of reactions for different types of rocks and particle sizes as well as establishing monitoring and verification processes (The Royal Society, 2018).

5.4.7 Risks and barriers

Amount of evidence

Agreement of evidence

Some of the silicate could be sourced from silicate wastes that are already available from mine waste, cement, slag or ash, although field trials are required to assess suitability. If suitable, the use of UK's silicate waste would allow an application rate to all arable land of 10 tonnes/ha/year (The Royal Society, 2018).

If these materials are not suitable, or if application rates are high and enhanced weathering is scaled up, mining of silicate rocks would be required for widespread ERW. If done in the UK, rock mining would require energy for rock extraction, grinding and transportation (for example, from the areas with suitable rock material in the north of England and Scotland to arable areas in the east of England). This energy could account for 10 to 30% of the amount of CO_2 sequestered (Beerling and Long, 2018). If silicate rocks are sourced in other countries, land requirements could lead to tropical deforestation, and transportation of materials would result in further emissions.

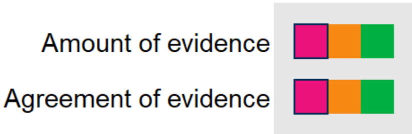
Widespread enhanced weathering could have implications for human health. These are related to the inhalation of rock dust. If inhaled, very small silicate rock particles can cause silicosis. This is a risk for those involved in mineral processing and in application to land.

Public opinion in the UK shows some support for research on ERW, on the condition that it is conducted in small-scale trials with careful monitoring and risk minimisation and that results are presented transparently (Pidgeon and Spence, 2017).

Environmental permits are required before spreading waste material on land. New policies will, therefore, be needed prior to widespread application of ERW (The Royal Society, 2018).

As of yet, ERW is not included in national or international carbon accounting agreements (The Royal Society, 2018). Further research on its efficacy and suitability is required.

5.4.8 Costs



Current cost estimates are highly uncertain. They also vary, with a detailed analysis of ERW potential in the UK estimating operational costs ranging between £44 and £361/tCO_2e in 2012 (Renforth, 2012). Costs depend on the prices of labour, diesel and electricity (Beerling and others, 2020).

Costs would likely decline as the market expands and technologies, such as more energy efficient and low-carbon machinery for rock grinding, develop.

5.5 Biochar

In this section on biochar we have not carried out a RAG rating for each of the different criteria. This is because this approach was not reviewed as extensively as the other approaches, due to being included at a later stage in the review.

Table 5-10 Summary results for biochar

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/ year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Removal	~70-270	~44[23]	High	Ready	Not ready	Immediate	Long term	Medium	Low

5.5.1 Approach overview

Biochar refers to charcoal used for soil amendment rather than for fuel. It is produced by heating biomass (to ~300-800°C) in low oxygen conditions, a process called ‘pyrolysis’. Biochar has 3 main potential benefits (Lehmann and Joseph, 2015), and, therefore, has been described by some as a ‘win-win-win solution’ (Laird, 2008):

- **carbon sequestration**: biochar is generally high in carbon and resistant to microbial decomposition so can sequester carbon in the soil for long periods of time
- **soil fertility**: biochar can improve soil nutrient and water retention and change soil pH
- **biofuel**: in addition to biochar, pyrolysis also produces syngas (a mixture of CO, CO_2, CH_4, H_2) and other minor components, which can be used as a biofuel to displace fossil fuels. However, syngas contains tars and carbon monoxide, which limit its use

[23] This assumes a biochar application rate of ~30t/ha/year, which is judged to be realistic but lower than any theoretical maximum. The main limitation is feedstock availability. This value also assumes that biochar is 60% carbon and 65% of this carbon lasts over 100 years.

There is also emerging evidence that biochar may reduce soil N_2O emissions, a potent GHG (Zhang and others, 2020). The main benefit of relevance to this report is carbon sequestration.

Four key factors influencing biochar net sequestration potential are (1) biochar yield from available feedstock, (2) biochar carbon content, (3) biochar residence time in the soil, (4) biochar influence on existing soil organic matter. These factors are extremely variable and depend on the biochar feedstock, the production conditions, the type of soil it is added to, the quantity in which it is added, and the climate. Biochar production yield can range from ~10 to 50% (Panwar and others, 2019); its carbon content can range from ~25 to 95% (Ippolito and others, 2015; Blanco-Canqui, 2020), and its residence time in the soil can range from 6 to over 5,000 years (Lehmann and others, 2015). In some situations, biochar can create a better environment for soil microbes, which increases the decomposition rate of soil organic carbon (SOC) and, therefore, reduces the net soil C sequestration achieved (Maestrini and others, 2015; Wang and others, 2016). Given this variability, it is hard to generalise about the sequestration potential of biochar.

Nevertheless, on average, biochar production yields are around 30% of dry biomass (Panwar and others, 2019); the carbon content is around 60% (Ippolito and others, 2015); and around 65 to 70% remains in the soil for over 100 years (Shackley and others, 2011). Assuming these averages and the availability of 3 to 20Mt of suitable feedstock, biochar has the potential to sequester ~2 to 13Mt of CO_2e per year in the UK, of which 1 to 9Mt may last over 100 years. Furthermore, biochar is unlikely to substantially increase SOC decomposition rates in the UK because, in general, they already have relatively good microbial activity.

Arguably, the main constraint on biochar carbon sequestration potential is the availability of feedstock. Additionally, using the available feedstock to make biochar may not have greater carbon abatement potential than other possible feedstock uses, and biochar production tends to be more expensive. Nevertheless, the potential co-benefits of biochar could add value.

5.5.2 Readiness for implementation

The technological readiness of biochar production is high. People have been making charcoal for thousands of years, and so the basic knowledge and equipment required already exists (Panwar and others, 2019). The technological readiness for using syngas (a by-product of biochar production) as a biofuel is low, due to the lack of efficient cleaning technologies to remove tars and carbon monoxide (Awais and others, 2018). Tars limit the use of syngas to external combustion engines, and carbon monoxide is poisonous.

There is currently no certification methodology for predicting and validating net biochar carbon sequestration. For prediction, further research is needed to understand the variability in biochar properties and how different biochars interact differently with different soil types. Additionally, robust life-cycle analyses of biochar production and a better understanding of biochar's wider impacts (for example, on existing SOC and N_2O emissions) will be needed to confidently calculate the net carbon abatement potential of biochar. For validation, soil carbon can be measured indirectly following biochar application, and the rate at which the carbon decomposes can be monitored.

5.5.3 Speed and scale

The speed of sequestration is immediate. Unlike most nature-based carbon removal approaches, biochar sequesters carbon as soon as it is produced. In contrast to other approaches, this carbon sequestration quantity may slowly decrease over time as some carbon is degraded.

The main constraint on the scale of biochar carbon sequestration potential is the availability of suitable feedstock. In theory, biochar can be made from anything organic, although the feedstock type will influence biochar yield and its physical/chemical properties. Feedstock can be either biowaste or purpose grown. However, given there are arguably much better uses of land, biowaste is preferable.

Estimating biowaste availability in the UK is challenging, particularly for commercial waste. Nevertheless, a 2011 paper estimated that around 3 to 20 Mt of biowaste feedstock suitable for biochar production is available in the UK each year (Shackley and others, 2011). Assuming a 30% yield of biochar from dry weight biomass (Panwar and others, 2019) and assuming that biochar is 60% carbon (Ippolito and others, 2015), this could hypothetically sequester ~2 to 13 Mt of CO_2 per year, of which 1 to 9Mt may last over 100 years. However, production yields and carbon contents are generally much lower for wet, non-woody biomass.

Although this quantity of biowaste is considered ‘available’, a large proportion of it is already allocated elsewhere and using this waste to produce biochar does not necessarily lead to greater carbon abatement than these alternative uses. High-level calculations suggest that using biowaste to displace fossil fuels via either anaerobic digestion or combustion could reduce CO_2 emissions by a similar amount to that which would be sequestered if this waste was converted into biochar. In fact, feeding biowaste to insects to displace soy may have a substantially greater carbon abatement potential than using it to make biochar (Palma and others, 2019). Furthermore, these options would likely be more economic than biochar production, which can be expensive. The most promising biochar feedstock is probably crop residues, such as straws, that are neither a particularly good feedstock for anaerobic digestion nor for combustion. If only straws were used for biochar production, the potential scale of carbon sequestration would be much lower.

5.5.4 Co-benefits

In addition to carbon sequestration, biochar can (1) improve soil fertility, (2) reduce soil N_2O emissions, and (3) produce syngas as a by-product, which can be used as a biofuel.

Biochar can improve a soil’s water and nutrient retention capacity (Lehmann and Joseph, 2015). Physically, biochar has a porous structure that can help retain water. Chemically, biochar tends to have a large negatively charged surface area, which can improve the soil’s ability to hold positively charged ions, including key plant nutrients. It can also act as a slow-release vehicle for fertilisers. Nevertheless, most nutrients in biochar are ‘locked up’ within its structure and are unavailable to plants. Therefore, whilst biochar can reduce nutrient leaching, it is not generally considered a nutrient source (Ippolito and others, 2015). Biochar generally increases soil pH, which can be an advantage or disadvantage depending on the soil type.

However, the effect of biochar on soil fertility is hugely variable, ranging from crop yield increases of 300% to yield decreases of 29% (Crane-Droesch and others, 2013; Jeffery and others, 2015, 2011). In many temperate areas, such as the UK, the soil already has a relatively high nutrient and water retention capacity and, therefore, biochar is unlikely to substantially improve soil fertility. A review by Jeffery and others (2017) found that biochar can increase crop yields in tropical regions by about 25% on average, but has little effect in temperate regions.

Biochar can also reduce soil N_2O emissions, by over 50% in some cases, though the mechanisms and timescales involved are still somewhat obscure (Zhang and others, 2020). Nevertheless, the Global Warming Potential of N_2O is approximately 300 times that of CO_2, so even very small effects persisting over several years may have important climate benefits.

The use of syngas as a biofuel has been proposed as a substantial co-benefit to biochar production (Lehmann and Joseph, 2015). However, syngas contains carbon monoxide, which is poisonous, and tar, which limits its use to external combustion engines (Awais and others, 2018). Furthermore, the yield of syngas depends on the production conditions, and decreases as biochar yield increases. Therefore, if the aim is to maximise biochar yield, the yield of syngas will be relatively low.

5.5.5 Confidence in the science

Over the last 20 years, biochar production and application has become increasingly well studied. Less than 200 articles about biochar were published before 2010, but about 13,000 articles (including both research articles and reviews) have been published between 2010 and mid-2020 (Blanco-Canqui, 2020). The results of these studies, however, show huge variability and some substantial knowledge gaps remain. Therefore, it is difficult to generalise the carbon impact of biochar addition to soils.

In particular, there are few long-term studies, few on temperate soils, few robust life-cycle analyses of biochar production, and few studies that investigate multiple, rather than just one, potential biochar benefits (Blanco-Canqui, 2020). Therefore, it is difficult to build a holistic picture of the net benefit of biochar addition, which could be used to confidently calculate its net carbon abatement potential and to assess the best use for biowaste feedstock.

5.5.6 Measuring impact

Soil carbon cannot be measured directly. At present, the most accurate method for measuring soil carbon is field sampling, followed by a dry combustion test (Chatterjee and others, 2009). A small sample of dry pulverized soil is heated to around 900°C, and the CO_2 gas that is produced from combustion is measured.

Established methods also exist for measuring N_2O emissions from soil, using small flux chambers and gas chromatography (Smith and others, 1995).

5.5.7 Risks and barriers

The main risks and barriers include: (1) feedstock availability, transportation, and alternative uses; (2) costs of biochar production and transportation; (3) potential for biochar to detrimentally affect soil fertility; (4) potential for producing toxic compounds during pyrolysis.

Estimating feedstock availability is challenging and comes with substantial uncertainty. Additionally, most biowaste feedstock in the UK already has an alternative use and making it into biochar may not provide greater environmental value. Although this is difficult to ascertain with any degree of certainty without more robust, holistic estimates of the environmental impact of biochar. Furthermore, feedstock is often widely distributed and can be expensive and challenging to gather and transport to a biochar production site.

Biochar production is expensive and this can be prohibitive. Its profitability would depend on potential revenue streams such as carbon offsets (which in turn rely on more reliable estimates of net carbon sequestration), possible crop yield increases (although these are likely to be limited in UK), and reduced costs of fertiliser/water treatment if nutrient leaching is reduced.

There is a risk that biochar application could negatively affect soil fertility in some areas, particularly due to unfavourable changes to pH in soils that are already alkaline.

Finally, pyrolysis of organic waste can produce toxic dioxins and furans, which are powerful carcinogens (Bucheli and others, 2015). The most toxic forms are often chlorinated, and this chlorine can come from the chlorophyll in green waste.

5.5.8 Costs

Cost has been identified as one of the main issues limiting widespread biochar use (Vochozka and others, 2016). Biochar production costs are dependent on a wide range of variables, with substantial uncertainty. One source suggests that industrially produced biochar costs around £500 to £800 per tonne of biochar, although this is not UK specific. In a UK context, the cost of biochar production (transport and spreading on the soil) may be around £100 to £400 per tonne (Shackley and others, 2011; UK Biochar Research Institute, 2011). Assuming biochar is 60% carbon and 65 to 70% of this carbon lasts over 100 years, the cost of abating one tonne of CO_2e in biochar may be around £70 to £270 per tonne. Although, this estimate is subject to substantial uncertainty.

Chapter 6. Built environment approaches

6.1 Household insulation

Table 6-1 Summary results for 'Household insulation'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	Per unit (tCO_2e/ha/ year)	National abatement potential	Technology	Certification method	Years		Evidence volume	Evidence agreement
Reduction	100-300	1	Very high	Ready	Not ready	Immediate	Short term	High	High

6.1.1 Approach overview

Heat in homes accounts for 15% of the UK's carbon emissions (Committee on Climate Change, 2019d). One way to reduce these emissions is by installing insulating material in a building's envelope, which reduces heat loss from the building. Under this approach, the purchaser of carbon credits would fund insulation projects that would not have happened otherwise, possibly in areas where economies of scale can be harnessed, like large blocks of flats or social housing complexes.

Installing insulation in buildings is generally technically simple and the materials and installation methods are well understood, although there is some variation in how complex the installation will be for buildings of different ages and types. Cavity wall insulation and loft insulation are among the cheapest ways of reducing heating emissions in homes, however these have been tackled to a large extent in the UK. Much of the potential for this kind of project is therefore in internal or external insulation for solid walled dwellings, which are generally harder to treat.

6.1.2 Readiness for implementation

Installing insulation is proven to reduce carbon emissions in buildings. However, it has not been used as a carbon offsetting approach in the UK, and a framework for reliably calculating and certifying emissions reductions must be developed.

Implementing this type of approach could be challenging, as it may prove difficult to find and approach large groups of properties needing insulation. This could be resolved through a partnership with a social housing operator, a local authority or a company already running large energy efficiency retrofit schemes (such as the Energy Company Obligation, ECO). Partnerships with organisations to carry out the installation would also be required.

6.1.3 Speed and scale

Amount of evidence

Agreement of evidence

Indicative values of the potential for abatement per household and nationally, broken down by type of insulation, are shown in Table 6-2.

Table 6-2 The carbon abatement potential for different types of insulation, per household and for the whole of the UK.

Insulation measure	Average abatement potential (tCO_2e/year/ household)[24]	National potential (million households)	Approximate national abatement potential ($MtCO_2e$)[25]	Proportion of total UK carbon emissions (2019)
Cavity wall insulation	0.5 - 1	4 – 5 [26] [27]	2	0.6%
Loft insulation	~0.1	2.9 [28]	0.6	0.2%
Solid wall insulation (external)	0.5 - 1	7.0-7.5 [29] [30]	3.2	0.9%
Solid wall insulation (internal)	0.5 - 1		1.6	0.5%
Total			7.4	2.2%

The potential for carbon emissions reduction achieved through insulating a home depends on:

- the carbon emissions of the home before (for example, the size of the home, its energy efficiency, its heating source). On average, heating a home in the UK produces about 2 tCO2e per year (Committee on Climate Change, 2014)
- how easy it is to cut these emissions (for example, the type and extent of the insulation that can be installed)

[24] Element Energy Limited and Energy Saving Trust (2013).
[25] Ibid.
[26] Element Energy Limited and Energy Saving Trust (2013).
[27] Committee on Climate Change (2016a).
[28] Ibid.
[29] Element Energy Limited and Energy Saving Trust (2013).
[30] Committee on Climate Change (2016a).

Carbon emissions from heating are reduced as soon as insulation is installed, meaning that funding insulation projects can be a very rapid approach to offsetting emissions (provided there is no serious ‘rebound’ in energy consumption).

There could be a limit to how quickly insulation could be installed in the UK because there are a limited number of qualified installers. Recent government funding commitments for domestic insulation may mean that uptake is high in the near future and the installer base could, therefore, be stretched. However, on a timescale of 10 years to meet a 2030 net zero target, this is unlikely to be impactful.

Baseline carbon emissions

The baseline level of emissions for this approach in the immediate term is the carbon emissions from heating the homes involved in the project. Into the future, establishing the baseline becomes more complex as the continued deployment of insulation and low carbon heating options means that the expected carbon emissions from heating fall with time. It is anticipated that insulation would be installed in a property anyway at some point before 2050 because of net zero commitments. The Committee on Climate Change states that a “15% reduction in energy used for heating existing buildings” is needed by 2030 to be on track for net zero (Committee on Climate Change, 2019e). Therefore, a household insulation offsetting project would not be able to continue claiming the initial level of emissions benefits indefinitely.

Establishing a baseline emissions pathway into the future must take into account government policy ambition and timelines that have not yet been set, as well as projections about the future technology and installation costs. A further consideration is the ‘base’ of homes being compared to. Is it the whole UK housing stock or only properties that had a similar performance before the insulation was installed? These assumptions will lead to uncertainty in estimates of the impact of the project.

Additionality and permanence

Initially, it can be assumed that the annual emissions reductions brought about by the project is the difference between the emissions from the energy consumed before and after insulation. Provided it could be shown that the funding from offsetting payments led to insulation being installed where it otherwise wouldn’t have been, the project will be additional. This additionality could be ensured by targeting particular buildings which would be very unlikely to be decarbonised in the near future without the additional funding. These may include:

- low-income homeowners who would not normally have the money available to make upgrades and who cannot access traditional sources of funding like the ECO
- homes that are very expensive to decarbonise but represent high potential for emissions reductions
- properties in the same building that are difficult to decarbonise individually and where more expensive measures must be installed collectively

Calculating the lifetime additionality of an offsetting-funded insulation project must take into account the falling baseline. One way to assess the ongoing emissions reductions that the project funding brings would be to track the national average performance of the building types in question and compare it to that of the insulated buildings. This could be done using a proxy value such as the proportion of buildings insulated or average energy consumption per m^2 of households.

Leakage

Improving energy efficiency can lead to a 'rebound' effect, where consumers use more energy after a measure has been installed thinking that the environmental/financial effect will be negated. In some instances, this will lead to excessive heating of a dwelling, whereas in others it may lead to healthier temperature levels. Clearly, the former needs to be avoided, while the latter is socially desirable as it will reduce negative health outcomes for residents. However, increased temperature levels will reduce the carbon benefits that could be claimed as the result of implementing a project.

As a result, a household insulation project should consider the potential for leakage both through overheating and through heating to a healthy temperature and whether these will significantly erode the potential benefits of the project. In all instances, part of the project should include educating recipients of insulation to avoid wasting heat.

There would also be embodied emissions associated with installing insulation arising from the products’ manufacture and transport. However, this is small compared to the lifetime benefits.

6.1.4 Co-benefits

Amount of evidence

Agreement of evidence

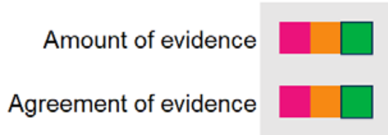
Insulated homes have lower fuel bills, which is particularly impactful for low-income households (in the UK more than 10% of households are in fuel poverty (Department for Business, Energy and Industrial Strategy, 2020a), meaning that they would fall below the poverty line if they sufficiently heated their homes throughout the year). According to the Energy Saving Trust, solid wall insulation saves around £225 a year in a typical house (Energy Saving Trust, 2020).

Insulation and retrofitting homes more generally could create many jobs. A recent report by the Energy Efficiency Infrastructure Group found that improving home energy efficiency could create and sustain 40,000 new jobs in the next 2 years as part of a coronavirus recovery stimulus package, with a total of 150,000 jobs created by 2050 (Energy Efficiency Infrastructure Group, 2020). Furthermore, for every £1 invested by government, GDP could increase by £3.20 and Treasury tax revenues by £1.25 (Verco and Cambridge Econometrics, 2014).

People who live in homes with higher energy efficiency tend to have better physical and mental health (Maidment and others, 2014). One study estimates that the net present value of improved health because of better domestic energy efficiency would be more than £4 billion in 2035 (Rosenow and others, 2017).

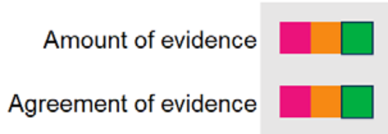
Finally, reduced energy consumption, and particularly consumption of natural gas for heating, could increase the UK’s energy independence from other countries.

6.1.5 Confidence in the science



The contribution of insulation to decarbonising a home is well understood. As described in section 6.1.3 Speed and scale, the project must establish how this contribution continues in time with respect to a baseline roll-out of insulation throughout the UK. A further major area for improving knowledge is modelling energy savings for a particular home with limited knowledge of its characteristics.

6.1.6 Measuring impact

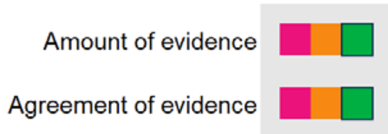


Carbon emissions cannot be practically measured using this approach. Reductions must be estimated via the following method:

- fuel/ electricity consumption is measured before and after insulation is installed (preferably for extended periods at multiple, equivalent points in the year)
- fuel consumption can then be converted to carbon emissions using simple conversion factors

Access to fuel consumption data could limit this method, particularly before insulation is installed. However, in properties managed by large institutions like local authorities or social housing companies, this should be not be an issue.

6.1.7 Risks and barriers



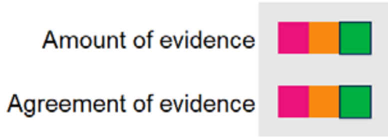
The rebound effect represents a serious risk to achieving carbon emissions through insulation. If insulation is not designed and installed properly it can be damaging, for instance if it does not allow sufficient ventilation and negatively impacts the health and comfort of occupants.

There is a risk that additionality will not be guaranteed in the long term because by 2050 nearly all homes will need to be at or near zero carbon to meet national carbon targets.

Paying to insulate buildings is not recognised as a method of carbon offsetting in the UK. For this project to become viable, a new framework for calculating and claiming the carbon benefits of insulation must be developed.

The UK government sees insulation as an important carbon reduction measure, as shown by its commitment to funding insulation as part of various programmes like the Green Deal, the ECO, and the recent £2 billion of ‘vouchers’ announced by the Treasury to support insulation installations (“Sunak to unveil £2bn home insulation scheme,” 2020). Therefore, it appears likely that insulation will continue to be supported by the UK government as a means of reducing carbon emissions. This may mean using insulation for offsetting could align with government priorities or undermine additionality.

6.1.8 Costs



Indicative ranges for the cost per measure and the marginal cost of abating one tonne of CO_2e are shown in Table 6-3. The range in the abatement costs arises from the fact that differences in dwellings make them more or less easy to tackle.

Table 6-3 Measure and marginal cost of carbon abatement.

Insulation measure	Cost (£/household) [31]	Approximate abatement cost (£/tCO_2e) [32]
Cavity wall insulation	500 - 1,000	Negative - 50
Loft insulation	200 - 400	Negative
Solid wall insulation (external)	12,000 - 15,000	350
Solid wall insulation (internal)	6,500 - 8,000	80

6.2 Household low carbon heating

Table 6-4 Summary results for 'Household low carbon heating'

Measure type	Approx. cost	Reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	£/tCO_2e	**Per unit (tCO_2e/ha/ year)**	**National abatement potential**	**Technology**	**Certification method**	**Years**		**Evidence volume**	**Evidence agreement**
Reduction	200-300	1	Very high	Ready	Not ready	Immediate	Short term	High	High

6.2.1 Approach overview

Low carbon heating is an essential part of decarbonising the UK. Heating domestic and non-domestic buildings (excluding industry) was responsible for 23% of the UK's emissions in 2016 (Department for Business, Energy & Industrial Strategy, 2018). Homes account for more than half of this, and are the focus of this approach (Committee on Climate Change, 2016b).

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[31] Department for Business, Energy and Industrial Strategy (2017).
[32] Element Energy Limited and Energy Saving Trust (2013).

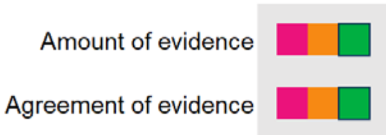
Conventionally, the first step in decarbonising a home is to improve its energy efficiency through measures like insulation, draught stripping and behavioural change. To further decarbonise a home once these measures have been put in place, it must be converted to use a low carbon heating system.

These technologies include heat pumps, heat networks and hydrogen boilers. Heat networks function at a large, infrastructural project level and are therefore likely to be too great a scale to be appropriate for carbon offsetting projects. Hydrogen boilers are not currently feasible because hydrogen gas is not produced at scale. This leaves heat pumps as the only viable low carbon heating option.

Heat pumps convert electricity into heat. They operate a cycle similar to that of a refrigerator in reverse: by using pumped refrigerant fluid to extract heat from the environment. An air source heat pump (ASHP) extracts this heat from the ambient air, a ground source heat pump (GSHP) from the ground. This section will concentrate on ASHPs because they are more widely applicable in the UK housing stock. They are generally easier to install as they don’t involve invasive ground works, and are cheaper than GSHPs. However, much of the analysis also applies to GSHPs.

A heat pump project could contribute to funding heat pump installation in those homes most in need, and least likely to be able to fund the change themselves. These homes could include social or low-income housing.

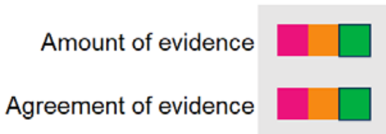
6.2.2 Readiness for implementation



Heat pumps are known to reduce greenhouse gas emissions (Rosenow, 2019) and will be a key technology in reaching net zero carbon emissions (Committee on Climate Change, 2015). The technology is well understood and no further R&D is needed to make it a viable approach for reducing carbon emissions. However, heat pump installation has not been used as an offsetting method in the UK, and new calculation and certification systems would need to be developed for the offset credits to be earned.

To implement this sort of offset approach partnerships would need to be set up to find and access the appropriate homes, as well as carrying out the work itself. Partnerships could, therefore, be developed with social housing providers or local authorities which do not have the budget or do not see it as economical to replace existing boiler heating. Alternatively, partnerships with building management companies or residents’ management companies could be established.

6.2.3 Speed and scale



On average, heating a home in the UK produces about 2 tCO_2e a year (Committee on Climate Change, 2014). Nearly all the 29 million homes already built in the UK will require some form of energy efficiency or heating improvement to reduce their carbon footprint (Committee on Climate Change, 2019e), and analysis suggests that as many as 19 million heat pumps could be economically installed by 2050 (Committee on Climate Change and others, 2019).

The total amount of current carbon emissions from home heating that must be abated by 2050 is approximately 50MtCO_2e/year (about 15% of current UK emissions) (Committee on Climate Change, 2019e) (Department for Business, Energy and Industrial Strategy, 2020b). Assuming that heat demand will be reduced by 15 to 30% (Committee on Climate Change and others, 2019) and that two-thirds of existing households will have a heat pump in 2050, it follows that heat pumps (alongside continuous electrification of the grid) could displace 30 to 40MtCO_2e/year of current national GHG emissions.

This translates to a GHG reduction potential for heat pumps of 1 to 1.5 tCO_2e/year per household. However, a heat pump installed today would not immediately lead to emissions of this magnitude, because the electricity grid is not fully decarbonised. Immediate annual emissions reductions would likely be about 20% of this (0.2 to 0.3tCO_2e/year) (Kelly and Cockroft, 2011).

The values stated here are uncertain because emissions cuts from heat pumps also depend on factors like the:

- size, construction type and age of the home
- energy efficiency of the home
- efficiency of the heat pump, and age and efficiency of the boiler being replaced
- existing heat distribution system (for example, radiators) of the home, and whether or not it needs to be replaced
- 'source' of the electricity (that is, whether or not it is from renewables)

The potential emissions reduction per household of a heat pump carbon offsetting project also depends on other, less tangible factors. For example, a project targeting lower income households, which often have very high energy consumption, could lead to a higher 'per-household' reduction than the national average. On the other hand, if (as the 'energy hierarchy' suggests it should) a heat pump is installed in tandem with insulation, then there are less 'available' emissions and the impacts of heat pump installation may be lower. Ongoing emissions reductions as a result of heat pump installation also depend on the baseline level of carbon emissions.

Carbon emissions reductions are immediate because heat pumps that use grid electricity have a lower carbon footprint per unit of energy supplied than boilers using natural gas.

Baseline carbon emissions

Heat from homes must be near-totally decarbonised by 2050 and it is likely that heat pumps will contribute a significant proportion of this (possibly as many as 19 million installations (Committee on Climate Change and others, 2019)). Because of this, it should be assumed that some of the homes involved in a project would receive heat pumps before 2050 without a carbon offsetting project, and that the project would, therefore, not be able to continue claiming additionality indefinitely.

The baseline is also affected by other factors. A heat pump's carbon emissions reductions will be lower in a home that uses less energy (that is, one that has high energy efficiency). National deployment of home insulation is anticipated up to 2050 and, therefore, the baseline energy demand, representing available emissions reductions, will fall.

The rate at which this heat pump and insulation deployment proceeds will depend on government policy and the future cost of technology, neither of which is certain. These factors must be taken into account when establishing the baseline carbon emissions.

Additionality and permanence

Initially, it could be assumed that the annual emissions reductions brought about by the project will be the difference between the emissions from the energy consumed before and after a heat pump has been installed (see section 6.2.6 Measuring impact). Provided it can be shown that the funding led to heat pumps being installed where it otherwise wouldn't have been, the project will be additional.

This additionality could be achieved by targeting particular buildings which would be very unlikely to be decarbonised in the near future without the additional funding. These may include:

- low-income homeowners who would not normally have the money available to make upgrades and who cannot access existing (or similar future) sources of funding like the ECO scheme because they do not meet eligibility criteria
- homes that are very expensive to decarbonise but represent high potential for emissions reductions
- properties in the same building that are difficult to decarbonise individually and where more expensive measures must be installed collectively

This additionality will decrease year on year as the baseline carbon emissions fall, which must be accounted for. One way to assess this could involve tracking the national average performance of the building types in question and comparing this to that of the buildings that had heat pumps installed as part of a project. This could be done using a proxy value like the proportion of buildings that have low carbon heating[33] or average heating emissions per m^2 of households. A further complication may arise when attributing the reductions earned when a home has insulation and a heat pump installed simultaneously.

Leakage

Improving energy efficiency can lead to a 'rebound' effect, where consumers use more energy after a measure has been installed thinking that the environmental/financial effect will be negated. A heat pump project should educate recipients of heat pumps of the dangers of this effect.

There would be embodied emissions associated with installing heat pumps due to the products' manufacture and transport. The refrigerants used in heat pumps are often

[33] For example, if 25% of maisonettes have a heat pump installed by the year 2035, a proxy baseline calculation method could be to assume that 25% of the heat in an Environment Agency-funded home would have come from low carbon sources in the baseline, and the Environment Agency funding provides the additional 75%.

greenhouse gases, and these leak during operation. The total contribution to global warming of the leaks is, however, low (Department of Energy & Climate Change and others, 2014).

6.2.4 Co-benefits

Amount of evidence

Agreement of evidence

A growing heat pump market would lead to higher demand for installers and create more jobs. Heat pumps do not create local air pollution, leading to improved health, however, this is difficult to quantify (Greater London Authority and Aecom, 2018).

6.2.5 Confidence in the science

Amount of evidence

Agreement of evidence

The contribution of heat pumps to decarbonising a home is well understood. However, as described above, it is less clear how big this saving will be over time, given it would be expected that there would be some heat pump installation even without the contribution of a project initiated for the purpose of carbon offsetting. This impact on the long-term savings of the measure would need to considered further before pursuing heat pumps as a project.

6.2.6 Measuring impact

Amount of evidence

Agreement of evidence

Carbon emissions from heating a home cannot be directly measured. Instead, understanding the change in energy consumption before and after the intervention will be required. To estimate this, the following process should be followed:

- fuel/electricity consumption for heating is measured before and after the heat pump is installed (ideally at equivalent points throughout the year)
- this consumption can be converted to carbon emissions using simple conversion factors

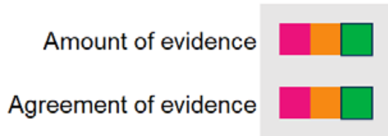
Given that this carbon offsetting approach is still in its infancy, there are no set of rules to follow on how monitoring should be completed. However, a sensible suggestion would be to monitor energy savings from a sample of homes that install a heat pump, and scale up the saving to the entire project. This would require some on-the-ground measurement activities that would need to be factored into the project design and costs.

It may be difficult to determine fuel or electricity consumption for some properties, particularly those using prepayment meters. This should be less of a problem with certain partners like social housing operators who manage energy consumption.

In order to estimate carbon emissions from the heat pump, assumptions would need to be made about the carbon factor associated with the electricity used. The government publishes data on these ‘grid factors’ (Department for Business, Energy & Industrial Strategy, 2020c). However, if a property has a ‘green’ energy tariff (in which the electricity used is associated with a Renewable Obligation Certificate showing that it came from a low carbon source), these associated emissions would be lower.

There are standard reporting techniques provided by international carbon reporting bodies which mean that these assumptions can be made in a consistent way (Greenhouse Gas Protocol, 2015).

6.2.7 Risks and barriers



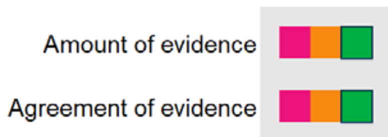
The major barrier of a heat pump carbon offsetting approach is public perception. Heat pumps are not a technology that has been widely deployed in the UK, and public understanding of the technology and how to interact with it is currently poor. Heat pumps are more efficient when operating at constant power, as opposed to the on/off operation of boilers, which is often unfamiliar to the occupants. If many people have unsatisfactory experiences with heat pumps as a result, this could lead to the technology gaining a poor reputation and limiting uptake. The project should incorporate engagement with individuals on the best way of using the heat pumps and their benefits.

Improving energy efficiency can lead to a 'rebound' effect, where consumers use more energy after a measure has been installed thinking that the environmental/financial effect will be negated; a heat pump offsetting project should educate recipients of heat pumps of the dangers of this effect.

There is a risk that additionality will not be guaranteed in the long term because by 2050 nearly all homes will need to have low carbon heating installed to meet national carbon targets.

Paying to install heat pumps in buildings is not recognised as a method of carbon offsetting in the UK. For this project to become viable, a new framework for calculating and claiming the carbon benefits must be developed. The UK government does not currently have a clear offsetting strategy, and so it is unclear whether it would be included on a future list of acceptable offset measures (should such a list be developed). However, the UK government considers heat pumps to be a key carbon reduction measure, as shown by its commitment to funding insulation as part of various programmes like the Renewable Heat Incentive (RHI), the ECO scheme, and the recent announcement of the Clean Heat Grant. Therefore, it appears almost certain that heat pumps will continue to be supported by the UK government as a means of reducing carbon emissions. This means that it is important to clearly demonstrate the additionality of a heat pump offsetting project.

6.2.8 Costs



An ASHP installation typically costs between £4,000 and £10,000 (compared to a gas boiler replacement cost of £2,500 to £4,500) (Heatable, 2020) (Building Services Research and Information Association (BSRIA), 2019). This wide range reflects the fact that the complexity of installation can vary based on the type of property and the amount of available space. Upfront costs are likely to fall in the future as the supply of installers in the UK increases, and companies produce heat pumps that are bespoke for the UK market.

The lifetime cost of carbon abatement of heat pumps varies because of these differences in capital cost, as well as variation in heat pumps' running costs. The Committee on Climate Change's current 'Speculative scenario'- in which all home heat emissions are cut by 2050 - finds that the majority of heat pump installations would abate carbon over their lifetime for less than £300/tCO_2e, with most installations having costs of £200 to £300/tCO_2e (Committee on Climate Change and others, 2019).

6.3 Other built environment carbon offsetting approaches

Presented in the following chapters are 4 further potential approaches to carbon offsetting. These are:

- renewable electricity production
- reducing water consumption
- building with timber
- low carbon transport

These approaches were not included in the detailed review (section 6.1 and 6.2). In the cases of renewable electricity production, reducing water consumption, building with timber, and low carbon transport, these were all considered possible reduction solutions which could be implemented as carbon offsetting projects, but have clear challenges which limited the need for a full review.

The later addition of these approaches to the review exercise means they have not been subject to the full review carried out for other approaches. However, they may still prove interesting opportunities to be considered in the future, and that is why they have been included in this report.

6.3.1 Renewable electricity production

Renewable electricity is generated from sources that are effectively inexhaustible and which produce a relatively small amount of GHG emissions per unit of energy produced compared with natural gas, the UK's principal fuel for energy supply. Electricity use accounts for 20% of the UK's GHG emissions (Department for Business, Energy & Industrial Strategy, 2020a).

A carbon offsetting approach involving renewable energy would see the installation of capacity (for example, solar or wind) and claiming the GHG emissions benefits arising from the displacement of fossil fuel-produced electricity. This method of carbon balancing is permitted on the international voluntary and compliance carbon offset markets, and there are established methods for assessing its carbon benefits.

Estimating the cost of carbon abatement from renewable technologies is difficult because there are assumptions which must be made about the lifetime cost of the technology, and the impact that increased renewable generation will have on the whole electricity system. However, it is likely that the cost of producing electricity from renewables, particularly solar PV and on- and off-shore wind, will continue the downward trend seen for the last 2 decades

(Department for Business, Energy and Industrial Strategy, 2020b). Funding renewable generation will most likely continue to be a cost-effective way of reducing carbon emissions over the coming decade. This is one reason why renewable energy carbon offsetting projects may not be viable in the UK: generating energy from renewable sources is increasingly cost competitive in the absence of carbon offsetting payments. Additionality is, therefore, likely to be hard to justify.

Furthermore, in the UK there are already systems for tracking and claiming the benefits of renewable electricity production: Renewable Energy Guarantees of Origin (REGOs). These certificates allow recording and reporting of how much electricity consumption has been generated from renewable sources. Separating out a REGO claim and an offsetting claim is likely to be complex and has the potential to hinder efforts to receive carbon offsetting payments for the installation of renewable generating capacity.

6.3.2 Reducing water consumption

Water is everywhere: every person, home and business in the UK relies on it to some extent. Water must treated for it to be suitable for consumption and used water must similarly be treated before it is allowed to return to the environment. This treatment uses energy and chemicals and, therefore, leads to carbon emissions, which can be reduced by cutting water consumption.

The UK government's emissions statistics indicate that, for every m^3 of water that is pre-treated, consumed and post-treated in the water network, sewerage and treatment operation emissions amount to about 1kgCO_2e (Department for Business, Energy & Industrial Strategy, 2020c).

These operations account for around 0.7% of the UK's carbon emissions (Ofwat, 2015). When the way water is used is accounted for, particularly hot water in homes, its carbon footprint increases to 5.5% of the UK total (Environment Agency, 2008). Reducing how much water we consume, particularly heated water, could, therefore, lead to a meaningful contribution to cutting the UK's carbon footprint.

Water consumption in the home could be reduced by improved metering, low flush toilets and other low-flow devices. These devices could together save 85kgCO_2e/year in a household, with savings coming predominantly from reduced demand for fossil fuels for heating (Environment Agency, 2008). Similarly, Waste & Resources Action Programme (WRAP) estimated that improved water use monitoring and demand-reduction technologies could save 30% of demand in businesses which have not made any previous attempts to curb consumption (WRAP, 2005).

Despite the excellent co-benefits of a water-reduction programme, water reduction measures for carbon offsetting purposes are unlikely to be possible at the necessary scale. With respect to business' water use (not including heated water), it would be challenging to demonstrate additionality in the long term. This is because the UK's water industry, which is responsible for the transfer and treatment of water used by homes and businesses in the UK, has declared its commitment to reach net zero carbon emissions by 2030 (Water UK, 2020). It could, therefore, be assumed that actions taken in this area would already be claimed as part of the water industry's emission reduction efforts.

6.3.3 Building with timber

The construction industry is directly responsible for around 10% of the UK's carbon emissions (Designing Buildings, 2020). Manufacturing steel and concrete, the most widely-used structural materials, requires large amounts of heat input and, therefore, has high associated emissions. Timber from coniferous trees can substitute for these materials in structural elements, which can lead to carbon emissions reductions in several ways, by:

- displacing high emissions intensity steel and concrete with low emissions intensity timber
- 'locking in' some carbon emissions to the structure of the building (however, this is only additional if that material would have been burnt an alternative scenario)
- reducing the operational emissions of buildings (that is, the energy used in the building)

Timber has been used as a construction material for many centuries, and continues to be prevalent in many countries despite falling out of fashion in the UK. It is no more technically complex to build houses and low-rise structures with timber, and the standards exist in the UK to do so.

It may be possible to develop carbon offsetting projects that balance emissions by funding construction with timber. This could take more than one form: funding could go to planting coniferous trees, with the ultimate aim being that the timber is used for construction. Alternatively, funding could encourage the construction industry in some way to expand its use of timber as a building material, leading to less steel and concrete use.

The complexity and uncertainty in calculating the emissions benefit arising from this approach precluded it from being reviewed in any more detail within this report. If coniferous trees were planted today, the material would not be ready until the 2050s at the earliest. Furthermore, increased coniferous planting could have detrimental impacts on the environment (Felton and others, 2010).

It is not clear that the alternative method, in which carbon offset funding is used to boost the use of pre-existing timber in the construction industry, could demonstrate additionality. Most of the coniferous timber grown in the UK is used in construction or similar products (Savills, 2019), meaning there is little potential for increased 'locking in' of emissions.

6.3.4 Low carbon transport

Transport, accounting for 18% of the UK's carbon emissions in the UK, is heavily reliant on fossil fuels (Department for Business, Energy & Industrial Strategy, 2020a). These emissions can be reduced by:

- reducing demand for transport, for example, by holding meetings online or reducing deliveries of consumable goods
- switching to active transport (walking, running, cycling)

- investing in or subsidising communal forms of transport like trains and buses, which have a lower carbon footprint per mile travelled per user
- switching to electric vehicles (EVs). Electric vehicles result in lower GHG emissions per mile than vehicles power by petrol or diesel. However, they still have very high embodied emissions (the emissions released in manufacturing cars and their parts), so it is not necessarily beneficial from a climate change perspective to dispose of a new fossil fuel car in place of a new EV

Reducing fossil fuel car use also has other benefits; combustion engines are sources of other emissions which reduce air quality (many places in the UK are regularly over legal air quality limits (Office for National Statistics, 2020)), while active transport leads to better health and fitness.

A carbon offsetting project using low carbon transport as a means of balancing carbon emissions could exist in several forms. The project could incentivise active travel through funding bikes or paying for cycle lanes. The money could be used to invest in EVs for individuals or businesses, for example.

Chapter 7. Conclusions

This evidence review has been collated to inform the development of the Environment Agency's carbon offsetting strategy. It has set out the current state of evidence for a variety of GHG reduction and removal approaches that could be implemented as carbon offsetting projects in the UK. Many of these approaches reviewed were not new in themselves, however there had been limited past research on their potential to be used for the purposes of carbon offsetting. This chapter summarises:

- main findings
- offsetting approach findings
- future research and evidence needs
- summary and next steps

7.1 Main findings

This review has found that all the approaches reviewed have strengths and weaknesses regarding their potential implementation for carbon offsetting. The evidence review shows that none of the approaches considered in this report meet all of the assessment criteria. This means that, in most cases, it will be necessary to pursue a range of approaches within an organisation's carbon offsetting strategy. This range would integrate the strengths of several approaches to maximise the likelihood of meeting the needs of the organisation which wishes to claim the climate benefits. This is particularly relevant for meeting short-term and interim net zero targets.

The exact range of offsetting approaches an organisation follows will depend on that organisation's needs, resources and objectives. How different organisations prioritise between, for example, scientific confidence in removal or reduction rates, short-term achievement of climate benefits, long-term permanence of impact, and project costs, will result in the selection of different approaches.

Two critical factors to consider are how quickly the approaches produce GHG emission reductions or removals, and the length of time that the climate benefits will be maintained (either in terms of additionality, or permanence). There is considerable variation between the approaches regarding these considerations. The interaction of these timing factors is important, particularly in the case of 2030 net zero targets (which are relatively short term). For many of the approaches reviewed to reach an impactful scale by 2030, implementation will have to begin very soon. Even if beginning to implement now, some approaches, such as woodland creation, may not produce large-scale GHG emission removals until beyond 2030.

The approaches reviewed include some that remove GHG emissions from the atmosphere, some that reduce the rate of GHG emissions to the atmosphere, and some that change between reductions and removals over time. Only carbon offsetting projects that remove GHG emissions will be compatible with true net zero emissions. This is where GHGs emitted into the atmosphere are balanced through equal removals of GHG emissions from the atmosphere. However, organisations may still find value in accelerating reductions elsewhere through carbon offsetting. This is especially the case in the shorter term, where the potential for GHG removals is more limited. Indeed, it may not be possible to achieve 2030 targets using purely GHG emission removal projects.

At present, only two of the approaches reviewed – woodland creation and upland peatland restoration - have certification standards that enable them to be used for offsetting in the UK. For other approaches reviewed, varying degrees of research and development will be required to progress with them for the purpose of certifiable carbon offsetting. This research and development may be needed in the area of fundamental scientific understanding of reduction or removal impacts, or in developing certification standards that provide a common process for calculating the climate benefits of a project.

It is important that care be taken to select carbon offsetting approaches that are suitable for local habitats. One land management approach will not suit all circumstances. This is an important reason why this study has reviewed a variety of potential offsetting approaches. Many of the approaches reviewed produce co-benefits such as biodiversity enhancement, and these should be taken into account when selecting approaches. It may be the case that further funding streams are available for supporting these co-benefits, and cost assessments should take these potential sources into account.

Working with landowners and land managers was also found to be vital. Understanding the needs of landowners and other local stakeholders, and designing projects that achieve outcomes for multiple beneficiaries, will help accelerate project implementation. The permanence of land use changes on agricultural land requires particular attention in these circumstances, to ensure climate benefits are maintained over time.

7.2 Offsetting approach findings

As well as these overarching conclusions, the evidence review has enabled us to make the following specific conclusions for the different offsetting approaches. Tables 7-1 and 7-2 summarise the main information collected for each approach.

7.2.1 Land-based carbon offsetting approaches

Woodland creation is the most advanced carbon offsetting approach in the UK - Woodland projects are advantaged by being certifiable through the Woodland Carbon Code. The science of sequestration is well understood, and permanence concerns are less pronounced than for many other approaches.

Upland peat restoration is also a well-developed approach - Projects can be certified through the Peatland Code, and substantial reductions to current emission rates can be achieved.

Lowland peat restoration also has good potential as a carbon offsetting approach - However, there has been less research into the carbon benefits of restoration, and restoration of fens is not currently included in the Peatland Code. Both upland and lowland peat restoration reduce GHG emissions, and may eventually remove GHG emissions, which needs to be considered in the context of net zero definitions that prioritise GHG removals.

7.2.2 River and coastal carbon offsetting approaches

Flood plain restoration presents an interesting GHG removal opportunity - Studies have indicated a high per hectare GHG sequestration rate. However, this approach has not been extensively reviewed in the scientific literature and the variety of factors influencing carbon fluxes means there is uncertainty regarding potential outcomes.

The GHG removal potential of constructed wetlands needs more research – There is currently a very limited evidence base. Present research indicates that constructed wetlands can act as both sources and sinks of GHG emissions. The variety of factors at play, such as the design of the wetland and vegetation types, make it challenging to firmly conclude preferable management for carbon sequestration.

Saltmarsh restoration has been shown to achieve relatively high rates of GHG removal - However, the challenges of coastal squeeze and sea level rise could affect the permanence of this habitat.

Seagrass restoration has strong co-benefits, but evidence of GHG removal is still subject to scientific debate - Some restoration projects have now begun and the outcomes of these should be used to expand this evidence base in the future. Measurement and monitoring processes may be more challenging in the marine environment than on land.

7.2.3 Agricultural carbon offsetting approaches

Carbon management practices on arable soils and pasture grassland offer considerable opportunities based on the applicable land area - There is a growing body of evidence to support these approaches but some uncertainty remains regarding the carbon benefits, due to factors such as the variety of starting land conditions, soil types, and range of potential interventions. Despite a large available land area, the per hectare sequestration benefit is less substantial than other approaches such as woodland or peat restoration. Some implementation challenges may need to be overcome, including high risk to permanence (for example, by reinstating tilling), and persuading landowners to alter their farming practices.

Increasing the prevalence of hedges and trees outside woodland offers further opportunities for carbon removal on agricultural land - Concerns regarding permanence would likely be limited as it is unlikely hedges and trees would be removed once established. Hedges and trees outside woodland would benefit from further scientific investigation of the scale of potential carbon removal to increase confidence in their benefits.

Enhanced terrestrial weathering has shown promise as a carbon offsetting approach in early studies - Relative to other approaches reviewed, knowledge is less developed about the benefit of enhanced terrestrial weathering, and further research will help address evidence gaps. In addition to uncertainty regarding carbon benefits, there is also a need to understand wider impacts on soil, water, and food chain contamination in more depth.

7.2.4 Built environment carbon offsetting approaches

Household insulation and heat pumps have the potential to reduce emissions in the UK substantially, and can do so quickly - Their benefit is achieved through reductions, and this would need to be considered in the context of net zero definitions. Scientific understanding of emission reduction processes is well developed, although there is not yet a carbon offsetting standard in the UK for these approaches. The additionality of these approaches may be shorter lived than natural environment approaches given the imperative of reducing emissions from the built environment.

7.3 Future research and evidence needs

To accelerate the domestic offsetting market there is a need for more in-situ measurement and monitoring of impacts in order to increase scientific confidence in the different offsetting approaches. For all the natural environment approaches reviewed, the variability of local conditions and landscape management practices means using generic factors to calculate GHG reductions and/or removals may lead to uncertainty regarding climate benefits. Measuring and monitoring processes using field-based measurements will help to increase confidence in the claims of projects. Measurement processes would need to establish the baseline emissions of the project, and be carried out periodically to track impacts over time. The costs and benefits associated with measurement and monitoring would need to be considered during project design.

There is currently limited information detailing the costs of implementation. In addition, there are inconsistencies between sources in what is included and excluded from cost assessments. More detailed consideration of implementation and operational costs would be required to move forward confidently with implementation for most approaches. These considerations should include the potential to combine multiple funding streams for multiple environmental outcomes. Whether land would need to be purchased to proceed with a project is another important consideration. The ownership and claiming of carbon benefits will also need to be considered as part of costs planning.

7.4 Next steps

The evidence collected within this report will now be used to support the development of the Environment Agency's carbon offsetting strategy. To develop this strategy, the Environment Agency will draw on insight into the international carbon offsetting market and a review of ethical and political considerations relating to carbon offsetting. It will also make use of an assessment of potential land-use changes on its own estate, and discussions with potential project implementation partners. Drawing the outcomes of these exercises together, the Environment Agency aims to develop a robust and innovative solution to the carbon offsetting component of its 2030 net zero target.

Table 7-1 A red-amber-green (RAG) analysis of the different offsetting approaches' performance against the key indicators, to inform decision making. A grey circle indicates that there is not enough reliable information in the literature to draw a conclusion. For a description of how RAG analysis was carried out see Appendix 2.

Approach	GHG emission reductions or removals	Abatement cost	Per-unit abatement potential	National abatement potential	Implementation readiness	Speed of impact	Longevity	Confidence in science
Upland peat restoration	Reductions and potential removals	🟢	🟢	🟢	🟢	🔴	🟢	🟠
Lowland peat restoration	Reductions and potential removals	⚪	🟢	🟢	🟠	🔴	🟢	🔴
Woodland creation	Removals	🟢	🟢	🟢	🟢	🔴	🟢	🟢
Grassland	Removals	⚪	🟠	⚪	🟠	🟠	🔴	🔴
Floodplain restoration	Removals	🔴	🟢	🟠	🟠	🔴	🟠	🔴
Constructed wetland	Removals	⚪	⚪	⚪	🟠	⚪	⚪	🔴
Saltmarsh restoration	Removals	⚪	🟠	🔴	🟠	🟠	🟢	🟠
Seagrass restoration	Removals	⚪	🟠	🔴	🔴	🔴	⚪	🔴

Approach	GHG emission reductions or removals	Abatement cost	Per-unit abatement potential	National abatement potential	Implementation readiness	Speed of impact	Longevity	Confidence in science
Kelp restoration	Removals	Grey	Amber	Grey	Red	Grey	Grey	Red
Soils management: arable	Removals and reductions	Amber	Red	Amber	Amber	Amber	Amber	Amber
Soils management: pasture	Removals and reductions	Amber	Amber	Green	Amber	Amber	Amber	Red
Hedges and trees outside of woodlands	Removals	Green	Amber	Green	Amber	Red	Green	Amber
Enhanced weathering	Removals	Amber	Green	Green	Red	Red	Amber	Red
Biochar	Removals	Amber	Green	Green	Amber	Green	Green	Amber
Household insulation	Reductions	Amber	Amber	Green	Amber	Green	Red	Green
Household low carbon heating	Reductions	Amber	Amber	Green	Amber	Green	Red	Green

Key for table:

- 'Per-unit' is per-hectare for natural environment measures and per-household for built environment measures.
- Appendix 2 describes the red, amber, green or grey rating.

Table 7-2 A summary of the key results of the review for the different carbon offsetting approaches

	Measure type	Cost (approx.)	Removal/reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	Reduction/ removal	£/tCO_2e	Per unit per year	Nationwide	Technology	Certification method	Years		Evidence volume	Evidence agreement
Upland peat restoration	Reduction and potentially removal	10-100	2-20	Very high	Ready	Ready	>10	Long	Med	High
Lowland peat restoration	Reduction and potentially removal	Uncertain	5-20	High	Ready	Not ready	>10	Long	Med	Low
Woodland	Removals	20-25	11	Very High	Ready	Ready	~10	Long	High	High
Grassland	Removals	n/a	2	n/a	Ready	Not ready	<10	Short/Med	Low	Low
Flood plain restoration	Removals	>1000	10	Moderate	Ready	Not ready	>10	Med/Long	Low	Low
Constructed wetlands	Removals	Uncertain	Uncertain	Uncertain	Ready	Not ready	Uncertain	Uncertain	Med	Low
Saltmarsh restoration	Removals	Uncertain	2-8	Low	Ready	Not ready	<10	Long	High	Med
Seagrass restoration	Removals	Uncertain	1.6	Low	Not ready	Not ready	>10	Uncertain	Low	Low
Kelp restoration	Removals	n/a	2.15	n/a	Not ready	Not ready	Uncertain	Uncertain	Low	Low
Soil management: Arable	Removals	100 – 1000+	0.5-1	Moderate	Ready	Not ready	<10	Med/Long	Med	Med

	Measure type	Cost (approx.)	Removal/reduction potential		Readiness		Speed of impact	Longevity	Confidence in science	
	Reduction/ removal	£/tCO_2e	Per unit per year	Nationwide	Technology	Certification method	Years		Evidence volume	Evidence agreement
Soil management: Pasture	Removals	10-1000+	0.2-4	Very high	Ready	Not ready	<10	Med/long	Med	Low
Hedges and trees outside woodlands	Removals	15-30	2-7	High	Ready	Not ready	>10	Long	Med	High
Enhanced weathering	Removals	40-360	6	Very high	Ready	Not ready	>10	Med/long	Low	Low
Biochar	Removals	70-270	44	High	Ready	Not ready	Immediate	Long	Med	Low
Household insulation	Reduction	100-300	1	Very high	Ready	Not ready	Immediate	Short	High	High
Household low carbon heating	Reduction	200-300	1	Very high	Ready	Not ready	Immediate	Short	High	High

Key for table:

- 'Per unit' refers to per hectare for natural environment, per household for built environment
- For 'national abatement potential': 'Low' corresponds to 0-1 MtCO_2e, 'Moderate' corresponds to 1-5 MtCO_2e, 'High' corresponds to 5-10 MtCO_2e and 'Very high' corresponds to more than 10 MtCO_2e
- For biochar, per unit assumes a biochar application rate of ~30t/ha/year, which is judged to be realistic but lower than any theoretical maximum. This value also assumes that biochar is 60% carbon and 65% of this carbon lasts over 100 years.

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List of abbreviations

ASHP	Air source heat pump
BAU	Business as usual
C	Carbon
CAP	Common Agricultural Policy
CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
COP	Conference of the Parties
DOC	Dissolved organic carbon
ECO	Energy Company Obligation
ELM	Environmental Land Management scheme
ERW	Enhanced rock weathering
EV	Electric vehicle
GDP	Gross domestic product
GHG	Greenhouse gas
GSHP	Ground source heat pump
GWP	Global warming potential
IPCC	International Panel on Climate Change
IUCN	International Union for Conservation of Nature
LED	Light emitting diode
NGO	Non-governmental organisation
N_2O	Nitrous oxide
PES	Payment for ecosystem services
POC	Particulate organic carbon
RAG	Red-amber-green
REGO	Renewable Energy Guarantees of Origin

RHCP Regional Habitat Compensation Programme

RHI Renewable Heat Incentive

RSPB Royal Society for the Protection of Birds

SAC Special Area of Conservation

SOC Soil organic carbon

UNFCCC United Nations Framework Convention on Climate Change

VCS Verified Carbon Standard

List of units

4p1000	4 per 1,000
£/tCO_2e	Pounds per tonne of carbon dioxide equivalent
GtC	Gigaton of carbon
Ha	Hectare
mgC/m^2/hour	Milligrams of carbon per metre squared per hour
MtCO_2e	Million tonnes of carbon dioxide equivalent
tC	Tonnes of carbon
tC/ha/year	Tonnes of carbon per hectare per year
C/m^2/year	Carbon per metre squared per year
tCO_2e/ha/year	Tonnes of carbon dioxide equivalent per hectare per year

Glossary

Additionality	A carbon offset project is additional if it leads to reductions or removals of greenhouse gases that would not have happened otherwise. Proving additionality is challenging as it requires comparison with a business as usual scenario.
Business as usual	Business as usual refers to the ongoing greenhouse gas emissions that would occur without any intervention.
Carbon dioxide equivalent (CO_2e)	A metric used to compare various greenhouse gases based on their global warming potential. The CO_2e quantity of any greenhouse gas is the amount of carbon dioxide that would produce the equivalent amount of global warming over a specified period (usually 100 years).
Carbon offsetting	The practice of reducing or removing greenhouse gas emissions to balance ongoing greenhouse gas emissions, to achieve claims such as climate neutrality or net zero.
Greenhouse gas emission reductions	Reducing the emissions of greenhouse gases compared to business as usual. The result is a lower quantity of greenhouse gases being emitted to the atmosphere.
Greenhouse gas emission removals	Removing greenhouse gases from the atmosphere and holding them in some form of long-term storage, such as plants, soils, oceans or geological features.
Global Warming Potential	A metric that enables comparisons of the global warming impacts of different greenhouse gases. It is a measure of how much energy the emissions of one tonne of a gas will absorb over a given period of time, relative to the emissions of one tonne of carbon dioxide. The time period is often 100 years, but this can underestimate the impact of short-lived greenhouse gases such as methane.
Leakage	When a GHG reduction or removal project in one place leads to increased GHG emissions elsewhere.
Permanence	A removal or reduction project is permanent if the impact on atmospheric greenhouse gas levels is not reversed at some point in the future. Permanence can vary, and not all carbon offsetting approaches may achieve indefinite permanence.

Appendix 1. Selecting the approaches

Not all potential carbon offsetting approaches could be reviewed within the scope of this report. To select those of most interest to the Environment Agency, a shortlisting process was carried out.

An initial longlist of potential approaches was developed. This list included the following natural environment approaches:

- agroforestry
- brownfield site to woodland
- conifer to broadleaf woodland
- enhanced weathering
- freshwater wetland
- grassland to constructed wetland
- grassland to woodland
- hedges and trees outside woodland
- lowland peatland
- saltmarsh restoration
- seagrass
- soil carbon practices: arable
- soil carbon practices: pasture
- upland peatland

The longlist also included the following built environment approaches:

- insulation (retrofit)
- renewable energy generation
- low carbon heating (retrofit)
- low carbon transport
- timber framed buildings
- low GWP refrigerants
- new build energy efficiency
- green walls
- bioenergy carbon capture and storage
- increasing recycling rates
- direct air carbon capture and storage
- biomass burial
- low carbon concrete
- bacterial in landfill

Based on the existing knowledge of the research team, each measure on the longlist was scored against several criteria:

- readiness of approach

- applicability to the Environment Agency estate
- applicability to Environment Agency partner estates
- ease of measuring carbon sequestered/reduced
- speed of sequestration/reduction
- co-benefits
- scalability
- costs
- permanence

The scores for each approach were then added together, with the project steering group discussing those that performed best to select the final approaches for review. The assessment criteria used in the research phase were changed slightly from the original shortlisting criteria bullet pointed above. These adjustments were made for the following reasons:

- as the understanding of important factors developed during the course of the shortlisting stage, additional factors were added to the list
- some restructuring of the assessment criteria made for a more logical flow of the report
- the focus on the Environment Agency's estate, and potential partner estates was removed. This having been established during shortlisting, it was not necessary to include it in the main report

Appendix 2. Scoring the approaches

A RAG (red-amber-green) analysis was developed to assess the performance of different offset approaches against the following criteria:

- abatement cost - the lifetime cost of reductions or removals achieved (£/tCO_2e)
- per-unit abatement cost - likely maximum reduction or removal rate (tCO_2e/ha/year)
- national abatement potential - maximum theoretical potential for carbon reduction/removal in the UK (MtCO_2e/year)
- speed of impact - time taken to have a considerable impact on net emissions (years)
- implementation readiness - in technical and methodological (for example, carbon accounting) terms, what state of readiness is this approach at?
- longevity - the length of time over which the measure reduces or removes GHGs at or near its maximum rate, when compared with the baseline
- confidence in the science – the score for the confidence in the science category (amount of evidence, agreement of evidence)

It is critical that each measure is assessed against narrowly defined criteria in order to minimise space of individual interpretation and, therefore, uncertainty in the results. In certain cases, arriving at a performance assessment involved deriving categorical scores from qualitative information and, therefore, the development of performance boundaries. The scoring ranges were created by taking into account ranges of values in the data and by choosing sensible boundaries. In order to differentiate the measures, we loosely aimed to have a relatively even distribution of 'reds', 'ambers', and 'greens'.

Where data were not available in the literature (and common-sense logic couldn't be applied), a 'grey' score was applied. In the context of this work, it is important to highlight where there is a lack of knowledge.

Where evidence for a measure suggests a range of values, and this range overlaps 2 categories (for example, the approach could fall into red or amber), the RAG score is allocated based on the value that researchers deemed most fairly represented the approach potential over its lifetime.

Table A-1 sets outs out the red, amber and green categories against the different criteria.

Table A-1 Description of red, amber and green categories against the different criteria

	Abatement cost	Per-unit abatement cost	National abatement potential	Speed of impact	Implementation readiness	Longevity	Confidence in the science
Red	>£1,000/tCO_2e	0.1-1tCO_2e/ha/year	0.1-1MtCO_2e/year	>10 years -	Insufficient understanding regarding how to technically implement the approach at scale. There are obstacles to quantifying the approach's GHG impacts which limit its ability to be used as an offsetting approach immediately.	<20 years	Low amount OR low agreement
Amber	£100-1,000/tCO_2e	1-5tCO_2e/ha/year	1-5MtCO_2e/year	<10 years	Technique is understood and could be implemented, but there are methodological obstacles to quantifying the approach's GHG impacts which limit its ability to be used as an offsetting approach immediately.	20-50 years approx.	Other
Green	£1-100/tCO_2e	>5tCO_2e/ha/year	>5MtCO_2e/year	Immediate impact (reduction/removal delivered at scale following implementation)	This is currently undertaken as an offsetting approach in the UK.	50+ years	High amount AND high agreement

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