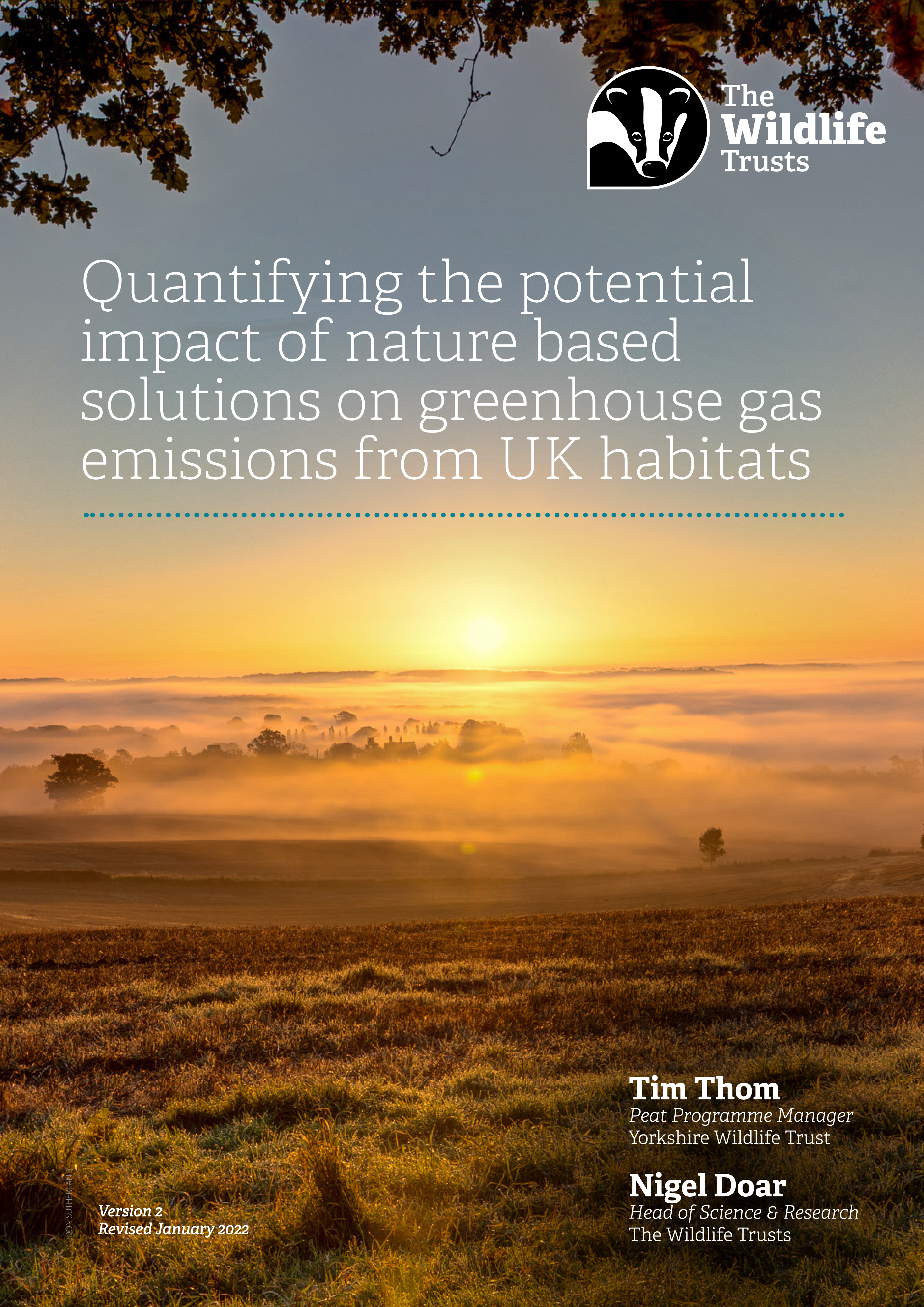

The Wildlife Trusts

Quantifying the potential impact of nature based solutions on greenhouse gas emissions from UK habitats

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DON SUTHERLAND

Contents

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1 Executive Summary

1.1 Background

During 2020 and the early part of 2021, The Wildlife Trusts refreshed their collective strategy — the guiding framework that unites the 46 individual Wildlife Trusts and the movement's central charity (the Royal Society of Wildlife Trusts, RSWT) — to re-focus and re-energise the movement's collective work over the coming decade.

Climate change is one of the biggest threats that nature faces in the decades ahead, even here in the UK. The natural world needs to recover so it can cope with climate change that is already happening, so its continuing decline doesn't make climate change worse and so it can contribute effectively to removing and storing carbon. The Wildlife Trusts are deeply committed to making this happen; Part of our vision is that wildlife and natural habitats play a valued role in addressing the climate and ecological emergencies.

This report was commissioned to give The Wildlife Trusts technical insight into the scientific evidence that is currently available on the relationship between habitat creation, restoration and management, and atmospheric greenhouse gas (GHG) levels . This evidence is necessary for the movement to make well informed decisions and have credible data on the climate benefits of different nature-based solutions. This report specifically sets out to present and interpret the best readily available relevant scientific evidence relating to the potential for UK habitat creation, restoration and management to contribute to nature-based sequestration of GHGs and reduction of land-use-related GHG emissions. A literature review was carried out to determine GHG fluxes for as many UK habitats as possible and to calculate the GHG emissions reduction/sequestration potential of those habitats under different management or restoration regimes.

1.2 Report Structure

Chapter 1 is an overall executive summary of the report's main elements, conclusions and recommendations.

Chapter 2 outlines the strategic context for the report and its relationship to The Wildlife Trusts' collective ambitions.

Chapter 3 sets out the purpose and scope of the report.

Chapter 4 explains key terminology relating to GHG accumulation, stocks and flux, how these are calculated and how they might be used in the development of GHG removal, emissions reduction, or offsetting schemes.

Chapter 5 sets out the review methodology.

Chapter 6 provides a detailed literature review of GHG emissions by habitat type.

Chapter 7 uses the information from Chapter 6 to determine the impacts of a range of nature-based solutions on land-use-related GHG fluxes. Four main approaches were looked at:

1. Protecting remaining high quality natural and semi-natural habitats.
2. Reducing emissions caused by the conservation management of high quality natural and semi-natural habitats.
3. Restoring degraded or creating/recreating natural and semi-natural habitats.
4. Reducing emissions from productive agricultural landscapes through better soil, crop and water management.

Chapters 8 and **9** summarise the main conclusions from the report, their implications for The Wildlife Trusts and recommendations for further work.

Table 13 (page 47) and Table 15 (Appendix 1) bring together the EFs derived from the literature reviewed, for different habitats and land uses, to calculate and present conservative estimates of the likely GHG gains that might be expected from implementing a number of different changes.

CHRIS GOMERSALL/2020VISION

1.3 Conclusions

There are three clear headline conclusions to be drawn from this review, two of which relate to the quantity, quality and applicability of available scientific evidence and one of which relates to the interpretation and use of the evidence that is available.

The first conclusion is that **there are too few consistent studies of GHG emissions from all the habitats reviewed, which leads to considerable variation and uncertainty in interpretation of the data**. It would be sensible, in developing practical initiatives intended to reduce land-related GHG emissions or to implement nature-based approaches to atmospheric GHG removal, to take a conservative approach. For its own carbon accounts, The Wildlife Trusts estimate the emissions from and removals by different broad habitat types by using the lowest Emission Factor (EF) values for removals (-ve value closest to zero), and the highest EF values (+ve value furthest from zero) for emissions, identified from the literature reviewed. For the implementation of nature-based solutions, we take the most conservative estimate of the benefit (avoided emissions or removals) likely to be achieved. The values in this review would be appropriate to give potential donors and other funders some initial confidence in their investments, within a "silver standard" (i.e. not necessarily the most stringent standard that can be applied, but the most practicable for Trusts to implement while still giving robust results) approach to GHG emissions reduction and/or removal. They would not yet be appropriate as the basis for a fully validated scheme. For some habitats, current evidence is too varied and inconclusive to confidently recommend any specific EF.

The second conclusion is that **a lot of the variation in the data is because habitats are extremely variable in biotic and abiotic factors over a range of spatial and temporal scales. For some habitat types, it may never be possible to create a single robust emissions or removals factor**. For these habitats, direct monitoring of sites managed with a specific intention of reducing GHG emissions and/or increasing sequestration — especially those going into codified schemes and funding arrangements designed specifically to achieve this — will be required. Additional research could also identify accurate, pragmatic, standardised approaches to the measurement of GHG fluxes, based on suitable proxies (such as satellite remote sensing of moisture content).

The third conclusion is that **despite the limitations and uncertainties in the available evidence, it is possible to identify habitat and land management changes that are, on the basis of best available evidence (reviewed here), likely to generate reductions in GHG emissions or increases in removals.** Conservative estimates of these values are presented in Table 13 (page 47), calculated from flux values given in the literature reviewed. For some of the land management changes where specific values could not be calculated reliably in this way, an alternative approach (using carbon stock values from the literature to estimate annual rates of stock-change) was used to calculate possible emission reductions or increases in removals (summarised in Table 15, Appendix 1).

The values suggested here are likely to be useful as scientifically credible initial estimates of the potential gains that might result from implementing particular changes in land-use and habitat condition.

Further conclusions from the review are as follows:

There are two types of GHG emissions reduction processes that need to be considered when developing a coherent Wildlife Trust approach:

- **Avoided emissions** — from changes in management that reduce emissions but don't necessarily lead to habitats becoming net carbon sinks.
- **Sequestration** — where carbon is sequestered from the atmosphere into storage within the habitat, in one of two ways::
  i. Short-term gains when a habitat changes from one equilibrium state with a lower soil organic carbon content, to another with a higher soil organic carbon content; and
  ii. ongoing sequestration of new CO_2 from the atmosphere into long-term increasing carbon stores.

The currently available evidence reviewed in this study indicates that only three broad habitats deliver ongoing sequestration:

- **Near-natural and pristine peatlands** which sequester small amounts of atmospheric carbon and continue to do this over thousands of years.
- **Woodlands (and forestry, depending on the fate of the harvested product)** which have high sequestration rates in younger growth phases and can store significant amounts of carbon in above-ground living biomass for centuries.
- **Saltmarsh** which continually sequesters carbon into long-term storage through high sedimentation rates.

Conversion of arable and intensive grasslands to extensive species-rich grasslands can lead to a period of sequestration while higher levels of soil organic carbon are accumulated, but this will tail off to a new state of equilibrium, probably within decades.

The restoration of degraded peatlands offers the largest potential for emissions reductions through avoided losses, rather than from new atmospheric CO_2 sequestration.

Ensuring the **protection of high nature value open habitats such as species-rich grasslands and heathlands is essential in preventing the loss of their soil organic carbon stores.** However, maintaining existing high nature-value habitats that are already in good condition (such as by grazing open habitats to prevent succession to scrub and woodland) is unlikely to sequester significant additional amounts of CO_2 from the atmosphere.

Altering management practices in existing high nature value habitats could reduce emissions but there is insufficient evidence to propose reliable estimates of the impact of these.

Altering management practices in arable and intensive grassland systems could lead to substantial emissions reductions, given their widespread extent — particularly where these are on peat.

There is currently **insufficient evidence to determine standard GHG fluxes in hedgerows, scrub, orchards, wood pasture, rivers, streams, floodplains or ponds accurately.**

The review also concluded that the development of a comprehensive Wildlife Trust movement-wide approach to habitats and carbon also needs to address the following two questions:

- How long does it take a habitat to "move" from one GHG emissions state to another?
- How do the GHG emissions change during the period of habitat creation, restoration, or management?

BEN HALL/2020VISION

2 Introduction:
The Wildlife Trusts, Nature's Recovery and the Climate Crisis

The Wildlife Trusts has a new, ambitious, collective Strategy to 2030. It affirms our vision of a thriving natural world, where our wildlife and natural habitats play a valued role in addressing the climate and ecological emergences, and people are inspired and empowered to take action for nature. The three overarching goals of the Strategy are to see:

- Nature in recovery;
- People taking action for nature and the climate; and
- Nature playing a central and valued role in addressing global issues such as climate change

Climate change is one of the biggest threats faced by wildlife in the UK and as such The Wildlife Trusts need to address it. Bringing wildlife and healthy ecosystems back across the land and seas of the UK is the focal purpose of The Wildlife Trusts, but:

a) This cannot be achieved without building resilience to current climate change into ecosystems, and will not be achievable at all if the global climate changes catastrophically.
b) Nature's recovery can make a significant and lasting positive contribution both to coping with the impacts of unavoidable climate change and to reducing future levels of atmospheric GHGs (leading to a stabilising climate).

The state of the natural world and the climate crisis are inextricably linked and both need to be addressed together. The natural world needs to recover so it can cope with the climate change that is already happening, so its continuing decline doesn't make climate change worse and so it can contribute to reducing atmospheric GHG concentrations.

The Wildlife Trusts must work out how they can best make a significant contribution to tackling climate change. This includes making sure that whatever we do and say in relation to it is factually correct, technically sound, and scientifically credible. As a leading environmental organisation, our supporters and allies will expect us to take a principled, effective approach and to demonstrate leadership in this area as well as championing nature's recovery and bringing wildlife back into people's daily lives.

The Wildlife Trusts have a very strong track record in and an increasing commitment to inspiring, empowering and enabling people to take effective action in the places where they live and work.

In 2019, The Wildlife Trusts directly managed 104,000 ha of land across the UK, the Isle of Man and Alderney, investing directly in the protection and management of land for nature. This means that collectively, we are among the UK's top ten land holders by area. The Wildlife Trusts also provided land management advice to other land managers (farmers, foresters, Local Authorities, etc.) that influenced the management of an additional 242,000 ha of land (The Wildlife Trusts, 2021). A further 13,000 ha of land was protected or enhanced for wildlife because of the movement's influence on built development through the local planning system. And the movement had further influence on the way in which land and the marine environment are managed (through advocacy and campaigning work locally and nationally, by providing advice and support, establishing partnerships with like-minded others, etc).

Increasingly, the movement has been demonstrating and showcasing what works and what can be achieved. This close link between direct practical experience of delivering change on the ground and advocating for and supporting others to do likewise has become a characteristic of the way The Wildlife Trusts achieve influence. Given this, it is crucial that our actions and approaches are grounded in good science and that our communications and claims are based on sound practical experience and solid evidence.

The Wildlife Trusts are committed both to securing nature's recovery and to addressing the climate crisis as a central (deeply embedded) part of our work. And the movement will play a significant role in both direct delivery and influencing the attitudes, behaviours, and actions of others. While The Wildlife Trusts are relatively small emitters of carbon dioxide and other GHGs, it is still vitally important that the movement reduces its emissions as far as possible and demonstrates both its commitment to the task and the impact the movement's actions in this area are achieving. (Though not the focus of this report, it is also essential that every Trust understands the risks from climate change to their assets, including reserves, and has in place plans to adapt to climate change.)

Given these things, it is important that The Wildlife Trusts understand (as far as is possible) what our GHG emissions are and what impact our habitat and land management work and advice could be having on atmospheric GHG levels.

3 This Project:
Understanding the Relationship Between Wildlife Habitats, Land Management for Wildlife and Atmospheric Greenhouse Gas Levels

This report was researched and largely written by Dr Tim Thom (Peat Programme Manager, Yorkshire Wildlife Trust), working on secondment to the movement's central charity, the Royal Society of Wildlife Trusts (RSWT). It was commissioned, partly written and edited by Nigel Doar (Head of Science & Research, The Wildlife Trusts) on behalf of The Wildlife Trust movement.

Early drafts of the final report were reviewed and discussed by members of The Wildlife Trusts' Habitats & Carbon Technical Working Group. This included a number of land management and policy specialists from individual Wildlife Trusts with a direct first-hand interest in the carbon benefits of wildlife habitats and their potential to contribute to stabilising the global climate. Early drafts were also shared with a number of external experts[1] who were invited (and in some cases contracted) to comment on the technical, scientific and policy content of the draft report. Their very valuable input was gratefully received and has been incorporated into the final report.

The report was commissioned to give The Wildlife Trusts sufficient technical insight into the scientific evidence that is currently readily accessible, for the movement to make well informed decisions and to make defensible public claims concerning the relationship between habitat creation, restoration and management (on the one hand), and atmospheric GHG levels (on the other). It is intended to provide a reasonable basis for immediate and imminent decisions, communications and advice concerning the GHG-related impacts of habitat creation, restoration, and management in the UK. It is first and foremost for use by The Wildlife Trusts, but may contain evidence, reasoning, interpretation and insights that others would find useful.

The Wildlife Trusts have interests in the GHG budgets of habitats both on land and at sea, including at the coast, where habitats like saltmarsh link the two. For simplicity, and to maintain focus, this paper concentrates on terrestrial, freshwater and saltmarsh habitats with other working groups and partnerships exploring marine systems. We do not therefore consider marine habitats such as kelp or seagrass habitats in this report.

While the primary purpose of the project is to provide an immediately available, credible ("good enough") basis for a wide range of actions, decisions and communications relating to habitats, GHG emissions and sequestration, there may be the potential to further augment the content presented here to become the technical basis of an income-generating accreditation scheme.

It is intended that this paper should make an early contribution to building the evidence needed for that (while recognising that various similar initiatives are developing elsewhere).

Some insights into how current GHG crediting arrangements work and the approaches taken by existing GHG offsetting standards and reporting guidance, such as the GHG Protocol, have informed parts of the approach taken here.

When looking to make informed decisions about habitat change and/or management, with an intention to achieve tangible GHG reduction benefits, four initial pieces of information are needed:

- What are the net GHG emissions from each habitat before and after an intervention?
- Which types of habitat creation, restoration and/or management provide the largest reduction in GHG emitted, captured, and stored?
- How long does it take a habitat to "move" from one GHG emissions "state" to another?
- How do the GHG emissions change during the period of habitat creation, restoration, or management?

The aim of this report is to provide initial answers to the first two of these questions, by carrying out a limited literature review to determine GHG emissions for as many habitats as possible and to calculate the GHG emissions reduction/sequestration potential of those habitats under different management or restoration regimes.

Further steps needed to answer the third and fourth questions are discussed briefly.

This is not a comprehensive literature review, given the limited time and resources available.

This review is intended as a "first filter"— a practical guide to help practitioners working in this space to access, interpret and apply the available scientific evidence in a way that is transparent. It is envisaged that the report will be updated over time as additional evidence becomes available, so the reader should note the report version.

[1] From the UK Centre for Ecology & Hydrology, the University of Manchester and the University of York.

ROSS HODDINOTT/2020VISION

4 Terminology

Quantifying GHG emissions from land use is complex and requires detailed knowledge of several processes which are currently poorly studied and not fully understood.

4.1 Carbon accumulation

Some habitats or land-use types have very high annual carbon accumulation rates (in the form of CO_2 photosynthesised into vegetation biomass), but also return this very rapidly to the atmosphere (such as through ecosystem respiration resulting from rapid decomposition by micro-organisms in the soil). In these cases, only small amounts of carbon may enter long-term storage (largely dependent on soil type and management practices). Agricultural land covers 71% of England's land area and one estimate suggests it stores around 583 MtC in arable soils and 686 MtC in the first 1 m of soils under permanent managed grasslands (Alonso *et al*, 2012). Given their large extent, even the loss of a relatively small proportion through unhelpful management practices may be significant. Even with very short-term temporary carbon storage benefits, better management of crops (including growing new types of crops) and soils in intensive agricultural systems could make a vital contribution to the reduction of GHG emissions from land.

Other land-use types (such as woodlands and forestry plantations) have the potential for very high annual accumulation rates leading to increased carbon storage in the woody living biomass which persists for many decades or even centuries — until the vegetation either dies or is removed and subsequently degrades. Even if little of this carbon is moved into long-term storage in soils these are clearly important ecosystems for inclusion in emissions reduction and GHG removal schemes.

In contrast, some habitats or land-use types have relatively low annual accumulation rates but, because ecosystem respiration through decomposition is very low, most of the accumulated carbon is stored for long periods (very long in the case of peatlands), leading to almost permanent stores of carbon. These are clearly important systems to include in emissions reduction and GHG removal schemes.

Of course, the ideal situation is high annual accumulation rates and large amounts of long-term storage in soils or sediments as, for example, in saltmarshes. If these systems can be properly maintained or restored they provide a great opportunity for significant emissions reductions and carbon sequestration.

For all these habitats or land-use scenarios, it is very easy to damage or degrade carbon sinks and stores to such an extent that they begin to release carbon from storage into the atmosphere, including through climate change impacts themselves like increasing wildfire risk. For some habitats this can lead to substantial releases of long-term carbon stores (e.g. damaged peatlands).

4.2 Carbon stocks

The overall land-based carbon stocks are made up of several components as follows:

1. Above ground biomass.
2. Below ground biomass.
3. Litter — the layer of dead and decaying organic matter that lies on the soil surface.
4. Soil organic carbon— this is the largest carbon stock in the UK, holding approximately 95% of land carbon (Ostle *et al,* 2009).

As discussed above, the size and turnover of these stocks will determine long-term carbon storage potential.

FERGUS GILL/2020VISION

4.3 Carbon fluxes

Carbon flux is a measure of the amount of carbon that is exchanged between a carbon stock (stored within a habitat) and the atmosphere.

There are several ways to measure and report carbon fluxes. These often cause confusion, with different types of measure often being compared incorrectly with each other. These are defined as follows:

1. Carbon is accumulated in land as a result of the conversion of carbon dioxide (CO_2) from the atmosphere into plant biomass, by photosynthesis. This is known as **Gross Primary Productivity (GPP)** and is usually expressed in units of Carbon per area per year.
2. All plants also respire, through a process known as **autotrophic respiration (R_A)** and so also emit CO_2 back to the atmosphere.
3. The balance of CO_2 exchange between these two processes is known as the **Net Primary Productivity (NPP)**, which is calculated as **GPP-R_A** and is also expressed in units of Carbon per area per year.
4. In addition, all the micro-organisms and animals that live in the land or habitat (largely in the soil) need to be accounted for. These consume the plant biomass and emit CO_2 and CH_4 through another form of respiration — **heterotrophic respiration (R_H)**. This is accounted for by calculating **NPP-R_H**, to give **Net Ecosystem Exchange (NEE)**, which in some cases is referred to as **Net Ecosystem Production (NEP),** again expressed in units of Carbon.
5. A further measure takes account of CO_2 losses due to periodic disturbances, **D_i**, such as fire or harvesting for food, or grazing (in the case of anthropogenic habitats). Harvested products that are converted into long-term construction products are not included here as they are not lost to the atmosphere. This **Net Biome Exchange (NBE)** or **Net Biome Production (NBP)** is calculated as **NEE (or NEP)-D_i** and expressed in units of Carbon as before.
6. **D_i** has an impact on the carbon emissions or removals resulting from a particular land use. If, for example, harvested wood products (and the carbon they contain) are embedded in buildings, they could be considered as part of the emissions reduction. If on the other hand, crops are consumed by livestock, then the carbon is returned to the atmosphere relatively quickly as CO_2 (through respiration) and as CH_4 (through enteric fermentation) and has not therefore contributed to emissions reduction.
7. Finally, there are several other pathways for carbon to be lost from the land to the atmosphere and this is taken account of in the **Net Ecosystem Carbon Balance (NECB)** which is calculated from **NEE (or NEP) -D_i-DOC-POC-VOC-methane** and expressed in units of Carbon where **DOC = dissolved organic carbon**, **POC = particulate organic carbon** and **VOC = volatile organic carbon**. Some habitats will have more or less of these pathways and some will have none of them.

In addition to these land-based fluxes, the carbon emitted because of management activities may need to be considered. For example, if management requires the intensive use of fuel in harvesting machinery this reduces the benefit of the land-based sequestration.

PHILIP PRECEY

4.4 Other GHG fluxes

Finally, CO_2 and CH_4 are not the only types of GHG emitted as a result of land-use, land management and land-use change. The most potent of the others is nitrous oxide (N_2O) and agricultural systems also use inorganic fertilisers, so the GHG "costs" of the manufacture of the fertiliser (in particular) may also need to be a consideration, along with other carbon costs of land management.

4.5 Calculating overall GHG fluxes

All the GHG fluxes discussed above need to be taken into account when trying to calculate the balance of GHG emissions and reductions within a given land-use or habitat.

For CO_2-derived pathways this is usually considered to be the same as the NEE (or NEP) or, if disturbances are added in, the NBE (or NBP), but expressed in tonnes of CO_2 by multiplying by 3.6667 (the ratio of the molecular masses of CO_2 and C = 44/12). Fluxes for CH_4 and N_2O are calculated separately, usually in kg CH_4-C ha^{-1} yr^{-1} and kg N_2O-N ha^{-1} yr^{-1}).

To calculate the total impact of GHG emissions from a system, the CO_2 equivalent values of each gas are calculated by multiplying their amounts by a weighting that takes account of their different warming effect relative to a pulse of CO_2 over a 100-year period — their **Global Warming Potential (GWP)**[2]. For the purposes of this report, the values used for CH_4 and N_2O are 25 and 298 times that of CO_2, respectively, as these are the values used by the UK GHG Inventory. Internationally, the IPCC periodically revises these values. The value used by IPCC for CH_4 in their 5th Assessment Report has gone up to 28 (based on direct warming impacts) or 33 (incorporating the indirect effect of warming by CH_4 on CO_2) (Myhre *et al*, 2013).

The emissions for a habitat can then be expressed by adding together the CO_2 eq for each of the relevant emissions/accumulation pathways. Defining this in terms of unit area over time (usually annual) gives an emissions rate often referred to as a combined **Emissions Factor (EF)**. The convention is that a negative combined EF value indicates a GHG-sequestering system while a positive combined EF value indicates a GHG-emitting system, over the given time period.

4.6 Using GHG stocks and/or fluxes for Offsetting and/or Emissions Reductions Schemes

The basic premise of a land-based GHG offsetting, removal and/or emissions-reduction scheme is that the activity should reduce GHG emissions beyond what is already happening against a baseline condition. There are several ways this could be achieved in the wildlife conservation and land management sectors.

4.6.1 Changes in carbon stocks

For some habitats or in agricultural systems, the biggest loss of GHG is carbon losses from soil organic matter, largely driven by increased cultivation and land-drainage. This can be due to direct soil erosion followed by oxidation or, in the case of peatlands, due to oxidation of exposed peat that was formed under anaerobic conditions. If these losses can be reduced or prevented, the "avoided losses" count as GHG emissions reductions.

Reducing emissions from damaged or degraded habitats is probably one of the most important actions for reducing land based GHG emissions.

Taking this approach a step further, if a change in land use or restorative management of a degraded or anthropogenic habitat brings about an increase in stored carbon stocks, the difference in carbon storage between the original state and the new state represents GHG sequestration, if it is permanent. For example, a change in agricultural practices or conversion to a semi-natural habitat that leads to a permanent increase in soil organic matter would have increased the long-term carbon stock (by becoming a GHG sink during the transition). If this is also combined with a semi-permanent increase in biomass storage (for example in conversion to woodland or forestry plantation), an even larger stock (or sink) can result.

Over time, however, most habitats reach an equilibrium state where natural sequestration and natural emissions of carbon are equal. At this point they can no longer be used in carbon emissions reduction or removal schemes as they are no longer storing any additional GHG each year. Reaching this equilibrium may be relatively rapid, for example in agricultural grasslands. In other habitats, such as new woodlands, this equilibrium can take centuries to be reached so they are, in essence, long-term carbon sinks even if the sequestration rate slows down over time.

[2] Even this gives only an approximate picture, as steady emissions (i.e. most ecosystem fluxes) have a different warming impact than pulsed emissions.

It is worth noting here that ongoing management of the *status quo* in existing habitats that are already at a GHG neutral equilibrium[3] could not reasonably be used in any emissions reduction/removal/offsetting scheme. The exception might be if it could be proven that ongoing direct management is essential to prevent an otherwise unavoidable increase in GHG emissions from the stock in question, and that financial support from the particular scheme is needed to sustain it. This combination of circumstances seems likely to be exceptionally rare.

4.6.2 GHG fluxes

One way of quantifying reductions in an emissions reduction scheme is to measure the carbon stock present in a habitat before and after an intervention and to count the difference between the two as the overall resulting GHG removals.

However, it may not be easy to quantify all the carbon stocks in a system properly. In addition, while some parts of the system may be sequestering CO_2 into plant biomass, other parts may be simultaneously emitting more powerful GHGs such as CH_4. The overall GHG impact of land use is determined by a combination of the amount and type of gases that are emitted to and removed from the atmosphere and their relative warming potential. In peatlands, for example, when they are in good condition, they sequester CO_2 over time which, due to low rates of decomposition, leads to a long-term and increasing carbon store. The overall (beneficial) impact of this on the warming potential of atmospheric GHGs is countered by the release of smaller amounts of CH_4 emitted through anaerobic respiration.

Because of its higher Global Warming Potential, this CH_4 makes the effective net value of the peatland GHG sink extremely small (less than 1 $tCO_2\ ha^{-1}yr^{-1}$ for even the most natural and sequestering of fens and bogs and even less for rewetted modified bog and near natural bog). Overall, this means that only a small amount of net GHGs is sequestered each year in a typical hectare of peatland even if it is stored for long periods. Understanding the different pathways (fluxes) of GHGs in a land management system and their differing global warming potentials is essential to understanding the net emissions from or sequestration to a particular land parcel.

4.6.3 A GHG management hierarchy

One of the first actions for any carbon-focused land management strategy is to maintain or prevent the loss of the carbon store. Most high-value natural or semi-natural wildlife habitats contain more stored carbon than habitats resulting from land uses with high levels of human intervention. Therefore, by maintaining these habitats in good condition and preventing their conversion to more intensive uses, most Wildlife Trusts are already helping to store significant stocks of carbon. This management cannot be "marketed" as part of an emissions reduction or carbon removal scheme because maintaining the *status quo* neither reduces emissions nor increases sequestration.

A carbon emission reduction or removal scheme needs to demonstrate a change in land management that has brought about additional emissions reductions or removals.

Here, we suggest that a hierarchy of priorities for achieving this could be as follows:

1. Restoration of existing land-use types or habitats to at least a carbon neutral state and, where possible, to one that sequesters carbon into long-term, essentially permanent stores (millennial timescales, e.g., peatlands).
2. Restoration of existing land-use types or habitats to at least a carbon neutral state and, where possible, to one that sequesters carbon into long-term, near-permanent stores (centennial timescales, e.g., woodlands).
3. Restoration of existing land-use types to at least a carbon neutral state (e.g., some types of high nature conservation value grasslands).
4. Conversion of carbon-emitting anthropogenic habitats to other habitats that are carbon-neutral or sequestering (e.g., converting drained lowland arable farmland to floodplain).
5. Implementing best practice land management activities or growing alternative crops to reduce emissions from anthropogenic land uses, ideally to zero (e.g., farming wetland crops on previously drained land or changing to no-till practices).

[3] That is they are not emitting any GHG and are unlikely to change to a sequestering state as a result of management action

5 Methods

5.1 Habitat/land-use priorities

Some work on prioritising the habitats that might be of primary concern to The Wildlife Trusts had already been undertaken (Prior, 2019) so this review used that prioritisation as its starting point and chose to keep its focus on terrestrial and freshwater habitats (plus saltmarsh — see below).

Marine habitats such as kelp, seagrass beds, maerl beds, biogenic reefs, cold-water coral reefs, shellfish beds or marine sediments are not considered here due to resource constraints and because the evidence base is changing quickly as more studies are focussing on these habitats at present.

Saltmarsh has been included in this review, as the main human impacts influencing its location and extent are essentially terrestrial (conversion to agricultural land through drainage and land reclamation). The techniques, licences, and permissions necessary to secure effective saltmarsh creation also relate largely to a terrestrial regulatory regime.

5.2 Stocks and fluxes

The initial brief for this report was to attempt to find scientific data on annual fluxes to establish "Emissions Factors (EFs)" that summarise the net annual CO2 equivalent GHG exchange with the atmosphere for the widest range of habitats possible. See Section 3, above.

After carrying out the initial literature search and feedback from reviewers of the first drafts of this report it was apparent that there was too little data relating directly to EFs for most habitats. To account for this, a second method was also employed for those habitats with little data on fluxes. This additional method looks at whether EFs can be inferred from evidence of changes in carbon stocks within a habitat between two points in time (which we have termed "the stocks approach").

5.3 GHG flux literature search

Various academic literature search databases (e.g., Web of Science and Science Direct) were used to search for literature from 2015-2020, using the search terms listed below. The resulting list of recent key references from these searches was combined with grey literature and other documents already known to the authors. The reference lists from those papers and reports were then used to work backwards in time rather than carrying out further database searches. For reviews, original sources were checked as thoroughly as time allowed. Various combinations of the following keyword searches were used: GHGs, Emissions Factors, GHG, LULUCF, AFOLU, Natural Climate Solutions, Nature based solutions, Carbon sequestration, Land-use.

This initial search produced over 1,000 hits. These were further filtered by relevance, initially of the title and then of the abstract, to give 120 references. These were then read in detail to search for data that could be used to derive Emissions Factors (EF) according to the methods outlined in section 5.4.

Only studies that were relevant to habitats known to occur in the UK were used, to avoid the application of emissions data from habitats from different climatic zones or with very different management histories and approaches.

5.4 Methods for the calculation and selection of GHG Emissions Factors

For the purposes of this report, Emissions Factors were expressed for each GHG in their original units and combined as tonnes CO_2 equivalents per hectare per year (t CO_2 eq ha^{-1} yr^{-1}). To do this an iterative process was taken when reviewing each paper or report:

1. Net CO_2 equivalent GHG data available and already presented in these units — use without further analysis.
2. Net GHG CO_2 equivalent GHG data available but not presented in these units — convert and then use.
3. Individual GHG CO_2 equivalent GHG data available covering all possible pathways — converted to these units and added together to give net GHG CO_2 equivalent.
4. Individual CO_2 (NEE for natural habitats, NEP for harvested), CH_4, N_2O plus Di data available covering all possible pathways — converted to correct units and then to net GHG by multiplying by Global Warming Potential and adding together.
5. Where studies did not have data of sufficient quality or certainty to cover all the important GHG pathways Emissions Factors could not be calculated.
6. Where net GHG values for each habitat covered a wide range from emission to sequestration (i.e., high levels of uncertainty) Emissions Factors could not be calculated.

5.5 Methods for the estimation of changing GHG emission and removal rates resulting from changes in land use or land management

Two approaches have been taken to calculating the carbon benefits of land use change here:

1. '**Flux measurement' approach** — Potential GHG gain (reduced annual GHG emissions and increased annual GHG removals) associated with different habitat and/or land-use changes were calculated as the difference between Emission Factors for the respective habitats before and after conversion. On the whole, it has been possible to use some form of Emission Factor to produce conservative estimates of likely changes in annual emissions and removals resulting from a range of changes in land management and habitat condition.
1. **'Stock change' approach** — As an alternative to the 'flux measurement' approach above, and in an attempt to fill some of the gaps left by it, implied changes in annual GHG emission and removal rates were also calculated for a number of habitat, land use and/or land management changes relating to "open" terrestrial habitats. Carbon stocks and rates of stock-change indicated in the literature reviewed were compared between two different points in time, before and after the change in land use. These were averaged out across the time over which the change occurred. This approach has not been applied more widely as there is too much uncertainty around, and variability in, stock estimates for other habitats.

Where the evidence reviewed suggests ranges of possible Emission Factors for a particular land-use or habitat state, the low and high limits of these have been used to calculate conservative estimates of implied changes in annual sequestration and/or emission rate between the starting land-use and the alternative (final) land-use. For example, where a starting land-use is emitting carbon (a positive value by convention) and the final-use is also still emitting, but at a lower rate, the gain would be in avoided emissions.

Where a starting land use was sequestering carbon (a negative value by convention) and the end-use was sequestering more, the gain would be the additional sequestration brought about by the land-use change. Some gains may include an element of each. On the other hand if the starting land-use was emitting less or sequestering more than the final land-use the change in management would be leading to increased emissions and/or reduced sequestration, i.e. no (or negative) gain.

For the 'stock-change' approach, the difference between the carbon stocks within the starting land-use and within the end land-use gives the stock change. As with the Emission Factors in the 'flux measurement' approach, low and high values have been calculated — in this case, for the total stock changes indicated as likely to occur as a result of each land use or habitat change.

To convert these total numbers into implied annual rates of net carbon flow (comparable with the 'flux measurement' approach above) we need to know how long each change is likely to take. A few studies provide this information (in which case implied annual rates of net carbon flow have been calculated directly from the published information).

For other changes, where the stock-change period cannot be deduced directly, a standard rate of 20 years has been used. This is based on the guidance in the GHG Protocol Agricultural Guidance (www.ghgprotocol.org) where the GGP Product Standard requires that "in the context of land use change, [changes should be amortized over] 20 years or the length of one harvest, whichever is longer". As more studies are carried out, these rates can be refined at a future point in time.

Across all estimated changes in the annual rate of GHG flux, comparisons between 'before' and 'after' rates have been based on the land use definitions and habitat classifications used by the researchers carrying out the primary research. These vary between studies, leading to a number of apparently similar transitions being included in the analysis. For precise definitions of the land uses and habitat classifications used, please refer to the original reference sources.


BARNEY WILCZAK

6 Carbon storage and greenhouse gas emissions by habitat

6.1 Terrestrial "open" habitats (not on peat)

6.1.1 Arable & Intensive Grasslands

Agricultural land covers 71% of England's land area and stores around 583 MtC in arable soils and 686 MtC in the first 1 m of soils under intensively managed grasslands (Alonso *et al*, 2012). The Climate Change Committee (2020) recommend that about 20% of agricultural land will need to be made available by 2050 for the reduction of GHG emissions.

As productive systems, crops (and intensive annual grass ley monocultures) can contribute a significant amount of CO_2 sequestration through the production of the crop or grass biomass. However, much of this is consumed relatively quickly through harvesting and consumption of crops and/or products from grazing livestock and consequently emitted back to the atmosphere. This does not contribute greatly to GHG sequestration or emissions reduction (if at all).

Most of the stored carbon in agricultural systems is in the soil. The impacts of land management on whether agricultural systems sequester to or release carbon from the soil store will depend on many different factors, making it very difficult to determine standardised measures of carbon stocks or sequestration rates. One of the most important factors will be the organic content of the soil. At one end of the spectrum are peatland soils which are discussed in a later section of this report, but some non-peatland soils also contain significant amounts of carbon. At the other end of the spectrum will be soils that have very little carbon content so will not contribute greatly to CO_2 fluxes.

The main losses of carbon from agricultural soils are through oxidation from topsoil and direct erosion due to disturbance from agricultural operations. The amount of carbon loss will be very variable depending on the machinery or livestock used. In addition, CO_2 is not the only GHG flux that needs to be considered in agricultural systems. Modern intensive agriculture uses large amounts of inorganic fertiliser. Denitrification of this can emit significant quantities of N_2O and, if the CH_4 produced by enteric fermentation in grazing livestock is also considered, then intensive agricultural systems can be significant sources of GHG emissions.

6.1.1.1 Arable

According to Bradley *et al* (2005), land in the UK under arable management stores around 120 tonnes of Carbon per hectare in the top 100 cm and accounts for 16% of carbon stored in UK soils. This is, however, very variable, depending on the organic content of the soil. In Scotland, with the dominance of peaty soils, the average soil carbon content is much higher, but the extent of arable land is less. Muhammed *et al* (2018) showed, using the Roth-C model, that carbon stocks under arable had declined over a long period but this loss had slowed between 1970 and 2010 to a rate of 0.08 t C ha^{-1} y^{-1}.

There are few studies of emissions from agricultural soils but when all the GHGs are accounted for, the figures summarised in Table 1 (for mineral and organo-mineral soils) and Table 4 (for peat soils) show that arable systems typically emit somewhere between 1.6 and 37.6 t CO_2 eq ha^{-1} yr^{-1}. This is clearly highly variable and depends on a wide range of factors including the original soil type, the crop, how it and the soil are managed during the crop cycle, the stone content of the soil, inputs of organic materials and the use of inorganic fertiliser (which is widely used in large quantities). The impact of N_2O on GHG fluxes has a particularly important impact in arable systems.

At present the evidence needed to develop standardised emission factors for arable systems is limited because their management is so varied. A more reliable approach would be through direct measurement of carbon stocks before, during and after an intervention on an individual field or farm. This is the approach being taken through, for example, the Farm Carbon Toolkit.

6.1.1.2 Agricultural grasslands

Bradley *et al* (2005) proposed that agricultural grasslands store an average of 160 tonnes of Carbon per hectare in the top 100 cm of soil and accounts for 29% of carbon stored in UK soils; significantly higher than under arable soils. As with arable soils, this is very variable; the average is significantly higher in Scotland due to the prevalence of peaty soils. It is perhaps unsurprising that grassland soils contain more carbon than arable ones because they are less disturbed during crop preparation and harvest.

As tables 1 and 4 show, those studies that exist for agricultural grasslands give widely varying emission factors, from -5.0 to 27.5 t CO_2 eq ha^{-1} yr^{-1}. This variation depends on a wide range of factors including the original soil type, the species mix, the cutting or grazing regime, the stone content of the soil and the inputs of organic materials and inorganic fertiliser. There is also the additional impact of enteric fermentation in ruminant grazing livestock, which converts carbon in the plant biomass consumed into CH_4 emissions.

If the impact of N_2O from artificial fertiliser and CH_4 from enteric fermentation is not included, the studies included in Table 1 suggest agricultural grasslands on mineral soils may sequester carbon at a rate of -0.19 to -8.32 t CO_2 eq ha^{-1} yr^{-1}.

The wide variability in the current evidence makes it impossible to develop standardised emissions factors for agricultural grasslands. It may be more appropriate to use site or farm-based direct measurement of soil carbon stock change instead, combined with developing a better understanding of the impacts of artificial fertiliser and enteric fermentation.

6.1.2 High Nature Value grasslands

The kinds of grasslands familiar to those who visit Wildlife Trust nature reserves are usually very species-rich in plants and invertebrates. They are usually very unproductive in biomass compared to arable or improved grasslands but receive little or no inputs and are managed through light grazing and/ or mowing and/or sometimes prescribed burning. There is often an ongoing programme of cutting or burning to hold back succession to scrub and woodland, to maintain their grassland biodiversity value. A proportion of the biomass in many of these grasslands is stored in an organic layer in the soil, or bound into the mineral subsoil. Overall, these grassland systems store more carbon than intensive agricultural systems (Table 1). In fact, the diversity of plant species — in particular, the presence of legumes in the sward — increases the GHG sequestration potential of semi-natural grasslands (DeDeyn *et al*, 2011). In some cases, e.g., grasslands on very thin chalk and limestone soils, there is very little organic storage in soils but there is clearly a lot of carbon in mineralised form in the underlying bedrock. Disturbance or erosion of this could lead to carbon losses.

Smith (2014) showed that grasslands are not a perpetual sink for carbon, but reach a new equilibrium following a change in management. A very lightly managed semi-natural grassland with few inputs is therefore likely to be close to carbon neutral and will not be sequestering GHG significantly. However, it could take up to 100 years for a grassland to reach a new equilibrium with a higher carbon stock, after a sustained change in management (Smith, 2014). During this time, the habitat is sequestering carbon (albeit at a gradually decreasing rate over time). Allard *et al* (2007), working on French upland sites showed that changing from intensive to extensive grassland management reduced N_2O and CH_4 emissions but also reduced CO_2 uptake so that overall annual carbon sequestration declined over time.

The limited number of studies summarised in Table 1 seem to confirm that where grasslands are maintained in an "undisturbed state" they can be considered to be carbon neutral (Sozanska-Stanton *et al*, 2016). Given all the interventions that may be needed to maintain grasslands and prevent succession, it would be relatively easy to tip grasslands over the threshold from sinks to sources of GHG, as also appears to be confirmed by the figures in Table 1 for "disturbed" grasslands (Sozanska-Stanton, 2016). The small number of studies suggests that their interpretation needs to be treated with caution.

Overall, such evidence as there is currently suggests that conversion from more intensively managed grassland habitats with lower soil carbon to more species-rich extensively managed habitats with higher soil carbon will, for a time, sequester GHG. Changes in long-term management practices may help to reduce GHG emissions from existing high nature conservation value grasslands, but once at equilibrium they are unlikely to turn into long-term significant GHG sequestering systems.

Of course, the biodiversity value of species-rich grasslands may dictate any management regime in the long-term and outweigh any GHG impacts. Given the limited distribution of these types of grasslands in the UK, they are unlikely to be a major source of GHGs.

6.1.3 Heathlands

Most heathlands in the UK are artificial — the result of centuries of human intervention. The area of heathland has declined since the 19th century, with the 20% of lowland heathland that still remains confined to small patches which are largely managed by conservation organisations. In the uplands there are still significant areas of heathland which are kept open in character through intensive management with grazing and burning.

The Countryside Survey (Emmett *et al*, 2010) suggests that heathlands store significant amounts of carbon (90t ha^{-1}) in the top 15 cm of soil (Table 1). Carbon sequestration in or emissions from heathlands are dependent on a range of factors including soil type, stage of vegetation growth and management interventions. There are very few GHG flux studies on heathlands (Table 2). Quin *et al* (2015) studied CO_2 fluxes but it was possible that one of their study sites was a peatland (organic layer 30-60 cm deep) so other GHGs such as methane would need to be accounted for. Sozanska-Stanton *et al* (2016) show that, generally, heathlands that are not on deep organic soils are GHG neutral but if they are disturbed by management, grazing, or burning they become a net GHG source, largely due to carbon losses from the soil.

Land Use	Soil Carbon Stocks (t C ha⁻¹)	Sources	Net GHG flux (t CO_2 eq ha^{-1} yr^{-1})	Sources
GB Arable & horticulture	0-15 cm depth = 47	Emmett *et al* (2010)		
GB Improved grassland	0-15 cm depth = 67			
GB Neutral Grassland	0-15 cm depth = 69			
GB Dwarf shrub heath	0-15 cm depth = 90			
UK Arable soils	0-30 cm depth = 80 (70-120) 0-100 cm depth = 120 (110-150)	Bradley *et al* (2005)		
UK Agricultural Grassland soils	0-30 cm depth = 100 (80-160) 0-100 cm depth = 160 (130-230)			
High plant diversity, extensively managed	0-100 cm depth = 414	Ward *et al* (2016)		
Intermediate plant diversity, intermediate level of management	0-100 cm depth = 446			
Low plant diversity, intensive management	0-100 cm depth = 403			
UK Arable — organo-mineral and mineral soils			1.62	Benetzen *et al* (2016)
UK Intensive Grassland — organo-mineral soils			-0.19 to -8.32 (-4.99to 5.25)*	Benetzen *et al* (2016), Chang *et al* (2015), Jones *et al* (2016)
Lowland calcareous grassland — "undisturbed"			0.01 to 0.04	Sozanska-Stanton *et al* (2016)
Lowland calcareous grassland —"disturbed" by management			0.66 to 4.96	
Upland calcareous grassland — "undisturbed"			0.01 to 0.04	
Upland calcareous grassland —"disturbed" by management			0.05 to 3.42	
Lowland meadow — "undisturbed"			0.01 to 0.04	
Lowland meadow —"disturbed" by management			0.17 to 4.40	
Upland meadow — "undisturbed"			0.04	
Upland meadow —"disturbed" by management			2.18	
Purple moor-grass — "undisturbed"			0.02 to 0.04	
Purple moor-grass —"disturbed" by management			1.15 to 6.22	
Lowland dry acid grassland — "undisturbed"			0.01 to 0.04	
Lowland dry acid grassland —"disturbed" by management			0.02 to 4.41	
Coastal machair — "undisturbed"			0.01	
Coastal machair —"disturbed" by management			0.09	
Mountain heath — "undisturbed"			0.01	
Mountain heath —"disturbed" by management			0.04 to 0.57	
Lowland heath — undisturbed"			0.02 to 0.20	
Lowland heath —"disturbed" by management			1.31 to 24.7	
Upland heath — undisturbed"			-5.60 to 0.20	
Upland heath —"disturbed" by management			0.35 to 23.2	

Table 1: Summary of review of soil carbon stocks and GHG fluxes from "open" non-peat land-uses. Positive numbers = emission, negative numbers = sequestration. *includes enteric fermentation and artificial fertilisers.

6.2 Rivers, streams, open water, and wetland (not on peat)

There are very few studies of the GHG emissions status of non-peatland "wet" habitats. Carbon cycling in aquatic systems is also poorly understood, making it difficult to determine reliable estimates of GHG emissions from these ecosystems.

Aquatic systems receive inputs of carbon from two sources — directly (as CO_2 from photosynthesis in algae and plants) and indirectly (as sediment washed in from surrounding land in the form of particulate and dissolved organic carbon — POC and DOC respectively).

6.2.1 Headwater streams

This review found no useful studies on carbon or GHG cycling in non-peat headwater streams.

6.2.2 Lowland rivers and streams

Rivers and streams used to be considered long-term stores of carbon as they transport POC for burial in downstream lakes, estuaries, or continental shelf seas. More recent evidence from Worral *et al* (2016) shows that most carbon entering rivers and streams is mineralised at some point during the transport phases and emitted back to the atmosphere, leading to smaller amounts reaching potential burial sites in deep water sediments. Taken on its own, the study suggests that these emissions should, therefore, be taken into account when quantifying emissions from land use in the surrounding catchments. Worral *et al* (2016) calculated that 1 tonne of fluvial POC leads to a median emissions factor of 6.6 t CO_2 eq yr^{-1}. Consequently, ever-increasing erosion into rivers and streams due to surrounding intensive land-uses means that rivers and streams may now provide an additional significant pathway for increasing emissions of GHGs driven by soil erosion caused by land management.

It follows that this fluvial loss of GHG to the atmosphere generated from POC from soil erosion should be factored in when determining GHG losses from terrestrial habitats, according to the Worral *et al* (2016) study.

6.2.3 Floodplain

In England and Wales, floodplain soils cover an area over 431,000 hectares (Burden *et al*, 2016). Where rivers can deposit their sediment loads into undisturbed floodplains there may be significant carbon storage potential. However, floodplains are also vulnerable to disturbance from agricultural activities and, in the UK, 42% of floodplains are no longer connected to rivers, largely due to flood defence infrastructure and development. One estimate suggests that eighty-two percent of floodplains in England and Wales has been drained for agriculture and development (Burden *et al,* 2016).

Floodplain is not a single habitat, but a combination of several habitats, with various land-uses, including intensively managed arable land, fen, bog, and grazing marsh.

Soils present will vary from mineral to deep peat. It is, therefore, not possible to provide generic values for carbon storage or GHG emissions for "floodplain" as these will be highly dependent on underlying soils, land management, connectivity, levels of alluvial deposition, inundation frequency and losses of soil through erosion.

Swinnen *et al* (2020) reported high carbon stocks of 323.27 ± 12.58 t C ha^{-1} in the floodplain of the River Dee in Scotland (Table 2) and Walling *et al* (2006, reported in Gregg *et al*, 2021) suggested floodplains in southern England were a carbon sink of -0.692 to -1.143 t C ha^{-1} y^{-1}. However, this review found no studies that provided full flux estimates for all GHG emissions from floodplains and it is possible that N_2O and CH_4 emissions from inundated floodplains may negate any GHG sink resulting from carbon sequestered in alluvial sediments.

6.2.4 Standing water

6.2.4.1 Upland lakes

This review found no studies of GHG cycling in upland lakes and tarns.

6.2.4.2 Lowland lakes

Most lakes are super-saturated with CO_2 and are net CO_2 sources to the atmosphere. Maberley *et al* (2012) showed that the main source of this excess oxidisable carbon entering productive lakes was direct input from the surrounding landscape. Increased nutrient inputs also lead to algal blooms and the decomposition of these can produce to significant emissions of CO_2 and CH_4.

However, some particulate organic carbon may be buried in lake sediments for centuries. Scott (2014) recorded burial rates of -0.68 t C ha^{-1} y^{-1} which would be a GHG sequestration rate of -2.5 t CO_2 eq ha^{-1} y^{-1} (Table 2), making them a large potential GHG sink, if all the POC coming off the land is buried. However, Worrall *et al* (2016) suggest this is not the case.

6.2.4.3 Ponds

The term pond is used for a wide variety of small waterbodies, including natural and artificially constructed pools for livestock drinking, intercepting floodwater and buffering against sediment loss or pollutants. A wide variety of management approaches across such a wide range of pond-types makes them very difficult to quantify in terms of GHG emissions.

None of the studies looked at as part of this review (Table 2) look at the whole GHG cycle; they focus mainly on carbon burial in pond sediments. Gilbert *et al* (2021) recorded carbon densities in pond sediments of between 30.1 and 59.2 t C ha^{-1} depending on the type of vegetation surrounding the pond and the amount of annual drying out.

Gilbert *et al* (2014) estimated carbon burial rates in ponds of -5. 5 t CO_2 eq ha^{-1} y^{-1}, which is similar to Taylor *et al* (2019) who recorded rates of -5. 2 t CO_2 eq ha^{-1} y^{-1}. However, Gilbert *et al* (2016) showed that ponds can rapidly switch from sink to source as they dry out. Eutrophic ponds with a high organic content may also generate CH_4 and N_2O, which may negate some of these high burial rates.

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Land Use	Soil Carbon Stocks (t C ha^{-1})	Sources	Net GHG flux (t CO_2 eq ha^{-1} yr^{-1})	Sources
Floodplain	323 ± 13	Swinnen *et al* (2020)		
Natural Pond in Naturalistic vegetation	30	Gilbert *et al* (2021)		
Natural Pond in Arable vegetation	30			
Natural Pond in Pasture vegetation	47			
Natural Pond in Dune vegetation	59			
Lake			-2.49	Scott (2014)
Floodplain			-2.49 to -4.19 (but doesn't include CH_4 or N_2O fluxes)	Walling *et al* (2006, reported in Gregg, *et al* 2021))
Ponds			-5.21 to -5.46 (but uncertain on CH_4 or N_2O fluxes)	Gilbert *et al* (2014); Taylor *et al* (2019)

Table 2: Summary of review of soil carbon stocks and burial rates in floodplain and standing water environments. Positive numbers = emission, negative numbers = burial.

6.3 Trees & Woodlands (not on peat)

This section is a summary and interpretation of the key findings of the Gregg *et al* (2021) report which was conducted by Natural England at the same time as this review was being conducted. There are several different "woody" habitats in the UK:

- Woodland
- Wood Pasture
- Hedgerows
- Orchards
- Scrub

6.3.1 Existing Native Woodland

Woodlands have the potential to sequester substantial amounts of carbon from the atmosphere, particularly during their early stages of development. Substantial amounts of carbon are stored in woodland soils (Table 3) but, in addition, unlike most other habitats, woodlands also store a substantial amount of carbon in the living above-ground biomass (e.g. tree trunks and branches). Under the right conditions, this can be stored for centuries. There is also long-term storage in dead wood and organic soil matter which can take decades to decompose. Long-term studies are sparse, but Thomas *et al* (2011) shows that ancient woodland can still be an ongoing GHG sink of around -4.4 to -6.6 t CO_2 eq ha^{-1} yr^{-1}, as it recovers from past management intervention or natural disturbance (Table 3).

6.3.2 Forestry and new woodland planting

In forestry and woodland creation schemes four main phases can be recognised where carbon sequestration rates differ significantly.

In the establishment phase, the process of planting trees may lead to carbon losses and increased emissions (such as emissions in the supply chain, and from direct soil disturbance during planting), which are not compensated by the early growth of small trees (though it is possible to reduce loss of soil resulting from drainage and also that other vegetation colonising the open spaces between the trees may compensate for this carbon loss). During the main growth phase for new woodland creation (which takes 20-30 years to achieve) primary productivity vastly outweighs respiration from decomposition, so large amounts of carbon are sequestered in the rapidly growing trees and the new woodland soil, easily compensating for the early losses. A study by Greig (2015) showed GHG sequestration rates of -2.2, -1. And -1.9 t CO_2 eq ha^{-1} yr^{-1} for Sitka spruce, Norway spruce and broadleaves respectively between 0-10 years after planting. This increased rapidly during the main growth phases to peaks of -24.8, -21.6 and -16.2 t CO_2 eq ha^{-1} yr^{-1} at years 40, 30 and 30 after planting for Sitka spruce, Norway spruce and broadleaves, respectively.

Conifers are normally harvested by clear-felling at about 40 years old, at which time sequestration will be reset and the phases begin again. However, if all of the harvested timber is not built into long-term construction products the sequestration gains will be lost back to the atmosphere. Broadleaved trees are generally left to grow on for longer and will continue to sequester carbon, but over time the build-up of organic matter in the soil leads to increased decomposition and associated respiration.

This, combined with the slower growth rate of the trees as they mature, slows down sequestration. The sequestration rates in the broadleaved trees studied by Greig (2015) dropped to just -1.42 t CO_2 eq ha^{-1} yr^{-1}.

However, this process can take centuries to reach an equilibrium state and mature woodlands go on sequestering carbon for considerable time periods. Gregg *et al* (2021) argue that across most of England native broadleaved woodlands can sequester as much carbon as conifer plantations, or even more. However, in wetter more acidic soils in the north and west, non-native conifers may sequester more (although see section 6.4 for the impact of plantations on peat).

Gregg *et al* (2021) use the Woodland Carbon Code and data on soil carbon from other sources to produce representative sequestration rates of -7 t CO_2 eq ha^{-1} y^{-1} and -14.5 t CO_2 eq ha^{-1} y^{-1} averaged over 100 years and 30 years, respectively for broadleaved woodland. The rates are higher for the shorter time period due to the high sequestration rates seen in the early growth phases after establishment.

6.3.3 Wood Pasture

Wood Pasture is an open, grazed habitat often with very old trees and lots of deadwood. The proportion of trees to open ground will depend on the type of grazing management carried out. Gregg *et al* (2021) were unable to find any studies of GHG cycling in wood pasture.

6.3.4 Hedgerows

Hedgerows are created by people as boundaries which were traditionally managed by "laying" and regular trimming to form an impenetrable barrier. They are now predominantly managed by mechanical cutting or flailing. In the UK they consist mainly of hawthorn (*Crataegus monogyna*) and blackthorn (*Prunus spinosa*) with a far wider range of species featuring in hedges in and around urban areas — particularly in parks and gardens. Since the Second World War, the extent of hedgerows in the UK has declined from 1.4 million km (in England and Wales alone) to a UK total of 656,000 km, with 200,000 km in a poor state. They continue to decline in quality (as summarised by Gregg *et al*, 2021).

Gregg *et al* (2021) highlight a lack of studies of carbon cycling in hedgerow vegetation and where these do exist, they are usually based on the extrapolation of data from woodland understorey species so may be inaccurate given that hedgerows are managed in a very different way to woodland understorey.

Estimates of carbon stocks in hedgerow soils vary widely but generally seem to reflect the soil stocks of the surrounding fields, except in the case of arable fields where hedgerow soil organic carbon stocks may be higher, reflecting the original state before the field was cultivated. Hedgerows have a limited ability to increase soil carbon stocks in a narrow strip of the field adjacent to the hedge.

Some attempts have been made to model carbon fluxes in hedgerows but, as these are largely based on woodland understorey, they are likely to be unreliable.

Even so, hedgerows may play a significant role in preventing soil erosion, which will have a wider landscape-scale benefit on GHG emissions — particularly if they prevent eroded material from entering fluvial systems.

6.3.5 Orchards

The role orchards play in carbon sequestration is very dependent on the type of orchard and its management. Robertson *et al*, (2013) show that intensively managed orchards may sequester more carbon during their lifetime than traditional orchards due to intensive pruning methods, the younger age of the trees and the higher planting density. However, much of this carbon is lost from the system through the fruit harvest and at the end of the shorter lifespan of intensively managed trees, they are often burnt, releasing their carbon back to the atmosphere. Longer-lived traditional orchard trees may store more carbon for longer, in their woody biomass.

Robertson *et al* (2013) also suggest that orchard soils under traditional management may store more carbon than those under intensive management, but soil sequestration rates may be higher under intensive management as they move towards equilibrium, from a disturbed state. Further research is needed to fully understand the carbon dynamics of orchards soils.

6.3.6 Scrub

Gregg *et al* (2021) found very few studies on carbon cycling in scrub in the UK and those from alpine areas showed variable results. Further research is needed for reliable estimates of stocks or fluxes in UK scrub environments to be made.

DON SUTHERLAND

Land Use	Soil Carbon Stocks ($t\ C\ ha^{-1}$)		Sources	Net GHG flux ($t\ CO_2\ eq\ ha^{-1}\ yr^{-1}$)	Sources
	Soil	Vegetation			
100-year Mixed native broadleaved woodland on mineral soil	15 cm depth = 50-59 1 m depth = 108-173	41-344	Gregg *et al* (2021); Poulton *et al* (2003); Vanguelova *et al* (2013)		
30-year Mixed native broadleaved woodland on mineral soil	15 cm depth = 50-59 1 m depth = 108-173	22-204			
Traditional Orchards	30 cm depth = 47-111	9-230	Robertson *et al* (2012)		
Forestry (mainly conifer) on organo-mineral soil				-6.9 to -29	Read *et al* (2009), Morrison *et al* (2012), Greig (2015)
Forestry (broadleaves) on non-organo-mineral soil				-8.64 to -18.50	Read *et al* (2009), Morrison *et al* (2012), Greig (2015)
Mixed native broadleaved woodland (100 year)				-2 to -13	Gregg *et al* (2021); Poulton *et al* (2003); Ashwood *et al* (2019)
Mixed native broadleaved woodland (30 years)				-2.5 to -25.5	Gregg *et al* (2021); Ashwood *et al* (2019)
Ancient Broadleaved Woodland				-4.4 to -6.6	Thomas *et al* (2011)
Traditional Orchard				-5.89 to 1.65	Robertson *et al* (2012)
Intensive Orchard				-4.21 to -7.77	Robertson *et al* (2012)

Table 3: Summary of review of soil carbon and above-ground biomass stocks and GHG fluxes from woodland and trees (not on peat) adapted from Gregg *et al* (2021). Positive numbers = emission, negative numbers = sequestration. *includes enteric fermentation and artificial fertilisers.

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6.4 Peatlands

Peatlands are the largest store of carbon of any UK land-use type, as they cover a large land area and their soils are extremely carbon-rich. Consequently, the loss of carbon from peatlands could have a huge impact on GHG emissions from land. UK peatlands cover 3 million hectares (12.2% of the total UK land area) and about 22% of these remain in a near-natural condition, sequestering CO_2 at a rate of about 1,800 kt CO_2 yr^{-1} (Evans *et al*, 2017). The remainder of peatlands, which are degraded to varying degrees (78%) are net emitters of GHGs, however.This section deals with peatlands in the following broad categories:

- Agricultural Peatlands
- Forested Peatlands
- Blanket Bog
- Raised Bog
- Fen

Peatland in good condition sequesters large amounts of carbon because the input of CO_2 exceeds output as a consequence of very slow decomposition rates resulting from low nutrient levels, water-logged anaerobic conditions and the presence of Sphagnum spp. Peatlands are dominated by hydrological conditions with the height of the water table largely determining the balance between sequestration and decomposition. If peatlands are drained, allowing greater oxidation, higher rates of decomposition occur and more CO_2 is returned to the atmosphere. The high water tables needed to maximise CO_2 sequestration also create the conditions for microbes to create CH_4 which is a more powerful GHG than CO_2. The balance between the draw-down of CO_2 and the emission of CH_4 results in near-natural peatlands being modest net sinks of GHGs, with a small net sequestration potential. However, unlike most habitats peatlands don't reach an equilibrium and can, in the right conditions, go on sequestering CO_2 in perpetuity, thereby storing large amounts of atmospheric carbon.

When damaged, peatlands can emit huge amounts of stored carbon. Around (41%) of the UK peat area is still in a semi-natural state but has been damaged or modified, largely by human activity (predominantly grazing, burning and drainage). They are now emitting GHGs at a rate of about 3,400 kt CO_2 eq yr^{-1}.

The bulk of emissions come from the estimated 31% of UK peatlands that have been converted to agriculture (arable and grassland) or forestry. These currently emit an underestimated 18,500 kt CO_2 eq yr^{-1} which is 75% of the GHG emissions from UK peatlands.

A further 1,200 kt CO_2 eq yr^{-1} comes from the 145,000 ha of current and historic peat cutting for domestic fuel and a small contribution from industrial peat extraction for horticulture (Evans *et al*, 2017).

As a result of these impacts, UK peatlands overall have switched from being a GHG sink estimated to be around 0.25 Mt CO_2 eq yr^{-1} before human impacts to a current source of 23 Mt CO_2 eq yr^{-1}. According to Evans *et al* (2017), this was enough to convert the whole of the UK Land-use, Land-use Change and Forestry (LULUCF) GHG inventory from a net sink to a net source of emissions (Department for Business, Energy & Industrial Strategy, 2021). This emphasises the urgent need to restore peatlands to reduce emissions as soon as possible.

The depth of peat varies considerably across UK peatlands, so accurate estimates of carbon stocks are not readily available. It is, however, very clear that peatlands store substantially more carbon than any other terrestrial or freshwater habitat in the UK (Table 3). Blanket and raised bogs are characterised by two layers — the peat layer (catotelm) and the living vegetation layer on top of this (acrotelm). It has been suggested previously that this acrotelm layer may contain an additional 2 t C ha^{-1} (Field *et al*, 2020), but Lindsay (2010) suggests that this could be a significant underestimate as it does not take account of the carbon stored in the Sphagnum component, which could be 45.5 t C ha^{-1} in a 15 cm deep acrotelm.

A comprehensive assessment of fluxes from peatlands was conducted by Evans *et al* (2017) and updated in 2021 (as reported in Gregg *et al*, 2021) (Table 4). This work was carried out to provide data to the national GHG inventory reporting process. The Emissions Factors developed provide the best currently available data for standardising the quantification of GHG emissions from peatlands in the UK for potential carbon removal or emissions reduction schemes, even though its applicability to local circumstances needs to be backed up by direct measurement.

6.4.1 Agricultural Peatlands

Most lowland peatlands are drained for agricultural production, either for crops or intensively managed grassland. These ongoing intensive management practices have caused considerable damage to lowland peatland habitats with substantial losses of the underlying peat and its stored carbon. Evans *et al* (2016) suggested that these peatlands could lose all their organic content within the next 100-200 years. Factoring in the effects of climate change, research commissioned by the Climate Change Committee concluded that the peat soils of England's fenlands may be lost within 30 to 60 years (CCC, 2013). Agricultural peatlands are one of the highest land-based sources of GHG emissions, with Emissions Factors in Evans *et al* (2017) of 37.61 and 27.54 t CO_2 eq yr^{-1} for arable and intensively managed grasslands, respectively.

6.4.2 Forested Peatlands

Gregg *et al* (2021) conclude that forestry planting on peatlands leads to a loss of the carbon store and Evans *et al* (2017, updated 2021) provide Emissions Factors that vary between 1.15 and 5.46 t CO_2 eq ha^{-1}. There is some evidence that a high-yielding plantation may sequester more carbon than is lost from the peat, but Gregg *et al* (2021) make the obvious but very important point that a long-term millennial-scale peatland carbon store is being replaced by a much less permanent system when it is afforested.

6.4.3 Blanket Bog

Blanket Bog clothes the gently sloping landscapes of upland Britain, as a consequence of high levels of rainfall that exceed the loss of water from the system. It is the most extensive form of peatland in England, covering approximately 355,000 ha (although this is probably an under-estimate due to degraded peatlands being misclassified as heathland and an arbitrary peat depth cut-off of 40 cm). Blanket bog has suffered considerable damage from burning, drainage, afforestation, excessive grazing and recreational pressure. Most of this damage occurred within the last 100 years, largely because of failed government-backed attempts to drain blanket bog for agriculture.

According to Artz *et al* (2014) 'pristine' bog may be sequestering carbon at a rate of -0.76±0.39 t CO_2 eq yr^{-1} although there are few studies of natural bogs. Table 4 shows that damaged bogs are emitting large amounts of GHG with Emissions Factors of between 2.31 and 13.28 t CO_2 eq yr^{-1} depending on the type of damage. When blanket bogs are restored through rewetting to near-natural conditions they can, once again, become a carbon sink of around -0.02 t CO_2 eq yr^{-1}.

6.4.4 Raised Bog

Raised bog is predominantly formed in the lowlands of the UK where drainage is impeded, leading to standing water or fen which then over time fills with peat and vegetation dominated by species such as Sphagnum. These places become bogs once the vegetation is separated from the minerotrophic groundwater sources. Over time, the bog forms a raised dome with some of the deepest peat in the UK.

A major hydrological difference between a raised bog and a blanket bog is the presence of a lag fen where the water draining from the domed bog collects at its edge, along with water from the surrounding landscape. These lag fens are often drained in agricultural landscapes, which leads to the drying out of the bog.

Raised bogs have been subjected to even more damage than blanket bogs largely through historical drainage for agriculture, development and removal of peat for domestic and industrial use.

Although they cover a smaller area than blanket bog, raised bogs store substantially more carbon per hectare due to the greater depth of peat (Table 4). Damage from extraction emits substantial amounts of carbon with Emission Factors of greater than 13 t CO_2 eq yr^{-1}.

Evans *et al* (2017, updated 2021, as reported by Gregg *et al*, 2021) does not differentiate between blanket and raised bog for the rest of their Emissions Factors, due to lack of evidence, but there may be climatic and altitudinal differences that need further investigation.

6.4.5 Fens

Fens occur on both peatland and non-peatland soils and, unlike bogs, receive water from both groundwater and surface water flows and can, therefore, be rich in minerals and nutrients, making them much more diverse.

There are few studies of carbon cycling in fen habitats. Estimates of carbon storage vary widely (Table 4) but clearly show that, like bogs, fens store significantly more carbon than other habitats.

The majority of the UK's fenland has been lost to agriculture and those fragments that are left are often surrounded by intensive agricultural management which impacts on the hydrological and nutrient status of the remaining fens. Evans *et al* (2017, updated 2021, as reported by Gregg *et al*, 2021) estimated Emissions Factors from rewetted fen of 8.05 t CO_2 eq yr^{-1} and near-natural fen to be a net sink of -0.93 t CO_2 eq yr^{-1} (Table 4). Evans *et al* (2016) showed that semi-natural fens could be an even larger sink of -1.71 to -10.31 t CO_2 eq yr^{-1} depending on the vegetation and nutrient status of the fen, but differences in water table depth could lead to semi-natural fens becoming GHG sources of 1.46 to 4.88 t CO_2 eq yr^{-1}.

Land Use	Carbon Stocks (tC ha^{-1})	Sources	Net GHG flux (t CO_2 eq ha^{-1} yr^{-1})	Sources
Blanket Bog	653 - 944	Heinemeyer *et al* (2020)		
Raised Bog	810 - 2530	Evans *et al* (2016)		
Fen	610 - 2820			
Cropland (Drained)			37.61	2021 update of Evans *et al* (2017) as reported in Gregg *et al* (2021)
Intensive Grassland (Drained)			27.54	
Extensive Grassland (Drained)			13.03	
Extracted Industrial (Drained)			13.28	
Extracted Domestic (Drained)			13.37	
Forested (mainly conifer plantation) (Drained)			1.15 to 5.46	
Eroding Modified Bog (Bare peat) (Drained)			13.28	
Heather & Grass dominated Modified Bog (Drained)			3.54	
Eroding Modified Bog (Bare peat) (Undrained)			12.17	
Heather & Grass Modified Bog (Undrained)			2.31	
Rewetted Fen			8.05	
Rewetted Modified Bog			-0.02	
Near natural bog			-0.02	
Near natural fen			-0.93	
"Pristine" intact bog			-0.76	Artz *et al* (2014)
"Semi-natural" fen			-10.31 to 4.88	Evans *et al* (2016)

Table 4: Summary of review of soil carbon stocks and Emissions Factor (EF) for peatlands. Positive numbers = emission, negative numbers = sequestration.

6.5 Tidal saltmarsh

Burden *et al* (2016) estimated that only 0.2% (40-45,000 ha) of the "Coastal Wetland" area of England Wales remained as intact saltmarsh, the rest having been lost mainly as a result of drainage and reclamation for agriculture. The remaining saltmarsh area is at risk of further loss due to sea level rise and coastal flooding, especially where intertidal habitats are unable to migrate inland due to hard sea-defences.

The largest areas of former saltmarsh are around the Wash, Humber, Severn, and Thames estuaries.

Saltmarshes are very effective and, until recently, very undervalued carbon sinks. They sequester carbon through high primary productivity and low decomposition due to anaerobic conditions, combined with trapping sediments from other terrestrial and marine habitats. In addition, unlike our other big carbon store in peatlands, they do not generate large amounts of CH_4 (Alonso *et al*, 2012).

Saltmarshes are often classified into different zones ranging from pioneer, newly developing marsh with colonising plant species to high marshes with mature, diverse vegetation. Pioneer marsh has high carbon fluxes dependent on the balance between erosion and accretion whereas high marsh will have lower fluxes but high stored soil carbon.

Recent studies show that saltmarsh GHG sequestration rates could be as high as 30 t CO_2 eq ha^{-1} yr^{-1} (Ouyang & Lee, 2014) (see Table 5) although estimates show considerable variation depending on the state and maturity of the saltmarsh, the rate of development, soil, and vegetation type and whether the marshes are formed naturally or as the result of a managed realignment approach.

Land Use	Soil Carbon Stocks (t C ha^{-1})			Net GHG flux ($t\ CO_2\ eq\ ha^{-1}\ yr^{-1}$)	Sources
	Soil	Vegetation	Sources		
Saltmarsh	0.1-93	0.01-1.3	Beaumont *et al* (2014); Ford *et al* (2012); Burden *et al* (2013); Ford *et al* (2019)		
Pioneer/Low Saltmarsh				-29.08 to -2.05	Adams *et al* (2012), Ouyang & Lee (2014), Burden *et al* (2019), Foster (2020)
Middle/High Saltmarsh				-9.31 to -0.70	Adams *et al* (2012), Ouyang & Lee (2014), Burden *et al* (2019), Foster (2020)

Table 5: Summary of review of soil carbon stocks and fluxes for saltmarsh. Positive numbers = emission, negative numbers = sequestration.

MARK HAMBLIN/2020VISION

7 Nature based solutions to land-use greenhouse gas emissions

The concept of Nature-Based Solutions (NBS) has developed over the last two decades as a framework for promoting the delivery of social and economic development through environmental protection and enhancement. It explicitly recognises that human beings are part of the natural world and dependent on it for their survival, wellbeing and prosperity. Consequently, it promotes the notion that protecting, restoring and maintaining ecologically functioning natural systems and processes is an integral part of addressing many of the social and economic challenges that people face. In short, NBS use ecosystems and the services they provide to address societal challenges such as climate change, food security, public health or natural disasters (The Wildlife Trusts, 2021).

Because of the increasing interest in NBS globally and their increasing prevalence in public policy, the IUCN has developed an overall definition, principles and operational framework for NBS, including an International Standard for their application and delivery (IUCN, 2020). NBS can have diverse and far-reaching applications to a multitude of societal issues (Cohen-Scham *et al* {eds}, 2016), one of which is the challenge posed by global climate change. Healthy (and restored) natural ecosystems are increasingly being recognised as vital contributors to efforts to reduce or cope with the emerging, imminent and likely future impacts of climate change (Seddon, N. *et al*, 2020).

Carbon naturally circulates between the atmosphere, the ocean, mineral deposits, soils and living systems, with different parts of the cycle taking place over anything from hours to millennia. Large quantities (trillions of tonnes) of carbon are stored within rocks and dissolved in the ocean. At any given time, smaller (but still very significant) amounts are held within the atmosphere, soils, marine sediments and living systems — mainly in vegetation on land and marine life at sea (Friedlingstein *et al* 2020).

At an ever-increasing rate, humans have been releasing carbon from the mineral store (by burning fossil fuels), from soils (through erosion and increased decomposition), sediments (by disturbing them and releasing their carbon back into circulation) and living systems (by removing and simplifying vegetation — mainly on land — and reducing the abundance of animals — mainly at sea (Friedlingstein *et al* 2020).

Given this relationship between the Earth's living systems and atmospheric levels of GHGs, it is clear that the pro-active maintenance, restoration and expansion of natural ecosystems and wildlife habitats would make a positive contribution to attempts to reduce atmospheric GHG levels and stabilise the climate. Having more living matter (of whatever kind — especially structurally complex plants) would lock up more carbon. And some living communities (such as developing woodlands and actively growing peatlands) not only accumulate and store carbon within the living tissue of their component plants, animals, fungi and microorganisms; over extended time periods, they also accumulate it in significant quantities within long-lasting soils (such as peat) and sediments (either at sea or in freshwater bodies), where it can be held for millennia (The Wildlife Trusts, 2021).

Nature Based Solutions to climate change are therefore increasingly being seen as direct contributors to the reduction of GHG emissions, the removal of atmospheric GHGs and their long-term storage. Girardin, *et al* (2021) characterised the potential of nature-based solutions as "three steps to natural cooling": improved **protection** of "intact lands", better **management** of "working lands" and the **restoration** of native vegetation cover.

The main approaches to NBS to land-use GHG emissions in the UK are:

- Protecting remaining high quality natural and semi-natural habitats.
- Reducing emissions caused by the conservation management of high quality natural and semi-natural habitats.
- Restoring degraded or creating/recreating natural and semi-natural habitats.
- Reducing emissions from productive agricultural landscapes through better soil, crop, and water management.

7.1 Protecting remaining high quality natural and semi-natural habitats.

It is clear from the review in section 6 (summarised in Table 6) that nearly all terrestrial natural and semi-natural habitats store significantly more carbon in their soils than agriculturally improved habitats, even with the uncertainty and variability around some of the estimates. This is likely to be because of generally increased soil disturbance and drainage in agricultural systems, combined with the removal of carbon as harvested crops. Habitats with woody vegetation have similar levels of soil organic matter to agricultural systems but store significant amounts of carbon in above-ground biomass (trunks, branches, woody stems).

Therefore, these less intensively managed areas of land need to be protected from loss or conversion to other land-uses to preserve their carbon stocks.

Natural and semi-natural habitats in the UK have been subjected to many pressures and the majority have declined to relatively small fragments which remain under threat. Many are protected and managed as nature reserves, have statutory protection and/or are maintained though agreements related to agri-environment schemes. Despite this, many of these habitats continue to decline in quantity and quality, often under pressure from development, agricultural intensification and increasingly from climate change (Hayhow *et al*, 2019).

While most species-rich grasslands are likely to be at an equilibrium in terms of GHG fluxes (Figure 1), woodlands and peatlands can go on sequestering carbon for considerable periods. Woodlands are also likely to reach an equilibrium state eventually (Figure 1), but this could take centuries and they continue to sequester carbon for over 100 years (see Table 3). Peatlands in good condition can continually sequester (Figure 1) small amounts of CO_2 (Table 4) each year and store them for thousands of years.

As discussed earlier (Sub Section 4.6.1), managing the *status quo* does not deliver against one of the key tenets of emissions reduction or carbon removal schemes — additionality. Emissions reduction savings should be additional to those likely to result from existing activities. Therefore, continued and expanded support for the protection and continued maintenance of existing habitats will still be needed alongside any emissions reduction or carbon removal scheme. This might include Wildlife Trust fund-raising, public support to land managers through grants and agri-environment schemes and/or corporate donations. The development of carbon emissions reduction or removal schemes should not be considered as a replacement for this ongoing support.

Land Use	Carbon Stocks (t C ha^{-1})
Arable soils (0-100m soil depth)	110-150
Agricultural Grassland soils (0-100m soil depth)	130-230
Low plant diversity, intensive management	403
Intermediate plant diversity, intermediate level management	446
High plant diversity, extensively managed	414
Floodplain	323 ± 13
Blanket Bog	653-944
Raised Bog	810-2530
Fen	610-2820
100-year Mixed native broadleaved woodland on mineral soil	
Soil	108-173
Vegetation	41-344
30-year Mixed native broadleaved woodland on mineral soil	
Soil	108-173
Vegetation	22-204

Table 6: Summary of carbon stocks in terrestrial soils under different land-uses.

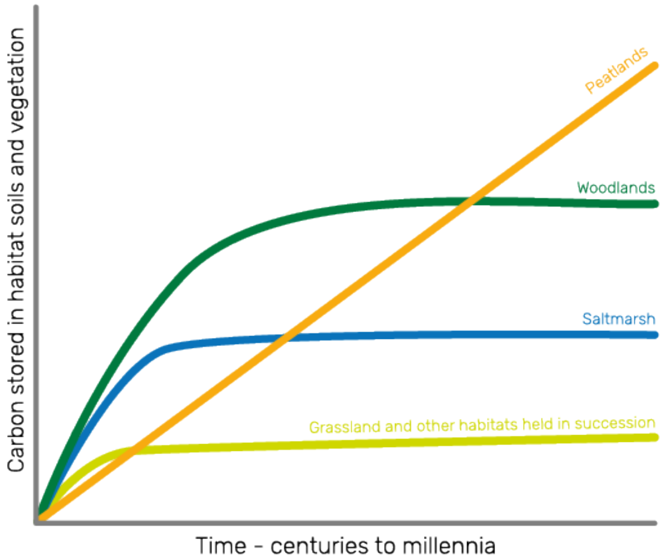

Figure 1: Conceptual model of habitat trajectory towards carbon stock equilibrium. The rate of sequestration and capacity to store carbon is different for different habitats, with every site having an equilibrium specific to its management, climate and soils. The exception to this is peatlands, which can continue to sequester carbon for many millennia. Note — this figure is conceptual: axes are for illustration and are not to scale. Trajectories assume no disturbance within the habitat. Reproduced with permission from Gregg *et al* (2021).

7.2 Reducing emissions resulting from the conservation management of high quality natural and semi-natural habitats.

For many habitats of high conservation value in the UK, interventionist management is often needed to retain the existing biodiversity interest. For species-rich grassland and heathland in particular this can involve mowing, grazing, scrub cutting and/or burning to prevent succession to woodland. Some established diverse woodlands may have a long history of management such as coppicing. All these interventions have a potential GHG impact (Figure 2) so reviewing and altering management practices while still retaining biodiversity interest could lead to GHG emissions reductions.

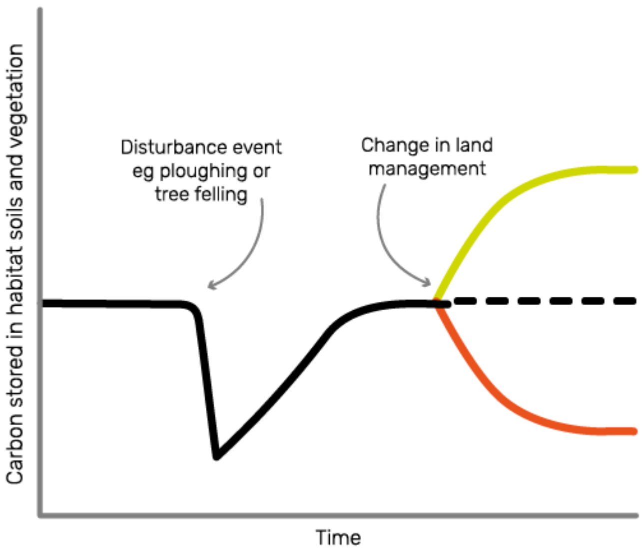

Figure 2: Conceptual model of habitat carbon stock equilibrium and land use change. Once a habitat achieves equilibrium, carbon stocks will maintain a steady state until disturbed or a new land management intervention is imposed. The green line indicates a change in land use to a habitat with a higher equilibrium (e.g., modified grassland conversion to woodland), and the red line a change to a lower equilibrium (e.g., semi-natural grassland to arable). The dashed line indicates a continuation in the established land management. Note — this figure is conceptual: axes are for illustration and are not to scale. Reproduced with permission from Gregg *et al* (2021).

For example, Sozanska-Stanton *et al* (2016) used literature values and readily available data combined with modelling approaches and the application of IPCC Tier 1 & Tier 2 methods to determine emissions from "undisturbed" habitats and the emissions associated with burning, drainage and/or grazing management (Table 7). Analysing potential GHG emissions relating to the implementation of the English Environmental Stewardship Scheme, Warner (2008) showed that less intensive management of grasslands could lead to greater GHG sequestration, but maintenance of some habitats for biodiversity reasons led to small amounts of increased emissions (Table 8). According to these data, reduction in the impacts of grazing management and drainage could have a significant impact on the emissions from some of these habitats (though the evidence is very variable so could not currently be used as a reliable estimate for a Wildlife Trusts emissions reduction or carbon removal scheme). In the ongoing work of Wildlife Trusts there will almost inevitably be trade-offs where management for important biodiversity reasons (such as maintaining open habitats required by specialist endangered species, by stopping succession), may lead to greater than if those habitats were left unmanaged, but can still be justified on the ground of needing to protect biodiversity and support nature recovery.

Habitat	GHG Emissions t CO_2 eq ha^{-1} yr^{-1}			
	Burning	Drainage	Grazing	Total
Lowland Calcareous Grassland			0.63-4.92	0.63-4.92
Upland Calcareous Grassland			0.02-3.40	0.02-3.40
Lowland Meadow			0.16-4.00	0.16-4.00
Upland Meadow			0.16	0.16
Purple Moor-grass Pastures	0-0.5	0-0.16	0.33-5.70	0.98-3.27
Lowland Dry Acid Grassland			0.01-4.40	0.01-4.40
Coastal Machair			0.08	0.08
Mountain Heath	0-0.5		0.03-0.07	0.03-0.12
Lowland Heath	0.2-0.5	0-18.8	0.69-3.78	1.12-22.2
Upland Heath	0-0.5	0-18.2	0.02-4.4	0.53-22.2

Table 7: GHG emissions from management impacts on semi-natural open habitats (data from Sozanska-Stanton *et al* (2016).

Habitat	Intervention	GHG Emissions (t CO_2 eq ha^{-1} yr^{-1})
In-bye Grassland	Managing with low inputs	-0.03 to -0.61
In-bye Grassland (organic)	Managing with low inputs	0.00 to -0.44
Semi-natural grassland	Maintenance	0.00
Rough grazing	Maintenance	0.00 to 0.04
Rough grazing (organic)	Maintenance	0.02
Rush pastures	Maintenance	0.01
Rush pastures (organic)	Maintenance	0.00 to 0.05
Lowland Heath	Maintenance	0.07

Table 8: GHG emissions from different types of Environmental Stewardship Scheme options for the management of “open” habitats (data from Warner, 2008). Negative values = sequestration, positive values = emissions.

7.3 Restoring degraded natural and semi-natural habitats

7.3.1 High nature value grasslands (not on peat)

There are few studies that quantify the GHG emissions associated with restoration management of high nature value grasslands. Those studies reviewed suggest that emissions are most likely to result from management impacts on the soil, so activities that disturb soil should be avoided.

Species-rich grasslands are usually maintained either by traditional grazing or cutting practices. No studies of the impact of changes in cutting regimes could be found in this review but Abdalla *et al* (2018) showed that any grazing of species-rich grasslands consistently reduced soil organic carbon in studies from across the globe.

Grazing impacts cannot be separated from other management activities. For example, the addition of lime and fertiliser can increase productivity, which may offset the losses from grazing. Ward *et al* (2016, Table 1) showed that extensive, traditional grassland management did lead to higher carbon stocks than intensive management but an intermediate level of management with the addition of some fertiliser and a more intensive grazing and cutting regime stored yet more. There may be a trade-off between optimal management for GHG emissions reduction or carbon removal and management targeted at maintaining species richness. The review by Sosanska-Stanton *et al* (2016) suggests that the maintenance of existing high nature value grasslands generally has a limited role to play in global GHG emissions reductions.

7.3.2 Heathland

The bulk of carbon in heathlands is in the soil, with very little stored in vegetation. The main objective of management of high nature value heathland is ultimately to prevent succession to woodland. Consistent with other habitats, any heathland management that involves soil disturbance is likely to lead to GHG emissions.

Restoration of heathland by reversing past succession to scrub and tree cover is also likely to lead to GHG emissions (Sosanska-Stanton *et al*, 2016; Warner, 2008).

There are, however, currently too few studies to justify including the restoration of degraded heathland in a GHG emissions reduction or carbon removal scheme. The focus should remain on restoration of degraded heathlands for biodiversity objectives.

7.3.3 Rivers, streams, open water, and wetland (not on peat)

This review was unable to find any studies of the GHG emissions associated with the restoration of degraded rivers, streams, open water and wetland systems. However, section 6 shows three things:

- rivers and streams act as fluvial pathways for the loss of carbon from terrestrial systems;
- GHG emissions from standing waters are largely dictated by inputs of sediment and nutrients from surrounding catchment land-use; and
- floodplains have the capacity to store significant amounts of carbon but are often not in the right condition to do so if they are managed intensively and/ or cut-off from fluvial systems by development and/or flood defence measures.

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Therefore, measures to reduce GHG emissions in catchments should focus on the restoration of terrestrial habitats to reduce sediment and nutrient input combined with naturalisation of fluvial systems and reconnection with less intensively managed floodplains. Reliable estimates of GHG emissions factors are not currently available for rivers, streams, open water or floodplain habitats, so in situ measurement could be particularly helpful here for restoration schemes.

7.3.4 Trees & Woodlands

The highest priority woodlands in the UK are ancient woodlands and these fall into three types — extant ancient woodland sites (AWS) that have been unmanaged for a long time; semi-natural ancient woodland sites (SNAWS) that have been in existence for centuries but may have been managed traditionally (e.g., coppice woodland) and plantations on ancient woodland sites (PAWS) that have been felled and replanted. The SNAWS and PAWS may subsequently have been neglected and are no longer managed, or they may be under conservation restoration management, or they may be managed as commercial forestry sites.

There are few studies of the impacts of management of existing native woodlands on GHG emissions. The variety of potential management options and the paucity of studies makes it impossible to come up with a standardised baseline set of GHG emissions for AWS under different regimes. As a result, it is equally impossible to quantify any likely change in emissions brought about by conservation management practices.

It is clear from Figure 3 that mature woodlands do not go on sequestering carbon forever (although even after 200 years NEE can still be as high as -4.4 to -6.6 t CO_2 eq ha^{-1} y^{-1} (Thomas *et al*, 2011)) with most of the carbon capture happening in the early decades of growth. The principal GHG benefit of ancient woodlands is the large amount of carbon they store in their soils and woody biomass, so management interventions that lead to soil disturbance or loss of the biomass will lead to GHG emissions.

The lack of GHG emissions studies and variability in findings for hedgerows, wood pasture, orchards, and scrub (see Gregg *et al*, 2021) mean that it is not possible to make meaningful conclusions around restoration management, although just allowing hedgerows to grow bigger (which has significant biodiversity benefits) will clearly store more above-ground biomass until they are re-laid or coppiced.

The Woodland Carbon Code (WCC) provides a recognised systematic practical method for estimating and reporting carbon sequestration within some kinds of intentionally created woodland, Its application to woodland in a number of situations is problematic, however, particularly when dealing with non-plantation mixed native broadleaved woodland with a diverse species mix and complex structure, and where woody vegetation occurs at lower densities or smaller sizes. While WCC can be applied to many woodland creation initiatives delivered or facilitated by The Wildlife Trusts, the most robust way for The Wildlife Trusts to accurately determine the GHG benefits of management to restore trees and woodlands may be through direct measurement on a site by site basis. reduction or carbon removal scheme is currently through direct measurements on a site-by-site basis.

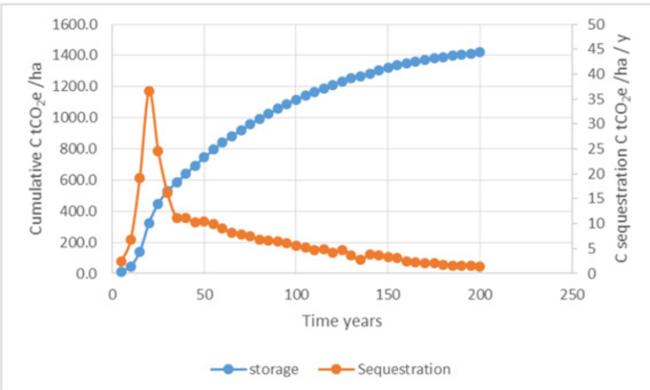

Figure 3: Illustration of how carbon sequestration of new woodlands peaks after a few decades, whereas carbon storage increases towards an equilibrium. (Based on Woodland Carbon Code data for un-thinned Yield Class 8 Oak in 5-year time intervals on a mineral soil with minimal soil emissions. Note: the modelling of early growth is limited by a lack of data so the timing and height of the early peak should only be treated as illustrative). Reproduced with permission from Gregg *et al* (2021).

7.3.5 Peatlands

As highlighted in section 6, peatlands are the largest carbon store in the UK and, as they are largely in a highly degraded state, they are currently a net major source of land based GHG emissions. Therefore, the restoration of degraded peatlands is urgently needed and represents one of the most important land-based actions for reducing GHG emissions. In addition, in the longer term, restored peatlands could once again become small annual GHG sinks, sequestering small amounts of carbon each year and storing it for thousands of years.

The Emissions Factors provided by Evans *et al* (2017, updated in 2021 as reported in Gregg *et al*, 2021) provide a relatively robust method for determining the GHG gains from restoring a degraded peatland habitat to a less-degraded condition with the aim of restoring to fully functioning GHG sequestering systems (Table 9). For the purposes of this report the conversion of arable cropland and grasslands on peat soils are considered to be habitat conversions rather than restoration of degraded peatland and are dealt with in section 7.4, below.

7.3.5.1 Blanket Bog

Peatland restoration techniques on upland blanket bogs have developed significantly over the last 20 years, with organisations like Moors for the Future and the Yorkshire Peat Partnership (managed by Yorkshire Wildlife Trust) developing an iterative approach that aims to revert degraded peatlands back to Sphagnum spp. dominated, hydrologically intact systems.

The blanket bog restoration process consists of:

- **Hydrological restoration** — blocking artificial drainage ditches and their associated erosion gullies with measures to slow the flow, enabling revegetation of gully floors and sides and reducing erosion and loss of DOC and POC to fluvial systems. As the gullies fill, repeat blocking should gradually raise water tables as well.
- **Stopping harmful management** — controlling management likely to inhibit vegetation recovery such as excessive grazing or burning.
- **Erosion control** — a combination of re-profiling gully and peat hag edges and bunding and revegetation of bare peat to prevent further losses.
- **Reintroduction of missing peatland species** — Cotton-grasses and dwarf-shrub species are seeded or planted to help with the erosion control process together with the re-introduction of Sphagnum species which help to further enhance the hydrology of the degraded peat and move the peatland towards a more natural functioning peat bog.

7.3.5.2 Raised Bogs

Raised bogs have suffered more significant management impacts from peat removal for fuel and horticultural use and from agricultural improvement. Drainage to facilitate peat extraction and the removal of functioning lag fen are significant issues in the restoration of degraded raised bogs.

The restoration process for raised bogs is very similar to the restoration requirements of blanket bogs although the hydrological management is a little more complex as degraded raised bogs have very little slope. Because of this, techniques involving cellular bunding with impermeable peat dams have been developed to help hold water and enable peatland vegetation to form. Scrub control is also often needed on drained raised bogs and lag fen restoration remains a significant problem in returning damaged raised bogs to full hydrological integrity.

7.3.5.3 Fen

As discussed in section 6, there is a lack of information on carbon cycling in fens, which makes it difficult to determine the GHG benefit of restoring degraded fens. It can also be difficult to differentiate precisely between a fen and a bog — particularly as most raised bogs develop from fens. It is known that fens can store a large amount of carbon, but most fens in lowland UK have been severely impacted by drainage and conversion to agricultural use. As with bogs, the status of fens is heavily influenced by hydrology, both in the surrounding landscape and through the impacts of drainage ditches within the fen itself. Restoration of hydrological conditions would therefore influence GHG emissions. Evans *et al* (2016) showed that, on average, lowering of the water table by 10 cm leads to a loss of about 4 t CO_2 eq ha^{-1} y^{-1}. Also, when the water table is raised to 25 cm below the surface, CH_4 emissions begin so that for every 1 cm rise in water table above this, additional CH_4 emissions (2 t CO_2 eq ha^{-1} y^{-1}) occur. Therefore, a careful balancing of hydrology would be needed to optimise GHG emissions reductions in fens if this is judged to be a primary aim for their restoration. This might not necessarily be in the interests of fen biodiversity.

Fen vegetation may also have an impact on carbon cycling. Taller vegetation such as reeds is likely to hold a larger biomass of carbon and some types of vegetation may increase or decrease CH_4 emissions. There are, however, too few studies to quantify this accurately.

Fens also frequently contain drainage ditches which can produce highly variable fluxes of both CH_4 and CO_2 ranging from emission to sequestration (Evans *et al* (2016), Peacock *et al*, 2016). This variability makes it difficult to quantify the impact of this on overall GHG emissions from fens, but emissions from ditches could make a significant contribution.

Therefore, although Evans *et al* (2017, updated 2021 as reported in Gregg *et al*, 2021) provide Emissions Factors for fens (Table 9) there is clearly still a great deal of uncertainty behind these numbers and there is a need for further research into fenland GHG fluxes.

7.3.6 Saltmarsh

Grazing is the predominant management activity on saltmarsh but there are too few studies to determine the impact of changes in grazing on GHG fluxes. There is some suggestion that "hotspots" of methane emissions could occur under higher grazing regimes (Gregg *et al*, 2021) but the impact of this on overall GHG fluxes is unknown.

Starting Land Use		Changed Land Use		Potential GHG Gain (t CO_2 eq ha^{-1} yr^{-1})		
Land Use (Soil Depth)	GHG flux (t CO_2 eq ha^{-1} yr^{-1})	Land Use	GHG flux (t CO_2 eq ha^{-1} yr^{-1})	Avoided emissions	Sequester	Total
Peatland — Extracted Domestic (drained)	13.37	Peatland — Rewetted Modified Bog	-0.02	-13.37	-0.02	-13.39
		Peatland — Rewetted Fen	8.05	-5.32	0	-5.32
Peatland — Extracted Industrial (drained)	13.28	Peatland — Rewetted Modified Bog	-0.02	-13.28	-0.02	-13.30
		Peatland — Rewetted Fen	8.05	-5.23	0	-5.23
Peatland — Eroding Modified Bog (Bare peat) (Drained)	13.28	Peatland — Heather & Grass Modified Bog (Undrained)	2.31	-10.97	0	-10.97
Peatland — Eroding Modified Bog (Bare peat) (Undrained)	12.17	Peatland — Heather & Grass Modified Bog (Undrained)	2.31	-9.86	0	-9.86
Peatland — Heather & Grass dominated Modified Bog (Drained)	3.54	Peatland — Heather & Grass Modified Bog (Undrained)	2.31	-1.23	0	-1.23
Peatland — Heather & Grass Modified Bog (Undrained)	2.31	Peatland — Near natural bog	-0.02	-2.31	-0.02	-2.33
Peatland — Rewetted Modified Bog	-0.02	Peatland — Near natural bog	-0.02	0	0	0
Peatland — Near natural bog	-0.02	Peatland — "Pristine" intact bog	-0.76	0	-0.74	-0.74
Rewetted Fen	8.05	Peatland — Near natural fen	-0.93	-8.05	-0.93	-8.98

Table 9: Potential changes in GHG emissions from the restoration of degraded peatlands. Negative GHG fluxes = sequestration, positive fluxes = emissions. Reductions in emissions through the restoration of degraded habitats constitute avoided emissions in a potential Wildlife Trust emissions reduction scheme and potentially lead to recovery of long-term sequestration potential.

7.4 Creating/recreating new carbon-rich natural and semi-natural habitats

A further step that could be taken beyond restoring degraded habitats to reduce GHG emissions would be to convert from a habitat with lower carbon stocks, lower GHG sequestration rates and/or higher emissions to a habitat with higher carbon stocks, higher sequestration rates and/or lower emissions.

Using the data from section 6, it should be possible to estimate the change in the amount of GHGs emitted and/or sequestered when creating new, restoring degraded, or altering the management of existing habitats. This would simply be the difference between the GHG emissions for the starting land-use or habitat state and the emissions for the new habitat or land-use. This difference would give the amount of CO_2 equivalent of GHG emissions likely to be avoided and/or the amount of additional CO_2 equivalent of GHG likely to be sequestered because of the land-use or land management change.

In this report the following criteria were used in selecting the habitat or land-use change scenarios:

1. The flux or stock data for the starting and new habitats are robust enough.
2. The change makes sense ecologically and environmentally.
3. The change is likely to be carried out for biodiversity enhancement reasons; and
4. Changes that are likely to damage the biodiversity value of habitats would not be included within any Wildlife Trust-led initiative.

These criteria were determined and applied by the authors, based on their own knowledge and experience. We welcome their review by others in order to refine the change scenarios. Two approaches have been taken to calculating the carbon benefits of land use change here, as outlined in Section 5.5:

1. **'Flux measurement' approach** — Tables 9 and 10 detail potential GHG gain for habitat and/or land-use changes, calculated from suitable 'before' and 'after' Emission Factors derived from the evidence reviewed. Table 9 deals with peatland habitats, and is based on data that is compatible with the 2019 UK GHG Inventory. Table 10 deals with non-peatland habitats and includes data from a wider variety of sources. Where the evidence reviewed suggests ranges of possible Emission Factors for a particular land-use, the low and high limits of these have been provided.

   There are high levels of uncertainty and variability in the evidence for some land uses on mineral and organo-mineral soils (in particular arable, intensively managed agricultural grasslands and some kinds of high-nature-value grasslands). These make it very difficult to deduce credible values for the changing rates of GHG emissions and removals that might result from converting arable to grassland or grassland to other habitats (on non-peat soils). As a result, values suggested for these changes should be treated with a high degree of caution.
2. **'Stock change' approach** — Table 14 in Appendix 1, shows carbon stock values and ranges derived from the evidence sources reviewed and indicates the annual GHG emission and removal rates (and ranges) that these would imply, for changes relating to a number of "open" terrestrial habitats (those where suitable stock data are available).. As with the 'flux measurement' approach above, the 'stock change' approach also brings with it considerable uncertainty in some of the stock values. These should also be treated with caution.

Given the level of uncertainty around many of the estimates in Table 10 and Table 14 (Appendix 1), and the generally accepted convention that GHG emissions and removals should be estimated conservatively, it would be appropriate to use the 'low gain' values where they are available. This will, of course, mean that some potentially high real-world sequestration rates may be under-estimated.

Both the 'flux measurement' and 'stock change' approaches (Tables 10 and 14 respectively) indicate that for some land use changes uncertainty is simply too high for reliable general conclusions to be drawn about the likely changes in emission and sequestration rates associated with them. It is highly likely (for instance) that complex variation in individual circumstances will determine whether the creation and ongoing management of calcareous grassland on currently arable farmland would lead to increased emissions or sequestration. This analysis of the evidence suggests that being conservative, in some circumstances, conversion of intensively managed grassland with low plant diversity to more extensive, high wildlife-value habitats (even woodland) may lead to lower sequestration or higher emissions. Such instances are highlighted grey in the tables. The GHG benefits of converting production forestry plantation to more natural broadleaved woodland is also unclear.

Tables 9 and 10 show that the largest net emission reductions are likely to be achieved by converting arable to woodland and, on peaty soils, to fen and bog, together with restoration of degraded peatlands. Creation of saltmarsh also has significant emissions reduction potential.

In the case of heathland restoration from productive forestry plantation, this might justifiably be carried out for wildlife conservation reasons, but Table 10 shows it is likely to cause a net increase in emissions, largely due to the removal of the woody biomass.The stocks change approach summarised in Table 14 (Appendix 1) provides clearer estimates for some arable and grassland land-use changes, albeit with some wide ranges and some uncertainty when trying to assess change to woodland — particularly when using data from 100cm soil depths.

CHRIS LAWRENCE

Starting Land Use			Changed Land Use			Potential GHG gain (t CO_2 eq ha^{-1} yr^{-1})				
Land Use (soil depth)	GHG Flux (t CO_2 eq ha^{-1} yr^{-1})		Land Use	GHG Flux (t CO_2 eq ha^{-1} yr^{-1})		GHG Emissions Avoided		GHG Sequestered		Total GHG Gain
	Low	High		Low	High	Low	High	Low	High	
Arable — on peat (drained)	37.61		Peatland — Rewetted Fen	8.05		-29.56		0		-29.56
			Extensive Grassland — on peat (drained)	13.03		-24.58		0		-24.58
			Intensive Grassland — on peat (drained)	27.54		-10.07		0		-10.07
Arable — organo-mineral and mineral soils	1.62		Middle/High Saltmarsh	-0.70	-9.31	-1.62	-1.62	-0.70	-9.31	-2.32 to -10.93
			Lowland dry acid grassland — "disturbed by management"	0.01	0.04	-1.58	-1.61	0	0	-1.58 to -1.61
			Lowland meadow — "disturbed by management"	0.02	4.41	2.79	-1.6	0	0	2.79 to -1.6
			Lowland calcareous grassland — "disturbed by management"	0.66	4.96	3.34	-0.96	0	0	3.34 to -0.96
			Lowland heath — "disturbed" by management	1.31	24.7	23.08	-0.31	0	0	23.08 to -0.31
			Mixed native broadleaved woodland (30 years)	-2.5	-25.5	-1.62	-1.62	-2.5	-25.5	-4.12 to -27.12
			Mixed native broadleaved woodland (100 years)	-2	-13	-1.62	-1.62	-2	-13	-3.62 to -14.62
UK Intensive Grassland — organo-mineral soils	-4.99	5.25	Lowland dry acid grassland — "disturbed by management"	0.01	0.04	5.03	-5.24	0	0	5.03 to -5.24
			Lowland meadow — "disturbed by management"	0.02	4.41	9.4	-5.23	0	0	9.4 to -5.23
			Lowland calcareous grassland — "disturbed by management"	0.66	4.96	9.95	-4.59	0	0	9.95 to -4.59
			Lowland heath — "disturbed" by management	1.31	24.7	29.69	-3.94	0	0	29.69 to -3.94
			Mixed native broadleaved woodland (30 years)	-2.5	-25.5	2.49	-5.25	0	-25.5	2.49 to -30.75
			Mixed native broadleaved woodland (100 years)	-2	-13	2.99	-5.25	0	-13	2.99 to -18.25
			Middle/High Saltmarsh	-0.70	-9.31	4.29	-5.25	0	-9.31	4.29 to -14.56
Intensive Grassland — on peat (drained)	27.54		Peatland — Rewetted Fen	8.05		-19.49		0		-19.49
			Extensive Grassland — on peat (drained)	13.03		-14.51		0		-14.51
Extensive Grassland — on peat (drained)	13.03		Peatland — Rewetted Modified Bog	-0.02		-13.03		-0.02		-13.05
			Peatland — Rewetted Fen	8.05		-4.98		0		-4.98
Upland Heath —"disturbed" by management	0.35	23.2	Mixed native broadleaved woodland (30 years)	-2.5	-25.5	-0.35	-23.2	-2.5	-25.5	-2.85 to -48.7
			Mixed native broadleaved woodland (100 years)	-2	-13	-0.35	-23.2	-2	-13	-2.35 to -36.2
Forestry (mainly conifer) on peat (drained)	1.15	5.46	Rewetted Modified Bog	-0.02		-1.15	-5.46	-0.02	-0.02	-1.17 to -5.48
Forestry (mainly conifer) on organo-mineral soil	-6.9	-29	Lowland heath — "disturbed" by management	1.31	24.7	53.7	8.21	0	0	53.7 to 8.21
			Upland heath — "disturbed" by management	0.35	23.2	52.2	7.25	0	0	52.2 to 7.25
			Mixed native broadleaved woodland (30 years)	-2.5	-25.5	0	0	26.5	-18.6	26.5 to -18.6
			Mixed native broadleaved woodland (100 years)	-2	-13	0	0	27	-6.1	27 to -6.1
Forestry (broadleaves) on organo-mineral soil	-8.64	-18.5	Lowland heath — "disturbed" by management	1.31	24.7	0	0	9.95	43.2	9.95 to 43.2
			Upland heath — "disturbed" by management	0.35	23.2	0	0	8.99	41.7	8.99 to 41.7
			Mixed native broadleaved woodland (30 years)	-2.5	-25.5	0	0	16	-16.86	16 to -16.86
			Mixed native broadleaved woodland (100 years)	-2	-13	0	0	16.5	-4.36	16.5 to-4.36
Intertidal sediments	-0.40		Pioneer/Low Saltmarsh	-2.05	-29.08	0	0	-1.65	-28.68	-1.65 to -28.68

Table 10: Changes in GHG emissions expected from creation/restoration of carbon-rich semi-natural habitats. Negative values indicate sequestration or emission reduction, positive values indicate emissions. Highlighted boxes indicate changes where a conservative interpretation and analysis of the evidence reviewed indicates that GHG gains may be possible, but that increased emissions or reduced sequestration is also a possibility. These changes should be treated with caution and taken on a case-by-case basis, as a reliable generic figure for the likely resulting GHG gain cannot be deduced from the evidence reviewed, using this method. Data in this table are derived from the primary evidence summarised for different habitat types and land uses, in tables 1-5 (Chapter 6).

7.5 Reducing GHG emissions from productive agricultural landscapes on organo-mineral (non-peat soils) through better/alternative soil and water management practices.

Clearly it is not going to be possible, or even desirable, to convert all areas of productive agriculture to wilder habitats, as food production will still be needed. However, if better management of crops (including growing new types) and soils can be achieved within intensive agricultural systems, to minimise carbon losses and/or maximise sequestration, this would make a significant contribution to the reduction of GHG emissions from land use. Particularly as agricultural land covers 71% of England's land area and stores around 583 Mt C in arable soils and 686 Mt C in the first 1 m of soils under intensively managed grasslands (Alonso *et al*, 2012). Muhammed *et al* (2018) calculated that arable land in the UK lost 0.08 t C ha^{-1} y^{-1} between 1970 and 2010.

Given that the aim of both arable and intensive grassland systems is to maximise the amount of crop and/or forage available which is then consumed, vegetation generally plays little part in net carbon sequestration in these systems. The key questions are how much carbon is lost from the soil as a result of agricultural operations, how much of the vegetation component can be transferred into soil storage, and what other GHGs are produced as a result of growing the crop.

Given the need to reduce the impacts of climate change on wildlife and the potential for Wildlife Trusts to help farmers and other land managers to play an active part in reducing carbon emissions from their operations, answering these questions and changing land management accordingly may be significant.

7.5.1 Changes in management of arable systems

7.5.1.1 Reducing erosion losses

Warner *et al* (2020) suggested a baseline loss of 0.7 t CO_2 eq ha^{-1} yr^{-1} from erosion from agricultural soils in England, largely from bare soils. Clearly any measures to reduce this erosion will reduce GHG emissions. Measures such as altering the direction of ploughing on slopes, retaining hedgerows or woodland shelter belts to act as barriers and ensuring continuous vegetation cover will reduce erosion losses.

7.5.1.2 Tillage

Gregg *et al* (2021) report that 3.9 million hectares of agricultural land in England and Wales is at risk from soil compaction due to excessive management in wet conditions, which can lead to increased nitrous oxide emissions and erosion, and prevents organic matter entering soil storage. Tillage is used to break up compaction but soil disturbance can lead to emissions. Reduced- or no-tillage practices have been advocated to reduce greenhouse emissions but as Gregg *et al* (2021) report, the evidence for this is variable and further research is needed.

7.5.1.3 Nitrogen

As Gregg *et al* (2021) rightly point out, the sequestration of carbon often requires increased inputs of nutrients including nitrogen which, when oxidised, is a significantly more powerful GHG than CO_2. This needs to be factored in when determining the GHG balance of agricultural systems.

7.5.1.4 Leys

Gregg *et al* (2021) reports on the potential use of leys (temporary grassland mixtures of grasses, legumes and broad-leaved forbs) as part of a rotation in mixed agricultural systems. These were once a commonplace method of rebuilding soil organic matter after periods of annual arable cropping. This system has tended to fall out of use as farms have specialised into arable or livestock, with an increasing reliance on artificial fertilisers.

A return to the use of leys and mixed farming (which brings inputs of organic matter through manure) may lead to increased sequestration of soil organic carbon although where ruminant livestock are used in the system, this will need to be balanced against the emission of CH_4 through enteric fermentation.

Gregg *et al* (2021) reported on a study by Johnston *et al* (2017) that showed increases in soil organic carbon content of 3-9% over 28 years, depending on the approach taken and whether or not it included livestock. They also referred to a study by Börjesson *et al* (2018), which showed increases in soil organic carbon concentrations of 0.36 and 0.59 t C ha^{-1} yr^{-1} (1.32 to 2.16 t CO_2 eq ha^{-1} yr^{-1}) in a ley system compared to a cereal monoculture, after 35 years of management (in this case with variation depending on soil structure).

Of course, these gains in soil organic matter may not be permanent, as the leys will periodically be returned to cropping or perhaps even revert to an arable cycle, depending on market demands and farmer preference.

The inclusion of leys in arable systems could therefore form part of an emissions reduction initiative (see Table 11) although, with the very limited number of studies these figures need to be treated with caution.

Starting Land Use		Management Change			Potential Total gain ($t C ha^{-1}$)		Average Annual Rate of Change $t CO_2$ eq $ha^{-1} yr^{-1}$ (years)
Land Use (soil depth)	Stock ($t C ha^{-1}$)	Change	Stock ($t C ha^{-1}$) Low	High	Low	High	
Arable & horticulture — organo-mineral soils (15cm)	47	Introduction of ley rotation (3-9% increase in stock, Johnston *et al*, 2017)	48.4	51.2	-1.4	-4.2	-0.1 to -0.2 (28 years)
		Introduction of ley rotation (increases in soil organic carbon concentrations of (1.32 to 2.16 t CO_2 eq $ha^{-1} yr^{-1}$, Börjesson *et al* 2018).					-1.32 to -2.16 (35 years)
		Use of cover/green manure crops (12% increase in stock, McClelland *et al*, 2020)	52.6		-5.6		
		Use of cover/green manure crops (1.17 t CO_2 eq $ha^{-1} y^{-1}$, Poeplau and Don (2015)					-1.17 (54 years)

Table 11: Estimates of potential total carbon stock changes in arable systems from changes in management. Estimates are based on a conservative minimum change to take account of uncertainty in the underlying data, so could be much higher in some cases. Negative values indicate sequestration of carbon into storage.

7.5.1.5 Cover crops

Gregg *et al* (2021) also report on the potential benefits of integrating cover or green manure crops (brassicas or legumes) into arable systems. These crops can potentially lead to increases in soil organic carbon content, reduce erosion and have some benefits for farmland birds and pollinating insects. Gregg *et al* (2021) reports on studies by McClelland *et al* (2020). which showed a 12 per cent increase in soil carbon content and Poeplau and Don (2015), which showed an annual change rate of 0.32±0.08 t C $ha^{-1} yr^{-1}$ (1.17 t CO_2 eq $ha^{-1} y^{-1}$) over 54 years where cover crops were included.

As with the incorporation of leys into arable systems, use of cover crops could form part of an emissions reduction initiative (see Table 11). Again, with the very limited number of studies, these figures need to be treated with caution.

7.5.2 Changes in management of intensive grassland systems

There are even fewer studies into GHG cycling in intensive grassland systems. The review in section 6 demonstrates that intensive grassland systems have higher soil organic carbon levels than arable systems but less than species-rich extensively managed grasslands. The evidence for the impact of changes in management on these soil stocks or sequestration rates is sparse.

7.5.2.1 Fertiliser use

This review found few studies of the impacts of changes in fertiliser use on GHG emissions although, as Gregg *et al* (2021) reports, increased nutrients lead to greater biomass productivity. This can lead to increased soil organic carbon in some circumstances and the use of manures and/or slurry may be more beneficial than artificial fertilisers. However, any potential increases in soil organic carbon due to the use of fertilisers then needs to be balanced against potential associated increases in emissions of N_2O and CH_4.

7.5.2.2 Plant diversity and grazing

This review could find no studies with quantified estimates of soil carbon stocks or sequestration rates due to changes in plant diversity or grazing rates. However, section 6 clearly shows that soil organic carbon levels are higher in species rich grasslands. Gregg *et al* (2021) also report some studies that show that inclusion of additional species can increase carbon stocks, but does not quantify this.

7.6 Reducing GHG emissions from productive agricultural landscapes on peat through better/alternative soil, water and crop management practices.

Section 6 clearly demonstrates that peatlands under agricultural management are currently emitting large amounts of GHG. Converting these damaged environments back to bog or fen would have major GHG emissions reduction benefits.

However, there are well-known opportunity costs of restoring all lowland peatland soils because they are currently within some of the most highly productive agricultural areas of the UK The challenge here is to reduce emissions from these areas through better management while they remain part of a productive agricultural system. This will also be of agricultural benefit as the current rate of soil-loss will rapidly turn these highly productive areas to low productivity within a generation or two. This will itself have a big impact on the agricultural economy.

It is, therefore, becoming increasingly important that different ways of managing agricultural peatlands are found.

7.6.1 Agricultural management
As with organo-mineral soils, measures that reduce soil erosion, include the use of leys and cover crops, limit the use of artificial fertilisers and reduce the impact of ruminant livestock will all reduce GHG emissions from agricultural systems on peat soils.

7.6.2 Raising water table
Productive agriculture on peatlands is made possible by drainage and most productive areas depend on sophisticated water level management through networks of drains, irrigation channels and pumping. Evans *et al* (2016) showed that a 10 cm lowering of the mean water table will increase CO_2 emissions by 3.7 t CO_2 ha^{-1} y^{-1}, but CH_4 emissions follow a counter-trend. At water table depths greater than 25 cm, CH_4 fluxes are consistently near zero, but with each 1cm rise above this they increase by 0.21 t CO_2 eq ha^{-1} y^{-1}. Maintaining water table depth at a level that maximises CO_2 sequestration but minimises CH_4 emissions is something of a fine balancing act. Using eddy covariance towers to assess carbon fluxes from a range of sites across the UK, Evans *et al* (2021) concluded that every 10cm increase in height of the water table could reduce the net impact of CO_2 and CH_4 emissions by 3 t CO_2 ha^{-1} y^{-1} or more, up to a water table depth of around 30 cm. Raising water tables still further, to 10 cm below the soil surface, retained a net cooling impact. Water tables less than 10 cm below the surface (particularly if they are high enough to result in surface water inundation), would lead to net warming impact due to the increased CH_4 emissions. So: raising the water table to between 10 cm and 30 cm below the surface may generate the optimum overall GHG emissions reductions.

Evans *et al* (2021) point out that many agricultural systems are "over-drained" (often with water tables at 2 m below the surface even when no crops are present). While changes in water table management may require new water management approaches, a halving of water table depth to an average of 1 m or less would have dramatic impacts on GHG emissions with potentially only minor impacts on existing crops.

For a summary of potential related GHG benefits, see Table 12.

7.6.3 Alternative and experimental approaches
A number of organisations are investigating alternative ways of reducing GHG emissions from agricultural peatlands. Three main issues are being addressed:

- Maximising C input — paludiculture
- Minimising decomposition
- Suppressing non CO_2 emissions (mainly CH_4)

7.6.3.1 Maximising C input — paludiculture

If water table depths are decreased to maximise CO_2 sequestration, leading to lowland peatlands becoming significantly wetter, farmers will need to develop productive and economically viable crops that can thrive in the wetter conditions. This is known as paludiculture. Mulholland *et al* (2020) reviewed a range of crops (including Sphagnum spp.) and assessed their viability in a range of potential commercial activities ranging from energy production, medicinal uses and construction. There are also a number of experiments looking at the viability of Sphagnum farming to produce media to replace peat in the horticultural sector. Paludiculture is in an early stage of experimental development, so estimates of GHG emissions reduction are uncertain. Initial findings suggest that paludiculture could have significant promise for the more sustainable use of lowland peatlands (Mulholland *et al*, 2020).

7.6.3.2 Minimising decomposition

Raising water tables reduces decomposition. In agricultural systems this could be enhanced by converting crop residues that would otherwise decompose to more recalcitrant forms of carbon. One way of achieving this is through pyrolysis to charcoal (biochar), which is then added back to the system. From a Wildlife Trust perspective, it may also be possible to convert waste residues from conservation activities (e.g. scrub removal from heathland restoration) into biochar, which could then be applied to agricultural systems which would then reduce net GHG emissions from restoration activities.

7.6.3.3 Suppressing non CO_2 emissions — CH_4

As previously discussed, CH_4 production is a significant problem when raising water tables both in agricultural systems and in the early stages of peatland restoration through rewetting. This could be prevented by the addition of gypsum ($CaSO_4$), which suppresses CH_4 production under anaerobic conditions, because SO_4^{2-} reducing bacteria outcompete methanogens for substrate (Gauci *et al*, 2002). With the right application rate, CH_4 emissions could be eliminated although further experimental work is needed to determine what this might be.

Starting Land Use		Management Change		Potential Total gain (t C ha^{-1})
Land Use (soil depth)	GHG Flux (t CO_2 eq ha^{-1} yr^{-1})	Change	GHG Flux (t CO_2 eq ha^{-1} yr^{-1})	
Arable — on peat (drained)	37.61	Decrease water table depth by 10cm	34.61	-3
		Decrease water table depth by 0.5m	22.61	-15
Intensive Grassland — on peat (drained)	27.54	Decrease water table depth by 10cm	24.54	-3
		Decrease water table depth by 0.5m	12.54	-15

Table 12: Reduction in GHG emissions from changes in water table management in drained arable and grassland systems on peat. Negative values indicate sequestration, positive fluxes = emissions. Based on a net reduction in GHG emissions of 3t CO_2 ha^{-1} y^{-1} for every 10cm reduction in water table depth to around 30cm.

CHRIS LAWRENCE

8 Conclusions

There are three clear conclusions to be drawn from this review, two of which relate to the quantity, quality and applicability of available scientific evidence and one of which relates to the interpretation and use of the evidence that is available.

8.1 The need for more evidence

The first conclusion is that **there are too few consistent studies of GHG emissions from all the habitats reviewed, which leads to considerable variation and uncertainty in interpretation of the data**. It would be sensible to take a conservative approach. For compilation and reporting of The Wildlife Trusts' own carbon accounts, this would be achieved by using the lowest Emission Factor (EF) values indicated for removals (-ve value closest to zero), and the highest EF values (+ve value furthest from zero) for emissions, identified from the literature reviewed.

In developing practical initiatives intended to reduce land-related GHG emissions or to implement nature-based approaches to atmospheric GHG removal, it would be wise to take the most conservative estimate of the benefit (avoided emissions or removals) likely to be achieved. This could be achieved by applying values for current (pre-intervention) emissions from the lower end of the indicated range (+ve value closest to zero), for current removals from the upper end of the indicated range (-ve value furthest from zero), and the reverse of these for forecast estimates of likely future conditions. Even this is unlikely to compensate fully for the high levels of variation and uncertainty in the evidence-base. Some of the current uncertainty makes it impossible to be certain (in general) whether some land use changes would result in increased emissions or increased sequestration.

The Emissions Factors and stock change values suggested here provide a useful basis for further thinking within The Wildlife Trusts, about the relationship between carbon and habitat management. The values in this review can be useful in making a first assessment of the potential for different nature-based solutions and of the potential impact of planned projects. They would be appropriate to give potential donors and other funders some initial confidence in their investments, within a "silver standard" approach (not the most stringently validated standard available, but sufficiently robust to be both practical and credible),. It is not unreasonable to use the conservative values suggested here as scientifically credible initial estimates of the potential gains that might result from implementing particular changes. This is, after all, the best evidence currently available to us. But they cannot be taken as a low-risk, reliable forecast of what will happen in any particular situation and it would not be appropriate to attempt to base a fully validated carbon offsetting or emissions reduction scheme (like the Peatland Code) on these at this stage.

There is clearly a need for a rigorous and systematic programme of additional scientific investigation across a full and representative sample range of habitat types to improve our understanding of the GHG fluxes from land-use.

The second conclusion is that **a lot of the variation in the data is because habitats are extremely variable in biotic and abiotic factors over a range of spatial and temporal scales. For some habitat stypes, it may never be possible to encapsulate this in a single Emissions Factor that would be applicable in all circumstances**.

Recommendation 1

The implication of this for The Wildlife Trusts is that **the emissions factors identified here could be used as an evidence-based starting point for communications, advice, and advocacy around nature-based solutions to climate change.** They provide pointers to how the movement might integrate climate change considerations into its habitat-related work and could be used to guide the targeting, implementation and promotion of specific carbon-related initiatives involving funders interested in achieving positive impacts for both wildlife and climate. However, we would not recommend that these values alone be used as the basis for reporting and verification of a carbon offsetting or similar scheme. There is also a clear need for further research and development work to identify and implement suitable (cheaper, simpler, more practical) approaches to the measurement of carbon fluxes associated with different habitats and habitat changes brought about by management interventions. These will almost certainly need to be developed in partnership with others, but The Wildlife Trusts have a significant role to play — particularly if credible, robust accredited (and verified) standards are going to be developed to provide investors in nature-based solutions to climate change the confidence to invest and defensible evidence of the impact achieved because of their investment.

For these habitats, direct monitoring of sites managed with a specific intention of reducing GHG emissions and/ or increasing sequestration — especially those going into codified schemes and funding arrangements designed specifically to achieve this — will be required.

Direct monitoring of GHG fluxes using chambers or flux towers is complex, time-consuming, and expensive. Therefore, if the direct measurement and demonstration of actual carbon storage and fluxes is to be rolled out more widely, accurate, pragmatic standardised methods based on proxies need to be developed. For example, in the Peatland Code, emissions are calculated based on the mapping of habitat types and erosion features. Other proxies could be based on satellite remote sensing of moisture content on peatlands.

Research would be needed to develop these proxies so that they accurately reflect real GHG fluxes.

8.2 Which types of habitat and land management activities provide the highest GHG emissions reductions?

The third conclusion is that **despite the limitations and uncertainties in the available evidence, it is possible to identify habitat and land management changes that are, on the basis of best available evidence (reviewed here), likely to generate specific quantified reductions in GHG emissions or increased removals.**

Table 13, below, and Table 15 (in Appendix 1) draw data and analysis from various places earlier in this review (Tables 9. 10, 11, 12 and 14) and synthesise from them a summary of the potential GHG emissions reductions for habitats and management interventions for which Emission Factors and stock-change estimates were found.. Low and high potential gain values are provided, with a suggestion that the low potential gain values should be used as a conservative approach to the estimation of habitat-related GHG emission reductions and sequestration within The Wildlife Trusts. Even using two approaches to the estimation of GHG benefits, generic values could not be deduced for a number of likely habitat conversions and land use changes. These will likely need to be determined individually on a case-by-case basis, supported by further monitoring, evaluation and research until such time as defensible values can be agreed (either for general application or for specific situations).

The review has shown that there are two types of GHG emissions reduction processes that need to be considered when developing a potential Wildlife Trust scheme:

- **Avoided emissions** — these will result from changes in management that reduce emissions but don't necessarily lead to habitats becoming net sinks — just smaller sources.
- **Sequestration** — where new CO_2 is sequestered from the atmosphere into storage, with habitats being net sinks of GHGs. Sequestration consists of two types:
  i. short-term gains when a habitat changes from one equilibrium state with a lower soil organic content, to a habitat with a higher soil organic carbon content. This will eventually reach a higher equilibrium state and sequestration will then cease; and
  ii. ongoing sequestration of new CO_2 from the atmosphere into long-term increasing carbon stores.

Currently available reliable evidence indicates that only three broad habitats deliver ongoing sequestration:

- Near-natural and pristine peatlands which sequester small amounts of atmospheric carbon and continue to do this over thousands of years, resulting in the build-up of very large carbon stores.
- Woodlands (and forestry depending on the fate of the harvested product) which have high sequestration rates in younger growth phases, and can store significant amounts of carbon in above-ground living biomass for centuries. Rates decline over time but this can take centuries, so woodlands can be considered as habitats with long-term sequestration potential.
- Saltmarsh which continually sequesters carbon through high sedimentation rates into long-term storage. Like woodlands, sequestration rates in saltmarsh slow as the marsh matures. The extent of saltmarsh, is, however, currently limited.

Ensuring the **protection of high nature value open habitats such as species-rich grasslands and heathlands is essential in preventing the loss of their soil organic carbon stores**. This is clearly one of the most important actions the Wildlife Trust movement take in managing the valuable species-rich habitats in its care. However, maintaining existing high nature-value habitats that are already in good condition (such as by grazing open habitats to prevent succession to scrub and woodland) is unlikely to sequester significant additional amounts of CO_2 from the atmosphere. In fact, the management activities themselves may lead to these habitats being small sources of emissions. In these circumstances, the biodiversity benefits of maintaining these habitats almost certainly outweigh GHG emissions reduction considerations, particularly as in most cases they are of limited extent, having suffered huge declines.

Altering management practices in existing high nature value habitats could reduce emissions but there is insufficient evidence to propose reliable estimates of the impact of these. Given their relatively limited extent, they are unlikely to be a significant part of the GHG emissions from the land management sector.

Conversion of arable and intensive grasslands to extensive species-rich grasslands can lead to a period of sequestration while higher levels of soil organic carbon are accumulated, following conversion, but this will tail off to a new state of equilibrium. Exactly how long this takes is unknown but is likely to be counted in decades.

The restoration of degraded peatlands offers the largest potential for emissions reductions through avoided losses, rather than from new atmospheric CO_2 sequestration.

There is **insufficient evidence to determine GHG fluxes in hedgerows, scrub, orchards, wood pasture, rivers, streams, floodplains or ponds accurately**. However, there is evidence for high levels of carbon storage in floodplain sediments and that rivers and streams are major fluvial pathways for emissions from sediments eroded from surrounding catchments. It would also seem obvious that the creation of "woody" habitats such as hedgerow or scrub should sequester more GHG, but more evidence is needed to quantify this.

Altering management practices in arable and intensive grassland systems could lead to substantial emissions reductions, given their widespread extent — particularly where these are on peat.

Recommendation 2

The implication of this for The Wildlife Trusts is that in general, the movement should accept that in the light of the best readily available evidence, **the majority of land over which they have direct management control does not contribute significantly to the ongoing direct capture and long-term storage of atmospheric carbon**.

An important part of The Wildlife Trusts' efforts to incorporate positive climate impact into their habitat management work may well be to identify and implement ways of reducing the carbon emissions associated with particular management practices (such as grazing, off-road vehicle use, scrub control, chemical use or timber extraction).

Recommendation 3

The implication of this for The Wildlife Trusts is that there are several ways in which the movement's approach to **land acquisition and the creation, restoration, and management of habitats (and the consultancy and advice services that Trusts provide) could contribute to future GHG emissions reduction and/or the increased future capture and long-term storage of atmospheric carbon.**

The restoration of damaged and degraded peatlands and the conversion of either arable or intensive agricultural grassland on mineral soils to more extensively managed wildlife habitats — particularly to broadleaved woodland and to saltmarsh in appropriate coastal locations — seem to offer the greatest potential.

This is likely to generate worthwhile carbon benefits as well as benefits for wildlife, whether implemented on land already owned and/or managed by The Wildlife Trusts, or on land owned and managed by third parties, and whether or not it is associated with a formal income-generating carbon standard or product.

In addition, significant reductions in emissions within a productive agricultural sector could be achieved through changes in management. Given that this will have an indirect beneficial effect on wildlife through climate change mitigation but only limited direct wildlife benefit, the Wildlife Trust movement will need to decide what role to play here and how far to actively engage with the wider agricultural sector on this agenda.

Land-use change or management intervention		Conservative estimate of potential GHG gain (t CO_2 eq ha^{-1} yr^{-1})			
		Potential change in GHG removal (sequestration)	Potential Change in GHG emissions (avoided emissions)	Total potential net GHG gain	
Starting Land Use	End Land Use or Intervention			Low (conservative)	High
Arable — on peat (drained)	Peatland — Rewetted Fen	0	-29.56	**-29.56**	
	Extensive Grassland — on peat (drained)	0	-24.58	**-24.58**	
	Decrease water table depth by 0.5m	0	-15	**-15**	
	Intensive Grassland — on peat (drained)	0	-10.07	**-10.07**	
	Decrease water table depth by 10cm	0	-3	**-3**	
Arable — organo-mineral and mineral soils	Mixed native broadleaved woodland (30 years)	-2.5 to -25.5	-1.62	**-4.12**	-27.12
	Mixed native broadleaved woodland (100 years)	-2 to -13	-1.62	**-3.62**	-14.62
	Middle/High Saltmarsh	-0.7 to -9.31	-1.62	**-2.32**	-10.93
	Lowland dry acid grassland — "disturbed by management"	0	-1.58 to -1.61	**-1.58**	-1.61
	Lowland meadow — "disturbed by management"	0	2.79 to -1.60	2.79	-1.60
	Lowland calcareous grassland — "disturbed by management"	0	3.34 to -0.96	3.34	-0.96
	Lowland heath — "disturbed" by management	0	23.08 to -0.31	23.08	-0.31
Intensive Grassland — on peat (drained)	Peatland — Rewetted Fen	0	-19.49	**-19.49**	
	Decrease water table depth by 0.5m	0	-15	**-15**	
	Extensive Grassland — on peat (drained)	0	-14.51	**-14.51**	
	Decrease water table depth by 10cm	0	-3	**-3**	
Extensive Grassland — on peat (drained)	Peatland — Rewetted Modified Bog	-0.02	-13.03	**-13.05**	
	Peatland — Rewetted Fen	0	-4.98	**-4.98**	
UK Intensive Grassland — organo-mineral soils	Mixed native broadleaved woodland (30 years)	0 to -25.5	2.49 to -5.25	2.49	-30.75
	Mixed native broadleaved woodland (100 years)	0 to -13	2.99 to -5.25	2.99	-18.25
	Middle/High Saltmarsh	0 to -9.31	4.29 to -5.25	4.29	-14.56
	Lowland dry acid grassland — "disturbed by management"	0	5.03 to -5.24	5.03	-5.24
	Lowland meadow — "disturbed by management"	0	9.4 to -5.23	9.4	-5.23
	Lowland calcareous grassland — "disturbed by management"	0	10.0 to -4.56	9.95	-4.56
	Lowland heath — "disturbed" by management	0	29.69 to -3.94	29.69	-3.94
Upland heath — "disturbed" by management	Mixed native broadleaved woodland (30 years)	-2.5 to -25.5	-0.35 to -23.2	**-2.85**	-48.7
	Mixed native broadleaved woodland (100 years)	-2 to -13	-0.35 to -23.2	**-2.35**	-36.2
Forestry (mainly conifer) on peat (drained)	Peatland — Rewetted Modified Bog	-0.02	-1.15 to -5.46	**-1.17**	-5.48
Forestry (mainly conifer) on organo-mineral soil	Mixed native broadleaved woodland (30 years)	26.5 to -18.6	0	26.5	-18.6
	Mixed native broadleaved woodland (100 years)	27 to -6.1	0	27	-6.1
Forestry (broadleaves) on organo-mineral soil	Mixed native broadleaved woodland (30 years)	16 to -16.9	0	16	-16.9
	Mixed native broadleaved woodland (100 years)	16.5 to -4.4	0	16.5	-4.4
Peatland — Extracted Domestic (drained)	Peatland — Rewetted Modified Bog	-0.02	-13.37	**-13.39**	
	Peatland — Rewetted Fen	0	-5.32	**-5.32**	
Peatland — Extracted Industrial (drained)	Peatland — Rewetted Modified Bog	-0.02	-13.28	**-13.30**	
	Peatland — Rewetted Fen	0	-5.23	**-5.23**	
Peatland — Eroding Modified Bog (Bare peat) (Drained)	Peatland — Heather & Grass Modified Bog (Undrained)	0	-10.97	**-10.97**	
Peatland — Eroding Modified Bog (Bare peat) (Undrained)	Peatland — Heather & Grass Modified Bog (Undrained)	0	-9.86	**-9.86**	
Peatland — Heather & Grass dominated Modified Bog (Drained)	Peatland — Rewetted Modified Bog	-0.02	-3.54	**-3.56**	
Peatland — Heather & Grass dominated Modified Bog (Undrained)	Peatland — Near-natural Bog	-0.02	-2.31	**-2.33**	
Peatland — Near-natural Bog	Peatland — "Pristine" intact bog	-0.74	0	**-0.74**	
Peatland — Rewetted Fen	Peatland — Near-natural Fen	-0.93	-8.05	**-8.98**	
Intertidal sediments	Pioneer/Low Saltmarsh	-1.65 to -28.68	0	**-1.65**	-28.68

Table 13: Summary overview of potential estimates of GHG emissions reductions from land-use change or alterations in management interventions using an Emissions Factor approach. Negative values indicate sequestration and/or avoided emissions, positive values indicate increased emissions. Highlighted boxes indicate changes where a conservative interpretation and analysis of the evidence reviewed indicates that GHG gains may be possible, but that increased emissions or reduced sequestration is also a possibility. These changes should be treated with particular caution and taken on a case-by-case basis, as a reliable generic figure for the likely resulting GHG gain cannot be deduced from the evidence reviewed, using this method. Recommended conservative estimates (in line with recommended GHG accounting practices) are indicated in **teal**. Data in this table are derived from the analysis presented in tables 9, 10 and 12 for different types of habitat or land management change.

9 Further Work

The development of a comprehensive Wildlife Trust movement-wide approach to habitats and carbon also needs to address the other two questions posed in the introduction to this report:

- How long does it take a habitat to "move" from one GHG emissions state to another?
- How do the GHG emissions change during the period of habitat creation, restoration, or management?

After a management intervention it is highly unlikely that a habitat will immediately switch from one state to another. It is more likely that this will take time and will be dependent on a range of biotic and abiotic factors specific to each habitat and potentially to each location. For example, the Peatland Code assumes it will take a minimum of 30 years to change from one peatland state to another, while the Woodland Carbon Code identifies a set of phases a woodland will go through as it matures. The GHG emissions will change as the habitat changes and it is essential that we understand the trajectory of that change to accurately calculate the emissions reduction and/or sequestration potential of a management intervention over the lifetime of the habitat.

A further literature review to try to understand the trajectory of change in GHG emissions in response to land management changes might help to inform the development of a TWT nature-based solutions scheme considerably. However, given the limited number of studies to date it is likely that there will be an equally limited number of studies that assess these trajectories of change.

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Appendix 1

The principal 'flux' method used in this study, to deduce credible generic emission factors for a range of land use and habitat changes in the UK, proved unable to identify sufficiently robust generic values for a number of transitions — especially those involving grasslands and a number of other 'open' habitats. In an attempt to plug some of the gaps in the results achievable using that method, a second 'stock-change' approach was also taken, based on reviewed evidence of carbon stock values held within different (mostly 'open') habitats.

The approach taken is explained and discussed in the main text of the review. With a few exceptions, the conclusions that can be drawn from taking this 'stock-change' approach don't differ significantly from those drawn from the main 'flux measurement' method. But for some changes of habitats and/or land use, this alternative approach does give greater clarity and may prove to be useful. To avoid confusion, Tables 14 and 15, which summarise the main points from the stock-change analysis, are presented in a separate appendix here. Table 14 shows the full detail of the upper and lower stock values found in the papers reviewed and the ranges of stock-change (in t C ha^{-1}) that might be calculated from them. Table 15 presents the upper and lower stock-change values for each habitat change in Table 14 and converts them to an implied annual GHG EF (in t CO_2 eq ha^{-1} yr^{-1}).

BARNEY WILCZAK

Starting Land Use: Land Use (soil depth)	Starting Land Use: Stock ($t\ C\ ha^{-1}$)	Management Change: Change	Management Change: Stock ($t\ C\ ha^{-1}$) Low	Management Change: Stock ($t\ C\ ha^{-1}$) High	Potential Total gain ($t\ C\ ha^{-1}$) Low	Potential Total gain ($t\ C\ ha^{-1}$) High	Average annual rate of change (20 year period unless stated)
Arable & horticulture — organo-mineral soils (15cm)	47	Dwarf shrub heath (15cm)	90		-43		-2.2
		30-year Mixed native broadleaved woodland (15cm)	72	263	-25	-216	-0.8 to -7.2 (30 years)
		100-year Mixed native broadleaved woodland (15cm)	91	403	-44	-356	-0.4 to -3.6 (100 years)
		Neutral Grassland (15cm)	69		-22		-1.1
		Improved grassland (15cm)	67		-20		-1.0
UK Arable soils — organo-mineral (0-100cm)	120	30-year Mixed native broadleaved woodland (100cm)	130	377	-10	-257	-0.3 to -8.6 (30 years)
		100-year Mixed native broadleaved woodland (100cm)	149	517	-29	-397	-0.3 to -4.0 (100 years)
		Permanent grassland, low plant diversity, intensive management (0-100cm)	403		-283		-14.2
Improved grassland — organo-mineral soils (15cm)	67	Dwarf shrub heath (15cm)	90		-23		-1.2
		30-year Mixed native broadleaved woodland (15cm)	72	263	-5	-196	-0.2 to -6.5 (30 years)
		100-year Mixed native broadleaved woodland (15cm)	91	403	-24	-336	-0.2 to -3.4 (100 years)
UK Agricultural Grassland soils — organo-mineral (0-100cm)	160	30-year Mixed native broadleaved woodland (100cm)	130	377	30	-217	1 to -7.2 (30 years)
		100-year Mixed native broadleaved woodland (100cm)	149	517	11	-357	0.1 to -3.6 (100 years)
		Permanent grassland, low plant diversity, intensive management (0-100cm)	403		-243		-12.2
Neutral Grassland (15cm)	69	30-year Mixed native broadleaved woodland (15cm)	72	263	-3	-194	-0.1 to -6.5 (30 years)
		100-year Mixed native broadleaved woodland (15cm)	91	403	-22	-334	-0.2 to -3.3 (100 years)
Permanent grassland, low plant diversity, intensive management (0-100cm)	403	30-year Mixed native broadleaved woodland (100cm)	130	377	273	26	9.1 to 0.9 (30 years)
		100-year Mixed native broadleaved woodland (100cm)	149	517	254	-114	2.5 to -1.1 (100 years)
		Permanent grassland, intermediate plant diversity, intermediate management (0-100cm)	414		-11		-0.6
Permanent grassland, intermediate plant diversity, intermediate management (0-100cm)	414	Permanent grassland, high plant diversity, extensive management (0-100cm)	446		-32		-1.6
Dwarf shrub heath (15cm)	90	30-year Mixed native broadleaved woodland (15cm)	72	263	18	-173	0.6 to -5.8 (30 years)
		100-year Mixed native broadleaved woodland (15cm)	91	403	-1	-313	-1 to -3.1 (100 years)

Table 14: Estimates of potential total carbon stock changes in terrestrial "open" land-use, from conversion to other habitats. Negative values indicate sequestration of., positive values indicate emissions. Highlighted boxes indicate changes that may lead to increased emissions. Unless otherwise stated, average annual rate of change is based on the assumptions in the GHG Reporting Protocol Agricultural Guidance (www.ghgprotocol.org) where GGP Product Standard requires that "in the context of land use change, [changes should be amortized over] 20 years or the length of one harvest, whichever is longer". Data in this table are derived from the primary evidence summarised for different habitat types and land uses, in tables 1-5 (Chapter 6)

Land-use change or management intervention		Estimated average annual rate of carbon stock change (t C ha⁻¹ yr⁻¹)		Estimated average annual rate of carbon stock change (t CO_2 eq ha^{-1} yr^{-1})	
Starting Land Use	End Land Use or Intervention	Low (conservative)	High	Low (conservative)	High
Arable & horticulture — organo-mineral soils (15cm)	Dwarf shrub heath (15cm)	-2.2		**-8.1**	
	30-year Mixed native broadleaved woodland (15cm)	-0.8	-7.2	**-2.9**	-26.4
	100-year Mixed native broadleaved woodland (15cm)	-0.4	-3.6	**-1.5**	-13.2
	Neutral Grassland (15cm)	-1.1		**-4.0**	
	Improved grassland (15cm)	-1.0		**-3.7**	
UK Arable soils — organo-mineral (0-100cm)	30-year Mixed native broadleaved woodland (100cm)	-0.3	-8.6	**-1.1**	-31.5
	100-year Mixed native broadleaved woodland (100cm)	-0.3	-4.0	**-1.1**	-14.7
	Use of cover/green manure crops	N/A	N/A	**-1.17**	
	Introduction of ley rotation	-0.027	-0.59	**-0.1**	-2.16
Improved grassland — organo-mineral soils (15cm)	Dwarf shrub heath (15cm)	-1.2		**-4.4**	
	30-year Mixed native broadleaved woodland (15cm)	-0.2	-6.5	**-0.7**	-23.8
	100-year Mixed native broadleaved woodland (15cm)	-0.2	-3.4	**-0.7**	-12.5
UK Agricultural Grassland soils — organo-mineral (0-100cm)	30-year Mixed native broadleaved woodland (100cm)	1.0	-7.2	3.67	-26.4
	100-year Mixed native broadleaved woodland (100cm)	0.1	-3.6	0.4	-13.2
Neutral Grassland (15cm)	30-year Mixed native broadleaved woodland (15cm)	-0.1	-6.5	**-0.4**	-23.8
	100-year Mixed native broadleaved woodland (15cm)	-0.2	-3.3	**-0.7**	-12.1
Permanent grassland, low plant diversity, intensive management (0-100cm)	100-year Mixed native broadleaved woodland (100cm)	2.5	-1.1	9.2	-4.0
	Permanent grassland, intermediate plant diversity, intermediate management (0-100cm)	-0.6		**-2.2**	
Permanent grassland, intermediate plant diversity, intermediate management (0-100cm)	Permanent grassland, high plant diversity, extensive management (0-100cm)	-1.6		**-5.9**	
Dwarf shrub heath (15cm)	30-year Mixed native broadleaved woodland (15cm)	0.6	-5.8	2.2	-21.3
	100-year Mixed native broadleaved woodland (15cm)	0.0	-3.1	**0.0**	-11.4

Table 15: Estimates of potential increased or decreased annual rates of stock-change resulting from land-use change or alterations in management interventions, for a number of changes involving "open" habitats (non-wooded and not on peat soils). Highlighted boxes indicate changes where a conservative interpretation of the evidence reviewed and the application of a stock-change comparison approach indicates that GHG gains may be possible, but that increased emissions or reduced sequestration is also a possibility. These changes should be treated with particular caution and taken on a case-by-case basis, as a reliable generic figure for the likely resulting GHG gain cannot be deduced from the evidence reviewed, using this method. Recommended conservative estimates (in line with recommended GHG accounting practices) are indicated in **teal**. Data in this table are derived from the analysis presented in tables 11 and 14 for different types of habitat or land management change.


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