ALIGNING THE PEATLAND CODE WITH THE UK PEATLAND INVENTORY

Chris Evans, Rebekka Artz, Annette Burden, Hannah Clilverd, Ben Freeman, Andreas Heinemeyer, Richard Lindsay, Ross Morrison, Jackie Potts, Mark Reed & Jennifer Williamson

Report to Defra and the IUCN Peatland Programme, March 2022
(Updated January 2023)

SUMMARY

The Peatland Code and the peatland elements of UK's national GHG emissions inventory both seek to determine the greenhouse gas (GHG) emissions and removals that occur as a result of land-use and land-use change. The approaches are conceptually similar, multiplying estimated areas of peatland in any given condition category by an associated set of Emission Factors (EFs) that are derived from an analysis of published flux data. The EFs used for the two approaches share a common ancestry; an initial collation of UK-relevant EFs compiled for the Peatland Code (Smyth et al., 2015) was subsequently expanded and updated (by the same core team) for the UK GHG inventory (Evans et al., 2017). However, the Peatland Code currently includes only a restricted range of restoration options and associated EFs, all of which relate to the restoration of drained or degraded bog habitats. Options for restoration of fen habitat are not currently included, and neither are more heavily modified pre-restoration habitats such as agricultural land, peat extraction sites or plantation forestry. Emission factors for these categories are included in the national inventory, but (in line with the requirements of national inventory reporting, and the constraints on data availability at this scale) generally apply single EFs to broad areas, effectively averaging out local-scale variability in site condition and management, including variations related to the effectiveness (or otherwise) of re-wetting and restoration interventions. Furthermore, the EFs used in the inventory are subject to periodic update to incorporate new data, whereas the Peatland Code EFs have not been updated since 2015.

In Section 1 of this report, we provide a comprehensive update of EFs for all Tier 2 (empirically based, country specific) peat condition categories included in the national inventory. This update incorporates new UK datasets, including a growing body of data from the UK flux tower network, as well as international data from climatically analogous regions. We also reviewed existing data and classifications used in EF database, resulting in the exclusion of a substantial number of cropland and grassland flux data for methodological reasons, and of data from 'flooded' sites (i.e. those with average annual water levels more than 5 cm above the peat surface) due to extreme levels of methane (CH_4) emissions associated with these conditions, which do not represent a desirable re-wetting endpoint.

In Section 2, we investigated the potential to include new reporting categories in the Peatland Code, including greater disaggregation of existing 'Tier 2' emissions reporting categories to reflect variations in site condition that might not be quantifiable at a national scale, such as the separation of 'modified bog' into heather, grass, rush or sedge-dominated subcategories, or the separate treatment of upland blanket bog and lowland raised bog, with different associated CO_2 and CH_4 fluxes. Unfortunately, this level of disaggregation remains impossible based on a Tier 2 approach, due to a continuing lack of sufficient GHG flux measurements from representative locations. While the growing flux tower network has greatly improved data coverage for some key categories such as cropland on lowland peat, similar coverage is not yet available for many upland bog habitat types. Resolving this problem will require concerted and coordinated investment in flux measurements to fill evidence gaps, rather than continued reliance on *ad hoc* data from individual research projects. Nevertheless, adoption of updated Tier 2 EFs for existing inventory classes by the Peatland Code should greatly broaden the range of peatland types and interventions for which climate mitigation benefits can be quantified, including the restoration of lowland fen peat.

In Section 3, we describe a proposed approach that would at least partly overcome these current data limitations to support more accurate reporting of emissions reductions resulting from restoration where detailed site level data are available. The approach is founded on a recent synthesis of eddy covariance CO_2 and static chamber CH_4 flux data from measurement sites across the British Isles (Evans et al., 2021) which derived quantitative relationships between mean fluxes of both GHGs and mean annual water table depth. The method is designed to be fully aligned with emissions inventory reporting, whereby each peat condition category has an assigned 'default' WTD at which CO_2 and CH_4 emissions will correspond to the Tier 2 EF values. Higher measured water tables will result in reduced CO_2 emissions (or greater CO_2 uptake in wetter categories) relative to the default values, whereas sites with lower water tables will have higher emissions. Conversely, CH_4 emissions increase non-linearly with rising water levels. The method also takes account of peat depth, allows peatlands to move between categories in response to restoration, and applies constraints on permitted water table values for each category to reduce the risk of misattribution (for example, classifying a site as 're-wetted' when water tables remain too deep to halt CO_2 loss). A spreadsheet-based 'carbon calculator' accompanying this report enables users to predict the carbon benefits of a restoration project in advance, or to evaluate outcomes based on measured data. The approach described is designed to offer a simple, empirically-based and inventory-aligned methodology to capture site-specific variations in both pre- and post-restoration condition. In effect, it permits the greater climate mitigation benefits of more effective restoration projects to be appropriately rewarded, potentially strengthening the economic basis for more ambitious (and expensive) restoration measures, while limiting the benefits that can accrue to less effective interventions.

Section 3.2 considers options to incorporate additional peat condition categories based on this approach. These include modified fen as a new category that is currently omitted from the UK inventory, and wet woodland and paludiculture as potential targets of interventions. At this stage, we recommend the use of a Tier 3 model-based approach to estimate pre-restoration emissions from conifer forests, and do not provide a method for reporting emissions during forest-to-bog restoration. We also recommend that large areas of permanent inundation (to a mean depth > 5 cm) be excluded from the 're-wetted' categories due to a risk of very high CH_4 emissions; such areas should ideally not form a component of Peatland Code restoration schemes, or if included that likely high associated CH_4 emissions should be taken into account in any assessment of overall climate mitigation.

In Section 3.3, we describe options and requirements for effective monitoring, reporting and verification (MRV) of restoration projects, based on a range of low cost monitoring techniques including broad-scale vegetation assessment, 'Eyes on the Bog' monitoring of water table depth and long-term peat elevation change, and the use of timelapse cameras to monitor short-term elevation change. The suite of methods described are intended to support the site-based emissions assessment method described above.

Section 3.4 considers a number of areas requiring further methodological development, including the treatment of emissions from prescribed and wild fires, site-specific estimation of N_2O emissions and waterborne carbon fluxes, and transitional changes in emissions during and after restoration. While there is some risk of elevated CH_4 emissions following restoration, we consider that transitional gains in ecosystem above and below ground carbon stocks following restoration have been overlooked, by both the Peatland Code and the inventory. These gains are theoretically large, and could result in restored peatlands sequestering CO_2 at much higher rates than are currently captured using the emission factor approach. Further empirical data is required, but if the evidence supports the inclusion of transitional carbon gains this could significantly enhance the overall climate mitigation potential of peat restoration, with the potential for net GHG removal in some circumstances.

Section 4 reviews the current status and future potential development of the Peatland Code in the context of a rapidly evolving policy environment and voluntary carbon market. A number of new UK 'Codes' are currently under development, including the Farm Soil Carbon Code, as well as multiple national and international carbon accounting tools and financial mechanisms relevant to peatlands. While the growing attention to peat restoration in carbon finance sector is welcome, there is a risk of fragmentation, overlap, methodological inconsistencies and the dilution of standards established by the Peatland Code. A key strength of the Peatland Code, which we have tried to retain in this assessment, is consistency of approach with UK national inventory reporting, which ensures credibility and alignment with government policy in this area. Given the rapid growth of carbon finance, and green finance in general, there is a need to define the future role and remit of the Peatland Code, and a number of options are presented.

Task 1. Update of Existing Peatland Code Emission Factors

1.1 Introduction

For this task, we undertook a full update of the emission factors (EFs) currently used for peatlands in the UK National Greenhouse Gas Inventory (Brown et al., 2021, see Annex A3.4.28). The Peatland Code (Smyth et al., 2015) currently provides EFs for four categories: Near Natural Bog, Modified Bog, Drained Bog, and Actively Eroding Bog. Pristine Bog was included as a placeholder, but without accompanying EFs as no data from truly pristine UK sites were considered to exist. The analysis undertaken for the Peatland Code subsequently fed into the development of EFs for the peatlands in the National Inventory (Evans et al., 2017), which was formally included in the UK's national emissions reporting for the first time for the 2019 reporting year (Brown et al., 2021). Both the Peatland Code and the National Inventory treat blanket bog and raised bog as a single peat type, due to data limitations, although they can be expected to function somewhat differently, notably in relation to their hydrology. Fens are treated as a single category for the same reason, although in reality peatlands occur along a continuum from fully rain-fed to groundwater fed, with considerable variability in key characteristics such as acidity and nutrient status.

While the treatment of peatland emissions in the Peatland Code and National Inventory are closely related, they are not identical. Key differences include the following:

1) The National Inventory includes additional EF categories spanning both bog and fen peat, and a wider range of management impacts (see below)
2) Pristine Bog (and Pristine Fen) were not included in the National Inventory
3) Due to difficulties in differentiating drained and undrained bogs sites in the source data used to derive EFs (i.e. modified bogs are typically affected by drying, either via active drainage or the indirect effects of land-use such as peat erosion) the National Inventory defined three categories of Modified Bog: Eroded, Heather Dominated and Grass Dominated. Each of these could then be defined as either drained or undrained, but differences between these subcategories were the result of increased CO_2 and CH_4 emissions via the drainage network, rather than any change in direct CO_2 or CH_4 fluxes between the peat surface and the atmosphere. Furthermore, while emissions from Heather-Dominated and Grass-Dominated Modified Bog were reported separately, they were assigned the same EFs due to a lack of sufficient GHG flux data from grass-dominated bogs (note that 'grass-dominated' in this case includes other graminoid species such as sedges, e.g. cotton grass, *Eriophorum spp.*).
4) The National Inventory 'Eroded Modified Bog' category was initially defined as areas of modified bog containing erosional features (assumed to occupy 15% of the overall area), to align with the available activity data. This created a mismatch with the Peatland Code, which reported on emissions from the erosional features themselves. While the overall outcome is mathematically the same (the Peatland Code reported higher emissions over a smaller area, and the National Inventory reported lower emissions from a larger area) this created a degree of confusion, and the National Inventory method has now been amended to report emissions from actively eroding areas specifically. We have followed the same approach for this report.
5) In the original analysis for the National Inventory (Evans et al., 2017) a single 'Rewetted Bog' category was defined. However subsequent updates to the EF database highlighted that the majority of the available data used to estimate emissions from this category came from studies involving restoration of comparatively degraded sites such as former plantation forest or grassland. These sites often retained quite high CO_2 and/or CH_4 emissions, and were considered unduly pessimistic for reporting of rewetting impacts in less degraded systems

such as ditched blanket bogs. Therefore the additional reporting category of 'Rewetted Modified Bog' was introduced, and (as an interim measure) assigned the same EFs as Near Natural Bog.

The additional reporting categories for peat included in the National Inventory are Cropland, Intensive and Extensive Grassland, Domestic and Industrial Peat Extraction, Near Natural and Rewetted Fen, and Woodland. With the exception of Woodland, all reporting follows a Tier 2 methodology, using empirically-derived EFs (i.e. a single set of estimated emissions or removals for each direct and indirect pathway of CO_2, CH_4 and N_2O emissions per category). For Woodland, the National Inventory follows a Tier 3 (model-based) approach, using the Forest Research CARBINE model. This approach provides consistency with reporting for woodlands on mineral soils, and also reflects the complexity and temporal variability of CO_2 uptake and removal over the forest management cycle. As a result of this (as well as a continuing shortage of full CO_2 balance data for UK-relevant forest systems, and methodological challenges of measuring the CO_2 balance of managed forests) we have not updated the Tier 2 EFs for Woodland in this analysis. Opportunities to include forest to bog restoration, as well as wet woodland as a restoration option, are considered in Section 3.4.

Note that (to date) a Modified Fen category has not been included in the National Inventory, due to a lack of both emissions data to derive a separate EF, and of suitable mapping information to provide activity data. This is problematic for an expanded Peatland Code, as it would be desirable to include restoration of degraded fen habitats as a potential intervention measure. Finally, paludiculture has not been included as an option for national-scale reporting, again because few flux data are available, and also because this activity currently remains too limited in extent to justify its inclusion in the National Inventory. It is also unclear whether paludiculture, as a continued productive use of peatland rather than a full restoration measure, should be included as an option in the Peatland Code. At this stage, we still do not have sufficient data to derive full Tier 2 EFs for either Modified Fen or Paludiculture categories at a broad scale. However, as options for future inclusion of these categories we describe an approach to enable reporting of GHG emissions and removals for both categories on a site-specific basis in Section 3.4.

Finally, the update of Tier 2 described here has been undertaken in parallel with an evaluation of new data collected as part of an ongoing BEIS-funded study to support the development of Tier 2 CO_2 EFs for Cropland and Intensive Grassland on wasted peat (Evans et al., 2022). Wasted peat, which typically occurs in areas of former lowland fen, occurs where the peat layer has been depleted by long-term drainage for agriculture to the extent that less than 40 cm of organic soil remain. While not immediately relevant to the Peatland Code, it is likely that some peat restoration projects will take place on areas of wasted peat. In these cases (and subject to a decision on their adoption for National Inventory reporting) we recommend that the new Tier 2 EF for cropland on wasted peat be used to define pre-restoration emissions, rather than the CO_2 EF for cropland described below, which primarily relate to areas that retain a peat cover > 40 cm. Separate EFs for grassland on wasted peat could not be derived (Evans et al., 2022). This is partly due to a lack of data, but also because mean water table depth in many grasslands is < 40 cm, differences in emissions between peat > and < 40 cm may be smaller than those for cropland.

1.2 Methods

GHG flux data collected and published in the period that had elapsed since original publication of the Peatland Code (Smyth et al., 2015) were compiled and integrated with the existing EF database, as published by Evans et al. (2017) and more recent updates to the emissions inventory led by Rebekka Artz for BEIS (for currently used EFs see Brown et al., 2021). Data classification followed the convention already in use in the UK emissions inventory, and data were used if they were collected from the temperate oceanic region of the Köppen-Geiger climate zone map (Figure 1.1), and were considered to be reasonably analogous to the UK situation in terms of their site characteristics and management. Given differences in climate, nitrogen deposition, historic and present-day management, some discrepancies are inevitable, but in most cases there remain insufficient data to support a purely UK-based approach. However, with the growing availability of data many categories are now well-populated by studies from the UK and Ireland. A higher-resolution version of the Köppen-Geiger map (Beck et al., 2018) was published since the original EF database was compiled, which led to some re-allocation of studies between regions in Central Europe.

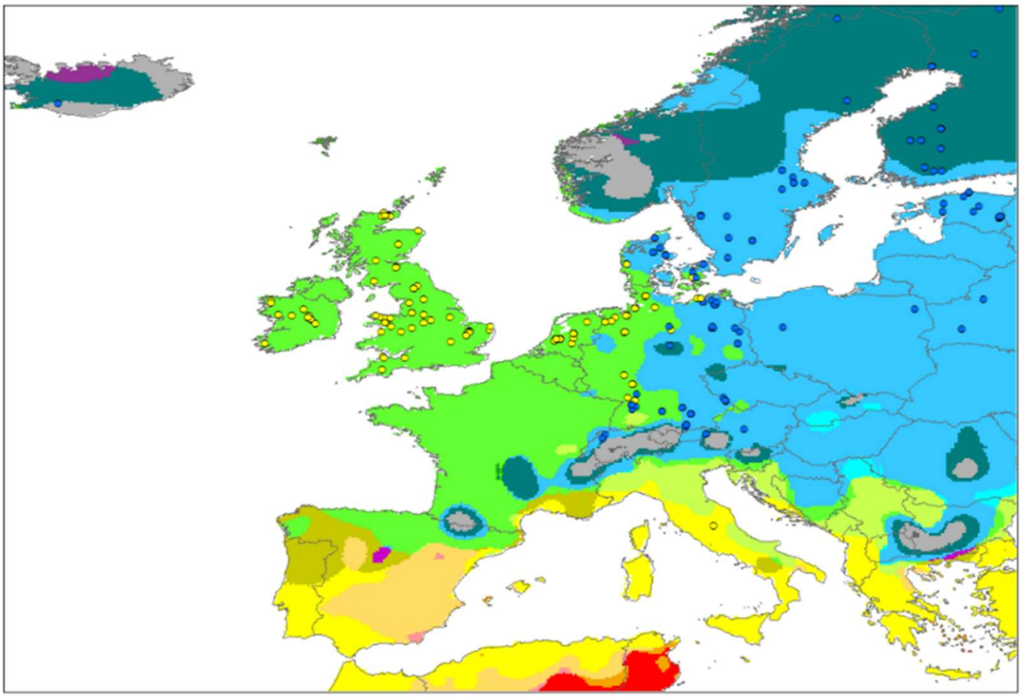

Figure 1.1. *Sites reporting peat GHG fluxes in Europe, overlaid on the high-resolution Köppen-Geiger climate zone map. Sites in the temperate oceanic zone (green) were considered climatically relevant to UK peatlands and were included in the database (yellow circles).*

We also undertook further QA assessment of some of the flux data that were included in the previous EF database update, which led to the identification of an apparently systematic positive bias in a large dataset of German chamber-based CO_2 flux estimates reported by Tiemeyer et al. (2020). These sites, which are primarily on agriculturally managed systems, had consistently higher emissions for a given water table depth when compared to most other published data, and in particular those obtained from flux tower studies, which we consider to be most reliable method of estimating annual GHG fluxes. Further investigation of the causes of this apparent bias revealed a number of issues with the modelling of CO_2 fluxes following biomass harvesting, which was undertaken manually inside the collars. The model assigned zero values of gross primary production (GPP) immediately post-harvest, which assumes a complete halt to photosynthesis at this time (something which flux tower data from grassland sites suggest is not correct) whereas ecosystem respiration (Reco) was considered to be unaffected by harvesting. Given that the net ecosystem exchange NEE of the ecosystem is calculated as the balance of Reco – GPP, these two assumptions introduce a potentially large positive bias. Paired chamber and flux tower CO_2 data from one of the sites, described in Poyda et al. (2017) support the presence of this positive bias in the chamber data, and show that they are large enough to skew annual flux estimates at agricultural sites. On this basis we made the decision to omit the Tiemeyer et al. (2020) data from the EF database. A fuller assessment of this issue is provided in the parallel assessment of emission factors for agriculturally managed wasted peat (Evans et al., 2022).

One further revision of the EF database involved the removal of a small number of sites (primarily re-wetted fen) where mean water tables were more than 5 cm above the peat surface. This decision was based on evidence (e.g. in Evans et al., 2016; 2021) that permanently inundated sites tend to have very high rates of CH_4 emission. While fen wetlands with tall emergent vegetation could continue to be peat-forming even under standing water, we currently lack data to quantify this uptake, and even where it does occur it is doubtful whether it will outweigh high rates of CH_4 emission in terms of the overall GHG balance. Very deep flooding (i.e. creation of lakes and ponds) will almost certainly impair CO_2 uptake, relative to that which could be achieved by creation of a functional peatland. On the basis that sustained flooding of restored peatlands should be avoided, we excluded these sites from the estimation of emission factors for near-natural and re-wetted bogs and fens, and recommend that restoration projects leading to this outcome over a substantial area (recognising that small areas of flooding may be unavoidable) should not be considered eligible for the Peatland Code. Further discussion of the potential treatment of these areas is included in Section 3.4

All retained data in the EF database were analysed using the same statistical method as for previous EF updates. For a description of this approach see Evans et al. (2017).

1.3 Results

Updated emission factor tables for on-site CO_2, CH_4 and N_2O emissions are shown in Tables 1.1 to 1.3. Combined GHG emissions, and including off-site CO_2 emissions and ditch CH_4 emissions (which have not been updated) are shown in Table 1.4[1]. All emission factors are now also updated to AR5 Global Warming Potentials as per the recent decision to switch to these for UK reporting (see Addendum) This table can be considered analogous to Table 1 of the Peatland Code methodology report (Smyth et al., 2015) to and Table 4.1 of the peatland inventory report (Evans et al., 2017).

For on-site CO_2 emissions, the revision of the data resulted in significantly lower EFs for agriculturally used peatland types (Table 1.1), in large part due to exclusion of a number of studies from Germany that had previously been included, as discussed above. Greater disaggregation of existing EFs was not feasible. For the modified bog category, we found no significant effect of type of vegetation (i.e. heather versus grass) cover. In part this may be because the number of measurements from grass-dominated bogs remains limited, although it is also possible that different types of grass (in reality graminoid) dominated bog may have different characteristic GHG fluxes; for example sedges such as *Eriophorum* tend to occur in wetter areas which may still be peat-forming, whereas (true) grasses are more likely to dominate in drier areas that are more likely to be losing carbon. Data from *Molinia*-dominated bog, which is widespread on the blanket bogs of Southwest England, South Wales and Southwest Scotland, some of which are undergoing restoration, remain insufficient to differentiate this potentially important category.

As for modified bog, there was no statistical support for a split in EF based on the starting land cover category before rewetting (i.e. Grassland, Woodland etc.). For Cropland and Intensive Grassland, there were insufficient data to split lowland fen peat from bog peat. This could be because peat type has a limited influence on emissions (e.g. Evans et al., 2021) but in many cases it was not possible to establish whether sites in published studies were on fen or bog peat, so firm conclusions could not be drawn. For extensive grassland, we observed a weak trend for higher emissions from fen peat than

[1] For completeness, we include alternative versions of the individual CO_2, CH_4 and N_2O emissions factor tables in Annex 1, but which incorporate data from studies with high methodological uncertainty and surface inundation in the emission factor calculations before the final QA process as described in 1.2.

bog types, however the datasets are also too small for a robust split in EFs. Data on specific interventions still remains sparse, with most observations from a very small number of sites, often clustered in one geographical location. While some evidence was captured for sites used for paludiculture, the number of studies remains very small and so these observations were grouped under the rewetted categories. A possible approach for paludiculture is described in Section 3.4.

Addendum: *During the update of the National Atmospheric Emission Inventory (NAEI) in September 2022 a further assessment was made of the source data used to derive emission factors for near-natural. A small number of sites from continental Europe were found to be exerting significant leverage on the CO_2 and CH_4 EFs for near-natural bog. On further inspection, the sites (although described as near-natural by the study authors) were observed to comprise relatively small fragments of raised bog surrounded by farmland, and therefore potentially unrepresentative of minimally disturbed UK bog habitats, which are predominantly upland blanket bog. Although the absolute differences in emissions are small, the relative differences between some categories (notably modified and near-natural bog) are more strongly affected. On this basis, and following consultation with BEIS, a decision was made to retain the existing Tier 2 EFs for near-natural bog in the current UK inventory submission. In light of this, and in order to maintain consistency between the Inventory and the Peatland Code, we have therefore retained the previous Inventory Tier 2 EFs in the tables below. An additional change was a decision by the NAEI to retain the Tier 1 emission factor for one of the component carbon fluxes for extracted peatlands. Finally, some minor differences due to rounding were observed and values in this report are now aligned with the NAEI methodology. Further work will be undertaken in 2023 to re-evaluate the data sources used to calculate EFs for near-natural bog, and potentially also modified bog and domestic peat extraction. Any proposed revisions will be presented to the Inventory Scientific Steering Group for approval. We expect that the Peatland Code EFs will be updated in late 2023 following the completion of the Inventory approval process.*

Following a decision by the UK NAEI to implement IPCC AR5 global warming potentials, we have also updated emission factors to account for the change to the global warming potentials for CH_4 and N_2O throughout the report.

***Table 1.1**. On-site CO_2 emission factors (t CO_2 ha^{-1} yr^{-1}) with standard errors and 95% confidence intervals. Note that Tier 2 values are not provided for Woodland, because the UK national inventory uses a Tier 3 model based approach for this category. Data with methodological issues were excluded (see Section 1.1). For comparative values with these included, see Annex 1. Cells shaded grey use Tier 1 EFs due to insufficient numbers of observations to derive Tier 2 EFs.*

Category	**Mean Tier 2 EF**	**Standard Error**	**Lower 95% CI**	**Upper 95% CI**
Cropland	27.06	7.94	9.58	44.54
Intensive Grassland	14.87	1.79	11.07	18.68
Extensive Grassland	11.78	3.31	4.98	18.58
Extracted (Domestic)	10.27	2.90	4.03	15.40
Extracted (Industrial)	5.44	0.84	3.69	7.19
Eroding Bare Peat[a]	5.44	0.84	3.69	7.19
Modified bog (heather or grass-dominated)	0.03	0.61	-1.17	1.24
Rewetted Modified Bog[c]	-3.54	0.75	-5.16	-1.91
Rewetted Bog	-0.58	0.80	-2.19	1.02
Rewetted Fen	-0.69	4.52	-10.07	8.69
Near-Natural Bog[b]	-3.54	0.75	-5.16	-1.91
Near-Natural Fen	-5.06	1.93	-9.35	-0.77

[a]The Eroding Bare Peat EF is the same as Extracted Industrial Peat as this is a compound EF from Bare Peat Surfaces regardless of origin of damage.

[b]Previous Inventory Tier 2 emission factors are retained for Near-Natural Bog pending further analysis of source data for the Inventory in 2023.

[c]The EF for Rewetted Modified Bog is assumed to be the same as Near-Natural Bog due to lack of data.

For on-site CH_4 (Table 1.2) and N_2O (Table 1.3), greater disaggregation of existing EFs into bog and fen types was not feasible for similar reasons; either the information could not be reliably found in the primary publications or the dataset as a whole was too small to be further disaggregated.

Table 1.2. *On-site CH_4 emission factors (kg CH_4 ha^{-1} yr^{-1}), with standard errors and 95% confidence intervals. Data from sites with flooding (mean annual WTD > 5 cm) or temporary inundation affecting the whole site were excluded (see Section 1.1). For comparative values with these included, see Annex 1. Cells shaded grey use Tier 1 EFs due to insufficient numbers of observations to derive Tier 2 EFs.*

Category	**Mean Tier 2 EF**	**Standard Error**	**Lower 95% CI**	**Upper 95% CI**
Woodland	2.50	1.61	-0.60	5.70
Cropland	1.96	1.16	-0.47	4.39
Intensive Grassland	29.03	19.10	-9.33	67.38
Extensive Grassland	35.91	16.60	2.24	69.58
Extracted (Industrial/Domestic)	42.72	19.54	-0.29	85.74
Eroding Bare Peat[a]	42.72	19.54	-0.29	85.74
Modified Bog (heather or grass-dominated)	61.75	11.62	38.34	85.16
Rewetted Modified Bog[c]	113.07	32.18	45.93	180.20
Rewetted Bog	111.11	24.36	61.80	160.41
Rewetted Fen	111.44	30.22	48.59	174.29
Near-Natural Bog[b]	113.07	32.18	45.93	180.20
Near-Natural Fen	143.25	16.16	107.69	178.82

[a]*The Eroding Bare Peat EF is the same as Extracted Industrial Peat as this is a compound EF from bare peat surfaces regardless of origin of damage.*

[b]*Previous Inventory Tier 2 emission factors are retained for Near-Natural Bog pending further analysis of source data for the Inventory in 2023.*

[c]*The EF for Rewetted Modified Bog is assumed to be the same as Near-Natural Bog due to lack of data.*

Table 1.3. *On-site N_2O emission factors (kg N_2O-N ha^{-1} yr^{-1}) with standard errors and 95% confidence intervals. Data from sites with flooding (mean annual WTD > 5 cm) or temporary inundation affecting the whole site were excluded (see Section 1.1). Cells shaded grey use Tier 1 EFs due to insufficient numbers of observations to derive Tier 2 EFs.*

Category	Mean Tier 2 EF	Standard Error	Lower 95% CI	Upper 95% CI
Woodland	1.48	0.81	-2.02	4.97
Cropland	16.28	4.42	6.98	25.57
Intensive Grassland	7.39	1.64	4.09	10.69
Extensive Grassland	1.82	0.99	-0.30	3.95
Extracted (Industrial/Domestic)	0.30	0.17	-0.03	0.64
Eroding Bare Peat[a]	0.30	0.17	-0.03	0.64
Modified Bog (heather or grass-dominated)	0.13	0.05	0.02	0.24
Rewetted Modified Bog[c]	0.00	0.00	0.00	0.00
Rewetted Bog	0.03	0.11	-0.19	0.25
Rewetted Fen	0.00	0.00	0.00	0.00
Near-Natural Bog[b]	0.00	0.00	0.00	0.00
Near-Natural Fen	0.00	0.00	0.00	0.00

[a]The Eroding Bare Peat EF is the same as Extracted Industrial peat as this is a compound EF from bare peat surfaces regardless of origin of damage.

[b]Previous Inventory Tier 2 emission factors are retained for Near-Natural Bog pending further analysis of source data for the Inventory in 2023.

[c]The EF for Rewetted Modified Bog is assumed to be the same as Near-Natural Bog due to lack of data.

Table 1.4. *Combined emission factors for all GHG source/sink pathways for each peat condition category, expressed in t CO_2 ha^{-1} yr^{-1} based on IPCC AR5 100-year Global Warming Potentials (28 for CH_4 and 265 for N_2O). Emission factors based on IPCC Tier 1 defaults are shown in italics. See also information above regarding the derivation of individual emission factors. Direct CH_4 EF corrected for (1-fraction of ditches) (see Equation 2.6, IPCC 2014)*

Peat Condition	**Drainage status**	**Direct CO_2**	**CO_2 from DOC**	**CO_2 from POC**	**Direct CH_4***	**CH_4 from Ditches**	**Direct N_2O**	**Total**
Near-Natural Bog	Undrained	-3.54	*0.69*	*0.00*	3.17	*0*	*0*	**0.32**
Near-Natural Fen	Undrained	-5.06	*0.69*	*0.00*	4.01	*0*	*0*	**-0.36**
Rewetted Bog	Rewetted	-0.58	*0.88*	*0.00*	3.11	*0*	0.01	**3.42**
Rewetted Modified Bog	Rewetted	-3.54	*0.69*	*0.00*	3.17	*0*	*0*	**0.32**
Rewetted Fen	Rewetted	-0.69	*0.88*	*0.00*	3.12	*0*	*0*	**3.31**
Modified Bog - grass/heather	Drained	0.03	*1.14*	*0.26*	1.69	*0.15*	0.05	**3.32**
	Undrained	0.03	*0.69*	*0.00*	1.73	*0*	0.05	**2.51**
Modified Bog – Eroding	Drained	5.44	*1.14*	*10.27*	1.14	*0.76*	*0.12*	**18.86**
	Undrained	5.44	*0.69*	*10.27*	1.20	*0*	*0.12*	**17.72**
Extracted – Domestic	Drained	*10.27*	*1.14*	*1.76*	1.14	*0.76*	*0.12*	**15.18**
Extracted – Industrial	Drained	5.44	*1.14*	*10.27*	1.14	*0.76*	*0.12*	**18.86**
Grassland – Extensive*	Drained	11.78	*1.14*	*0.51*	0.96	*0.74*	0.76	**15.88**
Grassland – Intensive*	Drained	14.87	*1.14*	*0.51*	0.77	*1.63*	3.08	**22.00**
Cropland (peat > 40 cm)	Drained	27.06	*1.14*	*0.51*	0.05	*1.63*	6.78	**37.17**
Cropland – wasted (peat < 40 cm)	Drained	15.98	*1.14*	*0.51*	0.05	*1.63*	6.78	**26.10**

**Note that separate EFs for grassland on wasted peat could not be derived (Evans et al., 2022). This is partly due to a lack of data, but also because mean water table depth in many grasslands is < 40 cm, differences in emissions between peat > and < 40 cm may be smaller than those for cropland.*

Task 2. Inclusion of new categories in the Peatland Code

This task was effectively subsumed into Task 1, in which we were able to materially revise the EFs across the entire suite of possible land use categories currently included in the National Inventory. These updates can serve both the UK GHG Inventory and the Peatland Code. In the case of the latter, this update permits the inclusion of a number of new pre-restoration categories covering upland and lowland peatlands, and bogs and fens: Cropland, Intensive Grassland, Extensive Grassland, Industrial and Domestic Extraction. The parallel report (Evans et al., 2022) describes new separate CO_2 emission factors for Cropland on peat > 40 cm ('extant peat') and peat ≤ 40 cm ('wasted peat'). At this stage, there are insufficient data to support a separate CO_2 emission factor Intensive Grassland on Wasted Peat, in part because mean water table depths under grassland are often < 40 cm in any case. The new emission factors can be used to determine the pre-restoration baseline for a greatly expanded range of Peatland Code restoration projects. The analysis also adds Rewetted Fen and Near Natural Fen as restoration targets, and (in line with the National Inventory) permits the use of a Rewetted Modified Bog category as a restoration target for less heavily degraded sites.

As described above, it was in general not possible (other than for wasted peat as noted above) to further disaggregate current National Inventory Tier 2 emission factor categories due to a lack of sufficient measurement data, and/or a lack of significant differentiation between prospective sub-categories based on the data that are available. However, in the following section we describe an alternative approach to emissions reporting at a site level which, while internally consistent with the Tier 2 inventory method, permits emissions of CO_2 and CH_4 to be adjusted upwards or downwards where measured site properties differ from the 'default' values that underpin the inventory. This approach also offers the possibility to estimate emissions and removals for a number of additional reporting categories, including Modified Fen and Paludiculture

Task 3. Assessment of opportunities for improved emissions reporting based on site-level data.

3.1. Introduction

In its current form, the Peatland Code assigns fixed 'Tier 2' emission factors for each peatland condition category, as described in the preceding tasks. This approach is appropriate for national-scale inventory reporting, where detailed site-specific information is not available and where the aim is to quantify the overall impacts of land-use and land-use change on peatlands. However at a project scale this approach is problematic, because it creates a 'one size fits all' situation where all interventions that lead to a change from one emission factor category to another produce the same outcome in terms of net climate mitigation, regardless of the specific attributes of the site before and after the intervention. For example, a heavily degraded modified bog that was subject to highly effective re-wetting, and additional restoration measures such as *Sphagnum* reintroduction, would apparently generate the same emissions savings as a far less effective set of interventions on a far less degraded site, despite potentially quite different outcomes for CO_2 and CH_4 fluxes; in both cases, the area would simply move from the standard Tier 2 EF for Modified Bog to that of Rewetted Modified Bog, which in the UK GHG inventory has an equivalent EF to Near-Natural Bog. An unintended consequence of this could be that 'basic' re-wetting of slightly degraded peatlands appears more economically favourable than more intensive (and expensive) restoration of heavily degraded peatlands, despite the greater potential mitigation abatement offered by the latter. At worst, some interventions that deliver genuine emissions abatement without causing the peatland to transition from one Tier 2 EF category to another would deliver no apparent abatement at all according to the current methodology. Furthermore, the limited range of condition classes in the current Peatland Code limits incentives to undertake more detailed monitoring and verification of outcomes in order to evidence higher rates of climate mitigation, or to identify areas where additional interventions are required.

At a project level, it is very likely that more detailed information on pre- and post-restoration ecosystem state are (or could be) collected than would be possible at the scale of national inventory reporting. Where site-level monitoring is sufficient to permit more accurate prediction of likely emissions changes following an intervention, this could enable the specific climate benefits of that intervention to be quantified in the Peatland Code, and for appropriate levels of payment (e.g. to reflect the greater climate benefits of a more expensive set of restoration measures) to be determined. This would also represent a change from the currently somewhat 'action based' reporting mechanism towards a fully 'outcome based' methodology, in line with the more stringent Monitoring, Reporting and Verification (MRV) requirements of carbon credit schemes.

In WP3, we describe a proposed methodology for the site-specific, outcome-based reporting of the GHG mitigation impacts of different peat restoration measures. The methodology has a clear and transparent empirical basis, is based on a low-cost MRV approaches, and aligns fully with existing inventory methods to ensure consistency between site and national-level emissions reporting.

3.2 Calculation of CO_2 and CH_4 emissions and removals pre- and post-restoration

3.2.1 CO_2 emissions and removals

To develop a site-specific method for calculating emissions impacts of restoration activities, we combined the updated 'Tier 2' EFs described in WP1 and WP2 with the empirical datasets describing relationships between CO_2, CH_4 and effective water table depth described by Evans et al. (2021) (Figure 3.1). For this study, a collation of UK and Irish flux tower data from a wide range of sites revealed a strong linear relationship between the Net Ecosystem Production of the peatland (NEP, i.e. its CO_2 balance, calculated as the sum of direct CO_2 emissions and removals from the ground surface, plus any lateral removal of carbon via biomass harvesting) and the mean annual effective water table depth (WTD*e*):

NEP = 0.4917 × WTD*e* – 6.34, R^2 = 0.90, < 0.001, *n* = 16 (Eq. 1)

Where NEP is expressed in t CO_2 ha^{-1} yr^{-1}, with positive values indicating net CO_2 emission, and WTD_e in cm, with positive values indicating a water table below the ground surface. The 'effective' water table depth is defined as whichever is the smaller of the water table depth and the peat depth (i.e. drainage below the base of the remaining peat layer is not considered to lead to additional CO_2 emissions). Note that in Evans et al. (2021) NEP was reported in units of t C ha^{-1} yr^{-1}, but here we converted values to units of t CO_2 ha^{-1} yr^{-1} (i.e. multiplying the original values by 3.667) for consistency with emissions reporting. The use of NEP in this way assumes that all harvested biomass is subsequently converted to CO_2, for example through the consumption of food or animal feed products, and therefore indirectly emitted to the atmosphere (although other outcomes are possible, for example through the incorporation of biomass in long-lived construction materials, which would reduce emissions accordingly). Note that Equation 1 suggests that peatlands act as net (direct) CO_2 sinks where WTD*e* < 13 cm, and become net (direct) CO_2 sources when WTD*e* is > 13 cm. This analysis does not however include other lateral pathways of carbon loss, namely DOC and POC leaching, which could result in peatlands that are acting as marginal net carbon sinks, in terms of their direct CO_2 exchange with the atmosphere, being small net C and CO_2 sources in reality once lateral C fluxes and off-site CO_2 emissions are considered. Revisiting these carbon pathways was beyond the scope of the current project, however, and we retained the existing methodology (used in both the Peatland Code and the National Inventory) of reporting these separately, based on existing emission factors. Options for a revised approach to these categories are discussed in Section 3.4.

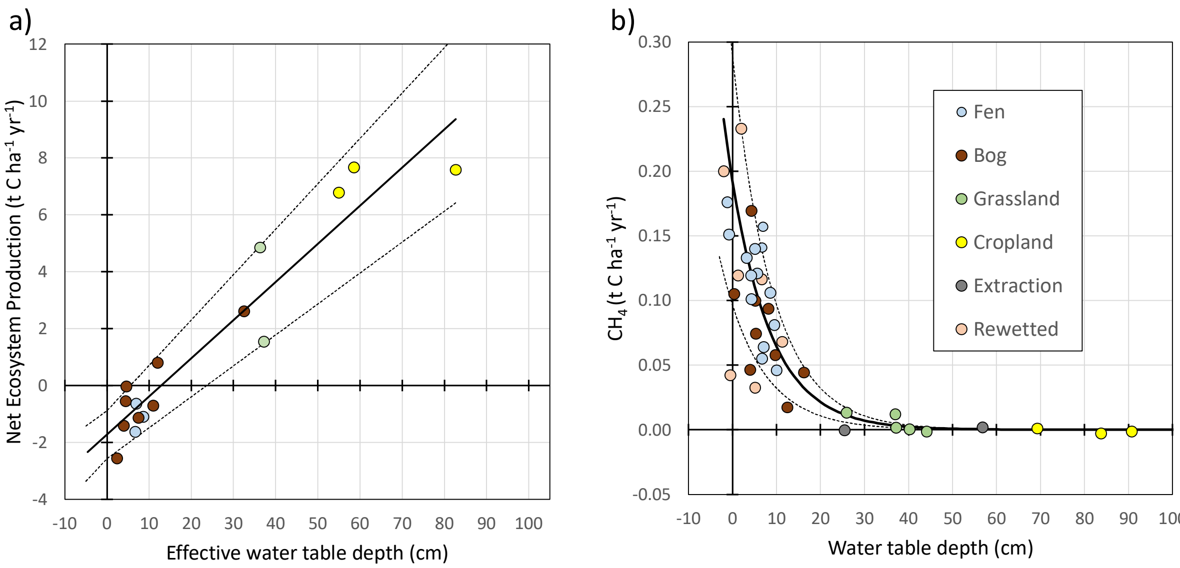

Figure 3.1. *Observed relationships between: a) annual Net Ecosystem Production (i.e. net CO_2 balance) and mean annual effective water table depth; and b) mean annual CH4 flux and mean annual water depth, for a range of peatland sites in the UK and Ireland. All data in a) are from flux tower, all data in b) are from static chambers. Figures are from Evans et al. (2021).*

To combine Tier 2 EFs with the Evans et al. (2021) approach, we reversed Equation 1 to express WTD*e* as a function of NEP, i.e.:

$$WTDe = (NEP + 6.43)/0.4917 \quad \text{(Eq. 2)}$$

Updated Tier 2 EFs for CO_2 (i.e. NEP values) from WP1 and WP2 were then used to calculate the implied mean WTD*e* value for that peat condition category (Figure 3.2). As a quality check, these predicted values were compared to observed mean WTD values (for those studies reporting data) in the EF database, shown in Annex 2. In general the predicted values correspond remarkably well with observations (Table 3.1), with implied WTD ranging from 3 cm in Near Natural Fen to 68 cm in Cropland. Some deviations are expected, given the uncertainties in both the empirical model of Evans et al. (2021) and the source data underpinning the Tier 2 EF estimates, and also the fact that only around 60% of the studies used to derive the Tier 2 EFs reported water table data. Overall we consider that this analysis demonstrates an acceptable level of consistency between the two approaches, and therefore supports the use of the Evans et al. (2021) response functions to scale Tier 2 EFs based on site-specific water level data.

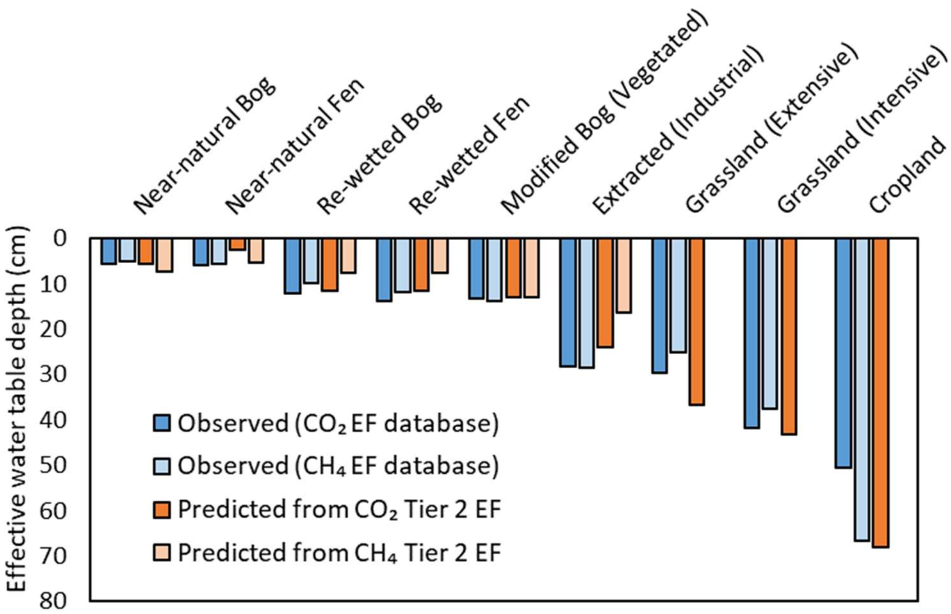

Figure 3.2. *A comparison of observed and predicted mean effective water table depth (WTDe) for Tier 2 reporting categories. 'Observed' values are the mean measured WTDe for sites reporting water table data used to derive Tier 2 EFs for CO_2 and CH_4 the emission factor database (see Section 2 and Annex 2). 'Predicted' WTDe values are those obtained by running these Tier 2 EFs through Equations 2 and 5 A close fit between observed and predicted values indicates high consistency between the Tier 2 EFs and the response functions derived by Evans et al. (2021). Note that we did not make predictions of WTD from CH_4 EFs for categories with a mean WTD > 25 cm because the non-linear relationship between CH_4 and WTD leads to very low CH_4 emissions and therefore unstable WTD predictions in this range.*

Based on this assessment, we therefore used Equation 1 to generate a set of lookup tables, which predict the CO_2 emission that would be predicted for any plausible combination of WTD*e* and peat condition category (an example is shown in Table 3.2). The tables have then been combined to create a simple spreadsheet-based 'Carbon Calculator', which accompanies this report. The methodology provides site-specific emissions estimates that are internally consistent with the Tier 2 EFs and the wider inventory methodology, and indeed could be considered as a simple empirical model-based ('Tier 3') approach to emissions reporting, where appropriate data are available.

Table 3.1. *Tier 2 emission factors for CO_2 and CH_4 derived from WP1 and WP2, with implied average effective water table depths (derived from EF $CO_{2\ (Tier2)}$ based on Equation 2); predicted average emissions of CH_4 for that effective water table depth (based on Equation 3); and the ratio of Tier 2 to predicted CH_4 emissions (RCH_4). See text for an explanation of terms and their derivation, and Tables 1.1 and 1.2 for uncertainty ranges on Tier 2 EFs. Note that the Woodland Tier 2 EF for CO_2 is taken from the previous analysis by Evans et al. (2017) but not used in the UK National Atmospheric Emissions Inventory, which uses a Tier 3 approach.*

Peat condition category	**EF $CO_{2\ (Tier\ 2)}$**	**WTD*e***	**EF $CH_{4\ (Tier\ 2)}$**	**EF $CH_{4\ (predicted)}$**	***R* CH_4**
	t CO_2 ha^{-1} yr^{-1}	cm	kg CH_4 ha^{-1} yr^{-1}	kg CH_4 ha^{-1} yr^{-1}	
Near-natural Bog	-3.54	5.7	113	138	0.82
Near-natural Fen	-5.06	2.6	143	193	0.74
Re-wetted Bog	-0.58	11.7	111	71	1.56
Re-wetted Fen	-0.69	11.5	111	73	1.53
Modified Bog - grass/heather	0.03	13.0	62	62	1.00
Modified Bog - eroding	5.44	23.9	43	19	2.30
Woodland	15.29	44.0	3	2	1.22
Extracted - Domestic	10.27	33.8	43	6	6.79
Extracted - Industrial	5.44	23.9	43	19	2.30
Grassland - Extensive	11.78	36.8	36	4	7.99
Grassland - Intensive	14.87	43.2	29	2	12.96
Cropland	27.06	67.9	2	0	13.28

The approach is relatively simple in that (in line with the conclusions of Evans et al., 2021) effective water table is considered to have an overriding role in determining CO_2 emissions, to the extent that other site attributes such as vegetation type are not considered to directly modify emissions for any given value of WTD*e*. On the other hand, the type of vegetation present on a peatland is to a substantial extent determined by the water table depth, so that for example *Sphagnum* would be unlikely to survive in very dry bogs, whereas *Calluna* is unlikely to be dominant where water table is very high. In managed agricultural and plantation forest landscapes, WTD*e* and vegetation type are both controlled by management practices but tend to be closely linked, with average drainage depths determined by the requirements of the crop.

Table 3.2. *Example lookup table for predicting direct CO_2 and CH_4 fluxes for Near-Natural Fen under a range of permitted water table depths. The 'default' values in bold type are the Tier 2 default emission factors for this category.*

	WTDe	**EF CO_2**	**EF CH_4**		**CO_2+CH_4**
	cm	t CO_2 ha^{-1} yr^{-1}	kg CH_4 ha^{-1} yr^{-1}	t CO_2e ha^{-1} yr^{-1}	t CO_2e ha^{-1} yr^{-1}
Default	**2.6**	**-5.1**	**143**	4.01	-1.05
Range	-5.0	-8.8	330	9.25	0.45
	-4.0	-8.3	296	8.29	-0.02
	-3.0	-7.8	265	7.43	-0.39
	-2.0	-7.3	238	6.66	-0.67
	-1.0	-6.8	213	5.96	-0.87
	0.0	-6.3	191	5.34	-1.00
	1.0	-5.9	171	4.79	-1.06
	2.0	-5.4	153	4.29	-1.07
	3.0	-4.9	137	3.84	-1.03
	4.0	-4.4	123	3.44	-0.93
	5.0	-3.9	110	3.08	-0.80
	6.0	-3.4	99	2.76	-0.63
	7.0	-2.9	88	2.48	-0.43
	8.0	-2.4	79	2.22	-0.19
	9.0	-1.9	71	1.99	0.07
	10.0	-1.4	64	1.78	0.35
	11.0	-0.9	57	1.60	0.66
	12.0	-0.4	51	1.43	0.99
	13.0	0.0	46	1.28	1.33

As a consequence of these associations, the range of WTD*e* values that can be selected for a given peat condition class in each of the lookup tables is constrained to only include values that could realistically arise under that land cover class (Table 3.3). Near-natural bog and fen are not permitted to have WTDe > 13 cm on the basis that (according to Equation 1) these areas can no longer be considered to be acting as direct CO_2 sinks, and are therefore modified (note that this water table threshold will be around 12 cm if lateral C losses are also included in the peat carbon balance). Similarly, sites could only be considered re-wetted if WTD < 20 cm (although ideally these sites would also have WTD < 13 cm). We also constrained the maximum WTD*e* value that could be assigned to all drained landcover classes (namely grassland, cropland, woodland, peat extraction areas and eroded bog) to 100 cm. This constraint reflects the limitations of the data used to parameterise the relationship of Evans et al. (2021), but also the possibility that CO_2 emissions may level off under very deep drainage as microbial respiration becomes limited by extremely low soil moisture levels (e.g. Renger et al., 2002; Tiemeyer et al., 2020). This constraint also avoids the risk of users of the Peatland Code claiming excessive CO_2 mitigation benefits from raising water levels in peatlands that have previously been exposed to extreme levels of drainage. Finally, we constrained all lookup tables covering near-natural and rewetted categories to a minimum WTD*e* value of -5 cm (i.e. average water levels 5 cm above the peat surface). Again, this partly reflects the limitations of the data used to parameterise the relationships described in Evans et al. (2021), but in addition it recognises that further increases in net CO_2 sequestration are highly unlikely to occur as peatland become increasingly inundated (also potentially leading to very high CH_4 emissions). Treatment of areas with higher water levels is discussed in the Section 3.2.4.

Table 3.3. *Defined ranges of minimum and maximum plausible effective water table depths for each Peatland Code category. Note that deep-drained categories are included here in order to estimate the emissions from pre-restoration land-use.*

Category	WTD*e* Min (cm)	WTD*e* Max (cm)	Justification
Near-Natural Bog	-5	13	Additional peat formation not anticipated to occur at WTD*e* < -5 cm, no peat formation anticipated at WTD > 13 cm.
Near-Natural Fen	-5	13	Additional peat formation not anticipated to occur at WTD*e* < -5 cm, no peat formation anticipated at WTD*e* > 13 cm.
Rewetted Bog	-5	20	Additional peat formation not anticipated to occur at WTD*e* < -5 cm; peat with WTD*e* > 20 cm cannot be considered re-wetted
Rewetted Fen	-5	20	Additional peat formation not anticipated to occur at WTD*e* < -5 cm; peat with WTD*e* > 20 cm cannot be considered re-wetted
Modified Bog (grass/heather)	5	50	Modified bog unlikely to be strongly peat forming (WTD*e* minimum value of 5 cm limits potential uptake to a maximum of 3.9 t CO_2 ha^{-1} yr^{-1}); bog vegetation likely to be lost with WTD*e* > 50 cm (assign site to grassland or woodland)
Modified Bog (eroding)	14	100	Eroding bog cannot be peat-forming (WTD*e* minimum value of 14 cm ensures that net CO_2 uptake cannot occur); no data on CO_2 emissions with WTD*e* > 100 cm, but emissions expected to level off under extreme drying
Modified Fen[1]	5	50	Modified fen unlikely to be strongly peat forming (WTD*e* minimum value of 5 cm limits potential uptake to a maximum of 3.9 t CO_2 ha^{-1} yr^{-1}); fen vegetation likely to be lost with WTD*e* > 50 cm (assign site to grassland or woodland)
Woodland (excluding commercially harvested plantations)[2]	-5	100	Wide range permitted as category encompasses a range of woodland types from wet fen woodland and birch woodland to drained conifer plantation. Maximum WTD*e* set to 100 cm as above.
Extracted (domestic)	14	100	Limits set as for eroding bog
Extracted (industrial)	14	100	Limits set as for eroding bog
Grassland (extensive)	30 (14)	100	Lower limit of WTD set to 20 cm, as this is considered to be the shallowest level of drainage that could support extensive grassland. However WTD*e* may be < WTD in wasted peat, with a minimum value of 14 cm to ensure that net CO_2 uptake cannot occur. Upper limit of 100 cm set as above.
Grassland (intensive)	30 (14)	100	Lower limit of WTD*e* set to 20 cm, as this is considered to be the shallowest level of drainage that could support intensive grassland. However WTD*e* may be < WTD in wasted peat, with a minimum value of 14 cm to ensure that net CO_2 uptake cannot occur. Upper limit of 100 cm set as above.

Cropland	30 (14)	100	Lower limit of WTD set to 30 cm, as this is considered to be the shallowest level of drainage that could support cropland. However WTD*e* may be < WTD in wasted peat, with a minimum value of 14 cm to ensure that net CO_2 uptake cannot occur. Upper limit of 100 cm set as above.
Paludiculture[1]	-5	30	Lower limit set to -5 cm (consistent with near-natural and re-wetted bog and fen). Upper limit set to 30 cm as deeper WTD values would not be considered paludiculture (assign site to cropland)

[1]Note that Modified Fen and Paludiculture do not have a Tier 2 EF due to lack of sufficient data to derive a category-specific EF, but are included here based on the response functions shown (Equations 1 and 4) to permit reporting of emissions based on measured water tables. For details see Section 3.2.4.

[2]Woodland is included as a potential category to allow for the possible inclusion of some wooded categories such as birch woodland or scrub (as pre-restoration categories) and wet carr woodland or wet willow coppice for paludiculture (as possible post-restoration categories); see Section 3.2.4. Conventionally managed plantation forest on peat is reported in the UK emission inventory using a Tier 3 approach based on the CARBINE model, to account for large variations in CO_2 uptake and removals over the management cycle. A similar approach is recommended to estimate pre-restoration emissions for forest-to-bog restoration.

3.2.2 CH_4 emissions

To generate site-specific estimates of CH_4 emissions from the peat surface, we again combined the Tier 2 EF approach with the empirical relationship between CH_4 and WTD obtained by Evans et al. (2021) for a dataset of chamber-based CH_4 flux measurements undertaken at UK and Irish sites. Note that actual WTD rather than WTD*e* is used for CH_4, because methane emissions are primarily determined by the depth at which anaerobic conditions occur, rather than the carbon content of the soil. Again, the original equation has been converted to standard units used for UK and IPCC inventory reporting of kg CH_4 ha^{-1} yr^{-1}:

$CH_4 = 445.3 \times 0.5^{(WTD+5)/6.31}$ Predicted vs observed $R^2 = 0.55$, $p < 0.001$, $n = 41$ (Eq. 3)

Note that this emission factor does not incorporate emissions from drainage ditches, which are accounted for separately in inventory methodology (see below) and is not applicable to inundated sites (mean water table > 5 cm above the peat surface) which are likely to have high emissions. It also does not incorporate the global warming potential (GWP) of CH_4, which is typically expressed as a ratio describing the relative warming impact of a specified gas relative to CO_2 on a mass basis and conventionally over a 100 year time horizon. The UK emissions inventory now uses a GWP of 28 for CH_4, based on the IPCC AR5 report. However, the most recent IPCC AR6 report (IPCC, 2021) recommends a GWP of 27.2 for methane derived from biogenic sources such as agriculture, on which basis 100 kg CH_4 ha^{-1} yr^{-1} equates to 2.72 t CO_2e ha^{-1} yr^{-1}. The use of GWPs (which were primarily intended to describe a pulse emission of a pollutant) to describe steady emissions from a natural or restored ecosystem is somewhat problematic, and range of other metrics have been developed to describe the warming impact of CH_4 (e.g. Neubauer and Megonigal, 2015; Cain et al., 2020). For this reason, and consistent with the IPCC Inventory approach, we report data in kg CH_4 ha^{-1} yr^{-1}. The accompanying carbon calculator allows different GWPs to be specified by the user.

To derive a specific CH_4-WTD function for each peat condition category, we firstly took the Tier 2 EF for CH_4 (CH_{4_Tier2}) obtained for the same peat condition category in WPs 1-2. Secondly, we used the implied WTD obtained from the Tier 2 EF for CO_2 (as described above) to generate a predicted CH_4 emission ($CH_{4\_predicted}$) based on Equation 2. This value can be considered the average CH_4 emission that would be expected for that WTD value ($CH_{4\_predicted}$), based on the broad dataset compiled by Evans et al. (2021). We then calculated the ratio of CH_{4_Tier2}/ $CH_{4\_predicted}$, hereafter termed RCH4, as an index of the relative tendency of that peat condition category to emit CH_4 at any given water table. Where $RCH_4 > 1$, this implies higher-than-average CH_4 emissions (e.g. due to the presence of aerenchymatous plant species which facilitate CH_4 transport from the anaerobic zone of the peat to the atmosphere, or to nutrient enrichment). Where $RCH_4 < 1$, this suggests a relatively low propensity to emit CH_4. For each category we assumed that RCH_4 remained constant over all water table depths, and rescaled Equation 3 accordingly, as:

$$CH_4 = 445.3 \times 0.5^{(WTD+5)/6.31} * RCH_4 \quad \text{(Eq. 4)}$$

As for CO_2, this equation can also be reversed in order to calculate the inferred WTD associated with each Tier 2 EF for CH_4:

$$WTD = 5 - 20.96 \log_{10}(CH_4/445.3) \quad \text{(Eq. 5)}$$

This equation was used (as a quality check only) to derive predicted WTD values as shown in Figure 3.2, for comparison to values predicted from CO_2 EFs, and observed values from the EF database.

A key assumption behind the approach used is that the mean WTD value for the source dataset used to derive EFs for CO_2 is the same as the mean WTD of the source dataset used to derive EFs for CH_4. For categories represented by a large number of studies, and/or where the same studies were used to derive both EFs, this assumption can be expected to hold. However for categories where we were reliant on fewer studies, or where the source data for CO_2 and CH_4 came from substantially different sites and datasets, there is risk that mismatches in the mean WTD of the source data could introduce biases in the model. Incomplete reporting of WTD values (just under 60% of the studies used in the emission factor database reported mean WTD values) limited the extent to which we could evaluate this issue. However for those sites that did report values, the correspondence was generally very good (compare blue bars in Figure 3.2; for data see Annex 2). For the wetter condition categories (those with WTD < 20 cm), measured WTDs tended to be quite similar between studies used to derive CO_2 fluxes and those used to derive CH_4 fluxes. There was a slight tendency for studies of CH_4 fluxes to be undertaken at slightly wetter sites than those used for CO_2 flux studies, but differences were small (< 1 cm for near-natural bog and fen, around 2 cm for re-wetted bog and fen; Annex 2).

For land-use classes with WTD > 25 cm, CH_4 emissions are consistently low and the non-linear relationship between CH_4 and WTD becomes insensitive to changes in WTD. Therefore we did not directly estimate RCH_4 from these data. Instead, we assumed that at higher water levels (i.e. where CH_4 emissions start to become significant) these land-classes would converge towards the most relevant semi-natural analogue, and could be assigned the RCH_4 value for this category. For cropland, intensive grassland and extensive grassland the semi-natural analogue was considered to be re-wetted fen, and for eroded and extracted peat it was considered to be re-wetted bog (note that this is consistent with assumptions used in the current inventory).

The values of RCH_4 obtained for wetter habitat types (Table 3.1) appear broadly plausible. Near-natural and Modified Bogs both have RCH_4 values at or below 1, implying that they broadly adhere to the general relationship shown in Equation 3 and Figure 2.1. Re-wetted Bogs, on the other hand, have a higher RCH_4 of 1.59, indicating that CH_4 emissions tend to be slightly higher than would be predicted from WTD alone. This seems to be in line with observations suggesting that re-wetted sites can sometimes have elevated emissions, for example due to colonisation of the site by cotton grass

(*Eriopohorum*), which acts as a conduit (so called 'shunt species') for CH_4 from the water table to the atmosphere (e.g. Cooper et al., 2014). There is some uncertainty as to whether this is a long-term or a transient effect. However, if a site undergoes sufficient ecological recovery to be reclassified as Near-Natural Bog, the associated RCH_4 would fall to 0.82 (for further discussion of the possible treatment of transitional effects of restoration, see Section 3.4). For Near-Natural Fen, the RCH_4 value is quite low (0.74), suggesting that natural fens do not emit CH_4 to the extent that their very high characteristic water levels would suggest. Re-wetted Fens have a higher RCH_4 of 1.53, similar to re-wetted bogs.

Overall, and subject to the uncertainties noted above, the approach taken appears to provide a plausible representation of the expected interactions between vegetation type, associated nutrient status, and drainage depth. We therefore used this approach to generate to predict site-specific CH_4 emission as a function of readily measureable site attributes (WTD*e* and vegetation) using the lookup table approach as described for CO_2 (e.g. Table 3.2) and all tables are included in the accompanying Excel file.

3.2.3. Quantifying changes in GHG fluxes as a result of restoration

In order to assess the site-specific impact of a restoration project on net GHG fluxes, it is necessary to compare the post-restoration to the pre-restoration state. Ideally this should be carried out *post hoc*, based on a comparison of measured data collected over at least a year before the intervention, and over a period of several years afterwards. However an initial prediction of expected GHG benefits of an intervention may be undertaken following an initial site assessment, and a prediction of the expected outcomes in terms of water table and vegetation changes. Assessments should be undertaken at a project-wide scale and based on a robust site survey and measurement programme, as described in Section 3.3. For direct emissions and removals of CO_2 and CH_4 (which typically comprise the largest components of the overall GHG budget) the approach described in the preceding sections offers a basis for more accurate, site-specific estimation of emissions abatement.

At the simplest level, emissions abatement can be calculated as a 'step change' from the pre-restoration state to the post-restoration state. In this case, restoration impacts are assumed to be instantaneous, and subsequently to remain constant over time. The net GHG impact of the changes is therefore calculated (for each area undergoing a restoration intervention) as:

$$\Delta EF\ CO_2 = EF\ CO_{2\,(post)} - EF\ CO_{2\,(pre)} \qquad \text{(Eq. 6)}$$

$$\Delta EF\ CH_4 = EF\ CH_{4\,(post)} - EF\ CH_{4\,(pre)} \qquad \text{(Eq. 7)}$$

Where ΔEFX represents the emission factor differential between the pre- and post-restoration emission factors, $EFX_{(pre)}$ and $EFX_{(post)}$ respectively, calculated from peat condition class and effective water table depth using Equations 2 and 4, or obtained from the lookup tables. As noted above the change in emission factor for CH_4 may be expressed in t CO_2 eq ha^{-1} yr^{-1} by multiplying the GWP of CH_4, and dividing by 1000 to convert kg to t. For restoration projects containing areas of more than one peat condition category either before or after restoration (or involving more than one intervention) this calculation should be carried out for each discrete area in turn, and the total mitigation benefit per unit area (for direct CO_2 and CH_4 emissions) can be calculated as:

$$\Delta E(CO_2 + CH_4) = \sum_{i=1}^{n} \Delta EF\ CO_2 i\ \times A_i + \sum_{i=1}^{n} \Delta EF\ CH_4 i \times A_i \qquad \text{(Eq. 8)}$$

Where $\Delta E(CO_2 + CH_4)$ is the total emission change in t CO_2eq yr^{-1} resulting from a restoration project comprising i = 1 to n combinations of pre- and post-restoration peat condition, each with an area A_i in hectares, and with ΔEFX_i calculated for each of these areas using Equations 6 and 7.

In principle this approach can also be expanded to include direct N_2O emissions, ditch CH_4 emissions and indirect CO_2 emissions from DOC and POC. In future it may be possible to derive reliable response functions to describe the relationships between these fluxes and measureable site properties (see below) but at this stage we recommend continued use of a Tier 2 emission factor approach based on site condition class to estimate these fluxes. A Tier 2 approach has been included for N_2O in the spreadsheet calculator tool which accompanies this report. Ditch CH_4 emissions and indirect CO_2 emissions from DOC and POC will be added to the calculator in future.

Where a time series of post-restoration water table depth and vegetation data have been collected (see below) a more sophisticated approach may be possible, enabling annual reporting of emissions mitigation benefits. The basic approach is the same as that outlined above, but calculations would be made on an yearly basis based on annual mean measured water tables, as well as observed changes in vegetation community where these are sufficient to support a change in peat condition category. For example, gradual hydrological recovery of the ecosystem might lead to a steady reduction in mean WTD (reducing CO_2 emissions or increasing CO_2 uptake, with some accompanying increase in CH_4 emissions) whilst vegetation reestablishment and change (either via natural succession or active planting) could lead to increased *Sphagnum* cover, sufficient to convert an area from Modified to Near-Natural Bog (and thereby likely reducing CH_4 emissions). In theory, this approach could also be used to capture any reversal of initial restoration benefits, for example as a result of the failure of a high proportion of dams in a rewetting project, ensuring that any claimed GHG mitigation benefits are subject to robust Monitoring, Reporting and Verification (MRV) and that project managers are incentivised to maintain site condition over the long term.

3.2.4. Incorporation of additional peat condition categories

It would be desirable to develop discrete EFs and calculators for a wider range of restoration targets, potential intermediate stages of restoration, and different final outcomes for peatland vegetation communities that would likely affect GHG fluxes (e.g. *Sphagnum* dominance vs sedge dominance). Unfortunately, our ability to do this remains constrained by the limited number of flux measurements for many peat condition classes, even at the coarse level of categorisation described by the current Tier 2 emission factors. Even for categories with a larger number of observations, a high level of within-category variability often makes it difficult to justify further subdivision. However, the strong overall relationships between GHG emissions and water table depth, as described in Evans et al. (2021) offer the possibility to include some additional categories based on an alternative analysis of the existing data.

3.2.4.1. Modified Fen

The lack of a 'modified fen' category from the emissions inventory has been identified as problematic because it does not permit emissions reductions resulting from fen restoration activities to be quantified. The reasons that modified fen was not included in the original peat inventory are firstly that insufficient flux data were available to define an EF; secondly that fen vegetation is intrinsically variable, making generalisation difficult; and finally that the land cover mapping data used to derive activity data for the inventory do not differentiate between different fen types, so national-scale reporting for this category was not possible. However, at a project level it should be possible to characterise fen condition, and to measure key attributes such as vegetation type and water table depth. On this basis, we have included a provisional Modified Fen category in the carbon calculator. This simply represents an extension of the existing Near Natural Fen category (including the same RCH_4 value) but with lower permitted water levels. While only a basic representation of this potentially complex condition category, the inclusion of Modified Fen should enable project managers to derive an estimate of GHG mitigation resulting from measures that raise water levels under existing fen vegetation.

3.2.4.2. Paludiculture

A second important category for which separate Tier 2 EFs cannot yet be developed is paludiculture. Paludiculture encompasses a wide range of productive uses of peatlands managed with high water tables, such as reed cultivation for thatch or other building materials, *Sphagnum* production for horticulture, and potentially the cultivation of novel crops for food or medicinal use (Mulholland et al., 2020). Tree cultivation on wet peat (e.g. willow withies for basket making or short rotation coppice management for biomass production) may also be considered as paludiculture crops. In future this category could also be expanded to include 'carbon farming' for GHG removal (see below). Paludiculture activities are likely to involve different peat types (fen vs bog) and different water management, so generalisation is difficult, and data on the GHG impacts of paludiculture management remain sparse. A small number of flux measurements from paludiculture studies were included in the derivation of Tier 2 EFs for Re-wetted Bog and Re-wetted Fen, but we are not yet able to derive paludiculture-specific EFs. However, where required we recommend that the GHG mitigation impacts of paludiculture may be estimated using EFs from the most appropriate re-wetted category, and using the emissions calculator spreadsheet with site-specific water table data where available. For CO_2, the calculated emissions should be adjusted for any biomass harvesting, because biomass carbon removed from the site will no longer be available to contribute to peat accumulation. This can be included as an annual flux (either in the case of an annually harvested crop, or as an average removal rate for a crop harvested less frequently) or as a time series based on annual management data.

3.2.4.3. Wet Woodland

As noted earlier, the UK national inventory uses a Tier 3 approach for estimating emissions from woodland, based on the Forest Research CARBINE model. This approach is required for annual reporting of emissions from managed plantation forests in particular, because their annual carbon balance changes greatly (potentially from CO_2 source to CO_2 sink and back) over the planting, growth and harvesting cycle. If forest-to-bog restoration projects are to be included in the Peatland Code, we recommend that the same approach is used to derive pre-restoration emissions, as well as a 'counterfactual' simulation of emissions and removals that would have occurred if plantation forest had remained on the site. At present, however, we do not have sufficient data to derive specific EFs for re-wetted peatlands on former forest. These are also likely to have distinct and temporally variable emissions and removals, for example due to the decomposition of brash after tree removal (Rigney et al., 2018), and may require a long period of time and/or active intervention to re-establish a functional peat-forming ecosystem. Consequently, a more sophisticated approach to quantify the net GHG impacts of forest-to-bog restoration is likely to be required than is provided here.

For other woodland types, however, it may be possible to apply a version of the simple carbon calculator approach described here. Wet (carr) woodland on fen peat is natural peat-forming ecosystem, which is still present in some areas such as the Norfolk Broads, and may be a target for restoration (for example associated with beaver reintroductions). While we still lack direct flux measurements from UK wet woodlands, Near-Natural or Re-wetted Fens could provide reasonable analogues where site-specific data are available. Given that wetland-adapted tree species such as alder have been shown to act as conduits for CH_4 emissions via their stems (e.g. Pangala et al., 2013) it may be appropriate to use the higher RCH_4 value associated with Re-Wetted Fen as a precautionary approach. In wet woodland ecosystems with periodic biomass removal (for example short-rotation coppice willow on wet peat), the annual CO_2 balance should be adjusted accordingly, as described above for other forms of paludiculture.

3.2.4.4. Flooded Land

The empirical relationships between CO_2, CH_4 and water table depth developed by Evans et al. (2021) were truncated at a WTD value of -5cm, i.e. data from sites where mean annual water table was more than 5 cm above the peat surface were excluded. The reasons for this were, firstly, that CO_2 uptake is not expected to continue to increase linearly with increasing inundation – at some point water depths are likely to become too great for emergent plant growth and peat formation. Secondly, available data suggest that CH_4 emissions can be extremely high from inundated peatland, which risks negating the carbon benefits of restoration (deeply flooded sites were also excluded from the emission factor database analysis for this reason). The IPCC 2019 inventory refinement (IPCC, 2019) does now include a reporting methodology for 'Flooded Lands', including constructed waterbodies, which could provide a basis for reporting emissions from flooded areas of peat restoration sites. However, flooding of re-wetted peatlands is not considered a desirable outcome from either a GHG or an ecological perspective, and based on a precautionary approach, we recommend that large-scale inundation of restored peatlands (to a depth of > 5 cm) should be avoided where possible. Where some degree of inundation may be unavoidable, for example in areas of uneven topography where it occurs as part of wider rewetting of the landscape, we recommend that the flooded areas are mapped, and (as a minimum) excluded from any calculation of overall GHG benefits of restoration. Note that this still risks creating hotspots for CH_4 emission that are not accounted for, so it should be avoided (or remedial action taken) where possible, or where unavoidable the resulting emissions should be estimated.

3.2.4.5. Restoration involving topsoil removal

Some restoration projects, particularly those aiming to restore former agricultural land to wetland, involve removal of topsoil as a means of removing nutrients and undesirable plant species from the site. This removal typically also involves the removal of a very large quantity of soil organic carbon from the ecosystem, which may be disposed of by application to farmland elsewhere. While not all of the removed carbon will be immediately oxidised to CO_2, it is highly likely that any peat-derived organic matter transferred to an aerobic soil environment elsewhere will eventually decompose leading to high off-site CO_2 emissions following restoration of the site. Any project involving topsoil removal should therefore aim to determine the fate of removed soil carbon, and to factor the resulting CO_2 emissions into an assessment of the overall GHG impact of restoration. It is highly likely that this will reduce or even negate the net climate benefits of restoration. While topsoil removal may be justified in terms of habitat and biodiversity benefits, we therefore advise extreme caution before including this measure in any project seeking support via the Peatland Code.

3.3. Monitoring, Reporting and Verification

In its current form the IUCN Peatland Code ensures that carbon benefits from participating projects are measured and monitored (based on indirect measurements of peat status that can be linked to empirically-based emission factors) for a minimum of 30 years following restoration interventions. At present, only blanket and raised bogs with a minimum peat depth of 50 cm, classified as drained and/or eroding, and/or with plantation forest removal planned, are eligible for inclusion into the Peatland Code. The current monitoring guidance is primarily based around measuring peat depth, and qualitative assignment of each monitoring location to a Peatland Code condition category.

This update to the recommended monitoring and reporting will allow the carbon and GHG benefits from additional restoration activities across a wider range of sites to subject to robust monitoring, reporting and verification (MRV). Ideally sites should incorporate a monitoring strategy that includes before versus after and control versus intervention (BACI) monitoring, to ensure that reported benefits can be unequivocally attributed to the restoration activity. This would require that, in addition to monitoring sites before and after restoration, as is the case at present, sites will include a control (or business as usual) area where restoration does not take place simultaneously. We recognise that this may not be possible in all cases, however, so this should be considered desirable rather than obligatory. Similarly, project-level measurements of change in carbon fluxes or stocks based on a BACI design would be highly beneficial in demonstrating real carbon and GHG benefits, but are unlikely to be affordable in many cases.

Given the considerable sums of public and private sector investment in peat restoration for climate mitigation, it seems reasonable to expect that a small proportion of the total project cost be invested in robust outcome monitoring, to ensure that the claimed mitigation benefits are genuine. At the same time, available budgets, as well as staffing capacity and skills, are likely to preclude full 'research grade' monitoring in most cases. For this reason, we propose that MRV methods should (as a minimum) follow the protocols set out in the IUCN's Eyes on the Bog monitoring guidance (Lindsay et al., 2019). At their most basic level, the Eyes on the Bog methods aim to provide sites with a low-cost, quantitative method to monitor change in vegetation (and hence infer peat condition category), peat stocks, and to provide information on water table depth in order to be able to modify the default emission factors to the conditions specific to the site. In this report (rather than replicate the existing guidance) we provide an overview of how this approach may be used to support the updated and expanded Peatland Code emissions reporting methods described above, as well as highlighting opportunities for enhanced monitoring of key environmental variables such as water table depth.

3.3.1. Vegetation monitoring

At present Peatland Code monitoring only requires that broad peatland condition assessment is made. As described above EFs are now available for a wider range of land uses on peat, including a larger number of both pre- and post-restoration categories. These may be more easily identified if a more detailed measure of vegetation cover is recorded. Being able to determine the trajectory of change in vegetation cover is an indicator of how effective restoration actions are (in terms of their impacts on hydrology and carbon cycling) across a site, even if there may not be sufficient change to suggest a change in categorisation. For example a reduction in *Calluna vulgaris* cover following ditch blocking would suggest that a site is getting wetter. If the 'carbon calculator' approach described above is applied at a project level, these broad-scale changes in vegetation may be used (together with point-scale measurements of water table depths) to infer changes in mean water table depth, as well as potential shifts between reporting categories, at the whole-project scale.

A rapid assessment methodology to allow bog condition assessment based on vegetation cover was developed by UKCEH as part of the Welsh Government's Sustainable Management Scheme (Burden et al., 2020). In this method the percentage cover of the main vegetation groups present within a $10m^2$ quadrat are recorded and then the condition category can be ascribed using the flow chart in Figure 3.3. At present, this scheme is primarily (although not exclusively) developed to support classification of blanket bog habitats, and in future it would be beneficial to develop a similar scheme for fens (including localised areas of fen vegetation within blanket peat landscapes) and some additional categories (such as scrub woodland) could be added for lowland raised bogs. In addition, many of the categories shown in Figure 3.3 (namely *Molinia*, Heather-dominated, Grass-dominated, Sedge-dominated and Rush-dominated) all fall within the single Modified Bog (vegetated) emission factor category, limiting the extent to which emissions from these classes can be disaggregated. However to the extent that these categories provide indirect information on mean water table depths, they may be useful in parameterising the carbon calculator approach at a project-wide (i.e. landscape) scale.

The classification scheme focuses on measurable site vegetation attributes, and therefore does not specifically take account of previous interventions such as re-wetting. Some further development may therefore be needed to fully align this scheme with the updated Peatland Code emissions reporting approach outlined above. In particular, there is a need to define a set of attributes that would enable site managers to reclassify areas from Re-wetted Bog or Fen (with high RCH_4 values and thus higher predicted CH_4 emissions for any given WTD) to Near-Natural Bog or Fen, with lower RCH_4 values. This would likely use similar criteria to those set out in Figure 3.3, such as percentage *Sphagnum* cover, but requires some further development.

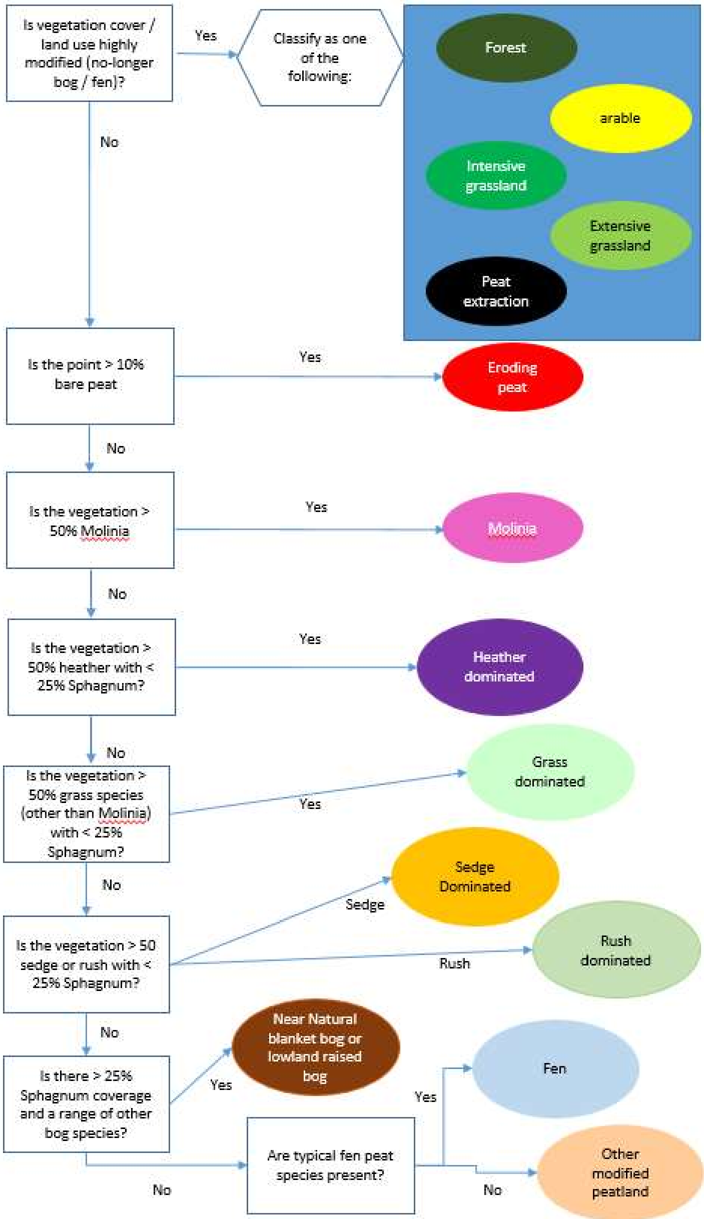

***Figure 3.3**. Flow chart describing attribution of peatland areas to broad peat condition categories based on broad vegetation attributes measured in 10 m^2 quadrats (from Williamson et al., 2021).*

3.3.2. Water table monitoring

As detailed in Section 3.2, water table depth has a strong influence on both CO_2 and CH_4 fluxes from peatlands and the empirical relationships defined in Evans et al. (2021) enable water table depth to be used as an indicative modifying factor for Tier 2 emissions estimates using the carbon calculator spreadsheet. In order to apply this approach, as a minimum it is necessary to obtain an estimate of the annual mean water table depth across the project area. The current peatland code methodology recommends taking measurements at the intersection of a 100 * 100 m grid across the site. Low cost water table monitoring could be incorporated into this by following the Eyes on the Bog rust rod methodology (Lindsay et al., 2019), whereby mild steel rods 50 – 100 cm in length (depending on anticipated water table depth range) are inserted into the peat at the monitoring point. When these are revisited the demarcation line between rusted and non-rusted on the bar will give an indication of aeration depth, and hence water table depth, at the location. Water table range, which is an important additional metric of peat condition (albeit one which cannot yet be empirically related to the C/GHG balance) can also be inferred from the overall height of the rust band. A network of these points can be revisited annually, or as funds allow, to determine whether the mean water table depth is changing over time.

If knowledge of the water table is integral to monitoring the required post-restoration condition of the site, or if more funding is being invested in MRV measures, manual or automated dipwells can be installed at all or a subset of the 100 m grid intersections. Annual means can be estimated from manual dipwell measurements provided that visits are spaced equally through the year and include the drier and wetter periods. Automated dipwell loggers provide the highest temporal frequency monitoring, and will provide a closer value to the "true" mean water table level and variation at a site but this comes with an increased financial cost to a project as commercial water table loggers are currently expensive, and staff time is needed to manage and interpret the data being collected. However, an option for lower-cost automated water table monitoring is outlined in the following section.

***Table 3.4**. Overview of currently available options for water table monitoring*

Monitoring method	Costs		Accuracy
	Equipment	***Staffing***	
Rust rods	Low	Low	Low - medium
Manual dip wells	Low	Moderate	Medium (depending on frequency)
Automated dip wells	High	Moderate to high	High

3.3.3. Peat depth and elevation change monitoring

The current peatland code monitoring methodology includes monitoring of peat depth using peat rods or other probes such as avalanche poles, at the same 100 * 100 m grid intersection as the condition categories. As a minimum this point monitoring should continue, as it would provide an estimate of carbon stocks at the site. However, peat depth can show high variability over small areas (e.g. where peat has formed over bedrock or glacial debris), and long-term changes in peat depth are typically very small compared to the overall size of the peat carbon pool, even where net carbon gains and losses are large. As a result, detecting the 'signal' of small changes from the 'noise' of high spatial variability in a very large pool is extremely difficult, and although it may be possible to detect changes over longer (~10 year) periods if repeat measurements are made under consistent hydrological conditions to avoid the confounding influence of peat shrinkage and swelling (Morton and Heinemeyer, 2019).

As a more practicable alternative, we recommend monitoring of change in the surface elevation of the peat, which we consider to be a more responsive and reliable measure of change in peat condition. Changes in peat surface elevation can occur both as a consequence of changes in peat carbon stock (for example by measuring long-term peat subsidence, as at the well-known Holme Post in the Fens, or by recording peat growth relative to a fixed datum) and also changes in the physical properties of the peat, such as bulk density (e.g. Morton and Heinemeyer, 2019). On shorter timescales, peat surfaces respond directly to changes in water levels due to drainage and re-wetting, as well as seasonal dry-wet cycles ('bog breathing') and even individual rain events. This short-term variability, while not directly associated with carbon cycling, is strongly related to the hydrological behaviour of the peat and may therefore provide useful proxy information on the condition of peat habitats (e.g. Howie and Hebda, 2019; Evans et al., 2021b).

To capture longer-term changes in peat elevation, the Eyes on the Bog methodology describes the use of peat surface rods. These painted steel threaded rods are inserted through the peat into the substrate below, and the present peat surface is marked using a washer held in place with nuts. This allows point monitoring of peat surface movement over time, and therefore the amount of long-term peat growth or subsidence that has occurred since restoration. Where the anticipated climate mitigation benefits of restoration are partly associated with avoided emissions, as will almost always be the case, we recommend (where conditions permit) the deployment of peat surface rods in unrestored control areas, in order to confirm continued rates of carbon loss from these areas as a counterfactual to the restoration. We recognise however that retaining unrestored control areas may be impossible in some projects, for example where the areas involved are small and/or hydrologically interconnected. In these cases, a longer period of pre-restoration baseline measurement would be desirable in order to demonstrate change.

Where possible – and potentially as part of an integrated programme of intensive and extensive measurements in combination with the Eyes on the Bog methods – higher-resolution measurements of peat elevation change can be made using a number of automated methods, such as time lapse cameras. This method was initially developed for work in Indonesia (Evans et al., 2021b) but is now being trialled at a range of sites around the UK, most recently as part of a pilot study for Natural England linked to the England Peat Map and the Natural Capital Ecosystem Assessment. The method comprises a time-lapse camera mounted on a platform anchored to the peat surface, which moves vertically with the peat. The camera takes daily (or sub-daily) photographs of a clearly demarcated rod fixed to the base of the peat (see Figure 3.4), which can be processed using image-processing software to record movements of the peat surface with sub-mm accuracy. An example of the data that can be obtained from these cameras, based on work undertaken in support of the Welsh Raised Bogs LIFE project, is shown in Figure 3.5. At this site, a raised bog in west Wales, contour bunds were installed on dates shown by vertical dashed lines on the graph alongside the camera situated at the edge of the raised bog. Prior to bunding the cameras at the edge and centre of the bog had recorded similar peat surface movement in response to rainfall but the installation of the bunds resulted in a sustained peat surface uplift of approximately 2 cm at the edge of the bog compared to the centre (J. Williamson, unpublished data). This rapid uplift is attributable to increased retention of water on the site rather than accumulation of new peat, and hence does not represent newly sequestered carbon, but shows an immediate impact of restoration on the bog which would otherwise be difficult or impossible to detect, and which can be expected to translate into long-term benefits to the carbon balance.

The camera system is currently being further developed by UKCEH to comprise a more robust, purpose-built system with additional sensors (including a low-cost water level sensor and soil moisture sensor), inbuilt data logger, data telemetry and the use of ARUCO markers to streamline image processing and improve data quality. By deploying cameras alongside flux towers for CO_2 and CH_4 measurement, the longer-term aim is to develop quantitative metrics of peatland function that relate directly to GHG emissions, and thus provide a lower cost (£100s) proxy method for estimating the GHG balance of peatland ecosystems that would otherwise require the use of flux towers (£10,000s). If successful, this approach could be combined with the carbon calculator method describe in the preceding section, and the rapid vegetation assessment and Eyes on the Bog methods described in this section, to provide an integrated and affordable basis for MRV of Peatland Code restoration projects.

***Figure 3.4**. Example of a prototype peat elevation camera in operation as part of the Welsh Raised Bogs LIFE project.*

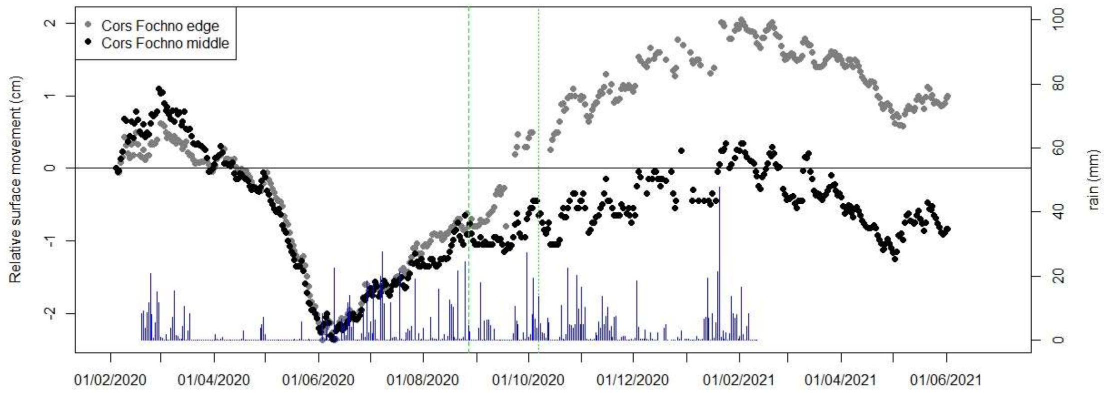

Figure 3.5*. Peat surface movement measurements taken between February 2020 and June 2021 at the edge and centre of the peat dome at Cors Fochno. Timing of bund interventions are shown by dashed green vertical lines, while daily rainfall totals are shown by blue bars (data only available until February 2021).*

3.3.4. Towards an integrated Peatland Code MRV methodology.

A comparison of the differences between the current peatland code monitoring methodology and the additional monitoring outlined above is shown in Figures 3.6 and 3.7. Although the monitoring effort is greater for the second method, the additional information that can be obtained will allow site reporting to move from merely reporting interventions carried out to being able to quantify the changes occurring as a result of the intervention activities.

One point to note is that additional monitoring has to take account of the scale of the sites being monitored. In this case the initial peat depth monitoring requires 157 points to be assessed. Clearly installing and checking 157 rust rods and surface level rods and recording 157 vegetation quadrats would be a large undertaking so in this example the additional monitoring would be carried out over a 200 m grid, resulting in 39 monitoring points. Smaller sites with more variable topography may still choose to carry out all monitoring over the original 100 m grid.

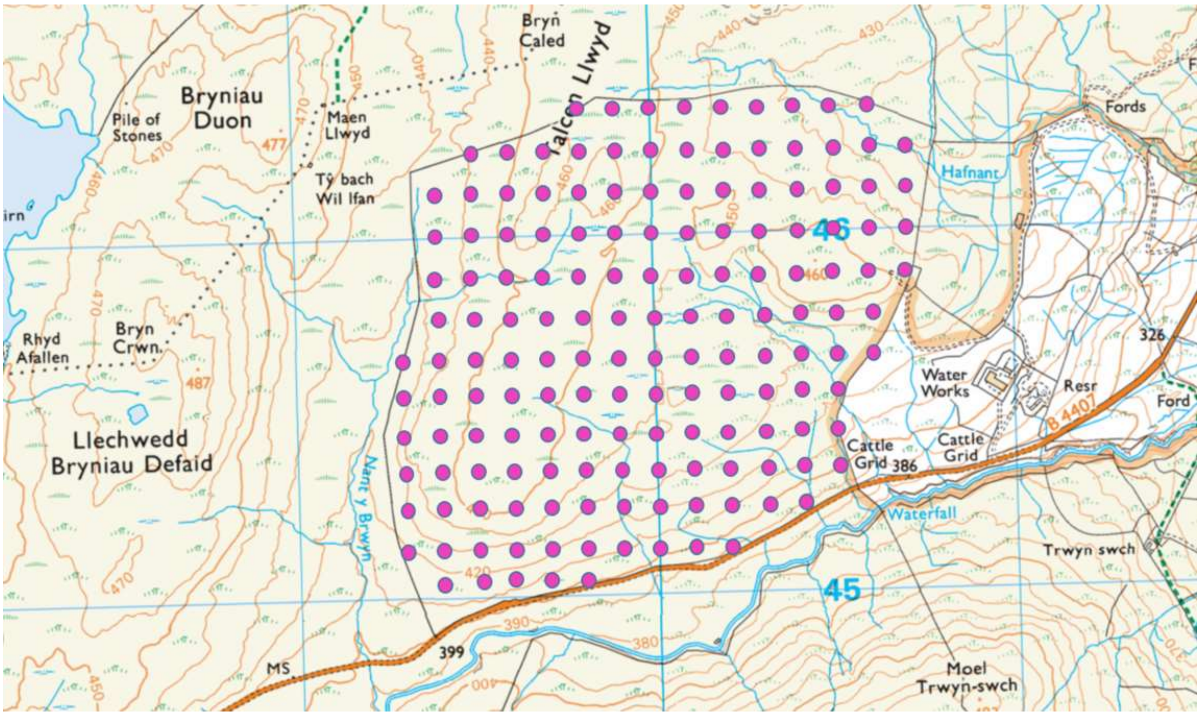

Figure 3.6. *Example of initial 100 m grid based monitoring as currently required by the Peatland Code for a potential restoration site in North Wales within the Migneint Arenig Dduallt SAC. The pink dots show where peat depth and estimation of peat condition should be made.*

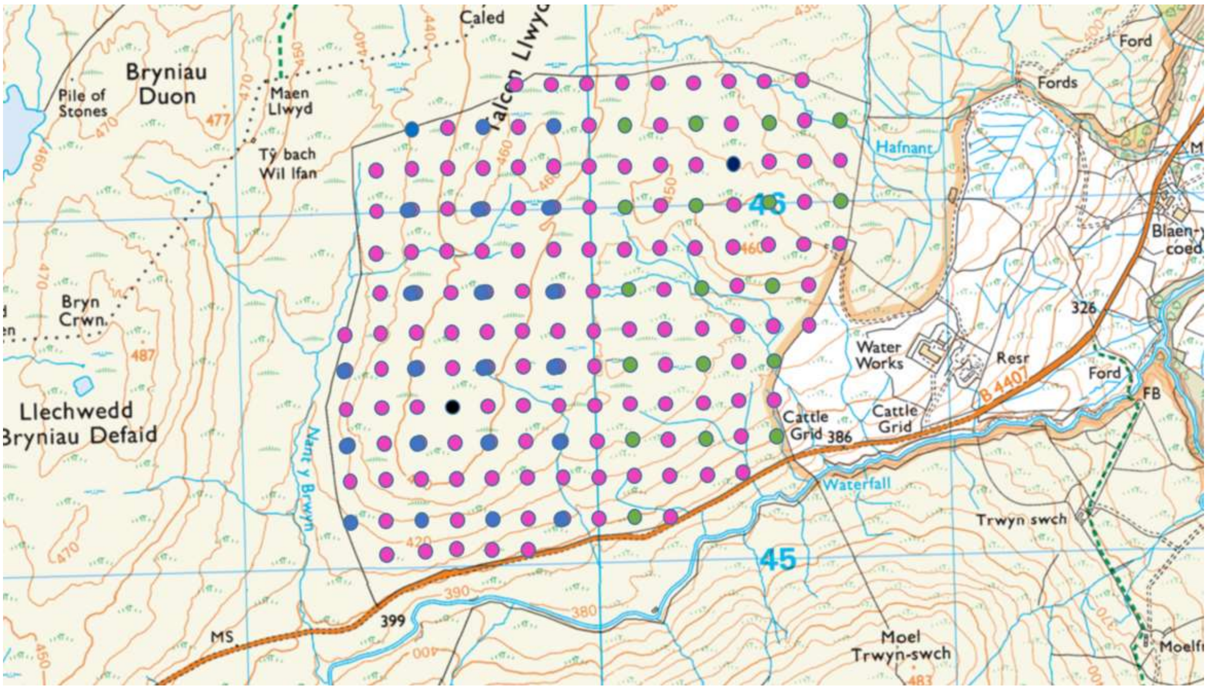

Figure 3.7. *Example of future expanded monitoring for the same potential restoration site in North Wales within the Migneint Arenig Dduallt SAC. In this case the site has the same base 100 m grid of points where initial peat depth monitoring takes place, but the site has been separated vertically with one side to be restored at a time, allowing for measurements of change between control and restored areas. The blue points are the control sites where additional surface level rods and rust rods are installed, while the green points are the restoration sites where additional surface level rods and rust rods are installed, and vegetation cover is recorded. Potential locations for peat motion cameras – one within the control site and one within the restoration site are shown as black points.*

In addition to ground based monitoring, the increased availability of remote sensing data will allow recording of site change from drones, aircraft and from space. Figure 3.8 shows an example of peat condition assessment for the Migneint Arenig Dduallt SAC using Sentinel 1 radar backscatter data. Use of these technologies for peat assessment is still at an early stage, but it is hoped that further development will allow detection of change in condition over time, and allow the more detailed site level modelling of GHG emissions based on WTD_e to be modelled on a national scale.

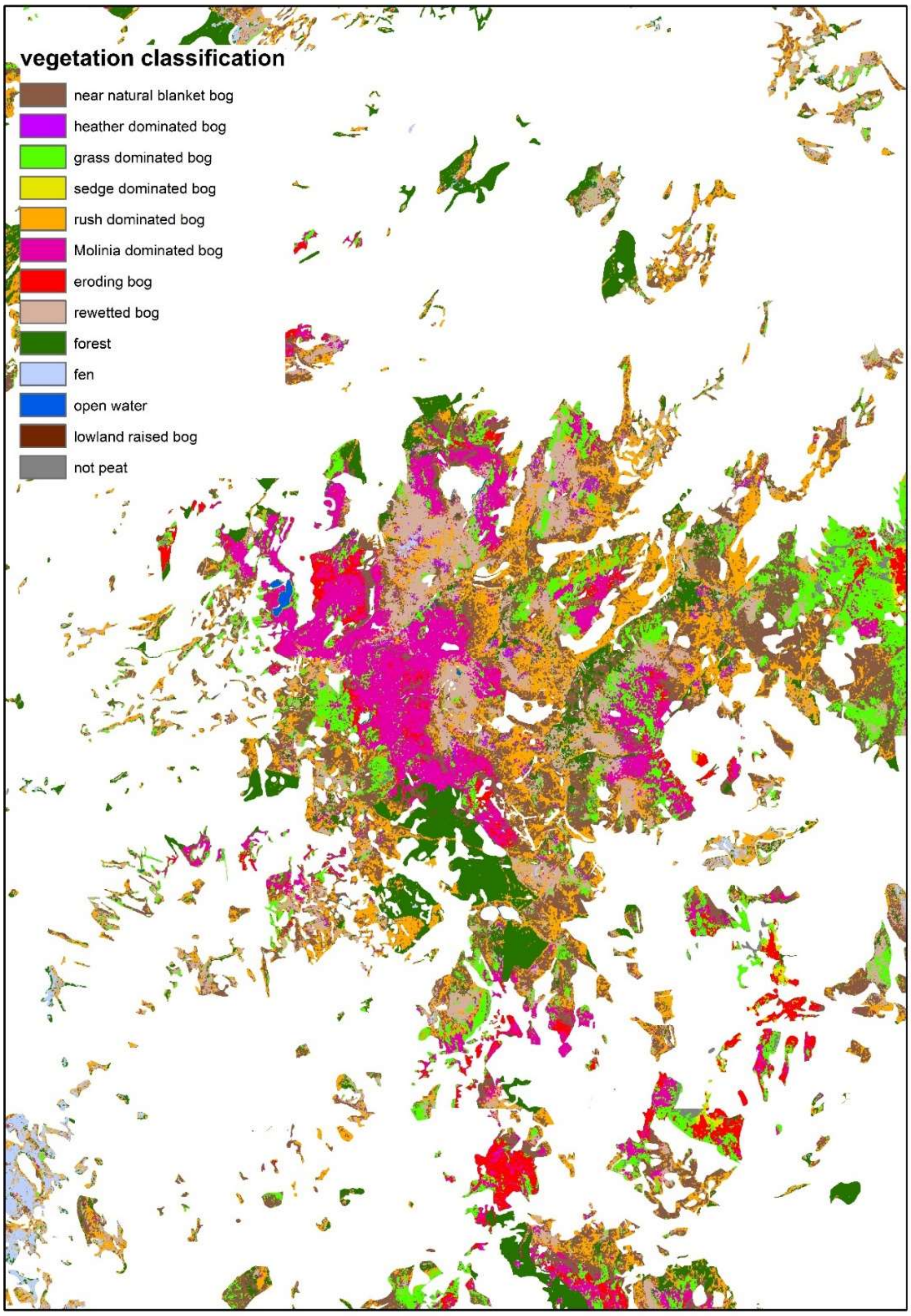

***Figure 3.8**. Peat condition classification from Sentinel 1 radar backscatter data for the Migneint Arenig Dduallt SAC (Williamson et al., 2021).*

3.4 Future methodological development

3.4.1. Fire emissions

At this stage, we have not developed a method to explicitly account for the effects of fire (either prescribed burns or wildfire) on peatland GHG emissions. The IPCC Wetland Supplement (IPCC, 2014) provides a Tier 1 method for estimating CO_2 and CH_4 emissions from fires, based on availability of dry matter (above ground biomass and dry peat) in drained and undrained landscapes, but does not provide default emission factors for prescribed fires. In the UK, the impact of managed burning (specifically for grouse moor management) has received a high degree of attention, but impacts on the peat carbon balance are contested (e.g. Davies et al., 2016; Harper et al., 2018; Heinemeyer et al., 2018; Marrs et al., 2018; Evans et al., 2019; Baird et al., 2019; Ashby and Heinemeyer, 2019). Based on detailed peat core records assessing peat physical and chemical in relation to fire events, Heinemeyer et al. (2018) argued that inputs of decay-resistant charcoal offset the loss of (more decomposable) litter from combusted biomass in the long-term, while other studies have argued that loss of *Sphagnum* and management of grouse moors to maximise heather cover negatively affect peat formation and may lead to peat drying (e.g. Baird et al., 2019). However, so far no study has measured fluxes over an entire management cycle or in relation to alternative management (cutting/no management). The previous Defra-funded study (Heinemeyer et al., 2019) was specifically set up to achieve this and other related aims as highlighted in Harper et al. (2018). The study of Marrs et al. (2019) showed a reduction in recent carbon accumulation rate at a long-term experimental burn experiment at Moor House from the equivalent of -1.72 t CO_2 ha^{-1} yr^{-1} at a site last burned 90 years previously to -1.32 t CO_2 ha^{-1} yr^{-1} at a site burned 6 times during that period, although this study did not incorporate potential charcoal impacts or bulk density changes. Their C accumulation rates are intermediate between the Tier 2 EFs for near-natural and modified bog (Moor House is considered to be modified, but recovering from past burning impacts) but not markedly different. Interactions with wildfire are also complex, with Marrs et al. (2019) arguing that abandonment of heather-dominated blanket bogs would increase above ground woody biomass (i.e. 'fuel load') and therefore the risk and severity of wildfire, whereas Baird et al. (2019) argued that re-establishment of *Sphagnum* cover and resulting suppression of heather growth should lead to a naturally fire-resistant ecosystem. Nevertheless the extent of carbon loss from peatlands following wildfire (as seen recently on Saddleworth Moor, Marsden Moor, Holme Moss etc., and also in Sweden; Granath et al., 2021) may be extremely large, so post-restoration management should be designed to minimise fire risk and to take account of potentially increasing frequency and severity of climate extremes.

In summary, while it would clearly be desirable to include direct impacts of peatland management and restoration in the Peatland Code, the empirical basis for doing so is not yet sufficiently conclusive to support this. On the other hand, the indirect effects of burn-management (insofar as they affect measurable site properties such as vegetation, bare peat extent and water table depth) may be captured within the proposed methodology described above. Wildfires remain difficult to incorporate directly in any carbon accounting methodology, but the associated risks (including risks to permanence of any carbon credits sold) need to be considered in the design of the Peatland Code (see Section 4).

3.4.2. Site-specific estimation of N_2O emissions

A number of previous analyses (e.g. Couwenberg et al., 2011; Leppelt et al., 2014; Liu et al., 2020; Lin et al., 2022) have shown that, in general, deeper-drained organic soils tend to have higher N_2O emissions. This effect is also evident in the emission factor database (Figure 3.9) which shows that very few sites have significant N_2O emissions until WTD is more than 20 cm below the peat surface, but very high variability below this depth. This is in line with the previous studies that have shown additional influences of intrinsic site properties such as fertility and pH, and management factors such as rates of fertiliser and manure application. However, these effects are hard to disentangle, because many of the properties that lead to high N_2O emissions are interrelated, for example drained and cultivated sites tend to also be fertilised. With additional data collection (e.g. during the ongoing BEIS wasted peat project) it may be possible to derive reliable predictive relationships in future based on measureable site properties in future. What is clear from virtually all studies, however, and from the data shown in Figure 3.9, is that N_2O emissions can be expected to fall to low levels in re-wetted peatlands, provided that water levels are raised to within 20 cm of the peat surface, under almost all circumstances. The utility of a more sophisticated approach to estimate N_2O emissions from re-wetting projects will therefore largely be in determining the pre-restoration emission rate, and therefore the level of avoided emission from re-wetting.

For fertile croplands or grassland systems where water levels have been raised but remain more than 20 cm below the surface, there is some risk of elevated N_2O emissions if this leads to intermediate soil moisture and aeration levels (which tend to cause N_2O production) within the rooting zone. In these situations, and until more measured data become available, we recommend applying a 'factor of safety' to the calculation of overall mitigation benefits of raised water levels, to allow for the risk of higher N_2O emissions partly offsetting CO_2 emission reductions. In general, any interventions that involve major disturbance of N-rich topsoil should be avoided, as these could lead to large pulses of N mineralisation, nitrification and subsequent N_2O production.

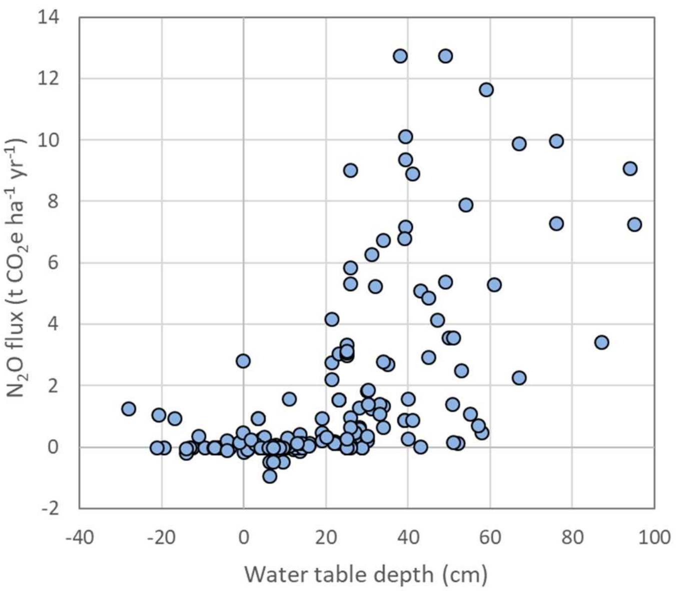

Figure 3.9. *Measured N_2O fluxes versus mean water table depth for sites included in the emission factor database (N_2O is expressed in t CO_2e ha^{-1} yr^{-1} based on the AR5 100-year GWP of 265, and positive values of mean annual water table depth indicate that the water table is below the peat surface. All data taken from the updated JHI Peatland Emission Factor Database.*

3.4.3. Site-specific estimation of waterborne carbon fluxes

The current Peatland Code methodology developed by Smyth et al. (2015) includes a simple Tier 2 type methodology for estimating pre- and post-restoration emissions associated with DOC and POC loss from different blanket bog categories. DOC emission factors follow Tier 1 default values for drained and rewetted temperate peatlands developed for the IPCC Wetland Supplement. These defaults were based partly on a small number of UK studies, and were also adopted for UK national inventory reporting. At this stage, we have not attempted to update these figures, as few additional UK studies have been published, and recent international studies (e.g. from Finland) fall outside the UK-relevant climatic region. However there is some recent UK evidence that DOC increases may be larger from afforested bogs, but smaller from drained blanket bogs (Williamson et al., 2020; Pickard et al. in prep.). It is also likely that DOC (and hence indirect CO_2) losses are lower from fen peatlands (both drained and undrained) due to lower concentrations and smaller water fluxes (Evans et al., 2016). These variations in DOC by site and management type have not yet been incorporated into the UK emissions inventory, however, and it is doubtful whether sufficient DOC flux data exist across the range of UK peat types and condition classes to support a full Tier 2 approach. The fate of DOC (i.e. the proportion ultimately converted to CO_2) also remains uncertain, although studies from other parts of the world do support the IPCC methodology in showing high mineralisation of peat-derived DOC to CO_2 in coastal seas, most of which is subsequently degassed to the atmosphere (Zhou et al., 2021).

For POC, the Peatland Code defines POC exports as zero for all categories except eroding peat, for which a value of 19.3 t CO_2 ha^{-1} yr^{-1} was assigned. In Appendix 2a.1 of the IPCC Wetland Supplement, a default POC export flux of 4 t C ha^{-1} yr^{-1} was proposed for bare peat surfaces based on studies on UK blanket bogs (Goulsbra et al., 2013) and the proportion of this POC subsequently mineralised to CO_2 was subsequently estimated at 70%, but with a high uncertainty (Evans et al., 2016), giving an emission factor of 2.8 t C ha^{-1} yr^{-1} (10.7 t CO_2e ha^{-1} yr^{-1}) for eroding peat surfaces. As for DOC, there are few additional POC flux estimates that would enable refinement of the IPCC default method, and we have not proposed further changes for this report. However, adoption of the current inventory figures for the Peatland Code would reduce the current default emission factor for POC loss from eroding bog by 45%. On the other hand, as a result of updates to other emission factors since the Peatland Code was implemented, the change in overall estimated GHG emissions from eroding peatlands is smaller (21% and 25% for drained and undrained actively eroding bog respectively). While this will still have some implications for estimated net GHG emissions reductions resulting from restoration of eroding bog, the net benefits will nevertheless still be large in all cases. It may also be possible to develop a more sophisticated approach to quantifying POC losses pre- and post-restoration based on monitoring and assessment of bare peat extent (e.g. using remote sensing approaches) as described in Section 3.3.

Finally, emissions of CH_4 from drainage ditches were not previously included in the Peatland Code, but have been fully included in the UK's emissions inventory based on the Tier 1 methodology described in the IPCC Wetland Supplement (see also Evans et al., 2016). Ditches can act as emissions hotspots within drained landscapes that do not otherwise release CH_4, and reductions in ditch CH_4 emissions may partly offset increase in emissions from the peat surface following rewetting. This is likely to vary by peat type and restoration method, however, with many lowland restoration projects retaining active ditch networks for water management purposes. In upland blanket bog, infilled ditches and shallow ponds behind peat dams may continue to act as CH_4 emissions hotspots (Peacock et al., 2013; Cooper et al., 2014; Chapman et al., 2022) although these emissions are expected to decline with vegetation succession from *Eriophorum* (and early colonist of many restored areas) to *Sphagnum*. Recent evidence from ditches and other constructed waterbodies such as ponds (Peacock et al., 2021) also suggests that CH_4 emissions are strongly affected by inputs of nutrients and labile organic matter from fertilisers and animal wastes, so restoration measures that reduce these inputs (for example in valley fens surrounded by farmland) may generate substantial reductions in CH_4 emissions from waterbodies.

3.4.4. Capturing transitional changes in GHG fluxes post-restoration

Previously, concern has been expressed about the potential for a 'methane spike' following restoration and re-wetting. This could occur if for example a large pool of labile, nutrient-rich organic matter (such as a farmed peat soil) is flooded, and may be a contributing factor in the high rates of measured CH_4 emission at some re-wetted fen sites included in the emission factor database. In upland blanket bogs, a similar peak of emissions could occur if re-wetted sites are initially colonised by aerenchymatous species such as *Eriophorum*, before reducing following natural succession to *Sphagnum,* as described above for infilled ditches. Active *Sphagnum* planting could accelerate this transition. However, a previous analysis of the database did not reveal a consistent pattern of elevated CH_4 at recently restored sites, or any clear relationship between emissions and time since rewetting (Artz et al., 2016), so a simple time-dependent function to describe a 'methane spike' is not considered to be appropriate at present. On the other hand, the proposed refinements to GHG emissions reporting set out above have the potential to capture some aspects of these transitional changes, for example due to gradual changes in water table depth or shifts in vegetation community sufficient to support a change in emissions reporting category. If additional data can be obtained over restoration transitions (e.g. flux towers to monitor CO_2 and CH_4 fluxes over restoration transitions, ideally based on a paired before-after control-intervention design) it may be possible to develop a more sophisticated empirical method to capture transitional impacts of re-wetting different peatland types on CH_4 emissions.

For CO_2, the methodology currently used for both the Peatland Code and the national inventory assumes that, at best, restoration will lead to rates of CO_2 uptake equivalent to those of a near-natural peatland. However the long-term carbon accumulation rate of a peatland is fairly low, and on the relatively short time-horizons implicit in the use of 100-year GWPs the CO_2 sequestration which does occur is largely counterbalanced by CH_4 emissions. As a result, the climate change mitigation of peatland restoration is currently considered to be mainly limited to abatement of current emissions. These emissions are large, and abating them is vital if the UK is to meet its emissions targets. However, with the growing emphasis (by countries, local authorities, NGOs and businesses) on achieving net zero emissions, investors in land-management for climate change mitigation are increasingly seeking options that can deliver net GHG removal (GGR). In this context, the Peatland Code would potentially secure greater private investment in peat restoration if it includes options that generate GGR. UKCEH are currently leading a large UKRI-funded project which will explore the potential to achieve GGR through a range of measures ranging from 'enhanced restoration' and paludiculture through to active 'carbon farming' of lowland peat that is currently under drainage-based agriculture, including measures such as production and application of biochar from wetland biomass, and the use of amendments to suppress CH_4 and N_2O emissions. While these techniques remain unproven, and are currently outside the scope of the Peatland Code, independent modelling studies by Heinemeyer et al. (2019) for the Defra Peatland-ES-UK (Defra BD5104) project, and Simon et al. (2021) for the BEIS review of UK GGR potential both suggested that degraded peatlands have the potential to accumulate carbon rapidly, and therefore that the CO_2 sequestration potential of peat restoration may have been significantly underestimated.

In summary, the models suggest that rates of CO_2 capture during the restoration transition may (perhaps greatly) exceed the long-term CO_2 uptake rate of a natural peatland. This is because there is often large potential to rebuild those elements of the peatland ecosystem that have been lost to degradation, including the above ground biomass, litter layer and surface peat layer (acrotelm). Heinemeyer et al. (2019) even suggest that CO_2 sequestration potential may be greatest in areas where peatlands have been completely lost as a result of historic land-use, although clearly restoration measures would have to be correctly targeted at areas where peat was previously present (i.e. areas in which peat could potentially re-form) and not towards areas where local conditions never supported peat formation. This would require either historic information, modelling of past conditions (e.g. Heinemeyer et al., 2019) or careful assessment of local topographic, hydrological and climatic factors.

In effect, rebuilding carbon stocks in degraded peatlands is analogous to planting a forest. Whilst a mature old-growth woodland may (like a near-natural peatland) have limited long-term CO_2 sequestration potential, the re-establishment of trees on a grassland will generate large transitional CO_2 capture. While less visually evident than tree-planting, peat restoration could generate similar CO_2 capture over similar timescales; for example a 20 cm acrotelm in a *Sphagnum* bog alone may contain as much carbon (around 60 t C ha^{-1}) as the standing crop of a mature Sitka spruce plantation on peat (Lindsay, 2010). Currently, carbon sequestration offered by the Woodland Carbon Code is entirely based on transitional carbon gains, whereas in the Peatland Code they are completely omitted. This creates an inconsistency between the Codes, and – by underestimating the carbon benefits – may lead to suboptimal levels of investment in peatland restoration.

In the analysis described in Simon et al. (2021), a simplified version of the DigiBog peatland carbon accumulation model (Young et al., 2017) was used to estimate rates of CO_2 accumulation in restored peatlands. The analysis, while theoretical, suggests that transitional CO_2 uptake rates as high as 10 t CO_2 ha^{-1} yr^{-1} could be possible, with high rates (> 5 t CO_2 ha^{-1} yr^{-1}) likely to persist for at least 50 years (i.e. beyond the 2050 target of the Paris Agreement and UK Net Zero strategy) (Figure 3.10). These rates compare to a long-term mean CO_2 accumulation rate for Northern peatlands of just -0.8 t CO_2 ha^{-1} yr^{-1} (Loisel et al., 2014) and an uptake rate based on our updated EF for near-natural bogs of 2.9 t CO_2 ha^{-1} yr^{-1}. Such high rates do not appear unrealistic, however; several studies in Finland and Canada have recorded average CO_2 uptake rates of 4-5 t CO_2 ha^{-1} yr^{-1} in the decades following peat restoration (Turunen et al., 2004; Kareksela et al., 2015; Beaulne et al., 2021), while one study in China reported a rate of 11.8 t CO_2 ha^{-1} yr^{-1} after 80 years (Li et al., 2018). The contemporary CO_2 uptake rate of 9.4 t CO_2 ha^{-1} yr^{-1} recorded at the Moor House (Evans et al., 2021) is also consistent with this prediction, and with observed ecological recovery at the site, including *Sphagnum* recolonisation (R. Lindsay, pers. comm.), although this measured uptake rate was based on only one full year of flux tower data.

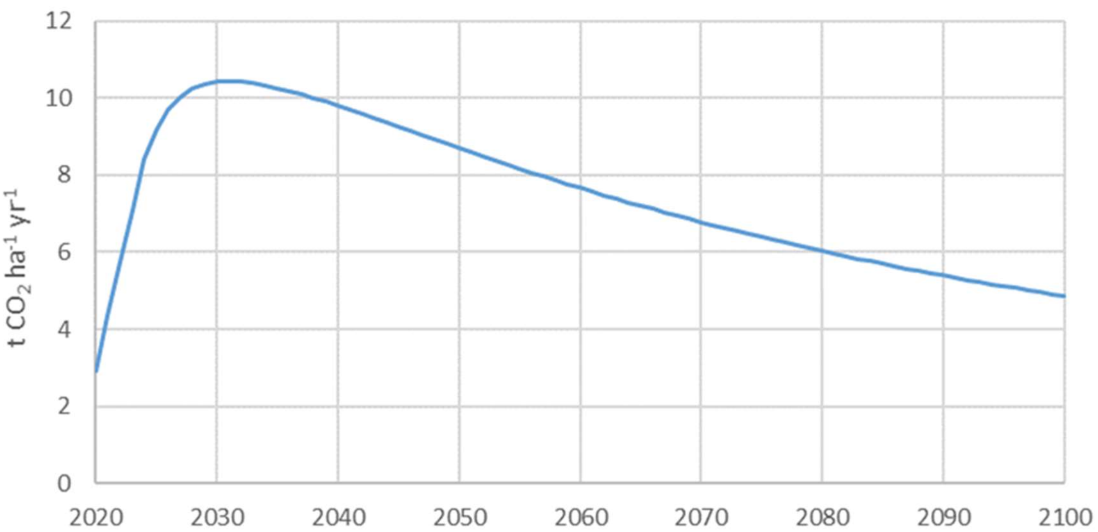

***Figure 3.10**. Modelled annual rates of CO_2 following restoration of a degraded peatland, simulated using a simplified version of DigiBog, based on the analysis described in Simon et al. (2021).*

Clearly, this simple model of transitional CO_2 uptake following peat restoration model needs to be refined and validated before it can be included in the Peatland Code, or used to support investment in peat restoration. However we consider that the theoretical basis for doing so is strong, and that the inclusion of transitional CO_2 uptake could significantly strengthen the overall case for peat restoration. According to the analysis of Simon et al. (2021), the projected rates of CO_2 uptake in this model substantially exceed expected CH_4 emissions from re-wetted peatlands based on 100 year GWPs, in which case peat restoration would be able to offer net GGR as well as avoided emissions. Perhaps counterintuitively, the greatest transitional CO_2 gains (and hence GGR) may occur in sites that have historically undergone the greatest degradation, because these have the greatest potential to capture CO_2 into previously depleted carbon pools (i.e. above ground biomass, litter layer, acrotelm. On the other hand, the most degraded sites may be the hardest to restore, and theoretical high rates of transitional CO_2 uptake will only be attained if a viable peat-forming ecosystem can be recreated. The inclusion of this approach in a future Peatland Code would therefore require robust MRV to ensure that the theoretical CO_2 benefits of restoration are delivered. On the other hand, it may help to support more effective (and typically expensive) restoration interventions such as *Sphagnum* planting if these can be shown to generate more rapid and/or substantial carbon benefits.

4. Comparison with other international peatland accounting methods

4.1 UK carbon markets in operation and under development

The UK participates in compliance markets such as the Emissions Trading Scheme and can engage with international voluntary carbon markets to meet its obligations under the Paris Agreement (the rule book for this was agreed under Article 6 at COP26). The majority of voluntary carbon market transactions take place via the Woodland Carbon Code, with the Peatland Code now rapidly scaling its operation. A number of initiatives are now underway to develop new carbon codes to expand the domestic voluntary carbon market in the UK, including:

- Development of a UK Farm Soil Carbon Code (UKFSCC) is being led by a consortium managed by the Sustainable Soils Alliance with FWAG South West, SRUC, University of Leeds and others funded by the Environment Agency's Natural Environment Investment Readiness Fund (NEIRF). Pilots are already running and they will be consulting on a template code in late 2022. In parallel with this, others are piloting Verra's Methodology for Improved Agricultural Land Management in the UK (VM0042);
- The Wilder Carbon standard has been developed by Kent Wildlife Trust and is due to be launched and piloted in March 2022. It will enable the generation of carbon credits from rewilding activities including woodland creation, peatland restoration and other forms of restoration, using metrics developed by the Woodland Carbon Code and Peatland Code, and requiring the collection of biodiversity data using Defra's biodiversity offsetting metric. In contrast to other UK domestic carbon markets, it requires buyer checks to ensure those investing in projects have done everything possible to reduce emissions at source before offsetting their residual emissions. It also has unusually long minimum contract lengths of 100 years, or 50 years with conservation covenants that would ensure projects are effectively permanent;
- The development of a Saltmarsh Code is being led by the UK Centre for Ecology & Hydrology with RSPB, Jacobs, SRUC and others funded by NEIRF. The goal is to enable the generation of carbon credits from saltmarsh restoration. The group will either adapt a Verra code for application in the UK in the first half of 2022 or pilot a UK Code in late 2022;
- The development of a Hedgerow Code is being led by the Game and Wildlife Conservation Trust's Allerton Project. While its initial development will focus on carbon in above ground biomass and soils, projects will monitor biodiversity benefits for potential use in Biodiversity Net Gain and similar programmes. There is no date set yet for the launch of the code;

- Adur District & Worthing Borough Councils were also awarded NEIRF funding to explore carbon market opportunities for sea kelp restoration, which has the potential to lead to the development of a new domestic market;
- The Woodland Carbon Code are in discussion with a team applying for NEIRF funding to explore the potential to extend the scope of the Code to include agroforestry, or develop a stand-alone Agroforestry Code;
- The Scottish Wildlife Trust's Riverwoods project is exploring the potential to finance the creation of riparian woodland via carbon markets. Given that the riparian zone itself is too narrow to qualify under the Woodland Carbon Code, it is likely that this would need to be done as part of a wider floodplain afforestation programme, which may be designed to attenuate flooding or help improve water quality, to be eligible under the Code. There are existing examples of riparian woodland planted as part of wider adjacent planting schemes already financed under the Woodland Carbon Code;
- Although there are also no UK domestic carbon markets that focus on species rich grasslands, there are a number of international voluntary carbon markets that have developed methodologies that could in theory be adapted for use in the UK (e.g. BCarbon and Verra's methodologies for Sustainable Grassland Management (VM0026) and Improved Agricultural Land Management in the UK (VM0042)). The UK Farm Soil Carbon Code could also be extended in future beyond its current arable focus to include conversion to species rich grasslands, given the evidence of soil carbon benefits from both conversion and nutrient management on existing grasslands.
- Although not UK specific, it is also worth noting the Verified Carbon Standard (VCS – also known as Verra) is a US based carbon accounting standard and organisation which has published methodologies designed to be applicable world-wide (not just US, although some are), covering a wide variety of technologies and measures from renewable energy projects, to land-use projects. VCS have two methodologies relevant to peatlands:
    - VM0036 (v1) for estimating change in GHG emissions associated with rewetting of drained temperate peatlands. The scope of this is limited to rewetting of peatlands that have been drained for forestry, peat extraction, or agriculture. Post-rewetting land use is limited to forestry, agriculture, nature conservation/recreation, or activities limited to those aiming at GHG emission reductions, or a combination of these activities.
    - VM0007 (v1.6) REDD+ framework quantifies GHG emissions from avoiding unplanned and planned deforestation and forest degradation. It includes specific modules for activities on forested peatlands.

4.1.1 The proposed Farm Soil Carbon Code (UKFSCC)

The proposed Farm Soil Carbon Code has particular relevance to the Peatland Code, as the only other voluntary domestic carbon market in the UK to focus on soil carbon. There is considerable debate over the evidence for soil carbon benefits from regenerative farming practices, and concerns over the capacity of soils to continue sequestering carbon in the long-term (most soils reach equilibrium levels of carbon after which they cannot sequester more) and the permanence of carbon gains (given the potential for reversal in response to future changes in land management). There is lack of evidence for the soil carbon gains of many popular regenerative farming practices, for example leys in crop rotation, and there are social or economic barriers to the adoption of some practices that are known to increase soil carbon e.g. paludiculture and bioenergy carbon capture and storage (Elliot et al., 2022).

A high level of caution is also needed in potentially transferring regenerative farming practices (even those proven to be effective) from mineral soils to organic soils, where processes and controls on the carbon balance may be different. For example, applying measures such as zero-till or cover cropping to cropland on organic soils, without also raising underlying water levels, could risk generating apparent carbon credits from a farming system that is still acting as a major carbon source.

However, Elliot et al. (2022) present robust evidence from systematic reviews of peer-reviewed literature that a number of practices can reliably increase soil carbon sequestration and storage, for example hedgerow planting, residue incorporation and biochar application. While the window of opportunity for soil carbon storage via techniques such as these is limited, it is estimated that residue incorporation could sequester -1.1-2.8 and biochar could sequester 6-41 Mt CO_2 per year until equilibrium levels of soil carbon are reached. A separate review of UK Greenhouse Gas Removal options for BEIS (Simon et al., 2022) estimated maximum technical potential carbon capture of 15.7 Mt CO_2 yr^{-1} for agricultural soil carbon sequestration in mineral soils, and 20 Mt CO_2 yr^{-1} (based on previously estimated rates of CO_2 uptake potential per hectare, and estimates of the availability of suitable land for both application and – in the case of biochar – production by 2050). While concerns remain over the permanence of these carbon stores, and the biodiversity impacts of biochar, these risks have been addressed in multiple soil carbon codes internationally using mechanisms such as minimum permanence periods enforced via contracts or other legal mechanisms (such as conservation covenants), carbon buffers and insurance products, leading to significant additional carbon storage from agriculture in other countries.

On this basis, the UK Environment Agency funded the development of a UK Farm Soil Carbon Code via its Investment Readiness Fund (NEIRF), to provide buyers and sellers with assurances as to the additionality and permanence of verifiable soil carbon gains and/or emission reductions. The Code is still under development and adapting to stakeholder feedback. The short term focus will be on over-arching standard for soil carbon codes in the UK, to which existing codes and other schemes already generating soil carbon credits could be assessed and benchmarked. This will ensure high standards around additionality, permanence, leakage, measurement, reporting and verification, increasing trust and helping scale high-integrity soil carbon markets. UKFSCC may look to verify its own carbon credits for the UK Land Carbon Registry (or any other registry) if there is a clear market demand, alongside existing codes and registries. For example, to meet UK specific needs at local authority level where cost-effective scaling could be aligned with other ecosystem carbon codes e.g. peatland, woodland etc.

The key interaction between the Peatland Code and the proposed Farm Soil Carbon Code concerns actions to reduce emissions or sequester and store carbon in lowland peats not currently covered by the Peatland Code. Of particular interest is wetland agriculture, or paludiculture, which has the potential to both reduce emissions and sequester carbon in lowland peats, but which is not currently included in the Peatland Code. The treatment of mitigation measures on lowland peat remaining under conventional agriculture currently fall outside the remit of the Peatland Code, but may fall under the remit of the UK Farm Soil Carbon Code. It may already be eligible under Verra's Methodology for Improved Agricultural Land Management in the UK (VM0042), although the costs of implementing this in a UK context are likely to be prohibitive.

4.2 Potential for integration of paludiculture with existing ecosystem markets

4.2.1 Potential for paludiculture to be integrated into the Farm Soil Carbon Code

The Farm Soil Carbon Code will assess and benchmark the existing and new soil carbon codes and schemes as they arise, and as such it would be possible to establish a paludiculture code which could be affiliated to the Farm Soil Carbon Code. Based on a review of international soil carbon standards (Kendal et al., in prep.), the Code is likely to allow significantly shorter permanence periods than the Peatland Code, but is likely to also allow much longer projects (e.g. up to 100 years). As such, the permanence periods in the Farm Soil Carbon Code are likely to be compatible with paludiculture projects, given that the blocking and re-routing of drainage systems to rewet soils is difficult to reverse and so (arguably) could be considered permanent. Additionality and leakage rules are likely to be similar to the Peatland Code, and the compatibility of paludiculture with these rules is discussed below.

The Farm Soil Carbon Code is currently being developed to account for carbon sequestration and storage in soils. It is not yet clear if MRV methods based on avoided emissions would be eligible under the code, although these are included under a number of international soil standards. However, given the early stage of development, it may be possible to incorporate this in the eligibility criteria for the Code. It would be important for avoided emissions to be included for projects to be financially viable, given that the majority of GHG emissions savings are likely to come from this source. If this barrier can be overcome, an advantage of integrating paludiculture with the Farm Soil Carbon Code (compared to the Peatland Code) is that it would be able to account for soil carbon sequestration and storage alongside avoided emissions. While it would be possible to incorporate sequestration in future iteractions of the Peatland Code, this would require a more fundamental revision of the Peatland Code, given that it currently only accounts for avoided emissions, compared to the Farm Soil Carbon Code, which will already be designed to account for sequestration. It is however important to note that there is limited evidence available for the likely prices per tonne of carbon that are likely to be reached under the Farm Soil Carbon Code. SoilCapital are insetting carbon for £23 per tonne in Europe, and prices in Australian and USA soil carbon markets are currently around £8 and £11 respectively, compared to typical prices of between £15-20 (and up to £30) for Pending Issuance Units under the Peatland Code (Kendall et al., in prep.).

4.2.2 Potential for paludiculture to be integrated into the Peatland Code

Since the Peatland Code is owned and operated by the IUCN UK Peatland Programme, it has prioritised extending the Code to cover semi-natural peatland habitats, focussing on restoration to near-natural conditions where possible. As a result, there are currently no plans to extend the operation of the Peatland Code to include agricultural mitigation measures, and no decision has been made on the potential inclusion of paludiculture. However, if there is sufficient evidence, including reliable emissions factors, that could be used to support inclusion of paludiculture in future (as discussed in section 3.2.4.2), the Executive Board has always been open to evidence-based extensions to the Code. Given that the Peatland Code team is currently expanding and emissions factors exist for some semi-natural wetlands in the UK that produce reeds (a form of paludiculture), it might be possible to consider extending the Code to include restoration of these wetlands to near-natural condition. It may then be possible to integrate other paludiculture systems into the Code in future, as reliable emissions factors can be developed, in the same way that other peatland systems are being integrated into version 2.0 of the code based on new evidence.

The integration of paludiculture to the Peatland Code would be compatible with its existing additionality criteria. Given that lowland peats under productive use are not covered by statutory designations in the UK, and the high opportunity costs of paludiculture compared to existing uses, the legal and economic alternative tests would be easily met, as long as projects have at least 15% carbon finance to meet the financial feasibility test. Although the minimum 30 year permanence period in the Peatland Code remains a barrier to the engagement of many private landowners in upland peats (Reed et al., 2022a), this would be less likely to pose a problem for paludiculture in situations involving long-term land use change (although note that some paludiculture activities may be theoretically reversible). It is likely that the majority of projects would opt for significantly longer project lengths (the long-term nature of re-wetting activities may make longer projects more attractive for paludiculture than they are for private owners of upland peats). It is difficult to assess the likelihood of leakage from paludiculture projects; while agricultural production is highly likely to be displaced onto other land, the extent of displacement will depend on current usage, soil type, and whether it affects fertiliser requirements or the need for food imports. Given that the vast majority of nearby lowland peats have already been drained, and the majority of undrained lowland peats are protected, leakage onto other undrained organic soils appears unlikely. Sequestration of carbon in above and below-ground biomass, or carbon capture into peat soils as part of a paludiculture/carbon farming system would not be included under the Peatland Code in its current form. However, research is underway to investigate the potential to quantify this potential (e.g. as part of the UKRI Peat Greenhouse Gas Removal Demonstrator project) and to develop MRV methods for sequestration in peat. If integrated in a future version of the Code, it is possible that soil carbon gains from paludiculture could be included in this way.

4.2.3 Potential for paludiculture to facilitate payments from other ecosystem markets

There is also potential for paludiculture to generate income from existing biochar and future saltmarsh and agroforestry carbon markets:

- There is currently a buoyant biochar carbon market that UK projects could already access, the European Biochar Certificate (https://www.european-biochar.org/en). Prices have been seen to reach 90 euros per tonne of carbon. The most cost-effective source of biochar in the UK is currently from food waste, and although currently around £100 per tonne, these costs are expected to reduce over time. An alternative (and potentially lower cost) source of biochar could be paludiculture itself, if carbon payments from the avoided losses, soil carbon sequestration and biochar could all be monetised via carbon markets to reach a sufficiently high overall value. The BEIS-funded 'Reverse Coal' project is exploring this potential to manage re-wetted peatlands for biomass/biochar production and carbon capture. It is also possible that biochar could be included in the UK Farm Soil Carbon Code as an alternative route to market.
- The majority of lowland peats in the East Anglian Fens are close to or below sea level, and as such, there may be an opportunity for saltmarsh creation in some locations, which could provide access to saltmarsh carbon markets, currently being trialled via an existing Verra code in the UK (see list of carbon markets under development above).
- Where short-rotation coppice is used in paludiculture systems, it may be possible to account for carbon in the above and below-ground biomass via a future agroforestry code (see list of carbon markets under development above).

In addition to this, it may also be possible to derive income for paludiculture from biodiversity markets. Similar to carbon, both voluntary and compliance markets exist for biodiversity. Compliance demand is driven by the Biodiversity Gain obligation in the Environment Act 2021 in England and is estimated to have a potential market of £100 to £300 million annually (The Nature Conservancy, 2021). Voluntary biodiversity offsetting is also growing in popularity, motivated by corporate interests in mitigating impacts or contributing to "nature positive" outcomes from their operations. Intermediaries selling voluntary biodiversity offsets include Environment Bank and Palladium. Since 2006, Environment Bank have delivered 19 voluntary projects via their Habitat Banking scheme (Reed et al., 2022b). While demand for biodiversity offsetting is likely to increase, supply for offsite biodiversity offsets may be constrained by the size, structure and certainty of payments relative to other sources of income, risk perception and contract length (the voluntary market currently tends to operate 30 year contracts). Having said this, demand for voluntary biodiversity offsets has so far been relatively low due to concerns about the robustness of Defra's biodiversity metric and the inability to use conservation burdens outside National Trust land to provide longer term permanence (zu Ermgassen et al., 2022).

Payments for water quality improvements resulting from paludiculture may also be viable. Water quality markets are primarily driven by nutrient neutrality regulation and drinking water investment requirements, with some water utilities paying landowners to change agricultural practice or restore habitats to tackle diffuse pollution at source, driving down water treatment costs and increasing regulatory compliance. For example, Wessex Water established EnTrade which operates a digital auction platform to generate water quality benefits via changes in agricultural practices in its catchments, which has now processed over £2M in transactions. It now provides services to other utility companies. Other schemes are under development to reduce diffuse pollution including the EnTrade Solent Nutrient Market Pilot and United Utilities Petteril projects under the Landscape Enterprise Networks platform.

There is potential for stacking of carbon and water quantity benefits, with a number of utilities investing in peatland restoration for carbon under the Peatland Code whilst justifying investment to Ofwat in part via anticipated improvements to water quality e.g. UU's SCAMP project and Southwest Water's Upstream Thinking project. Water utilities are estimated to have spent £3.5 billion from 2010 to 2016 on environmental improvement (Haigh, 2016). This pipeline could grow as a result of Ofwat's 2021 announcement of £2.7 billion in investment in environmental projects to support a green recovery.

There are, however, some doubts over the future of these markets due to ambiguity over what activities can be paid for by water companies, with 'fair share' responsibilities preventing them paying farmers to reduce nutrient loads so they can prove that their water quality measures do not hamper other sectors' ability to achieve their fair share of reducing a target nutrient load. There is also a lack of standardised measurement and verification methods for determining and attributing water quality improvements, and doubts over the extent to which some of the habitats and land use changes reviewed earlier in this report can reliably deliver water quality improvements. There is also significant variation in prices paid to landowners and a lack of transparency in pricing which may limit the supply of projects as this market begins to scale.

4.2.4 Stacking payments for multiple services

It may be possible to stack payments for multiple ecosystem services from paludiculture, including biodiversity and water quality and flood risk alleviation, with credits from the Peatland Code or future codes for soil carbon, biochar, saltmarsh creation or agroforestry. However, if stacking is done via markets governed by additionality criteria, this would only be possible in sites where costs are high and it can be demonstrated that projects would not be financially viable without both finance being combined from each source, to ensure that additionality criteria are met for each scheme.

Broadly speaking, there are two approaches to stacking:

1. Stacking may be done at the scale of an individual site, typically finding a single buyer for each ecosystem service, ensuring that additionality tests are met for each of the codes being used;

2. Stacking may occur at a landscape scale with pooled buyers via a "trading space" or "regional ecosystem market" approach where the terms and conditions (including payment rates, contract length and verification methods) are negotiated collectively by landowners and other rights holders from across the landscape with the investment pool. Multiple buyers are identified and their investment is pooled to achieve landscape scale benefits, and to ensure that payments for one service do not compromise the delivery of other services that are being demanded by investors in the pool. Landowners also need to be aggregated across property boundaries, and negotiate jointly with the investor pool to reach sufficiently attractive prices for the work. The dominant market model for this approach is Landscape Enterprise Networks (Reed et al., 2022b).

Alternatively, although not technically stacking, it may be possible to sell multiple ecosystem services by splitting up their delivery in time or space. It may be possible to: i) divide up land and apply different codes to different parcels of land; and ii) split over a restoration trajectory, for example using peatland carbon finance to go from eroding to modified condition, then using biodiversity funding to go from modified to near natural condition (Reed et al., 2022a).

4.3 Scaling and maintaining the integrity of the Peatland Code

There are a number of ways that the operation of the Peatland Code could be scaled, including the extension of the code to new habitats and systems, as discussed above, and the development of mechanisms and services to increase the supply of projects and govern increasing demand for peatland carbon.

To ensure strong future supply of projects into the Peatland Code to meet growing demand, the following suggestions may be considered:

- Training and other capacity building work with land advisory services, to enable them to develop projects and act as intermediaries for peatland carbon, and carbon and other ecosystem services across other ecosystem markets. This has the potential to help market Peatland Code projects (and where relevant, stack projects for other ecosystem markets), and simplify the interface between codes for landowners and managers.
- The Peatland Code brand could be trademarked and its intellectual property could be protected, to head off potential threats from new competing codes offering peatland carbon using metrics developed by the Peatland Code and intermediaries offering Peatland Code credits alongside peatland carbon without robust verification or other standards.
- The Peatland Code (along with other Codes) could evolve to become a recognised set of standards and metrics that would be available to all private or NGO-led schemes seeking to market carbon credits linked to peat restoration. In this scenario, the Peatland Code would move away from directly supporting individual projects towards more of a regulatory/compliance role. This could help to leverage greater private investment into peat restoration, while avoiding a 'race to the bottom' in which the cheapest, least regulated operators capture carbon investment at the expense of high-quality, verified schemes.
- A regulator, such as the Financial Conduct Authority, could be appointed to regulate carbon and other ecosystem markets in the UK, ensuring compliance with Peatland Code standards as described above.
- Standardised contract templates could be developed for the Peatland Code to level-up the contractual protections offered to landowners and managers by buyers and intermediaries.
- There is potential for central government to act as an owner and operator of carbon codes in the UK e.g. JNCC could review the emergence of new codes for integration with existing codes where possible, and maintain standards against Defra's Core Carbon Principles, currently being updated for integration with the Environmental Reporting Guidelines. This option needs

to be contrasted to the current market-based approach where there may be multiple competing codes for the same habitats, land uses and ecosystem services. A public sector owner of the Peatland Code and/or other ecosystem markets, could also provide the long-term stability of funding for operations needed by these markets.

To further stimulate and govern demand for peatland carbon:

- It may be possible to scale up private investment into the UK Nature based codes such as Peatland by standardising the legal framework, providing the technology and carbon insurance demanded by institutional investors and allowing the codes to work together.
- There is potential for Peatland Code projects to integrate with place-based schemes like LENs, which could also provide a mechanism for blending with public funding via ELMs, helping de-risk private investment (though potentially creating additionality issues)
- Blending mechanisms like a Peatland Carbon Guarantee could also be explored for integration into future rounds of the Climate for Nature Fund, to further de-risk and scale investment in peatland carbon.

References

Artz, R.R.E., Potts, J., Chapman, S., Wilson, D., Heinemeyer, A., Evans, C., Renou-Wilson, F. Emission factors applicable to UK peatlands. Report to the Department of Energy and Climate Change,

Ashby, M. and Heinemeyer, A., 2019. Prescribed burning impacts on ecosystem services in the British uplands: A methodological critique of the EMBER project. Journal of Applied Ecology, 57, 2112–2120.

Baird, A.J., Evans, C.D., Mills, R., Morris, P.J., Page, S.E., Peacock, M., Reed, M., Robroek, B.J.M., Stoneman, R., Swindles, G.T. and Thom, T., 2019. Validity of managing peatlands with fire. Nature Geoscience, 12, 884-885.

Beaulne, J., Garneau, M., Magnan, G. and Boucher, É., 2021. Peat deposits store more carbon than trees in forested peatlands of the boreal biome. Scientific reports, 11, 2657.

Beck, H.E., Zimmermann, N.E., McVicar, T.R., Vergopolan, N., Berg, A. and Wood, E.F., 2018. Present and future Köppen-Geiger climate classification maps at 1-km resolution. Scientific data, 5, 180214.

Brown, P., Cardenas, L., Choudrie, S., Del Vento, S., Karagianni, E., MacCarthy, J., Mullen, P., Passant, N., Richmond, B., Smith, H., Thistlethwaite, G., Thomson, A., Turtle, L., Wakeling, D., 2021. UK Greenhouse Gas Inventory, 1990 to 2019. Annual Report for Submission under the Framework Convention on Climate Change

Burden, A., Radbourne, A., Williamson, J., Evans, C. (2020). A rapid method for basic peatland condition and national-scale analysis. Report to Welsh Government for the Sustainable Management Scheme Peatlands project.

Cain, M., Lynch, J., Allen, M.R., Fuglestvedt, J.S., Frame, D.J. and Macey, A.H., 2019. Improved calculation of warming-equivalent emissions for short-lived climate pollutants. NPJ climate and atmospheric science, 2, 1-7.

Cooper, M., Evans, C., Zielinski, P., Levy, P., Gray, A., Peacock, M., Norris, D., Fenner, N. and Freeman, C., 2014. Infilled ditches are hotspots of landscape methane flux following peatland re-wetting. Ecosystems, 17, 1227-1241.

Couwenberg, J., Thiele, A., Tanneberger, F., Augustin, J., Bärisch, S., Dubovik, D., Liashchynskaya, N., Michaelis, D., Minke, M., Skuratovich, A. and Joosten, H., 2011. Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. Hydrobiologia, 674, 67-89.

Chapman, P.J., Moody, C.S., Turner, T.E., McKenzie, R., Dinsmore, K.J., Baird, A.J., Billett, M.F., Andersen, R., Leith, F. and Holden, J., 2022. Carbon concentrations in natural and restoration pools in blanket peatlands. Hydrological Processes, 36, e14520.

Davies, G.M., Kettridge, N., Stoof, C.R., Gray, A., Ascoli, D., Fernandes, P.M., Marrs, R., Allen, K.A., Doerr, S.H., Clay, G.D. and McMorrow, J., 2016. The role of fire in UK peatland and moorland management: the need for informed, unbiased debate. Philosophical Transactions of the Royal Society B: Biological Sciences, 371, 20150342.

Elliot J, Ritson J, Reed MS, Kennedy-Blundell O (2022). The opportunities of agri-carbon markets: policy and practice. Oxford Farming Conference, WWF-UK and Tesco. Available online at: https://www.ofc.org.uk/blog/ofc-launches-ground-breaking-reports-natural-capital.

Evans, C., Morrison, R., Burden, A., Williamson, J., Baird, A., Brown, E., Callaghan, N., Chapman, P., Cumming, A., Dean, H. and Dixon, S., Dooling, G., Evans, J., Gauci, V., Grayson, R., Haddaway, N., He, Y., Heppell, K., Holden, J., Hughes, S., Kaduk, J., Jones, D., Matthews, R., Menichino, N., Misselbrook, T., Page, S., Pan, G., Peacock, M., Rayment, M., Ridley, L., Robinson, I., Rylett, D., Scowen, M., Stanley, K., Worrall, F. 2016. Lowland peatland systems in England and Wales–evaluating greenhouse gas fluxes and carbon balances. Report to Defra on Project SP1210, Centre for Ecology and Hydrology, Bangor.

Evans, C.D., Renou-Wilson, F. and Strack, M., 2016. The role of waterborne carbon in the greenhouse gas balance of drained and re-wetted peatlands. Aquatic Sciences, 78, 573-590.

Evans, C., Artz, R., Moxley, J., Smyth, M.A., Taylor, E., Archer, E., Burden, A., Williamson, J., Donnelly, D., Thomson, A. Buys, G., Malcolm, H., Wilson, D., Renou-Wilson, F. 2017. Implementation of an emissions inventory for UK peatlands. Report to the Department for Business, Energy and Industrial Strategy, Centre for Ecology and Hydrology, Bangor.

Evans, C.D., Baird, A.J., Green, S.M., Page, S.E., Peacock, M., Reed, M.S., Rose, N.L., Stoneman, R., Thom, T.J., Young, D.M. and Garnett, M.H., 2019. Comment on: "Peatland carbon stocks and burn history: Blanket bog peat core evidence highlights charcoal impacts on peat physical properties and long-term carbon storage," by A. Heinemeyer, Q. Asena, WL Burn and AL Jones. Geo: Geography and Environment, 6, e00075.

Evans, C.D., Peacock, M., Baird, A.J., Artz, R.R.E., Burden, A., Callaghan, N., Chapman, P.J., Cooper, H.M., Coyle, M., Craig, E. and Cumming, A., 2021. Overriding water table control on managed peatland greenhouse gas emissions. Nature, 593, 548-552.

Evans, C.D., Callaghan, N., Jaya, A., Grinham, A., Sjogersten, S., Page, S.E., Harrison, M.E., Kusin, K., Kho, L.K., Ledger, M. and Evers, S., 2021b. A novel low-cost, high-resolution camera system for measuring peat subsidence and water table dynamics. Frontiers in Environmental Science, 9:630752.

Evans, C., Freeman, B., Artz, R., Burden, A., Burton, R., Chadwick, D., Clilverd, H., Cooper, H., Cowan, N., Cumming, A., Hudson, M., Jones, D., Kaduk, J., Newman, T., Page, S., Potts, J., Morrison, R., 2022. Wasted Peat Emission Factor Assessment. Interim Report to the Department for Business, Energy and Industrial Strategy.

Goulsbra, C., Evans, M. & Allott, T. (2013) Towards the estimation of CO_2 emissions associated with POC fluxes from drained and eroding peatlands. In: Emissions of greenhouse gases associated with peatland drainage waters. Report to Defra under project SP1205: Greenhouse gas emissions associated with nongaseous losses of carbon from peatlands – fate of particulate and dissolved carbon. Report to the Department of Environment, Food and Rural Affairs, UK.

Granath, G., Evans, C.D., Strengbom, J., Fölster, J., Grelle, A., Strömqvist, J. and Köhler, S.J., 2021. The impact of wildfire on biogeochemical fluxes and water quality in boreal catchments. Biogeosciences, 18, 3243-3261.

Haigh N, 2016. Water quality and agriculture: Payment for Ecosystems Services (PES) schemes to reduce the adverse water impacts of land management in England. Defra. Available online at: https://www.oecd.org/environment/resources/England-case-study-water-quality-and-agriculture-diffuse-pollution.pdf

Harper, A.R., Doerr, S.H., Santin, C., Froyd, C.A. and Sinnadurai, P., 2018. Prescribed fire and its impacts on ecosystem services in the UK. Science of the Total Environment, 624, 691-703.

Heinemeyer, A., Asena, Q., Burn, W.L. and Jones, A.L., 2018. Peatland carbon stocks and burn history: Blanket bog peat core evidence highlights charcoal impacts on peat physical properties and long-term carbon storage. Geo: Geography and Environment, 5, .e00063.

Heinemeyer A., Vallack H.W., Morton P.A., Pateman R., Dytham C., Ineson P., McClean C., Bristow C. and Pearce-Higgins J.W. (2019) Restoration of heather-dominated blanket bog vegetation on grouse moors for biodiversity, carbon storage, greenhouse gas emissions and water regulation: comparing burning to alternative mowing and uncut management. Final Report to Defra on Project BD5104, Stockholm Environment Institute at the University of York, York, UK.

Howie, S.A. and Hebda, R.J., 2018. Bog surface oscillation (mire breathing): A useful measure in raised bog restoration. Hydrological Processes, 32, 1518-1530.

IPCC, 2014. 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. Intergovernmental Panel on Climate Change

IPCC, 2019. Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, forestry and other land use (AFOLU). Intergovernmental Panel on Climate Change.

IPCC, 2021. Climate Change 2021: The Physical Science Basis. Intergovernmental Panel on Climate Change

Kareksela, S., Haapalehto, T., Juutinen, R., Matilainen, R., Tahvanainen, T. and Kotiaho, J.S., 2015. Fighting carbon loss of degraded peatlands by jump-starting ecosystem functioning with ecological restoration. Science of the Total Environment, 537, 268-276.

Kendall H, Parkhurst R, Black H, Rudman H, Chapman PJ, Ziv G, Cannon N, Yeluripati J, Orman M, Leeson A, Phelps J, Reed MS (in prep.) What makes an operational Farm Soil Carbon Code? Insights from a global comparison of existing soil carbon codes using a structured analytical framework. *Journal of European Public Policy*

Leppelt, T., Dechow, R., Gebbert, S., Freibauer, A., Lohila, A., Augustin, J., Drösler, M., Fiedler, S., Glatzel, S., Höper, H. and Järveoja, J., 2014. Nitrous oxide emission budgets and land-use-driven hotspots for organic soils in Europe. Biogeosciences, 11, 6595-6612.

Li, C., Huang, Y., Guo, H., Cui, L. and Li, W., 2018. Draining effects on recent accumulation rates of C and N in zoige alpine peatland in the Tibetan Plateau. Water, 10,.576.

Lindsay, R., 2010. Peatbogs and carbon: a critical synthesis to inform policy development in oceanic peat bog conservation and restoration in the context of climate change. University of East London, Environmental Research Group. Report for RSPB Scotland. Available at: https://repository.uel.ac.uk/item/862y6.

Lindsay, R., Clough, J., Clutterbuck, B., Bain, C., Goodyer, E., 2019. Eyes on the Bog. IUCN Peatland Programme, available at: https://www.iucn-uk-peatlandprogramme.org/sites/default/files/header-images/Eyes%20on%20the%20Bog%20Manual.pdf.

Lin, F., Zuo, H., Ma, X. and Ma, L., 2022. Comprehensive assessment of nitrous oxide emissions and mitigation potentials across European peatlands. Environmental Pollution, 119041.

Liu, H., Wrage-Mönnig, N. and Lennartz, B., 2020. Rewetting strategies to reduce nitrous oxide emissions from European peatlands. Communications Earth & Environment, 1, 1-7.

Loisel, J., Yu, Z., Beilman, D.W., Camill, P., Alm, J., Amesbury, M.J., Anderson, D., Andersson, S., Bochicchio, C., Barber, K. and Belyea, L.R., 2014. A database and synthesis of northern peatland soil properties and Holocene carbon and nitrogen accumulation. the Holocene, 24, 1028-1042.

Marrs, R.H., Marsland, E.L., Lingard, R., Appleby, P.G., , Piliposyan, G.T., , Rose, R.J., O'Reilly, J., Milligan, G., Allen, K.A., Alday, J.G., Santana, V., Lee, H., Halsall, K., Chiverrell, R.C. 2018. Experimental evidence for sustained carbon sequestration in fire-managed, peat moorlands. Nature Geoscience, 12, 108–112.

Morton, P.A. and Heinemeyer, A., 2019. Bog breathing: the extent of peat shrinkage and expansion on blanket bogs in relation to water table, heather management and dominant vegetation and its implications for carbon stock assessments. Wetlands Ecology and Management, 27, 467-482.

Mulholland, B., Abdul-Aziz, I., Lindsay, R., McNamara, N., Keith, A., Page, S., Clough, J., Freeman, B., Evans, C.D., 2020. An assessment of the potential of paludiculture in England and Wales. Literature Review: Defra Project SP1218. Available at https://lowlandpeat.ceh.ac.uk/sites/default/files/Defra%20LP2%20paludiculture%20report%20-%20April%202020.pdf.

The Nature Conservancy (2021) Biodiversity Net Gain in England: developing effective market mechanisms. Available online at: https://www.nature.org/content/dam/tnc/nature/en/documents/TNC_BiodiversityNetGain_England.pdf

Neubauer, S.C. and Megonigal, J.P., 2015. Moving beyond global warming potentials to quantify the climatic role of ecosystems. Ecosystems, 18, 1000-1013.

Pangala, S.R., Moore, S., Hornibrook, E.R. and Gauci, V., 2013. Trees are major conduits for methane egress from tropical forested wetlands. New Phytologist, 197, 524-531.

Peacock, M., Evans, C.D., Fenner, N. and Freeman, C., 2013. Natural revegetation of bog pools after peatland restoration involving ditch blocking—The influence of pool depth and implications for carbon cycling. Ecological engineering, 57, 297-301.

Peacock, M., Audet, J., Bastviken, D., Futter, M.N., Gauci, V., Grinham, A., Harrison, J.A., Kent, M.S., Kosten, S., Lovelock, C.E. and Veraart, A.J., 2021. Global importance of methane emissions from drainage ditches and canals. Environmental Research Letters, 16, 044010.

Poyda, A., Reinsch, T., Skinner, R.H., Kluß, C., Loges, R. and Taube, F., 2017. Comparing chamber and eddy covariance based net ecosystem CO2 exchange of fen soils. Journal of Plant Nutrition and Soil Science, 180, 252-266.

Reed MS, Fitton R, Khuri S (2022a) *Investing in Ecosystem Services of Peatland Restoration.* Report commissioned for Great North Bog partnership.

Reed MS, Curtis T, Kendall H, Gosal A, Andersen SP, Ziv G, Attlee A, Hay M, Hill D, Martin-Ortega J, Martino S, Olesen AS, Prior S, Rodgers C, Rudman H, Tanneberger F, Waylen K (2022b) Integrating ecosystem markets to deliver landscape-scale public benefits from nature. PLOS ONE 17(1): e0258334.

Renger, M., Wessolek, G., Schwärzel, K., Sauerbrey, R. and Siewert, C., 2002. Aspects of peat conservation and water management. Journal of Plant Nutrition and Soil Science, 165, 487-493.

Rigney, C., Wilson, D., Renou-Wilson, F., Müller, C., Moser, G. and Byrne, K.A., 2018. Greenhouse gas emissions from two rewetted peatlands previously managed for forestry. Mires and Peat, 21, 24.

Simon, R., Mitchell, A., Evans, C., Whitaker, J., Thomson, A. and Keith, A., 2021. Greenhouse gas removal methods and their potential UK deployment. Report to the Department for Business, Energy and Industrial Strategy by Element Energy and the UK Centre for Ecology and Hydrology, 120 pp.

Smyth, M.A., Taylor, E.S., Birnie, R.V., Artz, R.R.E., Dickie, I., Evans, C., Gray, A., Moxey, A., Prior, S., Littlewood, N. and Bonaventura, M., 2015. Developing peatland carbon metrics and financial modelling to inform the pilot phase UK peatland code. Report to Defra for Project NR0165. Crichton Carbon Centre, Dumfries.

Tiemeyer, B., Freibauer, A., Borraz, E.A., Augustin, J., Bechtold, M., Beetz, S., Beyer, C., Ebli, M., Eickenscheidt, T., Fiedler, S. and Förster, C., 2020. A new methodology for organic soils in national greenhouse gas inventories: Data synthesis, derivation and application. Ecological Indicators, 109, 105838.

Moore, T.R. and Turunen, J., 2004. Carbon accumulation and storage in mineral subsoil beneath peat. Soil Science Society of America Journal, 68, 690-696.

Williamson, J., Evans, C., Spears, B., Pickard, A., Chapman, P.J., Feuchtmayr, H., Leith, F. and Monteith, D., 2020. Will UK peatland restoration reduce dissolved organic matter concentrations in upland drinking water supplies? Hydrology and Earth System Sciences Discussions, 1-21.

Williamson, J., Burden, A., Evans, C., 2021. Using Earth Observation to map peatland condition in Wales. Report to Welsh Government for the Sustainable Management Scheme Peatlands project.

Young, D.M., Baird, A.J., Morris, P.J. and Holden, J., 2017. Simulating the long-term impacts of drainage and restoration on the ecohydrology of peatlands. Water Resources Research, 53, 6510-6522.

Zhou, Y., Evans, C.D., Chen, Y., Chang, K.Y. and Martin, P., 2021. Extensive remineralization of peatland-derived dissolved organic carbon and ocean acidification in the Sunda Shelf Sea, Southeast Asia. Journal of Geophysical Research: Oceans, 126, e2021JC017292.

zu Ermgassen, S., Drewniok, M., Bull, J.W., Walker, C.C., Mancini, M., Ryan-Collins, J. and Serrenho, A.C., 2022. A home for all within planetary boundaries: pathways for meeting England's housing needs without transgressing national climate and biodiversity goals. OSF Preprints DOI: 10.31219/osf.io/5kxce

Annex 1. Comparative emission factor data tables

This annex provides comparative emission factor data as per Section 1 (Table 1.1 to 1.3), but with data from studies for which confidence in the methodology was lower, or where there was evidence of sustained site flooding retained in the analysis. See Section 1 for an explanation of the rationale for excluding these data from the main tables.

Table A.1.1. *On-site, full dataset-based, CO_2 emission factors (t CO_2 ha^{-1} yr^{-1}) with standard errors and 95% confidence intervals. Note that Tier 2 values are not provided for Woodland, because the UK national inventory uses a Tier 3 model-based approach for this category. All data were included regardless of concerns about methodology to determine e.g., biomass offtake or gap-filling model (categories affected highlighted in yellow), validity of assignment to the relevant condition category (categories affected in brown), or observations from sites with flooding (mean annual WTD > 5 cm) or temporary inundation of the whole site (categories highlighted in blue). A further decision regarded whether the newly calculated EF was sufficiently robust to replace Tier 1 (categories affected in grey). See Section 1.2 for details.*

Category	Mean Tier 2 EF	Standard Error	Lower 95% CI	Upper 95% CI
Cropland	31.21	5.73	19.25	43.17
Intensive Grassland	27.91	4.39	19.07	36.75
Extensive Grassland	17.96	5.00	7.82	28.09
Extracted (Domestic)	6.02	0.97	-6.24	18.29
Extracted (Industrial)	5.44	0.83	3.69	7.18
Eroding Bare Peat	5.44	0.83	3.69	7.18
Modified bog (heather or grass-dominated)	0.03	0.61	-1.17	1.24
Rewetted Modified Bog	-2.87	0.98	-4.92	-0.82
Rewetted Bog	-0.46	1.09	-2.63	1.72
Rewetted Fen	-1.89	3.60	-9.25	5.48
Near-Natural Bog	-2.87	0.98	-4.92	-0.82
Near-Natural Fen	-5.29	1.70	-8.96	-1.61

Table A.1.2. *On-site, full dataset-based, CH_4 emission factors (kg CH_4 ha^{-1} yr^{-1}) with standard errors and 95% confidence intervals. Data from flooded (mean annual WTD > 5 cm) or temporarily but wholly inundated sites have been included (highlighted in blue) See Section 1.2 for details.*

Category	Mean Tier 2 EF	Standard Error	Lower 95% CI	Upper 95% CI
Woodland	2.65	1.98	-22.52	27.81
Cropland	1.96	1.16	-0.47	4.39
Intensive Grassland	29.03	19.10	-9.33	67.38
Extensive Grassland	35.91	16.60	2.24	69.58
Extracted (Industrial/Domestic)	42.72	19.54	-0.29	85.74
Eroding Bare Peat	42.72	19.54	-0.29	85.74
Modified Bog (heather or grass-dominated)	61.75	11.62	38.34	85.16
Rewetted Modified Bog	128.44	45.85	33.10	223.78
Rewetted Bog	171.71	45.20	80.92	262.50
Rewetted Fen	557.29	273.89	2.35	1112.24
Near-Natural Bog	128.44	45.85	33.10	223.78
Near-Natural Fen	152.78	20.33	108.86	196.70

Table A.1.3. *On-site N_2O emission factors (kg N_2O-N ha^{-1} yr^{-1}) with standard errors and 95% confidence intervals. Data from flooded (mean annual WTD > 5 cm) or temporarily but wholly inundated sites have been included (highlighted in blue) See Section 1.2 for details.*

Category	**Mean Tier 2 EF**	**Standard Error**	**Lower 95% CI**	**Upper 95% CI**
Woodland	1.48	0.81	-2.02	4.97
Cropland	16.28	4.42	6.98	25.57
Intensive Grassland	7.39	1.64	4.09	10.69
Extensive Grassland	1.82	0.99	-0.30	3.95
Extracted (Industrial/Domestic)	1.92	1.06	-2.65	6.49
Eroding Bare Peat	1.92	1.06	-2.65	6.49
Modified Bog (heather or grass-dominated)	0.13	0.05	0.02	0.24
Rewetted Modified Bog	0.03	0.02	-0.03	0.08
Rewetted Bog	0.26	0.17	-0.08	0.59
Rewetted Fen	1.79	1.57	-3.21	6.79
Near-Natural Bog	0.03	0.02	-0.03	0.08
Near-Natural Fen	1.00	1.00	0.00	0.00

Annex 2. Distribution of water table data in the Emission Factor Database

The following figures provide an assessment of the overall distribution of water table depth data in the updated Emission Factor Database. Approximately 60% of the values in the database had accompanying water table data.

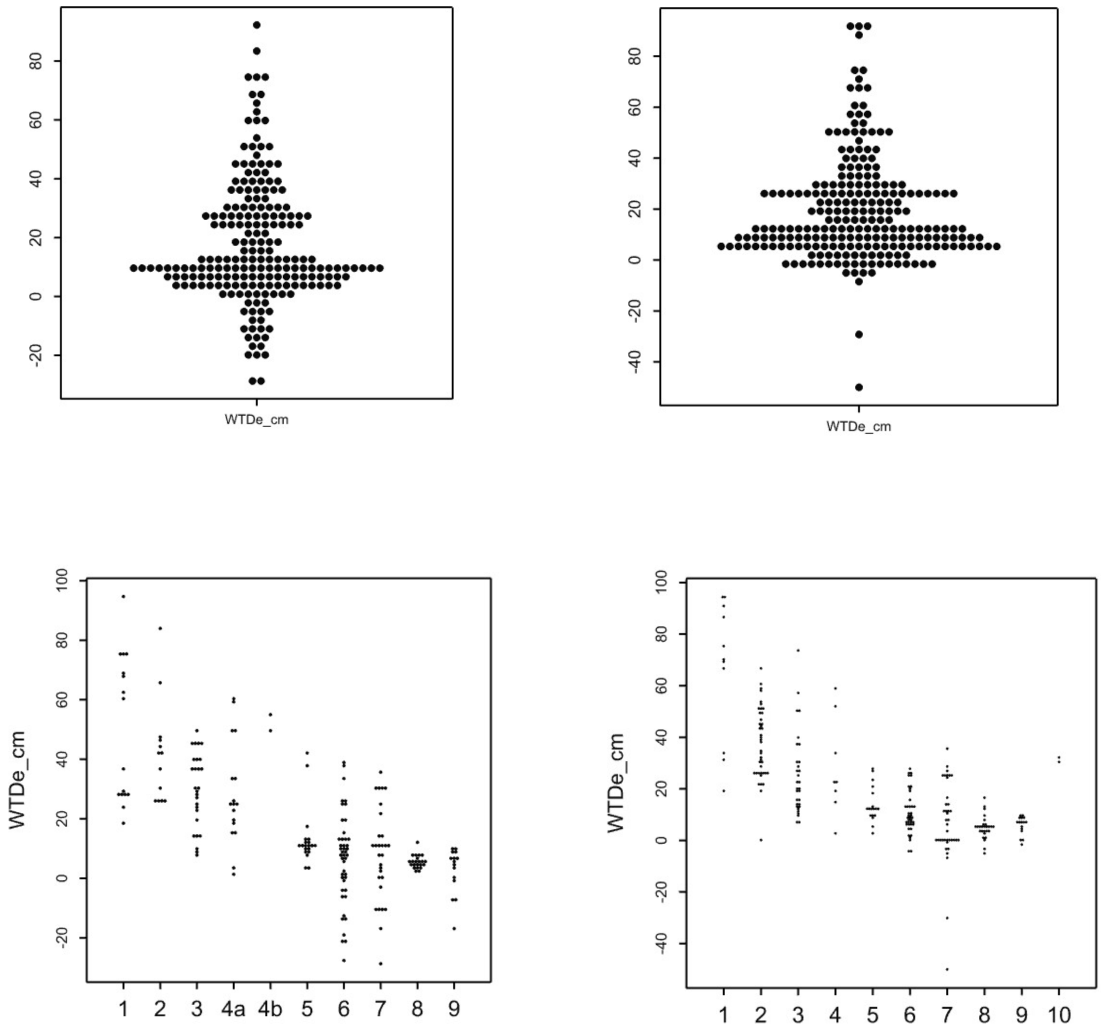

Figure A.2.1. *Histograms of the distribution of WTD depths (available for ~60% of entries) used to derive emission factors for CO_2 (left) and CH_4 (right). Upper plots show the entire dataset, lower plots show distributions by Tier 2 peat condition category (see table below for class codes). Note that studies for which average water levels were more than 5 cm above the peat surface (i.e. WTD < -5 cm) are included in the plots above but were excluded from subsequent emission factor calculations.*

Table A.2.1. *Number of observations that report WTD out of the total number of observations per land use class, with mean value for WTDe in cm below surface in brackets. Sites with flooding (mean annual WTD > 5 cm above surface) or temporary inundation (as mentioned in primary publication) are included in this table.*

Land use class	**NEP**	**CH_4**
1 – cropland	16/17 (50.06)	11/17 (66.54)
2- intensive grassland	13/19 (41.77)	41/51 (37.55)
3 – extensive grassland	27/31 (29.71)	31/40 (25.17)
4a – industrial extraction	17/21 (28.21)	8/13 (28.50)
4b – domestic extraction	2/3 (52.5)	
5 – modified	20/89 (13.21)	18/49 (13.89)
6 – rewetted bog	45/64 (6.615)	40/52 (10.81)
7 – rewetted fen	29/37 (8.128)	39/49 (6.925)
8 – near natural bog	22/24 (5.681)	27/28 (2.970)
9 – near natural fen	15/17 (2.597)	16/16 (5.735)
10 - woodland	N/A	2/5 (30.95)

Table A.2.2. *Number of observations that report WTD out of the total number of observations per land use class, with mean value for WTDe in cm below surface in brackets. Sites with flooding (mean annual WTD > 5 cm above surface) or temporary inundation (as mentioned in primary publication) were excluded.*

Land use class	**NEP**	**CH_4**
1 – cropland	16/17 (50.06)	11/17 (66.54)
2- intensive grassland	13/19 (41.77)	41/51 (37.55)
3 – extensive grassland	27/31 (29.71)	31/40 (25.17)
4a – industrial extraction	17/21 (28.21)	8/13 (28.50)
4b – domestic extraction	2/3 (52.5)	
5 – modified	20/89 (13.21)	18/49 (13.89)
6 – rewetted bog	36/55 (12.23)	32/42 (9.946)
7 – rewetted fen	23/31 (13.97)	25/33 (11.99)
8 – near natural bog	22/24 (5.681)	27/28 (4.970)
9 – near natural fen	12/14 (5.813)	14/14 (5.554)
10 - woodland	N/A	2/5 (30.95)