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A new methodology for organic soils in national greenhouse gas inventories: Data synthesis, derivation and application

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ABSTRACT

Drained organic soils are large sources of anthropogenic greenhouse gases (GHG) in many European and Asian countries. Therefore, these soils urgently need to be considered and adequately accounted for when attempting to decrease emissions from the Agriculture and Land Use, Land Use Change and Forestry (LULUCF) sectors. Here, we describe the methodology, data and results of the German approach for measurement, reporting and verification (MRV) of anthropogenic GHG emissions from drained organic soils and outline ways forward towards tracking drainage and rewetting. The methodology was developed for and is currently applied in the German GHG inventory under the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol.

Spatial activity data comprise high resolution maps of land-use, type of organic soil and mean annual water table (WT). The WT map was derived by a boosted regression trees model from data of more than 1000 dipwells. Emissions of carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) were synthesized from a unique national data set comprising more than 250 annual GHG balances from 118 sites in most land-use categories and types of organic soils. Measurements were performed with harmonized protocols using manual chambers. Non-linear response functions describe the dependency of CO_2 and CH_4 fluxes on mean annual WT, stratified by land-use where appropriate. Modelling results were aggregated into "implied emission factors" for each land-use category, taking into account the uncertainty of the response functions, the frequency distribution of the WT within each land-use category and further GHG sources such as dissolved organic carbon or CH_4 emissions from ditches. IPCC default emission factors were used for these minor GHG sources. In future, response functions could be applied directly when appropriate WT data is available. As no functional relationship was found for N_2O emissions, emission factors were calculated as the mean observed flux per land-use category. In Germany, drained organic soils emit more than 55 million tons of GHGs per year, of which 91% are CO_2. This is equivalent to around 6.6% of the national GHG emissions in 2014. Thus, they are the largest GHG source from agriculture

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and LULUCF. The described methodology is applicable on the project scale as well as in other countries where similar data are collected.

1. Introduction

Globally, drained peatlands and peat fires emit about 1 Gt carbon dioxide (CO_2)-equivalents per year which corresponds to 10% of the greenhouse gas (GHG) emissions from agriculture, land-use change and forestry (Smith et al., 2014). Furthermore, drainage and conversion to agriculture, forestry or peat extraction destroys valuable ecosystems with rare species (Succow and Joosten, 2001) and increases the losses of nutrients (Holden et al., 2004). Drained peatlands rank among the largest GHG sources from agriculture and forestry in many European and Asian countries, even when they cover only a small percentage of the national area (Tubiello et al., 2016; Drösler et al., 2008). To achieve implementation of the goals given by the Paris Agreement, reducing emissions from drained peatlands is urgently required. The current "4 per mille" initiative aims to increase carbon stocks in soils as a compensation for anthropogenic GHG emissions (Minasny et al., 2017). However, the protection of the large carbon stocks in natural peatlands and the reduction of emissions from drained organic soils by rewetting is direly needed to not counterbalance any potential success in the management of mineral soils.

National inventory reports under the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol report anthropogenic emissions from organic soils in the Agriculture and Land use, Land use change and Forestry (LULUCF) sectors. Basically, nitrous oxide (N_2O) emissions are reported in the sector agriculture, while carbon dioxide (CO_2) and methane (CH_4) emissions are reported in the LULUCF sector. Mandatory land use categories are forest land, cropland, grassland, wetland, settlement and other land.

According to the recent methodological guidance by the Intergovernmental Panel on Climate Change (IPCC) in the IPCC Wetlands Supplement (2014), rewetting is "the deliberate action of raising the water table on drained soils to re-establish water saturated conditions". In most cases, rewetting of peatlands significantly reduces GHG emissions, sometimes even to values close to zero (IPCC, 2014; Wilson et al., 2016) at moderate costs (Bonn et al., 2015). Estimating the emissions of drained peatland and thus the emission reduction potential prior to taking any measures is much more challenging as values strongly vary depending on hydrological dynamics, soil properties and other factors (Tiemeyer et al., 2016). The emission reductions are accountable under the Kyoto Protocol (UNFCCC, 1998) and additionally in Europe under the so-called LULUCF Decision (EC, 2013) and its recent amendment (EC, 2018). Peatland rewetting can contribute to various land-based activities under the Kyoto Protocol depending on the mandatory or chosen activities and on the land use in the base year: Forest Management, Grazing Land Management, Cropland Management and the newly introduced activity Wetland Drainage and Rewetting, which has specifically been designed for peatland rewetting (Decision 2/CMP.7, UNFCCC, 2012). Furthermore, the recent update of EU LULUCF regulation (EC, 2018) demands mandatory accounting of "managed wetlands" from 2026 onwards.

National GHG inventories must comply with the IPCC quality criteria of *transparency* – i.e. complete, accessible and timely documentation, *completeness* of anthropogenic GHG sources and sinks, *consistency* in time and space, *comparability* between countries, and *accuracy* (IPCC, 2000, 2003, 2006). This requires spatial and temporal activity data detailed enough to detect the drainage and rewetting activities, and adequate methodological detail to detect responses of all GHG species, sources and sinks to changes in land-use, management and water table.

In parallel, peatland rewetting projects have emerged in the voluntary carbon market. Several project standards have been developed for guiding monitoring, reporting and verification (MRV) of avoided or reduced GHG emissions. Examples are the Verified Carbon Standard (VCS, now Verra) (WWF, 2014), Tanneberger and Wichtmann (2011), MoorFutures (Joosten et al., 2013) and the Peatland Code (Smyth et al., 2015). The MRV requirements refer to the measurement and calculation of GHGs ("Monitoring"), the documentation and presentation ("Reporting") and the proof of correctness by an independent review ("Verification") and apply conceptually to projects and to national GHG inventories. The criteria of *transparency, consistency* and *comparability* exist as well in the voluntary carbon market – at least among projects in the same voluntary project standard. Voluntary projects also have to prove *additionality*, which means that the GHG mitigation would not have occurred without the project. The criteria of *completeness* and *accuracy*, however, have been deemed impracticable for voluntary projects and have been replaced by the criterion *conservativeness*. This means that GHG species, sources or sinks may be neglected if they are small or if it is clear that the project will reduce, but not increase emissions. For instance, the VCS standard only considers CO_2, but not CH_4 and N_2O for rewetting projects in tropical peatlands (WWF, 2014), while the MoorFutures standard and Tanneberger and Wichtmann (2011) consider CO_2 and CH_4, but not N_2O for rewetting projects in Northern Germany and Belarus.

So far, large-scale approaches to estimate GHG emissions have been restricted to emission factors stratified by coarse classifications of climate, nutrient status and drainage (IPCC, 2014), or to national classifications by peat type and land-use (Nielsen et al., 2016) or peat type and drainage status (Arets et al., 2018). Project scale approaches often suggest vegetation-based proxies (e.g. Couwenberg et al., 2011), for which data are not available in adequate detail, temporal resolution or wall-to-wall at national level. Thus, vegetation-based methods do not comply with the national requirements of *accuracy* and *completeness*. However, project standards also allow alternative approaches by direct measurements or such models as the ones described in this paper.

Consistent MRV from project to national scale could be based on emission factors or response functions related to site conditions for which national data are available. Response functions of GHGs to site conditions, in particular land-use, nutrient status and water table, have been developed for some GHGs, e.g. for N_2O from all land-use types in Europe (Leppelt et al., 2014), for CO_2 and CH_4 from German grasslands (Tiemeyer et al., 2016) and for the net climate effect for German peatlands (Drösler et al, 2013). Such functions depending on site conditions and management help designing effective mitigation strategies and are the prerequisite for reporting consistently on mitigation activities in projects and national GHG inventories.

This paper aims to describe a detailed and novel national methodology for reporting anthropogenic GHG emissions from drained and – in future – rewetted organic soils. The general approach has been developed for, and applied in, the German GHG inventory under the UNFCCC and the Kyoto Protocol (UBA, 2016) and has successfully passed the in-country review by the UNFCCC Secretariat in 2016 (FCCC, 2017). Rewetting is currently not considered in the German inventory due to a lack of national time series of drainage status, but here we provide emission factors for future use. As new GHG data has emerged since 2015, we will report updated response functions here. Thus, the results differ from the values reported for 2014 (UBA, 2016). While results are specific to Germany, the methodology can be applied elsewhere on project, regional, and national scale.

2. Material and methods

2.1. Methodological overview and IPCC requirements

The IPCC Guidelines (2006, 2014) describe the methodologies and present default emission factors (EF) for the national estimates ("Tier 1" level) of GHG emissions from organic soils. These represent minimum standards for the completeness and the degree of detail in the inventory. The default EFs are only meant for the calculation of minor national GHG sources to guarantee resource-efficient reporting. All other so-called "key categories" have to be reported based on national data and, if possible, with more detailed data or methodologies than the default ones. A "key category" is defined as one that contributes to the 95% major national GHG sources in terms of the absolute level of emissions, the trend in emissions, or the uncertainty in level or trend (IPCC, 2000). Briefly, key categories are identified by sorting emissions by their magnitude, calculating the cumulative emissions and identifying all categories contributing to 95% of the total emissions. National methodologies and data shall be representative of national circumstances. Methodologies are classified into "Tier 2" approaches with similar equations, and possibly finer aggregation levels as Tier 1 based on national data sources. The "Tier 3" approach includes more detailed emission response functions to driving factors or process-based models (IPCC, 1996). Our approach is

- a Tier 3 approach simplified to Tier 2,
- compliant with the Wetlands Supplement,
- based on an unprecedented large GHG data set,
- and, to our best knowledge, the first reporting method to combine representative water table distributions with response functions.

The IPCC Wetlands Supplement (IPCC, 2014) has provided globally applicable methodologies and EFs for national GHG inventories for drained and rewetted organic soils. The guidance is mandatory for GHG emission estimates for the eligible activity of Wetland Drainage and Rewetting under the Kyoto Protocol and voluntary for national GHG inventories until 2020 only. The basic calculations for the national inventories are the same under the UNFCCC and the Kyoto Protocol. Reporting land based activities under the Kyoto Protocol additionally sets a minimum size for the activities, which defines the necessary spatial resolution, and requires spatially explicit tracking of land on which the activities occur.

In the following, the methodology and terminology for GHG emission reporting on drained and rewetted organic soils based on the Wetlands Supplement (IPCC, 2014) is described in brief. Organic soils are stratified into drained and wet areas. A time series of the drainage status is needed to distinguish between drained, rewetted and naturally wet organic soils. The area of drained organic soils $A_{drained}$ and the respective EFs are stratified by climate zone, land-use, nutrient and drainage status where possible (IPCC, 2014). GHGs are converted to the common metrics of CO_2-equivalents ($CO_{2\text{-eq}}$) by their global warming potentials (GWP) over a time horizon of 100 years according to the 4th IPCC assessment report (Forster et al., 2007): 1 kg CH_4 = 25 kg CO_2, 1 kg N_2O = 298 kg CO_2. Here, uncertainties are reported as the 95% percentiles of either the modelled emissions (CO_2, CH_4) or the data (N_2O).

The Wetlands Supplement considers the GHGs CO_2 (Eq. (1)), CH_4 (Eq. (2)) and N_2O. Here, we use the terms "CO_2-$C_{organic}$", "$CH_{4\ organic}$" and "N_2O-$N_{organic}$" for the composite emission factors (e.g., t C ha^{-1} yr^{-1}), and not for the total emissions (e.g., t C yr^{-1}) as in the Wetlands Supplement which does not supply any specific terminology for composite emission factors themselves (see also Fig. 1).

$$CO_2\text{-}C_{organic} = CO_2\text{-}C_{onsite} + CO_2\text{-}C_{DOC} + L_{fire} \quad (1)$$

With

- CCO_2-$C_{organic}$:
  CO_2-C emissions from organic soils (t C ha^{-1} yr^{-1})
- CO_2-C_{onsite}:
  on-site CO_2-C emissions from organic soils (t C ha^{-1} yr^{-1})
- CO_2-C_{DOC}:
  indirect CO_2 emissions from leaching of dissolved organic carbon (DOC) (t C ha^{-1} yr^{-1})
- L_{fire} CO_2-C:
  emissions from burning of drained organic soils (t C ha^{-1} yr^{-1}), currently not reported in Germany.

Carbon dioxide emissions from organic soil areas comprise the direct CO_2 emissions from the drained organic soil area $A_{drained}$ [ha] itself including the ditch area (CO_2-C_{onsite} or, in the chapter on rewetted

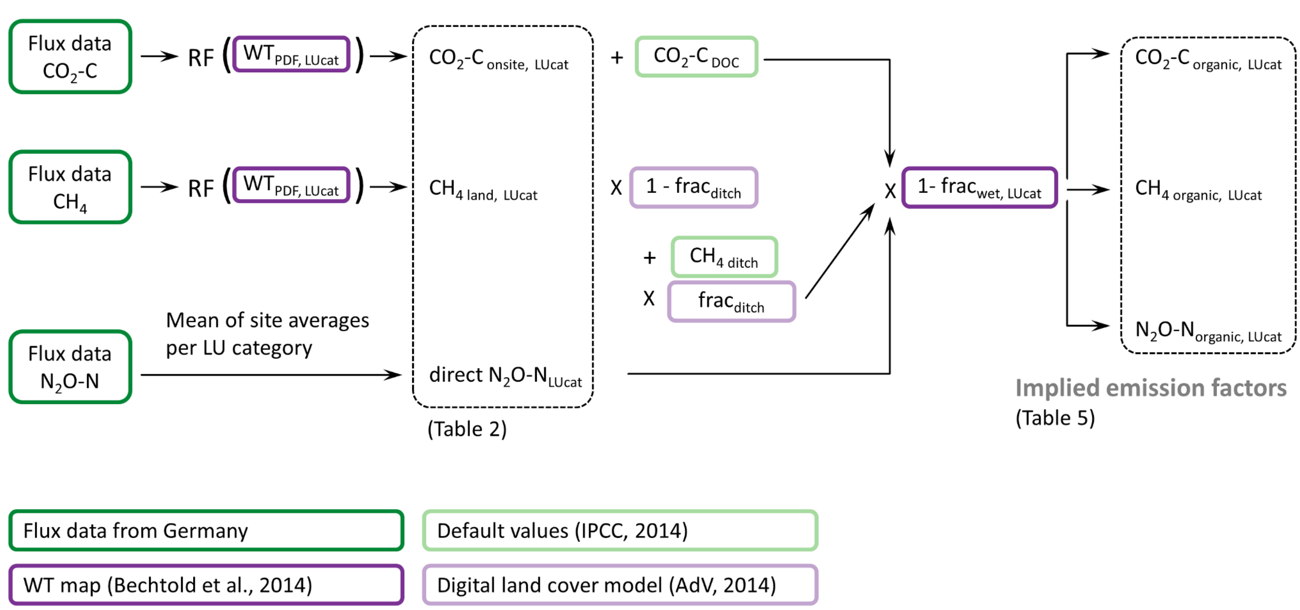

Fig. 1. Calculation of national implied emission factors for the individual land-use categories (LU_{cat}) of the German greenhouse gas inventory by response functions (RF) including data sources. WT_{PDF}: water table probability density distribution.

organic soils, CO_2-$C_{composite}$) as well as indirect CO_2 emissions from leaching of dissolved organic carbon (DOC) emitted downstream. CO_2-C_{onsite} includes the carbon export by harvested biomass.

Methane emissions from drained organic soil areas ($CH_{4\ organic}$ or, in the chapter on rewetted organic soils, $CH_{4\ soil}$) comprise the CH_4 emissions from the drained area ($CH_{4\ land}$) excluding ditches plus the CH_4 emissions from the ditches ($CH_{4\ ditch}$) which cover a certain fraction ($frac_{ditch}$) of the organic soil area (Eq. (2)). Methane emissions from peat fires are currently not reported in Germany, but guidance is available in IPCC (2014). In future, both CO_2 and CH_4 emissions from peat fire will be included in the German inventory. For rewetted areas, Eq. (2) can be simplified by neglecting the ditches, which are assumed to disappear over time or emit similar amounts of CH_4 as the surrounding land (IPCC, 2014).

$$CH_{4\ organic} = (1\text{-}frac_{ditch}) * CH_{4\,land} + frac_{ditch} * CH_{4\ ditch} \quad (2)$$

N_2O-$N_{organic}$ considers the nitrogen source from the mineralized peat. IPCC (2014) sets N_2O emissions from re-wetted areas to zero by default, but here we calculate emission factors for all land-use types.

2.2. Activity data

The German GHG inventory stratifies land into nine land-use categories: forest, cropland, grassland, shrubland, unutilized land (e.g. natural and degraded peatlands without a clear type of land-use, frequently influenced by former or surrounding drainage), water bodies, peat extraction areas, settlements, and other lands (e.g. rocks, glaciers; not occurring on organic soils).

Land-use data is derived from various sources for the entire national territory with complex decision trees according to data source quality. Most land-use data on organic soils originate from the Authoritative Topographic-Cartographic Information Systems – Digital Basic Landscape Model (ATKIS®-Basic DLM) with a spatial resolution of at least 1:25,000 (AdV, 2003). Land-use is represented in a spatially explicit way by grid-point sampling, which resulted in a sample raster of > 250,000 points for organic soils (UBA, 2016). The grid is derived from the national forest inventory. Each sample point represents roughly 6.4 ha ($A_{gridpoint}$). Ditch area is also estimated from the ATKIS®-Basic DLM, which contains ditches as line objects classified in three width classes.

Organic soils were defined according to IPCC (2006) as Histosols and other soils with histic horizons. The IPCC definition was matched as closely as possible with German soil types (Roßkopf et al., 2015). The area and location of organic soils is based on the map by Roßkopf et al. (2015), which has harmonized the best nationally available data provided by the German Federal States. The map shows organic soils on a conceptual 25 m grid, but is based on soil maps of heterogeneous spatial accuracy and age. The spatial scale ranges from 1:10,000 to 1:200,000, while the data age ranges from recently generated to the early 20th century. The area of organic soils was considered constant, neglecting a highly uncertain conversion of shallow organic soils into mineral soils by mineralization (Roßkopf et al., 2015).

The distribution of the mean annual water table (WT) of each land-use category was derived from the map of water tables in organic soils of Bechtold et al. (2014). This map provides the long-term WT representative of the situation of around 2010 as a 25 m grid. Using a machine learning algorithm (boosted regression trees), the WT map is based on information from the map of organic soils (Roßkopf et al., 2015), climate data, land use and drainage characteristics at landscape level, topographic information and long-term dipwell data from 1054 dipwells in 53 peatlands across Germany. The distributions of WTs per land use class used in this study represent the land-use specific variability, which has been obtained from the variation of the WT estimates (explained variability) plus their uncertainty (unexplained variability). Here, we define a water table below ground surface as negative and *vice versa*. Grid points were identified as drained organic soils by a WT deeper than −0.1 m below ground surface, while naturally wet and rewetted grid points are defined by a WT shallower than −0.1 m. The boundary was set according to the typical WT range in natural peatlands between 0 and −0.1 m (e.g. Jabłońska et al., 2011). So far, there is no time series of spatially explicit water table data available in Germany. In consequence, organic soils with naturally wet conditions cannot be differentiated yet from rewetted soils at national level. Similarly, we would not be able to identify newly drained areas, but as the few remaining natural or semi-natural peatlands are strictly protected, this is unlikely to have taken place after 1990. However, we can neither detect the effects of deepened drainage.

The amount of extracted peat was taken from national production statistics and converted to carbon by the IPCC (2006) default conversion factor of 0.07 t C m^{-3} air-dry peat. In Germany, peat is extracted nearly entirely for use in horticulture, but not for energy (Caspers and Schmatzler, 2009).

2.3. GHG measurement data

GHG flux data were synthesized from a unique data set of both published and unpublished GHG flux measurements in the five aggregated land-use categories forest and shrubland, cropland, grassland, unutilized land, and peat extraction areas, on bogs, fens and other organic soils in Germany (Tables S1 and S2). The data set included forest sites with sparse trees or large shrubs, which may not match the national forest definition. Therefore, forest and shrubland was combined into one joint land-use category here, which deviates from the national inventory where shrubland is combined with grassland.

The exchange of all GHGs was measured with harmonized protocols. Data originate from up to – depending on the gas – 21 different peatland areas, 149 sites and 320 annual GHG budgets over a wide range of site conditions representative of the organic soils in Germany (Tables 1 and S1). Data published previously (Beetz et al., 2013; Beyer, 2014; Beyer et al., 2015; Beyer and Höper, 2014; Drösler, 2005; Drösler et al., 2013; Eickenscheidt et al., 2014a, 2014b, 2015; Förster, 2016; Hoffmann et al., 2015; Leiber-Sauheitl et al., 2014; Metzger et al., 2015; Peichl-Brak, 2013; Poyda et al., 2016; Tiemeyer et al., 2016) were re-assessed and harmonized. For detailed methods of GHG measurements and calculation of annual GHG budgets see Tiemeyer et al. (2016) and the references listed in Table S1. The N_2O data were synthesized as part of a European study by Leppelt et al. (2014), but re-assessed and re-classified here to represent German conditions and the land-use categories in the GHG inventory. Additional recent data from Buchen et al. (2017) and Poyda et al. (2016) were added. Due to the small number of measurement sites (Hommeltenberg et al., 2014) forest on-site CO_2 emissions were derived from a general response function comprising all land-use categories.

Permanently flooded sites without or with lake vegetation were

Table 1
Number of annual GHG flux data by gas and land-use category: Number of annual budgets/number of sites/number of peatland areas (details on study areas and data sources in the supplementary data Table S1).

Land-use category	Number of annual GHG flux data		
	$CO_{2\ onsite}$	$CH_{4\ land}$	$N_2O_{organic}$
Forest and shrubland	0*	22/13/6	26/13/7
Cropland	34/15/6	42/17/7	43/19/9
Grassland	142/57/14	147/59/14	163/68/16
Unutilized land (undrained, degraded and rewetted sites)	81/44/11	81/46/10	84/47/12
Peat extraction	4/2/1	4/2/1	4/2/1
SUM	261/118/17	296/137/17	320/149/21

* data of Hommeltenberg et al. (2014) was not used for the derivation of emission factors due to differing methodological approaches, but for the verification of the forest emission factor.

excluded from this study as they represent flooded land on organic soils. GHG emissions from flooded land are excluded from mandatory reporting in GHG inventories as there is no agreed methodological guidance (IPCC, 2006, 2014). Sites with WT > 0 m and typical peatland vegetation were included. In line with the Wetlands Supplement (IPCC, 2014) and Wilson et al. (2016), undrained and rewetted peatlands were not distinguished for the derivation of EFs.

2.4. GHG emission response functions

Emission response functions were tested for CO_2-C_{onsite}, $CH_{4\ land}$ and direct N_2O emissions, for which national measurements were available. For CO_2-C_{onsite}, mean annual data of each site was used. Due to the strong non-linearity of the response individual annual values of $CH_{4\ land}$ were related to WT. The GHG response to multiple drivers was statistically analysed with univariate and multivariate linear and non-linear models, e.g. linear mixed effect models and fuzzy logic approaches (e.g. Leppelt et al., 2014; Tiemeyer et al., 2016). Here, drivers were restricted to those available at the national level:

- land use category
- type of organic soil (Roßkopf et al., 2015)
- Water table (WT, Bechtold et al., 2014)

Other drivers have also been identified as important proxies for some of the GHGs (soil properties, dynamic water table indicators, land use intensity, fertilization; Drösler et al., 2013; Leppelt et al., 2014, Tiemeyer et al., 2016), but are currently not available at national level. We parametrize non-linear response functions for both CO_2 (Gompertz, Eq. (3)) and CH_4 (exponential, Eq. (4)) in relation to WT with non-linear least squares (*nls*) estimation using R (R Core Team, 2016). Uncertainties of the function fit have been estimated by bootstrapping (n = 20,000) using the package *nlstools* (Baty et al., 2015).

$$CO_2\text{-}C(WT) = CO_2\text{-}C_{min} + CO_2\text{-}C_{diff}\, e^{-ae^{bWT}} \tag{3}$$

Here, CO_2-C_{min} is the lower asymptote, CO_2-C_{diff} the difference between upper and lower asymptote, while a and b are fitting parameters related to the displacement along the x-axis and the growth rate, respectively.

$$CH_4(WT) = CH_{4min} + ce^{-dWT} \tag{4}$$

$CH_{4\ min}$ is the lower asymptote and c and d are fitting parameters.

For CH_4, we fitted separate response functions to flux data from a) forest land, b) cropland and grassland, and c) unutilized organic soils and tested the difference of the bootstrapped parameter values by analysis of variance and a Tukey HSD (Honestly Significant Differences) post-hoc test (package *multcompView*, Graves et al., 2015)

WT can only act as driver of soil GHG emissions along the gradient of water saturation to dry conditions. Once the soil is saturated, WT relations lose their biogeochemical justification and would just indicate artefacts resulting from scatter in the multi-site data analysis (IPCC, 2014: Fig. 3A.2 for CO_2, Fig. 3A.4 for CH_4). Therefore, the parameterization of the CO_2 response function was constrained to an asymptotic value (CO_2-C_{min}). In the current German GHG inventory, CO_2 and CH_4 response functions were only applied for reporting emissions from drained organic soils (WT < −0.1 m). As long as undrained and rewetted soils cannot be distinguished, emissions from wet organic soils are assumed to be zero. Nonetheless, we have derived emission factors for rewetted organic soils in Germany by applying the CO_2 response functions for the water table range from −0.1 m to, in analogy with IPCC (2014), 0.2 m.

Nitrous oxide emissions did not show a clear response to any available driver. Therefore, the mean site averages of the N_2O measurements of each land-use category were used. As in the Wetlands Supplement, uncertainties were given by the 2.5 and 97.5 percentiles of the data. For rewetted organic soils, mean annual fluxes of all sites within the WT range of −0.1 to 0.2 m have been averaged.

2.5. Reporting: national emission factors for organic soils

IPCC default EFs (IPCC, 2014) were used as estimates for GHG sources which were assumed to be non-significant and for which German data was missing, i.e. downstream CO_2 emissions from DOC leaching (CO_2-C_{DOC}) and CH_4 emissions from drainage ditches ($CH_{4\ ditch}$) (Fig. 1). As there is no CO_2 data from forests in our data set (Table 1), we used the general CO_2 response function (Eq. (3)) describing CO_2 losses from the soil for forest land, too. Carbon uptake by trees is reported under the biomass, litter and dead wood carbon pool in the inventory. Settlements were treated as drained grassland.

A detailed Tier 3 approach is resource intensive and will only be reasonable if the anthropogenic activity data show a high temporal dynamics and are available at high detail and frequency, e.g. annually. As WT data is not available as time series, the Tier 3 approach is simplified to spatially representative Tier 2 EFs. National EFs for drained organic soils CO_2-C_{onsite} and $CH_{4\ land}$ were derived by applying the response functions for CO_2 and CH_4 to the WT at the grid points with a WT < −0.1 m of each land-use category and aggregating them to spatially representative EFs (Fig. 1, Eqs. (5) and (6)). For peat extraction areas mean values were used.

For wet organic soils, the same approach was used for grid points with a WT range of −0.1 to 0.2 m and, in case of CH_4, a response function with different coefficients (Eqs. (7) and (8)). Here, the WT distribution of the WT map was used to estimate the distribution of water tables within wet sites, but as we currently cannot distinguish between naturally wet and rewetted areas, these sites are currently not included in the emission inventory.

$$CO_2\text{-}C_{\text{onsite, LUcat}} = \frac{1}{A_{drained,LUcat}} \sum_{i=1}^{n_{grid,LUcat}} (CO_2\text{-}C_{min} + CO_2\text{-}C_{max} e^{-ae^{bWT_i}}) * A_{gridpoint} \tag{5}$$

$$CH_{4land,LUcat} = \frac{1}{A_{drained,LUcat}} \sum_{i=1}^{n_{grid,LUcat}} (CH_{4min,LUcat} + c_{LUcat} e^{-d_{LUcat} WT_i}) * A_{gridpoint} \tag{6}$$

With

- $n_{grid,LUcat}$:
  number of grid points for each drained land-use category
- $A_{drained,\ LUcat}$:
  drained area of each land-use category (ha)
- $A_{gridpoint}$:
  Area of each grid point (6.4 ha, see Section 2.2)

$$CO_2\text{-}C_{\text{onsite, rewetted}} = \frac{1}{A_{wet}} \sum_{i=1}^{n_{grid,wet}} (CO_2\text{-}C_{min} + CO_2\text{-}C_{max} e^{-ae^{bWT_i}}) * A_{gridpoint} \tag{7}$$

$$CH_{4soil} = \frac{1}{A_{wet}} \sum_{i=1}^{n_{grid,wet}} (CH_{4min,wet} + c_{wet} e^{-d_{wet} WT_i}) * A_{gridpoint} \tag{8}$$

With

- $n_{grid,wet}$:
  number of grid points with a WT range of −0.1 to 0.2 m
- A_{wet}:
  area of wet organic soils (ha)

Uncertainties of the EFs were estimated by the 2.5 and 97.5 percentiles of all grid values and reflect the variability of emissions caused by the WT distribution and its uncertainty of each land use class. Then, the EFs were combined with the IPCC defaults for CO_2-C_{DOC} and $CH_{4\ ditch}$ (Eqs. (1) and (2)). The wet land-use area with zero reportable emissions ($frac_{wet,\ LUcat}$) was fixed for each land-use category to the

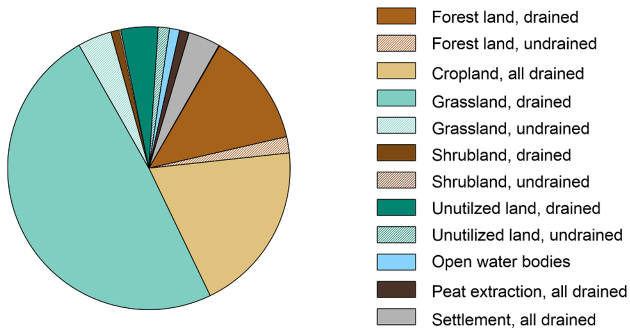

Fig. 2. Fractions of organic soils in Germany by land-use category (ATKIS®-Basic DLM 2014), split into "drained" (water table < −0.1 m) and "undrained" areas (water table ≥ −0.1 m).

percentage derived from the WT map (Fig. 2). Any drainage and rewetting activities from 2010 (WT estimate of Bechtold et al., 2014) onwards could be reported when new WT data or activity data will become available. The national implied EFs were calculated as weighted mean of the emissions from the drained and the wet land-use fraction without any anthropogenic GHG emissions (Fig. 1). To calculate total emissions for each land use category, the implied emission factors are multiplied with the total area [ha] of each land use category.

The GHG inventory tracks land-use changes on organic soils. Such land-use changes usually occur on drained land and are reported by a switch of the GHG emission EF from the old to the new land-use category. Accordingly, the land-use remains constant on undrained organic soils, but varies on drained organic soils. Therefore, the resulting time-series of GHG emissions from organic soils is derived from the time-series of land-use, while the WT distribution within each land-use category is assumed to be constant.

2.6. Verification

In the context of national emission inventories, "verification" means e.g. quality checks, independent reviews or the comparison against other data or other inventories. Verification is mandatory for national approaches. The national EFs were compared with the IPCC default values (IPCC, 2014) and with national approaches in The Netherlands (Arets et al., 2018), Denmark (Elsgaard et al., 2012; Nielsen et al., 2016) and the UK (Evans et al., 2017). The methodology was checked by reviewers of the UNFCCC Secretariat during the In Country Review of the German National Inventory under the UNFCCC and the Kyoto Protocol in September 2016 and was principally accepted (FCCC, 2017).

3. Results

We present the activity data, emission factors, methodology and the anthropogenic GHG emissions from drained organic soils in Germany for the land-use given by the ATKIS®-Basic DLM for 2014. Results are presented along the logical chain of Monitoring – Reporting – Verification.

3.1. Activity data

Organic soils as defined for the German GHG inventory cover 1.82 Mio ha, equivalent to 5.1% of Germanýs land area. This area differs slightly from Roßkopf et al. (2015) due to partially updated data sources. Dominant land-use categories are grassland (53% or 9691 km^2), cropland (20% or 3567 km^2), forests and shrubland (16% or 2927 km^2), while only 6% (1013 km^2) are classified as unutilized organic soils (Fig. 2). Most of the organic soils are drained for agriculture and forestry (Fig. 2). Drained organic soils dominate in all land-use categories except, of course, water bodies. Even the majority of the unutilized organic soils are drained (Figs. 2, 3). Only around 150,000 ha or 8% of organic soils are estimated to be undrained (WT ≥ −0.1 m).

Each land-use category encompasses a wide range of WT (Fig. 3). The fraction of undrained organic soil is largest in unutilized land (23% or 236 km^2), followed by forest and shrubland (12% or 362 km^2), and grassland (7% or 709 km^2). All croplands and settlements are drained (Figs. 2 and 3). When including undrained areas, mean water table depths decrease in the order unutilized land (−0.28 m), forest and shrubland (-0.38 m), grassland (−0.48 m), and cropland (−0.60 m) (Fig. 3). Mean WT excluding undrained land were −0.38 m (unutilized land), −0.43 m (forest and shrubland), −0.51 m (grassland) and −0.60 m (cropland). The relatively large share of very dry organic soils in most land use categories (Fig. 3) is caused by an artefact due to WT data transformation and by the uncertainty propagation of the WT map (methodological details see Bechtold et al., 2014). Due to the asymptotic shape of the response functions (Eqs. (3) and (4)) in dry conditions, this does not affect the results. Similarly, the relatively high percentage of wet organic soils might partially be a consequence of the error propagation.

Ditches covered 1.3% of the organic soil area (237 km^2). This value of $frac_{ditch}$ was applied to all land-use categories, as ditch spacing and width depend more on peat properties and terrain than on land-use. The German value for $frac_{ditch}$ is lower than both the indicative values given by IPCC (2014) and the Danish GHG Inventory (Nielsen et al., 2016) of 2.5 to 5%. As pipe drainage is common in Germany, a lower value of $frac_{ditch}$ is plausible.

3.2. GHG emission response functions and national emission factors

3.2.1. CO_2 emissions from drained and rewetted organic soils (CO_2-C_{onsite})

CO_2-C_{onsite} emissions increased steeply with deeper mean annual WT and level out at a WT of around −0.40 m where additional drainage would, on average, not further increase CO_2-C_{onsite} emissions (Fig. 4). There are, however, peatlands in our data set where further drainage up to nearly −1.0 m does strongly increase CO_2-C emissions (Tiemeyer et al., 2016). Under shallow drainage, CO_2-C_{onsite} emissions

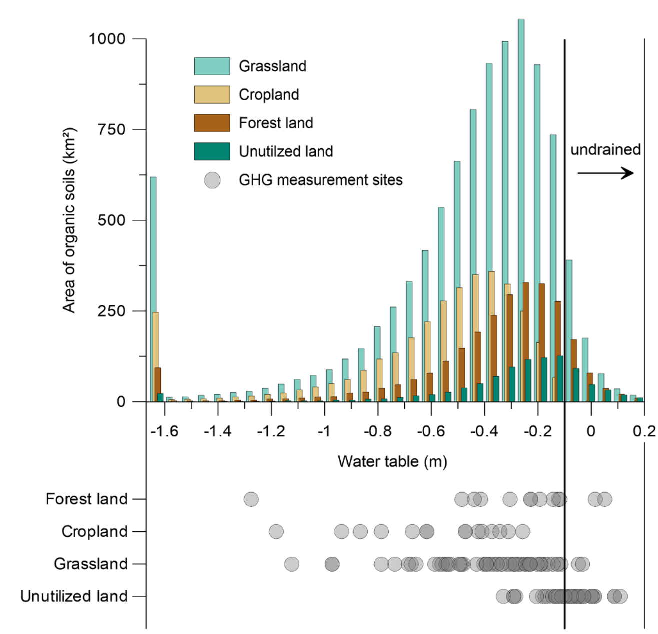

Fig. 3. Frequency distribution of mean annual water table (WT) of the major land-use categories in Germany (WT data from Bechtold et al., 2014) and mean annual water table of the GHG measurement sites with complete GHG budgets (forest land: sites with methane and nitrous oxide measurements).

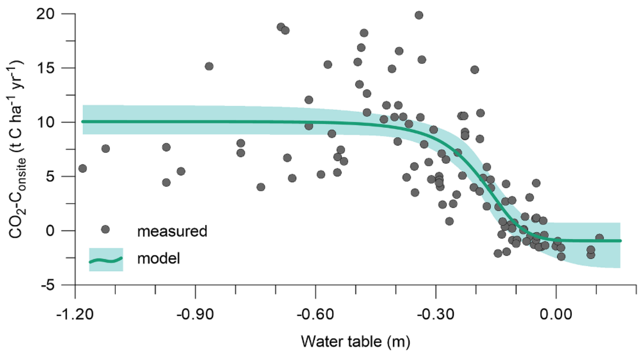

Fig. 4. Response of on-site CO_2-C emissions from organic soils to mean annual water table and coefficients of the fitted Gompertz function (Eq. (3)) with CO_2-C_{min} = −0.93 t C ha^{-1} yr^{-1}, CO_2-C_{diff} = 11.00 t C ha^{-1} yr^{-1}, a = 7.52 and b = 12.97 m^{-1}.

increased almost linearly with deeper WT. Overall, CO_2-C_{onsite} emissions showed a large scatter which can partially be explained by site conditions such as nutrient status and intra-annual WT dynamics, while CO_2-C emissions within single study areas clearly depended on WT (Leiber-Sauheitl et al., 2014; Tiemeyer et al., 2016). As there were no clear differences between the responses to WT in different land-use classes, all CO_2 data were lumped to derive one generic response function (Fig. 4). The relationship is statistically very robust despite substantial scatter in the underlying data.

CO_2-C emissions from wet (WT ≥ −0.1 m) organic soils were generally close to zero. Applying Eq. (7) resulted in an EF of −0.42 t CO_2-C ha^{-1} yr^{-1} for wet organic soils (Table 3). Taking into account the uncertainty ranges, this agrees reasonably well with the mean of the CO_2 flux data from sites within the same WT range (−0.38 t CO_2-C ha^{-1} yr^{-1}) and with the value of the response function at WT = 0 (−0.83 t CO_2-C ha^{-1} yr^{-1}).

The EFs were derived as the mean CO_2-C_{onsite} emission with the 95% percentiles from the frequency distribution of the grid points in the resulting CO_2-C_{onsite} emission map (Table 2). Thus, the German EFs generally show a wide range as this represents the spatial heterogeneity of the WT within each land use class.

3.2.2. CH_4 emissions from drained and rewetted organic soils ($CH_{4\ land}$)

As expected, annual methane fluxes range around zero or are small sinks in German deep drained organic soils (Fig. 5-c; Tiemeyer et al., 2016 for grassland). Starting at a WT of around −0.2 m, $CH_{4\ land}$ emissions generally increase either linearly (Levy et al., 2012) or exponentially (Drösler, 2005, Turetsky et al., 2014) with WT. This is also the case in this study (Fig. 5a). However, a high WT is only a necessary but not a sufficient condition for high emissions as low CH_4 fluxes also occur under wet conditions (Fig. 5c).

Table 3
Coefficients of the $CH_{4\ land}$ response functions.

Land-use category	$CH_{4\ min}$ (kg CH_4 ha^{-1} yr^{-1})	c (–)	d (m^{-1})
Forest land	−2.9	2260	−31.3
Cropland, grassland, settlement	3.5	17,055	−42.3
Drained unutilized land, rewetted organic soils	1.3*	292	−5.6

* Fixed at the mean value of all measurements with WT < −0.3 m.

Exponential response functions of $CH_{4\ land}$ to WT were fitted individually to the measured data from a) forest land, b) cropland and grassland, c) unutilized wet organic soils. The bootstrapped coefficients of Eq. (5) were significantly different from each other ($p < 0.01$) (Table 3). As the response functions for drained organic soils were fitted to data with a WT of −0.1 m or lower, they may not be extrapolated to sites with higher WT as the exponential function may produce unrealistically high values.

Methane emissions from fully saturated, rewetted organic soils could also be reported by using Eq. (5). However, we emphasise that this function should not be extrapolated as there is considerable uncertainty of the response function in the very wet range, especially when the mean WT is above ground surface (Fig. 5c, Wilson et al., 2016). Its applicability is thus restricted to organic soils sites with typical peatland vegetation and water table dynamics, but not to ditches or flooded land.

The national EFs for $CH_{4\ land}$ were derived analogously to the EF for CO_2-C_{onsite} by applying the response functions to the WT map of Bechtold et al. (2014). The EFs were derived as the mean $CH_{4\ land}$ emission with the 2.5 and 97.5% percentiles from the frequency distribution of the grid points in the resulting $CH_{4\ land}$ emission map (Table 2).

3.2.3. Direct N_2O emissions from drained and rewetted organic soils

As we could not find any robust functional relations between N_2O and nationally available drivers, the EFs were derived from the means of site average (measured data) by LU category (Table 2, Fig. 6).

3.2.4. Aggregated GHG emission factors for organic soils (implied emission factors)

In addition to CO_2-C_{onsite}, $CH_{4\ land}$ and direct N_2O-N, final national EFs consider all other anthropogenic GHG sources and sinks as shown in Eqs. (1)–(3); particularly CO_2 emissions from the export of dissolved organic carbon (drained: CO_2-C_{DOC} = 0.31 t C ha^{-1} yr^{-1}, rewetted: CO_2-C_{DOC} = 0.24 t C ha^{-1} yr^{-1}, IPCC, 2014) and CH_4 emissions from ditches (Table 4).

The so-called implied emission factors (IEFs) shown in Table 5 refer to all organic soils in each land-use category including eventual undrained fractions without any anthropogenic GHG emissions. They can

Table 2
German average emission factors and their 95% percentiles (values in brackets) for drained organic soils with a mean annual water table lower than −0.1 m below surface and for rewetted organic soils. CO_2-C_{onsite} and $CH_{4\ land}$ are derived from a Tier 3 methodology and compared with the IPCC default (Tier 1) emission factors for the temperate climate zone from the IPCC 2013 Wetlands Supplement (IPCC, 2014).

Land-use category	$CO_{2\ onsite}$ (t C ha^{-1} yr^{-1})		$CH_{4\ land}$ (kg CH_4 ha^{-1} yr^{-1})		direct N_2O (kg N_2O-N ha^{-1} yr^{-1})	
	German EF	Default EF (Tables 2.1 and 3.1)	German EF	Default EF (Tables 2.3 and 3.3)	German EF	Default EF (Table 2.5)
Forest land	7.7 (1.0–10.9)	2.6 (2.0–3.3)	4.0 (−12.4–45.7)	2.5 (−0.6–5.7)	2.0 (0.1–8.3)	2.8 (−0.57–6.1)
Cropland	9.2 (3.8–11.2)	7.9 (6.5–9.4)	5.5 (0.5–17.9)	0 (−2.8–2.8)	11.1 (1.8–40.5)	13 (8.2–18)
Grassland, settlement	8.3 (1.4–11.0)	3.6–6.1 (1.8–7.3)*	11.2 (0.6–86.4)	1.8–39 (−2.8–81)*	4.6 (0.3–22.2)	1.6–8.2 (0.56–11)*
Drained unutilized land	7.1 (0.7–10.8)	No EF	70.2 (1.3–184)	No EF	0.7 (−0.1–2.9)	No EF
Peat extraction**	1.3 (1.2–1.4)	2.8 (1.1–4.2)	4.2 (−0.4–13.1)	6.1 (1.6–11)	0.9 (0.3–1.4)	0.3 (0–0.6)
Rewetted organic soils	−0.4 (−2.4–1.3)	−0.23–2.5 (−0.71–1.71)**	279 (140–700)	123–288 (0–1141)***	0.1 (−0.5 to 1.0)	0

* values represent the range of (1) nutrient-poor, (2) shallow-drained, nutrient-rich and (3) deep-drained, nutrient rich-grasslands.
** without extracted peat, on-site emissions from peat deposits only
*** values represent the range of nutrient-poor and nutrient-rich rewetted organic soils.

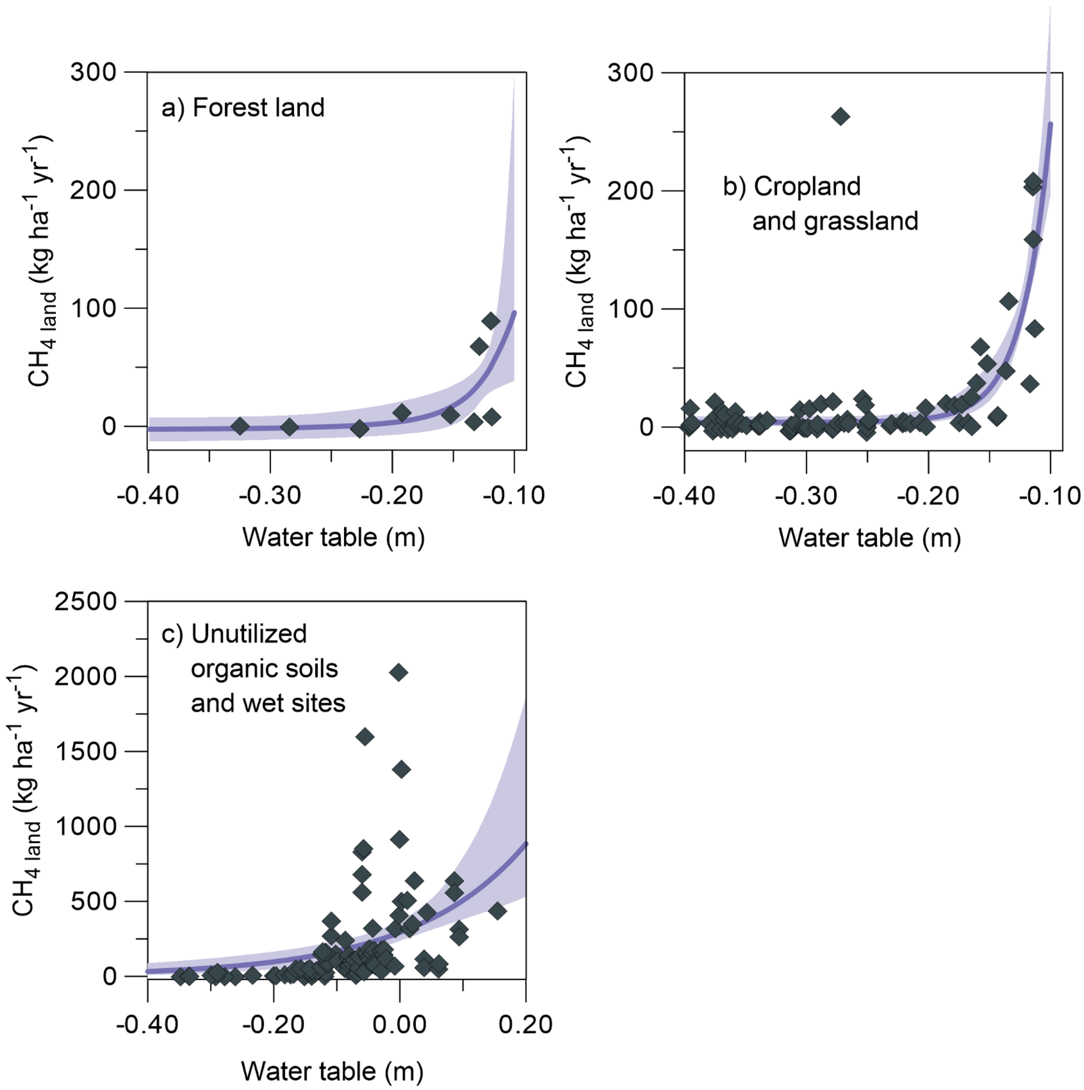

Fig. 5. Response of methane emissions from organic soils ($CH_{4\ land}$) to mean annual water table (WT) for a) forest land, b) cropland and grassland, c) unutilized organic soils and wet, i.e. semi-natural and rewetted sites. The grey bands show the bootstrapped 95% confidence intervals. Flux data with a WT < −0.40 m is not shown to improve clarity.

thus be multiplied with the total area of each land use category to derive total emissions. CO_2-C emissions from peat extraction areas exclude the peat extracted for horticulture.

3.3. Anthropogenic GHG emissions from drained organic soils in Germany

Fig. 7 shows the GHG emissions from German organic soils by land-use category and GHG including peat extracted for horticulture ($CO_{2\ peat\ offsite}$), which has been derived from the production statistics. The national emissions reflect the spatial extent of the land-use categories, i.e. 51% of the total emissions originate from grassland (30.5 Mio t CO_{2eq}), 24% from cropland (14.4 Mio t CO_{2eq}) and 13% from forest land (7.8 Mio t CO_{2eq}), while the other categories play a minor role. CO_2 accounts for most of the emissions (91% or 54.2 Mio t CO_{2eq}) and is thus by far the most important GHG. Mitigation measures should therefore primarily target CO_2. 93% of the CO_2 emissions could be attributed directly to the soil (CO_2-C_{onsite}), while DOC (CO_2-C_{DOC}) and peat extraction (CO_2-$C_{peat\ offsite}$) each account for 3 and 4%, respectively.

Nitrous oxide emissions account for 7% of the total emissions (4.2 Mio t CO_{2eq}) and are distinctly higher for cropland than for grassland. The contribution of CH_4 emissions is 2% of the total emissions (0.9 Mio t CO_{2eq}). More than half (55%) of the CH_4 emissions are estimated to originate from ditches, which cover only 1.3% of the land area. This is especially noticeable for cropland and peat extraction areas, where 73% and 63% of the CH_4 emissions originated from ditches.

Overall, drained organic soils annually emitted 59.3 Mio t CO_{2eq} which corresponded to 6.6% of the national GHG emissions in 2014 (including the net CO_2 sink in the LULUCF sector). The value is 26% above the 47.2 Mio t CO_{2eq} reported in the national inventory 2016 (UBA, 2016). One reason is the higher EF for forest and shrubland introduced here, and the second one the use of recent GHG data which was not yet available when first implementing the method in 2016. Thus, drained organic soils remain a significant GHG source. If other sectors continue to reduce GHG emissions, the relative importance of GHG emissions from organic soils will increase in the future. The time series of GHG emissions from organic soils is relatively stable as remaining unutilized and semi-natural areas are protected from more intensive use and as rewetting is not yet taking place at large scale and not yet considered outside land-use changes. Therefore, the GHG emission time series from organic soils is currently driven by land-use changes alone.

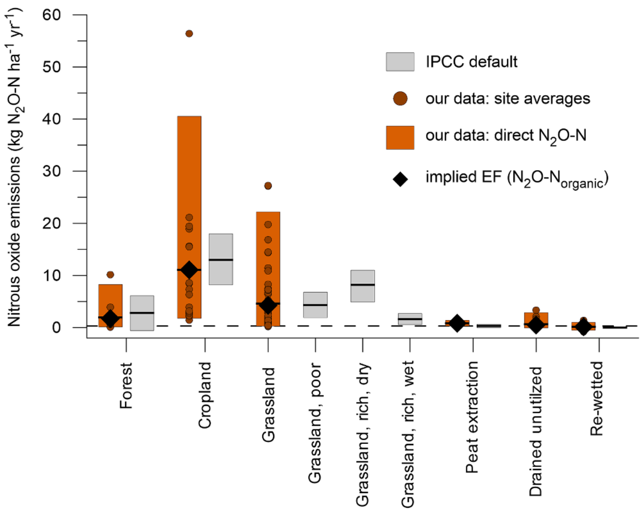

Fig. 6. Average annual N_2O budgets and implied national emission factors (EF) for the land-use categories in the German GHG inventory in comparison to the respective IPCC EFs with 95% percentiles. The IPCC default value for rewetted organic soils is zero.

4. Discussion

4.1. Verification of emission factors

4.1.1. Carbon dioxide

Verified against the IPCC default EFs (Table 2), the German CO_2 EFs for drained organic soils are similar for cropland, higher for forest land and grassland, and lower for peat extraction areas. While some of the data used to derive the German EFs have also been used to derive the IPCC default values, these data have largely been recalculated or amended by longer measurements at the same sites. The divergence from the defaults can be explained by a high drainage and land-use intensity in grasslands (Bechtold et al., 2014, Tiemeyer et al. 2016, Untenecker et al., 2017). The German forest EF agrees reasonably well with the long-term estimate 5.0 t CO_2-C ha^{-1} yr^{-1} for a nutrient-poor drained spruce stand by Hommeltenberg et al. (2014). In contrast, IPCC used comparatively wet and cold forest sites to derive the forest land default values (Section 4.3.2). In contrast to some other countries, peat extraction is restricted to nutrient-poor peat in Germany. Additionally, Wilson et al. (2015) derived an EF of 1.70 (± 0.47) t CO_2-C ha^{-1} yr^{-1} for industrial peat cutting in Ireland and the UK, which is much closer to our values than to the IPCC default value. Wilson et al. (2016) propose CO_2-C emission factors of −0.23 and 0.50 t CO_2-C ha^{-1} yr^{-1} for re-wetted nutrient-poor and nutrient-rich sites in the temperate zone. This is slightly less optimistic than our value of −0.4 t CO_2-C ha^{-1} yr^{-1}, but could be explained by the fact that we used an optimum WT range (> −0.1 m) for the derivation of our EF.

Based on the 8 sites measured by Elsgaard et al. (2012), the Danish National Inventory Report (Nielsen et al., 2016) applies emission factors of 11.5 ± 2.0 and 8.4 ± 1.0 t CO_2-C ha^{-1} yr^{-1} for cropland and grassland,

Table 5
Implied national emission factors (IEF) for CO_2-C$_{organic}$, $CH_{4\ organic}$, N_2O-N$_{organic}$ and total greenhouse gas emissions using global warming potentials (GWP) as given in IPCC AR4 (Forster et al., 2007). The IEFs are representative of the total area of each land-use category including undrained organic soil (reported as zero) and - for CH_4 - the ditch area.

Land use category	CO_2-C$_{organic}$ (t C ha^{-1} yr^{-1})	$CH_{4\ organic}$ (kg CH_4 ha^{-1} yr^{-1})	N_2O-N$_{organic}$ (kg N ha^{-1} yr^{-1})	GHG (t CO_{2eq} ha^{-1} yr^{-1})
Forest land	7.0	6.0	1.7	26.6
Cropland	9.5	20.6	11.1	40.4
Grassland	8.0	21.7	4.2	31.7
Drained unutilized land	5.7	55.3	0.5	22.5
Peat extraction*	1.6	11.2	0.9	6.5
Settlement	8.6	23.4	4.6	34.2
Rewetted organic soils	−0.4	279	0.1	5.5

* without extracted peat, emissions from peat deposits only.

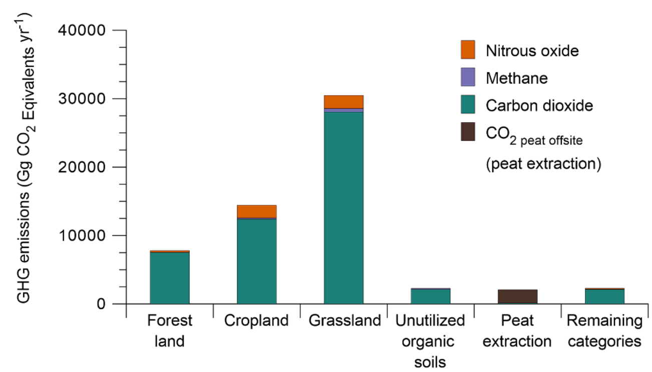

Fig. 7. Greenhouse gas (GHG) emissions from German organic soils by land-use category and GHG including peat extracted for horticulture ($CO_{2\ peat\ offsite}$) based on the land-use in 2014.

respectively, which agrees well with our EFs. However, they assume organic soils with a low content of soil organic carbon (6 to 12%) to emit only 50% of these values. This contradicts measurements at German agricultural sites where no difference between such soils and “true peat” (soil organic matter content > 30% according to the German classification system) could be found (Eickenscheidt et al., 2015; Leiber-Sauheitl et al., 2014; Tiemeyer et al., 2016). The EFs for grassland in the NIR of The Netherlands are based on subsidence measurements and are stratified according to drainage and nutrient status as well as substrate of the upper soil layer (Arets et al., 2018). Discounting clay-covered organic soils, values range from 4.9 to 7.2 t CO_2-C ha^{-1} yr^{-1}, which is slightly less than our values. Finally, UK emission factors for grassland were again lower than our values (3.6 and 6.4 t CO_2-C ha^{-1} yr^{-1} for nutrient-poor and nutrient-rich sites, respectively), which can again be explained by the relatively low land-use intensity especially of UK uplands (Evans et al., 2017). The cropland EF (7.2 t CO_2-C ha^{-1} yr^{-1}), on the other hand, is closer to our value of 9.2 t CO_2-C ha^{-1} yr^{-1}.

Table 4
Emission factors for emissions from ditches ($CH_{4\ ditch}$) used in German GHG inventory.

Land-use category	$CH_{4\ ditch}$ (kg CH_4 ha^{-1} yr^{-1})	Comment
Forest land	217	IPCC, 2014
Cropland	1165	IPCC, 2014
Grassland, settlement	948	Weighted average of shallow and deep drained grassland (IPCC, 2014) with 34% shallow drained grassland (Bechtold et al., 2014)
Drained unutilized land	217	as Forest land (no fertilization)
Peat extraction	542	IPCC, 2014
Rewetted organic soils	–	Refilled and blocked ditches assumed to be part of the landscape mosaic (IPCC, 2014)

4.1.2. Methane

Verified against the IPCC default EFs (Table 2), the German EFs for drained organic soils turn out relatively high for cropland. This is justified by occasional measurements of high CH_4 fluxes when strong rainfall led to waterlogged conditions on the soil surface and, in some cases, partial to complete dieback of the vegetation. In contrast, CH_4 emissions from grasslands and peat extraction tend towards the low end of the IPCC defaults (Table 2). The divergence from the defaults could also be explained by strong drainage intensity in grasslands and peat extraction sites. Our CH_4 EF for rewetted organic soil is well within the range of the IPCC default values.

4.1.3. Nitrous oxide

Verified against the IPCC default EFs (Fig. 6), the German EFs for drained organic soils turn out comparable to the IPCC default EFs but with a larger uncertainty range, particularly for cropland. Cropland EF derived for the UK inventory are clearly higher than our values (19.1 kg N_2O-N yr^{-1}, Evans et al., 2017). As there was no significant difference in the German EFs between the IPCC grassland types, a common EF is used for all grasslands. Given the high partially high land-use intensity of German grasslands on organic soils, the mean EF of 4.6 kg N_2O-N yr^{-1} is within the IPCC default range, but still somewhat surprising and will probably need some consideration in future (Section 4.3.2). Measured N_2O data support the IPCC default approach that direct N_2O emissions from fully saturated, rewetted organic soils could be reported as approximately zero (IPCC, 2014), although there is some variation around this value, probably due to atmospheric input or previous fertilization of the sites.

4.2. Importance of representativeness in methodology and data

The basis for developing our MRV system for organic soils is a unique, entirely harmonized nationally coordinated set of measurement data produced in two national joint projects and adjacent activities. This was amended by appropriate literature data (Leppelt et al., 2014; Poyda et al., 2016, Buchen et al., 2017). The measurement data set covers all important regions with organic soils and the most important land-use categories and GHG sources. Relatively extreme cases of management such as, for example very wet low intensity grassland systems or high intensity grassland systems with up to five cuts per year, or years with extreme weather conditions are also considered in the data set. This allowed for the first time to develop robust GHG response functions applicable from local to national level (Tier 3 methodology).

In parallel, substantial effort has been invested to improve the national activity data for organic soils, including a new high resolution map of organic soils in line with the IPCC definition of organic soils (Roßkopf et al., 2015) and a high resolution WT map (Bechtold et al., 2014). However, the improvement of activity data is lacking behind the progress in point-scale measurements (Section 4.4).

We could show that spatial resolution of soil maps can substantially bias land-use distribution on organic soils. In the past, Germany used a soil map at a scale of 1:1,000,000 (BGR, 2007) in its national GHG inventory to delineate organic from mineral soils. The total organic soil area amounted to 1.73 million ha, which is relatively close to the recent estimate of 1.82 million ha. The substantially improved resolution of 1:25,000, however, had drastic implications for the spatial position of the organic soils: new small areas of organic soils were 'detected' while the boundaries of larger peatlands were corrected. This had a major effect on the land-use distribution on organic soils: According to the coarse soil map (BGR, 2007), 37 and 34% of the organic soils are used as cropland and grassland, respectively, while the new map assigns only 20% to cropland, but 53% to grassland (Fig. 2). This does not have a large impact on the total GHG emissions, but on targeting appropriate mitigation measures.

Furthermore, we aimed at avoiding bias by using spatially weighted EFs. While we covered the whole range of WT within our flux data set (Fig. 3), mean and distribution of WT still differ between the national level and the measurement sites. Therefore, we used response functions to scale observations to nationally representative IEFs. If only the mean of the flux measurements had been used as it was the case for development of IPCC default EFs, this would have introduced only a small bias in the total CO_2 emissions when not considering forests (−3%), but a large bias in the CH_4 emissions (69%). The comparatively small bias in the CO_2 emission estimates can be explained by the relatively deep WT in most of the drained peatlands and "early" levelling out of the response function: For WTs of around −0.40 to −0.50 m and deeper, the calculated emissions reach the asymptotic value of Eq. (3) (Fig. 4), which represents the mean value of the measurement sites with such water tables. The bias in the CO_2 emissions would, however, have been large in the case of unutilized organic soils (−68%). This would hamper evaluating the effects of land-use changes, which currently drive the emission time-series due to the static representation of the WT. In the case of CH_4, spatially representative values deviated strongly from site means in the land use categories forest land and grassland, where the bias would have been very large (+101% and +107%, respectively) due to the strong non-linearity of the CH_4 response function. Neglecting spatial WT patterns would have a strong effect on evaluating mitigation measures and policies.

4.3. Towards reducing uncertainties

4.3.1. Response functions and differentiated emission factors for N_2O

While the CH_4 flux data from grassland, cropland, forest land and unutilized organic soils could be very well described with our response functions (Fig. 5a and b), there is substantial scatter in the CO_2 flux data (Fig. 4) and uncertainty in the response function for CH_4 emissions from wet organic soils (Fig. 5c).

For grassland sites, it could be shown that taking into account both soil properties and the intra-annual WT distribution substantially improves the prediction of CO_2 emissions (Tiemeyer et al., 2016). In particular, the aerated nitrogen (N) stock, i.e. the stock of N which is exposed to oxygen at any time of a year, has turned out to be an important parameter. Unfortunately, the idealized soil profiles linked to the map of organic soils do not contain any information on N concentrations or stocks (Roßkopf et al., 2015). Therefore, the development and application of more sophisticated response functions would require an update of these idealized soil profiles. Further, under very dry conditions, there might be lower emissions due to either moisture limitation or very shallow organic layers (Fig. 4), but accounting for such effects would also need parametrized soil maps. Stratifying the data by nutrient-poor and nutrient-rich peat only as done by IPCC (2014) did not reduce the scatter in the CO_2 response function although emissions from grassland on bog peat tended to be lower than from nutrient-rich fen peat, but intensively used bog grasslands are strongly underrepresented in our data set (Tiemeyer et al., 2016). However, due to their high bulk density and, accordingly, high aerated N stocks, "low carbon organic soils" emit as much CO_2 as "true" peat soils (Tiemeyer et al., 2016). Therefore, approaches such as reporting only 50% of the CO_2 emissions of true peat soils as done in the Danish NIR (Nielsen et al., 2016) are not supported by the German observations.

In the case of CH_4, WT became an uncertain predictor at very wet sites. This has also been observed when deriving the IPCC emission factor (Wilson et al., 2016) although the spread of the data is less obvious if presented with a logarithmic scale. The large scatter (Fig. 5c) might be explained by the nutrient status (Wilson et al., 2016), land-use history (e.g. peat extraction vs. nutrient-rich topsoil; Harpenslager et al., 2015), dieback of non-adapted vegetation (Tiemeyer et al., 2016), the influence of aerenchymous plants (Levy et al., 2012) or the combination of WT and leaf area of aerenchymous plants (Drösler, 2005). However, the data set is not yet large enough to derive stratified response functions or robust vegetation-based approaches due to the large variability especially at fen peat sites.

Similarly as in the case of CO_2, the N content (Tiemeyer et al., 2016), the N fertilization (Leppelt et al., 2014; Tiemeyer et al., 2016), and the pH-value (Leppelt et al., 2014) are likely to control N_2O emissions to a certain extent. Therefore, the development of stratified N_2O EFs requires parametrized soil maps. Finally, all N_2O data originates from chamber measurements with a comparatively low temporal resolution. Using automatic chambers (Brümmer et al., 2017) or the eddy covariance approach (Shurpali et al., 2016) could shed light on the temporal dynamics of N_2O emissions and even enable the development of appropriate models.

4.3.2. Carbon budget of forests on organic soils

We applied the general response function for CO_2 also to forest land, although none of the data points originates from a forested site. The resulting national EF of forest land is 7.7 t CO_2-C ha^{-1} yr^{-1} und thus significantly higher compared to the IPCC (2014) default EF of 2.6 t CO_2-C ha^{-1} yr^{-1} (Table 2).

While our approach certainly does have the shortcoming of missing actual measurements on forested sites, we are nonetheless convinced that the high emission factor is plausible for a number of reasons. First, the five studies used to derive the default EF (IPCC, 2014) reported a) colder mean annual air temperatures (sites range from −0.2 to 5.6 °C (von Arnold et al., 2005a,b; Minkkinen et al., 2007)) compared to German conditions (mean annual air temperature of 8.9 °C), and b) had higher mean WT (−0.15 m to −0.32 m (von Arnold et al., 2005a,b; Yamulki et al., 2013)) than German drained forests (−0.43 m according to the WT map, see Fig. 2). Second, some of the studied tree species such *Prunus* spp. (Glenn et al., 1993) are not common in Germany. Third, Hommeltenberg et al. (2014) estimated emissions of 5.0 t CO_2-C ha^{-1} yr^{-1} based on subsidence measurements for a Bavarian drained spruce forest (*Picea abies* L. Karst) on bog peat, which is in between our EF (7.7 t CO_2-C ha^{-1} yr^{-1}) and the IPCC default value (2.6 t CO_2-C ha^{-1} yr^{-1}). As the site of Hommeltenberg et al. (2014) is on bog peat and rather wet (−0.20 m), the lower value compared to the German EF is not surprising. Still, the assumption of high CO_2 emissions from forests on organic soils needs to be validated by measurements.

4.3.3. Importance of DOC losses and CH_4 emissions from ditches

Available data suggest that DOC losses of drained sites constitute only a minor part of the total carbon budget even at sites with high DOC concentrations (Frank et al., 2017, Tiemeyer and Kahle, 2014). For example Frank (2016) and Frank et al. (2017) report mean DOC losses of 430 kg ha^{-1} yr^{-1} at a deeply drained grassland on bog peat ("Ahlenmoor", Table S1), and of ~200 kg ha^{-1} yr^{-1} at a shallow drained grassland ("Großes Moor", Table S1), corresponding to 9% and 3% of the carbon budget of the respective site. In contrast to these relatively high values, Tiemeyer and Kahle (2014) measured DOC losses of 53 kg ha^{-1} yr^{-1} from a catchment with fen peat and other organic soils, which equals only around 1% of the carbon budget within this catchment ("Dummerstorf", Table S1). These numbers cover or even exceed the range of values (mean 340 kg ha^{-1} yr^{-1}, uncertainty range 210 to 510 kg ha^{-1} yr^{-1} before applying the factor $frac_{DOC\text{-}CO2}$ of 0.9 for the conversion of DOC to CO_2-C) suggested by IPCC (2014) for drained organic soils. Therefore, the IPCC default value may serve as a good first estimate for drained organic soils although the uncertainty at the scale of individual peatlands is likely very high. At wet sites with lower net ecosystem exchange (or even a slight uptake) DOC might become a more relevant component of the carbon budget as found in other studies (Evans et al., 2016). Frank (2016) measured average DOC losses of 120 kg ha^{-1} yr^{-1} from a bog rewetted after peat cutting ("Ahlenmoor", Table S1), which is clearly lower than the IPCC (2014) default value (mean 260 kg ha^{-1} yr^{-1}, uncertainty range 170 to 360 kg ha^{-1} yr^{-1}) for rewetted organic soils. To date, DOC data for drained and unutilized or rewetted sites is extremely sparse in Germany. Therefore, the IPCC default value has to be applied, but especially for rewetted sites and at the project scale, additional measurements might be useful.

Methane emissions from ditches constitute a "significant source" of the total CH_4 emissions, but data in Germany are still missing apart from those by Günther et al. (2017) for ditches in a Sphagnum cultivation site. According to IPCC (2014), ditches in rewetted peatlands are not reported separately, but interpreted as typical feature of the peatland's microtopography, and indeed the values of 48–144 kg CH_4 ha^{-1} yr^{-1} measured by Günther et al. (2017) are well within the uncertainty range of rewetted nutrient-poor peatlands (4–607 kg CH_4 ha^{-1} yr^{-1}, IPCC, 2014). Using IPCC (2014) default EFs (Table 4) and a ditch density of 0.013 m^2 m^{-2}, methane emission from ditches accounted for 55% of the total CH_4 emissions from drained organic soils, but they constituted only around 1% of the total GHG emissions. As the data base of the current EFs is rather sparse (Evans et al., 2016), additional measurements and more refined activity data on ditches could greatly improve emission estimates. There is substantial uncertainty in $frac_{ditch}$, not only due to the conversion of ditch classes of the ATKIS®-Basic DLM to ditch area, but also due to the uncertain ditch geometry and the (seasonally variable) ditch area actually covered by water. The default value of $frac_{ditch}$ was compared to results of an analysis of aerial photos around the GHG measurement sites: The average ditch density of 0.010 m^2 m^{-2} confirms that the order of magnitude in $frac_{ditch}$ seems to be adequate, keeping in mind that the ditch densities ranged from 0.002 to 0.029 m^2 m^{-2} may result in a variability of more than 100%.

4.4. Towards tracking drainage and rewetting

The EU LULUCF Decision (EC, 2013) implies that any changes (drainage, partial or full rewetting) in water table in organic soils on forest, cropland and grassland have to be reported and accounted for by EU Member States. The recent amendment (EC, 2018) of this regulation additionally requires "managed wetlands" to be accounted for from 2026 onwards. Emission factors for organic soils that were completely rewetted to near-natural hydrological conditions (as assumed by both the IPCC default values and our EF for rewetted organic soils, Tables 2 and 5) are now readily available. Due to the large number of measurement sites (Table 1), they might be even further stratified in future e.g. according to nutrient status, should suitable activity data be available at the scale of interest. The response functions presented in this paper could be applied for all other activities leading to deeper drainage and partial rewetting. Proxies such as ditch water levels could be used to estimate groundwater levels.

While considerable and extremely valuable effort has been put into improving EFs, the acquisition of suitable activity data has unfortunately received much less attention, especially considering the development of consistent time series. The WT map of Bechtold et al. (2014) is roughly representative of the situation in 2010, for which most dipwell data were available. This may include some rewetting since 1990 and is based on more wet than dry sites as monitoring activities generally concentrated on nature conservation or rewetting projects, but not on intensively managed and drained agricultural areas. While the WT map might have missed some deepened drainage since 1990, nature conservation legislation would not have allowed any new drainage of natural areas since before 1990. Overall, most drainage of German peatlands started decades to centuries ago, and was intensified after the Second World War (Succow and Joosten, 2001). However, deriving consistent time series of WT or even land-use of (German) organic soils is challenging for a number of reasons:

First, field drainage is not regulated by national or regional legislation in Germany. Water management is frequently in the hands of local associations or administrative units which might differ at the level of federal states, administrative regions or districts. Therefore, there is no central data archive of dipwell or ditch water table data, drainage maps or water management projects in organic soils for the whole country or for single federal states.

Second, rewetting projects have very diverse backgrounds, financing sources, levels of documentation and have mainly been conducted for nature conservation purposes so far (Belting and Freibauer, 2013). Generally, documentation is more comprehensive for biodiversity than for abiotic parameters except for some exemplary projects as e.g. Nature Park Drömling (Untenecker et al., 2016) or Schwäbisches Donaumoos (Mäck, 1998). Peatland conservation programmes of the federal states also comprise rewetting. While many sites have been target of rewetting measures of some kind, not all of them reach near-natural hydrological conditions. As in the case of drainage, there is no central database of rewetting projects at the national level. Generally, the highly diverse project and programme documentation does not comply with the IPCC quality criteria of consistency and completeness. However, good local or regional data is extremely valuable for the development and verification of large scale approaches (e.g. Untenecker et al., 2016, 2017).

Third, land-use data is only available from different data-sets with contrasting thematic and spatial resolution, which requires adequate methods to translate between different data sources (Untenecker et al., 2016). Historical tracking via land-use change may misestimate the changes in GHG emissions caused by drainage and rewetting, yet effects are likely to compensate each other to an unknown extent, even more so as land-use change does not always imply WT change (Untenecker et al., 2016). There is no national-scale biotope type or vegetation data (even less so as a time-series) which would enable the use of vegetation-based proxies.

Consequently, the most promising approach might be a combination of the use of airborne and spaceborne remote sensing data and a well-organised data base on rewetting projects. Remote sensing could aim at identifying vegetation classes or WT/soil moisture. As the most important land-use category "grassland" shows very variable emissions for the same vegetation class (Tiemeyer et al., 2016), the latter approach would be preferable. Still, the reconstruction of the moisture status in the current base year 1990 will remain challenging. In some federal states of Germany, existing classified colour infrared (CIR) data might be used, but extrapolation to other federal states is not advisable due to differences in political and agro-environmental trajectories (Untenecker et al., 2017). Although not yet complete, the attribute "wet soil" in the ATKIS®-Basic DLM which is available since 2008 might serve as a suitable conservative proxy of rewetting activities, but not of deepened drainage (Untenecker et al., 2016). In future, it will become even more important to both improve remote-sensing approaches (using e.g. Sentinel or other radar data, Bechtold et al., 2018) and to collect "ground-truthed" activity data on water management and WT data using the data base of Bechtold et al. (2014) as starting point. This should result in a nationally coordinated, representative dipwell monitoring to a) track drainage and rewetting, b) derive information on the *status quo* before any mitigation measure, c) account for land-based Kyoto activities, and d) serve as ground-truthing for remote sensing approaches. Ideally, this should be combined with the measurement of GHG emission proxies such as subsidence and the evaluation of policy measures.

4.5. Suitability for improving other national GHG inventories

The combination of considerable effort for improved activity data and a unique observational basis of GHG fluxes allow for a Tier 3 representation of Germany's largest GHG source from agriculture and land-use sectors as soon as time series of WT are available. Methodology and – under comparable climatic and agricultural conditions – EFs are generally transferable to other national GHG inventories.

Overall, the CO_2 EFs derived from the response function can partially deviate strongly from the IPCC (2014) default values: While the cropland EF is relatively close to the default value (16% higher), the grassland EF strongly exceeds the EFs for nutrient-poor grassland (56%), deep-drained nutrient-rich grassland (36%), and shallow-drained nutrient-rich grassland (130%). Similarly, the forest land EF was 195% higher than the default value (see Section 4.2.2). This demonstrates that the default IPCC EFs are inappropriate for Germany (even though some of the data were used for deriving the default values), and probably for other countries with similarly intensive drainage and comparable climatic and site conditions as well.

Methane and N_2O emission factors are closer to the default values (Table 2) and of minor importance for the national GHG balance of organic soils. However, CH_4 might gain in importance when scaling down to project level or when evaluating the effect of rewetting activities.

The developed methodology could be adopted by other countries by linking national activity data to our response function or using a similar approach to developing country specific response functions, possibly using data from our data set if conditions are comparable. GHG data might need to be harmonized, but probably much less effort might be needed for GHG fluxes than for activity data depending on data heterogeneity and availability of land management information. In a situation with a large share of drained organic soils, it might be advisable to focus on CO_2-C_{onsite} in the largest land-use categories as CO_2 emissions from soils are likely to be the most important GHG source. There might be exceptions, however, in situations where large peat fires occur frequently.

5. Conclusions

In this study, we presented the scientific background, data and methodology for a detailed GHG estimate for organic soils at national level based on spatially explicit modelling. This methodology is adequate for reporting a "key source" such as emissions from organic soils under UNFCCC and the Kyoto Protocol including land-based activities.

We found that CO_2 has by far the highest contribution to the total GHG emissions from organic soils in Germany (> 90%). Therefore, any mitigation measures should be targeted at reducing the CO_2 emissions which can be partially even higher than assumed so far by the IPCC default values. For inventory purposes, results have been aggregated to implied EFs ensuring Transparency, Consistency and Comparability. The methodology is also applicable at project level given that WT estimates are available. The general approach is suitable for other countries, but it might be necessary to adjust the response functions to national circumstances if an estimate of the WT is available. Under similar conditions with a large percentage of drained organic soils, monitoring should focus on CO_2. Improved and updated activity data from both remote sensing and a coordinated rewetting project data base is necessary to a) improve the inventory by using more sophisticated response functions or other spatially explicit models, b) adequately represent water management measures including rewetting, and c) evaluate policies and measures.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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