Estimation of greenhouse gas emission reductions based on vegetation changes after rewetting in Drentsche Aa brook valley

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SUMMARY

Rewetting can effectively reduce greenhouse gas (GHG) emissions from drained peatlands. Reliable emissions estimation approaches are needed for accounting of such reductions and for evaluating the potential in terms of carbon credits. Annual mean water level and vegetation are reliable and widely used proxies for emissions estimation. However, indications of water level based on plant species (e.g. Ellenberg Indicator Values) are qualitative with large variances, and there are insufficient high-quality flux measurement data to support the direct use of vegetation as a proxy for GHG fluxes. Here we combine vegetation and water level proxies to estimate emissions, by using bioindication of vegetation communities for water level together with the linear correlation between annual mean water level and GHG fluxes. This approach is demonstrated in the Drentsche Aa brook valley in The Netherlands, where peatlands were rewetted to restore rich fen vegetation. Biodiversity of the landscape was monitored by repeated vegetation mapping before and after rewetting, which enables the estimation of emissions reduction as a co-benefit. Mean annual water level values are assigned to mapped vegetation types using existing data on water level dynamics from measurements on corresponding plant communities. GHG emissions are estimated using linear regression models of gas fluxes against mean annual water levels. This approach provides spatially explicit and quantitative estimation of mean annual water levels and GHG fluxes. When combined with information on spatial patterns and variances, the resulting estimations can promote recognition of the carbon co-benefits of biodiversity restoration while facilitating more site-specific optimisation of management practices.

KEY WORDS: emissions estimation, GHG, groundwater level, spatial pattern, vegetation restoration

INTRODUCTION

Peatlands and greenhouse gas emissions

At global scale, peatland has the most important soil carbon stock, covering only 3 % of Earth's land surface and containing approximately 600 Gt of carbon, which is in the same order as the global carbon stock in vegetation (Yu *et al.* 2010, Yu 2012, Page & Baird 2016). Besides their role in global climate regulation, peatlands also provide various important ecosystem services such as fresh water and biomass provisioning, flood control, biodiversity and recreational opportunities (Whitfield *et al.* 2011, Reed *et al.* 2014). However, 10–20 % of all peatlands have been degraded due to human activities including agricultural and forestry drainage. This disturbance has transformed peatlands into a net greenhouse gas (GHG) source equivalent to approximately 0.1 Pg yr^{-1} of carbon in total (Frolking *et al.* 2011). Drainage strongly affects various environmental factors in peatlands (Landry & Rochefort 2012). Global CO_2 emissions from drained peatlands had increased by more than 20 % in 2008 (Joosten 2010), and drainage alone will lead to a release of 80.8 Pg carbon cumulatively, assuming a CO_2-eq emission rate of 1.91 Pg yr^{-1} (Leifeld & Menichetti 2018). In addition to hampering global emissions goals, this is causing significant environmental crises including soil subsidence (Hoogland *et al.* 2012) and biodiversity losses (Minayeva *et al.* 2017).

Rewetting of peatlands is one of the most successful measures for reducing GHG emissions (Couwenberg *et al.* 2011, Wilson *et al.* 2016b, Cui *et al.* 2017, Renou-Wilson *et al.* 2019). By re-establishing high water levels (e.g. by ditch blocking; Worrall *et al.* 2010), peat mineralisation could be reduced and peat-forming vegetation could start to recover at some locations (Kozlov *et al.* 2016, Guo *et al.* 2017). In most cases rewetting has converted peatlands back into CO_2 sinks, although they have remained net GHG sources because of increased CH_4 emissions. Even so, their Global Warming Potential (GWP) could still be significantly reduced (Vanselow-Algan *et al.* 2015, Wilson *et al.* 2016b, Lee *et al.* 2017, Kandel *et al.* 2019). Peatland restoration through rewetting is predicted to have a maximum (CO_2-eq) climate mitigation potential of 815 Tg ha^{-1} yr^{-1} (Griscom *et al.* 2017). Considering the extent of peatland at large scales (e.g. Europe, Tanneberger *et al.* 2017), rewetting of peatlands could greatly influence the regional voluntary carbon market by producing emissions reduction credits (carbon credits; Worrall *et al.* 2010, Bonn *et al.* 2014, Günther *et al.* 2018). However, the economic benefits would be neutralised by expensive and laborious GHG flux measurements (Günther *et al.* 2018). Thus, reliable estimation methods are needed for cheaper accounting of the emissions reductions achieved after rewetting.

The IPCC guidelines provide direct emission factors for peatlands which have been applied in large-scale accounting of emissions (Wijedasa *et al.* 2018). However, the original 2006 guidelines substantially under-estimated emissions from organic soils and employed inconsistent definitions (Couwenberg 2011). The 2013 supplement (IPCC 2013) provides guidance for accounting GHG emissions from wetlands and drained organic soils in which peatlands are categorised according to climate zone, peatland type and drainage conditions, and emission factors are assigned to these categories. Drainage conditions are divided into deep-drained and shallow-drained with mean water level 30 cm below surface as the threshold. Such categorisation can support GHG accounting at national and regional scales, but the high variation within categories (Tiemeyer *et al.* 2016) and calibration data gaps for specific countries/regions (Wilson *et al.* 2016a) make the IPCC inventory unsuitable as a basis for fine-scale estimations.

Vegetation as a proxy for GHG emissions

Proxies are also commonly used for emissions estimation. For example, mean annual water level is a strong predictor of annual GHG flux because the two factors are strongly correlated (Couwenberg *et al.* 2011, Couwenberg & Fritz 2012). Vegetation indicates a wide range of environmental factors that are strongly correlated to GHG emissions (Berglund & Berglund 2011, Bartelheimer & Poschlod 2016) and is directly involved in GHG flux processes (Couwenberg & Fritz 2012, Dunn *et al.* 2016, Strack *et al.* 2017). Therefore, the structure, composition and traits of vegetation can also serve as proxies for GHG flux estimation (e.g. Dias *et al.* 2010, Gray *et al.* 2013, Karu *et al.* 2014, Goud *et al.* 2017). For example, Audet *et al.* (2013) demonstrated reliable prediction of yearly CH_4 emissions from vegetation assemblages in riparian wetlands, using generalised linear models and weighted average regressions.

Currently, the most practical vegetation proxy for GHG emissions is the system of GHG Emission Site Types (GESTs) developed by Couwenberg *et al.* (2011). The GESTs are vegetation types with relationships to water level based on 'bioindication' by ecological plant groups (Koska *et al.* 2001) or plant species (Ellenberg *et al.* 1992). Emission factors were assigned to these vegetation types and verified or specified by regression models of GHG flux against mean annual water level. Mapping of the GESTs could thus be directly linked to flux values, providing a highly standardised and practical approach but also presenting many uncertainties. On the one hand, using Ellenberg Indicator Values (EIVs) introduces uncertainties about water level indication and thus the vegetation groups. Originally designed for central Europe, the EIVs have been calibrated for The Netherlands (e.g. Ertsen *et al.* 1998) to improve local fitness. However, the species-based ordinal values are still largely qualitative and can provide only ranges and classes of water level with unmeasurable variances. On the other hand, the emission factors could be calibrated for different locations and validated by direct measurements. However, the time, money and labour required for GHG emission measurements (Günther *et al.* 2018) make both the expansion of flux values and the ground truthing difficult and not cost-effective.

In the present study we used a similar approach to link vegetation composition to groundwater levels but, instead of using plant species indications such as those proposed by Ellenberg *et al.* (1992), we used data that directly linked local vegetation types to measured (mean) groundwater levels (e.g. Grootjans & Ten Klooster 1980). Holtland *et al.* (2010) also used measured indicator values in the Iteratio model, but by calculating weighted averages of species-based values rather than using vegetation types directly. Vegetation types are combinations of plant species with similar ecological requirements. The bioindication of a combination of species is more accurate than that of separate species (Niemann 1973). Furthermore, the indication for wetness of species may differ between geographical regions (Kotowski *et al.* 1998). For this reason, we used data from north-western and central Europe only. Meanwhile, the correlation of GHG emissions with annual mean water levels has been updated continuously with measurements from various geological and climatic regions (Couwenberg *et al.* 2011, Wilson *et al.* 2016a). Therefore, combining bioindication of water levels from vegetation types and the correlation of water levels with GHG emissions could provide a reliable approach to estimating GHG emissions that circumvents the problems of vague bioindication by EIVs and lack of high-quality GHG flux measurements.

This article aims to describe a GHG emissions estimation approach that combines vegetation types and water level proxies. The approach is illustrated by a case study in the Drentsche Aa brook valley (Netherlands) which was originally a fen system, later intensively used for agriculture, and rewetted since the 1990s. The main questions of this article are: 1) is the bioindication of vegetation for annual mean water level a reliable proxy for GHG emissions compared to existing tools; and 2) to what extent did rewetting of the Drentsche Aa brook valley provide emissions reduction as a co-benefit.

METHODS

Study area

The Drentsche Aa brook valley (53° 7' 12.39" N, 6° 37' 34.45" E) is located in the province of Drenthe, north-east Netherlands. The natural mires in the area have been progressively drained and fertilised for intensive agriculture since the Middle Ages, and the hydrological system has been disturbed by groundwater abstraction (for drinking water) close to the brook valley, causing various degradation problems e.g. vegetation losses, desiccation and eutrophication. Starting in 1965, the degraded peatlands were gradually converted into nature protection areas, and in 2002 around 32,000 ha of the brook valley was designated as National Landscape with 35 % nature reserves. The National Landscape includes villages and areas still in agricultural use. In the Drentsche Aa brook valley, more than 600 ha of meadows have been rewetted since 1996 by blocking drainage ditches. Mowing of biomass and topsoil removal have also been undertaken, aiming to facilitate the recovery of species-rich meadows.

The results of the restoration efforts have been monitored over time by repeated detailed vegetation mapping on more than 2,000 ha of the natural areas. Within each of the mapped vegetation types, dominant, co-dominant and local plant communities were recorded according to their appearance and abundance. The maps have shown a reduced level of desiccation and recovery of rare species, but the reduction of GHG emissions has not been discussed. This could be a substantial co-benefit of rewetting, alongside the results in conservation of vegetation biodiversity, especially as the potential of climate benefits turning into new business models involving the carbon market is gaining interest amongst local land managers.

The emissions estimation was restricted to peatlands within the designated natural areas of the National Landscape. The vegetation maps used were made from two rounds of vegetation surveys carried out in 1994–1996 (before rewetting) and 2015–2016 (after rewetting), covering 2143 ha and 2481 ha respectively, and overlapping by 1102 ha. The distribution of peat soil in the brook valley was extracted from a peat thickness map of the northern part of the Netherlands (de Vries *et al.* 2014), which is a 50 m × 50 m raster dataset with peat thickness values interpolated from available soil survey data up to 2014. Areas with less than 40 cm of peat soil were excluded from the calculations in order to ignore peat soil undergoing depletion, which may have a different emission mechanism and would be a minor influence on the long-term carbon budget. After overlaying and exclusion, the remaining 561 ha of peat soils in natural areas of the National Landscape were extracted. All spatial analysis was conducted using ArcGIS 10.3 software.

Linking vegetation to mean annual water levels

In the present study, information on the relationship between plant communities and mean annual groundwater levels was derived from studies published in West European literature (mainly from The Netherlands; Grootjans & Ten Klooster 1980, de Haan 1992) that had related measured groundwater levels over several years to vegetation types (at the level of associations or sub-associations). Data from clay and loam soils were discarded because their groundwater characteristics differ markedly from those of peaty and sandy soils. All groundwater fluctuation data were converted to cumulative frequency diagrams (duration lines) representing an approximation of the period that a certain water level had been exceeded. Annual means and standard deviations of water level were calculated from multiple duration lines. Means of annual highest/lowest water levels were also calculated to show the extremes of inter-annual water level variation. Woody plant species were not included due to lack of water level monitoring data. The resulting database was then extrapolated for plant communities that had insufficient or no data but species composition similar to that of analysed communities according to their dominant and characteristic species, based on expert opinion and field experience. Using the resulting water level indication inventory, annual mean water level values were assigned to patches of the vegetation maps based on the dominant plant communities recorded, then water level changes were examined by overlaying and subtracting the two maps across their matching spatial extent.

GHG emissions estimation

Agricultural management and chemical fertilisation were withdrawn from the designated natural areas of the brook valley National Landscape in 1965, so we expected the main GHG emissions to be CO_2 and CH_4 from peat decomposition rather than N_2O and CH_4 reflecting influence of remnants from former agricultural activities (cf. Audet *et al.* 2013). Therefore, regression models of net annual CO_2 and CH_4 fluxes against annual mean water level from Couwenberg *et al.* (2011) were adopted and N_2O fluxes were neglected because of their low magnitude in natural and rewetted peatlands without agricultural inputs (Schrier-Uijl *et al.* 2014, Wilson *et al.* 2016b). This is supported by analysis of water samples collected in the brook valley during 2014 and 2015, which showed very low nitrogen contents (average concentrations of nitrate and ammonium in 32 groundwater samples were 1.4 and 0.2 mg L^{-1}, respectively; Elshehawi *et al.* 2019). Based on meta-analysis of year-round flux data collected in temperate Europe, the following functions were adapted for our datasets from Couwenberg *et al.* (2011):

$$\text{Net annual } CO_2 \text{ fluxes} = 752 \times \text{MWL} - 4750 \quad (\text{MWL} \geq 0) \qquad [1]$$

$$\text{Net annual } CH_4 \text{ fluxes} = 16.7 \times (20 - \text{MWL}) \quad (0 \leq \text{MWL} \leq 20) \qquad [2]$$

where MWL stands for annual mean groundwater level (cm below ground surface) and greenhouse gas fluxes are expressed in kg ha^{-1} yr^{-1}. These functions do not describe fluxes under conditions of standing water, since there was no area in the brook valley with annual mean water level above the ground surface (see Results). However, some of the plant communities indicate possible seasonal inundation with mean highest water levels up to 20 cm above surface, although this happens only occasionally for a short period of time. This could lead to underestimation of CH_4 fluxes as relatively high CH_4 emissions have been observed on inundated peatlands after rewetting, especially in eutrophic fens (Jurasinski *et al.* 2016). In addition, the CH_4 fluxes function applies only when MWL $\leq$ 20 cm; MWL larger than 20 cm was thus taken to indicate a CH_4 flux of 0 instead of uptake. Net GHG emissions were calculated from the fluxes as Global Warming Potential (GWP) using 25 as the CO_2-equavelent of CH_4. Emissions reductions were evaluated by overlaying and subtracting fluxes and GWP maps from before and after rewetting.

Comparison with other estimation approaches

Besides estimating landscape-scale changes in GHG fluxes and GWP using the above methodology, the estimations of emissions reduction were compared with results from other approaches. Estimations based on emission factors from the IPCC Wetlands Supplement (IPCC 2013) and based on water level monitoring data from a previous study (Hoetz 2013) were considered (Table 1). The comparison was restricted to the 135 ha of nature reserve located in middle stream areas of the brook valley where the local land manager (Dutch Staatsbosbeheer) reported that rewetting management had been implemented. Overall GWP changes and spatial patterns resulting from this management were estimated.

Table 1. Estimation approaches for comparison.

Reference	Approach	Data / factors used
IPCC 2013	Categorise the landscape according to the IPCC guideline as temperate nutrient-rich grasslands that are deep-drained/shallow-drained/rewetted according to the indicated water levels from vegetation maps, then apply the corresponding emission factors.	CO_2-eq emission factors (t ha^{-1} yr^{-1}): Deep-drained (WL > 30 cm) 22.77 Shallow-drained (0 < WL ≤ 30 cm) 14.18 Rewetted (WL ≤ 0 cm) 7.23
Hoetz 2013	Extrapolate water level monitoring data points (DINOlokot.nl) to landscape scale for spatial data on water level changes, then apply general regression models of water level - GHG fluxes.	Five measurements from three locations were used for water level changes after excluding data for deeper aquifer and measurement sites too close to ditches.

RESULTS

Annual mean water level changes

The indication of water level based on plant communities is shown in Table A1 (Appendix). Annual mean water levels (cm below ground surface) are presented as the indicator values, together with means and standard errors of annual highest/lowest water levels and number (n) of duration lines accounted to represent uncertainties. In total, ten vegetation types are given annual water level values and variances, while other types appearing in the vegetation maps which were rare in the landscape or had no measurement data were given interpolated annual water level values based on characteristic species composition according to expert experience. Applying annual mean water level values from Table A1 to the vegetation maps resulted in water level maps covering an area of 250 ha. Mean water level changes were calculated and divided into five sub-areas along the brook valley. Annual mean water levels in the brook valley ranged from surface level (0 cm) to 40 cm below surface, with averages of 24.4 cm and 19.9 cm below surface before and after rewetting, respectively. The annual mean water level rise was on average 5.5 cm throughout the landscape, with approximately 80 % of the total area having maintained or raised water levels after rewetting. Annual mean water level changes were different for each part of the brook valley (Figure 1). In the middle reaches of the brook valley system, where most of the rewetting measures were applied, mean annual water level had risen considerably, by 9.3 cm on average, although a few small patches had lowered water levels. The source area and the transition areas from upper/lower stream to the middle stream had experienced modest rises in mean annual water level of 5.8 cm and 6 cm/5.4 cm, respectively. A negligible 0.7 cm rise in mean annual water level was estimated for the downstream area, where spatial variation was higher and larger patches showed lowered mean annual water levels (Figure 1).

GHG emissions reduction

GHG emissions showed various changes across the catchment (Figure 2). Overall, the landscape remained a net carbon source dominated by CO_2 emission. Annual mean CO_2 fluxes ranged from 25 t ha^{-1} yr^{-1} to -5 t ha^{-1} yr^{-1} with an average of 10 t ha^{-1} yr^{-1} after rewetting (Table 2; negative fluxes indicate uptake). However, CO_2 fluxes were substantially reduced at an average of 5 t ha^{-1} yr^{-1} after rewetting. While CO_2 reduction was generally high in the middle reaches and upstream areas, more patches downstream showed neutral or increased CO_2 fluxes. CH_4 flux increased significantly (the annual average of 90 kg ha^{-1} yr^{-1} is three times the value before rewetting; Table 2), although this emission was relatively small in magnitude. In a considerable number of patches, spatially concentrated in the middle reaches, CH_4 flux increased to more than 150 kg ha^{-1} yr^{-1}, while more variation was observed

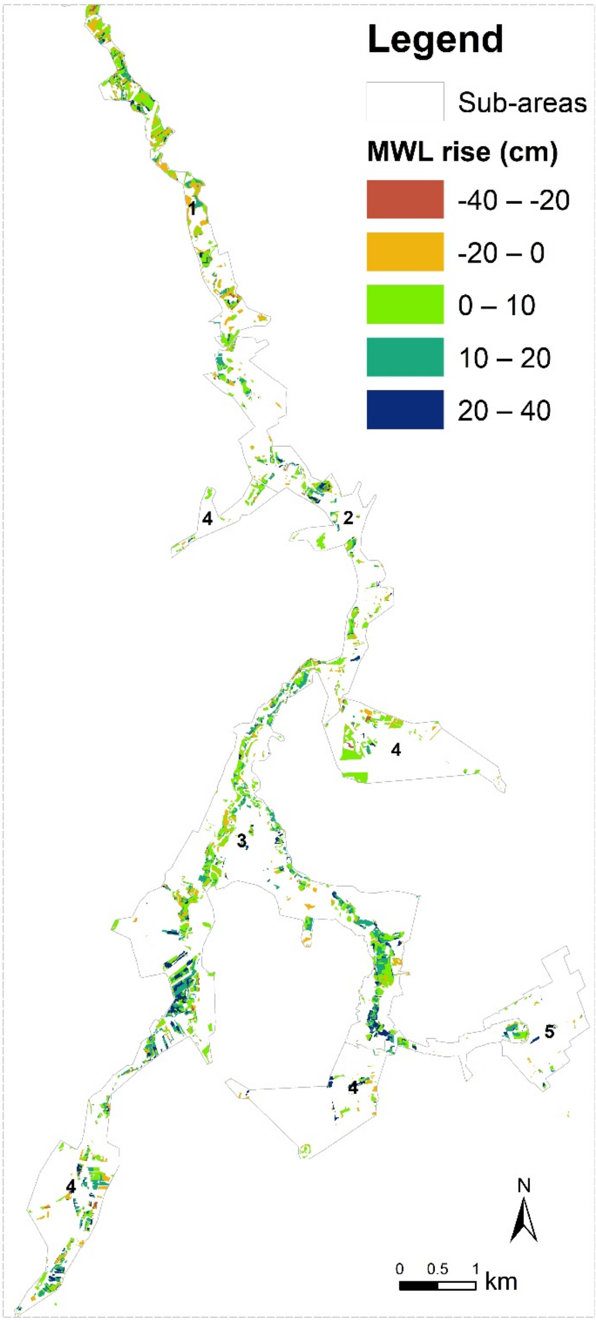

Figure 1. Changes in mean water level with green and blue colours indicating raised water level. The landscape is clustered into sub-areas: 1 = downstream, 2 = transition down-middle reaches, 3 = middle reaches, 4 = transition middle-upper reaches, 5 = source area.

in downstream areas (Figure 2b). In terms of GWP, the increase in CH_4 fluxes (expressed as CO_2-eq) offset 33 %, or nearly 400 t yr^{-1}, of the overall reduction in CO_2 emissions (Table 2). Nonetheless, rewetting of the Drentsche Aa brook valley still effected a 20 % emissions reduction overall.

Comparison with other estimation approaches

Extracting the rewetted areas from the overlaid maps indicated that the annual average (CO_2-eq) GWP reduction was 5 t ha^{-1} yr^{-1} (Figure 3a). Applying the IPCC emission factors resulted in a GWP reduction of 2 t ha^{-1} yr^{-1}. The GWP reduction estimated from monitoring data, derived by extrapolating rises in water level of 10 cm from 'Measurement A & B', 15 cm from 'Measurement C & D' and 5 cm from 'Measurement E' (Figure 3c) to nearby locations and applying the general regression models, was 4 t ha^{-1}

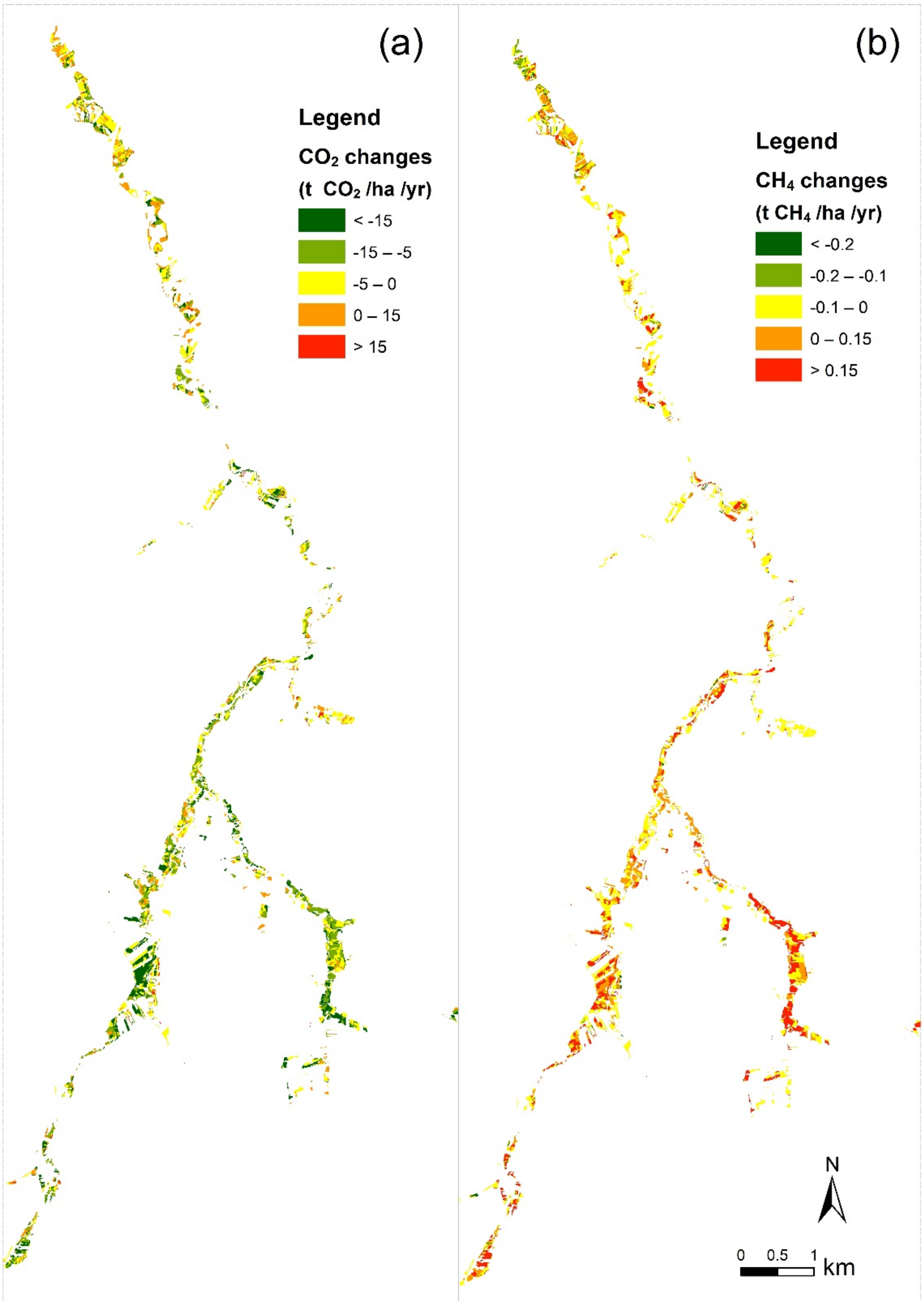

Figure 2. GHG fluxes changes of (a) CO_2 and (b) CH_4 after rewetting. Green and yellow colours show reduction and reddish colours show increase on corresponding GHG fluxes.

Table 2. Changes in mean GHG fluxes and total emissions before and after rewetting.

	Mean fluxes			**Annual total emissions**		
	CO_2 ($t\ ha^{-1}\ yr^{-1}$)	**CH_4 ($kg\ ha^{-1}\ yr^{-1}$)**	**CO_2-eq GWP ($t\ ha^{-1}\ yr^{-1}$)**	**CO_2 (t)**	**CH_4 (t)**	**CO_2-eq GWP (t)**
Before	14.39	30	15.10	3584	7	3761
After	9.60	90	11.88	2392	23	2958
Changes	-4.79	60	-3.22	-1192	16	-803

yr^{-1}. Despite having the same order of magnitude, the resulting estimates of emission reductions differed significantly in terms of spatial patterns.

Compared to our approach, the IPCC emission factor approach showed a similar pattern but at lower resolution, since the land categories failed to capture minor changes in water level (Figure 3b). The extrapolation of water level measurements showed a highly homogeneous pattern without the recognisable trend shown by the other two approaches (Figure 3c).

DISCUSSION

Linking vegetation types to mean annual water levels

The occurrence and abundance of plant species can provide integrated indication values for various environmental parameters over longer time periods (Schaffers & Sýkora 2000, Bartelheimer & Poschlod 2016) but, since new species need time to disperse and establish, vegetation changes may 'limp behind' shifts in environmental conditions (Diekmann 2003). Although the vegetation maps may not represent the latest site conditions, the interval of more than ten years between the two maps (1994 and 2015) is sufficient to give representative water level values integrating conditions before and after rewetting in 1996. By taking annual mean water level values from direct observations or interpolations based on plant communities rather than individual species, our approach can arguably improve the accuracy of bioindication. Although many other factors besides water level also influence the occurrence of plant communities, there will be no effect on accuracy because the plant association will change with such factors (e.g. pH, nutrient levels, etc.) and corresponding water level measurements would be considered according to our approach. However, extra uncertainties will be introduced by the grouping up of sub-associations to obtain the ten main types presented in Table A1.

More than 70 plant communities were considered in this study, which has given a rather complex spatial pattern of the water levels and their changes. Although the landscape-scale average change of annual mean water level isn't substantial, changes of the sub-areas are in accordance with the rewetting measures implemented, since middle stream areas where ditch closure was applied had the highest annual mean water level rise, while water levels of lower stream areas that are strongly influenced by the surrounding agricultural land and human settlements showed hardly any change. The dynamics of groundwater differ between different landscape or hydrological units with different topographic positions, for example distance from stream (Blumstock *et al.* 2016). This may have resulted in lower water levels occurring even in small patches of the middle stream after rewetting. Thus, the real-world situation cannot be summarised by one average value. Higher resolution provided by our more detailed indication could highlight contrasting changes within different landscape units, which could help with troubleshooting of the management and provide reference states for later-stage planning.

GHG emissions reduction and its spatial patterns

The annual mean CO_2-eq GWP values derived from the vegetation - water level proxies before and after rewetting were 15.10 and 11.88 $t\ ha^{-1}\ yr^{-1}$, respectively. This matches the 12.57 $t\ ha^{-1}\ yr^{-1}$ CO_2-eq emission factor for rewetted nutrient-rich temperate grassland updated from the IPCC Wetlands Supplement by Wilson *et al.* (2016a). Annual mean GWP was dominated by CO_2 both before and after rewetting, with annual mean CO_2 emissions of 14.39 and 9.60 $t\ ha^{-1}\ yr^{-1}$ contributing 95 % and 81 % of the GHG balances, respectively. Peacock *et al.* (2019) reported a CO_2 balance roughly in the same order of magnitude (4.88 $t\ ha^{-1}\ yr^{-1}$) and negligible CH_4 fluxes for a rewetted cropland. Such

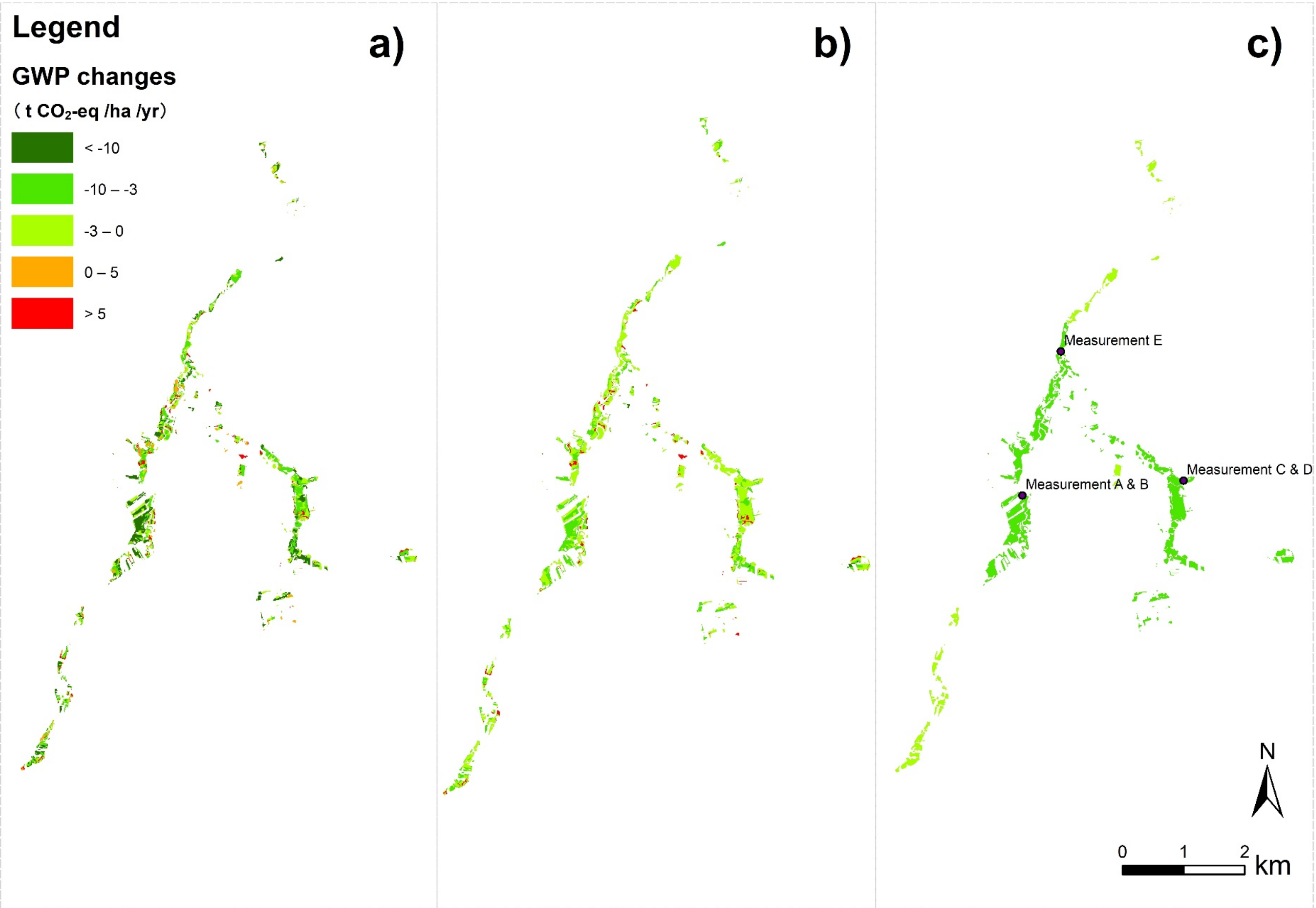

Figure 3. Comparison of GWP reduction from 3 approaches at rewetted areas in the middle reaches of the Drentsche Aa: a) the vegetation - water level proxy approach; b) the IPCC emission factors approach; c) the measurement-based approach. Points represent locations of measurement wells used.

carbon balances with high CO_2 fluxes are closer to the GHG budgets of grasslands on peat (e.g. Teh *et al.* 2011, Schrier-Uijl *et al.* 2014, Tiemeyer *et al.* 2016) than to the CH_4-dominant balances reported from some rewetted cases (e.g. Vanselow-Algan *et al.* 2015, Windham-Myers *et al.* 2018), although they did show increasing importance of CH_4 fluxes. They are far from the re-established net carbon sink (despite relatively high CH_4 emissions) that could potentially be achieved (e.g. Schrier-Uijl *et al.* 2014, Wilson *et al.* 2016b, Windham-Myers *et al.* 2018).

Higher water level (especially higher than 20 cm depth) and presence of aerenchyma plants can substantially increase CH_4 production and effluxes (Waddington & Day 2007, Couwenberg & Fritz 2012, Shao *et al*. 2017, Strack *et al.* 2017). For example, on a temperate wetland rewetted for 12 years, Kandel *et al.* (2019) observed CO_2 fluxes similar to our estimation (8.07 t ha^{-1} yr^{-1}) but CH_4 fluxes an order of magnitude higher (CO_2-eq 14.85 t ha^{-1} yr^{-1}), due to annual mean water level close to surface and an inundation period in winter. In our case, the relatively low CH_4 fluxes were mainly due to a smaller effect of rewetting at catchment scale, leading to a generally lower water level than in rewetted sites where emissions are CH_4 dominated. Notably, influence of plant species and possible high water level from the inter-annual fluctuation were not considered in our estimation. Therefore, when incorporating emissions in future conservation management, extra caution is needed to prevent high CH_4 emissions since the increase of CH_4 emissions already neutralised almost one third of the CO_2 reduction in our case (Table A1). Carbon fluxes at landscape scale are strongly dependent on spatial heterogeneity and diversity (Premke *et al.* 2016). Hotspots of increasing CH_4 fluxes (Figure 2) should have more attention for a spatial-specific mitigation measure. For instance, downstream locations with lower average water level should receive more intensive rewetting measures, while water management in middle stream areas with generally higher water level should focus towards a more even and stable water level. Meanwhile, inter-annual fluctuation of water level that may lead to temporarily high water levels should also be taken into account, for identifying the possible peaks of CH_4 fluxes that could influence the accuracy of an annual emissions estimation.

Notably, the 21 % reduction of total GWP (Table A1) estimated in this study is solely a side-effect of the restoration measures designed for the recovery of rich fen vegetation with no consideration for emission reduction. The potential for co-benefits of biodiversity conservation and climate change mitigation has been frequently noted (Bryan *et al.* 2016, Essl *et al.* 2018). Such potential could be better exploited with additional attention on emission reduction to create a combined strategy. The possible use of such reductions as carbon credits in carbon markets (Newell *et al.* 2014) also opens up opportunities to expand the current business model of conservation management. In this case our approach provided an option for using existing knowledge, i.e. biodiversity monitoring data, as alternative input (rather than using direct measurements) to produce reliable quantification of the emission reductions.

Comparison with other estimation approaches

The problem of applying the IPCC Wetlands Supplement was the adoption of its category system to the landscape. At landscape scale, the brook valley cannot be defined as ‘wet soil’ according to the IPCC guideline (IPCC 2013) with a rather ‘moist’ water level condition. Applying the ‘rewetted’ category widely on the landscape based on management would give a substantial overestimation of the reductions. Applying the IPCC categories according to water level definitions showed a lower sensitivity, because small annual mean water level changes will not result in category changes, and the corresponding emission changes will thus be ignored. This explains the overall underestimation from the IPCC approach. Results from the measurement-based approach were more similar to our estimation. However, extrapolation of very limited number of measurements resulted in a highly uniform spatial pattern. Therefore, although measurements from groundwater level wells have been widely used for plot-scale flux analysis (Wilson *et al.* 2015, Minke *et al.* 2016, Chimner *et al.* 2017), they would not be reliable as input for finer-scale estimations considering their lack of spatial variation and representativeness. Substitution of both detailed water level indication and continuous emission functions resulted in loss of spatial information in this comparison. Small patches identified by the proxy approach that have increased emissions or higher reductions do not have major effects on the average GWP reduction value but will affect the evaluation of conservation by providing information on the specific locations of strong effects or failure. Although more detailed inputs are required, such spatially explicit information is more valuable for landscape-scale assessments than an average value with management-oriented aims.

Prospects

As a compromise to insufficient measurement data and model uncertainties, our approach focuses on providing details on spatial patterns that could contribute to the optimisation of a location-specific strategy for long-term landscape management. However, the accuracy of this approach still needs improvements or at least evaluation. It has been shown that the inclusion of flux measurements in a carbon credit scheme could still be profitable, and a higher price of certificates could be achieved with more reliable methodology (Günther *et al*. 2018). One direct way is to validate and calibrate the relationships that the proxy relies on, i.e. the bioindication of groundwater level by vegetation and the correlations between water level and GHG fluxes, by collecting higher-quality region-specific flux data with adequate monitoring of environmental factors. On the other hand, given the difficulty of gathering enough flux measurements, reliability of the estimation could also be enhanced by quantifying the uncertainties. Besides providing mean annual flux values, emissions could be estimated as confidence intervals if more statistical characteristics of the input data (e.g. annual mean water level for each vegetation type) and model parameters (e.g. coefficients of the linear models) could be identified, which could help decision makers to fully grasp the possible effects of their conservation strategies and the associated benefits or risks.

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AUTHOR CONTRIBUTIONS

APG supervised the project and provided data on the relationship between vegetation and groundwater fluctuation. HE and NdV provided the vegetation maps and developed the vegetation typology and GIS datasets. WL performed the GIS analyses and wrote the first draft of the manuscript. APG, CF and HE contributed to writing of the manuscript.

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Appendix

Table A1. Water level bioindicator values of vegetation types. Water levels of main and sub types are presented as annual means, together with years of measurements (n), standard deviations, maximum and minimum values and references.

	Vegetation type (Scientific name)	Years of measurement (n)	Annual mean water level (cm below surface)	Standard deviation	Max	Min	References
1	**Vegetation of open water**	-	0	-	0	0	-
2	**Reed communities (Typho-Phragmitetum)**						
	with *Typha*	13	11	12	-15	20	Geurts & Fritz 2018
	with *Glyceria*		5		-20	40	
	with *Phragmites*	3	12	10	0	20	Geurts & Fritz 2018
3	**Tall sedges (Caricetum gracilis)**						
	with *Carex gracilis*	1	5	-	-20	10	Tüxen 1954, Balátova-Tuláčková 1968, Wiedenroth 1971
	with *Carex aquatilis*	1	5	-	-20	10	Grootjans & Ten Klooster 1980
	with *Carex acutiformis*	1	20	-	0	45	
	with *Phalaris*	1	30	-	10	70	Everts & de Vries 1991
4	**Small sedges (Caricetum nigrae)**						
	typicum	15	6	8	-7	20	de Haan 1992
	with *Carex lasiocarpa*	4	13	4	8	18	de Haan 1992, Dierssen & Dierssen 1985
	with *Carex aquatilis*	1	9	-	0	18	Grootjans & Van Tooren 1984
	with *Carex rostrata*	1	9	-	0	18	de Haan 1992, Zarzycki 1958
5	**Acid small sedges**	1	7	-	0	30	Grootjans & Ten Klooster 1980
6	**Wet meadows (Calthion palustris)**	1	19	-	0	42	Tüxen 1954, Balátova-Tuláčková 1968, Grootjans & Ten Klooster 1980
7	**Poor wet meadows (Cirsio-Molinietum)**	13	26	7	14	36	de Haan 1992
8	**Drained wet meadows**	1	30	-	-20	70	Grootjans *et al*. 1985
9	**Flooded fertilised meadows**	1	40	-	20	80	Everts & de Vries 1991
10	**Unmanaged meadows**	1	15	-	0	45	Everts & de Vries 1991