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The role of waterborne carbon in the greenhouse gas balance of drained and re-wetted peatlands

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Abstract

Accounting for greenhouse gas (GHG) emissions and removals in managed ecosystems has generally focused on direct land-atmosphere fluxes, but in peatlands a significant proportion of total carbon loss occurs via fluvial transport. This study considers the composition of this 'waterborne carbon' flux, its potential contribution to GHG emissions, and the extent to which it may change in response to land-management. The work describes, and builds on, a methodology to account for major components of these emissions developed for the 2013 Wetland Supplement of the Intergovernmental Panel on Climate Change. We identify two major components of GHG emissions from waterbodies draining organic soil: i) 'on site' emissions of methane (and to a lesser extent CO_2) from drainage ditches located within the peatland; and ii) 'off site' emissions of CO_2 resulting from downstream oxidation of dissolved and particulate organic carbon (DOC and POC) within the aquatic system. Methane emissions from ditches were found to be large in many cases (mean 60 g CH_4 m^{-2} yr^{-1} based on all reported values), countering the view that methane emissions cease following wetland drainage. Emissions were greatest from ditches in intensive agricultural peatlands, but data were sparse and showed high variability. For DOC, the magnitude of the natural flux varied strongly with latitude, from 5 g C m^{-2} yr^{-1} in northern boreal peatlands to 60 g C m^{-2} yr^{-1} in tropical peatlands. Available data suggest that DOC fluxes increase by around 60% following drainage, and that this increase may be reversed in the longer-term through re-wetting, although variability between studies was high, especially in relation to re-wetting response. Evidence regarding the fate of DOC is complex and inconclusive, but overall suggests that the majority of DOC exported from peatlands is converted to CO_2 through photo- and/or bio-degradation in rivers, standing waters and oceans. The contribution of POC export to GHG emissions is even more uncertain, but we estimate that over half of exported POC may eventually be converted to CO_2. Although POC fluxes are normally small, they can become very large when bare peat surfaces are exposed to fluvial erosion. Overall, we estimate that waterborne carbon emissions may contribute about 1 to 4 t CO_2-eq ha^{-1} yr^{-1} of additional GHG emissions from drained peatlands. For a number of worked examples this represented around 15 to 50% of total GHG emissions.

Keywords: waterborne carbon, peatlands, drainage, greenhouse gases, DOC, methane

1. Introduction

1.1 Management impacts on the peatland greenhouse gas balance

Peatlands occur in many parts of the world, where the suppression of decomposition by waterlogged conditions leads to an excess of primary production over that lost to respiration. Although they occupy only around 3% of the land surface, it is thought that they hold 470-620 Pg of carbon, making them the single largest terrestrial carbon store (Page et al., 2011). Additionally, and in contrast to most other soils, peatlands can continue to sequester small but sustained amounts of CO_2 for millennia after their initiation, whilst also acting as sources of methane (CH_4). On the 100 year time window over which global warming potentials (GWPs) are commonly calculated, these two opposing fluxes mean that natural peatlands are approximately climate-neutral, or may even have a net warming effect where CH_4 emissions are large (e.g. Nilsson et al., 2008). However, as noted by Frolking et al. (2006), the long-term sequestration of CO_2 into stable organic matter gradually outweighs the warming effect of CH_4, due to the shorter atmospheric lifetime of the latter, so that natural peatlands exert a net cooling impact on the atmosphere over longer periods.

Globally, peatlands have been modified to support production of crops, livestock and timber, and the peat itself removed for use as a fuel or in horticulture. Historically, the greatest modification of peatlands has occurred in Europe, with the majority of all peatlands in continental Europe now converted to agriculture, and large areas of European Russia and Fenno-Scandia drained to support production forestry; for the European region as a whole Joosten and Clark (2002) estimated that 26% of the peatland area has been converted to agriculture, and 16% to forestry. Significant peatland conversion has also taken place in North America, Northeast Asia and Africa (Joosten and Clark, 2002), and most recentlyin Southeast Asia; it has been estimated that almost half of Southeast Asian peat swamp forests have now been cleared, drained, often burnt, and in many cases converted to oil palm and pulpwood plantations (Hooijer et al, 2010). In almost all cases, these land-use changes involve drainage of the peatland, exposing organic matter which has accumulated under anaerobic conditions to oxygen. Drainage permits the establishment of more productive and/or commercially valuable vegetation types, but simultaneously causes the accelerated aerobic decomposition of the peat, leading in many cases to large and sustained rates of C loss (e.g. Byrne et al., 2004; Bridgham et al., 2006), as well as land subsidence (e.g. Waltham, 2000).The contribution of drained peatlands to global anthropogenic CO_2 emissions is significant; the recent IPCC Assessment Report (IPCC, 2014b) estimated total CO_2 emissions from drained peatlands of 1.1 to 1.5 Gt CO_2 yr^{-1}, representing around one third of net GHG emissions from the entire Forestry and Other Land Use (FOLU) sector. On the other hand, drainage tends to reduce CH_4 emissions from the peatland surface to near-zero values; the IPCC Guidelines for Greenhouse Gas Inventories (IPCC, 2006) considered that CH_4 emissions from all drained organic soils (i.e. former peatlands) were zero.

1.2. The role of waterborne carbon in the peatland greenhouse gas balance

Direct (gaseous) fluxes of GHGs from the surface of undrained and drained peatlands have now been fairly well quantified, particularly for northern temperate and boreal systems (e.g. Alm et al., 2008; Couwenberg et al., 2011; Yu, 2012; IPCC, 2014a), and increasingly for tropical peatlands (e.g. Jauhiainen et al., 2012). Within natural peatlands in general, there is also growing evidence that the waters draining these systems represent an important conduit for carbon loss, and that this 'waterborne carbon' flux can significantly influence the overall carbon balance of the peatland. For example, Billett et al. (2004) measured a waterborne carbon flux of 30 g C m^{-2} yr^{-1}, which they estimated was sufficient to turn their study site, a raised bog in Southern Scotland, from an apparent carbon sink (based on direct land-atmosphere fluxes alone) into an actual carbon source. A number of subsequent flux measurement studies in other near-natural peatlands (Roulet et al., 2007; Nilsson et al., 2008; Koehler et al., 2011), as well as further measurements at the same site (Dinsmore et al., 2012), confirm that waterborne carbon is, to varying degrees, a quantitatively important component of the carbon balance. The omission of waterborne fluxes leads to a systematic bias in calculated

carbon balancesunder-estimating carbon losses from many ecosystems (e.g. Ciais et al., 2008; Gielen et al., 2011; Kindler et al., 2011). However the issue is most pronounced for peatlands.

Several issues limit our current understanding of the role of waterborne carbon in the peatland carbon balance, and also its overall importance in terms of GHG fluxes. Firstly, 'waterborne carbon' incorporates a number of different carbon forms, including free gaseous CO_2 and CH_4, dissolved inorganic carbon (DIC, comprising bicarbonate, HCO_3^-, and carbonate, CO_3^{2-}), dissolved organic carbon (DOC) and particulate organic carbon (POC). Secondly, these various carbon forms have differing sources within the peatland, exhibit different physical, chemical and biological behaviour, and have differing fates. This latter issue is of particular importance to the peatland GHG balance, since waterborne carbon which is ultimately converted to (or directly emitted as) CO_2 or CH_4 will contribute to the overall GHG emission from the peatland, whereas any which is re-deposited in a stable form (e.g. in lake or marine sediments) may not. Thirdly, while waterborne carbon fluxes have been fairly well studied within natural peatland systems, fewer data are available describing how these fluxes may change in response to drainage and associated land-use change.

1.3. 'On-site' and 'Off-site' emissions

In our treatment of waterborne carbon, we follow the IPCC's terminology in considering 'on-site' and 'off-site' emissions (IPCC 2006, 2014).'On-site' emissions of GHGs are associated with gas transfer across the water surface to the atmosphere from water bodies located within the peatland itself. These emissions may include degassing of CO_2 and CH_4 (and potentially also the non-carbon GHG nitrous oxide,N_2O) carried in water transported from the peat matrix, and/or gases produced within the water body or its underlying sediment as labile organic substrates are metabolised. Gases may be emitted either through diffusive fluxes across the water surface, due to over-saturation of the gas in the water column relative to the atmosphere (e.g. Billett and Moore, 2008), or via the physical movement of bubbles (ebullition) through the water column (e.g. Vermaat et al., 2011). In accordance with reporting frameworks used for the FOLU sector under the United Nations Framework Convention on Climate Change (UNFCCC),, GHG emissions associated with natural peatland processes are not accounted for, which in this context means that only on-site emissions from artificial water bodies such as drainage ditches need to be considered. However, much of the research on GHG emissions from peatland water bodies has been carried out within natural features such as streams, pools and lakes (e.g. Hope et al., 2001; Billett and Moore, 2008; Repo et al., 2007; Dinsmore et al., 2010; Buffam et al., 2011; Koehler et al., 2011; Juutinen et al., 2013; Wallin et al., 2013). While this has enhanced our understanding of fundamental processes, natural fluxes and measurement techniques, it has not provided direct data on GHG emissions from artificial water bodies within managed peatlands, notably drainage ditches. Drainage ditches are designed to alter the hydrological functioning of peatlands by lowering and/or controlling the level of the water table, and facilitating the transfer of water out of (or occasionally into) the peat body. They generally increase the spatial density of the natural drainage network, and thus the area of surface water from which GHG emissions may occur (although this may be counterbalanced by the drainage of natural peat pools). Ditches are usually slow-flowing, so low levels of water turbulence may resulti in low rates of gas evasion, although zones of higher turbulence can occur where sluices, weirs or pumps are used to control water levels, or in the higher-gradient ditches present in some blanket bogs.

'Off-site' emissions of waterborne carbon primarily occur through the chemical or biological transformation of peat-derived organic matter downstream of the peatland, leading to the formation and emission of CO_2 or CH_4. Although dissolved CO_2 and CH_4 may also be transported into drainage systems, if present in excess of atmospheric concentrations they tend to rapidly evade,

until equilibrium is reached. Although CO_2 evasion from streams with peat catchments can be large (e.g. Dinsmore et al., 2010; Wallin et al., 2013), most of this evasion occurs within or close to the peatland itself, and may therefore be considered as mainly an 'on-site' emission.

In most peatlands (whether natural or managed), DOC forms the largest component of waterborne carbon export (e.g. Dawson *et al.*, 2004; Jonsson *et al.*, 2007; Dinsmore *et al.*, 2010). It is produced through biological activity including plant exudation and decomposition processes, although its transport into drainage waters is also influenced by hydrological factors (e.g. Clark et al., 2007) and chemical solubility controls (Evans et al., 2012). Once in the drainage network, it is susceptible to photochemical breakdown (e.g. Köhler et al., 2002; Cory et al., 2014) and may also be utilised as an energy and carbon source by heterotrophic organisms (e.g. Battin et al., 2008).

In most natural and managed peatlands, POC exports are small, but exceptions occur where bare peat surfaces are exposed through erosional processes, peat extraction, burning, forest operations or cultivation. In these circumstances, POC losses may be very high; for example, Worrall et al. (2011) recorded POC fluxes of over 100 g C m^{-2} yr^{-1} from an eroding blanket bog, far exceeding typical DOC fluxes from these systems. There is less evidence that drainage alone (i.e. without accompanying bare peat exposure) can generate large POC fluxes, although the ditches themselves may mobilise POC if they erode into the peat body. Furthermore, because POC mobilisation is largely a physical process, it is possible that this material will be unreactive within the fluvial system, and will simply sediment out on floodplains, within lakes or in the oceans, thereby making little short-term contribution to GHG emissions.

In waters draining bog peats, DIC fluxes (other than rapidly-evaded CO_2) are generally negligible due to the low solubility of CO_2 at low pH. A different situation arises in fen peats, where high pH values are associated with the presence of HCO_3^- and (in the most alkaline systems) CO_3^{2-}. The significance of this flux in terms of the peatland carbon balance depends on its source. Much of the DIC in fen drainage derives from mineral weathering in groundwaters feeding the fen, and does not therefore represent a net carbon flux from the peat itself. On the other hand, CO_2 produced through peat respiration under alkaline conditions may equilibrate with water to form HCO_3^-, and thus a component of the DIC flux may represent carbon loss from the peatland (e.g. Worrall et al., 2003; Fiedler et al., 2008). Subsequent incorporation into biomass through autotrophic activity may reintroduce this DIC into the biological carbon cycle, and indirectly lead to CO_2 emission, or or alternatively it may remain in solution and be transported to the ocean, potentially resulting in sedimentary re-burial. In this case, peat-derived DIC exported from fens would represent a carbon loss from the peatland, but not a source of CO_2 emissions to the atmosphere.

1.4. Aims and scope

Here, we provide an overview of current understanding of the role of waterborne carbon within the peatland carbon balance, with particular emphasis on the effects of peat drainage on these fluxes, and the consequent implications for the carbon and GHG balance of drained organic soils. The review makes reference to the recent IPCC 2013 Supplement to the 2006 Guidelines for National Greenhouse Gas Inventories: Wetlands (IPCC, 2014a), describing the treatment of waterborne carbon within this assessment, and expanding on the scientific rationale for the methods adopted. We consider the different chemical forms that comprise the carbon flux in peatland runoff; the available evidence regarding the influence of drainage (and associated management) on the flux of each of these forms; and the extent to which we are able to determine their eventual fate in terms of GHG emissions. In accordance with UNFCCC and Kyoto Protocol reporting frameworks, we focus on GHG emissions associated with land-management activities, rather than those associated with natural peatland processes. In each case we identify the main sources of GHG emissions, describe a method for including these fluxes in GHG accounting methods, and provide 'Tier 1' estimates of key

emissions sources based on current knowledge. Finally, we critically assess the overall importance of waterborne carbon in the GHG balance of drained peatlands, and identify current research gaps, areas of greatest uncertainty, and issues requiring further methodological development.

2. Methods

2.1. Literature collation

Literature data were collated to support the quantification of two key waterborne carbon fluxes, namely CH_4 emissions from drainage ditches and DOC fluxes from natural and drained peatlands. Whilst this assessment falls short of a full systematic review, we attempted to capture as many relevant publications as possible through a combination of literature searches (using Web of Science and Google Scholar), searches of grey literature, and direct contact with other researchers.

2.2. Estimation of on-site methane emissions from drainage ditches

Studies reporting CH_4 emissions from drainage ditches were categorised by a combination of peat and land-use type, with the level of stratification reflecting the (generally low) number of available studies. We classified temperate and boreal peatlands into five categories – drained blanket bog, drained forest, peat extraction, low-intensity and high-intensity grassland – but could only derive a single value for drained tropical peatlands (incorporating agriculturally cleared and plantation sites) due to data limitations. For temperate and tropical sites, measurements were generally made throughout the year, or on a subset of representative sampling dates, so that reasonable estimates of annual mean fluxes could be obtained. For the majority of boreal studies, fluxes were measured during the growing season only. In these cases, we estimated the mean annual flux as the mean instantaneous flux reported, multiplied by the length of the growing season as reported by the study authors, or implied by the duration of the measurements. This approach assumes negligible CH_4 emissions during the dormant period, which may be realistic if ditches are frozen, but could lead to an under-estimation of annual emissions (e.g. Minkinnen and Laine, 2006; Strack and Zuback, 2013).

To estimate mean emissions per category, we applied two approaches. First, we took account of the number of 'sites' within each study (where 'site' was taken to represent a discrete peat unit, rather than multiple measured ditches within a single peat unit), in order to calculate a weighted mean emission (i.e. each 'site' was treated as a single observation in the calculation of the mean). However, because some studies reported mean values for a set of measurement sites, rather than individual values, a true standard error could not be derived using this approach. For this reason, we also calculated mean emission values where the mean reported by each study was treated as a single observation, such that a standard error (of n studies) could be calculated. The implications and limitations of this approach are discussed below.

The overall CH_4 flux from drained peatlands can be expressed as:

$$EF_{CH_4(TOTAL)} = EF_{CH_4(DITCH)} \bullet Frac_{DITCH} + EF_{CH_4(LAND)} \bullet (1 - Frac_{DITCH}) \quad (1)$$

Where $EF_{CH_4(TOTAL)}$ is the 'emission factor' for CH_4 for the drained peatland , $EF_{CH_4(DITCH)}$ is the emission expressed per unit area of ditch surface, and $EF_{CH_4(LAND)}$ is the emission per unit area of the peat surface between ditches (all expressed in this study in g CH_4 m^{-2} yr^{-1}). $Frac_{DITCH}$ represents the proportion of the peatland area occupied by ditches, effectively a function of ditch spacing, configuration and width. Indicative values for this parameter were obtained for each aggregated land-use category as the mean of reported values from each publication. For grasslands, we

excluded studies from the Netherlands with exceptionally high ditch areas (up to 25%) as we considered these to be atypical of drained grasslands elsewhere.

2.3. Estimation of off-site CO_2 emissions from DOC and POC

2.3.1. Estimation of baseline DOC fluxes from near-natural sites

Estimates of aerial mean solute fluxes from peatlands are prone to significant errors, due to the difficulty of accurately defining catchment areas, especially in low-relief systems and in fens with complex subsurface hydrology. In drained peatlands this problem is exacerbated by the artificially modified drainage pathways and often very small hydrological units created by ditching. Where the water flux is uncertain, and in some cases physically implausible (i.e. where estimated runoff either exceeds precipitation or is implausibly low by comparison), the resulting DOC fluxes are also uncertain (e.g. Wilson et al., 2011; Worrall et al., 2011). Therefore, we took the approach of first deriving a set of catchment-scale measurements of DOC flux from near-natural peatlands, in order to quantify baseline rates of DOC export from unmanaged systems. Temperature and rainfall data were also collated for all sites where values were reported, or available for that location from other sources. The peatland type at each site (bog, fen, mixed mire, blanket bog, swamp forest) was also recorded, as a factor potentially contributing to between-site differences.

2.3.2. Estimation of DOC response to drainage

To evaluate responses of DOC leaching to drainage, we collated published data from paired studies of drained and undrained sites. Given large observed spatial variations in DOC fluxes among natural sites, a paired approach was considered more likely to provide a detectable signal than a simple collation of reported fluxes from all drained and undrained sites. Studies were included if they were based on direct field measurements of surface waters or peat pore waters. We also accepted field-based, vegetated mesocosm experimental studies with a minimum duration of a year, but excluded laboratory experiments, short-term (seasonal) field manipulations and studies which compared dry with wet years at a single site. Where experimental studies included more than one level of drainage intensity, we used data from the more intensive drainage treatments.

Given the additional issues highlighted above in relation to the estimation of DOC fluxes from small hydrologic units in drained peatlands, we found relatively few paired studies with reliable comparisons of DOC fluxes. On the other hand, a reasonable number of paired studies have reported mean DOC concentrations. In wetter and cooler areas, where precipitation is much higher than evapotranspiration, it is likely that peatland drainage has relatively little impact on the water flux. In this case, proportional changes in DOC concentration should provide a reasonable proxy for proportional changes in DOC flux. On the other hand, drainage of peatlands in warmer areas (notably in the tropics, and especially when also associated with changes in vegetation cover) may lead to large shifts in the peatland water balance (e.g. Moore et al., 2013). In these instances concentration changes are likely to be a poor indicator of flux changes. Differences in concentration responses and flux responses to drainage could also arise in drier continental mire systems, despite cool conditions (e.g. Rantakari et al., 2010). Therefore we gave precedence to flux measurements where available, but relied on concentration data where flux data were unavailable. Additional information on the extraction of data from individual published studies is provided in the Supplementary Material.

2.3.3. Estimation of total off-site CO_2 emissions associated with DOC export

An estimate of the CO_2 flux associated with DOC export from drained peatlands was obtained using the following equation (IPCC, 2014):

$$EF_{DOC} = DOC_{FLUX\_NATURAL} \bullet (1 + \Delta DOC_{DRAINAGE}) \bullet Frac_{DOC-CO_2} \quad (2)$$

Where EF_{DOC} is the 'emission factor' for DOC (expressed as the annual carbon flux per unit area, in this case as g CO_2-C m^{-2} yr^{-1}); $DOC_{FLUX\_NATURAL}$ is the flux of DOC from a natural, undrained peatland (g CO_2-C m^{-2} yr^{-1}); $\Delta DOC_{DRAINAGE}$ is the proportional increase in DOC flux from drained sites relative to un-drained sites, and $Frac_{DOC\text{-}CO_2}$ represents the fraction of DOC exported which is ultimately converted to CO_2, following export from the site. A similar ratio-based approach was used to quantify DOC responses to the harvesting of forested peatlands, relative to a reference level, by Schelker et al. (2014).

The value of $Frac_{DOC\text{-}CO_2}$ is fundamental to the calculation of CO_2 emissions associated with peatland DOC export, and thus to the potential contribution of DOC to GHG emissions following peatland drainage. Estimating this value is challenging, however, given the long continuum of potential locations (headwaters to the oceans) and timescales (hours to years) over which DOC processing (physico-chemical and biological) may occur. In the absence of a previous, systematic study of the fate of peat-derived DOC through this aquatic continuum, we reviewed a range of available literature spanning headwaters, lakes, large rivers systems and oceans, from which we derived initial estimates of this parameter.

2.3.4. Estimation of total off-site CO_2 emissions associated with POC export

For POC, a conceptually similar approach was followed to that for DOC, however published data were insufficient to allow estimates of the natural flux or average drainage responses to be derived. Instead, we took a previously published response function describing POC fluxes as a function of bare peat area (Evans et al., 2014a) and used this to estimate drainage responses by assuming that drainage channels were equivalent to exposed bare peat. An estimate of the fraction of POC converted to CO_2 ($Frac_{POC\text{-}CO_2}$) was derived from published literature on organic matter processing in a range of environments, as for DOC.

3. Results and Discussion

3.1. On-site methane emissions from ditches

We identified surprisingly few studies reporting GHG fluxes from drainage ditches. In total, just 19 different studies reported CH_4 flux data for drained sites (Table S1). Some of these studies reported fluxes from more than one land-use category, and a number reported fluxes for multiple sites within a category, giving a total of 69 individual sites (defined as measurements from separate peat units). In a small number of cases, different studies reported fluxes measured at different times from the same general peatland area; in these cases, the values were treated as independent. The geographical and typological distribution of sites was highly uneven; of the 67 flux values collated, over half were collected from the Netherlands, many from the single large-scale assessment of Vermaat et al. (2011). Similarly, over half of all measured values were derived from drained low- and high-intensity grasslands (mostly from the Netherlands), although forestry-drained peatlands and extraction sites were also reasonably well-represented (12 and 10 sites respectively). In contrast, only three values were obtained from drained mires (two from Cooper et al., 2014 and one from Huotari et al., 2014) and three from tropical peatlands (all from Jauhianen and Silvennoinen, 2012). Data from drained mires and forests were merged in the analysis, but tropical peatlands were retained as a separate category despite the small number of data points.

Mean CH_4 fluxes from drainage ditches are shown by land-use category, expressed per unit ditch surface area (Figure 1a) and per unit peatland area (Figure 1b); source data are given in the Supplementary Table 1. Because standard errors could only be calculated on a 'per-study' basis, Figure 1 shows both a simple mean of the mean values from each study, and a weighted mean taking account of the number of individual sites within each study. In the absence of individual reported values from some multiple-site studies, it was only possible to estimate confidence intervals for the 'mean of means' value. Differences between the two values were only evident for the high-intensity grassland category, where the weighted mean is 38% lower than the unweighted mean. This occurred because one study reporting data from 16 sites (Vermaat et al., 2011) gave much lower emissions compared to two studies reporting data from single sites (Teh et al., 2011; Chistotin et al., 2006).

Whilst recognising the considerable limitations and uncertainties associated with this comparatively small dataset, a number of observations can be made. Firstly, almost all studies where comparisons were made recorded higher CH_4 emissions from drainage ditches than from adjacent peatlands (e.g. Roulet and Moore, 1995; Sundh et al., 2000; Minkinnen and Laine, 2006; Schrier-Uijl et al., 2010; Teh et al., 2011). In some cases ditches were found to act as the source of most or all CH_4 emissions from the site. The fluxes are also in some cases considerably higher than those reported from undrained peatlands. For example, Roulet et al. (2007) and Nilsson et al. (2008) reported mean CH_4 emissions from intact boreal mires of 5 and 15 g CH_4 m^{-2} yr^{-1} respectively, whereas mean ditch emissions exceeded 30 g CH_4 m^{-2} yr^{-1} for all peat/land-use categories other than drained blanket bog (Figure 1).

A second observation from the collated dataset for temperate and boreal peatlands is that mean ditch CH_4 emissions increase with intensity of land-use, from drained blanket bogs and forestry-drained peatlands to peat extraction sites and grasslands. Although the highest emissions were observed from ditches in tropical peatlands, the very small number of data points for tropical sites precludes any assessment of land-use effects on emissions in this region. Similarly, we were unable to locate any measurements from temperate and boreal peatlands under arable cultivation. However the generally higher CH_4 emissions from ditches in peatlands converted to grassland could indicate that site fertility, together with increased levels of labile organic matter transport into ditches (notably from livestock) lead to increased rates of ditch CH_4 emission. In this respect, it is worth noting that different authors have reached different conclusions about the likely source of CH_4 emissions from ditches. Roulet and Moore (1995) noted that ditch CH_4 emissions could derive from methanogenesis within the peat itself, followed by the lateral transport of this CH_4 in groundwater seepage and subsequent evasion from the ditch, or alternatively from *in situ* CH_4 production within the ditch sediments. For a forestry-drained peatland, Minkinnen and Laine (2006) concluded that the first of these mechanisms was more important. However authors working in more intensive grassland systems have generally attributed high fluxes to *in situ* production. As part of a seasonal survey, Schrier-Uijl et al. (2011) found that high CH_4 emissions coincided with nutrient-enriched ditch sediments, and suggested that reducing nutrient and labile substrate inputs to ditches could reduce CH_4 production. Vermaat et al. (2011) noted that a large proportion of total emissions occurred through ebullition, again implying that ditch sediments were acting as the main CH_4 source. Other factors likely to influence CH_4 emission rates include water flow rates (e.g. Minkinnen and Laine, 2006) and the presence of emergent vegetation (Vermaat et al., 2011).

As described in Equation 1, the significance of ditches as a source of CH_4 emissions depends not only on the emissions per unit surface area of ditch, but also on the proportion of the peatland area occupied by ditches, $Frac_{DITCH}$. This area varies considerably as a function of both peat type and local land-use. Amongst the studies collated, reported values of $Frac_{DITCH}$ ranged from 0.02 in drained tropical peatlands (Jauhianen and Silvennoinen, 2012) and 0.025 in forestry drained mires (e.g.

Roulet and Moore, 1995; Minkinnen and Laine, 2006; Sirin et al., 2013) to over 0.2 in some grasslands (Van den Pol-Van Dasselaar, 1999; Schrier-Uijl et al., 2010; Vermaat et al., 2011), although such high values appear unusual outside the Netherlands, and indicative values of 0.05 were suggested for grasslands and extraction sites in the IPCC Wetland Supplement (IPPC, 2014a). Differences in drainage ditch density between categories reflect differences in the characteristics of the peat; for example, fibrous tropical peatlands can be effectively drained by a few large but widely spaced canals, whereas highly humified blanket bogs require a much higher density of smaller ditches to effectively lower water tables (Evans et al., 2014b). Grasslands in some flat former fen landscapes, such as the Netherlands, tend to contain large areas of open water. Figure 1b shows estimated landscape-scale emission of ditches per land-use category based on the collated studies, and associated values of $Frac_{DITCH}$, giving a range from 0.1 g CH_4 m^{-2} yr^{-1} in blanket bogs to 5.7 g CH_4 m^{-2} yr^{-1} in intensive grassland systems. Landscape-scale fluxes from drainage ditches in grassland and drained tropical peatlands are thus of a similar order to those from undrained peatlands, and may exceed pre-drainage fluxes at sites, such as those in the Netherlands, where ditches occupy a greater proportion of the landscape. The importance of ditch spacing was also noted by Roulet and Moore (1995), who observed that if ditches in an afforested bog were less than ~35m apart, the net effect of drainage would be to increase CH_4 emissions relative to undrained conditions.

Finally, it is worth noting that the contribution of drainage ditches to CH_4 emissions may not cease after re-wetting, as many forms of re-wetting leave either an active ditch system (now used to maintain high water tables) or a relic network of blocked of infilled channels. We could not obtain sufficient data to derive estimates of mean emissions from ditches in re-wetted peatlands, as this flux will vary according to the antecedent land-use, and the form of re-wetting undertaken. At a set of Dutch sites returned to semi-natural reed and sedge fen, Vermaat et al. (2011) measured lower CH_4 emissions from ditches compared to sites under grassland. On the other hand Waddington and Day (2007) measured higher (albeit highly variable) CH_4 emissions from ditches in a re-wetted peat extraction site compared to an adjacent cutover area, and Cooper et al. (2014) measured very high emissions from an infilled ditch on blanket bog recolonised by *Eriophorum vaginatum*, but comparatively low fluxes from areas remaining under bare peat or colonised by *Sphagnum*. Thus both the type of site restoration and the resulting plant community seem likely to determine CH_4 emissions from ditches in re-wetted landscapes. The IPCC Wetland Supplement (IPCC, 2014) did not provide guidance to account for ditch emissions in re-wetted peatlands, but our analysis suggests that this emission may continue to exceed that from the re-wetted peat mass itself, potentially leading to higher overall CH_4 emissions (at least temporarily following re-wetting) than would have occurred before the site was drained (Cooper et al., 2014).

3.2. On-site emissions of other GHGs from drainage channels

We did not collate sufficient data on fluxes of either CO_2 or N_2O from drainage ditches to allow average estimates to be obtained. However, those studies that have reported data on these fluxes from ditches have tended to indicate that these fluxes are not significantly different from those in the adjacent drained peat. For example both Teh et al. (2011) and Hyvönen et al. (2013) found that ditch CO_2 and N_2O emissions were lower per unit surface area than from the terrestrial peat surface, for a drained intensive grassland and a former extraction site under a bioenergy crop, respectively. Sundh et al. (2000) recorded similar aerial CO_2 emissions from ditches and terrestrial areas across a range of active extraction sites, and Best and Jacobs (1997) measured lower emissions from ditches at grassland sites. Vermaat et al. (2011) measured substantial CO_2 emissions (circa 300 g C m^{-2} yr^{-1}) from ditches in reed/sedge fen and low-intensity grassland, but slight net CO_2 uptake by ditches in high-intensity grassland.

A number of studies have measured CO_2 evasion from areas of open water in natural peatlands, with fluxes ranging from 1.5 g C m^{-2} yr^{-1} in two Swedish boreal peatlands (Wallin et al., 2010) and 4.8 g C m^{-2} yr^{-1} in a Canadian peatland (Dinsmore et al., 2009) to 12.7 g C m^{-2} yr^{-1} in a Scottish blanket peatland (Dinsmore et al., 2010). The only comparative CO_2 evasion estimates for drained and undrained peatlands (based on dissolved CO_2 concentrations rather than direct evasion measurements) were made by Rantakari et al. (2010). Their results give mean CO_2 evasion rates of 12 g C m^{-2} yr^{-1} for four undrained peatlands, and 24 g C m^{-2} yr^{-1} for four drained peatlands, although the latter was influenced by a very high value for one site. The authors attributed higher fluxes to extensive ditch networks and higher discharge rates in drained peatlands. However, most evasion studies suggest that stream CO_2 emissions are highest from steeper channels with turbulent water flow (e.g. Wallin et al., 2010; Billett and Harvey, 2013), which could explain lower direct measurements of CO_2 emissions from drainage ditches in the studies described above. Comparing two drained peatlands under grassland, Renou-Wilson et al. (2014) recorded mean excess CO_2 in drainage water of 4.3 g C m^{-2} yr^{-1} in a nutrient rich site, and 16 g C m^{-2} yr^{-1} in a nutrient-poor site, the latter representing around 60% of the total fluvial C flux during a dry year.

In summary, it is possible that ditches may act as sources of both CO_2 and N_2O under some circumstances, but we did not find evidence that ditches act as 'hotspots' for emission of these GHGs to the same extent as has been observed for CH_4. If, as suggested by some of the studies cited above, CO_2 and N_2O fluxes from ditches are similar to those from the adjacent drained peat surface, then a single 'emission factor' applied to the whole peatland area may be sufficient for reporting. However, more data are clearly required in order to establish whether ditches can act as significant CO_2 or N_2O sources under some circumstances; for example, there are few N_2O flux measurements from ditches draining fertilised peatlands.

While there is clear evidence that CO_2 evasion can represent a significant flux in natural peatlands, it has not yet been quantified for a sufficient number of drained sites to allow this to be included in GHG accounting. Furthermore, dissolved CO_2 in peatland drainage waters may not be degassed until turbulent water flow occurs downstream of the peatland area, in which case it could be considered an 'off-site' emission). This could also lead to some risk of double-counting, if some of the measured CO_2 flux is derived from breakdown of DOC or POC in the water column.

3.3. Off-site CO_2 emissions from DOC

3.3.1 DOC fluxes from natural peatlands

We collated data from 27 published estimates of DOC flux from near-natural peatlands, ranging from a subarctic fen to tropical peat swamps (Supplementary Table 2). Annual precipitation and mean temperature data were collated from the publications, or from nearby meteorological records, wherever possible. Results indicate that the natural DOC flux from peatlands varies by over an order of magnitude. Fluxes followed a clear latitudinal and climatic gradient, from a minimum of 5 g C m^{-2} yr^{-1} in subarctic and some boreal peatlands, to a maximum of around 60 g C m^{-2} yr^{-1} in tropical peatlands. Fluxes were moderately correlated with mean annual rainfall (R^2 = 0.47, $p < 0.001$), and strongly correlated with mean annual temperature (R^2 = 0.82, $p < 0.001$) (Figure 2). The correlation with precipitation is weakest for temperate blanket bogs, which have a similarly high precipitation to tropical peat swamps, but much lower mean annual temperatures. This suggests that temperature may be the main fundamental control on natural peatland DOC fluxes, although we note that the cluster of data points from tropical sites exert a strong leverage on the regression. Some previous studies have noted lower DOC concentrations in water draining natural fens compared to bogs (e.g. Glaser et al. 1981; Pastor et al., 2003), although this was not particularly evident in the data collated

here possibly due to the inclusion of a number of poor fen (e.g. Strack et al., 2008; Nilsson et al., 2008) and mixed mire systems in the dataset (Supplementary Table 2).

3.3.2 Effects of peatland drainage and rewetting on DOC fluxes

We identified a total of 15 published studies which provided sufficient data to calculate ratios of either DOC concentration or DOC flux between comparable drained and un-drained peat sites (Supplementary Table 3). Three studies reported paired data for more than one peat or land-use type, giving a total of 19 values of $\Delta DOC_{DRAINAGE}$. Although this dataset includes measurements from boreal and temperate raised bogs and fens, blanket bogs, and tropical peats, and drainage for both peat extraction and land-use change to agriculture, the number of studies is insufficient to draw any clear conclusions with regard to peat-type, land-use or inter-regional patterns. Furthermore, the range of water types analysed (including porewaters, ditches and streams; see Supplementary Table 3), as well as study design, duration (i.e. length of study, and whether seasonal or annual measurements were made), and levels of land-use/water table disturbance, add considerable uncertainty to the analysis and interpretation of results. Despite this, we observed a surprising degree of consistency in the direction and magnitude of observed DOC responses to drainage (Figure 3a). Taking concentration data alone, the mean increase in DOC for drained versus undrained sites was 48% (n = 17, 95% confidence interval (CI) 29-67%). Taking the smaller dataset of flux comparisons gave a mean increase of 67% (n = 5, 95% CI 42-91%). Merging the two datasets (i.e. giving precedence to flux data in studies reporting both measures) gave a mean increase of 62% (n = 17, 95% CI 48-76%). We therefore conclude that a $\Delta DOC_{DRAINAGE}$ value of around 0.6 represents a reasonable default for estimating DOC exports and associated CO_2 emissions from drained peatlands.

It is notable that the only two studies reporting negative or negligible changes in DOC concentration (Pastor et al., 2003; Moore et al., 2013) both showed large increases in DOC flux, indicating that increases in water flux at these sites (dry boreal mire and tropical peat swamp respectively) were the primary driver of increased DOC export. Similarly, at the dry, high-altitude Chinese peatland studied by Lou et al. (2014), observed DOC flux increases exceeded observed DOC concentration increases in drained field mesocosms, as water discharge also increased. Joensuu et al. (2002) suggest that increased runoff is a general response to ditching of boreal peatlands, unless offset by increased tree growth and subsequent evapotranspiration. If this 'amplifying' effect of increased water fluxes as well as increased DOC concentrations is common to other dry continental peatlands, a higher value of $\Delta DOC_{DRAINAGE}$ may be more appropriate than the default obtained here based partly on concentration changes. On the other hand, only one of the studies collated (Wallage et al., 2006) was based on a blanket bog, and additional data are needed to establish whether the fairly high default value obtained largely from continental peatlands is applicable to these more oceanic, high-rainfall systems, where it would imply a large absolute increase in DOC flux.

We identified just nine studies reporting DOC responses to peat re-wetting, of which seven reported concentration changes and four reported flux changes (Figure 3b, Supplementary Table 4). In a number of cases the estimation of DOC responses to re-wetting was quite convoluted (for example based on deviations from predicted DOC based on control sites, rather than absolute changes) and this adds uncertainty to the interpretation of results. Furthermore, we excluded several additional studies due to methodological issues (see Supplementary Material). Different treatment of some of these data sources could, given the small overall number of studies, lead to a different overall interpretation. However, we did find reasonably consistent evidence for an inhibitory effect of re-wetting on rates of DOC loss, albeit with considerable variability between studies. On average, re-wetting reduced DOC concentrations by 25% (n = 7, 95% CI -6% to -45%), and DOC fluxes by 31% (n = 4, 95% CI -12% to -50%). As was the case for drainage, the three studies which reported both

concentration and flux data (which comprised one study where DOC concentration increased, and two where it marginally decreased) all gave larger decreases in DOC flux. Merging the two sets of measurements and giving precedence to flux data where available, as above, gave a mean DOC change of -37% (n = 9, 95% CI -24% to -50%). This is close to the inverse of the mean 62% increase in DOC observed in response to drainage, suggesting that DOC increases due to drainage may be largely reversible through re-wetting.

Whilst these findings appear quite consistent, a number of caveats apply. Firstly, a number of studies have shown that DOC losses actually increase in the immediate post re-wetting period (e.g. Worrall et al., 2007; Zak and Gelbrecht, 2007). Secondly, studies which report decreases in DOC fluxes but not concentrations require that water yields also declined after re-wetting. This hydrologic response has clearly been demonstrated at the Bois-de-Bel site in Canada (Waddington et al., 2008), but is harder to explain at high rainfall blanket bogs, where evapotranspiration accounts for only a small fraction of precipitation, giving limited scope for peatland management to alter overall water balances. In two studies that have shown reduced DOC fluxes in small drained catchments (Gibson et al., 2009; Turner et al., 2014) the authors noted the likelihood that reduced water fluxes down the blocked drains were being offset by increased water flows elsewhere (e.g. within the peat or over the surface). Depending on the DOC concentration of this 'leakage' flow, overall DOC losses might not necessarily change as much as the results included in our analysis, based on small drain 'catchments', would suggest.

Additional complexity in the response of peatlands to drainage and re-wetting activities is likely to be associated with related land-use activities. In a study of a highly modified German peatland, Frank et al. (2014) measured much higher porewater DOC concentrations under high-intensity grassland (196 mg l^{-1}) compared to low-intensity grassland (89 mg l^{-1}). Similarly, Renou-Wilson et al. (2014) measured DOC fluxes of 38 g C m^{-2} yr^{-1} from a nutrient rich drained grassland site in Ireland, compared to 18 g C m^{-2} yr^{-1} from a nutrient-poor site. These results suggest that land-use intensity (e.g. deeper drainage, fertiliser use or presence of livestock) could further increase rates of DOC loss. Forest management may also influence DOC losses; Baker et al. (2008) measured higher DOC concentrations in a drained forest stream compared to an adjacent drained moorland stream, and a number of studies have observed elevated DOC after forestry operations in peaty catchments (e.g. Nieminen, 2004; Schelker et al., 2012; Nieminen et al., 2014). Within semi-natural peatlands, Clay et al. (2012) measured higher DOC concentrations in an eroded compared to a natural peat catchment, and some studies have suggested that moorland burning may enhance DOC loss rates (e.g. Holden et al., 2012 and references therein). All of these results suggest that increasing intensity of peatland disturbance will tend to increase waterborne DOC losses.

Finally, peat type may significantly alter the nature of DOC response to drainage. For fen peats in particular, which receive water via lateral inputs, drainage may actually reduce water fluxes through the peat by hydrologically isolating former groundwater-fed areas, altering abiotic conditions such as acidity and nutrient status (e.g. Laiho, 2006) with uncertain overall consequences for DOC production. Given the inherent difficulties of quantifying net DOC exports from systems receiving lateral water inputs, our conclusions are therefore considered tentative in relation to fens. If drains in any peatland type intercept permeable underlying mineral soils, increased DOC retention onto mineral surfaces may actually lead to a net reduction in waterborne DOC losses (Åström et al., 2001; Joensuu et al., 2001); this situation represents a clear exception to the general pattern observed here.

3.3.3 Contribution of peat DOC fluxes to CO_2 emissions

The analysis above provides preliminary estimates of $DOC_{FLUX\_NATURAL}$ and $\Delta DOC_{DRAINAGE}$. The conversion of these estimates into CO_2 emissions requires an estimate of the proportion of exported DOC converted to CO_2, $Frac_{DOC\text{-}CO_2}$. As described in Section 2.3.3., the long spatial and temporal continuum over which DOC breakdown can occur through the river-lake-estuary-ocean system makes definitive quantification of this parameter virtually impossible. Ultimately, however, the major sinks for DOC exported into the aquatic system are either conversion to CO_2, or transfer into lake or marine sediments, where low decomposition rates allow carbon to accumulate (e.g. Tranvik et al., 2009). The balance of these two sinks effectively determines the contribution of DOC export to CO_2 emissions. Whilst the DOC exported from peatlands tends to be biologically unreactive, recent work has emphasised the importance of photochemical processes in river systems (Cory et al., 2013), and a growing number of studies indicate that peat-derived DOC is particularly susceptible to photodegradation. Photochemical processing may also make the residual DOC more susceptible to biological utilisation (e.g. Fasching and Battin, 2012). Köhler et al. (2002), Moody et al. (2013) and Jones et al. (this volume) all showed very high rates of DOC loss in samples from peat streams exposed to light, with average DOC removal ranging from 33% to 75% over periods of up to 10 days. Since much of this degradation occurs within the first 48 hours, this could be sufficient to convert most peat-derived DOC to CO_2 before it enters the sea. Dawson et al. (2001) estimated that 12-18% of DOC was removed within a 2 km peat stream reach, and Moody et al. (2013) estimated that 50-70% of all DOC would be mineralised within the residence time of an 800 km^2 UK river system. Jonsson et al. (2007) estimated that 45% of all terrestrially-derived organic carbon was mineralised and evaded as CO_2 within a 3000 km^2 mixed boreal catchment, with sedimentation negligible and the remainder exported to the sea. For a 6400 km^2 lake-rich catchment in the Northern United States, Buffam et al. (2011) estimated that 33% of terrestrial C inputs to the aquatic system were degassed as CO_2, 2% as CH_4, 26% accumulated in sediments and 40% transported downstream. For a set of 21 large Swedish boreal catchments, Algesten et al. (2003) estimated that 50% of all organic carbon entering the aquatic system was removed, with CO_2 degassing accounting for 90% of removal and sedimentation for 10%. In a global analysis, Gudasz et al. (2010) estimated a 20% average 'burial efficiency' of total organic carbon in lake sediments.

Globally, Tranvik et al. (2009) estimated that 48% of terrestrial C input was evaded to the atmosphere as CO_2, 21% buried in sediments, and 31% exported to the ocean. These values incorporate multiple forms of aquatic carbon (i.e. DOC, POC and DIC) and multiple sources, and it is thus difficult to relate these results specifically to peat-derived DOC. If it is assumed that POC is relatively susceptible to sedimentation; that a large amount of weathering-derived DIC will be transported through freshwaters to the ocean as unreactive HCO_3^- or CO_3^{2-}; and that peat-derived DOC has an above-average susceptibility to photo-degradation, then the rate of peat DOC to CO_2 evasion should considerably exceed this global mean. Finally, the processing of terrestrial organic matter does not end with its transfer from freshwater to marine systems. Some DOC may precipitate out at the freshwater-seawater interface (e.g. Sholkovitch et al., 1978), although studies of DOC removal in estuaries receiving water from peat catchments provide conflicting evidence of the importance of this process (e.g. Álvarez-Selgado and Miller, 1998; Spencer et al., 2007; Palmer et al., this volume) making it difficult to generalise. For terrestrial DOC that does reach the ocean, it appears that (although specific values for peat-derived DOC are not available) most DOC is microbially processed, on a timescale of years to decades (e.g. Bianchi, 2011; Opsahl and Benner, 1997). Burdige (2005) estimated that sedimentation accounted for only around 15-30% of all (dissolved and particulate) terrestrial organic matter inputs to the ocean, with the remainder mineralised to CO_2. Schlunz and Schneider (2000) provided an even lower estimate, of 10%.

Taken together, these observations suggest: i) that peat-derived DOC can be very rapidly photodegraded in headwater systems; ii) that CO_2 emissions consistently exceed sedimentary C burial in large peaty catchments where both fluxes have been quantified, even where lakes are

present; iii) that flocculation in estuaries is probably a minor sink for DOC, and iv) that much more of the DOC reaching the ocean is mineralised to CO_2 than is buried in marine sediments. On this basis, we conclude that a very high overall fraction of DOC exported from peatlands will be converted to CO_2, across a range of locations, and over a range of timescales from hours to years. This assessment provided the basis for assigning a default value of 0.9 ($\pm$ 0.1) for $Frac_{DOC\text{-}CO_2}$ in the IPCC Wetland Supplement (IPCC, 2014a), implying that the global warming potential of DOC export from peatlands may approach that of CO_2 on a carbon equivalent basis. Applying this value, along with those for $DOC_{FLUX\_NATURAL}$ and $\Delta DOC_{DRAINAGE}$ in Equation 1, yields estimates of CO_2 emissions due to DOC export of 0.41 (95% CI 0.22 – 0.67) t CO_2-eq ha^{-1} yr^{-1} in drained boreal peatlands, 1.05 (95% CI 0.65 – 1.57) t CO_2-eq ha^{-1} yr^{-1} in drained temperate peatlands, and 2.77 (95% CI 1.87 – 3.92) t CO_2-eq ha^{-1} yr^{-1} in drained tropical peatlands.

3.4 Off-site CO_2 emissions from POC

3.4.1. Effects of peatland drainage on POC fluxes

Although waterborne POC losses from eroding peatlands can be very large, there is less evidence that POC losses due to peatland drainage alone are likely to generate large POC fluxes, or therefore large GHG emissions. A simple empirically-based linear relationship between POC and exposed peat surface area developed for UK blanket bogs (Evans et al., 2014a) suggests that each 1% of bare peat (as a proportion of total peat area) will increase POC losses by 4 g C m^{-2} yr^{-1}, with a 35% uncertainty range. For a typical $Frac_{DITCH}$ value of 0.05, this generates a POC flux of 20 ($\pm$7) g C m^{-2} yr^{-1}. Moore et al. (2013) measured an increase in POC flux from 1 to 10 g C m^{-2} yr^{-1} for drained versus undrained tropical peatlands with a somewhat lower ditch density, broadly in line with these estimates. However, activities associated with drainage that lead to additional peat exposure, such as peat extraction, arable cultivation and forest operations may substantially increase this flux (e.g. Marttila and Kløve, 2008; 2010). Similarly, the development of erosional features such as gullies and peat pipes following drainage has been shown to increase POC losses from blanket bogs (Holden, 2006). Additional data are needed to quantify POC losses as a function of peatland drainage and management more generally.

3.4.2 Contribution of peat POC fluxes to CO_2 emissions

Quantifying the fate of POC, in terms of its contribution to off-site CO_2 emissions, is a further area of high uncertainty. As a product of physical erosion it may be less reactive than DOC, and carbon accumulation rates in water bodies downstream of actively eroding peatlands indicate a high rate of sedimentary re-burial in some cases (e.g. Yeloff et al., 2005, and references therein). As POC tends to be mobilised during high flow events, a significant proportion is likely to be re-deposited on downstream floodplains; Walling et al. (1998) gave estimates of 39-49% for sediment deposition to floodplains for two large UK river catchments (although this was not specifically for organic sediments). Although floodplain deposition could represent a return to stable storage, re-deposited material is likely to be more exposed to aerobic decomposition than it was in the peatland, and Goulsbra et al. (2013) estimated that around 80% of floodplain-deposited POC will be oxidised over a 30 year period. For POC remaining in the aquatic system, physico-chemical and biological breakdown processes may also lead to oxidation. Worrall et al. (2014) estimated that 23% of POC in UK river systems is mineralised before reaching the estuary, but did not provide an estimate of burial rates in freshwater sediments. These burial rates are likely to be higher in lake-dominated boreal regions than in UK rivers, therefore we estimated freshwater POC burial at 20% of riverine input (range 10-30%) based on values reported by Gudasz et al. (2010), Algesten et al. (2003), Buffam et al. (2011) and Tranvik et al. (2009). For estuaries, we took the estimate of Worrall et al. (2014), based on a previous analysis by Tappin et al. (2003), that 45% of POC reaching estuaries is mineralised before it

reaches the sea, and 4% buried in estuarine sediments. For POC reaching the ocean, the 15-30% burial efficiency for terrestrial organic matter from Burdige (2005) potentially provides an under-estimate, as POC is more likely to be sedimented out than DOC. Combining this estimate with a global estimate by Meybeck (1982) that 40% of terrestrial organic matter enters the ocean as POC, and assuming that DOC is preferentially oxidised, yields a range for ocean POC oxidation of 44 $\pm$ 19%. This is consistent with the estimate of Bianchi (2011) that 50% of terrestrial POC inputs are remineralised. Assigning arbitrary 25% ranges on values where uncertainties were not specified (reflecting also our reliance on data that were not specific to peat-derived POC in many cases), the combination of these yields an overall $Frac_{POC\text{-}CO_2}$ of 0.70, with a range of 0.49 to 0.91 (Figure 4).

We recognise that there is huge uncertainty in these calculations, which are based on a combination of small-scale studies at individual peatland sites, and global budget calculations that are not peat-specific. Nevertheless, it appears reasonable to reject the 'null hypothesis' that POC exported from peatlands is chemically inert, and thus to infer that POC losses contribute to GHG emissions from drained peatlands. It is also worth noting that the estimation of $Frac_{POC\text{-}CO_2}$ is relatively insensitive to the estimate of floodplain POC deposition, since estimates of burial efficiency on floodplains and in the aquatic system as a whole are fairly similar. Taking the estimates of POC loss for drained peatlands above would suggest that this contribution is minor, in the region of 0.1 to 0.5 t CO_2-eq ha^{-1} yr^{-1}. However in actively eroding peatlands, much larger emissions are possible, perhaps up to 5 t CO_2-eq ha^{-1} yr^{-1} for a fully exposed bare peat surface.

4. Synthesis and Conclusions

Estimates of GHG emissions from drained peatlands associated with waterborne carbon losses are summarised in Figure 5, for ditch CH_4 emissions, CO_2 evasion, DOC and POC export. Note that the estimates for CO_2 evasion are simply the mid-point and range of the reported values in Section 3.2, which are not specific to drained systems and thus provide only a crude indication of the likely order of magnitude of this flux. POC emissions should also be considered approximate, as they are based on the simple response function included in Evans et al. (2014a) rather than a comprehensive literature assessment.

Although the categorisation of GHG emissions by site type differs for CH_4 and DOC, as a function of the available data and observed controls on fluxes, we were able to produce estimates of overall GHG emissions from waterborne carbon for some illustrative land-use examples in Figure 6. This analysis suggests that total waterborne GHG emissions vary considerably according to site type, with fairly low emissions from forestry-drained boreal bog (circa 1 t CO_2-eq ha^{-1} yr^{-1}), but substantially higher emissions (3 to 4.5 t CO_2-eq ha^{-1} yr^{-1}) for the boreal peat extraction, temperate grassland and cleared tropical forest examples. For drained boreal forest, however the waterborne GHG flux is approximately equal to the sum of direct gaseous flux of CO_2, CH_4 and N_2O calculated from IPCC Tier 1 emission factors for this category (Figure 6b). For the other examples shown, direct gaseous emissions are considerably higher (10 to 20 t CO_2-eq ha^{-1} yr^{-1}), but waterborne emissions are nonetheless estimated to make up 30% of total GHG emissions from the boreal peat extraction example, and 17% of total emissions from both temperate grassland and cleared tropical forest examples. Additional data will be needed in order to provide more comprehensive and specific fluxes according to factors such as peat type (e.g. raised bog, blanket bog, fen, tropical swamp), climatic conditions, land-use and management intensity.

The relative importance of different components of the waterborne emission varies between the examples shown in Figure 6. For peat extraction, POC represents 65% of the total estimated flux. As noted above this estimate carries a high uncertainty, and actual loss rates may also depend on site management, such as measures to reduce peak flow rates (e.g. Marttila and Kløve, 2008). For the

intensive grassland, 46% of emissions are due to CH_4 emissions from ditches. This estimate assumes a $Frac_{DITCH}$ value of 0.05, so for the much higher values (up to 0.25) recorded in some Dutch peatlands, CH_4 emissions are likely to dominate the overall waterborne flux, and to make a major contribution to total GHG emissions. For the tropical peat example, DOC makes up 65% of the estimated waterborne flux, and has been estimated to comprise around 20% of total CO_2 emissions from drained tropical sites (Moore et al., 2013). However, it is possible that the CO_2 evasion flux has been under-estimated, as we used a range of values from temperate and boreal peatlands which may not be representative for these systems.

Overall, our analysis has highlighted the need for additional measurements of a range of key fluxes and processes contributing to GHG emissions from peatlands via fluvial pathways. Measurements of ditch CH_4 emissions were particularly scarce, and unevenly distributed, with high representation of grasslands sites in the Netherlands, and low representation of non-European (and especially tropical) sites. The number of studies from which data on drainage or re-wetting impacts on DOC could be obtained was insufficient to define different responses according to peat type or land-use type, and necessitated the use of both concentration and flux data, each of which carry considerable uncertainties. There is a particular need for long-term studies, and for year-round measurements to allow calculation of annual fluxes. Whilst recognising the limitations of the analysis presented, and the need for further work, we can nevertheless draw the following general conclusions:

1) That waterborne carbon makes a quantitatively significant contribution to total GHG emissions from drained peatlands, under all land-uses and in all climatic regions
2) That this flux comprises a number of distinct components, the relative importance of which will vary according to land-use, peat type and climate zone
3) That both dissolved and particulate organic carbon exported from peatlands appear to be 'climatically active', with the majority of both carbon forms likely to be oxidised to CO_2 within a range of aquatic environments and over a range of timescales.
4) That increases in waterborne carbon losses following drainage and land-use conversion are likely to be at least partly reversible through re-wetting and restoration
5) That CO_2 evasion rates are not well quantified for drained peatlands, or for tropical peatlands in general.
6) That significant additional work, including field measurements, experiments and process studies will be needed in order to provide more complete and robust estimates of GHG emissions associated with waterborne carbon fluxes.

In relation to the final point, we note that the importance of waterborne carbon is not restricted to peatlands; for example lateral carbon movements into and out of coastal wetlands are likely to be important components of their overall carbon balance. We also note that waterborne fluxes do not necessarily represent the only important pathway for lateral carbon loss; for example airborne losses of POC may be substantial from peatlands subject to erosion, fire, extraction or cultivation, but remain largely unquantified. Finally, we were not able to consider the potential role of aquatic DOC and POC as substrates for downstream CH_4 production. Methane emissions from reservoirs, treated as 'flooded lands' in IPCC terminology, were excluded from the Wetland Supplement (IPCC 2014a), but can act as major emission sources in some circumstances. Barros et al. (2011) identified DOC as a significant explanatory factor in observed rates of both CH_4 and CO_2 emission from a global hydroelectric reservoir dataset, and similar relationships have been observed in lakes (Sobek et al., 2009). Given our finding that peatland drainage tends to increase both DOC and POC supply to downstream waterbodies, there is potential for this to generate increased CH_4 emissions from these systems, effectively amplifying the global warming impact of waterborne carbon loss.

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References

Algesten G, Sobek S, Bergström AK, Ågren A, Tranvik L, Jansson M (2003) Role of lakes for organic carbon cycling in the boreal zone. Global Change Biol 10: 141-147.

Alm A, Shurpali N, Minkinnen K, Aro L, Hytönen J, Laurila T, Lohila A, Maljanen M, Martikainen PJ, Mäkiranta P, Penttilä T, Saarnio S, Silvan N, Tuitilla ES, Laine J (2008) Emission factors and their uncertainty for the exchange of CO_2, CH_4 and N_2O in Finnish managed peatlands. Boreal Env Res 12: 191-209.

Álvarez-Selgado XA, Miller AEJ (1998) Dissolved organic carbon in a large macrotidal estuary (the Humber, UK): Behaviour during estuarine mixing. Marine Pollut Bull 37: 216-224.

Armstrong, A., Holden, J., Kay, P., McDonald, A.T., Gledhill, S., Foulger, M., Walker, A., 2010. The impact of peatland drain-blocking on dissolved organic carbon loss and discolouration of water; results from a national survey. Journal of Hydrology, 381, 112-120.

Åström M, Aaltonen EK, Koivusaari (2001) Effect of ditching operations on stream-water chemistry in a boreal forested catchment. Sci Total Environ 279: 117-129.

Baker A, Bolton L, Newson M, Spencer RGM (2008). Spectrophotometric properties of surface water dissolved organic matter in an afforested upland peat catchment. Hydrol Process 22: 2325-2336.

Banaś K, Gos K (2004) Effect of peat-bog reclamation on the physico-chemical characteristics of ground water in peat. Polish Journal of Ecology, 52: 69-74

Barros N, Cole JJ, Tranvik LJ, Prairie YT, Bastviken D, Huszar Vl, del Giorgio P, Roland F (2011) Carbon emission from hydroelectric reservoirs linked to reservoir age and latitude. Nature Geosci 9: 593-596.

Battin TJ, Kaplin LA, Findlay S, Hopkinson CS, Marti E, Packman AI, Newbold DI, Sabater S (2008). Biophysical controls on organic carbon fluxes in fluvial networks. Nature Geosci 1: 95-100.

Best, E.P.H. and Jacobs, F.H.H. 1997. The influence of raised water table levels on carbon dioxide and methane production in ditch dissected peat grasslands in the Netherlands. Ecological Engineering 8: 129-144.

Bianchi TS (2011). The role of terrestrially derived organic carbon in the coastal ocean: A changing paradigm and the priming effect. Proceedings of the National Academy of Science 108: 19473–19481.

Billett MF, Palmer SM, Hope D, Deacon C, Storeton-West R, Hargreaves KJ, Flechard C, Fowler D (2004). Linking land-atmosphere-stream carbon fluxes in a lowland peatland system. Global Biogeochemical Cycling 18: GB1024.

Billett MF, Charman DJ, Clark JM, Evans CD, Evans MG, Ostle NJ, Worrall F, Burden A, Dinsmore KJ, Jones T, McNamara NP, Parry L, Rowson JG, Rose R (2010). Carbon balance of UK peatlands: current state of knowledge and future research challenges. Climate Research, 45, 13-29.

Billett MF, Harvey FH (2013) Measurements of CO_2 and CH_4 evasion from UK Peatland headwater streams. Biogeochemistry 114: 165-181.

Billett MF, Moore TR (2008) Supersaturation and evasion of CO_2 and CH_4 in surface waters at Mer Bleue peatland, Canada. Hydrol Process 22: 2044-2054.

Bridgham S, Megonigal JP, Keller JK, Bliss NB, Trettin C (2006) The carbon balance of North American wetlands. Wetlands 26: 889-916.

Buffam I, Turner MG, Desai AR, Hanson PC, Rusak JA, Lottig NR, Stanley EH, Carpenter SR (2011). Integrating aquatic and terrestrial components to construct a complete carbon budget for a north temperate lake district. Global Change Biol 17: 1193-1211.

Burdige DJ (2005) Burial of terrestrial organic matter in marine sediments: A re-assessment. Global Biogeochem Cycl 19, GB4011.

Byrne KA, Chojnicki B, Christensen TR, Droesler M, Freibauer A, Fribourg T, Frlking S, Lindroth A, Mailhammer J, Malmer N, Selin P, Turunen J, Valentini R & Zetterberg L (2004) EU peatlands: current carbon stocks and trace gas fluxes. Carbo Europe Discussion Paper for Concerted Action CarboEurope-GHG, Lund, Sweden, October 2003.

Chistotin M.V., Sirin A.A. and Dulov L.E. 2006. Seasonal dynamics of carbon dioxide and methane emission from a peatland in Moscow Region drained for peat extraction and agricultural use. Agrokhimija 6: 54–62.

Ciais P, Borges AV, Abril G, Meybeck M, Folberth G, Hauglustaine D, Janssen IA (2008). The impact of lateral carbon fluxes on the European carbon balance. Biogeosciences 5: 1259-1271.

Clark JM, Lane SN, Chapman PJ, Adamson JK (2007) Export of dissolved organic carbon from an upland peatland during storm events: Implications for flux estimates. J Hydrol 347: 438-447.

Clay GD, Dixon S, Evans MG, Rowson JG, Worrall F (2012) Carbon dioxide fluxes and DOC concentrations of eroding blanket peat gullies. Earth Surf. Process. Landforms 37: 562–571.

Cooper M, Evans CD, Zieliński P, Levy PE, Gray A, Peacock M, Fenner N, Freeman C (2014) Infilled ditches are hotspots of landscape methane flux following peatland restoration. Ecosystems 17: 1227-1241.

Cory RM, Ward CP, Crump BC, Kling GW (2014) Sunlight controls water column processing of carbon in arctic fresh waters. Science 345: 925-928.

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

Dawson JJC, Bakewell T, Billett MF (2001). Is in-stream processing an important control on spatial changes in carbon fluxes in headwater catchments? Science of the Total Environment 265: 153-167.

Dawson JJC, Billett MF, Hope D, Palmer SM, Deacon CM (2004). Sources and sinks of aquatic carbon in a peatland stream continuum. Biogeochemistry 70: 71–92.

Dinsmore KJ, Billett MF, Moore TR (2009) Transfer of carbon dioxide and methane through the soil-water-atmosphere system at Mer Bleue peatland, Canada. Hydrol Process 23: 330-341.

Dinsmore KJ, Billett MF, Skiba UM, Rees RM, Drewer J, Helfter C (2010). Role of the aquatic pathway in the carbon and greenhouse gas budgets of a peatland catchment. Global Change Biol. 16: 2750-2762.

Evans CD, Jones TG, Burden A, Ostle N, Zieliński P, Cooper MDA, Peacock M, Clark JM, Oulehle F, Cooper D, Freeman C (2012). Acidity controls on dissolved organic carbon mobility in organic soils. Global Change Biology, 18: 3317–3331.

Evans CD, Bonn A, Holden J, Reed M, Evans M, Worrall F, Couwenberg J, Parnell M (2014a). Relationships between anthropogenic pressures and ecosystem functions in UK blanket bogs: Linking process understanding to ecosystem service valuation. Ecosystem Services 9: 5-19.

Evans CD, Page SE, Jones T, Moore S, Gauci V, Laiho R, Hruška J, Allott TEH, Billett MF, Tipping E, Freeman C, Garnett MH (2014b) Contrasting vulnerability of drained tropical and high-latitude peatlands to fluvial loss of stored carbon. Global Biogeochem Cycles doi 10.1002/2013GB004782.

Fasching C, Battin T (2012) Exposure of dissolved organic matter to UV-radiation increases bacterial growth efficiency in a clear-water Alpine stream and its adjacent groundwater. Aquat Sci 74: 143-153.

Fiedler S, Höll BS, Freibauer A, Stahr K, Drösler M, Schloter M, Jungkunst HF (2008). Particulate organic carbon (POC) in relation to other pore water carbon fractions in drained and rewetted fens in Southern Germany. Biogeosciences, 5: 1615–1623.

Frank S, Tiemeyer B, Gelbrecht J, Freibauer A (2014). High soil solution carbon and nitrogen concentrations in a drained Atlantic bog are reduced to natural levels by 10 years of rewetting. Biogeosciences, 11, 2309-2324.

Frolking S, Roulet N, Fuglestvedt (2006) How northern peatlands influence the Earth's radiative budget: Sustained methane emission versus sustained carbon sequestration. J Geophys Res 111: G01008.

Gibson HS, Worrall F, Burt TP, Adamson JK (2009). DOC budgets of drained peat catchments: implications for DOC production in peat soils. Hydrological Processes, 23: 1901-1911.

Gielen B, Neirynck J, Luyssaert S, Janssens IA (2011) The importance of dissolved organic carbon fluxes for the carbon balance of a temperate Scots pine forest. Agric Forest Meteorol 151: 270-278.

Glagolev MV, Chistotin MV, Shnyrev NA, Sirin AA (2008). The emission of carbon dioxide and methane from drained peatlands changed by economic use and from natural mires during the summer-fall period (on example of a region of Tomsk oblast). Agrokhimija 5: 46-58.

Glaser PH, Wheeler GA, Gorham E, Wright HE (1981). The patterned mires of the Red Lake peatland., Northern Minnesota: Vegetation, water chemistry and landforms. Journal of Ecology, 69: 575-599.

Glatzel S, Kalbitz K, Dalva M, Moore T (2003). Dissolved organic matter properties and their relationship to carbon dioxide efflux from restored peat bogs. Geoderma 113, 397-411.

Goulsbra CS, Evans MG, Allott TEH (in prep.). Using laboratory experiments to investigate the potential for transformation of POC to dissolved and gaseous forms. Aquat Sci (this issue)

Gudasz C, Bastviken D, Steger K, Premke K, Sobek S, Tranvik LJ (2010). Temperature-controlled organic carbon mineralization in lake sediments. Nature 466: 478–481.

Heikkinen, K., 1990. Transport of organic and inorganic matter in river, brook and peat mining water in the drainage basin of the River Kiiminkijoki. Aqua Fennica, 20, 143-155.

Holden J (2006) Sediment and particulate carbon removal by pipe erosion increase over time in blanket peatlands as a consequence of land drainage. J Geophys Res 111: F02010.

Holden J, Chapman PJ, Palmer SM, Kay R, Grayson R (2012) The impacts of prescribed moorland burning on water colour and dissolved organic carbon: A critical synthesis. J Environ Manage 101: 92-103.

Hope D, Palmer SM, Billett MF, Dawson JJC (2001). Carbon dioxide and methane evasion from a temperate peatland stream. Limnol Oceanogr 46: 847-857.

Hooijer A, Page S, Canadell JG, Silvius M, Kwadijk J, Wösten H, Jauhianen (2010) Current and future CO_2 emissions from drained peatlands in Southeast Asia. Biogeosciences, 7: 1505–1514.

Huotari J, Nykänen H, Forsius M, Arvola L (2013) Effect of catchment characteristics on aquatic carbon export from a boreal catchment and its importance in regional carbon cycling. Global Change Biol 19: 3607-3620.

Hyvönen NP, Huttunen JT, Narasinha NJ, Lind SE, Marushchak ME, Heitto L, Martikainen PJ (2013) The role of drainage ditches in greenhouse gas emissions and surface leaching losses from a cutaway peatland cultivated with a perennial bioenergy crop. Boreal Env Res 18: 109-126.

IPCC (2006). 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K (eds). Intergovernmental Panel on Climate Change, IGES, Japan.

IPCC (2014a). 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. Hiraishi T, Krug T, Tanabe K, Srivastava N, Baasansuren J, Fukuda M, Troxler TG (eds). Intergovernmental Panel on Climate Change, Switzerland.

IPCC (2014b) Chapter 11, IPCC Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change .Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Jauhiainen J, Hooijer A, Page SE (2012) Carbon dioxide emissions from an Acacia plantation on peatland in Sumatra, Indonesia. Biogeosciences 9, 617-630.

Jauhiainen J, Silvennoinen H (2012). Diffusion GHG fluxes at tropical peatland drainage canal water surfaces. Suo 63, 93-105.

Joensuu S, Ahti E, Vuollekoski M (2001) Long-term effects of maintaining ditch networks on runoff water quality. Suo 52: 17-28.

Joensuu S, Ahti E, Vuollekoski M (2002) Effects of ditch network maintenance on the chemistry of run-off water from peatland forests. Scand J For Res 17: 238-247

Jones TG, Evans CD, Jones D, Hill PW (in prep.) Transformation and loss of peat-derived DOC by solar radiation and biofilm uptake: evidence from a 14C isotope tracer study. Aquat Sci (this issue)

Jonsson A, Algesten G, Bergström AK, Bishop K, Sobek S, Tranvik LJ, Jansson M (2007). Integrating aquatic carbon fluxes in a boreal catchment carbon budget. J. Hydrol. 334: 141-150.

Joosten H, Clarke D (Eds) (2002). Wise use of mires and peatlands, background and principles including a framework for decision-making. International Mire Conservation Group and International Peat Society, Finland.

Juutinen S, Väliranta M, Kuutti V, Laine AM, Virtanen T, Seppä H, Weckström J, Tuitilla ES (2013). Short-term and long-term carbon dynamics in a northern peatland-stream-lake continuum: A catchment approach. J Geophys Res: Biogeosci 118: 171-183.

Kane ES, Turetsky MR, Harden JW, McGuire AD, Waddington JM (2010). Seasonal ice and hydrologic controls on dissolved organic carbon and nitrogen concentrations in a boreal-rich fen. Journal of Geophysical Research 115: G04012.

Kindler R et al. (2011). Dissolved carbon leaching from soil is a crucial component of the net ecosystem carbon balance. Global Change Biol 17: 1167-1185.

Koehler AK, Murphy K, Kiely G, Sottocornola M (2009). Seasonal variation of DOC concentration and annual loss of DOC from an Atlantic blanket bog in South Western Ireland. Biogeochemistry 95: 231–242.

Koehler AK, Sottocornola M, Kiely G (2011). How strong is the current carbon sequestration of an Atlantic blanket bog? Global Change Biol 17: 309–319.

Köhler S, Buffam I, Jonsson A, Bishop K (2002) Photochemical and microbial processing of stream and soil water dissolved organic matter in a boreal forested catchment in northern Sweden. Aquat Sci 64: 1-13.

Laiho R (2006) Decomposition in peatlands: Reconciling seemingly contrasting results on the impacts of lowered water levels. Soil Biol Biochem 38, 2011-2024.

Marttila H, Kløve B (2008). Erosion and delivery of deposited peat sediment. Water Resour Res 44, W06406.

Marttila H, Kløve B (2010). Dynamics of erosion and suspended sediment transport from drained peatland forestry. J Hydrol 388: 414-425.

Meybeck M (1982) Carbon, nitrogen and phosphorus transport by the world's rivers. Amer J Sci 282: 401-450.

Moody CS, Worrall F, Evans CD, Jones TJ (2013). The rate of loss of dissolved organic carbon (DOC) through a catchment. J Hydrol 492: 139-150.

Minkkinen K, Laine J (2006) Vegetation heterogeneity and ditches create spatial variability in methane fluxes from peatlands drained for forestry. Plant and Soil: 289–304.

Moore S, Evans CD, Page SE, Garnett MG, Jones TG, Freeman C, Hooijer A, Wiltshire A, Limin S, Gauci V (2013) Deep instability of deforested tropical peatlands revealed by fluvial organic carbon fluxes. Nature 493: 660-664.

Moore TR, Clarkson BR (2007). Dissolved organic carbon in New Zealand peatlands. New Zealand J. Marine Freshwater Res. 41: 137-141.Moore, T.R., Matos, L., Roulet, N.T. 2003. Dynamics and chemistry of dissolved organic carbon in Precambrian Shield catchments and an impounded wetland. Can J Fish Aquat Sci 60: 612-623.

Nieminen M (2004) Export of dissolved organic carbon, nitrogen and phosphorus following clear-cutting of three Norway spruce forests growing on drained peatlands in southern Finland. Silva Fennica 38: 123–132.

Nieminen M, Koskinen M, Sarkkola S, Laurén A, Kaila A, Kiikkilä O, Nieminen TM, Ukonmaanaho L (2015) Dissolved organic carbon export from harvested peatland forests with differing site characteristics. Water Air Soil Pollut 226: 181.
Nilsson M. Sagerfors J, Buffam I, Laudon H, Eriksson T, Grelle A, Klemedtsson L, Weslien P, Lindroth A (2008). Contemporary carbon accumulation in a boreal oligotrophic minerogenic mire - a significant sink after accounting for all C-fluxes. Global Change Biol 14: 2317–2332.

Opsahl S, Benner R (1997) Distribution and cycling of terrigenous dissolved organic matter in the ocean. Nature 386: 480-482.

Page SE, Rieley JO, Banks CJ (2011) Global and regional importance of the tropical peatland carbon pool. Global Change Biol 17: 798-818.

Palmer SM. Evans CD, Chapman PJ, Burden A, Jones TG, Allott TEH, Evans MG, Moody CS, Worrall F, Holden J (in prep.) Sporadic hotspots for physico-chemical retention of aquatic organic carbon: from peatland source to sea, Aquat Sci (this issue)

Pastor J, Solin J, Bridgham SD, Updegraff K, Harth C, Weishampel, Dewey B (2003) Global warming and the export of dissolved organic carbon from boreal peatlands. Oikos 100: 380-386.

Rantakari M, Mattsson T, Kortelainen P, Piirainen S, Finér L, Ahtiainen M (2010). Organic and inorganic carbon concentrations and fluxes from managed and unmanaged boreal first-order catchments. Sci Total Environ408: 1649–1658.

Renou-Wilson F, Barry C, Müller C, Wilson D (2014) The impacts of drainage, nutrient status and management practice on the full carbon balance of grasslands on organic soils in a maritime temperate zone. Biogeosciences 11(16): 4361-4379.

Repo ME, Huttunen JT, Naumov AV et al (2007) Release of CO_2 and CH_4 from small wetland lakes in western Siberia. Tellus 59:788–796.

Roulet NT, LaFleur PM, Richards PJ, Moore TR, Humphreys ER, Bubier J (2007) Contemporary carbon balance and late Holocene carbon accumulation in a northern peatland. Global Change Biol 13, 397-411.

Roulet NT, Moore TR (1995) The effect of forestry drainage practices on the emission of methane from northern peatlands. Can J For Res 25: 491-499.

Schelker J, Eklöf K, Bishop K, Laudon H (2012) Effects of forestry operations on dissolved organic carbon concentrations and export in boreal first-order streams. J Geophys Res 117: G01011.

Schelker J, Öhman K, Löfgren S, Laudon H (2014). Scaling of increased dissolved organic carbon inputs by forest clear-cutting – What arrives downstream? J Hydrol 508: 299-306.

Schlünz B, Schneider RR (2000). Transport of terrestrial organic carbon to the oceans by rivers: re-estimating flux- and burial rates. Int J Earth Sci 88: 599–606.

Schrier-Uijl AP, Kroon PS, Leffalaar PA, van Huissteden JC, Berendse F, Veenendal EM (2010) Methane emissions in two drained peat agro-ecosystems with high and low agricultural intensity. Plant Soil 329: 509-520

Schrier-Uijl AP, Veraart AJ, Leffelaar PJ, Berendse F, Veenendaal EM (2011) Release of CO_2 and CH_4 from lakes and drainage ditches in temperate wetlands. Biogeochemistry 102: 265–279.

Sholkovitch ER, Boyle EA, Price NB (1978) The removal of dissolved humic acids and iron during estuarine mixing. Earth Planet Sci Lett 40: 130-136.

Sirin AA, Suvorov GG, Chistotin MV, Glagolev MV (2012). Values of methane emission from drainage ditches. Environ Dynamics Climate Change 3: 1–10

Sobek S, Algesten G, Bergström AK, Jansson M, Tranvik LJ (2003) The catchment and climate regulation of pCO_2 in boreal lakes. Global Change Biol 9: 630-641.

Spencer RGM, Ahad JME, Baker A, Cowie GL, Ganeshram R, Upstill-Goddard RC, Uher G (2007) The estuarine mixing behaviour of peatland derived dissolved organic carbon and its relationship to chromophoric dissolved organic matter in two North Sea estuaries (U.K.). Estuarine Coastal Shelf Sci 74: 131-144

Strack M, Waddington JM, Bourbonniere RA, Buckton L, Shaw K, Whittington P, Price JS (2008) Effect of water table drawdown on peatland dissolved organic carbon export and dynamics. Hydrol Process, 22: 3373-3385.

Strack M, Zuback YCA (2013). Annual carbon balance of a peatland 10 yr following restoration. Biogeosciences 10: 2885–2896

Sundh I, Nilsson M, Mikkelä C, Granberg G, Svensson BH (2000) Fluxes of methane and carbon dioxide on peat-mining areas in Sweden, Ambio 29: 499–503.

Tappin AD, Harris JRW, Uncles RJ (2003) The fluxes and transformations of suspended particles, carbon and nitrogen in the Humber estuarine system (UK) from 1994 to 1996: results from an integrated observation and modelling study. Sci Total Environ 314: 665-713.

Teh YA, Silver WL, Sonnentag O, Detto M, Kelly M, Baldocchi DD (2011). Large greenhouse gas emissions from a temperate peatland pasture. Ecosystems 14: 311-325.

Tranvik LJ, Downing JA, Cotner JB *et al*. (2009) Lakes and reservoirs as regulators of carbon cycling and climate. Limnol Oceanogr 54: 2298–2314.

Turner EK, Worrall F, Burt TP (2013). The effect of drain blocking on the dissolved organic carbon (DOC) budget of an upland peat catchment in the UK. J Hydrol 479: 169-179.

Urban NR, Bayley SE, Eisenreich SJ (1989) Export of dissolved organic carbon and acidity from peatlands. Water Resour Res 25: 1619-1628.

Urbanová Z, Picek T, Bárta J (2011). Effect of peat re-wetting on carbon and nutrient fluxes, greenhouse gas production and diversity of methanogenic archaeal community. Ecol Engin 37: 1017-1026.

van den Pol-van Dasselaar A, van Beusichem ML, Oenema O (1999) Methane emissions from wet Grasslands on peat soil in a nature preserve. Biogeochemistry 44: 205–220.

Vermaat JE, Hellmann F, Dias ATC, Hoorens B, van Logtestijn RSP, Aerts R (2011). Greenhouse gas fluxes from Dutch peatland water bodies: Importance of the surrounding landscape. Wetlands 31: 493-498.

Waddington JM, Tóth K, Bourbonniere R (2008) Dissolved organic carbon export from a cutover and restored peatland. Hydrol Process 22: 2215-2224.

Waddington JM, Day SM (2007) Methane emissions from a peatland following restoration. J Geophys Res 112, G03018.

Wallage ZE, Holden J, McDonald AT (2006) Drain blocking: An effective treatment for reducing dissolved organic carbon loss and water discolouration in a drained peatland. Sci Total Environ 367: 811-821.

Wallin M, Buffam I, Öquist M, Bishop K (2010) Temporal and spatial variability of dissolved inorganic carbon in a boreal stream network: Concentrations and downstream fluxes. J Geophys Res: Biogeosci 115: G02014.

Wallin M, Grabs T, Buffam I, Laudon H, Ågren A, Oquist G, Bishop K (2013). Evasion of CO_2 from streams - the dominant component of the carbon export through the aquatic conduit in a boreal landscape. Global Change Biology 19: 785-797.

Walling DE, Owens PN, Leeks GJL (1998). The role of channel and floodplain storage in the suspended sediment budget of the River Ouse, Yorkshire, UK. Geomorphology 22: 225-242.

Waltham T (2000). Peat subsidence at the Holme Post. Mercian Geol. 15: 49-51.

Wilson L, Wilson J, Holden J, Johnstone I, Armstrong A, Morris M (2011). Ditch blocking, water chemistry and organic carbon flux: Evidence that blanket bog restoration reduces erosion and fluvial carbon loss. Sci Total Environ 409: 2010-2018.

Worrall F, Reed M, Warburton J, Burt T (2003) Carbon budget for a British upland peat catchment. Sci Total Envirob 312: 133-146.

Worrall F, Armstrong A, Holden J (2007) Short-term impact of peat drain-blocking on water colour, dissolved organic carbon concentration, and water table depth - post drainage DOC flux. J Hydrol 337: 315-325.

Worrall F, Rowson JG, Evans MG, Pawson R, Daniels S, Bonn A (2011) Carbon fluxes from eroding peatlands – the carbon benefit of revegetation following wildfire. Earth Surf Process Landf 36: 1487–1498.

Worrall F, Burt TP, Howden NJK (2014) The fluvial flux of particulate organic matter from the UK: Quantifying in-stream losses and carbon sinks. J Hydrol 519: 611-625.

Yeloff DE, Labadz JC, Hunt CO, Higgitt DL, Foster IDL (2005) Blanket peat erosion and sediment yield in an upland reservoir catchment in the southern Pennines, UK. Earth Surf Process Landforms 30, 717–733.

Yu ZC (2012) Northern peatland carbon stocks and dynamics: a review. Biogeosciences 9: 4071-4085.

Zak D, Gelbrecht J (2007) The mobilisation of phosphorus, organic carbon and ammonium in the initial stage of fen rewetting (a case study from NE Germany). Biogeochemistry 85: 141-151.

FIGURE CAPTIONS

Figure 1a) Mean drainage ditch methane emissions, expressed per unit ditch surface area, for a range of peat/land-use categories. White bars represent mean values weighted by the number of sites in each study (no error bars), grey bars represent a mean of reported values from each study. The number of sites/studies in each category is shown above the respective bars. Error bars show 95% confidence intervals for the per-study means only, and where n > 1; **b)** The same data expressed per unit peatland area, based on typical values of $Frac_{DITCH}$ for each category.

Figure 2. Relationships between reported DOC fluxes from near-natural peatlands and **a)** annual precipitation, **b)** mean annual temperature.

Figure 3. Estimated percentage change in DOC concentration and/or flux from a set of **a)** drained vs undrained comparison studies; **b)** re-wetted versus drained comparisons**.** Sites are labelled according to peat type and climate region; for the full list of data sources and calculations see Supplementary Material (Supplementary Table 3).

Figure 4. Schematic illustration of the fate of a unit mass of peat carbon lost as POC to the fluvial system. Bold underlined values represent best estimates of the fraction of carbon associated with each pathway, italic values beneath represent lower and upper estimates. For derivation of values and literature sources see text.

Figure 5. Collated estimates of GHG emissions, expressed as CO_2 equivalents, for a range of waterborne carbon fluxes. CH_4 emissions were converted to CO_2 equivalents using a 100 year Global Warming Potential of 25. For explanation of how estimates and error ranges were derived, see text.

Figure 6a) Estimated total waterborne GHG emissions for four illustrative peat type/land-use categories, based on the emission factors shown in Figure 4. Ditch CH_4 emissions were calculated for the 'default' $Frac_{DITCH}$ values given in Supplementary Table 1.; **b)** IPCC Tier 1 default emission factors for 'terrestrial' GHG emissions from comparable peat type/land-use categories (boreal nutrient poor drained forest, boreal peatland managed for extraction, temperate nutrient-rich shallow-drained grassland, tropical drained and cleared forest land) (IPCC, 2014a). Uncertainty ranges shown are calculated as the sum of 95% CI values for each individual flux.

Figure 1

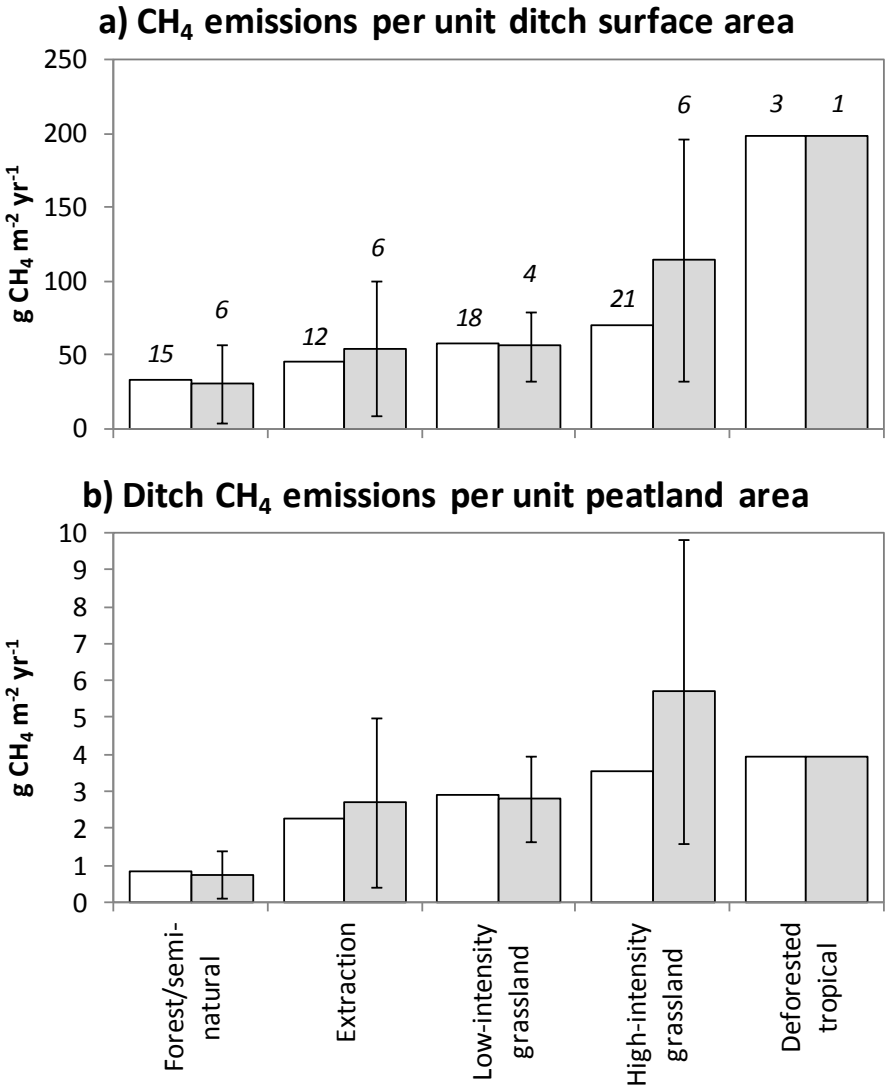

a) CH4 emissions per unit ditch surface area
g CH4 m-2 yr-1
250
200
150
100
50
0
15
6
12
6
18
4
21
6
3
1
b) Ditch CH4 emissions per unit peatland area
g CH4 m-2 yr-1
10
9
8
7
6
5
4
3
2
1
0
Forest/semi-natural
Extraction
Low-intensity grassland
High-intensity grassland
Deforested tropical

Figure 2

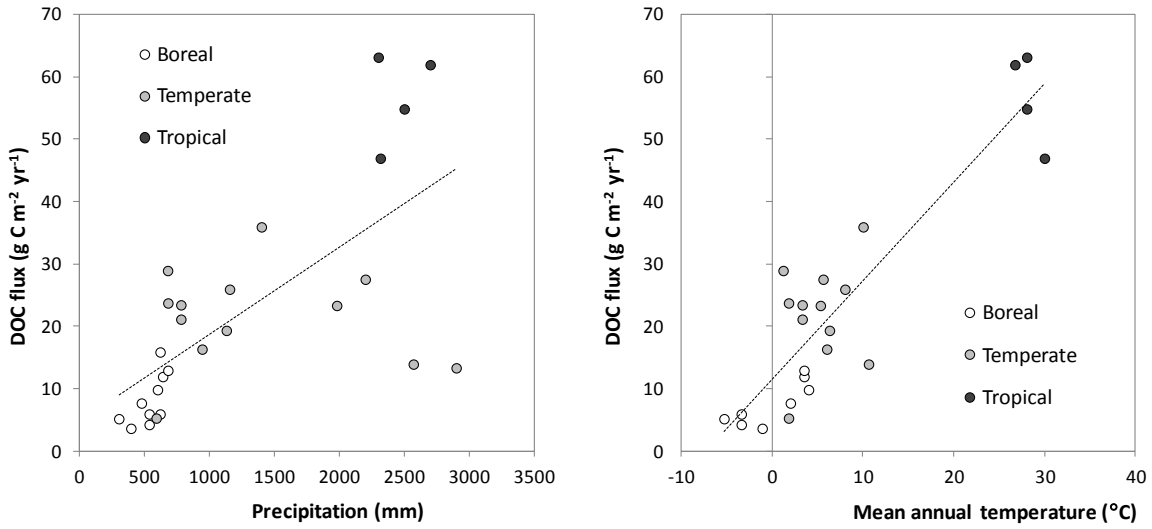

Boreal
Temperate
Tropical
DOC flux (g C m-2 yr-1)
Precipitation (mm)
0
10
20
30
40
50
60
70
0
500
1000
1500
2000
2500
3000
3500
Boreal
Temperate
Tropical
DOC flux (g C m-2 yr-1)
Mean annual temperature (°C)
-10
0
10
20
30
40

Figure 3

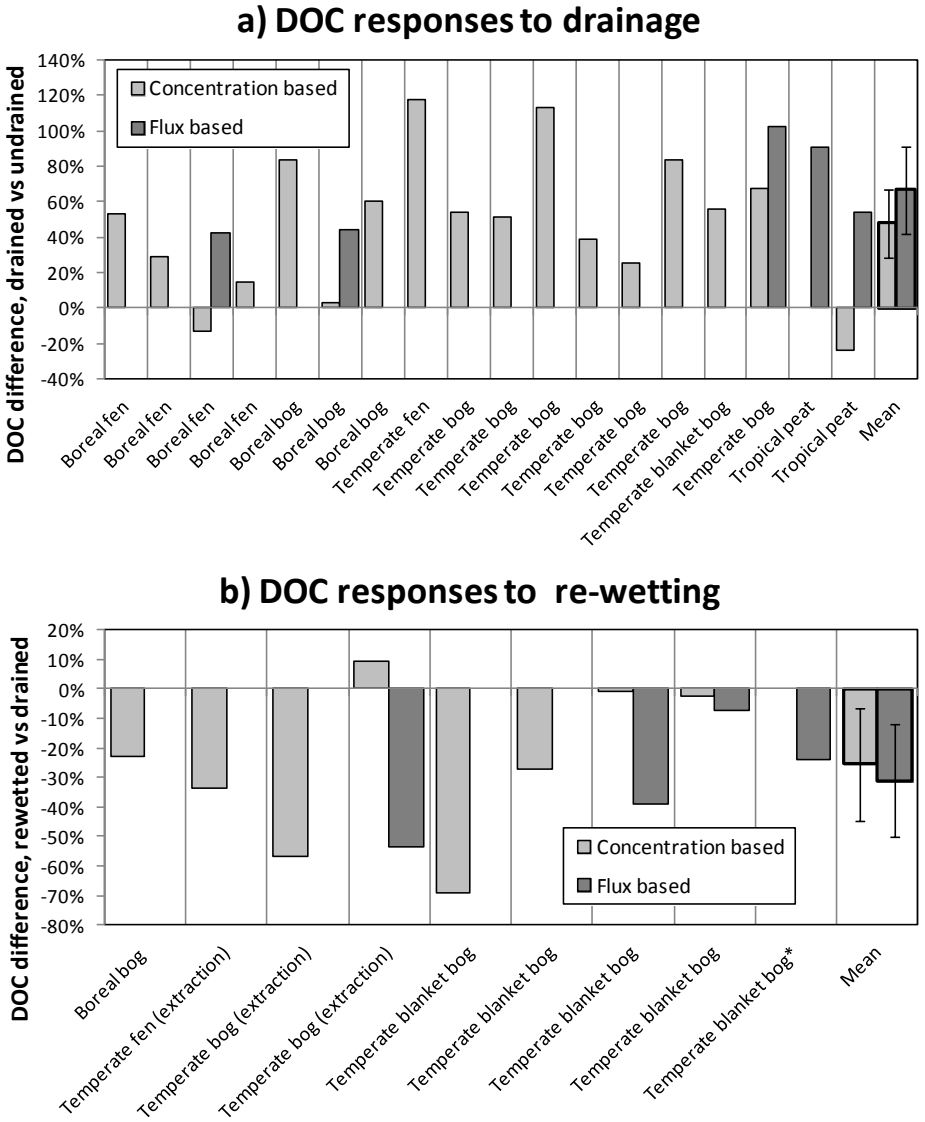

a) DOC responses to drainage
DOC difference, drained vs undrained
140%
120%
100%
80%
60%
40%
20%
0%
-20%
-40%
Concentration based
Flux based
Boreal fen
Boreal fen
Boreal fen
Boreal fen
Boreal bog
Boreal bog
Boreal bog
Temperate fen
Temperate bog
Temperate bog
Temperate bog
Temperate bog
Temperate bog
Temperate bog
Temperate blanket bog
Temperate bog
Tropical peat
Tropical peat
Mean
b) DOC responses to re-wetting
DOC difference, rewetted vs drained
20%
10%
0%
-10%
-20%
-30%
-40%
-50%
-60%
-70%
-80%
Concentration based
Flux based
Boreal bog
Temperate fen (extraction)
Temperate bog (extraction)
Temperate bog (extraction)
Temperate blanket bog
Temperate blanket bog
Temperate blanket bog
Temperate blanket bog
Temperate blanket bog*
Mean

Figure 4

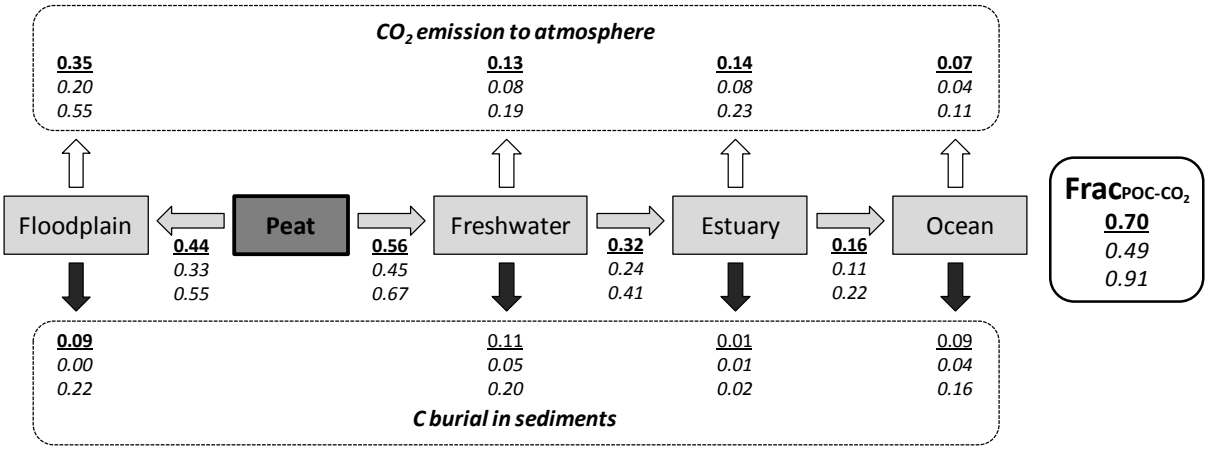

CO_2 emission to atmosphere
0.35
0.20
0.55
0.13
0.08
0.19
0.14
0.08
0.23
0.07
0.04
0.11
Floodplain
0.44
0.33
0.55
Peat
0.56
0.45
0.67
Freshwater
0.32
0.24
0.41
Estuary
0.16
0.11
0.22
Ocean
$Frac_{POC\text{-}CO_2}$
0.70
0.49
0.91
0.09
0.00
0.22
0.11
0.05
0.20
0.01
0.01
0.02
0.09
0.04
0.16
C burial in sediments

Figure 5

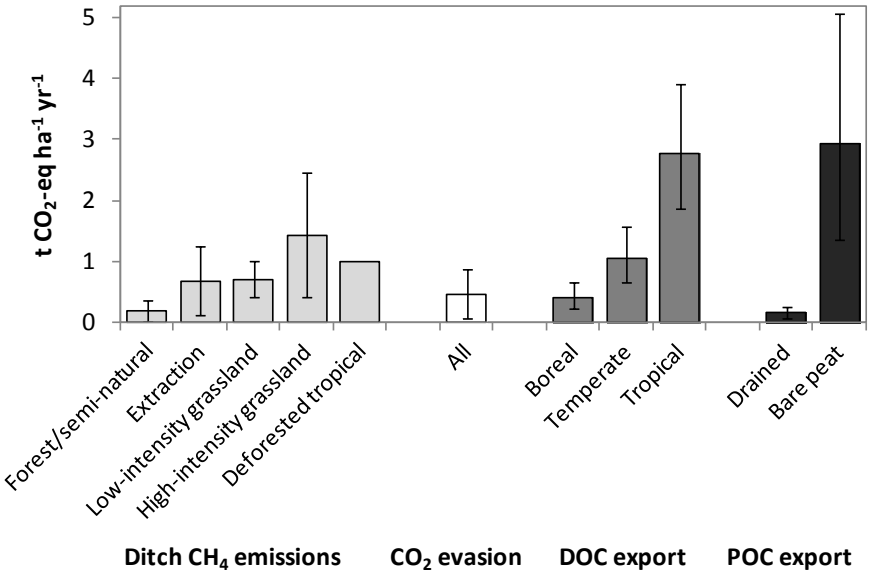
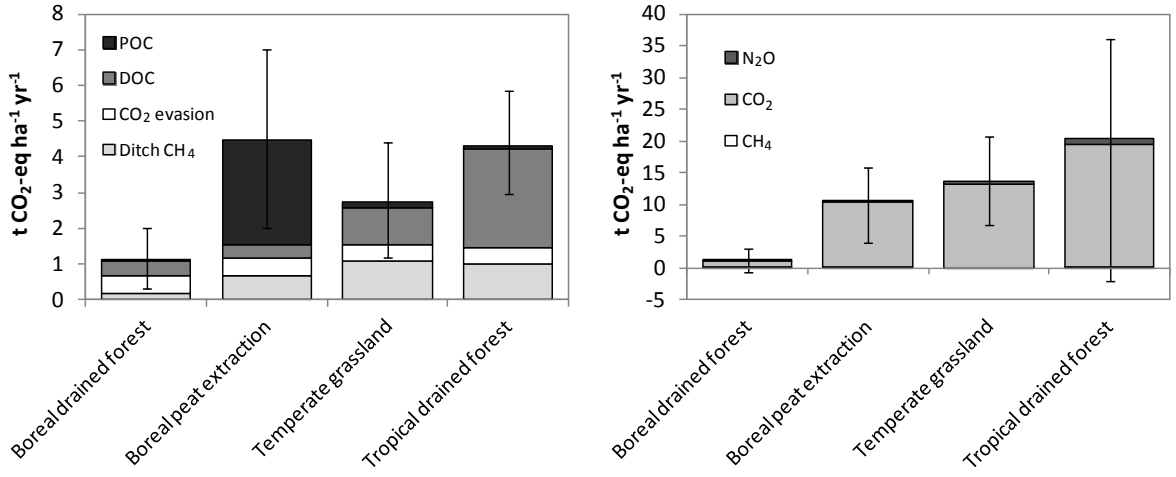

t CO2-eq ha-1 yr-1
5
4
3
2
1
0
Forest/semi-natural
Extraction
Low-intensity grassland
High-intensity grassland
Deforested tropical
All
Boreal
Temperate
Tropical
Drained
Bare peat
Ditch CH4 emissions
CO2 evasion
DOC export
POC export

Figure 6


t CO2-eq ha-1 yr-1
POC
DOC
CO2 evasion
Ditch CH4
Boreal drained forest
Boreal peat extraction
Temperate grassland
Tropical drained forest
t CO2-eq ha-1 yr-1
N2O
CO2
CH4
Boreal drained forest
Boreal peat extraction
Temperate grassland
Tropical drained forest