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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 CO₂) from drainage ditches located within the peatland; and ii) ‘off site’ emissions of CO₂ 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 CH₄ 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 CO₂ 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 CO₂. 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 CO₂-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 CO₂ for millennia after their initiation, whilst also acting as sources of methane (CH₄). 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 CH₄ emissions are large (e.g. Nilsson et al., 2008). However, as noted by Frolking et al. (2006), the long-term sequestration of CO₂ into stable organic matter gradually outweighs the warming effect of CH₄, 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 recently in 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 CO₂ emissions is significant; the recent IPCC Assessment Report (IPCC, 2014b) estimated total CO₂ emissions from drained peatlands of 1.1 to 1.5 Gt CO₂ 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 CH₄ emissions from the peatland surface to near-zero values; the IPCC Guidelines for Greenhouse Gas Inventories (IPCC, 2006) considered that CH₄ 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 balances under-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 CO₂ and CH₄, dissolved inorganic carbon (DIC, comprising bicarbonate, HCO₃⁻, and carbonate, CO₃²⁻), 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) CO₂ or CH₄ 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 CO₂ and CH₄ (and potentially also the non-carbon GHG nitrous oxide, N₂O) 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 result 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 CO₂ or CH₄. Although dissolved CO₂ and CH₄ may also be transported into drainage systems, if present in excess of atmospheric concentrations they tend to rapidly evade,

until equilibrium is reached. Although CO₂ 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 CO₂) are generally negligible due to the low solubility of CO₂ at low pH. A different situation arises in fen peats, where high pH values are associated with the presence of HCO₃⁻ and (in the most alkaline systems) CO₃²⁻. 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, CO₂ produced through peat respiration under alkaline conditions may equilibrate with water to form HCO₃⁻, 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 CO₂ emission, 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 CO₂ 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 CH₄ 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 CH₄ 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 CH₄ 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 CH₄ 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 CH₄ 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 CH₄ 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 CO₂ 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 CO₂ emissions associated with DOC export

An estimate of the CO₂ 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 CO₂-C m⁻² yr⁻¹); $DOC_{FLUX_NATURAL}$ is the flux of DOC from a natural, undrained peatland (g CO₂-C m⁻² yr⁻¹); $\Delta DOC_{DRAINAGE}$ is the proportional increase in DOC flux from drained sites relative to un-drained sites, and $Frac_{DOC-CO_2}$ represents the fraction of DOC exported which is ultimately converted to CO₂, 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 $Frac_{DOC-CO_2}$ is fundamental to the calculation of CO₂ 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 CO₂ 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 CO₂ ($Frac_{POC-CO_2}$) 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 CH₄ 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 CH₄ 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 CH₄ 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 CH₄ 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 CH₄ emissions from intact boreal mires of 5 and 15 g CH₄ m⁻² yr⁻¹ respectively, whereas mean ditch emissions exceeded 30 g CH₄ 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 CH₄ 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 CH₄ 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 CH₄ emission. In this respect, it is worth noting that different authors have reached different conclusions about the likely source of CH₄ emissions from ditches. Roulet and Moore (1995) noted that ditch CH₄ emissions could derive from methanogenesis within the peat itself, followed by the lateral transport of this CH₄ in groundwater seepage and subsequent evasion from the ditch, or alternatively from *in situ* CH₄ 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 CH₄ emissions coincided with nutrient-enriched ditch sediments, and suggested that reducing nutrient and labile substrate inputs to ditches could reduce CH₄ 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 CH₄ source. Other factors likely to influence CH₄ 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 CH₄ 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, $\text{Frac}_{\text{DITCH}}$. This area varies considerably as a function of both peat type and local land-use. Amongst the studies collated, reported values of $\text{Frac}_{\text{DITCH}}$ 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 (IPCC, 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 F_{DITCH} , giving a range from 0.1 g CH₄ m⁻² yr⁻¹ in blanket bogs to 5.7 g CH₄ 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 CH₄ emissions relative to undrained conditions.

Finally, it is worth noting that the contribution of drainage ditches to CH₄ 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 or 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 CH₄ emissions from ditches compared to sites under grassland. On the other hand Waddington and Day (2007) measured higher (albeit highly variable) CH₄ 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 CH₄ 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 CH₄ 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 CO₂ or N₂O 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 CO₂ and N₂O 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 CO₂ 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 CO₂ emissions (circa 300 g C m⁻² yr⁻¹) from ditches in reed/sedge fen and low-intensity grassland, but slight net CO₂ uptake by ditches in high-intensity grassland.

A number of studies have measured CO₂ 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 CO₂ evasion estimates for drained and undrained peatlands (based on dissolved CO₂ concentrations rather than direct evasion measurements) were made by Rantakari et al. (2010). Their results give mean CO₂ 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 CO₂ 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 CO₂ emissions from drainage ditches in the studies described above. Comparing two drained peatlands under grassland, Renou-Wilson et al. (2014) recorded mean excess CO₂ 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 CO₂ and N₂O 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 CH₄. If, as suggested by some of the studies cited above, CO₂ and N₂O 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 CO₂ or N₂O sources under some circumstances; for example, there are few N₂O flux measurements from ditches draining fertilised peatlands.

While there is clear evidence that CO₂ 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 CO₂ 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 CO₂ flux is derived from breakdown of DOC or POC in the water column.

3.3. Off-site CO₂ 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 $\Delta\text{DOC}_{\text{DRAINAGE}}$. 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 $\Delta\text{DOC}_{\text{DRAINAGE}}$ 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 $\Delta\text{DOC}_{\text{DRAINAGE}}$ 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 $38 \text{ g C m}^{-2} \text{ yr}^{-1}$ from a nutrient rich drained grassland site in Ireland, compared to $18 \text{ g C m}^{-2} \text{ yr}^{-1}$ 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 CO₂ emissions

The analysis above provides preliminary estimates of $\text{DOC}_{\text{FLUX_NATURAL}}$ and $\Delta\text{DOC}_{\text{DRAINAGE}}$. The conversion of these estimates into CO_2 emissions requires an estimate of the proportion of exported DOC converted to CO_2 , $\text{Frac}_{\text{DOC-CO}_2}$. 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 km² 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 km² mixed boreal catchment, with sedimentation negligible and the remainder exported to the sea. For a 6400 km² 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, Gudas 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-Salgado 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 CO₂ 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 CO₂, 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 $\text{Frac}_{\text{DOC-CO}_2}$ in the IPCC Wetland Supplement (IPCC, 2014a), implying that the global warming potential of DOC export from peatlands may approach that of CO₂ on a carbon equivalent basis. Applying this value, along with those for $\text{DOC}_{\text{FLUX_NATURAL}}$ and $\Delta\text{DOC}_{\text{DRAINAGE}}$ in Equation 1, yields estimates of CO₂ emissions due to DOC export of 0.41 (95% CI 0.22 – 0.67) t CO₂-eq ha⁻¹ yr⁻¹ in drained boreal peatlands, 1.05 (95% CI 0.65 – 1.57) t CO₂-eq ha⁻¹ yr⁻¹ in drained temperate peatlands, and 2.77 (95% CI 1.87 – 3.92) t CO₂-eq ha⁻¹ yr⁻¹ in drained tropical peatlands.

3.4 Off-site CO₂ 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 $\text{Frac}_{\text{DITCH}}$ 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 CO₂ emissions

Quantifying the fate of POC, in terms of its contribution to off-site CO₂ 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 Gudas 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 underestimate, 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 $\text{Frac}_{\text{POC-CO}_2}$ 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 $\text{Frac}_{\text{POC-CO}_2}$ 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 $\text{CO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$. However in actively eroding peatlands, much larger emissions are possible, perhaps up to 5 t $\text{CO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$ 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 $\text{CO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$), but substantially higher emissions (3 to 4.5 t $\text{CO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$) 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 $\text{CO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$), 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 CH₄ emissions from ditches. This estimate assumes a $\text{Frac}_{\text{DITCH}}$ value of 0.05, so for the much higher values (up to 0.25) recorded in some Dutch peatlands, CH₄ 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 CO₂ emissions from drained tropical sites (Moore et al., 2013). However, it is possible that the CO₂ 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 CH₄ 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 CO₂ 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 CO₂ 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 CH₄ 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 CH₄ and CO₂ 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 CH₄ emissions from these systems, effectively amplifying the global warming impact of waterborne carbon loss.

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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 $\text{Frac}_{\text{DITCH}}$ 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' $\text{Frac}_{\text{DITCH}}$ 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

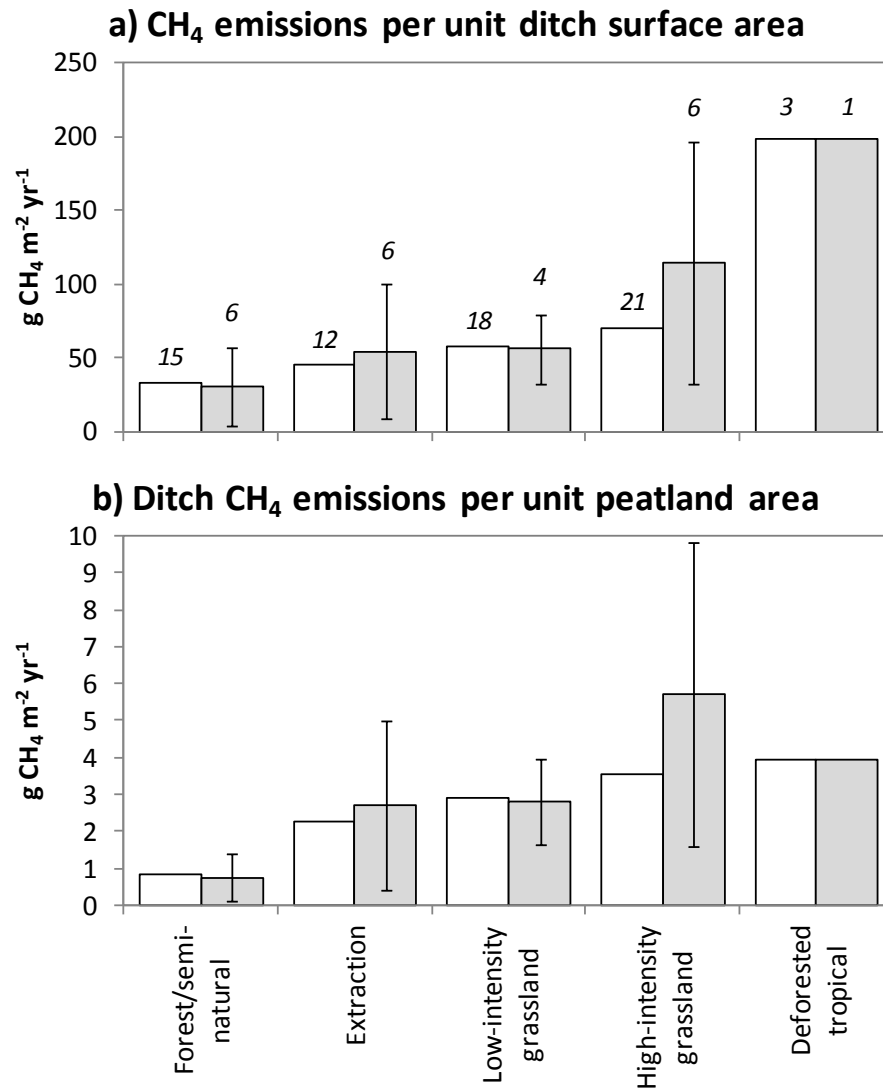


Figure 2

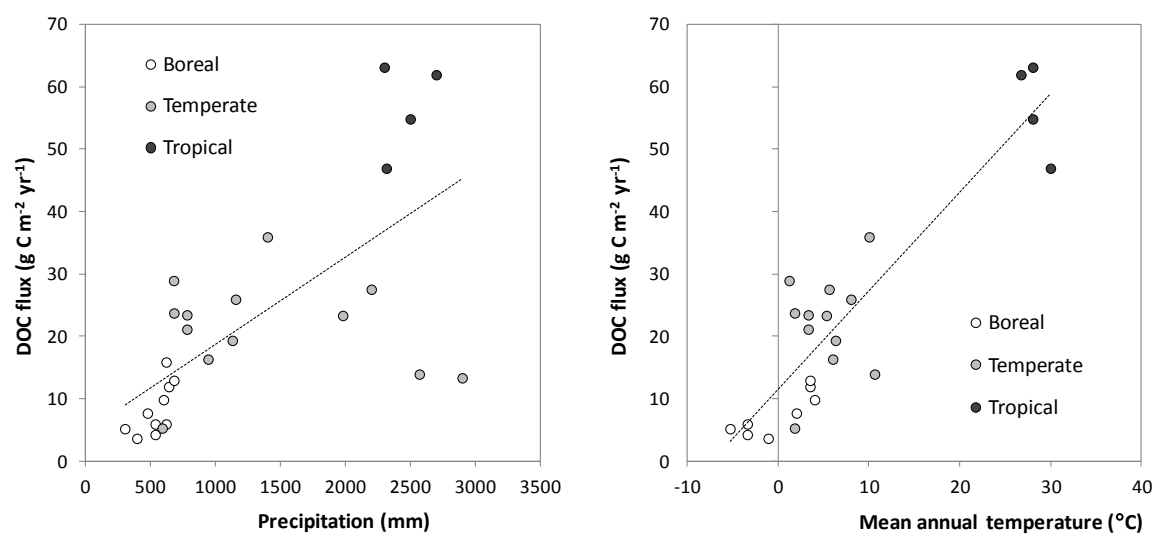
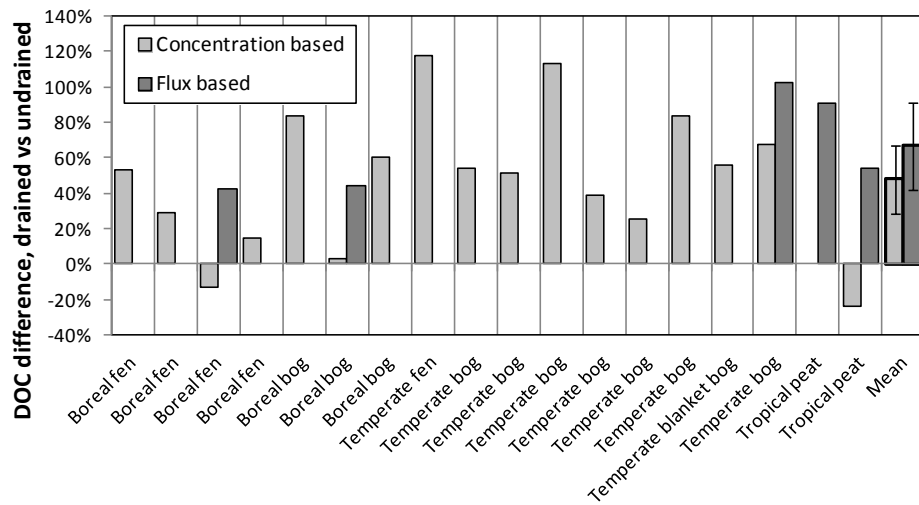


Figure 3

a) DOC responses to drainage



b) DOC responses to re-wetting

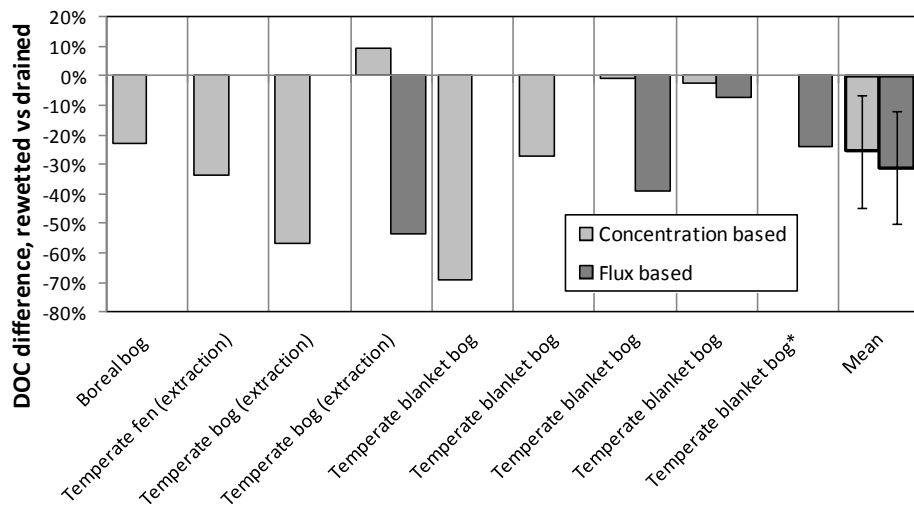


Figure 4

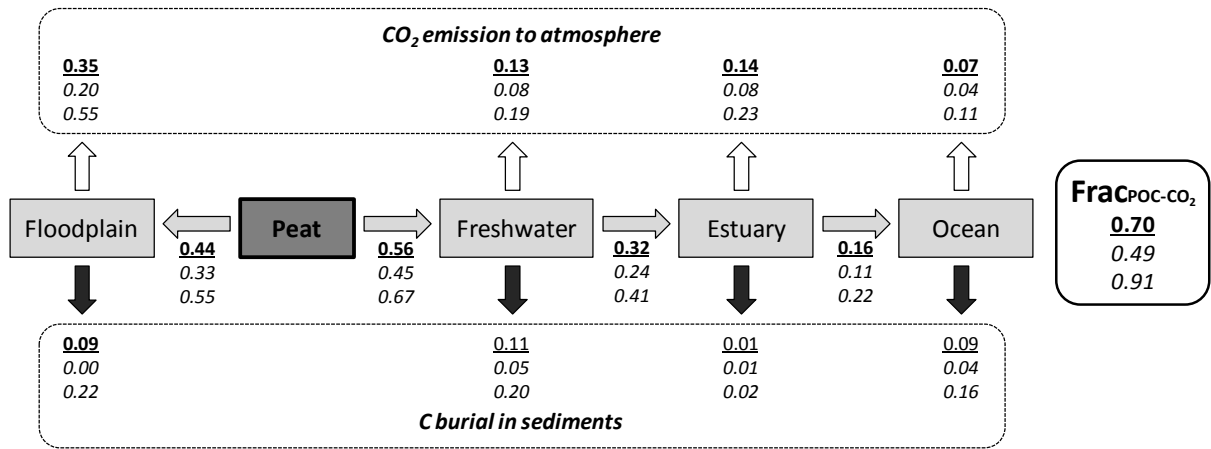


Figure 5

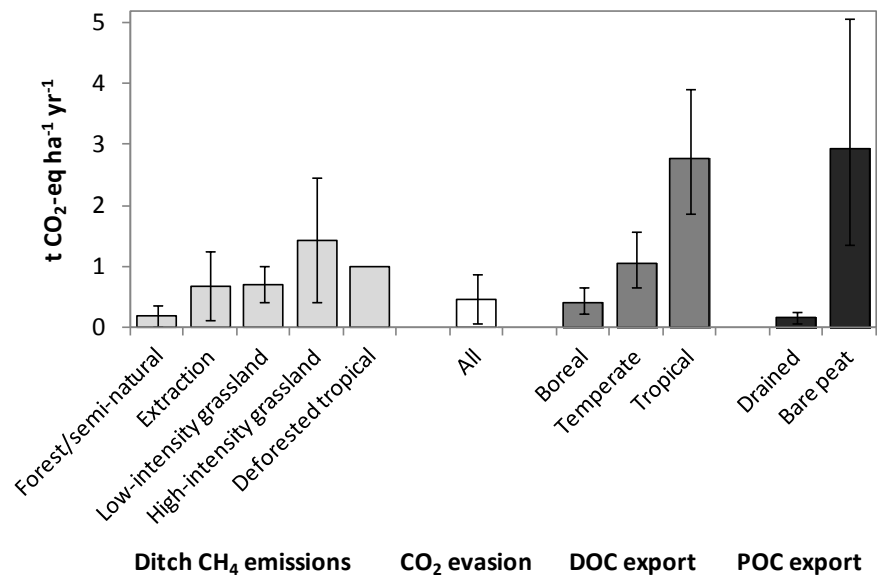


Figure 6

