Article (refereed) - postprint

Cooper, Mark D.A.; Evans, Christopher D.; Zieliński, Piotr; Levy, Peter E.; Gray, Alan; Peacock, Mike; Norris, David; Fenner, Nathalie; Freeman, Christopher. 2014. **Infilled ditches are hotspots of landscape methane flux following peatland re-wetting**. *Ecosystems*, 17 (7). 1227-1241. 10.1007/s10021-014-9791-3

Infilled ditches are hotspots of landscape methane flux following peatland re-wetting

Mark. D. A. Cooper[1, 2], Christopher. D. Evans[1], Piotr Zielinski[2, 3], Peter. E. Levy[4], Alan Gray[4], Mike Peacock[2,5], David Norris[1], Nathalie Fenner[2] and Christopher Freeman[2]

[1]*Centre for Ecology and Hydrology, Environment Centre Wales, Bangor, Gwynedd, LL57 2UW, U.K.*

[2]*Wolfson Carbon Capture Laboratory, School of Biological Sciences, Bangor University, Bangor Gwynedd LL57 2UW, U.K.*

[3]*Institute of Biology, University of Bialystok, Swierkowa 20B, 15-950 Bialystok, Poland*

[4]*Centre for Ecology and Hydrology, Bush Estate, Penicuik, Midlothian, EH26 0QB, U.K.*

[5]*Centre for Earth, Planetary, Space and Astronomical Research (CEPSAR), Department of Environment, Earth and Ecosystems, The Open University, Walton Hall, Milton Keynes, MK7 6AA.*

Abstract. Peatlands are large terrestrial stores of carbon, and sustained CO_2 sinks, but over the last century large areas have been drained for agriculture and forestry, potentially converting them into net carbon sources. More recently, some peatlands have been re-wetted by blocking drainage ditches, with the aims of enhancing biodiversity, mitigating flooding and promoting carbon storage. One potential detrimental consequence of peatland re-wetting is an increase in methane (CH_4) emissions, offsetting the benefits of increased CO_2 sequestration. We examined differences in CH_4 emissions between an area of ditch-drained blanket bog, and an adjacent area where drainage ditches were recently infilled. Results showed that *Eriophorum vaginatum* colonisation led to a 'hotspot' of CH_4 emissions from the infilled ditches themselves, with smaller increases in CH_4 from other re-wetted areas. Extrapolated to the area of blanket bog surrounding the study site, we estimated that CH_4 emissions were around 60 kg CH_4 ha^{-1} yr^{-1} prior to drainage, reducing to 44 kg CH_4 ha^{-1} yr^{-1} after drainage. We calculated that fully re-wetting this area would initially increase emissions to a peak of around 120 kg CH_4 ha^{-1} yr^{-1}, with around two thirds of the increase (and 90% of the increase over pre-drainage conditions) attributable to CH_4 emissions from *Eriophorum vaginatum*-colonised infilled ditches, despite these areas only occupying 7% of the landscape. We predicted that emissions should eventually decline towards pre-drainage values as the ecosystem recovers, but only if *Sphagnum* mosses displace *Eriophorum vaginatum* from the infilled ditches. These results have implications for peatland management for climate change mitigation, suggesting that restoration methods should aim, if possible, to avoid the colonisation of infilled ditches by aerenchymatous species such as *Eriophorum vaginatum*, and to encourage *Sphagnum* establishment.

Key Words

Methane, carbon, peatland, blanket bog, re-wetting, restoration, *Eriophorum*, *Sphagnum*

Introduction

Peatlands are characterised by an anoxic catotelm, where water-logged conditions and high acidity inhibit biological processes, suppressing decomposition and enabling organic material to accumulate (e.g. Freeman et al., 2001; Belyea and Baird, 2006). Since the end of the last ice age, peatlands have acted as a net sink for carbon dioxide (CO_2) wherever decomposition has been slower than litter formation. The carbon stock of peatlands in the Northern hemisphere is estimated to be 473-621 Pg C (Yu et al., 2010), representing 40% of global soil carbon, despite only covering 3% of the Earth's terrestrial surface.

The conditions within peatlands that favour CO_2 sequestration also favour the production of methane (CH_4), a powerful greenhouse gas, through the anaerobic decomposition of organic matter by methanogenic microbes (Denman et al., 2007). According to the most recent assessment report of the Intergovernmental Panel on Climate Change (IPCC), CH_4 has a 100 year global warming potential 28 times that of CO_2, rising to 34 if feedbacks from CH_4-driven warming on oceanic and terrestrial CO_2 release are taken into account (see Myrhe et al., 2013, and references therein).

CH_4 may enter the atmosphere via diffusion, ebullition, or plant-mediated transport. CH_4 may be oxidised by methanotrophic microbes as it diffuses through the aerobic acrotelm, such that almost all of the CH_4 produced may be removed before reaching the atmosphere (Calhoun and King, 1997). Alternatively, CH_4 may effectively by-pass oxidation if transported via ebullition (i.e. transport in biogenic gas bubbles; Baird et al., 2006) or through gas-transporting plant stems (aerenchyma). The difference between CH_4 production and oxidation in the ecosystem determines the net flux of CH_4 between terrestrial ecosystem and the atmosphere. Estimates of annual global CH_4 emissions from peatlands to the atmosphere range from 38 to 157 Tg CH_4 year^{-1} (Petrescu et al., 2010).

Over recent centuries, large areas of Northern peatlands have been damaged through drainage and extraction, most notably in Northern Europe and European Russia, but also in parts of North America and Asia (e.g. Joosten, 2010). In these areas the water level is typically lowered by the digging of drainage channels or ditches. Draining peatlands can promote decomposition by increasing the depth of the anaerobic zone, exposing stored organic matter to oxidation and potentially resulting in high rates of CO_2 emission. On the other hand, this expansion of the aerobic zone also increases rates of CH_4 oxidation (Sundh et al., 1995).

More recently, re-wetting of some drained peatlands has taken place via the blocking of draniage ditches, in an attempt to return these ecosystems to a near-natural state, and thereby to enhance their biodiversity and conservation value (Carroll et al., 2011). A variety of methods have been tried, including the use of dams (made from peat blocks or plastic sheeting), in-filling with materials such as wood brash or bales of locally harvested vegetation such as *Calluna vulgaris*, or ‘re-profiling’ the ditch by transferring peat material into the ditch from adjacent areas (Armstrong et al., 2009). The expectation in all cases is that restoring water-logged conditions will promote the functioning of peatlands as net CO_2 sinks, or at least reduce net CO_2 emissions, but there is a risk that increased CH_4 emissions may reduce, or negate, the benefits of peatland restoration in terms of the net greenhouse gas balance. A number of studies have quantified the direct effects of blocking drainage ditches on CH_4 emissions. Waddington and Day (2007) found evidence to suggest that CH_4 emissions increased 4.6 times following restoration of a cutover peatland site in Bois-des-Bel, Canada. Tuittila et al. (2000) observed an increase in CH_4 flux with the colonisation of *Eriophorum vaginatum* after restoration of a cut-over peatland in Southern Finland. Best and Jacobs (1997) observed a 3.4 fold increase in CH_4 production seven years after raising the water level in a grass dominated peatland in the Netherlands, while Wilson et al. (2008) found that CH_4 fluxes increased from near-zero in an area of bare cutaway peat to between 4 and 39 g CH_4 m^{-2} yr^{-1} in re-wetted areas, depending on the plant community present. Mesocosm studies also indicate that rewetting blanket bog peat leads to an increase in CH_4 flux to the atmosphere (Dinsmore et al., 2009; Green et al., 2011).

The potential to develop policy and economic instruments to support peatland re-wetting as a mechanism for reducing greenhouse gas (GHG) emissions has received increasing recent attention (e.g. Bain et al., 2011; Dunn and Freeman, 2011). Wetland re-wetting has now been adopted as a voluntary reporting activity under the Kyoto Protocol of the United Nations Framework Convention on Climate Change (UNFCCC), and new guidance provided by the Intergovernmental Panel on Climate Change on methods to support GHG accounting for drained and re-wetted organic soils (IPCC, 2013). In the UK, policy impetus towards peat restoration was provided by the recent International Union for Conservation of Nature (IUCN) Commission of Enquiry on Peatlands (Bain et al., 2011), leading to the development of a pilot ‘Peatland Code’ to underpin a ‘Payment for Ecosystem Services’ scheme to provide financial support to peatland restoration (Reed et al., 2013). All of these international and national-scale initiatives depend, however, on robust and representative underpinning data on the effects of both drainage and ditch-blocking on CO_2 and CH_4 fluxes, as the basis for calculating ‘emission factors’ (estimates of the mean annual GHG flux per unit of land surface within a particular land-use category) for use in national greenhouse gas inventories and other accounting schemes. In the UK, and particularly for the upland blanket bog peatlands that make up a large part of the UK peat resource (JNCC, 2011) these data are largely lacking.

In this study, we therefore aimed to provide suitable data to support the development of emission factors for GHG accounting in drained and re-wetted blanket bogs, with a focus on CH_4, based on an experimental peat restoration site in North Wales, UK. We also aimed to enhance current understanding of the processes that drive changes in CH_4 emissions following peat re-wetting, particularly in relation to vegetation changes. We hypothesised: 1) that the ditch-blocked area would emit greater amounts of CH_4 than the drained area, as wetter conditions favour methanogenesis and restrict methane oxidation; and 2) that the actual rate of CH_4 emission within the ditch-blocked area would vary according to the extent and type of vegetation re-establishment within the site, due to the role of plants as sources of labile substrate for methanogenesis, and (in the case of aerenchymatous species) in providing pathways for CH4 transport to the atmosphere. To provide a comparison with more natural (i.e. pre-drainage) conditions, further data were collected at a nearby site with minimal drainage impacts.

Materials and Methods

Site Description

Research was undertaken within the catchment of Llyn Serw (52° 58′ 09″ N, 3° 49′ 00″ W), a small lake within the Migneint blanket bog, which drains to the River Conwy. The Migneint, which lies within a European Special Area of Conservation, largely comprises heterogeneous upland wet heathland on blanket peat, with dominant plant species being *Calluna vulgaris*, *Eriophorum spp*, *Juncus spp*. and *Sphagnum spp*. The Llyn Serw study site lies at an altitude of 450-460m, in an area strongly affected by historic peat drainage, which was undertaken during the 20th century with the intention of enhancing grazing productivity. The site is entirely comprised of *Calluna*-dominated mire with an average canopy height of 50 cm (UK National Vegetation Classification (NVC) class M19 (Rodwell, 1998)). Average peat thickness is around 2 m, with underlying bedrock consisting of Ordovician shales and volcanic tuffs (Lynas, 1973).

The hillslope on which the study took place was previously drained by two sets of intersecting drainage ditches (Figure 1). The first set of parallel ditches was dug in the 1920s-1930s running downslope from northeast to southwest. Some vegetation recolonisation and in-filling of these ditches subsequently took place, although in general they have continued to function. A second, deeper set of ditches was dug in the 1970s-1980s, running diagonally across the slope from southeast to northwest. The latter ditches are approximately 40 m apart and flow into Llyn Serw via a single ditch flowing northwest to southeast. The area was affected by a wildfire in 2003, so that the regenerating *Calluna* heathland is at a relatively early growth stage. In August 2008, as part of a pilot restoration study, four of the deeper southeast-northwest ditches were completely blocked with heather bales and then re-profiled, whereby the upper layers of peat adjacent to the ditch were scraped over the heather bales to entirely fill the ditch. The remaining two ditches (those closest to the lake) were left open as a control, visible in Figure 1.

In addition to the Llyn Serw site, further data were collected over the same time period from the nearby Nant y Brwyn catchment (52° 59′ 51″ N, 3°48′ 0″ W), located 3 km to the North

East. This catchment is topographically and botanically similar to Llyn Serw, but has been less affected by drainage, and recent land-management activities have been minimal. The site therefore served as a relatively undisturbed reference site for the drained (and now re-wetted) Llyn Serw catchment. Mean annual air temperature at an automatic weather station located at the Nant y Brwyn (altitude 415 m) is 5.6 ^{o}C, with monthly averages ranging from -1.2 °C in December to 12.2 °C in July. The mean annual precipitation is around 2200 mm. For further details of the Migneint area see also Ellis and Tallis (2001; Billett et al., 2010; Evans et al., 2012).

Experimental design

Two sampling transects were established, each approximately 8 m in length, following the slope and extending approximately 4 m either side of an unblocked and a blocked ditch from the second, deeper set of ditches (Figure 1,2). Along each transect, a sequence of measurement points (points A-F, in a sequence from upslope to downslope) were established, within which replicate fixed collars for static chamber measurement and co-located dipwells and pore-water samplers were deployed. Sampling points were also established within the unblocked and blocked ditches themselves. The aim of the experimental design was to obtain a set of integrated and comparable measurements, focusing primarily on CH_4 emissions, at a set of points subject to varying degrees of water table drawdown, during a two year period closely following the re-wetting of a part of the site.

Methane flux measurements

A total of 26 sets of CH_4 flux measurements were made over a 27-month period from June 2009 to August 2011, using the static chamber approach (Livingston and Hutchinson, 1995).

Sampling was intensified during the growing season, in order to capture the period of anticipated higher CH_4 emissions, and reduced during winter. At sampling points A-F, two replicate gas sampling collars, 30 cm in diameter, were installed in March 2009 and allowed a two-month settling period before sampling commenced. The collars were inserted approximately 5 cm into the ground, with 5cm above-ground. The subsurface part of the collar was perforated to allow subsurface water throughflow and to prevent ponding within the collar during high rainfall. When making flux measurements, we followed the method described by Ward et al. (2007) for the same habitat type, using a chamber modified from a garden cloche (a domed plastic cover designed to protect plants from frost) which was attached to the collar using a rubber seal, to make a closed chamber with a maximum height of 31 cm, and an internal volume of 19 litres. Each chamber contained a vent covered in an expandable polythene skin to allow air pressure inside and outside the chamber to equilibrate. During enclosure, the internal chamber air temperature was monitored using Tinytag temperature loggers (Gemini Data Loggers (UK) Ltd, Chichester, UK) installed in eight of the chambers, with two further loggers monitoring ambient air temperature. Sampling was undertaken from adjacent boardwalk to minimise disturbance and the risk of inducing ebullition. Gas samples of 30 ml were extracted through SubaSeal septa using a syringe and needle. Ambient air samples were also collected during gas sampling, and assumed to represent initial gas concentrations. During the first part of the study, three 30 ml gas samples were extracted from the chamber headspace over an enclosure time of two hours. In 2011, in order to shorten the enclosure time, increase temporal resolution and ensure detection of any non-linear CH_4 concentration changes, four measurements were extracted over a 30 minute period. Extracted gas samples were immediately transferred into pre-evacuated 22 ml air-tight glass vials, and analysed within a week of sampling whenever possible.

The additional sampling points G-H were established within the blocked drain in June 2010, on bare peat and re-colonising vegetation respectively, while a point G was also established on bare peat within the open drain in January 2011. Due to the flow of water and sediment, *in situ* collars were not used in the open drain; instead, chamber lids were placed directly onto the ditch base during sampling. Evidence of an air tight seal was demonstrated by suction resistance when the chambers were removed at the end of sampling. Further CH_4 measurements (following the same methods) were made in a ditch in the Nant y Brwyn catchment that had naturally infilled with *Sphagnum fallax*, and from a nearby area of *Calluna-Sphagnum* blanket bog that was minimally influenced by drainage. These data were used as a 'reference' for the Llyn Serw site, on the assumption that it will eventually resemble this less disturbed site.

Gas analysis was carried out using a Perkin Elmer Clarus 500 Gas Chromatograph (GC). CH_4 was detected using FID (flame ionisation detector) at 375°C, and the sample oven at 40°C, equipped with a methaniser. The calibration of the GC for CH_4 involved three standard concentrations (5 ppm, 20 ppm and 50 ppm; Cryoservice, UK) and calibration was accepted at $r^2 > 0.99$. Standard gas concentrations were analysed after every ten samples to assess accuracy of the calibration. The flux was calculated from the time series of CH_4 concentrations within the chamber using linear regression (Levy et al., 2011). A flux was accepted if the coefficient of determination (r^2) was at least 0.70. However Alm et al. (2007) highlighted that low fluxes (particularly those close to zero) generally have a low r^2, and should not therefore be excluded, as this can lead to an over-estimate of mean fluxes. Therefore, fluxes with $r^2 < 0.7$ were retained provided that the residual variance did not exceed a threshold (based on an inspection of the typical variability in the dataset) of 30 (μmol mol^{-1})^2. Mean fluxes were calculated for individual sampling points and for each landscape category (e.g. drained *Calluna*-*Sphagnum* bog, *Eriophorum vaginatum*-colonised infilled ditch) by first aggregating retained flux measurements into three time periods, namely October to March, April to June, and July to September. The longer period over which data were aggregated during winter was a consequence of the smaller number of measurements made during this period. For each time period, individual collar mean CH_4 fluxes were calculated, and used to derive a seasonal mean flux and associated standard error for each landscape category. Annual mean fluxes were calculated by taking a time-weighted average of the three seasonal subset means. This approach was taken in order to eliminate any potential bias resulting from the greater frequency of summer versus winter measurements.

Other measurements

For each flux measurement, soil temperature was measured at 10 cm depth, and water table depth was measured in a dipwell adjacent to each chamber on each sampling occasion. Mean annual water levels were calculated per dipwell, and aggregated by landscape class, following the same seasonal subset approach applied to CH_4 fluxes. Freely draining pore water was collected using shallow piezometers comprising 2.5cm diameter perforated PVC pipes coated with filter gauze, sampling water at a depth of 15-20 cm. The top of each piezometer was covered with a polypropylene lid, with a vent hole, to avoid contamination. One piezometer was installed next to each set of gas sampling collars at Llyn Serw, including the vegetated and unvegetated sampling points within the infilled ditch. Samples from the intact drainage ditch were collected by directly sampling the surface water with a syringe. At the Nant y Brywn reference site, two piezometers were installed in the *Calluna*-*Sphagnum* bog, and two in the *Sphagnum*-filled ditch.

All piezometers were emptied and allowed to refill before sampling, and samples collected monthly from October 2009 until November 2010 using tubes attached to a 50 ml syringe. Samples were transferred to pre-washed (10% hydrochloric acid) polyethylene bottles for transfer to the laboratory, where they were analysed for pH using an Orion 720A pH meter, and remaining sample filtered using 0.45 μm cellulose membrane filter (Minisart, Sartorius Stedim Biotech, Germany) and stored at 4°C prior to analysis. Dissolved organic carbon (DOC) was analysed with a Thermalox 5001.03 carbon analyser (Analytical Sciences Limited, Cambridge, UK) using the non-purgable organic carbon (NPOC) method, whereby samples were acidified to pH 2.0 and purged with oxygen to drive off any inorganic carbon prior to analysis for DOC. Sulphate concentrations were measured using a Metrohm 850 Ion Chromatograph equipped with a Dionex AS14A analytical column. Meteorological data were obtained from the automatic weather station in the Nant y Brwyn catchment.

Within each of the sampling collars, the percent cover was recorded for each species, both in the field and using photographs taken in June 2010 and June 2011. The number of *Eriophorum vaginatum* spikelets (flower clusters, visible in Figure 3 below) was also recorded from these photographs.

Statistical analysis

A simple linear mixed-effects model (Pinheiro and Bates, 2000) was used to analyse the CH_4 dataset in relation to land cover category (i.e. undrained, drained and re-wetted blanket bog, active ditch, and infilled ditches with bare peat, *Eriophorum* and *Sphagnum*). Land-cover category was treated as a fixed effect, and repeated measurements at each individual collar location as a random effect. Mean CH_4 flux from each measurement point was analysed against mean measured water table depth and *Eriophorum* cover using simple linear regression.

Area-weighted flux estimation

Area-weighted fluxes were calculated in order to estimate the overall CH_4 emission from the blanket bog landscape in the vicinity of the measurement transects, taking account of the differing proportions of different landscape features, for a number of pre- and post-drainage scenarios. As the basis for this landscape upscaling, we defined a rectangular area around our measurement sites, with a total area of 11000 m^2 (Figure 1). Although the boundaries of this area are essentially arbitrary, they encompass a fairly typical and homogenous 'target area' of the drained blanket bog, much of which was ditch-blocked during the restoration. Adjacent natural wetland flushes, dominated by *Juncus effusus*, were excluded. Within the target area, a high-resolution (0.5 m pixel size) LiDAR digital elevation dataset (National Trust, unpublished data) was used to map the ditches, and to calculate total ditch length. Average width of unblocked and blocked ditches was recorded on the ground, and used to calculate total ditch areas within the target area before and after ditch blocking. Note that blocked ditches remained as shallow, broader features within the landscape following the infilling process (Figure 2). Within these infilled ditches, the proportion of the area occupied by bare peat and recolonising vegetation was also quantified, and measured CH_4 fluxes from the G and H collars used to calculate emissions for each category. The *Calluna*-dominated bog between the ditches was considered as a homogenous landscape component in terms of CH_4 fluxes, which were therefore calculated from the mean of all measured fluxes from collars A-F within the drained and re-wetted areas respectively. Although ditch blocking was only carried out on part of the study site, for the purposes of evaluating CH_4 fluxes from drained and re-wetted sites we calculated landscape-scale emissions within the target area for the original fully-drained condition, and for a fully re-wetted scenario. In addition, we applied three alternative ‘long-term restored’ scenarios. The first of these assumed that the wet depressions created by ditch-blocking would become fully colonised by a persistent *E. vaginatum* dominated community. The second scenario assumed that these depressions would ultimately become dominated by *S. fallax*, as has been observed at naturally infilled ditches elsewhere on the Migneint. The third scenario additionally assumed that the re-wetted blanket bog would eventually attain the same level of CH_4 emission as an undrained system. These calculations utilised measured CH_4 fluxes from the ‘reference’ site in the nearby Nant y Brwyn catchment.

Results

Ecological Observations

Over the 27 month sampling period, distinct changes in vegetation were recorded within the blocked drains. Following the disturbance associated with reprofiling, the surface of the blocked ditches was largely bare. Over the course of the study, vegetation cover increased, notably by *E. vaginatum* which was the main colonising species on the perturbed peat on the infilled ditches. By 2010, this had led to visible ‘white stripes’ across the blanket bog along the former ditch lines (Figure 3). Despite this, surface flow continued to occur along parts of the infilled ditches, leading to the persistence of substantial areas of bare peat (also visible on the left of Figure 3) and very limited *Sphagnum* spp. recolonisation. This situation has persisted up to the time of writing (summer 2013), although given that only five years have elapsed since the ditch-blocking took place, it is probable that vegetation composition at the site is still in transition.

Water level and water chemistry

Water levels were clearly lowered either side of the unblocked ditch, with mean water table 5.5 cm below the surface upslope of the ditch, and 14.7 cm below at the downslope sampling points (Table 1). The greatest water table drawdown was observed at sampling point D, immediately downslope of the open ditch, on average 25 cm below the surface (Figure 2). Comparing the between-ditch sampling points at the two transects, mean water table over the full measurement period was 7 cm higher around the blocked ditch compared to the unblocked ditch, and this difference was consistent throughout the year (Figure 4a). Mean water table depths of around 3 cm either side of the blocked ditch were close to those measured at the Nant y Brwyn reference site (around 1 cm), indicating that the ditch-blocking has been fairly successful in raising water levels towards natural levels, producing a shallow and relatively uniform water table across the blocked-ditch transect, and reducing interception of downslope flows by ditches running laterally across the hillslope.

Water chemistry measurements from within the ditches indicated that DOC concentrations were lowest within the open ditch, and in pore water from the *S. fallax*-infilled reference ditch, and highest in pore water in areas of the infilled ditch where *E. vaginatum* re-colonisation had occurred (Table 1). In the blanket bog, mean porewater DOC concentrations were lower (< 50 mg l^{-1}) at the undrained bsite and downslope of the infilled ditch, and higher (> 70 mg l^{-1}) upslope of the infilled ditch and either side of the open ditch. Pore water pH varied slightly between plots, but there was no clear relationship with vegetation cover or water table. Sulphate concentrations were somewhat higher in the ditch-blocked transect, particularly downslope of the ditch where water table drawdown was greatest, but were similarly high at the undrained site.

Methane fluxes

Methane fluxes showed a high degree of both spatial and temporal variability. Data aggregated into three seasonal time periods (Figure 4b-c) show a general tendency for emissions to be lowest during the winter period (October to March), intermediate during April to June, and highest in July to September. This seasonal pattern was consistent across the undrained, drained and re-wetted *Calluna*-*Sphagnum* bog, and also in areas of infilled ditch occupied by *E. vaginatum* or *Sphagnum* spp. Emissions from unvegetated areas (open ditches and areas of bare peat in the infilled ditches) showed less consistent seasonal patterns, although maximum fluxes were again recorded during the growing season.

Analysis using the mixed-effects model showed the *Eriophorum*-dominated infilled ditches to be the only significantly different land cover category; the 95 % confidence intervals on the other groups were overlapping. Because the study design was not replicated in a strict sense, we place limited emphasis on significance testing of differences, but consider the trends between the groups. For the *Calluna*-*Sphagnum* bog, average fluxes during all seasons were in the order Re-wetted > Undrained > Drained, albeit with fairly high spatial variability within each category (Figure 4b). Estimated mean annual fluxes ranged from 43.7 kg CH_4 ha^{-1} yr^{-1} at the drained site to 74.4 kg CH_4 ha^{-1} yr^{-1} at the re-wetted site. At the drained site, mean CH_4 fluxes were higher upslope of the ditch compared to downslope (Table 2), corresponding to greater water table drawdown in the downslope locations. Contrasts between fluxes upslope and downslope of the blocked ditch were more subdued.

For the within-ditch measurements, by far the highest CH_4 emissions were recorded from areas of *E. vaginatum*-colonised infilled ditch, with an estimated annual mean of 720 kg CH_4 ha^{-1} yr^{-1}. In contrast, mean emissions from the active, infilled bare peat and *S. fallax*-colonised ditch sites were much smaller but similar (ranging from 43 to 51 kg CH_4 ha^{-1} yr^{-1}).

Relationships between CH_4 fluxes and other measured variables

Mean CH_4 flux measured at each individual collar showed a negative, non-linear relationship with water table, but with very high variability in mean flux among collars with a mean flux at or close to the ground surface (Figure 5a). A significant positive correlation was observed between mean flux and estimated cover of *Eriophorum* spp. (adjusted $R^2 = 0.70$, $p < 0.001$; Figure 5b). Comparing the number of recorded *Eriophorum* spikelets gave a considerably stronger correlation (adjusted $R^2 = 0.89$, $p < 0.001$; Figure 5c), and was particularly effective at differentiating fluxes between collars with a high percentage *Eriophorum* cover.

Area-weighted flux estimates

Data from the drained and drain-blocked transects at Llyn Serw, together with data from the Nant y Brwyn reference site, were used to generate estimates of the overall CH_4 flux from the defined ‘target area’ (Figure 1). Methane fluxes for Scenario 1 were calculated for a pre-disturbance condition, when the entire area would have been occupied by *Calluna-Sphagnum* blanket bog, as at the Nant y Brwyn. Based on the LiDAR data, a total ditch length of 1559 m was estimated to be present within this 11000 m^2 area prior to restoration, which ground observations indicated had a mean width of 0.5 m, giving a total of 7.1% of the target area occupied by ditches (Scenario 2). The ditch-blocking process resulted in the formation of shallower depressions with a mean width of approximately 1m. Thus, for a fully re-wetted site, the proportional area occupied by the infilled ditches would increase to 14.1% of the total area. As noted in the methods, this area was estimated to be equally comprised of bare peat and re-colonised *E. vaginatum* after two years of restoration (Scenario 3). The three alternative future scenarios (4a – re-wetted blanket bog with infilled ditches fully colonised by *E. vaginatum*; 4b – re-wetted blanket bog with infilled ditches fully colonised by *S. Fallax*; 4c – CH_4 fluxes equivalent to those from an undrained blanket bog, with infilled ditches fully colonised by *S. Fallax*) essentially represent 'worst', 'intermediate' and 'best' cases in terms of CH_4 emissions. Note that these scenarios are not time-specific, since the actual trajectory of ecosystem recovery at the site is unknown; in principle these states could occur sequentially, or could represent alternative stable end-points.

Results suggest that drainage of the blanket bog reduced CH_4 emissions by a relatively modest amount, from 61 kg CH_4 ha^{-1} yr^{-1} to 44 kg CH_4 ha^{-1} yr^{-1}. Re-wetting is estimated to generate a large increase in overall landscape-scale flux, to approximately 117 kg CH_4 ha^{-1} yr^{-1} (for a fully re-wetted area), almost double the pre-drainage emission. Around 43% of the total landscape CH_4 emission at this point derives from the estimated 7.1% of the area occupied by *E. vaginatum* in infilled ditches. Of the net increase in estimated CH_4 emissions from the fully re-wetted site, relative to the pre-drainage baseline, an estimated 90% is attributable to the *E. vaginatum*-colonised infilled ditches.

Taking the 'worst-case' scenario of the infilled ditches becoming fully colonised by *E. Vaginatum* (Scenario 4a), the predicted CH_4 flux would further increase to 166 kg CH_4 ha^{-1} yr^{-1}. On the other hand, were the infilled ditches ultimately to become colonised by *Sphagnum* (Scenario 4b), the predicted landscape CH_4 flux would reduce from current levels to around 71 kg CH_4 ha^{-1} yr^{-1}. Further assuming that CH_4 emissions from the blanket bog return to pre-drainage levels (Scenario 4c), the estimated landscape flux closely approaches its original level, and could even be marginally lower (on the basis that measured CH_4 emissions from residual, *Sphagnum*-filled ditch lines were lower than those measured from undrained blanket bog).

Discussion

Effects of re-wetting on CH_4 emissions

Our results suggest that blocking ditches in a drained Welsh blanket bog has led to substantial increases in CH_4 emissions in the years immediately following re-wetting. We recognise that our measurements were (given logistical constraints) limited to a single study site, and thus lacked true replication. Additionally, the measurement method did not permit us to quantify fluxes associated with ebullition, which tend to occur as short, infrequent pulses and therefore require a different, longer-term sampling technique (e.g. Baird et al., 2004). This may have led to some under-estimation of the total CH_4 emission from the peatland. Nevertheless, the relative changes in CH_4 flux observed in our study were generally clear, and were to a large degree consistent with results from other peatland types, including re-wetted cutover sites in Canada (Waddington and Day, 2007), Ireland (Wilson et al., 2009) and Finland (Tuittila et al., 2000), all of which reported an overall increase in CH_4 emissions following re-wetting. For the blanket bog at Llyn Serw, spatial extrapolation of the measurements suggests that landscape-scale CH_4 emissions increased by a factor of 2.7 when comparing the second year post re-wetting to the drained condition. By comparison, Waddington and Day (2007) observed a near fivefold increase in CH_4 when comparing a re-wetted site to a drained cutover site, based on measurements made a similar time after re-wetting. Their landscape-scaled mean CH_4 emission from the restored site at this time was very similar to that estimated in our study (127 vs 117 kg CH_4 ha^{-1} yr^{-1}), with the greater relative difference due mainly to lower emissions from their unrestored site (a largely unvegetated cutover peatland) compared to our drained but largely intact blanket bog (27 vs 44 kg CH_4 ha^{-1} yr^{-1}). The study by Wilson et al. (2009) gave lower CH_4 emissions from areas of a re-wetted cutover peatland dominated by an *Eriophorum/Carex* mix (32 – 43 kg CH_4 ha^{-1} yr^{-1}), but markedly higher emissions from areas occupied by tall fen species (184 – 388 kg CH_4 ha^{-1} yr^{-1}).

The role of vegetation

Per unit area, our results clearly showed that the largest source of CH_4 emissions derived from blocked ditches containing *E. vaginatum,* which colonised the disturbed bare peat during the first two years after re-wetting. This colonisation by *E. vaginatum* after re-wetting has also been observed on a similar timeframe elsewhere on the Migneint (Peacock et al., 2013); three years after restoration of a peat harvesting site in Eastern Canada (Marinier et al., 2004); and following restoration of a harvest site in a raised bog in Southern Finland (Tuittila et al., 2000). Our results, suggesting that infilled ditches colonised by *E. vaginatum* generate almost half of the total CH_4 emission from less than 10% of the re-wetted blanket bog land surface, are consistent with a number of previous studies showing that CH_4 fluxes tend to be highest where this species occurs (Tuittila et al., 2000; McNamara et al., 2008; Green and Baird, 2012; Ström et al., 2012). Waddington and Day (2007) found that CH_4 emissions from a re-wetted cutover peatland were overwhelmingly derived from areas of the peat surface colonised by herbaceous species, of which *E. vaginatum* was the main constituent, together with ditches that had also been colonised by vascular plants, whereas the

~50% of the peatland colonised by moss species acted as a marginal net CH_4 sink. For blanket bogs, McNamara et al. (2008) estimated that around 75% of CH_4 emissions from a semi-natural catchment were associated with *Eriophorum*, and that wet gully areas (42% of which were covered by *Eriophorum* spp.) generated over 95% of all CH_4 emissions from less than 10% of the landscape. Our results thus provide similar evidence of spatially heterogeneous CH_4 emissions, associated with the presence of the same species, within a more human-modified blanket bog landscape. The strong observed correlation between CH_4 emissions and *Eriophorum* spikelet numbers has particular value for upscaling, as the spikelets can be detected in aerial imagery during the flowering period, and have been used to map CH_4 emissions elsewhere (Kalacska et al., 2013).

Potential mechanisms for increased CH_4 emissions

Several mechanisms may contribute to higher CH_4 emissions in areas of *Eriophorum* cover. These include the role of aerenchymatous tissue in transporting CH_4 from the anaerobic zone to the atmosphere, the active production of methanogenic substrate by the plants, or simply the tendency for *Eriophorum* spp. to grow within wetter, and hence CH_4-producing, areas within the bog. Our results, showing a stronger relationship between mean CH_4 fluxes and the presence of *Eriophorum* spp. (in particular the number of spikelets) than with mean water table or other measured environmental variables (Figure 5), support previous conclusions that higher fluxes are not simply due to *Eriophorum* spp. occupying wetter niches within the peatland landscape, but reflect an active influence of the plant on CH_4 emissions (Greenup et al., 2000; Marinier et al., 2004; Ström et al., 2012). Higher measured porewater DOC beneath an *E. vaginatum*-vegetated area compared to a bare peat areas of the infilled ditch in our study (Table 1) provides some support to the hypothesis that the plants increase the supply of substrate for methanogenesis, and appear consistent with the results of Ström et al. (2012), who found greater concentrations of acetate, (a substrate for methanogenesis) around *Eriophorum* roots. Lower DOC concentrations from bare peat areas on the infilled ditch suggest that these higher DOC concentrations were not solely related to their location within the infilled ditches, or to decomposition of the underlying *Calluna* bales.

The observation that CH_4 fluxes were more closely related to the number of *Eriophorum* spikelets than to percentage cover alone suggests that the vitality and productivity of the plants, rather than simply their presence, may be important. However, this observation could be explained by a number of factors, namely: (i) that the number of spikelets provides a better proxy for aerenchymatous conduit area than our estimates of percent cover; (ii) that the spikelet stems themselves act as a substantial conduit for CH_4 and oxygen exchange between rhizosphere and atmosphere; and (iii) that spikelet numbers are correlated with the rate of root exudate production. Further work is required to differentiate these potential influences on the rate of CH_4 emission, as well as the net greenhouse gas balance implications if higher CH_4 emissions are associated with more productive plants, which may offset these emissions via a greater uptake of CO_2.

CH_4 emissions at the landscape scale

Our results demonstrate the importance of both water table (e.g. comparing fluxes from undrained, drained and re-wetted *Calluna*-*Sphagnum* bog) and vegetation (e.g. comparing fluxes from bare peat, *Eriophorum* and *Sphagnum*-filled ditches) in determining rates of CH_4 emission. The observed role of vegetation type supports previous attempts to use peatland flora as a proxy to estimate CH_4 flux (Dias et al., 2010; Couwenberg et al., 2011; Levy et al., 2012; Gray et al., 2013). Our upscaled flux estimates support the general observation that peatland drainage substantially reduces total CH_4 emissions. In contrast to several previous studies (e.g. Roulet and Moore, 1995; Schrier-Uijl et al., 2011; Teh et al., 2011), we did not observe higher emissions from the active ditches themselves compared to the adjacent land surface. This could reflect the higher ditch gradients in blanket bog (reducing both water residence times and mean water depths, and thus potential for *in-situ* methanogenesis) or relatively low substrate quality and nutrient levels when compared to the agriculturally-drained peatlands studied by Schrier-Uijl et al. (2011) and Teh et al. (2011). On the other hand, some CH_4 *was* emitted from the active ditch, and in general our data support the inclusion of CH_4 emissions from both drained peatland surfaces and drainage ditches in GHG accounting methods (IPCC, 2013), in place of the previous assumption that drained peatlands do not emit any CH_4 (IPCC, 2006).

The positive impact of peatland re-wetting on CH_4 emissions is clear from our results, which suggest that overall emissions from the re-wetted area are about 2.7 times higher than under drained conditions, and 1.9 times higher than from an undrained site. However, the evidence that a very high proportion of this increased emission is associated specifically with *Eriophorum* colonisation of the infilled ditches highlights the importance of successional changes in vegetation following peat re-wetting. *E. vaginatum* is a pioneer species, and was the first to establish within the infilled ditches, and it is possible that it will be displaced, or at least reduced in cover, as other species establish. If this were to happen, at least a part of the increased CH_4 emissions measured in the two years after ditch-blocking could be considered transient. Other factors including disturbance of the peat, and addition of labile organic matter during the restoration process (in this case, heather bales) could also contribute to a transient pulse of emissions. The slightly (albeit non-significantly) higher measured CH_4 flux from the re-wetted *Calluna-Sphagnum* blanket bog, when compared to a botanically similar undrained location, would appear to support this interpretation.

Two of our scenarios for landscape-scale CH_4 fluxes (Scenarios 4b and 4c, Table 3), in which *S. fallax* is assumed to colonise the infilled ditches, suggest that emissions may eventually reduce towards pre-drainage levels. However the validity of the assumptions underpinning these scenarios, and also the time it may take to achieve a final vegetation community, remain uncertain. Haapalehto et al. (2011) observed that cover of *E. vaginatum* was continuing to increase 10 years after the re-wetting of a bog in Finland, suggesting the CH_4 emissions at our site might in fact continue to rise. For our worst-case scenario (4a), in which *E. vaginatum* expands to cover the entire infilled ditch area, the landscape-scale estimated CH_4 flux is almost 3.8 times higher than from the drained bog, and 2.7 times higher than the from undrained bog. The establishment of *Sphagnum* within infilled ditches thus appears critically important; as well as suppressing the cover of aerenchymatous species such as *E. vaginatum*, *Sphagnum* has been shown to support methanotrophic (CH_4 consuming) bacteria, reducing the release of CH_4 from the anaerobic zone to the atmosphere (Raghoebarsing et al., 2005). The marked contrast in CH_4 emissions between vegetation types, and the uncertain trajectory of future vegetation changes, suggests that active management to facilitate re-colonisation by *Sphagnum* might be beneficial. On the Migneint, more recent re-wetting activities have been undertaken using an alternative ditch-blocking method, involving the ‘reprofiling’ of ditches to form shallower depressions, interspersed with peat dams and small pools. Peacock et al. (2013) found that *Eriophorum* species were unable to colonise the deeper pools, which instead tended to develop a *Sphagnum* cover, with probable benefits in terms of CH_4 emissions.

Finally, it is important to emphasise that, although we observed an increase in CH_4 flux following ditch-blocking at our site, this does not necessarily indicate that peatland re-wetting has had a net warming effect in terms of overall GHG emissions. Waddington et al. (2010) found that an increase in the CO_2 sink after restoration outweighed the increase in CH_4 emissions, and Wilson et al. (2013) found *Eriophorum* to be substantial sink of carbon, offsetting its role as a source of CH_4 emissions. It is therefore feasible that the re-wetting of our study site, and similar sites elsewhere, may be having a net cooling impact, despite increasing CH_4 emissions with increasing *Eriophorum* cover.

Conclusions

Infilling drainage ditches increased water table elevation and landscape-scale CH_4 flux in the two years following blocking. The increased CH_4 emissions observed were driven by the creation of CH_4 'hotspots' that occurred where *E. vaginatum* tussocks colonised the infilled ditch. It is unknown whether this phenomenon is a long-term effect; CH_4 fluxes from the ditch-blocked area were also higher than in a nearby undrained area, suggesting that at least part of the observed increase may be transient. A large part of the uncertainty in attempting to extrapolate these effects over longer time scales is the uncertain trajectory and time course of plant succession on the blocked ditches, together with the apparent but uncertain links between plant species composition and CH_4 flux. Active vegetation management may therefore exert a considerable influence on the greenhouse gas balance of re-wetted peatlands.

Acknowledgements

The authors are grateful to Trystan Edwards, Andrew Roberts and others at the National Trust for carrying out the ditch blocking for this pilot study, providing access to LiDAR data and for support throughout the study. The authors would also like to thank Tim Jones, Rob Mills, Andy Baird and two anonymous reviewers for useful discussions, comments and suggestions at various stages during the development of this manuscript.

References

Alm, J., Shurpali, N.J., Tuittila, E.S., Laurila, T., Maljanen, M., Saarnio, S., Minkkinen, K., 2007. Methods for determining emission factors for the use of peat and peatlands - flux measurements and modelling. Boreal Environment Research 12, 85-100.

Armstrong, A., Holden, J., Kay, P., Foulger, M., Gledhill, S., McDonald, A.T., Walker, A., 2009. Drain-blocking techniques on blanket peat: A framework for best practice. Journal of Environmental Management 90, 3512-3519.

Bain, C.G., Bonn, A., Stoneman, R., Chapman, S., Coupar, A., Evans, M., Gearey, B., Howat, M., Joosten, H., Keenleyside, C., Labadz, J., Lindsay, R., Littlewood, N., Lunt, P., Miller, C.J., Moxey, A., Orr, H., Reed, M., Smith, P., Swales, V., Thompson, D.B.A., Thompson, P.S., Van de Noort, R., Wilson, J.D., Worrall, F., 2011. IUCN UK Commission of Inquiry on Peatlands. IUCN UK Peatland Programme, Edinburgh.

Baird, A.J., Beckwith, C.W., Waldron, S., Waddington, J.M., 2004. Ebullition of methane-containing gas bubbles from near-surface *Sphagnum* peat. Geophysical Research Letters 31, L21505.

Belyea, L.R., Baird, A.J., 2006. Beyond "The Limits of Peat Bog Growth": Cross-scale Feedback in Peatland Development. Ecological Monographs 76, 299-322.

Best, E.P.H., 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.

Billett, M.F., Charman, D.J., Clark, J.M., Evans, C.D., Evans, M.G., Ostle, N.J., Worrall, F., Burden, A., Dinsmore, K.J., Jones, T., McNamara, N.P., Parry, L., Rowson, J.G., Rose, R.,

2010. Carbon balance of UK peatlands: current state of knowledge and future research challenges. Climate research 45, 13-29.

Carroll, M.J., Dennis, P., Pearce-Higgins, J.W., Thomas, C.D., 2011. Maintaining northern peatland ecosystems in a changing climate: effects of soil moisture, drainage and drain blocking on craneflies. Glob. Change Biol. 17, 2991-3001.

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

Denman, K.L., Brasseur, G., Chidthaisong, A., Ciais, P., Cox, P.M., Dickinson, R.E., Hauglustaine, D., Heinze, C., Holland, E., Jacob, D., Lohmann, U., Ramachandran, S., da Silva Dias, P.L., Wofsy, S.C., Zhang, X., 2007. Couplings Between Changes in the Climate System and Biogeochemistry. *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Dias, A.T.C., Hoorens, B., Van Logtestijn, R.S.P., Vermaat, J.E., Aerts, R., 2010. Plant Species Composition Can Be Used as a Proxy to Predict Methane Emissions in Peatland Ecosystems After Land-Use Changes. Ecosystems 13, 526-538.

Dinsmore, K.J., Skiba, U.M., Billett, M.F., Rees, R.M., 2009. Effect of water table on greenhouse gas emissions from peatland mesocosms. Plant and Soil 318, 229-242.

Dunn, C., Freeman, C., 2011. Peatlands: our greatest source of carbon credits? Carbon Management 2, 289-301.

Ellis, C.J., Tallis, J.H., 2001. Climatic control of peat erosion in a North Wales blanket mire. New Phytologist 152, 313-324.

Evans, C.D., Jones, T.G., Burden, A., Ostle, N., Zieliński, P., Cooper, M.D.A., Peacock, M., Clark, J.M., Oulehle, F., Cooper, D., Freeman, C., 2012. Acidity controls on dissolved organic carbon mobility in organic soils. Glob. Change Biol. 18, 3317-3331.

Freeman, C., Ostle, N., Kang, H., 2001. An enzymic 'latch' on a global carbon store - A shortage of oxygen locks up carbon in peatlands by restraining a single enzyme. Nature 409, 149-149.

Gray, A., Levy, P.E., Cooper, M.D.A., Jones, T., Gaiawyn, J., Leeson, S.R., Ward, S.E., Dinsmore, K.J., Drewer, J., Sheppard, L.J., Ostle, N.J., Evans, C.D., Burden, A., Zieliński, P., 2013. Methane indicator values for peatlands: a comparison of species and functional groups. Glob. Change Biol. 19, 1141-1150.

Green, S., Baird, A., 2012. A mesocosm study of the role of the sedge *Eriophorum angustifolium* in the efflux of methane—including that due to episodic ebullition—from peatlands. Plant and Soil 351, 207-218.

Green, S., Boardman, C., Baird, A., Gauci, V., 2011. Investigation of peatland restoration (grip blocking) techniques to achieve best outcomes for methane and greenhouse gas emissions/balance. Controlled Environment (Mesocosm Experiment. Final Report to Defra Project code SP1202, Defra.

Greenup, A.L., Bradford, M.A., McNamara, N.P., Ineson, P., Lee, J.A., 2000. The role of *Eriophorum vaginatum* in CH_4 flux from an ombrotrophic peatland. Plant and Soil 227, 265-272.

Haapalehto, T.O., Vasander, H., Jauhiainen, S., Tahvanainen, T., Kotiaho, J.S., 2011. The Effects of Peatland Restoration on Water-Table Depth, Elemental Concentrations, and Vegetation: 10 Years of Changes. Restoration Ecology 19, 587-598.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. IGES.

IPCC, 2013. 2013 Supplement to the 2006 IPCC guidelines for national greenhouse gas inventories: Wetlands. Methodological guidance on lands with wet and drained soils, and constructed wetlands for wastewater treatment. IGES, Japan.

JNCC, 2011 Towards an assessment of the state of UK Peatlands, Joint Nature Conservation Committee report No. 445, Peterborough.

Joosten, H. 2010. The global peatland CO_2 picture: Peatland status and drainage related emissions in all countries of the world. Wetlands International, Ede (www.wetlands.org)

Kalcska, M., Arroyo-Mora, J.P., de Gea, J., Snirer, E., Herzog, C., Moore, T.R., 2013. Videographic analysis of eriophorum vaginatum spatial coverage in an ombotrophic bog, Remote Sens. 5, 6501-6512.

Levy, P.E., Burden, A., Cooper, M.D.A., Dinsmore, K.J., Drewer, J., Evans, C., Fowler, D., Gaiawyn, J., Gray, A., Jones, S.K., Jones, T., McNamara, N.P., Mills, R., Ostle, N., Sheppard, L.J., Skiba, U., Sowerby, A., Ward, S.E., Zieliński, P., 2012. Methane emissions from soils: synthesis and analysis of a large UK data set. Glob. Change Biol. 18, 1657-1669.

Levy, P.E., Gray, A., Leeson, S.R., Gaiawyn, J., Kelly, M.P.C., Cooper, M.D.A., Dinsmore, K.J., Jones, S.K., Sheppard, L.J., 2011. Quantification of uncertainty in trace gas fluxes measured by the static chamber method. Eur. J. Soil Sci. 62, 811-821.

Livingston, G.P., Hutchinson, G.L., 1995. Enclosure-based measurement of trace gas exchange: applications and sources of error. In: Matson P.A., R.C., H. (Eds.), Biogenic trace gases: measuring emissions from soil and water. Blackwell, Oxford, pp. 14-51.

Lynas, B.D.T., 1973. The Cambrian and Ordovician rocks of the Migneint area, North Wales. Journal of the Geological Society 129, 481–503.

Marinier, M., Glatzel, S., Moore, T.R., 2004. The role of cotton-grass (*Eriophorum vaginatum*) in the exchange of CO_2 and CH_4 at two restored peatlands, eastern Canada. Ecoscience 11, 141-149.

McNamara, N.P., Plant, T., Oakley, S., Ward, S., Wood, C., Ostle, N., 2008. Gully hotspot contribution to landscape methane (CH_4) and carbon dioxide (CO_2) fluxes in a northern peatland. Sci. Total Environ. 404, 354-360.

Myhre, G.,D. Shindell, F.-M. Bréon, W. Collins, J. Fuglestvedt, J. Huang, D. Koch, J.-F. Lamarque, D. Lee, B. Mendoza, T. Nakajima, A. Robock, G. Stephens, T. Takemura, Zhang, H., 2013. Anthropogenic and Natural Radiative Forcing. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

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

Petrescu, A., Van Beek, L., Van Huissteden, J., Prigent, C., Sachs, T., Corradi, C., Parmentier, F., Dolman, A., 2010. Modeling regional to global CH_4 emissions of boreal and arctic wetlands. Glob. Biogeochem. Cycle 24.

Pinheiro, J.C., and Bates, D.M. 2000. Mixed-Effects Models in S and S-PLUS, Springer, New York.

Reed, M.S., Bonn, A., Evans, C., Joosten, H., Bain, B., Farmer, J., Emmer, I., Couwenberg, J., Moxey, A., Artz, R., Tanneberger, F., von Unger, M., Smyth, M., Birnie, R., Inman, I., Smith, S., Quick, T., Cowap, C., Prior, S., 2013. Peatland Code Research Project Final Report, Defra, London.

Rodwell, J.S., 1998. British Plant Communities: Mires and Heaths. Cambridge University Press.

Roulet, N.T., Moore, T.R., 1995. The effect of forestry drainage practices on the emission of methane from northern peatlands. Canadian Journal of Forest Research 25, 491-499.

Schrier-Uijl, A.P., Veraart, A.J., Leffelaar, P.A., Berendse, F., Veenendaal, E.M., 2011. Release of CO_2 and CH_4 from lakes and drainage ditches in temperate wetlands. Biogeochemistry 102, 265-279.

Ström, L., Tagesson, T., Mastepanov, M., Christensen, T.R., 2012. Presence of *Eriophorum scheuchzeri* enhances substrate availability and methane emission in an Arctic wetland. Soil Biology and Biochemistry 45, 61-70.

Sundh, I., Nilsson, M., Mikkela, C., Granberg, G., Svensson, B.H., 2000. Fluxes of methane and carbon dioxide on peat-mining areas in Sweden. Ambio 29, 499-503.

Teh, Y.A., Silver, W.L., Sonnentag, O., Detto, M., Kelly, M., Baldocchi, D.D., 2011. Large Greenhouse Gas Emissions from a Temperate Peatland Pasture. Ecosystems 14, 311-325.

Tuittila, E.S., Komulainen, V.M., Vasander, H., Nykanen, H., Martikainen, P.J., Laine, J., 2000. Methane dynamics of a restored cut-away peatland. Glob. Change Biol. 6, 569-581.

Waddington, J.M., Day, S.M., 2007. Methane emissions from a peatland following restoration. Journal of Geophysical Research-Biogeosciences 112.

Waddington, J.M., Strack, M., Greenwood, M.J., 2010. Toward restoring the net carbon sink function of degraded peatlands: Short-term response in CO_2 exchange to ecosystem-scale restoration. Journal of Geophysical Research-Biogeosciences 115.

Ward, S., Bardgett, R.D., McNamara, N.P., Adamson, J.K., Ostle, N.J., 2007. Long-term consequences of grazing and burning on northern peatland carbon dynamics. Ecosystems 10, 1069-1083.

Wilson, D., Alm, J., Laine, J., Byrne, K.A., Farrell, E.P., Tuitilla, E-S. Rewetting of cutaway peatlands: Are we re-creating hot spots of methane emissions? Restoration Ecology 17, 796-806.

Wilson, D., Farrell, C., Mueller, C., Hepp, S., Renou-Wilson, F., 2013. Rewetted industrial cutaway peatlands in western Ireland: a prime location for climate change mitigation? Mires and Peat 11, http://www.mires-and-peat.net/map11/map_11_01.htm.

Yu, Z.C., Loisel, J., Brosseau, D.P., Beilman, D.W., Hunt, S.J., 2010. Global peatland dynamics since the Last Glacial Maximum. Geophys. Res. Lett. 37.

Table 1. Annual mean water table depth (± standard error for locations with multiple dipwells), and porewater pH, DOC, SO_4 concentrations measured at 20 cm depth. 'Number of samples' corresponds to total number of pore water chemistry samples used to calculate each mean value. Water table was recorded manually from one dipwell per plot at the time of sampling.

Sampling location (plot codes)	**Water table (cm)**	**pH**	**DOC (mg l^{-1})**	**SO_4 (mg l^{-1})**	**Number of piezometers**	**Number of samples**
Drained site (Llyn Serw)						
Blanket bog, upslope (A-C)	5.5 ± 0.7	4.91	71.4	1.00	3	28
Within ditch (G)	*At surface*	4.81	30.3	0.64	1*	10
Blanket bog, downslope (D-F)	14.7 ± 2.6	4.84	77.0	1.86	3	21
Re-wetted site (Llyn Serw)						
Blanket bog, upslope (A-C)	3.0 ± 0.5	4.94	80.7	0.90	3	30
Within ditch unvegetated (G-H)	*At surface*	4.88	35.3	0.38	1	10
Within ditch vegetated (G-H)	*At surface*	4.92	52.1	0.97	1	10
Blanket bog, downslope (D-F)	2.7 ± 0.3	4.90	48.8	0.54	3	30
Undrained reference site (Nant y Brwyn)						
Blanket bog, undrained	1.0	4.67	44.1	1.91	2	17
Within ditch, *Sphagnum*	1.2	4.59	27.6	0.86	2	18

*Within-ditch sample from the open ditch was collected directly from surface water using a syringe.

Table 2. Annual mean CH_4 fluxes (± standard error) expressed as mg CH_4 m^{-2} day^{-1} and kg CH_4 ha^{-1} yr^{-1}, together with the number of collars within each category for which measurements were made, and the number of individual chamber tests for which it was possible to calculate fluxes after screening.

Sampling location (plot codes)	CH_4 (mg CH_4 m^{-2} day^{-1})	CH_4 (kg CH_4 ha^{-1} yr^{-1})	Number of collars	Number of chamber tests
Unblocked ditch (Llyn Serw)				
Blanket bog, upslope (A-C)	15.1 ± 2.0	55.3 ± 7.3	6	111
Within ditch (G)	16.3 ± 5.6	59.7 ± 20.6	4	24
Blanket bog, downslope (D-F)	8.8 ± 1.2	32.1 ± 4.4	6	108
Blocked ditch (Llyn Serw)				
Blanket bog, upslope (A-C)	23.7 ± 8.1	86.7 ± 29.5	6	102
Within ditch unvegetated (G-H)	10.3 ± 2.7	37.7 ± 10.0	4	17
Within ditch vegetated (G-H)	197.0 ± 31.1	719.5 ± 113.5	4	28
Blanket bog, downslope (D-F)	16.3 ± 3.9	59.6 ± 14.1	6	98
Reference site (Nant y Brwyn)				
Blanket bog, undrained	16.7 ± 2.5	61.1 ± 9.2	8	89
Within ditch, *Sphagnum*	13.9 ± 7.5	50.7 ± 27.2	4	39

Table 3. Estimated area occupied by each land-cover category within the target area shown in Figure 1, and estimated CH_4 emissions, for a sequence of pre-, during- and post-drainage scenarios

	Drainage Scenario						
Area occupied by each land-cover category (%)	**1) Pre-drainage**	**2) Drained**	**3) Recently re-wetted**	**4a) Long-term re-wetted 1**	**4b) Long-term re-wetted**	**4b) Long-term rewetted 3**	**CH_4 flux by land-cover category (kg CH_4 ha^{-1} yr^{-1})**
Blanket bog (undrained)	100.0					85.9	61.1
Blanket bog (drained)		92.9					43.7
Blanket bog (re-wetted)			85.9	85.9	85.9		74.4
Active ditch		7.1					43.1
Infilled ditch (bare peat)			7.1				37.7
Infilled ditch (*Eriophorum*)			7.1	14.1			719.5
Infilled ditch (*Sphagnum*)					14.1	14.1	50.7
Landscape CH_4 flux by scenario (kg CH_4 ha^{-1} yr^{-1})	61.1	43.7	117.4	165.6	71.0	59.6	

FIGURE CAPTIONS

Figure 1. Lidar hillshade image of the Llyn Serw study site, part of the Migneint blanket bog in North Wales, UK. Two open ditches (running Southeast to Northwest) are visible as sharp linear features, whilst the infilled ditches running on the same trajectory appear as shallower, broader features. Ditches running from Northeast to Southwest are older, shallower and partly infilled, but continue to have exert some influence on water table in the Southwestern part of the site. Measurement transects are also shown, along with the 'target area' of relatively homogenous drained bog used for upscaling.

Figure 2. Indicative surface elevation (thinner black line, derived from LiDAR cross-sections) for the two Llyn Serw sampling transects, showing infilled and open ditches in cross-section. Ditches run diagonally across the hillslope. Static chamber sampling locations between ditches (A-E), and within ditches (G-H) are shown. Water table elevation (thicker blue line) is approximate, based on mean annual water table depth relative to the LiDAR surface at dipwells located within the bog, and observations of water table at or slightly above the surface in the infilled and open ditches respectively. The two transects are separated by a buffer section of approximately 40 m, including a blocked ditch (see Figure 1).

Figure 3. Recolonisation of *Eriophorum vaginatum* on an infilled ditch (photograph taken in July 2010, 24 months after infilling took place). Note presence of bare peat on left of ditch.

Figure 4. Seasonal and spatial variations in a) mean water table depth from undrained, drained and re-wetted *Calluna*-*Sphagnum* blanket bog; b) mean CH_4 flux from the same locations, c) mean CH_4 flux from active and infilled ditches, and. Error bars indicate standard

error among replicate sampling collars and dipwells within each category (note that standard errors could not be calculated from water table in undrained *Calluna-Sphagnum* bog, as only one dipwell was deployed here).

Figure 5. Mean annual CH_4 flux for each sampling collar versus a) water table, b) *Eriophorum* cover and c) number of *Eriophorum* spikelets recorded in each collar during June 2011.

Figure 1

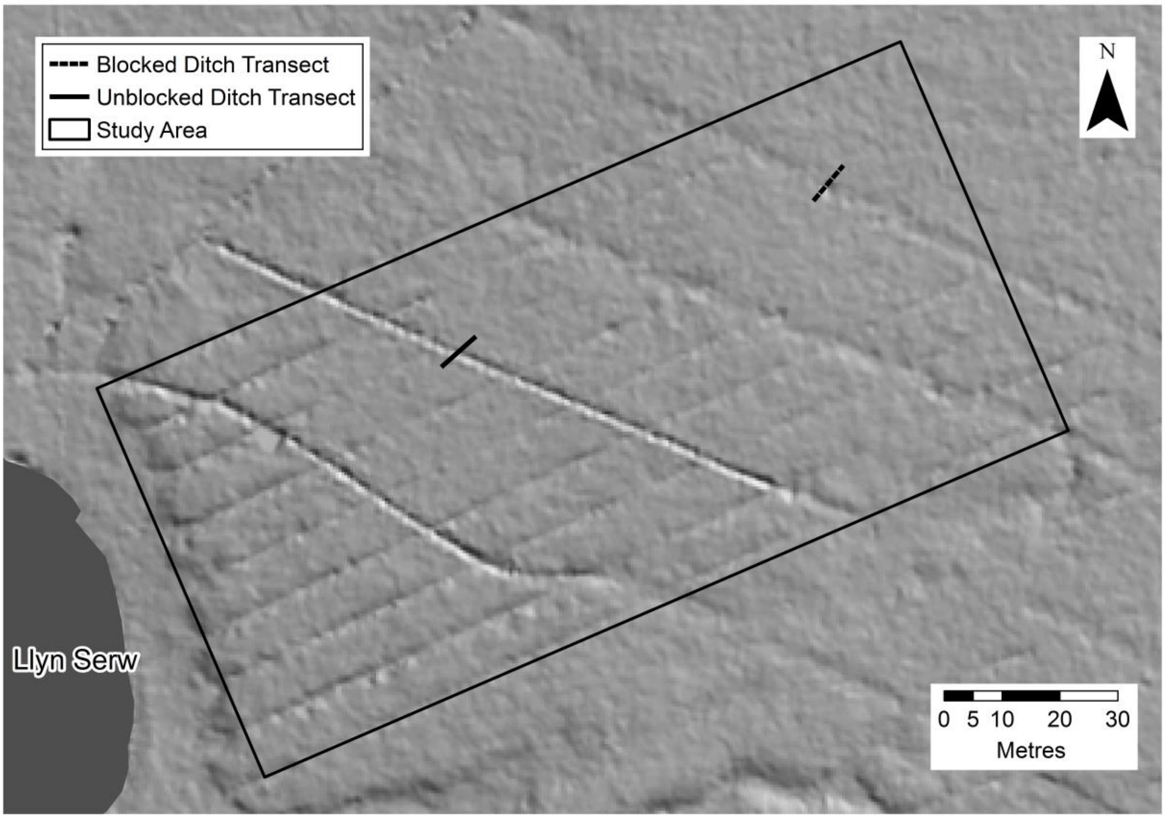

Blocked Ditch Transect
Unblocked Ditch Transect
Study Area
N
Llyn Serw
0 5 10 20 30
Metres

Figure 2

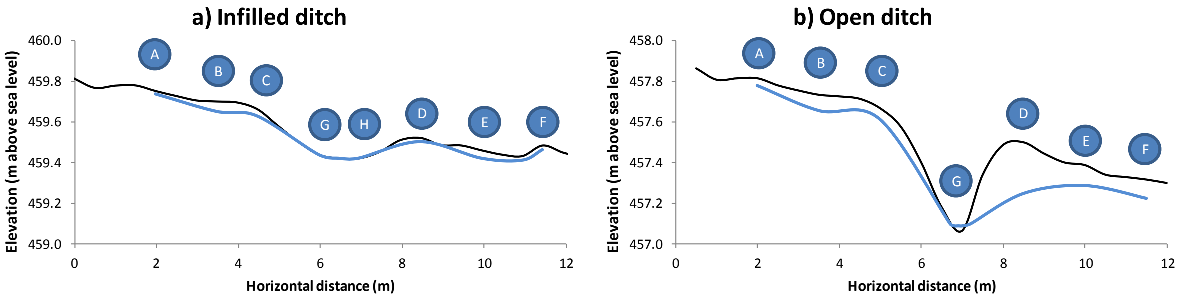

a) Infilled ditch
b) Open ditch
Elevation (m above sea level)
Horizontal distance (m)
A
B
C
D
E
F
G
H
460.0
459.8
459.6
459.4
459.2
459.0
458.0
457.8
457.6
457.4
457.2
457.0
0
2
4
6
8
10
12

Figure 3

Figure 4

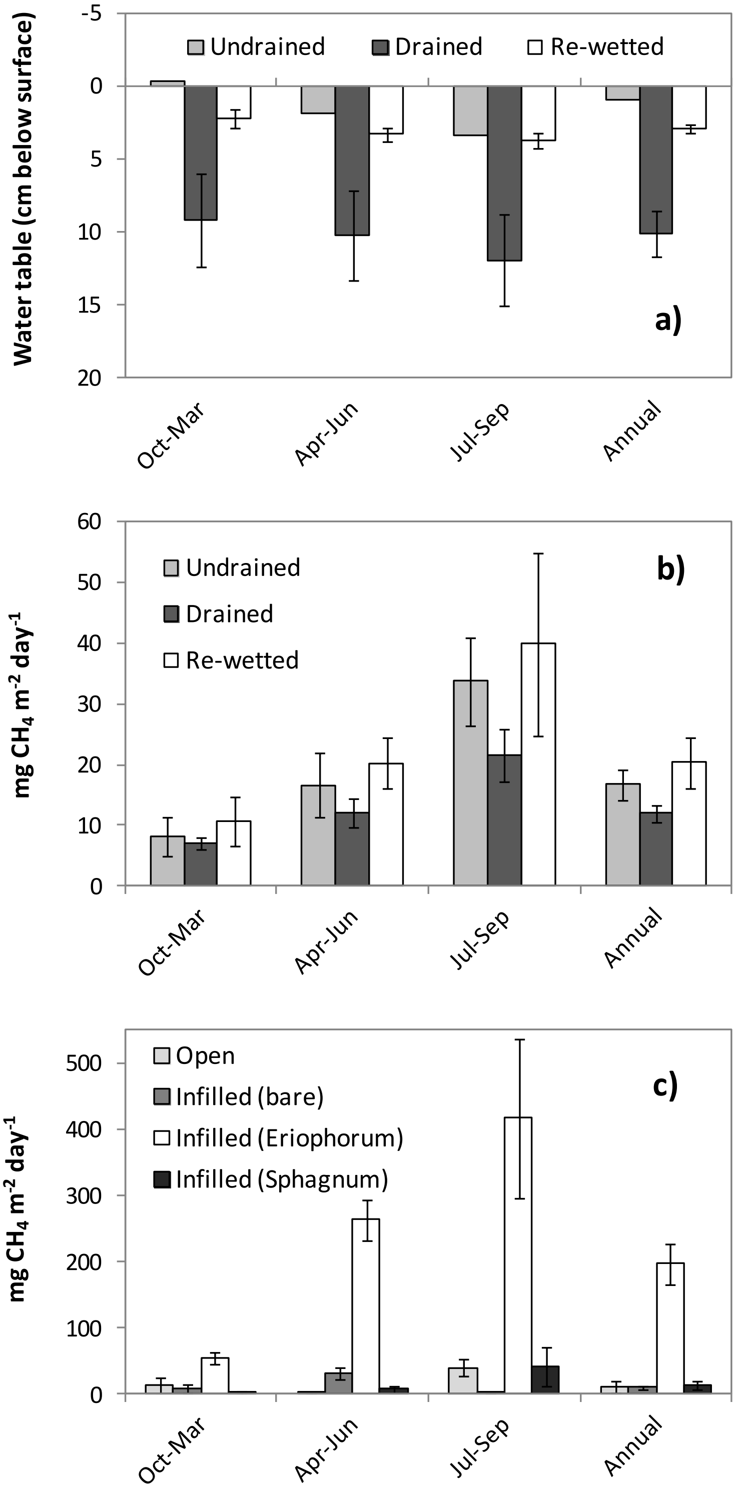
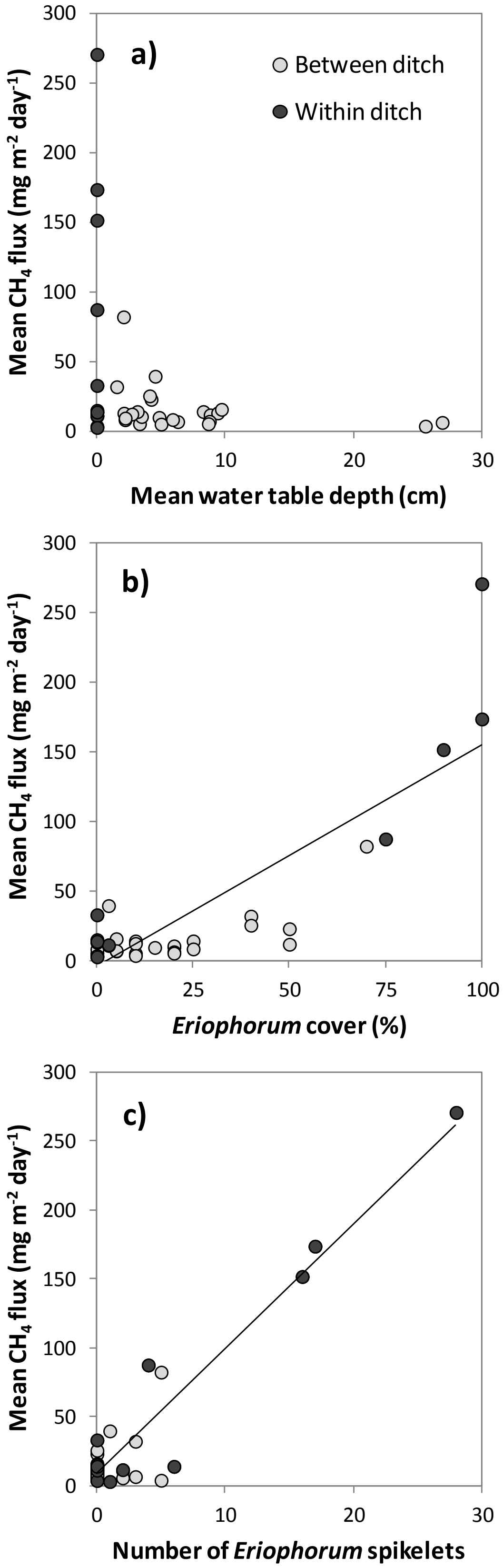

Water table (cm below surface)
-5
0
5
10
15
20
Undrained
Drained
Re-wetted
a)
Oct-Mar
Apr-Jun
Jul-Sep
Annual
mg CH_4 m^{-2} day^{-1}
60
50
40
30
20
10
0
Undrained
Drained
Re-wetted
b)
Oct-Mar
Apr-Jun
Jul-Sep
Annual
mg CH_4 m^{-2} day^{-1}
500
400
300
200
100
0
Open
Infilled (bare)
Infilled (Eriophorum)
Infilled (Sphagnum)
c)
Oct-Mar
Apr-Jun
Jul-Sep
Annual

Figure 5



a)
Between ditch
Within ditch
Mean CH_4 flux (mg m^{-2} day^{-1})
Mean water table depth (cm)
b)
Eriophorum cover (%)
c)
Number of *Eriophorum* spikelets