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Title Page

Title: Carbon balance of a restored and cutover raised bog: Comparison to global trends

Running Head: C BALANCE Of A RESTORED AND CUTOVER BOG

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Abstract

All major aspects of the carbon balance – net ecosystem exchange (NEE), CH_4 flux, losses of dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC), and open water CO_2 evasion – were measured for several distinct ecotypes in a restored unharvested raised bog and an adjacent historically abandoned cutover bog over a two year period. The average annual ecotype carbon balance at the Sub-Central ecotype, with eco-hydrological characteristics most similar to a high quality raised bog, was the largest net carbon sink of -32 ±65 g C m^{-2} yr^{-1} while the Calluna Cutover ecotype, with the characteristics of a well-drained peatland site was the largest net carbon source of 239 ±83 g C m^{-2} yr^{-1}. The annual carbon balance from all ecotype study locations was found to be controlled by mean annual water table (MAWT). Also, significant negative correlation was observed between the plot global warming potential and percent *Sphagnum* moss cover, highlighting the importance of regenerating this keystone genus as a climate change mitigation strategy in peatland restoration. The data from this study was then compared to the rapidly growing number of peatland carbon balance studies across Boreal and Temperate regions. The trend in NEE and CH_4 flux with respect to MAWT was compared for the five ecotypes in this study was and literature data from degraded/restored peatlands, intact peatlands, and bare peat sites.

1. Introduction

Peatlands are important to the global carbon cycle as they act as significant stores of carbon (C) and sources or sinks of carbon dioxide (CO_2) and methane (CH_4) (Gorham 1991). Despite covering only ~3% of the earth's terrestrial surface, it is estimated that between 500 and 700 billion tonnes of C are stored as organic soil within the global peatland expanse (Yu et al., 2010; Paige and Baird, 2016; Leifeld and Menichetti. 2018). However, at present, human activity is either draining or mining ~10% of global peatlands, transforming them from long-term C sinks into sources (Joosten, 2010; Leifeld and Menichetti. 2018). In Europe, a high percentage (~46%) of the remaining peatlands are degraded to the point whereby peat is no longer actively being formed (Tanneberger et al. 2017),

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and in Ireland whilst ~20% of the land area is peatland, over 95% of it has been degraded through anthropogenic activities such as drainage for agriculture, forestry and peat extraction (Connolly and Holden, 2009; Connolly and Holden, 2017).

The carbon cycle and greenhouse gas (GHG) dynamics of degraded peatlands is often substantially different compared to pristine peatlands (Blodau, 2002; Baird et al., 2009) making them significant with respect to national and global GHG budgets and emission reporting (Wilson et al. 2013; Billet et al. 2010). Moreover, degraded peatlands can continue to emit C for decades to centuries following drainage, and current estimates are that degraded peatlands store globally ~80.8 Gt soil C and emit ~1.91 (0.31–3.38) Gt CO_2-eq. yr^{-1} (Leifeld and Menichetti. 2018). Soil carbon sequestration through peatland restoration is increasingly recognized as an important strategy to tackle climate change (Dise, 2009; Leifeld and Menichetti. 2018), and in recent years there has been a substantial increase in money being invested in peatland projects across the world (Anderson *et al*., 2017). With the increase in global active peatland management, there is a need for more studies examining how drainage and restoration alters the eco-hydrology of degraded peatlands systems and their carbon balances (Baird et al., 2009; Young et al., 2017).

The land atmosphere CO_2 flux, or net ecosystem exchange (NEE) in peatlands is related to water table level, as inundation creates anaerobic conditions which suppresses the decomposition of soil organic matter (Lain et al., 1996). This can result in a net CO_2 sink (negative NEE) whereas a low water table can result in a net CO_2 source (positive NEE). Thus, water table has been correlated to spatial (Strack et al., 2014; Junkurst and Fielder, 2007; Silvola et al., 1996) and temporal (McVeigh et al. 2014; Peichl et al., 2014; Lund et al., 2012; Strachan et al., 2016; Helftler et al., 2015) variation in the NEE of both intact and degraded peatlands. However, anaerobic conditions due to a high water table can also increase the land atmosphere CH_4 flux (Frenzel and Karofeld 2000). Both NEE and CH_4 flux are also affected by plant ecology, as the extent of aerenchymatous vegetation cover such as *Eriophorum spp.* is correlated with increased CH_4 flux (Cooper et al. 2014; Frenzel and Karofeld 2000, Waddington and Day 2007, McNamera et al. 2008, Gray et al. 2013). *Sphagnum spp.*, however, often exhibit lower CH_4 fluxes (Frenzel and Rudolph et al. 1998) due to a symbiotic relationship with methanotrophic bacteria (Raghoebarsing et al. 2005). Also, *Sphagnum spp.* coverage may correspond to an increase in the CO_2 sink function of natural sites (Strack et al. 2016) as much of the peat in northern peatlands is derived from this genus (Vitt et al., 2000, Bacon et al., 2017). Furthermore, the extent of vegetation cover is an important factor affecting the NEE (Waddington and Day, 2010; Tuitili et al., 1999, Strack et al., 2016). This is relevant to degraded and restored peatlands because harvested peatlands can have large areas of bare peat (Wilson et al., 2015).

Climatic variables such as the frequency of cloudiness, temperature, and length of growing season have also been found to be important controlling factors of NEE (Charman et al., 2013; Zhaojun et al., 2011; McVeigh et al., 2014; Helftler et al., 2015). However, climate variables cannot be controlled at a specific site, and therefore, may not be as relevant when considering climate change mitigation actions.

Although N_2O emissions can be an important aspect of the GHG emissions from organic soils (Pärn et al. 2018), this study focuses only on aspects of the carbon balance. In low nutrient, semi-natural sites like in this study, N_2O emissions are typically low (Haddaway et al., 2014) but can be higher for deeply drained (Vanselow-Algan et al., 2015) or high nutrient sites (Danevčič et al., 2010). The radiative impact of different GHGs can be normalized by converting them into a CO_2 equivalents in terms of the 100-year global warming potential (GWP) in tonnes CO_2-eq ha^{-1} yr^{-1}: over a hundred year horizon, CO_2 = 1, CH_4 = 34, and N_2O = 298 (after Wilson et al., 2016b from IPCC 2014).

Intact peatlands are a net CO_2 sink [typical annual average NEE range -31.9 to -66 g C-CO_2 m^{-2} yr^{-1}, from Helftler et al., (2015)] and a CH_4 source [average of 9.2 g C-CH_4 m^{-2} yr^{-1}, (95% CI 0.3 to 44.5 g C-CH_4 m^{-2} yr^{-1}) for low nutrient temperate peatlands from Wilson et al., (2016a)]. By contrast, drained peatlands are a CO_2 source [the average annual NEE of +81 to +151 g C-CO_2 m^{-2} yr^{-1} reported in Renou-Wilson et al., (2018a) is typical] with very low CH_4 emissions (Baird et al., 2009). However, it should be noted that this can be offset by high methane emissions from active drains of ~60 g CH_4 m^{-2} yr^{-1} (Evans et al., 2016). Degraded/drained peatlands typically have a larger GWP compared to natural sites or rewetted sites because a large positive NEE outweighs the reduced CH_4 emissions (Renau-Wilson et al., 2018a). The NEE and CH_4 fluxes from restored peatlands can be similar to pristine peatlands, but exhibit greater variability (Wilson et al., 2016a; Strack et al., 2016).

Several studies have suggested the hypothesis that time since restoration is an important factor in the GWP of peatlands (Augustin & Joosten, 2007; Bain et al., 2011; Waddington and Day 2007). In particular, the restored sites may go through an initial period of high methane production and high GWP because restored peatlands are often rapidly colonized by aerenchymatous vegetation, such as *Eriophorum spp.* (Waddington and Day 2007, Cooper et al. 2014). This is followed by a period of decreasing GWP as mosses and other peatland species become established (Augustin & Joosten, 2007; Bain et al., 2011). To test this hypothesis, more data is needed for peatlands "restored more than 10 years previously" (Bacon et al., 2017). Also, it is valuable to have studies which directly compare adjacent raised bog and cutover bog with different site histories.

Aquatic losses of carbon include dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) in runoff as well as CO_2 evasion from open water. These have not been measured as frequently as NEE and CH_4 flux (Dinsmore et al., 2010), but can represent a key component of the net ecosystem carbon budget (NECB) (Kindler et al., 2011). Ignoring the aquatic carbon losses would result in an overestimate of the carbon sink function of peatlands (Billet et al. 2010). The DOC losses from temperate peatlands range from 5-36 g C m^{-2} yr^{-1} and are lower for boreal peatlands (Range: 4-13 g C m^{-2} yr^{-1}) (Evans et al., 2016). Few studies have simultaneously measured a complete NECB for a peatland including the DIC flux (Nilsson et al. 2008) and CO_2 evasion from open water (Dinsmore et al. 2010), even though CO_2 evasion has been found to be important to the overall carbon balance, with a reported 2-year average of 12.8 g C m^{-2} yr^{-1} for an intact peatland in Scotland (Dinsmore et al. 2010). Further, these studies have focused on intact rather than restored or recovering peatlands.

The growing body of scientific research on the GHG and carbon balance of peatlands and the importance to global climate change means that it is increasingly important to consider new data in the context of global studies. Often, boreal and temperate peatlands have similar conditions: hydrological (consistently high water table), chemical (high carbon, often acidic peat soils), ecological (often ground cover of *Sphagnum* mosses, with low nutrient sedges and ericaceous shrubs). As similar factors (i.e. water table, plant ecology, growing season length, soil temperature, etc.) are often cited as controlling factors for greenhouse gas fluxes, it may be possible to identify global trends across boreal and temperate peatlands (e.g. Junkurst and Fieldler 2007).

The goal of this work is to quantify all of the major aspects of the carbon balance (NEE, CH_4 flux, and aquatic losses as DOC, DIC, and CO_2 evasion), for a historically (ca. 1960) abandoned cutover bog compared to an adjacent more recently restored (2009) raised bog. This study also presents the measurements in the context of global studies on boreal and temperate peatlands with the aim of identifying trends in NEE and CH_4 flux based on land condition (drained, restored, pristine), mean annual water table, and vegetation cover (presence/lack of vegetation).

2. Materials and Methods

2.1 Site Description

Abbeyleix Bog (N 52.89714, W 7.35022, elevation approx. 90 m) is a natural peatland area in Co. Laois, Ireland containing both un-harvested raised bog and historically harvested cutover bog (Fig. 1). This site is located in a temperate, oceanic climate with a mean annual rainfall of 844 mm and a mean annual temperature of 9.9° C. Acidic, low nutrient, histosol, peat soils remain throughout the raised and cutover bog with 5.0-8.5 m depth on the raised bog and 1-3 m depth on the cutover bog. The areas of raised bog were impacted by surface drainage in the 1980's in preparation for industrial extraction. Surface drains were installed at 15 m spacing to a depth of 1 m, and connected with older, and deeper drains along a historic railway track and the margins of the cutover bog. The plans for industrial extraction of the peat were abandoned due to resistance from the local community, and the surface drains were blocked in 2009, 7 years before the start of this study. The raised bog is mostly surrounded by cutover bog, which was domestically harvested for peat between the 1870's and 1960's, and then abandoned (Ryle, 2013).

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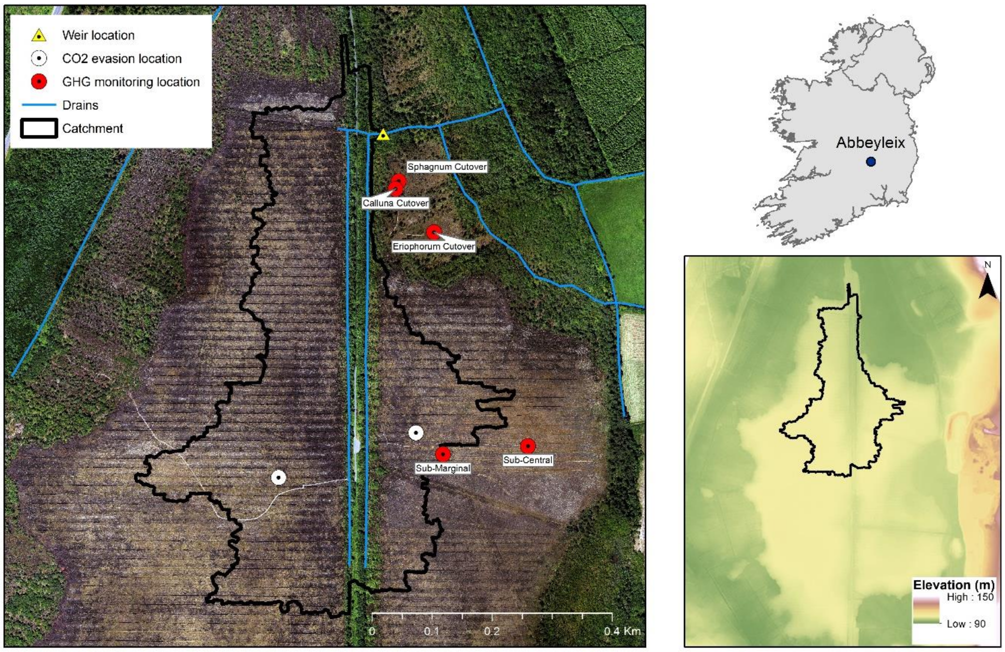

Figure 1. Location of the study site in Ireland; elevation map of Abbeyleix Bog (bottom right) showing the uncut raised bog surrounded by lower cutover bog and the higher esker complex to the east; an aerial photograph of the study site showing the weir catchment area, major drains, and sampling locations. In the aerial photograph the blocked surface drainage network on the raised bog can be seen as a set of horizontal lines and the historic railroad track can be seen as a vertical line through the middle of the photograph.

2.2 Sampling Locations

Five sampling locations were chosen to quantify GHG emissions, two on the uncut raised bog and three on the cutover bog. These locations were chosen to represent 5 ecotypes, where the ecotype refers to a distinct set of hydro-physical and ecological conditions. These 5 areas were chosen to represent common ecotypes on raised and cutover bogs in Ireland with the help of ecologists from the National Parks and Wildlife Service (NPWS).

On the raised bog, one study location was chosen in a Sub-Central ecotype, which is defined as having a continuous *Sphagnum spp.* cover and continuously high water table but lacking the micro-topography of hummocks and hollows. The Sub-Central ecotype is the highest quality bog conditions found at this site. Another study location was chosen in a Sub-Marginal ecotype, which is defined as having a discontinuous *Sphagnum spp.* moss cover and a mixed presence of both relatively wet and dry bog vegetation (Table 1). Further description of raised bog ecotypes can be found in Schouten et al. (2002).

On the cutover bog, three sampling locations were chosen based on distinctions in the plant ecology. The Sphagnum Cutover ecotype contains a continuous *Sphagnum spp.* cover (primarily as hummocks of *Sphagnum capilifolium* with some *Sphagnum subnitens and Sphagnum magellanicum*) and a mixture of plant species similar to the Sub-Central ecotype. The Calluna Cutover ecotype contains a low diversity of plant species characteristic of a well-drained peat soil, dominated by heather (*Calluna vulgaris*), bare peat, and lichens (mostly *Cladonia portenosa*) similar to a facebank ecotype on a raised bog. The Eriophorum Cutover ecotype is dominated by *Eriophorum angustifolium*, and contains a moderate percent (21-54% in this study) cover of *Sphagnum spp.* (Table 1). All sampling locations were chosen in open areas, excluding any large trees, shrubs or other vegetation that could not fit under the gas sampling chambers (see Section 2.3). Six collars were installed for each ecotype except for the Calluna Cutover ecotype where 5 collars were installed. Collar locations were chosen to represent ecological variability within each ecotype. Plant ecology was characterized for all collars in June 2016 and again in June 2017 with the help of ecologists from the NPWS. The plant ecology was determined in terms of the percent cover of every species present, averaged over the two years.

Table 1. Summary of the plant ecology for each ecotype in this study. Data is reported as the mean (range) of the collars within each ecotype.

Ecotype	Percent *Spahgnum spp.* cover	Percent *Eriophorium spp.* cover	Percent *Calluna vulgaris* cover	Percent Total Plant Cover
Sphagnum Cutover	94 (78 to 100)	8 (3 to 23)	16 (5 to 30)	119 (103 to 134)
Calluna Cutover	0	2 (0 to 3)	35 (8 to 50)	51 (18 to 68)
Eriophorium Cutover	35 (21 to 54)	51 (21 to 80)	6 (2 to 15)	103 (77 to 140)

Sub-Marginal	57 (15 to 89)	13 (4 to 37)	9 (2 to 15)	100 (69 to 114)
Sub-Central	98 (93 to 100)	8 (1 to 39)	2 (0 to 8)	124 (107 to 151)

2.3 Meteorological Field Data

On site, hourly measurements of air temperature and humidity (CS215 probe, Campbell Scientific, Loughborough, UK), rainfall (ARG100 Tipping Bucket Raingauge, Campbell Scientific), barometric pressure (PTB110 Barometer, Vaisala, Oyj, Finland), and soil temperature at 5 and 10 cm (PT100 temperature probes, Campbell Scientific) were recorded by a CR1000 Data logger (Campbell Scientific). Soil temperature was also recorded at ecotypes by two LogBoxAA data loggers (Novus, Miami, USA). Hourly phreatic water table was recorded in 5 cm diameter stilling wells located at each of the five ecotypes by an Orphius Mini Level Logger (vented transducer, 0.1% error, OTT Hydromet, Kempten, Germany). The ground elevation at the center of each collar was surveyed and compared to the stilling well using an RTK GPS with ± 2mm accuracy (TDL 450L, Trimble, Sunnyvale, CA), and the hourly water table at each collar was offset by this difference in elevation. All collars were located within 8 m of the ecotype water table logger.

The hourly light intensity was measured in the field in units of W/m^2 using an LP02 Pyranometer (Hukseflux Thermal Sensors, Delft, Netherlands). This sensor was calibrated to the photosynthetically active radiation (PPFD) sensor (TPR-2, PP Systems) in units of (μmol m^{-2} s^{-1}) used during the field chamber measurements. A linear calibration between these two sensors was found for both sunny and overcast days (n=27, r^2=0.82), which was used to convert hourly light intensity to hourly PPFD.

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2.4 Greenhouse Gas Flux Measurements

The closed static chamber method was used to measure greenhouse gas fluxes from all plots, comparable to methods used in a large number of other studies, particularly on peatlands in Ireland (e.g. Wilson et al. 2016b). A stainless steel collar was permanently installed 20 cm into the ground at least two weeks before the start of sampling. This collar had a water trough to ensure a suitable seal with the chamber. The chambers (60 x 60 x 30 cm equipped with a fan) were constructed in house of clear polycarbonate for CO_2 measurements and opaque polystone^{tm} for CH_4. A system of wooden platforms was constructed 6-7 weeks before the start of sampling so that each collar could be accessed without putting pressure on the ground surface adjacent to it. Platforms were placed on piles to the base of the peat in the Sub-Central ecotype to prevent sinking into the bog. For CO_2 flux measurements, chambers were gently set on the collar and any pressure differential between the chamber headspace and the ambient atmosphere was vented using a 5 cm^2 hole set in the side of the chamber. The chamber was then sealed and the CO_2 concentration was recorded in the field every 15 seconds for a period of 105 seconds using an EGM-4 infra-red gas analyser (PP Systems, Amesbury, USA). CO_2 flux was calculated from the slope of the linear increase in CO_2 flux over time. In order to maintain a constant temperature, particularly under high irradiance, a cooling system was installed in this chamber which pumped water from an ice bath through a small radiator located behind the fan to keep the variance of the chamber temperature to within 1°C during the measurement. The CO_2 flux measurement was repeated under a range of light levels by artificially shading the chamber. Ecosystem respiration is assumed to be the CO_2 flux where the light transmitted into the chamber was zero. CO_2 flux measurements were conducted over 63 field days between January 2016 and August 2017. A total of 3358 quality checked chamber measurements for CO_2 flux were conducted over 29 collar locations.

For CH_4 flux measurements, gas samples of 20 mL each were extracted from the chamber every 10 minutes beginning 5 minutes after the chamber had been placed on the collar and sealed. These samples were later analyzed in the lab on an Agilent Gas Chromatograph instrument with a flame ionization detector and a 30 m long Elite-plot Q GC column. Samples were collected over 17 field days between April 2017 and January 2018.

Additionally, the soil temperature at 5 and 10 cm depth, water table adjacent to the collar, air temperature, and light level inside the chamber (for CO_2 flux measurements) were recorded for each chamber closure at the time of sampling.

2.5 NEE Modelling

The NEE was modelled on an hourly basis to account for the expected diurnal variations, which is driven by PPFD and temperature for the daytime uptake and night time release, respectively. Field measurements of CO_2 flux were used to build collar specific empirical models of gross primary production (GPP) and ecosystem respiration (ER). Hourly measurements of field variables were input into these empirical models to calculate hourly GPP and ER, which were then summed to calculate NEE.

Several models of GPP and ER were tested to fit the data (see Supplemental Section 1). These models were judged based on the sum of the squares of the residuals and r^2 values. Models were also checked to ensure that there was no bias or trend in the residuals with respect to independent variables. The GPP model in Eq. (1) was found to best explain the variance in the field data for all of the 29 collars.

$$GPP = -(a + c * \sin((JDAY + 215)/365 * 2\pi)) * \frac{PAR}{PAR+b} * exp(T_{5cm} * d) * (1 + WT * e) \qquad (1)$$

where ***a***, ***b***, ***c***, ***d***, and ***e*** are collar specific empirical fitted model parameters and JDAY is the Julian day of the year, PPFD is the light level in (µmol m^{-2} s^{-1}), T_{5cm} is the soil temperature at 5 cm, and WT is the water level in cm below ground surface at the collar. The r^2 value of the modelled versus measured data using Eq. (1) ranged between 0.77 and 0.94 for each of the 29 collars (Table S3).

For ER, the model in Eq. (2) was found to best explain the variance in the field data for all of the 29 collars. For this ER model, the r^2 values ranged from 0.74 to 0.94 for each of the collars.

$$ER = \left(a + d * \sin\left(\frac{JDAY + 215}{182.5} * 2\pi\right) + e * WT\right) * \exp\left(c * \left(\frac{1}{(283.15 - 227.13)} - \frac{1}{(TK5cm - 227.13)}\right)\right) + b * WT \quad (2)$$

where ***a***, ***b***, ***c***, ***d***, and ***e*** are fitting parameters, and other variables are as above. Fitting parameters and more information on the GPP and ER models tested can be found in Supplemental Section 1.

Hourly water level, T_{5cm}, PAR, and Julian day data were input into Eq. (1) and Eq. (2) (with the collar specific fitting parameters) to calculate hourly GPP and ER at each collar.

2.6 Methane Modelling

In contrast to CO_2 flux, CH_4 fluxes are expected to be much more constant throughout the day (Pypker et al. 2013) with apparently random variation. Therefore, CH_4 fluxes from each collar could be calculated from the average measured flux over a given time period (as in Strack et al. 2014). However, in this case, methane flux measurements were not conducted over the entire 2 year time period because of equipment issues. Thus, a model was constructed for the purpose of extrapolating the field data to the entire study period. The field data of CH_4 flux from all collars were normalized by the collar average CH_4 flux and lumped together to model the average temporal variation in CH_4 flux. The variations in CH_4 flux was modelled according to the Julian day of year and soil temperature (Eq. S11). Due to limited data, methane flux variations were assumed to follow the same temporal trend across all ecotypes. The overall average temporal variation was then multiplied by the average measured methane flux at a given collar. The model gave little difference between 2016 and 2017, and as field data was only collected in 2017, it was assumed that the methane flux from both years was the same for the purposes of calculating annual carbon balance and GWP.

2.7 Aquatic Carbon Losses

A thin plate V-notch weir was installed to measure hourly discharge from a 249,000 m^2 catchment area onsite (as shown in Fig. 1). The weir catchment area was delineated in ARC-GIS using a digital terrain map based on LiDAR survey data from 2013. The majority of this catchment area was composed of marginal and sub-marginal uncut raised bog (>90%) as well as lightly forested drains along a bog road (<10%). Aquatic carbon losses as DOC and DIC were quantified at this location only, and assumed to be the same for all ecotypes (even those adjacent to but outside of this catchment area) given the difficulty in resolving the relative contributions of each ecotype to the total DOC flux. The DOC concentration was measured weekly in 2016 and every 12 hours (with a few gaps) from January through November 2017. DOC samples were filtered in the field using a 0.45 µm cellulose syringe filter after rinsing the syringe and filter with 20 mL of sample. Samples were then acidified to pH 2 using 10% HCl to preserve them and stored under refrigeration at 4° C and analysed within two months. The DOC concentration was measured by UV absorbance as in other studies (e.g. Jager et al. 2008, Koehler et al., 2009) at wavelength 254 nm. A site specific calibration curve was determined between 254 nm UV absorbance and DOC concentration measured using a Vario Total Organic Carbon (TOC) Select Analyzer (Elementar, Langenselbold, Germany). This was undertaken on samples collected from January 2016 to April 2016, July 2016, and July 2017 (r^2 =0.997, n=76). The error of this method was ± 1.1 mg C/L based on the standard

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deviation of the residuals. The hourly discharge at the weir was multiplied by the most recent DOC concentration measurement to calculate a carbon flux as DOC from the catchment. This value was then divided by the catchment area to calculate the aquatic carbon loss as DOC per m^2.

The DIC concentration at the weir was calculated from the aqueous partial pressure of CO_2 as well as the pH and temperature using equations from Gelbrecht et al. (1998) as in Nillson et al. (2008) where dissolved CO_2 was included as part of DIC. Partial pressure of CO_2, was measured onsite in triplicate by filling, then sealing a 250 mL bottle with 200 mL of water sample. Circulated air was bubbled through the sample and the change in CO_2 concentration in the headspace was measured over time using an EGM-4 infra-red gas analyser (PP Systems, Amesbury, USA) until the concentration was constant (10-12 minutes). The initial partial pressure of dissolved CO_2 in the sample was then back calculated from the total change in CO_2 concentration in the headspace. A total of 7 DIC measurements were taken at the weir between November 2016 and October 2017. The average DIC concentration was multiplied by the hourly discharge and divided by the catchment area to calculate the aquatic carbon loss as DIC per m^2.

CO_2 evasion occurred from the open water areas of blocked drains on the raised bog and from the functioning drain network upstream of the weir. CO_2 evasion was measured in triplicate with a CPY-4 (PP systems, Amesbury, USA) chamber fitted to a small floating raft and EGM-4 gas analyser. A total of 15 measurements of CO_2 evasion were conducted between two locations of blocked drains on the raised bog (Fig. 1), and 8 measurements were conducted just upstream of the weir from November 2016 to July 2017.

For the calculation of the global warming potential, 90% of the DOC loss is assumed to be converted to CO_2 and 10% to longer term storage (after Evans et al. 2016), while 100% of the DOC flux is included in the calculation of the carbon balance for the system. All of the DIC loss is assumed to be converted to atmospheric CO_2 as DIC is almost entirely composed of dissolved supersaturated CO_2.

2.8 Statistical Analysis

The ecotype variance in the NEE can be calculated as the sum of the within collar variance and the between collar variance. The within collar variance was calculated from the sum of model error and the error of input field variables. The annual model error was calculated from the standard deviation of the residuals for GPP and ER models for each collar on an hourly time step and propagated for the entire year. Similarly, the field inputs into the NEE models were assumed to have a hourly random variation of ± 1°C, ±1 cm WT, and ± 5% PAR. The effect of which on the NEE, was calculated from sensitivity analysis, which was run for all models and propagated for the entire year. The variance in the ecotype CH_4 flux was also calculated from the sum of the within collar variance and the between collar variance. The annual within collar standard deviation of CH_4 flux was assumed to be ± 30% of the collar average annual CH_4 flux, or 2.8 g C-CH_4 m^{-2} yr^{-1}, which was applied to all collars. For the carbon balance and GWP, the variance in NEE and CH_4 flux was summed with the variance due to measurement error in the DOC flux, DIC flux, and CO_2 evasion. Significant differences between ecotype annual carbon balance, and GWP was determined using 1-way ANOVA and Bonferroni confidence intervals. The significance of the linear regressions was determined with Minitab 18 Statistical Software.

3. Results

3.1 Environmental Monitoring

The annual rainfall at Abbeyleix Bog was 746 mm in 2016 and 840 mm in 2017, compared to the 2001-2016 annual average of 838 mm at the Ballyroan (Oatlands) daily rainfall station, located approximately 5 km NE of the site. The mean annual temperature at Abbeyleix bog was 9.6° C and

9.7° C in 2016 and 2017, similar to the (1978-2007) average of 9.9° C. Mean daily PPFD, air temperature, and monthly rainfall are shown in Figure 2 over the study period. The mean annual water table (MAWT) was within 2 cm at all ecotypes between the two years. The winter (Oct-Mar) water table was higher than summer (Apr-Sep) water table, as expected (Fig. 3). The average soil pore water pH was 4.7 (range: 4.4-5.1) for all ecotypes.

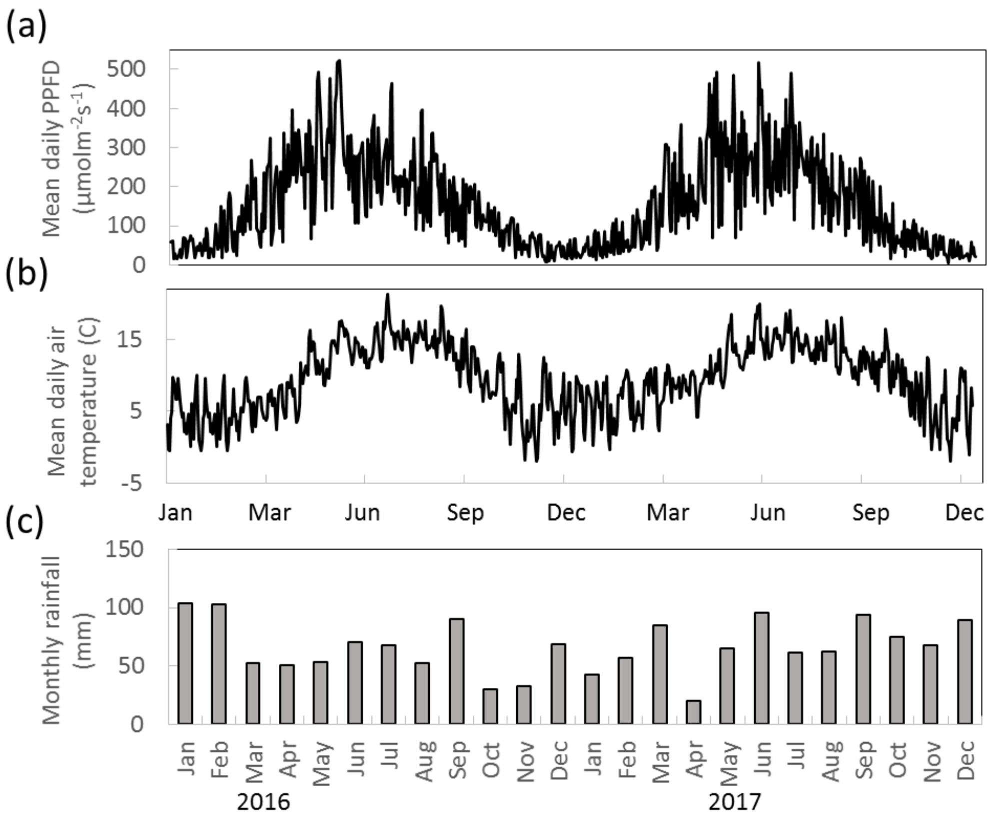

Figure 2. (a) Mean daily PPFD, (b) mean daily temperature, and (c) monthly rainfall at Abbeyleix Bog in 2016 and 2017.

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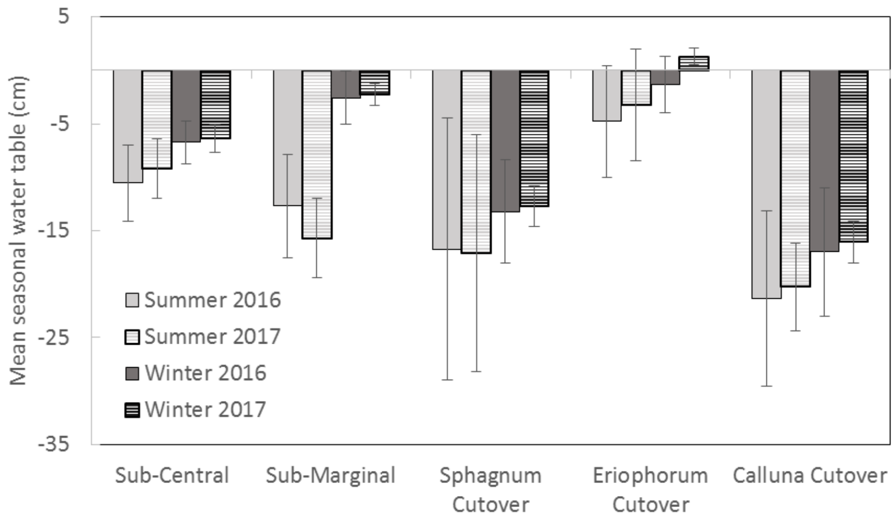

Figure 3. Mean seasonal water table for each of the ecotypes for summer (Apr-Sep) and winter (Oct-Mar), where the mean annual water table is measured with respect to the springtime peat surface or sphagnum surface (if present).

3.2 CO_2 and CH_4 Gas Fluxes

The modeled annual GPP, ER, and NEE for each collar is shown in (Table S5). The ecotype CO_2 fluxes were calculated as the average of all collars in each ecotype. The seasonal trend in modeled monthly GPP and ER were similar among all ecotypes increasing in the summer and decreasing in the winter (Fig. 4a & 4b). The Sphagnum Cutover ecotype had the largest monthly GPP from January to June both years. The monthly ER was highest at the Calluna Cutover ecotype, especially during the summer months. The ecotypes show different seasonal trends in cumulative NEE (Fig. 4c). The Sphagnum Cutover and the Sub-Central ecotypes were net CO_2 sinks (negative slope) from March (March 27 for Sub-central and March 4 for Sphagnum Cutover) to October 24, 2016 and April 24 to October 7, 2017 and CO_2 sources the rest of the year, showing an overall similar pattern to other studies of intact peatlands (e.g. Gažovič et al. 2013). By contrast, the Calluna Cutover ecotype was the strongest CO_2 source during the summer months. The Sub-Marginal ecotype is an overall moderate CO_2 source both years with a minor net CO_2 uptake occurring during summer of 2017. The Eriophorum Cutover ecotypes is approximately CO_2 neutral for much of the year with short periods of CO_2 uptake during the summer months.

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The temporal variation in methane flux was captured reasonably well (r^2 =0.61) by the model (Fig. 5). The methane data was extrapolated to an annual period using this model. Annual methane fluxes by ecotype are shown in Figure 6 and annual methane emissions for each collar are shown in Table S5. The methane emissions are highest for the Eriophorum Cutover (14.2 ±4.8 g C-CH_4 m^{-2} yr^{-1}) and Sub-Central ecotypes (12.6 ±7.9 g C-CH_4 m^{-2} yr^{-1}), which have the highest mean annual water table. The annual CH_4 flux at the Sub-Central ecotype is highly variable with a range of 1.2 to 19.3 g C-CH_4 m^{-2} yr^{-1} between collars. The annual methane flux is lowest for the Calluna Cutover ecotype (2.7 ±1.4 g C-CH_4 m^{-2} yr^{-1}).

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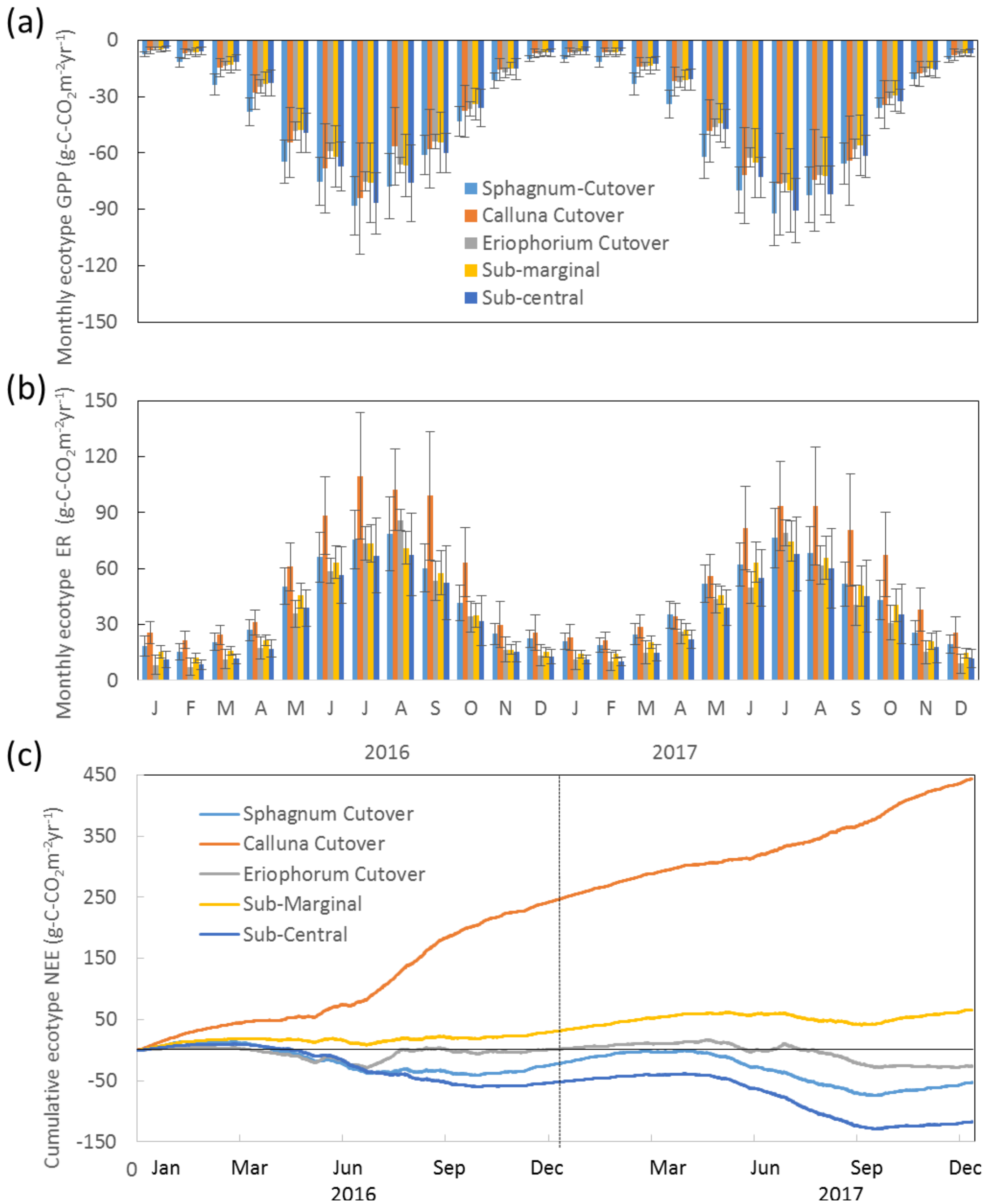

Figure 4. Monthly (a) GPP and (b) ER, and (c) cumulative NEE for each ecotype for 2016 and 2017, where the ecotype values are the average of all collars in the ecotype.

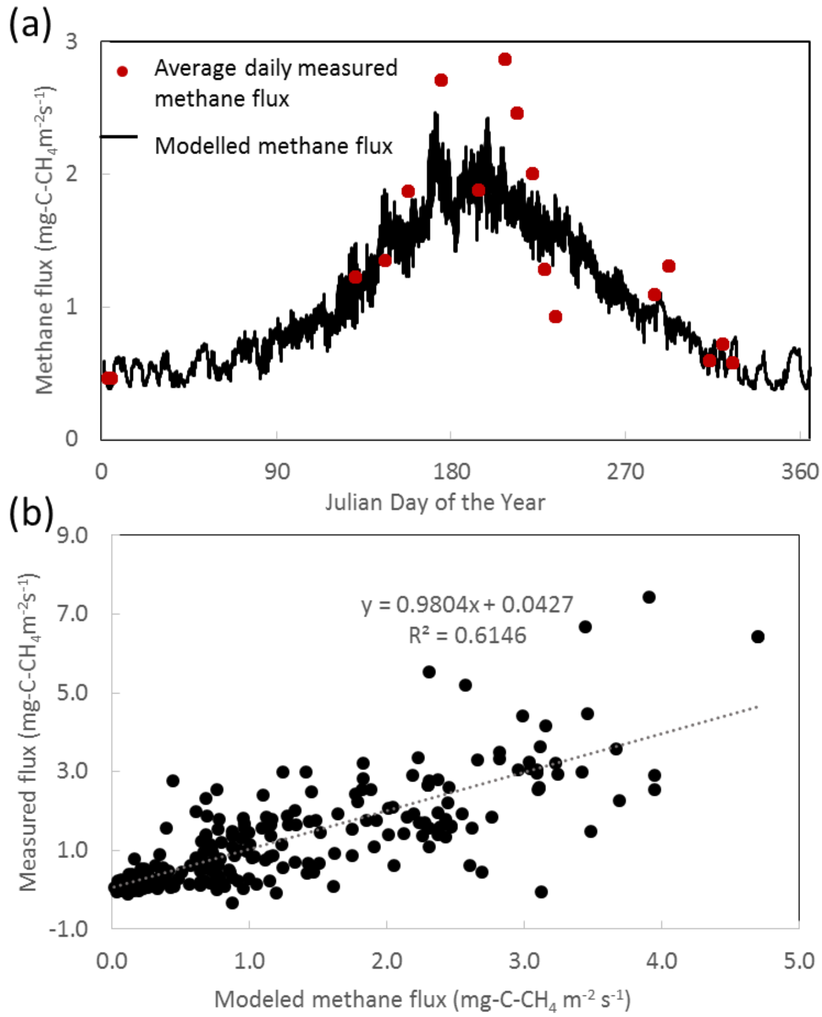

Figure 5. (a) The average daily methane flux compared to the modelled temporal fluctuations in methane flux, and (b) the modelled vs. measured methane flux when the temporal variation in multiplied by the collar average flux.

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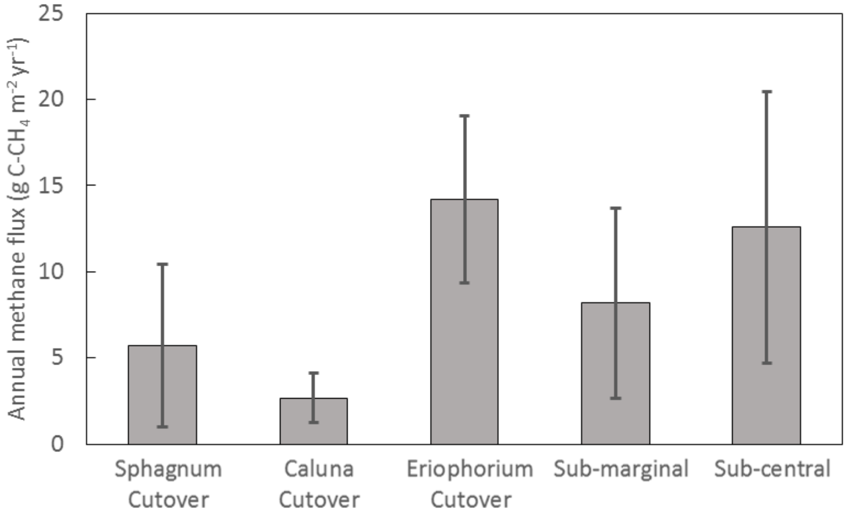

Figure 6. Annual methane flux for each ecotype averaged over all collars in the ecotype.

3.3 Aquatic Carbon Losses

The DOC concentrations showed a seasonal trend for both years - higher between approx. June and November (46.0 ±3.0 mg L^{-1}) and lower between December and May (34.5 ±2.3 mg L^{-1}) (Fig. 7).

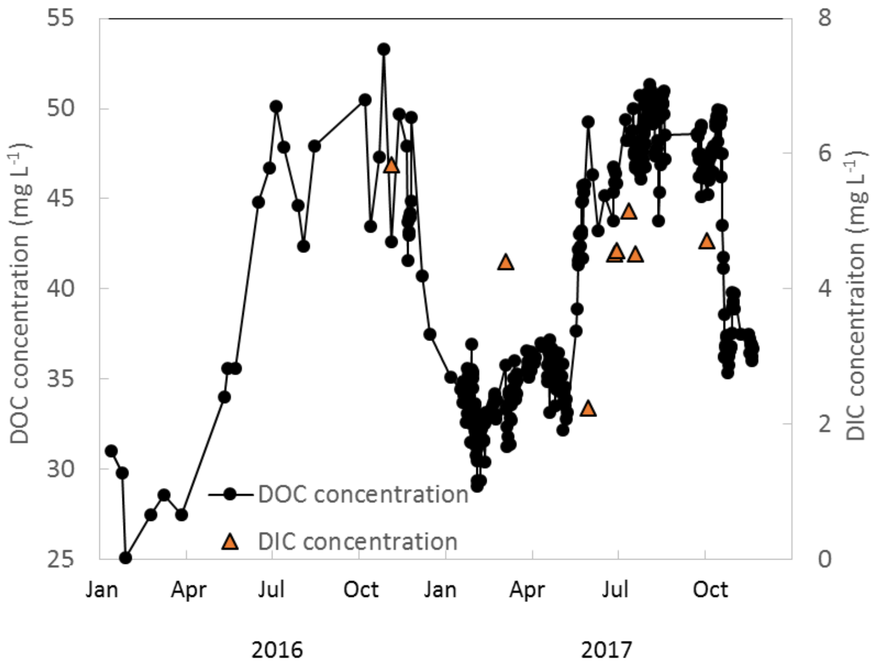

Figure 7. Measured DOC and DIC concentrations (mg L^{-1}) over a two year period (2016 and 2017) at the weir.

No trend was observed with respect to discharge. The discharge at the weir site was much higher in the winter months, with a resulting higher total DOC flux over those months. Annual losses of DOC were 8.0 ±1.6 and 12.8 ±2.5 g C m^{-2} yr^{-1} for 2016 and 2017, respectively. Seven DIC measurements were conducted at the weir site between November 2016 and October 2017. The average DIC concentration at the weir was 4.6 ±1.1 mg L^{-1}, excluding 1 low outlier (2.2 mg L^{-1}) on June 2, 2017 (Fig. 7). Based on this limited amount of data there is no significant trend in DIC concentration with respect to season, temperature, or discharge, so it was assumed constant throughout the 2 year study period. Annual carbon losses as DIC were 1.1 ±0.2 and 1.5 ±0.3 g C m^{-2} yr^{-1}. These values of annual aquatic carbon loss for DOC and DIC were applied to each of the ecotypes equally when calculating the carbon balance and GWP. Open water CO_2 evasion was measured for two blocked drains on the raised bog and just upstream of the weir. The average CO_2 evasion rate from the two blocked drains (n=15) was 5.1 x 10^{-3} ±2.9 x 10^{-3} mg C-CO_2 m^{-2} s^{-1} and was somewhat higher at the weir (n=8) as 9.2 x 10^{-3} ±3.2 x 10^{-3} mg C-CO_2 m^{-2} s^{-1} (Fig. 8).

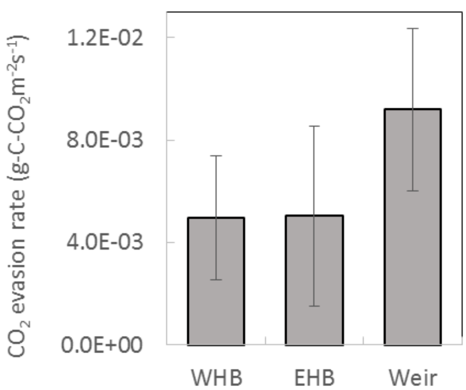

Figure 8. CO_2 evasion rate measured at two blocked drains on the high bog (WHB and EHB) and just upstream of the weir. Data was collected between March and July 2017 at the WHB location (n=7), November 2016 and July 2017 at the EHB location (n=8), and December 2016 and July 2017 at the weir location (n=8).

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Based on this limited data set, there was no significant trend in evasion rate with respect to season, temperature, or (at the weir site) discharge. CO_2 evasion rate was thus assumed constant and extrapolated to give an annual carbon loss as CO_2 evasion of 162 ±91 g C-CO_2 m^{-2} yr^{-1} and 290 ±100 g C-CO_2 m^{-2} yr^{-1} for open water blocked ditches and active drain network of the weir, respectively. The open water areas in the drain network contributing to the weir were ~0.9% of the total catchment area to give a carbon loss of 2.7 ±0.9 g C-CO_2 m^{-2} yr^{-1} for the weir catchment area as a whole. As above, this was applied equally all ecotypes. Open water areas of blocked drains only occurred near one of the ecotypes (Sub-Marginal), where they were estimated to be 2.8% of the total surface area. This gives an additional carbon loss in the Sub-Marginal ecotype of 4.5 ±2.6 g C-CO_2 m^{-2} yr^{-1}.

3.4 Carbon Balance and GWP by Ecotype

The NEE, CH_4 fluxes, and the aquatic losses of carbon were compiled to calculate the carbon balance and GWP for each ecotype (Fig. 9). The Calluna Cutover ecotype was a substantial carbon source of 260 ±70 g C-CO_2 m^{-2} yr^{-1} and 218 ±78 g C-CO_2 m^{-2} yr^{-1} for 2016 and 2017, respectively. This ecotype was significantly higher than all the other ecotypes in 2016 ($p < 0.001$) and 2017 ($p=0.011$). The annual carbon balance for the other ecotypes was not significantly different from carbon neutral. However, four of the six collars at the Sub-Central ecotype were significant carbon sinks both of the years (range -25 to -97 g C-CO_2 m^{-2} yr^{-1}). One collar in the Sub-Central ecotype was found to be a significant carbon source both of the measured years (51 and 62 g C-CO_2 m^{-2} yr^{-1}).There is substantial variation between collars within each ecotype for NEE and CH_4 flux, which is the largest source of error in ecotype carbon balance and GWP.

All ecotypes had an average positive GPW both years, with the lowest average GWP of 2.1 ± 2.4 tons CO_2-eq m^{-2} yr^{-1} at the Sphagnum Cutover ecotype and the highest average GWP occurring at the Calluna Cutover ecotype of 9.8 ± 3.5 tons CO_2-eq m^{-2} yr^{-1}. The GWP at the Calluna Cutover ecotype was significantly higher than the Sphagnum Cutover ($p = 0.002$) and Sub-Central ecotype (p =

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0.028) in 2016 and only the Sphagnum Cutover ecotype in 2017 (p = 0.018) (Fig. 9b). Methane emissions account for 12% and 14% of the GWP at the Calluna Cutover ecotype in 2016 and 2017, respectively. Methane emissions account for the majority of the total GWP in all other ecotypes (65-146%).

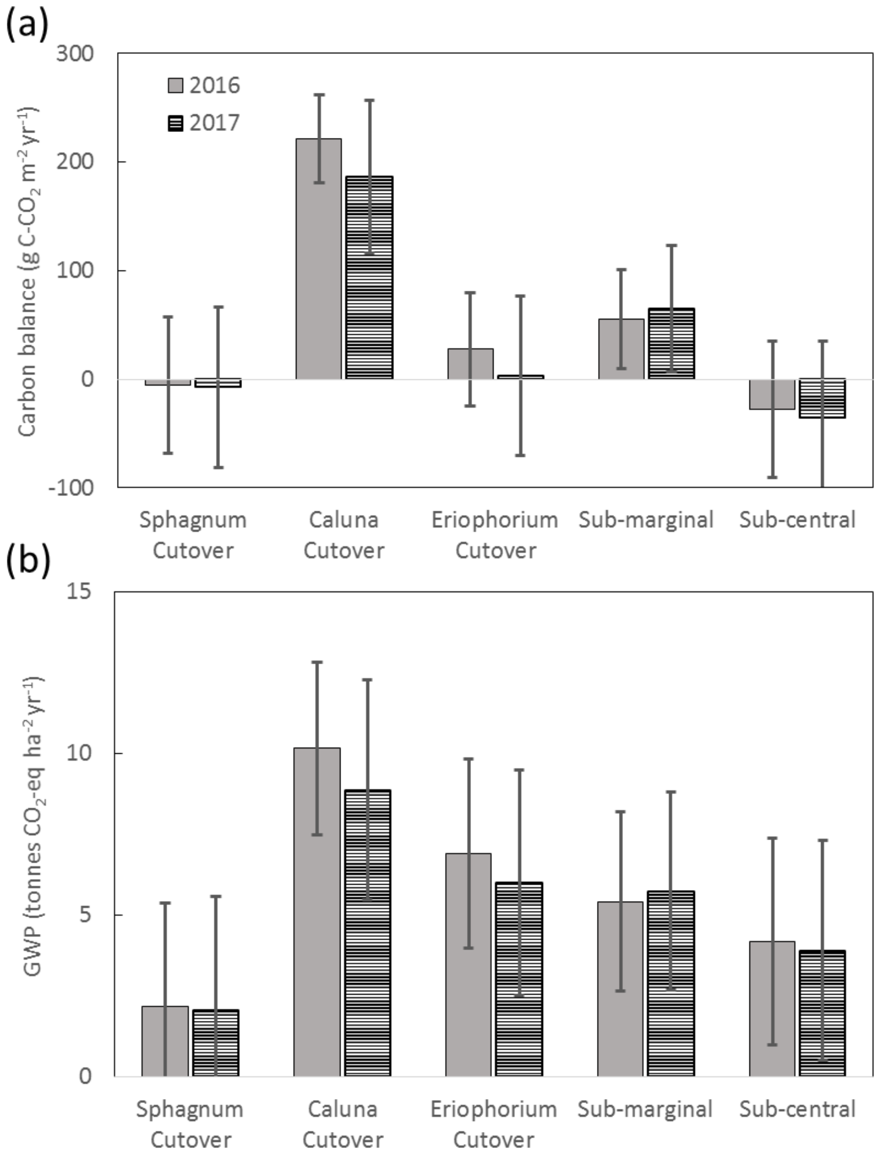

Figure 9. (a) Carbon balance for each ecotype including NEE, CH4 flux, aquatic losses as DOC and DIC, and open water CO_2 evasion. (b) Global warming potential for each ecotype.

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3.5 Drivers of NEE and GWP

Environmental drivers of the annual carbon balance, CH_4 flux, and GWP were analyzed by comparing the data from each of the 29 collars. There is a significant ($p=0.015$) but weak ($r^2 = 0.20$) negative linear correlation between the two year average annual carbon balance and the average MAWT (Fig. 10a). This particular data set is skewed by the Sphagnum Cutover ecotype, where there is a relatively low water table and an overall neutral carbon balance due to the presence of *Sphagnum spp.* hummocks. If the Sphagnum Cutover ecotype is excluded, the linear regression between average carbon balance and MAWT is highly significant ($p<0.001$) with a stronger correlation ($r^2 = 0.41$). The annual CH_4 flux has a significant ($p < 0.001$) positive linear correlation ($r^2=0.51$) with the average MAWT (Fig. 10b). The trends in CH_4 flux and carbon balance with respect to MAWT offset each other such that there is no trend ($p = 0.91$, $r^2 < 0.01$) in GWP with respect to mean annual water table (Fig. 10c).

The collar annual average GWP has a highly significant ($p < 0.001$) negative linear correlation ($r^2 = 0.58$) with the percent *Sphagnum spp.* cover in the collar (Fig. 10f). The percentage *Sphagnum spp.* cover and *Eriophorum spp.* cover in the collar seem to be correlated in a non-linear fashion with the average annual carbon balance and the annual CH_4 flux, respectively (Fig. 10 d,e). In particular, the annual CH_4 flux is greater than ~9 g $C\text{-}CH_4$ m^{-2} yr^{-1} for all collars where the percentage *Eriophorum spp.* cover is higher than 10%.

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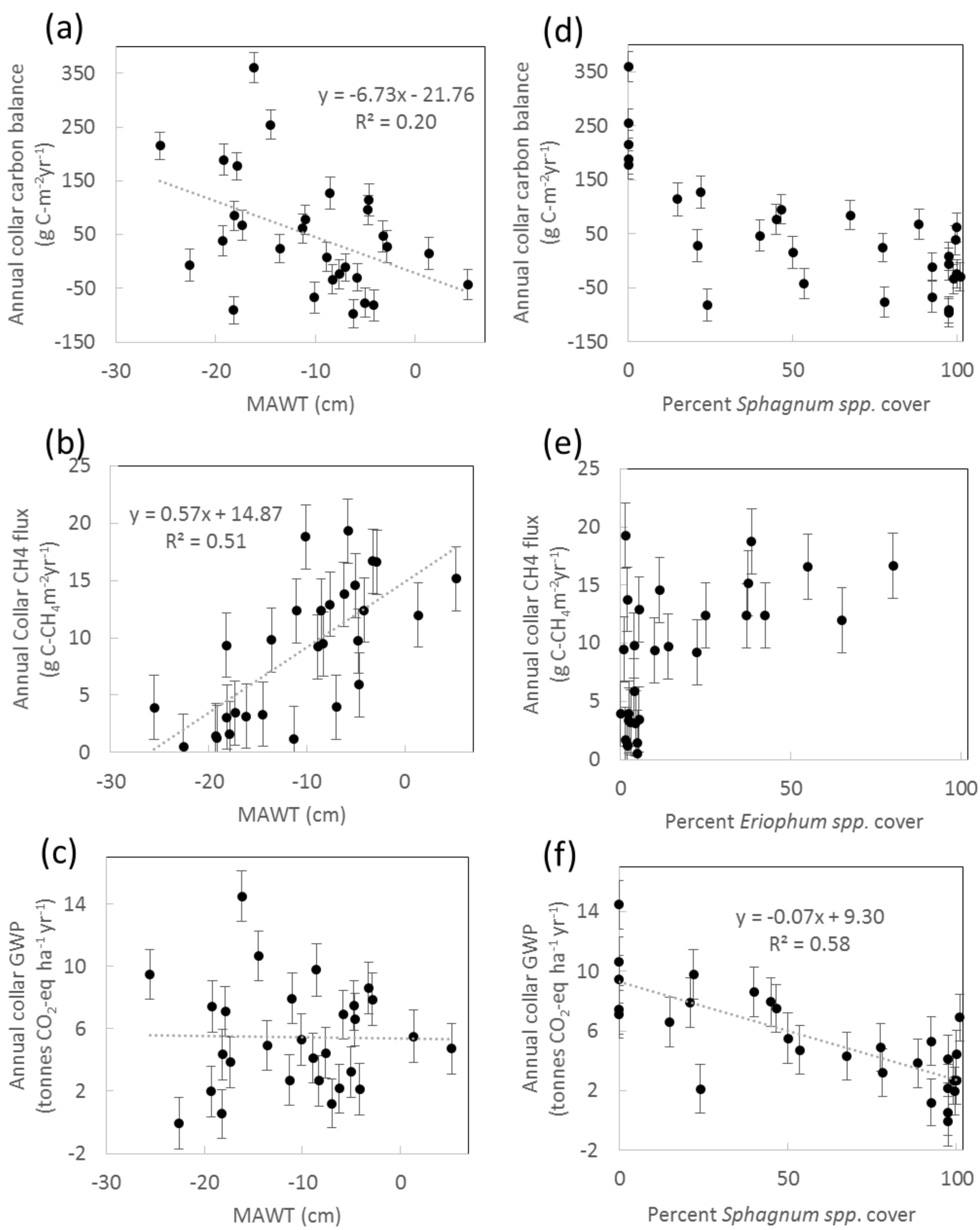

Figure 10. Trends in collar annual carbon balance, CH_4 flux, and GWP plotted against mean annual water table (MAWT) (a-c) and percent genus cover (d-f). Data is averaged over the two year period.

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4. Discussion

4.1 Carbon balance and GWP

For the first time, all of the major aspects of the carbon balance were measured simultaneously in a recovering peatland. Also, the carbon balance of ecotypes with different degradation histories was compared for a naturally recovering old cutover bog and a restored unharvested raised bog. Although the NEE is the most variable component of the carbon balance and drives the trends in the carbon balance, it is not necessarily the largest component of the carbon balance. Other aspects of the carbon balance become proportionally more important when the NEE is near neutral. For example, the NEE at the Eriophorum Cutover ecotype in 2016 was +2 ±53 g C-CO_2 m^{-2} yr^{-1}. The magnitude of the aquatic carbon loss in 2016 (11.8 ±1.8 g C m^{-2} yr^{-1}) was actually larger than the NEE at this ecotype. The total NECB was also measured for a Boreal oligotrophic mire in northern Sweden (Nilsson et al. 2008) and Auchencorth Moss, a lowland bog in Scotland (Dinsmore et al. 2010). The average annual DOC losses found in this study (10.4 g C m^{-2} yr^{-1}) are comparable to the average annual losses reported in Nillson et al. (2008) of 13.0 g C m^{-2} yr^{-1} and lower than those reported Dinsmore et al. (2010) of 25.4 g C m^{-2} yr^{-1}. The DIC losses in this study (1.3 g C m^{-2} yr^{-1}, including super-saturated CO_2 as DIC) are lower than the values reported in Nilsson et al. (2008) and Dinsmore et al. (2010) of 2.0 and 4.6 g C m^{-2} yr^{-1}, respectively. This is partially because the average DIC concentration measured in this study (4.6 ±1.1 mg C/L) is somewhat lower than that reported in Nillson et al. (2008) of 9.6 mg C/L and at Auchencorth Moss (Dinsmore et al. 2013) of 8.65 mg C/L. The annual open water CO_2 evasion found in this study (2.7 or 7.2 g C m^{-2} yr^{-1}) is lower than what was reported in Dinsmore et al. (2010) (12.7 g C m^{-2} yr^{-1}), but this is dependent on the geometry of the system as water surface area is a factor in the calculation. Also, the floating chamber method used in this study may underestimate total CO_2 evasion (Dinsmore et al. 2010). The overall two year average aquatic carbon loss found in this study (14.4 g C m^{-2} yr^{-1}) is comparable to Nilsson et al. (2008) (17.8 g C m^{-2} yr^{-1}) and lower than Dinsmore et al. (2010) (43.8 g C m^{-2} yr^{-1}).

The Calluna Cutover ecotype was found to be a substantial carbon source and this is likely due to a lower water table and a plant ecology reflective of a degraded peatland. The Eriophorum Cutover ecotype has the highest mean annual water table and the highest *Eriophorum spp.* cover; both of which are related to an increase in the observed methane flux. Even with the increased methane flux at the Eriophorum Cutover ecotype, the GWP at this ecotype was not higher than the Calluna Cutover ecotype. This agrees with Wilson et al. (2016b), where a rewetted bog in Ireland was found to have a lower GWP than a well-drained site even where *Eriophorum angustifolium* developed. The *Sphagnum spp.* dominated ecotypes (Sphagnum Cutover and Sub-Central) were the lowest average GWP sources, and *Sphagnum spp.* cover was negatively correlated to the GWP at the collar scale. In terms of restoration, this suggests that there is GHG benefit for both raising the water table as well as establishing high quality bog vegetation such as *Sphagnum spp.*

The eco-hydrological conditions seem to be what determines GHG emissions, rather than time since restoration. The data here do not support the hypothesis that time since restoration/abondonment *per se* is an important factor in the GHG emissions (once vegetation is established as discussed below). For example, all three of the cutover sites were presumably abandoned at the same time (circa 1960's). However, these three sites have very different CO_2 and CH_4 emissions despite their close physical proximity (within 200 m), similarities in soil, and a shared site history. Similarly, the raised bog ecotypes (Sub-Central and Sub-marginal) were restored more recently by drain blocking in 2009. The average carbon balance and GWP of the Sub-Marginal ecotype falls within the range of the much older cutover ecotypes, and the Sub-Central ecotype has a similar average GWP to the Sphagnum Cutover ecotype. This hypothesis would only be true if is there is an eco-hydrological trajectory in the years post restoration/abandonment where *Eriophorum spp.* cover decreases or *Sphagnum spp.* cover increases, for example. Further, although the Calluna Cutover location is much higher carbon source than the Sub-Marginal or Sub-Central locations. This area is similar ecologically (and presumably in terms of hydrologic conditions) to the large areas of the uncut raised bog, which are heavily degraded. This type of habitat seems to be the most common habitat in the cutover areas in Abbeyleix bog, and is probably similar to much of the degraded bog areas in Ireland. In the absence of restoration works, this ecotype remains a large carbon source more than 5 decades after abandonment.

There is a need for simple methodologies to predict greenhouse emissions from peatlands for policy and management, particularly from data that are available at the regional or national scale. Water table, vegetation cover, and soil temperature have been previously suggested as potential predictive metrics of GHG fluxes from peatlands (Strack et al. 2016). Hence, a simple linear regression based on MAWT (in cm below ground level) and percent genus cover was fit to the data from the 29 collars in this study to predict the annual carbon balance (Eq. (3), $r^2 = 0.71$) and CH_4 flux (Eq. (4), $r^2 = 0.56$).

$$\text{Annual carbon balance} = 117.9 - 6.23*(\text{MAWT}) - 2.1*(\text{Percent } \textit{Sphagnum spp.}) \quad (3)$$

$$\text{Annual } CH_4 \text{ Flux} = 12.23 + 0.440*(\text{MAWT}) + 0.0754*(\text{Percent } \textit{Eriophorum spp.}) \quad (4)$$

where annual carbon balance and annual CH_4 flux are in units of g C m^{-2} yr^{-1}. While these coefficients are site specific, these metrics may be useful for comparison to future studies.

4.2 Comparisons with Global Studies of Boreal and Temperate Peatlands

The annual NEE, CH_4 flux, and water table data from the ecotypes in this study were compared to global studies of boreal and temperate peatlands. The data from global studies was divided into three generic categories as follows:

- Pristine/Intact peatlands; - those peatlands that have not been harvested, undergone intensive agriculture or forestry, and are not heavily impacted by drainage or other disturbance;
- Bare peat sites; - previous peat extraction sites where there is an absence of vegetation cover;
- Restored/Degraded/Recovering peatlands; - all other peatlands are grouped into this category for this comparison.

This compilation of data focuses on low nutrient (if specified, pH<6) natural and semi-natural sites and excludes sites that are actively used for intensive agriculture or forestry.

For both vegetated and bare peat sites, there is a negative correlation between MAWT and NEE (Fig. 11). Both intact peatlands and variously degraded/recovering peatlands fall on the same trend line, agreeing with Wilson et al. (2016a). Annual NEE for vegetated sites followed a linear trend with respect to MAWT with slope of -4.5 g C-CO_2 m^{-2} yr^{-1} per cm rise in MAWT and an intercept of -92 g C-$CO_2$$m^{-1}$$yr^{-1}$. The slope is similar to that reported from a review of studies of peatlands with MAWT higher than -30 cm (Wilson et al. 2016a) of -2.0 ±1.0 and -5.0 ±2.0 g C-CO_2 m^{-2} yr^{-1} per cm rise in MAWT for boreal and temperate peatlands, respectively. However, the trend in NEE with respect to MAWT should be interpreted with some caution because of the difficulty of generalizing across sites based on simple water table proxies (Wilson et al. 2016a). For example, there was a "highly peatland-specific dependency (i.e., with different offsets and slopes) of the CO_2 response to water table depth" for grassland peatlands in Germany (Tiemeyer et al. 2017), although, that study looked at grasslands, which may have much more variability in soil type, land management, nutrient status, etc. than the natural and semi-natural low nutrient sites shown in Figure 11. This trend may also break down as MAWT becomes too low (e.g. Tiemeyer et al. 2017) because soil respiration can be limited if the soil is too dry (Briones et al., 2014). Thus, climate patterns could be an important factor in CO_2 response to water table (Tiemeyer et al. 2017).

Based on the data collected in Figure 11, intact peatlands occur at a narrower range of mean annual water table and NEE. This is expected because degraded peatlands can have a wider range of site histories and eco-hydrological conditions (Wilson et al. 2016a). This agrees with Strack et al. (2016) who reported greater variation in CO_2 and CH_4 fluxes at restored plots when compared to either unrestored or natural plots. As with data from this study, this may suggest that restoration of high quality peatland ecology has an additional NEE benefit beyond raising the water table.

The Sub-Central ecotype in this study has continuous *Sphagnum spp.* lawns similar to an intact peatland. This ecotype has a mean annual NEE of -57 g C-CO_2 m^{-2} yr^{-1} and a mean annual water table of -8.2 cm. This is close to the overall average NEE (-60 g C-CO_2 m^{-2} yr^{-1}) and mean annual water table (-9 cm) for intact/pristine peatlands in this figure. This comparison is valuable for validating the data for the other ecotypes because the carbon balance of natural bogs is comparatively better characterized than degraded systems and the potential for systematic bias in chamber measurements. The Calluna Cutover ecotype from this study has an exceptionally high NEE (222 g C-CO_2 m^{-2} yr^{-1}) for the mean annual water table (-18.6 cm) compared to the NEE (-5 g C-CO_2 m^{-2} yr^{-1}) predicted from the best fit trend line of vegetated sites.

Also, as shown in Figure 11, bare peat sites have a higher NEE than vegetated sites at a given MAWT, and these trend lines diverge at higher MAWT. As it can take decades for vegetation to be established in industrially harvested peatlands (Wilson et al., 2015), this data would suggest that restoration to encourage plant colonization could reduce the short term CO_2 emissions even if no other restoration works are undertaken. This data set could be used to predict the CO_2 reduction

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from raising the water table as well as establishing vegetation on bare peat sites. Further, peatlands may be large carbon sinks in the years immediately post restoration as vegetation recovers due to the rapid, subsequent increase in vegetation biomass. For example, an annual NEE of -473 g C-CO_2 m^{-2} yr^{-1} was reported by Waddington et al. (2010) one year post restoration for sites where herbaceous vegetation increased dramatically. This may explain some of the low outliers in Figure 11 for restored/recovering sites. Three of the low outliers in Figure 11 are from Strack et al. (2014), which is 4 years post restoration with a growing season NEE of -162, -121- and -126 g C-CO_2 m^{-2} for mean seasonal water tables of -21.3, -24.9 and -28.2 cm, respectively.

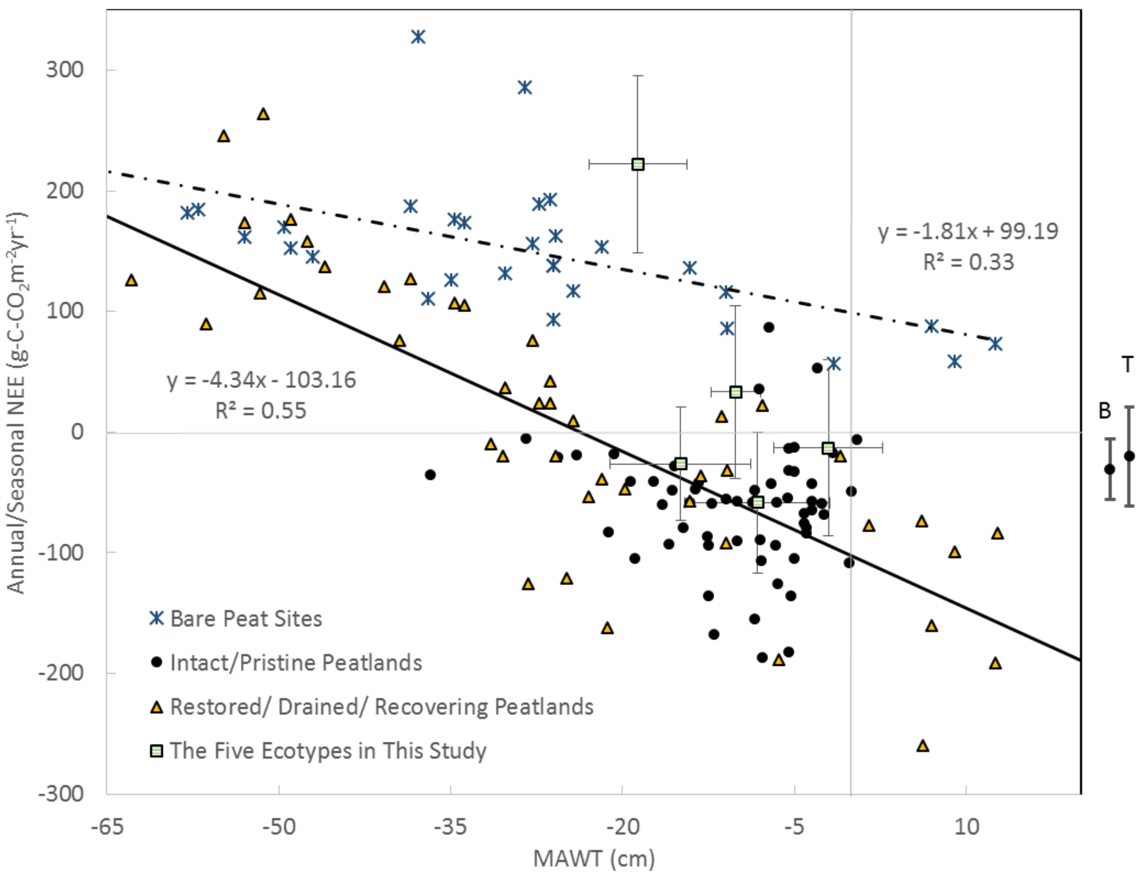

Figure 11. Mean annual water table vs. the annual NEE for the 5 ecotypes in this study (error bars are standard deviation) compared to global studies from boreal and temperate peatlands. The solid line shows

the best fit linear trend line from all vegetated sites and the dashed line shows the best fit trend line for bare peat sites. (Data from: Wilson et al., 2015; Wilson et al., 2016; Vanslow-Algan et al., 2015; Tuittili et al., 1999; Waddington et al., 2010; Strack et al., 2014; Nilsson et al., 2008; Dinsmore et al., 2010; Koehler et al., 2011; Chimner et al., 2017; Gazovic et al., 2013; Lund et al., 2012; Levy and Grey et al., 2015; McVeigh et al., 2014; Helftler et al., 2015; Piechl et al., 2014; Stranchen et al., 2016; Roulet et al., 2007; Waddington and Roulet, 2000; for more details and additional studies see Supplemental Table S6 in Supplemental Section 3). Also, shown to the right of the figure is the mean and 95% CI NEE from nutrient poor, wet (MAWT >-30 cm) boreal (B) and temperate (T) peatlands (from the review paper, Wilson et al. 2016a).

There are a few cautionary notes that should accompany this plot. First, some of this data was collected using the closed chamber method and some collected using eddy covariance methods. Although both methods measure the same metric (NEE), closed chamber methods are inherently micro-scale while eddy-covariance methods are inherently landscape scale, as are the water table measurements accompanying them. Eddy-covariance measurements spatially integrate the micro-variations within the landscape compared to closed chamber measurements, and much of the NEE data reported in Figure 11 for intact peatlands is from eddy-covariance flux towers while there are very few studies that have used this technique on degraded or recovering peatlands. This may cause apparently higher variation in NEE for restored/recovering peatlands. Second, many of the studies on boreal peatlands report only growing season NEE and water table because of frozen winter conditions. Seasonal values from these studies are assumed to approximately represent annual values because inter fluxes at boreal sites are probably of minor importance to the annual fluxes. Third, this figure contains data points from different locations as well as the same location over multiple years where data is available.

Similarly, annual/seasonal methane emissions are plotted against MAWT (Fig. 12). The data from the ecotypes in this study fall well within the range of the CH_4 flux values in this compilation of data. Reported methane emissions from drained peatlands are quite low and typically do not exceed 0.6 g $C\text{-}CH_4$ m^{-2} yr^{-1} when the mean annual water table is below -30 cm.

There is a high degree of variability in methane emissions in sites where the MAWT is higher than -20 cm. Thus, a high MAWT seems to be a prerequisite for high methane emissions but does not necessarily result in a high methane emissions, which agrees with Tiemeyer et al. (2017). As in Wilson et al. (2016a), there does not seem to be a difference between restored and intact peatlands for the CH_4 flux data presented in Figure 12, excluding infilled ditches, which can be hotspots for methane emissions (Waddington and Day, 2000). For example, Cooper et al., (2014) reports 53.9 g $C\text{-}CH_4\ m^{-2}\ yr^{-1}$ for infilled ditches (Cooper et al. 2014). The low methane emissions from rewetted bare peat soils suggests that the methanogenesis is limited by substrate availability in cutover peatlands (Tuittila et al. 2000; Tuittila et al. 1999).

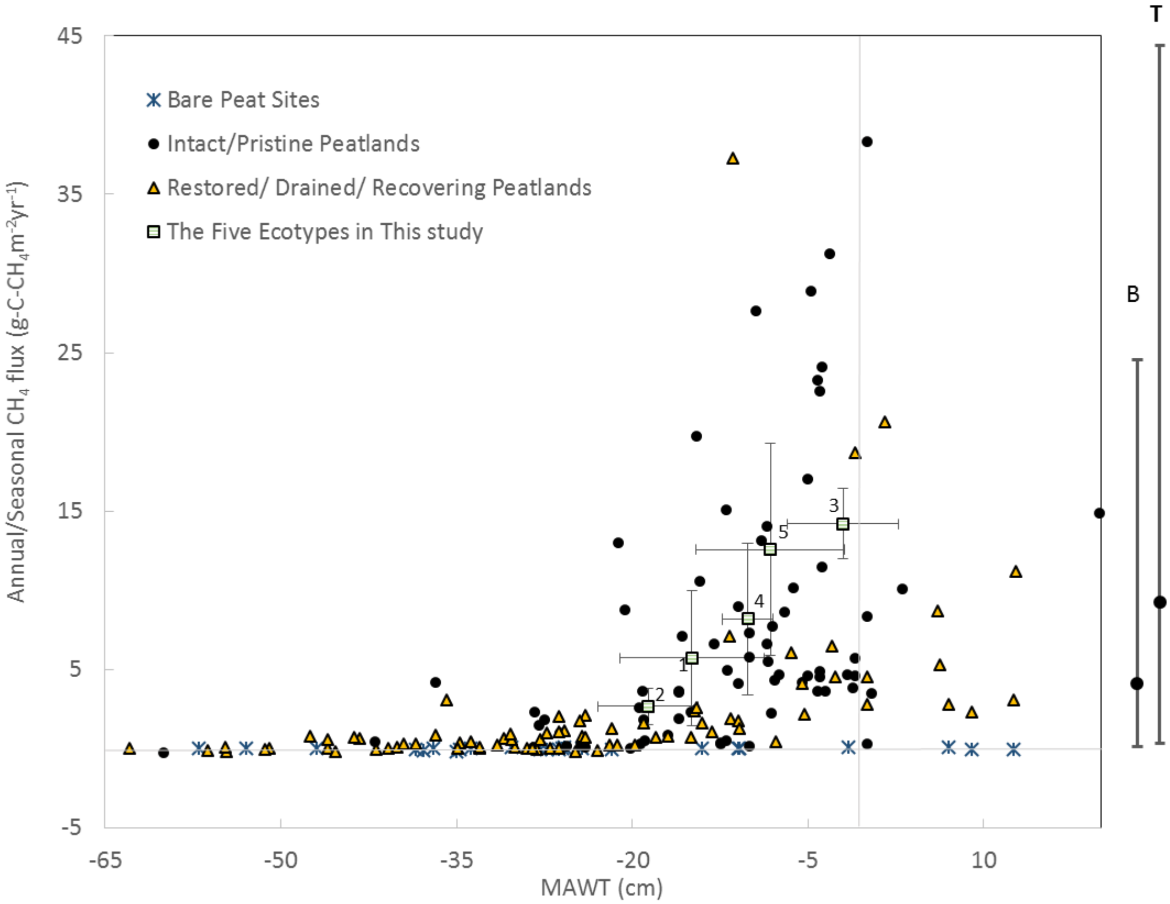

Fig. 12. This figure shows the mean annual water table plotted against the measured annual methane emissions for each ecotype for each ecotype in this study (error bars are standard deviations) and from global studies of temperate and boreal peatlands (Sources: Flessa et al. 1998, Fieldler et al. 1998, Wilson et al. 2016; Tuitili et al., 1999; Wilson et al., 2018; Danevic et al., 2010; Von Arnold et al., 2005; Laine et al., 1996; Yamulki et al., 2012; Nykanen et al., 1998; Fieldler et al., 2007; Cooper et al., 2014; Waddington and Day, 2007; Chimner et al., 2017; Waddington and Roulet 2000; for more details see Table S6 in Supplemental Section 3. Also, shown to the right of the figure is the mean and 95% CI of methane emissions from nutrient poor, wet (MAWT >-30 cm) boreal (B) and temperate (T) peatlands (from the review paper, Wilson et al. 2016a).

As with the NEE data in Figure 11, this figure contains both annual and seasonal fluxes, where seasonal fluxes are more often reported for boreal sites. This figure excludes methane emissions from infilled ditches. There are few studies that have reported methane emissions from bare peat sites, and the results are generally low (mean of -0.03 g C-CH_4 m^{-2} yr^{-1}) even at high water table. The

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data used to compile Figs. 11 and 12 and additional studies can be found in Supplemental Section 3, Table S6.

4.3 Implications for Peatland Management and Restoration

Peatland management and restoration is primarily able to alter 1) the hydrology, typically managing the water table through drainage or drain blocking and 2) the plant ecology, through revegetation efforts, managing water table, and controlling invasive species (Andersen et al. 2017). If peatland management is used as a climate change mitigation tool [as suggested in Birkin et al. (2011); Wilson et al. (2013); Leifeld and Menichetti, (2018)], the impact of these things must be considered. The wide range of methane emissions reported in the literature at high water tables means that generalizations cannot be made about GWP for restored vs. pristine peatlands or GWP as a function of water table. For example, reported values of annual CH_4 fluxes in Fig. 12 from sites with a MAWT above -10 cm range from 0.3 (Nykanen et al., 1995) to 38.3 g C-CH_4 m^{-2} yr^{-1} (calculated from Junkurst and Fieldler, 2007) for intact peatlands and 0.4 (Strack et al. 2014) to 20.6 g C-CH_4 m^{-2} yr^{-1} (Renou-Wilson et al. 2018a) for restored peatlands. This corresponds to a 100-year GWP of 0.1 to 17.3 and 0.2 to 9.3 tonnes CO_2-eq ha^{-1} yr^{-1}, respectively. This range is larger than the largest reported CO_2 sink for intact peatlands of -6.9 tonnes CO_2-eq ha^{-1} yr^{-1} (calculated from Levy and Grey 2015) and far larger than the average CO_2 sink for intact peatlands of -2.2 tonnes CO_2-eq ha^{-1} yr^{-1} reported in Figure 11. Still, a GWP decrease is often observed following rewetting (Wilson et al. 2016a, Wilson et al. 2016b, Renou-Wilson et al. 2018b). Additionally, the data from this study would suggest that the presence of *Sphagnum spp.* corresponds to a decreased GWP. Junkurst and Fielder (2007) conducted a review of CO_2 and CH_4 flux for boreal and temperate peatlands. They state that the methane fluxes in temperate peatlands are "usually found to be three orders of magnitude lower than simultaneously measured CO_2 emissions." Thus, they conclude that the suppressed CO_2 emission from higher water table would outweigh the GWP effect of increased methane emissions. This conclusion seems unlikely to be generally true based on the data shown Figure 11 and 12.

There is some debate about the use of GWP as a metric for natural peatlands because this metric focuses on a 100-year time window, which may not be appropriate. For example, to quote from Evans et al. (2016), "as noted by Frolking et al. (2006), the long-term sequestration of CO_2 into stable organic matter gradually outweighs the warming effect of CH_4, due to the shorter atmospheric lifetime of the latter, so that natural peatlands exert a net cooling impact on the atmosphere over longer periods." This means that the long term climate benefit of peatlands is primarily controlled by NEE. However, this logic would only apply to restoration works if these impact the eco-hydrological conditions on time scales >> 100 years.

The ecotypes of the uncut raised bog at Abbeyleix were mapped by Bord na Móna in 2009 just after restoration works blocking surface drains and again in 2014. During this time the extent of Sub-Central area increased by approx. 2.1 ha largely at the loss of the Sub-Marginal ecotype. Assuming the values found in this study are representative of all years, the restoration works resulted in a reduction of 7.0 ± 7.7 tonnes yr^{-1} CO_2 although a smaller reduction (3.3 ± 7.6 tonnes yr^{-1}) of CO_2 equivalents. Additionally, there is a potential reduction in CO_2 emissions due to raising the water table throughout the entire 108 ha of raised bog area. The change in water table from these restoration works was not directly measured, but based on the typical depths of water in the blocked drains, there was an estimated 10–40 cm rise in water table. For the 108 ha of raised bog area, this could result in an additional reduction of 166–664 tonnes yr^{-1} of CO_2 based on the trends in Fig. 11. The impact of increased methane emissions in this case is probably minimal because the majority (67%) of the raised bog area, although with a higher water table than previously, remains as deeply drained ecotypes (Marginal or Facebank).

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5. Conclusions

All the major components of the carbon balance were measured at several different ecotypes, on restored and cutover raised bog with different land use and degradation histories. Trends in annual NEE and CH_4 fluxes were observed with respect to both ecological and hydrological conditions. In particular, higher water level and intact *Sphagnum* vegetation seem to be related to higher carbon sink and lower GWP. The data from ecotypes in this study were compared to a large number of studies on boreal and temperate peatlands with respect to MAWT. In this broader comparison, negative trends were observed in NEE with respect to MAWT for both vegetated and bare peat sites, while CH_4 fluxes were more variable at high MAWT.

Data availability

Much of the data on the various aspects of the annual carbon balance including all the data behind Fig. 6, Fig. 9, Fig. 10, Fig. 11, and Fig. 12 can be found in the supplemental material. All other data used in this study are archived by the authors and are available on request (swensonm@tcd.ie).

Supplemental Information

Section S1. A description of the NEE and CH_4 flux models tested and the thought behind these models. Also, for each of the 29 collars in this study, the empirical fitting parameters, the r^2, and the standard deviation of the residuals is shown for the best GPP and ER models.

Section S2. Eco-hydrological conditions and carbon balance terms for all collars and both years of this study.

Section S3. Data collected from literature: measurements aspects of peatland greenhouse gas balance. This section includes the data behind Fig. 11 and Fig. 12 as well as other studies.

Author contribution

Michael Swenson collected and analyzed the majority of the field data and prepared the manuscript with contributions from other co-authors. Shane Regan attained the grant award, determined the field site location, and contributed to setting up the field equipment and measuring infrastructure. Dirk Bremmers collected methane flux data in the field and analyzed gas samples in the lab. Jenna Lawless collected field measurements of DIC and CO_2 evasion. Shane Regan, Matt Saunders and Laurence Gill contributed technical advice and guidance throughout the project implementation and manuscript writing stages.

Competing interests

The authors declare that they have no conflict of interest.

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