Growing season carbon dioxide and methane exchange at a restored peatland on the Western Boreal Plain

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A R T I C L E   I N F O

A B S T R A C T

Boreal peatlands represent globally important long term sinks of carbon; however, horticultural peat extraction disrupts this carbon sink function, converting these ecosystems to large sources of greenhouse gases. Peatland restoration mitigates these emissions but to date no measurement of greenhouse gas exchange has been conducted on restored peatlands in western Canada, a region where continental climate could impact restoration success. We measured CO_2 and CH_4 fluxes during the growing season in a restored, cutover peatland in northern Alberta (Boreal Plain Ecozone) and compared these to fluxes measured on a neighboring unrestored area. Restoration resulted in a shift in mean growing season fluxes from 378 g CO_2—C and −0.2 g CH_4—C at the unrestored site to −30 g CO_2—C and 3.7 g CH_4—C at the restored site, where positive values indicate flux of carbon from the peatland to the atmosphere. Carbon dioxide exchange was correlated to vascular vegetation cover that varied depending on local water table position. Water table was also related to CH_4 flux, with higher emissions from wet sites. Restoration activities should avoid creating very dry microsites where greenhouse gas emissions will remain high, while very wet sites may accumulate carbon as CO_2 but will likely create areas of high CH_4 flux.

1. Introduction

Peatlands are wetland ecosystems where the productivity of the vegetation community exceeds the rate of decay resulting in the accumulation of organic matter, or peat, over thousands of years (Vasander and Kettunen, 2006; Vitt, 2006). This accumulation of peat makes these ecosystems important long-term sinks of atmospheric carbon; however, peat harvesting may degrade the ecosystem sufficiently to turn it into a source of atmospheric carbon (Waddington et al., 2010).

Canada has an estimated 120 million ha of peatland of which 25,000 ha, or 0.02% have been drained for horticultural peat harvesting, with ∼14,000 ha currently in operation (Environment Canada, 2013a). The draining of peatlands typically results in an increase in net CO_2 emission and a decrease in CH_4 efflux, except in drainage ditches where increased CH_4 flux has been reported (Mahmood and Strack, 2011; Waddington and Day, 2007). The majority of ongoing horticultural peat extraction in Canada is by vacuum harvesting, requiring an extensive network of drainage ditches.

Without immediate remediation, cutover peatlands will become persistent sources of CO_2 and result in huge carbon losses to the atmosphere (Waddington et al., 2002). Depending on the hydrochemistry of the residual peat, abandoned cutover peat fields may experience spontaneous recolonization by, predominantly, vascular plants (Graf et al., 2008; Mahmood and Strack, 2011); however, many sites remain poorly revegetated and largely devoid of mosses decades after peat extraction has ceased (Poulin et al., 2005). Spontaneous recolonization of harvested peatlands has been found to increase CH_4 flux (Mahmood and Strack, 2011) by providing an escape through plant pathways, while it may also decrease CO_2 flux (Bortoluzzi et al., 2006) as a result of increased productivity.

Considering CO_2 and CH_4 exchange from northern peatlands it has been concluded that these ecosystems have resulted in net atmospheric cooling over the Holocene (Frolking and Roulet, 2007). Extracted peatlands represent a persistent source of CO_2 (Waddington et al., 2002) and given modern concerns about the role of these greenhouse gases (GHGs) in accelerating climate change, there is a need to develop methods for restoring extracted peatlands. This increase in CO_2 emission results from the removal of vegetation and the drawing down of the water table. Therefore the process of restoring a peatland must include the reestablishment

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of hydrological conditions typical of a natural peatland system and the re-establishment of natural peatland vegetation communities (Waddington et al., 2010). While spontaneous revegetation of harvested peatlands may be beneficial in restoring a site back to a net carbon sequestering system (Bortoluzzi et al., 2006; Graf et al., 2008; Lavoie et al., 2001), active restoration practices may be necessary to enhance recovery (Rochefort et al., 2003; Waddington et al., 2010). This has led to the development of a North American approach to the restoration of cutover peatlands (Quinty and Rochefort, 2003). This method of peatland restoration involves restoring local hydrology by blocking drainage ditches and resurfacing the cutover peatland; collecting, introducing and protecting collected diaspores (any part of a plant that can generate a new individual); and introducing fertilizer to encourage the growth of vascular plants and *Polytrichum* moss that act as nurse species for *Sphagnum* moss.

Although the North American peatland restoration method has been applied for almost two decades, most restoration has taken place in eastern Canada with application to western provinces only recently. These regions have very different climate regimes. For example, peatlands along the southern shore of the St. Lawrence River in Québec experience mean annual precipitation of ~900 to >1200 mm and mean annual temperature of ~4 °C (Environment Canada, 2013b). In contrast, peat extraction areas in Alberta receive an average of 500–600 mm of precipitation annually and mean annual temperatures of 0–3.4 °C (Environment Canada, 2013b). Given the limited number of restoration projects in western Canada and the lack of GHG flux measurements at these sites, it remains unclear what constraint these climatic conditions place on restoration success and the carbon balance of these restored areas.

The goal of this study was to evaluate the effectiveness of restoration techniques based on Quinty and Rochefort (2003) as applied on a cutover peat bog in northern Alberta. Based on previous research on restored peatlands in Québec (Mahmood and Strack, 2011; Strack and Zuback, 2013; Waddington et al., 2010) we hypothesized that: (1) restoration would decrease the CO_2 source from peat fields and possibly result in a CO_2 sink, (2) restoration would increase CH_4 efflux and (3) water table and plant cover would be significant controls on rates of GHG exchange.

2. Study site

This study was conducted in a cutover peatland north of Wandering River, Alberta, Canada (55.293° N, 112.475° W, Fig. 1). Measurements were made across the ~10 ha restored section and compared to measurements at a neighboring unrestored section. The study site is situated in the Boreal Plains ecozone (Ecological Stratification Working Group, 1996). The 30-yr normal annual precipitation at Wandering River is 522 mm and mean annual temperature is 0.17 °C (The Weather Network, 2013). The restored site is owned and operated by Sun Gro Horticulture and had been previously drained by a series of ditches around the site and crossing it longitudinally along its major axis. This area was previously under active vacuum-extraction and was restored by the site operator in 2008 according to the North American Peatland Restoration Guide by Quinty and Rochefort (2003). Briefly, restoration involved filling ditches on the restored area, spreading diaspores from a neighboring ombrotrophic bog in a ratio of 1:10 (1 ha of collected material over 10 ha of restored area), covering material with straw mulch, adding phosphate rock fertilizer (150 kg ha⁻¹), and blocking perimeter ditches.

Twelve sample plots were established systematically on the restored site (R) in approximately four rows (A–D) of three collars with each row spaced ~100 m apart along a main transect west

to east along the site's major axis on the south side of the field (Fig. 1). The first row was ~100 m due east of the site access road. This design was created with the intent of capturing the clearly visible hydrologic gradient at the site, where the west end was dry and the east end was largely ponded. Furthermore, sites were chosen to capture the variety of vegetation on the site (Table 1). Rows A and C were inset ~60 m from the main transect, and each plot was spaced ~40 m apart (Fig. 1). Each sample plot was bounded by a 0.36 m² (60 cm × 60 cm) stainless steel collar with grooves, onto which a clear (CO_2) or opaque (CH_4) chamber was placed during flux measurement. Boardwalks were installed across waterlogged areas, and platforms were constructed next to each collar to limit any soil disturbance during C flux measurement.

Three additional plots were established on a neighboring unrestored (U) peat field (Fig. 1) that had experienced no spontaneous recolonization by native vegetation. These were used to establish a baseline for bare peat flux for CH_4 and CO_2 when no restoration efforts are undertaken. Each of these plots was roughly in line with rows A, B and C, respectively, of the restored peatland. Wells were installed at each plot; however, collars were not installed in 2011. Instead, a portable collar was inserted ~9 cm into the peat surface each time C flux measurements were taken. Permanent collars were installed for measurements made in 2012.

Carbon flux measurements were completed between July 13–September 9, 2011 and May 16–July 21, 2012. During this period CO_2 and CH_4 fluxes were measured at each sampling plot seven to eight times.

3. Methods

3.1. Carbon dioxide exchange

Net ecosystem exchange of CO_2 (NEE) was determined using the closed chamber technique. Briefly, a transparent acrylic chamber (60 cm × 60 cm × 30 cm) was placed on the sampling plot and CO_2 concentration in the headspace monitored for 105 s using a portable infrared gas analyzer (EGM-4, PP Systems, Massachusetts, USA). A battery operated fan mixed the headspace during flux measurement. Flux was determined from the linear change in CO_2 concentration over time correcting for chamber volume and ambient temperature as recorded with a thermocouple inserted into the chamber. Short chamber closure times were used to limit heating inside the chamber headspace and data do not provide evidence of a deviation from a linear pattern of concentration change over time.

During each flux measurement photosynthetically active radiation (PAR) was measured with a quantum sensor connected to the EGM-4. The measurement was repeated under a variety of light levels created using shades and under an opaque tarp to determine ecosystem respiration (ER). Gross ecosystem photosynthesis (GEP) was determined as the difference between NEE and ER. At the unrestored site only ER was determined as no vegetation was present. As mentioned above, in 2011 ER was determined using a smaller portable chamber-collar combination (~10 L total volume). In 2012, 60 cm × 60 cm collars were installed and the same chamber was used at both the restored and unrestored sites.

3.2. Methane flux

The closed chamber technique was also used to determine CH_4 fluxes at each plot. An opaque 60 cm × 60 cm × 30 cm chamber was used and equipped with a battery-operated fan to mix the headspace air. Headspace was sampled at 7, 15, 25, and 35 min after chamber closure through tubing sealed with a three-way
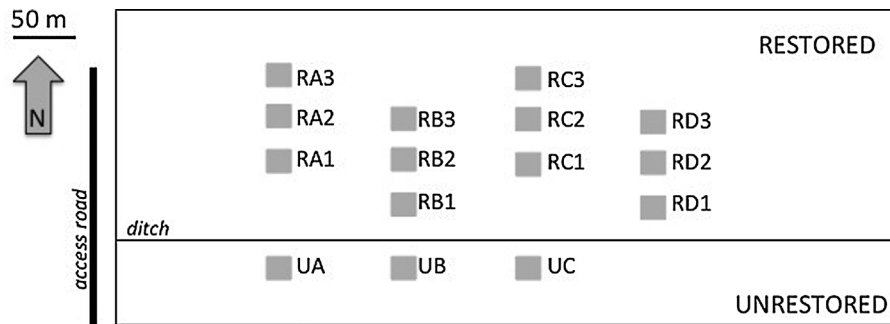
Fig. 1. Schematic of the study site. Gray squares represent collar locations. The eastern portion of the restored site was largely flooded during the entire study period.

valve. Samples were stored in pre-evacuated Exetainers (Labco Ltd., UK) until analysis for CH_4 concentration on a gas chromatograph equipped with flame ionization detector (Varian 3800, Varian Canada, etc.). Methane flux was determined from the linear change in concentration over time, corrected for chamber volume and temperature monitored with a thermocouple inserted in each chamber. Flux measurements with concentration outliers that suggested ebullition were removed as it was unclear whether or not these resulted from disturbance during sampling. This resulted in loss of less than 5% of data.

3.3. Environmental variables

Soil temperature was measured at 2, 5, 10, 15 and 20 cm depths adjacent to each sampling plot during each flux measurement. Water table was measured in a PVC standpipe installed adjacent to each plot.

A meteorological station was installed in an undisturbed bog ~10 km from the study site. The station recorded PAR (LICOR quantum sensor), air temperature (Campbell Scientific, 107B), and precipitation (tipping bucket rain gauge), measured every minute and averaged each half hour (Campbell Scientific, CR1000 data logger). Water table was measured in a PVC standpipe every half hour with a pressure transducer corrected for barometric pressure

(Solinst, levelogger junior and barologger). Soil temperature was monitored with HOBO temperature loggers.

3.4. Vegetation community

In August 2012, 4 yr after initial restoration, vegetation survey was conducted by dividing each collar into quadrants. Cover of each species was estimated visually within each quadrant to the closest 1% and estimates were averaged for the entire collar. Coverage of mosses, graminoids, and shrubs was computed based on cover of individual species.

3.5. Data analysis

Growing season CO_2 exchange was estimated by modeling GEP and ER for each sample plot. Gross ecosystem photosynthesis was related to PAR using a rectangular hyperbola:

$$GEP = \frac{PAR \times Q \times GP_{max}}{(PAR \times Q + GP_{max})} \tag{1}$$

where Q is the quantum efficiency and describes the initial slope of the hyperbola and GP_{max} is the theoretical maximum rate of GEP and represents the asymptote of the hyperbola.

Table 1
Ecohydrological conditions of study plots.

Plot no.	Water table[a]	Moss cover (%)	Vascular cover (%)	Total plant cover (%)	Dominant species[b]
Restored					
RA1	−62.9 (−114 to −45)	4	31	36	*Oxycoccus microcarpus, Salix pedicellaris*
RA2	−56.3 (−106 to −39)	34	79	112	*S. pedicellaris, Polytrichum strictum, Carex canescens*
RA3	−40.8 (−65 to −27)	34	1	35	*P. strictum, Pohlia nutans*
RB1	−21.3 (−55 to −2)	44	41	85	*P. nutans, Carex aquatilis*
RB2	−23.0 (−50 to −4)	104	57	161	*P. nutans, C. canescens, C. aquatilis, Sphagnum angustifolium*
RB3	−7.8 (−19 to 0)	50	89	139	*C. canescens, S. angustifolium Eriophorum vaginatum, S. pedicellaris*
RC1	−13.2 (−42 to 2)	0	80	80	*C. aquatilis, Lophozia ventricosa, E. vaginatum*
RC2	−19.8 (−47 to 0)	0	56	59	*C. aquatilis*
RC3	−6.4 (−40 to 13)	0	72	90	*C. aquatilis, S. pedicellaris, L. ventricosa*
RD1	−11.4 (−68 to 10)	0	58	81	*Carex utricularia, Agrostis scabra, Rorippa islandica*
RD2	−24.9 (−73 to −3)	47	59	120	*P. strictum, S. pedicellaris, E. vaginatum, Marchantia polymorpha*
RD3	−28.2 (−54 to −8)	3	58	60	*C. aquatilis, S. pedicellaris*
Unrestored					
UA	−97.8 (−106 to −93)	Bare			
UB	−38.7 (−70 to −22)	Bare			
UC	−35.0 (−79 to −17)	Bare			

[a] Water table given as mean (minimum to maximum) in cm below the soil surface.
[b] Includes all species with >12% cover in the plot. Listed in order of declining coverage.

Ecosystem respiration was modeled using multiple linear regression with water table (WT) and soil temperature at 5 cm (T_5) depth according to:

$$ER = aT_5 + bWT + c \qquad (2)$$

where a, b and c are parameters fitted by least squares regression.

In order to evaluate differences in CO_2 exchange between sites and potential controls on these fluxes, GEP and NEE fluxes under full light conditions (PAR photon flux density > 1000 μmol m^{-2} s^{-1}, Bubier et al., 2003) were considered. These fluxes are referred to as GEP_{max} and NEE_{max}. Differences in ER and CH_4 flux were also evaluated. We evaluated differences in CO_2 flux between unrestored plots and wet and dry restored plots using a general linear model with pairwise comparisons and a family α of 0.05. Methane fluxes were not normally distributed. Thus, differences were evaluated using a Kruskal–Wallis test with pairwise comparisons using Mann–Whitney tests with Bonferroni correction resulting in α of 0.017. Previous research has suggested that water table position shallower than 20 cm below the surface leads to large increases in CH_4 flux (e.g. Couwenberg and Fritz, 2012) and thus we used this water table position as the cutoff for wet and dry restored plots. Mean flux values for the study period at each sampling plot were used for all analyses.

Controls (e.g. vegetation cover, water table position, soil temperature) on GHG fluxes were evaluated using linear regression. Minitab 14.1 was used for all statistical analyses.

Seasonal CO_2 exchange was estimated over a 130 day growing season (May 29–October 5) using environmental conditions measured at the meteorological station in 2011 combined with equations 1 and 2 and parameters determined for each sampling plot. At the majority of plots there were no significant regressions between daily CH_4 flux values and soil temperature or water table position. Thus, seasonal CH_4 flux was estimated by multiplying mean CH_4 flux determined over the sampling period by 130 days in the growing season. These growing season estimates for CO_2 and CH_4 exchange are based on a limited number of field measured fluxes and care should be taken in their interpretation.

The GHG balance based on CO_2 and CH_4 fluxes for each sampling plot during the growing season was determined using the global warming potential (GWP) approach. The CO_2 balance at each plot, as determined from modeled fluxes and environmental conditions

in 2011, was added to the CH_4 flux multiplied by 25 (IPCC, 2007) to calculate a GWP in units of g CO_2-e m^{-2}.

4. Results

4.1. Ecohydrological conditions

Water table and vegetation cover varied between plots (Table 1). At the unrestored site, mean water table position during the study period was between 35.0 and 97.8 cm below the peat surface. At the restored site, mean water table was between 6.4 and 62.9 cm below the surface (Table 1). At both sites water table position was highest in July 2011, gradually declined into September 2011, remained low in spring 2012, but recovered slightly by July 2012. At all plots the frost table was within the upper 25 cm during May 2012 measurements. While the restored site soil thawed by late June 2012, the frost table remained within the upper 70 cm at the unrestored site into the July sampling date in 2012.

Restored plots had plant cover between 35 and 160%, most of which consist of spontaneously regenerated species. Common species observed on the restored site included *Carex aquatilis*, *Carex canescens*, *Eriophorum vaginatum*, *Salix pedicellaris*, *Polytrichum strictum* and *Pohlia nutans*. Moss cover remained low at many plots (Table 1) but reached 40% cover or greater in four of the 12 plots studied.

Water table position was significantly positively related to graminoid cover (linear regression, R^2(adj) = 0.43, p = 0.012) but was not a significant predictor of total plant cover, total vascular cover, moss cover or shrub cover. Comparing wet and dry restored plots, moss and shrub cover was generally higher in dry restored plots, while total vascular plant cover and graminoid cover was higher in wet restored plots. However, only graminoid cover was statistically significantly different between wet and dry restored plots (t-test, $T = -3.36$, $p = 0.007$).

4.2. Carbon dioxide fluxes

Mean growing season GEP_{max} varied at restored plots from -3.0 to -50.0 g CO_2 m^{-2} d^{-1}. As no plants were present at the unrestored plots GEP was assumed to be 0. Mean NEE_{max} was between -23.4 g and 2.2 CO_2 m^{-2} d^{-1} at restored plots and 3.8 and 11.6 g CO_2 m^{-2} d^{-1} at unrestored plots (Fig. 2). Both GEP_{max} and NEE_{max} were significantly different between sites (GEP_{max}: $df = 2$,
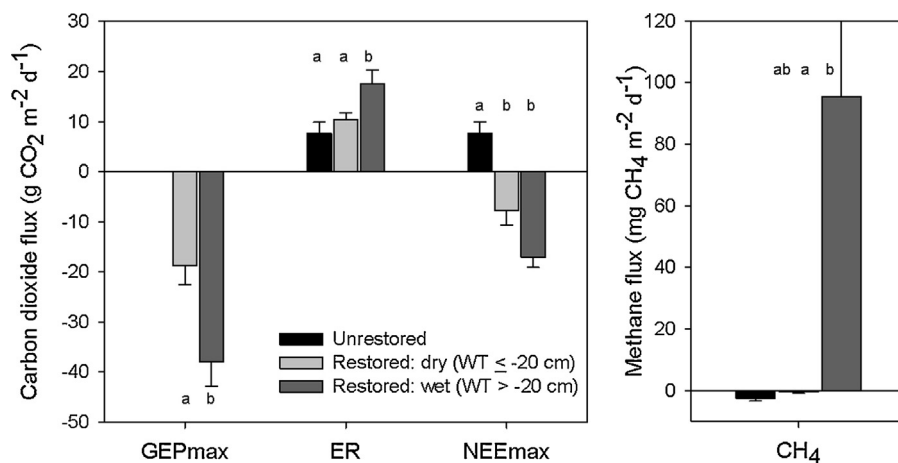
Fig. 2. Mean gross ecosystem photosynthesis (GEP), ecosystem respiration (ER), net ecosystem exchange (NEE) and methane flux (CH_4) from unrestored and restored plots. Dry restored plots had mean water table deeper than -20 cm. GEP_{max} and NEE_{max} are based on measurements of CO_2 exchange when photosynthetically active radiation photon flux density was greater than 1000 μmol m^{-2} s^{-1}. Error bars give standard error. Groups are significantly different from each other if they share no letters in common. Letters should only be compared within one flux component.

$F = 16.15$, $p < 0.001$; NEE: $df = 2$, $F = 14.55$, $p = 0.001$). In both cases, the unrestored site was significantly different from the restored site, while wet and dry restored plots were different for GEP$_{max}$, but not statistically significantly different for NEE$_{max}$ ($p = 0.06$). Ecosystem respiration was also significantly different between sites ($df = 2$, $F = 5.56$, $p = 0.02$) with wet restored plots being significantly higher than unrestored plots. Dry restored plots were not statistically significantly different from either wet restored or unrestored plots (Fig. 2).

Combining all sites and using mean growing season values of CO_2 exchange and water table position, GEP$_{max}$ (Fig. 3b) and NEE$_{max}$ were both significantly related to water table (linear regression, $p < 0.05$). Ecosystem respiration was significantly related to GEP$_{max}$ (Fig. 3c, linear regression, R^2(adj) = 0.67, $p < 0.001$). Although not significantly related to water table position alone (Fig. 3a), inclusion of this parameter with GEP$_{max}$ in multiple regression explained an additional 11% of the variation in ER between sample plots (linear regression, R^2(adj) = 0.78, $p < 0.001$). The correlation between ER and GEP$_{max}$ was further supported by a significant linear regression between ER and vascular plant cover (Fig. 4, R^2(adj) = 0.28, $p = 0.024$). Vascular plant cover was also a significant predictor of $\log$(GEP$_{max}$) (Fig. 4, linear regression, R^2(adj) = 0.49, $p = 0.007$). Graminoid cover was also significantly related to $\log$(GEP$_{max}$) (R^2(adj) = 0.47, $p = 0.008$).

Using environmental conditions from the meteorological station recorded in 2011 and equations 2 and 3, growing season CO_2 exchange was estimated for the period May 29–October 5 (130 days total). Models described the data well for most sampling plots (Table 2). Standard error was 0.4–2.7 g C m^{-2} d^{-1} for estimated GEP. For estimated ER, standard error was 0.2–1.3 and 0.4–1.0 g C m^{-2} d^{-1} at restored and unrestored sites, respectively. Gross ecosystem photosynthesis was −57 to −423 and −277 to −595 g CO_2—C m^{-2} at dry and wet restored plots, respectively. Ecosystem respiration at dry restored plots was 121–346 g CO_2 m^{-2} and at wet restored plots was 299–608 g CO_2—C m^{-2}. This resulted in estimates of growing season NEE of −162 to 127 at dry restored and −189 to 23 g CO_2—C m^{-2} at wet restored plots. Unrestored plots released 126–680 g CO_2—C m^{-2} as ER.

4.3. Methane flux

Mean CH$_4$ flux during the sample period was between −1.77 and 394.68 mg CH$_4$ m^{-2} d^{-1} at the restored plots, where negative values indicate consumption of CH$_4$ from the atmosphere. All unrestored plots had mean CH$_4$ flux indicating net consumption ranging between −1.13 and −4.00 mg CH$_4$ m^{-2} d^{-1}. Methane flux was significantly different between sites (Fig. 2; Kruskal–Wallis, $df = 2$, $H = 10.71$, $p = 0.005$). Pairwise comparison indicated that the unrestored site CH$_4$ flux was not significantly different from dry restored (Mann–Whitney, $p = 0.11$) or wet restored sites (Mann–Whitney, $p = 0.04$) although the latter is likely due only to the low number of replicate plots at the unrestored site. Wet restored plots had significantly higher CH$_4$ flux than dry restored plots (Mann–Whitney, $p = 0.006$).

Mean seasonal CH$_4$ flux varied with water table with $\log$ (total season CH$_4$ flux + 1) increasing as mean season water table position became shallower (Fig. 5; R^2(adj) = 0.25, $p = 0.057$). Mean seasonal CH$_4$ flux was not significantly related to total plant cover, total vascular plant cover, nor total graminoid cover.

Multiplying average fluxes by the 130 day growing season period resulted in estimated growing season CH$_4$ flux of −0.38 to −0.11, −0.17 to 0.21, and 0.25–37 g CH$_4$—C m^{-2} at unrestored, dry restored and wet restored sites, respectively.
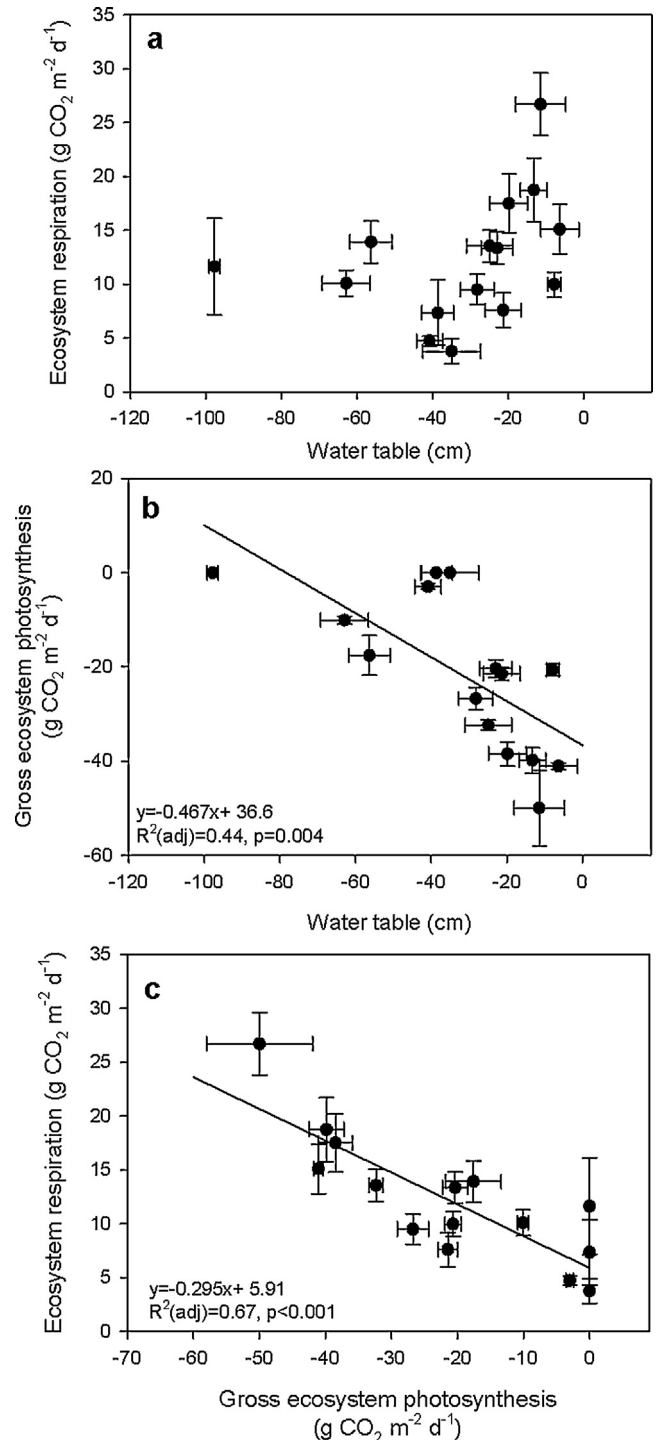
Fig. 3. Regression between mean ecosystem respiration, mean gross ecosystem photosynthesis (when photosynthetically active radiation photon flux density was greater than 1000 μmol m^{-2} s^{-1}) and mean water table position. Error bars give standard error of the mean.

4.4. Growing season carbon balance and global warming potential

Plot carbon balance between May and October 2012 was largely driven by CO_2 exchange and thus patterns closely follow those of estimated growing season NEE. Unrestored plots were sources of carbon losing 126–680 g C m^{-2}. Restored plots acted as both sources and sinks of carbon at both dry and wet restored plots,

Table 2
Parameters for CO_2 exchange models.

| Plot no.	GEP parameters			ER parameters			
	GP_{max}	Q	R^2	a	b	c	R^2
Unrestored							
UA		n.a.		0.953	4.272	412.39	0.99
UB		n.a.		0.525	1.281	43.65	0.86
UC		n.a.		0.429	0.016	−2.116	0.35
Wet restored							
RB3	−0.042	−35.098	0.89	0.220	−0.222	4.415	0.24
RC1	−0.044	−151.340	0.94	1.323	−0.656	−10.450	0.71
RC2	−0.043	−133.722	0.91	1.100	−0.357	−7.156	0.66
RC3	−0.062	−93.331	0.94	0.708	−0.350	1.511	0.86
RD1	−0.047	−716.092	0.78	1.936	−0.218	−9.176	0.80
Dry restored							
RA1	−0.036	−12.105	0.77	0.613	−0.069	−4.266	0.56
RA2	−0.055	−24.383	0.57	0.902	−0.239	−13.740	0.72
RA3	−0.021	−4.140	0.49	−0.018	0.060	7.608	0.41
RB1	−0.046	−33.736	0.81	0.493	−0.268	−6.360	0.89
RB2	−0.037	−40.075	0.85	0.331	−0.229	0.334	0.78
RD2	−0.054	−67.751	0.93	0.626	−0.159	−1.727	0.70
RD3	−0.048	−52.515	0.87	0.309	−0.175	−0.861	0.40

although the average across all restored plots was a net sink. Dry restored plots had carbon balance between −162 and 89 g C m^{-2}, while wet restored plots were −183 to 23 g C m^{-2}.

Global warming potential (GWP) based on CO_2 and CH_4 exchange was 456–2481, −589 to 465 and −490 to 1292 g CO_2-e m^{-2} at unrestored, dry restored and wet restored plots, respectively. High GWP at unrestored and dry restored plots largely resulted from large CO_2 effluxes, whereas CH_4 efflux contributed more to GWP at wet restored plots.

5. Discussion

Ongoing research on greenhouse gas exchange following peatland rewetting and restoration suggests that these activities greatly reduce carbon and greenhouse gas losses to the atmosphere (e.g. Samaritani et al., 2011; Strack and Zuback, 2013) and may return peatlands to carbon sinks over time (Tuittila et al., 1999). On the other hand, rewetting and colonization by aerenchymatous species (e.g. *Carex* spp, *Eriophorum* spp) can lead to large increases in CH_4 flux (Marinier et al., 2004; Tuittila et al., 2000; Waddington and Day, 2007). The present study supports these trends with restoration resulting in a shift in mean growing season net CO_2 and CH_4 fluxes from 378 g CO_2—C and −0.2 g CH_4—C at the unrestored site to −30 g CO_2—C and 3.7 g CH_4—C at the restored site.

Although no undisturbed site was monitored for comparison in the present study, mean growing season CO_2 and CH_4 fluxes for both dry and wet restored sites fall close to ranges reported for peatlands in Alberta. Adkinson et al. (2011) report growing season NEE of −110 g C m^{-2} for an Alberta poor fen whereas a
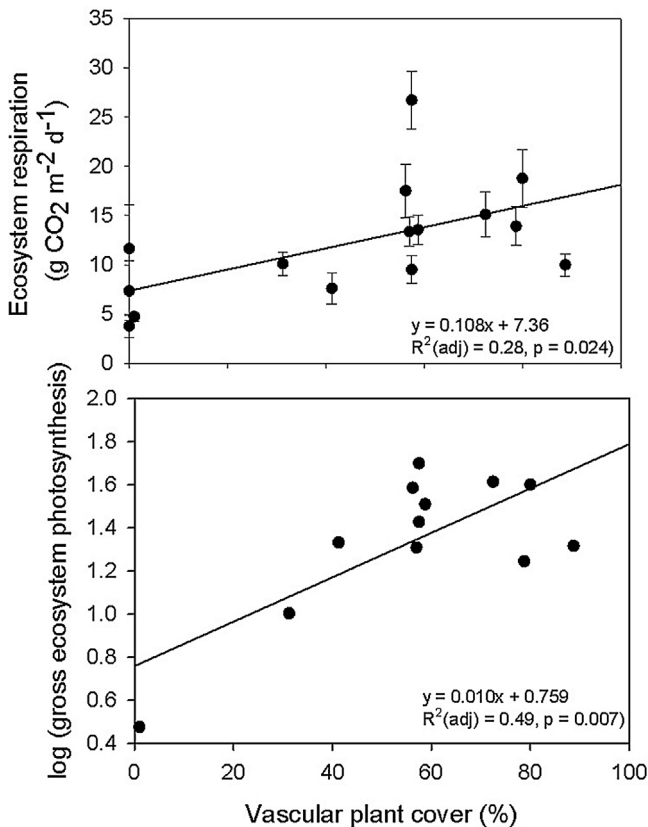
Fig. 4. Linear regression between mean seasonal ecosystem respiration and log (mean seasonal gross ecosystem photosynthesis) and vascular plant cover.
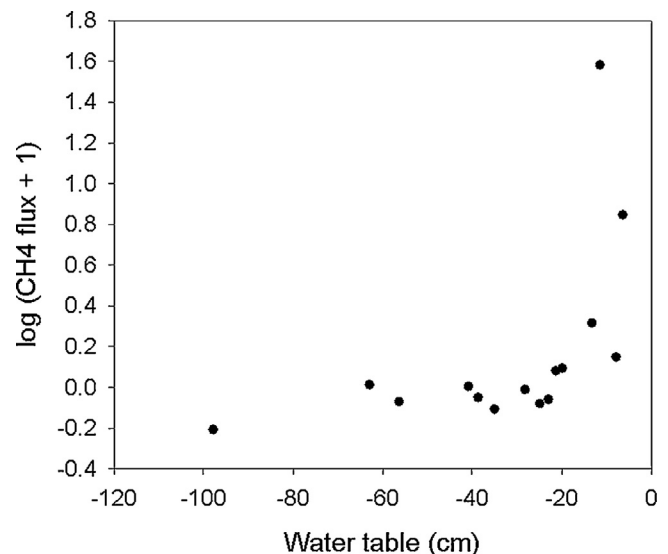
Fig. 5. Log (seasonal mean CH_4 flux + 1) vs. mean seasonal water table position.

nearby extreme rich fen had measured growing season NEE of −34.5 to −153.5 g C m^{-2} over 3 yr of measurements with varying weather. Growing season NEE of −207 to −282 g C m^{-2} was reported for a moderately rich treed fen in Alberta (Cai et al., 2010). Using a combination of flux measurements and tree growth on a chronosequence of Alberta bogs following wildfire, Wieder et al. (2009) estimated NEE of −120 to −221 g C m^{-2}. Long et al. (2010) measured CH$_4$ flux using eddy covariance in a moderately rich Alberta fen and report emission of 3.2 g CH$_4$ m^{-2} (2.4 g CH$_4$−C m^{-2}). A recent literature review of CH$_4$ fluxes across the northern high latitude region (Olefeldt et al., 2013) reports mean fluxes of 23.0 and 37.1 mg CH$_4$ m^{-2} d^{-1} for non-permafrost bogs and fens, respectively. Based on these literature values, 3 yr post-restoration, the restored site in the present study was a smaller CO$_2$ sink than undisturbed peatlands with a CH$_4$ flux similar to that reported for northern fens.

Growing season fluxes of CO$_2$ and CH$_4$ from the restored site were also similar to those reported for other temperate and boreal restored cutover peatlands. Wilson et al. (2013) report annual NEE and CH$_4$ fluxes of −14 to −588 g CO$_2$−C m^{-2} and 5–12 g CH$_4$−C m^{-2} for vegetated microsites and 37–82 g CO$_2$ m^{-2} and 0.1 g CH$_4$−C m^{-2} for bare peat areas on a rewetted cutover peatland in Ireland restored 7–10 yr prior to the study. Growing season NEE in a cutover peatland in Finland 1–3 yr following rewetting was 44 to −65 g CO$_2$−C m^{-2} (Tuittila et al., 1999). As with the present study, Tuittila et al. (1999) also reported a link between vascular plant cover and wetness resulting in increased CO$_2$ sequestration in wetter zones. Methane efflux during the growing season from the same study site was 0.4–1.0 g CH$_4$−C m^{-2} in the third year following rewetting (Tuittila et al., 2000). Samaritani et al. (2011) report growing season NEE for 29, 42 and 51 yr old regenerating plots on a cutover bog in the Swiss Jura mountains of 40, −222 and −209 g CO$_2$−C m^{-2}, respectively. In Canada, Waddington et al. (2010) report development of a growing season CO$_2$ sink of −20 g C m^{-2} in the third year post-restoration for a restored cutover bog in Quebec, while CH$_4$ emissions in the same year were 3.1 g CH$_4$−C m^{-2} (Waddington and Day, 2007). These values are similar to those reported in the present study suggesting that the continental climate does not greatly impact the effect of restoration on CO$_2$ and CH$_4$ fluxes, at least in the short term. Based on measurements at the same Quebec site 10 yr after restoration, Strack and Zuback (2013) determined that the site was annually a source of 102 g CO$_2$−C m^{-2} and 0.8 g CH$_4$−C m^{-2}, likely due to the dry conditions during the study period. This illustrates the sensitivity of greenhouse gas flux estimates to the weather during the study period (e.g. Roulet et al., 2007; Urbanová et al., 2013) indicating the importance of long-term studies to determine robust estimates of peatland carbon balance. In order to determine a full carbon balance of the present study site, year round measurement of CO$_2$ and CH$_4$ fluxes and determination of waterborne losses of carbon would be required.

A detailed vegetation survey of the restored site was not conducted in the present study. However, species composition of the measured gas flux plots revealed that, although many wetland species were present on the site, very few were representative of the donor material applied during the restoration (Table 1). Donor material for this site was collected from an ombrotrophic bog, which in this region would be dominated by black spruce (*Picea mariana*), ericaceous shrubs (e.g. *Ledum groenlandicum, Chamadaphne calyculata, Vaccinium vitis-idaea*), small herbs (*Rubus chamaemorus, Smilacina trifolia*) and *Sphagnum* moss (*Sphagnum fuscum, Sphagnum angustifolium, Sphagnum magellanicum*) (e.g. Szumigalski and Bayley, 1996; Wieder et al., 2009; Xu, unpublished data). Aside from *P. strictum*, a small amount of *S. angustifolium* and *Oxycoccus microcarpus*, the majority of the plants

on site likely spontaneously colonized the restored area following rewetting. This probably occurred due to the peat chemistry on the site that has pH in surface water and shallow subsurface water of 6.71 and 5.39. This pH is higher than would be found in ombrotrophic conditions (bog pH 3.9–4.0, Vitt et al., 1995) and may indicate more minerotrophic or fen-like conditions on site. This often occurs for cutover peatlands if a large portion of the *Sphagnum* peat layer is removed resulting in extraction down to former fen peat (Graf et al., 2008; Wind-Mulder et al., 1996). Vegetation surveys on abandoned cutover fens indicate that these sites are more likely to spontaneously re-vegetate than cutover bogs, particularly when drainage is blocked (Graf et al., 2008). Thus, when planning peatland restoration it is important for managers to measure chemistry of the residual peat and choose an appropriate donor site to increase the likelihood of propagule survival, and reduce cost and effort on sites where application of donor material may not be required.

Despite the fact that most plants on the restored site likely did not result from the donor material, cover of both mosses and vascular plants was similar to that reported for other peatland restoration projects in Canada. For example, Graf et al. (2012) report mean total moss cover for nine restored peatlands near Rivière-du-Loup, Québec, Canada at ~20% and total vegetation cover for the same sites at just over 30% 3 yr post-restoration. The presence of many wetland species not characteristic of bogs following restoration has also be reported in previous studies (e.g. Poulin et al., 2012).

Although spontaneous revegetation can occur following peatland extraction, moss cover often remains low (Graf et al., 2008; Poulin et al., 2005). Mosses are important components of many undisturbed peatlands, make a major contribution to peat accumulation (Vitt, 2000), and *Sphagnum* has been called a keynote genus for peatland restoration (Rochefort, 2000). Because of the very different porosity and water retention of near surface *Sphagnum* moss, newly formed *Sphagnum* moss carpets on restored cutover peatlands and the residual cutover peat, establishment of a thick new *Sphagnum* moss layer appears critical to return hydrologic function to restored sites (Lucchese et al., 2010; McCarter and Price, 2013). However, it remains unclear how important this near-surface moss layer is for fen hydrology (Graf et al., 2012). On the other hand, the present study highlights the importance of vascular plants, particularly graminoid cover in this case, for carbon sequestration. This has been reported elsewhere, with *E. vaginatum* often playing an important role in initial high rates of CO$_2$ uptake following restoration (e.g. Tuittila et al., 1999). Yli-Petäys et al. (2007) even suggest that reduced rates of CO$_2$ uptake several decades following restoration are partially linked to reduction in vascular plant cover. Thus, while a return to long-term ecohydrological function of restored peatlands, particularly bogs, likely requires successful moss establishment, vascular plants clearly play an important role in post-restoration carbon sequestration. On the other hand, vascular plant litter is generally more easily decomposed compared to moss litter (e.g. Thormann et al., 2001) and bogs have been reported to accumulate peat at greater rates than fens (e.g. Turunen et al., 2002). However, Bauer et al. (2003) actually report greater rates of peat accumulation for rich fen areas compared to *Sphagnum* dominated areas in a peatland complex of the Western Boreal Plain. While these differences were likely hydrologically driven as *Sphagnum* dominated areas tend to have deeper water tables than rich fens in this region, results do indicate that fen-type vegetation can result in relatively rapid accumulation of peat with large herbaceous contribution. Evaluating the importance of mosses for long term rates of peat accumulation will require both year round monitoring of carbon fluxes and monitoring of restored sites for longer periods post-restoration.
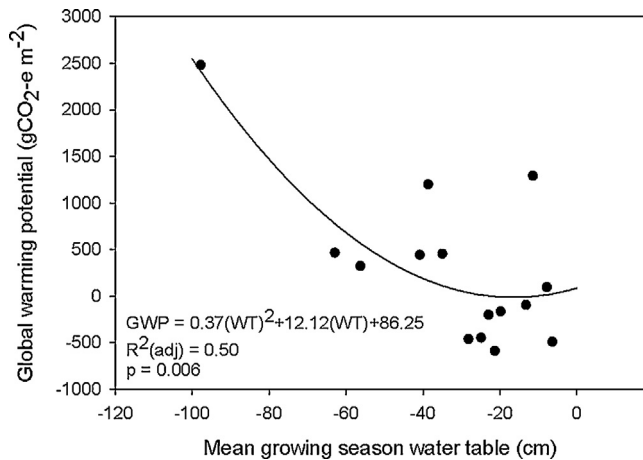
Fig. 6. Global warming potential (for CO_2 and CH_4) vs. mean growing season water table position across all unrestored and restored plots.

Graminoid cover accounted for a large proportion of total plant cover and was correlated to water table position, and this was likely an important driver for the relationship between water table and CO_2 exchange (Fig. 3). However, Chimner and Cooper (2003) report water table control on graminoid net primary productivity in a mountain fen suggesting that shallow water table is also likely to result in higher GEP once plants are established. Water table also largely controls CH_4 flux (this study, Couwenberg and Fritz, 2012) and thus, while wet restored sites were larger seasonal sinks for CO_2, they were also much larger sources of CH_4. As peatland restoration may be implemented with a goal to reduce GHG emissions from disturbed sites, it is important to consider the impact of management decisions on resulting greenhouse gas budgets. While on average restoration reduced GWP from CO_2 and CH_4 at both dry and wet restored plots, both very dry and very wet plots remained GHG sources due to continued high emissions of CO_2 (dry sites) or high CH_4 efflux (wet sites).

Total GHG balance in a peatland will also depend on fluxes of nitrous oxide (N_2O), which was not measured in this study. Although several studies report low N_2O emissions from undisturbed northern peatlands, drainage tends to increase emissions (Maljanen et al., 2010). Even given this increase, reported values from active and abandoned cutover peatlands are low, between 0.06 and 0.09 g $N_2O\,m^{-2}\,yr^{-1}$ (Maljanen et al., 2010; Salm et al., 2012). Given that the GWP of N_2O is 298 times that of CO_2 on a 100 yr time scale (IPCC, 2007), these fluxes would contribute 18–27 g CO_2-e $m^{-2}\,yr^{-1}$ to measured GWP, having little effect on most estimated values presented in this study. No N_2O flux estimates are available for cutover peatlands restored using moss-transfer. Investigation of nitrogen cycling in a restored cutover bog in Finland suggests that vascular plants compete with microbes for nitrate possibly reducing N_2O flux (Silvan et al., 2005) suggesting vegetation establishment following restoration would lower flux. Emission of N_2O is also well correlated to soil C/N ratio with very low emission below a ratio of 25 (Klemedtsson et al., 2005). Given that Bayley et al. (2005) report C/N ratios of 29–79 in the upper 20 cm of Alberta peatlands including both minerotrophic and ombrotrophic sites, it is likely that N_2O emissions from the study site are low. In general, more data on N_2O emissions from restored and rewetted peatlands are required.

Plotting GWP based on CO_2 and CH_4 flux against mean seasonal water table results in a unimodal relationship (Fig. 6) that suggests that GHG sink at this site is optimized when water table is 17 cm below the surface. When only restored plots are included, the relationship is no longer statistically significant, but

suggests an optimum water table of 28 cm below the surface. This is consistent with the finding that CH_4 emissions are significantly higher from sites with mean annual water table shallower than −20 cm (Couwenberg and Fritz, 2012). Thus, in order to achieve greenhouse gas emission reductions (and possibly create sinks) managers should take care to level the peatland prior to restoration in order to avoid creation of very dry microsites where the potential to develop a sink for either carbon and GHG is likely to be limited. At wet microsites carbon accumulation is likely to be encouraged, but CH_4 emissions can also be high. More data on GHG fluxes from restored areas with very shallow water tables are needed to evaluate whether an upper water table limit should be considered to minimize CH_4 emissions and enhance GHG sinks. Finally, although peatland restoration provides an opportunity for possible GHG mitigation (e.g. Wilson et al., 2013), this is only one function of peatland ecosystems and restoration decisions should also consider other peatland ecosystem services such as hydrologic regulation and habitat provision.

6. Conclusions

Application of methods outlined in the North American Restoration Guide (Quinty and Rochefort, 2003) to a cutover peatland in the Western Boreal Plain Ecozone in Alberta, Canada resulted in a switch from a large growing season CO_2 source on unrestored plots to a growing season CO_2 sink 3 yr following restoration. Restoration also resulted in a shift from a small CH_4 sink to a CH_4 source of 3.7 g CH_4—C m^{-2} over the growing season. Plant cover increased post restoration with mean moss and vascular cover of restored plots of 27% and 57%, respectively. These results suggest that, despite the dry climate of the Western Boreal Plain, restoration methods developed in eastern Canada can be applied in this region.

Water table position on the restored peatland was well correlated to both CO_2 and CH_4 flux. Graminoid cover was positively correlated to water table resulting in wetter sites having higher graminoid cover while often also having higher vascular plant and total plant cover in general, and thus higher rates of net CO_2 uptake. On the other hand, some of the wettest plots were also very large sources of CH_4. Thus, care should be taken to avoid very dry microsites following restoration in order to maximize GHG emission reduction. Although wet sites will likely accumulate more carbon, there is potential for large CH_4 efflux. Finally, managers should determine chemistry of the residual peat layer prior to restoration in order to match donor material to local conditions and/or minimize restoration efforts.

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