



Annual carbon balance of a peatland 10 yr following restoration

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Abstract. Undisturbed peatlands represent long-term net sinks of carbon; however, peat extraction converts these systems into large and persistent sources of greenhouse gases. Although rewetting and restoration following peat extraction have taken place over the last several decades, very few studies have investigated the longer term impact of this restoration on peatland carbon balance. We determined the annual carbon balance of a former horticulturally-extracted peatland restored 10 yr prior to the study and compared these values to the carbon balance measured at neighboring unrestored and natural sites. Carbon dioxide (CO_2) and methane (CH_4) fluxes were measured using the chamber technique biweekly during the growing season from May to October 2010 and three times over the winter period. Dissolved organic carbon (DOC) export was measured from remnant ditches in the unrestored and restored sites. During the growing season the restored site had greater uptake of CO_2 than the natural site when photon flux density was greater than $1000 \mu\text{mol m}^{-2} \text{s}^{-1}$, while the unrestored site remained a source of CO_2 . Ecosystem respiration was similar between natural and restored sites, which were both significantly lower than the unrestored site. Methane flux remained low at the restored site except from open water pools, created as part of restoration, and remnant ditches. Export of DOC during the growing season was 5.0 and 28.8 g m^{-2} from the restored and unrestored sites, respectively. Due to dry conditions during the study year all sites acted as net carbon sources with annual balance of the natural, restored and unrestored sites of 250.7 , 148.0 and 546.6 g C m^{-2} , respectively. Although hydrological conditions and vegetation community at the restored site remained intermediate between natural and unrestored conditions, peatland restoration resulted in a large reduction in annual carbon loss from the system resulting in a carbon balance more similar to a natural peatland.

1 Introduction

Peatlands play an important role in the global carbon cycle storing an estimated 469 to 486 Gt of carbon (Page et al., 2011), emitting approximately 10% of all global methane (CH_4) emissions (Mikaloff Fletcher et al., 2004) and acting as large sources of particulate and dissolved organic carbon to downstream ecosystems (e.g. Billett et al., 2004). In addition, peat is an important mineral and biological resource that is extracted for fuel and horticultural uses. In Canada, peat is primarily used for horticulture with 24 000 ha of peatland disturbed for horticultural peat extraction of which 14 000 ha are currently active (Environment Canada, 2010). Extraction of peat for horticultural use involves removing surface vegetation and draining the peatland (Waddington and Price, 2000). Once the peatland is no longer economical for extraction it is abandoned (Waddington and Price, 2000). These drained peatlands usually have a depth of peat remaining, and the dry aerated soil mineralizes at a higher rate than natural peatlands, resulting in a large persistent source of carbon dioxide (CO_2) (Waddington et al., 2002).

Peatland restoration projects following peat extraction have been undertaken in North America and Europe for several decades (e.g. Tuittila et al., 1999; Waddington and Price, 2000; Cobbaert et al., 2004; Wilson et al., 2009; Samaratani et al., 2011). Many projects involve simply rewetting the peatland; however, in North America most projects follow the peatland restoration process outlined by Quilty and Rochefort (2003). Briefly, this process involves leveling of the peat surface, spreading vegetative material collected from a donor site in a ratio of 1 : 10, covering the introduced material with straw mulch and blocking drainage ditches (Rochefort et al., 2003). In some cases dykes are also built to hold back surface water and phosphorus fertilizer may be applied to encourage establishment of *Polytricum* moss and

vascular plants that act as nurse plants for *Sphagnum* moss (Quinty and Rochefort, 2003).

The North American goals of peatland restoration in the short-term are to (1) establish a plant cover composed of typical peatland species and (2) restore hydrology characteristic of undisturbed peatlands, with the long-term goal of returning peat and carbon accumulation functions (Rochefort et al., 2003). Application of rewetting and/or additional restoration measures appears to improve site water balance and near-surface moisture conditions (e.g. Shantz and Price, 2006), vegetation cover and species composition (e.g. Tuittila et al., 2000; Poulin et al., 2012), microbial community structure (Andersen et al., 2010), accumulation of fresh biomass on the peat surface (Lucchese et al., 2010) and may re-establish carbon accumulation in the short-term (Tuittila et al., 2000; Waddington et al., 2010).

Tuittila et al. (1999) measured net growing season carbon accumulation in a Finnish peatland three years after restoration. In contrast, measurements on a revegetated cutover peatland five decades post-extraction determined that the site was a net source of carbon possibly due to a low cover of vascular plants, resulting in limited productivity (Yli-Petays et al., 2007). Measurements made in a section of a cutover peatland in the Swiss Jura mountains after 29–51 yr of regeneration suggest that up to 50 yr may be required before carbon accumulation function is regained (Samaratani et al., 2011). Waddington et al. (2010) report that a restored peatland in Quebec, Canada was a net sink of CO₂ during the growing season and estimated that the site would be a net annual carbon sink 6–10 yr post-restoration.

Since very few studies have determined how a restored peatland functions in the longer term it remains unclear when, or even if, the restored ecosystem will have carbon fluxes similar to a natural peatland. Moreover, most studies have focused only on growing season CO₂ fluxes, and thus the complete annual carbon balance of the peatland cannot be assessed. Therefore, the objective of this study was to determine the annual carbon balance of a restored peatland 10 yr after restoration.

2 Study site

The study was conducted at the Bois-des-Bel (BDB) peatland (47.9671° N, 69.4285° W) located approximately 11 km northeast of Rivière-du-Loup, Quebec, Canada. The 11.5 ha cutover section of peatland is part of a 200 ha open and treed bog complex. Horticultural peat extraction began in 1972 and continued until 1980. The cutover peatland was divided into two sections (Fig. 1) and in 1999 restoration activities took place according to the North American peatland restoration guide (Quinty and Rochefort, 2003) on the 7.5 ha restored site. This restored site was subdivided into four sections with the construction of dykes to hold back snowmelt water. The eastern three sections were restored in autumn 1999,

while the remaining westernmost section was restored in autumn 2000. Prior to restoration, the restored site was cleared of all vegetation. In addition to the restoration steps outlined in the Introduction, eight open water pools were also created (Fig. 1). A 1.8 ha section was left untouched and will be referred to as the unrestored site. The restored and unrestored sites were separated by a buffer strip (Fig. 1). Measurements were also made within an open section (un-treed) of the undisturbed peatland referred to as the natural site.

Determination of CO₂ and CH₄ flux was carried out at plots distributed across the sites. At least one plot was installed in each of the fields at the restored site with additional plots chosen to represent the diversity of vegetation cover and microtopography that was observed. Triplicate plots were also placed on the open water pools and ditches. In total, 14 plots were installed on the restored peat fields with an additional three on pools and three on ditches for a total of 20 restored site plots (Fig. 1). At the unrestored site, a vegetation gradient was observed, with very little spontaneous recolonization in the northeast portion of the site, much higher vegetation cover at the southwest portion and intermediate coverage between these areas. In each of the two unrestored fields, one plot was randomly placed in each of these three vegetation areas for a total of six unrestored plots. At the natural site, six plots were installed to capture the microtopographic gradient with triplicate plots at each of hummocks and hollows. Boardwalks were installed next to each plot to reduce disturbance during measurements.

At all plots measurements were made several times per month between 15 May and 15 October, 2010. Non-growing season measurements of CO₂ and CH₄ flux were conducted three times, January, February and March, on a subset of the sampling plots.

3 Methods

3.1 Carbon dioxide exchange

Carbon dioxide exchange was determined using the closed chamber method. The net ecosystem exchange (NEE) of CO₂ was determined with a clear acrylic chamber (60 × 60 × 30 cm) placed on a stainless steel collar (60 × 60 cm) permanently installed at each sampling plot. A groove in the collar held the chamber and was filled with water to create a seal. A battery-operated fan installed inside the chamber circulated the headspace air throughout the measurement period and the chamber was lifted from the collar between each measurement and allowed to equilibrate to ambient CO₂ concentration and temperature. The concentration of CO₂ was determined inside the chamber at 15 s intervals for a maximum of 105 s using a portable infrared gas analyzer (EGM-4, PPSystems, Massachusetts, USA). The linear change in CO₂ concentration over time was used to calculate NEE. Shrouds were used to reduce the incoming

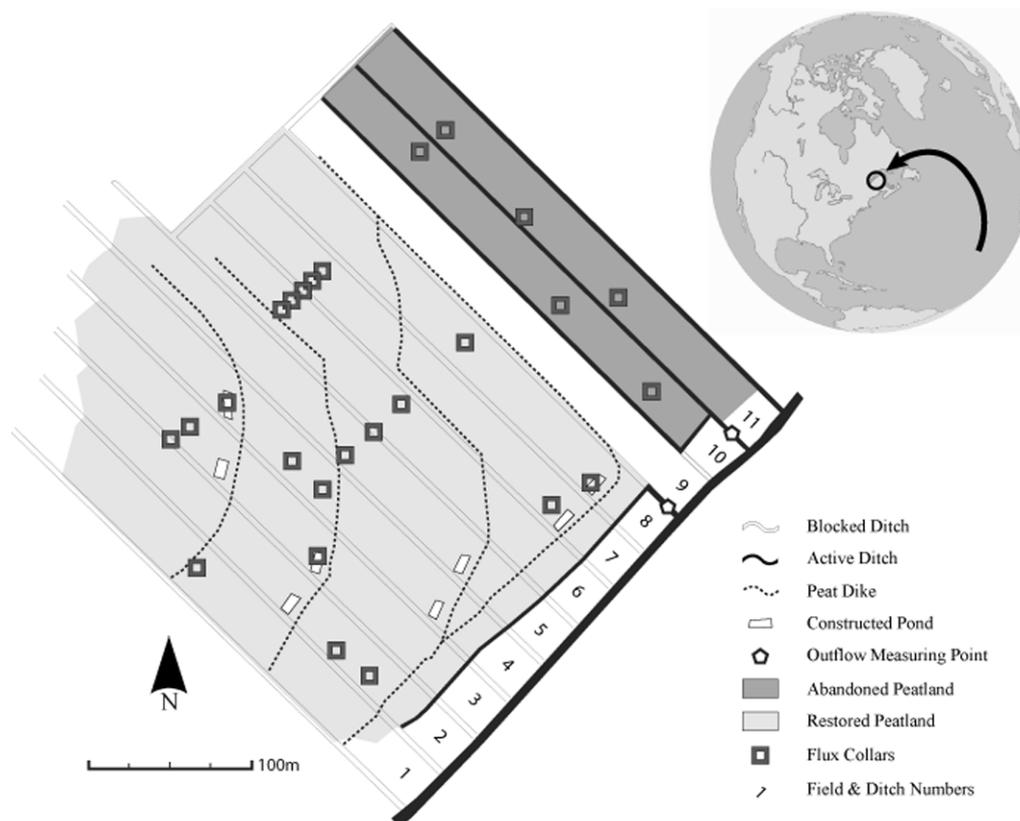


Fig. 1. Bois-des-Bel study site showing restored and unrestored (abandoned) sites. The natural site is located approximately 1.7 km west of the restored site in the neighboring undisturbed peatland

radiation inside the chamber and the measurement was repeated. Ecosystem respiration (ER) was determined by darkening the chamber with an opaque shroud. Gross ecosystem photosynthesis (GEP) was calculated as the sum of NEE and ER. We use the convention that negative values indicate uptake of CO_2 from the atmosphere by the ecosystem (net sink).

3.2 Methane flux

Methane flux was determined using the closed chamber method at the collars described above. For CH_4 flux determination opaque stainless steel chambers (60 cm \times 60 cm \times 30 cm) were used. A battery-operated fan circulated the headspace air during the measurement period. Chambers were put in place for 35 min with gas samples collected at 7, 15, 25 and 35 min following chamber closure. Gas samples were stored in pre-evacuated vials (Exetainers, Labco Ltd., UK) and sent to the University of Calgary for determination of CH_4 concentration on a Varian 3800 gas chromatograph equipped with a flame ionization detector. Methane flux was determined from the linear change in CH_4 concentration in the headspace over time.

3.3 Dissolved organic carbon export

Remnant ditches at both the restored and unrestored sites continued to discharge water from the cutover peatland. All discharge from each site was collected at two separate outflow weirs (Fig. 1). Discharge was measured manually, bi-weekly at the weirs and regressed against continuously monitored water level (Levellogger, Solinst) to determine discharge over the study period (see also McCarter and Price, 2013). Water samples were collected weekly during base-flow for determination of DOC concentration. Five storms were also monitored with water samples collected prior to the storm, and every four hours during the event up to 16 h following the cessation of precipitation.

For DOC concentration determination, water samples were filtered through 0.4 μm glass fiber filters (Macherey-Nagel GF-5). Absorbance by each sample was determined at 400 nm on a Perkins-Elmer 3B Lambda UV-Visible Spectrophotometer. A subset of samples was preserved by acidification and analyzed for DOC content on a total organic carbon analyzer (Shimadzu TOC-V) following sparging of inorganic carbon. Absorbance at 400 nm was regressed against DOC concentration for these samples and used to determine DOC concentration in all samples.

When all samples were combined there was no significant relationship between discharge and DOC concentration. Thus, growing season (May–October) DOC export (DOC) was estimated using method 5 outlined in Walling and Webb (1985):

$$\text{DOC} = \left[\frac{K \sum_{i=1}^n (C_i Q_i)}{\sum_{i=1}^n Q_i} \right] Q_r, \quad (1)$$

where K is a correction factor (d season^{-1}) to convert from daily to seasonal time step, C_i is the instantaneous DOC concentration (g L^{-1}), Q_i is the instantaneous discharge (L d^{-1}) and Q_r is the mean discharge over the sampling period (L d^{-1}). Dividing DOC by site (drainage) area (m^2) results in an area based DOC export estimate ($\text{g m}^{-2} \text{ season}^{-1}$).

3.4 Environmental variables

During each CO_2 and CH_4 exchange measurement, water table was determined in a dipwell adjacent to each sampling plot. A soil temperature profile was measured with a thermocouple soil probe at 5 cm intervals to a depth of 20 cm. Temperature inside the chamber was measured with a thermocouple thermometer. Photosynthetically active radiation (PAR) was monitored with a quantum sensor attached to the infrared gas analyzer. Water table position, precipitation, air and soil temperature, and PAR were measured continuously and averaged every half hour at meteorological stations (CR10X, Campbell Scientific Canada, Edmonton, AB, Canada) located at the restored and natural sites.

In July, a vegetation survey was carried out at each of the sampling plots. All species present were identified and their cover estimated visually to the closest 1 %.

3.5 Annual carbon balance

Carbon dioxide exchange during the growing season was estimated using empirical models parameterized for each sampling plot. Gross ecosystem photosynthesis was estimated according to (modified from Riutta et al., 2007):

$$\text{GEP} = \frac{\text{PAR} \times P_{\max}}{(\text{PAR} + k)} \times e^{\left[-0.5x \left(\frac{\text{WT} - \text{WT}_{\text{opt}}}{\text{WT}_{\text{tol}}} \right)^2 \right]} \times e^{\left[-0.5x \left(\frac{T - T_{\text{opt}}}{T_{\text{tol}}} \right)^2 \right]}, \quad (2)$$

where P_{\max} is the maximum rate of GEP ($\text{g CO}_2 \text{ m}^{-2} \text{ d}^{-1}$) when water table and temperature are not limiting, k is the level of PAR ($\mu\text{mol m}^{-2} \text{ s}^{-1}$) at which half of the maximum rate of GEP occurs, WT is the water table position (cm), WT_{opt} and WT_{tol} are parameters in a Gaussian response of GEP to water table representing the water table when GEP is optimized and the width of the Gaussian curve, T is the soil temperature at 5 cm below the surface ($^{\circ}\text{C}$) and T_{opt} and T_{tol} represent optimum temperature and width of the Gaussian response as described for water table. Ecosystem respiration

was estimated from multiple linear regression with water table position and temperature at 5 cm below the soil surface according to

$$\text{ER} = a \times T + b \times \text{WT} + c, \quad (3)$$

where a , b and c are regression parameters.

Both GEP and ER were estimated based on Eqs. (2) and (3) for each half hourly period between 1 May and 31 October, averaged daily and summed for a growing season total. Growing season NEE was determined by summing seasonal GEP and ER estimates.

At many sampling plots there was no significant relationship between daily CH_4 flux and water table position or soil temperature. Thus, growing season CH_4 flux was estimated by weighting each flux measurement based on the number of days between measurements and summing all values for the seasonal total flux.

Export of DOC during this period was determined using Eq. (1). Since the open, natural portion of the peatland was located in the center of the peatland with no clear outflow, DOC export from the natural portion was not estimated in this study.

Non-growing season (November–April) fluxes of CO_2 and CH_4 were estimated by multiplying the mean wintertime flux value for each site (natural, restored and unrestored) by the number of days during this period. Losses of DOC during winter were assumed to be negligible. Snowmelt DOC fluxes were not measured in this study, but it is clear that snowmelt may account for a substantial portion of annual DOC export (Waddington et al., 2008; Dyson et al., 2011). An estimate of DOC export during snowmelt was obtained by weighting the snowmelt flux reported by Waddington et al. (2008) for the reported snowfall at the Rivière-du-Loup weather station for the winter of 2009–2010 (Environment Canada, 2012).

3.6 Statistical analyses

As P_{\max} determined in Eq. (2) is a theoretical maximum rate of GEP which may never actually be attained, we evaluated differences in maximum rates of GEP and NEE by comparing CO_2 flux when PAR photon flux density was greater than $1000 \mu\text{mol m}^{-2} \text{ s}^{-1}$ (GEP_{\max} , NEE_{\max}) according to Bubier et al. (2003a). Data for CO_2 and CH_4 fluxes were non-normally distributed, thus non-parametric Kruskal–Wallis and Mann–Whitney tests with Bonferroni correction for multiple comparisons were used to evaluate differences between sites resulting in a corrected p value of 0.005. Linear and non-linear regression was used to evaluate potential controls (water table, vegetation cover, etc.) on rates of CO_2 and CH_4 exchange. All analyses were performed in Minitab 14.1.

4 Results

4.1 Weather and site characteristics

Overall, 2010 was slightly warmer and drier than the 30 yr normal. Long-term data (1971–2000) were available for the Cacouna meteorological station ~ 8 km west of the study site, while 2010 data were only available at Rivière-du-Loup station ~ 22 km further west (Environment Canada, 2012). Based on these data, average annual temperature and total precipitation was 5.2°C and 886 mm for 2010 compared the 30 yr normal of 3.2°C and 963 mm. July and August were particularly dry, receiving only 54 and 10 % of normal precipitation, respectively. In contrast, September was quite wet, receiving more than 200 % of normal precipitation.

The dry summer conditions resulted in deep water tables in July and August at all sites. Average water table position between May and October was -15.3 , -26.5 and -47.7 cm at natural, restored and unrestored sites, respectively (Table 1).

Vegetation community varied greatly between sites and plots. While moss cover at the unrestored site remained very low, vascular plant cover was as high as 70 % on one plot. Both the restored and natural sites had extensive moss cover, with slightly higher sedge cover at the restored site while the natural site had a larger proportion of shrubs (Table 1). Detailed description of vegetation communities can be found in Poulin et al. (2012).

4.2 Carbon dioxide exchange

During the growing season (May–October) when photon flux density of PAR was greater than $1000 \mu\text{mol m}^{-2} \text{s}^{-1}$ both GEP_{max} and NEE_{max} were significantly different between sites (Fig. 2; GEP : Kruskal–Wallis, $H = 50.62$, $p < 0.001$; NEE : Kruskal–Wallis, $H = 87.70$, $p < 0.001$). While GEP_{max} was similar at the unrestored site to restored ditches and pools, it was significantly lower than both restored fields (Mann–Whitney, $p < 0.0001$) and the natural peatland (Mann–Whitney, $p = 0.003$). Restored fields had significantly greater GEP_{max} than the natural site (Mann–Whitney, $p < 0.0001$). The unrestored site acted as a source of CO_2 even under full light conditions, having significantly lower CO_2 uptake as NEE_{max} than either the restored (Mann–Whitney, $p < 0.0001$) or natural sites (Mann–Whitney, $p < 0.0001$). The restored fields had greater uptake of CO_2 than the natural peatland plots (Mann–Whitney, $p = 0.0007$).

Average growing season GEP_{max} was significantly related to vascular plant cover determined in July (Fig. 3a; linear regression, $R^2 = 0.37$, $p < 0.001$) where higher plant cover resulted in higher productivity. A similar pattern was observed between NEE_{max} and vascular plant cover although this was not statistically significant. Using all individual flux measurements from all plots, NEE_{max} was significantly correlated with water table position where shallower water ta-

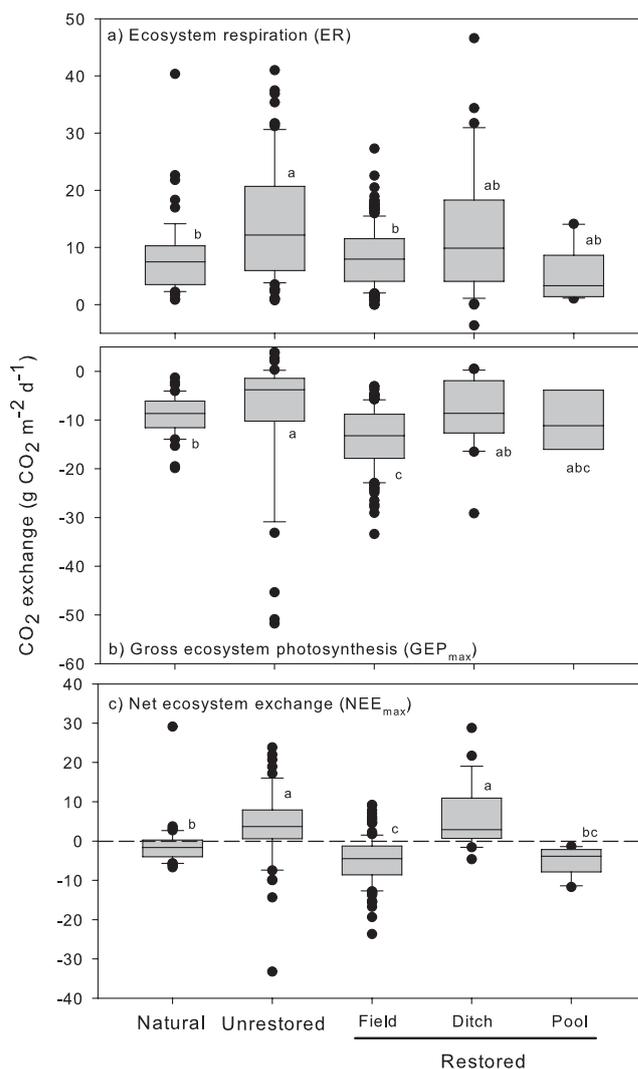


Fig. 2. Ecosystem respiration (a), gross ecosystem photosynthesis (b) and net ecosystem exchange (c) at natural, unrestored and restored sites during the growing season (May–October). Values for gross ecosystem photosynthesis and net ecosystem exchange represent only those measured when photon flux density of photosynthetically active radiation was greater than $1000 \mu\text{mol m}^{-2} \text{s}^{-1}$. Box plots indicate 10th and 90th percentile with bars, 25th and 75th percentiles with top and bottom of box and median with center line. Medians are significantly different from each other if they share no letters in common. Letters should only be compared within one panel.

ble resulted in increased CO_2 uptake, although this explained only 4.1 % of the variation in fluxes (Fig. 3b; linear regression, $p < 0.001$).

Ecosystem respiration was also significantly different between sites (Fig. 2, Kruskal–Wallis, $H = 31.75$, $p < 0.001$). The unrestored site had higher ER than both restored (Mann–Whitney, $p < 0.0001$) and natural sites (Mann–Whitney, $p < 0.0001$), which were not significantly different from

Table 1. Site characteristics^a.

Site	Water table (cm)	Total moss cover (%)	Total vascular plant cover	Total shrub cover	Total sedge cover
Natural	−15.3 (−10.8 to −22.2)	90.3 (61 to 104)	21.9 (10 to 47)	12.9 (3 to 37)	2.5 (0 to 9)
Unrestored	−47.7 (−40.7 to −54.8)	0.1 (0 to 0.5)	30.1 (4 to 71)	24.8 (0 to 60)	0
Restored	−26.5 (−7.7 to −43.4)	88.4 (43 to 114)	20.3 (8 to 37)	10.8 (0 to 36)	7.5 (0 to 30)
-field	−6.8 (−3.2 to −9.2)	46.7 (30 to 60)	10.8 (8 to 17)	1.0 (0 to 3)	0
-ditch	48.5 (43.8 to 55.2)	53.3 (0 to 90)	5.0 (0 to 10)	0	1.2 (0 to 3)
-pool					

^a Value given is the mean May to October water table, or July vegetation cover over all study plots at each site. Ranges of observed mean water table or vegetation cover for plots at each site are given in brackets.

each other. Restored pools and ditches had rates of ER that were not significantly different from any of the other sites. A deeper water table position resulted in higher ER (linear regression, $R^2 = 0.17$, $p < 0.001$), while warmer soil temperature at 5 cm depth also resulted in higher ER (linear regression, $R^2 = 0.19$, $p < 0.001$). Combining both water table and temperature explained 29% of variation in ER among plots and sampling dates (linear regression, $p < 0.001$).

Models of GEP according to Eq. (2) were all statistically significant ($p < 0.01$ in all cases). They generally explained greater than 70% of the variation in the data except for one unrestored, one ditch and one restored plot where only 40, 45 and 64% of the variation was accounted for, respectively. Standard error of the estimate for GEP was 0.51–1.48, 0.34–3.38 and 0.51–1.16 $\text{g C m}^{-2} \text{d}^{-1}$ for models at the natural, unrestored and restored site, respectively. Variation in ER was similarly well accounted for based on Eq. (3) although almost no correlation between ER and temperature or water table was observed for one ditch plot. Standard error of the estimate for ER was 0.26–1.21, 0.28–1.48, and 0.40–0.99 $\text{g C m}^{-2} \text{d}^{-1}$ for models at natural, unrestored and restored plots. Combining these models with measured environmental variables from the meteorological stations installed at the restored and natural sites resulted in growing season estimates of GEP, ER and NEE for the period May 1 to October 31, 2010. Modeled growing season GEP was -162.6 to $-529.5 \text{ g C m}^{-2}$ at natural collars, -42.6 to $-617.5 \text{ g C m}^{-2}$ at unrestored collars and -112.9 to $-411.5 \text{ g C m}^{-2}$ at restored collars. Modeled ER for the same time period was 270.0 to 937.0 g C m^{-2} , 188.4 to 1305.3 g C m^{-2} and 185.6 to 582.4 g C m^{-2} at natural, unrestored and restored sites, respectively. Adding these values together resulted in estimated growing season NEE of 107.5 to 407.5 g C m^{-2} for the natural site, 145.8 to 687.8 g C m^{-2} at the unrestored site and -24.2 to 203.4 g C m^{-2} at the restored site. Ditches and pools on the

restored site had an average modeled growing season GEP of -178.2 and $-137.7 \text{ g C m}^{-2}$, respectively. Ditch ER was on average 628.9 g C m^{-2} resulting in NEE at restored ditches of 450.7 g C m^{-2} over the growing season. At pools, mean ER was 216.2 g C m^{-2} while mean growing season NEE was 78.4 g C m^{-2} (Table 2).

Mean non-growing season flux of CO_2 was 0.9, 1.2 and 0.8 $\text{g CO}_2 \text{ m}^{-2} \text{d}^{-1}$ from natural, unrestored and restored sites, respectively. Although CO_2 flux was slightly higher in January (Fig. 4) than the other two sampling dates, there were no significant differences in CO_2 flux with date (Kruskal–Wallis, $p = 0.079$) or site (Kruskal–Wallis, $p = 0.469$). Mean ditch CO_2 flux during the non-growing season was 0.9 $\text{g CO}_2 \text{ m}^{-2} \text{d}^{-1}$ with no measurements made on pools during this period. Applying measured flux values to the period from 1 November to 30 April resulted in estimated non-growing season emission of 46.0, 57.9 and 39.9 g C m^{-2} from the natural, unrestored and restored sites (Table 2).

4.3 Methane flux

Growing season mean (standard deviation) CH_4 flux was 35.9 (27.6), -1.3 (3.2), 1.8 (4.1), 38.6 (48.6), and 164.7 (145.6) $\text{mg CH}_4 \text{ m}^{-2} \text{d}^{-1}$ at natural, unrestored, restored field, ditch and pool plots, respectively. Methane flux was significantly different between sites (Kruskal–Wallis, $H = 93.94$, $p < 0.001$). The natural site had significantly higher CH_4 flux than both the restored fields (Fig. 5; Mann–Whitney, $p < 0.0001$) and unrestored site (Mann–Whitney, $p < 0.0001$). However, there was no significant difference between unrestored and restored fields (Mann–Whitney, $p = 0.025$). Restored ditches had significantly higher CH_4 flux than both restored and unrestored sites, but were not statistically different than the natural site, while pool CH_4 efflux was higher than all other sites (Fig. 5). Mean growing season CH_4 flux was non-linearly related to mean water table

Table 2. Carbon flux estimates for 2010^a.

	Natural			Unrestored			Restored			
		Field	Ditch ^b	Site ^c	Field	Ditch	Pool	Site ^c		
Growing Season (1 May–31 October)										
NEE (g CO ₂ -C m ⁻²)	198.1 (114.0)	459.7 (320.5)	450.7 (80.5)	459.2 (309.0)	83.0 (70.2)	450.7 (80.5)	78.4 (44.8)	101.7 (70.8)		
CH ₄ (g CH ₄ -C m ⁻²)	5.5 (4.4)	-0.2 (0.4)	4.9 (7.3)	0.06 (0.7)	0.4 (0.8)	4.9 (7.3)	15.7 (10.9)	0.8 (1.2)		
DOC (g C m ⁻²)	n.d.			28.8				5.0		
<i>Total</i> ^d (g C m ⁻²)	203.6 (118.4)			488.1 (309.7)				107.5 (72.0)		
Non-growing Season (1 November–30 April)										
NEE (g CO ₂ -C m ⁻²)	46.0 (54.5)	58.6 (31.6)	43.6 (56.4)	57.9 (32.8)	39.6 (39.3)	43.6 (56.4)	n.d.	39.9 (40.3)		
CH ₄ (g CH ₄ -C m ⁻²)	1.1 (1.1)	0.05 (0.2)	10.6 (17.8)	0.6 (1.0)	-0.04 (0.1)	10.6 (17.8)	n.d.	0.6 (1.1)		
<i>Total</i> ^d (g C m ⁻²)	47.1 (55.6)			58.5 (33.8)				40.5 (41.4)		
Annual Total (g C m ⁻²)	250.7 (174.0)			546.6 (343.5)				148.0 (113.4)		

^a Values are the mean of all measurements plots with standard deviation given in brackets. Natural site, $n = 6$; unrestored field, $n = 6$; ditches, $n = 3$; pools, $n = 3$; restored field, $n = 14$.

^b Restored ditch values used to estimate fluxes at unrestored ditches.

^c Weighted values based on area occupied by each feature. At the unrestored site, fields accounted for 95.2% of the area, with ditches occupying 4.8%. At the restored site, ditches and pools accounted for 5.1 and 0.9% of the area, respectively, with restored fields making up the remaining 94%.

^d Additional DOC loss during snowmelt was not measured, but estimated to account for 19.4 and 4.0 g C m⁻² at the unrestored and restored sites, respectively.
n.d. = not determined

position with very low fluxes when water table fell below approximately -20 cm (Fig. 6). There was no clear relationship between mean growing season CH₄ flux and July vascular plant cover (data not shown).

Mean total CH₄ emissions over the growing season (1 May–31 October 2010) were estimated as 5.5, -0.2, and 0.4 g CH₄-C m⁻² at the natural, unrestored and restored sites (Table 2). Restored ditches and pools were estimated to release on average 4.9 and 15.7 g CH₄-C m⁻² over the growing season, respectively.

Non-growing season CH₄ flux was on average 7.8, 0.4 and -0.3 mg CH₄ m⁻² d⁻¹ from natural, unrestored and restored sites. This resulted in mean non-growing season CH₄ flux of 1.1 for natural plots, 0.06 at unrestored plots and -0.05 g CH₄-C m⁻² at restored field plots. Ditches released an estimated 14.2 g CH₄-C m⁻² over the same period. There were no significant differences in CH₄ flux between the three sampling dates (Fig. 4); however, CH₄ flux was significantly higher at the natural site than either the unrestored (Mann–

Whitney, $p = 0.008$) or the restored sites (Mann–Whitney, $p = 0.002$).

4.4 Dissolved organic carbon export

The concentration of DOC in discharge from the unrestored site ranged from 75.2 to 134.8 mg L⁻¹ with a mean of 100.6 mg L⁻¹. At the restored site, concentration was between 49.2 and 129.3 mg L⁻¹ with mean 86.3 mg L⁻¹. Using Eq. (1), total DOC export between May and October was 28.8 and 5.0 g C m⁻² at the unrestored and restored sites, respectively. Continuously measured discharge between late June and early September (66 days) resulted in total discharge of 37 mm at unrestored and 7 mm at the restored site (McCarter and Price, 2013).

4.5 Annual carbon flux estimate

Annual carbon flux estimates were determined for each site based on modeled growing season NEE, estimated losses of

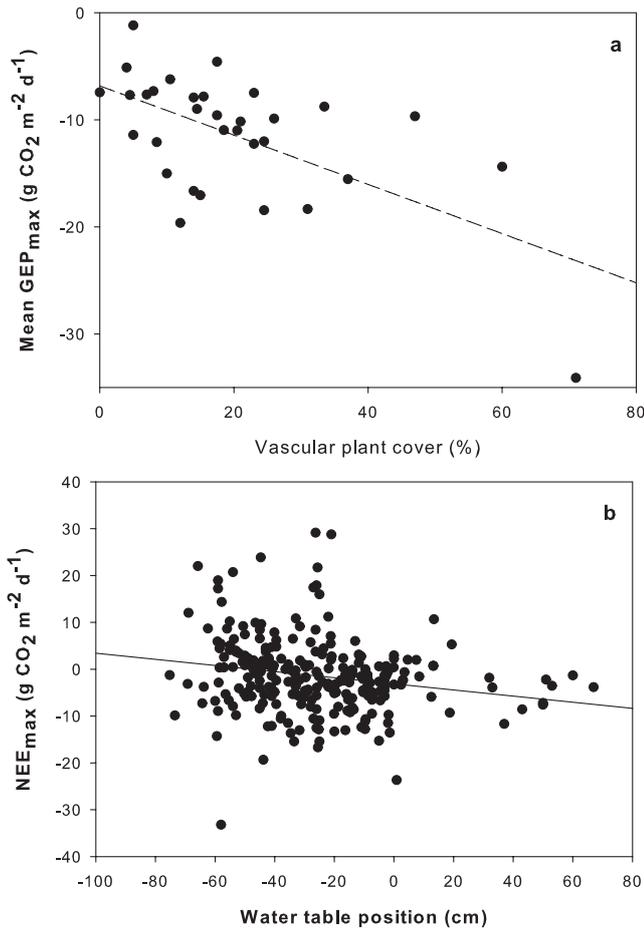


Fig. 3. (a) Regression between mean growing season gross ecosystem photosynthesis when photon flux density of photosynthetically active radiation was greater than $1000 \mu\text{mol m}^{-2} \text{s}^{-1}$ (GEP_{max}) and vascular plant cover in July ($\text{GEP}_{\text{max}} = -0.23$ (vascular cover) $- 6.8415$; $n = 32$, $R^2 = 0.37$, $p < 0.001$). (b) Regression between net ecosystem exchange when photon flux density of photosynthetically active radiation was greater than $1000 \mu\text{mol m}^{-2} \text{s}^{-1}$ (NEE_{max}) and the water table position ($\text{NEE} = -0.065$ (water table) $- 3.13$; $n = 261$, $R^2 = 0.04$, $p = 0.001$). Negative values indicate uptake of CO₂ by the ecosystem and water table position below the soil surface, respectively.

CH₄ during growing and non-growing season, CO₂ emissions during the non-growing season and growing season export of DOC. Fluxes from fields, ditches and pools were included based on their relative areal extent at the un-restored and restored sites. All sites were sources of carbon during the year 201 losing an estimated 250.7, 546.6 and 148.0 g C m⁻² yr⁻¹ at the natural, un-restored and restored sites, respectively. Non-growing season carbon losses accounted for 11–30 % of the total emissions depending on the site.

We did not measure DOC losses during the non-growing season nor during snowmelt and this will account for an

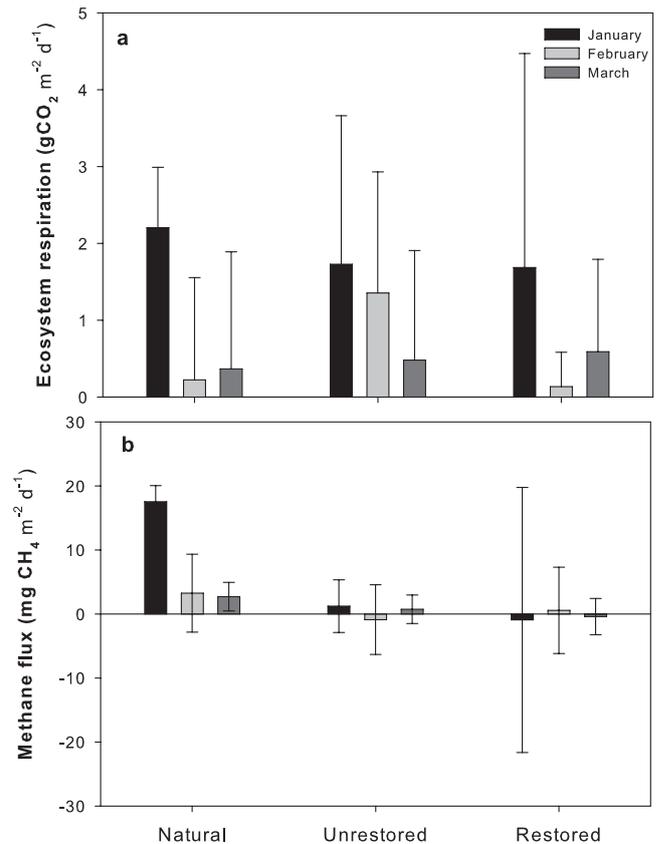


Fig. 4. Non-growing season ecosystem respiration (a) and methane flux (b) at natural, un-restored and restored field plots. Error bars give standard deviation of the mean.

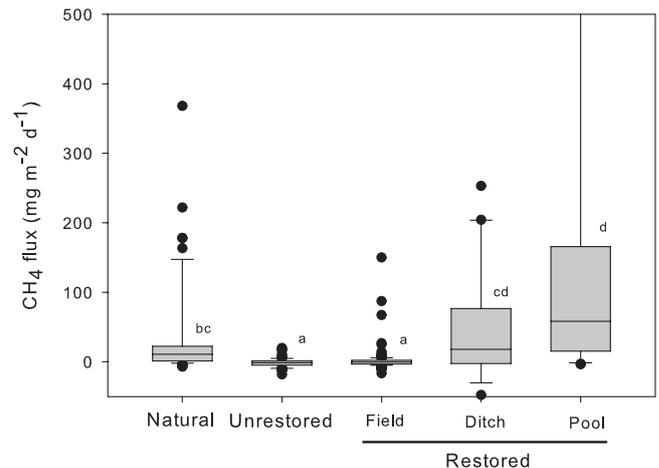


Fig. 5. Methane flux from natural, un-restored and restored plots during the growing season (May–October). Box plots indicate 10th and 90th percentile with bars, 25th and 75th percentiles with top and bottom of box and median with centerline. Medians are significantly different from each other if they share no letters in common.

additional loss of carbon from the site. Waddington et al. (2008) estimated snowmelt DOC export of 43.6 g C m^{-2} from 150 mm snow water equivalents (SWE) at the unrestored site and 8.3 g C m^{-2} from 137 mm at the restored site. Using snowfall data from the Rivière-du-Loup station (Environment Canada, 2012), snow on the ground on March 24, 2010 was 67 cm ($\sim 67 \text{ mm SWE}$). If DOC export is weighted to account for this smaller volume of snowmelt water, it can be estimated that snowmelt export may have been 19.4 and 4.0 g C m^{-2} from the unrestored and restored sites, respectively.

5 Discussion

Restoration of a former horticultural peatland has resulted in a substantial reduction in carbon losses 10 yr following the restoration activities. Furthermore, although all monitored sites acted as carbon sources during the study period, the restored site released less carbon than a neighboring natural peatland. The fact that all sites were sources of carbon in 2010 is likely linked to the dry midsummer conditions during the study period. Other studies have reported that peatlands may act as net sources of carbon during dry years (e.g. Alm et al., 1999; Waddington and Price, 2000; Bubier et al., 2003b). Although water tables were deeper at the restored site than the natural site (Table 1), estimated losses of carbon as CO_2 were greater from the former. This may result from differences in vegetation composition between the sites, a factor that has been observed to influence drought response (Bubier et al., 2003a; Strack et al., 2006). The restored site continues to have higher diversity than the natural site due to a combination of species introduced during restoration and those that have spontaneously colonized the site (Poulin et al., 2012). This may allow productivity to continue under a wider range of moisture conditions reflected in the higher rate of productivity under full light (GEP_{max}) at the restored site compared to the natural site. Specific factors contributing to the observed changes in each component of the carbon balance at the restored site are discussed in the subsequent sections.

5.1 Effect of restoration on CO_2 exchange

Restoration has significantly increased GEP and decreased ER on restored peat fields compared to the unrestored site (Fig. 2a, b). This has resulted in net uptake of CO_2 under full light conditions (NEE_{max}) at the restored site, while the unrestored site remained a source of CO_2 . Variability in CO_2 exchange remained high at the restored site due to differences in vegetation cover across the site and large water table variability over the season. Overall, productivity has increased following restoration largely due to an increase in both moss and vascular cover at the restored site (Table 1) resulting from the active reintroduction and protection of plant mate-

rial. In fact, there was a significant correlation between mean growing season GEP_{max} and vascular plant cover across all measurement plots. Although some unrestored plots had high vascular plant cover, other areas were poorly colonized and moss cover remained minimal across the site (see also Poulin et al., 2012). In contrast, the restored field plots had similar moss, sedge and shrub cover to plots at the natural site, although species composition continues to differ. For example, moss cover at restored plots has a higher proportion of *Polytrichum strictum* than natural plots that are almost completely *Sphagnum* covered.

Rewetting of the site through the restoration process has facilitated the successful establishment of plants. Ditch blocking and the creation of dykes has reduced discharge and increased water storage on the site resulting in shallower growing season water table position at restored fields (-26.5 cm) compared to unrestored fields (-47.7 cm). This shallower water table has led to significantly lower ER at the restored site due to a reduction in heterotrophic respiration. Although restoration has rewetted the site, water tables remained lower than the natural peatland plots (Table 1). This is likely caused by differences in soil structure between the sites in which the restored site continues to have a sharp transition between largely undecomposed moss in the upper 10–30 cm of the soil profile to highly compressed, decomposed peat below. The lower porosity and specific yield of this deeper soil results in large water table fluctuations when the water table falls below the newly formed moss layer (e.g. Lucchese et al., 2010; McCarter and Price, 2013), resulting in a deeper average water table over the growing season. Ditches remaining on site, although blocked, may also continue to provide some drainage to the peat fields during times of drought. Despite this deeper water table, ER was not significantly different between the natural and restored sites (Fig. 2), possibly due to the low substrate quality of this deeper peat at the restored site (Andersen et al., 2006).

Pools and ditches at the restored site had lower GEP than restored fields, possibly because inundated conditions limited plant colonization and productivity. In particular, ditches remained large sources of CO_2 despite shallow water tables and/or flooding throughout the growing season. It is likely that substrate collects in these ditches from the surrounding peat fields and is mineralized, resulting in high ER from these locations. High emission of CO_2 from ditches was reported previously at this site (Waddington et al., 2010) and in other drained peatlands (Sundh et al., 2000; Schrier-Uijl et al., 2011).

5.2 Effect of restoration on methane emissions

Peatland drainage and extraction generally greatly reduces CH_4 flux (Sundh et al., 2000; Maljanen et al., 2010; Waddington and Day, 2007) and may convert peat fields from CH_4 sources to sinks (e.g. Lohila et al., 2011). Rewetting and revegetation of a site following restoration can increase

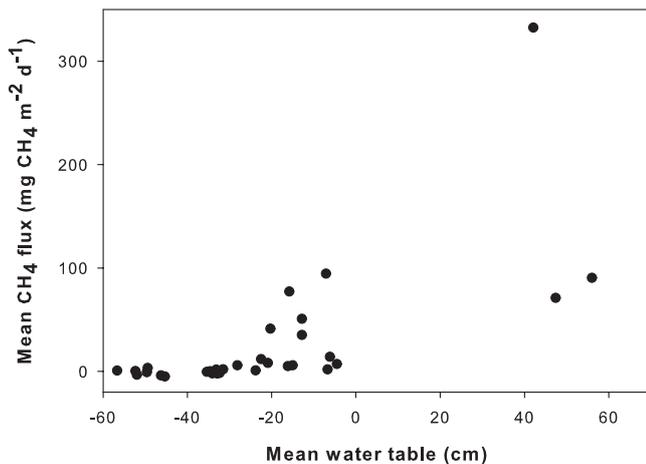


Fig. 6. Mean growing season methane flux versus mean growing season water table position including all natural, restored and unrestored plots. Negative water table indicates a water level below the soil surface.

CH₄ flux relative to unrestored areas (Tuittila et al., 2000; Wilson et al., 2009; Waddington and Day, 2007); however, as observed in the present study, emissions often remain lower than from undisturbed peatlands. The continued lower CH₄ flux from the restored site compared to the natural site (Fig. 5) is likely linked to deeper water table position at the former. The observed water table–CH₄ flux relationship suggests that CH₄ emission is greatly reduced when water table position falls below -20 cm (Fig. 6). Literature analysis of published CH₄ fluxes has reported a similar pattern (Couwenberg et al., 2011). Since mean growing season water table position at the restored site was -26.5 cm, the water table was often deep enough that very little CH₄ flux would be expected. Moreover, this depth in the peat profile generally occurs below the depth of new peat accumulation (Lucchese et al., 2010), suggesting that substrate quality could be limited, further reducing CH₄ production and flux.

Despite low CH₄ fluxes from restored peat fields, CH₄ emission from created pools on the restored site and remnant drainage ditches was the highest observed at any measured plots. Inundated conditions in pools combined with substrate supply from aboveground vegetation likely contributed to high CH₄ flux. Similarly, shallow water table and/or inundation at ditches and possibly substrate supply from not only vegetation within ditches, but also water flow from neighboring peat fields encouraged high ditch fluxes. High CH₄ fluxes from ditches in drained peatlands have been reported widely in the literature (Minkinen et al., 1997; Sundh et al., 2000; Waddington and Day, 2007; Schrier-Uijl et al., 2011) and should be included when determining ecosystem scale estimates of CH₄ flux from managed peatlands where ditches are present.

5.3 Effect of restoration on dissolved organic carbon export

Restoration has greatly reduced the export of DOC compared to the unrestored site. Despite an increase in soil water DOC concentration in situ following restoration (data not shown), DOC concentration in discharge water was lower from the restored site than the unrestored site. Discharge was also greatly reduced due largely to blocking the active drainage network and creating structures (pools, dykes) to increase water storage on the site. In general, differences in DOC export between the restored and unrestored site scaled closely to the reduction in discharge observed following restoration. As a result, DOC export from the restored site was estimated as only 5.0 g C m^{-2} over the growing season. Even including the estimated additional 4 g C m^{-2} loss from snowmelt results in an annual export value similar to the range of DOC export, $11.9\text{--}14.9 \text{ g C m}^{-2}$, reported for natural peatlands (Roulet et al., 2007; Nilsson et al., 2008; Koehler et al., 2011). Dissolved organic carbon export 10 yr following restoration was similar to that reported three years post-restoration (Waddington et al., 2008) suggesting that continued vegetation community changes and peat accumulation has had little influence on hydrologic carbon exports from the site. However, the impact of these ecohydrological changes on the chemistry of exported DOC requires further study.

6 Conclusions

Due to dry conditions during the growing season, the natural, unrestored and restored sites all acted as annual carbon sources during the study year. However, restoration greatly reduced carbon losses relative to the unrestored site, with losses even lower than those estimated for the natural peatland. This reduction in carbon emission has come primarily from a large decline in CO₂ efflux facilitated by a significant increase in plant productivity and decrease in ecosystem respiration linked to revegetation and rewetting of the restored site. Dissolved organic carbon export has also been greatly reduced by restoration through a decrease in discharge. In contrast, CH₄ emissions at the restored site, although higher than unrestored plots, remain much lower than the natural peatland, likely due to the deeper water table position at the restored site. Methane fluxes from open water pools and ditches were elevated and it is important to account for these features in overall budgets of carbon and greenhouse gas exchange. These results suggest that, although hydrological processes and rates of CH₄ efflux from the restored site are still intermediate between an unrestored and natural system, peatland restoration resulted in a large reduction in annual carbon loss from the system resulting in a carbon balance more similar to the natural peatland.

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