Annual greenhouse gas budget for a bog ecosystem undergoing restoration by rewetting

Sung-Ching Lee[1], Andreas Christen[1], Andy T. Black[2], Mark S. Johnson[3,4], Rachhpal S. Jassal[2], Rick Ketler[1], Zoran Nesic[1,2], Markus Merkens[5]

[1]Department of Geography / Atmospheric Science Program, The University of British Columbia, Vancouver, Canada
[2]Faculty of Land and Food Systems, The University of British Columbia, Vancouver, Canada
[3]Institute of Resources, Environment and Sustainability, The University of British Columbia, Vancouver, Canada
[4]Department of Earth, Ocean and Atmospheric Sciences, The University of British Columbia, Vancouver, Canada
[5]Parks, Planning and Environment Department, Metro Vancouver, Vancouver, Canada

Correspondence to: S.-C. Lee (sungching.lee@geog.ubc.ca)

Abstract. Many peatlands have been drained and harvested for peat mining, which has turned them from carbon (C) sinks into C emitters. Rewetting of disturbed peatlands facilitates their ecological recovery, and may help them revert to carbon dioxide (CO_2) sinks. However, rewetting may also cause substantial emissions of the more potent greenhouse gas (GHG) methane (CH_4). Our knowledge on the exchange of CO_2 and CH_4 following rewetting during restoration of disturbed peatlands is currently limited. This study quantifies annual fluxes of CO_2 and CH_4 in a disturbed and rewetted area located in the Burns Bog Ecological Conservancy Area in Delta, BC, Canada. Burns Bog is recognized as the largest raised bog ecosystem on North America's West Coast. Burns Bog was substantially reduced in size and degraded by peat mining and agriculture. Since 2005, the bog has been declared a conservancy area, with restoration efforts focusing on rewetting disturbed ecosystems to recover *Sphagnum* and suppress fires. Using the eddy-covariance (EC) technique, we measured year-round (16th June 2015 to 15th June 2016) turbulent fluxes of CO_2 and CH_4 from a tower platform in an area rewetted for the last 8 years. The study area, dominated by sedges and *Sphagnum*, experienced a varying water table position that ranged between 7.7 (inundation) and -26.5 cm from the surface during the study year. The annual CO_2 budget of the rewetted area was -179 g CO_2-C m^{-2} year^{-1} (CO_2 sink) and the annual CH_4 budget was 16 g CH_4-C m^{-2} year^{-1} (CH_4 source). Gross ecosystem productivity (GEP) exceeded ecosystem respiration (R_e) during summer months (June-August), causing a net CO_2 uptake. In summer, high CH_4 emissions (121 mg CH_4-C m^{-2} day^{-1}) were measured. In winter (December-February), while roughly equal magnitudes of GEP and R_e made the study area CO_2 neutral, very low CH_4 emissions (9 mg CH_4-C m^{-2} day^{-1}) were observed. The key environmental factors controlling the seasonality of these exchanges were downwelling photosynthetically active radiation and 5-cm soil temperature. It appears that the high water table caused by ditch blocking suppressed R_e. With low temperatures in winter, CH_4 emission was more suppressed than R_e. Annual net GHG flux from CO_2 and CH_4 expressed in terms of CO_2 equivalents (CO_2e) during the study period totaled to -55 g CO_2e m^{-2} year^{-1} (net CO_2e sink) and 1147 g CO_2e m^{-2} year^{-1} (net CO_2e source) by using 100-year and 20-year global warming potential values,

respectively. Consequently, the ecosystem was almost CO_2e neutral during the study period expressed on a 100-year time horizon but was a significant CO_2e source on a 20-year time horizon.

1 Introduction

35

Wetland ecosystems play a disproportionately large role in the global carbon (C) cycle compared to the surface area they occupy. Wetlands cover only 6% − 7% of the Earth's surface (Lehner and Döll, 2004; Mitsch et al., 2010), but they act as a major sink for the long-term C storage by sequestering carbon dioxide (CO_2) from the atmosphere. For example, strong C sinks (896 to 1139 g CO_2-C m^{-2} yr^{-1} and 1236 g CO_2-C m^{-2} $year^{-1}$) were found in Southeast USA and Eastern France,

40 respectively (Grasset et al., 2016; Mitsch et al., 2013). Other wetlands around the world sequester around 100 g CO_2-C m^{-2} $year^{-1}$ (Bortolotti et al., 2016; Lu et al., 2016; Petrescu et al., 2015). C storage in wetlands has been estimated to be up to 450 Gt C or approximately 20% of the total C storage in the terrestrial biosphere (Bridgham et al., 2006; Lal, 2008; Wisniewski and Sampson, 2012). However, wetlands emit significant quantities of methane (CH_4), a powerful greenhouse gas (GHG), due to anaerobic microbial decomposition (Aurela et al., 2001; Rinne et al., 2007). CH_4 emissions from wetlands are

45 responsible for 30% of all global CH_4 emissions (Bergamaschi et al., 2007; Bloom et al., 2010; Ciais et al., 2013). Peatlands are the most widespread of all wetland types in the world, representing 50 to 70% of global wetlands (Roulet, 2000; Yu et al., 2010). Their dynamics have played an important role in the global C cycle during the Holocene period (Gorham, 1991; Menviel and Joos, 2012; Yu, 2011), and it has been shown that it is crucial to include peatlands in the modelling and analysis of the global C cycle (Frolking et al., 2013; Kleinen et al., 2010; Wania et al., 2009).

50 Many peatlands have been harvested and continue to be disturbed by the extraction of peat for horticultural use. In the case of Burns Bog, peat was also used for fire bombs during World War II (Cowen, 2015). Generally, during harvesting, the surface vegetation is removed, and then wetlands are drained by a network of ditches (Price and Waddington, 2000; Waddington and Roulet, 2000). When no longer economical, many harvested peatlands are abandoned and kept at artificially low water tables due to the drainage ditches. This environmental condition limits the disturbed and abandoned peatlands

55 ability to return to their prior state. Drainage results in increased oxidation in peat soils, which then can become a strong source of CO_2 (Langeveld et al., 1997; Petrescu et al., 2015; Tapio-Biström et al., 2012). Additionally, degraded peat increases the risk of peatland fires, which could consequently cause significant CO_2 emissions (Gaveau et al., 2014; Page et al., 2002; van der Werf et al., 2004). These consequences could be worse if nothing is done after the peat extraction. Therefore, and for reasons of conservation ecology (unique habitat), disturbed peatlands may be restored.

60 Restoration efforts typically rely on elevating the water table and managing vegetation. The water table depth and the amount of vegetation are the most important factors affecting land-atmosphere C exchange. Rewetting by ditch blocking can have an immediate impact on the C exchange between the peatland surface and the atmosphere (Limpens et al., 2008). Rewetting has strong direct and indirect effects on CO_2 and CH_4 fluxes. Raising the water level has been found to suppress the CO_2 efflux from the soil and result in an increase in net CO_2 uptake by native bog vegetation (Komulainen et al., 1999).

65 CH_4 emissions from rewetted sections in a bog in Finland were three times higher than the release from the disturbed and dry area (Tuittila et al., 2000). Another study found similar rates of CH_4 production in disturbed and restored wetlands in the southern United States (Schipper and Reddy, 1994). Re-vegetation of degraded peat leads to faster re-establishment of peat formation that can have significant effects on C exchange. However, the increased above- and below-ground biomass of plants and litter enhances organic matter oxidation, which raises CO_2 emissions (Finér and Laine, 1998; Minkkinen and

70 Laine, 1998). In other studies, re-establishing the conditions permitting peat formation also initially increased CH_4 emission, but the C exchange did not reach the level of seasonal emissions from pristine peatlands (Crill et al., 1992; Dise et al., 1993; Shannon and White, 1994).

Very few studies provide continuous, long-term measurements to determine how restored and rewetted peatland ecosystems recover in terms of their productivity and GHG exchange. It remains unclear when, or even if, restored peatland

75 ecosystems could show a similar magnitude of C fluxes as in pristine (undisturbed) peatland ecosystems. Furthermore, most investigation focusing on GHG exchange of restored peatlands only measured CO_2 and/or CH_4 fluxes during short periods, e.g. the growing season. There are few studies that measured continuously and year-round fluxes (Anderson et al., 2016; Järveoja et al., 2016; Knox et al., 2015; Richards and Craft, 2015; Strack and Zuback, 2013), relying instead on sporadic, or repeating chamber measurements, which are difficult to upscale to annual totals.

80 In this study, we a) quantified seasonal and annual CO_2 and CH_4 fluxes, using the eddy covariance (EC) technique, in a disturbed ecosystem that is representative of areas subject to recent restoration efforts (ditch blocking for the last 8 years), b) identified key environmental controls and their effects on CO_2 and CH_4 fluxes, and c) quantified whether the study ecosystem is net source or sink of C and its net climate forcing at different time scales by considering GWPs of CO_2 and CH_4.

2 Study area

85 Burns Bog in Delta, BC, on Canada's Pacific Coast, is part of a remnant peatland ecosystem that is recognized as the largest raised bog ecosystem (2,042 ha) on North America's west coast. During the last century, it was significantly disturbed as a result of it being used for housing, peat mining and agriculture (MetroVancouver, 2007). The Burns Bog Ecological Conservancy Area (BBECA) was established in 2005 to conserve this large coastal raised bog and restore ecological integrity to the greatest extent possible. Christen et al. (2016) measured summertime CO_2 and CH_4 exchanges using

90 primarily chamber systems in several plots representative of disturbed areas of the BBECA, where some plots were rewetted and others were not. The study found substantial emissions of CH_4 primarily in recently rewetted plots, with highest emissions associated with high water tables. Nevertheless, a significant spatial and temporal variability was found between and within plots. In order to constrain these emission estimates, it was suggested to extend the year-round monitoring of CO_2 and CH_4 exchanges using EC technique to provide spatially more representative fluxes at a recently rewetted plot.

95 The current study site is located in a harvested, disturbed, and rewetted area in the centre of the BBECA (122°59'05.87"W, 49°07'47.20"N, WGS-84) with dimensions of 400 m by 250 m (Fig. 1). The field is surrounded by a

windbreak to the west and an abandoned (now blocked) drainage ditch to the north (see supplementary material, Fig. S1 and S2). The study area was harvested between 1957 and 1963 using the Atkins-Durbrow Hydropeat method to remove the peat (Heathwaite and Göttlich, 1993). In 2007, the study site was rewetted via ditch-blocking using dams built with plywood and using wooden stakes as bracing (Howie et al., 2009). Based on the weather data for 1981 to 2010 from the closest Environment Canada weather station, Vancouver International Airport, the average annual temperature was 10.4 ℃ and average annual precipitation was 1189 mm. Following rewetting, water table height (WTH) in the study area fluctuates between 30 cm above ground and 20 cm below ground over the year. In all years since rewetting started in 2007, water table positions were lower in late summer and early fall and high all winter and spring. WTH decreases steadily between June and September. In September and October, a water table rise due to the increase in precipitation and reduced evapotranspiration (ET) (Fig. 2) as a consequence of reduced available energy and senescence of sedges was observed, which is similar to water table observations in other temperate wetlands (Lafleur et al., 2005; Rydin et al., 2013) . The depth of peat at the study site is 5.83 m. A silty clay layer is located below the peat layer (Chestnutt, 2015). The plant communities in the study ecosystem are dominated by *Sphagnum spp.* and *Rhynchospora alba*. The average height of the vegetation during the growing season is about 0.3 m (Madrone Consultants Ltd., 1999). Plants are separated by shallow open water pools, some of them populated by algae developing. Birch trees are dispersed and appear to be growing on the remnants of baulks but none of them was taller 2 m. Sphagnum covers over 25% of the surface inside the study area (Hebda et al., 2000). The area of the open water ponds was estimated to be about 20% of the surface in summer by aerial photo.

3 Materials and methods

3.1 Climate measurements

Weather variables were continuously measured in order to determine climatic controls of CO_2 and CH_4 fluxes. Four components of radiation (shortwave/longwave, incoming and outgoing) were continuously measured by a four-component net radiometer (CNR1, Kipp and Zonen, Delft, Holland) on top of the tower. Two quantum sensors (LI-190, LI-COR Inc., Lincoln, NE, USA) measured incoming and outgoing photosynthetically active radiation (PAR). Precipitation was measured with an unheated tipping bucket rain gauge (TR-525M, Texas Electronics, Dallas, TX, USA) at 1 m height, 10 m north of the tower. Air temperature (T_a) and relative humidity (RH, HMP-35 A, Vaisala, Finland) were measured at the heights of 2.0 m and 0.3 m, and soil thermocouples (type T) were recording soil/water temperatures at the depths of 0.05, 0.10 and 0.50 m. A pressure transducer (CS400, CSI) was installed on July 28th 2015 in an observation well west of the tower to continuously measure WTH for the remainder of the study period. A soil volumetric water content (θ_w) sensor (CS616, CSI) was inserted vertically to measure integrated θ_w from the surface to a depth of 0.30 m.

3.2 Eddy-covariance measurements

Over the entire annual study period, from 16th June 2015 to 15th June 2016, a long-term eddy-covariance system (EC-1) was operated on a floating scaffold tower (Fig. 1) at a height of 1.8 m (facing south). The EC-1 system consisted of an ultrasonic anemometer-thermometer (CSAT-3, Campbell Scientific Inc. (CSI)) and an open-path CO_2/H_2O infrared gas analyzer (IRGA, LI-7500, LI-COR Inc.). The path separation between CSAT-3 and LI-7500 was 5 cm. The CSAT-3 measured three-dimensional wind (u, v, w, in m s^{-1}) and sonic temperature (T_s, in °C) at 60 Hz and output data at 10 Hz. The IRGA measured water vapor density (ρ_v) and CO_2 density (ρ_c) at 10 Hz. The 10-Hz data from both instruments were sampled on a data logger (CR1000, CSI) and processed fluxes of CO_2 (F_c) were calculated in post-processing of 30-min data blocks following the procedures documented in Crawford et al. (2013) .

An additional, independent EC system (EC-2) was added on June 10th 2015 to measure CH_4 fluxes. The EC-2 system was also located at a height of 1.8 m, 1.8 m to the west of EC-1, and faced south (Fig. 1). EC-2 consisted of a similar ultrasonic anemometer-thermometer (CSAT-3, CSI, 20 Hz), an enclosed-path H_2O/CO_2 IRGA (LI-7200, LI-COR Inc., 20 Hz) and an open-path gas analyzer to measure the partial density of CH_4 (ρ_m) (LI-7700, LI-COR Inc., 20 Hz). The northward-separation of LI-7200 was 20 cm. The northward-separation of LI-7700 was 40 cm and eastward-separation of LI-7700 was 20 cm. Data from EC-2 were collected by an analyzer interface unit (LI-7550, LI-COR Inc.) and processed on-site. Fluxes of CH_4 (F_m) were processed in advanced mode using EddyPro® (V6.1.0, LI-COR Inc.) with a missing sample allowance of 30%. F_m data were quality checked using the flagging system proposed by Mauder and Foken (2004).

3.3 Gap filling algorithms

Some gaps in climate and flux measurements are unavoidable due to challenging weather and low-light situations (the station was solar powered), and need to be filled in for estimating seasonal and annual fluxes. Gaps in the climate data (<1% of the year) were filled using measurements at nearby climate stations. Small gaps (<60 minutes) of missing CO2 and CH4 fluxes were filled by linear interpolation. Longer gaps were filled using empirical relationships between CO_2 or CH_4 fluxes and environmental variables. Two-year (from July 2014 to June 2016) of measurements of CO_2 fluxes were used for modelling R_e and GEP to achieve better statistical relationships. Since there were two EC systems running with redundant fluxes of CO_2, the sensitivity of different combinations of data (EC-1 vs. EC-2 or using an average of the two) have been explored in Lee et al. (2016). For the data presented in this study, CO_2 fluxes, H, LE from EC-1 and CH_4 fluxes EC-2 were used. Valid data from EC-1 was obtained for 59% of the year (after quality control). Valid data from EC-2, which was restricted by power availability, was 32% of the year (after quality control). In this study, net fluxes of CO_2 and CH_4 toward the ecosystem surface are negative and net fluxes from the ecosystem surface to the stmosphere are positive. Therefore, negative NEE and F_m represent net CO_2 and CH_4 uptake, respectively.

3.3.1 Gap filling of CO_2 flux data

For gaps longer than 2 hours in CO_2 fluxes, the CO_2 flux (e.g., net ecosystem exchange, NEE) was modelled as the difference between ecosystem respiration (R_e) and gross ecosystem productivity (GEP) i.e. NEE $= R_e -$ GEP. Nocturnal NEE values were R_e as there is no photosynthesis (GEP) at night.

160 R_e was modelled based on soil temperature at the 5-cm depth ($T_{s,5cm}$) using a logistic fit (Neter et al., 1988):

$$R_e = \frac{1}{r_1 r_2^{T_{s,5cm}} + r_3} \tag{1}$$

A comparable logistic function was proposed and used by FLUXNET Canada (Barr et al., 2002; Kljun et al., 2006). In this

165 study, we used this logistic model available in IDL (version 8.5.1, Exelis Visual Information Solutions, Boulder, Colorado). r_1, r_2, and r_3 are empirical parameters; r_1 controls the slope of exponential phase; r_2 decides where the transitional phase starts; and r_3 determines the height of plateau phase. The empirical parameters r_1, r_2, and r_3 were determined separately for each day of the year, using a moving window of 120 days (60 days into past and 60 days into future) based on all measured nighttime data from 2014 to 2016 when friction velocity was higher than 0.08 m s^{-1}. Lee (2016) determined the effect of

170 using different window sizes (60, 90, 120 and full year) on the annual modelled and gap-filled R_e and showed that a moving window size of 120 days was least sensitive to errors while still allowing for seasonal changes. However, sensitivity of choosing different window sizes on gap filled R_e was small, varying the annual value between 226 and 245 g C m^{-2} year^{-1}.

GEP was modelled using the photosynthetic light-response curves (Ö gren and Evans, 1993) based on photosynthetic photon flux density (PPFD in µmol m^{-2} s^{-1}):

175

$$GEP = \frac{MQY \cdot PPFD + P_M - ((MQY \cdot PPFD + P_M)^2 - 4 \cdot C_v \cdot MQY \cdot PPFD \cdot P_M)^{0.5}}{2 \cdot C_v} \tag{2}$$

Maximum photosynthetic rate at light saturation (P_M) and maximum quantum yield (MQY) are fitted parameters with GEP estimated as measured daytime NEE minus daytime R_e calculated using Eq. 1. Convexity (C_v) was fixed at 0.7 (Farquhar et

180 al., 1980). The time-varying parameters MQY and P_M were fitted separately for each day, using a moving window of 90 days using all data from 2014 to 2016 when friction velocity was higher than 0.08 m s^{-1}. The sensitivity of window size on gap filled GEP was small, resulting in annual value to vary between 385 and 415 g C m^{-2} year^{-1}.

3.3.2 Gap filling of CH_4 flux data

CH_4 fluxes with quality flags 0 and 1 according to Mauder and Foken (2004) were plotted against all related variables

185 including WTH, θ_W, T_a, and $T_{s,5cm}$. Since the main control was $T_{s,5cm}$, it was used to build a model to fill the gaps in CH_4 fluxes:

$$F_m = ae^{bT_{s,5cm}}$$ (3)

where F_m is the CH_4 flux, $T_{s,5cm}$ is the soil temperature at the 5 cm depth, and a and b are empirical parameters for the annual relationship.

3.4 Calculating CO₂e

The combined effect all long-lived greenhouse gases was compared for CO_2 and CH_4 by converting the molar fluxes of CO_2 and CH_4 into time-integrated radiative forcing (e.g. global warming potential, GWP) expressed on a mass basis in terms of CO_2 equivalents (g CO_2e m^{-2} s^{-1}) as follows:

$$CO_2\text{e (g)} = m_{CO_2}F_{CO_2} + GWP_{CH_4}m_{CH_4}F_{CH_4}$$ (4)

GWP_{CH_4} is the mass-based GWP for the CH_4 (g g^{-1}), m_{CO_2} is the molecular mass of CO_2 (44.01 g mol^{-1}), and m_{CH_4} is the molecular mass of CH_4 (16.04 g mol^{-1}). In this study, a 100-year GWP of CH_4 of 28, and 20-year GWP of CH_4 of 84, were used respectively (IPCC, 2014). N_2O fluxes have been neglected in this study because previous chamber-based measurements during the growing season found no significant emissions or uptake of N_2O in all study plots in the BBECA (Christen et al., 2016).

4 Results and Discussion

4.1 Weather

During the study period (June 16th 2015 to June 15th 2016), the site experienced an annual average T_a (2 m height) of 11.3 °C. Mean monthly T_a ranged between 4.4 (Jan 2016) and 19.3 °C (Jul 2015). The study site received a total annual precipitation of 1061.7 mm, of which 16% (173.4 mm) fell during the warm half year (Apr-Sep) and 84% (888.3 mm) during the cold half year (Oct-Mar). There was no lasting snow cover during the study year. However, the surface was frozen over ten days in January 2016, with an ice thickness of up to 5 cm.

Winds at this site were often influenced by a sea-land breeze circulation. Under sea-breeze situations, wind mainly came from the south (40% of all cases). Sometimes, however, the sea-land breeze blew from the west, primarily between 17:00 and 19:00 PST. The wind direction on average turned to east during the nighttime (land-breeze), and generally at night, the winds were weaker.

4.2 Surface conditions

4.2.1 Turbulent flux footprints

Cumulative turbulent source areas were calculated using the analytical turbulent source area (turbulent footprint) model (Kormann and Meixner, 2001) following the procedure outlined in Christen et al. (2011). The 80% contour line (enclosing 80% of the cumulative probability for a unit source) was entirely inside the field in spring and summer. It reached beyond the ditches at the north side in fall and winter. Unstable conditions during daytime allowed for a more constrained footprint surrounding the tower. Stable conditions at night led to larger footprints, primarily from East. The cumulative footprint for each of the four seasons for the EC-1 overlaid on the satellite image of the site are documented in Fig. S1 (supplementary material).

4.2.2 Vegetation cover and water table changes

Mosses and white beak sedge (the common name of *Rhynchospora alba*) started to grow in March and grasses grew up to a maximum of 0.3 m height in summer. In summer, vegetation covered almost the entire study area of the surface, including ponds (some with algae), so the surface was less patchy in summer compared to other seasons, when standing water ponds were intermixed with vegetation in fall, winter and spring (see supplementary material, Fig. S2).

Winter was the wettest season when WTH was mostly above the bare soil (reference surface). The highest water table position was 7.7 cm above the reference surface in December. In the dry season, the water table position dropped to 26.5 cm beneath the bog surface in August. The WTH decreased in spring, and dry hummocks could be seen from April to September. The water table started to rise above the surface after receiving the fall precipitation. The study site was flooded in winter during the study year.

4.3 CO$_2$ exchange

4.3.1 Annual, seasonal and monthly NEE, R_e and GEP

Overall, the study area was a CO$_2$ sink in spring (MAM, -1.10 g C m^{-2} day^{-1}) and in summer (JJA, -0.82 g C m^{-2} day^{-1}). Net CO$_2$ fluxes were near zero in fall (SON, +0.03 g C m^{-2} day^{-1}) and winter (DJF, -0.07 g C m^{-2} day^{-1}). Over the entire year, the annual CO$_2$-C budget (i.e., NEE) was -179 g C m^{-2} yr^{-1}. Almost in each month of the calendar year, the site was a weak sink for CO$_2$ except in October, November and December (Fig. 3, Table 1). Monthly net fluxes of CO$_2$ (NEE) ranged from +1.77 g C m^{-2} month^{-1} in November 2015 to -56.20 g C m^{-2} month^{-1} in May 2016.

The annual R_e and GEP during the study year were 236 and 415 g C m^{-2} yr^{-1}, respectively. The relative changes in R_e and GEP were closely linked to the seasonality of the plant phenology. Based on GEP trends, we can divide the study period into three segments, 'winter' (Oct-Mar), 'early growing season' (Apr-Jun), and 'late growing season (Jul-Sep). The rising temperature triggered growth in the early growing season (GEP = 59.73 g C m^{-2} month^{-1}), while the later growing season had limited growth (GEP = 25.08 g C m^{-2} month^{-1}). Winter had lowest productivity (GEP = 7.58 g C m^{-2} month^{-1}) (Table 1).

Despite a large seasonal amplitude in monthly GEP, R_e showed less variability over the year. The highest rate of increase in the magnitude of NEE and the highest magnitude of NEE both occurred early in growing season (Fig. 4). This was caused by the onset of R_e being delayed compared to GEP, resulting in the greatest imbalance between respiratory and assimilatory fluxes in May.

250   Table 2 compares annual NEE, R_e and GEP at the study site to Fluxnet sites over other land covers in the same region that experienced similar climate forcings, although from different years. An unmanaged grassland site 15 km to the west of the study area in the Fraser River Delta (Westham Island, Delta, BC, Crawford et al., 2013) had about 1.3 times higher NEE than this rewetted area. Annual R_e and GEP values at this grassland site were higher than the study site by a factor of 5.2 and 3.5. A mature 55-year-old Douglas-fir forest on Vancouver Island (200 km NW of the study area; Krishnan et al., 2009)

255   showed an NEE of 1.8 times higher than the study area. The R_e and GEP were even higher by factors of 7.8 and 5.2, respectively. A young forest plantation (Buckley Bay, 150 km W of the study area; Krishnan et al., 2009), which was a weak C source, had R_e and GEP of six- and three-fold higher than the study site, respectively. Compared to these other sites under similar climatic conditions, the rewetted area of the bog was not an ecosystem of high productivity but one with considerably limited R_e that permits more efficient CO_2 sequestration (-NEE is 43 % of GEP, as opposed to 15% for the unmanaged

260   grassland site and mature forest).

The annual NEE in this study was more negative than in the majority of previously reported NEE values for pristine temperate peatlands, which were weak sinks, typically in the range of -50 g C m^{-2} year^{-1} (Christensen et al., 2012; Humphreys et al., 2014; Matthias et al., 2014; McVeigh et al., 2014; Pelletier et al., 2015; Roulet et al., 2007). Values that are comparable to the current restored wetland were reported in five pristine temperate wetlands: −248 g C m^{-2} year^{-1}

265   (Lafleur et al., 2001), −234 g C m^{-2} year^{-1} (Campbell et al., 2014), -210 g C m^{-2} year^{-1} (Fortuniak et al., 2017), −189 g C m^{-2} year^{-1} (Flanagan and Syed, 2011), and −103 g C m^{-2} year^{-1} (Lund et al., 2010). The few datasets in the literature for NEE of restored wetlands showed a wide range of values. Some were CO_2 sources, with NEE ranging from +103 g C m^{-2} year^{-1} to +142 g C m^{-2} year^{-1} (Järveoja et al., 2016; Richards and Craft, 2015; Strack and Zuback, 2013). Other measurements in restored wetlands, however, were sinks, all of them stronger than in this study, with NEE values ranging from -804 g C m^{-2}

270   year^{-1} to -270 g C m^{-2} year^{-1} (Anderson et al., 2016; Badiou et al., 2011; Hendriks et al., 2007; Herbst et al., 2013; Knox et al., 2015). In this study, values of R_e and GEP were lower than those found for a restored wetland at a comparable latitude in the central Netherlands with slightly lower annual temperature and precipitation (Hendriks et al., 2007). R_e and GEP in this study area were also lower than values for most pristine peatlands at comparable latitudes (Helfter et al., 2015; Levy and Gray, 2015). Comparably low R_e and GEP were reported from the 'Mer Bleue' boreal raised bog (Lafleur et al., 2001; Moore,

275   2002) and from an Atlantic blanket bog (McVeigh et al., 2014; Sottocornola and Kiely, 2010), both of which experienced a lower mean annual temperature.

It is important to estimate dissolved organic carbon (DOC) to determine a more complete ecosystem C budget. DOC lost from restored and pristine peatlands have been found typically to range from 3.4 to 16.1 g C m^{-2} year^{-1} (Hendriks et al., 2007; Koehler et al., 2011; Roulet et al., 2007; Waddington et al., 2010), although, Chu et al. (2014) reported a net DOC import for

280   a marsh of 23 ± 13 g C m^{-2} year^{-1}. D'Acunha et al. (2016) estimated DOC export for the current study area for Jan – Dec 2016 to be 22.4 g C m^{-2} year^{-1} (15% of annual NEE).

4.3.2 Diurnal variability in CO$_2$ fluxes

The seasonally-changing diurnal course of gap-filled NEE with isopleths over time of day and year is shown in Fig. 4. The daily maximum in GEP changed with season resulting in the high magnitude of NEE during midday between May and July

285   (~ -3.5 μmol m^{-2} s^{-1}) with the highest magnitude of NEE occurring in May. Nighttime NEE, i.e., R_e, showed relatively small variation with season, and on average was ≤ 1 μmol m^{-2} s^{-1} for most of the study period. The rapid decrease in monthly R_e from May to June was caused by low R_e in early morning or at nightfall in June.

4.3.3 Ecosystem respiration

Figure 5 shows the relationship between nighttime R_e and $T_{s,5cm}$ using the data for the entire study period. R_e increased with

290   increasing $T_{s,5cm}$ as expected, and annually followed a logistic curve rather than an exponential relationship. R_e response curves were also calculated every two months (see supplementary material, Fig. S3). R_e showed different curves depending on season. In winter, R_e varied little with $T_{s,5cm}$ and was close to zero. From February to May, the relationship became closer to logistic. In June and July, due to general warm condition (>15°C), R_e remained nearly constant at ~1 μmol m^{-2} s^{-1} (the fitted curve stayed in the plateau phase). The study area had the highest R_e in these two months. In fall, R_e curves were closer

295   to an exponential relationship, which could be due in part to leaf senescence (Shurpali et al., 2008). Decomposition of dead plant organic matter on the soil surface may have caused a higher R_e in fall compared to spring and winter at the same $T_{s,5cm}$. Another factor could be the WTH, which in fall was not high enough to suppress R_e as it did in winter (Juszczak et al., 2013). The differences between March and September at the same $T_{s,5cm}$ were up to 0.4 μmol m^{-2} s^{-1}.

Two other controls on R_e explored were air temperature (T_a) and WTH. The role of WTH was described above and T_a

300   had a similar impact on R_e as $T_{s,5cm}$ when $T_a < 16$°C, but for warmer temperatures, T_a did not correlate with R_e. The explanation for this is that heterotrophic component of R_e depends on T_s, not the rapidly changing T_a (Davidson et al., 2002; Edwards, 1975; Lloyd and Taylor, 1994).

It is widely reported that in most terrestrial ecosystems, the activity of soil microbes is also governed by soil moisture status, having little activity when the soil is excessively dry or excessively wet. Accordingly, and like other wetlands, R_e was

305   small when the water table was above the surface because this situation suppressed aerobic decomposition of peat (Rochefort et al., 2002; Weltzin et al., 2000). When the water table was below surface, R_e increased to near 1 μmol m^{-2} s^{-1} and became stable no matter how low the water table position was. This relationship was also found in many other peatlands (Bridgham et al., 2006; Ellis et al., 2009; Strack et al., 2006). There was no obvious relationship between θ_w (integrated from 0-30 cm depth) and R_e. R_e slightly decreased from 1.0 to 0.6 μmol m^{-2} s^{-1} when θ_w increased from 84% to 88%. Other than this range,

310   θ_w had no more impact on R_e.

4.3.4 Gross ecosystem productivity

Figure 6 shows the average light response curve, with half-hourly GEP as a function of PPFD. Due to different phenology over the year and the changes in solar altitude, light response curves were also calculated every two months (see supplementary material, Fig. S4). GEP reached a maximum in May with 92.63 g C m^{-2} month^{-1}, and a minimum of 2.79 g C m^{-2} month^{-1} in December (Fig. 3, Table 1). GEP at light saturation reached roughly 5.09 μmol m^{-2} s^{-1} in summer, and remained below 2.49 μmol m^{-2} s^{-1} in winter, due to reduced leaf area, flooding, and lower temperatures. From March to May, GEP increased much more rapidly than R_e. In fall, GEP decreased faster than R_e. The magnitude of R_e already was close to GEP in the late August to make the study area become CO_2 neutral in late summer.

Other possible controls on GEP explored were WTH and T_a. We found that WTH was not a control on GEP in the current study as the study area remained fairly wet throughout the year. Furthermore, the effects of T_a on GEP were approximately limited between 10 and 15 °C.

4.4 CH$_4$ exchange

4.4.1 Annual and seasonal CH$_4$ budgets

Overall, the study area was a source of CH_4 in each of the twelve months (Table 1). The annual CH_4-C budget was 16 g CH_4-C m^{-2} yr^{-1}. CH_4 emissions were close to zero in winter (8.7 mg CH_4-C m^{-2} day^{-1}). Seasonally, it was a weaker CH_4 source in fall (21.5 mg CH_4-C m^{-2} day^{-1}) and spring (29.4 mg CH_4-C m^{-2} day^{-1}), and then became a significant source in summer (120.9 mg CH_4-C m^{-2} day^{-1}). Monthly emissions of CH_4 ranged from 66 (November) to 4436 (July) mg CH_4-C m^{-2} month^{-1}. CH_4 fluxes showed a seasonal pattern, which was linked to phenology and temperature. The rising T_a did not trigger CH_4 production immediately, and CH_4 fluxes remained low in April and May. But once the subsurface and water became warm enough, CH_4 emissions increased from to 1.5 to 3.0 g CH_4-C m^{-2} month^{-1} in June (Table 1). CH_4 emissions reached the peak in July (4.4 g CH_4-C m^{-2} month^{-1}) and held similar magnitude (3.7 g CH_4-C m^{-2} month^{-1}) in August even though the T_a had dropped.

The annual CH_4 flux in this study area was lower than CH_4 fluxes reported for other restored wetlands (Anderson et al., 2016; Hendriks et al., 2007; Knox et al., 2015; Mitsch et al., 2010). Despite the study area being flooded for most of the study year, CH_4 emissions were closer to fluxes measured over drained peatlands (Kroon et al., 2010; Schrier-Uijl et al., 2010). Only Herbst et al. (2013) reported an annual CH_4 flux from a restored wetland in Denmark that was lower than in this study (9 to 13 g CH_4-C m^{-2} year^{-1}). Our annual CH_4 flux at 16 g CH_4-C m^{-2} year^{-1} was comparable to an average natural temperate wetland CH_4 flux, which is typically around 15 g CH_4-C m^{-2} year^{-1} (Abdalla et al., 2016; Fortuniak et al., 2017; Nicolini et al., 2013; Turetsky et al., 2014). The CH_4 fluxes from a number of temperate and tropical pristine wetlands exceeded the CH_4 fluxes reported in this study, including emissions from marshes in the Southwestern US (130 g CH_4-C m^{-2} year^{-1}, Whiting & Chanton, 2001), tropical wetlands in Costa Rica (82 g CH_4-C m^{-2} year^{-1}, Nahlik & Mitsch, 2010), and

marshes in the Midwestern US (50 g CH_4-C m^{-2} year^{-1}, Koh et al., 2009). However, all these studies were conducted using chambers and the sampling frequency was at most once per month.

4.4.2 Diurnal variability in CH_4 fluxes

345 The ensemble diurnal courses of the CH_4 fluxes measured by the EC-2 system are shown in Fig. 7 from 16th June 2015 to 15th June 2016. Surprisingly, there was only small diurnal variation observed for CH_4 fluxes in the summer months, as has been found in other studies (Juutinen et al., 2004; Long et al., 2010; Sun et al., 2013; Wang and Han, 2005). In the current study area, with changes in WTH and vegetation growth occurring during the year, there were likely several processes affecting CH_4 transport, which masked the diurnal pattern of CH_4 fluxes. Furthermore, $T_{s,5cm}$ appeared to be the main

350 environmental control on CH_4 fluxes in this study but did not have as strong effect on CH_4 emissions as found in previous studies. Thus CH_4 was continuously emitted at a similar rate during daytime and nighttime. From January to March and October to December, the winter half-year, the study site had constant CH_4 emissions of less than 50 nmol m^{-2} s^{-1}, and almost no diurnal variation was observed. July had the greatest CH_4 emissions, and the highest magnitude (>150 nmol m^{-2} s^{-1}) appeared in the evening (3 pm to 9 pm). This corresponded to the lagged effect of soil temperature and may be partly due to

355 convective turbulent mixing caused by cooling during the evening (Godwin et al., 2013).

4.5 CO_2e exchange

Figure 8a and 8b show CO_2 and CH_4 fluxes expressed in terms of CO_2e using 100-year and 20-year GWPs, respectively. Considering fluxes of both GHGs together, this rewetted area was annually near to CO_2e neutral at 100-year scale with a net uptake by CO_2 (-656 g CO_2e m^{-2} year^{-1}) nearly the same as CH_4 emissions (601 g CO_2e m^{-2} year^{-1}). On shorter time horizon

360 of 20 years, the study area represented a significant net climatic forcing in CO_2e terms as the net uptake of CO_2 (-656 g CO_2e m^{-2} year^{-1}) was one-third that of CH_4 emissions (1803 g CO_2e m^{-2} year^{-1}). In late spring and early summer, the early onset of CO_2 sequestration in May and the time lag in CH_4 fluxes combined to represent a negative net GHG forcing, no matter which GWP time horizon was considered. The quick drop in CO_2 sequestration in August and September allowed the highest net GHG forcing to be observed at both time horizons in late summer. In short, the critical time period for both, CO_2

365 and CH_4 fluxes in terms of CO_2e, was the growing season when magnitude of fluxes changed differently across the growing season. The results show that measurements made during a part of the growing season are not necessarily representative for the entire growing season or the year.

Using GWP to classify a study area as a net GHG source or sink is useful; however, the appropriateness of this method in computing the actual radiative forcing has been questioned (e.g. sustained step-change in CO_2 and CH_4 fluxes can not be

370 evaluated) and alternative models were proposed (Frolking and Roulet, 2007; Fuglestvedt et al., 2000; Neubauer and Megonigal, 2015; Petrescu et al., 2015; Smith and Wigley, 2000).

5 Conclusions

The study area, a rewetted plot in the BBECA undergoing ecological restoration, was a net CO_2 sink over the study period (-179 g CO_2-C m^{-2} $year^{-1}$). The study area was not a highly productive ecosystem (annual GEP = 415 g CO_2-C m^{-2} $year^{-1}$) but exhibited low R_e (annual R_e = 236 g CO_2-C m^{-2} $year^{-1}$), likely due to oxygen limitations. The annual CO_2 fluxes reported here from a restored and rewetted peatland are comparable with data reported from pristine temperate peatlands in temperate mid latitudes (Alm et al., 1997; Lafleur et al., 2001; Pihlatie et al., 2010; Shurpali et al., 1995). The study area sequestered less CO_2 than the few other restored wetlands reported in the literature (Anderson et al., 2016; Järveoja et al., 2016; Knox et al., 2015; Richards and Craft, 2015; Strack and Zuback, 2013). The major controls on CO_2 fluxes were PAR irradiance and $T_{s,5cm}$. The magnitude of PAR strongly controlled GEP, and the $T_{s,5cm}$ regulated R_e. WTH also had influence on R_e especially when the ecosystem was flooded.

Annual CH_4 emissions were 16 g CH_4-C m^{-2} $year^{-1}$, which is lower than those reported for other restored wetlands (Anderson et al., 2016; Knox et al., 2015). CH_4 emissions in summer months were 60 times stronger than in winter. The ditch blocking permitted anaerobic conditions with the water table within 30 cm of the surface throughout the year. Effects of changing WTH on CH_4 fluxes at the study area were not clearly apparent. $T_{s,5cm}$ explained CH_4 fluxes best (R^2 = 0.66) – although both $T_{s,5cm}$ and WTH changed seasonally.

In terms of the C balance, our results suggest that our study area in BBECA was a net C sink (-163 g C m^{-2} $year^{-1}$) during the 8th year following rewetting. These results are consistent with those of several disturbed peatlands that have become a net annual C sink after following restoration by rewetting (Karki et al., 2016; Schrier-Uijl et al., 2014; Wilson et al., 2013). In terms of net climate forcing of the system related to CO_2 and CH_4 fluxes expressed by GWPs, our results show that the ecosystem was almost CO_2e neutral (-55 g CO_2e m^{-2} $year^{-1}$) over a 100-year time horizon during the study period after a 7-year restoration. However, the rewetted area was a substantial net CO_2e source (1147 g CO_2e m^{-2} $year^{-1}$) on a 20-year time horizon due to the stronger GWP of CH_4 on shorter timescales.

Acknowledgements

This research was primarily funded through research contracts between Metro Vancouver and UBC (PI: Christen). Selected equipment was supported by the Canada Foundation for Innovation (Christen, Johnson) and NSERC RTI (Christen). Financial support through scholarships and training were provided by UBC Faculty of Graduate and Postdoctoral Studies and UBC Geography. We appreciate the substantial technical and logistical support by Joe Soluri (Metro Vancouver) in operating the site, and scientific contributions and data provided by C. Reynolds (Metro Vancouver) and S. Howie (Delta, BC).

References

Abdalla, M., Hastings, A., Truu, J., Espenberg, M., Mander, Ü., and Smith, P.: Emissions of methane from northern peatlands: a review of management impacts and implications for future management options, Ecology and Evolution, 6, 7080-7102, 2016.

Alm, J., Talanov, A., Saarnio, S., Silvola, J., Ikkonen, E., Aaltonen, H., Nykänen, H., and Martikainen, P. J.: Reconstruction of the carbon balance for microsites in a boreal oligotrophic pine fen, Finland, Oecologia, 110, 423-431, 1997.

Anderson, F. E., Bergamaschi, B., Sturtevant, C., Knox, S., Hastings, L., Windham-Myers, L., Detto, M., Hestir, E. L., Drexler, J., Miller, R. L., Matthes, J. H., Verfaillie, J., Baldocchi, D., Snyder, R. L., and Fujii, R.: Variation of energy and carbon fluxes from a restored temperate freshwater wetland and implications for carbon market verification protocols, Journal of Geophysical Research: Biogeosciences, 121, 777-795, 2016.

Badiou, P., McDougal, R., Pennock, D., and Clark, B.: Greenhouse gas emissions and carbon sequestration potential in restored wetlands of the Canadian prairie pothole region, Wetlands Ecol Manage, 19, 237-256, 2011.

Barr, A. G., Griffis, T. J., Black, T. A., Lee, X., Staebler, R. M., Fuentes, J. D., Chen, Z., and Morgenstern, K.: Comparing the carbon budgets of boreal and temperate deciduous forest stands, Canadian Journal of Forest Research, 32, 813-822, 2002.

Bergamaschi, P., Frankenberg, C., Meirink, J. F., Krol, M., Dentener, F., Wagner, T., Platt, U., Kaplan, J. O., Körner, S., Heimann, M., Dlugokencky, E. J., and Goede, A.: Satellite chartography of atmospheric methane from SCIAMACHY on board ENVISAT: 2. Evaluation based on inverse model simulations, Journal of Geophysical Research: Atmospheres, 112, n/a-n/a, 2007.

Bloom, A. A., Palmer, P. I., Fraser, A., Reay, D. S., and Frankenberg, C.: Large-Scale Controls of Methanogenesis Inferred from Methane and Gravity Spaceborne Data, Science, 327, 322-325, 2010.

Bortolotti, L. E., St. Louis, V. L., Vinebrooke, R. D., and Wolfe, A. P.: Net Ecosystem Production and Carbon Greenhouse Gas Fluxes in Three Prairie Wetlands, Ecosystems, 19, 411-425, 2016.

Bridgham, S., Megonigal, J. P., Keller, J., Bliss, N., and Trettin, C.: The carbon balance of North American wetlands, Wetlands, 26, 889-916, 2006.

Campbell, D. I., Smith, J., Goodrich, J. P., Wall, A. M., and Schipper, L. A.: Year-round growing conditions explains large CO2 sink strength in a New Zealand raised peat bog, Agricultural and Forest Meteorology, 192–193, 59-68, 2014.

Chestnutt, C.: For peat's sake: A water balance study and comparison of the eddy covariance technique and semi-empirical calculation to determine summer evapotranspiration in Burns Bog, British Columbia., BSc, The University of Edinburgh, The University of British Columbia, 2015.

Christen, A., Coops, N. C., Crawford, B. R., Kellett, R., Liss, K. N., Olchovski, I., Tooke, T. R., van der Laan, M., and Voogt, J. A.: Validation of modeled carbon-dioxide emissions from an urban neighborhood with direct eddy-covariance measurements, Atmospheric Environment, 45, 6057-6069, 2011.

Christen, A., Jassal, R. S., Black, T. A., Grant, N. J., Hawthorne, I., Johnson, M. S., Lee, S. C., and M., M.: Summertime greenhouse gas fluxes from an urban bog undergoing restoration through rewetting., Mires and Peat, 18, 1-24, 2016.

Christensen, T. R., Jackowicz-Korczyński, M., Aurela, M., Crill, P., Heliasz, M., Mastepanov, M., and Friborg, T.: Monitoring the Multi-Year Carbon Balance of a Subarctic Palsa Mire with Micrometeorological Techniques, AMBIO, 41, 207-217, 2012.

Chu, H., Chen, J., Gottgens, J. F., Ouyang, Z., John, R., Czajkowski, K., and Becker, R.: Net ecosystem methane and carbon dioxide exchanges in a Lake Erie coastal marsh and a nearby cropland, Journal of Geophysical Research: Biogeosciences, 119, 722-740, 2014.

Ciais, P., Sabine, C., Bala, G., Bopp, L., Brovkin, V., Canadell, J., Chhabra, A., DeFries, R., Galloway, J., Heimann, M., Jones, C., Le Quéré, C., Myneni, R. B., Piao, S., and Thornton, P.: Carbon and Other Biogeochemical Cycles. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Stocker, T. F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S. K., Boschung, J., Nauels, A., Xia, Y., Bex, V., and Midgley, P. M. (Eds.), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 2013.

Cowen, G. J.: Social and environmental interaction in urban wetlands, Burns Bog Conservation Society., 2015.

Crawford, B., Christen, A., and Ketler, R.: Processing and quality control procedures of turbulent flux measurements during the Vancouver EPiCC experiment, The University of British Columbia, 2013. http://hdl.handle.net/2429/45079.

Crill, P., Bartlett, K., and Roulet, N.: Methane flux from boreal peatlands, International workshop on carbon cycling in boreal peatlands and climatic change, Hyytiaelae, Finland, 10, 1992.

D'Acunha, B., Johnson, M. S., Lee, S.-C., and Christen, A.: Carbon fluxes in dissolved and gaseous forms for a restored peatland in British Columbia, Canada: Net ecosystem carbon balance (NECB) determined using eddy covariance for CO2 and CH4 and dissolved C fluxes, 2016 AGU Fall meeting, San Francisco, 2016.

Davidson, E. A., Savage, K., Verchot, L. V., and Navarro, R.: Minimizing artifacts and biases in chamber-based measurements of soil respiration, Agricultural and Forest Meteorology, 113, 21-37, 2002.

Dise, N. B., Gorham, E., and Verry, E. S.: Environmental factors controlling methane emissions from peatlands in northern Minnesota, Journal of Geophysical Research: Atmospheres, 98, 10583-10594, 1993.

Edwards, N. T.: Effects of Temperature and Moisture on Carbon Dioxide Evolution in a Mixed Deciduous Forest Floor1, Soil Science Society of America Journal, 39, 361-365, 1975.

Ellis, T., Hill, P. W., Fenner, N., Williams, G. G., Godbold, D., and Freeman, C.: The interactive effects of elevated carbon dioxide and water table draw-down on carbon cycling in a Welsh ombrotrophic bog, Ecological Engineering, 35, 978-986, 2009.

Farquhar, G. D., von Caemmerer, S., and Berry, J. A.: A biochemical model of photosynthetic CO2 assimilation in leaves of C3 species, Planta, 149, 78-90, 1980.

Finér, L. and Laine, J.: Root dynamics at drained peatland sites of different fertility in southern Finland, Plant and Soil, 201, 27-36, 1998.

Flanagan, L. B. and Syed, K. H.: Stimulation of both photosynthesis and respiration in response to warmer and drier conditions in a boreal peatland ecosystem, Global Change Biology, 17, 2271-2287, 2011.

Foken, T., Gockede, M., Mauder, M., Mahrt, L., Amiro, B. D., and Munger, J. W.: Post-field data quality control. In: Handbook of Micrometeorology: A Guide for Surface Flux Measurements, Lee, X. (Ed.), Kluwer Academic Publishers, Dordrecht, 2004.

Fortuniak, K., Pawlak, W., Bednorz, L., Grygoruk, M., Siedlecki, M., and Zieliński, M.: Methane and carbon dioxide fluxes of a temperate mire in Central Europe, Agricultural and Forest Meteorology, 232, 306-318, 2017.

Frolking, S., Roulet, N., and Lawrence, D.: Issues Related to Incorporating Northern Peatlands into Global Climate Models. In: Carbon Cycling in Northern Peatlands, American Geophysical Union, 2013.

Frolking, S. and Roulet, N. T.: Holocene radiative forcing impact of northern peatland carbon accumulation and methane emissions, Global Change Biology, 13, 1079-1088, 2007.

Fuglestvedt, J. S., Berntsen, T. K., Godal, O., and Skodvin, T.: Climate implications of GWP-based reductions in greenhouse gas emissions, Geophysical Research Letters, 27, 409-412, 2000.

Gaveau, D. L. A., Salim, M. A., Hergoualc'h, K., Locatelli, B., Sloan, S., Wooster, M., Marlier, M. E., Molidena, E., Yaen, H., DeFries, R., Verchot, L., Murdiyarso, D., Nasi, R., Holmgren, P., and Sheil, D.: Major atmospheric emissions from peat fires in Southeast Asia during non-drought years: evidence from the 2013 Sumatran fires, Scientific Reports, 4, 6112, 2014.

Godwin, C. M., McNamara, P. J., and Markfort, C. D.: Evening methane emission pulses from a boreal wetland correspond to convective mixing in hollows, Journal of Geophysical Research: Biogeosciences, 118, 994-1005, 2013.

Gorham, E.: Northern Peatlands: Role in the Carbon Cycle and Probable Responses to Climatic Warming, Ecological Applications, 1, 182-195, 1991.

Grasset, C., Abril, G., Guillard, L., Delolme, C., and Bornette, G.: Carbon emission along a eutrophication gradient in temperate riverine wetlands: effect of primary productivity and plant community composition, Freshwater Biology, 61, 1405-1420, 2016.

Heathwaite, A. L. and Göttlich, K.: Mires: process, exploitation, and conservation, Wiley, 1993.

Hebda, R. J., Gustavson, K., Golinski, K., and Calder, A. M.: Burns Bog Ecosystem Review Synthesis for Burns Bog, Fraser River Delta, South-western British Columbia, Canada, Environmental Assessment Office, Victoria, B.C., 2000.

Helfter, C., Campbell, C., Dinsmore, K. J., Drewer, J., Coyle, M., Anderson, M., Skiba, U., Nemitz, E., Billett, M. F., and Sutton, M. A.: Drivers of long-term variability in CO_2 net ecosystem exchange in a temperate peatland, Biogeosciences, 12, 1799-1811, 2015.

Hendriks, D. M. D., van Huissteden, J., Dolman, A. J., and van der Molen, M. K.: The full greenhouse gas balance of an abandoned peat meadow, Biogeosciences, 4, 411-424, 2007.

500 Herbst, M., Friborg, T., Schelde, K., Jensen, R., Ringgaard, R., Vasquez, V., Thomsen, A. G., and Soegaard, H.: Climate and site management as driving factors for the atmospheric greenhouse gas exchange of a restored wetland, Biogeosciences, 10, 39-52, 2013.

Howie, S. A., Whitfield, P. H., Hebda, R. J., Munson, T. G., Dakin, R. A., and Jeglum, J. K.: Water Table and Vegetation Response to Ditch Blocking: Restoration of a Raised Bog in Southwestern British Columbia, Canadian Water Resources

505 Journal / Revue canadienne des ressources hydriques, 34, 381-392, 2009.

Humphreys, E. R., Charron, C., Brown, M., and Jones, R.: Two Bogs in the Canadian Hudson Bay Lowlands and a Temperate Bog Reveal Similar Annual Net Ecosystem Exchange of CO_2, Arctic, Antarctic, and Alpine Research, 46, 103-113, 2014.

IPCC: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of

510 Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Field, C.B., V.R. Barros, D.J. Dokken, K.J. Mach, M.D. Mastrandrea, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 2014.

Järveoja, J., Peichl, M., Maddison, M., Soosaar, K., Vellak, K., Karofeld, E., Teemusk, A., and Mander, Ü.: Impact of water

515 table level on annual carbon and greenhouse gas balances of a restored peat extraction area, Biogeosciences, 13, 2637-2651, 2016.

Juszczak, R., Humphreys, E., Acosta, M., Michalak-Galczewska, M., Kayzer, D., and Olejnik, J.: Ecosystem respiration in a heterogeneous temperate peatland and its sensitivity to peat temperature and water table depth, Plant and Soil, 366, 505-520, 2013.

520 Juutinen, S., Alm, J., Larmola, T., Saarnio, S., Martikainen, P. J., and Silvola, J.: Stand-specific diurnal dynamics of CH_4 fluxes in boreal lakes: Patterns and controls, Journal of Geophysical Research: Atmospheres, 109, n/a-n/a, 2004.

Karki, S., Elsgaard, L., Kandel, T. P., and Lærke, P. E.: Carbon balance of rewetted and drained peat soils used for biomass production: a mesocosm study, GCB Bioenergy, 8, 969-980, 2016.

Kleinen, T., Brovkin, V., von Bloh, W., Archer, D., and Munhoven, G.: Holocene carbon cycle dynamics, Geophysical

525 Research Letters, 37, L02705, 2010.

Kljun, N., Black, T. A., Griffis, T. J., Barr, A. G., Gaumont-Guay, D., Morgenstern, K., McCaughey, J. H., and Nesic, Z.: Response of Net Ecosystem Productivity of Three Boreal Forest Stands to Drought, Ecosystems, 9, 1128-1144, 2006.

Knox, S. H., Sturtevant, C., Matthes, J. H., Koteen, L., Verfaillie, J., and Baldocchi, D.: Agricultural peatland restoration: effects of land-use change on greenhouse gas (CO_2 and CH_4) fluxes in the Sacramento-San Joaquin Delta, Global Change

530 Biology, 21, 750-765, 2015.

Koehler, A.-K., Sottocornola, M., and Kiely, G.: How strong is the current carbon sequestration of an Atlantic blanket bog?, Global Change Biology, 17, 309-319, 2011.

Koh, H.-S., Ochs, C. A., and Yu, K.: Hydrologic gradient and vegetation controls on CH4 and CO2 fluxes in a spring-fed forested wetland, Hydrobiologia, 630, 271-286, 2009.

535 Komulainen, V.-M., Tuittila, E.-S., Vasander, H., and Laine, J.: Restoration of Drained Peatlands in Southern Finland: Initial Effects on Vegetation Change and CO2 Balance, Journal of Applied Ecology, 36, 634-648, 1999.

Kormann, R. and Meixner, F. X.: An analytical footprint model for non-neutral stratification, Boundary-Layer Meteorology, 99, 207-224, 2001.

Krishnan, P., Black, T. A., Jassal, R. S., Chen, B., and Nesic, Z.: Interannual variability of the carbon balance of three

540 different-aged Douglas-fir stands in the Pacific Northwest, Journal of Geophysical Research: Biogeosciences, 114, n/a-n/a, 2009.

Kroon, P. S., Schrier-Uijl, A. P., Hensen, A., Veenendaal, E. M., and Jonker, H. J. J.: Annual balances of CH4 and N2O from a managed fen meadow using eddy covariance flux measurements, European Journal of Soil Science, 61, 773-784, 2010.

545 Lafleur, P. M., Hember, R. A., Admiral, S. W., and Roulet, N. T.: Annual and seasonal variability in evapotranspiration and water table at a shrub-covered bog in southern Ontario, Canada, Hydrological Processes, 19, 3533-3550, 2005.

Lafleur, P. M., Roulet, N. T., and Admiral, S. W.: Annual cycle of CO2 exchange at a bog peatland, Journal of Geophysical Research: Atmospheres, 106, 3071-3081, 2001.

Lal, R.: Carbon sequestration, Philosophical Transactions of the Royal Society B: Biological Sciences, 363, 815-830, 2008.

550 Langeveld, C. A., Segers, R., Dirks, B. O. M., van den Pol-van Dasselaar, A., Velthof, G. L., and Hensen, A.: Emissions of CO2, CH4 and N2O from pasture on drained peat soils in the Netherlands. In: Developments in Crop Science, Ittersum, M. K. v. and Geijn, S. C. v. d. (Eds.), Elsevier, 1997.

Lee, S.-C.: Annual greenhouse gas budget for a bog ecosystem undergoing restoration by rewetting, MSc, Geography, UBC, Vancovuer, 2016.

555 Lehner, B. and Döll, P.: Development and validation of a global database of lakes, reservoirs and wetlands, Journal of Hydrology, 296, 1-22, 2004.

Levy, P. E. and Gray, A.: Greenhouse gas balance of a semi-natural peatbog in northern Scotland, Environmental Research Letters, 10, 094019, 2015.

Limpens, J., Berendse, F., Blodau, C., Canadell, J. G., Freeman, C., Holden, J., Roulet, N., Rydin, H., and Schaepman-Strub,

560 G.: Peatlands and the carbon cycle: from local processes to global implications – a synthesis, Biogeosciences, 5, 1475-1491, 2008.

Lloyd, J. and Taylor, J. A.: On the Temperature Dependence of Soil Respiration, Functional Ecology, 8, 315-323, 1994.

Long, K. D., Flanagan, L. B., and Cai, T.: Diurnal and seasonal variation in methane emissions in a northern Canadian peatland measured by eddy covariance, Global Change Biology, 16, 2420-2435, 2010.

565 Lu, W., Xiao, J., Liu, F., Zhang, Y., Liu, C. a., and Lin, G.: Contrasting ecosystem CO2 fluxes of inland and coastal wetlands: a meta-analysis of eddy covariance data, Global Change Biology, doi: 10.1111/gcb.13424, 2016. n/a-n/a, 2016.

Lund, M., Lafleur, P. M., Roulet, N. T., Lindroth, A., Christensen, T. R., Aurela, M., Chojnicki, B. H., Flanagan, L. B., Humphreys, E. R., Laurila, T., Oechel, W. C., Olejnik, J., Rinne, J., Schubert, P. E. R., and Nilsson, M. B.: Variability in exchange of CO2 across 12 northern peatland and tundra sites, Global Change Biology, 16, 2436-2448, 2010.

570   Madrone Consultants Ltd.: Burns Bog Ecosystem Review. Plants and Plant Communities. 1999.

Matthias, P., Mats, Ö., Mikaell Ottosson, L., Ulrik, I., Jörgen, S., Achim, G., Anders, L., and Mats, B. N.: A 12-year record reveals pre-growing season temperature and water table level threshold effects on the net carbon dioxide exchange in a boreal fen, Environmental Research Letters, 9, 055006, 2014.

McVeigh, P., Sottocornola, M., Foley, N., Leahy, P., and Kiely, G.: Meteorological and functional response partitioning to

575   explain interannual variability of CO2 exchange at an Irish Atlantic blanket bog, Agricultural and Forest Meteorology, 194, 8-19, 2014.

Menviel, L. and Joos, F.: Toward explaining the Holocene carbon dioxide and carbon isotope records: Results from transient ocean carbon cycle-climate simulations, Paleoceanography, 27, PA1207, 2012.

MetroVancouver: Burns Bog Ecological Conservancy Area Management Plan. 2007.

580   Minkkinen, K. and Laine, J.: Long-term effect of forest drainage on the peat carbon stores of pine mires in Finland, Canadian Journal of Forest Research, 28, 1267-1275, 1998.

Mitsch, W., Nahlik, A., Wolski, P., Bernal, B., Zhang, L., and Ramberg, L.: Tropical wetlands: seasonal hydrologic pulsing, carbon sequestration, and methane emissions, Wetlands Ecol Manage, 18, 573-586, 2010.

Mitsch, W. J., Bernal, B., Nahlik, A. M., Mander, Ü., Zhang, L., Anderson, C. J., Jørgensen, S. E., and Brix, H.: Wetlands,

585   carbon, and climate change, Landscape Ecol, 28, 583-597, 2013.

Moore, P. D.: The future of cool temperate bogs, Environmental Conservation, 29, 3-20, 2002.

Neter, J., Wasserman, W., and Whitmore, G. A.: Applied Statistics, Allyn & Bacon, Newton, Massachusetts, 1988.

Neubauer, S. C. and Megonigal, J. P.: Moving Beyond Global Warming Potentials to Quantify the Climatic Role of Ecosystems, Ecosystems, 18, 1000-1013, 2015.

590   Nicolini, G., Castaldi, S., Fratini, G., and Valentini, R.: A literature overview of micrometeorological CH4 and N2O flux measurements in terrestrial ecosystems, Atmospheric Environment, 81, 311-319, 2013.

Ögren, E. and Evans, J. R.: Photosynthetic light-response curves, Planta, 189, 182-190, 1993.

Page, S. E., Siegert, F., Rieley, J. O., Boehm, H.-D. V., Jaya, A., and Limin, S.: The amount of carbon released from peat and forest fires in Indonesia during 1997, Nature, 420, 61-65, 2002.

595   Pelletier, L., Strachan, I. B., Roulet, N. T., Garneau, M., and Wischnewski, K.: Effect of open water pools on ecosystem scale surface-atmosphere carbon dioxide exchange in a boreal peatland, Biogeochemistry, 124, 291-304, 2015.

Petrescu, A. M. R., Lohila, A., Tuovinen, J.-P., Baldocchi, D. D., Desai, A. R., Roulet, N. T., Vesala, T., Dolman, A. J., Oechel, W. C., Marcolla, B., Friborg, T., Rinne, J., Matthes, J. H., Merbold, L., Meijide, A., Kiely, G., Sottocornola, M., Sachs, T., Zona, D., Varlagin, A., Lai, D. Y. F., Veenendaal, E., Parmentier, F.-J. W., Skiba, U., Lund, M., Hensen, A., van

600   Huissteden, J., Flanagan, L. B., Shurpali, N. J., Grünwald, T., Humphreys, E. R., Jackowicz-Korczyński, M., Aurela, M. A.,

Laurila, T., Grüning, C., Corradi, C. A. R., Schrier-Uijl, A. P., Christensen, T. R., Tamstorf, M. P., Mastepanov, M., Martikainen, P. J., Verma, S. B., Bernhofer, C., and Cescatti, A.: The uncertain climate footprint of wetlands under human pressure, Proceedings of the National Academy of Sciences, 112, 4594-4599, 2015.

605 Pihlatie, M. K., Kiese, R., Brüggemann, N., Butterbach-Bahl, K., Kieloaho, A. J., Laurila, T., Lohila, A., Mammarella, I., Minkkinen, K., Penttilä, T., Schönborn, J., and Vesala, T.: Greenhouse gas fluxes in a drained peatland forest during spring frost-thaw event, Biogeosciences, 7, 1715-1727, 2010.

Price, J. S. and Waddington, J. M.: Advances in Canadian wetland hydrology an biogeochemistry, Hydrological Processes, 14, 1579-1589, 2000.

Richards, B. and Craft, C. B.: Greenhouse Gas Fluxes from Restored Agricultural Wetlands and Natural Wetlands, 610 Northwestern Indiana. In: The Role of Natural and Constructed Wetlands in Nutrient Cycling and Retention on the Landscape, Vymazal, J. (Ed.), Springer International Publishing, Cham, 2015.

Rochefort, L., Campeau, S., and Bugnon, J.-L.: Does prolonged flooding prevent or enhance regeneration and growth of Sphagnum?, Aquatic Botany, 74, 327-341, 2002.

Roulet, N.: Peatlands, carbon storage, greenhouse gases, and the Kyoto Protocol: Prospects and significance for Canada, 615 Wetlands, 20, 605-615, 2000.

Roulet, N. T., Lafleur, P. M., Richard, P. J. H., Moore, T. R., Humphreys, E. R., and Bubier, J.: Contemporary carbon balance and late Holocene carbon accumulation in a northern peatland, Global Change Biology, 13, 397-411, 2007.

Rydin, H., Jeglum, J. K., Jeglum, J. K., and Bennett, K. D.: The Biology of Peatlands, 2e, OUP Oxford, 2013.

Schipper, L. A. and Reddy, K. R.: Methane Production and Emissions from Four Reclaimed and Pristine Wetlands of 620 Southeastern United States, Soil Sci. Soc. Am. J., 58, 1270-1275, 1994.

Schrier-Uijl, A. P., Kroon, P. S., Hendriks, D. M. D., Hensen, A., Van Huissteden, J., Berendse, F., and Veenendaal, E. M.: Agricultural peatlands: towards a greenhouse gas sink – a synthesis of a Dutch landscape study, Biogeosciences, 11, 4559-4576, 2014.

Schrier-Uijl, A. P., Kroon, P. S., Leffelaar, P. A., van Huissteden, J. C., Berendse, F., and Veenendaal, E. M.: Methane 625 emissions in two drained peat agro-ecosystems with high and low agricultural intensity, Plant and Soil, 329, 509-520, 2010.

Shannon, R. and White, J.: A three-year study of controls on methane emissions from two Michigan peatlands, Biogeochemistry, 27, 35-60, 1994.

Shurpali, N. J., Hyvö Nen, N. P., Huttunen, J. T., Biasi, C., Nykä Nen, H., Pekkarinen, N., and Martikainen, P. J.: Bare soil and reed canary grass ecosystem respiration in peat extraction sites in Eastern Finland, Tellus B, 60, 200-209, 2008.

630 Shurpali, N. J., Verma, S. B., Kim, J., and Arkebauer, T. J.: Carbon dioxide exchange in a peatland ecosystem, Journal of Geophysical Research: Atmospheres, 100, 14319-14326, 1995.

Smith, S. J. and Wigley, M. L.: Global Warming Potentials: 1. Climatic Implications of Emissions Reductions, Climatic Change, 44, 445-457, 2000.

Sottocornola, M. and Kiely, G.: Hydro-meteorological controls on the CO2 exchange variation in an Irish blanket bog, Agricultural and Forest Meteorology, 150, 287-297, 2010.

Strack, M., Waddington, J. M., Rochefort, L., and Tuittila, E. S.: Response of vegetation and net ecosystem carbon dioxide exchange at different peatland microforms following water table drawdown, Journal of Geophysical Research: Biogeosciences, 111, n/a-n/a, 2006.

Strack, M. and Zuback, Y. C. A.: Annual carbon balance of a peatland 10 yr following restoration, Biogeosciences, 2013. 12, 2013.

Sun, L., Song, C., Miao, Y., Qiao, T., and Gong, C.: Temporal and spatial variability of methane emissions in a northern temperate marsh, Atmospheric Environment, 81, 356-363, 2013.

Tapio-Biström, M. L., Joosten, H., Tol, S., Food, Project, A. O. o. t. U. N. M. o. C. C. i. A., and International, W.: Peatlands: Guidance for Climate Change Mitigation Through Conservation, Rehabilitation and Sustainable Use, Food and Agriculture Organization of the United Nations, 2012.

Tuittila, E.-S., Komulainen, V.-M., Vasander, H., Nykänen, H., Martikainen, P. J., and Laine, J.: Methane dynamics of a restored cut-away peatland, Global Change Biology, 6, 569-581, 2000.

Turetsky, M. R., Kotowska, A., Bubier, J., Dise, N. B., Crill, P., Hornibrook, E. R. C., Minkkinen, K., Moore, T. R., Myers-Smith, I. H., Nykänen, H., Olefeldt, D., Rinne, J., Saarnio, S., Shurpali, N., Tuittila, E.-S., Waddington, J. M., White, J. R., Wickland, K. P., and Wilmking, M.: A synthesis of methane emissions from 71 northern, temperate, and subtropical wetlands, Global Change Biology, 20, 2183-2197, 2014.

van der Werf, G. R., Randerson, J. T., Collatz, G. J., Giglio, L., Kasibhatla, P. S., Arellano, A. F., Olsen, S. C., and Kasischke, E. S.: Continental-Scale Partitioning of Fire Emissions During the 1997 to 2001 El Niño/La Niña Period, Science, 303, 73-76, 2004.

Waddington, J. M. and Roulet, N. T.: Carbon balance of a boreal patterned peatland, Global Change Biology, 6, 87-97, 2000.

Waddington, J. M., Strack, M., and Greenwood, M. J.: Toward restoring the net carbon sink function of degraded peatlands: Short-term response in CO2 exchange to ecosystem-scale restoration, Journal of Geophysical Research: Biogeosciences, 115, n/a-n/a, 2010.

Wang, Z.-P. and Han, X.-G.: Diurnal variation in methane emissions in relation to plants and environmental variables in the Inner Mongolia marshes, Atmospheric Environment, 39, 6295-6305, 2005.

Wania, R., Ross, I., and Prentice, I. C.: Integrating peatlands and permafrost into a dynamic global vegetation model: 1. Evaluation and sensitivity of physical land surface processes, Global Biogeochemical Cycles, 23, n/a-n/a, 2009.

Weltzin, J. F., Pastor, J., Harth, C., Bridgham, S. D., Updegraff, K., and Chapin, C. T.: Response of bog and fen plant communities to warming and water-table manipulations, Ecology, 81, 3464-3478, 2000.

Whiting, G. J. and Chanton, J. P.: Greenhouse carbon balance of wetlands: methane emission versus carbon sequestration, 2001, 53, 2001.

Wilson, D., Farrell, C., Mueller, C., Hepp, S., and Renou-Wilson, F.: Rewetted industrial cutaway peatlands in western Ireland: a prime location for climate change mitigation?, Mires and Peat, 11, 2013.

Wisniewski, J. and Sampson, R. N.: Terrestrial Biospheric Carbon Fluxes Quantification of Sinks and Sources of CO2, Springer Netherlands, 2012.

Yu, Z.: Holocene carbon flux histories of the world's peatlands: Global carbon-cycle implications, The Holocene, doi: 10.1177/0959683610386982, 2011. 2011.

Yu, Z., Loisel, J., Brosseau, D. P., Beilman, D. W., and Hunt, S. J.: Global peatland dynamics since the Last Glacial Maximum, Geophysical Research Letters, 37, n/a-n/a, 2010.

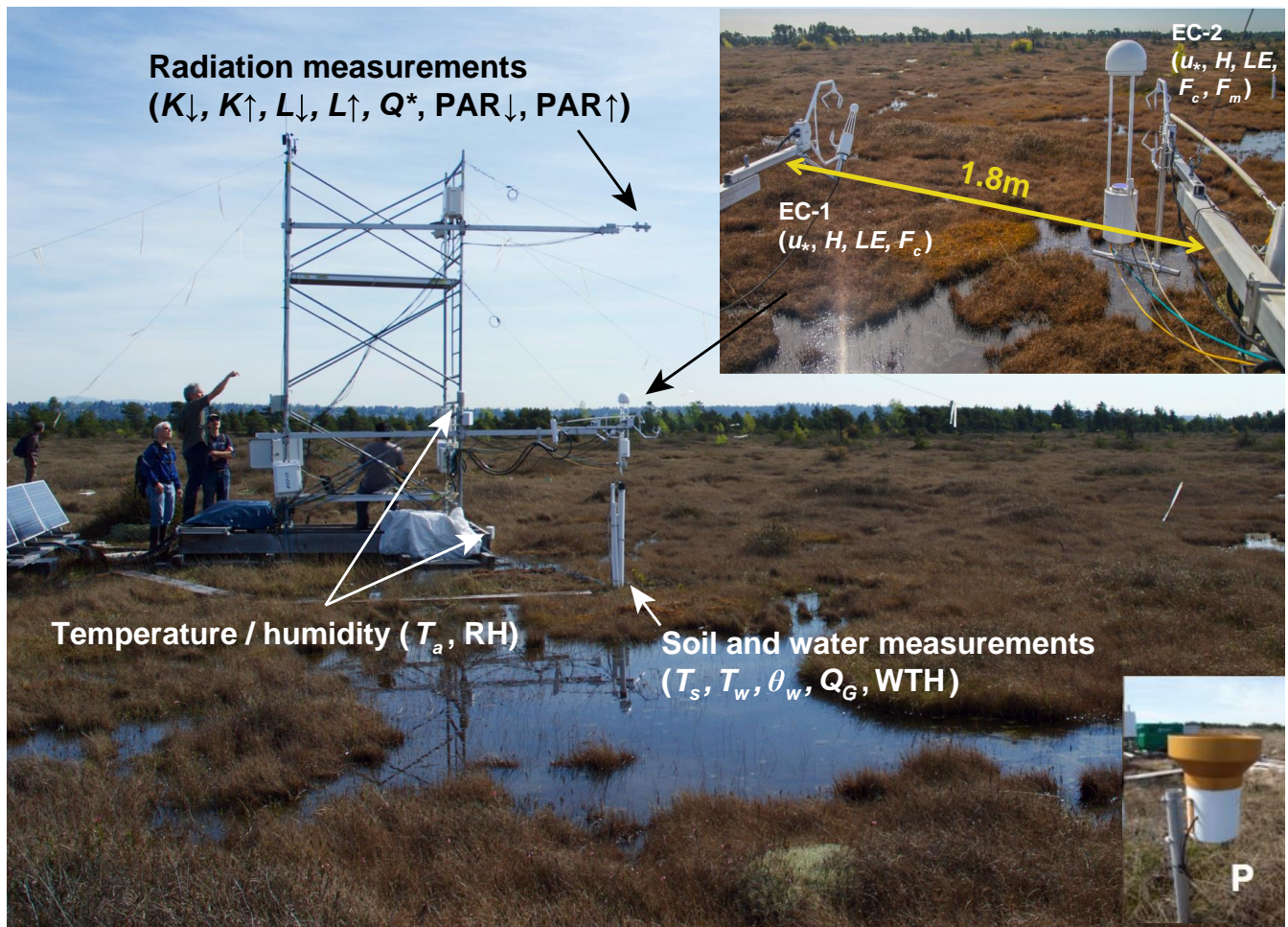
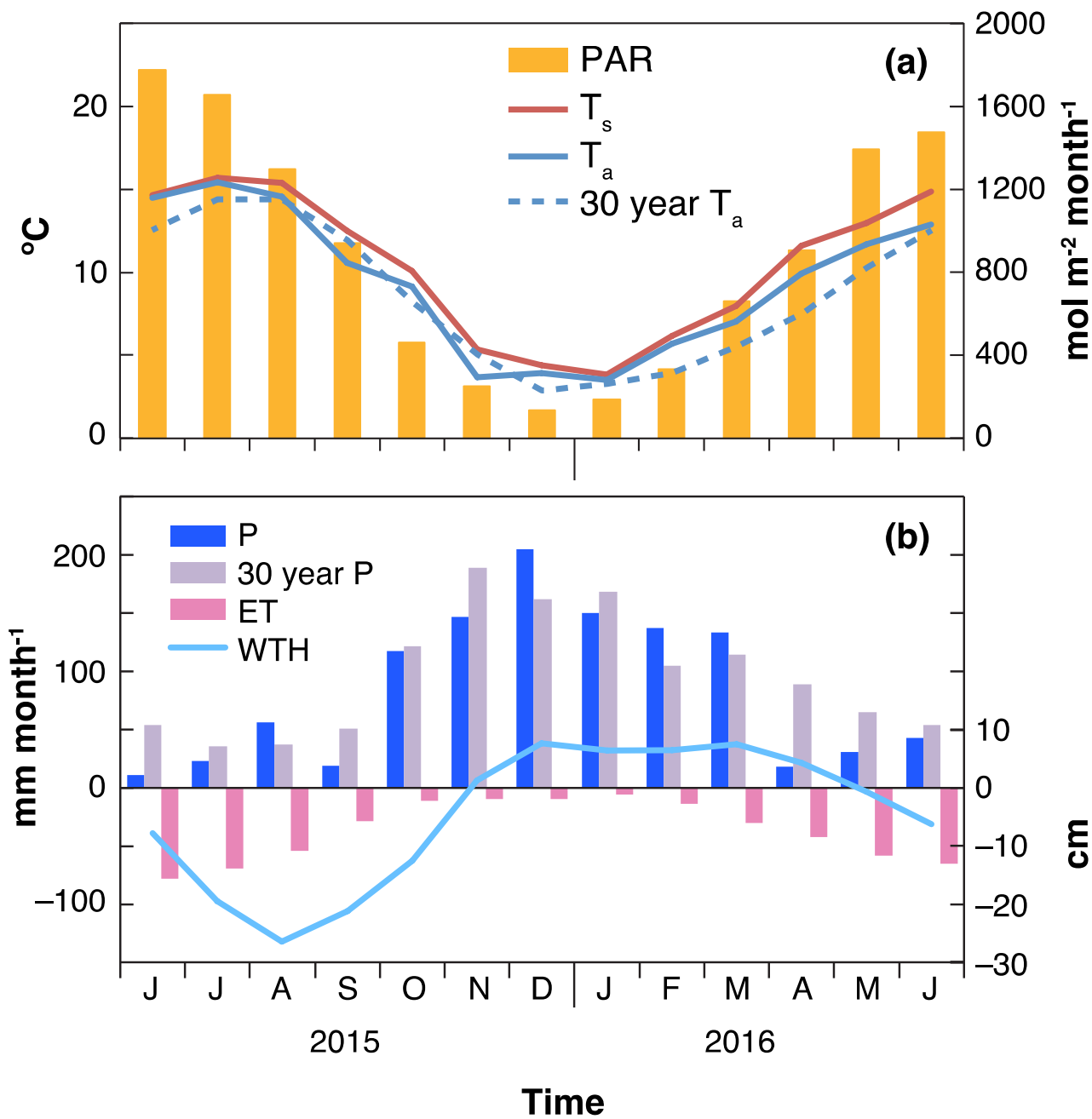



Figure 1: Flux tower on floating platform with EC-1 and EC-2 systems facing south and instruments that measured climate variables indicated (friction velocity (u_*), sensible heat flux (H), latent heat flux (LE), CO_2 flux (F_c), CH_4 flux (F_m), incoming shortwave radiation (K $\downarrow$), outgoing shortwave radiation (K $\uparrow$), incoming longwave radiation (L $\downarrow$), outgoing longwave radiation (L $\uparrow$), net all-wave radiation (Q*), incoming PAR (PAR $\downarrow$), outgoing PAR (PAR $\uparrow$), air temperature (T_a), relative humidity (RH), soil temperature (T_s), water temperature (T_w), soil water content (θ_w), soil heat flux (Q_G), water table height (WTH), and precipitation (P)).

685   Figure 2: The annual course of weather variables (T_a, T_s, P, and PAR) and WTH. The 30-year climate normals (30-year T_a and P) were measured at Vancouver International Airport (Data: Environment Canada).

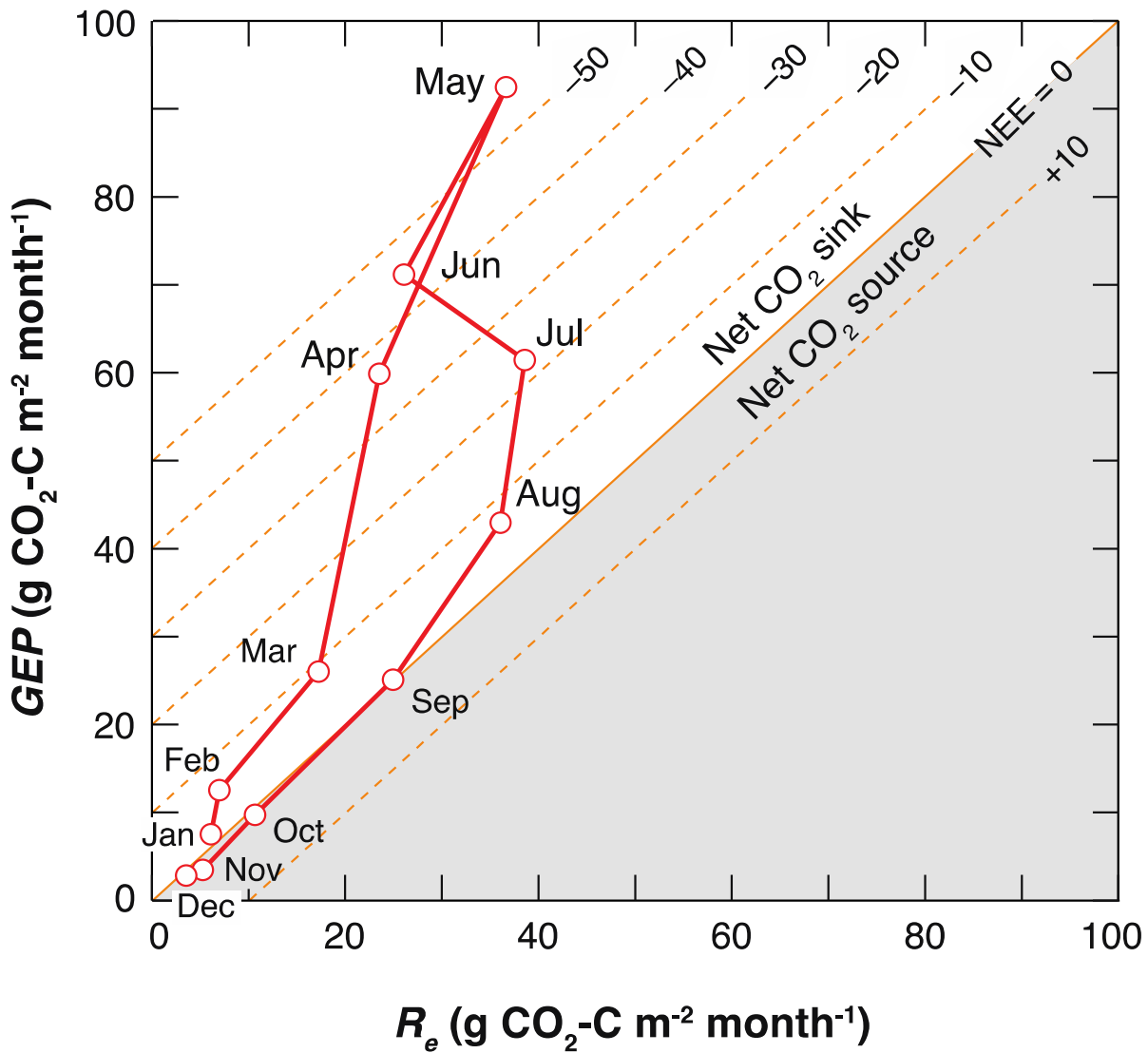



Figure 3: Monthly gap-filled R_e (x-axis) drawn against GEP (y-axis). The resulting NEE can be read off the diagonal lines. The thick 1:1 line shows carbon neutrality, while lines in the upper right are of increasingly negative NEE (uptake) and lines towards the lower right are positive NEE (net source).

690

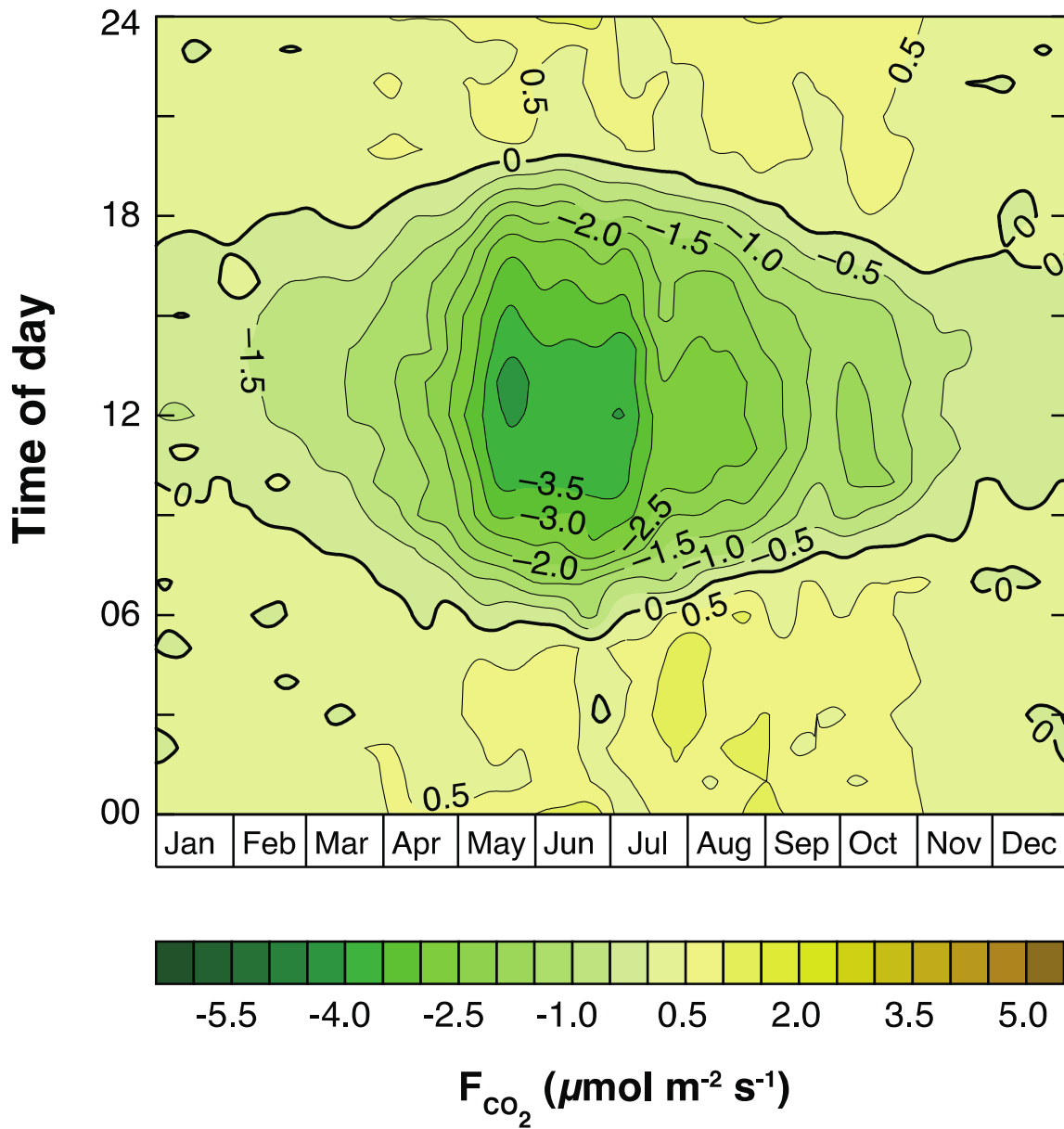



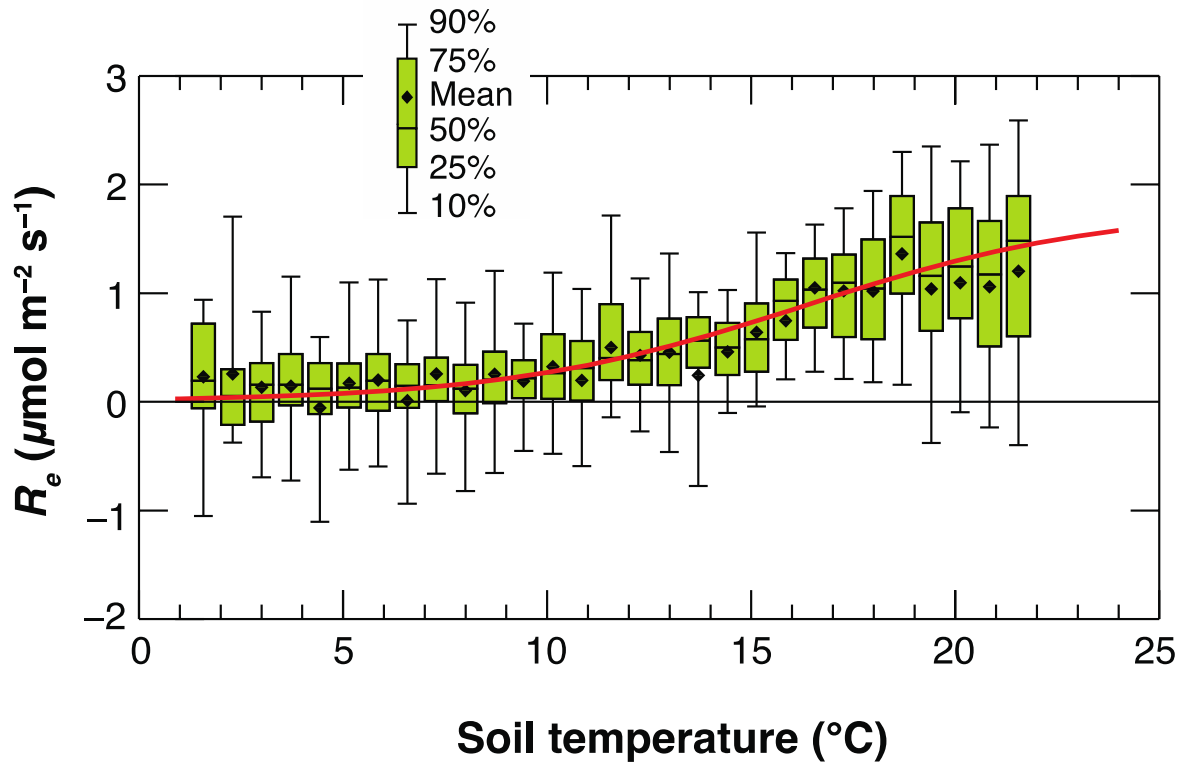
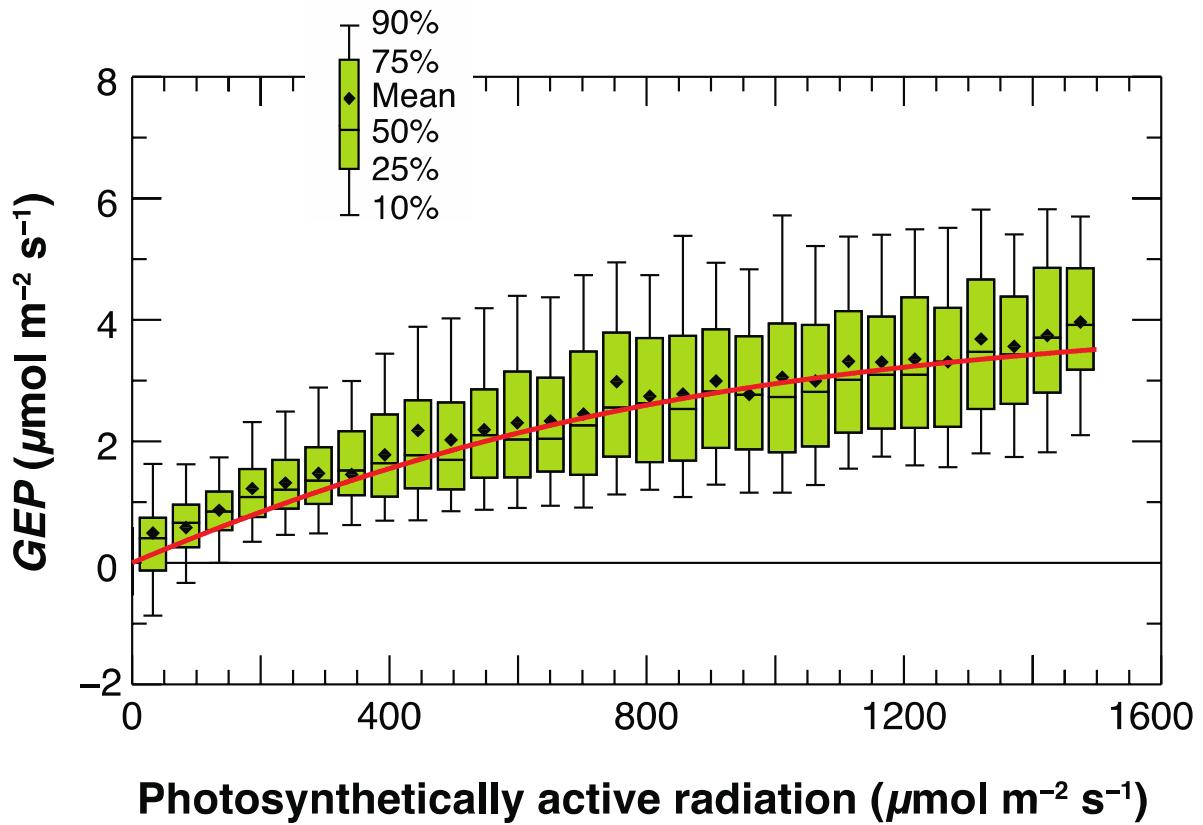
Figure 4: Isopleths of gap-filled NEE (net CO$_2$ fluxes) from the EC-1 system plotted as a composite in the study year. The graph uses a Gaussian filter of $\sigma = 45$ days (which conserves total NEE) to graphically smooth horizontal variations.

695   Figure 5: Relationship between R_e (nighttime 30-minute CO_2 flux measurements) and $T_{s,5cm}$ during the entire study period. The u_* threshold was 0.08 m s^{-1}. The fitted curve is a logistic relationship following Eq. 1. $T_{s,5cm}$ was binned for 32 classes from minimum of $T_{s,5cm}$ to maximum of $T_{s,5cm}$. See Fig. S3 in supplement for seasonal differences. Negative R_e values were caused by measurement uncertainties.

700

Figure 6: Annual light response curve determined from the daytime 30-minute NEE measurements and Eq. 1, i.e., GEP = R_e + -NEE. The curves are the best fit of the Eq. 2. PPFD was binned for 30 classes from 0 to 1500 μmol m^{-2} s^{-1}. Annual MQY was 4.00 mmol C mol^{-1} photons, P_M was 4.68 umol m^{-2} s^{-1}, and C_v was 0.7 (fixed).

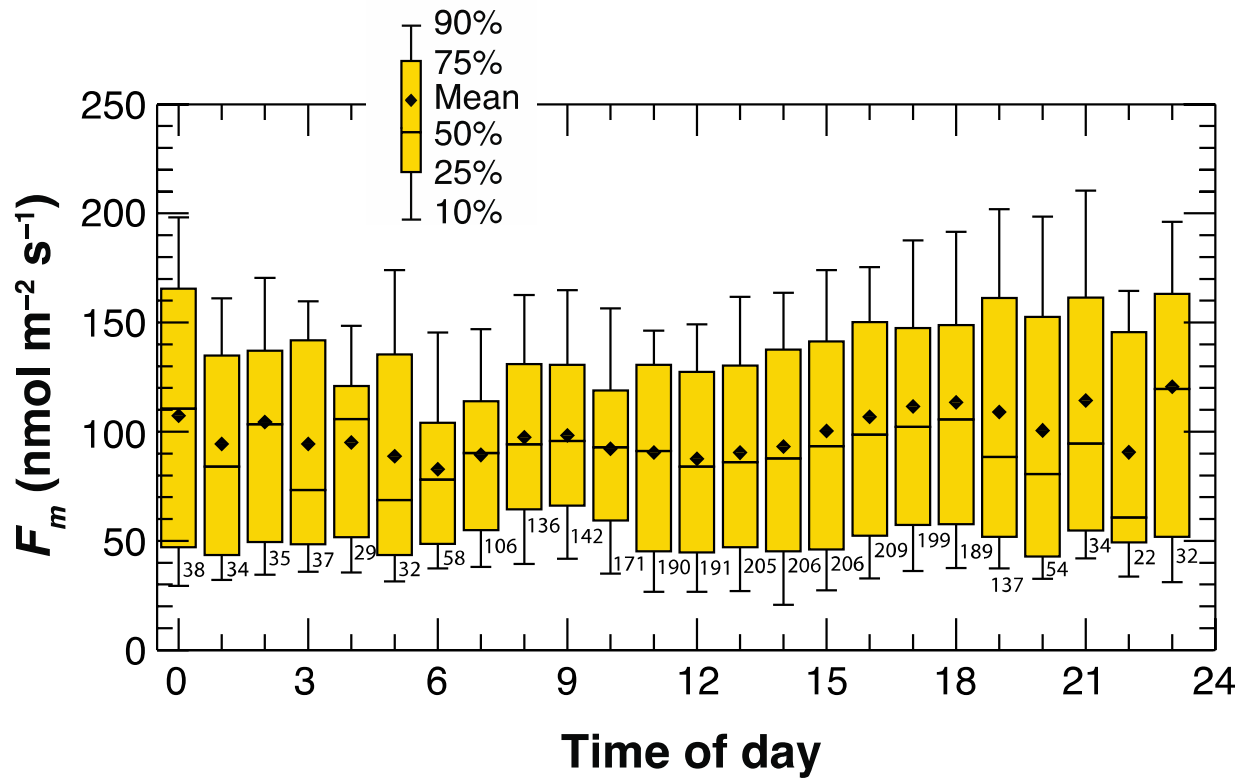



Figure 7: Diurnal course of measured CH$_4$ fluxes from the EC-2 system during the study period.

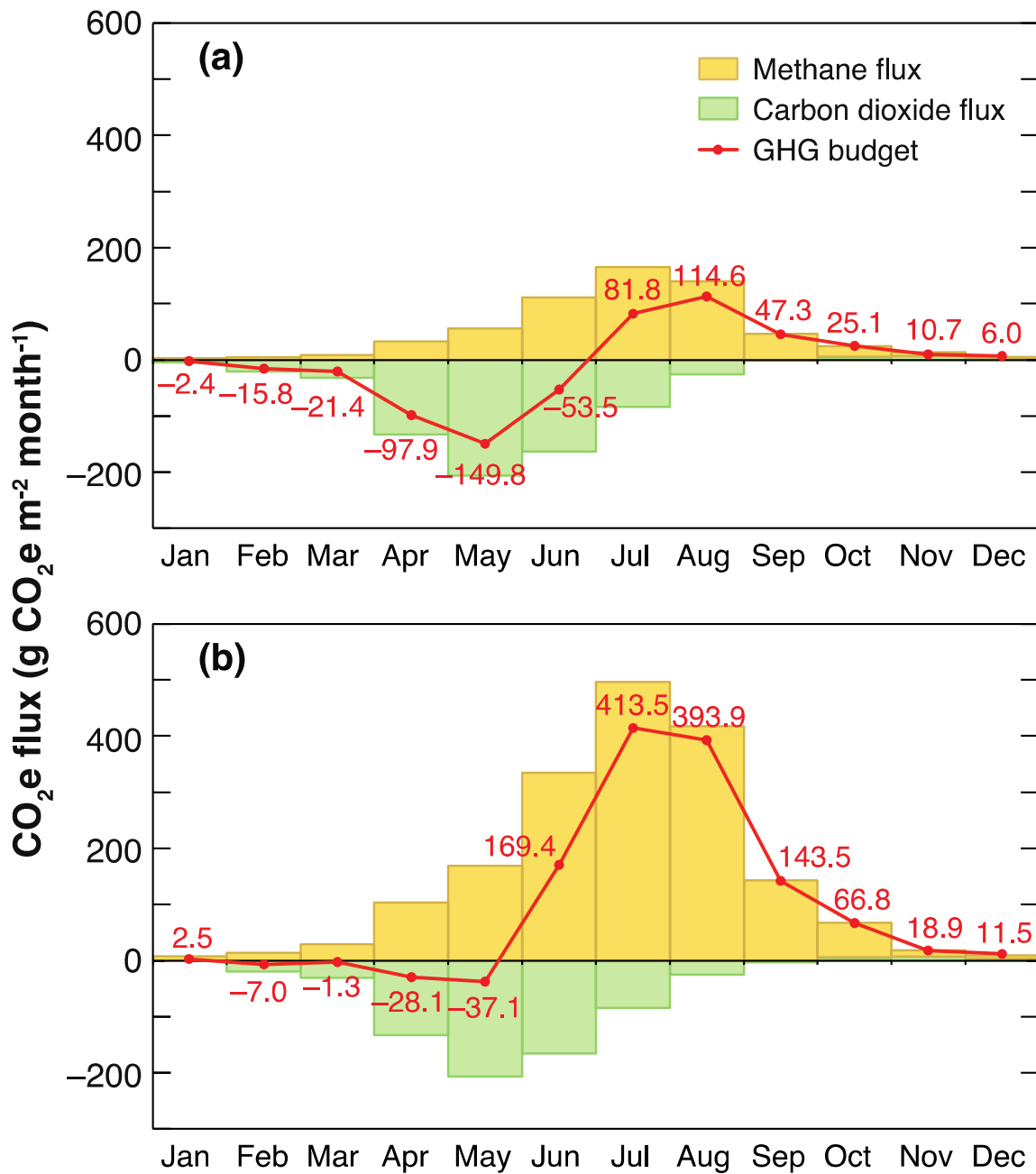



Figure 8: EC-measured monthly CO_2, CH_4 and net GHGs fluxes shown as CO_2e totals by using (a) 100-year and (b) 20-year GWPs. Missing data were gap-filled.

Table 1: Monthly EC-measured and gap-filled NEE (CO_2 fluxes), CH_4 fluxes, CO_2e fluxes using 20-year GWP, and CO_2e fluxes using 100-year GWP at the study site during the study period.

Month	R_e	GEP	NEE	CH_4 fluxes	20-year CO_2e fluxes	100-year CO_2e fluxes
	(g CO_2-C m^{-2} month^{-1})			(mg CH_4-C m^{-2} month^{-1})	(g CO_2e m^{-2} month^{-1})	(g CO_2e m^{-2} month^{-1})
Jan	6.17	7.50	-1.33	66	2.5	-2.4
Feb	6.94	12.46	-5.52	118	-7.0	-15.8
Mar	17.33	25.89	-8.59	269	-1.3	-21.4
Apr	23.52	59.73	-36.21	933	-28.0	-97.8
May	36.46	92.63	-56.20	1506	-37.0	-149.6
Jun	26.13	71.10	-44.97	2980	169.5	-53.3
Jul	38.53	61.47	-22.94	4436	413.6	81.8
Aug	36.15	42.97	-6.82	3734	393.9	114.6
Sep	24.84	25.08	-0.21	1286	143.5	47.3
Oct	10.76	9.58	1.18	557	66.8	25.2
Nov	5.16	3.39	1.77	111	18.9	10.6
Dec	3.63	2.79	0.87	74	11.5	6.0
Study	g CO_2-C m^{-2} year^{-1}			g CH_4-C m^{-2} year^{-1}	g CO_2e m^{-2} year^{-1}	
year	236	415	-179	16	1147	-55

715

Table 2: Comparison of annual NEE, R_e and GEP, over different ecosystems (vegetation covers) in the Vancouver region using EC measurements. Sorted by magnitude of -NEE/GEP ratio.

Site	Land cover	NEE	R_e	GEP	-NEE/GEP
		g C m^{-2} year^{-1}			
Burns Bog **(this study)** Delta, BC	Rewetted raised bog ecosystem	-179	236	415	43%
Westham Island (CA-Wes)[*] Delta, BC	Unmanaged grassland	-222	1215	1438	15%
Campbell River (CA-Ca1)[*] Vancouver Island	Douglas-fir forest (~55 yrs)	-328[+]	1830[+]	2158[+]	15%
Buckley Bay (CA-Ca3)[*] Vancouver Island	Douglas-fir forest (~15 yrs)	64[+]	1487[+]	1423[+]	-4%

[*] Site identifier in global FLUXNET database (http://fluxnet.ornl.gov).

720   [+] Data from Krishnan et al., 2009 before fertilisation.