Wetlands Ecology and Management

Methane emissions from fens in Alberta’s boreal region: Reference data for functional evaluation of restoration outcomes

--Manuscript Draft--

Manuscript Number:	WETL-D-19-00231R1
Full Title:	Methane emissions from fens in Alberta’s boreal region: Reference data for functional evaluation of restoration outcomes
Article Type:	Original Research
Section/Category:	Biogeochemistry
Keywords:	CH4; dissolved methane; ecological restoration; peatland; reclamation
Corresponding Author:	Maria Strack University of Waterloo Waterloo, ON CANADA
Corresponding Author Secondary Information:	
Corresponding Author's Institution:	University of Waterloo
Corresponding Author's Secondary Institution:	
First Author:	Aneta Bienida
First Author Secondary Information:	
Order of Authors:	Aneta Bienida
	Vinay Daté
	Roxane Andersen
	Felix Nwaishi
	Jonathan Price
	Md. Sharif Mahmood
	Maria Strack
Order of Authors Secondary Information:	
Funding Information:	Natural Sciences and Engineering Research Council of Canada (418557) — Dr. Jonathan Price
Abstract:	The aim of the study was to document methane (CH_4) dynamics from fen ecosystems in the Athabasca Oil Sands Region (AOSR) in northern Alberta to create a reference database for evaluation of peatland restoration and reclamation projects in the region. The study included three types of fens commonly occurring in this region: poor fen (open and treed), moderately-rich treed fen, and open saline fen. We quantified CH_4 fluxes, pore water concentration (PW[CH_4]), and production potential together with ecohydrological variables that may influence CH_4 dynamics over four growing seasons. Mean (standard deviation) fluxes for open and treed poor fen (99.8 (269.7) and 68.3 (118.8) mg CH_4 m^{-2} d^{-1}, respectively) were higher than for treed rich (32.8 (63.7) mg CH_4 m^{-2} d^{-1}) and open saline fens (34.6 (91.3) mg CH_4 m^{-2} d^{-1}). The total growing season CH_4 emissions from these fens ranged between 3.7 and 11.3 g CH_4 m^{-2}. Methane production potential varied from 0.1 (0.1) µmol CH_4 g peat^{-1} d^{-1} at the saline fen to 4.6 (0.8) µmol CH_4 g peat^{-1} d^{-1} at the treed rich fen. The variability of CH_4 fluxes and pore water concentration between study sites and years was mostly controlled water table (WT) and soil temperature indicating that these variables should be used to assess the expected CH_4 flux in peatland reclamation projects. Large inter-annual variability in CH_4 flux illustrates the importance of multi-year records for data used in functional

	evaluation of restoration outcomes.
Response to Reviewers:	Thank-you for the opportunity to revise our manuscript. The comments from the reviewer have helped to clarify many points in the manuscript. We present each reviewer comment, followed by our response. Line numbers in our response refer to those in the revised version. Reviewer report: This report specifically examines methane dynamics in four different fen peatland ecosystem types in the Athabasca Oil Sands region, over four years. The principal objective was to establish "reference" conditions for methane efflux to be used as benchmarks for restoration; this goal appears to be appropriate for 'Wetland Ecology and Management'. The writing is concise and describes the data quite well. The figures and tables are straightforward, and there is enough detail to replicate the field monitoring protocols. The authors place their findings in the broader context of northern fen research (table 4). I offer minor comments by line number, below, and hope they are helpful with revision. Line 164: Might be good to specify that this work focuses on diffusive efflux, and does not reflect ebullition events. Response: A good suggestion and we have added the following sentence at the end of this paragraph to clarify what is included in the fluxes based on our quality control methods: “Therefore, results presented here largely represent diffusive and plant-mediated fluxes as sporadic ebullition events would have been removed from the data set based upon our quality control criteria; steady ebullition, if it occurred, would be included as it would result in a linear increase in concentration change over the chamber closure period”. (lines 168-172) Line 174-175: It would be good to specify the micron size, or brand, of nylon screening. Response: The screening had a 250 μm mesh size and this has now been specified in the text (line 179). Line 188-189: It would be nice to briefly provide a little information on how the incubation headspace N2 was flushed (glovebox? Replacing headspace?). Response: The jars were prepared and sealed in a glovebox flushed with N2. We have clarified this in the text (line 193). Line 202-203: This is noted again on line 343, but it would be good to note right up front that since canopy is not captured in the 30 cm tall chambers, and yet tree roots are, ecosystem respiration is likely higher than what is represented by the plants present, and GEP is therefore likely to be an underestimation. Response: Since we only use GEP in our analysis of potential controls on CH4 flux (and not ecosystem respiration), we have chosen to only highlight here the underestimation of ecosystem GEP and not the effects of tree roots on respiration. Since including tree root respiration in our chamber measurements will not effect the GEP estimate directly, we do not want to confuse the reader by highlighting that the respiration measured is higher than that from only the ground layer plants as it is not relevant to the data used in further analysis. In short, we have modified the last sentence in this paragraph so that it now reads: “As these measurements included only the vegetation within the collar, they represent GEP of the understory vegetation only and do not include trees present at TPF and TRF and therefore underestimate total ecosystem GEP.” Line 333: Would it be appropriate to briefly list the statistic used to establish if a value was truly an "outlier" (Cook's D statistic, or similar?)? Response: These values were greater than 2 orders of magnitude different that other samples from the same study site and so appeared as true outliers. We did not use a specific statistical test, but have now clarified how far outside the other replicates they lie in the text (line 340). Line 436: The authors introduce the "multifaceted role" that different plant functional groups play on CH4 efflux/oxidation; perhaps this could be a little more detailed here, coming back to aerenchymatous plants?

	Response: We have added the following sentence and associated reference here to provide more detail on plant roles in CH4 cycling "For example, while aerenchymatous plants have been shown to increase the transport of CH4 from the soil to the atmosphere, in some cases, CH4 emissions can be reduced due to greater oxidation in the rooting zone (Bhullar et al., 2013)." (lines 444-447) Line 445: I think these references pertaining to trees venting CH4 to the atmosphere are for tropical trees? I don't think black spruce or tamarack have pneumatophores or lenticels? Response: More and more recent research is suggesting that even trees without pneumatophores or lenticels may vent methane from wetland soils, although little research specific to black spruce or tamarack has been conducted. While the Pangala reference is related to tropical wetlands, the Gauci reference actually refers to alders in a temperate fen. We also already acknowledge here that more research is needed in boreal peatlands to determine if trees are really playing an important role, so we feel this addresses the reviewer's concerns and have not made any further changes here. Line 450-465: It seems there is something interesting going on with pore water [CH4] in 2014, with a large spike that doesn't necessarily correspond with CH4 efflux. Can the authors speculate as to what may be contributing to the spike in PW[CH4] in that year? Response: We did not specifically investigate what might cause this decoupling, but hypothesize that it could reflect the potentially long mean residence time of porewater CH4 (months to years). So, we posit that the porewater pool lags hydrological conditions slightly and that the continuing increase in porewater concentration in 2014 may reflect increases in CH4 production in 2013 and continued favourable conditions as most sites in 2014, allowing the pool to continue to grow, while falling water table would enhance oxidation and start to reduce emissions. We have added the following to this section: "However, there appears to be some decoupling between CH4 emissions and PW[CH4] (Figure 3). The continuing increase in PW[CH4] in 2014 while emissions declined compared to the previous year may reflect the fact that mean residence time of CH4 in peat can be months to years (Strack and Waddington, 2008). Therefore, the dissolved CH4 pool in 2014 potentially reflects the favourable CH4 production conditions in 2013 when all sites had shallow water tables." (lines 475-479). Line 478: See comment above about apparent decoupling of PW and CH4 efflux for some of the sites in 2014. Response: We agree and have addressed this in detail in the previous comment. Here, we now say "In our study PW[CH4] was generally linked to fluxes but showed less spatial and temporal variability", adding the word "generally" to reflect that there is some level of decoupling.

Methane emissions from fens in Alberta's boreal region: Reference data for functional evaluation of restoration outcomes

Aneta Bienida[1], Vinay Daté[1], Roxane Andersen[2], Felix Nwaishi[1,3], Jonathan Price[1], Md. Sharif Mahmood[4], Maria Strack[1,4]

[1] University of Waterloo, Department of Geography and Environmental Management, 200 University Avenue West, Waterloo, ON, N2L 3G1, Canada
[2]University of Highlands and Islands, Environmental Research Institute, Thurso, UK
[3]Mount Royal University, Earth and Environmental Sciences, Calgary, AB, Canada
[4]University of Calgary, Department of Geography, Calgary, AB, Canada

Correspondence to: Maria Strack (mstrack@uwaterloo.ca)

Keywords: CH_4, dissolved methane, ecological restoration, peatland, reclamation

Abstract

The aim of the study was to document methane (CH_4) dynamics from fen ecosystems in the Athabasca Oil Sands Region (AOSR) in northern Alberta to create a reference database for evaluation of peatland restoration and reclamation projects in the region. The study included three types of fens commonly occurring in this region: poor fen (open and treed), moderately-rich treed fen, and open saline fen. We quantified CH_4 fluxes, pore water concentration (PW[CH_4]), and production potential together with ecohydrological variables that may influence CH_4 dynamics over four growing seasons. Mean (standard deviation) fluxes for open and treed poor fen (99.8 (269.7) and 68.3 (118.8) mg CH_4 m^{-2} d^{-1}, respectively) were higher than for treed rich (32.8 (63.7) mg CH_4 m^{-2} d^{-1}) and open saline fens (34.6 (91.3) mg CH_4 m^{-2} d^{-1}). The total growing season CH_4 emissions from these fens ranged between 3.7 and 11.3 g CH_4 m^{-2}. Methane production potential varied from 0.1 (0.1) µmol CH_4 g $peat^{-1}$ d^{-1} at the saline fen to 4.6 (0.8) µmol CH_4 g $peat^{-1}$ d^{-1} at the treed rich fen. The variability of CH_4 fluxes and pore water concentration between study sites and years was mostly controlled water table (WT) and soil temperature indicating that these variables should be used to assess the expected CH_4 flux in peatland reclamation projects. Large inter-annual variability in CH_4 flux illustrates the importance of multi-year records for data used in functional evaluation of restoration outcomes.

Introduction

Northern peatlands play an important role in the global carbon cycle by acting as large soil carbon stocks, contributing significant amounts of dissolved carbon to downstream ecosystems and accounting for 5 – 10% of global CH_4 emissions (Blodau 2002). Boreal and subarctic peatlands release 17 – 61 Tg of CH_4 per year and inter-annual variations in these emissions may contribute to fluctuations in atmospheric CH_4 concentration (Bridgham et al. 2013). In the Athabasca Oil Sands Region (AOSR) of Alberta, Canada, open pit mining for oil sands extraction has disturbed 895 km^2 boreal forest (Government of Alberta 2018), ~50% of which is covered by peatlands (Vitt et al. 1996), with 90% of these peatlands being fens (Vitt et al. 2000). The Alberta government requires land disturbed by oil sands extraction to be returned to equivalent land capability (Province of Alberta 2018), with recently more focus placed on including peatlands in the post-mining landscape (e.g., Daly et al. 2012). Therefore, understanding and quantifying key processes in fen ecosystems in near-pristine condition is essential to develop reference baselines for future evaluation of reclaimed landscapes (Nwaishi et al. 2015). This study focuses specifically on fen CH_4 dynamics in the AOSR.

Methane is produced in soils under highly reducing conditions by methanogenic *Archaea* (Rosenberry et al. 2006; Lai 2009). The saturated soil conditions in peatlands allow for CH_4 production; however, presence of alternative terminal electron acceptors (TEAs), such as NO_3^-, SO_4^{2-}, Fe^{3+}, and anaerobic bacteria that utilize them, can inhibit or reduce rates of methanogenesis by creating conditions where the reaction is not thermodynamically favoured (Bridgham et al. 2013; Madigan et al. 2009; Minderlein & Blodau 2010). As fens receive water from ground and surface water sources, concentrations of TEAs can be higher than in bogs, thereby reducing CH_4 production (Estop-Aragonés et al. 2013). Methane production is also dependent on organic matter substrate quality, plant community composition and productivity (Bridgham et al. 2013; Tuittila et al. 2000). The presence of highly-productive graminoids in many fens provides large quantities of fresh substrate through root exudates and litter accumulation that should enhance CH_4 production (Strack et al. 2017).

The atmospheric flux of CH_4 from a peatland is dependent not only on production, but also on CH_4 oxidation rate and transport pathways. Methane is oxidized by methanotrophic bacteria, with greatest rates usually measured just above the mean water table position, where there is a source of both CH_4 and oxygen (Andersen et al. 2013; Clymo & Bryant 2008; Sundh et al. 1995). Since the majority of CH_4 is oxidized in the unsaturated zone, water table is often a good predictor of flux (Couwenberg & Fritz 2012). Plants with aerenchymatous tissue, which transports oxygen to roots growing in saturated soils, can contribute to CH_4 oxidation below the water table, as radial oxygen loss from roots supports methanotrophic activity (Popp et al. 2000). Anaerobic oxidation has also been reported from peatlands (Gupta et al. 2013) and is likely linked to reduction of TEAs including NO_3^-, SO_4^{2-}; however, its role in reducing CH_4 flux *in situ* remains unclear.

Once produced, CH_4 can be transported to the atmosphere via diffusion through the peat matrix, plant-mediated transport, and ebullition. Diffusion through peat is slow due to its high water content, and has the potential to result in high rates of CH_4 oxidation, particularly if the water table is deep (Lai 2009). Plant-mediated transport is the movement of CH_4 through plant tissue as diffusion through aerenchyma, pressure-driven flow, or dissolved in water lost through transpiration (Lai 2009). Emission by plants can account for the majority of CH_4 flux from peatlands, particularly when plants with aerenchyma are present (Couwenberg & Fritz 2012). Ebullition may also account for a large proportion of CH_4 emission; Glaser et al. (2004) estimated that several large, episodic ebullition events accounted for over 50% of annual emissions from a bog. The importance of ebullition in fens is less clear, but losses of CH_4 through plant-mediated transport may reduce subsurface CH_4 pools (Strack et al. 2017), potentially limiting bubble accumulation and thus ebullition.

Following oil sands extraction in Alberta, government regulations require the return of the landscape to equivalent land capability. While this does not necessarily require the return to conditions identical to those present pre-disturbance, focus on returning peatland ecosystems to the post-mining landscape has increased in recent years (Daly et al. 2012; Environment and Parks 2017). Reclamation criteria for particular disturbances related to oil sands extraction, such as well-pads and associated roads, focus largely on returning appropriate vegetation communities (Environment and Parks 2017); however, Nwaishi et al. (2015) argue that outcomes should be evaluated using functional indicators. Methane production and emission indicates decomposition of organic matter under high-reduced, anoxic conditions, those characteristics of peat-forming conditions, indicating that CH_4 accumulation in pore water and atmospheric flux are useful indicators of peatland function. However, to be used as a functional indicator, data from representative reference ecosystems, in this case fens in boreal western Canada, is required. A review of previous measurements of CH_4 flux in northern fens reports mean annual emissions of

15.4 g CH_4-C m^{-2} yr^{-1}, with large variation between sites being significantly related to water table position (Abdalla et al. 2016). According to Abdalla et al.'s (2016) compiled data set, Canadian fens emitted 0 to 154 g CH_4-C m^{-2} yr^{-1}; however, this only incorporated one reported measurement from Alberta, namely 2.8 g CH_4-C m^{-2} yr^{-1} from a treed moderately-rich fen (Long et al. 2010). Given the limited available data on fen CH_4 flux from the AOSR and continued disturbance of natural peatlands, it is critical to quantify CH_4 fluxes from a range of fen types in the region that could be used as reference ecosystems and generate data for evaluation of current and future reclamation projects. Therefore, the objectives of this study were to: 1) quantify CH_4 flux, pore water concentration and potential production rates from representative fens in the AOSR, 2) evaluate spatial and temporal variation in CH_4 dynamics at each fen type, and 3) investigate the relationship of CH_4 dynamics to ecohydrological conditions.

Methods

Study sites

In May 2011, three main study sites (poor fen, moderate-rich fen and saline fen) were chosen in the AOSR that represented a range of fen types in the region, had large sections where hydrology was unaffected by human disturbance, and were sufficiently accessible to allow for frequent measurements. The poor fen sites included distinct open (OPF) and treed (TPF) areas, whereas the moderately-rich fen was treed (TRF), and the saline fen was open (SF). We acknowledge that resource exploration and extraction is widespread in the region, therefore it is virtually impossible to find truly undisturbed sites; however, in all cases, sampling plots were located more than 50 m from any disturbance (e.g. road, cutlines).

The poor fen (Pauciflora fen, see also Wells et al. 2017) is located ~40 km south of Fort McMurray (56° 22.610 N, 111°14.164 W). This fen receives discharge from peatlands upstream and a forested upland surrounding the fen. The peat is 4 m deep on average; however, thickness varies widely ranging from <1 m to >10 m. The site is situated close to a road at its north end, leading to wetter conditions in this portion of the site (Bocking et al. 2017). Plant species include *Sphagnum* spp., *Chamaedaphne calyculata*, *Carex* spp., *Picea mariana* and *Betula pumila.* The poor fen basin is dominated by *Sphagnum* moss species (*Sphagnum fuscum* and *Sphagnum angustifolium*); however, distinct plant communities are observed in the north and south of the basin. TPF occupies the central part of the fen, where water table (WT) is deeper than in the northern and southern parts, which are wetter and dominated by sedges (OPF). Mean pH and corrected electrical conductivity (corrEC; Sjors 1950) were 5.6 and 45 µS cm^{-1} at OPF and 4.9 and 25 µS cm^{-1} at TPF.

The rich fen (TRF; Poplar fen, Elmes et al. 2018) is located ~20 km north of Fort McMurray (56° 56.330 N, 111° 32.934 W). This site was disturbed by cutlines and a pipeline and dirt roads passing through the broader fen boundaries, although the actual study area has not been directly impacted. The site's vegetation is dominated by *Larix laricina, Betula pumila*, *Equisetum fluviatile*, *Smilacina trifolia*, *Carex* spp. and brown mosses, dominated by *Tomenthypnum nitens*. The peat was about 1 to 1.5 m thick, mean pH was 7.0 and corrEC was 330 µS cm^{-1}.

The saline fen (SF) is located 10 km south of Fort McMurray (56° 34.398 N, 111° 16.518 W) and is dominated by *Juncus balticus, Calamagrostis stricta* and *Triglochin maritima*. It is an extremely saline site due to its geological setting that causes discharge of saline groundwater (Wells & Price 2015). The site is surrounded by forested peatland but there are no trees in the study area. Peat depth was 0.75 to 1.5 m, pH was 6.1 and corrEC was 12,000 µS cm^{-1}. Although saline fens are not widespread in the region, they represent a potentially important reference system for peatland reclamation, as construction materials in the post-mining AOSR landscape will include tailings sand, which represents a source of salinity (Simhayov et al. 2018), and is likely to result in saline wetlands (Trites & Bayley 2009).

At each of the four sites, three replicate pairs of sampling plots were established, each pair encompassing a hummock/ridge and a hollow/depression. The plots included collars for greenhouse gas measurement (GHG) where vegetation surveys were also undertaken in 2011 and 2014, pore-water samplers, temperature measurements, dipwells, and were also used for peat sampling in 2014.

Methane flux

All study sites were monitored for CH_4 flux between June 2011 and August 2014. Gas samples were collected weekly to biweekly in 2011 from June 23rd to August 11th, resulting in 6 – 7 measurements at each plot. In 2012 samples were also collected weekly to biweekly between May 9th and August 25th, with an additional measurement in mid-October, resulting in 9 – 15 measurements at each plot. In 2013 and 2014, sampling frequency declined to once every three weeks between May 19th and August 22nd, with 5 – 6 measurements made at each plot.

Methane flux was measured using static closed chambers. At each sampling location, stainless steel collars (60 cm x 60 cm) were installed 10 – 15 cm deep in the peat in early June 2011 and left in place for the remainder of the study. During a measurement, an opaque acrylic chamber (60 cm x 60 cm x 30 high) was placed on the collar and the collar was filled with water to prevent air leakage. A hole in the top of the chamber prevented over-pressurization during chamber placement. Once the chamber was in place, the hole was blocked with a stopper equipped with tubing sealed with a three-way valve. The headspace was mixed with a battery-operated fan and samples were collected using a syringe at 7, 15, 25, and 35 minutes post-chamber closure, and immediately injected into pre-evacuated Exetainers (Labco Ltd. UK). Samples were analyzed for CH_4 content on a Varian 3800 gas chromatograph (GC, Varian 3800) equipped with a flame ionization detector. The GC was calibrated every eight samples and standards were within +/- 10% of known concentrations. Methane flux was estimated from the linear change in CH_4 concentration in the headspace over time after correcting for actual headspace volume and temperature, except when concentration change was less than the precision of the GC, in which case flux was assigned a value of 0. Patterns of concentration change suggesting disturbance during the measurement period (e.g., 7-minute concentration > 5 ppm with concentration falling over the rest of the closure period or rapid increase in the middle of the closure period followed by decline) were removed from the data set. This resulted in a loss of 20% of the data over the entire study period. Therefore, results presented here largely represent diffusive and plant-mediated fluxes as sporadic ebullition events would have been removed from the data set based upon our quality control criteria; steady ebullition, if it occurred, would be included as it would result in a linear increase in concentration change over the chamber closure period.

Pore water CH_4 concentration

Pore water CH_4 concentration was determined from water samples (Strack et al. 2004) collected from samplers installed ~10 cm deeper than the WT position in early June 2011. As WT fluctuated over the study period, some samplers were occasionally above the WT, and could not be sampled at these times. Samplers consisted of 20 cm long segments of 2.5 cm diameter plastic pipe with holes drilled in the middle 10 cm. Samplers were sealed at both ends and covered in synthetic nylon screening (250 μm mesh size) to prevent clogging. Tubing extended from the bottom end of the sampler to the soil surface where it was sealed with a three-way valve. The entire sampler was filled with water and the valve was closed to prevent oxygen leakage to the sampler. To collect a sample, 40 – 60 ml of water was removed from the sampler using a syringe and discarded. Then, 20 ml was collected and equilibrated with 20 ml of ambient air in the syringe by shaking for 5 minutes. The headspace in the syringe was then transferred to a pre-evacuated Exetainer, and CH_4 concentration determined on the GC. Pore water concentration was calculated according to Kampbell and Vandergrift (1998).

Methane production potential

Five composite replicate peat samples per site were collected at OPF, TRF, and SF in July 2014. Each sample consisted of five homogenized cores 15 cm long and 10 cm in diameter, three of them being collected adjacent to collars. Fresh peat was stored in plastic zipper bags at 4°C in the dark until analysed. From each sample, triplicate sub-samples were prepared for incubation by placing 10 g of peat in Erlenmeyer flasks and adding sufficient Milli-Q water to reach saturation. The flasks were prepared in a glovebox flushed with N_2 to ensure anoxic conditions. Flasks were closed with butyl rubber stoppers in the glovebox prior to incubation in the dark at 24.5°C for 6 weeks. Gas was sampled at 0, 1, 2, 5, 7, 14, 21, 28, 38, and 42 days of incubation using a syringe flushed three times with N_2. During sampling, 20 mL of gas from the headspace was transferred into a pre-evacuated Exetainer and replaced with 20 mL of 99% N_2. Methane concentration was measured using a Shimadzu GC (GC-2014) with flame ionization detector.

Gross Ecosystem Productivity (GEP)

Productivity of the vegetation within the collars was estimated under full light conditions using dynamic transparent closed chambers (60 cm x 60 cm x 30 cm) to determine net ecosystem exchange of carbon dioxide. Measurements were carried out over 2 minute closure periods with CO_2 concentration measured every 15 seconds using a portable infrared gas analyzer (EGM-4, PPSystems). Ecosystem respiration was determined by placing an opaque shroud over the chamber and GEP calculated as the difference between net exchange and respiration (see Munir et al. 2015 for further details). As these measurements included only the vegetation within the collar, they represent GEP of the understory vegetation only and do not include trees present at TPF and TRF and therefore underestimate total ecosystem GEP.

Environmental conditions

Water table position was measured in wells installed adjacent to each collar during each flux measurement. Water table was also monitored hourly in one additional well at all representative fen types using a pressure transducer (Solinst levelogger), corrected for barometric pressure (Solinst barologger). This continuous water table record was regressed with manual measurements at each plot and was expressed relative to the soil surface of hummocks at each study site. Soil temperature was measured adjacent to each collar during flux measurements using an Omega HH200A temperature probe at depths from -5 cm to -30 cm with 5 cm depth increments. Meteorological data for the region was compiled from Environment and Climate Change Canada's climate data, using the Fort McMurray AWOS station for 2011 – 2012 and its replacement station Fort McMurray A for 2013 – 2014 (Environment and Climate Change Canada 2018)

Growing season CH_4 emissions

The growing season length was determined according to Robeson (2002) as the number of days between the last freeze in spring and first freeze in autumn. A freeze was defined as a threshold of daily minimum temperatures (Linderholm 2006); in this study, a threshold of 0 °C was used. The average length of the growing season in 2011 – 2014 was calculated to be 113 days, using data recorded at the Fort McMurray CS meteorological station (56°39'04''N, 111°12'48''W, elevation 368.80 m, Environment and Climate Change Canada 2018). CH_4 emission per growing season was estimated by multiplying the mean flux for each site by 113 days.

Data analysis

Statistical analysis, graphs and tables were prepared using R (R Core Team 2017). Temporal and spatial variability in CH_4 flux and dissolved CH_4 concentration in pore water (PW[CH_4]) was analyzed using linear mixed effect (lme) models built with the 'nlme' package (Pinhero et al. 2017). In all cases, the mean for each plot for each study season was used. We chose to use seasonal means as several studies have observed stronger correlations between peatland CH_4 flux and environmental conditions on monthly to seasonal time scales compared to shorter time periods (Treat et al. 2007; Turetsky et al. 2015). Moreover, the aim was to develop understanding of fen CH_4 flux as a functional indicator for restoration/reclamation outcomes. In this case it is the overall CH_4 emission expected under a given set of environmental conditions established in the reclaimed site that is of interest, as opposed to the daily variation in that flux. The model included site, year, microform, and paired interactions between them. Separate lme models were created to investigate the significance of environmental factors on CH_4 flux and PW[CH_4]. These models contained fixed effects of GEP, WT, T5, T20, and categorical (site, year) variables and paired interactions between them. All lme models in the study included plot as a random factor to account for multiple sampling at the same locations. F-values for models and type III (marginal) errors were generated using the 'anova' function. To determine the variables explaining important amounts of variation in CH_4 flux between collars, non-significant controls and their interactions were removed from the model one at a time, in order of the highest p-value (Zuur et al. 2009). The R^2 of each model was calculated using the 'r.squaredGLMM' function in the 'MuMIn' package (Barton 2016). All models were built for measured CH_4 flux and PW[CH_4] and for log transformed data (log CH_4 and log PW[CH_4]). Models were validated for normality and distribution of residuals; only those incorporating log transformed values were used for further spatial-temporal and environmental analysis, and only the results from the models using log transformed data are reported here.

To evaluate which sites and years within sites showed similar CH_4 and PW[CH_4] patterns and which ones varied significantly from each other, Tukey pairwise comparison was conducted using the 'lsmeans' function in the 'lsmeans' package (Lenth 2016).

The significance of single environmental factors in CH_4 flux and PW[CH_4] was likely obfuscated by the presence of significant interactions between model components. When such interaction occurred between a categorical (site or year) and a continuous variable, regression was calculated for each category using the 'lstrends' function in the 'lsmeans' package (Lenth, 2016). The significance of regression was assessed based on the 0.95 confidence interval. Further comparison (e.g., how the relationship between CH_4 flux and water table differs between sites given the significance of water table x site interaction) was conducted using Tukey pairwise comparisons.

Graphs were prepared using the 'ggplot2' (Wickham 2009), 'cowplot' (Wilke 2016) and 'gridExtra' (Auguie 2016) packages. The 'tables' package (Murdoch 2017) was used to calculate descriptive statistics for parameters reported in Table 1. Although the data are not normally distributed, which is common for greenhouse gas fluxes, we have included mean and standard deviation values to enable comparison with other published data on CH_4 pore water concentration and fluxes, which are frequently reported as mean values.

Potential rate of CH_4 production was calculated as a slope of CH_4 concentration increase over time. For both sets of data, linear models were built using the 'lm' function in the 'nlme' package (Pinheiro et al. 2017), followed by analysis of variance ('anova' function) to investigate if the rates of CH_4 production varied significantly between sites. If the relationship was found to be significant ($P < 0.05$), Tukey pairwise comparison was conducted using the 'lsmeans' function in the 'lsmeans' package (Lenth 2016) to identify significant differences between sites.

Results

Environmental conditions

Over the 2011 – 2014 study period, mean monthly growing season (May – October) temperatures were similar to or warmer than the 30-year average (1980-2010; Table 1). It was 1 – 2 °C warmer most months in 2012 and 2014, while 2013 had a particularly warm May and June. With only 213.5 mm of precipitation, 2011 was relatively dry compared to the long-term average of 313.5 mm over the same period (Table 1). Conditions were wetter than average in 2012, receiving over 50 mm more than normal, 2013 and 2014 received just over the average amount of precipitation. However, in both 2012 and 2013, there were several summer months with > 100 mm of precipitation, which led to shallow water table position and frequent inundation at many of the study sites by 2013 (Figure 1).

Water table position at OPF remained close to the surface throughout all study years, falling slightly in August and September (Figure 1). At TPF, water table was generally 10 to 20 cm below the surface, also varying comparatively little between years. Water table at TRF was much more variable, declining from just below the surface in 2011 to deeper than -40 cm by September 2012, followed by largely flooded conditions in 2013 and 2014. At SF, water table also dropped below the surface throughout 2011 and early 2012, but rapidly rebounded with the large amount of precipitation in July 2012. This led to standing water in many locations that persisted into 2013. Water table at SF then gradually declined throughout 2014.

Methane flux

Generally, CH_4 fluxes increased from May into June and July, declining by late August (Figure 2). Measured fluxes in 2011 did not exceed 125 mg m^{-2} d^{-1} at sites TPF and TRF, were close to 0 mg m^{-2} d^{-1} at SF, and were up to ~350 mg m^{-2} d^{-1} at OPF. In 2012, CH_4 fluxes did not increase until the end of June (days 158 – 168). The peak in CH_4 emission overlapped with increased precipitation in July and decreased gradually towards the end of August. In 2013, increased CH_4 emission at both OPF and TPF overlapped with high precipitation in June and July (~170 mm and ~80 mm, respectively, Table 1). Mean CH_4 fluxes in 2013 at SF gradually increased during the growing season from ~25 mg m^{-2} d^{-1} to ~200 mg m^{-2} d^{-1}. Mean CH_4 fluxes in 2014 were lower than in 2013 at all sites except TRF.

While CH_4 flux from hollows was generally higher than from hummocks, the linear mixed effects models built to characterize the spatio-temporal patterns in CH_4 flux and PW[CH_4] indicated that microform type (i.e., hummock vs. hollow) did not explain a significant amount of variation. Thus, microforms were removed from the models. There was a significant interaction between year and site ($F_{9,60} = 5.2$, $p < 0.001$) and CH_4 fluxes varied significantly between sites ($F_{3,20} = 16.4$, $p < 0.001$), but not years ($F_{3,60} = 1.5$, $p = 0.22$). The highest fluxes were measured at OPF in 2012 with mean (standard deviation) of 152.7 (446.6) mg CH_4 m^{-2} d^{-1}, while other sites showed relatively low fluxes that year (Table 2). The lowest flux was observed at SF in 2011, a mean flux of -0.02 (0.7) mg CH_4 m^{-2} d^{-1} (negative values indicate net removal of CH_4 from the atmosphere). The CH_4 fluxes were similar across years at OPF, but were significantly lower in 2011 than in the other years at TPF and SF, and significantly higher in 2013-2014 than 2011-2012 at TRF (Figure 3A). Over all study years, mean CH_4 at OPF was significantly higher than at TRF and SF (99.8, 32.8, and 34.6 mg CH_4 m^{-2} d^{-1}, respectively), while on average 68.3 mg CH_4 m^{-2} d^{-1} was emitted from TPF, which was not significantly different from mean fluxes at OPF and TRF (Table 2, Figure 3A).

Pore water CH_4 concentrations

Concentrations of CH_4 dissolved in pore water generally followed the same pattern as CH_4 fluxes: the highest mean PW[CH_4], over 5 mg CH_4 L^{-1}, were observed at OPF and TPF sites, while TRF and SF had lower values (2.7 (4.0) and 0.9 (1.8) mg CH_4 L^{-1}, respectively, Table 2). Pore water samples were taken less frequently than CH_4 fluxes, thus temporal trends are not as clear as for CH_4 fluxes; however, mean PW[CH_4] increased over the growing season at TPF and TRF in 2011, 2012, and 2013 (Figure 4). A few measurements taken in 2014 at all sites showed much higher values than in other years, but no consistent patterns of increase or decrease over the study period in mean PW[CH_4], aside from decreasing values at the TPF.

In the linear mixed effects model, year ($F_{3,53} = 10.8$, $p < 0.001$), site ($F_{3,20} = 30.0$, $p < 0.001$) and their interaction ($F_{9,53} = 7.4$, $p < 0.001$) explained a significant fraction of spatio-temporal variability of PW[CH_4]. At OPF and TPF, PW[CH_4] was significantly higher in 2014 compared to the early study years (Figure 3B). A similar pattern was observed at TRF; however, at this site PW[CH_4] was also significantly lower in 2012 compared to 2011 and 2013. At SF, concentration was significantly lower in 2011 than all other study years. Over the whole study period, both poor fen sites had significantly higher PW[CH_4] than TRF and SF, which were also significantly different from each other (Figure 3B).

Potential CH_4 production

The mean potential CH_4 production was 0.8 (0.5), 0.1 (0.1) and 4.6 (0.8) µmol CH_4 g peat^{-1} d^{-1} at OPF, SF, and TRF, respectively. Outliers (30.9 and 0.023 µmol CH_4 g peat^{-1} d^{-1} from OPF and TRF, respectively) were rejected before the calculation of the mean values; in both cases, the removed values were two orders of magnitude differ than the other replicates from the same study site. The potential rate of CH_4 production was significantly higher at TRF than OPF and SF ($p = 0.0025$ and $p = 0.0009$, respectively).

Relationship of CH_4 flux and pore water concentrations to environmental conditions

Generally, all parameters, except temperature at -5 and -20 cm depth, varied widely between years and across sites (Table 2). Ground layer GEP was negative at all sites and in all years, meaning that there was net uptake of CO_2 from the atmosphere, which is expected for natural peatlands. SF had the highest CO_2 uptake (from -21.5 to -38.2 g CO_2 m^{-2} d^{-1}), and TRF the lowest (from -10.1 to -17.3 g CO_2 m^{-2} d^{-1}). However, it should be noted that as the overstory production was not included at TRF, total GEP would be underestimated.

Methane flux was significantly related to pore water CH_4 concentration and WT position across all plots (Table 3, Figure 5), but there was also a significant WT-site interaction, indicating that the relationship was site specific. Within each site, WT explained a significant amount of the variation in CH_4 flux, but at OPF, shallower WT resulted in lower CH_4 emission, while the opposite was true at all other study sites; therefore, the slope of the regression at OPF was significantly different than all other sites (OPF vs. SF, $p < 0.001$, OPF vs. TPF, $p = 0.002$, OPF vs.

TRF, p = 0.005). Moreover, the slope of the WT-CH_4 flux relationship was steeper at SF, resulting in a significant difference from the slope at TRF (p = 0.001).

Variation in pore water CH_4 concentration was significantly explained by WT, soil temperature at 20 cm (T20) and interactions between temperature at both 5 cm (T5) and T20 with site. Across the whole data set, PW[CH_4] was higher when WT was shallow (Figure 5) and T20 was cooler. The regression for T5 and PW[CH_4] was only significant at SF, where cooler temperatures resulted in higher concentrations. Considering T20, regressions were significant for all sites except TPF. At OPF and TRF, cooler temperatures resulted in higher concentration, while at SF, warmer temperatures results in higher PW[CH_4].

Discussion

Results from the present study provide a valuable starting point for building a reference database of CH_4 flux and pore water concentrations for fens in the AOSR specifically, and western Canada in general. Following oil sands extraction, the function of restored and constructed peatlands can be assessed in comparison to these reference ecosystems. While measurements of CH_4 flux in western Canada are limited, our results agree well with previous studies in the region (Table 4). OPF had similar carbon emissions to an open poor fen in the Northwest Territories (mean 99 mg CH_4 m^{-2} $year^{-1}$, Liblik et al. 1997), while TRF (mean 32.8 mg CH_4 m^{-2} d^{-1}) had slightly greater mean flux compared to a moderate-rich treed fen in boreal Alberta (mean 25.6 mg CH_4 m^{-2} d^{-1}; Long et al. 2010). Measured PW[CH_4] concentrations also were similar to literature values at sites across North America (1.3 – 6.4 mg L^{-1}; Chasar et al. 2000; Murray et al. 2017a; Strack et al. 2004).

Mean flux from data compilation across boreal and temperate fens was approximately 80 mg CH_4 m^{-2} d^{-1} (Turetsky et al. 2015), with the mean across study years in the present study of 68.3 to 99.8 at poor fen sites and 32.8 mg CH_4 m^{-2} d^{-1} at the treed rich fen. We also compared our fluxes to data estimated on an annual scale by estimating annual emissions of 9.9, 6.8, 3.3 and 3.4 g CH_4-C m^{-2} $year^{-1}$ for OPF, TPF, TRF, and SF, respectively, assuming that the non-growing season flux contributed 15% of the total yearly flux (Saarnio et al. 2007). This assumption is similar to the median non-growing season contribution of 16% determined through data compilation across northern wetlands (Treat et al. 2018), but may also represent an underestimate of non-growing season fluxes given that some studies report up to 47% of CH_4 flux can occur outside the growing season (Treat et al. 2018). When compared to mean fluxes for northern peatlands (7.6 – 15.7 g CH_4-C m^{-2} $year^{-1}$, where lower values represented bogs and higher fens; Abdalla et al. 2016), our values were closer to natural northern bogs than to fens. In the context of peatland reclamation, comparing values with averages from northern fens dispersed at latitudes above 45°N may therefore not be relevant, and highlights the importance of compiling data from local reference ecosystems to assess ecosystem progress following intervention (e.g., Gorham & Rochefort 2003). Furthermore, a wide range of mean annual CH_4 emission has been reported from northern poor fens (e.g., 1.5 g CH_4-C m^{-2} $year^{-1}$ (Godin et al. 2012) and 31 g CH_4-C m^{-2} $year^{-1}$ (Treat et al. 2007)) and rich fens (e.g., 4.1 g CH_4-C m^{-2} $year^{-1}$ (Pelletier et al. 2007) and 154 g CH_4-C m^{-2} $year^{-1}$ (Godin et al. 2012)) suggesting that the type of fen alone cannot be a proxy for the level of emitted CH_4. Instead, if CH_4 flux is used as a functional indicator for restoration or reclamation evaluation, specific environmental conditions prevailing at each site should be considered. Indeed, the factors that significantly controlled CH_4 flux in our research, e.g., WT, and soil temperature, have been previously recognized in other peatland studies (Pelletier et al. 2007; Strack et al. 2004; Treat et al. 2007; Whalen 2005; Rinne et al. 2018).

Water table is a widely reported control on peatland CH_4 flux, and our data further supports this observation (Table 3; Figure 4). Interestingly, WT affected CH_4 fluxes differently at OPF than the other three sites (Table 4). Generally, CH_4 emission increases at sites with shallower WT (Abdalla et al. 2016), as deep WT position increases the size of the oxic zone and the likelihood that CH_4 will be oxidized to CO_2 before it reaches the peat surface (Whalen 2005). This pattern was observed at TPF, TRF and SF. At OPF, CH_4 emission declined with increasing WT. While this site was characterized by more stable (Wells et al. 2017) and shallower WT than the other sites, it also had a complete cover of *Sphagnum*, which are known to host symbiotic CH_4 oxidizing bacteria (Kip et al. 2010). These communities can support high rates of CH_4 oxidation even in submerged conditions (Parmentier et al. 2011) when CH_4-rich pore water comes in contact with the living part of *Sphagnum.* This CH_4 emission pattern could also be caused by reduced vegetation productivity under inundated conditions (Strack et al. 2004) that would reduce substrate availability for methanogens and thus limit CH_4 production. These wet conditions at OPF, and other sites in some study years, may have also contributed to the lack of a statistically significant effect of microform type in spatial variability of CH_4 fluxes. Although fluxes > 50 mg CH_4 m^{-2} d^{-1} were observed in hollows during the dry growing season of 2011, large fluxes were measured at both hummocks and hollows in wetter years, with hummocks emitting large amounts of CH_4 at the OPF. At this site, hummocks were often only a few cm higher than hollows, resulting in limited WT differences between microforms.

Differences in the slope of the CH_4 flux-WT and PW[CH_4]-T20 relationship between sites may also be driven by chemical differences between the study fens. The poor fen sites were more acidic than TRF and SF, where pH of 6 – 7 is optimal for methanogens (Blodau 2002). This suggests that under saturated conditions, CH_4 emissions should be higher from TRF and SF. Although fluxes at all sites were similar when WT was near the surface (Figure 4), high CH_4 production potential at TRF does suggest more favourable substrate availability and/or chemical conditions than the other sites. CH_4 production and flux can also be limited by the availability of terminal electron acceptors (TEAs), such as nitrate, iron and sulphate (Lai 2009), that are more likely to be present in higher concentrations at TRF and SF than the poor fen sites. In fact, Murray et al. (2017a) found a strong negative relationship between sulphur availability and CH_4 flux and PW[CH_4] across OPF and SF. The relatively steep slope of the CH_4 flux-WT relationship at SF may reflect the importance of sulphate as a control on fluxes at this site; once WT drops below the surface, reduced sulphur could be oxidized to sulphate, quickly limiting CH_4 production. The presence of sulphate could also account for the low CH_4 production potential measured at SF. Given that studies of fen reclamation projects in the AOSR report high sulphate concentrations (Nwaishi et al. 2015; Murray et al. 2017a; Clark et al. 2019), measuring pore water chemistry, specifically for TEAs will be important when using CH_4 as a functional indicator of reclamation outcomes.

Substrate availability for methanogens is also a control on CH_4 production and hence PW[CH_4] and CH_4 flux (Lai 2009). Some previous studies have found a significant positive correlation between plant productivity and CH_4 flux (e.g., Whiting & Chanton 1993; Bellisario et al. 1999), but GEP was not a significant predictor of variation in mean flux between measured plots in the present study. This could be due to the multifaceted role that plants play in CH_4 flux, both increasing production and transport (Waddington et al. 1996, Strack et al. 2017), while also enhancing oxidation (Popp et al. 2000, Sutton-Grier & Megonigal 2011). For example, while aerenchymatous plants have been shown to increase the transport of CH_4 from the soil to the atmosphere, in some cases, CH_4 emissions can be reduced due to greater oxidation in the rooting zone (Bhullar et al. 2013). When WT is favourable for CH_4 production, other environmental variables such as soil temperature and GEP can drive spatial and temporal variation in CH_4 flux

(Strack et al. 2004). Since WT varied widely between sites and years, it may be that this acted as an overriding control compared to GEP. It should be noted that measured GEP did not include the overstory productivity, which is likely substantial at TPF and TRF (Murray et al. 2017b). Including an estimate of total ecosystem GEP potentially would further explain differences in CH_4 flux between the studied fens. Further, some studies indicate that trees can vent CH_4 in wetlands (Gauci et al. 2010; Pangala et al. 2013), suggesting that the role that trees play in total CH_4 emissions in western Canadian peatlands also requires further investigation.

One of the most striking patterns observed was the large inter-annual variability in CH_4 fluxes. In the year following the data collection period presented here, Murray et al. (2017a) measured CH_4 emission at some of these same sites and report a flux of 23.9 mg CH_4 m^{-2} d^{-1} at OPF and 4.4 mg CH_4 m^{-2} d^{-1} at SF. Both, if used to recalculate mean CH_4 flux for years 2011 – 2015, would reduce the flux by ~20% at OPF and ~10% at SF. This inter-annual variability can be captured only with long-term studies, illustrating their importance to the development of robust reference datasets. Wells et al. (2017) also underlined the importance of long-term studies on peatland hydrology (main driver of CH_4 fluxes) in relation to long-term climate variability.

The studied fens also varied in their CH_4 flux response to inter-annual variation in summer precipitation, a pattern likely driven by local hydrogeologic setting. Hydrological studies at TRF indicate that it has a longer "memory" than the other study fens, with WT position carrying over from available moisture in previous years (Elmes et al. 2018). For example, while most sites exhibited wetter conditions in 2012 than 2011, TRF remained dry and consequently had lower CH_4 emissions than the other sites in this year. Furthermore, availability of TEAs at TRF, particularly after long periods of low water table may have also contributed to lower CH_4 emissions.

Compared to inter-annual variation in flux, PW[CH_4] was less variable; values remained relatively similar over the first 3 years of study, but increased in 2014 at TPF, OPF, and TRF. This may indicate that PW[CH_4] is a useful functional indicator for reclamation outcomes as it appears to be more stable in response to inter-annual hydrologic variations. However, there appears to be some decoupling between CH_4 emissions and PW[CH_4] (Figure 3). The continuing increase in PW[CH_4] in 2014 while emissions declined compared to the previous year may reflect the fact that mean residence time of CH_4 in peat can be months to years (Liblik et al. 1997; Strack and Waddington 2008). Therefore, the dissolved CH_4 pool in 2014 potentially reflects the favourable CH_4 production conditions in 2013 when all sites had shallow water tables.

Conclusions

We measured CH_4 production potential and CH_4 flux and pore water CH_4 concentration (PW[CH_4]) over four growing seasons, across a variety of fen types in the Athabasca Oil Sands Region (AOSR) in order to contribute to the development of a reference fen dataset that can be used for functional evaluation of fen restoration and reclamation projects in the region. Mean flux over the study period was 32.8 to 99.8 mg CH_4 m^{-2} d^{-1}, while mean PW[CH4] was 0.9 to 5.5 mg CH_4 L^{-1}. While differences within and between fens and across study years were largely driven by WT position, this relationship varied between sites likely linked to local plant community and chemistry. The high level of variability in CH_4 flux between study years indicates the importance of multi-year studies of CH_4 flux, not only for developing reference datasets, but also for measuring ecosystem function in reclamation projects. In our study, PW[CH_4] was generally linked to fluxes but showed less spatial and temporal variability. Thus, PW[CH_4] may be useful independently as a functional indicator for reclamation. Our study generated a multi-year baseline for fens in the AOSR, which could be used by the energy industry in the context of post-mining management of disturbed or reclaimed fens in this region. Ideally, similar baseline CH_4 emission and PW[CH_4] databases, including high frequency of measurements, would need to be constructed for other regions.

Acknowledgments

Funding for this project was provided by a Natural Sciences and Engineering Research Council of Canada (NSERC) Collaborative Research and Development (CRD) grant (#418557) to JP and MS co-funded by Suncor Energy Inc., Imperial Oil Resources Limited, and Shell Canada Energy. The authors would like to acknowledge Canada’s Oil Sands Innovation Alliance (COSIA) for its support of this project. Funders had no role in study design, data analysis or decision to publish the results. We thank Peter Macleod and Mendel Perkins for help collecting field data. Suggestions from anonymous reviewers improved the manuscript.

Data availability

All data is available from the corresponding author by request.

Author contributions

MS, MSM and RA conceived the field study and implemented the design. JP, VD, RA and FN conceived the incubation study. VD, RA, FN, MSM and MS collected data and contributed to data analysis. AB completed final statistical analysis and wrote the first draft of the paper. All authors edited the final manuscript.

References

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

Andersen R, Chapman SJ, Artz R (2013) Microbial communities in natural and disturbed peatlands: A review. Soil Biology and Biochemistry 57: 979-94.

Auguie B (2016) gridExtra: Miscellaneous Functions for "Grid" Graphics. R package version 2.2.1. URL: https://CRAN.R-project.org/package=gridExtra.

Barton K (2016) MuMIn: Multi-Model Inference. R package version 1.15.6. URL: https://CRAN.R-project.org/package=MuMIn.

Bellisario LM, Bubier JL, Moore TR, Chanton JP (1999) Controls on CH_4 emissions from a northern peatland. Global Biogeochemical Cycles 13: 81-91.

Bhullar GS, Edwards PJ, Venterink HO (2013) Variation in plant-mediated methane transport and its importance for methane emission from intact wetland peat mesocosms. Journal of Plant Ecology 6: 298-304.

Blodau C (2002) Carbon cycling in peatlands A review of processes and controls. Environmental Reviews 10: 111-134.

Bocking E, Cooper DJ, Price J (2017) Using tree ring analysis to determine impacts of a road on a boreal peatland. Forest Ecoloogy and Management 404: 24-30.

Bridgham SD, Cadillo-Quiroz H, Keller JK, Zhuang Q (2013) Methane emissions from wetlands: Biogeochemical, microbial, and modeling perspectives from local to global scales. Global Change Biology 19: 1325-1346.

Chasar LS, Chanton JP, Glaser PH, Siegel DI (2000) Methane concentration and stable isotope distribution as evidence of rhizospheric processes: Comparison of a fen and bog in the glacial lake Agassiz peatland complex. Annals of Botany 86: 655-663.

Clark MG, Humphrey E.R, Carey SK (2019) Low methane emissions from a boreal wetland constructed on oil sand mine tailings, Biogeosciences Discussions, https://doi.org/10.5194/bg-2019-271.

Clymo RS, Bryant C (2008) Diffusion and mass flow of dissolved carbon dioxide, methane, and dissolved organic carbon in a 7-m deep raised peat bog. Geochimica et Cosmochimica Acta 72: 2048-2066.

Couwenberg J, Fritz C (2012) Towards developing IPCC methane 'emission factors' for peatlands (organic soils). Mires and Peat 10: 03.

Daly C, Price J, Rezanezhad F, Pouliot R, Rochefort L, Graf MD (2012) Initiatives in oil sand reclamation: considerations for building a fen peatland in post mined oil sands landscape. In: Vitt DH, Bhatti J (eds) Restoration and Reclamation of Boreal Ecosystems, Cambridge University Press, New York.

Elmes MC, Thompson DK, Sherwood JH, Price JS (2018) Hydrometeorological conditions preceding wildfire, and the subsequent burning of a fen watershed in Fort McMurray, Alberta, Canada. Natural Hazards and Earth System Sciences 18: 157-170.

Environment and Climate Change Canada (2018) Historical Climate Data, [cited 2018 Jul 19]. Available from: http://climate.weather.gc.ca/.

Environment and Parks (2017) Reclamation Criteria for Wellsites and Associated Facilities for Peatlands, March, 2017, Edmonton, Alberta, pp 142.

Estop-Aragonés C, Knorr K-H, Blodau C (2013) Belowground in situ redox dynamics and methanogenesis recovery in a degraded fen during dry-wet cycles and flooding, Biogeosciences 10: 421-436.

Gauci V. Gowing DJG, Hornibrook ERC, Davis JM, Dise NB (2010) Woody stem methane emission in mature wetland alder trees. Atmospheric Environment 44: 2157-2160.

Glaser PH, Chanton JP, Morin P, Rosenberry DO, Siegel DI, Ruud O, Chasar LI, Reeve AS (2004) Surface deformations as indicators of deep ebullition fluxes in a large northern peatland, Global Biogeochemical Cycles 18: GB1003.

Godin A, McLaughlin JW, Webster KL, Packalen M, Basiliko N (2012) Methane and methanogen community dynamics across a boreal peatland nutrient gradient. Soil Biology and Biochemistry 48: 96-105.

Gorham E, Rochefort L (2003) Peatland restoration: A brief assessment with special reference to sphagnum bogs. Wetlands Ecology and Management 11: 109-119.

Government of Alberta (2018) [cited 2018 Jul 19]. Available from: https://www.energy.alberta.ca/OS/AOS/Pages/FAS.aspx#Environment.

Gupta V, Smemo KA, Yavitt JB, Fowle D, Branfireun B, Basiliko N (2013) Stable isotopes reveal widespread anaerobic methane oxidation across latitude and peatland type. Environmental Science and Technology 47: 8273-8279.

Kampbell DH, Vandegrift SA (1998) Analysis of dissolved methane, ethane, and ethylene in ground water by a standard gas chromatographic technique. Journal of Chromatographic Science 36: 253-256.

Kip N, van Winden JF, Pan Y, Bodrossy L, Reichart G, Smolders AJP, Jetten MSM, Damste JSS, Op den Camp HJM (2010) Global prevalence of methane oxidation by symbiotic bacteria in peat-moss ecosystems. Nature Geoscience 3: 617-621.

Lai D (2009) Methane dynamics in northern peatlands: A review. Pedosphere 19: 409-421.

Lenth RV (2016) Least-Squares Means: The R Package lsmeans. Journal of Statistical Software 69: 1-33.

Liblik LK, Moore TR, Bubier JL, Robinson SD (1997) Methane emissions from wetlands in the zone of discontinuous permafrost: Fort Simpson, Northwest Territories, Canada. Global Biogeochemical Cycles 11: 485-494.

Linderholm HW (2006) Growing season changes in the last century. Agricultural and Forestry Meteorology 137: 1-14.

Long KD, Flanagan LB, Tiebo C (2010) Diurnal and seasonal variation in methane emissions in a northern Canadian peatland measured by eddy covariance. Global Change Biology 16: 2420-2435.

Madigan MT (2009) Brock biology of microorganisms.12th ed. San Francisco (CA), Pearson/Benjamin Cummings, 1061 pp.

Minderlein S, Blodau C (2010) Humic-rich peat extracts inhibit sulfate reduction, methanogenesis, and anaerobic respiration but not acetogenesis in peat soils of a temperate bog. Soil Biology and Biochemistry 42: 2078-2086.

Munir TM, Perkins M, Kaing E, Strack M (2015) Carbon dioxide flux and net primary production of a boreal treed bog: Responses to warming and water-table-lowering simulations of climate change. Biogeosciences 12, 1091-1111.

Murdoch D (2017) tables: Formula-Driven Table Generation. R package version 0.8.3. URL: https://CRAN.R-project.org/package=tables.

Murray KR, Barlow N, Strack M (2017a) Methane emissions dynamics from a constructed fen and reference sites in the Athabasca oil sands region, Alberta. Science of the Total Environment 583: 369-381.

Murray KR, Borkenhagen AK, Cooper DJ, Strack M (2017b) Growing season carbon gas exchange from peatlands used as a source of vegetation donor material for restoration**.** Wetlands Ecology and Management 25: 501-515.

Nwaishi F, Petrone R, Price J, Andersen R (2015)Towards developing a functional-based approach for constructed peatlands evaluation in the Alberta oil sands region, Canada. Wetlands 35: 211-225.

Pangala SR, Moore S, Hornibrook ERC, Gauci V (2013) Trees are major conduits for methane egress from tropical forested wetlands. New Phytologist 197: 524-531.

Parmentier FJW, van Huissteden J, Kip N, Op den Camp HJM, Jetten MSM, Maximov TC, Dolman AJ (2011) The role of endophytic methane-oxidizing bacteria in submerged *Sphagnum* in determining methane emissions of Northeastern Siberian tundra. Biogeosciences 8: 1267-1278.

Pelletier L, Moore TR, Roulet NT, Garneau M, Beaulieu-Audy V (2007) Methane fluxes from three peatlands in the la Grande Rivière watershed, James Bay lowland, Canada. Journal of Geophysical Research 112: 1-12.

Pinheiro J, Bates D, DebRoy S, Sarkar D, R Core Team (2017) nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-131, URL: https://CRAN.R-project.org/package=nlme.

Popp TJ, Chanton JP, Whiting GJ, Grant N (2000) Evaluation of methane oxidation in the rhizosphere of a Carex dominated fen in north central Alberta, Canada. Biogeochemistry 51: 259-281.

Province of Alberta (2018) Environmental Protection and Enhancement Act, Conservation and Reclamation Regulation. Alberta Queen's Printer, Edmonton, AB.

R Core Team (2017) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: https://www.R-project.org/.

Rinne J, Tuittila E-S, Peltola O, Li X, Raivonen M, Alekseychik P, Haapanala S, Pihlatie M, Aurela M, Mammarella I, Vesala T (2018) Temporal variation of ecosystem scale methane emission from a boreal fen in relation to temperature, water table position, and carbon dioxide fluxes. Global Biogeochemical Cycles 32: 1087–1106.

Robeson SM (2002) Increasing growing-season length in Illinois during the 20th century. Climatic Change 52: 219-238.

Rosenberry DO, Glaser PH, Siegel DI (2006) The hydrology of northern peatlands as affected by biogenic gas: Current developments and research needs. Hydrological Processes 20: 3601-3610.

Saarnio S, Morero M, Shurpali NJ, Tuittila , Mäkilä M, Alm J (2007) Annual CO2 and CH4 fluxes of pristine boreal mires as a background for the lifecycle analyses of peat energy. Boreal Environment Research 12: 101-113.

Simhayov RB, Weber TKD, Price JS (2018) Saturated and unsaturated chemical non-equilibrium salt transport in peat from a constructed fen. Soil 4: 63-81.

Sjörs H (1950) On the relation between vegetation and electrolytes in north Swedish mire waters. Oikos 2: 241-258.

Strack M, Mwakanyamale K, Hassanpour Fard G, Bird M, Bérubé V, Rochefort L (2017) Effect of plant functional type on methane dynamics in a restored minerotrophic peatland. Plant and Soil 410: 1-16.

Strack M, Waddington JM (2008) Spatiotemporal variability in peatland subsurface methane dynamics. Journal of Geophysical Research 113: G02010, doi: 10.1029/2007JG000472.

Strack M, Waddington JM, Tuittila E-S (2004) Effect of water table drawdown on northern peatland methane dynamics: Implications for climate change. Global Biogeochemical Cycles 18: GB4003, doi: 10.1029/2003GB002209.

Sundh I, Mikkelä C, Nilsson M, Svensson BH (1995) Potential aerobic methane oxidation in a sphagnum-dominated peatland--controlling factors and relation to methane emission. Soil Biology and Biochemistry 27: 829-837.

Sutton-Grier AE, Megonigal JP (2011) Plant species traits regulate methane production in freshwater wetland soils. Soil Biology & Biochemistry 43: 413-420.

Treat CC, Bubier JL, Varner RK, Crill PM (2007) Timescale dependence of environmental and plant-mediated controls on CH4 flux in a temperate fen. Journal of Geophysical Research 112: G01014, doi: 10.1029/2006JG000210.

Treat CC, Bloom AA, Marushchak ME (2018) Nongrowing season methane emissions – a significant component of annual emissions across northern ecosystems. Global Change Biology 24: 3331-3343.

Trites M, Bayley SE (2009) Vegetation communities in continental boreal wetlands along a salinity gradient: Implications for oil sands mining reclamation, Aquatic Botany 91: 27-39.

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

Vitt DH, Halsey LA, Bauer IE, Campbell C (2000) Spatial and temporal trends in carbon storage of peatlands of continental western Canada through the Holocene. Canadian Journal of Earth Sciences 37: 683-693.

Vitt D, Halsey L, Thormann M, Martin T (1996) Peatland inventory of Alberta. Phase 1: Overview of peatland resources in the natural regions and subregions of the province, University of Alberta, Edmonton, AB, Canada.

Waddington JM, Roulet NT, Swanson RV (1996)Water table control of CH_4 emission enhancement by vascular plants in boreal peatlands. Journal of Geophysical Research 101: 22775-22785.

Wells C, Ketcheson S. Price J (2017) Hydrology of a wetland-dominated headwater basin in the boreal plain, Alberta, Canada. Journal of Hydrology 547: 168-183.

Wells CM, Price JS (2015) A hydrologic assessment of a saline-spring fen in the Athabasca oil sands region, Alberta, Canada – a potential analogue for oil sands reclamation. Hydrological Processes 29: 4533-4548.

Whalen SC (2005) Biochemistry of methane exchange between natural wetlands and the atmosphere. Environmental Engineering Science 22: 73-94.

Whiting GJ, Chanton JP (1993) Primary production control of methane emission for wetlands. Nature 364: 794-795.

Wickham H (2009) ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York.

Wilke CO (2016) cowplot: Streamlined Plot Theme and Plot Annotations for 'ggplot2'. R package version 0.7.0. URL: https://CRAN.R-project.org/package=cowplot.

Zuur AF, Ieno EN, Walker NJ, Saveliev AA, Smith GM (2009) Mixed Effects Models and Extensions with R. Springer, New York.

Tables

Table 1. Mean monthly precipitation (P, mm) and temperature (T, °C)

Month	Year 2011		2012		2013		2014		Normal[b]	
	P	T	P	T	P	T	P	T	P	T
May	23.5	11.0	19.6	10.6	4.7	12.7	82.9	7.9	36.5	9.9
June	51.0	14.6	38.2	15.3	165.9	16.1	69.7	15.3	73.3	14.6
July	53.0	17.2	130.8	19.2	87.1	16.9	55.7	19.0	80.7	17.1
August	61.5	16.0	16.4	16.8	3.9	17.6	36.3	17.0	57.1	15.4
September	21.0	12.5	116.9	12.1	35.0	13.4	62.8	9.4	39.7	9.5
October	3.5	6.2	42.8	-0.2	21.1	3.6	30.3	5.2	26.2	2.3
Growing season[a]	213.5	12.9	364.7	12.3	317.7	13.4	337.7	12.3	313.5	11.5

a. total P and average T for May to October

b. normal average for years 1981 – 2010

Table 2. Descriptive statistics for measured variables

		WT (cm)			T5[a] (°C)			T20[b] (°C)			GEP[c] (g CO_2 m^{-2} d^{-1})			CH_4 flux (mg CH_4 m^{-2} d^{-1})			PW[CH_4][d] (mg L^{-1})		
Site	Year	n	mean	sd	n	mean	sd	n	mean	sd	n	mean	sd	n	mean	sd	n	mean	sd
OPF	2011	36	-2.3	6.7	38	17.2	3.9	38	15.2	2.8	24	-23.0	10.6	31	50.0	75.6	46	4.5	1.7
	2012	45	-7.5	4.7	38	16.1	3.5	38	12.7	5.1	48	-18.2	16.1	42	152.7	446.6	32	4.5	2.0
	2013	42	-3.9	7.9	42	16.2	3.9	42	13.8	3.5	40	-16.0	10.7	29	104.1	103.7	26	4.4	2.0
	2014	29	-1.5	4.6	29	16.2	2.9	29	13.0	4.8	30	-19.3	17.9	24	66.3	97.9	12	12.8	6.8
TPF	2011	36	-9.9	12.0	36	16.4	3.2	36	13.8	2.1	37	-21.6	22.5	28	13.1	21.9	48	3.9	1.8
	2012	74	-11.6	8.1	52	15.3	3.7	52	12.0	6.1	54	-17.7	9.5	49	38.0	54.4	37	4.8	2.5
	2013	46	-3.0	8.7	46	16.9	3.7	46	14.3	3.1	36	-20.8	10.7	38	140.0	171.6	30	5.0	2.4
	2014	30	-3.4	7.1	30	17.4	2.0	30	13.4	4.2	30	-25.1	18.5	26	79.9	127.1	15	13.5	10.5
TRF	2011	28	-7.9	11.2	28	16.5	3.2	28	12.5	2.4	51	-17.3	20.0	32	20.2	32.8	29	1.9	1.7
	2012	87	-23.6	17.8	63	13.4	5.4	63	10.4	6.5	55	-15.5	16.1	65	7.7	12.3	31	1.0	3.6
	2013	36	2.2	10.8	36	13.6	3.3	36	11.3	4.2	35	-10.1	10.7	29	66.3	105.2	29	2.5	2.6
	2014	24	1.4	7.2	28	13.6	3.1	28	10.4	4.5	29	-13.8	12.0	26	73.8	74.0	14	9.1	4.8
SF	2011	13	-13.5	5.1	42	19.1	3.2	42	17.2	2.2	50	-38.2	16.0	37	0.0	0.7	37	0.2	0.1
	2012	66	-3.6	11.4	55	16.5	4.6	55	15.0	5.4	55	-29.5	20.6	55	11.1	19.9	24	0.2	0.3
	2013	38	1.0	8.3	42	16.7	2.4	42	16.0	2.2	30	-23.6	14.2	34	117.6	159.1	34	2.0	2.2
	2014	30	-6.3	10.1	30	16.9	4.8	30	15.7	4.4	32	-21.7	16.9	19	21.8	41.4	8	2.3	3.6
OPF	2011 - 2014	152	-4.1	6.6	147	16.4	3.6	147	13.7	4.2	142	-18.6	14.4	126	99.8	269.7	116	5.3	3.7
TPF	2011 - 2014	186	-7.8	9.7	164	16.4	3.4	164	13.3	4.4	157	-20.7	15.6	141	68.3	118.9	130	5.5	5.0
TRF	2011 - 2014	175	-12.4	18.5	155	14.1	4.3	155	11.0	5.1	170	-14.6	16.0	152	32.8	63.7	103	2.8	4.0
SF	2011 - 2014	147	-3.8	10.6	169	17.3	4.0	169	15.9	4.0	167	-29.5	18.5	145	34.6	91.3	103	0.9	1.9

a. T5 = soil temperature at 5 cm depth
b. T20 = soil temperature at 20 cm depth
c. GEP = gross ecosystem photosynthesis
d. PW[CH_4] = CH_4 concentration in pore water

Table 3. Results from the linear mixed effect models for CH_4 flux and pore water CH_4 concentration

Factor[a]	F[b]	p-value[b]
$\log_{10}$[CH_4 flux (mg m^{-2} d^{-1})]; $R^2_{GLMM} = 0.66$[c]		
Site	$F_{3,20} = 0.4$	0.77
WT	$F_{1,60} = 7.5$	0.008
$\log_{10}$(PW[CH_4])	$F_{1,60} = 10.4$	0.002
Site x WT	$F_{3,60} = 10.6$	<0.0001
Intercept	$F_{1,60} = 24.8$	<0.0001
$\log_{10}$[pore water CH_4 concentration (mg L^{-1})]; $R^2_{GLMM} = 0.74$[c]		
Site	$F_{3,20} = 2.1$	0.13
WT	$F_{1,56} = 24.9$	<0.0001
T5	$F_{1,56} = 0.10$	0.76
T20	$F_{1,56} = 6.41$	0.014
Site x T5	$F_{3,56} = 4.9$	0.004
Site x T20	$F_{3,56} = 6.08$	0.001
Intercept	$F_{1,56} = 10.4$	0.002

a. Factors considered included water table (WT), soil temperature at 5 cm (T5), soil temperature at 20 cm (T20), ground layer gross ecosystem photosynthesis (GEP), pore water CH_4 concentration (PW[CH_4])

b. Results from the linear mixed effect models. All models included year, site, WT, T5, T20, GEP and two-way interaction with site, as fixed factors and plot as a random factor. The flux model also included $\log_{10}$(PW[CH_4]) and its interaction with site. Factors were removed sequentially starting with the highest p-value.

c. Reported R^2_{GLMM} is the marginal value, representing variation described by fixed factors only.

Table 4. Compilation of literature values for western Canadian fen methane flux

Fen type	**Location**	**No. seasons measured, months covered**	**Mean CH_4 flux** (std. error) mg CH_4 m^{-2} d^{-1}	**Reference**
Open poor fen	56.38, 111.24	4, May-Oct	99.8 (269.7)	This study
Open poor fen	61.8, 121.4	1, Jul-Aug	162.5 (94.3)[c]	Liblik et al. 1997
Open poor fen	56.67, 113.53	1, May-Oct	23.5 (18.7)	Malhotra 2010
Open rich fen (graminoid)	61.8, 121.4	1, Jul-Aug	63.5 (36.2)[c]	Liblik et al. 1997
Open rich fen (low shrub)	61.8, 121.4	1, Jul-Aug	19.0 (13.9)[c]	Liblik et al. 1997
Open rich fen	55.85, 107.68	2, Jul-Oct	0.2 (0.1)	Turetsky et al. 2002
Open fen	54, 113	3, May-Oct	56.2 (11.2)[a]	Whiting & Chanton, 2001
Open saline fen	56.57, 111.28	4, May-Oct	91.3 (103)	This study
Patterned rich fen	53.77, 104.60	1, Apr-Oct	121.8 (85.9)	Rask et al. 2002
Patterned rich fen	53.95, 105.95	2, Apr-Oct	194.4[d]	Sukyer et al. 1996
Treed poor fen	56.38, 111.24	4, May-Oct	68.3 (118.9)	This study
Treed moderate rich fen	56.94, 111.55	4, May-Oct	63.7 (103)	This study
Treed moderate rich fen	56.40, 116.89	2; May-Sep	30.4 (41.6)	Strack et al. 2018
Treed moderate rich fen	54.82, 113.52	1, May-Sep	25.4[b]	Long et al. 2010
Treed moderate rich fen	56.94, 111.55	2; May-Aug	34.3 (18.4)	Murray et al. 2017b
Treed rich fen	61.8, 121.4	1, Jul-Aug	3.7 (3.8)[c]	Liblik et al. 1997

a. Mean and standard error calculated from mean flux for each year presented in the publication
b. Daily mean calculated from seasonal total measured by eddy covariance, divided by number of days in the study period. No uncertainty in the estimate was presented in the original publication
c. Value in brackets represents standard deviation as reported in the original publication
d. Measured with eddy covariance, no uncertainty presented in original publication

Figure captions

Figure 1: Daily mean water table position during the growing season over each study year. Water table is presented relative to the surface of hummocks at each site, where negative values indicate depth below the surface. When available, data from May 1 (day of year 121) to October 31 (day 304) is presented. Sensor failure in early 2013 at TRF resulted in a large data gap; manual measurements are plotted as points during this period.

Figure 2: Measured CH_4 fluxes over the growing seasons 2011 – 2014 at each site. Sites are open poor fen (OPF), treed poor fen (TPF), treed rich fen (TRF) and open saline fen (SF). DOY is day of year. Note the log scale used on the y-axis. In order to apply the log scale to negative fluxes, a value of 10 was added to each flux; therefore, 10 represents a zero flux.

Figure 3. Mean log CH_4 fluxes (A) and mean log PW[CH_4] (B) at each site separated by years. Significantly different sites are labelled with no capital letter in common. Significantly different years are marked with no lower-case letter in common and should be compared within one site. The upper and lower edges of the boxes show 25th and 75th percentile, respectively, and the median (50th percentile) is located between them. The extent of the upper and lower whiskers away from the box to the most extreme data is no longer than 1.5 times the length of the box, thus outliers are present in the graphs.

Figure 4. Measured pore water CH_4 concentration over the growing seasons of 2011-2014 at each site. Sites are open poor fen (OPF), treed poor fen (TPF), treed rich fen (TRF) and open saline fen (SF). DOY is day of year.

Figure 5. Water table relationship to CH_4 fluxes (A) and PW[CH_4] (B). A: $y=0.04x + 1.6415$, $r^2=0.2775$; B: $y=0.0184x + 0.6799$, $r^2=0.175$.

Figure 1

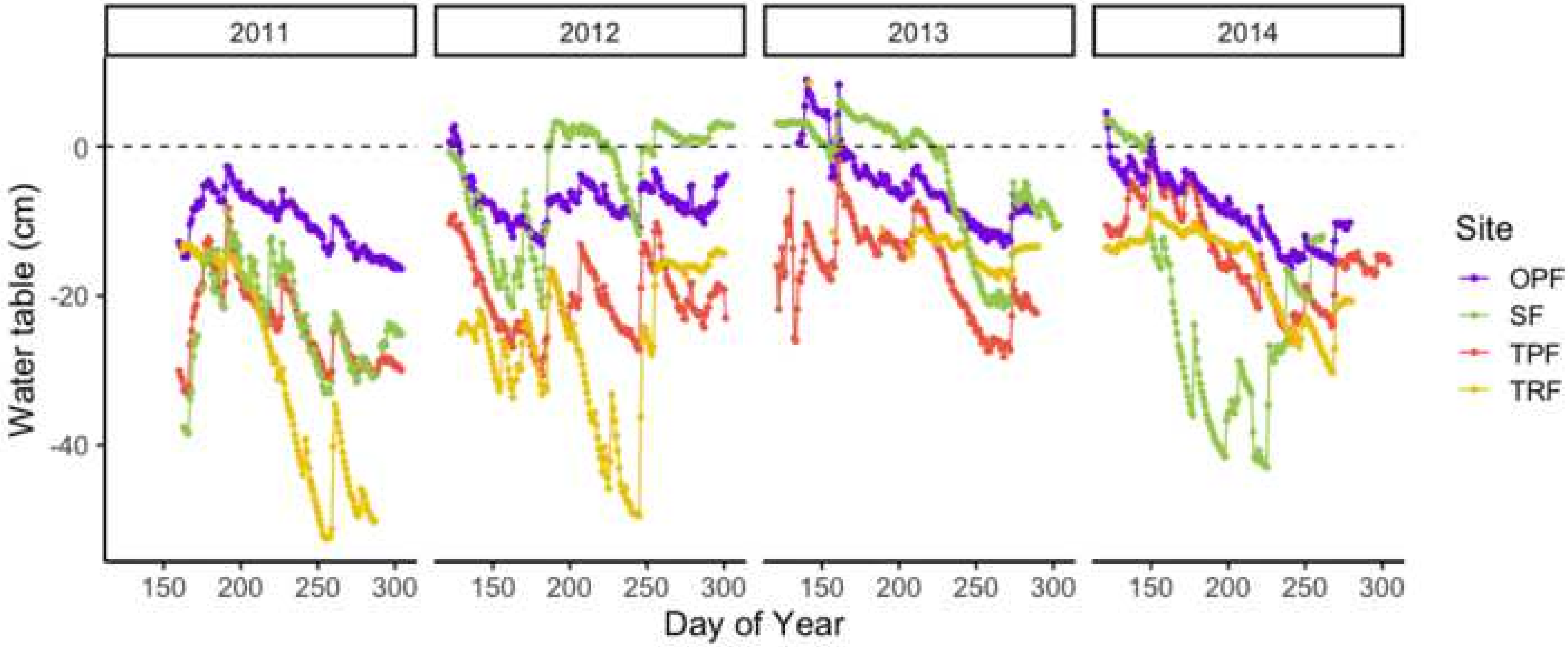

2011
2012
2013
2014
Water table (cm)
0
-20
-40
150
200
250
300
Day of Year
Site
OPF
SF
TPF
TRF

Figure 2

Click here to access/download;Figure;Figure 2.tiff

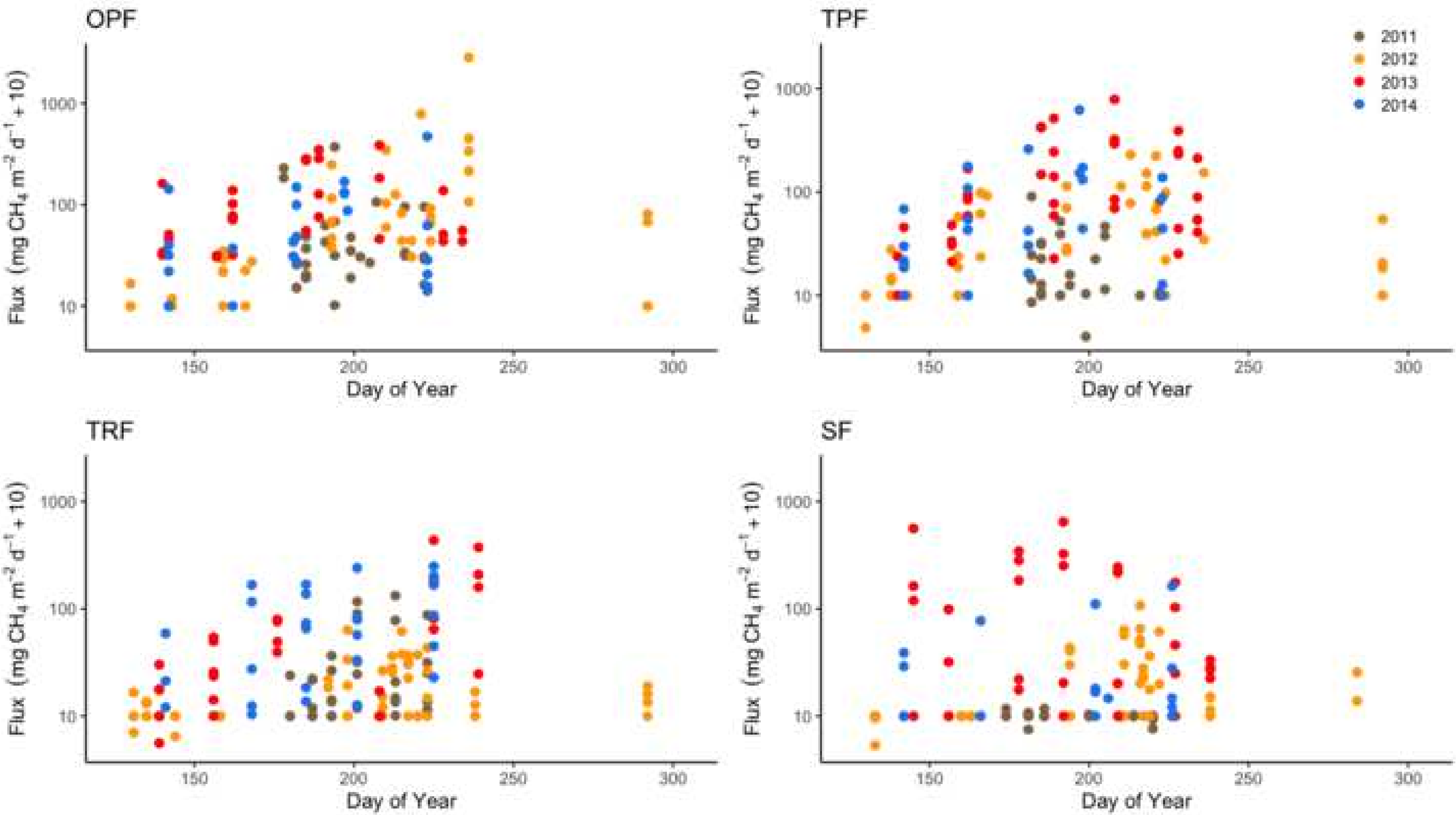


Figure 3

Click here to access/download;Figure;Figure 3.eps

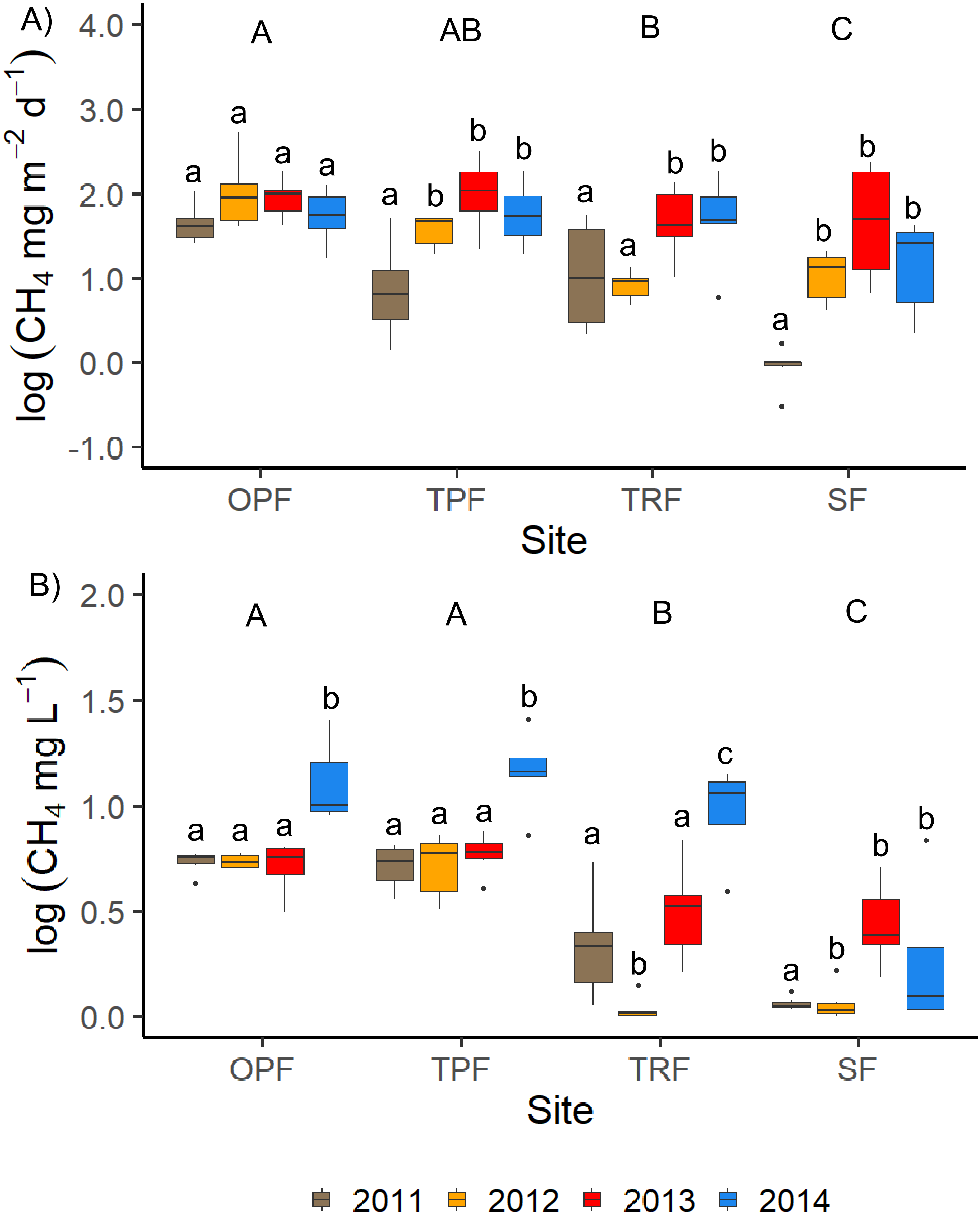


Figure 4

Click here to access/download;Figure;Figure 4.tiff

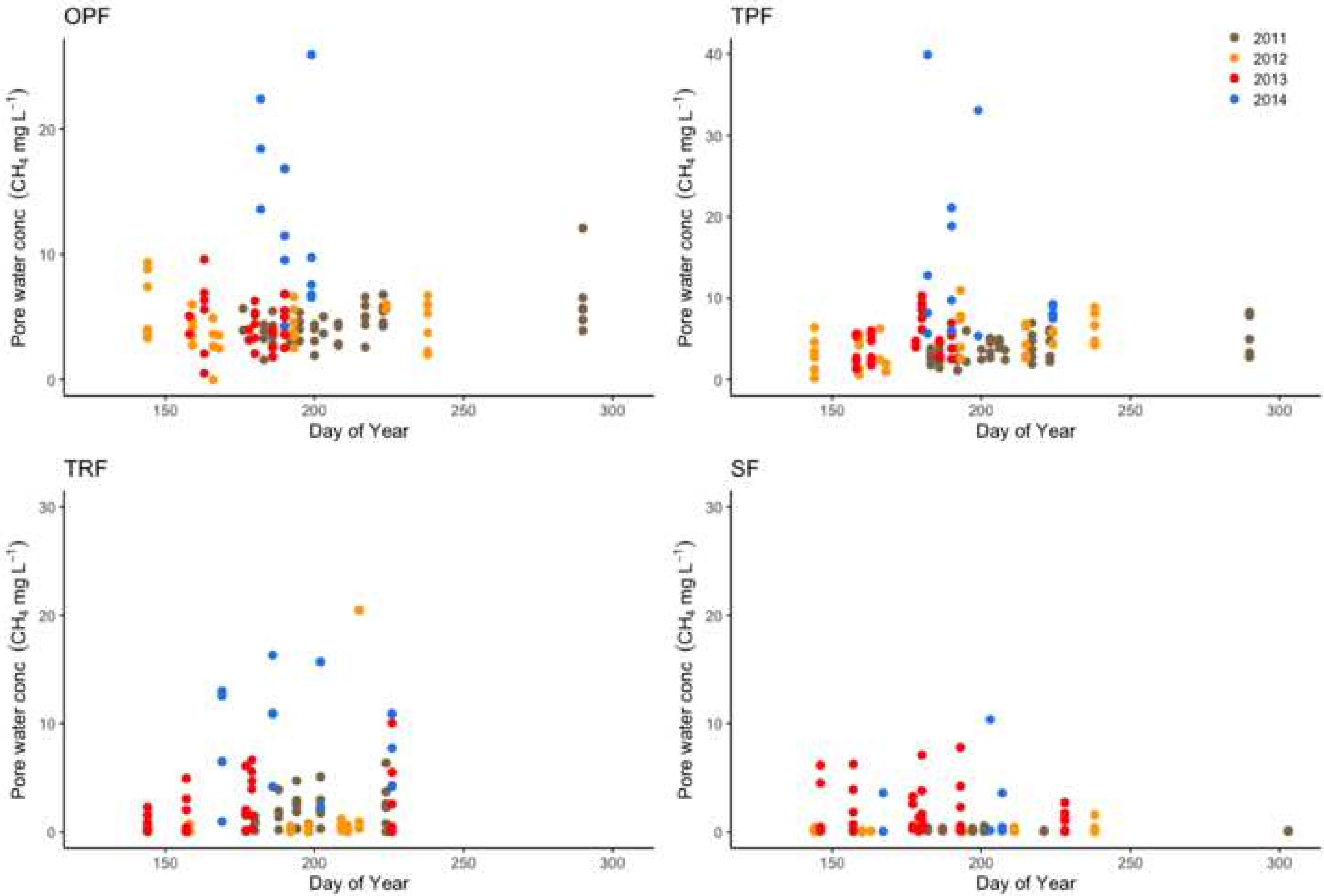
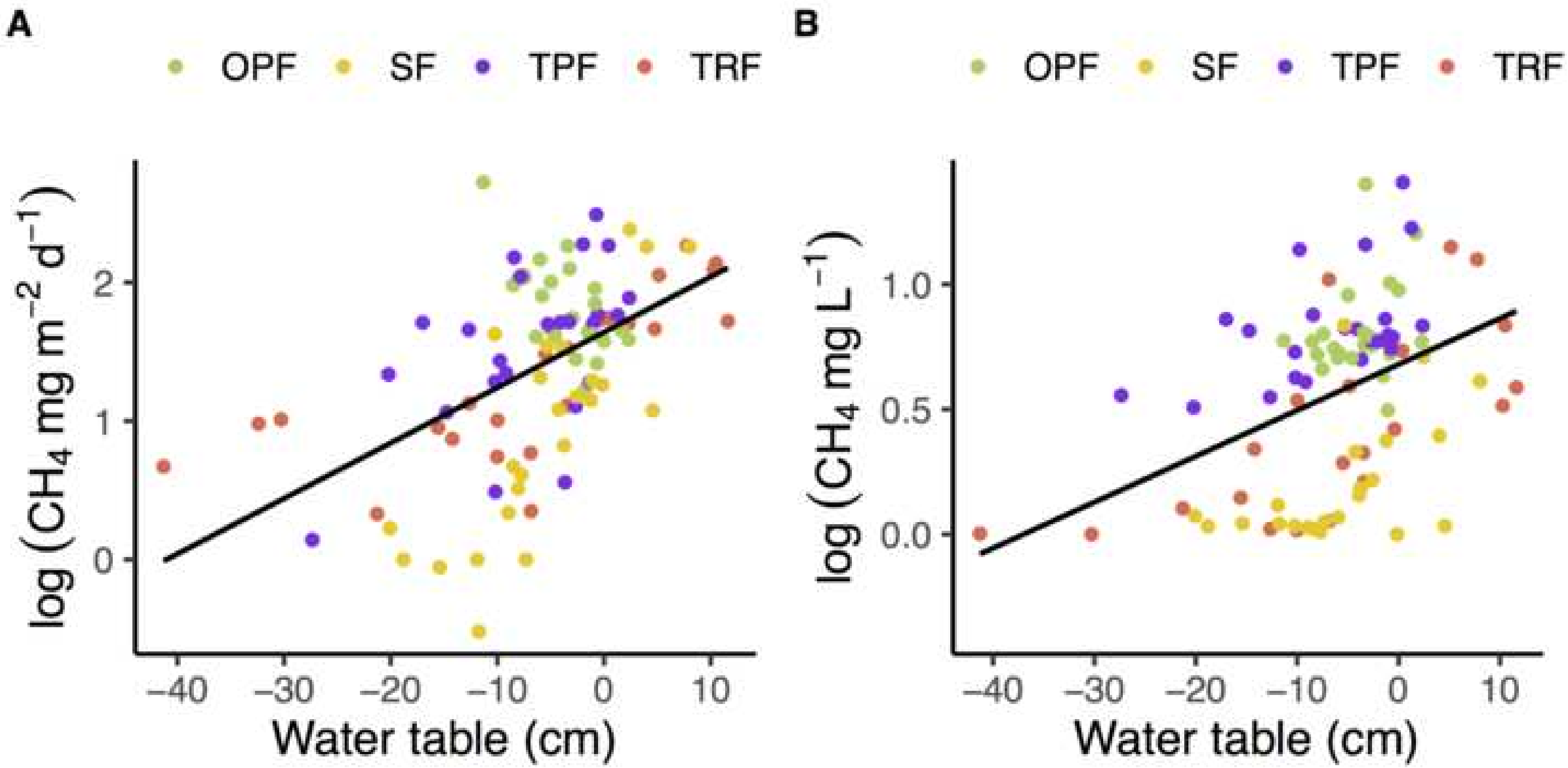


Figure 5



A
OPF
SF
TPF
TRF
log (CH4 mg m−2 d−1)
2
1
0
−40
−30
−20
−10
0
10
Water table (cm)
B
OPF
SF
TPF
TRF
log (CH4 mg L−1)
1.0
0.5
0.0
−40
−30
−20
−10
0
10
Water table (cm)

Methane emissions from fens in Alberta's boreal region: Reference data for functional evaluation of restoration outcomes

Funding sources:
Funding for this project was provided by a Natural Sciences and Engineering Research Council of Canada (NSERC) Collaborative Research and Development (CRD) grant (#418557) to JP and MS co-funded by Suncor Energy Inc., Imperial Oil Resources Limited, and Shell Canada Energy. The authors would like to acknowledge Canada's Oil Sands Innovation Alliance (COSIA) for its support of this project. Funders had no role in study design, data analysis or decision to publish the results.