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# Abstract

Many quantitative relations in the environmental sciences, and specifically in watershed

# 1. Introduction

Wetlands are important global carbon (C) stores, accounting for 20-30% of the total terrestrial C storage in soils despite only covering 4-6% of the Earth’s land surface (refs). While peatlands are responsible for the majority of C stored in wetland soils, freshwater mineral soil (FWMS) wetlands are also globally significant C stores. Furthermore, FWMS wetlands are typically much more productive compared to peat forming wetlands (Mitsch and Gosselink 2000; Rocha and Goulden 2009). Globally, average C sequestration in established temperate FWMSWs is estimated to range between 100 and 250 (Bernal and Mitsch, 2012; Zhang et al., 2016; Lu et al., 2017).

The North American Prairie Pothole Region (PPR) extends from north-west Iowa in the USA into central Alberta in Canada and covers an area of ~800,000 km (Badiou et al., 2011). This region is dotted with millions of FWMS wetlands, generally refereed to as prairie pothole wetlands. Relative to other wetland ecosystems such as swamps, bogs, and northern peatlands, fewer studies have focused on prairie pothole wetlands despite their high C sequestration capacity (Bansal et al., 2016).

Although peatlands account for the majority of wetland area in Canada, it is estimated that ~20 million ha of FWMSWs have been lost in Canada since European settlement (~1800), compared to 1.4 million ha of peatlands (National Wetlands Working Group (NWWG) 1988), resulting in significant emissions of to the atmosphere (refs). Conversely, restoring FWMSWs can reverse soil C loss and sequester atmospheric (refs). Several studies have shown that restored wetlands in the Prairie Pothole Region of North America are particularly proficient at sequestering C (Gleason et al., 2006), with C sequestration rates ranging between 110-305 (Euliss et al., 2006; Badiou et al., 2011; Tangen and Bansal, 2020). These high C sequestration rates are driven by high productivity and low decomposition rates created by anoxic conditions. However, the same conditions which allow PPR wetlands to accumulate large amounts of C also promote the production and emission of methane ().

Methane fluxes from PPR wetlands have been observed to be among the highest reported for freshwater wetlands, although emissions show considerable spatial and temporal variability (Bansal et al., 2016, Badiou et al., 2011, Pennock et al., 2010). Notably, emissions in the PPR are significantly inversely correlated to concentrations of wetland waters (Pennock et al. 2010, Bansal et al., 2016). PPR wetlands have a wide range of sulfate-dominated salinities due to undulating topography and groundwater interactions with sulfur and carbonate rick glacial till (Winter and Rosenberry 1998;Goldhaber et al.2014). Higher sulfate concentrations are typically linked to reduced emissions as sulfate-reducing bacteria out compete methanogens for primary substrates such as acetate and hydrogen (refs).

While there have been a growing number of studies focused on C cycling in the PPR wetlands, to date observations of greenhouse gas (GHG) fluxes in the region have only been conducted using chamber-based methods (e.g., Bansal et al., 2016, more refs). While chambers are advantageous for assessing spatial variability in GHG exchange and treatment effects on fluxes, they are discrete in time, cover only a small area and are challenging to conduct over tall, emergent vegetation which dominate PPR wetlands. These limitations present challenges for estimating robust annual GHG budgets at the ecosystem level (Baldocchi 2003). Conversely, eddy covariance measurements can provide GHG flux estimates that are near-continuous and at ecosystem-scale flux measurements, without interfering with the system they are measuring. This makes this approach well-suited for estimating accurate GHG budgets and informing nature-based climate solutions (Novick et al., 2022). Furthermore, coupling these quasi-continuous flux measurements with ancillary biophysical measurements can provide new insights into the controls on GHG fluxes across a range of temporal scales (Knox et al., 2021)

Here we present the first eddy covariance estimates of carbon dioxide () and fluxes from two geographically isolated freshwater marshes in the grasslands and croplands of the PPR of Canada. Our objectives are to: (1) assess the annual GHG budget of these two wetland sites, and (2) identify the biophysical drivers of and fluxes at these sites and if/how they differ between sites.

# 2. Methods

## 2.1 Site description

MBPPW1, located at 50.3623˚N, -100.20242˚W, is an isolated cropland marsh in the PPR of Manitoba, Canada. This wetland site is entirely dominated by emergent vegetation, primarily populated by Schoenoplectus tabernaemontani and Typha spp. The water chemistry in this wetland is characterized by high sulfate concentrations.

MBPPW2, located at 50.3705˚N, -100.5339˚W, is an isolated grassland marsh about 24 km East of MBPPW1. However, this site is much more heterogeneous than MBPPW1 and is characterized by a combination of open water and emergent vegetation. Large mats of submersed macrophytes are found near the open water surface during the growing season, and the emergent vegetation is dominated by Typha spp. This site is characterized by lower sulfate concentrations.

## 2.2 Eddy covariance measurements

## 2.3 Gap-filling, NEE partitioning, and annual budget computation

## 2.4 Supporting measurements

### 2.4.1 Water quality measurements

ABS 280 - does higher mean more recalcitrant DOC?

## 2.5 Statistical methods

All data processing and statistical analyses were conducted using R (R Core Team, 2019). A two-sample Wilcoxon test was used to assess differences in water quality parameters between sites (NOTE TO DELETE: this is for variables with only 1 year) due to the small sample size and the fact that a number of quality parameters were not normally distributed.This was done using the wilcox.test function from the stats package.

When water quality parameters were available for multiple years, significant differences in water quality parameters between sites, years and the interaction between site and years were assessed using a two-way ANOVA with Type III Sums of Squares on the rank-transformed data, as ranking can be used to transform data that do not meet the assumptions of normality (ref). Here, each original data value was replaced by its rank, with 1 for the smallest value to N for the largest, where N is the combined data sample size. This rank-based approach is robust to non-normal errors, resistant to outliers, and is effective for many distributions.

Multivariate associations of water quality parameters across sites were analyzed using principal component analysis (PCA). All variables were centered and scaled, and PCA was done using the prcomp function in base R. As certain water quality parameters were highly correlated (Figure [2](#fig-PCA)), we only selected a subset of water quality parameters for the hierarchical mixed model. To capture the primary differences in water quality parameters between sites, only , TP, and pH were included in the model.

# 3. Results

## 3.1 Weather and hydrological conditions

### 3.1.1 Water quality

Significant differences were observed in water quality parameters across sites (Table [1](#tbl-WQ), Figure [2](#fig-PCA)). Averaged across years, Hogg had significantly higher , specific conductivity, dissolved organic carbon (DOC), total dissolved nitrogen (TDN), and Specific ultraviolet absorbance at 280 nm (ABS 280) than Young. Conversely, dissolved reactive phosphorus (DRP), total dissolved phosphorus (TDP) concentrations than Young, total phosphorus (TP) was significantly lower at Hogg than Young. No significant differences in pH, NO3 or NO2, or NH4 were observed between sites.

Table 1: Differences in water quality parameters across sites.

| site | year | pH | SO4 | Specific\_cond | DOC | TDN | NO3\_NO2\_N | NH4\_N | DRP | TDP | TP | ABS\_280nm |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| Hogg | 2021 | 9 | 1211 | 3848 | 95 | 5 | 15 | 116 | 19 | 94 | 181 | 2 |
| Hogg | 2022 | NaN | 279 | NaN | NaN | NaN | NaN | NaN | NaN | NaN | NaN | NaN |
| Young | 2021 | 9 | 340 | 1017 | 30 | 2 | 4 | 44 | 289 | 387 | 505 | 0 |
| Young | 2022 | NaN | 198 | NaN | NaN | NaN | NaN | NaN | NaN | NaN | NaN | NaN |

Figure [1](#fig-SO4).

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| Figure 1: SO4. |

Figure [2](#fig-PCA).

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| Figure 2: PCA. |

Figure [3](#fig-NEE).

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| Figure 3: NEE. |

# 4. Discussion

Notes:

Also, waters of the permanentwetlands in the PPR tend to contain relatively high sulfateconcentrations (Phillips and Beeri2008;Pennocketal.2010 ), which suppresses methane production. Thus, chemicalvariation among PPR wetlands may play a key role in moder-ating greenhouse gas emissions from wetlands, especiallyfrom those with longer periods of ponding

Badiou et al., 2011 Additionally, thedilution effect resulting from this increase in waterlevel greatly reduced concentrations of nutrients andmajor anions and cations. This is important as sulfatereduction is known to at least partially inhibit CH4

production (Gauci et al.2004). A recent studyconducted in ephemeral prairie pothole wetlands inSK by Pennock et al. (2010) has demonstrated thatCH4emissions decrease as sulfate (SO42-) concen-trations increase. Rapid increases in CH4emissionsassociated with increased spring runoff leading to

depletion in SO42-concentrations has also beendocumented by Phipps (2006) for a permanentwetland located in the St. Denis National WildlifeArea in Saskatchewan, Canada

# References

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, <https://doi.org/10.1007/s11273-011-9214-6>, 2011.

# Tables