

The Forces Behind the Flux: Methane, Carbon Dioxide, and Nitrous Oxide Dynamics and
Their Environmental Drivers in Restored Agricultural Wetlands

by

Shayna Bree Meinzing

A thesis

presented to the University of Waterloo

in fulfilment of the

thesis requirement for the degree of

Master of Science

in

Earth Sciences

Waterloo, Ontario, Canada, 2026

© Shayna Bree Meinzing 2026

Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Wetlands provide vital ecosystem services such as water filtration and flood mitigation but are also significant natural sources of greenhouse gases (GHGs), particularly methane (CH_4). This study examined seasonal and spatial patterns of CH_4 , carbon dioxide (CO_2), and nitrous oxide (N_2O) emissions from seven restored agricultural wetlands in the Ontario portion of the Lake Erie Basin, focusing on diffusive and ebullitive flux pathways. Emissions were measured across four seasons, alongside water quality parameters used to identify key environmental drivers. Dissolved oxygen (DO) emerged as a strong driver of GHG fluxes, with lower DO concentrations consistently promoting higher CH_4 , CO_2 , and N_2O emissions. Duckweed cover also enhanced CH_4 production by creating anoxic conditions. GHG emissions peaked during summer months with heightened biological activity, while winter fluxes, though reduced, remained detectable, emphasizing the contribution of cold-season processes. N_2O emissions remained consistently low throughout the year. Across sites, methane emissions were generally low relative to natural temperate wetlands, except at one nutrient-enriched outlier (Site MA). Spatial variation within wetlands was minimal, suggesting that sampling from a single representative location may be sufficient for long-term monitoring. These findings show that restored agricultural wetlands can act as both carbon sinks and GHG sources depending on local biogeochemical conditions. By identifying major environmental controls on emissions, this study advances understanding of GHG dynamics in restored wetlands, informs efficient monitoring and modelling considerations, and strengthens national inventory and restoration policy development.

Keywords: Greenhouse gas emissions, restored agricultural wetlands, environmental drivers, wetland restoration, emission modelling

Acknowledgements

I would like to sincerely acknowledge that this research is funded by a Natural Sciences and Engineering Research Council of Canada (NSERC) Alliance and Mitacs grant ('Designing Climate and Water Smart Wetland Restoration Scenarios for Canada's Agricultural Landscapes') with PI Nandita Basu, co-PI Tonya DelSontro and collaborators, Pascal Badiou and Bryan Page from Ducks Unlimited Canada.

I gratefully acknowledge Tonya DelSontro for her invaluable guidance and leadership throughout this project. She conceptualized the initial framework for the field sampling campaigns, which served as the foundation for the study design and was further developed over the course of the project. Her mentorship and consistent support were instrumental to the successful completion of this research.

I extend my sincere thanks to my committee members — Maddy Rosamond, Rebecca Rooney, and Nandita Basu — for their valuable guidance, insightful recommendations, and support during the development of the project proposal. I am grateful for their continued involvement and enthusiasm throughout the course of this research.

I am also grateful to the landowners who generously granted access to their wetlands for fieldwork. Their openness and enthusiasm for this research were deeply appreciated and made this work possible.

I am deeply indebted to my field team for their support in completing six extensive field campaigns. The data collection for this project was a strong team effort, and everyone involved was integral to this project's success. Mackenzie Sluys, Sara Abate, Cooper Pickering, Anisha Karkera, Finan Turnage-Barney, Jordan Li, Chris Bisson, Tim Siembida, Isabella Sciorilli, Ruby Collins, Al Meinzinger, Lara Dobre and Joey Ruck contributed to field sampling for this project

while I held a team leader role. Their dedication, hard work, and collaboration were invaluable, and this thesis would not have been possible without their commitment and support.

Many thanks to Richard Elgood for providing invaluable insight throughout this project. His expertise and readiness to assist in developing more effective sampling protocols were instrumental in the success of this research.

This work was made possible by the contributions of numerous individuals and laboratories involved in the comprehensive sample analyses. We extend our sincere gratitude to Richard Elgood, Holly Grierson, Laura Berkovitz, William Mark, Ryan Hutchins, Justin Harbin, Katherine Velghe, and Gena Braun for their invaluable support.

I would like to acknowledge Rachel McNamee for her expert guidance in chlorophyll analysis and for coordinating laboratory access, which facilitated the processing of project samples. In addition, I would like to thank Jumi Gogoi for her support and expertise in Random Forest modelling. Thank you for generously sharing your time and knowledge.

I acknowledge the use of AI tools (ChatGPT, OpenAI; Grammarly) to assist with literature searches, R code development and troubleshooting, and to enhance clarity and grammar in parts of this thesis.

The author declares no conflicts of interest.

Land Acknowledgement

This land acknowledgement has provided me with another avenue of learning, allowing me to increase my awareness about the importance of land, knowledge and history. As a student studying earth science, my work is focused on exploring the physical processes of the Earth, but I recognize that this research alone cannot fully capture the long histories and profound wisdom held by the Indigenous peoples who have lived in relationship with these lands for millennia. I strive to conduct research that respects Indigenous knowledge and sovereignty.

It is important to recognize the injustices that Indigenous peoples within Canada have faced throughout this long history, including the displacement from their lands, forced assimilation through residential schools, loss of language and culture, and ongoing marginalization. These injustices have had profound and lasting impacts on Indigenous communities, and it is crucial to acknowledge them as part of the reconciliation process and to actively work towards restoring justice, equity, and healing.

I acknowledge that this research was conducted on the traditional territories of the Anishinabewaki, Mississauga, Ho-de-no-sau-nee-ga (Haudenosaunee) and Attiwonderonk (neutral) peoples. Specifically, the area encompasses lands covered by the London Township Treaty (1796), McKee Treaty (1790), Huron Tract Treaty (1827), Treaty 21 (1819), Treaty 3 (1792), and the Dish with One Spoon Covenant Wampum. I honour and respect the enduring presence and contributions of the Chippewas of the Thames First Nation, Oneida Nation of the Thames, Munsee-Delaware Nation, and all Indigenous peoples who have cared for this land for generations.

By conducting this research, I recognize that I am engaging with a space that holds layers of history, culture, and meaning far beyond my immediate scientific focus. I am committed to understanding and respecting this legacy as part of my research journey.

Dedication

To my advisor, Dr. DelSontro, who provided me with an opportunity to grow both professionally and personally.

To my parents, Joy and Al Meinzinger, for providing me with a loving home and unwavering support, no matter how far apart we are.

To Mackenzie Sluys, for being by my side during this journey and helping me navigate the challenges and celebrate the successes.

Table of Contents

Author's Declaration.....	ii
Abstract	iii
Acknowledgements.....	iv
Land Acknowledgement	vi
Dedication	viii
List of Figures	xiii
List of Tables	xviii
List of Abbreviations	xix
List of Symbols	xx
Chapter 1: Literature Review.....	1
1.1 Greenhouse Gas Production and Emission Pathways in Wetlands	1
1.2 Environmental Drivers of Wetland GHG Emissions.....	1
<i>Temperature</i>	2
<i>Wetland Size</i>	3
<i>Nutrients</i>	4
<i>Chlorophyll-a</i>	5
<i>Cations and Anions</i>	6
<i>pH</i>	7
<i>Substrate</i>	7
<i>Oxygen Levels</i>	8

<i>Hydrological Drivers</i>	9
<i>$\delta^{18}O$ and δ^2H of H_2O</i>	11
<i>Additional Environmental Drivers</i>	12
1.3 Environmental Drivers of GHG Emissions in Agricultural Wetlands.....	13
1.4 Research Gaps.....	14
1.5 Objectives	15
Chapter 2: Greenhouse Gas Dynamics in Restored Agricultural Wetlands	16
2.1 Background and Context.....	16
2.2 Methods.....	21
2.2.1 Study Design.....	21
<i>Study Sites</i>	21
<i>Sampling Strategy</i>	22
2.2.2 GHG Sampling Strategy	23
<i>Dissolved Gas Sampling</i>	23
<i>Diffusive GHG Fluxes</i>	23
<i>Ebullitive CH_4 Fluxes</i>	26
<i>Statistical Analysis</i>	27
2.2.3 Water Chemistry	28
2.2.4 Environmental Driver Analysis on Diffusive Emissions.....	29
<i>Kendall Correlation</i>	29
<i>Random Forest</i>	32
2.3 Results.....	33

<i>Summary of Environmental Conditions Across Wetlands</i>	33
2.3.1 Diffusive GHG Flux Results.....	35
<i>Diffusive GHG Flux Spatial Variability</i>	38
<i>Diffusive GHG Flux Seasonal Variability</i>	41
2.3.2 Ebullitive GHG Flux Results	45
<i>Ebullitive GHG Flux Spatial Variability</i>	45
<i>Ebullitive GHG Flux Seasonal Variability</i>	47
2.3.3 Environmental Drivers Analysis Results	48
<i>Kendall Correlation During the Peak GHG Production Season</i>	48
<i>Random Forest Modelling of Diffusive GHG Fluxes</i>	50
2.4 Discussion	56
2.4.1 Flux Variability in Restored Agricultural Wetlands of Southern Ontario	56
2.4.2 The Force Behind the Flux: Environmental Drivers.....	59
2.4.3 Modelling Considerations.....	62
<i>Spatial Considerations for GHG Modelling</i>	62
<i>Influence of Duckweed on GHG Emissions</i>	64
<i>Seasonal Considerations for GHG Modelling</i>	66
<i>Winter Impacts on GHG Fluxes</i>	67
2.4.4 Recommendations and Relevance of Findings	70
2.4.5 Management and Restoration Considerations	71
Chapter 3: Conclusion.....	73
3.1 Summary of Key Findings	73

3.2 Implications of Research.....	74
3.3 Limitations	76
3.4 Future Research	76
References	78
Appendix.....	108

List of Figures

Figure 1.1: Summary of directional relationships between environmental drivers and GHG fluxes (CH ₄ , CO ₂ , and N ₂ O) synthesized from the literature review. Arrows indicate the general trend reported across studies (↑ increase, ↓ decrease, → no consistent relationship). Overall, most studies exhibited similar directional patterns; however, several papers reported contradictory or context-dependent results.....	14
Figure 2.1: Study sites in restored agricultural wetlands of southwestern Ontario, Canada, with an inset showing the regional context. Wetlands are labelled by DUC site ID (Page et al., 2023).	22
Figure 2.2: Bar plots showing the average surface (orange) and above bottom (i.e., depth in purple) water quality parameters measured at the middle location across multiple campaigns (1 in June 2024; 2 in July 2024; 3 in August 2024; 4 in October 2024; 5 in January/February 2025; 6 in April 2025). Values represent mean water temperature (°C), specific conductivity (μS/cm), pH, and dissolved oxygen (DO) saturation (%).	34
Figure 2.3: Maximum water depth measured across wetland sites over six monitoring campaigns.	34
Figure 2.4: Boxplots showing the overall variability of diffusive GHG fluxes (CH ₄ , CO ₂ , N ₂ O) across wetland sites. Data includes all inlet, middle, and outlet locations combined for each site over 6 campaigns spanning one year. Measurements taken on duckweed, snow, ice, and after augering a hole in winter are included. N ₂ O fluxes were measured at each sampling, but the majority were close to zero and hence hard to visualize here (<i>see Figure 2.6 for more detail</i>). Each box represents the interquartile range, the horizontal line indicates the	

median, and whiskers extend to the minimum and maximum values within 1.5× the interquartile range. Outliers beyond this range are shown as individual points. 36

Figure 2.5: Boxplots of CH₄ (top) and CO₂ (bottom) fluxes (mmol m⁻² day⁻¹) by site and location category across the annual sampling period. Winter measurements include only ice and snow fluxes, representing the minimum fluxes expected during the coldest season. Winter measurements were only taken at the middle location. Red points indicate back-calculated data used when direct chamber measurements were not feasible. Duckweed locations at site MA are represented with a dark green point. Neon-green points represent back-calculated duckweed locations. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within 1.5× the interquartile range. Outliers beyond this range are shown as individual points..... 37

Figure 2.6: Boxplots display N₂O fluxes (mmol m⁻² day⁻¹) by site and location category across the annual sampling period. Duckweed locations at site MA were excluded. Winter measurements include only ice and snow fluxes from the middle location, representing the lowest fluxes during the coldest season. Red points indicate back-calculated values used when direct chamber measurements were infeasible. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within 1.5× the interquartile range. Outliers beyond this range are shown as individual points. 38

Figure 2.7: Seasonal variations in diffusive CH₄, CO₂ and N₂O emissions across restored wetland sites in the middle location. Light blue bars represent fluxes from the middle location of each wetland, including the ice hole drilled in January and February. Fluxes

taken on top of snow/ice or duckweed are distinguished by dark blue and green bars, respectively. Dark blue bars for ice and snow measurements are present but not visible at this scale, indicating extremely low emissions of CH₄, CO₂, and N₂O from snow/ice at those sites. Note that majority of N₂O fluxes are near zero and thus not quite visible at the reported scales. 42

Figure 2.8: Boxplots of CH₄, CO₂, and N₂O fluxes (mg m⁻² d⁻¹) across six sampling campaigns at the middle locations. All measurements were included to show overall variability. CH₄ fluxes have been visually zoomed to improve clarity, while extreme values remain included in the calculations. A red arrow indicates the maximum value for each month in the CH₄ panel that is not visualized. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within 1.5× the interquartile range. Outliers beyond this range are shown as individual points. 42

Figure 2.9: Dissolved CH₄ profiles at each restored wetland site across all sampling seasons. In October, wetlands exhibited their lowest maximum water depths (at BL, only surface samples were collected). During the January/February campaign, winter conditions compromised samples at BL and FE, resulting in profiles with only surface or limited depth measurements. 44

Figure 2.10: Boxplots showing the total gas released (ml/day/cm²) across sampling sites. Each box represents the distribution of measured gas release per site encompassing all funnels on the transect. In winter, there were no measurements taken. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within 1.5× the interquartile range. Outliers beyond this range are shown as individual points. 46

Figure 2.11: Seasonal variations in ebullitive emissions across restored wetland sites. Fluxes are shown from all locations across the transect. No measurements were taken during winter. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within $1.5 \times$ the interquartile range. Outliers beyond this range are shown as individual points..... 47

Figure 2.12: Kendall rank correlations between diffusive fluxes of CH₄, CO₂, and N₂O and their associated environmental drivers. Correlations are filtered by significance thresholds ($p < 0.05$ for CO₂; $p < 0.10$ for CH₄ and N₂O) using base R's cor.test function. Data are restricted to Campaigns 1–3 (June–August) to ensure consistent summer temperatures and include only middle locations where YSI data were collected. Measurements taken on duckweed were excluded. Orange bars represent negative correlations, while blue bars represent positive correlations. Unless specified, all variables were measured at the surface of the wetland..... 49

Figure 2.13: Variable importance for Random Forest prediction of diffusive GHG emissions. Importance is measured as the mean decrease in model accuracy (%IncMSE) when each variable is permuted. Categorical predictor 'Site' was excluded in one scenario and included in another to test for any site-specific parameters not measured. Variables in red text differ between the two scenarios, while green text indicates variables shared across both models. Site is highlighted when it appears in importance. Measurements collected on duckweed were excluded from analysis, although duckweed was retained as a categorical variable. Winter measurements were limited to those taken on ice during the winter season. 50

Figure 2.14: Partial-dependence plots for the top 10 predictor variables in the Random Forest models *without* site included as a predictor variable. The y-axis represents the predicted

response values averaged across all observations for each predictor value. The histograms at the bottom of each plot display the distribution of the predictor values, highlighting the density of observations that inform the plotted relationships. The quick drop in values (for example, CH₄ for dissolved oxygen) is an artifact of the outlier results from Site MA. **54**

Figure 2.15 Conceptual schematic illustrating seasonal patterns in greenhouse gas emission pathways from an agricultural wetland system. Relative diffusive fluxes of CH₄ (red arrows), CO₂ (green arrows), and N₂O (blue arrows) are shown for each season, while ebullitive emissions are depicted as dashed red arrows (CH₄ only). Arrow lengths indicate relative magnitudes and are comparable only within the same gas and emission pathway. Insets show representative dissolved oxygen (blue) and temperature (orange) depth profiles, highlighting summer stratification and spring and fall mixing events. Seasonal context reflects the surrounding agricultural land use and the progression of the growing season. **58**

List of Tables

Table 2.1: Environmental parameters included in the Kendall correlation analysis, grouped by category.	30
Table 2.2: Random Forest model performance metrics for predicting diffusive GHG (CH ₄ , CO ₂ , N ₂ O) in the Middle of each wetland. R ² indicates the proportion of variance explained, and RMSE (Root Mean Squared Error) quantifies prediction error. Metrics are shown for three scenarios: using Middle predictors only (baseline), Inlet predictors, and Outlet predictors. No YSI variables were included; dissolved gas measurements were included. Higher R ² and lower RMSE indicate better predictive performance.....	55
Table 2.3: Comparison of site-specific characteristics highlighting the unique features of MA relative to the other study wetlands (Page et al., 2023).	65

List of Abbreviations

Abbreviation	Definition
GHG	Greenhouse Gas
CH ₄	Methane
CO ₂	Carbon Dioxide
N ₂ O	Nitrous Oxide
SRP	Soluble Reactive Phosphorus
DP	Dissolved Phosphorus
TP	Total Phosphorus
TN	Total Nitrogen
DN	Dissolved Nitrogen
DOC	Dissolved Organic Carbon
DOM	Dissolved Organic Matter
DIC	Dissolved Inorganic Carbon
POM	Particulate Organic Matter
SPC	Specific Conductivity
DW	Duckweed
RF	Random Forest
KT	Kendall Tau

List of Symbols

Symbol	Definition
δ	Isotopic Composition
‰	Permil
%IncMSE	Percent Increase in Mean Squared Error



Chapter 1: Literature Review

1.1 Greenhouse Gas Production and Emission Pathways in Wetlands

Methane (CH₄), carbon dioxide (CO₂), and nitrous oxide (N₂O) are all produced in wetlands through distinct microbial processes but share similar emission pathways to the atmosphere. CH₄ is generated in wetlands through methanogenesis, which requires available substrate and anoxic conditions, with methanogenic archaea active across a wide temperature range (−2 °C to 122 °C) (Hoehler et al., 2018; Kotsyurbenko et al., 2019). CO₂ is primarily produced by heterotrophic and autotrophic respiration (Bao et al., 2023), while N₂O arises from microbial transformations of carbon and nitrogen stocks, including nitrification, denitrification, nitrifier denitrification, and dissimilatory nitrate reduction to ammonia (Bahram et al., 2022; Huang et al., 2013). Despite these differences in production, the three gases are emitted from wetlands via the same major pathways: ebullition, diffusion across the water–air interface, and transport through plant aerenchyma (Limpert et al., 2020; Yin et al., 2023).

1.2 Environmental Drivers of Wetland GHG Emissions

Greenhouse gas (GHG) emissions from wetlands can be affected by a variety of natural and anthropogenic factors. This background literature review centers on wetland ecosystems, with particular emphasis on restored agricultural wetlands. To capture relevant insights into GHG dynamics within systems of similar scale and function to the study sites, the review also incorporates research on systems such as, but not limited to, small ponds, lakes, ditches, and constructed wetlands.

Temperature

Elevated temperatures may lead to increased GHG emissions from wetlands. The existing body of research consistently demonstrates that warmer temperatures increase CH₄ emissions (Gedney et al., 2004; Moss et al., 2011; Z. Zhang et al., 2017, 2023). A simulation, considering current climate change mitigation approaches, forecasts a doubling of CH₄ emissions from wetlands by the year 2090 (Z. Zhang et al., 2023). Zhang et al. (2023) proposed the likelihood of a positive feedback loop between CH₄ emissions from wetlands and the ongoing processes of global warming and shifts in precipitation patterns. A study of small artificial water bodies shows CH₄ emissions increase with temperature and decline in winter (Peacock et al., 2021). In North American wetlands, soil temperature positively correlates with CH₄ emissions, as shown by a large database of 19,000 chamber measurements (Bansal, Post van der Burg, et al., 2023) and in northwestern Eurasian and Greenlandic wetlands (Christensen et al., 2003). CH₄ ebullition increases with temperature, with studies showing an 11% rise per 1 °C in Chinese rivers and lakes (G. Wang et al., 2021), exponential relationships in a shallow lake (Praetzel et al., 2021), and a positive correlation with sediment temperature (DelSontro et al., 2016). Continuing, 90% of diffusive CH₄ fluxes occur during warmer months in Estonian constructed wetlands (Mander et al., 2024). Eutrophic conditions in the waterbody are significant, as a 2°C rise in temperature is linked to an increase in CH₄ emissions by 47-183%, depending on the lake's eutrophic status (Sepulveda-Jauregui et al., 2018).

Elevated temperatures and extreme weather events, such as droughts, can accelerate vegetation decomposition and alter hydrological conditions, potentially transforming wetlands from carbon sinks to carbon sources by increasing both CO₂ and CH₄ emissions (Flanagan & Syed, 2011; Peacock et al., 2021; Salimi et al., 2021). Demonstrating this, CO₂ emissions in

constructed wetlands were observed to be higher during warmer months (SØvik et al., 2006). Moreover, a study investigating 37 urban ponds in Denmark identified a temperature-related increase in CO₂ emissions (Audet et al., 2020).

Elevated temperatures have the potential to induce heightened microbial activity in wetlands, leading to increased N₂O emissions through elevated rates of nitrification and denitrification. This phenomenon was evident in a constructed wetland study where N₂O fluxes were notably higher in the summer compared to the winter (SØvik et al., 2006). Similarly, a study of 37 urban ponds found an increase in N₂O flux with rising temperatures (Audet et al., 2020). Additionally, research on a freshwater wetland demonstrated a positive correlation between N₂O emissions and increasing temperature (Khoiyangbam & Chingangbam, 2022).

Despite literature supporting a positive correlation between GHG flux and temperature, some studies diverge from this trend. Baron et al. (2022) found no correlation between CH₄ emissions and increased temperature in small shallow ponds (Baron et al., 2022). Similarly, DelSontro et al. (2016) and van Bergen et al. (2019) reported no significant correlation between CO₂ flux and temperature in studies about urban ponds, northern ponds, and lakes (DelSontro et al., 2016; van Bergen et al., 2019).

Wetland Size

Smaller aquatic systems generally produce higher GHG emissions. Studies of ponds, ditches, and wetlands show CH₄ emissions increase as waterbody size decreases (Bansal, Post van der Burg, et al., 2023; Bastviken et al., 2004; DelSontro et al., 2018; Gorsky et al., 2019; Holgerson & Raymond, 2016; Peacock et al., 2021; Rasilo et al., 2015), supported by data from a North American wetland complex comprising 19,000 chamber measurements (Bansal, Post van

der Burg, et al., 2023). Smaller water bodies have greater perimeters relative to volume, enhancing terrestrial carbon inputs, mixing, and ebullitive CH₄ release from sediments (Holgerson & Raymond, 2016). CO₂ emissions were shown to decrease with increasing lake size across productivity levels (DelSontro et al., 2018), as well as decrease with increasing wetland size (Peacock et al., 2021).

N₂O shows more variable patterns, increasing with lake size in some syntheses (DelSontro et al., 2018) and remaining low in small nitrogen-rich reservoirs (Webb et al., 2019); However, in wetlands with lower water levels, N₂O emissions are more prevalent and actively driven (Kasak et al., 2022).

Nutrients

Studies of small ponds, ditches, and restored wetlands show that elevated nutrient concentrations, including carbon, nitrogen, and phosphorus, are associated with increased GHG fluxes (Peacock et al., 2021). Total phosphorus correlates positively with CH₄ in small artificial water bodies and drainage ditches (Köhn et al., 2021; Malyan et al., 2022; Peacock et al., 2021), while total nitrogen correlates positively with N₂O in freshwater wetlands wetland (Khoiyangbam & Chingangbam, 2022; Malyan et al., 2022). Nitrate also influences emissions, showing positive correlations with CO₂ and N₂O (Stadmark & Leonardson, 2005, 2007; H. Tian et al., 2015), although some studies report no or negative correlations with CH₄ (Audet et al., 2020; Köhn et al., 2021; Stadmark & Leonardson, 2007). Sources of ammonium have been shown to stimulate N₂O emissions in agricultural streams, and are more dominant than nitrate sources (S. Wang et al., 2024).

Eutrophic water bodies, with high nutrient loads, generate low-oxygen zones that favour methanogens, enhancing CH₄ emissions in systems (Beaulieu et al., 2019; Davidson et al., 2018; Y. Li et al., 2021; Sepulveda-Jauregui et al., 2018). In lakes, eutrophication can amplify CH₄ responses to temperature increases (Sepulveda-Jauregui et al., 2018), while CO₂ flux may decrease due to enhanced photosynthetic carbon fixation (Colas et al., 2021; H. Sun et al., 2021). Lake Erie, for example, has experienced large increases in CH₄ and N₂O emissions across all seasons due to eutrophication (Fernandez et al., 2019). Higher temperatures further favour algal growth in shallow lakes, reinforcing the link between temperature, eutrophication, and CH₄ emissions; lower nutrient concentrations are needed to stimulate blooms as temperatures rise, especially near agricultural lands (Kosten et al., 2009).

Chlorophyll-a

Chlorophyll-*a* is a valuable indicator of a water body's trophic state (Karimian et al., 2023) and has been linked to GHG dynamics across aquatic ecosystems. A synthesis of 223 studies on lakes and impoundments found a positive correlation between N₂O emissions and chlorophyll-*a* concentrations (DelSontro et al., 2018). Contrastingly, observations from Swedish wetlands showed that higher chlorophyll-*a* was associated with lower N₂O concentrations (Åhlén et al., 2023).

Commonly, CH₄ emissions have been strongly predicted by chlorophyll-*a* concentration. Research on Chinese lakes reported that chlorophyll-*a* was positively associated with CH₄ emissions but negatively correlated with CO₂ flux, likely due to algal photosynthesis enhancing CO₂ fixation into oxygen (Kumar et al., 2023). This negative correlation can be explained by the notion that increased algae levels contribute to the fixation of CO₂ into oxygen through photosynthesis. In wetlands and lakes more broadly, chlorophyll-*a* is often assumed to be

positively related to CH₄ emissions (Badiou et al., 2019; DelSontro et al., 2018), although some studies report no relationship (Bastviken et al., 2004). In contrast, observations from Swedish wetlands revealed that higher chlorophyll-*a* levels were associated with lower CH₄ concentrations, with peak levels occurring during the summer months (Åhlén et al., 2023).

Cations and Anions

Anions and cations play complex roles in GHG dynamics in wetland environments, largely by influencing microbial activity and redox conditions.

Sulfate has been shown to suppress CH₄ emissions in small lentic systems by acting as a competing electron acceptor in anaerobic microbial respiration. For example, sulfate concentrations were negatively correlated with CH₄ flux in a shallow pond (Baron et al., 2022) and in prairie pothole wetlands (Logozzo et al., 2025). More broadly, elevated salinity—often composed of ions such as sulfate, sodium, and chloride—can inhibit CH₄ production in wetlands and agricultural ponds, likely due to shifts in microbial metabolic pathways (Soued et al., 2024). Indeed, alternative electron acceptors such as sulfate, nitrate, nitrite, and manganese have all been associated with reduced methanogenesis under anaerobic conditions (Sapkota et al., 2022). Magnesium application via magnesium-modified wheat straw biochar in acidic soils increased CO₂ emissions but decreased N₂O emissions (Khan et al., 2022).

Fluoride presents a contrasting case. High fluoride concentrations, particularly under saline-alkali conditions, have been linked to increased CH₄ emissions in constructed wetland experiments, likely due to disruptions in microbial community composition and function (Zhou et al., 2023). This suggests that not all anions suppress CH₄ production; in some cases, they may enhance it by altering ecosystem processes.

Potassium appears to have some potential to reduce CH₄ emissions under engineered conditions. In an integrated vertical-flow constructed wetland, the use of potassium peroxymonosulfate led to decreased CH₄ fluxes, suggesting a possible oxidizing or microbial regulatory role (X. Li et al., 2023).

pH

Generally, pH can either promote or suppress GHG emissions, depending on the tolerance and preferences of the microbial communities responsible for producing each gas. Two river-based studies in Mongolia and northern Taiwan similarly concluded that there was no significant relationship between CH₄ emissions and pH (Hao et al., 2021; S.-S. Yang et al., 2015). The association between CO₂ flux and pH in various water bodies, such as lakes, stormwater ponds, urban ponds, and reservoirs, has produced inconsistent results. In a compilation of various water body references by Malyan et al. (2022), CO₂ was shown to be negatively correlated with pH in 5 studies compared to 2 studies indicating a positive correlation (Malyan et al., 2022). A study on small artificial water bodies, which closely resemble the systems at our proposed study sites, demonstrated a negative relationship between pH and CO₂ flux (Peacock et al., 2021). N₂O emissions do not show a clear trend with pH and can either be positively or negatively correlated (Khoiyangbam & Chingangbam, 2022; Ma et al., 2018; H. Tian et al., 2015).

Substrate

CH₄ production is influenced by both autochthonous and allochthonous carbon inputs (Sepulveda-Jauregui et al., 2018). Organic fertilizers have been shown to possibly enhance CH₄ emissions by providing more availability of carbon substrates to methanogens (Yuan et al.,

2018). The addition of organic fertilizers stimulates acetolactic pathways and suppresses hydrogenotrophic pathways of CH₄ production (Yuan et al., 2018). Nitrogen fertilizers, by comparison, suppress CH₄ emissions by stimulating denitrifying bacteria that utilize nitrogen as an energy pathway rather than utilizing substrates that create CH₄ as a byproduct (Tiwari et al., 2020). In a study of northern wetlands, a Pearson correlation showed organic acid concentrations had a consistent, significant correlation with CH₄ emissions from wetlands (Christensen et al., 2003).

Several studies have reported a positive correlation between CH₄ flux and dissolved organic matter (DOM), notably in small artificial waterbodies by Peacock et al. (Gorsky et al., 2019; Obrador et al., 2018; Peacock et al., 2019; H. Tian et al., 2015; G. Wang et al., 2021; S.-S. Yang et al., 2015). Similarly, both N₂O and CO₂ flux have been positively correlated with increasing DOM levels (Borges et al., 2015; Gorsky et al., 2019; Ma et al., 2018; Obrador et al., 2018), notably in small pond-ditch systems. In wetlands, CH₄ production exhibits a greater dependence on DOM than on CO₂ (Hu et al., 2024). Therefore, wetland soils with lower carbon-nitrogen ratios may favour increased CH₄ and CO₂ emissions, as it is indicative of higher decomposition rates (Bansal, Creed, et al., 2023; Hu et al., 2024). In organic soils of natural and semi-natural ecosystems, N₂O fluxes peaked at around 19 soil C:N ratio with low emissions under 10 and over 30 soil C:N ratio (Z. Yao et al., 2022).

Oxygen Levels

CH₄ flux has been negatively correlated with oxygen levels in a water body (Audet et al., 2020; Peacock et al., 2021; H. Tian et al., 2015; S.-S. Yang et al., 2015). However, one study by Hao et al. (2021) focused on a river reported no correlation in this regard (Hao et al., 2021). Methanogens are recognized as obligate anaerobes, but there are cases in which aerobic CH₄

production occurs. There are hypotheses that CH₄ might be produced aerobically in aquatic systems as a by-product of photosynthesis, in pockets of oxygenated microhabitats within anoxic zones, and during bacteria's acquisition of phosphorus from methyl-phosphonate (Bižić et al., 2020; Donis et al., 2017; M. Yao et al., 2016). In two rivers, CO₂ flux was negatively correlated with increasing oxygen levels (Hao et al., 2021; S.-S. Yang et al., 2015). In a wetland complex in Brazil, a higher saturation of CO₂ and CH₄ was found to be higher with less dissolved oxygen (Belger et al., 2011). This relationship was exhibited again in five mature lakes, where an excess of oxygen resulted in an undersaturation of CO₂ (Colas et al., 2021).

N₂O flux, on the other hand, has demonstrated both negative and positive correlations with increasing oxygen levels in different studies (Hao et al., 2021; Ma et al., 2018; H. Tian et al., 2015). In a study focusing on a constructed wetland, N₂O was negatively correlated with increasing oxygen levels, with the highest emissions reported under anoxic conditions (Huang et al., 2013).

Hydrological Drivers

CH₄ and CO₂ emissions are affected by changing water levels in wetlands (X. Yao & Song, 2022). In a river-connected wetland in China, CH₄ and N₂O were positively correlated with water level (Jin et al., 2023). In contrast, a low water level was shown to be the main driver of N₂O emissions in a treatment wetlands study (Kasak et al., 2022). In a study of the effect of water table depth on alpine wetlands in the Qinghai-Tibetan Plateau, an increased water table depth led to more CO₂ emissions (J. Zhang et al., 2024); specifically, CO₂ emissions rose by 31% in fens with an increase of 20 cm in the water table (Matysek et al., 2019). In contrast, an increasing water table depth lowered CH₄ emissions and N₂O emissions (J. Zhang et al., 2024). In a global meta-analysis, a critical water level where CH₄ emissions were the highest was

identified at 50 cm (Calabrese et al., 2021). In a study of seasonal inundation on CH₄ fluxes from freshwater wetlands, inundation extent and duration promoted CH₄ emissions (Hondula et al., 2021).

Water levels may affect the sensitivity of CH₄ emissions to temperature. A higher water table resulted in a higher temperature sensitivity for CH₄ emissions in wetlands (H. Chen et al., 2021). Water levels did not affect the dependence of temperature on CO₂ emissions in wetlands (H. Chen et al., 2021).

Climate warming will increase the frequency of precipitation events, expanding wetland areas, and ultimately creating more anoxic zones available for CH₄ production (Z. Zhang et al., 2023). Conversely, carbon may be trapped more effectively with increased rainfall (Salimi et al., 2021). Rewetting wetlands stops CO₂ emissions, but re-vitalizes CH₄ emissions (Günther et al., 2020). Droughts may release carbon from wetlands, changing them from carbon sinks to sources (Salimi et al., 2021).

In a study of pristine wetlands under global warming conditions, changes in wetland CO₂ flux have a significant positive relationship with the mean annual precipitation regardless of different plant functional types in the ecosystem. This indicates a stronger source of CO₂ flux in wetter conditions (Bao et al., 2023). In this study, there was no correlation found between CH₄ or N₂O emissions regarding mean annual precipitation. Considering wetlands in North America, there was a positive relationship between water-filled pore space and CH₄ emissions in addition to hydroperiod and CH₄ emissions (Bansal, Post van der Burg, et al., 2023). Lastly, in a study of wetlands, an analysis revealed that increased precipitation generally reduces GHG emissions (M. Chen et al., 2020). However, with mean annual precipitation ranging between 400 mm to 700

mm, CH₄ and N₂O emissions increased with increasing mean annual precipitation (M. Chen et al., 2020).

δ¹⁸O and δ²H of H₂O

δ¹⁸O and δ²H isotopes are used to understand hydrological processes over multiple scales, including the identification of vegetation water sources, partitioning of evapotranspiration, separating hydrographs, and quantifying atmospheric processes (C. Tian et al., 2018). These isotopes lend insight into a system's water cycling mechanisms, past climate conditions, (Aron et al., 2021) and surface water evaporation rates (Z. Sun et al., 2021). Deuterium excess is another variable that can be used to further understand temporal and spatial variations in hydrological processes and hydroclimatic conditions (C. Tian et al., 2018). δ¹⁸O helps to show sensitivities to equilibrium and kinetic fractionation processes (C. Tian et al., 2018). If a wetland is enriched in heavier isotopes, this would lend insight that lots of evaporation has occurred in the system (Gallart et al., 2024). Additionally, precipitation at higher latitudes is expected to have more negative δ¹⁸O values due to the latitude effect (Clark & Fritz, 1997).

In a study of precipitation samples over two years, δ¹⁸O showed a seasonal trend with more enrichment in warmer months (C. Tian et al., 2018); δ¹⁸O and δ²H were significantly affected by temperature (C. Tian et al., 2018). In a study on Lake Chad, δ¹⁸O became highly enriched and correlated positively with evaporation rates and salinity (Fontes et al., 1970). In a study characterizing the runoff generation process in Old Crow Flats, Yukon, d-excess gave insight into the source of the runoff. A declining d-excess was associated with lake drainage, a stable d-excess was associated with negligible runoff, and an increasing d-excess was associated with rainfall-generated runoff (Turner et al., 2014). In a study by Behrens et al., a variable δ-excess gave insight into regional groundwater and surface water, reflecting changes in

precipitation sources and evaporation (Behrens et al., 2021). One wetland study found there was a strong evaporative signature in their stable water isotopes in the summer (Miranda & Whitfield, 2024).

Additional Environmental Drivers

Few studies exist on the effect of constructed wetland age on GHG emissions. A study of constructed wetlands compared GHG fluxes at 5 and 15 years after reconstruction. From years 5 to 15, mean CH₄ emissions tripled and mean CO₂ emissions doubled (Liikanen et al., 2006). After 15 years, the average CH₄ release rates were just slightly higher than the range of average emissions for a wetland in the same region; However, the highest recorded CH₄ spike in the study occurred at year 5 and was considerably higher than the normal reported values (Liikanen et al., 2006). In a longer timescale (over 100 years), studies have shown that wetlands act as a sink for GHGs (Brix et al., 2001; Mitsch et al., 2013). More recent studies show restored wetlands have a net cooling effect on average, 141 to 525 years post restoration (Schuster et al., 2024).

Wetland habitat type also affects CH₄ production. The average CH₄ flux compiled for Northern wetlands by Bartless and Harriss (1993) shows that sedge meadow habitats have the highest average flux out of ~115 measurements of different habitats (Bartlett & Harriss, 1993). After trying to model global CH₄ emissions from tropical wetlands, it was seen that emissions are affected by many unique environmental drivers, which results in variable flux measurements, making it difficult to upscale models (Murguia-Flores et al., 2023).

After an intensive study compiling information on CH₄ emissions from 71 northern, temperate, and subtropical wetlands, it was demonstrated that CH₄ flux controls are diverse

among wetland types and complex to model (Turetsky et al., 2014). Therefore, the most accurate models are those developed for localized zones with a narrow range of wetland types and in conditions that are well constrained.

1.3 Environmental Drivers of GHG Emissions in Agricultural Wetlands

Wetlands embedded in agricultural crop lands are exposed to unique conditions, and it has been shown that they have double the aquatic diffusive emissions of GHG compared to perennial landcover, largely driven by CH₄ (Logozzo et al., 2025). Agricultural wetlands are typically eutrophic, as they receive input from the surrounding farm fields of excess nutrients. Wetlands in agricultural zones have been shown to have elevated phosphate and ammonium content derived from fertilizer (Logozzo et al., 2025). A study of agricultural ditches concluded nutrient input was the primary driver stimulating GHG production and emissions (Wu et al., 2023). A study conducted in an agricultural wetland in Iowa found that CH₄ emissions were predominantly generated within the wetland system itself, while N₂O fluxes were significantly influenced by surface water inflows. Nonetheless, the wetland exhibited efficient nitrate (NO₃⁻) transformation, resulting in lower N₂O production compared to surrounding cropland and downstream areas (Ampe et al., 2025). While restored agricultural wetlands vary in size, they are typically shallow, making them more susceptible to rapid warming than deeper waterbodies. From the literature review, a rising temperature and a smaller wetland size generally increase GHG emissions. A warmer temperature also supports algal blooms at lower nutrient levels (Kosten et al., 2009), feeding this positive feedback cycle of an allied attack between climate change and eutrophication (Moss et al., 2011). **Figure 1.1** provides a summary of directional relationships between environmental drivers and GHG emissions discussed in the literature review.

Temperature	↑	↑	↑
Wetland Size	↓	↓	→
Nutrients	↑	→	↑
pH	→	↓	→
Oxygen	↓	↓	→
Organic Matter	↑	↑	↑
	CH ₄	CO ₂	N ₂ O

Figure 1.1: Summary of directional relationships between environmental drivers and GHG fluxes (CH₄, CO₂, and N₂O) synthesized from the literature review. Arrows indicate the general trend reported across studies (↑ increase, ↓ decrease, → no consistent relationship). Overall, most studies exhibited similar directional patterns; however, several papers reported contradictory or context-dependent results.

1.4 Research Gaps

The number of studies investigating GHG emissions from agricultural wetlands has increased globally in recent years (Logozzo et al., 2025; Page et al., 2023; Wu et al., 2023), reflecting a growing recognition of the importance of the contribution of agricultural wetlands to global GHG budgets. There is, however, a noticeable scarcity of research specifically addressing GHG emissions and environmental drivers in restored agricultural wetlands, farm ponds, and agricultural reservoirs in Ontario, which hosts >25 % of Canada’s agricultural land (Government of Canada, 2022). Moreover, many of the reviewed studies above did not simultaneously assess all three major GHGs within the same site, making it challenging to identify specific environmental drivers in well-defined contexts. CH₄ ebullition and oxidation pathways are not

consistently addressed in GHG dynamics studies, and information specifically on CH₄ oxidation in wetlands is particularly scarce.

Studies show conflicting relationships between environmental drivers and GHG emissions, and certain drivers (such as ions like bromide, lithium, and fluoride, as well as stable isotopes) in restored agricultural wetlands remain largely underexplored, leaving a significant gap in understanding their influence on GHG dynamics.

Many studies overlook temporal variability, yet the importance of environmental drivers changes over time and across climate zones, influencing GHG dynamics regionally. The complexity of emissions in restored agricultural wetlands, shaped by multiple interacting drivers, makes it difficult to generalize. Developing localized models for newly constructed wetlands is therefore crucial, as these systems may not yet have reached baseline conditions, and understanding these processes is essential for accurate regional scaling given climate and land-use influences.

1.5 Objectives

This study aims to investigate GHG emissions from restored wetland systems located on or near agricultural lands, also considered farm pond systems. We focus on two of the three major GHG emission pathways - diffusive and ebullitive - while characterizing both spatial heterogeneity (across and within wetlands) and temporal variability (across seasons). Our objectives are to (1) quantify, compare, and interpret spatial and seasonal trends in GHG fluxes across restored agricultural wetlands in southern Ontario; (2) identify the key environmental drivers influencing these fluxes; and (3) discuss the broader considerations of these findings for wetland emissions modeling, management, and monitoring efficiency.

Chapter 2: Greenhouse Gas Dynamics in Restored Agricultural Wetlands

2.1 Background and Context

Wetlands are functional ecosystems that provide numerous ecosystem services, including but not limited to recreation, water filtration, flood control, and carbon storage (Mitsch et al., 2015; Ramsar Convention Secretariat, 2001; Sharma et al., 2021; Valach et al., 2021).

Globally, an estimated 3.4 million km² of wetlands have been lost since 1700, primarily due to conversion to cropland (Fluet-Chouinard et al., 2023). In Ontario, agricultural activity is largely concentrated in the southern region (Statistics Canada, 2022), which has led to extensive wetland loss. By 2019, approximately 68% of Southern Ontario's wetlands had been lost, with the greatest declines occurring in areas of intensive agricultural expansion (Penfound & Vaz, 2022).

The Canadian Government has a variety of funding programs and incentives to support wetland restoration (Environment and Climate Change Canada, 2016, 2021a). The Nature Smart Climate Solutions Fund supports the conservation, restoration, and enhancement of wetlands, peatlands, and grasslands to promote carbon storage. Wetland ecosystems can be strong carbon sequestrators through multiple biogeochemical processes (Villa & Bernal, 2018). The federal government also recognizes the role of wetlands as nature-based solutions beyond climate mitigation, such as reducing nutrient runoff before it reaches sensitive ecosystems like Lake Erie (Environment and Climate Change Canada, 2018). The capacity of wetlands to intercept and retain nutrients from adjacent agricultural landscapes suggests that these ecosystems play a significant role in mitigating harmful algal blooms (HABs), which have re-emerged as a persistent issue in Lake Erie (Michalak et al., 2013; Smith et al., 2019; Steffen et al., 2014). Consequently, the restoration of wetlands within agricultural watersheds is hypothesized to

provide a dual benefit by reducing both the frequency and severity of HABs and contributing to climate change mitigation efforts.

Wetlands, however, are the largest natural source of CH₄, a highly potent GHG (Gedney et al., 2004; Jackson et al., 2020; Rosentreter et al., 2021; Saunois et al., 2025), with a global warming potential approximately 80 times greater than CO₂ over 20 years due to its short atmospheric lifetime and chemical properties (Forster et al., 2021). This makes CH₄ a critical target for short-term climate mitigation efforts (Dlugokencky et al., 2011). A model estimates that by 2100, wetland CH₄ emissions may increase by 50% to 80% driven by climate warming (Koffi et al., 2020).

A restored wetland represents an effort to rehabilitate the ecosystem to its prior state and to re-establish the natural functions it historically provided. The reestablishment of anoxic conditions during wetland restoration results in the production of CH₄ and N₂O (Schuster et al., 2024), thereby complicating the trade-offs between ecosystem service provision and GHG emissions mitigation. With a push to restore wetlands, a meta-analysis estimates the impact will increase CH₄ emissions, especially in the early years following restoration (He et al., 2024).

CH₄ emissions from freshwater aquatic systems have the potential to offset carbon uptake, thereby minimizing the net carbon sequestration benefits of these ecosystems (Bastviken et al., 2011). Considering restored wetlands, the overall GHG balance can vary substantially depending on environmental factors such as hydrology, vegetation, and climate conditions (Y. Zhang et al., 2025).

For example, Schuster et al. (2024) reported that restored freshwater wetlands may not exhibit a net cooling effect until 141 to 525 years after restoration, indicating limited benefits for

short-term climate mitigation goals (Schuster et al., 2024). Similarly, Hemes et al. (2018) found that restored, impounded freshwater wetlands in California—ranging from 5 to 21 years old—emitted enough CH₄ to function as net GHG sources on an annual basis. Their projections suggest these systems may only become net climate sinks after one to two centuries (Hemes et al., 2018). Northern peatland restorations present yet another long-term trajectory: while they initially act as net GHG sources, their warming effect peaks around 50 years post-restoration and gradually transitions to a net cooling influence over timescales of several millennia (Frolking et al., 2006).

In contrast, other studies suggest that net carbon sequestration can occur more rapidly depending on wetland type and climate. Valach et al. (2021) found that highly productive restored freshwater marshes in a Mediterranean climate acted as strong carbon sinks over a 21-year monitoring period (Valach et al., 2021). Likewise, Yang et al. (2020) showed that coastal wetland restorations can become net carbon sinks within just four years, suggesting more favourable outcomes in saline or brackish environments (H. Yang et al., 2020). Most topical, the restoration of seasonal through to permanent prairie pothole wetlands in Canada was found to potentially reduce net GHG emissions over the near term (Badiou et al., 2011).

Continuing, restored wetlands can act as sources or sinks of CO₂ to or from the atmosphere. Once established, restored wetlands generally act as net sinks for CO₂. According to He et al. (2024), most wetlands begin sequestering more CO₂ than they emit approximately four years after restoration. When considering the full GHG balance of wetland ecosystems, studies suggest that CO₂ sequestration can offset CH₄ and N₂O emissions within 40 to 80 years following restoration (Günther et al., 2020; Zou et al., 2022). These findings contrast with Schuster et al. (2024), who estimate that net cooling in freshwater wetlands occurs over a longer

timescale of 141 to 525 years. This discrepancy arises from differences in modelling frameworks, with Schuster employing a more conservative approach. Additionally, vegetation type plays a significant role in influencing CO₂ fluxes in restored wetlands (Silan et al., 2024).

Wetland restoration has been shown to reduce N₂O emissions by 68.6 % (He et al., 2024). This reduction potential is particularly high on agricultural lands (Kasak et al., 2021), however, the presence of residual nitrogen fertilizer in the soils and fluctuating environmental conditions can enhance N₂O production (Bonetti et al., 2022). N₂O emissions in wetlands are regulated by multiple factors, including substrate availability, pH, temperature, and moisture (Gutknecht et al., 2006). Shallow wetlands and areas with water depths less than 9 cm can contribute significantly to system-wide N₂O emissions, with water table levels and temperature serving as key controlling variables (Kasak et al., 2022). Prairie wetlands exhibited minimal or negative N₂O fluxes in this study, suggesting that these wetland types are likely a very small net source of N₂O (Tangen & Bansal, 2022).

Assessing GHG dynamics in newly restored wetlands is essential, particularly in agricultural landscapes where most restoration opportunities are found in Southern Ontario (Statistics Canada, 2022). The region has experienced the greatest loss of wetland area due to conversion to agriculture, resulting in extensive areas with potential for restoration (Ducks Unlimited Canada, 2010; Fluet-Chouinard et al., 2023; Snell, Elizabeth A., 1987). Studies using wetland suitability indices indicate that substantial areas of Southern Ontario are classified as having medium to high suitability for wetland restoration, highlighting significant opportunities for ecological recovery in this region (Medland et al., 2020). However, our understanding of GHG emission dynamics from restored wetland systems in Southern Ontario farmland is limited.

Estimating CH₄ emissions from small waterbodies, such as restored wetlands on or near agricultural lands, remains challenging for several reasons (Saunois et al., 2025). Because these wetlands are often small and pond-like in structure, studies of similar small aquatic systems are particularly relevant. First, these systems exhibit highly variable CH₄ fluxes compared to larger water bodies, making it difficult to generalize findings across sites (Holgerson & Raymond, 2016). Second, there is a limited number of studies focused on CH₄ emissions from restored agricultural wetlands, and existing research often lacks sufficient spatial representation or fails to report statistical uncertainty, further complicating our understanding (Saunois et al., 2025). Third, methodological approaches are not standardized between studies, which introduces additional uncertainty into emission estimates (Lauerwald et al., 2023). Furthermore, ebullition—CH₄ release via bubbling—remains particularly difficult to constrain and model accurately due to its complex and episodic nature, representing an understudied pathway of CH₄ emission from small, shallow aquatic systems (Wik et al., 2016).

This study investigates GHG emissions from restored wetlands on or near agricultural lands, including farm pond systems. We measure diffusive and ebullitive fluxes across multiple locations and seasons to capture spatial and temporal variability. The objectives are to characterize spatial and seasonal GHG flux patterns in restored agricultural wetlands in southern Ontario and identify their key environmental drivers to inform improved modeling and management.

2.2 Methods

2.2.1 Study Design

Study Sites

This study was conducted at seven restored wetlands near London, Ontario, within the Lake Erie Basin (**Figure 2.1**), located across the Thames River, Sydenham River, Kettle Creek, and Catfish Creek watersheds (Page et al., 2023). All sites are situated within agricultural landscapes and receive runoff from surrounding fields: four via both tile and overland flow, and three via overland flow only. The wetlands range in age from about four to eight years since restoration and in basin volumes from 927 to 5,365 m³ (Page et al., 2023). Each wetland holds a standing body of water with an average surface area between 0.14 to 0.78 ha, making them comparable to small ponds that have recently become important in the literature (Holgerson & Raymond, 2016). Although the surface area of each wetland is relatively similar, the contributing area of the wetlands spans an order of magnitude, ranging from 2.2 to 30 ha. Outflow structures vary in complexity across sites, and wetland vegetation has been established at all wetlands. The surrounding agricultural fields cultivate a variety of crops, with some undergoing seasonal rotation while others maintain the same crop throughout the year (see Page et al. 2023 for more site-specific information).

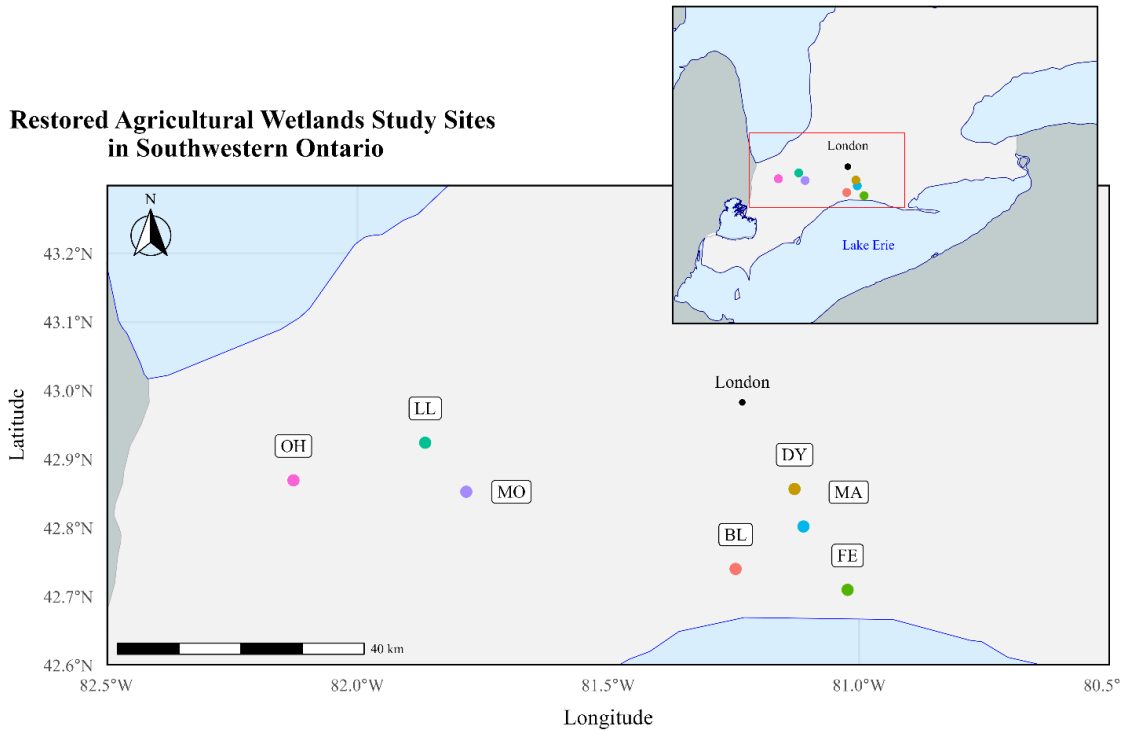


Figure 2.1: Study sites in restored agricultural wetlands of southwestern Ontario, Canada, with an inset showing the regional context. Wetlands are labelled by DUC site ID (Page et al., 2023).

Sampling Strategy

Data was collected during six campaigns spread over one year that covered each season, but with a higher frequency in the warm growing season (June, July, August, October of 2024, and January/February and April of 2025). Wetlands were visited in randomized order within two geographic clusters to minimize bias with sampling at each wetland taking one full day during the campaigns. Atmospheric temperature, pressure, and average wind speed were taken each sampling day using a Kestrel handheld anemometer. Water and dissolved gas samples were taken from three locations: inlet, outlet, and the middle or deepest location (if the deepest location was not in the middle). The inlet of each wetland was typically within the wetland itself, near an inflowing stream or culvert pipe. The outlet sample would be from the culvert pipe if

accessible and running, or the inside of the wetland near the pipe or tile drain location. Vertical profiles of temperature, dissolved oxygen (mg/L and %), specific conductance ($\mu\text{S}/\text{cm}$), and pH were measured using a YSI EXO2 Sonde at the middle/deepest point of each wetland. Ice covered the wetlands in January and February, and YSI profiles were collected after augering a hole.

2.2.2 GHG Sampling Strategy

Dissolved Gas Sampling

Vertical profiles of dissolved GHG (i.e., CH_4 , CO_2 , and N_2O) were measured at the deepest point of each location and surface dissolved GHG was measured at the inlet and outlet of each site. Dissolved gases were sampled using the headspace equilibration method (McAulliffe, 1971), in which a 140 mL syringe was used to equilibrate 100 mL of a water sample with 40 mL of pure nitrogen gas following three minutes of shaking. The 40 mL headspace was transferred to a 12 mL evacuated exetainer, and triplicate samples were taken at each location or depth. Exetainers were analyzed for CH_4 , CO_2 , and N_2O using a Scion Instruments GHG Analyzer (Scion 456-GC), and dissolved gas concentrations were calculated using Henry's Law of Solubility and the true pressure and temperature.

Diffusive GHG Fluxes

Diffusive fluxes of CH_4 , CO_2 , and N_2O were measured at the air–water interface at three locations within each wetland (inlet, middle, and outlet) using floating chambers (28 cm diameter, 10.9 L total volume) connected to gas analyzers in a closed loop system. The chambers were custom-made and equipped with a layer of aluminum foil tape on the surface to reduce heat

gain from solar radiation. The same chamber was used for all measurements, and both the chamber volume and surface area were specific to that individual chamber.

To access the middle location, a small inflatable raft was used. At inlet and outlet locations, chambers were typically deployed from the shoreline. In occasional cases where water was too shallow or inaccessible from the bank, field staff waded into the wetland to find a deeper patch. In these instances, sufficient time was allowed for any sediment-disturbed ebullition to subside before placing the chamber well away from the walking path, ensuring minimal disturbance.

CO₂ and CH₄ concentrations were measured using a Picarro G4301 or an ABB OA-ICOS GLA131, while N₂O was measured with a LI-COR LI-7820. During part of Campaign 3 (August), an ABB OA-ICOS GLA131 series analyzer temporarily replaced the Picarro. In January and February, when temperatures were below the operating range of the Picarro or LGR, a Li-COR LI-7810 was used. Using different analyzers for CH₄ or CO₂ flux does not discredit the flux data calculated, as flux measurements are dependent on the slope of the concentration change rather than the absolute concentration values.

At each sampling location, three chamber measurements (~3 minutes each) were conducted. Fluxes were classified as diffusive if gas concentrations increased linearly over time. Valid flux measurements required a minimum of 40 seconds of linear concentration increase. Sudden, stepwise increases—indicative of captured bubbles—resulted in a reset of the chamber deployment. Floating chambers can inadvertently trap ebullitive bubbles during placement, leading to artificially elevated readings. Any such measurements were excluded from analysis.

Ambient environmental conditions, including air temperature, pressure, and wind speed, were recorded at the time of sampling using a handheld Kestrel anemometer. The molar volume was derived by the Ideal Gas Law utilizing the site's atmospheric conditions at the time of the diffusive flux measurement.

Fluxes (F , $\text{mmol m}^{-2} \text{ day}^{-1}$) were calculated following DelSontro et al. (2016) using the equation: $F = (\frac{\delta C}{\delta t} \times V_{CH} \times 86400) / (V_m \times A \times 1000)$, where $\frac{\delta C}{\delta t}$ is the concentration change over time (ppm s^{-1}), V_{CH} is the chamber volume (L), V_m is the molar volume (L mol^{-1}), and A is the chamber surface area (m^2). A positive F would be indicative of an emission from the water into the atmosphere, while a negative F would indicate gas sequestration from the atmosphere into the water column.

Gas analyzer malfunctions on a few occasions resulted in missing diffusive flux measurements for certain locations and dates, accounting for approximately 5% of CH_4 and CO_2 data and 10% of N_2O data. For the 9 runs on duckweed, 44% were missing for CH_4 and CO_2 data and 0% of N_2O data. Across all campaigns, only about 3% of data could not be collected using either the floating flux measurements or back-calculation. Some missing data for duckweed is due to its absence in certain campaigns.

When flux data were unavailable—such as during periods of excessive bubbling or analyzer malfunction—diffusive emissions were estimated from water concentrations using the equation from Sieczko et al. (2016): $F_{gas} = k * (C_w - C_{eq})$, where F_{gas} is the gas flux ($\text{mmol m}^{-2} \text{ day}^{-1}$), k is the gas exchange velocity (m day^{-1}), C_w is the observed concentration of dissolved gas in the water (mmol L^{-1}), and C_{eq} is the gas concentration of surface water in equilibrium with the atmosphere (mmol L^{-1}).

The gas transfer velocity (k) was derived by back-calculating from valid, site- and location-specific flux measurements. For cases where flux data were missing, a representative k value was selected based on the back-calculated k from the flux measurement with the closest matching dissolved gas concentration and campaign at that site and location. C_{eq} was calculated as the product of the partial pressure of atmospheric CH_4 and the temperature-corrected Henry's Law constant for CH_4 . An average C_{eq} of duplicate readings were utilized.

Ebullitive CH_4 Fluxes

Ebullitive CH_4 emissions were measured using suspended, custom-made bubble traps in two sizes: small (21 cm diameter) and large (66 cm diameter). Large traps were constructed from circular plastic trays cut once along the radius, overlapped and glued together, then fitted with flexible PVC couplings and a threaded connector to hold an inverted 500 ml glass bottles. Small traps were created by inserting a syringe closed with a stopcock through a pre-made funnel and sealing binding it to the funnel with waterproof sealant. Weights—hockey pucks or tubing filled with sand—were attached around the trap perimeter to submerge and stabilize it, while foam flotation tubes suspended the funnels 0.5 to 1 m below the water surface. Traps were deployed in a transect hanging from a rope that stretched across the wetland's deepest location. Five traps were deployed per site: two small with one on each end of the transect in the shallowest locations and three large traps equally spaced between the two small traps. A second parallel rope served as a guide for boat access to avoid disturbing the traps or sediment. Traps were deployed for 24 hours, and if sufficient gas accumulated, triplicate samples were collected. For small traps, a sampling syringe was directly connected to the funnel syringe to extract gas. For larger traps, the glass bottle was carefully unscrewed from the funnel while remaining partially submerged to keep the accumulated gas within the bottle. Then a sampling syringe with an 8-inch tubing was

submerged under water to fill the tubing with water before guiding it into the collection bottle. Accumulated gas was drawn from this bottle while it remained partially submerged to avoid atmospheric contamination. For both sampling methods, the gas was transferred to a dry syringe before being injected into pre-evacuated exetainers. The volume and CH₄ concentration of collected gas were used to estimate ebullitive CH₄ flux, with gas concentrations analyzed using a Scion Instruments GHG Analyzer. Ebullitive CH₄ flux (ml day⁻¹ cm⁻²) was calculated as the product of the total gas volume collected in the trap ($V_{\text{collected}}$) and the fraction of CH₄ in the sample (C_{CH_4}), divided by the trap area (A_{trap}) and deployment time (t):

CH₄ Flux (ml day⁻¹ cm⁻²) = $\frac{V_{\text{collected}} \times C_{\text{CH}_4}}{A_{\text{trap}} \times t}$; where $V_{\text{collected}}$ is in milliliters, C_{CH_4} is unitless (e.g., 0.02 for 2 %), A_{trap} is in cm², and t is in days. As the results of the CH₄ concentrations have not yet been received, this paper will focus on discussing the volume of gas released.

Statistical Analysis

GHG flux data were analyzed in R (version 4.4.2) to assess differences among and within wetland sites and identify statistical outliers. Prior to analysis, data were filtered to exclude duckweed (DW) measurements; for Campaign 5, only “Ice” and “Snow” locations were retained for the Middle location to maintain consistency between environmental driver analysis.

For each gas (CH₄, CO₂, and N₂O), nonparametric analyses were conducted using the Aligned Rank Transform (ART-ANOVA) to evaluate site-level effects, followed by post-hoc pairwise comparisons using Holm-adjusted p -values. Results were cross-validated with Kruskal–Wallis tests and Dunn’s multiple comparisons. Outliers were identified within each site using both Grubbs’ test and interquartile range (IQR) or median absolute deviation (MAD) criteria.

Additionally, Mann–Whitney U tests were used to compare Site MA against all other sites combined.

2.2.3 Water Chemistry

Water samples were collected from the inlet, outlet, middle surface, and middle bottom of each wetland. Samples were stored in dark bottles to prevent photodegradation. Samples were filtered within 24 hours of collection (0.2 μm filter) and kept in the refrigerator or freezer until processing. Particulate organic matter (POM) and chlorophyll-*a* filters were prepared within hours of water collection and stored in the freezer until processing.

Water samples were analyzed for dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), cations (Li^+ , Na^+ , K^+ , Mg^{2+} , Ca^{2+} , NH_4^+) and anions (F^- , Cl^- , Br^- , NO_2^- , NO_3^- , PO_4^{3-} , SO_4^{2-}), total phosphorus (TP), total nitrogen (TN), soluble reactive phosphorus (SRP), dissolved phosphorus (DP), chlorophyll-*a* and phaeophytin, stable water isotopes ($\delta^{18}\text{O}$, $\delta^2\text{H}$), and particulate organic matter (POM) isotopes ($\delta^{13}\text{C}$, $\delta^{15}\text{N}$).

DOC and DN were measured on an Analytik-Jena Multi N/C 3100 analyzer at Wilfrid Laurier University. DOC was run as non-purgeable organic carbon on filtered water. DIC was analyzed using a Dohrmann DC-190 Total Carbon Analyzer in inorganic carbon mode at the University of Waterloo's Environmental Geochemistry Lab. Cations and anions were analyzed via ion chromatography (Metrohm 940 IC Vario) at the Urban Water Research Centre, Toronto Metropolitan University.

Phosphorus samples were analyzed using a standard protocol (Wetzel and Likens, 2000) at the GRIL-Université du Québec à Montréal (UQAM) analytical laboratory. Nitrogen samples were analyzed at the GRIL- UQAM analytical laboratory with a continuous flow analyzer (OI

Analytical Flow Solution 3100 ©) using an alkaline persulfate digestion method, coupled with a cadmium reactor, following a standard protocol (US EPA Method 353.2, Patton and Kryskalla, 2003).

Chlorophyll-*a* and phaeophytin samples were prepared by filtering water through 0.45 µm GF/F filters, extracted in 90% acetone, and quantified using a Turner Designs TD 10AU Fluorometer at the University of Waterloo Wetland Science Laboratory.

Stable isotopes of water and POM were analyzed at the University of Waterloo Environmental Isotope Lab. $\delta^{18}\text{O}$ and $\delta^2\text{H}$ of water was measured using a Triple Liquid Water Isotope Analyzer (LGR T-LWIA-45-EP) following Lis et al. (2008). $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ of POM were measured using an IsoPrime isotope ratio mass spectrometer coupled to a Costech elemental analyzer following filtration through 0.45 µm QMA filters and drying. Results were normalized to international standards and reported with typical analytical errors of $\pm 0.2\text{‰}$ for $\delta^{13}\text{C}$ and $\pm 0.3\text{‰}$ for $\delta^{15}\text{N}$.

2.2.4 Environmental Driver Analysis on Diffusive Emissions

Kendall Correlation

To assess relationships between GHG diffusive fluxes and environmental parameters during summer, Kendall rank correlations were applied using data from Campaigns 1 to 3 (June – August) at middle wetland locations. These campaigns were selected to control temperature variability, as they were the three campaigns closest in temperature; therefore, seasonal variability is minimized, allowing us to identify which variables other than temperature influence GHG emissions. No duckweed measurements were included in this analysis. Kendall's tau was chosen due to non-linear relationships in the data and its robustness with small sample sizes ($n \approx$

21). Correlations were computed using the dplyr and stats packages in R, with missing values handled on a per-variable-pair basis.

Environmental parameters included in the analysis are outlined in **Table 2.1**. Variables directly related to GHG concentration measurements were excluded from the analysis to avoid circularity. Site was excluded from the Kendall correlation analysis because it is a categorical variable and cannot be meaningfully used in Kendall's rank-based correlation, which requires numeric data.

Table 2.1: Environmental parameters included in the Kendall correlation analysis, grouped by category.

Category	Variable Name
<i>Water Chemistry</i>	Dissolved Oxygen Surface (%)
	Dissolved Oxygen Depth (%)
	Water Temperature (°C)
	pH
	Specific Conductivity (µS/cm) (SPC)
<i>Stable Water Isotopes</i>	Oxygen-18 Isotope
	Hydrogen-2 Isotope
<i>POM Isotopes</i>	POM Carbon-13 Isotope
	POM Nitrogen-15 Isotope
	C:N Ratio
<i>Physical Parameters</i>	Max Depth (m)
	Secchi Depth (m)
<i>Nutrients</i>	Dissolved Phosphorus (µg/L) (DP)
	Total Nitrogen (ppm) (TN)
	Soluble Reactive Phosphate (µg/L) (SRP)
	Total Phosphorus (µg/L) (TP)
	Dissolved Organic Carbon (mg/L) (DOC)
	Dissolved Nitrogen (mg/L) (DN)
	Chlorophyll- <i>a</i> (µg/L)
	Phaeophytin (µg/L)
	Dissolved Inorganic Carbon (ppm C/L) (DIC)
	Nitrate (ppm)
	Nitrite (ppm)
	Fluoride (ppm)
	Chloride (ppm)
<i>Major Ions</i>	Bromide (ppm)
	Lithium (ppm)
	Sodium (ppm)
	Ammonium (ppm)
	Potassium (ppm)
	Calcium (ppm)
	Magnesium (ppm)
	Phosphate (ppm)

Random Forest

Random Forest regression models were applied to identify key environmental drivers of diffusive CH₄, CO₂, and N₂O fluxes, incorporating all seasonal data beyond the summer-only Kendall correlation analysis. Duckweed measurements were included by creating a numerical indicator of duckweed presence, and only winter measurements taken on ice or snow were included (prior to augering). Different sets of predictor variables were selectively included or excluded to assess their influence on model performance. Predictors included physicochemical parameters, nutrient concentrations, isotopic signatures, and site metadata (see **Table 2.1**).

For each gas, initial Random Forest models were fit to rank predictors by mean decrease in accuracy (%IncMSE), with higher %IncMSE indicating greater importance. The top 10 predictors were visualized as horizontal bar plots to highlight the most influential environmental drivers. To assess the role of wetland *Site*, models were run both with and without *Site* as a categorical predictor, allowing comparison of the top predictors and evaluation of how much *Site* contributed to explaining variation in diffusive GHG fluxes.

To evaluate the predictive power of Inlet and Outlet measurements for Middle diffusive fluxes, the Middle fluxes were treated as response variables and the corresponding Inlet and Outlet measurements as predictors. The dataset was cleaned by removing variables measured only at Middle locations (e.g., YSI variables) and excluding predictors with fewer than five non-missing values. Random Forest models were trained on Inlet or Outlet predictors and tested against observed Middle fluxes to calculate R² and RMSE. For comparison, baseline models used only Middle measurements as predictors. This approach quantified the relative ability of Inlet and Outlet chemistry to explain Middle GHG fluxes.

2.3 Results

Summary of Environmental Conditions Across Wetlands

Sampling occurred across four seasons with air temperatures ranging from -11.8°C to 36.5°C , covering the typical climate gradient experienced in Southern Ontario. Observed water temperatures reflected a similar wide range in our shallow systems (surface water = -0.3°C to 32.3°C). The highest average air temperatures occurred during the summer (27 to 29°C). In winter, the average air temperature reached a low of -7°C . Dissolved oxygen (DO) saturation at the surface ranged from 0% to 171.9%, while at depth, a smaller range from 0% to 126.6% was observed. Specific conductance (SPC) ranged from 57.3 to 944 $\mu\text{S}/\text{cm}$, and pH values ranged from 6.62 to 9.42. Air pressure ranged from 973.7 to 1006.7 hPa. Observed wind speeds covered a typical range for the region, from no wind, particularly during warmer months, to $>8\text{ m/s}$.

The winter campaign, conducted in January and early February, had an average air temperature of -6.84°C , with a minimum of -11.8°C . Winter recorded the lowest water temperatures at both the surface and the depth of the wetland. Surface dissolved oxygen levels were low during winter, like those in July (Campaign 2). Dissolved oxygen saturation ranged from 30.4% to 107.7% just below the ice, and from 0% to 83% at depth. pH remained stable year-round, with no significant changes seen in winter. The highest SPC was observed at depth during winter (**Figure 2.2**).

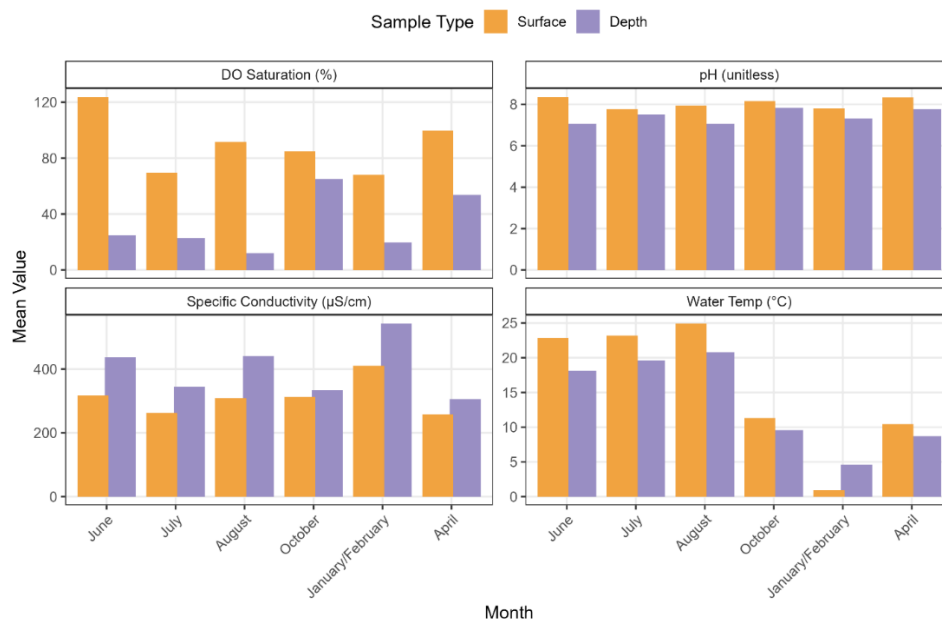


Figure 2.2: Bar plots showing the average surface (orange) and above bottom (i.e., depth in purple) water quality parameters measured at the middle location across multiple campaigns (1 in June 2024; 2 in July 2024; 3 in August 2024; 4 in October 2024; 5 in January/February 2025; 6 in April 2025). Values represent mean water temperature (°C), specific conductivity (µS/cm), pH, and dissolved oxygen (DO) saturation (%).

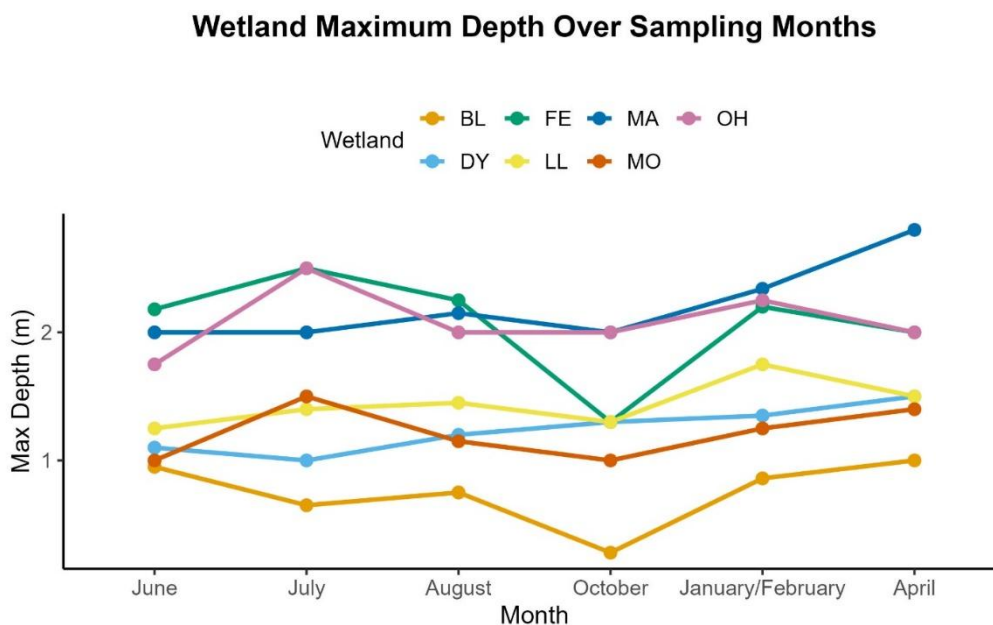


Figure 2.3: Maximum water depth measured across wetland sites over six monitoring campaigns.

Maximum water depths for 3 of the wetlands ranged from 1.5 – 2.5 m, another 3 ranged from 1-1.5 m and one site (BL) was only a few centimeters deep in fall and ~1 m at maximum (**Figure 2.3**). Some wetlands displayed minimal depth variation throughout the year, while others experienced larger fluctuations; however, no wetland ever went dry during the sampling period. Shallowest wetland depths occurred in the fall (October), following a loss of 0.15 to 0.95 m from August. Ice thickness ranged from approximately 10 to 20 cm.

Average nutrient concentrations and ion concentrations per wetland and campaign are in Appendix A1 and A2, respectively. Total phosphorus (TP) ranged from approximately 16 to 1060 $\mu\text{g/L}$, while soluble reactive phosphorus (SRP) ranged from 2 to 248 $\mu\text{g/L}$, comprising roughly 12–23% of TP throughout the year. SRP peaked at Site MA during January and February, whereas patterns were more stable across other sites, with a slight increase during July. Total nitrogen (TN) ranged from 0 to 46 ppm, dissolved inorganic carbon (DIC) from 7 to 70 ppm C/L, and non-purgeable organic carbon (NPOC) from 3 to 9 mg/L over the same period. Major ion concentrations varied among wetland sites, reflecting differences in hydrology and potential anthropogenic inputs such as road salt or agricultural runoff. Among all measured ions, calcium exhibited the highest mean concentration across wetland sites over the sampling period. Most of the systems would be considered eutrophic to hyper-eutrophic based on the range in TP, with SRP comprising up to a quarter of TP. TN and DOC also indicate that these are highly productive systems.

2.3.1 Diffusive GHG Flux Results

Six of the sites (BL, DY, FE, LL, MO, OH) exhibited similar orders of magnitude for diffusive emissions of all three GHGs, with the exception being site MA, which displayed the highest fluxes for all gases (**Figures 2.4, 2.5, 2.6**). MA is a distinct outlier in this set of study

wetlands. Annually, diffusive CH_4 and CO_2 fluxes spanned over 2 orders of magnitude, but only by ~ 1.5 – 1.7 when excluding Site MA (duckweed locations excluded). N_2O fluxes showed the greatest variability, spanning just about 3 orders of magnitude overall and dropping to ~ 1.8 when Site MA was removed. This order-of-magnitude analysis excluded extreme values by considering only the 5th–95th percentiles, included MA flux values when duckweed was removed, and included only winter fluxes measured on top of ice or snow (i.e., not from an ice hole). Therefore, MA fluxes may be overestimated relative to when duckweed coverage could act as a flux barrier, and the winter estimates could be underestimating if any freeze–thaw events occurred during winter.

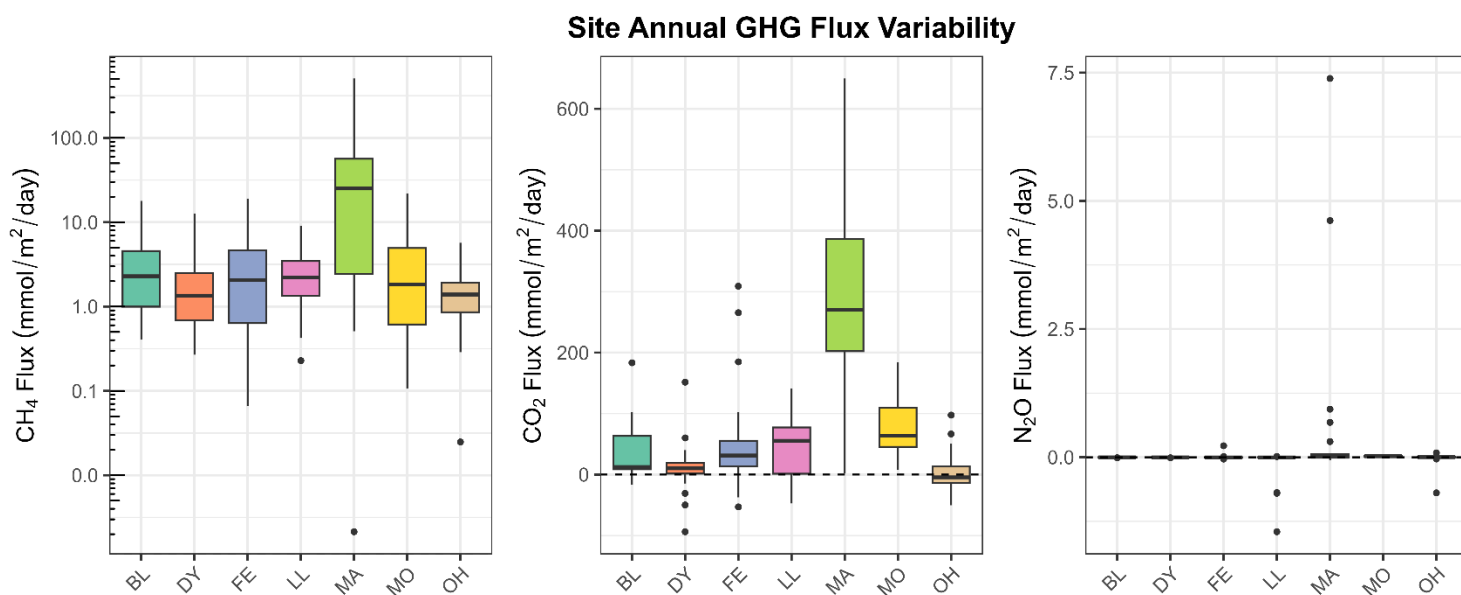


Figure 2.4: Boxplots showing the overall variability of diffusive GHG fluxes (CH_4 , CO_2 , N_2O) across wetland sites. Data includes all inlet, middle, and outlet locations combined for each site over 6 campaigns spanning one year. Measurements taken on duckweed, snow, ice, and after augering a hole in winter are included. N_2O fluxes were measured at each sampling, but the majority were close to zero and hence hard to visualize here (*see Figure 2.6 for more detail*). Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within $1.5\times$ the interquartile range. Outliers beyond this range are shown as individual points.

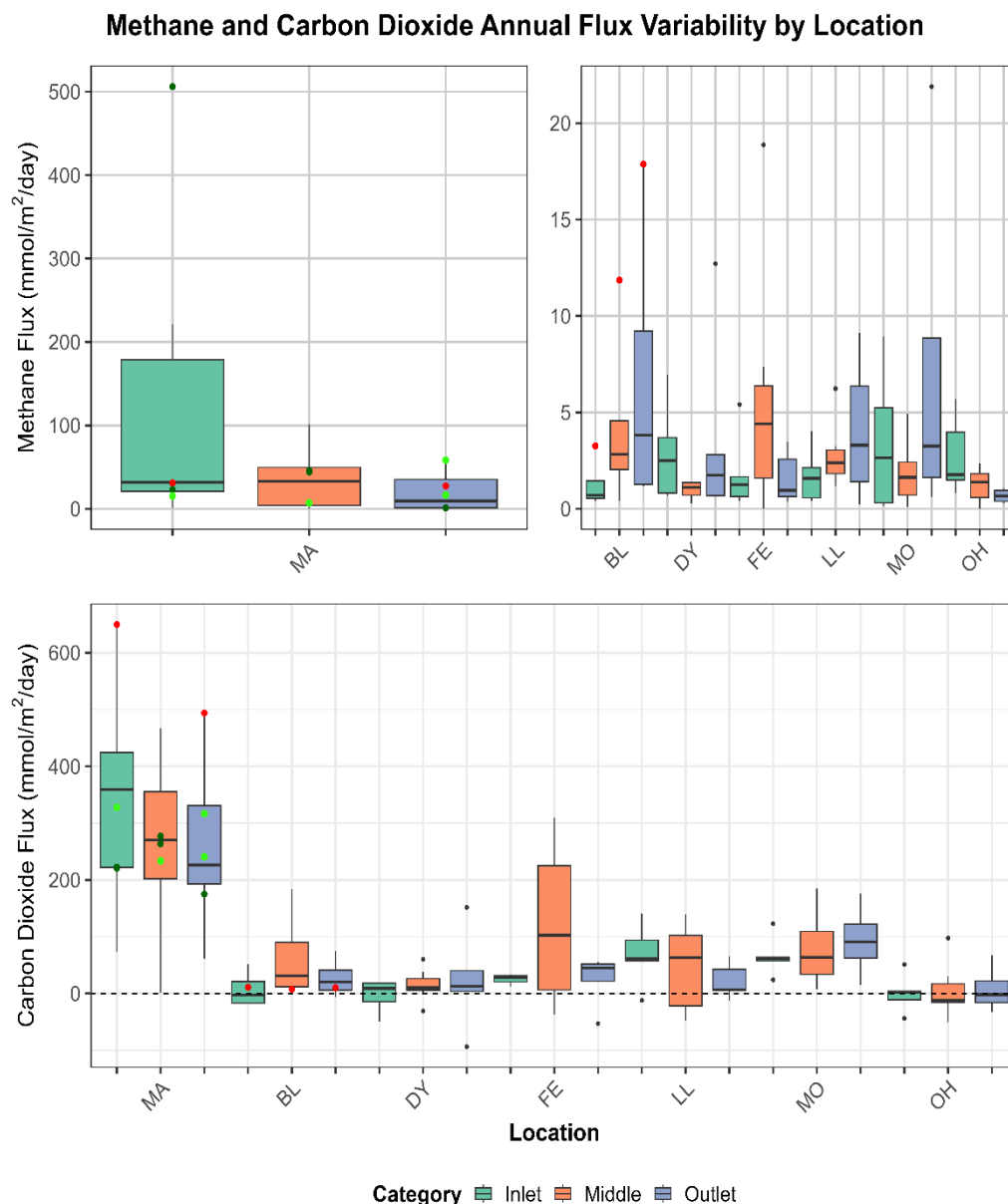


Figure 2.5: Boxplots of CH₄ (top) and CO₂ (bottom) fluxes (mmol m⁻² day⁻¹) by site and location category across the annual sampling period. Winter measurements include only ice and snow fluxes, representing the minimum fluxes expected during the coldest season. Winter measurements were only taken at the middle location. Red points indicate back-calculated data used when direct chamber measurements were not feasible. Duckweed locations at site MA are represented with a dark green point. Neon-green points represent back-calculated duckweed locations. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within 1.5× the interquartile range. Outliers beyond this range are shown as individual points.

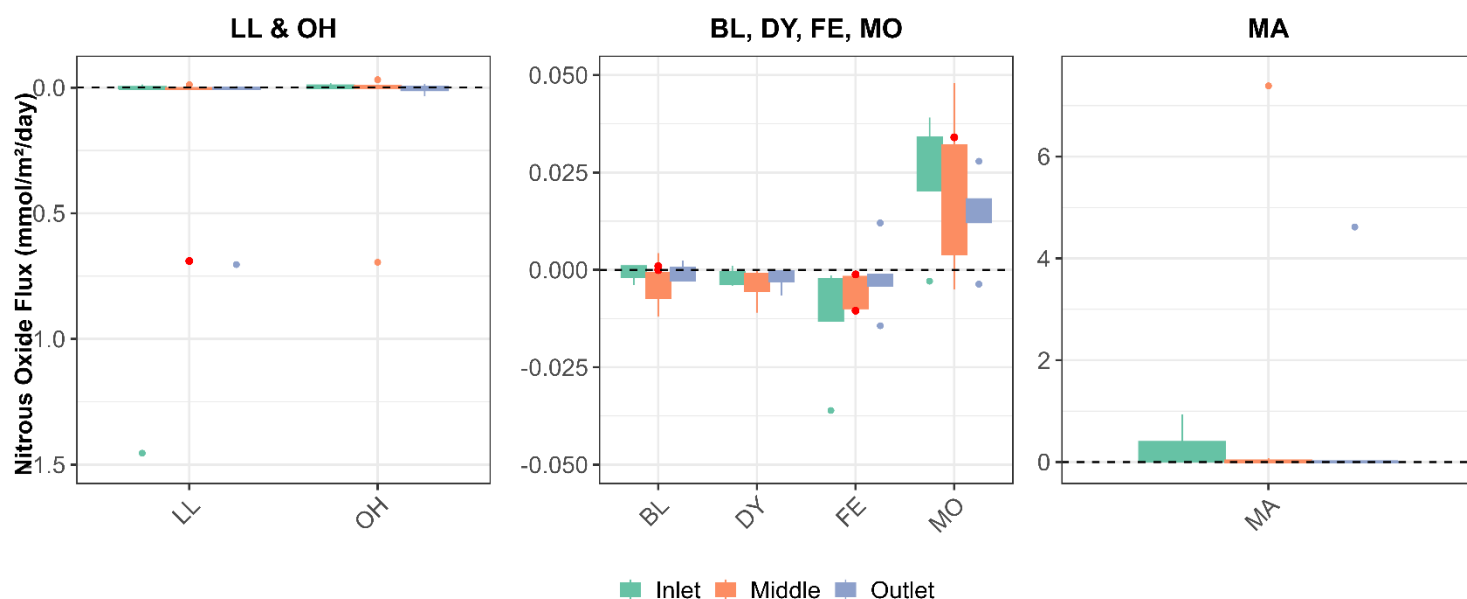


Figure 2.6: Boxplots display N_2O fluxes ($\text{mmol m}^{-2} \text{day}^{-1}$) by site and location category across the annual sampling period. Duckweed locations at site MA were excluded. Winter measurements include only ice and snow fluxes from the middle location, representing the lowest fluxes during the coldest season. Red points indicate back-calculated values used when direct chamber measurements were infeasible. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within $1.5 \times$ the interquartile range. Outliers beyond this range are shown as individual points.

Diffusive GHG Flux Spatial Variability

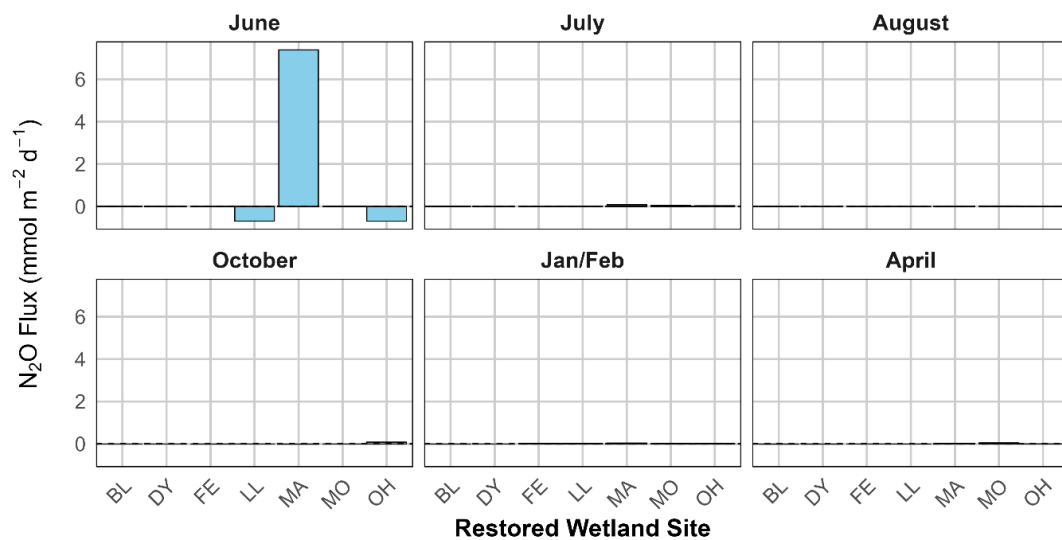
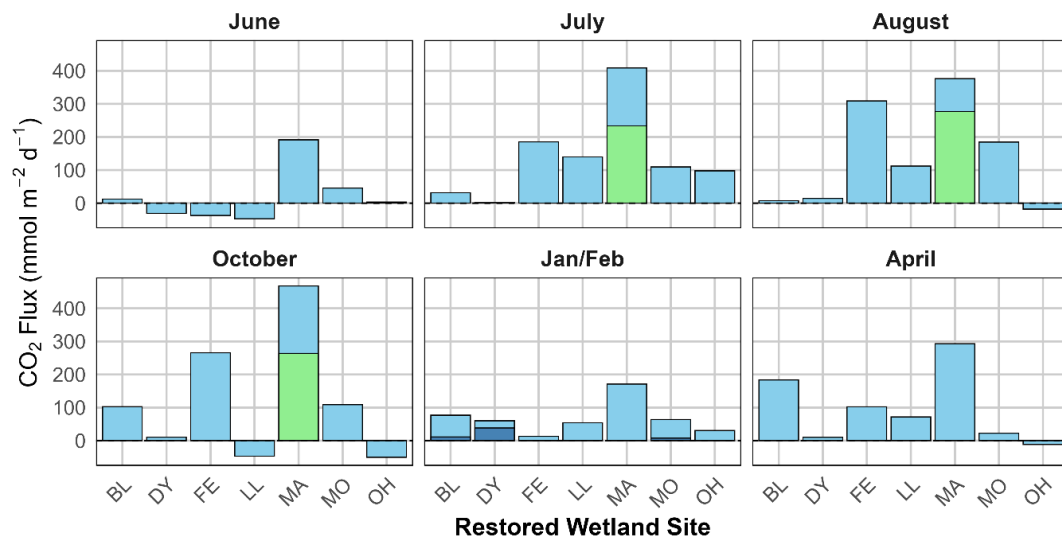
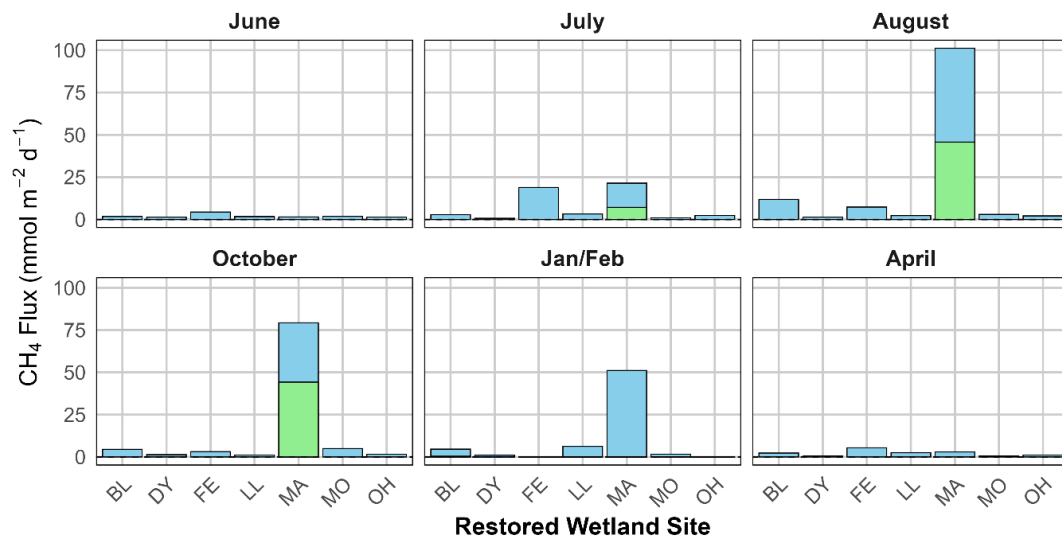
Diffusive CH_4 fluxes across all wetlands ranged from 0 to $221.7 \text{ mmol/m}^2/\text{day}$, with all wetlands acting as CH_4 sources year-round, including winter (see Appendix A3). MA had the highest mean observed emissions and standard deviation ($46.8 \text{ mmol/m}^2/\text{day}$, $\text{SD} = 65.6$), whereas the six other sites ranged in mean emission from 1.7 to $4.0 \text{ mmol/m}^2/\text{day}$ ($\text{SD} 1.5 - 5.5$). The standard deviation indicates that LL and OH were the least variable sites, while MA is the most variable. Diffusive CO_2 fluxes ranged from -93.8 to $649.7 \text{ mmol/m}^2/\text{day}$, acting as a source or sink depending on site and campaign (see Appendix A3), but the majority of measurements were positive. Wetland OH had the lowest mean observed annual flux ($2.8 \text{ mmol/m}^2/\text{day}$) because of several negative flux measurements, indicating significant CO_2 uptake at this site. In

addition, Site OH was the least variable site ($SD = 40.4$). As with CH_4 , MA emitted the highest observed mean diffusive CO_2 flux and was also the most variable (mean = $315.4 \text{ mmol/m}^2/\text{day}$; $SD = 177.8$). MO was the second-highest CO_2 emitter and the only other site besides MA at which no CO_2 uptake was observed during any campaign. Diffusive N_2O fluxes ranged from -177.7 to $815.9 \text{ } \mu\text{mol/m}^2/\text{day}$, indicating minor uptake or emission (see Appendix A3). All wetlands (except MA) had very low mean observed emissions annually (under $20 \text{ } \mu\text{mol/m}^2/\text{day}$), indicating that these systems do not strongly emit N_2O .

Between wetlands, CH_4 flux did not significantly vary by site (ART-ANOVA: $F_{6,101} = 1.83$, $p = 0.101$, Kruskal-Wallis: $\chi^2_6 = 10.47$, $p = 0.106$); however, post-hoc tests indicate that comparisons involving the MA site were the only ones approaching significance. Mann-Whitney U test ($W = 1072$, $p = 0.0037$) confirmed that MA's CH_4 flux distribution significantly differed from other sites. CO_2 flux varied significantly ($F_{6,101} = 12.28$, $p < 0.001$; Kruskal-Wallis: $\chi^2_6 = 45.13$, $p < 0.0005$). Post-hoc tests identified the highest emitter MA as distinct from all five sites ($p < 0.001$) except the second highest emitter MO, and that the wetland that exhibited the most uptake, OH, also differed from MO ($p = 0.0002$) and FE ($p = 0.0440$). DY also differed significantly from MO ($p = 0.0024$). The Mann-Whitney U test exhibited MA as having a unique CO_2 flux distribution from other sites ($W = 1350$, $p < 0.001$). N_2O flux varied significantly (ART-ANOVA: $F_{6,117} = 4.4$, $p < 0.001$; Kruskal-Wallis: $\chi^2_6 = 22.66$, $p < 0.001$). Site MA had the largest range (-0.02 to 7.4) and the highest recorded flux (7.4), LL and OH had the lowest mean annual fluxes, and Dunn's pairwise comparison test showed MO differed from DY, FE, and LL ($p < 0.13$). Statistical analysis in this section excluded duckweed measurements, and for winter, only snow and ice measurements were considered.

Within wetlands, diffusive CH₄ fluxes were generally within one order of magnitude among locations across sites and campaigns (1–1.4), except MA in June (2.0). CO₂ fluxes were also within one order of magnitude, except at DY in July (1.02), FE in August (1.4), OH in October (1.4), and LL in April (1.1). N₂O fluxes showed similar consistency (1–1.1), with higher variation at FE in June (1.3), MA in July (1.5), and OH in June (2.3). For each gas, localized emission hotspots were observed, with some middle locations exhibiting the highest and others the lowest fluxes. Overall, Kruskal-Wallis tests indicated no significant differences among inlet, middle, and outlet locations within any wetland during each campaign for any GHG (all $p > 0.05$).

Diffusive GHG Flux Seasonal Variability



■ Middle ■ Snow/Ice ■ Duckweed Middle

Figure 2.7: Seasonal variations in diffusive CH₄, CO₂ and N₂O emissions across restored wetland sites in the middle location. Light blue bars represent fluxes from the middle location of each wetland, including the ice hole drilled in January and February. Fluxes taken on top of snow/ice or duckweed are distinguished by dark blue and green bars, respectively. Dark blue bars for ice and snow measurements are present but not visible at this scale, indicating extremely low emissions of CH₄, CO₂, and N₂O from snow/ice at those sites. Note that majority of N₂O fluxes are near zero and thus not quite visible at the reported scales.

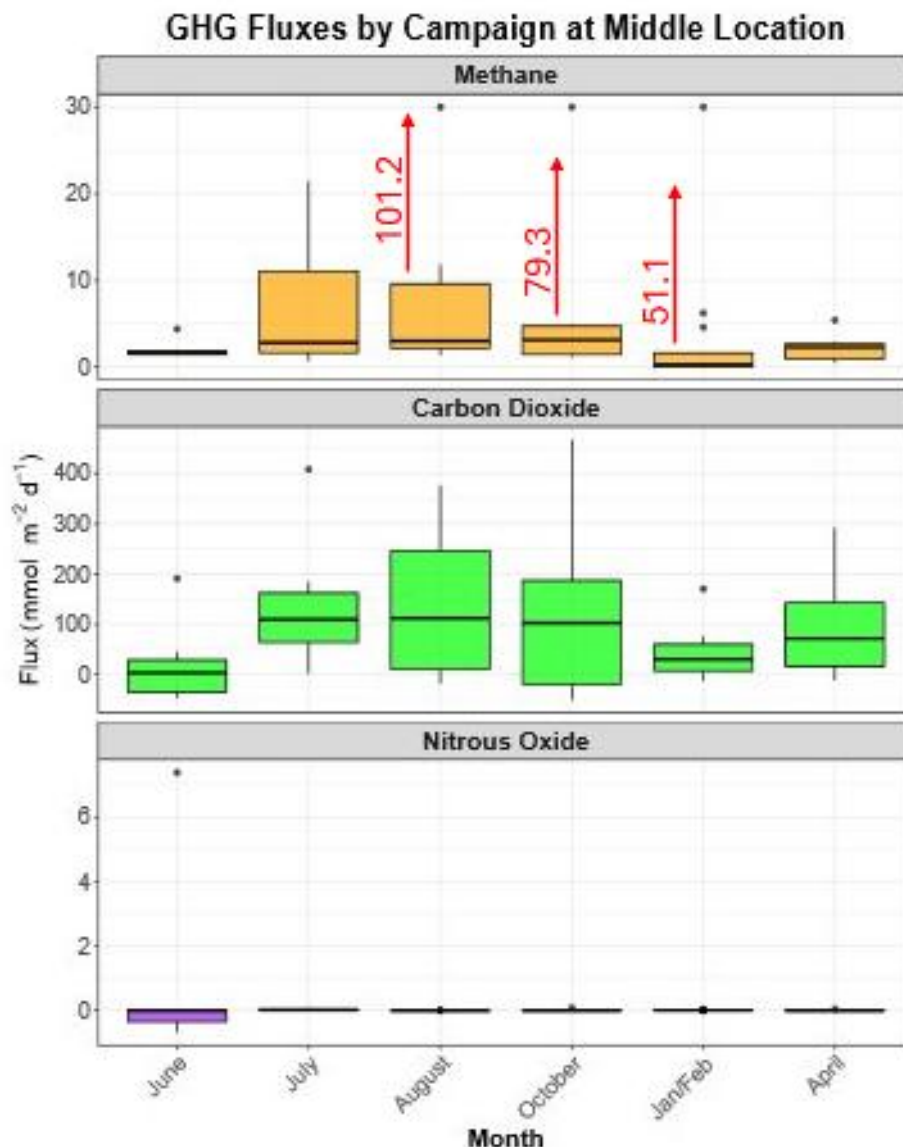


Figure 2.8: Boxplots of CH₄, CO₂, and N₂O fluxes (mg m⁻² d⁻¹) across six sampling campaigns at the middle locations. All measurements were included to show overall variability. CH₄ fluxes have been visually zoomed to improve clarity, while extreme values remain included in the calculations. A red arrow indicates the maximum value for each month in the CH₄ panel that is

not visualized. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within $1.5\times$ the interquartile range. Outliers beyond this range are shown as individual points.

Diffusive CH_4 flux varies by month, with the highest emissions and variability in August (SD = 51.8, IQR = 7.6) (Appendix A4). For winter, only ice and snow measurements were included as those were the realistic conditions on the site at that time. No duckweed measurements were included. Visually, the data suggests an annual cycle, with low emissions at the start of the growing season, a rise through the summer, a decline in the fall, a sharp drop in winter, and a return to low levels at the onset of the following growing season (**Figure 2.8**). In winter, diffusive flux measurements taken over snow and ice showed the lowest recorded emissions, representing the realistic flux from the wetland water body under continuous frozen conditions without freeze-thaw events. The light blue bar in January/February represents flux measurements taken from open water after drilling through the ice, capturing the effect of periodic ice melting events during the winter (**Figure 2.7**). Winter had the lowest recorded emissions and variability (SD = 0.44).

The highest CO_2 emissions occurred in October, but were very similar in July and August (mean = 116.2, 107.1 and 103.3 mmol/m²/day respectively), while the lowest were observed during winter when considering over snow/ice fluxes (dark blue bars, **Figure 2.7**). Interestingly, emissions from the ice holes (light blue bars, **Figure 2.7**) indicated that thawing events occurring during winter could lead to emissions comparable to those in April (**Figure 2.7**). Early in the growing season (i.e., June) shows some sites actively sequestering CO_2 . As temperatures rise through July and August, most sites exhibit high emissions, except for OH in August, which continues to uptake. In the fall, emissions generally stay elevated at most sites, though LL and OH continue to act as carbon sinks. Winter fluxes are the lowest across the year, but not all sites

show near-zero emissions; notably, BL, DY, and MO have higher emissions on snow and ice during winter compared to other sites. In April, despite cooler temperatures, emissions resemble those of August, with Site BL even exhibiting higher emissions in April than in August. CO₂ measurements display greater variability than CH₄, reflected in larger standard deviations each month, with October recording the highest standard deviation (SD = 190.5).

As the majority of seasonal N₂O fluxes were close to zero (either negative or positive), the most notable aspects of the seasonal N₂O emissions were the extreme emissions from MA in June but also significant uptake in LL and OH also in June.

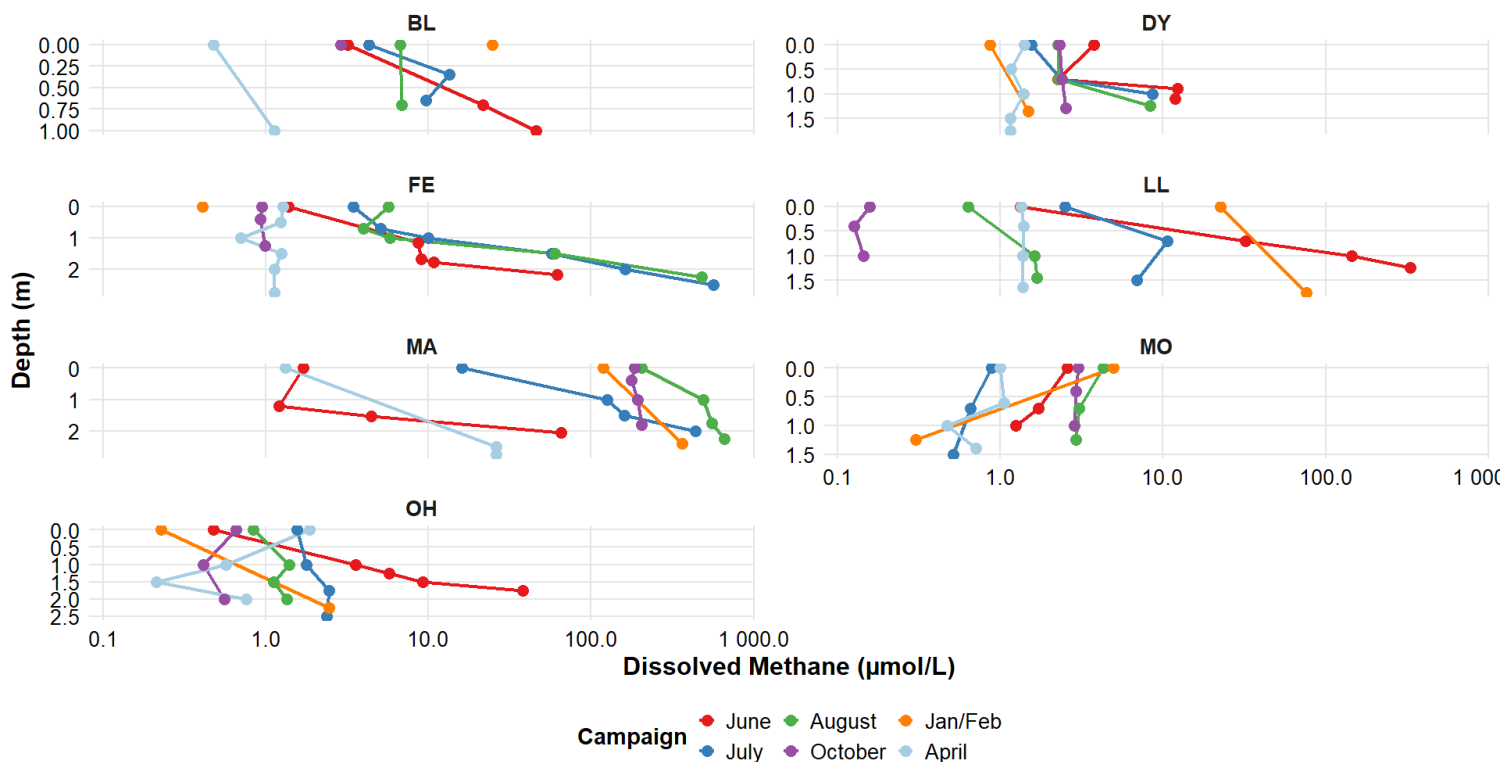


Figure 2.9: Dissolved CH₄ profiles at each restored wetland site across all sampling seasons. In October, wetlands exhibited their lowest maximum water depths (at BL, only surface samples were collected). During the January/February campaign, winter conditions compromised samples at BL and FE, resulting in profiles with only surface or limited depth measurements.

Elevated concentrations of dissolved gases were recorded near the bottom of the water column during winter in some wetlands (MA, LL, OH), occasionally exceeding fall levels, indicating gas accumulation beneath the ice cover (**Figure 2.9**). Dissolved gases were generally stratified during the summer, despite the shallow depths of the wetlands, and became well-mixed in the fall. In winter, stratification re-emerged in all wetlands, except at BL and FE, where data were unavailable due to sampling failures. MO stands out as having extremely unique dissolved gas profiles, having lower concentrations at depth throughout the entire year.

2.3.2 Ebullitive GHG Flux Results

Ebullitive GHG Flux Spatial Variability

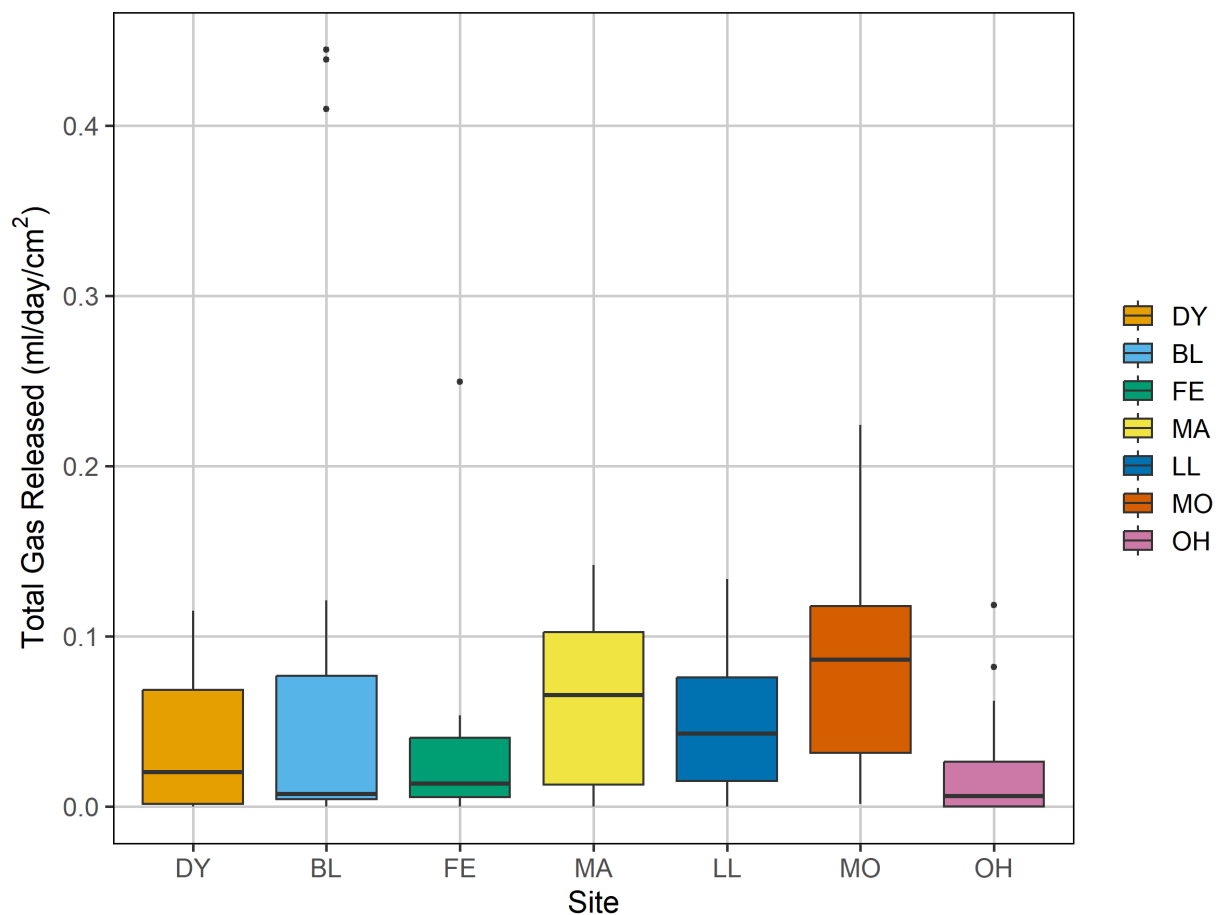


Figure 2.10: Boxplots showing the total gas released (ml/day/cm²) across sampling sites. Each box represents the distribution of measured gas release per site encompassing all funnels on the transect. In winter, there were no measurements taken. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within 1.5× the interquartile range. Outliers beyond this range are shown as individual points.

Ebullitive gas release varied across campaigns and sites, with average values ranging from near 0 up to 0.444 ml/day/cm² (Appendix A5 and A6). Site BL consistently showed some of the highest fluxes, particularly in Campaigns 2 and 3, while MO also exhibited elevated emissions in campaigns 2, 4 and 6 (**Figure 2.10**). Outlier points in Site BL may be due to its fluctuating water depth through the year, resulting in pressure changes. Sites LL and OH generally had lower average fluxes, though some individual measurements exceeded typical ranges, as indicated by MAD and IQR outlier analyses. Site OH was consistently one of the deepest wetlands in contrast to BL. Several samples were identified as outliers, indicating localized peaks and variability in gas release, with some observations noting funnel placement or sampling conditions that could have affected the measurements. Overall, gas release showed both spatial (site-to-site) (ART-ANOVA: $F_{6,139} = 42430.9$, $p = 0.0003$) and temporal (campaign-to-campaign) variability, with lower emissions in October and April compared to the summer months (June, July, August).

Ebullitive GHG Flux Seasonal Variability

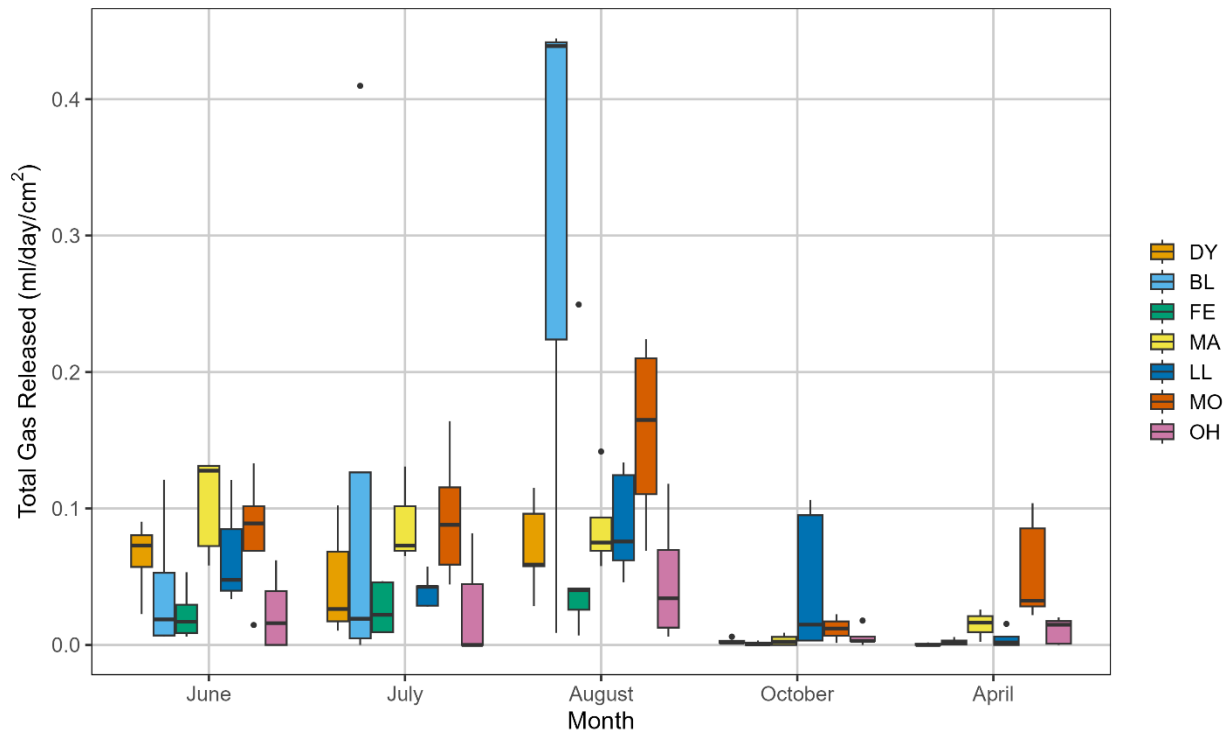


Figure 2.11: Seasonal variations in ebullitive emissions across restored wetland sites. Fluxes are shown from all locations across the transect. No measurements were taken during winter. Each box represents the interquartile range, the horizontal line indicates the median, and whiskers extend to the minimum and maximum values within $1.5 \times$ the interquartile range. Outliers beyond this range are shown as individual points.

The highest mean release of gas occurred in August ($0.107 \text{ ml/day/cm}^2$). October and April released the least amount of gas throughout the year; however, we can assume that in winter, there was also low bubble release due to the ice layer on top of the wetland (**Figure 2.11**). There is greater variability in August ($SD = 0.11$) than in April ($SD = 0.025$), indicating that during these cooler months, gas release is more consistently low. The largest IQR range was recorded for August, showing a wider spread of gas release values. This range is highly influence by BL in August, most likely due to BL being the shallowest wetlands with fluctuating water depth through the year, resulting in pressure changes.

2.3.3 Environmental Drivers Analysis Results

Kendall Correlation During the Peak GHG Production Season

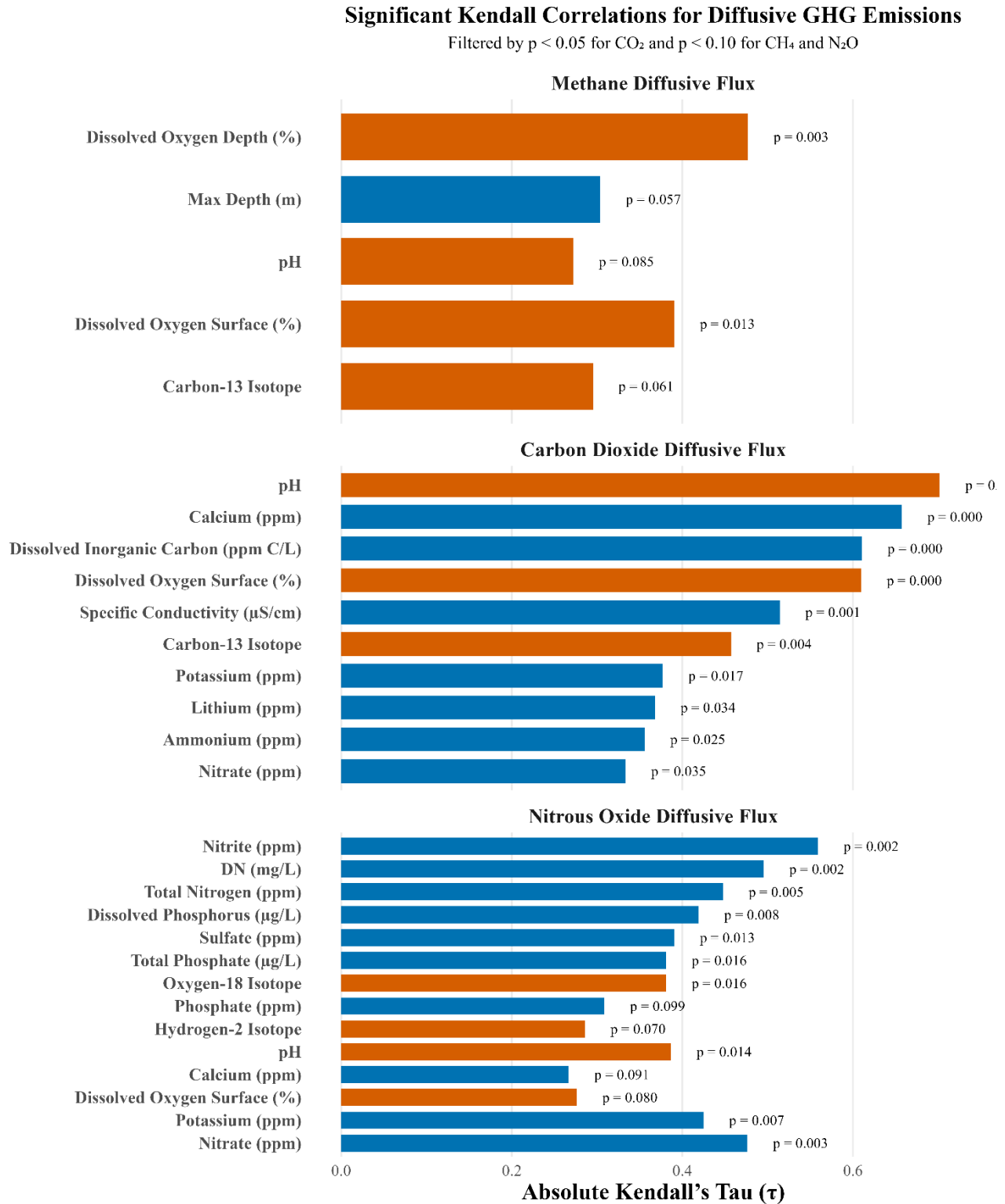


Figure 2.12: Kendall rank correlations between diffusive fluxes of CH₄, CO₂, and N₂O and their associated environmental drivers. Correlations are filtered by significance thresholds ($p < 0.05$ for CO₂; $p < 0.10$ for CH₄ and N₂O) using base R's cor.test function. Data are restricted to Campaigns 1–3 (June–August) to ensure consistent summer temperatures and include only middle locations where YSI data were collected. Measurements taken on duckweed were excluded. Orange bars represent negative correlations, while blue bars represent positive correlations. Unless specified, all variables were measured at the surface of the wetland.

Figure 2.12 visualizes the results of the Kendall correlation for diffusive CH₄, CO₂, and N₂O emissions for the three summer campaigns (June, July, August) to control for temperature (see Appendix 7). The importance of duckweed was not considered until Campaign 2, and thus, this analysis was conducted without duckweed measurements to ensure a consistent flux type across this analysis. Kendall Tau correlation ranking found that the most significant parameters impacting diffusive CH₄ flux in a negative direction were dissolved oxygen saturation at the surface and depth of the wetland (%), pH, and the $\delta^{13}\text{C}$ of particulate organic matter (POM), whereas depth positively correlated with diffusive CH₄ emissions. Like CH₄, CO₂ flux was negatively correlated with pH, dissolved oxygen at the surface, $\delta^{13}\text{C}$ of particulate organic matter (POM). Several more positive correlates were found for diffusive CO₂ than for CH₄: Calcium, DIC, SPC, potassium, lithium, ammonium and nitrate. N₂O flux negatively correlated with pH and dissolved oxygen at the surface (like both CH₄ and CO₂) as well as stable water isotopes. N₂O flux correlated most significantly and positively with different N (TN, NO₂, NO₃, DN) and P (TP, DP, PO₄) species found in the water column. Importantly, the correlation between pH and gas fluxes reflects shared underlying processes rather than direct causation.

Random Forest Modelling of Diffusive GHG Fluxes

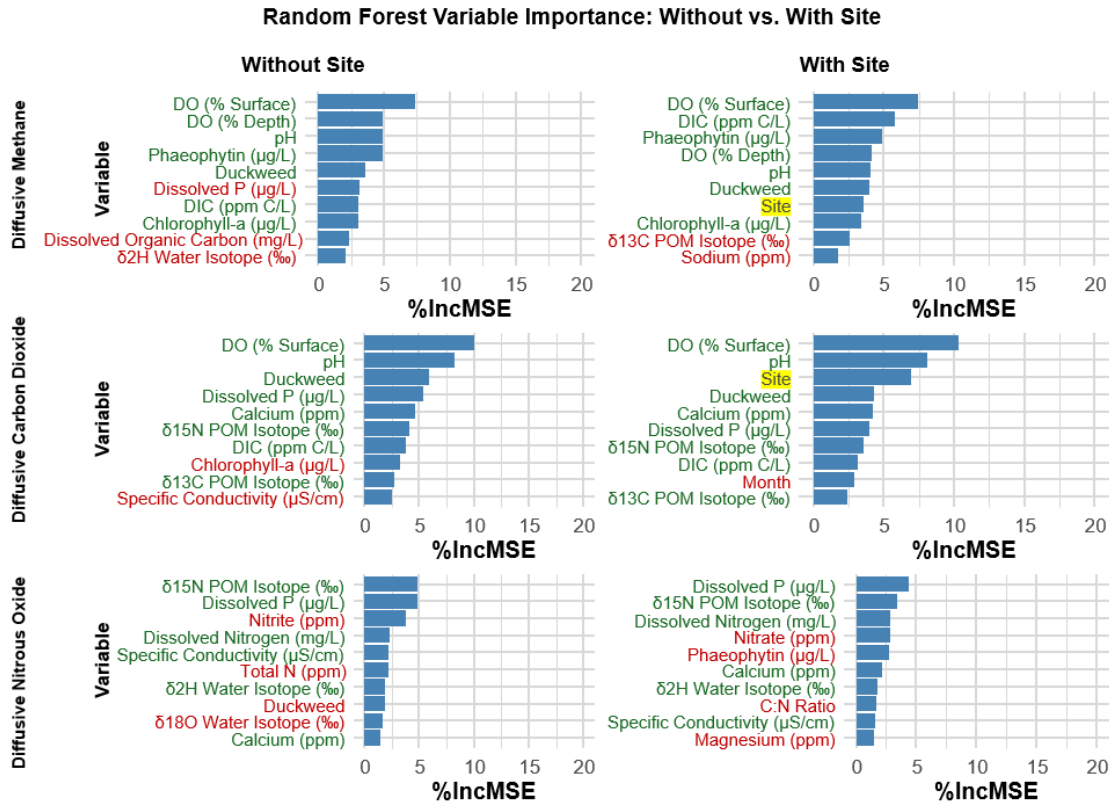


Figure 2.13: Variable importance for Random Forest prediction of diffusive GHG emissions. Importance is measured as the mean decrease in model accuracy (%IncMSE) when each variable is permuted. Categorical predictor ‘Site’ was excluded in one scenario and included in another to test for any site-specific parameters not measured. Variables in red text differ between the two scenarios, while green text indicates variables shared across both models. Site is highlighted when it appears in importance. Measurements collected on duckweed were excluded from analysis, although duckweed was retained as a categorical variable. Winter measurements were limited to those taken on ice during the winter season.

Random Forest results excluding *Site* as a predictive variable were most comparable to the Kendall correlation results, although the latter only included summer data when GHG emissions are presumed to be highest. A complementary Random Forest (RF) analysis was conducted, as this method provides a more robust approach for identifying key environmental drivers, particularly given the non-linear relationships and relatively small dataset.

When *Site* was excluded from the Random Forest (RF) models, both RF and Kendall's tau (KT) analyses identified similar key predictors across gases. For CH₄, dissolved oxygen (surface and depth) and pH were common between methods, with all three ranking among the top predictors in RF (though pH was slightly less important in KT, ranking fifth). KT identified fewer significant variables overall (five total), while the RF analysis highlighted more, including productivity-related parameters such as chlorophyll-*a*, phaeophytin, DIC, duckweed cover, and DP, suggesting broader environmental influences beyond those captured in the summer-only KT analysis. For CO₂, six of the top ten predictors overlapped between KT and RF, including dissolved oxygen, pH, $\delta^{13}\text{C}$ -POM, DIC, calcium, and specific conductance (SPC), although their relative importance varied (e.g., DO ranked first in RF but fourth in KT). KT uniquely identified nitrate, ammonium, and other ions (K, Li), whereas RF emphasized chlorophyll-*a*, $\delta^{15}\text{N}$ -POM, DP, and duckweed cover—again pointing to productivity and vegetation-related drivers that may not be captured in the summer-only KT dataset. For N₂O, seven of the ten RF predictors overlapped with KT, including nitrite, dissolved nitrogen, dissolved phosphorus, total nitrogen, $\delta^2\text{H}$ -H₂O, $\delta^{18}\text{O}$ -H₂O, and calcium. Water isotopes were identified in both analyses, though with lower overall importance. KT highlighted primarily nitrogen species (DO, TN, TP) among its top predictors, whereas RF ranked $\delta^{15}\text{N}$ -POM and DP highest. The absence of nitrate in the RF results may reflect its low concentration outside the summer campaigns used in the KT analysis. Again, the correlation between pH and gas fluxes reflects shared underlying processes rather than direct causation.

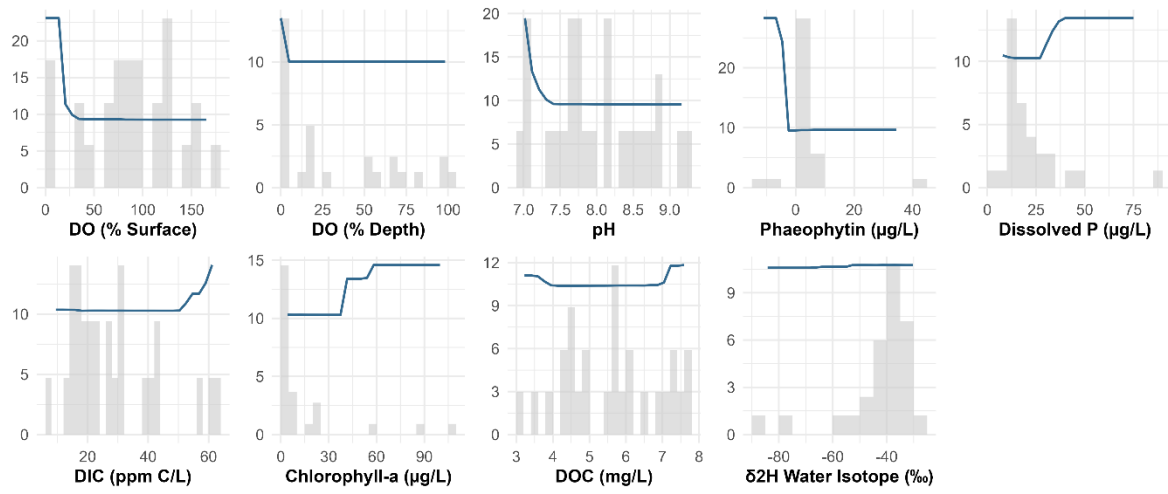
Regarding the Random Forest results, variable importance analysis indicates that *Site* is among the top 10 predictors for CH₄ and CO₂, ranking seventh most important for CH₄ and third

for CO₂ (**Figure 2.13**). Including *Site* changed the top 10 predictors: for CH₄ and CO₂, three and two variables were removed, respectively, while two and one new variables were added.

A

Partial Dependence for Diffusive Methane

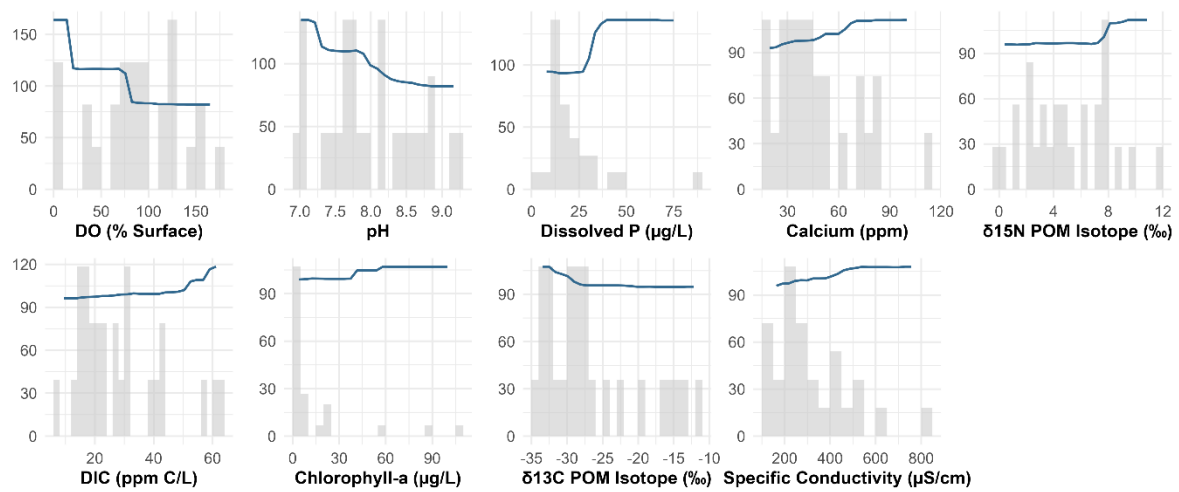
Predicted Diffusive Methane ($\text{mg m}^{-2} \text{d}^{-1}$)



B

Partial Dependence for Diffusive Carbon Dioxide

Predicted Diffusive Carbon Dioxide ($\text{mg m}^{-2} \text{d}^{-1}$)



C

Partial Dependence for Diffusive Nitrous Oxide

Predicted Diffusive Nitrous Oxide ($\text{mmol m}^{-2} \text{d}^{-1}$)

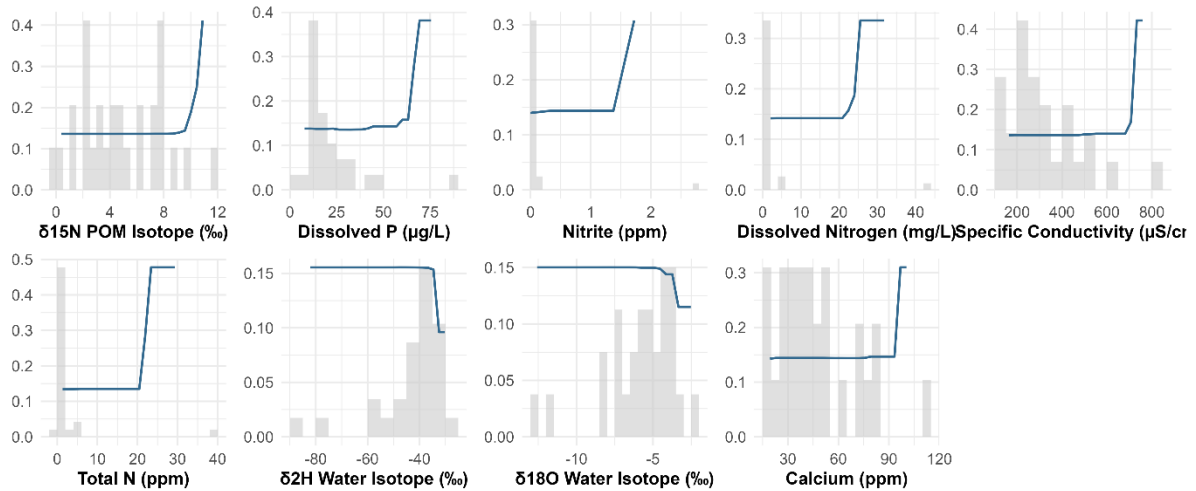


Figure 2.14: Partial-dependence plots for the top 10 predictor variables in the Random Forest models *without* site included as a predictor variable. The y-axis represents the predicted response values averaged across all observations for each predictor value. The histograms at the bottom of each plot display the distribution of the predictor values, highlighting the density of observations that inform the plotted relationships. The quick drop in values (for example, CH₄ for dissolved oxygen) is an artifact of the outlier results from Site MA.

Partial dependence analysis for diffusive CH₄ (**Figure 2.14A**) indicates that emissions increase with higher levels of dissolved inorganic carbon (DIC), chlorophyll-*a*, dissolved phosphorus (DP), and dissolved organic carbon (DOC). Diffusive CH₄ also rises with more positive $\delta^2\text{H}$ water isotope values and duckweed cover (noting that only one out of seven sites had duckweed). Emissions begin to rise sharply when DIC exceeds approximately 25 ppm C/L, chlorophyll-*a* surpasses ~35 $\mu\text{g/L}$, and DP increases beyond 25 $\mu\text{g/L}$, levelling off around 40 $\mu\text{g/L}$ DP. In contrast, CH₄ emissions decrease sharply with higher surface and depth oxygen levels (% DO) and with increasing pH or phaeophytin. The steepest declines occur when surface DO reaches about 25% and pH exceeds 7.5.

Partial dependence analysis for diffusive CO₂ (**Figure 2.14B**) shows that emissions increase with higher calcium, dissolved phosphorus, DIC, specific conductivity, chlorophyll-*a*, and more positive $\delta^{13}\text{C}$ POM isotope values. CO₂ emissions begin to rise noticeably above approximately 25 $\mu\text{g/L}$ DP and 40 ppm C/L DIC. In contrast, CO₂ emissions decrease with increasing surface DO and pH, indicating reduced flux under more oxygenated or alkaline conditions. CO₂ emissions also decrease with increasing $\delta^{13}\text{C}$ POM isotope values.

Partial dependence analysis for diffusive N₂O (**Figure 2.14C**) shows that emissions increase with higher $\delta^{15}\text{N}$ POM isotope, dissolved phosphorus, nitrite, total nitrogen, dissolved nitrogen, specific conductivity, and calcium. In contrast, N₂O emissions decrease with more

positive $\delta^2\text{H}$ and $\delta^{18}\text{O}$ water isotope values, suggesting that isotopically lighter waters are associated with greater N_2O fluxes.

Random Forest models revealed that both Inlet and Outlet measurements explained a substantial portion of the variation in Middle diffusive fluxes, often outperforming models using only Middle measurements (**Table 2.2**).

Table 2.2: Random Forest model performance metrics for predicting diffusive GHG fluxes (CH_4 , CO_2 , N_2O) in the Middle of each wetland. R^2 indicates the proportion of variance explained, and RMSE (Root Mean Squared Error) quantifies prediction error. Metrics are shown for three scenarios: using Middle predictors only (baseline), Inlet predictors, and Outlet predictors. No YSI variables were included; dissolved gas measurements were included. Higher R^2 and lower RMSE indicate better predictive performance.

Gas	R^2 <i>Middle</i>	RMSE <i>Middle</i>	R^2 <i>Inlet</i>	RMSE <i>Inlet</i>	R^2 <i>Outlet</i>	RMSE <i>Outlet</i>
CH_4	0.52	16.26	0.94	7.56	0.98	5.97
CO_2	0.59	89.82	0.95	47.55	0.95	43.70
N_2O	-0.05	1.45	0.98	0.47	0.99	0.47

For CH_4 and CO_2 , R^2 values were higher when using upstream predictors, indicating that water chemistry at the wetland edges strongly influences Middle fluxes. N_2O showed weaker predictability, particularly for the Middle-only baseline. Overall, these results suggest that upstream water quality can serve as a useful proxy for GHG dynamics in the center of restored wetlands, although local factors remain important for certain gases.

2.4 Discussion

2.4.1 Flux Variability in Restored Agricultural Wetlands of Southern Ontario

This study collected data from restored agricultural wetlands in Ontario, Canada. These systems are comparable in size and function to those described in the literature as farm ponds, shallow ponds, prairie pothole wetlands, and both natural and restored wetlands of similar size (Appendix A8). Site MA exhibited the highest measured CH₄ flux at the inlet location—approximately five times greater than the highest value reported in the literature. However, the median CH₄ flux at Site MA was lower (25.4 mmol CH₄ m⁻² day⁻¹), remaining within the range reported for similar systems, including agricultural reservoirs in Saskatchewan, constructed stormwater wetlands in British Columbia, and small artificial ponds influenced by duckweed (Appendix A8). When Site MA is excluded, the average median CH₄ flux across the remaining wetlands is much lower—about 3 mmol CH₄ m⁻² day⁻¹—closely aligning with values reported for beaver pond systems (DelSontro et al., 2016; Weyhenmeyer, 1999) and prairie wetland ponds (Badiou et al., 2011).

Site MA exhibited the highest measured CO₂ flux at the inlet location; however, this value remains within the range reported in the literature for natural and restored wetland ponds as well as constructed stormwater wetlands. In comparison, some systems in the literature validation have shown maximum fluxes of up to 1350 mmol CO₂ m⁻² day⁻¹, exceeding even the highest fluxes recorded at Site MA. Across all sites, CO₂ emissions fell within the range reported for comparable Canadian systems, including agricultural reservoirs, natural wetland ponds, and restored wetlands, as summarized in a modified table from Jensen et al. (2023) (Appendix A8). When measuring CO₂ diffusive fluxes, our floating chambers were initially opaque, which likely led to an overestimation of emissions over duckweed-covered areas, as photosynthetic CO₂

uptake was not accounted for. To address this limitation, sampling could be updated to include both clear and opaque chambers, enabling more accurate flux measurements in vegetated zones. Regardless, our study exhibited the greatest CO₂ sinking potential among comparable systems, with values reaching $-93.8 \text{ mmol CO}_2 \text{ m}^{-2} \text{ day}^{-1}$, whereas natural wetlands have shown maximum sinking potentials of approximately $-60 \text{ mmol CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Bortolotti et al., 2016).

Excluding Site MA, N₂O diffusive fluxes in this study ranged from -1450 to $200 \text{ } \mu\text{mol m}^{-2} \text{ day}^{-1}$ (mean = $-1.4 \text{ } \mu\text{mol m}^{-2} \text{ day}^{-1}$), indicating a net uptake overall. The lowest value likely represents an outlier or invalid measurement, but the mean uptake aligns with similar Canadian systems, closely matching the $-0.9 \text{ } \mu\text{mol m}^{-2} \text{ day}^{-1}$ reported for prairie pothole ponds (Miranda and Whitfield, 2024). Agricultural reservoirs have shown greater variability (-12 to $166 \text{ } \mu\text{mol m}^{-2} \text{ day}^{-1}$; Webb et al., 2019), and our maximum value—excluding MA—slightly exceeds this range. In contrast, Site MA exhibited substantial net emissions (-61 to $7390 \text{ } \mu\text{mol m}^{-2} \text{ day}^{-1}$; median = 5.4), making it one of the stronger N₂O sources compared to other systems, though its mean value remains within reported ranges.

Our findings suggest strong seasonal controls on GHG emissions, consistent with other GHG studies (Guo et al., 2025; Jensen et al., 2022; Logozzo et al., 2025). **Figure 2.15** shows a conceptual schematic depicting our observed seasonal patterns in GHG fluxes.

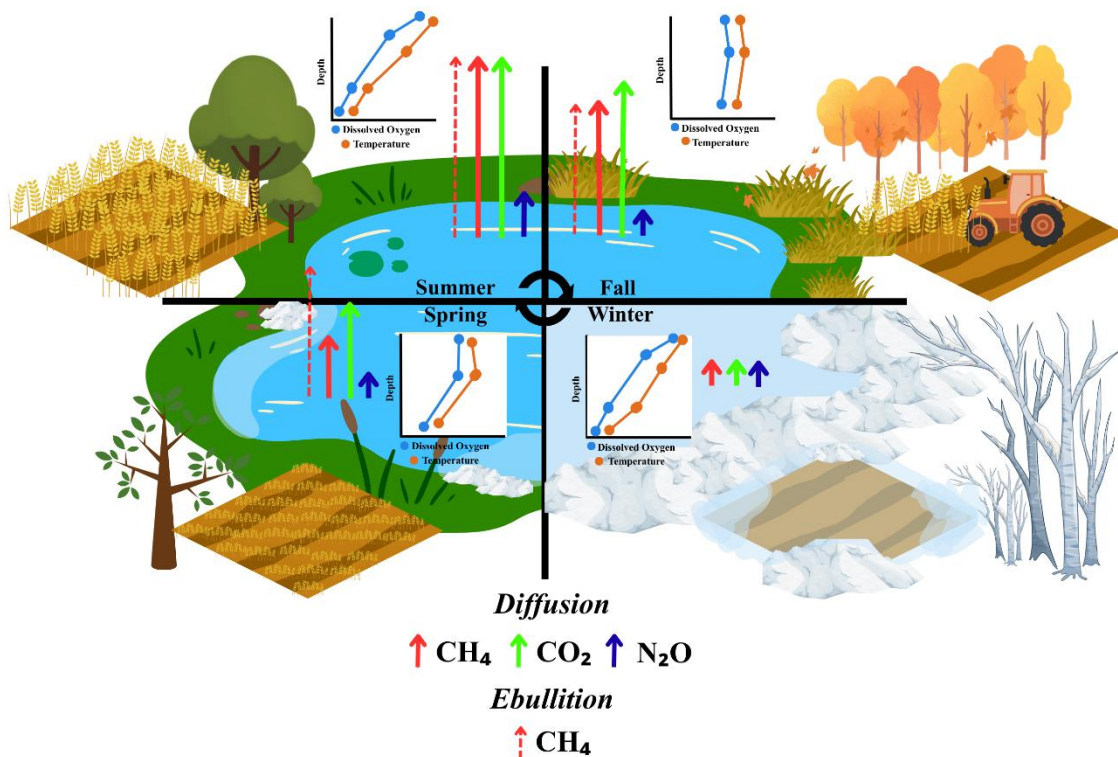


Figure 2.15 Conceptual schematic illustrating seasonal patterns in greenhouse gas emission pathways from an agricultural wetland system. Relative diffusive fluxes of CH₄ (red arrows), CO₂ (green arrows), and N₂O (blue arrows) are shown for each season, while ebullitive emissions are depicted as dashed red arrows (CH₄ only). Arrow lengths indicate relative magnitudes and are comparable only within the same gas and emission pathway. Insets show representative dissolved oxygen (blue) and temperature (orange) depth profiles, highlighting summer stratification and spring and fall mixing events. Seasonal context reflects the surrounding agricultural land use and the progression of the growing season.

Overall, our results show that summer fluxes exhibit greater variability than other seasons (Figure 2.15), and there is also clear spatial variability, with Site MA standing out as a pronounced outlier, emitting substantially more GHGs than the other wetlands. Understanding the environmental drivers behind this summer variability is particularly important, as these months account for the peak periods of GHG production. Importantly, there is a notable scarcity of studies focusing on analogous wetland systems in Ontario, despite a thorough review of the

literature. Moreover, winter flux data for these systems in Ontario remain severely underrepresented, highlighting a critical knowledge gap.

2.4.2 The Force Behind the Flux: Environmental Drivers

During the summer, when GHG emissions were at their peak (**Figure 2.15**), several key environmental drivers were identified across the wetlands. Redox conditions primarily governed CH₄ emissions. Low dissolved oxygen levels in both surface and deeper waters promoted methanogenesis, as oxygen inhibits CH₄ production under aerobic conditions (Thauer et al., 2008). Emissions tended to increase in deeper or more stratified areas where anoxic zones develop (Moore et al., 2011), although there is literature that suggests that peak emissions will peak with shallower maximum depths (~0.5 m) (Calabrese et al., 2021). Higher pH appeared to suppress CH₄ production, consistent with the inhibitory effects of alkaline conditions (Qiu et al., 2023), most likely reflecting system changes driven by factors like dissolved CO₂. Stronger CH₄ fluxes were associated with fresher particulate organic matter, suggesting that recently produced or less decomposed material fuels microbial activity.

CO₂ fluxes were largely influenced by oxygen availability, acidity, and inorganic carbon dynamics (Boyd, 2015). Elevated CO₂ emissions occurred under low-oxygen and more acidic conditions, reflecting enhanced microbial respiration and the accumulation of dissolved CO₂. High CO₂ in the water column lowers pH, releasing calcium and potassium from minerals, which elevates conductivity and fuels CO₂ production. Nutrient inputs—particularly nitrogen and potassium—further supported microbial metabolism, likely amplified by agricultural runoff from surrounding farmland (Bonetti et al., 2022). As with CH₄, higher CO₂ emissions coincided with fresher organic material, highlighting the importance of recent carbon inputs to respiratory processes.

For N_2O , nutrient enrichment emerged as the dominant driver. Elevated nitrogen and phosphorus concentrations were strongly linked to greater N_2O fluxes, pointing to enhanced nitrification and denitrification fueled by fertilizer-derived inputs (Feng et al., 2025; Hénault et al., 2019). Acidic and low-oxygen conditions favoured incomplete denitrification, increasing N_2O accumulation (Čuhel et al., 2010). The influence of calcium and sulfate suggests a connection to mineral-mediated buffering and microbial redox reactions (Z. Li et al., 2020; Linquist et al., 2012; H. Wang et al., 2022). Hydrological factors also played a role—periods following rewetting likely triggered short-term pulses of N_2O , reflecting moisture-driven shifts in microbial activity (Pinto et al., 2021) as reflected in the stable water isotope relationships.

Random Forest analysis suggests that site-specific factors play an important role in controlling CH_4 and CO_2 fluxes, with *Site* emerging as a top predictor for both gases. This indicates that there are likely influential environmental variables not captured in our measurements. Potential missing factors include vegetation characteristics, local agricultural practices, and wetland geomorphology, all of which could affect biogeochemical processes and gas exchange. CO_2 appears particularly sensitive to these unmeasured site-specific influences, suggesting that plant-mediated processes may be especially important for controlling its fluxes.

Significant overlap was identified in environmental drivers exhibited by the Kendall Tau and Random Forest analysis (not including site): Both KT correlations and RF partial dependence analyses identified oxygen as a strong, consistent negative control on diffusive CH_4 , CO_2 , and N_2O emissions, as well as pH, which most likely reflect underlying system changes from dissolved CO_2 and mineral dynamics. However, the RF model—accounting for annual-scale variability—revealed several additional nonlinear and threshold relationships not captured in the summer-only KT results. For CH_4 , both KT and RF analyses indicated strong negative

effects of DO and pH. The RF model additionally identified positive, nonlinear responses to DIC, DP, DOC, $\delta^2\text{H}$ of water, and chlorophyll-*a*. In contrast, the KT analysis showed a positive relationship with maximum wetland depth and a negative relationship with $\delta^{13}\text{C}$ of POM, which differed from the RF results.

For CO_2 , both methods showed that emissions decreased with surface DO and pH, but neither was correlated with DO at depth. Emissions increased with calcium, DIC, and SPC in both analyses. The KT analysis additionally indicated positive relationships with potassium, lithium, ammonium, and nitrate during the summer, which were not reflected in the annual RF model. Conversely, the RF identified positive relationships with DP, chlorophyll-*a*, and $\delta^{15}\text{N}$ of POM that were absent in the KT. $\delta^{13}\text{C}$ of POM exhibited a consistent negative relationship across both methods.

For N_2O , the KT analysis showed negative relationships with surface DO and pH, patterns not observed in the RF model. Both methods indicated negative relationships with the stable water isotopes and positive associations with nitrogen species (DP, nitrite, DN, and TN) as well as calcium. The RF additionally showed positive relationships with $\delta^{15}\text{N}$ of POM and SPC, while the KT identified further positive associations with nitrate, potassium, sulfate, TP, and phosphate. The additional predictors not evident in the KT analysis suggest seasonal or productivity-related drivers that were not fully represented in the temperature-controlled summer campaigns.

Overall, the RF results highlight the dominant role of nutrient enrichment in driving annual variability of GHG emissions from wetlands. For CH_4 , key environmental drivers indicate strong links with eutrophication, where elevated nutrients promote plant growth (e.g., high chlorophyll-*a*, duckweed cover) and create anoxic, hypereutrophic conditions favorable for

methanogenesis. Chlorophyll-*a* concentrations exceeding 30 µg/L are indicative of hypereutrophic systems (Brehob et al., 2024), while total phosphorus >100 µg/L has similarly been used to define hypereutrophic thresholds (Wetzel, Robert G., 1975). Seasonal transitions into such conditions likely explain observed CH₄ emission peaks. For CO₂, emissions increased sharply around 7‰ δ¹⁵N, a range consistent with nitrogen derived from fertilizers (0–10‰) and organic soil sources (–7 to 8‰; Townsend et al., 2003), suggesting agricultural runoff as a key contributor to elevated CO₂ and CH₄ fluxes. For N₂O, nutrient forms such as nitrate and nitrite emerged as primary drivers, directly fueling denitrification and nitrification processes. The strong positive effect of specific conductivity (SPC) for CO₂ further points to increased ionic strength and nutrient loading, while hydrogen and oxygen water isotopes among the top predictors for N₂O and CH₄ indicate that hydrological variability—through precipitation and evaporation—modulates production dynamics.

2.4.3 Modelling Considerations

Spatial Considerations for GHG Modelling

Most wetlands exhibited relatively consistent GHG fluxes within close orders of magnitude of one another; however, Site MA markedly increased the overall variability and order-of-magnitude range for all three gases, underscoring its influence as an outlier. This suggests a consistent baseline across most wetlands for modelling diffusive CH₄, with exceptions at outlier sites like MA. Outlier sites like MA, with exceptionally high emissions and unique vegetation (e.g., abundant duckweed), may occur across Southern Ontario. Such sites can skew regional models if not properly accounted for, as their emissions may be influenced by legacy nutrients or distinct site conditions. Within locations in a wetland, the results indicate that although emissions vary across different wetlands and their locations collectively, emissions

within each wetland are fairly uniform, implying that spatial heterogeneity within wetlands plays a minor role when scaling emissions.

Although statistical tests did not reveal significant differences in GHG fluxes across locations within individual wetlands, patterns suggest that certain locations consistently exhibit higher emissions—potentially acting as localized hotspots. This subtle spatial heterogeneity, even if not statistically significant, can complicate efforts to upscale point measurements to whole-wetland emissions. If location-specific differences are present, even modestly, models may require site-specific calibration or alternative spatial approaches to accurately reflect these dynamics. Consequently, relying on limited point measurements may lead to under- or overestimation of total wetland emissions. Such uncertainty should be explicitly considered in GHG inventories and management decisions to ensure robust and informed policy and mitigation planning. This study recognizes that back-calculated values via the gas-exchange formula showed considerable uncertainty when compared to field chamber measurements. These values often represented the extremes of our observed ranges—either the highest or lowest points—but occasionally fell near the mid-range, reflecting variability in the estimates.

The partial dependence plots from the RF model were based on data from the middle of each wetland. Therefore, to ensure reliability, water sampling should ideally occur at the wetland center—a process that poses logistical challenges without equipment such as a raft. This raises the question of whether edge sampling could provide comparable insights and simplify fieldwork. Random Forest results indicate that Inlet and Outlet predictors can explain Middle fluxes with 94% and 98% accuracy, respectively (excluding YSI variables). Dissolved gas concentrations also correlated positively and linearly with diffusive fluxes (Spearman's $r^2 = 0.61$ for CH₄, 0.61 for CO₂, and 0.94 for N₂O). However, this relationship is largely driven by Site

MA; when MA is excluded, correlations drop substantially (CH_4 $r^2 \approx 0.14$, CO_2 $r^2 \approx 0.09$, N_2O $r^2 \approx 0.005$). Despite this, across larger areas with greater measurement gradients and inherent outliers, these relationships can still provide meaningful insights into GHG flux patterns. Consequently, collecting dissolved gas samples only from wetland edges may not reliably approximate central fluxes; including a few additional environmental drivers can improve estimates, as indicated by the Random Forest model. This variability highlights the challenge of accurately representing emissions from heterogeneous wetland environments when relying solely on perimeter or central sampling locations.

Influence of Duckweed on GHG Emissions

MA was the dominant GHG emitter among the study wetlands, exhibiting significantly higher mean diffusive fluxes of CH_4 , CO_2 , and N_2O across most campaigns. It was the only site where duckweed persisted through multiple stages of the annual cycle—a factor linked to roughly threefold higher carbon output in small aquatic systems (J. Rabaey & Cotner, 2022).

MA was also the most anoxic wetland, with near-zero surface DO and the highest dissolved CH_4 concentrations. Unlike other sites where CH_4 peaked at depth, MA showed elevated concentrations at the surface as well as at depth (**Figure 2.9**), likely due to duckweed limiting oxygen diffusion at the air-water interface. Duckweed restricts light penetration, thereby suppressing photosynthesis of submerged vegetation or algae, and simultaneously releases the oxygen it produces to the atmosphere rather than the water column, which intensifies anoxia and inhibits significant CH_4 oxidation (Hillman, 1961; J. Rabaey & Cotner, 2022). The forest surrounding site MA likely further limits wind mixing and helps sustain the low-oxygen conditions brought on by the presence of duckweed (see **Table 2.3** for more site-specific characteristics of site MA).

Table 2.3: Comparison of site-specific characteristics highlighting the unique features of MA relative to the other study wetlands (Page et al., 2023).

Category	MA Site Characteristics
Restoration Age	10 years post-restoration; within study range of 7 – 11 years (shared with LL and MO)
Basin Area	0.18 ha (within study range of 0.14–0.78 ha); relatively small basin area
Basin Area Group	Small: MA, MO, DY, BL
Basin Volume	927 m ³ (within study range of 827–5,365 m ³); one of only two wetlands < 1,000 m ³
Basin Volume Group	Low volume: MA, MO, DY, BL High volume: OH, LL, FE
Inflow Channel	Present (defined inflow channel)
Outflow Structure	Present (culvert); similar to 6 of 8 study wetlands
Sediment Phosphorus	Highest sediment P concentration among all study wetlands

Nutrient enrichment appears central to MA’s elevated GHG emissions. It consistently exhibited the highest TP (213 µg/L) and DP (102 µg/L) concentrations, comparable to values in nutrient-rich duckweed ponds (Rabaey & Cotner, 2022). Elevated TN (7.9 mg/L) and NH₄⁺ (0.14 mg/L), coupled with high DIC, provide ample substrate for GHG production. Sediment data confirms the highest P concentrations among all wetlands (Page et al., 2023), suggesting legacy nutrient loading that continues to fuel eutrophication and duckweed growth. Phosphate was the limiting nutrient for duckweed growth under natural, pre-anthropogenic conditions, leading to their adaptation for efficient assimilation and storage of this ion (Ziegler et al., 2023). As duckweed decomposes, it releases the nutrients it previously assimilated back into the system. Kruskal–Wallis and post-hoc Dunn’s tests confirmed that MA had significantly higher dissolved nutrient concentrations than most other wetlands.

The range of $\delta^{15}\text{N}$ values observed (-1.2 to 11.76‰) suggests variable nitrogen sources across sampling sites such as soil organic nitrogen, and fertilizer (Townsend et al., 2003). Elevated $\delta^{15}\text{N}$ values near 11.76‰ at the MA middle site likely reflect one occurrence of a larger influence of volatilization, animal waste, and/or denitrification.

Site emerged as a key predictor in the Random Forest model, suggesting that unmeasured, site-specific factors influenced GHG fluxes. These findings highlight duckweed as a potential environmental control that may be reflected in the model's results. Spatial context and measurement location are therefore critical when estimating GHG fluxes, as high spatial variability in CH_4 emissions is often linked to vegetation type in wetlands (Stewart et al., 2024) and lakes (Desrosiers et al., 2022). Incorporating vegetation characteristics could have improved the model's explanatory power.

Seasonal Considerations for GHG Modelling

GHG emissions have been shown to be strongly influenced by seasonality in temperate zones (Guo et al., 2025; McNicol et al., 2017) and the seasonal temporal variability evident in this study suggests the same. Summer GHG emissions, however, show greater variability (based on SD) during the peak productive period compared to the cooler months (October–April). This increased summer variability introduces greater uncertainty in emission estimates, highlighting the need for representative data during this period to maximize model accuracy as summer contributes the majority of annual emissions.

Interestingly, water temperature did not emerge as an important variable in the RF models, despite seasonality—strongly influenced by temperature—being hypothesized as a key control on GHG emissions. In scenarios that included YSI parameters (surface water temperature, dissolved oxygen, pH, SPC), temperature variables did not rank highly in

importance. It is possible that the variability in nutrients and other environmental factors such as dissolved oxygen, which appear to play a more dominant role in these restored agricultural wetlands, indirectly reflect seasonal temperature changes.

Figure 2.13 shows that Month emerged among the top predictors for CO₂ at the middle locations when site was included. Month may capture seasonal effects on temperature, plant productivity, or other biotic changes throughout the year that influence CO₂ flux. Differences in macrophyte types across wetlands, along with their seasonal growth and decomposition, likely contribute to CO₂ dynamics more than other GHG.

Winter Impacts on GHG Fluxes

Continuing, incorporating winter fluxes is important when modelling wetland GHG emissions, as fluxes during cold seasons may not be negligible, but it is rarely accounted for in studies. Southern Ontario experiences cold winters with an approximate daily average of -5.4 °C to -4.8 °C in January and February (Environment and Climate Change Canada, 2025); thus, water bodies tend to freeze over for at least a portion of the winter. In January and February of 2025, we had to drill through 10 to 20 cm of ice in each pond to access water for sampling, but the true GHG fluxes during that time would have been through ice/snow. Interestingly, we did measure a flux from some of these surfaces, albeit very small (CH₄: -0.003 to 1.1, CO₂: -12.7 to 38.4, N₂O: -0.01 to 0.001 mmol/m²/day). During the thawing event, however, we would expect to see emissions similar to those measured in our ice holes, which were significantly higher than over snow/ice emissions as the water beneath the ice was still highly saturated in all 3 GHGs (CH₄: 0.02 to 51.07, CO₂: 13.1 to 171, and N₂O: -0.005 to 0.04 mmol/m²/day).

Winter CH₄ fluxes observed in our study are broadly comparable to previous reports from cold-season wetlands. Beaver ponds in Ontario, the most comparable system, emitted CH₄ at ~0.76 mmol m⁻² day⁻¹ over a three-month winter under ice cover (Weyhenmeyer, 1999). Other cold-season wetland studies, conducted either under snowcover or in unfrozen water during cold conditions, including agricultural dams, peatlands, and marshes, reported winter CH₄ fluxes ranging from ~0.008 to 0.76 mmol m⁻² day⁻¹, CO₂ fluxes of 3.9–16 mmol m⁻² day⁻¹, and N₂O fluxes from small sinks to ~0.74 mmol m⁻² day⁻¹ (Miao et al., 2012; Ollivier et al., 2019; Weyhenmeyer, 1999). These reported ranges are consistent with the winter CH₄, CO₂, and N₂O fluxes observed in our study under ice and snow conditions.

As air temperatures start to cool in autumn into winter, cooling surface water typically promotes mixing, bringing oxygen to deeper layers—opposite to the thermal stratification usually observed in summer (Ford et al., 2002). An increase of oxygen in bottom waters will enhance methanotrophy and inhibit methanogenesis in former anoxic zones (Bastviken et al., 2004). Interestingly, we observed oxygen levels beneath winter ice comparable to those from other campaigns, with the Jan/Feb surface O₂ saturation as low as that during July. This indicates that either anoxia persisted following the freezing of the surface or re-established itself following mixing and freezing (Woolway et al., 2022).

This seasonal mixing, followed by ice formation, can alter substrate availability, and when comparing winter (Campaign 5 – January/February) to the average of all other campaigns, several notable differences emerge. Dissolved phosphorus (DP), soluble reactive phosphorus (SRP), and particularly phosphate were elevated in winter, with phosphate concentrations approximately 560% higher (0.086 mg/L vs. 0.013 mg/L). These elevated levels may reflect reduced biological uptake under ice cover. However, total phosphorus (TP) concentrations were

nearly identical between seasons (112 $\mu\text{g/L}$ in winter vs. 111 $\mu\text{g/L}$), suggesting that while bioavailable forms increased (SRP), overall P stocks remained stable. Total nitrogen (TN), nitrate, and nitrite levels showed little seasonal variation.

Rabaey and Cotner (2024) showed in this study of small ponds that nutrient concentrations (TP, TDN, DOC) peaked in winter beneath the ice. Some ponds also showed elevated conductivity under ice during winter. Similarly, shallow reservoirs exhibited declining oxygen early in winter, followed by increases in specific conductance (SPC), ammonium, and SRP (Cavaliere & Baulch, 2021). Our results showed higher SPC at depth in winter, indicating increased ionic concentration likely caused by ice trapping salts below and limiting water mixing (DeGrandpre et al., 2021). Additionally, runoff from nearby roads may contribute salts during winter, as we observed elevated calcium, DIC, potassium, sulfate, chloride, sodium, lithium, and bromide, with only minor decreases in fluoride and ammonium.

This aligns with increased GHG build-up beneath the ice cover. Despite frozen surface conditions, microbial activity persists beneath the ice, producing GHGs at appreciable rates. Methanogenesis has been found to occur between 2°C and 37°C in wetlands, with the most activity between 2°C and 10°C (Westermann, 1993). During the winter, water temperature at depth ranged from 3.95°C to 5.95°C. Other studies have found similar build-up of CH₄ and CO₂ beneath the ice during winter (Phelps et al., 1998; J. S. Rabaey & Cotner, 2024).

CH₄ may be able to diffuse through ice slowly (Noguchi et al., 2019), but larger emissions can occur through cracks or openings in the ice during winter. CH₄ bubbles can become trapped and frozen within the ice as they accumulate beneath it. Frozen bubbles can have very high levels of CH₄ concentrations up to 95% (Wik et al., 2011). Measuring precise CH₄ concentrations in frozen bubbles is an emerging area of research, with methodologies still under

development (Greene et al., 2014; Schüpbach et al., 2009). In a warming climate, earlier and more rapid ice melt may enhance CH₄ emissions, as the shorter duration of ice cover may reduce the time available for CH₄ oxidation before release (Greene et al., 2014).

2.4.4 Recommendations and Relevance of Findings

This study provides a framework for assessing GHG fluxes in restored agricultural wetlands in Southern Ontario. While direct field measurements and Random Forest modeling are time- and labor-intensive, our results identify the most influential environmental drivers, offering opportunities to streamline future data collection. By focusing on key variables—such as dissolved oxygen, duckweed presence, and nutrient concentrations—researchers can obtain reliable estimates of wetland GHG fluxes with reduced effort, improving efficiency for monitoring programs and informing wetland management strategies. However, the *Site* predictor may reflect the importance of the agricultural landscape or an unmeasured variable, warranting further investigation to determine whether additional factors should be incorporated into the sampling plan. However, for model upscaling, sample collection should ideally be simplified, reducing the need for intensive parameter measurements. This simplification would increase accessibility, expand the number of sites that can be sampled, and facilitate broader participation, including the potential involvement of citizen scientists. Future field sampling should prioritize variables identified as key explanatory factors through variable importance analyses, thereby facilitating the development of more accessible models with streamlined data collection protocols.

The Kruskal–Wallis analysis showed no statistically significant differences in GHG emissions among sampling locations within each wetland. Similarly, Random Forest model results indicated that measurements taken from either the inlet or outlet can adequately represent

average emissions across the wetland. These findings suggest that sampling strategies can be simplified without compromising data quality.

Moreover, our study highlights that duckweed strongly influences GHG emissions, with minimal research existing in this space, further work is needed to fully understand the role of duckweed in wetland biogeochemistry. Additionally, most of our young (<10 years of age at time of sampling), restored wetlands are emitting relatively low amounts of methane compared to undisturbed temperate wetlands (Turetsky et al., 2014), which is unusual and particularly promising given Southern Ontario's extreme climate and wetland age. These results suggest that restoration-focused nature-based solutions (NBS) wetlands can possibly be supported under challenging climatic conditions without producing excessive GHGs.

2.4.5 Management and Restoration Considerations

As wetland restoration efforts expand—particularly on Canadian farmland where there has been large historical wetland loss—it is important to recognize that restoring wetlands in areas with high nutrient pollution or legacy nutrient accumulation can strongly influence GHG emissions. Restoration efforts could aim to target areas with less nutrient-rich soils to help prevent the development or exacerbation of eutrophic conditions in newly restored wetlands.

Vegetation (e.g., duckweed) could have complex effects on GHG emissions. Plans to limit duckweed growth such as removing it periodically or planting low emitting plants could be a part of the restoration plan. Duckweed has been shown to uptake large amounts of 99.1% of TN and 90.8% of TP from water in polyculture (G. Chen et al., 2018). Removing duckweed from the surface of waterbodies helped reduce nutrient concentrations, since decomposing duckweed would otherwise release the nutrients, it had absorbed back into the water.

Wetland bathymetry could be further explored to determine the most effective configurations for minimizing GHG emissions. In this study, maximum water depth was found to influence summer CH₄ emissions, suggesting that wetland shape and depth distribution may play a key role in regulating gas dynamics. Oxygen bubblers could be incorporated into restored wetland designs to help maintain oxygen-saturated conditions, addressing one of the key drivers of GHG emissions identified in this study.

Chapter 3: Conclusion

This comprehensive field study provided a detailed quantification of greenhouse gas (GHG) dynamics in restored agricultural wetlands across the Ontario portion of the Lake Erie basin throughout all four seasons. Methane (CH₄), carbon dioxide (CO₂), and nitrous oxide (N₂O) fluxes were measured across seven restored wetlands, and the key environmental drivers influencing these emissions were identified, providing insight into opportunities to improve emission modeling for restored wetland systems in the region. Seasonal and spatial measurements captured both diffusive and ebullitive pathways, providing a thorough understanding of GHG dynamics in wetlands within agricultural landscapes. These findings refine regional GHG inventories for restored agricultural wetlands and inform targeted modeling, management, and monitoring strategies.

3.1 Summary of Key Findings

When assessing GHG emissions in restored agricultural wetlands, both spatial and temporal variability are important considerations for model upscaling.

Spatial variability: Within individual wetlands, Kruskal–Wallis tests showed no statistically significant differences across locations, though small local variations exist. This suggests that sampling from a single representative location—such as the inlet, outlet, or middle—may be sufficient for estimating overall wetland emissions, particularly in saturated zones. Across sites, most wetlands exhibited consistently low methane emissions compared to undisturbed temperate wetlands (Turetsky et al., 2014), with the exception of Site MA, which acted as an outlier. Notably, these are young wetlands (7–11 years post-restoration), and low emissions at this early stage are unusual compared to other restored systems, which often display higher fluxes before equilibrating to natural wetland similar emissions. This finding has important implications for

restoration policies in Southern Ontario, indicating that early-stage wetlands can possibly achieve low GHG outputs even under extreme climatic conditions.

Seasonal variability: GHG emissions peaked during summer months, reflecting higher biological activity and productivity. Winter emissions were lower but still detectable, indicating that cold-season processes, including sub-ice microbial activity and nutrient dynamics, contribute meaningfully to annual fluxes.

Environmental drivers of GHG flux: Dissolved oxygen was identified as a strong driver of CH₄ and CO₂ fluxes and thus has the potential for mitigating wetland carbon emissions.

Duckweed emerged as a unique environmental driver, strongly influencing GHG emissions by promoting anoxic conditions and mediating nutrient cycling. Nutrient enrichment, particularly nitrogen and phosphorus, consistently increased emissions, suggesting that wetland restoration on sites with low legacy nutrient loading may help minimize GHG output.

Considerations for monitoring and modeling: This study highlights that careful consideration of sampling location, season, and key environmental drivers can allow for simplified and effective monitoring strategies.

3.2 Implications of Research

Canada has committed to achieving net-zero GHG emissions by 2050 (Canada, 2020), in line with international agreements such as the Paris Agreement and the Global CH₄ Pledge, which aims to reduce CH₄ emissions by 30% below 2020 levels by 2030 (Environment and Climate Change Canada, 2021b; Government of Canada, 2016; United Nations, 2018). To meet these ambitious targets, national GHG inventories must accurately account for all significant sources and sinks, including smaller and often-overlooked systems such as agricultural ponds.

Our study addresses a critical blind spot in current GHG accounting by quantifying GHG emissions from farm ponds in Southern Ontario—systems that are common across agricultural landscapes but remain underrepresented in national and global CH₄ assessments (Saunois et al., 2025). We designed our study with recommendations from literature, including the need for spatially distributed measurements (Lauerwald et al., 2023), increased research attention to farm ponds and small water bodies (Saunois et al., 2025), and winter conditions (under ice measurements, fluxes in the winter season) (Denfeld et al., 2018; Weyhenmeyer, 1999). This study also measured all three major GHG in tandem, a limitation identified by Logozzo et al. (2025). This research provided critical seasonal field observations from restored agricultural wetlands across Southern Ontario, which is especially important in a region with extensive opportunities for wetland restoration, given the substantial historical wetland loss.

This research contributes to the broader SOLUTIONSCAPES initiative — *Designing climate- and water-smart agricultural solutions in complex working landscapes* — by providing critical field-based GHG data from underrepresented systems. By capturing real-world emissions data, this work helps to reduce uncertainty in the global CH₄ budget and directly inform the development of improved predictive models for agricultural GHG emissions within SOLUTIONSCAPES.

Continuing, by evaluating the most common environmental drivers of restored agricultural wetlands in Ontario, this research provides insights that can inform restoration frameworks and management practices aimed at mitigating GHG emissions across Canada. Ultimately, this work advances our understanding of GHG dynamics in restored wetlands within agricultural landscapes, strengthening GHG budgeting and modelling efforts while also

informing restoration practices, guiding policy development, and inspiring future approaches to wetland management.

3.3 Limitations

Several limitations should be considered when interpreting our results. First, more duckweed-dominated sites would be needed to fully assess the influence of vegetation cover on GHG fluxes. Second, while clear chambers were incorporated in later campaigns to account for photosynthetic uptake, initial measurements relied solely on opaque chambers, which may have biased CO₂ flux estimates. Third, hydrological factors were not measured in this study, yet groundwater inputs and other subsurface processes could influence GHG dynamics. For example, dissolved gas profiles at Site MO suggest potential groundwater influence (see dissolved gas profiles), highlighting the need for further investigation. Ongoing analysis of gas isotopes will provide additional insights into the production and consumption of GHGs in restored wetlands. Finally, there is potential for bias in flux calculations, and implementing automated or scripted processing could help minimize errors in future analyses.

3.4 Future Research

Future work should focus on addressing several key knowledge gaps identified in this study. The literature review highlighted limited research on certain environmental drivers of GHG emissions, particularly stable water isotopes and specific ionic components. Continued data collection from the existing study sites, as well as new sampling within the wetland inundation zone, will help clarify how saturation influences GHG fluxes—an important driver for emissions in restored wetlands (specifically N₂O emissions).

Future research should also further explore winter processes, such as the formation and release of CH₄ bubbles trapped in ice, which may contribute significantly to cold-season emissions. Analyzing the concentration of GHG and isotope compositions from funnel and bubble tap gas samples will be completed. This analysis will provide valuable insights into the production and oxidation pathways of these gases. Finally, continued data collection should support the development and calibration of a predictive model for estimating GHG emissions from restored wetlands. This model could be refined using field measurements to improve understanding of emission dynamics and guide wetland restoration and management practices.

References

- Åhlén, D., Peacock, M., Brodin, Y., & Hambäck, P. A. (2023). Wetland productivity determines trade-off between biodiversity support and greenhouse gas production. *Ecology and Evolution*, 13(10), e10619. <https://doi.org/10.1002/ece3.10619>
- Ampe, H. L., Crumpton, W. G., Stenback, G. A., & Green, D. I. S. (2025). Nitrous oxide and methane production and emission in wetlands receiving nonpoint source nitrate loads. *Journal of Environmental Quality*, n/a(n/a). <https://doi.org/10.1002/jeq2.70081>
- Aron, P. G., Poulsen, C. J., Fiorella, R. P., Levin, N. E., Acosta, R. P., Yanites, B. J., & Cassel, E. J. (2021). Variability and Controls on $\delta^{18}\text{O}$, d-excess, and $\Delta^{17}\text{O}$ in Southern Peruvian Precipitation. *Journal of Geophysical Research: Atmospheres*, 126(23), e2020JD034009. <https://doi.org/10.1029/2020JD034009>
- Audet, J., Carstensen, M. V., Hoffmann, C. C., Lavaux, L., Thiemer, K., & Davidson, T. A. (2020). Greenhouse gas emissions from urban ponds in Denmark. *Inland Waters*, 10(3), 373–385. <https://doi.org/10.1080/20442041.2020.1730680>
- Badiou, P., McDougal, R., Pennock, D., & Clark, B. (2011). Greenhouse gas emissions and carbon sequestration potential in restored wetlands of the Canadian prairie pothole region. *Wetlands Ecology and Management*, 19(3), 237–256. <https://doi.org/10.1007/s11273-011-9214-6>
- Badiou, P., Page, B., & Ross, L. (2019). A comparison of water quality and greenhouse gas emissions in constructed wetlands and conventional retention basins with and without submerged macrophyte management for storm water regulation. *Ecological Engineering*, 127, 292–301. <https://doi.org/10.1016/j.ecoleng.2018.11.028>
- Bahram, M., Espenberg, M., Pärn, J., Lehtovirta-Morley, L., Anslan, S., Kasak, K., Kõljalg, U., Liira, J., Maddison, M., Moora, M., Niinemets, Ü., Öpik, M., Pärtel, M., Soosaar, K.,

- Zobel, M., Hildebrand, F., Tedersoo, L., & Mander, Ü. (2022). Structure and function of the soil microbiome underlying N₂O emissions from global wetlands. *Nature Communications*, 13(1), 1430. <https://doi.org/10.1038/s41467-022-29161-3>
- Bansal, S., Creed, I. F., & Tangen, B. A. (2023). Practical Guide to Measuring Wetland Carbon Pools and Fluxes. *Wetlands*, 43, 105. <https://doi.org/10.1007/s13157-023-01722-2>
- Bansal, S., Post van der Burg, M., Fern, R. R., Jones, J. W., Lo, R., McKenna, O. P., Tangen, B. A., Zhang, Z., & Gleason, R. A. (2023). Large increases in methane emissions expected from North America's largest wetland complex. *Science Advances*, 9(9), eade1112. <https://doi.org/10.1126/sciadv.ade1112>
- Bao, T., Jia, G., & Xu, X. (2023). Weakening greenhouse gas sink of pristine wetlands under warming. *Nature Climate Change*, 13(5), 462–469. <https://doi.org/10.1038/s41558-023-01637-0>
- Baron, A. A. P., Dyck, L. T., Amjad, H., Bragg, J., Kroft, E., Newson, J., Oleson, K., Casson, N. J., North, R. L., Venkiteswaran, J. J., & Whitfield, C. J. (2022). Differences in ebullitive methane release from small, shallow ponds present challenges for scaling. *Science of The Total Environment*, 802, 149685. <https://doi.org/10.1016/j.scitotenv.2021.149685>
- Bartlett, K. B., & Harriss, R. C. (1993). Review and assessment of methane emissions from wetlands. *Chemosphere*, 26(1), 261–320. [https://doi.org/10.1016/0045-6535\(93\)90427-7](https://doi.org/10.1016/0045-6535(93)90427-7)
- Bastviken, D., Cole, J., Pace, M., & Tranvik, L. (2004). Methane emissions from lakes: Dependence of lake characteristics, two regional assessments, and a global estimate. *Global Biogeochemical Cycles*, 18, null. <https://doi.org/10.1029/2004GB002238>

- Bastviken, D., Tranvik, L., Downing, J., Crill, P., & Enrich-Prast, A. (2011). Freshwater Methane Emissions Offset the Continental Carbon Sink. *Science*, *331*, 50–50.
<https://doi.org/10.1126/science.1196808>
- Beaulieu, J. J., DelSontro, T., & Downing, J. A. (2019). Eutrophication will increase methane emissions from lakes and impoundments during the 21st century. *Nature Communications*, *10*(1), Article 1. <https://doi.org/10.1038/s41467-019-09100-5>
- Behrens, D., Langman, J. B., Brooks, E. S., Boll, J., Waynant, K., Moberly, J. G., Dodd, J. K., & Dodd, J. W. (2021). Tracing $\delta^{18}\text{O}$ and $\delta^2\text{H}$ in Source Waters and Recharge Pathways of a Fractured-Basalt and Interbedded-Sediment Aquifer, Columbia River Flood Basalt Province. *Geosciences*, *11*(10), Article 10. <https://doi.org/10.3390/geosciences11100400>
- Belger, L., Forsberg, B. R., & Melack, J. M. (2011). Carbon dioxide and methane emissions from interfluvial wetlands in the upper Negro River basin, Brazil. *Biogeochemistry*, *105*(1), 171–183. <https://doi.org/10.1007/s10533-010-9536-0>
- Bižić, M., Klintzsch, T., Ionescu, D., Hindiyeh, M. Y., Günthel, M., Muro-Pastor, A. M., Eckert, W., Urich, T., Keppler, F., & Grossart, H.-P. (2020). Aquatic and terrestrial cyanobacteria produce methane. *Science Advances*, *6*(3), eaax5343.
<https://doi.org/10.1126/sciadv.aax5343>
- Bonetti, G., Limpert, K. E., Brodersen, K. E., Trevathan-Tackett, S. M., Carnell, P. E., & Macreadie, P. I. (2022). The combined effect of short-term hydrological and N-fertilization manipulation of wetlands on CO_2 , CH_4 , and N_2O emissions. *Environmental Pollution (Barking, Essex: 1987)*, *294*, 118637.
<https://doi.org/10.1016/j.envpol.2021.118637>

- Borges, A. V., Darchambeau, F., Teodoru, C. R., Marwick, T. R., Tamoooh, F., Geeraert, N., Omengo, F. O., Guérin, F., Lambert, T., Morana, C., Okuku, E., & Bouillon, S. (2015). Globally significant greenhouse-gas emissions from African inland waters. *Nature Geoscience*, 8(8), 637–642. <https://doi.org/10.1038/ngeo2486>
- Bortolotti, L. E., St. Louis, V. L., Vinebrooke, R. D., & Wolfe, A. P. (2016). Net Ecosystem Production and Carbon Greenhouse Gas Fluxes in Three Prairie Wetlands. *Ecosystems*, 19(3), 411–425. <https://doi.org/10.1007/s10021-015-9942-1>
- Boyd, C. E. (2015). pH, Carbon Dioxide, and Alkalinity. In C. E. Boyd (Ed.), *Water Quality: An Introduction* (pp. 153–178). Springer International Publishing. https://doi.org/10.1007/978-3-319-17446-4_8
- Brehob, M. M., Pennino, M. J., Handler, A. M., Compton, J. E., Lee, S. S., & Sabo, R. D. (2024). Estimates of Lake Nitrogen, Phosphorus, and Chlorophyll-a Concentrations to Characterize Harmful Algal Bloom Risk Across the United States. *Earth's Future*, 12(8), e2024EF004493. <https://doi.org/10.1029/2024EF004493>
- Brix, H., Sorrell, B. K., & Lorenzen, B. (2001). Are Phragmites-dominated wetlands a net source or net sink of greenhouse gases? *Aquatic Botany*, 69(2), 313–324. [https://doi.org/10.1016/S0304-3770\(01\)00145-0](https://doi.org/10.1016/S0304-3770(01)00145-0)
- Calabrese, S., Garcia, A., Wilmoth, J. L., Zhang, X., & Porporato, A. (2021). Critical inundation level for methane emissions from wetlands. *Environmental Research Letters*, 16(4), 044038. <https://doi.org/10.1088/1748-9326/abedea>
- Canada, S. (2020, November 19). *Net-zero emissions by 2050*. <https://www.canada.ca/en/services/environment/weather/climatechange/climate-plan/net-zero-emissions-2050.html>

- Cavaliere, E., & Baulch, H. M. (2021). Winter in two phases: Long-term study of a shallow reservoir in winter. *Limnology and Oceanography*, 66(4), 1335–1352.
<https://doi.org/10.1002/lno.11687>
- Chen, G., Fang, Y., Huang, J., Zhao, Y., Li, Q., Lai, F., Xu, Y., Tian, X., He, K., Jin, Y., Tan, L., & Zhao, H. (2018). Duckweed systems for eutrophic water purification through converting wastewater nutrients to high-starch biomass: Comparative evaluation of three different genera (*Spirodela polyrhiza*, *Lemna minor* and *Landoltia punctata*) in monoculture or polyculture. *RSC Advances*, 8(32), 17927–17937.
<https://doi.org/10.1039/c8ra01856a>
- Chen, H., Xu, X., Fang, C., Li, B., & Nie, M. (2021). Differences in the temperature dependence of wetland CO₂ and CH₄ emissions vary with water table depth. *Nature Climate Change*, 11(9), 766–771. <https://doi.org/10.1038/s41558-021-01108-4>
- Chen, M., Chang, L., Zhang, J., Guo, F., Vymazal, J., He, Q., & Chen, Y. (2020). Global nitrogen input on wetland ecosystem: The driving mechanism of soil labile carbon and nitrogen on greenhouse gas emissions. *Environmental Science and Ecotechnology*, 4, 100063. <https://doi.org/10.1016/j.esa.2020.100063>
- Christensen, T. R., Ekberg, A., Ström, L., Mastepanov, M., Panikov, N., Öquist, M., Svensson, B. H., Nykänen, H., Martikainen, P. J., & Oskarsson, H. (2003). Factors controlling large scale variations in methane emissions from wetlands. *Geophysical Research Letters*, 30(7). <https://doi.org/10.1029/2002GL016848>
- Clark, I. D., & Fritz, P. (1997). *Environmental Isotopes in Hydrogeology*. CRC Press.
<https://doi.org/10.1201/9781482242911>

- Colas, F., Baudoin, J.-M., Bonin, P., Cabrol, L., Daufresne, M., Lassus, R., & Cucherousset, J. (2021). Ecosystem maturity modulates greenhouse gases fluxes from artificial lakes. *Science of The Total Environment*, 760, 144046. <https://doi.org/10.1016/j.scitotenv.2020.144046>
- Čuhel, J., Šimek, M., Laughlin, R. J., Bru, D., Chèneby, D., Watson, C. J., & Philippot, L. (2010). Insights into the Effect of Soil pH on N₂O and N₂ Emissions and Denitrifier Community Size and Activity. *Applied and Environmental Microbiology*, 76(6), 1870–1878. <https://doi.org/10.1128/AEM.02484-09>
- Davidson, T. A., Audet, J., Jeppesen, E., Landkildehus, F., Lauridsen, T. L., Søndergaard, M., & Syväranta, J. (2018). Synergy between nutrients and warming enhances methane ebullition from experimental lakes. *Nature Climate Change*, 8(2), 156–160. <https://doi.org/10.1038/s41558-017-0063-z>
- DeGrandpre, E. L., DeGrandpre, M. D., Colman, B. P., & Valett, H. M. (2021). Observations of River Solute Concentrations during Ice Formation. *ACS ES&T Water*, 1(8), 1695–1701. <https://doi.org/10.1021/acsestwater.1c00064>
- DelSontro, T., Beaulieu, J. J., & Downing, J. A. (2018). Greenhouse gas emissions from lakes and impoundments: Upscaling in the face of global change. *Limnology and Oceanography Letters*, 3(3), 64–75. <https://doi.org/10.1002/lol2.10073>
- DelSontro, T., Boutet, L., St-Pierre, A., del Giorgio, P. A., & Prairie, Y. T. (2016). Methane ebullition and diffusion from northern ponds and lakes regulated by the interaction between temperature and system productivity. *Limnology and Oceanography*, 61(S1), S62–S77. <https://doi.org/10.1002/lno.10335>

- Denfeld, B. A., Baulch, H. M., del Giorgio, P. A., Hampton, S. E., & Karlsson, J. (2018). A synthesis of carbon dioxide and methane dynamics during the ice-covered period of northern lakes. *Limnology and Oceanography Letters*, 3(3), 117–131.
<https://doi.org/10.1002/lol2.10079>
- Desrosiers, K., DelSontro, T., & del Giorgio, P. A. (2022). Disproportionate Contribution of Vegetated Habitats to the CH₄ and CO₂ Budgets of a Boreal Lake. *Ecosystems*, 25(7), 1522–1541. <https://doi.org/10.1007/s10021-021-00730-9>
- Dlugokencky, E. J., Nisbet, E. G., Fisher, R., & Lowry, D. (2011). Global atmospheric methane: Budget, changes and dangers. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, 369(1943), 2058–2072.
<https://doi.org/10.1098/rsta.2010.0341>
- Donis, D., Flury, S., Stöckli, A., Spangenberg, J. E., Vachon, D., & McGinnis, D. F. (2017). Full-scale evaluation of methane production under oxic conditions in a mesotrophic lake. *Nature Communications*, 8(1), 1661. <https://doi.org/10.1038/s41467-017-01648-4>
- Ducks Unlimited Canada. (2010, March). *Southern Ontario Wetland Conversion Analysis*. Ducks Unlimited Canada.
- Environment and Climate Change Canada. (2016, March 24). *National Wetland Conservation Fund* [Program descriptions]. <https://www.canada.ca/en/environment-climate-change/services/wildlife-habitat/national-wetland-conservation-fund.html>
- Environment and Climate Change Canada. (2018, February 22). *Canada-Ontario Lake Erie action plan*. <https://www.canada.ca/en/environment-climate-change/services/great-lakes-protection/action-plan-reduce-phosphorus-lake-erie.html>

- Environment and Climate Change Canada. (2021a, March 18). *Nature Smart Climate Solutions Fund* [Grants and funding opportunities]. <https://www.canada.ca/en/environment-climate-change/services/environmental-funding/programs/nature-smart-climate-solutions-fund.html>
- Environment and Climate Change Canada. (2021b, October 11). *Canada confirms its support for the Global Methane Pledge and announces ambitious domestic actions to slash methane emissions* [News releases]. <https://www.canada.ca/en/environment-climate-change/news/2021/10/canada-confirms-its-support-for-the-global-methane-pledge-and-announces-ambitious-domestic-actions-to-slash-methane-emissions.html>
- Environment and Climate Change Canada. (2025, June 10). *Canadian Climate Normals 1991-2020 Data*. https://climate.weather.gc.ca/climate_normals/results_1991_2020_e.html?searchType=stnProx&txtRadius=25&optProxType=city&selCity=42%7C59%7C81%7C14%7CLondon&selPark=&txtCentralLatDeg=&txtCentralLatMin=0&txtCentralLatSec=0&txtCentralLongDeg=&txtCentralLongMin=0&txtCentralLongSec=0&txtLatDecDeg=&txtLongDecDeg=&stnID=184000000&dispBack=0
- Feng, Y., Song, Y., Zhu, M., Li, M., Gong, C., Luo, S., Mei, W., Feng, H., Tan, W., & Song, C. (2025). Microbes drive more carbon dioxide and nitrous oxide emissions from wetland under long-term nitrogen enrichment. *Water Research*, 272, 122942. <https://doi.org/10.1016/j.watres.2024.122942>
- Fernandez, J., Townsend-Small, A., Zastepa, A., Watson, S., & Brandes, J. (2019). *Large increases in emissions of methane and nitrous oxide from eutrophication in Lake Erie*. <https://doi.org/10.1101/648154>

- Flanagan, L. B., & Syed, K. H. (2011). Stimulation of both photosynthesis and respiration in response to warmer and drier conditions in a boreal peatland ecosystem. *Global Change Biology*, 17(7), 2271–2287. <https://doi.org/10.1111/j.1365-2486.2010.02378.x>
- Fluet-Chouinard, E., Stocker, B. D., Zhang, Z., Malhotra, A., Melton, J. R., Poulter, B., Kaplan, J. O., Goldewijk, K. K., Siebert, S., Minayeva, T., Hugelius, G., Joosten, H., Barthelmes, A., Prigent, C., Aires, F., Hoyt, A. M., Davidson, N., Finlayson, C. M., Lehner, B., ... McIntyre, P. B. (2023). Extensive global wetland loss over the past three centuries. *Nature*, 614(7947), Article 7947. <https://doi.org/10.1038/s41586-022-05572-6>
- Fontes, J. C., Gonfiantini, R., & Roche, M.-A. (1970). *Deutérium et oxygène-18 dans les eaux du lac Tchad*.
- Ford, P. W., Boon, P. I., & Lee, K. (2002). Methane and oxygen dynamics in a shallow floodplain lake: The significance of periodic stratification. *Hydrobiologia*, 485(1), 97–110. <https://doi.org/10.1023/A:1021379532665>
- Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D. J., Mauritsen, T., Palmer, M. D., Watanabe, M., Wild, M., Zhang, H., Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S. L., Péan, C., Berger, S., & Caud, N. (2021). *The Earth's Energy Budget, Climate Feedbacks, and Climate Sensitivity. In Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (L. G. Chen, M. I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J. B. R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, & B. Zhou, Eds.). Cambridge University Press. <https://doi.org/10.1017/9781009157896.009>.

- Frolking, S., Roulet, N., & Fuglestad, J. (2006). How northern peatlands influence the Earth's radiative budget: Sustained methane emission versus sustained carbon sequestration. *Journal of Geophysical Research: Biogeosciences*, 111(G1).
<https://doi.org/10.1029/2005JG000091>
- Gallart, F., González-Fuentes, S., & Llorens, P. (2024). Technical note: Isotopic fractionation of evaporating waters: effect of sub-daily atmospheric variations and eventual depletion of heavy isotopes. *Hydrology and Earth System Sciences*, 28(1), 229–239.
<https://doi.org/10.5194/hess-28-229-2024>
- Gedney, N., Cox, P. M., & Huntingford, C. (2004). Climate feedback from wetland methane emissions. *Geophysical Research Letters*, 31(20). <https://doi.org/10.1029/2004GL020919>
- Gorsky, A. L., Racanelli, G. A., Belvin, A. C., & Chambers, R. M. (2019). Greenhouse gas flux from stormwater ponds in southeastern Virginia (USA). *Anthropocene*, 28, 100218.
<https://doi.org/10.1016/j.ancene.2019.100218>
- Government of Canada. (2016, January 6). *The Paris Agreement* [Program descriptions; international treaties]. <https://www.canada.ca/en/environment-climate-change/services/climate-change/paris-agreement.html>
- Government of Canada, S. C. (2022, November 23). *Agriculture—Canada at a Glance, 2023*.
<https://www150.statcan.gc.ca/n1/pub/12-581-x/2023001/sec20-eng.htm>
- Greene, S., Walter Anthony, K. M., Archer, D., Sepulveda-Jauregui, A., & Martinez-Cruz, K. (2014). Modeling the impediment of methane ebullition bubbles by seasonal lake ice. *Biogeosciences*, 11(23), 6791–6811. <https://doi.org/10.5194/bg-11-6791-2014>
- Günther, A., Barthelmes, A., Huth, V., Joosten, H., Jurasinski, G., Koebsch, F., & Couwenberg, J. (2020). Prompt rewetting of drained peatlands reduces climate warming despite

- methane emissions. *Nature Communications*, 11(1), 1644.
<https://doi.org/10.1038/s41467-020-15499-z>
- Guo, C., Wang, Y., Lin, S., Du, Q., Wei, Z., Ye, Z., Xu, Y., Yin, B., Zhang, L., & Zhu, Q. (2025). Seasonal variation and influencing factors of gas emission from free water surface flow constructed wetlands. *Ecological Engineering*, 219, 107716.
<https://doi.org/10.1016/j.ecoleng.2025.107716>
- Gutknecht, J. L. M., Goodman, R. M., & Balser, T. C. (2006). Linking soil process and microbial ecology in freshwater wetland ecosystems. *Plant and Soil*, 289(1), 17–34.
<https://doi.org/10.1007/s11104-006-9105-4>
- Hao, X., Ruihong, Y., Zhuangzhuang, Z., Zhen, Q., Xixi, L., Tingxi, L., & Ruizhong, G. (2021). Greenhouse gas emissions from the water–air interface of a grassland river: A case study of the Xilin River. *Scientific Reports*, 11(1), 2659. <https://doi.org/10.1038/s41598-021-81658-x>
- He, T., Ding, W., Cheng, X., Cai, Y., Zhang, Y., Xia, H., Wang, X., Zhang, J., Zhang, K., & Zhang, Q. (2024). Meta-analysis shows the impacts of ecological restoration on greenhouse gas emissions. *Nature Communications*, 15(1), 2668.
<https://doi.org/10.1038/s41467-024-46991-5>
- Hemes, K. S., Chamberlain, S. D., Eichelmann, E., Knox, S. H., & Baldocchi, D. D. (2018). A Biogeochemical Compromise: The High Methane Cost of Sequestering Carbon in Restored Wetlands. *Geophysical Research Letters*, 45(12), 6081–6091.
<https://doi.org/10.1029/2018GL077747>
- Hénault, C., Bourennane, H., Ayzac, A., Ratié, C., Saby, N. P. A., Cohan, J.-P., Eglin, T., & Gall, C. L. (2019). Management of soil pH promotes nitrous oxide reduction and thus

- mitigates soil emissions of this greenhouse gas. *Scientific Reports*, 9(1), 20182.
<https://doi.org/10.1038/s41598-019-56694-3>
- Hillman, W. S. (1961). The Lemnaceae, or duckweeds. *The Botanical Review*, 27(2), 221–287.
<https://doi.org/10.1007/BF02860083>
- Hoehler, T., Losey, N. A., Gunsalus, R. P., & McInerney, M. J. (2018). Environmental Constraints that Limit Methanogenesis. In A. J. M. Stams & D. Sousa (Eds.), *Biogenesis of Hydrocarbons* (pp. 1–26). Springer International Publishing.
https://doi.org/10.1007/978-3-319-53114-4_17-1
- Holgerson, M., & Raymond, P. (2016). Large contribution to inland water CO₂ and CH₄ emissions from very small ponds. *Nature Geoscience*, 9.
<https://doi.org/10.1038/ngeo2654>
- Hondula, K. L., Jones, C. N., & Palmer, M. A. (2021). Effects of seasonal inundation on methane fluxes from forested freshwater wetlands. *Environmental Research Letters*, 16(8), 084016. <https://doi.org/10.1088/1748-9326/ac1193>
- Hu, H., Chen, J., Zhou, F., Nie, M., Hou, D., Liu, H., Delgado-Baquerizo, M., Ni, H., Huang, W., Zhou, J., Song, X., Cao, X., Sun, B., Zhang, J., Crowther, T. W., & Liang, Y. (2024). Relative increases in CH₄ and CO₂ emissions from wetlands under global warming dependent on soil carbon substrates. *Nature Geoscience*, 17(1), 26–31.
<https://doi.org/10.1038/s41561-023-01345-6>
- Huang, L., Gao, X., Guo, J., Ma, X., & Liu, M. (2013). A review on the mechanism and affecting factors of nitrous oxide emission in constructed wetlands. *Environmental Earth Sciences*, 68(8), 2171–2180. <https://doi.org/10.1007/s12665-012-1900-z>

- Jackson, R. B., Saunio, M., Bousquet, P., Canadell, J. G., Poulter, B., Stavert, A. R., Bergamaschi, P., Niwa, Y., Segers, A., & Tsuruta, A. (2020). Increasing anthropogenic methane emissions arise equally from agricultural and fossil fuel sources. *Environmental Research Letters*, 15(7), 071002. <https://doi.org/10.1088/1748-9326/ab9ed2>
- Jensen, S. A., Webb, J. R., Simpson, G. L., Baulch, H. M., Leavitt, P. R., & Finlay, K. (2022). Seasonal variability of CO₂, CH₄, and N₂O content and fluxes in small agricultural reservoirs of the northern Great Plains. *Frontiers in Environmental Science*, 10. <https://www.frontiersin.org/articles/10.3389/fenvs.2022.895531>
- Jensen, S. A., Webb, J. R., Simpson, G. L., Baulch, H. M., Leavitt, P. R., & Finlay, K. (2023). Differential Controls of Greenhouse Gas (CO₂, CH₄, and N₂O) Concentrations in Natural and Constructed Agricultural Waterbodies on the Northern Great Plains. *Journal of Geophysical Research: Biogeosciences*, 128(4), e2022JG007261. <https://doi.org/10.1029/2022JG007261>
- Jin, Q., Liu, H., Xu, X., Zhao, L., Chen, L., Chen, L., Shi, R., & Li, W. (2023). Emission dynamics of greenhouse gases regulated by fluctuation of water level in river-connected wetland. *Journal of Environmental Management*, 329, 117091. <https://doi.org/10.1016/j.jenvman.2022.117091>
- Karimian, H., Huang, J., Chen, Y., Wang, Z., & Huang, J. (2023). A novel framework to predict chlorophyll-a concentrations in water bodies through multi-source big data and machine learning algorithms. *Environmental Science and Pollution Research*, 30(32), 79402–79422. <https://doi.org/10.1007/s11356-023-27886-2>
- Kasak, K., Espenberg, M., Anthony, T. L., Tringe, S. G., Valach, A. C., Hemes, K. S., Silver, W. L., Mander, Ü., Kill, K., McNicol, G., Szutu, D., Verfaillie, J., & Baldocchi, D. D.

- (2021). Restoring wetlands on intensive agricultural lands modifies nitrogen cycling microbial communities and reduces N₂O production potential. *Journal of Environmental Management*, 299, 113562. <https://doi.org/10.1016/j.jenvman.2021.113562>
- Kasak, K., Kill, K., Uuemaa, E., Maddison, M., Aunap, R., Riibak, K., Okiti, I., Teemusk, A., & Mander, Ü. (2022). Low water level drives high nitrous oxide emissions from treatment wetland. *Journal of Environmental Management*, 312, 114914. <https://doi.org/10.1016/j.jenvman.2022.114914>
- Khan, M. N., Huang, J., Shah, A., Li, D., Daba, N. A., Han, T., Du, J., Qaswar, M., Anthonio, C. K., Sial, T. A., Haseeb, A., Zhang, L., Xu, Y., He, Z., Zhang, H., & Núñez-Delgado, A. (2022). Mitigation of greenhouse gas emissions from a red acidic soil by using magnesium-modified wheat straw biochar. *Environmental Research*, 203, 111879. <https://doi.org/10.1016/j.envres.2021.111879>
- Khoiyangbam, R. S., & Chingangbam, S. S. (2022). Assessing seasonal variation of diffusive nitrous oxide emission from freshwater wetland in Keibul Lamjao National Park, Manipur Northeast India. *Atmospheric Environment: X*, 13, 100147. <https://doi.org/10.1016/j.aeaoa.2022.100147>
- Koffi, E. N., Bergamaschi, P., Alkama, R., & Cescatti, A. (2020). An observation-constrained assessment of the climate sensitivity and future trajectories of wetland methane emissions. *Science Advances*, 6(15), eaay4444. <https://doi.org/10.1126/sciadv.aay4444>
- Köhn, D., Welpelo, C., Günther, A., & Jurasinski, G. (2021). Drainage Ditches Contribute Considerably to the CH₄ Budget of a Drained and a Rewetted Temperate Fen. *Wetlands*, 41(6), 71. <https://doi.org/10.1007/s13157-021-01465-y>

- Kosten, S., Kamarainen, A., Jeppesen, E., Van Nes, E. H., Peeters, E. T. H. M., Mazzeo, N., Sass, L. L., Hauzwell, J., Hansel-Welch, N., Lauridsen, T. L., Sondergaard, M., Bachmann, R. W., Lacerot, G., & Scheffer, M. (2009). Climate-related differences in the dominance of submerged macrophytes in shallow lakes. *Global Change Biology*, 15(10), 2503–2517. <https://doi.org/10.1111/j.1365-2486.2009.01969.x>
- Kotsyurbenko, O. R., Glagolev, M. V., Merkel, A. Y., Sabrekov, A. F., & Terentieva, I. E. (2019). Methanogenesis in Soils, Wetlands, and Peat. In A. J. M. Stams & D. Z. Sousa (Eds.), *Biogenesis of Hydrocarbons* (pp. 211–228). Springer International Publishing. https://doi.org/10.1007/978-3-319-78108-2_9
- Kumar, A., Mishra, S., Bakshi, S., Upadhyay, P., & Thakur, T. K. (2023). Response of eutrophication and water quality drivers on greenhouse gas emissions in lakes of China: A critical analysis. *Ecohydrology*, 16(1), e2483. <https://doi.org/10.1002/eco.2483>
- Lauerwald, R., Allen, G. H., Deemer, B. R., Liu, S., Maavara, T., Raymond, P., Alcott, L., Bastviken, D., Hastie, A., Holgerson, M. A., Johnson, M. S., Lehner, B., Lin, P., Marzadri, A., Ran, L., Tian, H., Yang, X., Yao, Y., & Regnier, P. (2023). Inland Water Greenhouse Gas Budgets for RECCAP2: 1. State-Of-The-Art of Global Scale Assessments. *Global Biogeochemical Cycles*, 37(5), e2022GB007657. <https://doi.org/10.1029/2022GB007657>
- Li, X., Liu, X., Zhang, K., Luo, H., Pu, A., Zhuang, D., Jiang, B., Li, M., Chen, W., Fan, L., Qing, J., Zhang, X., Chen, F., & Zhang, X. (2023). Controlling methane emissions from Integrated Vertical-Flow Constructed Wetlands by using potassium peroxymonosulfate as oxidant. *Journal of Environmental Management*, 325, 116444. <https://doi.org/10.1016/j.jenvman.2022.116444>

- Li, Y., Shang, J., Zhang, C., Zhang, W., Niu, L., Wang, L., & Zhang, H. (2021). The role of freshwater eutrophication in greenhouse gas emissions: A review. *Science of The Total Environment*, 768, 144582. <https://doi.org/10.1016/j.scitotenv.2020.144582>
- Li, Z., Xia, S., Zhang, R., Zhang, R., Chen, F., & Liu, Y. (2020). N₂O emissions and product ratios of nitrification and denitrification are altered by K fertilizer in acidic agricultural soils. *Environmental Pollution*, 265, 115065. <https://doi.org/10.1016/j.envpol.2020.115065>
- Liikanen, A., Huttunen, J. T., Karjalainen, S. M., Heikkinen, K., Väisänen, T. S., Nykänen, H., & Martikainen, P. J. (2006). Temporal and seasonal changes in greenhouse gas emissions from a constructed wetland purifying peat mining runoff waters. *Ecological Engineering*, 26(3), 241–251. <https://doi.org/10.1016/j.ecoleng.2005.10.005>
- Limpert, K. E., Carnell, P. E., Trevathan-Tackett, S. M., & Macreadie, P. I. (2020). Reducing Emissions From Degraded Floodplain Wetlands. *Frontiers in Environmental Science*, 8. <https://doi.org/10.3389/fenvs.2020.00008>
- Linquist, B. A., Adviento-Borbe, M. A., Pittelkow, C. M., van Kessel, C., & van Groenigen, K. J. (2012). Fertilizer management practices and greenhouse gas emissions from rice systems: A quantitative review and analysis. *Field Crops Research*, 135, 10–21. <https://doi.org/10.1016/j.fcr.2012.06.007>
- Logozzo, L. A., Soued, C., Bortolotti, L. E., Badiou, P., Kowal, P., Page, B., & Bogard, M. J. (2025). Agricultural Land Use Impacts Aquatic Greenhouse Gas Emissions From Wetlands in the Canadian Prairie Pothole Region. *Global Biogeochemical Cycles*, 39(3), e2024GB008209. <https://doi.org/10.1029/2024GB008209>

- Ma, L., Tong, W., Chen, H., Sun, J., Wu, Z., & He, F. (2018). Quantification of N₂O and NO emissions from a small-scale pond-ditch circulation system for rural polluted water treatment. *Science of The Total Environment*, 619–620, 946–956.
<https://doi.org/10.1016/j.scitotenv.2017.11.192>
- Malyan, S. K., Singh, O., Kumar, A., Anand, G., Singh, R., Singh, S., Yu, Z., Kumar, J., Fagodiya, R. K., & Kumar, A. (2022). Greenhouse Gases Trade-Off from Ponds: An Overview of Emission Process and Their Driving Factors. *Water*, 14(6), Article 6.
<https://doi.org/10.3390/w14060970>
- Mander, Ü., Maddison, M., Valach, A. C., Soosaar, K., Kill, K., & Kasak, K. (2024). High methane emissions as trade-off for phosphorus removal in surface flow treatment wetlands. *Aquatic Botany*, 190, 103719. <https://doi.org/10.1016/j.aquabot.2023.103719>
- Matysek, M., Leake, J., Banwart, S., Johnson, I., Page, S., Kaduk, J., Smalley, A., Cumming, A., & Zona, D. (2019). Impact of fertiliser, water table, and warming on celery yield and CO₂ and CH₄ emissions from fenland agricultural peat. *Science of The Total Environment*, 667, 179–190. <https://doi.org/10.1016/j.scitotenv.2019.02.360>
- McAullife, C. (1971). GC determination of solutes by multiple phase equilibration. *Chem Tech*, Jan, 46–51.
- McNicol, G., Sturtevant, C. S., Knox, S. H., Dronova, I., Baldocchi, D. D., & Silver, W. L. (2017). Effects of seasonality, transport pathway, and spatial structure on greenhouse gas fluxes in a restored wetland. *Global Change Biology*, 23(7), 2768–2782.
<https://doi.org/10.1111/gcb.13580>

- Medland, S. J., Shaker, R. R., Forsythe, K. W., Mackay, B. R., & Rybarczyk, G. (2020). A multi-Criteria Wetland Suitability Index for Restoration across Ontario's Mixedwood Plains. *Sustainability*, 12(23), Article 23. <https://doi.org/10.3390/su12239953>
- Miao, Y., Song, C., Wang, X., Sun, X., Meng, H., & Sun, L. (2012). Greenhouse gas emissions from different wetlands during the snow-covered season in Northeast China. *Atmospheric Environment*, 62, 328–335. <https://doi.org/10.1016/j.atmosenv.2012.08.036>
- Michalak, A. M., Anderson, E. J., Beletsky, D., Boland, S., Bosch, N. S., Bridgeman, T. B., Chaffin, J. D., Cho, K., Confesor, R., Daloğlu, I., DePinto, J. V., Evans, M. A., Fahnenstiel, G. L., He, L., Ho, J. C., Jenkins, L., Johengen, T. H., Kuo, K. C., LaPorte, E., ... Zagorski, M. A. (2013). Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. *Proceedings of the National Academy of Sciences*, 110(16), 6448–6452. <https://doi.org/10.1073/pnas.1216006110>
- Miranda, L. T., & Whitfield, C. J. (2024). Seasonal patterns and key chemical predictors of dissolved greenhouse gases in small prairie pothole ponds. *FACETS*, 9, 1–18. <https://doi.org/10.1139/facets-2023-0134>
- Mitsch, W. J., Bernal, B., & Hernandez, M. E. (2015). Ecosystem services of wetlands. *International Journal of Biodiversity Science, Ecosystem Services & Management*, 11(1), 1–4. <https://doi.org/10.1080/21513732.2015.1006250>
- Mitsch, W. J., Bernal, B., Nahlik, A. M., Mander, Ü., Zhang, L., Anderson, C. J., Jørgensen, S. E., & Brix, H. (2013). Wetlands, carbon, and climate change. *Landscape Ecology*, 28(4), 583–597. <https://doi.org/10.1007/s10980-012-9758-8>

- Moore, T. R., De Young, A., Bubier, J. L., Humphreys, E. R., Lafleur, P. M., & Roulet, N. T. (2011). A Multi-Year Record of Methane Flux at the Mer Bleue Bog, Southern Canada. *Ecosystems*, 14(4), 646–657. <https://doi.org/10.1007/s10021-011-9435-9>
- Moss, B., Kosten, S., Meerhoff, M., Battarbee, R. W., Jeppesen, E., Mazzeo, N., Havens, K., Lacerot, G., Liu, Z., De Meester, L., Paerl, H., & Scheffer, M. (2011). Allied attack: Climate change and eutrophication. *Inland Waters*, 1(2), 101–105. <https://doi.org/10.5268/IW-1.2.359>
- Noguchi, M., Tachibana, S., & Nagahara, H. (2019). Diffusivity and solubility of methane in ice Ih. *GEOCHEMICAL JOURNAL*, 53(1), 83–89. <https://doi.org/10.2343/geochemj.2.0537>
- Obrador, B., von Schiller, D., Marcé, R., Gómez-Gener, L., Koschorreck, M., Borrego, C., & Catalán, N. (2018). Dry habitats sustain high CO₂ emissions from temporary ponds across seasons. *Scientific Reports*, 8, 3015. <https://doi.org/10.1038/s41598-018-20969-y>
- Ollivier, Q. R., Maher, D. T., Pitfield, C., & Macreadie, P. I. (2019). Winter emissions of CO₂, CH₄, and N₂O from temperate agricultural dams: Fluxes, sources, and processes. *Ecosphere*, 10(11), e02914. <https://doi.org/10.1002/ecs2.2914>
- Page, B., Badiou, P., & Steele, O. (2023). Nutrient retention of newly restored wetlands receiving agricultural runoff in a temperate region of North America. *Ecological Engineering*, 195, 107060. <https://doi.org/10.1016/j.ecoleng.2023.107060>
- Peacock, M., Audet, J., Bastviken, D., Cook, S., Evans, C. D., Grinham, A., Holgersson, M. A., Högbom, L., Pickard, A. E., Zieliński, P., & Futter, M. N. (2021). Small artificial waterbodies are widespread and persistent emitters of methane and carbon dioxide. *Global Change Biology*, 27(20), 5109–5123. <https://doi.org/10.1111/gcb.15762>

- Peacock, M., Audet, J., Jordan, S., Smeds, J., & Wallin, M. B. (2019). Greenhouse gas emissions from urban ponds are driven by nutrient status and hydrology. *Ecosphere*, 10(3), e02643. <https://doi.org/10.1002/ecs2.2643>
- Penfound, E., & Vaz, E. (2022). Analysis of 200 years of change in ontario wetland systems. *Applied Geography*, 138, 102625. <https://doi.org/10.1016/j.apgeog.2021.102625>
- Phelps, A. R., Peterson, K. M., & Jeffries, M. O. (1998). Methane efflux from high-latitude lakes during spring ice melt. *Journal of Geophysical Research: Atmospheres*, 103(D22), 29029–29036. <https://doi.org/10.1029/98JD00044>
- Pinto, R., Weigelhofer, G., Brito, A. G., & Hein, T. (2021). Effects of dry-wet cycles on nitrous oxide emissions in freshwater sediments: A synthesis. *PeerJ*, 9, e10767. <https://doi.org/10.7717/peerj.10767>
- Praetzel, L., Schmiedeskamp, M., & Knorr, K.-H. (2021). Temperature and sediment properties drive spatiotemporal variability of methane ebullition in a small and shallow temperate lake. *Limnology and Oceanography*, 66. <https://doi.org/10.1002/lno.11775>
- Qiu, S., Zhang, X., Xia, W., Li, Z., Wang, L., Chen, Z., & Ge, S. (2023). Effect of extreme pH conditions on methanogenesis: Methanogen metabolism and community structure. *Science of The Total Environment*, 877, 162702. <https://doi.org/10.1016/j.scitotenv.2023.162702>
- Rabaey, J., & Cotner, J. (2022). Pond greenhouse gas emissions controlled by duckweed coverage. *Frontiers in Environmental Science*, 10. <https://doi.org/10.3389/fenvs.2022.889289>

- Rabaey, J. S., & Cotner, J. B. (2024). The influence of mixing on seasonal carbon dioxide and methane fluxes in ponds. *Biogeochemistry*, 167(10), 1297–1314.
<https://doi.org/10.1007/s10533-024-01167-7>
- Ramsar Convention Secretariat. (2001). *Wetland Ecosystem Services – An Introduction* (Ramsar Briefing Note No. 1). Ramsar Convention on Wetlands (Ramsar, Iran, 1971).
https://www.ramsar.org/sites/default/files/documents/library/services_01_e.pdf
- Rasilo, T., Prairie, Y. T., & del Giorgio, P. A. (2015). Large-scale patterns in summer diffusive CH₄ fluxes across boreal lakes, and contribution to diffusive C emissions. *Global Change Biology*, 21(3), 1124–1139. <https://doi.org/10.1111/gcb.12741>
- Rosentreter, J. A., Borges, A. V., Deemer, B. R., Holgerson, M. A., Liu, S., Song, C., Melack, J., Raymond, P. A., Duarte, C. M., Allen, G. H., Olefeldt, D., Poulter, B., Battin, T. I., & Eyre, B. D. (2021). Half of global methane emissions come from highly variable aquatic ecosystem sources. *Nature Geoscience*, 14(4), Article 4. <https://doi.org/10.1038/s41561-021-00715-2>
- Søvik, A. K., Augustin, J., Heikkinen, K., Huttunen, J. T., Necki, J. M., Karjalainen, S. M., Kløve, B., Liikanen, A., Mander, Ü., Puustinen, M., Teiter, S., & Wachniew, P. (2006). Emission of the Greenhouse Gases Nitrous Oxide and Methane from Constructed Wetlands in Europe. *Journal of Environmental Quality*, 35(6), 2360–2373.
<https://doi.org/10.2134/jeq2006.0038>
- Salimi, S., Almuktar, S. A. A. A. N., & Scholz, M. (2021). Impact of climate change on wetland ecosystems: A critical review of experimental wetlands. *Journal of Environmental Management*, 286, 112160. <https://doi.org/10.1016/j.jenvman.2021.112160>

- Sapkota, Y., Duball, C., Vaughan, K., Rabenhorst, M., & Berkowitz, J. (2022). Indicator of Reduction in Soil (IRIS) devices: A review. *Science of The Total Environment*, 852, 158419. <https://doi.org/10.1016/j.scitotenv.2022.158419>
- Saunois, M., Martinez, A., Poulter, B., Zhang, Z., Raymond, P. A., Regnier, P., Canadell, J. G., Jackson, R. B., Patra, P. K., Bousquet, P., Ciais, P., Dlugokencky, E. J., Lan, X., Allen, G. H., Bastviken, D., Beerling, D. J., Belikov, D. A., Blake, D. R., Castaldi, S., ... Zhuang, Q. (2025). Global Methane Budget 2000–2020. *Earth System Science Data*, 17(5), 1873–1958. <https://doi.org/10.5194/essd-17-1873-2025>
- Schüpbach, S., Federer, U., Kaufmann, P. R., Hutterli, M. A., Buiron, D., Blunier, T., Fischer, H., & Stocker, T. F. (2009). A new method for high-resolution methane measurements on polar ice cores using continuous flow analysis. *Environmental Science & Technology*, 43(14), 5371–5376. <https://doi.org/10.1021/es9003137>
- Schuster, L., Taillardat, P., Macreadie, P. I., & Malerba, M. E. (2024). Freshwater wetland restoration and conservation are long-term natural climate solutions. *Science of The Total Environment*, 922, 171218. <https://doi.org/10.1016/j.scitotenv.2024.171218>
- Sepulveda-Jauregui, A., Hoyos-Santillan, J., Martinez-Cruz, K., Walter Anthony, K. M., Casper, P., Belmonte-Izquierdo, Y., & Thalasso, F. (2018). Eutrophication exacerbates the impact of climate warming on lake methane emission. *Science of The Total Environment*, 636, 411–419. <https://doi.org/10.1016/j.scitotenv.2018.04.283>
- Sharma, S., Phartiyal, M., Madhav, S., & Singh, P. (2021). Global Wetlands. In *Wetlands Conservation* (pp. 1–16). John Wiley & Sons, Ltd. <https://doi.org/10.1002/9781119692621.ch1>

- Silan, G., Buosi, A., Bertolini, C., & Sfriso, A. (2024). Dynamics and drivers of carbon sequestration and storage capacity in *Phragmites australis*-dominated wetlands. *Estuarine, Coastal and Shelf Science*, 298, 108640.
<https://doi.org/10.1016/j.ecss.2024.108640>
- Smith, R. B., Bass, B., Sawyer, D., Depew, D., & Watson, S. B. (2019). Estimating the economic costs of algal blooms in the Canadian Lake Erie Basin. *Harmful Algae*, 87, 101624.
<https://doi.org/10.1016/j.hal.2019.101624>
- Snell, Elizabeth A. (1987). *Wetland distribution and conversion in southern Ontario* (Nos. En73-4/48E-PDF; Working Paper ; No. 48). Environment Canada, Canada Land Use Monitoring Program, Inland Waters and Lands Directorate.
- Soued, C., Bogard, M. J., Finlay, K., Bortolotti, L. E., Leavitt, P. R., Badiou, P., Knox, S. H., Jensen, S., Mueller, P., Lee, S. C., Ng, D., Wissel, B., Chan, C. N., Page, B., & Kowal, P. (2024). Salinity causes widespread restriction of methane emissions from small inland waters. *Nature Communications*, 15(1), 717. <https://doi.org/10.1038/s41467-024-44715-3>
- Stadmark, J., & Leonardson, L. (2005). Emissions of greenhouse gases from ponds constructed for nitrogen removal. *Ecological Engineering*, 25(5), 542–551.
<https://doi.org/10.1016/j.ecoleng.2005.07.004>
- Stadmark, J., & Leonardson, L. (2007). Greenhouse gas production in a pond sediment: Effects of temperature, nitrate, acetate and season. *Science of The Total Environment*, 387(1), 194–205. <https://doi.org/10.1016/j.scitotenv.2007.07.039>
- Statistics Canada. (2022). *Land use, Census of Agriculture, 2021 (Table 32-10-0406-01)* (Nos. 32-10-0406–01). Statistics Canada.
<https://www150.statcan.gc.ca/t1/tbl1/en/tv.action?pid=3210040601>

- Steffen, M. M., Belisle, B. S., Watson, S. B., Boyer, G. L., & Wilhelm, S. W. (2014). Status, causes and controls of cyanobacterial blooms in Lake Erie. *Journal of Great Lakes Research*, 40(2), 215–225. <https://doi.org/10.1016/j.jglr.2013.12.012>
- Stewart, G. A., Sharp, S. J., Taylor, A. K., Williams, M. R., & Palmer, M. A. (2024). High spatial variability in wetland methane fluxes is tied to vegetation patch types. *Biogeochemistry*, 167(12), 1589–1607. <https://doi.org/10.1007/s10533-024-01188-2>
- Sun, H., Lu, X., Yu, R., Yang, J., Liu, X., Cao, Z., Zhang, Z., Li, M., & Geng, Y. (2021). Eutrophication decreased CO₂ but increased CH₄ emissions from lake: A case study of a shallow Lake Ulansuhai. *Water Research*, 201, 117363. <https://doi.org/10.1016/j.watres.2021.117363>
- Sun, Z., Hu, C., Wu, D., Chen, G., Lu, X., & Liu, X. (2021). Estimation of evaporation losses based on stable isotopes of stream water in a mountain watershed. *Acta Geochimica*, 40(2), 176–183. <https://doi.org/10.1007/s11631-021-00452-8>
- Tangen, B. A., & Bansal, S. (2022). Prairie wetlands as sources or sinks of nitrous oxide: Effects of land use and hydrology. *Agricultural and Forest Meteorology*, 320, 108968. <https://doi.org/10.1016/j.agrformet.2022.108968>
- Thauer, R. K., Kaster, A.-K., Seedorf, H., Buckel, W., & Hedderich, R. (2008). Methanogenic archaea: Ecologically relevant differences in energy conservation. *Nature Reviews Microbiology*, 6(8), 579–591. <https://doi.org/10.1038/nrmicro1931>
- Tian, C., Wang, L., Kaseke, K. F., & Bird, B. W. (2018). Stable isotope compositions ($\delta^2\text{H}$, $\delta^{18}\text{O}$ and $\delta^{17}\text{O}$) of rainfall and snowfall in the central United States. *Scientific Reports*, 8(1), 6712. <https://doi.org/10.1038/s41598-018-25102-7>

- Tian, H., Chen, G., Lu, C., Xu, X., Ren, W., Zhang, B., Banger, K., Tao, B., Pan, S., Chu, M., Zhang, C., Bruhwiler, L., & Wofsy, S. (2015). Global methane and nitrous oxide emissions from terrestrial ecosystems due to multiple environmental changes. *Ecosystem Health and Sustainability*, 1(1). <https://doi.org/10.1890/EHS14-0015.1>
- Tiwari, S., Singh, C., & Singh, J. S. (2020). Wetlands: A Major Natural Source Responsible for Methane Emission. In A. K. Upadhyay, R. Singh, & D. P. Singh (Eds.), *Restoration of Wetland Ecosystem: A Trajectory Towards a Sustainable Environment* (pp. 59–74). Springer. https://doi.org/10.1007/978-981-13-7665-8_5
- Townsend, M., Young, D., & Macko, S. (2003). Kansas Case Study Applications of Nitrogen-15 Natural Abundance Method for Identification of Nitrate Sources. *Journal of Hazardous Substance Research*, 4(1). <https://doi.org/10.4148/1090-7025.1029>
- Turetsky, M. R., Kotowska, A., Bubier, J., Dise, N. B., Crill, P., Hornibrook, E. R. C., Minkinen, 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., & Wilmking, M. (2014). A synthesis of methane emissions from 71 northern, temperate, and subtropical wetlands. *Global Change Biology*, 20(7), 2183–2197. <https://doi.org/10.1111/gcb.12580>
- Turner, K. W., Edwards, T. W. D., & Wolfe, B. B. (2014). Characterising Runoff Generation Processes in a Lake-Rich Thermokarst Landscape (Old Crow Flats, Yukon, Canada) using $\delta^{18}\text{O}$, $\delta^2\text{H}$ and d-excess Measurements. *Permafrost and Periglacial Processes*, 25(1), 53–59. <https://doi.org/10.1002/ppp.1802>
- United Nations. (2018, November 29). *The Paris Agreement—Publication* | UNFCCC. <https://unfccc.int/documents/184656>

- Valach, A. C., Kasak, K., Hemes, K. S., Anthony, T. L., Dronova, I., Taddeo, S., Silver, W. L., Szutu, D., Verfaillie, J., & Baldocchi, D. D. (2021). Productive wetlands restored for carbon sequestration quickly become net CO₂ sinks with site-level factors driving uptake variability. *PLOS ONE*, *16*(3), e0248398. <https://doi.org/10.1371/journal.pone.0248398>
- van Bergen, T. J. H. M., Barros, N., Mendonça, R., Aben, R. C. H., Althuizen, I. H. J., Huszar, V., Lamers, L. P. M., Lüring, M., Roland, F., & Kosten, S. (2019). Seasonal and diel variation in greenhouse gas emissions from an urban pond and its major drivers. *Limnology and Oceanography*, *64*(5), 2129–2139. <https://doi.org/10.1002/lno.11173>
- Villa, J. A., & Bernal, B. (2018). Carbon sequestration in wetlands, from science to practice: An overview of the biogeochemical process, measurement methods, and policy framework. *Ecological Engineering*, *114*, 115–128. <https://doi.org/10.1016/j.ecoleng.2017.06.037>
- Wang, G., Xia, X., Liu, S., Zhang, L., Zhang, S., Wang, J., Xi, N., & Zhang, Q. (2021). Intense methane ebullition from urban inland waters and its significant contribution to greenhouse gas emissions. *Water Research*, *189*, 116654. <https://doi.org/10.1016/j.watres.2020.116654>
- Wang, H., Oertelt, L., & Dittert, K. (2022). The addition of magnesium sulfate and borax to urea reduced soil NH₃ emissions but increased N₂O emissions from soil with grass. *The Science of the Total Environment*, *803*, 149902. <https://doi.org/10.1016/j.scitotenv.2021.149902>
- Wang, S., Lan, B., Yu, L., Xiao, M., Jiang, L., Qin, Y., Jin, Y., Zhou, Y., Armanbek, G., Ma, J., Wang, M., Jetten, M. S. M., Tian, H., Zhu, G., & Zhu, Y.-G. (2024). Ammonium-derived nitrous oxide is a global source in streams. *Nature Communications*, *15*(1), 4085. <https://doi.org/10.1038/s41467-024-48343-9>

- Webb, J. R., Hayes, N. M., Simpson, G. L., Leavitt, P. R., Baulch, H. M., & Finlay, K. (2019). Widespread nitrous oxide undersaturation in farm waterbodies creates an unexpected greenhouse gas sink. *Proceedings of the National Academy of Sciences*, 116(20), 9814–9819. <https://doi.org/10.1073/pnas.1820389116>
- Westermann, P. (1993). Temperature regulation of methanogenesis in wetlands. *Chemosphere*, 26(1), 321–328. [https://doi.org/10.1016/0045-6535\(93\)90428-8](https://doi.org/10.1016/0045-6535(93)90428-8)
- Wetzel, Robert G. (1975). *Limnology*. W.B. Saunders.
- Weyhenmeyer. (1999). *Methane emissions from beaver ponds: Rates, patterns, and transport mechanisms* [Global Biogeochemical Cycles]. Wiley Online Library. <https://agupubs.onlinelibrary.wiley.com/doi/10.1029/1999GB900047>
- Wik, M., Crill, P. M., Bastviken, D., Danielsson, Å., & Norbäck, E. (2011). Bubbles trapped in arctic lake ice: Potential implications for methane emissions. *Journal of Geophysical Research: Biogeosciences*, 116(G3). <https://doi.org/10.1029/2011JG001761>
- Wik, M., Thornton, B. F., Bastviken, D., Uhlbäck, J., & Crill, P. M. (2016). Biased sampling of methane release from northern lakes: A problem for extrapolation. *Geophysical Research Letters*, 43(3), 1256–1262. <https://doi.org/10.1002/2015GL066501>
- Woolway, R. I., Denfeld, B., Tan, Z., Jansen, J., Weyhenmeyer, G. A., & La Fuente, S. (2022). Winter inverse lake stratification under historic and future climate change. *Limnology and Oceanography Letters*, 7(4), 302–311. <https://doi.org/10.1002/lol2.10231>
- Wu, W., Niu, X., Yan, Z., Li, S., Comer-Warner, S. A., Tian, H., Li, S.-L., Zou, J., Yu, G., & Liu, C.-Q. (2023). Agricultural ditches are hotspots of greenhouse gas emissions controlled by nutrient input. *Water Research*, 242, 120271. <https://doi.org/10.1016/j.watres.2023.120271>

- Yang, H., Tang, J., Zhang, C., Dai, Y., Zhou, C., Xu, P., Perry, D. C., & Chen, X. (2020). Enhanced Carbon Uptake and Reduced Methane Emissions in a Newly Restored Wetland. *Journal of Geophysical Research: Biogeosciences*, 125(1), e2019JG005222. <https://doi.org/10.1029/2019JG005222>
- Yang, S.-S., Chen, I.-C., Ching-Pao, L., Liu, L.-Y., & Chang, C.-H. (2015). Carbon dioxide and methane emissions from Tanswei River in Northern Taiwan. *Atmospheric Pollution Research*, 6(1), 52–61. <https://doi.org/10.5094/APR.2015.007>
- Yao, M., Henny, C., & Maresca, J. A. (2016). Freshwater Bacteria Release Methane as a By-Product of Phosphorus Acquisition. *Applied and Environmental Microbiology*, 82(23), 6994–7003. <https://doi.org/10.1128/AEM.02399-16>
- Yao, X., & Song, C. (2022). Effect of different factors dominated by water level environment on wetland carbon emissions. *Environmental Science and Pollution Research*, 29(49), 74150–74162. <https://doi.org/10.1007/s11356-022-20289-9>
- Yao, Z., Yan, G., Ma, L., Wang, Y., Zhang, H., Zheng, X., Wang, R., Liu, C., Wang, Y., Zhu, B., Zhou, M., Rahimi, J., & Butterbach-Bahl, K. (2022). Soil C/N ratio is the dominant control of annual N₂O fluxes from organic soils of natural and semi-natural ecosystems. *Agricultural and Forest Meteorology*, 327, 109198. <https://doi.org/10.1016/j.agrformet.2022.109198>
- Yin, X., Jiang, C., Xu, S., Yu, X., Yin, X., Wang, J., Maihaiti, M., Wang, C., Zheng, X., & Zhuang, X. (2023). Greenhouse Gases Emissions of Constructed Wetlands: Mechanisms and Affecting Factors. *Water*, 15(16), Article 16. <https://doi.org/10.3390/w15162871>

- Yuan, J., Yuan, Y., Zhu, Y., & Cao, L. (2018). Effects of different fertilizers on methane emissions and methanogenic community structures in paddy rhizosphere soil. *Science of The Total Environment*, 627, 770–781. <https://doi.org/10.1016/j.scitotenv.2018.01.233>
- Zhang, J., Chen, H., Wang, M., Liu, X., Peng, C., Wang, L., Yu, D., & Zhu, Q. (2024). An optimized water table depth detected for mitigating global warming potential of greenhouse gas emissions in wetland of Qinghai-Tibetan Plateau. *iScience*, 27(2), 108856. <https://doi.org/10.1016/j.isci.2024.108856>
- Zhang, Y., Zhang, X., Fang, W., Cai, Y., Zhang, G., Liang, J., Chang, J., Chen, L., Wang, H., Zhang, P., Wang, Q., & Zhang, Y. (2025). Carbon sequestration potential of wetlands and regulating strategies response to climate change. *Environmental Research*, 269, 120890. <https://doi.org/10.1016/j.envres.2025.120890>
- Zhang, Z., Poulter, B., Feldman, A. F., Ying, Q., Ciais, P., Peng, S., & Li, X. (2023). Recent intensification of wetland methane feedback. *Nature Climate Change*, 13(5), Article 5. <https://doi.org/10.1038/s41558-023-01629-0>
- Zhang, Z., Zimmermann, N. E., Stenke, A., Li, X., Hodson, E. L., Zhu, G., Huang, C., & Poulter, B. (2017). Emerging role of wetland methane emissions in driving 21st century climate change. *Proceedings of the National Academy of Sciences*, 114(36), 9647–9652. <https://doi.org/10.1073/pnas.1618765114>
- Ziegler, P., Appenroth, K. J., & Sree, K. S. (2023). Survival Strategies of Duckweeds, the World's Smallest Angiosperms. *Plants*, 12(11), 2215. <https://doi.org/10.3390/plants12112215>
- Zou, J., Ziegler, A. D., Chen, D., McNicol, G., Ciais, P., Jiang, X., Zheng, C., Wu, J., Wu, J., Lin, Z., He, X., Brown, L. E., Holden, J., Zhang, Z., Ramchunder, S. J., Chen, A., &

Zeng, Z. (2022). Rewetting global wetlands effectively reduces major greenhouse gas emissions. *Nature Geoscience*, 15(8), Article 8. <https://doi.org/10.1038/s41561-022-00989-0>

Appendix

Appendix A1: This table summarizes the mean nutrient concentrations measured at each wetland site across all sampling campaigns. Values represent the average of all locations collected per site during each campaign.

Site	Campaign	TP (ug/L)	SRP (ug/L)	DP (ug/L)	TN (ppm)	DN (mg/L)	DOC (mg/L)	DIC (ppm C/L)
BL	1	33.32	5.89	14.68	0.75	0.74	4.88	17.40
	2	37.48	2.77	15.53	0.83	0.90	5.45	15.47
	3	40.07	3.87	15.77	0.16	1.06	7.06	17.40
	4	40.00	14.91	13.56	0.85	0.85	5.54	14.17
	5	37.73	5.74	11.60	0.87	0.72	4.52	38.27
	6	50.92	3.73	10.26	0.47	0.43	7.36	17.54
DY	1	51.96	8.74	21.17	0.90	0.82	4.30	18.37
	2	89.79	2.18	16.47	0.92	0.82	4.39	20.27
	3	105.25	6.17	15.86	0.10	0.79	4.95	31.00
	4	85.68	27.07	15.64	0.94	0.69	7.36	15.82
	5	123.66	15.37	19.52	2.35	0.45	6.58	49.30
	6	91.35	6.47	10.40	0.81	0.45	7.04	40.56
FE	1	20.80	11.00	11.69	0.55	0.62	3.13	18.97
	2	25.37	4.38	11.17	0.45	0.59	3.21	26.77
	3	23.94	5.55	11.35	-0.14	0.76	4.20	33.77
	4	20.58	32.39	9.60	0.53	0.55	7.32	15.82
	5	17.14	7.75	8.69	0.50	0.64	8.63	39.74
	6	31.17	3.96	8.84	0.43	0.39	6.74	30.47
LL	1	38.64	5.91	15.98	0.89	0.92	4.98	19.70
	2	123.31	22.11	17.68	1.08	1.21	5.30	26.37
	3	45.95	4.18	15.68	-0.07	0.76	4.08	27.23
	4	45.11	6.51	17.69	0.57	0.62	7.60	7.68
	5	43.41	5.61	10.54	0.59	0.60	8.53	36.97
	6	34.07	3.78	11.60	0.45	0.51	6.28	32.15
MO	1	71.00	3.53	17.09	1.57	1.67	5.28	39.33
	2	492.25	57.07	31.78	3.74	4.18	6.23	34.30
	3	61.68	8.30	18.51	0.82	1.74	5.81	55.30
	4	152.37	6.55	31.95	2.07	1.44	7.67	17.22
	5	26.64	10.16	4.73	4.23	5.19	7.01	43.48
	6	40.62	4.65	9.68	4.24	5.28	6.11	40.78
OH	1	116.98	16.64	29.91	1.39	1.67	8.05	27.87

	2	606.22	37.70	40.77	1.41	1.33	4.66	12.85
	3	76.39	5.73	25.40	0.39	1.02	5.73	20.57
	4	164.35	20.90	30.75	2.72	2.05	6.36	9.25
	5	226.09	6.68	19.38	1.81	1.44	5.65	26.29
	6	116.26	4.78	14.81	1.26	1.15	4.85	27.51
	6	116.26	4.78	14.81	1.26	1.15	4.85	27.51
MA	1	146.11	93.37	100.35	36.92	39.69	4.29	42.33
	2	237.62	38.00	68.46	2.36	2.35	5.20	60.23
	3	124.25	55.19	85.85	0.29	1.09	6.48	64.30
	4	354.66	84.24	65.77	1.26	1.01	6.73	22.27
	5	309.62	248.30	274.51	3.12	5.79	4.72	56.05
	6	260.58	53.04	111.76	2.41	2.48	7.98	32.28

Appendix A2: This table summarizes the mean ion concentrations measured at each wetland site across all sampling campaigns. Values represent the average of all locations collected per site during each campaign.

Site	Campaign	Fluoride (ppm)	Chloride (ppm)	Bromide (ppm)	Nitrite (ppm)	Nitrate (ppm)	Phosphate (ppm)	Sulfate (ppm)
BL	1	0.12	0.27	0.00	0.01	0.33	0.00	0.50
	2	0.11	1.83	0.00	0.00	0.09	0.00	0.48
	3	0.12	1.01	0.00	0.00	0.07	0.00	0.45
	4	0.12	3.35	0.00	0.00	0.05	0.00	0.82
	5	0.10	3.93	0.00	0.00	0.10	0.00	1.46
	6	0.10	4.17	0.00	0.00	0.10	0.00	1.01
DY	1	0.14	8.32	0.00	0.00	0.17	0.00	10.98
	2	0.14	8.37	0.00	0.00	0.09	0.00	8.40
	3	0.16	10.13	0.00	0.00	0.05	0.00	8.84
	4	0.15	11.91	0.00	0.00	0.07	0.00	12.82
	5	0.14	10.91	0.00	0.00	0.06	0.00	10.96
	6	0.14	11.20	0.00	0.00	0.16	0.00	10.77
FE	1	0.14	7.80	0.00	0.00	0.85	0.00	0.61
	2	0.13	9.17	0.00	0.00	0.11	0.00	0.50
	3	0.13	10.74	0.00	0.00	0.05	0.00	0.43
	4	0.14	11.14	0.00	0.00	0.02	0.00	0.15
	5	0.12	12.62	0.00	0.00	0.20	0.00	0.87
	6	0.11	10.00	0.00	0.00	0.18	0.00	0.87
LL	1	0.15	1.34	0.00	0.00	0.03	0.00	0.50

	2	0.18	5.01	0.01	0.06	0.80	0.00	2.52
	3	0.21	0.43	0.00	0.00	0.07	0.00	0.45
	4	0.22	0.74	0.00	0.00	0.06	0.00	0.51
	5	0.18	3.64	0.04	0.00	0.09	0.00	1.35
	6	0.15	2.25	0.00	0.00	0.08	0.00	1.76
MO	1	0.21	9.09	0.04	0.09	1.84	0.00	50.10
	2	0.20	2.80	0.01	0.07	21.70	0.00	15.73
	3	0.21	7.62	0.05	0.01	2.17	0.00	30.55
	4	0.23	9.60	0.08	0.00	0.28	0.00	31.35
	5	0.21	17.88	0.09	0.04	30.28	0.00	77.26
	6	0.25	14.18	0.05	0.11	31.28	0.00	134.21
OH	1	0.23	17.06	0.07	0.02	0.48	0.00	17.25
	2	0.18	3.25	0.02	0.07	1.47	0.00	6.58
	3	0.20	9.33	0.06	0.00	0.06	0.00	12.96
	4	0.24	14.00	0.06	0.28	6.44	0.00	20.80
	5	0.18	9.58	0.03	0.04	2.52	0.00	21.16
	6	0.21	20.34	0.06	0.05	2.04	0.03	43.09
MA	1	0.12	15.42	0.04	2.35	173.06	0.15	21.98
	2	0.12	12.18	0.05	0.54	7.20	0.07	13.01
	3	0.13	9.95	0.05	0.00	0.09	0.00	3.56
	4	0.12	10.51	0.06	0.00	0.04	0.00	3.95
	5	0.13	17.78	0.05	0.46	27.14	0.60	15.81
	6	0.14	9.63	0.01	0.08	12.00	0.20	10.22

Site	Campaign	Lithium (ppm)	Sodium (ppm)	Ammonium (ppm)	Potassium (ppm)	Calcium (ppm)	Magnesium (ppm)
BL	1	0.01	0.83	0.03	1.92	25.19	2.57
	2	0.00	0.76	0.02	1.09	27.36	2.58
	3	0.01	0.24	0.03	0.81	24.53	2.67
	4	0.00	0.73	0.03	4.65	44.39	4.34
	5	0.01	1.22	0.03	5.30	60.27	4.74
	6	0.01	2.06	0.07	3.31	29.42	2.95
DY	1	0.01	3.70	0.03	0.59	21.41	12.03
	2	0.00	3.91	0.02	1.20	26.40	11.82
	3	0.01	4.13	0.03	2.43	37.16	12.87
	4	0.02	4.96	0.02	3.08	47.64	15.17
	5	0.01	4.44	0.03	2.60	80.62	15.89
	6	0.00	3.70	0.06	0.52	68.63	12.76
FE	1	0.01	0.75	0.06	0.19	24.49	7.98

	2	0.01	1.26	0.03	0.52	39.70	9.10
	3	0.03	1.32	0.09	0.83	45.95	9.07
	4	0.01	1.40	0.01	0.85	52.14	9.39
	5	0.01	2.04	0.03	0.48	62.73	10.19
	6	0.01	1.52	0.06	0.32	53.22	8.04
<i>LL</i>	1	0.01	2.31	0.03	0.64	18.80	7.61
	2	0.00	3.33	0.04	2.69	34.71	5.65
	3	0.01	3.00	0.14	1.04	36.17	5.73
	4	0.02	2.50	0.02	1.91	20.17	5.39
	5	0.00	3.43	0.03	6.14	50.40	8.73
	6	0.00	2.92	0.05	3.73	45.92	8.05
<i>MO</i>	1	0.00	12.37	0.23	1.29	46.47	24.86
	2	0.00	4.27	0.13	2.04	50.85	10.47
	3	0.00	8.68	0.54	2.09	73.07	19.67
	4	0.02	8.96	0.18	3.96	55.33	19.54
	5	0.02	13.46	0.17	0.26	82.70	24.88
	6	0.02	18.40	0.05	0.20	83.29	34.45
<i>OH</i>	1	0.00	13.90	0.17	2.53	37.85	8.88
	2	0.00	3.19	0.08	1.59	20.25	3.40
	3	0.00	7.87	0.05	1.73	32.69	5.93
	4	0.00	10.02	0.07	3.31	39.67	7.25
	5	0.01	9.00	0.30	2.67	42.10	7.79
	6	0.01	14.44	0.16	0.56	54.42	11.43
<i>MA</i>	1	0.00	7.17	0.57	6.94	112.60	17.13
	2	0.01	7.32	0.31	4.45	87.98	15.45
	3	0.02	6.14	0.11	3.57	77.56	14.98
	4	0.02	4.74	0.06	4.80	68.42	11.52
	5	0.03	6.36	0.11	9.66	72.63	14.00
	6	0.00	3.56	0.08	3.14	57.36	8.58

Appendix A3: Descriptive statistics for diffusive CH₄, CO₂, and N₂O fluxes were calculated by site using R. Metrics include the mean, standard deviation (SD), median, interquartile range (IQR), minimum, and maximum. This dataset encompasses the entire sampling year, providing a mean of the annual emissions. For winter conditions, only snow and ice measurements were used

to capture actual fluxes at the time. Duckweed (DW) measurements were excluded because they were only available for campaigns 2 and 3.

Descriptive Statistics of Gas Fluxes by Site							
Site	Mean	Std Dev	Median	IQR	Minimum	Maximum	Coeff. of Var.
Diffusive CH ₄ Flux (mmol/m ² /day)							
BL	3.959	5.056	2.030	3.276	0.403	17.871	1.277
DY	2.487	3.176	1.354	1.848	0.632	12.705	1.277
FE	3.533	4.642	2.109	4.015	0.000	18.875	1.314
LL	2.667	2.386	2.138	1.974	0.228	9.129	0.895
MA	46.799	65.625	24.458	63.465	0.002	221.701	1.402
MO	4.049	5.545	2.234	4.430	0.107	21.897	1.370
OH	1.661	1.475	1.389	1.072	-0.003	5.687	0.888
Diffusive CO ₂ Flux (mmol/m ² /day)							
BL	34.791	54.469	11.875	38.639	-16.793	183.388	1.566
DY	8.752	50.771	10.194	21.173	-93.816	151.572	5.801
FE	66.879	101.662	32.590	49.050	-53.014	309.026	1.520
LL	45.137	61.686	57.073	86.022	-47.265	140.814	1.367
MA	315.369	177.819	374.686	227.068	1.623	649.739	0.564
MO	79.667	54.282	62.893	72.652	7.610	184.597	0.681
OH	2.762	40.394	-11.010	20.759	-50.300	97.570	14.623
Diffusive N ₂ O Flux (μmol/m ² /day)							
BL	-2.050	4.419	-1.405	3.763	-12.021	4.289	-2.156
DY	-3.016	3.093	-2.422	3.838	-10.988	1.046	-1.026
FE	7.828	58.235	-3.586	8.331	-36.102	223.000	7.440
LL	-177.653	414.980	-3.122	7.479	-1,454.628	12.785	-2.336
MA	815.909	2,094.431	7.325	146.361	-16.724	7,387.895	2.567
MO	18.625	17.053	18.762	23.422	-5.057	47.914	0.916
OH	-38.457	183.395	1.757	16.519	-694.925	84.867	-4.769

Appendix A4: Descriptive statistics of diffusive CH₄, CO₂, and N₂O fluxes across sampling campaigns (June–April). Sites with duckweed (DW) were excluded, and for the winter campaign (January/February), only Ice and Snow locations were included. Values represent the mean, standard deviation, median, interquartile range (IQR), minimum, and maximum flux for each month.

Descriptive Statistics of Gas Fluxes by Month						
Month	Mean	Std Dev	Median	IQR	Minimum	Maximum
Diffusive CH ₄ Flux (mmol/m ² /day)						
June	11.195	35.249	2.812	3.860	0.580	164.635
July	5.722	8.689	2.000	2.546	0.152	32.250
August	21.802	51.827	3.004	7.573	0.414	221.701
October	9.123	18.504	1.553	4.370	0.228	79.253
January/February	0.271	0.438	0.055	0.338	−0.003	1.102
April	1.560	1.298	1.063	1.580	0.303	5.415
Diffusive CO ₂ Flux (mmol/m ² /day)						
June	45.752	118.132	12.525	103.939	−93.816	421.703
July	107.087	117.540	64.646	111.767	−16.711	408.611
August	103.292	148.205	18.406	168.910	−31.930	432.202
October	116.199	190.493	51.904	92.432	−50.300	649.739
January/February	7.645	17.168	4.616	10.169	−12.682	38.430
April	51.501	74.674	21.769	58.672	−11.684	293.063
Diffusive N ₂ O Flux (μmol/m ² /day)						
June	0.445	1.947	−0.001	0.023	−1.455	7.388
July	0.027	0.070	0.010	0.029	−0.036	0.304
August	−0.001	0.009	−0.004	0.004	−0.017	0.020
October	0.001	0.020	−0.003	0.002	−0.013	0.085
January/February	−0.001	0.004	0.000	0.000	−0.010	0.001
April	0.005	0.016	0.001	0.008	−0.014	0.048

Appendix A5: Descriptive statistics of bubble gas release measured across sites, encompassing all funnels on the transects. Summary metrics include mean, standard deviation (SD), median, interquartile range (IQR), minimum, and maximum. No measurements were taken in winter.

Descriptive Statistics of Bubble Gas Release by Site						
Site	Mean	Std Dev	Median	IQR	Minimum	Maximum
Total Gas Released (ml/day/cm ²)						
BL	0.101	0.174	0.007	0.072	0.000	0.444
DY	0.036	0.039	0.020	0.067	0.000	0.115
FE	0.031	0.054	0.013	0.035	0.000	0.249
LL	0.049	0.042	0.043	0.061	0.000	0.134
MA	0.063	0.051	0.065	0.090	0.000	0.142
MO	0.086	0.062	0.086	0.086	0.002	0.224
OH	0.022	0.031	0.006	0.026	0.000	0.118

Appendix A6: Descriptive statistics of bubble gas release measured across months, encompassing all funnels on the transects. Summary metrics include mean, standard deviation (SD), median, interquartile range (IQR), minimum, and maximum. No measurements were taken in winter.

Descriptive Statistics of Bubble Gas Release by Month						
Month	Mean	Std Dev	Median	IQR	Minimum	Maximum
Total Gas Released (ml/day/cm ²)						
June	0.058	0.044	0.053	0.069	0.000	0.133
July	0.058	0.076	0.044	0.054	0.000	0.410
August	0.107	0.110	0.071	0.082	0.006	0.444
October	0.012	0.027	0.003	0.005	0.000	0.106
April	0.015	0.025	0.002	0.019	0.000	0.104

Appendix A7: Significant Kendall rank correlations (τ) between diffusive GHG fluxes and environmental parameters at middle wetland locations during summer (June to August). Only variables with $p < 0.10$ are shown. Positive τ values indicate a direct relationship (both variables increase or decrease together), while negative τ values indicate an inverse relationship (one variable increases as the other decreases). Kendall's τ and corresponding p-values were calculated using the `cor.test()` function (`method = "kendall"`) in the R package `stats`. Unless otherwise stated, variables were measured at the surface of the wetland.

Diffusive Flux	Variable	Tau	p-value
CH₄	Dissolved Oxygen Depth (%)	-0.48	0.003
	Dissolved Oxygen Surface (%)	-0.39	0.013
	Max Depth (m)	0.30	0.057
	POM Carbon-13 Isotope	-0.30	0.061
	pH	-0.27	0.085
CO₂	pH	-0.70	0.000
	Calcium (ppm)	0.66	0.000
	Dissolved Inorganic Carbon (ppm C/L)	0.61	0.000
	Dissolved Oxygen Surface (%)	-0.61	0.000
	Specific Conductivity (μS/cm)	0.51	0.001
	POM Carbon-13 Isotope	-0.46	0.004
	Potassium (ppm)	0.38	0.017
	Lithium (ppm)	0.37	0.034
	Ammonium (ppm)	0.36	0.025
	Nitrate (ppm)	0.33	0.035
	Nitrite (ppm)	0.56	0.002
	Dissolved Nitrogen (mg/L)	0.50	0.002
N₂O	Nitrate (ppm)	0.48	0.003
	Total Nitrogen (ppm)	0.45	0.005
	Potassium (ppm)	0.42	0.007
	Dissolved Phosphorus (μg/L)	0.42	0.008
	Sulfate (ppm)	0.39	0.013
	pH	-0.39	0.014
	Oxygen-18 Isotope	-0.38	0.016
	Total Phosphorus (μg/L)	0.38	0.016
	Phosphate (ppm)	0.31	0.099
	Hydrogen-2 Isotope	-0.29	0.070
	Dissolved Oxygen Surface (%)	-0.28	0.080
	Calcium (ppm)	0.27	0.091

Appendix A8: Summary of GHG flux measurements across study sites. Diffusive flux values are reported for CO₂, CH₄, and N₂O, while ebullitive fluxes are reported only for CH₄. Methods used for diffusive flux include the floating chamber (FC) and headspace extraction (HE) techniques; methods used for ebullitive flux include the floating chamber (FC) and bubble trap (BT). * = mean value; ** = modeled value (adapted from (Jensen et al., 2023)).

Citation	System	Region	Method	Diffusive CO ₂ (mmol m ⁻² day ⁻¹)	Diffusive CH ₄ (mmol m ⁻² day ⁻¹)	Diffusive N ₂ O (μmol m ⁻² day ⁻¹)	CH ₄ Ebullition (mmol m ⁻² day ⁻¹)
This Study (excluding Site MA)	Restored Agricultural Wetlands	Southern Ontario	FC, HE, BT	-93.8 to 309 (23.2)	-0.003 to 21.9 (3)	-1450 to 220 (-1.4)	
Outlier Site MA				1.6 to 649.7 (270.3)	0.002 to 506.1 (25.3)	-61 to 7390 (5.4)	
Weyhenmeyer (1999)	Beaver Ponds	Central Ontario	FC, HE, BT	-	0.42 to 4.21	-	-
Jensen et al. (2023)	Agricultural Reservoirs	Saskatchewan, Canada	HE	-30 to 464	0 to 13.5	-9.66 to 11.61	
Jensen et al. (2023)	Natural Wetland Ponds	Saskatchewan, Canada	HE	-31 to 784	0.13 to 14.11	-13.5 to 2.2	
Webb, Hayes, et al. (2019)	Agricultural Reservoirs	Saskatchewan, Canada	HE	-	-	-12 to 166	-
Webb, Leavitt, et al. (2019)	Agricultural Reservoirs	Saskatchewan, Canada	HE	-21 to 466	0.14 to 92	-	-
Bortolotti et al. (2016)	Natural and Restored Wetland Ponds	Saskatchewan, Canada	HE	-60 to 1,350	0.7 to 13.3	-	-
Badiou et al. (2011)	Prairie Wetland Ponds	Prairie Pothole Region, Canada	FC	-	0.11 to 3.81	1.43 to 20.0	-
Miranda and Whitfield (2024)	Prairie Pothole Ponds	Prairie Pothole Region, Canada	HE	-38 to 710 (5.0)	0.003 to 100 (0.41)	-11 to 420 (-0.9)	-
Prėskienis et al. (2021)	Tundra Lakes and Ponds	Nunavut, Canada	HE, BT	-9.25 to 23.39	0.108 to 0.955	-	0.04 to 5.255
D'Acunha and Johnson (2019)	Constructed Stormwater Wetlands	British Columbia, Canada	HE	10.41 to 1,266	0 to 55.5	-7.44 to 57.47	-
DelSontro et al. (2016)	Beaver Ponds	Quebec, Canada	FC, BT	57*	4.2*	-	4.6*
McClure et al. (2020)	Eutrophic Reservoir	North America	FC, BT	-	0.03 to 1.37	-	0.023 to 2.24
Villa et al. (2021)	Temperate Freshwater Marsh	Ohio, USA	FC	-	0.78*	-	0 to 95.04
McNicol et al. (2017)	Restored Wetland	California, USA	HE, FC	95 to 3,773	0.035 to 7.42	-8.64 to 103.68	1.15 to 2.3
Dalcin Martins et al. (2017)	Prairie Pothole Wetlands	North Dakota, USA	FC	60*	7.0*	-	-
Tangen et al. (2015)	Pothole Basin	North Dakota, Minnesota, Iowa, USA	FC	-	69*	14*	-
Rabaei and Cotner. (2022)	Small Artificial & Natural Ponds with Duckweed Influence	Minnesota, USA	FC, HE, BT	-2.16*	54*	14.2*	4.59*