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Hydrologic Variability Drives Differential Methane Dynamics in Agricultural Reservoirs of the Northern Great Plains

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Key Points:

- Increased spring runoff in a wet year resulted in 65% deeper water columns compared to a dry year
- This deeper water column caused increased anoxia and CH_4 concentrations in deepwater areas of agricultural reservoirs
- Deep CH_4 was not correlated with surface diffusive fluxes, likely due to offsetting mechanisms of CH_4 oxidation

Supporting Information:

Supporting Information may be found in the online version of this article.

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Abstract Climate variability can regulate aquatic methane fluxes as increasing temperatures can elevate microbial metabolic rates, including methanogenesis. It is less well known how climate-induced variability in seasonal precipitation and runoff might affect methane concentrations and fluxes in aquatic ecosystems. Here, we measured seasonal methane concentrations and calculated diffusive fluxes from 20 agricultural reservoirs in the northern Great Plains in contrasting wet and dry summers. Relative to the dry year, water column depths increased 65% (from 1.7 to 2.6 m) in the wet year and was associated with stronger stratification and increased anoxia at depth. Solute concentrations also declined during the wet year, with sulfate concentrations less than half that observed in the dry year (645 mg SO_4^{2-} L^{-1} vs. 1620 mg SO_4^{2-} L^{-1}). Together, the more profound anoxia combined with lower sulfate concentrations resulted in significantly higher hypolimnetic CH_4 concentrations in the wet year (40.3 μM) compared with the dry year (18.1 μM), particularly in August (30-fold higher). Despite these patterns, surface CH_4 concentrations and estimated diffusive emissions did not significantly increase in wet summers (1.13 μM and 2.31 mmol m^{-2} yr^{-1}) relative to dry summers (3.78 μM and 5.71 mmol m^{-2} yr^{-1}), likely owing to offsetting mechanisms of increased CH_4 storage and oxidation through the deeper water column. Climate-driven changes in precipitation and runoff are expected to modify the physical factors controlling methanogenesis and methanotrophy. Our findings show corresponding minimal effects on diffusive fluxes of methane, but future studies should also address ebullition and seasonal turnover to capture the full CH_4 budget of inland waters.

Plain Language Summary Inland waters are estimated to contribute >50% to the total global methane emissions, but it is unclear how current and future climate change may affect these rates. Previous research suggests that increasing global temperatures are likely to increase methane emissions from these ecosystems, but the impact of changing precipitation and runoff patterns is much less clear. Here, we demonstrate that a wetter climate resulted in deeper water depths in small agricultural reservoirs, which created conditions more favorable for methane production (e.g., low oxygen levels in deep waters and dilution of sulfate that can otherwise suppress methane production). Interestingly, the elevated methane concentrations in deep waters did not result in concurrently higher surface methane concentrations or estimated diffusive fluxes to the atmosphere, possibly due to offsetting mechanisms created by the deeper water column, namely increased methane storage, and oxidation. Future work should additionally evaluate how bubbling of methane and seasonal water column mixing affect full methane budgets.

1. Introduction

Inland waters are estimated to collectively contribute ~53% to the total global CH_4 emissions from natural and anthropogenic sources (Rosentreter et al., 2021). Small water bodies <1 km^2 (Holgerson et al., 2022) in particular, are known to disproportionately contribute to carbon and methane cycling (Gilbert et al., 2021; Holgerson & Raymond, 2016). Small agricultural reservoirs can produce some of the highest freshwater CH_4 emissions (Malerba et al., 2022) but are understudied relative to natural aquatic ecosystems (Peacock et al., 2021). Although reservoirs account for <10% of the global inland water area (Mendonça et al., 2017), small reservoirs have demonstrated higher rates of methanogenesis than natural lakes at the regional scale (DelSontro et al., 2016) arising from elevated organic carbon concentrations, reduced O_2 within shallow water columns (Holgerson, 2015), and

more contact with shallow sediment and littoral zones (Malyan et al., 2022; Peeters et al., 2019; Yakimovich et al., 2020). Continued climate and land-use changes are further complicating our ability to predict and scale methane emissions, as temperature can differentially affect rates of methanogenesis and methanotrophy, and precipitation may indirectly affect these processes.

Previous research investigating the effects of climate change on aquatic methane emissions has primarily focused on temperature effects (Feron et al., 2024), with less work examining the effects of changing seasonal precipitation patterns (Niu et al., 2025; Sanches et al., 2019). Small, closed-basin waterbodies in the northern Great Plains (NGP) may be particularly sensitive to variations in intensity and frequency of precipitation events with climate change, which can dramatically affect water levels (2–10 m: Bjoerndahl et al., 2022; Gushulak et al., 2024; van der Kamp et al., 2008), although the relationships between precipitation and water level can be complex in the NGP (Pomeroy et al., 2007). Generally, in deeper water bodies, the pathway of CH_4 through oxic water from the deep-water to the surface becomes longer and nutrient concentrations are lower, leading to reduced diffusive CH_4 emissions (Li, Huang, et al., 2020; Li, Peng, et al., 2020). Increased rainfall can also lower temperatures and dilute solutes, reducing both diffusive and ebullitive CH_4 fluxes (Niu et al., 2025). The impact of varying water levels on methane emissions has been examined in managed reservoirs, where effects of drawdown are shown to increase CH_4 emissions, through reduction of methane oxidation through the water column and increased rates of ebullition (Harrison et al., 2017). These quick drawdown effects on CH_4 emissions, however, may not be directly comparable to effects of water level change driven by seasonal or annual variation in precipitation and runoff, which also affect watershed loading and the concentration of solutes from evaporation (evapoconcentration). In the future, the NGP is predicted to experience more extreme precipitation events (Bonsal et al., 2011; Laird et al., 2003; St-Jacques et al., 2018) and industrial irrigation or wetland drainage (Spence et al., 2022). Such climate and anthropogenic forcing will have profound direct and indirect impacts on water levels, thermal stratification, and solute concentrations with expected, but as yet unquantified, impacts on CH_4 emissions.

There are three main pathways by which variability in seasonal precipitation regimes and water fluxes could impact aquatic CH_4 concentrations in small reservoirs in the NGP. First, increased water-column stability and stratification may increase anoxia in the hypolimnion and provide optimal redox conditions for methanogenesis (D'Ambrosio et al., 2022; Duc et al., 2010; Hounshell et al., 2021; Sobek et al., 2012; Webb et al., 2019b) and methane storage (Bastviken et al., 2008). Second, increased watershed loading of labile organic carbon and nutrients may increase substrate availability for methanogenesis (Berberich et al., 2020; Thottathil et al., 2019; Webb et al., 2019b; West et al., 2016; Yakimovich et al., 2020). Third, and in contrast to the previous point, increased water availability may dilute solute concentrations and alter stratification regimes, possibly reducing available substrates or competitive inhibitors (e.g., sulfate; Hammer, 1986). Differentiating between these mechanisms is important for understanding how future climate change may impact CH_4 emissions from these small waterbodies.

Here, we examine the effect of changing interannual patterns of runoff on water body depth, CH_4 concentrations, and associated diffusive emissions in small agricultural reservoirs of the NGP during the open water season. This work builds on previous regional studies that demonstrated these water bodies are significant sources of CH_4 with concentration and fluxes correlated with oxygen availability, productivity, sulfate concentrations, and water column stability (Jensen et al., 2022; Soued et al., 2024; Webb et al., 2019b). By comparing CH_4 concentrations in surface and deep waters in 20 agricultural reservoirs between hydrologically dry (2021) and wet (2022) years, we evaluated the magnitude of impact of water level changes on CH_4 concentrations and potential diffusive fluxes. Specifically, our objectives were to: (a) determine the effect of deeper water depths on water column stability and hypolimnetic oxygen levels; (b) evaluate the effect of increased runoff on solute concentrations (total dissolved nitrogen and phosphorus (TDN, TDP) and SO_4^{2-}); (c) evaluate the effect of (a, b) on methane concentrations at the sediment-water interface, and; (d) quantify the fate of dissolved methane at depth and the impact on net CH_4 emission rates via diffusion at the water-air interface. Together, this research will help inform predictions of future methane emissions from understudied small artificial waterbodies under changing hydrological regimes.

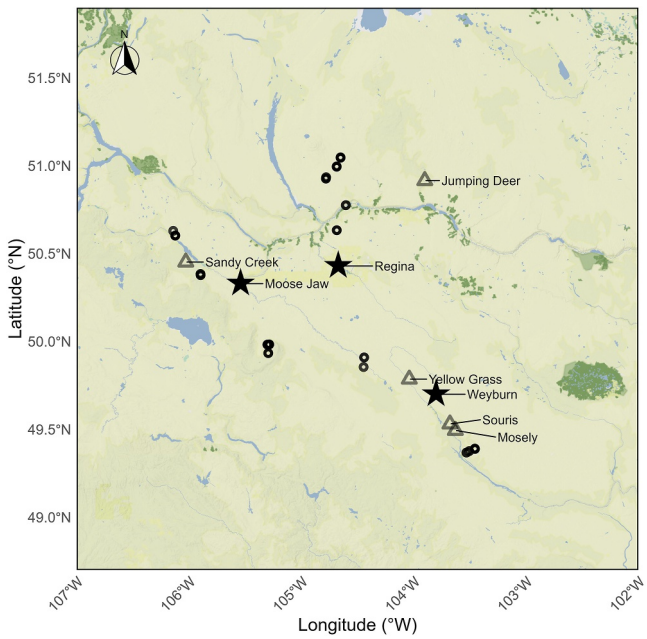

Figure 1. Location of study sites. Open circles indicate locations of 20 agricultural reservoirs included in this study. Several locations included more than one reservoir in close proximity. Stars represent Environment and Climate Change Canada weather station sites, and open triangles represent stream gauging sites monitored by the Saskatchewan Water Security Agency.

2. Methods

2.1. Study Site

This study was conducted in southern Saskatchewan, Canada, located in the NGP. The region is known for its vast and diverse agricultural land use, including extensive cattle ranching and wheat, canola, and pulse crops. The region is characterized by sub-humid conditions (Sauchyn & Kerr, 2016), with high evaporation rates (water deficit of 20–60 cm yr^{-1}, Haig et al., 2021). Small, endorheic water bodies in this region can experience extreme fluctuations in water level owing to variations in winter and summer precipitation and lack of connectivity (Bjoerndahl et al., 2022), and many small wetland ponds are ephemeral (Stewart & Kantrud, 1971). Current annual precipitation averages 365.3 mm yr^{-1} over the past several decades (ECCC, 2023), and is highly variable, with differences between years >300 mm (Gushulak et al., 2024). Long-term trends in annual precipitation demonstrate increasing spring precipitation, while winter precipitation has steadily decreased since the 1980s (Gushulak et al., 2024), but precipitation does not always equate to increasing water levels (Pomeroy et al., 2007).

For this study, we contrasted methane concentrations and calculated diffusive fluxes in 20 small constructed agricultural ponds in 2 years of varying runoff and water levels. The reservoirs were constructed between 10 and 50 years ago, based on communication with landowners. The sediments of 17 similar sites were probed with a telescoping pole in 2017 (J. Webb unpublished data) and sediment depths were observed to range from 17 to 115 cm (mean = 53 cm), suggesting that there is sufficient organic matter for microbial activity. One site (T1) was dredged in 2021, and thus the change in depth for this reservoir cannot be attributed solely to changing runoff levels. The sites were all located within a 200 km radius of the city of Regina, Saskatchewan, Canada (Figure 1).

Cumulative precipitation (rain and snow) from Jan – May at three weather stations was 67.1 mm (±47.8 mm) in 2021, and 109.4 mm (±28.3) in 2022. Although annual precipitation was not unusually high, we classify 2022 here as a hydrologically wet year due to much higher (~7,000%) runoff volumes than what was observed in 2021 at 5 unregulated stream gauging sites in the region (Figure S4 in Supporting Information S1, data from the Saskatchewan Water Security Agency), and local water levels were observed to be much higher in 2022. Average air temperatures between April-Aug did not differ significantly among years (14.4 and 13.6°C in 2021 and 2022, ANOVA $p = 0.11$). Wind direction (2021: 200.33 ± 10.55°; 2022: 210.97 ± 0.81°), and speed (2021: 7.76 ± 6.86 km/hr, 2022: 6.99 ± 5.97 km/hr) did not differ among years (all $p > 0.05$).

2.2. Field Collection

The 20 study sites were sampled once per month from May to August in 2021 and 2022 between the hours of 9:00–13:00, covering the open-water season but excluding spring and fall turnover. Water sampling was conducted at the deepest location in the reservoir as determined by a handheld depth meter, and Secchi disc depths were taken at each site. Profiles of water quality measurements were collected using a Yellow Springs Instrument (YSI, Yellow Springs, OH, USA) multiparameter probe. We measured dissolved oxygen (DO), temperature, conductivity, and pH in the water column at 0.5 m intervals from ~0.25 m below the surface to ~0.25 m above the sediment. Surface and bottom water samples were collected using a 2 L Van Dorn at 0.25 m below the surface, and 0.25 m above the sediment into rinsed carboys for transport to the laboratory. When water depths were between 0.5 and 1.0 m, we collected samples directly below the water surface, and directly above the sediment surface (~10 cm), and sites that were <0.5 m in depth were not included in the analyses focusing on surface and deep measurements.

Surface and deep methane concentrations were collected using the headspace equilibrium method, similar to Webb et al. (2019) and Jensen et al. (2022). At each site, 1.2 L glass-serum bottles were slowly filled to overflowing to evacuate any air bubbles in the sample. The bottle was sealed using a rubber stopper with two 3-lock

valves. The bottle was inoculated with 60 mL of atmospheric air using a syringe and simultaneously removing 60 mL of water. This headspace was equilibrated with aqueous CH_4, by shaking for 2 min. Samples were then extracted from the headspace and evacuated into 12 mL exetainer vials (Ho et al., 2021). Duplicate samples were collected at each site and results were averaged for one measurement of CH_4 concentration at each sample-date.

2.3. Laboratory Analyses

A minimum of 2 L of surface and deep water was filtered through 0.45 µm cellulose fiber filters and stored at 4°C until laboratory analyses within 6 months of collection. Filtered water was analyzed for dissolved nutrients using a Lachat QuickChem Ion analyzer for total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP), and sulfate concentrations were analyzed using the SmartChem 200 Wet Chemistry Analyzer. Dissolved organic carbon (DOC) was analyzed using standard analytic procedures on a Shimadzu model 5000A Total Carbon Analyzer. All chemical analyses were conducted at the Institute of Environmental Change and Society (IECS) at the University of Regina (U of R) following methods outlined in Swarbrick et al. (2022).

Duplicate water samples for chlorophyll *a* (Chl *a*) were filtered on to a Whatman GF/C filter (1.2 µm nominal pore size), wrapped in foil, and frozen (−10°C) until analysis. Chlorophyll *a* concentration was measured using standard trichromatic spectrophotometric methods (Finlay et al., 2009; Jeffrey & Humphrey, 1975) using a Shimadzu Spectrophotometer in the Environmental Quality Analysis Laboratory (EQAL) at the University of Regina.

Methane gas samples were analyzed at the Global Institute for Water Security, University of Saskatchewan as detailed in Jensen et al. (2022, 2023). Briefly, headspace gas samples were analyzed for the dry molar fraction of each gas using a fully calibrated Scion 456 Gas Chromatograph (Bruker Ltd.) with Combipal autosampler (CTC Analytics—PAL System), using argon as the carrier gas. All gas samples were stored at room temperature until analysis, within 2 months of collection.

2.4. Numerical Analyses

The dry molar fraction of CH_4 was estimated following methods reported in Webb et al. (2019b) and Jensen et al. (2022, 2023). Dissolved CH_4 concentrations were used to estimate the diffusive flux (f_C) of CH_4 using the gas transfer velocity (k_c), gas concentration of the water (C_{water}), and the equilibrium concentration (C_{eq}) following equation (Equation 1):

$$f_C = k_c(C_{water} - C_{eq}). \quad (1)$$

Mean CH_4 concentration in air was 1.868 µatm. Diffusive flux was calculated using the gas transfer velocity as measured in comparable small agricultural reservoirs in the region for CH_4 (1.64 m day^{-1}; Webb et al., 2019a). We chose to standardize gas transfer velocity based on site-specific measurements as common wind-based empirical developed for lakes and reservoirs (e.g., Cole & Caraco, 1998; Vachon & Prairie, 2013) may underestimate k600 in small lentic systems (Holgerson et al., 2017; Vachon & Prairie, 2013). Ebullitive fluxes were not measured in this study.

Water-column stratification was estimated using the squared Brunt-Väisälä buoyancy frequency (BF_{max}, s^{-2}). Here, we examined the steepest density gradient measured for water temperature profiles taken at 0.5-m intervals and was calculated using the *rLakeAnalyzer* package (Read et al., 2012) in R (version 4.3.2; R Core Team, 2021) following Webb et al. (2019b). To classify the mixing regime of the reservoirs, we used 0.287 kg m^{-3} m^{-1} as the maximum water density difference threshold following Holgerson et al. (2022). Sites that were never stratified over the 4 months were classified as "mixed," while those that alternated between stratified and unstratified were classified as "polymictic" and those that were either stratified over all 4 months, or were unstratified in May, and stratified from June-August were classified as "stratified." Sites that became fully or nearly desiccated (<0.25 m) were removed from the analyses as it was not possible to evaluate differences in surface and deep conditions.

To determine the differences in water body characteristics and CH_4 concentrations between dry (2021) and wet (2022) years, we used a repeated measures analysis of variance (RM-ANOVA), with month and year as "within" and "between" factors, using the *rstatix* (Kassambara, 2023) package in R. The RM-ANOVAs were performed on log-10 (most variables) or square root (BF_{max}) transformed variables. Significance is reported at $p < 0.05$ and

using a Bonferroni correction for multiple comparisons ($p < 0.002$). Gas concentrations were not collected in July 2021, so comparisons of CH_4 concentrations and fluxes focused on May, June, and August samples only.

Generalized additive models (GAMs, Wood, 2017; Wood et al., 2016) were used to examine potential environmental drivers of surface and deep CH_4 concentrations in the agricultural reservoirs. Predictor variables in the models were selected based on prior knowledge of factors known to influence production or consumption of CH_4 (Jensen et al., 2023; Soued et al., 2024; Webb et al., 2019b). Deep CH_4 concentration was modeled as a function of deep dissolved O_2, deep SO_4^{2-}, deep DOC:TDN (as a measure of an indicator of organic carbon availability and lability), and deep water temperature. We predicted that surface CH_4 concentrations should increase with CH_4 production in anoxic (deep) waters but that surface CH_4 levels would be reduced by oxidation during diffusion, and, as such, would be related to differences in water column thermal stability, and availability of dissolved oxygen. Thus surface CH_4 concentration was modeled as a function of deep CH_4 concentration, surface dissolved O_2 and SO_4^{2-}, water body depth, and buoyancy frequency. GAMs were used for these analyses because of their ability to model linear, nonlinear, and nonmonotonic relationships between response and predictor variables (Monteith et al., 2014; Orr et al., 2015; Webb et al., 2019a, 2019b). All modeling was done using the *mgcv* package (Wood, 2011; Wood et al., 2016) in R (version 4.3.2; R Core Team, 2021).

Year was included as a parametric fixed effect in the models, and a factor-smooth interaction was used for each smooth term to allow the smooth effects of each covariate in the model to vary between years. The gamma (loglink function) distribution was used to describe the CH_4 concentration resulting in a GAM, that is, suitable for positive continuous responses. The gam.check() function was used to assess agreement with assumptions by examining basis size, dispersion of residuals, homogeneity of variance, and the relationship between the observed and predicted response. Residual marginal likelihood (REML) was used for selection of smoothness parameters (Wood, 2011), and the double penalty approach of Marra and Wood (2011) was used to aid in model selection. Parameters predicting CH_4 concentrations were considered significant at 95% confidence level ($\alpha = 0.05$) for each waterbody.

3. Results

Water depths in 2022 were significantly higher than those recorded in 2021, resulting in significant differences in physicochemical conditions and biogeochemical variables in the reservoirs among year. 2021 experienced low water levels ($\overline{Z}_{max}$ = 1.7 m, SD = 1.0 m, min = 0.1 m, max = 3.8 m), while 2022 demonstrated statistically significantly higher water levels ($\overline{Z}_{max}$ = 2.6 m, SD = 1.0 m, min = 0.7 m, max = 5.8 m) (*t*-test $p = 0.019$) (Table 1, Figure 2, and Figure S1 in Supporting Information S1). Overall, Secchi depth was significantly greater (RM-ANOVA $p < 0.05$) in 2022 (1.07 vs. 0.55 m) while deepwater temperature (14.55°C vs. 16.83°C), chlorophyll *a* (13.84 μg L^{-1} vs. 68.31 μg L^{-1}), and surface (417.7 mg L^{-1} vs. 1619.4 mg L^{-1}) and deep SO_4^{2-} concentrations (645.3 mg L^{-1} vs. 1619.4 mg L^{-1}) were lower in wet 2022. Several variables had significant year-month interactions, suggesting strong seasonality, including buoyancy frequency, secchi disc depth, surface and deep temperature, surface and TDP, TDN, dissolved oxygen, and dissolved methane (Table 1).

Deeper water depths were associated with elevated water-column stability (0.0078 s^{-2} vs. 0.0069 s^{-2}) and reduced oxygen availability at depth (2.84 mg L^{-1} vs. 3.67 mg L^{-1}) in 2022 (Figure S2 in Supporting Information S1). As well, more reservoirs were classified as "stratified" in 2022 versus 2021 ($n = 8$) than in 2021 ($n = 3$), with fewer in the "polymictic" category in ($n = 1$ vs. 5). The remaining sites were categorized as "mixed." Deep O_2 concentrations were inversely correlated with buoyancy frequency ($R^2 = 0.178$, $p < 0.0001$). The degree of decline in deepwater oxygen concentrations was greater later in summer and was particularly notable in August 2022 (Figure 2d).

Higher water levels in the wet year resulted in a significantly lower concentrations of TDN and SO_4^{2-}, but higher concentrations of TDP (Table 1 and Figures 3a and 3b). On average, deepwater SO_4^{2-} concentrations were less than half in 2022 when compared to 2021 (e.g., 645.3 mg L^{-1} vs. 1619.4 mg L^{-1}), while TDN concentrations in 2022 ranged from 50% to 80% of concentrations in 2021 (Table 1). Both surface and deep CH_4 concentrations showed a significant year-month interaction in the RM-ANOVAs (Table 1). The largest difference in CH_4 concentrations between years was observed in August, where mean deepwater CH_4 concentration was 112.0 μM (SD = 196.4 μM) in the wet year but was only 4.4 μM (SD = 9.4 μM) in the dry year. Interestingly, while deep CH_4 concentrations were ~ two-fold greater in the wet year, surface CH_4 concentrations were marginally lower in 2022,

Table 1
Mean and Standard Deviation (SD) of Major Physicochemical Parameters in 20 Agricultural Reservoirs in the NGP

			RM-ANOVA *p*-value		
Variable	2021 (dry)	2022 (wet)	Year	Month	Year-Month
Depth (m)	1.79 (1.03)	2.61 (1.01)	**0.0190**	**0.00018***	0.114
Buoyancy Frequency (BF; s^{-2})	0.006973 (0.007922)	0.007851 (0.006074)	0.249	0.107	**0.027**
Secchi (m)	0.55 (0.51)	1.07 (0.54)	**<0.001***	0.341	**0.013**
Chlorophyll *a* (μg L^{-1})	68.31 (137.7)	13.84 (33.16)	**0.022**	0.351	0.119
Surface temp (°C)	19.48 (4.42)	19.61 (3.68)	0.08	**<0.001***	**<0.001***
Deep temp (°C)	16.83 (3.86)	14.45 (4.94)	**0.023**	**<0.001***	**0.004**
Surface pH	8.66 (0.68)	8.85 (0.71)	0.140	0.194	**0.034**
Deep pH	8.14 (0.83)	8.07 (0.90)	0.95	0.14	**0.0002***
Surface TDP (μg L^{-1})	430.2 (710.7)	567.8 (825.9)	0.472	0.137	**0.003**
Deep TDP (μg L^{-1})	507.6 (837.9)	904.3 (1246.6)	0.179	0.154	**0.015**
Surface TDN (μg L^{-1})	3949.3 (2966.5)	1957.4 (1238.0)	**<0.001***	**0.026**	**0.001***
Deep TDN (μg L^{-1})	4138.7 (2716.2)	3457.2 (5434.7)	0.189	**0.032**	0.186
Surface DO (mg L^{-1})	8.13 (4.58)	7.45 (2.99)	0.564	**0.013**	**0.010**
Deep DO (mg L^{-1})	3.67 (3.56)	2.84 (3.87)	0.094	0.369	**0.033**
Surface DOC (mg L^{-1})	29.65 (20.36)	19.07 (9.51)	**0.043**	**0.002***	0.099
Deep DOC (mg L^{-1})	31.75 (21.86)	22.58 (21.53)	**0.024**	0.125	0.270
Surface SO_4^{2-} (mg L^{-1})	1149.5 (1414.8)	417.7 (581.7)	**0.021**	0.081	0.342
Deep SO_4^{2-} (mg L^{-1})	1619.4 (1748.2)	645.3 (1399.8)	**0.011**	0.457	0.369
Surface CH_4 (μM)	3.78 (6.61)	1.13 (1.97)	0.196	0.601	**0.009**
Deep CH_4 (μM)	18.41 (52.32)	40.28 (115.26)	0.602	0.096	**0.0004***
Surface CH_4 flux (mmol m^{-2} day^{-1})	5.711 (14.86)	2.31 (3.65)	0.182	0.627	0.073

Note. *P*-values indicate results from RM-ANOVA considering year (2021 vs. 2022), month (May–August) and year-month interaction. Surface indicates measurements taken at 0.25 m below the surface, while deep samples were collected 0.25 m above the sediment. Bolded values indicate significance at $p < 0.05$, * indicates significance with Bonferroni correction $p < 0.002$.

mostly early in the summer (Figures 3c and 3d and Figure S3 in Supporting Information S1) and resulted in no significant difference in diffusive methane fluxes between the wet and dry years (2.31 vs. 5.71 mmol m^{-2} day^{-1}, Table 1).

The GAM for deep CH_4 had 69.8% deviance explained, and year was a significant fixed effect. The partial plots (Figure 4) indicate that CH_4 concentrations were correlated significantly and negatively with deep dissolved O_2 concentrations and deepwater SO_4^{2-} concentrations (Figures 4a–4d). Overall, deep CH_4 concentrations were corelated with DOC to TDN ratios, but the direction of the relationships differed between 2022 (negative) and 2021 (unimodal) (Figures 4e and 4f). Temperature was a not a significant predictor in the model.

The GAM for surface CH_4 concentration explained 73.1% of deviance, with surface CH_4 concentrations being correlated positively with deep CH_4 concentrations, and negatively with surface sulfate and chlorophyll *a* concentrations (Figure 5). BFmax, surface O_2, and water column depth were all non-significant predictors in the model.

When examined more closely, surface CH_4 concentrations are positively correlated with deep CH_4 concentrations (mixed: $R^2 = 0.73$, $p < 0.0001$; polymictic: $R^2 = 0.39$, $p < 0.0001$; stratified: $R^2 = 0.12$, $p = 0.042$), but these relationships were all shallower than 1:1 line (Figure 6).

Major axis regression models indicated that slope estimates for polymictic lakes (0.09) were significantly steeper than for stratified lakes (0.004, $p < 0.001$). Mixed lakes (0.04) were intermediate, not significantly different from either group.

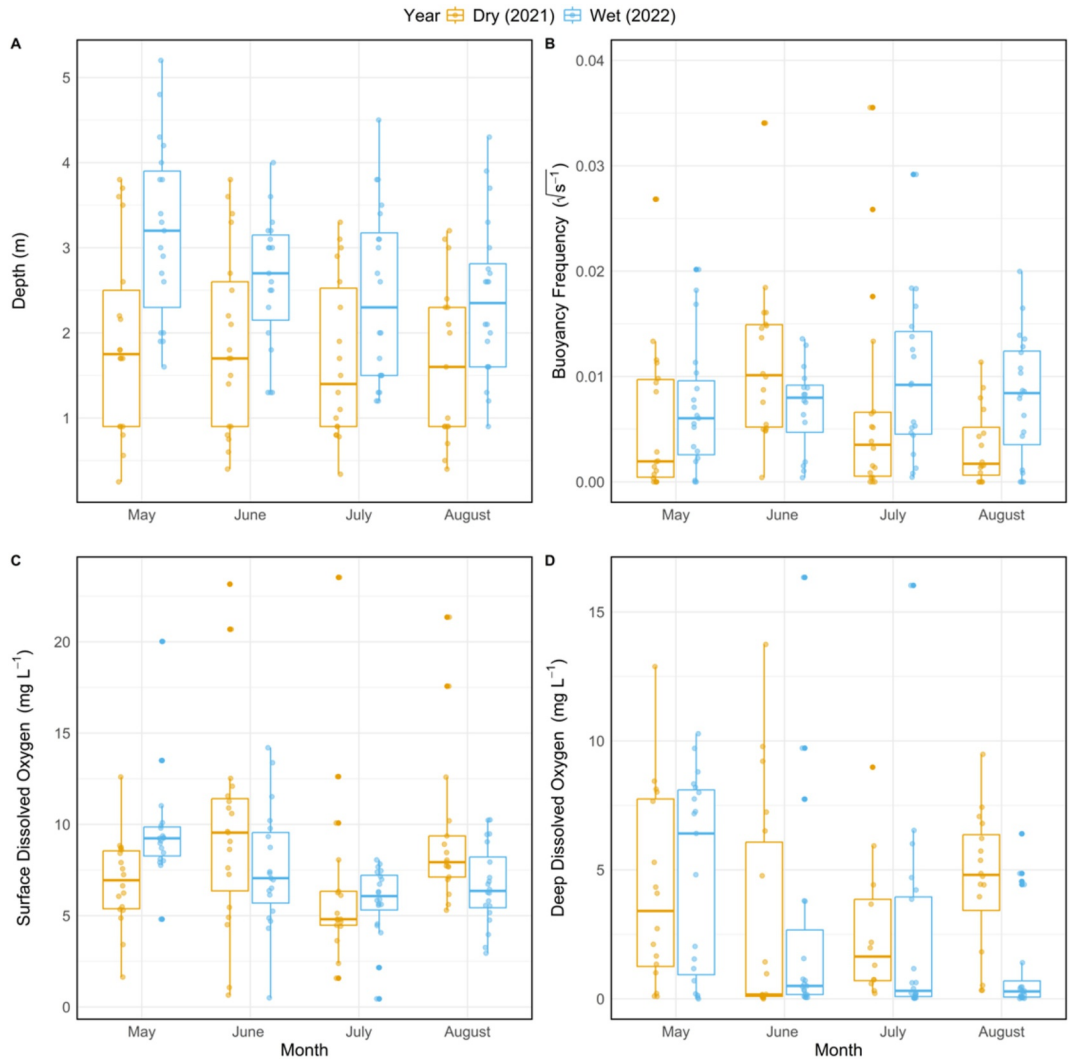

Figure 2. Water column depth (a), buoyancy frequency (b), surface dissolved oxygen (c), and deep dissolved oxygen concentrations in each summer month of 2021 (orange) and 2022 (blue bars). RM-ANOVA analyses indicating significant differences are summarized in Table 1.

4. Discussion

We found that a year with higher spring runoff resulted in significantly deeper water column (65%) in small agricultural ponds of the Northern Great Plains, which was associated with increased thermal stratification, enhanced anoxia in hypolimnetic waters, reduced solute concentrations, and increased deep water concentrations of methane. Although surface CH_4 concentrations were found to be positively correlated with hypolimnetic CH_4 concentrations, this relationship was much shallower than a 1:1 relationship, and we did not measure significantly higher surface methane concentrations or inferred diffusive fluxes during the wet year. These results indicate that while climate change may induce substantial variation in water level of small prairie ecosystems, these changes in depth potentially supported both increased methanogenesis and methanotrophy, resulting in no net change in immediate CH_4 diffusive flux at the reservoir surface.

4.1. Effect of Reservoir Depth on Physical and Chemical Characteristics

Increased water column depth resulted in greater water column stability, depleted oxygen at depth, and more favorable habitat for methanogenesis. This is a common observation as water depth increases across different water body types, including large freshwater systems that develop a stronger oxycline as water depth increases

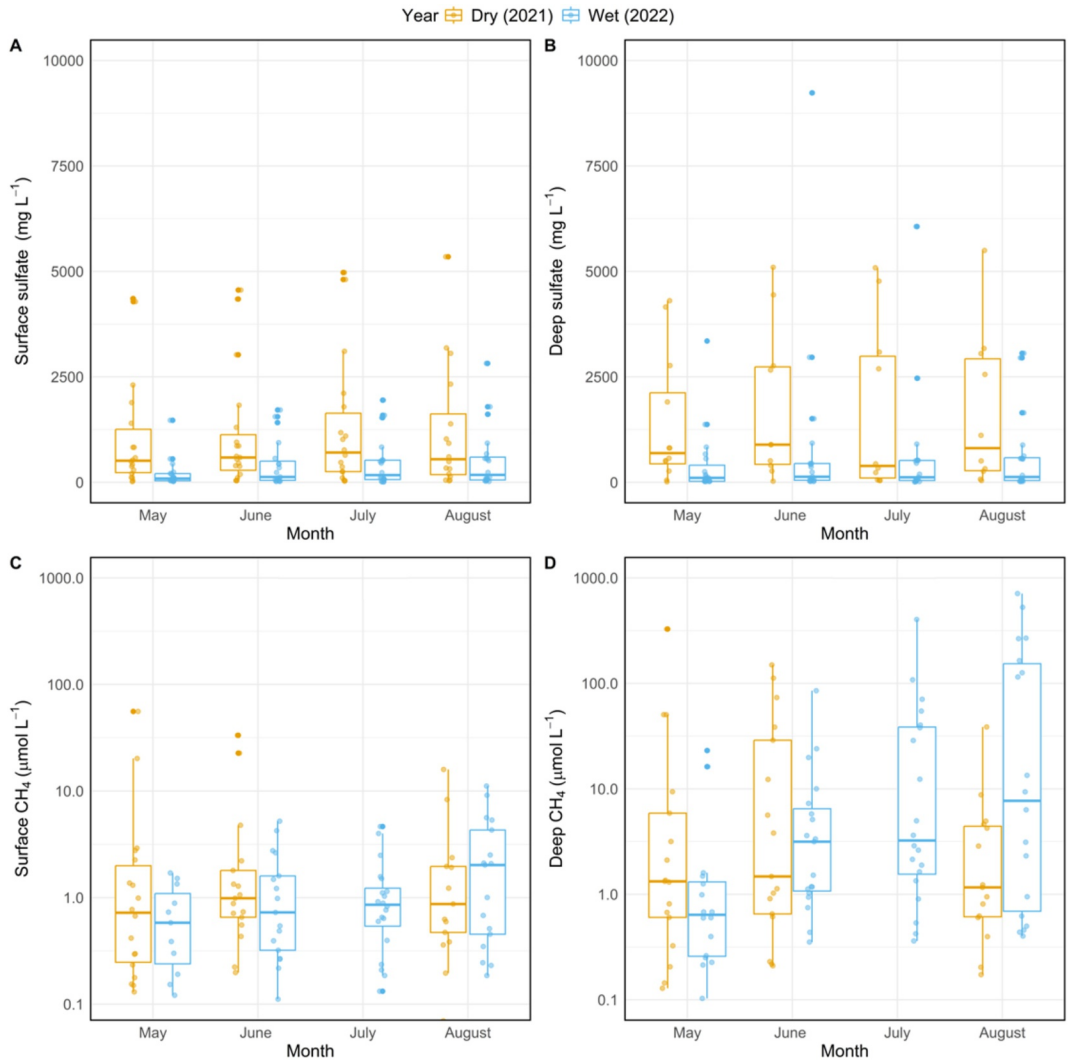

Figure 3. Boxplots of surface (a) and deep (b) sulfate (mg L^{-1}) and surface (c) and deep methane (d) for 20 agricultural reservoirs in a dry year (2021 orange boxes) and a wet year (2022 blue boxes). RM-ANOVA analyses indicating significant differences are summarized in Table 1.

(Li, Huang, et al., 2020; Li, Peng, et al., 2020). The induced stratification in wet conditions is amplified through the summer (Hounshell et al., 2021) as surface waters become warmer, and deeper waters remain cooler, leading to conditions conducive to methanogenesis. Previous research suggests that with climate warming, enhanced thermal stratification and associated hypoxia will become stronger as water layers become more segregated (Hounshell et al., 2021; Jenny et al., 2015).

Increased runoff reduced the concentration of several solutes in the agricultural reservoirs, including sulfate, TDN, and DOC (Table 1). Increasing precipitation has been observed to cause the dilution of solutes (including DOC), lowering of pH, and reduction of gas concentrations and water transparency (Holgerson, 2015; Thomas & Ratcliffe, 1973). However, depending on runoff conditions in the surrounding landscape, higher precipitation can also increase the loading of labile DOC (Holgerson, 2015). Although rarely measured in other studies, we additionally observed a decrease in sulfate concentrations in the wet year (Table 1, Figure 2). Interestingly, we observed an increase in TDP in the wet year (Table 1), suggesting that increased anoxia resulted in increased P release from sediments (Prairie et al., 2001). Thus, while increasing precipitation can either dilute solutes, or increase loading from the catchment, our observations primarily support a mechanism of solute dilution.

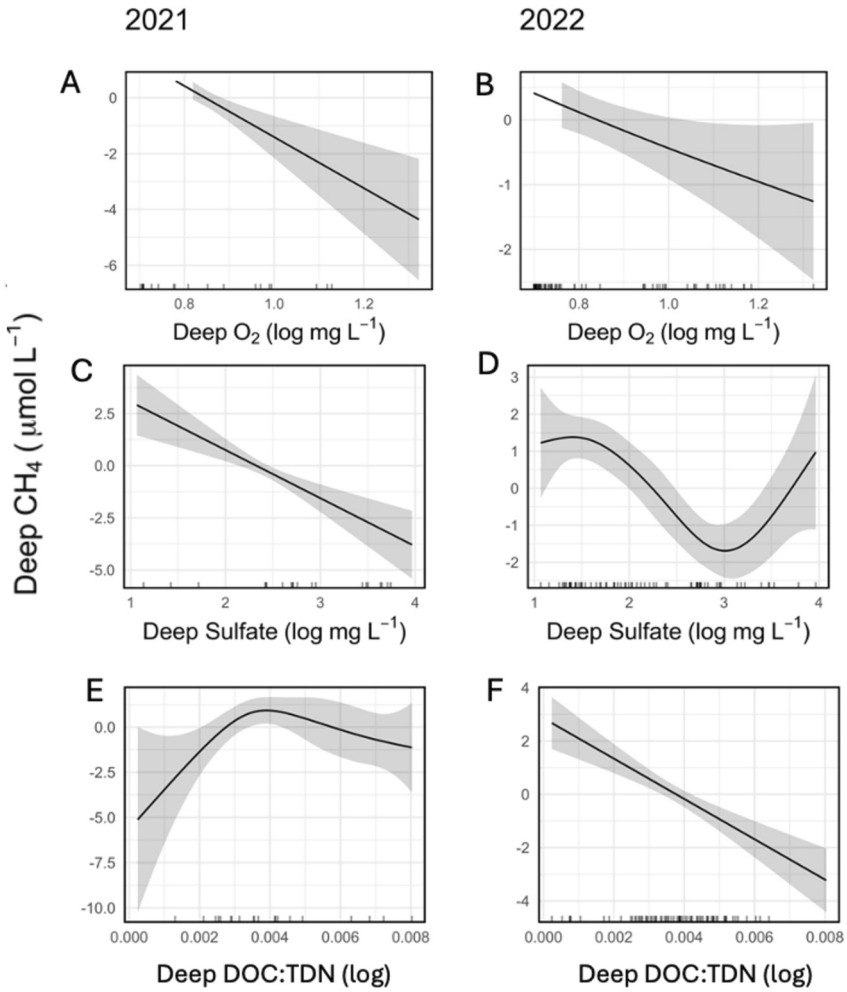

Figure 4. Partial effects plots from the generalized additive model relating deep CH_4 concentrations in 2021 (left) and 2022 (right) to water column conditions. Overall deviance explained = 69.8%. Gray shaded region represents the 95% Bayesian credible interval. Ticks along the *x*-axis represent the measured data points. Covariates included deep dissolved oxygen (a, b), sulfate (c, d), and DOC:TDN ratios (e, f).

4.2. Effects of Depth on Deep Methane Concentrations

In both 2021 and 2022 we observed a negative relationship between deep CH_4 and sulfate concentrations (Figure 4), but the relationship was non-linear in 2022 (Figures 4c and 4d). Sulfate concentrations have been demonstrated to mediate methanogenesis (Baron et al., 2022; Borrel et al., 2016), as sulfate reduction is energetically favorable over methanogenesis (Abram & Nedwell, 1978; Baron et al., 2022) and can also contribute to methane oxidation (Timmers et al., 2016). In 2022, CH_4 concentrations actually increased as sulfate increased above ~1,000 mg L^{-1} sulfate, perhaps indicating the effects of other regulatory processes (Figure 4d). Specifically, there was a significant positive relationship between DOC and sulfate (regression $R^2 = 0.52$, $p < 0.001$), so it is likely that the extremely high sulfate ponds additionally had sufficient DOC to support both sulfate reduction and methanogenesis (Martins et al., 2017).

The DOC:TDN ratios were used as an indicator of OC quality and lability, where higher DOC:TDN ratios suggest nutrient limitation for DOC degradation, and less ideal conditions for methanogenesis. While a negative correlation between CH_4 concentrations and DOC:TDN was observed in 2022, the relationship was nonlinear and was only marginally significant in 2021. Elsewhere, methanogenesis was elevated with high availability of DOC (Polishchuk et al., 2018; Zhou et al., 2023), and labile carbon (Bertolet et al., 2020; Deemer & Holgerson, 2021; Mendoza-Pascual et al., 2021).

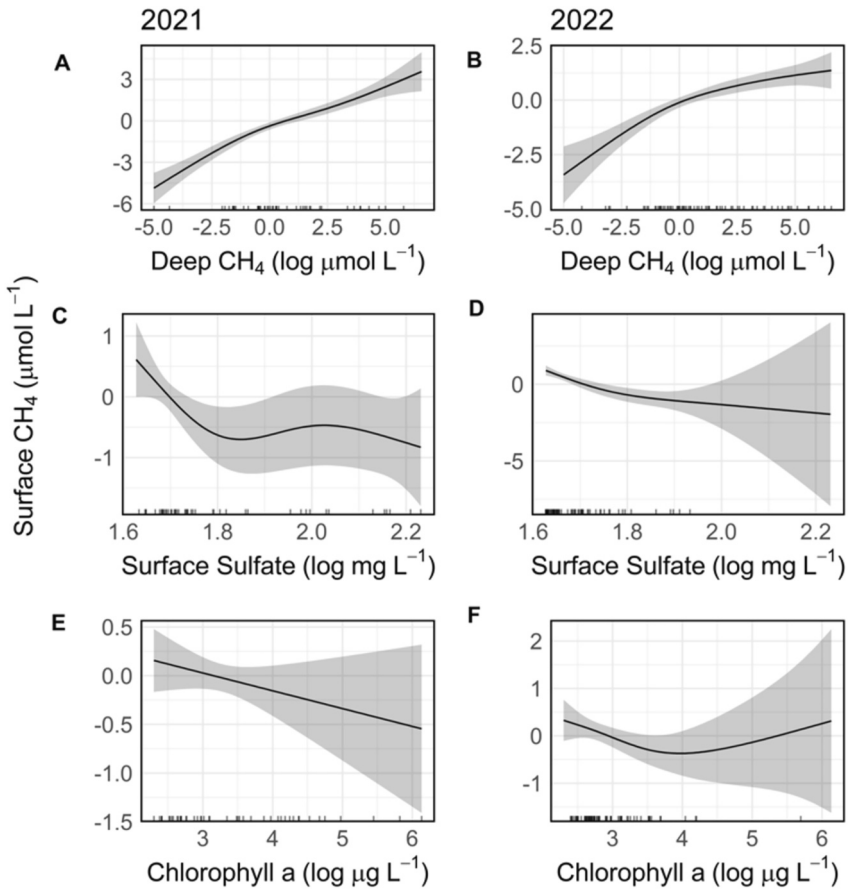

Figure 5. Partial effects plots from the generalized additive model relating surface CH_4 in 2021 (left) and 2022 (right) to water column conditions. Overall deviance explained = 79.7%. Gray shaded region represents the 95% Bayesian credible interval. Ticks along the *x*-axis represent the measured data points. Covariates included deep CH_4 concentration (a, b), sulfate concentration (c, d), and chlorophyll *a* concentrations (e, f).

Interestingly, despite many observations globally that warmer waters increase rates of methanogenesis (DelSontro et al., 2016; Fan et al., 2022; Mendoza-Pascual et al., 2021; Yvon-Durocher et al., 2014), temperature was not a significant predictor variable of CH_4 levels in either year of observation. Other studies have similarly found stronger correlations of methane emissions with water body productivity and temperature exerting less control over broad geographic scales (Deemer et al., 2016; Deemer & Holgerson, 2021; DelSontro et al., 2016; Webb et al., 2023). We may not have observed a strong relationship with temperature owing to the limited biological relevance of the temperature differences among years in deep water (14.5°C vs. 16.8°C).

Overall, the combination of a more stable water column, reduced hypolimnetic oxygen, and lower sulfate in 2022 resulted in a more than doubled CH_4 concentration in deep reservoir waters. In general, methane concentrations increased during the summer months (Figure 3d) consistent with several previous studies that found that stratified conditions accumulate CH_4 below the oxycline (Bastviken et al., 2008; Li, Huang, et al., 2020; Li, Peng, et al., 2020; Mendoza-Pascual et al., 2021; Ray et al., 2023; Thottathil et al., 2019). Stratification is integral to hypolimnion formation and promotion of CH_4 production and storage (Cadieux et al., 2015; Denfeld et al., 2020; Montes-Pérez et al., 2022), therefore increased water column stability in the wet year favoured isolation of the hypolimnion and depletion of dissolved oxygen. At present, we cannot determine to which degree accumulation of CH_4 during the summer, versus greatly elevated production in the warmest month, resulted in maximal deepwater concentrations in August.

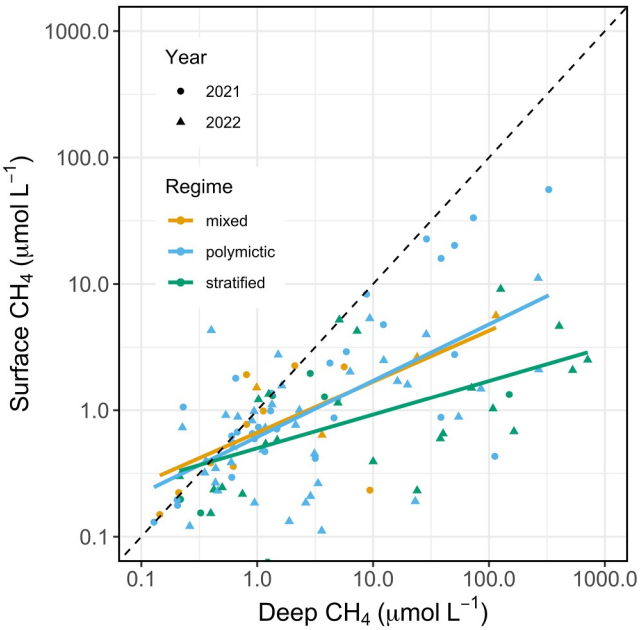

Figure 6. Surface versus deep CH_4 concentrations in agricultural reservoirs in 2021 and 2022. Sites are grouped as continuously mixed (orange points, no stratification over the summer months), polymictic (blue points, temporary stratification), or stratified (green points, water column stratified over the summer months). Major axis regression lines are added to for each mixing regime type. All regression lines were significant at $p < 0.05$. The 1:1 line is depicted by the dashed black line.

4.3. Drivers of Surface CH_4 Concentrations

Although deepwater CH_4 was greater in 2022 and was a significant predictor of surface CH_4 concentrations overall, we did not see differences in surface water CH_4 among years (see also Li, Huang, et al., 2020; Li, Peng, et al., 2020). One reason for this observation was that the relationship between surface and deep CH_4 concentrations has shallow slopes, and was strongest when the reservoirs were mixed or polymictic (Figure 6). These results could suggest that CH_4 is accumulated and trapped under thermocline depths (Breton et al., 2009; Hounshell et al., 2023; Ray et al., 2023) and that stratification can reduce methane concentrations at the surface, reducing emissions (Vachon et al., 2019). Previous studies have observed that high production of CH_4 at depth does not necessarily translate to elevated emission rates as the accumulated CH_4 can be oxidized at rates ranging from 39% to 99% during stratification (Bastviken et al., 2008; Vizza et al., 2022; West et al., 2016), and never reach the atmosphere (Bastviken et al., 2008; Reeburgh et al., 1993). Our calculated CH_4 diffusive flux rates support this observation, with no significant difference in fluxes among years (Table 1).

4.4. Implications for Annual Diffusive CH_4 Fluxes

Although we did not observe elevated surface CH_4 concentrations or diffusive fluxes in the hydrologically wet year, it is not clear whether the accumulated CH_4 in the hypolimnion could be released during mixing or fall turnover. During the fall, recirculation events disrupt stratification and cause potential partial or complete loss of the stored CH_4. In large reservoirs, a fraction of CH_4 is oxidized or is released into the atmosphere (Mayr et al., 2020; Montes-Pérez et al., 2022; Vachon et al., 2019). With highly oxygenated surface waters, turnover can cause large oxidation events (Kankaala et al., 2006). The rate of turnover is important as indicated by Vachon et al. (2019), who determined whether stored CH_4 is partially (quick turnover) or completely (gradual turnover) oxidized. Previous work has indicated that 15%–46% (Encinas Fernández et al., 2016; Vachon et al., 2019) of this CH_4 can be lost to the atmosphere. Our study systems, however, are polymictic (Holgerson et al., 2022), suggesting that these emissions may not be emitted as a single pulse during the fall, but that the intermittent mixing may periodically release CH_4 as has been seen previously in reservoirs (Montes-Pérez et al., 2022) and lakes (Davidson et al., 2024; Mendoza-Pascual et al., 2021).

4.5. Limits to Study Design

Our results were based on observations at the deepest part of the agricultural reservoir, and did not include estimates of spatial or diel variability within the reservoirs, littoral CH_4 emissions, or ebullition pathways. While epilimnetic sediments have been demonstrated to contribute directly to CH_4 emissions (Bastviken et al., 2008), our study sites are constructed reservoirs that are dredged with steep sides to minimize shallow littoral zones (Government of Alberta, 2015), and so are unlikely to contribute to large spatial differences in methanogenesis. Additionally, we did not consider other pathways of CH_4 release, including ebullition and plant-mediated flux. It is possible that the hypolimnetic CH_4 accumulation observed here may also be correlated with CH_4 bubble production in the sediments. Davidson et al. (2024) observed an order of magnitude increase in CH_4 ebullition during the short periods of stratification in a small pond. Other research has found that CH_4 ebullition rates in these ponds is similarly correlated with sulfate concentrations (Soued et al., 2024), but the role of water depth in emissions rates is not well established here. Future work will need to evaluate how the effects of depth affect the interaction of methanogenesis and bubble release through changes in hydrostatic pressure.

5. Conclusions

Climate change is anticipated to induce warming and affect runoff regimes that can alter CH_4 cycling in freshwater lentic systems. In deep systems, warming can induce longer stratification periods as surface waters become warmer, causing prolonged periods of methanogenesis. Here, we found that varying spring runoff patterns had a

strong effect on waterbody depth and stratification, impacting deep CH_4 concentrations, but this did not translate into elevated surface CH_4, likely due to offsetting mechanisms of increased CH_4 storage in the hypolimnion and increased oxidation through the deeper water column. This is the first study to demonstrate such an effect on small artificial agricultural waterbodies, which are model freshwater systems characterized by heightened climatic and anthropogenic forcing. Extreme water level changes are commonly observed in this region of NGP, particularly in closed-basin systems. Historical climate analyses indicate a transition to warmer temperatures, increasing summer precipitation, and decreasing winter precipitation leading to increased evaporation and susceptibility to droughts (Dumanski et al., 2015; Gushulak et al., 2024; Pham et al., 2009; Tanzeeba & Gan, 2012). Thus, we anticipate general trends of decreasing water levels in this region, but that extreme precipitation events may cause dramatic and flashy systems in future. We anticipate that the reservoirs in the NGP will likely experience lower water levels, elevated SO_4^{2-} concentrations, and reduced methanogenesis due to projected reductions in winter precipitation. However, our study demonstrated that runoff-driven water level changes may not translate to changes in surface diffusive emissions due to deep CH_4 being oxidized in the water column.

Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

Data Availability Statement

Data and R code are available at https://doi.org/10.5281/zenodo.17344880 (Rimas & Finlay, 2026).

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References

Abram, J. W., & Nedwell, D. B. (1978). Hydrogen as a substrate for methanogenesis and sulphate reduction in anaerobic saltmarsh sediment. *Archives of Microbiology*, *117*(1), 93–97. https://doi.org/10.1007/BF00689357

Baron, A. A. P., Dyck, L. T., Amjad, H., Bragg, J., Kroft, E., Newson, J., et al. (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

Bastviken, D., Cole, J. J., Pace, M. L., & Van de Bogert, M. C. (2008). Fates of methane from different lake habitats: Connecting whole-lake budgets and CH4 emissions. *Journal of Geophysical Research*, *113*(G2), G02024. https://doi.org/10.1029/2007JG000608

Berberich, M. E., Beaulieu, J. J., Hamilton, T. L., Waldo, S., & Buffam, I. (2020). Spatial variability of sediment methane production and methanogen communities within a eutrophic reservoir: Importance of organic matter source and quantity. *Limnology & Oceanography*, *65*(6), 1336–1358. https://doi.org/10.1002/lno.11392

Bertolet, B. L., Olson, C. R., Szydlowski, D. K., Solomon, C. T., & Jones, S. E. (2020). Methane and primary productivity in lakes: Divergence of temporal and spatial relationships. *Journal of Geophysical Research: Biogeosciences*, *125*(9), e2020JG005864. https://doi.org/10.1029/2020JG005864

Bjoerndahl, J. A., Gushulak, C. A. C., Mezzini, S., Simpson, G. L., Haig, H. A., Leavitt, P. R., & Finlay, K. (2022). Abrupt changes in the physical and biological structure of endorheic upland lakes due to 8-m lake-level variation during the 20th century. *Limnology & Oceanography*. https://doi.org/10.1002/lno.12054

Bonsal, B. R., Wheaton, E. E., Chipanshi, A. C., Lin, C., Sauchyn, D. J., & Wen, L. (2011). Drought research in Canada: A review. *Atmosphere-Ocean*, *49*(4), 303–319. https://doi.org/10.1080/07055900.2011.555103

Borrel, G., Adam, P. S., & Gribaldo, S. (2016). Methanogenesis and the wood–ljungdahl pathway: An ancient, versatile, and fragile association. *Genome Biology and Evolution*, *8*(6), 1706–1711. https://doi.org/10.1093/gbe/evw114

Breton, J., Vallières, C., & Laurion, I. (2009). Limnological properties of permafrost thaw ponds in northeastern Canada. *Canadian Journal of Fisheries and Aquatic Sciences*, *66*(10), 1635–1648. https://doi.org/10.1139/F09-108

Cadieux, S. B., White, J. R., & Pratt, L. M. (2015). The effect of warm summer 2012 on seasonal and annual methane dynamics in adjacent small lakes on the ice-free margin of Greenland. In *2015 AGU fall meeting*. AGU.

Cole, J. J., & Caraco, N. F. (1998). Atmospheric exchange of carbon dioxide in a low-wind oligotrophic lake. *Limnology & Oceanography*, *43*(4), 647–656. https://doi.org/10.4319/lo.1998.43.4.0647

D'Ambrosio, S. L., Henderson, S. M., Nielson, J. R., & Harrison, J. A. (2022). In situ flux estimates reveal large variations in methane flux across the bottom boundary layer of a eutrophic lake. *Limnology & Oceanography*, *67*(10), 2119–2139. https://doi.org/10.1002/lno.12102

Davidson, T. A., Søndergaard, M., Audet, J., Levi, E., Esposito, C., Bucak, T., & Nielsen, A. (2024). Temporary stratification promotes large greenhouse gas emissions in a shallow eutrophic lake. *Biogeosciences*, *21*(1), 93–107. https://doi.org/10.5194/bg-21-93-2024

Deemer, B. A., Harrison, J. A., Li, S., Beaulieu, J. J., DelSontro, T., Barros, N., et al. (2016). Greenhouse gas emissions from Reservoir water surfaces: A new global synthesis. *BioScience*, *66*(11), 949–964. https://doi.org/10.1093/biosci/biw117

Deemer, B. R., & Holgerson, M. A. (2021). Drivers of methane flux differ between lakes and reservoirs, complicating global upscaling efforts. *Journal of Geophysical Research: Biogeosciences*, *126*(4), e2019JG005600. https://doi.org/10.1029/2019JG005600

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 & Oceanography*, *61*(S1), S62–S77. https://doi.org/10.1002/lno.10313

Denfeld, B. A., Lupon, A., Sponseller, R. A., Laudon, H., & Karlsson, J. (2020). Heterogeneous CO2 and CH4 patterns across space and time in a small boreal lake. *Inland Waters*, *10*(3), 348–359. https://doi.org/10.1080/20442041.2020.1731789

Duc, N. T., Crill, P., & Bastviken, D. (2010). Implications of temperature and sediment characteristics on methane formation and oxidation in lake sediments. *Biogeochemistry*, *100*(1–3), 185–196. https://doi.org/10.1007/s10533-010-9415-8

Dumanski, S., Pomeroy, J. W., & Westbrook, C. J. (2015). Hydrological regime changes in a Canadian Prairie basin. *Hydrological Processes*, *29*(18), 3893–3904. https://doi.org/10.1002/hyp.10567

Encinas Fernández, J., Peeters, F., & Hofmann, H. (2016). On the methane paradox: Transport from shallow water zones rather than in situ methanogenesis is the major source of CH4 in the open surface water of lakes. *Journal of Geophysical Research: Biogeosciences*, *121*(10), 2717–2726. https://doi.org/10.1002/2016JG003586

Environment and Climate Change Canada (ECCC). (2023). *Canadian climate normals 1981-2020 station data*. Moose Jaw, Saskatchewan. Retrieved from https://climate.weather.gc.ca/climate_normals/results_1981_2010_e.html?searchType=stnName&txtStationName=moose&searchMethod=contains&txtCentralLatMin=0&txtCentralLatSec=0&txtCentralLongMin=0&txtCentralLongSec=0&stnID=2967&dispBack=0

Fan, M., Zhang, W., Wu, J., & Zhou, J. (2022). Agricultural land use and pond management influence spatial-temporal variation of CH4 and N2O emission fluxes in ponds in a subtropical agricultural headstream watershed. *Frontiers in Environmental Science*, *10*, 2132. https://doi.org/10.3389/fenvs.2022.850070

Feron, S., Malhotra, A., Bansal, S., Fluet-Chouinard, E., McNicol, G., Knox, S. H., et al. (2024). Recent increases in annual, seasonal, and extreme methane fluxes driven by changes in climate and vegetation in boreal and temperate wetland ecosystems. *Global Change Biology*, *30*(1), e17131. https://doi.org/10.1111/gcb.17131

Finlay, K., Leavitt, P. R., Wissel, B., & Prairie, Y. T. (2009). Regulation of spatial and temporal variability of carbon flux in six hard-water lakes of the northern Great Plains. *Limnology & Oceanography*, *54*(6part2), 2553–2564. https://doi.org/10.4319/lo.2009.54.6_part_2.2553

Gilbert, P. J., Taylor, S., Cooke, D. A., Deary, M. E., & Jeffries, M. J. (2021). Quantifying organic carbon storage in temperate pond sediments. *Journal of Environmental Management*, *280*, 111698. https://doi.org/10.1016/j.jenvman.2020.111698

Government of Alberta. (2015). Quality farm dugouts (3rd ed.). Retrieved from https://open.alberta.ca/dataset/a55d220d-b8c7-405d-90b4-e216b7fa1776/resource/770e7737-9c69-455c-98f5-56c5cccfe589/download/716-b01.pdf

Gushulak, C. A. C., Chegoonian, A. M., Wolfe, J., Gray, K., Mezzini, S., Wissel, B., et al. (2024). Impacts of hydrologic management on the eutrophication of shallow lakes in an intensive agricultural landscape (Saskatchewan, Canada). *Freshwater Biology*, *69*(7), 984–1000. https://doi.org/10.1111/fwb.14260

Haig, H. A., Hayes, N. M., Simpson, G. L., Yi, Y., Wissel, B., Hodder, K. R., & Leavitt, P. R. (2021). Effects of seasonal and interannual variability in water isotopes (δ2H, δ18O) on estimates of water balance in a chain of seven prairie lakes. *Journal of Hydrology X*, *10*, 100069. https://doi.org/10.1016/j.hydroa.2020.100069

Hammer, T. U. (1986). Saline Lake resources of the Canadian prairies. Can. *Water Resources Journal*, *11*(1), 43–57. https://doi.org/10.4296/cwrj1101043

Harrison, J. A., Deemer, B. R., Birchfield, M. K., & O'Malley, M. T. (2017). Reservoir water-level drawdowns accelerate and amplify methane emission. *Environmental Science & Technology*, *51*(3), 1267–1277. https://doi.org/10.1021/acs.est.6b03185

Ho, L., Jerves-Cobo, R., Morales, O., Larriva, J., Arevalo-Durazno, M., Barthel, M., et al. (2021). Spatial and temporal variations of greenhouse gas emissions from a waste stabilization pond: Effects of sludge distribution and accumulation. *Water Research*, *193*, 116858. https://doi.org/10.1016/j.watres.2021.116858

Holgerson, M. A. (2015). Drivers of carbon dioxide and methane supersaturation in small, temporary ponds. *Biogeochemistry*, *124*(1–3), 305–318. https://doi.org/10.1007/s10533-015-0099-y

Holgerson, M. A., & Raymond, P. (2016). Large contribution to inland water CO2 and CH4 emissions from very small ponds. *Nature Geoscience*, *9*(3), 222–226. https://doi.org/10.1038/ngeo2654

Holgerson, M. A., Richardson, D. C., Roith, J., Bortolotti, L. E., Finlay, K., Hornbach, D. J., et al. (2022). Classifying mixing regimes in ponds and shallow lakes. *Water Resources Research*, *58*(7), e2022WR032522. https://doi.org/10.1029/2022wr032522

Holgerson, M. A., Zappa, C. J., & Raymond, P. A. (2017). Gas transfer velocities in small forested ponds. *Journal of Geophysical Research: Biogeosciences*, *122*(5), 1011–1021. https://doi.org/10.1002/2016JG003734

Hounshell, A. G., D'Acunha, B. M., Breef-Pilz, A., Johnson, M. S., Thomas, R. Q., & Carey, C. C. (2023). Eddy covariance data reveal that a small freshwater reservoir emits a substantial amount of carbon dioxide and methane. *Journal of Geophysical Research: Biogeosciences*, *128*(3), e2022JG007091. https://doi.org/10.1029/2022jg007091

Hounshell, A. G., McClure, R. P., Lofton, M. E., & Carey, C. C. (2021). Whole-ecosystem oxygenation experiments reveal substantially greater hypolimnetic methane concentrations in reservoirs during anoxia. *Limnology and Oceanography Letters*, *6*(1), 33–42. https://doi.org/10.1002/lol2.10173

Jeffrey, S. W., & Humphrey, G. F. (1975). New spectrophotometric equations for determining chlorophylls a, b, c1 and c2 in higher plants, algae and natural phytoplankton. *Biochemie und Physiologie der Pflanzen*, *167*(2), 191–194. https://doi.org/10.1016/S0015-3796(17)30778-3

Jenny, J.-P., Francus, P., Normandeau, A., Lapointe, F., Perga, M.-E., Ojala, A., et al. (2015). Global spread of hypoxia in freshwater ecosystems during the last three centuries is caused by rising local human pressure. *Global Change Biology*, *22*(4), 1481–1489. https://doi.org/10.1111/gcb.13193

Jensen, S. A., Webb, J. W., Simpson, G. L., Baulch, H. M., Leavitt, P. R., & Finlay, K. (2022). Seasonal and diel patterns of greenhouse gas content and fluxes in small agricultural reservoirs of the northern Great Plains. *Frontiers in Environmental Science*, *10*, 895531. https://doi.org/10.3389/fenvs.2022.895531

Jensen, S. A., Webb, J. W., Simpson, G. L., Baulch, H. M., Leavitt, P. R., & Finlay, K. (2023). Carbon dioxide, methane, and nitrous oxide emissions in natural and constructed waterbodies on the northern Great Plains. *Journal of Geophysical Research: Biogeosciences*, *128*(4), e2022JG007261. https://doi.org/10.1029/2022JG007261

Kankaala, P., Huotari, J., Peltomaa, E., Saloranta, T., & Ojala, A. (2006). Methanotrophic activity in relation to methane efflux and total heterotrophic bacterial production in a stratified, humic, boreal lake. *Limnology & Oceanography*, *51*, 1195–1204. https://doi.org/10.4319/lo.2006.51.2.119

Kassambara, A. (2023). rstatix: Pipe-friendly framework for basic statistical tests. *R package version 0.7.2*. Retrieved from https://rpkgs.datanovia.com/rstatix/

Laird, K. R., Cumming, B. F., Wunsam, S., Rusak, J. A., Oglesby, R. J., Fritz, S. C., & Leavitt, P. R. (2003). Lake sediments record large-scale shifts in moisture regimes across the northern prairies of North America during the past two millennia. *Proceedings of the National Academy of Sciences*, *100*(5), 2483–2488. https://doi.org/10.1073/pnas.0530193100

Li, M., Peng, C., Zhu, Q., Zhou, X., Yang, G., Song, X., & Zhang, K. (2020). The significant contribution of lake depth in regulating global lake diffusive methane emissions. *Water Research*, *172*, 115465. https://doi.org/10.1016/j.watres.2020.115465

Li, N., Huang, T., Li, Y., Si, F., Zhang, H., & Wen, G. (2020). Inducing an extended naturally complete mixing period in a stratified reservoir via artificial destratification. *Science of the Total Environment*, *745*, 140958. https://doi.org/10.1016/j.scitotenv.2020.140958

Malerba, M. E., de Kluyver, T., Wright, N., Schuster, L., & Macreadie, P. I. (2022). Methane emissions from agricultural ponds are underestimated in national greenhouse gas inventories. *Communications Earth & Environment*, *3*, 306. https://doi.org/10.1038/s43247-022-00483-1

Malyan, S. K., Singh, O., Kumar, A., Anand, G., Singh, R., Singh, S., et al. (2022). Greenhouse gases trade-off from ponds: An overview of emission process and their driving factors. *Water*, *14*(6), 970. https://doi.org/10.3390/w14060970

Marra, G., & Wood, S. N. (2011). Practical variable selection for generalized additive models. *Computational Statistics & Data Analysis*, *55*(7), 2372–2387. https://doi.org/10.1016/j.csda.2011.02.004

Martins, P. D., Hoyt, D. W., Bansal, S., Mills, C. T., Tfaily, M. M., Tangen, B. A., et al. (2017). Abundant carbon substrates drive extremely high sulfate reduction rates and methane fluxes in Prairie Pothole Wetlands. *Global Change Biology*, *23*(8), 3107–3120. https://doi.org/10.1111/gcb.13633

Mayr, M. J., Zimmermann, M., Dey, J., Brand, A., Wehrli, B., & Bürgmann, H. (2020). Growth and rapid succession of methanotrophs effectively limit methane release during lake overturn. *Communications Biology*, *3*(1), 108. https://doi.org/10.1038/s42003-020-0838-z

Mendonça, R., Müller, R. A., Clow, D., Verpoorter, C., Raymond, P., Tranvik, L. J., et al. (2017). Organic carbon burial in global lakes and reservoirs. *Nature Communications*, *8*, 1694. https://doi.org/10.1038/s41467-017-01789-6

Mendoza-Pascual, M. U., Itoh, M., Aguilar, J. I., Padilla, K. S. A. R., Papa, R. D. S., & Okuda, N. (2021). Controlling factors of methane in tropical lakes of different depths. *Journal of Geophysical Research: Biogeosciences*, *126*(4), e2020JG005828. https://doi.org/10.1029/2020JG005828

Monteith, D. T., Evans, C. D., Henrys, P. A., Simpson, G. L., & Malcolm, I. A. (2014). Trends in the hydrochemistry of acid-sensitive surface waters in the UK 1988–2008. *Ecological Indicators*, *37*, 287–303. https://doi.org/10.1016/j.ecolind.2013.10.026

Montes-Pérez, J. J., Marcé, R., Obrador, B., Conejo-Orosa, T., Díez, J. L., Escot, C., et al. (2022). Hydrology influences carbon flux through metabolic pathways in the hypolimnion of a Mediterranean reservoir. *Aquatic Sciences*, *84*, 36. https://doi.org/10.1007/s00027-022-00855-0

Niu, X., Yan, Z., Wu, W., Hua, Z., Hao, L., & Rosentreter, J. A. (2025). Rainfall can significantly reduce pond methane emissions by depressing ebullition. *Water Research*, *285*, 124167. https://doi.org/10.1016/j.watres.2025.124167

Orr, H. G., Simpson, G. L., des Clers, S., Watts, G., Hughes, M., Hannaford, J., et al. (2015). Detecting changing river temperatures in England and Wales. *Hydrological Processes*, *29*(5), 752–766. https://doi.org/10.1002/hyp.10181

Peacock, M., Audet, J., Bastviken, D., Cook, S., Evans, C. D., Grinham, A., et al. (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

Peeters, F., Encinas Fernandez, J., & Hofmann, H. (2019). Sediment fluxes rather than oxic methanogenesis explain diffusive CH4 emissions from lakes and reservoirs. *Scientific Reports*, *9*(1), 243. https://doi.org/10.1038/s41598-018-36530-w

Pham, S. V., Leavitt, P. R., McGowan, S., Wissel, B., & Wassenaar, L. I. (2009). Spatial and temporal variability of prairie lake hydrology as revealed using stable isotopes of hydrogen and oxygen. *Limnology & Oceanography*, *54*(1), 101–118. https://doi.org/10.4319/lo.2009.54.1.0101

Polishchuk, Y., Bogdanov, A. N., Muratov, I. N., Polishchuk, V., Lim, A., Manasypov, R. M., et al. (2018). Minor contribution of small thaw ponds to the pools of carbon and methane in the inland waters of the permafrost-affected part of the Western Siberian Lowland. *Environmental Research Letters*, *13*(4), 045002. https://doi.org/10.1088/1748-9326/aab046

Pomeroy, J. W., Gray, D. M., Brown, T., Hedstrom, N. R., Quinton, W. L., Granger, R. J., & Carey, S. K. (2007). The cold regions hydrological model: A platform for basing process representation and model structure on physical evidence. *Hydrological Processes*, *21*(19), 2650–2667. https://doi.org/10.1002/hyp.6787

Prairie, Y. T., de Montigny, C., & Del Giorgio, P. A. (2001). Anaerobic phosphorus release from sediments: A paradigm revisited. *SIL Proceedings, 1922-2010*, *27*(7), 4013–4020. https://doi.org/10.1080/03680770.1998.11901750

Ray, N. E., Holgerson, M. A., Andersen, M. R., Bikše, J., Bortolotti, L. E., Futter, M., et al. (2023). Spatial and temporal variability in summertime dissolved carbon dioxide and methane in temperate ponds and shallow lakes. *Limnology & Oceanography*, *68*(7), 1530–1545. https://doi.org/10.1002/lno.12362

R Core Team. (2021). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing. Retrieved from https://www.R-project.org/

Read, J. S., Winslow, L. A., Hansen, G. J. A., Van Den Hoek, J., Hanson, P. C., Bruce, L. C., & Markfort, C. D. (2012). rLakeAnalyzer: Package for the analysis of lake physics. *R package version*. https://doi.org/10.32614/CRAN.package

Reeburgh, W. S., Whalen, S. C., & Alperin, M. J. (1993). The role of methylotrophy in the global methane budget. In *Microbial growth on C1 compounds* (pp. 1–14).

Rimas, R. I., & Finlay, K. (2026). Hydrologic variability drives differential methane dynamics in agricultural reservoirs of the northern Great Plains [Dataset and Software]. *Zenodo*. https://doi.org/10.5281/zenodo.17344880

Rosentreter, J. A., Borges, A. V., Deemer, B. R., Holgerson, M. A., Liu, S., Song, C., et al. (2021). Half of global methane emissions come from highly variable aquatic ecosystem sources. *Nature Geoscience*, *14*(4), 225–230. https://doi.org/10.1038/s41561-021-00715-2

Sanches, L. F., Guenet, B., Marinho, C. C., Barros, N., & de Assis Esteves, F. (2019). Global regulation of methane emission from natural lakes. *Scientific Reports*, *9*(1), 255. https://doi.org/10.1038/s41598-018-36519-5

Sauchyn, D., & Kerr, S. (2016). Canadian prairies drought from a paleoclimate perspective. *Vulnerability and Adaptation to Drought*, *39*.

Sobek, S., DelSontro, T., Wongfun, N., & Wehrli, B. (2012). Extreme organic carbon burial fuels intense methane bubbling in a temperate reservoir. *Geophysical Research Letters*, *39*(1). https://doi.org/10.1029/2011GL050144

Soued, C., Finlay, K., Bortolotti, L. E., Jensen, S., Leavitt, P. R., Mueller, P., et al. (2024). Salinity causes widespread restriction of methane emissions from inland waters. *Nature Communications*, *15*(1), 717. https://doi.org/10.1038/s41467-024-44715-3

Spence, C., He, Z., Shook, K. R., Pomeroy, J. W., Whitfield, C. J., & Wolfe, J. D. (2022). Assessing runoff sensitivity of North American Prairie Pothole Region basins to wetland drainage using a basin classification-based virtual modeling approach. *Hydrology and Earth System Sciences*, *26*(21), 5555–5575. https://doi.org/10.5194/hess-26-5555-2022

Stewart, R. E., & Kantrud, H. A. (1971). *Classification of natural ponds and lakes in the glaciated prairie region*. Resource Publication #92. US Department of Interior, Fish and Wildlife Service-Bureau of Sport Fisheries and Wildlife.

St-Jacques, J.-M., Andreichuk, Y., Sauchyn, D. J., & Barrow, E. (2018). Projecting Canadian prairie runoff for 2041-2070 with North American Regional Climate Change Assessment Program (NARCCAP) data. *JAWRA Journal of the American Water Resources Association*, *54*(3), 660–675. https://doi.org/10.1111/1752-1688.12642

Swarbrick, V. J., Bergbusch, N. T., & Leavitt, P. R. (2022). Spatial and temporal patterns of urea content in a eutrophic stream continuum on the Northern Great Plains. *Biogeochemistry*, *157*(2), 171–191. https://doi.org/10.1007/s10533-021-00868-7

Tanzeeba, S., & Gan, T. Y. (2012). Potential impact of climate change on the water availability of South Saskatchewan River Basin. *Climatic Change*, *112*(2), 355–386. https://doi.org/10.1007/s10584-011-0221-7

Thomas, J. D., & Ratcliffe, P. J. (1973). Observations on the limnology and primary production of a small man-made lake in the West African savanna. *Freshwater Biology*, *3*(6), 573–612. https://doi.org/10.1111/j.1365-2427.1973.tb00079.x

Thottathil, S. D., Reis, P. C., & Prairie, Y. T. (2019). Methane oxidation kinetics in northern freshwater lakes. *Biogeochemistry*, *143*(1), 105–116. https://doi.org/10.1007/s10533-019-00552-x

Timmers, P. H. A., Suarez-Zuluaga, D. A., van Rossem, M., Diender, M., Stams, A. J. M., & Plugge, C. M. (2016). Anaerobic oxidation of methane associated with sulfate reduction in a natural freshwater gas source. *The ISME Journal*, *10*(6), 1400–1412. https://doi.org/10.1038/ismej.2015.213

Vachon, D., Langenegger, T., Donis, D., & McGinnis, D. F. (2019). Influence of water column stratification and mixing patterns on the fate of methane produced in deep sediments of a small eutrophic lake. *Limnology & Oceanography*, *64*(5), 2114–2128. https://doi.org/10.1002/lno.11172

Vachon, D., & Prairie, Y. T. (2013). The ecosystem size and shape dependence of gas transfer velocity versus wind speed relationships in lakes. *Canadian Journal of Fisheries and Aquatic Sciences*, *70*(12), 1757–1766. https://doi.org/10.1139/cjfas-2013-0241

van der Kamp, G., Keir, D., & Evans, M. S. (2008). Long-term water level changes in closed-basin lakes of the Canadian Prairies. *Canadian Water Resources Journal*, *33*(1), 23–38. https://doi.org/10.4296/cwrj3301023

Vizza, C., Jones, S. E., Hart, J. A., West, W. E., & Lamberti, G. A. (2022). Pond methane dynamics, from microbial communities to ecosystem budget, during summer in Alaska. *Limnology & Oceanography*, *67*(2), 450–467. https://doi.org/10.1002/lno.12003

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

Webb, J. R., Leavitt, P. R., Simpson, G. L., Baulch, H. M., Haig, H. A., Hodder, K. R., & Finlay, K. (2019). Regulation of carbon dioxide and methane in small agricultural reservoirs: Optimizing potential for greenhouse gas uptake. *Biogeosciences*, *16*(21), 4211–4227. https://doi.org/10.5194/bg-16-4211-2019

Webb, J. R., Quayle, W. C., Ballester, C., & Wells, N. S. (2023). Semi-arid irrigation farm dams are a small source of greenhouse gas emissions. *Biogeochemistry*, *166*(2), 123–138. https://doi.org/10.1007/s10533-023-01100-4

West, W. E., Creamer, K. P., & Jones, S. E. (2016). Productivity and depth regulate lake contributions to atmospheric methane. *Limnology & Oceanography*, *61*(S1), S51–S61. https://doi.org/10.1002/lno.10247

Wood, S. N. (2011). Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. *Journal of the Royal Statistical Society: Series B*, *73*(1), 3–36. https://doi.org/10.1111/j.1467-9868.2010.00749.x

Wood, S. N. (2017). *Generalized additive models: An introduction with R* (2nd ed.). Chapman and Hall/CRC.

Wood, S. N., Pya, N., & Säfken, B. (2016). Smoothing parameter and model selection for general smooth models. *Journal of the American Statistical Association*, *111*(516), 1548–1563. https://doi.org/10.1080/01621459.2016.1180986

Yakimovich, K. M., Orland, C., Emilson, E. J., Tanentzap, A. J., Basiliko, N., & Mykytczuk, N. C. (2020). Lake characteristics influence how methanogens in littoral sediments respond to terrestrial litter inputs. *The ISME Journal*, *14*(8), 2153–2163. https://doi.org/10.1038/s41396-020-0680-9

Yvon-Durocher, G., Allen, A., Bastviken, D., Conrad, R., Gudasz, C., St-Pierre, A., et al. (2014). Methane fluxes show consistent temperature dependence across microbial to ecosystem scales. *Nature*, *507*(7493), 488–491. https://doi.org/10.1038/nature13164

Zhou, X. D., Kumar, A., Wang, H. Y., Knorr, K. H., Chen, Y. B., Liu, Y. X., et al. (2023). Greenhouse gas emissions hotspots and drivers of urban freshwater bodies in areas of the Yangtze River delta, China. *Ecohydrology*, *16*(2), e2498. https://doi.org/10.1002/eco.2498