Associate Editor Decision: Reconsider after major revisions (16 Sep 2019) by Ji-Hyung Park

Comments to the Author:

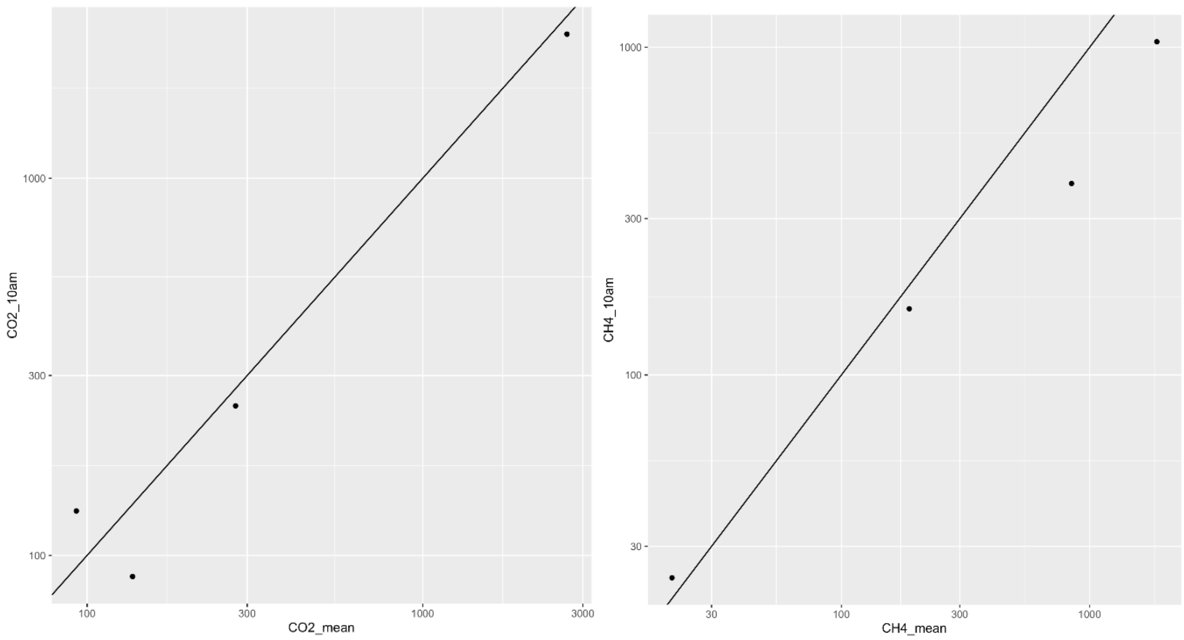
Thank you for providing detailed responses to the comments and suggestions offered by two reviewers.

Both reviewers recognized the scientific value and novelty of your manuscript, but the second reviewer also raised several critical issues. I agree that you need to pay more attention to uncertainties in estimating CO2 and CH4 fluxes when you evaluate the sink or source capacity of the studied reservoirs. I thought that the manuscript would require a substantial revision to address all the raised issues and a number of other comments, so I recommend 'reconsider after major revisions'.

Response: We thank the Associate Editor for these suggestions and consideration of this manuscript after revisions. We have addressed each comment below in further detail.

With regard to your assessment of net CO2 eq sink (19%), I would suggest that you consider uncertainties associated with CH4 loss via ebullition (as suggested by the second reviewer) and potential temporal (both diurnal and seasonal) variations in CO2 to provide ranges of estimates rather than one single estimate. You measured CO2 only in summer months, so you may have different (probably higher) values in other seasons due to changes in primary production. Please refer to other studies (or your own studies if you have) to estimate potential seasonal variations. You did not provide any detail about sampling frequency and time (Once per sampling, per season? Or repeated samplings to cover diurnal variation?). It would be quite misleading if you provide one single value out of uncertain estimates even though your sampling did not cover seasonal and diurnal variations.

Response: We followed your suggestion and assessed the range in CO_2-e sink capacity of our farm reservoirs based on preliminary seasonal data. Firstly, our diurnal data is greatly limited to only four sites, yet suggests far less variation than seasonal. The plots below show CO_2 and CH_4 concentrations at time = 0 (10 am) versus the average concentration over the 24 hr time series. Note that 4 sites were sampled over a 24-h period, and gases were collected every 6 hours (5 samples). We show that values collected at 10 am were not systematically higher or lower than the mean daily concentrations for a given site, suggesting low variability. This was likely due to the high alkalinity (a dominant characteristic for most farm reservoirs in this survey), buffering potentially large CO_2 fluctuations.



Therefore, the following paragraph has been added to the discussion.

"*On average, 8% of farm reservoirs were acting as CO_2-e sinks on the range of -0.6 to 79 g CO_2 m^{-2} d^{-1} during the time of sampling. This number offers a snapshot of the potential for farm reservoirs to act as a net CO_2-e sink and it is important to consider how seasonal variation influences the GHG sink/source status. Preliminary data on seasonal variation in CO_2 and CH_4 concentrations from a smaller number of farm reservoirs indicate variation (represented as the standard deviation related to the mean), ranging between 20 to 200% and 40 to 200% for CO_2 and CH_4, respectively. Here, this variation represents monthly sampling between the periods of ice melt and ice formation on lakes in Saskatchewan. Applying the average observed seasonal variation of 78% and 93% to our current spatial dataset suggests that CO_2-e emissions from farm reservoirs may vary between -1.7 and 150 g CO_2 m^{-2} d^{-1}, or 0 to 44% as acting net CO_2-e sinks. Further study into the consistency of potential farm reservoir CO_2 sinks on the temporal scale is required to better assess the overall GHG impact.*" Line 400

Because we have not undertaken any direct ebullition measurements, we feel that providing an assessment of uncertainties associated with ebullition is too speculative to apply to this quantitative dataset. Instead we highlight the importance of measuring this pathway to further inform management strategies and design:

"It is important to note that the CH_4 contribution to CO_2-e emissions is likely underestimated here as ebullition emissions were not measured. In farm reservoirs, ebullition flux can contribute >90% of total CH_4 emissions and is often highest in the smallest size classes (Grinham et al., 2018a). However, the sporadic nature of this pathway remains difficult to constrain for one single type of waterbody and may be a minor contributor in reservoirs and ponds > 3-5 m deep (Joyce and Jewell, 2003; DelSontro et al., 2016). This reinforces that design and management strategies that focus on reducing all pathways of CH_4 emissions will be most effective in curbing total CO_2-e emissions. Deeper farm dams with steep side slopes will likely be effective in reducing ebullition events due to a limited macrophytes, reduced bottom water temperature in summer, and supressed bubble release with higher water pressure (Joyce and Jewell, 2003; Natchimuthu et al., 2014; Grinham et al., 2018b)." Line 401

Finally, we have added additional information on the frequency and timing of sampling for this study:

"Each site was sampled once during this period, between the daylight hours of 10:00 to 15:00." Line 96

Considering the critical role of phytoplankton in reservoir CO2 budgets, you might also need to provide more descriptions and discussion on the relationship between Chl a and CO2. It appears that your model (and also your discussion) does not consider this important relationship. Please check and discuss any lack or hidden relationship between Chl a and CO2 to assess the role of phytoplankton as a CO2 sink, particularly in relation to nutrient levels in the studied reservoirs (for instance, in lines 198-199 you can provide more information about how CO2 varies with Chl a and nutrient levels). Your discussion on nutrient control over phytoplankton and CO2 levels (lines 262-270) focuses on the positive relationship between N and CO2. Please refer to other studies reporting various relationships between nutrients (both N and P) and phytoplankton uptake and release of CO2 (and CH4) to provide a more in-depth discussion of the observed patterns (your succinct data presentation does not allow readers to find out detailed information on this topic).

Response: We agree that autotrophic activity plays an active role in reservoir CO_2 budgets. The role of phytoplankton was initially tested using the parameter chlorophyll *a* (a measure of phytoplankton biomass) in the correlation tests for CO_2. Readers can find this presentation of the data in Supplementary Tables S1 and 2. It had a significant relationship with dissolved oxygen (DO), both representing the role of autotrophic activity. Because DO represented a more direct measure of primary productivity at the time of sampling, and was more significantly correlated with reservoir CO_2 concentration, this parameter of primary production was selected for the final model, rather than Chl *a*.

- Line 22: It is not clear which optimal design and management can minimize GHG impact. Please elaborate on the implication of your findings in the context of GHG emission mitigation. You stated "evaluating the potential for reservoir design to minimize CO2-equivalent (CO2-e) emissions (line 71) as a primary goal of your study. However, as you mentioned in the following sentence ("By identifying the driving characteristics of farm dams that support reduced C emissions, our findings provide the first step to developing management strategies to help minimize farm carbon emissions."), your results appear to provide some baseline information that could be useful in opting for emission mitigation strategies. Because this baseline information is not specific enough to suggest "the potential for reservoir design to minimize" GHG emissions, a more cautious wording would help readers grab some practical ramifications of your scientific findings.

Response: The sentence in the abstract on design and management has now been elaborated to read:

"From our models, we show that the GHG impact of farm reservoirs can be greatly minimised with overall improvements in water quality and consideration to position and hydrology within the land scape." Line 25

We have also revised the wording in the study goal sentence to read:

"Our aim was to identify the key environmental conditions regulating CO_2 and CH_4 fluxes, and to evaluate this baseline data in the context of emission mitigation strategies." Line 76

- Line 73 (& 178-):. Is this (GAMs) a new approach proposed in this study? Please clarify whether you propose this approach here for the first time or simply follow other studies (then cite relevant references)

Response: GAMs are a fairly standard modelling tool in ecology statistics (Pedersen et al., 2019. https://doi.org/10.7717/peerj.6876). We have added the following statement when introducing GAMs in methods:

"GAMs are not constrained by prescribed assumptions associated with parametric models such as linearity of link-scale effects in generalized linear models. Instead, the functional form of the partial relationships between covariates and the response are determined from the data. The more flexible modelling approach is useful where the effects of covariates on the response are non-linear and has been applied to complex aquatic datasets assessing GHGs (Wiik et al., 2018; Webb et al., 2019)." Line 197

- Line 17 "address and manage their potential importance" – Please specify what specific importance you want to "address and manage" (?).

Response: "…in agricultural GHG budgets" has been added to the sentence (Line 17).

- Line 22 "eutrophication-driven CH4": Do you mean "eutrophication-driven production of CH4"?

Response: Statement has been revised to read "…a positive association between eutrophication and CH_4 production".

- Lines 28-54: These two paragraphs may be reversed in order.

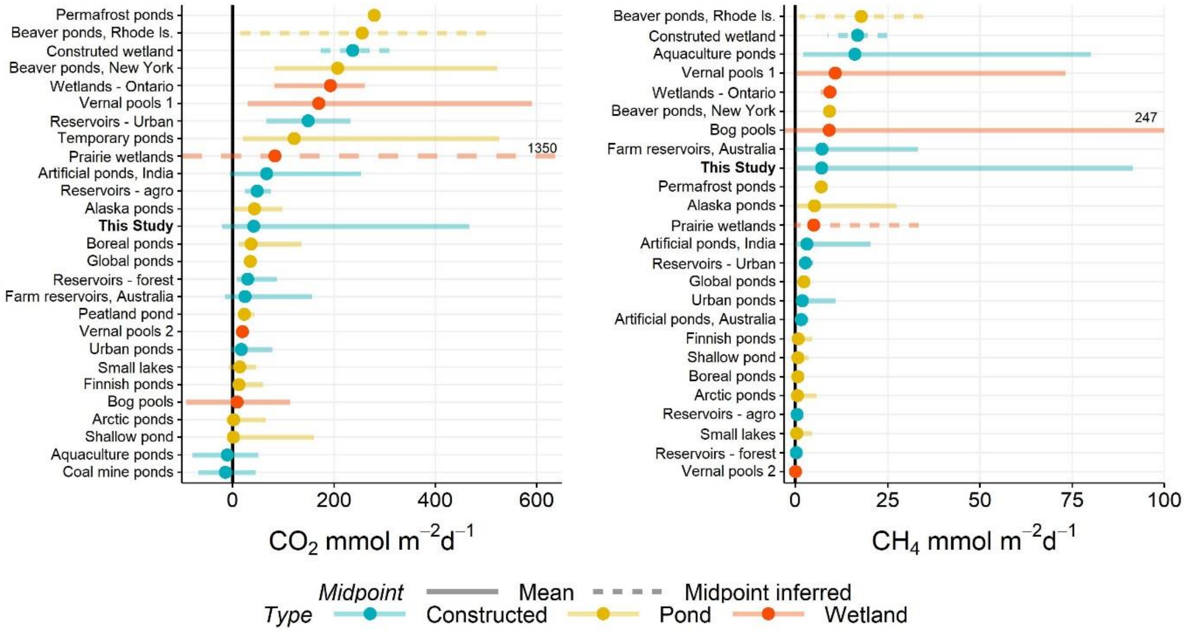
Response: Corrected.

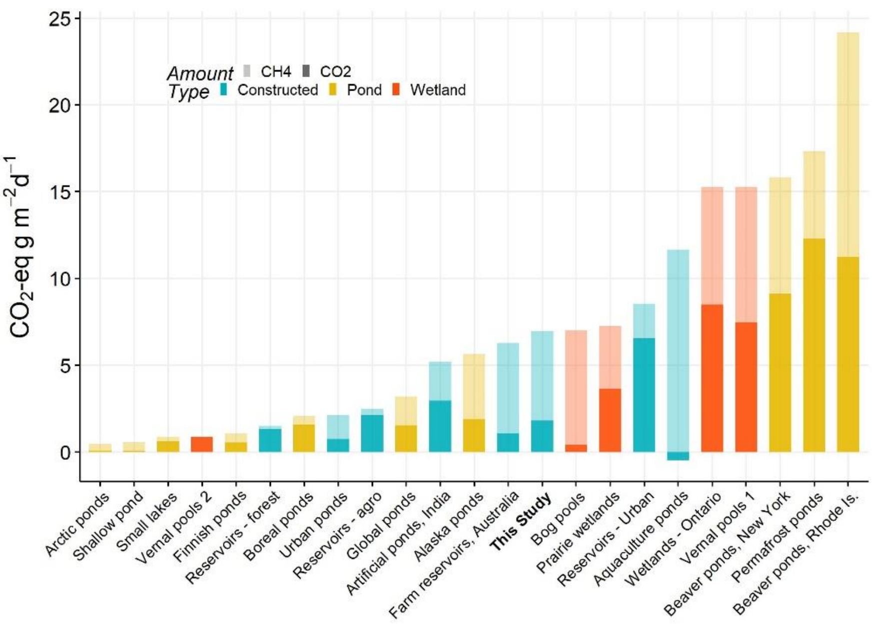
- Line 172 "NOx": Have you defined this earlier? Please note that NOx usually refers to nitrogen oxides in environmental science.

Response: We have added the following definition in parenthesis after first mention of NO_x here: "…(NO_2 + NO_3)", Line 191

- Figs. 4-5: Can you make the data of the label "Farm reservoirs (this study)" stand out by using some special symbol or color?

Response: We have now highlighted this study in bold on Figures 5 and 6:





End of Associate Editor response

Anonymous Referee #1

Received and published: 12 August 2019

The paper by Webb et al presents CH4 and CO2 data from 101 farm ponds. Alongside these GHG measurements are an impressive array of variables of water chemistry, hydrological characteristics, and landscape attributes. The authors investigate these variables as drivers of the GHG emissions. The paper is well written and I enjoyed reading it. It is within the scope of BG, and presents novel data insomuch as the fact that more pond GHG data is needed (and this point was explicitly raised in the recent IPCC refinement). If small, artificial waterbodies can be designed to minimise CH4 emissions, and to act as CO2 sinks, then this could lead to them acting as natural climate solutions.

Methods and analysis are well explained with sufficient detail, and the results support the conclusions. Presentation is good, language is fluent, abstract is suitable. The work is mostly well referenced (I suggest two older references of farm pond emissions that the authors may have missed). I particularly enjoyed reading the succinct and to-the-point results section, which was enough to get the authors' points over without endlessly writing numbers out, as so many results sections do. The one thing I find lacking from the paper is a visual presentation of the underlying CO2 and CH4 data, and in my comments I suggest a way to address this. I think it is important that readers are offered an easy way to understand the variation in the GHG data across all 101 waterbodies. I suggest the paper is acceptable following minor revisions. Below are my detailed comments.

Response: We thank the reviewer for their positive review and their constructed comments and suggestions offered. Detailed responses to the comments are addressed in blue font below.

L29. "Small waterbodies have recently been recognised as substantial contributors to global carbon emissions from inland waters." This is true, and missing from somewhere in the introduction (and discussion) is a mention that the recent 2019 IPCC Refinement explicitly addresses the issue of CH4 emissions from artificial ponds. The Refinement can be found at the link below, and the relevant chapter is in vol. 4 (AFOLU), chapter 7 (Wetlands). The emission factor given for artificial ponds is 183 kg CH4/ha/yr, but there is currently not enough data to disaggregate pond emissions by climate zone. How does your data compare to this emission factor? https://www.ipccnggip.iges.or.jp/public/2019rf/index.html

Response: We appreciate the reviewer raising awareness of the latest IPCC estimate. The following sentence has now been added to the introduction:

"*The recent 2019 IPCC Refinement has assigned a CH_4 emission factor of 183 kg ha^{-1} yr^{-1} to constructed waterbodies, however data is greatly limited, both geographically and in number (n = 68), that climatic-zone emission factors cannot be estimated (IPCC, 2019).*" Line 61

We also now compare our average farm dam CH_4 emission with the IPCC estimate in the discussion:

"Average CH_4 fluxes from our farm reservoirs correspond to 417 kg CH_4 ha^{-1} yr^{-1}, which is greater than the current IPCC emission factor estimate of 183 kg CH_4 ha^{-1} yr^{-1} (IPCC, 2019). Considering the skewness of our CH_4 data, our median value of 184 kg CH_4 ha^{-1} yr^{-1} agrees with the emission factor of other artificial ponds." Line 368

L36. It's worth noting the recent paper by van Bergen et al who measured CH4 (including ebullition) and CO2 emissions, and C burial of an urban pond. Ideally we need studies that quantify GHG emissions and C burial, so the net balance can be calculated. van Bergen, T.J., Barros, N., Mendonça, R., Aben, R.C., Althuizen, I.H., Huszar, V., Lamers, L.P., Lürling, M., Roland, F. and Kosten, S.,

2019. Seasonal and diel variation in greenhouse gas emissions from an urban pond and its major drivers. Limnology and Oceanography.

Response: The van Bergen reference has now been added to the following sentences in the introduction.

"Artificial reservoirs have the potential to be potent sources of CO2 and CH4 (Downing et al., 2008; Holgerson and Raymond, 2016). This can be demonstrated by a carbon budget estimate from an urban pond where carbon emissions (both diffusive and ebullitive for CH_4) offset carbon burial by >1,000% (van Bergen et al., 2019)." Line 59

L60. "Currently, only three studies have comprehensively assessed C fluxes from small agricultural reservoirs." What does "comprehensively" mean in this case? These three studies are slightly different – Ollivier et al did not measure ebullition whilst the other two studies did. Ollivier et al and Paneer Selvam et al were 'snapshot' studies whilst Grinham included some temporally repeated measurements (but didn't measure CO2). So are they all comprehensive really? I accept this is a minor point of language but it does matter. Additionally, there are two other papers that have measured farm ponds. Stadmark et al made repeated measurements of CH4 and CO2 emissions from agricultural ponds created to retain N: Stadmark, J. and Leonardson, L., 2005. Emissions of greenhouse gases from ponds constructed for nitrogen removal. Ecological Engineering, 25(5), pp.542-551. There is also data in an old and rather blandly titled paper from two farm ponds. Baker-Blocker, A., Donahue, T.M. and Mancy, K.H., 1977. Methane flux from wetlands areas. Tellus, 29(3), pp.245-250. L62. "Large fractions of CH4 being released." Fractions seems like an odd and unsuitable word. Change for "volumes", "amounts", "quantities", etc?

Response: We have removed "comprehensively" and replaced with "at regional scales" in the sentence which now reads:

"Currently, only three studies have assessed C fluxes from small agricultural reservoirs at regional scales and these support the notion that they are important landscape sources of GHGs (Panneer Selvam et al., 2014; Grinham et al., 2018a; Ollivier et al., 2019)." Line 64

Because here we are referring to studies with a high number of sites spanning a regional scale, we will not refer to the other two studies mentioned given they only measured a couple of sites.

L80. The study region occupies a large area, but seeing as temperatures are given it would also be good to give a value (or range) for annual precipitation. Reading on, I see the results says "precipitation ~60% less than the long-term climate average of 390 mm in Regina." Please give the value in the methods.

Response: The following sentence has been added to site description:

"Average annual precipitation in the area ranges from 354 to 432 mm." Line 89

L86. It says 101 ponds were sampled, but in table 1 some variables have N = 102. Where does 102 come from?

Response: We did sample 102 sites but lost GHG measurements from one. Because we are focusing of CO_2 and CH_4 samples in this study, we will refer to total number of sites as 101 and replace 102 in Table 1.

L113, L118. Floating chambers are not "incubations". This word should be altered to something like "deployments" or similar. L121. It says DO was measured in mg/l but in table 1 it is given as %. The methods text should be amended to % instead.

Response: "Incubations" have now been replaced with "deployments". DO units have also been amended to read % saturation in Methods text.

L149. Inflow is mentioned here. Do these systems have inflows? Is water pumped in for storage, or do they simply collect rainwater?

Response: With the water isotope mass balance method, inflow here refers to precipitation, snowmelt, and groundwater inputs. These farm reservoirs are designed collect most water than falls on the landscape due to being positioned in depressional area.

L183. "To avoid multicollinearity, correlation coefficients between pairs from Pearson linear correlation tests was used to guide covariate choice before model fitting." This is vague. Did you use a Pearson correlation coefficient of a certain value to decide when multicollinerity was present?

Response: Here if the correlation was significant then it was decided that multicollinerity was present. We have added that detail to the sentence, which now reads:

"To avoid multicollinearity, correlation coefficients and statistical significance (p <0.05) between pairs from Pearson linear correlation tests was used to guide covariate choice before model fitting (Table S1-3)." Line 199

L197. Something I desperately miss from the paper is a figure allowing the reader to visualise the raw CH4 and CO2 data and its distribution. I strongly advise the addition of a figure to show this. It could take numerous forms, such as a scatter plot of CH4 vs CO2 for all 101 ponds, or a box plot of GHGs (grouped by pond size, or pasture vs cropland), or even a bar plot showing individual concs for 101 ponds (large and unwieldy perhaps, but visually useful). Reading on I see figure.3 has a very small land-use graph, but I think a more obvious, up-front figure would be better.

Response: We have now added a figure (Figure 2) to illustrate the distribution of CO_2 and CH_4 concentrations across all sites. Additionally, we have added Figure S4 and S5 to Supplementary Materials which illustrates scatterplots of all data used in the models.

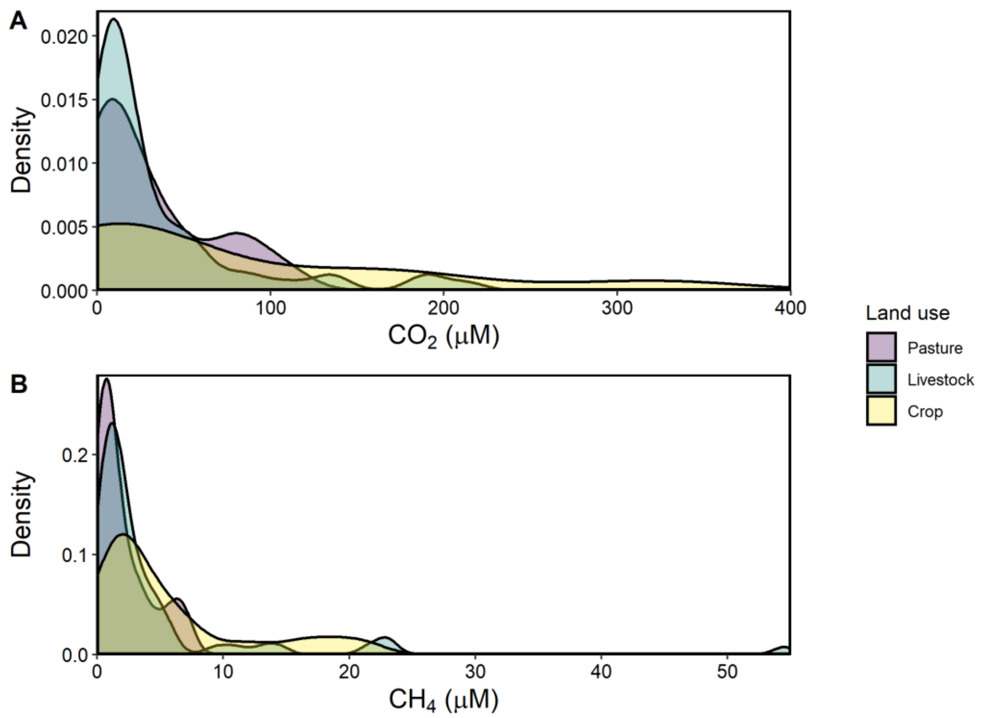


Figure 2: Kernel density estimates of CO_2 and CH_4 concentrations measured in 101 farm reservoirs grouped by land use.

Fig 2 and fig 3. In part this relates to my point above. Wouldn't these figures be improved by adding the underlying data points on to these figures as a scatter? That way the reader can see the model, and the raw data. It would help the reader visually determine the robustness of the models.

Response: While we understand where the reviewer is coming from regarding underlying data points, we chose to avoid adding these here as adding raw data to partial effects plots of GAMs does not provide a meaningful way to represent model fit. These figures illustrate the partial effects transformed on the response scale and the fitted relationship between each covariate and the response is affected by all covariates in the model. Instead, we have now provided diagnostic plots in the supplementary material (Figs. S2 and S3) to allow readers to visually assess the robustness of each model. One of these plots shows the observed versus predicted values of our CO_2 and CH_4 concentrations with the model, where the non-constant variance of the response is visible as increased spread of observations around the 1:1 line (not shown) at higher values of the response.

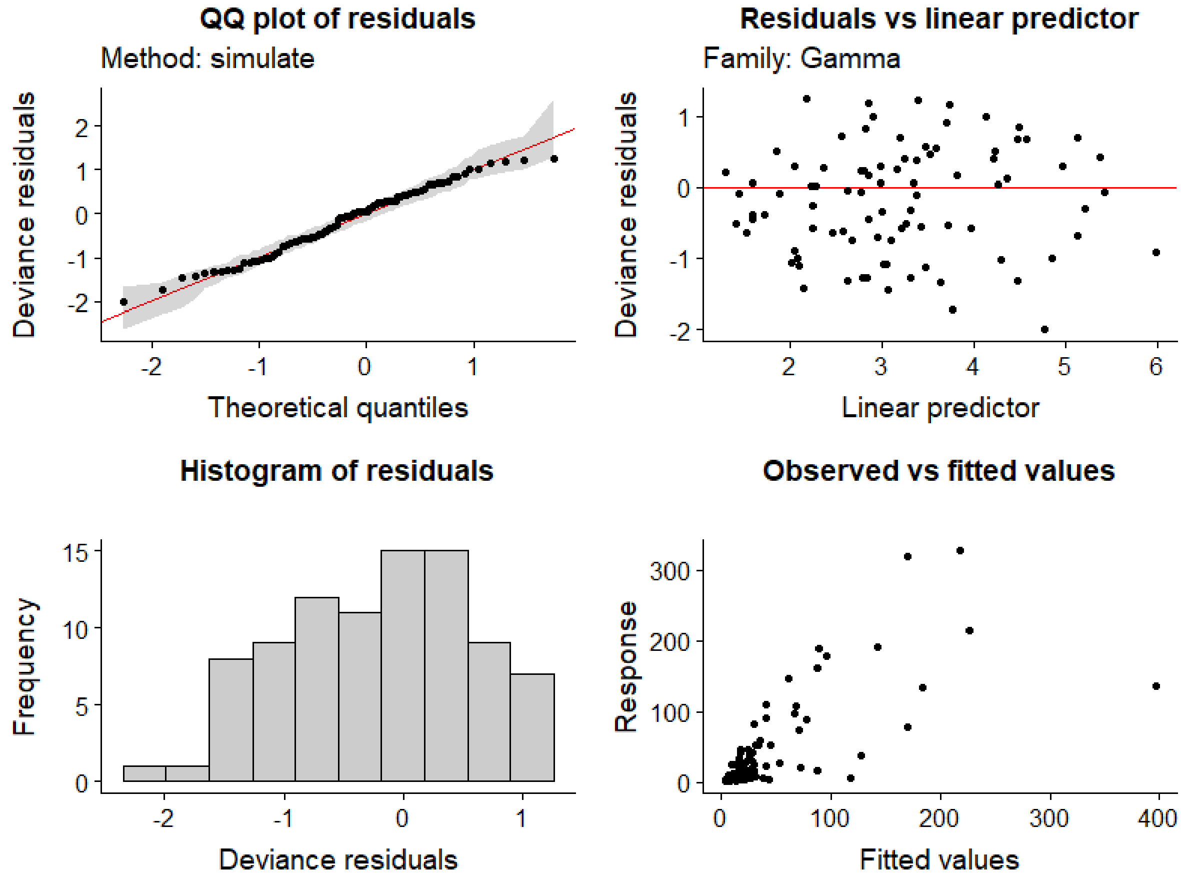


Fig. S2: R output of diagnostic plots for carbon dioxide model

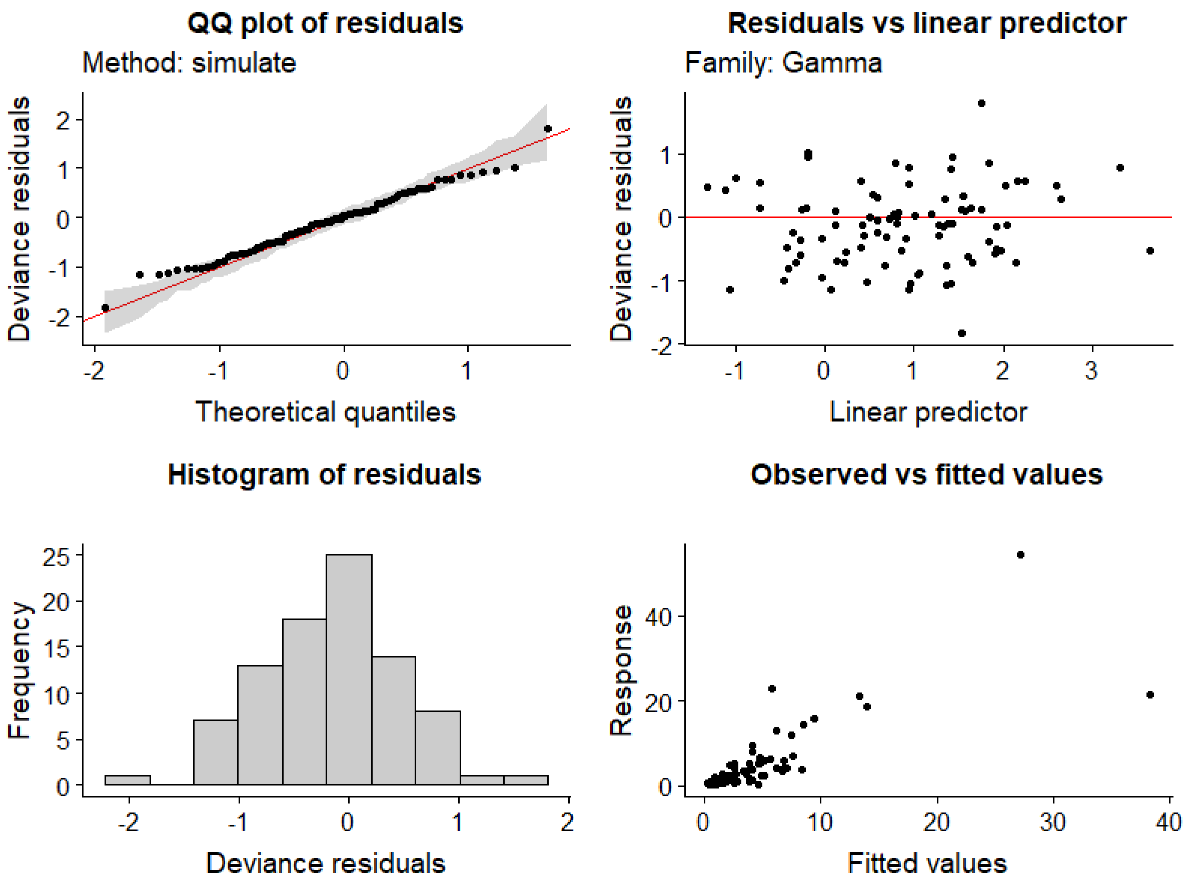


Fig. S3: R output of diagnostic plots for methane model

L210. "CO2concentrations displayed a positive response with. . .NOx" Whilst the upper 95% credible interval continues to increase, the black line presumably suggests that CO2 decreases at the highest NOx levels. Is there a mechanism that can explain this? Figure 3 has a land use graph, but figure two doesn't. Even if there is no difference in CO2 between land use a figure would still be interesting to see, and there is room for an extra panel at the bottom right anyway. For the land use panel in figure 3, the categories are pasture, livestock and cropland. However, line 87 in the methods only mentions pasture (n = 80) and cropland (n = 21). Where do these livestock ponds come from?

Response: Credible intervals always flair out to some extent as they are the extremes of the data as the estimated smooth function is less-constrained there because there are no additional data beyond the observed range to constrain the fitted function. You would see the same thing in a linear model with a small negative effect (slope), but flaring credible interval. The estimated smooth is that which has highest posterior density and reflects the best estimate given the data of the partial effect of NO_x on CO_2; the interval simply reflects the greater uncertainty in the estimate. We have not quantified the probability that the effect is an increasing one here, but given the shape of the upper credible interval, the posterior probability that the smooth effect is increasing is very small, perhaps on the order of a few %. The addition of supplementary figures S4 and S5 shows the distributions and correlations between covariate pairs to demonstrate this.

Fig. S4: Scatterplot matrices of covariate data used in the CO_2 model showing distribution and correlation pairs

Fig S5. Scatterplot matrices of covariate data used in the CH_4 model showing distribution and correlation pairs.

A plot for CO_2 land use model results, Figure 3I, has now been included. We have also corrected the definition of land use types in methods which now mentions livestock:

"We sampled 101 farm reservoirs between July and August 2017, ranging in surface area from 158 – 13,900 m^2 (Table 1), including basins in pasture (n = 18), pastures with livestock (n = 62) and cropland (n = 21) sites." Line 93

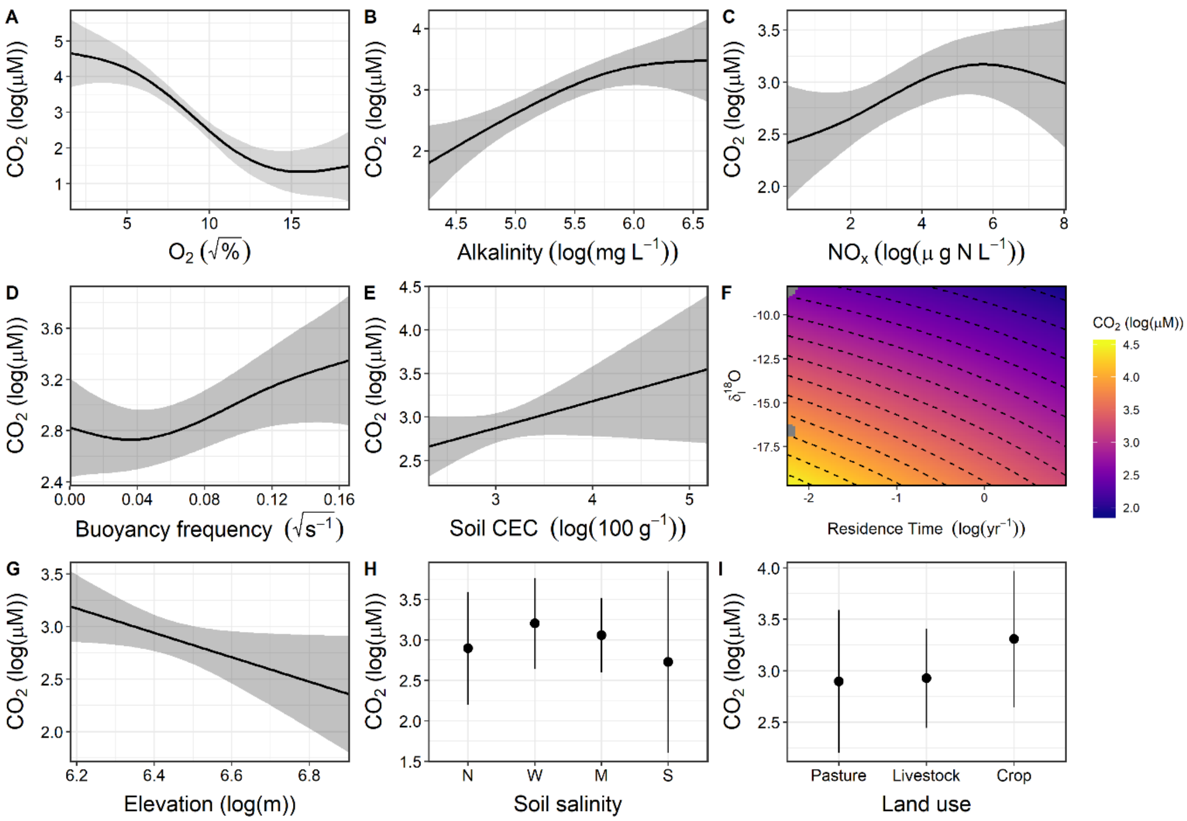


Figure 3: Response patterns farm reservoir CO_2 concentrations with abiotic, biotic, hydromorphological, and landscape variables based on GAMs. CO_2 was best estimated by a combination of a) DO saturation, b) alkalinity, c) NOx, d) buoyancy frequency, e) interaction between δ_I and WRT, f) soil CEC, g) and elevation, with soil salinity (h) and land use (I) not significant. Model deviance explained was 66.5%. The response patterns shown are the partial effect splines from the GAM (solid line) and shaded area indicated 95% credible intervals. See Table S4 and Figure S2 for summary of model statistics and model fit with observed data.

L224. "Our comprehensive spatial analysis revealed wide variations among CO2 and CH4 concentrations between farm reservoirs" As per my previous comment, there's currently no easy way to assess this until the raw data is more visible in a figure.

Response: A new figure (Figure 2), has been provided as suggested previously and is now referenced in that text.

L227. "CH4 was most correlated by internal abiotic and biotic mechanisms" Should this not be "most correlated with"?

Response: Corrected.

L282. "Additionally, smaller waterbodies with shorter WRT can support higher rates of internal CO2 production due higher rates of allochthonous DOC mineralisation" Needs amending to read "due to"

Response: Corrected.

L285. “This mechanism is also suggested by the observation that higher reservoir CO2concentrations are predicted in high CEC soils Alkaline high CEC soils retain more calcium ions within clay particles which releases carbonates and bicarbonates into soil porewater” It seems like something has gone awry in the writing here, and this should be two sentences or some words need removing.

Response: Yes, this sentence should be separated into two. This has now been corrected.

L331. “The effect potential effect of sulfate” The first “effect” needs deleting

Response: Corrected.

L336. “In contrast to the external drivers found for CO2, local land use had a significant effect on CH4 concentrations in farm reservoirs (Fig.3I), with significantly higher CH4 levels in cropland waterbodies than those in pasture. This finding contrasts with those from Australian farm reservoirs where diffusive CH4 fluxes were 250% higher in reservoirs with livestock compared to crops,” I find this section of the discussion interesting. As the authors write, the intensive agricultural practices associated with cropland could be expected to result in elevated CH4 concentrations. Conversely, pasture/livestock emissions would depend on the system (intensive or extensive), livestock, etc. Intensive grassland systems could easily result in high emissions, whilst low-level grazing might result in emissions being less than those from cropland. So cropland > grassland and grassland < cropland are both explicable it seems to me.

Response: We agree that for all land use types, the intensity of agricultural production likely governs the effect on methane in the reservoirs, perhaps more so than land use type itself. Although assessing the intensity of each land use is beyond the scope of this research, we have expanded this section of the discussion with mention to livestock intensity:

“Our finding contrasts with those from Australian farm reservoirs where diffusive CH_4 fluxes were 250% higher in reservoirs with livestock compared to crops, although the mechanisms responsible for observed differences were inconclusive (Ollivier et al., 2019). This difference could be the result of the intensity of agricultural production, where farm reservoirs supporting high intensity grazing may also experience high CH_4 production as demonstrated by a couple of high CH_4 concentrations observed in our livestock pasture reservoirs (Fig. 2). In this case it’s likely that CH_4 levels are more influenced by nutrient loading from the landscape which stimulates eutrophication (Huttunen et al., 2003), as suggested by the biotic variables in our model (Fig. 4). The intensity of agricultural production under different land use types should be an area of further exploration for external controls on farm reservoir GHG production.” Line 357

Figure 4 and fig. 5. The study by Grinham et al of Australian ponds is referenced in the text but doesn’t seem to be included in these figures. Is there any reason their data was left out?

Response: The Grinham et al., 2018 study is included in Figure 4 under “Artificial ponds, Australia” for the CH_4 fluxes. We now realise this reference is not included in the supplemental table referred to in the figure caption. Reference details to this study is now included in Table S6.

L365. “Here, CH4 fluxes were converted to CO2-efluxes using the sustained-flux global warming potential over 100 years” I am not familiar with this metric, and suggest a few lines are included in the methods as to what it is and how it is calculated.

Response: We have added details to how the CH_4 fluxes were converted to CO_2-equivalent fluxes in the methods:

“For comparing CO_2-equivalent fluxes, CH_4 fluxes were converted using the 100-year sustained-flux global warming potential (SGWP, Neubauer and Megonigal, 2015). This metric offers a more attainable measure of ecosystem climatic forcing, assuming gas flux persists over time instead of

occurring as a single pulse as quantified using traditional global warming potentials (GWP, Myhre et al., 2013). Here, a SGWP multiplier of 45 was applied to all CH_4 fluxes in the literature comparison, which is slightly higher than the traditional GWP of 32 over a 100-year time frame (Myhre et al., 2013)." Line 130

Section 4.4. What (if any) vegetation colonises these pools? Is there no role for encouraging certain plant species that might promote C uptake? For instance, Moore & Hunt say: "The carbon sequestration assessment of constructed stormwater wetlands and ponds suggests that emergent vegetation is a significant source to the soil carbon pool (compared to allochthonous sources) and a critical component of carbon sequestration in these systems." Moore, T.L. and Hunt, W.F., 2012. Ecosystem service provision by stormwater wetlands and ponds–A means for evaluation?. Water research, 46(20), pp.6811-6823.

Response: We agree that vegetation likely plays an important role in sequestering carbon in sediments and have added the following paragraph to the discussion in section 4.4:

"Studies have also shown the importance of emergent vegetation plant species in sequestering carbon in sediments. Emergent vegetation was found to contribute significantly to the soil carbon pool of stormwater ponds compared to allochthonous sources (Moore and Hunt, 2012). However, in our CH_4 model, the significant effect of sediment C:N ratios suggested that an autochthonous organic matter source from either phytoplankton or submerged macrophytes supports greater CH_4 production in farm reservoirs. The ability of farm reservoirs to have a negative climate forcing will rely on the balance between GHG fluxes and sediment carbon accumulation. The effect different plant species and other aquatic primary producers have on both these processes needs to be evaluated in future studies as the current design of farm dams within the study area minimises growth of emergent vegetation through steep sides and slopes." Line 426

L392. "The flux of N2O was constrained in our earlier study (Webb et al., 2019), which found a small CO2-e sink (-89 to -3 mg CO2m-2d-1) for the majority of these farm reservoirs despite high N concentrations." Something of a diversion here, but doesn't this depend on how the data are interpreted though? In your earlier study the median N2O flux was negative, but the mean was positive (with 33% of ponds emitting N2O), whilst in this study (figs 4 and 5) you present mean CH4 and CO2. There's probably a debate to be had concerning what average is most appropriate to use, but note the IPCC Refinement used a mean value calculated from log-transformed values.

Response: We thank the reviewer for their insight but have respectfully retained our original presentation. As noted above, we presented both median and mean in the Webb et al. 2019 publication because we wanted to make a clear point that most small agricultural reservoir was, unexpectedly, not a major source of N_2O. This result is not highly dependent on the form of the summary statistic (weak sink, weak source; neither are large). Similarly, in this paper, we focus on the mechanisms predicting variation in the C-based GHG fluxes rather than the absolute values. Thus, while we agree that the 'optics' of the presentation (interpretation by readers) of median and mean are slightly different, we feel that this is a 'side issue' better left for the IPCC committees to debate.

End of Referee #1 response

Anonymous Referee #2

Received and published: 19 August 2019

This paper describes CO2 and CH4 concentration measurements made during the summer season on 101 farm reservoirs in an agricultural region of Saskatchewan, Canada. The authors then use a series of floating chamber measurements to infer diffusive fluxes of these two greenhouse gases at the pond surface via estimations of gas transfer. The authors also collect data on a number of abiotic and biotic landscape/waterbody characteristics that may help predict farm pond GHG concentrations. They then use general additive modeling to describe controls on waterbody concentration. While not currently emphasized, this paper follows up on a previous article that described novel N2O uptake dynamics in these same ponds. The authors emphasize a few findings: 1) more than half of farm ponds are net CO2 sinks, 2) some (19%) farm ponds are net CO2-eq sinks when looking at diffusive emissions, 3) CO2 concentrations are governed most by hydrology/landscape position, 4) CH4 emissions are governed most by autochthonous production.

Response: We thank the reviewer for their critical analysis of our study and appreciate suggestions that further link this work to the broader literature. Detailed responses to comments are provided below.

The current framing of this paper is difficult for me to digest given the complete lack of any CH4 ebullition measurements from these systems (and given that fluxes were estimated based on highly uncertain estimates of gas transfer). While the authors acknowledge that their estimates of CO2-eq emissions are likely low due to the lack of ebullition measurements, this is done at the very end of their paper. I think this point should be made sooner as it is an important detail that influences the interpretation of their findings. The relative contribution of ebullition to total methane flux can vary widely from system to system and the controls on the proportion of methane flux that is ebullitive are not well understood (Deemer et al. 2016 BioScience). It would be helpful to know if the authors observed any evidence of ebullition events during their floating chamber surveys? How much ebullition would have to be observed to push the net CO2-eq sink systems towards net-source? Also, what is the uncertainty in sink vs. source estimations due to uncertainty in system gas transfer velocity? To this same end, it is difficult to see the 19% of systems that are net CO2-eq sinks by looking at the authors' figures. Is this because the net CO2-eq sink is very small? For example, Figure 4 does not seem to show that over 50% of the systems in your study were net CO2 sinks. I suggest adding a zero line to your figures and possibly creating an additional figure that shows fluxes site-by-site for the farm ponds in your study. The visual aids currently offered for showing the distribution of your own dataset are sort of overshadowed by a comparison with the broader literature.

Response: We agree that ebullition can be a major methane flux pathway and plan on investigating this in future field studies. Because the focus of the study was to assess the mechanistic drivers of CO_2 and CH_4 concentrations, the survey was designed to optimise data collection from a large number of sites and ebullition measurements were not carried out. Based on your suggestion, we now highlight this detail earlier in the Methods section:

"To compare with the literature and assess the source/sink behaviour of the reservoirs, diffusive fluxes of carbon dioxide and methane fluxes were estimated for each water body. Given that the focus of the study was to investigate drivers of CO_2 and CH_4 concentrations across farm reservoirs, ebullition events were not measured during this survey and as such total CH_4 fluxes are likely underestimated. Diffusive fluxes were estimated using water column concentrations (C_{water}) and average farm reservoir gas transfer velocity (k_c) using the following equation:

$$f_C = k_c(C_{water} - C_{air}), \qquad (1)"$$

Line 112

We agree that the highly variable nature of gas transfer velocities is the greatest source of uncertainty in flux calculations. As previously mentioned in the manuscript, k600 values for CO_2 and CH_4 were 1.50 ± 1.34 m d^{-1} and 1.64 ± 1.14 m d^{-1}, respectively. These data, along with the median, range, and calculated CO_2 and CH_4 fluxes, have now been added to Table 1 (highlighted in bold below) to provide more transparency to the reader. Please also note that flux and k600 data are provided in a GitHub repository (https://github.com/JackieRWebb/Dugouts-CO2-CH4) which will be publicly available upon publication. Finally, we respectfully note that application of uncertainty values for k600 to our fluxes will increase or decrease the sink or source capacity of the systems, but will not change the number of reservoirs that are CO_2-eq sinks/sources.

Table 1: Farm reservoir and landscape physical, hydrological, and chemical characteristics of the study sites (n = 101)

	Units	*N*	*Mean*	*Median*	*Min*	*Max*
Area	m^2	101	1,312	1,040	158	13,900
Depth	m	101	2.08	2.10	0.18	5.10
Buoyancy frequency	s^{-2}	99	0.01	0.005	0.00	0.03
$\delta^{18}O$ inflow	‰	101	-13.37	-13.33	-19.39	-8.40
Evaporation to inflow		101	0.46	0.43	0.04	1.58
Water residence time	Years	100	0.76	0.66	0.08	2.51
CO_2	µM	101	42.2	14.6	1.3	326.1
CH_4	µM	101	4.3	1.9	0.1	54.5
Flux CO_2						
Positive	**mmol m^{-2} d^{-1}**	**47**	**100.1**	**58.1**	**0.1**	**466.2**
Negative	**mmol m^{-2} d^{-1}**	**54**	**-11.9**	**-13.3**	**-21.3**	**-0.1**
Flux CH_4	**mmol m^{-2} d^{-1}**	**101**	**7.1**	**3.2**	**0.4**	**91.5**
k600- CO_2	**m d^{-1}**	**15**	**1.50**	**0.98**	**0.20**	**4.12**
k600- CH_4	**m d^{-1}**	**23**	**1.64**	**1.25**	**0.38**	**4.14**
Temperature	°C	101	20.1	19.9	15.7	29.5
Dissolved O_2	%	101	92.6	88.9	2.3	344.0
Salinity	ppt	101	0.9	0.5	0.1	8.6
pH		101	8.75	8.75	6.95	10.19
Chlorophyll a	µg L^{-1}	101	99.1	36.9	2.2	2,483
NH_3	µg N L^{-1}	100	354.7	100.0	10.0	5,930
NO_x	µg N L^{-1}	98	196.6	34.1	1.2	3,188
TP	µg P L^{-1}	98	285.2	80.0	8.7	6,480
TN	µg N L^{-1}	98	3,082	2,360	417.5	14,280
DOC	mg C L^{-1}	99	31.8	29.3	4.6	90.4
Sediment organic carbon	%	101	5.2	3.9	0.6	31.4
Sediment organic nitrogen	%	101	0.6	0.4	0.1	2.8
Alkalinity	mg L^{-1}	96	245.4	219.2	71.0	755.5
Soil CEC	M-eq $100g^{-1}$	98	24	24	10	180
K_{sat}	cm hr^{-1}	101	9.9	5.0	0.0	39.7
Elevation	m	101	627.6	598.0	484.0	997.0

As suggested a solid line indicating the threshold between positive and negative fluxes has been added to Figure 5 for better visualisation. The >50% reservoirs that were found to be sinks may be hard to distinguish because our data is highly skewed by some very high concentrations/fluxes. As per the suggestion of Reviewer 1, this is demonstrated more clearly by the addition of a density plot (Figure 2).

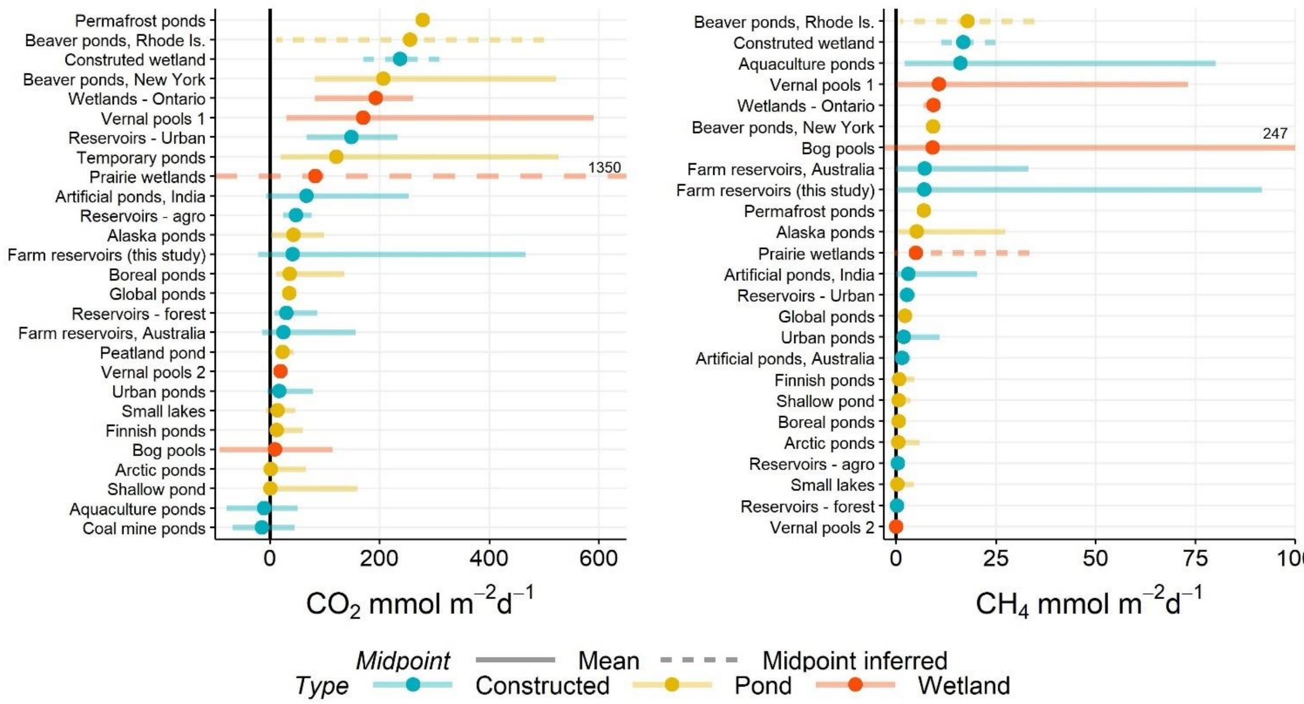


Figure 5: Range of CO_2 and CH_4 (diffusive) fluxes observed in natural and constructed small (<0.01 km^2) waterbodies, including this study (farm reservoirs). Dots represent the mean reported in each study and error bars the range. If no mean value was reported, then the midpoint was inferred as the middle of range (dashed lines). Solid black line distinguished between positive and negative fluxes. All data is from the published literature and references can be found in the Table S6.

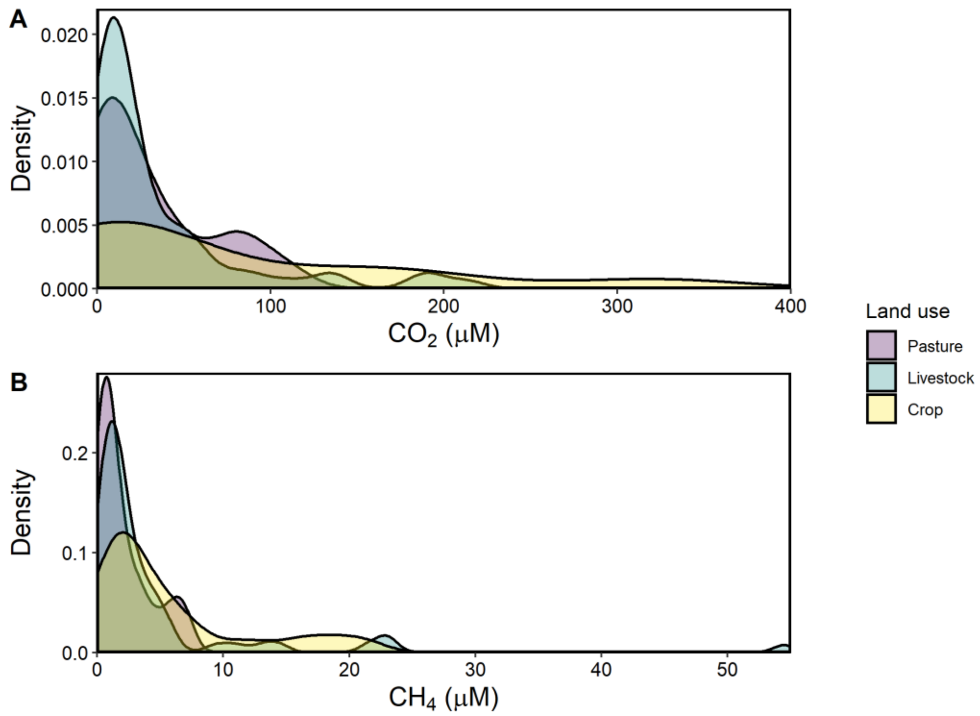


Figure 2: Kernel density estimates of CO_2 and CH_4 concentrations measured in 101 farm reservoirs grouped by land use.

Also, while I am not very familiar with GAMs, I found this analysis a bit opaque and difficult to interpret as currently described. For example, were both N and P variables put into the model and NOx/DIN came out as more important? Also, how were the variables plotted in figures 2 and 3 selected? From what I can gather, you have plotted more than just the variables in the best model. For the sake of discussion, it would be nice to see a consistent set of variables and their relationship to both CH4 and CO2.

Response: Variables for each model were selected based on previous knowledge from the literature on the potential mechanisms controlling CO_2 or CH_4 in freshwater bodies. The model is designed to test the hypothesis of selected environmental controls and included variables representing water chemistry and biology (Table S1), hydrology (Table S2), and external landscape factors (Table S3). As described in the methods, correlation analysis of covariate pairs was first carried out to guide variable selection in the final models as a) some variables represent the same mechanism and are highly correlated (e.g. total N and total P) and b) provided a first assessment of what variables correlated strongest with the response variable within each group of environmental factors. Results of these correlation analysis is provided in Supplementary materials (Table S1-S3). Finally, all variables plotted in Figs 3 and 4 represent those that were included in the GAM and therefore need to be presented, even if some variables came out as non-significant. This reflects modelling best-practice; were we to remove non-significant covariates we would be implying & assuming that the effect(s) on the response were exactly equal to zero, and yet given our data we do not estimate zero effects for these covariates. The model summary statistics and credible intervals on estimated smooth functions or parametric effects presented in the paper include the additional uncertainty that arises from our ignorance of exactly which covariates had the strongest controls on CO_2 or CH_4. It is from here that we learn what the most important mechanisms are for potentially controlling gas concentrations.

To me, the more novel part of this data set is the high fraction of ponds that are net CO2 sinks. This is also a finding that is most strongly backed by the data that was collected since the conclusion doesn't rely as much on gas transfer estimates and since CO2 ebullition is typically an extremely small fraction of total CO2 emission. The extent of the CO2 sink in these small agricultural ponds could be compared to the lesser extent reported in the global data set of artificial reservoir GHG dynamics (Deemer et al. 2016). It is also interesting that the CO2 sink seems to scale more with landscape and hydrological factors than with ecosystem productivity. While multiple other studies have already emphasized the potential importance of nutrient management/eutrophication on lake, pond, and reservoir methane emissions (see Beaulieu et al. 2019 for a very recent global scale discussion), the findings you present in this paper suggest that landscape placement of farm reservoirs may help buffer GHG emissions independent of trophic status (via carbonate buffering and groundwater DIC chemistry dynamics). See paper by Pacheco et al 2013 in Inland Waters (which asks if eutrophication can reverse the aquatic C budget). To this end, it would also be nice to see plots comparing emission by land use for both CH4 and CO2 (right now the plot is only shown for CH4).

Response: We agree and have expanded the following paragraph in discussion to emphasize our findings on CO_2 uptake:

"The negative fluxes observed in our farm dams represents one of the few studied small waterbodies that exhibit CO_2 sink behaviour, with most showing net heterotrophy (Fig. 5). Although other studies have noted CO_2 sink behaviour in artificial ponds and reservoirs (Peacock et al., 2019; Ollivier et al., 2019), this is the first study to capture such a high proportion (>52%) of CO2 uptake in such systems, with negative fluxes estimated to range between -21 to -0.1 (mean -12) mmol m-2 d-1 for CO_2 (Table 1). These flux ranges compare to CO_2 uptake of -1 to -11 mmol m-2 d-1 in agricultural eutrophic lakes of North America (Finlay et al., 2010; Pacheco et al., 2013). Studies have shown the importance

of eutrophication, leading to net autotrophy, in enhancing CO_2 uptake and reversing carbon budgets in lakes (Pacheco et al., 2013). However, a global analysis of GHG fluxes from lakes and reservoirs revealed that the consequence of increased CH4 emissions with increasing trophic status often outweighs the impact of negative CO_2 fluxes (Deemer et al., 2016). Here, our model shows the potential importance of reservoir placement within the landscape as a way of reducing CO_2 emissions via hydrological and geochemical controls without the added consequence of increased CH_4 emissions." Line 372

A suggested by yourself and Reviewer 1, land use in now included in Figure 3 for the CO_2 model. In addition, the new Figure 2 also shows the raw data distribution for CO_2 concentrations by land use.

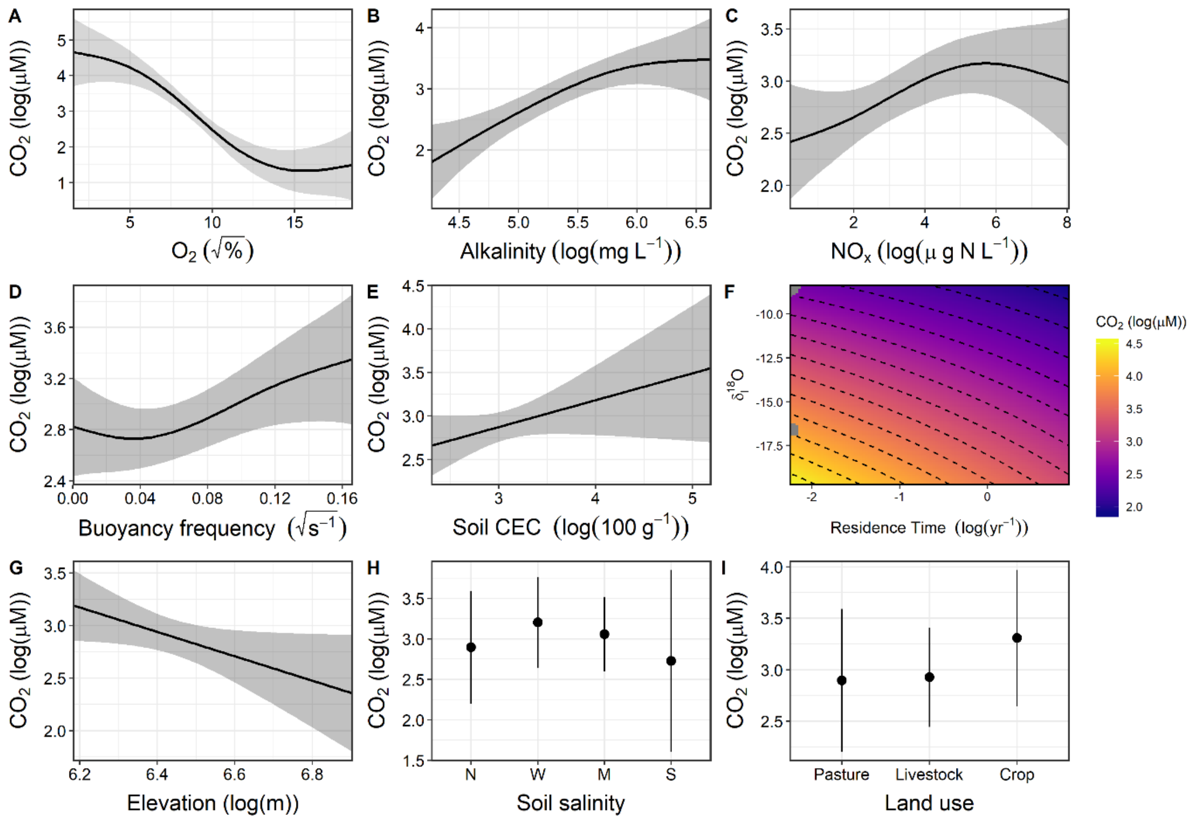


Figure 3: Response patterns farm reservoir CO_2 concentrations with abiotic, biotic, hydromorphological, and landscape variables based on GAMs. CO_2 was best estimated by a combination of a) DO saturation, b) alkalinity, c) NOx, d) buoyancy frequency, e) interaction between δ_I and WRT, f) soil CEC, g) and elevation, with soil salinity (h) and land use (I) not significant. Model deviance explained was 66.5%. The response patterns shown are the partial effect splines from the GAM (solid line) and shaded area indicated 95% credible intervals. See Table S4 and Figure S2 for summary of model statistics and model fit with observed data.

The comparison between human-made and natural waterbodies is also interesting and novel. I think it would be good to more thoroughly introduce this question/concept (that the systems might fundamentally differ from each other) earlier in the paper and then come back to it in the discussion. A good reference for comparing human-made and natural waterbodies is Hayes et al. 2017 L&O Letters as well as Doubek & Carey 2017 Inland Waters.

Response: We agree that human-made and natural waterbodies function differently from each other on a range of ecological scales. However, our discussion of the literature review focuses on CO_2 and CH_4 fluxes only and to date have revealed few differences between constructed and natural systems, mainly because both systems have highly variable flux rates (Lines 382, 388). Given our focus on CO_2 and CH_4 fluxes here, we did not want to add overly speculative text on the potential impact of human-made and natural waterbodies.

Line by Line Edits

Line 18: add “surface” before “concentrations”

Response: Corrected

Lines 20-21: this is a little misleading since pH was actually a better predictor

Response: the term “best” has been removed.

Lines 23-24: state the timescale over which you are calculating CO2-equivalents

Response: “100-year radiative forcing” has been added.

Line 26: bringing up depth doesn’t seem appropriate here since depth didn’t come out as a significant predictor variable in your models

Response: Depth has been removed from this sentence and revised to more accurately reflect our model findings:

“From our models, we show that the GHG impact of farm reservoirs can be greatly minimised with overall improvements in water quality and consideration to position and hydrology within the land scape.” Line 25

Line 30-31: Holgerson and Raymond 2016 didn’t look at ebullition

Response: We have now clarified that this reference refers to diffusive fluxes only: *“Current assessments estimate that diffusive CO_2 and CH_4 emissions from small ponds (<0.001 km^2) account for 15% and 40% of global emissions from lakes, respectfully (Holgerson and Raymond, 2016).”* Line 30

Line 45-46: Also check out Couto and Olden 2018. . . there aren’t really global papers that distinguish surface area of small farm reservoirs/ponds from small hydropower.

Response: We have added “artificial reservoirs” to this sentence to be clear that this global estimate does not just refer to farm reservoirs.

Lines 46-47: I suggest listing out numbers of reservoirs by country since the current phrasing is difficult to interpret. Either that or use a word like “collectively” to indicate that 8 million is the sum across multiple countries.

Response: “collectively” has been added.

Line 51: What does It mean to create reservoirs at a rate of up to 60% of standing stock? I’m a bit confused by this wording.

Response: “standing stock” has been replaced with “existing reservoirs”.

Lines 56-57: It is a bit awkward to suggest that eutrophication results in potent CO2 release since autochthonous production actually works to fix CO2 (see Pacheco et al. 2013).

Response: The mention to eutrophication has been removed from the sentence.

Lines 76-77: I suggest clarifying: you are identifying drivers of surface water concentration, not total flux. Although these are related, they are not the same thing.

Response: “fluxes” have been replaced with “concentrations”.

Lines 86-87: How did you select your sites? Randomly?

Response: Sites were selected from a database of farm reservoirs collected by a survey of regional landowners, as well as from sites on federal lands. Site selection was refined by ensuring a relatively even spatial distribution across the study area, while also considering ease of access.

Lines 197-202: What were N:P ratios like in these systems?

Response: Total N to P ratios (by mass) varied from 1.4 to 126. Readers will be able to refer to all raw data provided in a Github repository ((https://github.com/JackieRWebb/Dugouts-CO2-CH4) which will be made public upon publication.

Results section: I suggest including a summary of the fluxes you estimate (and associated gas transfer rates from the floating chamber surveys). Can you estimate how variability in k might affect variability in your flux estimates? Are there cases where you have both a floating chamber and a concentration based estimate of flux? How much did these differ from each other?

Response: As suggested by the reviewer, we have added the summary statistics for both fluxes and measured gas transfer velocities to Table 1. In the results section, we have focused on describing gas concentrations and model results. Instead, description of fluxes are presented later in the paper to aid with comparison of literature values.

Line 227: change “by” to “with”

Response: Corrected

Line 246: Not a complete sentence.

Response: Sentence corrected to read *“Here, we see evidence for both linked and divergent processes (Fig. 3A).”* Line 261

Lines 261-262: This doesn’t seem like a very satisfying explanation to me. Is it also possible that differing hydrology leads to the more stratified systems also being the ones that are higher in CO2?

Response: We agree that this sentence is speculative and have removed it.

Line 269: add “of” between “effect” and “increased”

Response: Corrected

Line 270: Nitrification doesn’t produce CO2; it is an autotrophic process.

Response: “nitrification” has been removed.

Line 272: This is a pretty vague topic sentence. It would be helpful to be a little more specific.

Response: Sentence has been revised to read: *“Hydrological controls were found to be important regulators of CO_2 concentrations in these farm reservoirs.”* Line 286

Line 303: get rid of “by”

Response: Corrected

Lines 306-307: Deemer et al. 2016 and Beaulieu et al. 2019 are also good references here.

Response: References have been added

Lines 312-315: Higher CH4 from higher C:N sediments suggests more (not less) important role for allochthonous C right?

Response: Our C/N ratios (8.5 to 13.4) were low enough to still be in the range of autochthonous C based on Liu et al., 2018. However, we have added a sentence to account for the input of allochthonous C contributing to higher C/N ratios: *"This suggests that in situ rather than terrestrial organic matter (OM) was likely the main source of C fuelling methanogenesis in these reservoirs, although increasing CH_4 concentrations with C/N may also represent a larger contribution of terrestrial OM."* Line 328

Line 318-319: I would expect thermal stratification to influence bottom water CH4 concentration more than surface water CH4, but you only have surface water concentrations in your model.

Response: Yes, this is most likely the case. We have clarified the sentence to read:

"Thermal stratification of the water column did not significantly influence surface CH_4 concentrations in small farm reservoirs (Fig. 4E)." Line 333

Line 331: Get rid of second "effect"

Response: Corrected

Line 334-335: Avoid using the word "clearly". Also, it would be helpful to show the relationship between CH4 and salinity in your Figure 3 to support this discussion.

Response: "Clearly" has been removed from the sentence which now reads: *"Evidently, the biological influence on CH_4 concentrations appears less pronounced in these larger, low-flow dams."* Line 349. The inclusion of conductivity in the CH_4 model already represents a potential sulfate effect and supports this discussion.

Lines 365-366: State the actual factor that you used here too. Was it 34?

Response: At the suggestion of Reviewer 1 for additional information on the calculation of CO_2-equivalent emissions, this has now been provided in the Methods:

"For comparing CO_2-equivalent fluxes, CH_4 fluxes were converted using the 100-year sustained-flux global warming potential (SGWP, Neubauer and Megonigal, 2015). This metric offers a more attainable measure of ecosystem climatic forcing, assuming gas flux persists over time instead of occurring as a single pulse as quantified using traditional global warming potentials (GWP, Myhre et al., 2013). Here, a SGWP multiplier of 45 was applied to all CH_4 fluxes in the literature comparison, which is slightly higher than the traditional GWP of 32 over a 100-year time frame (Myhre et al., 2013)." Line 129

Lines 392-393: It seems like it would be nice to mention this parallel study earlier in your paper and give it a bit more discussion.

Response: We agree and now bring attention to this study in the Introduction:

"This study builds on from our previous research farm reservoir GHG research which found an unexpected nitrous oxide (N_2O) sink in 67% of reservoirs (Webb et al., 2019)." Line 72

Lines 378-383: This all seems very speculative. As do lines 400-403.

Response: We agree that some of the mechanistic narrative is speculative; however, we also feel that our analysis is robust and that these statements provide promising avenues for further testing of tangible solutions for GHG reduction, both by ourselves and other researchers. Consequently, we have respectfully decided to retain this material, unless the editor feels strongly that it should be removed.

We now clarify the mention of building deeper reservoirs as a way to increase water residence time, which was a parameter in our model found to be related to lower CO_2 and CH_4 concentrations:

"Increasing WRT by creating deeper reservoirs may promote primary production through increased water clarity (Dirnberger and Weinberger, 2005), facilitate CH_4 oxidation through the water column (Bastviken et al., 2008), and reduce the impact of watershed-derived solutes, terrestrial OM and benthic respiration." Line 407

End of Referee #2 response

Regulation of carbon dioxide and methane in small agricultural reservoirs: Optimizing potential for greenhouse gas uptake

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Abstract. Small farm reservoirs are abundant in many agricultural regions across the globe and have the potential to be large contributing sources of carbon dioxide (CO_2) and methane (CH_4) to agricultural landscapes. Compared to natural ponds, these artificial waterbodies remain overlooked in both agricultural greenhouse gas (GHG) inventories and inland water global carbon (C) budgets. Improved understanding of the environmental controls of C emissions from farm reservoirs is required to address and manage their potential importance in agricultural GHG budgets. Here, we conducted a regional scale survey (~235,000 km^2) to measure CO_2 and CH_4 surface concentrations and diffusive fluxes across 101 small farm reservoirs in Canada's largest agricultural area. A combination of abiotic, biotic, hydromorphologic, and landscape variables were modelled using generalized additive models (GAMs) to identify regulatory mechanisms. We found that CO_2 concentration was estimated by a combination of internal metabolism and groundwater-derived alkalinity (66.5% deviance explained), while multiple lines of evidence support a positive association between eutrophication and CH_4 production (74.1% deviance explained). Fluxes ranged from -21 to 466 and 0.14 to 92 mmol m^{-2} d^{-1} for CO_2 and CH_4, respectively, with CH_4 contributing an average of 74% of CO_2-equivalent (CO_2-e) emissions based on a 100-year radiative forcing. Approximately 8% of farm reservoirs were found to be net CO_2-e sinks. From our models, we show that the GHG impact of farm reservoirs can be greatly minimised with overall improvements in water quality and consideration to position and hydrology within the land scape.

1 Introduction

The expansion of agriculture and urban land use has introduced a new type of lentic system that remains relatively unexplored – small artificial waterbodies (Clifford and Heffernan, 2018). These artificial aquatic systems have been created through human modification of the hydrological landscape and include small farm reservoirs and urban ponds. Farm reservoirs are earthen excavations designed to store water for later use (BC Ministry of Agriculture, 2013). The global abundance of these systems remains uncertain (Verpoorter et al., 2014), but statistical extrapolation suggest there may be around 16 million artificial reservoirs worldwide (Lehner et al., 2011). Regional-scale inventories indicate that collectively upwards of 8 million farm reservoirs exist in the USA (Brunson, 1999; Smith et al., 2002), China (Chen et al., 2019), India (Anbumozhi et al., 2001), South Africa (Mantel et al., 2017), and Australia alone (Lowe et al., 2005; MDBA, 2008; Grinham et al., 2018a). The density of farm reservoirs can exceed 30% of agricultural area in some regions such as China where food demand is high (Chen et al., 2019). Small agricultural reservoirs are estimated to cover 77,000 km^2 globally and are being created at rates up to 60% of existing reservoirs per annum in some regions (Downing et al., 2008). Given their abundance, these artificial systems may contribute substantially to landscape biogeochemical cycles, including fluxes of GHG. In particular, very little is known of the capability of these systems to act as GHG sinks to partially offset the otherwise strong carbon efflux associated with intensive agriculture (Robertson et al., 2000).

Small waterbodies have recently been recognised as substantial contributors to global carbon emissions from inland waters. Current assessments estimate that diffusive CO_2 and CH_4 emissions from small ponds (<0.001 km^2) account for 15% and 40% of global emissions from lakes, respectfully (Holgerson and Raymond, 2016). Other estimates suggest emissions from small lakes and impoundments (0.001 to 0.01 km^2) could constitute 40% of global CO_2 emissions and 20% of global CH_4 emissions from lentic ecosystems (DelSontro et al., 2018). Extreme CO_2 and CH_4 supersaturation is characteristic of small waterbodies due to greater contact with the sediment and littoral zone (Downing et al., 2008; Holgerson, 2015), often making them disproportionately important in landscape carbon (C) budgets (Hamilton et al., 1994; Premke et al., 2016; Kuhn et al., 2018). Conversely, ponds may have the capacity to store landscape-significant amounts of carbon, with burial rates 20–30 times higher than wetlands and large lakes (Gilbert et al., 2014; Taylor et al., 2019). While these assessments have stimulated a growing area of research on small waterbodies, much work is still needed to revise estimates of their carbon emissions due to limited knowledge on their regional distribution and variability, as well as their overall global extent (Verpoorter et al., 2014). This is particularly true for greenhouse gas (GHG) emissions from human-created small waterbodies.

Understanding the controls and rates of carbon fluxes from small artificial waterbodies is the first step required to understand their landscape and eventually global importance. Further, estimates of CO_2 and CH_4 flux are complicated by high variation among reservoirs and regions in the importance of groundwater, littoral macrophytes, and local land use practises (Pennock et al., 2010; Badiou et al., 2019). Artificial reservoirs have the potential to be potent sources of CO_2 and CH_4 (Downing et al., 2008; Holgerson and Raymond, 2016). This can be demonstrated by a carbon budget estimate from an urban pond where carbon emissions (both diffusive and ebullitive for CH_4) offset carbon burial by >1,000% (van Bergen et al., 2019). The recent 2019 IPCC Refinement has assigned a CH_4 emission factor of 183 kg ha^{-1} yr^{-1} to constructed waterbodies, however data is greatly limited, both geographically and in number (n = 68), that climatic-zone emission factors cannot be estimated (IPCC, 2019). Currently, only three studies have assessed C fluxes from small agricultural reservoirs at regional scales and these support the notion that they are important landscape sources of GHGs (Panneer Selvam et al., 2014; Grinham et al., 2018a; Ollivier et al., 2019). All studies found large fractions of CH_4 being released, and large mean CO_2 emissions on the order of 24 and 99 mmol m^{-2} d^{-1}, comparable to the global average flux rate of very small natural ponds (35 mmol m^{-2} d^{-1}, Holgerson and Raymond, 2016). However, carbon fluxes from farm reservoirs remain unaccounted in agricultural GHG inventories and global inland water carbon budgets. To facilitate their inclusion in agricultural and global budgets, we need to further constrain flux rates and mechanisms across a broad geographic area.

Here, we present a large-scale assessment of CO_2 and CH_4 concentrations from small farm reservoirs in the Northern Great Plains, the largest agricultural region in Canada. This study builds on from our previous farm reservoir GHG research which found an unexpected nitrous oxide (N_2O) sink in 67% of reservoirs (Webb et al., 2019). The hydroclimate, lithology and edaphic features are vastly different compared to previous studies of agricultural areas (Australia, India, USA), with factors that favour CO_2 uptake by alkaline surface waters (Finlay et al., 2009; Finlay et al., 2015) and lead to high variability in CH_4 fluxes from regional wetlands (Pennock et al., 2010; Badiou et al., 2019). Our aim was to identify the key environmental conditions regulating CO_2 and CH_4 fluxes, and to evaluate this baseline data in the context of emission mitigation strategies. To achieve this goal, we carried out an extensive survey of CO_2 and CH_4 concentrations across 101 farm reservoirs and used generalized additive models (GAMs) to assess the effects of abiotic, biotic, hydromorphological and land use properties. Our findings show that farm dams were not always strong sources of carbon emissions and in certain cases can be carbon neutral or sinks in terms of CO_2-equivalent (CO_2-e) emissions. By identifying the driving characteristics of farm dams that support reduced C emissions, our findings provide the first step to developing management strategies to help minimise farm carbon emissions.

2 Methods

2.1 Study site

Farm sites were surveyed across the agricultural region of Saskatchewan, Canada (Fig. 1). This region covers an area of 235,000 km^2 in the southern half of the province, where agriculture accounts for ~80% of land use. The region has a sub-humid to semi-arid climate (Köppen D*fb* classification), with short warm summers (~18°C) and long winters (~-17°C) resulting in 4.5 to 5.5 months of ice cover on surface waters (Finlay et al., 2015). Average annual precipitation in the area ranges from 354 to 432 mm.

Small farm reservoirs (known locally as ‘dugouts’) are a prominent feature of the landscape, with densities up to 10 per km^2 (Fig. 1B). Up until 1985, over 110,000 farm reservoirs had been constructed in Saskatchewan (Gan, 2000), although subsequent densities are unknown. We sampled 101 farm reservoirs between July and August 2017, ranging in surface area from 158 – 13,900 m^2 (Table 1), including basins in pasture (n = 18), pastures with livestock (n = 62) and cropland (n = 21) sites. Each site was sampled once during this period, between the daylight hours of 10:00 to 15:00. Saskatchewan farm reservoirs are typically uniform in shape and morphometry, dug to a depth of 4 to 6 m with steep sides (1.5:1 slopes). Most shallow wetlands and lakes in the region exhibit water balances dominated by evaporation and limited inflow from winter precipitation or groundwater (Conly and van der Kamp, 2001; Pham et al., 2009). Farm reservoirs differ from small natural waterbodies in that they have a higher ratio of water volume to surface area, designed to minimise evaporation losses. Despite this feature, arid conditions persisted during the sampling year, with reduced (34-65%) annual rainfall such that many reservoirs were only half their designed depth. Natural waterbodies also tend to be high pH hard-water systems, owing to the soils which consist of glacial till high in carbonates (Last and Ginn, 2005). The same was observed for the majority of farm reservoirs, with an average pH of 8.75 (Table 1).

2.2 CO_2 and CH_4 measurements

Dissolved gas samples were collected using the in-field headspace extraction method (Webb et al., 2019). Briefly, water was collected from ~30 cm below the surface using a submersible pump which filled a 1.2-L glass-serum bottle, ensuring the bottle overflowed and no air bubbles were present. The bottle was sealed with a rubber stopper fitted with two three-way stopcock valves. Using two 60-mL air-tight syringes, atmospheric air was added to the bottle whilst simultaneously extracting 60-mL of water. The bottle was then shaken for 2 minutes to ensure gas equilibration in the headspace. Two analytical replicates were extracted and stored in 12-mL evacuated Exetainer vials with double-wadded caps. Headspace concentrations of CO_2 and CH_4 were measured using gas chromatography with a Scion 456 Gas Chromatograph (Bruker Ltd.) and calculated using standard curves. Dry molar fractions were corrected for dilution and converted to concentrations according to solubility coefficients (Weiss, 1974; Yamamoto et al., 1976).

To compare with the literature and assess the source/sink behaviour of the reservoirs, diffusive fluxes of carbon dioxide and methane fluxes were estimated for each water body. Given that the focus of the study was to investigate drivers of CO_2 and CH_4 concentrations across farm reservoirs, ebullition events were not measured during this survey and as such total CH_4 fluxes are likely underestimated. Diffusive fluxes were estimated using water column concentrations (C_{water}) and average farm reservoir gas transfer velocity (k_c) using the following equation:

$$f_C = k_c(C_{water} - C_{air}), \quad (1)$$

where f_c is the flux of CO_2 or CH_4 (mmol m^{-2} d^{-1}) and C_{air} is the ambient air concentration. The average global mixing ratios for the sampling period of 406 and 1.85 µatm were used for ambient concentrations for CO_2 and CH_4 respectively (Mauna Loa NOAA station, June to August 2017). Site-specific gas transfer velocity (k_c) was determined from 30 individual floating- chamber (area = 0.23 m^2, volume = 0.046 m^3) measurements carried out on a subset of 10 reservoirs. During each 10-minute deployment, changes in gas concentrations were measured at 2.5-min intervals by taking samples using syringes and dispensing gases into pre-evacuated 12-mL vials. The flux (mmol m^{-2} d^{-1}) was calculated from the observed rate of change in the dry mole fraction of the respective gas (Lorke et al., 2015). The gas transfer velocity normalised to a Schmidt number of 600 (k_{600}) for each respective gas was then determined using measured flux, *in situ* gas concentrations, atmospheric concentration, Henry's constant, and Schmidt numbers, assuming a Schmidt exponent of 0.67. The average k_{600} calculated from the floating chamber deployments was 1.50 ± 1.34 m d^{-1} and 1.64 ± 1.14 m d^{-1} for CO_2 and CH_4, respectively (Table 1).

For comparing CO_2-equivalent fluxes, CH_4 fluxes were converted using the 100-year sustained-flux global warming potential (SGWP, Neubauer and Megonigal, 2015). This metric offers a more attainable measure of ecosystem climatic forcing, assuming gas flux persists over time instead of occurring as a single pulse as quantified using traditional global warming potentials (GWP, Myhre et al., 2013). Here, a SGWP multiplier of 45 was applied to all CH_4 fluxes in the literature comparison, which is slightly higher than the traditional GWP of 32 over a 100-year time frame (Myhre et al., 2013).

2.3 Abiotic and biotic variables

A range of abiotic and biotic parameters were measured at each site. Water quality variables including temperature (ºC), pH, dissolved O_2 (DO; % saturation), conductivity (µS cm^{-2}), and salinity were measured at 0.5-m intervals from the surface to the bottom using a YSI (Yellow Springs Instruments, OH, USA) multi-probe meter. Surface (0.5 m) samples for water chemistry were collected using a submersible pump. Upon collection, samples for dissolved nitrogen (NO_3+NO_2, NH_4, total dissolved N; µg N L^{-1}), soluble reactive phosphorus (SRP; µg P L^{-1}) and total dissolved P (TDP; µg P L^{-1}), dissolved organic and inorganic carbon (DOC, DIC; mg C L^{-1}), alkalinity (OH + HCO_3 + CO_3; mg L^{-1} as $CaCO_3$), and water isotopes (δ^2H, δ^{18}O; ‰) were filtered through a 0.45-µm pore membrane filter. Nutrient and dissolved carbon samples were stored in a dark bottle at 4°C until analysis. Chlorophyll *a* (Chl-*a*) samples were collected on GF/C glass-fiber filters (nominal pore size 1.2 µm) and frozen (-10°C) until analysis. Sediment samples were collected at the centre of each reservoir, the uppermost 10 cm using an Ekman grab sampler, and were frozen at -10°C until analysis.

Most analyses were carried out at the University of Regina Institute of Environmental Change and Society (IECS). Water nutrient and dissolved carbon concentrations were measured on a Lachat QuikChem 8500 and Shimadzu model 5000A total carbon analyzer, following standard analytical procedures, respectively (Patoine et al., 2006;Finlay et al., 2009). Alkalinity was measured using standard methods of the US Environmental Protection Agency (EPA) on a SmartChem 200 Discrete Analyser (WestCo) and estimated as the concentration of $CaCO_3$ (EPA, 1974). Chl-*a* was analysed using standard trichromatic methods (Finlay et al. 2009). The total carbon and nitrogen content (% dry weight) of freeze-dried sediment samples were determined on a NC2500 Elemental Analyzer (ThermoQuest, CE Instruments).

2.4 Hydromorphology

Morphometric parameters of reservoirs were estimated for each site. The depth of each farm reservoir was measured during using a portable ultrasonic depth sounder, taken at the deepest section in the centre of the reservoir. Surface area was determined using Google Earth satellite imagery. Reservoir volume was calculated using the formula for a prismoid by assuming that all sites maintained their original shape, including slopes of 1.5:1 ratio (Andresen et al., 2015). From these measurements, an Index of Basin Permanence (IBP) was calculated (Kerekes, 1977).

The degree of water-column mixing or vertical stratification was determined by calculating the squared Brunt-Väisälä buoyancy frequency (N^2, s^{-2}). The strongest density gradient was calculated based on vertical temperature measurements at 0.5-m depth intervals using the package *rLakeAnalyzer* (Read et al., 2012) in R (version 3.5.2; R Core Team 2018).

The hydrology of farm reservoirs was estimated through analysis of $\delta^{18}O$ and $\delta^{2}H$ isotope values of water. Samples were collected from 0.5 m below the surface, filtered (0.45-µm pore) and stored in amber borosilicate jars at 4°C until analysis using a Picarro L2120-I cavity ring-down spectrometer (CRDS). Hydrological parameters, including evaporation to inflow ratio (E/I), residence time (years), and inflow volume (m^3), deuterium (^2H) excess (d-excess), and $\delta^{18}O$ inflow (δ_I) values, were calculated using the coupled isotope tracer method (Yi et al., 2008) and conventional isotopic water-balance methods (Gibson et al., 2001). All methods assumed that reservoirs were headwater systems in hydrological steady-state (Yi et al., 2008). Model inputs included information about the local water meteoric line (LWML), the trajectory of evaporation along a local evaporative line (LEL), and regional meteorological conditions. From here, the water mass balance of a given waterbody can be quantified based on its relative position along the LEL (Gibson et al., 2001).

Briefly, the isotopic inflow values were estimated by the intercept between the LWML and site-specific LEL as determined by $\delta^{18}O$ evaporation value (δ_E) and $\delta^{18}O$ reservoir water value at each site (Yi et al., 2008). The E/I ratio was calculated by using headwater isotopic models of the water mass balance ($(\delta_I - \delta_L) * (\delta_E - \delta_L)^{-1}$). Hydrologic residence time was estimated from the reservoir volume and the water isotopic values of waterbodies, inflow, and evaporation. Deuterium excess (d-excess ‰ = $\delta^2H - 8*\delta^{18}O$) was calculated as an additional indicator of evaporation losses, where lower values (< -10‰) indicate isotopic enrichment from precipitation (Brooks et al., 2014).

2.5 Landscape properties

Landscape soil data was obtained from The National Soil DataBase, Government of Canada (http://sis.agr.gc.ca/cansis/nsdb/dss/v3/index.html) using ArcGIS to extract the soil attributes at each site. Extracted variables included soil salinity, soil pH, soil organic carbon content, saturated hydraulic conductivity (K_{sat}), cation exchange capacity (CEC), and the total composition of soil from sand, silt, and clay fractions (%). Reservoir elevation (m, a.s.l.) was determined using ArcGIS and the Canadian Digital Elevation Model (CDEM, v1.1). Local land use in the immediate area surrounding each reservoir was categorised into three types based on local observations at the time of sampling. Categories included pasture land used for either livestock grazing or hay harvesting, pasture where livestock have direct access to the waterbody, and crop fields.

2.6 Statistical analyses

Environmental variables were selected based on known or presumed influence on CO_2 and CH_4 concentrations in lakes and small waterbodies. Both biotic and abiotic predictors that influence production or consumption of CO_2 and CH_4 were selected, including DO, alkalinity, NO_X (NO_2 + NO_3), NH_4, dissolved inorganic nitrogen (DIN), TDN, TDP, Chl-*a*, DOC, conductivity, pH, and sediment organic C:N ratio. The influence of reservoir hydrology and morphology were also examined, including measures of surface area, basin permanence, hydrologic regime (E/I), water source (δ_I), and degree of mixing (or stratification). Finally, potential effects of the surrounding terrestrial landscape were estimated in models using soil properties, elevation, and land use practises to account for any localised landscape drivers. Before testing relationships, all predictors were transformed as needed using either $\log_{10}$ or square root to remove skewness.

The relationships between covariates and CO_2 and CH_4 were estimated using generalised additive models (GAMs). GAMs provide an ideal approach to model non-linear associations between predictor variables and responses, using the sum of unspecified smooth functions to estimate trends. GAMs are not constrained by prescribed assumptions associated with parametric models such as linearity in generalized linear models, and instead use information from the current set of data to draw predictions. The more flexible modelling approach is useful for uncovering non-standard relationships between predictor and response variables and has been applied to complex aquatic datasets assessing GHGs (Wiik et al., 2018; Webb et al., 2019). GAMs were developed with a gamma distribution for the response and the log link function. Each model included covariates that represented hydromorphological, abiotic and biotic, and landscape controls. To avoid multicollinearity, correlation coefficients and statistical significance (p <0.05) between pairs from Pearson linear correlation tests was used to guide covariate choice before model fitting (Table S1-3). Candidate variables were then selected for each model to test which variables best estimate variability in CO_2 and CH_4 concentrations. All model coefficients were estimated using restricted marginal likelihood with the *mgcv* package (Wood, 2011; Wood et al., 2016) for R (version 3.5.2; R Core Team 2018).

3 Results

The region experienced a drier than average year during sampling, with recorded average annual precipitation ~60% less than the long-term climate average of 390 mm in Regina, Saskatchewan (Government of Canada, http://climate.weather.gc.ca). Consequently, while most farm reservoirs were constructed to ~5 m depth the mean water-column depth was 2.1 m (0.2-5.1, Table 1). Despite this, isotopic analysis of water revealed that 93% of waterbodies exhibited an E/I < 1.0, suggesting that reservoirs were gaining more water than was lost via evaporation. In general, water residence time was ~8 months, although the range in this value was large (29 days to 2.5 years). Estimates of inflow $\delta^{18}O$ (δ_I) indicated variable water sources, with 79% derived from rain (>-15.66‰), 6% from snowmelt or groundwater (<-17.9‰), and 15% intermediate between sources (-17.9 to -15.6‰).

Carbon dioxide and methane concentrations spanned three orders of magnitude across surveyed reservoirs, with concentrations ranging between 1.3 to 326.1 and 0.1 to 54.5 µM for CO_2 and CH_4, respectively (Fig. 2). Most waterbodies were alkaline, with a mean pH of 8.8 (7.0 to10.2) and carbonate alkalinity between 71 and 755 mg L^{-1} (Table 1). Many waters were highly eutrophic, with means for Chl-*a* of 99 µg L^{-1} (range 2 to 344 µg L^{-1}), total nitrogen of >3,000 µg N L^{-1} (418 to 14,280), and total phosphorus of 285 µg P L^{-1} (9 to 648). Dissolved O_2 in the surface layer varied by three orders of magnitude among basins with 32% exhibiting oversaturation (>100%).

3.1 Models

Regional variation in CO_2 concentrations were best estimated in a GAM including pH alone, with 86.3% of deviance explained and a strongly declining CO_2 at pH above 8 (Fig. S1). Exclusive of the model with pH, the detailed mechanistic GAM for estimating CO_2 concentrations across farm reservoirs included a combination of DO saturation, alkalinity, NO_x, thermal stratification (buoyancy frequency), basin hydrology (the interaction between δ_I and WRT), and landscape features (soil CEC, elevation, soil salinity) (Fig. 3). Overall, the model explained 66.5% of deviance in CO_2 concentrations (Table S4, Fig. S2). All covariates had a significant effect except soil salinity, with DO, alkalinity, and the interaction between δ_I and WRT being the strongest predictors (p <0.001). CO_2 concentrations displayed a positive response with increasing alkalinity, NO_x, buoyancy frequency, and soil CEC, with a generally negative response to increasing DO and elevation. The effect of DO on CO_2 was particularly distinct between 25 and 100% O_2 saturation (Fig. 3A). The interactive effect of hydrology parameters suggests that sites with elevated rain inflows ($\delta^{18}O$ > -12.5‰) and longer WRT will exhibit undersaturated CO_2 concentrations.

Variation in CH_4 concentrations among waterbodies were explained by a combination of DO saturation, sediment C/N ratio, DIN, conductivity, the interaction between δ_I and WRT, and local land use (Fig. 4), with buoyancy frequency, soil K_{sat}, and elevation not significant. Overall, the GAM explained 74.1% of the deviance in CH_4 (Table S5, Fig. S3). Concentrations of CH_4 increased with sediment C/N and DIN and decreased with conductivity. The significant unimodal relationship with DO indicates that the highest observed CH_4 concentrations occurred under both anoxic and supersaturated O_2 environments (Fig. 4A), while low CH_4 levels were seen when inflow was more composed of snowmelt or groundwater (depleted isotope values) and WRT was long (Fig. 4F). In contrast to the CO_2 model, soil properties and elevation were not significant drivers, yet local land use was significant, with crop sites having significantly higher CH_4 compared to pastures.

4 Discussion

Our comprehensive spatial analysis revealed wide variations among CO_2 and CH_4 concentrations between farm reservoirs (Fig. 2). Significant modelled environmental drivers suggested CO_2 was primarily controlled by pH, with strong independent models indicating mechanisms associated with primary productivity, the hydrological regime, and landscape elevation. In contrast, CH_4 was most correlated with internal abiotic and biotic mechanisms. We discuss these potential drivers in detail and from our evidence suggest management strategies that may help reduce the net GHG effect of these farm reservoirs.

4.1 Environmental drivers of CO_2 concentrations

As seen in other hardwater ecosystems, variations in CO_2 were strongly coupled to differences among sites in water-column pH (Finlay et al., 2015; Müller et al., 2016). We demonstrate this with the strong correlation observed between CO_2 and pH in a separate GAM of only water pH as a covariate, explaining 86.3% of deviance (Fig. S1). As expected, the role of pH in regulating CO_2 content is most pronounced at values between 8.6-9.0, the transition point where the predominant species of DIC shifts from free CO_2 to HCO_3^- (Duarte et al., 2008; Finlay et al., 2015). Above this value, carbonate buffering increasingly regulates pH and restricts CO_2 to only trace fractions of total DIC (Stumm and Morgan 1970). However, direct changes in CO_2 concentrations can also alter water-column pH, such as biological metabolism (Talling, 2010). Therefore, given the direct chemical relationship between pH and CO_2 concentrations (Stumm and Morgan, 1970), we opted to leave pH out of our model to further investigate the underlying biological, chemical, hydrological, and land use mechanisms.

The detailed GAM showed that variance in CO_2 concentrations among farm reservoirs was estimated (66.5% of deviance) by a combination of predictors related to water-column productivity and microbial metabolism (DO saturation, alkalinity, NO_x), thermal stratification (buoyancy frequency), basin hydrology (the interaction between δ_I and WRT), and landscape features (soil CEC, elevation) (Fig. 3), but not local soil salinity. This was shown by the DO, alkalinity, δ_I and WRT covariates having the most significant effect at $p<0.001$, while CO_2 concentrations did not vary significantly between different soil salinity levels (Table S4, Fig. 3).

Carbon dioxide and dissolved oxygen are closely linked by biological metabolism in aquatic systems and diverge when other chemical or physical processes occur. Here, we see evidence for both linked and divergent processes (Fig. 3A). The tight linear relationship between CO_2 and O_2 at 25 to 100% saturation indicates close coupling between the gases. This likely represents control via metabolic processes such as net ecosystem production (NEP) or chemical oxidation of reduced species (Stets et al., 2017). In contrast, relationships between CO_2 and O_2 were less well defined a both high and low oxygen saturations, conditions which may indicate a greater contribution from anaerobic production of CO_2 (Torgersen and Branco, 2008; Holgerson, 2015). Alternatively, alkalinity buffering can mediate the effect of NEP on CO_2 concentrations at both extreme ranges of the DO spectrum (Marcé et al., 2015). Alkalinity buffering is most likely to affect CO_2-DO relationships in waters where alkalinity is >2000 µeq L^{-1} (Stets et al., 2017) which was the case for ~90% of our sites (Table 1; Fig. 3).

Stratification can also weaken the impact of DO as a driver for CO_2 by regulating the effect of sediment respiration on epilimnetic chemistry (Huotari et al., 2009; Holgerson, 2015). Our model shows that those sites that were most stratified (elevated buoyancy frequency) exhibited higher CO_2 concentrations (Fig. 3D). This pattern contrasts those observed in other small lentic systems where elevated epilimnetic CO_2 concentrations were observed during and after breakdown of water-column stratification (Huotari et al., 2009; Glaz et al., 2016). Preliminary seasonal studies of some farm reservoirs in 2018 show that stratification is strong and persistent throughout the summer, with no obvious diurnal mixing events. Such strong stratification can maintain anoxic conditions throughout most of the water column, which supports intense anaerobic respiration and CO_2 production.

The positive association between NO_x and CO_2 found in our reservoirs is consistent with similar patterns seen with dissolved inorganic N species in other artificial waterbodies (Ollivier et al., 2019; Peacock et al., 2019) and regional prairie lakes (Wiik et al., 2018). In some lakes, high N loading favoured elevated heterotrophy, despite simultaneous boosts in primary production which draws down free CO_2 (Huttunen et al., 2003; Cole et al., 2000). The effect of a high N influx on CO_2 may be heightened in smaller or shallow lentic waters which are more influenced by sedimentary processes (Torgersen and Branco, 2008). Further, high N availability can increase algal biomass and the deposition of fresh OM made increasingly available for bacterial respiration (Cole et al., 2000). As a result, the effect of increased benthic respiration offsets CO_2 uptake by primary producers, while extremely high influx of dissolved N can also favour microbial processes such as denitrification which increase CO_2 evolution (Bogard et al., 2017).

Hydrological controls were found to be important regulators of CO_2 concentrations in these farm reservoirs. Sites which received most of their inflow from snowmelt or groundwater, and which had short WRT supported supersaturated CO_2 concentrations (Fig. 3F). Such patterns may reflect increased inputs of groundwater which are typically supersaturated with CO_2 (Macpherson, 2009). Long WRT is associated with larger, deeper systems. These sites are usually less influenced by the terrestrial-aquatic interface, take longer to concentrate the effect of any catchment-derived solutes (Junger et al., 2019), and have higher biotic assimilation of nutrients (Devito and Dillon, 1993; Fairchild and Velinsky, 2006). Larger waterbodies may also be able to better mediate stream or groundwater C inputs through longer chemical processing times and transformations. For example, agricultural reservoirs with the highest WRTs tended to be hydrologically closed systems (E/I > 1) and any watershed derived DIC delivered from previous water sources is likely to be consumed by primary production which encourages atmospheric CO_2 uptake (Macrae et al., 2004) Additionally, smaller waterbodies with shorter WRT can support higher rates of internal CO_2 production due to higher rates of allochthonous DOC mineralisation (Weyhenmeyer et al., 2015; Vachon et al., 2017).

Groundwater delivery of DIC-rich porewater is the most likely hydrological source resulting in CO_2 enrichment of small farm reservoirs. This mechanism is also suggested by the observation that higher reservoir CO_2 concentrations are predicted in high CEC soils. Alkaline high CEC soils retain more calcium ions within clay particles which releases carbonates and bicarbonates into soil porewater (Kelley and Brown, 1934). Although regional snowmelt and groundwater have similar isotopic signatures (Pham et al., 2009; Jasechko et al., 2017), the positive correlation of CO_2 with alkalinity suggests groundwater as the main source. Edaphic sources of inorganic carbon can result in farm waterbodies accumulating dissolved CO_2, bicarbonates, and carbonates, and therefore alkalinity, from the surrounding soils via groundwater discharge (Miller et al., 1985). Other studies have found strong evidence for groundwater inputs driving CO_2 supersaturation in small lentic systems (Perkins et al., 2015; Peacock et al., 2019) and watershed-derived alkalinity driving CO_2 supersaturation in lakes (Marcé et al., 2015).

Finally, landscape elevation had a significant external effect on reservoir CO_2 and may represent diverse weak controls related to landscape setting. Lower CO_2 concentrations at higher elevations are common in 'perched' ecosystems with smaller contributing catchment areas (Diem et al., 2012) and low rates of allochthonous carbon influx (Rose et al., 2015). Conversely, waterbodies low in the landscape may receive more watershed C via groundwater influx due to topographical gradient (Winter and LaBaugh, 2003; van der Kamp and Hayashi, 2009). The effect of elevation could also be related to changes in vegetation composition within the local landscape, with the lowest lying catchments exhibiting higher abundance of marginal wetland vegetation (Zhang et al., 2010) which favours higher inputs of terrestrial C (Magnuson et al., 2006; Abril et al., 2014).

4.2 Environmental drivers of CH_4 concentrations

The GAM suggested that CH_4 concentrations were primarily related to internal biogeochemical processes and the influence of the hydrological regime. For example, factors related to water column productivity (DO, sediment C/N, DIN, conductivity) had the most significant effect (p <0.01), while some of the broader landscape features such as soil K_{sat} and elevation had no significant effect on CH_4 levels. The nutrient status of waterbodies is often a primary driver of high CH_4 emissions in lakes, impoundments, and ponds (Deemer et al., 2016; Beaulieu et al., 2019; Peacock et al., 2019). Consequently, high nutrient availability is likely fuelling elevated values in both O_2 saturation and CH_4 (Fig. 4A). High CH_4 concentrations at low O_2 saturation reflects the development of anoxic habitats which favours methanogenesis (Huttunen et al., 2003; Bastviken et al., 2004). This is likely the result of rapid biomass production which both enriches epilimnion with O_2 and depletes O_2 in the hypolimnion by providing fresh labile organic matter for decomposition.

In support of eutrophication-driven CH_4 production, our model indicated that high proportions of autochthonous organic matter in sediments were associated with elevated concentrations of CH_4 (Fig. 4B). Overall, sedimentary C/N ratios were in the range (8.5 to 13.4) expected for both phytoplankton and submerged macrophytes (Liu et al., 2018). This suggests that *in situ* rather than terrestrial organic matter (OM) was likely the main source of C fuelling methanogenesis in these reservoirs, although increasing CH_4 concentrations with C/N may also represent a larger contribution of terrestrial OM. Strong associations of labile autochthonous C and CH_4 production in sediments (Due et al., 2010; Crowe et al., 2011) also suggests a direct link between eutrophication and CH_4 production in small farm waterbodies.

Thermal stratification of the water column did not significantly influence surface CH_4 concentrations in small farm reservoirs (Fig. 4E). This finding contrasts with observations from other small waterbodies where limited mixing favours CH_4 accumulation (Kankaala et al., 2013). Although some small systems exhibit diurnal mixing patterns with turnover at night (Glaz et al., 2016), the wide range of buoyancy frequency values (0.00 to 0.16) suggests that at least some farm reservoirs are continuously stratified, particularly in deeper ponds (Kankaala et al., 2013), as noted for CO_2 distributions (see above and Fig. 3D). Taken together, our findings suggest that variability in the biological production of CH_4 likely exerts a stronger influence over CH_4 concentrations across farm reservoirs than does physical mixing, and further supports the hypothesis that the prevailing sediment and water chemistry are the primary controls of CH_4 concentrations.

Although the hydrological regime of small water bodies is rarely measured, we find that water source (rain, snow/groundwater) and reservoir retention time interact to influence CH_4 concentrations (Fig. 4F). In particular, CH_4 concentrations were lowest when WRT was long (>1 year) and water was derived mainly from snow or groundwater sources ($\delta^{18}O$ depleted). This may be due to a combination of reasons, including the prevalence of sulfate delivered from groundwater (Pennock et al., 2010), dilution of waterbody from snow melt inflow, and sediments depleted in labile carbon due to longer biogeochemical processing times in the dams. The potential effect of sulfate limiting methanogenesis is in agreement with the strong negative relationship found between CH_4 and conductivity in our model (Fig. 4D). Sulfate makes up a large portion of the ionic composition of groundwater in the Prairie Pothole Region due to pyrite oxidation (Goldhaber et al., 2014). Evidently, the biological influence on CH_4 concentrations appears less pronounced in these larger, low-flow dams.

In contrast to the external drivers found for CO_2, local land use had a significant effect on CH_4 concentrations in farm reservoirs (Fig. 4I), with significantly higher CH_4 levels in cropland waterbodies than those in pasture. Catchment land use regulates the physioco-chemical properties of ponds (Novikmec et al., 2016) by influencing the degree of local vegetative cover and associated influx of allochthonous C to waterbodies (Whitfield et al., 2011). Similarly, regions with crops undergo more intensive agricultural modification, with fertilisation, crop rotations, and mechanical disturbance of soil which all lead to greater nutrient runoff and soil erosion. Our finding contrasts with those from Australian farm reservoirs where diffusive CH_4 fluxes were 250% higher in reservoirs with livestock compared to crops, although the mechanisms responsible for observed differences were inconclusive (Ollivier et al., 2019). This difference could be the result of the intensity of agricultural production, where farm reservoirs supporting high intensity grazing may also experience high CH_4 production as demonstrated by a couple of high CH_4 concentrations observed in our livestock pasture reservoirs (Fig. 2). In this case it's likely that CH_4 levels are more influenced by nutrient loading from the landscape which stimulates eutrophication (Huttunen et al., 2003), as suggested by the biotic variables in our model (Fig. 4). The intensity of agricultural production under different land use types should be an area of further exploration for external controls on farm reservoir GHG production.

4.3 Emissions from farm reservoirs compared to other small waterbodies

To date, small waterbodies on farms have been shown to be large emitters of both CO_2 and CH_4 (Fig. 5). However, in our study we show that this is not always the case. Diffusive fluxes varied -21 to 466 and 0.14 to 92 mmol m^{-2} d^{-1} for CO_2 and CH_4, respectively. These findings are consistent with other small artificial waterbodies which are strong CH_4 sources that exhibit a large range of variability from 0.02-33 mmol m^{-2} d^{-1} (Grinham et al., 2018a; Ollivier et al., 2019). Average CH_4 fluxes from our farm reservoirs correspond to 417 kg CH_4 ha^{-1} yr^{-1}, which is greater than the current IPCC emission factor estimate of 183 kg CH_4 ha^{-1} yr^{-1} (IPCC, 2019). Considering the skewness of our CH_4 data, our median value of 184 kg CH_4 ha^{-1} yr^{-1} agrees with the emission factor of other artificial ponds.

The negative fluxes observed in our farm dams represents one of the few studied small waterbodies that exhibit CO_2 sink behaviour, with most showing net heterotrophy (Fig. 5). Although other studies have noted CO_2 sink behaviour in artificial ponds and reservoirs (Peacock et al., 2019; Ollivier et al., 2019), this is the first study to capture such a high proportion (>52%) of CO_2 uptake in such systems, with negative fluxes estimated to range between -21 to -0.1 (mean -12) mmol m^{-2} d^{-1} for CO_2 (Table 1). These flux ranges compare to CO_2 uptake of -1 to -11 mmol m^{-2} d^{-1} in agricultural eutrophic lakes of North America (Finlay et al., 2010; Pacheco et al., 2013). Studies have shown the importance of eutrophication, leading to net autotrophy, in enhancing CO_2 uptake and reversing carbon budgets in lakes (Pacheco et al., 2013). However, a global analysis of GHG fluxes from lakes and reservoirs revealed that the consequence of increased CH_4 emissions with increasing trophic status often outweighs the impact of negative CO_2 fluxes (Deemer et al., 2016). Here, our model shows the potential importance of reservoir placement within the landscape as a way of reducing CO_2 emissions via hydrological and geochemical controls without the added consequence of increased CH_4 emissions.

When CO_2 and CH_4 fluxes from small artificial waterbodies are compared with natural small waterbodies, no apparent trend exists in which group produces more or less carbon emissions (Fig. 5). Natural ponds and constructed waterbodies have a similar range in variability of mean fluxes for both gases, while wetlands exhibit some of the greatest within-study variability. Constructed waterbodies often have lower net CO_2 efflux, suggesting that these systems more often switch between net autotrophy and heterotrophy than small natural systems. Small artificial waterbodies have disproportionately higher CO_2 and CH_4 emissions than other natural waterbodies due to the direct impact of agricultural and urban land use (Wang et al., 2017). However, analysis of the limited literature shows that is not the case. We suggest that the lack of a clear distinction between constructed and naturally-occurring small water bodies arises because of geographical variation in the relative importance of the diverse factors regulating carbon metabolism (Figs. 3, 4).

When assessing the GHG impact of constructed waterbodies, it is important to consider the relative contribution to CO_2-equivalent (CO_2-e) fluxes between CO_2 and CH_4. Here, CH_4 fluxes were converted to CO_2-e fluxes using the sustained-flux global warming potential over 100 years (Neubauer and Megonigal, 2015). On average, 8% of farm reservoirs were acting as CO_2-e sinks on the range of -0.6 to 79 g CO_2 m^{-2} d^{-1} during the time of sampling. This number offers a snapshot of the potential for farm reservoirs to act as a net CO_2-e sink and it is important to consider how seasonal variation influences the GHG sink/source status. Preliminary data on seasonal variation in CO_2 and CH_4 concentrations from a smaller number of farm reservoirs indicate variation (represented as the standard deviation related to the mean) ranging between 20 to 200% and 40 to 200% for CO_2 and CH_4, respectively. Here, this variation represents monthly sampling between the periods of ice melt and ice formation on water bodies in Saskatchewan. Applying the average observed seasonal variation of 78% and 93% to our current spatial dataset suggests that CO_2-e emissions from farm reservoirs may vary between -1.7 and 150 g CO_2 m^{-2} d^{-1}, or 0 to 44% as acting net CO_2-e sinks. Further study into the consistency of potential farm reservoir CO_2 sinks on the temporal scale is required to better assess the overall GHG impact.

Small natural ponds and wetlands have some of the highest CO_2-e emission rates, with particular importance of contributions from CH_4 (Fig. 6). On average our farm reservoirs had one of the highest CH_4 contribution to CO_2-e fluxes (74%), in agreement with the one other farm reservoir study (83%) of CH_4 contribution (Ollivier et al., 2019). This large contribution from CH_4 is similar to patterns recorded from lakes and impoundments globally, where large freshwater bodies contribute to

75% of all CO_2-e efflux (DelSontro et al., 2018). Fortunately, because the factors that regulate CH_4 emissions are becoming better identified (Fig. 4), there exists the possibility that artificial wetlands can be constructed to minimize CH_4-related CO_2-e emissions and mitigate the overall large rate of CO_2-e emissions from agriculture (Robertson et al., 2000).

4.4 Minimising emissions: potential management solutions

A combination of factors, including landscape position, construction, and management, could optimize features to minimize carbon emissions from reservoirs and potentially enhance the carbon storage on farms. From our models, we suggest that key variables including the degree of water column stratification (buoyancy frequency), WRT, water source, land use, and elevation are all suitable parameters for management. For example, strategizing landscape positioning to favour groundwater influx of sulfate to reduce methanogenesis. Increasing WRT by creating deeper reservoirs may promote primary production through increased water clarity (Dirnberger and Weinberger, 2005), facilitate CH_4 oxidation through the water column (Bastviken et al., 2008), and reduce the impact of watershed-derived solutes, terrestrial OM and benthic respiration. Additionally, deeper and larger artificial waterbodies tend to have lower nutrient concentrations due to longer processing times (Chiandet and Xenopoulos, 2016). Finally, modest increases in pH may further enhance CO_2 capture (Supporting Information), while having limited effect on CH_4 fluxes (Fig. 4).

Agricultural and urban waterbodies are highly susceptible to nutrient enrichment due to their direct proximity to intensified land uses. Reducing nutrient loading from the landscape will likely have one of the greatest impacts in minimising C emissions from farm dams given that both CO_2 and CH_4 were strongly predicted by inorganic N-species. In Australian farm reservoirs, for example, a 25% reduction of nitrates can reduce CO_2-e emissions by 50% (Ollivier et al., 2019). Similarly, removing direct livestock access to farm waterbodies will improve water quality overall through reducing direct DIN inputs and dam infilling.

Nitrogen loading can also have a direct influence on nitrous oxide (N_2O), the third most potent greenhouse gas that can contribute substantially to CO_2-e emissions in farm systems (Robertson et al., 2000). The flux of N_2O was constrained in our earlier study (Webb et al., 2019), which found a small CO_2-e sink (-89 to -3 mg CO_2 m^{-2} d^{-1}) for the majority of these farm reservoirs despite high N concentrations. Similar to our CO_2 model, stratification and primary production were important regulators in driving N_2O uptake (Webb et al., 2019). Therefore, the potential to achieve net GHG sinks weighs mostly on the ability to reduce CH_4 emissions in these systems.

Studies have also shown the importance of emergent vegetation plant species in sequestering carbon in sediments. Emergent vegetation was found to contribute significantly to the soil carbon pool of stormwater ponds compared to allochthonous sources (Moore and Hunt, 2012). However, in our CH_4 model, the significant effect of sediment C:N ratios suggested that an autochthonous organic matter source from either phytoplankton or submerged macrophytes supports greater CH_4 production in farm reservoirs. The ability of farm reservoirs to have a negative climate forcing will rely on the balance between GHG fluxes and sediment carbon accumulation. The effect different plant species and other aquatic primary producers have on both these processes needs to be evaluated in future studies as the current design of farm dams within the study area minimises growth of emergent vegetation through steep sides and slopes.

It is important to note that the CH_4 contribution to CO_2-e emissions is likely underestimated here as ebullition emissions were not measured. In farm reservoirs, ebullition flux can contribute >90% of total CH_4 emissions and is often highest in the smallest size classes (Grinham et al., 2018a). However, the sporadic nature of this pathway remains difficult to constrain for one single type of waterbody and may be a minor contributor in reservoirs and ponds > 3-5 m deep (Joyce and Jewell, 2003; DelSontro et al., 2016). This reinforces that design and management strategies that focus on reducing all pathways of CH_4 emissions will be most effective in curbing total CO_2-e emissions. Deeper farm dams with steep side slopes will likely be effective in reducing ebullition events due to a limited macrophytes, reduced bottom water temperature in summer, and supressed bubble release with higher water pressure (Joyce and Jewell, 2003; Natchimuthu et al., 2014; Grinham et al., 2018b).

5 Conclusion

Until recently, carbon emissions from small farm reservoirs have been an overlooked, yet potentially important source of CO_2 and CH_4 emissions within agricultural carbon budgets. To date, development of management strategies to reduce GHG emissions from waterbodies has been limited by lack of knowledge about the mechanisms regulating CO_2 and CH_4 production in these systems. By utilising adaptive modelling techniques across a broad range of environmental variables (abiotic, biotic, hydromorphological, landscape properties), we were able to explain a high degree of deviance in reservoir CO_2 and CH_4 concentrations. We found that *in situ* water chemistry and local hydrological regime had the strongest impact on CO_2 and CH_4 concentrations. In agreement with previous studies, CH_4 fluxes were the largest contributor to CO_2-e emissions. However, in 19 reservoirs the net CO_2-e emissions were found to be sinks. We suggest that with optimal reservoir design and management the climatic impact of farm reservoir C-emissions has the potential to be a carbon net sink. To further develop farm reservoir management practices that are locally effective, we express a need for more widespread farm waterbody GHG measurements across the globe to cover other continents and land uses.

Data availability: All data used in the models is available online in a GitHub repository (https://github.com/JackieRWebb/Dugouts-CO2-CH4). Public access to this repository will be made available upon publication and a DOI will be generated at this time.

Supplement: The supporting information related to this study will be published online.

Author contributions*:* J.R.W., G.L.S., P.R.L., H.M.B., and K.F. designed research; J.R.W. performed research and wrote the paper; H.M.B. contributed new reagents/analytic tools; H.A.H., P.R.L., G.L.S., and K.F. contributed towards ideas and data analysis; K.R.H performed GIS analysis; and G.L.S. developed models.

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References

Abril, G., Martinez, J.-M., Artigas, L. F., Moreira-Turcq, P., Benedetti, M. F., Vidal, L., Meziane, T., Kim, J.-H., Bernardes, M. C., and Savoye, N.: Amazon River carbon dioxide outgassing fuelled by wetlands, Nature, 505, 395, 2014.

Anbumozhi, V., Matsumoto, K., and Yamaji, E.: Towards Improved Performance of Irrigation Tanks in Semi-Arid Regions of India: Modernization Opportunities and Challenges, Irrigation and Drainage Systems, 15, 293-309, 10.1023/a:1014420822465, 2001.

Andresen, B., Buchanan, B., Corkal, D., Fairley, B., Fortin, R., Hilliard, C., Kidd, J., Pasquill, R., Sketchell, J., and Thompson, T.: Quality Farm Dugouts. Forestry, A. A. a. (Ed.), Alberta Government, Alberta, 2015.

Badiou, P., Page, B., and Ross, L.: 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, 2019.

Bastviken, D., Cole, J., Pace, M., and Tranvik, L.: Methane emissions from lakes: Dependence of lake characteristics, two regional assessments, and a global estimate, Global biogeochemical cycles, 18, 2004.

Bastviken, D., Cole, J. J., Pace, M. L., and Van de Bogert, M. C.: Fates of methane from different lake habitats: Connecting whole-lake budgets and CH4 emissions, Journal of Geophysical Research: Biogeosciences, 113, 2008.

Beaulieu, J. J., DelSontro, T., and Downing, J. A.: Eutrophication will increase methane emissions from lakes and impoundments during the 21st century, Nature Communications, 10, 1375, 10.1038/s41467-019-09100-5, 2019.

Bogard, M. J., Finlay, K., Waiser, M. J., Tumber, V. P., Donald, D. B., Wiik, E., Simpson, G. L., del Giorgio, P. A., and Leavitt, P. R.: Effects of experimental nitrogen fertilization on planktonic metabolism and CO_2 flux in a hypereutrophic hardwater lake, PLOS ONE, 12, e0188652, 10.1371/journal.pone.0188652, 2017.

Brooks, J. R., Gibson, J. J., Birks, S. J., Weber, M. H., Rodecap, K. D., and Stoddard, J. L.: Stable isotope estimates of evaporation : inflow and water residence time for lakes across the United States as a tool for national lake water quality assessments, Limnology and Oceanography, 59, 2150-2165, doi:10.4319/lo.2014.59.6.2150, 2014.

Brunson, M. W.: Managing Mississippi farm ponds and small lakes, 1999.

Chen, W., He, B., Nover, D., Lu, H., Liu, J., Sun, W., and Chen, W.: Farm ponds in southern China: Challenges and solutions for conserving a neglected wetland ecosystem, Science of The Total Environment, 659, 1322-1334, https://doi.org/10.1016/j.scitotenv.2018.12.394, 2019.

Chiandet, A. S., and Xenopoulos, M. A.: Landscape and morphometric controls on water quality in stormwater management ponds, Urban Ecosystems, 19, 1645-1663, 10.1007/s11252-016-0559-8, 2016.

Clifford, C., and Heffernan, J.: Artificial Aquatic Ecosystems, Water, 10, 1096, 2018.

Cole, J. J., Pace, M. L., Carpenter, S. R., and Kitchell, J. F.: Persistence of net heterotrophy in lakes during nutrient addition and food web manipulations, Limnology and Oceanography, 45, 1718-1730, doi:10.4319/lo.2000.45.8.1718, 2000.

Conly, F. M., and van der Kamp, G.: Monitoring the Hydrology of Canadian Prairie Wetlands to Detect the Effects of Climate Change and Land Use Changes, Environmental Monitoring and Assessment, 67, 195-215, 10.1023/a:1006486607040, 2001.

Crowe, S., Katsev, S., Leslie, K., Sturm, A., Magen, C., Nomosatryo, S., Pack, M., Kessler, J., Reeburgh, W., and Roberts, J.: The methane cycle in ferruginous Lake Matano, Geobiology, 9, 61-78, 2011.

Duarte, C. M., Prairie, Y. T., Montes, C., Cole, J. J., Striegl, R., Melack, J., and Downing, J. A.: CO_2 emissions from saline lakes: A global estimate of a surprisingly large flux, Journal of Geophysical Research: Biogeosciences (2005–2012), 113, 10.1029/2007jg000637, 2008.

DelSontro, T., Beaulieu, J. J., and Downing, J. A.: Greenhouse gas emissions from lakes and impoundments: Upscaling in the face of global change, Limnology and Oceanography Letters, 3, 64-75, doi:10.1002/lol2.10073, 2018.

DelSontro, T., Boutet, L., St-Pierre, A., del Giorgio, P. A., and Prairie, Y. T.: Methane ebullition and diffusion from northern ponds and lakes regulated by the interaction between temperature and system productivity, Limnology and Oceanography, 61, S62-S77, 10.1002/lno.10335, 2016.

Devito, K. J., and Dillon, P. J.: Importance of Runoff and Winter Anoxia to the P and N Dynamics of a Beaver Pond, Canadian Journal of Fisheries and Aquatic Sciences, 50, 2222-2234, 10.1139/f93-248, 1993.

Diem, T., Koch, S., Schwarzenbach, S., Wehrli, B., and Schubert, C. J.: Greenhouse gas emissions (CO_2, CH_4, and N_2O) from several perialpine and alpine hydropower reservoirs by diffusion and loss in turbines, Aquatic Sciences, 74, 619-635, 10.1007/s00027-012-0256-5, 2012.

Dirnberger, J. M., and Weinberger, J.: Influences of lake level changes on reservoir water clarity in Allatoona Lake, Georgia, Lake and Reservoir Management, 21, 24-29, 2005.

Downing, J. A., Cole, J. J., Middelburg, J. J., Striegl, R. G., Duarte, C. M., Kortelainen, P., Prairie, Y. T., and Laube, K. A.: Sediment organic carbon burial in agriculturally eutrophic impoundments over the last century, Global Biogeochemical Cycles, 22, 10.1029/2006gb002854, 2008.

Due, N. T., Crill, P., and Bastviken, D.: Implications of temperature and sediment characteristics on methane formation and oxidation in lake sediments, Biogeochemistry, 100, 185-196, 2010.

EPA: Method 310.2: Alkalinity (Colorimetric, Automated, Methyl Orange) by Autoanalyzer. Agency, U. S. E. P. (Ed.), 1974.

Fairchild, G. W., and Velinsky, D. J.: Effects of Small Ponds on Stream Water Chemistry, Lake and Reservoir Management, 22, 321-330, 10.1080/07438140609354366, 2006.

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

Finlay, K., Vogt, R. J., Bogard, M. J., Wissel, B., Tutolo, B. M., Simpson, G. L., and Leavitt, P. R.: Decrease in CO2 efflux from northern hardwater lakes with increasing atmospheric warming, Nature, 519, 215-218, 10.1038/nature14172, 2015.

Gan, T. Y.: Reducing Vulnerability of Water Resources of Canadian Prairies to Potential Droughts and Possible Climatic Warming, Water Resources Management, 14, 111-135, 10.1023/a:1008195827031, 2000.

Gatland, J. R., Santos, I. R., Maher, D. T., Duncan, T. M., and Erler, D. V.: Carbon dioxide and methane emissions from an artificially drained coastal wetland during a flood: Implications for wetland global warming potential, Journal of Geophysical Research: Biogeosciences, 119, 1698-1716, 10.1002/2013jg002544, 2014.

Gibson, J., Vincent, W., and Pienitz, R.: Hydrologic control and diurnal photobleaching of CDOM in a subarctic lake, Archiv für Hydrobiologie, 152, 143-159, 2001.

Gilbert, P. J., Taylor, S., Cooke, D. A., Deary, M., Cooke, M., and Jeffries, M. J.: Variations in sediment organic carbon among different types of small natural ponds along Druridge Bay, Northumberland, UK, Inland Waters, 4, 57-64, 10.5268/IW-4.1.618, 2014.

Glaz, P., Bartosiewicz, M., Laurion, I., Reichwaldt, E. S., Maranger, R., and Ghadouani, A.: Greenhouse gas emissions from waste stabilisation ponds in Western Australia and Quebec (Canada), Water Research, 101, 64-74, https://doi.org/10.1016/j.watres.2016.05.060, 2016.

Goldhaber, M. B., Mills, C. T., Morrison, J. M., Stricker, C. A., Mushet, D. M., and LaBaugh, J. W.: Hydrogeochemistry of prairie pothole region wetlands: Role of long-term critical zone processes, Chemical Geology, 387, 170-183, https://doi.org/10.1016/j.chemgeo.2014.08.023, 2014.

Grinham, A., Albert, S., Deering, N., Dunbabin, M., Bastviken, D., Sherman, B., Lovelock, C. E., and Evans, C. D.: The importance of small artificial water bodies as sources of methane emissions in Queensland, Australia, Hydrol. Earth Syst. Sci., 22, 5281-5298, 10.5194/hess-22-5281-2018, 2018a.

Grinham, A., Dunbabin, M., and Albert, S.: Importance of sediment organic matter to methane ebullition in a sub-tropical freshwater reservoir, Science of The Total Environment, 621, 1199-1207, https://doi.org/10.1016/j.scitotenv.2017.10.108, 2018b.

Hamilton, J. D., Kelly, C. A., Rudd, J. W. M., Hesslein, R. H., and Roulet, N. T.: Flux to the atmosphere of CH_4 and CO_2 from wetland ponds on the Hudson Bay lowlands (HBLs), Journal of Geophysical Research: Atmospheres (1984–2012), 99, 1495-1510, 10.1029/93JD03020, 1994.

Holgerson, M. A.: Drivers of carbon dioxide and methane supersaturation in small, temporary ponds, Biogeochemistry, 124, 305-318, 10.1007/s10533-015-0099-y, 2015.

Holgerson, M. A., and Raymond, P. A.: Large contribution to inland water CO_2 and CH_4 emissions from very small ponds, 9, 222, 10.1038/ngeo2654 https://www.nature.com/articles/ngeo2654#supplementary-information, 2016.

Huotari, J., Ojala, A., Peltomaa, E., Pumpanen, J., Hari, P., and Vesala, T.: Temporal variations in surface water CO_2 concentration in a boreal humic lake based on high-frequency measurements, 2009.

Huttunen, J. T., Alm, J., Liikanen, A., Juutinen, S., Larmola, T., Hammar, T., Silvola, J., and Martikainen, P. J.: Fluxes of methane, carbon dioxide and nitrous oxide in boreal lakes and potential anthropogenic effects on the aquatic greenhouse gas emissions, Chemosphere, 52, 609-621, https://doi.org/10.1016/S0045-6535(03)00243-1, 2003.

IPCC: 2019 Refinement to the 2006 Guidelines for National Greenhouse Gas Inventories, *Chapter 7; Wetlands*, https://www.ipcc-nggip.iges.or.jp/public/2019rf/index.html, 2019

Jasechko, S., Wassenaar, L. I., and Mayer, B.: Isotopic evidence for widespread cold-season-biased groundwater recharge and young streamflow across central Canada, Hydrological Processes, 31, 2196-2209, 10.1002/hyp.11175, 2017.

Joyce, J., and Jewell, P. W.: Physical controls on methane ebullition from reservoirs and lakes, Environmental & Engineering Geoscience, 9, 167-178, 2003.

Junger, P. C., Dantas, F. d. C. C., Nobre, R. L. G., Kosten, S., Venticinque, E. M., Araújo, F. d. C., Sarmento, H., Angelini, R., Terra, I., Gaudêncio, A., They, N. H., Becker, V., Cabral, C. R., Quesado, L., Carneiro, L. S., Caliman, A., and Amado, A. M.: Effects of seasonality, trophic state and landscape properties on CO_2 saturation in low-latitude lakes and reservoirs, Science of The Total Environment, 664, 283-295, https://doi.org/10.1016/j.scitotenv.2019.01.273, 2019.

Kankaala, P., Huotari, J., Tulonen, T., and Ojala, A.: Lake-size dependent physical forcing drives carbon dioxide and methane effluxes from lakes in a boreal landscape, Limnology and Oceanography, 58, 1915-1930, 10.4319/lo.2013.58.6.1915, 2013.

Kelley, W., and Brown, S.: Principles governing the reclamation of alkali soils, Hilgardia, 8, 149-177, 1934.

Kerekes, J.: The index of lake basin permanence, Internationale Revue der Gesamten Hydrobiologie und Hydrographie, 62, 291-293, 1977.

Kuhn, M., Lundin, E. J., Giesler, R., Johansson, M., and Karlsson, J.: Emissions from thaw ponds largely offset the carbon sink of northern permafrost wetlands, Scientific Reports, 8, 9535, 10.1038/s41598-018-27770-x, 2018.

Last, W. M., and Ginn, F. M.: Saline systems of the Great Plains of western Canada: an overview of the limnogeology and paleolimnology, Saline systems, 1, 10, 2005.

Lehner, B., Liermann, C. R., Revenga, C., Vörösmarty, C., Fekete, B., Crouzet, P., Döll, P., Endejan, M., Frenken, K., Magome, J., Nilsson, C., Robertson, J. C., Rödel, R., Sindorf, N., and Wisser, D.: High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management, Frontiers in Ecology and the Environment, 9, 494-502, 10.1890/100125, 2011.

Liu, W., Li, X., Wang, Z., Wang, H., Liu, H., Zhang, B., and Zhang, H.: Carbon isotope and environmental changes in lakes in arid Northwest China, Science China Earth Sciences, 1-14, 2018.

Lorke, A., Bodmer, P., Noss, C., Alshboul, Z., Koschorreck, M., Somlai-Haase, C., Bastviken, D., Flury, S., McGinnis, D. F., and Maeck, A.: Technical note: drifting versus anchored flux chambers for measuring greenhouse gas emissions from running waters, 2015.

Lowe, L., Nathan, R., and Morden, R.: Assessing the impact of farm dams on streamflows, Part II: Regional characterisation, Australasian Journal of Water Resources, 9, 13-26, 2005.

Macpherson, G. L.: CO_2 distribution in groundwater and the impact of groundwater extraction on the global C cycle, Chemical Geology, 264, 328-336, http://dx.doi.org/10.1016/j.chemgeo.2009.03.018, 2009.

Macrae, M. L., Bello, R. L., and Molot, L. A.: Long-term carbon storage and hydrological control of CO_2 exchange in tundra ponds in the Hudson Bay Lowland, Hydrological Processes, 18, 2051-2069, doi:10.1002/hyp.1461, 2004.

Magnuson, J. J., Kratz, T. K., and Benson, B. J.: Long-term dynamics of lakes in the landscape: long-term ecological research on north temperate lakes, Oxford University Press on Demand, 2006.

Mantel, S. K., Rivers-Moore, N., and Ramulifho, P.: Small dams need consideration in riverscape conservation assessments, Aquatic Conservation: Marine and Freshwater Ecosystems, 27, 748-754, 2017.

Marcé, R., Obrador, B., Morguí, J.-A., Lluís Riera, J., López, P., and Armengol, J.: Carbonate weathering as a driver of CO_2 supersaturation in lakes, Nature Geoscience, 8, 107, 10.1038/ngeo2341 https://www.nature.com/articles/ngeo2341#supplementary-information, 2015.

MDBA: Mapping the growth, location, surface area and age of man made water bodies, including farm dams, in the Murray-Darling Basin, Murray-Darling Basin Commission, Canberra. MDBC Publication, 2008.

Miller, J. J., Acton, D. F., and St. Arnaud, R. J.: The effect of groundwater on soil formation in a morainal landscape in Saskatchewan, Canadian Journal of Soil Science, 65, 293-307, 10.4141/cjss85-033, 1985.

Müller, B., Meyer, J. S., and Gächter, R.: Alkalinity regulation in calcium carbonate-buffered lakes, Limnology and Oceanography, 61, 341-352, 2016.

Myhre, G., Shindell, D., Bréon, F. M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J. F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., and Zhang, H.: Anthropogenic and natural radiative forcing. Stocker, T. F., Qin, D., Plattner, G. K., Tignor, M. M. B., Allen, S. K., Boschung, J., Nauels, A., Xia, Y., Bex, V., and Midgley, P. M. (Eds.), Cambridge University Press, Cambridge, UK, 2013.

Natchimuthu, S., Panneer Selvam, B., and Bastviken, D.: Influence of weather variables on methane and carbon dioxide flux from a shallow pond, Biogeochemistry, 119, 403-413, 10.1007/s10533-014-9976-z, 2014.

Neal, C., House, W. A., and Down, K.: An assessment of excess carbon dioxide partial pressures in natural waters based on pH and alkalinity measurements, Science of The Total Environment, 210-211, 173-185, https://doi.org/10.1016/S0048-9697(98)00011-4, 1998.
Neubauer, S. C., and Megonigal, J. P.: Moving Beyond Global Warming Potentials to Quantify the Climatic Role of Ecosystems, Ecosystems, 1-14, 2015.
Nitzsche, K. N., Kalettka, T., Premke, K., Lischeid, G., Gessler, A., and Kayler, Z. E.: Land-use and hydroperiod affect kettle hole sediment carbon and nitrogen biogeochemistry, Science of the Total Environment, 574, 46-56, 2017.
Novikmec, M., Hamerlík, L., Kočický, D., Hrivnák, R., Kochjarová, J., Oťaheľová, H., Paľove-Balang, P., and Svitok, M.: Ponds and their catchments: size relationships and influence of land use across multiple spatial scales, Hydrobiologia, 774, 155-166, 10.1007/s10750-015-2514-8, 2016.
Ollivier, Q. R., Maher, D. T., Pitfield, C., and Macreadie, P. I.: Punching above their weight: Large release of greenhouse gases from small agricultural dams, Global Change Biology, 25, 721-732, doi:10.1111/gcb.14477, 2019.
Pacheco, F. S., Roland, F., and Downing, J. A.: Eutrophication reverses whole-lake carbon budgets, Inland Waters, 4, 41-48, 10.5268/IW-4.1.614, 2014.
Panneer Selvam, B., Natchimuthu, S., Arunachalam, L., and Bastviken, D.: Methane and carbon dioxide emissions from inland waters in India-Implications for large scale greenhouse gas balances, Global change biology, 2014.
Patoine, A., Graham, M. D., and Leavitt, P. R.: Spatial variation of nitrogen fixation in lakes of the northern Great Plains, Limnology and Oceanography, 51, 1665-1677, 10.4319/lo.2006.51.4.1665, 2006.
Peacock, M., Audet, J., Jordan, S., Smeds, J., and Wallin, M. B.: Greenhouse gas emissions from urban ponds are driven by nutrient status and hydrology, Ecosphere, 10, e02643, 10.1002/ecs2.2643, 2019.
Pennock, D., Yates, T., Bedard-Haughn, A., Phipps, K., Farrell, R., and McDougal, R.: Landscape controls on N_2O and CH_4 emissions from freshwater mineral soil wetlands of the Canadian Prairie Pothole region, Geoderma, 155, 308-319, http://dx.doi.org/10.1016/j.geoderma.2009.12.015, 2010.
Perkins, A. K., Santos, I. R., Sadat-Noori, M., Gatland, J. R., and Maher, D. T.: Groundwater seepage as a driver of CO_2 evasion in a coastal lake (Lake Ainsworth, NSW, Australia), Environmental Earth Sciences, 74, 779-792, 2015.
Pham, S. V., Leavitt, P. R., McGowan, S., Wissel, B., and Wassenaar, L. I.: Spatial and temporal variability of prairie lake hydrology as revealed using stable isotopes of hydrogen and oxygen, Limnology and Oceanography, 54, 101-118, doi:10.4319/lo.2009.54.1.0101, 2009.
Premke, K., Attermeyer, K., Augustin, J., Cabezas, A., Casper, P., Deumlich, D., Gelbrecht, J., Gerke, H. H., Gessler, A., Grossart, H. P., Hilt, S., Hupfer, M., Kalettka, T., Kayler, Z., Lischeid, G., Sommer, M., and Zak, D.: The importance of landscape diversity for carbon fluxes at the landscape level: small-scale heterogeneity matters, Wiley Interdisciplinary Reviews: Water, 3, 601-617, 10.1002/wat2.1147, 2016.
Psenner, R., and Catalan, J.: Chemical composition of lakes in crystalline basins: a combination of atmospheric deposition, geologic background, biological activity and human action, 1994.
R Core Team. R: A language and environment for statistical computing. (R Foundation for Statistical Computing, Vienna, Austria, 2018).
Read, J. S., Hamilton, D. P., Desai, A. R., Rose, K. C., MacIntyre, S., Lenters, J. D., Smyth, R. L., Hanson, P. C., Cole, J. J., Staehr, P. A., Rusak, J. A., Pierson, D. C., Brookes, J. D., Laas, A., and Wu, C. H.: Lake-size dependency of wind shear and convection as controls on gas exchange, Geophysical Research Letters, 39, 10.1029/2012GL051886, 2012.
Reverey, F., Grossart, H.-P., Premke, K., and Lischeid, G.: Carbon and nutrient cycling in kettle hole sediments depending on hydrological dynamics: a review, Hydrobiologia, 775, 1-20, 2016.
Robertson, G. P., Paul, E. A., and Harwood, R. R.: Greenhouse Gases in Intensive Agriculture: Contributions of Individual Gases to the Radiative Forcing of the Atmosphere, Science, 289, 1922-1925, 10.1126/science.289.5486.1922, 2000.
Rose, K. C., Williamson, C. E., Kissman, C. E. H., and Saros, J. E.: Does allochthony in lakes change across an elevation gradient?, Ecology, 96, 3281-3291, 2015.
Smith, S. V., Renwick, W. H., Bartley, J. D., and Buddemeier, R. W.: Distribution and significance of small, artificial water bodies across the United States landscape, Science of The Total Environment, 299, 21-36, https://doi.org/10.1016/S0048-9697(02)00222-X, 2002.
Stets, E. G., Butman, D., McDonald, C. P., Stackpoole, S. M., DeGrandpre, M. D., and Striegl, R. G.: Carbonate buffering and metabolic controls on carbon dioxide in rivers, Global Biogeochemical Cycles, 31, 663-677, 10.1002/2016GB005578, 2017.
Stumm, W., and Morgan, J. J.: Aquatic chemistry; an introduction emphasizing chemical equilibria in natural waters, 1970.
Talling, J. F.: pH, the CO_2 System and Freshwater Science, 2, BIOONE, 133-146, 114 pp., 2010.
Taylor, S., Gilbert, P. J., Cooke, D. A., Deary, M. E., and Jeffries, M. J.: High carbon burial rates by small ponds in the landscape, Frontiers in Ecology and the Environment, 17, 25-31, doi:10.1002/fee.1988, 2019.
Torgersen, T., and Branco, B.: Carbon and oxygen fluxes from a small pond to the atmosphere: Temporal variability and the CO_2/O_2 imbalance, Water Resources Research, 44, doi:10.1029/2006WR005634, 2008.

Vachon, D., Prairie, Y. T., Guillemette, F., and del Giorgio, P. A.: Modeling Allochthonous Dissolved Organic Carbon Mineralization Under Variable Hydrologic Regimes in Boreal Lakes, Ecosystems, 20, 781-795, 10.1007/s10021-016-0057-0, 2017.

van Bergen, T. J. H. M., Barros, N., Mendonça, R., Aben, R. C. H., Althuizen, I. H. J., Huszar, V., Lamers, L. P. M., Lürling, M., Roland, F., and Kosten, S.: Seasonal and diel variation in greenhouse gas emissions from an urban pond and its major drivers, Limnology and Oceanography, 0, 10.1002/lno.11173,

van der Kamp, G., and Hayashi, M.: Groundwater-wetland ecosystem interaction in the semiarid glaciated plains of North America, Hydrogeology Journal, 17, 203-214, 10.1007/s10040-008-0367-1, 2009.

Verpoorter, C., Kutser, T., Seekell, D. A., and Tranvik, L. J.: A global inventory of lakes based on high-resolution satellite imagery, Geophysical Research Letters, 41, 6396-6402, doi:10.1002/2014GL060641, 2014.

Wang, X., He, Y., Yuan, X., Chen, H., Peng, C., Yue, J., Zhang, Q., Diao, Y., and Liu, S.: Greenhouse gases concentrations and fluxes from subtropical small reservoirs in relation with watershed urbanization, Atmospheric Environment, 154, 225-235, https://doi.org/10.1016/j.atmosenv.2017.01.047, 2017.

Webb, J. R., Hayes, N. M., Simpson, G. L., Leavitt, P. R., Baulch, H. M., and Finlay, K.: Widespread nitrous oxide undersaturation in farm waterbodies creates an unexpected greenhouse gas sink, Proceedings of the National Academy of Sciences, 201820389, 10.1073/pnas.1820389116, 2019.

Weiss, R. F.: Carbon dioxide in water and seawater: the solubility of a non-ideal gas, Marine Chemistry, 2, 203-215, http://dx.doi.org/10.1016/0304-4203(74)90015-2, 1974.

Weyhenmeyer, G. A., Kosten, S., Wallin, M. B., Tranvik, L. J., Jeppesen, E., and Roland, F.: Significant fraction of CO_2 emissions from boreal lakes derived from hydrologic inorganic carbon inputs, Nature Geoscience, 8, 933, 2015.

Whitfield, C. J., Aherne, J., and Baulch, H. M.: Controls on greenhouse gas concentrations in polymictic headwater lakes in Ireland, Science of The Total Environment, 410, 217-225, http://dx.doi.org/10.1016/j.scitotenv.2011.09.045, 2011.

Wiik, E., Haig, H. A., Hayes, N. M., Finlay, K., Simpson, G. L., Vogt, R. J., and Leavitt, P. R.: Generalized Additive Models of Climatic and Metabolic Controls of Subannual Variation in pCO_2 in Productive Hardwater Lakes, Journal of Geophysical Research: Biogeosciences, 123, 1940-1959, 10.1029/2018jg004506, 2018.

Winter, T. C., and LaBaugh, J. W.: Hydrologic considerations in defining isolated wetlands, Wetlands, 23, 532, 2003.

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

Wood, S. N., Pya, N., and Säfken, B.: Smoothing Parameter and Model Selection for General Smooth Models, Journal of the American Statistical Association, 111, 1548-1563, 10.1080/01621459.2016.1180986, 2016.

Yamamoto, S., Alcauskas, J. B., and Crozier, T. E.: Solubility of methane in distilled water and seawater, Journal of Chemical and Engineering Data, 21, 78-80, 1976.

Yi, Y., Brock, B. E., Falcone, M. D., Wolfe, B. B., and Edwards, T. W. D.: A coupled isotope tracer method to characterize input water to lakes, Journal of Hydrology, 350, 1-13, https://doi.org/10.1016/j.jhydrol.2007.11.008, 2008.

Zhang, J., Ma, K., and Fu, B.: Wetland loss under the impact of agricultural development in the Sanjiang Plain, NE China, Environmental Monitoring and Assessment, 166, 139-148, 10.1007/s10661-009-0990-x, 2010.

Tables and Figures

Table 1: Farm reservoir and landscape physical, hydrological, and chemical characteristics of the study sites (n = 101)

	Units	*N*	*Mean*	*Median*	*Min*	*Max*
Area	m^2	101	1,312	1,040	158	13,900
Depth	m	101	2.08	2.10	0.18	5.10
Buoyancy frequency	s^{-2}	99	0.01	0.005	0.00	0.03
$\delta^{18}O$ inflow	‰	101	-13.37	-13.33	-19.39	-8.40
Evaporation to inflow		101	0.46	0.43	0.04	1.58
Water residence time	Years	100	0.76	0.66	0.08	2.51
CO_2	µM	101	42.2	14.6	1.3	326.1
CH_4	µM	101	4.3	1.9	0.1	54.5
Flux CO_2						
Positive	mmol m^{-2} d^{-1}	47	100.1	58.1	0.1	466.2
Negative	mmol m^{-2} d^{-1}	54	-11.9	-13.3	-21.3	-0.1
Flux CH_4	mmol m^{-2} d^{-1}	101	7.1	3.2	0.4	91.5
k600- CO_2	m d^{-1}	15	1.50	0.98	0.20	4.12
k600- CH_4	m d^{-1}	23	1.64	1.25	0.38	4.14
Temperature	°C	101	20.1	19.9	15.7	29.5
Dissolved O_2	%	101	92.6	88.9	2.3	344.0
Salinity	ppt	101	0.9	0.5	0.1	8.6
pH		101	8.75	8.75	6.95	10.19
Chlorophyll a	µg L^{-1}	101	99.1	36.9	2.2	2,483
NH_3	µg N L^{-1}	100	354.7	100.0	10.0	5,930
NO_x	µg N L^{-1}	98	196.6	34.1	1.2	3,188
TP	µg P L^{-1}	98	285.2	80.0	8.7	6,480
TN	µg N L^{-1}	98	3,082	2,360	417.5	14,280
DOC	mg C L^{-1}	99	31.8	29.3	4.6	90.4
Sediment organic carbon	%	101	5.2	3.9	0.6	31.4
Sediment organic nitrogen	%	101	0.6	0.4	0.1	2.8
Alkalinity	mg L^{-1}	96	245.4	219.2	71.0	755.5
Soil CEC	M-eq $100g^{-1}$	98	24	24	10	180
K_{sat}	cm hr^{-1}	101	9.9	5.0	0.0	39.7
Elevation	m	101	627.6	598.0	484.0	997.0

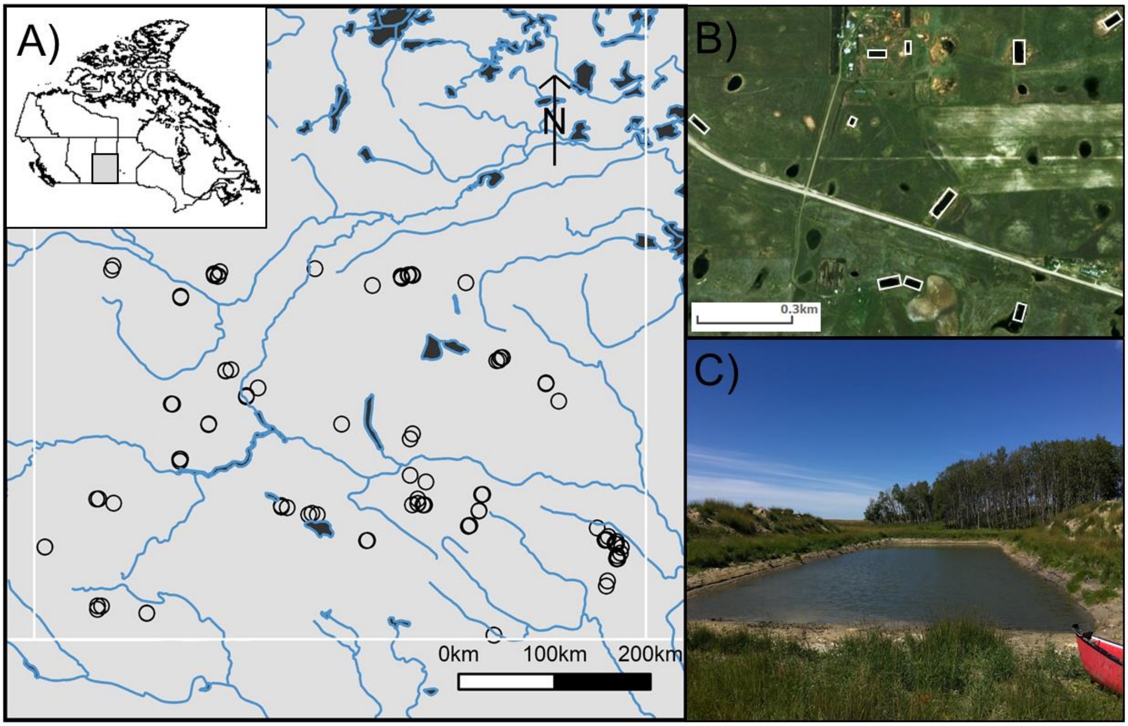

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Figure 1: A) Map of southern Saskatchewan in Canada showing the distribution of studied farm reservoirs, B) aerial image showing 10 farm reservoirs delineated by white rectangles within a 1 km^2 area, and C) general size and shape of farm reservoirs with two characteristic side mounds of excavated materials.

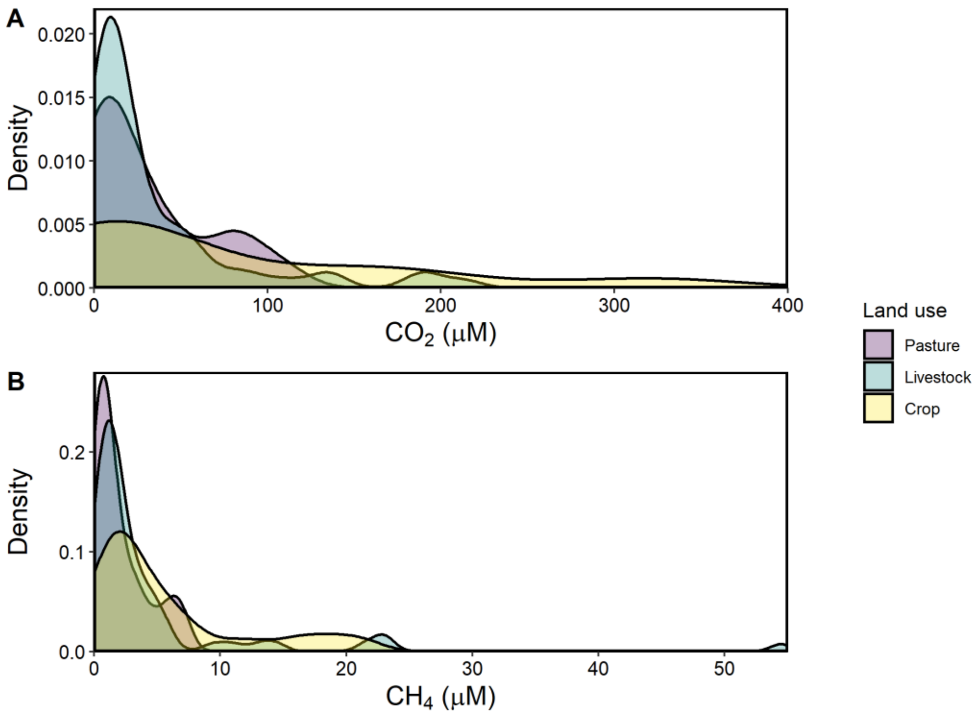

Figure 2: Kernel density estimates of CO_2 and CH_4 concentrations measured in 101 farm reservoirs grouped by land use.

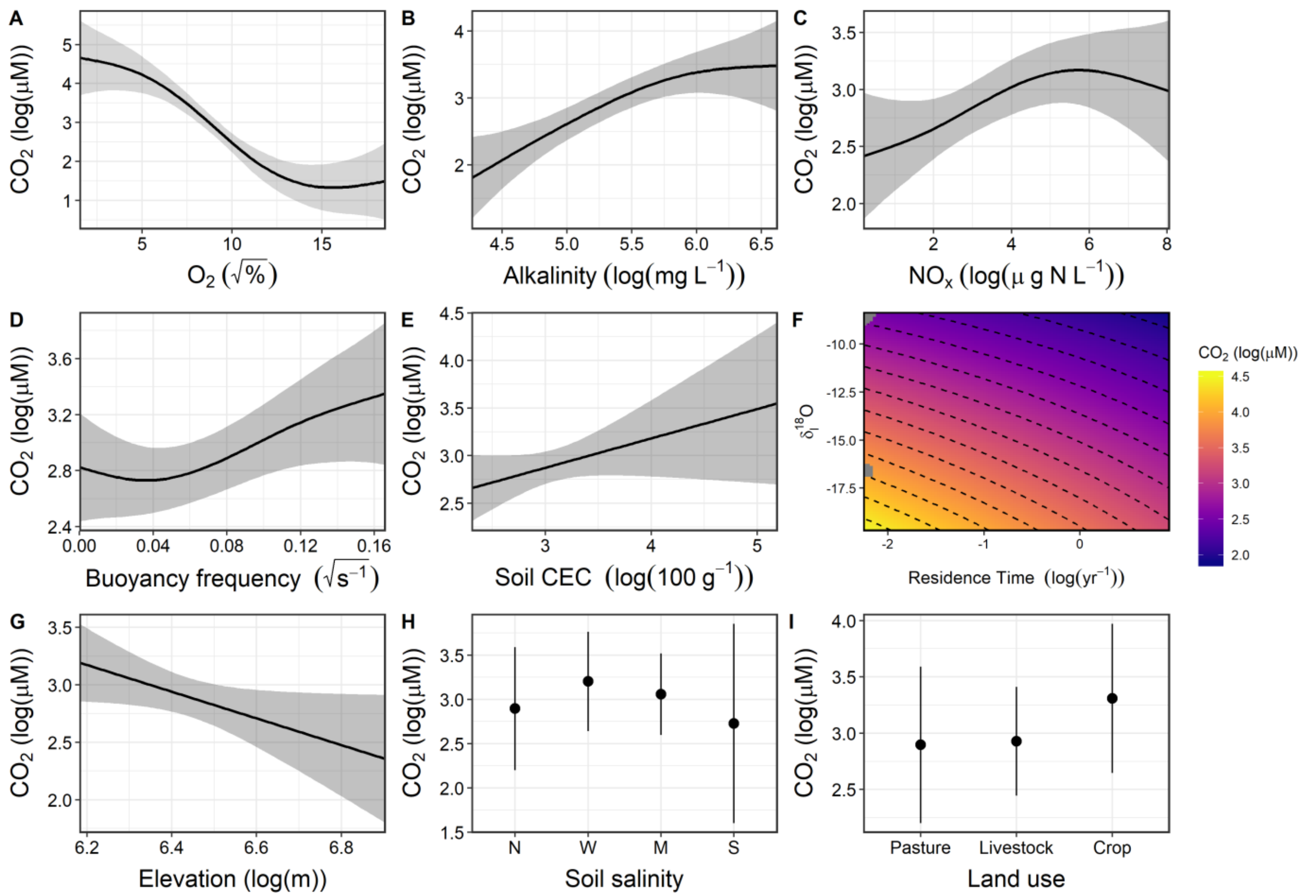

Figure 3: Response patterns farm reservoir CO_2 concentrations with abiotic, biotic, hydromorphological, and landscape variables based on GAMs. CO_2 was best estimated by a combination of a) DO saturation, b) alkalinity, c) NOx, d) buoyancy frequency, e) interaction between δ_I and WRT, f) soil CEC, g) and elevation, with soil salinity (h) and land use (I) not significant. Model deviance explained was 66.5%. The response patterns shown are the partial effect splines from the GAM (solid line) and shaded area indicated 95% credible intervals. See Table S4 and Figure S2 for summary of model statistics and model fit with observed data.

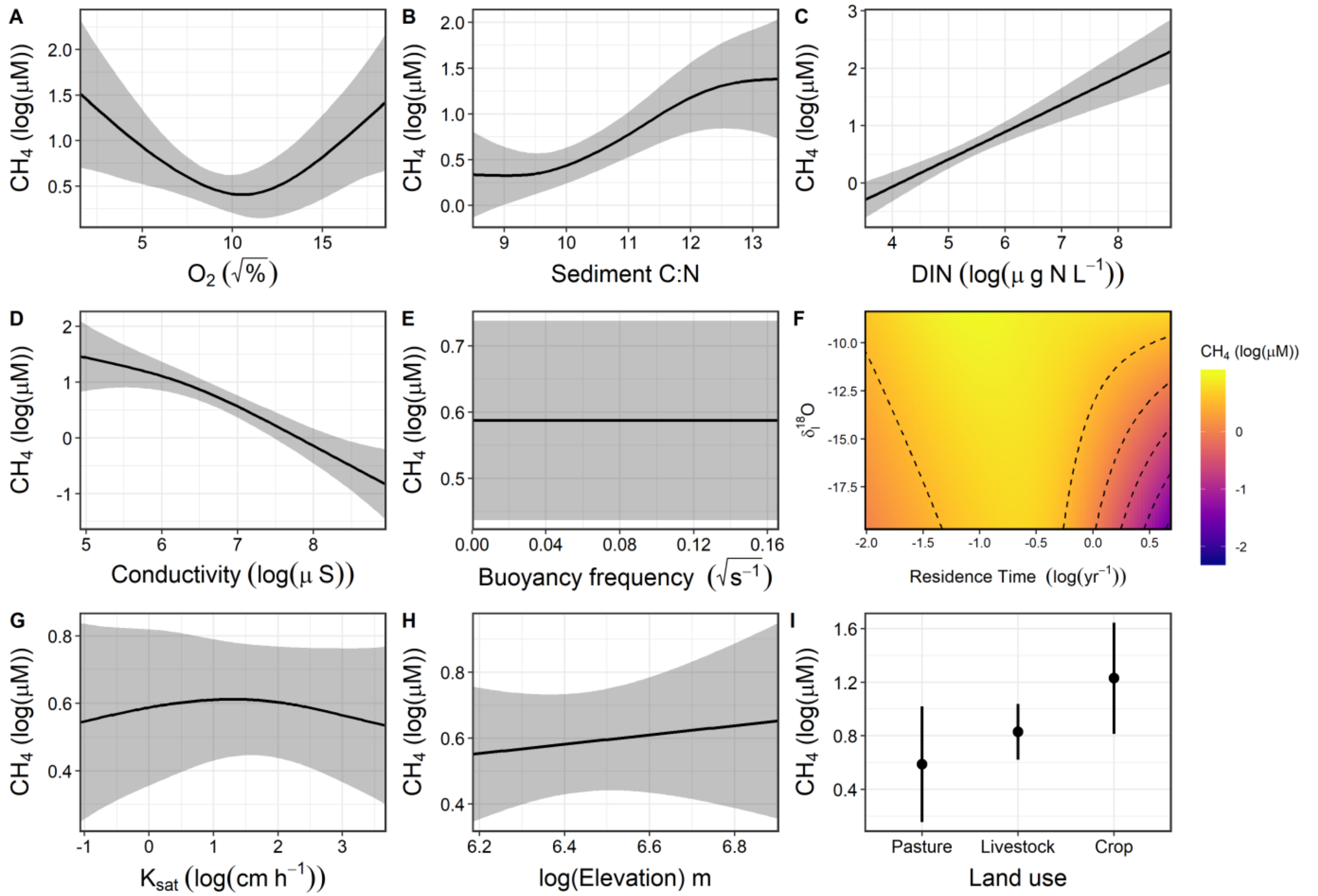

Figure 4: Response patterns farm reservoir CH_4 concentrations with abiotic, biotic, hydromorphological, and landscape variables based on generalised additive models (GAMs). CH_4 was explained by a combination of a) DO saturation, b) sediment C/N, c) DIN, d) conductivity, e) buoyancy frequency (not significant, f) interaction between δ_I and WRT, g) soil Ksat (not significant), h) elevation (not significant), and i) local land use. Model deviance explained was 74.1%. The response patterns shown are the partial effect splines from the GAM (solid line) and shaded area indicated 95% credible intervals. See Table S5 and Figure S3 for summary of model statistics and model fit with observed data.

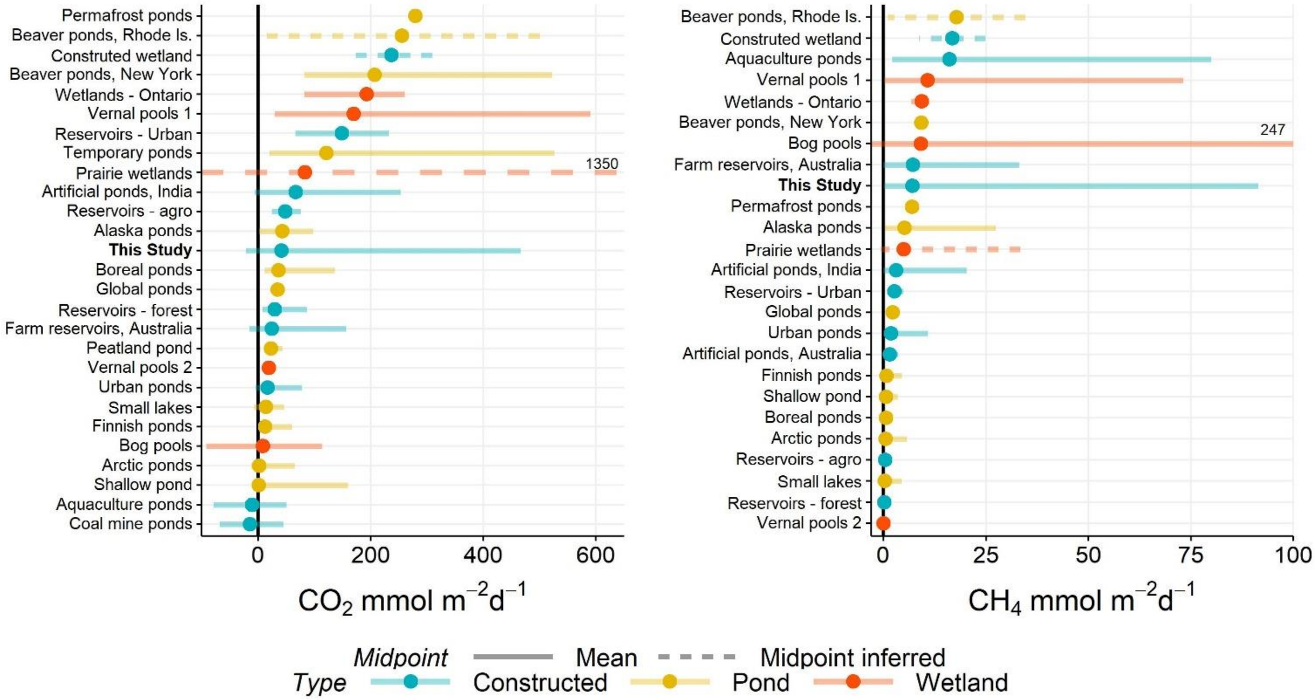

Figure 5: Range of CO_2 and CH_4 (diffusive) fluxes observed in natural and constructed small (<0.01 km^2) waterbodies, including this study (farm reservoirs). Dots represent the mean reported in each study and error bars the range. If no mean value was reported, then the midpoint was inferred as the middle of range (dashed lines). Solid black line distinguished between positive and negative fluxes. All data is from the published literature and references can be found in the Table S6.

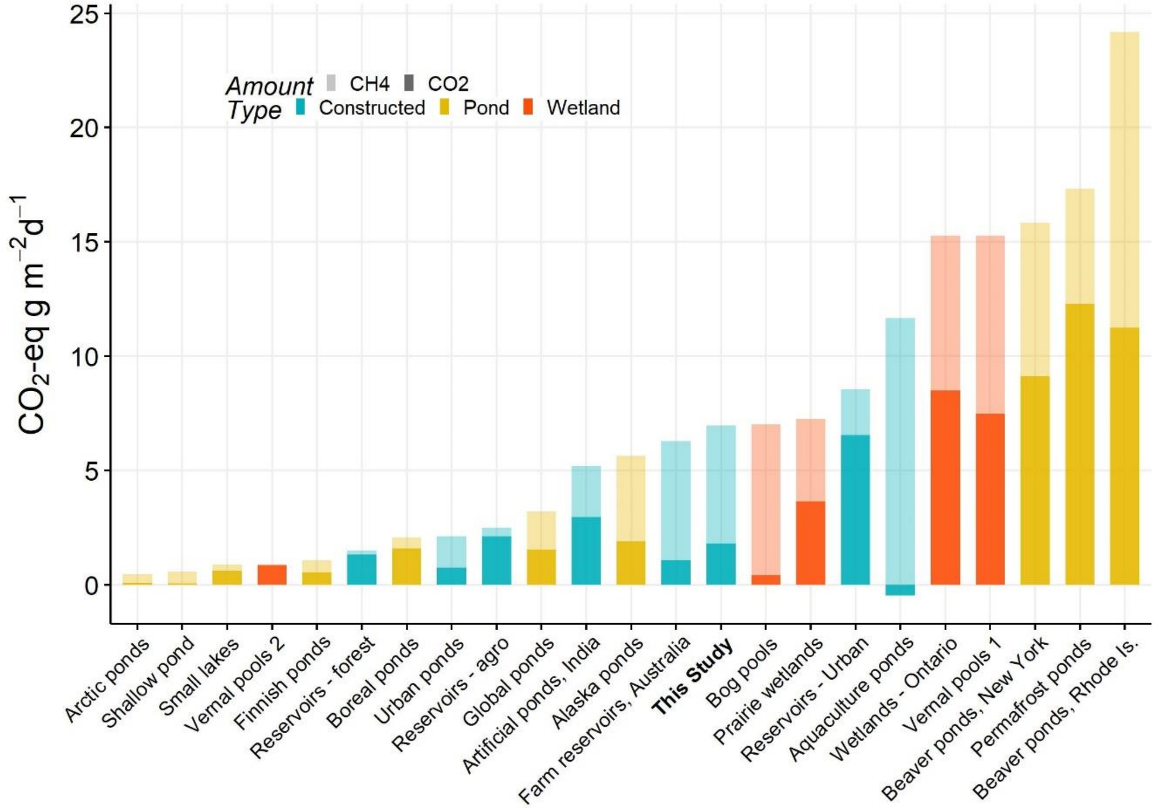

Figure 6: Total average CO_2 equivalent fluxes of CO_2 and CH_4 (diffusive) measured in natural and artificial small waterbodies (<0.01 km^2). CO_2-e fluxes were calculated based on 100 year sustained-flux global warming potentials in Neubauer and Megonigal (2015). Relative proportions of each gas are indicated by shading, and waterbody type is given by colour. All data is from the published literature and references can be found in the Table S6.