

Geographic variability in freshwater methane hydrogen isotope ratios and its implications for global isotopic source signatures

Peter M.J. Douglas¹, Emerald Stratigopoulos¹, Jenny Park¹, Dawson Phan¹

¹Earth and Planetary Sciences, McGill University, Montreal, H3A 0E8, Canada

Correspondence to: Peter M. J. Douglas (peter.douglas@mcgill.ca)

Abstract. There is growing interest in developing spatially resolved methane (CH₄) isotopic source signatures to aid in geographic source attribution of CH₄ emissions. CH₄ hydrogen isotope measurements ($\delta^2\text{H-CH}_4$) have the potential to be a powerful tool for geographic differentiation of CH₄ emissions from freshwater environments, as well as other microbial sources. This is because microbial $\delta^2\text{H-CH}_4$ values are partially dependent on the $\delta^2\text{H}$ of environmental water ($\delta^2\text{H-H}_2\text{O}$), which exhibits large and well-characterized spatial variability globally. We have refined the existing global relationship between $\delta\text{D-CH}_4$ - $\delta\text{D-H}_2\text{O}$ by compiling a more extensive global dataset of $\delta^2\text{H-CH}_4$ from freshwater environments, including wetlands, inland waters, and rice paddies, comprising a total of 129 different sites, and compared these with measurements and estimates of $\delta^2\text{H-H}_2\text{O}$, as well as $\delta^{13}\text{C-CH}_4$ and $\delta^{13}\text{C-CO}_2$ measurements. We found that estimates of $\delta^2\text{H-H}_2\text{O}$ explain approximately 42% of the observed variation in $\delta^2\text{H-CH}_4$, with a flatter slope than observed in previous studies. The inferred global $\delta^2\text{H-CH}_4$ vs $\delta^2\text{H-H}_2\text{O}$ regression relationship is not sensitive to using either modelled precipitation $\delta^2\text{H}$ or measured $\delta^2\text{H-H}_2\text{O}$ as the predictor variable. The slope of the global freshwater relationship between $\delta^2\text{H-CH}_4$ and $\delta^2\text{H-H}_2\text{O}$ is similar observations from in incubation experiments, but is different from pure culture experiments, and is consistent with previous suggestions that variation in the $\delta^2\text{H}$ of acetate controlled by environmental $\delta^2\text{H-H}_2\text{O}$ is important in determining variation in $\delta^2\text{H-CH}_4$. The relationship between $\delta^2\text{H-CH}_4$ and $\delta^2\text{H-H}_2\text{O}$ leads to significant differences in the distribution of freshwater $\delta^2\text{H-CH}_4$ between the northern high latitudes (60-90 °N), relative to other global regions. We estimate a flux-weighted global freshwater $\delta^2\text{H-CH}_4$ of $-310\pm 15\%$, which is higher than many previous estimates. Comparison of the residual variability in $\delta^2\text{H-CH}_4$ with $\delta^{13}\text{C}$ measurements of both CH₄ and CO₂ does not support a dominant role for either differential isotopic fractionation related to methanogenesis pathways or methane oxidation in controlling variation in $\delta^2\text{H-CH}_4$, but instead suggests that residual $\delta^2\text{H-CH}_4$ variation is the result of complex interactions between these and other biogeochemical variables. We observe significantly higher distribution of $\delta^2\text{H-CH}_4$ values, corrected for $\delta^2\text{H-H}_2\text{O}$, in inland waters relative to wetlands, and suggest this difference is caused by more prevalent CH₄ oxidation in inland waters. We used the expanded freshwater CH₄ isotopic dataset to calculate a bottom-up estimate of global CH₄ $\delta^2\text{H}$ and $\delta^{13}\text{C}$ sources that includes spatially resolved isotopic signatures for freshwater CH₄ sources. The bottom-up global source $\delta^2\text{H-CH}_4$ estimate is higher than a previous estimate using a similar approach, as a result of a more enriched global freshwater $\delta^2\text{H-CH}_4$ signature. However, it is in agreement with top-down estimates of global source $\delta^2\text{H-CH}_4$ based

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98 [on atmospheric measurements and estimated atmospheric sink fractionations. In contrast our bottom-up global source \$\delta^{13}\text{C}\$ -](#)
99 [CH₄ estimate is lower than top-down estimates, partly as a result of a lack of \$\delta^{13}\text{C}\$ -CH₄ data from C₄ plant dominated](#)
00 [ecosystems. In general, we find there is a particular need for more data to constrain isotopic signatures for low-latitude](#)
01 [microbial CH₄ sources.](#)

02 1 Introduction

03 Methane (CH₄) is an important greenhouse gas that accounts for approximately 25% of current anthropogenic global
04 warming, but we do not have a complete understanding of the current relative or absolute fluxes of different CH₄ sources to
05 the atmosphere (Schwietzke et al., 2016;Saunois et al., 2019), nor is there consensus on the causes of recent decadal-scale
06 changes in the rate of increase in atmospheric CH₄ (Kai et al., 2011;Pison et al., 2013;Rice et al., 2016;Schaefer et al.,
07 2016;Worden et al., 2017;Thompson et al., 2018;Turner et al., 2019). Freshwater ecosystems are an integral component of
08 the global CH₄ budget. They are one of the largest sources of atmospheric CH₄ and are [unequivocally](#) the largest natural, or
09 non-anthropogenic, source (Bastviken et al., 2011;Saunois et al., 2019). At the same time the geographic distribution of
10 freshwater CH₄ emissions, changes in the strength of this source through time, and the relative importance of wetland versus
11 inland water CH₄ emissions all remain highly uncertain (Pison et al., 2013;Schaefer et al., 2016;Ganesan et al., 2018;Saunois
12 et al., 2019;Turner et al., 2019). Gaining a better understanding of freshwater CH₄ emissions on a global scale is of great
13 importance for understanding potential future climate feedbacks related to CH₄ emissions from these ecosystems (Bastviken
14 et al., 2011;Koven et al., 2011;Yvon-Durocher et al., 2014;Zhang et al., 2017). It is also necessary in order to better constrain
15 the quantity and rate of change of other CH₄ emissions sources, including anthropogenic sources from fossil fuels,
16 agriculture, and waste (Kai et al., 2011;Pison et al., 2013;Schaefer et al., 2016).

17 Isotopic tracers, particularly $\delta^{13}\text{C}$, have proven to be very useful in constraining global CH₄ sources and sinks (Kai
18 et al., 2011;Nisbet et al., 2016;Rice et al., 2016;Schaefer et al., 2016;Schwietzke et al., 2016;Nisbet et al., 2019). However,
19 $\delta^{13}\text{C}$ source signatures cannot fully differentiate CH₄ sources, leaving residual ambiguity in source apportionment (Schaefer
20 et al., 2016;Schwietzke et al., 2016;Worden et al., 2017;Turner et al., 2019). Applying additional isotopic tracers to
21 atmospheric CH₄ monitoring has the potential to greatly improve our understanding of CH₄ sources and sinks (Saunois et al.,
22 2019;Turner et al., 2019). [Recently developed laser-based methods, including cavity ringdown spectroscopy, quantum](#)
23 [cascade laser absorption spectroscopy, and tunable infrared laser direct absorption spectroscopy](#) (Chen et al., 2016;
24 Röckmann et al., 2016; Yacovitch et al., 2020) [could greatly enhance the practicality of atmospheric \$\delta^2\text{H}\$ -CH₄ measurements](#)
25 [at greater spatial and temporal resolution, similar to recent developments for \$\delta^{13}\text{C}\$ -CH₄ measurements \(Zazzeri et al., 2015;](#)
26 [Miles et al., 2018\).](#) $\delta^2\text{H}$ -CH₄ measurements have proven useful in understanding past CH₄ sources in ice-core records
27 (Whiticar and Schaefer, 2007; Mischler et al., 2009; Bock et al., 2010; Bock et al., 2017), but have seen only limited use in
28 modern atmospheric CH₄ budgets (Kai et al., 2011; Rice et al., 2016), in part because of loosely constrained source terms, as
29 well as relatively sparse atmospheric measurements. [Atmospheric inversion models have shown that increased spatial and](#)

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35 [temporal resolution of \$\delta^2\text{H-CH}_4\$ measurements could provide substantial improvements in precision for global and regional](#)
36 [methane budgets](#) (Rigby et al., 2012).

37 $\delta^2\text{H-CH}_4$ measurements could prove especially useful in understanding freshwater CH_4 emissions. Freshwater $\delta^2\text{H-CH}_4$
38 CH_4 is thought to be highly dependent on $\delta^2\text{H-H}_2\text{O}$ (Waldron et al., 1999a; Whiticar, 1999; Chanton et al., 2006). Since $\delta^2\text{H-}$
39 H_2O exhibits large geographic variation as a function of temperature and fractional precipitation (Rozanski et al., 1993;
40 Bowen and Revenaugh, 2003), $\delta^2\text{H-CH}_4$ measurements have the potential to differentiate freshwater CH_4 sources by latitude.
41 This approach has been applied in some ice core studies (Whiticar and Schaefer, 2007; Bock et al., 2010), but geographic
42 source signals remain poorly constrained, in part because of small datasets and because of incompletely understood
43 relationships between $\delta^2\text{H-H}_2\text{O}$ and $\delta^2\text{H-CH}_4$. In contrast, recent studies of modern atmospheric $\delta^2\text{H-CH}_4$ have typically not
44 accounted for geographic variation in freshwater CH_4 sources (Kai et al., 2011; Rice et al., 2016). Relatedly, other studies
45 [have found an important role for variation in \$\delta^2\text{H-H}_2\text{O}\$ in controlling \$\delta^2\text{H-CH}_4\$ from biomass burning](#) (Umezawa et al., 2011)
46 [and from plants irradiated by UV light](#) (Vigano et al., 2010), [as well as the \$\delta^2\text{H}\$ of \$\text{H}_2\$ produced by wood combustion](#)
47 (Röckmann et al., 2016).

48 [In addition to variance caused by \$\delta^2\text{H-H}_2\text{O}\$, a number of additional biogeochemical variables have been proposed to](#)
49 [influence \$\delta^2\text{H-CH}_4\$ in freshwater environments. These include differences in the predominant biochemical pathway of](#)
50 [methanogenesis](#) (Whiticar et al., 1986; Whiticar, 1999; Chanton et al., 2006), [the extent of methane oxidation](#) (Happell et al.,
51 1994; Waldron et al., 1999a; Whiticar, 1999; Cadieux et al., 2016), [isotopic fractionation resulting from diffusive gas](#)
52 [transport](#) (Waldron et al., 1999a; Chanton, 2005), and differences in the thermodynamic favorability or enzymatic
53 reversibility of methanogenesis (Valentine et al., 2004b; Stolper et al., 2015; Douglas et al., 2016). These influences on $\delta^2\text{H-}$
54 CH_4 have the potential to complicate geographic signals, but also provide the potential to differentiate ecosystem sources if
55 [specific ecosystems are characterized by differing rates and pathways of methanogenesis, rates of \$\text{CH}_4\$ oxidation, or gas](#)
56 [transport processes](#). A recent study proposed that freshwater $\delta^{13}\text{C-CH}_4$ could be differentiated geographically based on
57 ecosystem differences [in the prevalence of different methanogenic pathways and in the predominance of \$\text{C}_4\$ plants](#), in
58 addition to the geographic distribution of wetland ecosystems (Ganesan et al., 2018). $\delta^2\text{H-CH}_4$ measurements have the
59 potential to complement this approach by providing an additional isotopic parameter for [differentiating ecosystem and](#)
60 [geographic \$\text{CH}_4\$ source signatures](#).

61 In order to use $\delta^2\text{H-CH}_4$ as an indicator of freshwater ecosystem contributions to global and regional CH_4 emissions
62 budgets, a clearer understanding of freshwater $\delta^2\text{H}$ source signals, and how they vary by geographic location, ecosystem
63 type, and other variables is needed. In order to address this need we have assembled and analyzed a dataset of [897 \$\delta^2\text{H-CH}_4\$](#)
64 [measurements from 129 individual ecosystems, or sites, derived from 40 publications](#) (Schoell, 1983; Woltemate et al.,
65 1984; Burke Jr and Sackett, 1986; Whiticar et al., 1986; Burke Jr et al., 1988; Burke Jr, 1992; Burke Jr et al., 1992
66 ;Lansdown et al., 1992; Lansdown, 1992; Martens et al., 1992; Wassmann et al., 1992; Happell et al., 1993; Levin et al.,
67 1993; Happell et al., 1994; Wahlen, 1994; Bergamaschi, 1997; Chanton et al., 1997; Hornibrook et al., 1997; Tyler et al.,

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1997; Zimov et al., 1997; Bellisario et al., 1999; Popp et al., 1999; Waldron et al., 1999b; Chasar et al., 2000; Marik et al., 2002; Nakagawa et al., 2002b; Nakagawa et al., 2002a; Chanton et al., 2006; Walter et al., 2006; Walter et al., 2008; Alstad and Whiticar, 2011; Brosius et al., 2012; Sakagami et al., 2012; Bouchard et al., 2015; Stolper et al., 2015; Wang et al., 2015; Cadieux et al., 2016; Douglas et al., 2016; Thompson et al., 2016; Lecher et al., 2017). We have advanced existing datasets of freshwater $\delta^2\text{H-CH}_4$ (Whiticar et al., 1986; Waldron et al., 1999a; Sherwood et al., 2017) in the following key attributes: 1) compiling a significantly larger dataset than was previously available; 2) compiling paired $\delta^{13}\text{C-CH}_4$ data for all sites, $\delta^{13}\text{C-CO}_2$ data for 50% of sites, and $\delta^2\text{H-H}_2\text{O}$ data for 47% of sites; 3) compiling geographic coordinates for all sites, providing the ability to perform spatial analyses and compare with gridded datasets of precipitation isotopic composition; and 4) classifying all sites by ecosystem and sample type (dissolved vs. gas samples), allowing for a clearer differentiation of how these variables influence $\delta^2\text{H-CH}_4$.

Using this data set we applied statistical analyses to address key questions surrounding the global distribution of freshwater $\delta^2\text{H-CH}_4$, the variables that control this distribution, and its implications for atmospheric $\delta^2\text{H-CH}_4$. Specifically, we investigated the nature of the global dependence of $\delta^2\text{H-CH}_4$ on $\delta^2\text{H-H}_2\text{O}$, and whether this relationship results in significant differences in freshwater $\delta^2\text{H-CH}_4$ by latitude. We also assessed whether variability in $\delta^{13}\text{C-CH}_4$, $\delta^{13}\text{C-CO}_2$, and α_c , was correlated with $\delta^2\text{H-CH}_4$, and whether there are significant differences in $\delta^2\text{H-CH}_4$ between different ecosystem and sample types. Finally, we used our dataset, combined with other isotopic datasets (Sherwood et al., 2017) and flux estimates (Saunois et al., 2020), to estimate the global $\delta^2\text{H-CH}_4$ and $\delta^{13}\text{C-CH}_4$ of global emissions sources, and compared this with previous estimates based on atmospheric measurements or isotopic datasets (Whiticar and Schaefer, 2007; Rice et al., 2016; Sherwood et al., 2017).

2 Methods

2.1 Isotope Nomenclature

The isotope notation used in this study is briefly introduced here. Hydrogen and carbon isotope ratios are primarily discussed as delta values, using the generalized formula (Coplen, 2011):

$$\delta = \frac{(R_{\text{sample}} - R_{\text{standard}})}{R_{\text{standard}}} \quad (1)$$

where R is the ratio of the heavy isotope to the light isotope, and the standard is Vienna Standard Mean Ocean Water (VSMOW) for $\delta^2\text{H}$ and Vienna Pee Dee Belemnite (VPDB) for $\delta^{13}\text{C}$. δ values are expressed in per mil (‰) notation.

We also refer to the isotopic fractionation factor between two phases, or α , which is defined as:

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Deleted: 1) it is significantly larger than previous datasets, with 83 additional sites relative to Waldron et al., (1999); 2) paired $\delta^{13}\text{C-CH}_4$ data is available for all sites, $\delta^{13}\text{C-CO}_2$ data is available for 50% of sites, and $\delta^2\text{H-H}_2\text{O}$ data is available for 47% of sites; 3) all sites are geo-located, providing the potential to perform spatial analyses and compare with gridded isotopic datasets of precipitation isotopic composition; and 4) we classify all sites by ecosystem and sample type (dissolved vs. gas samples), allowing for a clearer differentiation of how these variables influence $\delta^2\text{H-CH}_4$.

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$$\alpha_{a-b} = \frac{R_a}{R_b} = \frac{\delta_a + 1}{\delta_b + 1} \quad (2)$$

Specifically, we discuss the carbon isotope fractionation factor between CO₂ and CH₄ (α_C) and the hydrogen isotope fractionation factor between H₂O and CH₄ (α_H).

2.2 Dataset Compilation

2.2.1 Literature Survey

To identify datasets we used a set of search terms (methane OR CH₄ AND freshwater OR wetland OR peatland OR swamp OR marsh OR lake OR pond OR 'inland water' AND 'hydrogen isotope' OR 'δD' OR 'δ²H') in Google Scholar to find published papers that discussed this measurement. We also identified original publications using previously compiled datasets (Waldron et al., 1999a; Sherwood et al., 2017). Data for 90% of sites were from peer-reviewed publications. Data from 13 sites were from a [Ph.D. dissertation](#) (Lansdown, 1992).

2.2.2 Dataset structure

Most samples were associated with geographic coordinates in data tables or text documentation, or with specific geographic locations such as the name of a town or city. In a few cases we identified approximate geographic locations based on text descriptions of sampling sites, with the aid of Google Earth software. Sampling sites were defined as individual water bodies or wetlands as identified in the relevant study. In some cases where a number of small ponds were sampled from the same location, we grouped ponds of a given type as a single site (Bouchard et al., 2015). [We divided sampling sites into six ecosystem categories: 1\) lakes and ponds \(hereafter lakes\), 2\) rivers and floodplains \(hereafter rivers\), 3\) bogs, 4\) fens, 5\) swamps and marshes, and 6\) rice paddies. Most data \(7 of 8 sites\) in the rivers category are from floodplain lake or delta environments.](#) Swamps and marshes were combined as one category because of a small number of sites, and because there is no clear indication of biogeochemical differences between these ecosystems. To make these categorizations we relied on site descriptions in the data source publications. We also analyzed data in two larger environment types, inland waters (lakes and rivers) and wetlands (bogs, fens, swamps and marshes, and rice paddies), [which correspond to two flux categories \(freshwaters and natural wetlands\) documented by](#) Saunois et al. (2020). While rice paddies are an anthropogenic ecosystem, they are wetlands where microbial methanogenesis occurs under generally similar conditions to natural wetlands, and therefore we included them [as wetlands](#) in our analysis. In some cases the type of wetland was not specified. We did not differentiate between ombrotrophic and minerotrophic peatlands since most publications did not specify this difference, although it has been inferred to be important for δ¹³C-CH₄ distributions (Hornibrook, 2009). [For studies of bogs and fens that sampled by soil depth we have only included sample measurements from the upper 50 cm. This is based on the observation](#)

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11 of large-scale isotopic variability with soil depth in these ecosystems (Hornibrook et al., 1997;Waldron et al., 1999b), and
12 the observation that shallow peat is typically the dominant source of atmospheric emissions (Waldron et al., 1999b;Bowes
13 and Hornibrook, 2006;Shoemaker et al., 2012), which is our primary focus in this study. Other wetland ecosystems were not
14 sampled by soil depth.

15 We also categorized samples by the form in which CH₄ was sampled, differentiating between dissolved CH₄ and
16 CH₄ emitted through diffusive fluxes, which we group as dissolved CH₄, and gas-phase samples, including bubbles sampled
17 either by disturbing sediments or by collecting natural ebullition fluxes. In some cases the sampling method or type of
18 sample was not specified, or samples were a mix of both categories, which we did not attempt to differentiate.

19 Where possible (78% of sites), δ²H-CH₄ and δ¹³C-CH₄ values, as well as δ¹³C-CO₂ and δ²H-H₂O, were gathered
20 from data files or published tables. In a number of publications, representing 22% of sites, data were only available
21 graphically. For these studies we used Webplot Digitizer (<https://automeris.io/WebPlotDigitizer/>) software to extract data for
22 these parameters. Previous studies have shown that user errors from Webplot Digitizer are typically small, with 90% of user
23 generated data within 1% of the actual value (Drevon et al., 2017). Based on this, we estimate a typical error for δ²H-CH₄
24 data of less than 3%. Studies where data were derived from graphs are identified in Supplementary Table S1 (Douglas et al.,
25 2020).

26 2.2.3 Estimates of δ²H-H₂O and its effects on δ²H-CH₄

27 To estimate δ²H-H₂O for sites where it was not measured we relied on estimates of the isotopic composition of
28 precipitation (δ²H_p), derived the Online Isotopes in Precipitation Calculator v.3.1.1 (OIPC3.1; www.waterisotopes.org; Bowen
29 and Wilkinson, 2002; Bowen and Revenaugh, 2003; Bowen et al., 2005). Inputs for δ²H_p estimates are latitude, longitude,
30 and elevation. We estimated elevation for each site surface elevation at the site's geographic coordinates reported by Google
31 Earth. We tabulated estimates of both annual precipitation-amount weighted δ²H_p, and growing season precipitation-amount
32 weighted δ²H_p, where the growing season is defined as months with a mean temperature greater than 0 °C. We then analysed
33 whether annual or growing season δ²H_p is a better estimate of environmental δ²H-H₂O for both wetlands and inland waters
34 by comparing these values with measured δ²H-H₂O for sites with measurements (See Sect. 3.2).

35 Based on this analysis, we then identified a 'best-estimate' δ²H-H₂O value for each site, using an approach similar to that of
36 Waldron et al. (1999a). Namely, we apply measured δ²H-H₂O where available, and estimates based on the regression
37 analyses detailed in Section 3.2 for sites without measurements.

38 To account for the effects of δ²H-H₂O on δ²H-CH₄, we introduce the term δ²H-CH_{4,w0}, which is the estimated δ²H-
39 CH₄ of a sample if it had formed in an environment where δ²H-H₂O = 0‰. This is defined by the equation:

$$40 \delta^2\text{H-CH}_{4,w0} = \delta^2\text{H-CH}_4 - (b \times \delta^2\text{H}_2\text{O}) \quad (3)$$

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56 where $\delta^2\text{H-H}_2\text{O}$ is the 'best-estimate' value for each site described above, b is the slope of the regression relationship of
57 $\delta^2\text{H-H}_2\text{O}$ vs. $\delta^2\text{H-CH}_4$ for the entire dataset, as reported in Sect. 3.3. We also performed the same calculation separately for
58 the subset of sites with measured $\delta^2\text{H-H}_2\text{O}$. We analyze $\delta^2\text{H-CH}_4$ instead of α_H because, as discussed in Sect. 3.3.1, the
59 global relationship between $\delta^2\text{H}_p$ vs. $\delta^2\text{H-CH}_4$ does not correspond to a constant value of α_H , and therefore deviations from
60 the global empirical relationship are more clearly expressed as a residual as opposed to a fractionation factor.

62 2.3 Statistical analyses

63 For all statistical analyses we use site-level mean isotopic values. This avoids biasing our analyses towards sites with a large
64 number of measurements, since there are large differences in the number of samples analyzed per site (n ranges from 66 to
65 1). To calculate α_C we used average $\delta^{13}\text{C-CH}_4$ and $\delta^{13}\text{C-CO}_2$ at a given site. This approach entails some additional
66 uncertainty in this variable, but was necessary because at many sites these measurements were not made on the same
67 samples.

68 We perform a set of linear regression analyses $\delta^2\text{H-CH}_4$ against other isotopic variables, in addition to latitude. All
69 statistical analyses were performed in Matlab. We considered $p < 0.05$ to be the threshold for identifying significant
70 regression relationships. We chose to perform unweighted regression, as opposed to weighted regression based on the
71 standard error of sample measurements, for two reasons. First, a small number of sites with a large number of measurements,
72 and therefore small standard error, had a disproportionate effect on weighted regression results. Second, in environmental
73 research unweighted regression is frequently less biased than weighted regression (Fletcher and Dixon, 2012). Based on a
74 test proposed by Fletcher and Dixon (2012) unweighted regression is appropriate for this dataset. We used analysis of
75 covariance to test for significant differences ($p < 0.05$) between regression relationships.

76 To compare isotopic data ($\delta^2\text{H-CH}_4$ and $\delta^{13}\text{C-CH}_4$) between groups (i.e. latitudinal bands, ecosystem types, sample
77 types) we used non-parametric statistical tests to test whether the groups were from different distributions. We used non-
78 parametric tests because some sample groups were not normally distributed, as determined by a Shapiro-Wilk test (Shapiro
79 and Wilk, 1965). For comparing differences between the distributions of two groups we used the Mann-Whitney U-test
80 (Mann and Whitney, 1947), whereas when comparing differences between the distributions of more than two groups we
81 used the Kruskal-Wallis H-test (Kruskal and Wallis, 1952), combined with Dunn's test to compare specific sample group
82 pairs (Dunn, 1964). We considered $p < 0.05$ to be the threshold for identifying groups with significantly different
83 distributions.

84 When comparing $\delta^{13}\text{C-CH}_4$ by latitude and ecosystem we combined the data from this study with additional data
85 from Sherwood et al. (2017) (32 additional sites) where $\delta^2\text{H-CH}_4$ was not measured to make our dataset as representative as
86 possible. To our knowledge this combined dataset is the largest available compiled dataset of freshwater $\delta^{13}\text{C-CH}_4$, although

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- 2.3.3 Sample set comparison tests -
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- 2.3.3 Sample set comparison tests -
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19 there are many more $\delta^{13}\text{C}-\text{CH}_4$ measurements that have not yet been aggregated. We did not include these additional data
20 when analysing differences by sample type, as sample type was not specified in the dataset of Sherwood et al. (2017).

21 2.4 Estimation of global atmospheric CH_4 $\delta^2\text{H}$ and $\delta^{13}\text{C}$ source values

22 To better understand how latitudinal differences in wetland isotopic source signatures influence atmospheric $\delta^2\text{H}-\text{CH}_4$ and
23 $\delta^{13}\text{C}-\text{CH}_4$, we calculated a 'bottom-up' mixing model of $\delta^2\text{H}-\text{CH}_4$ and $\delta^{13}\text{C}-\text{CH}_4$. For this calculation we ascribed all CH_4
24 sources a flux (derived from Saunois et al., 2020; see details below) and a $\delta^2\text{H}$ and $\delta^{13}\text{C}$ value, and calculated the global
25 atmospheric source value using an isotopic mixing model. Because of non-linearity when calculating mixtures using $\delta^2\text{H}$
26 values, we performed the mixing equation using isotopic ratios (see Sect. 2.1). The mixing equation is as follows:

$$27 R_{\text{mix}} = f_1 R_1 + f_2 R_2 + \dots + f_n R_n \quad (4)$$

28 where f_n is the fractional flux for each source term (i.e. the ratio of the source flux to total flux), and R_n is the isotope ratio for
29 each source term.

30 Values for the flux, $\delta^2\text{H}$, and $\delta^{13}\text{C}$ applied for each source term are shown in Table 1. We used bottom-up source
31 fluxes from Saunois et al. (2020) for the period 2008-2017. For categories other than wetlands, inland waters, and rice
32 paddies, we used global fluxes and isotope values, since geographically resolved isotopic source signature estimates are not
33 available. For these sources we used $\delta^2\text{H}$ values published by Sherwood et al. (2017), using the mean value for each source
34 term. For wetlands, inland waters, and rice paddies, we used geographically resolved (60-90 °N; 30-60 °N, 90° S-30°N)
35 fluxes derived from Saunois et al. (2019) for the period 2008-2017, and mean $\delta^2\text{H}-\text{CH}_4$ for these latitudinal bands from this
36 study.

37 To calculate mean $\delta^{13}\text{C}-\text{CH}_4$ from wetlands, inland waters, and rice paddies for different latitudinal bands we
38 combined the data from this study along with additional data from Sherwood et al. (2017) (32 additional sites) to make our
39 estimated source signatures as representative as possible. To our knowledge this combined dataset is the largest available
40 compiled dataset of freshwater $\delta^{13}\text{C}-\text{CH}_4$ (See Sect. 2.3). Sites dominated by C_4 plants are notably underrepresented in this
41 combined dataset. In addition, the biomass burning dataset of Sherwood et al. (2017) contains very few data from C_4 plant
42 combustion. We performed a separate estimate of global source $\delta^{13}\text{C}-\text{CH}_4$ that attempted to correct for these likely biases by
43 making two adjustments: 1) using the estimated low-latitude wetland $\delta^{13}\text{C}-\text{CH}_4$ signature of Ganesan et al., (2018) (-56.7‰),
44 which takes into account the predicted spatial distribution of C_4 plant dominated wetlands; and 2) using the biomass burning
45 $\delta^{13}\text{C}-\text{CH}_4$ signature of Schwietzke et al., (2016) (-22.3‰), which is weighted by the predicted contribution from C_4 plant
46 combustion. We did not attempt to take into account $\delta^{13}\text{C}-\text{CH}_4$ from ruminants feeding on C_4 plants. For these C_4 plant
47 corrections we applied the same uncertainties that are reported in Table 1.

48 Since fluxes from other natural sources are not differentiated for the period 2008-2017, we calculated the
49 proportional contribution of each category of other natural sources for the period 2000-2009 (Saunois et al., 2020), and

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65 | applied this to the total flux from other natural sources for 2008-2017. Inland waters and rice paddies do not have
 66 | geographically resolved fluxes reported in Saunio et al. (2019). Therefore, we calculated the proportion of *other natural*
 67 | *sources* attributed to inland waters from 2000-2009 (71%), and applied this proportion to the geographically resolved fluxes
 68 | of *other natural sources*. Similarly, we calculated the proportion of *agricultural and waste sources* attributed to rice
 69 | agriculture from 2008-2017 (15%), and applied this to the geographically resolved fluxes of *agricultural and waste fluxes*.

70 | To estimate uncertainty in the modelled total source $\delta^2\text{H}$ and $\delta^{13}\text{C}$ values we conducted Monte Carlo analyses
 71 | (Thompson et al., 1992). We first estimated the uncertainty for each flux, $\delta^2\text{H}$, and $\delta^{13}\text{C}$ term. Flux uncertainties were
 72 | defined as one half of the range of estimates provided by Saunio et al., (2020). For sources where fluxes were calculated as
 73 | a proportion of a larger flux, we applied the same proportional calculation to uncertainty estimates. In cases where one half
 74 | of the range of reported studies was larger than the flux estimate, we set the uncertainty to be equal to the flux estimate to
 75 | avoid negative fluxes in the mixing model. Isotopic source signal uncertainties were defined as the 95% confidence interval
 76 | of the mean value for a given source category. For some sources there is insufficient data to calculate a 95% confidence
 77 | interval, and we applied a conservative estimate of uncertainty for these sources, as detailed in Table 1. We then recalculated
 78 | the $\delta^2\text{H}$ and $\delta^{13}\text{C}$ mixing models 10,000 times, each time sampling inputs from the uncertainty distribution for each variable.
 79 | We assumed all uncertainties were normally distributed. We interpret the 2-sigma standard deviation of the resulting Monte
 80 | Carlo distributions as an estimate of the uncertainty of our total atmospheric CH_4 source isotopic values. To examine how
 81 | the Monte Carlo analyses were specifically influenced by uncertainty in isotopic source signatures vs. flux estimates, we
 82 | conducted sensitivity tests where we set the uncertainty in either isotopic source signatures or flux estimates to zero. We also
 83 | used the mixing model and Monte Carlo method to estimate the mean flux-weighted freshwater $\delta^2\text{H-CH}_4$ and $\delta^{13}\text{C-CH}_4$,
 84 | using only the inputs for freshwater environments (Wetlands, Inland Waters, and Rice Cultivation) from Table 1 (See Sect.
 85 | 3.5).

Table 1: Estimates of source-specific fluxes, $\delta^2\text{H-CH}_4$, and $\delta^{13}\text{C-CH}_4$, and their uncertainties, used in mixing models and Monte Carlo analyses

Category	Flux		$\delta^2\text{H}$ signature		$\delta^{13}\text{C}$ signature	
	(Tg/Yr)	Uncertainty	(‰, VSMOW)	Uncertainty	(‰, VPDB)	Uncertainty
Wetlands (<30N)	115	37.5	-301	15	-64.4	1.9
Wetlands (30-60N)	25	16.5	-324	14	-61.8	2.6
Wetlands (>60N)	9	8.0	-374	10	-62.7	3.0
Inland Waters (<30N)	80	39.4	-301	12	-57.1	3.0
Inland Waters (30-60N)	64	31.9	-308	18	-62.0	3.8
Inland Waters (>60N)	16	7.5	-347	9	-65.0	1.8
Geological (onshore) ^a	38	13.0	-189	44	-43.8	10.0
Wild animals ^b	2	2.0	-316	28	-65.4	3.5

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Termites ^c	9	6.0	-343	50	-63.4	3.5
Permafrost soils (direct) ^d	1	0.5	-37.4	15	-64.4	1.7
Geological (offshore) ^a	7	7.0	-189	44	-43.8	10.0
Biogenic open and coastal ^c	6	3.0	-200	50	-80.0	20.0
Enteric fermentation and manure	111	5.0	-308	28	-65.4	3.5
Landfills and waste	65	4.5	-297	6	-56.0	4.9
Rice cultivation (<30N)	19	1.2	-324	8	-55.0	6.5
Rice cultivation (30-60N)	12	0.5	-325	8	-62.3	2.1
Coal mining	42	15.5	-232	5	-49.5	1.0
Oil and gas	79	13.0	-189	2	-43.8	0.5
Industry ^f	3	3.0	-189	2	-43.8	0.5
Transport ^f	4	4.0	-189	2	-43.8	0.5
Biomass burning ^g	17	6.0	-211	15	-26.2	2.0
Biofuel burning ^g	12	2.0	-211	15	-26.2	2.0

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a-No specific isotopic measurements in the database (Sherwood et al., 2017). We applied the mean isotopic values for oil and gas, and applied the standard deviation of for oil and gas as the uncertainty

b-No specific isotopic measurements in database (Sherwood et al., 2017). We used the isotopic values and uncertainties from livestock

c-Only one $\delta^2\text{H}$ measurement in database (Sherwood et al., 2017). We applied 50‰ as a conservative uncertainty estimate.

d- No specific isotopic measurement in database (Sherwood et al., 2017). We used the isotopic values and uncertainties for high-latitude wetlands

e- No specific isotopic measurements in database (Sherwood et al., 2017). We applied approximate isotopic values based on Whiticar, (1999), and conservatively large uncertainty estimates.

f-No specific isotopic measurements in database (Sherwood et al., 2017). We used the isotopic values and uncertainties for oil and gas.

g-We applied all isotopic measurements of biomass burning to both the biomass burning and biofuel burning categories. We did not correct for the relative proportion of C_3 and C_4 plant combustion sources (See Sect. 2.4)

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86 3 Results and Discussion

87 3.1 Dataset distribution

88 The dataset is primarily concentrated in the northern hemisphere (Fig. 1A), but is distributed across a wide range of
 89 latitudes between 3 °S to 73 °N (Fig. 1C). The majority of sampled sites are from North America (Fig. 1B), but there are
 90 numerous sites from Eurasia. A much smaller number of sites are from South America and Africa. We define three
 91 latitudinal bands for describing geographic trends: low latitudes (3 °S to 30 °N); mid-latitudes (30 °to 60 °N); and high-
 92 latitudes; (60° to 90° N). This definition was used primarily because it corresponds with a commonly applied geographic
 93 classification of CH_4 fluxes (Saunois et al., 2020).

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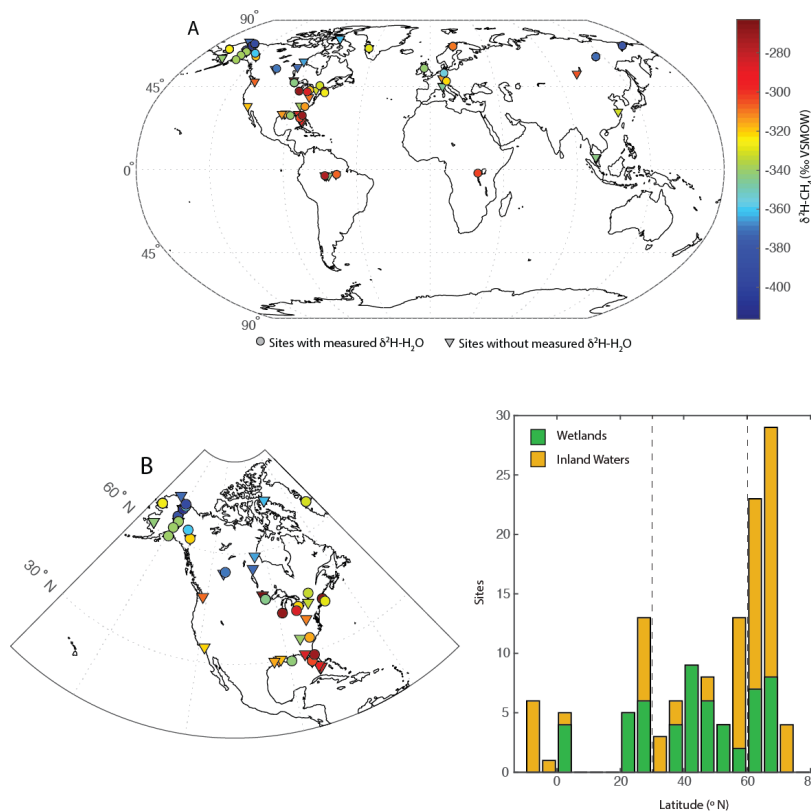


Figure 1: Distribution of sites shown; A) on a global map, with site mean $\text{CH}_4-\delta^2\text{H}$ values indicated in relation to a color bar. Sites with and without measured $\delta^2\text{H}-\text{H}_2\text{O}$ are differentiated; B) on a map of North America; and C) as a histogram of sites by latitude, differentiated between wetlands and inland waters. Dashed lines in C) indicate divisions between low-latitude, mid-latitude, and high-latitude sites.

74 of 129 sites are classified as inland waters, primarily lakes ($n = 66$), with a smaller number from rivers ($n = 8$). To our knowledge, all of the inland water sites are natural ecosystems and do not include reservoirs. 55 sites are classified as wetlands, including 16 bogs, 14 swamps and marshes, 12 fens, and 8 rice paddies. For the majority of sites ($n = 84$) gas samples were measured, whereas studies at 36 sites measured dissolved CH_4 or diffusive fluxes.

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28 As discussed in Sect. 2.2.3, we compared modelled annual and growing season $\delta^2\text{H}_p$ with measured $\delta^2\text{H-H}_2\text{O}$ to
 29 determine which is a better estimator for sites where $\delta^2\text{H-H}_2\text{O}$ is not measured. We performed this analysis separately for
 30 wetland and inland water environments because these broad environmental categories have distinct hydrological
 31 characteristics. For all comparisons we found strong correlations, with R^2 values between 0.82 to 0.88 (Fig. 2). For wetlands,
 32 regression using annual $\delta^2\text{H}_p$ produces a slightly better fit, and also produces a slope within error of 1 (Fig. 2A), suggesting
 33 that variation in annual $\delta^2\text{H}_p$ scales proportionately with variation in measured $\delta^2\text{H-H}_2\text{O}$. However, the intercept of this
 34 relationship was significantly greater than 0 (19±9 ‰). We interpret this intercept as indicating that evaporative isotopic
 35 enrichment is generally important in controlling $\delta^2\text{H-H}_2\text{O}$ in wetlands. A slope slightly greater than 1 is also consistent with
 36 evaporative enrichment, since greater evaporation rates would be expected in low-latitude environments with higher $\delta^2\text{H-}$
 37 H_2O . These results are consistent with detailed studies of wetland isotope hydrology that indicate a major contribution from
 38 groundwater, with highly dampened seasonal variability relative to precipitation, but also indicate evaporative enrichment of
 39 water isotopes in shallow soil water (Sprenger et al., 2017; David et al., 2018).

40 For inland waters, regression with growing season $\delta^2\text{H}_p$ produces a relationship that is within error of the 1:1 line
 41 (Fig. 2C), in contrast to annual $\delta^2\text{H}_p$, which produces a flatter slope (Fig. 2D). We infer that seasonal differences in $\delta^2\text{H}_p$ are
 42 important in determining $\delta^2\text{H-H}_2\text{O}$ in the inland water environments analyzed, especially at high latitudes, implying that
 43 these environments generally have water residence times on subannual timescales. This finding is generally consistent with
 44 evidence for seasonal variation in lake water isotopic compositions that is dependent on lake water residence times (Tyler et
 45 al., 2007; Jonsson et al., 2009). Lake water residence times vary widely, primarily as a function of lake size, but isotopic data
 46 implies that small lakes have water residence times of less than a year (Brooks et al., 2014), resulting in seasonal isotopic
 47 variability (Jonsson et al., 2009). Isotopic enrichment of lake water is highly variable, but is typically minor in humid and
 48 high-latitude regions (Jonsson et al., 2009; Brooks et al., 2014), which characterizes most of our study sites.

49 Based on these results we combine measured and estimated $\delta^2\text{H-H}_2\text{O}$ to determine a 'best-estimate' value for each
 50 site, an approach similar to that of Waldron et al. (1999a). For sites with measured $\delta^2\text{H-H}_2\text{O}$ values we use the measured
 51 value. For inland water sites without measured $\delta^2\text{H-H}_2\text{O}$ we use modeled growing season $\delta^2\text{H}_p$ since the regression of this
 52 against measured $\delta^2\text{H-H}_2\text{O}$ is indistinguishable from the 1:1 line (Fig. 2D). For wetland sites without measured $\delta^2\text{H-H}_2\text{O}$ we
 53 estimate $\delta^2\text{H-H}_2\text{O}$ using the regression relationship with annual precipitation $\delta^2\text{H-H}_2\text{O}$ shown in Fig. 2A. The root mean
 54 square errors (RMSE) of these relationships (16‰ for wetlands, 22‰ for inland waters) provide an estimate of the
 55 uncertainty associated with estimating $\delta^2\text{H-H}_2\text{O}$ using $\delta^2\text{H}_p$. Given the uncertainty associated with estimating $\delta^2\text{H-H}_2\text{O}$ using
 56 $\delta^2\text{H}_p$, for all analyses presented below that depend on $\delta^2\text{H-H}_2\text{O}$ values we also analyse the dataset only including sites with
 57 measured $\delta^2\text{H-H}_2\text{O}$.

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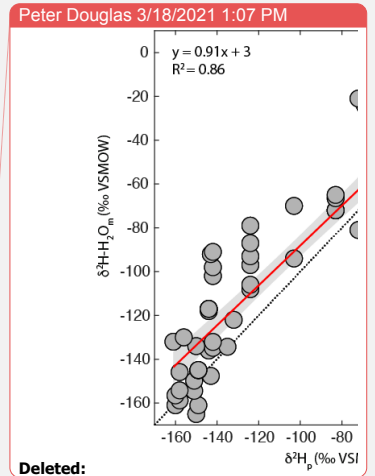
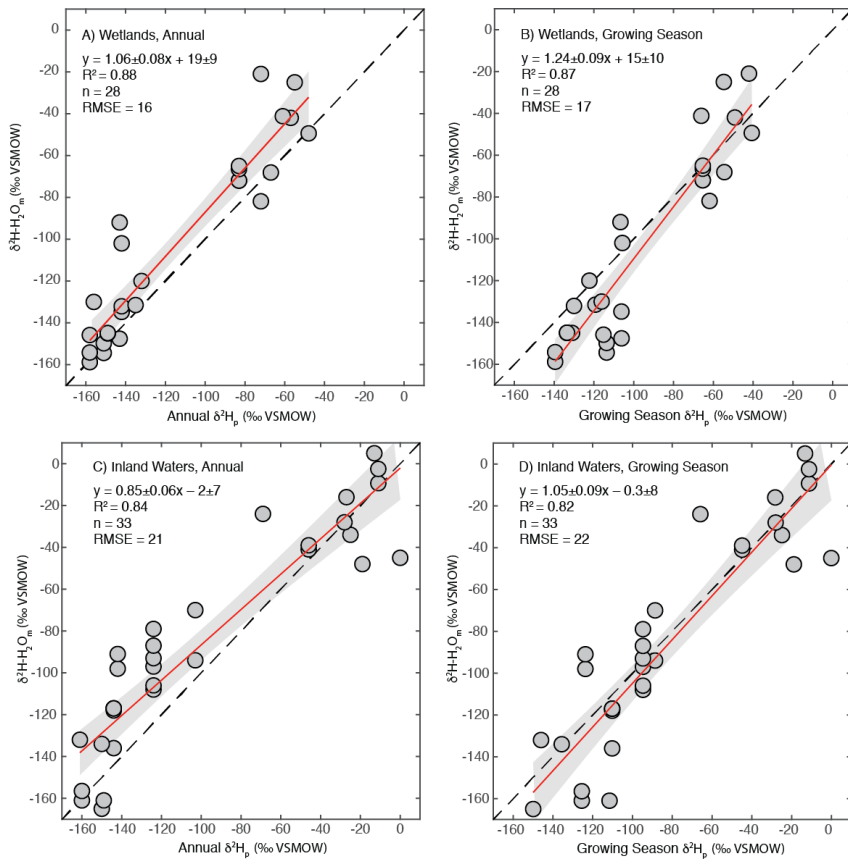
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Deleted: Identifying the relationship between modelled $\delta^2\text{H}_p$ and $\delta^2\text{H-CH}_4$ is of value because global distributions of $\delta^2\text{H}_p$ are routinely estimated using isotope enabled Earth system models (Zhu et al., 2017), and could potentially be used to predict the distribution of $\delta^2\text{H-CH}_4$ under past and future global climates. In addition, this approach allows us to examine co-variation of $\delta^2\text{H-CH}_4$ and $\delta^2\text{H-H}_2\text{O}$ at sites where $\delta^2\text{H-H}_2\text{O}$ was not measured. However, modelled annual $\delta^2\text{H}_p$ is not a perfect estimator of $\delta^2\text{H-H}_2\text{O}$ in freshwater environments. Comparing modelled $\delta^2\text{H}_p$ with measured $\delta^2\text{H-H}_2\text{O}$ for 62 sites indicates generally good agreement at low and high values of $\delta^2\text{H}_p$, but also that $\delta^2\text{H-H}_2\text{O}$ is generally higher than predicted at intermediate values of $\delta^2\text{H}_p$ (Fig. 2). We infer that this is because in our dataset wetlands are concentrated in the mid-latitudes (Fig. 1B), and because wetlands could be more likely to experience changes in $\delta^2\text{H-H}_2\text{O}$ on a seasonal basis because of overall smaller water volumes than lakes (Clay et al., 2004). However, the root mean square error (RMSE) for the 1:1 line (23‰) is not substantially larger than the RMSE for the best-fit linear relationship shown in Fig. 2 (19‰). Based on this, and on the rationale for analyzing $\delta^2\text{H-CH}_4$ in terms of $\delta^2\text{H}_p$ as described above, we proceed with this analysis. However, we also examine how the relationship between $\delta^2\text{H}_p$ vs. $\delta^2\text{H-CH}_4$ differs from that for $\delta^2\text{H-H}_2\text{O}_m$ vs. $\delta^2\text{H-CH}_4$ (Sect. 3.3).



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93 **Figure 2: Scatter plots of annual or growing season $\delta^2\text{H}_p$ vs. measured $\delta^2\text{H}-\text{H}_2\text{O}$ for wetland (A,B) and inland water (C,D) sites.** The red lines indicates the best fit, with a 95% confidence interval (gray envelopes), and the dashed black lines are the 1:1 relationship.

96 **3.3 Relationship between $\delta^2\text{H}-\text{H}_2\text{O}$ and $\delta^2\text{H}-\text{CH}_4$**

97 We carried out regression analyses of $\delta^2\text{H}-\text{H}_2\text{O}$ vs. $\delta^2\text{H}-\text{CH}_4$, both using 'best-estimate' $\delta^2\text{H}-\text{H}_2\text{O}$ as described in sect. 3.2
98 (Fig. 3A), and only including sites with measured $\delta^2\text{H}-\text{H}_2\text{O}$ (Fig. 3B). In addition we analysed the relationship for all sites
99 using annual (Fig. 3C) and growing season (Fig. 3D) $\delta^2\text{H}_p$. Identifying the relationship between modelled $\delta^2\text{H}_p$ and $\delta^2\text{H}-\text{CH}_4$
00 is of value because this could be used to create gridded global predictions of $\delta^2\text{H}-\text{CH}_4$ based on gridded datasets of $\delta^2\text{H}_p$

08 (Bowen and Revenaugh, 2003), as well as to predict the distribution of $\delta^2\text{H-CH}_4$ under past and future global climates using
09 isotope enabled Earth system models (Zhu et al., 2017).

10 $\delta^2\text{H-CH}_4$ is significantly positively correlated with $\delta^2\text{H-H}_2\text{O}$ when using all four methods of estimating $\delta^2\text{H-H}_2\text{O}$
11 (Fig. 3, Supplemental Table 2). This is the case when analysing all sites together, as well as when analysing wetlands and
12 inland waters separately (Supplemental Table 2, Fig. 4). There is no significant difference in regression relationships, based
13 on analysis of covariance, when $\delta^2\text{H-CH}_4$ is regressed against best-estimate $\delta^2\text{H-H}_2\text{O}$, measured $\delta^2\text{H-H}_2\text{O}$, or modelled $\delta^2\text{H}_p$,
14 nor is there a major difference in R^2 values or RMSE (Supplemental Table S2). Wetland sites consistently have a steeper
15 regression slope than inland water sites (Supplemental Table S2), but this difference is not significant. Regression with
16 wetland sites also consistently results in a higher R^2 values and lower RMSE.

17 Given the similar results when regressing with estimated or measured $\delta^2\text{H-H}_2\text{O}$, we infer that using either the 'best-
18 estimate' $\delta^2\text{H-H}_2\text{O}$ or modelled $\delta^2\text{H}_p$ instead of measured $\delta^2\text{H-H}_2\text{O}$ to predict $\delta^2\text{H-CH}_4$ does not result in substantial
19 additional error. This implies that isotope-enabled Earth Systems models (ESMs) could be used to predict the distribution of
20 freshwater $\delta^2\text{H-CH}_4$ under past and future climates based on modeled $\delta^2\text{H}_p$, although the substantial scatter in Figures 3C
21 and D should be taken into account. The southern hemisphere is highly underrepresented in available $\delta^2\text{H-CH}_4$ data.
22 However, the mechanisms linking $\delta^2\text{H-CH}_4$ with $\text{H}_2\text{O-}\delta^2\text{H}$ should not differ in the southern hemisphere, and we argue that
23 the relationships observed in this study are suitable to predict southern hemisphere freshwater $\delta^2\text{H-CH}_4$. The choice of
24 predicting $\delta^2\text{H-CH}_4$ using growing-season vs. annual precipitation $\delta^2\text{H}_p$ could be important, with steeper slopes overall when
25 regressing against growing season $\delta^2\text{H}_p$. Based on our analysis in sect. 3.2, we suggest that annual $\delta^2\text{H}_p$ may be more
26 appropriate for estimating wetland $\delta^2\text{H-CH}_4$, while growing season $\delta^2\text{H}_p$ may be more appropriate for estimating inland
27 water $\delta^2\text{H-CH}_4$. Future research will combine gridded datasets of wetland distribution (Ganesan et al., 2018), modeled
28 annual $\delta^2\text{H}_p$ (Bowen and Revenaugh, 2003), and the regression relationships from this study to predict spatially-resolved
29 wetland $\delta^2\text{H-CH}_4$ at a global scale.

30 Overall, our results are consistent with those of Waldron et al., (1999a), and confirm the finding of that study that
31 $\delta^2\text{H-H}_2\text{O}$ is the predominant predictor of global variation in $\delta^2\text{H-CH}_4$. All of the regression slopes produced using our dataset
32 are flatter than the regression relationship found by Waldron et al. (1999a) using a smaller dataset (0.68 ± 0.1), although the
33 slopes are not significantly different based on analysis of covariance. Based on this result we infer that the true global
34 relationship is likely flatter than that estimated by Waldron et al. (1999a), but more data will be needed to further constrain
35 this relationship. The difference between the regression relationships reported here and that of Waldron et al. (1999a) is
36 largely a result of a much greater number of samples from the high latitudes (Fig. 1C), where $\delta^2\text{H-H}_2\text{O}$ values are typically
37 lower. The small number of high-latitude sites sampled by Waldron et al. (1999a) are skewed towards the low end of the
38 high-latitude $\delta^2\text{H-CH}_4$ data from this study (Fig. 3). A similarly flatter slope (0.54 ± 0.05) was found by Chanton et al. (2006)
39 when combining a dataset of $\delta^2\text{H-CH}_4$ from Alaskan wetlands, which are included in this study, with the dataset of Waldron
40 et al. (1999a). As discussed below in sect. 3.3.1, our regression relationship slopes are very similar to that of the 'in-vitro'

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65 line of Waldron et al. (1999a). Based on the range of R^2 values shown in Figure 3, we estimate that $\delta^2\text{H-H}_2\text{O}$ explains
 66 approximately 42% of variability in $\delta^2\text{H-CH}_4$, implying substantial residual variability, with greater residual variability
 67 inland water sites than in wetlands (Supplemental Table 2).
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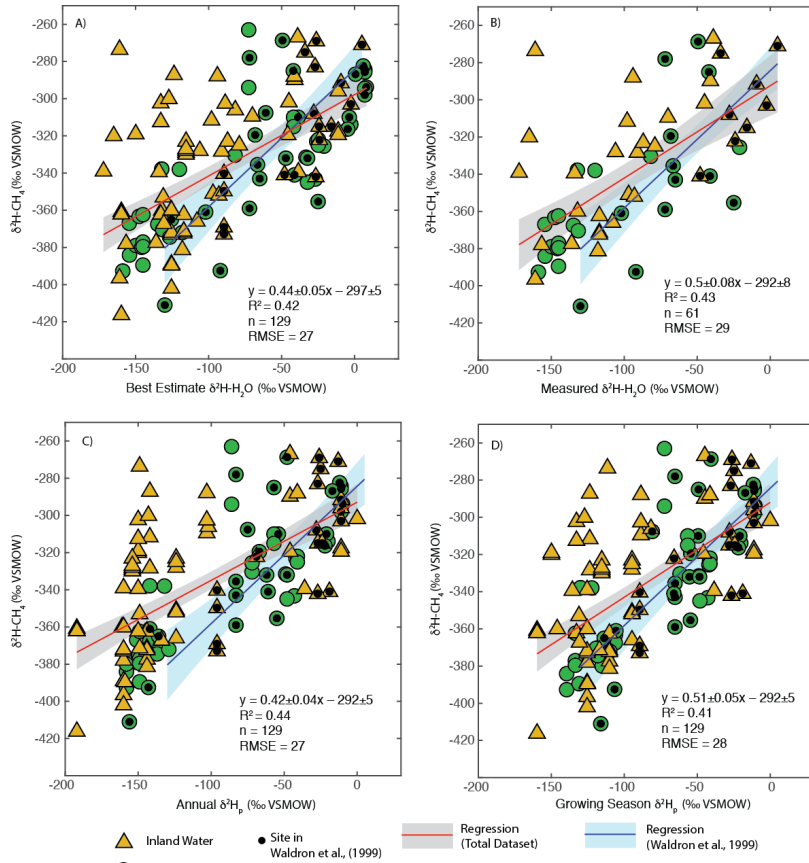
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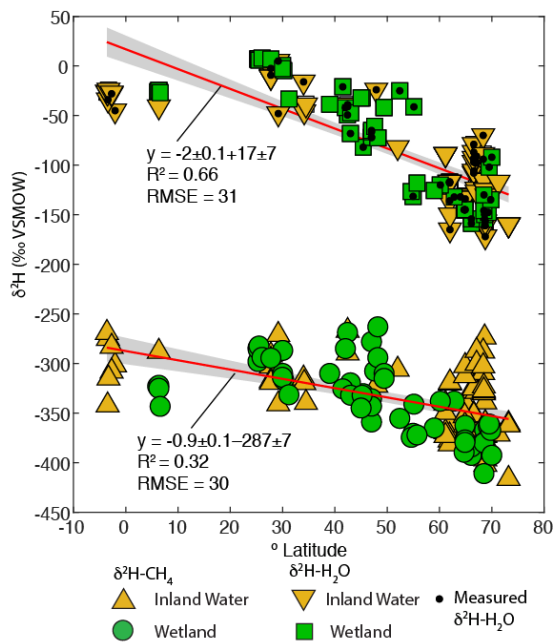
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 70 **Figure 3:** Scatter plots of $\delta^2\text{H-CH}_4$ vs. (A) best-estimate $\delta^2\text{H-H}_2\text{O}$; (B) measured $\delta^2\text{H-H}_2\text{O}$; (C) annual $\delta^2\text{H}_p$; and (D) growing
 71 season $\delta^2\text{H}_p$. Sites that were included in the analysis of Waldron et al. (1999a) are indicated. The regression relationship for the
 72 total dataset in each plot is shown by the red line, with its 95% confidence interval (grey envelope). The regression relationship
 73 and confidence interval for the dataset of Waldron et al., (1999a) is shown in blue. Uncertainties for reported regression
 74 relationships are standard errors.

87 Given that $\delta^2\text{H-H}_2\text{O}$ is strongly influenced by latitude, although it is also influenced by other geographic and
 88 climatic variables, we examined whether $\delta^2\text{H-CH}_4$ is also significantly correlated with latitude. There is indeed a significant,
 89 negative relationship between latitude and $\delta^2\text{H-CH}_4$, indicating an approximate decrease of $0.9\text{‰}/^\circ$ latitude (Fig. 4). The
 90 slope is significantly flatter than that for latitude vs. $\delta^2\text{H-H}_2\text{O}$ in this dataset ($-2\text{‰}/^\circ$ latitude), which is consistent with the
 91 inferred slope for $\delta^2\text{H-H}_2\text{O}$ vs. $\delta^2\text{H-CH}_4$ (0.44 to 0.5). There is greater scatter in $\delta^2\text{H-CH}_4$ at higher latitudes, especially for
 92 inland waters, but it is unclear if this is simply a result of a larger sample set or of differences in the underlying processes
 93 controlling $\delta^2\text{H-CH}_4$. We discuss latitudinal differences in $\delta^2\text{H-CH}_4$ in further detail in Sect. 3.5



94 **Figure 4:** Scatter plots of $\delta^2\text{H-CH}_4$ and best estimate $\delta^2\text{H-H}_2\text{O}$ vs. latitude ($^\circ\text{N}$). Sites with measure $\delta^2\text{H-H}_2\text{O}$ are indicated.
 95 **Envelopes indicate 95% confidence intervals for regression lines.**

96 **3.3.1 Comparison of $\delta^2\text{H-H}_2\text{O}$ vs $\delta^2\text{H-CH}_4$ relationships between environmental and experimental studies**

97 To further understand the processes controlling the observed $\delta^2\text{H-H}_2\text{O}$ vs. $\delta^2\text{H-CH}_4$ relationships we compared
 98 them to results from pure culture and incubation experiments across a wide range of $\delta^2\text{H-H}_2\text{O}$ values (Fig. 5), focusing on
 99 regression against best-estimate $\delta^2\text{H-H}_2\text{O}$. The regression slopes for both wetlands and inland waters (0.5 and 0.42) are
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16 within error of the 'in-vitro' relationship compiled by Waldron et al. (1999a) (0.44), based on laboratory incubations from
17 three separate studies (Schoell, 1980; Sugimoto and Wada, 1995; Waldron et al., 1998). The intercept for the wetland and
18 inland water regressions is higher than that for the 'in-vitro' relationship, although only the difference with inland waters is
19 significant. In contrast, the regression slope for pure-culture acetoclastic methanogenesis experiments is much flatter (0.18 to
20 0.2) (Valentine et al., 2004b; Gruen et al., 2018), consistent with the prediction that one hydrogen atom is exchanged between
21 water and the acetate methyl group during CH₄ formation (Pine and Barker, 1956; Whiticar, 1999). The large difference in
22 intercept between the two acetate pure culture datasets is likely a function of differences in the δ²H of acetate, but could also
23 be influenced by differences in kinetic isotope effects (Valentine et al., 2004b).

24 Pure culture hydrogenotrophic methanogenesis experiments (Gruen et al., 2018) yield a regression slope that is
25 consistent with a constant α_H value, although α_H clearly varies depending on experimental or environmental conditions
26 (Valentine et al., 2004b; Stolper et al., 2015; Douglas et al., 2016). The wetland, inland water, and 'in-vitro' regression
27 relationships are not consistent with a constant value of α_H (Fig. 5). Our comparison supports previous inferences that the in-
28 vitro line of Waldron et al. (1999a) provides a good estimate of the slope of environmental δ²H-H₂O vs. δ²H-CH₄
29 relationships. This slope is likely controlled by the relative proportion of acetoclastic and hydrogenotrophic methanogenesis,
30 the net kinetic isotope effect associated with these two methanogenic pathways, and variance in δ²H of acetate (Waldron et
31 al., 1998; Waldron et al., 1999a; Valentine et al., 2004a), but the relative importance of these variables remains uncertain.

32 In particular, the δ²H of acetate methyl hydrogen is probably influenced by environmental δ²H-H₂O, and therefore
33 likely varies geographically as a function of δ²H_p, as originally hypothesized by Waldron et al. (1999a). To our knowledge
34 there are no measurements of acetate or acetate-methyl δ²H from natural environments with which to test this hypothesis. In
35 general, variability in the δ²H of environmental organic molecules in lake sediments and wetlands, including fatty acids and
36 cellulose, is largely controlled by δ²H-H₂O (Huang et al., 2002; Sachse et al., 2012; Mora and Zanazzi, 2017), albeit with
37 widely varying fractionation factors. The δ²H of methoxyl groups in plants has also been shown to vary as a function of δ²H-
38 H₂O (Vigano et al., 2010). Furthermore, culture experiments with acetogenic bacteria imply that there is rapid isotopic
39 exchange between H₂ and H₂O during chemoautotrophic acetogenesis (Valentine et al., 2004a), implying that the δ²H of
40 chemoautotrophic acetate is also partially controlled by environmental δ²H-H₂O. Incubation experiments, such as those
41 included in the 'in-vitro line' (Schoell, 1980; Sugimoto and Wada, 1995; Waldron et al., 1998), probably contain acetate-δ²H
42 that varies as a function of ambient δ²H-H₂O, given that the acetate in these incubation experiments was actively produced
43 by fermentation and/or acetogenesis during the course of the experiment. This differs from pure cultures of methanogens,
44 where acetate is provided in the culture medium and therefore would not vary in its δ²H value (Valentine et al., 2004b; Gruen
45 et al., 2018).

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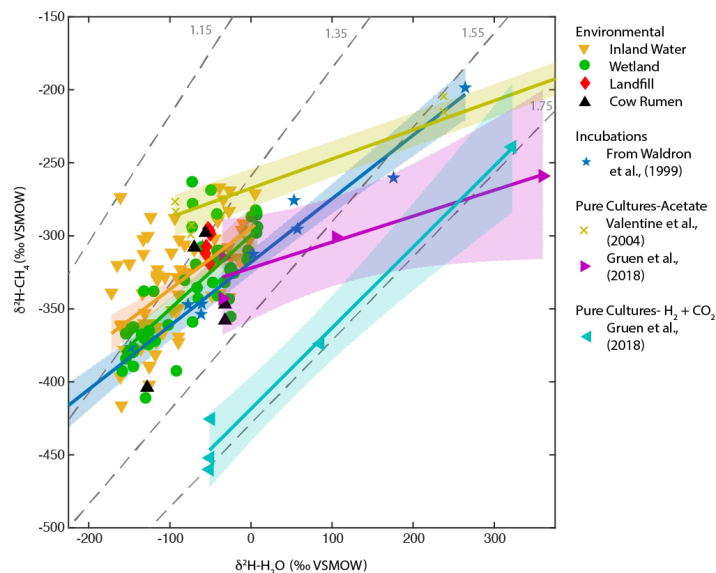
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58 **Figure 5: Scatter plots of $\delta^2\text{H-CH}_4$ vs. $\delta^2\text{H-H}_2\text{O}$ for wetlands, inland waters, landfills, and cow rumen, compared with incubation**
 59 **and pure-culture experiments. Regression lines and confidence intervals corresponding to each dataset (except landfills and cow**
 60 **rumen) are shown. Dashed gray lines indicate constant values of α_H . Regression line statistics are listed in Supplemental Table S2.**
 61 **Plotted $\delta^2\text{H-H}_2\text{O}$ values are ‘best-estimate’ values for wetlands and inland waters, measured values for culture experiments, and a**
 62 **combination of measured values and annual $\delta^2\text{H}_p$ for landfills and cow rumen (See supplemental Table S3 for more details).**

64 **3.4 Relationship of $\delta^2\text{H-CH}_4$ with $\delta^{13}\text{C-CH}_4$, $\delta^{13}\text{C-CO}_2$, and α_C**

65 As shown in Fig. 3, there is a large amount of residual variability in $\delta^2\text{H-CH}_4$ that is not explained by $\delta^2\text{H-H}_2\text{O}$.
 66 Several biogeochemical variables have been proposed to influence freshwater $\delta^2\text{H-CH}_4$ independently of $\delta^2\text{H-H}_2\text{O}$, including
 67 the predominant biochemical pathway of methanogenesis (Whiticar et al., 1986;Whiticar, 1999;Chanton et al., 2006), the
 68 extent of methane oxidation (Happell et al., 1994;Waldron et al., 1999a;Whiticar, 1999;Cadieux et al., 2016), isotopic
 69 fractionation resulting from diffusive gas transport (Waldron et al., 1999a;Chanton, 2005), and differences in the
 70 thermodynamic favorability or reversibility of methanogenesis (Valentine et al., 2004b;Stolper et al., 2015;Douglas et al.,
 71 2016). These variables are also predicted to cause differences in $\delta^{13}\text{C-CH}_4$, $\delta^{13}\text{C-CO}_2$, and α_C . Therefore, we analysed co-
 72 variation between $\delta^2\text{H-CH}_4$ (see definition in Sect. 2.2.3) and $\delta^{13}\text{C-CH}_4$, $\delta^{13}\text{C-CO}_2$, and α_C to see if it could partially
 73 explain the residual variability in $\delta^2\text{H-CH}_4$ (Fig. 6).

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Figure 4: Scatter plots of $\delta^2\text{H-CH}_4$, $\delta^2\text{H-H}_2\text{O}_m$, CH_4 , $\delta^2\text{H-H}_2\text{O}_m$, and $\delta^2\text{H}_p$ vs. latitude. Envelopes indicate 95% confidence intervals for regression lines.

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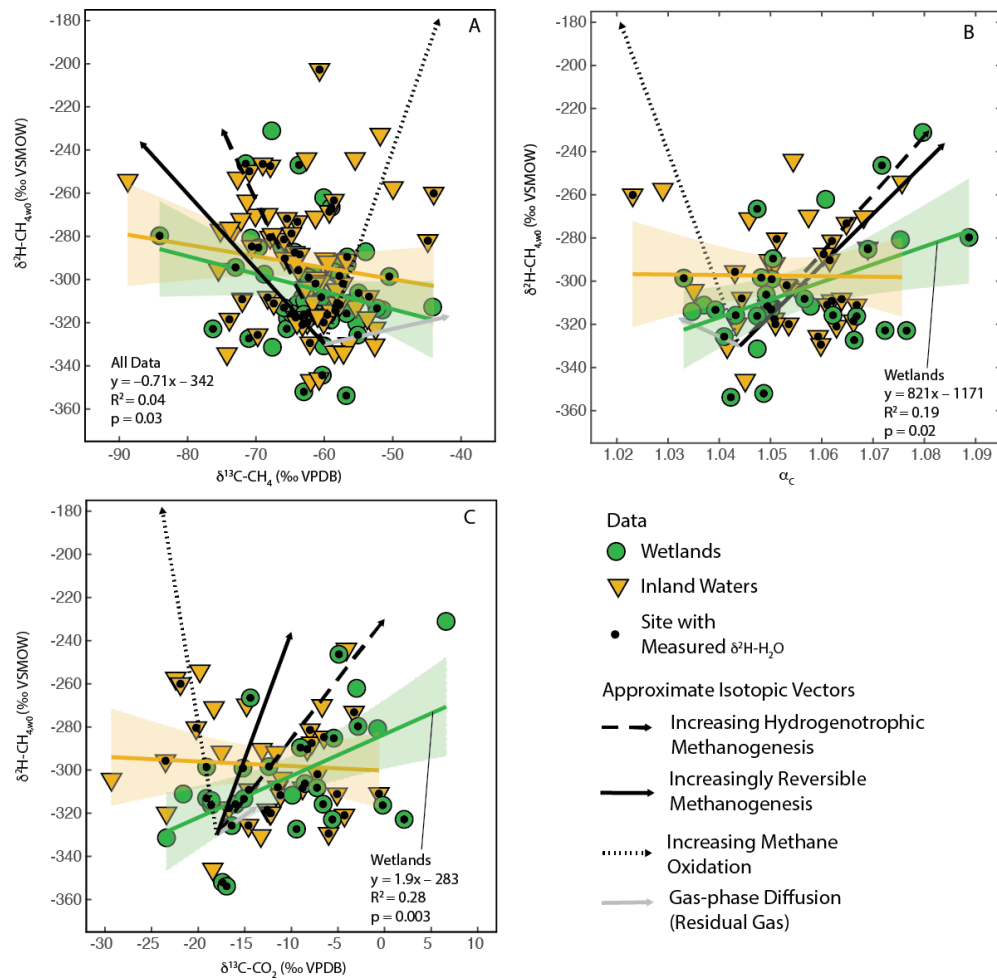
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'94 In order to facilitate interpretation of isotopic co-variation, we estimated approximate vectors of predicted isotopic
'95 co-variation for the four variables being considered (Fig. 6). We emphasize that these vectors are uncertain, and while they
'96 can be considered indicators for the sign of the slope of co-variation and the relative magnitude of expected isotopic
'97 variability, they are not precise representations of the slope or intercept of isotopic co-variation. In reality, isotopic co-
'98 variance associated with these processes likely varies depending on specific environmental conditions, although the sign of
'99 co-variance should be consistent. The starting point for the vectors is arbitrarily set to typical isotopic values for inferred
'00 acetoclastic methanogenesis in freshwater systems (Whiticar, 1999). We based the vectors for differences in the dominant
'01 methanogenic pathway and methane oxidation on Figures 8, 5, and 10 in Whiticar (1999). These figures are widely applied
'02 to interpret environmental isotopic data related to CH₄ cycling. However, we note that both environmental and experimental
'03 research has questioned whether differences in the dominant methanogenic pathway has an influence on δ²H-CH₄ (Waldron
'04 et al., 1998;Waldron et al., 1999a). Differences in δ²H-CH₄ between hydrogenotrophic and acetoclastic methanogenesis are
'05 likely highly dependent on both the δ²H of acetate and the carbon and hydrogen kinetic isotope effects for both
'06 methanogenic pathways, both of which are poorly constrained in natural environments and are likely to vary between sites
'07 (see Sect. 3.3.1). We did not differentiate between anaerobic and aerobic methane oxidation, and the vectors shown are
'08 similar to experimental results for aerobic methane oxidation (Wang et al., 2016).

'09 The vector for isotopic fractionation related to gas-phase diffusion is based on the calculations of Chanton (2005),
'10 and indicates isotopic change for residual gas following a diffusive loss. Gas-liquid diffusion is predicted to have a much
'11 smaller isotopic effect (Chanton, 2005). The vector for differences in enzymatic reversibility are based on experiments where
'12 CH₄ and CO₂ isotopic compositions were measured together with changes in methane production rate or Gibbs free energy
'13 (Valentine et al., 2004b;Penning et al., 2005). We note that these studies did not measure δ³H-CH₄ in the same experiments
'14 as δ¹³C-CH₄ or δ¹³C-CO₂, implying large uncertainty in the co-variance vectors. More detail on the estimated vectors is
'15 provided in the Supplementary Text.

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18 **Figure 6: Scatter plots of $\delta^2\text{H-CH}_{4,\text{are}}$ vs. (A) $\delta^{13}\text{C-CH}_4$, (B) α_c , and (C) $\delta^{13}\text{C-CO}_2$. Approximate vectors for isotopic co-variation**
 19 **related to four biogeochemical variables are shown. See details in Sect. 3.4 and the supplemental text. Regression relationships are**
 20 **shown for wetland and inland water sites, with envelopes indicating 95% confidence intervals. Regression statistics are shown here**
 21 **for relationships with significant correlations ($p < 0.05$). All regression statistics are detailed in Supplemental Table S4.**

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22 We observe significant positive correlations between $\delta^2\text{H-CH}_4\text{w}_0$, calculated using best estimate $\delta^2\text{H-H}_2\text{O}$, and both
23 $\delta^{13}\text{C-CO}_2$, and α_c for wetland sites (Fig. 6B,C; Supplemental Table S4). We do not observe a significant correlation between
24 these variables for inland water sites or for the dataset as a whole. We also observe a very weak, but significant, negative
25 correlation, between $\delta^2\text{H-CH}_4\text{w}_0$ and $\delta^{13}\text{C-CH}_4$ for all sites, but not for data disaggregated into wetlands and inland water
26 categories (Fig. 6A). The significant correlations shown in Figure 6 should be interpreted with caution, since repeating this
27 analysis only using sites with measured $\delta^2\text{H-H}_2\text{O}$ does not result in any significant correlations (Supplemental Table S4). It
28 is unclear whether this different result when using best-estimate or measured $\delta^2\text{H-H}_2\text{O}$ represents a bias related to estimating
29 $\delta^2\text{H-H}_2\text{O}$ using $\delta^{13}\text{C}$, or is an effect of the much smaller sample size for sites with $\delta^2\text{H-H}_2\text{O}$ measurements. If accurate, the
30 observed significant positive correlations in Figures 6B and C suggest that residual variability in $\delta^2\text{H-CH}_4$ in wetlands is
31 more strongly controlled by biogeochemical variables related to methanogenesis, namely differences in methanogenic
32 pathway or thermodynamic favorability, than post-production processes such as diffusive transport and CH_4 oxidation.
33 However, the residual variability in $\delta^2\text{H-CH}_4$ explained by $\delta^{13}\text{C-CO}_2$ and α_c in wetlands is relatively small, specifically
34 between 19 to 28% based on the R^2 values in Figures 6B and C. For inland water sites our analysis suggests that no single
35 biogeochemical variable has clear effect in controlling residual variability in $\delta^2\text{H-CH}_4$. It is intriguing that we observe the
36 strongest correlation in wetlands between $\delta^2\text{H-CH}_4\text{w}_0$ and $\delta^{13}\text{C-CO}_2$, since it is probable that a wide range of biotic and
37 abiotic processes unrelated to methane cycling influence $\delta^{13}\text{C-CO}_2$. This suggests that measurements of $\delta^{13}\text{C-CO}_2$ are
38 important for future research on environmental variables controlling wetland $\delta^2\text{H-CH}_4$.

39 Overall, our results are not consistent with arguments that residual variability in freshwater $\delta^2\text{H-CH}_4$ is dominantly
40 controlled by either differences in methanogenic pathway (Chanton et al., 2006), or post-production processes (Waldron et
41 al., 1999a). Instead they highlight the combined influence of a complex set of variables and processes that are difficult to
42 disentangle on an inter-site basis using $\delta^{13}\text{C}$ measurements alone. It is also important to note the likely importance of
43 variables that could influence $\delta^{13}\text{C-CH}_4$ or $\delta^{13}\text{C-CO}_2$ but not necessarily affect $\delta^2\text{H-CH}_4$, including variance in the $\delta^{13}\text{C}$ of
44 soil or sediment organic matter (Conrad et al., 2011; Ganesan et al., 2018), diverse metabolic and environmental sources and
45 sinks of CO_2 in aquatic environments, and Rayleigh fractionation associated with CH_4 carbon substrate depletion (Whiticar,
46 1999). Finally, the possible role of other carbon substrates, such as methanol, in CH_4 production could be important in
47 controlling isotope variability. Culture experiments suggest that CH_4 produced from methanol has low $\delta^{13}\text{C}$ and $\delta^2\text{H}$ values
48 relative to other pathways (Krzycki et al., 1987; Penger et al., 2012; Gruen et al., 2018), although the importance of this
49 difference in environmental CH_4 is unclear.

50 Further research examining intra-site isotopic co-variation, which largely avoids complications associated with
51 estimating $\delta^2\text{H-H}_2\text{O}$, would help to more clearly resolve the relative importance of these processes, and how they vary
52 between environments. Expanded research using methyl fluoride to inhibit acetoclastic methanogenesis (Penning et al.,

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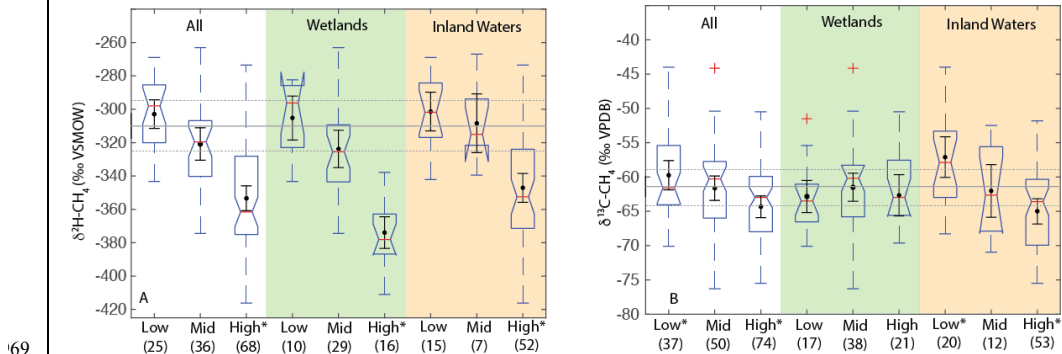
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53 2005; Penning and Conrad, 2007; Conrad et al., 2011), with a particular focus on $\delta^2\text{H-CH}_4$ measurements, would also help to
 54 clarify the importance of methanogenic pathway on isotopic co-variation. Finally, an expanded application of measurements
 55 of clumped isotopes, which have distinctive patterns of variation related to these processes (Douglas et al., 2016; Douglas et
 56 al., 2017; Young et al., 2017; Douglas et al., 2020), would also be of value in determining their relative importance in
 57 controlling $\delta^2\text{H-CH}_4$ values in freshwater environments.

58
 59 **3.5 Differences in $\delta^2\text{H-CH}_4$ and $\delta^{13}\text{C-CH}_4$ by latitude**

60 When analysing all sites together we found a significant difference in the distribution of $\delta^2\text{H-CH}_4$ between high-
 61 latitude sites (median: -351‰) and both low (median: -298‰) and mid-latitude sites (median: -320‰) (Fig. 7A). However,
 62 we did not find a significant difference in the distribution of low- and mid-latitude sites. Similar differences were found
 63 when the data were disaggregated into wetland and inland water sites. We also found that the distribution of $\delta^{13}\text{C-CH}_4$ for
 64 low latitude sites (median: -61.6‰) was significantly higher than for high latitude sites (median: -63.0‰), but that mid-
 65 latitude sites (median: -60.3‰) were not significantly different from the other two latitudinal zones (Fig. 7B). The observed
 66 difference by latitudinal zone in $\delta^{13}\text{C-CH}_4$ appears to be driven primarily by latitudinal differences between inland water
 67 sites, where a similar pattern is found. In wetland sites we found no significant differences in the distribution of $\delta^{13}\text{C-CH}_4$ by
 68 latitude.



70 **Figure 7: Boxplots of (A) $\delta^2\text{H-CH}_4$ and (B) $\delta^{13}\text{C-CH}_4$ for sites differentiated by latitude, for all data, wetlands, and inland waters.**
 71 Numbers in parentheses indicate the number of sites for each category. Red lines indicate medians, boxes indicate 25th and 75th
 72 percentiles, whiskers indicate 95th and 5th percentiles, and outliers are shown as red crosses. Notches indicate the 95% confidence
 73 intervals of the median value; where notches overlap the edges of the box this indicates the median confidence interval exceeds the
 74 75th or 25th percentile. Black points and error bars indicate the category mean and 95% confidence interval of the mean. Gray
 75 lines indicate the estimated flux-weighted mean values for global freshwater CH_4 , and dashed lines indicate the 95% confidence
 76 interval of this value. Asterisks in (A) indicate that high-latitude sites have significantly different distributions from other
 77 latitudinal bands. Asterisks in (B) indicate groups that have significantly different distributions from one another, within a specific
 78 environmental category. Two extremely low outliers ($<-80\text{‰}$, high latitude wetland and inland water) are not shown in (B).

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 Figure 5: Scatter plots of $\delta^2\text{H-CH}_{4, \text{w0}}$ vs. (A) $\delta^{13}\text{C-CH}_4$, (B) α_c , and (C) $\delta^{13}\text{C-CO}_2$. Predicted variation for variation in methanogenic pathway and CH_4 oxidation are shown by colored parallelograms, with details on predicted values in Supplementary Table 2 (Douglas et al., 2020). Significant ($p < 0.05$) piece-wise regression relationships are shown in (B) and (C), with breakpoints shown by dashed lines. No significant relationships were observed in (A). Gray envelopes indicate 95% confidence interv... [10]
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50 Estimates of flux-weighted mean freshwater $\delta^2\text{H-CH}_4$ and $\delta^{13}\text{C-CH}_4$, calculated using the Monte Carlo approach
51 described in Sect. 2.4, are $-310\pm 15\%$ (Fig. 7A) and $-61.5\pm 3\%$ (Fig. 7B) respectively. Flux weighted mean values for natural
52 wetlands (not including inland waters or rice paddies) are $-310\pm 25\%$ for $\delta^2\text{H-CH}_4$ and $-63.9\pm 3.3\%$ for $\delta^{13}\text{C-CH}_4$. Flux
53 weighted mean values for inland waters are $-309\pm 31\%$ for $\delta^2\text{H-CH}_4$ and $-60\pm 5.7\%$ for $\delta^{13}\text{C-CH}_4$. As discussed in Sect. 2.4
54 there are limited data in our dataset or that of Sherwood et al., (2017) from C_4 plant dominated wetlands, and therefore our
55 low-latitude and flux-weighted mean $\delta^{13}\text{C-CH}_4$ values for wetlands are probably biased towards low values.

56 Differences in $\delta^2\text{H-CH}_4$ by latitude has the potential to aid in geographic discrimination of freshwater methane
57 sources, both because it is based on a clear mechanistic linkage with $\delta^2\text{H-H}_2\text{O}$ (Figs. 3 and 4), and because geographic
58 variation in $\delta^2\text{H-H}_2\text{O}$ is relatively well understood (Bowen and Revenaugh, 2003; Bowen et al., 2005). However, recent
59 studies of atmospheric $\delta^2\text{H-CH}_4$ variation have typically not accounted for geographic variation in source signals. As an
60 example, Rice et al., (2016) apply a constant $\delta^2\text{H-CH}_4$ of -322% for both low-latitude ($0\text{-}30^\circ\text{N}$) and high latitude ($30\text{-}90^\circ\text{N}$)
61 wetland emissions. Based on our dataset this estimate is an inaccurate representation of wetland $\delta^2\text{H-CH}_4$ for either $0\text{-}30^\circ\text{N}$
62 (mean: $-305\pm 13\%$) or $30\text{-}90^\circ\text{N}$ (mean: $-345\pm 11\%$). Studies of ice core measurements have more frequently differentiated
63 freshwater $\delta^2\text{H-CH}_4$ values as a function of latitude. For example, Bock et al., (2010) differentiated $\delta^2\text{H-CH}_4$ between
64 tropical (-320%) and boreal (-370%) wetlands. This tropical wetland signature is significantly higher than our estimate of
65 low-latitude wetland $\delta^2\text{H-CH}_4$, although the boreal wetland signature is similar to our mean value for high-latitude wetlands
66 ($-374\pm 10\%$). Overall, our results imply that accounting for latitudinal variation in freshwater $\delta^2\text{H-CH}_4$, along with accurate
67 latitudinal flux estimates, is important for developing accurate estimates of global freshwater $\delta^2\text{H-CH}_4$ source signatures.

68 Our analysis indicates significant differences in the distribution of freshwater $\delta^{13}\text{C-CH}_4$ between the low- and high-
69 latitudes, but mid-latitude sites cannot be differentiated. Furthermore our results do not indicate significant latitudinal
70 differences in $\delta^{13}\text{C-CH}_4$ for wetland sites in particular, whereas we do observe significant differences between the low- and
71 high-latitudes for inland water sites. This is in contrast to previous studies that have inferred significant differences in
72 wetland $\delta^{13}\text{C-CH}_4$ by latitude (Bock et al., 2010; Rice et al., 2016; Ganesan et al., 2018). An important caveat is that we have
73 not analyzed a comprehensive dataset of freshwater $\delta^{13}\text{C-CH}_4$, for which there are much more published data than for $\delta^2\text{H-}$
74 CH_4 . However, our analysis does comprise the largest dataset of freshwater $\delta^{13}\text{C-CH}_4$ compiled to date (See Sect. 2.3). In
75 addition, our analysis does not take into account the geographic distribution of different ecosystem categories, although we
76 do not find significant differences in $\delta^{13}\text{C-CH}_4$ between ecosystem categories (Fig. 8; Sect. 3.6). Low-latitude ecosystems
77 dominated by C_4 plants are especially underrepresented both in our dataset and that of Sherwood et al., (2017), and
78 accounting for this would likely lead to a more enriched low-latitude wetland $\delta^{13}\text{C-CH}_4$. In contrast, high-latitude
79 ecosystems, including bogs, are relatively well represented in these datasets (Fig. 8), and we suggest that inferences of
80 especially low $\delta^{13}\text{C-CH}_4$ in high-latitude wetlands (Bock et al., 2010; Rice et al., 2016; Ganesan et al., 2018) are not
81 consistent with the compiled dataset of in-situ measurements. However, we note that atmospheric estimates of high-latitude

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.01 wetland $\delta^{13}\text{C-CH}_4$ ($\sim -68 \pm 4\%$) (Fisher et al., 2011) are lower than the median or mean value shown in Figure 7B, and are in
.02 close agreement with the relatively low values predicted by (Ganesan et al., 2018). Ombrotrophic and minerotrophic
.03 peatlands have distinctive $\delta^{13}\text{C-CH}_4$ signatures (Bellisario et al., 1999; Bowes and Hornibrook, 2006; Hornibrook, 2009), with
.04 lower signatures in ombrotrophic peatlands. We did not differentiate peatlands by trophic status, and it is possible that the
.05 dataset of high-latitude wetland in-situ measurements is biased towards minerotrophic peatlands with relatively high $\delta^{13}\text{C-}$
.06 CH_4 .

.07 Latitudinal differences in $\delta^{13}\text{C-CH}_4$ inferred by Ganesan et al. (2018) were based on two key mechanisms: (1)
.08 differences in methanogenic pathway between different types of wetlands, especially between minerotrophic fens and
.09 ombrotrophic bogs; and (2) differential inputs of organic matter from C_3 and C_4 plants. Because inferred latitudinal
.10 differences in $\delta^{13}\text{C-CH}_4$ and $\delta^2\text{H-CH}_4$ are caused by different mechanisms, they could be highly complementary in validating
.11 estimates of freshwater emissions by latitude. It is also important to note that previous assessments of latitudinal differences
.12 in $\delta^{13}\text{C-CH}_4$ did not include inland water environments. Our analysis suggests that latitudinal variation in $\delta^{13}\text{C-CH}_4$ in inland
.13 waters may be more pronounced than in wetlands, although the mechanisms causing this difference will need to be
.14 elucidated with further study. A benefit of geographic discrimination based on $\delta^2\text{H-CH}_4$ is that the same causal mechanism
.15 applies to all freshwater emissions, including both wetlands and inland waters.

.16 3.5.1 Potential for geographic discrimination of other microbial methane sources based on $\delta^2\text{H-CH}_4$

.17 We speculate that latitudinal differences in $\delta^2\text{H-CH}_4$ should also be observed in other fluxes of microbial methane from
.18 terrestrial environments, including enteric fermentation in livestock and wild animals, manure ponds, landfills, and termites.
.19 This is because microbial methanogenesis in all of these environments will incorporate hydrogen from environmental water,
.20 and therefore will be influenced by variation in precipitation $\delta^2\text{H}$. There are limited data currently available to test this
.21 prediction, but $\delta^2\text{H-CH}_4$ data from cow rumen and landfills are available with either specified locations or $\delta^2\text{H-H}_2\text{O}$ (Burke
.22 Jr, 1993; Levin et al., 1993; Liptay et al., 1998; Bilek et al., 2001; Wang et al., 2015; Teasdale et al., 2019). These data plot in a
.23 range that is consistent with the $\delta^2\text{H-CH}_4$ vs. $\delta^2\text{H-H}_2\text{O}$ relationships for freshwater CH_4 (Fig. 5). Landfill data are only
.24 available for a very small range of estimated $\delta^2\text{H-H}_2\text{O}$, making it impossible to assess for geographic variation currently.
.25 $\delta^2\text{H-CH}_4$ data from cow rumen span a much wider range, and express substantial variation that is independent of $\delta^2\text{H-H}_2\text{O}$.
.26 However, the cow rumen data span a range that is similar to that observed in freshwater environments. Based on these
.27 limited data, variation observed in incubation studies that simulate landfill conditions (Schoell, 1980; Waldron et al., 1998),
.28 and our understanding of the influence of $\delta^2\text{H-H}_2\text{O}$ on microbial $\delta^2\text{H-CH}_4$ (Fig. 6), we suggest that both landfill and cow
.29 rumen $\delta^2\text{H-CH}_4$ likely vary geographically as a function of $\delta^2\text{H-H}_2\text{O}$. If validated, this variation could also be used to
.30 distinguish these CH_4 sources geographically. More data are clearly needed to test this conjecture, and it will also be
.31 important to evaluate how closely annual or seasonal $\delta^2\text{H}_2$ corresponds to environmental $\delta^2\text{H-H}_2\text{O}$ in both landfills and cow

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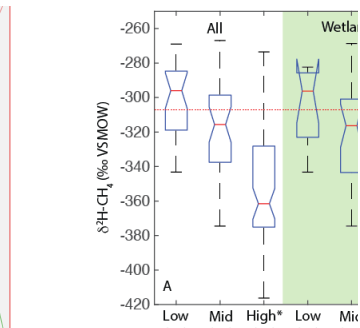
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35 rumen. Similarly, the $\delta^2\text{H}$ of CH_4 emitted by biomass burning or directly by plants has also been shown to vary as a function
36 of $\delta^2\text{H}\text{-H}_2\text{O}$ (Vigano et al., 2010; Umezawa et al., 2011).

37 3.6 Differences in $\delta^2\text{H}\text{-CH}_4$ and $\delta^{13}\text{C}\text{-CH}_4$ by ecosystem

39 When comparing ecosystems, we analyze $\delta^2\text{H}\text{-CH}_{4,\text{w0}}$ values to account for variability related to differences in $\delta^2\text{H}\text{-H}_2\text{O}$.
40 Ecosystem types are not evenly distributed by latitude, and therefore have different distributions of $\delta^2\text{H}\text{-H}_2\text{O}$ values. There
41 are differences in the median values by ecosystem, with rivers (-283‰) exhibiting relatively enriched median $\delta^2\text{H}\text{-CH}_{4,\text{w0}}$,
42 and fens (-314‰) and rice paddies (-314‰) exhibiting relatively low median values. However, given the small sample sizes
43 and large variance in most of these categories, our analysis does not infer a significant difference in the distribution of $\delta^2\text{H}\text{-}$
44 $\text{CH}_{4,\text{w0}}$ between ecosystems. Comparing the broader categories of inland waters and wetlands with a we do find a significant
45 difference in $\delta^2\text{H}\text{-CH}_{4,\text{w0}}$ distributions, with inland waters shifted towards higher values (median: -296‰) than wetlands
46 (median: -311‰). We repeated this analysis only including sites with measured $\delta^2\text{H}\text{-H}_2\text{O}$ and found the same results
47 in terms of category differences (Supplemental Figure S1). We did not observe any significant differences in $\delta^{13}\text{C}\text{-CH}_4$
48 distributions between ecosystems, nor was there a significant difference in $\delta^{13}\text{C}\text{-CH}_4$ distributions between inland waters
49 and wetlands. The median $\delta^{13}\text{C}\text{-CH}_4$ value for bogs was relatively low (-66‰), while median values for fens (-60.3‰) and
50 rice paddies (-60.3‰) were relatively high, but there was a large range in values for all of these ecosystems.

51 The significant difference in the distribution of $\delta^2\text{H}\text{-CH}_{4,\text{w0}}$ between the overarching categories of inland waters and
52 wetlands is primarily a result of the difference in $\delta^2\text{H}\text{-CH}_4$ between these environments in the high latitudes (Figs. 3, 4, and
53 7). We are unsure of the mechanism causing this difference, though it is likely related to a greater overall prevalence of CH_4
54 oxidation in inland waters. As shown in Figure 6, the lack of positive co-variation between $\delta^2\text{H}\text{-CH}_{4,\text{w0}}$ and $\delta^{13}\text{C}\text{-CO}_2$, and
55 α_{C} could be interpreted to support a greater role for CH_4 oxidation to control $\delta^2\text{H}\text{-CH}_{4,\text{w0}}$ in inland waters relative to
56 wetlands, although this result requires further validation. In lakes that undergo seasonal overturning and water column
57 oxygenation there may be a greater overall effect of CH_4 oxidation than there are in wetlands typically. The absence of
58 significant differences in $\delta^2\text{H}\text{-CH}_{4,\text{w0}}$ distributions between specific ecosystem categories could be the result of small
59 samples sizes for most ecosystems. Further study could be targeted towards verifying and testing the apparent differences
60 shown in Figure 8A. Generally lower $\delta^2\text{H}\text{-CH}_{4,\text{w0}}$ in rice paddies and fens could reflect a greater proportion of acetoclastic
61 methanogenesis inferred for these ecosystems (Conrad and Klose, 1999; Hornibrook, 2009; Ganesan et al., 2018), or possibly
62 more thermodynamically favorable methanogenesis related to high carbon substrate, H_2 , or nutrient concentrations. Both of
63 these explanations would be consistent with the relatively high median $\delta^{13}\text{C}\text{-CH}_4$ values in these ecosystems (Fig. 7B, see
64 also Fig. 6A). High median values in river ecosystems, in contrast, may be a function of generally greater rates of oxidation,
65 given that these environments are also characterized by relatively high $\delta^{13}\text{C}\text{-CH}_4$ (Fig. 8B), and the potential for greater



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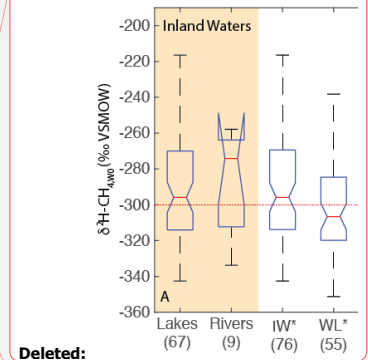
56 water-column oxygenation in fluvial environments with turbulent flow (Devol et al., 1987). However, our river dataset is
 57 highly biased towards the Amazon river basin, and drawing firm conclusions will require a larger and more widely
 58 distributed dataset.

59 The absence of significant differences between ecosystems in terms of $\delta^{13}\text{C-CH}_4$ (Fig. 8B) is in contrast to previous
 60 studies that have suggested that fens and bogs in particular have distinctive $\delta^{13}\text{C-CH}_4$ (Ganesan et al., 2018). Bogs in
 61 particular have a very wide distribution of $\delta^{13}\text{C-CH}_4$ that could represent differences between minerotrophic and
 62 ombrotrophic bogs (Hornibrook, 2009), which we did not differentiate in our dataset. This result should be interpreted with
 63 caution given that our dataset is not a comprehensive compilation of published $\delta^{13}\text{C-CH}_4$ data, although it is the largest
 64 compiled dataset available (Sect. 2.3). We argue that inferred differences in $\delta^{13}\text{C-CH}_4$ between wetland ecosystem categories
 65 should be further verified with more comprehensive data assimilation and additional measurements.

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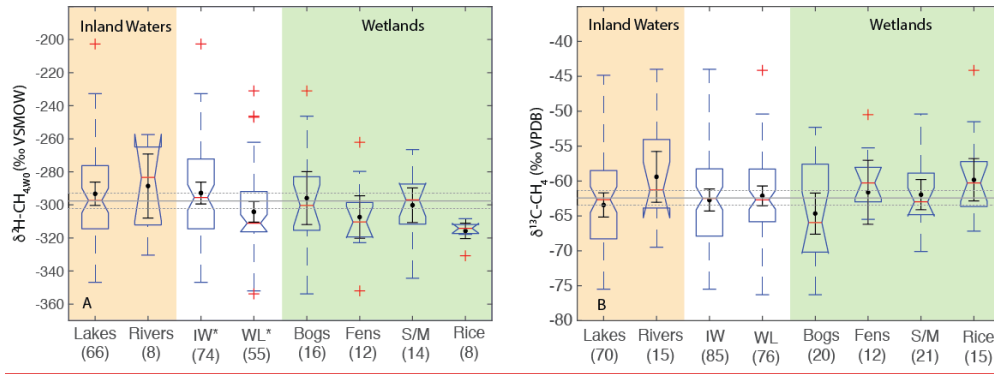


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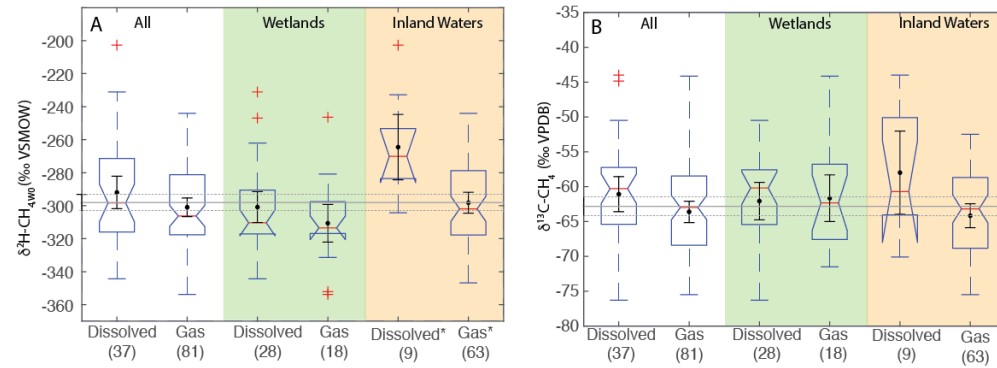
66 **Figure 5:** Boxplots of (A) $\delta^2\text{H-CH}_{4,w0}$ and (B) $\delta^{13}\text{C-CH}_4$ for sites differentiated by ecosystem type. Numbers in parentheses
 67 indicate the number of sites for each category. Boxplots parameters are as in Fig. 7. Black points and error bars indicate the
 68 category mean and 95% confidence interval of the mean. Gray lines indicate the mean values across all categories and the dashed
 69 lines indicate the 95% confidence interval of this value. Two extremely low outliers ($<-8\text{‰}$; lake and fen) are not shown in (B).
 70 IW- Inland Waters; WL- Wetlands; S/M- Swamps and marshes. Asterisks in A indicate that inland waters and wetlands have
 71 significantly different distributions.

72 **3.7. Differences in $\delta^2\text{H-CH}_4$ and $\delta^{13}\text{C-CH}_4$ by sample type**

74 As with comparing ecosystems, when comparing sample types we analyze $\delta^2\text{H-CH}_{4,w0}$ values to normalize for variability
 75 related to differences in $\delta^2\text{H-H}_2\text{O}$, since sample types are not distributed evenly by latitude. When comparing sample types,
 76 dissolved CH_4 samples do not have a significantly different $\delta^2\text{H-CH}_{4,w0}$ distribution for the dataset as a whole, nor is there a
 77 significant difference between these groups in wetland sites (Fig. 9A). There is, however, a significant difference in inland

94 water sites, with dissolved CH₄ samples having a more enriched distribution (median: -270‰) vs. gas samples (median: -
 95 302‰). We repeated this analysis only including sites with measured δ²H-H₂O and found the same results in terms of
 96 category differences (Supplemental Figure S2). We did not observe a significant difference in the distribution of δ¹³C
 97 between dissolved and gas-phase CH₄ samples, either for the dataset as a whole or when the dataset was disaggregated into
 98 wetlands and inland waters (Fig. 9B). We suggest that the higher δ²H-CH_{4,w0} in dissolved vs. gas samples for inland waters
 99 could be a result of generally greater oxidation of dissolved CH₄ in inland water environments, potentially as a result of
 00 longer exposure to aerobic conditions in lake or river water columns. This is in contrast to wetlands, where aerobic
 01 conditions are generally limited to the uppermost layers of wetlands proximate to the water table. However, our dataset for
 02 inland water dissolved CH₄ is quite small (n=9), and more data are needed to test this hypothesis. Furthermore, it is unclear
 03 why oxidation in inland water dissolved CH₄ would be more strongly expressed in terms of δ²H-CH_{4,w0} (Fig. 9A) than δ¹³C
 04 (Fig. 9B).

05 Overall, our data imply that isotopic differences between dissolved and gas phase methane are relatively minor on a
 06 global basis, especially in wetlands. This result could imply that the relative balance of diffusive vs. ebullition gas fluxes
 07 should not have a large effect on the isotopic composition of freshwater CH₄ emissions. However, our study does not
 08 specifically account for isotopic fractionation occurring during diffusive or plant-mediated transport (Hornibrook, 2009), and
 09 most of our dissolved sample data are of *in-situ* dissolved CH₄ and not diffusive fluxes. More isotopic data specifically
 10 focused on diffusive methane emissions, for example using measurements of gas sampled from chambers, would help to
 11 resolve this question, as would more comparisons of the isotopic composition of diffusive and ebullition CH₄ emissions from
 12 the same ecosystem.



13 **Figure 9:** Boxplots of (A) δ²H-CH_{4,w0} and (B) δ¹³C-CH₄ for sites differentiated by sample type. Numbers in parentheses indicate
 14 the number of sites for each category. Boxplots parameters are as in Fig. 7. Black points and error bars indicate the category mean
 15 and 95% confidence interval of the mean. Gray lines indicate the mean values across all categories and the dashed lines indicate
 16 the 95% confidence interval of this value. Two extremely low outliers (<-80‰; dissolved wetland and gas inland water) are not
 17 shown in (A).

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32 shown in (B). Asterisks in A indicate that dissolved and gas-phase CH₄ samples from inland water sites have significantly different
33 distributions.

34 3.8 Estimates of global emissions source δ²H-CH₄ and δ¹³C-CH₄

35 Our mixing model and Monte Carlo analyses estimate a global source δ²H-CH₄ of -27.8±1.5‰, and a global source δ¹³C-CH₄
36 of -56.4±2.6‰ (Fig. 10). Monte Carlo sensitivity tests that only included uncertainty in either isotopic source signatures or
37 flux estimates suggest that larger uncertainty is associated with isotopic source signatures (1.2‰ for δ²H; 2.2‰ for δ¹³C) than
38 with flux estimates (8‰ for δ²H; 1.4‰ for δ¹³C). When correcting for wetland and biomass burning emissions from C₄ plant
39 ecosystems, as described in Section 2.4, our estimate of global source δ¹³C-CH₄ increases to -55.2±2.6‰.

40 Our estimate of global source δ²H-CH₄ is substantially higher than a previous bottom-up estimate using a similar
41 approach (-295‰; Fig. 10) (Whiticar and Schaefer, 2007). This difference can be largely attributed by the application of
42 more depleted δ²H-CH₄ source signatures for tropical wetlands (-360 ‰), and to a lesser extent boreal wetlands (-380 ‰), by
43 Whiticar and Schaefer (2007). Another key difference is their inclusion of a relatively large flux and enriched δ²H-CH₄
44 signature from aerobic methane production from plants by Whiticar and Schaefer (2007), which is not included as a CH₄
45 source in our calculations.

46 Our bottom-up estimate of global source δ²H-CH₄ substantially overlaps the range of top-down estimates (-258 to -
47 289‰) based on atmospheric δ²H-CH₄ measurements from 1977-2005 and a box model of sink fluxes and kinetic isotope
48 effects (Rice et al., 2016) (Fig. 10). It is also within error of simpler top-down estimates calculated based on mean
49 atmospheric measurements and estimates of a constant sink fractionation factor (Whiticar and Schaefer, 2007; Sherwood et
50 al., 2017). Sherwood et al., (2017) estimate a very wide range of possible global source δ²H-CH₄ values based on a relatively
51 large atmospheric sink fractionation with large uncertainty (-235±80‰). This range overlaps with our bottom up estimate,
52 although its mid-point is substantially lower than our estimate. We argue that the box-model method used to account for sink
53 fractionations applied by Rice et al. (2016) probably provides a more accurate representation of global-source isotopic
54 composition than the other top-down estimates shown in Figure 10 (Whiticar and Schaefer, 2007; Sherwood et al., 2017).
55 The estimates of Rice et al. (2016) are also supported by the results of a global inversion model. Overall, the overlap
56 between our bottom-up estimate of global source δ²H-CH₄ with top-down estimates is encouraging, and suggests that the
57 estimates of emission source δ²H-CH₄ signatures applied in this study are reasonably accurate. However, as discussed below,
58 there is still substantial room to further constrain these estimates and to reduce uncertainty.

59 Our bottom-up estimate of global source δ¹³C-CH₄ is lower than the other top-down and bottom-up estimates shown
60 in Figure 10. As discussed above, there is likely a bias in our freshwater CH₄ isotopic database in that it includes very few
61 wetland sites from C₄-plant dominated ecosystem. When correcting for this, as well as CH₄ emissions from biomass burning
62 (Fig. 10), our estimate shifts to a more enriched value that is within uncertainty of other estimates. Clearly, accounting for
63 the effect of C₄ plants in wetland and biomass burning CH₄ emissions, and potentially also in enteric fermentation emissions,

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.76 is important for accurate estimates of global source $\delta^{13}\text{C}\text{-CH}_4$. As discussed below, other sources of error in both isotopic
.77 source signatures and inventory-based flux estimates could also partially account for our relatively low global source $\delta^{13}\text{C}\text{-}$
.78 CH_4 estimate.

.79 Previous studies have argued, on the basis of comparing atmospheric measurements and emissions source $\delta^{13}\text{C}\text{-CH}_4$
.80 signatures, that there are biases in global emissions inventories, specifically that fossil fuel emissions estimates are too low,
.81 and that either microbial emissions estimates are too high (Schwietzke et al., 2016), or that biomass burning estimates are too
.82 high (Worden et al., 2017). We argue that greater analysis of $\delta^2\text{H}\text{-CH}_4$ measurements could be valuable for evaluating these
.83 and other emissions scenarios, as has been suggested previously (Rigby et al., 2012). This is especially true for determining
.84 the relative proportion of fossil fuel and microbial emissions, since these sources have widely differing $\delta^2\text{H}\text{-CH}_4$ signatures
.85 (Table 1). Currently, atmospheric $\delta^2\text{H}\text{-CH}_4$ measurements are not a routine component of CH_4 monitoring programs, but we
.86 argue that based on both their value in constraining emissions sources and sinks (Rigby et al., 2012), and the increasing
.87 practicality of high-frequency measurements (Chen et al., 2016; Röckmann et al., 2016; Yacovitch et al., 2020), that there
.88 should be a renewed focus on these measurements.

.89 The uncertainty in our bottom-up estimates, the overall greater uncertainty associated with isotopic source
.90 signatures in our Monte Carlo calculations, and the apparent discrepancies for $\delta^{13}\text{C}\text{-CH}_4$ shown in Figure 10, also imply that
.91 isotopic source signatures for specific sources could be greatly improved. As noted by Rigby et al. (2012), the impact of
.92 improved isotopic source signatures increases as measurement precision improves. We have discussed above the importance
.93 of increased data assimilation and measurements from tropical wetlands, with a particular focus on C_4 plant dominated
.94 ecosystems. Using the isotopic source signal uncertainties and emissions fluxes shown in Table 1, we identified the sources
.95 with the greatest flux-weighted uncertainty in isotopic signatures. Based on this analysis, the greatest uncertainty for global
.96 source $\delta^2\text{H}\text{-CH}_4$ estimates comes from source signatures for enteric fermentation and manure, low-latitude wetlands, onshore
.97 geological emissions, low-latitude and mid-latitude inland waters, termites, and landfills. We identified the same source
.98 categories as having the greatest flux-weighted uncertainty for $\delta^{13}\text{C}\text{-CH}_4$, with the exception of termites, but repeat the
.99 caveat that the underlying dataset is less comprehensive for $\delta^{13}\text{C}\text{-CH}_4$. We argue that these source categories should be
.00 considered priorities for future emissions source isotopic characterization through data assimilation and additional
.01 measurements. In particular, as discussed in Sect. 3.5.1, evaluation of possible latitudinal variation in enteric fermentation
.02 and landfill $\delta^2\text{H}\text{-CH}_4$ is particularly promising.

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Figure 9: Boxplots of (A) $\delta^2\text{H}\text{-CH}_{4,\text{w0}}$ and (B) $\text{CH}_{4,\text{w0}}$ and (B) $\delta^{13}\text{C}\text{-CH}_4$ for sites differentiated by sample type. Boxplots parameters are as in Fig. 7. Red dashed lines indicate the median values for all sites. One extremely high outlier (-160‰, gas inland water) is not shown in (A). Two extremely low outliers (<-84‰; dissolved wetland and gas inland water) are not shown in (B). ... [26]

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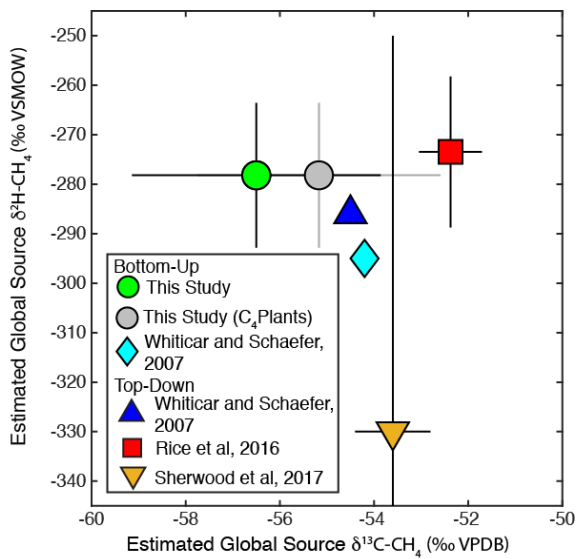


Figure 6. Comparison of estimates of dual-isotope global source $\delta^2\text{H-CH}_4$ and $\delta^{13}\text{C-CH}_4$ from this and previous studies. Error bars from this study indicate the 2σ standard deviation from Monte Carlo analysis. Gray dot and error bars indicate an estimate corrected for the lack of data from wetlands and biomass burning in C_4 plant environments, as described in Sect. 2.4. Error bars for Rice et al., (2016), indicate the range of values estimated in that study between 1977-2005. Error bars for Sherwood et al., (2017) reflect the combined measurement uncertainty and uncertainty in sink fractionations reported in that study. Whiticar and Schaefer (2007) did not provide uncertainties for their estimates.

5 Conclusions

Our analysis of an expanded isotopic dataset for freshwater CH_4 confirms the previous finding that $\delta^2\text{H-H}_2\text{O}$ is the primary determinant of $\delta^2\text{H-CH}_4$ on a global scale (Waldron et al., 1999a), but also finds that the slope of this relationship is probably flatter than was inferred previously (Fig. 3). This flatter slope is primarily the result of the inclusion of a much larger number of high-latitude sites with low $\delta^2\text{H-H}_2\text{O}$ in our dataset. We find that the inferred relationship between $\delta^2\text{H-CH}_4$ and $\delta^2\text{H-H}_2\text{O}$ is not highly sensitive to whether measured $\delta^2\text{H-H}_2\text{O}$, modeled $\delta^2\text{H}_p$, or a combination of the two (i.e. a best-estimate) is used to estimate $\delta^2\text{H-H}_2\text{O}$. This implies that gridded datasets of $\delta^2\text{H}_p$ or isotope-enabled climate models could be used to predict the distribution of $\delta^2\text{H-CH}_4$ in the present, as well as under past and future climates. Our analysis also suggests that annual $\delta^2\text{H}_p$ may be a better predictor for wetland $\delta^2\text{H-CH}_4$, while seasonal $\delta^2\text{H}_p$ may be a better predictor of inland water $\delta^2\text{H-CH}_4$.

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 Deleted: (B), and dual isotope signatures (C). Histograms in (A) and (B) show probability distributions from Monte-Carlo analyses. Colored lines indicate estimated isotopic trends for increasing emissions from specific sources, with all other emissions sources remaining constant. Colored vectors in (C) indicate the dual-isotope trend for increasing emissions fluxes from specific sources by 40 Tg/yr, with all other emissions sources remaining constant. Gray lines and points indicate top-down estimates of emissions source isotopic signatures from 1977 to 2005 (Ric ... [28]
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The slope of $\delta^2\text{H-CH}_4$ vs. $\delta^2\text{H-H}_2\text{O}$ in both wetlands and inland waters agrees well with that found in incubation experiments (Schoell, 1980; Sugimoto and Wada, 1995; Waldron et al., 1998; Waldron et al., 1999a), and we concur with previous inferences that this slope is partly controlled by variation in the $\delta^2\text{H}$ of acetate as a function of $\delta^2\text{H-H}_2\text{O}$ (Waldron et al., 1999a). Analysis of co-variation of $\delta^2\text{H-CH}_4$ with $\delta^{13}\text{C-CH}_4$, $\delta^{13}\text{C-CO}_2$, and α_C suggest that residual variation in $\delta^2\text{H-CH}_4$ is influenced by a complex set of biogeochemical variables, including both variable isotopic fractionation related to methanogenesis, and post-production isotopic fractionation related to CH_4 oxidation and diffusive gas transport. A significant positive correlation between $\delta^2\text{H-CH}_4$ and both $\delta^{13}\text{C-CO}_2$ and α_C in wetlands suggests that variable fractionation related to methanogenesis pathway and thermodynamics may be more important in these environments, but this result is dependent on the method used to estimate $\delta^2\text{H-H}_2\text{O}$ and requires further validation.

The dependence of $\delta^2\text{H-CH}_4$ on $\delta^2\text{H-H}_2\text{O}$ leads to clear latitudinal differences in $\delta^2\text{H-CH}_4$, with particularly low values from high latitude sites (Fig. 4; Fig. 7A). The mechanism for latitudinal differences in $\delta^2\text{H-CH}_4$ is distinct from proposed mechanisms for latitudinal differences in $\delta^{13}\text{C-CH}_4$ (Ganesan et al., 2018), implying that these two isotopic tracers are complementary in differentiating geographic emissions sources. We estimate a global flux-weighted $\delta^2\text{H-CH}_4$ signature from freshwater environments of $-3.10 \pm 1.5\%$, which is enriched relative to values used in previous source apportionment studies (Rice et al., 2016; Bock et al., 2017). We observe a significantly higher $\delta^2\text{H-CH}_4$ distribution in inland waters relative to wetlands (Fig. 8A), which we suggest is a result of greater rates of CH_4 oxidation. We do not find significant differences between more specific ecosystem categories, but there are apparent differences between some wetland ecosystems that could be verified with larger datasets. We also do not find significant differences in $\delta^2\text{H-CH}_4$ between sample types (Fig. 9A), with the exception of higher values in dissolved CH_4 relative to gas-phase CH_4 in inland water environments.

Our bottom-up estimate of the global $\delta^2\text{H-CH}_4$ source signature, $-27.8 \pm 1.5\%$, is higher than previous bottom-up estimates (Whiticar and Schaefer, 2007), but is within the range of top-down estimates based on atmospheric measurements and modeled sink fractionations (Rice et al., 2016). In contrast, our bottom-up estimate of global $\delta^{13}\text{C-CH}_4$, $-56.4 \pm 2.6\%$, is low relative to top-down estimates, which is partially explained by a lack of data from C_4 plant-dominated ecosystems in the freshwater CH_4 isotopic dataset. The agreement between bottom-up and top-down global $\delta^2\text{H-CH}_4$ estimates suggests that our current understanding of $\delta^2\text{H-CH}_4$ source signatures, when combined with inventory-based flux estimates (Saunois et al., 2020), is consistent with atmospheric measurements. This supports the argument that increased measurements and modeling of atmospheric $\delta^2\text{H-CH}_4$ could help to constrain global CH_4 budgets (Rigby et al., 2012). However, there is clearly a need to better constrain source signatures for both $\delta^2\text{H-CH}_4$ and $\delta^{13}\text{C-CH}_4$, especially from low-latitude microbial sources.

Data Availability: The datasets used in this paper (Supplementary Tables 1-4) are publicly available: Douglas, Peter; Stratigopoulos, Emerald; Park, Jenny; Phan, Dawson (2020): Data for geographic variability in freshwater methane

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'63 hydrogen isotope ratios and its implications for [global isotopic source signatures](https://doi.org/10.6084/m9.figshare.13194833.v1),
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'66 **Author Contribution:** PMJD designed the project, assisted with compiling the data, analyzed the data, and wrote the
'67 manuscript; ES and JP compiled the data, and assisted with analyzing the data and editing the manuscript; DP developed
'68 code for mixing model and Monte Carlo calculations, and assisted with analyzing the data and editing the manuscript.

'70 **Competing Interests:** The authors declare they have no competing interests.

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'73 Table 1). [We also thank Susan Waldron, Edward Hornibrook, and an anonymous reviewer for constructive feedback.](#) This
'74 research was partially funded by McGill Science Undergraduate Research Awards to ES and JP and by NSERC Discovery
'75 Grant 2017-03902 to PMJD.

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