Critical inundation level for methane emissions from wetlands

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Abstract

Global methane (CH$_4$) emissions have reached approximately 600 Tg per year, 20%–40% of which are from wetlands. Of the primary factors affecting these emissions, the water table level is among the most uncertain. Here we conduct a global meta-analysis of chamber and flux-tower observations of CH$_4$ emissions and employ a novel mechanistic model to show that wetlands have maximum emissions at a critical level of inundation and discuss its origin. This maximum arises from an interplay between methanogenesis, methanotrophy, and transport, whose rates vary differently with the inundation level. The specific location of the critical water level above the soil surface may differ depending on wetland characteristics, for example temperature or the presence of macrophytes with aerenchyma. However, data suggest that globally a water level of about 50 cm is the most favorable to CH$_4$ emissions. Keeping the water level away from this critical value could reduce methane emissions in human-made wetlands, which comprise at least one fifth of the global wetland area.

1. Introduction

Increased atmospheric CH$_4$ concentrations are the result of an imbalance between sources and sinks (Kirschke et al. 2013, Melton et al. 2013, Saunois et al. 2016), raising concerns for the Earth’s energy balance (Ringeval et al. 2011, Berner and Berner 2012). Environmental conditions, such as temperature (Schütz et al. 1999, Oertel et al. 2016) and water table level (Moore and Dalva 1993, Kelley et al. 1995), affect the CH$_4$ budget in soils, impacting both production, during anaerobic digestion of organic matter (resulting in biological methane production by methanogens), consumption (i.e., biological oxidation by methanotrophs), and transport to the atmosphere (Le Mer and Roger 2001, Brady and Weil 2016). The hydrologic regime affects all terms of the methane budget from formation to emission, but its overall role is still unclear and represents a major uncertainty in the estimation of soil contribution to the CH$_4$ global budget (Bloom et al. 2010, Saunois et al. 2020).

According to recent estimates, wetlands are one of the largest single CH$_4$ sources (on average 145 and 194 Tg CH$_4$ per year from bottom-up and top-down approaches, respectively), but is also one of the most uncertain, along with the other natural sources (Jackson et al. 2020, Saunois et al. 2020). Furthermore, freshwater systems, such as lakes and ponds, contribute 159 Tg CH$_4$ per year, while rice cultivation (typically considered an anthropogenic source) adds another 30 Tg CH$_4$ each year (Saunois et al. 2020). Despite the dominant role of these sources in the atmospheric CH$_4$ budget, synthesizing the role of various environmental factors in regulating CH$_4$ emissions across sites has been challenging. It has been particularly difficult to explain the effect of the water table level, when this is above the soil surface (i.e., inundation), with observations often showing contrasting trends across different wetland and ecosystem types (Turetsky et al. 2014, Knox et al. 2019).

To better understand the role of the water table level on regulating CH$_4$ emissions from wetland...
systems, we performed a global analysis of CH \(_4\) emissions as a function of inundation level, informed by a mechanistic model of CH \(_4\) formation, decay and emission. Supported by field observations in a variety of environmental settings, our analysis reveals and explains the maximum CH \(_4\)-emission rate occurring for a critical level of inundation. This critical level originates from two competing effects, one related to the availability of methanogenic substrates as influenced by the saturated portion of soil column and the other linked to the gas transport in and out of the soil. These effects limit CH \(_4\) emissions above and below the critical level.

While the presence of this critical water table level is interesting from the fundamental standpoint of the underlying biogeochemical processes, it is also useful for improving wetland CH \(_4\) emissions and climate projections and for informing water management strategies, which could maintain the water level away from this critical value. In rice fields, this strategy has the potential to provide substantial benefits in addition to the ones deriving from traditional strategies, or to suggest alternative strategies to fully draining and re-inundating the soil that avoid fertility losses and limit nitrous oxide (N\(_2\)O) emissions (Hu et al 2017).

2. Experimental evidence

We examined the existing literature relating CH \(_4\) emission to the water level. We considered both chamber and flux tower measurements for sites encompassing a wide range of latitudes, temperature, substrate availability, and water table regimes. Sites where CH \(_4\) emissions were measured through chambers (42 sites) include marshes, wetprairies and sawgrass marshes along the Shark River Slough in the Everglades (Florida, USA) (Harriss et al 1988, Li and Mitsch 2016), cypress swamps in the Southeastern United States (South Carolina, Georgia, Florida) (Harriss and Sebacher 1981, Villa and Mitsch 2014), as well as coastal marshes, fen and bog wetlands, and peatlands in Ontario and northern Québec (Canada) (Moore and Roulet 1993, Moore et al 1994), boreal ponds and lakes in Finland (Huttunen et al 2003, Kankaala et al 2013), United Kingdom (Casper et al 2000), Québec (Canada) (Bartosiewicz et al 2015), Antarctica (Zhu et al 2010), Pantanal region (South America) (Bastviken et al 2010), North India (Singh et al 2000), and the Colorado Rockies (USA) (Smith and Lewis Jr 1992).

Flux tower measurements were retrieved from FLUXNET-CH \(_4\) (Knox et al 2019), which has assembled a global CH \(_4\) database from numerous eddy-covariance sites (38) that also span different wetland/ecosystem types. These sites include wet tundras, peat plateaus, bogs, fens, swamps, agricultural wetlands, upland soils, salt marshes, and water bodies.

The spatial distribution of the sites is illustrated in figure 1, while a complete list of the sites and their characteristics is provided in a supplementary table (available online at https://doi.org/10.4211/hs.c7050702af674a7a88068a6fd48450f4). There is a higher representation of sites in temperate and boreal ecosystems, but we show in the discussion that temperature does not alter our main conclusion on the existence and location of the critical water level.

The interactions between all biological (e.g. activities of microbial communities and plants), geochemical (e.g. redox potential, labile C content and form), and hydro-climatic factors (e.g. water level, temperature) affecting CH \(_4\) production, oxidation, and emission generate highly variable CH \(_4\) dynamics, with fluctuations in CH \(_4\) emissions at various frequencies. The presence of such multi-frequency fluctuations makes it hard to interpret experimental observations and understand the key features of CH \(_4\) dynamics. Therefore, to capture the dominant dependence of
CH$_4$ emissions on water level, we averaged out small timescale fluctuations (hourly to daily) and focused on the average CH$_4$ emissions with respect to the average water level (see also appendix A). Accordingly, we temporally averaged CH$_4$ emissions and corresponding water level depending on data availability (see table S1). These temporal averages were generally computed over a season (e.g. wet season or ice-free season) for chamber measurements and over the year for flux tower measurements. However, in some datasets, e.g. Moore and Roulet (1993) and Moore et al (1994), only single daily averages were available.

The results (shown in figure 2 in linear and log-linear scales) strongly support the widespread existence and influence of a critical water table level, approximately 50 cm above the soil surface, for which CH$_4$ emissions are maximum. Emission rates sharply increase up to the critical level, followed by a slower decrease. Especially from the critical water level up, observations tend to scatter, reflecting the considerably different conditions among sites, such as vegetation type, temperature, and biome. Below, we interpret these observations based on our quantitative understanding of the processes of CH$_4$ production, oxidation, and transport to the atmosphere.

3. Origin of the critical water level

We introduce an analytical model to interpret the above observations of CH$_4$ emissions and water table level. The model is described here conceptually, while the mathematical details are provided in appendix A and B.

Consider an inundated soil as in figure 3(a). The level of the water table is identified with $h_{w}$, measured with reference to the soil surface. We assume for simplicity that the system is homogeneous with respect to the physical and chemical characteristics of the soil, including the available substrate for microbial degradation and temperature (appendix A). Such simplifications are made to facilitate our conceptual model development, but do not affect the main conclusions.

We first consider CH$_4$ production by methanogenesis, as illustrated in figure 3(b). Since it occurs in the submerged, anoxic soil layers, CH$_4$ production greatly depends on the vertical extension of the saturated zone determined by the water table level, along with other physico-chemical factors (Yavitt et al 2000, Le Mer and Roger 2001, Whalen 2005). Higher water table levels imply that a greater portion of soil column is exposed to anoxic conditions (due to drawdown of atmospheric derived O$_2$ from microbial respiration), a fact that favors methanogenesis. As the level of the water table is increased, methanogenesis reaches the highest rates when the water level coincides with the soil surface ($h_w = h_s$), as supported by numerous studies (e.g. Sass et al 1992, Mander et al 2011). During inundated conditions ($h_w > h_s$), a further increase in the level of the water table is likely to slow down methanogenesis. At these high water table levels above the soil surface, methanogenesis could decrease due to decreased productivity of aquatic vegetation (e.g. light attenuation with increasing depth, and damaged, stressed, or flooded plants) and thus decreased formation of plant-derived organic substrates for methanogens (Bastviken 2009, Schlesinger and Bernhardt 2013). Additionally, it is possible that the concentration of the organic substrate (e.g. acetate and methylated compounds) may be diluted by the additional water, to an extent that depends on the degree of mixing. This effect may also decrease methanogenesis by reducing the availability of substrates in the soil layers where methanogens
are active. The resulting trend in CH$_4$ production as a function of $h_w$ (equations (B.1) and (B.3) in appendix B), which results from combining the above mechanisms, is illustrated in figure 3(b).

The trend of CH$_4$ oxidation is illustrated in figure 3(c). In the presence of O$_2$, CH$_4$ oxidation is a thermodynamically spontaneous process (Mancinelli 1995, Hakemian and Rosenzweig 2007), part of which takes place in the soil through methanotrophy before being released to the atmosphere. This aerobic process occurs mostly in unsaturated zones of the soil column, but also in the saturated zone and in the water column, see for instance the oxidation along the water column in the ocean (Reeburgh 2007, Valentine 2011, He et al 2012). The level of the water table controls the portion of soil directly exposed to O$_2$ as well as the amount of O$_2$ that can diffuse in, thus exerting a major control on total oxidation. The resulting specific rate of CH$_4$ oxidation, $k_{ox}$ (d$^{-1}$), is larger when the soil is unsaturated ($h_w < 0$) and decreases with the depth of the water table (figure 3(c)). This trend is described by equation (B.4) in appendix B.

Finally, the water table level also controls the diffusion of CH$_4$ to the atmosphere, which is shown in figure 3(d). This trend is due to the fact that the coefficient of mass transport is inversely proportional to the squared length of the diffusion path which increases with the water table. The diffusive flux is hence negatively controlled by $h_w$ (see equation (B.5) in appendix B). In parallel with diffusion, CH$_4$ also reaches the atmosphere following a concentration gradient through gas conduits provided by arenchymous plants (Brix et al 1992, Jackson and Armstrong 1999) or through stems of wetland trees (Pitz et al 2018, Mitra et al 2020), and through ebullition. The plant-mediated transport is practically independent of the water table level (unless of course the plant is completely submerged), as the mass-transfer coefficient in plants mostly depends on the plant developmental stage (Jackson and Armstrong 1999, Pangala et al 2013), while for ebullition (i.e. bubbling up of supersaturated CH$_4$) it is typical to assume to be proportional to the amount of CH$_4$ (Walter and Heimann 2000, Zhuang et al 2004) (see equation (B.6)), even if the role of physical factors affecting bubble formation, trapping, and displacement is not fully understood (Van Der Nat and Middelburg 2000, Christensen et al 2003, Rosenberry et al 2003).

The above considerations reveal an interplay between production and oxidation and transport that generates maximum CH$_4$ emissions at a critical level of the water table, $h_{cr}$, just above the soil surface (figures 2 and 3(e)). The relation between CH$_4$ emissions and water table level, obtained by combining the above trends, is described by equation (B.10) in appendix B. These theoretical considerations both support and explain the experimental evidence of the critical level. The rapid increase of emissions for water levels up to the critical level can be attributed to rapidly improving conditions for methanogenesis and reduced oxidation. By contrast, for water levels above the critical level, methanogenesis is negatively affected because of reduced availability of substrate to methanogens, while the water table begins to behave as a barrier to emissions due to the inverse relationship between mass transport and diffusion path length. Such a resistance to transport favors methanotrophy.
The scatter of observations around the best-fit curve (blue line in figure 2) can also be captured by varying the specific CH\textsubscript{4} production rate, see shaded area in figure 2. In agreement with observations, the width of the shaded area increases as the water table approaches the critical level and then remains stable above it. This behavior readily suggests that, above the critical level, other factors affecting the CH\textsubscript{4} production rate (e.g. availability of substrate, temperature) also exert an important control on emissions. Values outside the shaded band in figure 2 are very unlikely, as these would require extreme values of methanogenesis, which are rarely observed in these ecosystems.

4. Discussion and conclusion

Wetlands provide a variety of ecosystem services and their protection from over-exploitation has been a major concern for decades (Dugan and Dugan 1990). They operate as major carbon (C) sinks (Kayranli et al 2010, Mitsch et al 2013), but this benefit can be offset by their high rates of CH\textsubscript{4} emissions. Given the large uncertainties in these fluxes, it is important that the biogeochemical and hydrological processes underlying such emissions are more fully understood. Here we focused on the water level and showed, by means of a combined data-model approach, that the water level, whether below or above the soil surface, controls CH\textsubscript{4} emissions by modulating simultaneously production, oxidation, and transport. In particular, a water table at about 50 cm appears to be the most favorable to CH\textsubscript{4} emissions.

In considering a wide range of water table levels, the analysis necessarily spans different wetland ecosystems (figure 4). These may differ not only in their typical water levels but also in terms of vegetation, hence microbial communities and substrate for methanogenesis and dominant mechanisms of CH\textsubscript{4} transport to the atmosphere. Despite such differences, each of the major wetland types (bog, fen, marsh, and swamp) and small lakes follow the trend described by the model, pointing to the important role of the water level on wetland biogeochemistry. Because of their typical water level (i.e. \( h_w < h_{cr} \)), bogs and fens tend to lie on the increasing branch of the curve (figure 4(a)). Marshes and swamps exhibit a wide range of water table levels, passing right through the critical level. The combination of water level near the critical level (i.e. \( h_w \approx h_{cr} \)), presence of aquatic plants, and higher temperature in some of the marsh sites (e.g. Florida) all contribute to make marshes among the top CH\textsubscript{4} emitters (see median in figure 4(b)). With their high water levels (i.e. \( h_w > h_{cr} \)), ponds and small lakes are found on the decreasing branch of the curve. These ecosystems are also the ones exhibiting the largest variability in CH\textsubscript{4} emissions figure 4(b), a fact that can be explained by the broad range of observed water levels.

The water table level of course is not the only factor controlling CH\textsubscript{4} emissions, a fact that is also evident from our analysis. At low water tables (below and near the surface), the water level is a primary factor controlling the emissions, as demonstrated by the data points (and the shaded area) tending to align with the blue line, despite differences in wetland characteristics. However, for higher water tables (\( h_w > h_{cr} \)) other factors also become important and may alter the location of the critical water level. For example, figure 5(a) shows how the specific CH\textsubscript{4} production rate (e.g. determined by substrate availability and temperature) affects the dependence of CH\textsubscript{4} emissions on the water table level. The specific CH\textsubscript{4} production rate has a greater influence at high water levels (\( h_w > h_{cr} \)), where a two order of magnitude difference in production translates into almost a three order of magnitude difference in emissions. However, the location of the critical level remains unaffected.
Another factor controlling the dependence of emissions on the water level is the occurrence of fast transport to the atmosphere, through ebullition, the presence of macrophytes with aerenchyma, or tree stems. In this case, reduced transport by these two mechanisms affects CH$_4$ emissions at low and intermediate water levels, by changing the emission rates (i.e. higher emissions for faster transport) as well as the position of the critical water level (figure 5(b)). This suggests that differences in vegetation density and type (e.g. across wetland types), which may affect the relevance of plant-mediated CH$_4$ transport, may have limited effects on the overall CH$_4$ emissions at high water levels (such as in swamps, ponds, and small lakes), justifying the use of a constant emission coefficient to fit the observations across water table levels (see appendix C).

As the areal extent of human-made wetlands is increasing and currently exceeds one fifth of the global wetland area (Davidson et al. 2018), it is imperative that these are designed and managed so as to control the CH$_4$ budget, for example by controlling the water table and maintaining its level away from the critical value, $h_c$. This strategy is applicable for example to restored and managed wetlands, where the water flow and thus the water level are partly controlled. Evidence for the potential reduction of emission of this alternative strategy is clearly visible from figure 2, where according to the best-fit trend maintaining the water level $\approx$30 cm below the critical level reduces CH$_4$ emissions by almost 70%, and a 30% reduction is achieved by increasing the water level to $\approx$1 m. More than half the global extent of human-made wetlands are rice paddies, where to some extent such strategy is also feasible by controlling irrigation. The cost of this water management strategy in wetlands would be limited to the adjustment and maintenance of the irrigation and controlled drainage systems already in place.

Since multiple factors may affect the specific location and variation of the critical level, detailed site-specific studies are necessary for strategies of this type. For instance, the substrate providing C and energy to methanogens may not be homogeneously distributed across the different soil layers. This might be the case for non-agricultural wetlands where soils are not mechanically mixed. Temporal variability in climate (e.g. seasonality) and C source (vegetation dynamics) may also impact the location of the critical level over time. Obviously, controlling the water level not only has implications for CH$_4$ emissions, but also for other ecosystem services (e.g. biodiversity) (Doney et al. 2018), so that multiple objectives always need to be considered simultaneously.

**Data availability statement**

The data that support the findings of this study are openly available at the following URL/DOI: https://doi.org/10.4211/hs.e7050702af674a7a88068a6fd48450f4.

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**Appendix A. The methane budget**

We begin by integrating over a soil column the reactive transport equation for methane (Zhuang et al. 2004, Riley et al. 2011, Stumm and Morgan 2012). This provides the mass balance equation per unit ground area of the CH$_4$ content over the soil column,

\[
\frac{dM(t)}{dt} = P - OX - E, \quad \text{(A.1)}
\]
where $M(t)$ is the total CH$_4$ content in the soil column at time $t$, $P$ and $OX$ are the total production and oxidation rates, and $E$ is the emission rates of CH$_4$ out of the soil surface. The terms $P$ and $OX$ derive from the integral over the soil column of methanogenesis and oxidation rates, respectively. For instance, if $P$ is the CH$_4$ production rate at depth $z$, the total CH$_4$ production is computed as $P(h_w) = \int p(z, t; h_w)dz$. The term $E$ includes the flux of CH$_4$ out of the control volume (i.e. the soil column), by diffusion (the integral of the diffusive term in the transport equation), ebullition, and plant transport through the aerenchyma.

The interactions between all biological (e.g. activities of microbial communities and plants), geochemical (e.g. redox potential, labile C content), and hydro-climatic factors (e.g. water level, temperature) affecting CH$_4$ production, oxidation, and emission generate highly variable CH$_4$ dynamics, with fluctuations in CH$_4$ emissions at various frequencies. The presence of such multi-frequency fluctuations makes it hard to interpret experimental observations and understand the key features of CH$_4$ dynamics. To capture the dominant behavior of CH$_4$ budget in relation to the controlling factors affecting it, it becomes essential to average out the low frequency fluctuations (e.g. hourly or daily variability) and focus on the slow-varying dynamics driven by seasonal to inter-annual changes in environmental conditions, such as water level, temperature, or labile C availability. Over this longer timescale, the fluxes of CH$_4$ are likely to balance out, so that (A.1) reduces to:

$$P = OX + E.$$  \hfill (A.2)

### Appendix B. Methane fluxes

We consider methanogenesis to occur only in the saturated layers, according to a first order kinetics with respect to the concentration of the C substrate, $p = k_p[C]$. Assuming that $[C]$ is homogeneous across the soil column, the total production can be expressed as the product of the local rate $p$ and the height of the saturated soil column, $h_{sat}$,

$$P = p \cdot h_{sat} = k_p[C]h_{sat}. \quad \text{(B.1)}$$

When only a part of the soil column is saturated, $h_{w} < h_{sat}$, the term $h_{sat} = h_{w}$, while in flooded conditions, $h_{w} \geq h_{sat} = h_{w}$. Thus assuming a homogeneous distribution of the substrate $C$ within the soil column, as the water table rises the production rate increases linearly, $P = k_p Ch_w$ until $h_{w} = h_{w}$, for which $P = P_{max} = k_p Ch_w$. For water table higher than the soil column, $h_{w} > h_{sat}$, part of the C substrate may be diluted by this additional water depending on the level of mixing of the substrate. If the substrate is well-mixed within the column of water, then its concentration is $[C] = C \times h_{w}/h_{w}$ and the production:

$$P = k_p Ch_{w}^{2}/h_{w}, \quad \text{(B.2)}$$

such that the rate of methanogenesis decays hyperbolically with $h_{w}$ in flooded soils. In the total absence of mixing, flooded conditions do not impact the production of CH$_4$, which remains equal to $P = k_p Ch_w$. The well-mixed and no mixing cases represent two extreme scenarios bounding the actual effect of flooded conditions. In a realistic setting, it is likely that there would be an intermediate degree of mixing and production rate (see figure 3(b)). For water table above the soil surface, the primary productivity of plants also decreases (Bastviken et al. 2009, Schlesinger and Bernhardt 2013) and oxygen entrained at the water level surface may impact the activity of methanogens, possibly inducing a further decrease in methane production. To account for these combined effects, we include an exponent $\alpha$ in equation (B.2),

$$P = k_p Ch_{w}^{(h_{w}/h_{w})^\alpha}, \quad \text{(B.3)}$$

so that, for $\alpha = 0$ the water table does not effect $P$ (i.e. no dilution and no effect on productivity), while as $\alpha$ increases these factors increasingly reduce methane production.

Methane oxidation is modeled through a first order reaction with respect to methane concentration (Das and Adhya 2012), i.e. $OX = k_{ox} \cdot O_{2} \cdot M$, in which $O_{2}$ represent the overall availability of oxygen throughout the water column. In completely unsaturated conditions, $h_{w} = 0$, the O$_2$ concentration in the soil air is approximately equal to the atmospheric one, and as a result the availability of O$_2$ is maximum, $O_{2} \approx O_{2}^{max}$. As the water table rises, the availability of O$_2$ decreases and it tends to zero in flooded conditions. Modeling the availability of oxygen as an exponential decay with the water table, the oxidation rate is:

$$OX = k_{ox} O_{2}^{max} e^{-\beta h_{w}} M = k_{ox}^{max} e^{-\beta h_{w}} M, \quad \text{(B.4)}$$

where $\beta$ is an empirical parameter and $k_{ox}^{max} = k_{ox} O_{2}^{max}$.

The emission $E$ occurs through three different pathways (Whalen 2005): diffusion, ebullition and plant transport, $E = DIFF + EB + PT$. The diffusive flux, $DIFF$, is controlled by the diffusion in the saturated zone, $h_{w}$, as the diffusion coefficient is much lower than in air, $D_{sat} \ll D_{air}$. Using finite differences, $DIFF = D_{sat} \cdot ([M]_{air}^{max} - [M]_{air}^{sat})/L$, being $L = h_{w}/2$, and given that $M_{air}^{max} \gg M_{air}^{sat}$, one obtains:

$$DIFF = D \cdot [M]_{air}/h_{w} = k_d M, \quad \text{(B.5)}$$

where $D = D_{sat}/2$, and $k_d = D/h_{w}^{2}$ is the mass transfer coefficient. Ebullition and plant transport are considered proportional to the methane content,

$$EB = k_d M, \quad \text{PT} = k_p M, \quad \text{(B.6)}$$
where \( k_{eb} \) and \( k_{pt} \) are empirical constants. The overall CH\(_4\) emission can then be expressed as:

\[
E = \text{DIFF} + \text{EB} + \text{PT} = (k_d + k_{eb} + k_{pt}) \cdot M = k_e \cdot M, \tag{B.7}
\]

in which \( k_e = (k_d + k_{eb} + k_{pt}) \) depends on \( h_w \) through the coefficient \( k_e \). Substituting production, oxidation, and emission rates into (A.2) yields:

\[
P = k_{ox} M + k_e M, \tag{B.8}
\]

so that the CH\(_4\) content per unit ground area in the soil column can be computed as:

\[
M = \frac{P}{k_{ox} + k_e} = \begin{cases} k_F \cdot h_w \left( \frac{h_w}{h_s} \right) & h_w \leq h_s, \\ k_F \cdot h_w \left( \frac{h_w}{h_s} \right)^\alpha & h_w > h_s, \end{cases} \tag{B.9}
\]

where we introduced \( k_F = k_p C \), assuming that \( C \) is constant throughout the soil column. From equation (B.9), CH\(_4\) emissions, \( E = k_e M \), are computed as:

\[
E = \begin{cases} \frac{k_F \cdot h_w}{k_{ox} + k_e} & h_w \leq h_s, \\ \frac{k_F \cdot h_w}{(k_{ox} + k_e)} \left( \frac{h_w}{h_s} \right)^\alpha & h_w > h_s. \end{cases} \tag{B.10}
\]

The resulting CH\(_4\) production rate, oxidation and emission coefficients, and CH\(_4\) emission rate are shown in figures 3(b)–(e) respectively.

**Appendix C. Data analysis and model fitting**

To explore the relationship between CH\(_4\) emissions and water table level, we compiled an extended data set of chamber measurements of CH\(_4\) emissions and corresponding water table level across different wetland types (bog, fen, marsh, swamp) from the literature (see supplementary table), for a total of 42 sites. Data were generally provided as averages over hydro-periods (e.g. a season or ice-free periods for high latitude wetlands). To complement the dataset, we included a recent compilation of eddy-covariance measurements from the FLUXNET community (Knox et al 2019), adding another 38 sites across wetland types. These flux tower measurements, and water table level, were averaged the entire measurement period. The spatial distribution of the sites included in the analysis is shown in figure 1. One limitation is that the sites mostly span temperate and boreal regions of the northern hemisphere, highlighting the need for more measurements of CH\(_4\) emissions and water table level over the tropics and southern hemisphere. All CH\(_4\) emissions measurements are plotted against the water table level in figure 2.

To find the ‘average’ relation between CH\(_4\) emissions and water table level, we fitted the model to the observations, including both chamber and flux tower measurements. The maximum oxidation rate constant and diffusion coefficient are known, \( k_{ox}^{\text{max}} = 200 \text{ d}^{-1} \) (Bender and Conrad 1992) and \( D = 1.3 \times 10^{-4} \text{ m}^2 \text{ d}^{-1} \) (Cussler and Cussler 2009). The oxidation constant decays with increasing water table level, due to decreasing dissolved O\(_2\) concentration, according to an assumed decay coefficient \( \beta = 0.1 \). Since we are fitting the model to different sites with various degree of mixing, we simply assumed \( \alpha = 1 \). This assumption is reasonable considering the fact that we focus on long timescales (i.e. seasonal to yearly), while mixing in water occurs over shorter timescales (i.e. sub-daily). The only parameters that needed calibration then were \( k_F \) and \( k_e \). Because there is a substantially higher concentration of sites at water levels \( \approx 0 \) or below, we binned the CH\(_4\) emissions according to water table level intervals of 0.1 cm and considered median emissions within each interval. The parameters were then calibrated using a standard least squares method focusing on water level range \( \approx 3 \text{ m} \). The number of sites above 3 m is too low), which yielded \( k_F \approx 300 \text{ mg m}^{-3} \text{ d}^{-1} \) and \( k_e \approx 0.1 \text{ d}^{-1} \).

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