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Wetland hydrological dynamics and methane emissions



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Wetlands are the largest and most uncertain biological source of atmospheric methane, with hydrological fluctuations exacerbating this uncertainty. Here we critically explore the complex relationship between hydrological fluctuations and methane emissions in wetlands by integrating observations from 31 FLUXNET wetland sites with a comprehensive literature review. We present the prevalence and patterns of water table fluctuations and their contribution to uncertainty in methane fluxes. We also highlight key pathways through which these fluctuations affect methane production and emission, such as soil redox heterogeneity, changes in substrate availability and alternative electron acceptor pool, the contribution of different methane transport pathways, and the non-linear responses of community structure and activity of methanogens and methanotrophs to hydrological fluctuations. This review aims to improve the accuracy of wetland methane emission reports by carefully assessing biogeochemical kinetics under hydrological fluctuations.

Methane (CH₄) in the atmosphere contributes ~16–25% of global warming to date^{1,2}. Importantly, over the past decade, there has been a continuous increase in atmospheric CH₄ concentration, coupled with notable variations in growth rates^{3–6}. Isotope fractionation-based analysis suggests that biogenic CH₄ emissions may strongly contribute to this dynamic^{7–10}. Wetlands, as the largest natural biogenic source of CH₄¹¹, are considered as the main contributor to the recent atmospheric CH₄ growth rate anomaly^{12,13}. However, their emissions exhibit high uncertainty and vary across spatial and temporal scales^{14,15}.

Variations in wetland CH₄ emissions are also implicated as the primary driver of observed interannual variability in global CH₄ fluxes^{16,17}. Critical gaps arise from the fact that existing observational networks and their methodologies, whether bottom-up approaches based on process modeling or top-down approaches based on atmospheric inversions, often exhibit substantial differences. This variance makes it challenging to pinpoint specific changes in CH₄ sources at large scales, particularly when dealing with conditions of hydrological fluctuations^{4,18–20}. Indeed, diurnal, seasonal, and annual fluctuations in meteorological factors such as air temperature, precipitation, and evapotranspiration^{21,22}, along with extreme events under climate change conditions like flooding and droughts²³, result in dynamic hydrological processes at the regional or catchment scale. These variations introduce considerable uncertainties in large-scale CH₄ emission assessments, especially for wetlands in different climatic zones, thus complicating the understanding and prediction of atmospheric CH₄ dynamics⁵.

Nowadays, advancements, such as the use of ecosystem-scale eddy covariance measurement systems, enable continuous monitoring and capture of changes in CH₄ fluxes across various time scales, particularly under disturbances caused by hydrological, meteorological, and management activities^{24–27}. This may help us understand the response of CH₄ fluxes to hydrological fluctuations.

Despite longstanding knowledge about the limitations of microbial thermodynamics and energetics, traditional wetland CH₄ flux predictions rigidly assign CH₄ production to the anoxic zone below the water table and CH₄ oxidation to the oxic zone above it^{28,29}. Recent discoveries in microbial CH₄ processes over the past decade are challenging these boundaries. For instance, it is now understood that methanogenesis can take place in the microaerobic zone, often referred to as the methane paradox, although its presence in freshwater wetlands remains somewhat speculative^{30–32}. Additionally, CH₄ oxidation can largely occur in reductive zones using alternative electron acceptors occurring in wetlands, such as nitrate³³, nitrite³⁴, sulfate³⁵, iron³⁶, and more, from both autochthonous and allochthonous sources (e.g., anthropogenic eutrophication^{37,38}, saltwater intrusion³⁹, and atmospheric deposition⁴⁰). Of greater significance, variations in wetland hydrological conditions at different time scales (spanning from hourly to annual)⁴¹, as well as changes in intensity of period, amplitude, and the ratio of oxic to anoxic periods⁴², can lead to even more intricate and nonlinear responses in CH₄ fluxes. Moreover, climate warming influences wetland CH₄ cycle not only through precipitation-driven hydrological fluctuations, but also

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through directly influencing the kinetics of CH₄ biogeochemical reactions due to rising temperatures⁴³ and indirectly affecting plant community composition and plant traits such as biomass^{44,45}. This complexity adds an additional layer of uncertainty, which in turn diminishes our ability to accurately assess the impact of wetlands on climate change. Thus, a complete understanding of the connection between hydrological fluctuations and the CH₄ cycle in wetlands, influenced by both biotic and abiotic environmental factors, is lacking. This knowledge gap poses a substantial barrier to accurately reporting CH₄ emissions and implementing effective mitigation measures, especially in the face of climate change, marked by increasingly frequent extreme events.

In this review, we aim to bridge this gap by: (i) presenting a comprehensive exploration of the intricate relationship between CH₄ flux and hydrological dynamics, and (ii) providing a detailed analysis of the impacts of hydrological fluctuations on the CH₄ cycle, along with the underlying mechanisms. By shedding light on these aspects, we hope to contribute to a more comprehensive understanding of the complex interplay between wetland hydrology and CH₄ emissions, facilitating improved management and mitigation in the context of our changing climate.

Hydrological fluctuations in wetland ecosystems

Prevalence of hydrological fluctuations

Wetlands, serving as the typical transitional zones between the permanent wet environment of aquatic ecosystems and the dry environment of terrestrial ecosystems, are susceptible to longitudinal, lateral, and vertical hydrological fluctuations^{46–49}. These dynamics are shaped by the interplay of hydrometeorological factors (e.g., precipitation and evapotranspiration) and key topographic and hydrogeologic features (e.g., elevation, microtopography, and connections to groundwater and adjacent water systems)⁵⁰.

Climate change adds a layer of complexity to wetland hydrology. Worldwide, rising temperatures are amplifying both the overall precipitation and its variability, fundamentally altering the hydrological dynamics within wetlands⁵¹. Notably, in tropical regions, meteorological changes at a large scale are accentuating the hydrological cycle of tropical wetlands^{52–56}. This phenomenon is triggering hydroclimatic events, such as seasonal rainfall anomalies, fostering the expansion in certain tropical wetlands⁵⁷ while others experience loss⁵⁸. Overall, in a warmer future, the increased evaporative demand may outweigh changes in precipitation, potentially leading to a substantial reduction (~34,000 km²) in tropical wetland area by the end of the century⁵⁹. North American inland wetlands are also expected to experience a 5–10% reduction in average area by the end of the century compared to historical contemporaries⁶⁰. Moreover, the accelerated glacial melting (including ice caps and mountain glaciers) due to global warming greatly alters water inputs to wetlands⁶¹, while thawing permafrost can increase the hydrogeological connectivity of wetlands by opening up previously blocked vertical and lateral flow pathways, in addition to recharging groundwater and reducing surface runoff^{62–64}. It is also important to note that glacier melting and thermal expansion of oceans are driving sea level rise⁶⁵, which is projected to increase 0.26–0.82 m by 2100^{66,67}. Although wetlands can naturally keep pace with changes in sea level through accretion processes, climate change may upset this balance, accelerating water table rise in coastal and estuarine wetlands, and eventually resulting in wetlands being submerged, pushed inland or covered with salt.

Beyond these natural factors, anthropogenic activities such as land-use change⁶⁸, groundwater extraction⁶⁹, dam construction⁷⁰, and petroleum exploration^{71–74} can also exert strong impacts on the water balance in wetlands. In particular, land-use change has led to the long-term degradation of 21% of wetlands globally from 1700 to 2020 due to drainage primarily for intensive agriculture development⁷⁵. However, large-scale wetland restoration and conservation initiatives are working to re-raise water table⁷⁶. As a result, hydrological fluctuations have become a prevalent occurrence in wetlands under the influence of natural processes, climate change, and anthropogenic activities.

Patterns of hydrological fluctuations

Hydrological fluctuation patterns in wetlands vary widely across wetlands due to the spatiotemporal variations of different hydrometeorological and hydrogeologic factors⁷⁷. Two key variables, amplitude, and periodicity/frequency are commonly used to characterize hydrological fluctuations in wetlands²³. Seasonal fluctuations in wetlands can vary widely, ranging from several tens of centimeters^{78,79} to over a meter^{80–82}, with the extent largely depending on climate conditions and wetland types. Tropical wetlands, for instance, often experience more dramatic fluctuations compared to boreal wetlands⁸³, primarily due to pronounced seasonal rainfall patterns within tropical regions. Additionally, rainfall patterns in wetlands are subject to the impact of interannual and interdecadal climate fluctuations, further amplifying the variability in fluctuation amplitudes⁸⁴. By synthesizing water table data from 31 wetland sites in the FLUXNET database (including six bog sites, seven fen sites, three swamp sites, seven marsh sites, three salt marsh sites, and five rice sites)^{26,27}, Fig. 1 illustrates differences in the water table fluctuation amplitudes and density distributions of water table data across various wetland types^{85,86}, typically attributed to variations in water supply pathways⁸⁷, proximity to water sources⁸⁸, vegetation communities⁸⁹, and soil properties⁹⁰.

Moreover, water regime can determine wetland vegetation structure and composition. In turn, vegetation can influence wetland water balance through rainfall interception, stemflow, and transpiration⁹¹. Besides, differences in vegetation ecological characteristics and adaptation strategies can further impact hydrological fluctuations. For instance, the encroachment of woody shrubs (e.g., Carolina willow) into graminoid-dominated subtropical wetlands can increase transpiration water loss, which may drive a sustained water table decline⁹². In tropical papyrus wetlands, the papyrus mat system can enhance water storage capacity during the rainy season, allowing it to withstand high water volume change and water table fluctuations⁹³. Additionally, *Sphagnum* can form a dense cover on some boreal wetlands, with its unique cellular structure enabling it to absorb and retain water, acting like a sponge⁹⁴. This can impede the infiltration of low-intensity rainfall and thus influence the fluctuation amplitude⁹⁵.

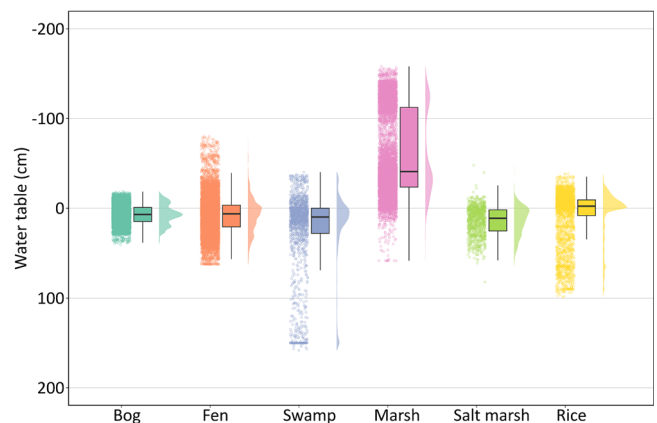


Fig. 1 | Water table fluctuations in six wetland types. These wetland sites include six bog sites with 6717 water table observations (green), seven fen sites with 10,679 water table observations (orange), three swamp sites with 1543 water table observations (blue), seven marsh sites with 9108 water table observations (pink), three salt marsh sites with 947 water table observations (light green), and five rice field sites with 3495 water table observations (yellow). Data are daily mean values and were collected from FLUXNET-CH₄ database^{26,27}. Site information is in Supplementary Table 1. The raincloud plot for each wetland type includes a jitter plot showing individual data points (colored dots), a half-violin plot presenting density distributions of data (shaded areas), and a box plot illustrating summary statistics (boxes with whiskers). The box plots show the median (line within the box), interquartile range (box edges), and whiskers (1.5 times the interquartile range). Note: positive values of the water table represent distance below the ground surface and negative values represent surface flooding.

Water table in wetlands generally exhibits periodic fluctuations of different intensities over various time scales, attributable to diurnal, seasonal, and annual water inputs and outputs. Diurnal water table fluctuations, usually a few centimeters, are generally described as a function of evapotranspiration and periodic changes in atmospheric pressure^{96,97}. During the summer, wetlands often experience active evapotranspiration throughout the day, leading to a typical pattern of gradual decline in water table starting at dawn (Supplementary Fig. 1)⁹⁸. Seasonal water table fluctuations are primarily governed by the temporal dynamics of precipitation and temperature. Water table rises substantially during wet seasons such as the rainy season or snowmelt, and declines accordingly during dry seasons when precipitation decreases and evapotranspiration increases (Fig. 2a, b). This implies that the seasonal fluctuation pattern of water table is closely related to the local climatic conditions at the time.

Moreover, climate change can reshape rainfall patterns, such as increasing rainfall in northern Europe and reducing rainfall in southern Europe¹. Increased rainfall generally leads to increased flooding frequency in inland wetlands⁹⁹, especially in low-elevation wetlands rather than high-elevation wetlands¹⁰⁰. Over long timescales, this interannual variability and gradual alterations in climate conditions contribute to long-term changes in water table. On the other hand, changes in rainfall patterns may increase the risk of extreme weather events, causing dramatic fluctuations in water table in the short term^{101,102}.

Uncertainty of methane fluxes under hydrological fluctuations

In wetlands, water table has long been considered a primary driver of CH₄ flux variations^{103–105}. This recognition stems from the straightforward association observed between CH₄ flux and water table in specific scenarios. Particularly, it has been repeatedly observed that CH₄ fluxes from wetland soils increase as the water table approaches the soil surface^{106–111}. For instance, a significant positive linear correlation between water table and CH₄ flux ($p < 0.01$) emerged in a blanket bog with water table ranging from 50 cm below the soil surface to 0 cm¹⁰⁶. Additionally, CH₄ emissions in a coastal wetland exhibited an exponential increase with rising water table in the 40 cm below the soil surface to 0 cm range ($p < 0.001$)¹⁰⁷. This correlation under sub-surface water table conditions has also been verified in cross-wetland analyses, including different wetland types^{108,112–114}, diverse climate zones¹¹⁵, and different soil management practices^{113,114,116}.

However, nonmonotonic relationships between water table and CH₄ flux have also been widely reported in wetlands. For example, CH₄ fluxes from permanently inundated wetlands appear to lack a consistent or strong direct relationship with water level^{108,117}. In other words, CH₄ fluxes do not consistently increase with water table rise, but rather there exists a critical water table where CH₄ emissions increase as the water table rises up to this threshold, followed by high variability^{19,118,119}. Based on data analysis of 42 wetland sites (including bogs, fens, marshes, swamps, and ponds/lake), the critical water table could be 50 cm above the soil surface¹¹⁸. Notably, such an extensive analysis may not fully capture the specific influence of wetland type on the critical water table. The critical water table, as it turns out, is highly dependent on the specific wetland ecosystem, as evident in separate studies exploring the relationship between water table and CH₄ fluxes in different wetland types, involving 51 and 83 wetland sites^{113,120}, respectively. The critical water table tended to be below the soil surface in bogs, close to the soil surface in fens and marshes, while in swamps, it was usually above the soil surface.

On the other hand, hydrological fluctuations in wetlands can amplify CH₄ flux uncertainty through lag effects on different time scales. During the transition of hydrological state in wetlands, the temporal lag in CH₄ emissions could be a few days^{24,119,121,122} or a few months^{123–125}, which is closely related to factors such as the intensity and duration of fluctuation events and soil properties like availability of alternative electron acceptors, soil texture, and initial moisture¹²². For instance, seasonal CH₄ flux changes could lag water table fluctuations by about 17 ± 11 days across 23 wetland sites²⁴, while the temporal lag between

CH₄ fluxes and precipitation in tropical wetlands was up to eight months¹²⁵. This short-term lag effect can even lead to a 5-fold lower CH₄ emission during the transition from drought to inundation compared to the steady wet conditions¹²⁶. In addition, extreme drought events can exert long-term effects on CH₄ emissions through potential impacts on vegetation composition¹²⁷ and microbial community structure and activity¹²⁸, leading to temporal lags of up to several years¹²⁹.

Moreover, episodic CH₄ pulses, such as ebullition events, associated with hydrological fluctuations like dramatic drops in water table¹³⁰, transient flooding¹³¹, spring thaw in most northern peatlands¹³², and rainfall anomalies in tropical regions¹³³, can lead to short-term outbursts of CH₄ fluxes, thereby complicating the relationship with water table. For example, the release of trapped CH₄ in the ice layer of Finnish mires during the spring thaw contributed 11% of the annual CH₄ emissions¹³⁴. Further, the seasonal CH₄ pulses in East Africa due to rainfall anomalies accounted for over 25% of the global CH₄ emission growth in 2019¹³³. This implies that, when exclusively relying on the average water table over an extended period (e.g., annual mean water table), the assessment of CH₄ emissions may ignore the considerable impact of high emissions from episodic events¹³⁵.

Thus, subjected to typical periodical and atypical dramatic hydrological fluctuations, CH₄ fluxes readily exhibit nonmonotonicity, lagged response, and pulsation. Additionally, under climate change, rising temperatures may override the influence of water table on CH₄ emissions, especially at high latitudes¹³⁶, by affecting the reaction rates of CH₄-related biogeochemical processes. Even in some cases where two wetland sites have completely opposite seasonal water table trends, their CH₄ flux patterns do not necessarily mirror the fluctuations in water table⁸⁵ (Fig. 2a, b), but rather align with soil temperature variations⁸⁵ (Fig. 2c, d). In tropical wetlands, increased temperatures and reduced precipitation together explain 49% of interannual CH₄ flux variations¹²⁵. This further complicates the relationship between CH₄ flux and water table, highlighting its strong dependence on the specific wetland types and local environment, as demonstrated across the 6 wetland types in this study⁸⁵ (Fig. 3). Nevertheless, water table remains a key explanatory variable for variations in CH₄ fluxes within current multivariate models^{137,138}. This is partly due to the simplicity and cost-effectiveness of water table measurements, as well as the convenience of applying simple binary relationships at large scales⁹⁷. More importantly, in some practical engineering scenarios, such as peatland rewetting/restoration, water table remains the primary consideration for realizing the potential climate benefits that rewetting could offer¹¹⁴. Based on this logic, it is imperative to comprehend and modify the relationship between water table and CH₄ cycle before identifying a more suitable explanatory variable. In the following sections, we aim to gain insights into how hydrological fluctuations specifically decouple the subsurface CH₄-related biogeochemical processes from surface CH₄ fluxes.

Response of soil redox conditions to hydrological fluctuations

The occurrence of CH₄ production and oxidation in wetlands is determined by the soil oxygenation status that can be indicated by the redox potential (Eh). Basically, Eh levels in wetland soils is largely determined by oxygen availability, which is controlled by diffusion from the atmosphere into the soil profile and by consumption through soil respiration¹³⁹. In seasonally flooded or chronically submerged wetland soils, soil respiration and organic matter degradation rapidly deplete available oxygen, thereby forming the anoxic environment. Subsequent anaerobic respiration processes continue the organic matter decomposition using alternative electron acceptors (e.g., nitrate, iron, manganese, and sulphate), resulting in a gradual decline in soil Eh until a highly reduced environment (typically below -200 mV) for methanogenesis¹⁴⁰. As water table declines, atmospheric oxygen penetrates into the reducing zone through diffusion pathways in water-unsaturated soils, thereby elevating soil Eh at depth and suppressing CH₄ production^{127,141}. In addition, oxygen availability plays a crucial role in determining the pathway of CH₄ oxidation, influencing whether it occurs

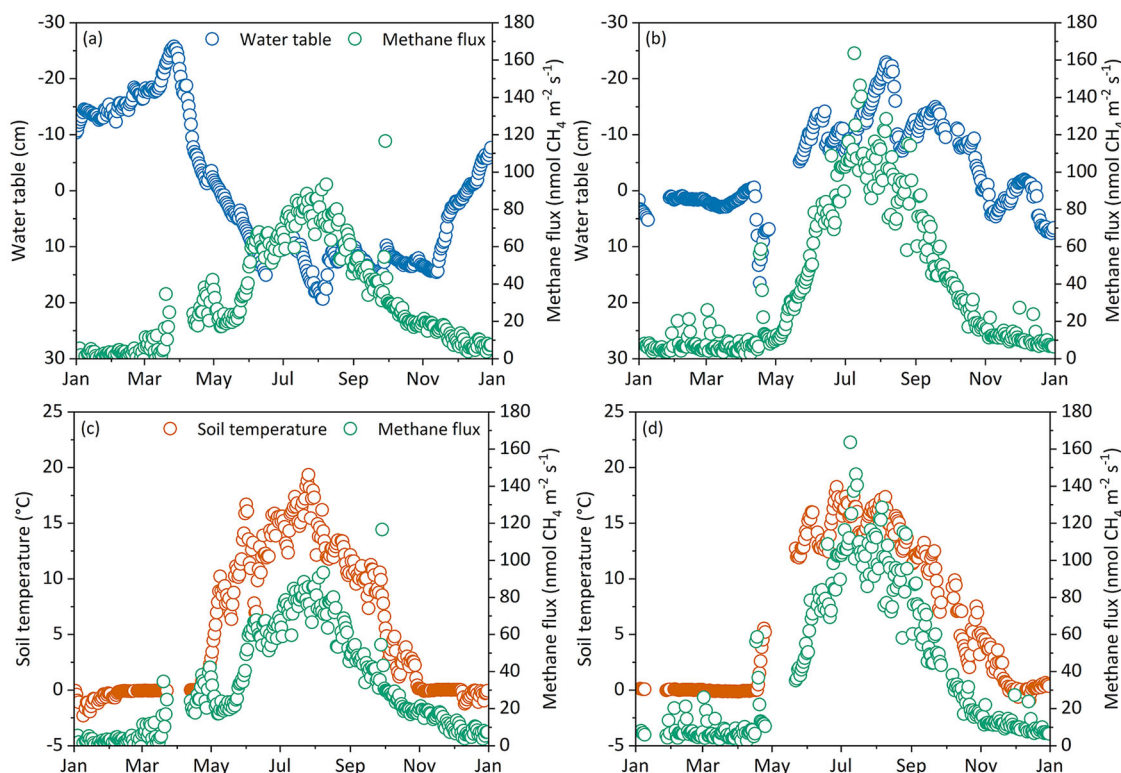


Fig. 2 | Seasonal water table and soil temperature fluctuations with corresponding CH₄ emission patterns. **a** Seasonal water table fluctuations (blue circles) and corresponding CH₄ emission patterns (bluish green circles) at the SE-Deg site (Västerbotten, Sweden) in 2016. **b** Seasonal water table fluctuations (blue circles) and corresponding CH₄ emission patterns (bluish green circles) at the FI-Sii site (Pirkanmaa, Finland) in 2013. **c** Seasonal soil temperature fluctuations (vermillion circles) and corresponding CH₄ emission patterns (bluish green circles) at the SE-

Deg site (Västerbotten, Sweden) in 2016. **d** Seasonal soil temperature fluctuations (vermillion circles) and corresponding CH₄ emission patterns (bluish green circles) at the FI-Sii site (Pirkanmaa, Finland) in 2013. Site information is in Supplementary Table 1. Note: positive values of the water table represent distance below the ground surface and negative values represent surface flooding. Soil temperature probe depth at SE-Deg and FI-Sii were 0.05 cm and 0.02 cm below the surface, respectively.

aerobically in the presence of oxygen or anaerobically with alternative electron acceptors¹⁴².

However, oxygen diffusion in wetland soils is a function of both soil properties (e.g., soil water content, soil structure, and texture) and environmental conditions (e.g., hydrological status and vegetation physiological characteristics). For example, oxygen diffuses 10,000 times faster in air-filled pores than in water-filled pores¹⁴³, indicating that the original moisture content of wetland soils largely determines the oxygen diffusion rate. In addition, compared to macropores with low tortuosity and high pore connectivity in soil structural units, microporous with high tortuosity and low pore connectivity are more likely to retain water and elongate the diffusion pathways for oxygen, thereby hampering oxygen supply^{144,145}. In these conditions, anoxic microsites can be established even within well-drained soils, thus forming steep Eh gradients between macropores and micropores⁴¹. Furthermore, the abundance of anoxic microsites is positively correlated with clay content. This is because soils with higher clay content exhibit elevated levels of microaggregates, whose micropore networks can strongly reduce the oxygen diffusion rate¹⁴⁴. Thus, anoxic microsites may be prevalent in fine-textured wetland soils with high organic carbon content during periodic declines in water table, allowing for the maintenance of methanogenic activity¹⁴⁶.

Conversely, the influx of oxygen-rich water, such as precipitation and lateral subsurface flow^{147,148}, can establish oxic microsites within otherwise anoxic zones, even in extremely wet or submerged soil conditions. In addition, the formation of oxic microsites is easier in coarser-textured wetlands, attributed to more macropores and faster oxygen transport rates compared to finer-textured wetland soils^{146,149}. In the hyporheic zone of wetlands, the mixing of oxygen-rich surface water and shallow groundwater can also facilitate the deep penetration of dissolved oxygen into the reduced

zone^{150,151}. The input of oxygen-rich water implies that transient water table rises might not necessarily result in increased CH₄ production, but in some cases can even reduce CH₄ emission¹⁴⁷. Besides, the aerenchyma in wetland aquatic plants can transport oxygen from the air or from plant photosynthesis to the root tip and the rhizosphere, thus developing oxic microsites in the rhizosphere¹⁵².

Thus, redox conditions in wetland soils under hydrological fluctuations are highly dynamic, characterized by pronounced shifts, periodic oscillations between oxic and anoxic states, or high heterogeneity due to the prevalence of aggregate-scale oxic and anoxic microsites. This implies that CH₄ production and oxidation are not strictly depth-stratified (Supplementary Table 2). In addition, the presence of anoxic microsites is even considered to be a potential explanation for CH₄ production in wetland oxic zones/water columns (i.e., methane paradox)^{30,31}. Similarly, anaerobic CH₄ oxidation is not restricted to deep soils, but can be prevalent in surface soils due to the presence of anoxic microsites¹⁵³.

Effects of hydrological fluctuations on methane production

Substrate availability

Early investigations have repeatedly shown that the availability of substrate, primarily derived from plant residues and root exudates, is one of the critical environmental variables controlling CH₄ fluxes in wetlands^{118,154–156}. Notably, common substrates for methanogens are typically found in the dissolved organic matter (DOM) fraction, rather than in the organic matter in the bulk soil¹⁴². Besides, characteristics of DOM are closely linked to the composition of wetland vegetation communities^{154,157}. For example, DOM derived from different plant species contains different levels of organic acids, which are the more

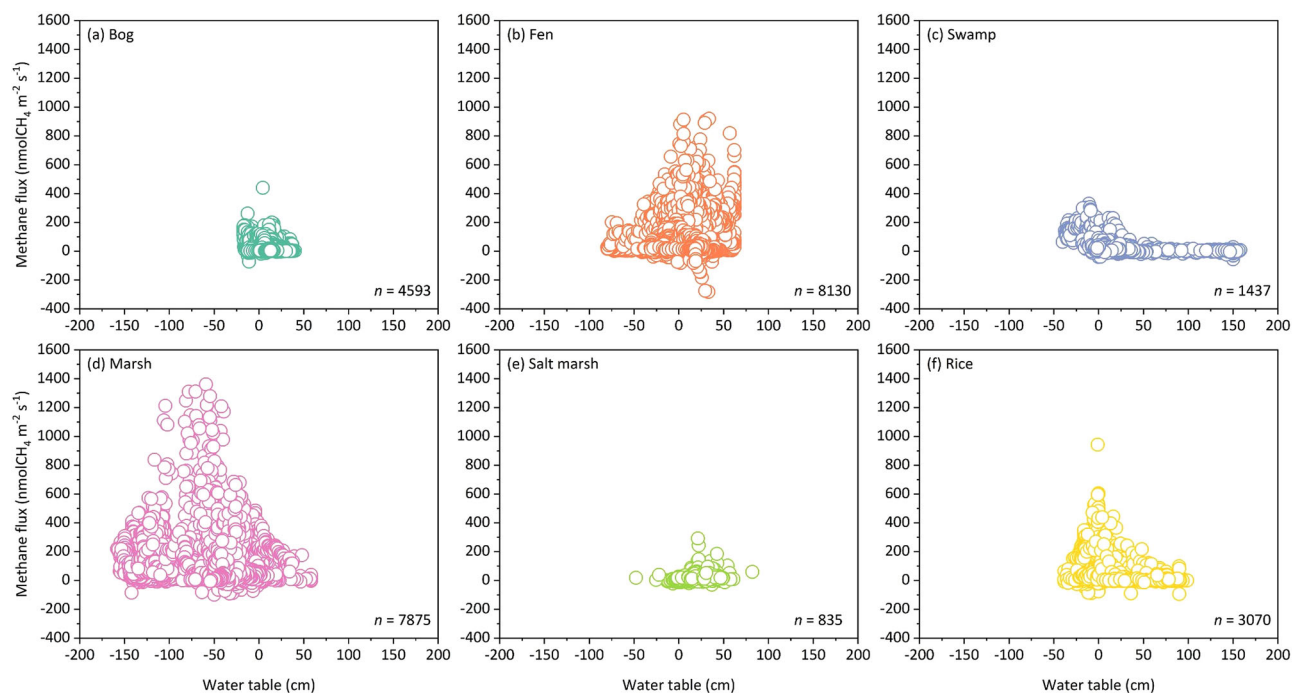


Fig. 3 | CH₄ fluxes at different water tables in six wetland types. **a** Bog: six sites with 4593 paired observations of CH₄ flux (green circles) and water table. **b** Fen: seven sites with 8130 paired observations of CH₄ flux (orange circles) and water table. **c** Swamp: three sites with 1437 paired observations of CH₄ flux (blue circles) and water table. **d** Marsh: seven sites with 7875 paired observations of CH₄ flux (pink

circles) and water table. **e** Salt marsh: three sites with 835 paired observations of CH₄ flux (light green circles) and water table. **f** Rice: five sites with 3070 paired observations of CH₄ flux (yellow circles) and water table. Site information is in Supplementary Table 1. Note: positive values of the water table represent distance below the ground surface and negative values represent surface flooding.

bioavailable fraction of DOM, thus leading to differences in DOM reactivity. Both long-term (often years, decades, or even centuries)^{158,159} and short-term (often days to months)¹⁶⁰ hydrological fluctuations can drive vegetation community succession¹⁶¹. For example, prolonged water table drawdown gradually leads to the transformation of vascular plant communities into woody plants^{162,163} and the long-term loss of aquatic *Sphagnum* species, such as *S. cuspidatum*¹⁶³. Consequently, water table fluctuations inevitably lead to differences in DOM composition and decomposition by indirectly controlling vegetation communities¹⁶⁴.

Hydrological fluctuations can also provide additional substrates for CH₄ production. Examples include the input of exogenous organic carbon (e.g., plant debris and terrestrial DOM)¹⁶⁵ and the release of previously sequestered labile organic carbon within permafrost¹⁶⁶ during thawing contributed to increased methanogenic activity in wetlands. This activity can be further exacerbated by rising soil temperatures¹⁶⁷. Lateral water exchange in floodplains also led to higher CH₄ fluxes compared to adjacent tundra wetlands, attributed to the additional supply of particulate organic matter¹⁶⁸. Similar phenomena can manifest during rainstorms and flooding events¹⁶⁹.

On the other hand, the penetration of oxygen under hydrological fluctuations can alter substrate availability through biotic and abiotic pathways and therefore affect methanogenic activity. For instance, redox fluctuations could trigger functional changes in the peat microbiome that contribute to the biotic degradation of organic carbon rich in aromatic and polyphenolic compounds¹⁷⁰. Additionally, the co-existence of Fe(II)-containing minerals and oxygen can trigger Fenton-like reactions that generate hydroxyl radicals ([•]OH) to intensify abiotic organic carbon mineralization in wetland soils^{170–172}. Similarly, tidal hydrology-driven redox cycling in intertidal soils can activate thermodynamically stable iron minerals (e.g., pyrite and goethite) to generate redox-active metastable iron phases with efficient electron transfer capacity, which can effectively activate oxygen to produce [•]OH¹⁷³. Moreover, during hydrological fluctuation periods, reactive iron minerals can precipitate or

dissolve in response to changes in redox conditions, leading to the capture or mobilization of organic carbon^{62,174,175}.

Alternative electron acceptors

Even lacking environmental studies, prolonged periods of inundation (e.g., 50 days) are believed to largely deplete alternative electron acceptors through anaerobic respiration¹⁷⁶. However, hydrological fluctuations can induce the repeated renewal of alternative electron acceptor pools, enabling alternative electron acceptors to re-engage in anaerobic respiration processes during post-drought inundation¹⁷⁷. This dynamic process inhibits and delays CH₄ production, potentially explaining the lagged response of CH₄ emissions to water table fluctuations^{176,178}. The inhibitory effect of anaerobic respiration is closely linked to the size of alternative electron acceptor pools, with a focus on iron, manganese, sulfate, and organic alternative electron acceptors due to their significance in natural wetlands.

Regarding iron content in wetlands, groundwater-recharged fens exhibit higher concentrations than precipitation-recharged bogs^{179–181}. In such cases, iron can serve as a crucial alternative electron acceptor under anoxic conditions, but readily undergoes redox cycling through various biotic and abiotic pathways under hydrological fluctuations (Supplementary Fig. 2)^{182–186}. The iron turnover due to water table fluctuations has been shown to inhibit CH₄ production by stimulating microbial anaerobic respiration^{176,178}. Shorter fluctuation periods generally lead to higher iron reduction rates than long fluctuation periods^{187,188}. In addition, compared to the low fluctuation amplitude (1% oxygen), the high fluctuation amplitude (21% oxygen) during the oxic phase induces faster Fe(II) oxidation and produces more short-range ordered iron minerals, while possessing a higher Fe(III) reduction rate during the subsequent anoxic phase⁴². However, it should also be noted that the continuously fluctuating redox environment can induce short-range ordered iron minerals to gradually transform into thermodynamically more stable iron-containing minerals, thereby reducing reactivity¹⁸⁹. Similarly, manganese turnover, prevalent in mineral-rich wetlands, can competitively inhibit CH₄ production¹⁹⁰.

Bacterial sulfate reduction is the most important mechanism of anaerobic decomposition and methanogenesis suppression in brackish wetlands due to the continuous supply of sulfate by tidal exchange^{142,191}. Saltwater inundation is therefore considered a useful strategy to mitigate CH₄ emissions from rewetted wetlands^{192–194}. For instance, compared to freshwater-flooded soils emitting around 303 Mg CH₄ yr⁻¹, the emissions of CH₄ from saltwater-flooded soils can be negligible¹⁹³. Besides differences in water levels and substrate availability, substantial differences in sulfate concentrations are the key factor in this phenomenon. Despite small sulfate pools in freshwater wetlands, periodic hydrological fluctuations, transport of oxygen-rich water in hyporheic zones, and alternating oxic-anoxic conditions in plant root zones can fuel rapid sulfur cycling by active iron cycling or direct oxidation via aerobic sulfur-oxidizing organisms^{151,195–199}, potentially reducing CH₄ production¹⁹⁸.

DOM is considered an important class of organic alternative electron acceptors in organic-rich wetlands, containing electron-donating phenolic and electron-accepting quinone moieties²⁰⁰. They can inhibit CH₄ production either directly via the competition between quinone-reducing bacteria and methanogens for electron donors²⁰¹, or indirectly by chemically oxidizing H₂S to oxidized sulfur species to fuel the dissimilatory sulfate reduction process²⁰², or by acting as an electron shuttle to stimulate microbial Fe(III) reduction²⁰³. Reduced DOM during the inundation period can be oxidized after water table drops²⁰⁴ or by oxygen transported through the vascular aerenchyma²⁰⁵, thereby competitively inhibiting CH₄ production. Importantly, DOM enable reversible and sustainable electron transfer even under continuous redox cycling^{206,207}. This ability allows DOM to reduce 10–166% of the average CH₄ flux in the northern peatlands by inhibiting hydrogenotrophic methanogenesis²⁰⁷. Similarly to natural DOM, pyrogenic carbon, produced by wildfire or prescribed burning and constituting for 5–13% or even up to 50% of the soil organic carbon pool in peatlands^{208,209}, is likewise recognized as a redox-active substance that can be recharged repeatedly, which allows it to competitively inhibit CH₄ production by 13–24%²¹⁰.

In the context of changing climate conditions, redox conditions characterized by spatiotemporal dynamics may strongly drive the regeneration of alternative electron acceptors. This hints that current models that do not include redox cycling of iron, manganese, sulfur, and organic alternative electron acceptors may misestimate CH₄ emissions²¹¹.

Effects of hydrological fluctuations on methane oxidation

Aerobic methane oxidation (MOx) is a biogeochemical process driven by aerobic methanotrophs using oxygen as the terminal electron acceptor. During hydrological fluctuations, MOx inevitably exhibit spatial variations as the availability of oxygen and CH₄ varies with water table. Generally, the overlapping region between the oxygen and CH₄ gradients, where both gases are present in sufficient concentrations, show increased MOx rates^{212,213}. This region is often located close to the water table fluctuation zone (e.g., the mesotelm in peatlands), implying that pulsating soils with CH₄ oxidation hotspots possess higher CH₄ consumption capacity than permanently flooded soils^{140,214}. For example, the highest MOx rate (~160 nmol CH₄ g⁻¹ dry soil h⁻¹) in a drained fen with seasonal flooding was observed at the water table fluctuation layer (50–60 cm below the soil surface)²¹⁵. However, this is not always the case, and to some extent CH₄ availability appears to play a more important role in regulating MOx than oxygen availability, which results in MOx sometimes being higher in wet soils than in dry soils^{216–219}.

CH₄ oxidation can also occur under anoxic conditions (i.e., anaerobic methane oxidation, AOM) using alternative electron acceptors^{220–227}. Compared to dry conditions, the submerged conditions during hydrological fluctuations can create a conducive anoxic environment for AOM as well as abundant electron donors, i.e., CH₄. The availability of alternative electron acceptors is another key determinant of AOM efficiency. Hydrological fluctuations can introduce exogenous alternative electron acceptors, such as increased sulfate availability due to sea level rise³⁹ and saltwater rewetting¹⁹³,

and nitrate/nitrite inputs from agricultural runoff and leaching²²⁸, thereby intensifying corresponding AOM processes. For instance, high sulfate concentrations like 20 mmol L⁻¹ could consume up to 96% of CH₄ in the sulfate-CH₄ transition zone of coastal wetlands through sulfate-dependent AOM processes²²⁰. Nitrogen inputs have been shown to increase the contribution of denitrifying AOM processes to CH₄ consumption in coastal wetlands and rice paddies to ~34% and ~72%, respectively²²⁹. In addition, throughout hydrological fluctuations, the regeneration of alternative electron acceptors not only enhances anaerobic respiration but also offers a continuous supply of electron acceptors for AOM processes. This is why, despite the fact that sulfate-driven AOM is not thermodynamically more favorable than AOM driven by other alternative electron acceptors such as nitrate, nitrite, iron, and manganese, sulfates that rely on rapid sulfur cycling can still be the most important electron acceptor for AOM in wetland hyporheic zones²³⁰. This dynamic interaction between CH₄, alternative electron acceptors, and AOM under hydrologic fluctuations is also well documented in estuarine/coastal wetlands. Material fluxes between oxygenated tidal water and intertidal surface sediments induced by periodic tidal pulses can provide sufficient carbon and nitrogen sources thereby facilitating denitrification-driven AOM processes²³¹.

Effects of hydrological fluctuations on methane cycling microbes

Methanogens

While methanogens have been identified in various anoxic environments worldwide, only *Methanobacteriales*, *Methanomassiliococcales*, *Methanocellales*, *Methanomicrobiales*, and *Methanosarcinales* are commonly found in wetlands^{232,233}. Generally, methanogens are divided into three groups according to their catabolic pathways, including hydrogenotrophic methanogens, methylotrophic methanogens, and acetotrophic methanogens^{233–235}. Traditionally, it is assumed that the acetotrophic and hydrogenotrophic methanogenesis are dominant pathways for controlling CH₄ production in most freshwater wetlands, followed by methylotrophic methanogenesis with a smaller contribution^{233,236}. Hydrogenotrophic methanogenesis may be particularly important in acidic peatlands with incomplete organic matter decomposition and high H₂ partial pressures²³⁷.

The relative abundance of methanogens generally increases with increasing water table and saturation duration^{238,239}, and decreases after water table drops^{240–242}. For instance, the abundance of methanogens in submerged soils was 4.2 and 43.2 times higher than that in the soil-water interface and emergent soils, respectively²⁴³. Conversely, the diversity of methanogens is lower in hydrologically stable zones that are chronically saturated or submerged compared to water table fluctuation zones^{239,244,245}, which is associated with heterogeneous environments with both anoxic and oxic ecological niches caused by water table fluctuations²⁴⁶. Moreover, water table fluctuations can alter the community structure of methanogenic populations. These differences occur due to changes in soil environmental conditions (e.g., pH, nutrients, and organic carbon content)^{244,247}, variations in vegetation distribution²⁴⁸, and particular physiological characteristics exhibited by certain methanogens (e.g., relative higher resistance to oxygen toxicity/exposure/stress)^{249–251}.

Although the abundance and community structure of methanogens are closely affected by water table, they can exhibit certain resistance to drought, and re-establish and further reactivate methanogenic processes after water table has re-emerged, but with varying recovery times ranging from days to years. For example, methanogenesis was rapidly recovered within 3–6 days after wetland inundation in summer²¹, or more than 3 months after the water table exceeded a threshold of 5 cm below soil surface in shrub bogs²⁵². Even 10–12 years after the rewetting of forestry drainage peatlands, methanogens were still not fully re-established²¹³. Differences in the recovery time of methanogens' viability are related to a range of environmental variables, such as the duration and intensity of water table drop, the content of organic substrate, and the regeneration of alternative electron acceptor pools^{253,254}.

Methanotrophs

Generally, aerobic methanotrophs are categorized into Gammaproteobacteria, Alphaproteobacteria, and phylum Verrucomicrobia²⁵⁵. According to the phylogenetic variability, genera of aerobic methanotrophs can be further grouped into type Ia (*Methylobacter*, *Methylomonas*, *Methylomicrobium*, and *Methylosarcina* species within Gammaproteobacteria class), type Ib (*Methylocaldum* and *Methylococcus* species within Gammaproteobacteria class), and type II (*Methylocystis* and *Methylosinus* species within Alphaproteobacteria class)^{216,256}.

Aerobic methanotrophs are somewhat resilient to stress caused by water table fluctuations (including prolonged drought or dry-wet alternation)²⁵⁷, while their activity, community composition, and abundance can be greatly affected and related to the frequency and intensity of water table fluctuations. For example, under the context of increased frequency of wet-dry alternation (i.e., fortnightly drying to weekly drying), type Ia methanotrophs exhibited the greatest resilience, being the least affected compared to other methanotrophs. In contrast, the abundance of type II and type Ib methanotrophs decreased by about two orders of magnitude, but the former gradually recovered over time while the latter kept declining²⁵⁸. In extreme cases, such as the severe drought experienced in Europe in 2018, the abundance of aerobic methanotrophs and the transcript abundance of the *pmoA* gene encoding the β -subunit of the particulate methane monooxygenase enzyme showed an overall downward trend in two rewetted fens, and did not recover after the end of the drought²⁵⁹. In addition, the interaction network of the recovered aerobic methanotrophs was profoundly altered compared to the pre-disturbance period, with a reduction in complexity and modularity in a rewetted bog²⁶⁰, or an increase in complexity but a reduction in modularity in dry-wet rice soils²⁶¹. This implies that water table fluctuations also affect the interactions between methanotrophs and accompanying microorganisms, which in turn affects the recovery of methanotrophs activity and resilience to future environmental disturbances. On the other hand, water table fluctuations can affect aerobic methanotrophs hosted on *Sphagnum* mosses. Although the symbiotic relationship between methanotrophs and *Sphagnum* mosses actively contributes to in situ CH₄ oxidation in *Sphagnum*-dominated wetlands, this effect is more pronounced when the water table is close to *Sphagnum* moss layer rather than during dry conditions^{262,263}.

The AOM process is first associated with sulfate-reducing bacteria in the sulfate-CH₄ transition zone of marine ecosystems. Here, the microbial consortium, consisting of sulfate-reducing bacteria and ANME belonging to ANME-1, ANME-2a/b/c, and ANME-3 clades^{264–269}, utilizes downward-diffusing sulfate as electron acceptors to oxidize CH₄²²⁰. The sulfate-dependent AOM process is also found in brackish wetlands^{220,270}. In the sulfate-poor freshwater wetlands, the AOM process can be coupled with nitrate²²¹ and nitrite^{34,222}, which are catalyzed by *Candidatus* Methanoperedens nitroreducens and *Candidatus* Methyloimrabilis oxyfera (phylum NC10), respectively. *Candidatus* M. nitroreducens are also capable of coupling the reduction of iron and manganese to the AOM process²²³. Moreover, *Candidatus* Methanoperedens ferrireducens²⁷¹ and *Candidatus* Methanoperedens manganireducens/manganicus²⁷² have been identified to independently mediate iron- and manganese-dependent AOM processes, respectively. The reductions of chromium²²⁵, DOM²²⁴, and arsenic^{226,227} are also successively found to involve ANME-2d clade-driven AOM processes.

Considering that anaerobic methanotrophs prefer to inhabit oxygen-limited environments, they are expected to show a positive correlation between their relative abundance with water table and saturation duration^{230,239}. For instance, the abundance of NC10 bacteria in the water table fluctuation zones can increase from 3.2×10^3 to 5.3×10^4 copies g⁻¹ dry soil after about six months of inundation time²⁷³. However, continuous inundation could reduce their abundances due to substrate depletion²³¹. On the other hand, prolonged oxygen exposure can result in anaerobic methanotrophs being subjected to oxidative stress, leading to a decrease in AOM rates^{274,275}. However, intermittent shifts between oxic and anoxic conditions, which occur in oxygen minimum zones, may create suitable

living conditions for anaerobic methanotrophs. In periodically flooded mangrove wetlands, for example, NC10 was observed to dominate in the upper soil layers (i.e., 0–20 cm) due to the presence of trace oxygen²⁷⁶. In addition to oxygen, the distribution of electron acceptors (e.g., nitrite and nitrate) also affects the abundance and activity of anaerobic methanotrophs. In Zoige peatlands, for instance, lower nitrite concentrations in deeper soils can result in the relative abundance of NC10 first increasing and then decreasing with soil depth, peaking around 50–60 cm²⁷⁷. Similarly, the effect of water table on the distribution of electron donors (i.e., CH₄) inevitably shapes the spatial distribution of anaerobic methanotrophs²⁷⁸. Thus, hydrological fluctuations can affect anaerobic methanotrophs by influencing the formation of anoxic environments and the distribution of both electron acceptors and donors.

Effects of hydrological fluctuations on methane transport

CH₄ fluxes from wetlands are the result of the balance between methanogenic and methanotrophic processes in soils, sediments, and water columns⁵. The CH₄ transport pathways, which include plant-mediated transport, ebullition, and diffusion through the water column, determine whether and what proportion of CH₄ can be oxidized, as well as the CH₄ emission rate^{130,279}. Both plant-mediated transport and ebullition allow CH₄ to be rapidly released into the atmosphere almost without undergoing oxidation²⁸⁰, in contrast to the slower diffusion process.

Molecular diffusion, pressurized flow, and transpiration-driven flow constitute the three main pathways for plant-mediated transport. The vegetation composition largely depends on wetland hydrology and varies along a gradient from dry to wet soil conditions²⁸¹, leading to corresponding changes in the primary CH₄ transport mechanisms²⁸². The efficiency of molecular diffusion depends on CH₄ concentration gradients, but is usually lower than pressurized flow. As water levels rise, pressurized flow becomes evident in some emergent plants inhabiting shallow water areas, albeit with a relatively lower flow rate. With further increases in water levels, some emergent plants, as well as floating or even submerged plants in deep water areas, can exhibit higher pressurized flow rates²⁸². Consequently, plant-mediated transport dominates CH₄ transport from some emergent plant-dominated wetlands compared to ebullition and diffusion^{137,283,284}. Trees also play an important role in CH₄ transport by molecular diffusion and transpiration-driven flow that is associated with sap flow in the xylem tissue²⁸⁵. Notably, water table fluctuations are a key driver of changes in tree-mediated CH₄ fluxes^{286–288}. In forested wetlands, tree-mediated CH₄ fluxes can contribute up to ~50–70% of total CH₄ fluxes during wet periods, much higher than the ~3–30% during dry periods²⁸⁶. Thus, not only differences in wetland vegetation composition induced by long-term water table changes can affect CH₄ fluxes via plant-mediated transport, but also short-term seasonal water table changes impact the contributions of plant-mediated transport²⁸⁹.

Ebullition is considered a critical release mechanism especially when the water table is above the soil surface^{290–295}. Ebullition events are closely linked to the potential for CH₄ production in sediments, and can be further exacerbated (6–20% per 1 °C increase) by climate warming due to enhanced microbial metabolism and substrate availability, as well as reduced CH₄ solubility²⁹⁴. Another key contributor to ebullition is hydrostatic changes caused by hydrological fluctuations^{293,295,296}. For example, ebullition caused by falling water table was the main driver of the episodic CH₄ fluxes in some fens^{297,298}, and even contributed up to 65% of the seasonal CH₄ fluxes²⁹⁹. However, when the water table drops below the soil surface, the ebullition pathway would stop as the water-filled pores above the water table drain out and the pore spaces become filled with air instead. Differently, diffusion processes, especially Fickian transport, plays an important role in CH₄ transport when the water table is below the surface. This is mainly because the diffusion coefficient in unsaturated soils is much higher than that in water³⁰⁰. Additionally, the diffusion coefficient is negatively related to the diffusion path length, which increases as the water table rises¹¹⁸.

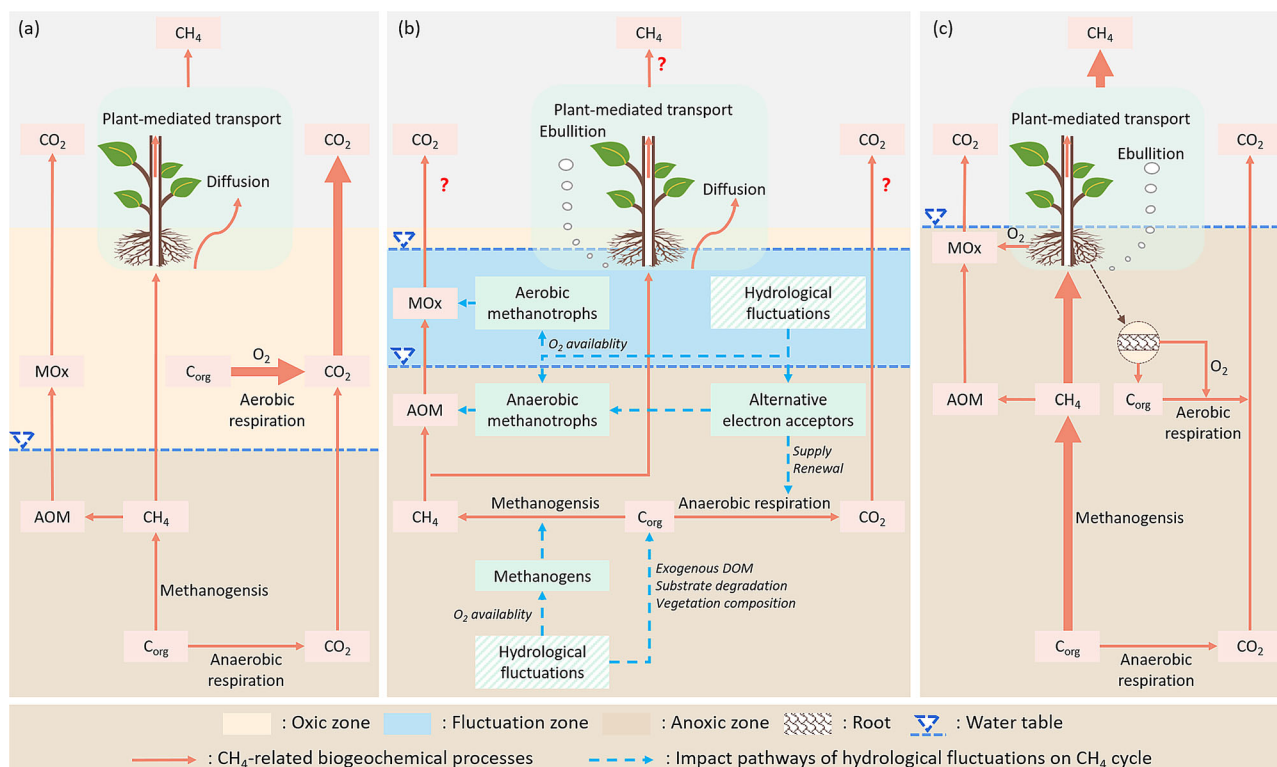


Fig. 4 | Conceptualization of the biogeochemical CH₄ cycle in wetlands under different hydrological scenarios. **a** Prolonged drought. **b** Hydrological fluctuations. **c** Prolonged flooding. The diagrams illustrate the CH₄-related biogeochemical processes (red arrows) and the impact pathways of hydrological fluctuations on the CH₄ cycle (blue dashed arrows). The different zones are marked as oxic zone (light brown), fluctuation zone (blue), anoxic zone (dark brown), and root zone (patterned area). Plant-mediated transport is indicated by plant diagrams, ebullition by bubbles, and diffusion by curved arrows. Aerobic respiration tends to be more active in oxic zones during prolonged drought, while methanogenesis is more prevalent in anoxic

zones during prolonged flooding. Under hydrological fluctuations, CH₄-related biogeochemical processes become more complex due to changes in soil redox state, substrate availability, the pool of alternative electron acceptors, contribution of different CH₄ transport routes, and the community structure and activity of methanogens and methanotrophs. Note: rhizosphere oxygen is particularly critical for aerobic processes under prolonged flooding conditions and is therefore highlighted in the panel c. AOM anaerobic methane oxidation, MOx aerobic methane oxidation, DOM dissolved organic matter, C_{org} organic carbon.

Conclusions and perspectives

This work summarized the possible pathways through which hydrological fluctuations prevalent in wetlands can interfere with the CH₄ cycle (Fig. 4). Compared to prolonged drought with CO₂ as the dominant greenhouse gas (Fig. 4a) and prolonged flooding where CH₄ takes that role (Fig. 4c), the assessment of CH₄ fluxes under hydrological fluctuations appears more challenging. Several factors caused by those fluctuations contribute to this complexity (Fig. 4b): (i) the spatial heterogeneity of soil redox conditions, closely linked to wetland soil properties; (ii) alterations in substrate availability for methanogenesis due to changes in vegetation composition, exogenous substrate supply, and substrate degradation efficiency through chemical reactions and mineral transformations; (iii) changes in the size of alternative electron acceptor pools for anaerobic CO₂ production and the AOM process through the regeneration of alternative electron acceptors or the introduction of exogenous ones; (iv) variations in community structure and activity of CH₄-cycling microorganisms; (v) changes in CH₄ transport pathways and their respective contributions. This reminds us of the need for advanced scientific understanding of hydrological fluctuations, particularly in the context of climate change. To this end, we believe that the following key knowledge gaps still need to be addressed:

(i) To better comprehend the hydrology of wetlands, it is essential to increase the monitoring of water table changes, particularly in tropical wetlands with distinct seasonality¹⁵. This requires transitioning from manual measurements to remote sensing technologies like satellite imagery and Interferometric Synthetic Aperture Radar^{301,302}, which offers a comprehensive view of water table changes at large scales and

high temporal resolutions, along with real-time data from automated sensors and telemetry systems.

- (ii) To deeply understand the biogeochemical reactions of CH₄ production and consumption under different patterns of hydrological fluctuations, we need to explore the impact of electron acceptor pool renewal on the electron flow pathways of organic matter mineralization and AOM processes. This requires more extensive measurements of electron acceptor pools, revealing which biogeochemical processes are occurring in wetlands. Such an electron perspective differs from simple redox potential measurements, which might fail to capture local heterogeneity such as oxic and anoxic microsites¹²³.
- (iii) To better understand seasonal variation patterns and microbial driving mechanisms of CH₄ fluxes, more research is needed to explore the seasonal activity of methanogens and methanotrophs in response to seasonal dynamics of environmental conditions^{43,303–306}.
- (iv) To clearly understand the complex interactions between hydrological fluctuations and CH₄ transport mechanisms, more research is needed to observe the impact of water table fluctuations on three CH₄ transport pathways, i.e., plant-mediated transport, ebullition, and diffusion. Particularly, the current understanding of wetland tree-mediated CH₄ emissions, including transport mechanisms, control factors, and quantitative methods, is still in its infancy. In addition, our knowledge regarding the impact of emergent macrophytes, such as papyrus, on CH₄ production and transport in tropical wetlands³⁰⁷ remains limited.
- (v) To accurately assess global CH₄ budgets, existing process-based models need to be updated by incorporating the understanding of CH₄-related

biogeochemical processes under hydrological fluctuations. Furthermore, it is evident from our review that the current approach of relying solely on annual mean water table depth for calculating CH₄ budgets is insufficient. Instead, we propose implementing alternative approaches that account for more detailed temporal variability (such as daily and seasonal) in water table depth and consider other relevant hydrological variables, such as hydroperiod and inundation events.

- (vi) The impact of hydrological fluctuations on CH₄ cycling may vary under different climatic conditions with different temperatures³⁰⁸. However, the current information remains challenging in deciphering the interactive effects of these two factors. Thus, further research is essential to explore the temperature-affected hydrological effect on CH₄ emissions.

In conclusion, understanding the impact of hydrological fluctuations on CH₄ cycle are essential to accurately estimate CH₄ emissions from wetlands and reduce their contribution to global greenhouse gas emissions. Ignoring this issue would compromise our ability to achieve climate change mitigation targets and sustainable management of wetland ecosystems.

Data availability

The source data for Figures are available as excel files in the Figshare repository (<https://doi.org/10.6084/m9.figshare.26359939>).

Code availability

The code used for generating the raincloud plot in Fig. 1 is available in the Figshare repository (<https://doi.org/10.6084/m9.figshare.26364376>). The code was run using R version 4.1.3. Key packages and their versions include ggplot 3.4.1 for data visualization. Detailed documentation of the parameters and variables used in the scripts is provided within the code files.

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References

- IPCC & Masson-Delmotte, V. et al. (eds). Climate Change 2021: The Physical Science Basis. In *Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (Cambridge University Press, 2021).
- Etminan, M., Myhre, G., Highwood, E. J. & Shine, K. P. Radiative forcing of carbon dioxide, methane, and nitrous oxide: A significant revision of the methane radiative forcing. *Geophys. Res. Lett.* **43**, 12–614 (2016).
- Lan, X., Thoning, K.W. & Dlugokencky, E.J. *Trends in globally-averaged CH₄, N₂O, and SF₆ determined from NOAA Global Monitoring Laboratory measurements*. Version 2024-02, <https://doi.org/10.15138/P8XG-AA10>.
- Tumer, A. J., Frankenberg, C. & Kort, E. A. Interpreting contemporary trends in atmospheric methane. *Proc. Natl Acad. Sci.* **116**, 2805–2813 (2019).
- Dean, J. F. et al. Methane feedbacks to the global climate system in a warmer world. *Rev. Geophys.* **56**, 207–250 (2018).
- Crippa, M., et al. *GHG emissions of all world countries* (Publications Office of the European Union, 2023).
- Schaefer, H. et al. A 21st-century shift from fossil-fuel to biogenic methane emissions indicated by ¹³CH₄. *Science* **352**, 80–84 (2016).
- Oh, Y. et al. Improved global wetland carbon isotopic signatures support post-2006 microbial methane emission increase. *Commun. Earth Environ.* **3**, 159 (2022).
- Basu, S. et al. Estimating emissions of methane consistent with atmospheric measurements of methane and δ ¹³C of methane. *Atmos. Chem. Phys.* **22**, 15351–15377 (2022).
- Nisbet, E. G. et al. Atmospheric methane: Comparison between methane's record in 2006–2022 and during glacial terminations. *Glob. Biogeochem. Cycles* **37**, e2023GB007875 (2023).
- Saunois, M. et al. The global methane budget 2000–2017. *Earth Syst. Sci. Data* **12**, 1561–1623 (2020).
- Peng, S. et al. Wetland emission and atmospheric sink changes explain methane growth in 2020. *Nature* **612**, 477–482 (2022).
- Feng, L., Palmer, P. I., Zhu, S., Parker, R. J. & Liu, Y. Tropical methane emissions explain large fraction of recent changes in global atmospheric methane growth rate. *Nat. Commun.* **13**, 1378 (2022).
- Zhu, Q. et al. Estimating global natural wetland methane emissions using process modelling: Spatio-temporal patterns and contributions to atmospheric methane fluctuations. *Glob. Ecol. Biogeogr.* **24**, 959–972 (2015).
- Murguia-Flores, F., Jaramillo, V. J. & Gallego-Sala, A. Assessing methane emissions from tropical wetlands: uncertainties from natural variability and drivers at the global scale. *Glob. Biogeochem. Cycles* **37**, e2022GB007601 (2023).
- Hamdan, L. J. & Wickland, K. P. Methane emissions from oceans, coasts, and freshwater habitats: New perspectives and feedbacks on climate. *Limnol. Oceanogr.* **61**, S3–S12 (2016).
- Zhang, Z. et al. Development of the global dataset of Wetland Area and Dynamics for Methane Modeling (WAD2M). *Earth Syst. Sci. Data* **13**, 2001–2023 (2021).
- Ganesan, A. L. et al. Advancing scientific understanding of the global methane budget in support of the Paris Agreement. *Glob. Biogeochem. Cycles* **33**, 1475–1512 (2019).
- Hondula, K. L., Jones, C. N. & Palmer, M. A. Effects of seasonal inundation on methane fluxes from forested freshwater wetlands. *Environ. Res. Lett.* **16**, 084016 (2021).
- Guo, M., Li, J., Sheng, C., Xu, J. & Wu, L. A review of wetland remote sensing. *Sensors* **17**, 777 (2017).
- Watras, C. J., Morrison, K. A., Rubsam, J. L. & Buffam, I. Estimates of evapotranspiration from contrasting Wisconsin peatlands based on diel water table oscillations. *Ecohydrology* **10**, e1834 (2017).
- Arsenault, J. et al. The spatial heterogeneity of vegetation, hydrology and water chemistry in a peatland with open-water pools. *Ecosystems* **22**, 1352–1367 (2019).
- Wei, G.-W. et al. Growth responses of eight wetland species to water level fluctuation with different ranges and frequencies. *PLoS one* **14**, e0220231 (2019).
- Knox, S. H. et al. Identifying dominant environmental predictors of freshwater wetland methane fluxes across diurnal to seasonal time scales. *Glob. Change Biol.* **27**, 3582–3604 (2021).
- Hemes, K. S., Runkle, B. R. K., Novick, K. A., Baldocchi, D. D. & Field, C. B. An ecosystem-scale flux measurement strategy to assess natural climate solutions. *Environ. Sci. Technol.* **55**, 3494–3504 (2021).
- Knox, S. H. et al. FLUXNET-CH₄ synthesis activity: Objectives, observations, and future directions. *Bull. Am. Meteorological Soc.* **100**, 2607–2632 (2019).
- Delwiche, K. B. et al. FLUXNET-CH₄: a global, multi-ecosystem dataset and analysis of methane seasonality from freshwater wetlands. *Earth Syst. Sci. Data* **13**, 3607–3689 (2021).
- Morel, X. et al. A new process-based soil methane scheme: Evaluation over Arctic field sites with the ISBA land surface model. *J. Adv. Model. Earth Syst.* **11**, 293–326 (2019).
- Yang, W. H. et al. Evaluating the classical versus an emerging conceptual model of peatland methane dynamics. *Glob. Biogeochem. Cycles* **31**, 1435–1453 (2017).
- Angle, J. C. et al. Methanogenesis in oxygenated soils is a substantial fraction of wetland methane emissions. *Nat. Commun.* **8**, 1–9 (2017).
- Perez-Coronel, E. & Michael Beman, J. Multiple sources of aerobic methane production in aquatic ecosystems include bacterial photosynthesis. *Nat. Commun.* **13**, 6454 (2022).

32. Ordóñez, C. et al. Evaluation of the methane paradox in four adjacent pre-alpine lakes across a trophic gradient. *Nat. Commun.* **14**, 2165 (2023).
33. Raghoebarsing, A. A. et al. A microbial consortium couples anaerobic methane oxidation to denitrification. *Nature* **440**, 918–921 (2006).
34. Ettwig, K. F. et al. Nitrite-driven anaerobic methane oxidation by oxygenic bacteria. *Nature* **464**, 543–548 (2010).
35. Boetius, A. et al. A marine microbial consortium apparently mediating anaerobic oxidation of methane. *Nature* **407**, 623–626 (2000).
36. Beal, E. J., House, C. H. & Orphan, V. J. Manganese- and iron-dependent marine methane oxidation. *Science* **325**, 184–187 (2009).
37. Serrano, L. et al. A new tool for the assessment of severe anthropogenic eutrophication in small shallow water bodies. *Ecol. Indic.* **76**, 324–334 (2017).
38. Zhu, B. et al. Anaerobic oxidization of methane in a minerotrophic peatland: enrichment of nitrite-dependent methane-oxidizing bacteria. *Appl. Environ. Microbiol.* **78**, 8657–8665 (2012).
39. Weston, N. B., Vile, M. A., Neubauer, S. C. & Velinsky, D. J. Accelerated microbial organic matter mineralization following salt-water intrusion into tidal freshwater marsh soils. *Biogeochemistry* **102**, 135–151 (2011).
40. Gauci, V. et al. Sulfur pollution suppression of the wetland methane source in the 20th and 21st centuries. *Proc. Natl Acad. Sci.* **101**, 12583–12587 (2004).
41. Zhang, Z. & Furman, A. Soil redox dynamics under dynamic hydrologic regimes—A review. *Sci. Total Environ.* **763**, 143026 (2021).
42. Chen, C., Meile, C., Wilmoth, J., Barcellos, D. & Thompson, A. Influence of pO₂ on iron redox cycling and anaerobic organic carbon mineralization in a humid tropical forest soil. *Environ. Sci. Technol.* **52**, 7709–7719 (2018).
43. Chadburn, S. E. et al. Modeled microbial dynamics explain the apparent temperature sensitivity of wetland methane emissions. *Glob. Biogeochem. Cycles* **34**, e2020GB006678 (2020).
44. Noyce, G. L. & Megonigal, J. P. Biogeochemical and plant trait mechanisms drive enhanced methane emissions in response to whole-ecosystem warming. *Biogeosciences* **18**, 2449–2463 (2021).
45. Sutton-Grier, A. E. & Megonigal, J. P. Plant species traits regulate methane production in freshwater wetland soils. *Soil Biol. Biochem.* **43**, 413–420 (2011).
46. Rasmussen, T. *Methods for Evaluating Wetland Condition: Wetland Hydrology*, 1–45 (United States Environmental Protection Agency Office of Water, 2008).
47. Brinson, M. Hydrogeomorphic wetland classification system: An overview and modification to better meet the needs of the Natural Resources Conservation Service. USDA Technical Note NO. 190-8-76. Natural Resources Conservation Service Washington, DC, USA (2008).
48. Brinson, M. M. *A hydrogeomorphic classification for wetlands*. Wetlands Research Program Technical Report WRP-DE-4. U.S. Army Corps of Engineers, Washington, DC (1993).
49. Meng, B., Liu, J. L., Bao, K. & Sun, B. Methodologies and management framework for restoration of wetland hydrologic connectivity: a synthesis. *Integr. Environ. Assess. Manag.* **16**, 438–451 (2020).
50. Nygren, M. et al. Changes in seasonality of groundwater level fluctuations in a temperate-cold climate transition zone. *J. Hydrol. X* **8**, 100062 (2020).
51. Zhang, W. et al. Increasing precipitation variability on daily-to-multiple-year time scales in a warmer world. *Sci. Adv.* **7**, eabf8021 (2021).
52. Cai, W. et al. Pantropical climate interactions. *Science* **363**, eaav4236 (2019).
53. Lu, B. & Ren, H. L. What caused the extreme Indian Ocean dipole event in 2019? *Geophys. Res. Lett.* **47**, e2020GL087768 (2020).
54. Barichivich, J. et al. Recent intensification of Amazon flooding extremes driven by strengthened Walker circulation. *Sci. Adv.* **4**, eaat8785 (2018).
55. Bradley, R. S. & Diaz, H. F. Late Quaternary Abrupt Climate Change in the Tropics and Sub-Tropics: The Continental Signal of Tropical Hydroclimatic Events (THEs). *Rev. Geophys.* **59**, e2020RG000732 (2021).
56. Mitsch, W. J. et al. Tropical wetlands: seasonal hydrologic pulsing, carbon sequestration, and methane emissions. *Wetl. Ecol. Manag.* **18**, 573–586 (2010).
57. Nisbet, E. G. Climate feedback on methane from wetlands. *Nat. Clim. Change* **13**, 421–422 (2023).
58. Lázaro, W. L., Oliveira-Júnior, E. S., Silva, C. J. D., Castrillon, S. K. I. & Muniz, C. C. Climate change reflected in one of the largest wetlands in the world: an overview of the Northern Pantanal water regime. *Acta Limnologica Brasiliensia* **32**, e104 (2020).
59. Xi, Y., Peng, S., Ciais, P. & Chen, Y. Future impacts of climate change on inland Ramsar wetlands. *Nat. Clim. Change* **11**, 45–51 (2021).
60. Xu, D. et al. Climate change will reduce North American inland wetland areas and disrupt their seasonal regimes. *Nat. Commun.* **15**, 2438 (2024).
61. Xing, T. et al. Sink or Source: Alternative Roles of Glacier Foreland Meadow Soils in Methane Emission Is Regulated by Glacier Melting on the Tibetan Plateau. *Front. Microbiol.* **13**, 862242 (2022).
62. Patzner, M. S. et al. Seasonal Fluctuations in Iron Cycling in Thawing Permafrost Peatlands. *Environ. Sci. Technol.* **56**, 4620–4631 (2022).
63. Walvoord, M. A. & Kurylyk, B. L. Hydrologic impacts of thawing permafrost—A review. *Vadose Zone J.* **15**, 20 (2016).
64. Lamontagne-Hallé, P., McKenzie, J. M., Kurylyk, B. L. & Zipper, S. C. Changing groundwater discharge dynamics in permafrost regions. *Environ. Res. Lett.* **13**, 084017 (2018).
65. Brunna, S. E., Schröter, J., Timmermann, R., Rietbroek, R. & Kusche, J. Modeled steric and mass-driven sea level change caused by Greenland Ice Sheet melting. *J. Geodyn.* **59**, 219–225 (2012).
66. Change, I. Climate change 2007: The physical science basis. *Agenda* **6**, 333 (2007).
67. Hinkel, J. et al. Sea-level rise scenarios and coastal risk management. *Nat. Clim. Change* **5**, 188–190 (2015).
68. Dohong, A., Aziz, A. A. & Dargusch, P. A review of the drivers of tropical peatland degradation in South-East Asia. *Land Use Policy* **69**, 349–360 (2017).
69. Auterives, C., Aquilina, L., Bour, O., Davranche, M. & Paquereau, V. Contribution of climatic and anthropogenic effects to the hydric deficit of peatlands. *Hydrological Process.* **25**, 2890–2906 (2011).
70. Zhang, X., Liu, H., Baker, C. & Graham, S. Restoration approaches used for degraded peatlands in Ruergai (Zoige), Tibetan Plateau, China, for sustainable land management. *Ecol. Eng.* **38**, 86–92 (2012).
71. Strack, M. et al. Petroleum exploration increases methane emissions from northern peatlands. *Nat. Commun.* **10**, 2804 (2019).
72. Lovitt, J. et al. UAV remote sensing can reveal the effects of low-impact seismic lines on surface morphology, hydrology, and methane (CH₄) release in a boreal treed bog. *J. Geophys. Res. Biogeosci.* **123**, 1117–1129 (2018).
73. van Rensen, C. K., Nielsen, S. E., White, B., Vinge, T. & Lieffers, V. J. Natural regeneration of forest vegetation on legacy seismic lines in boreal habitats in Alberta's oil sands region. *Biol. Conserv.* **184**, 127–135 (2015).
74. Strack, M., Softa, D., Bird, M. & Xu, B. Impact of winter roads on boreal peatland carbon exchange. *Glob. Change Biol.* **24**, e201–e212 (2018).

75. Fluet-Chouinard, E. et al. Extensive global wetland loss over the past three centuries. *Nature* **614**, 281–286 (2023).
76. Kreyling, J. et al. Rewetting does not return drained fen peatlands to their old selves. *Nat. Commun.* **12**, 1–8 (2021).
77. Batzer, D. P., Rader, R. B. & Wissinger, S. A. *Invertebrates in freshwater wetlands of North America: ecology and management* (John Wiley & Sons, 1999).
78. Zakharova, E. A., Kouraev, A. V., Rémy, F., Zemtsov, V. A. & Kirpotin, S. N. Seasonal variability of the Western Siberia wetlands from satellite radar altimetry. *J. Hydrol.* **512**, 366–378 (2014).
79. Ratcliffe, J. L., Campbell, D. I., Clarkson, B. R., Wall, A. M. & Schipper, L. A. Water table fluctuations control CO₂ exchange in wet and dry bogs through different mechanisms. *Sci. Total Environ.* **655**, 1037–1046 (2019).
80. Burt, T. P. et al. Water table fluctuations in the riparian zone: comparative results from a pan-European experiment. *J. Hydrol.* **265**, 129–148 (2002).
81. Chamberlain, S. D., Boughton, E. H. & Sparks, J. P. Underlying ecosystem emissions exceed cattle-emitted methane from subtropical lowland pastures. *Ecosystems* **18**, 933–945 (2015).
82. Amatya, D. M., Chescheir, G. M., Williams, T. M., Skaggs, R. W. & Tian, S. Long-term water table dynamics of forested wetlands: drivers and their effects on wetland hydrology in the southeastern Atlantic coastal plain. *Wetlands* **40**, 65–79 (2020).
83. Walter, B. P., Heimann, M. & Matthews, E. Modeling modern methane emissions from natural wetlands: 1. Model description and results. *J. Geophys. Res. Atmosp.* **106**, 34189–34206 (2001).
84. van der Valk, A. G., Volin, J. C. & Wetzel, P. R. Predicted changes in interannual water-level fluctuations due to climate change and its implications for the vegetation of the Florida Everglades. *Environ. Manag.* **55**, 799–806 (2015).
85. Shihao, C. Wetland hydrological dynamics and methane emissions. Dataset. <https://doi.org/10.6084/m9.figshare.26359939.v3> (2024).
86. Shihao, C. Raincloud plot code for water table fluctuations in different wetlands. Software. <https://doi.org/10.6084/m9.figshare.26364376.v1> (2024).
87. Price, J. S., McCarter, C. P. R. & Quinton, W. L. *Groundwater in Peat and Peatlands* (Groundwater Project, 2023).
88. Duval, T. P. & Waddington, J. M. Extreme variability of water table dynamics in temperate calcareous fens: Implications for biodiversity. *Hydrological Process.* **25**, 3790–3802 (2011).
89. Ridolfi, L., D’Odorico, P. & Laio, F. Effect of vegetation–water table feedbacks on the stability and resilience of plant ecosystems. *Water Resour. Res.* **42**, W01201 (2006).
90. Johansen, O. M., Andersen, D. K., Ejrnæs, R. & Pedersen, M. L. Relations between vegetation and water level in groundwater dependent terrestrial ecosystems (GWDTEs). *Limnologica* **68**, 130–141 (2018).
91. Le Maitre, D. C., Scott, D. F. & Colvin, C. Review of information on interactions between vegetation and groundwater. *Water SA* **25**, 137–152 (1999).
92. Budny, M. L. & Benscoter, B. W. Shrub encroachment increases transpiration water loss from a subtropical wetland. *Wetlands* **36**, 631–638 (2016).
93. Kayendeke, E. J., Kansime, F., French, H. K. & Bamutaze, Y. Spatial and temporal variation of papyrus root mat thickness and water storage in a tropical wetland system. *Sci. Total Environ.* **642**, 925–936 (2018).
94. Dise, N. B. Peatland response to global change. *Science* **326**, 810–811 (2009).
95. Tu, N. et al. Effects of moss overlay on soil patch infiltration and runoff in karst rocky desertification slope land. *Water* **14**, 3429 (2022).
96. Carlson Mazur, M. L., Wiley, M. J. & Wilcox, D. A. Estimating evapotranspiration and groundwater flow from water-table fluctuations for a general wetland scenario. *Ecohydrology* **7**, 378–390 (2014).
97. Baird, A. J. & Low, R. G. The water table: Its conceptual basis, its measurement and its usefulness as a hydrological variable. *Hydrological Process.* **36**, e14622 (2022).
98. Ahmad, S. et al. Meteorological controls on water table dynamics in fen peatlands depend on management regimes. *Front. Earth Sci.* **9**, 630469 (2021).
99. Junk, W. J. et al. Current state of knowledge regarding the world’s wetlands and their future under global climate change: a synthesis. *Aquat. Sci.* **75**, 151–167 (2013).
100. Luo, F.-L., Jiang, X.-X., Li, H.-L. & Yu, F.-H. Does hydrological fluctuation alter impacts of species richness on biomass in wetland plant communities? *J. Plant Ecol.* **9**, 434–441 (2016).
101. Koebsch, F. et al. The impact of occasional drought periods on vegetation spread and greenhouse gas exchange in rewetted fens. *Philos. Trans. R. Soc. B* **375**, 20190685 (2020).
102. Chen, J. et al. The influence of the 2022 extreme drought on groundwater hydrodynamics in the floodplain wetland of Poyang Lake using a modeling assessment. *J. Hydrol.* **626**, 130194 (2023).
103. Zhang, T., Liu, X. & An, Y. Fluctuating water level effects on soil greenhouse gas emissions of returning farmland to wetland. *J. Soils Sediment.* **20**, 3857–3866 (2020).
104. Sabrekov, A. F., Runkle, B. R. K., Glagolev, M. V., Kleptsova, I. E. & Maksyutov, S. S. Seasonal variability as a source of uncertainty in the West Siberian regional CH₄ flux upscaling. *Environ. Res. Lett.* **9**, 045008 (2014).
105. Moore, T. R. et al. A multi-year record of methane flux at the Mer Bleue bog, southern Canada. *Ecosystems* **14**, 646–657 (2011).
106. MacDonald, J. A. et al. Methane emission rates from a northern wetland; response to temperature, water table and transport. *Atmos. Environ.* **32**, 3219–3227 (1998).
107. Zhao, M. et al. Responses of soil CO₂ and CH₄ emissions to changing water table level in a coastal wetland. *J. Clean. Prod.* **269**, 122316 (2020).
108. Taylor, M. A. et al. Quantifying the effects sizes of common controls on methane emissions from an ombrotrophic peat bog. *J. Geophys. Res.* **128**, e2022JG007271 (2023).
109. Gao, W., Gao, D., Cai, T. & Liang, H. Driving Factors on Greenhouse Gas Emissions in Permafrost Region of Daxing’an Mountains, Northeast China. *J. Geophys. Res.* **127**, e2021JG006581 (2022).
110. Tian, W. et al. Water table level controls methanogenic and methanotrophic communities and methane emissions in a Sphagnum-dominated peatland. *Microbiol. Spectr.* **11**, e01992–01923 (2023).
111. Strack, M., Keith, A. M. & Xu, B. Growing season carbon dioxide and methane exchange at a restored peatland on the Western Boreal Plain. *Ecol. Eng.* **64**, 231–239 (2014).
112. Sun, X., Mu, C. & Song, C. Seasonal and spatial variations of methane emissions from montane wetlands in Northeast China. *Atmos. Environ.* **45**, 1809–1816 (2011).
113. Turetsky, M. R. et al. A synthesis of methane emissions from 71 northern, temperate, and subtropical wetlands. *Glob. Change Biol.* **20**, 2183–2197 (2014).
114. Evans, C. D. et al. Overriding water table control on managed peatland greenhouse gas emissions. *Nature* **593**, 548–552 (2021).
115. Inubushi, K. et al. Factors influencing methane emission from peat soils: comparison of tropical and temperate wetlands. *Nutrient Cycl. Agroecosyst.* **71**, 93–99 (2005).

116. Tiemeyer, B. et al. A new methodology for organic soils in national greenhouse gas inventories: Data synthesis, derivation and application. *Ecol. Indic.* **109**, 105838 (2020).
117. Jackowicz-Korczyński, M. et al. Annual cycle of methane emission from a subarctic peatland. *J. Geophys. Res.* **115**, G02009 (2010).
118. Calabrese, S., Garcia, A., Wilmoth, J. L., Zhang, X. & Porporato, A. Critical inundation level for methane emissions from wetlands. *Environ. Res. Lett.* **16**, 044038 (2021).
119. Moore, T. R. & Dalva, M. The influence of temperature and water table position on carbon dioxide and methane emissions from laboratory columns of peatland soils. *J. Soil Sci.* **44**, 651–664 (1993).
120. Bao, T., Jia, G. & Xu, X. Wetland heterogeneity determines methane emissions: a pan-arctic synthesis. *Environ. Sci. Technol.* **55**, 10152–10163 (2021).
121. Boon, P. I., Mitchell, A. & Lee, K. Effects of wetting and drying on methane emissions from ephemeral floodplain wetlands in south-eastern Australia. *Hydrobiologia* **357**, 73–87 (1997).
122. Zhu, X., Song, C., Chen, W., Zhang, X. & Tao, B. Effects of water regimes on methane emissions in peatland and gley marsh. *Vadose Zone J.* **17**, 1–7 (2018).
123. Blodau, C. & Moore, T. R. Micro-scale CO₂ and CH₄ dynamics in a peat soil during a water fluctuation and sulfate pulse. *Soil Biol. Biochem.* **35**, 535–547 (2003).
124. Knorr, K.-H., Oosterwoud, M. R. & Blodau, C. Experimental drought alters rates of soil respiration and methanogenesis but not carbon exchange in soil of a temperate fen. *Soil Biol. Biochem.* **40**, 1781–1791 (2008).
125. Zhu, Q. et al. Interannual variation in methane emissions from tropical wetlands triggered by repeated El Niño Southern Oscillation. *Glob. Change Biol.* **23**, 4706–4716 (2017).
126. Tangen, B. A. & Bansal, S. Hydrologic lag effects on wetland greenhouse gas fluxes. *Atmosphere* **10**, 269 (2019).
127. Ballantyne, D. M., Hribljan, J. A., Pypker, T. G. & Chimner, R. A. Long-term water table manipulations alter peatland gaseous carbon fluxes in Northern Michigan. *Wetl. Ecol. Manag.* **22**, 35–47 (2014).
128. Unger, V. et al. Congruent changes in microbial community dynamics and ecosystem methane fluxes following natural drought in two restored fens. *Soil Biol. Biochem.* **160**, 108348 (2021).
129. Kalthori, A. et al. Temporally dynamic carbon dioxide and methane emission factors for rewetted peatlands. *Commun. Earth Environ.* **5**, 62 (2024).
130. Bubier, J. L. & Moore, T. R. An ecological perspective on methane emissions from northern wetlands. *Trends Ecol. Evolution* **9**, 460–464 (1994).
131. Chamberlain, S. D. et al. Influence of transient flooding on methane fluxes from subtropical pastures. *J. Geophys. Res.* **121**, 965–977 (2016).
132. Tokida, T. et al. Episodic release of methane bubbles from peatland during spring thaw. *Chemosphere* **70**, 165–171 (2007).
133. Lunt, M. F. et al. Rain-fed pulses of methane from East Africa during 2018–2019 contributed to atmospheric growth rate. *Environ. Res. Lett.* **16**, 024021 (2021).
134. Hargreaves, K. J., Fowler, D., Pitcairn, C. E. R. & Aurela, M. Annual methane emission from Finnish mires estimated from eddy covariance campaign measurements. *Theor. Appl. Climatol.* **70**, 203–213 (2001).
135. Feng, X. et al. Climate sensitivity of peatland methane emissions mediated by seasonal hydrologic dynamics. *Geophys. Res. Lett.* **47**, e2020GL088875 (2020).
136. Bloom, A. A., Palmer, P. I., Fraser, A., Reay, D. S. & Frankenberg, C. Large-scale controls of methanogenesis inferred from methane and gravity spaceborne data. *Science* **327**, 322–325 (2010).
137. Ueyama, M. et al. Modeled production, oxidation and transport processes of wetland methane emissions in temperate, boreal, and Arctic regions. *Global Change Biol.* **29**, 2313–2334 (2023).
138. Hill, A. C., Forbrich, I., Schäfer, K. V. R. & Vargas, R. Empirical Dynamic Modeling Reveals Complexity of Methane Fluxes in a Temperate Salt Marsh. *J. Geophys. Res.* **129**, e2023JG007630 (2024).
139. Barczok, M., Smith, C., Di Domenico, N., Kinsman-Costello, L. & Herndon, E. Variability in soil redox response to seasonal flooding in a vernal pond. *Front. Environ. Sci.* **11**, 1114814 (2023).
140. Chowdhury, T. R. & Dick, R. P. Ecology of aerobic methanotrophs in controlling methane fluxes from wetlands. *Appl. Soil Ecol.* **65**, 8–22 (2013).
141. Strack, M. & Waddington, J. M. Response of peatland carbon dioxide and methane fluxes to a water table drawdown experiment. *Global Biogeochem. Cycles* **21**, GB1007 (2007).
142. Bridgman, S. D., Cadillo-Quiroz, H., Keller, J. K. & Zhuang, Q. Methane emissions from wetlands: biogeochemical, microbial, and modeling perspectives from local to global scales. *Glob. Change Biol.* **19**, 1325–1346 (2013).
143. Currie, J. A. Gaseous diffusion in the aeration of aggregated soils. *Soil Sci.* **92**, 40–45 (1961).
144. Keiluweit, M., Gee, K., Denney, A. & Fendorf, S. Anoxic microsites in upland soils dominantly controlled by clay content. *Soil Biol. Biochem.* **118**, 42–50 (2018).
145. Lacroix, E. M. et al. Consider the Anoxic Microsite: Acknowledging and Appreciating Spatiotemporal Redox Heterogeneity in Soils and Sediments. *ACS Earth Space Chem.* **7**, 1592–1609 (2023).
146. Daugherty, E. E. et al. Hydrogeomorphic controls on soil carbon composition in two classes of subalpine wetlands. *Biogeochemistry* **145**, 161–175 (2019).
147. Itoh, M. et al. Hydrologic effects on methane dynamics in riparian wetlands in a temperate forest catchment. *J. Geophys. Res.* **112**, G01019 (2007).
148. Sakabe, A. et al. Controlling factors of seasonal variation of stem methane emissions from *Alnus japonica* in a riparian wetland of a temperate forest. *J. Geophys. Res.* **126**, e2021JG006326 (2021).
149. White, I., Melville, M. D., Wilson, B. P. & Sammut, J. Reducing acidic discharges from coastal wetlands in eastern Australia. *Wetl. Ecol. Manag.* **5**, 55–72 (1997).
150. Yang, C., Zhang, Y. K., Liu, Y., Yang, X. & Liu, C. Model-based analysis of the effects of dam-induced river water and groundwater interactions on hydro-biogeochemical transformation of redox sensitive contaminants in a hyporheic zone. *Water Resour. Res.* **54**, 5973–5985 (2018).
151. Torgeson, J. M. et al. Hydrobiogeochemical interactions in the hyporheic zone of a sulfate-impacted, freshwater stream and riparian wetland ecosystem. *Environ. Sci. Processes Impacts* **24**, 1360–1382 (2022).
152. Dong, C., Zhu, W., Zhao, Y. Q. & Gao, M. Diurnal fluctuations in root oxygen release rate and dissolved oxygen budget in wetland mesocosm. *Desalination* **272**, 254–258 (2011).
153. Gauthier, M., Bradley, R. L. & Šimek, M. More evidence that anaerobic oxidation of methane is prevalent in soils: Is it time to upgrade our biogeochemical models? *Soil Biol. Biochem.* **80**, 167–174 (2015).
154. Ström, L., Tagesson, T., Mastepanov, M. & Christensen, T. R. Presence of *Eriophorum scheuchzeri* enhances substrate availability and methane emission in an Arctic wetland. *Soil Biol. Biochem.* **45**, 61–70 (2012).
155. Liu, D., Ding, W., Yuan, J., Xiang, J. & Lin, Y. Substrate and/or substrate-driven changes in the abundance of methanogenic archaea cause seasonal variation of methane production potential in

- species-specific freshwater wetlands. *Appl. Microbiol. Biotechnol.* **98**, 4711–4721 (2014).
156. Welti, N., Hayes, M. & Lockington, D. Seasonal nitrous oxide and methane emissions across a subtropical estuarine salinity gradient. *Biogeochemistry* **132**, 55–69 (2017).
 157. Chanton, J. P. et al. Radiocarbon evidence for the importance of surface vegetation on fermentation and methanogenesis in contrasting types of boreal peatlands. *Global Biogeochemical Cycles* **22**, GB4022 (2008).
 158. Valk, A. G. v. d. Water-level fluctuations in North American prairie wetlands. *Hydrobiologia* **539**, 171–188 (2005).
 159. Steinman, A. D., Ogdahl, M. E., Weinert, M. & Uzarski, D. G. Influence of water-level fluctuation duration and magnitude on sediment–water nutrient exchange in coastal wetlands. *Aquat. Ecol.* **48**, 143–159 (2014).
 160. Garssen, A. G., Verhoeven, J. T. A. & Soons, M. B. Effects of climate-induced increases in summer drought on riparian plant species: A meta-analysis. *Freshw. Biol.* **59**, 1052–1063 (2014).
 161. Ryberg, E. E. et al. Postglacial peatland vegetation succession in Store Mosse bog, south-central Sweden: An exploration of factors driving species change. *Boreas* **51**, 651–666 (2022).
 162. Kokkonen, N. A. K. et al. Responses of peatland vegetation to 15-year water level drawdown as mediated by fertility level. *J. Vegetation Sci.* **30**, 1206–1216 (2019).
 163. Kokkonen, N. et al. Two Mechanisms Drive Changes in Boreal Peatland Photosynthesis Following Long-Term Water Level Drawdown: Species Turnover and Altered Photosynthetic Capacity. *Ecosystems* **25**, 1601–1618 (2022).
 164. LaCroix, R. E. et al. Shifting mineral and redox controls on carbon cycling in seasonally flooded mineral soils. *Biogeosciences* **16**, 2573–2589 (2019).
 165. Chen, H. et al. Methane emissions during different freezing–thawing periods from a fen on the Qinghai–Tibetan Plateau: Four years of measurements. *Agric. For. Meteorol.* **297**, 108279 (2021).
 166. Walter, K. M., Smith, L. C. & Stuart Chapin III, F. Methane bubbling from northern lakes: present and future contributions to the global methane budget. *Philos. Trans. R. Soc. A Math. Phys. Eng. Sci.* **365**, 1657–1676 (2007).
 167. Tang, X. et al. Changing microbiome community structure and functional potential during permafrost thawing on the Tibetan Plateau. *FEMS Microbiol. Ecol.* **99**, fiad117 (2023).
 168. Van Huissteden, J., Maximov, T. C. & Dolman, A. J. High methane flux from an arctic floodplain (Indigirka lowlands, eastern Siberia). *J. Geophys. Res.* **110**, G02002 (2005).
 169. Li, Y. et al. Changes in Water Chemistry Associated with Rainstorm Events Increase Carbon Emissions from the Inflowing River Mouth of a Major Drinking Water Reservoir. *Environ. Sci. Technol.* **56**, 16494–16505 (2022).
 170. Wilmoth, J. L. et al. The role of oxygen in stimulating methane production in wetlands. *Glob. Change Biol.* **27**, 5831–5847 (2021).
 171. Yu, G.-H. & Kuzyakov, Y. Fenton chemistry and reactive oxygen species in soil: Abiotic mechanisms of biotic processes, controls and consequences for carbon and nutrient cycling. *Earth Sci. Rev.* **214**, 103525 (2021).
 172. Naughton, H. R. et al. Reactive iron, not fungal community, drives organic carbon oxidation potential in floodplain soils. *Soil Biol. Biochem.* **178**, 108962 (2023).
 173. Zhao, G. et al. Redox Oscillations Activate Thermodynamically Stable Iron Minerals for Enhanced Reactive Oxygen Species Production. *Environ. Sci. Technol.* **57**, 8628–8637 (2023).
 174. Hömberg, A., Broder, T., Knorr, K.-H. & Schaller, J. Divergent effect of silicon on greenhouse gas production from reduced and oxidized peat organic matter. *Geoderma* **386**, 114916 (2021).
 175. Patzner, M. S. et al. Iron mineral dissolution releases iron and associated organic carbon during permafrost thaw. *Nat. Commun.* **11**, 6329 (2020).
 176. Knorr, K.-H. & Blodau, C. Impact of experimental drought and rewetting on redox transformations and methanogenesis in mesocosms of a northern fen soil. *Soil Biol. Biochem.* **41**, 1187–1198 (2009).
 177. Peiffer, S. et al. A biogeochemical–hydrological framework for the role of redox-active compounds in aquatic systems. *Nat. Geosci.* **14**, 264–272 (2021).
 178. Knorr, K.-H., Lischeid, G. & Blodau, C. Dynamics of redox processes in a minerotrophic fen exposed to a water table manipulation. *Geoderma* **153**, 379–392 (2009).
 179. Curtinrich, H. J., Sebestyen, S. D., Griffiths, N. A. & Hall, S. J. Warming stimulates iron-mediated carbon and nutrient cycling in mineral-poor peatlands. *Ecosystems* **25**, 44–60 (2022).
 180. Wang, H., River, M. & Richardson, C. J. Does an ‘iron gate’ carbon preservation mechanism exist in organic-rich wetlands? *Soil Biol. Biochem.* **135**, 48–50 (2019).
 181. Hájek, M., Horsák, M., Hájková, P. & Dítě, D. Habitat diversity of central European fens in relation to environmental gradients and an effort to standardise fen terminology in ecological studies. *Perspect. Plant Ecol. Evol. Syst.* **8**, 97–114 (2006).
 182. Karvinen, A., Saarnio, S. & Kankaala, P. Sediment iron content does not play a significant suppressive role on methane emissions from boreal littoral sedge (*Carex*) vegetation. *Aquat. Bot.* **127**, 70–79 (2015).
 183. Kappler, A. et al. An evolving view on biogeochemical cycling of iron. *Nat. Rev. Microbiol.* **19**, 360–374 (2021).
 184. Lueder, U., Jørgensen, B. B., Kappler, A. & Schmidt, C. Photochemistry of iron in aquatic environments. *Environ. Sci.: Process. Impacts* **22**, 12–24 (2020).
 185. Borch, T. et al. Biogeochemical redox processes and their impact on contaminant dynamics. *Environ. Sci. Technol.* **44**, 15–23 (2010).
 186. Huang, J. et al. Fe(II) redox chemistry in the environment. *Chem. Rev.* **121**, 8161–8233 (2021).
 187. Barcellos, D., Cyle, K. T. & Thompson, A. Faster redox fluctuations can lead to higher iron reduction rates in humid forest soils. *Biogeochemistry* **137**, 367–378 (2018).
 188. Ginn, B., Meile, C., Wilmoth, J., Tang, Y. & Thompson, A. Rapid iron reduction rates are stimulated by high-amplitude redox fluctuations in a tropical forest soil. *Environ. Sci. Technol.* **51**, 3250–3259 (2017).
 189. Thompson, A., Chadwick, O. A., Rancourt, D. G. & Chorover, J. Iron-oxide crystallinity increases during soil redox oscillations. *Geochim. Cosmochim. Acta* **70**, 1710–1727 (2006).
 190. Lovley, D. R. Dissimilatory Fe(III) and Mn(IV) reduction. *Microbiological Rev.* **55**, 259–287 (1991).
 191. Torres-Alvarado, R., Ramírez-Vives, F., Fernández, F. J. & Barriga-Sosa, I. Methanogenesis and methane oxidation in wetlands: implications in the global carbon cycle. *Hidrobiológica* **15**, 327–349 (2005).
 192. Pönisch, D. L. et al. Nutrient release and flux dynamics of CO₂, CH₄ and N₂O in a coastal peatland driven by actively induced rewetting with brackish water from the Baltic Sea. *Biogeosci. Discuss.* **2022**, 1–45 (2022).
 193. Petersen, S. G. G., Kristensen, E. & Quintana, C. O. Greenhouse gas emissions from agricultural land before and after permanent flooding with seawater or freshwater. *Estuaries Coasts* **46**, 1459–1474 (2023).
 194. Kroeger, K. D., Crooks, S., Moseman-Valtierra, S. & Tang, J. Restoring tides to reduce methane emissions in impounded wetlands: A new and potent Blue Carbon climate change intervention. *Sci. Rep.* **7**, 11914 (2017).

195. Prietzel, J. et al. Sulfur speciation in well-aerated and wetland soils in a forested catchment assessed by sulfur K-edge X-ray absorption near-edge spectroscopy (XANES). *J. Plant Nutr. Soil Sci.* **172**, 393–403 (2009).
196. Smemo, K. A. & Yavitt, J. B. Anaerobic oxidation of methane: an underappreciated aspect of methane cycling in peatland ecosystems? *Biogeosciences* **8**, 779–793 (2011).
197. Hansel, C. M. et al. Dominance of sulfur-fueled iron oxide reduction in low-sulfate freshwater sediments. *ISME J.* **9**, 2400–2412 (2015).
198. Pester, M., Knorr, K.-H., Friedrich, M. W., Wagner, M. & Loy, A. Sulfate-reducing microorganisms in wetlands—fameless actors in carbon cycling and climate change. *Front. Microbiol.* **3**, 72 (2012).
199. Karimian, N., Johnston, S. G. & Burton, E. D. Iron and sulfur cycling in acid sulfate soil wetlands under dynamic redox conditions: A review. *Chemosphere* **197**, 803–816 (2018).
200. Walpen, N., Getzinger, G. J., Schroth, M. H. & Sander, M. Electron-donating phenolic and electron-accepting quinone moieties in peat dissolved organic matter: quantities and redox transformations in the context of peat biogeochemistry. *Environ. Sci. Technol.* **52**, 5236–5245 (2018).
201. Cervantes, F. J., Van Der Velde, S., Lettinga, G. & Field, J. A. Competition between methanogenesis and quinone respiration for ecologically important substrates in anaerobic consortia. *FEMS Microbiol. Ecol.* **34**, 161–171 (2000).
202. Heitmann, T. & Blodau, C. Oxidation and incorporation of hydrogen sulfide by dissolved organic matter. *Chem. Geol.* **235**, 12–20 (2006).
203. Lovley, D. R. et al. Humic substances as a mediator for microbially catalyzed metal reduction. *Acta Hydrochim. Hydrobiol.* **26**, 152–157 (1998).
204. Heitmann, T., Goldhammer, T., Beer, J. & Blodau, C. Electron transfer of dissolved organic matter and its potential significance for anaerobic respiration in a northern bog. *Glob. Change Biol.* **13**, 1771–1785 (2007).
205. Agethen, S., Sander, M., Waldemer, C. & Knorr, K.-H. Plant rhizosphere oxidation reduces methane production and emission in rewetted peatlands. *Soil Biol. Biochem.* **125**, 125–135 (2018).
206. Tan, W. et al. Molecular-weight-dependent redox cycling of humic substances of paddy soils over successive anoxic and oxic alternations. *Land Degrad. Dev.* **30**, 1130–1144 (2019).
207. Klüpfel, L., Piepenbrock, A., Kappler, A. & Sander, M. Humic substances as fully regenerable electron acceptors in recurrently anoxic environments. *Nat. Geosci.* **7**, 195–200 (2014).
208. Leifeld, J. et al. Pyrogenic carbon contributes substantially to carbon storage in intact and degraded northern peatlands. *Land Degrad. Dev.* **29**, 2082–2091 (2018).
209. Gao, C., Cong, J., Sun, Y., Han, D. & Wang, G. Variability in pyrogenic carbon properties generated by different burning temperatures and peatland plant litters: implication for identifying fire intensity and fuel types. *Int. J. Wildland Fire* **31**, 395–408 (2022).
210. Sun, T. et al. Suppressing peatland methane production by electron snorkeling through pyrogenic carbon in controlled laboratory incubations. *Nat. Commun.* **12**, 1–9 (2021).
211. Sulman, B. N. et al. Simulated hydrological dynamics and coupled iron redox cycling impact methane production in an Arctic soil. *J. Geophys. Res.* **127**, e2021JG006662 (2022).
212. Freeman, C. et al. Contrasted effects of simulated drought on the production and oxidation of methane in a mid-Wales wetland. *Soil Biol. Biochem.* **34**, 61–67 (2002).
213. Juottonen, H. et al. Methane-cycling microbial communities and methane emission in natural and restored peatlands. *Appl. Environ. Microbiol.* **78**, 6386–6389 (2012).
214. Altor, A. E. & Mitsch, W. J. Methane and carbon dioxide dynamics in wetland mesocosms: effects of hydrology and soils. *Ecol. Appl.* **18**, 1307–1320 (2008).
215. Jerman, V., Danevčič, T. & Mandić-Mulec, I. Methane cycling in a drained wetland soil profile. *J. Soils Sediment.* **17**, 1874–1882 (2017).
216. Siljanen, H. M. P. et al. Hydrology is reflected in the functioning and community composition of methanotrophs in the littoral wetland of a boreal lake. *FEMS Microbiol. Ecol.* **75**, 430–445 (2011).
217. Siljanen, H. M. P., Saari, A., Bodrossy, L. & Martikainen, P. J. Seasonal variation in the function and diversity of methanotrophs in the littoral wetland of a boreal eutrophic lake. *FEMS Microbiol. Ecol.* **80**, 548–555 (2012).
218. Jensen, S. & Siljanen, H. M. P. & Dörsch, P. Activity and abundance of methanotrophic bacteria in a northern mountainous gradient of wetlands. *Environ. Microbiol. Rep.* **15**, 206–215 (2023).
219. Cui, H. et al. Comparative analyses of methanogenic and methanotrophic communities between two different water regimes in controlled wetlands on the Qinghai-Tibetan Plateau, China. *Curr. Microbiol.* **75**, 484–491 (2018).
220. La, W. et al. Sulfate concentrations affect sulfate reduction pathways and methane consumption in coastal wetlands. *Water Res.* **217**, 118441 (2022).
221. Haroon, M. F. et al. Anaerobic oxidation of methane coupled to nitrate reduction in a novel archaeal lineage. *Nature* **500**, 567–570 (2013).
222. Hu, B.-L. et al. Evidence for nitrite-dependent anaerobic methane oxidation as a previously overlooked microbial methane sink in wetlands. *Proc. Natl Acad. Sci.* **111**, 4495–4500 (2014).
223. Ettwig, K. F. et al. Archaea catalyze iron-dependent anaerobic oxidation of methane. *Proc. Natl Acad. Sci.* **113**, 12792–12796 (2016).
224. Valenzuela, E. I. et al. Anaerobic methane oxidation driven by microbial reduction of natural organic matter in a tropical wetland. *Appl. Environ. Microbiol.* **83**, e00645–00617 (2017).
225. Lu, Y.-Z. et al. Cr (VI) reduction coupled with anaerobic oxidation of methane in a laboratory reactor. *Water Res.* **102**, 445–452 (2016).
226. Shi, L.-D. et al. Coupled anaerobic methane oxidation and reductive arsenic mobilization in wetland soils. *Nat. Geosci.* **13**, 799–805 (2020).
227. Zhou, Y. et al. Anaerobic methane oxidation coupled to arsenate reduction in paddy soils: Insights from laboratory and field studies. *Chemosphere* **311**, 137055 (2023).
228. Hefting, M. M., van den Heuvel, R. N. & Verhoeven, J. T. A. Wetlands in agricultural landscapes for nitrogen attenuation and biodiversity enhancement: opportunities and limitations. *Ecol. Eng.* **56**, 5–13 (2013).
229. Wang, J. et al. Nitrogen input promotes denitrifying methanotrophs' abundance and contribution to methane emission reduction in coastal wetland and paddy soil. *Environ. Pollut.* **302**, 119090 (2022).
230. Su, G. *Anaerobic oxidation of methane in lake environments: rates, pathways, environmental controls and microorganisms*. Diss. University_of_Basel (2020).
231. Chen, F. et al. Effects of periodic drying-wetting on microbial dynamics and activity of nitrite/nitrate-dependent anaerobic methane oxidizers in intertidal wetland sediments. *Water Res.* **229**, 119436 (2023).
232. Evans, P. N. et al. An evolving view of methane metabolism in the Archaea. *Nat. Rev. Microbiol.* **17**, 219–232 (2019).
233. L. Bräuer, S., Basilliko, N., Mp Siljanen, H. & H. Zinder, S. Methanogenic archaea in peatlands. *FEMS Microbiol. Lett.* **367**, fnaa172 (2020).

234. Antony, C. P., Murrell, J. C. & Shouche, Y. S. Molecular diversity of methanogens and identification of *Methanlobus* sp. as active methylotrophic Archaea in Lonar crater lake sediments. *FEMS Microbiol. Ecol.* **81**, 43–51 (2012).
235. Liu, P., Klose, M. & Conrad, R. Temperature-dependent network modules of soil methanogenic bacterial and archaeal communities. *Front. Microbiol.* **10**, 496 (2019).
236. Villa, J. A. Functional representation of biological components in methane-cycling processes in wetlands improves modeling predictions. *J. Geophys. Res.* **125**, e2020JG005794 (2020).
237. Conrad, R. Importance of hydrogenotrophic, acetoclastic and methylotrophic methanogenesis for methane production in terrestrial, aquatic and other anoxic environments: a mini review. *Pedosphere* **30**, 25–39 (2020).
238. Rey-Sanchez, C. et al. The ratio of methanogens to methanotrophs and water-level dynamics drive methane transfer velocity in a temperate kettle-hole peat bog. *Biogeosciences* **16**, 3207–3231 (2019).
239. Maietta, C. E., Hondula, K. L., Jones, C. N. & Palmer, M. A. Hydrological conditions influence soil and methane-cycling microbial populations in seasonally saturated wetlands. *Front. Environ. Sci.* **8**, 593942 (2020).
240. Tian, J. et al. Response of archaeal communities to water regimes under simulated warming and drought conditions in Tibetan Plateau wetlands. *J. Soils Sediment.* **15**, 179–188 (2015).
241. Yang, G. et al. Peatland degradation reduces methanogens and methane emissions from surface to deep soils. *Ecol. Indic.* **106**, 105488 (2019).
242. Wang, H. et al. Molecular mechanisms of water table lowering and nitrogen deposition in affecting greenhouse gas emissions from a Tibetan alpine wetland. *Glob. Change Biol.* **23**, 815–829 (2017).
243. Zhang, W. et al. Soil water content, carbon, and nitrogen determine the abundances of methanogens, methanotrophs, and methane emission in the Zoige alpine wetland. *J. Soils Sediments* **22**, 470–481 (2022).
244. Zhang, S. et al. Shifts of soil archaeal nitrification and methanogenesis with elevation in water level fluctuation zone of the three Gorges Reservoir, China. *J. Environ. Manag.* **339**, 117871 (2023).
245. Chen, Y., Qiu, K., Zhong, Z. & Zhou, T. Influence of environmental factors on the variability of archaeal communities in a karst wetland. *Front. Microbiol.* **12**, 675665 (2021).
246. Ye, F. et al. Shifts of archaeal community structure in soil along an elevation gradient in a reservoir water level fluctuation zone. *J. Soils Sediment.* **16**, 2728–2739 (2016).
247. Cui, H. et al. Response of methanogenic community and their activity to temperature rise in alpine swamp meadow at different water level of the permafrost wetland on Qinghai-Tibet Plateau. *Front. Microbiol.* **14**, 1181658 (2023).
248. Høj, L., Rusten, M., Haugen, L. E., Olsen, R. A. & Torsvik, V. L. Effects of water regime on archaeal community composition in Arctic soils. *Environ. Microbiol.* **8**, 984–996 (2006).
249. Conrad, R. et al. Response of the methanogenic microbial communities in a mazonian oxbow lake sediments to desiccation stress. *Environ. Microbiol.* **16**, 1682–1694 (2014).
250. Liu, F., Zhang, Y., Liang, H. & Gao, D. Resilience of methane cycle and microbial functional genes to drought and flood in an alkaline wetland: A metagenomic analysis. *Chemosphere* **265**, 129034 (2021).
251. Yuan, Y., Conrad, R. & Lu, Y. Transcriptional response of methanogen *mcrA* genes to oxygen exposure of rice field soil. *Environ. Microbiol. Rep.* **3**, 320–328 (2011).
252. Wang, H., Ho, M., Flanagan, N. & Richardson, C. J. The effects of hydrological management on methane emissions from southeastern shrub bogs of the USA. *Wetlands* **41**, 87 (2021).
253. Kitson, E. & Bell, N. G. A. The response of microbial communities to peatland drainage and rewetting. A review. *Front. Microbiol.* **11**, 582812 (2020).
254. Estop-Aragonés, C. & Blodau, C. Effects of experimental drying intensity and duration on respiration and methane production recovery in fen peat incubations. *Soil Biol. Biochem.* **47**, 1–9 (2012).
255. Kalyuzhnaya, M. G., Gomez, O. A. & Murrell, J. C. The methane-oxidizing bacteria (methanotrophs). Taxonomy, Genomics and Ecophysiology of Hydrocarbon-Degrading Microbes. *Handbook of Hydrocarbon and Lipid Microbiology* (eds McGenity, T. et al.) 245–278 (Springer, Cham., 2019).
256. Dedysh, S. N. & Knief, C. Diversity and phylogeny of described aerobic methanotrophs. *Methane Biocatalysis: Paving the Way to Sustainability* (Kalyuzhnaya, M. & Xing, X. H.) 17–42 (Springer, Cham., 2018).
257. Emsens, W.-J. et al. Recovery of fen peatland microbiomes and predicted functional profiles after rewetting. *ISME J.* **14**, 1701–1712 (2020).
258. Ho, A., Van den Brink, E., Reim, A., Krause, S. M. B. & Bodelier, P. L. E. Recurrence and frequency of disturbance have cumulative effect on methanotrophic activity, abundance, and community structure. *Front. Microbiol.* **6**, 1493 (2016).
259. Wang, H. et al. Linking Transcriptional Dynamics of Peat Microbiomes to Methane Fluxes during a Summer Drought in Two Rewetted Fens. *Environ. Sci. Technol.* **57**, 5089–5101 (2023).
260. Kaupper, T. et al. Recovery of methanotrophic activity is not reflected in the methane-driven interaction network after peat mining. *Appl. Environ. Microbiol.* **87**, e02355–02320 (2021).
261. Kaupper, T. et al. When the going gets tough: emergence of a complex methane-driven interaction network during recovery from desiccation-rewetting. *Soil Biol. Biochem.* **153**, 108109 (2021).
262. Larmola, T. et al. The role of Sphagnum mosses in the methane cycling of a boreal mire. *Ecology* **91**, 2356–2365 (2010).
263. Kip, N. et al. Global prevalence of methane oxidation by symbiotic bacteria in peat-moss ecosystems. *Nat. Geosci.* **3**, 617–621 (2010).
264. Knittel, K. & Boetius, A. Anaerobic oxidation of methane: progress with an unknown process. *Annu. Rev. Microbiol.* **63**, 311–334 (2009).
265. Michaelis, W. et al. Microbial reefs in the Black Sea fueled by anaerobic oxidation of methane. *Science* **297**, 1013–1015 (2002).
266. Orphan, V. J. et al. Comparative analysis of methane-oxidizing archaea and sulfate-reducing bacteria in anoxic marine sediments. *Appl. Environ. Microbiol.* **67**, 1922–1934 (2001).
267. Orphan, V. J., House, C. H., Hinrichs, K.-U., McKeegan, K. D. & DeLong, E. F. Multiple archaeal groups mediate methane oxidation in anoxic cold seep sediments. *Proc. Natl Acad. Sci.* **99**, 7663–7668 (2002).
268. Schreiber, L., Holler, T., Knittel, K., Meyerdierks, A. & Amann, R. Identification of the dominant sulfate-reducing bacterial partner of an anaerobic methanotrophs of the ANME-2 clade. *Environ. Microbiol.* **12**, 2327–2340 (2010).
269. Hinrichs, K. U. & Boetius, A. The anaerobic oxidation of methane: new insights in microbial ecology and biogeochemistry. In *Ocean Margin Systems* (eds Wefer, G.) 457–477 (Springer, Berlin, Heidelberg, 2002).
270. Krause, S. J. E. & Treude, T. Deciphering cryptic methane cycling: Coupling of methylotrophic methanogenesis and anaerobic oxidation of methane in hypersaline coastal wetland sediment. *Geochimica et Cosmochimica Acta* **302**, 160–174 (2021).
271. Cai, C. et al. A methanotrophic archaeon couples anaerobic oxidation of methane to Fe (III) reduction. *ISME J.* **12**, 1929–1939 (2018).

272. Leu, A. O. et al. Anaerobic methane oxidation coupled to manganese reduction by members of the Methanoperedenaceae. *ISME J.* **14**, 1030–1041 (2020).
273. Wang, Y. et al. Nitrite-dependent anaerobic methane oxidizing bacteria along the water level fluctuation zone of the Three Gorges Reservoir. *Appl. Microbiol. Biotechnol.* **100**, 1977–1986 (2016).
274. Luesken, F. A. et al. Effect of oxygen on the anaerobic methanotroph ‘*Candidatus Methyloirabilis oxyfera*’: kinetic and transcriptional analysis. *Environ. Microbiol.* **14**, 1024–1034 (2012).
275. Liu, Y. et al. Anaerobic methane-oxidizing bacterial communities in sediments of a drinking reservoir, Beijing, China. *Ann. Microbiol.* **70**, 1–13 (2020).
276. Zhang, M. et al. Molecular and stable isotopic evidence for the occurrence of nitrite-dependent anaerobic methane-oxidizing bacteria in the mangrove sediment of Zhangjiang Estuary, China. *Appl. Microbiol. Biotechnol.* **102**, 2441–2454 (2018).
277. Zhong, Q. et al. Structure and distribution of nitrite-dependent anaerobic methane oxidation bacteria vary with water tables in Zoige peatlands. *FEMS Microbiol. Ecol.* **96**, faa039 (2020).
278. Zhu, G. et al. Denitrifying anaerobic methane oxidizing in global upland soil: sporadic and non-continuous distribution with low influence. *Soil Biol. Biochem.* **119**, 90–100 (2018).
279. Askaer, L. et al. Soil heterogeneity effects on O₂ distribution and CH₄ emissions from wetlands: In situ and mesocosm studies with planar O₂ optodes and membrane inlet mass spectrometry. *Soil Biol. Biochem.* **42**, 2254–2265 (2010).
280. Bastviken, D. et al. The importance of plants for methane emission at the ecosystem scale. *Aquat. Bot.* **184**, 103596 (2023).
281. Regmi, T., Shah, D. N., Doody, T. M., Cuddy, S. M. & Shah, R. D. T. Hydrological alteration induced changes on macrophyte community composition in sub-tropical floodplain wetlands of Nepal. *Aquat. Bot.* **173**, 103413 (2021).
282. Vroom, R. J. E., van den Berg, M., Pangala, S. R., van der Scheer, O. E. & Sorrell, B. K. Physiological processes affecting methane transport by wetland vegetation—a review. *Aquat. Bot.* **182**, 103547 (2022).
283. Whiting, G. J. & Chanton, J. P. Plant-dependent CH₄ emission in a subarctic Canadian fen. *Glob. Biogeochem. Cycles* **6**, 225–231 (1992).
284. Jeffrey, L. C. et al. Wetland methane emissions dominated by plant-mediated fluxes: Contrasting emissions pathways and seasons within a shallow freshwater subtropical wetland. *Limnol. Oceanogr.* **64**, 1895–1912 (2019).
285. Barba, J. et al. Methane emissions from tree stems: a new frontier in the global carbon cycle. *N. Phytologist* **222**, 18–28 (2019).
286. Jeffrey, L. C. et al. Large methane emissions from tree stems complicate the wetland methane budget. *J. Geophys. Res.* **128**, e2023JG007679 (2023).
287. Pangala, S. R. et al. Large emissions from floodplain trees close the Amazon methane budget. *Nature* **552**, 230–234 (2017).
288. Gauci, V. et al. Non-flooded riparian Amazon trees are a regionally significant methane source. *Philos. Trans. R. Soc. A* **380**, 20200446 (2022).
289. Jeffrey, L. C., Maher, D. T., Tait, D. R., Euler, S. & Johnston, S. G. Tree stem methane emissions from subtropical lowland forest (*Melaleuca quinquenervia*) regulated by local and seasonal hydrology. *Biogeochemistry* **151**, 273–290 (2020).
290. Wik, M., Varner, R. K., Anthony, K. W., MacIntyre, S. & Bastviken, D. Climate-sensitive northern lakes and ponds are critical components of methane release. *Nat. Geosci.* **9**, 99–105 (2016).
291. Yang, G. et al. Characteristics of methane emissions from alpine thermokarst lakes on the Tibetan Plateau. *Nat. Commun.* **14**, 3121 (2023).
292. DelSontro, T. et al. Spatial heterogeneity of methane ebullition in a large tropical reservoir. *Environ. Sci. Technol.* **45**, 9866–9873 (2011).
293. Harrison, J. A., Deemer, B. R., Birchfield, M. K. & O’Malley, M. T. Reservoir water-level drawdowns accelerate and amplify methane emission. *Environ. Sci. Technol.* **51**, 1267–1277 (2017).
294. Aben, R. C. H. et al. Cross continental increase in methane ebullition under climate change. *Nat. Commun.* **8**, 1682 (2017).
295. Varadharajan, C. & Hemond, H. F. Time-series analysis of high-resolution ebullition fluxes from a stratified, freshwater lake. *J. Geophys. Res.* **117**, G02004 (2012).
296. Maeck, A., Hofmann, H. & Lorke, A. Pumping methane out of aquatic sediments—ebullition forcing mechanisms in an impounded river. *Biogeosciences* **11**, 2925–2938 (2014).
297. Windsor, J., Moore, T. R. & Roulet, N. T. Episodic fluxes of methane from subarctic fens. *Can. J. Soil Sci.* **72**, 441–452 (1992).
298. Treat, C. C., Bubier, J. L., Varner, R. K. & Crill, P. M. Timescale dependence of environmental and plant-mediated controls on CH₄ flux in a temperate fen. *J. Geophys. Res.* **112**, G01014 (2007).
299. Moore, T., Roulet, N. & Knowles, R. Spatial and temporal variations of methane flux from subarctic/northern boreal fens. *Glob. Biogeochemical Cycles* **4**, 29–46 (1990).
300. Walter, B. P., Heimann, M., Shannon, R. D. & White, J. R. A process-based model to derive methane emissions from natural wetlands. *Geophys. Res. Lett.* **23**, 3731–3734 (1996).
301. Mohammadimanesh, F., Salehi, B., Mahdianpari, M., Brisco, B. & Motagh, M. Wetland water level monitoring using interferometric synthetic aperture radar (InSAR): A review. *Can. J. Remote Sens.* **44**, 247–262 (2018).
302. Lee, H., Yuan, T., Yu, H. & Jung, H. C. Interferometric SAR for wetland hydrology: An overview of methods, challenges, and trends. *IEEE Geosci. Remote Sens. Mag.* **8**, 120–135 (2020).
303. Gontijo, J. B. et al. Not just a methane source: Amazonian floodplain sediments harbour a high diversity of methanotrophs with different metabolic capabilities. *Mol. Ecol.* **30**, 2560–2572 (2021).
304. Täumer, J. et al. Linking transcriptional dynamics of CH₄-cycling grassland soil microbiomes to seasonal gas fluxes. *ISME J.* **16**, 1788–1797 (2022).
305. Lyautey, E., Billard, E., Tissot, N., Jacquet, S. & Domaizon, I. Seasonal dynamics of abundance, structure, and diversity of methanogens and methanotrophs in lake sediments. *Microb. Ecol.* **82**, 559–571 (2021).
306. Wang, L. et al. The synergism between methanogens and methanotrophs and the nature of their contributions to the seasonal variation of methane fluxes in a wetland: the case of Dajiuhu subalpine Peatland. *Adv. Atmos. Sci.* **39**, 1375–1385 (2022).
307. Helfter, C. et al. Phenology is the dominant control of methane emissions in a tropical non-forested wetland. *Nat. Commun.* **13**, 133 (2022).
308. Chen, H., Xu, X., Fang, C., Li, B. & Nie, M. Differences in the temperature dependence of wetland CO₂ and CH₄ emissions vary with water table depth. *Nat. Clim. Change* **11**, 766–771 (2021).

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Author contributions

Shihao Cui, Pengfei Liu, and Shubiao Wu designed the framework of the paper. Shihao Cui did the literature review, data extraction, analysis, and

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Competing interests

The authors declare no competing interests.

Additional information

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