



Impact of Water Table on Methane Emission Dynamics in Terrestrial Wetlands and Implications on Strategies for Wetland Management and Restoration

Tao Yang¹ · Qiang He² · Jing Jiang³ · Lianxi Sheng¹ · Haibo Jiang¹ · Chunguang He¹ 

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Abstract

Methane is a potent greenhouse gas. Wetlands are considered as significant sources of methane emission, prompting the need to understand determinants of methane flux in these critical ecosystems. The importance of the water table in methane emission has been noted in terrestrial wetlands; however, the role of the water table in methane emission remains to be clarified in order for the development of strategies to mitigate methane emission from wetland ecosystems. This review examines the current literature on factors influencing methane emission in terrestrial wetlands. The water table was illustrated as an overriding factor that controls both methane generation and consumption. The contribution of other main factors, including substrate characteristics, wetland plants and temperature, to methane emission was also discussed. Building upon the growing understanding of processes underlying methane emission, strategies centered around the control of water table was proposed to minimize methane emission in wetland management and restoration efforts to maximize the ecological value of wetlands.

Keywords Methane · Water Table · Terrestrial Wetlands · Wetland Restoration

Introduction

Wetlands have irreplaceable and distinctive functions such as hydrological regulation, water purification, biodiversity preservation and carbon sequestration. Among all terrestrial ecosystems, wetlands have the highest carbon density because of the slow rate of carbon mineralization in anoxic wetland soils (Mitsch et al. 2013). Wetlands store vast quantities of carbon (approximately 15×10^{14} kg) in a relatively small land area (estimated between 2 and 6% of global land area) (Schlesinger 1990; Schlesinger and Bernhardt 2013).

Compared to carbon pools in forests, organic carbon stored in wetlands remains stable for a longer period of time due to anaerobic conditions prevalent in wetlands. Therefore, wetland ecosystems can serve as stable carbon sinks.

However, the anaerobic condition in wetlands can also facilitate methane production. Methanogens produce CH_4 efficiently under strictly anaerobic conditions ubiquitous in wetlands. It is currently agreed that global wetland ecosystems emit substantial amounts of CH_4 that accounts for the largest natural source of atmospheric CH_4 (Bruhwiler et al. 2014; Dean et al. 2018), up to 70% of all natural emissions at an estimated rate of approximately 1.45×10^{11} kg $\text{CH}_4\text{-C yr}^{-1}$ (Bridgham et al. 2013; Turetsky et al. 2014).

Methane is an important greenhouse gas (GHG) due to its high global warming potential (Neubauer and Megonigal 2015). The concentration of methane in the atmosphere has increased from 722 ppb during the pre-industrial period to 1909.2 ppb in March 2022 (https://www.esrl.noaa.gov/gmd/ccgg/trends_ch4/) which has more than doubled. Recent calculations indicate that atmospheric methane accumulation is responsible for approximately 16% to 20% of Earth's warming induced by long-lived greenhouse gases (Yvon-Durocher et al. 2014; WMO 2021). As the concentration of methane in the atmosphere continues to increase rapidly, it can be

✉ Qiang He
qianghe@utk.edu

✉ Chunguang He
he-cg@nenu.edu.cn

¹ State Environmental Protection Key Laboratory of Wetland Ecology and Vegetation Restoration, Northeast Normal University, Changchun, Jilin, China

² Department of Civil and Environmental Engineering, The University of Tennessee, Knoxville, TN, USA

³ Jilin Shize Environmental Protection Group Co., Ltd., Changchun, Jilin, China

expected that methane may have an increasing impact on global climate change in the future, highlighting the potential benefits of reducing methane emission from wetland ecosystems, which could offset some of the anthropogenic origins warming effects.

It should be noted that, owing to the dual effects of global climate change and human activities, the hydrological structure of wetland ecosystems has suffered severe interference and the water table has changed which further leads to changes in the function of wetlands in the natural carbon and nitrogen cycle which causes significant changes in fluxes of greenhouse gases such as CO₂, CH₄, and N₂O (Parn et al., 2018; Lin et al. 2022). Aiming to reverse the trend of wetland losses to protect biodiversity and ecological health, wetland management and restoration projects have been carried out around the world. In the context of global climate change, the impact of wetland restoration projects on GHGs emissions should be taken into account especially for methane as most restoration projects involve raising the water table, which may lead to more methane emissions (Vanselow-Algan et al. 2015), however this is not typically done. As a large number of studies show that methane emissions in wetlands are strongly regulated by the water table, it is generally accepted that hydrological management and restoration is a primary step to restore the ecological services of wetlands. However, there remains uncertainties in the relationship between wetland methane emission and the water table, which makes it difficult to set optimal water table in wetland management and restoration efforts to mitigate methane emission.

This review summarizes the variability and uncertainty of methane emission in terrestrial freshwater wetlands in relation to hydrological responses, laying the foundation for the development of potential strategies to mitigate methane emission for wetland management and restoration efforts, which will be informative for engineers, ecologists, scientists, researchers, as well as policy makers involved in the planning, design, and implementation of wetland management and restoration projects.

Main Factors Affecting Methane Emissions from Terrestrial Wetlands

The magnitude of CH₄ flux is determined by the following three processes: CH₄ production, consumption, and transport (Dunfield et al. 1993; Borrel et al. 2011). In wetlands, CH₄ is emitted to the atmosphere through different pathways, including molecular diffusion, ebullition, and passage through aerenchyma of vascular plants (Philipp et al. 2017), when the balance between methane production and consumption is positive (Whalen 2005; Bao et al. 2021; Minick et al. 2021; Zhang et al. 2022c).

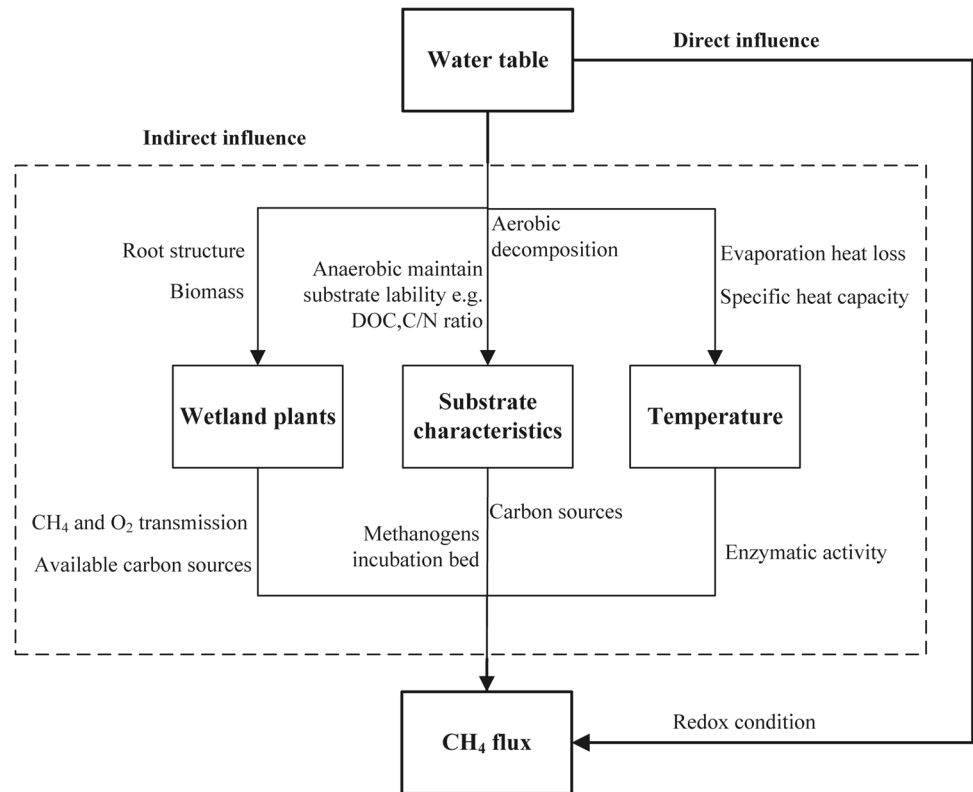
Previous studies on the dynamics of methane emission from wetlands have observed that factors such as depth of water table (Abdalla et al. 2016; Evans et al. 2021; Morin et al. 2022; Mwagona et al. 2021), temperature (Yvon-Durocher et al. 2014; Zhu et al. 2021), substrate characteristics (Nielsen et al. 2017; Oikawa et al. 2017; Xu et al. 2021), wetland plants (Jitka et al. 2017; McInerney and Helton 2016; Derby et al. 2021), and microorganisms present in wetlands (Dedysh et al. 1998; Hao et al. 2020; Xie et al. 2020) are the main determinants of methane emission from terrestrial freshwater wetlands. Among these the water table are overriding factor (Evans et al. 2021) that not only directly controls methane generation and consumption, but also profoundly affects methane flux mediately by changing wetland soil physicochemical properties (Ellis et al. 2009), forming wetland plant community (Yu and Ehrenfeld 2010), and altering temperature sensitivity of methane emission (Chen et al. 2021). Therefore, when discussing the effect of water table on methane emissions, here we also discuss other factors including wetland plants, substrate characteristics, and temperature that are influenced by water table (Fig. 1).

Water Table

Water table directly controls the redox condition in the wetland substrate (Yang et al. 2013; Gulley et al. 2020), which in turn controls methanogenesis (Olefeldt et al. 2017). Previous studies found that net methane fluxes from wetlands show a clear relationship to water level. In most cases, higher water tables increase the emission of CH₄ while lower water tables accelerate carbon mineralization and decrease CH₄ flux (Table 1).

A microcosms simulate study of CH₄ fluxes under manipulated water table treatments in the Sanjiang Plain freshwater marshes showed that the average CH₄ emission was 7.43 mg m⁻² h⁻¹, and higher water table condition (+2 to +14 cm) emits 75% more CH₄ than at low water table (-11 to 0 cm), however the situation is reversed for CO₂ and N₂O emissions which were 120% and 60% lower respectively (Yang et al. 2013). Field greenhouse gas flux monitoring in the Dajihu wetland Shennongjia Forest District, Hubei Province, China shows that methane fluxes were significant different ($p < 0.05$) between flooded and low water tables. When the peat was flooded (high water table), the methane flux was high, ranging from 30 to 130 nmol/m²/s ($n = 230$); when the water table was below the surface, the methane flux was much lower, with maxima < 5 nmol/m²/s ($n = 222$), even negative when the water table dropped below 10 cm, the methane flux reached a minimum of -2.5 nmol/m²/s (Zhang et al. 2022d). CH₄ emissions at different water table from artificial restoration wetlands in Hangzhou Bay, northern Zhejiang Province, China shows that high water table condition (10-30 cm above the surface) emit much

Fig. 1 Direct and indirect effects of water table on CH₄ fluxes in terrestrial wetlands



more CH₄ (2 ~ 17 mg CH₄ m⁻² h⁻¹) than low water table which just close to the surface (0 ~ 2 mg CH₄ m⁻² h⁻¹) (Xiong et al. 2022). Field researches conducted in Wicken Fen National Nature Reserve compared the CH₄ emissions of an undisturbed nature fen named Wicken Sedge Fen and a former cropland restored fen named Baker's Fen. The Wicken Sedge Fen which is the oldest nature reserve in the UK has never been agriculturally drained, it emit 10.6 g C m⁻² y⁻¹ CH₄, the water table (positive value means above the surface, negative value means below the surface) range recorded by the researchers are -40 ~ -80 cm in summer and 5 ~ -10 cm in winter/spring; The Baker's Fen was drained in the mid-nineteenth century for agriculture, resulting in extensive peat loss and subsidence, it emit 0.15 g C m⁻² y⁻¹ CH₄, the water table range recorded by the researchers are -70 cm in summer and 0 ~ -40 cm in winter/spring (Peacock et al. 2019). A microcosm simulate study on a coastal wetland in Kenli County, Shandong Province, China shows a results of linear relationship between water table and methane emissions. Soil CH₄ emission at -40, -30, -20, -10 and 0 cm water table (negative value means below the surface) are 0.07, 0.1, 0.12, 0.19 and 0.22 nmol CH₄ m⁻² s⁻¹ respectively (Zhao et al. 2020). CH₄ fluxes of drained and rewetted temperate bog in Himmelmoor, northwestern Germany measured by eddy covariance tower are 7.2 ~ 12.1 and 13.3 ~ 18.3 g m⁻² a⁻¹ CO₂ Eq. (34 to convert FCH₄ into CO₂ eq. release), which indicates higher water table leads to

increased methane emissions (Holl et al. 2020). Microcosm simulate study on CH₄ emissions from the gley marsh and peatland in Sanjiang Plain northeast China shows methane emissions under steady water table conditions were significantly lower in the drawdown treatment in both peatland and gley marshes but did not differ between the 10 cm and 0 cm treatments. The CH₄ emissions at -10, 0 and 10 cm water table (positive value means above the surface, negative value means below the surface) from gley marsh and peatland are 9.1, 17.4 ± 3.0, 15.0 ± 2.3 and 11.3 ± 1.3, 23.8 ± 2.7, 23.0 ± 3.4 mg C m⁻² h⁻¹ (Zhu et al. 2018). This is probably due to lower water tables and thicker aerated top layer would increasing methanotrophs in aerobic soil layers.

However, some of the studies have observed weak or even the opposite relationships between CH₄ emission and the water table (Zhao et al. 2020; Luta et al. 2021; Mwagona et al. 2021; Wang et al. 2021). For example, microcosm study on a tropical peatland in Sarawak, Malaysia shows averaged soil CH₄ emission under a low water table were higher compared with that of high water table. When water table close to the surface, it emit 0.3–0.5 t CH₄ ha⁻¹ yr⁻¹; when water table is 90 cm below the surface, it emit 1.9–2.0 t CH₄ ha⁻¹ yr⁻¹ (Luta et al. 2021). The high spatial and temporal variability in methane flux afield sites could contribute to the inconsistencies, which may be further complicated by the interactions between the water table and other factors, including temperature, plant community and substrate characteristics. Wetland CH₄ flux

Table 1 Some recently published data on methane fluxes from wetlands at different water tables

Study area	Coordinate	Wetland types	Vegetation Types	Field or Microcosm	Monitoring methods
Sacramento-San Joaquin Delta of California	38.0366°N, 121.7540°W	Drained peatland pasture	<i>Hordeum murinum</i> L., <i>Lepidium latifolium</i> L.	Field	EC (eddy covariance)
	38.1047°N, 121.6433°W	Conventional drained agricultural(Pasture)	<i>Zea mays</i>	Field	
	38.10875°N, 121.6530°W	Conventional drained agricultural(Corn)	<i>Oryza sativa</i>	Field	
	38.0498°N, 121.7650°W	Agricultural wetland(Rice)	<i>Schoenoplectus acutus</i>	Field	
Sherman Island, Northern California	38.1074°N, 121.6469°W	Restored wetland	<i>Typha</i> spp. <i>Schoenoplectus acutus</i>	Field	Chamber(trace gas pool dilution technique)
	38.04°N, 121.75°W	Drained temperate peatland pasture	pepperweed (<i>Lepidium latifolium</i>) and the annual mouse barley grass (<i>Hordeum murinum</i>)	Field	Static chamber
North Carolina, USA	N/A	Pocosins, freshwater evergreen shrub bogs	trees, including loblolly bay [<i>Gordonia lasianthus</i> (L.) Ellis]	Field	Static chamber
		natural	shrubs dominate, including inkberry [<i>Ilex glabra</i> (L.) A. Gray]	Field	
		restored (rewetting)	western brackenfern [<i>Pteridium aquilinum</i> (L.) Kuhn] with scattered woody shrubs like winged sumac (<i>Rhus copallinum</i> L.)	Field	
Sanjiang Plain, Northeastern China	132°00'22"-132°24'46"E, 46°39'45"-46°48'24"N	drained	Shrubs	Microcosm	Gas chromatograph
		Surface soil (0–5 cm) of flooding period in some natural and restored sites		Microcosm	Gas chromatograph
Hauraki Plains of the North Island/Te Ika a Māui, New Zealand	37°55.5'S, 175°22.2'E	Riparian freshwater marsh	<i>C. angustifolia</i> <i>P. australis</i>	Microcosm	Gas chromatograph
		Ombrotrophic, raised bog (peat depths up to 14 m)	<i>C. pseudo-curaica</i>	Field	EC
Cors Erdreintog, UK	53.31, -4.30	Fen	The jointed wire rush, <i>Empodisma robustum</i>	Field	EC
	52.49, -4.03	Raised bog	Phragmites australis, Cladium mariscus, Molinia caerulea; brown moss hollows, Sphagnum hummocks and Juncus species	Field	Static chambers
The Dajuhu wetland Shennongjia Forest District, Hubei Province, China	31°25' ~ 31°32' N, 109°58' ~ 110°08' E	Sphagnum palustre peatland	<i>Sphagnum palustre</i> , <i>Carex argyi</i> , <i>Juncus effusus</i> , <i>Euphorbia esula</i> , and <i>Sanguisorba officinalis</i>	Field	Chamber coupled with ultrapotable Greenhouse Gas Analyzer

Table 1 (continued)

Study area	Coordinate	Wetland types	Vegetation Types	Field or Microcosm	Monitoring methods
Hangzhou Bay, northern Zhejiang Province, China	121°09'132"E, 30°19'74"N	Natural wetlands	<i>Scirpus maritimus</i> , <i>Phragmites australis</i> , <i>Spartina alterniflora</i>	Field	Field
Naigombwa wetland, Iganga District, Southeastern Uganda	N/A	Reclamation-restored wetlands Natural	<i>Phragmites australis</i>	Field	Field
Sesang, Saratok, Sarawak, Malaysia	N/A	Rice paddy Tropical peatland	<i>Papyrus</i> <i>Typha</i> <i>Phragmites</i> Rice Moris pineapple	Field Microcosm	Field Microcosm
Biebrza National Park wetlands, northeastern Poland	53°35'30.8" N, 22°53'32.4" E, 109 m a.s.l	Temperate mire	Reeds, sedges and rushes typical of Biebrza wetlands	Field	Field
Kenli County, Shandong Province, China	37°45'50" N, 118°59'24" E	Coastal wetland	<i>Suaeda salsa</i> , <i>Phragmites australis</i> , <i>Tamarix chinensis</i> and <i>Imperata cylindrica</i>	Microcosm	Microcosm
Himmelmoor, northwestern Germany	53°44'23.3" N, 9°50'55.8" E	Temperate bog	Sphagnum spp. peat mosses	Field	Field
Wicken Sedge Fen, Wicken Fen National Nature Reserve	52.31°N, 0.28°E, area = 61 ha, 2–3 m above sea level	Fen, oldest nature reserve in the UK	Saw sedge <i>Cladium mariscus</i> , common reed <i>Phragmites australis</i>	Field	Field
Baker's Fen, Wicken Fen National Nature Reserve	52.30°N, 0.29°E, area = 56 ha, 0–1 m above sea level	Fen, former cropland	<i>Agrostis stolonifera</i> with <i>Arrhenatherum elatius</i>	Field	Field
Sanjiang Plain in northeast China	47°31' N, 133°22' E	Gley marsh Peatland	<i>Carex lasiocarpa</i> Ehrh	Microcosm	Microcosm
Ob' River floodplain, Khanty-Mansiysk city, Russia	61.08977°N, 69.45513°E	Riparian wetlands	Sedges or bare	Field	Field
Southwest of Fairbanks, Alaska, USA	64.82°N, 147.87 W	Rich fen	Marsh cinquefoil (<i>Potentilla palustris</i>), wheat sedge (<i>Carex altherodes</i>), water horsetail (<i>Equisetum fluviatile</i>)	Field	Field

Table 1 (continued)

Study area	*Water table(cm)	Methane Flux	Time Scale /Monitoring frequency	Soil pH	Reference
Sacramento-San Joaquin Delta of California	-65 (-92 ~ -37)	5.84 ± 1.51 to 11.4 ± 2.66(g C m ⁻² Y ⁻¹)	1 March 2012–1 March 2013	N/A	(Knox et al. 2015)
	-82 (-86 ~ -76)	N/A	9 May 2012–9 May 2013		
	9 (-30 ~ 17)	5.30 ± 0.80(g C m ⁻² Y ⁻¹)	1 March 2012–1 March 2013		
	26 (6 ~ 32)	53.0 ± 0.78(g C m ⁻² Y ⁻¹)	1 August 2012–1 August 2013		
	107 (102 ~ 109)	38.7 ± 1.10(g C m ⁻² Y ⁻¹)	1 March 2012–1 March 2013		
Sherman Island, Northern California	Maintained around 50 cm depth by continual pumping	180.3 ± 81.2 mg C m ⁻² d ⁻¹ in drainage ditches to -0.7 ± 1.2 mg C m ⁻² d ⁻¹ in the highest landform	October 2007, January 2008, March 2008, and May 2008	N/A	(Yang et al. 2017b)
	-25 ± 17 (-60 ~ -100)	0.128 mg CH ₄ m ⁻² h ⁻¹	Monthly from August 2011 to January 2013	3.3	(Wang et al. 2021)
North Carolina, USA	16 ± 7	0.024 mg CH ₄ m ⁻² h ⁻¹		4 ~ 5	
	-38 ± 10	0.018 mg CH ₄ m ⁻² h ⁻¹			
	-55 ± 14	0.015 mg CH ₄ m ⁻² h ⁻¹			
	-68 ± 8 (-150 ~ -50)	0.005 mg CH ₄ m ⁻² h ⁻¹		4 ~ 5	
	-70 ± 10 (-150 ~ -50)	0.005 mg CH ₄ m ⁻² h ⁻¹			
	-84 ± 11 (-150 ~ -50)	0.008 mg CH ₄ m ⁻² h ⁻¹			
	2 ~ 3 (3 months of lab incubation)	0.104 mg CH ₄ m ⁻² h ⁻¹	Soil cores in pocosin after 3 months of lab incubation under a saturated condition	N/A	
	5 ~ 20(1–5 days of lab incubation)	0.18 (-0.02 ~ 0.90)mg/m ² h in C	1–5 days of lab incubation	N/A	(Mwagoma et al. 2021)
		0.63(-4.20 ~ 18.96) mg/m ² h			
		0.42(-2.57 ~ 5.21) mg/m ² h			
Hauraki Plains of the North Island/ Te Ika a Māui, New Zealand	2012 summer	150 mg m ⁻² d ⁻¹	From 4 February 2012 to 4 August 2014	N/A	(Goodrich et al. 2015)
	2012 winter	40–60 mg m ⁻² d ⁻¹			
	2013/2014 summer	140–170 mg m ⁻² d ⁻¹			
Cors Erdreimog,UK	7	209 kgCH ₄ ha ⁻¹ yr ⁻¹	3 years, 12–16 times per year	5.9–6.6	(Evans et al. 2021)
	9	188 kgCH ₄ ha ⁻¹ yr ⁻¹			
	3	177 kgCH ₄ ha ⁻¹ yr ⁻¹			
	10	61 kgCH ₄ ha ⁻¹ yr ⁻¹			
	7	73 kgCH ₄ ha ⁻¹ yr ⁻¹			
	6	161 kgCH ₄ ha ⁻¹ yr ⁻¹			
	4	159 kgCH ₄ ha ⁻¹ yr ⁻¹			
	5	187 kgCH ₄ ha ⁻¹ yr ⁻¹			
	7	85 kgCH ₄ ha ⁻¹ yr ⁻¹			
	10	108 kgCH ₄ ha ⁻¹ yr ⁻¹			
	-1	235 kgCH ₄ ha ⁻¹ yr ⁻¹			
	4	135 kgCH ₄ ha ⁻¹ yr ⁻¹			
	5	133 kgCH ₄ ha ⁻¹ yr ⁻¹			
	0	140 kgCH ₄ ha ⁻¹ yr ⁻¹			
	5	99 kgCH ₄ ha ⁻¹ yr ⁻¹			
	16	59 kgCH ₄ ha ⁻¹ yr ⁻¹			

Table 1 (continued)

Study area	*Water table(cm)	Methane Flux	Time Scale /Monitoring frequency	Soil pH	Reference
The Dajiuhu wetland Shennongjia Forest District, Hubei Province, China	June (WTD = -2.4 cm, flooded), July (WTD = -1.6 cm, flooded), April (WTD = 0.5 cm), September (WTD = 11.0 cm), October (WTD = 16.8 cm)	flooded (negative WTD), the methane flux was high, ranging from 30 to 130 nmol/m ² /s (n = 230); water table below the peat surface, < 5 nmol/m ² /s (n = 222)	April, June, July, September and October in 2015	5.6–5.9	(Zhang et al. 2022d)
Hangzhou Bay, northern Zhejiang Province, China	N/A	-1.5 ± 2 mg CH ₄ m ⁻² h ⁻¹	From April to December 2014, once per month from April to December, with three replicates per collection	8.6–8.9	(Xiong et al. 2022)
Naigombwa wetland, Iganga District, Southeastern Uganda	0 cm 10–30 cm Surface water level (cm) 7.5–31.5 cm 7.1–29.1	0–2 mg CH ₄ m ⁻² h ⁻¹ 2–17 mg CH ₄ m ⁻² h ⁻¹ 15.6 ± 1.3 CH ₄ mg C m ⁻² h ⁻¹ 14.9 ± 1.2 mg C m ⁻² h ⁻¹ 12.1 ± 0.8 mg C m ⁻² h ⁻¹	February, March, April, August, September and October 2019, twice a month	5.98–6.87	(Were et al. 2021)
Sesang, Saratok, Sarawak, Malaysia	0.4–10.9	3.6 ± 0.4 mg C m ⁻² h ⁻¹	July, August, September and December 2015	N/A	(Luta et al. 2021)
Biebrza National Park wetlands, northeastern Poland	90 cm	0.3–0.5 tCH ₄ ha ⁻¹ yr ⁻¹ 1.9–2.0 tCH ₄ ha ⁻¹ yr ⁻¹ 27.4 mg CH ₄ m ⁻² d ⁻¹	6-years of continuous measurement, 2013–2018	5.0–6.7	(Fortuniak et al. 2021)
Kent County, Shandong Province, China	water table level - 14.1	0.07 nmol CH ₄ m ⁻² s ⁻¹ 0.1 nmol CH ₄ m ⁻² s ⁻¹ 0.12 nmol CH ₄ m ⁻² s ⁻¹ 0.19 nmol CH ₄ m ⁻² s ⁻¹ 0.22 nmol CH ₄ m ⁻² s ⁻¹	Once every 10 days from 23 April to 23 September 2018	7.78–8.18	(Zhao et al. 2020)
Himmelmoor, northwestern Germany	drained	7.2–12.1 g m ⁻² a ⁻¹ CO ₂ Eq. 34 to convert FCH ₄ into CO ₂ eq. release	1 June 2012 to 31 May 2013 and 1 June 2013 to 31 May 2014	N/A	(Holl et al. 2020)
Wicken Sedge Fen, Wicken Fen National Nature Reserve	rewetted WTD 40–80 in summer -5–10 in winter/spring	13.3–18.3 g m ⁻² a ⁻¹ CO ₂ eq 10.6 g C m ⁻² y ⁻¹	31 occasions at Sedge Fen, (every 2–3 weeks) in summer and (every 4–8 weeks) in winter	7.54	(Peacock et al. 2019)
Baker's Fen, Wicken Fen National Nature Reserve	WTD 70 in summer 0–40 in winter/spring	0.15 g C m ⁻² y ⁻¹	37 occasions at Baker's Fen, (every 2–3 weeks) in summer and (every 4–8 weeks) in winter	7.1	
Sanjiang Plain in northeast China	10 cm 0 cm -10 cm 10 cm 0 cm -10 cm	15.0 ± 2.3 mg C m ⁻² h ⁻¹ 17.4 ± 3.0 mg C m ⁻² h ⁻¹ 9.1 mg C m ⁻² h ⁻¹ 23.0 ± 3.4 mg C m ⁻² h ⁻¹ 23.8 ± 2.7 mg C m ⁻² h ⁻¹ 11.3 ± 1.3 mg C m ⁻² h ⁻¹	Every 10 d from late June to early October in 2013	5.25–5.87	(Zhu et al. 2018)
Ob' River floodplain, Khatyn-Mansysk city, Russia	water table level (negative values mean that water stands above the surface) - 250–100	0–17.5 mg C m ⁻² h ⁻¹	2015 to 2016	N/A	(Terentjeva et al. 2018)
Southwest of Fairbanks, Alaska, USA	Water table interval < -25 -25--10 0--10 > 0	0.7 ± 1.0 mg CH ₄ m ⁻² d ⁻¹ 3.3 ± 2.7 mg CH ₄ m ⁻² d ⁻¹ 7.6 ± 5.5 mg CH ₄ m ⁻² d ⁻¹ 12.5 ± 6.6 mg CH ₄ m ⁻² d ⁻¹	2005 to 2013, a total of 918 measurements of F _{CH4}	(surface water pH) 5.2–5.4	(Olefeldt et al. 2017)

responses to the water table differed across bogs, marshes, fens, and swamps. The estimated maximum CH₄ effluxes were the largest in fens ($85.9 \pm 2.3 \text{ mg mg m}^{-2} \text{ d}^{-1}$), followed by bogs ($75.0 \pm 3.5 \text{ mg m}^{-2} \text{ d}^{-1}$), marshes ($65.3 \pm 8.9 \text{ mg mg m}^{-2} \text{ d}^{-1}$), and the lowest in swamps ($16.2 \pm 1.2 \text{ mg mg m}^{-2} \text{ d}^{-1}$). The optimal water table position for CH₄ effluxes is below the ground surface in bogs ($-12.6 \pm 0.9 \text{ cm}$), close to the ground surface in marshes ($-6.9 \pm 11.1 \text{ cm}$) and fens ($1.0 \pm 4.3 \text{ cm}$), and above the ground surface in swamps ($17.8 \pm 2.7 \text{ cm}$) (Bao et al. 2021). This might be the result of heterogeneities in vegetation types and soil properties in different wetland types.

High water tables keep oxygen from contacting with wetland substrate, thereby maintaining anaerobic conditions. Conversely, lower water tables allow oxygen to more readily diffuse into the substrate, creating aerobic conditions in the substrate (Yang et al. 2013; Gulley et al. 2020). Methanogenesis in wetlands is limited to zones under anaerobic conditions, thereby controlled by the water table. It is therefore anticipated that CH₄ emission in wetlands is sensitive to the hydrologic regime. Lower water tables reduce CH₄ emissions by reducing anaerobic zones and suppressing methanogenesis, at the same time increasing CH₄ oxidation with greater availability of oxygen in the overlaying zones above the water table (Olefeldt et al. 2017). Indeed, many studies have observed that, when the water table drops considerably below the substrate surface, the wetlands may change from being a source to a sink for methane due to increased methane oxidation (Whalen 2005; White et al. 2008; Lai 2009). For example, studies focused on methane flux dynamics of the Great Dismal Swamp show the peat in this wetland system is a methane sink when the water table is below the surface of the peat during dry periods. In contrast, when the peat is saturated with water, it becomes an important methane source (Happell and Chanton 1993; Gutenberg et al. 2019).

The water table also has strong influences on vegetation traits (Blodau et al. 2004; Strakova et al. 2012), substrate characteristics (Hou et al. 2000a, 2000b; Williams et al. 2000; Von Arnold et al. 2005) and microbial activity in wetlands (Roslev and King 1996; Goodrich et al. 2015; Jerman et al. 2017), therefore playing a significant role in the regulation of methane emissions from terrestrial wetlands. Lowering water table by drainage has been reported to enhance soil aeration and increase soil temperatures, leading to reduced CH₄ emissions (Von Arnold et al. 2005).

Substrate Characteristics

Owing to the dual effects of climate change and human activities, the hydrological structure of wetland ecosystems has suffered severe interference and the water table has changed which further leads to changes of the redox conditions in wetland soils which further leads to altered SOC content and bioavailability in the substrate. SOC content and labile

fractions in the substrate determine the quality of wetland substrates and provides carbon source and electron donors for methanogenesis. It has been observed that poor substrate quality, such as the prevalence of recalcitrant organic materials, is more likely to lead to low methane emission rates in wetlands (Esterle and Ferm, 1994; Couwenberg et al. 2009; Yule and Gomez 2009). It is noted that the distribution of recalcitrant versus labile carbon fractions of the organic matter in wetland substrates is an important determinant of the rate and magnitude of methane production (Jerman et al., 2017; Lupascu et al. 2020). Furthermore, the presence and distribution of alternative electron acceptors, such as SO₄²⁻, Fe³⁺, Mn⁴⁺, NO₃⁻ and NO₂⁻, in wetland substrate, may strongly affect the production and consumption of methane via methanogenesis and anaerobic methane oxidation process (Noyce and Megonigal 2021; Rush et al. 2021). These alternative electron acceptors typically suppress methanogens and decrease methane production (Mwagona et al. 2021) while enable methane oxidation under anaerobic conditions (Dowrick et al. 2006).

Wetlands with higher water table are likely to breed more microbes and their activities increased the soil organic matter decomposition which led to higher concentrations of labile fractions (Wang et al. 2007; Gao et al. 2014a). Soil labile organic carbon refers to the part of organic carbon that was fast-moving, unstable, easily oxidized, and easily mineralizable. It occupies a relatively small proportion of the total soil carbon, but is of great significance to greenhouse gas emissions and soil carbon availability and quality (Huo et al. 2017; Xi et al. 2018). What's more, soil processes involving labile organic carbon may strongly control the carbon source/sink functions of wetland ecosystems (Jones and Willett 2006; Said-Pullicino et al. 2016).

Dissolved organic carbon (DOC) and soil microbial biomass carbon (MBC) are two important parts of soil labile organic carbon pool (Huo et al. 2017; Wickland et al. 2018). DOC is the organic carbon extracted using water or diluted salt solution and can pass through 0.45 μm filter, which is more important for biogeochemical processes and the ecosystem function. DOC originates from plant root exudation and decomposition of litter and soil organic matter (Kalbitz et al. 2000). Methanogens in wetlands predominantly use acetate (acetoclastic), H₂/CO₂ (hydrogenotrophic) and methylated substances as substrates for methane production (Bridgham et al. 2013), DOC or further fermentation of DOC from root exudates or plant residues can directly provided these methanogens required substrates (Ye and Horwath 2017). Therefore the positive relationship between DOC concentrations and methane emissions is likely due to DOC providing highly bioavailable labile substrates for methanogens. MBC presents the carbon in the living soil microorganism whose volume is $< 5 \times 10^3 \mu\text{m}^3$, which plays a fundamental role in soil

organic carbon dynamics and is often regarded as a useful indicator for the changes in soil carbon stabilization and nutrient dynamics (Liu et al. 2016). Besides that, some other researches have divided the labile carbon into water soluble organic carbon (WSOC) and light fraction organic carbon (LFOC) which were also available C for soil microorganism reproduction activity and decomposition, most of the WSOC and LFOC concentrations were depleted coincident with wetland degradation which usually accompanied by a drop in water table (McDowell et al. 2004; Zhang et al. 2007; Fang et al. 2009).

Research on labile organic carbon of estuarine wetlands with different groundwater tables in the Yellow River Delta shows that water table and soil water content synergistically affect soil labile organic carbon (MBC and DOC) dynamic, the response of labile organic carbon to water conditions varied with different water tables. (Yu et al. 2020). A study of differences in soil labile carbon between degraded and non-degraded wetlands observed that a significant linear decrease in the water soluble organic carbon (WSOC) and light fraction organic carbon (LFOC) values with the water table drops, deeper water table resulted in significantly smaller average water soluble organic carbon (WSOC) and light fraction organic carbon (LFOC) concentrations. In the degraded wetland compared to the non-degraded wetland the light fraction organic carbon (LFOC) concentrations significantly differed, this was attributed to the difference in water table between the degraded and non-degraded wetland. (Gao et al. 2014a). Laboratory incubation experiments with samples collected from two bogs and a poor fen in northern Minnesota found that increasing DOC concentration from 0 to 50% significantly enhanced methane production in the surface peat layer (25–50 cm) indicating that DOM acts as a primary driver of surface methanogenesis in peatlands (Hopple et al. 2019). In addition, several studies have found similar radiocarbon ages of methane and DOC in peat columns which supports that DOC is an important substrate source for methanogenesis in peatland (Charman et al. 1999; Wilson et al. 2016). Field observations in boreal peatlands also found a positive correlation between DOC concentrations and methane emissions (Pastor et al. 2003). A study of DOC dynamics and their relationship to wetland methane emissions in a subtropical wetland complex in central China shows during floods and high water table conditions, DOC concentrations in the surface peat increased significantly. The DOC concentrations of the porewater from the depth of 0–10 cm were higher during July and August (higher water table) compared to the other months (lower water table). The averaged DOC concentration reached the maximum of ca. 40 mg/L in the wet August, meanwhile methane emissions also increased. Their research results suggest that DOC concentration in the porewater of surface peat layers have affected the flux of methane emission from wetlands. They

further discussed that when methanogenesis is no longer primarily limited by temperature and water table, substrate availability, and hence DOC content, may become the limiting factor for methanogenesis (Zhang et al. 2022b). A similar positive relationship between DOC concentrations and methane emissions has also been found in paddy soils (Lu et al. 2000; Said-Pullicino et al. 2016; Bertora et al. 2018).

Soil DOC:TN ratio influences nutrient supply of the decomposition process of methanogenic microorganisms. (Zhang et al. 2022a). Soil DOC was an active and labile chemical component and could be readily utilized by microbes in the decomposition process (Yang et al. 2017a), but the positive effects of DOC on CH₄ emissions were dependent on the status of inorganic N in the soil (Wu et al. 2017). Methane production depends strongly on the ratio oxidizing: reducing capacity of the soil, it can be influenced by e.g. addition of sulphate, which inhibits methanogenesis. The type and application mode of mineral fertilizers may also affect methane emissions. Addition of organic matter in the form of compost or straw causes an increase of methane emissions, but methane production is lower for materials with a low C/N ratio (Bouwman 1991). Many literatures have proved that C:N ratios affects methane emissions. Some studies shows that CH₄ flux are positively correlated with soil C:N ratios (Lv et al. 2018; Wanyama et al. 2019), while others have found negative correlation (Kato et al. 2011; Grasset et al. 2019). A study conducted in Zoige alpine wetland, situated on the eastern edge of the Qinghai–Tibetan Plateau, Southeast China revealed the range effect of C:N ratios on the CH₄ flux and methanogenic and methanotrophic abundances. Their research results show that the CH₄ flux and methanogenic and methanotrophic abundances maintained the high levels when soil C:N ratio was in the range of 11–24 and decreased exponentially with the increase of soil DOC:TN ratio, the CH₄ fluxes out of the specific C:N ratio range were almost under low soil water content conditions, which indicate that the CH₄ flux are simultaneously dependent on soil C:N ratios and soil water content (Zhang et al. 2022a). The correlation between C:N ratios and CH₄ flux was primarily related to either carbon or nitrogen limitation, which might affect the carbon supply for the microbial activity of CH₄ production (Girkin et al. 2019). Under low DOC:TN ratio conditions, DOC was low but may not a limiting factor when the N content was high, thus supplying more available N that enhanced the microbial activity of litter decomposition (Zhang et al. 2017; Aronson et al. 2019). Low DOC:TN ratios are typically observed under high water table conditions, meanwhile relatively low content of soil-pore oxygen could be depleted by the enhanced microbial activities, and then the consequent anaerobic conditions favored the methanogenesis and thus increased CH₄ flux (Wang et al. 2017).

SOC is a universal nutrient source for methanogenesis presenting the balance between the input of organic matter and output of the organic carbon losses through many processes such as decomposition, leaching and dissolved carbon export (Turetsky 2004; Wang et al. 2010b). In order to recover the function of the degraded wetlands for carbon storage, recent wetland restoration has occurred worldwide by means of restoring hydrology and vegetation and limiting overgrazing. Restoration of cultivated land is usually accompanied by changes in water table. A study on the response of soil organic carbon dynamics to the abandonment of cultivated land in Northeast China showed that the abandonment of cultivated wetlands resulted in an increase in SOC and the availability of C. The SOC content increased to 31, 44, and 107 g kg⁻¹ after these cultivated wetlands were abandoned for 1, 6, and 13 years, respectively, as compared to an SOC content of 28 g kg⁻¹ in the soil that had been cultivated on for 9 years. The root biomass was the key factor affecting SOC concentration according to the observation made during the recovery of cultivated soil that was abandoned. LFOC was the most sensitive to the changes of organic carbon following the abandonment of cultivated lands in the northeast China (Zhang et al. 2007). However, studies have also found that SOC values did not differ significantly between the reference (491.8 g C kg⁻¹) and the restored wetland (418.5 g C kg⁻¹) (Gao et al. 2014b). Freshwater addition (water table rise) enhanced ($p < 0.001$) the accumulation of SOC in the Yellow River Delta, China. The SOC mean concentration was 5.44 ± 0.57 g/kg for the freshwater treated wetlands (higher water table), and 3.45 ± 0.24 g/kg for the reference wetlands for the entire 0–40 cm soil depth (Wang et al. 2011). Lowering of the water table e.g. for agricultural use accelerates aerobic secondary peat decomposition and processes of earthification. Peatlands change from C sinks to C sources. Total organic C decreased with increasing human impact and intensity of drainage, in contrast, the more recalcitrant fractions increased (Heller and Zeitz 2012).

Wetland Plants

The effects of plants on methane emissions from wetlands are multifaceted and complex, directly or indirectly affecting CH₄ production, oxidation and transport to the atmosphere from wetlands (Fig. 2). CH₄ fluxes from wetland may vary widely under different dominant plant species. Research comparing four major swamp types based on data compilation of methane fluxes from swamps with different dominant plant species in North America: needle-leaved, broad-leaved, mixedwood and shrub/thicket swamps shows that broad-leaved swamps having the largest CH₄ flux of 126.8 ± 33.9 mg CH₄ m⁻² d⁻¹, whereas needle-leaved swamps have the smallest CH₄ flux at 13.5 ± 10.3 mg CH₄ m⁻² d⁻¹ (Davidson et al. 2022).

Wetland plants provides plant litter and root exudates containing labile organic compounds that facilitate methanogenesis (Wang et al. 2010a; Zhai et al. 2013; Turner et al. 2020). Studies also show that most CH₄ emitted from wetlands are generated from recently fixed carbon, which is directly linked to existing vegetation (Chanton et al. 1995; Strom et al. 2003; Campeau et al. 2017). While the aerenchyma of wetland plants transports oxygen to the rhizosphere (Joabsson et al. 1999; Kim et al. 2018), it could also serve as a low-resistance pathway for CH₄ to be transported from the rhizosphere to the atmosphere (Schimel 1995; Evans 2004; Iqbal et al. 2021), bypassing overlaying the methane oxidizing layer in the wetlands (Schimel 1995; Stepniewska et al. 2018). Studies of various wetland types support that aerenchyma tissues of wetland plants mediate the process of methane emissions to the atmosphere (Purvaja et al. 2004; Yu et al. 2013; Maier et al. 2018).

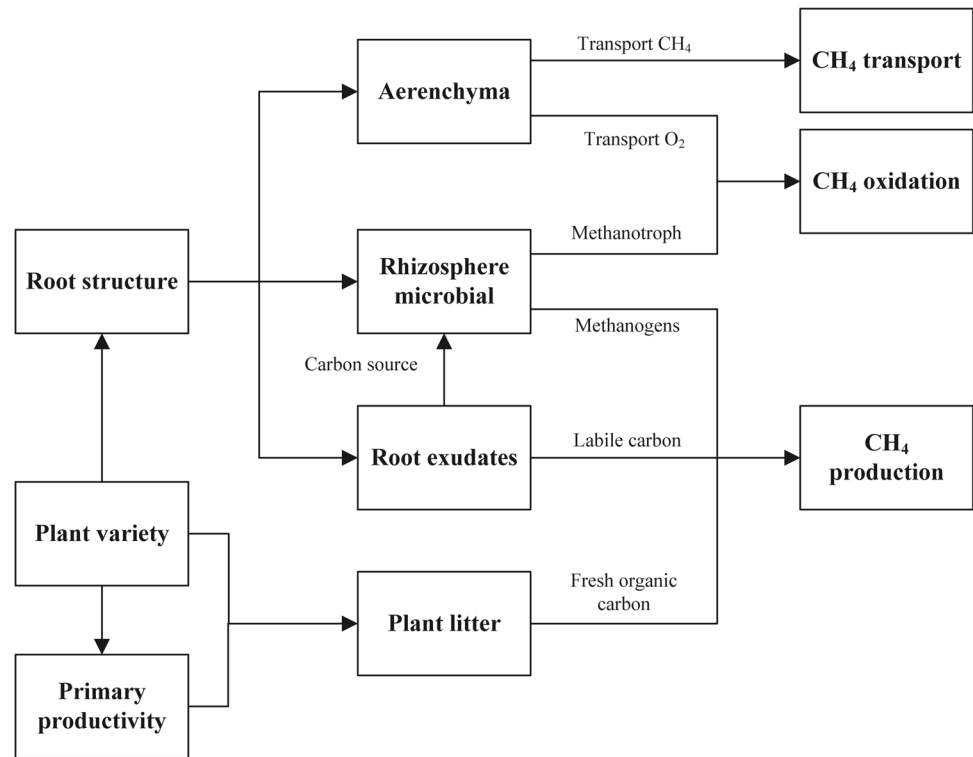
Wetland plants community structure has been shown to have an unequal effect on methane emission (Andrews et al. 2013; Vanselow-Algan et al. 2015). There are differences in the composition and quantity of root exudates in various plant communities (De Deyn et al. 2008), supporting specific rhizosphere microbial communities that utilize the exudates as growth substrates, some of which could be used for methanogenesis (Haichar et al., 2008; Berendsen et al. 2012). Recent research points out that differences in plant root structure may also affect methane emissions from wetlands (Lin et al. 2021). Root structures such as pneumatophores can suppress methane production and stimulate methane oxidation by oxygen transport into the root zone.

Recent research has demonstrated that the root zone size affects aerenchyma area; therefore root transverse section area could be used as an indicator for wetland plant species with reduced methane emission via transport through the aerenchyma (Kim et al. 2018). In one study, genetically engineered rice varieties were generated with less aerenchymas, which were shown to emit less methane (Fig. 3) (Iqbal et al. 2021), suggesting a promising strategy to control methane emission in wetlands through the use of plants with low methane emission potential.

Variability in the Response of Methane Emissions to the Water Table in Terrestrial Wetland

Strong correlation between methane emission and water table has been observed in previous studies (Philipp et al. 2017; Zhu et al. 2018; Griffis et al. 2020; Evans et al. 2021; Shahariar et al. 2021); however some studies reported otherwise (Zhao et al. 2020; Luta et al. 2021; Mwagona et al. 2021; Wang et al. 2021). For example, a study of three wetlands in Northeast China showed no significant effect of

Fig. 2 Multifaceted effects of plants on methane emissions from terrestrial wetlands



water table on CH_4 emission (Liu et al. 2015). Some studies have only found weak correlations between methane fluxes and the water table, with considerable inconsistencies (Counenberg et al. 2009).

One potential contributor to the observed variability in the response of methane emission to water table is the high spatial and temporal variability characteristic of field wetland sites. It has been further suggested that this variability could be attributable to the interactions between the water table temperature, vegetation and substrate quality (Mwagona et al. 2021), which requires further examination of the existing literature.

Coupling between the Water Table and Temperature

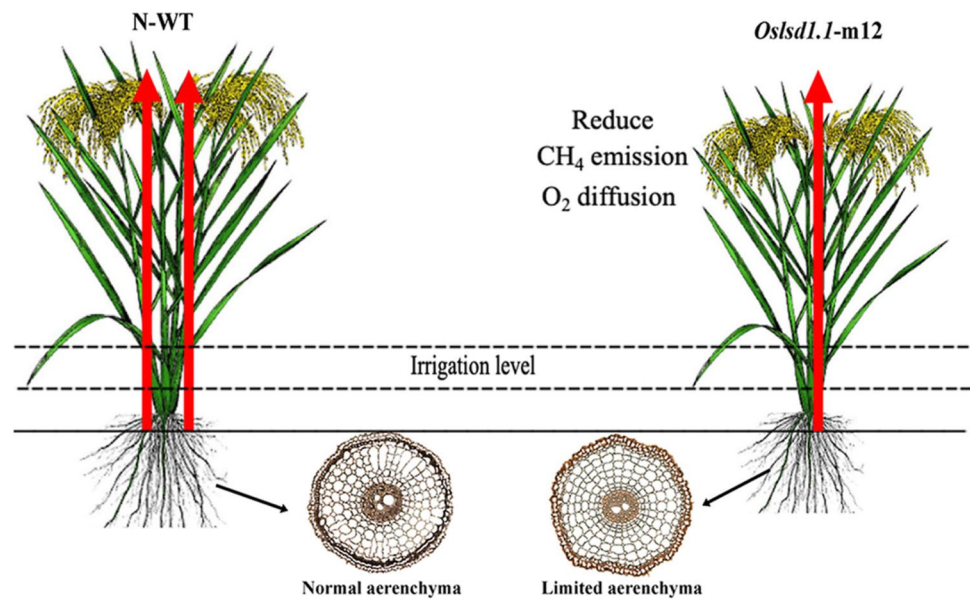
Temperature influences all metabolic reactions, including primary production and microbial respiration. Chemical kinetics dictate that reaction rates increase with increasing temperature in physiologically relevant ranges. Indeed, CH_4 emission flux measured in a wide range of wetland ecosystems show temperature correlations, since temperature affects methanogenesis, methane oxidation and methane transport simultaneously (Yvon-Durocher et al. 2014). Most studies revealed that the emissions of CH_4 from the decomposition of organic matter in wetlands increase with increasing temperature, which is consistent with kinetics predictions (Inglett et al. 2012; Cui et al. 2015; Bansal et al. 2016; Zhu et al. 2021). Temperature as a key factor

for almost all biochemical reactions, directly affecting the rate of methanogenesis.

Water table indirectly affects CH_4 fluxes by influencing soil temperature and the temperature sensitivity of methane emission. Study shows that there are differences in soil temperature under different water tables, this may be due to different water table resulting in different soil evapotranspiration rates (Mengistu et al. 2018). Water table affects soil temperature and thus methane fluxes. A coupled hydrological and biogeochemical model of wetland greenhouse gases shows that higher water table lead to increased soil temperatures, thereby raise CH_4 emissions (Grant 2015). Olefeldt et al.'s empirical models indicate that a 3.5 °C increase at 25 cm soil depth when the water table is level with the peat surface leads to 85% to 120% increases in methane emissions (Olefeldt et al. 2017).

In addition, studies indicate temperature sensitivity of methane emission is strongly regulated by the water table (Chen et al. 2021). A 2.5-year methane flux study of a raised peat bog in New Zealand indicated the relationship between temperature and methane flux was regulated by the water table. Results from this study show that 100 mm is a critical threshold for water table to regulate the temperature sensitivity of methane emission. When the water table was within 100 mm of the peat surface, a significant correlation between methane flux and temperature is established. However, there is no significant correlation between methane

Fig. 3 Normal versus limited aerenchyma (Iqbal et al. 2021) showing limited aerenchyma with a smaller root zone size and narrower aerenchyma



flux and temperature at deeper water tables (Goodrich et al. 2015), providing insights into potential strategies to control methane emission in wetlands when temperature effects are important.

Coupling between the Water Table and Wetland Plants

Plants play a key role in controlling methane emissions in wetlands (Robroek et al. 2015). However, the impact of wetland plants on methane emission has been observed to be dependent on the water table. Usually, different water table conditions raising varied plant communities in terrestrial freshwater swamps, and then water table coupled with plant species synergistically contribute to CH_4 production, oxidation and transport processes. CH_4 emissions from wetlands dominated by vascular plants rely more on species composition than that dominated by non-vascular plants. Wetlands with greater abundance of graminoids (e.g., fens) have higher CH_4 emissions than tree-dominated wetlands (e.g., swamps) (Bao et al. 2021). Field research in Nai-gombwa wetland, Iganga District, Southeastern Uganda shows that CH_4 fluxes ($\text{mg C m}^{-2} \text{ h}^{-1}$) were 15.6 ± 1.3 , 14.9 ± 1.2 , and 12.1 ± 0.8 from *Papyrus* communities with water table 7.5–31.5 cm, *Typha* communities with water table 7.1–29.1 cm and *Phragmites* communities with water table 1.9–21 cm, respectively (Were et al. 2021).

In general, high water tables result in increased methane emissions whereas low water tables reduce methane emissions. However, declining water tables were unexpectedly observed to increase CH_4 emissions when associated with enhanced root growth of aerenchymous species, such as *Juncus effusus* (Strack et al. 2006; Petersen et al. 2012). It

is likely that the extensive presence of the aerenchymatous pathway in wetlands contributed to the connection between the deep anaerobic methanogenic layers and the atmosphere, thus bypassing the upper aerobic layers where CH_4 otherwise would be oxidized (Henneberg et al. 2016; Waldo et al. 2019). These studies suggest that the selection of non-aerenchymous plant species could be a strategy to mitigate potential increases in methane emission in wetlands where high water tables are desirable.

Research in a tidal freshwater wetland suggest that changes in rates of Fe(III) reduction and methanogenesis were directly affected by plant mediated processes, plant regulation of Fe(III) reduction may be one of the mechanisms by which plants influence the carbon cycling and greenhouse gas production in wetlands (Neubauer et al. 2005). Plants growing in typically anoxic wetland soils have the potential to increase the abundance of NO_3^- , Fe(III), SO_4^{2-} , and other electron acceptors by introducing O_2 via their roots. They are also a major source of organic carbon, a universal electron donor for which heterotrophic microbes compete.

Coupling between the Water Table and Substrate Characteristics

Water table drawdown in wetlands has been shown to accelerate aerobic decomposition of organics in the substrate with increases in substrate aeration (Ellis et al. 2009), promoting microbial mineralization of organic substrates in wetlands and subsequent reduction in the potential of methanogenesis (Moore and Dalva 1997; Hribljan et al. 2014; Bragazza et al. 2016). For example, bacteria and fungi capable of decomposition of organic matter were shown with greater richness

under drier conditions in wetlands, again evidence of accelerated organic decomposition with lower water tables (Fenner et al. 2005; Barnard et al. 2013; Asemaninejad et al. 2017; Jassey et al. 2018). Research shows that high water table (close to surface) is beneficial to maintain substrate lability whereas long-term water table drawdown results in degradation of substrate quality with regard to the availability of labile organic carbon (Hribljan et al. 2017).

In contrast, high water tables may facilitate methane generation in wetlands. In particular, abundance of fresh surface litter with high water table was shown to contribute to high methane flux in wetlands, as the highest rates of methane production were found at surface substrates with fresh organic material accumulation and a water table close to the wetland surface (Glatzel et al. 2004). Consistent with the suggested importance of surface substrates in methane production in wetlands, it is observed that most of the methane fluxes take place in the relatively thin layers near the surface of wetlands (Vavrova et al. 2009), highlighting the importance of substrate characteristics in methane production.

The water table, an essential determinant of soil-organic-carbon dynamics interacts with soil organic carbon. It is not only the wetland water table that affects the wetland soil organic carbon composition dynamics, but in turn the organic carbon accumulation also affects the maintenance of the water table. Because of the high water-holding capacity of peat and its low hydraulic conductivity, accumulation of soil organic carbon raises the water table, which lowers decomposition rates of soil organic carbon in a positive feedback loop (Ise et al. 2008).

The impact of water table on methane emission in wetlands could be complicated by the presence of alternative electron acceptors that may suppress microbial methanogenesis. As discussed above, high water tables limit oxygenation and promote highly reducing conditions conducive to methanogenesis; but with the presence of alternative electron acceptors, which thermodynamically outcompete methanogenesis, methanogenesis favored by high water table could be suppressed (Sahrawat 2004; Noyce and Megonigal 2021; Rush et al. 2021). Specifically, NO_3^- , an alternative electron acceptor readily generated by microbial nitrification, was shown to outcompete methanogenesis when O_2 availability is low with high water tables (Lozanovska et al. 2016; Vroom et al. 2018). Other studies have demonstrated that higher sulfur (S) content in wetland substrate may favor sulfate reduction against methanogenesis, thus lowering CH_4 emissions (Shannon and White 1996; Dowrick et al. 2006). A classic example of microbial competition for electron donors occurs in river–estuarine systems. Methanogenesis is generally negligible at high salinity because SO_4^{2-} reducers outcompete methanogens for electron donors (Kelley et al. 1990). It should be noted that the formation of sulfate is strongly affected by the water table (Estop-Aragones et al. 2013). For example, sulfate is generally generated during water table drawdown, and depleted after a rise in water table (Estop-Aragones et al. 2016).

Microbial respiration is a fundamental process that influences the capacity of ecosystems to store soil carbon,

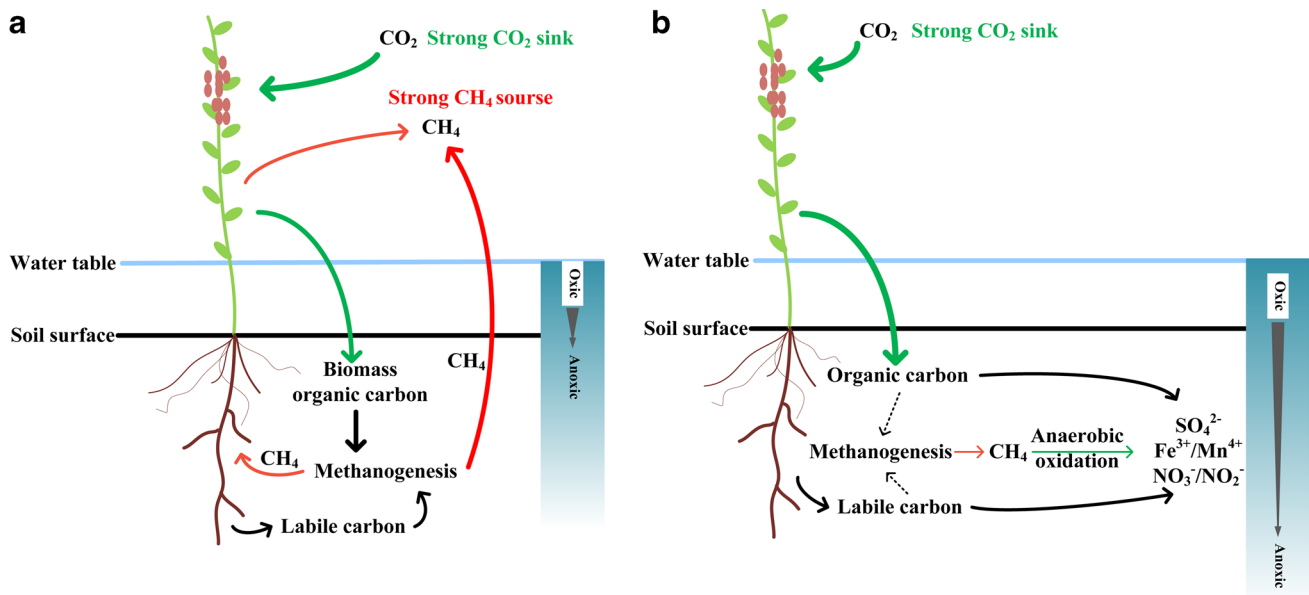


Fig. 4 WTMS 1—the water table is above the upper surface of the wetland substrate, resulting in complete inundation. **a)** Without the presence of alternative electron acceptors; and **b)** with the presence of alternative electron acceptors

Fig. 5 WTMS 2—The water table is below the substrate surface but above the plant root zone

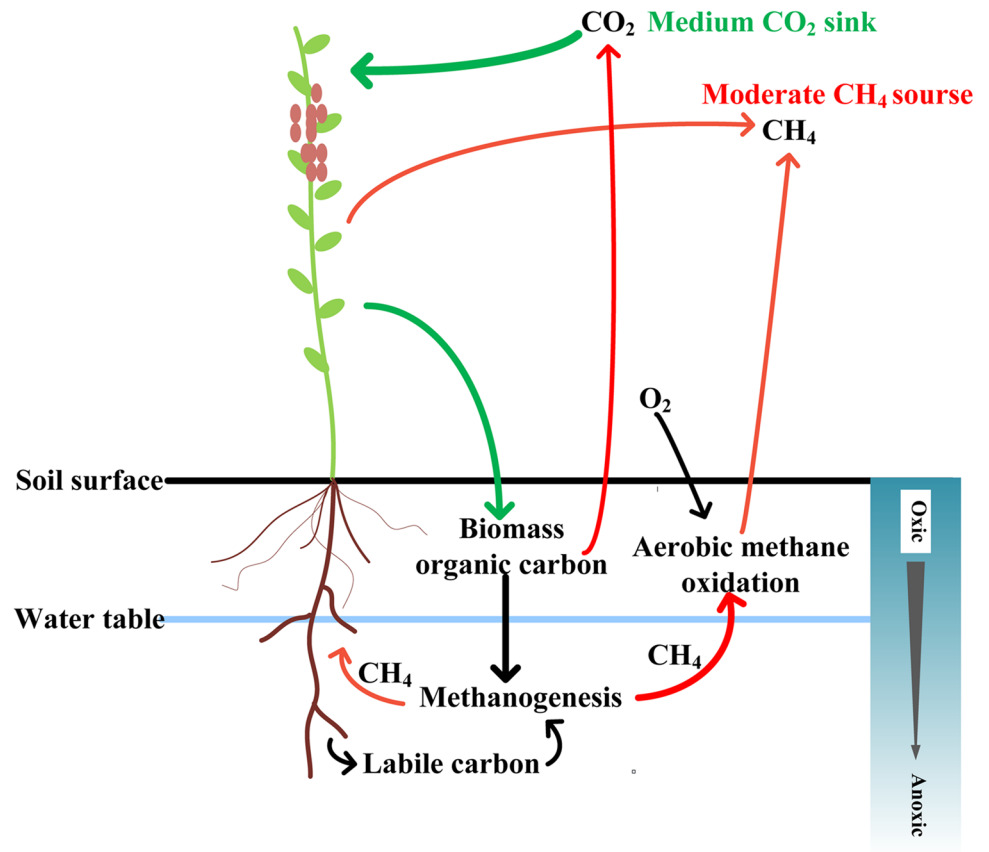


Fig. 6 WTMS 3—Water table is below the root zone where labile organic carbon is readily available

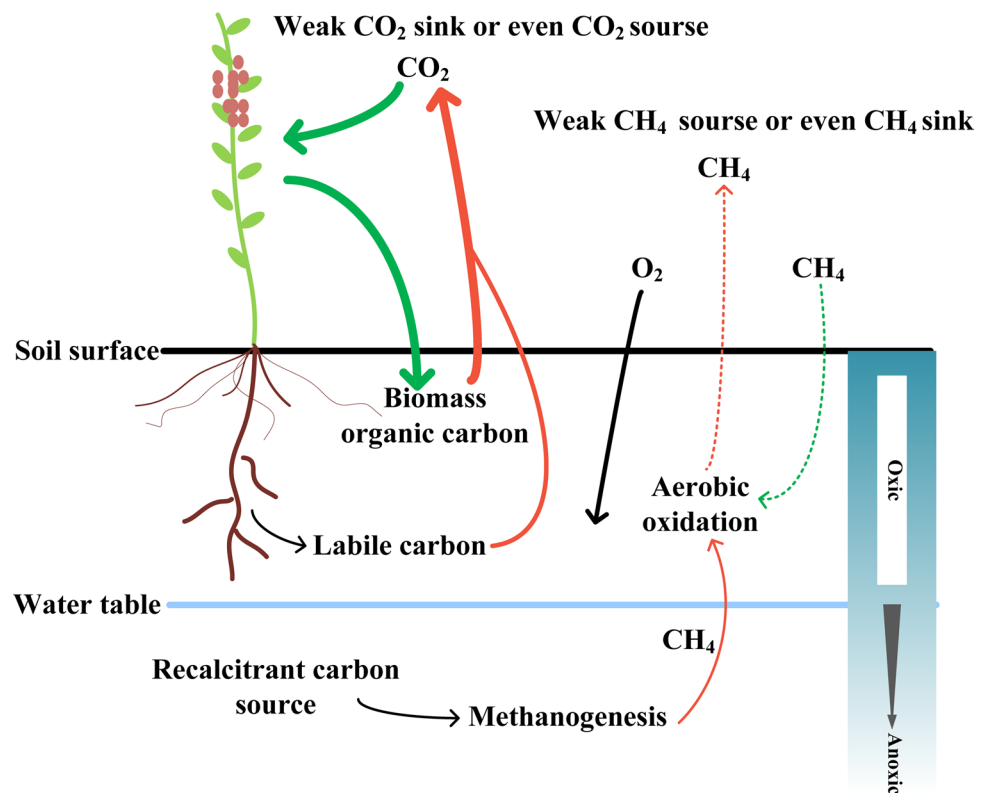
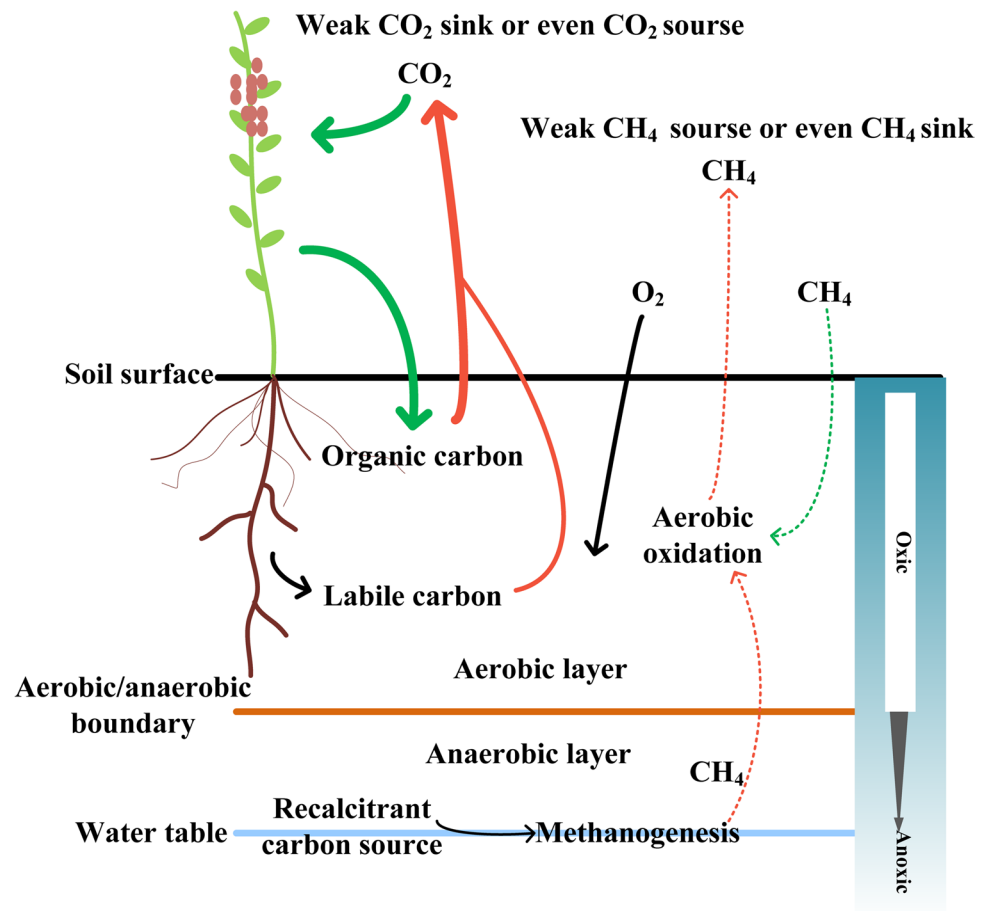


Fig. 7 WTMS 4—Water table is lowered to zones decoupled from the aerobic/anaerobic boundary



mineralize nutrients, and produce greenhouse gases. A central tenet of microbial ecology is that respiration is regulated by supplies of both electron acceptors and electron donors, and by competition between microbial groups for these resources (Hedin et al. 1998). Thermodynamic theory dictates that the outcome of microbial competition for substrates depends on the energetic efficiency of individual metabolic pathways and therefore suggests predictable patterns of microbial activity with changes in the abundance of electron acceptors and donors (Megonigal et al. 2004). The energy yield of terminal electron accepting processes (TEAPs) for the oxidation of a given substrate follows the sequence aerobic > anaerobic respiration: Mn reduction > NO₃⁻ reduction > Fe(III) reduction (FeR) > Sulfate reduction (SR) > Methanogenesis (Hoehler et al. 1998).

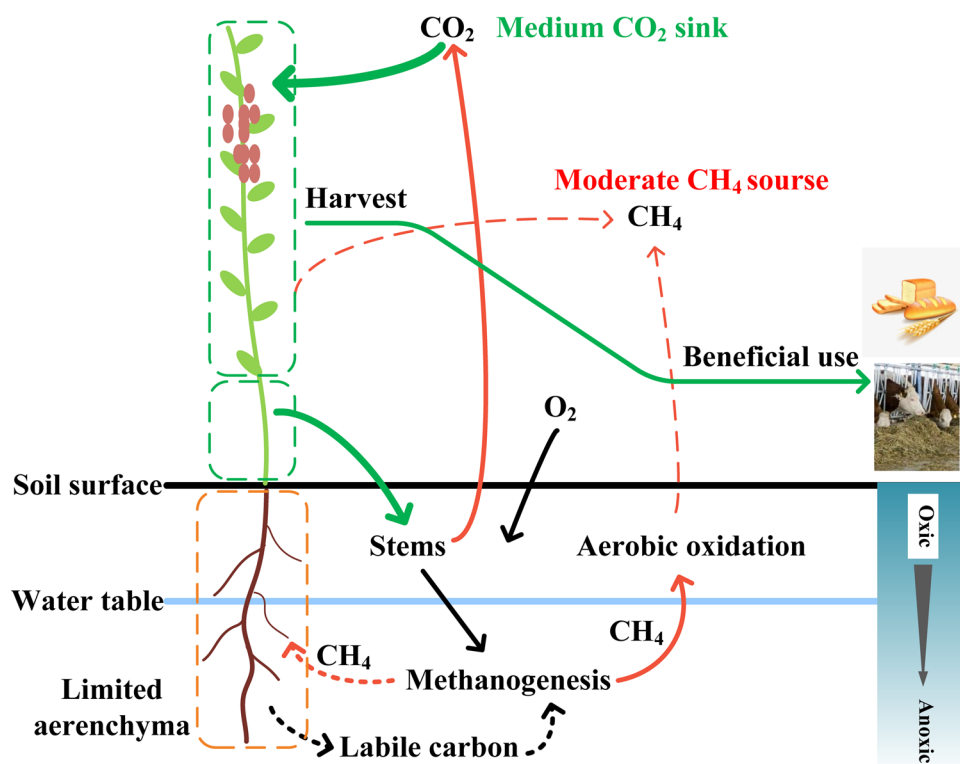
Another more important alternative electron acceptor is Fe(III), the suppressive effect of Fe(III) reduction on CH₄ production in wetland systems has mitigating global warming application potential (Roden and Wetzel 1996). High reduction rate of Fe(III) can effectively suppress methanogenesis (Neubauer et al. 2005). Microbial Fe(III) reduction can suppress both SO₄²⁻ reduction (Kostka et al.

2002) and methanogenesis (Frenzel et al., 1999), suggesting that Fe(III) reduction has the potential to dominate anaerobic carbon metabolism in wetland systems.

Seasonal biogeochemical patterns at the brackish marsh were affected by water table and iron-sulfur interactions (Neubauer et al. 2005). When the water table rises from a lower level to a near-surface state, the peat compaction controlled oxygen penetration, solid phase content of reactive iron controlled regeneration of electron acceptors and total reduced inorganic sulfur and organic matter content controlled the recovery of CH₄ production. High water table (flooding) led to accumulation of acetate and H₂, promoted CH₄ production and strengthened the co-occurrence of iron and sulfate reduction and methanogenesis (Estop-Aragones et al. 2013).

Water table fluctuation may result in hydrostatic pressure changes, which may cause lag effects on CH₄ emissions. Olefeldt et al.'s research shows their linear model over- and underestimated CH₄ emissions by ~25% when the water table had raised or dropped by 5 cm over the preceding 5 days, respectively (Olefeldt et al. 2017). In addition to this, differential growth rates between methanogens and methanotrophs (Segers 1998) and alternative electron

Fig. 8 Proposed strategy for sustainable freshwater wetland management and restoration



acceptors suppression effect would also contribute to lag effects (Knorr and Blodau 2009; Knorr et al. 2009; Deppe et al. 2010a, 2010b).

The coupling effect of the water table and substrate characteristics on methane emission suggests the need of site-specific surveys of wetland substrates for the development of methane emission mitigation strategies.

Critical Zone for Methane Emission in Wetlands

A nonmonotonic (Gaussian) relationship has been proposed between the water table and methane fluxes at the ecosystem scale, suggesting that there is a critical zone for the water table above and below which methane fluxes would decline (Brown et al. 2014). Water table dropping below the critical zone for an extended period expected to induce significant decreases in methane fluxes. The relationships between CH₄ fluxes and factors such as water table and vegetation would be overruled once the water table dropped below the critical threshold for CH₄ release (Zhu et al. 2014). For example, rapid and short-lived precipitation events might have no immediate influence on CH₄ emissions, which depends on whether the water table fluctuation crosses the critical zone or not.

The soil aeration zone, plant root zone, and moisture conditions contribute to the form of the critical zone for methane emission in wetlands (Zhu et al. 2014). Cluster roots of various plants exude large amounts of labile organic carbon

which could support methanogenesis (Lambers et al. 2012). Water table dictates the redox condition of the site-dependent critical zone thereby controlling whether methanogens or aerobic bacteria have access to labile organic carbon from cluster root exudation and then affecting methane emissions (Goodrich et al. 2015). Previous studies have suggested that when the water table was below the root zone of vegetation which in most cases results in the aerobic zone extended beneath the root zone, methanogens no longer had access to labile root exudates, and the plant transport flux pathway was also cut off (Lai et al. 2014), consequently resulting in a significant reduction in methane emissions.

Implications of Water Table on Methane Emission for Wetland Management and Restoration

Raising the water table or rewetting is a common practice for wetland management for climate mitigation, in particular to promote CO₂ sequestration. However, uncertainties with regard to methane emission due to higher water tables subsequent to re-wetting, warrant further assessment of the implications of the water table on CH₄ emissions during wetland management and restoration efforts (Evans et al. 2021).

Using peatland restoration as examples, rewetting and raising the water table have been reported to result in higher

CH₄ emissions (Strack and Zuback 2013; Holl et al. 2020). In comparison, drainage would significantly reduce CH₄ emissions to the atmosphere, on average by 84%. Restoration of drained peatlands by raising the water table through rewetting was shown to increase CH₄ emission on average by 46% compared to pre-management CH₄ fluxes (Abdalla et al. 2016). Peatland rewetting has been identified as a cost-effective measure to curb emission of CO₂, but would re-establish the emission of methane. In light of the stronger impact of CH₄ on global warming as compared with CO₂, it may seem imprudent to knowingly create or restore an additional source. Wetland restoration projects that convert drained land-use types into wetlands will help reduce the net emissions of CO₂ due to oxidation of organic matter in the converted wetlands. However, higher water tables are expected to increase the emissions of methane (Herbst et al. 2013; Miller 2011; Teh et al. 2011), a GHG with a global warming potential (GWP) 28 times greater than CO₂ over a 100-year time scale (Forster et al. 2021). Therefore, even relatively modest increases in methane emissions could offset the benefit of CO₂ sequestration in terms of the net greenhouse gas effect. It has been reported that prudent control of the water table during wetland management and restoration could prevent methane emission to a large extent. In a study of restored shrub bogs, non-inundated rewetting in drained/degraded low-latitude shrub bogs, such as pocosins (Wang et al. 2021), was implemented by maintaining the water table at least 5 cm below soil surface for about 90% of the year. It was demonstrated that CH₄ emission is negligible from these shrub bogs. This study suggests that water table control, via non-inundated rewetting in this case, could be an effective strategy to prevent methane emission from wetland management and restoration.

It should be noted that the overall methane budgets of wetland ecosystems remain highly uncertain (Bridgman et al. 2006). Despite a growing interest in wetland restoration for carbon sequestration (Maljanen et al. 2010), few studies have measured near-continuous CO₂ and CH₄ fluxes from restored wetlands (Waddington and Day 2007; Hendriks et al. 2007; Miller 2011; Herbst et al. 2013). Further studies are needed to systematically monitor the dynamics of GHG emission in wetlands with various management and restoration practices.

Comparisons of Methane Emission in Wetlands with Contrasting Water Table Management Schemes

With the significance of the water table in methane emission in wetland ecosystems, the impacts of the water table on methane production, oxidation, and transports are illustrated and compared in the following four water table management schemes (WTMSs).

WTMS 1

When the water table rises above the surface of the substrate (Fig. 4), all wetland substrates, including fresh organic carbon in the upper layer, would be fully inundated, creating an anaerobic environment conducive for methanogenesis with limited methane oxidation potential. Under this WTMS, significant emission of methane would be expected. If alternative electron acceptors, such as sulfate and nitrate, are present, methanogenesis could be temporarily suppressed until the alternative electron acceptor is gradually exhausted, resulting in a delayed increase in methane emission. The duration of this delay is dependent on site conditions that dictate the biogeochemical cycling of the specific alternative electron acceptors.

WTMS 2

When the water table is below the substrate surface but above the plant root zone where labile organic carbon is abundant, a strong positive correlation is anticipated between the water table and methane emissions (Fig. 5). Within this range, lower water table would shrink the anaerobic layer within the root zone, limit access of anaerobic microbial populations to labile organic carbon, thus reducing methanogenesis and methane production. On the other hand, lower water table allows deeper transport of oxygen into the wetland substrate, hence greater potential for methane oxidation. The combination of reductions in methane production and increases in methane oxidation would contribute to declines in methane emission with lower water tables. In contrast, rise in water table would reverse the above processes and lead to higher methane fluxes. WTMS 2 is characterized by the strong correlation between the water table and both methane production and methane oxidation, providing an effective control strategy to mitigate methane emission in wetlands where the water table can be manipulated.

WTMS 3

When the water table drops below the root zone (or the substrate layer where labile organic carbon is readily available), anaerobic conditions required for microbial methanogenesis would be limited to zones without ample supply of labile organic carbon (Fig. 6). Under this circumstance, methane production in the anaerobic zone would be limited by the availability of substrates, leading to significantly reduced rates of methanogenesis. On the other hand, the overlaying oxic zone would expand, resulting in greater potential for methane oxidation. WTMS 3 is thus characterized by low rates of methane emission

due to the lack of access to labile organic substrate by anaerobic microbial communities that carry out methanogenesis. Since methanogenesis is hindered by limited substrate availability while aerobic oxidation capacity is maintained in the overlying oxic zone, the flux of methane might even become negative which means the wetland could become a methane sink (Goodrich et al. 2015).

WTMS 4

Lowering the water table further would eventually lead to the decoupling of the water table from the aerobic/anaerobic boundary in wetlands (Fig. 7). The specific depth that the decoupling may occur is related to the physical properties of the wetland substrate. Once the water table is low enough to be decoupled from the aerobic/anaerobic boundary, the continuing drop of the water table will no longer cause changes in the aerobic and anaerobic layers. Thus, the methane flux is likely no longer affected by the water table under WTMS 4.

Implications for Wetland Management and Restoration from the Perspective of Controlling Methane Emissions

With the urgency to mitigate climate change, wetlands as one of largest natural sources of methane emission have been important targets for GHG mitigation. It is a sensible choice to develop responsible hydrological management strategy for wetland restoration to reduce greenhouse gas emissions, while making a vital contribution to global climate change mitigation. Wetland agriculture strategies are required to adapt crops to the wetland characters, and balance GHG mitigation against productivity (Freeman et al. 2022). However, outcomes of wetlands management and restoration efforts are uncertain with regard to methane emission. Building on the understanding of processes that influence methane emission in wetlands, particularly the water table, we envision strategies to minimize emission of CH₄ from terrestrial freshwater wetlands while retaining a certain carbon sequestration function, at the same time providing desirable ecological services.

Specifically, it is necessary to conduct preliminary investigations of the target wetland before implementing management and restoration practices. Based on current understanding of methane production, oxidation, and transport in wetland ecosystems, the following wetland characteristics are recommended for the preliminary investigation: substrate quality, depth profile of labile organic carbon, occurrence and distribution of alternative electron acceptors, composition of vegetation, and root zone profiles.

Given the importance of the water table in modulating methane emission in wetlands (Figs. 4, 5, 6 and 7), controlling the water table is an effective and implementable strategy to mitigate methane emission. WTMS 2 is recommended with the capacity to both reduce methane production and enhance methane oxidation, maximizing methane mitigation potential (Fig. 8). Additional advantages of WTMS 2 include the ability to support normal plant growth in the wetland by maintaining the water table above the root zone, thus preserving the ecological functions of healthy vegetation in wetlands. Given the significance of methane transport via aerenchyma, it is recommended that, if the selection of plant variety is practically permissible, wetland plants with limited aerenchymous tissues should be favored to minimize methane emission via aerenchymous transport. Other recommendations for wetland management include the harvesting of readily biodegradable biomass to reduce the input of labile carbon into the critical zone of methane production. It should be cautioned that these recommendations are based on current understanding of processes underlying methane emission in freshwater wetland ecosystems, where considerable uncertainties remain. With progress being made in the monitoring and modeling of biogeochemical cycling of carbon and nutrients in various types of wetlands, recommendations for strategies and practices to mitigate methane emission in wetlands need to be revisited periodically.

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Data Availability The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

Declarations

Competing Interests The authors have no relevant financial or non-financial interests to disclose.

Conflict of Interest The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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