



Modelling methane emissions from natural wetlands by development and application of the TRIPLEX-GHG model

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Abstract. A new process-based model TRIPLEX-GHG was developed based on the Integrated Biosphere Simulator (IBIS), coupled with a new methane (CH₄) biogeochemistry module (incorporating CH₄ production, oxidation, and transportation processes) and a water table module to investigate CH₄ emission processes and dynamics that occur in natural wetlands. Sensitivity analysis indicates that the most sensitive parameters to evaluate CH₄ emission processes from wetlands are r (defined as the CH₄ to CO₂ release ratio) and Q_{10} in the CH₄ production process. These two parameters were subsequently calibrated to data obtained from 19 sites collected from approximately 35 studies across different wetlands globally. Being heterogeneously spatially distributed, r ranged from 0.1 to 0.7 with a mean value of 0.23, and the Q_{10} for CH₄ production ranged from 1.6 to 4.5 with a mean value of 2.48. The model performed well when simulating magnitude and capturing temporal patterns in CH₄ emissions from natural wetlands. Results suggest that the model is able to be applied to different wetlands under varying conditions and is also applicable for global-scale simulations.

(Hartmann et al., 2013). Wetlands, as an important component of the terrestrial ecosystem, play a vital role in the global carbon cycle, which includes a CH₄ budget (Zhang et al., 2002; Ciais et al., 2013). CH₄ emissions from natural wetlands are the main drivers of the global interannual variability of CH₄ emissions with high confidence and contribute largely to interannual variations and anomalies of atmospheric CH₄ concentrations (Ciais et al., 2013). Therefore, it is vital to improve existing CH₄ emission quantification methods for wetlands to better understand the global CH₄ budget (Chen et al., 2013; Kirschke et al., 2013; Nisbet et al., 2014).

Over the last decades, three approaches have generally been used in estimating CH₄ emissions from wetlands across different scales: (1) an extrapolation of flux measurements approach, which uses actual CH₄ emission measurements to calculate a global estimation, (2) a bottom-up approach, which uses process-based models to calculate CH₄ fluxes based on understanding of CH₄ emissions and their environmental controls, and (3) a top-down approach, which uses inverse models to estimate the distribution of CH₄ sources and sinks by incorporating atmospheric observations (e.g. satellite observations), an atmospheric transport model and prior estimates of source distributions and magnitudes (Arneeth et al., 2010; EPA, 2010; Kirschke et al., 2013). The first approach can be unreliable in scaling from point measurements up to regional or global scales due to limitations in spatial and temporal coverage of measurements (Cao et al., 1996).

1 Introduction

Methane (CH₄) is an important greenhouse gas, with a 100-year global warming potential 28 times stronger than that of carbon dioxide (CO₂) (Myhre et al., 2013). Atmospheric CH₄ concentration in 2011 is 150 % greater than before 1750

The currently used top-down approach – generally believed proficient in covering large regions – may inadvertently include some incomplete observations and error amplifications during inverse modelling processes (Chen and Prinn, 2005; Ciais et al., 2013). Process-based models can be used to improve CH₄ emission estimation under different climatic regimes and at the same time cope with the complex interactions that take place between soil, vegetation, and hydrology under CH₄ production and consumption processes. The development and application of process-based models could be a practical alternative approach when extrapolating results from site scale to regional or global scale (Cao et al., 1996; Li, 2000; Zhang et al., 2002).

Several process-based models have been developed to estimate global CH₄ emissions. Each has its own strategy and features to deal with wetland system complexity and CH₄ flux processes (Li, 2000; Walter and Heimann, 2000; Zhuang et al., 2004; Meng et al., 2012). Cao et al. (1995, 1996) developed a CH₄ emission model for rice paddies based on C substrate level, soil organic matter (SOM) degradation and environmental control factors and improved it for global natural wetland simulation; but the model has no specific CH₄ emission process. Walter and Heimann (2000) and Walter et al. (2001a, b) developed a 1-D process-based climate-sensitive model to estimate global long-term CH₄ emissions from natural wetlands, forced with net primary production derived by a separate model. Li (2000) developed a denitrification–decomposition model (DNDC) to simulate CH₄ emissions but only for rice paddies. Zhang et al. (2002) adopted the DNDC model and some of its key components to simulate wetland ecosystem emissions. Zhuang et al. (2004) have considered the important freeze–thaw processes and integrated methanogenesis modules into the Terrestrial Ecosystem Model (TEM), but only applied it to estimate net CH₄ emissions in the high-latitude area of the Northern Hemisphere. A process model (PEATLAND) was developed to simulate CH₄ flux from peat soils (van Huissteden et al., 2006) and up-scaled for global boreal and Arctic wetland simulations (Petrescu et al., 2010), although the model did not include explicit soil biogeochemical processes. Wania et al. (2010) integrated a CH₄ emission module into the modified dynamic global vegetation model Lund-Potsdam-Jena (LPJ) to simulate CH₄ emissions from northern peatlands with consideration of permafrost dynamics, peatland hydrology and peatland vegetation. This model was then modified to simulate global net CH₄ emissions for northern peatlands, naturally inundated wetlands and rice agriculture soils (Spahni et al., 2011). To characterize uncertainties and feedbacks between CH₄ flux and climate, Riley et al. (2011) developed a CH₄ biogeochemistry model (CLM4Me) and integrated it into the land component of the Community Earth System Model (CESM) and further analyses were conducted by Meng et al. (2012), but specific plant functional types have not been considered in wetlands.

The Wetland and Wetland CH₄ Inter-comparison of Models Project (WETCHIMP), which simulates and compares large-scale wetland characteristics and corresponding CH₄ emissions, reported that large uncertainties indeed still exist when estimating CH₄ emissions (Melton et al., 2013; Wania et al., 2013). These uncertainties are generally introduced from large temporal and spatial variations in CH₄ flux, with the complex processes that underlie CH₄ emissions and also the limited inherent range of field and laboratory measurements (Arneth et al., 2010; Wania et al., 2010; Spahni et al., 2011; Meng et al., 2012). Therefore, further development of process-based CH₄ emission models is critical (Walter and Heimann, 2000; Ito and Inatomi, 2012). A dynamic global vegetation model (DGVM) generally combines vegetation dynamics, biogeochemistry, and biogeography processes to predict terrestrial ecosystem response to rapid climate change (Prentice et al., 1989; Cramer et al., 2001). Integrating the CH₄ emissions module into a DGVM would be an efficient approach to reflect interactions between hydrology, vegetation, soil and CH₄-related processes, and subsequently reducing uncertainties in CH₄ emission estimation at different spatial and temporal scales (Arneth et al., 2010; Tian et al., 2010; Wania et al., 2010). It would also be a practical approach to apply when predicting spatial and temporal patterns of CH₄ emissions under different future climate change scenarios (Gedney et al., 2004; Shindell et al., 2004; Tian et al., 2010; Stocker et al., 2013). To date, only a few DGVM models have an integrated CH₄ emission module (Wania et al., 2010; Spahni et al., 2011).

Therefore, a new model development framework of TRIPLEX-GHG in which a CH₄ emission model based on a synthesis of the previous studies discussed was integrated into a DGVM of the Integrated Biosphere Simulator (IBIS). The IBIS is designed to integrate a variety of terrestrial ecosystem functions within a physically consistent modelling framework and represents land surface processes, canopy physiology, vegetation phenology, long-term vegetation dynamics and carbon exchange (Foley et al., 1996; Kucharik et al., 2000). In addition, a water table simulation module based on the approach developed by Granberg et al. (1999) was also integrated into the IBIS and a new plant function type for wetland was added as well. The objectives of the current study are to: (1) integrate biogeochemical-based methanogenesis processes into a DGVM, which includes explicit description of the processes of CH₄ production, oxidation and transportation, for the interactions between hydrology (e.g. water table), vegetation (e.g. specific plant function type in wetlands, primary production) and soil biogeochemistry; (2) make the model applicable throughout global natural wetlands by adjusting a few sensitive parameters.

2 Model description and key processes

The TRIPLEX-GHG model (Peng et al., 2013) is based on the legacy of well-established and published models that include IBIS (Foley et al., 1996), DNDC (Li, 2000), TRIPLEX (Peng et al., 2002) and CASACNP (Wang et al., 2010). However, the scope of this study was only to introduce the development of the new wetland water table and the methanogenesis modules. The basic concept and structure of CH₄ emission and water table models and their integration into the IBIS are presented in Fig. 1 and details are described below.

IBIS represents vegetation with plant functional types (PFTs) characterized in terms of biomass and leaf area index (LAI) to simulate changes in vegetation structure on an annual time step through PFT sunlight and water competition (Foley et al., 1996; Kucharik et al., 2000, 2006). For wetland simulations, a new PFT was added in the model. Most of the PFT phenological and physiological parameters were adopted from the C3 grass PFT in the original IBIS model. The definition of inundation stress effects on gross primary productivity (GPP) of the added PFT in wetlands followed the assumption made by Wania et al. (2009a), that sphagnum and C3 graminoids photosynthesis will increase or decrease when water table rises or drops.

2.1 Water table module

Since the water table is an essential factor in determining anoxic and oxic soil zone extent where CH₄ is produced and oxidized, respectively, a water table simulation module was integrated into the IBIS. The module primarily follows the approach developed by Granberg et al. (1999) and the approach has been applied to studies by Zhuang et al. (2004), Weiss et al. (2006) and Wania et al. (2009b). Although applications of the water table simulation approach by Granberg et al. (1999) have been primarily carried out for peatlands or mires, the method was extended for water table simulations of natural wetlands in addition to peatlands for this study. Water balance is the basis of water table simulations. Two zones separated by water table surface were specified for wetlands: the anoxic zone (saturated zone) and the oxic zone (unsaturated zone). Positioning of the water table is subject to soil moisture change, i.e. the input and output volume of water in a specific location. The assumption is that standing water can occur above the land surface and that the drainage process through the bottom of the soil layers can be omitted. When the position of water table is higher than maximum standing water, excess water will be released as runoff. The water budget in the wetland soil profile was derived from the deficit of water input (precipitation) and water output (evapotranspiration and runoff). The water table is estimated by the equation provided below, which was also described in the studies of Granberg et al. (1999) and Wania et al. (2009b):

$$\text{Water_Table} = \begin{cases} V_{\text{tot}} - Z_{\text{acro}}\phi & \text{if } WT > 0 \\ -\sqrt{\frac{3.0 \cdot (Z_{\text{acro}}\phi - V_{\text{tot}})}{2.0 \cdot A_z}} & \text{if } WT \leq Z_{\theta_{s,\text{min}}} \\ -\frac{3.0 \cdot (Z_{\text{acro}}\phi - V_{\text{tot}})}{2.0 \cdot (\phi - \theta_{s,\text{min}})} & \text{if } WT > Z_{\theta_{s,\text{min}}} \end{cases} \quad (1)$$

where V_{tot} is total water content in the soil profile (cm), Z_{acro} is the maximum water table depth (30 cm, Frohling and Crill, 1994; Granberg et al., 1999; Zhuang et al., 2004), ϕ is the soil porosity (fraction), $\theta_{s,\text{min}}$ is the minimum volumetric water content at the soil surface (0.25, Granberg et al., 1999), $Z_{\theta_{s,\text{min}}}$ is the maximum depth where evaporation influences soil moisture (10 cm, Granberg et al., 1999). A_z is the gradient in the linearly decreasing interval, calculated as $A_z = (\phi - \theta_{s,\text{min}})/Z_{\theta_{s,\text{min}}}$. A negative/positive value of the water table indicates that the water table is below/above the soil surface, respectively.

2.2 Methane module

The CH₄ emissions module was adapted and integrated from a number of studies and models (Li, 2000; Walter and Heimann, 2000; Zhang et al., 2002; Wania et al., 2010; Riley et al., 2011; Spahni et al., 2011). Three major processes that include CH₄ production, transport (ebullition, diffusion, and plant-mediated transport) and oxidation were coupled with the IBIS. CH₄ is produced in each soil layer when soil conditions are favourable. The soil layers is 6 and the soil depth is set to 4 m as in the original IBIS model. The thermal and water balance processes were inherited from IBIS model. For the wetlands methane module, to simulate the dynamics of the water table, we divided the soil profile into 30 layers (1 cm per layer) above the maximum water table depth (30 cm). Thus 30 layers were used to simulate the water table changes and methane emission process. The soil above maximum water table depth was separated into anoxic and oxic zones, where CH₄ is produced and oxidized, respectively. The change in CH₄ for each time step in each soil layer is determined by the CH₄ production magnitude (Pro_{CH_4}), oxidation (Oxi_{CH_4}), and three transportation pathways (Ebullition: Ebu_{CH_4} , Diffusion: Dif_{CH_4} , and Plant Mediated Transport: PMT_{CH_4}). For each soil layer, CH₄ flux is the difference between production (Pro_{CH_4}) and consumption/emission ($\text{Pro}_{\text{CH}_4} - \text{Oxi}_{\text{CH}_4} - \text{Ebu}_{\text{CH}_4} - \text{Dif}_{\text{CH}_4} - \text{PMT}_{\text{CH}_4}$). Total CH₄ released to the atmosphere is the sum of the three-way transportation ($\text{Ebu}_{\text{CH}_4} + \text{Dif}_{\text{CH}_4} + \text{PMT}_{\text{CH}_4}$) of the total soil profile.

2.2.1 Methane production

CH₄ production is considered as the final stage of organic matter mineralization under anaerobic conditions (Cao et al., 1996). It depends not only on carbon substrate supply from plant primary production, but also on soil heterotrophic respiration rates and soil environmental conditions (such as the water table, soil temperature and hydrological regimes which

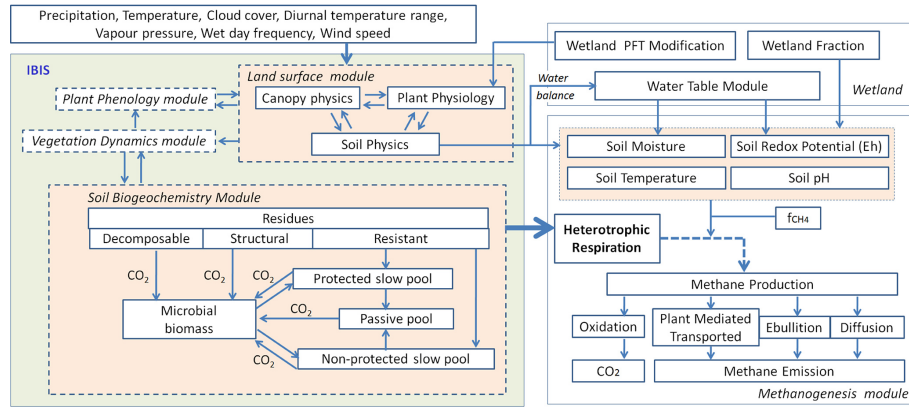


Figure 1. Basic structural concept and integration of CH₄ emission and the water table modules into DGVM of IBIS in TRIPLEX-GHG.

define the essential anaerobic conditions for methanogenesis (Moore and Knowles, 1990; Sass et al., 1990; Whiting and Chanton, 1993; Cao et al., 1996; Walter and Heimann, 2000; Walter et al., 2001b).

CH₄ production was calculated as a proportion of heterotrophic respiration (CO₂-C) along with soil temperature, Eh and pH modification factors as follows:

$$Pro_{CH_4} = R_H \cdot r \cdot f_{STP} \cdot f_{pH} \cdot f_{EhP}, \quad (2)$$

where R_H is the soil heterotrophic respiration rate ($gC\ m^{-2}\ day^{-1}$), calculated as the change in soil carbon pool size for each time step by the biogeochemical module in the IBIS. Along with the decomposition processes, carbon flows between different pools including above-ground and below-ground litter pools, a microbial pool, protected and nonprotected pools, and a passive pool. The carbon in litter and soil organic matter is partitioned between microbial biomass and respiration in decomposition processes, with assigned microbial efficiencies for each transformation among pools (Kucharik et al., 2000). A nitrogen feedback is added to control both above-ground carbon assimilation and below-ground soil organic carbon decomposition (Liu et al., 2005). f_{STP} , f_{pH} and f_{EhP} represent CH₄ production factors of soil temperature, pH and redox potential, respectively. r is the release ratio of CH₄ to CO₂.

Many studies have shown that CH₄ production is smaller or significantly lower than that found during growing seasons when soil temperature is below 0 °C (Whalen and Reeburgh, 1992; Shannon and White, 1994). CH₄ emissions during winter are suggested to be produced during the previous growing seasons and stored in the soil profile (Dise, 1992; Melloh and Crill, 1996). Therefore, CH₄ production was only permitted in the module when soil temperature was above the freezing point (0 °C) and below an extremely high temperature limit (45 °C in this study). The relationship between soil temperature and CH₄ production was adapted from Zhang et al. (2002) as described below:

$$f_{STP} = \begin{cases} 0 & \text{if } T_{soil} < 0 \\ 0 & \text{if } T_{soil} > T_{max} \\ vt^{xt} \cdot \exp(xt \cdot (1 - vt)) & \text{if } 0 \leq T_{soil} \leq T_{max}, \end{cases} \quad (3)$$

where

$$vt = \left(\frac{T_{max} - T_{soil}}{T_{max} - T_{opt}} \right) \quad (4)$$

$$xt = \left[\log(Q_{10}) \cdot (T_{max} - T_{opt}) \right]^2 \times \frac{\left[1.0 + (1.0 + 40.0 / [\log(Q_{10}) \cdot (T_{max} - T_{opt})])^{1/2} \right]^2}{400} \quad (5)$$

and T_{max} and T_{opt} are the highest temperature and optimum temperature for CH₄ production with values of 45 °C and 25 °C, respectively. T_{soil} is soil temperature (°C). For this study, a base Q_{10} value of 3.0 was used (Zhang et al., 2002) for simulations at different calibration sites.

Soil pH affected methanogenesis with a tolerance range between 5.5 and 9.0 while optimal values ranged between 6.4 and 7.8 with peak values ranging between 6.9 and 7.1 (Wang et al., 1993; Cao et al., 1995). Walter and Heimann (2000) included the effects of pH on CH₄ production in the tuning parameter. The approach by Cao et al. (1995) was adopted here to express the relationship between soil pH and CH₄ production:

$$f_{pH} = \begin{cases} 0 & \text{if } pH_{soil} > pH_{high} \text{ or } pH_{soil} < pH_{low} \\ \left(\frac{pH_{soil} - pH_{low}}{pH_{opt} - pH_{low}} \right) \cdot \left(\frac{pH_{high} - pH_{soil}}{pH_{high} - pH_{opt}} \right)^{\frac{pH_{high} - pH_{opt}}{pH_{opt} - pH_{low}}} & \end{cases} \quad (6)$$

where pH_{low} and pH_{high} represent low (4.0) and high (9.0) limitations of pH effect intervals. The optimal value was set at 7.0. Soil pH is taken from the soil properties data set.

Inundation causes low redox potential and promotes an anaerobic soil environment that will stimulate methanogenesis. The soil layer is considered to be inundated below the water table surface and the inundated condition of each soil layer changes with the dynamics of the water table. A linear relationship between soil water table position and CH₄ production was previously used to represent the effects of redox potential (Cao et al., 1996). In this study, redox potential changes are determined by water table position, water-filled pore space (WFPS) fraction, as well as root distribution (Zhang et al., 2002; Zhuang et al., 2004). For the effects of redox potential on methanogenesis, Cao et al. (1995) applied a switch value (−200 mV) for CH₄ production processes, while Li (2000) assumed that CH₄ production takes place with a soil Eh level below −150 mV. The relationship between redox potential and CH₄ production used in this study was generalized by Zhang et al. (2002), based on the studies by Fiedler and Sommer (2000) and Segers (1998). When redox potential is within a range between −200 mV and −100 mV, its effect on methanogenesis diminishes linearly from 1.0 to 0.0. Otherwise, the factor is equal to 1.0 and 0.0 when redox potential is less than −200 mV and greater than −100 mV, respectively.

The assumption in calculation of the parameter r in the CH₄ production adopted here was used in the CLM4Me (Riley et al., 2011; Meng et al., 2012) and LPJ-WHyMe (Wania et al., 2010; Spahni et al., 2011) models, in which CH₄ production within the anaerobic portion of the soil column relates to soil heterotrophic respiration, and where the soil carbon for methanogenesis is considered as a fraction of soil heterotrophic respiration.

2.2.2 Methane oxidation

CH₄ is oxidized by aerobic methanotrophic activities in the soil, taking place in the unsaturated zone above the water table (Cao et al., 1996; Li, 2000; Zhuang et al., 2004). The rate of CH₄ oxidation can be calculated based on a linear relationship with GPP (Cao et al., 1996) or as a function of soil CH₄ concentration and Eh (Li, 2000). Given that CH₄ oxidation is primarily controlled by CH₄ concentration, redox potential and soil temperature (Segers, 1998), the equation used here is as follows:

$$\text{Oxi}_{\text{CH}_4} = A_{\text{CH}_4} \cdot f_{\text{CH}_4} \cdot f_{\text{STO}} \cdot f_{\text{EhO}}, \quad (7)$$

A_{CH_4} is the amount of CH₄ in each soil layer ($\text{gC m}^{-2} \text{ layer}^{-1}$). The CH₄ concentration factor (f_{CH_4}) is represented by a Michaelis–Menten kinetic relationship: $C_{\text{CH}_4} / (K_{\text{CH}_4} + C_{\text{CH}_4})$, where C_{CH_4} is the CH₄ concentration ($\mu\text{mol L}^{-1}$) and K_{CH_4} is the half-saturation coefficient with respect to CH₄ and set to $5 \mu\text{mol L}^{-1}$ (Walter and Heimann, 2000; Zhang et al., 2002; Riley et al., 2011). A Q_{10} value of 2.0 based on results from previous studies (Segers, 1998; Walter and Heimann, 2000; Zhang et al., 2002) was used to quantify soil temperature effects on CH₄ oxidation (f_{STO}).

The term for redox potential effects on CH₄ oxidation (f_{EhO}) was adopted from a general relationship between redox potential and CH₄ oxidation reported by Zhang et al. (2002), which was taken from Fiedler and Sommer (2000) and Segers (1998). The redox potential factor was set to zero and 1.0 when Eh was below −200 mV and above 200 mV, respectively. In the remaining range of −200 to −100 mV and −100 to 200 mV, the effective factors of redox potential were represented by two simple linear functions, varying from 0 to 0.75 and from 0.75 to 1, respectively.

2.2.3 Methane emission processes

In early studies, no specific CH₄ emission process in CH₄ modelling existed (Cao et al., 1995, 1996). Generally, major CH₄ emission processes, including diffusion, ebullition and plant-mediated transportation, were formulated by the release of more recent models (Li, 2000; Walter and Heimann, 2000; Wania et al., 2010; Riley et al., 2011), as have been considered in this study.

CH₄ diffusion between soil layers was estimated here using Fick's law based on the CH₄ concentration gradient in the soil profile (Walter and Heimann, 2000; Zhuang et al., 2004). The diffusion coefficient for each soil layer was modelled as follows:

$$D_i = D_a \cdot f_{\text{coarse}} \cdot f_{\text{tort}} \cdot \text{SoilPorosity} \cdot (1 - \text{WFPS}_i) + D_w \cdot \text{WFPS}_i, \quad (8)$$

where D_a and D_w are the CH₄ molecular diffusion coefficients in air with a value of $0.2 \text{ cm}^2 \text{ s}^{-1}$ and in water with a value of $0.00002 \text{ cm}^2 \text{ s}^{-1}$, respectively (Walter and Heimann, 2000). D_a and D_w reflect differences in the rate of CH₄ molecular diffusion through unsaturated versus saturated soil layers. f_{coarse} is the relative volume of coarse pores depending on soil texture (fraction) (Zhuang et al., 2004). f_{tort} is the tortuosity coefficient with a value of 0.66 (Walter and Heimann, 2000). Lastly, SoilPorosity is soil porosity and WFPS is the water-filled pore space.

Being a relatively rapid channel for CH₄ emissions, bubbles will form as soon as CH₄ concentrations in the soil profile exceed a certain threshold (Walter and Heimann, 2000). For CH₄ ebullition emission processes, a constant threshold value of $750 \mu\text{mol L}^{-1}$ (Walter and Heimann, 2000; Zhang et al., 2002) was used in this study.

Vascular plants provide an effective pathway for CH₄ transport to the atmosphere (Shannon et al., 1996; Walter and Heimann, 2000). The plant-mediated flux is proportional to CH₄ concentration in the soil and is related to the concentration gradient between the soil and the atmosphere (Walter and Heimann, 2000). A simple equation was used to describe plant-mediated emissions based on the plant aerenchyma factor:

$$\text{PMT}_{\text{CH}_4} = f_{\text{rhi}} \cdot f_{\text{aer}} \cdot \text{CH}_{4\text{gra}}, \quad (9)$$

where f_{rhi} is the rhizospheric oxidation factor, suggesting that a relatively large proportion of CH_4 will be oxidized in the highly oxic rhizospheric zone before entering plant tissue (Wania et al., 2010). The factor is dependent on plant type and can range between 20 % and 100 % (Strom et al., 2005; Wania et al., 2010) and a constant value of 0.5 was used here (Zhang et al., 2002). f_{aer} is the plant aerenchyma factor estimated as a function of root length density (converted from root biomass using a specific root length of 2.1 cm mg^{-1}), the area of the cross section of a typical fine root (assumed as a constant of 0.0013 cm^2) and the degree of gas diffusion from root to atmosphere (a scalar determined by the aerenchyma condition of plants) (Zhang et al., 2002). A simplified constant of 0.5 was used as the degree of gas diffusion from root to atmosphere in this study, since the value should be 1 for the plants with well-developed aerenchyma (e.g. grasses and sedges) and be 0 for the plants without aerenchyma (e.g. sphagnum and moss). $\text{CH}_{4\text{gra}}$ is the CH_4 deficit between the soil profile and the atmosphere ($\text{gC m}^{-2} \text{ layer}^{-1}$).

2.2.4 Major parameters in methane module

The major parameters for the CH_4 module are presented in Table 1. Some of the parameters adopted values that have been fully discussed and supported in previous studies (Cao et al., 1996; Segers, 1998; Walter and Heimann, 2000; Zhang et al., 2002; Zhuang et al., 2004; Wania et al., 2010; Meng et al., 2012).

3 Data and method

3.1 Study sites

Observed CH_4 emissions data of natural wetlands based on previous studies or field work (19 sites worldwide from 35 studies) were collected for model sensitivity analysis, parameter fitting and calibration. Information related to these wetland sites, including location, wetland type, measurement method and references are summarized in Table 2. The selected sites have a wider geographical spread over low- to mid- to high-latitude regions.

Many of the sites listed in Table 2 have been also used in other CH_4 emission modelling tests. Since there were no observed detailed individual plot data in the studies carried out in the Amazon Basin (Bartlett et al., 1988, 1990; Devol et al., 1988; Melack et al., 2004) and South Florida (Burke Jr. et al., 1988; Harriss et al., 1988), with only observed emission rate throughout the regions, region-based average emission rates were compared between prediction and estimation. For some sites where different studies were carried out by different groups in the same or different time period (e.g. Sanjiang plain, Stordalen, Minnesota, Table 2), comparisons were made between the same modelling results and those results from different studies. Observation period modelling results were used for parameter fitting and model

performance evaluation. Monthly and yearly emission rates were calculated from simulated daily rates when comparing with monthly and yearly observed data.

3.2 Input data

For the regional simulations conducted in the Amazon Basin and South Florida, the CRU-TS 3.1 Climate Database (<http://badc.nerc.ac.uk/data/cru>) was adopted to construct monthly climate input data for the two regions. Selected variables include cloud cover, diurnal temperature range, precipitation, temperature, vapour pressure and wet-day frequency. For the site-based simulations in China, daily climate data were obtained from the Meteorological Bureau of China. For the other sites, since directly measured climate data was unavailable, the daily climate data (mean, max, min temperature, precipitation, mean wind speed, dew point) were downloaded from the nearest stations to the evaluation sites in the global data set of Global Summary of the Day (GSOD) (<http://www7.ncdc.noaa.gov/CDO/cdoselect.cmd?datasetabbv=GSOD&countryabbv=&georegionabbv=>) to drive the model. Being unavailable in GSOD, the station-based cloud cover data are extracted from the CRU-TS data set for each evaluation site. Climate data used for the model spin-up period were mean observed data for site-based simulation or 30-year (1961–1990) means of the CRU data (http://www.ipcc-data.org/obs/get_30yr_means.html) for regional simulation in Florida and the Amazon Basin.

CO_2 concentration data for the simulation period were composed of two parts. Observed CO_2 concentrations were used for the period covering 1958–2009, derived by Keeling et al. (2005) from in situ air measurements taken at Mauna Loa Observatory, Hawaii. CO_2 concentrations before 1958 were adopted from the IS92a global CO_2 concentration yearly data set, derived using a spline fit of Mauna Loa and ice core data (Enting et al., 1994).

The soil classification map used was based on the Digital Soil Map of the World (DSMW), generated from the FAO-UNESCO Soil Map of the World (<http://www.fao.org/geonetwork/srv/en/metadata.show?id=14116>). DSMW attributes were connected with the soil properties data set contributed by Batjes (2006) that describes characteristics of soil texture (soil clay, sand and silt fraction) and soil pH. A global soil data set (IGBP-DIS, 2000) was adopted to generate soil carbon data for model initialization.

A global digital elevation model (DEM) with an approximate 1 km spatial resolution (GTOPO30) was used for the topographic input data.

3.3 Initial sensitivity analysis for parameters fitting

For the model parameter fitting, initial sensitivity analysis experiments were conducted to obtain the most sensitive parameters in order to simplify the fitting processes.

Table 1. List of major parameters in CH₄ production, oxidation and transportation.

| Process | Parameters | Values | Unit | Description | References |
|--------------------|---------------------------|---------|---------------------------------|---|---|
| Methane production | T_{\max} | 45 | °C | highest temperature for methane production | This study |
| | T_{opt} | 25 | °C | optimum temperature for methane production | This study |
| | pH_{high} | 9 | – | highest pH for methane production | Cao et al. (1996), Zhang et al. (2002), Zhuang et al. (2004) |
| | pH_{low} | 4 | – | lowest pH for methane production | Cao et al. (1996), Zhang et al. (2002), Zhuang et al. (2004) |
| | pH_{opt} | 7 | – | optimum pH for methane production | Cao et al. (1996), Zhang et al. (2002), Zhuang et al. (2004) |
| | r | 0.1–0.4 | – | ratio of CH ₄ and CO ₂ | Wania et al. (2010), Zhang et al. (2002) |
| Methane oxidation | Q_{10} | 1.7–16 | – | Q_{10} for methane production | Dunfield et al. (1993), Walter and Heimann (2000) |
| | K_{CH_4} | 5 | µmol | Michaelis–Menten coefficients | Walter and Heimann (2000), Zhang et al. (2002) |
| Methane transport | Q_{10} | 1.4–2.4 | – | Q_{10} for methane oxidation | Meng et al. (2012), Walter and Heimann (2000), Zhang et al. (2002), Segers (1998) |
| | f_{ort} | 0.66 | – | tortuosity coefficient | Walter and Heimann (2000) |
| | D_{a} | 0.2 | cm ² s ^{−1} | molecular diffusion coefficient of methane in air | Walter and Heimann (2000) |
| | D_{w} | 0.00002 | cm ² s ^{−1} | molecular diffusion coefficient of methane in water | Walter and Heimann (2000) |
| | f_{rhi} | 0.5 | – | factor of rhizospheric oxidation | Wania et al. (2010), Zhang et al. (2002) |

Based on analyses carried out in previous studies, the Q_{10} (for processes of CH₄ production and oxidation) and the release ratio of CH₄ to CO₂ (r) were selected for initial sensitivity analysis for parameter fitting. Obtaining an optimal sensitive parameter combination at each individual site was then attempted. Previous studies have shown that CH₄ production and oxidation are primarily dependent on temperature, which is controlled by Q_{10} values (Cao et al., 1996; Walter and Heimann, 2000; Walter et al., 2001a; Riley et al., 2011). The Q_{10} values have large uncertainty with broad ranges (Dunfield et al., 1993; Westermann, 1993), which may be caused by substrate availability (Valentine et al., 1994), as well as the influence of temperature on plant growth and organic matter decomposition (Cao et al., 1996). Some studies have shown observed Q_{10} values ranging from 1.7 to 16 (Dunfield et al., 1993; Valentine et al., 1994) or from 1.7 to 4.7 (Valentine et al., 1994). Cao et al. (1996) used a Q_{10} value of 2.0 while Walter and Heimann (2000) and Walter et al. (2001b) used a Q_{10} value of 6.0. Zhuang et al. (2004) used an ecosystem-specific Q_{10} coefficient to evaluate soil temperature effects on CH₄ production at northern high latitudes. The parameter r in the CH₄ production was considered as the most important and influencing parameter for CH₄ emissions (Wania et al., 2010; Spahni et al., 2011). Cao et al. (1995, 1996) used a constant ratio to represent the proportion of decomposed organic carbon that can be converted to CH₄. Walter and Heimann (2000) and Walter et al. (2001b) used a tuning parameter calculated using a simple multiple linear regression of soil organic carbon and mean annual temperature to adjust the amplitude of simulated CH₄ emissions.

Zhuang et al. (2004) used an ecosystem-specific potential rate for CH₄ production.

Sensitivity is generally expressed as the ratio between a relative change of model output and a relative change of a parameter. The sensitivity index described in Lenhart et al. (2002) was used to quantify sensitivity in this study. The sensitivity index (I) is expressed as a finite difference in approximation of a partial derivative, which indicates the dependence of a variable (y) from a parameter (x):

$$I = \frac{(y_2 - y_1)/y_0}{2\Delta x/x_0}, \quad (10)$$

where y_0 is the model output with an initial parameter of x_0 . The initial parameter value varied by $\pm\Delta x$ ($x_1 = x_0 - \Delta x$ and $x_2 = x_0 + \Delta x$) with corresponding values y_1 and y_2 . The sign of sensitivity index (I) indicates the direction of the model's reaction to parameter change. According to Lenhart et al. (2002), calculated sensitivity indices are ranked into four classes (small to negligible, medium, high, very high). Model sensitivity to a specific parameter is small to negligible when the absolute value of the sensitivity index is less than 0.05 but very high when the absolute value of the sensitivity index is greater than or equal to 1.0.

3.4 Methods for parameter fitting evaluation

After the initial sensitivity analysis, parameter fitting processes for the selected most sensitive parameters was conducted to find the best combination of site-specific parameters. Indices including root mean square error (RMSE), coefficient of determination (R^2) and index of agreement (D)

Table 2. Information of collected sites for model parameter fitting. The information of nearest GSOD station for each site appended.

| ID | Sites | Location | Lat | Long | Type | Biome | Time | Method | Reference | GSOD station ID | GSOD station name |
|----|------------------------|-----------|--------------|--------------|----------------------------|-----------|------------------------------|--|------------------------------------|-----------------|---------------------|
| 1 | Stordalen | Sweden | 68°20' N | 19°03' E | subarctic mire | Boreal | 1974/1994/1995 | static chamber | Svensson et al. (1999) | 020200 | Katerjåkck |
| 2 | Degerö Stormyr | Sweden | 64°11' N | 19°33' E | boreal mire, fen | Boreal | 1995–1997 | static chamber | Jackowicz-Korczynski et al. (2010) | 022740, 022860 | Vindeln, Umeå |
| 3 | Salmisuo mire | Finland | 62°47' N | 30°56' E | boreal fen | Boreal | 1993 | static chamber | Grandberg et al. (2001) | 029290 | Juonsuu |
| 4 | Ruovesi | Finland | 61°50' N | 24°12' E | boreal fen | Boreal | 2005 | eddy covariance technique | Saario et al. (1997) | 029450 | Halli |
| 5 | Pionrkovo West Siberia | Russia | 57°00' N | 82°00' E | ombrotrophic bogs | Boreal | Jul 1993–1997 | static chamber | Rinne et al. (2007) | 293280, 296310 | Bakchar, Kolyvan |
| 6 | Fairbanks Alaska | USA | 64°48' N | 147°42' W | wetland | Boreal | 1987–1990 | static chamber | Parkov and Dedysb (2000) | 702610 | Fairbanks Int Arpt |
| 7 | BOREAS SSA | Canada | 53°48' N | 104°37' W | peatland, fen | Boreal | 1994–1995 | eddy covariance technique & static chamber | Sellers et al. (1997) | 718690 | Prince Albert Arpt |
| 8 | BOREAS NSA | Canada | 55°55' N | 98°23' W | peatland, fen | Boreal | 1994/1996 | static chamber | Sellers et al. (1997) | 710790 | Thompson Arpt |
| 9 | Quebec | Canada | 53°54' N | 78°46' W | peatland | Boreal | 2003 | static chamber | Pelletier et al. (2007) | 716278 | Kujjuarapik Arpt |
| | | | 53°38' N | 77°43' W | peatland | | 2003 | static chamber | | 718270 | La Grande Riviere |
| | | | 53°34' N | 76°08' W | peatland | | 2003 | static chamber | | 718113 | Nemisau Aprt (Sawr) |
| 10 | Sanjiang plain | China | 47°35' N | 133°31' E | marshland | Temperate | 2002–2005 | open-bottom chamber | Huang et al. (2010) | – | – |
| | | | | | natural freshwater wetland | | 2002–2005 | dark chamber and gas chromatography techniques | Song et al. (2009) | – | – |
| | | | | | marshland | | 2001 | static chamber | Wang et al. (2002b) | – | – |
| | | | | | marshland | | 1995–1996 | static chamber | Cui et al. (1997) | – | – |
| | | | | | freshwater marsh | | 2001–2002 | open-ended plexiglass chambers | Ding et al. (2004) | – | – |
| | | | | | marshland | | 2003 | static chamber | Yang et al. (2006) | – | – |
| | | | | | marshland | | 2002 | static chamber | Hao et al. (2004) | – | – |
| 11 | Qinghai–Tibet | China | 32°47' N | 102°32' E | peatland | Temperate | 2001 | static chamber | Wang et al. (2002a) | – | – |
| | | | | | peatland | | 2001–2002 | open-ended plexiglass chambers | Ding et al. (2004) | – | – |
| | | | | | peatland | | 2005–2006 | static chamber | Chen et al. (2008) | – | – |
| 12 | Minnesota | USA | 47°32' N | 93°28' W | peatland | Temperate | 1988–1990 | static chamber | Dise (1993) | 727450 | Duluth Int Airport |
| | | | | | peatland | | 1991–1992 | eddy covariance technique & static chamber | Clement et al. (1995) | – | – |
| 13 | Michigan | USA | 42°27' N | 84°01' W | peatland, duck hollow bog | Temperate | 1991–1993 | static chamber | Shannon and White (1994) | 725395 | Jackson Co Reynolds |
| 14 | Saltics Fen | USA | 43°12.5' N | 71°3.5' W | fen | Temperate | 1994–2001 | static chamber | Zhuang and Cihl (2008) | 726055 | Pease Int. Treadpor |
| 15 | Lact Vale, Colorado | USA | 40°17' N | 105°39' W | subalpine wetland | Temperate | 1996–1998 | closed chamber | Wickland et al. (2001) | 724675 | Eagle Co Regl |
| 16 | Mer Bleue | Canada | 45°41' N | 75°48' W | peatland, bog | Temperate | 2004–2007 | static chamber | Moore et al. (2011) | 710630 | Ottawa Recreation C |
| 17 | Ruans I Billingsong | Australia | 36°07' S | 146°58' E | freshwater wetland | Temperate | Apr 1993–May 1994 | portable gas filter correlation (GFC) measurement system | Boon (1995) | 948990 | Corryong Parish Lam |
| 18 | Florida | USA | 25°00' N | 81°00' W | everglade | Tropical | Jan–Feb, 1984; Dec, 1985 | static chamber | Bartlett et al. (1989) | – | – |
| | | | | | everglade | | Jan, Nov 1986; Mar, Aug 1987 | static chamber | Burke Jr et al. (1988) | – | – |
| | | | | | everglade | | 1980; 1982; 1985 | gas filter correlation infrared absorption analyzer integrated with an open bottom chamber | Harris et al. (1988) | – | – |
| 19 | Amazon | Brazil | Amazon Basin | Amazon Basin | flooded plain | Tropical | 1979–1987 | satellite inverse | Meack et al. (2004) | – | – |
| | | | | | | | Jul–Aug, 1985 | static chamber | Devol et al. (1988) | – | – |
| | | | | | | | 18 Jul–2 Sep, 1985 | static chamber and gas filter correlation | Bartlett et al. (1988) | – | – |
| | | | | | | | Apr–May, 1987 | static chamber and gas filter correlation | Bartlett et al. (1990) | – | – |

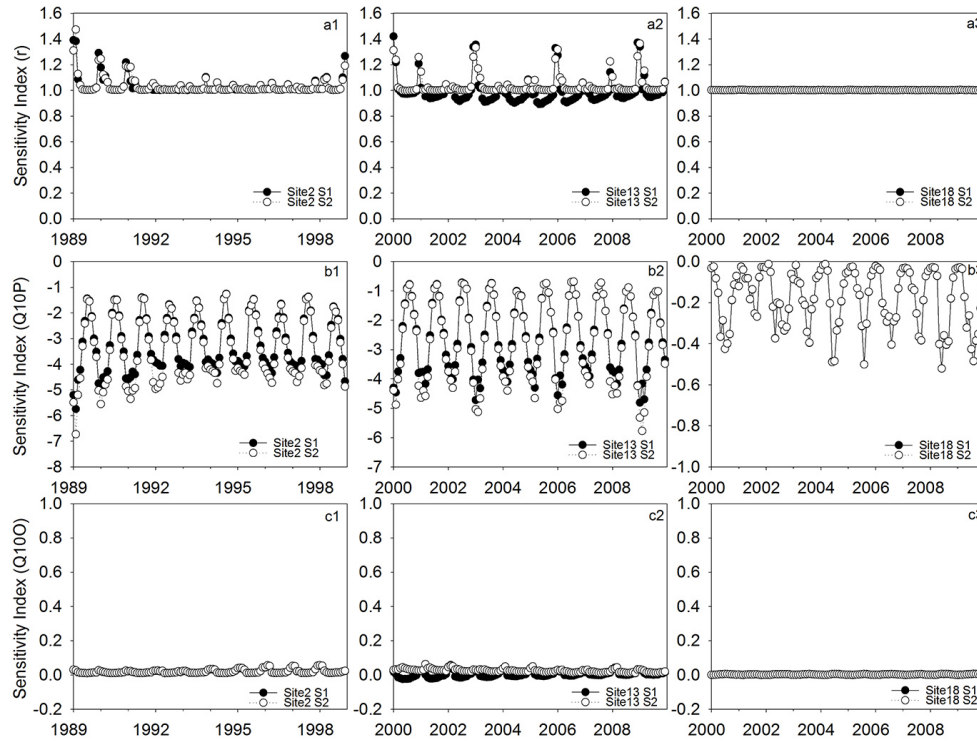


Figure 2. Sensitivity analysis of the three parameters (r , $Q_{10}P$, $Q_{10}O$) in the CH_4 module at three selected sites for different biome regions: boreal (site 2), temperate (site 12), and tropical (site 18). S1: scenario with single Δx ; S2: scenario with double Δx .

were used to evaluate the model’s performance. The RMSE was calculated as

$$RMSE = \sqrt{\frac{\sum_{i=1}^n (S_i - O_i)^2}{n}}, \quad (11)$$

where S are the simulated and O the observed values; n is the number of data. The D is calculated as

$$D = 1 - \frac{\sum_{i=1}^n (S_i - O_i)^2}{\sum_{i=1}^n (|S_i - \bar{O}| + |O_i - \bar{O}|)^2}, \quad (12)$$

where \bar{O} is the average of observed values. D varies between 0 and 1 and it is overly sensitive to extreme values (Willmott, 1981). A D value of 1 indicates a perfect match, and 0 indicates no agreement at all.

4 Results

4.1 Initial sensitivity analysis

Based on the initial values, Δx was set at 0.2 for Q_{10} in CH_4 production ($Q_{10}P$) and CH_4 oxidation ($Q_{10}O$), and set at 0.05 for the release ratio of CH_4 to CO_2 (r). Two scenarios

were considered, with single Δx (S1) and double Δx (S2). The initial sensitivity tests were conducted at three sites (site 2, site 13 and site 18, Table 2) by the biome type (boreal, temperate and tropical regions).

Over different biomes, sensitivity analysis results indicate that r (Fig. 2a1–a3) and $Q_{10}P$ (Fig. 2b1–b3) were very sensitive, while $Q_{10}O$ had a very low model sensitivity with a sensitivity index (I) of less than 0.05 (Fig. 2c1–c3). Seasonal patterns of I show that the model was more sensitive to $Q_{10}P$ during winter than other seasons in boreal (Fig. 2b1) and temperate (Fig. 2b2) regions. In contrast, the model was more sensitive during the summer than other seasons in the tropical region (Fig. 2b3). To simplify parameter fitting and make processes efficient as well as to assess model performance while reducing fluctuating parameters to as few as possible, $Q_{10}O$ was set as a constant value (2.0) and only two adjustable parameters were chosen (r and $Q_{10}P$) during the parameter fitting discussed below.

4.2 Model parameterization and calibration

Calibration sites were categorized into three main biome regions: tropical, temperate and boreal (Table 2). Parameters r and $Q_{10}P$ were adjusted to yield the best agreement for each site between simulation and observations (Figs. 3–5). The best parameter combination as well as the initial conditions of soil carbon and biomass after spin-up running for

Table 3. List of calibrated values for the parameters of r and $Q_{10}P$, the initial soil carbon and biomass conditions, and the model performance criteria indices for each site.

| ID | Sites | r (CH ₄ /CO ₂) | $Q_{10}P$ | Initial Soil Carbon (kgC m ⁻²) | Initial Biomass(kgC m ⁻²) | RMSE | R^2 | D | Counts |
|----|-------------------------|---|-----------|--|---------------------------------------|------|-------|------|--------|
| 1 | Stordalen | 0.13 | 2.00 | 17.12 | 1.36 | 0.03 | 0.67 | 0.83 | 17 |
| 2 | Degero Stormyr | 0.12 | 2.00 | 38.45 | 4.04 | 0.01 | 0.79 | 0.94 | 13 |
| 3 | Salmisuo mire | 0.27 | 1.70 | 31.23 | 5.77 | 0.04 | 0.55 | 0.83 | 6 |
| 4 | Ruovesi | 0.10 | 1.60 | 17.44 | 6.66 | 0.01 | 0.90 | 0.93 | 10 |
| 5 | Plotnikovo West Siberia | 0.20 | 2.00 | 24.00 | 6.19 | 0.09 | 0.52 | 0.73 | 6 |
| 6 | Fairbanks Alaska | 0.20 | 2.00 | 22.53 | 3.03 | 0.01 | 0.88 | 0.88 | 37 |
| 7 | BOREAS SSA | 0.22 | 2.00 | 27.46 | 6.08 | 0.06 | 0.54 | 0.85 | 12 |
| 8 | BOREAS NSA | 0.25 | 2.00 | 47.86 | 7.30 | 0.04 | 0.32 | 0.69 | 10 |
| 9 | Quebec | 0.10 | 3.00 | 20.91 | 4.69 | 0.02 | 0.93 | 0.94 | 9 |
| 10 | Sanjiang plain | 0.45 | 3.50 | 12.99 | 2.54 | 0.10 | 0.74 | 0.90 | 22 |
| 11 | Qinghai–Tibet | 0.21 | 3.00 | 31.57 | 3.45 | 0.07 | 0.54 | 0.76 | 20 |
| 12 | Minnesota | 0.14 | 2.30 | 13.07 | 8.30 | 0.03 | 0.70 | 0.91 | 44 |
| 13 | Michigan | 0.39 | 2.00 | 11.68 | 5.63 | 0.09 | 0.66 | 0.90 | 36 |
| 14 | Sallies Fen | 0.18 | 2.00 | 17.01 | 9.26 | 0.23 | 0.28 | 0.50 | 74 |
| 15 | Loch Vale,Colorado | 0.70 | 2.00 | 8.08 | 7.74 | 0.04 | 0.88 | 0.90 | 25 |
| 16 | Mer Bleue | 0.10 | 3.00 | 10.43 | 9.05 | 0.11 | 0.05 | 0.42 | 27 |
| 17 | Ryans 1 Billagong | 0.35 | 3.50 | 8.71 | 2.80 | 0.05 | 0.86 | 0.93 | 14 |
| 18 | Florida | 0.15 | 3.00 | 24.02 | 1.20 | – | – | – | – |
| 19 | Amazon Basin | 0.15 | 4.50 | 10.24 | 11.52 | – | – | – | – |

each site are listed in Table 3. The evaluation indices for the model performance were calculated with monthly results as also listed in Table 3. The release ratio of CH₄ to CO₂ ranged from 0.1 to 0.7 with a mean value of 0.23, and the Q_{10} value for CH₄ production ranged from 1.6 to 4.5 with a mean value of 2.48. Considering the different biome regions of boreal, temperate and tropical, parameter r has mean values of 0.18, 0.32, and 0.15, and parameter $Q_{10}P$ has mean values of 2.03, 2.66, and 3.75, respectively.

4.2.1 Boreal region

Data on five natural European wetland sites located in the boreal region were collected, including one subarctic mire site and one boreal mire site in Sweden (Svensson et al., 1999; Granberg et al., 2001; Jackowicz-Korczyński et al., 2010), two boreal fen sites in Finland (Saarnio et al., 1997; Rinne et al., 2007) and one ombrotrophic bog site in Russia (Panikov and Dedysh, 2000).

Svensson et al. (1999) and Jackowicz-Korczyński et al. (2010) took CH₄ emission measurements at the same site during different time periods (Fig. 3a, b). The mean emission rate of four wet and semi-dried sites in Svensson et al.'s (1999) study and data observed with the automatic chamber system in Jackowicz-Korczyński et al.'s (2010) study were used for comparison (Fig. 3a, b). CH₄ emission rates ranged approximately from 0.02 to 4.9 gC m⁻² month⁻¹ in the study by Svensson et al. (1999) and approximately from 0.9 to 5.3 gC m⁻² month⁻¹ in the study by Jackowicz-Korczyński et al. (2010). It was difficult for the model to catch peak values for some years at this location (Fig. 3a, b). For another study (Granberg et al., 2001) carried out in a boreal mire in Sweden, however, the model

simulated seasonal CH₄ emission variability with reasonable accuracy (RMSE = 0.01, D = 0.94, Fig. 3c).

We have compared modelled results from the current study with mean emission rates of four different vegetation surfaces (hummocks, flarks, *Eriophorum* lawns and *Carex* lawns) from a study by Saarnio et al. (1997) (Fig. 3d). Modelled daily CH₄ emission rates agreed reasonably with ranges of observed scatter points (RMSE = 0.04, R^2 = 0.55, D = 0.83) (Fig. 3d), while there was a time lag for the peak simulation.

A CH₄ flux study by Rinne et al. (2007) was measured using the eddy covariance technique. Gap-filling data using linear interpolation in that study was adapted for the modelling test carried out in the current study. Variation in modelled CH₄ emissions was consistent with that observed before early July and after mid-September (Fig. 3e). During the period from early July to mid-September, there was a drop in observed emission rates (Fig. 3e). Modelled emission rates showed higher values than observed emission rates during this period (Fig. 3e). Overall, emission rates agreed well between observation and simulation (RMSE = 0.01, R^2 = 0.9, D = 0.93).

In a study by Panikov and Dedysh (2000), annual variation in CH₄ emissions was relatively high, even during the same season. For example, emission rates in July ranged from 3.2 gC m⁻² month⁻¹ (1997) to 10.8 gC m⁻² month⁻¹ (1995) (Fig. 3f). The model simulated CH₄ emissions at this site reasonably well (RMSE = 0.09, D = 0.73) with the exception of those years that exhibited extremely high variation (e.g. 1993 and 1995) (Fig. 3f).

Figure 3g and h show the comparison of CH₄ emission rate between model simulations and observations obtained from the Boreal Ecosystem-Atmosphere Study

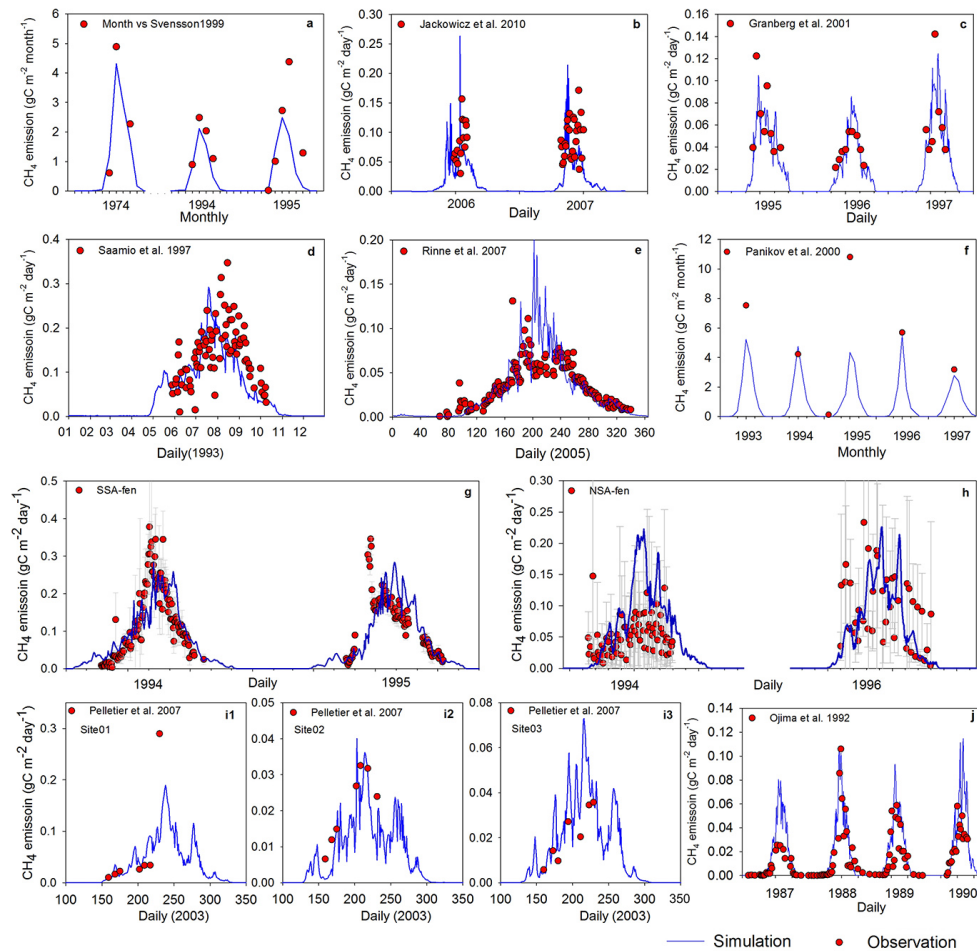


Figure 3. Comparison of modelled and observed CH_4 emissions for the sites located in the boreal region.

(BOREAS), a large-scale international interdisciplinary experiment situated in the northern boreal forests of Canada (Sellers et al., 1997). Daily observed CH_4 emission rates collected from the Southern Study Area (SSA-fen) and the Northern Study Area (NSA-fen) were used for comparison (http://daac.ornl.gov/BOREAS/bhs/BOREAS_Home.html). The magnitude of observed CH_4 emission rates for SSA was approximately twice that for NSA. For NSA, the mean CH_4 emission rate for 1996 was nearly twice that of 1994. These patterns indicate high temporal and spatial variation associated with CH_4 flux in these areas. The model had better performance in SSA ($R^2 = 0.54$, $D = 0.85$) than in NSA ($R^2 = 0.32$, $D = 0.69$). For SSA, the model failed to capture the 1994 peak emission. There was an approximate one month delay for the simulated peak emission rate when compared to observed data from 1995. For NSA, the model overestimated the 1994 CH_4 emissions while 1996 showed relatively better performance.

From a study by Pelletier et al. (2007), observed CH_4 emission rates measured on peatland covered by different vegetation types (excluding pools) was averaged for com-

parison. Based on the monthly mean emission rates comparison, the simulations agreed well with the observations ($\text{RMSE} = 0.02$, $R^2 = 0.93$, $D = 0.94$) (Fig. 3i1–i3).

Observational data of the Fairbanks site (Fig. 3j) was obtained from The United States Trace Gas Network (TRAGNET) online database (<http://www.nrel.colostate.edu/projects/tragnet/>) described in a workshop reported by Ojima et al. (1992). The model performed well ($\text{RMSE} = 0.01$, $R^2 = 0.88$, $D = 0.88$) except for some cases of overestimation during growing seasons.

4.2.2 Temperate region

Two sites in China were selected to test the model, one in northeast China and one in the Qinghai–Tibet Plateau. Natural wetlands and floodplains in China are primarily located in these two key regions (Ma et al., 2012). Figure 4a and b show comparisons between field measurements and model simulations in the Sanjiang Plain in northeast China. Figure 4a shows mean annual CH_4 emissions from wetlands of different plant types in a study by Huang et al. (2010) compared to

annual CH₄ emission rates of inundated marshes in a study by Song et al. (2009). CH₄ emission rates reported by Huang et al. (2010) were relatively low. Simulated CH₄ emissions for 2002–2003 were much higher than observed. For 2003–2004, CH₄ emission rates were higher for Song et al. (2009) by an approximate magnitude of 2.5 compared to those of Huang et al. (2010). Simulated results were within the range of these two independent studies.

The wetland plant type used in the studies was *Carex lasiocarpa* (Cui, 1997; Wang et al., 2002b; Ding et al., 2004; Hao et al., 2004; Yang et al., 2006). The peak values (growing season) of observed data in a study by Ding et al. (2004) were slightly higher than modelled results for 2001–2002 while the observed data in a study by Hao et al. (2004) were lower than those reported in a study by Ding et al. (2004) as well as this study's simulation for 2002. The model captured the main variations of observations and agreed well with the mean emission of all observations ($R^2 = 0.74$, $D = 0.90$).

For those studies carried out in the Qinghai–Tibet Plateau, modelled peak values were slightly higher than observed data collected from studies with the exception of a 2005 comparison reported in a study by Chen et al. (2008). It was observed that CH₄ emission rates reported by Chen et al. (2008) for 2005 were much higher than their reported 2006 emission rates and also higher than emission rates observed by Wang et al. (2002a) and Ding et al. (2004), even though site locations were situated very close to each other (Fig. 4c, d). By comparing all the observations conducted at Qinghai–Tibet Plateau with the model simulations, the model presented relatively low agreement index ($D = 0.76$) due to the mismatch of peak values.

Data on four natural American wetland sites located in the temperate region were collected. Observational data of Sallie's fen was obtained from the National Center for Ecological Analysis and Synthesis (NCEAS) data repository (Zhuang and Crill, 2008). The remaining observed data were obtained from corresponding citations (Fig. 4e–i) (Burke Jr. et al., 1988; Harriss et al., 1988; Bartlett et al., 1989; Dise, 1993; Shannon and White, 1994; Clement et al., 1995; Wickland et al., 2001). Although the model overestimated or underestimated emission rates for some peak values, Fig. 4e–i show that general seasonal patterns of CH₄ emissions simulated by the model were consistent with observations.

For the site located in north central Minnesota, the mean emission rate was calculated based on observations carried out under different vegetation conditions from a study by Dise (1993). Simulated CH₄ emissions were slightly lower than the observed emissions for winter (Fig. 4e) while modelled results agreed better for static chamber measurement than for eddy correlation measurement (Fig. 4f). The model reached an overall agreement index of 0.91 by combining chamber and eddy correlation data.

For the site located in Michigan, observed CH₄ emissions collected from three individual flux chambers distributed approximately 10 m apart in a study by Shannon and

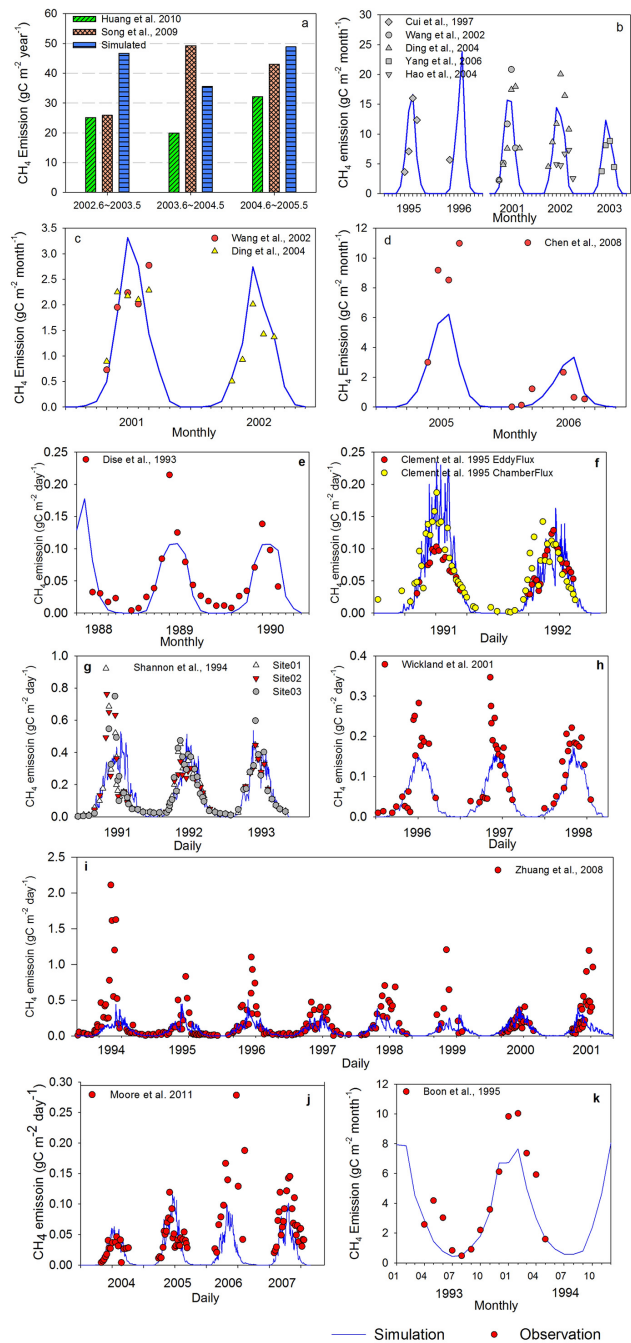


Figure 4. Comparison of modelled and observed CH₄ emissions for the sites located in the temperate region.

White (1994) were compared to simulated CH₄ emissions on a daily scale for the period 1991–1993 (Fig. 4g). Although the model failed to capture the 1991 summer emission pulse, it performed well for both 1992 and 1993 (Fig. 4g) and showed a good overall agreement index of 0.9.

For the site in a study reported by Wickland et al. (2001), the model again failed to capture peak emissions during the growing season, but it simulated quite well in the remaining

seasons (Fig. 4h). Based on the monthly mean emission rates, simulated results agreed well with observations ($R^2 = 0.88$, $D = 0.90$).

For the Sallie's fen site (Zhuang and Crill, 2008), simulated seasonal and annual variation agreed well with an 8-year uninterrupted observation (Fig. 4i). However, it is difficult for the model to capture several peak values during the growing season at this site, especially in 1994 (Fig. 4i), which accounts for the relatively low level of the evaluation index ($R^2 = 0.28$, $D = 0.50$).

In comparison with results reported by Moore et al. (2011), the model underestimated CH_4 emissions from 2006 to 2007, especially in 2006 (note: three observed value points that were much greater than $0.3 \text{ gC m}^{-2} \text{ day}^{-1}$ in 2006 are not shown on the graph), but it agreed well with observations from 2004 and 2005 (Fig. 4j). Overall, in the comparison, the index of agreement is low ($D = 0.42$) because the model did not capture the largest and smallest observations.

Additionally, a set of observed CH_4 emission data related to freshwater wetlands in southeastern Australia was collected from a study by Boon (1995) where field work was carried out over a 14-month period in 1993 and 1994. Seasonal and annual variation patterns for model simulations and observations stood in good agreement with each other (RMSE = 0.05, $R^2 = 0.86$, $D = 0.93$) (Fig. 4k).

4.2.3 Tropical region

Studies carried out in the Florida Everglades have no precise geographical point location; instead, regional averaged modelling results around the Everglades National Park were used for comparison (Fig. 5). Variations in observations ranged from $3.0 \text{ mgC m}^{-2} \text{ day}^{-1}$ to $481.5 \text{ mgC m}^{-2} \text{ day}^{-1}$. However, the highest emission rate ($481.5 \text{ mgC m}^{-2} \text{ day}^{-1}$) was measured in an open-water site while rapid ebullition was observed on all sampling dates (Burke Jr. et al., 1988). This produced extremely high CH_4 flux at this particular site. Emissions ranged from $3.0 \text{ mgC m}^{-2} \text{ day}^{-1}$ to $186.0 \text{ mgC m}^{-2} \text{ day}^{-1}$ if excluding the open-water site. Based on grid statistics, the simulated mean CH_4 emission for this area was approximately $60.0 \text{ mgC m}^{-2} \text{ day}^{-1}$ with a variation of approximately $20.0 \text{ mgC m}^{-2} \text{ day}^{-1}$. Modelled emission rate ranges were generally consistent with observed data (Fig. 5a).

Since precise information on field site locations was unavailable for the Amazon Basin, a simple comparison of monthly mean CH_4 emissions was made between mean field observations and regional averaged modelled results (Fig. 5b). Modelled monthly CH_4 emission rates were close to flooded forest emissions rates reported by Bartlett et al. (1988, 1990). Melack et al. (2004) reported that total annual CH_4 emissions from the central Amazon Basin had a mean value of 6.8 TgC yr^{-1} , based on remote sensing estimation. Total emissions for the same area where Melack et al. (2004) carried out their study was evaluated by the current

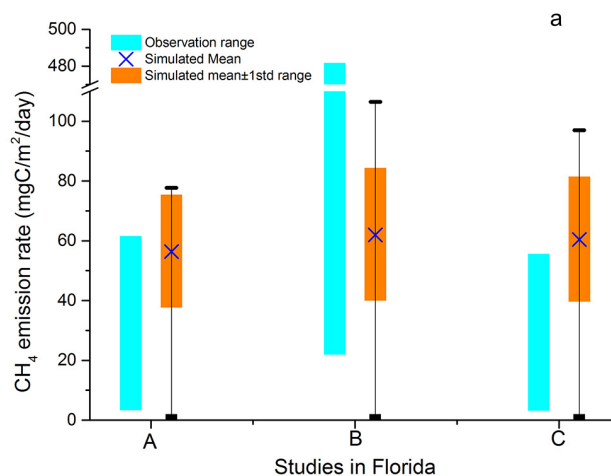


Figure 5a. Comparisons between simulation results of mean (cross symbol) ± 1 SD (orange column) and maximum and minimum (short bar) emission rates and observed emissions range (light blue column) of studies A: Bartlett et al. (1989), B: Burke Jr. et al. (1988), C: Harriss et al. (1988) in Florida everglade.

study, based on regional simulations. Modelled annual total emissions were 6.32 TgC yr^{-1} , which was close to the value (6.8) reported by Melack et al. (2004).

5 Discussion

Certain assumptions and equations from previous studies and models were adopted by this study in order to construct the CH_4 emission model before being integrated into a DGVM (IBIS) model. Parameters relatively high in model sensitivity were determined through initial sensitivity analysis. Wania et al. (2010) tested seven parameters using more than 2000 different parameter combinations; Riley et al. (2011) tested different parameters for CH_4 production, consumption and transportation processes; Meng et al. (2012) tested eight parameters for sensitivity within certain possible ranges. For this study, only three parameters were selected (r , Q_{10P} , and Q_{10O}) to conduct a series of initial sensitivity analyses. Results show that Q_{10} for CH_4 production had a much higher sensitivity level than Q_{10} for CH_4 oxidation, and the release ratio of CH_4 to CO_2 was highly sensitive to CH_4 emission processes and had direct impacts on CH_4 production. Much stronger temperature dependence during production compared to oxidation was also reported in previous studies (Walter and Heimann, 2000; Walter et al., 2001a; Wania et al., 2010; Riley et al., 2011; Meng et al., 2012), as was the release ratio of CH_4 to CO_2 , being the parameter that most influenced CH_4 emissions, with the ratio varying in a large range (Wania et al., 2010; Riley et al., 2011; Spahni et al., 2011; Meng et al., 2012). Therefore, the release ratio of CH_4 to CO_2 and Q_{10} for CH_4 production were selected for

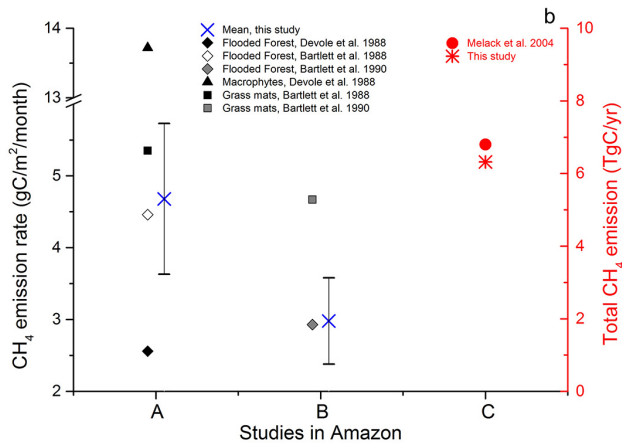


Figure 5b. Comparisons between simulation results of mean (cross symbol) ± 1 SD (short bar) and observations from studies of A: Devol et al. (1988) and Bartlett et al. (1988), and B: Bartlett et al. (1990); C: total annual CH_4 emission comparison between simulation and the study of Melack et al. (2004) in the Amazon.

parameter fitting in order to ascertain the best combination to test model performance and calibrate for individual sites.

Additionally, sensitivity levels indicate that Q_{10P} is much more sensitive in winter than summer for northern middle- to high-latitude regions, while the reverse pattern is observed in tropical regions. Furthermore, Q_{10P} has a higher sensitivity level in boreal and temperate regions than in tropical regions. The gradually increasing parameter Q_{10P} from boreal to temperate to tropical implies that the CH_4 emissions from wetlands are more temperature dependent in high-latitude regions than in the tropical region. Parameter r indicates that the release ratio of CH_4 to CO_2 is higher in the temperate region than the others.

Some simulated results failed to capture peak observed CH_4 emission values during the growing season. The underestimation of CH_4 emission pulse may be partly due to, on the one hand, external environmental triggers and CH_4 contribution from microbial mat systems during summer not being included in the model (Shoemaker and Schrag, 2010; Tian et al., 2010). On the other hand, the ebullition events of short duration, which are often recorded as very high fluxes in the observation, are hard to reproduce (Moore et al., 2011), since some dependent factors (e.g. the density of nucleation site) in CH_4 ebullition process are difficult to simulate (Wania et al., 2010). At the same time, some simulated peak emission values were not seen in the observations. One possible explanation could be the low sampling frequency typical of field work. Although some comparisons were carried out between simulations and observations using the high-frequency eddy covariance technique rather than low-frequency flux chambers, the footprint associated with eddy flux estimation depends on wind properties, boundary layers, surface roughness, etc. (Riley et al., 2011). Nevertheless, the

model was able to capture seasonal CH_4 emission variation patterns effectively. For example, a drop in observed emission rates was detected in a study by Rinne et al. (2007) during early July to mid-September. It was suggested that the major reason for this emission drop may be due to the fact that the methanogenic microbe population exceeded growth in available substrates (Rinne et al., 2007). At the same time, the green area of the vascular plants reached its maximum (Rinne et al., 2007), which means that more oxygen can be transported to the tips of the roots of the vascular plants. The subsequently high CH_4 oxidation rate will also contribute to the emission drop. A drop pattern was also found in our simulation during the same period.

Another reason for the differences between simulation and site observations may be attributed to the uncertainties of driving data. For some sites, climate data were obtained from the nearest meteorological station since climate observations in situ were unavailable. The differences of climate condition between modelling sites and meteorological sites could be large due to the effects of topography and geological location. For the model simulations in the tropical region, a global-scale climate data set was used. The climate data set cannot reflect the details of the daily or seasonal variation in temporal and spatial respects for the given large grid. Local climates may differ significantly from grid climates, especially in regions of high relief (Wania et al., 2010). Biases may be subsequently added to simulations of net primary productivity (NPP), heterotrophic respiration, soil water and thermal processes, which are important parameters for CH_4 production. Therefore, it is difficult for the model to capture full emission patterns. The soil property is also difficult to be replicated accurately at site level using a global soil data set, which is also a possible cause for the disagreement between model simulation and observation.

Uncertainties from field observations should also be taken into account in such comparisons. In many studies, for example, observations taken in the first year typically differ from subsequent years. This may be due to the fact that conditions of observational systems are unstable during initial setup. Larger differences are exhibited in the first year between simulations and observations.

It is important to note that two parameter values analysed in this study (r and Q_{10P}) are spatially heterogeneous. For the parameter fitting process, we tried to find the best site-specific parameter combination and to show how well the model can perform at the local site condition. This will continue to be an important issue for model simulation and development at both regional and global scales. Using different parameter sets or an overall uniform parameter set (e.g. average values for all sites) for site-, region- or global-scale simulations should be considered. Collecting more observed CH_4 emissions data sets from different geographic locations and wetland types could produce more reliable parameter sets under different conditions and locations, after which spatial distribution layers for highly sensitive parameters could be

constructed by a specific land surface classification (e.g. wetland types, biome types, etc.), which is something that has not been considered in recent studies (Melton et al., 2013; Wania et al., 2013), but which is important for the evaluation of wetland CH₄ emissions, especially at the regional or global scale. Better parameterization and evaluation of the model can potentially reduce the uncertainties in wetland CH₄ emissions response to projected climate change (Melton et al., 2013).

Further modelling, observational and data collecting works should be carried out to reduce uncertainties and enable more accurate simulations. Firstly, studies have pointed out that CH₄ emissions can be predicted well, but with incorrect contributions of production, oxidation and different transport (Riley et al., 2011; Melton et al., 2013). Further investigation should be conducted to examine methane fluxes from different transport pathways, the proportion of CH₄ production and oxidation, as well as the spatial patterns of these processes across wide geographic locations. Also, support is needed from explicit site observations of CH₄ emission processes, and relative physical and biogeochemical state variables (Melton et al., 2013). Secondly, the site-based simulations indicate that the model struggles to capture the peak emissions that are largely contributed by ebullition process. Though the process is hard to predict, preliminary improvement should be considered in the next stage. For example, a lower threshold could be used in the growing season, thus with higher ebullition (Spahni et al., 2011). Thirdly, in this current study, only one PFT was added for wetlands without considering specific wetland plants type (e.g. graminoids, sedges, sphagnum, moss). The model could be improved by considering the spatial characteristics of wetland vegetation, as well as including different effects of vascular and nonvascular plants on CH₄ emissions processes, especially for the oxidation and plant-mediated transport processes. Fourthly, CH₄ oxidation rate was simulated as a function of CH₄ concentration, soil temperature and soil redox potential in this study. Redox potential was used to represent the condition of soil electron acceptor. Although methane oxidation can happen under anaerobic conditions with available alternative electron acceptors (Zhuang et al., 2004), modelling dynamic changes of oxygen in the profile, including diffusion through soil layers and transport from atmosphere via vascular plants, is essential in the methanotrophic process (Wania et al., 2010). These processes need to be considered explicitly in further model development. Finally, more data from literature reviews or field measurements are needed to conduct the parameterization of the water table module and test model simulation performance, because the water table is an important factor controlling CH₄ production, consumption and transportation processes.

6 Summary and conclusions

This study has introduced the successful integration of a CH₄ biogeochemistry module incorporating CH₄ production, oxidation and transportation processes into an existing DGVM (IBIS). Factors controlling CH₄ emission processes, such as soil temperature, redox potential and pH, were specified into the model. A water table module was also integrated into DGVM to improve hydrological processes for wetland simulation. Sensitivity analysis indicates that the release ratio of CH₄ to CO₂ and Q_{10} for CH₄ production are two major controlling factors in CH₄ emission modelling. These two parameters were subsequently calibrated to data obtained from 19 sites collected from approximately 35 studies across different wetlands globally, which is a more extensive sampling than previous models have encompassed. Having a heterogeneous spatial distribution, r ranged from 0.1 to 0.7 with a mean value of 0.23, and the Q_{10} for CH₄ production ranged from 1.6 to 4.5 with a mean value of 2.48. We found that the TRIPLEX-GHG was able to capture patterns in temporal variation of CH₄ emission, but was unable to simulate daily details or the emission pulse. Results suggest that the TRIPLEX-GHG model can be applied to different wetlands under varying conditions. It should contribute to the scientific modelling community by accounting for GHG exchange and the budgeting of terrestrial ecosystems, especially the CH₄ budget at both regional and global scales.

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