

Analyzing the ecosystem carbon and hydrologic characteristics of forested wetland using a biogeochemical process model

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Abstract

A comprehensive biogeochemical model, Wetland-DNDC, was applied to analyze the carbon and hydrologic characteristics of forested wetland ecosystem at Minnesota (MN) and Florida (FL) sites. The model simulates the flows of carbon, energy, and water in forested wetlands. Modeled carbon dynamics depends on physiological plant factors, the size of plant pools, environmental factors, and the total amount and turnover rates of soil organic matter. The model realistically simulated water level fluctuation, forest production, carbon pools change, and CO₂ and CH₄ emission under natural variations in different environmental factors at two sites. Analyses were focused on parameters and inputs potentially cause the greatest uncertainty in calculated change in plant and soil C and water levels fluctuation and shows that it was important to obtain accurate input data for initial C content, climatic conditions, and allocation of net primary production to various forested wetland components. The magnitude of the forest responses was dependent not only on the rate of changes in environmental factors, but also on site-specific conditions such as climate and soil. This paper explores the ability of using the biogeochemical process model Wetland-DNDC to estimate the carbon and hydrologic dynamics of forested wetlands and shifts in these dynamics in response to changing environmental conditions.

Key words: biogeochemical modeling, carbon cycling, carbon pools, forested wetlands, greenhouse gases emission, hydrological process

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Introduction

Forested wetlands are recognized for their considerable potential to sequester C and their ability to affect global carbon budgets (Birdsey & Heath, 2001). Although they comprise a small proportion (i.e. 2–3%) of earth's terrestrial surface, they contain a significant proportion (35%) of the terrestrial carbon pool and contribute approximately 22% of the annual global methane emissions (Matthews & Fung, 1987; Bartlett & Harris, 1993). Certain types of wetlands contain large, historic reservoirs of carbon in above ground biomass, litter,

peats, soils, and sediments. Net carbon sequestration occurs as long as rates of conversion exceed decomposition and external transport of materials from wetlands. Carbon pools, contained in wetland forests, are a function of complex interactions of inherent soil processes, climate, vegetation, time, and disturbance regimes. The water level in the wetland governs many processes in this ecosystem such as, photosynthesis (Rydin & McDonald, 1985), partitioning of the surface energy and water balances (Kim & Verma, 1996), and the heat flux and temperature within the soil (Den Hartog *et al.*, 1994). As water level and temperature govern most biological processes, they also are strongly connected to emissions of greenhouse gases such as methane and carbon dioxide (Bubier & Moore, 1994; Granberg *et al.*, 1997).

Linking forested wetland hydrology with soil carbon dynamics is needed to provide a comprehensive

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modeling basis for evaluating how ecosystem C pools will respond to stressors (e.g. changes in anaerobic status in wetland soil will be determined by changes in water table dynamics, precipitation, and evapotranspiration, the latter of which will be affected by changes in surface energy exchange driven by changes in atmospheric conditions. Changes in soil moisture will affect C and nutrient mineralization by changing the rate at which oxygen (O₂) is supplied to microbial communities. Gross primary productivity will be strongly affected by changes in atmospheric CO₂ concentration, air temperature, and soil moisture). Therefore, models used in this ecosystem should include a fully coupled simulation of heat and water exchange between forested wetland surfaces and the atmosphere, and through the soil profile. Models should be able to explicitly simulate the biological processes by which atmospheric CO₂ concentration, temperature, and water stress affect forest productivity at a time scale appropriate to the large diurnal variation in these effects. Models should simulate gas transfer and exchange (O₂, CO₂, CH₄, N₂O, NH₃, N₂) in gaseous and aqueous phases of soils and roots.

A range of such models has been developed (e.g. McMurtrie *et al.*, 1990; Landsberg & Waring, 1997; Battaglia & Sands, 1997, 1998), and new models continue to come forward (e.g. Parton *et al.*, 1987; Letts *et al.*, 2000; Li *et al.*, 2000; Grant *et al.*, 2001; Grant & Roulet, 2002). Some models, such as BIOMASS (McMurtrie *et al.*, 1990), can successfully predict water use and carbon gain of stands, but do not model nutrient dynamics. Other models, such as CENTURY (Parton *et al.*, 1987, 1988), have been devised to simulate soil organic matter turnover, but have only simple procedures to handle plant processes and the important ecosystem components (i.e. wetlands, riparian zones). Some studies have examined the rates of decomposition for a particular wetland type or plant litter, or on organic matter accumulation rates in peatlands (Clymo, 1984). They have not specifically focused on how rates of decomposition vary and fluctuate over time, depending upon a variety of interrelated factors such as temperature, water levels, hydroperiod, flow of water and nutrients. In a recent study supported through the US Forest Service Southern Global Change Program, we reviewed 12 prominent soil C models (i.e. CANDY, CENTURY, DAISY, DNDC, ITE, MBL-GEM, NCSOIL, QSOIL, RothC, SOMM, VVV, and WMEM) to assess their suitability to hydric forest soils. Results demonstrated that most of these models do not account for anoxia, alternating hydroperiods, and complex interactions of soil chemistry that are inherent to wetland soils. Accordingly, those models do not include important wetland attributes and will yield erroneous

estimates of C pools and fluxes for critical components of the landscape. Although the critical role of hydrology in wetland functions is widely acknowledged, hydrologic considerations are inadequate in assessing C dynamics in wetlands and virtually absent from soil models (Trettin *et al.*, 2001).

A comprehensive wetland C model, which includes anoxia, surface, and ground water flows, soil chemistry, and vegetation dynamics will allow an assessment of the role wetlands have in C cycling. This capability is extremely important because of the unique biogeochemistry of forested wetlands, which make them an important C sequestration pathway with a disproportional influence on terrestrial C storage. In this paper, we explore the ability of a biogeochemical process model, Wetland-DNDC, to estimate the hydrologic and C dynamics of forested wetlands in different climate conditions. The strategies were evaluated primarily on three aspects. First, insight into the importance of processes and mechanisms: what kind of hydrologic and biogeochemical characteristics about the functioning of the forested wetland ecosystem can be derived from the model. Second, model performance: how well did the Wetland-DNDC model describe the measured data in forested wetland. Third, explores the application possibilities of the model and potentials for a reliable assessment of wetland ecosystem dynamics.

Methodology

Model description

A biogeochemical model, Wetland-DNDC has been constructed by integrating forest biogeochemical and hydrological processes in forested wetland (Fig. 1). Wetland-DNDC is a process model that simulates C, N, and water fluxes of forested wetland ecosystems. It is an extended version for use with different forest types in wetland, which may be identified and partially characterized by hydrological functions. The basic structure of the model incorporates the hydrological functions and at the same time, has been modified for spatial simulations – that is, it is prepared for use in a landscape or regional context. Wetland-DNDC tracks the flow of C through various components of forested wetland ecosystem including wood products, debris, soil, and accounts for the effects of harvest, multiple rotations, and water management (Zhang *et al.*, 2002; Li *et al.*, 2004). Major functions for simulating the forest physiology and phenology, biogeochemical processes, and hydrologic cycle in Wetland-DNDC listed in the Appendix.

The algorithms for photosynthesis, respiration in Wetland-DNDC were adopted from PnET (Aber &

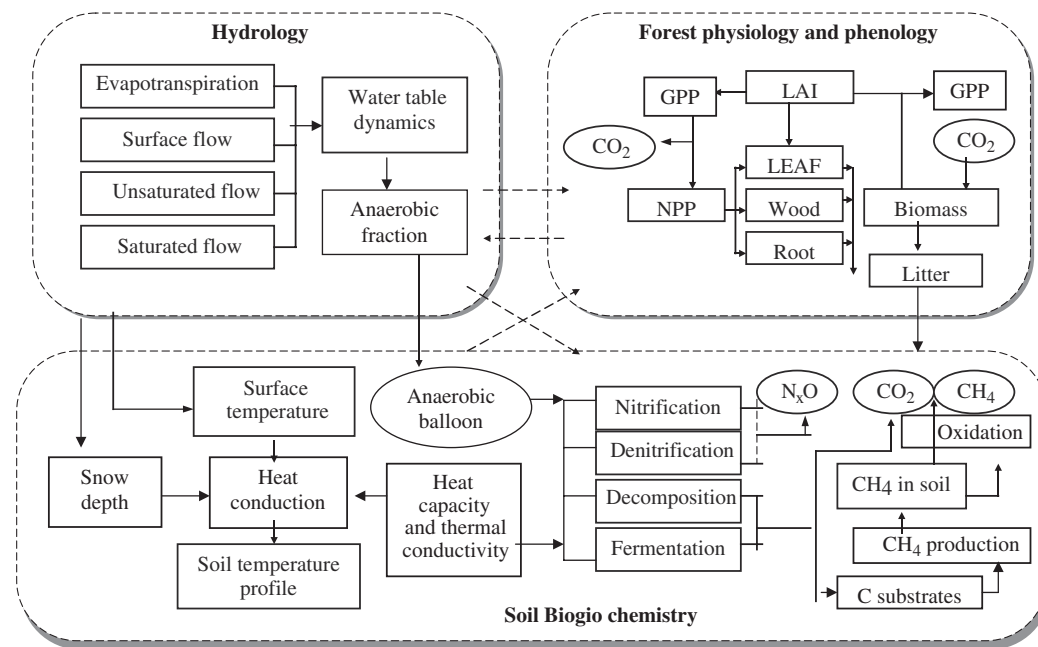


Fig. 1 Wetland-DNDC was constructed by integration three sub-models, namely, the forest physiology and phenology sub-model, the soil biogeochemical sub-model, and hydrological sub-model.

Federer, 1992; Aber *et al.*, 1996). Foliar N determines A_{\max} , which is then separated into potential gross photosynthesis and dark respiration. Potential gross photosynthesis is reduced for suboptimal conditions of light, temperature, and vapor pressure deficit (VPD) to give realized gross photosynthesis. Light levels, in the simulated layered canopy, are determined by ambient photosynthetically active radiation (PAR), cumulative leaf area index (LAI) and the light attenuation constant. Respiration is modified by temperature using a Q_{10} function. Basal respiration of foliage is assumed to be 10% of the maximum net photosynthesis rate, so $Gross\ Psn_{\max} = 1.1\ NetPsn_{\max}$. Maximum gross photosynthesis per unit leaf area is modified for suboptimal environmental conditions. NEP is the difference between the gross input of carbon in photosynthesis, GPP, and the sum of the losses of carbon in autotrophic respiration (RA) and heterotrophic respiration (RH). Autotrophic respiration was the sum of maintenance and growth respiration of the different parts of the plant (canopy, stem, roots). Maintenance respiration of each plant compartment was computed as a function of the compartment's N content and temperature. Actual evapotranspiration (AET) was estimated as the daily sum of plant transpiration and evaporation from soil and canopy. The Penmon-Monteith equation (Monteith, 1973) was used to calculate both the evaporation and transpiration rates. These computations were based on absorbed photosynthetically active radiation, daily

average surface temperature, and surface resistances to the transport of sensible heat and water vapor, respectively.

Wetland-DNDC calculates rates of litterfall and root slough and the fraction of litter and dead root components that are decomposable or resistant. Decomposable and resistant litter pools described in Wetland-DNDC are transferred to the soluble, cellulose and lignin input pools. Following decomposition of above-ground litter, and thus some losses of C because of respiration, decomposition products enter the humus pool (equations see Li *et al.*, 1992, 2000). Decomposition, and associated C loss because of respiration, is predicted using the algorithms within DNDC (Li *et al.*, 1992, 2000); it includes multiple soil layers, moisture, and temperature controls on organic matter turnover, provisions for methane emission, and a nitrogen submodel. Carbon entering the humus, decomposable, and resistant pools may be transferred between the humus and biomass pools, upon which some C is lost because of microbial respiration. Heterotrophic respiration included decomposition of both litter and soil and was related to their chemical composition, to their C:N ratios, to soil mineral N availability, and to soil moisture and temperature. Thus, Wetland-DNDC was able to simulate the effects of a number of abiotic (temperature, soil water, solar radiation, atmospheric CO_2 concentration, and atmospheric N deposition) and biotic (LAI, soil C and N contents) controls on net C flux.

Hydrologic control is the principal factor that differentiates uplands from wetlands. With respect to forested wetland ecosystem simulations, the model has been corroborated for a hydrological and anaerobic C cycle components. Improvements over the existing models include: (1) anaerobic conditions for different soil layers are taken into account; (2) redox potential (E_h) is used to control the decomposition and methane emission factor. If a soil is flooded, the E_h value will decrease because of depletion of oxides in the soil. Wetland-DNDC regulates the E_h decrease rate for each layer based on its depth, temperature and organic matter content, as well as flooding duration and plant aerenchyma development. When E_h is lower than -150 mv, CH_4 will be produced in the soil layer. The oxidation rate is regulated by the E_h value at the layer. CH_4 is allowed to diffuse between layers based on the concentration gradients. Two classical equations, the Nernst equation and the Michaelis–Menten equation, were integrated into the model algorithm (Li *et al.*, 2004), with a simple kinetic scheme, which is defined as an anaerobic volumetric fraction of a soil profile, used to link these two equations. The Nernst equation calculates soil E_h based on the concentrations of dominant oxidants and reductants. Wetland soil is divided into two parts: anaerobic microsites within the anaerobic volumetric fraction and aerobic microsites outside of it. Wetland-DNDC allocates the substrates (e.g., DOC, NO_3^- , NH_4^+ , etc.) into the aerobic and anaerobic microsites based on the size proportion. The substrates allocated within the anaerobic volumetric fraction were involved in the reductive reaction (e.g. denitrification, methanogenesis), others participated in the oxidation (e.g. nitrification, methanotrophy). The rates of the reactions occurring within and outside of the anaerobic volumetric fraction were determined by the Michaelis–Menten equation. By tracking the formation and deflation of a series of anaerobic volumetric fraction driven by depletions of oxygen, NO_3^- , Mn^{4+} , Fe^{3+} , and SO_4^{2-} , Wetland-DNDC is able to quantify soil E_h dynamics as well as net production of N_2O or CH_4 under fluctuated soil water status.

In the latest version of Wetland-DNDC, most simulated ecosystem activity occurs at a daily time step, driven by daily values for maximum and minimum temperatures, precipitation, and solar radiation. Examples of processes assessed daily are soil water balance, photosynthesis, allocation, litterfall, and C and N dynamics in the litter and soil. The model is designed to require only standard meteorological data, i.e. daily maximum–minimum temperature, precipitation, and solar radiation, so that the model may be applied beyond those sites with sophisticated instrumentation.

Research sites and model parameterizations

The Wetland-DNDC was implemented at the two sites to explore possible controls on site carbon budgets and to help explain the observed and modeled differences in hydrological dynamics and carbon storage and carbon fluxes. Two forested wetland sites were selected where extensive measurements are available. The first site is MEF-Bog (MN), which is a forested bog located at the Marcell Experimental Forest in Minnesota. MEF-Bog is completely forested with black spruce (*Picea mariana*), and the surface is covered by mosses (*Sphagnum* spp.) and sedge species (*Carex* spp.). MEF-Bog has a perched water table above the regional aquifer, and has been monitored since 1961. The second site is the Gator National Forest (FL), in Florida. FL site is located 15 km north of Gainesville. Topographical slopes range from 0% to 1.6%. The average annual temperature is 21 °C. Average annual rainfall is 1330 mm. The dominant tree canopy was slash pine with an understory of saw palmetto. Detailed descriptions of the site can be found in Shurpali *et al.* (1993, 1995). Both CO_2 and CH_4 fluxes were measured by the eddy covariance technique (Shurpali *et al.*, 1993, 1995; Suyker *et al.*, 1996, 1997; Newcomer *et al.*, 1999).

Wetland-DNDC was initialized for MN and FL sites with ecophysiological parameters (Table 1) and with site-specific sets of parameters, including geographic coordinates, atmospheric N deposition, and soil texture and depth. In this study, the simulated forested wetlands differed by climate, forest age, soil depth, and soil texture, as well as by amount of atmospheric N deposition. Daily surface weather variables are the primary model drivers, including maximum and minimum air temperature, precipitation. Most of the parameters required as input to the vegetation ecophysiology component of Wetland-DNDC were measured at the two sites, or could be derived from measurements at the sites. In some cases, parameters that were not measured onsite were estimated from other published observations. Given that we were interested in the simulations of a forest stand of a certain age, the tree C pools were consequently initialized with the initial values of the C pools, the model simulations were also performed for the same number of years with all C pools initialized from the steady-state condition. In order to simulate the changes in soil carbon pools during long-term period, 100-year climate scenarios were constructed by repeating the relevant 20 years meteorological data (1980–2000) at MN and FL sites. Then the differences between outputs of the two sites were analyzed.

Table 1 Values of ecophysiological parameters initialized for FL and MN sites

Parameters	Description	MN	FL
Leaf	Initial leaf biomass (kg C ha ⁻¹)	7410.6	2964.2
Wood	Initial woody biomass (kg C ha ⁻¹)	146216	162462
Root	Initial root biomass (kg C ha ⁻¹)	7410.65	1778.55
MaxL	Maximum leaf biomass (kg C ha ⁻¹)	9880.8	3705.3
MinL	Minimum leaf biomass (kg C ha ⁻¹)	7410.65	1235.11
PlantN	Initial plant N storage (kg C ha ⁻¹)	78.38	124.06
BudC	Initial available C stored in buds (kg C ha ⁻¹)	2470.2	2470.2
WoodC	Initial available C stored in woody biomass (kg C ha ⁻¹)	6175.54	6916.6
PlantC	Initial available C stored in forest (kg C ha ⁻¹)	8645.75	9368.82
Initial leaf N content %	Initial N concentration in foliage (% by weight)	1.2	1.4
$A_{\max A}$, and $A_{\max B}$	Coefficients for photosynthesis curve	9.3; 21.5	-22; 71.9
Optimum Psn temperature	Optimum temperature for photosynthesis (°C)	24	24
Minimum Psn temperature	Minimum temperature for photosynthesis (°C)	2	4
A_{\max} fraction	Daily A_{\max} as a fraction of instantaneous A_{\max}	0.76	0.76
Growth respiration fraction	Growth respiration as a fraction of gross photosynthesis	0.25	0.25
Wood maintain respiration fraction	Wood maintenance respiration as a fraction of gross photosynthesis	0.07	0.07
Root maintain respiration fraction	Root maintenance respiration as a fraction of gross photosynthesis	1	1
Light half saturation constant	Half saturation light intensity ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	200	200
Respiration Q_{10}	Effect of temperature on respiration	2	2
Canopy light attenuation k	Light attenuation constant	0.5	0.58
Water use efficiency	Water demand for producing a unit of biomass	13.9	13.9
D_{VPD1} and D_{VPD2}	Coefficients for calculating VPD	0.05, 2	0.05, 2
Max N storage	Maximum N content in forest (kg N ha ⁻¹)	200	200
Max leaf growth rate	Maximum foliage growth rate (% yr ⁻¹)	0.3	0.95
Max wood growth rate	Maximum wood growth rate (% yr ⁻¹)	0.9	1
Leaf start TDD	Accumulative TDDs for starting leaf growth	250	400
Wood start TDD	Accumulative TDDs for starting wood growth	250	400
Leaf end TDD	Accumulative TDDs for ceasing leaf growth	250	1300
Wood end TDD	Accumulative TDDs for ceasing wood growth	1400	1300
Leaf N retranslocation	Fraction of leaf N transferred to plant N storage during senescence	0.5	0.15
Senescence start day	Starting Julian day for senescence	270	260
Leaf C/N	C/N ratio in foliage	37	30
Wood C/N	C/N ratio in woody biomass	200	100
Leaf retention (years)	Time span of leaf retention, years	4	1.5
C reserve fraction	Fraction of available C for plant reserve	0.75	0.75
C fraction of dry matter	C/dry matter ratio	0.45	0.45
Specific leaf weight (g m ⁻²)	Specific leaf weight (g dry matter m ⁻² leaf)	170	200
Min wood/leaf	Minimum wood/leaf ratio	1.25	1.4
Leaf geometry	Leaf geometry index	1	2

TDD, thermal degree day; VPD, vapor pressure deficit.

Characteristics of forested wetland at MN and FL

Forested wetlands at MN and FL are quite varied in physical and chemical properties, geomorphic settings, and hydrologic regimes. Correspondingly, biogeochemical processes and many of the associated functions vary. A significant variation, especially the water table fluctuations associated with the hydro period of the

wetland during the dry and/or wet seasons, can result in biogeochemical processes changes.

Hydrologic characteristics of MN and FL wetlands

Hydrologic factors such as orientation, surrounding soil characteristics, storm characteristics, adjacent land use patterns, and man-made alterations (such as land use

changes) affect wetland hydrology. In addition to physical shape and form, major factors that influence the hydrology of forested wetlands are precipitation, surface water inflows and outflows, groundwater exchange, evapotranspiration as well as flood hydrographs, water level fluctuations and hydroperiods related to the meteorological conditions of each site.

Precipitation inputs to wetlands exhibit extreme spatial and temporal variability, even over small areas during a single storm event. The Wetland-DNDC model generates appropriate responses of the wetland water levels to variations in intensity and time of rainfalls. Figure 2 presents the response of the wetland water levels to many single light rainfall events occurring in FL and MN sites. The water level rose rapidly after a rainfall with intensity so large that it exceeded the storage and/or the infiltration capacity of the soil. This is shown as an example in Fig. 2, where it is noted that on DOY 303 at FL, daily rainfalls of 94 mm generated a commensurate rise of 16 cm in the water level. Gradual percolation causes a regulating effect on wetlands and its hydroperiod. In a flood event, the runoff rate drastically increases when water levels exceed a system's normal barriers to flow. Evapotranspiration combines process of evaporation from vegetation, land, water surface, and transpiration by plants. It depends on its microclimate (relative humidity, air and soil temperature, wind velocity and its duration), the soil moisture content and the type and density of the vegetation. All the processes above were fairly integrated in the Wetland-DNDC model. Analysis of the modeled and observed daily average ET showed that the Wetland-DNDC model explained 78–88% of variance in the measured water fluxes. In general, the simulated water levels by Wetland-DNDC compared favorably with the observed water levels in FL and MN sites. The important hydrological processes involved in the wetland water balance are represented well by the model. Figure 2 shows the simulated and measured water level variations in FL and MN wetlands for the period from 1986. The water level fluctuations over entire year are reproduced. The scatter plot of the calculated vs. measured water levels indicates a high R^2 value ($R^2 = 0.884$ at FL and 0.786 at MN sites), and a general clustering of the data about the 1:1 line but with some slight tendency for underestimation of water level. Simulation results can and do provide insight into the effect of climate change on the wetland water levels. Two factors may have contributed to the poor match in peaks of water levels. Firstly, the simulation assumes the snowfall to be distributed uniformly throughout the contributing watershed (for the MN case). The redistribution of snow into the wetland as a result of wind is not considered in the model. Secondly, meteorological

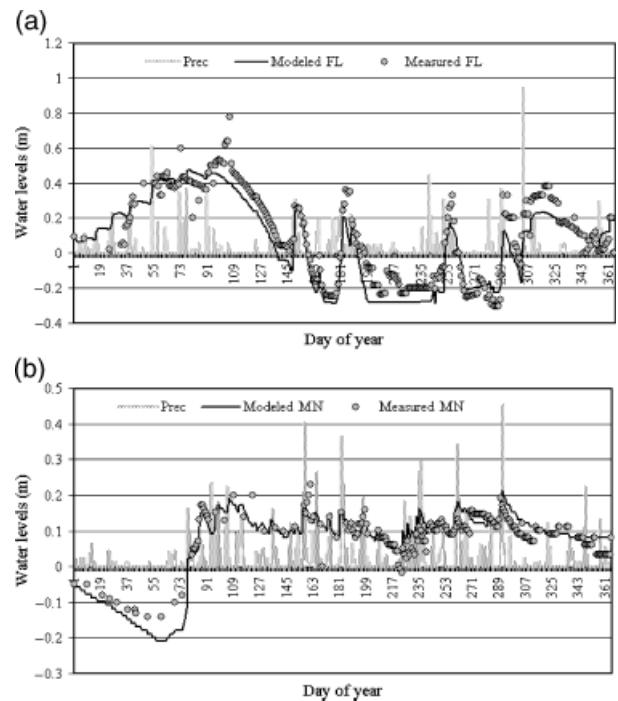


Fig. 2 Simulated (lines) and measured (symbols) water level variations in (a) FL and (b) MN wetlands. The modeled response of (i) the wetland water levels to many single light rainfall events occurring during the summer (ii) water level rose rapidly after a rainfall with intensity.

conditions collected likely differed on many occasions from those occurring at both of the FL and MN sites, particularly for summer precipitation events that occurred during convective storms, because these tend to be localized.

Carbon cycling in MN and FL forested wetland

Carbon is contained in the standing vegetations and in litter, peats, organic soils, and sediments. The magnitude of storage depends upon wetland type and size, vegetation, the depth of wetland soils, ground water levels, nutrient levels, pH and other factors. The net carbon sequestering vs. carbon release roles of forest wetlands are complex. Wetlands often provide longer-term carbon storage when it is saturated, high acidity, and low temperatures hinder decomposition processes.

Carbon storage by plants at FL and MN. The processes of photosynthesis and respiration are functions of several environmental and plant variables, including solar radiation, air and soil temperature and humidity, availability of water and nutrients, leaf area, and foliar nutrition. Productivity among wetland forest types varies widely, reflecting differences in climate,

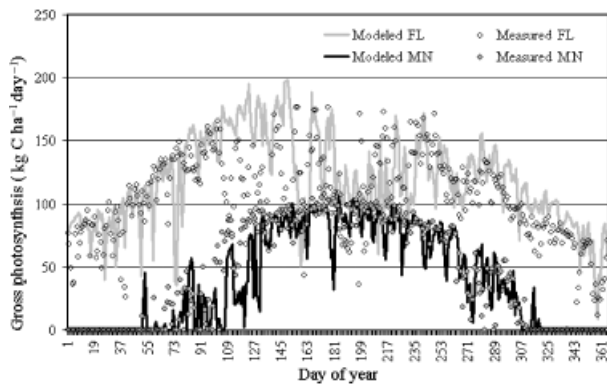


Fig. 3 Simulated gross photosynthesis (lines) at FL and MN were within the standard error of measured (symbols). Linear regression analysis of modeled and observed daily average NEP showed a good relation.

hydrology, and vegetation communities. Figure 3 shows that simulated gross photosynthesis at FL and MN, which were strongly affected by temperature. In carrying out photosynthesis, wetland vegetations convert atmospheric carbon dioxide into biomass. Trees and other vegetation grow quickly in FL wetland with ample sunlight, nutrients, water and warm temperatures. In contrast, the growth of trees and other vegetation is slow for high latitude wetlands (e.g. MN) with colder temperatures. NEP is a measure of the production of the total ecosystem and is equivalent to net primary production (NPP) minus soil microbial respiration. Simulation shows that daily NEP modeled with Wetland-DNDC at FL and MN sites gave the closest estimate of the observed NEP. Linear regression analysis of modeled and observed daily average NEP showed a good relation for the two sites, with $R^2 = 0.683$ at FL and 0.808 at MN. Despite the significantly different LAI, annual mean NEP did not vary as much between the two sites, because forests with a higher LAI also generate more litter and consequently have higher respiration rates. Minnesota had a considerably lower LAI and NEP than Florida because of its lower mean annual temperatures.

Soil carbon dynamics at FL and MN. The critical characteristic of forested wetland is the anoxic soil aeration regime. Soil moisture conditions control the rate of oxygen diffusion into the soil, hence flooding or prolonged saturation tends to increase the reduction capacity of the soil. Oxidation and reduction reactions correspond to the aeration regime, and those reactions control chemical processes that affect important wetland functions.

The Wetland-DNDC model was sensitive to inputs of turnover fractions of tree components. Although

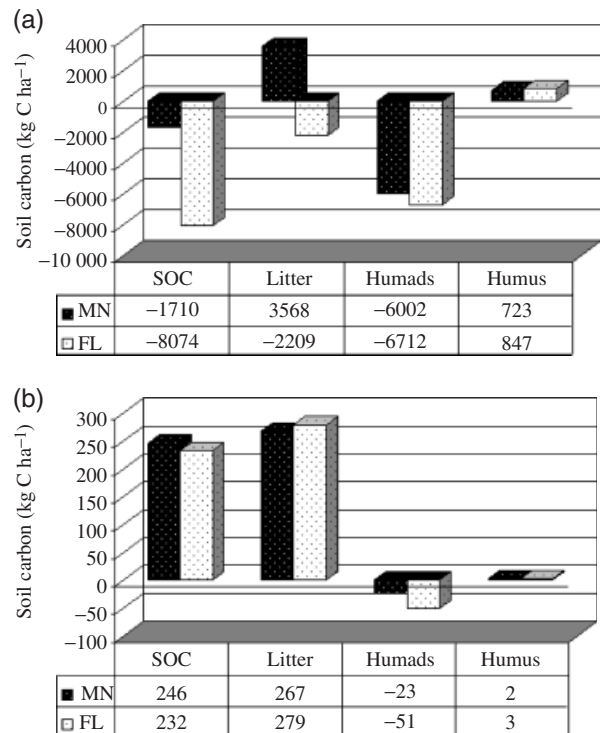


Fig. 4 Annual average soil C changes simulated by Wetland-DNDC during 100-year runs, (a) changes of soil carbon pools on forest floor, and (b) changes of soil carbon pools on the mineral soil.

increased leaf and coarse root turnover generally increased the predicted stock of soil C, increased leaf turnover in the FL case study resulted in a decrease in soil C stocks. This was because during long rotations where allocation of NPP to leaves was relatively low (<10%), an increase in leaf turnover eventually resulted in decreased leaf area and ultimately a decrease in NPP. For material with a given turnover rate, C content, and decomposability, Wetland-DNDC predicts about 16% more C is respired during decomposition when this material is treated as dead roots rather than above-ground litter. Figure 4 shows the annually average soil C changes at FL and MN simulated by Wetland-DNDC during 100-year runs. In FL, average decrease of SOC at forest floor pool was 78% greater than MN and at mineral soil pool SOC was increased 246 and 232 kg C ha⁻¹ yr⁻¹ at MN and FL, respectively. Average decrease of Humads at forest floor pool was 6002 kg C ha⁻¹ yr⁻¹ at MN and 6712 kg C ha⁻¹ yr⁻¹ at FL. The hydric soil conditions, active anaerobic microbial populations, and adapted vegetation interact affect the distribution and amount of soil C, the pathways of C fluxes from the soil, and the rates of transfer (Trettin *et al.*, 1996).

In addition to having a high sensitivity, the large range of observed values for soil C under forest wet-

land makes initial soil C one of the most likely causes of uncertainty in model predictions. Furthermore, historical measurements of soil C usually do not include estimates of inert C, and thus uncertainty in the size of this pool is also relatively high. Site-specific information about the anthropogenic factors influencing ecosystems would improve current predictions of soil C dynamics. At the two sites, predicted change in soil C was also generally insensitive to the initial mass of tree components, and the initial mass of C within the plant debris or microbial pools. Also, an increase in C accumulation in aboveground biomass does not necessarily mean a long-term gain in soil C pools.

Analysis from MN and FL wetland sites showed that varying the humification rate substantially affected the predicted change in soil C. Uncertainty in parameter values affecting microbial assimilation efficiency (the fraction of C lost to the atmosphere during decomposition) during both belowground and aboveground decomposition, also significantly affected the predicted change in soil C. One of the consequences of anoxic soil conditions in wetlands is that the rate of organic matter decomposition is reduced. Predicted change in soil C was particularly sensitive to the parameters in Wetland-DNDC defining decomposition of soil SOC, Litter, Humads, and Humus pools. The evaporation factor used in Wetland-DNDC was also important in the calculated change of soil C. In these case studies, predicted changes in soil C were also significantly affected by the clay content of soil because this affects microbial assimilation efficiency, decomposition rates of pools and soil moisture deficit.

The annual C balance in wetlands is sensitive to minor changes in climatic conditions that alter the hydrologic regime. Temperature affects the rate of a number of processes that in turn affect soil C dynamics. The higher temperature at FL site caused a decrease in SOC ($8074 \text{ kg C ha}^{-1} \text{ yr}^{-1}$) because of a greater proportion of C was respired during decomposition of soil pools. Radiation extinction coefficient for absorption also had a significant influence on predicted change in soil C. Increasing the amount of solar radiation increased amounts of soil C. However, given that air temperature is positively correlated to solar radiation, the resultant increase in the amount of C transferred to soil is partly negated by increased decomposition. For a given rainfall, increasing evaporation leads to an increase in soil C stocks because the rate of decomposition of soil C is decreased. Soil C balance is thought to be a function of overall site productivity; hence any changes in soil productivity would affect soil C pools.

Carbon losses as CO₂ and CH₄ at FL and MN. Carbon dioxide and methane are the end products of organic

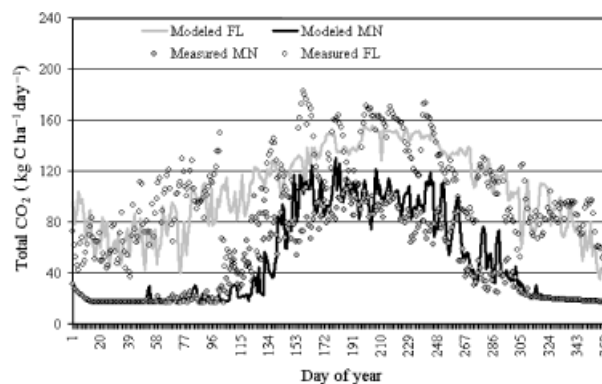


Fig. 5 Modeled carbon dioxide emission (lines) was reasonable agreement with carbon dioxide measured (symbols) at the FL and MN sites.

matter decomposition in forested wetland soils. CO₂ emission rates from forested wetlands reflect variations in hydrology, temperature, and fertility regimes. To estimate losses of gaseous CO₂ from soil C pools, Wetland-DNDC partitioned soil microbial respiration from root respiration. Figure 5 shows that simulated total carbon dioxide emission was within the standard error of carbon dioxide measured at the FL and MN sites. Total carbon dioxide emission measured and simulated varied daily from annually average influxes of $105 \text{ kg C ha}^{-1} \text{ day}^{-1}$ at FL to $50.4 \text{ kg C ha}^{-1} \text{ day}^{-1}$ at MN, standard deviation (SD) of simulated vs. measured CO₂ fluxes = $7.6 \text{ kg C ha}^{-1} \text{ day}^{-1}$. The larger influxes simulated during DOY 173 at MN, DOY 198 at FL were consistent with the higher radiation and temperatures recorded on this date (Fig. 5). Daily CO₂ effluxes remained low during winter in the model (for the MN case). Total CO₂ effluxes at FL in March and April that were consistent with mean values over MN of $63.2 \text{ kg C ha}^{-1} \text{ day}^{-1}$ in March and $73 \text{ kg C ha}^{-1} \text{ day}^{-1}$ in April. The water table is another principal factor affecting CO₂ fluxes from wetlands, which have shown a strong positive relationship between CO₂ fluxes and water-table depth. Wetlands may supply large amounts of carbon to the atmosphere if water levels are lowered result in oxidation of soils.

Methane is another source of gaseous loss of C from wetland soils. In wetland soils, varied and fluctuating water levels cut off the oxygen supply from the atmosphere, resulting in anaerobic fermentation of soil organic matter. Methane, a major end product of anaerobic fermentation, is released from submerged soils. The factors controlling CH₄ emissions are soil redox conditions, pH, the availability of readily mineralizable substrate, and temperature. Methane fluxes vary considerably over the sites, with summer exhibiting

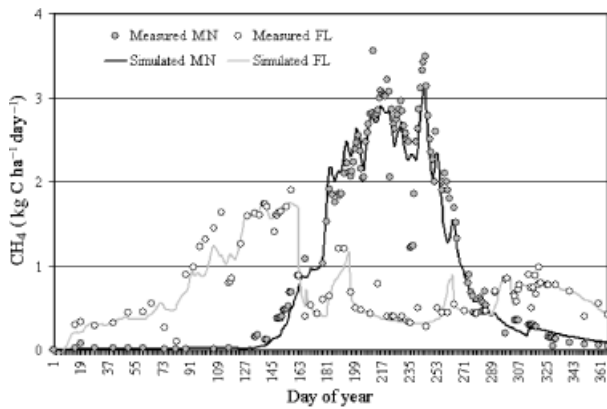


Fig. 6 Comparison of methane fluxes between FL and MN sites. Methane fluxes vary considerably over the FL and MN sites, with summer exhibiting the great emissions. Modeled Methane fluxes were reasonable agreement with simulated at the two sites.

the greatest losses (Fig. 6). The highest methane emissions, up to $3.5 \text{ kg C ha}^{-1} \text{ day}^{-1}$, were simulated at sites MN in the middle of August during a warm and dry period with the lowest WT. Snowpack and cold temperatures had been assumed to decrease CH_4 losses in MN wetland. CH_4 emissions from wetlands are sensitive to water-table depth. Flooding drastically reduces the diffusion of atmospheric oxygen into the soil, and facultative and anaerobic microorganisms sequentially reduce soil substrates. The redox potential is a quantitative indicator measuring the tendencies of different oxidations and reductions to occur. Redox potential characterizes the processes that bring about a given chemical and biochemical milieu in a soil. The higher the value of the redox potential, the greater the presence of strong oxidizing agents in a soil. The most important redox buffer system in wetland soils is comprised of iron and organic compounds. When the water table is below 15–30 cm depth, CH_4 diffuses through unsaturated soil, where it is oxidized to CO_2 . Methane production is negatively correlated with soil-redox potential and positively correlated with soil temperature, soil carbon content.

Conclusion and discussions

The biogeochemical model, Wetland-DNDC, used here brought advances to the simulation of wetland forests productivity, carbon and hydrologic dynamics in different forested wetland sites FL and MN. This model has allowed the interactive effects of climatic factors, nutrient and water status on carbon dynamics to be simulated and so improves confidence in projections of forested wetlands processes during climate change. In

this study, we evaluated the ability of the Wetland-DNDC model to estimate the hydrologic and C dynamics of wetland forests at FL and MN, compares in these dynamics in response to different environmental conditions at different sites. There is clearly considerable variation in the estimates of two sites of forested wetland. The model predicted daily C and water fluxes reasonably well at two sites.

Results indicated good estimates are required for the inputs of initial C, climatic data (temperature, rainfall, solar radiation, and frost days), optimal temperature for growth, maximum available soil water content, etc. The sensitivity rank of examined parameters for the process of CO_2 emission is soil organic C, soil clay content, mean annual temperature, annual precipitation and for the process of decomposition is initial organic C, soil temperature, soil moisture, dry period duration, respectively. Predicted change in soil C was most sensitive to parameters determining the fraction of the various pools of litter that transfer to the soil each month, parameters used in the Wetland-DNDC sub-model to predict NPP, decomposition of C within pools of litter and soil, and the C fraction of tree components are sensitive for prediction. Identifying the most important inputs and parameters in the model affecting predicted change in soil C has been valuable and necessary to most efficiently target further calibration and verification needs of Wetland-DNDC, and to further its application as a predictive tool for any given set of site, species, and management conditions. Clearly, it is most important to decrease the uncertainty in the values of parameters to which the model is most sensitive. Therefore, the highest priority for calibration of parameters used in Wetland-DNDC is to decrease the uncertainty in the values of parameters used to describe decomposition of pools of C within litter, dead roots and soil under forest wetland systems.

Wetlands have been considered as rather different ecosystems and the attempts to model their C dynamics have used very different approaches. Part of the distinction lies in the need to cope with anaerobic decomposition. More accurate estimates of the rates of carbon loss and accumulation are necessary. Calibration of decomposition rates of litter and soil pools can be achieved although collating data on mass loss from litterbag studies in the different sites, and on litter accumulation and change in soil C observed in long-term field studies. In particular, a tighter control should be made on the estimated gains and losses from soil organic matter at the respective 'slow' and 'fast' rates. These values should be based upon appropriate and well-researched studies. The limited evidence available would suggest that the 'fast' loss rate is too fast. Also, it seems likely that the 'one size fits all' may not be

appropriate and that different soils will gain or lose C at different rates. Simultaneous measurements of C pools of the forest wetland ecosystems will help to refine model algorithms, and eventually reduce uncertainty in the C budget for the further application of the model. A more accurate assessment is required of the spatial distribution of carbon stored in organic soils in forested wetland. This is in terms of organic soils as C pool and as a potential CO₂ source following land-use change. This requires reevaluation of the areas, depths, bulk densities, and C contents of wetlands. New algorithms, which allow real-time adjustment in the model's state variables using on-ground and remote observations of soil information, vegetation structure and processes, should reduce uncertainty in the model estimates and improve its accuracy. Further work is also required to explore the mechanism of relationship between nutrients release. For instance, phosphorous is another nutrient to be considered in the freshwater systems of the Everglades. It is suggested that elevated P levels increase the size and activity of the microbial biomass in soils. Accelerated activity of microbes in P enriched areas resulted in rapid turnover of organic matter and resulted in release of bioavailable nutrients. These results suggest the microbial pool was responsible for liberating greater amounts of inorganic N, which can increase macrophyte growth and spread. This mechanism of N release is likely to continue to spread as P released from the most nutrient-rich areas is carried in surface waters further down the hydrologic gradient. Further studies to clarify the role of forested wetlands as GHG emitters or fixers would allow better judgments of the effects of land-use change involving wetlands. The new algorithms for simulating the C balance over wetlands would seem to be one which could be extended to give better answers under different land use changes given its built-in integrative nature.

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Appendix A

Major functions used in Wetland-DNDC for simulating the forest physiology and phenology, biogeochemical and hydrologic processes (See Table A1).

Table A1 Major functions used in Wetland-DNDC

Function	Equation
Maximum net photosynthesis	$\text{NetPsn}_{\max} = -5.98 + 4.86 \text{ N\%}$
Maximum gross photosynthesis	$\text{GrossPsn} = \text{GrossPsn}_{\max} \times D_{\text{temp}} \times D_{\text{water}} \times D_{\text{VPD}}$
Effects of temperature	$D_{\text{temp}} = (\text{Psn}T_{\max} - T_{\text{day}})(T_{\text{day}} - \text{Psn}T_{\min}) / (\text{Psn}T_{\max} - \text{Psn}T_{\min})^2$
Effects of water availability	$D_{\text{water}} = \text{Trans}_i / \text{PotTrans}$
Effects of VPD	$D_{\text{VPD}} = \text{VPDEffK} \times \text{VPD}$
Basal respiration of foliage is assumed to be 10% of the maximum net photosynthesis rate	
Light attenuation	$I_i = I_0 \times e^{-k(\text{LAI}_i)}$
Anaerobic effects on decomposition	$f_{\text{dec}} = 0.2 + 0.05 \times \exp(E_h/250)$
E_h dynamics	$E_h = E_0 + RT/nF \times \ln([\text{oxidant}]/[\text{reductant}])$ $E_h = 0.82 + 0.015 \log[\text{O}_2]$
Effect of canopy on soil temperature	$dT(\text{soil}) = -0.0037 + 0.2422 dT(\text{air})$, coniferous forests; $dT(\text{soil}) = -0.0058 + 0.1827 dT(\text{air})$, deciduous forests;
Partitioning of fresh litter into soil litter pools	Litter C (vl) = FL_C; if FL_CN < CN(vl); Litter C (vl) = 1.25 P1 - 0.25FL_C; if CN(vl) <= FL_CN < CN(l); Litter C (l) = FL_C - P1; Litter C (r) = 0.25 (FL_C - P1); P1 = FL_C (K1 - FL_C) (K1 - CN (vl) CN(vl)/FL_CN); K1 = 2 CN(l) CN(r)/(CN(l) + CN(r)); Litter C (vl) = 0.25 (FL_C - P2); if CN(l) <= FL_CN < CN(r); Litter C (l) = FL_C - P2; Litter C (r) = 1.25P2 - 0.25 FL_C; P2 = FL_C (K2 - FL_C) (K2 - CN (r) CN (r)/FL_CN); K2 = 2 CN(vl) CN(l)/(CN(vl) + CN(l)); Litter C (r) = FL_C; if FL_CN >= CN(r);
Oxygen diffusion coefficient in soil	$D_{s[\text{L}]} = D_{\text{air}} \text{afps}[\text{L}]^{3.33} / \text{afps}_{\max[\text{L}]}^{2.0}$

(Continued)

Table A1 (Contd.)

Function	Equation
Oxygen diffusion rate affected by frost	$D_{s[L]} = D_{s[L]} F_{\text{frost}}; 0 < D_{s[L]} < 1$ $F_{\text{frost}} = 1.2; \text{ if } T > 0^\circ\text{C}$ $F_{\text{frost}} = 0.8; \text{ if } T \leq 0^\circ\text{C}$
Oxygen partial pressure	$d(pO_{2[L]})/dt = (d(D_{s[L]} d(pO_{2[L]})/dz)dz - R)/afps;$
Volumetric fraction of anaerobic microsities	$anvf_{[L]} = a(1 - (b pO_{2[L]}/pO_{2\text{air}}));$
Relative growth rate of nitrifiers	$u_g = u_{\text{max}} ([\text{DOC}]/(1 + [\text{DOC}]) + F_m/(1 + F_m));$
Relative death rate of nitrifiers	$u_d = a_{\text{max}} B_n/(5 + [\text{DOC}])/(1 + F_m);$
Net increase in nitrifiers biomass	$u_b = u_g - u_d B_n F_t F_m;$
Nitrification rate	$R_n = R_{\text{max}}[\text{NH}_4]B_n \text{ pH};$
Temperature factor	$F_t = ((60 - T)/25.78)^{3.503} e^{(3.503(T - 34.22)/25.78)};$
Moisture factor	$F_m = 1.01 - 0.21 \text{ wfps}; \text{ if } \text{wfps} > 0.05$ $F_m = 0; \text{ if } \text{wfps} \leq 0.05$
NO production from nitrification	$\text{NO} = 0.0025 R_n F_t;$
N ₂ O production from nitrification	$\text{N}_2\text{O} = 0.0006 R_n F_t \text{ wfps};$
Relative growth rate of NO _x denitrifiers	$u_{\text{NO}_x} = u_{\text{NO}_x(\text{max})} ([\text{DOC}]/(K_c + [\text{DOC}]))([\text{NO}_x]/(K_n + [\text{NO}_x]));$
Relative growth rate of total denitrifiers	$u_g = F_t (u_{\text{NO}_3} F_{\text{pH}_1} + u_{\text{NO}_2} F_{\text{pH}_2} + u_{\text{NO}} F_{\text{pH}_2} + u_{\text{N}_2\text{O}} F_{\text{pH}_3});$ $F_t = 2^{((T - 22.5)/10)}$ $F_{\text{pH}_1} = 1 - 1/(1 + e^{(\text{pH} - 4.25)/0.5});$ $F_{\text{pH}_2} = 1 - 1/(1 + e^{(\text{pH} - 5.25)/1.0});$ $F_{\text{pH}_3} = 1 - 1/(1 + e^{(\text{pH} - 6.25)/1.5});$
Denitrifier growth rate, death rate, and consumption rate of soluble carbon	$R_g = u_g \times B_d;$ $R_d = M_c \times Y_c \times B_d;$ $R_c = (u_g/Y_c + M_c) \times B_d;$ $R_{\text{NO}_x} = (u_{\text{NO}_x}/Y_{\text{NO}_x} + M_{\text{NO}_x} [\text{NO}_x]/[\text{N}])B_d;$
Consumption rates of N oxides	$q_N = R_g/\text{CN};$
Nitrogen assimilation rate	$v = D_{\text{max}} afps (1 - anvf) F_{\text{clay}} 2^T/20;$
Ges diffusion factor	$F_{\text{clay}} = 0.13 - 0.079 \text{ clay};$
CH ₄ content of each soil layer (ΔM)	$\Delta M = M_{\text{PRD}}M_{\text{OXD}}M_{\text{DFS}}M_{\text{EBL}}M_{\text{PLT}}$
Microbial biomass death rate because of freezing	$d(\text{micro})/dt = 0.001 \text{ ASOC RBO}; \text{ if } T < 0^\circ\text{C}$
Overall water balance for each soil layer	$V_{Sj}(t) = V_{Sj}(t - 1) + I_j(t) - I_{j+1}(t) - E_j(t) - Q_j(t)$
Governing equation for the groundwater flow	$K_x \frac{\partial^2 H}{\partial x^2} + K_y \frac{\partial^2 H}{\partial y^2} + K_z \frac{\partial^2 H}{\partial z^2} = S_s \frac{\partial H}{\partial t}$

Notation: NetPsn_{max}, maximum net photosynthesis; Gross Psn_{max}, maximum gross photosynthesis; D_{temp} , D_{water} , D_{VPD} vary between 0 and 1 and express the effects of temperature, water availability and vapor pressure deficit; $\text{Psn}T_{\text{max}}$ and $\text{Psn}T_{\text{min}}$ are the maximum and minimum temperatures; VPDefk , system-specific constant; $afps$, air-filled porosity; $afps_{\text{max}}$, porosity; $anvf$, volumetric fraction of anaerobic microsities; D_{air} , oxygen diffusion rate in the air; D_s , oxygen diffusion coefficient in soil; F_{frost} , frost factor; L , layer number; pO_2 , oxygen partial pressure; R , oxygen consumption rate ($\text{kg C ha}^{-1} \text{ h}^{-1}$); t , time (h); z , soil depth (m); a_{max} , maximum death rate for nitrifiers; B_n , biomass of nitrifiers (kg C ha^{-1}); $[\text{DOC}]$, concentration of dissolved organic C (kg C ha^{-1}); $[\text{NH}_4]$, concentration of ammonium (kg C ha^{-1}); NO , NO production from nitrification; N_2O , N_2O production from nitrification; pH , soil pH; R_n , nitrification rate; R_{max} , maximum nitrification rate (1 h^{-1}); T , soil temperature ($^\circ\text{C}$); wfps , water-filled porosity; u_{max} , maximum growth rate for nitrifiers; u_b , net increase in nitrifiers biomass; u_d , relative death rate of nitrifier; u_g , relative growth rate of nitrifiers; B_d , denitrifier biomass (kg C m^{-3}); clay , clay fraction in the soil; CN , C/N ratio in denitrifiers; D_c , consumption rate of soluble carbon by denitrifiers ($\text{kg C m}^{-3} \text{ h}^{-1}$); D_{max} , maximum diffusion rate in air ($\text{m}^2 \text{ h}^{-1}$); D_{NO_x} , consumption rate of N oxides by denitrifiers ($\text{kg C m}^{-3} \text{ h}^{-1}$); $[\text{DOC}]$, soluble C concentration (kg C m^{-3}); K_c , half-saturation value of soluble carbon; K_n , half-saturation value of N oxides; M_c , maintenance coefficient on carbon; $[\text{N}]$, concentration of all NO_x (kg N m^{-3}); $[\text{NO}_x]$, concentration of NO_3^- , NO_2^- and N_2O (kg N m^{-3}); q_n , nitrogen assimilation rate ($\text{kg N ha}^{-1} \text{ h}^{-1}$); v , gas diffusion factor (%); Y_c , maximum growth rate of denitrifiers on soluble carbon; M_{NO_x} , maintenance coefficient on N oxides; R_d , denitrifier death rate; R_g , denitrifier growth rate; Y_{NO_x} , maximum growth rate on N oxides; u_{NO_3} , u_{NO_2} , u_{NO} , $u_{\text{N}_2\text{O}}$, relative growth rate of NO_3^- , NO_2^- and N_2O denitrifiers; ASOC , active soil organic carbon (kg C ha^{-1}); $\text{CN}(\text{vl})$, C/N ratio of very labile litter pool; $\text{CN}(\text{l})$, C/N ratio of labile litter pool; $\text{CN}(\text{r})$, C/N ratio of resistant litter pool; $dT(\text{soil})$, daily increase in surface soil temperature ($^\circ\text{C}$); $dT(\text{air})$, daily increase in air temperature ($^\circ\text{C}$); FL_C , fresh litter C content (kg C ha^{-1}); FL_CN , fresh litter C/N ratio; $\text{LitterC}(\text{vl})$, organic C content in very labile litter pool (kg C ha^{-1}); micro , death rate of microbes because of freezing; RBO , microbial fraction of active organic carbon.