Biogeochemistry (2005) 72: 147–167
DOI 10.1007/s10533-004-0367-8

Linkage of MIKE SHE to Wetland-DNDC for carbon budgeting and anaerobic biogeochemistry simulation

JIANBO CUI[1,]*, CHANGSHENG LI[1], GE SUN[2] and CARL TRETTIN[3]
*[1]Institute for the Study of Earth, Oceans, and Space, University of New Hampshire, Durham, NH 03824, USA; [2]Department of Forestry, North Carolina State University, Raleigh, NC 27606, USA; [3]USDA, Forest Service, Center for Forested Wetlands Research, Charleston, SC 29414, USA; *Author for correspondence. Present address: Ecological Modelling and Carbon Science Laboratory, Institut des sciences de l'environnement, Université du Qubéc à Montréal, Case Postale 8888, succursale Centre-Ville Montreal (Quebec), Canada H3C 3P8 (e-mail: jcui@eos.sr.unh.edu; phone: (514)-987-4717; fax: (514)-987-4718)*

Received 12 December 2003; accepted in revised form 23 March 2004

Key words: Biogeochemical modeling, Carbon dynamics, Forest wetland, Greenhouse gases emission, Mitigation strategies

Abstract. This study reports the linkage between MIKE SHE and Wetland-DNDC for carbon dynamics and greenhouse gases (GHGs) emissions simulation in forested wetland.Wetland-DNDC was modified by parameterizing management measures, refining anaerobic biogeochemical processes, and was linked to the hydrological model – MIKE SHE. As a preliminary application, we simulated the effect of water table position and forest management practices on GHGs emissions and carbon dynamics to test the capabilities of the models for simulating seasonal and long-term carbon budget. Simulation results show that water table changes had a remarkable effect on GHGs fluxes. Anaerobic conditions in forested wetland soils reduce organic matter decomposition and stimulate CH_4 production. Decrease in the water table from the wetland surface decreases methane flux, while CO_2 emission was lower with a rise in the water table. When there is a drop in water availability, wetlands can become a net source of atmospheric CO_2 as photosynthesis is decreased and respiration loss enhanced. Forest management activities i.e. harvest, fertilization and reforestation practices were parameterized in the model. We predicted carbon fluxes and stores on a pine forest under different forest management scenarios during 160 years. Results show that average long-term carbon storage in ecosystem pools increased with increasing rotation length; Soil carbon showed only minor, long-term responses to harvesting events. In contrast, carbon sequestered in tree biomass and litter fluctuated widely, in concert with the harvest cycle. Application of nitrogen fertilizer increased average carbon storage in all ecosystem pools and wood products. We presented the linkage of MIKE SHE and Wetland-DNDC as a way to use of simulation modeling tools for assessing GHGs mitigation strategies, carbon budgeting and forest management.

Introduction

Increasing emissions of carbon dioxide, methane, and other greenhouse gases (GHGs) are believed to contribute to global warming. The forested wetlands can offer a number of options for reducing GHGs, particularly C emissions. It serves as the removal of CO_2 from the atmosphere into carbon pools, which can be living, aboveground biomass, living biomass or recalcitrant organic and inorganic carbon in soils and deeper subsurface environments. In forested wetlands, dynamics of carbon fluxes are affected by complex interactions between abiotic and biotic environmental factors and actual processes, e.g. methane production, methane oxidation and transport from wetland to atmosphere. Studies on the carbon budget of forested wetlands to date have employed a variety of techniques (Nieveen et al. 1998; Schreader et al. 1998; Waddington and Roulet 2000; Aurela et al. 2001; Lafleur et al. 2001, 2003; Bubier et al. 2004). Mathematical models provide a powerful tool to predict long-term carbon dynamics under climate change. Many types of models have been developed and used to simulate carbon cycle on different spatial and temporal scales (Chertov 1990; De Willigen 1991; Cao et al. 1996; McClain et al. 1996; Arah and Stephen 1998; Grant 1998; Walter and Heimann 2000; Granberg et al. 2001; Segers and Leffelaar 2001a, b; Segers et al. 2001; Walter et al. 2001a, b; Frolking et al. 2002). Nevertheless, predicting the responses of environmental drivers and their effects on carbon dynamics presents a number of challenges, largely because the primary responses lead to secondary effects which form a complicated network of feedbacks and indirect responses, often operating at a number of spatial and temporal scales. For example, decomposition, CO_2, CH_4 emission relevant reactions consist of environmental forces deriving from soil temperature, moisture, pH, Eh, substrate concentration, and other soil environmental factors. Soil environmental factors are controlled by several ecological drivers including climate, soil physical properties, vegetation, and anthropogenic activities, which are different spatially and temporally. Forested wetlands are characterized by permanent or temporal flooding, by soil conditions different from those in upland soils. Simulation results indicate that forested wetlands differ in their response to changes in water-table elevation in terms of above- and below-ground net primary production, thermal energy flux and evapotranspiration (ET), atmospheric carbon flux and gross primary production, and ecosystem carbon and nutrient budgets (Weltzin et al. 2000). Water level is the major factor controlling carbon allocation, organic matter decomposition and C fluxes in wetland (Moore and Dalva 1993; Kettunen et al. 1999; Weltzin et al. 2000). The water table fluctuation determine saturated and unsaturated proportion of the wetland both spatially and temporally, thus significantly affects physical, chemical and biogeochemical processes that controlling carbon dynamics in the forested wetland (e.g. aerobic mineralization versus fermentation, reductive processes such as sulfate-, nitrate-, Mn- or Fe-reduction). Consequently, the function and element balances of forested wetland ecosystems may drastically change in response to alterations in water table. Despite the importance of forested wetland in the global carbon cycle, no widely applicable ecosystem model exists for this ecosystem. The existing models do not explicitly describe the connection between the vegetation, water table and carbon fluxes (Trettin et al. 2001). Most of the models do not take into account the dynamic effects of changes in water table patterns on GHGs fluxes.

Meanwhile, humans have the potential through forest management to change the magnitude and direction of forest carbon pools and flux, and thus alter their role in the carbon cycle and their potential to impact climate change. There is growing interest in the role that forest management practices can play in preventing global warming. Many forest management practices have been reported to enhance carbon mitigation. Studies have showed that drainage of forested wetlands can enhance tree growth significantly, but the net ecosystem carbon changes are less clear – some studies report large net gains while others indicate large net losses of carbon to the atmosphere (Silvola et al. 1996; Zoltai and Martikainen 1996; Whiting and Chanton 2001; Minkinnen et al. 2002). Less information is available on impacts of alternative practices on soil and total ecosystem carbon dynamics. In addition to climate change scenarios, the results of forest management studies are currently needed for estimation the effects of land use changes on the C balance, e.g. given the impact of harvesting on carbon storage in many forested ecosystems, there is growing concern over how present harvesting can restore and maintain a substantial carbon sink. Although the theoretical foundations for the interactions involved in forested wetland are preliminarily established, there are few opportunities to use models to simulate the long-term dynamics of carbon in forested wetlands by improved management.

The objectives of the research were to (1) modify the Wetland-DNDC model and link it to MIKE SHE for the quantification of anaerobic biogeochemical processes in forested wetland ecosystem. Model results are combined with efforts to improve estimates of carbon biomass and procuctivity in this system; (2) examine long-term ecosystem responses and GHGs emissions to water table fluctuation; and (3) evaluate the effect of forest management practices on carbon dynamics in a long-term period. Simulation results were used to suggest GHG mitigation strategies in forest wetlands, as well as to address the use of simulation modeling tools in forest management.

Methods

Model modifications

The process-based model, Wetland-DNDC, was developed and modified for predicting C dynamics in forested wetland ecosystems. This process-oriented ecosystem model simulates the flows and storage of carbon in all the vegetation and soil components of the forested wetland. Anaerobic and aerobic decomposition are computed separately, along with production, oxidation, emission of methane, and dissolved organic carbon production and loss with drainage. Some model parameters are considered as general to all wetlands (e.g. PSN and respiration functions for vegetation types), others are site specific (e.g. vegetation type, maximum leaf area index, and biomass). Water table is a key variable as it determines the relative contributions of aerobic and anaerobic decomposition, and influences most vegetation processes. So it has been given specific attention in the model. The basic functions adopted by Wetland-DNDC for simulating forest growth, soil biogeochemistry processes have been well validated in previous publications (Li et al. 1992; Aber et al. 1996; Li et al. 2000). The original version of Wetland-DNDC focused on natural wetlands with few management practices simulation included. In order to make the model capable of predicting impacts of management on water, C and N biogeochemical cycles in forested wetland ecosystems, we modified Wetland-DNDC by parameterizing management practices, refining anaerobic biogeochemical processes, and linking it to the hydrological model – MIKE SHE.

Modeling anaerobic biogeochemistry

Wetland soils, in contrast to upland soils are generally anoxic and thus support the production of trace gases (CH_4, N_2O, and NO) by anaerobic bacteria such as fermenters, methanogens, acetogens, sulfate reducers, and denitrifiers. NH_4^+ is the dominating N-species in anoxic soils. It may be oxidized to NO_2^- (and eventually to NO_3^-) not only by classical nitrifiers, but also by methanotrophs. A simplified scheme of the interactions of nitrification and denitrification in the proportions of aerobic/anaerobic microsites in soil matrix was depicted by Li et al. (2000). Methane- and ammonia-oxidizing bacteria play major roles in the carbon and nitrogen cycles. These bacteria convert the most reduced carbon and nitrogen compounds (CH_4 and NH_4^+) to oxidized forms (CO_2 and NO_2^-). Apart from their primary substrates, CH_4 and NH_4^+, both groups of bacteria need oxygen for growth. CH_4 production was largely controlled by the availability of methanogenic substrates, which in turn was under control of competing electron acceptors like iron. The temporal pattern of CH_4 oxidation was consistent with a limitation by nutrient (NH_4^+) availability. Net emission of CH_4 is determined by the balance between production and consumption. Similar interactions between anaerobic and aerobic processes apply for the cycling of reduced and oxidized species of nitrogen, iron, and others. N_2O reduction to N_2 by denitrifying bacteria; NO consumption by either reduction to N_2O in denitrifiers or oxidation to nitrate in heterotrophic bacteria.

Net increase in nitrifiers biomass:

$$u_b = u_g - u_d B_n F_t F_m,$$

where u_b, net increase in nitrifiers biomass; u_g is relative growth rate of nitrifiers: $u_g = u_{max}([DOC]/(1 + [DOC]) + F_m/(1 + F_m))$; u_d is relative death rate of nitrifiers: $u_d = a_{max} B_n/(5 + [DOC]/(1 + F_m))$; F_t is temperature factor: $F_t = ((60 - T)/25.78)^{3.503} e^{(3.503\ (T-34.22)/25.78)}$; F_m is moisture factor: $F_m = 1.01–0.21$ wfps; if wfps > 0.05; $F_m = 0$; if wfps ≤ 0.05; [DOC], concentration of dissolved organic C (kg C ha^{-1}); B_n, biomass of nitrifiers (kg C ha^{-1});

NO production from nitrification:

$$NO = 0.0025 R_n F_t,$$

N_2O production from nitrification:

$$N_2O = 0.0006 R_n F_t \text{ wfps},$$

where R_n is nitrification rate; wfps is water-filled porosity.
Relative growth rate of total denitrifiers:

$$u_g = F_t(u_{NO_3} F_{PH1} + u_{NO_2} F_{PH2} + u_{NO} F_{PH2} + u_{N_2O} F_{PH3}),$$

where relative growth rate of NO_x denitrifiers: $u_{NO_x} = u_{NO_{x(max)}}([DOC)]/(K_c + [DOC])[NO_x]/(K_n + [NO_x])$; $F_t = 2^{((T-22.5)/10)}$; $F_{PH1} = 1 - 1/(1 + e^{((PH-4.25)/0.5)})$; $F_{PH2} = 1 - 1/(1 + e^{((PH-5.25)/1.0)})$; $F_{PH3} = 1 - 1/(1 + e^{((PH-6.25)/1.5)})$; u_{NO_3}, u_{NO_2}, u_{NO}, u_{N_2O}, relative growth rate of NO_3^-, NO_2^-, NO^- and N_2O denitrifiers.

Consumption rates of N oxides:

$$R_{NO_x} = (u_{NO_x}/Y_{NO_x} + M_{NO_x}[NO_x]/[N])B_d,$$

where [N], concentration of all NO_x (kg N m^{-3}); $[NO_x]$, concentration of NO_3^-, NO_2^- and N_2O (kg N m^{-3}); Y_{NO_x}, maximum growth rate on N oxides.

Forested wetland soil is characterized by the presences of a saturated zone, which is determined by the fluctuated water table. This feature significantly affects C and N dynamics in this ecosystem. Flooding and draining practices cause dramatic changes in the soil redox potential (i.e., Eh) conditions, and hence affect production and consumption of the GHGs in the soils (Anderson and Levine 1986; Yagi and Minami 1990; Sass et al. 1991; Wassmann et al. 1993; Holland and Schimel 1994; Bollmann and Conrad 1998). In order to quantify the Eh dynamics and its impacts on N_2O or CH_4 production, we integrated two classical equations, the Nernst equation and the Michaelis–Menten equation, into the model algorithm. The Nernst equation is a basic thermodynamic formula defining soil Eh based on concentrations of the existing oxidants and reductants in the soil liquid phase (Stumm and Morgan 1981). The Michaelis–Menten equation describes kinetics of microbial growth with dual nutrients (Paul and Clark 1998). A simple kinetic scheme was adopted in Wetland-DNDC to link these two equations. The kinetic scheme is defined to be an anaerobic volumetric fraction of a soil. Based on the concentrations of dominant oxidants and reductants, the Nernst equation calculates soil Eh.

$$Eh = E_0 + RT/nF \cdot \ln([\text{oxidant}]/[\text{reductant}])$$

where Eh is redox potential of the oxidation–reduction system (V), E_0 is standard electromotive force (V), R is a constant (8.313), T is absolute temperature (273 + t, °C), n is transferred electron number, F is constant (96,500k), [oxidant] is concentration (mol/l) of dominant oxidant in the system, and [reductant] is concentration (mol/l) of dominant reductant in the system.

Based on the Eh value, the soil is divided into two parts: relatively anaerobic microsites within the anaerobic volumetric fraction and relatively aerobic microsites outside of the anaerobic volumetric fraction. Wetland-DNDC allocates the substrates (e.g., DOC, NO_3^-, NH_4^+, etc.) into the aerobic and anaerobic microsites in the soil based on the size proportion. The substrates allocated within the anaerobic volumetric fraction will be involved in the reductive reaction (e.g., denitrification, methanogenesis); and those allocated outside of the anaerobic volumetric fraction will participate in the oxidation (e.g., nitrification, methanotrophy). The Michaelis–Menten equation is used to determine the rates of the reactions occurring within and outside of the anaerobic volumetric fraction. 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 Eh dynamics as well as net production of N_2O or CH_4 under complex soil water conditions.

Parameterization of management practices
Harvest, fertilization, reforestation practices were parameterized in Wetland-DNDC to predict impacts of management practices on C dynamics and trace gas emissions. Wetland-DNDC simulates forest biomass dynamics by tracking the growth of upperstory, understory (e.g., bushes or shrubs), and ground-level vegetation (e.g., moss, herbaceous or lichens) based on their competition for light, water and N. Forest harvest in the model is defined with its timing and harvested percent of the above-ground woody biomass. When a harvest event occurs, the modeled above-ground woody mass will be removed from the ecosystem, and the litter (e.g., dead leaves, branches, and roots) is incorporated in the soil profile (i.e., the forest floor and mineral soil). The litter is partitioned into the soil organic matter (SOM) pools based on their C/N ratio values (Li et al. 2000). Reforestation is defined with its timing and the type and age of planted trees. Reforestation can be applied with or without understory component. Fertilization is defined by the timing and amount of fertilizer applied. The fertilized nutrients join the inorganic N pools to support the tree growth or soil microbial activities during the simulations. The new features for management practices have been linked to the forest growth and soil biogeochemistry sub-models originally embedded in the Wetland-DNDC model, and it effectively interacts with the water, C and N cycles in the simulated ecosystems.

Linking with hydrological model – MIKE SHE
We created a new interface for Wetland-DNDC to link it to spatial hydrological models. MIKE SHE (Refsgaard and Storm 1995), is a comprehensive, distributed, and physically based model capable of simulating both surface and ground water with precision equal to that of models focused separately on either surface water or ground water. The MIKE SHE modeling system simulates most major hydrological processes of water movement, including canopy and land surface interception after precipitation, snowmelt, evapotranspiration, overland flow, channel flow, unsaturated subsurface flow, and saturated ground water flow. A grid network represents spatial distributions of the model parameters, inputs, and results with vertical layers for each

grid. The unsaturated zone processes play a critical role in forest wetland because it couples the surface flow system to the saturated zone. Even though the unsaturated zone does not store significant volume of water, it acts as a conduit for water flow. The dynamics of how the water table responds to precipitation, evaporation, and surface flow depends on the unsaturated flow and storage processes. MIKE SHE uses the one-dimensional unsaturated flow equation (Richard's equation). Extraction of moisture for transpiration and soil evaporation is introduced via sink terms at the node points in the root zone. Infiltration rates are found by the upper boundary that may be either flux controlled or head controlled. The lowest node point included in the finite difference scheme depends on the pyretic surface level, and allowance is made for the unsaturated zone to disappear in cases where the pyretic surface rises to the ground surface. The saturated zone module of MIKE SHE simulates three-dimensional groundwater flow under both unconfined and confined conditions. An implicit finite difference scheme is used in the numerical solution of saturated groundwater flow equation (Boussinesq equation). Implicit solution schemes allow for the use of any grid size and computational time steps without affecting convergence and stability of the solution. The model runs on a daily time step for short or long-term predictions and operates in a distributed manner to account for spatial differences in soils, land use, crops, topography, channel morphology, and weather conditions.

This analysis involves a linkage between the modeling tools employed by MIKE SHE for water table dynamics analyses and Wetland-DNDC for carbon dynamics and GHGs mitigation analyses in forested wetland. The water table fluctuation results simulated from the MIKE SHE were input to Wetland-DNDC for the carbon dynamics studies. The biogeochemical processes in both anaerobic and aerobic scenarios were simulated by Wetland-DNDC.

Model verification

Study site

Data sets of water table dynamics, forest productivity, and CO_2 and CH_4 fluxes observed at wetland site in Florida (latitude 29.0°N) were used to compare our simulation results. The site is the Gator National Forest (GNF), Florida 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; dominant tree canopy was slash pine with an understory of saw palmetto. The detailed features and information about this site can be found in (Sun et al. 1998).

Daily maximum and minimum air temperatures, precipitation were derived from the continuous data records for the site and used to generate model results. Mean daily air temperature was estimated as the average of the measured daily maximum and minimum air temperatures. CO_2 and CH_4 emission were measured at approximate 15 days intervals during 1994–1996. Other measurements included aboveground biomass, species composition and soil water content. The water table has been monitored since 1991.

Water table dynamics

Daily outflows and water table depths predicted by the hydrological model (MIKE SHE) were compared with the measured data for that field to test the reliability of the model. Daily outflows were underpredicted during days 180–240 (summer 1992) and days 420–480 (spring 1993) resulting in total reduction of cumulative outflow by 9.6% for the three-year (1992–1995) period. This could be due to errors in the soil hydraulic properties obtained from the literature and used in the model for this field. The daily water tables predicted by the MIKE SHE model for 5 years (1991–1996) were in reasonable agreement with measured data (Figure 1). The square of the correlation coefficient, between actual and modeled groundwater level curves (R^2) is 0.71 indicating that the model explains 71% of the changes in actual water table dynamics. Model predictions for water table position compare favorably with measurements of water table from 1991 throughout 1996. Rapid rates of decline in water table during DOY 800-881 and 1281-1524 can be captured by MIKE SHE, but predictions are somewhat less consistent with measured rates of decline in water table during 1992. In addition, rapid increases in simulated water table following large precipitation events are not observed in field measurements of 1992 nor for mid-September 1995.

Some other discrepancies during the summer and fall were attributed to potential errors in both rainfall and estimates of PET. Note that the measured data were obtained by using rainfall from only one station in the watershed. Relatively large differences in annual rainfall were observed between gauges at the some watershed; therefore, it seems reasonable that a part of the difference could have been due to the use of the site rainfall data (which is available for testing) for the entire watershed. Other potential problems in this sub-watershed may be due to heterogeneities in soil types, land cover, water management practices (not taken into detailed account). On an annual basis the

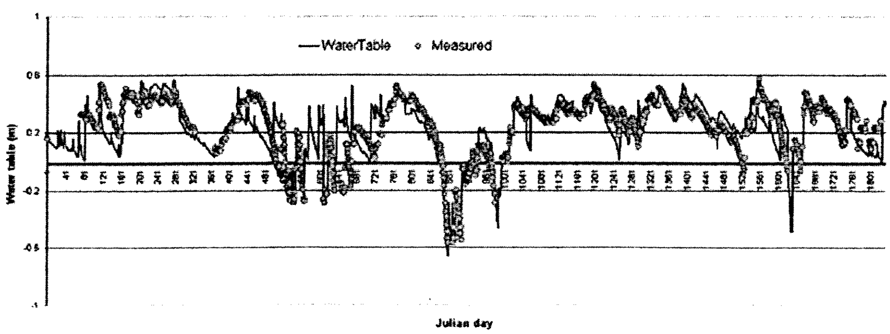

Figure 1. Model predictions for water table position compare favorably with measurements of water table from 1991 to 1996.

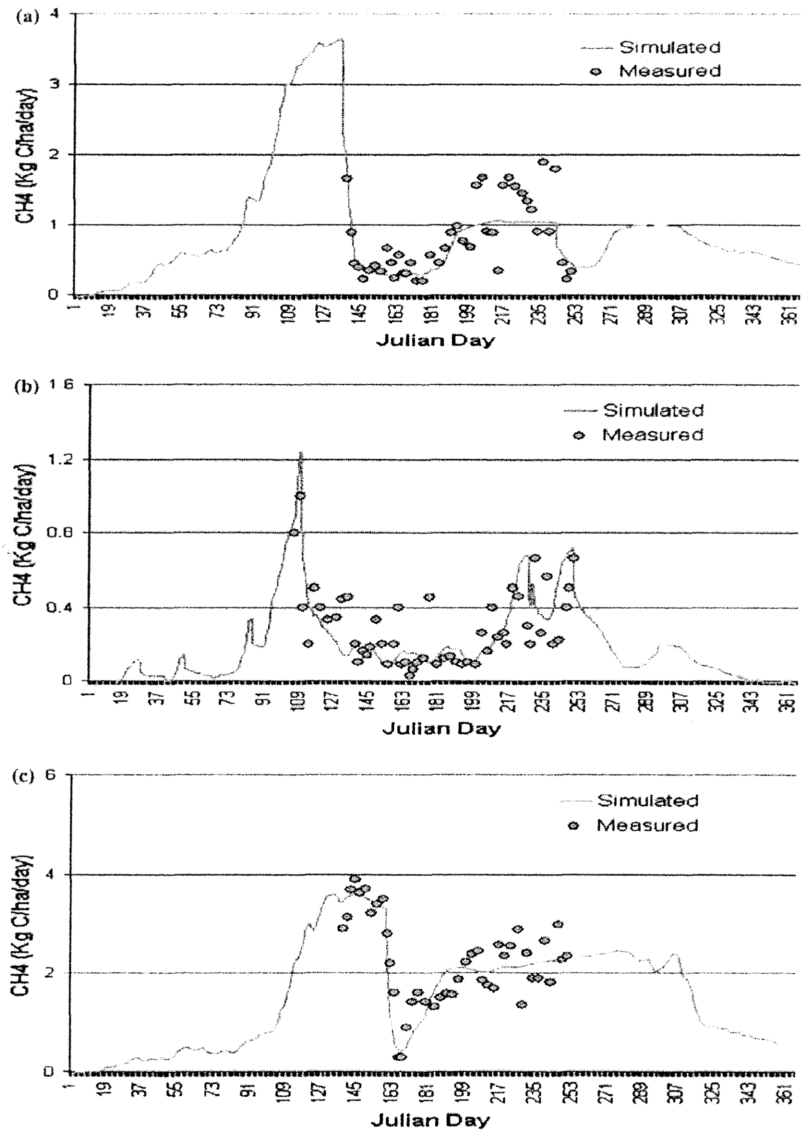

Figure 2. Comparison between the modeled and measured CH_4 fluxes (kg C ha^{-1} day^{-1}): (a) 1994, (b) 1995, and (c) 1996.

predictions of water table were satisfactory. Assuming consistency in model structure, the results clearly demonstrate the need for detailed field information for making a more reliable prediction of daily water table. This is especially true on wetlands with heterogeneous soils, crops and complex management practices.

Emission of CH_4
We tested the model against the CH_4 emission measured at Florida site. The average absolute daily deviations between the measured and predicted CH_4 for each of the years 1994, 1995 and 1996 were 0.23, 0.18 and 0.11 kg C ha^{-1} day^{-1}, respectively. Similarly, the square of the correlation coefficients (R^2) between the measured and predicted daily CH_4 for these 3 years were 0.69, 0.50 and 0.67. The measured data has a similar pattern of the model prediction in all instances (Figure 2). Both measured and estimated concentrations ranged mostly between 0 and 5 kg C ha^{-1}. But in all instance, measured data was slightly higher than that of the model prediction.

Net ecosystem exchange of carbon
The input to the carbon budget in forest wetland ecosystem is the photosynthetic fixation of CO_2 by the forest canopy. Outputs are all in the form of respired CO_2, coming either from plant tissues due to growth or maintenance respiration, or from the litter and soil carbon pools as the result of heterotrophic respiration. The net ecosystem exchange of carbon (NEC) represents the net accumulation or loss of carbon by the entire soil–stand system and is determined as the difference between GPP (the total gain of carbon to the system by net photosynthesis) and Rtot (sum of the maintenance, heterotrophic and growth respiration components). Positive fluxes in this investigation denote a net uptake of carbon by the system while negative fluxes denote a net loss. The Wetland-DNDC output generally agreed with the measured NEC fluxes, although maximum simulated fluxes were somewhat low. The average absolute daily deviations between the measured and predicted NEC for each of the years 1994, 1995 and 1996 were 11, 12.4 and 8 kg C ha^{-1} day^{-1}, respectively. Although there was a relatively large scatter, the covariance coefficients were high (R^2 are 0.86 for 1995 and 0.71, 0.69 for 1994 and 1996) (see Figure 3).

Model application

Simulation scenarios design

For the purposes of this study we start our multiple temporal scales modeling strategy with the seasonal runs, as that has been the focus of our past work. This analysis will be linked to the long-term scale by 'looking upward' to determine the effects of water table fluctuation and management practices on GHG emissions and potential stores of carbon, as well as production and decomposition rates from 1961 to 2100. We set the simulated water table in 1961–2110 as a baseline (160-year average water table: −39.33 cm) by repeating the 20-year (1961–1980) water table dynamics at Gator National Forest (GNF), Florida. The water table was assumed to be at the same level as the measured water table at GNF. Changes in hydrological conditions were studied simulation the system with a constant raise water table of 10, 20, 30

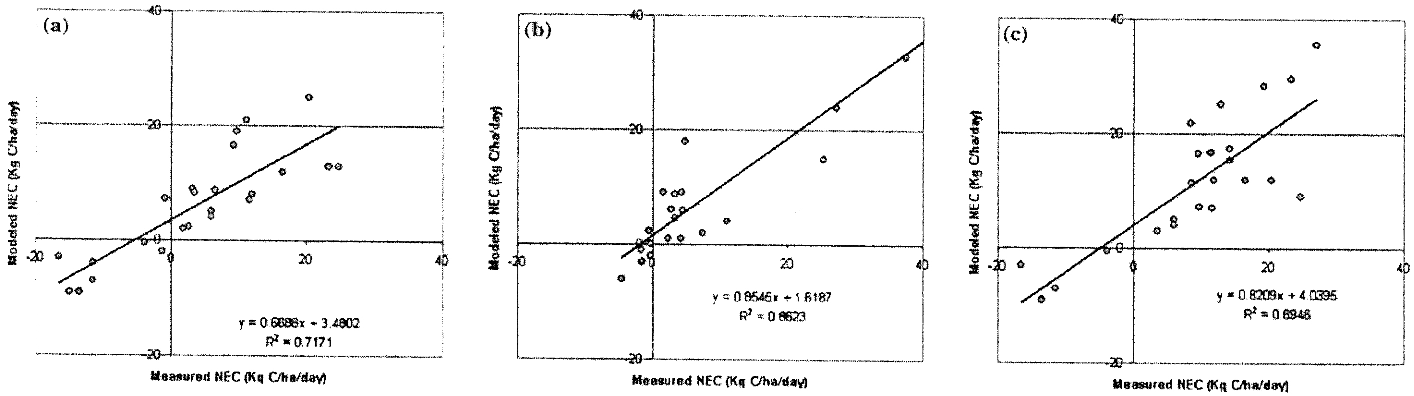

Figure 3. Comparison between the modeled and measured NEC (kg C ha^{-1} day^{-1}): (a) 1994, (b) 1995, and (c) 1996. The bold lines are linear regressions fitted to the data points. The regression parameters and correlation coefficients (R^2) are given in the figure.

Table 1. Scenarios made to evaluate the effect of water table position and forest management practices on GHGs emissions and average carbon storage in forest wetland ecosystem

Scenario	Description	Questions to answer
Water table position		
Baseline water table	Repeat the 20-year (1961–1980) water table fluctuation at Gator National Forest, Florida	Effect of water table position on GHGs emission
WA	Baseline water table + 10 cm	
WB	Baseline water table + 20 cm	
WC	Baseline water table + 30 cm	
WD	Baseline water table + 40 cm	
Forest management practices		
	Changes in harvest and deforestation frequency (Pine forest)	Average long-term carbon storage in ecosystem pools
HA	No harvest during 160 years	
HB	Harvest in year of 120, than deforestation	
HC	Harvest in year of 80, 160, deforestation	
HD	Harvest in year of 40, 80, 120, 160, deforestation	
	Response to application of fertilizer	
FA	No nitrogen fertilizer applied	
FB	Nitrogen fertilizer applied (200 kg N ha^{-1} 6 year)	

and 40 cm as Scenario WA, WB, WC, WD, respectively. Model seasonal scenarios were based on site characteristics, including species composition, site location, elevation, and climate. 160-year climate scenarios were constructed by repeating the relevant 20 years meteorological data.

At the site level, we are using the Wetland-DNDC model to simulate the effects of changes forest management practices, such as harvest, fertilization, and reforestation on carbon flux over 160 years. Particular attention is directed at the impact of alternative management scenarios including longer rotations, more intensive management activities and no-management. Fluxes in forested wetland ecosystem are solved as changes in live biomass, and forest products pools over time. The scenarios are list below, along with the questions they were intended to answer, and summarized in Table 1.

Effect of water table fluctuation on CO_2, CH_4 and N_2O emission

CO_2, CH_4 and N_2O emission

The effect of water table position on CH_4 emission was significant throughout the growing season (Figure 4c). At high water table, CH_4 emission was significantly greater than baseline. If the water table stayed at wetland surface throughout the season, methane flux would increase. Results in Figure 4c emphasize the importance of the uppermost 20 cm, as decrease in the water table from the wetland surface to 20 cm below the wetland surface decreases methane flux by $\sim$5%, while further decrease from 20 to 30 cm changes the flux from -24% of the reference to -46% of the reference.

Table 2 shows a summary of the total flux of GHGs emission during 160 years as simulated by the Wetland-DNDC. Methane emissions varied widely among water table scenarios throughout the 160 years. CH_4 emissions is low if the water table remains deep below the surface (Baseline). Scenario WB and Scenario WC increased emissions to 4.87- and 8.1-fold, and Scenario WD increased emissions to 12.1-fold in comparison to emissions from the Pine vegetation having a deep water table (Baseline). If forested wetlands keep higher water table, it would represent a long-term net source of atmospheric methane.

The seasonal variation in net CO_2 exchange followed the seasonal variation in water table (Figure 4b). The reconstruction of CO_2 fluxes over the growing season demonstrates the sensitivity of carbon dynamics to water table variation. Scenario WA, WB and WC lowered CO_2 emission from wetland surface about 4.95, 9.2, and 12.7%, respectively, and by raising the water table close to the surface (Scenario WD), the emission was reduced by over 15.4% (Table 2). In dense Pine vegetation, the simulated seasonal CO_2 balance was positive in each year 1961–2110. The CO_2 balance was 2–3 times higher with Scenario WD than with Scenario WB. The possible mechanism is that oxygen concentration in vertical soil profile depends on water table position and moisture profile, when a soil is shifting from unsaturated to saturated conditions, oxygen concentration in water saturated wetland could be assumed constantly so low, and hence a series of reductive reactions will be occurred. These reductive reactions usually include reductions of manganese (Mn^{4+}), iron (Fe^{3+}) and sulfate (SO_4^{2-}), and methanogenesis. The processes reduce soil CO_2 emission due to the depressed microbial respiration, and elevate CH_4 emission due to the enhanced denitrification or methanogensis. When a soil is shifting from saturated to unsaturated conditions, the oxidative reactions (e.g., methanotrophy, nitrification, decomposition, etc.) will be enhanced. That will increase soil CO_2 emission and decrease CH_4 emissions.

Nitrous oxide emissions from soils are caused principally by microbial nitrification and denitrificaion. These processes are controlled by several factors – particularly soil water-filled pore space, which depends on the balance between the amount of water entering the soil through precipitation or irrigation and the combined effect of evapotranspiration and drainage. Modeled N_2O fluxes were low (0.58 kg N ha^{-1} $year^{-1}$) on Scenario WD. Some of the N_2O fluxes modeled on this Scenario were greatest during the DOY 70-131 and 190-280 when the water table were below the surface. Fluxes during these periods contributed to around 80% of the total N_2O annual budget. The flux of N_2O decreased with increasing water table throughout the season and in a situation when the water table stayed at wetland surface, N_2O emissions are close to zero. As soon as the soils were drained (Scenarios C, B, A), rapid decomposition of the accumulated labile SOC released a great amount of DOC, ammonium and nitrate, which stimulated both nitrification and denitrification to elevate N_2O emissions (Figure 4d).

Global warming potential

We used the global warming potential (GWP) for demonstrating the sum of the warming forces of all the three GHGs. GWP is a measure of how much a given mass of GHG is estimated to contribute to global warming. The higher GWP of lower-emitting GHGs significantly increases their contributions to the greenhouse effect. According to IPCC (1991) over a 100-year time horizon, nitrous oxide N_2O is 310 times and CH_4 is 21 times more effective than carbon dioxide at trapping heat in the atmosphere (Table 2). Rise the water table from the baseline to 30 cm below the wetland surface decreases GWP by −4.4%, while further increase from 30 to 10 cm changes the GWP from −4.65% of the reference to −2.55% of the reference. At high water table, GWP increased by 1.99% because of the CH_4 emission was significantly greater than baseline.

Effect of forest management practices on carbon storage in ecosystem pools

Values for carbon in the litter and debris, vegetation and soil pools response to harvesting events

Long-term carbon storage in ecosystem pools was sensitive to changes in harvest frequency. The consequences of changing the schedule of harvesting for products including a no harvest option are shown average total carbon storage increased

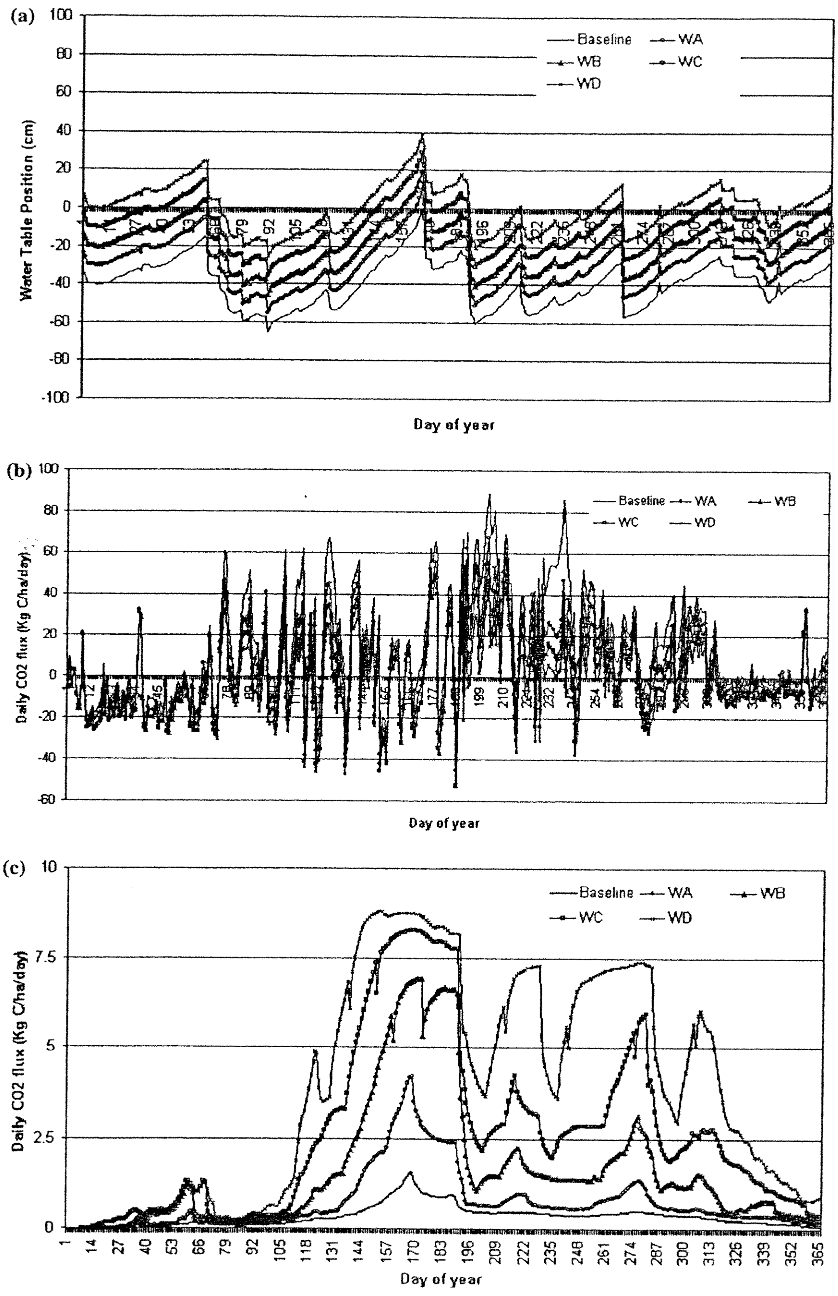

Figure 4. The annual variation in CO_2, CH_4 and N_2O emissions followed the variation in water table: (a) elevated water table (cm, negative values indicate that water table remains below wetland surface) and measured water tables (circles); (b) CO_2 (kg C ha^{-1}); (c) CH_4 (kg C ha^{-1}) and (d) N_2O (g N ha^{-1}).

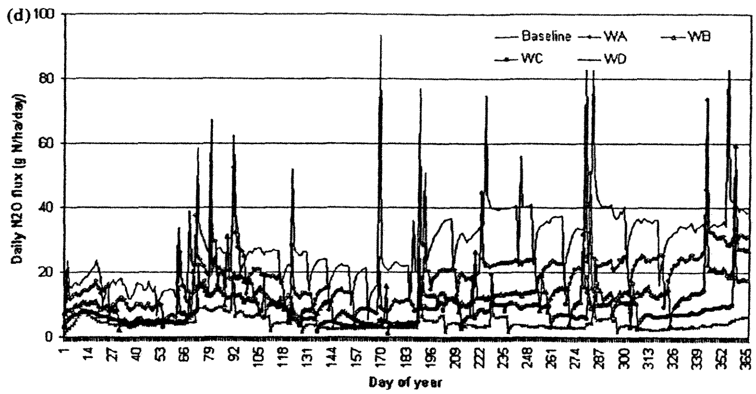

Figure 4. Continued.

with increasing rotation length. Figure 5 shows the net forest carbon for different management rotations. Maximum total ecosystem carbon storage was observed in Scenario HA with no harvest applied. Conversely, use of short rotation periods (Scenario HB and Scenario HC) led to substantial declines in long-term ecosystem carbon storage. Our analysis of carbon stores in forest wetland ecosystems shows that carbon levels in Scenario HB, Scenario HC and Scenario HD have declined 57.9, 64.2 and 76.4%, respectively, compare to Scenario HA.

Carbon storage increase was attributable largely to the vegetation pool, though carbon stored in the litter and soil pools was also generally higher at longer rotation lengths. In our analysis using Wetland-DNDC, there were clear differences in vegetation carbon storage patterns between harvest management. Forests store carbon as they accumulate biomass until disturbances or natural mortality more than offsets growth. Scenario HA is shown to store the most carbon in the forest with the pool decreasing with shorter rotations. In Scenario HA, average vegetation pools was 8% greater than Scenario HB and 24% greater than Scenario HC. In scenario HD, vegetation carbon was 33% less than original stores at 160 years.

In addition to reducing average storage in vegetation pools, intensive harvesting negatively impacted carbon stored in litter and debris pools. The Scenario HB was the only scenario in which average storage was roughly equivalent to Scenario HA (the natural disturbance scenario). The proportion of carbon sequestered in litter pools was roughly equivalent between the natural disturbance and management scenarios. Despite considerable fluctuation of carbon in vegetation and litter pools following harvest, in the short-term soil carbon was unaffected by management activities. Similarly, in a recent study of forest management and its relation to soil carbon, Johnson et al. (2002) concluded that harvesting, on average, caused little or no decline in the organic carbon content in soils. The modified Wetland-DNDC captured this feature. The long-term effect of management activities on soil carbon in our simulation is dependent on the existing quantity of SOM. In our study, harvesting lead to

Table 2. Wetland-DNDC modeled greenhouse emissions with different water table scenarios in comparison to emissions from the baseline*

Scenario	Greenhouse gases emission			
	CO_2	CH_4	N_2O	GWP**
WA	−4.95%	2.18	−0.71	−4.40%
WB	−9.20%	4.87	−0.51	−4.65%
WC	−12.70%	8.06	−0.40	−2.55%
WD	−15.40%	12.10	−0.31	1.99%

*CH_4 and N_2O are showed as folds in comparison with the baseline, CO_2 and GWP are showed as percent change, respectively.
**The GWP value for each scenario is calculated as follows: $GWP_i = CO_{2i}/12 * 44 + N_2O_i/28 * 44 * 310 + CH_{4i}/12 * 16 * 21$; where GWP_i (kg CO_2 equivalent ha^{-1} $year^{-1}$) is the global warming potential induced by scenario *i*; CO_{2i}, N_2O_i and CH_{4i} are CO_2 flux (kg C ha^{-1} $year^{-1}$), N_2O flux (kg N ha^{-1} $year^{-1}$) and CH_4 flux (kg C ha^{-1} $year^{-1}$), respectively, induced by scenario *i*.

a net decline (0.2–4%) in soil carbon content during the 160-year simulation period, the greatest losses occurring in short rotation Scenario HD. This decline in soil carbon resulted from the cumulative impact of decreased litter production associated with harvesting.

Values for carbon in the litter, biomass and soil pools response to application of fertilizer

Application of nitrogen fertilizer to watershed increased average carbon storage in all ecosystem pools and wood products. Most of the increase was attributable to increased storage in wood products (23%) and biomass (16%) pools. When the carbon costs of N fertilizer production are accounted for (1.5 units C for every unit of N fertilizer produced), the net increase in carbon

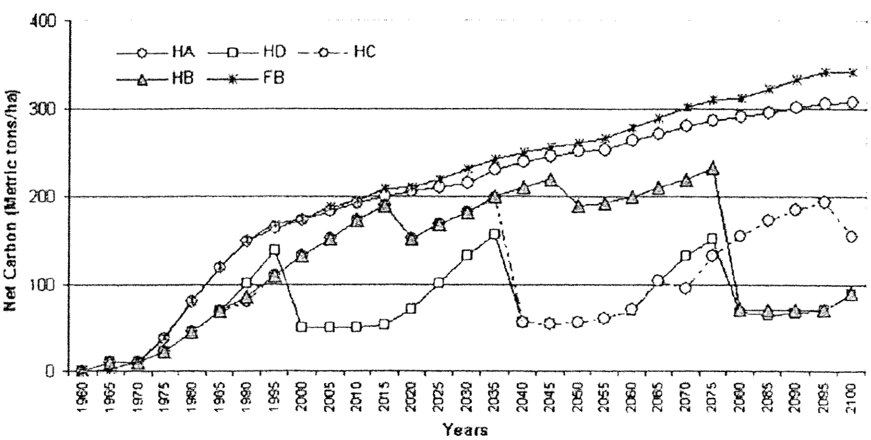

Figure 5. Effect of rotation length and nitrogen application on the net carbon change over the 160-year aimulation period. Description of scenarios HA, HB, HC, HD and FB, see Table 1.

storage drops from 5.4%. The proportion of total carbon stored in the ecosystem pools and wood products is shown in Figure 5. There was proportionately less storage in soil and greater storage in the biomass pools.

Discussions and conclusions

The MIKE SHE and Wetland-DNDC models are modified to estimate the effects of water table fluctuation and forest management regime on the potential carbon dynamics and GHGs emissions. We constructed scenarios to simulate changes in water table position and forest management practices to show how they influence carbon dynamics characteristics in forested wetlands. These analysis focuses on simulations studied the effects of an increase or a decrease in water table position, without change in other environmental factors. Simulation results should indicate the degree of water table position influences GHGs emissions and the carbon dynamics and productivity in different management scenarios. We are also exploring the linkage of MIKE SHE and Wetland-DNDC to estimate of carbon dynamics and GHG emission as a way to test the ability of using models to predict carbon stores and dynamics in forested wetland ecosystem.

It is clear that the potential impact of climate change on the water balance of forested wetland ecosystems is a crucial question for future C balance research. The spatial–temporal distribution of near-surface soil moisture or water table is central to the regulation of land–atmosphere water, energy, and carbon interaction. Hydrological changes had a remarkable effect on GHGs fluxes, even if everything else remained the same. Anaerobic conditions in wetland soils reduce organic matter decomposition and stimulate CH_4 production. Our results support previous findings on wetland CO_2 flux response to water deficits (Shurpali et al. 1995; Joiner et al. 1999). Drainage of a wetland would provide a pathway for water to exit wetlands, thereby lowering the water table, will most likely result in a reduction in CH_4 emissions and an increase in CO_2 emissions from soils. Harvesting of wet forests also involves their drainage. The consequences of the increased soil aeration in wetland is enhanced soil oxidation and subsidence associated with the loss of buoyancy of the overlying material as the water table dropped. Although both processes decrease water storage capacity, they also have the effect of increasing soil bulk density, and therefore decreasing saturated and unsaturated hydraulic conductivity, and increasing water retention.

Based on our simulation, maintain anaerobic conditions and management of forest wetland environments to enhance primary production and increase SOM, are the options involved as GHGs mitigation strategies in forest wetland. Carbon stores increased as rotation length increased. Our results confirm that long-term carbon storage in ecosystem pools was sensitive to changes in harvest frequency. In this respect, allowing managed forests to accumulate greater biomass through longer rotations should be an effective means of increasing carbon storage. In addition to reducing storage in biomass pools, intensive harvesting negatively impacted carbon stored in litter and dead wood pools. Carbon sequestration is the balance of inputs and outputs from forested wetland ecosystem. If decomposition is inhibited because of nitrogen inputs, then increased biological nitrogen fixation, nitrogen fertilization and nitrogen deposition may promote carbon sequestration.

In this paper, we present the model simulation results of the water table fluctuation and forest management activities that can be implemented to mitigate carbon emissions and thus serve as potential carbon-offset projects. Hydrological model – MIKE SHE's reasonable prediction of water table dynamics and its linkage with biogeochemical model, Wetland-DNDC, indicated that this system can be used to predict seasonal/annual and long-term cumulative carbon dynamics and provide a means to understand how different parts of the ecosystem respond to environmental drivers. Such estimates are important in understanding the future potential fluxes of carbon into terrestrial biomes; they can be used for the carbon budgeting of planting and forest management policies, in long-term planning of forest resources, etc. and thereby improve both strategic and tactical planning for managing natural resources in a sustainable and environmentally sound manner.

Acknowledgments

The authors are grateful to the Southern Global Change Program, USDA Forest Service and the National Council for Air and Stream Improvement (NCASI) for proving financial support for this project; and would like to thank Dr Marsha Hosner (DHI Inc), who kindly provided the MIKE SHE model and were always open for fruitful discussion. Thanks are also given to the two anonymous reviewers whose comments stimulated us to not only rewrite the paper but also reorganize the model tests.

References

Aber J.D., Reich P.B. and Goulden M.L. 1996. Extrapolating leaf CO_2 exchange to the canopy: a generalized model of forest photosynthesis compared with measurements by eddy correlation. Oecologia 10: 257–265.

Anderson I.C. and Levine J.S. 1986. Relative rates of nitric oxide and nitrous oxide production by nitrifiers, denitrifiers, and nitrate respirers. Appl. Environ. Microbiol. 51: 938–945.

Arah J.R.M. and Stephen K. 1998. A model of athe processes leading to methane emission from peatland. Atmos. Environ. 32: 3257–3264.

Aurela M., Laurila T. and Tuoviner J.P. 2001. Seasonal CO_2 balances of a subarctic mire. J. Geophys. Res. 106: 1623–1638.

Bollmann A. and Conrad R. 1998. Influence of O_2 availability on NO and N_2O release by nitrification and denitrification in soils. Global Change Biol. 4: 387–396.

Bubier J.L., Bhatia G., Moore T.R., Roulet N.T. and Lafleur P.M. 2004. Between year and site variability in growing season net ecosystem CO_2 exchange at a large peatland. Ontario, Canada, Ecosystems, in press.

Cao M., Marshall S. and Gregson K. 1996. Global carbon exchange and methane emissions from natural wetlands: application of a process-based model. J. Geophys. Res. 101(D9): 14399–14414.

Chertov O.G. 1990. SPECOM – A single tree model of pine stand /raw humus soil ecosystem. Ecol. Model. 50: 107–132.

De Willigen P. 1991. Nitrogen turnover in the soil-crop system: comparison of fourteen simulation models. Fertilizer Res. 27: 141–149.

Frolking S., Roulet N., Moore T., Lafleur P., Bubier J. and Crill P. 2002. Modeling the seasonal to annual carbon balance of northern peatlands. Global Biogeochem. Cycles 16(3), doi: 10.1029/2001GB1457.

Granberg G., Ottosson-Lofvenius H., Grip H., Sundh I. and Nilsson M. 2001. Effect of climatic variability from 1980 to 1997 on simulated methane emission from a boreal mixed mire in northern Sweden. Global Biogeochem. Cycles 15(4): 977–991.

Grant R.F. 1998. Simulation of methanogenesis in the mathematical model ECOSYS. Soil Biol. Biochem. 30(7): 883–896.

Holland E.A. and Schimel D.S. 1994. Ecosystem and physiological controls over methnae production in northern wetlands. J. Geophys. Res. 99: 1563–1571.

IPCC 1991. Intergovernmental Panel on Climate Change, Guidelines for National Greenhouse Gas Inventories. OECD/ODCE, Paris.

Johnson D.W., Knoepp J.D., Swank W.T., Shan J., Morris L.A., Van Lear D.H. and Kapeluck P.R. 2002. Effects of forest management on soil carbon: results of some long-term resampling studies. Environ. Pollut. 116: S201–S208.

Joiner D.W., Lafleur P.M., McCaughey J.H. and Bartlett P.A. 1999. Inter annual variability in carbon dioxide exchanges at a boreal wetland in the BORES northern study area. J. Geophys. Res. 104: 27663–27672.

Kettunen A., Kaitala V., Lehtinen A., Lohila A., Alm J., Silvola J. and Martikainen P.J. 1999. Methane production and oxidation potentials in relation to water table fluctuations in two boreal mires. Soil Biol. Biochem. 31: 1741–1749.

Lafleur P.M., Roulet N.T., Bubier J.L., Frolking S. and Moore T.R. 2003. Interannual variability in the peatland-atmosphere carbon dioxide exchange at an ombrotrophic bog. Global Biogeochemical Cycles 17(2): 1036, doi: 10.1029/2002GB001.

Lafleur P.M., Roulet N.T. and Admiral S.W. 2001a. Annual cycle of CO_2 exchange at a bog peatland. J. Geophys. Res. 106: 3071–3081.

Li C., Aber J., Stange F., Butterbach-Bahl K. and Papen H. 2000. A process-oriented model of N_2O and NO emissions from forest soils: 1, Model development. J. Geophys. Res. 105: 4369–4384.

Li C., Frolking S. and Frolking T.A. 1992. A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity. J. Geophys. Res. 97: 9759–9776.

McClain M.E., Mayorga E., Logsdon M.G. and Richey J.E. 1996. A conceptual framework for modelling organic dynamics in large river systems'. In: Bottrell S.H. (ed.), Fourth International Symposium on the Geochemistry of the Earth's Surface. University of Leeds, Leeds, pp. 323–326.

Minkinnen K., Korhonen R., Savolainen I. and Laine J. 2002. Carbon balance and radiative forcing of Finnish peatlands 1900–2100 – the impact of forestry drainage. Global Change Biol. 8: 785–799.

Moore T.R. and Dalva M. 1993. The influence of temperature and water table position on carbon dioxide and methane emissions from laboratory columns of peatland soil. J. Soil Sci. 44: 651–664.

Nieveen J.P., Jacobs C.M.J. and Jacobs A.F.G. 1998. Diurnal and seasonal variation of carbon dioxide exchange from a former true raised bog. Global Change Biol. 4: 823–850.

Paul E.A. and Clark F.E. 1989. Soil Microbiology and Biochemistry, 2nd ed. Academic Press, San Diego, Boston, New York, Tokyo, Toronto, pp. 157–166.

Refsgaard J.C. and Storm B. 1995. In: Singh V.P. (ed.), MIKE SHE Computer Models in Watershed Hydrology. Water Resource Publications, Colorodo, USA, pp. 806–846 .

Sass R.L., Fisher F.M., Turner F.T. and Jund M.F. 1991. Methane emission from rice fields as influenced by solar radiation, temperature, and straw incorporation. Global Biogeochem. Cycles 5: 335–350.

Schreader C.P., Rouse W.R., Griffis T.J., Boudreau L.D. and Blanken P.D. 1998. Carbon dioxide fluxes in a northern fen during a hot, dry summer. Global Biogeochem. Cycles 12: 729–740.

Segers R. and Leffelaar P.A. 2001a. Modeling methane fluxed in wetland with gas-transporting plants. 1. Single root scale. J. Geophys. Res. 106(D4): 3511–3528.

Segers R. and Leffelaar P.A. 2001b. Modeling methane fluxed in wetland with gas-transporting plants. 3. Plot scale. J. Geophys. Res. 106(D4): 3541–3558.

Segers R., Rappoldt C. and Leffelaar P.A. 2001. Modeling methane fluxed in wetland with gas-transporting plants 2. Soil layer scale. J. Geophys. Res. 106(D4): 3529–3540.

Shurpali N.J., Verma S.B., Kim J. and Arkebauer T.J. 1995. Carbon dioxide exchange in a peatland ecosystem. J. Geophys. Res. 100: 14319–14326.

Silvola J., Alm J., Ahlholm U., Nykanen H. and Martikainen P.J. 1996. CO_2 fluxes from peat in boreal mires under varying temperature and moisture conditions. J. Ecology 84: 219–228.

Stumm W. and Morgan J.J. 1981. Aquatic Chemistry: An Introduction Emphasizing Chemical Equilibria in Natural Waters, 2nd ed. John Wiley & Sons, New York, pp. 418–503.

Sun G., Riekerk H. and Comerford N.B. 1998. Modeling the hydrologic impacts of forest harvesting on Florida flatwoods. J. Am. Water Resour. Assoc. 34(4): 843–854.

Trettin C.C., Song B., Jurgensen M.F. and Li C. 2001. Existing soil carbon models do not apply to forested wetlands. Gen. Tecsh. Rep. SRS-46 US Department of Agriculture, Forest Service, Southern Research Station, Asheville, NC, 10 p.

Waddington J.M. and Roulet N.T. 2000. Carbon balance of a boreal patterned peatland. Global Change Biol. 6: 87–97.

Walter B.P. and Heimann M. 2000. A process-based, climate sensitive model to derive methane emissions from natural wetlands: application to five wetland sites, sensitivity to model parameters, and climate. Global Biogeochem. Cycles 14(3): 745–765.

Walter B.P., Heimann M. and Matthews E. 2001a. Modeling modern methane emissions from natural wetlands: 1. Model description and results. J. Geophys. Res. 106(24): 34189–34206.

Walter B.P., Heimann M. and Matthews E. 2001b. Modeling modern methane emissions from natural wetlands: 2 Interannual variations 1982–1993. J. Geophys. Res. 106(24): 34207–34219.

Wassmann R., Wang M.X., Shangguan X.J., Xie X.L., Shen R.X., Wang Y.S., Papen H., Rennenberg H. and Seiler W. 1993. First records of a field experiment on fertilizer effects on methane emission from rice fields in Hunan-province (PR China). Geophys. Res., Lett. 20: 2071–2074.

Weltzin J.F., Pastor J., Harth C., Bridgham S.D., Updegraff K. and Chapin C.T. 2000. Response of bog and fen plant communities to warming and water-table manipulations. Ecology 81: 3464–3478.

Whiting G.J. and Chanton J.P. 2001. Greenhouse carbon balance of wetlands: methane emission versus carbon sequestration. Tellus 53B: 521–528.

Yagi K. and Manami K. 1990. Effect of organic matter application on methane emission from some Japanese paddy fields. Soil Sci. Plant Nutr. 36: 599–610.

Zoltai S.C. and Martikainen P.J. 1996. Estimated extent of forested peatlands and their role in the global carbon cycle. In: Apps M.J. and Price D.T. (eds), Forest Ecosystems, Forest Management and the Global Carbon Cycle, NATO Advanced Science Institutes Series, Vol. 140. Springer Verlag, Heidelberg, pp. 47–58.