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Modeling Impacts of Management on Carbon Sequestration and Trace Gas Emissions in Forested Wetland Ecosystems

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ABSTRACT / A process-based model, Wetland-DNDC, was modified to enhance its capacity to predict the impacts of management practices on carbon sequestration in and trace gas emissions from forested wetland ecosystems. The modifications included parameterization of management practices (e.g., forest harvest, chopping, burning, water management, fertilization, and tree planting), inclusion of detailed anaerobic biogeochemical processes for wetland soils, and utilization of hydrological models for quantifying water table variations. A 150-year management scenario consisting of three stages of wetland forest, deforestation/drainage, and wetland restoration was simulated with the Wetland-DNDC for two wetlands in Minnesota and Florida, USA. The impacts of the management scenario on carbon ecosystem exchange, methane emission, and nitrous oxide emission were quantified and assessed. The results suggested that: (1) the same management scenario produced very different consequences on global warming due to the contrast climate conditions; and (2) methane and nitrous oxide fluxes played nonnegligible roles in mitigation in comparison with carbon sequestration.

Forests are recognized for having considerable potential to sequester carbon (C) (Birdsey and Heath 2001). In the conterminous United States, wetlands comprise approximately 16% of the forested area, but they contain over 50% of the total soil C (Trettin and Jurgensen 2003). Unfortunately, analyses of C dynamics in forested landscapes typically presume upland soil conditions, thereby overlooking the potential and complexity associated with hydric soils. Accordingly, presuming aerated conditions across the forested landscape overlooks processes that are critical to the terrestrial C cycle, and completely discounts the complex interactions of management practices with C cycling in hydric soils.

The forested wetland resource is not static. Both natural and management-induced disturbance regimes are common and necessary. Until recently, disturbance regimes were considered to be detrimental to soil C in wetland forests. However, long-term field studies have now demonstrated that increased forest productivity on managed peatlands can result in soil C gains (Minkkinen and Laine 1998). Many forest management practices have been reported to enhance carbon mitigation. Recently, more comprehensive studies emphasized the importance of complete accounting for the entire carbon flows in and out of the system and the analysis of long-term patterns. Restoration of forested wetlands is now considered a means to enhance terrestrial C sequestration. Although there are no long-term field studies, short-term studies suggest high rates of sequestration (Trettin and Jurgensen 2003). A computer simulation model is an alternative tool addressing this type of questions.

There are few opportunities to augment the accumulation of carbon in wetlands by improved management, although the theoretical foundations for the interactions involved in watershed are reasonably well established. Draining 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. Experimental results are also limited by the technical difficulty of controlling CO_2 and climate at appropriate spatial scales, and by the very long

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time constants for some of the component processes. While the limitations of earlier studies are now evident, data for comprehensive analysis at the regional scale are not yet available. Accordingly, understanding how disturbance or restoration regimes affect the C dynamics in forested wetlands is critical to accurate assessments of C balance, whether at the site or landscape level. To quantify the complex impacts of management practices on C dynamics in wetlands, we realized utilizing process-based models was unavoidable. We recognized that the prevalent soil C models were not applicable to forested wetlands (Trettin and others 2001) and proceeded to develop a new modelling framework based on PnET-N-DNDC, a process-oriented model simulating C and nitrogen (N) dynamic and trace gas emissions in upland forest ecosystems (Li and others 2000). The new model, Wetland-DNDC, considers C dynamics as driven by hydrology, soil biogeochemistry, and vegetation processes in wetland ecosystems (Zhang and others 2002). Wetland-DNDC was designed for both mineral and organic soils, where water table dynamics, soil properties, vegetation, and climate affect the C cycling and the partitioning among greenhouse gases. Zhang and others (2002) validated the model for boreal-forested wetlands, demonstrating the effectiveness of this process-based model. In this paper, we report an improved version of the Wetland-DNDC model that has incorporated common management practices in forested wetlands. Our objectives are: (1) to demonstrate the model utility to simulate the forest soil C cycle under managed conditions; and most importantly, (2) to demonstrate the sensitivity of forested wetland C cycles to climate gradient and management practices.

Methods

Model Modifications

A process-based model, Wetland-DNDC, was developed for predicting C and N biogeochemistry in forested wetland ecosystems. The model was constructed by incorporating hydrological features into an upland forest model, PnET-N-DNDC (Li and others 2000, Stange and others 2000). The basic functions adopted by Wetland-DNDC for simulating forest growth, soil biogeochemistry, and hydrological processes have been evaluated in previous publications (Li and others 1992, Li 2000, Zhang and others 2002). The original version of Wetland-DNDC focused on natural wetlands with few management options. 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 hydrological models.

Parameterization of management practices. Common forest management practices such as harvest, chopping, burning, water management, wetland restoration, fertilization, reforestation, etc. (Trettin and Jurgensen 2003) were parameterized in Wetland-DNDC to enable it to predict impacts of change in management on C sequestration 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 plants, or lichens) based on their competition for light, water, and N (Zhang and others 2002). A forest harvest event is defined by its timing and percent of the above-ground woody biomass harvested. 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 and others 2000). If fire is used for deforestation, it will be defined by its timing and burned fractions of the above-ground biomass as well as the forest floor mass. We assumed the understory and ground-growth would survive from fires and start regrowth after burning. Chopping has been parameterized as a means for preparing planting beds. Chopping cleans the understory and leaves all of the litter in the soil. Reforestation is defined with its timing and the type and age of planted trees. Reforestation can be applied with or without an understory component. Water management is defined by drainage duration and water table depth. As soon as drainage is applied in the model simulations, the controlled water table will override the natural water table depth. Termination of drainage will resume the observed or modeled natural water table dynamics. Fertilization is defined by its timing and amount of N applied. The fertilized nutrients join the inorganic N pools to support the tree growth or soil microbial activities during the simulations. A wide range of management scenarios can be constructed by combining the above-listed management practices. For example, a wetland-to-farmland conversion scenario can be combined with drainage and deforestation, and a wetland restoration scenario can include drainage termination and reforestation. The new features for management practices have been linked to the forest growth and soil biogeochemistry submodels originally embedded in the

Wetland-DNDC model (Zhang and others 2002, Li and others 2000).

Modeling anaerobic biogeochemistry. Wetland soil is characterized by the presence of a saturated zone. The saturated proportion of a soil profile is determined by the water table fluctuations. This dynamic water table feature greatly affects C and N dynamics and processes in the wetland soils. When a soil is shifting from unsaturated to saturated conditions, the soil oxygen (O_2) will be depleted, and hence a series of reductive reactions will be stimulated. These reductive reactions usually include denitrification; 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 N_2O or CH_4 emissions due to the enhanced denitrification or methanogensis, respectively. When a soil is shifting from saturated to unsaturated conditions, the oxidative reactions (e.g., methanotrophy, nitrification, decomposition, etc.) will be enhanced, resulting in increased soil CO_2 emission and decreased N_2O or CH_4 emissions. In wetland soils, flooding and draining practices cause dramatic changes in the soil redox potential (i.e., Eh) conditions, and hence affect the production and consumption of greenhouse gases in the soils (Anderson and Levine 1986, Bollmann and Conrad 1998, Sass and others 1991, Wassmann and others 1993, Yagi and Manami 1990, Holland and Schimel 1994). To quantify the Eh dynamics and its impacts on N_2O or CH_4 production, we integrated two classical equations, the Nernst equation (Equation 1) and the Michaelis-Menten equation (Equation 3), into one 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 is a widely applied formula describing kinetics of microbial growth with dual nutrients (Paul and Clark 1998).

$$\text{Nernst equation: } Eh = E_0 + RT/nF \cdot \ln([\text{oxidant}]/[\text{reductant}]) \quad (1)$$

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 the transferred electron number, F is a constant (96,500 k), [oxidant] is the concentration (mole/liter) of the dominant oxidant in the system, and [reductant] is the concentration (mole/liter) of the dominant reductant in the system.

For a system that has several pairs of oxidant–reductant, the species that has the lowest Gibbs free energy for changing electrons usually dominates the Eh of the entire system. For example, since oxygen has the lowest Gibbs free energy change in comparison with other common oxidants in soils, oxygen dominates the soil Eh in the aerobic soils. If we set temperature and pH to be constants, the Nernst equation can be simplified as

$$Eh = 0.82 + 0.015 \log[O_2] \quad (2)$$

where $[O_2]$ is oxygen concentration (mole/liter) in soil water (pH 7.0, temperature = 25°C). According to Bader (1978), dual-substrate Michaelis-Menten kinetics equation can be used to estimate rates of the biochemical reductions, including denitrification and methanogenesis. Based on the equation, rate of each reduction is controlled by the oxidant content and available C concentration:

$$f[\text{oxidant}] = a\,[\text{DOC}/(b + \text{DOC})] * [\text{oxidant}/(c + \text{oxidant})] \quad (3)$$

where f[oxidant] is fraction of the oxidant reduced during a time step, DOC is available C content, and a, b, and c are coefficients.

Since both the Nernst and the Michaelis-Menten equations share a common factor, oxidant concentration, they can be merged. A simple kinetic scheme was adopted in Wetland-DNDC to realize the conjunction with computing language. The kinetic scheme is defined to be the anaerobic volumetric fraction of a soil. Based on concentrations of the dominant oxidants and reductants in a soil, the Nernst equation calculates the soil bulk Eh. 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). Based on the size proportion, Wetland-DNDC allocates the substrates (e.g., DOC, NO_3^-, NH_4^+, etc.) into the aerobic and anaerobic microsites in the soil. We defined that the substrates allocated within the anaerobic volumetric fraction can only be involved in the reductive reaction (e.g., denitrification, methanogenesis); and the substrates allocated outside of the anaerobic volumetric fraction can only 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. Since the anaerobic volumetric fraction only swells or shrinks as driven by the reduction–oxidation reactions, we simply called it an "anaerobic balloon" (Figure 1). When a soil is irrigated or flooded, its oxygen content will decrease and that will drive the anaerobic balloon to swell. As soon as the oxygen is depleted, the anaerobic balloon will

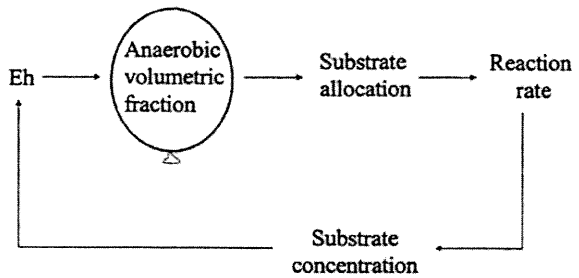

Figure 1. Anaerobic volumetric fraction (anaerobic balloon) links soil redox potential (Eh) to the rates of oxidative–reductive reactions through allocating substrates (e.g., DOC, ammonium, nitrate, etc.) in or outside of the balloon. The soil bulk Eh value is estimated by the Nernst equation based on the substrate concentrations in the soil liquid phase. The oxidative or reductive reaction rate is calculated with the Michaelis-Menten equation based on availability of the substrates within the balloon (for reductive reactions) or outside of the balloon (for oxidative reactions). When the balloon swells, more of the substrates will be allocated within the balloon to enhance the reductive reactions, less substrates left outside of the balloon to participate in the oxidative reactions, and the gas products (e.g., N_2O, NO, CH_4) will take a longer time to diffuse out of the balloon. The latter effect will further reduce some of the gases (e.g., N_2O and NO) to the end products (e.g., N_2) through dentrification. When the balloon shrinks, all of the trends will be reverse. With the ever swelling or shrinking anaerobic balloon in the soil matrix, Wetland-DNDC calculates both the reductive reactions (e.g., dentrification, methane production) and the oxidative reactions (e.g., nitrification, methane oxidation) occurring simultaneously within a soil.

reach its maximum and burst. At this moment, a new oxidant (i.e., NO_3^-) will become the dominant species in the soil, and a new anaerobic balloon will be created and swell as driven by the NO_3^- depletion. By tracking the formation and deflation of a series of anaerobic balloons driven by depletions of oxygen, NO_3^-, Mn^{4+}, Fe^{3+}, and SO_4^{2-}, Wetland-DNDC estimates soil Eh dynamics as well as production and consumptions of the products from the reductive–oxidative reactions, including N_2O and CH_4. With the anaerobic balloon, Wetland-DNDC is well linked to the soil water regime to trace gas emissions for wetland ecosystems.

Linking with hydrological models. Water table depth is a key factor for defining soil aeration and hence a series of soil biogeochemical processes in wetland ecosystems. The ground water table divides the soil profile into saturated and unsaturated zones, which have fundamentally different biochemical and geochemical reactions due to the different redox potential conditions. Several approaches have been adopted for Wetland-DNDC to obtain water table dynamics. Observed water table depth values can be directly utilized as input or convert to empirical equations to drive biogeochemical processes in the model simulations (Zhang and others 2002). Both approaches are limited to small scale studies. To overcome this shortcoming, we tested several spatially distributed hydrological models such as FLATWOODS (Sun and others 1998), SWAT 2003 http://www.brc.tamus.edu/swat/index.html), and MIKE SHE http://www.dhisoftware.com/mikeshe/). These models predict the spatial distributions of water table depths by using spatial databases of topography, climate, soil, and vegetation at a watershed scale. The major advantage of linking a spatially distributed hydrologic model is that we can examine the spatial dynamics of the biogeochemical cycles and their response to management at a landscape scale. For the study reported in this paper, we used the empirical equations to generate daily water table depths to support the model simulations.

Site and Management Scenario

Two forested wetland sites with different climate regimes were selected for testing the impacts of management practices on C sequestration in and trace gas emissions from the wetlands. The Marcell wetland as reported in Zhang and others (2002) is located in Minnesota (latitude 47.5°N), and the other wetland is in Florida (latitude 29.0°N). The 150-year climate scenarios for the Minnesota (MN) and Florida (FL) sites were constructed by repeating the 10-year (1990–1999) meteorological data. A 150-year management scenario was designed to include deforestation and wetland restoration for the two sites. The scenario started with a 90-year-old pine forest growing in soil with a 1-m-thick organic layer (207 tons of SOC) in the forest floor and 0.5 m mineral soil underneath the forest floor. The water table seasonally fluctuated around the ground surface from −20 cm to +20 cm. The pine forest had a 10-year-old understory (spruce). In May of year 50, deforestation practices were applied, which included harvest of the pine forest and drainage of the ground water to keep the water table at 1 m depth under the ground surface. During the following 50 years after deforestation, the drained lands were left fallow with only the understory (spruce) growing. In year 100, the drainage was terminated to allow the water table dynamics to return to the natural condition, the understory (spruce) was chopped for bed preparation, and new pine trees were planted. The same management scenario was applied for the two tested sites, MN and FL, to observe how climatic conditions could affect the impacts of management on C sequestration and trace

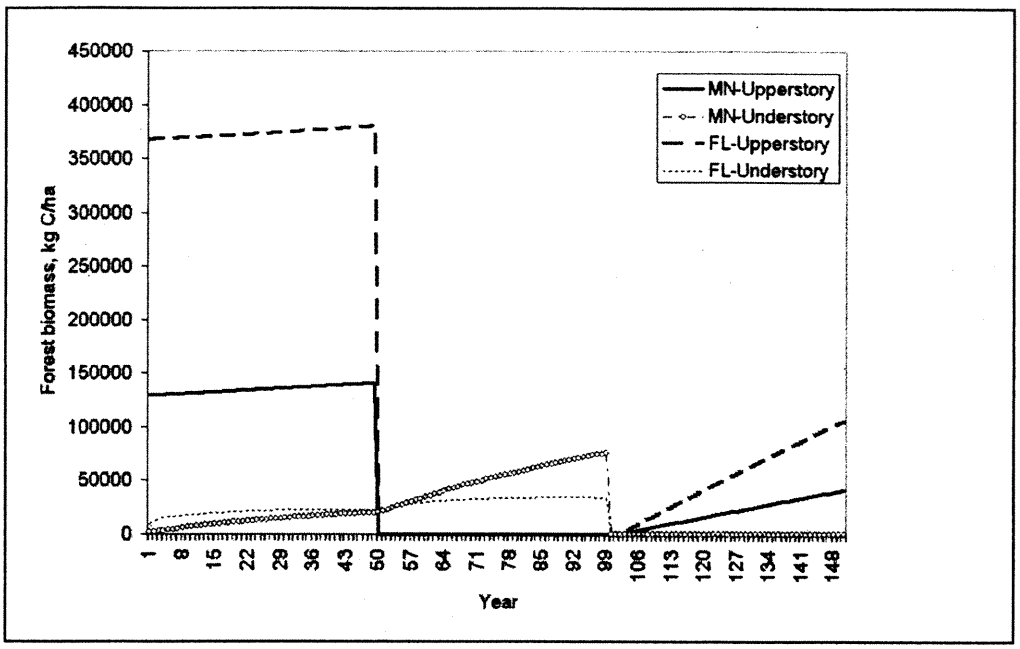

Figure 2. Modeled 150-year forest biomass dynamics at the wetland sites in Minnesota (MN) and Florida (FL). The upperstory (pine) at the FL site (thick dotted line) grew faster than that at the MN site (thick solid line) under the wetland conditions in years 1–50 and 100–150 due to the warmer weather in FL. Growth rates of the understory (spruce) increased after harvest of the upperstory in year 50 at both MN (thin line with circles) and FL (dotted thin line), although the spruce grew much better under the cool weather conditions at MN.

gas emissions for the boreal and sub-tropical forested wetland ecosystems.

Results

Forest Biomass

The simulated scenario started from a pine forest at age 90 growing at the MN and FL sites. The initial forest biomass, which was estimated by Wetland-DNDC as default, was higher at the FL site due to the favorable climate conditions. During the simulated first 50 years, the forest growth rates were 237 and 262 kg C/ha/yr for MN and FL, respectively. During the drained years from years 51 to 100, there was only the understory (spruce) growing with rates 1118 and 227 kg C/ha/yr for MN and FL, respectively. The spruce grew better at the cool MN site than at the hot FL site in the draining years. At MN, the clear-cutting of the upperstory (pine) apparently created better conditions for the understory (spruce) to grow (Figure 2). After year 100 when the new pine trees were planted, the pine growth rate at FL was almost double of that at MN due to the higher radiation and temperature.

Soil Organic Carbon

Soil C storage is determined by the balance between litter input and soil heterotrophic respiration. In the simulated first 50 years, since soil respiration at the both sites was depressed due to the year around submerged conditions, the soil C accumulation rate was mainly driven by litter incorporation. The Wetland-DNDC-modeled soil C accumulation rate at the FL site (3873 kg C/ha/yr) was higher than that at the MN site (786 kg C/ha/yr) due to the difference in their litter production rates. After the forest harvest with drainage set up in year 51, soil aeration became dominant. The soil C storage at the both sites rapidly decreased, although the FL site had a higher C loss rate (5959 kg C/ha/yr) than MN (2865 kg C/ha/yr) due to the difference in temperature. At the end of the draining period (i.e., year 100), SOC contents at the two sites approached a similar level (775 tons C/ha). During the 50 years of drainage/deforestation, the FL site lost 292 tons C/ha, and the MN site lost 140 tons C/ha. After the wetland restoration, soil C storage resumed at the both sites, although with a faster rate in the FL site. In years 50 and 100, there were sudden increases in SOC

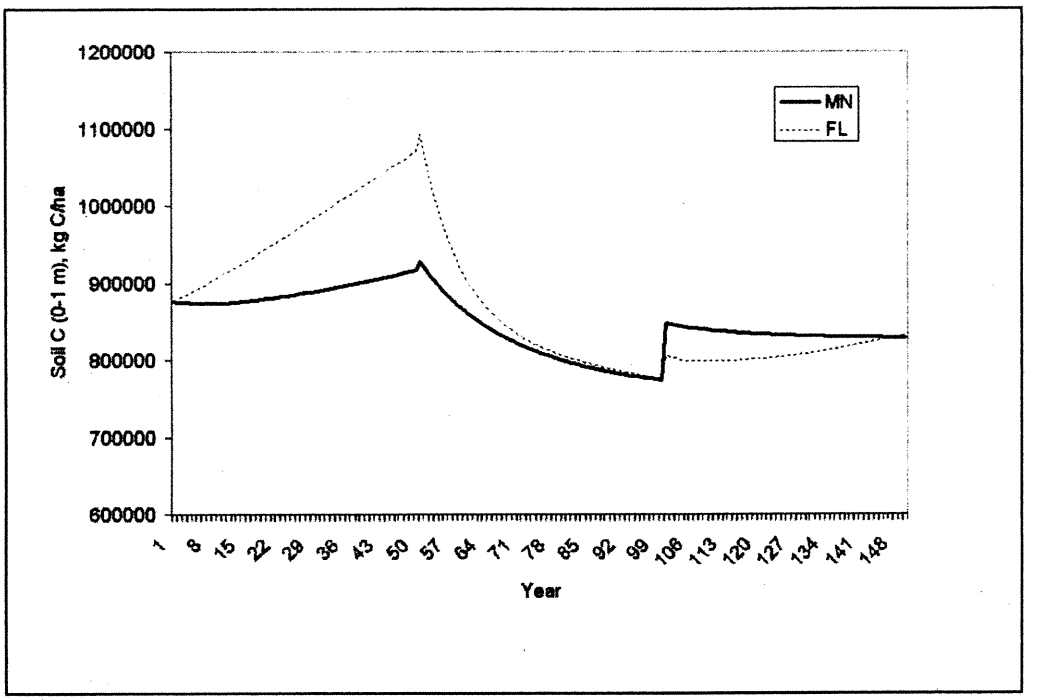

Figure 3. Modeled 150-year soil organic carbon (SOC) dynamics at the wetland sites in Minnesota (MN) and Florida (FL). SOC was accumulated under wetland conditions in years 1–50, decreased during the draining period from 51–100 years, and resumed after the wetland was reinstalled in year 100 at MN (thick line) and FL (thin dotted line) sites. The sudden increase in SOC in year 100 at the both sites were caused by chopping of the understory (spruce) as part of the bed preparation before replanting pine in year 100.

due to the litter incorporation following deforestation of the upperstory and chopping of the understory (Figure 3). Since the understory (spruce) grew very well at MN, a lot of litter was left and incorporated in the soil after chopping in year 100. That caused slight decrease in SOC even after reinstallation of the wetland.

Net Emission of CO_2

Carbon dioxide emission is defined as net ecosystem C exchange in this study. Daily CO_2 emission is the difference between forest net primary production (NPP) and soil heterotrophic respiration. Annual CO_2 emission is obtained by summing up the daily net CO_2 fluxes. We did not count deforestation as a direct source of atmospheric CO_2, as the harvested wood may not be converted to CO_2 immediately. Figure 4 indicates: (1) both MN and FL were sinks of atmospheric CO_2 with average sink rates 1076 and 4260 kg C/ha/yr, respectively, in years 1–50; (2) the MN and FL sites became sources with average emission rates of 2049 and 5450 kg C/ha/yr, respectively, in years 51–100 when the soils were drained, and (3) the two sites became sinks again with rates of 444 and 2647 kg C/ha/yr for MN and FL, respectively, as soon as the wetland was reinstalled in year 100 (Table 1). During the simulated 150 years with the wetland–draining–wetland scenario, the MN site lost 26 tons C/ha into the atmosphere, and the FL site sequestered 1.5 tons C/ha from the atmosphere.

CH_4 Emissions

Wetland-DNDC simulates CH_4 emission or oxidation based on the modeled soil Eh, soil temperature, and available C from root and microbial respiration (Zhang and others 2002, Li 2000). Methane can be produced from the saturated soil layers. As soon as a soil layer turns into aeration, it will become a sink of CH_4. Under the wetland conditions in years 1–50, MN and FL were sources of atmospheric CH_4, with average annual emission rates of 57 and 785 kg CH_4/ha, respectively. The modeled CH_4 rates were much higher in FL than in MN because of the higher temperature and tree growth rate. During years 101–150, the MN and FL sites became weak sinks of atmospheric CH_4. The FL soil oxidized more CH_4 due to the higher methane oxidation activity under the higher temperature. As soon as

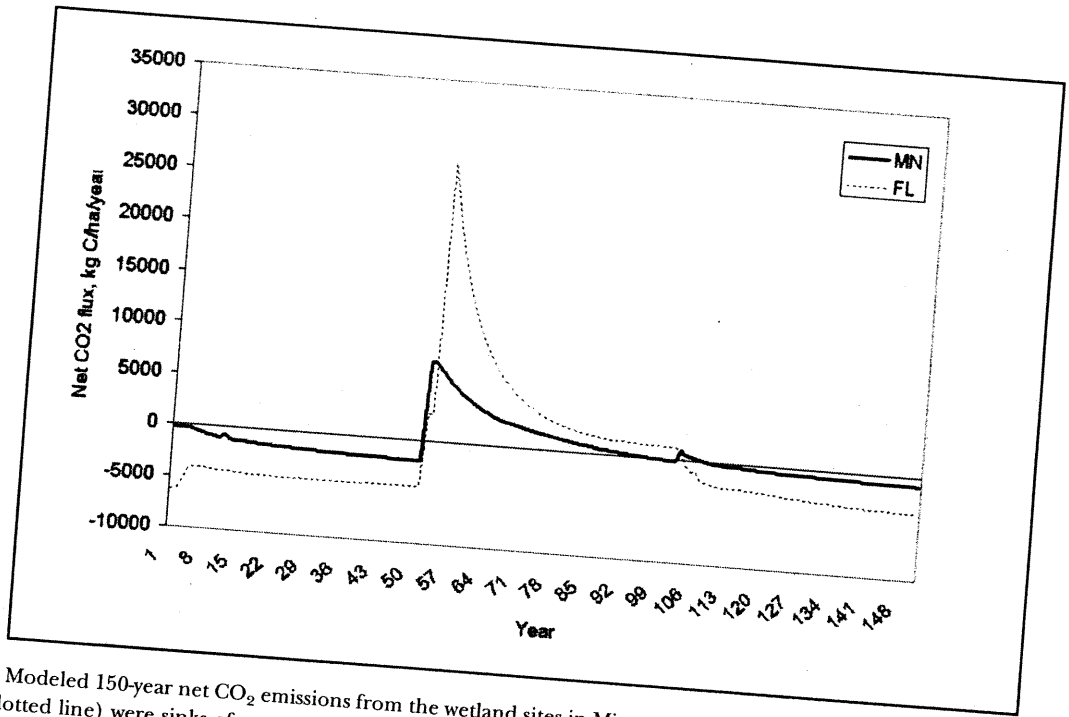

Figure 4. Modeled 150-year net CO_2 emissions from the wetland sites in Minnesota (MN) and Florida (FL). MN (thick line) and FL (thin dotted line) were sinks of atmospheric CO_2 under wetland conditions in years 1–50 and 100–150, but became sources due to the accelerated decomposition of the organic soils when the soils were drained from year 50 to 100.

Table 1. Modeled greenhouse emissions from wetland sites in Minnesota (MN) and Florida (FL) during different management stages in 150-year simulations[a]

	Average annual gas flux and global warming potential (GWP)								
Management stage	Wetland forest Year 1–50			Deforestation/drainage Year 51–100			Wetland restoration Year 101–150		
	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O
MN site	−1076	56.7	0.7	2049	−7.3	6.9	−444	40.9	0.01
Net GWP	(−3945)	(1587)	(358)	(7514)	(−204)	(3373)	(−1628)	(1146)	(3)
FL site	−4260	−2001			10684			−479	
Net GWP	(−15620)	784.8	0.6	5450	−14.7	5.3	−2647	676	0.00
		(21974)	(270)	(19982)	(−412)	(2587)	(−9707)	(18939)	(1)
		6623			22157			9233	

[a]Units for average annual CO_2, CH_4, N_2O, and GWP fluxes are kg C/ha, kg C/ha, kg N/ha, and kg CO_2 emission equivalents/ha/yr, respectively.

the wetlands were reinstalled, MN and FL became CH_4 sources again, with emission rates of 41 and 676 kg C/ha/yr (Figure 5 and Table 1). During the entire simulated 150 years, MN and FL emitted 4880 and 73,040 kg CH_4C/ha, respectively.

N_2O emissions

Wetland-DNDC simulates N_2O emissions based on soil Eh, temperature, and concentrations of ammonium and nitrate. Ammonium and nitrate contents are calculated based on decomposition and nitrification rates (Li 2000, Zhang and others 2002, Li and others 2000). Under submerged conditions, soil Eh is low, and decomposition and nitrification are depressed. The deep anaerobic conditions drive the sequential reactions of denitrification all the way to its end product, N_2. As soon as the soil is drained, decomposition and nitrification resume, and the freshly produced nitrate will fuel denitrification to produce N_2O. During wetland years 1–50 and 100–150, the modeled N_2O fluxes

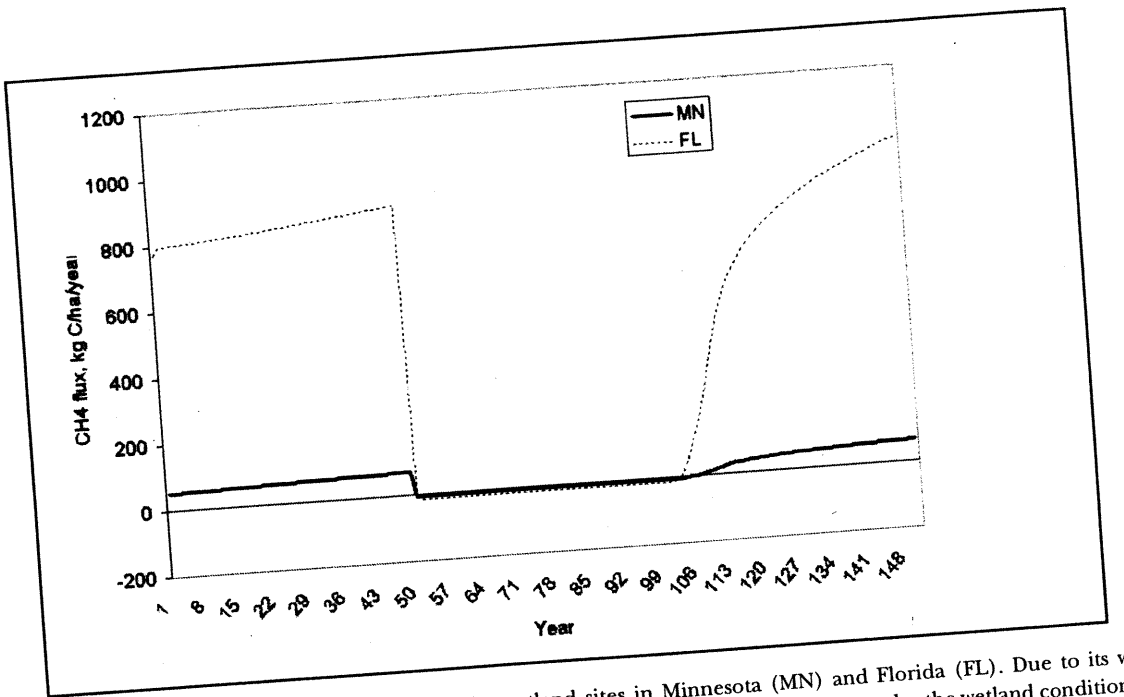

Figure 5. Modeled 150-year CH_4 emissions from the wetland sites in Minnesota (MN) and Florida (FL). Due to its warmer weather, FL (thin dotted line) was a stronger source of atmospheric CH_4 than MN (thick line) under the wetland conditions from year 1 to 50. Both sites became weak sinks when the soils were drained during years 51–100, but became sources again after the wetland reinstallation in years 100–150.

were very low (0.01–0.7 kg N/ha/yr). As soon as the soils were drained, rapid decomposition of the accumulated labile SOC released a large amount of DOC, ammonium, and nitrate, which stimulated both nitrification and denitrification to elevate N_2O emissions. During the draining years 50–100, MN and FL emitted 345 and 265 kg N/ha, with average annual rates of 6.9 and 5.3 kg N/ha, respectively (Table 1). The annual emission rates were higher at beginning of the draining period, and gradually decreased along with the consumption of the labile SOC (Figure 6). The modeled high N_2O emissions caused by the drainage lasted for about 40 years, with the highest rate of 14 kg N/ha in year 56 for MN. In comparison, the N_2O peak at the FL site was higher (20 kg N/ha/yr) but lasted less time (20 years). The difference in N_2O dynamics between MN and FL was related to the SOC decomposition rates in the two climatic zones. The total N_2O fluxes during the draining years were 380 and 295 kg N/ha for MN and FL, respectively. The modeled high N_2O fluxes at the drained organic soils were consistent with field observations. Terry and others (1981) measured very high N_2O fluxes from an organic soil in a Florida farmland, which was converted from a wetland in the Everglades.

The modeled results indicated substantial differences between MN and FL regarding their C sequestration capacity, CH_4 fluxes, N_2O fluxes during the simulated 150 years under a same management scenario.

Discussion

Since a change in management practices can simultaneously affect C sequestration and CH_4 and N_2O fluxes, the net effect of each practice on global warming will need to be assessed. We define the net impact as the sum of the warming forces of all the three greenhouse gases based on the concept of global warming potential (GWP). According to IPCC (1997), with a 100-year time horizon, the warming forces of 1 kg CH_4 and N_2O are 21 and 310 times higher than that of 1 kg CO_2. The GWP value for each scenario is calculated as follows:

$$GWP_i = CO_{2i}/12*44 + N_2O_i/28*44*310 + CH_{4i}/12*16*21 \quad (4)$$

where GWP_i (kg CO_2 equivalent/ha/yr) is the global warming potential induced by scenario i; $CO_{2\,i}$, N_2O_i, and $CH_{4\,i}$ are CO_2 flux (kg C/ha/yr), N_2O flux (kg N/ha/yr) and CH_4 flux (kg C/ha/yr), respectively, induced by scenario i.

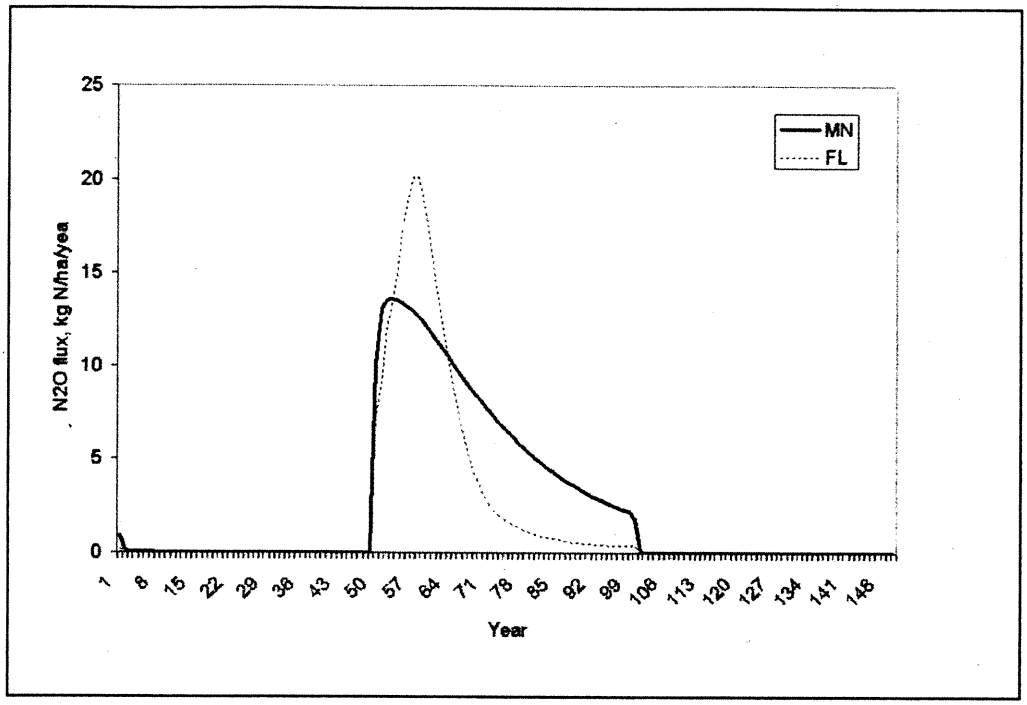

Figure 6. Modeled 150-year N_2O emissions from the wetland sites in Minnesota (MN) and Florida (FL). The both sites emitted little N_2O under submerged conditions in years 1–50 and 100–150. When the soils were drained after year 50, surges of N_2O emissions appeared at the both sites due to the suddenly increased decomposition, nitrification, and denitrification. Driven by the warmer weather, the drainage-induced N_2O surges at FL (thin dotted line) were higher but lasted shorter than that at MN (thick line).

The CO_2 item stands for net C loss from the ecosystem, which is equivalent to the modeled net CO_2 emission rate. A negative GWP value means the ecosystem mitigates global warming, and a positive GWP value means elevates global warming (Table 1).

During the wetland forest period (year 1–50), the MN site mitigated global warming while the FL site elevated global warming (Table 1). During this time period, both MN and FL sequestered atmospheric CO_2 and emitted CH_4. FL sequestered more C (4260 kg C/ha/yr) than did MN (1076 kg C/ha/yr) but emitted more CH_4 (785 kg C/ha/yr) than MN (57 kg C/ha/yr). At the MN site, C sequestration overwhelmed CH_4 emissions. At the FL site, the CH_4 emissions offset all of the benefit gained from the C sequestration (Table 1).

During the deforestation/drainage period (year 51–100), both MN and FL elevated global warming although the contribution from FL was double of that from MN (Figure 7). The two sites emitted similar amounts of N_2O, although the loss of SOC was 2.6 times higher in FL than in MN. Soil C loss dominated the GWP values during this draining period although N_2O had significant contributions. During the first 10–20 years following the deforestation and drainage in year 50, the sudden increase in GWP at FL and MN was almost equally attributed to CO_2 and N_2O fluxes (Figure 7). Oxidation of CH_4 had a relatively small contribution to GWP in this time period for both sites.

The reinstallation of wetland brought the water table back to its natural state, which was above the ground surface seasonally. That immediately stimulated CH_4 production, although the production rates gradually increased along with the increase in SOC content and plant root activity (Figure 5). The CH_4 increase rate in FL was much higher than that in MN due to the higher temperatures as well as the higher rates of SOC accumulation and root development under the subtropical climate conditions (Figure 5). During the reinstallation period (year 101–150), SOC loss was substantially moderated. At FL, CH_4 dominated the GWP values. In contrast, MN still remained a mitigator (Table 1). After wetland restoration in year 100, the MN site continued to be a sink while the FL site rapidly became a source of atmospheric CO_2 equivalent.

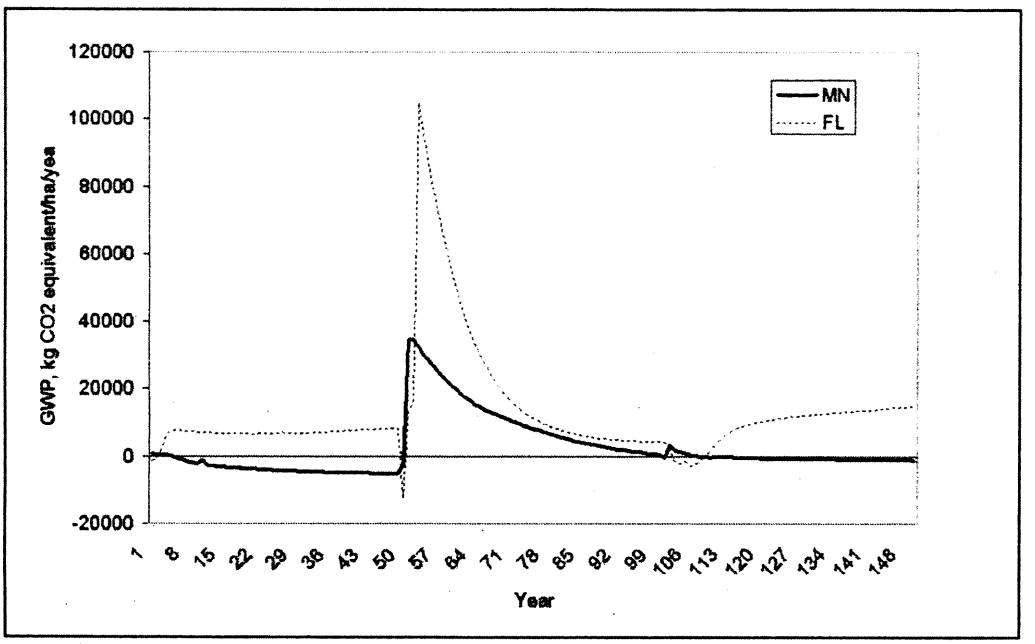

Figure 7. Modeled 150-year global warming potential (GWP) of the wetland sites in Minnesota (MN) and Florida (FL). The MN site (thick line) mitigated global warming under the wetland conditions (years 1–50 and 100–150), while the effect was reversed when the soil was drained in years 50–100. The FL site (thin dotted line) always increased global warming during the simulated 150 years, although the effect was greatly enhanced during the draining years. High CH_4 and CO_2 emissions from FL dominated GWP from the subtropical site under the wetland and draining conditions, respectively.

Conclusion

During the simulated 150 years with a wetland forest–drainage–reinstallation scenario, the total contributions of the MN and FL sites to global warming were 410 and 1900 tons CO_2 emission equivalents/ha, respectively. The MN wetland mitigated global warming except when the land was drained. The FL wetland always elevated global warming due to its high CH_4 emissions under wetland conditions and high CO_2 emissions under draining conditions. We concluded that (1) the same management scenario may have very different effects on global warming when applied at different locations with different climate/soil/vegetation conditions; and (2) counting only C sequestration may not be adequate to assess impacts of management practices on global warming.

The case study with two wetland sites in boreal and subtropical areas demonstrates a complex picture of the impacts on global warming from management alternatives applied to forested wetlands across climatic zones even though the two sites shared the same forest type, soil properties, and management scenario. If several factors (e.g., forest type, water table, soil texture, management, climate, etc.) are simultaneously altered, the impacts of management practices would become more complex. The findings of this study confirm the importance of the water table position, and associated aerobic and anaerobic zones, in controlling CO_2, CH_4, and N_2O emissions from forest wetlands. Process-based models, such as Wetland-DNDC, provide a powerful tool to test management strategies across climatic zones, hydrological regimes, soil properties, and forest types. We present the simulation results of forest management activities that can be implemented to mitigate carbon emissions, and thus serve as potential carbon-offset projects, and the environmental benefits that accrue in addition to mitigating carbon emissions. The model's prediction within the 150-year period indicated that this modeling system can be used to predict long-term cumulative carbon dynamics and provide a means to understand how different parts of the ecosystem respond to environmental drivers, thereby improving both strategic and tactical planning for managing natural resources in a sustainable and environmentally sound manner. To apply the Wetland-DNDC model to a landscape or watershed scale, we will make new efforts to develop GIS databases, which will store and provide spatially differentiated information on climate, topog-

raphy, hydrology, soil, vegetation, and management to support the model runs at a large scale. Upscaling of the model predictions will produce more valuable information to understand the role of managed wetlands and to assist in the assessment of alternatives and policy making.

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