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Modeling water, carbon, and nitrogen dynamics for two drained pine plantations under intensive management practices

Shiying Tian [a,*], Mohamed A. Youssef [a], R. Wayne Skaggs [a], Devendra M. Amatya [b], George M. Chescheir [a]

[a] Department of Biological and Agricultural Engineering, NCSU. D.S. Weaver Labs, Campus Box 7625, Raleigh, NC 27695, United States
[b] USDA Forest Service, Center for Forested Wetlands Research, 3734 Highway 402, Cordesville, SC 29434, United States

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ABSTRACT

This paper reports results of a study to test the reliability of the DRAINMOD-FOREST model for predicting water, soil carbon (C) and nitrogen (N) dynamics in intensively managed forests. The study site, two adjacent loblolly pine (*Pinus taeda* L.) plantations (referred as D2 and D3), are located in the coastal plain of North Carolina, USA. Controlled drainage (with weir and orifice) and various silvicultural practices, including nitrogen (N) fertilizer application, thinning, harvesting, bedding, and replanting, were conducted on the study site. Continuous collection of hydrological and water quality data (1988–2008) were used for model evaluation. Comparison between predicted and measured hydrologic variables showed that the model accurately predicted long-term subsurface drainage dynamics and water table fluctuations in both loblolly pine plantations. Predicted mean and standard deviation of annual drainage matched measured values very well: 431 ± 217 *vs.* 436 ± 231 mm for D2 site and 384 ± 152 *vs.* 386 ± 160 mm for D3 site. Nash–Sutcliffe coefficients (NSE) were above 0.9 for drainage predictions on annual and monthly basis and above 0.86 for predictions of daily water table fluctuations. Compared to measurements in other similar studies, the model also reasonably estimated long-term dynamics of organic matter pools on forest floor and in forest soil. Predicted mean and standard deviation of annual nitrate exports were comparable to measured values: 1.6 ± 1.3 *vs.* 1.5 ± 1.5 kg ha^{-1} for D2 site, and 1.4 ± 1.3 *vs.* 1.3 ± 1.1 kg ha^{-1} for D3 site, respectively. Predicted nitrate export dynamics were also in excellent agreement with field measurements as indicated by NSE above 0.90 and 0.84 on annual and monthly bases, respectively. The model, thus successfully tested, was applied to predicted hydrological and biogeochemical responses to drainage water management and silvicultural practices. Specifically, the model predicted reduced rainfall interception and ET after clear cutting, both of which led to increased water yield and elevated water table, as expected. The model also captured temporary changes in nitrogen transformations following forest harvesting, including increased mineralization, nitrification, denitrification, and decreased plant uptake. Overall, this study demonstrated that DRAINMOD-FOREST can predict water, C and N dynamics in drained pine forests under intensive management practices.

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1. Introduction

A large portion of forested lands in coastal Southeastern USA are plantation forests (Smith et al., 2004). These forests are usually intensively managed with silvicultural practices such as fertilization, thinning, harvesting, site preparation, bedding, and regeneration. Large areas of these plantations are on naturally poorly drained soils and require artificial drainage (drainage ditches) to improve trafficability and increase forest productivity. Intensively managed plantation forests may have adverse environmental impacts including contamination of receiving water bodies (Amatya et al., 1998; Nieminen, 2004; Stednick, 2008; Beltran et al., 2010), affecting hydrological services of forest ecosystems (Bosch and Hewlett, 1982; Sun et al., 2001; Brown et al., 2005; Moore and Wondzell, 2005; Eisenbies et al., 2007; NRC, 2008), and altering forest soil carbon (C) and nitrogen (N) (Johnson and Curtis, 2001; Johnson et al., 2002, 2003). These potential risks may impair the ecological functions of managed forest ecosystems and consequently impede their long-term sustainability (Peng et al., 2002; Driscoll et al., 2003; Gundersen et al., 2006; Diochon et al., 2009). Fortunately, both onsite and offsite negative impacts of forest management can be minimized through implementation of best management practices (Amatya et al., 1998; Wynn et al., 2000; Johnson and Curtis, 2001; Sun et al., 2001; Fox et al., 2007). Nevertheless, developing environmentally "friendly" management strategies relies upon our knowledge and understanding of processes and mechanisms controlling the hydrological and biogeochemical responses of managed forests to silvicultural and/or land management practices.

* Corresponding author. Tel.: +1 919 986 0048; fax: +1 919 515 7760.
E-mail addresses: stain@ncsu.edu, stian.tsy@gmail.com (S. Tian).

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Experimental studies have been useful in providing valuable information about the complex and interacting physical, chemical, and biological processes influencing the hydrology, biogeochemistry, and productivity of intensively managed forest ecosystems (Bosch and Hewlett, 1982; Amatya et al., 1996, 1998; Johnson and Curtis, 2001; Johnson et al., 2002, 2003; Sun et al., 2001; Brown et al., 2005; Nieminen, 2004; Moore and Wondzell, 2005; Eisenbies et al., 2007; NRC, 2008; Stednick, 2008; Beltran et al., 2010). Many forest ecosystem models have also been developed and utilized to reproduce, verify, and extend experimental findings across various spatiotemporal scales. Most of these modeling studies have focused on predicting effects of forest harvesting on either forest hydrology (McCarthy and Skaggs, 1992; Storck et al., 1998; Sun et al., 1998, 2000; Lavigne et al., 2004) or biogeochemistry (Romanya et al., 2000; Aber et al., 2002; Peng et al., 2002; Laurén et al., 2005; Palosuo et al., 2008; Johnson et al., 2010; Dai et al., 2011). These modeling studies demonstrated that computer models are useful and essential scientific tools for evaluating long-term effects of management practices on the hydrology and biogeochemistry of managed forest ecosystems. However, current models rarely integrate comprehensive modeling for both hydrological and biogeochemical dynamics in forest ecosystems under intensive management practices. In other words, current models are seldom "well balanced" in representing the water, C, and N cycles with comparable levels of detail (Tiktak and van Grinsven, 1995; Landsberg, 2003). This is expected to limit the ability of many of these models to fully address responses of forest ecosystems to changes in climate, land use, and/or management practices (Wallman et al., 2005; Waring and Running, 2007).

Until recently, comprehensive mechanistic models have been seldom developed and applied to predict water, C and N dynamics in artificially drained forest ecosystems under intensive silvicultural practices, despite their large area and proximity to nutrient sensitive surface waters (Amatya et al., 1998). The DRAINMOD-FOREST model was developed to simulate hydrologic, C and N cycles in drained forests under various silvicultural and land management practices (Tian, 2011; Tian et al., 2009). It is a field scale, quasi-process based model that integrates a forest growth model to the hydrologic model, DRAINMOD (Skaggs, 1978; 1999), and the soil C and N dynamics model, DRAINMOD-N II (Youssef et al., 2005). The model has been successfully tested for an artificially drained loblolly pine (*Pinus taeda L.*) plantation under limited management practices (Tian et al., 2010). It is essential to further test the model for drained forests under intensive silvicultural practices, characteristic to much of the plantation sites in the southeastern US. The objective of this study, therefore, was to evaluate the performance of the DRAINMOD-FOREST model for predicting water, soil C, and N cycling in intensively managed forests. This field evaluation of the model was conducted using a 21-year data set (from 1988 to 2008) collected from two intensively managed loblolly pine plantations located in Eastern North Carolina, United States.

2. Materials and methods

2.1. Study sites

The data set used for model testing was collected from two intensively managed loblolly pine plantations (referred to as D2 and D3), adjacent to a reference site (referred to as D1). The research sites are located in the Atlantic Coastal Plain of North Carolina, USA (34° 48' N, 76° 42' W). Each site is approximately 25 ha and relatively flat (<0.1% slope). Long-term mean annual precipitation at the site is 1517 mm and mean annual evapotranspiration is about 1057 mm (Amatya and Skaggs, in press; Tian et al., 2010). The soil at the site is hydric and naturally poorly drained (deloss fine sandy loam). The two small watersheds are artificially drained by four 1.2 m deep parallel lateral ditches spaced 100 m apart (Fig. 1). Loblolly pine trees were planted in 1974 at a density of 2100 trees ha^{-1}. Both sites underwent a pre-commercial thinning (thinned to about 988 trees ha^{-1}) and a commercial thinning (thinned to about 370 trees ha^{-1}) in 1981 and late 1988, respectively. After the second thinning operation, Urea-N fertilizer (225 kg N ha^{-1}) was applied to both sites in 1989 (Amatya et al., 1996, 1998). In October 1995, D2 site was clear cut, followed by site preparations in spring of 1996 and regeneration in early 1997 at a density of about 2100 trees ha^{-1} (Amatya et al., 2006) In 2002, D3 was thinned for the third time to reduce the stocking number to approximately 185 trees ha^{-1} (Amatya and Skaggs, 2008) In 2005, both D2 and D3 sites received urea-N fertilizer at rates of 118 and 175 kg N ha^{-1}, respectively (Beltran et al., 2010). In addition to these silvicultural practices, water management practices including controlled drainage and orifice weir treatment were also implemented at these sites in early 1990s. The controlled drainage was carried out by adjusting weir elevation at the outlet of the main ditch draining each watershed. The orifice weir treatment was implemented by installing a rectangular weir with an orifice near the ditch bottom to dampen peak drainage rates during storm events (Amatya et al., 2003). History and management practices of the two sites are summarized in Table 1. The reader is referred to McCarthy (1990), Amatya et al. (1996) and Amatya et al. (2003) for a detailed description of the study sites.

2.2. Data collection

Rainfall was measured with a tipping bucket rain gauge located in an open area of each watershed (Fig. 1). Additional manual gauges have been used to verify rainfall data measured by the automatic rain gauge (Amatya and Skaggs, in press; Amatya et al., 1996). Other climatological data including air temperature, relative humidity, wind speed and direction, and solar and net radiation were measured and recorded every half-hour. Before middle of 1997, the weather data were obtained from a station located about 800 m away from the site. Thereafter, weather data had been collected using an on-site station until September 2005 when Hurricane Ophelia damaged the station's tower. Weather data between 2005 and 2008 were obtained from another station, about 2 miles away from the site. Since August 2008, the weather data have been collected using a new 3 m high on-site weather station (Amatya and Skaggs, in press).

Drainage flow rates were measured using a 120° V-notch weir mounted on a water level control structure installed at the outlet of the collector ditch of each site. The bottom of the V-notch weir was placed about 1.2 m below the average ground surface. Automatic stage recorders were installed to record the head upstream and downstream of the weir. Groundwater table elevations were measured using two wells equipped with automatic water level recorders, located at two experimental plots midway between the inner field ditches of each watershed (Fig. 1). Detailed description of the hydrologic measurements is given elsewhere (Amatya et al., 1996, 2003).

Drainage water quality was intensively monitored during 1989–1994. Automatic ISCO-2700 samplers were used to collect drainage water samples every 2 h during each storm event. Every four consecutive samples were mixed to make one composite sample, making three water quality samples per day. Grab samples were collected every two weeks during flow events for the whole study period. Water samples were analyzed for nitrate and nitrite, ammonium and dissolved organic nitrogen (TKN), phosphorous, and sediments. Detailed procedures of event sampling and laboratory analyses are documented by Amatya et al. (1998, 2003) and

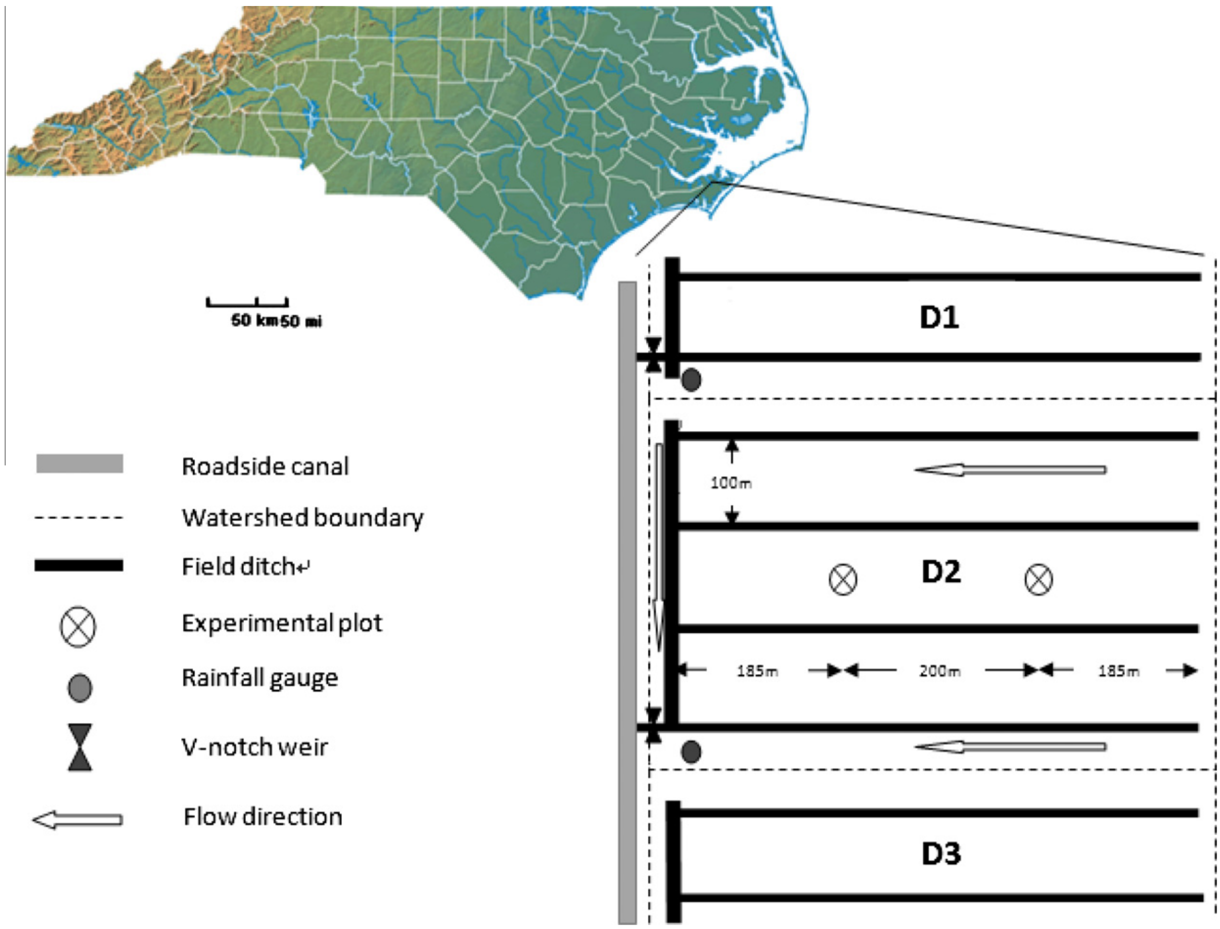

Fig. 1. Location (up right: Carteret County, North Carolina, USA) and schematic diagram of the study watersheds (after Amatya and Skaggs, 2001). Experimental design of the three sites D1,D2 and D3 are identical.

Table 1
A summary of management practices implemented at the study sites.

Managements	Time	D2[a]	D3[a]	Description
Plantation	1974	✓	✓	2100 trees ha^{-1}
Pre-commercial thinning	1980	✓	✓	Thinned to about 1000 trees ha^{-1}
Fertilization	1981	✓	✓	Aerial applied 169 kg N/ha
Commercial thinning	1988	✓	✓	Thinned to about 400 trees ha^{-1}
Fertilizer application	1989	✓	✓	Ground applied 225 kg N/ha
Control drainage	1990–1994	✓	✓	Weir raised during growing season
Harvesting	1995	✓		Whole tree cutting
Orifice control drainage	1995–1999		✓	Hole (4 in. diameter) placed near weir bottom
Site preparation	1996	✓		Bedding
Regeneration	1997	✓		2100 trees ha^{-1}
Thinning	2002		✓	Thinned to about 185 trees ha^{-1}
Fertilization	2005	✓	✓	Aerial applied 118 (D2) and 175 (D3) kg N ha^{-1}

[a] "✓" means the practice was implemented at the site.

Beltran et al. (2010). Daily loads of nitrate only in this study were calculated by multiplying measured daily drainage by daily effluent nitrate concentration. Monthly and annual nitrate loads were estimated by summing daily loads (Amatya and Skaggs, in press).

2.3. Model description

The DRAINMOD-FOREST model is a comprehensive, quasi process-based, and stand level model that integrates a forest growth model with DRAINMOD (Skaggs, 1978, 1999) and DRAINMOD-N II (Youssef, 2003; Youssef et al., 2005) models to simulate hydrological processes, soil C and N dynamics, and tree growth for drained forest lands under silvicultural and water management practices (Tian, 2011; Tian et al., 2009). DRAINMOD conducts a water balance at the soil surface and along a soil column midway between two parallel drains on a daily or hourly basis. At each time step, simulated key hydrological processes include rainfall interception, throughfall, infiltration, evapotranspiration (ET), subsurface drainage, surface runoff, subirrigation, deep and lateral seepage, water table fluctuation, and soil water distribution in the vadose zone. Infiltration is estimated using the Green-Ampt equation. Subsurface drainage rates are calculated using the Hooghoudt's equation for water table drawdown and Kirkham's equations for ponded surface conditions (Skaggs, 1999). Surface runoff is estimated as the difference between throughfall and infiltration rates, and a function of a site specific depressional storage, which must be filled before runoff can start (Skaggs, 1999). A detailed description of DRAINMOD is given by Skaggs (1978, 1999). In DRAINMOD-FOREST, rainfall interception was simulated using the sparse Gash's method (Tian et al., 2009). Daily PET is simulated using Penman–Monteith method (Monteith, 1965) with canopy conductance estimated as a function of climatologically-regulated stomatal conductance and leaf area index (LAI) that is predicted by the new forest growth model (Tian et al., 2009). The standard FAO method for tall grasses (Allen et al., 1998) was used to simulate PET during the period between forest harvesting and regeneration.

DRAINMOD-N II simulates a detailed N cycle including atmospheric deposition, application of mineral N fertilizers, and organic N sources, plant uptake, mineralization, immobilization, nitrification, denitrification, ammonia volatilization, and N losses via subsurface drainage and surface runoff (Youssef, 2003; Youssef et al., 2005). Reactive transport of inorganic N is simulated using a multiphase form of the one dimensional advection–dispersion-reaction equation. Soil C dynamics is simulated using a soil C submodel adapted from the CENTURY model (Parton et al., 1993). The soil C submodel divides organic matter into three soil pools (active, slow, and passive), two above- and below-ground residue pools (metabolic and structural), and a surface microbial pool. Each organic matter pool is characterized by user specified organic C content, potential rate of decomposition, and C:N ratio (Youssef et al., 2005). The reader is referred to Youssef (2003) and Youssef et al. (2005) for a detailed description of DRAINMOD-N II model.

The tree growth and forest productivity component is a stand level model that divides the forest canopy into two or three layers depending upon the tree height and stocking number. Radiation use efficiency methods are used to simulate gross primary production (GPP) as a function of intercepted radiation, air temperature, and the availability of soil water and nitrogen (Tian et al., 2009). Intercepted radiation was simulated using Beer–Lambert law as a function of LAI and canopy fraction. The net primary production (NPP) of midstory and upstory species can either be estimated as a constant fraction of GPP or determined as the difference between GPP and plant respiration (the total of maintenance, construction, and growth respiration). The NPP of the two understory groups considered in the model (shade-tolerant and shade-intolerant species) is estimated empirically in terms of light availability and a user defined maximum productivity. The model partitions C assimilated by trees into foliage, stem, and root biomass using tree species-dependent allometric functions, which are also regulated by soil water and nitrogen status. Carbon loss through foliage litterfall is estimated as a function of leaf longevity, while fine root turnover is quantified based on fine root lifespan. DRAINMOD-FOREST simulates the effects of commonly used silvicultural practices on hydrological and biogeochemical processes in the forest ecosystem. Thinning, pruning and harvesting remove certain live biomass based on user assigned management intensity. The amount of foliage, woody litter, and dead root biomass produced during these practices are estimated using user-specified "left" fractions and added to surface and below-ground litter pools of the soil C and N model DRAINMOD-NII. The effects of site preparation on mixing surface litter and enhancing the decomposition of OC within the topsoil are simulated using the tillage and residue management component of DRAINMOD-N II (Youssef et al., 2005). The fertilizer application is also simulated using the fertilizer component of the DRAINMOD-NII model. The model simulates short-term processes associated with fertilizer application including fertilizer dissolution, urea hydrolysis, and pH changes following the application of urea. Fertilizer dissolution follows a zero-order rate once soil water content reaches a threshold value. Urea hydrolysis is simulated using Michaelis–Menten kinetics. Water management practices such as artificial drainage, subirrigation, and controlled drainage are simulated using the hydrologic model DRAINMOD (Skaggs, 1999). The forest growth model of DRAINMOD-FOREST is described in detail by Tian (2011) and Tian et al. (2011).

The three component models DRAINMOD, DRAINMOD-NII, and the forest growth model are fully integrated with internal feedback reflecting the interaction among soil water, soil C and N, and vegetation. LAI, plant height, and canopy fraction predicted by the forest growth model are used by DRAINMOD to predict potential evapotranspiration (PET) using the Penman–Monteith equation (Monteith, 1965). DRAINMOD predicted hydrological variables including soil water conditions and drainage are used by DRAINDMOD-N II model to simulate the reactive transport of soil N and predict mineral N leaching losses. Predicted ET and PET are used for representing water stress when simulating canopy photosynthesis and C allocation under water deficit conditions. Litterfall and root turnover simulated by the plant model are sources of organic matter for the soil C and N cycles simulated by DRAINMOD-N II model, which predicts soil nutrient status for simulating plant growth.

2.4. Model parameterization

Prior to 1990, the three sites D1, D2 and D3 had the same management history (McCarthy, 1990; Amatya et al., 1996). Thus, the initial conditions describing the hydrology, soil C and N, and vegetation for the three sites were assumed to be the same. Tian et al. (2011) calibrated and validated the model using 21 yr data collected from the adjacent site D1, which has been kept under limited silvicultural practices during the entire simulation period. The procedures of model initialization, calibration, and validation are reported in detail by Tian et al. (2011). For this study, the model was calibrated using hydrological (water table depth and drainage) and water quality (nitrate export) data measured for D2 and D3 sites during 1988–1997, and validated using 1998–2008 data. Because of the similar conditions of the three sites (D1, D2 and D3), most of model inputs for simulating the two sites D2 and D3 including drainage system design, hydrologic parameters, soil C and N transformations parameters, and plant related parameters were based on values obtained from the simulation of D1 site (Tian et al., 2010). Model inputs specific to D2 and D3 sites are listed in Table 2. Year of 2003 was excluded from model validation because of inaccurate flow measurements due to frequent weir submergence at the watershed outlet. DRAINMOD was evaluated by comparing predicted daily, monthly, and yearly drainage and daily water table depth to measured values. DRAINMOD-N II was evaluated by comparing measured and predicted nitrate-N losses via drainage water. Nash–Sutcliffe coefficients (NSE), degree of agreements (d), mean absolute error (MAE) and normalized percent error (NPE) were used as goodness-of-fit statistics to assess model performance (Moriasi et al., 2007).

Controlled drainage was implemented at D2 site during the growing season to conserve water for tree growth (Amatya et al., 1996). In D3 site, controlled drainage was used during both spring and growing season to reduce drainage outflows and N loss to downstream surface water (Amatya et al., 1998). Parameters of controlled drainage including timing and weir depth are listed in Table 2. Calibrated hydraulic conductivity values are comparable to estimates by Skaggs et al. (2006). Calibrated drainable porosity of each soil layer was in the range of field measurements conducted by McCarthy (1990) and Blanton et al. (1998). Parameters of harvesting and thinning intensities were set according to field operations. The model user is required to specify fractions left on the ground to simulate the amount of logging residues following thinning and harvesting. In this study, fractions left were

Table 2
Model inputs characterizing silvicultural and water management practices implemented at D2 and D3 sites.

Water management practice	Management period		Weir depth settings (m)	
Controlled drainage in D2 site	Jun-16 to Nov-30		0.6	
Controlled drainage in D3 site	Mar-16 to Jun-15		0.4	
	Jun-16 to Nov-30		0.8	
Soil hydraulic properties				
	0–50 cm	50–100 cm	100–300 cm	
Effective hydraulic conductivity (m d^{-1})	60	56	1.6	
Drainable porosity	0.06	0.06	0.08	
Forest harvesting and thinning				
	Intensity (%)	Left fraction of each tree component (%)		
		Leaf	Stem	Root
Forest harvesting	100	100	5	100
Thinning 1	50	100	5	100
Thinning 2	50	100	5	100
Site preparation in D2 site				
Mixing intensity			65	
Mixing depth (cm)			15	

empirically determined based on values given by Peng et al. (2002) who predicted soil carbon response to different harvesting scenarios using CENTURY model. The stem fraction left of 5% represented the remaining litter of small branches after harvesting and thinning. Site operations could inevitably disturb soil physical and chemical conditions through mixing surface litter into topsoil and accelerating decomposition rates of soil organic matter due to increased soil aeration and increased contact area between organic material and soil particles (Barnes et al., 1998). The model uses a mixing intensity factor, ranges from 0 to 1.0 to quantify the level of mixing and soil disturbance caused by operation associated with site preparation. In this simulation, we used a mixing factor of 0.65, which means that the forest management operation would mix 65% of the litter on the forest floor into the topsoil.

3. Results

3.1. Hydrologic predictions

Both visual (Fig. 2) and statistical (Table 3) comparisons indicated that predicted annual drainage rates were in very good agreement with field measurements in most years. The means and standard deviations of predicted and measured annual subsurface drainage matched very well for both D2 site (predicted 431 ± 217 mm *vs.* measured 436 ± 231 mm) and D3 site (predicted 384 ± 152 *vs.* measured 386 ± 160 mm). Excellent model performance was demonstrated by the high values of NSE (>0.9) and d (>0.92) and by the relatively lower mean values of MAE (<45 mm yr^{-1}), compared to standard deviation of measured annual drainage. Normalized percent errors (NPE) indicated that annual drainage was over-predicted in 10 out of the 20 years for both D2 and D3 sites, suggesting no bias (systematic errors) in model predictions of annual drainage. Absolute NPEs were less than 10% for 15 years over the whole study period for both sites. The largest NPEs in predicting annual drainage (89.3% for D2 and 58.6% for D3) occurred in the extremely dry year of 2001. The second largest NPEs (about 35% for both sites) and the largest absolute error (152 mm yr^{-1} for D2 and 160 mm yr^{-1} for D3) occurred in 2002. Following the method given by Amatya et al. (1996), we conducted a simple water balance to estimate annual ET from measured rainfall, drainage, and difference in soil water storage. Results showed that annual ET in 2002 was about 1250 mm yr^{-1} for both sites, which was higher than the long-term mean annual ET (1074 mm yr^{-1}) of the study site. However, it is comparable to measurement of Sun et al. (2010) who reported annual ET of 1226 mm for a mid-rotation loblolly pine plantation (13–15 years old) located in the lower coastal plain of North Carolina. In contrast, annual ET predictions in 2002 were 1015 mm and 1078 mm for D2 and D3 sites, respectively. The under-prediction of ET was mainly caused by possible over-predicted reduction in LAI after extended drought in 2001 (Tian et al.2010). The good agreement between the predicted and measured annual flows for 1995–1999 on the D3 watershed (Fig. 2) with the orifice-weir type of controlled drainage treatment clearly demonstrates the model's ability to simulate drainage for that type of water management treatment.

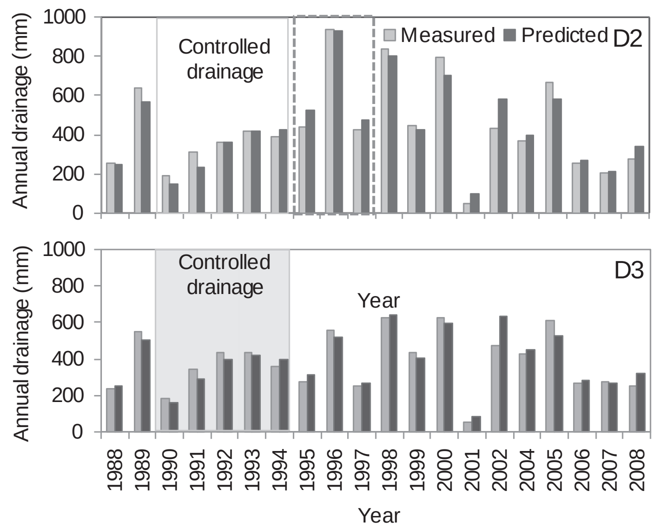

Fig. 2. Comparisons between predicted and measured annual drainage for D2 and D3 sites over the study period. The box with dashed line covers harvesting period for D2 site.

Table 3
Summary of statistical measures of model performance for predicting annual and monthly drainage over the study period.

	Annual predictions				Monthly predictions		
	NSE	*D*	MAE	NPE (%)	NSE	*d*	MAE
D2	0.93	0.96	44.5	4.2	0.9	0.94	11.1
D3	0.90	0.95	36.9	4.5	0.9	0.93	9.5

A scatter diagram comparing predicted and measured monthly drainage for D2 and D3 sites is given in Fig. 3. Statistical measures of model performance are summarized in Table 3. For both sites, predictions of monthly drainage were in very good agreement with field measurements as indicated by the high values of NSE (=0.9) and d (>0.93) and the significantly smaller values of the MAE (<11 mm mo^{-1}) compared to the standard deviations of measured monthly drainage (44 mm mo^{-1} for D2 and 38 mm mo^{-1} for D3).

Model predictions of daily water table depth were in very good agreement with measured data (Fig. 4). Goodness-of-fit indices were NSE = 0.90, d = 0.92, MAE = 0.12 m for D2 site, and NSE = 0.86, d = 0.91, MAE = 0.15 m for D3 site. These statistical indices indicated that the model accurately predicted daily water table fluctuation for both D2 and D3 sites. The discrepancy between model predictions and field measurements of water table depths were attributed to errors in representation of drainable porosity and simulation of ET (Amatya and Skaggs, 2001; Tian et al., 2010).

According to model predictions, mean annual ET is the largest water balance component, accounting for 65% of mean annual precipitation. This is consistent with experimental results obtained from other coastal forests located in Eastern United States (e.g., Sun et al., 2010). Thus, accurate prediction of ET is critical for accurately simulating the other hydrological processes in the forest ecosystem. DRAINMOD-FOREST simulates daily PET using Penman–Monteith method (Monteith, 1965) with canopy conductance estimated as the product of predicted stomatal conductance and LAI (Tian et al., 2009). According to model predictions, maximum stomatal conductance of each year for D2 and D3 sites ranged from 73.5 to 113.8 mmol m^{-2} s^{-1} with a mean of 85 mmol m^{-2} $s^{-1}$and a standard deviation of 13.6 mmol m^{-2} s^{-1}. Magnitudes and seasonal variations of predicted stomatal conductance are comparable to field measurements (e.g., Amatya and Skaggs, 2001; Domec et al., 2009). Predicted annual mean LAI ranged from 0 m^2 m^{-2} after forest harvesting to 6.6 m^2 m^{-2} for D2, and 3.4 m^2 m^{-2} to 6.4 m^2 m^{-2} for D3. The capability of DRAINMOD-FOREST to simulate long-term LAI dynamics has been demonstrated by Tian et al. (2010). However, model predictions of the temporal changes in stomatal conductance need to be verified by comparing model predictions with field measurements.

3.2. Carbon pool dynamics

According to model predictions, OC pool on forest floor of both sites had a similar magnitude and temporal variations from 1988 to 1995 (Fig. 5). The model predicted that OC pool on forest floor of D2 site was somewhat greater than that on D3 site from late 1988 to 1995 (Fig. 5), although OC pools on forest floors of D2 and D3 sites were assumed to be the same at the beginning of model simulation. The difference was probably caused by higher microbial decomposition rates under more favorable soil moisture condition caused by consistently higher water table in D3 site than D2 site during this period (Amatya et al., 1996; 2006). After harvesting of D2 site, its OC pool on forest floor abruptly increased due to large amounts of fresh logging slash and debris. Thereafter, predicted forest floor OC pool size in D2 site have steadily decreased from the forest harvesting in 1995 until the site preparation in October 1996. Then the forest floor OC storage suddenly dropped to a very low value because the model assumed that site preparation incorporated 65% of the total surface OC storage into

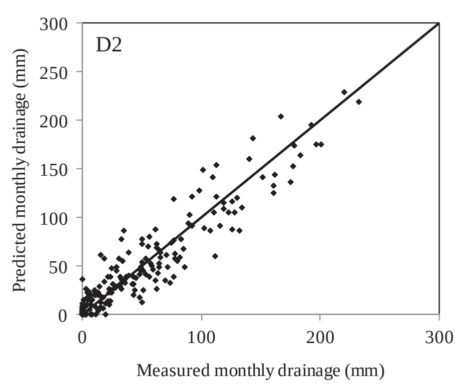

Fig. 3. Comparison between predicted and measured monthly subsurface drainage for D2 and D3 sites over the study period.

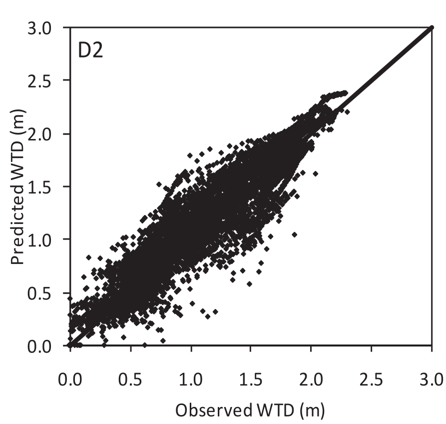

Fig. 4. Comparisons between predicted and measured daily water table depth in D2 and D3 site over the study period.

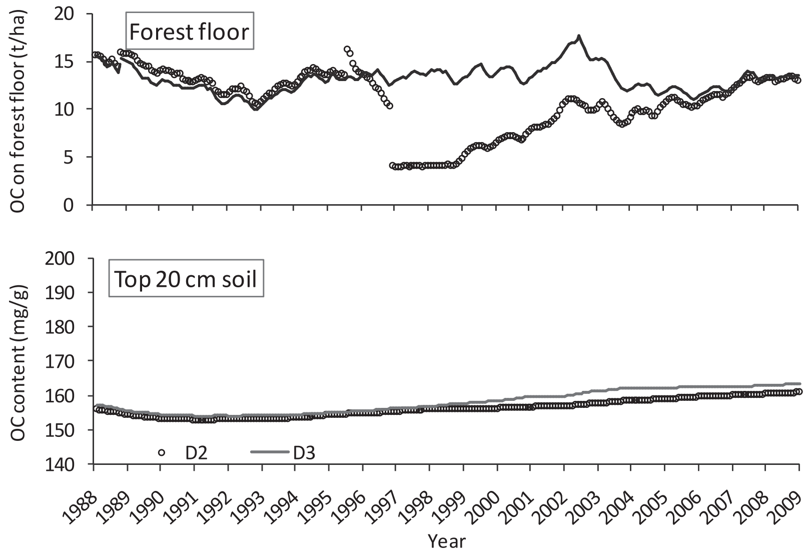

Fig. 5. Comparison between predicted organic carbon dynamics on forest floor and in top 20 cm soil for D2 and D3 sites.

forest soil. After the forest regeneration in February 1997, the OC pool on the forest floor of D2 site was predicted to remain constant for two years and to steadily accumulate thereafter. Model predictions indicated that it took about 10 years for surface OC pool on D2 site to reach its pre-harvesting level. Predicted fluctuation of OC content on forest floor is comparable to field measurements conducted for loblolly pine stands located in the Upper Coastal Plain of Alabama (Zerpa, 2005).

Predicted OC pool on forest floor of the two forests shows distinct intra- and inter-annual fluctuations. Seasonal dynamics of OC pool on forest floor were caused not only by seasonal litterfall dynamics that is primarily regulated by plant physiology (Gresham, 1982), but also by variations in microbial decomposition rates, which are influenced by changes in weather conditions (Vanhala, 2002; Lal, 2005). In contrast, inter-annual fluctuations of OC on forest floor were mainly affected by extreme climate conditions and silvicultural practices. For instance, extended drought during the growing season of 1993 and the whole year of 2001 significantly constrained microbial decomposition, leading to large accumulation of OC on forest floor (Fig. 5). Forest management practices including thinning and harvesting increased OC pool by 1.5 t ha^{-1} and 3.2 t ha^{-1} on forest floor, respectively, due to sudden input of logging slashes. On the other hand, the predicted surface OC pool on D2 site in 1997 decreased by approximately 6.8 t ha^{-1} after site preparation that mixed stored OC on forest floor into topsoil.

In contrast to obvious temporal variations of OC pool on forest floor, predicted soil OC content was relatively stable (Fig. 5). Predicted soil OC content increased over the 21 years from 156 mg g^{-1} to 161 mg g^{-1} and 163 mg g^{-1} in D2 and D3 sites, respectively. This increase in soil OC is only 3.2% for D2 and 4.5% for D3. The standard deviation of soil OC content at the end of each month in D2 and D3 sites are 2.4 mg $g^{-1}$and 3.3 mg g^{-1}, respectively, which are less than 2% of their long-term mean values (156.4 mg $g^{-1}$and 158.2 mg g^{-1}, respectively). Soil OC in both sites was predicted to decrease slightly during the first 5 years followed by a small but gradual increase. The initial decreasing trend of soil OC content was mainly caused by the reduced litterfall caused by the forest thinning in 1988 (Tian et al., 2010). Moreover, it is clear that soil OC in D2 had continuously lower OC content (by 3 mg g^{-1}) compared to D3 site after 1998. This was probably caused by reduced root turnover in the harvested site.

3.3. Nitrogen export predictions

3.3.1. Annual nitrate export

Predictions of ammonium export were excluded from the following analysis because both predicted and measured annual average ammonium losses were smaller than 0.2 kg ha^{-1}. Model predictions of annual nitrate loading are in good agreement with measured values (Fig. 6 and Table 4). The means and standard deviations of predicted and measured annual nitrate export closely matched for D2 (predicted: 1.6 ± 1.3 *vs.* measured 1.5 ± 1.5 kg ha^{-1}) and D3 (predicted: 1.4 ± 1.3 *vs.* 1.3 ± 1.1 kg ha^{-1}). The high NSE and d values (>0.9) indicated DRAINMOD-FOREST accurately predicted long-term nitrate losses from the two loblolly pine plantations. The small MAE values (<0.22 kg ha^{-1}), compared with the standard deviation

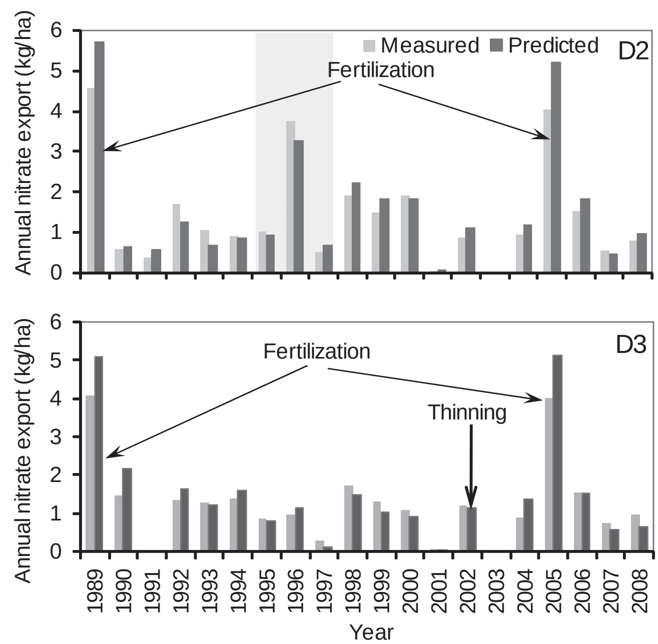

Fig. 6. Predicted and measured annual nitrate losses via subsurface drainage; gray area indicates forest harvest and site preparation period.

Table 4
Summary of statistical measures of model performance in predicting annual and monthly nitrate losses over the study period.

	Annual predictions				Monthly predictions		
	NSE	*d*	MAE	NPE (%)	NSE	*d*	MAE
D2	0.92	0.95	0.22	15	0.86	0.96	0.08
D3	0.90	0.93	0.21	7	0.84	0.96	0.06

of measured annual nitrate losses (>1.1 kg ha^{-1}), also suggests good model performance. The largest MAEs in predicting nitrate losses from both sites consistently occurred in years with N fertilizer application. Mean annual NPEs were 15% and 7%, for D2 and D3 sites, respectively. The NPE in predicting annual nitrate export ranged from −35.2% to 100% for D2 site and −31.6% to 70% for D3 site. The largest NPEs in predicting nitrate loadings from both D2 and D3 sites occurred in 2001, consistent with the largest percent errors of predicted annual drainage (Fig. 2). However, the relatively large error in predicting nitrate losses of 2001 is of no practical significance because of the measured load during this year was practically negligible (<0.2 kg ha^{-1}). Of the 20-year simulation period, the model under predicted annual nitrate export from D2 and D3 sites in 8 and 10 years, respectively, indicating almost no bias in annual nitrate export predictions.

DRAINMOD-FOREST reasonably captured the spike in nitrate leaching losses following N fertilizer application. Although the model over predicted nitrate export from D2 and D3 sites in 1989 and 2005 by approximately 15%, predictions of nitrate export during years with fertilizer application closely matched measured nitrate losses from both sites. In 1989, predicted and measured annual nitrate losses were 5.6 and 4.5 kg ha^{-1} for D2 site, and 5.2 and 4.1 kg ha^{-1} for D3 site, respectively. In 2005 predicted and measured annual nitrate exports were 5.2 and 4.0 kg ha^{-1} for D2 site, and 5.1 and 4.0 kg ha^{-1} for D3 site, respectively. Predicted and measured annual nitrate loads in 1989 and 2005 were significantly higher ($p < 0.001$) than the long-term mean annual nitrate export (for D2 and D3 sites).

The model also captured the increase in annual nitrate export from D2 site during forest harvesting and site preparation period. With the exception of 1995, 1996, and 1997, both field measurements and model predictions showed that there were no significant differences between annual nitrate losses from D2 and D3 sites ($p > 0.4$, $df = 36$) according to a two-tailed student *t*-test. In 1996, the first year after forest harvesting, predicted and measured nitrate losses from D2 site spiked to 3.8 and 3.5 kg ha^{-1}, respectively, compared with the low nitrate losses from D3 site during this year (predicted loss = 1.0 kg ha^{-1}, measured loss = 1.1 kg ha^{-1}). The difference between nitrate export from D2 and D3 site during 1995, 1996, and 1997 was possibly caused by forest harvesting and site preparation in D2 site, and orifice weir treatment in D3 site (Amatya et al., 2003). Good agreement between predicted and measured annual nitrate export in those three years showed that the model was able to accurately predict the effects of harvesting and site preparation on N export from the drained pine plantation.

3.3.2. Monthly nitrate export

Results indicated that model predictions of monthly nitrate loads were also in very good agreement with field measurements (Fig. 7 and Table 4). NSEs for predicted monthly nitrate export from D2 and D3 sites were 0.86 and 0.84, respectively. Degrees of agreement values for monthly predictions of nitrate export from both sites were above 0.95. MAEs were 0.06 and 0.05 kg ha^{-1} mo^{-1} for D2 and D3, respectively, which are relatively low compared to standard deviations of 0.32 and 0.36 kg ha^{-1} mo^{-1} for measured monthly nitrate losses from D2 and D3 sites, respectively.

Urea fertilizer application significantly increased nitrate loss during wet months right after fertilization. According to both model predictions and field measurements, monthly nitrate losses via subsurface drainage in most months were lower than 1.0 kg ha^{-1} (Fig. 7). The number of months with measured nitrate export higher than 1.0 kg ha^{-1} were 7 and 4 for D2 and D3 site, respectively, most of which occurred in months with large storm events after nitrogen fertilizer application or forest harvesting (Amatya et al., 1998; 2006; Amatya and Skaggs, 2008) as highlighted in Fig. 7. Predicted monthly nitrate export after N fertilizer application was in good agreement with measured values in both sites (Fig. 7). Nitrate export from both D2 and D3 sites during wet months after Urea-N fertilizer application was noticeably over predicted (Fig. 7). In September of 1989, predicted and measured nitrate losses were 1.66 and 1.12 kg ha^{-1} for D2 site, and 1.39 and 0.83 kg ha^{-1} for D3 site, respectively. In December of 1989, predicted and measured nitrate losses were 1.46 and 1.02 kg ha^{-1} for D2 site, and 2.14 and 1.61 kg ha^{-1} for D3 site, respectively. Predicted and measured monthly nitrate losses in October of 2005 were 3.9 and 3.3 kg ha^{-1} for D2 site, and 4.1 and 3.5 kg ha^{-1} for D3 site, respectively.

The relative similarity between statistical measures of drainage (Table 3) and nitrate losses predictions (Table 4) indicated that hydrological predictions are critical in controlling model performance of nitrate export predictions. However, smaller magnitudes of NSE and *d*, as well as higher NPE, suggest that factors other than hydrological predictions also contributed to the discrepancies between model predictions and field measurements of nitrate losses. The other factors were most likely associated with simulations of N transformations including mineralization, nitrification, denitrification and plant uptake (Youssef et al., 2006).

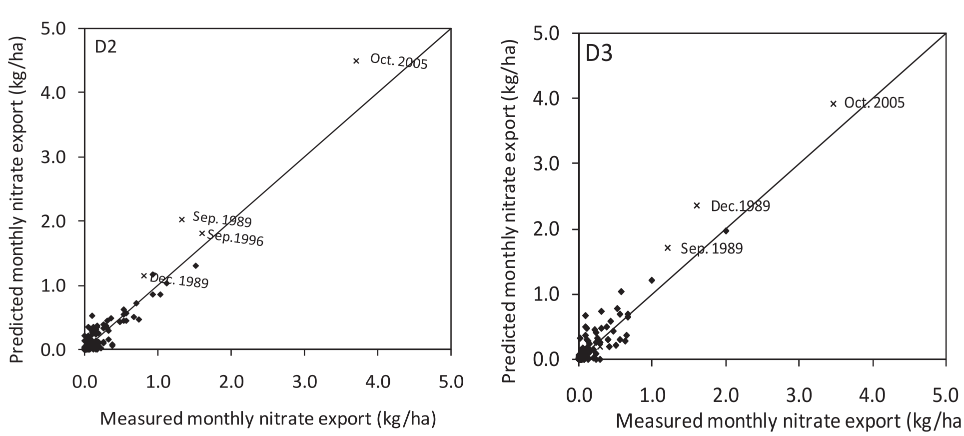

Fig. 7. Comparisons between predicted and measured monthly nitrate export. Crosses represent months with heavy storm events after fertilization.

4. Discussion

This study demonstrated the applicability of DRAINMOD-FOREST for predicting long-term water, soil organic matter and nitrate export dynamics for two artificially drained forests under intensive management. The model accurately predicted annual and monthly drainage (Figs. 2 and 3), daily water table fluctuations (Fig. 4), as well as annual and monthly nitrate export dynamics from both sites (Figs. 6 and 7). Compared with published studies, model predictions of the temporal changes in OC pools on forest floor and in the topsoil were also reasonable (Fig. 5). In this section, model predictions are thoroughly analyzed to evaluate the validity of model simulations of hydrological and biogeochemical processes as affected by water and silvicultural management practices.

4.1. Measured and simulated effects of controlled drainage

Controlled drainage reduces nitrate export from drained forest through reducing drainage volume (Amatya et al., 1998). DRAINMOD-FOREST accurately predicted drainage reductions caused by controlled drainage. Predicted mean annual drainage during period under controlled drainage in D2 and D3 sites were 362.6 and 367.5 mm, respectively, both of which closely matched field measurements (Fig. 2) and were approximately 30% lower than predicted mean annual drainage of 540.2 mm for D1 site, which was under conventional drainage (Tian et al., 2010). Although controlled drainage reduced drainage outflow and slightly elevated water table (Amatya et al., 1996), model predictions show no significant impact of the practice on N transformations. On average over the controlled drainage period (1991–1994), predicted annual net mineralization rates for D2 and D3 sites were 60.7 and 63.5 kg ha^{-1}, respectively. These values were comparable to the mean of predicted annual net mineralization during the corresponding period for D1 site (61.2 kg ha^{-1}), which was under conventional drainage (Tian et al., 2010). The average of predicted annual denitrification rates was about 2.3 kg ha^{-1} for both D2 and D3 sites, which is substantially higher than predicted denitrification (0.97 kg ha^{-1}) for the reference site during the same period.

Since it only slightly influenced N budget, controlled drainage reduced nitrate export by primarily reducing drainage rates. The averages of predicted annual nitrate export from D2 and D3 during period of controlled drainage were 0.8 and 1.5 kg ha^{-1}, respectively, both of which were comparable to field measurements (1.0 kg ha^{-1} for D2 and 1.3 kg ha^{-1} for D3). The reduction in nitrate export due to controlled drainage was less than 16% compared to nitrate export from the reference site (Amatya et al., 1998), which was lower than the percent reduction in annual drainage. This was mainly because a substantial amount of the nitrate losses occurred during winter when controlled drainage was not applied (Amatya et al., 1998).

4.2. Measured and simulated effects of nitrogen fertilizer application

4.2.1. Effects of urea-N fertilization on nitrate losses

The application of fertilizers has been widely used to enhance forest productivity (Fox et al., 2007). Numerous studies have shown that environmental effects of fertilization are short lived under sound management practices (Binkley et al., 1999; Fox et al., 2007). However, the potential adverse environmental consequences of fertilizer application have been a public concern (Amatya et al., 1998; Stednick, 2008; Beltran et al., 2010). Results presented in Figs. 6 and 7 show that the model performed well in predicting nitrate losses over the whole study period. The model also nearly successfully captured the measured temporary spike in nitrate losses following the urea-N fertilizer application in years 1989 and 2005.

The model, however, overestimated effects of fertilization on annual and monthly nitrate losses (Figs. 6 and 7). One possible cause of this over-prediction is the fertilization application practice of the landowner. To minimize the risk of the fertilizer drift into the drainage ditches, fertilizer was not applied to a 30-m wide buffer zone adjacent to the ditches (Beltran et al., 2010). This practice prevents the dissolved nitrate originating from the applied fertilizer from quickly reaching the drainage ditch through the short groundwater flow paths occurring close to the ditches. Thus, any nitrate that moves with infiltrated precipitation down through the vadose zone and reaches the groundwater will follow long flow paths to the drainage ditch. This increases the nitrate residence time in the shallow groundwater, enhancing plant uptake and reducing overall nitrate loss to the receiving surface water. This phenomenon cannot be simulated by a one-dimensional model such as DRAINMOD-FOREST; a two-dimensional model would be required but such models are not practically applicable because of the excessive number of input parameters. Moreover, discrepancies between predicted and measured daily water table depths, especially at shallow depths (Fig. 4) may lead to inaccurate predictions of nitrate losses since approximately 50% of total N fertilizer is stored on the forest floor and in the top soil layer (Will et al., 2006). Further, the model may have overestimated nitrification rates following the urea fertilizer application.

4.2.2. Effects of fertilizer application on nitrogen transformations

Fertilizer application can alter N biogeochemical processes including mineralization, immobilization, nitrification, and denitrification. Predicted N transformations in years with N fertilization are summarized in Table 5. Compared to long-term mean annual N transformation rates, predicted annual net mineralization of soil organic matter was substantially lower in years (1989, 2005) with Urea-N fertilizer application, because of higher immobilization rates (Table 5). The predicted reduction in annual net mineralization rates appears to contradict findings from lab and field experiments that reported increased net N mineralization after applying Urea-N fertilizer to forest soil (Gurlevik et al., 2002; Lee and Jose, 2006). These studies, however, assumed that ammonium originated from urea hydrolysis is a product from net mineralization. This assumption was based on the fact that urea is an organic form of N. In contrast, DRAINDMOD-N II model regards ammonium produced from urea hydrolysis as an independent N source. Predicted annual production rate of ammonium were computed for D2 and D3 by adding the mass rate of ammonium produced from urea hydrolysis to the mass rate of ammonium produced from the mineralization of organic matter. For D2 and D3, these rates were found to be 117 and 121 kg ha^{-1} in 1989, and 87 and 105 kg ha^{-1} in 2005, respectively. These values were comparable to values reported by Gurlevik et al. (2002) who studied effects of urea fertilizer application on N mineralization in a loblolly pine plantation located in the piedmont region of North Carolina. However, effects of N fertilization on net mineralization are subjected to many factors including fertilizer form, application amount and frequency, site-specific climatic conditions, soil type and vegetation (Lee and Jose, 2006). Lee and Jose (2006) found that the application of mixed N fertilizer (47% as urea, 29% as ammonium and 24% as nitrate) had no impact on annual net mineralization rates in a loblolly pine plantation located in Florida, USA.

Predicted annual nitrification rates were higher due to the high concentrations of ammonium following Urea-N fertilizer application (Table 5). As expected, annual denitrification rates also increased due to increased nitrate in the soil profile (Table 5). However, the denitrification rates were still substantially lower compared to agricultural fields (Barton et al., 1999). Model

Table 5
Comparison between predicted N transformations during the year with N fertilizer application and the long-term mean annual N transformation rates.

	D2				D3			
	Net-Min ($kg\ ha^{-1}\ yr^{-1}$)	Immob ($kg\ ha^{-1}\ yr^{-1}$)	Nit ($kg\ ha^{-1}\ yr^{-1}$)	Denit ($kg\ ha^{-1}\ yr^{-1}$)	Net-Min ($kg\ ha^{-1}\ yr^{-1}$)	Immob ($kg\ ha^{-1}\ yr^{-1}$)	Nit ($kg\ ha^{-1}\ yr^{-1}$)	Denit ($kg\ ha^{-1}\ yr^{-1}$)
1989	30.4	138.4	87.0	3.4	28.2	132.1	95.8	4.6
2005	45.4	76.2	61.3	2.6	34.8	105.1	55.3	1.9
Mean[a]	67.2	9.8	30.6	1.3	61.9	7.9	27.7	1.3

[a] Mean values represent the long-term mean of transformation processes over the study period. Note that all rates were cumulative values on predicted daily rates on annual basis; Net-Min: net mineralization, Immob: immobilization, Nit: nitrification, Denit: denitrification.

predictions of increased annual denitrification following fertilizer application were consistent with findings by Mohn et al. (2000) who reported increase in annual denitrification in fertilized spruce forests ($2.9\ kg\ ha^{-1}\ yr^{-1}$) compared to unfertilized plots ($1.7\ kg\ ha^{-1}\ yr^{-1}$).

4.2.3. Other possible consequences of fertilization

Repeated N fertilization can increase LAI significantly (Vose and Allen, 1988; Albaugh et al., 1998), which will potentially modify hydrological processes through increasing rainfall interception and plant transpiration. However, there were no noticeable differences in the results of hydrological simulations after applying N fertilization in 1989 and 2005. This is probably due to the limited effects of one time fertilizer application. Furthermore, N fertilizer application did not noticeably change predicted soil OM pool as indicated in Fig. 5. This was consistent with findings of Sartori et al. (2007) who found no significant changes of soil C pool over 4–16 years with annual fertilizer application to loblolly pine plantations located in the Piedmont region of Georgia, USA. Lee and Jose (2003) also concluded that fertilizer application had little effect on soil OC pool in a loblolly pine plantation located in northwest Florida, USA. Jandl et al. (2007) concluded that effects of fertilization on soil C pool are variable and depend upon many site-specific factors regulating subsequent soil biogeochemical processes.

4.3. Measured and simulated effects of thinning treatments

Forest thinning can alter hydrological processes (Grace et al., 2006a,b, 2007). D2 site was subjected to one thinning treatment in 1989. D3 was thinned twice in 1989 and 2002, respectively. Field measurements indicated that the 1989 thinning operation reduced LAI from 4.2 to 2.1, and canopy coverage by 50% (Amatya et al., 1996). The reduced LAI and canopy fraction correspondingly modify hydrological processes like ET through reducing rainfall interception and canopy transpiration. Consider the hydrological effects of thinning in site D3 in 2002 as an example. Measured annual drainage in 2002 from D3 site was 470 mm, compared to 426 mm measured from D1 (reference site). Predicted values were 580 and 631 mm for D1 and D3, respectively. Both field measurements and model predictions suggested that thinning in D3 site led to an increase in annual drainage of approximately 50 mm. Model predictions were comparable to analysis by Amatya and Skaggs (2008) for the same thinned site (D3). In contrast, Grace et al. (2006a,b) found that forest thinning increased mean daily flow by 100% in a loblolly pine plantation located in Washington County, NC, USA. The thinning treatment reduced the stocking number in their study site from 1060 to 320 trees ha^{-1} (Grace et al., 2006a,b). It seems logical that hydrological impacts of thinning treatment is subject to the thinning intensity, pre-treatment site conditions including vegetation, soil and hydrology, as well as post-treatment recovery of canopy closure. Differences in these factors would explain the contrast in hydrological impacts of thinning between observations of Grace et al., 2006a,b and those observed and predicted in this study.

Although thinning in 2002 altered forest hydrology in D3 to some extent, it did not noticeably change nitrate export according to either field measurements or model predictions (Fig. 6). Amatya and Skaggs (2008) reached similar conclusions with respect to effects of thinning on water quality in drainage flow from D3. This was probably due to the fact that forest ecosystems are N limited and thinning might not significantly alleviate the N deficit that already exists prior to the operation. Also, understory species usually responds quickly to canopy removal in a managed forest (Thomas et al., 1999), which could largely compensate for decreased nitrogen uptake by removed loblolly pine. This was consistent with the speculation of Sampson et al. (in press) who reported the potential influence of emerging understory vegetation on site nutrient balance. In addition, alterations in soil moisture and temperature caused by the thinning treatment might be insufficient to stimulate microbial activities to noticeably change N transformations.

Grace et al. (2006a,b, 2007) found that mechanical operations during thinning caused changes in soil properties such as hydraulic conductivity and porosity that may affect hydrological processes. Such soil physical changes were not considered in this application of DRAINMOD-FOREST, and may constitute a source of model error. Further verification of such observations by field experiments is thus needed. Nevertheless, DRAINMOD-FOREST model reasonably simulated effects of thinning treatments on hydrological processes, and soil N and C dynamics in drained forest ecosystems, indicating that the model can be used to predict effect of thinning loblolly pine with reasonable reliability.

4.4. Measured and simulated effects of harvesting, site preparation and regeneration

Compared to other management practices simulated in this study, forest harvesting, which is usually followed by site preparation and regeneration, has long been recognized as the major anthropogenic disturbance in managed forest ecosystems (e.g., Roberts, 2007). This treatment suddenly removes large amount of live forest biomass and disturbs forest floor and topsoil, which substantially influence both hydrological (Bosch and Hewlett, 1982; Sun et al., 2001; Brown et al., 2005; Moore and Wondzell, 2005; Eisenbies et al., 2007; NRC, 2008) and biogeochemical processes (Johnson, 1992; Johnson et al., 2002; Grenon et al., 2004; Lapointe et al., 2005; Gundersen et al., 2006). Using paired watershed approach, Amatya et al. (2006) concluded that forest harvesting and regeneration in D2 site substantially increased drainage volume, raised water table depth, and increased nitrate export. These observed changes were successfully captured by DRAINMOD-FOREST, as indicated by the good agreement between model predictions and field measurements (Figs. 2, 4 and 6). Further discussions were necessary to investigate underlying mechanisms that are responsible for hydrological and biogeochemical changes caused by forest harvesting and regenerations. Taking D2 watershed as an example, following sections compare model predictions of detailed hydrological and biogeochemical processes under two scenarios: one represents the real condition in which forest harvesting, site preparation and regeneration had been carried out, and the other one is a hypothetical control scenario without these practices. Simulations for both scenarios were carried out using calibrated model parameters. The simulated effects of these management operations were determined as the difference between model predictions for hydrological and biogeochemical processes for the two scenarios.

4.4.1. Effects of forest harvesting on hydrology

Fig. 8 illustrates model predictions of daily drainage from 1995 to 1999 for the harvesting and no harvesting scenarios. Field measurements of daily drainage are also included in Fig. 8 to demonstrate the level of agreement between measured and predicted daily drainage for the forest harvesting scenario. Model predictions of daily drainage closely matched field measurements (Fig. 8) for each of the years since harvesting in July 1995. There were apparent differences between predicted daily drainage under the two scenarios, especially during growing seasons. According to model predictions, storm events during the fall of 1995, and summer and fall seasons of 1996, and summer of 1997 generated considerably larger drainage volumes for the forest harvesting scenario, compared to the scenario without harvesting. For example, forest harvesting increased drainage rate by as much as 20 mm day^{-1} during the period from June to September of 1996. In contrast, there was no noticeable difference between magnitudes of peak discharge of storm events during non-growing seasons. These results support conclusions by Sun et al. (2001) and Brown et al. (2005) that forest harvesting usually poses significant influence on drainage processes during growing seasons. Jones and Post (2004) also found that changes in daily stream flow after forest

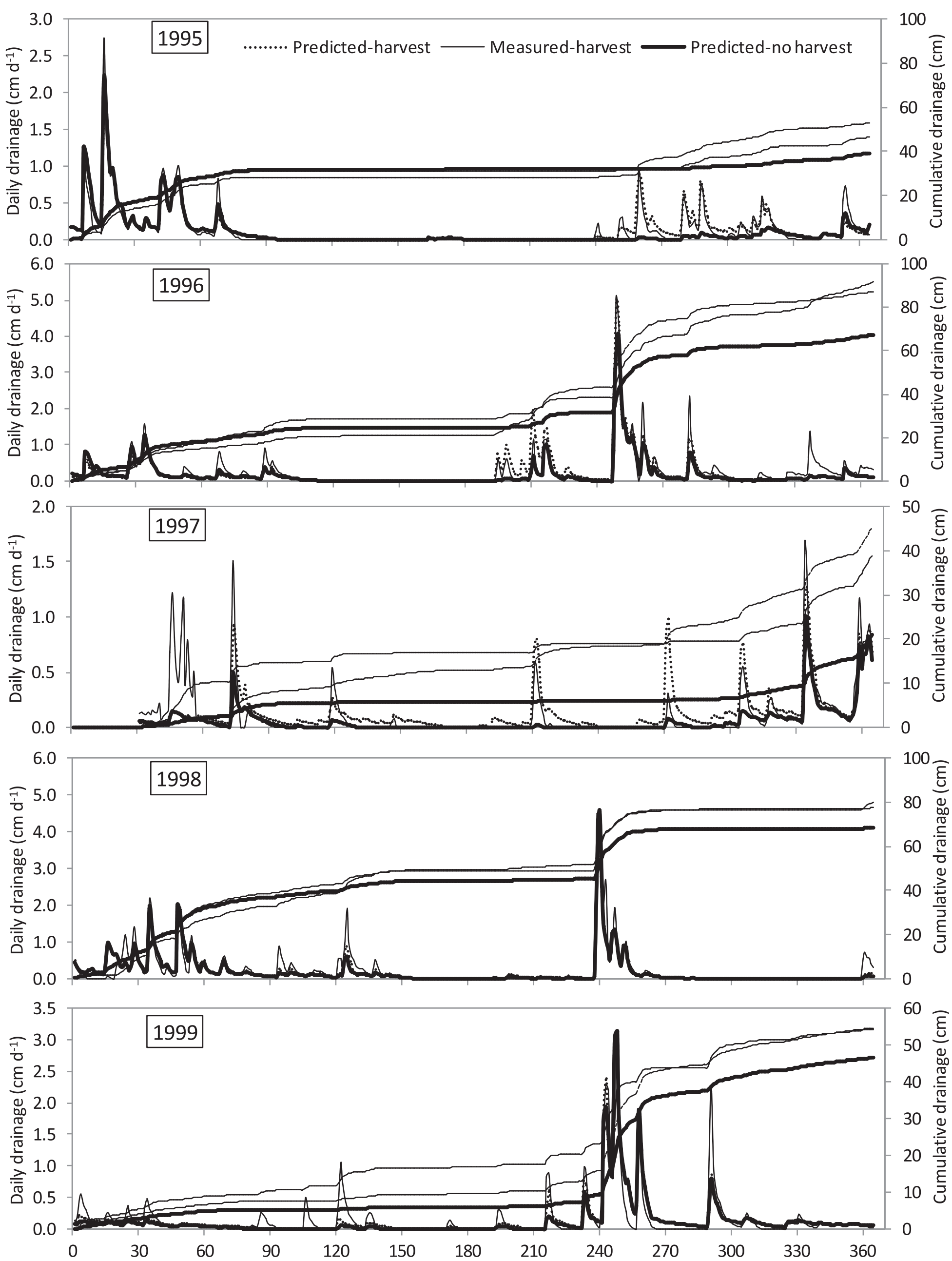

Fig. 8. Daily drainage for D2 site during 1995–1999 (measured and predicted for the harvesting scenario and predicted for the no-harvesting scenario).

cutting and regrowth in eastern United States were mainly concentrated during warm and wet seasons when soil moisture and air temperature are conducive to evapotranspiration.

Fig. 9 shows measured and predicted daily water table depths for the harvesting scenario and predicted water table depth for the no-harvesting scenario during 1995–1999. Predicted daily water table depth matched field measurements for most of the time since forest harvesting in July 1995. However, there was a relatively large discrepancy between model predictions and field measurements during spring of 1999 (Fig. 10). According to field measurements, 1.7 cm of rainfall was recorded from day 80 to day 86 in 1999 resulting in a measured rise in water table of about 20 cm. The model did not predict a corresponding rise in water table, which increased the difference between predicted and measured water table depths from 24.0 cm just before these small rainfall events to 51.0 cm. This difference was mostly carried over during the summer of 1999. This discrepancy between predicted and measured water table depths were most likely caused by either errors in rainfall measurement or overestimation of ET during the period. Nevertheless, the impacts of forest harvesting on water table fluctuations are clearly illustrated by the comparison (Fig. 9). Predicted water table during summer months was substantially shallower under forest harvesting compared to the scenarios without harvesting. For instance, from June to October of 1997, predicted mean daily water table depth for the harvest scenario was approximately 27 cm shallower, compared to the no-harvesting scenario. The effects of forest harvesting on water

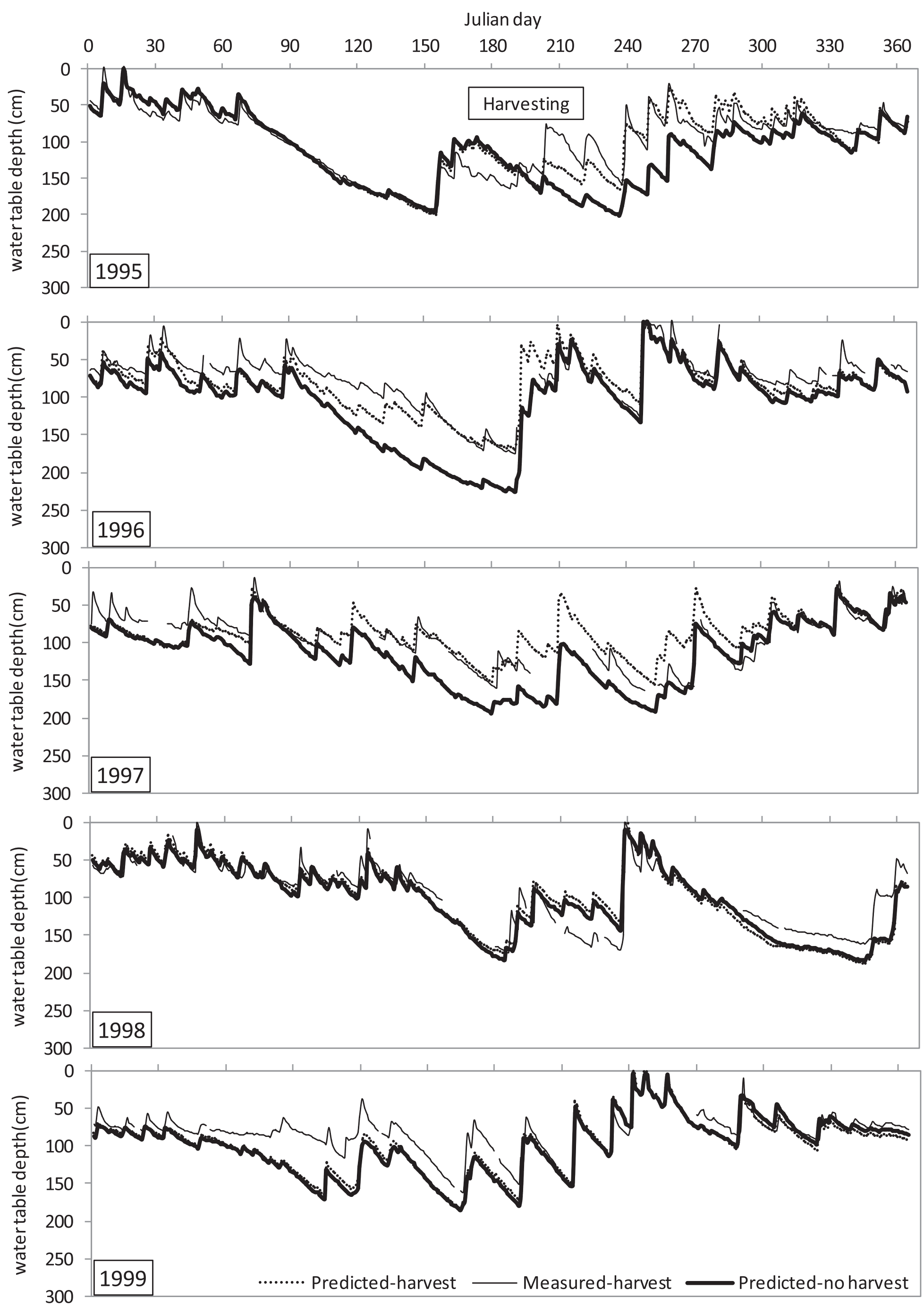

Fig. 9. Daily water table depths for the D2 site during 1995–1999 (measured and predicted for the harvesting scenario and predicted for the no-harvesting scenario).

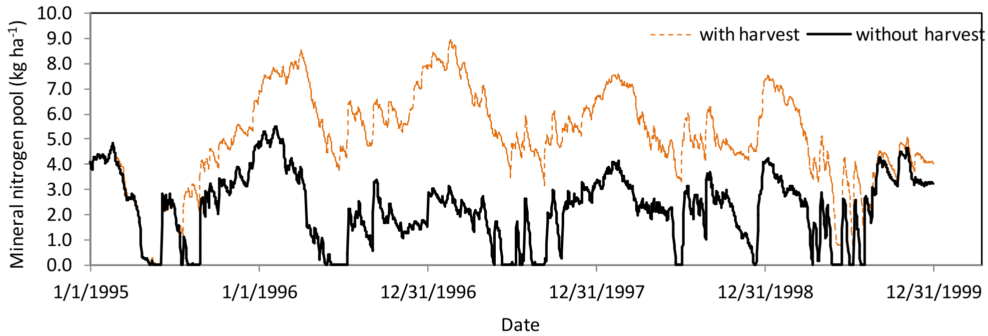

Fig. 10. Predicted soil mineral nitrogen pool dynamics under scenarios with and without forest harvesting.

table depth were greatly attenuated during dormant seasons with small and indistinguishable ET demands for both harvesting and no-harvesting scenarios. The model predictions were consistent with experimental studies investigating the effects of forest harvesting on groundwater levels (Sun et al., 2000 and Xu et al., 2002). These studies attributed the substantial rise in groundwater level to the reduced canopy transpiration and interception following harvesting.

Amatya et al. (2006) investigated effects of harvesting on hydrological processes in D2 site; They pointed out that water table elevation increased by about 20 cm while drainage increased by 9.1 cm in the first 6 months following harvest. The hydrological impacts of harvesting were mainly attributed to decreased ET due to canopy removal (Amatya et al., 2006). Predicted main hydrological processes such as rainfall interception, ET, and subsurface drainage under both scenarios are summarized in Table 6. According to model predictions, forest harvesting significantly increased subsurface drainage by 11.7 cm in the second half of year 1995, which was slightly higher than field measurements (Amatya et al., 2006). Predicted annual drainage was increased by 39% in 1996, 136% in 1997, 31% in 1998 and 22% in 1999. Hydrology approximately returned to baseline level in 2000 because no significant differences were detected in monthly subsurface drainage in D2 site ($p > 0.5$). Rainfall interception was substantially reduced after forest harvesting. During the period after harvesting and before regeneration (1995 and 1996), model predictions indicated no rainfall interception under the forest harvesting scenario, comparing to 9.5 and 19.8 cm rainfall interception under the control scenario. From 1997 to 1999, the first three years after regeneration, predicted rainfall interception values were still lower under the harvesting scenario than those under the control scenario. According to model predictions, ET was reduced by 17% in 1995, 15% in 1996, 27% in 1997, 12% in 1998 and 10% cm in 1999. Based on hydrologic predictions summarized in Table 6, we conclude that both reduced rainfall interception and canopy transpiration made substantial contributions to raised water table and increased water yield after forest harvesting. Predicted annual changes of simulated ET and rainfall interception are mainly attributed to LAI dynamics. According to model predictions, annual maximum LAI of replanted loblolly pine steadily increased from less than 0.1 $m^2\,m^{-2}$ in 1997 to 4.7 $m^2\,m^{-2}$ in 2001. This increase of LAI gradually offsets the hydrological alterations resulted from canopy removal.

Swank et al. (2001) pointed out that vegetation species in early succession after canopy removal could affect ET and reduce magnitude of hydrological responses during summer seasons after forest harvesting. Similar observations at this site were reported by Sampson et al. (in press) who reported that LAI of understory vegetation from 1995 to 1999 varied from 0.5 to 6 with an average of about 2.5. The DRAINMOD-FOREST model simulated potential evapotranspiration (PET) during the period between forest harvesting and regrowth (June 1995 to February 1997) using the Food and Agriculture Organization (FAO) Peman-Monteith method for a hypothetical reference plant with a height of 0.12 m and LAI of about 3 $m^2\,m^{-2}$ (Allen et al., 1998). This approximation reflected to some extent the hydrological influences of early successional plants, consistent with recent data (average around 2.5 $m^2\,m^{-2}$) published by Sampson et al. (in press) for the emerging vegetation. Nevertheless, ignoring rainfall interception of successional plants may introduce some errors in model predictions as the one possibly occurred in 1997.

Forest harvesting and site preparation can change soil physical properties (Blanton et al., 1998; Xu et al., 2002; Grace et al., 2006a,b, 2007; Skaggs et al., 2006a,b). Blanton et al. (1998) studied changes in soil hydraulic properties during forest harvesting and regenerating period (1995–1998) on the D2 site. The authors reported that these management practices led to approximately 50% decrease of soil drainable porosity in top 60 cm and significantly altered storm drainage hydrographs (Blanton et al., 1998). Disturbances of soil physical properties and corresponding effects on hydrological behaviors are beyond the scope of the DRAINMOD-FOREST model. Thus the model deals with inputs such as soil bulk density, porosity, and hydraulic conductivity as constants rather than functions of forest management practices. Considering the complexity of mechanistic model development and parameterization, the necessity of using detailed mechanistic methods to

Table 6
Comparison between predicted main hydrological processes for the harvesting and no-harvesting scenarios.

	Rainfall (cm yr^{-1})	Rainfall interception (cm yr^{-1})		Evapotranspiration (cm yr^{-1})		Drainage (cm yr^{-1})	
		Harvest	No-harvest	Harvest	No-harvest	Harvest	No-harvest
1995	85.8	0.0	9.2	35.9	43.1	51.0	39.3
1996	164.6	0.0	19.8	70.8	82.9	93.2	67.1
1997	138.2	12.1	26.1	55.1	75.8	49.0	20.7
1998	160.8	10.2	18.3	67.1	76.8	79.4	60.5
1999	144.3	15.3	18.5	74.4	83.0	49.4	40.6

Note: (a) Only the period of 1995 after forest harvesting was included in this analysis; (b) the sum of separate contribution of rainfall interception and ET may not equal to 100% because of annual difference in soil water storage; (c) the model does not consider rainfall interception of understory species.

quantify hydrological impacts of early successional plants growth after harvesting and changes of soil physical properties needs further investigation.

4.4.2. Effects of forest harvesting on nitrogen dynamics

Forest harvesting unavoidably disturbs soil profile near the surface and increases soil exposure to solar radiation. These changes could alter soil moisture, temperature and aeration conditions and significantly affect soil organic matter decomposition and soil nutrients status (Peng et al., 2002; Laiho et al., 2003). Numerous studies have shown that forest canopy removal increases the soil mineral N pool and prompts N leaching to receiving surface waters (Likens et al., 1970; Knoepp and Swank, 1997; Prescott, 1997; Grenon et al., 2004; Lapointe et al., 2005; Hazlett et al., 2007; Gundersen et al., 2006). The DRAINMOD-FOREST model reasonably predicted the response of nitrate export after harvesting the D2 site (Figs. 6 and 7). Aside from the reason of increased drainage fluxes due to forest removal (Fig. 2), the increased size of the soil mineral N pool was at least partially responsible for increased nitrate export.

Fig. 10 displays predicted mineral N pool dynamics from 1995 to 1999 under scenarios with and without forest harvesting. As expected, there was no difference between predicted mineral N dynamics prior to the harvesting. The predicted mineral N pool for the two scenarios continuously followed similar temporal dynamics, both before and after the forest harvesting. However, the predicted amount of mineral N pool in the soil profile after the forest harvesting was much larger than that without forest harvesting. The predicted mineral N pool size after harvesting ranged from 1.3 to 9.1 kg $ha^{-1}$for years between 1996 and 1998, compared to 0 to 4.2 kg ha^{-1} without forest harvesting. After the site was bedded and replanted in February 1997, the difference between predicted mineral N pools under the two scenarios was slowly reduced, especially during growing seasons. In the growing season of 1999, predicted mineral N pools under the two scenarios were very close. The model predictions are supported by numerous studies that have been conducted to evaluate effects of clear cutting on soil mineral N dynamics (Likens et al., 1970; Knoepp and Swank, 1997; Prescott, 1997; Grenon et al., 2004; Lapointe et al., 2005; Hazlett et al., 2007) and comparable to other modeling studies (e.g., Laurén et al., 2005). Variations between predicted mineral N pools under the two scenarios clearly suggested changes in nitrogen transformation processes after forest harvesting.

The most straightforward explanation of mineral N flush after forest removal may be that harvesting significantly reduced N uptake during growing seasons (Likens et al., 1970; Hazlett et al., 2007). N mineralization and nitrification rates may have also been affected by changes in soil moisture content (Grenon et al., 2004; Hazlett et al., 2007) and/or increased soil temperature caused by forest harvesting (Vitousek and Matson, 1985; Grenon et al., 2004), as well as reduced inputs of fresh carbon (Prescott, 1997; Grenon et al., 2004; Lapointe et al., 2005; Hazlett et al., 2007). Additionally, the site preparation and bedding processes, which involved mixing organic material on forest floor into forest soil, is expected to increase the rates of organic matter decomposition (Vitousek and Matson, 1985; Youssef et al., 2005).

Further comparison of predicted N transformations under the two scenarios provides meaningful insights about the underlying mechanisms controlling changes in mineral N pool due to forest harvesting. Table 7 summarizes predicted N transformation processes under scenarios with and without forest harvesting. As demonstrated by many field experimental studies, predicted annual net N mineralization under forest harvesting can either be lower or higher than that under scenarios without forest harvesting. During the first two years after forest harvesting, predicted net N mineralization rates were about 15% lower than those under the control scenario. After site preparation and planting in 1997, however, predicted net N mineralization rates were about 8% higher than the rates under the no-harvest scenario. This is in contrast to field findings of Carter et al. (2002) who reported immediate increase in net N mineralization after forest harvesting and site preparation in two loblolly pine forests located in US Gulf region. According to model predictions, canopy removal increased soil temperature by approximately 1.2 °C and substantially raised soil water table level (Fig. 9), both of which potentially increased gross N mineralization. However, an increased gross N mineralization rate does not necessarily increase net N mineralization (Grenon et al., 2004). Labile carbon availability is another critical factor regulating net mineralization in disturbed forest ecosystems (Gurlevik et al., 2002; Grenon et al., 2004). The decreased net N mineralization predicted for 1995 and 1996 was mainly due to increased N immobilization caused by sudden intensive inputs of fresh labile carbon in the form of logging slash and underground roots after forest harvesting. Idol et al. (2001) reported that N immobilization can be significantly increased through mixing logging residuals into forest soil. Carmosini et al. (2002) found that gross N mineralization balanced ammonium immobilization even 1.5 years after forest cut in a boreal forest located in western Canada. Model predictions in this study indicated that forest harvesting-induced enhancements in N mineralization surpassed that in N immobilization after two years, which led to increased N net mineralization in years of 1997, 1998, and 1999.

Forest harvesting in this study also altered nitrification, denitrification, and N uptake processes (Table 7). Elevated nitrification rates after forest harvesting are responsible for increased mineral N export (Laurén et al., 2005). According to model predictions (Table 7), annual nitrification rates were substantially increased because of enhanced substrate level, increased soil temperature and moisture. Annual nitrification was increased by forest harvesting, irrespective of changes in net mineralization. This conclusion is consistent with finding of field experiments by Carter et al. (2002). In addition, predicted mean annual denitrification rate was approximately 2.7 times higher after forest harvesting. This can be attributed to two possible causes: the increased nitrate pool size and the elevated water table (Fig. 9), which resulted in favored anoxic conditions. As expected, nitrogen uptake was significantly decreased under forest harvesting compared to no harvest. For

Table 7
Comparison between predicted N transformation processes under scenarios with and without forest harvesting.

	Net mineralization (kg ha^{-1} yr^{-1})		Nitrification (kg ha^{-1} yr^{-1})		Denitrification (kg ha^{-1} yr^{-1})		Plant uptake (kg ha^{-1} yr^{-1})		Nitrate export (kg ha^{-1} yr^{-1})	
	With	Without	With	Without	With	Without	With	Without	With	Without
1995[a]	26.3	33.7	15.9	10.1	1.4	0.4	15.6	28.1	0.7	0.3
1996	52.7	61.4	28.3	17.1	4.2	1.0	37.1	68.5	3.8	1.3
1997	65.8	60.9	31.9	12.8	3.0	0.5	62.0	66.5	0.5	0.1
1998	73.2	68.1	31.3	17.2	2.1	0.7	62.3	73.7	2.3	1.3
1999	61.5	56.4	24.7	15.9	2.1	0.9	65.9	62.6	0.7	0.8

[a] Only the second half year of 1995 after forest harvesting was considered in this analysis.

instance, predicted nitrogen uptake was about 45% lower under harvesting scenario in 1995 and 1996, than the non-harvest scenario. Nitrogen uptake was restored to levels similar to that under the control scenario after bedding and regeneration in February 1997, which is largely attributed to the temporary regrowth of understory species (Sampson et al., in press). Although we do not have field measurements of all N compartments to verify model prediction, predicted large contribution of understory vegetation to N uptake after forest harvesting is comparable to Carneiro et al. (2009) and other studies reviewed therein. We found minor differences between predicted nitrogen transformation processes under the two scenarios in and after 1999. This result is consistent with findings of several other comparative studies. Shepard (1994) concluded that forest harvesting would raise nutrient concentrations and this effect can last 1–4 years consistent with Amatya et al. (2006). Aust and Blinn (2004) pointed out that water quality would be restored within 2–5 years after forest operation disturbances, which was also consistent with Amatya et al. (2006).

The DRAINMOD-FOREST model reasonably predicted the mechanisms governing biogeochemical responses to forest harvesting and associated operations. However, other factors such as successional growth of plants and prescribed burning after forest harvesting should not be overlooked (Grenon et al., 2004; Carter et al., 2002; Johnson et al., 2010; Sampson et al., in press). For instance, Prescott et al. (1989) found that N uptake rates by successional ground vegetation and shrubs in a clear-cut site were comparable to that of mature forests. The DRAINMOD-FOREST model simulated N uptake by successional growth of plants using an empirical method (Tian et al., 2009). It is easy to use, but is an approximate method and may not adequately represent the role of successional growth of understory species in the study sites. Additionally, prescribed burning can increase N mineralization rate significantly (e.g., Schoch and Binkley, 1989; Deluca and Zouhar, 2000). Nevertheless, the feasibility of incorporating a detailed understory growth module and simulating prescribed burning has to be evaluated with caution since it could substantially increase the complexity of the model and the uncertainty in its prediction.

5. Conclusion

The DRAINMOD-FOREST model was evaluated using a 20-year data set collected from two intensively managed coastal loblolly pine (*P. taeda* L.) plantations located in Carteret county of North Carolina, USA. Management practices on the site during the 20-year period included controlled drainage, the application of N fertilizer, thinning, forest harvesting, site preparation, and regeneration. The hydrologic and water quality effects of all these practices were simulated and evaluated using the model. Predicted annual and monthly drainage as well as daily water table depth were in very good agreement with measured values. Predicted C pool dynamics on forest floor and in topsoil reasonably responded to forest managements and climatic conditions. The model accurately predicted nitrate losses through subsurface drainage on both annual and monthly bases. The validity of predicted hydrological and biogeochemical responses to controlled drainage and silvicultural practices was also verified. Special attention was given to model predictions of hydrological and biogeochemical processes after forest harvesting. The model reasonably predicted reduced rainfall interception and ET after clear cutting, both of which led to increased water yield and elevated water table. The model also reasonably captured alterations of nitrogen transformations caused by forest harvesting, such as increased mineralization, nitrification, and denitrification, and decreased plant uptake. In summary, this study demonstrated that the DRAINMOD-FOREST model can be utilized to predict water, C and N dynamics in drained forest ecosystems under intensive management practices.

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