Simulated Effects of Nitrogen Management and Soil Microbes on Soil Nitrogen Balance and Crop Production

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Searching for environmentally friendly N management practices in the midwestern United States is an ongoing task in the agricultural community. Many practices have shown promise in reducing N in tile drainage that may contribute to hypoxia in the Gulf of Mexico. In this study, an agricultural system in central Iowa, managed using a corn (*Zea mays* L.)–soybean [*Glycine max* (L.) Merr.] rotation from 1996 to 2005 was evaluated with the Root Zone Water Quality Model (RZWQM) for crop yield, grain N, annual N loss in tile drainage flow, and residual soil NO_3–N under high (H, 199 kg N ha^{-1}), medium (M, 138 kg N ha^{-1}), and low (L, 69 kg N ha^{-1}) N application rates shortly after planting, and a split (S, 69 kg N ha^{-1} shortly after planting and again at midseason) N treatment. The model adequately simulated the responses of yield and N loss to N application rates. Simulated N losses to drainage flow from 1996 to 2005 were 348, 277, and 228 kg N ha^{-1} for the H, M, and L treatments, respectively, compared with corresponding measured values of 369, 265, and 201 kg N ha^{-1}. The S treatment had simulated and measured total N losses in drainage flow of 194 and 172 kg N ha^{-1}, respectively, from 1999 to 2005. The study also demonstrated that RZWQM without soil microbial growth produced very similar simulation results for crop production and soil N and water balances as RZWQM with dynamic soil microbial growth.

Abbreviations: FWANC, flow-weighted annual nitrogen concentrations; ME, Nash–Sutcliffe model efficiency; RMSE, root mean square error, RZWQM, Root Zone Water Quality Model.

Nitrogen loss to the environment has been a major concern for many decades. Fertilizer use has increased rapidly in the 1990s worldwide, which has led to increased N losses to the environment (Dinnes et al., 2002). The contribution of N from agricultural lands has been a significant part of N in surface and groundwater bodies and the hypoxia in the Gulf of Mexico (Rabalais et al., 1996). To address this problem, comprehensive water quality studies have been conducted in several management systems evaluation areas (MSEAs) as a part of the Midwest Water Quality Initiative in the last decade (Watts et al., 1999), focused on tile-drained agricultural lands. Management practices that have been or are currently being evaluated in the Midwest are winter cover crops (Kaspar et al., 2007), tillage (Karlen et al., 1998), N application rate and timing (Jaynes et al., 2001, 2004; Jaynes and Colvin, 2006; Thorp et al., 2007), manure application (Bakhsh et al., 1999, 2000), and drainage water management (Ma et al., 2007c).

Modeling of the agricultural systems and management practices for N loss in tile flow and crop production has been studied under several weather conditions and soils (Ma et al., 2007b, 2007c, 2007d; Malone et al., 2007a, 2007b; Saseendran et al., 2007; Skaggs et al., 1995). The Root Zone Water Quality Model (RZWQM) is one model widely used to simulate management effects on water quality (Ma et al., 2000, 2007a), especially in Iowa. One of the MSEA sites was the Nashua site in northeast Iowa, where the RZWQM was used to study the effects of crop rotation, tillage, drainage water management, and manure management on crop yield and N and pesticide losses in tile drainage (Ma et al., 2007b, 2007c, 2007d; Malone et al., 2007a, 2007b; Saseendran et al., 2007). Other MSEA sites studied with the RZWQM were the Walnut Creek watershed south of Ames, IA (Bakhsh et al., 2004a, 2004b), and Story City in central Iowa (Bakhsh et al., 2001; Thorp et al., 2007). In these studies, the RZWQM was capable of simulating the effects of N rate on crop yield and N loss in tile drainage; however, the responses of the RZWQM to N application timing were not evaluated. In the Story City study, Bakhsh et al. (2001) only analyzed data from 1996 to 1999 with a generic crop growth model initially released with the RZWQM, whereas Thorp et al. (2007) used the RZWQM-DSSAT3.5 to simulate crop yield and N loss in tile flow at three different N application rates shortly after planting from 1996 to 2005, but they did not simulate the midseason N application that was also part of the study (Jaynes and Colvin, 2006).

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For each of these applications, the RZWQM has to be initialized for soil organic matter and soil microbial pools so that management effects can be correctly simulated (Ma et al., 1998), since these pools cannot be easily measured experimentally. Ma et al. (1998) suggested 10 to 12 yr of simulation with previous management practices and weather data for model initialization. Such an initialization has two purposes. First, it can stabilize (to steady state) the soil C and N pools based on management and weather history of the field. Second, it makes sure that the microbial pools and the soil C and N pools are synchronized so that the equations to describe soil C and N dynamics are valid. The initialization can be arbitrary, however, due to unknown management practices and weather data before the experiment.

Although the addition of dynamic soil microbial population growth made the RZWQM more responsive to the soil environment, the model is difficult to implement without proper initialization of the organic and microbial pools (two crop residue pools, three soil humus pools, and three microbial pools). Therefore, it would be advantageous to have a soil C and N module with fixed soil microbial pools. Thus, the objectives of this study were (i) to further evaluate N application rate and timing on crop production and soil N balance in Story City, IA, using the newly developed RZWQM-DSSAT4.0, and (ii) to test the performance of the RZWQM-DSSAT4.0 with dynamic and constant soil microbial populations for soil C and N balances.

MATERIALS AND METHODS

Field Experimentation

The study area was a 22-ha privately owned field in central Iowa (93°35′50″ N, 42°11′48″ W). The soil is mainly in the Kossuth (fine-loamy, mixed, superactive, mesic Typic Endoaquolls)–Ottosen (fine-loamy, mixed, superactive, mesic Aquic Hapludolls) association. These clay loam soils were formed on nearly level, alluvial or lacustrine sediments, range from very poorly to somewhat poorly drained, and have surface soil organic C contents of 29 g kg^{-1}. In 1992, new subsurface drainage lines were installed in the field at a depth of 1.45 m, with drainage spacing of 27.4 m. Twelve lengths of 10.2-cm-diameter plastic corrugated drainpipe (tile) were installed along an east–west direction across the field (Jaynes et al., 2001; Jaynes and Colvin, 2006). Composite water samples from each tile were collected automatically and returned to the laboratory on a weekly or shorter basis, depending on tile flow rate, and chilled to 4°C until analysis. Water samples were analyzed for NO_3 using a Lachat 8000 (Zellweger Analytics, Lachat Instrument Div., Milwaukee, WI). Rainfall was measured, starting in 1996, with a tipping bucket rain gauge and recorded every hour at a location <0.5 km from the field. Missing temperature and precipitation data were obtained from the National Climatic Data Center (NCDC) for a weighing rain gauge located 2 km away. For 1985 to 1990, maximum and minimum daily temperature, solar radiation, and daily precipitation were obtained from a weather data set for the Iowa State University Agronomy and Agricultural Engineering Research Center (AAERC) 25 km southwest of our study site. Wind run and relative humidity for 1961 through 1990 were obtained from a NCDC weather station at Des Moines International Airport, 75 km south of the study site. For 1991 though 1995, maximum and minimum daily temperatures from AAERC were used, but information for solar radiation, daily precipitation, wind run, and relative humidity were obtained from a data set for the Walnut Creek watershed located 30 km south of the study site.

The field was planted to corn in even years and soybean in odd years from 1996 to 2005 and was in a 2-yr corn–soybean rotation before this time. Primary tillage consisted of fall chisel plowing after soybean, except for 1997 when moldboard plowing was used. After corn harvest, moldboard plowing was implemented in 1996 and chisel plowing in 1998. A field cultivator was used to prepare the soil for planting corn and to incorporate herbicide in the spring; a row crop cultivator was used several times during the early growing season for weed control in corn. Corn was planted on a 76-cm row spacing at a rate of 75,000 seeds ha^{-1}. Soybean was planted at an approximate rate of 370,000 seeds ha^{-1}.

Fertilizer rates were 199, 138, and 69 kg N ha^{-1} for the high (H), medium (M), and low (L) N treatments, respectively, except for 1998 when N rates were 172, 114, and 57 kg N ha^{-1}, respectively. The H rate was equivalent to the farmer's normal practice and the M rate was, on average, the economic optimum N rate (Jaynes et al., 2001). A fourth treatment was applied to simulate a split (S) application consisting of 69 kg N ha^{-1} applied at the same time as the other treatments followed by a midseason (at V16 stage) application of another 69 kg N ha^{-1}. The S treatment plots had received the H rate in 1996 and 1998 before the S treatment was initiated (Jaynes and Colvin, 2006). In 1995, anhydrous NH_3 was applied 1 wk before corn planting. After 1995, 28% urea–NH_4NO_3 (UAN) was slot applied to the field between the V1 and V3 growth stages of corn using a coulter applicator. The midseason N application was applied by dribbling liquid UAN (28%) in a narrow band between the rows using a high-clearance sprayer with drop hoses. Liquid fertilizer was used because of its better uniformity of application compared with the more commonly used anhydrous NH_3 (Weber et al., 1995). No N was applied to soybean.

Tile drainage flow collection was started in 1996 for the H, M, and L treatments and in 1999 for the S treatment. In addition, grain yield and N in grain were measured. Six soil cores were taken randomly in November after harvest from each N-treatment plot. The soil cores were taken midway between rows to a depth of 1.2 m. The soil cores were cut into 150-mm-long sections for determination of soil water and NO_3 (Jaynes and Colvin, 2006).

Soil Carbon and Nitrogen Dynamics in RZWQM

The RZWQM is a one-dimensional model emphasizing management effects on soil water quality and crop production. It simulates water infiltration using the Green–Ampt equation and water redistribution by solving Richards' equation. It also simulates plant water uptake using the Nimah–Hanks equation and tile drainage flow using the steady-state Hooghoudt equation (Ahuja et al., 2000). Pseudo groundwater lateral flow below the tiles is simulated based on a user-defined lateral hydraulic gradient (Ma et al., 2007b). Macropore flow is also simulated if rainfall or irrigation exceeds soil hydraulic conductivity. A generic plant growth model was originally developed by Hanson (2000) and later the DSSAT crop growth models (Version 3.5) were linked to RZWQM (Ma et al., 2005, 2006). The RZWQM-DSSAT3.5 was used to study N management by Thorp et al. (2007). Recently, RZWQM was updated to DSSAT4.0 for its plant growth modules (RZWQM-DSSAT4.0).

The RZWQM has five soil organic C pools: two for surface residue (fast and slow) and three for soil humus pools (fast, intermediate, and slow). Decomposition of soil organic C is simulated individually for each pool with first-order kinetics (Ma et al., 2001):

$$r_i = -k_i C_i \quad [1]$$

where r_i is the decay rate of the ith pool (mg C kg^{-1} d^{-1}) (i = 1 for slow surface residue pool, i = 2 for fast surface residue pool, i = 3 for fast humus pool, $i = 4$ for intermediate humus pool, and $i = 5$ for slow humus pool); C_i is the C concentration (mg C kg^{-1} soil), and k_i is a first-order rate coefficient (s d^{-1}) and is calculated from

$$k_i = f_{\mathrm{aer}}\left(\frac{k_{\mathrm{B}}T}{h_{\mathrm{P}}}\right)A_i \exp\left(-\frac{E_{\mathrm{a}}}{R_{\mathrm{g}}T}\right)\frac{[\mathrm{O_2}]}{\left[\mathrm{H}^{kh}\gamma_1{}^{kh}\right]}P_{\mathrm{het}} \qquad [2]$$

where A_i is the rate constant for pool i, [O_2] is the O_2 concentration in the soil water with the assumption that O_2 in the soil air is not limited (mol O_2 L^{-1} pore water), H is the hydrogen ion concentration (mol H L^{-1} pore water), γ_1 is the activity coefficient for monovalent ions ($1/\gamma_1{}^{kh}$ = 3.1573 × 10^3 if pH > 7.0 and $1/\gamma_1{}^{kh}$ = 1.0 if pH ≤ 7.0), kh is the H ion exponent for decay of organic matter (0.167 for pH ≤ 7.0 and −0.333 for pH > 7.0), P_{het} is the population of aerobic heterotrophic microbes (no. of organisms g^{-1} soil, minimum 50,000, default value 100,000), k_{B} is the Boltzman constant (1.383 × 10^{-23} J K^{-1}), T is soil temperature (K), h_{P} is the Planck constant (6.63 × 10^{-34} J s), R_{g} is the universal gas constant (1.99 × 10^{-3} kcal mol^{-1} K^{-1}), E_{a} (= 15.1 + 12.3U, where U is ionic strength [mol]) is the apparent activation energy (kcal mol^{-1}), and f_{aer} is a soil aeration factor estimated from Linn and Doran (1984).

Based on Eq. [1] and [2] and assigned values for each variable in the equations, the model developers derived a set of rate constants (A_i). Constants A_1 and A_2 are derived from measured residue decomposition data and A_3, A_4, and A_5 are derived by assuming a turnover time of 5, 20, and 2000 yr for the fast, intermediate, and slow humus pools. The derived A_i values with a derived unit of seconds per day are A_1 = 1.67 × 10^{-7}, A_2 = 8.14 × 10^{-6}, A_3 = 2.5 × 10^{-7}, A_4 = 5.0 × 10^{-8}, and A_5 = 4.5 × 10^{-10}. Decayed organic C is either transformed into another C pool, released as inorganic CO_2, or assimilated into microbial biomass (immobilization).

In the RZWQM, a zero-order kinetics is used for nitrification (the nitrification rate is constant with respect to soil NH_4 concentration) and a first-order kinetics for denitrification (the denitrification rate is proportional to soil NO_3 concentration). The corresponding zero- and first-order rate coefficients are

$$k_0 = f_{\mathrm{aer}}\left(\frac{k_{\mathrm{B}}T}{h_{\mathrm{P}}}\right)A_{\mathrm{nit}} \exp\left(-\frac{E_{\mathrm{an}}}{R_{\mathrm{g}}T}\right)\frac{[\mathrm{O_2}]^{1/2}}{\left[\mathrm{H}^{kh}\gamma_1{}^{kh}\right]}P_{\mathrm{aut}} \qquad [3]$$

and

$$k_1 = f_{\mathrm{anaer}}\left(\frac{k_{\mathrm{B}}T}{h_{\mathrm{P}}}\right)A_{\mathrm{den}} \exp\left(-\frac{E_{\mathrm{den}}}{R_{\mathrm{g}}T}\right)\frac{\mathrm{C_s}}{\left[\mathrm{H}^{kh}\gamma_1{}^{kh}\right]}P_{\mathrm{ana}} \qquad [4]$$

where P_{aut} is the autotrophic microbial population (nitrifiers) (no. of organisms g^{-1} soil, minimum 500, default value 1000); P_{ana} is the population of anaerobic microbes for denitrification (no. of organisms g^{-1} soil, minimum 5000, default value 10,000); E_{an} and E_{den} are the apparent activation energies for nitrification and denitrification processes, respectively; C_s is the weighted soil organic C in the soil; and f_{anaer} is the anaerobic factor (Ma et al., 2001). The rate constants, A_{nit} and A_{den} for nitrification and denitrification respectively, have calibrated values of 1.0 × 10^{-9} and 1.0 × 10^{-13} s d^{-1}. These A_i values are only approximated within an order of magnitude.

The first-order rate coefficient for urea hydrolysis (hydrolysis rate is proportional to soil urea concentration) in the RZWQM is written as (Ma et al., 2001)

$$k_{\mathrm{urea}} = f_{\mathrm{aer}}\left(\frac{k_{\mathrm{B}}T}{h_{\mathrm{P}}}\right)A_{\mathrm{urea}} \exp\left(-\frac{E_{\mathrm{u}}}{R_{\mathrm{g}}T}\right) \qquad [5]$$

where E_{u} is the activation energy for urea hydrolysis (12.6 kcal mol^{-1}) and A_{urea} is the rate constant for urea hydrolysis (2.5 × 10^{-4} s d^{-1}).

In the RZWQM, NH_3 volatilization (VOL) is simulated based on the partial pressure gradient of NH_3 in the soil (P_{NH3}, atm [1 atm = 0.101 MPa]) and air (P'_{NH3}, 2.45 × 10^{-8} atm):

$$\mathrm{VOL} = -K_{\mathrm{v}}T_{\mathrm{f}}(P_{\mathrm{NH_3}} - P'_{\mathrm{NH_3}})C_{\mathrm{NH_4}} \qquad [6]$$

where C_{NH4} is the concentration of NH_4 in the soil (mol N L^{-1} pore water) and K_{v} is a volatilization rate coefficient affected by wind speed (W, km d^{-1}) and soil depth (Z, cm):

$$K_{\mathrm{v}} = 4.0\times10^3 \ln(W)\exp(-0.25Z) \qquad [7]$$

The temperature factor, T_{f}, is calculated from

$$T_{\mathrm{f}} = 2.9447\times10^4 \exp\left[\frac{-6.0}{1.99\times10^{-3}(T+273)}\right] \qquad [8]$$

where T is soil temperature (°C). The NH_3 partial pressure gradient for the soil, P_{NH3} is calculated by equilibrating soil NH_4 and NH_3.

Microbial populations are controlled by daily growth and death rates. The growth of P_{het} and P_{ana} are determined by assuming a fraction of daily decomposed soil organic C to be converted to microbial biomass, and the growth of P_{aut} is proportional to the daily nitrification rate (Ma et al., 2001). The death rate of each microbial population is assumed to be first order with respect to its population size, and death rate coefficients are functions of soil C, soil temperature, soil pH, and soil O_2 concentration (Ma et al., 2001). A feedback mechanism is thus built into the RZWQM model through the dependency between reaction rates and microbial populations; however, this dependency also makes it is difficult to apply the RZWQM without some sort of initialization (Ma et al., 1998). In this study, we tested the RZWQM with constant soil microbial populations rather than dynamic soil microbial population growth (originally implemented in the RZWQM) in search of a simpler parameterization of the soil C and N module in the RZWQM.

Model Implementation

To evaluate the RZWQM-DSSAT4.0 with and without microbial growth, a previous study of Thorp et al. (2007) was used. Thorp et al. (2007) applied an older version (RZWQM-DSSAT3.5) to the H, M, and L treatments of the same experiment, but not the S treatment. We found that the parameters of Thorp et al. (2007) worked well for the new RZWQM-DSSAT4.0, since there were no major changes in the plant growth models between DSSAT3.5 and DSSAT4.0 (Lopez-Cedron et al., 2005). We did, however, make adjustments of two parameters—increasing the denitrification coefficient from 1 × 10^{-14} to 3 × 10^{-13} s d^{-1} to increase denitrification, and decreasing the decay rate of the slow organic matter pool from 2.4 × 10^{-9} to 4.5 × 10^{-10} s d^{-1} to reduce total yearly mineralization, so that the simulated average yearly denitrification (~10 kg N ha^{-1} yr^{-1}) and mineralization (around 100 kg N ha^{-1} yr^{-1}) were similar to a previous study in Iowa by Ma et al. (2007d). Soil hydraulic properties, initial soil C, microbial populations, and plant parameters for corn and soybean were also from Thorp et al. (2007), with the exception of the lateral hydraulic gradient, which was reduced from 1 × 10^{-5} to 5 × 10^{-6} m m^{-1}. The model was run from 1 Jan. 1985 to 31 Dec. 2005, and simulated results from 1996 to 2005 were compared with experimental results when data were available. Simulation from 1985 to 1995 with the same corn–soybean rotation and 199 kg N ha^{-1} fertilization after corn planting was used to initialize the soil C and microbial pools (Ma et al., 1998). These additional years of initialization were necessary because of the above changes made to the three parameters (i.e., denitrification coefficient, decay rate of the slow C pool, and lateral hydraulic gradient) and the new DSSAT4.0 crop growth modules. It should be noted that these calibrated parameters are only one of many reasonable realizations of the model for the given experimental conditions and may vary from user to user based on criteria of calibration. For the purpose of comparing model behavior with and without microbial growth, these calibrated soil and plant parameters are sufficient and were used for the simulation (1985–2005).

To decouple soil microbial growth from soil C and N dynamics, we assumed constant microbial populations as suggested on the RZWQM user interface (the default populations), i.e., 100,000 microbes g^{-1} soil for aerobic heterotrophs (P_{het}); 1000 microbes g^{-1} soil for nitrifiers (P_{aut}); and 10,000 microbes g^{-1} soil for anaerobic heterotrophs (P_{ana}) in the topsoil horizon. Microbial populations in the soil profile were proportional to its organic C content compared with the topsoil horizon. We also disabled microbial growth and death in the model to maintain constant populations during the simulations. Reaction rates with constant microbial populations are still functions of soil pH, soil moisture, soil temperature, soil O_2, and soil C as shown in Eq. [2–5]. A test of the RZWQM with constant soil microbial populations showed that the denitrification coefficient needed to be adjusted from 3×10^{-13} to 1×10^{-13} s d^{-1}. All the other parameters were kept the same as with dynamic microbial growth to obtain similar soil C and N balances.

The Nash–Sutcliffe model efficiency (Nash and Sutcliffe, 1970) was used to quantify goodness of model prediction:

$$\mathrm{ME}=1-\frac{\sum_{i=1}^{n}\left(O_i-S_i\right)^2}{\sum_{i=1}^{n}\left(O_i-\overline{O}\right)^2} \qquad [9]$$

where ME is the Nash–Sutcliffe model efficiency, n is the number of data points, O and S are observed and simulated values, and $\overline{O}$ is the average observed value; ME = 1 indicates a perfect match between observed and simulated values and ME = 0 suggests that simulation results are as good as using the mean value. Root mean square error (RMSE) was also used to quantify derivation between simulated and measured results.

RESULTS AND DISCUSSION

Simulated Soil Water Balance

Simulated soil water balance did not vary much between the two microbial growth options. Rainfall was the only water input to the systems, with annual rainfall amounts ranging from 58 cm in 2000 to 102 cm in 1996 during the experimental period from 1996 to 2005, with the lowest measured tile flow in 2000 (0.1–2.7 cm). The average highest tile flow (31.8 cm) was measured in 1999 with a rainfall amount of 78 cm. Lower drainage flow in 1996 (27.2 cm) could be due to the relatively dry years in 1994 (63 cm rain) and 1995 (68 cm rain), whereas there was 87 cm of rainfall in 1998. Coefficients of determination (r^2) between measured annual tile flow and annual rainfall ranged from 0.61 to 0.81. Simulated annual tile flow followed a similar pattern to that measured and was correlated to rainfall with r^2 = 0.58 to 0.76 (Fig. 1). Thus, the model correctly simulated the annual tile flow amount with a RMSE of 3.4 cm and a ME of 0.89 (Fig. 1). Simulated yearly (actual) evapotranspiration ranged from 44 to 57 cm, which was in agreement with Ma et al. (2007d) for northeast Iowa and matched measured values in the region as stated in Thorp et al. (2007). Simulated surface runoff ranged from 0.2 cm in 2000 to 9.3 cm in 1996 and was correlated with rainfall, with r^2 = 0.71. Approximately 95% of the runoff water in 1996 was generated on 16 June due to a 16-cm rainfall. Simulated lateral flow below the tiles at the given lateral hydraulic gradient ranged from 5 to 8 cm yr^{-1}.

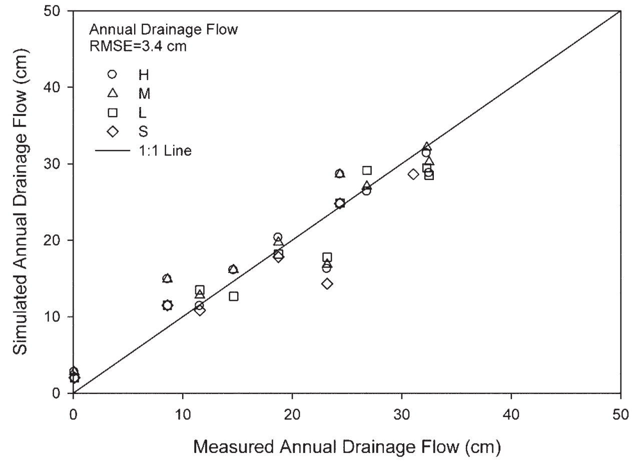

Fig. 1. Measured and simulated annual tile drainage for the high (H), medium (M), low (L), and split (S) N treatments. The results were simulated with dynamic soil microbial growth.

Model Results with Dynamic Soil Microbial Population Growth

Simulated total N losses to drainage flow from 1996 to 2005 were 348, 277, and 228 kg N ha^{-1} for the H, M, and L treatments (Table 1), respectively, which were close to the corresponding measured values of 369, 265, and 201 kg N ha^{-1}. For the S treatment, simulated and measured total N losses were 194 and 172 kg N ha^{-1}, respectively, from 1999 to 2005. During the same period (1999–2005), measured total N losses for the H, M, and L treatments were 214, 148, and 109 kg N ha^{-1}, respectively, with corresponding simulated values of 224, 188, and 149 kg N ha^{-1}. Although the model simulated the relative response of N loss to N rate, the simulated N losses

Table 1. Nitrogen balance from 1996 to 2005 for high (H), medium (M), low (L), and split (S) N treatments with (WMG) and without soil microbial growth (WOMG).

Treatment	N from rainfall	N from fertilizer	Net gain from mineralization	N fixation	Plant N uptake	N loss to denitrification	N loss to tile flow	N loss to lateral flow	N loss to volatilization	N loss to runoff	Δ Soil NO_3–N
						kg N ha^{-1}					
H-WMG	67.8	984	1073	1283	2918	110.4	348.0	55.3	4.4	1.6	− 18.0
M-WMG	67.8	676	993	1328	2662	86.9	277.0	49.8	1.7	1.6	− 3.0
L-WMG	67.8	344	889	1367	2341	66.6	228.4	48.9	0.3	1.6	− 10.9
S-WMG	67.8	801	1050	1303	2797	92.8	317.6	54.5	2.1	1.6	− 32.3
H-WOMG	67.8	984	1042	1285	2845	149.8	362.8	59.9	9.5	1.6	− 47.6
M-WOMG	67.8	676	968	1327	2580	118.9	309.9	56.1	4.1	1.6	− 30.4
L-WOMG	67.8	344	880	1360	2257	93.3	277.8	56.7	1.1	1.6	− 34.0
S-WOMG	67.8	801	1014	1306	2718	122.7	344.3	60.1	5.2	1.6	− 60.6

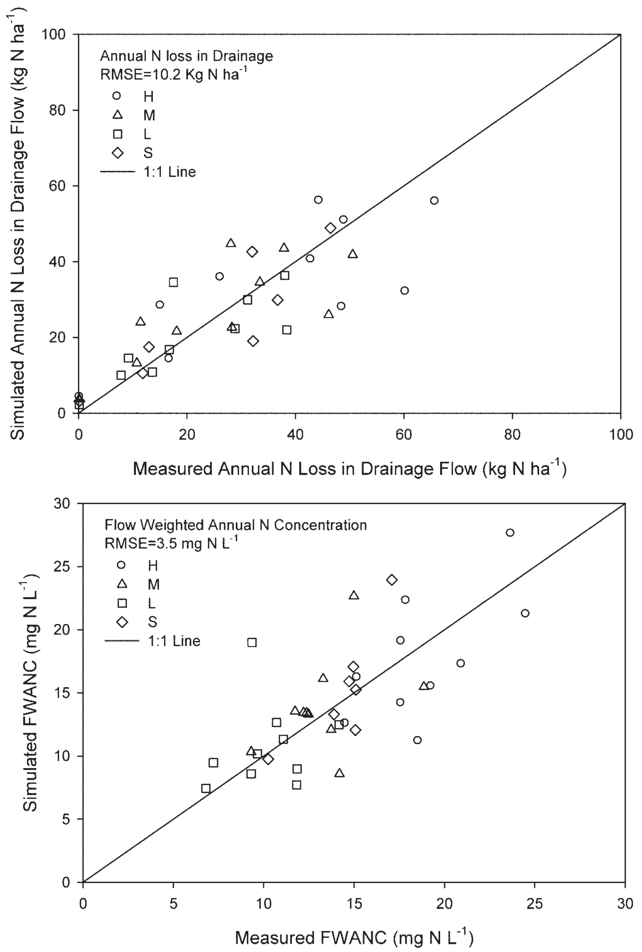

Fig. 2. Measured and simulated annual N loss in tile flow and flow-weighted annual N concentration (FWANC, total annual N load divided by total annual flow volume) for the high (H), medium (M), low (L), and split (S) N treatments. The results were simulated with dynamic soil microbial growth.

were higher than measured values. Both measured and simulated N loss to drainage flow in the S treatment were higher than in the M treatment, which is in agreement with experimental observation (Jaynes and Colvin, 2006).

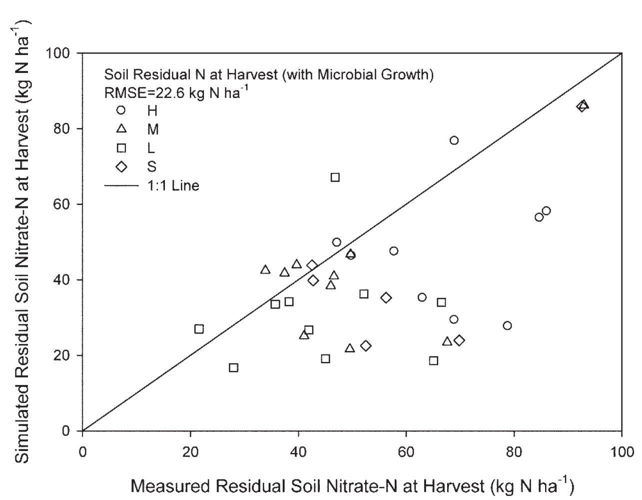

Fig. 3. Measured and simulated residual soil NO_3–N after harvest in the 120-cm soil profile for the high (H), medium (M), low (L), and split (S) N treatments. The results were simulated with dynamic soil microbial growth.

For each N treatment, measured annual N loss in tile flow was highly correlated with annual rainfall amounts, with r^2 ranging from 0.78 to 0.86. Simulated annual N losses in tile flow and flow-weighted annual N concentrations (FWANC, total annual N load divided by total annual flow volume) were also highly correlated with measured values (Fig. 2), with RMSE values of 10.2 kg N ha^{-1} and 3.5 mg N L^{-1}, respectively. Simulated annual N loss in tile flow, however, was less correlated with annual rainfall for each treatment, with r^2 = 0.45 to 0.62, compared with measured annual N losses. The highest FWANC was simulated for the H treatment, followed by M and L treatments. Very comparable values were obtained for both simulated and measured FWANCs between the M and S treatments (Jaynes and Colvin, 2006). Model efficiencies for annual N loss and FWANC were 0.66 and 0.26, respectively; however, the model underestimated residual soil NO_3–N at harvest, with a RMSE of 22.6 kg N ha^{-1} and a ME of −0.32 (Fig. 3). A negative ME suggested that an average residual soil N was better than the simulated residual soil N. Since standard errors among replicates in measured residual soil NO_3–N ranged from 4 to 42 kg N ha^{-1} (with an average value of 14 kg N ha^{-1}), a RMSE of 22.6 kg N ha^{-1} was still acceptable. Simulated residual soil NH_4–N was close to zero, whereas measured soil NH_4 ranged from 9 to 25 kg N ha^{-1}, which indicated that the model simulated more nitrification than occurred in the field by harvest.

Simulated corn and soybean yields and grain N responded well to N rate and timing (Fig. 4 and 5). In general, corn yield was overpredicted for the H and S treatments and underpredicted for the L treatment, indicating that the model responded to N application rate more than experimentally observed. The S-shaped curve for corn yield in Fig. 4 was not observed when the RZWQM-DSSAT3.5 version was used (Thorp et al., 2007), which could be due to differences in soil C and N parameters and additional initialization of the pools discussed above. Thorp et al. (2007) simulated higher soil N mineralization than in this study due to the higher rate coefficient. Such a difference in simulated yields between the previous study of Thorp et al. (2007) and this study demonstrates the importance of model parameterization and initialization on simulation results when a complex model is used. Nonetheless, the RMSEs shown in these figures were well within measurement errors in the field (Ma et al., 2007d) and comparable to simulation errors reported by Thorp et al. (2007). The ME for simulated corn yield was 0.52, suggesting that the model simulated the response of crop yield to N treatments and weather variability among years. The model failed, however, to simulate the observed lower corn yields in the S treatment than in the M treatment in 2002 and 2004 with the same total amount of N application, suggesting a need to improve crop N response in the S treatment. Therefore, the simulated and observed higher corn yields by the RZWQM and APSIM models for a split N treatment in northeast Iowa were probably due to a higher N application rate in the split treatment than in a single N application treatment (Saseendran et al., 2007; Malone et al., 2007a). Simulated corn grain N had a RMSE of 28.7 kg N ha^{-1} and ME of −0.64, suggesting that using an average grain N was better than simulated values (Fig. 5). Simulated corn grain N concentrations were within the range of measured results, with

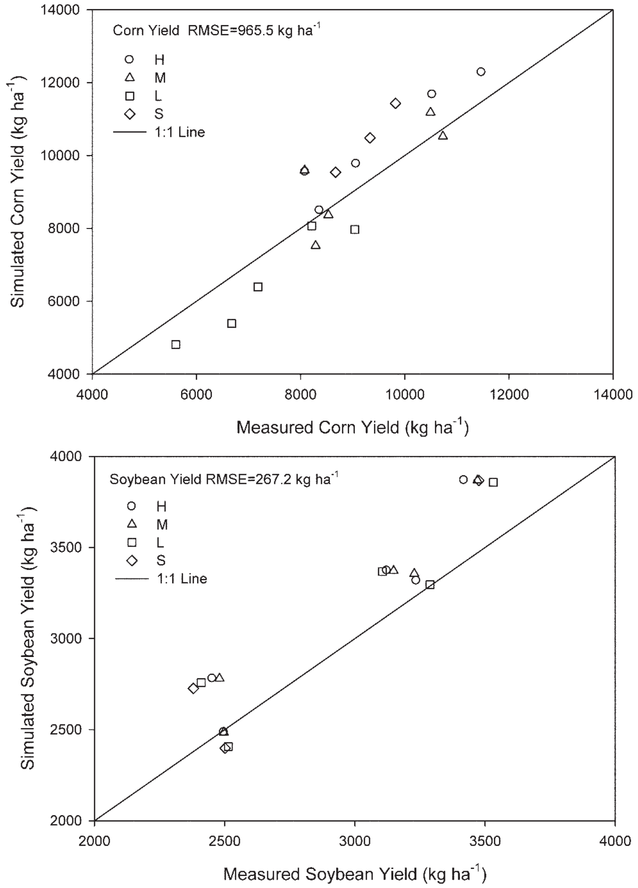

Fig. 4. Measured and simulated corn and soybean yield for the high (H), medium (M), low (L), and split (S) N treatments. The results were simulated with dynamic soil microbial growth.

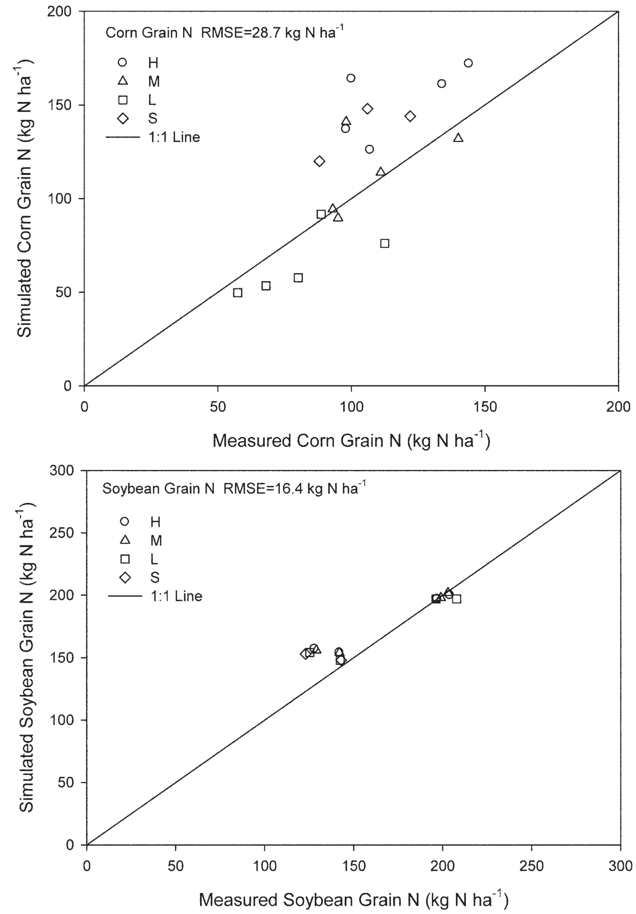

Fig. 5. Measured and simulated grain N of corn and soybean for the high (H), medium (M), low (L), and split (S) N treatments. The results were simulated with dynamic soil microbial growth.

a RMSE of 0.33%, but the ME was –2.98 (Fig. 6). Soybean yield and grain N did not change much from year to year and the model provided adequate prediction for both, with RMSEs of 267.2 kg ha^{-1} and 16.4 kg N ha^{-1}, respectively, and corresponding MEs of 0.62 and 0.76 (Fig. 4 and 5). The ME for soybean grain N concentration was 0.16 (Fig. 6).

Average simulated net N mineralization per year was 109, 99, 99, and 106 kg N ha^{-1} for the H, M, L, and S treatments, respectively, from 1996 to 2005. The model also simulated an immobilization rate of 22 to 25 kg N ha^{-1} yr^{-1}. Corresponding denitrification per year was 11, 9, 6, and 9 kg N ha^{-1} for the four treatments, which are within the range given by Svensson et al. (1991). Average annual N loss to tile flow was 35, 28, 23, and 32 kg N ha^{-1} yr^{-1} for the H, M, L, and S treatments, respectively. Simulated N fixation in the soybean years was about 265 kg N ha^{-1} yr^{-1}. All these results were in close agreement with Ma et al. (2007d) for a corn–soybean rotation in northeast Iowa. The model also simulated an average lateral flow of 6 cm with a corresponding N loss of 8 kg N ha^{-1}, which were about half of that reported by Ma et al. (2007d). Soil organic N (summation of the fast, intermediate, and slow humus pools) increased by 280 kg N ha^{-1} (or a 1.1% increase), and the increase did not vary with N treatment. Simulated total N loss in runoff was 1.6 kg N ha^{-1} from 1996 to 2005. Simulated NH_3 volatilization ranged from 0.3 to 4.4 kg N ha^{-1}, which was in agreement with estimates made by Jaynes and Colvin (2006) (Table 1).

Model Results without Soil Microbial Growth

The above simulations were conducted with the original soil C and N dynamics module, where dynamic soil microbial population growth was an essential part of all the reaction equations (see Eq. [2–5]). After setting the soil denitrification coefficient to 1×10^{-13} s d^{-1}, the RZWQM without microbial growth simulated a similar soil N balance as the model with

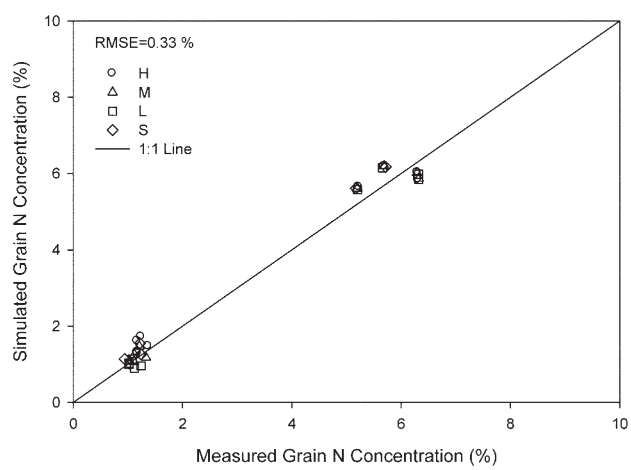

Fig. 6. Measured and simulated grain N concentration of corn and soybean for the high (H), medium (M), low (L), and split (S) N treatments. The results were simulated with dynamic soil microbial growth.

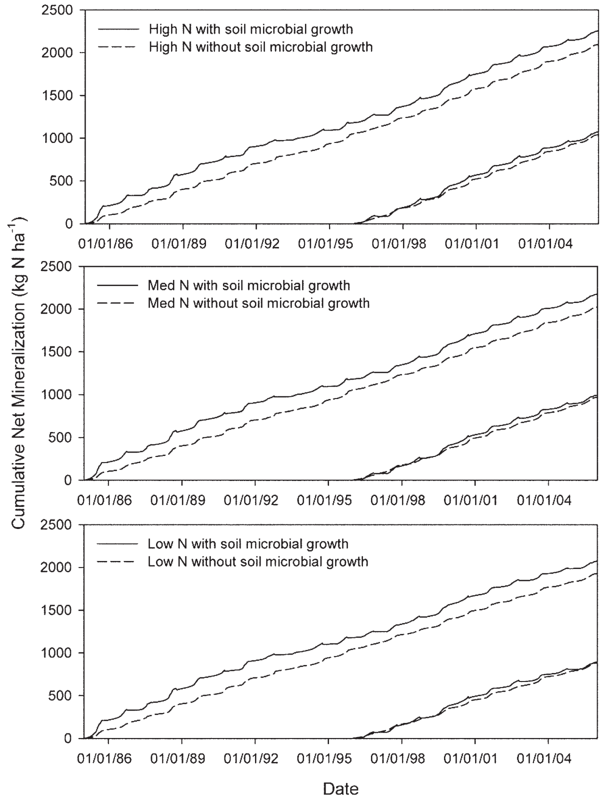

Fig. 7. Simulated cumulative net mineralization with and without soil microbial growth for the high (H), medium (M), and low (L) N treatments from 1996 to 2005 (lower curves) and from 1985 to 2005 (upper curves).

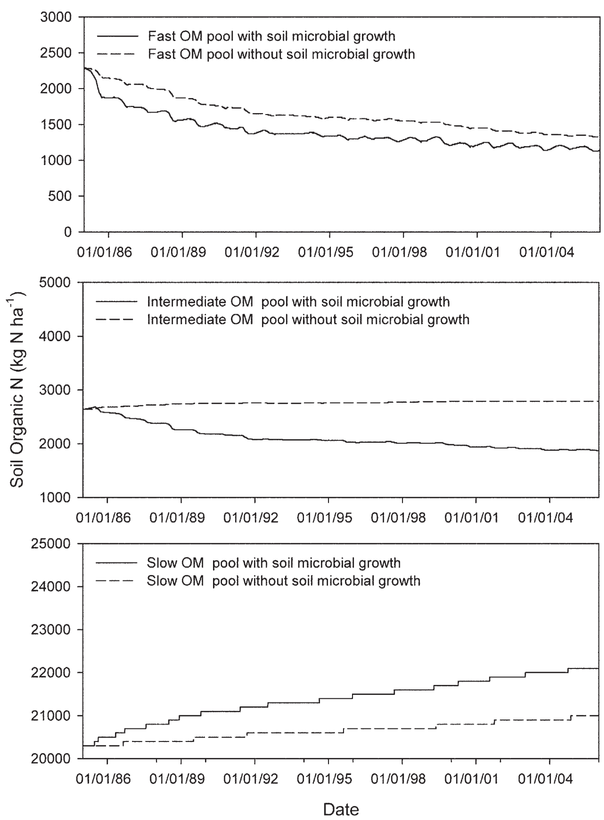

Fig. 8. Simulated organic matter (OM) pool dynamics with and without soil microbial growth for the high-N treatment from 1985 to 2005.

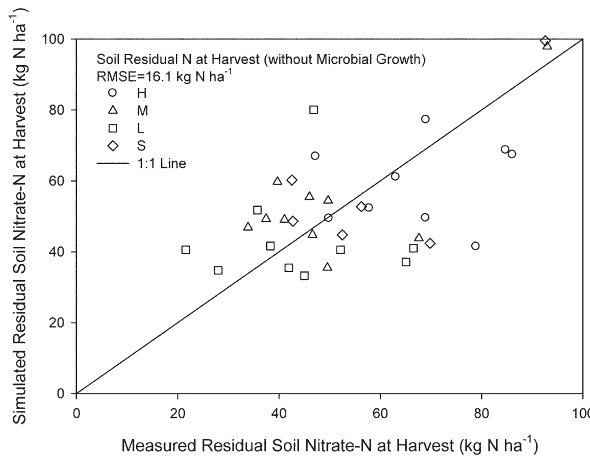

Fig. 9. Measured and simulated residual soil NO_3–N after harvest for the high (H), medium (M), low (L), and split (S) N treatments without soil microbial growth.

dynamic soil microbial growth (Table 1). Net N mineralization rates without microbial growth were lower in the first few years, but were comparable to the results with dynamic soil microbial growth after the initialization period from 1985 to 1995 (Fig. 7). Annual net N mineralization was 104, 97, 88, and 102 kg N ha^{-1} from 1996 to 2005 (Table 1). The dynamics of the organic pools were different, however, between the microbial growth options (Fig. 8). When dynamic microbial growth was simulated, the fast and intermediate pools lost N more rapidly than when soil microbial growth was not simulated, which was caused by simulation of much higher microbial populations. Simulated microbial populations stabilized at about six, two, and two times higher than the constant populations used in this study for P_{het}, P_{aut}, and P_{ana}, respectively. As a result, the slow pool gained N more rapidly from the fast and intermediate pools when dynamic microbial population growth was simulated. The change in total soil organic N during the experimental period of 1996 to 2005 was 100 kg N ha^{-1} (0.4%) without soil microbial growth, compared with 280 kg N ha^{-1} (1.1%) with dynamic soil microbial growth. As a result, the model without soil microbial growth simulated slightly higher residual soil NO_3–N at harvest than the model with dynamic soil microbial growth, which improved residual soil NO_3–N simulation, with a RMSE of 16.1 kg N ha^{-1} (ME = 0.34) compared with 22.6 kg N ha^{-1} (ME = −0.32) with dynamic soil microbial growth (Fig. 9).

Simulated annual N loss in tile flow was similar to the results with dynamic microbial growth, with a RMSE of 10.2 (ME = 0.66) and 9.6 kg N ha^{-1} (ME = 0.70), respectively. Simulated FWANC without soil microbial growth was comparable to those with dynamic microbial growth, with a RMSE of 3.5 mg N L^{-1} (ME = 0.26). In general, simulated N loss and FWANC without soil microbial growth were highly correlated with the results obtained with microbial growth, with r^2 values of 0.94 and 0.90, respectively (Fig. 10). Simulated crop yield was also similar to that obtained with dynamic microbial growth, with RMSEs of 1137.6 kg ha^{-1} (ME = 0.38) and 249.2 kg ha^{-1} (ME = 0.67) for corn and soybean, respectively. The major difference in simulated yield between the two microbial growth options was for the L treatment in 1998,

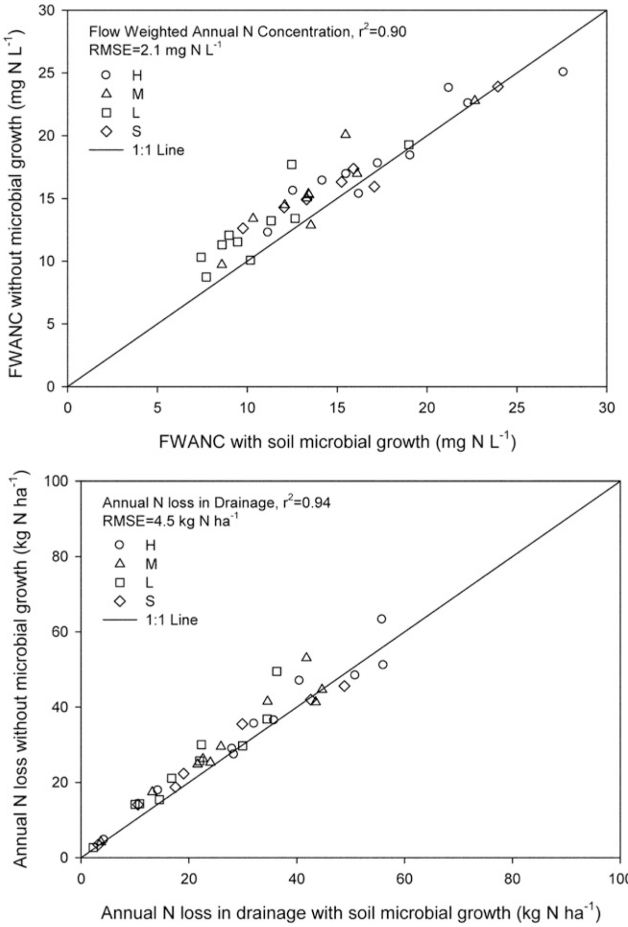

Fig. 10. Simulated flow-weighted annual N concentration (FWANC) and annual N loss in tile drainage with and without soil microbial growth for the high (H), medium (M), low (L), and split (S) N treatments.

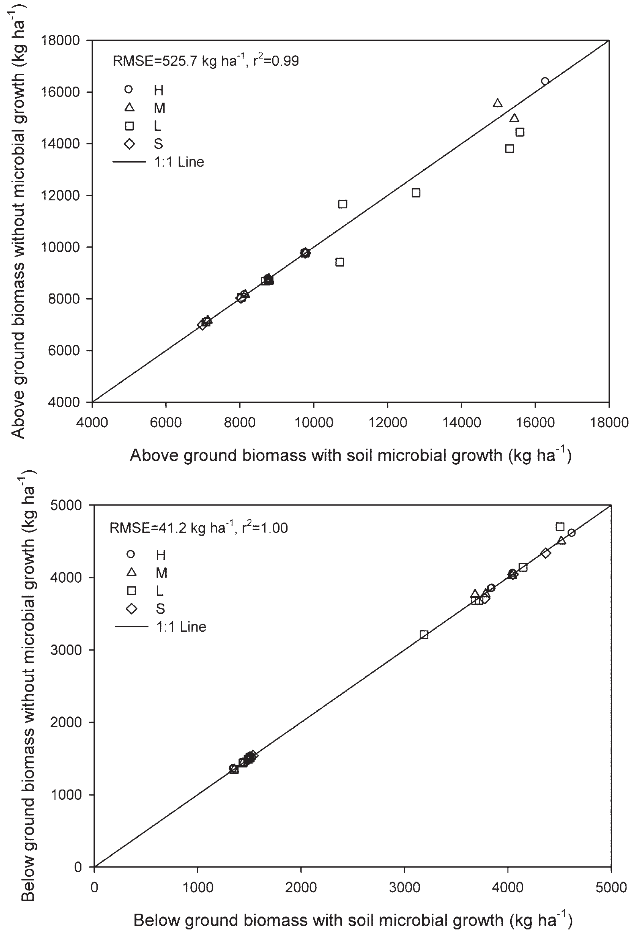

Fig. 11. Simulated above- and belowground biomass with and without soil microbial growth for the high (H), medium (M), low (L), and split (S) N treatments.

when simulated corn yield was 4319 kg ha^{-1} without microbial growth and 5388 kg ha^{-1} with microbial growth, compared with a measured yield of 6676 kg ha^{-1}. Simulated crop biomass with and without soil microbial growth was also similar, with an r^2 of 0.99 (Fig. 11). The model-simulated N loss in lateral flow was 15 to 20% of the N loss in tile flow, which suggested that most of the N loss was discharged to surface water via tile flow compared with the Nashua site in northeast Iowa from the study by Ma et al. (2007b, 2007c, 2007d), who simulated as much N loss in lateral flow as in tile flow. Simulated volatilization was doubled without soil microbial growth, but the N mass lost to volatilization was still a very small amount (1.1–9.5 kg N ha^{-1}). Simulated yearly net N mineralization was similar in most years between the two microbial growth options, except for the first year (Fig. 12). Therefore, using the constant soil microbial population option in the RZWQM may provide similar but more consistent results with minimal model initialization.

Although the RZWQM was initially developed to use up-to-date science for its simulated processes, users need to be trained to correctly use the model. The uncertainty associated with soil organic C pools directly affects the performance of the model, such as response to N application. By reducing the mineralization rate in this study, the model became more sensitive to the N application rate than previous results obtained by Thorp et al. (2007) for the same data (H, M, and L), suggesting that results from both studies may be questionable without knowing the exact mineralization rate in the experimental field. If the actual mineralization rate is lower than simulated by Thorp et al. (2007), the unbiased N responses simulated in their study were compensated by an increasing mineralization rate. The differences between this study and that of Thorp et al. (2007) also demonstrate the importance of model initialization and preference in model calibration by users, although both studies came to the same conclusion. Since the objective of this

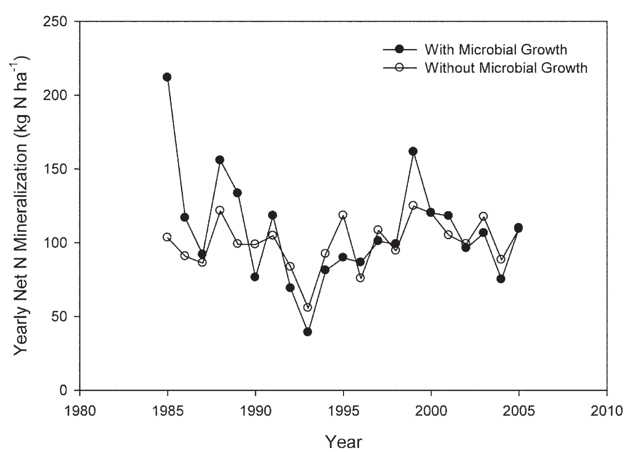

Fig. 12. Annual mineralization of soil organic N simulated for the high-N treatment with and without soil microbial growth.

study was to compare the model behavior without microbial growth, we did not recalibrate the soil and plant parameters except for the three parameters mentioned above. There is no question that the results can be improved with further calibration, such as using different root growth factors for different crops in DSSAT4.0, as suggested by Thorp et al. (2007). Another issue is that the model did a poor job in simulating plant N uptake, with a negative ME, which may need to be improved first before worrying about small differences in N loading in tile flow (Malone and Ma, 2008).

CONCLUSIONS

In this study, the newly developed RZWQM-DSSAT4.0 was tested for simulating crop yield, grain N, annual N loss in tile drainage, annual tile drainage flow, and residual soil NO_3–N after harvest under four N treatments (H, M, L, and S). Model simulation results (with microbial growth) showed that corn yield and corn grain N uptake were slightly overestimated for the H treatment and underestimated for the L treatment. Soybean yield and N uptake did not change much from year to year and the model provided adequate predictions for both; however, the model underestimated residual soil NO_3–N at harvest. Simulated N losses in tile drains responded well to N application rates, with the highest simulated FWANC for the H treatment and the lowest FWANC for the L treatments. Other simulated N fates were also within experimental observations reported in the literature; however, the model failed to simulate the lower yield under the S treatment than under the M treatment, as observed by Jaynes and Colvin (2006). Therefore, there is a need to improve crop responses to the timing of N application, as is the case in the S treatment.

The new option in the RZWQM without soil microbial growth provided very similar results to those of the model with dynamic soil microbial growth, suggesting that it may not be necessary to simulate soil microbial growth for long-term studies. Simulated results with and without soil microbial growth were highly correlated, with $r^2 > 0.90$. The errors between the two options were well within the experimental errors measured in the field. For residual soil NO_3–N, simulation results without soil microbial growth were better than the ones with dynamic soil microbial growth. Further evaluation of using the RZWQM without soil microbial growth is needed for other soils and climate conditions, especially where soil water is limited and soil organic C is different. The constant microbial populations may have to be set differently for different soil organic matter contents. This study was more for proof of the concept by demonstrating the possibility of using constant soil microbial populations for the complex soil C and N module in the RZWQM. Rigorous evaluation of each soil C and N process (e.g., mineralization, nitrification, denitrification) and each soil C and N pool against experimental data should be done before the role of microbial growth can be fully understood in the RZWQM.

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