Manure Management in an Irrigated Silage Corn Field: Experiment and Modeling

Liwang Ma, M. J. Shaffer,* J. K. Boyd, R. Waskom, L. R. Ahuja, K. W. Rojas, and C. Xu

ABSTRACT

On agricultural lands, animal waste disposal as fertilizer has been practiced since the beginning of agriculture. However, the practice has been an environmental concern in recent years due to over disposal of animal waste in some instances. This study evaluated soil NO_3 response to beef-manure application on a corn (*Zea mays* L.) field and tested the Root Zone Water Quality Model (RZWQM) for manure management. The experiment site was located in Northeastern Colorado on a silage-corn field with a history of fertilization with beef manure every fall after corn harvest. To study the residual effect of long-term manure application, 582 kg ha^{-1} of manure-N was applied to the east side of the field in the Fall of 1993, 1994, and 1995, while the west side received manure in 1993 only. Average silage-corn yields from the west site were 25.4, 31.9, and 22.5 Mg ha^{-1} for 1994, 1995, and 1996, respectively, which were not significantly different from that harvested from the east site (25.1, 30.9, and 24.3 Mg ha^{-1}, respectively). Average soil NO_3 concentrations decreased significantly from 14.9 to 8.5 mg N kg^{-1} in the top 30 cm of soil, and from 5.4 to 3.7 mg N kg^{-1} in the 30- to 60-cm soil profile after stopping manure application. No significant difference in soil NO_3 concentrations between the manured and not-manured sites was found below 60 cm. Average plant N uptake ranged from 140 to 362 kg N ha^{-1} and was not significantly different between the two sites. The RZWQM was calibrated on the basis of the measured silage-corn yield and plant N uptake, and was then used to predict soil NO_3 concentration and total water storage in the soil profile. Generally, the calibrated model provided adequate predictions for both NO_3 and soil water content with $r^2 > 0.83$. The model was further used to evaluate alternative scenarios of manure and water management.

ANIMAL WASTES are among the most economical fertilizers for plants and also enhance long-term productivity of soils. However, extensive use of these materials on agricultural land is likely to result in potential contamination of the atmosphere, groundwater, and surface water bodies. As a result, numerous studies have been conducted to monitor N and P concentrations in soils after manure applications, and mathematical models have been used as tools to evaluate the environmental impact of various manures under field conditions (Yoon et al., 1994; Minkara et al., 1995; Gangbazo et al., 1995). Yoon et al. (1994) and Minkara et al. (1995) applied the GLEAMS (Groundwater Loading Effects of Agricultural Management System) model to NO_3 and NH_4 losses in surface and subsurface runoff from poultry litter applications. Large differences between experimental observation and model simulation were obtained for all manure application rates and experimental sites. Edwards et al. (1994) used the EPIC (Erosion Productivity Impact Calculator) model to predict runoff transport of NO_3 in a poultry-manured tall fescue (*Festuca arundinacea* Schreber) field and obtained poor prediction of NO_3 in runoff losses. Jabro et al. (1993) evaluated the LEACHM (Leaching Estimation And CHemical Model) and the NCSWAP (Nitrogen, Carbon, Soil, Water And Plant) models for predicting NO_3 leaching in a corn field applied with liquid dairy manure. They found that both models provided poor predictions of NO_3 leaching below the 1.2-m soil depth, with LEACHM statistically performing better than NCSWAP. Lengnick and Fox (1994) found that it was difficult to calibrate both soil-NO_3 and leachate-NO_3 concentrations to experimental observations using the NCSWAP model. They attributed the simulation errors to the model's incapability of describing macropore flow and N dynamics. Boyd (1996) was able to budget N in a continuously manured corn field using the NLEAP (Nitrate Leaching and Economic Analysis Package) model, and he also suggested the best manure-management practices based on his simulation results. Parton and Rasmussen (1994) tested a long-term simulation model (CENTURY) on N prediction in a wheat–fallow system with 56 yr of barnyard manure application, and obtained excellent agreement with experimental measurements.

The RZWQM is a field-oriented research model developed in response to increasing needs for scientific support in agricultural management and consulting. The model incorporates the state-of-the-science knowledge related to agricultural systems. Several of its components have been satisfactorily validated, such as water transport (Ahuja et al., 1993, 1995), pesticide movement (Ahuja et al., 1995, 1996; Ma et al., 1995, 1996; Azevedo et al., 1997), evapotranspiration (ET) (Farahani and Bausch, 1995), subsurface drainage (Johnsen et al., 1995; Singh and Kanwar, 1995a,b; Singh et al., 1996), organic matter (OM)–N cycling (Hansen et al., 1995; Shaffer et al., 1992), and plant growth (Nokes et al., 1995). Overall performance of the model under field conditions was also evaluated by Farahani et al. (1995) and Buchleiter et al. (1995). The objectives of this study were to evaluate soil-NO_3 response to manure application on a corn field and to test the RZWQM for manure and water management.

MATERIALS AND METHODS

Root Zone Water Quality Model Description

The RZWQM is a process-based agricultural management model. It contains water, chemical, and heat transport modules; a plant growth module; an ET module; an equilibrium chemistry processes module; an OM–N cycling module; a pesticide fates module; and a management practices module. Some of these are still in the process of being tested. Selected

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Abbreviations: ET, evapotranspiration; OM, organic matter; RZWQM, Root Zone Water Quality Model.

processes are described below and more information is available in the RZWQM technical documentation (RZWQM Team, 1992) and the RZWQM user's manual (RZWQM Team, 1995).

Water and Chemical Movement

Water and chemical transport has been described and tested by Ahuja et al. (1993, 1995). A modified Brooks–Corey equation is used to describe hydraulic properties of the soil. Saturated hydraulic conductivity (K_s, cm h^{-1}) is calculated from effective porosity ϕ_e (Ahuja et al., 1989). Maximum water infiltration rate during rainfall or irrigation events is calculated with a modified Green-Ampt equation (Ahuja et al., 1995). When the rainfall or irrigation rate is less than the maximum infiltration rate, the actual infiltration rate is set to the rainfall rate or irrigation rate. If the rainfall rate is greater than the maximum infiltration rate, excess water is assigned to macropore flow or surface runoff. Water content is updated at each time interval depending on infiltration rate and water deficiency. Infiltrated water is then partitioned between mobile (called *mesopores* in the model, macropores are absent in this study) and immobile (called *micropores* in the model) water. The microporosity can be either provided by the user or set to soil water content at water tension of 200 kPa using the Brooks–Corey equation. Water is first assigned to fill the micropores and then to mesopores. Between rainfall or irrigation events, soil moisture is redistributed according to the Richards' equation. Root water uptake is evaluated using the approach of Nimah and Hanks (1973).

For the purpose of solute transport, NO_3 is assumed to be conservative and has an adsorption constant (K_d) value of zero. To account for physical nonequilibrium, the soil solution is divided into mobile (water in the mesopores of the model) and immobile fractions (water in the micropores). Prior to infiltration, NO_3 concentrations in the mobile and immobile waters are in equilibrium. During the infiltration period, only about 50% or less of the mesopores are assumed to be piston-displaced followed by an instantaneous mixing of solution in the mesopores. At the end of an infiltration event, water and NO_3 in the meso- and micropores are allowed to equilibrate. Nitrate is transported with water from layer to layer.

Organic Matter–Nitrogen Cycling

In RZWQM, residues (crop stover, manure, and other organics) are partitioned into fast and slow pools based on their C/N ratios. The fraction of residue materials to the fast residue pool (S) is calculated from:

$$S = \left(\frac{1}{C_N(\text{new})} - \frac{1}{C_{NS}}\right) \Big/ \left(\frac{1}{C_{NF}} - \frac{1}{C_{NS}}\right) \qquad [1]$$

where C_N(new) is the C/N ratio of new added materials to the soil, C_{NS} and C_{NF} are the C/N ratios of slow and fast residue pools, respectively. The fast residue pool has a C/N ratio of 80, and the slow one has a C/N ratio of 8 (modified to account for manure).

There are three soil OM, or humus, pools with C/N ratios of 8 (fast pool), 10 (medium pool), and 12 (slow pool), respectively. These five pools are dynamically linked together as shown in Fig. 1. In addition, there are three living microorganism pools for aerobic heterotrophs (Microbial pool 1), autotrophs (Microbial pool 2), and facultative heterotrophs (Microbial pool 3). Residue and OM pools are subject to a first-order decay with respect to its C concentration:

$$r(i) = k_d(i)C(i) \quad 1 < i < 5 \qquad [2]$$

where $r(i)$ is the decay rate of pool i (μg C g^{-1} d^{-1}), and $C(i)$ is C concentration (μg g^{-1} soil). The $k_d(i)$ is a first-order decay rate coefficient (d^{-1}) and is affected by soil water O_2 concentration, soil pH, ion strength, heterotrophic microbial population, soil temperature, and degree of soil water saturation (Hansen et al., 1995; Shaffer et al., 1992).

A fraction of the decayed residue and OM is transferred to other pools, as denoted by R_{14}, R_{23}, R_{34}, and R_{45} in Fig. 1. Because the C/N ratios are different among pools, adjustments to maintain N conservation are made during the transformations. The rest of the C goes off as CO_2 and the model also uses CO_2 as a C source. Nitrogen is released as NH_4 from the residue and OM pools during the decaying processes. Ammonium is then nitrified to NO_3 following a zero-order equation as a function of soil temperature, soil O_2 concentration, soil pH, ion strength, autotrophic microbial population, and degree of water saturation (Shaffer et al., 1992). Nitrate from nitrification or applied commercial fertilizers is subject to denitrification under anaerobic conditions, and the denitrification rate is described by a first-order equation. The denitrification rate coefficient is a function of soil temperature, soil pH, ion strength, total soil C, anaerobic microbial population, and the degree of water saturation (Shaffer et al., 1992).

Minimum microbial populations set in RZWQM are 50 000, 500, and 5000 organisms g^{-1} soil for Microbial Pools 1, 2, and 3, respectively. There is no further microbial death if populations are less than their respective minimum populations. The growth of heterotrophs (heterotrophic decomposers and facultative anaerobes) is calculated from OM decay by assuming a fraction of decayed OM components being transferred to microbial biomass. Decayed OM-C has three possible fates: (i) transfer to another OM pool, (ii) microbial biomass, and (iii) CO_2. Autotroph (nitrifier) growth is proportional to nitrification rate. Denitrifier (facultative anaerobes) can also grow under anaerobic conditions and decompose soil residue and soil OM. Death rates for the three microbial populations are calculated as first-order equations with respect to their biomass and the rate coefficients are functions of soil temperature, soil pH, ion strength, soil O_2 concentration, total soil C, NO_3 concentration, and the degree of water saturation. Ammonia volatilization is estimated from the partial pressure gradient of NH_3 in the soil and air, with due consideration of wind speed and soil depth (Crane et al., 1981; Hoff et al., 1981).

Evapotranspiration

The RZWQM ET subroutine has been described in detail and tested by Farahani and Bausch (1995). Total potential ET is calculated from Shuttleworth and Wallace (1985) and partitioned between evaporation and transpiration based on energy received by the canopy and soil. Evaporation is further divided between bare soil and surface residues. Actual transpiration is conditional on water availability and plant root activity. Actual residue evaporation is equal to potential residue evaporation if the amount of water in the residue surface is equal to or greater than the demand. Actual soil evaporation is equal to potential soil evaporation if the soil conductivity is sufficient to produce the demand at the surface, otherwise it is equal to the maximum amount of water allowed by the soil water flux.

If the soil is dry and cannot use the energy to produce evaporative demand, the model assumes 60% of the unused energy is available first to the residue, and then to the plant canopy. If the residue is not able to produce its evaporative demand, 60% of the unused energy is available for the plant canopy (DeCoursey, 1992).

General Plant Growth Model

This submodel of RZWQM is a generic crop-production model capable of predicting the relative responses of corn and soybean [*Glycine max* (L.) Merr.] to environmental variance and management practices (Nokes et al., 1995; Hanson and Hodges, 1992). It divides a plant into seven phenological growth stages: (i) dormant stage, (ii) germinating stage, (iii) emergence stage, (iv) four-leaf stage, (v) vegetative growth stage, (vi) reproductive stage, and (vii) senescent stage. Plant development is driven by growing degree days and plant growth is driven by photosynthesis. A plant requires a certain amount of degree days before advancing from one phenological stage to another. Photosynthesis is assumed to start when fully functional plant leaves are established at the four-leaf stage. Plants in one growth stage can remain alive in the current stage, pass to the next stage, or die. Plant population development is estimated from a transition probabilities matrix, which is updated daily according to environmental stresses and phenological growth stage (Hanson and Hodges, 1992).

Nitrogen uptake by plants is passive if the amount of N flow into the plants through water transpiration meets plant N demand, otherwise, active N uptake is required according to the Michaelis–Menton equation (Hanson and Hodges, 1992). The total amount of N taken up by plants is then partitioned into each soil layer in proportion to the root density function. In addition, the model assumes equal availability of NO_3 and NH_4 to plants.

Agricultural Management Practices

Organic wastes (manure and beddings) are treated as residues and partitioned into slow and fast residue pools according to Eq. [1], whereas the amount of NH_4 in the manure is added into the NH_4 pool directly. Surface residues are incorporated into the soil (residue pools) through tillage or biological activities. Beside its effects on residue mass, tillage also reduces soil bulk density, and prevents surface crusting and continuity of microporosity. Soil reconsolidation after rainfall or irrigation is also simulated in the model (Rojas et al., 1992).

Field Experiment Design

A 4-ha field was selected in Weld County near Lucerne, CO. For the last decade, this field has had a history of beef-manure fertilization every fall after silage-corn harvest, without any inorganic fertilizer application. The experiment plots were on a Vona sandy loam soil (coarse-loamy, mixed, mesic, Ustollic Haplargid). The water table was approximately 8 m below the ground level. The experiment plots were positioned in the middle of the field. Three plots (15 by 15 m) were located on the east half and three plots on the west half of the field. The field was irrigated in alternate furrows with ditch water (1.3 μg L^{-1} NO_3–N) by placing 5-cm-diam. siphon tubes on both the south and north ends of the furrows. Each irrigation event lasted for 12 h with a total application of 20 cm. The farmer usually irrigated four to six times during the months of July and August.

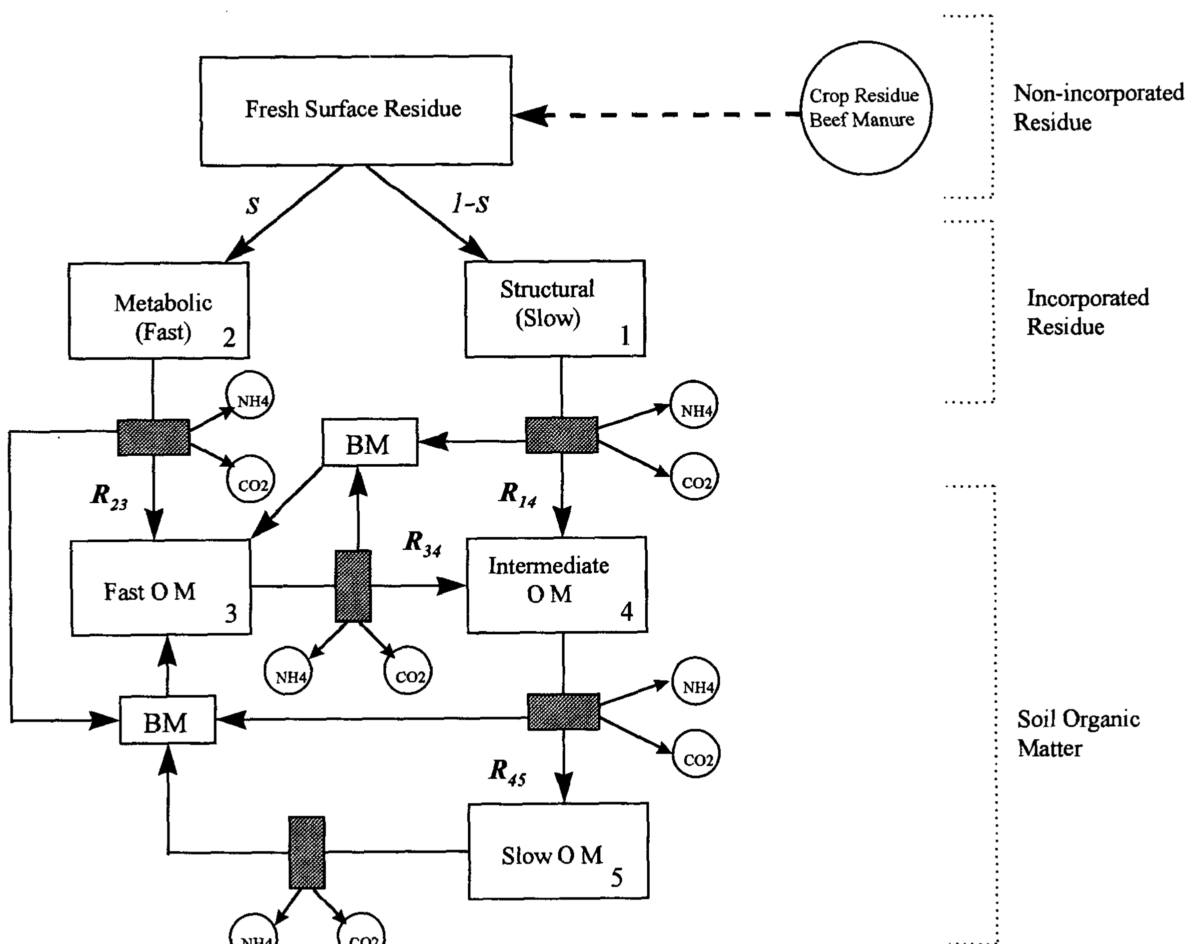

Fig. 1. A schematic diagram of residue and soil organic matter (OM) pools. R_{14}, R_{23}, R_{34}, and R_{45} are inter-pool mass transfer coefficients. BM is biomass of microbial pools.

The experiment was set up in the Fall of 1993 and finished in the fall of 1996. The farmer applied ≈44.8 Mg ha^{-1} of beef manure (on dry weight basis) to both the west and east sides of the field in the fall of 1993 (mid-October), as he had in the last 10 yr. In the fall of 1994 and 1995, the farmer applied 44.8 Mg ha^{-1} only to the east half of the field, whereas the west half of the field received no manure. The applied manure was incorporated into the soil after 1 to 2 d with a moldboard plow. A sample of the manure was collected and analyzed for total N, NH_4–N, NO_3–N, and moisture content.

Soil samples for soil NO_3 measurement were collected to a depth of 150 cm three times in 1994, four times in 1995, and three times in 1996. At least three soil cores (3.8-cm diam.), composited at 30-cm increments, were taken from each experiment plot.

Nitrate in the soil was extracted with 1.0 *M* KCl and analyzed using the Cd-reduction method on a Lachat Instruments Autoanalyzer (Mequon, WI) (Nelson, 1983). Silage-corn yield at harvest was estimated from aboveground biomass sampled from three randomly selected 4-m-long rows. Nitrogen content of the corn silage was measured from total Kjeldahl N and used to calculate total N uptake. Soil water content was monitored once every 2 wk with a neutron probe (503DR Hydroprobe Moisture Soil Depth Gauge, CPN International Inc., Martinez, CA) during the irrigation season and by the oven-dry method during the nonirrigation season (Boyd, 1996).

Soil OM content, soil pH, and soil texture were measured by the Colorado State University Soil, Water, and Plant Testing Laboratory. The modified Walkey and Black method was used for soil OM content (Allison, 1965), and soil texture was measured with the hydrometer method (Gee and Bauder, 1986). Soil bulk density was measured using the core method (3.8 cm in diam.), and 33 kPa soil water content was estimated using the ceramic pressure plate method. Climatic data were obtained from a weather station 0.4 km southwest of Lucerne. Figure 2 shows measured daily rainfall and daily average air temperature from 1993 to 1996. Agricultural management information such as manure application, irrigation, tillage, and planting and harvesting dates, were recorded.

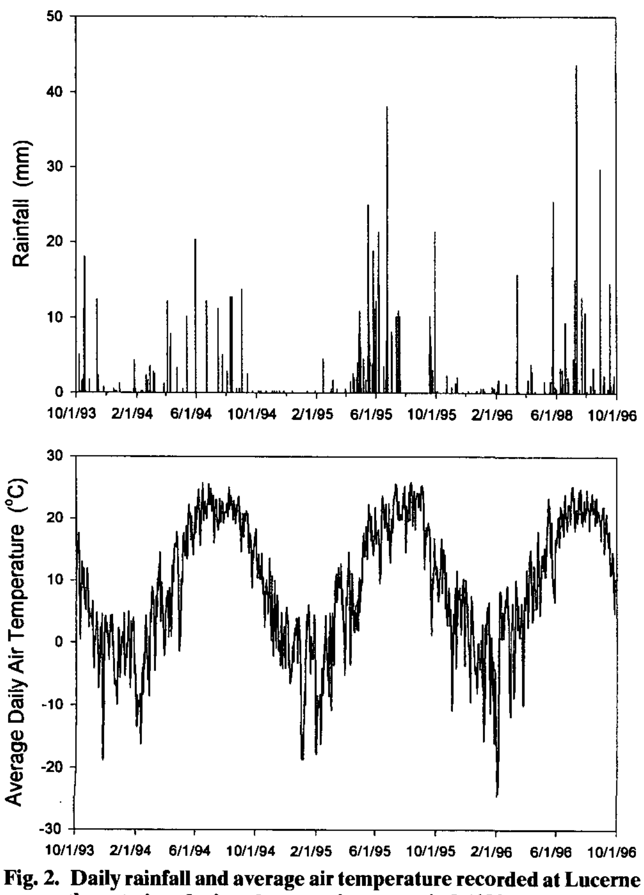

Fig. 2. Daily rainfall and average air temperature recorded at Lucerne weather station during the experiment period (1993–1996).

Table 1. Measured soil physical properties used in the Root Zone Water Quality Model (RZWQM).

Soil depth (cm)	0–30	30–60	60–90	90–120	120–150
Bulk density (g cm^{-3})	1.4	1.5	1.5	1.5	1.5
Organic matter (%)	1.7	0.7	0.6	0.3	0.3
Sand (%)	63	61	41	64	48
Silt (%)	15	15	28	16	28
Clay (%)	22	24	31	20	24
33 kPa water content (cm^3 cm^{-3})	0.227	0.222	0.279	0.188	0.279

RESULTS AND DISCUSSION

Experimental Results

Applied manure had a C/N ratio of 8:1 and 21 g kg^{-1} water content. Total N applied was 582 kg ha^{-1}, with 27 kg ha^{-1} identified as NH_4–N. Measured soil texture, bulk density, OM content, and 33 kPa soil moisture content are listed in Table 1. During the 3 yr of the study, soil cores were sampled in the crop rows for residual NO_3–N analysis. To check the effect of sampling

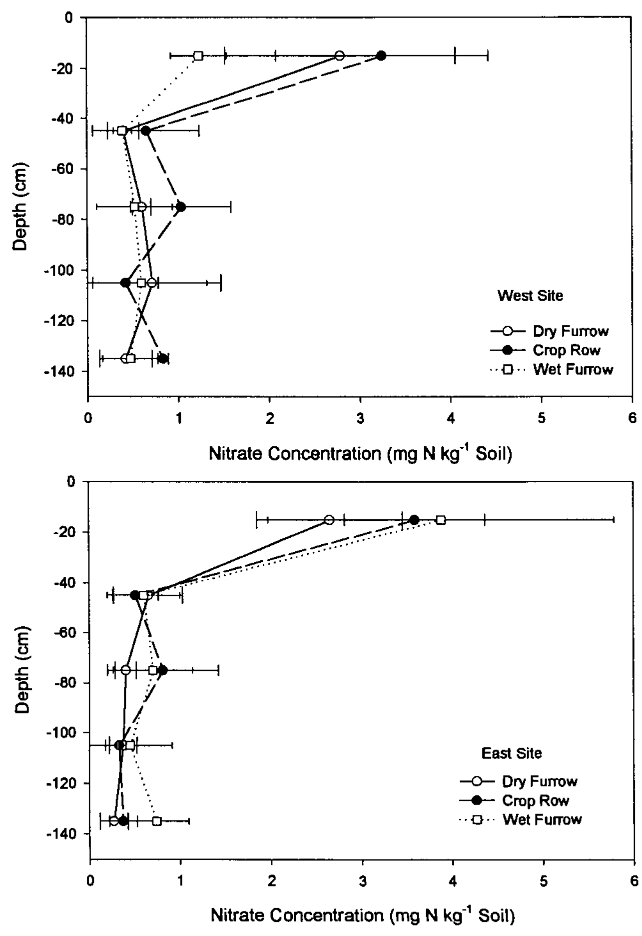

Fig. 3. Nitrate concentration measured from crop rows, dry and wet furrows. No significant differences were found among sampling locations (P = 0.254). East site received manure applications in Fall of 1993, 1994, and 1995; west site received manure application in Fall 1993 only.

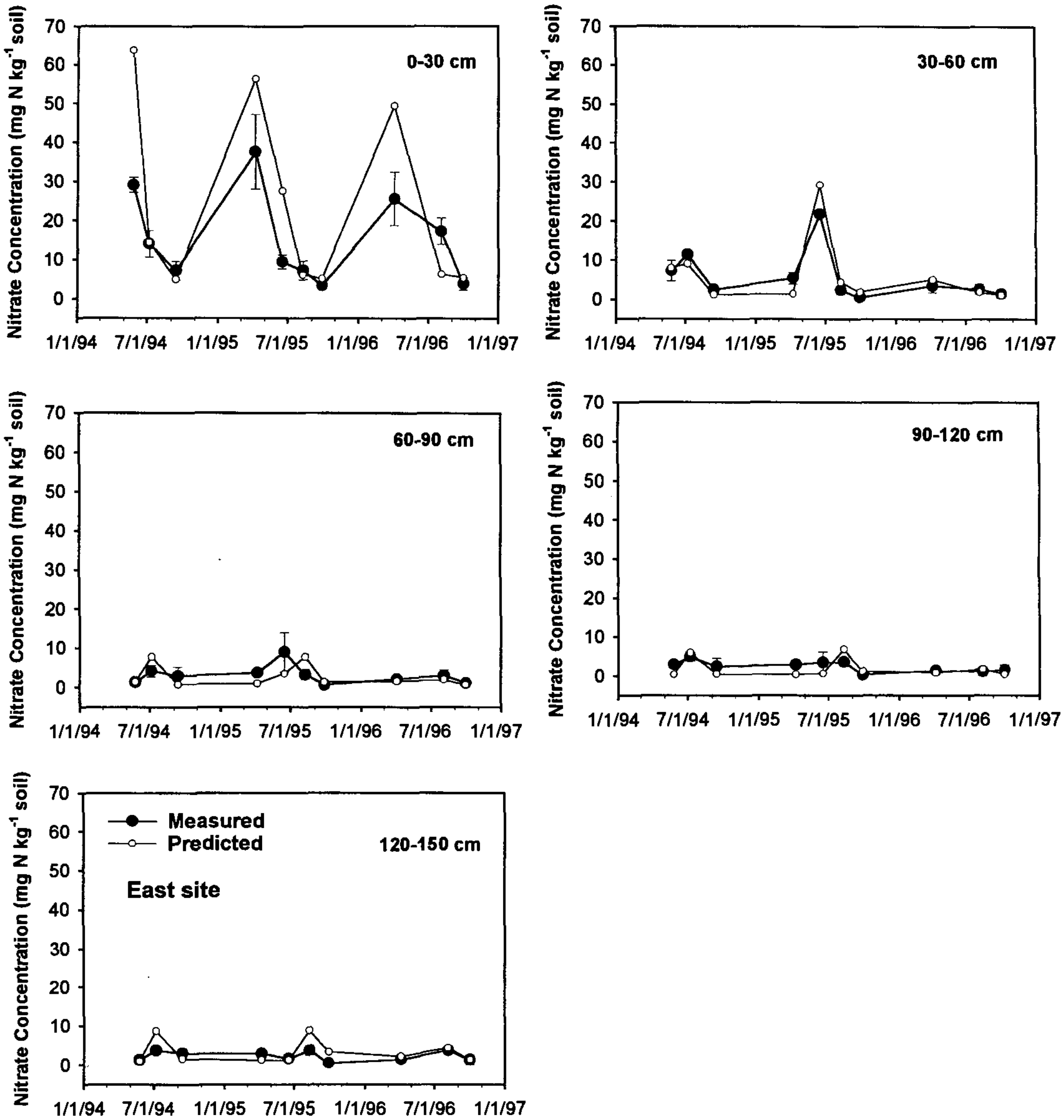

Fig. 4. Measured and predicted NO_3 concentrations in the soil profile in the east site where manure was applied every year. East site received manure applications in Fall of 1993, 1994, and 1995.

location on residual soil NO_3 measurement, soil samples from the crop rows were compared with those from dry and wet furrows on the sampling date of 28 Sept. 1995. The NO_3–N concentrations at each 30-cm increment of the soil profile are shown in Fig. 3. Using analysis of variance (ANOVA), there were no significant differences in residual NO_3 concentrations between samples from the crop rows and the furrows (P = 0.254). In addition, no significant difference was found between the west and east sites (P = 0.306). Measured NO_3 concentrations were significantly higher in the top 30-cm soils than that in the lower soil layers ($P < 0.001$). No significant differences in measured NO_3 concentrations were found among the lower soil depths ($P < 0.05$).

Soil NO_3 concentrations at different soil depths are presented in Fig. 4 and 5 along with standard errors. During the 3 yr of the experiment, NO_3 seasonal changes showed similar patterns for both the west and east sites. No significant differences in measured NO_3 concentrations were found between the west and east sites in 1994 (P = 0.110–0.862) for all the soil depths, probably because the same manure treatment was used for both the west and east sites prior to 1994. However, ANOVA analysis showed significantly higher NO_3 concentrations in the east site (manured) than in the west site (not manured) at soil depths of 0 to 30 cm and 30 to 60 cm ($P < 0.001$) after the zero manure treatment initiated in 1994. No significant differences were observed at soil depths of 60 to 90 cm (P = 0.451), 90 to 120 cm (P = 0.175), and 120 to 150 cm (P = 0.648). Therefore, the effects of manure on soil NO_3 were limited to only the top 60 cm. In addition, NO_3 concentration in the soil profile was distributed in the order of: 0 to 30 cm > 30 to 60 cm > 60 to 90 cm > 90 to 120 cm = 120 to 150 cm.

Total silage yields and total N uptake (above ground) are shown in Fig. 6. There was no significant difference in corn yields between the east and west sites (P = 0.925). However, the yields were significantly different between 1994 and 1995, and between 1995 and 1996 ($P < 0.001$). The yield difference between 1994 and 1996 was not significant (P = 0.313). Yield increase in 1995 was probably due to a wet spring in 1995 (Fig. 2) or

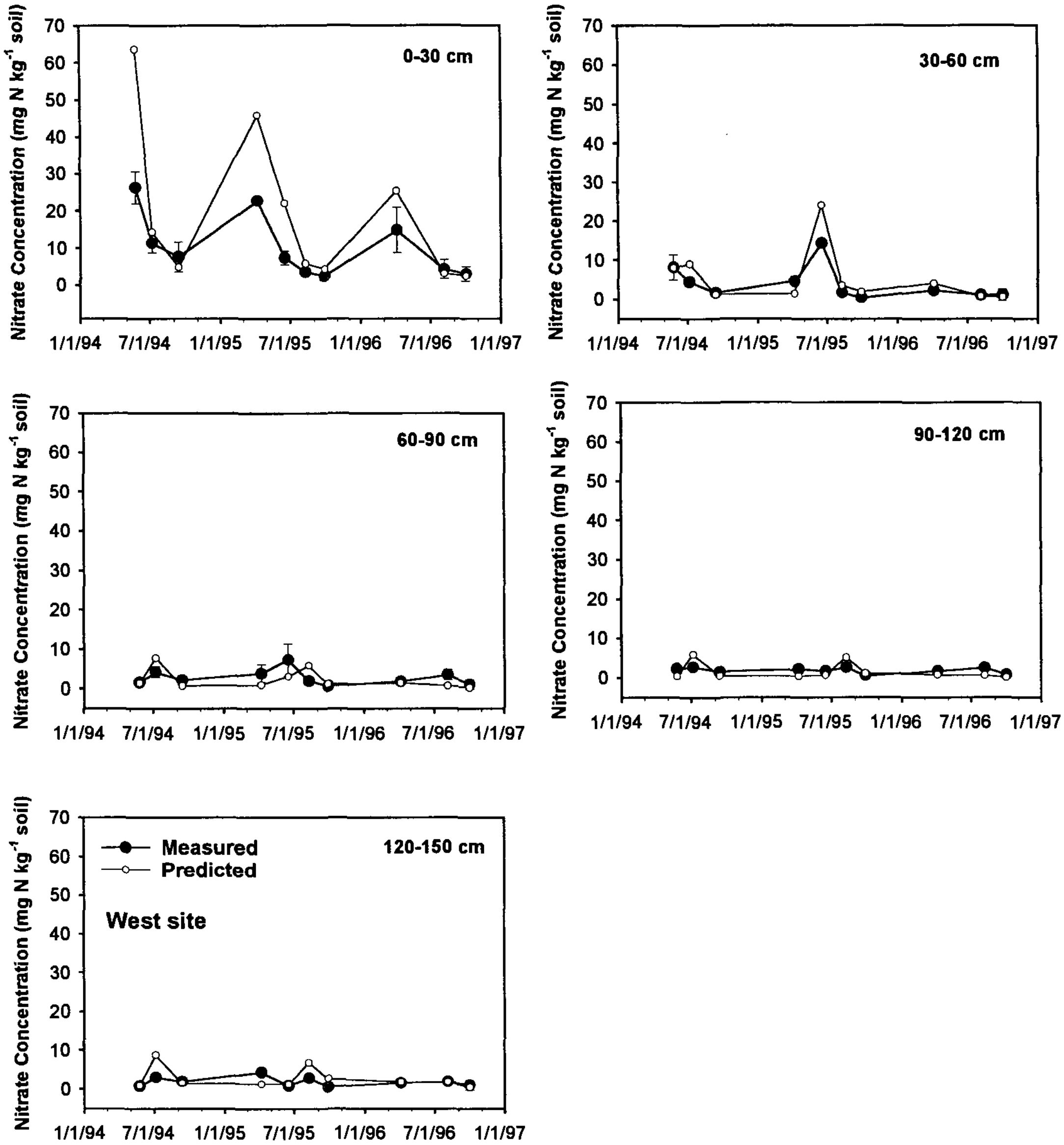

Fig. 5. Measured and predicted NO_3 concentrations in the soil profile in the west site where manure was not applied after 1993. West site received manure application in Fall 1993 only.

sampling error since the farmer's average selling yields were 22, 22, and 20 Mg ha^{-1} for 1994, 1995, and 1996, respectively. Nevertheless, silage-corn yield was not reduced significantly when manure was not applied for 2 yr following a decade of manure application. Nitrogen uptake was not significantly different between the east and west sites either (P = 0.165), although there was a significant difference in soil NO_3 concentrations between the west and east sites at the top 60-cm soil profile. Thus, plant N uptake was not limited by soil NO_3. While there was no significant difference in silage yields between 1994 and 1996, N uptake was significantly lower in 1996 than in 1994 (P = 0.025). The highest N uptake in 1995 was due to estimated high silage yield.

Modeling Results

To simulate the experiment results, we needed to characterize the RZWQM for the experiment site. At first, soil water contents were measured during the crop growing seasons in 1994 and 1995 with a neutron probe, and then used to check the hydraulic properties estimated from Brooks–Corey equations using measured soil bulk density, soil texture, and soil water contents at 33 kPa. Because only daily rainfall was recorded and no water runoff was observed, we assumed an average 2-h duration for all the rainfall events to eliminate water runoff during model simulation. Figure 7 shows simulated and measured water storage in the 150-cm soil profile. Model prediction is highly correlated with experimental measured values (r = 0.979), with better prediction for 1994 than for 1995. Simulated ET during the crop growing season was about 630 mm, which is higher than the 509 mm reported for grain corn by Farahani and Bausch (1995). Such a high prediction of ET may be attributed to the high leaf area index of silage corn.

Secondly, the model needed to be characterized for soil OM and microbial pools. Initially, soil OM was divided into three pools: fast (5%), intermediate (10%),

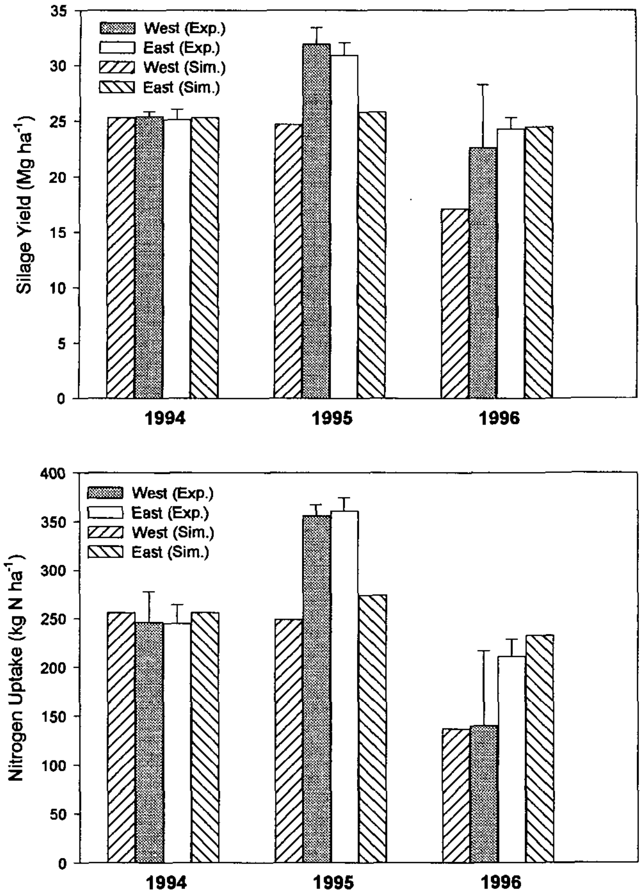

Fig. 6. Measured and simulated silage yield and total plant N uptake for 1994, 1995, and 1996 growing seasons. East site received manure applications in Fall of 1993, 1994, and 1995; west site received manure application in Fall 1993 only.

and slow (85%) pools, based on the guidelines in the RZWQM manual. Minimum microorganism populations of 50 000, 500, and 5000 organism g^{-1} soil were assigned to Microorganism Pools 1, 2, and 3, respectively (RZWQM Team, 1995). The decay rate coefficients for residue pools were based on C/N ratios, while those for the three OM (humus) pools were derived by assuming

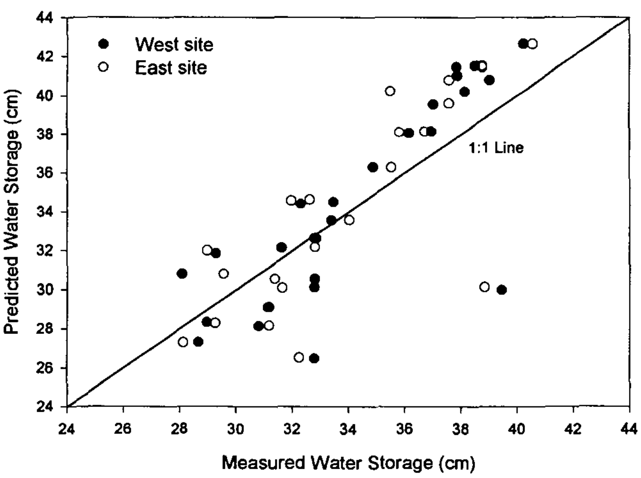

Fig. 7. Measured and predicted total water storage in the 150-cm soil profile in the 1994 and 1995 growing seasons.

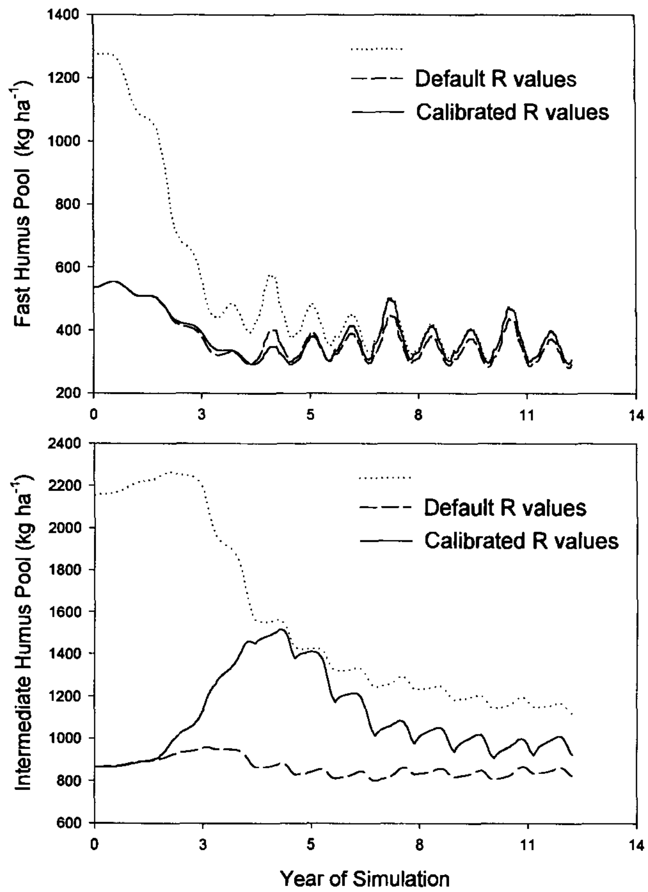

Fig. 8. Stability analysis of soil organic matter (OM) pools as affected by initial conditions and model parameters. Default R values are R_{14} = 0.1, R_{23} = 0.1, R_{34} = 0.6, and R_{45} = 0.4, and Calibrated R values are R_{14} = 0.6, R_{23} = 0.1, R_{34} = 0.1, and R_{45} = 0.1. Dash lines are simulations from the default R values with two different initial conditions, and the solid line is simulation from the calibrated R values.

a turnover time of 5, 20, and 2000 yr for the fast, intermediate, and slow pools, respectively (RZWQM Team, 1995). The model initialized using this method does not guarantee a dynamic steady state with respect to the fast soil OM and microorganism pools. Therefore, it is recommended to run the model for several simulation years prior to the experiment to stabilize the soil OM and microbial pools under constant management practices (RZWQM Team, 1998). Since we did not have recorded weather data for the experiment site prior to 1992, we used the weather data from 1993 to 1996 for the years prior to the experiment. To test the validity of such an approach, we obtained 15 yr of weather data from a nearby station in Greeley (8 km southwest of Lucerne) and applied them to our experimental site. Simulation results (not shown) indicated that OM and microbial pools reached the same steady states whether 15 yr of continuous weather data were used or 3 yr of weather data were repeated five times. Thus, the variability of weather data from 1993 to 1996 may be large enough to represent the long term uncertainty of a weather sequence under Colorado conditions.

Figures 8 and 9 show the dynamics of OM and microbial pools over 12 yr using weather information from

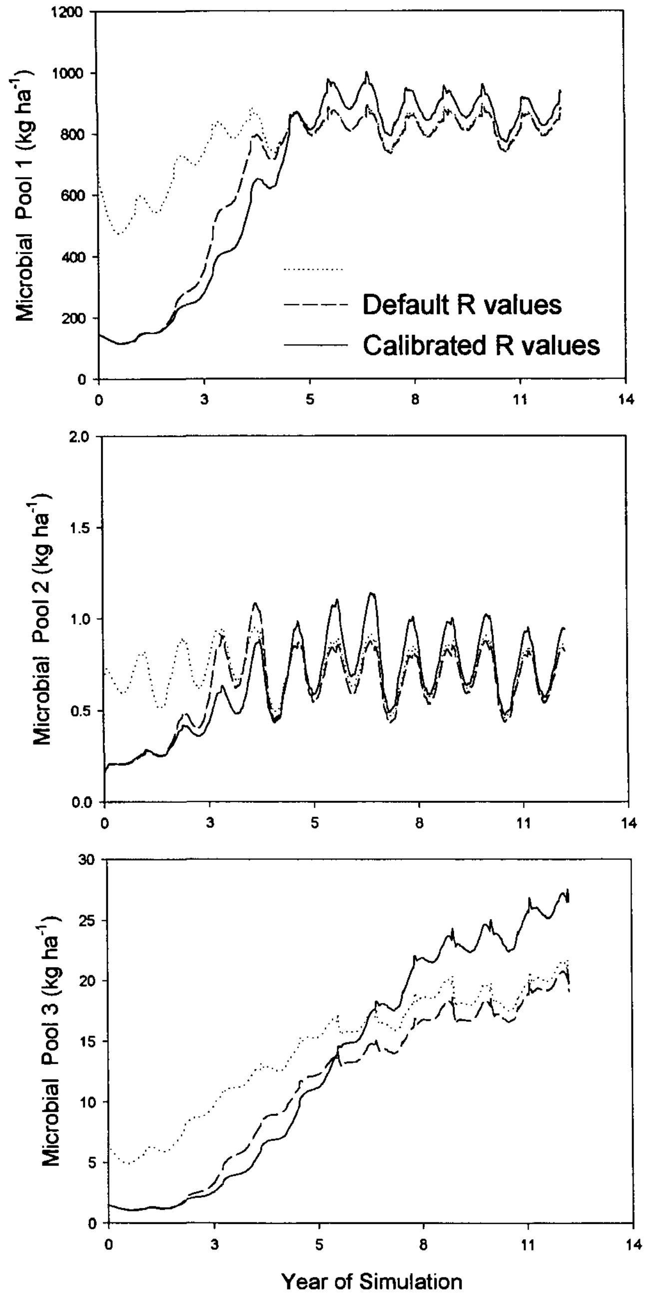

Fig. 9. Stability analysis of soil microbial pools as affected by initial conditions and model parameters. Default R values are R_{14} = 0.1, R_{23} = 0.1, R_{34} = 0.6, and R_{45} = 0.4, and Calibrated R values are R_{14} = 0.6, R_{23} = 0.1, R_{34} = 0.1, and R_{45} = 0.1. Dash lines are simulations from the default R values with two different initial conditions, and the solid line is simulation from the calibrated R values.

1993 to 1996. Generally, the OM and microbial pools were not at steady states initially, which probably was due to the unknown initial microbial pool sizes. With the self adjustment of microbial pools from given OM pools and environmental conditions, both OM and microbial pools were approaching their respective steady states. The fast soil organic pool reached a steady state after 6 yr, while the intermediate pool reached a steady state after 9 yr (Fig. 8). The slow organic pool took about 600 yr to reach steady-state (not shown). Thus, the initialization was primarily for the fast and intermediate pools. Once these two pools are stabilized, the slow humus pool should also be parameterized based on the amount of total OM content in the soil. The microbial pools reached a quasi steady state within 12 yr as well. After the 12 yr of initialization, the effect of initial values on pool stability (or simulation results) was minimized (Fig. 8 and 9).

Thirdly, the transfer coefficients between pools (R_{14}, R_{23}, R_{34}, and R_{45}) are particularly important for N cycling because they can be used to adjust overall manure and OM degradation rates. Initially, default R values (R_{14} = 0.1, R_{23} = 0.1, R_{34} = 0.6, and R_{45} = 0.4) were used. Then, the parameters were varied independently from 0.1 to 0.9 to check their effects on crop yield predictions. Because of possible interdependency of R parameters and initial conditions (Fig. 8 and 9), the model was initialized for 12 yr for each set (or combination) of model parameters to avoid the effect of initial conditions on simulated results. Figure 10 shows the effects of R_{14}, R_{23}, R_{34}, and R_{45} on predicted crop yields in 1995. Since we did not observe any yield decrease in 1995 for the west (not manured) site, calibrated R_{14}, R_{23}, R_{34}, and R_{45} should provide no or minimal yield reduction in 1995 at the west site. As a result, R_{14} = 0.6, R_{34} = 0.1, and R_{45} = 0.1 were selected. Since simulated yield was not sensitive to R_{23}, the default R_{23} = 0.1 was used. The calibrated new R values affected the steady states of OM and microbic pools as shown in Fig. 8 and 9 and other sensitivity analysis results (not shown). After 12 yr of initialization using the calibrated R values, OM pool distributions for the top 30 cm of soil were 2% for the fast pool, 11% for the intermediate pool, and 87% for the slow pool, with a total soil OM content of 1.7%. Ma et al. (1998) used the calibrated R values for manure management in a tall fescue field in Arkansas and obtained good prediction of manure effects on soil NO_3 concentration.

Fourthly, plant N uptake was used to calibrate the Michaelis–Menton equation. Calibrated maximum daily plant N uptake was 0.5 g plant^{-1} d^{-1} Simulated silage yields and N uptake are shown in Fig. 6. The model provided good descriptions for both 1994 and 1996, although we did not use 1996 data to calibrate the R values. However, the yield and N uptake were underestimated in 1995 by the RZWQM, which could be due to either model insensitivity to the wet spring in 1995 or experimental error in yield measurement. As mentioned above, the farmer did not observe high yields in 1995.

Now that the model was characterized for the experiment site, it could be used to simulate manure management effects. As shown in Fig. 4 and 5, the model correctly predicted NO_3 dynamics in the soil profile with r^2 of 0.83 and 0.86 for the east and west sites, respectively. However, the model slightly overpredicted the peak NO_3 concentrations. Since NO_3 peaks were observed before corn planting and after a 6-mo winter season, overprediction of the peaks for both the manured and not-manured sites suggested that the RZWQM may not properly represent the temperature response of N cycling in the winter season.

Although other fates of N were not experimentally

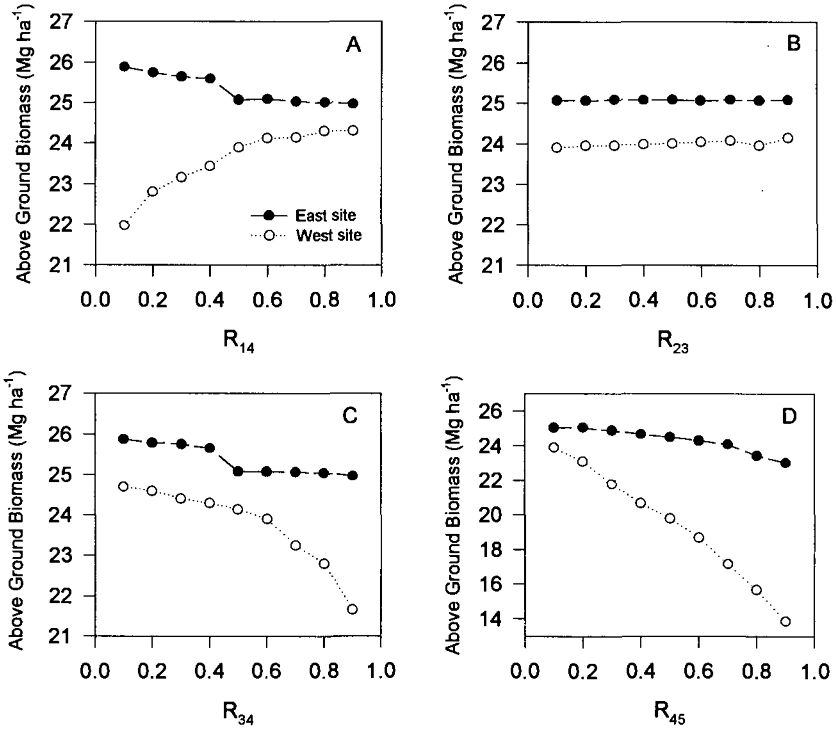

Fig. 10. Simulated corn yields in 1995 as affected by (A) R_{14}, (B) R_{23}, (C) R_{34}, and (D) R_{45}. Sensitive analysis was performed with defaulted R values R_{14} = 0.1, R_{23} = 0.1, R_{34} = 0.6, and R_{45} = 0.4. East site received manure application in Fall of 1993, 1994, and 1995; west site received manure application in Fall 1993 only.

measured, the model can be used to infer manure effects on other N sinks as well. As shown in Fig. 11, N losses due to N leaching, denitrification, and volatilization were significantly reduced in the west site because there were no manure applications in 1994 and 1995, whereas corn yield was not significantly reduced (Fig. 6). Also shown in Fig. 11 are yearly N mineralization rates. The amount of N mineralized in the site that was not manured decreased 29 and 57%, compared with the manured site in 1995 and 1996, respectively. To estimate the amount of manure N mineralized to NH_4 in the first year, the model was run for 3 yr without manure applications to eliminate manure effects on N mineralization, and then run for another 3 yr with either manure application in the first year or no manure application. The difference in N mineralization between the manure and no-manure simulations was assumed to be contributed from the manure application. Estimated manure-N mineralization rates were 22% in the first year and 18% in the second year, which are close to the ranges (20–35% in the first year and 10–15% in the second year) given by Schepers and Mosier (1991) for beef manure with 13 g kg^{-1} N content.

Manure and Water Management

One of the advantages of mathematical models is that they can be used to propose environmentally acceptable management practices to farmers without expensive long-term experimental measurements. To examine the possible environmental impacts of agricultural management, the calibrated model was run for an additional 12 yr with ten different water and manure managements (Table 2). Here, the model was allowed to continue with the current management practice or to suddenly shift to an alternative practice. Suggested alternatives were to reduce the irrigation water, to reduce the manure application rate, and to split the manure application. These management practices were evaluated based on their impact on corn yield and N losses to the environment in the next 12 yr after shifting from present practices.

As shown in Table 2, reducing irrigation water to 50% (10 cm) does not affect farmer's yields (Management 1–5 vs. 6–10), which is understandable given the simulation results that 50% of the 20 cm irrigated water was leached during the same period of time. However, only 10% of irrigated water was leached below 150 cm at the 10-cm irrigation rate. At the end of 12 yr, NO_3 storage increased five times and plant N uptake also increased 20% at the 10-cm irrigation rate. Nitrate leaching losses were reduced by 30 to 48% at the 10-cm irrigation rate. With 22.4 Mg ha^{-1} manure application each year, average silage yield was reduced by 13%, whereas denitrification loss of N was reduced by 63%, volatilization by 68%, and leaching by 46 to 58%. Therefore, high losses of N under current manure and water management were due to both elevated N concentrations in the soil profile and excess amounts of water leached. Split manure application (22.4 Mg ha^{-1} on 15 October and 1 April, respectively) did not benefit corn production and environmental quality, which may be due to low temperature and rainfall during the winter. Applying 44.8 Mg ha^{-1} of manure every other year reduced corn yield by 10% with a return of 58, 57, and

53 to 66% decreases in denitrification, volatilization, and leaching losses, respectively. The farmer could probably suffer a 22 to 24% loss in yield if he applies manure every 3 yr at 44.8 Mg ha^{-1}. The environmental gain with respect to N contamination is not favorable because of the loss in corn silage yield.

DISCUSSION

During the development of RZWQM, many processes were simplified due to a lack of understanding of the processes. Since agriculture is a complex system with great uncertainty, it is not possible to predict a specific agricultural event in the field with high accuracy as in controlled laboratory conditions. For the same reason, field measurements usually contain large experimental errors. Although each module of the RZWQM has been tested independently under certain experimental conditions, the testing by no means has been extensive. Furthermore, an individually tested module does not guarantee the overall performance of RZWQM using integrated modules. With a reasonable effort of model parameterization, the model provided good predictions of silage yield and plant N uptake for 1996, and NO_3 concentrations in the soil profile for all 3 yr. Model simulations for manure and water management can be used for making recommendations to farmers in Colorado or other locations with similar soil and environmental conditions. Therefore, the RZWQM is capable of describing an agricultural system, and the results can provide guidelines and confidence for those who desire to use the RZWQM in their specific agricultural environment (Ma et al., 1998).

One of the major difficulties with such a dynamic agricultural-system model is how to initialize the model. Since we cannot measure all the model parameters for a given study, we have to estimate many of the parameters. These estimates may not be internally consistent, and such an internal inconsistency makes sensitivity analysis of model parameters more difficult. Thus, model initialization becomes important. First, since not all the model parameters are known for a given experimental condition, running the model for a few times will internally adjust some of the parameters, such as microbial and soil OM pool sizes. Secondly, the initial condition of any experiment is the accumulative result

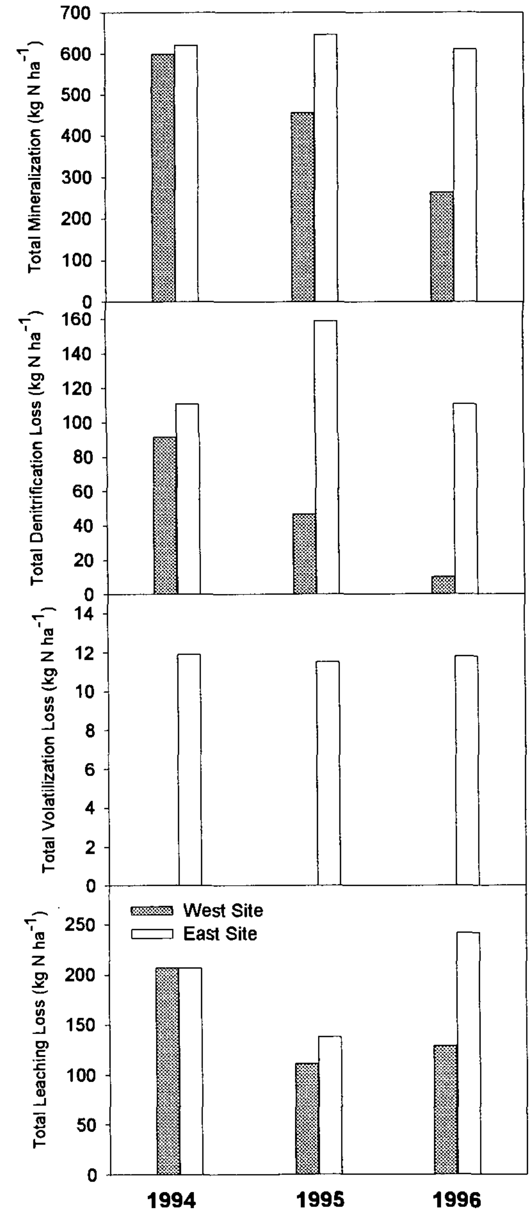

Fig. 11. Simulated N losses and gains in 1994, 1995, and 1996 from the west and east sites. East site received manure applications in Fall of 1993, 1994, and 1995; west site received manure application in Fall 1993 only.

Table 2. Average yearly summary for manure and water management based on Root Zone Water Quality Model (RZWQM). Full manure application rate was 44.8 Mg ha^{-1}. Manure was applied on October 15 for one application and October 15 and April 1 for split application (22.4 Mg ha^{-1} each). Alternated 1993–1996 management practices and weather data (Fig. 2) were used in the 12-yr simulation.

Management	Manure	Water applied	Silage yield	Nitrogen uptake	N-seepage	Denitrification	Volatilization
		cm/event	Mg ha^{-1} yr^{-1}	kg N ha^{-1} yr^{-1}			
1	Full rate every year	20	25.2	254	196	130	11.1
2	Half rate every year	20	21.9	182	106	48	3.5
3	Fall rate every two years	20	22.8	192	91	57	4.7
4	Full rate every three years	20	19.0	153	83	34	3.2
5	Full rate, split application	20	25.1	250	188	140	7.1
6	Full rate every year	10	25.8	307	116	141	11.1
7	Half rate every year	10	22.7	228	49	50	3.5
8	Full rate every two years	10	22.9	232	39	59	4.7
9	Full rate every three years	10	20.1	186	40	35	3.2
10	Full rate, split application	10	25.6	302	109	150	7.1

from previous management on that site. Thus, running the model with previous management prior to the experiment setup will create a better initial state of the system.

Many factors may contribute to the goodness of model predictions, such as model capabilities and the representability of field measurements. For example, we used an average water application rate of 20 cm for each irrigation event, which could vary from 10 to 30 cm across the field. Manure application may not be uniform either. Our model parameters (especially R values) were obtained based on corn yield and N uptake in 1994 and 1995 assuming 12 yr of continuous manure application at 44.8 Mg ha^{-1} per year before the experiment. However, calibrated model parameters may be different if more experimental data become available or different criteria are selected, such as NO_3 distribution in the soil profile rather than crop yield. In addition, since the model has many degrees of freedom, a set of site-specific measured parameters should improve model simulation results (Ahuja et al., 1995).

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