Effects of Subsurface Drainage Systems on Water and Nitrogen Footprints Simulated with RZWQM2

K. J. Craft, M. J. Helmers, R. W. Malone, C. H. Pederson, L. R. Schott

Abstract. *Developing drainage water management (DWM) systems in the Midwest to reduce nitrogen (N) transport to the northern Gulf of Mexico hypoxic zone requires understanding of the long-term performance of these systems. Few studies have evaluated long-term impacts of DWM, and the simulation of controlled drainage (CD) with the Root Zone Water Quality Model (RZWQM) is limited, while shallow drainage (SD) has not been examined. We tested RZWQM using nine years (2007-2015) of field data from southeast Iowa for CD, SD, conventional drainage (DD), and undrained (ND) systems and simulated the long-term (1971-2015) impacts. RZWQM accurately simulated N loss in subsurface drainage, and the simulations agreed with field data that CD and SD substantially reduced N loss to drainage. As indicated by the field data, the SD N concentration was predicted to be greater than DD and CD, likely due to reduced time of travel to shallower drains. The long-term simulations show that CD and SD reduced annual N lost via tile drainage by 26% and 40%, respectively. Annual reductions in N lost via tile drainage ranged from 28% in the driest years to 22% in the wettest years for CD and from 56% in the driest years to 35% in the wettest years for SD. Considering spring N loading for the purpose of addressing hypoxia in the Gulf of Mexico, CD was found to be less effective than SD, and in many years CD exported more N in the spring than DD. Spring N loading (April through June) was indicated by the EPA Science Advisory Board to have the greatest impact on hypoxia in the northern Gulf of Mexico. Therefore, improvement of CD systems within the months of April through June to reduce N loss via drainage across the upper Midwest landscape may be required. Limited research in the upper Midwest has addressed spring N loading under controlled drainage systems (CD). This research will help model developers, model users, and agricultural scientists more clearly understand N transport under different systems, including CD, SD, and ND, which will aid in developing the design and management of drainage systems to reduce N transport from tile-drained agriculture to surface waters.*

Keywords. *Agricultural simulation model, Drainage water management, Nonpoint-source pollution, Northern Gulf of Mexico hypoxic zone, Nutrient reduction, Subsurface drainage.*

Subsurface drainage systems are common throughout the Midwestern U.S. Corn Belt region and have converted naturally poorly drained soils into highly productive cropping systems that are among the most productive in the world. Of the roughly 1 billion tons of maize (*Zea mays* L.) and 300 million tons of soybean (*Glycine max* L. Merr.) produced each year, globally 36% and 33%, respectively, come from the U.S., and the majority of this production is within the central area of the Midwest Corn Belt (USDA, 2016; Goolsby et al., 1999). Estimates indicate that a quarter of all cropland in the U.S. and Canada requires enhancing the soil's poor natural drainage with an artificial drainage system in order to produce crops (Skaggs et al., 1994). Within the state of Iowa's 9.4 million ha of land area used for maize and soybean row-crop agriculture, it is estimated that 5.2 million ha (55%) are drained via artificial subsurface drainage systems (USDA, 2012, 2015). During wet growing seasons, artificial drainage removes excess soil moisture from the field to prevent water stress on growing crops and subsequent production loss. However, a negative side effect of these drainage systems throughout the Midwest is that the highly soluble nitrate-nitrogen (NO_3-N) carried within drainage water is short-circuited to surface water bodies, where it contributes to the northern Gulf of Mexico hypoxic zone (Goolsby et al., 1999).

Two alternative approaches to the conventional design and management of artificial subsurface drainage systems have been found to reduce NO_3-N loss from agricultural fields: (1) controlled drainage raises the outlet elevation of the drainage system with an outlet control structure during periods of the year when drainage is unnecessary for produc-

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tion purposes, and (2) shallow drainage raises the outlet elevation of the drainage system with tile lines installed closer to the ground surface. Three processes are responsible for nutrient reduction with these alternative drainage systems: (1) an increased anaerobic zone in the soil profile, which enhances denitrification; (2) a reduced volume of drainage water leaving the field via drains; and (3) a reduced depth of the soil profile that drainage water infiltrates through, potentially reducing its exposure to soil NO_3-N (Dinnes et al., 2002). The reduction in NO_3-N loss in controlled drainage experiments has been most commonly attributed to a reduction in drainage volume rather than a reduction in NO_3-N concentration (Evans et al., 1995; Skaggs et al., 2012). The Iowa Nutrient Reduction Strategy science assessment summarized multiple field studies and states in which controlled drainage and shallow drainage have been shown to reduce nitrate loads by 33% and 32%, respectively (Thompson et al., 2016). These reductions have been shown to depend on weather and precipitation, and nutrient reductions may be greatest in drier years and lowest in the wettest years (Evans et al., 1995).

Current field research on drainage-related nutrient reduction strategies is presently limited to a narrow range of time periods, climates, and soil types. Agricultural simulation modeling tools will be essential to advance research and scientific understanding of the effects of drainage systems by extending beyond the limits of field research. Early drainage simulation studies found comparable results to field-based monitoring, including NO_3-N loss reduction and increased denitrification with drainage systems (Skaggs and Gilliam, 1981; Wright et al., 1992). Simulation studies for both controlled drainage and shallow drainage have been successful in correctly predicting the reduction in drainage as well as excess water stress from higher water tables, which can generate yield loss in some soils and climates (Luo et al., 2010; Singh et al., 2006). The Root Zone Water Quality Model (RZWQM) (Ahuja et al., 2000a) has proven successful in simulating crop rotation and yield, daily and annual drainage volume, as well as nitrogen (N) dynamics, such as NO_3-N loss in artificial subsurface drain flow (Ma et al., 2007b; Qi et al., 2011). With RZWQM, Fang et al. (2012) successfully simulated the effect of a controlled drainage system on monthly tile drainage volume, N loss, and flow-weighted annual NO_3-N concentration (FWANC) and found that the percent reduction in N loss compared to an unmanaged system was higher with increased annual rainfall.

Simulation modeling research has demonstrated the use of long-term datasets to gain additional information from short-term, field-calibrated subsurface drainage datasets (Ma et al., 2007c; Randall and Iragavarapu, 1995). As water quality issues continue to persist in the state of Iowa and globally, it is vital for researchers to extend the knowledge gained in field studies with simulation modeling. Long-term evaluations of DWM systems are lacking in related research. Additionally, RZWQM simulations of controlled drainage systems are limited to a few studies, and none currently exist for shallow drainage systems. This study had three specific objectives: (1) calibrate RZWQM2 (ver. 3.29) for a naturally poorly drained soil in southeast Iowa with a conventional drainage system, (2) test the parameterized model for DWM of shallow and controlled drainage and for systems without artificial drainage, and (3) apply the calibrated model to a 45-year historical dataset to examine the long-term impacts of drainage systems in southeast Iowa.

Materials and Methods

Overview of RZWQM2

RZWQM2 was developed as a field-scale, one-dimensional agricultural simulation model by the USDA Agricultural Research Service (USDA-ARS) and is capable of simulating biological, chemical, and physical processes within the soil root zone. RZWQM2 has been well tested for its ability to simulate various agricultural management scenarios as well as the transport and transformation of water, nutrients, and pesticides (Ma et al., 2005, 2006, 2007b; Malone et al., 2007). For this study, DSSAT (ver. 4.0) crop growth models were used, including CERES-Maize (Ma et al., 2006) and CROPGRO-Soybean (Ma et al., 2005). Daily potential evapotranspiration (ET) is estimated with a modified Shuttleworth-Wallace equation that extends from the Penman-Monteith concept, as described by Farahani and DeCoursey (2000). The soil water balance is modeled using the Green-Ampt equation for infiltration into the soil profile, with the Richards equation for redistribution within the profile and the steady-state Hooghoudt equation for tile drainage (Ahuja et al., 2000c). The Organic Matter/Nitrogen Cycling (OMNI) model (Shaffer et al., 2000) simulates the cycling of carbon (C) and nitrogen (N) throughout three microbial pools, two surface residue pools, and three soil humus pools. Nutrient cycling is controlled by C:N ratios and rate coefficients. The microbial populations simulated in OMNI include aerobic heterotrophs, which are soil decomposers; anaerobic heterotrophs, which are denitrifying facultative bacteria; and autotrophs, which perform nitrification (Ahuja et al., 2000b). The primary processes involved with C and N cycling simulated within OMNI are sensitive to environmental conditions and include the death and growth of microbial populations, nitrification of ammonium (NH_4) to nitrate (NO_3), denitrification to produce nitrogen (N_2) or nitrous oxide (N_2O) gases, mineralization-immobilization of organic material, and production or consumption of methane (CH_4) or carbon dioxide (CO_2) gases (Kumar et al., 1998).

Site Description and Management

The experimental field site is located on the Iowa State University (ISU) Southeast Research and Demonstration Farm (SERF) near Crawfordsville, Iowa. The field slope is less than 1%, and soil types at this site include Taintor (silty clay loam, fine smectitic, mesic Vertic Argiaquolls) and Kalona (silty clay loam, fine, smectitic, mesic Vertic Endoaquolls), which are both poorly drained soil types. The subsurface drainage systems were installed in 2006 and have been monitored for crop growth and nitrogen and water dynamics from 2007 to 2015. The cropping system included an annual rotation of corn (*Zea mays* L.) and soybean (*Glycine max* [L.] Merr.), and a split-plot design warranted that each plot be cropped half in corn and half in soybeans within every year. The 17 ha field site consisted of eight experimental plots, each 1.2 to 2.4 ha in size, which included two replications of three drainage treatments and one naturally drained, or undrained, treatment (ND). The two crop rotations within each plot constituted a total of 16 subplots. Drainage flow and nutrient analysis was collected on a plot (rather than subplot) basis, meaning that drainage analysis came from half corn ground and half soybean ground each year. The three drainage treatments included conventional drainage (DD) with 1.2 m drain depth and 18 m spacing, shallow drainage (SD) with 0.76 m depth and 12.2 m spacing, and controlled drainage (CD) with 1.2 m depth and 18 m spacing with a controlled-outlet structure. All subsurface drainage systems were designed with a maximum daily drainage coefficient of 1.9 cm d^{-1}.

For the CD system, control boards or gates within the controlled-outlet structure were opened about two weeks before planting in the early spring, typically mid-April, to allow winter or early spring drainage water to drain from the field and allow field entry. The boards were then replaced to set the outlet height at 0.76 m deep after planting, typically in early June. If water was being held back by the controlled-outlet gates in the fall, the boards were removed again to permit field entry for harvest and then replaced after fall tillage, commonly in early November. For SD plots, the tile lines were installed closer to the ground surface, not as deep as the conventional drainage tiles, and were 6 m closer together to accommodate a similar drainage intensity between all plots and drainage treatments in the study. All drainage systems included a parallel tile drainage layout with curtain tiles installed between plots to eliminate influence and cross-contamination of neighboring drainage treatments. Each spring, 150 kg N ha^{-1} was injected as anhydrous ammonia to the corn half of each plot. Field cultivation was carried out every spring to both corn and soybean ground, and corn residue was chisel plowed each fall.

Data Collection

To run simulations in RZWQM2, the minimum required climate data include minimum and maximum air temperatures, wind speed, shortwave solar radiation, relative humidity, and rainfall hydrographs (Ahuja et al., 2000a). Historical weather data for SERF were collected from three different climate data networks based on the best available data over time. The meteorology dataset used data from the legacy network of the ISU Ag Climate automated weather station and the ISU Soil Moisture Network automated station, both located on site, as well as the National Weather Service (NWS) Cooperative Observer Program (COOP), located approximately 18 km away in Washington, Iowa. The ISU Ag Climate station was installed on site in 1988 and replaced with the new ISU Soil Moisture Network station in December 2013. The rainfall breakpoint or hydrograph data were acquired from manual readings taken on site for 2007 to 2013 and from the ISU Soil Moisture Network automated tipping-bucket rain gauge for 2014 and 2015. Hourly rainfall data were only available for 2014 and 2015; therefore, the breakpoint curves for 2007 to 2013 were created using typical rainfall intensities and durations commonly observed in the region. Quality control of weather data is essential in agricultural simulation modeling, as Malone et al. (2011) showed that a data bias of 10% in humidity, solar radiation, and rainfall can account for a 40% error in simulated tile drainage and NO_3-N loss in Iowa cropping system simulations. Only weather data that had been quality controlled by data specialists were used, and these data were checked against multiple nearby sources when available. Solar radiation was used for PET calculations, which is a large factor in the water and energy balances. The dataset was only considered acceptable once annual averages were within a regionally acceptable range and below a theoretical maximum for the region (Malone et al., 2011).

Soil chemical analysis was completed in 2011, 2013, and 2015 and texture analysis in 2011 by collecting push probe samples from each plot and measuring the fraction of particle sizes. For simplicity in calibration and testing, measured values were averaged across all plots in order to create a representative set of soil properties for the field site. This was justified due to the limited variation in soil properties among the eight plots, including soil texture, bulk density, and total carbon (Schott et al., 2015). Hydraulic properties of bulk density and soil water content at 10, 33, and 1500 kPa matric potentials were measured by collecting undisturbed soil cores in 2011, 2013, and 2015 (Schott et al., 2015). The saturated hydraulic conductivity (K_{sat}) was estimated using the USDA-ARS Rosetta model (ver. 1.0) pedotransfer function (PTF). The estimates for K_{sat} were applied cautiously because much faster infiltration rates have been observed at the site. Calibration of K_{sat} values within the upper and bottom layers is discussed in the Model Calibration and Evaluation section. Soil properties were measured in the field to a depth of 60 cm for bulk density and soil texture properties and to a depth of 20 cm for soil water retention properties. Below these depths, values were either repeated or estimated by adjusting from default values for a silty clay loam soil within the literature review of soil properties completed by Rawls et al. (1982). Measured and estimated soil properties are provided in table 1.

Drainage from the interior tiles of each plot was continuously monitored for flow rate during the drainage season. Most years, drainage monitoring began near spring thaw (March or early April) and usually continued through December. Each of the six monitored tile lines drained a single plot, with half of the plot cropped to corn and half to soybeans each year. Grab samples were taken weekly for NO_3-N analysis, as described by Helmers et al. (2012), and NO_3-N concentrations were linearly interpolated between grab samples. The NO_3-N concentration and representative tile drainage volume were multiplied to calculate NO_3-N load from each plot. FWANC was quantified using the annual NO_3-N loss from each plot, normalized to the annual drainage volume. Also described by Helmers et al. (2012), the depth to the water table was continuously monitored with pressure transducers in observation wells installed directly in the center of the plots. Sub-hourly water table data were averaged to determine the daily depth from the ground surface. Average daily soil water storage was derived from continuously monitored volumetric water content within each plot using Decagon 5TM sensors (Decagon Devices, Inc., Pullman, Wash.) (Schott et al., 2015).

Grain yield was recorded for each year of the nine-year

Table 1. Measured and estimated soil hydraulic properties.[a]

Depth (cm)	Silt (cm^3 cm^{-3})	Clay (cm^3 cm^{-3})	SOC (%)	BD (g cm^{-3})	Porosity (cm^3 cm^{-3})	P_b (cm)	K_{sat} (cm h^{-1})	LK_{sat} (cm h^{-1})	θ_{res} (cm^3 cm^{-3})	θ_{sat} (cm^3 cm^{-3})	θ_{33} (cm^3 cm^{-3})	θ_{1500} (cm^3 cm^{-3})
0-5	0.48	0.39	2.95	1.1	0.585	-1	5	10	0.025	0.573	0.253	0.153
5-20	0.49	0.38	2.81	1.348	0.491	-32	1.5	3	0.04	0.467	0.340	0.209
20-40	0.46	0.4	1.99	1.344	0.493	-32	0.5	1	0.04	0.463	0.331	0.198
40-60	0.47	0.39	0.93	1.383	0.478	-32	0.7	1.4	0.04	0.430	0.308	0.186
60-90	0.47	0.39	0.23	1.383	0.478	-32	2	4	0.04	0.430	0.308	0.186
90-100	0.47	0.39	0.23	1.383	0.478	-42	5	10	0.04	0.430	0.320	0.192
100-140	0.47	0.39	0.23	1.383	0.478	-42	2	4	0.04	0.430	0.320	0.192
140-180	0.47	0.39	0.12	1.45	0.453	-42	1.5	3	0.05	0.408	0.307	0.190
180-260	0.47	0.39	0.01	1.6	0.396	-42	0.01	2	0.07	0.357	0.276	0.182

[a] SOC = soil organic carbon, BD = bulk density, P_b = bubbling pressure, K_{sat} = saturated hydraulic conductivity, LK_{sat} = lateral hydraulic conductivity, θ_{res} = residual water content, θ_{sat} = saturated water content, θ_{33} = water content at 33 kPa, and θ_{1500} = water content at 1500 kPa.

field study with a combine yield monitor (HarvestMaster, Logan, Utah). The yield measurement was limited to the center rows to minimize effects from neighboring plots (Helmers et al., 2012). From 2011 to 2015, corn and soybean biomass was collected prior to harvest to obtain measurements of total aboveground biomass production and N content (Schott et al., 2015). N content was measured for crop grain, which was conducted by the ISU Soil and Plant Analysis Laboratory (Ames, Iowa).

Model Initialization

RZWQM2 was initialized with a 26-year dataset of historical weather data (1981-2006). During initialization, the cropping and tillage system in place during the study period was executed in order to set up the microbial and residue pools and to establish the hydraulic and nutrient cycles within the soil profile. Measured soil carbon content was allocated between the fast, medium, and slow organic matter pools and initialized as 5%, 35%, and 60%, respectively, based on similar methods described by Kumar et al. (1999) and Hanson et al. (1999). Percent SOM was measured at depth increments of 0-10 cm, 10-20 cm, 20-40 cm, and 40-60 cm in 2011, 2013, and 2015 at the field site. Due to little variation between years, an average SOM value by depth increment was used. A carbon-to-nitrogen (C:N) ratio of 60 and a conversion factor for soil organic matter (SOM) to soil organic carbon (SOC) of 0.58 was used based on typical estimates for corn-soybean system residue (Christianson et al., 2012). Default C:N ratios were used for microbial and humus pools.

Based on modeling methods demonstrated by Landa et al. (1999) and Hanson et al. (1999), multiple iterations of 26 years of typical management and climate data were continuously run until the soil humus and biota pools reached a steady state and the sum of the humus carbon pools were close to measured values, or around 2% to 3% (table 1). Minor iterative adjustments to microbial death rates, humus decay rates, and organic matter inter-pool transfer coefficients were required until the slow humus pool size was stable and the microbial and organic matter pool sizes reached equilibrium (Hanson et al., 1999; Kumar et al., 1999). These rate coefficients were also used to simulate an acceptable rate of annual soil N mineralization for the general region, which was presumed to be within the range of 112 to 504 kg N ha^{-1} (Helmers and Castellano, 2015). Qi et al. (2011) simulated an average annual N mineralization rate of 140 kg N ha^{-1} for an artificially drained corn-soybean system with silty clay loam and clay loam soils in north-central Iowa. Ma et al. (2007c) simulated an annual average of 109 kg N ha^{-1} with a 26-year range between 67 to 223 kg N ha^{-1} for a similar system in northeast Iowa. The inter-pool transfer coefficients were slightly adjusted based on those calibrated by Ma et al. (2007a) and Thorp et al. (2007), including adjusting the slow residue to intermediate SOM to 0.2, the fast residue to fast SOM to 0.5, the fast SOM to intermediate SOM to 0.5, and the intermediate SOM to slow SOM to 0.7. The adjustment to the intermediate SOM to slow SOM transfer coefficient aided in simulating N mineralization rates (Hanson et al., 1998).

The denitrification rate coefficient was adjusted to 1.8×10^{-13} to fit to acceptable annual denitrification rates. Thorp et al. (2007) modeled an annual denitrification rate of 6.8 ±5.2 kg N ha^{-1} for a tile-drained corn-soybean system simulated with RZWQM, and other modeled estimates for the Midwest have been found to be between 6 and 30 kg N ha^{-1} (Christianson et al., 2012). The background chemistry of rainwater was set using National Atmospheric Deposition Program estimates, including pH of 5.1, 0.5 mg N L^{-1} for NH_4, and 1.3 mg N L^{-1} for NO_3-N (NADP, 2015). Initial chemical status of the soil profile was set with three-year averages of measured pH and CEC values, which were obtained from chemical analysis performed by the ISU Soil and Plant Analysis Laboratory (Ames, Iowa) in 2011, 2013, and 2015. Initial moisture conditions were entered based on field capacity and saturated water contents to initialize a water table within the soil profile.

Model Calibration and Evaluation

Manual parameterization was iteratively carried out using methods described by Ma et al. (2003) with observed data for the DD system from 2007 to 2015. Two replicate scenarios were executed to model the split-plot cropping system, with each replication having one of two cropping rotations: corn following soybeans or soybeans following corn. Numerical results from the two scenarios were averaged for hydrology information in order to compare simulation results with field-collected data, as each field plot consisted of an equal area planted in corn and soybeans within each year. Simulated crop production estimates for both corn and soybeans were calibrated within each year. Field-collected data included crop production, nutrient, and hydrology measurements. Crop production measurements included corn and soybean yield, biomass production, harvest index, and grain N uptake. Nutrient measurements involved NO_3-N load and FWANC as well as fall soil nitrate. Lastly, hydrology measurements included tile drainage volume at the annual, monthly, and daily levels, soil water content to a depth of 60 cm, and depth to water table.

The quality of simulation was determined by evaluating discrepancies between simulated and measured or observed data and was based on quantitative and qualitative measures of goodness-of-fit, similar to methods described by Bakhsh et al. (2001). Crop production simulations were considered satisfactory when percent error (PE) was within ±15%, based on methods described by Ahuja et al. (2000b), and the relative root mean square error (n-RMSE) was <30%, per Liu et al. (2011). Hydrology and NO_3-N load simulations were considered satisfactory when the Nash-Sutcliffe efficiency (NSE) was >0.50, percent bias (PBIAS) was within ±25%, and the root mean square error normalized to the standard deviation of the observed dataset (RSR) was ≤0.70 (Moriasi et al., 2007). The coefficient of determination (R^2) was used to quantify the proportion of variance in the measured data explained by the model, which was deemed acceptable when >0.50 (Moriasi et al., 2007). Qualitative goodness-of-fit tests included review of graphical representations of model simulation differences from observed measurements.

Both corn and soybean yields were calibrated for each year from 2007 to 2015 for the DD system. Previously parameterized crop models for maize (IB1 068 Dekalb 521) and soybean (990002 M Group 2) were used, and some minor parameter adjustments were carried out following methods described by Ma et al. (2006) and Thorp et al. (2007). Parameters were initialized with values used for simulations in north-central Iowa by Qi et al. (2011) and adjusted to match observed annual grain yields (2007-2015) as well as aboveground biomass and harvest index (2011-2015). To simulate nine years of corn yield measurements for SERF, it was necessary to simulate two maize hybrids with minor differences in maturity length because the relative hybrid maturity ratings of corn hybrids planted at SERF ranged from 106 to 113. Minor adjustments were required for P1, P5, G2, G3, and PHINT for the corn model. The only adjusted parameter for the soybean model was LFMAX, which was calibrated to 0.725 to fit soybean yield and harvest index. The simulation parameters of the two maize models and the soybean model are provided in table 2. The soil root growth factor (SRGF) was adjusted for the maize and soybean models in order to fit the N concentration in grain to a reasonable level and to measured values for 2011-2015 for corn and 2012-2015 for soybean. For depths of 5, 15, 30, 40, 60, 90, and 120 cm, the SRFG was set to 1, 1, 0.32, 0.2, 0.12, 0.12, and 0.0 for soybeans and to 1, 1, 0.8, 0.4, 0.27, 0.1, and 0.06 for corn, respectively, by making minor adjustments from settings in past corn and soybean system modeling work by Qi et al. (2011), Malone et al. (2010), and Ma et al. (2006). These SRGF parameter settings simulated average rooting depths of around 90 cm for soybeans and 115 cm for corn.

Soil hydraulic information was parameterized for annual, monthly, and daily tile drainage as well as daily depth to water table and soil water content. To simulate a water table within the soil profile, a constant flux boundary condition was chosen for the redistribution model. An impermeable layer was set at the bottom of the soil profile with a K_{sat} value of 0.01 cm h^{-1}, similar to methods described by Ma et al. (2007a), and a water table leakage rate of 1E-6 cm h^{-1} was used. Similar to work by Singh et al. (1996), the impermeable layer was defined at 2.6 m below the ground surface based on field observations and the USDA Web Soil Survey (USDA, 2013) estimates of depth to the restrictive layer. The lateral hydraulic conductivity (LK_{sat}), drainable porosity ($\theta_{sat} - \theta_{1/3}$), and lateral hydraulic gradient (LHG) have been found to be highly sensitive parameters for simulation of tile flow in RZWQM (Ma et al., 2007a; Singh and Kanwar, 1995), which was also observed for calibration of SERF. Additionally, LK_{sat} has been found to be most important for simulating tile flow and NO_3-N loss at the soil layer containing the tile drain, followed by the layer directly above (Ma et al., 2007a). For this reason, it was necessary to calibrate LK_{sat} values for simulation of tile drainage in layers 6 and 7, as the conventional tile depth is within layer 7. Within layer 7, LK_{sat} was calibrated to match a peak daily drainage rate of 1.9 cm d^{-1}. Setting LK_{sat} for layers 4 and 5 was done similarly to aid in simulation of drainage for the alternative drainage systems during the testing period.

The lateral hydraulic gradient (LHG) was parameterized to a value of 1.2E-5 to fit annual observed drainage volumes for the conventional system. To facilitate soil water flow to the tile drains and maintain a lateral loss pathway, LK_{sat} was set to twice the value of K_{sat} within each layer. Maintaining high LK_{sat} values throughout the soil profile was done to promote tile drainage as well as additional lateral losses in the system other than subsurface drainage, as we hypothesized subsurface lateral loss to be the second largest subsurface loss pathway after tile drainage. Based on methods described by Bakhsh et al. (2004), the drainable porosity of the upper 5 cm profile was calibrated to limit evaporation from the soil surface. Additionally, setting a sufficient drainable porosity and high K_{sat} value in the upper layer minimized runoff by increasing infiltration into the soil profile. To correctly simulate evapotranspiration (ET), albedo coefficients were adjusted based on simulations by Qi et al. (2011) and Thorp et al. (2007) to fit ET within acceptable ranges for the region, including the albedo of dry soil, wet soil, mature crop, and

Table 2. Calibrated DSSAT crop parameters for corn and soybean.

Crop[a]	Parameter	Description	Corn 1[b]	Corn 2[b]
Corn	P1	Thermal time from seedling emergence to the end of the juvenile phase (°C above 8°C base temp.)	225	225
	P2	Delay in development (days h^{-1}) for each hour that day length is greater than 12.5 h (0 to 1)	0.4	0.4
	P5	Thermal time from silking to physiological maturity (°C days above 8°C base temperature)	750	795
	G2	Maximum possible number of kernels per plant	810	810
	G3	Kernel filling rate during the linear grain filling stage and under optimum conditions (mg d^{-1})	7	8
	PHINT	Phylochron interval in thermal time (°C days) between successive leaf tip appearances	52	52
Soybean	LFMAX	Maximum leaf photosynthesis rate at 30°C, 350 vpm CO_2, and high light (mg CO_2 m^{-2} s^{-1})	0.725	

[a] Maize cultivar IB1 068 Dekalb 521 and soybean cultivar 990002 M Group 2.

[b] Corn 1 is the calibrated parameter set for simulation of a shorter maturity group, and Corn 2 is for simulation of a longer maturity group.

Table 3. Management dates used for 2007 to 2015 simulations and for long-term simulation (1960-2006), including management of the outlet control structure for the controlled drainage system (CD).

	Corn		Soybean		CD - Spring Control		CD - Fall Control
Year	Planting	Harvest	Planting	Harvest	Open (1.2 m)	Close (0.76 m)	Close (0.3 m)
2007	May 5	Nov. 5	June 2	Oct. 11	Apr. 30	June 2	Jan. 7, 2008
2008	May 9	Nov. 5	June 6	Oct. 11	Apr. 14	June 5	Nov. 19
2009	Apr. 17	Oct. 13	May 31	Oct. 20	Apr. 15	May 29	Nov. 5
2010	Apr. 15	Sep. 30	May 28	Oct. 2	Apr. 15	June 24	Oct. 18
2011	May 3	Sep. 29	May 11	Oct. 3	Apr. 25	June 1	-
2012	Apr. 18	Sep. 24	May 15	Oct. 24	Apr. 5	June 14	-
2013	May 17	Oct. 04	June 12	Oct. 2	-	-	-
2014	May 6	Nov. 7	May 9	Oct. 10	-	-	-
2015	Apr. 30	Sep. 15	May 2	Oct. 7	Mar. 31	May 22	-
1960-2006	May 1	Oct. 19	May 20	Oct. 8	Apr. 15	June 14	-

fresh residue, which were set to 0.2, 0.1, 0.25, and 0.8, respectively.

Using the parameterized RZWQM2 model, we tested for shallow and controlled drainage systems and an undrained system. To simulate SD, the depth was decreased from 120 to 76 cm and the tile spacing was decreased from 18 to 12.2 m to maintain the same drainage intensity, as was done in the field experiment. For the CD simulation, the timing and depth of outlet structure management from 2007 to 2015 were used, which can be seen in table 3. Generally, the boards in the control structure were removed to set an outlet depth of 120 cm in the early spring before planting and then replaced shortly after planting in the later spring to set an outlet depth of 76 cm. In 2007 to 2010, management was also carried out in the fall to raise the outlet to a depth of 30 cm; however, there was rarely enough rainfall during the fall and winter to be held back with the outlet structure during this time, and therefore fall management was not continued for 2011 to 2015. The goodness-of-fit parameters used in calibration were also used in model testing.

Table 4. Goodness-of-fit statistics for crop growth for calibration with conventional drainage (DD) and testing with controlled drainage (CD), shallow drainage (SD), and undrained (ND) systems from 2007 to 2015 for corn (C) and soybean (SB).[a]

	Statistic	DD	CD	SD	ND
Corn yield	PE	-3%	-2%	-2%	3%
	n-RMSE	13%	15%	13%	10%
Corn grain N	PE	10%	11%	12%	*25%*
	n-RMSE	19%	24%	19%	28%
Corn AGB	PE	-6%	-7%	-6%	-2%
	n-RMSE	20%	23%	20%	24%
Corn HI	PE	4%	5%	4%	5%
	n-RMSE	13%	12%	11%	18%
Soybean yield	PE	-1%	1%	2%	12%
	n-RMSE	14%	13%	14%	16%
Soybean grain N	PE	5%	6%	3%	5%
	n-RMSE	14%	13%	10%	17%
Soybean AGB	PE	-1%	-10%	-1%	1%
	n-RMSE	19%	23%	15%	11%
Soybean HI	PE	-2%	8%	-3%	8%
	n-RMSE	15%	19%	13%	22%

[a] PE = percent error, n-RMSE = relative root mean square error, AGB = aboveground biomass, and HI = harvest index. Corn AGB and HI measurements were only available for 2012-2015, and soybean AGB and HI measurements were only available for 2013-2015. Unsatisfactory goodness-of-fit statistics are italicized.

MODEL APPLICATION

To carry out long-term simulations, 45 years of historical climate and management data (1971 to 2015) were used after the model had been initialized for organic matter and microbial pools. Additionally, eleven years of weather and management data were added from 1960 to 1970 to initialize the processes within RZWQM prior to the long-term summary period. To simulate long-term effects of CD, the average dates of controlled-outlet structure management were taken from the nine-year study at SERF; this included opening the control boards to a depth of 120 cm on April 15 and replacing the boards to a depth of 76 cm on June 14 (table 3). With this analysis, we wanted to observe the impacts of artificial drainage systems and of a poorly drained system without artificial drainage.

RESULTS AND DISCUSSION

MODEL CALIBRATION AND EVALUATION

Crop Growth and Yield

Corn and soybean production was well simulated for the nine-year dataset, as shown by the goodness-of-fit statistics in table 4. Corn grain yield was satisfactorily simulated for calibration with DD as well as the three tested systems, which all have PE and n-RMSE values within error limits of 15% and 30%, respectively. All nine years of DD corn yield are within 20% PE, and seven of the nine years are within 16% PE.

Corn grain yield simulations correctly predicted a yield loss in each of the three tested systems compared to the conventional system (DD), which was observed in some years with the nine-year field study. Helmers et al. (2012) reported statistically significant corn yield losses with CD (6%) and ND (7%) from 2007 to 2010, and Schott et al. (2017) reported a statistically significant 6% yield loss with ND from 2011 to 2015. Over the eight-year period from 2007 to 2014, statistically significant corn yield losses were reported as 4% with CD, 3% with SD, and 6% with ND. Schott et al. (2015) and Helmers et al. (2012) hypothesized that the reductions in corn yield were due to higher water tables in CD, SD, and ND that reached within the root zone of the growing crops, generating excess water stress in high rainfall years. Additionally, it was reported that the yield loss with CD may have been mitigated with additional management of the controlled-outlet elevation during the growing season (Helmers et al., 2012; Schott et al., 2015). The measured corn yield losses from CD, SD, and ND over the nine-year study were 5%, 4%, and 8%, respectively, while the related RZWQM2-predicted impacts were 3% for all three systems. In the wettest year (2009), the measured yield losses were 12% for CD, 6% for SD, and 20% for ND and were statistically significant for CD and ND (Helmers et al., 2012). The simulated yield losses in 2009 were reasonable at 17% for CD, 13% for CD, and 21% for ND. The PE of the nine-year average corn simulations were -2.0%, -1.8%, and 3.0% for CD, SD, and ND, respectively. These simulations of water excess yield reductions are reasonable compared to the field study as well as with other simulation work, such as Singh et al. (2006, 2007) and Luo et al. (2009) with DRAINMOD, where it was also found that excess water stress hindered corn yield in a CD system with higher water tables.

Soybean yield simulations for calibration and testing were acceptable; however, it was noted that unlike the DSSAT-Maize model, the DSSAT-Soybean model may not simulate excess water stress sufficiently (Malone et al., 2010). This may have caused some of the model overprediction in yield in high precipitation years, such as 2010 when wetness reduced soybean yield. Soybean yield reductions in the three alternative systems were measured over the nine-year field study and were 2%, 3%, and 12% for CD, SD, and ND, respectively. RZWQM2 was unable to simulate the observed soybean yield loss from excess water stress experienced within the CD, SD, and ND systems, as no yield differences were found among the four systems. Although deficit water stress seems to have been well developed with the CROPGO-Soybean model (Nielsen et al., 2002), little research has been done regarding this model in high soil water conditions.

Measured data for aboveground biomass (AGB) and harvest index (HI) were only available from 2012 to 2015 for corn and from 2013 to 2015 for soybeans. The available AGB and HI data were well simulated for both corn and soybeans for the calibration as well as the three tested systems. Corn grain N uptake was overestimated, as the percent nitrogen of the corn grain in the calibration dataset was overestimated at 1.4%, on average, for 2012 to 2015, while the corresponding measured value was only 1.2%. However, regional variation in corn N content has been cited to be within 1.2% to 1.6% N (Christianson et al., 2012). Similar overestimation of corn grain N content occurred in the three tested systems. The calculated PE and n-RMSE indicate a satisfactory simulation, with a slight overprediction of N uptake in corn grain for DD, CD, and SD. Predictions of corn grain N in the ND system were not satisfactory and also indicated overestimation error. Similar issues with RZWQM2 overprediction of corn grain N uptake have been found in multiple simulations studies, as described by Thorp et al. (2007). The N content of soybean grain (6.1%) was well simulated compared to measured N content (5.9%). These simulations also agree with regional estimates, which are within 6% to 6.5% N (Christianson et al., 2012).

Hydrology

The average annual components of the hydrologic balance are provided in table 5 for calibration with DD and testing with CD, SD, and ND. In table 5, the percent of precipitation is given for each water balance component to indicate its relative magnitude as an input/output pathway to and from the system. Additionally, the average annual simulated change in soil profile water storage is provided as ΔS, which equals precipitation minus lateral seepage, runoff, ET, and tile drainage (TD).

Precipitation was on average 105 cm for the nine-year study. The years 2007 to 2010 as well as 2015 experienced average to above-average rainfall, while 2011 to 2014 experienced average to below-average rainfall. Over the nine-year study, 65% of annual rainfall occurred in March through August, with 28% in March through May, and 36% in June through August. The greatest loss from the system was ET. The annual nine-year average ET was simulated as 51 cm year^{-1}, which is close to measured estimates by Bakhsh et al. (2004) for a tile-drained corn-soybean system in central Iowa, which included a range of 33.4 to 49.3 cm year^{-1}. This simulation was also near the model estimates reported by Thorp et al. (2007), with an average ET of 46.8 cm year^{-1}. A modeled water-balance method was used by Sanford and Selnick (2013) to estimate ET across the conterminous U.S. and reported a range of 51 to 70 cm year^{-1}, or 60% to 69% of annual rainfall for southeast Iowa. Simulated estimates of ET accounted for 48% of precipitation, on average, which is less than the regional water-balance estimates of percent of precipitation.

Annual drainage was underpredicted for calibration data by a total of 35 cm over nine years. Simulated cumulative drainage was 248 cm, which accounted for 26% of precipitation and ranged from 9% to 35% of annual precipitation. The calibration with the DD annual drainage dataset was well simulated as a post-manual calibration; however, only eight years of tile drainage measurements were used. Data from 2014 were not used because of sensor measurement error, which caused the measured drainage to be inexplicably high. With the exclusion of 2014, the goodness-of-fit statistics for annual and monthly drainage volume are satisfactory for the calibration dataset (table 6). Including all nine years, daily measurements of DD drainage compared well to simu-

Table 5. Simulated average annual hydrologic components from 2007 to 2015 for conventional drainage (DD), controlled drainage (CD), shallow drainage (SD), and undrained (ND) along with observed tile drainage for comparison and the percent of precipitation for each component.

	DD		CD		SD		ND	
	cm	%	cm	%	cm	%	cm	%
Precipitation	105	-	105	-	105	-	105	-
Actual ET[a]	50	48%	48	46%	50	48%	53	51%
Runoff	4	4%	10	10%	9	9%	18	18%
Lateral seepage	22	21%	26	25%	30	29%	35	33%
Tile drainage	28	26%	20	19%	15	15%	-	-
Tile drainage observed[b]	32	31%	16	15%	15	14%	-	-
ΔS[c]	0.9	-	0.3	-	0.0	-	-1.1	-

[a] ET = evapotranspiration.
[b] Measured value for tile drainage.
[c] Average annual change in soil water profile storage.

Table 6. Goodness-of-fit statistics for annual and monthly tile drainage, nitrate loss, and FWANC for conventional drainage (DD), controlled drainage (CD), and shallow drainage (SD).[a]

		DD	CD	SD
Tile drainage, annual (cm)	PBIAS	3.3%	-17.1%	3.8%
	NSE	0.93	0.62	0.61
	RSR	0.27	0.61	0.63
	R^2	0.88	0.68	0.57
Tile drainage, monthly (cm)	PBIAS	16.5%	-19.1%	2.1%
	NSE	0.78	0.66	0.62
	RSR	0.53	0.58	0.61
	R^2	0.74	0.78	0.67
N loss, annual (kg N ha^{-1})	PBIAS	9.4%	-5.9%	8.4%
	NSE	0.73	0.72	0.54
	RSR	0.52	0.53	0.68
	R^2	0.53	0.55	*0.31*
N loss, monthly (kg N ha^{-1})	PBIAS	14.3%	-9.3%	-2.9%
	NSE	0.63	0.71	*0.31*
	RSR	0.68	0.54	*0.83*
	R^2	0.60	0.78	0.55
FWANC (mg L^{-1})	PBIAS	6.8%	10.8%	-6.9%
	NSE	*-0.46*	*-0.06*	*0.12*
	RSR	*1.21*	*1.03*	*0.94*
	R^2	*0.01*	*0.13*	*0.00*

[a] Of the nine years of data (2007 to 2015), the annual datasets for DD, CD, and SD exclude the years 2014, 2015, and 2011, respectively, while monthly datasets include all nine years. PBIAS = percent bias, NSE = Nash-Sutcliffe efficiency, RSR = root mean square error normalized to the standard deviation of the observed dataset, R^2 = coefficient of determination, and FWANC = flow-weighted annual NO_3-N concentration. Unsatisfactory goodness-of-fit statistics are italicized.

lated values with PBIAS, NSE, RSR, and R^2 values of 14.0%, 0.69, 0.56, and 0.70, respectively. Table 6 provides goodness-of-fit statistics for annual and monthly tile drainage and NO_3-N loss for DD and the tested drainage systems (CD and SD).

For the CD annual drainage dataset, 2015 was not included in the calculations for goodness-of-fit statistics for CD annual tile drainage due to an unusually low drainage from the hypothesized sensor error. For the SD annual dataset, the year 2011 was not used, as there were some missing drainage data in the SD system that made the 2011 data less reliable. With the adjusted eight-year datasets, both CD and SD were satisfactorily simulated for annual tile drainage; additionally, tile drainage was well simulated at the monthly level for CD and SD for all nine years of data (table 6).

For water mass balance purposes, all nine years were used to compare all four simulated systems. Cumulative tile drainage was overpredicted over the nine years in simulations by about 40 cm for CD and 7 cm for SD. For CD, average annual tile drainage volume was simulated at 20 cm year^{-1}, which is 4 cm year^{-1} higher than the observed average annual drainage of 16 cm year^{-1}, and most of the error comes from 2015. For SD, average annual tile drainage volume was simulated and observed at 15 cm year^{-1}. Similar to RZWQM2 simulations reported by Fang et al. (2012), most of the error in CD simulation of tile drainage and N loss occurred within months in which control boards were removed to allow free drainage in the spring. This may indicate that the model routines for simulation of drainage volume in a controlled-outlet drainage system should be examined. Of the 4 cm year^{-1} of drainage that was overpredicted for CD, nearly 80% of this overprediction occurred within the months of April, May, and June. Although this overpredicted spring volume is relatively small (3.2 cm year^{-1}), it is likely that winter and spring drainage water within the CD system is subject to losses that are not well predicted. Losses such as surface runoff, ET, and lateral flow have been found to cause issues with drainage volume simulation in CD systems (Singh et al., 2007; Woli et al., 2010). To improve the modeling of CD systems, attempts at in-field quantification of these spring losses in controlled-outlet drainage systems are needed.

Over the nine years, simulations of DD lost 26% of precipitation via tile drainage, while the losses for CD and SD were less at 20% and 15%, respectively. Figure 1 presents the monthly observed and simulated tile drainage for each drainage system as well as monthly precipitation. As shown by the PBIAS values (table 6), annual drainage volume was underpredicted for DD and SD and overpredicted for CD, which caused discrepancies in the percent reduction estimates between the observed and simulated datasets. For the nine-year dataset, the CD system's average annual drainage reduction was simulated as 26%, which is lower than the observed reduction of 47%. That of the SD system was simulated as 46%, or lower than the observed reduction of 50%. The simulated annual percent reduction in tile drainage, compared to DD, was higher in the below-average and average rainfall years of 2011 to 2014, at 30% for CD and 50% for SD. This tile drainage reduction was lower in the above-average rainfall years of 2007 to 2010 and 2015, at 23% for CD and 44% for SD. These simulations of annual drainage volume for DD, CD, and SD were similar to findings in the measured dataset (Schott et al., 2017) and are reasonable compared to DRAINMOD simulations by Youssef et al. (2006) with NSE values greater than 0.60 and PBIAS within 17% (table 6). The simulations of monthly tile drainage compared well with Singh et al. (2006), with overall NSE values between 0.56 and 0.89 and coefficient of mass residual values between 0.04 to -0.18 for calibration and validation. Improved estimates of ET may have aided in the simulation of drainage volume, as minor changes in ET can greatly impact drainage volume (Thorp et al., 2009).

In addition to drainage volume, this dataset also allows model assessment of simulation of depth to water table and 60 cm soil water storage. Using daily average measurements of the depth to water table from April through October for 2008 to 2015, the calibration statistics are satisfactory except for R^2 (table 7). Depth to water table for calibration is presented in figure 2 and shows that many of the simulated daily peaks and falls track with the observed data. Note that data for some years within dry periods of the year, usually after July in 2011 through 2014, were not available due to the observation well drying out, and no data were collected by the pressure transducers.

Daily depth to water table was also well simulated for CD and SD (table 7) but was not well simulated for the ND system. For CD and SD, the RZWQM error is generally due to simulations of depth to water table being too shallow in the spring and too deep in the summer. This likely relates to the overestimation bias in tile drainage for CD and SD, as most drainage occurs in the spring. The simulations for ND water table had much more variability in daily estimates than what

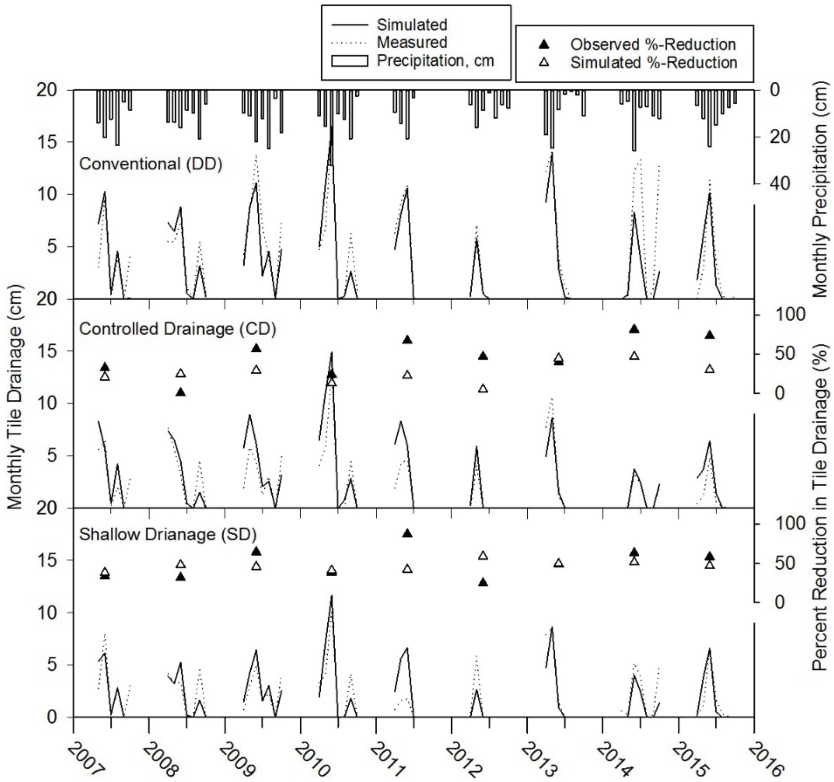

Figure 1. Monthly simulated and measured tile drainage for conventional drainage (DD), controlled drainage (CD), and shallow drainage (SD) for April through October 2007-2015, as well as monthly precipitation (bar graph at top) and percent reductions in tile drainage in CD and SD.

Table 7. Goodness-of-fit statistics for daily average depth to water table and daily average soil water storage in the 60 cm soil profile.[a]

		DD	CD	SD	ND
Depth to water table	PBIAS	-2.2%	2.1%	-9.0%	*-34.2%*
	NSE	0.82	0.80	0.72	*0.27*
	RSR	0.43	0.45	0.53	*0.86*
	R^2	*0.39*	0.56	0.57	0.47
Soil water storage	PBIAS	-9.8%	-12.6%	-17.2%	-14.4%
	NSE	*0.26*	*-0.64*	*-1.08*	*-0.03*
	RSR	*0.86*	*1.28*	*1.44*	*1.01*
	R^2	0.52	0.53	0.69	0.53

[a] Depth to water table data includes 2008 to 2015, while soil water storage data include 2011 to 2015. Unsatisfactory goodness-of-fit statistics are italicized.

was found in the measured data, causing high simulation error. In general, spring ND water table was fairly well simulated; however, summer drawdown was overestimated by the model. The simulation of summer water table depth may have been improved with better estimates of crop ET (Johnsen et al., 1995; Malone et al., 2004). Although there are some concerns with this simulation of daily fluctuation in water table, overall these simulations are encouraging, as few studies have tested RZWQM2 at a highly frequent daily measurement scale. With NSE values greater than 0.7, the simulations of daily depth to water table for DD, CD, and SD are comparable with DRAINMOD simulations by Youssef et al. (2006), which were considered very good due to the quick fluctuations that occurred.

Estimates of 60 cm soil water storage from field-collected volumetric water content at 10, 20, 40, and 60 cm compared well with simulated soil water storage, although simulations often overpredicted soil water storage. For the five years of calibration data, the PBIAS and R^2 values are satisfactory (table 7); however, NSE and RSR are unsatisfactory. These RZWQM2 simulations of soil water storage are similar to simulations presented by Qi et al. (2011) and Ma et al. (2003). Soil water storage was simulated fairly well for CD, SD, and ND, with acceptable R^2 (fig. 3) and PBIAS values (table 7). Similar to the water table, the daily soil water storage fluctuations track well between observed and simulated soil water storage estimates.

A generally large proportion of the simulated nine-year calibration water balance (21% of precipitation, or 201 cm) was lost via the lateral subsurface pathway. For CD, 25% of precipitation was lost laterally, similar to 29% lost for SD and 33% for ND. Compared to the DD system, both alternative drainage systems increased the proportion lost via the subsurface lateral pathway. The simulated percentages lost laterally in DD, CD, and SD were similar to or greater than their simulated percentages lost via tile drainage. Similar results have been found in simulation studies of controlled-outlet drainage systems (Ma et al., 2007c; Wahba et al., 2001). Simulated runoff in calibration was nominal, accounting for only 4% of precipitation, or 37 cm over nine years, which was expected in the well-drained plots at SERF due to the minimal ground slope. Due to the higher water table simulated with the CD, SD, and ND systems, the model simulated increased runoff compared to the little runoff simulated with conventional drainage (table 5), which was also reported by Ma et al. (2007b). Fang et al. (2012) and Ma et al. (2007b) also reported an increase in lateral seepage and runoff from a CD system.

Nitrogen Dynamics

Table 8 gives the simulated N balances for the nine-year

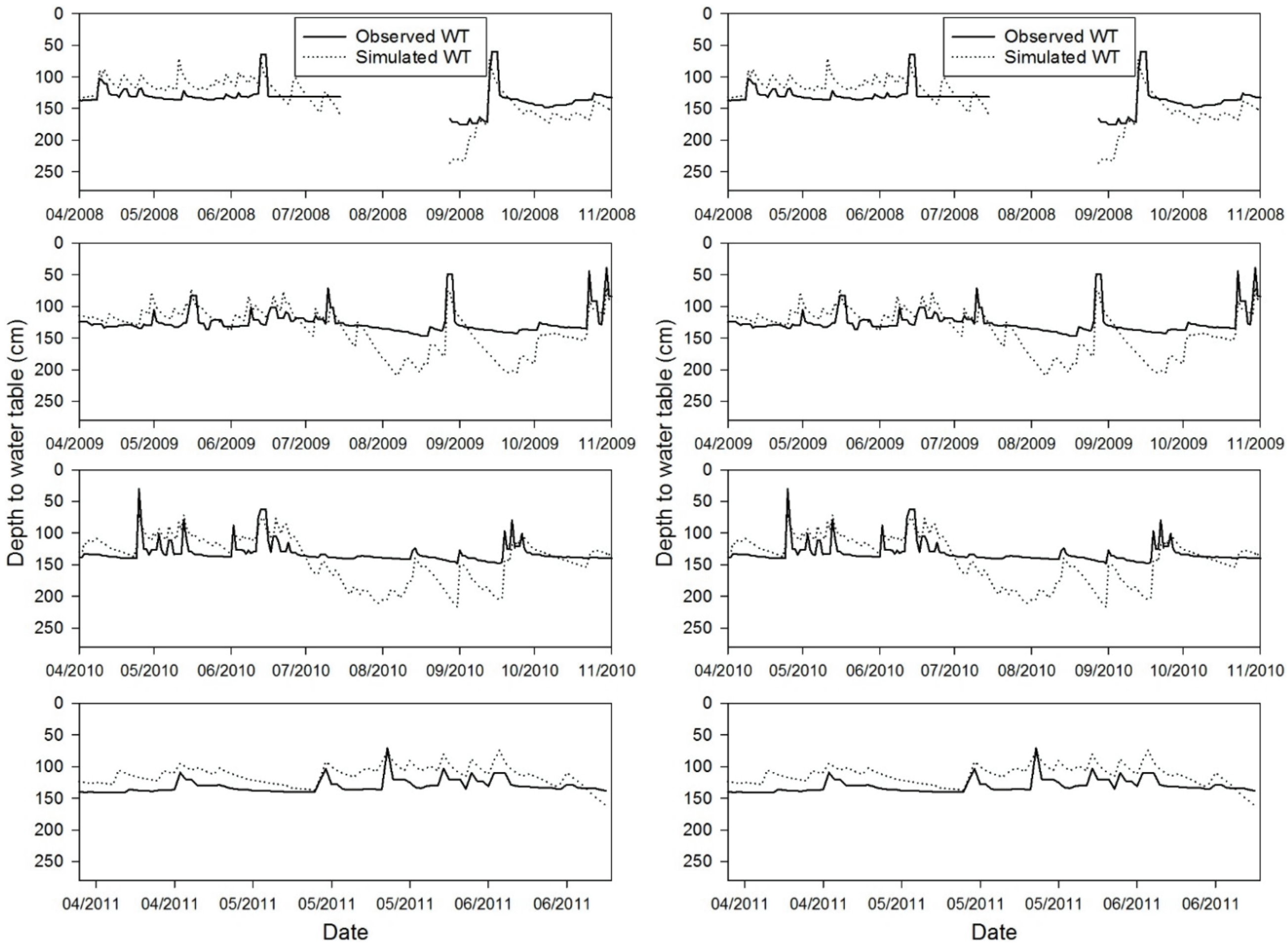

Figure 2. Daily average measured (solid line) and simulated (dotted line) depth to water table (cm) for calibration with the DD system.

datasets of the DD system along with the three tested systems. Inputs of N from fertilizer and soybean fixation were summarized together each year by averaging the fertilizer added to the corn with the fixation by the soybeans. The only other input to the system includes N deposited from the atmosphere with precipitation. Outputs from the system include N lost with lateral flow and deep seepage, N in runoff, denitrification, export of N via grain harvest, and N lost via tile drainage. Grain N export was used to give a broad view of the N balance on a whole-system level. Again, for mass balance purposes, all nine years were used to compare all four simulated systems.

For testing, annual NO_3-N loads and FWANC simulations were analyzed with the same eight years as annual tile drainage simulations (table 6). The simulated average annual N loss for DD was 25 kg N ha^{-1}, which was underpredicted compared to the measured value of 32 kg N ha^{-1}. An average annual load of 17 kg N ha^{-1} was simulated for CD, which compared well to the measured value of 16 kg N ha^{-1}. For SD, the average annual load was simulated as 18 kg N ha^{-1}, similar to the average field measurement of 17 kg N ha^{-1}. Similar to findings by Youssef et al. (2006), most of the N load simulation errors were due to errors in drainage volume simulation. Like drainage volume, most of the 1 kg N ha^{-1} overprediction occurred in the spring months. The simulated percent reduction for NO_3-N load compared to DD was again underpredicted for both CD and SD, similar to drainage volume. The CD system's average annual N load reduction was simulated as 31%, which is lower than the observed reduction of 48%. The SD system's average annual N load reduction was simulated as 28%, which was lower than the observed reduction of 44%. This simulation in percent reduction in annual N loss is reasonable compared with the simulation of CD by Fang et al. (2012), in which the 22% reduction was simulated as 32% with an NSE value greater than 0.5 for annual and monthly N loss for DD and less than 0.5 for CD.

The annual nitrate loss normalized to annual drainage, or FWANC (mg N L^{-1}), was well simulated for calibration with DD, as the simulated nine-year average FWANC of 10.4 mg N L^{-1} is close to the observed value of 10.2 mg N L^{-1}. It is noteworthy that RZWQM2 predicted high FWANC in dry years, which was also observed in field data by Schott et al. (2017). The worst simulated FWANC occurred in 2013, when FWANC was overpredicted by 6 mg N L^{-1}. This may have been because the prior year (2012) experienced rainfall that was 30 cm below average, which caused a buildup of residual soil NO_3-N in the system and a high simulated FWANC in 2013. The 2012 fall soil NO_3-N measurement was higher than the other two measurements taken in 2011 and 2014, and the same was found with the RZWQM2 simulations of fall soil NO3-N for those years. Soil NO3-N in

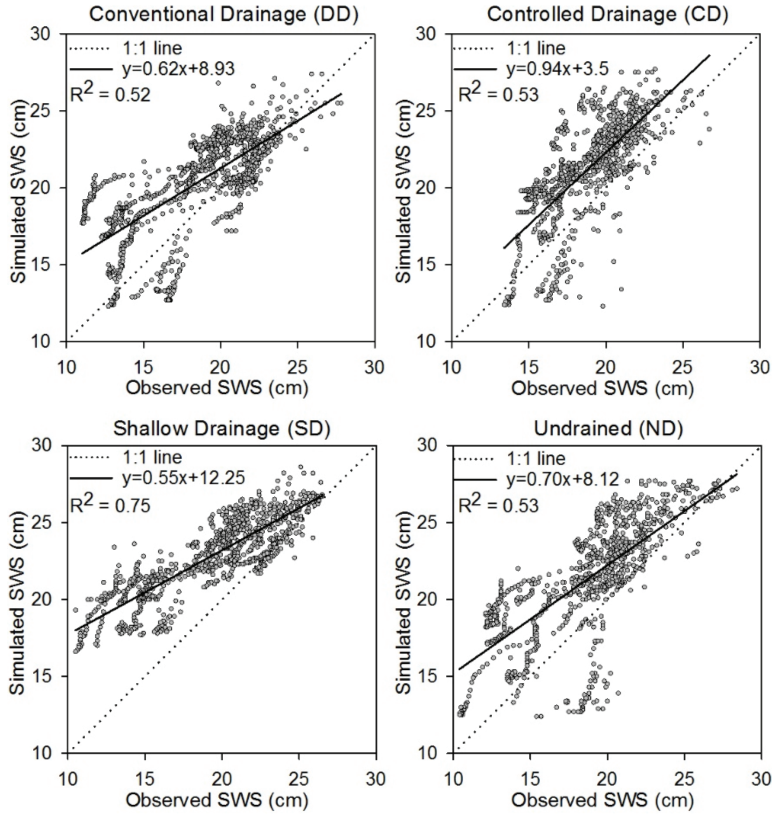

Figure 3. Daily average simulated versus measured 60 cm soil water storage (SWS) for conventional drainage (DD), controlled drainage (CD), shallow drainage (SD), and undrained (ND) with *x*=*y* lines, regression analysis equations, and R^2 values.

Table 8. Simulated average annual N components from 2007 to 2015 for conventional drainage (DD), controlled drainage (CD), shallow drainage (SD), and undrained (ND), along with observed N loss via tile drainage for comparison and the percent of inputs for each loss component.

	DD		CD		SD		ND	
	kg N ha^{-1}	%	kg N ha^{-1}	%	kg N ha^{-1}	%	kg N ha^{-1}	%
Precipitation N	16	-	16	-	16	-	16	-
Fertilizer + fixation	196	-	196	-	196	-	196	-
Lateral seepage N	17	8%	20	9%	21	10%	26	12%
Runoff N	1	0.4%	2	0.9%	2	0.8%	3	1.6%
Denitrification	14	7%	21	10%	19	9%	27	13%
Grain N	157	74%	154	73%	155	73%	158	75%
Tile drainage N	25	12%	17	8%	18	8%	-	-
Tile drainage N observed[a]	32	15%	16	7%	17	8%	-	-
ΔS[b]	-3.3	-	-2.4	-	-1.9	-	-2.5	-
Net N mineralization[c]	104.7	-	97.4	-	92.9	-	89.7	-

[a] Measured value for tile drainage N loss.
[b] Average annual change in soil profile N.
[c] Average annual net N mineralization, equal to N mineralization minus N immobilization.

the fall of 2012 was measured as 27.8 kg N ha^{-1}, while RZWQM2 simulated 33.3 kg N ha^{-1} with 19.6% PE. Randall et al. (2003) demonstrated that soil NO_3-N can accumulate in the profile in a dry year due to mineralization and fertilization and is then flushed out with subsequent rain, causing an unusually high FWANC in the following year. It is unclear why measured FWANC was not higher in 2013, but this could be due to overestimation of tile drainage in 2013. In general, FWANC was overpredicted in above-average rainfall years (2007, 2008, 2009, 2010, 2015) and underpredicted in below-average rainfall years. For the calibration FWANC dataset, the goodness-of-fit statistics are acceptable only for PBIAS at 6.83%. Other research has shown difficulty in simulating the year-to-year variability in FWANC, as the variability is relatively low (Thorp et al., 2007).

FWANC was not as well simulated for CD and SD based on all goodness-of-fit statistics (table 6). However, the eight-year average simulated FWANC for CD (9.3 mg N L^{-1}) is close to the observed average of 10.8 mg N L^{-1}, and the range in simulations (6.3 to 12.6 mg N L^{-1}) is similar to the direct measurements (6.3 to 13.6 mg N L^{-1}). Consistent with the findings of Ma et al. (2007b) and Fang et al. (2012), there was little difference in simulated FWANC between DD and CD. For SD, the average FWANC over eight years was predicted as 12.7 mg N L^{-1} and compared well to the observed average of 11.9 mg N L^{-1}. Measured values of FWANC for SD ranged from 9.2 to 15.5 mg N L^{-1}, while simulated values ranged from 6.8 to 20.2 mg N L^{-1}. Observations in the field study of higher FWANC with the SD system were well simulated. Schott et al. (2015) hypothesized that this is due to a reduced retention time in the soil profile after a rainfall-infiltration event, thus limiting the vulnerability of infiltrating water to undergo denitrification or dilution. This hypothesis was based on evidence provided by Burchell et al. (2005) of preferential flow and shorter flow lines from the surface to the drains in a system with tiles that are shallower and closer together. Drainage water short-circuiting to shallower drains seems to be well simulated in the model, as the increased FWANC in the SD system was simulated, potentially indicating that less time is allowed for N to be lost via other pathways and carried directly to the tile drains. The simulation of FWANC for DD, CD, and SD is encouraging, as the complexities of modeling N cycling and transformations have been found to make N concentration challenging to predict (Fang et al., 2012). Although NSE values were low for FWANC, other simulation studies have reported successful calibration with low or negative NSE values (Fang et al., 2012; Li et al., 2008; Thorp et al., 2007).

Soil NO_3-N measurements, measured in the fall of 2011, 2012, 2014, and 2015, were well simulated for the 0-90 cm layer (table 9). However, the distribution in the upper 90 cm layer was not well simulated, as the 0-30 cm estimate was underpredicted and the 30-90 cm layer was overpredicted. In RZWQM2 simulations, residual NO_3-N in the upper soil profile can be greatly affected by crop N uptake, generating underprediction of NO_3-N with an overprediction of crop N uptake (Thorp et al., 2007). In 2012, the 30 cm soil NO_3-N was the most underpredicted, by around 8.3 kg N ha^{-1}, likely because grain N uptake was highly overpredicted by 33.8 kg N ha^{-1} in 2012. Simulated N mineralization was highest in 2012 of the nine years at 154 kg N ha^{-1}, followed by 2013 at 128 kg N ha^{-1}, likely because, as these were the two driest years, the soil profile was well aerated due to a consistently deep water table. Although higher residual soil nitrate could be expected to high N mineralization, it seems that high corn N uptake offsets the addition of available N from mineralization, as the two years with the highest mineralization are also among the years with the greatest N concentration in corn (1.6% in 2012, 1.4% in 2007, and 1.4% in 2013). As discussed by Ma et al. (2007c), the error in 2012 fall soil NO_3-N may also be partly due to difficulty in simulation because of the drought conditions experienced in that year. Fall soil NO_3-N simulations were also unsatisfactory in the upper 30 cm soil profile for CD, SD, and ND (table 9), indicating underprediction of residual NO_3-N. The 90 cm fall soil nitrate simulations were better for each system; however, the statistics were unsatisfactory for SD and ND.

Table 9. Goodness-of-fit statistics for fall soil nitrate within the 0-30 cm and 0-90 cm soil profile.[a]

	Statistic	DD	SD	ND	CD
0-30 cm	PE	*-21%*	-14%	-9%	*-28%*
	n-RMSE	*32%*	*76%*	*56%*	*48%*
0-90 cm	PE	-2%	1%	*17%*	-1%
	n-RMSE	15%	*35%*	15%	10%

[a] Unsatisfactory goodness-of-fit statistics are italicized.

Long-Term Simulations

Corn grain yield was reduced, on average over the 45 years, by 209 kg ha^{-1} (2.2%) in the undrained system, by 146 kg ha^{-1} (1.5%) in the controlled drainage system, and by 116 kg ha^{-1} (1.2%) in the shallow drainage system, which was due again to the higher simulated water tables generating excess water stress on the corn root system (table 10). The long-term corn yield loss is less than that of the nine-year simulations, likely because the annual average precipitation for the nine-year dataset is 105 cm, or higher than the 45-year dataset average of 90 cm, and six years of the nine-year dataset had above-average rainfall. Within the long-term dataset, the lowest annual rainfall occurred in 1988, with only 47 cm, and the highest annual rainfall was within the nine-year field study (2009), with 137 cm of rain. This long-term simulation of corn yield reduction is likely an underestimation, as yield loss was underpredicted in the nine-year dataset simulations. It is possible that more work needs to be done in the development of excess moisture stress within the maize model in addition to the soybean model.

The 45-year average annual water balances for the four systems are presented in table 11, and the N balances are provided in table 12. The simulated average annual percent reduction in tile drainage volume over the 45 years was 18% for CD and 48% for SD (table 11), while the average annual percent reduction in NO_3-N loss in tile drainage was 26% for CD and 40% for SD (table 12). These reductions are in good agreement with other simulation studies, such as simulations of 20% to 30% reductions in annual drainage NO_3-N loss in CD and SD systems in south-central Minnesota by Luo et al. (2010) and a simulated 30% reduction with CD by Ma et al. (2007b). The simulated reductions in volume are lower than the reductions found in field studies (Helmers et al., 2012; Schott et al., 2015), mostly due to underprediction of drainage volume in the DD system and overprediction of drainage volume in the CD system. On average, the DD system lost 20 kg N ha^{-1} annually via tile drainage water flow in the 45-year simulation, which was less than the simulated tile drainage N loss in the DD system for the nine-year study, which was 25 kg N ha^{-1} annually. This also occurred in the CD and SD systems, with 15 kg N ha^{-1} lost long-term and 17 kg N ha^{-1} lost in the nine-year simulation for CD and 12 kg N ha^{-1} lost long-term and 18 kg N ha^{-1} lost in the nine-year simulation for SD.

The N balances for the long-term simulations showed av-

Table 10. Average annual crop production for 45-year simulations (1971-2015) of corn-soybean systems with conventional drainage (DD), controlled drainage (CD), shallow drainage (SD), and undrained (ND), along with the differences in CD, SD, and ND compared to DD.

	DD	CD	CD-DD Difference		SD	SD-DD Difference		ND	ND-DD Difference	
Crop Production[a]	(kg ha^{-1})	(kg ha^{-1})	(kg ha^{-1})	(%)	(kg ha^{-1})	(kg ha^{-1})	(%)	(kg ha^{-1})	(kg ha^{-1})	(%)
Corn grain yield	9551	9405	-146	-1.5	9434	-116	-1.2	9341	-209	-2.2
Corn AGB	19352	18789	-563	-2.9	18936	-416	-2.1	18515	-837	-4.3
Corn BGB	3872	3955	82	2.1	3897	25	0.6	3842	-31	-0.8
Soybean grain yield	3495	3516	21	0.6	3494	-1	0.0	3487	-9	-0.2
Soybean AGB	7439	7458	19	0.3	7432	-7	-0.1	7411	-28	-0.4
Soybean BGB	1096	1090	-6	-0.6	1091	-5	-0.5	1080	-16	-1.5

[a] AGB = aboveground biomass production, and BGB = belowground biomass production.

Table 11. Annual hydrologic components for 45-year simulations (1971-2015) of corn-soybean systems with conventional drainage (DD), controlled drainage (CD), shallow drainage (SD), and undrained (ND), along with the differences in CD, SD, and ND compared to DD.

	DD	CD	CD-DD Difference		SD	SD-DD Difference		ND	ND-DD Difference	
	(cm)	(cm)	(cm)	(%)	(cm)	(cm)	(%)	(cm)	(cm)	(%)
Precipitation	90	90	-	-	90	-	-	90	-	-
Actual ET[a]	53	50	-3.1	-6	53	0.3	1	55	1.8	3
Actual evaporation	11	11	0.7	6	11	0.6	6	14	2.9	28
Actual transpiration	42	39	-3.7	-9	42	-0.3	-1	41	-1.1	-3
Potential ET[a]	119	111	-7.2	-6	119	0.9	1	122	3.1	3
Potential evaporation	36	36	0.0	0	37	1.0	3	39	3.2	9
Potential transpiration	43	39	-3.8	-9	42	-0.4	-1	42	-1.2	-3
Runoff	2	4	2.2	112	4	2.4	121	9	7.4	374
Lateral seepage	20	23	3.2	16	26	6.1	31	29	9.0	45
Tile drainage	19	16	-3.5	-18	10	-9.2	-48	0	-	-
ΔS[b]	-4	-2	1.1	-32	-3	0.4	-12	-3	0.7	-21

[a] ET = evapotranspiration.

[b] Average annual change in soil water profile storage.

Table 12. Annual N dynamics for 45-year simulations (1971-2015) of corn-soybean systems with conventional drainage (DD), controlled drainage (CD), shallow drainage (SD), and undrained (ND), along with the differences in CD, SD, and ND compared to DD.

	DD	CD	CD-DD Difference		SD	SD-DD Difference		ND	ND-DD Difference	
	(kg N ha^{-1} $year^{-1}$)	(kg N ha^{-1} $year^{-1}$)	(kg N ha^{-1} $year^{-1}$)	(%)	(kg N ha^{-1} $year^{-1}$)	(kg N ha^{-1} $year^{-1}$)	(%)	(kg N ha^{-1} $year^{-1}$)	(kg N ha^{-1} $year^{-1}$)	(%)
Precipitation N	14	14	-	-	14	-	-	14	-	-
Fertilization	77	77	0.0	0	77	0.0	0	77	0.0	0
Fixation	120	123	-3.6	-3	122	-2.0	-2	122	-2.5	-2
Lateral seepage N	18	20	-1.8	-10	23	-4.8	-26	27	-8.7	-47
Runoff N	0.3	0.7	-0.4	-119	0.8	-0.5	-138	1.8	-1.4	-427
Denitrification	14	19	-4.2	-29	19	-4.5	-31	25	-10.3	-71
Volatilization	0	0	0.0	-	0	0.0	-	0	0.0	-
Grain N export	171	169	1.5	1	169	2.0	1	168	2.4	1
Tile drainage N	20	15	5.1	26	12	7.9	40	-	-	-
ΔS[a]	-13	-9	-4.3	32	-11	-2.5	19	-9	-4.5	33
Net N mineralization[b]	120	114	5.2	4	116	3.5	3	113	6.4	5
	(mg N L^{-1})	(mg N L^{-1})	(mg N L^{-1})	(%)	(mg N L^{-1})	(mg N L^{-1})	(%)			
FWANC[c]	10.5	9.2	1	12	11.8	1	13			

[a] Average annual change in soil profile N.

[b] Average annual net N mineralization, equal to N mineralization minus N immobilization.

[c] Flow-weighted annual NO_3-N concentration, calculated as annual N load normalized to the annual drainage volume.

erage annual losses in soil N storage of 13 kg N ha^{-1} $year^{-1}$ for DD, 11 kg N ha^{-1} $year^{-1}$ for SD, 11 kg N ha^{-1} $year^{-1}$ for CD, and 11 kg N ha^{-1} $year^{-1}$ for ND, which are within the range of soil N storage changes across the Midwest region reported by Thorp et al. (2008). In addition, similar to findings by Thorp et al. (2008), the loss in soil nitrate storage was greater in a conventional drainage system than in a DWM system. The differences in soil N storage between DD and the other three systems in these long-term simulations was mostly due to decreased N mineralization of soil profile organic material as well as increased denitrification in the CD, SD, and ND systems. The simulated 5.2 kg N ha^{-1} $year^{-1}$ decrease in N mineralization is similar to the 9 kg N ha^{-1} $year^{-1}$ reported by Thorp et al. (2008) and the 6.2 kg N ha^{-1} $year^{-1}$ reported by Fang et al. (2012) for a CD system. The 4.2 and 4.5 kg N ha^{-1} $year^{-1}$ increases in denitrification from CD and SD, respectively, are similar to the 5 kg N ha^{-1} $year^{-1}$ increase in denitrification reported as a regional estimate by Thorp et al. (2008). Annual lateral subsurface losses increased from 18 kg N ha^{-1} in DD to 20 kg N ha^{-1} in CD and 23 kg N ha^{-1} in SD, and N lost via surface runoff was minimal (<1 kg N ha^{-1} $year^{-1}$) in all three drainage systems.

In the ND system, the N inputs were lost via denitrification, lateral seepage, and runoff. In the ND system, on average, 13% was lost via a subsurface pathway or lateral seepage, 1% via runoff, and 12% via denitrification, compared to 17% subsurface, <1% runoff, and 8% denitrification, on average, across the three tile drainage systems. This indicates that the pathways of N loss in the system without tile drainage converted some of the subsurface losses to surface and gaseous loss pathways. However, the subsurface seepage loss is still significant in the ND system. This diffuse subsurface seepage loss may be available for other loss pathways over time, such as denitrification or plant uptake; however, there may also be lateral seepage into a waterway or to groundwater sources. Research is needed to collect more information on the potential destinations of subsurface losses in undrained systems that are naturally poorly drained but use a subsurface lateral seepage pathway for a large proportion of their water and N balances.

Within the 45-year simulations, the five wettest and the five driest precipitation years were summarized to give the range of impacts of the two alternative drainage systems. Within the five driest years of the 45-year dataset (precipitation ranged from 47 to 69 cm), an average reduction of 77% in annual tile drainage volume and 56% reduction in NO_3-N loss was seen in the SD system. In these years, the CD system only reduced tile drainage by 20% and NO_3-N loss by 28%. Within the five wettest years (precipitation ranged from 119 to 137 cm), SD and CD reduced annual tile drainage volume by 43% and 18%, respectively, and annual NO_3-N loss decreased by 35% with SD and by 22% with CD. It is important to note that the percent reductions in annual drainage and N load were greatest in dry years and lowest in wet years, especially for the SD system. This same phenomenon was observed in southern Minnesota on a similar silty clay loam soil, as Sands et al. (2008) found that a shallow drainage system reduced annual drainage volume from 16.4% to 30.4% over a five-year study and that the years with the largest rainfall produced the least percent reduction in drainage volume. For the five wet years, the FWANC variation was minor between drainage systems: 7.8 mg L^{-1} for DD, 7.5 mg L^{-1} for CD, and 8.9 mg L^{-1} for SD. The dry years had much greater variation: 14.6 mg L^{-1} for DD, 12.1 mg L^{-1} for CD, and 26.5 mg L^{-1} for SD. The CD system likely reduced FWANC because denitrification increased by 5.4 and 2.4 kg N ha^{-1} in the wet and dry years, respectively, and N mineralization decreased by 6.7 and 3.2 kg N ha^{-1} in the wet and dry years, respectively. This increase in denitrification and decrease in N mineralization was similar in the SD system. As mentioned before, the high FWANC simulated for the SD system may be due to infiltrating water short-circuiting to the shallower tile drains.

An annual exceedance probability (AEP) curve for N load was estimated from the 45-year simulations for DD, CD, and SD. We also analyzed the spring exceedance probability (SEP) for N load. Spring N loading included the months of April, May, and June, as the EPA Science Advisory Board indicated that this period has the greatest impact on hypoxia in the northern Gulf of Mexico (Dale et al., 2007). The probability of exceedance curves for N load are provided in figure 4, including both the exceedance probabilities for annual N load (fig. 4a) as well as spring N load (fig. 4b).

In the CD system at SERF, the control boards were managed to release the water held back in the tiles to a depth of 120 cm around April 15 and to replace the boards to 76 cm

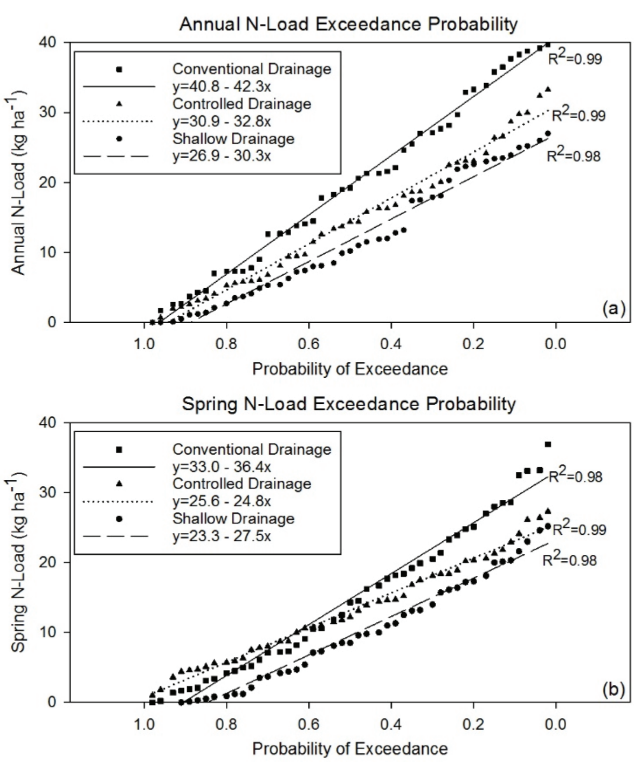

Figure 4. (a) Annual and (b) spring tile drainage N load versus calculated probability of exceedance from 45-year simulations of conventional drainage (DD), controlled drainage (CD), and shallow drainage (SD) with regression analysis equations and R^2 values.

below the ground surface around June 14. This means that the N load reduction potential in tile drainage for CD may be limited from mid-April to mid-June. This limitation was highlighted in a long-term simulation across the Midwest region, in which a controlled-outlet system was found to be most effective in southern areas of the Midwest, such as Missouri, Illinois, Indiana, and Ohio (reductions of 35.8 to 46.3 kg N ha^{-1} $year^{-1}$), and least effective in the northern areas, such as Iowa, Minnesota, and Wisconsin (reductions of 7.1 to 20.4 kg N ha^{-1} $year^{-1}$), which was mostly attributed to differences in the timing of precipitation and drainage (Thorp et al., 2008). In west-central Indiana, DRAINMOD modeling by Ale et al. (2010) showed that the majority (>80%) of N loss reduction with DWM occurred during the non-growing season (November through April), which is also typically when the majority of annual precipitation and drainage occurs (Adeuya et al., 2012). However, in regions similar to southeast Iowa, the timing of the majority of annual precipitation (50%) and drainage (70%) (April, May, and June) generally coincides with the release of drainage water through the control structure to allow for spring field activities (Helmers et al., 2005; Thorp et al., 2008).

The long-term simulation AEP and SEP curves demonstrate the importance of spring drainage for the upper Midwest region, as the majority of annual N loss occurred in the spring months. Additionally, the curves demonstrate that the N loss reduction of CD is poorer in the spring months. The SEP curves show that there was a high likelihood within the 45-year simulations that the CD system generated higher N loads than DD at the high probability, low drainage, and low N load discharges within the spring. This may mean that, with CD, drainage water was held back from the previous fall or winter and released during the spring months, while the DD or SD system allowed this discharge to leave prior to spring. Similarly, Ale et al. (2010) found that April had an increase in drain flow in a controlled drainage system after the controlled outlet was opened on March 31. Research on this effect of spring N load from controlled drainage is lacking. On average, the CD system reduced spring N load by only 11% and increased N load in about half of the years, compared to DD. However, the SD system was effective in reducing spring N load, with an average reduction of 35%. Improvement of controlled-outlet systems may be required to reduce drainage N loading within the critical spring months across the upper Midwest. Although there were issues with the RZWQM2 simulation of spring drainage volume with the CD system, these long-term findings warrant additional research regarding the performance of CD in spring months to fully address hypoxia in the northern Gulf of Mexico. Additionally, because Fang et al. (2012) also reported issues with RZWQM2 simulation of CD volume during the spring when the control boards were released, simulation model routines governing drainage in a controlled-outlet system may need to be re-examined.

SUMMARY AND CONCLUSION

This work adds unique contributions to the relevant literature by demonstrating that RZWQM2 is capable of simulating a shallow drainage system. Additionally, we attempted to quantify the long-term impacts of DWM systems, which only a few studies have done. This work points out potential limitations of RZWQM2-DSSAT simulations of crop production losses under high moisture stress conditions, especially with the CROPGRO-Soybean model. Modeling this excess moisture stress will be vital for simulation models in order to fully understand the impacts and potential drawbacks of future adoption of DWM practices. Additionally, based on these findings, as well as RZWQM2 simulations by Fang et al. (2012), drainage model routines within a controlled-outlet system may need to be re-examined to better simulate drainage volume when the outlet is opened in the spring.

Based on the long-term simulations, the possibility of meeting the Iowa Nutrient Reduction Strategy goal of a 41% reduction in annual TN loss using DWM in southeast Iowa is promising. Considering the long-term 26% N load reduction with controlled drainage and the 40% reduction with shallow drainage (5.1 and 7.9 kg N ha^{-1} $year^{-1}$, respectively), shallow drainage seems to be more consistent in reducing N lost via tile drainage, yet both systems showed substantial reductions in annual N loading. In the long-term simulations, drainage volume was reduced by 18% with controlled drainage and by 48% with shallow drainage, and corn yield losses were predicted as 1.5% with controlled drainage, 1.2% with shallow drainage, and 2.2% with no drainage. It is possible that the yield loss with controlled drainage may be prevented with additional management of the control boards during the growing season to prevent excess moisture stress on the crop, as hypothesized by Schott et al. (2015) and Helmers et al. (2012).

From the long-term simulations, controlled drainage was found to be ineffective at reducing N loss during the spring period in years with low N load, which is the critical period for reducing the impact on hypoxia in the northern Gulf of Mexico. Currently, research that focuses on spring N load of a controlled-outlet drainage system is limited and therefore should be addressed in future research.

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