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Integrated BIosphere Simulator (IBIS) Yield and Nitrate Loss Predictions for Wisconsin Maize Receiving Varied Amounts of Nitrogen Fertilizer

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ABSTRACT

Agriculture in the U.S. Midwest faces the formidable challenge of improving crop productivity while simultaneously mitigating the environmental consequences of intense management. This study examined the simultaneous response of nitrate nitrogen (NO_3–N) leaching losses and maize (*Zea mays* L.) yield to varied fertilizer N management using field observations and the Integrated BIosphere Simulator (IBIS) model. The model was validated against six years of field observations in chisel-plowed maize plots receiving an optimal (180 kg N ha^{-1}) fertilizer N application and in N-unfertilized plots on a silt loam soil near Arlington, Wisconsin. Predicted values of grain yield, harvest index, plant N uptake, residue C to N ratio, leaf area index (LAI), grain N, and drainage were within 20% of observations. However, simulated NO_3–N leaching losses, NO_3–N concentrations, and net N mineralization exhibited less interannual variability than observations, and had higher levels of error (20–65%). Potential effects of 30% higher (234 kg N ha^{-1}) and 30% lower (126 kg N ha^{-1}) fertilizer N use (from optimal) on NO_3–N leaching loss and maize yield were simulated. A 30% increase in fertilizer N use increased annual NO_3–N leaching by 56%, while yield increased by only 1%. The NO_3–N concentration in the leachate solution at 1.4 m below the soil surface was 30.7 mg L^{-1}. When fertilizer N use was reduced by 30% (from optimal), annual NO_3–N leaching losses declined by 42% after seven years, and annual average yield only decreased by 8%. However, NO_3–N concentration in the leachate solution remained above 10 mg L^{-1} (11.3 mg L^{-1}). Clearly, nonlinear relationships existed between changes in fertilizer use and NO_3–N leaching losses over time. Simulated changes in NO_3–N leaching were greater in magnitude than fertilizer N use changes.

CURRENTLY, a multitude of crop simulation models are available to study the effects of climate and farm management practices on carbon (C) and nitrogen (N) cycling (Molina et al., 1983; Parton et al., 1987), crop yield (Jones and Kiniry, 1986; Jones et al., 1988; Sharpley and Williams, 1990; Rosenberg et al., 1992; Easterling et al., 1996; Mearns et al., 1992, 1999), and solute transport (Dodds et al., 1998; Pang et al., 1998; Shirmohammadi et al., 1998; Zhao et al., 2000). The EPIC (Sharpley and Williams, 1990), CERES-Maize (Jones and Kiniry, 1986), and GLYCIM (Acock and Trent, 1991; Haskett et al., 1995, 1997) crop models are some of the more recognized tools used in past assessments of climate change on U.S. agriculture (Cooter, 1990; Rosenzweig, 1990; Rosenzweig and Tubiello, 1996; Easterling et al., 1993, 1996; Haskett et al., 1995, 1997; Brown and Rosenberg, 1997; Mearns et al., 1999; Brown et al., 2000). With the large number of tools available, one might question the need for another crop model. However, other crop models are limited by the scale at which they can operate, and by the empirical approach they take (rather than a process-based approach).

Previously mentioned crop models also are somewhat limited by a lack of mechanistic modeling of physiological (i.e., plant photosynthesis and stomatal conductance) and biophysical processes (i.e., water, energy, C, and N balances) (Boote et al., 1996); often require cultivar-specific data (e.g., genetic coefficients); and in other cases, are not capable of simulating continuous field conditions (i.e., year to year) (Zhao et al., 2000; Chung et al., 2001). For example, the CERES-Maize and DRAINMOD-N models are limited by an inability to simulate frozen soils (Pang et al., 1998; Zhao et al., 2000). Past evaluation of these crop models has focused on measured versus predicted yield comparisons, with much less information reported on how the models capture water and N cycling (Zhao et al., 2000), or how they capture feedbacks between integrated biogeochemical processes. Similarly, a study by Zhao et al. (2000) using DRAINMOD-N evaluated the effect of N fertilization rate on N leaching, but failed to characterize the response of crop yield to N. Nonetheless, these models and studies have contributed to tremendous progress in understanding how crops might respond to future climatic change (Rosenzweig, 1990; Easterling et al., 1996; Mearns et al., 1999; Southworth et al., 2000) and management decisions.

Our purpose here was to develop the capability to study the simultaneous interactions between climate variability, land management, soils, crop growth, C and N cycles, and agrochemical leaching at several important scales. The goal was to develop a process-based model based primarily on differences in C_3 and C_4 plant physiology and crop phenology that was responsive to management options (e.g., irrigation, fertilizer application, planting date) and environmental stresses (e.g., climate and water and nitrogen limitations). Our approach takes advantage of the mechanistic nature of a well-tested

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Abbreviations: CP, chisel-plowed; CV, coefficient of variation; IBIS, Integrated BIosphere Simulator; LAI, leaf area index; NT, no tillage.

dynamic global ecosystem model (DGEM), the Integrated BIosphere Simulator (or IBIS) (Foley et al., 1996, Delire and Foley, 1999; Kucharik et al., 2000, 2001; Lenters et al., 2000), and minimizes the number of variables that control crop growth and behavior. Many crop models are reliant on numerous empirical parameters that require adjustment depending on species, hybrid, and geographic location. The structure of the IBIS modeling framework allows for other crop types (e.g., soybean [*Glycine max* (L.) Merr.] and spring and winter wheat [*Triticum aestivum* L.]) to be simulated in addition to maize.

Our approach, which does not require hybrid-specific input constants, is necessary because the IBIS model is frequently used to simulate interactions of the soil–plant–atmosphere system across continental scales at coarse resolution (0.5° or 5-min terrestrial grid). Detailed crop growth parameters such as heat units to maturity and the response of specific hybrids to water and nitrogen limitations are not currently available as gridded datasets at these scales. This type of ecosystem modeling approach is essential to studying land-use changes on regional hydrology, the effect of N fertilizer use on nitrate flux to waterways (S. Donner and C. Kucharik, unpublished data, 2002), and the effects of irrigation and climate variability on crop growth in water-limited regions such as the Mississippi and Lake Chad basins (Coe and Foley, 2001), and it gives significant capability to diagnose the influence of land management on C cycling and C sequestration potential (Kucharik et al., 2001).

In general, simulations of plant behavior within 0.5° grid cells (approximately 2500 km^2) are limited more by aggregation of soils data (e.g., texture and physical properties) and current climate information. However, model implementation was also desired for much finer scales, both at an individual-field scale (approximately 100 m^2) and a precision agriculture scale (approximately 25 m^2), where some flexibility in altering model parameters was desired. Generally, crop hybrid information such as heat units required to silking and physiological maturity is readily available at the field scale. Thus, the modeling approach does allow for these types of variables to be readily adjusted if desired. While the IBIS model can be used at a variety of spatial scales to simulate crop growth and behavior, as the model is applied at finer resolution, modelers must decide whether additional field scale processes should be included to provide more realism. For example, a precision agriculture version of IBIS (the Precision-Agricultural Landscape Modeling System [PALMS]) operates on an individual field with 5-m grid cells and includes a diffusive wave runoff model (Julien et al., 1995) with ponding.

Version 2 of IBIS (Kucharik et al., 2000) provided the framework for crop model development. The original IBIS model included, in a single integrated framework for natural vegetation, representations of land-surface processes (energy, water, and momentum exchange between soil, vegetation, and the atmosphere), canopy physiology (canopy photosynthesis and conductance), natural vegetation phenology, vegetation dynamics (allocation, turnover, and competition between plant types), and terrestrial C balance (net primary production, tissue turnover, soil C, and organic matter decomposition) (Foley et al., 1996; Kucharik et al., 2000). These processes are organized in a hierarchical framework and operate at time steps ranging from one hour to one year. This approach allows for explicit coupling among ecological, biophysical, and physiological processes occurring on different timescales. This modeling framework was adapted and modified to provide the capability to simulate typical C_3 and C_4 crop types across the central USA. Figure 1 shows the IBIS model structure, adapted for cropping systems. Model output includes crop yield, dry matter production (leaves, stem, root, and grain), harvest index, daily LAI, root growth and turnover, total plant N uptake, net N mineralization, plant tissue C and N, evapotranspiration, soil C and N, and soil CO_2 flux.

The connection between N fertilizer management at the farm level in the upper Midwest and its influence on aquatic systems at the large-river-basin or watershed scale has already been well documented (Rabalais et al., 1996; Goolsby et al., 1999, 2000). Across the U.S. Midwest, it has been suggested that the Mississippi River transports a considerable amount of NO_3–N from farms to the Gulf of Mexico, contributing to hypoxia and the expansion of the Gulf "dead zone" (Goolsby et al., 2000). In the immediate future, inorganic N fertilizer will need to be applied with more precision, and probably at a reduced rate throughout the U.S. Midwest (Showstack, 2000). However, the effects of such management on NO_3–N leaching and crop yield have yet to be quantified over the entire region, particularly in Wisconsin. Economic losses are an immediate concern for growers when adapting new management practices. Local ground water contamination is a widespread and serious problem. Currently in Wisconsin, as many as 14% of wells in the state have ground water NO_3–N concentrations greater than 10 mg L^{-1}. In the Lake Mendota watershed, within which the city of Madison is located, 40% of wells are above the USEPA health standard (Seely, 2001).

Numerous studies in the past decade have examined the influence of N fertilizer usage and the effects of crop rotations on NO_3–N leaching in agricultural systems. For example, studies by Toth and Fox (1998), Owens et al. (1995, 2000), and Klocke et al. (1999) have examined the effects of crop rotations on NO_3–N leaching losses. Toth and Fox (1998) showed that NO_3–N leaching varied from 55 to 81 kg N ha^{-1} yr^{-1} in a continuous maize system in Pennsylvania with mean concentrations of NO_3–N in leachate of 15 mg L^{-1}. However, their study concluded that including alfalfa (*Medicago sativa* L.) in rotation with maize would have resulted in reducing NO_3–N leaching by 75% compared with the continuous maize system. Owens et al. (2000) reported that leachate NO_3–N concentrations were 9.9 mg L^{-1} for a corn–soybean rotation on silt loam soils in Ohio. Klocke et al. (1999) reported that previous legume N credits might be too low for soybean because their continuous corn plots showed an annual NO_3–N leaching rate of 52 kg

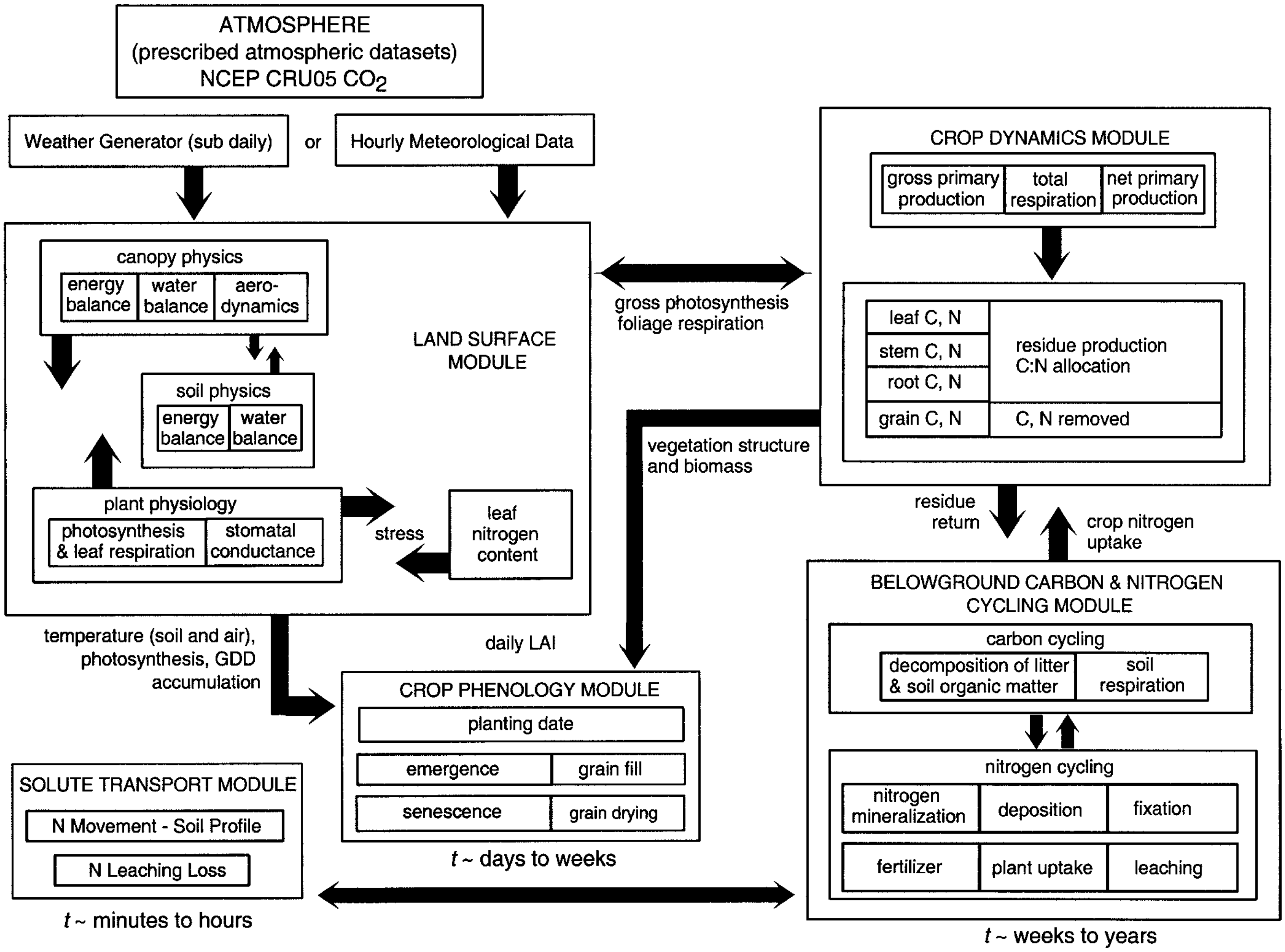

Fig. 1. Schematic of the dynamic ecosystem model IBIS (the Integrated BIosphere Simulator) adapted for agroecosystems.

N ha^{-1} compared with 91 kg N ha^{-1} in a corn–soybean rotation. Other studies that have examined continuous maize systems generally show high NO_3–N leaching rates, and NO_3–N concentrations in leachate that are much above 10 mg L^{-1} (Owens et al., 1995). Ritter et al. (1993) reported that NO_3–N leached from no-tillage and conventional-tillage irrigated maize on sandy loam soils ranged from 55 to 94 kg N ha^{-1} yr^{-1}. A study by Jemison and Fox (1994) showed that even at economic optimum N fertilizer rates (EON), NO_3–N concentrations in leachate averaged 18.8 and 19.3 mg L^{-1} for nonmanured and manured maize, respectively. Two separate studies (Randall and Iragavarapu, 1995; Sogbedji et al., 2000) demonstrated how cropping system changes and climate variability can lead to short-term elevated NO_3–N leaching rates in continuous maize systems due to rapid mineralization of organic N and increased storage of soil NO_3–N during relatively dry periods.

Multiple investigations have examined how to optimize N fertilizer usage and other crop management decisions (e.g., tile spacing and irrigation) to maintain yield and minimize NO_3–N losses through a combination of both field studies and modeling exercises (Ferguson et al., 1991; Vanotti and Bundy, 1994a; Olness et al., 1995, 1998; Schepers et al., 1995; Pang et al., 1998; Rasse et al., 1999; Sogbedji et al., 2000; Zhao et al., 2000). Of particular interest are the studies of Ferguson et al. (1991) and Rasse et al. (1999) that suggested maize yield was weakly correlated to significant changes in N fertilizer application rates from the optimum recommended rate. Ferguson et al. (1991) reported that yield was affected more by NO_3–N storage soils, and irrigation NO_3–N concentration and amount.

In this study, we investigated the interactions between maize C assimilation and N cycling, water movement, and NO_3–N leaching, and characterized the long-term effects of varied fertilizer N use and climate variability on these natural processes. The first objective of this study was to evaluate the reliability and performance of IBIS at the local field scale (100 m^2) in predicting such quantities as yield, drainage, and NO_3–N leaching. Point-location validation of the IBIS crop model used an extensive array of field data obtained at an agricultural research site in southern Wisconsin between 1995 and 2000 (Kucharik et al., 2001). Using the validated model, the second objective was to examine how maize yield and NO_3–N loss would be affected by four scenarios of potential future N fertilizer management (±30% relative to optimal, optimal, and unfertilized) in a continuous maize system on a typical silt loam soil in southern Wisconsin. The present model does not account for varied tillage practices, and has been calibrated to represent typical chisel-plowed crop systems in Wisconsin and the upper Midwest. Thus, only simulations of this type of tillage management are presented in this study. However, field data collected in no-tillage maize are presented as a benefit to other researchers to demonstrate the effects of tillage in these ecosystems, and to help diagnose whether the effects of tillage will need to be accounted for in future model improvements.

SITE DESCRIPTION

Field data were collected during a 6-yr period from 1995 through 2000 in four maize agroecosystems at the University of Wisconsin's Agricultural Research Station near Arlington, WI (43°17′ N, 89°22′ W). The field experiment was conducted on a Plano silt loam soil (fine-silty, mixed, superactive, mesic Typic Argiudoll) with $<2\%$ slope. A randomized complete block was established for maize tillage treatments of conventional chisel-plowed (CP) and no tillage (NT) in fall 1994 (Brye et al., 2000). A 105-d relative maturity hybrid maize variety was planted for each tillage treatment at two fertilizer N application levels to represent optimal and deficient N requirements for maize (Kelling et al., 1991). Nitrogen-fertilized tillage treatment combinations received 180 kg N ha^{-1} yr^{-1} of surface broadcast-applied ammonium nitrate (NH_4NO_3) immediately following planting, while the N-unfertilized tillage treatments received no supplemental N. Grain was harvested annually and residue was returned to the soil surface. For the CP treatments, tillage occurred in the fall following harvest and the seed bed was prepared by disking in the spring before planting.

FIELD MEASUREMENTS

Leaf area index was measured weekly in the maize agroecosystems with a Li-Cor (Lincoln, NE) LAI-2000 plant canopy analyzer (Welles and Norman, 1991). Aboveground whole-plant samples were collected in the maize agroecosystems following physiological maturity. Ten maize plants per plot were cut at random and the fresh weights of the 10-plant bundles were recorded in the field. The bundle was passed through a tractor-mounted stalk chopper and a subsample of chopped maize tissue was obtained, dried for 48 h at 63°C, and weighed to compute aboveground dry matter. Subsamples of grain were obtained following harvest, dried, and weighed to compute yield. Residue was also collected from inside a randomly placed 0.25-m^2 metal frame in all maize plots following harvest and stalk chopping but before plowing, in 1996 and 1998, and dried for 48 h at 63°C.

Subsamples of dried plant material collected from the maize agroecosystems were ground to pass through a 1-mm mesh screen. Maize grain samples were pulverized to pass through a 1-mm mesh screen. Whole-plant N was determined by a micro-Kjeldahl digestion procedure (Nelson and Sommers, 1973). Ammonium N determinations were made by colorimetric analysis with a continuous-flow ion analyzer (Lachat Instruments, 1993a). Whole-plant C and grain and residue C and N were determined by high-temperature catalytic combustion with a Carlo-Erba (Milan, Italy) Model NA 1500 C and N analyzer. Vegetation C and N concentrations were multiplied by their dry mass to compute C and N content on a dry-mass basis.

In situ net N mineralization was measured monthly throughout the growing season between 1995 and 1998 in the maize agroecosystems with an in situ soil-core and ion-exchange-resin bag (ISC/IERB) technique similar to that of DiStefano and Gholz (1986). Monthly net N mineralization rates measured in the top 20 cm were scaled to the soil profile by assuming that N mineralization rates were proportional to soil profile C storage in the maize agroecosystems (Brye, 1999).

Postharvest soil samples 2 cm in diameter were collected at 30-cm increments to 1.2 m for extractable soil inorganic N (i.e., nitrate and ammonium). Two soil cores per plot were collected, thoroughly mixed, and composited into one sample per plot. Soil samples were dried for 48 h in a forced-draft soil dryer at 33°C. Dried soil samples were ground to pass through a 2-mm mesh screen. Sieved soil was extracted with potassium chloride (KCl) and shaken for 1 h (Bundy and Meisinger, 1994). The KCl–soil solutions were filtered and aliquots of the filtrate were refrigerated until colorimetric analysis could be performed with a continuous-flow ion analyzer (Lachat Instruments, 1986, 1987). Incremental inorganic N concentrations were multiplied by fixed bulk density (i.e., 1.3 g cm^3) and summed for the whole profile.

Drainage and inorganic N leaching losses were quantified with equilibrium-tension lysimeters (ETLs) (Brye et al., 1999; Brye et al., 2001). Two 0.2-μm, porous, stainless steel plate ETLs (0.25 × 0.76 m) were installed in the N-fertilized and N-unfertilized plots in 1995 and 1999, respectively, at a depth of 1.4 m. Suction on the ETLs was maintained continuously with a regulated vacuum system placed at each field site to keep the suction in the ETLs as near as possible to the tensions in undisturbed soil at the same depth. Lysimeters were sampled approximately every 14 d, or more frequently depending on precipitation events, during spring, summer, and fall. During the winter, ETLs were sampled roughly every 30 d. Leachate volumes collected from ETLs were recorded in the field. Filtered leachate was refrigerated until colorimetric analysis was performed for NO_3–N concentrations with a continuous-flow ion analyzer (Lachat Instruments, 1993b).

MODEL IMPLEMENTATION AND VALIDATION

The IBIS model (Version 2.5) and its adaptations (i.e., crop phenology, allocation, and solute transport) for maize ecosystems are described in the appendix, below (Fig. 1). A comprehensive review of IBIS can be found in Foley et al. (1996) and Kucharik et al. (2000). The IBIS crop model was validated with field measurements of maize grain yield, harvest index, plant N uptake, postharvest residue C to N ratio, LAI development, net N mineralization, extractable soil NO_3–N, annual drainage, annual NO_3–N leaching, fall soil inorganic N storage, and flow-weighted mean annual NO_3–N concentration in leachate solutions, all collected at the Arlington study site between 1995–2000 (Brye, 1999; Kucharik et al., 2001). A soil profile depth of 1.4 m was used to compare predicted NO_3–N concentration, drainage, and NO_3–N leaching with field measurements.

Soil layer structure, textural data, and hydraulic and physical properties, which are required inputs for the IBIS solute transport submodel and representative of the study site, are summarized in Table 1. All simulations accounted for the effects of tillage (chisel-plowed) on soil physical properties. Simulations were not performed for no-tillage maize plots. Aboveground plant residue (minus grain) was returned to the soil surface after harvest on 1 November of each year.

Table 1. Soil physical and hydraulic properties according to defined layer structure for chisel-plowed maize at the Arlington Agricultural Research Station, Arlington, WI.

Soil depth	Bulk density	Sand	Silt	b†	Air-entry potential	K_{sat}
cm	Mg m^{-3}		%		J kg^{-1}	kg s m^{-3}
0–30	1.34	16	59	6.4	−1.5	5.0×10^{-5}
30–60	1.39	14	55	6.4	−1.5	5.0×10^{-3}
60–200	1.34	18	55	6.4	−1.5	5.0×10^{-3}

† The term b is defined in the context of the equation $\Psi_m = \Psi_e(\theta/\theta_s)^{-b}$, where soil matric potential (Ψ_m) is related to soil air entry potential (Ψ_e), volumetric water content (θ), and saturated water content (θ_s) (Campbell, 1985).

Model Initialization

The necessary boundary conditions needed to simulate crop growth and response to changing N dynamics require knowledge about current soil organic C and N storage pools to a 1-m depth, a general history of past N inputs (e.g., deposition and fertilizer use) and land management for at least the previous 50 years, and current N fertilizer management. These requirements are necessary so that the model adequately represents current stores of inorganic and organic N pools. Because the IBIS soil biogeochemistry submodel simulates net N mineralization as a function of soil C decomposition and residue quality (and fine root C to N ratios), either soil C pool sizes and decay constants could be used as inputs, or as was used in this study, a numerical acceleration procedure can be used to bring soil C and N pools into equilibrium so they are representative of data measured at field sites. We recommend using the numerical acceleration technique described by Kucharik et al. (2000) because decay constants for C pools are difficult to measure and are not always readily available. Alternatively, the model can simulate crop growth assuming that soil inorganic N is not a limiting environmental condition. In this scenario, it is not necessary to bring the soil biogeochemistry dynamics into equilibrium. However, net N mineralization, NO_3–N leaching, residue and fine root C to N ratio, and plant N uptake will not be properly simulated.

In this study, simulations used an hourly timestep and an initial 55-year spin-up period (1931–1985) assuming that maize was planted each spring at the site (representative of past land use history). During this period, soil C and N dynamics established equilibrium using the numerical acceleration technique (Kucharik et al., 2000, 2001). During this period, monthly climate data (temperature, precipitation, relative humidity, radiation, and wind speed) from the Climate Research Unit's (CRU) dataset (CRU05; New et al., 1999, 2000) for the 0.5° grid cell corresponding to Arlington, WI were used as input for the weather generator within IBIS (WGEN; Richardson and Wright, 1984). Additional empirical equations that describe the diurnal cycle of the simulated meteorological variables (e.g., sine functions) were used to derive hourly weather conditions in combination with WGEN (Campbell and Norman, 1998). The CRU05 dataset provides monthly quantities of each meteorological variable, allowing for interannual variability to be characterized. To provide more realism to crop simulations that are generally more responsive than natural vegetation to weekly weather and soil conditions, the National Center for Environmental Prediction (NCEP) reanalysis of daily weather events between 1958–1985 was used in combination with the CRU climate dataset for the Arlington region. This ensured that realistic daily weather events were patterned after the NCEP analysis, but statistically preserved the monthly anomalies (i.e., actual quantities) from the CRU05 dataset.

A weather station near the site (within <0.5 km) collected hourly average measurements of air temperature (2-m height), wind speed, relative humidity, precipitation, and solar radiation from 1986 through 2000 and provided hourly weather data to drive model simulations during the later 15-yr period after model initialization. Maize planting dates were prescribed in the model during 1995–2000 in accordance with actual planting dates of 6 May in 1995 and 1996, 29 April in 1997 and 2000, and 30 April in 1998 and 1999. Maize planting dates prior to 1995 were derived from a combination of several temperature requirements described in the appendix.

To account for historical changes in fertilizer usage from 1931 through 1985 at the site, we assumed that 0 kg N ha^{-1} fertilizer was used prior to 1950, with a linear increase to 180 kg N ha^{-1} by 1985, based on records of N fertilizer usage in this region prior to 1986 (Alexander and Smith, 1990), and from observations recorded at the study site during 1986–1994 (T. Andraski, personal communication, 2000). For the period from 1986 through 1994, unfertilized and fertilized maize field plots were subjected to similar management and received an optimum fertilizer application of 180 kg N ha^{-1} yr^{-1}. At planting, N fertilizer was applied as a pulse, broadcast application of NH_4NO_3 to the soil surface. Irrigation was not necessary at the field site. Atmospheric nitrogen deposition data collected at the study site averaged 9.6 kg ha^{-1} (standard error = 2.4) and atmospheric N deposition was used as an additional N source to the crop in the model (Brye, 1999). For all other years, atmospheric N deposition was calculated with an empirical equation as a function of annual precipitation (Kucharik et al., 2000).

Maize Yield Calibration

A calibration procedure was used to modify the effects of leaf N concentration on maximum plant photosynthetic capacity (V_{max}) so that maize yields of 12 to 15 Mg ha^{-1} (assuming grain contained 15% moisture content and 45% C) could be attained under nonstressed (e.g., by water or N), optimum conditions (180 kg N ha^{-1} yr^{-1} fertilizer), and 4 Mg ha^{-1} were obtained in N limited (0 kg N ha^{-1} yr^{-1} fertilizer) conditions. These are reflective of data collected during the field experiment at Arlington during the late 1990s. Thus, the model has been calibrated to characterize late-1990s yields that are representative of the region, and a process that was independent of soil type was used. There was no calibration performed for the phenological stages of plant growth (e.g., genetic parameters for the specific cultivar grown). Maize physiological and growth model parameters are listed in Tables A1 and A2, respectively.

RESULTS AND DISCUSSION

Model Output Comparisons with Field Observations

Soil Inorganic Nitrogen, Net Nitrogen Mineralization, Plant Nitrogen Uptake, Nitrogen Grain Removal, and Residual Soil Nitrate Nitrogen

Comparisons of IBIS-simulated N cycling quantities for CP maize are compared with field measurements in Table 2. Model performance was judged by calculating percent error from observed on an annual basis, aver-

Table 2. Observed and simulated components of the nitrogen cycle for fertilized and unfertilized maize agroecosystems at the Arlington Agricultural Research Station, Arlington, WI. The terms NT and CP represent no-tillage and chisel-plowed maize, respectively. Standard errors are shown in parentheses for field measurements when available. The term MD denotes the average mean difference between measured and predicted values, SE is the standard deviation, CV is the coefficient of variation.

	Soil inorganic N†‡			Net N mineralization‡			Plant N uptake‡			N grain removal‡			Residual soil NO$_3$–N§		
Year	IBIS	NT	CP	IBIS	NT	CP	IBIS	NT	CP	IBIS	NT	CP	IBIS	NT	CP
	kg ha^{-1} yr^{-1}														
							Fertilized								
1995	126.1	–	–	142.9	38.3	83.1	220	–	–	153.5	128.6 (2.1)	139.6 (10.3)	−2.1	–	–
1996	120.1	77.7	91.9	104.0	42.2	224.2	207	201	230	114.3	120.3 (6.8)	117.7 (6.6)	19.8	119.5	150.9
1997	142.4	169.8	130.3	108.9	153.1	205.1	196	245	259	101.7	104.1 (5.5)	114.1 (8.3)	44.6	10.6	49.1
1998	133.3	59.3	83.4	162.8	−36.1	–	237	172	192	152.7	136.3 (25.0)	152.8 (8.0)	18.9	7.8	67.0
1999	125.2	–	–	133.7	–	–	245	175.3	226.2	174.6	93.4 (13.3)	119.5 (13.9)	12.1	–	–
2000	95.5	–	–	104.9	–	–	203	–	–	139.5	–	–	−0.8	–	–
Mean¶	123.8			126.2			218			139.4			15.4		
Mean (% error)#	131.9	102.3 (−29)	101.9 (−29)	118.6	77.9 (−52)	170.8 (31)	221	198.3 (−1)	226.8 (3)	139.4	116.5 (−20)	128.7 (−8)	27.8	46.0 (40)	89.0 (69)
MD		−29.6	−30.0		−40.7	52.2		−22.8	5.7		−22.9	−10.7		18.2	61.2
SE	6.48	34.18	14.43	9.87	38.99	44.20	7.94	16.85	13.72	11.05	7.89	7.49	6.99	36.78	31.38
CV	0.13	0.58	0.25	0.19	1.00	0.45	0.09	0.17	0.12	0.19	0.15	0.13	1.11	1.38	0.61
							Unfertilized								
1995	61.0	–	–	144.1	96.4	−14.4	138	–	–	91.2	62.5 (3.4)	69.8 (9.4)	−94.3	–	–
1996	32.2	82.0	54.6	91.3	77.2	113.1	99	87	85	33.4	41.0 (0.6)	48.0 (4.5)	−36.0	36.6	86.3
1997	30.5	108.9	92.6	97.4	133.8	164.1	90	121	152	32.7	44.9 (4.6)	56.2 (4.4)	−5.9	−31.9	−37.5
1998	44.7	85.4	61.2	146.5	68.7	114.5	114	77	95	52.4	54.8 (3.1)	64.0 (4.4)	13.2	−74.2	−24.2
1999	32.0	–	–	120.2	–	–	117	78.9	95.2	58.0	36.6 (2.1)	61.1 (11.6)	−9.3	–	–
2000	23.4	–	–	103.2	–	–	93	–	–	33.4	–	–	−8.0	–	–
Mean¶	37.3			117.1			109			48.8			−23.4		
Mean (% error)#	35.8	92.1 (61)	69.5 (48)	119.8	94.0 (−27)	94.3 (−27)	105	91.0 (−15)	106.8 (2)	53.5	48.0 (12)	59.9 (11)	−9.6	−23.2 (−59)	8.2 (217)
MD		56.3	33.7		−25.8	−25.5		−14.0	1.8		−5.5	6.4		−13.6	17.8
SE	5.51	8.46	11.72	9.15	14.47	38.13	7.43	10.24	15.25	9.34	4.72	3.69	15.56	32.28	39.24
CV	0.36	0.16	0.29	0.20	0.31	0.81	0.17	0.23	0.29	0.46	0.22	0.14	−1.63	−2.41	8.29
Difference††	−69.9	−9.9	−31.8	−7.2	20.7	−44.8	−50.2	−54.1	−52.9	−63.9	−58.8	−53.4	−251	−150	−91

† Both simulated and measured soil inorganic nitrogen include the surface to 1.4-m profile. Measurements were made during fall season; IBIS values are for 1 October.

‡ Values represent total inorganic nitrogen ($NO_3 + NH_4$).

§ Residual fall soil inorganic nitrogen is calculated by subtracting nitrogen outputs (leaching and plant uptake) from nitrogen inputs (mineralization, deposition, fertilizer).

¶ Mean for IBIS simulations 1995–2000.

Mean calculated for IBIS only includes years that had comparative field data. Error = [(measured mean − simulated mean)/measured mean] × 100.

†† Difference = [(unfertilized mean quantity − fertilized mean quantity)/fertilized mean quantity] × 100. The row "Mean" is used for simulated differences; the row "Mean (% error)" is used for measured differences.

aged over the 6-yr period. In accordance with previous modeling studies, a 20% error level was used to categorize satisfactory model performance (Chung et al., 2001). The coefficient of variation (CV) was calculated independently for simulated and measured quantities using mean values (mean divided by the standard deviation) during the 6-yr period to assess which quantities were most variable.

In both fertilization treatments, IBIS generally performed satisfactorily in capturing annual average values and interannual variability of plant N uptake and N grain removal as model error was generally less than 10% for the CP case. The CV was comparable for simulated (0.09) and measured (0.12) N-fertilized plant N uptake, and also for N-unfertilized cases (simulated = 0.17, measured = 0.29). Similarly, the CV for N-fertilized CP grain N removal was small and comparable between simulated (0.19) and measured (0.13) quantities (Table 2). A higher CV was noted in the simulated N-unfertilized case (0.46). It is clear that tillage significantly influenced measured quantities, and model error was generally higher when compared with NT data (Table 2), which was expected. IBIS simulated 50 and 65% reductions in plant N uptake and grain N removal, respectively, when switching from optimal recommended fertilizer rates to zero N for CP maize. Field data support these simulated differences, showing a 53% reduction in both N uptake and grain N removal for the 6-yr period (Table 2).

We note that in the comparisons of net N mineralization, soil inorganic N, and residual soil NO_3–N, more significant model error was observed, on average, during the 6-yr period. Simulated soil inorganic N is the sum of both solution-contained and soil-bound N on 1 November. Measured quantities of soil inorganic N were obtained in October or November of each year. Model error was between 27 and 61% for soil inorganic N and net N mineralization, and higher for residual soil NO_3–N. There was a model bias of having more soil inorganic N in fall than measured in N-fertilized plots (29%), and a negative bias in N-unfertilized plots (−48%).

It was clear that the simulations did not capture the large range of interannual variability in some N cycling quantities as found in field observations. There are several potential reasons for this behavior. Clearly, soil inorganic N and residual soil NO_3–N are affected by all other N budget components in any particular year. For example, in 1996, the simulated values of NO_3–N loss (Table 3), grain N removal, and plant N uptake for the N-fertilized case were within 10% of measured values. However, simulated and measured net N mineralization differed by a factor of 2 (104 vs. 224 kg N ha^{-1} yr^{-1}), which probably accounted for the large discrepancy in

Table 3. Observed and simulated annual NO_3–N and drainage properties for fertilized and unfertilized maize agroecosystems at the Arlington Agricultural Research Station, Arlington, WI. The terms NT and CP represent no-tillage and chisel-plowed maize, respectively. Standard errors for field measurements are reported in parentheses when available. The term MD denotes the average mean difference between measured and predicted values, SE is the standard error, and CV is the coefficient of variation.

	NO_3–N concentration†‡			NO_3–N loss‡			Drainage‡		
Year	IBIS	NT	CP	IBIS	NT	CP	IBIS	NT	CP
	mg L^{-1}			kg ha^{-1} yr^{-1}			mm yr^{-1}		
				Fertilized					
1995	19.4	–	–	105.7	–	–	466.7	–	–
1996	18.8	28.8 (29.4)	23.9 (46.9)	57.9	63.9 (15)	61.8 (15)	260.3	219.4 (51)	258.7 (32)
1997	21.2	17.1 (12.2)	13.6 (10.5)	48.3	30.5 (5.5)	33.6 (8.7)	192.1	178.2 (45)	247.3 (83)
1998	20.2	36.3 (1625)	22.5 (22.1)	86.0	102.0 (52)	75.5 (19)	369.2	281.3 (3.2)	335.3 (86)
1999	19.6	2.6 (50.0)	2.6 (8.5)	56.8	5.9 (2.6)	8.5 (3.7)	242.4	235.1 (5.2)	327.8 (43.2)
2000	18.8	7.9 (18.6)	10.9 (5.4)	82.6	23.8 (3.1)	56.6 (10.4)	377.7	300.0 (16.8)	519.6 (192.4)
Mean§	19.6			72.9			318.1		
Mean (% error)§	19.7	18.6 (−7)	14.7 (−34)	66.3	45.2 (−47)	47.2 (−40)	288.3	242.8 (−19)	337.7 (15)
MD		−1.2	−5.0		−20.6	−19.1		−45.5	49.4
SE	0.37	6.30	3.92	9.00	17.02	11.80	42.12	21.83	48.79
CV	0.05	0.76	0.60	0.30	0.84	0.56	0.32	0.20	0.32
				Unfertilized					
1995	17.9	–	–	99.8	–	–	476.5	–	–
1996	9.1	3.8	3.0	28.3	–	–	266.3	–	–
1997	5.9	1.2	3.3	14.0	–	–	198.4	–	–
1998	4.2	–	–	18.8	–	–	388.7	–	–
1999	4.3	–	–	12.7	–	–	248.5	–	–
2000	4.1	0.99 (0.62)	3.2 (4.0)	18.7	3.2 (1.6)	4.2 (3.2)	395.0	325.7 (256.2)	130.0 (57.2)
Mean§	7.6			32.0			328.9		
Mean (% error)§	6.4	2.0 (−220)	3.2 (−100)	18.7	3.2 (−484)	4.2 (−345)	395.0	325.7 (−21)	130.0 (−204)
MD		−4.4	−3.2		−15.5	−14.5		−69.3	−265.0
SE	2.21	0.82	0.10	13.73	–	–	43.64	–	–
CV	0.71	0.78	0.07	1.05	–	–	0.32	–	–
Difference#	−61.4	−87.4	−78.5	−56.0	−92.9	−91.1	3.4	34.1	−61.5

† Simulated and measured concentration is the flow-weighted mean annual average.

‡ Measurements were made with equilibrium tension lysimeters at a 1.4-m soil depth; IBIS simulations depict nitrate N and drainage passing through the 1.4-m soil layer.

§ Mean for IBIS simulations 1995–2000.

¶ Mean calculated for IBIS only includes years that had comparative field data. Error = [(measured mean − simulated mean)/measured mean] × 100.

Difference = [(unfertilized mean quantity − fertilized mean quantity)/fertilized mean quantity] × 100. The row "Mean" is used for simulated differences; the row "Mean (% error)" is used for measured differences.

simulated and actual residual soil NO_3–N values. In general, the largest year-to-year differences between simulated and measured N budget components were for net N mineralization (Table 2). Net N mineralization field data exhibited significant year-to-year variation in both N-fertilized (CV of 1.0 and 0.48 for NT and CP, respectively) and N-unfertilized (CV of 0.31 and 0.81 for NT and CP, respectively) treatments. In particular, the effects of tillage and fertilization rate combined showed a significant effect on measured net N mineralization. There was a 21% increase in net N mineralization rate for N-unfertilized NT plots when compared with N-fertilized NT plots, while a 45% decrease was observed in CP treatments (Table 2).

In contrast to field measurements, model simulations demonstrated that net N mineralization was one of the least variable quantities on a year-to-year basis, and there was only a 7% difference between N-fertilized and N-unfertilized simulations on average for the 6-yr period. The CV for simulated annual net N mineralization for N-fertilized and N-unfertilized scenarios was only 0.19 and 0.20, respectively (Table 2). There are several potential explanations for discrepancies between simulated and observed annual net N mineralization. First and foremost, the soil biogeochemistry module in IBIS contains several pools of soil C and N (e.g., fast, slow, and passive), which is an oversimplification of the continuum of varied ages of cohorts of soil C and N. While simulated N and C mineralization were dependent on general water and temperature functions that approximate the response of microbial activity (Lloyd and Taylor, 1994; Linn and Doran, 1984), it is possible that the soil microbial biomass at Arlington may respond differently to environmental conditions than these relationships depict. Second, IBIS does not explicitly simulate C pool dynamics as a function of soil depth. Rather, the temperature and moisture functions that control microbial activity and decomposition are calculated with weighted averages of soil temperature and moisture based on the fine root profile. Third, simulations assume that all aboveground residue becomes immediately available to microbes for decomposition and the residue layer does not affect soil boundary conditions (e.g, heat and energy exchange). Clearly, surface residue can have an immediate effect on soil moisture and temperature conditions near the surface and could affect microbial activity. Lastly, the soil biogeochemistry submodel assumes that C to N ratios of the microbial biomass and soil C and litter pools remain constant throughout simulations. The effects of SOM on soil physical properties are also not accounted for.

In addition, net N mineralization was quite difficult to measure and most variable in the field (Brye, 1999). This was particularly evident in our field data that show significant interannual variability in addition to significant influences of fertilizer rate and tillage practice. All of these management practices probably disrupt the functioning of the microbial community, which are not explicitly captured by IBIS.

Unfertilized plots showed a tendency in both field data and model simulations to be losing NO_3–N, which would be expected from the change of optimal N fertilization for years previous to 1995, to 0 kg N ha^{-1} at the start of the measurement period. Model simulations also captured the excess residual NO_3–N calculated for the N-fertilized systems (89 kg ha^{-1} for 1996–1998), but on average, simulated values were 69% lower. A similar bias was evident in the unfertilized case, but model error was even larger (217%). Large CVs were evident in both simulated and measured residual soil NO_3–N values (Table 2).

Yield, Harvest Index, Leaf Area Index, and Residue Carbon to Nitrogen Ratio

Model performance for vegetation structure and crop production agreed reasonably with field observations. Table 4 shows simulated residue C to N ratio (i.e., post physiological maturity), peak LAI, yield, and harvest index, which was defined as yield biomass divided by total aboveground biomass, for N-fertilized and N-unfertilized CP scenarios in comparison with field observations. Capturing the correct residue C to N ratio was particularly important in IBIS because it directly affected rates of C accumulation and net N mineralization. The average simulated error during the 6-yr period was generally less than 20% for all quantities, and in many cases, model error was below 10%. Reasonable agreement was achieved between simulated and measured yield and harvest index.

Simulated average yield for N-fertilized CP maize was 9.9 Mg ha^{-1}, while average yield was 10.4 Mg ha^{-1} from field data (Table 4). Field measurements depicted that NT yields for the 6-yr period were 7 and 21% lower on average for N-fertilized and N-unfertilized CP plots, respectively. Simulated (5.6 Mg ha^{-1}) and measured (5.6 Mg ha^{-1}) yield averages for CP were 43 and 46% less, respectively, for N-unfertilized plots compared with the optimally fertilized case. While IBIS simulated a 26% reduction in harvest index when shifting to the N-unfertilized scenario over the 6-yr period, harvest index (HI) decreased by 10% in the measured CP N-unfertilized plots. In comparison, the measured NT N-unfertilized HI decreased by 22%. Field data suggested that suboptimal soil N conditions within the N-unfertilized plots were more influential on yield and HI in NT treatments compared with CP. IBIS predictions of HI were more sensitive to suboptimal soil N conditions in N-unfertilized CP maize than field observations. The average model prediction was only 8% less than field data in the N-fertilized CP simulations, but was 24% lower in the N-unfertilized CP. Regardless of these differences, model error was within an acceptable range (i.e., $\leq$20%), and IBIS satisfactorily captured year-to-year variability evident in field observations. The CVs for simulated maize yield in N-fertilized and N-unfertilized scenarios were 0.19 and 0.22, respectively, while measured CVs were 0.23 and 0.15 for N-fertilized and N-unfertilized CP, respectively. Similar CVs were calculated for harvest index and were comparable between simulated and measured values (Table 4).

Simulated peak LAI varied between 3.3 and 3.9 m^2

Table 4. Observed and simulated crop growth quantities for fertilized and unfertilized maize agroecosystems at the Arlington Agricultural Research Station, Arlington, WI. The terms NT and CP represent no-tillage and chisel-plowed maize, respectively. Standard errors are reported in parentheses for field measurements when available. The term MD denotes the average mean difference between measured and predicted values, SE is the standard error, and CV is the coefficient of variation.

	Residue C to N ratio			Peak LAI†			Maize yield			Harvest index‡		
Year	IBIS	NT	CP	IBIS	NT	CP	IBIS	NT	CP	IBIS	NT	CP
				m^2 m^{-2}			Mg ha^{-1} yr^{-1}					
							Fertilized					
1995	72.8	–	–	3.31	3.82	3.92	11.0	10.1 (0.1)	10.5 (0.3)	0.53	0.57	0.52
1996	70.9	73.9 (6.3)	69.6 (4.4)	3.58	3.89	3.8	8.2	9.0 (0.2)	8.7 (0.3)	0.39	0.42	0.38
1997	71.5	–	–	3.31	3.33	3.28	7.3	8.0 (0.3)	8.4 (0.9)	0.39	0.38	0.38
1998	63.2	91.8 (2.1)	80.6 (7.9)	3.46	3.31	3.17	10.9	13.4 (1.9)	14.7 (1.1)	0.47	0.60	0.61
1999	71.7	–	–	3.56	–	–	12.5	8.2 (1.3)	11.0 (1.0)	0.52	0.50	0.58
2000	78.5	–	–	3.86	–	–	9.9	8.8 (0.4)	8.9 (0.7)	0.45	–	–
Mean§	71.4			3.51			10.0			0.46		
Mean (% error)§	67.1	82.9 (19)	75.1 (11)	3.42	3.59 (4)	3.54 (3)	10.0	9.6 (−4)	10.4 (4)	0.46	0.49 (7)	0.49 (7)
MD		15.8	8.0		0.17	0.12		−5.9	6.7		0.03	0.03
SE	2.00	8.95	5.5	0.08	0.16	0.19	0.79	0.82	0.97	0.03	0.04	0.05
CV	0.07	0.15	0.10	0.06	0.09	0.11	0.19	0.21	0.23	0.13	0.19	0.22
							Unfertilized					
1995	72.8	–	–	3.31	3.42	3.42	7.3	5.8 (0.2)	6.5 (0.6)	0.43	0.46	0.53
1996	99.4	83.2 (9.1)	98.5 (5.7)	3.58	3.24	3.32	4.4	4.1 (0.1)	4.8 (0.2)	0.27	0.32	0.38
1997	100.0	–	–	3.31	2.97	3.08	3.9	4.0 (0.4)	5.0 (0.3)	0.27	0.28	0.31
1998	99.8	85.2 (7.6)	98.0 (3.0)	3.46	3.13	3.2	6.2	5.3 (0.4)	6.5 (0.6)	0.35	0.47	0.49
1999	99.4	–	–	3.56	–	–	6.2	3.4 (0.2)	6.2 (0.9)	0.36	0.38	0.54
2000	98.7	–	–	3.86	–	–	5.8	4.3 (0.3)	4.9 (0.9)	0.35	–	–
Mean§	95.0			3.51			5.6			0.34		
Mean (% error)§	99.6	84.2 (−18)	98.3 (−1)	3.42	3.19 (−7)	3.26 (−5)	5.6	4.5 (−26)	5.6 (0)	0.34	0.38 (11)	0.45 (24)
MD		−15.4	−1.3		−0.23	−0.16		−18.5	−0.0		0.04	0.11
SE	4.45	1.00	0.25	0.08	0.10	0.07	0.51	0.36	0.35	0.03	0.04	0.05
CV	0.11	0.02	0.004	0.06	0.06	0.05	0.22	0.20	0.15	0.18	0.22	0.22
Difference#	33.0	1.6	30.8	0.0	−11.1	−8.1	−43.4	−53.3	−45.6	−26.2	−22.6	−8.9

† Single-sided leaf area index = (m^2 leaf area/m^2 ground area).

‡ Harvest index = (yield biomass/total aboveground biomass).

§ Mean for IBIS simulations 1995–2000.

¶ Mean calculated for IBIS only includes years that had comparative field data. Error = [(measured mean − simulated mean)/measured mean] × 100.

Difference = [(unfertilized mean quantity − fertilized mean quantity)/fertilized mean quantity] × 100. The row “Mean” is used for simulated differences; the row “Mean (% error)” is used for measured differences.

m^{-2}, but was identical for N-unfertilized and N-fertilized CP scenarios because the model used a thermal time equation to simulate the rate of leaf expansion (see appendix for complete details). If leaf area development at that rate was unable to be supported by the rate of C assimilation and C partitioning factors (i.e., N stress conditions), the specific leaf area (SLA) was allowed to vary from its initial value of 7 m^2 kg^{-1} up to a maximum of 12 m^2 kg^{-1}. Field data showed that total aboveground dry matter production was 62% lower in unfertilized plots (data not shown), but peak LAI was only 11 and 8% lower in N-unfertilized NT and CP plots, respectively (Table 4). This suggested that maize plants adjusted their specific leaf area in the absence of available soil N to help maximize total LAI. The CV for peak LAI was generally the lowest for all quantities analyzed in this study; the CV was 0.06 for both N-fertilized simulations during the 6-yr period, and 0.11 and 0.05 for measured N-fertilized and N-unfertilized CP, respectively (Table 4).

Simulated and measured residue C to N ratios for the CP treatment were 48 and 30% higher, respectively, in N-unfertilized plots compared with the optimally fertilized case. These increases may partially explain the lower rates of net N mineralization in the CP N-unfertilized maize (Table 2).

Nitrate Nitrogen Concentration, Drainage, and Nitrate Nitrogen Leaching

The model performance varied considerably for NO_3–N concentration, drainage, and NO_3–N loss for the N-fertilized systems, but was more difficult to assess for the N-unfertilized plots because of a lack of field data. Monthly measured precipitation at the site and simulated drainage for fertilized CP maize are shown in Fig. 2. On average during the 6-yr study period, simulated monthly drainage was 61% of monthly precipitation (standard error = 0.16).

For the N-fertilized CP simulations, annual flow-weighted mean NO_3–N concentrations at 1.4 m (hereafter referred to as NO_3–N concentration) varied from a low of 18.8 mg L^{-1} in 1996 and 2000, to a high of 21.2 mg L^{-1} in 1997 (Table 3). This range was considerably smaller than field measurements depicted as lysimeter measurements reached a low of 2.6 mg L^{-1} in 1999, and a high of 23.9 mg L^{-1} in 1996. These NO_3–N concentrations are similar to values reported by Klocke et al. (1999) and Toth and Fox (1998) for continuous maize systems. The CV for the simulated N-fertilized case was only 0.05, but was 0.60 and 0.76 for measured CP and NT, respectively (Table 3). The low CV for simulated N-fertilized NO_3–N concentrations could potentially be attributed to the current way that NH_4–N and NO_3–N are characterized in the IBIS solute transport submodel. At any given timestep, 10% of the total soil inorganic N is assumed to be NO_3–N, mobile in soil solution (see appendix), and subject to leaching loss. The remainder of soil inorganic N is assumed to be NH_4–N and immobile. Thus, no explicit nitrification process is simulated, and plants are allowed to have access to both mobile and immobile inorganic N pools for uptake. Furthermore, no explicit volatilization or denitrification function is used (e.g., Riley and Matson, 2000). Based on our results in this study, future improvements will be necessary so that a more dynamic, robust model can be developed to better capture the dynamics between soil NO_3–N and NH_4–N pools (Johnsson et al., 1987; Hutson and Wagenet, 1991). This is obviously crucial to adequately capturing the effects of N fertilizer management on NO_3–N leaching losses. Annual mean simulated NO_3–N concentration was 34% less than N-fertilized CP field measurements.

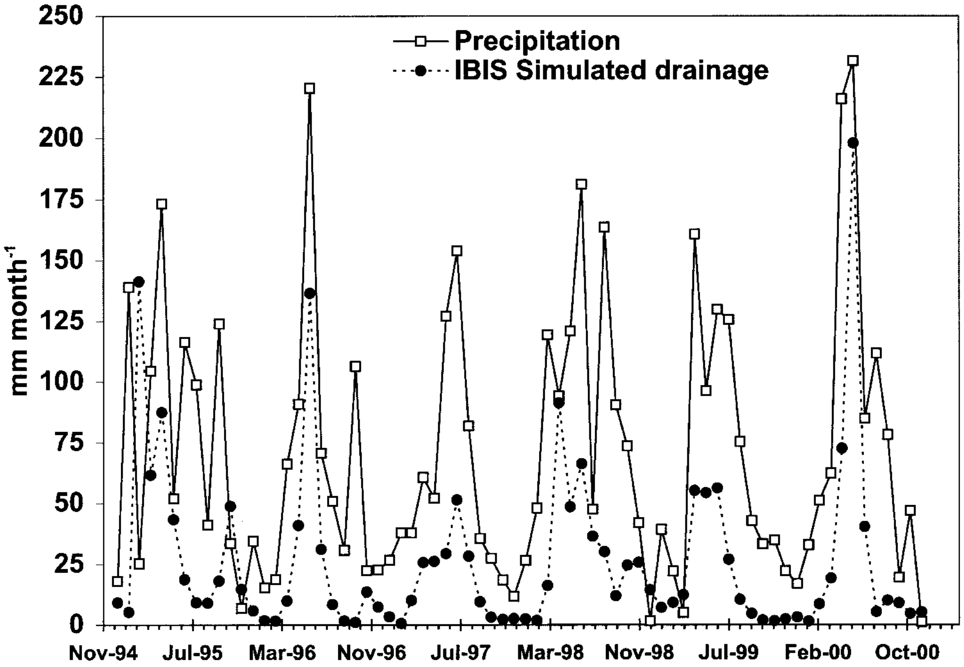

Fig. 2. Monthly total precipitation measured at the Arlington (WI) Agricultural Research Station field site for 1995–2000 compared with IBIS simulated drainage at a 1.4-m soil depth for chisel-plowed maize.

For the N-unfertilized case, IBIS simulations showed a decreasing trend in NO_3–N concentrations, resulting from the change in fertilizer management beginning in 1995 (i.e., 180 kg N ha^{-1} prior to 1995, to 0 kg N ha^{-1}) as stored inorganic N was depleted from the soil. The NO_3–N concentration dropped from a maximum of 17.9 mg L^{-1} in 1995 to a low of 4.1 mg L^{-1} in 2000 (Table 3). Measured values in N-unfertilized NT and CP averaged 2.0 and 3.2 mg L^{-1}, respectively, for the three years of data collected in 1996–1997 and 2000. Overall, simulated NO_3–N concentration declined by 62% over the 6-yr study period when fertilizer application was reduced to 0 kg N ha^{-1}. Simulated NO_3–N levels appeared to stabilize seven years after the management change. These changes were reflected in the high CV for the N-unfertilized simulations (0.71). Measured NO_3–N concentrations dropped by 89 and 78% for NT and CP plots, respectively, with the change in fertilizer use. It appeared that measured NO_3–N levels responded more quickly (i.e., within two years) to the fertilizer use change than simulations reflected.

Simulated soil water drainage and NO_3–N leaching were similar to the interannual variability captured by field measurements and demonstrated significant year-to-year fluctuations. Measured and simulated CVs for drainage in the N-fertilized CP were identical (0.32) (Table 3). However, the measured CV for NO_3–N leaching was higher (0.55) than simulated (0.30) for the N-fertilized CP. Simulated annual drainage was only 15% lower (288.3 mm yr^{-1}) than measurements (337.7 mm yr^{-1}) for the same treatments. Simulated annual NO_3–N loss in N-fertilized CP was, on average, 40% higher (66.3 kg N ha^{-1}) than measured values (47.2 kg N ha^{-1}). These results were indicative of the model simulating slightly more NO_3–N in solution than field observations depicted. Annual NO_3–N leaching losses reported here are typical of rates reported in other studies (e.g., Owens et al., 1995, 2000; Toth and Fox, 1998; Klocke et al., 1999; Sogbedji et al., 2000) that ranged from 31 to 94 kg N ha^{-1} yr^{-1}. Nitrate N leaching during 1999 appeared to be anomalously low compared with other years, especially for those years with similar rainfall. Drainage was also significantly affected by tillage practices in this study, where annual average drainage was 28% lower in NT than CP plots (Brye et al., 2001).

Overall, simulated NO_3–N leaching was reduced by 56% over the 6-yr study period when fertilizer use was reduced to 0 kg N ha^{-1}. Based on only one year of data collected in N-unfertilized treatments, measured NO_3–N leaching dropped by 93 and 91% for NT and CP plots, respectively, with the change in fertilizer use. Likewise, fertilizer management effects on drainage could not be determined from measurements because drainage was only measured in 2000 in the N-unfertilized plots. However, simulations suggested that water uptake by maize in N-unfertilized CP was reduced enough to cause annual drainage to increase on average by 10% (Table 3). This suggested some potential for higher leached NO_3–N during years when plant growth is reduced due to environmental stress, although many other competing factors affect leaching, such as timing of fertilizer application and precipitation timing, intensity, and duration.

FUTURE EFFECTS OF VARIED FERTILIZER APPLICATIONS

Modeling Approach

To quantify the future response of N cycling, drainage, and maize yield to varied levels of N fertilizer use, we imposed four different scenarios of fertilizer use beginning in 1995 and simulated 21 years of maize cropping with these management changes. The model initialization procedure for years prior to 1995 was identical to that described in the model validation section. The control case for 1986 through 2015 was the 180 kg N ha^{-1} fertilization rate, which is optimum for maize yield on silt loam soil in Wisconsin. The three varied rates imposed during this period were N unfertilized (0 kg N ha^{-1}), a 30% reduction relative to optimum (126 kg N ha^{-1}), and 30% excess relative to optimum (234 kg N ha^{-1}). A recent study of Wisconsin farm fertilizer N use found that farmers on average apply an excess of 43 kg N ha^{-1} (Shepard, 2000), which is 24% above the recommended optimum rate and comparable with the 30% excess imposed for future simulations. However, 14% of farmers surveyed were applying 357 kg N ha^{-1}, which is about 100% greater than recommended (Shepard, 2000). This series of IBIS simulations was designed to establish a range of potential responses of NO_3–N leaching, NO_3–N concentrations in subsurface drainage, and crop yield for current and futuristic N management within this highly intensive agricultural region.

For a basis of comparison with a natural, unmanaged vegetation type, a simulation of natural grassland was performed to show the extent to which agriculture has disrupted the natural terrestrial N cycle. A recent study by Kucharik et al. (2001) illustrated how IBIS could be used with confidence to simulate C and N cycling in grassland ecosystems across the region. Tallgrass prairies once dominated the southern Wisconsin landscape before agricultural expansion took place in the mid-1800s (Kucharik et al., 2001). Prairie and grassland restoration is now rapidly gaining popularity in Wisconsin as a means to help improve water quality, control soil erosion, and enhance biodiversity on abandoned agricultural land.

Hourly weather data collected at the site was used to drive IBIS for the entire 21-yr simulation period. For the years 1995–2000, each year's weather record was used. For the period of 2001–2015, a randomly chosen weather record file from the 1986–2000 database was used for each simulation year as a surrogate for future conditions. Thus, the effects of climate change or atmospheric CO_2 increases on maize growth and hydrologic processes were not imposed in the future simulations.

Outlook

Table 5 and Fig. 3 through 6 depict simulated changes in soil inorganic N storage, maize yield, annual drainage, NO_3–N leaching, and NO_3–N concentrations resulting

Table 5. Simulated changes in yield, NO_3–N cycling, and drainage for varied fertilizer management in chisel-plowed maize agroecosystems at the Arlington Agricultural Research Station, Arlington, WI. The control case was a fertilizer N application of 180 kg ha^{-1} at planting; changes in fertilizer management were introduced in 1995. The term SE denotes standard error, and CV is coefficient of variation.

	N fertilizer management (kg ha^{-1} yr^{-1})				Percent change at different N fertilizer management schemes (kg ha^{-1} yr^{-1})		
Parameter†	234	180	126	0	234	126	0
						%	
		1995–2000					
Total NO_3–N leaching, kg N ha yr^{-1}	585	437	318	192	33.8	−27.4	−56.1
Fraction of total N applied	0.42	0.40	0.42	–			
Average NO_3–leaching, kg N ha^{-1} yr^{-1}	98	73	53	32			
NO_3–N concentration, mg L^{-1}	26.6	19.6	13.9	7.6	35.3	−29.3	−61.4
Maize yield, Mg ha^{-1}	10.1	9.9	9.0	5.6	1.3	−9.8	−43.3
Harvest index	0.46	0.46	0.44	0.34	1.1	−4.7	−26.2
Accumulated drainage, mm	1908	1908	1918	1973	−0.04	0.52	3.4
Average drainage, mm	318	318	320	329			
SE NO_3–N leaching, kg N ha^{-1} yr^{-1}	17.11	13.12	12.35	13.39			
SE NO_3–N concentration, mg L^{-1}	2.29	0.60	1.21	2.08			
SE yield, Mg ha^{-1}	17.65	17.26	14.67	9.00			
SE harvest index	0.03	0.03	0.03	0.03			
SE annual drainage, mm	59.82	59.81	60.01	61.72			
		1995–2015					
Total NO_3–N leaching, kg N ha^{-1} yr^{-1}	2151	1377	810	367	56.2	−41.2	−73.3
Fraction of total N applied	0.44	0.36	0.31	–			
Average NO_3–N leaching, kg N ha^{-1} yr^{-1}	103	66	39	18			
NO_3–N concentration, mg L^{-1}	30.7	19.5	11.3	4.6	57.4	−42.2	−76.1
Maize yield, Mg ha^{-1}	9.7	9.7	8.8	5.4	0.7	−8.8	−44.2
Harvest index	0.46	0.46	0.44	0.34	0.6	−4.4	−25.7
Accumulated drainage, mm	6126	6129	6162	6373	0.0	0.6	4.0
Average drainage, mm	291.7	291.8	293.5	303.5			
SE NO_3–N leaching, kg N ha^{-1} yr^{-1}	9.91	6.23	4.51	4.31			
SE NO_3–N concentration, mg L^{-1}	1.23	0.66	0.68	0.74			
SE yield, Mg ha^{-1}	6.52	6.44	5.43	3.31			
SE harvest index	0.01	0.01	0.01	0.01			
SE annual drainage, mm	28.35	28.36	28.51	29.30			
CV NO–N leaching	0.44	0.43	0.54	1.13			
CV yield	0.19	0.19	0.18	0.18			

† Leaching, drainage, and nitrate concentrations are reported for a soil depth of 1.4 m.
‡ Percent change = [(Fertilizer management − control)/control] × 100.

from imposed changes to fertilizer management. Figure 3a depicts the annual precipitation for the site (actual values 1986–2000; random thereafter) and soil water drainage for the control case. Results for the 1986–1994 period are shown for comparison and represent the optimum fertilization rate.

Figure 3b shows how relatively dry years (1988–1989) followed by wet years (1990–1993) can lead to a gradual buildup of soil inorganic N, followed by high rates of NO_3–N leaching (Fig. 3c). This result mimics a study by Randall and Iragavarapu (1995) for a clay loam soil in Minnesota that showed how flow-weighted mean NO_3–N concentration increased in 1990–1991 following the relatively dry period of 1987–1989. Leaching in 1993 was particularly high because above-average spring rainfall immediately followed planting and fertilizer application. Clearly, simulated natural prairie systems do not store excessive amounts of inorganic N; thus NO_3–N concentration in subsurface drainage is generally less than 1 mg L^{-1} and NO_3–N leaching is low (Fig. 3c). These simulated results for a natural prairie are in agreement with measurements collected by Brye et al. (2000).

The data reported in Table 5 show how the changes in fertilizer management affected various quantities during two time periods; the first represents the first six years after management changes (1995–2000), while the second time period is the cumulative results for the 21-yr simulation period. Clearly, the 30% increase in fertilizer use (234 kg N ha^{-1}) did not have a significant effect on yield during the first six years, which only increased annual average yield by 1.2%. However, the total NO_3–N leaching during the first six years after the change increased by 34.2%. A study by Ferguson et al. (1991) in Nebraska also showed that a 75% increase in N fertilizer usage above the optimal recommended rate only increased yield by 1.3%. Figure 3d suggests a new higher equilibrium for drainage-water NO_3–N concentration was established about seven years after fertilizer use changed. More encouraging were the results of the 30% reduction in N fertilizer usage, where during the first six years, annual average NO_3–N leaching declined by 27.6%, but annual average yield only decreased by 9%. Ferguson et al. (1991) only reported a 2.6% decline in maize yield for a 30% reduction (from optimum) of N fertilizer use. A change to N-unfertilized maize helped reduce leaching rates by 56.4% during the first six years but yield also declined 42%; obviously an unlikely future fertilizer management scenario. Cumulative results for the 21-yr simulation period showed that N cycling equilibrium did not occur until at least seven years after management changes occurred (Fig. 4a) while this response time was independent of the N fertilizer manage-

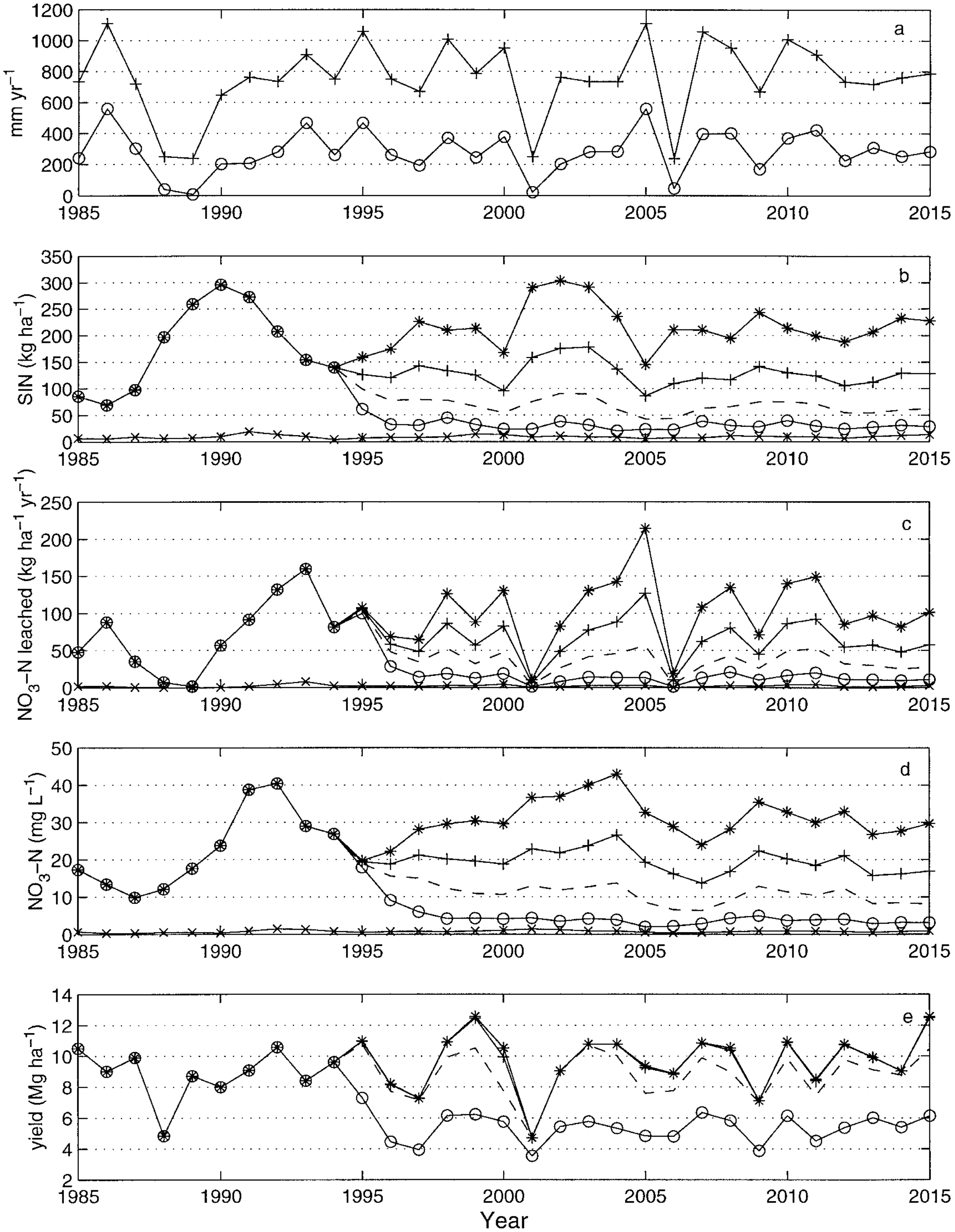

Fig. 3. Results of IBIS simulations showing effects of varied fertilizer N use beginning in 1995 on (*b*) total soil inorganic N storage in top 1.4 m of soil, (*c*) annual NO$_3$–N leaching, (*d*) NO$_3$–N concentration, and (*e*) maize yield. Part (*a*) shows annual precipitation used to drive model simulations compared with predicted drainage for the control case (180 kg N ha^{-1}). Maize received 180 kg N ha^{-1} fertilizer N for years prior to 1995. In (*a*), the symbol + denotes annual precipitation, and ○ is IBIS predicted drainage at 1.4 m for chisel-plowed N-fertilized maize receiving 180 kg N ha^{-1}. For (*b*)–(*d*), * is 234 kg N ha^{-1}, + is 180 kg N ha^{-1}, — is 126 kg N ha^{-1}, ○ is N unfertilized, and × is prairie.

ment scenario. Maize yield response appeared to be immediately affected by the change in fertilizer use (Fig. 4b). The 30% excess-fertilizer-use case showed that compared with the first six years, NO$_3$–N leaching continued to rise on average, and over the entire simulation period, was 56.2% higher than the optimum-fertilizer-use case. The average NO$_3$–N concentration in the drainage water was 30.7 mg L^{-1}, about three times higher than the USEPA health standard of 10 mg L^{-1}. The increase in fertilizer only increased annual average maize yield by 0.7%. Annual average NO$_3$–N leaching loss for the optimum-fertilizer-use case was 65.6 kg N ha^{-1}, and NO$_3$–N concentrations were still above the acceptable USEPA level (19.5 mg L^{-1}). The 30% reduc-

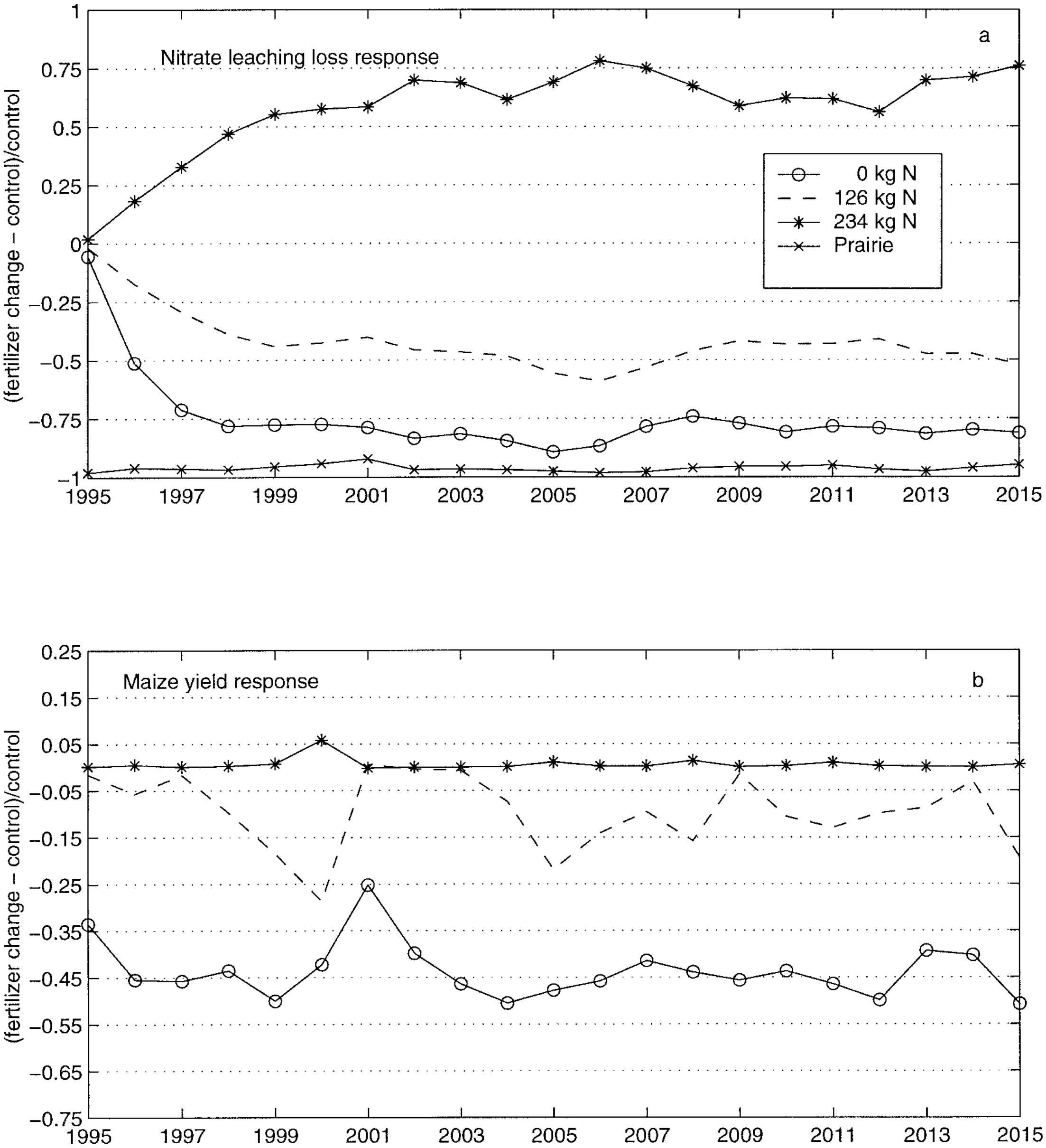

Fig. 4. Fractional change of (*a*) annual NO_3–N leaching and (*b*) maize yield normalized by predictions for the control case (180 kg N ha^{-1}) during the 21-yr simulation. The figure legend applies to both (*a*) and (*b*).

tion case showed that annual average NO_3–N leaching at Arlington could be reduced by 41.2% over the next 15 years, with only an 8.8% reduction in yield. The total amount of NO_3–N leaching losses amounted to 43.7, 36.4, and 30.6% of the amount of N fertilizer applied over the 21-yr period for the 234, 180, and 126 kg N ha^{-1} treatments, respectively (Table 5).

Vanotti and Bundy (1994a) reported a significant linear relationship between maize yield and residual soil NO_3–N and fertilizer applications. While our model was calibrated independently of this dataset, the simulated response of yield to fertilizer use showed close agreement with the Vanotti and Bundy (1994a) study. In applying their regression analysis, we assumed that 30 kg N ha^{-1} was residual NO_3–N based on our model simulations, although the regression model is most sensitive to N fertilizer use. The average yield for the recommended fertilizer application at Arlington was 9.6 Mg ha^{-1}, which was almost identical to our simulated average of 9.7 Mg ha^{-1} (Table 5) during the 21-yr simulation. For a 30% reduction and 30% increase in fertilizer N rate, we used their regression analysis and calculated yield responses to be −7.7% (average yield 9.1 Mg ha^{-1}) and 2.6% (average yield 9.9 Mg ha^{-1}), respectively. These results lend significant confidence to our simulations at Arlington. However, in a N-unfertilized case, predicted yield with their equation was 6.7 Mg ha^{-1} (assuming no residual NO_3–N). This was about 20% higher than our simulated N-unfertilized maize yield, but it was also much higher than the measured average yield at Arlington in the N-unfertilized plots from 1995–2000.

Simulations depicted that for several years, maize yield was controlled more by weather conditions (e.g.,

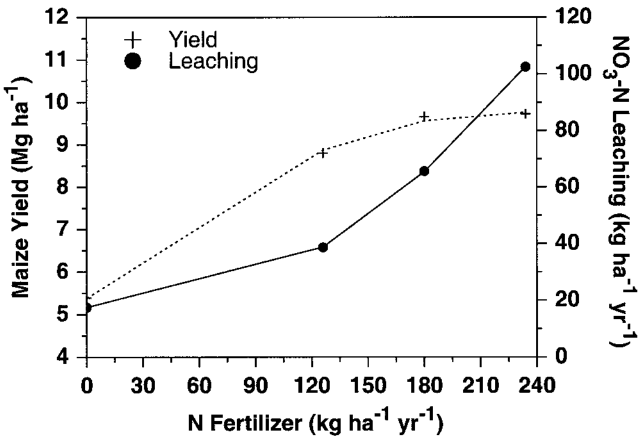

Fig. 5. IBIS simulated average maize yield and NO_3–N leaching as a function of fertilizer N use on the Plano silt loam soil at the Arlington Agricultural Research Station, Arlington, WI.

1997, 2001–2003) than by fertilizer rate (Fig. 3e, 4b). This idea could also be derived from measurements of yield from 1995–2000, where in 1997 poor weather conditions appeared to have a greater effect on yield in the N-fertilized plots compared with the N-unfertilized plots. These results were also depicted by Vanotti and Bundy (1994a), who showed a relatively weak response of maize yield to fertilizer use in 1988 at Arlington, at which time drought conditions persisted during much of the growing season. Simulations also suggested that a 30% reduction would not be significant enough to reduce NO_3–N concentrations below 10 mg L^{-1}, as the annual average during the period was 11.3 mg L^{-1} (Table 5). Small increases in total soil water drainage were simulated as fertilizer use decreased with subsequent small decreases in plant growth and water usage, but the largest annual average increase was only 4.0% in the N-unfertilized case.

It is apparent that a nonlinear relationship exists between changes in fertilizer use and the reduction or increase in NO_3–N leaching losses (Fig. 5). According to our simulations, 30% increases or reduction in fertilizer use will not amount to a corresponding 30% change in NO_3–N leaching. In both cases, the changes in leaching were greater in magnitude than the fertilizer-use change. Additionally, maize yield did not respond linearly with excessive fertilizer N inputs, which has also been shown in previous studies (Vanotti and Bundy, 1994a,b; Pang et al., 1998).

There is a positive correlation between total fertilizer use and the standard error of annual average yield and NO_3–N leaching losses (Table 5). Clearly, the increase in fertilizer use helps attain higher yield, but it also increases the likelihood of higher NO_3–N leaching, and greater year-to-year variability in yield (Table 5). The cumulative probability of simulated NO_3–N losses, drainage water NO_3–N concentration, and maize yield for the 21-yr simulation (1995–2015) are shown in Fig. 6a–c, respectively, for the four applied N fertilizer rates. The effects of more anomalous weather (e.g., 20% cumulative probability is reflective of a colder or drier year, and 80% probability is a warmer year with optimal precipitation) are more evident on yield and NO_3–N losses in simulations that received increasing amounts of N fertilizer. A cumulative probability of 50% refers to median year weather, and the values shown in Fig. 6 at 50% are the median reported in Table 5 for the 21-yr simulation period. Simulations in Fig. 6a show that in 9 out of 10 years (90% cumulative probability), annual NO_3–N leaching could be reduced by 75% if a 30% reduction in fertilizer usage from the optimal rate occurred. Similarly, for the same change, Fig. 6c shows that grain yield loss would only be about 10% at a 90% cumulative probability.

While the CV is only slightly different between the 30% excess and reduction (i.e., from optimum) in fertilizer N use scenarios, meaning that the variability, when normalized by average yield, has not changed, these simulations still depict a trend that has been occurring since the 1950s. The relationship is quite clear: Reliance on N fertilizer has made growers particularly vulnerable to changes in weather, most notably precipitation. Figure 7 shows the average yield (Fig. 7a), 10-yr running average standard error (Fig. 7b), CV (Fig. 7c), and the estimated change in N fertilizer use for the Arlington research station in Columbia Co., WI (Fig. 7d). Observed maize yields were obtained using the National Agricultural Statistics Service (NASS) county yield estimates between 1950 and 2000 (USDA, 2001). While a grower in the 1950s and 1960s may have only experienced year-to-year maize yield variability of approximately 0.1 Mg ha^{-1} in southern Wisconsin, today, the level of variability (approximately 0.4 Mg ha^{-1}) and the likelihood of significant NO_3–N leaching events have increased (Fig. 7).

CONCLUSIONS

The IBIS model was originally designed as a dynamic global ecosystem model, with no representation of managed ecosystems. Future applications of the model in regions such as the U.S. Midwest will require careful and rigorous testing for a variety of cropping systems. The additions made to the model in this study aided in evaluating the water balance, nitrogen fluxes, food production, and responses of natural processes to changes in land management and climate. Field data collected within representative maize ecosystems of the region helped to evaluate model performance for the U.S. upper Midwest.

Although most comparisons between simulated and measured quantities were encouraging, there were several inconsistencies. Simulated annual residue C to N ratio, peak LAI, yield, harvest index, N grain removal, and plant N uptake all generally met the criteria of 20% or less error, and had comparable CVs. Somewhat larger variability (i.e., 25–60%) existed in measured quantities of net N mineralization, soil inorganic N, residual soil NO_3–N, and NO_3–N leaching and concentration.

The IBIS model can be a valuable predictive tool. However, more research will be needed to analyze seasonal variations in biochemical equilibrium between stored inorganic soil N and soil solution N. Model treat-

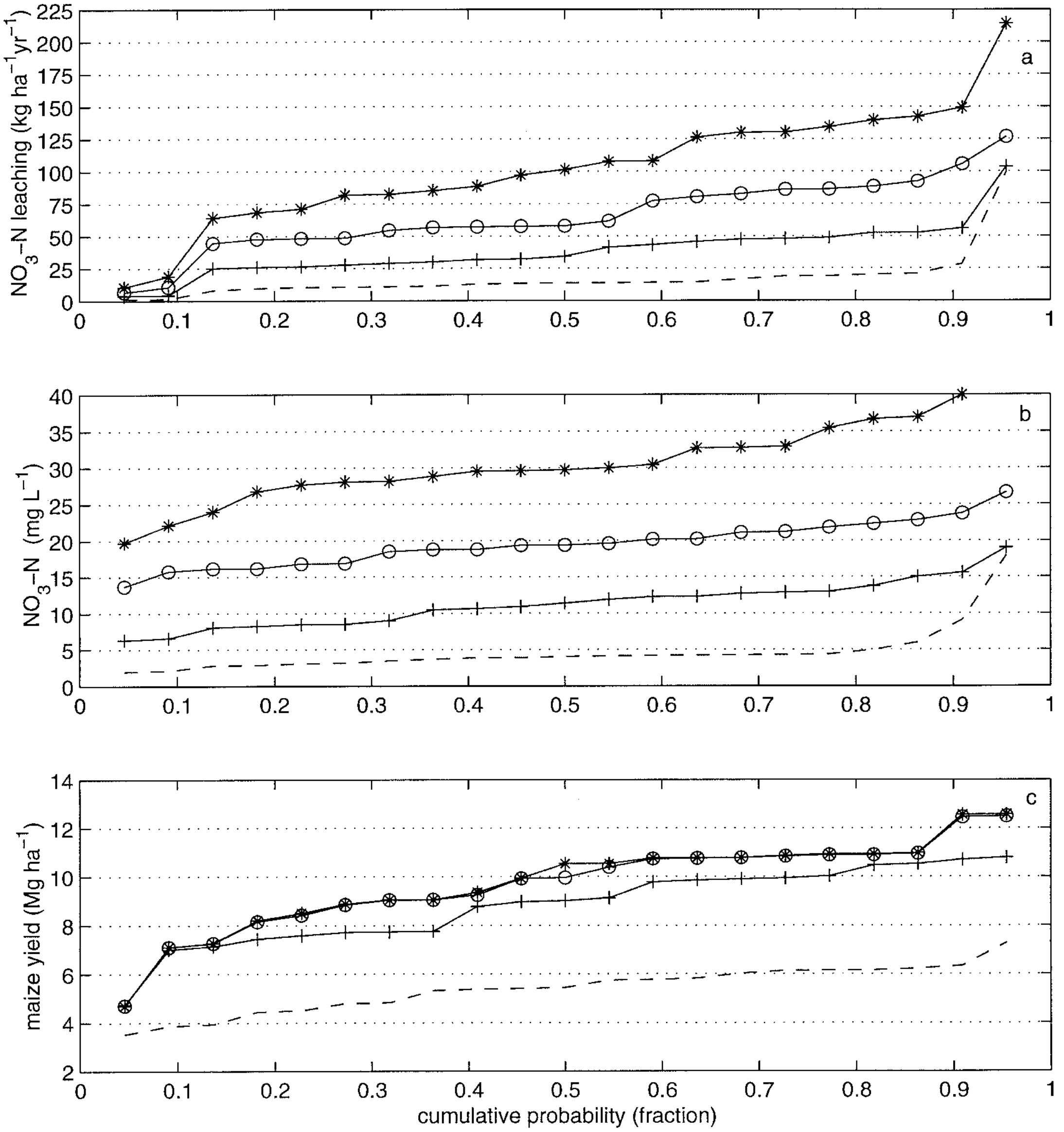

Fig. 6. IBIS simulated cumulative probabilities of (*a*) annual NO_3–N losses at a 1.4-m soil depth, (*b*) flow-weighted annual mean NO_3–N concentration at a 1.4-m soil depth, and (*c*) maize yield on the Plano silt loam soil at the Arlington Agricultural Research Station, Arlington, WI. For all figures, * is 234 kg N ha^{-1}, + is 180 kg N ha^{-1}, — is 126 kg N ha^{-1}, and ○ is N unfertilized.

ment of the partitioning of NO_3–N between solution and soil-bound pools, which is a modifiable process with one empirical parameter, was the likely cause of simulated errors in NO_3–N leaching rather than the drainage flux. The partitioning of total soil inorganic N between the soil solution and that which is soil-bound or immobilized immediately upon applying fertilizer N is probably complex and dynamic. Future model improvements will need to assess how this phenomenon can be linked to other soil processes or quantities such as microbial biomass, soil texture, soil temperature, and soil moisture conditions.

Overall, IBIS demonstrated strong capability to replicate long-term effects of fertilizer N use on maize yield, NO_3–N leaching and concentration, soil water drainage, and other related crop growth quantities. Field measurements collected in both N-fertilized (180 kg N ha^{-1}) and N-unfertilized maize allowed the model to be calibrated so that simulated yield response to soil inorganic N availability was appropriately captured. We used field data for NT maize to compare with CP data and CP simulations to better understand whether the effects of tillage would need to be accounted for in future model revisions. In summary, the largest differences in NT and CP field data were found in yield, drainage, and net N mineralization. Drainage (measured at 1.4 m) in the CP plots was 39% higher than NT over the 6-yr field study (Table 3), and yield was 24% higher in the unfertilized CP case (Table 4). Yield in fertilized CP was only 8% higher than fertilized NT, however, during the six years. There were no significant differences in peak LAI or NO_3–N leaching loss between tillage treatments. However, in fertilized plots, the data showed that net N mineralization in NT was approximately 45% of that

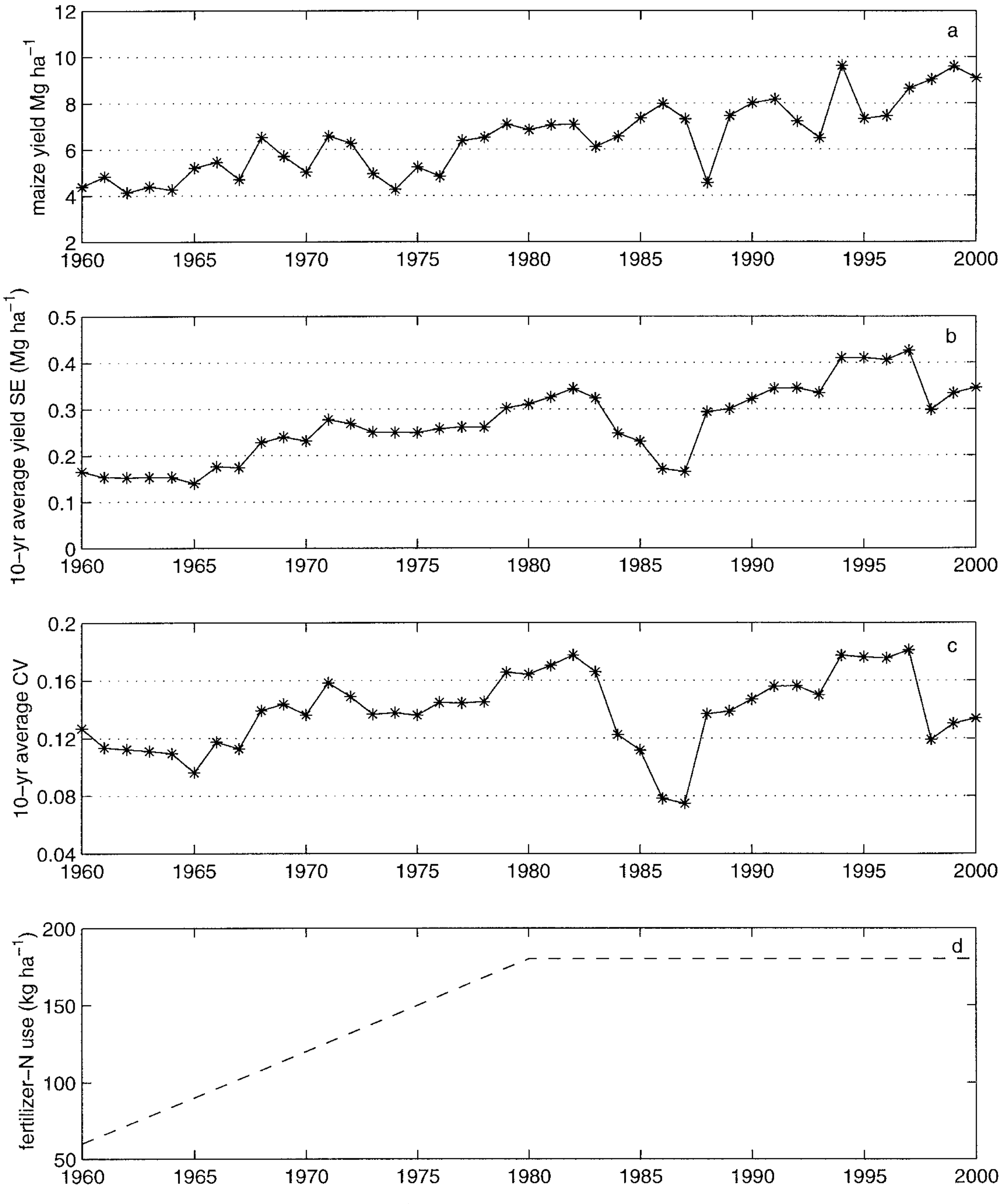

Fig. 7. (*a*) Historical USDA reported average maize yield, (*b*) the computed 10-yr running average standard error of yield, (*c*) the 10-yr running average coefficient of variation of yield, and (*d*) estimated fertilizer N use for Columbia County, Wisconsin. The Arlington Agricultural Research Station is located in the southeastern corner of the county.

occurring in the CP plots. Based on these results, additions of a surface residue layer and accounting for increased soil surface aeration and drainage due to tillage should be added to the model in the future because of the effects these probably have on microbial dynamics near the surface.

It was unknown whether the nonlinear response of leaf N concentration on the photosynthetic capacity of maize in IBIS will adequately represent plant growth at fertilizer rates that are less than optimum, but more than 0 kg N ha^{-1}. Because field data generally do not show a marked increase in yield with N fertilizer application in excess of the optimum, IBIS was originally calibrated to produce maximum measured yields for N usage at 180 kg N ha^{-1}. In this approach, any excess soil inorganic N will not increase plant production under optimal field conditions. Upon comparison with an independent dataset and empirical relationship that related crop yield to fertilizer usage at Arlington, much confidence was gained in simulated responses.

The bottom line resulting from this study was that nonlinear relationships existed between changes in fertilizer use and NO_3–N leaching losses over time. Simulated changes in NO_3–N leaching were generally greater in magnitude than fertilizer-use changes. Altered tillage regimes also had little effect on NO_3–N leaching, but had significant effects on yield in unfertilized plots. We suggest that future field experiments be designed so that the effects of altered N fertilizer use can be addressed, in particular for effects of reduced fertilizer use on yield and NO_3–N. Moreover, our results indicate that the response time of management changes also needs to be quantified. Until then, simulation models will continue to have uncertainty in addressing future effects, particularly in scenarios of future environmental change.

APPENDIX

IBIS Crop Model Description

This appendix briefly details the IBIS crop modeling approach. The maize plant physiology and phenology described in this appendix are generally applicable to Midwest and Great Plains agriculture in the USA. More detailed descriptions of the IBIS modeling framework can be found in Kucharik et al. (2000, 2001).

Weather, Soils, and Planting Date

In the absence of subdaily meteorological driver data, IBIS uses a stochastic weather generator based on Richardson (1981), Richardson and Wright (1984), and Geng et al. (1985) in combination with additional empirical equations that describe diurnal cycles (Campbell and Norman, 1998) to derive meteorological quantities (temperature, precipitation, radiation, wind speed, relative humidity) from monthly mean datasets. The interrelationships between the random variations in atmospheric parameters and their persistence over time are determined from a serial autocorrelative approach (Kucharik et al., 2000). A fully adjustable multilayer soil formulation is used to simulate the diurnal and seasonal variations of heat and moisture in the top 2 m (Kucharik et al., 2000). At any time step, each layer is described in terms of soil temperature, volumetric water content, and ice content (Pollard and Thompson, 1995; Foley et al., 1996). The Richards equation is used to calculate the time rate of change of liquid soil moisture, and the vertical flux of water is modeled according to Darcy's law (Campbell and Norman, 1998). The soil water budget is controlled by the rate of infiltration, evaporation of water from the soil surface, the transpiration stream originating from plants, and redistribution of soil water. Soil texture is classified into 1 of 11 categories (based on sand and clay fractions; Gerakis and Baer, 1999) so that soil physical and hydraulic properties can be assigned according to Rawls et al. (1992) and Campbell and Norman (1998).

Crop planting date can either be prescribed or determined by comparing 10-d running averages of daily mean air temperature ($T_{\mathrm{ave},10}$) and minimum temperature ($T_{\mathrm{min},10}$) to thresholds needed to plant. In addition, planting cannot take place before 1 April based on the typical growing season across the eastern USA since $T_{\mathrm{ave},10}$ must usually be greater than 10°C and $T_{\mathrm{min},10}$ must be greater than 3°C. All three conditions described must be met for planting to occur.

Carbon Assimilation, Allocation, and Plant Phenology

The IBIS model uses a mechanistic treatment of canopy photosynthesis (Farquhar et al., 1980; Farquhar and Sharkey, 1982) and a semimechanistic model of stomatal conductance (Ball et al., 1986). Following Collatz et al. (1992), the photosynthesis rate of C_4 plants is determined from three potential capacities to fix carbon. Parameters for maize physiology are found in Table A1. Plant photosynthetic rate is decreased under water-limited conditions. A stress factor between 0.0 (high stress) and 1.0 (no stress) is calculated using the integral of plant-available water content over each soil layer. The contribution of each layer to the overall plant water stress is weighted by the soil layer root fraction. The value for $V_{\max}$ (maximum photosynthetic rate) is adjusted by this stress factor to reduce plant productivity under water stress conditions.

Table A1. Maize physiological parameters used in IBIS model (15°C base).

Parameter	Value
Intrinsic quantum efficiency (dimensionless)	0.05
Leaf respiration coefficient	0.10
Coefficient for stomatal conductance (m)	4.0
Coefficient for stomatal conductance (b)	0.03
Minimum stomatal conductance	1×10^{-5}
Photosythesis coupling coefficients	
θ	0.97
β	0.80
$V_{\max}$, μmol m^{-2} s^{-1}†	70
CO_2 to O_2 specificity ratio	4500.0
O_2/CO_2 kinetic parameters, mol mol^{-1}	
K_c	1.5×10^{-4}
K_o	2.5×10^{-1}

† Maximum photosynthetic rate.

The daily summation of assimilated C is partitioned between leaf, stem, root, or grain, and is dependent on the physiological age of the plant. Crop growth is divided into three basic phases, defined by temperature sums (daily average temperate minus base temperature; Ritchie and NeSmith, 1991; Tollenaar and Dwyer, 1999) similar to the CERES-Maize (Jones and Kiniry, 1986) and EPIC (Sharpley and Williams, 1990; Cabelguenne et al., 1999) models. Thirty-year monthly mean climatology (period 1961–1990) from the Climate Research Unit (CRU) climate database (New et al., 2000) is interpolated to obtain daily average temperatures on a 0.5° terrestrial grid. These data are used to initialize the average growing degree-day (AGDD) summation (8°C base temperature for maize [GDD_8]) during the period 1 April through 30 September, inclusive (hereafter, these average seasonal summations are denoted by $AGDD_8$). This calendar period encompasses the general planting-through-maturity cycle for the eastern two-thirds of the USA. These GDD summations are calculated with temperature limits of 30°C maximum, and 8 or 10°C minimum, depending on crop type (Plett, 1992).

Growth stages for crops are defined as total degree days needed to reach the beginning of each developmental phase (e.g., Cutforth and Shaykewich, 1989; Carlson and Gage, 1989; Hayhoe and Dwyer, 1990). All GDD summations are relative to the crop planting date. Stage one is the period between planting and leaf emergence. Leaf emergence occurs when 3% of $AGDD_8$ for a location is reached (Stewart et al., 1998). In this initial phase, the GDD summation is based on the difference between daily average soil temperature of the top layer and the respective base temperature for each crop type (Table A1). The second phase is the period between leaf emergence and the end of silking when leaf area development is most rapid. The end of Phase 2 occurs at 60% of $AGDD_8$, at which time grain fill begins. Phase 3 encompasses the period of grain fill to physiological maturity (e.g., black layer in maize), which is reached at approximately 95% of AGDD. Plant growth is terminated as soon as the 5-d running mean minimum air temperature reaches 5°C or less, or the crop is harvested.

At the beginning of the growing season, the initial C partitioning to roots is 40% in maize. Remaining carbon allocation

Table A2. Maize crop growth parameters used in the IBIS model.

Parameter	Value
Crop growth constants	
Base temperature for growing degree-day and growth, °C	**8**
Carbon fraction in dry matter (leaf and stem)†	**0.45**
Carbon fraction in grain	**0.45**
Maximum leaf area index, m^2 m^{-2}‡	**5.0**
Maximum crop height, m‡	**2.5**
Leaf expansion constant, °Cd leaf^{-1}§	
Leaves 1–4	**65.0**
Leaves $\geq$ 5	**42.0**
Duration of Phase 1 (planting to emergence)	**0.03**
Duration of Phase 2 (emergence to silking)	**0.57**
Duration of Phase 3 (grain fill to maturity)	**0.40**
Maximum harvest index	**0.61**
Allocation to reproductive tissue that is seed or grain	**1.00**
Specific leaf area, m^2 kg^{-1} dry matter	**7–12**
Carbon allocation	
Initial root C allocation	**0.40**
Final root C allocation	**0.05**
Final leaf C allocation	**0.00**
Final stem allocation	**0.00**
Ratio to leaf to stem C allocation	**60:40**
End of season reproductive allocation	**0.90**
Nitrogen allocation, g N kg^{-1} dry matter	
Maximum N in leaf	**40.0**
Optimum N in leaf (no stress)	**20.0**
Maximum N in grain	**15.0¶**
Optimum grain N concentration	**12.0¶**
Minimum N in leaf and stem	**5.0¶**
Minimum N in roots	**1.0**
Stem to leaf N allocation ratio	**0.50¶**
Root to leaf N allocation ratio	**0.10**

† Penning DeVries et al. (1989).
‡ Sharpley and Williams (1990), Cabelguenne et al. (1999).
§ Carberry et al. (1989), Carberry (1991), Ritchie and Nesmith (1991), Jones and Kiniry (1986).
¶ Muchow (1994), Sinclair and Muchow (1995).

is divided between leaves and stem in a 60:40 ratio for maize (Table A2), partially based on the plant N allocation (Muchow, 1994; Sinclair and Muchow, 1995). Leaf area index changes according to carbon assimilation, allocation, and specific leaf area (e.g., Carberry et al., 1989; Carberry, 1991; Garnier et al., 1997; Birch et al., 1998). As the plant reaches physiological maturity, carbon allocated to fine roots decreases 5% in maize plants as a function of GDD (Sharpley and Williams, 1990). In maize, all reproductive allocation past the end of silking is assumed to go to grain. Harvest index is computed at the end of the growing season as the ratio of carbon allocated to grain to total aboveground biomass. Based on field data, maximum values 0.61 are used for maize (Sharpley and Williams, 1990; Jones and Kiniry, 1986; Cabelguenne et al., 1999). If excess carbon is allocated to grain past these allowable fractions, it is divided equally and allocated to roots and stems. Other crop-specific parameters relating dry matter to assimilated carbon are reported in Table A2.

Nitrogen Cycling and Plant Nitrogen Uptake

Four potential sources of inorganic N available to plants originate from atmospheric deposition, fixation, fertilizer application, and mineralization of soil organic matter. These are balanced by plant uptake, denitrification, and leaching. Deposition of inorganic N is a linear function of daily rainfall amount, and is based on empirical equations in the CENTURY model (Parton et al., 1987). Nitrogen inputs of broadcast fertilizer (single pulse) and atmospheric deposition enter the surface soil layer and are transported through the soil profile as a function of water flow. Plant-rooting profiles are used to partition the total input of inorganic N via N fixation and mineralization to each soil layer. Nitrogen mineralization is computed as a total input to the soil–plant system and is mediated by soil temperature and moisture.

The four crop vegetation compartments (leaf, stem, fine roots, and grain) have optimum N requirements and upper limits that affect how partitioning of plant N uptake occurs (Table A2), but are independent of growth stage. Nitrogen supply to the crop is based on N availability in the soil and plant water uptake. Typical plant tissue N content is based on nominal field measurements at the University of Wisconsin Agricultural Research Station at Arlington and a survey of the literature (Muchow, 1994; Sinclair and Muchow, 1995; Muchow and Sinclair, 1995; Liang et al., 1996; Plenet and Lemaire, 1999). As C is assimilated from the atmosphere, the plant extracts available water and inorganic N from each soil layer independently. The magnitude is dependent on the soil layer root fraction and the availability of inorganic N, and the rate of uptake is a linear function of the plant transpiration stream. Total plant N concentration is calculated from the total accumulated N uptake and total dry matter production. The leaf N concentration is set as the difference between the total plant N and N allocated to grain, stem, and root.

Plants begin to experience nitrogen stress when the leaf N concentration (mass basis) falls below an optimum value of 20 g N kg^{-1}. Nitrogen in fine root and stem biomass is accumulated having fixed ratios to N in leaf biomass. For fine roots, the N concentration is set to 10% of that in leaves, whereas in stems, the ratio is 50% (Muchow, 1994; Sinclair and Muchow, 1995). The maximum allowable N concentration in leaves is 40 g N kg^{-1}, and the minimum requirement is 5 g N kg^{-1} in leaf and stem, and 1 g N kg^{-1} in fine roots (Table A2). If leaf N stress occurs, plant photosynthesis is adjusted accordingly. The value of $V_{\max}$ (e.g., 70 μmol m^{-2} s^{-1} for maize) is multiplied by the most limiting stress factor, either N or water stress.

Grain is produced with a constant N concentration of 12 g N kg^{-1} as long as sufficient N is available to meet other plant N requirements. If excess N is available, grain N concentration is allowed to increase up to 15 g N kg^{-1} (Muchow, 1994; Sinclair and Muchow, 1995). During the growing season, the plant requires that vegetative tops have a C to N ratio of approximately 100 or less during growth (5 g N kg^{-1}; Muchow, 1994; Sinclair and Muchow, 1995). Less N will be allocated to the grain pool (12 g N kg^{-1}) to help meet this requirement late in the growing season, which can result in grain N levels less than 10 g N kg^{-1}, depending on soil inorganic N levels.

The crop model for maize in this version of IBIS has been calibrated with 1990s yield data available from the USDA National Agriculture Statistics Service (NASS) website and field data collected at the University of Wisconsin Agricultural Research Station at Arlington, WI (Brye, 1999). Generally, N fertilizer inputs of 160 to 220 kg ha^{-1} minimize N deficiencies for maize across the upper Midwest and Great Plains soils (Vanotti and Bundy, 1994a,b) under favorable soil moisture conditions.

Solute Transport

A convective transport model is used to simulate the movement of NO_3–N between soil layers in response to water fluxes. Plant-available N is partitioned between a temporarily immobile quantity of stored soil N and a mobile quantity of soil solution N, which is available to leach through the profile. However, both pools are accessible by the plant for N uptake. Because NH_4–N is rapidly fixed by plants and taken up through the transpiration stream, we track only inorganic NO_3–N in soil solution and subsequent leaching through the profile. An equilibrium factor (0.10) is applied to stored soil NO_3–N to maintain a reasonable fraction of NO_3–N in soil solution at all times (Brye, 1999). As water moves between soil layers (hourly timestep), soil solution NO_3–N is transported with the water flux. After NO_3–N either leaves or enters into a soil layer, the NO_3–N partitioning between the immobile pool and soil solution is readjusted. At each timestep, mass balance is maintained through the summation of additions (mineralization, fertilization, deposition) and losses (leaching and plant uptake).

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