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Integration of nitrogen dynamics into the Noah-MP land surface model v1.1 for climate and environmental predictions

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Abstract. Climate and terrestrial biosphere models consider nitrogen an important factor in limiting plant carbon uptake, while operational environmental models view nitrogen as the leading pollutant causing eutrophication in water bodies. The community Noah land surface model with multi-parameterization options (Noah-MP) is unique in that it is the next-generation land surface model for the Weather Research and Forecasting meteorological model and for operational weather/climate models in the National Centers for Environmental Prediction. In this study, we add a capability to Noah-MP to simulate nitrogen dynamics by coupling the Fixation and Uptake of Nitrogen (FUN) plant model and the Soil and Water Assessment Tool (SWAT) soil nitrogen dynamics. This model development incorporates FUN’s state-of-the-art concept of carbon cost theory and SWAT’s strength in representing the impacts of agricultural management on the nitrogen cycle. Parameterizations for direct root and mycorrhizal-associated nitrogen uptake, leaf retranslocation, and symbiotic biological nitrogen fixation are employed from FUN, while parameterizations for nitrogen mineralization, nitrification, immobilization, volatilization, atmospheric deposition, and leaching are based on SWAT. The coupled model is then evaluated at the Kellogg Biological Station – a Long Term Ecological Research site within the US Corn Belt. Results show that the model performs well for environmental predictions.

Introduction

Over the past several decades, eutrophication — high concentrations of nutrients in freshwater bodies leading to severe oxygen depletion from the resultant algal blooms — has become a worldwide problem facing river, lake, and coastal waters (Conley et al., 2009; Howarth et al., 2006). As one of the greatest threats to freshwater and coastal ecosystems, eutrophic conditions lower biotic diversity, lead to hypoxia and anoxia, increase the incidence and duration of harmful algal blooms, and change ecological food webs that reduce fish production (Diaz and Rosenberg, 2008; National Research Council, 2000). These eutrophic conditions are attributed to excessive fertilizer leaching in river basins (Boesch et al., 2009; Boyer et al., 2006). To complicate this further, climate variation and climate change also determine the variation of hypoxia extent (Donner and Scavia, 2007); higher temperatures may extend the thermal stratification period and deepen the thermocline, thereby resulting in the upwelling of nutrients from sediment and increasing the concentration of nutrients in bottoms of water in lakes (Komatsu et al., 2007). Further, higher precipitation produces more runoff, and very likely more nutrients are delivered to the ocean as well (Donner and Scavia, 2007).

Nitrogen (N) is recognized as the leading nutrient causing eutrophication. Without human interference, N cycling is relatively slow, as most ecosystems are efficient at retaining this in-demand nutrient. N enters soils regularly either through atmospheric wet and dry deposition or through atmospheric N2 fixation by microorganisms (occurring mostly in legume plants). N taken up by plants is confined to relatively slow processes (e.g., growth, decay, and mineralization); in some regions or during the growing season, N may also limit plant growth, which reduces carbon sequestration over land (Fisher
et al., 2012). In addition, N cycling plays a crucial role in nitrous oxide (N₂O), which is considered one of the important greenhouse gases responsible for climate warming. These facts make the N cycle important for studying the response of the climate to the elevated greenhouse gas concentrations. With human tillage of soils, mineralization and nitrification of N are amplified, which results in the reduction of N storage in soil (Knops and Tilman, 2000; Scanlon et al., 2008). In addition, a large amount of N fertilizer is applied in specific areas within a short period of time; as a result, a massive excess of N is leached to the aquatic systems through discharge and erosion, which contributes to the eutrophication in aquatic systems.

Many of these N processes have been included in land surface, hydrologic, and water quality models developed particularly for environmental, climate, and agricultural applications (Bonan and Levis, 2010; Dickinson et al., 2002; Fisher et al., 2010; Kronvang et al., 2009; Schoumans et al., 2009; Thornton et al., 2007; Wang et al., 2007; Yang et al., 2009). These developments are still in their infancy, and large-scale climate models lack N leaching parameterizations that are comparable to those used in water quality models. Thus, large-scale models are not feasible for inherently fine-scale applications such as agricultural fertilization management and water quality prediction. Therefore, the present study improves these weaknesses by incorporating the strength of agriculture-based models into large-scale land surface models (LSMs).

The community Noah LSM with multi-parameterization options (Noah-MP) (Niu et al., 2011; Yang et al., 2011) is used as an exemplar of LSMs because it is the next-generation LSM for the Weather Research and Forecasting (WRF) meteorological model (Rasmussen et al., 2014) and for the operational weather and climate models in the space-time, temperature, and soil moisture – it is logical to augment this scheme with N limitation and realistic plant N uptake and fixation. The state-of-the-art vegetation N model is the Fixation and Uptake of Nitrogen (FUN) model of Fisher et al. (2010), which is embedded into the Joint UK Land Environment Simulator (JULES) (D. B. Clark et al., 2011) and the Community Land Model (CLM) (Shi et al., 2016). Modeling the impacts of agricultural management (e.g., fertilization use) on N leaching is the strength of the Soil and Water Assessment Tool (SWAT) (Neitsch et al., 2011). Therefore, this study incorporates into Noah-MP both FUN’s strength in plant N uptake and SWAT’s strength in soil N cycling and agricultural management.

Our objective is to develop and utilize a land surface modeling framework for simultaneous climate (carbon) and environmental (water quality) predictions. We first describe the nitrogen dynamic model which combines equations used in FUN and SWAT. We then focus on evaluating the new integrated model at a cropland site, because fertilizer application on croplands globally contributes approximately half of the total N input to soil, with the other half coming from natural processes (i.e., atmospheric deposition and biological N fixation) (Fowler et al., 2013; Gruber and Galloway, 2008). Furthermore, cropland is a major source of N loading in water bodies. We evaluate the new model against observed soil moisture content, concentration of soil nitrate, concentration of nitrate leaching from soil bottom, and annual net primary productivity (NPP). We then analyze the impacts of the addition of N dynamics on the carbon and water cycles. To guide the use of this model on regional scales, we also analyze the impacts from different fertilizer application scenarios. Finally, we discuss other model behaviors, i.e., N uptake from different pathways and the major soil nitrate fluxes.

2 Models, data, and methods

2.1 Noah-MP

The Noah-MP model was augmented from the original Noah LSM with improved physics and multi-parameterization options (Niu et al., 2011; Yang et al., 2011), based on a state-of-the-art multiple-hypothesis framework (M. P. Clark et al., 2011). Noah-MP provides users with multiple options for parameterization in leaf dynamics, canopy stomatal resistance, soil moisture factor for stomatal resistance, and runoff and groundwater. Until this work, Noah-MP did not include any N dynamics. The only N-related parameterization is in the calculation of the maximum rate of carboxylation (\( V_{\text{max}} \)), an important factor in estimating the total carbon assimilation (or photosynthesis) rate (Niu et al., 2011):

\[
\frac{\text{leaf biomass}}{\text{mass}} = \frac{\text{NOAA/National Centers for Environmental Prediction. Be}}{\text{land area}}
\]

\[
V_{\text{max}} = V_{\text{max,25}} \alpha_v^{10}
\]

\[
f(\text{N}) = f(T) \beta
\]

which predicts the leaf area index (LAI) as a function of space where \( V_{\text{max,25}} \) is the maximum carboxylation rate at 25°C (\( \mu\text{mol} \text{ CO}_2 \text{ m}^{-2} \text{s}^{-1} \)), \( \alpha_{\text{max}} \) is a temperature-sensitive parameter, \( f(T) \) is a function that mimics the thermal breakdown of metabolic processes, and \( f(N) \) is a foliation nitrogen factor (\( f(N) = 1 \)), and \( \beta \) is the soil moisture controlling factor. Since there were no N dynamics in the model, \( f(N) \) was set as a constant 0.67, which translates to a constant 33% of \( V_{\text{max}} \) down-regulation due to N stress. This factor was originally used in Running and Coughlan (1988) and adapted into LSMs by Bonan (1991).

Our modifications to the original Noah-MP mainly concern the sub-models dealing with dynamic leaf and subsurface runoff. The dynamic leaf option is turned on to provide NPP and biomass to the newly coupled N dynamic sub-model. In the original Noah-MP model, subsurface runoff from each soil layer was not an explicit output, but it is now a new output in the updated model. However, N concentrations...
are different among soil layers, which affects the amount of N removed from each soil layer by subsurface runoff. There- fore, in conjunction with the runoff scheme options 1 (TOPMODEL with groundwater) and 2 (TOPMODEL with an equilibrium water table), the lumped subsurface runoff for all four layers is first calculated, and then the water is removed from each soil layer weighted by hydraulic conductivity and soil layer thickness.

2.2 Nitrogen dynamics

In Noah-MP, the soil N model structure is the same as in SWAT, which includes five N pools consisting of two inorganic forms (\(\text{NH}_4^+\) and \(\text{NO}_3^-\)) and three organic forms (active, stable, and fresh pools). The N processes employed from SWAT are mineralization, decomposition, immobilization, nitrification, denitrification, and atmospheric deposition. The N processes employed from FUN are uptake and symbiotic biological N fixation, which can be further divided into active and passive soil N uptake, leaf N retranslocation, and symbiotic biological N fixation. Figure 1 shows the flow chart of the nitrogen dynamic model. In this section, we describe the core equations. The full description for plant N uptake and soil N dynamics is available in Fisher et al. (2010) and Neitsch et al. (2011), respectively. Table 1 shows the model input variables and parameters. Most of these parameters use the values recommended by Fisher et al. (2010) and Neitsch et al. (2011), while some of them are adjusted to best repre- sent the site condition and hence match site observation. The important adjusted parameters include the \(v_{\text{ Merry}}\) (threshold value of soil water factor for denitrification to occur), \(\beta_{\text{ min}}\) (rate coefficient for mineralization of the humic organic nitrogen), and \(\beta_{\text{ max}}\) (rate coefficient for mineralization of the fresh organic nitrogen in residue).

2.2.1 Nitrogen uptake and fixation

Plant N uptake and fixation follow the framework of Fisher et al. (2010), which determines N acquired by plants through Eq. (3), advection (passive uptake); Eq. (4), symbiotic bi- space

\[
N_{\text{ passive}} = N_{\text{ soil}} \frac{E_T}{s_d}
\]

where \(N_{\text{ soil}}\) is the available soil N for the given soil layer (kg N m\(^{-2}\)), \(E_T\) is transpiration rate (m s\(^{-1}\)), and \(s_d\) is the soil water depth (m). This pathway is typically a minor con- tributor except under very high soil N conditions.

If \(N_{\text{ passive}}\) is less than \(N_{\text{ demand}}\), then the remaining required N must be obtained from retranslocation \(N_{\text{ resorb}}\), active uptake \(N_{\text{ active}}\), or biological N fixation space.
space(\(N_{\text{fix}}\), kg N m\(^{-2}\)), all of which are associated with energetic cost and hence require C expenditure (C cost). The C costs of fixation (\(\text{Cost}_{\text{fix}}\), kg C kg N\(^{-1}\)), active uptake (\(\text{Cost}_{\text{active}}\), kg C kg N\(^{-1}\)), and resorption (\(\text{Cost}_{\text{resorb}}\), kg C kg N\(^{-1}\)) are calculated:

\[
\text{space}(\text{Cost}_{\text{fix}}, \text{Cost}_{\text{active}}, \text{Cost}_{\text{resorb}}) = \text{space}(\text{Cost}_{\text{fix}} \cdot \text{Cost}_{\text{active}} \cdot \text{Cost}_{\text{resorb}})
\]

(8)

Fresh organic residue is broken down into simpler organic components via decomposition. The plant-unavailable organic N is then converted into plant-available inorganic N via mineralization by microbes. Plant-available inorganic N can also be converted into plant-unavailable organic N via immobilization by microbes.

**Table 4.** Mineralization, decomposition, and immobilization.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Definition</th>
<th>Controlling process</th>
<th>Unit</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>(N_{\text{fix}})</td>
<td>kg N m(^{-2})</td>
<td>Leaf 27, root 45</td>
<td>kg C m(^{-1})</td>
<td>1.0</td>
</tr>
<tr>
<td>(N_{\text{active}})</td>
<td>kg C kg N(^{-1})</td>
<td>Fixation</td>
<td>kg C m(^{-2})</td>
<td>1.0</td>
</tr>
<tr>
<td>(N_{\text{resorb}})</td>
<td>kg C kg N(^{-1})</td>
<td>Fixation</td>
<td>g N m(^{-2})</td>
<td>6.7</td>
</tr>
<tr>
<td>(T_{\text{soil}})</td>
<td>deg K</td>
<td>Fixation</td>
<td>g N m(^{-2})</td>
<td>12.4</td>
</tr>
<tr>
<td>(\gamma)</td>
<td>kg C kg N(^{-1})</td>
<td>Fixation</td>
<td>g N m(^{-2})</td>
<td>5.3</td>
</tr>
<tr>
<td>(\beta)</td>
<td>kg C kg N(^{-1})</td>
<td>Fixation</td>
<td>g N m(^{-2})</td>
<td>0.76</td>
</tr>
<tr>
<td>(\phi)</td>
<td>kg C kg N(^{-1})</td>
<td>Fixation</td>
<td>kg N m(^{-2})</td>
<td>1.3</td>
</tr>
<tr>
<td>(\rho)</td>
<td>kg C kg N(^{-1})</td>
<td>Fixation</td>
<td>1.4</td>
<td></td>
</tr>
</tbody>
</table>

Note: some parameters are not described in the paper. The values for \(K_{\text{acq}}\) and \(p_{\text{b}}\) are for the four soil layers.
The nutrient-cycling water factor for soil layer ly, \( \gamma_{\text{sw,ly}} \), is calculated as follows:

\[
\gamma_{\text{sw,ly}} = \frac{s_{\text{ly}}}{\theta_{\text{ly}}}, \quad (10)
\]

where \( s_{\text{ly}} \) is the water content of soil layer ly (mm H\(_2\)O) and \( \theta_{\text{ly}} \) is the water content of soil layer ly at field capacity (mm H\(_2\)O).

The mineralized N from the humus active organic N pool, \( N_{\text{min,ly}} \) (kg N m\(^{-2}\)), is calculated as follows:

\[
space \Theta_{\text{fr,vol,ly}} = 1 - \exp[-\eta_{\text{vol,ly}}], \quad (17)
\]

\[
space \Theta_{\text{fr,nit,ly}} = 1 - \exp[-\eta_{\text{nit,ly}}]. \quad (18)
\]

### 2.2.4 Denitrification

Denitrification is the process of bacteria removing N from soil (converting NO\(_3^–\) to N\(_2\) or N\(_2\)O gases). Denitrification rate, \( N_{\text{denit,ly}} \) (kg N m\(^{-2}\)), is calculated as follows:

\[
N_{\text{denit,ly}} = \frac{\text{NO}_{3,\text{ly}} \cdot \left[1 - \exp(-\beta_{\text{denit}} \cdot \gamma_{\text{mp,ly}} \cdot \text{orgC}_{\text{ly}})\right]}{\gamma_{\text{sw,ly}}} \quad \text{if} \quad \gamma_{\text{sw,ly}} \geq \gamma_{\text{sw,thr}} \quad \text{(19)}
\]

\[
N_{\text{min,ly}} = \beta_{\text{min,ly}} \cdot \gamma_{\text{mp,ly}} \cdot \text{orgC}_{\text{ly}} \quad \text{if} \quad N_{\text{denit,ly}} = 0
\]

\[
N_{\text{min,ly}} = \frac{\text{orgC}_{\text{ly}} \cdot \gamma_{\text{mp,ly}}}{\gamma_{\text{sw,ly}}} \quad \text{if} \quad \gamma_{\text{sw,ly}} \geq \gamma_{\text{sw,thr}}
\]

\[
N_{\text{min,ly}} = 0.8 \cdot \delta_{\text{ntr,ly}} \cdot N_{\text{fon,ly}}, \quad (11)
\]

where \( \beta_{\text{min}} \) is the rate coefficient for mineralization of the humus active organic nutrients and \( N_{\text{fon,ly}} \) is the amount of N in the active organic pool (kg N m\(^{-2}\)).

The mineralized N from the residue fresh organic N pool, \( N_{\text{min,ly}} \) (kg N m\(^{-2}\)), is calculated as follows:

\[
N_{\text{min,ly}} = 0.8 \cdot \delta_{\text{ntr,ly}} \cdot N_{\text{fon,ly}}, \quad (12)
\]

where \( \delta_{\text{ntr,ly}} \) is the residue decay rate constant, and \( N_{\text{fon,ly}} \) is the amount of N in the fresh organic pool (kg N m\(^{-2}\)).

The decomposed N from the residue fresh organic N pool, \( N_{\text{dec,ly}} \) (kg N m\(^{-2}\)), is calculated as follows:

\[
N_{\text{dec,ly}} = 0.2 \cdot \delta_{\text{ntr,ly}} \cdot N_{\text{fon,ly}}
\]

### 2.2.5 Atmospheric deposition

While the mechanism of atmospheric deposition is not fully understood, the uncertainty is parameterized into the concentration of nitrate/ammonium in the rain for wet deposition, and the nitrate/ammonium deposition rate for dry deposition.

The amounts of nitrate and ammonium added to the soil through wet deposition, NO\(_3\)\(_{\text{vol}}\) (kg N m\(^{-2}\)) and NH\(_4\)\(_{\text{vol}}\) (kg N m\(^{-2}\)), are calculated as follows:

\[
NO_{3\text{vol}} = 0.01 \cdot R_{NO_3} \cdot P, \quad (20)
\]

\[
NH_{4\text{vol}} = 0.01 \cdot R_{NH_4} \cdot P, \quad (21)
\]

where \( R_{NO_3} \) is the concentration of nitrate in the rain (mg N L\(^{-1}\)), \( R_{NH_4} \) is the concentration of ammonium in the rain (mg N L\(^{-1}\)), and \( P \) is the amount of precipitation. The values for \( R_{NO_3} \) and \( R_{NH_4} \) used in this study are listed in Table 1.

### 2.2.6 Fertilizer application

The N fertilizer application process is included in the new model as well. If real fertilizer application data (timing and amount for a specific year) are available, they can be used as model inputs. Otherwise, a fixed amount of N fertilizer (e.g., 7.8 g N m\(^{-2}\) yr\(^{-1}\) in this study) is applied at a fixed time of a year (e.g., 20 June in this study).

### 2.2.7 spaceLeaching

N leaching from land to water bodies is a consequence of soil weathering and erosion processes. In particular, organic N attached to soil particles is transported to surface water through soil erosion. Therefore, the modified universal soil loss equation (USLE) (Williams, 1995) is used to determine...
N in nitrate form can be transported with surface runoff, lateral runoff, or percolation, which is calculated as follows:

\[
\text{NO}_3_{\text{surf}} = \beta_{\text{NO}_3} \cdot \text{conc}_{\text{NO}_3,\text{mobile}} \cdot Q_{\text{surf}},
\]

\[
\text{NO}_3_{\text{lat},\text{ly}} = \beta_{\text{NO}_3}
\]

space\text{ 1.2 m of soil depth, is available from 1995 to 2013. These two measurements are used to evaluate model-simulated concentrations of soil nitrate for the top 25 cm and nitrate leach- ing from the soil bottom. Soil N mineralization, which mea- sures the net mineralization potential and is available from 1989 to 2012, is compared with the modeled mineralization rate qualitatively.

In addition, soil moisture content is sampled from the sur- face to 25 cm soil depth and is calculated on a dry-weight basis. In order to compare with model output, it is converted to volumetric soil moisture by applying the soil bulk density. Annual NPP is converted from annual crop yields (1989–2013) by assuming a harvest index and a root-to-whole-plant ratio for each crop type. The harvest indices for corn, soy- bean, space\text{bean,}

\[
Q_{\text{lat},\text{ly}} = \text{spaceconc}_{\text{NO}_3,\text{mobile}} \cdot \text{conc}_{\text{NO}_3,\text{mobile}} \cdot Q_{\text{surf}},
\]

for top layer \text{NO}_3_{\text{lat},\text{ly}} = \text{spaceconc}_{\text{NO}_3,\text{mobile}} \cdot \text{conc}_{\text{NO}_3,\text{mobile}} \cdot \text{surf}

\]

soil erosion. The details of the calculation are described in Neitsch et al. (2011).

This site features multiple N-related measurements. Soil inorganic N concentration, which is sampled from the surface to 25 cm soil depth, is available from 1989 to 2012. Concentra- tion of inorganic N leaching at bedrock, which is sampled spaceWest et al., 2010). Although N uptake cannot be evaluated directly at this site, by evaluating the annual NPP, we can see the model’s performance in representing the N limitation effect on plant growth.

Noah-MP requires the following atmospheric forcing data at least at a 3-hourly time step: precipitation, air temperature, specific humidity, surface air pressure, wind speed, incom- ing solar radiation, and incoming longwave radiation. The weather station at the site measures all of these except for incoming longwave radiation, but it does not cover the entire period from 1989 to 2014 (e.g., hourly precipitation data are only available since 2007), when the N data are available. Therefore, atmospheric forcing data are extracted from the 0.125° x 0.125° gridded forcing data from the North American Land Data Assimilation System (NLDAS; Xia et al., 2012). Table 2 compares the atmospheric forcing data be- tween NLDAS and site measurements for 2008–2014. We can see that the differences in precipitation and air temperature – the two most important forcing fields for N cycling – are very small, with relative biases 0.4 and 4.2 %, respectively.

Finally, the site management log records the detailed oper- ational practices such as soil preparation, planting, fertilizer application, pesticide application, and harvest. N fertilizer application data include the date of application, rate, fertil- izer type, and equipment used. The fertilizer application date and rate are used as model inputs.

\[\text{Precipitation} (\text{mm})\]

\[\text{Air Relative Pressure} (\text{hPa})\]

\[\text{Source}\]

\[\text{www.geosci-model.net/9/1/2016/}\]

\[\text{Comparison of annual averaged atmospheric forcing data (2008–2014) between site observation and NLDAS.}\]

\[\text{Table 2.}\]

\[\text{3 Results and analyses}\]

\[\text{3.1 Evaluation of soil moisture}\]

Modeled volumetric soil moisture, which is important for nu- trient cycling and plant growth, is compared to measured soil moisture (Fig. 2). The model performs reasonably well on both treatments (i.e., with and without tillage) in terms of capturing the mean and seasonal variation, which is consis-
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Table 1. Observed and simulated soil moisture from 1989 to 2012 for (a) treatment 1 – cropland with conventional tillage – and (b) treatment 2: cropland without tillage. The grey shaded area shows the observational ranges from up to six replicates for each treatment.

<table>
<thead>
<tr>
<th></th>
<th>Site obs.</th>
<th>NLDAS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature (°C)</td>
<td>937.19</td>
<td>924.45</td>
</tr>
<tr>
<td>Humidity (%)</td>
<td>9.15</td>
<td>9.55</td>
</tr>
<tr>
<td>Site obs.</td>
<td>73.44</td>
<td>76.50</td>
</tr>
</tbody>
</table>

3.2 Evaluation of soil nitrate

Soil nitrate concentration is the outcome of all N-related processes that occur in soil such as decomposition, mineralization, nitrification, denitrification, and uptake. It determines the available N that plants can use. The skills in modeling the soil nitrate concentration reflect the overall performance of the model in simulating the N cycle. Figure 3 shows the comparison of the model-simulated soil nitrate concentration with site observations for both T1 and T2. The model captures the major variations of the soil nitrate. N fertilizer application is responsible for the high peaks. These high peaks drop very fast at first and then drop slowly, which can sustain crop growth for a few months.

The multi-year average of modeled soil nitrate concentration is 5.77 mg kg\(^{-1}\) (4.90 mg kg\(^{-1}\)) for T1 (T2), which is consistent with the observed 5.61 mg kg\(^{-1}\) (4.81 mg kg\(^{-1}\)). Correlation coefficients are 0.58 and 0.56 for T1 for T2, respectively. From the wide spread of the range error bars, we space...
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N leaching can be transported to rivers through surface and subsurface runoff and to groundwater through percolation from soil bottom. Only the last pathway is measured at this site. Figure 4 shows the comparison of concentrations of the leached solution from the soil bottom between model simulation and observation. The averaged concentration of N leaching from the soil bottom for T1 (T2) is 12.84 mg kg$^{-1}$ (8.86 mg kg$^{-1}$) from model simulation and 13.57 mg kg$^{-1}$ (9.26 mg kg$^{-1}$) from observation. The correlation coefficients are 0.43 and 0.40 for T1 and T2, respectively. Although these skills may not be considered satisfactory, the model can still produce comparable results with observation.

The peak in 2003 is extremely high and long lasting. This is probably due to the abnormal pattern of precipitation distribution in 2003. In a normal year, storms higher than 50 mm usually occur in either summer or fall. However, in 2003, there was an early storm on 4 April which reached 61 mm in 1 day. As we can see from Fig. 3, the soil nitrate concentration from 1989 to 2011 for (a) treatment 1 – cropland with conventional tillage – and (b) treatment 2: cropland without tillage. The grey shaded area shows the observational ranges from up to six replicates for each treatment.

Table 3. Annual averages of Noah-MP-simulated major nitrogen fluxes and NPP. The NPP within the parentheses is from observation.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>NPP (gC m$^{-2}$)</th>
<th>Uptake</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Passive (gN m$^{-2}$)</td>
<td>Active (gN m$^{-2}$)</td>
</tr>
<tr>
<td>T1</td>
<td>432 (437)</td>
<td>6.18</td>
</tr>
<tr>
<td>T2</td>
<td>441 (471)</td>
<td>6.62</td>
</tr>
</tbody>
</table>

Figure 3. Observed and model-simulated soil nitrate concentration from 1989 to 2011 for (a) treatment 1 – cropland with conventional tillage – and (b) treatment 2: cropland without tillage. The grey shaded area shows the observational ranges from up to six replicates for each treatment.

space

can see that soil N dynamics may be affected by a variety of complicated factors, which makes it difficult to model. Therefore, although the correlation coefficients are not considered high skills relative to the soil moisture statistics, they are still reasonable.

While both treatments show very similar patterns (Fig. 3), T1 with conventional tillage tends to have higher soil nitrate concentration. This is understandable because tillage practices redistribute water and nutrients in soil, which accelerates the N cycling. Table 3 shows annual averages of major N fluxes for both treatments. T1 has higher rates of humus mineralization and residue decomposition, but, at the same time, it also has higher rates of denitrification and leaching. Therefore, it produces more N$_2$O (a greenhouse gas) and more N runoff to rivers. Particularly, with higher N leaching, less soil nitrate is available for passive uptake by plants. As a result, plants need to acquire more N through active uptake.

space

3.3 Evaluation of nitrate leaching from soil bottom


Figure 4. Observed and model-simulated nitrate leaching from bottom of soil profile from 1995 to 2013 for (a) treatment 1 – cropland with conventional tillage – and (b) treatment 2: cropland without tillage. The grey shaded area shows the observational ranges from up to six replicates for each treatment.

space

The peak in 2003 is extremely high and long lasting. This is probably due to the abnormal pattern of precipitation distribution in 2003. In a normal year, storms higher than 50 mm usually occur in either summer or fall. However, in 2003, there was an early storm on 4 April which reached 61 mm in 1 day. As we can see from Fig. 3, the soil nitrate concentration from 1989 to 2011 for (a) treatment 1 – cropland with conventional tillage – and (b) treatment 2: cropland without tillage. The grey shaded area shows the observational ranges from up to six replicates for each treatment.

space

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We also notice that, without tillage, N leaching is about one-third lower than that with tillage. Without tillage, the temporal variation is also smaller.

### 3.4 Evaluation of annual NPP

NPP indicates the amount of C that is assimilated from the atmosphere into plants and thus is important in studying not only crop and ecosystem productivity but also climate change feedbacks. NPP is mainly determined by plant photosynthesis and autotrophic respiration. It is also affected by water and nutrient stresses. In this study, N stress on plant growth is calculated by the reduction of NPP due to N acquisition, which can be considered another form of plant respiration. Figure 5 shows the comparison of simulated annual NPP against observation. Since the original Noah-MP without N dynamics also simulates NPP, its results are also shown here as a reference. The mean annual NPP simulated by the original Noah-MP is 544 gC m\(^{-2}\) (the same simulation for both treatments as original Noah-MP does not distinguish tillage and no tillage). By including the N dynamics, simulated annual NPP is reduced to 432 gC m\(^{-2}\) (441 gC m\(^{-2}\)) for T1 (T2), which is more consistent with observed 437 gC m\(^{-2}\) (471 gC m\(^{-2}\)). The correlation coefficient increased to 0.77 for T1, and from 0.30 to 0.72 for T2, which is a significant improvement. This improvement is due to the better characterization of the amount of carbon allocated to N acquisition instead of growth.

The modeled rate of NPP down-regulation – the fraction of NPP reduction due to N limitation – is 35.4 and 34.7 % for T1 and T2, respectively. These rates are close to the 33 % of down-regulation rate used in the default Noah-MP. By dynamically simulating the demand and supply of N with time, these become even closer to the observations.

Surprisingly, even with slower N cycling, T2 produces slightly higher NPP (Table 3), which is consistent between model and observation. If this is the case, except for drying up soil, releasing more N\(_2\)O gas, and producing more N leaching, is there any benefit from tillage? The answer is yes. Less N fertilizer is needed for cropland with tillage. Based on the site management log, in total there was 194.8 gN m\(^{-2}\) of N fertilizer applied to T2 during the same period.

### 3.5 Impacts of nitrogen dynamics on carbon cycle

The coupling of the N dynamics into Noah-MP not only adds N-related modeling but also affects other components of the model, i.e., the carbon and water cycles. This is because the change in NPP affects leaf biomass and hence LAI.

---

**Figure 5.** Observed and modeled annual NPP from 1989 to 2013 for (a) treatment 1 – cropland with conventional tillage – and (b) treatment 2: cropland without tillage. The error bars show the observational ranges from up to six replicates for each treatment.
applied after late June and thus plants encountered high N stress during the first half of the growing season.

3.6 Impacts of nitrogen dynamics on water cycle

Through the changes in LAI and soil organic matters (SOMs), the addition of N dynamics affects not only the carbon cycle but also the water cycle. The change in SOM is not currently considered, and therefore the impacts on the water cycle are from the change in LAI only, as shown in Fig. 7. These impacts are most pronounced on plant transpiration, which is increased by 33 mm yr\(^{-1}\). The increase mostly occurs during and after the peak growing season. In Cai et al. (2014a), Noah-MP-simulated evapotranspiration (ET) over croplands increases too fast during the first half of the growing season and reaches peak about 1 month earlier than observation. The delayed peaks of LAI and ET can partly mitigate this issue. As there is more water extracted from soil by transpiration, soil moisture further decreases during the second half of the growing season. Therefore, less water is available and thus soil evaporation is decreased by 9 mm yr\(^{-1}\). With the increase in ET, runoff is decreased by 13 mm yr\(^{-1}\).

Therefore, besides the great implications for C modeling and the potential for being used in environmental predictions, the addition of N dynamics can improve the hydrological simulations as well.

Figure 6. (left column) Monthly and (right column) climatologically seasonal cycle of model-simulated (a) LAI, (b) NPP, (c) GPP, and (d) NEE from default Noah-MP and enhanced Noah-MP with N dynamics. The values in the right column indicate annual mean for each term (black: Noah-MP without N dynamics; red: Noah-MP with N dynamics).

Figure 7. Same as Fig. 6 except for (a) soil moisture, (b) transpiration, (c) soil evaporation, and (d) runoff.

3.7 Impacts of nitrogen fertilizer application

Observed N fertilizer application data are used in this study. However, this type of data is not always available, especially when models are applied in large regions. Often we only know the approximate amount of N fertilizer applied, without information on the exact dates. To guide the future large-scale application of this model, two additional experiments are run: (1) N fertilizer is applied on 20 June every year, and
space(2) N fertilizer is applied on 29 June every year. The first experiment is designed because in this site a large amount of N fertilizer is applied mostly during mid-June and early July. Other dates are also reported in the literature; therefore, we use 15 April as another example. Both experiments use the same amount of N fertilizer as in the management log, which on average is 7.8 g N m⁻² yr⁻¹.

Figure 8 shows comparison of some of the most relevant results between the two experiments and the one (real) with space

**Figure 8.** (left column) Monthly and (right column) climatologically seasonal cycle of model-simulated (a) NPP, (b) N uptake, (c) N leaching, and (d) soil nitrate with different dates for N fertilization: real, 20 June, and 15 April. The values in the right column indicate annual mean for each term (black: real; red: 20 June; blue: 15 April). (e) Actual nitrogen fertilizer application amounts and dates as recorded in the agronomic log.

recorded dates and amount of N fertilizer application. Despite the different application time, the two experiments produce very consistent NPP with the real case. The 20 June experiment is much closer to the real case; even the seasonal variation is identical. The largest discrepancy is in 1993 and 1996. Based on the management log, in these two years, a large amount of N fertilizer was applied, which resulted in much higher NPP than results from the two experiments. Since 15 April is much earlier than the primary fertilizer application dates, NPP from this experiment is flattened out through the year. This also confirms the statement in Sect. 3.5 that later N fertilizer applications delay plant growth. Simulated N uptake from both experiments shows exactly the same story as NPP.

The simulated N leaching shows the opposite pattern to NPP. The default simulation produces the highest leaching, followed by the 20 June experiment and then the 15 April experiment. This is very likely because the fertilizer application dates are closer to the flood season for the former two cases and the chance of fertilized N being flushed out is higher. The difference in N fertilization dates also clearly affects the simulations of total soil nitrate. In the 20 June experiment, soil nitrate reaches the lowest level in May because no N fertilizer is applied before 20 June. In the default case, N fertilizer can actually be applied as early as April, but with a smaller amount before mid-June, which prevents the soil nitrate concentration from getting too low. Besides a large amount of N fertilizer applied in later months, the other reason that the default simulation reaches the highest concentration of soil nitrate is because it produces higher NPP, which can be returned to soil for decomposition.

Overall, the default simulation grows better plants (higher NPP) because N fertilizer is applied based on expert judgment of plants’ demand. At the same time, however, it produces more N leaching than the two experiments, which is significant (insignificant) with respect to the 15 April (20 June) experiment at 90% confidence level. The experiment with closer dates of N fertilizer application to reality can better reproduce the N dynamics in observation. Therefore, although we cannot always know the exact dates of N fertilizer application, a survey on this can help to improve model simulation.

**Figure 9.** Daily climatology (1989–2013) of nitrogen uptake by pathways expresses as (a) actual amount of uptake and (b) percentage of total uptake.

3.8 Analysis of nitrogen uptake

As described in Sect. 2.2.1, plants can get N for growth from four pathways: passive uptake, active uptake, fixation, and

space


space

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References


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\[ V_{\text{max}} = \frac{25}{T_{\text{d}}} \times a_{\text{max}} \]

\[ f(N) = f(T_{\text{d}}) \beta \]

\[ N_{\text{demand}} = \frac{C_{\text{NPP}}}{r_{C:N}} \]

\[ N_{\text{passive}} = \frac{E_{T}}{s_{\text{d}}} \]

where \( C_{\text{NPP}} \) is the carbon:nitrogen ratio of the whole plant, \( r_{C:N} \) is the C:N ratio for each component (leaf, root, and woody), and \( N_{\text{passive}} \) is the plant proportionally to the biomass. C:N ratios for different components of the plant for each vegetation type are given by Oleson et al. (2013).

Because no extra energetic cost is needed, passive nitrogen \( N_{\text{passive}} \) is the first and preferred source for a plant depleting.

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### Table 1. Model input variables and parameters.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>$r_{C:N}$</td>
<td>C : N ratios for each component of the</td>
</tr>
<tr>
<td>$a$</td>
<td>Empirical curve-fitting parameter</td>
</tr>
<tr>
<td>$b$</td>
<td>Empirical curve-fitting parameter</td>
</tr>
<tr>
<td>$c$</td>
<td>Empirical curve-fitting parameter</td>
</tr>
<tr>
<td>$s$</td>
<td>Scaling factor</td>
</tr>
<tr>
<td>$k_N$</td>
<td>Empirical curve-fitting parameter</td>
</tr>
<tr>
<td>$k_C$</td>
<td>Empirical curve-fitting parameter</td>
</tr>
<tr>
<td>$N_{\text{no3}}$</td>
<td>Initial value for NO$_3$ concentration in s</td>
</tr>
<tr>
<td>$N_{\text{non}}$</td>
<td>Initial value for humic organic N in soil</td>
</tr>
<tr>
<td>$N_{\text{fon}}$</td>
<td>Initial value for fresh organic N in soil</td>
</tr>
<tr>
<td>$C_{\text{org}}$</td>
<td>Initial organic carbon content in soil la;</td>
</tr>
<tr>
<td>$\rho_b$</td>
<td>Bulk density of the soil layer</td>
</tr>
<tr>
<td>$f_{\text{nh4,n}}$</td>
<td>Fraction of mineral N in fertilizer that i</td>
</tr>
<tr>
<td>$f_{\text{surfn}}$</td>
<td>Fraction of fertilizer that is applied to t</td>
</tr>
<tr>
<td>$\epsilon_{\text{mix}}$</td>
<td>Mixing efficiency of tillage operation</td>
</tr>
<tr>
<td>$\beta_{\text{min}}$</td>
<td>Rate coefficient for mineralization of t;</td>
</tr>
<tr>
<td>$\beta_{\text{frd}}$</td>
<td>Rate coefficient for mineralization of t;</td>
</tr>
<tr>
<td>$\beta_{\text{denit}}$</td>
<td>Rate coefficient for denitrification</td>
</tr>
<tr>
<td>$\gamma_{\text{sw,thr}}$</td>
<td>Threshold value of soil water factor for</td>
</tr>
<tr>
<td>$R_{\text{no3}}$</td>
<td>Concentration of nitrate in the rain</td>
</tr>
<tr>
<td>$R_{\text{nh4}}$</td>
<td>Concentration of ammonium in the rain</td>
</tr>
<tr>
<td>$D_{\text{no3}}$</td>
<td>Constant of nitrate rate with dry deposi</td>
</tr>
<tr>
<td>$D_{\text{nh4}}$</td>
<td>Constant of ammonium rate with dry d</td>
</tr>
<tr>
<td>$\theta_e$</td>
<td>Fraction of porosity from which anions</td>
</tr>
<tr>
<td>$\beta_{\text{no3}}$</td>
<td>Nitrate percolation coefficient</td>
</tr>
</tbody>
</table>

Note: some parameters are not described in the paper. The values for

$(N_{\text{fix}}, \text{kg N m}^{-2})$, all of which are associated cost and hence require C expenditure (C cos of fixation (Cost$_{\text{fix}}, \text{kg C kg N}^{-1}$), active upt kg C kg N$^{-1}$), and resorption (Cost$_{\text{resorb}}, \text{kg}$ calculated as follows:

\[
\text{Cost}_{\text{fix}} = s \{ \exp[a + b \cdot T_{\text{soil}} \cdot (1 - 0.5 \cdot T_{\text{soil}})] \}
\]

\[
\text{Cost}_{\text{active}} = \left( \frac{k_N}{N_{\text{soil}}} \right) \left( \frac{k_C}{C_{\text{root}}} \right),
\]

\[
\text{Cost}_{\text{resorb}} = \frac{k_R}{N_{\text{leaf}}}.
\]
Immobilization is incorporated into mineralization (net mineralization). Mineralization, which are only allowed to occur when the temperature is above 0°C, are constrained by water availability. The nutrient-cycling temperature layer \( \gamma_{\text{tmp,ly}} \) is calculated as follows:

\[
\gamma_{\text{tmp,ly}} = 0.9 \cdot \frac{T_{\text{soil,ly}}}{T_{\text{soil,ly}} + \exp[9.93 - 0.312 \cdot T_{\text{soil,ly}}]} +
\]

where \( T_{\text{soil,ly}} \) is the temperature of soil layer \( ly \).

The nutrient-cycling water factor for soil layer \( ly \) calculated as follows:

\[
\gamma_{\text{sw,ly}} = \frac{\theta_{ly}}{\theta_{s,ly}},
\]

where \( \theta_{ly} \) is the water content of soil layer \( ly \) and \( \theta_{s,ly} \) is the water content of soil layer \( ly \) at (mm H\(_2\)O).

The mineralized N from the humus active organic matter \( N_{\text{mina,ly}} \) (kg N m\(^{-2}\)), is calculated as follows:

\[
N_{\text{mina,ly}} = \beta_{\text{mina,ly}} \left( \gamma_{\text{tmp,ly}} \cdot \gamma_{\text{sw,ly}} \right)^{1/2} \cdot N_{\text{aon,ly}},
\]

where \( \beta_{\text{mina}} \) is the rate coefficient for mineralization of humus active organic nutrients and \( N_{\text{aon,ly}} \) is N in the active organic pool (kg N m\(^{-2}\)).

The mineralized N from the residue fresh matter \( N_{\text{minf,ly}} \) (kg N m\(^{-2}\)), is calculated as follows:

\[
N_{\text{minf,ly}} = 0.8 \cdot \delta_{\text{ntr,ly}} \cdot N_{\text{fon,ly}},
\]

where \( \delta_{\text{ntr,ly}} \) is the residue decay rate constant.

The decomposed N from the residue fresh matter \( N_{\text{dec,ly}} \) (kg N m\(^{-2}\)), is calculated as follows:

\[
N_{\text{dec,ly}} = 0.2 \cdot \delta_{\text{ntr,ly}} \cdot N_{\text{fon,ly}}.
\]

### 2.2.3 Nitrification and ammonia volatilization

Using a first-order kinetic rate equation, the

\[
\begin{align*}
\text{NO}_3,_{\text{surf}} &= \beta_{\text{NO}_3} \cdot \text{conc}_{\text{NO}_3,\text{mobile}} \cdot Q_{\text{surf}}, \\
\text{NO}_3,_{\text{lat,ly}} &= \beta_{\text{NO}_3} \cdot \text{conc}_{\text{NO}_3,\text{mobile}} \cdot Q_{\text{lat,ly}} \quad \text{for top layer} \\
\text{NO}_3,_{\text{lat,ly}} &= \text{conc}_{\text{NO}_3,\text{mobile}} \cdot Q_{\text{lat,ly}} \quad \text{for lower layers} \\
\text{NO}_3,_{\text{perc}} &= \text{conc}_{\text{NO}_3,\text{mobile}} \cdot w_{\text{perc,ly}},
\end{align*}
\]

\[\text{www.geosci-model.net/9/1/2016/}\]