# Impact of transient soil water simulation to estimated nitrogen leaching and emission at high- and low-deposition forest sites in Southern California

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[1] Soil water dynamics and drainage are key abiotic factors controlling losses of atmospherically deposited N in Southern California. In this paper soil N leaching and trace gaseous emissions simulated by the DAYCENT biogeochemical model using its original semi-dynamic water flow module were compared to that coupled with a finite element transient water flow module (HYDRUS), for two mixed conifer forests with annual deposition rates of about 70 and 9 kg N ha$^{-1}$, in the San Bernardino National Forest. Numerical solution of the Richards equation implemented in HYDRUS water module could improve response of surface soil water dynamics to precipitation pattern, compared to the original, and consequently it resulted in annual N gaseous emission loss about 1.5 ~ 2 times higher. While the two flow modules predicted similar amounts of annual water drainage, the HYDRUS water module simulated more frequent, but smaller drainage fluxes, which favors soil mineralization and downward transport. In normal precipitation years, annual leaching losses predicted by the HYDRUS coupled DAYCENT model was about 5–18 kg N ha$^{-1}$ higher due to different temporal patterns of daily water drainage. In dry and wet years, leaching losses were similar. Our analysis suggests that it is necessary to fully capture dynamics of transient water flow (e.g., by numerically solving the transient Richards equation) in order to adequately estimate soil N gaseous emissions and N transport and thus leaching, although it requires more computational resources while the uncertainty in model improvement is still large due to lack of measurements.

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## 1. Introduction

[2] In southern California, natural ecosystems in regions downwind of Los Angeles and nearby agricultural areas suffer from heavily polluted air, which produces the "N saturation" phenomenon due to the highest atmospheric N deposition rates in the United States [*Fenn et al.*, 1996, 2003]. Besides field investigations on the fate of anthropogenic N additions in these ecosystems, the modeling approach has been used [e.g., *Arbaugh et al.*, 1999; *Fenn et al.*, 2008]. The modeling adopted for these forest ecosystems is similar to applications in agricultural systems [*Chen et al.*, 2008; *Del Grosso et al.*, 2005, 2006, 2009], and grasslands [*Parton et al.*, 2001]. Although this sort of modeling approach has been widely used to increase our understanding of the fate of N-fertilizer applications in these managed systems, it is less known how it performs when applied to natural ecosystems, such as forests under continuous atmospheric N deposition.

[3] It is well known that the movement and losses of excess soil N depend upon soil water dynamics, because water flow is the primary soil solute transport pathway [*Feyen et al.*, 1998]. Additionally, soil water-filled pore space (WFPS) is a primary factor controlling N trace gas production [*Dobbie and Smith*, 2003] and soil water contents near saturation favor primary N trace gas emissions [e.g., *del Prado et al.*, 2006], although aerobic conditions may produce non-ignorable trace gaseous N from soil nitrification in these systems. Therefore a reliable description of soil water content and fluxes in both wetting and drying processes is necessary in order to adequately represent the N cycle in soil-plant systems. What, however, are the appropriate spatial and temporal scales that need to be considered to sufficiently represent important water processes in biogeochemical models for various purposes [*Chen et al.*, 2008]? This problem is especially critical for the estimation

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of gaseous emissions, which are highly variable in time; and even pulsed for sudden wetting or drying events during and after rainfall or snowmelt [e.g., *Frolking et al.*, 1998; *Smith et al.*, 1998; *Stolk et al.*, 2011].

[4] It has been shown that dynamic soil water models based on the numerical solution of the Richards equation can adequately simulate transient water flow and thus soil water dynamics [*de Jong and Bootsma*, 1996; *Ranatunga et al.*, 2008; *Scanlon et al.*, 2002], and these approaches have been extensively applied in land surface models [*Shao and Henderson-Sellers*, 1996; *Lee and Abriola*, 1999] and agricultural and environmental modeling systems [*Ranatunga et al.*, 2008; *Gu and Riley*, 2010; *Gu et al.*, 2009]. The disadvantage of the Richards equation based water flow models is that they require knowledge of unsaturated soil hydraulic properties, which are rarely available, and daily or even shorter time steps and fine soil horizon discretization for numerical solution of the Richards equation [*Lee and Abriola*, 1999]. These approaches require significant computational resources and more data inputs as well, and thus are not preferred in complex ecosystem models for regional or global application or for effective and exhaustive analysis of soil, water, plant, and climatic interactions on multiple biogeochemical processes [*de Jong and Bootsma*, 1996; *Rodriguez-Iturbe et al.*, 2007]. On the other hand, the commonly used tipping bucket models are too simple to represent soil water dynamics [*de Jong and Bootsma*, 1996; *Shao and Henderson-Sellers*, 1996]. Thus semi-dynamic water models were extensively developed to allow for a robust representation of soil water dynamics and fast computation in such circumstances, e.g., DSSAT [*Jones et al.*, 2003; *Ritchie*, 1998], DAYCENT [*Parton et al.*, 1998], and EU-Rotate_N [*Doltra and Muñoz*, 2010]. These models adopted the field capacity concept from a tipping-bucket model, but allow multiple soil layers with more realistic and mechanical modifications regarding infiltration, soil internal water flow, bottom drainage, and evapotranspiration in order to catch the dynamic features of infiltration (during rainfall or snowmelt) and redistribution of soil water. *Pachepsky et al.* [2006] and *Doltra and Muñoz* [2010] demonstrated that these water flow models could achieve similar accuracy as Richards equation based models for simulation of soil water fluxes. However, the concept of field capacity is not only ambiguous in its definition for the purpose of mathematical modeling [*Twarakavi et al.*, 2009]; it also essentially states that soil water above this point would drain from the soil, which is true when soil water flow reaches steady state in a period of a few days upon soil internal flow [*Ritchie* 1998]. It implies that soils are either fully saturated such as after an intense rainfall, or at or below field capacity [e.g., *Doltra and Muñoz*, 2010, Figure 1]. This assumption could be inappropriate for estimating N trace gas emissions, since simulated soil water content would rarely be in ranges close to saturation, a condition favoring anaerobic N gaseous emissions during and after rainfall or snowmelt [*del Prado et al.*, 2006]. This approach may also produce a very different drainage pattern than those based on the solution of the Richards flow equation, especially at smaller time scales [*Scanlon et al.*, 2002] or different ecosystems than those in the work by *Pachepsky et al.* [2006].

[5] There exist many modeling studies that coupled transient water dynamics and complicated N biogeochemical processes in order to investigate water effects on N biogeochemical cycles especially for gaseous N emission relevant processes due to its high temporal variability [e.g., *Gu and Riley*, 2010; *Gu et al.*, 2009; *Hendriks et al.*, 2007; *Stolk et al.*, 2011]. It appears that model improvements of N emission process responding to water conditions and soil conditions are still in need. On the other hand, *Doltra and Muñoz* [2010] investigated effects of transient water dynamic model (in HYDRUS-2D) and semi-dynamic water flow module in EU-Rotate_N model in N-fertilized vegetable fields, showing that N uptake difference resulted from evapotranspiration simulation could cause observable difference on soil nitrate and leaching losses. However, it is still not clear to what degree the important biogeochemical processes (e.g., other soil N processes besides N uptake) would be affected at different temporal scales if biogeochemical models are coupled with either a Richards equation based water model or a simple tipping-bucket or semi-dynamic water model alone. Analysis on such an issue would be useful to decide what sort of soil water or hydrological algorithms should be included in biogeochemical or ecological models.

[6] In this study soil excess N losses (both leaching and emission) simulated by a well-established biogeochemical model (DAYCENT [*Parton et al.*, 2001]) in mixed-coniferous forest ecosystem under substantial N deposition will be investigated, by implementing a numerical solution of the one-dimensional (1D) transient soil water flow equation (based on the Richards equation, from HYDRUS-1D [*Šimůnek et al.*, 2008]) into it as an alternative to its original water flow module [*Parton et al.*, 1998], which can be categorized as a semi-dynamic tipping bucket model [*de Jong and Bootsma*, 1996]. Comparisons of results produced by these two model versions, hereafter referred to as DAYCENT models with the "original" and "HYDRUS" water flow module, were conducted with regards to effects on simulated soil N accumulation and leaching and emission losses. The objective is to illustrate the importance of adequately simulating transient soil water dynamics and flows to estimate gaseous N emissions and N leaching and other N processes.

## 2. Materials and Methods

### 2.1. Site Description

[7] The San Bernardino National Forest (SBNF) is located east of Los Angeles, California. It is a mixed conifer forest, with dominant tree species ponderosa (*Pinus ponderosa* Laws) and Jeffrey pine (*P. jeffreyi* Grev. and Balf.). Logging has not occurred for many years and forest fires have been suppressed since the early 1900s. Nitrogen deposition monitoring has demonstrated that in the San Bernardino Mountains there is an N deposition gradient from west to east and south with decreased deposition, with increasing distance from the urban region. Precipitation declines along this gradient as well [*Fenn et al.*, 2003]. In this study, a high N deposition site, Camp Paivika (CP), on the western end of this gradient, and a relatively low deposition site, Barton Flats (BF), 46 km east of CP, were chosen. The current N deposition levels at CP and BF were estimated at 70 and 8.8 kg N $ha^{-1}$ $yr^{-1}$ [*Fenn et al.*, 2008], of which about 5.4 (CP) and 2.4 (BF) kg N $ha^{-1}$ $yr^{-1}$ occur in the form of wet deposition (M. E. Fenn, unpublished data).

[8] The soils are generally sandy loam in texture, with 70–72% sand, 8–9% silt, and 19–21% clay for CP, and 66–69%

**Table 1.** Soil Physical Properties, Soil Organic Matter, and Root Fraction for Soil Layers Required by DAYCENT, Based on the Measurement of *Grulke et al.* [1998] at Experimental Sites Camp Paivika, Barton Flats, and James Mountain Reserves [*Vargas and Allen*, 2008] in or Near the San Bernardino National Forest, California

| Layers (cm) | Bulk Density ($g\ cm^{-3}$) | Silt | Sand | Clay | Soil Organic Matter (%) | Root Fraction |
|---|---|---|---|---|---|---|
| | *Camp Paivika* | | | | | |
| 0–1, 1–4 | 1.32 | 0.09 | 0.70 | 0.21 | 0.21 | 0.6 |
| 4–15 | 1.43 | 0.09 | 0.71 | 0.20 | 0.20 | 0.2 |
| 15–30, 30–45 | 1.53 | 0.09 | 0.72 | 0.19 | 0.19 | 0.1 |
| 45–60, 60–75, 75–90, 90–105 105–120 | 1.55 | 0.09 | 0.72 | 0.19 | 0.10 | 0.1 |
| | *Barton Flats* | | | | | |
| 0–1, 1–4 | 1.47 | 0.12 | 0.66 | 0.22 | 0.22 | 0.6 |
| 4–15 | 1.53 | 0.09 | 0.67 | 0.24 | 0.25 | 0.2 |
| 15–30 | 1.58 | 0.08 | 0.69 | 0.23 | 0.24 | 0.1 |
| 30–45, 45–60, 60–75, 75–90, 90–105, 105–120 | 1.59 | 0.08 | 0.69 | 0.23 | 0.10 | 0.1 |
| | *James Mountains Reserve* | | | | | |
| 0–2, 2–8, 8–16 | 1.20 | 0.10 | 0.83 | 0.07 | 0.08 | 0.9 |
| 16–26, 26–36, 36–46 | 1.30 | 0.15 | 0.82 | 0.03 | 0.04 | 0.1 |

sand, 8–12% silt, and 22–25% clay for BF [*Grulke et al.*, 1998]. For applying DAYCENT, it was suggested that the surface 15 cm soil layers are divided into 3 thin layers (e.g., 0–1, 1–4, and 4–15 cm in this study) in order to simulate the rapid changes of near surface water and temperature conditions, which are critical to gaseous N emissions [*Parton et al.*, 1998]. The soil physical properties, SOM, and root fractions of the soil profiles needed by the models are shown in Tables 1 and 2.

[9] In order to assess how well the newly incorporated soil water module can simulate soil moisture dynamics, a model simulation was conducted and compared with soil water observations at the University of California James San Jacinto Mountains Reserve (http://www.jamesreserve.edu) located in the San Jacinto Mountains in Riverside County, California [*Kitajima et al.*, 2010]. The site is an old-growth stand of California semiarid mixed conifer forest, surrounded by the San Bernardino Mountains. Soils are shallow Entisols (Mixed, mesic, shallow Typic Xeropsamment) with a loamy-sand texture and underlay by weathered granitic bedrock. The soil water content was measured using soil moisture ECHO probes (ONESET®, S-SMA-M005) every 5–15 min at 2, 8, and 16 cm depth since the fall 2004 [*Allen et al.*, 2007; *Kitajima et al.*, 2010]. Weather data such as air temperature and precipitation are also monitored in situ or nearby station. In this study soil water content during period of 2006–2009 was compared, with measured or derived soil properties at the site (Tables 1 and 2).

[10] The two DAYCENT model versions are compared with regards to soil solution nitrate concentration at CP and BF sites. In 2000 at each site 6 soil pits were dug deep enough, and 3 lysimeter soil solution collectors were installed at a depth of 1 m. The lysimeter collectors were horizontally put into tunnels of the pit walls, following standard procedure to be sure the lysimeter being contacted with soils well. From 2001–2004, the soil solutions were extracted 13 times at CP and 11 times at BF with at least one effective sample (i.e., enough water volume for record and analysis), for which then the $NH_4^+$-N and $NO_3^-$-N concentrations were analyzed. The $NH_4^+$-N in these samples was minor and can be neglected,

**Table 2.** Soil Bulk Density and Texture (see Table 1) Derived Hydraulic Properties and Parameters for the 10 Layers Used by DAYCENT ($\theta$r, $\theta$wp, $\theta$fc, $\theta$s, Ks) and HYDRUS ($\theta$r, $\theta$s, Ks, $\alpha$, n, l) at Experimental Sites Camp Paivika, Barton Flats, and James Mountain Reserves in or Near the San Bernardino National Forest, California

| Layers (cm) | $\theta$r[a] | $\theta$wp[a] | $\theta$fc[a] | $\theta$s[a] | Ks[b] | $\alpha$, n, l[c] |
|---|---|---|---|---|---|---|
| | | | *Camp Paivika* | | | |
| 0–1, 1–4 | 0.061 | 0.170 | 0.280 | 0.450 | 57.71 | 0.0296, 1.292,0.5 |
| 4–15 | 0.060 | 0.151 | 0.240 | 0.416 | 46.87 | 0.0381, 1.320, 0.5 |
| 15–30, 30–45 | 0.054 | 0.130 | 0.212 | 0.389 | 40.48 | 0.0417, 1.348, 0.5 |
| 45–60, 60–75, 75–90, 90–105 105–120 | 0.052 | 0.125 | 0.202 | 0.371 | 34.24 | 0.0423, 1.357, 0.5 |
| | | | *Barton Flats* | | | |
| 0–1, 1–4 | 0.058 | 0.155 | 0.250 | 0.404 | 31.80 | 0.0326, 1.302, 0.5 |
| 4–15 | 0.058 | 0.164 | 0.254 | 0.386 | 24.80 | 0.0332, 1.290, 0.5 |
| 15–30 | 0.055 | 0.151 | 0.233 | 0.369 | 24.15 | 0.0370, 1.308, 0.5 |
| 30–45, 45–60, 60–75, 75–90, 90–105, 105–120 | 0.055 | 0.144 | 0.223 | 0.366 | 23.90 | 0.0386, 1.319, 0.5 |
| | | | *James Mountains Reserve* | | | |
| 0–2, 2–8, 8–16 | 0.019 | 0.099 | 0.153 | 0.475 | 274.4 | 0.0399, 1.678, 0.5 |
| 16–26, 26–36, 36–46 | 0.020 | 0.060 | 0.123 | 0.461 | 303.9 | 0.0456, 1.661, 0.5 |

[a]$\theta$r, $\theta$wp, $\theta$fc and $\theta$s are soil residual water content, wilting point, field capacity, and saturated water content, respectively.
[b]Ks is soil saturated hydraulic conductivity ($cm\ day^{-1}$).
[c]$\alpha$ ($cm^{-1}$), n, l, are parameters of the van Genuchten type analytical functions.

so only the $NO_3^-$-N concentrations are compared to our model simulations of deep soil nitrate concentration averaged over soils from 60–120 cm depths. So the model may be evaluated with regard to its ability to capture deep soil nitrate accumulation with time, rather than N transport, which needs estimation of water flow.

### 2.2. Overview of DAYCENT's N Biogeochemical Module

[11] The DAYCENT biogeochemical model [*Del Grosso et al.*, 2000, 2001; *Parton et al.*, 1998, 2001] had been applied to simulate plant growth, litter-fall, soil organic matter dynamics and the tightly coupled soil-plant N cycle. The model had also been successfully applied to estimate N trace gas emissions for agricultural systems across the United States [*Del Grosso et al.*, 2006] and globally [*Del Grosso et al.*, 2009]. An important feature of DAYCENT is its N cycling algorithm, which considers $N_2O$, $NO_x$, and $N_2$ emissions resulting from nitrification and denitrification [*Parton et al.*, 1988a, 2001], and detailed soil N mineralization-immobilization processes associated with various organic matter pools (including microbial), which are distinguished by their decomposition rates [*Parton et al.*, 1988b]. In this study, the model was calibrated mainly by modifying parameters relevant to tree growth, N adsorption by litter, and litter and SOM decomposition rates in order to achieve reasonably well-fitted outputs with observations of tree biomass and allocation, litter and SOM accumulation, and C/N ratios as previously reported [*Fenn et al.*, 2008]. Otherwise, for other N processes it was not calibrated and parameters used those already included in the model as reported by *Parton et al.* [1998] and others [e.g., *Del Grosso et al.*, 2002, 2009].

[12] DAYCENT simulates N mineralization from released N bonded with 3 SOM components (active, slow, and passive pools) and metabolic residue of litter materials when respired, assuming that C/N ratios remain fixed, while N immobilization from soil inorganic N pools when litter structural residue decomposes [*Parton et al.*, 1988b]. The net mineralized N from SOM decomposition and/or respiration adds into soil inorganic $NH_4^+$-N pools, which is further nitrified into $NO_3^-$-N. Together with N inputs by bio-fixation, deposition, and/or fertilization (N sources), soil $NH_4^+$-N and $NO_3^-$-N can be taken up by roots. The net N addition (i.e., mineralization + sources − uptake) stores in soil excess N pools, which are subjected to downward transport with water flow (even leaching out of root zones with drainage) and gaseous emissions. When water flows between layers, the nitrate portion of flowing water over the upper layer's total water volume transports into the next layers. Although an empirical factor in the DAYCENT model accounting for nitrate transport not associated with water flow may be applied by setting a constant fraction of nitrate moving downward to the lower horizons, we arbitrarily turned this parameter off in order to consider the direct effects of drainage (and internal water flow) by the two contrasting water flow modules (see below) on nitrogen leaching.

[13] DAYCENT's N gas submodel is based on the leaky pipe concept, in which total N gas emissions are proportional to soil N transformation rates (nitrification and denitrification) and soil gas diffusivity determines the ratios of respective N gas species (mainly $N_2O$, $N_2$, and $NO_x$) [*Parton et al.*, 2001]. Soil $NO_3^-$-N and $N_2$ will be final products of nitrification and denitrification, respectively, in both of which $N_2O$ and $NO_x$ will be produced (leaking) as by-products. The model first calculates $N_2O$ emission rates from those two processes, and then the ratio of $NO_x$:$N_2O$ is calculated according to an empirical relationship with gas diffusivity. The ratio of $NO_x$:$N_2O$ in nitrification is also controlled by a pulse multiplier counting for the effect of rain into previous dry soil. In all those N processes (including gas diffusivity), soil WFPS (i.e., soil moisture), and/or soil texture are the major environmental control factors, while soil temperature only affects soil N mineralization and nitrification in the model. The details and parameters refer to the work by *Parton et al.* [2001].

### 2.3. Incorporation of HYDRUS-1D Water Module into DAYCENT and Parameterization

[14] The DAYCENT original soil water module [*Parton et al.*, 1998] has a two-stage process to describe infiltration and redistribution of water. During periods of rainfall or snowmelt, water infiltrates into a surface soil horizon until it is saturated and then flows into the next layer at saturated conductivity (Ks in Table 2). After infiltration, soil water in any horizon above its field capacity will be drained into the next layer and finally drained out of the soil bottom at saturated conductivity as well. Any extra rainfall or snowmelt water is removed as run-off. Following this infiltration/percolation stage, bidirectional unsaturated flux proceeds according to the Darcy equation, which also multiplied by a so-called "damping multiplier" ($dmp_{flux}$) (default 0.000001) [*Parton et al.*, 1998], as following,

$$flux_i = \frac{dmp_{flux} * \left(h_{poti-1} - h_{poti}\right) * av_{condi}}{dist_i} \quad (1)$$

where, $hpot_{i-1}$ and $hpot_i$ are the hydraulic potential (i.e., sum of soil matric potential and gravitational head) of two adjacent layers, i − 1 and i, $av_{condi}$ is the weighted average of unsaturated conductivity of layer (i) and its upper (i − 1) and lower ones (i + 1), and $dist_i$ is the distance of the middle points of two layers. The success of this approach depends upon the carefully designed vertical soil layer discretization and calculation time steps for the two processes [*de Jong and Bootsma*, 1996; *Ranatunga et al.*, 2008]. Practically, in the DAYCENT water module, drainage rates are adjustable using bottom layer's base flow or stormflow factors (ranging from 0 to 1), and internal flow rates by changing "damping multiplier."

[15] In this study, in order to better understand the impact of soil water contents and overall ecosystem water balance on these N processes and its losses, the soil water module of HYDRUS-1D [*Šimůnek et al.*, 2008] was successfully incorporated as an alternative submodule to DAYCENT's two modules for infiltration and redistribution (unsaturated flow). The HYDRUS water module uses the method of linear mass-lumped finite elements to numerically solve the highly nonlinear Richards equation describing both saturated and unsaturated water flow,

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x}\left[K\left(\frac{\partial h}{\partial x} + 1\right)\right] - S \quad (2)$$

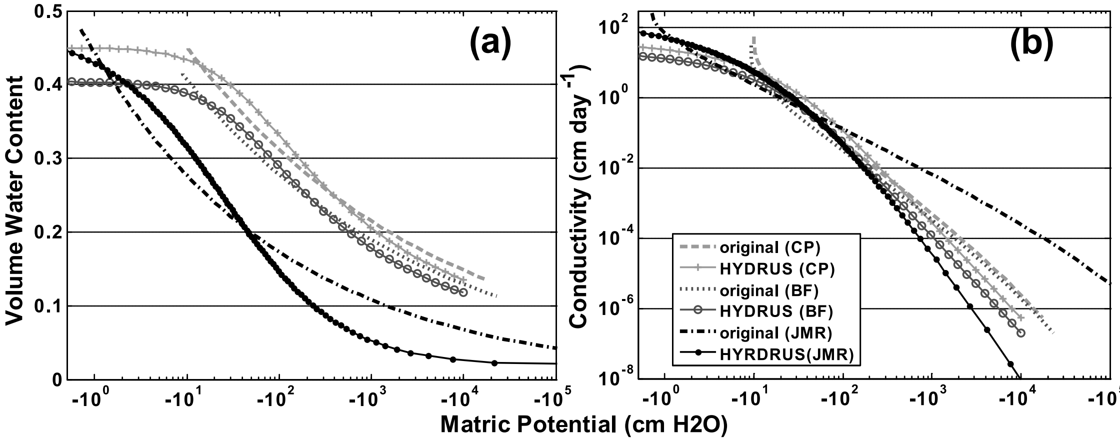


**Figure 1.** Comparison of surface (0–4 or 2 cm) soil hydraulic property functions derived from the soil texture: (a) soil water retention curves, and (b) unsaturated conductivity, used by DAYCENT original and HYDRUS water flow module at Camp Paivika (CP), Barton Flats (BF), and James Mountain Reserves (JMR), in or near the San Bernardino National Forest, California.

where h is the water pressure head, $\theta$ is the volumetric water content, t is the time, x is the distance, S is the water sink/source term, and K is the unsaturated conductivity. Both $\theta$ and K are dependent on pressure head h, which can be expressed in the van Genuchten functions as,

$$\theta(h)\theta_r + \frac{\theta_s - \theta_r}{[1 + |\alpha h|^n]^m} \tag{3a}$$

and

$$k(h) = Ks\ Se^l \Big[1 - \Big(1 - Se^{1/m}\Big)^m\Big]^2 \tag{3b}$$

where $\theta$r and $\theta$s are residue and saturated water content, Se is effective saturation (i.e., = $(\theta - \theta r)/(\theta s - \theta r)$), m = 1 − 1/n, and $\alpha$, n, l are the empirical parameters. All of those independent parameters can be derived from pedotransfer functions [e.g., *Schaap et al.*, 2001; *Šimůnek et al.*, 2008; *Wösten et al.*, 2001]. In this water module, runoff and drainage can be estimated correspondingly, without any empirical adjustment as the original DAYCENT water module.

[16] Both water modules were driven by the same water input and potential evapotranspiration, which were calculated in the original DAYCENT water submodels; then, actual evapotranspiration and soil water content were used to drive plant growth and soil C and N biogeochemical processes. Due to differences in soil water content predicted by the two models, actual evapotranspiration is expected to be different as well, although the potential evapotranspiration and root distribution are identical.

[17] The soil hydraulic property functions, characterizing soil's retention and conductive properties, are different in two water modules. While the DAYCENT water module uses the Saxon-Rawls method, HYDRUS uses the van Genuchten type analytical functions (see equation (3) above) or others (e.g., Brooks and Corey, or the modified van Genuchten type) [see *Šimůnek et al.*, 2008, and references therein]. The bulk density (Table 1), the wilting point, and field capacity were derived based on soil texture and soil organic matter (SOM) using the Saxon-Rawls method [*Saxton et al.*, 1986], except for saturated conductivity (Table 2). The Rosetta module, included in the HYDRUS-1D package, was applied to determine the van Genuchten parameters required by the HYDRUS water module [*Schaap et al.*, 2001; *Šimůnek et al.*, 2008], using soil texture, bulk density, and soil water content at −33 kPa (i.e., field capacity as determined above) (Table 2). In this procedure of estimating soil hydraulic properties, the common parameters for the two modules were forced to be the same in order to allow the hydraulic functions to be as similar as possible so that soil water flow calculation affected by those functions could be minimized, as reported by *Doltra and Muñoz* [2010].

[18] However, the hydraulic functions are still different. For example, while soil water retention curves and hydraulic conductivity functions for the depth of 0–4 cm at CP and BF site intersect at around field capacity and the pressure head of −10 cm (Figure 1), they show relatively large differences near saturation. While this may cause differences in predicted soil water dynamics and movement, it is not possible to predict differences in modeling output by comparing these functions directly. While HYDRUS-1D applies these functions in the entire range of soil water contents in its water modules, DAYCENT uses them only between the residual soil water content and field capacity. Additionally, DAYCENT's water flow rates in the unsaturated flow stage are calculated using Darcy's equation and then multiplied by a so-called "damping multiplier" (default 0.000001) as described above.

## 2.4. Model Simulations

[19] Inputs required by DAYCENT include daily maximum/minimum temperatures and precipitation, current and historical land use, disturbances and/or management, and site-specific soil properties and hydraulic characteristics. For the driving weather data, we used a 65-year daily data set from the weather stations in the Lake Arrowhead area (1942–2006) downloaded from the NOAA National Climate Data Center (http://www.ncdc.noaa.gov/oa/climate/stationlocator.html). Because our two SBNF sites are away from the weather stations in the Lake Arrowhead area, precipitation for CP and BF needs to be extrapolated. We extracted monthly precipitation

**Table 3.** Annual Precipitation and Simulated Drainage, Runoff, Actual Evapotranspiration, and Soil Water Change by HYDRUS or DAYCENT Original Soil Water Module for Years 2001–2005 at Experimental Sites Camp Paivika and Barton Flats in the San Bernardino National Forest, California[a]

| | | Drainage | | Runoff | | Actual Evapotranspiration | | Soil Water Change | |
|---|---|---|---|---|---|---|---|---|---|
| Year | Precipitation | H | D | H | D | H | D | H | D |
| | | | | *Camp Paivika* | | | | | |
| 2001 | 67.2 | 42.8 | 44.4 | 0 | 0 | 18.1 | 15.7 | 6.2 | 7.0 |
| 2002 | 29.7 | 12.7 | 11.9 | 0 | 0 | 19.0 | 19.3 | −2.1 | −1.5 |
| 2003 | 76.6 | 52.5 | 49.4 | 0 | 4.2 | 22.1 | 20.6 | 1.9 | 2.3 |
| 2004 | 29.8 | 9.3 | 11.0 | 0 | 0 | 18.6 | 18.6 | 1.8 | 0.1 |
| 2005 | 141.0 | 116.3 | 123.0 | 0 | 2.3 | 8.6 | 8.1 | 15.9 | 7.5 |
| | | | | *Barton Flats* | | | | | |
| 2001 | 53.8 | 27.4 | 24.8 | 0 | 1.3 | 26.3 | 27.4 | 0.1 | 0.3 |
| 2002 | 15.6 | 0.34 | 0.003 | 0 | 1.5 | 20.2 | 20.1 | −5.0 | −6.1 |
| 2003 | 69.3 | 41.7 | 35.0 | 0 | 6.6 | 20.7 | 17.1 | 6.9 | 10.5 |
| 2004 | 32.1 | 10.7 | 10.4 | 0 | 0 | 24.9 | 25.0 | −3.5 | −3.3 |
| 2005 | 151.5 | 111.9 | 99.3 | 16.4 | 30.1 | 15.3 | 15.7 | 7.5 | 6.3 |

[a]HYDRUS, H; DAYCENT Original, D. Annual precipitation measured from previous June to July. All measurements are in centimeters.

for the two sites from the PRISM database [*Di Luzio et al.*, 2008] and then downscaled to daily values assuming the same time variation pattern as at the Lake Arrowhead station. Annual precipitation averaged 88.2 cm at CP and 59.1 cm at BF, with more than 70% of precipitation falling from December to March. Since both sites exhibited a Mediterranean climate, the yearly cycle considered in this paper started in July and ended the following June. For the period from 2001–2005, the annual precipitation (from previous July to June) was 67.2, 29.7, 76.6, and 29.8, and 141.0 cm at CP, and 53.8, 15.6, 69.3, 32.1, and 151.5 cm at BF (see Table 3 and Figures 4a and 4d for the daily patterns). Years 2002 and 2004 were dry, 2001 and 2003 were nearly average, and 2005 was wet. For the James San Jacinto Mountains Reserve site (JMR), the air temperature data from 2006–2009 was directly obtained from the in situ observations and precipitation data was from the nearby Keenwild station (Figure 2a). The weather driving data beyond this period was obtained from the PRISM database and adjusted accordingly using the same procedure as for CP and BF.

[20] Model runs were initialized and stabilized by repeatedly using site-specific weather data of all available years for 900 years (i.e., initially started from 1000 AD). A background N deposition rate of 0.1 kg N $ha^{-1}$ $yr^{-1}$ and fire occurrence of every 100 years were used. For the final model runs (2001–2005), the forests were assumed to start growing after a fire in 1895 without any additional fire. The model was calibrated to obtain historical tree growth, litter C, SOM and C/N ratios in

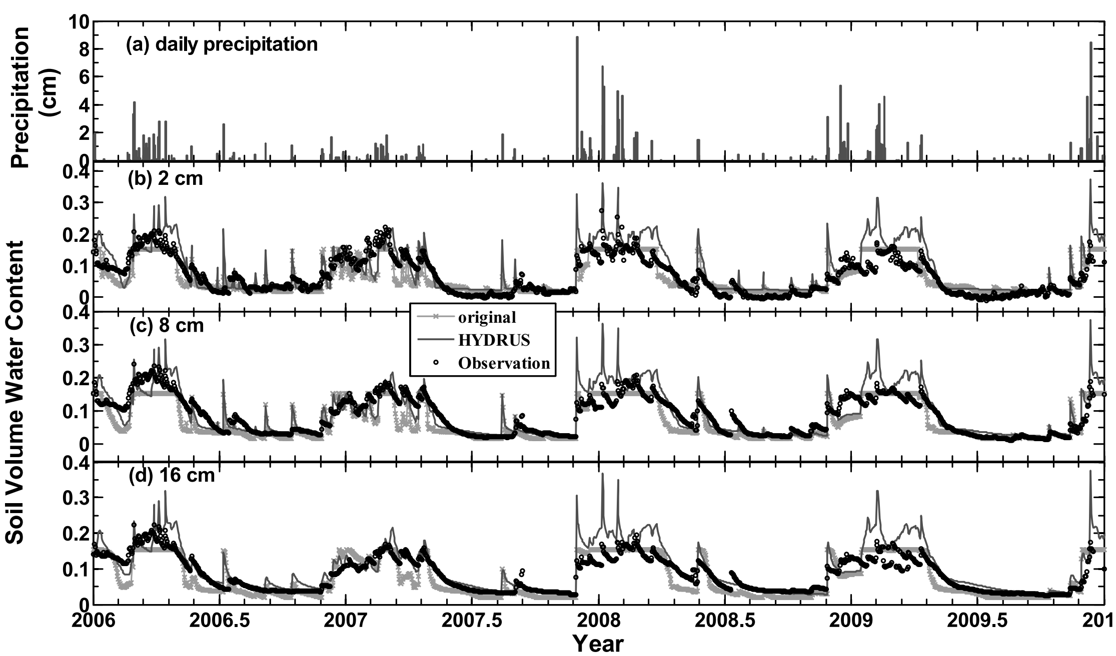


**Figure 2.** Precipitation (cm $day^{-1}$) and soil volume water contents (VWC) at 2, 8, and 16 cm depths observed and simulated by DAYCENT original and HYDRUS water flow module at the James San Jacinto Mountains Reserves (JMR) near the San Bernardino National Forest, California, from 2006 to 2009.

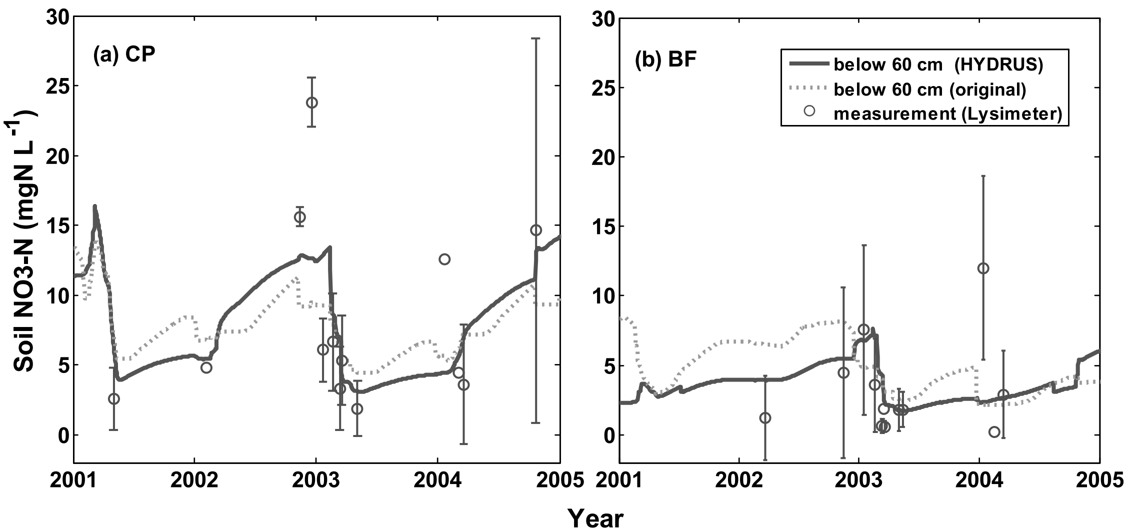


**Figure 3.** Soil solution $NO_3^-$-N concentrations (mg $L^{-1}$) measured in the lysimeter (mean ± standard deviation; no error bar if only one observation) and simulated using the DAYCENT model with the original or HYDRUS water flow modules at (a) Camp Paivika (CP) and (b) Barton Flats (BF), San Bernardino National Forest, California, from 2001 to 2004.

the study area, and some parameters, mainly tree growth and mortality, were modified in order to achieve reasonably well-fitted outputs with these observations [*Fenn et al.*, 2008]. The model inputs assumed that gradually increasing N deposition began in 1930 and in 1980 reached the current levels of 70.0 and 8.8 kg N $ha^{-1}$ $yr^{-1}$ for CP and BF, respectively, and remained constant thereafter [*Fenn et al.*, 2008].

## 3. Results

### 3.1. Model Evaluation in Soil Water Dynamics and Deep Nitrate Simulation

[21] As described above, the original soil water module in DAYCENT treats water movement as a two-stage process. This algorithm produces surface soil saturation during periods of continuous and large rainfall (or snowmelt) and thereafter a rapid drop to field capacity (Figure 2). At the James San Jacinto Mountains Reserve (JMR), surface saturation occurred very rarely from 2006–2009, probably due to its sandy texture with very high saturated conductivity (Table 2). During most of the rainy season, the soil surface remained at field capacity or below. Apparently the HYDRUS water flow module produced soil moisture patterns during the rainy season that were more realistic and corresponded better to the precipitation pattern (Figure 2). Compared to the measurements, the HYDRUS water module appeared to over-estimate soil moisture during large rain events. And both water modules showed time-lagged high soil water content during the late precipitation seasons, probably because DAYCENT assumed some snow accumulation in winter if temperature dropped to zero, which was melting later on. The observation demonstrated this could be incorrect. Since the two water flow modules produced different water content patterns (especially during rainy periods), it can be expected that this will result in different estimates of soil nitrogen gaseous emissions, because generally near-saturation conditions promote gaseous emissions in the DAYCENT model.

[22] At CP and BF sites, simulated soil solution nitrate concentrations, calculated as the soil $NO_3^-$-N content divided by the amount of soil water in between 60 and 120 cm depths, are compared in Figure 3 with measured $NO_3^-$-N concentrations, averaged over the sampled lysimeters at a depth of 1 m (samples varied from 1–6, because very often soil solution samples could not be extracted from all installed lysimeters). Both measurements and simulations showed that late fall rainfall events transported nitrate downward so that deep soil nitrate concentrations reached an annually high value at the beginning of the winter rainy season. Further precipitation then leached soil nitrate out of the profile, resulting in very low nitrate concentrations below 60 cm. It appears that, in general, both modules adequately represented deep soil nitrate dynamics (time series), although the absolute nitrate concentration comparison might be inappropriate because (1) there was large variance in the lysimeter collected measurements, and (2) the soil solution collected by lysimeters installed at 1 m depth may not identically represent the simulated soil portions. However, it still provided useful information for model evaluation. For an example, the measurements were almost carried out in the 2003 rainy season, during which the HYDRUS water flow module simulated a higher deep soil $NO_3^-$-N accumulation prior to drainage but lower by the end of early spring drainage period compared to the original module, especially at the high-N deposition CP site (Figure 3a). This result might be related to more N downward movement and final leaching simulated by the HYDRUS module (see below). Apparently, neither model simulated the deep soil $NO_3^-$-N peaks in the dry 2004 year at both sites (Figure 3), nor high concentrations observed in the wet 2003 year (see Table 3) at the CP site, indicating that further improvement in DAYCENT's solute transport module may be needed. Both simulations and lysimeter measurements showed that although the annual N deposition rate at the CP site (70 kg N $ha^{-1}$) was nearly 8 times larger than at the BF site (8.8 kg N $ha^{-1}$), deep soil nitrate accumulation was not very

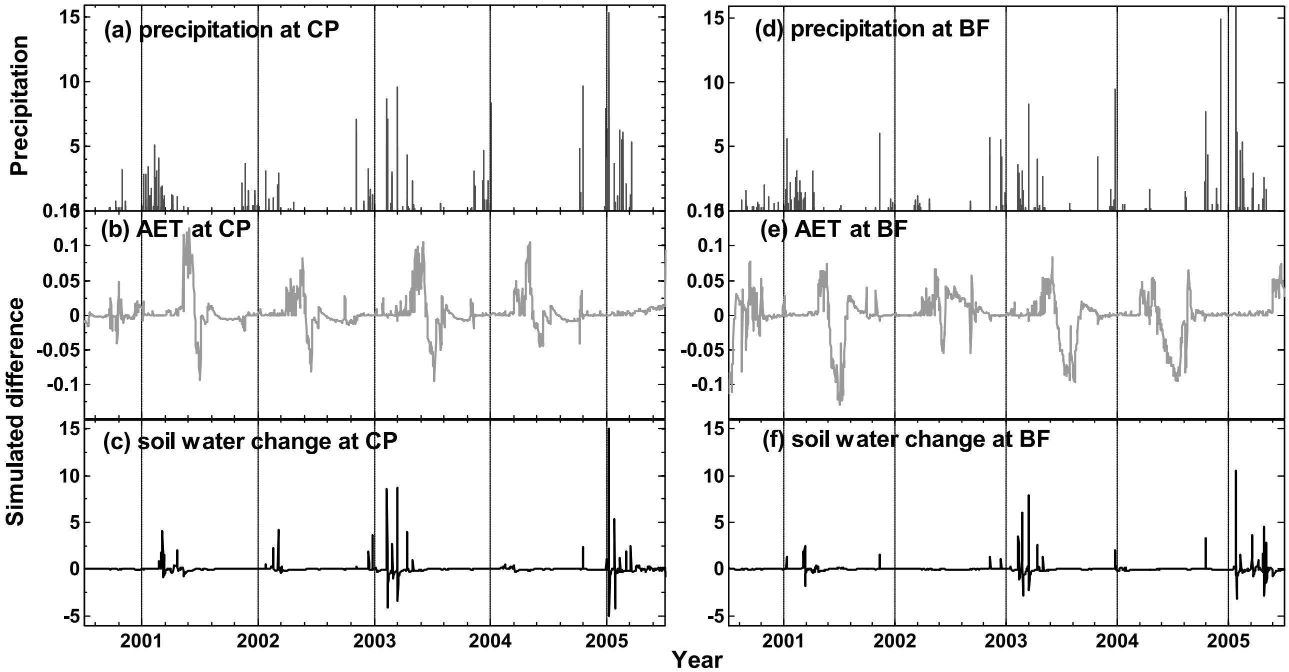


**Figure 4.** (a–f) Daily precipitation (cm day$^{-1}$) and simulated differences of actual evapotranspiration (AET) and soil water change by the DAYCENT original and HYDRUS water flow modules at Camp Paivika (CP) and Barton Flats (BF), San Bernardino National Forest, California, from 2001 to 2005.

different from both modeling and observations. This result implies that nitrate transported downward would eventually be leached out of the profile, irrespective of the rates of external N sources and/or mineralization at these two sites.

### 3.2. Comparison of Soil Water Drainage Simulation by Two Water Modules

[23] At our two study sites (CP and BF), the HYDRUS water module did not produce runoff except for very wet 2005 year at BF, and the original water module only simulated a very small amount of runoff (Table 3). Different soil water content simulated by the two modules did not significantly affect estimates of actual evapotranspiration (AET) either; less than 0.15 cm day$^{-1}$ and 2.4 cm per year at most (Figure 4b and Table 3). But it did affect the timing and pattern of soil water change (Figure 4c) and thus drainage (Figure 5) remarkably, although the differences at yearly were small (Table 3). After a long dry period in summer and early fall, initial precipitation events did not produce much water flow out of the system until early spring, when soil water storage had risen and snowmelt or rain continued (Figure 4a versus Figure 5a for CP site, Figure 4d versus Figure 5b for BF site). Drainage predicted by the DAYCENT original water module stopped earlier and was stronger than by the HYDRUS water module, in which drainage lasted until May, except for the wet year 2005 (Figures 5a and 5b).

[24] HYDRUS water module simulated small drainage fluxes on more days. It, however, simulated fewer large drainage events, compared to the original water module (Figures 5c and 5d). For the time period (total 1827 days) of 2001–2005 at the CP site, daily drainage of less than 0.25, from 0.25 to 1.0, and more than 1.0 cm per day, simulated by the HYDRUS module, occurred on 278, 238, and 19 days, respectively. The original water module simulated drainage for the same intervals on 67, 198, and 26 days, respectively (Figure 5c). At the BF site, the same ranges of drainage were simulated by the HYDRUS module on 308, 142, and 20 days, and by the original water module on 37, 108, and 27 days, respectively (Figure 5d). The original water flow module simulated drainage fluxes of higher intensity and smaller frequency, i.e., more pulsed, than the HYDRUS water module. This result was especially true during drier 2002 and 2004 years, but less so during the wetter 2005 (Figures 5a and 5b).

[25] By integrating the daily drainage over monthly and yearly periods for 2001–2005 at the CP site, the coefficient of variation decreased from 4.752 to 2.127 (monthly) and 0.924 (yearly) for the HYDRUS water module simulations, and from 6.633 to 2.345 (monthly) and 0.986 (yearly) for the original water module simulations. At the BF site, the coefficient of variation for daily, monthly, and yearly time-scale was 5.477, 2.277, and 1.102 for the HYDRUS module, and 7.587, 2.630, and 1.123 for the original module, respectively. These results show that the simulated difference in drainage volumes between the two modules decreased with increased time averaging period, until there was no significant difference at the annual time scale (Table 3).

### 3.3. Difference of DAYCENT Simulated N Leaching Losses with Two Water Modules

[26] In the simulations, most N leaching in 2001 through 2005 at CP and BF sites occurred in the late rainfall season (from late winter to early spring), corresponding to drainage events (Figure 6b versus Figure 5a for CP site, and Figure 6e versus Figure 5b for BF site) rather than net addition to soil excess N pool (i.e., mineralization + source − uptake) (Figure 6a and Figure 6d). The rates of N loss generally followed the

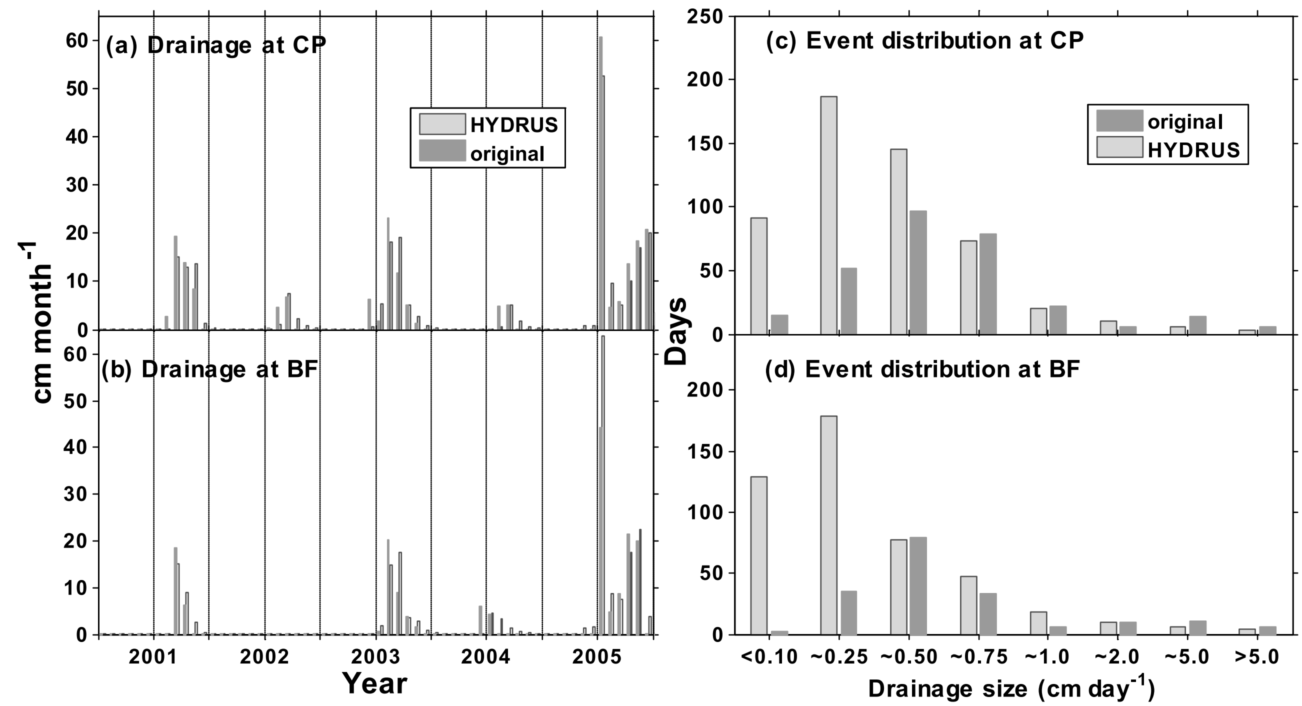


**Figure 5.** (a–d) Comparison of monthly soil water drainage rates (cm month$^{-1}$) and number of days when daily drainage rates were in specified ranges simulated by the DAYCENT original and HYDRUS water flow modules at Camp Paivika (CP) and Barton Flats (BF), San Bernardino National Forest, California, from 2001 to 2005.

time trends in drainage rates. Consequently, N leaching losses in the dry years of 2002 and 2004 were low compared to high N deposition amounts (see Figure 6 and Tables 3 and 4). In the wet year of 2005, very high N leaching in December was consistent with the intensive drainage during the same period, but it was not proportional to drainage. While drainage was 2–3 times greater during the wet year compared to the normal year, it did not produce correspondingly larger nitrate leaching losses.

[27] The 2002 and 2004 years were dry, so the leaching losses predicted by the two water modules were very low and only slightly different. However, during the average

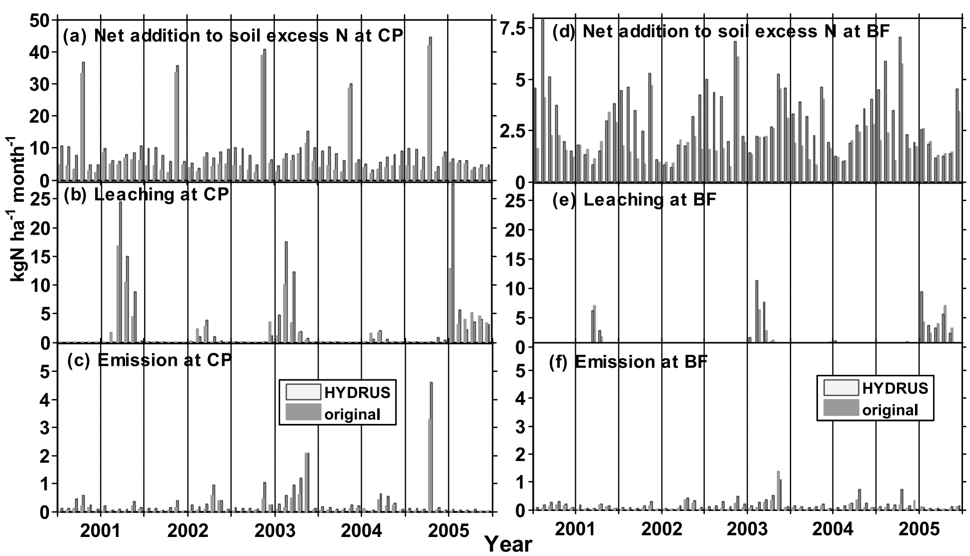


**Figure 6.** (a–f) Monthly N losses (leaching at 120 cm soil bottom and gaseous emissions) (kg ha$^{-1}$ month$^{-1}$) simulated by the DAYCENT model with either the original or HYDRUS water flow module at Camp Paivika (CP) and Barton Flats (BF), San Bernardino National Forest, California, from 2001 to 2005.

**Table 4.** Annual N Net Addition to Soil Excess N Pools and Their Losses Due to Leaching and Gaseous Emission, Simulated by HYDRUS or DAYCENT Original Soil Water Module, at the High-N Deposition Site Camp Paivika and the Low-N Deposition Site Barton Flats in the San Bernardino National Forest, California, in 2001–2005[a]

| | Net Addition to Soil Excess N Pool | | Leaching | | Emission (kg N $ha^{-1}$) | |
|---|---|---|---|---|---|---|
| Year | H | D | H | D | H | D |
| | | | *Camp Paivika* | | | |
| 2001 | 123.3 | 88.3 | 48.6 | 33.3 | 2.42 | 0.84 |
| 2002 | 116.4 | 80.3 | 6.24 | 5.26 | 2.94 | 1.42 |
| 2003 | 135.0 | 99.2 | 38.7 | 20.2 | 6.85 | 4.23 |
| 2004 | 107.6 | 69.9 | 3.35 | 3.31 | 2.51 | 1.30 |
| 2005 | 115.7 | 89.7 | 47.5 | 33.0 | 5.39 | 3.42 |
| | | | *Barton Flats* | | | |
| 2001 | 36.9 | 25.8 | 9.35 | 8.65 | 1.72 | 0.87 |
| 2002 | 33.8 | 20.4 | 0.00 | 0.00 | 1.68 | 0.92 |
| 2003 | 42.8 | 29.3 | 21.8 | 10.7 | 3.77 | 2.59 |
| 2004 | 33.6 | 23.3 | 2.29 | 1.46 | 2.17 | 1.08 |
| 2005 | 37.8 | 26.6 | 25.4 | 20.4 | 1.82 | 0.94 |

[a]Net addition is Mineralization plus Sources minus Uptake. Leaching is at 120 cm soil bottom. Emissions (kg N $ha^{-1}$) are measured from previous July to June. HYDRUS, H; DAYCENT Original, D. Camp Paivika is 70 kg N $ha^{-1}$ $yr^{-1}$, and Barton Flats is 8.8 kg N $ha^{-1}$ $yr^{-1}$.

years of 2001 and 2003 and the wet year of 2005, the rates of soil nitrogen leaching losses predicted by the HYDRUS flow module were very different compared to those calculated by the original water module (Figure 6b and 6e). In these 3 years, model runs that used the HYDRUS water flow module simulated higher N leaching losses than those that used the original water flow module. Cumulative annual N leaching losses (from previous July to June) (Table 4) at the high-N CP site simulated using the HYDRUS flow module were about 39 ~ 49 kg N $ha^{-1}$, while those simulated using the original DAYCENT water flow module were about 14 ~ 18 kg N $ha^{-1}$ lower. At the low-N deposition BF site, leaching N losses were highly variable, ranging from 9 kg N $ha^{-1}$ in 2001 to more than 20 kg N $ha^{-1}$ in both 2003 and 2005. In simulations with the HYDRUS water flow module the leaching was about 0.7, 11, and 5 kg N $ha^{-1}$ higher in 2001, 2003, and 2005, respectively, than in simulations with the DAYCENT water flow module (Table 4).

[28] The difference of net N additions to soil excess pool by two water modules could affect the absolute N leaching losses at CP and BF site (Table 4 and Figure 6), but did not primarily contribute to the difference of leaching estimation by two water modules. By contrast, Figure 7 showed that the simulated N leaching differences by the HYDRUS and original water modules was mostly correlated to the water drainage differences of two modules at daily time step (Figure 7a) rather than at monthly (Figure 7b) or yearly (Figure 7c). At the wetter CP site, regression $R^2$ of N leaching difference and drainage difference between two modules decreased from 0.292 at daily to 0.01 at monthly and 0.00005 at yearly, and at the relatively drier BF site, it changed from 0.382 at daily to 0.169 at monthly and 0.259 at yearly time step. Figure 7 also showed that soil drainage difference between two water flow modules mostly varied within 10 cm $H_2O$ regardless of time step, while the resulted N leaching loss estimation differences varied greatly at various time steps. It clearly demonstrated that short-term water flow dynamics would remarkably affect the N downward transport and its leaching losses.

### 3.4. Difference of DAYCENT Simulated N Gaseous Emissions with Two Water Modules

[29] Although N gaseous losses were relatively small, they occurred throughout almost the entire precipitation season, during which two major peaks appeared; one at the start and another at the end of the rainy season (Figure 6c and 6f). Usually the late spring N emissions were much higher than those in the previous fall, except for the very wet year 2005. The low N emission between these two peaks occurred during the period of very strong N leaching losses. The timing and amounts of leaching and emission compensated for each other (Figure 6b versus Figure 6c for CP site, Figure 6e versus Figure 6f for BF site). The 2005 rainy season started earlier than in other years, and stimulated larger production of trace N gases. The strong precipitation events that started in the middle of the rainy season then almost exhausted the soil nitrate near the soil surface, leaving not enough N for an observable gaseous emission peak in the late spring of 2005, as shown in other years.

[30] With the two-stage water flow algorithm used in the DAYCENT original water module, very few precipitation

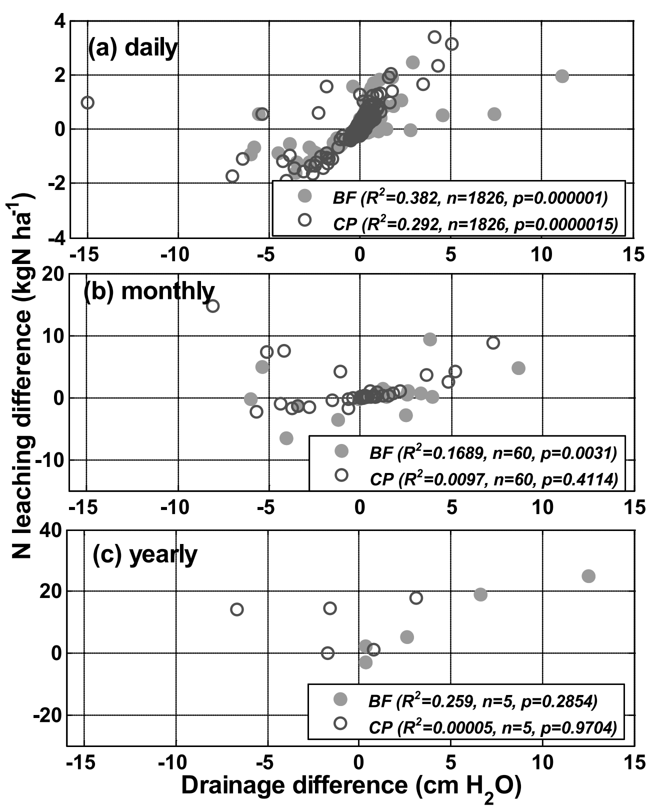


**Figure 7.** Comparison of the DAYCENT simulated N leaching difference and the soil drainage difference, between the original and HYDRUS water flow module, at (a) daily, (b) monthly, and (c) yearly time step, at Camp Paivika (CP) and Barton Flats (BF), San Bernardino National Forest, California, from 2001 to 2005.

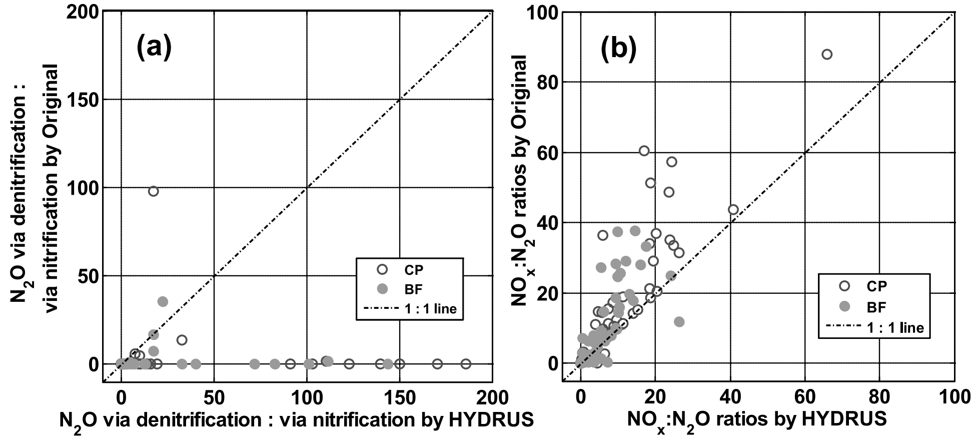


**Figure 8.** Comparison of the monthly DAYCENT simulated (a) N2O ratios of via denitrification to via nitrification, and (b) ratios of NOx to N2O, between the original and HYDRUS water flow module, at Camp Paivika (CP) and Barton Flats (BF), San Bernardino National Forest, California, from 2001 to 2005. Note that high $NO_x$:$N_2O$ ratios generally imply low water-filled pore space (WFPS) in Figure 8b [see *Parton et al.*, 2001].

events caused soil saturation, and even when such conditions occurred, soil moisture would very quickly drop back to field capacity, which would then persist or be further reduced (see Figure 2). As expected, calculations that used the HYDRUS water module simulated higher N gaseous emissions over the entire study period than those that used the original DAYCENT water module (Figures 6c and 6f and Table 4). In this study, DAYCENT with the HYDRUS water module predicted the annual emission of 2.4 ~ 6.8 kg N $ha^{-1}$ at CP and 1.7 ~ 3.8 kg N $ha^{-1}$ at BF, while with the original module of 0.8 ~ 4.2 kg N $ha^{-1}$ at CP and 0.9 ~ 2.6 kg N $ha^{-1}$ at BF (Table 4). In other words, the difference was about 1.2 ~ 2.6 and 0.7 ~ 1.2 kg N $ha^{-1}$ $year^{-1}$ for the high-N CP site and the low-N BF site, respectively, or about 1.5 ~ 2 times higher with the HYDRUS water module. Thus differences in trace N emissions estimated by the two water modules cannot be ignored in this heavily N deposition ecosystem.

[31] In this study, of total N emissions, $NO_x$ gas was the major species with 80–85% at CP and 75% at BF, $N_2O$ was about 15–18% at CP and 22% at BF, and the rest 1–3% was in form of $N_2$. As reasonably expected from soil water simulations, especially during late rainy season (Figures 4c and 4f), N gaseous emission difference between two water modules described above was mainly related to soil water controlled N biochemical processes. As briefly described previously, DAYCENT calculated $N_2O$ emissions as leaking from both aerobic nitrification and anaerobic denitrification processes. Figure 8a compared monthly $N_2O$ ratios of via denitrification: via nitrification, indicating that the model with its original water module only simulated large denitrification produced $N_2O$ occasionally, while with incorporated HYDRUS water module it produced much more $N_2O$ from denitrification. On the other hand, ratios of $NO_x$:$N_2O$ by DAYCENT with the original water module were about 56% larger than those with HYDRUS water module (Figure 8b). Generally the ratios of $NO_x$:$N_2O$ decrease with increasing WFPS, because gas diffusivity decreases with WFPS [*Parton et al.*, 2001]. Our results indicated that the original DAYCENT model produced higher $NO_x$:$N_2O$ ratios because of lower soil WFPS simulated, compared to the HYDRUS water module incorporated model.

## 4. Discussion

### 4.1. Soil Water Content Dynamic and Drainages

[32] The original and new HYDRUS soil water flow modules produced different soil water and drainage temporal patterns because of different assumptions involved in each of these modules, and thus different governing flow equations being solved, and different parameters (see Table 2 and Figure 1) used to characterize soil hydraulic properties. For example, in the tipping bucket model, field capacity is a key parameter, but it is not used in models like HYDRUS solving Richards equation for water flow. In many cases, this parameter is even not rigorously defined [*Twarakavi et al.*, 2009]. These differences resulted in soil moisture patterns produced by these two modules that were quite different, especially after rainfall events (Figure 2). The field observations showed that water contents higher than field capacity would last for a longer time period as simulated by the HYDRUS module, and that quick infiltration of water from top to bottom as predicted by the original flow module didn't capture observed water content variations with time (Figure 2). This difference in transient soil water status near saturation would likely have large effects on soil N states and dynamics of soil biogeochemical processes, such as mineralization and nitrification occurring mainly under aerobic conditions and denitrification under anaerobic conditions, both of which would affect N downward transport and gaseous emissions. In DAYCENT this issue could be overcome by: (i) dividing the topsoil layer into very thin layers (e.g., 0–1 and 1–4 cm, or more detailed horizon discretion), and (ii) adjusting the precipitation event period (e.g., from 4 to 2 h), so that the surface soils could be saturated even with small rainfall events [*Parton et al.*, 1998]. Obviously, these model tuning techniques may not be appropriate or workable for various situations. For example, this study demonstrated that the very thin top layer division did not improve much in

the two study sites with coarse soils. For fine soils, this approach may be viable because of very low infiltration rates, but it may remain a challenge even for an experienced modeler. A more realistic approach with this sort of water models may be to modify the hydraulic conductivity derived from the hydraulic functions rather than assuming saturated flow.

[33] As for soil water redistribution or the drying process, the original water module appeared to dry soil too quickly compared to the HYDRUS water module (Figure 2). It is generally accepted that physically based flow models that numerically solve the Richards equation [*Scanlon et al.*, 2002] or other analytical water flow equation [e.g., *Bastiaanssen et al.*, 2007] are superior to models not based on numerical solutions, especially when applied to arid regions. A specific difficulty with models that do not use the Richards equation may lie in how to continuously move water between layers under unsaturated conditions. DAYCENT water module applies Darcy's Law, but the fluxes are adjusted by a "damping multiplier" probably in order to reasonably fit well with observation, which adds challenges to the model calibration process.

[34] The yearly water drainage volumes simulated by the two modules were similar (Table 3), implying that the two-stage (infiltration-redistribution) water flow module (in the original DAYCENT) is appropriate for long-term water balance calculations. *Doltra and Muñoz* [2010] also had proved that both semi-dynamic and transient dynamic water modules could achieve very similar drainage simulation as long as using hydraulic functions derived from same pedotransfer functions. However, the semi-dynamic water modules cannot adequately describe the short-term dynamics (Figure 2). In our study, the two-stage water flow module of DAYCENT produced more pulsed water drainage fluxes, while the HYDRUS flow model simulated more continuous, but smaller fluxes. Further comparison of internal soil water fluxes also showed that, similarly as for bottom fluxes (i.e., drainage), the HYDRUS module simulated more frequent and less intensive water fluxes than the DAYCENT water flow module.

### 4.2. Soil Water Flow (and Drainage) and N Transport (and Leaching)

[35] In this study, actual evapotranspiration (AET) from two water modules was very similar (see Figure 4b and Table 3) because the same potential ET and root distributions for water uptake were used. The runoff from two modules was a small portion of water balance (Table 3). So the difference of soil water change was mainly caused by water drainage, which remarkably occurred during late precipitation period (see Figure 4). Therefore the characteristics of continuous, frequent, and less intensive water flow produced by the HYDRUS water module would primarily be responsible to the increase of the transport of soil surface nitrate downward and then its leaching out of the soil profile in average and wet years. As shown above, differences in the annual N leaching loss simulated by the two water modules were largest during the average year of 2003 (over 30%). For another average year 2001, the difference was still very large (18%) at the CP site, while at the BF site it was much smaller, probably because the dry conditions reduced drainage events and drainage amounts (Table 4) to a critical threshold at which further dry conditions would not significantly affect the annual N leaching loss. These simulated differences decreased in the very wet 2005 (Tables 3 and 4).

[36] Our analysis revealed that N leaching difference between two water modules were actually related to the difference of water drainage at the daily time step rather than longer simulation intervals (Figure 7). In the DAYCENT model, each water flow event will deplete the topsoil nitrate and stimulate soil mineralization. This process occurs primarily in the topsoil, and produces more soluble N, which in turn provides more nitrates for leaching. Because the plant N uptake for these old-growth trees was a small portion of total N budget (less than 4 kg N $ha^{-1}$ $yr^{-1}$) and same amount of N deposition was received at each site, the difference of simulated net N addition to soil was mainly the consequence of large difference in N mineralization by two modules (Table 4). Figures 5a and 5d showed that with HYDRUS water module, the N net addition to excess N pools was higher than that with the original water module, partially caused by this effect of water depletions. So relatively more frequent water flow, predicted by HYDRUS compared to the original water module, can transport more nitrate downward, so that the nitrate concentrations are higher in the bottom layer and lower in the top layer, except in dry years (see Figure 3 for comparison of deep nitrate concentrations) when little movement occurs due to lack of water.

[37] However, under intense precipitation conditions (e.g., the very wet 2005), initial very large leaching events produced by the models could exhaust soil inorganic nitrogen and leave less N for continuous loss later on (see Figures 6b and 6e). Additionally, soil N mineralization, which can restore soil mobile N pools, could be reduced due to more anaerobic soil conditions in very wet years, as indicated by actually lower net N addition to soils in 2005. Also, in the wet 2005, the relative difference in the number of effective drainage days produced by the two water flow modules was significantly reduced compared to 2001 and 2003. Therefore, simulated N leaching in the very wet year 2005 was comparable to those in the average years of 2001 (except for the BF site) and 2003, although the drainage amount almost doubled or tripled (Tables 3 and 4). The medium rainfall in the average years caused soil water conditions and water infiltration and drainage, which is favorable for N mineralization and downward movement and leaching. This is because generally, while N leaching increases with drainage (i.e., rainfall), it does not do so linearly [*Simmelsgaard*, 1998], mostly due to the limitation of available N.

[38] Lysimeter observations of high nitrate concentrations at the beginning of the rainy season prior to large leaching events (see Figure 3) imply that the actual nitrate leaching from the soil surface to deeper soil layers may be higher than simulated. In other words, both modules might underestimate soil nitrate leaching losses, probably due to soil nitrate diffusion and dispersion not explicitly included in the current N module of DAYCENT. But generally it is small compared to advective solute transport.

#### 4.2.1. Transient (Top) Soil Water Dynamics and N Gaseous Emissions

[39] Nitrogen gaseous emissions are mainly produced by a series of anaerobic processes as well as through the leaky pipe of nitrification in this study because ammonia volatilization was not favored in these forests (pH 4–5 at CP and 5.8–6 at BF [*Grulke et al.*, 1998]). The $NO_x$ emission in the study sites

might be associated with nitrification and dominate the N emission during dry season [e.g., *Fenn et al.*, 1996], as shown in our simulations that N gaseous emission occurred throughout the entire year (see Figures 6c, 6f, and 8b). But apparently emissions during the rainy season (especially in the spring) were more important, implying that simulating soil water near-saturation is critical in this ecosystem. The late-fall gaseous emissions were due to the early fall rainfall that induced near-saturation conditions and not much inorganic N leaching loss or plant uptake. With initial leaching in the middle of the precipitation season, nitrate accumulation in the topsoil layers was quickly reduced due to solute transport to deep soil layers. Also the soil temperature, which was very low at that time (i.e., winter period) at both study sites, constrained the production of trace N gases due to temperature regulated soil $NO_3^-$-N production through nitrification. When water drainage and thus N leaching started to decline in the spring, another N emission peak (usually stronger) appeared, because the rainfall season continued (so that soil water content was still high) and the weather was warmer, all of which favored soil nitrification, denitrification, and other trace N gaseous emission processes within the model.

[40] With the two-stage water flow process considered in the DAYCENT original water module, very few precipitation events caused soil saturation, and N emissions were lower than those predicted using the HYDRUS water module. This implies that the approach used in the original water module may be inappropriate for properly driving highly spontaneous and pulsed soil N gaseous processes in these two coarse-soil study sites. Even when the modeled soil profiles were divided into layers as suggested (i.e., two very thin top layers 1 and 3 cm thick), this modeling comparison study showed that the original approach was not as dynamical as the finite element algorithm used in HYDRUS for simulating water flow. The HYDRUS approach produced moisture patterns that allowed us to properly represent the pulses of soil N emissions at the daily time step, as theoretically and practically expected.

#### 4.2.2. Transient Soil Water or Semi-dynamical Water Modules for N Biogeochemical Cycle?

[41] As investigated in this study, many studies had already demonstrated that Richards equation or other based numerical soil water flow modules are needed to capture dynamic features and complexity of N biogeochemical processes under highly variable water conditions [e.g., *Gu and Riley*, 2010]. There are also many modeling studies showing that more comprehensive representation of soil-plant N processes [e.g., *Doltra and Muñoz*, 2010; *Stolk et al.*, 2011] would be required, especially for those N gaseous emission related. The challenge is still remaining largely due to lack of continuous and reliable field measurements on N losses, especially N gaseous emissions, for model development and evaluation.

[42] Major concerns or obstacles regarding the application of Richards equation-based numerical soil water flow modules in large-scale (both spatially and temporally) biogeochemical models, such as DAYCENT, lie in the computational demands of these modules, which usually require much longer computational time than tipping bucked modules [e.g., *Crevoisier et al.*, 2009; *de Jong and Bootsma*, 1996; *Scanlon et al.*, 2002]. This was also the case in our study. For examples, when running DAYCENT for 900 spin-up years and then 109 transient years at JMR sites with 6 soil layers, as shown in Table 1 on a laptop with Intel® Core2 Duo CPU of 2.26 GHz and RAM of 2 GB, with the original water module it took an average of 150 s (10 duplicates), while with HYDRUS water module it needed about 420 s. Then when running at CP and BF sites with 10 soil layers, the average computation time for the original semi-dynamical water module was about 210–230 s, but for the transient HYDRUS water module the runtime needed rose up to 800–920 s. This could present a serious obstacle when applying these sorts of models regionally [*Del Grosso et al.*, 2006] and globally [*Del Grosso et al.*, 2009]. However, considering the dependence of highly variable N biogeochemical processes on transient soil water contents and fluxes, it was necessary and worthwhile, as demonstrated in this modeling comparison study, to incorporate a more dynamic soil water module to properly simulate soil water flow and status. Recently, *Crevoisier et al.* [2009] demonstrated a fast non-iterative approach to numerically solve the Richards equation. Such an approach may prove to be a very good alternative to tipping bucket water models for use in biogeochemical model applications if experimentally proven workable.

## 5. Conclusions

[43] In the San Bernardino National Forest (SBNF) in southern California, the mixed conifer forests have experienced the highest atmospheric N deposition in North America. To be able to track the fate of N, it is crucial that soil water and its movement (and drainage) are appropriately modeled for this Mediterranean type climate. This importance stems from the fact that (1) soil water dynamics determine redox conditions, an important factor controlling N gaseous emissions, and (2) downward water fluxes (drainage) transport nitrate below the root zone and toward groundwater and ultimately surface runoff, resulting in N leaching loss.

[44] After replacing the original DAYCENT's semi-dynamic (two-stage) water flow module with a finite element based transient water flow module from HYDRUS-1D (based on the solution of the Richards equation), daily soil water contents were more dynamically simulated, better representing the time series of soil water contents measured at a shallow soil site of an old-growth mixed coniferous forests surrounded by SBNF, especially during winter precipitation periods. The two-stage water flow algorithm used in the original DAYCENT model did not allow the topsoil horizon to remain above field capacity for a long period of time. When comparing simulated deep soil nitrate concentrations with the lysimeter measurements conducted at the two study sites (CP and BF) from 2001 to 2004, simulations that used the HYDRUS water flow module produced a slightly higher deep nitrate concentration prior to, but lower concentration after, the drainage period, compared to the original DAYCENT module. However, both models captured the time patterns of deep nitrate accumulation indicated by the measurements.

[45] DAYCENT model with the original water module estimated trace N gaseous losses 18–48% lower than when the HYDRUS water flow module was incorporated, which estimated soil N emissions of 2.4 ~ 6.8 kg N ha$^{-1}$ year$^{-1}$ at the high-N CP site and 1.7 ~ 3.8 kg N ha$^{-1}$ year$^{-1}$ at the low-N BF site. This simulated N gaseous emission difference between two water modules largely due to the transient water

flow module simulated more favorable water conditions (near saturation) for denitrification relevant N gaseous emissions.

[46] The semi-dynamic water module of DAYCENT generally simulated less frequent but higher volume (i.e., more spontaneous) water fluxes and drainage compared to the HYDRUS water module. Although the annual water drainage simulated by the two modules was not significantly different, the HYDRUS module predicted more N mineralization and nitrate downward movement and thus higher leaching losses. This is because frequent but small water flows promoted continuous nitrate transport and likely soil N mineralization at topsoils. With contrasted drainage time patterns, the difference of N leaching between the two models was about 14 ~ 18 kg N $ha^{-1}$ $yr^{-1}$ at the high-N CP site, while about 5 ~ 12 kg N $ha^{-1}$ $yr^{-1}$ at the low-N BF site, during normal years (with 60 ~ 80 cm annual precipitation). During either dry (annual precipitation of less than 30 cm) or wet (about 120 cm) years, the differences in model predictions in N leaching were relatively small.

[47] This modeling study suggests that estimation of water balance over time periods longer than one month will not likely be impacted whether semi-dynamical or fully transient water modules are used. However, it appears that numerical solutions based on the transient Richards equation are preferred when soil water flow and solute transport are coupled and predictions of soil-plant N cycling are needed at time scales of days or less.

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