Modeling N$_2$O emission from grazed grassland

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The DLO Research Institute for Agrobiology and Soil Fertility (AB-DLO) is part of the Dutch Agricultural Research Department (DLO-NL) of the Ministry of Agriculture, Nature Management and Fisheries.

The institute was founded on 1 November 1993 by the amalgamation of the Centre for Agrobiological Research (CABO-DLO) in Wageningen and the Institute for Soil Fertility Research (IB-DLO) in Haren.

The DLO organization generates new knowledge and develops and maintains the expertise needed for implementing government policies, for improving the agro-industry, for the planning and management of rural areas and for protecting the environment.

AB-DLO, with locations in Wageningen and Haren, will carry out research into plant physiology, soil science and agro-ecology with the aim of improving the quality of soils and agricultural produce and of furthering sustainable plant production systems.

Key areas in AB-DLO expertise are: plant physiology, soil biology, soil chemistry and physics, nutrient management, crop and weed ecology, grassland science and agrosystems research.

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Introduction

This report is the final result of project 852078, of the Dutch Climate Change Research Programme (NPO-I), on modeling dinitrogen oxide (N\textsubscript{2}O) emission from grassland. This project formed part of an integrated study of N\textsubscript{2}O fluxes from grassland, within the framework of NOP-I. Participating institutes were, besides AB-DLO, Theoretical Production Ecology (Wageningen Agricultural University), Nutrient Management Institute (NMI), Environmental Sciences (TNO-MW), and Public Health and Environmental Protection (RIVM).

Chapter 1 gives a general introduction of the subject, and places the emission of N\textsubscript{2}O from grassland within the wider context of nitrogen losses from dairy cattle farming. It also indicates how to reduce these nitrogen losses, including the emission of N\textsubscript{2}O.

Chapter 2 describes the components of an integrated model, which are necessary as a framework to simulate the emission of N\textsubscript{2}O from mown and from grazed grassland. Two examples of such more or less complete models from the literature are compared with two modeling approaches, that were followed at AB-DLO.

Chapter 3 describes the simulation model SONICG, that was developed at AB-DLO to describe the emission of N\textsubscript{2}O from grassland soils. Details are presented of the modules used in SONICG, followed by results of two preliminary simulation runs for grassland without and with urine application.

Chapter 4 describes results from a simulation model for carbon and nitrogen turnover in mown grassland, developed at AB-DLO by Verberne. This is done because some results from this model are used as input data for the model SONICG.

Chapter 5 describes a recent modification of the model for mown grassland, which makes it suitable to simulate grazed grassland and that discriminates areas unaffected by animal excreta from urine spots.

Instead of a long list of references at the end of the report, we have chosen to inform the reader about the relevant literature sources that we used immediately following each (sub-)chapter of the text.
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1. Nitrogen losses from grazed grassland, a general introduction

Nitrogen losses from grazed grassland, including \(N_2O\) emission, are caused by:
- high \(N\) inputs, leading to periods with high concentrations of mineral \(N\) in the soil and increasing the chances of \(N\) loss via nitrification and denitrification,
- the basically low \(N\) efficiency of milk and meat production by cows,
- the limited capacity of grass and soil to act as sinks for mineral \(N\), especially during autumn and winter,
- the concentration of high \(N\) inputs in urine and manure spots in the field, with negative effects on grass growth, a reduced \(N\) uptake and evapotranspiration, increasing the chances of \(N\) loss.

Nitrogen losses from grazed grassland, including \(N_2O\) emission, can be reduced:
- at the national scale by reducing feed imports,
- at the farm level by appropriate management including a reduction of external \(N\) inputs and by making optimum use of internal \(N\) inputs,
- at the field level by reducing spatial variability of \(N\) inputs.

Nitrogen losses from grazed grassland should be considered at three scales:
- the farming system (scale about 50 ha),
- the grassland field (scale about 1 ha),
- urine and feces spots within a field (scale < 1 m²).

The farming system scale

Reduction of nitrogenous emissions to the environment (atmosphere, surface- and groundwater) should be achieved by appropriate management at the farm-system level. A good example of this approach is the research on dairy farming with minimal emissions on the experimental dairy farm "De Marke" (Biewinga et al., 1992). Nitrogen losses from dairy farming are strongly related to the intensity of \(N\) turnover as shown by the \(N\)-flow schemes for intensive and extensive dairy farms (Fig. 1.1, adapted from Aarts et al., 1992).

Management to minimize \(N\) emissions to the environment should consider:
- the number of animals per ha of grassland,
- the feed production on the farm versus imported feed, including the integration of arable farming with grassland utilization,
- a reduction of external inputs, e.g. of mineral fertilizers and of imported animal feed,
- optimal use of internal inputs as animal manure and nitrogen fixation by clover,
- the production capacity of the soil, determined by characteristics as soil type, groundwater level and drainage, water availability for grass production,
- the integration of mowing and grazing,
- continuous grazing versus limited grazing.

A reduction of feed import from other countries will also reduce the input surplus of nutrients on the national nutrient balances.

The grassland field-scale

The potential for grass production and \(N\) uptake will strongly depend on the soil characteristics and on the weather conditions in a given year (Verberne, 1992). Grass production will strongly affect \(C\) and \(N\) turnover in the soil as well as the availability of water by evapotranspiration.
Figure 1.1. Top: Nitrogen flows for the experimental dairy farm with minimal emissions to the environment "De Marke"; from Biewinga et al. (1992). Bottom: Nitrogen flows for specialized dairy farms on sandy soil in the Netherlands, average values for the period 1983-1986; based on data from LEI-DLO.

All values are given in kg ha\(^{-1}\) y\(^{-1}\).
Grass is the main supplier of carbon to the soil by root exudation, dead roots and above-ground litter. N uptake by the grass is a potentially very large sink for soil mineral N. Under optimum conditions for growth, grass can annually take up as much as 800 kg N per ha. Periods with bad conditions for grass growth will lower net production and N uptake and will increase C and N recycling of dead grass litter into the soil.

A shift from arable soil to grassland will lead to an increased level of soil organic matter and to net N immobilization by the build-up of soil organic N. Turning grassland back to arable use will lead to a rapid decrease of soil organic matter and to net N mineralization. Reversal of old grassland to arable land may lead to substantial N losses when the amount of N mineralized from accumulated soil organic matter is much higher than can be taken up by an arable crop.

The apparent N recovery in mown grasslands is high, up to 90%, suggesting that only limited losses to the environment may occur ('t Mannetje & Jarvis, 1990). Grazing increases the recycling of C and N in grassland. Of the N taken up by dairy cows about 70-85% is returned to the soil, in feces and urine, and only about 15-30% is removed as milk and meat. The low efficiency of N utilization in grazed grassland explains the high N losses to the environment by NH₃ emission, nitrate leaching and denitrification (van der Meer, 1991). Local deposition of urine and feces can affect grass development, positively as well as negatively, and thus indirectly affect the recycling of C and N. Urine and feces spots are a main source of spatial heterogeneity in the field. Moreover, the deposition of urine and feces is sometimes concentrated on part of the field around a water supply or a milking point. Figure 1.2 shows the nitrogen cycle in grazed grassland and indicates where gaseous N emissions may occur.

Figure 1.2. The nitrogen cycle in grazed grassland.
Urine and feces spots as active centres for N loss

Nitrogen inputs into the soil of grazed grassland are concentrated in urine spots (about 0.5 m²) and feces spots (about 0.05 m²). Much more nitrogen is deposited there than can be taken up by one grass crop (Deenen & Middelkoop, 1992). Thus a surplus amount of soil mineral N can be present during several months (Afzal et al., 1993) and may lead to N losses (van der Meer & Whitehead, 1990).

Positive as well as negative effects on grass growth can occur (Deenen, 1990; Lantinga et al., 1987), depending on the actual urine concentration and on weather conditions. The nitrogen from urine is rapidly converted into ammonium-N, which may be partly lost by NH₃ volatilization. According to Vertregt & Rutgers (1991) on average about 10% of the urine-N may be lost from urine spots by NH₃ emission. After microbial nitrification of urinary ammonium to nitrate, the latter may partly be lost by leaching and by denitrification. The nitrogen from cow feces mainly consists of organic-N, that is only slowly mineralized, and a fraction of about 10% ammonium-N, that can be lost completely by NH₃ emission (van der Meer, 1991). As shown in figure 1.2, nitrogenous oxides can be produced as a by-product of nitrification (N₂O) or as intermediate products of denitrification (NO, N₂O).

Relevance of emission of N oxides from grassland to atmospheric chemistry

Groffman (1991) discussed the environmental factors that are controlling nitrification and denitrification activity at different scales (Table 1.1). In intensively managed grassland the plant community consists only of one or two productive grass species. According to Groffman, both nitrification and denitrification are likely to occur only in ecosystems with high or excess N availability, as is the case in intensively managed grazed grassland.

In grazed grassland effects of urine and feces on grass growth and senescence affect the field scale controlling factors soil water content, nitrate concentration and carbon supply. Nitrification of large amounts of ammonium in urine spots may take periods of weeks to months. However, as soon as large amounts of nitrate are formed, nitrate concentration is no longer a controlling factor for denitrification, leaving only soil water content and carbon supply as controlling factors. Thus, weeks to months after urine deposition to the soil a period of high rainfall intensity may cause a denitrification peak. The production of N₂O versus its reduction to N₂ will then depend on the carbon availability and the emission of N₂O on the gas diffusion characteristics of the soil and on the depth of the N₂O production/reduction zones. Rainfall during a period of rapid nitrification might also increase N₂O production through nitrifier-denitrification, when oxygen supply through gas diffusion limits complete nitrification (Poth & Focht, 1985).

References

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<td>Soil water</td>
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<td>Biome type, Climate</td>
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From: P.M. Groffman (1991)


2. Modeling $N_2O$ emission from (grazed) grassland

2.1 Introduction

In a first literature review several approaches relevant to modeling of $N_2O$ emission from grazed grassland at the field scale, have been discussed (Van Faassen, 1993). Specific attention was directed to modeling of processes in urine patches, as sites where $N_2O$ emission might be largely concentrated.

The emission of $N_2O$ from grazed grassland is the result of the complex interaction of physical, chemical and biological processes. Therefore, to model $N_2O$ emission from grazed grassland we need models that integrate all these processes. This can be done by using submodels (modules), that describe grass development, soil physico-chemical conditions and processes, and soil microbiological carbon and nitrogen cycling.

Integrated models

A large number of integrated models have been published for arable crop-soil ecosystems (Engel et al., 1993; Engel and Baldioli, 1993; Groot et al., 1991). Although many parts of these models are useful for comparative purposes, none of them specifically considers grazed grassland or $N_2O$ emission. Only two extensive models are known to us that specifically simulate the emission of $N_2O$ from soil as part of larger models (Grant, 1994; Li et al., 1992 a,b). Table 2.1 compares these models with the two modeling approaches followed at AB-DLO by Verberne (1992) and by Bril and Van Faassen (this report). A model of nitrogen flows in grassland under grazing or mowing (Thornley & Verberne, 1989) extensively simulated pasture development and effects of grazing by sheep but had only a limited soil submodel.

2.2 Two modeling approaches followed at AB-DLO

Starting from a model for mown grassland

- On the one hand we used the integrated model FSE-GRASS-SOM-WATER (Verberne, 1992), simulating C and N turnover in mown grassland. This model was based on three interacting submodels (Figure 2.1):
  - GRASS simulating C and N turnover of grass,
  - WATNIT simulating transport of water and dissolved mineral N,
  - SOM simulating C and N turnover in soil.

Results from simulation runs with this model, for a sandy soil and a clay soil in years with rather different weather conditions, are given in chapter 4. At the moment we develop a modified version of this model that can be used to simulate grazed grassland.
Development of a detailed model for C and N turnover in grassland soil

- On the other hand, Bril and Faassen (chapter 3) developed the model SONICG for detailed C and for N turnover in soil (Figure 2.2). SONICG includes the B and C modules and module A2 of Table 2.1. Unique is the presence of module B5 on soil chemistry, which calculates equilibrium concentrations of chemical substances in soil solution, adsorbed to the cation exchange complex and in the gas phase. The pH of the soil solution is also calculated and may be used to describe pH effects on soil processes. Physico-chemical equilibria strongly determine the partial pressures of the gases CO$_2$, NH$_3$ and N$_2$O, and thus are essential to modeling gaseous emissions. By using data on grass development as input, the model of Bril can describe N$_2$O, CO$_2$ and NH$_3$ emission from (grazed) grassland. A more detailed description of SONICG is given in chapter 3.

<table>
<thead>
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<th>References</th>
<th>Modules:</th>
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<tbody>
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<td>Grant (1991/a/b, 1994);</td>
<td>Grant et al. (1993/a/b/c/d, 1994)</td>
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<td>Li et al. (1992/a,b)</td>
<td>in x in</td>
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<tr>
<td>Bril &amp; Van Faassen</td>
<td>x = included; - = not included; x- = partly included; in = input needed</td>
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Module A: GRASS OR CROP DEVELOPMENT
A1 = carbon (C) and nitrogen (N) turnover (photosynthesis, metabolism, N uptake, litter recycling, removal of C and N with mown grass).
A2 = uptake by the grass of some macronutrients (K+Na, Ca+Mg,...).
A3 = effects of grazing and of management (positive and/or negative effects of urine and manure, fertilizer and slurry applications, grass intake, grass regrowth after grazing, different litter recycling under grazing).

Module B: SOIL PHYSICAL CONDITIONS AND PROCESSES
B1 = heat transport (soil temperature profile).
B2 = soil water balance and water transport (water input by rainfall and irrigation; water output by evapotranspiration and leaching).
B3 = transport of substances (nutrients and gases), dissolved in the soil water phase by mass flow and by diffusion.
B4 = transport of gases by gas diffusion through the soil gas phase.
B5 = soil chemical equilibrium calculations of concentrations of substances in the water phase (including pH), in the solid phase (including adsorption to soil CEC) and in the gas phase.

Module C: SOIL MICROBIOLOGICAL TURNOVER PROCESSES
C1 = soil MIT (Mineralization Immobilization Turnover of C and N, including the dynamics of the microbial biomass).
C2 = nitrification, including the dynamics of the nitrifier population.
C3 = denitrification (turnover of organic matter by a part of the microbial biomass included in C1, with the use of nitrate and N$_2$O as electron acceptors instead of oxygen).
Figure 2.1. Schematic representation of the model structure, which comprises four submodels simulating grass growth and the dynamics of soil moisture, soil organic matter and soil inorganic nitrogen (From Verberne, 1992)

Figure 2.2. Conceptual scheme of N transformations and N exchanges in the model SONICG (Simulation Of the Nitrogen Cycle in Grassland); Bril & Van Faassen
References


3. Simulation of the nitrogen cycle in (grazed) grassland soil (SONICG)

3.1 Introduction

As a main part of NOP-project 852078, on modeling N\textsubscript{2}O emission from grazed grassland, the simulation model SONICG, on carbon (C) and nitrogen (N) cycling in grassland soil, was developed by Bril and Van Faassen (this chapter). SONICG mechanistically layerwise simulates the relevant soil physical, chemical and biological processes. SONICG uses the model of Verberne (1992, see chapter 4) on grass development and a hydrology module to supply necessary daily input data, and concentrates itself on the nitrogen cycle processes that play a central role in the model SONICG: nitrification and denitrification, including the production and the reduction of N\textsubscript{2}O (Figure 2.2).

SONICG simulates processes in homogeneous grassland areas at the field scale. Therefore, separate simulation runs have to be made when large horizontal differences occur within a grassland field, as in the case of grazed grassland (see Chapter 5). The model considers 30 soil layers of 2.5 cm, each with its own properties, but homogeneous properties are used within a layer. The lower border is assumed to be permeable for water but not for gases. Thus gases can only flow out at the bottom of the profile dissolved in water. Above the soil a gas layer of 2.5 cm high is present in the model. Transport of gases through this layer to the atmosphere is assumed to occur only by molecular diffusion, with the rates of diffusion depending on the gas concentration gradients between the atmosphere and the gas phase of the first soil layer.

The model SONICG consist of the following modules:
1. Hydrology
2. Soil temperature
3. Soil organic matter turnover
4. Plant uptake of nutrients
5. Nitrification
6. Denitrification
7. Gas transport
8. Transport of dissolved substances
9. Chemical equilibria

Figure 3.1 shows the flow chart of the model calculations for each timestep of one day. The flow chart shows the order in which the modules are used, where the information needed for each module comes from and where the output of each module goes to. After initialization of the model, input data are READ from the hydrology module as well as other necessary input data. In the following lines a short explanation of each of the modules is given.
3.2 Hydrology

In this module the water balance is calculated daily for each soil layer. At day 1 (January 1) calculations start with a wet soil profile. Calculations start at layer 1. Rainfall and surface application of irrigation water, urine and slurry are water inputs for layer 1. Outputs are the physical evaporation of water and water taken up by plant roots and transpired by the grass. Each layer has a maximum and a minimum water holding capacity (WHC). Above the maximum WHC water freely flows to the next lower layer. At the minimum WHC no more water can be evaporated or taken up by plant roots and only inflow from neighbouring layers can occur. Water uptake by the grass is distributed over all the rooted layers, more or less proportional to root area and water availability in each layer. If water content of a layer has increased during a timestep, X% of the increase flows to the next lower layer. If water content has decreased during a timestep, an inflow of Y% of the decrease may occur from the next lower layer, if that layer is wetter. The values of X and Y depend on the physical characteristics of the soil (layer).
After calculations for layer 1, calculations for layer 2 follow and so on till the last layer. Gradients in water content between neighbouring layers are then slightly leveled off.

Physical water evaporation at the soil surface from layer 1 only (mm d⁻¹) is calculated from average daily temperature \(T_{\text{air}}\) and soil water-filled porosity (WFPS) according to equation 1:

\[
\text{water evaporation} = 0.1 \times T_{\text{air}} \times [\text{WFPS}(1)]^2
\] (1)

Water transpiration by the grass (mm d⁻¹) is calculated by using equation 2, which is a good approximation of the reference water transpiration for grassland according to Makkink (1975). To limit the total yearly water evaporation by the grass to a realistic value, a reduction factor (RF) is added that depends on the water supply capacity of the soil during the growing season.

\[
\text{water transpiration} = RF \times [0.65 \times \sin(2 \pi (\text{day}-30)/365) + 0.16 \times T_{\text{air}}]
\] (2)

Water transpired by the grass is taken up from all the rooted layers. In case of sandy soil a maximum rooting depth of 45 cm is used. A constant amount of roots is simulated, with a total surface area of 50 m² per m² soil. Root distribution over the rooted layers is calculated according to equation 3:

\[
\text{ROOTS}(i) = 50 \times \left[\frac{0.45}{\text{depth}(i)}\right]^{25} \sum \left[\frac{0.45}{\text{depth}(i)}\right]^{25}
\] (3)

\text{ROOTS}(i) is the root surface area in layer i; \text{depth}(i) is the average distance of layer i from the soil surface.

The hydrology module uses daily weather data as input: rainfall, minimum and maximum temperature. A weighted daily average temperature \(T_{\text{air}}\) is calculated from minimum and maximum temperature and day length. In case of urine deposition 5 mm water addition is used as input. A small amount of extra rainfall is given near the end of the year to return to the same amount of water stored in the soil profile as at the start.

References


3.3 Soil temperature

Soil temperature is modeled dynamically, using a one-dimensional convection-dispersion equation. First the initial temperature profile on January 1 is calculated from equation 4:

\[
T(i) = T_{\text{avg}} + \text{Amp} \times \sin(2\pi(-121/365) - d(i)/2) \times \exp\{-d(i)/2\}
\] (4)

\(T(i)\) is the temperature of layer i; \(T_{\text{avg}}\) is the mean annual temperature, taken to be 10 °C; \(\text{Amp}\) is the amplitude of the yearly temperature variation, also taken as 10 °C; \(d(i)\) is the depth of layer i in meters. It is assumed that the damping depth of the temperature wave is 2 meters for sandy soils.

The dynamic calculation of soil temperatures is based on equation 5:

\[
\frac{\partial T}{\partial t} = (\kappa C_p) \times \frac{\partial^2 T}{\partial x^2}
\] (5)
Here $\lambda/C_w$ is the heat diffusion coefficient, $\lambda$ is the heat conduction coefficient and $C_w$ is the heat capacity of the soil. The value of $\lambda/C_w$ is calculated each timestep for each soil segment. $C_w$ is calculated from the summed heat capacities of water, organic matter and mineral parts in a segment. Specific $C_w$ values of 1000, 600 and 480 kcal m$^{-3}$°C$^{-1}$ are used for water, organic matter and minerals, respectively (De Vries, 1963).

$\lambda(i)$ is calculated from the water content of segment $i$ with equation 6:

$$\lambda(i) = 0.04 + 0.35 \times \frac{[WFPS(i)]^2}{(0.065 + [WFPS(i)]^2)}$$  \hspace{1cm} (6)

Equation 6 was fitted to the curve of $\lambda$ for sandy soil, given by Duin (1956). Given the upper and lower boundary conditions as the air temperature and the temperature of the last model segment at the last timestep, a new temperature profile can be calculated.

References


3.4 Soil mineralization immobilization turnover (MIT)

Turnover of organic matter by soil microbes is simulated in this module, with inputs from (dead) grass roots, senesced grass litter and organic manure. The MIT module is a variant of a soil organic matter turnover model by Verberne et al. (1990). Figure 3.2 schematically shows the structure of this module, and indicates which products are formed upon degradation of a certain pool of organic matter. Each organic matter pool contains C as well as N with a fixed C:N ratio. Turnover is described in terms of C, and the resulting N turnover is calculated from the C:N ratios of substrates degraded and products formed.

When the products contain less N than the substrates, mineral N is liberated in the form of ammonium (N mineralization). When the products contain more N than the substrates, mineral N becomes immobilized (N immobilization).

All (external) inputs of organic matter to the soil are considered to consist of three fractions: rapidly Decomposable Plant Material (DPM), more slowly decomposable Structural Plant Material (SPM) and rather Resistant Plant Material (RPM). Distribution of inputs over these fractions is based on the overall (C/N)-ratio of the input (C/N). First, the fraction RPM-C is calculated using equation 4:

$$\text{fraction RPM-C} = 0.4 \times (C/N)^{1.5} \times [200 + (C/N)^{1.5}]$$  \hspace{1cm} (4)

Next, the fractions SPM-C and DPM-C can be calculated, using the fixed C/N-ratio's of the three fractions. Figure 3.3 shows the distribution of organic matter input over DPM, RPM and SPM as a function of C/N.

The turnover of these input fractions by soil microbes is separated into turnover by protected (BIOP) and non-protected (BION) microbial biomass. BIOP can be regarded as microbes that are protected against predation by clay particles or because they live in small soil pores, that cannot be entered by their predators. The fraction of the microbial biomass that is protected, fprot, is therefore described as a function of the fraction clay particles by equation 5:

$$f_{prot} = 0.8 \times (\text{fraction particles} < 2\mu m)^{0.5}$$  \hspace{1cm} (5)
Figure 3.2. Detailed scheme of the MIT module on Mineralization Immobilization Turnover of organic matter in the soil

Figure 3.3. Distribution of organic matter input over different pools.

SPM = structural plant material, mainly cellulose and hemicellulose; DPM = rapidly decomposable plant material, mainly proteins; RPM = resistant plant material, mainly lignin

The carbon degradation rates of DPM, SPM, and RPM by BIOP and by BION, respectively, are given by equations 6a and 6b:

\[
\frac{d(C)}{dt} = k_{ip} \cdot f(T) \cdot f(WFPS) \cdot C_i \cdot fprot
\]  

(6a)
\[
\frac{d(C)}{dt} = k_p \cdot f(T) \cdot f(WFPS) \cdot C \cdot (1 - \text{fprot}) \tag{6b}
\]

were \( C \) stands for either DPM-C, SPM-C or RPM-C; \( k_p \) and \( k_n \) are the maximum first-order rate constants \( (d^{-1}) \) for protected and non-protected fractions; \( f(T) \) the temperature rate reduction functions and \( f(WFPS) \) is the moisture rate reduction function. Thus, the degradation of DPM, SPM and RPM is attributed to BIOP and BION in the ratio \( [k_p \cdot \text{fprot}] : [k_n \cdot (1 - \text{fprot})] \).

Table 3.1 specifies the proportions of products formed, including microbial biomass, the first-order rate constants used for each transformation, and the C/N ratios of the organic matter pools.

Microbial biomass decreases at a death rate that is taken proportional to the size of this pool. Note that the death rate constant of protected microbes is assumed to be less than that of unprotected microbes. This can be explained as a result of the protection against predation, for microbes that live in small pores. Dead microbial biomass C is added in equal amounts to the pools of DPM and OMP or OMN and part of the dead biomass N becomes liberated at the same time.

Table 3.1. Biodegradation reactions of the M.I.T. module (Figure 2), reaction rate constants \((d^{-1})\) and carbon to nitrogen ratios of organic matter pools.

<table>
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<th>Biodegradation equations for carbon</th>
<th>Reaction rate constants (d^{-1}(*))) protected</th>
<th>non-protected</th>
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<tr>
<td>DPM-C ( \rightarrow 0.60 \text{CO}_2\text{-C} + 0.35 \text{BIO-C} + 0.05 \text{OM-C})</td>
<td>(k_p = 0.1)</td>
<td>(k_n = 0.1)</td>
</tr>
<tr>
<td>SPM-C ( \rightarrow 0.75 \text{CO}_2\text{-C} + 0.20 \text{BIO-C} + 0.05 \text{OM-C})</td>
<td>(k_p = 0.02)</td>
<td>(k_n = 0.02)</td>
</tr>
<tr>
<td>RPM-C ( \rightarrow 0.40 \text{CO}_2\text{-C} + 0.05 \text{BIO-C} + 0.55 \text{OM-C})</td>
<td>(k_p = 0.004)</td>
<td>(k_n = 0.004)</td>
</tr>
<tr>
<td>OM-C ( \rightarrow 0.80 \text{CO}_2\text{-C} + 0.10 \text{BIO-C} + 0.10 \text{SOM-C})</td>
<td>(k_p = 0.004)</td>
<td>(k_n = 0.02)</td>
</tr>
<tr>
<td>SOM-C ( \rightarrow 0.98 \text{CO}_2\text{-C} + 0.02 \text{BIO-C})</td>
<td></td>
<td>(k_p = 2\times10^{-5})</td>
</tr>
<tr>
<td>BIO-C ( \rightarrow 0.50 \text{DPM-C} + 0.50 \text{OM-C})</td>
<td></td>
<td>(k_p = 0.01)</td>
</tr>
</tbody>
</table>

\(*\) under optimum conditions of temperature and moisture

Carbon to Nitrogen ratios of organic matter pools (C/N)

<table>
<thead>
<tr>
<th>ORGANIC MATTER POOL</th>
<th>atom ratio</th>
<th>kg C (kg N)^{-1}</th>
</tr>
</thead>
<tbody>
<tr>
<td>DPM = rapidly Degradable Plant Material</td>
<td>C100 N16</td>
<td>5.4</td>
</tr>
<tr>
<td>SPM = Structural Plant Material</td>
<td>C100 N0.5</td>
<td>171.4</td>
</tr>
<tr>
<td>RPM = Resistant Plant Material</td>
<td>C100 N1</td>
<td>85.7</td>
</tr>
<tr>
<td>OMP = young soil Organic Matter, Protected</td>
<td>C100 N9</td>
<td>9.5</td>
</tr>
<tr>
<td>OMN = young soil Org.Matter, Non-protected</td>
<td>C100 N6</td>
<td>14.3</td>
</tr>
<tr>
<td>SOM = old Soil Organic Matter</td>
<td>C100 N5</td>
<td>17.1</td>
</tr>
<tr>
<td>BIOP= microbial BIOmass, Protected</td>
<td>C100 N20</td>
<td>4.3</td>
</tr>
<tr>
<td>BION= microbial BIOmass, Non-protected</td>
<td>C100 N12</td>
<td>7.1</td>
</tr>
</tbody>
</table>

The degradation rates of the protected and unprotected soil organic matter pools OMP, OMN and SOM, by BIOP and by BION, are described by equation 7:
\[
\frac{d(C_i)}{dt} = k_i \cdot f(T) \cdot f(WFPS) \cdot C_i
\]  

(7)

\(C_i\) is the carbon concentration in pool \(i\) (OMP, OMN and SOM); \(k_i\), the maximum first-order rate constant, equals \(k_{jp}\) for protected pools and \(k_{jn}\) for non-protected pools.

In contrast to other models, the effect of soil temperature on turnover rates is different for different organic matter pools (Nicolardot et al., 1994). Three temperature rate reduction functions \(f(T)\) are used for materials of different stability (Figure 3.4):

\[f(T)_1 = 0.01095 \cdot T^{1.5} / (1 + 3e-5 \cdot T^3), \text{ for DPM};\]

\[f(T)_2 = 0.00376 \cdot T^{1.75} / (1 + 6e-7 \cdot T^4), \text{ for SPM, RPM, OMN, and OMP}.\]

\[f(T)_3 = 0.001235 \cdot T^2 / (1 + 1e-8 \cdot T^5), \text{ for SOM}.\]

As can be seen from Figure 3.4, the more stable an organic matter pool, the more its relative degradation rate is shifted to higher temperatures. This can be explained by the higher energy of activation necessary to start degradation of more stable organic compounds.

The effect of the fraction soil water-filled pore space (WFPS) on turnover rates is described by one function for all organic matter pools (Figure 3.5):

\[f(WFPS) = 6 \cdot (P_w)^2 / [1 + 9 \cdot (P_w)^4]\]

The function was derived from data of Doran et al. (1990). As shown in Figure 3.5, the biodegradation of organic matter increases with increasing availability of soil water, until oxygen supply becomes rate limiting at WFPS = \(\sqrt{1/3}\). From that point onwards denitrification starts to participate in biodegradation and increases with the decreasing oxygen supply by gas diffusion. The exact form of \(f(WFPS)\) will still depend on soil type, soil dry bulk density and the aerobic biological activity of the soil, together determining the maximum of WFPS above which oxygen starts to limit oxygen consuming processes (Doran et al., 1990; Neilson & Pepper, 1990).
Thus, $f(WFPS)$ will also depend on soil temperature.

\[ \frac{1}{3} \]

\begin{figure}[h]
\centering
\includegraphics[width=\textwidth]{figure3.5.png}
\caption{Functional dependence of organic matter decay and denitrification rates on soil water-filled pore space}
\end{figure}

The temperature and moisture rate reduction functions are used independently of each other, thus actual rates are calculated by multiplying the maximum rate constants with both functions.

Daily input data of grass residues were taken from a grass growth model of Verberne (1992), which uses the same daily weather data and the same fertilizer (and urine) N inputs as SONICG.

References


3.5 Plant uptake of nutrients

This module describes the grass uptake of mineral N and the concomittant uptake of some macroelements (K+Na, Ca+Mg, ...), based on the given composition of the grass. Input data on potential daily N uptake by the grass were taken from a grass growth model of Verberne (1992), which uses the same daily weather data and fertilizer N inputs as SONICG.
Uptake of dissolved cations and anions by the grass roots is calculated based on a diffusive flux through a water film to the root surface. The average diffusion length is calculated from root surface area and water-filled porosity. The maximum nutrient supply by diffusive flux is limited to 10% of the total amount of a nutrient, present in the water phase plus the solid phase. Uptake of N as ammonium versus nitrate is based on the ratio of their concentrations in the soil solution.

For each rooted layer a potential daily N supply from the diffusive flux is calculated as well as the sum for all rooted layers. The maximum N uptake equals the potential daily uptake when the supply from the diffusive flux is (more than) sufficient. Then the same necessary fraction of the N supply is taken from all contributing layers. Else, the actual N uptake equals the calculated N supply. Nutrients are only taken up from layers where water is taken up. The maximum uptake of other nutrients than N is calculated on the basis of fixed ratios with actual N uptake. If the supply of other nutrients is insufficient to realize the fixed ratio with N uptake, only the actual supply is taken up.

References


3.6 Nitrification

Nitrification is described, without taking into account the population development of the nitrifying microbes, as a function of soil temperature T, pH of the soil solution, the fraction water-filled pore space, and total (dissolved + adsorbed) ammonium concentration. Nitrate and N$_2$O were regarded as the only products of nitrification, since sufficient data on nitrite as an intermediate product were lacking. The description includes inhibition of nitrification, at high concentrations of ammonium-N (NH$_4$-N). The final nitrification function developed is shown by equation 8:

$$\frac{dNH_4}{dt} = f(T) \cdot f(pH) \cdot f(WPFS) \cdot k_1 \cdot \frac{[NH_4]}{(k_2 + [NH_4]^2)} \quad (8)$$

In equation 8, [NH$_4$] represents the total amount of NH$_4$-N present in the soil water phase plus the solid phase (in mmol per kg moist soil). Experimental data of Mahli and McGill (1982) were used to develop equation 8. The 'constant' k$_1$ could as follows be fitted as a function the Cation Exchange Capacity (CEC, in meq per 100 g dry soil) of the soil: k$_1$ = 12.3 - 2.9 * log(CEC), for Canadian soils, but for Dutch soils we used: k$_1$ = 20.0 - 3.0 * log(CEC), because in Dutch soils larger nitrifier populations and nitrification rates were expected as a result of generally higher ammonium inputs. For k$_2$ a constant value of 195 (mmol NH$_4$)² (kg wet soil)$^{-2}$ was fitted.

The temperature dependence of nitrification is described by equation 8a:

$$f(T) = 3.15e-3 \cdot T^2 / (1 + 1.0e-7 \cdot T^2) \quad (8a)$$

The optimum temperature for Dutch conditions is taken about 5 °C higher than for Canadian conditions of Mahli and McGill, because of an expected adaptation of nitrifiers to the different soil temperatures of both countries.

The pH-dependence of nitrification is given by a sigmoidal function, with half the maximum rate at pH 4.7, as given by equation 8b:

$$f(pH) = 1/(1 + 5 \cdot 10^4 \cdot 10^{pH}) \quad (8b)$$
Thus \( f(pH) \) reflects that nitrification is generally rapid at \( pH > 6 \) and slows down as a result of acidification caused by the nitrification process. Since soil solution pH values are calculated in SONICG by the chemical equilibria module, the model can account for pH effects on nitrification.

The dependence of nitrification on WFPS has the same optimum as organic matter decay, but, as oxygen is essential for nitrification, the rate goes to zero at \( f(WFPS) = 1 \), as shown in Figure 3.6. \( f(WFPS) \) is described by equation 8c:

\[
f(WFPS) = \left( \sin(\pi \cdot WFPS^{1.75}) \right)^{1.75}
\]

As already mentioned by Mahli and McGill (1982), population development of the nitrifying microbes should be taken into account when a large amount of ammonium is added to the soil at one moment. Table 9 of their paper indeed showed increased average rates of nitrification in such cases. For grazed grassland, the area affected by urine gets such high amounts of ammonium that population development of nitrifiers should be modeled. A second module for nitrification, which includes the population development of ammonium-oxidizers and of nitrite-oxidizers has been made (Van Faassen, unpublished), but has not yet been included in SONICG.

Modeling the development of both nitrifying populations gives insight into conditions that may cause temporary accumulation of nitrite (Belser, 1979), as has been found in experiments where soil was incubated after addition of artificial urine (Monaghan & Barraclough, 1992). Thus, temporary accumulation of nitrite in the soil can be explained from conditions that cause a slower development of nitrite-oxidizing bacteria than of ammonium-oxidizing bacteria. Such inhibitory conditions will often be present in urine spots and should be taken into account when sufficient data are available.

![Figure 3.6](image)

**Relative rate**

**Total Nitrification**

**NO\(_3\)** production

**N\(_2\\)O production

Figure 3.6. Functional dependance of total nitrification, nitrate production and \( N_2O \) production rates on soil water-filled pore space.

**Production of \( N_2O \) during nitrification**

A production of \( N_2O \) by "nitrifier-denitrification" has been shown to occur even at atmospheric oxygen pressure (pO\(_2\)), and to increase with decreasing pO\(_2\) (Blackmer et al., 1980; Goreau et al.,
1980; Klemedtsson et al., 1988; Poth & Focht, 1985). On the field-scale pO₂ is a function of WFPS and to simulate the mentioned literature data at the field-scale we described the fraction of ammonium-N transformed into N₂O in layer i, α(i), by equation 9:

$$\alpha(i) = 0.005 + 0.4\times[(WFPS(i)-0.6)/0.4]^3$$

(9)

Equation 9 is used when WFPS(i) > 0.6 and when WFPS(i) < 0.6, α(i) is given a constant value of 0.005. The relative production rates of N₂O-N and nitrate-N by nitrification as a function of WFPS are shown in Figure 3.6.

As mentioned for aerobic organic matter degradation, the exact form of f(WFPS) for nitrification and N₂O production will still depend on a number of variables as soil type, soil dry bulk density and the rate of oxygen consuming processes. In case of a high input of NH₄-N, the oxygen need for high nitrification rates may contribute considerably to total oxygen consumption of the soil.

References


Goreau, T.J., 1980. Production of N₂ and N₂O by nitrifying bacteria at reduced concentrations of oxygen. Ibid., 526-532.


3.7 Denitrification

Starting point in modeling denitrification is the assumption that the aerobic biodegradation of organic matter is partly taken over by denitrification with decreasing pO₂ in soil. Thus, at the field-scale both soil respiration and denitrification can be described as functions of WFPS (Figure 3.5). As shown in Figure 2.2, N₂O is taken as the only and obligatory intermediate product in the denitrification of nitrate to N₂. When nitrate is fully reduced to N₂, the nitrate disappearance rate can be described as a fraction of organic carbon degradation rate, dC/dt, as shown in equation 10, derived from experimental data of Mahli et al. (1990):

$$-5\times d(NO_3)/dt = 10\times d(N_2)/dt = NO_3/(k_{d1} + NO_3)*f(WFPS)*4*dC/dt$$

(10)

Thus denitrification follows Michaelis-Menten kinetics, with k_{d1} the amount of nitrate at which half the maximum rate is reached at optimal water content, when f(WFPS) = 1. The value of dC/dt is calculated in the MIT module.
The dependence of denitrification rates on WFPS, above optimum WFPS for aerobic degradation, as used in SONICG is shown in Figure 3.5. The form of \(f(WFPS)\) is related to the decrease of the oxygen supply rate by gas diffusion, which is a function of gas-filled pore space, GFPS \((= 1 - WFPS)\). As already mentioned for aerobic degradation, the exact form of \(f(WFPS)\) for denitrification can also depend on soil type and on soil water hysteresis effects (Sexstone et al., 1988; Groffman and Tiedje, 1988).

The use of nitrate and of \(N_2O\) as electron-acceptors competing with \(O_2\) is calculated on the basis of electronequivalents (Cho, 1982). The factors 5, 10 and 4 in equation 10 are conversion factors from moles to electron-equivalents. When \(N_2O\) as well as \(N_2\) are produced the left-hand side of equation 10 becomes:

\[ -4\frac{d(NO_3^\text{-})}{dt} + 2\frac{d(N_\text{2})}{dt} \quad (10a) \]

The maximum amount of nitrate that can be denitrified per day is limited in the model to 25% of the amount present.

The ratio of \(N_2O\) production to \(N_2\) production

The ratio \(N_2O/N_2\) depends on the fractions of electrons accepted by nitrate and by \(N_2O\), respectively, and is known to increase with decreasing pH. The production rates of \(N_2O\) from nitrate and of \(N_2\) from \(N_2O\), in terms of electron equivalents, can be described by the first-order rate equations 11a and 11b:

\[ r_1 = -\frac{d(NO_3^\text{-})}{dt} = k_a \cdot [NO_3^\text{-}] \cdot 4 \quad (11a) \]
\[ r_2 = \frac{d(N_\text{2})}{dt} = k_b \cdot f(pH) \cdot [N_2O] \cdot 2 \quad (11b) \]

Thus, the reduction rate of nitrate is assumed to be independent of pH, whereas the reduction rate of \(N_2O\) is known to depend on pH (Blackmer & Bremner, 1978). The fraction of electrons going to \(N_2O\), \(f_{rel}(N_2O)\), is equal to \(r_2(r_2+r_1)\) and by substituting \(r_1\) and \(r_2\) from equations 11a and 11b we get:

\[ f_{rel}(N_2O) = f(pH) \cdot [N_2O]/(f(pH) \cdot [N_2O] + (k_a/k_b) \cdot 2\cdot[N_\text{3}]) \quad (11c) \]

When the total denitrification rate is known from equation 10, multiplication with \(f_{rel}(N_2O)\) from equation 11c gives the amount of \(N_2O\) that is reduced to \(N_2\), after conversion from electron equivalents to moles. Subtraction of \(N_2O\) reduction from total denitrification gives nitrate reduced to \(N_2O\).

From data of Blackmer & Bremner (1978) we derived that \(f(pH)\) can be described by equation 11a:

\[ f(pH) = 10^{(pH \cdot 6.5/9)} \quad (11d) \]

\(N_2O\) in the gas phase is considered to be as available as dissolved \(N_2O\) to accept electrons, because the exchange of \(N_2O\) between gas and water phase is assumed to be very rapid. Thus, the total amount of \(N_2O\) in a layer is used to calculate the \(N_2O\) concentration in equation 11c.

An alternative way is to describe the ratio of the first-order rate constants for \(N_2O\) and nitrate reduction, respectively, as can be derived from equations 11a and 11b and is shown in equation 11d:

\[ 2\cdot r_2[N_2O] = (k_b/k_a) \cdot f(pH) \cdot r_1[N_\text{3}] \quad (11e) \]
Again, the absolute amounts of nitrate and of N$_2$O reduced during denitrification can be calculated from equations 10 and 11e.

References


3.8 Gas transport

Gas diffusion in the soil is calculated with a one-dimensional, fully implicit transport model using zero- and first-order reaction terms. Equilibrium with the water phase is simulated with Henry constants, which are used as first-order withdrawal terms, whereas the amount in solution at t-1 is a zero-order input term. Equation 12 generally describes the change in concentration of a gas during a timestep for each soil layer:

$$\frac{\partial C_g}{\partial t} = D_{\text{eff}} \frac{\partial^2 C_g}{\partial z^2} - V_{\text{adv}} \frac{\partial C_g}{\partial z} + R_0 - R_1 \cdot C_g$$  (12)

The right-hand side of equation 12 consists of a diffusion term, an advection term, a zero-order production term and a first-order withdrawal term. Gas transport is calculated for the gases CO$_2$, NH$_3$ and N$_2$O. For CO$_2$, the production term $dC/dt$ from the decay of organic matter, as calculated by the MIT module, is included as a zero-order input into each layer. For N$_2$O, the production from nitrification plus denitrification is included as zero-order input to each layer, whereas the consumption of N$_2$O by denitrification to N$_2$ is included as a first-order withdrawal from each layer.

The effective gas diffusion coefficient, $D_{\text{eff}}$ ($m^2 d^{-1}$), for each interface in the soil column is calculated from equation 13:

$$D_{\text{eff}} = a \cdot D_{\text{air}} \cdot (\text{GFPS}_{\text{avg}})^b$$  (13)

where $D_{\text{air}}$ is the binary diffusion coefficient for the gas in free air and GFPS$_{\text{avg}}$ is the average gas-filled porosity of the two layers that border the interface. The constants a and b depend on the soil type (Washington et al., 1994; Bruckler et al., 1989; Campbell, 1985).

Bakker et al. (1987) summarize a lot of gas diffusion measurements on different Dutch soils. Variations in the total porosity of a soil can occur as a result of compaction. The effect of total porosity is included in constant a. The constant b reflects the continuity of pores. Blockage of pores by water has been shown to depend on the way of moistening. Ball (1981) found higher $D_{\text{eff}}$ values in a silt loam at pH 3.7 when soil of pH 4 was moistened than when saturated soil was dried to pH 3.7. This hysteresis effect can be important in denitrification.

Often, pores in the soil are not homogeneously distributed and pore systems of different order may exist together. This can be the case in aggregated soils, where large inter-aggregate pores and small intra-aggregate pores can be present. The same type of differences can be found in
rooted versus unrooted soils or between soils with and without worm holes. Soils that form cracks upon drying may allow gas transport already at low overall GFPS.

In SONICG equation 13 was used for $D_{eff}$ of a sandy soil, taking $a = 1$ and $b = 2$, as a best fit to measurements of Bruckler et al. (1989). The soil column is assumed to be closed for gas transport at the bottom. However, gases dissolved in water can leave the soil column at the bottom with drainwater. The model includes a gas layer of 2.5 cm on top of the soil column, which is used only as a "buffer" for gas transport between the soil and the atmosphere. This gas layer more or less simulates the gas layer in the field situation, where the horizontal component of the windspeed is very low. Thus, effects of windspeed and of atmospheric pressure variations are not considered.

References


Ball, B.C., 1981. Pore characteristics of soils from two cultivation experiments as shown by gas diffusivities and permeabilities and air-filled porosities. Journal Soil Science 32: 483-498.


3.9 Transport of dissolved substances

Water flow between the segments of the schematization is provided by the hydrological model. Since the dissolved concentrations of all components at t-1 are known (from the last performed equilibrium calculation), the mass transported between segments i and i+1 is, when the flow is from segment i to segment i+1 (downward flow):

\[ M_{\text{trans}(i,i+1)} = \text{flow}_{i,i+1} \times C_{t-1} \]  

(implicit transport)

When the flow is upwards, then

\[ M_{\text{trans}(i+1,i)} = \text{flow}_{i+1,i} \times C_{t-1} \]

However, this approach is only valid when the flow in the timestep is small compared to the volume of water in the source segment. This is generally true for upward flow, but not for downward flow. When this is not the case, then an implicit transport scheme is adopted:

\[ M_{\text{trans}(i,i+1)} = \text{flow}_{i,i+1} \times C_{t} \]  

(implicit transport)

$C_{t}$ is calculated assuming conservative mixing behavior for all dissolved components with a numerical onedimensional transport model.

For each segment the mass balance for each component is (downward flow):

\[ M_{c,t} = M_{c,t-1} + M_{\text{trans}(i-1,i)} - M_{\text{trans}(i,i+1)} \]

Since the transport scheme is numerical, numerical dispersion is introduced. The size of the
numerical dispersion depends on the segment size, and the flow rate. The calculation assumes the numerical dispersion to account for diffusion/dispersion in the soil solution, so no diffusion/dispersion term is included in the transport equation.

3.10 Chemical equilibria

Each layer of the physical schematization is considered to be an equilibrium system, where cation exchange, complexation in solution, precipitation/dissolution of calcite and the exchange of gases between the soil porewater and the soil atmosphere is in chemical equilibrium each timestep. The chemical equilibrium system distinguishes components (building blocks of matter), species (the existing substances) and phases (chemically homogeneous, physically distinguishable parts of an equilibrium system). The components of the equilibrium system which are used in SONICG are: H₂O, H⁺, Ca**, K**, NH₄⁺, NO₃⁻, N₂O, CO₃⁻, Cl⁻, CEC (cation exchange capacity) and Ig (inert gas, i.e. all gasses except the explicitly modeled gasses: CO₂, NH₃ and N₂O). Species in the water phase are: H₂O, H⁺, OH⁻, NH₄⁺, NH₂⁺, N₂O, CO₂, Cl⁻, HCO₃⁻, CO₃⁻, NO₃⁻, K⁺, Ca**, CaHCO₃⁻, CaCO₃⁰.

Cation exchange is calculated with 4 cations: H⁺, K⁺, Ca** and NH₄⁺. Cation exchange is modeled according to the so-called Gapon convention. For the reaction of replacing sorbed calcium with potassium the equilibrium relation is:

\[ \text{Ca}_5\text{CEC} + K^+ = \text{KCEC} + 0.5 \text{Ca}^{**} \]

The Gapon exchange constant for this reaction is taken as 5.7 l⁵.mol⁻⁵. The same reaction with ammonium instead of potassium has an exchange constant of 2.1 l⁵.mol⁻⁵. For H⁺ the constant is 3000 l⁵.mol⁻⁵.

The solubility product of calcium carbonate is taken as 10⁻⁶.05 mol².l⁻² (T = 25 °C, 1 atm, at ionic strength I = 0), which makes calcium carbonate more soluble than the theoretical solubility (Kₚ = 10⁻⁶.35). It is our experience that calculations with this increased solubility give more realistic results.

The Henry equilibrium constants used for the gasses are: for CO₂: Kₖ = 10⁻¹.48 l.mol⁻¹, for NH₃: Kₖ = 10⁻¹.76 l.mol⁻¹, and for N₂O: Kₖ = 10⁻¹.615 l.mol⁻¹.

Temperature corrections for equilibrium constants are performed using the van 't Hoff relation:

\[ \ln K_T - \ln K_0 = -(1/T - 1/T_0) \times \Delta H/R \]

Here T = temperature in °K, T₀ = 298.15 °K, ΔH = the enthalpy of the reaction (J.mol⁻¹) and R the universal gas constant (8.3144 J. °K⁻¹.mol⁻¹).

Ionic strength effects are calculated with the extended Debye-Hückel equation:

\[ \log \gamma = - A \times z^2 \times \sqrt{l} / (1 + B \times \delta \times \sqrt{l}) \]

where γ is the activity coefficient of an ionic species, z is the charge of that ionic species, l is the ionic strength of the solution, A and B are temperature-dependent constants for water (at 25 °C, A = 0.509, B = 0.328), and δ is an ion-specific constant, describing the effective hydrated ionic radius in Ångstrom (e.g. Ca**) = 6). Activities aᵢ of chemical species are used in all the equilibrium calculations, with aᵢ = γᵢ * Cᵢ.
3.11 Some first simulation results from SONICG

Input of the model

Daily C and N inputs with grass residues as well as potential daily N uptake by the grass have been taken from the model for mown grassland of Verberne (see Chapter 4). The simulations have been run for a sandy soil and the same daily weather conditions ("Wageningen, 1989") were used as in the model of Verberne.

Two preliminary simulation runs

Since the programming of SONICG was completed only recently (November '94), we were not yet able to test the model extensively. A first impression that the model functioned rather well was obtained from two simulations that are discussed in the following.

One simulation run was made for mown grassland, fertilized at a rate of 480 kg N per ha. A second simulation run was made where, in addition to mineral N applications, urine N was applied to mown grassland on day 140 at a rate of 420 kg per ha. The second run should simulate a urine spot in grazed grassland. However, effects of grazing on grass development (Chapter 5) were not yet included. These simulation runs were used to test the overall performance of SONICG and more specifically some of the modules, as discussed under model output.

Output of the model

The model simulation output has two possibilities:
- some parameters of interest can be followed on a daily basis on the PC screen during the simulation run for a year. Thus, the variation of some key parameters is made visible;
- pool sizes can be given at the end of a simulation run, or even on a daily basis, and yearly flows can be calculated as well as balances for water, carbon, nitrogen and other elements.

The following parameters have been selected for dynamic representation on the PC screen:
- pH of the soil solution for each layer. This parameter is a sensitive element for the performance of the soil chemistry module. The limited pH-buffering capacity of the soil solution led to pH-oscillations when urine spots were simulated. The emission of CO$_2$ led to rapid increase of pH, which promoted rapid NH$_3$ emission, the latter causing a rapid pH decrease. To limit these oscillations, the percentage of a gas that could leave the water phase during a timestep was limited to 60%;
- water-filled soil porosity for each layer was chosen because it affects the relative activities of aerobic degradation, nitrification and denitrification via the rate-reduction functions f(WFPS). The maximum water holding capacity of the upper three soil layers had to be decreased to limit denitrification for a sandy soil to realistic values;
- nitrate concentration in the soil solution for each layer was chosen because it is a prerequisite for N$_2$O production via denitrification;
- the total amount of N$_2$O for each layer was chosen because it forms the driving force for the emission of N$_2$O.

The day-to-day variation of these four parameters for each soil layer gives a rapid and informative view of "what is going on" during a simulation year.

Emission of N$_2$O

The main parameter of interest, the daily emission rate of N$_2$O from the soil to the atmosphere is shown on the screen in a fifth graph. In this case the daily values are plotted as a continuous graph that develops on the screen during the simulation. Thus, at the end of a simulation run
this graph on the screen shows the pattern of N₂O emission during the year. Figure 3.7 gives an example of two simulation runs for grassland without and with application of urine at day 140. As shown in Figure 3.7, a much larger N₂O emission results from urine treated grassland than from the control grassland with mineral N fertilizer only.

Figure 3.7. Daily fluxes of N₂O-N from grassland unaffected by urine and from urine spots, as simulated by SONICG for a sandy soil and weather conditions of Wageningen 1989. Urine was applied on day 140, at a dosage rate corresponding to 420 kg of N per ha of urine-affected area.

Figure 3.7 shows that the large peaks in N₂O emission appeared more than 100 days after urine application. This can be explained from the fact that complete nitrification of NH₄-N from urine takes considerable time. In autumn high nitrate concentrations will thus be present in the soil, leading to N₂O production when rainfall makes soil conditions conducive to denitrification. The smaller peaks in N₂O emission in spring are related to the application of mineral fertilizer N, and are partly the result of N₂O production by nitrification. The model may calculate how much nitrification and denitrification each contributed to the production of N₂O. The model may also calculate how much of this N₂O is reduced to N₂ and how much is leached with water at the bottom of the modeled profile.

It must be stressed that the result shown in Figure 3.7 is just a first preliminary result of SONICG, without possible adjustments of model parameters. However, the characteristic peaks that were found in the field are also shown in these first results.

The model can even produce a screen graph of both N₂O and NH₃ emission. The emission of CO₂ can also be presented on the screen, but because of the different scale cannot be combined with the graphs of the N emissions.

3.12 Evaluation, validation and further development of SONICG

The SONICG model will be used to simulate the emission of N₂O from the grassland locations where extensive measurements have been made by Velthof (NOP project 852073). By using the local input data and weather conditions gathered by Velthof, several simulation outputs of
SONICG will be compared with the actual values, for instance of N$_2$O emission rates, soil water contents and N uptake by the grass. The evaluation, validation and further development of SONICG will be made as part of the DLO Research Programme on Climate Change 1995-1999.
4. Simulation of carbon and nitrogen turnover in mown grassland

4.1 Introduction

A mechanistic simulation model for carbon and nitrogen turnover in mown grassland, developed by Verberne (1992), was run for three different years and for two soil types, a sandy soil and a clay soil. The model made use of daily weather data from a weather station near Wageningen for three different years: a dry year (1982), an average year (1989) and a wet year (1965). The results of these six simulation runs showed that conditions favouring good grass production led to low N losses to the environment, whereas conditions unfavourable to or limiting grass production led to large accumulations of mineral N in the soil, with the risk of large N losses by nitrate leaching and/or denitrification. However, the model considered nitrate leaching as the only nitrogen loss to the environment and showed it to vary potentially from 0-250 kg ha\(^{-1}\), depending on the interaction of weather conditions, grass development and C and N turnover in the system.

4.2 Results and discussion

Yearly N balances and N flows

Total N input to grassland (Table 4.1) differed mainly because of differences in fertilizer application, based on the number of grass harvests. Only two or three harvests were possible in the dry year 1982, compared to eight harvests for both soils in the wet year 1965 (Table 4.2).

The availability of water was a main limiting factor for grass production in these simulations, especially for the clay soil. The modeled soil profiles had a storage/water supply capacity of about 120 mm for the clay soil and 220 mm for the sandy soil.

Except for clay soil in 1982, N uptake by the grass was (much) higher than the total N input (Table 4.3). This showed that under favourable conditions for grass growth this crop could represent a large sink for mineral N. Even the amount of N removed with harvested grass was higher than the total N input in two cases (Table 4.2).

Leaching of nitrate below 100 cm (Table 4.2) was relatively low at the end of each year, because the main part of nitrate leaching should occur in the period January till May of the next year.

The potential for further N losses, either by leaching or by denitrification, was indicated by the increased storage of mineral N in the soil profile at the end of 1982 for both soils and at the end of 1989 for the clay soil (Table 4.2). Unfavorable conditions for grass growth, especially prolonged drought periods, increased senescence of the grass and the recycling of dead grass litter in 1982 for clay soil and in 1982 for both soils (Table 4.3).

Net mineralization of N from soil organic matter pools (microbial biomass and humus) was always larger in the clay soil than in the sandy soil (Table 4.3). This can be explained from a much higher amount of soil organic nitrogen in the clay soil than in the sandy soil: about 7000 and 4000 kg ha\(^{-1}\), respectively. The highest net N mineralization for the clay soil in 1989 might be explained by the higher average temperature and by the larger input of grass residues, the latter leading to more (N mineralization from) microbial biomass and young humus (Table 4.3).
The first step in turnover of grass litter in the soil always caused some net immobilization of mineral N in the clay soil, but only once (1989) in the sandy soil (Table 4.3). This difference between soil types had to do with the lower average C/N ratio of the soil microbial biomass, growing on grass litter, in the clay soil than in the sandy soil. A difference in C/N ratio of grass residues also played some role: under unfavorable conditions for grass growth less N from senescent litter could be reused by growing plant parts and then litter with a lower C/N ratio was recycled into the soil, and increased the chance of net N mineralization.

Table 4.1. Rainfall (RAIN), average daily air temperature (TEMP) and N input (kg ha\(^{-1}\)) with mineral fertilizer (FERT-N) and with rainfall (RAIN-N).

<table>
<thead>
<tr>
<th>SOIL TYPE</th>
<th>SAND</th>
<th>SAND</th>
<th>SAND</th>
<th>CLAY</th>
<th>CLAY</th>
<th>CLAY</th>
</tr>
</thead>
<tbody>
<tr>
<td>RAIN (mm)</td>
<td>566</td>
<td>697</td>
<td>989</td>
<td>566</td>
<td>697</td>
<td>989</td>
</tr>
<tr>
<td>TEMP (°C)</td>
<td>9.7</td>
<td>10.3</td>
<td>8.6</td>
<td>9.7</td>
<td>10.3</td>
<td>8.6</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>N-INPUT</th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>FERT-N</td>
<td>400</td>
<td>480</td>
<td>480</td>
<td>320</td>
<td>400</td>
<td>480</td>
</tr>
<tr>
<td>RAIN-N</td>
<td>28</td>
<td>35</td>
<td>49</td>
<td>28</td>
<td>35</td>
<td>49</td>
</tr>
<tr>
<td>TOTAL-N</td>
<td>428</td>
<td>515</td>
<td>529</td>
<td>348</td>
<td>435</td>
<td>529</td>
</tr>
</tbody>
</table>

Table 4.2. N output with grass harvests (HARV-N) and nitrate leaching (LEACH-N), and changes in storage of N in grass (GRASS-N), N in soil organic matter (SOILORGN) and in soil mineral N (SOILNMIN) in (kg ha\(^{-1}\)).

<table>
<thead>
<tr>
<th>HARVESTS</th>
<th>3</th>
<th>6</th>
<th>8</th>
<th>2</th>
<th>4</th>
<th>8</th>
</tr>
</thead>
<tbody>
<tr>
<td>HARV-N</td>
<td>261</td>
<td>487</td>
<td>584</td>
<td>134</td>
<td>339</td>
<td>602</td>
</tr>
<tr>
<td>LEACH-N</td>
<td>20</td>
<td>7</td>
<td>30</td>
<td>25</td>
<td>20</td>
<td>12</td>
</tr>
<tr>
<td>GRASS-N</td>
<td>+32</td>
<td>+34</td>
<td>-1</td>
<td>+26</td>
<td>+19</td>
<td>0</td>
</tr>
<tr>
<td>SOILORGN</td>
<td>+24</td>
<td>-13</td>
<td>-84</td>
<td>-70</td>
<td>-39</td>
<td>-91</td>
</tr>
<tr>
<td>SOILNMIN</td>
<td>+92</td>
<td>-5</td>
<td>-4</td>
<td>+233</td>
<td>+96</td>
<td>+1</td>
</tr>
<tr>
<td>TOTAL-N</td>
<td>428</td>
<td>515</td>
<td>529</td>
<td>348</td>
<td>435</td>
<td>529</td>
</tr>
</tbody>
</table>
Table 4.3. Internal N flows (kg ha\(^{-1}\)): N uptake by the grass (N-\text{UPTAKE}), N recycled in the form of above-ground litter (aLITT-N), below-ground litter (bLITT-N), net mineralization from soil organic matter (MINER-N), and nett immobilization as a result of crop residu turnover (IMMOB-N).

<table>
<thead>
<tr>
<th></th>
<th>477</th>
<th>713</th>
<th>707</th>
<th>298</th>
<th>587</th>
<th>731</th>
</tr>
</thead>
<tbody>
<tr>
<td>N-\text{UPTAKE}</td>
<td>477</td>
<td>713</td>
<td>707</td>
<td>298</td>
<td>587</td>
<td>731</td>
</tr>
<tr>
<td>aLITT-N</td>
<td>170</td>
<td>180</td>
<td>114</td>
<td>121</td>
<td>216</td>
<td>120</td>
</tr>
<tr>
<td>bLITT-N</td>
<td>15</td>
<td>13</td>
<td>9</td>
<td>17</td>
<td>14</td>
<td>9</td>
</tr>
<tr>
<td>MINER-N</td>
<td>160</td>
<td>206</td>
<td>236</td>
<td>236</td>
<td>291</td>
<td>263</td>
</tr>
<tr>
<td>IMMOB-N</td>
<td>0</td>
<td>0</td>
<td>28</td>
<td>29</td>
<td>22</td>
<td>42</td>
</tr>
</tbody>
</table>

Distribution of mineral N fertilizer applications in the simulations: 120 - 100 - 100 - (80) - (80) - kg ha\(^{-1}\). The first application was when the sum of average daily temperature was more than 280 °C. The other applications were after each harvest of about 2000 kg grass dry matter per ha.

Nitrogen turnover in mown grassland: summary of results from six simulation runs with the model FSE-GRASS-SOM-WATER (Verberne, 1992)

Dynamics of carbon and nitrogen within a year (some examples)

The conditions for grass growth were very favorable during the wet year 1965, as shown by the almost linear increase of the cumulative amount of carbon assimilated by the grass (Figure 4.1, top). Very unfavorable growth conditions however, around week 30 in the dry year 1982, almost stopped gross C assimilation by the grass and resulted in a very limited grass production (Figure 4.1, bottom). The amounts of assimilated C lost by respiration and by senescence, the net amount of C resulting in the grass, the amount of grass-C harvested and the amount of grass-C present in the standing crop are also indicated in Figures 4.1a and 4.1b.

The (cumulative) N uptake by the grass on the clay soil in 1965 and 1982, respectively, showed the same trends as the (cumulative) gross C-assimilation (Figures 4.2, top and 4.2, bottom compared to 4.1, top and 4.1, bottom). Thus, under favorable conditions for grass growth, as in 1965, almost all the available mineral N is taken up by the grass, leaving very low concentrations of mineral N in the soil from week 35 onwards. Grass growth being impaired by drought in 1982, on the other hand, led to very limited N uptake and mineral N concentrations in soil increased almost continuously. Such high concentrations of soil mineral N, in spite of limited N fertilization in 1982, make large N losses very likely when later on conditions become favorable for denitrification or leaching of nitrate.
Figure 4.1. Simulated carbon turnover by grass in mown grassland, in a favorable (1965, top) and in an unfavorable year (1982, bottom) for grass production.

**CUMASS** = grass-C (at t = 0) + cumulative gross C assimilation

**net CUMASS** = CUMASS - (respired C)

**CUMLIVEC** = net CUMASS - (grass-C recycled as dead litter)

**CUHARC** = cumulative harvested grass-C

**CGRASS** = carbon in the standing grass crop
Figure 4.2. Simulated nitrogen turnover by grass in mown grassland, in a favorable (1965, top) and an unfavorable year (1982, bottom) for grass production.

CUMNIN = mineral N (at t = 0) + fertilizer N + net N mineralization + N deposition with rainfall;
CUMNUP = cumulative N uptake by the grass;
net CUMNUP = CUMNUP - (grass-N recycled as dead litter);
CUHARN = cumulative grass-N removed by mowing;
NGRASS = N in the standing grass crop;
NMSOIL = mineral N in the 0-100 cm layer of the soil.
Figure 4.3. Simulated (cumulative) net N mineralization from soil organic matter and from grass residues, in three years with quite different weather conditions: 1989 ('average'), 1982 ('dry') and 1965 ('wet'); negative values indicate net N immobilization.

Figure 4.4. Simulated rates of N turnover in a sandy grassland soil: input of N from grass residues (NRESINPUT), N mineralization from grass residues (NMINRES) and from soil organic matter (NMINSON) and total N mineralization in the soil (NMINTOT).
The cumulative mineralization of N from soil organic matter and immobilization of N resulting from litter (dead grass residues) turnover in the clay soil is shown in Figure 4.3. Differences between the years are the overall result of differences in soil temperature and moisture and in the amounts and composition of grass litter inputs. For the sandy soil in 1989 the weekly(!) rates of N input with grass litter, of N mineralization from soil organic matter and of N immobilization/mineralization from grass litter is shown in Figure 4.4. A large N input with grass litter around week 34 indicates a period where considerable senescence of the grass occurred, leading to a peak in N mineralization from these residues around week 39. Till week 26 turnover of grass litter only led to N immobilization. Compared with a maximum daily(!) N uptake by the grass of about 6 kg ha$^{-1}$ in the simulations, it can be seen from Figure 4.4 that optimum grass growth also needs an external N supply in addition to N supply from the soil.

Conclusion

The results of six simulation runs for carbon and nitrogen turnover in mown grassland showed strong interactions between C and N turnover, by the grass and in the soil, with weather conditions that influenced the moisture status of the soil and the water availability to the grass. Therefore, only models that consider all these interacting factors together can give reliable estimates of N losses to the environment as a function of N input and other management factors.

Reference

5. Simulation of grazed grassland

5.1 Introduction

The effects of nitrogen fertilizer application on grassland has been subject of study for many years now. Because of the complexity of the highly interacting systems, it is hard to study these effects through direct empirical research. Mechanistic models can be used to understand the behavior of the system under any given combination of management strategies.

The objective of this chapter is to give a description of the grazing submodel which is used as an addition to the mechanistic grass growth model as proposed by Verberne (1992). The main difference between grazing and mowing is the fact that in case of grazing nitrogen is partly returned to the pasture in urine and feces spots. In these spots the N input is very high: about 40-200 g m\(^{-2}\) (Lantinga et al., 1987). Because most of the excreted N is present in urine, and urine spots cover an about 10 times larger area than feces spots, the latter are neglected in the model. Another difference is a higher loss of litter in case of grazing then in case of mowing.

Figure 5.1. shows the structure of the submodel, including the links with the existing submodels (Verberne, 1992).

5.2 Modifications compared to mown grassland

Fertilization

The fertilization of the soil takes place one day after a harvest, or one day after a grazing period. The fertilization scheme, as recommended by Verberne (1992), is used to determine the amount of nitrogen fertilizer application. To simulate the effect of urine two model runs are necessary. The first run determines the grazing scheme (number of grazing periods and total
Grazing by dairy cows

When the option switch ‘GRAZE’ is set to one, the animal intake will begin when more than 3500 kg DM ha\(^{-1}\) is available. The duration of the grazing period equals the number of days it takes to reduce the standing crop to less than 3500 kg DM ha\(^{-1}\), increased with a variable number of days which is the minimum grazing period (parameter ‘DUGR’). The stocking density is calculated using equation 1:

\[
\frac{\text{n} - (\text{WSHBT} - 2000)}{(\text{DUGR} \times \text{G}_{\text{max}})}
\]

Where \(\text{G}_{\text{max}}\) is the maximum daily animal intake (kg DM animal\(^{-1}\) day\(^{-1}\)) and WSHBT is the standing crop (kg DM ha\(^{-1}\)).

The actual animal intake, \(\text{G}\) (kg DM animal\(^{-1}\) day\(^{-1}\)), is assumed to be determined by the maximum daily intake, \(\text{G}_{\text{max}}\) and the leaf area index, \(\text{L}\) (Thornley & Verberne, 1989):

\[
\text{G} = \text{G}_{\text{max}} \times \frac{\left(\frac{\text{L}}{\text{K}_{\text{g}}}\right)^{q}}{1 + \left(\frac{\text{L}}{\text{K}_{\text{g}}}\right)^{q}}
\]

\(\text{K}_{\text{g}}\) determines the half-maximum response (if \(\text{L} = \text{K}_{\text{g}}\), \(\text{G}\) equals \(\text{G}_{\text{max}}/2\)), and \(q\) determines the sigmoidicity of the response.

It is assumed that the material removed by the livestock (\(n\text{G}\)) is taken from the four age categories of the shoot material only. Let \(G_{\text{dmi}}\) denote the dry matter intake due to grazing of the age categories \(i\) (kg DM ha\(^{-1}\) day\(^{-1}\)):

\[
\text{nG} = \sum_{i=1}^{4} G_{\text{dmi}}
\]

The dry matter intake from age category \(i\) (\(G_{\text{dmi}}\)) depends on the leaf area index of age category \(i\) (\(L_i\)), and on a affinity-index (\(a_i\)) allowing different removal due to selectivity or availability:

\[
G_{\text{dmi}} = \frac{a_i \times L_i}{\sum_{i=1}^{4} a_i \times L_i} \times \text{nG}
\]
Carbon and nitrogen intake

The total carbon and nitrogen concentrations ($C_{plant}$ and $N_{plant}$) in the lamina, sheath and stem of the plant are determined by the total substrate carbon and substrate nitrogen content ($W_c$ and $W_N$, respectively), the total structural dry matter ($W_T$) and the total dry matter of shoot parts of the plant ($W_T$; consisting of shoot and substrates):

$$C_{plant} = \frac{W_c + f_c W_g}{W_T} \quad \quad N_{plant} = \frac{W_N + f_N W_g}{W_T}$$

(5)

$f_c$ and $f_N$ are fractional C and N contents of shoot parts of the plants structural dry matter.

The animal intake of nitrogen and carbon per day is (using equations 3 and 5):

$$I_c = C_{plant} \cdot nG \quad \quad I_N = N_{plant} \cdot nG$$

(6)

The fraction of the N intake retained by the animal ($f_{N,ret}$) has a constant low value and can be used to calculate the retained N and C:

$$F_{N,ret} = f_{N,ret} \cdot I_N \quad \quad F_{C,ret} = \lambda_{CN,ret} \cdot F_{N,ret}$$

(7)

The parameter $\lambda_{CN,ret}$ is the C/N ratio of the retained biomass; the value of $\lambda_{CN,ret}$ is constant and reflects the C/N ratio of proteins, the main component of meat and milk.

The yield coefficient $Y_{animal}$ (kg C in product [kg C assimilated]$^{-1}$) is introduced to reflect the loss of C due to animal respiration. The respiration rate ($R_{animal}$ kg C ha$^{-1}$ day$^{-1}$) can now be calculated:

$$R_{animal} = F_{C,ret} \cdot \frac{(1 - Y_{animal})}{Y_{animal}}$$

(8)

Excretions

The total excreted N flux, $F_{N,exc}$, is defined as the difference between the nitrogen intake ($I_N$) and the retained N:

$$F_{N,exc} = I_N - F_{N,ret}$$

(9)

The amount of N excreted in urine is calculated with the use of a regression curve (Valk, 1991; Lantinga et al., 1987). With the $f_{N,urine}$ known, the expressions for the fluxes of C and N to urine and feces (kg C or N ha$^{-1}$ day$^{-1}$) can easily be defined (Thornley & Verberne, 1989):
\[ F_{\text{N,urine}} = f_{\text{N,urine}} \cdot F_{\text{N,ex}} \]  \tag{10}

\[ F_{\text{C,urine}} = \left( \frac{12}{14} \right) \cdot F_{\text{N,urine}} \]  \tag{11}

\[ F_{\text{C,faeces}} = l_{\text{C}} \cdot F_{\text{C,ret}} - R_{\text{animal}} - F_{\text{C,urine}} \]  \tag{12}

\[ F_{\text{N,faeces}} = (1 - f_{\text{N,urine}}) \cdot F_{\text{N,ex}} \]  \tag{13}

The C and N in urine are assumed to be present in the form of urea only and thus C in urine can be calculated (eq. 11) when urine N is known (eq. 10).

### Ammonia volatilization

The volatilization of ammonia is modelled after Bussink (1994). The relationship between the total \( \text{NH}_3 \) volatilization \( V \) kg N ha\(^{-1}\) yr\(^{-1}\) and the total N excreted over a year \( F_{\text{N,tot,ex}} \) can adequately be described by a power function of the type \( V = x \cdot F_{\text{N,tot,ex}}^y \). To determine the values of the parameters \( x \) and \( y \) datasets from 1987, 1988 and 1990 were used (Bussink, 1994):

\[ V = 7.70 \times 10^{-6} \times (F_{\text{N,tot,ex}})^{2.491} \]  \tag{14}

### 5.3 Modeling example

To run the model an extended set of input parameters is necessary. The parameter set can roughly be divided into: pasture data, hydrology parameters, grazing parameters, organic matter parameters and weather parameters. The model itself also requires a couple of control parameters which contain the names of in- and output files, the size of the integration step, the simulated year, the soil type (clay or sand) etc. All parameters (except the parameters necessary for the simulation of grazing) are listed by Verberne (1992).

During the model's run, the output, in the form of numerical values of variables, is produced. This output can be made graphical by using other software. The output variables can be selected by adjusting the model and re-compiling it. After linking the newly compiled version, a new set of output data can be calculated.

In this example different parameter sets were used to calculate the difference in the nitrogen balance between grazed and a mown grassland. In the first simulation run no grazing took place, just harvesting. The fertilization scheme according to Verberne (1992) was used to determine the inorganic nitrogen input. The first fertilization is when the temperature sum equals 280 °C. The next fertilizer applications take place after each harvest, but only if the precipitation equals 5 mm or more. The grass is harvested when more than 4000 kg DM ha\(^{-1}\) (shoot material) is available. After harvesting the shoot material is reduced to 2000 kg DM ha\(^{-1}\).

The nitrogen balance of a mown grassland field in 1989 is schematically shown in Figure 5.2. Each numerical value has the dimension kg N ha\(^{-1}\) year\(^{-1}\), and the fertilization scheme is put between parentheses.
When the option 'GRAZE' is set to one, animals will begin with the intake of grass when more than 3500 kg DM ha\(^{-1}\) is available. The urine and feces produced by the animals can optionally be returned to the pasture, but in this example the excreta are not fed back directly. Instead, the urine spots were simulated by an extra N input of 420 kg N ha\(^{-1}\).

Figure 5.3 shows the N balance of a urine spot in grazed pasture and Figure 5.4 shows the N balance for the area that was unaffected by urine. Effects of feces were not yet taken into account, but are presumed to be small. From the number of cow grazing days the area affected by urine spots can be calculated, when we assume that one cow affects 6.8 m\(^2\) of pasture daily by urine (Lantinga et al., 1987). Next, the N balance for a grazed pasture can be calculated by summation of N balances for urine spots and area unaffected by urine, on an area basis. For this example about 0.21 ha was affected by urine, with an input of 88.2 kg N, or 420 kg ha\(^{-1}\). Thus, the chosen dosage rate of urine-N in Figure 5.3 (420 kg ha\(^{-1}\)) was about 12 % less than it should have been.
Figure 5.3. Nitrogen balance of urine spots in a grazed pasture on a sandy soil for the year 1989.

Figure 5.4. Nitrogen balance of the area in a grazed pasture that is not affected by urine on a sandy soil for the year 1989.
Parameter values

Parameter values used in modeling effects of grazing are given in Table 5.1.

Table 5.1. Summary of parameter values used to simulate grazing

<table>
<thead>
<tr>
<th>Parameter values</th>
<th>Units</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>$G_{max}$</td>
<td>18.</td>
<td>kg DM animal$^{-1}$ day$^{-1}$ maximum daily intake</td>
</tr>
<tr>
<td>$DUGR$</td>
<td>10.</td>
<td>day         minimum length of grazing</td>
</tr>
<tr>
<td>$k$</td>
<td>1.</td>
<td>m$^2$ m$^2$ determines the half-maximum response</td>
</tr>
<tr>
<td>$q$</td>
<td>3.</td>
<td>-           sigmoidicity parameter</td>
</tr>
<tr>
<td>$a_i$</td>
<td>0.4, 0.3, 0.2, 0.1</td>
<td>- affinity index for leaf class $i$</td>
</tr>
<tr>
<td>$\lambda_{CNO_i}$</td>
<td>3.42</td>
<td>kg C [kg N]$^{-1}$ C/N ratio of retained biomass</td>
</tr>
<tr>
<td>$Y_{animal}$</td>
<td>0.5</td>
<td>kg C prod. [kg C substr.$]^{-1}$ carbon yield of assimilated C</td>
</tr>
<tr>
<td>$f_{\text{ret}}$</td>
<td>0.15</td>
<td>- nitrogen fraction retained in milk and meat</td>
</tr>
</tbody>
</table>

References


