Nitrogen Dynamics and Nitrate Leaching in Intensive Vegetable Rotations in Highlands of Central Java, Indonesia

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ABSTRACT

High rainfall intensity is major factor governing leaching process, where leaching is often the most important process of N loss from the field and lead to agricultural environmental pollution. In order to measure the movement of mineral-N in soil profile, a field research had been conducted in two sites of center vegetable farming area with six farmer cooperators in Central Java, Indonesia. Regular soil sampling was done from Improve Practice (IP) and Farmer Practice (FP) treatment for three planting seasons during 2007. Almost all treatments FP applied higher rate of N fertilizer compare to IP, but it was not reflected in N profile. Comparison of predicted and measured mineral N content was simulated using Burns α model, then the closeness of the estimation and measured calculated using Coefficient of Residual Mass (CRM) calculation as an indicator with 0 as ideal value. Out of 9 measurements of IP and FP treatment, eight and seven measurements had negative CRM representing slight overestimation. The NO$_3$-N loss estimated using the Burns α model for IP and FP was in average of 67% for IP and 71% for FP of total N fertilizer added or 67% for IP and 76% for FP of total-N surplus, respectively. The calculation of potential nitrate concentration (PNC) at 1 m soil depth at the end of the third season showed a high concentration with significant different of IP and FP having mean value of 59.8 and 82.5 mg N L$^{-1}$. From the gathered data it was obvious that over N fertilization had negative effect to agricultural environment.

Keywords: N loss, improve practice, farmer practice, over N fertilization

Kata kunci: Kehilangan N, Improve Practice, Farmer Practice, pemupukan N yang berlebih

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INTRODUCTION

N is a major plant nutrient required for high yields of agricultural crops. Nitrogen (N) is the nutrient that is most susceptible to transformations affecting plant availability. These transformations include mineralization, immobilization, nitrification, and denitrification, as well as leaching and NH₃ volatilization (Petersen et al. 1998). The N in NO₃ is soluble and mobile and susceptible to transport into groundwater, which has become increasingly degraded (Strebel et al. 1989; Spalding and Exner, 1993). Sources of NO₃ in agricultural soils are inorganic-N fertilizer, manure, crop residue, and soil organic matter. Agriculture is considered to be the primary contributor to NO₃ contamination of groundwater (Strebel et al. 1989; Fraters et al. 1998).

Nitrate leaching and contamination of groundwater and surface water has become a major environmental problem in Europe and the USA and also increasingly in Asian countries, including China and Indonesia due to over-application of N fertilizers and farmyard manure in rainfed areas (Kurokura et al. 2001; Di and Cameron 2002; Zhu and Chen 2002; Arsanti 2008). The threshold recommended by the World Health Organization for drinking water is 50 mg NO₃-L⁻¹, similar to a maximum admissible concentration defined by European Community (EC) legislation and the US Environmental Protection Agency (10 mg N L⁻¹). Rass et al. (1999) stated that NO₃ contamination of groundwater is closely related to corresponding agricultural management practices.

Nitrogen over-fertilization, particularly in vegetable production, occurs in many countries, including Indonesia. Excessive N fertilizer application is therefore very common, while N use efficiency is often low. Data gathered from farmers’ practices (Widowati et al. 2012), revealed an average ANUE value 16% for two consecutive years; therefore, more than 80% of N added is susceptible to loss. Application of N fertilizer in excess of crop requirements can result in the accumulation of NO₃-N in the soil profile (Gillard et al. 1995; Malhi et al. 2001). Although organic material is considered as a slow-release fertilizer, excessive application may cause unintended NO₃ leaching (Gerke et al. 1999) with the percolating water. Leaching is often the most important means of nitrogen loss from field soils (Aulakh et al. 2000; Chowdary et al. 2004). The movement of a solute is strongly interlinked with the movement of water. Nitrate movement depends on water movement below the rooting depth of the crops. A greater water supply than the crop requirement (by rain and/or irrigation) is a very simple indicator of the leaching potential (Song et al. 2009; Sipahutar et al. 2013).

Nitrate leaching in the Andisols of intensive vegetable production regions in Indonesia is hypothesized to be the primary N loss process, as they have low bulk density, are highly permeable and because the climate is monsoonal with high rainfall intensity (>2,500 mL yr⁻¹). However, no information is available to date on the impact of high organic and inorganic N fertilizer application on N leaching in intensive vegetable cropping systems in Indonesian (i.e., tropical) Andisols.

The objectives of this study were therefore to monitor NO₃-N profiles in intensively managed Andisols because leaching cannot be directly derived from such mineral N profiles. The second objective was to quantify NO₃ leaching losses from model calculations using a simple yet robust leaching model.

MATERIALS AND METHODS

Two Andisols located in Wonosobo and Kopeng were used for the measurement of N movement through the soil profile. Samples for initial soil characteristics were taken the day after harvest of the third planting season of the year 2006.

The soils under study have low clay contents. They contain allophone as an indicator of low bulk density soils, and the BD values were between 0.65 and 0.9 g cm⁻³ in the 25 cm upper layer (Table 1). Another physical characteristic of these soils is a high permeability index (from medium to fast), dominated by very fast pore drainage.

Soil Sampling for Physical Properties and Mineral N Profiles

Soil was collected at both locations from Improved Practice (IP) and Farmer Practice (FP) plots, established at each of three farmers’ fields, with each plot consisting of 10 individual beds. During 2007, the beds were not covered; however, in 2008, they were covered with plastic mulch to protect the beds from erosion.

The physical characterization of soils was performed at each farmer’s field before the start of the experiment, in layers of 25 cm to a depth of 1 m (Table 1). The soil sampling for mineral nitrogen profiles was conducted during the year 2007 with sampling intervals of 3 weeks. Each soil sample was collected using an auger at four soil depths: 0-25, 25-50, 50-
Maximum care was taken to minimize the contamination of samples by the soil material of the overlying layers. The sampling points were carefully selected to avoid contamination with recently applied mineral or organic fertilizer. The soil samples were then analyzed for \( \text{NH}_4^+ + \text{NO}_3^- \). Ammonium nitrogen was extracted with 1 M KCl followed by colorimetric analysis with a spectrophotometer at wavelength 636 nm, and \( \text{NO}_3^- \) N was extracted with 0.01 M CaCl\(_2\) followed by colorimetric analysis with a UV spectrophotometer at wavelengths of 210 and 270 nm (Hitachi U-2010, Japan).
To estimate the movement of \( \text{NO}_3^- \), we selected the Burns \( \alpha \) model for its simplicity and versatility. The Burns \( \alpha \) leaching model is an adaptation of the Burns leaching model (Burns 1974) and was chosen because it requires only readily available soil and meteorological data (De Neve and Hofman, 1998; Moreels et al. 2003; Chaves et al. 2006) as presented in Table 2. Of the numerous leaching models published, this model is one of the few that has been applied to actual field conditions (Scotter et al. 1993). Moreels et al. (2003) successfully used the Burns \( \alpha \) model to predict moisture and nitrate contents in bare fallow soils, and Chaves et al. (2006) used this model to predict N release from N-rich crop residues and organic wastes.

One major drawback of the Burns model is that no water content above field capacity can be simulated, thus limiting its use to light textured soils (Moreels et al. 2003). To adapt this model to other soil textures, the model was adjusted by adding the \( \alpha \) parameter, which denotes the proportion of water above field capacity that drains to the underlying layer (a varies between 0 and 1), which must be specified for each soil layer. This adjustment allows simulations of moisture contents between field capacity and saturation. The value of \( \alpha \) for each layer is obtained by calibrating the model using measured soil moisture contents. In this research, we used data from a bromide leaching experiment on these soils for the calibration of \( \alpha \) (data not shown). The calibration based on Br- leaching yielded a value of 1 for all layers and all plots, thus indicating very rapid leaching.

De Neve and Hofman (1998) extended the model with a mineralization module, which calculates and takes into account the N mineralization from soil organic matter and from added organic materials. Moreels et al. (2003) further adapted the model to also calculate N losses by denitrification. Because the model was developed for simulating leaching only in bare fallow soils, we need to correct the calculations for N uptake by the crop as follows: each day, the \( \text{NO}_3^- \) N concentration in the top layer was calculated, taking into account inputs by mineralization and fertilizer and outputs by leaching from the previous day. This daily \( \text{NO}_3^- \) N concentration was then diminished by the daily N uptake by the crop (calculated as total N uptake divided by the length of the cropping cycle in days; i.e., assuming a linear crop N uptake pattern) before calculations for that day were continued. The data on N mineralization from soil organic matter and organic materials were taken from Widowati et al. 2012, the total crop N uptake was taken from Widowati et al. 2011, and the denitrification rates were obtained from Anggria (2007). Data used in the running model were from the first and third planting seasons of the year 2007; i.e., only during the rainy season, as there was no leaching in the second planting season.

**Meteorological Data**

Meteorological data for 2007-2008, namely relative humidity, temperature, evapotranspiration, and cloudiness, were obtained from the closest climate station, while precipitation (which is much more spatially and temporally variable) was collected by the farmer co-operators directly. Data for the Kopeng site were obtained from Meteorological and Geophysical Agency Area II, which is located at Ungaran Sub-district (± 35 km distance), and climate data for the Kejajar-Wonosobo site were obtained from the Tambi Tea Plantation Climate Station, which is located less than 3 km from the Wonosobo sites.

**Estimation of Nitrate Concentrations in Soil Water at 1 m Depth**

A rough estimation of the nitrate concentration was obtained using the total N and water balance equation according to OECD (1999), as in Maeda et al. (2003). The equation for predicting the \( \text{NO}_3^- \) N concentration in soil water at 1-m depth is:

\[
PNC = \frac{PNP}{EW} \times 100
\]

where PNC (mg L\(^{-1}\)) is the potential nitrate concentration; PNP (kg N ha\(^{-1}\) yr\(^{-1}\)) is the potential nitrate nitrogen present in soil, which in this calculation represents the total N-balance within one year; and EW (mm yr\(^{-1}\)) is the excess water (the difference between water input and evapotranspiration). Evapotranspiration was calculated using the Penmann equation. Maeda et al. (2003) obtained PNP by subtracting the amount of N uptake, which refers to N in all parts of the crops removed from the plots from that of the total N application in a year. However, the authors assumed all N from the applied organic matter in their study to be available within one observation, which obviously is an oversimplification. Here, the total N-balance for the calculation of the PNP was obtained directly from the N-balance results (Table 3). Based on these N-balance calculations, the data used here to calculate the PNP more realistically represent the potential nitrate present in the soil, as
the parameters of N-balance were calculated in a more comprehensive way than in Maeda et al. (2003).

**Statistical Analysis**

Significant differences in those parameters between IP and FP were calculated by t-tests using SPSS v15.0. To evaluate model performance, an analysis of coefficient of residual mass (CRM) and the modeling efficiency (EF) were conducted. The CRM and EF were calculated as (Vereecken et al. 1991; Moreels et al. 2003):

$$\text{CRM} = 1 - \frac{\sum_{j=1}^{N} P_j}{\sum_{j=1}^{N} O_j}$$

$$\text{EF} = 1 - \frac{\sum_{j=1}^{N} (P_j - O_j)^2}{\sum_{j=1}^{N} (O_j - \bar{O})^2}$$

Where $P_j$ are the simulated values, $O_j$ are the measured values, $\bar{O}$ is the average of the observed values and $N$ is the number of data pairs. The EF provides a comparison of the efficiency of the model to the efficiency of describing the data as the mean of the observation. The optimum value for EF is one, while the CRM should be as close to zero as possible. The EF can become negative, indicating that the observed mean is a better estimate of the observations than the model predictions. The coefficient of residual mass CRM indicates whether the observed data are overestimated (CRM<0) or underestimated (CRM>0) by the simulations (bias).
Figure 1. Monthly precipitation (mm) at Wonosobo and Kopeng sites year 2007-2008.

Figure 2. Mean daily temperature of Wonosobo and Kopeng sites year 2007-2008.
RESULTS AND DISCUSSION

Rainfall and Temperature Pattern

The Wonosobo site received more rain on average (3034 mm year$^{-1}$) than the Kopeng site (2234 mm year$^{-1}$), and the rain was concentrated in November to May (Figure 1). There are significant differences concerning the rainfall intensity during the wet and dry seasons. During the dry season (June to October), some farmers decided to leave the field bare, primarily where irrigation water was not available.

The average temperature at the Wonosobo experimental site ranged from 13 to 21 °C, with an average of 17.5±1.4 °C; at the Kopeng site, the temperature ranged from 15 to 21 °C with an average of 18.4±1.2 °C (Figure 2). These temperatures are appropriate for optimum growth of vegetables adapted to the highland climate.

Soil Mineral Nitrogen Profile

The soil mineral N profile (NO$_3^-$N) in these Andisols was highly dynamic, both between locations and temporally (Figure 3 and 4). Almost all FP treatments applied much higher rates of N fertilizer.
compared to the IP treatments (Widowati et al. 2011), but this was surprisingly not clearly reflected in the total mineral N in the soil profiles. Of the 36 growing seasons, only five had significantly greater total mineral N in FP, whereas in the rest of the plots, only a tendency towards higher mineral N was observed. During the third week after planting, the mineral N of the plots tended to increase. The
second fertilization was applied during the fourth week after planting but was not followed by an increase in the mineral N content in the second observation (42 days).

Levels of NO$_3$ measured in the soil monitored at a depth of 0-25 cm have been found to change very rapidly. Several researchers report an increased concentration of residual NO$_3$N in soil profiles after the application of large amounts of N to different cropping systems (Malhi et al. 2001; Gillard et al. 1995) and N mineralization from soil organic matter and organic fertilizer, while decreased nitrate content occurs due to plant uptake, immobilization, denitrification, and leaching.

Mineral nitrogen in the soil is easily transformed and translocated, which is reflected in the N-profile measurements over time. Although a portion of the N (NH$_4^+$-N) from fertilizer and organic matter is adsorbed by soil particles and taken up by the plants, the remaining N was estimated to be highly concentrated in the form of NO$_3$ and susceptible to leaching into a deeper layer. A simulation of the estimation of factors affecting nitrate leaching was conducted by LilBurnse et al. (2003); they reported that soil type, climate, and sowing date explained approximately equal amounts of the variance in nitrate leaching, whereas fertilizer application explained only approximately one-third of the variance of the other inputs.

The results of the mineral N dynamics between IP and FP indicated that there was no specific pattern. Although FP typically applied higher total-N, a greater accumulation of mineral nitrogen did not always occur. The reason for this is not very clear. One reason may be that residual N from previously applied manure was mineralized and may have been available for leaching in the continuously managed fields (Angle et al. 1993, Thomsen et al. 1993; Bergstrom and Kirchmann, 1999). Another reason may be the very specific nature of these soils, as discussed below.

| Site/Farmer name | N-balance | PNC |
|------------------|-----------|-----|
|                  | IP kg N ha$^{-1}$ | FP mg N L$^{-1}$ |
| **Wonosobo**     |           |     |
| Nurhakim         | 1505      | 1807 |
| Sudarto          | 1128      | 1866 |
| Sucipto          | 1351      | 1869 |
| **Kopeng**       |           |     |
| Nano             | 811       | 901  |
| Lukas            | 795       | 1104 |
| Ngatemin         | 790       | 1267 |

Nitrate Leaching Simulation Using the Burns $\alpha$ Model

Nitrate loss simulated from IP and FP averaged 252.3 kg NO$_3$N ha$^{-1}$ (ranged from 56 to 442 kg NO$_3$N ha$^{-1}$) and 372.7 kg NO$_3$N ha$^{-1}$ (ranged from 48 to 984 kg NO$_3$N ha$^{-1}$), respectively, but were not significantly different (Table 3). Comparing the simulated N losses to the total N added generated a percent loss varying from 14 to 137% (average 67%) for IP and 9% to 183% (average 71%) for FP. The calculation of N-balances (Widowati et al. 2011) revealed large N surpluses, and the relative percentage of simulated NO$_3$ loss to N-surplus produced varied from 11 to 117% (average 67%) for IP and 11 to 141% (average 76%) for FP.

The simulation results of N leaching using the Burns$_\alpha$ model obtained eight and seven negative CRM values over nine seasons for IP and FP, respectively. The CRM value of IP ranged from -1.869 to 0.165, and the value of FP ranged from -2.030 to 0.165. The EF (which is ideally equal to 1) was low in most of the cases, and the largest EF value reached 0.725.
measuring NO$_3^-$ leaching is by the use of weighing lysimeters. However, such instruments are very expensive and only available at a very limited number of locations.

As the leaching of soil NO$_3^-$-N is very difficult to measure directly in situ, a simulation model like the Burns$_\alpha$ model has to be used to predict nutrient losses to the environment (Moreels et al. 2003). Efforts to simulate the movement of NO$_3^-$-N using the Burns$_\alpha$ model were not very successful for the soils in this study, as reflected by the mostly overestimated results (negative CRM) and often poor EF values. Given the very low N use efficiency in these rotations, we would expect most of the surplus to be lost through leaching over time (1-2 years). However, the total N loss by leaching was much less than the N surplus in most cases, and was always less than the total N added. Because of the relatively large mineralization rates, we would expect most of the N added to be lost by leaching. This is in contrast with the good results obtained with the model in previous studies (e.g. De Neve and Hofman 1998; Moreels et al. 2003; Chaves et al. 2006), and here we discuss a number of reasons for the limited success of the simulations.

A first reason for the relatively poor simulation results could be the nature of the soils in this study. To our knowledge, this simulation model has not yet been applied in Andisols, and in general there have been very little studies on NO$_3^-$ leaching in Andisols using simulation models. The nature of these soils (high permeability) combined with the very intensive rainfall during the rainy season result in very fast leaching. Possibly the model was not able to capture this fast leaching rate. The calibration of the model (using leaching experiments with Br$^-$ as a tracer) gave an $\alpha$ value of 1, which is the maximum. This value corresponds to immediate leaching of all water above field capacity to the underlying layer within one-time step. Perhaps the time step used normally with this model (1 day) was too coarse to capture the very fast dynamics of leaching. Another reason related to the nature of the soils could be the difficulty in measuring accurate values of field capacity. Because of the very low bulk density, physical soil properties have rather high uncertainties. If the real field capacity of these soils is lower than measured here, than the actual leaching would be faster than the leaching simulated.

Another reason for the lack of agreement between measured and simulated NO$_3^-$ profiles may be the way the N dynamics in soil were implemented in the model. The original Burns$_\alpha$ model did not include a plant N uptake module. The crop N uptake was implemented in the model in a simplified way, namely dividing the total crop N uptake over the entire growing season over the number of days, and this amount was subtracted each day from the amount of NO$_3^-$-N present in the soil profile. We were also unable to take into account the rooting depth of the different crops. However, the error introduced by an inaccurate simulation of the crop N uptake would have a relatively small impact on the simulation of NO$_3^-$ leaching, because crop N uptake often was only a minor fraction in the total N balance of these rotations.

Finally, anion exchange of NO$_3^-$ could also influence simulation results, because Andisols may have significant anion exchange capacity. However, lab measurements indicated that the anion exchange of NO$_3^-$ was negligible in these soils (data not shown).

**Estimation of NO$_3^-$ Concentration in the Soil Water at 1 m Depth**

The average total evaporation was 1033 mm year$^{-1}$ and 942 mm year$^{-1}$ for Wonosobo and Kopeng, respectively. The average excess water (EW) was 2073 mm year$^{-1}$ for the Wonosobo sites and 1275 mm year$^{-1}$ for the Kopeng sites, respectively.

The IP and FP nitrogen balance results were highly positive, as the nitrogen fertilizer was heavily applied (Widowati et al. 2012). The Potential Nitrate Concentration (PNC) calculation results for IP and FP therefore demonstrated high concentrations, with averages of 59.8 mg N L$^{-1}$ and 82.5 mg N L$^{-1}$ in 2007, respectively (Table 4). The average PNC values of IP and FP were significantly different ($P < 0.05$).

**Potential Nitrate Concentration**

To estimate the movement of nitrate, nitrogen surplus accumulated as nitrate-nitrogen can be calculated in the soil at 1 m soil depth (PNC) (Maeda et al. 2003). Agricultural activity using organic and inorganic N fertilizers affects this PNC value. Improved practice tended to decrease the PNC value compared to farmer practice. However, IP still had a high PNC value compared to the maximum admissible concentration of 11.3 mg NO$_3^-$N L$^{-1}$ (the 1991 Directive 91/676 – European Community (EC) legislation), which was extended to apply to surface freshwater and groundwater intended for abstraction for drinking water and to freshwater, estuaries and coastal waters liable to eutrophication (Rodda et al. 1995). This is an indication that the vegetable farming systems in both locations experience over-fertilization; therefore, inefficiency occurred and
contributed to very high soil pollution. Maeda et al. (2003) also reported that excessive N from chemical fertilizer applied to Andisols can cause NO₃-leaching at 1-m depth under the Japanese climate (Asian monsoon).

CONCLUSIONS

Nitrate leaching is a problem that often occurs in intensive vegetable production centers in Indonesia. Measurements must be taken to monitor and provide evidence to farmers to improve the understanding of this problem. Although the FP always applied more nitrogen, a greater mineral N content was not always observed in the soil profile. Simulation results obtained from the Burns-a model on Andisols in general did not agree very well with the measured nitrate profiles, resulting mostly in overestimations of the actual values. The likely factors responsible for these overestimations were the very particular nature of these volcanic soils (low BD, uncertainties about soil physical properties), exceptionally high rainfall intensity and the way in which the crop N uptake was taken into account. Nitrate leaching can also be approximated by measuring the potential nitrate concentration (PNC) at 1 m soil depth. PNC values calculated here were very large. Although only approximate, the amounts of nitrate leaching and high PNC values found here are quite alarming, and farmers and policy makers should be made more aware of this issue that is threatening water quality and is a sign of highly inefficient N use.

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