

## NITROGEN LEACHING

### NITROGEN LEACHING IN AN AQUATIC TERRESTRIAL TRANSITION ZONE

SHORT TITLE: NITROGEN LEACHING

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**Abstract.** Large parts of East Germany are characterised by sandy soils with a high hydraulic conductivity. The risk of nitrogen leaching and groundwater pollution may be minimised by organic farming, which has been spread out in Germany during the last years. The study was conducted on an organically farmed rye field next to a lake in the state of Brandenburg between 2002 and 2004. In order to show how far organic farming may affect lake water quality, soil inorganic nitrogen (CaCl<sub>2</sub> extraction) and denitrification (acetylene inhibition method) were studied along an aquatic terrestrial transition zone (A = field site: 5 m above water level, B = field site: 1 m above water level, C = riparian zone with macrophytes: 0.5 m above water level). Although the field did not receive any organic and mineral fertiliser there was a short-term shifting of nitrogen by leaching from the field to the groundwater and the lake. Nitrogen loss during the winter was 29 kg N ha<sup>-1</sup> y<sup>-1</sup> and 12 kg N ha<sup>-1</sup> y<sup>-1</sup> in 2002/03 and 2003/04, respectively. Deviation between the two years seemed to be caused by great differences in precipitation. No nitrogen loss was observed from a control site. High denitrification was measured at sites B and C indicating an efficient nitrogen removal capacity within the riparian buffer zone.

**Keywords:** organic farming, N<sub>min</sub>, leaching, mineralisation, denitrification, groundwater, riparian zone

## 1. Introduction

Intensive farming in Germany leads to N surplus, which is up to more than 100 kg N ha<sup>-1</sup> y<sup>-1</sup>. Excessive N can be lost via two pathways. It is released as gas during microbiological transformation or it undergoes leaching with the seepage water. N leaching is accelerated in sandy soils due to high rates of precipitation. Primarily lowland sites and aquatic terrestrial transition zones demand special attention because they are nearby to groundwater and surface water. Since large parts of the northeast of Germany belong to the subglacial lowland with sandy soils, there is a considerable risk of nitrogen translocation due to leaching (Wendland et al. 1994). On the other hand, long residence times of the groundwater and good hydrogeochemical conditions may support N removal mechanisms.

Organic and integrated farming are becoming more important in order to meet environmental requirements. Particularly organic farming has been well established in the study area with numerous lakes. By abdication of mineral fertiliser, organic farming is appropriate for sandy soils of the federal state Brandenburg. In order to evaluate the close vicinity of agricultural land and environmentally protected areas, a small lake with a size of 3 ha was selected. The lake is surrounded by a 5-40 m wide riparian macrophyte strip. Wetlands and ecosystems such as found in the study area are considered to be effective for nonpoint source pollution control (Hill 1996, Blackwell et al. 1999, Hoffmann et al. 2000, Hefting 2003). Next to the lake, the fields were organically farmed for rye production in the years 2002 and 2003. 2002 was a wet year in contrast to 2003 with an extreme low precipitation rate. Such climatic distinction might have affected both, the microbiological activity in the soil (Hellebrand et al. 2005) and the extent of nitrogen leached out from the soil (Shepherd 1996, Kleinhenz et al. 1997).

The objective of this study is to estimate the output of nitrogen from the agricultural field and to compare it with the N output from nearby undisturbed grassland (control). It should be shown how far organic farming and climatic factors may affect groundwater and lake water quality and how far the littoral vegetation works as buffer strip removing nitrogen.

## 2. Methods

Soils of one organically farmed field and one non-cultivated control area were studied during a two-year period from April 2002 to March 2004. Three sampling sites were located along two transects with a slope of 5% meeting the groundwater fed lake. Each transect had an upland site (A) 5 m above water

level, a lowland site (B) 1 m above water level and a riparian site (C) 0.5 m above water level. Site A had a Kf value of  $4.0 \times 10^{-6} \text{ m s}^{-1}$  and  $2.5 \times 10^{-7} \text{ m s}^{-1}$  on the field transect and the control transect, respectively. Soil samples were taken with a corer ( $\varnothing = 2 \text{ cm}$ ) from 0-30 cm and 30-60 cm depths for four times and then they were mixed to obtain composite sample for the two layers. Soil samples were stored in a refrigerator before the analysis on the next day.

Usually the  $N_{\min}$  method is used to obtain the plant available nitrogen in the soil and to give some information about the N demand to the farmer. During the winter time, immobilisation and uptake of N by plants can be neglected. Therefore, the difference of  $N_{\min}$  between October/November and following March can be considered as N loss. According to the agricultural monitoring programme of the Brandenburg state, mineral nitrogen ( $N_{\min}$ ) is defined as the sum of ammonium and nitrate nitrogen that is determined after the extraction of wet soil samples with 0.0125 M  $\text{CaCl}_2$  solution (shaken for 1 hour).  $\text{NH}_4\text{-N}$  was measured photometrically by the indophenol blue method and  $\text{NO}_3\text{-N}$  by ion chromatography. Total carbon (TC) and total nitrogen (TN) were measured by dry combustion method using an elemental analyser.

Gaseous nitrogen release from the soil was detected after incubation in a nitrogen atmosphere for 4 hours at  $22^\circ \text{C}$ . Two assays, each with 3 replicates, were applied to obtain the rate and the potential of denitrification. Denitrification was measured using the acetylene-blockage technique (15%  $\text{C}_2\text{H}_2$ ) described by Yoshinari & Knowles (1976). To obtain the denitrifying potential 1 mmol  $\text{NO}_3$  was amended to the incubation vessels. Gas samples of 1 ml were injected into a gas chromatograph (Fisons Instruments, GC 8340) equipped with a 3 m long packed column (Haye Sepe D 100/120 mesh). The oven temperature was programmed between  $70^\circ \text{C}$  and  $120^\circ \text{C}$ .  $\text{N}_2\text{O}$  is detected by a  $^{63}\text{Ni}$  electron capture detector (ECD) operating at  $320^\circ \text{C}$ . The carrier gas was helium with a flow rate of  $15 \text{ ml min}^{-1}$ . All data for extractable nitrogen and denitrification rates were related to soil dry weight after drying at  $105^\circ \text{C}$ .

Lake water was analysed for  $\text{NH}_4$  and  $\text{NO}_3$  at monthly intervals directly as mentioned above. Climatic factors were monitored by a weather station in Potsdam (TOSS GmbH Potsdam) located 30 km north of the study area.

### 3. Results and discussion

#### 3.1. CLIMATIC CONDITIONS

During the study period, air temperature did not differ very much which were  $10.9^\circ \text{C}$  and  $10.6^\circ \text{C}$  in 2002 and 2003, respectively. However, there was a major difference in precipitation with a relative high rate of 634 mm in 2002

and a very low rate of 272 mm in 2003. The extreme dryness in 2003 was observed in most regions of Germany leading to heavy crop losses.

### 3.2. EXTRACTABLE NITROGEN

From the high elevated site (A) to the riparian site (C) total and extractable nitrogen ( $N_{\min}$ ) increase along both transects. On the field of winter rye an average of 0.2% of total N in the soil was extractable N compared to 0.1% on the control transect (Tab. 1).

TABLE 1. Mean values of soil characteristics within 0-60 cm depths of the two transects at the lake. Upland sites (A), lowland sites (B) and the riparian sites (C) were sampled from April 2002 to March 2004.

		A	B	C	A	B	C
		Winter rye		Littoral	Control		Littoral
<b>pH</b>		<b>6.0</b>	<b>5.5</b>	<b>5.3</b>	<b>5.9</b>	<b>6.4</b>	<b>5.7</b>
<b>Electrical conductivity</b>	$\mu\text{S cm}^{-1}$	<b>30</b>	<b>58</b>	<b>83</b>	<b>41</b>	<b>48</b>	<b>70</b>
<b>Water content</b>	%	<b>4.5</b>	<b>18.5</b>	<b>34.6</b>	<b>4.9</b>	<b>5.3</b>	<b>36.9</b>
<b>Total C</b>	$\text{g kg}^{-1}$	<b>4.1</b>	<b>35.0</b>	<b>74.7</b>	<b>5.4</b>	<b>5.1</b>	<b>46.2</b>
<b>Total N</b>	$\text{mg kg}^{-1}$	<b>352</b>	<b>1,685</b>	<b>2,836</b>	<b>400</b>	<b>517</b>	<b>2,340</b>
<b>NH<sub>4</sub>-N</b>	$\text{mg kg}^{-1}$	<b>0.20</b>	<b>0.77</b>	<b>4.09</b>	<b>0.24</b>	<b>0.38</b>	<b>1.96</b>
<b>NO<sub>3</sub>-N</b>	$\text{mg kg}^{-1}$	<b>0.64</b>	<b>1.97</b>	<b>0.21</b>	<b>0.20</b>	<b>0.38</b>	<b>0.13</b>
<b>N<sub>min</sub></b>	$\text{mg kg}^{-1}$	<b>0.84</b>	<b>2.74</b>	<b>4.30</b>	<b>0.44</b>	<b>0.76</b>	<b>2.09</b>
<b>N removal (Oct.-March)</b>	$\text{kg N ha}^{-1}$	<b>8.7</b>	<b>32.4</b>	<b>47.6</b>	<b>-0.5</b>	<b>-4.1</b>	<b>28.1</b>

The low organic matter content, reflected by low total carbon and the low water content at the site A of both transects indicated a high hydraulic conductivity of the soil. With a rate of 70% sand on the control transect and 75% sand on the field transect, the soils have a little water holding capacity implying a high risk of nutrient leaching.

On the field site A, the mean extractable nitrogen content was  $0.84 \text{ mg kg}^{-1}$  during the two-year study period. In contrast to this relatively low value,  $N_{\min}$  content was 3 to 4 times higher at the low field site B where a sink of nitrogen can be assumed. Some of the nutrients possibly have been derived by leaching from the site A and subsequent groundwater runoff. In contrast to the control sites of A and B where both  $\text{NH}_4$  and  $\text{NO}_3$  are forming similar amounts of  $N_{\min}$ ,  $\text{NO}_3$  is predominating on the rye planted sites. Situation at site C is quite different within the littoral. The mean  $N_{\min}$  content is  $4.3 \text{ mg kg}^{-1}$  and consists primarily of  $\text{NH}_4$ . Low  $\text{NO}_3$  concentrations can be explained by both, a

suppression of nitrification in the waterlogged environment, plant uptake and/or a high denitrifying activity. This pattern can also be observed at the control site (Fig. 1). In contrast to the impact of riparian forest buffers on agricultural nonpoint source pollution (Snyder et al. 1998, Addy et al. 1999) relatively little is known about the function of herbaceous buffer strips (Flite et al. 2001). Our results provide some information that noncultivated herbaceous riparian zones can reduce the  $\text{NO}_3$  concentrations of waters draining from upslope cultivated agricultural soils. Enhanced  $\text{NH}_4$  concentrations at site C of the field transect in 2003/04 may be explained by accelerated mineralisation following warming and aeration of the organic soil due to the lack of rainwater.

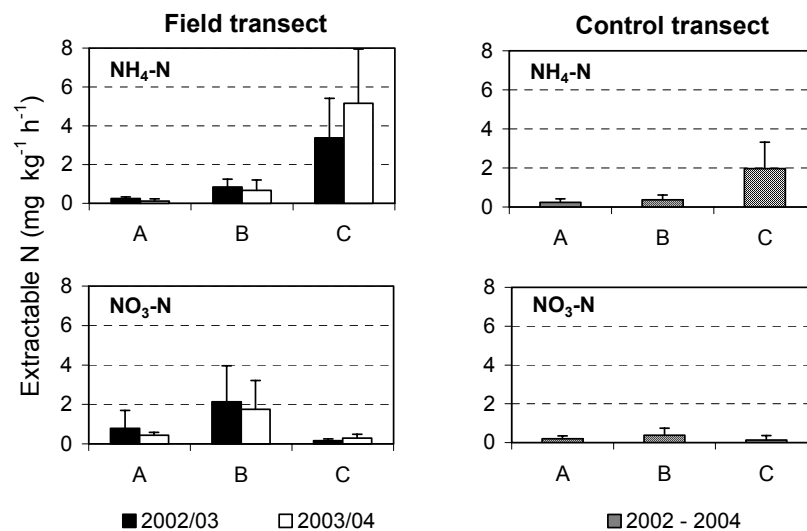


Figure 1. Extractable  $\text{NH}_4$  and  $\text{NO}_3$  in the 0-60 cm soil depths along the field and the control transect.

$\text{NO}_3$  concentrations were highest at the field site B in 2002/03 and 2003/04.  $\text{NO}_3$  concentrations of this site were 5 fold greater than those of control site B and it could be influenced by more elevated sites such as site A as a result of leaching and run-off. Since the field was not fertilised during the study period and total N of the field and control transect were in the same range, the enhanced  $\text{NO}_3$  concentrations at the field sites A and B probably arise from the much more intensive mineralisation due to soil cultivation. Organic residues, which remain on the field after harvest, can explain the higher amount of easily available N on the field. Part of this enhanced soil  $\text{N}_{\text{min}}$  may be subject to leaching (Shepherd & Lord 1996, Kavdir et al. 2005).

Nitrogen also becomes available after mineralisation of soil organic matter (SOM) which contains a considerable amount of N. Estimation of N mineralisation could be made using SOM content values. If 2% of the total organic N in the surface soil is mineralised annually, a soil with 1% SOM content can mineralize about  $45 \text{ kg N ha}^{-1} \text{ y}^{-1}$  (Schepers & Mosier, 1991). Therefore if we calculate the potential N mineralisation of study soils using the data for Table 1 (soil bulk density of  $1.5 \text{ g cm}^{-3}$  and 0-30 cm depth), we can find that N mineralisation potential from A and B (rye) and C is 32, 152 and  $255 \text{ kg N ha}^{-1} \text{ y}^{-1}$ , respectively. In the same range is N mineralisation from control sites A, B and C with 36, 47 and  $211 \text{ kg N ha}^{-1} \text{ y}^{-1}$ , respectively. Consequently tillage operations in planted areas seem to have increased  $\text{NO}_3$  concentrations on the field sites compared to control sites. Such high amounts of N which may become available by mineralisation can explain both, N removal by harvest (average of sites A and B =  $39 \text{ kg N ha}^{-1} \text{ y}^{-1}$ , Kern 2004) and N loss (site A =  $8.7 \text{ kg N ha}^{-1} \text{ y}^{-1}$  and site B =  $32.4 \text{ kg N ha}^{-1} \text{ y}^{-1}$ ) during the winter. However, these are rough estimates, which will vary due to temperature, precipitation and tillage (Wienhold & Halvorson, 1999).

### 3.3. NITROGEN REMOVAL

The most important removal paths for nitrogen from the field are plant uptake and harvest, leaching, surface run-off and denitrification. The amount of N scavenged depends in part on plant type, plant growth, soil type, amount of fall soil inorganic N, and weather. Removal of scavenged nitrogen by harvest was  $64 \text{ kg N ha}^{-1}$  in 2002 and only  $13 \text{ kg N ha}^{-1}$  in 2003 due to low water supply and bad plant growth (Kern 2004). Similar differences in the loss of N during the winter are calculated by changes in  $N_{\text{min}}$  from October/November to March of the following year. Taking the means of field site A and B, there is an overwinter N loss of  $29.1 \text{ kg N ha}^{-1}$  in the first winter and  $11.9 \text{ kg N ha}^{-1}$  in the second winter of the study period. It becomes apparent that the dry weather reduced the N loss considerably. But even under such extreme conditions, the N loss on the field transect was much higher compared to the control transect. In the same range as on our field transect, Wendland et al. (2004) estimated a leaching rate of  $15 \text{ kg N ha}^{-1} \text{ y}^{-1}$  within the catchment area of the River Elbe. The authors stressed, however, that about 90% of the diffuse N input into the groundwater can be degraded in the aquifer system before entering the surface water. Contrasting results were presented by Brye et al. (2001) who found that less than 25% of the N leached below the root zone in an agro-ecosystem was subjected to denitrification. Limited denitrification was substantiated by insufficient lengths of saturated soil conditions and the supply of dissolved organic carbon.

Much lower N uptake by plants and lower amount of leaching/denitrification in 2003/04 can be explained by the extreme dry weather in 2003, when the region of Potsdam received only 272 mm water by precipitation. Thus the total outflow of water and dissolved nutrients was considerably low.

Anthropogenic N input from the atmosphere is suggested to be about 30 kg N ha<sup>-1</sup> in Germany (Dannowski 1995). This amount of N should have been completely consumed in 2002. However, it is even possible that N is being enriched on the field sites due to atmospheric deposition in 2003 (dry year). The N<sub>min</sub> removal data presented in Table 1 are integrated values, which do not allow distinguishing between the removal paths such as leaching and gaseous release. It is rather difficult to present a real N balance, which at least should include precise data of one removal path.

A part of the biomass produced at the riparian site of the lake does not undergo a complete mineralisation due to the waterlogged environment. Carbon and nutrients may accumulate yielding in a peat production. In the case of nitrogen, an immobilisation of nitrogen can be estimated for the last 60 years, which would explain all the N<sub>min</sub> lost during the study period and extrapolated to a period of 60 years.

#### 3.4. GASEOUS NITROGEN TRANSFORMATION

Although the objective of this study was not to quantify N leaching losses, it is possible to get a better understanding of the N dynamic along the aquatic terrestrial transition zone by studying microbiological processes, particularly denitrification.

Denitrification was restricted to the 0-30 cm uppermost layer at all sites indicating that there was no lateral flow of NO<sub>3</sub>-rich water in deeper soil layers. Denitrification does not play an important role in N release from the control site, in contrast to the field site (Fig. 2). Highest rates of denitrification were measured at field site B according to highest NO<sub>3</sub> concentrations in the soil. That means that N can be efficiently removed on this low field site before the mobile NO<sub>3</sub> pass through the riparian site C reaching the lake afterwards. Alternative wetland buffer zones can be even more effective for NO<sub>3</sub> removal than riparian zones adjacent to the water body as reported by Blackwell et al. (1999). Disadvantages of conventional riparian buffer zones may derive from by-passing due to pathways which follow drains and ditches.

Although the riparian site is characterised by a high carbon content, which is favourable for denitrification (Maag et al. 1997, Burt et al. 1999), denitrifying activity is quite low in contrast to the field site B. This was caused by very low concentrations of NO<sub>3</sub>, which seems to be limited for denitrification in the soil of site C. NO<sub>3</sub> supply should have been interrupted because nitrification is

suppressed under anaerobic conditions in waterlogged environments (Phillips 1999). The assay with the amendment of  $\text{NO}_3$  showed that site C had the same ability to denitrify as site B, confirming the results from Davidsson et al. (2002) on flooded and drained peatland soils. Furthermore, the denitrifying potential of site B was nearly as same as the denitrification rate without addition of  $\text{NO}_3$ . Consequently, the denitrification capacity of site B fully used all  $\text{NO}_3$  supply in both study years. Evidence for a high denitrifying potential of study soils supports the low amount of  $\text{NO}_3$  supply to the riparian zone C. High concentrations of  $\text{NO}_3$  in the soil and the lake were observed only in October 2002 when  $\text{NO}_3$  not sufficiently denitrified. As a result excess  $\text{NO}_3$  could be leached into the lake.

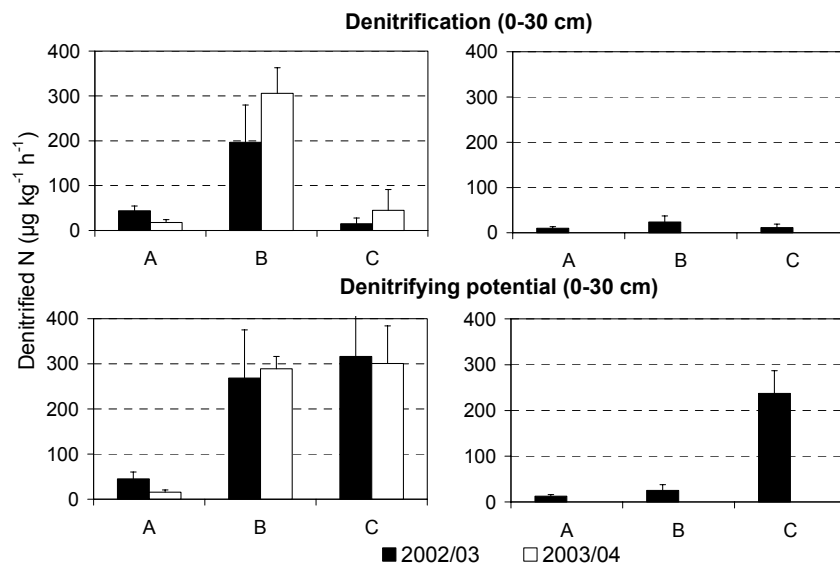


Figure 2. Denitrifying activity and its potential after  $\text{NO}_3$  amendment in the soil layer 0-30 cm.

#### 4. Conclusions

A good balance between the current organic farming N supply and N uptake by crop plants can be concluded for the study area. Both, little N leaching of the root zone and the low slope characterise the agro-ecosystem under study as an effective sink for N. Even if there is no mineral and organic fertilisation, N translocation from the field to the groundwater cannot be completely excluded. But the nitrogen that is leached from the study field can be denitrified or fixed

as organic matter in the riparian zone. Thus the lake under study seems to be well protected from N pollution.

The key factors controlling the subsurface NO<sub>3</sub> retention are residence time, low oxygen concentration and high content of electron donors such as organic carbon. These factors are correlated to hydrogeological site conditions. By prevailing glacio-fluviatile sands and moraine deposits, the whole European Pleistocene Lowland can be considered as NO<sub>3</sub> removing aquifer.

This study is an example how far an aquatic transition zone may serve as a buffer against NO<sub>3</sub> loaded groundwater. The leaching of NO<sub>3</sub> into the groundwater must not be necessarily a problem if the pollution of drinking water resources can be excluded. NO<sub>3</sub> can be removed with high efficiency as long as groundwater-born NO<sub>3</sub> meets favourable conditions for denitrification. In this way, NO<sub>3</sub> inflow into surface water particularly into river systems with its low denitrifying capacity can be prevented.

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