

Extended abstract No. 39

NITRATE UTILIZATION IN A PEAT SOIL UNDER REWETTING CONDITIONS – A RESULT OF A ¹⁵N-NITRATE TRACER APPROACH AT LABORATORY AND FIELD SCALE

Dr. Nadine Tauchnitz, Martin-Luther-University Halle-Wittenberg, Faculty of Natural Sciences III, Institute of Agricultural and Nutritional Sciences, Julius-Kühn-Str. 23, D-06112 Halle (Saale), Tel.: +493455522773, Email: nadinetauchnitz@alice-dsl.net

Dr. Rolf Russow, Dr. Oliver Spott, Helmholtz Centre for Environmental Research-UFZ, Dept. Soil Physics

Dr. Sabine Bernsdorf, Martin-Luther-University Halle-Wittenberg, Faculty of Natural Sciences III, Institute of Agricultural and Nutritional Sciences

Prof. Dr. Ralph Meissner, Helmholtz Centre for Environmental Research-UFZ, Dept. Soil Physics, Lysimeter Station

SUMMARY

The present study was aimed at determining the consumption of nitrate (NO_3^-) and pathways of NO_3^- transformation using a [¹⁵N] NO_3^- /bromide double-tracer approach. Under field conditions in the rewetted peat sites ¹⁵ NO_3^- consumption occurred rapidly with more than 50 % NO_3^- consumed within 2 days and 100 % within 8 days after tracer application. Concomitantly ¹⁵N abundances of different soil water N compounds (ammonium, nitrite) increased considerably and lead to the conclusion that, besides commonly known NO_3^- reduction to nitrite (i.e. denitrification), a reduction of NO_3^- to ammonium based on added ¹⁵ NO_3^- has simultaneously taken place.

KEY WORDS: denitrification, dissimilatory nitrate reduction, peatlands

INTRODUKTION

Nitrogen (N) inputs by atmospheric deposition have increased markedly in Europe over the last five decades (e.g. Pitcairn *et al.*, 1995) and may affect sensitive ecosystems that are adapted to nutrient poor conditions such as peatlands (e.g. Bragazza *et al.*, 2004). Recent concerns, however, emphasise a potential risk that an additional input of nitrate (NO_3^-) by atmospheric deposition may increase denitrification rates and thus, enhance the emission of the greenhouse gas nitrous oxide (Aerts, 1997). Within peat soils NO_3^- represents a very labile species, since pronounced anaerobic conditions favour a fast microbial consumption of NO_3^- with denitrification as the major process assumed (Aerts, 1997; Rückauf *et al.*, 2004). However, recent studies suggest that soil NO_3^- can also be transformed during dissimilatory NO_3^- reduction to ammonium (NH_4^+ , DNRA) (Rütting *et al.*, 2008; Sgouridis *et al.*, 2009) but only little is known concerning its presence in peatlands (Ambus *et al.*, 1992; Davidsson and Ståhl, 1999; Rückauf *et al.*, 2004).

The present study was aimed at determining NO₃⁻-N consumption and transformation in a transition bog under drained (water table -30 cm) and rewetted (water table 0 cm) conditions by using a [¹⁵N]NO₃⁻/bromide double-tracer approach.

MATERIAL AND METHODS

Study site

Experiments were performed in the Harz Mountains (central Germany) in a transition bog (670 m a.s.l.) in direct vicinity to the Mt. Brocken (1142 m a.s.l., 51°48'N, 10°37'E). The peat soil was drained in 1750 to 1776 by drainage ditches, then planted with spruces for forestry use and recently partially rewetted by closing the ditches beginning of the year 2005. It extends to an area of 19 ha and is classified as nutrient poor and acidic.

Experimental design

For the laboratory experiments 16 intact soil cores (20 cm diameter, 50 cm height) were cut out of the peat soil at a drained peat site. Field experiments were performed at both drained (FE 1) and rewetted peat sites (FE 2). For the field experiments V2A steel tubes (30 cm diameter, 80 cm height) were pushed 70 cm deep into the peat soil layer (five replicates). Introduced tubes were open at the bottom to permit a natural water table adjustment within the tubes during an experiment. Peat soil columns were then labelled with [¹⁵N]NO₃⁻, equivalent to 30 kg N ha⁻¹ (¹⁵N abundance of 48 at. %). Concomitantly to NO₃⁻ the conservative tracer Br⁻ was applied to calculate the effective NO₃⁻ consumption in a peat soil column. For soil water sampling miniature suction cups were placed within the tubes in different depths.

Chemical and ¹⁵N analysis

Concentration of bromide (Br⁻) and NO₃⁻ were performed by ion chromatography (DX 500). The N concentration and ¹⁵N abundance of NH₄⁺, nitrite (NO₂⁻), and NO₃⁻ in soil water samples was analysed by the SPINMAS approach (Russow, 1999; Stange *et al.*, 2007).

Calculation of NO₃⁻ consumption

NO₃⁻ consumption was calculated based on measured Br⁻ and [¹⁵N]NO₃⁻ concentrations in peat soil water samples and Br⁻ and ¹⁵NO₃⁻ amounts applied at the beginning of an experiment. First, dilution (DL) of the Br⁻ tracer was calculated according to Eq. (1).

$$DL = \frac{[Br^-_{sample}] - [Br^-_{zero}]}{[Br^-_{applied}]} \quad (1)$$

[Br⁻_{sample}] – concentration of Br⁻ in peat soil water sample; [Br⁻_{zero}] - concentration of Br⁻ in peat soil water prior to tracer application; [Br⁻_{applied}] - concentration of Br⁻ tracer applied.

Afterwards dilution corrected ¹⁵NO₃⁻ concentration was calculated according to Eq. (2) and then used to calculate ¹⁵Nitrate consumption (C_{NO3-}) express in percent according to Eq. (3).

$$[^{15}NO_3^-_{corr.}] = \frac{[^{15}NO_3^-_{sample}]}{DL} \quad (2)$$

$[^{15}\text{NO}_3^-]_{\text{sample}}$ - concentration of $^{15}\text{NO}_3^-$ in peat soil water.

$$C_{\text{NO}_3^-} = \frac{([^{15}\text{NO}_3^-]_{\text{corr.}} - [^{15}\text{NO}_3^-]_{\text{applied}})}{[^{15}\text{NO}_3^-]_{\text{applied}}} \cdot 100 \quad (3)$$

$[^{15}\text{NO}_3^-]_{\text{applied}}$ - concentration of $[^{15}\text{N}]\text{NO}_3^-$ tracer applied.

RESULTS

NO_3^- consumption

At the drained peat sites (FE 1) NO_3^- consumption was very low and peaked at only 47 % at the end of FE 1 (Fig. 1). By contrast, at the rewetted field site (FE 2) a more rapid rise of NO_3^- consumption was observed, reaching a total consumption of more than 50 % two days and up to 100 % five days after tracer application (Fig. 1).

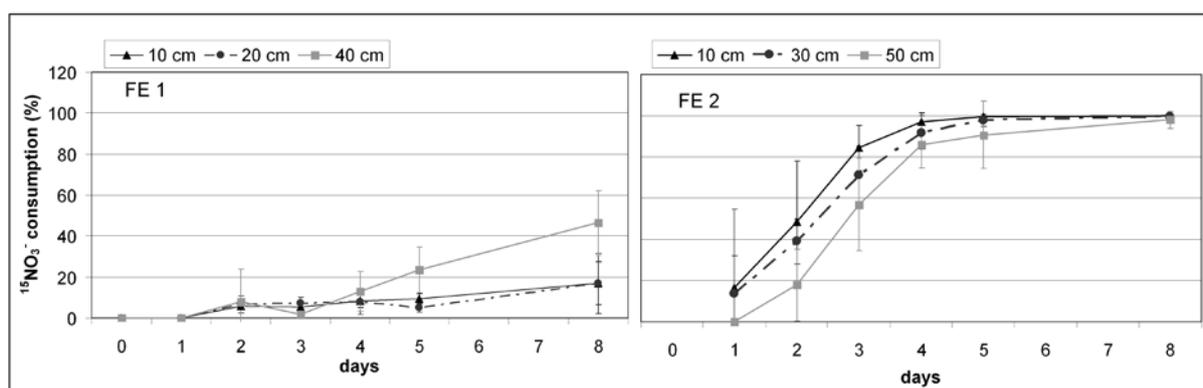


Figure 1. Nitrate (NO_3^-) consumption at the drained (FE 1) and rewetted peat sites (FE 2) (shown as mean and standard deviation). Tracers were applied at day 1.

NO_3^- transformation

The $^{15}\text{NH}_4^+$ concentrations in the peat soil water of the rewetted peat site (FE 2) increased markedly in all considered depths to maximum values of up to $0.005 \text{ mmol L}^{-1}$ (-10 cm) (data not shown). In the peat soil water of the drained peat site smaller $^{15}\text{NH}_4^+$ concentrations of maximally $0.002 \text{ mmol L}^{-1}$ (-10 cm) were observed (data not shown).

At the rewetted peat site (FE 2) $^{15}\text{NH}_4^+$ abundances increased in all considered depths to a maximal value of 4.2 at. % at the third day of the experiment (Fig. 2a).

Calculated NH_4^+ formation from NO_3^- increased from 3.7 % (-10 cm), 3.3 % (-30 cm) and 3.2 % (-50 cm) at the first day of the experiment to 12 % (-10 cm), 9.2 % (-30 cm) and 7.4 % (-50 cm) at the fourth day of the experiment at the rewetted site (Table 1). At the drained site NH_4^+ formation from NO_3^- was maximally 3.9 % (20 cm) with a very high standard deviation (Table 1).

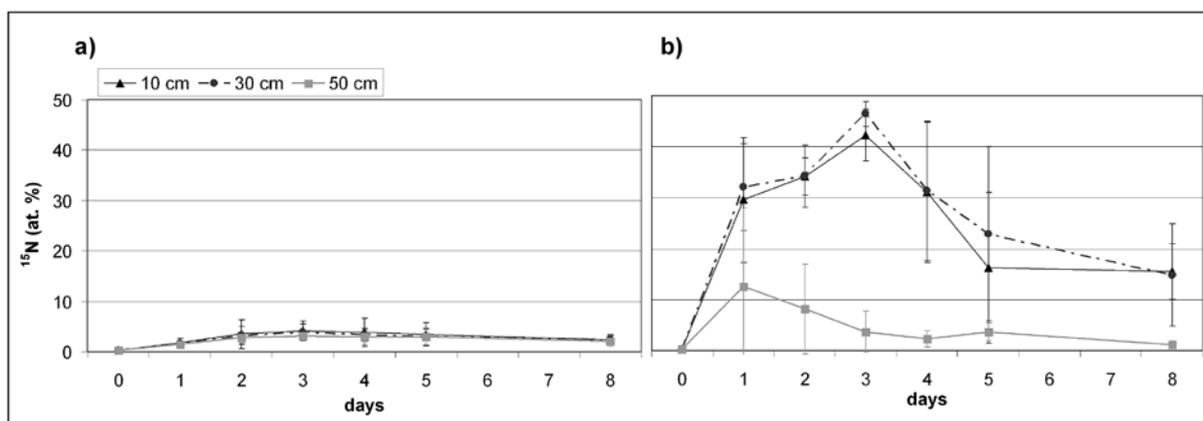


Figure 2. ^{15}N abundances of a) ammonium (NH_4^+) and b) nitrite (NO_2^-) in peat soil water at the rewetted peat site (FE 2) (shown as mean and standard deviation). Tracers were applied at day 1.

Calculated NO_2^- formation based on NO_3^- at the rewetted peat site increased from 62.1 % (-10 cm), 67.2 % (-30 cm) and 59.0 % (-50 cm) at the first day of the experiment to maximal values of 90.3 % (-10 cm), 96.1 % (-30 cm) and 97.0 % (-50 cm) reached at the third of the experiment (data not shown). Our study furthermore showed that ^{15}N abundances of NO_2^- in the peat soil water at the rewetted peat site (FE 2) significantly increased mainly in the first days of the experiment reaching maximal value of 46 at. % (Fig. 2b).

Table 1: Ammonium (NH_4^+) formation derived from nitrate (NO_3^-) at the drained (FE 1) and rewetted (FE 2) peat site (shown as mean and standard deviation). Tracer was applied at day 1.

	NH_4^+ formation derived from NO_3^- (in %)					
	FE 1			FE 2		
Depth	10 cm	20 cm	40 cm	10 cm	30 cm	50 cm
Day 0	0	0	0	0	0	0
1	1.4 ± 0,2	-	0	3.7 ± 3	3.3 ± 1	3.2 ± 1
2	1.6 ± 0,5	1.7 ± 1	2.3 ± 0.4	7.3 ± 7	6.9 ± 2	5.9 ± 2
3	2.3 ± 0,2	2.7 ± 2	-	9.0 ± 6	8.2 ± 3	6.8 ± 3
4	-	-	-	11.8 ± 10	9.2 ± 4	7.4 ± 3
5	-	-	-	n. c.	n. c.	n. c.
8	2.8 ± 1	3.9 ± 5	3.0 ± 1	n. c.	n. c.	n. c.

n.c.: not exactly calculable because of low $^{15}\text{NO}_3^-$ abundances, - : not enough sampling water

DISCUSSION

NO_3^- consumption rates differed significantly between both peat sites with more than 50 % NO_3^- consumed within 2 days and 100 % within 8 days after tracer application at the rewetted peat site (see Fig. 1). Reasons for the clear differences between drained and rewetted sites are most likely related to the O_2 availability. At the drained peat site (FE 1) observed redox potentials were significantly higher than observed under water-logged conditions at the rewetted site (data not shown) that have been well in the range reported for microbial NO_3^- reduction (e.g. denitrification) (Mitsch and Gosselink, 2000). It has been shown in several studies that at

high soil water contents and low O₂ concentration (as often found in peatland soils) NO₃⁻ is rapidly utilized (e.g. reduced by denitrifier).

The present study revealed furthermore that under water-logged conditions at the rewetted peat site (FE 2) the ¹⁵N abundance of NH₄⁺ increased markedly during the experiment suggesting the microbial process of DNRA (see Fig. 2a, Table 1) whereas under drained conditions the dissimilatory NO₃⁻ reduction seems to be less important. The higher O₂ availability observed at the drained peat site is assumed as the reason for the differences between both sites. It was shown in previous studies that DNRA was restricted to strongly reducing conditions. Furthermore NO₃⁻ limiting conditions with C:NO₃⁻ ratios >12 associated with a shortage of the electron acceptor NO₃⁻ seems to favour DNRA over common denitrification (Davidsson and Ståhl, 2000). But whereas the importance of DNRA was clearly pointed out for many ecosystems (particularly for river sediments, riparian zones as well as paddy and forest soils; e.g. Rütting *et al.*, 2008; Sgouridis *et al.*, 2009), there are only a few studies available which consider DNRA in peatland soils (Ambus *et al.*, 1992; Davidsson and Ståhl, 2000). In contrast to our results DNRA was found to be an insignificant process in previous studies (Davidsson and Ståhl, 2000; Rückauf *et al.*, 2004). Besides DNRA we observed increasing ¹⁵N abundances of NO₂⁻ at the rewetted peat site to values that were nearly high than the applied NO₃⁻ tracer. These results indicate a reduction of the applied NO₃⁻ by microbial degradation processes. Denitrification processes are known to be of major importance for the microbial degradation of NO₃⁻ favoured by anaerobic conditions in peat soils (Aerts, 1997; Rückauf *et al.*, 2004) and are suggested as the main reason for the observed high NO₃⁻ consumption at the rewetted peat sites (see Fig. 1).

ACKNOWLEDGEMENTS

We want to thank the German Science Foundation (DFG) for funding the present study. In addition, we would like to thank Mr. B. Apelt (Helmholtz Centre for Environmental Research-UFZ, Department Soil Physics) for conducting ¹⁵N analyses of water samples and also many thanks to the Department of Analytic (Helmholtz Centre for Environmental Research-UFZ) for analysis of water samples.

REFERENCES

- Aerts, R. (1997). Atmospheric nitrogen deposition affects potential denitrification and N₂O emission from peat soils in the netherlands. *Soil Biology and Biochemistry* **29**, 1153-1156.
- Ambus, P., Mosier, A. R. and Christensen, S. (1992). Nitrogen turnover rates in a riparian fen determined by ¹⁵N dilution. *Biology and Fertility of Soils* **14**, 230-236.
- Bragazza, L. and Limpens, J. (2004). Dissolved organic nitrogen dominates in European bogs under increasing atmospheric deposition. *Global Biogeochemical Cycles* **18**, 1-5.
- Davidsson, T.E. and Ståhl, M. (2000). The influence of organic carbon on nitrogen transformations in five wetland soils. *Soil Science Society of America Journal* **64**, 1129-1136.
- Mitsch, J.W. and Gosselink, J.G. (ed.) (2000). *Wetlands*, third ed. John Wiley & Sons, New York.

Pitcairn, C.E.R., Fowler, D. and Grace, J. (1995). Deposition of fixed atmospheric nitrogen and foliar nitrogen content of bryophytes and *Calluna vulgaris*. *Environmental pollution* **88**, 193 – 205.

Rückauf, U., Augustin, J., Russow, R. and Merbach, W. (2004). Nitrate removal from drained and reflooded fen soils affected by soil N transformation processes and plant uptake. *Soil biology and biochemistry* **36**, 77-90.

Russow, R. (1999). Determination of ¹⁵N in ¹⁵N-enriched nitrite and nitrate in aqueous samples by reaction continuous-flow quadrupole mass spectrometry. *Rapid Communication in Mass Spectrometry* **13**, 1334-1338.

Rütting, T., Hyugens, D., Müller, C., Van Cleemput, O. and Godoy, R. (2008). Functional role of DNRA and nitrite reduction in a pristine south Chilean *Nothofagus* forest. *Biogeochemistry* **90**, 243-258.

Sgouridis, F., Heppel, C.M. , Trimmer, M. and Wharton, G. (2009). Dissimilatory Nitrate Reduction to Ammonium (DNRA) potential in the re-connected floodplain of the River Cole (Oxfordshire, UK). *Geophysical Research Abstracts* **11**.

Stange, C.F., Spott, O., Apelt, B. and Russow, R. (2007). Automated and rapid online determination of ¹⁵N abundance and concentration of ammonium, nitrite or nitrate in aqueous samples by the SPINMAS technique. *Isotopes in Environmental and Health Studies* **43**, 227-236.