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Little fertilizer response but high N loss risk of maize on a productive organic-sandy soil

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Abstract Information on the environmental impact of maize production is actually inconsistent. Indeed some experiments report good nitrogen (N) efficiencies and small residual N. Other experiments show large leaching losses, while in practice maize production is often coupled with an intensive production and large N surpluses. Here, we present data from a 4-year experiment with silage maize on a sandy soil of high mineralization potential. The experimental set-up included three N input forms, mineral, cattle and pig slurry and four rates of total N, of 0, 80, 160 and 240 kg N ha⁻¹ year⁻¹ and the use of suction cups. Results show that dry matter and N yields for N0–N240 were relatively high and consistent (158–192 kg N ha⁻¹). Further findings show large residual soil mineral nitrogen, of 138–237 kg N ha⁻¹, and high nitrate concentrations in leaching water during winter, of 39–73 mg NO₃-N L⁻¹, corresponding to leaching losses of 86–152 kg N ha⁻¹. Response to N input was small with apparent N recoveries of 14–22% for manures and mineral fertilizers. We conclude that caution is needed when maize production is

extended to fields with an apparently high potential for mineralization and that use as grassland would be a better alternative with regard to N leaching losses.

Keywords Maize · Nitrogen losses · Manure · Leaching · Organic matter · Mineralization

Abbreviations

ANR	Apparent nitrogen recovery
CAN	Calcium–ammonium–nitrate treatment
CS	Cattle slurry treatment
DM	Dry matter
N0, N80, N160, N240	Total N input treatment levels
PS	Pig slurry treatment
RNFV	Relative N fertilizer value
SMN	Soil mineral nitrogen

1 Introduction

The area of land cultivated with maize in Germany has increased steadily in recent years, a trend that is expected to continue (Offermann et al. 2010). Maize (*Zea mays*) is not only used for providing feed as corn or silage but also for production of biogas. The production and use of maize for biogas is more cost efficient compared with other crops and the cultivation, harvest and storage of maize is well established with farmers. Consequently, the biogas boom has helped to make maize the most cultivated crop with 424,000 ha in Lower Saxony, Germany, in 2007. Deep-ploughed soils from shallow moors in northwest Germany are well qualified for use as grassland or arable land for the cultivation of maize. Until 1978, about 200,000 ha of peat soils have been deep ploughed to become a sand-mix culture

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(Hagemann 1978). After drainage and under agricultural use, these soils can have an increased potential for mineralization. Generally, maize can use mineral and organic fertilizer nitrogen (N) quite effectively (Schröder et al. 1998; Mädl et al. 1999; Nevens 2003; Wachendorf et al. 2006b). However, N uptake slows down and eventually ceases about 2 weeks after silking, which is often in mid-August (Ma et al. 1999) and N mineralized later from manures, mineral fertilizers or soil might be lost via leaching or in gaseous forms. Often, both, quantity and timing of N fertilizer application are not well adjusted to the site-dependent N requirements of the maize crop and cause large residual soil mineral N (SMN) and N leaching, especially on coarse textured soils (Schepers et al. 1991; Sticksel et al. 1994; Schröder et al. 1998; Sogbedji et al. 2000; Nevens 2003). In some soils with a potential for N mineralization large residual SMN were found even when N application was well adapted to the requirements of the crop (Richards et al. 1999). Moreover, in a survey of silage maize fields in northern Germany a N balance at the field scale could not explain high residual SMN (Schiermann 2004). With the ongoing trend to cultivate maize, the consideration of site effects and their contribution to N supply and N surplus become increasingly important for an efficient and environmentally friendly cultivation of maize, especially in combination with manures (Richards et al. 1999). In a 4-year field experiment, we tested the hypothesis that different fertilizer types (pig slurry, cattle slurry and mineral N) and increasing N levels have an effect on maize yields, residual soil mineral N and N leaching on a productive sandy soil.

The results add to the understanding of manure effects on maize yields and N leaching losses on sandy soils with a dynamic N mineralization.

2 Materials and methods

2.1 Site

The experimental site was located in northwest Germany (52°56'44" N and 7°50'17" E). The original soil type can be described as a Gleyic Podzol (WRB; ~Typic Haplaquod, Soil Taxonomy) covered with a relatively shallow layer (0–50 cm) of degraded peat (high moor) which had been converted into an Anthric Podzol (WRB) by deep ploughing in 1975. From that time onwards it had been used as arable land for crop rotations based predominantly on cereals followed by catch crops and occasionally maize and forage grass; input of manure was moderate to high (120–160 kg total N ha⁻¹). The soil profile consists of a recent plough layer (0–30 cm), a deep-ploughed subsoil layer (30–80 cm) consisting of almost vertical layers of peat and sand, and an undisturbed sand

layer below 80 cm; the groundwater level is at 95–140 cm depth. The texture of the topsoil is sand to loamy sand with on average 87% sand (50% fine sand), 11% silt and 2% clay. Deep-ploughed soils are characterized by 6–8% of organic matter, here 6.8%, good natural drainage and good storage capacities for nutrients and water (Scheffer and Schachtschabel 2002). The N mineralization in the top layer and potential N leaching are actually higher in a mature sand-mix-culture (>20 years) compared with a young soil (<5 year) (Müller and Raissi 2002). The mineralization of the peat layer in the subsoil will be quite small/low due to the low pH and the wide C/N ratio. During the course of the experiment all treatments had sufficient to high amounts of macro nutrients in the topsoil (Table 1).

2.2 Set-up of the experiment

The experiment had a two-factorial split-plot design investigating the form of the N input (mineral, cattle and pig slurry; main plots) and the amount of N input (0, 80, 160 and 240 kg N ha⁻¹; sub-plots) and was conducted over a period of 4 years (Table 2); parts of the experiment were extended to 6 years and for comparison some 6-year data are presented where appropriate. Nitrogen input is based on total N; the proportion of NH₄N of the slurries was on average 60% for cattle slurry and 75% for pig slurry of total N. Treatments were repeated four times and sub-plot size was 72 m²; plots and treatments were fixed over all years. Maize (*Z. mays* L. cv. Magda) was cultivated in monoculture and planted within ±5 days of 1 May and harvested within ±5 days of 1 October. The main parameters were dry matter (DM) yield, nutrient concentration in harvested plant material, N offtake with harvest, SMN in autumn and nitrate leaching. Three suction cups had been installed at 70 cm depth in each sub-plot to measure nitrate concentrations in leaching water during winter (October–April).

Table 1 Some chemical soil properties of the experimental site

	Mean	Standard deviation	
P	148	16.6	mg kg soil ⁻¹
K	83	23.8	mg kg soil ⁻¹
pH (CaCl ₂)	4.8	0.12	
TC	4.04	0.495	%
TN	0.19	0.023	%
C/N ratio	21:1		

Sampling depth is 0–30 cm; values refer to oven-dry soil; $n=20$; means and standard deviation, based on CAN and CS plots at the end of the experiment

TC total carbon (corresponds here to total organic carbon), TN total N, double lactate soluble P and K

Table 2 Nutrient input per year with mineral fertilizers and cattle and pig slurries

Form	N input (kg ha ⁻¹)	P input (kg ha ⁻¹)	K input (kg ha ⁻¹)
CAN	0	30 ^a	200 ^a
	80	30 ^a	200 ^a
	160	30 ^a	200 ^a
	240	30 ^a	200 ^a
CS ^b	0	30 ^a	200 ^a
	80	16+30 ^a	93
	160	32+30 ^a	187
	240	48+30 ^a	280
PS ^b	0	30 ^a	200 ^a
	80	23+30 ^a	46
	160	46+30 ^a	91
	240	69+30 ^a	137

Slurries and CAN (mineral N) were applied in two doses; 70% after ploughing in April and 30% in June into the growing stand. Phosphorus (P) was applied as a side dressing with 30 kg P ha⁻¹ to all plots at the time of sowing

CAN calcium–ammonium–nitrate

^a In mineral form as triplephosphate (P) or KCl, muriate of potash (40)

^b CS Cattle slurry—5.7% DM, 3.0 N, 0.6 P, and 3.5 K (kg m⁻³); PS pig slurry—6.3% DM, 6.3 N, 1.8 P, and 3.6 K (kg m⁻³)

Samples from each cup were taken on a weekly basis. The nitrate concentration was determined photometrically with an EPOS 5060 auto-analyzer (Eppendorf). The date for leaching to start was inferred from soil and meteorological data. The nitrate concentrations represent the environmental effect and the potential impact on the groundwater quite well. The nitrate leaching losses were calculated as the product of the nitrate concentration and the amount of water percolating through the profile during a given time. It was assumed that after the soil water content had reached field capacity in autumn, daily drainage equaled rainfall minus evapotranspiration (Haude; DVWK 1996). Summing the nitrate leaching for all sample dates while percolation occurred gave a total loss over winter.

At the beginning of each leaching period in autumn and at the start of the growing season in spring, soil samples were taken from the layers 0–30, 30–60 and 60–90 cm to determine soil mineral nitrogen content (sum of NO₃-N and NH₄-N; SMN).

The N fertilization was divided into two applications, with 70% applied after ploughing and 30% given at a later date before the closing of plant rows in June. Phosphorus and potassium were provided either by cattle or pig slurry or added in mineral form to plots that received no nitrogen (N0) or received N as calcium–ammonium–nitrate (CAN). Thoroughly mixed slurry was applied prior to sowing by a special splash plate and incorporated within 2 h into soil to reduce NH₃ emissions. Application in the growing stand

was done by trailing hose with no further tillage. NH₄-N concentrations were determined before application by a quick test and slurry samples taken for laboratory analysis. The need for P was met by the application of a band of mineral fertilizer placed 5 cm below and 5 cm to the side of the maize seed with 30 kg P ha⁻¹ to all treatments. K in mineral form was applied broadcast and amounted to 200 kg K ha⁻¹. Weeds were controlled by herbicides.

Table 3 shows the climatic conditions for the experimental years. For the experimental site meteorological data from a station of the German Weather Service about 10 km away were obtained. Apart from the second winter, which was exceptionally cool and dry, winter seasons of the other years were relatively mild with average to high rainfall. The summer of year three differed from the other years as it was quite cold and precipitation was below average apart from August and also October, which had incidents of high rainfall.

2.3 Chemical and statistical analyses and calculations

The pH in soil samples was determined in a 0.01 M CaCl₂ solution. Total carbon content (TC) was determined by means of an infrared cell in a LECO SC 444 analyzer (Leco Ltd.). For the non-calcareous sandy soils in this experiment the total organic carbon content might be taken as equal to the TC. All total nitrogen contents (TN) in soil, and plant material were determined through the automated N analyser macro N (Heraeus) following an altered method of Dumas. For the determination of soil mineral nitrogen (SMN), NO₃⁻ and NH₄⁺ were extracted from a 150 g soil sample with 600 mL of 0.0125 M CaCl₂. Filtered extracts were analysed for NO₃⁻ and NH₄⁺ photometrically with an EPOS 5060 auto-analyzer. Potassium and phosphorus were extracted from soil samples following the DL-method (double lactate) as described by Hoffmann (1991). All plant materials were oven dried at 60°C after sampling. The dried and ground material (<1 mm) was analysed for DM content at 105°C and for TN directly.

The apparent nitrogen recovery (ANR) was calculated as (Nyield Nx - Nyield N0)/(N fertilizer supply × 100); x stands for N input treatment and N fertilizer supply refers to the actual N input in kg ha⁻¹ at the specific treatment level. The relative N fertilizer value (RNFV) of slurries was defined as the ratio of ANR of slurry to ANR of mineral fertilizer.

Statistical data analysis was carried out using the Genstat 6.1 software package. Analysis of variance (ANOVA) considered two factors in a split-plot design with four replications. Each block (replicate) was divided into three main plots for the three fertilizer forms (F), and the different levels of N (amount of fertilizer, A) were randomly allocated in sub-plots within each main plot. We checked the assumptions of the statistical models using residual plots. To consider the effect of four consecutive experimen-

Table 3 Mean daily temperature, annual rainfall, and estimated leaching for 4 years

Year (May–April)	Mean daily temperature (°C)	Annual rainfall (mm)	Estimated leaching (October–April; mm)
Year 1	10.0	959.0	408
Year 2	8.5	542.3	66
Year 3	8.7	715.6	216
Year 4	10.4	704.5	232
Long-term average	8.7	783.1	265

tal years, we applied the procedure ‘Arepmeasure’ in Genstat which deals with repeated measurements. Webster (2006) recommends for experiments with graded treatments to not make multiple comparisons. Additionally to comparing individual means for the N forms and quantitative N levels, a polynomial contrast analysis was carried out to investigate whether the effects of the N levels responded linearly or quadratically.

3 Results and discussion

In our experiment we evaluated the contribution of mineral fertilizer, pig and cattle slurry and N from mineralization to yield, residual SMN and nitrate concentrations in leaching water of silage maize. The supply of N from soil resources was very high and had a pronounced effect on the efficiency of fertilizer and manure N.

Results from an ANOVA revealed no significant effect of the fertilizer form, that is, CAN, cattle slurry or pig slurry on yield parameters and residual SMN, while the effect of the amount of fertilizer N was highly significant (Table 4). There was no interaction between N form and amount of N. The $\text{NH}_4\text{-N}$ proportion of the SMN was independent of fertilizer form and of amount of fertilizer and reflected background mineralization from the soil resources. Due to

some loss of original data only averages over all blocks of each treatment on each sampling date were available for data analysis of N leaching and thus no ANOVA was performed.

The response of DM yield, N yield, SMN in autumn, and Nitrate-N concentration in leaching water to increasing N input in three different fertilizer forms is shown for the four experimental years and their average in Figs. 1, 2, 3 and 4; trendlines describe the means. In Table 5, means for the average of 4 years are presented with least significant differences.

3.1 Yields

Although the proportion of $\text{NH}_4\text{-N}$ of total N supplied with cattle and pig slurry was only about 0.60 and 0.75, respectively, maize dry matter yields and N yields did not differ significantly from plots that received mineral N (Fig. 1). The proportion of maize yield of cattle slurry treatments to CAN was 0.96 and that of pig slurry treatment was 0.98 for applications of 80–240 kg Nha^{-1} .

The form of N input had no significant effect on the dry matter yields of silage maize over the 4 years. A contrast analysis showed that the relationship between the amount of N supplied and the dry matter yields was linear. However, the DM yields for N_0 were already quite large

Table 4 Results of ANOVA (repeated measurements considered) for dry matter (DM) yield, N yield, and soil mineral nitrogen (SMN) in autumn as averaged over four experimental years; N balance is N fertilizer+N deposition–N yield

	DM yield (t ha^{-1})		N yield (kg ha^{-1})		N balance (kg ha^{-1})		SMN (kg ha^{-1})	
	v.r.	<i>P</i>	v.r.	<i>P</i>	v.r.	<i>P</i>	v.r.	<i>P</i>
Fertilizer form (F)	0.18	0.843	0.68	0.544	0.68	0.544	2.37	0.174
Amount of fertilizer (A)	15.27	<0.001	30.60	<0.001	1,106.7	<0.001	21.73	<0.001
Year (Y)	61.42	<0.001	49.46	<0.001	49.46	<0.001	90.28	<0.001
F*A	1.09	0.393	1.72	0.154	1.72	0.154	1.02	0.433
Y*F	0.91	0.475	1.58	0.180	1.58	0.180	2.55	0.027
Y*A	1.28	0.266	2.59	0.019	2.59	0.019	5.42	<0.001
Y*F*A	0.48	0.944	0.73	0.736	0.73	0.736	1.29	0.218

N deposition=30 kg $\text{N ha}^{-1} \text{ year}^{-1}$, A. Kayser (Lower Saxony Water Management, Coastal Defence and Nature Conservation Agency), personal communication

v.r. variance ratio (the ratio of the mean squares of treatment and residual), *P* probability level

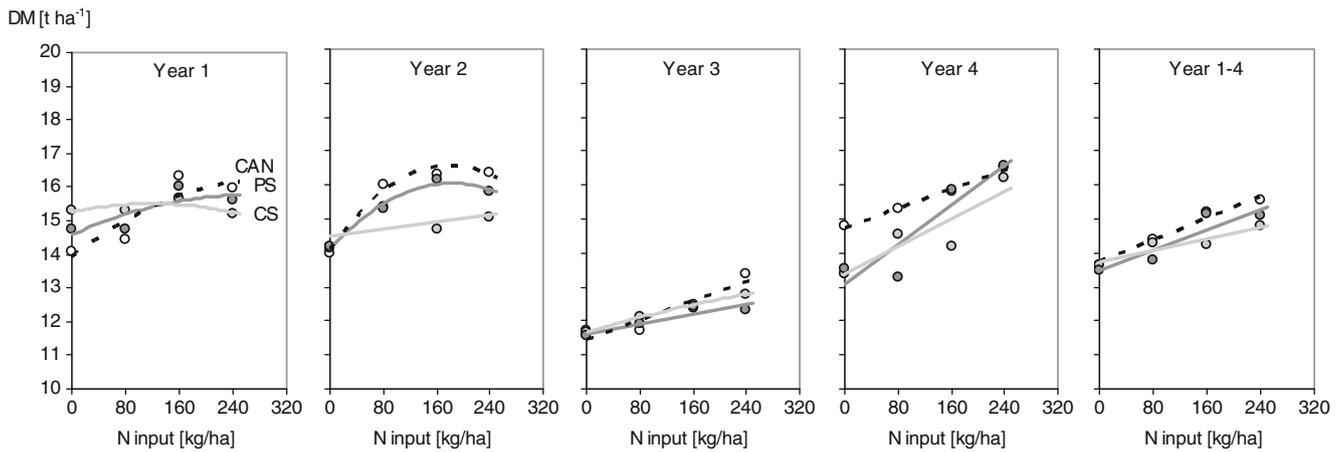


Fig. 1 Dry matter yields (*DM*) of silage maize depending on form of N input (*CAN* calcium–ammonium–nitrate (mineral N), *CS* cattle slurry, *PS* pig slurry) and level of N input (0–240 kg N ha⁻¹). We give trendlines for the single years and for the average of years 1–4 as a linear response (except years 1 and 2) on the basis of means. For the

average of 4 years, a polynomial contrast analysis indicates that there is a linear significant increase in *DM* yield with increasing N input (nlev, $p < 0.001$). *CAN*, $y = 0.0082x + 13.71$; *CS*, $y = 0.0043x + 13.73$; *PS*, $y = 0.0076x + 13.47$

and consequently the slope of the response curve is relatively flat. At higher N rates cattle slurry seemed to be less effective compared with mineral N and pig slurry, which was due to the smaller proportion of NH₄-N in cattle slurry. Differences in *DM* yield between years were pronounced. Year 3 especially, showed distinctively smaller yields, which were affected by an exceptionally cool summer with high rainfall in August and October. The yield response to added N was generally small: every 80 kg N ha⁻¹ gained on average 0.51 t *DM* ha⁻¹. This was coupled with increases in N yield of on average 11.2 kg N ha⁻¹.

N yields increased by 0.17, 0.15 and 0.10 kg N ha⁻¹ per kg N ha⁻¹ year⁻¹ when N was applied with *CAN*, pig slurry and cattle slurry, respectively (Fig. 2). Nitrogen concen-

trations in harvested material (not shown) were influenced by N input and were highest in year 3 and lowest in year 4.

With no N input from fertilizer or slurries, N offtake amounted to 161 kg N ha⁻¹ on average, which was provided by mineralization and atmospheric deposition. Apparent N mineralization (N yield+SMN autumn–(N input+SMN spring+N deposition)) amounted at N0 to 180 kg N ha⁻¹ (140 kg N ha⁻¹ for 6 years of *CAN* and *CS*) as averaged over the three fertilizer forms and at N240 to 45 kg N ha⁻¹ (40 kg N ha⁻¹). Wachendorf et al. (2006a) and Nevens and Reheul (2005) measured an N yield of 60 and 56 kg N ha⁻¹ at N0 compared with 160 kg N ha⁻¹ in our experiments; the SMN and N concentrations in the leached water were likewise much smaller.

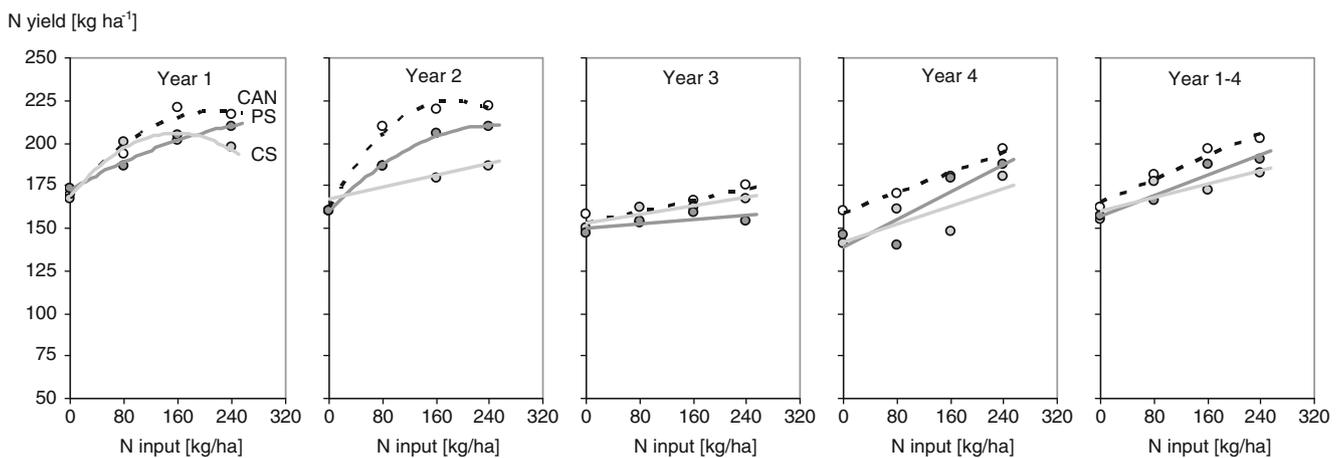


Fig. 2 N yields of silage maize depending on form of N input (*CAN* calcium–ammonium–nitrate (mineral N), *CS* cattle slurry, *PS* pig slurry) and level of N input (0–240 kg N ha⁻¹). We give trendlines for the single years and for the average of years 1–4 as a linear response (except years 1 and 2) on the basis of means. For the

average of 4 years, a polynomial contrast analysis indicates that there is a linear significant increase in N yield with increasing N input (nlev, $p < 0.001$). *CAN*, $y = 0.1704x + 165.23$; *CS*, $y = 0.0991x + 160.03$; *PS*, $y = 0.1498x + 157.18$

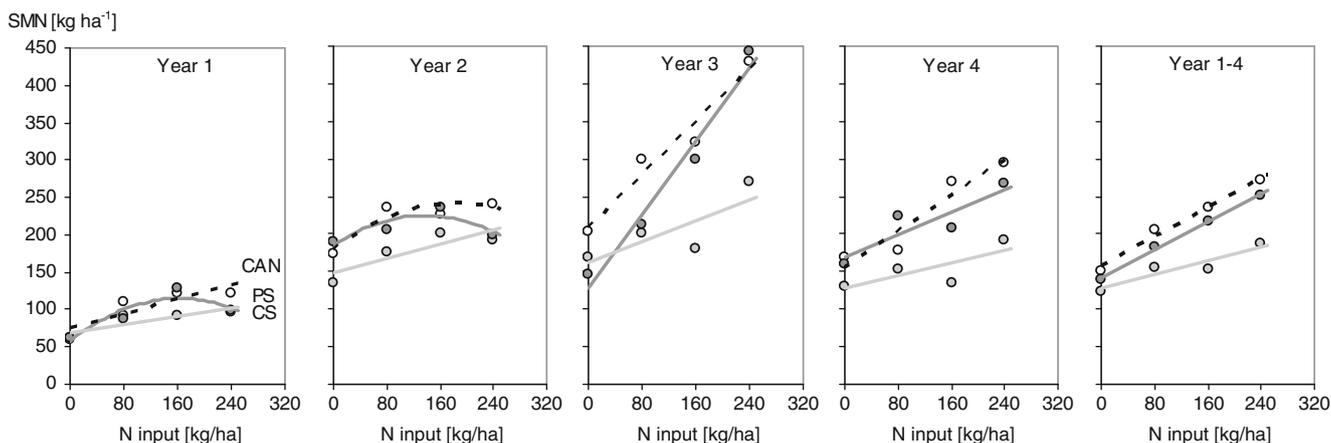


Fig. 3 Soil mineral nitrogen (SMN) in autumn after harvest of silage maize depending on form of N input (CAN calcium–ammonium–nitrate (mineral N), CS cattle slurry, PS pig slurry) and level of N input (0–240 kg N ha⁻¹). We give trendlines for the single years and for the average of years 1–4 as a linear response (except years 1 and 2) on the

basis of means. For the average of 4 years, a polynomial contrast analysis indicates that there is a linear significant increase in SMN with increasing N input (nlev, $p < 0.001$). CAN, $y = 0.4863x + 157.4$; CS, $y = 0.2326x + 126.86$; PS, $y = 0.4666x + 141.68$

3.2 Apparent N recovery

The ANR was generally low and was greater for mineral fertilizer compared with slurries and negatively related to N input (Table 6). Average ANR was 21% for CAN, 17% for cattle slurry, and 15% for pig slurry; the respective values for the N input levels 80, 160 and 240 kg N ha⁻¹ were 21%, 17% and 14%. The high average RNFV for cattle slurry can be explained by the good N recovery at the 80 kg N input

level; at N 240 RNFV were 70% for cattle slurry and 82% for pig slurry.

Schröder et al. (1998) in an analysis of 25 field experiments calculated an ANR of 53% at the economically optimal sum of N input (soil SMN in spring + mineral N + NH₄-N from slurry applications in spring) and an ANR of 73% when the optimal input was reduced by 100 kg N ha⁻¹. They concluded that adjusting the N input to a level below the economically optimal rate can reduce the risk for N losses to the environment with a limited effect on silage yields.

SMN autumn [kg ha⁻¹]

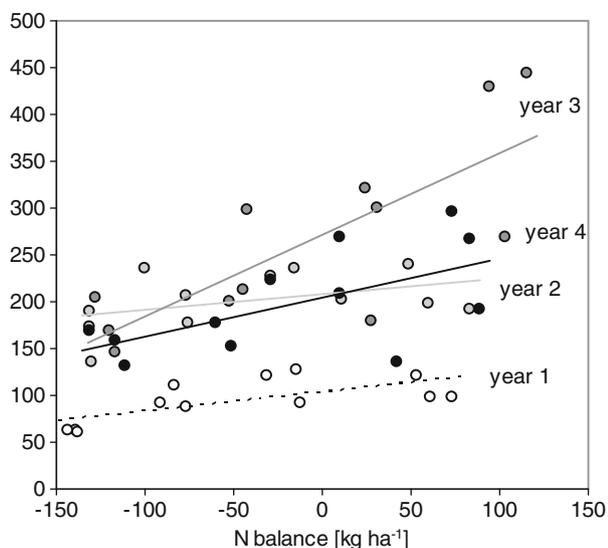


Fig. 4 Relationship between N balance (N input + N deposition - N yield) and soil mineral nitrogen (SMN) in autumn for the four experimental years (year 1, open circles; year 2, light grey; year 3, darker grey; year 4, black); based on means of all N input forms and N levels. Coefficients of determination are 0.46 for year 1; 0.18 for year 2; 0.61 for year 3; and 0.37 for year 4

In the present investigation the slurry treatments resulted in dry matter yields of 90% of that of mineral fertilizer; this implies that some of the organic N in the slurry must have been mineralized and become available to the plants. It has also been suggested that adding slurry to the soil enhances mineralization as a priming effect and that this is larger in soils with higher C and N (Kuzakov et al. 2000; Plaza et al. 2002). Application of pig slurry resulted in DM yields of silage maize that were almost similar to those receiving mineral N. This is astonishing as the NH₄ proportion in pig slurry was 75% of total N. The effect of cattle slurry was smaller but still stronger than expected (Sommer and Hutchings 2001). Schröder et al. (2005) investigated the effects of long-term nitrogen supply from cattle slurry and model calculations indicated that the RNFV of cattle slurry rises from approximately 55–60% when first applied to approximately 80% after continuous application over 6–8 years. The experimental field had a history of manure applications at least over the previous two decades. These findings call for a consideration of the total N in the slurry on sites with a history of manure application, relatively high organic matter content in the soil and a rather shallow groundwater level, instead of considering the NH₄-N proportion alone. This would have implica-

Table 5 Dry matter (DM) yield, N yield, soil mineral nitrogen (SMN) in autumn, and N leaching losses over winter

Fertilizer N form	N level (kg ha ⁻¹)	DM yield (t ha ⁻¹)	N yield (kg ha ⁻¹)	SMN (kg ha ⁻¹)	N leaching (kg ha ⁻¹)
CAN	0	13.65	162	152	98
	80	14.37	182	205	117
	160	15.20	197	235	124
	240	15.56	203	272	166
CS	0	13.63	155	124	78
	80	14.32	177	156	96
	160	14.26	173	152	103
	240	14.81	183	187	113
PS	0	13.50	157	139	81
	80	13.80	167	182	107
	160	15.13	187	217	135
	240	15.08	190	252	176
l.s.d.		1.938 (0.902) ^a	30.9 (13.4) ^a	75.0 (44.4) ^a	
CAN		14.70	186	216	126
CS		14.26	172	155	97
PS		14.38	175	198	125
l.s.d.		1.885	30.2	70.4	
	0	13.59	158	138	86
	80	14.16	175	181	106
	160	14.87	185	201	121
	240	15.15	192	237	152
l.s.d.		0.521	7.8	25.6	

Means over 4 years with least significant differences (l.s.d.); no ANOVA results for N leaching as only already averaged values were available from each sampling date

l.s.d. least significant difference ($P=0.05$), CAN calcium–ammonium–nitrate (mineral N), CS cattle slurry, PS pig slurry

^a When comparing within the same level of N form

tions for the livestock management of the farm. Generally, the efficiency of N from slurry is likely to be reduced by volatile losses and denitrification after application—influenced by plant uptake and weather conditions—and transformation processes of organic N in the soil. When mineral N fertilizers (CAN) and slurries are compared, gaseous losses should be considered. Slurry applications were split, with 70% of total N applied before sowing and 30% applied to the growing

stand before closing of the rows. Total gaseous losses could have been at least 16% and 10% of applied total N for CS and PS, respectively. For CAN, we might assume losses of 5% of NH₃ and N₂/N₂O for applied N. These figures are based on assumed losses of 20% for CS and 8% for PS of ammoniacal N for broad spreading with a splash plate and incorporation within 1–4 h. For application of slurry with a trailing hose system in a growing stand we assume 25% and

Table 6 Apparent N recovery (ANR) and relative N fertilizer values (RNFV) for fertilizer application forms (CAN, CS, and PS) at different levels of N input

Fertilizer N form	ANR (%)					RNFV (4 years)
	N 80	N 160	N 240	Mean (4 years)	Mean (6 years)	
CAN	24 (25) ^a	22 (25) ^a	17 (19) ^a	21	23	100
CS	28 (27) ^a	11 (13) ^a	12 (15) ^a	17	18	82
PS	12	19	14	15		71

For slurries, total N was considered. Treatments CAN and CS were extended to 6 years

CAN calcium–ammonium–nitrate (mineral N), CS cattle slurry PS pig slurry

^a Values in brackets refer to means of years 1–6

10% losses for CS and PS, respectively. Unavoidable losses of N_2 and N_2O were considered with 5% of ammoniacal N (Gutser and Ebertseder 2002).

3.3 Soil mineral nitrogen in autumn

The SMN for N0 input ranged from 60 (year 1) to 173 (year 3), indicating a relatively high supply of N from mineralization processes (Fig. 3). For every kg N added with fertilizer, SMN increased by about 0.48 kg N ha⁻¹ for CAN and PS and 0.23 kg N ha⁻¹ for cattle slurry. Differences between years were pronounced and after the poor harvest of year 3, SMN was highest. The spring SMN (not shown) as averaged over the fertilizer forms was for N0: 44, 184, 94 and 99 kg N ha⁻¹ and for N240: 44, 273, 172 and 138 kg N ha⁻¹, for years 1–4, respectively.

In the experiments of Wachendorf et al. (2006b), soil mineral N for the mineral fertilizer treatments ranged from 35 kg N ha⁻¹ for N0 to 55 kg N ha⁻¹ at input of 150 kg N ha⁻¹. In our experiments, corresponding SMN values for 0 and 150 kg N ha⁻¹ input of mineral fertilizer were 157 and 230 kg N ha⁻¹. Some treatments in our study were extended to 6 years, but still there was hardly a trend to smaller proportions of N0 to N160 in N yields. Likewise, SMN in autumn in years 5 and 6 was lower than the 4-year average, but the proportion of N0 to N160 stayed at the same level, which indicates effects of weather and sampling time. Figure 4 shows the relationship between N balance and SMN in autumn in our experiments.

Apart from year 3 with small dry matter yields, due to unfavourable weather conditions, the response of SMN to N surpluses was quite weak in all other years (an effect that continued in years 5 and 6 for CAN and PS). The good response to fertilizer input and the relatively small SMN values and leaching losses in the experiments of Wachendorf et al. (2006a, b) are mainly due to less ‘background’ mineralization than in the results presented here. The differences between sites and soil conditions can have pronounced effects on the performance of silage maize and subsequent SMN and N leaching. Richards et al. (1999) conducted 15 field trials with silage maize in England, and assessed the contribution of four levels of N input with manure and fertilizer on yield and SMN. The soil mineral N after harvest varied widely between sites and soils and ranged from 43 to 369 kg N ha⁻¹ when no N was applied. Response of yield to N input was generally rather weak and was attributed partly to the extent of net mineralization after drilling and partly to the small effects of manure to SMN (Richards et al. 1999). Schiermann (2004) draws the conclusion from extensive survey data that site-related factors contributed substantially to the variability in SMN in autumn, while the influence of N rates were rather small or masked by other factors.

3.4 Nitrate concentrations in leachate

Nitrate concentrations that may enter the upper groundwater level were quite high and exceeded the limit in the EU Nitrate Directive of 11.3 mg NO₃-N L⁻¹ (50 mg L⁻¹ nitrate) three- to sevenfold (Fig. 5). N concentrations were likely to be influenced by the amount of mineral N left in the soil after harvest, continuing mineralization during late autumn and winter and the amount and pattern of rainfall during the leaching period (October to April).

Estimated leaching losses of NO₃-N were in a range of 20 to 290 kg N ha⁻¹ with the smallest losses corresponding to small amounts of water leached and little N input as in year 2 and largest losses occurring in year 3 with relatively high NO₃-N concentrations in the leaching water. When averaged over the four experimental years calculated, N leaching losses amounted to 86 kg N ha⁻¹ for N0, 106 kg N ha⁻¹ for N80, 121 kg N ha⁻¹ for N160, and 152 kg N ha⁻¹ for N240; N losses for CAN and pig slurry were in a similar range (Table 5). These calculated potential losses are quite large and constitute a substantial loss of nutrients from the field and a serious imposition to the environment, especially water quality.

The relationship between SMN in autumn and NO₃-N concentrations in leached water averaged over the years at the treatment level was good ($R^2=0.91$) and showed a positive linear trend ($y=0.2923x-1.7728$). An average NO₃-N concentration of 11.3 mg L⁻¹ would correspond to an SMN (0–90 cm) in autumn of 45 kg ha⁻¹.

It often appears to be difficult to achieve NO₃-N concentrations much below 11.3 mg L⁻¹ when sandy soils are used for arable crop production (Köhler et al. 2006). Nitrate concentrations in water leaching beyond 75 cm were at a range of 24–120 mg NO₃-N L⁻¹ in our experiments. Even the treatment with no N input from mineral fertilizer or slurry had an average concentration of 38 mg NO₃-N L⁻¹. While we report only on nitrate leaching during the winter period, N losses in early summer can be substantial after high precipitation on soils with a low water holding capacity and a slow juvenile development of the maize (Herrmann et al. 2005). Wantulla et al. (1988) report nitrate-N concentrations under maize cultivation with different fertilizer strategies on a stagno-gleyic Luvisol in northwest Germany of 23 mg L⁻¹ three years after the last slurry application. N losses summed up for the whole year were in a range of 113–208 kg N ha⁻¹ for a year with very high precipitation and amounted to 69–108 kg N ha⁻¹ in a normal year with good maize yields (Wantulla et al. 1988). Wachendorf et al. (2006b) found that nitrate leaching losses under maize on a gleyic podzol soil increased exponentially with increasing amounts of N input and mineral N fertilizer. Highest leaching losses at an N fertilization with 40 m³ slurry and 150 kg mineral N

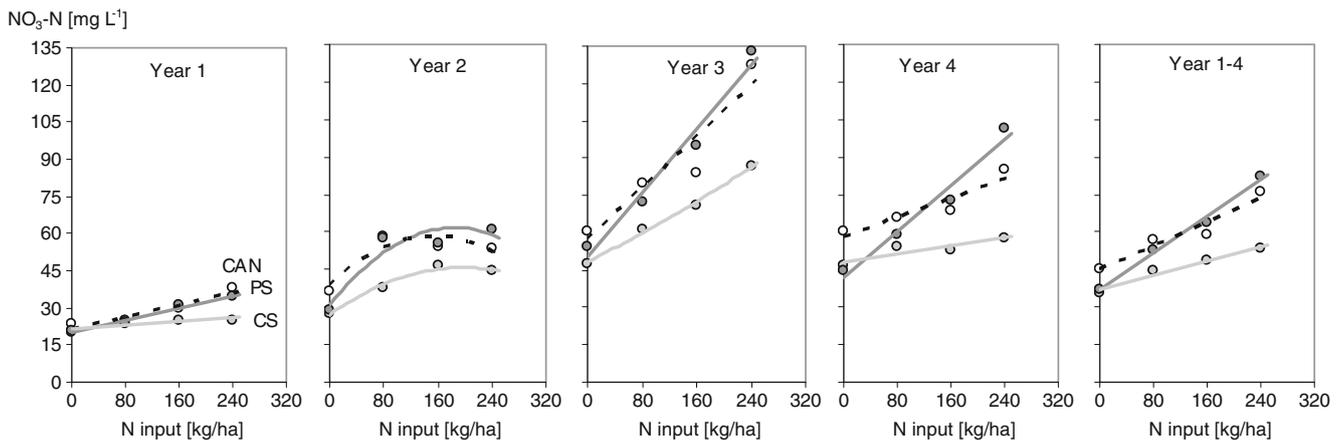


Fig. 5 Flow-weighted $\text{NO}_3\text{-N}$ concentrations in leached water during winter depending on form of N input (*CAN* calcium–ammonium–nitrate (mineral N), *CS* cattle slurry, *PS* pig slurry) and level of N input (0–240 kg N ha^{-1}). We give trendlines for the single years and for the

average of years 1–4 as a linear response (except year 2) on the basis of means. For the average of 4 years, the trendlines can be defined as: *CAN*, $y=0.1177x+45.24$; *CS*, $y=0.073x+36.79$; *PS*, $y=0.183x+37.10$

and no cover crops amounted to 30 kg $\text{NO}_3\text{-N}$ ha^{-1} (17 mg NL^{-1}); no slurry and omission of mineral N with understorey resulted in losses of 2.5 kg $\text{NO}_3\text{-N}$ ha^{-1} (1.4 mg NL^{-1}). These losses appear to be remarkably small and demonstrate that an effective use of N and a strong response to reduced N input allows a maize production with little environmental impact on soils where mineralization is not substantial.

3.5 Denitrification and mineralization

Denitrification has probably prevented most of the nitrate from reaching deeper groundwater. At the experimental site denitrification rates in the shallow groundwater (166–220 cm in depth) as determined in laboratory and in situ studies were 140–152 $\mu\text{g N kg}^{-1}$ and 14–119 $\mu\text{g N kg}^{-1} \text{ day}^{-1}$, respectively (Well et al. 2003). However, this ability to substantially reduce the nitrate charge to the groundwater is finite. Denitrification in soils with a high nitrate load and a limited amount of reducing agents can lead to higher N_2O , which is also undesirable (Mehranfar 2003). Due to intensive agricultural use there probably has been considerable N translocation for some time from sources like mineralization and fertilizer input at the expense of the denitrification capacity. Mehranfar (2003) concludes from laboratory experiments and modelling scenarios that the limited reservoir of reducing agents at the site might end in 10–20 years assuming a continuing N surplus of 50–100 kg N $\text{ha}^{-1} \text{ year}^{-1}$.

Deep-ploughed organic soils (sand-mix culture) have a higher yield capacity than comparable sandy soils but mineralization and consequently N leaching might also be higher (Müller and Raissi 2002). While the mineralization potential and N leaching of older sand-mix cultures from high moors is regarded as intermediate, this can be high if

the origin is from low moors. The experimental site actually has some features of a low moor, as it was a part of a slightly depressed area.

With N from mineralization being as high as on our experimental site, the effects of measures like reducing the N input or the use of cover crops seem to be limited in the short term. However, grass as a cover crop might take up 60 kg N ha^{-1} in above-ground and below-ground biomass as well as preventing wind and water erosion and influence water infiltration and evapotranspiration (Schröder et al. 1996; Wachendorf et al. 2006b). Mitigation options like this are of particular interest in areas with high concentration of livestock where a certain amount of manure would be applied in practice.

The loss of the denitrification buffer in the zone above the shallow groundwater would stress the importance of a balanced land use for groundwater protection even further. This affects the fertilization practices as well as land management on the field, farm and regional scale, for example, the ongoing trend of turning grassland into arable land. A neighbouring field to the maize site had been used for a cut grassland experiment and N leaching was five times less than for comparable N inputs with silage maize. A further expansion of maize production to former grassland sites with higher organic matter content could exacerbate the situation on a regional scale.

4 Conclusions

Fields with a relatively high total C and total N content in the soil as well as a high biological activity, have to be regarded as problematic for cultivation of maize and need, at least, special consideration with respect to management activities. In our experiments, amounts of residual soil

mineral nitrogen were large and subsequently the N concentrations in soil leachate during winter were high. This was probably buffered by denitrification before reaching the groundwater. However, these processes might contribute to increased N₂O emission and rely on a limited reservoir of reducing agents which might end in 10–20 years at the experimental site. Therefore, reducing the residual N that is prone to leaching from sources like mineralization, animal manures and mineral fertilizers is still at the core of groundwater protection measures. The site responded only weakly to reductions in N fertilization. Thus, using it as grassland would be desirable and much more sustainable with respect to groundwater protection.

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