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ORIGINAL PAPER

Reduced nitrogen losses from drained temperate agricultural peatland after mineral soil coverage

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Abstract

Draining peatlands for agriculture induces peat decomposition, subsidence, and carbon (C) and nitrogen (N) losses, thereby contributing to soil degradation and climate change. To sustain the agricultural productivity of these organic soils, coverage with mineral soil material has increasingly been used. To evaluate the effect of this practice on the N flows within the plant–soil system, we conducted a ^{15}N tracer experiment on a drained peatland that was managed as an intensive meadow. This peatland was divided into two parts, either without (reference "Ref") or with ~ 40 cm mineral soil cover (coverage "Cov"). We applied $^{15}NH_4{}^{15}NO_3$ on field plots to follow the fate of ^{15}N in plant–soil system over 11 months. In addition, N mineralization was determined by laboratory incubation. The field experiment showed that Cov lost less ^{15}N ($p < 0.05$) than Ref, even though plant ^{15}N uptake was similar at both sites. The lower net N loss from the Cov site was accompanied by higher soil ^{15}N retention. The laboratory incubation revealed a ~ 3 times lower N mineralization at Cov than at Ref, whereas the N release per unit soil N was around two times higher at Cov than at Ref, suggesting a faster SOM turnover rate at Cov. Overall, the mineral soil cover increased the retention of fertilizer-N in the soil, thus reducing the system N losses. Our result indicates that agricultural production on drained peatland is less harmful to the environment with mineral soil coverage than using drained peatland directly.

Keywords Organic soil · ^{15}N recovery · N mineralization · Intensively managed grassland

Introduction

Although peatlands only cover approximately 3% of the terrestrial surface area, they are an essential soil organic matter (SOM) pool and can store 8–14 Gt N globally (Yu et al. 2010; Loisel et al. 2014; Leifeld and Menichetti 2018). However, long-term drainage for agricultural production has already resulted in ~ 51 Mha degraded peatlands worldwide, with the highest share occurring in tropical and temperate regions, where around half of the initial peatland surface has been disturbed due to agricultural production, forestry, or peat extraction (Kasimir et al. 2018; Leifeld and Menichetti 2018). Peatland degradation is typically associated with peat decomposition, which result in C and, to a smaller extent, in N losses, as well as strong soil subsidence. As a consequence, soil C to N ratios decrease (Klemedtsson et al. 2005; Leifeld 2018). The decomposition of peat is a substantial contributor to the N supply for agricultural production in drained peatlands. Therefore, the soil N supply and plant N uptake from drained peatlands might be higher than in mineral soil.

Around 30% of the agriculturally used peatland is managed as grassland globally (Leifeld and Menichetti 2018; Evans et al. 2021). For the temperate zone, plant N uptake in grasslands has been widely explored in both mineral soils and organic soils. It has been reported from grasslands on mineral soil that plant biomass accumulated up to ~ 130 kg N ha^{-1} $year^{-1}$ without fertilization in Germany (Bessler et al. 2012). Müller et al. (2011) found that, based on a 38-year field observation in Germany, the aboveground grass N uptake ranges from 50 to 200 kg N ha^{-1} $year^{-1}$ with N application of ~ 200 kg N ha^{-1}. In a study on grasslands on organic soil, Sonneveld and Lantinga (2011) reported an aboveground grass N uptake of 342 kg N ha^{-1} based on a 3-year field experiment in drained peatland without

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fertilization in the Netherlands. Schothorst (1977) even reported an aboveground grass N uptake of ~ 400 kg N ha^{-1} from a non-fertilized drained peatland in the Netherlands. These data tentatively suggest that plant N uptake in drained organic soil might be generally higher than in mineral soil, which might be related to the higher soil N supply in drained peatland through organic matter decomposition. Higher soil N mineralization often leads to a supply of N exceeding grass uptake, which consequently results in greater N losses to the environment of grass produced on drained organic soil compared to production on mineral soil (Pijlman et al. 2020). It has been estimated that with the ongoing agricultural use of degraded peatland, 9.7 Mt N $year^{-1}$ will be released to the environment annually, and c. 2.3 Gt N will be released cumulatively with the full degradation of all currently managed peatland (Leifeld and Menichetti 2018). Therefore, it is vitally important to evaluate how the N losses from drained peatland can be reduced.

In order to compensate for continued soil subsidence of drained organic soils and thereby to maintain agricultural productivity, adding mineral soil as a cover fill with a thicknesses of 0.2–0.5 m on the surface of organic soil has increasingly been adopted by farmers working in Switzerland and other European countries (Schindler and Müller 1999; Ferré et al. 2019). With mineral soil cover, the soil N balance of drained peatlands may change due to various factors. First, the smaller surface soil C and N content in the mineral soil cover material supports smaller microbial biomass and microbial activity (Wardle 1998). This may result in lower SOM mineralization rates with mineral soil coverage compared with the surface soil from non-covered drained organic soil. Second, mineral soil cover might increase fertilizer N retention in drained peatland owing to its overall smaller N content. Third, a cover fill may also change other physical–chemical soil properties (e.g., clay content, soil pore volume, and soil cation exchange capacity) that feedback into soil N dynamics (Barrett and Burke 2002). Finally, for the peat layer underneath the mineral soil coverage, the addition of mineral soil material may compress the peat layer and push it deeper into zones with lower oxygen availability, thereby reducing the mineralization of easily degradable N in those peat layers. A prior study conducted at the same site as studied here proved that mineral soil cover induced a substantial reduction of N_2O emissions (Wang et al. 2022), underpinning a strong influence of mineral soil coverage on the N balance in the soil–plant system of the drained peatland. However, a mechanistic understanding of the impact of mineral soil cover on the N cycling in the plant–soil system of drained organic soils is still missing.

In this study, we examined the N dynamics and N loss in plant–soil system in a drained peatland under grassland use both with and without mineral soil coverage. We did so by using isotopically labeled ^{15}N fertilizer in combination with measurements of the corresponding N pools in soil, roots, and harvests. The application of ^{15}N-enriched fertilizer is considered a useful and targeted tool for tracing the fate of applied N in plant–soil systems (Rahman and Parsons 1999; Wesselsperelo et al. 2006; Sebilo et al. 2013; Rowlings et al. 2016; Kalu et al. 2021). The specific objectives of this study were to (1) determine the fertilizer N recovery and fertilizer allocation in the plant–soil system in drained peatland with and without mineral soil coverage; (2) assess the soil mineral N (N and ^{15}N) release in drained peatland with and without mineral soil coverage; and (3) quantify the impact of mineral soil cover on the total plant–soil system N loss from drained peatland.

Materials and methods

Field site

The field experiment was carried out in the Swiss Rhine Valley, at the site Rüthi (47° 17′ N, 9° 32′ E), a drained fen with a peat thickness of ~ 10 m. The site has a cool temperate-moist climate with a mean annual precipitation of 1297 mm and a mean annual temperature of 10.1 °C (1981–2010, https://www.meteoswiss.admin.ch; for precipitation and temperature during the experimental period please see Fig. S1). The site was drained with ditches before 1890 (https://map.geo.admin.ch). In 1973, an intensive drainage system with pumps and pipes was built. The site was used as pasture, and since 2013 as an intensively managed meadow. From 2006 to 2007, one part of the field (~ 2 ha) was covered with mineral soil material (without mixing with the peat underneath) to improve the agricultural usability. We established the field experiment at this mineral soil coverage site (Cov, with mineral soil coverage thickness ~ 40 cm) and used the adjacent drained organic soil (~ 9 ha) without mineral soil coverage as the reference (Ref, see Fig. S2 A). The basic soil properties for both sites are provided in Table 1. Both sites have similar vegetation and identical farming practices with 5–6 cuts per year and ~ 230 kg N ha^{-1} fertilizer application, both as slurry (applied with a splash plate) and as mineral fertilizer (ammonium nitrate or ammonium sulfate, applied with fertilizer sprayer). The atmospheric N deposition at the study site for 2015 is 20–30 kg N ha^{-1} $year^{-1}$ (Rihm and Künzle 2019). Dominant grass species are *Lolium perenne*, *Alopecurus pratensis*, *Festuca arundinacea*, *Trifolium spec*., and *Festuca pratensis*.

Experimental design and field management

The study was conducted from July 2020 to July 2021. In July 2020, eight separate plots (four for Cov, four for Ref; size, 3.5 m × 1.5 m) were distributed on the experimental site. Each plot was divided into two subplots (1.5 m × 1.5 m, Fig. S2.B), and the distance between the two subplots was

Table 1 Soil properties of drained organic soil with (Cov) and without (Ref) mineral soil coverage ($n = 16$)

Parameter	Cov	Ref
Sand, 3–8 cm (%)[a]	31.8	0.6
Silt, 3–8 cm (%)[a]	52.3	67.3
Clay, 3–8 cm (%)[a]	15.9	32.1
Total pore volume, 3–8 cm (%)[a]	58.4 ± 1.5	75.1 ± 0.5
pH, 0–25 cm[b]	7.2	5.2
pH, 25–50 cm[b]	7.0	5.1
pH, 50–75 cm[b]	5.8	4.7
NH_4^+ (N mg kg^{-1} dry soil)[a]	2.62 ± 0.96	37.33 ± 12.07
NO_3^- (N mg kg^{-1} dry soil)[a]	2.81 ± 1.21	5.16 ± 1.13
Bulk density (g cm^{-3})	0.7 ± 0.09	0.3 ± 0.06
N stock, 0–15 cm (t ha^{-1}) [c]	6.5 ± 0.03	14.4 ± 0.5
Total N content cm (%) [c]	0.36 ± 0.02	1.4 ± 0.09
C to N ratios [c]	18.5 ± 0.7	18.2 ± 5.0

[a]Data from Wang et al. (2022)

[b]Soil pH was determined based on the samples from Wang et al. (2021)

[c]Total N content, C content, C to N ratios, and soil physical data down to 60 cm are provided in Supplementary Table S1 and S2

0.5 m. At each plot, one subplot received ^{15}N double-labeled ammonium nitrate ($^{15}NH_4{}^{15}NO_3$) as a treatment plot, and the other one received the same amount of non-labeled ammonium nitrate (NH_4NO_3) as a control plot. For these treatment plots, $^{15}NH_4{}^{15}NO_3$ was dissolved in water, and the salt solution was applied in three campaigns at the same time the farmer fertilized the overall field. We dissolved 1.35 g, 1.35 g, and 0.8 g 98 atom $\%^{15}$ N $^{15}NH_4{}^{15}NO_3$ in 2.25 L water for each application, which is equivalent to 1 mm precipitation per application. The control plot (i.e., the plot without a label) always received the same amount of NH_4NO_3 solution. The additional N input rate was chosen based on the typical field fertilizer N application. A total of extra ^{15}N input of 0.57 g N m^{-2} for each plot were chosen; this was equivalent to ~ 2.5% of the regular field fertilizer N input, and it was assumed that this small extra dose would not cause a major disturbance in the N cycle of the ecosystem. The plot received the same regular fertilizer as the overall field. Extra $^{15}NH_4{}^{15}NO_3$ salt solution was sprayed directly onto the ground on 10 September 2020, 25 March 2021, and 13 May 2021. In order to spread the fertilizer solution homogenously, each subplot was divided into 15 units (0.3 m × 0.5 m, Fig. S2.C), and the same amount of the fertilizer solutions was applied to each unit.

Plant and soil sample collection and analysis

During the experimental period, soil and plant samples were taken 1 day before the regular field harvest events in October of year 2020, May, June, and July of year 2021. In addition, extra soil samples were taken on August 2020 and August 2021. Soil samples from the first sampling were used to determine the background ^{15}N signature over all experimental plots, and those from the last sampling were used to determine the soil bulk density from 0–5 cm to 5–15 cm. For each subplot, composite soil samples from 3 units were collected by using a 6.5-cm-diameter corer for 0–20 cm depth and a 2.6-cm-diameter corer for 20–60 cm depth. Those samples were divided into 4 layers: 0–5 cm, 5–15 cm, 15–30 cm, and 30–60 cm. The samples from Cov were additionally divided at the boundary of mineral soil cover and underlying peat by a stainless steel knife during field sampling. After sampling, soil samples were stored at 4 °C in a cooling room overnight, and visible root and stones were removed from composite soil samples the next day. Soil samples were then dried at 105 °C for 72 h, ground with mortar and pestle, milled in a ball mill (Retsch, MM 400, Germany) at 25 rotation s^{-1} for 3 min, and finally loaded in a tin capsule to determine the soil N and ^{15}N content via elemental analysis isotope ratio mass spectrometry (EA-IRMS) (vario PYRO cube, Elementar, Germany and isoprime precisION, Elementar, Germany).

For each subplot, aboveground biomass samples from three units were harvested by grass clippers to a height of 3 cm. At the same unit, root samples were collected by taking soil cores with a 6.5-cm-inner-diameter corer down to a depth of 20 cm. Composited grass samples were dried at 60 °C in the oven for 72 h to determine the dry biomass and to calculate the aboveground biomass based on the covered area of the three units (0.15 m^2 each). Dried plant samples were cut into small pieces, milled in a ball mill (Retsch, MM 400, Germany) at 25 rotation s^{-1} for 3 min, and then loaded into a tin capsule to determine the grass N and ^{15}N content with elemental analysis isotope ratio mass spectrometry (EA-IRMS) (vario PYRO cube, Elementar, Germany and isoprime precisION, Elementar, Germany). Roots were extracted from each soil core in the lab. To do so, soil material from the soil core was removed by hand, and the remaining root from the removed soil material was picked out. The left soil core and the roots that were picked out from the soil were submerged in distilled water for 2–3 h, and then put on a fine mesh screen to be washed with a gentle water shower until the residual soil material was removed. The bare roots were dried at 60 °C in the oven until the constant weight (~ 48 h) to determine the dry biomass and calculate the biomass of the root based on the covered area of the three soil cores (0.033 m^2 each) for each subplot, and then, the root N and ^{15} N content was determined by following the same procedure described above.

Laboratory incubation

To determine the net N and ^{15}N mineralization rate of the surface soil at both sites, the 0–5 cm and 5–15 cm soil samples, which were collected in October 2020, were incubated for 28 days. Five duplicated ($n = 160$) soil samples equivalent to 10 g dry soils were weighted into 50-ml PET containers with soil moisture adjusted to 60% of their water holding capacity. Water holding capacity was determined following Franzluebbers (2020). The PET containers were incubated at 25 °C, and soil moisture was adjusted every 2 days by adding distilled water. After 0, 7, 14, 21, and 28 days of incubation, the soil samples were suspended in 80 ml 0.01 M $CaCl_2$ salt solution to extract soil N and ^{15}N (Steffens et al. 1996), shaken at 160 cycles min^{-1} for 30 min, and filtered. Total N and ^{15}N from the soil extracts were determined by EA-IRMS (vario TOC cube, Elementar, Germany and iso TOC cube, Elementar, Germany). Daily net N and ^{15}N mineralization rates (N_{r_min}, mg N kg^{-1} soil day^{-1}; $^{15}N_{r\_min}$, mg ^{15}N kg^{-1} soil day^{-1}) from two sites and depth were calculated based on the regression slope of the five total dissolved N (mg N kg^{-1} soil) and ^{15}N (mg ^{15}N kg^{-1} soil) against their incubation times (day). The specific N and ^{15}N mineralization rate (specific N_{r_min}, mg N g^{-1} soil N day^{-1}; specific $^{15}N_{r\_min}$, mg ^{15}N g^{-1} soil ^{15}N day^{-1}) was calculated as the regression slope of the five specific dissolved N (mg N kg^{-1} soil N) and ^{15}N (mg ^{15}N kg^{-1} soil ^{15}N) against their incubation times (day). The specific extractable N was determined as the ratio of the total extractable N (mg N kg^{-1} soil) and the soil N content (%). Correspondingly, the specific extractable ^{15}N was determined as the ratio of the total dissolved ^{15}N (mg ^{15}N kg^{-1} soil) and the soil ^{15}N content (%).

Isotope calculation and statistics

The N isotope ratios of the samples are presented by using the δ notation (Fry 2006).

$$\delta^{15}N(‰) = \left(\frac{R_{sample}}{R_{standard}} - 1\right) \times 1000 \quad (1)$$

where R_{sample} and $R_{standard}$ are the ratios between ^{15}N and ^{14}N of the sample and the standard, respectively. Here, atmospheric N_2 is used as a standard with $R_{standard} = 0.003665$ (Mariotti 1983).

The isotope enrichment in the sample from the treatment plot ($\delta^{15}N_{sample}$) is expressed as ^{15}N enrichment relative to that of the control plot ($\delta^{15}N_{control}$).

$$^{15}N \text{ enrichment}(‰) = \left(\frac{\delta^{15}N_{sample} - \delta^{15}N_{control}}{\delta^{15}N_{control} + 100}\right) \times 1000 \quad (2)$$

The recovery of the ^{15}N fertilizer in the labeled N pools is calculated as follows:

$$^{15}N(\%) = \left(\frac{\left(\%^{15}N_{sample} - \%^{15}N_{control}\right) \times M_{pool}}{\left(\%^{5}N_{label} - \%^{15}N_{control}\right) \times M_{label}}\right) \times 100 \quad (3)$$

where $\%^{15}N_{sample}$ is ^{15}N atom percent in the soil sample from the labeled plot; $\%^{15}N_{control}$ is ^{15}N atom percent in the corresponding control plot; M_{label} is the amount of the ^{15}N applied to the treatment plot (g ^{15}N m^{-2}); and $\%^{15}N_{label}$ is the ^{15}N atom percent in the labeled fertilizer, and M_{pool} is the N amount of the labeled pool (g N m^{-2}). In this study, there are three labeled pools, $M_{pool,grass}$, $M_{pool,root}$ and $M_{pool,soil}$. $M_{pool,grass}$ and $M_{pool,root}$ were determined based on the dry biomass for each plot. $M_{pool,soil}$ was calculated as follows:

$$M_{pool,soil} = \sum BD_i \times L_i \times N_{sample} \quad (4)$$

BD_i is the soil bulk density (g cm^{-3}) at four different soil depths (0–5 cm, 5–15 cm, 15–30 cm, and 30–60 cm). Soil bulk density from 15–30 cm to 30–60 cm was determined plot wise based on the correlation between soil bulk density and the soil organic carbon from Wang et al. (2021). L_i is the thickness of each depth (cm), and N_{sample} is the total N content of the soil sample from the labeled plot (%).

The mass balances of ^{15}N in the system were used to quantify the ^{15}N recovery in the system; any ^{15}N which was not retained in the plant and soil system was defined as losses. Plot-based ^{15}N losses (N_{losses}) were calculated as the difference between ^{15}N input through ^{15}N tracer application, as well as the N output from harvest and the ^{15}N retained in soil and roots. The ^{15}N input through regular fertilizer and atmospheric ^{15}N deposition is accounted for by the ^{15}N abundance from the associated control plot. The cumulative N losses at each harvest event are calculated as follows:

$$N_{losses,i} = \sum_{i=1}^{i=n} {}^{15}N_{fer,i} - \sum_{i=1}^{i=n} {}^{15}N_{grass,i} - {}^{15}N_{root,i} - {}^{15}N_{soil,i} \quad (5)$$

where $N_{losses,\ i}$ is the N losses after the i th harvest event, $n = 1, 2, 3, 4$; $^{15}N_{fer}$ is the cumulative ^{15}N input through fertilization; $^{15}N_{grass}$ is the cumulative ^{15}N uptake through harvest; and $N_{root,\ i}$ and $N_{soil,\ i}$ are the ^{15}N retained in roots and soil at the i th harvest event, respectively.

Statistical analysis and data visualization were performed using the open source software R (version 4.1.3). Significant differences between the two sites for soil and plant N content, $\delta^{15}N$ content, ^{15}N enrichment, net N mineralization rate, ^{15}N recovery, and ^{15}N losses were determined using a t-test. Significant differences in the ratio of the specific $^{15}N_{r\_min}$: N_{r_min}, N and ^{15}N release rate, and specific N and ^{15}N mineralization rate in soil layers 0–5 cm and 5–15 cm between the two sites were determined through ANOVA.

In case of a significant effect, a Tukey's HSD test was performed for multiple pairwise comparisons between different sampling dates. The error probability was set as $p < 0.05$. The results were always reported as mean ± 1 standard error (se).

Results

Effect of mineral soil coverage on plant biomass, N uptake, and plant ^{15}N enrichment

During the experimental period, the cumulated grass yield was not different between sites (Table 2); only in June 2021, the yield at Cov was higher ($p < 0.05$) than at Ref. The harvested grass took up 274.34 ± 22.78 kg N ha^{-1} from the mineral soil coverage (Cov) site over four harvest events, which was significantly higher than that taken up from the drained peatland (Ref) site (229.97 ± 10.56 kg N ha^{-1}). The higher grass N uptake of Cov was not observed in all harvest events; it was significant only for the harvest in June 2021, whereas for the rest of the cuts, no significant difference between Cov and Ref was found. The ^{15}N enrichment of the grass varied largely between the different harvest events. It was higher at Ref compared to Cov in October 2020, whereas no significant differences were found for the other harvest events.x

For roots, the differences in biomass, N uptake, and ^{15}N enrichment were not constant between the two sites. In June 2021, root biomass and ^{15}N enrichment were significantly higher ($p < 0.05$) at Cov than at Ref, and a significantly higher root N content was found at Ref in July 2021.

Effect of mineral soil coverage on soil ^{15}N enrichment

Applications of ^{15}N labeled fertilizer induced an increase in the soil ^{15}N signature. At both sites, the highest soil ^{15}N signature was found in July 2021 after the three labeling events were finished, although no ^{15}N tracer was applied directly before that sampling event. At 0–5 cm soil depth, the ^{15}N enrichment was 146.0 ± 13.3‰ at Cov and 49.4 ± 13.7‰ at Ref (Fig. 1D). At 5–15 cm soil depth, the ^{15}N enrichment was 32.7 ± 8.8‰ at Cov and 7.4 ± 1.4‰ at Ref (Fig. 1D). The higher ^{15}N signature was only found at the surface 0–30 cm, below 30 cm depth, and the soil ^{15}N enrichment was similar to the value prior the ^{15}N tracer application, which was near zero (Fig. 1). The surface (0–5 cm, 5–15 cm) soil ^{15}N enrichment was higher ($p < 0.05$) at Cov than at Ref, whereas below 15 cm, the difference in ^{15}N enrichment between the sites was less pronounced at any sampling date (Fig. 1).

Table 2 Plant biomass, N uptake, and ^{15}N enrichment of a drained peatland with (Cov) and without mineral soil coverage (Ref)

Sample	Item	Site	October 2020	May 2021	June 2021	July 2021	Total
Above-ground biomass	Biomass	Cov	2015 ± 133	4044 ± 541	4905 ± 183	2853 ± 111	13,817 ± 738
	(kg ha−1)	Ref	2144 ± 89	3755 ± 238	4063 ± 264	3049 ± 156	13,011 ± 290
			ns	ns	*	ns	ns
	N uptake	Cov	57.76 ± 5.86	74.82 ± 10.49	81.23 ± 5.04	54.59 ± 5.28	274.34 ± 22.78
	(kg N ha^{-1})	Ref	61.74 ± 3.62	62.40 ± 4.34	55.26 ± 4.81	49.00 ± 3.11	229.97 ± 10.56
			ns	ns	*	ns	*
	^{15}N enrichment	Cov	1977.63 ± 124.24	2252.37 ± 170.86	2187.33 ± 193.50	705.53 ± 146.43	-
	(‰)	Ref	2518.91 ± 153.99	2212.39 ± 143.02	2393.74 ± 100.52	760.24 ± 25.15	-
			*	ns	ns	ns	-
Root	Biomass	Cov	4133 ± 548	4893 ± 404	5396 ± 616	3604 ± 312	-
	(kg ha^{-1})	Ref	4371 ± 576	4781 ± 442	4127 ± 322	3933 ± 224	-
			ns	ns	*	ns	-
	N uptake	Cov	73.5 ± 10.6	53.2 ± 3.9	45.9 ± 3.7	27.7 ± 2.5	-
	(kg N ha^{-1})	Ref	92.1 ± 12.7	53.0 ± 5.2	43.3 ± 3.9	42.8 ± 5.0	-
			ns	ns	ns	*	-
	^{15}N enrichment	Cov	901.67 ± 77.97	1847.36 ± 303.83	2073.23 ± 195.26	1403.81 ± 155.81	-
	(‰)	Ref	710.57 ± 70.95	1613.24 ± 160.84	1445.56 ± 192.88	1408.22 ± 102.89	-
			ns	ns	*	ns	-

Significant differences between the two sites over the experimental period are indicated with asterisks ("**", $p < 0.01$, "*", $p < 0.05$, "ns", no significant difference).

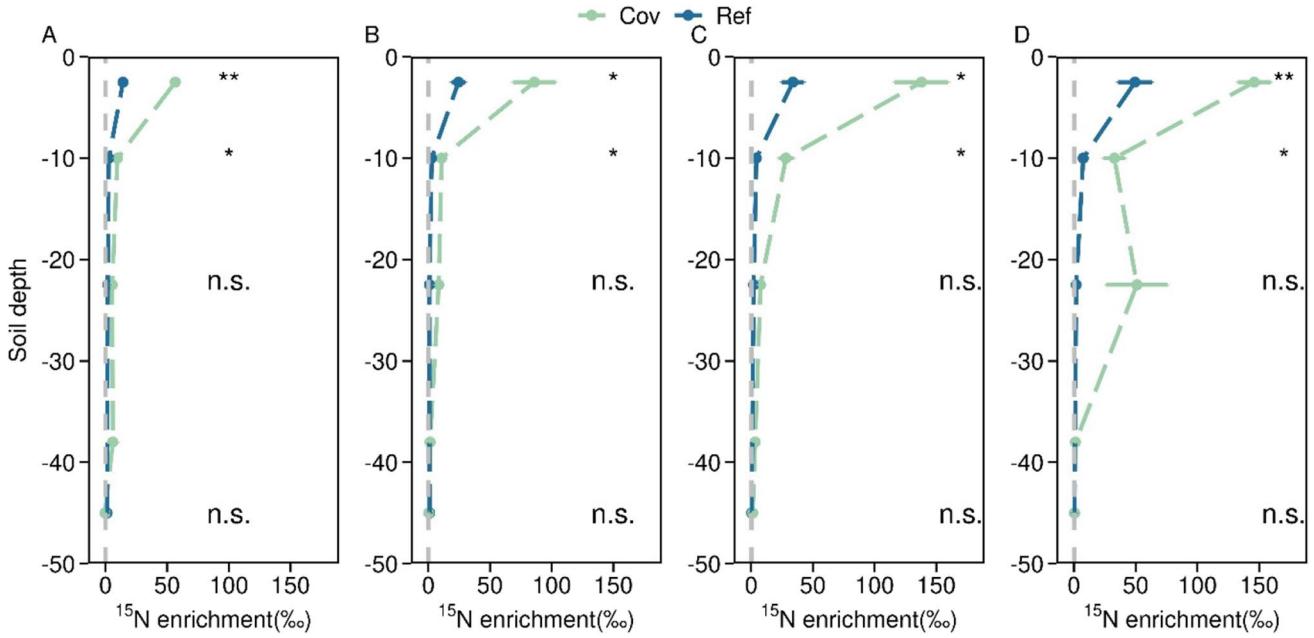

Fig. 1 Soil profile (0–60 cm) ^{15}N enrichment (mean ± se, $n = 4$) at sampling dates in October 2020 (**A**), May 2021 (**B**), June 2021 (**C**), and July 2021 (**D**) from the drained peatland with (Cov) and without mineral soil coverage (Ref). The symbols always denote the middle depth of each sampled segment. The dashed gray line indicates the background ^{15}N enrichment before ^{15}N application, which is zero. For each sampling date, significant differences between two sites at different soil depths are indicated with asterisks ("**" $p < 0.01$, "*" $p < 0.05$, "ns" no significant difference)

At Cov, the ^{15}N signal moved from the surface soil to the deeper (15–30 cm) layer during the growing season and resulted in a slightly higher ^{15}N enrichment (49.4 ± 13.7‰) at 15–30 cm depth compared with the upper layer (32.7 ± 8.8‰) at the last sampling date (July 2021). However, no such trend was found for Ref (Fig. 1D).

The effect of mineral soil coverage on ^{15}N recovery from drained organic soil

During the experimental period, the recovery of ^{15}N in plants (aboveground biomass and roots) was not different between Cov and Ref. The cumulative tracer exports through aboveground biomass harvest accounted for 32.2 ± 2.2% and 30.0 ± 0.3% of the applied ^{15}N for Cov and Ref, respectively (Fig. 2A). Roots took up 2.5 ± 0.3% and 3.9 ± 0.5% of the applied ^{15}N from Cov and Ref, respectively, after the three labeling events were finished (Fig. 2A). Hence, a significant part of the applied ^{15}N was not used by the plants. A share of 10–20% was incorporated into the soil N pool. At site Cov, 19.8 ± 2.0% of the tracer remained in the soil N pool, more ($p < 0.05$) than at Ref (9.8 ± 3.2% see Fig. 2B). Overall, site Cov showed smaller N losses ($p < 0.05$) compared to Ref. At Cov, 45.4 ± 3.0% of the applied labeled mineral fertilizer was lost outside the plant–soil system boundary of the study, whereas at Ref, the loss accounted for 56.2 ± 3.1% (Fig. 2C).

Nitrogen mineralization

The average amount of soil N_{r_min} was significantly higher ($p < 0.05$) at Ref (6.07 ± 0.84 mg N kg^{-1} day^{-1}) than at Cov (2.10 ± 0.15 mg N kg^{-1} day^{-1}) at the 0–5 cm depth (Fig. 3A). In addition, in the deeper layer, the average amount of soil N_{r_min} was significantly higher ($p < 0.01$) at Ref (5.45 ± 0.26 mg N kg^{-1} day^{-1}) compared to Cov (1.71 ± 0.13 mg N kg^{-1} day^{-1}; Fig. 3B). A similar trend to that of the total N release was found for the average release of soil ^{15}N. Ref released ^{15}N at higher rates (0.027 ± 0.001 mg ^{15}N kg^{-1} day^{-1} at 0–5 cm; 0.022 ± 0.002 mg N kg^{-1} day^{-1} at 5–15 cm) than at Cov (0.009 ± 0.001 mg ^{15}N kg^{-1} day^{-1} at 0–5 cm; 0.005 ± 0.0004 mg N kg^{-1} day^{-1}, at 5–15 cm) (Fig. 3C and D).

In addition, the average amount of N and ^{15}N release showed no difference at the 0–5 cm depth for the two sites (73.67 ± 6.27 mg N m^{-2} day^{-1}, 0.31 ± 0.02 mg ^{15}N m^{-2} day^{-1} at Cov and 77.71 ± 8.08 mg N m^{-2} day^{-1}, 0.35 ± 0.04 mg ^{15}N m^{-2} day^{-1} at Ref, respectively). However, it was significantly ($p < 0.05$) higher at Ref (204.68 ± 10.53 mg N m^{-2} day^{-1}, 0.81 ± 0.06 mg ^{15}N m^{-2} day^{-1}) than at Cov (155.70 ± 15.41 mg N m^{-2} day^{-1}, 0.48 ± 0.05 mg ^{15}N m^{-2} day^{-1}) at the 5–15 cm depth (Table 3).

Specific N mineralization

The specific soil N mineralization (specific soil N_{sr_min}) was significantly higher with mineral soil

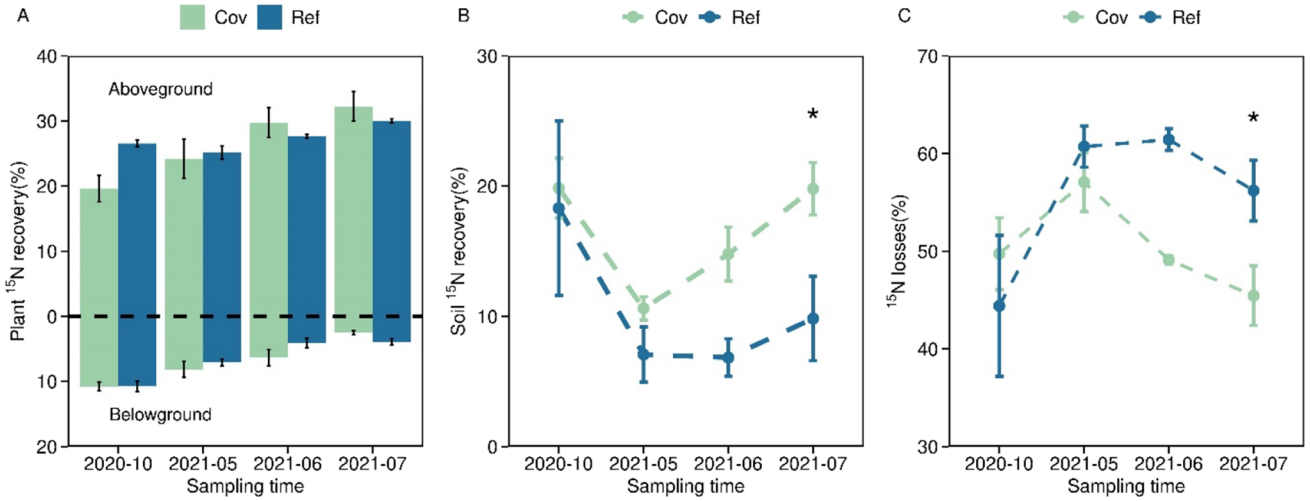

Fig. 2 Budget of ^{15}N tracer based on mass and isotope balances for plants (**A**), soil (**B**), and N losses (**C**) from drained peatland with (Cov) and without mineral soil coverage (Ref). Significant differences in cumulative soil ^{15}N recovery and ^{15}N losses between the two sites over the experimental period are indicated with asterisks ("**" $p < 0.01$, "*" $p < 0.05$, "ns" no significant difference) in **B** and **C**. The dashed line in **A** separates the aboveground biomass ^{15}N recovery and the belowground root ^{15}N recovery

coverage (Table 3) for both soil layers (0–5 cm and 5–15 cm). Throughout 28 days of incubation, the specific soil N_{sr_min} at site Cov was 0.60 ± 0.07 mg N g^{-1} N day^{-1} and 0.35 ± 0.03 mg N g^{-1} N day^{-1} at Ref at a soil depth of 0–5 cm. A similar difference was also found at 5–15 cm soil depth, where Cov released 0.58 ± 0.03 mg N g^{-1} N day^{-1}, significantly more than Ref (0.36 ± 0.03 mg N g^{-1} N day^{-1}). In addition, specific soil $^{15}N_{sr\_min}$ was also significantly higher with mineral soil coverage (Table 3) for both soil layers (0.68 ± 0.10 mg ^{15}N g^{-1} ^{15}N day^{-1} at 0–5 cm depth and 0.71 ± 0.14 mg ^{15}N g^{-1} ^{15}N day^{-1} at 5–15 cm depth) than at Ref (0.39 ± 0.02 mg ^{15}N g^{-1} ^{15}N day^{-1} at 0–5 cm depth and 0.37 ± 0.02 mg ^{15}N g^{-1} ^{15}N day^{-1} at 5–15 cm depth).

The ratio of the specific soil $^{15}N_{sr\_min}$ release to the N_{r_min} release was above one for both layers and sites (Fig. 4). No difference was found between the two sites; however, the ratio of specific $^{15}N_{sr\_min}$ and N_{sr_min} from the 5–15 cm soil layer was lower than from the 0–5 cm soil layer. Significant differences among the two soil layers and the two sites are indicated with lowercase letters (ANOVA and Tukey's honest significant differences)

Discussion

The effect of mineral soil cover on soil ^{15}N retention and N mineralization

Soil ^{15}N retention

The field ^{15}N tracer experiment showed that 10% of the applied ^{15}N tracer resided in the soil pool in the Ref site, of which more than 90% was found in the top 30 cm of soil. This result was similar to the ^{15}N retention from the drained fen peatland reported by Augustin et al. (1997) who found that 10–20% of the applied labeled ^{15}N fertilizer were recovered in the soil pool in a ^{15}N tracer experiment from two drained peatlands in Germany, of which more than 90% was located at the 0–20 cm depth. To the best of our knowledge, soil ^{15}N recovery from drained peatland with mineral soil coverage has never been studied. Our results indicate that the soil ^{15}N recovery (~ 20% of the applied ^{15}N tracer) from the Cov site was generally significantly higher than at Ref at the end of the study period, suggesting a better retention of the fertilizer-N though mineral soil coverage. The recovery was at the lower end of the range of data reported from ^{15}N tracer (with ^{15}N enriched fertilizer or slurry) studies in grassland on mineral soil in Europe. This included 20–25% soil ^{15}N retention from a grassland in southern England (Jenkinson et al. 2004), ~ 15% from grassland in the Netherlands (De Vries et al. 2011), and 30–40% from grassland in Germany (Zistl-Schlingmann et al. 2020).

However, we observed a downward movement of the ^{15}N tracer to the deeper layer at Cov, whereas no such trend was found at Ref (Fig. 1D) over the course of the experiment. This indicates that, despite a higher overall recovery in the studied soil layers, fertilizer N might leach faster in the covered mineral soil material at Cov than in the drained peatland at Ref. This may, after longer periods, also change the overall recovery once the leachate leaves the investigated zone of 0–60 cm. The higher ^{15}N leaching from Cov may be attributed to the low absorption rate of the mineral cover material compared to the degraded peat, as the sand content of the mineral soil coverage

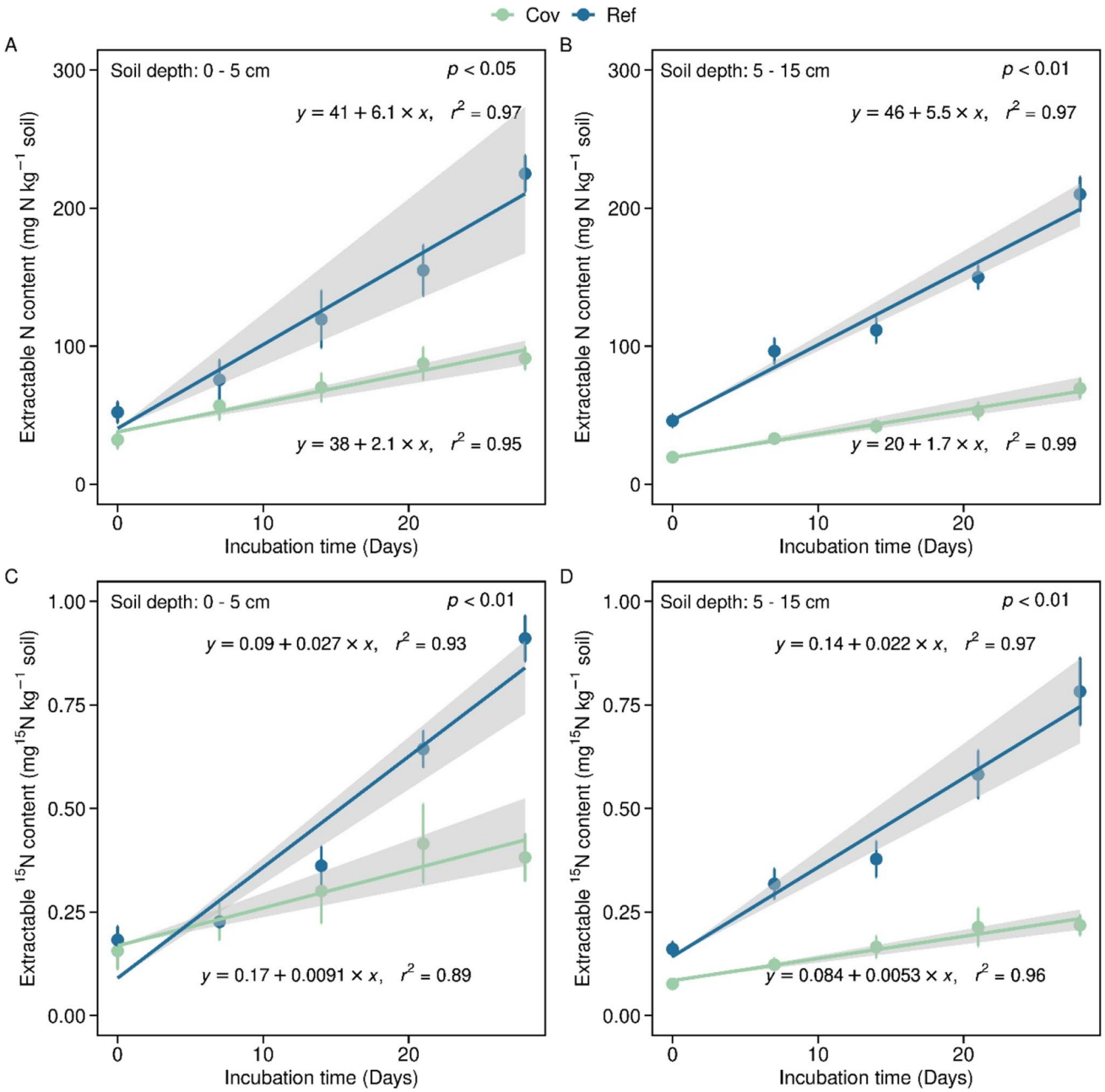

Fig. 3 Extractable soil N and ^{15}N (mean ± se, $n = 4$) for different incubation days in soil layers 0–5 cm (**A, C**) and 5–15 cm (**B, D**) from drained peatland with (Cov) and without mineral soil coverage (Ref). The shaded area indicates the maximum and minimum rate of soil N_{r_min} and $^{15}N_{r\_min}$ release from soil. The p values indicate significant differences (t-test) of soil N_{r_min} and $^{15}N_{r\_min}$ release between the two sites

is much higher than that of the peat at Ref (Table 1). Hence, the low adsorption potential of the sand for anions compared to organic materials with higher anion sorption capacity may have induced a higher N leaching at the mineral soil layer from Cov.

Higher soil ^{15}N recovery at Cov might be due to a higher microbial ^{15}N uptake. Microbial N use is positively correlated with soil substrate availability and soil pH (Elrys et al. 2022). Compared with Ref, the SOM pool from Cov is small (Table 1), but relatively young and labile as previous shown from a $^{14}CO_2$ measurement on the same site (Wang et al. 2021). The higher availability of labile SOM stimulates soil microbial activity, which ultimately promotes microbial N uptake (Barrett and Burke 2000; Booth et al. 2005; Yang et al. 2022). Moreover, the relative old and stable SOM pool from Ref may exist in forms that the microorganism cannot easily use (Baldock and Skjemstad 2000; Fontaine et al. 2007). This may result in a

Table 3 Average soil N release rate and specific N mineralization rate in soil layers at depths of 0–5 cm and 5–15 cm from drained peatland with (Cov) and without mineral soil coverage (Ref)

	Cov		Ref	
	0–5 cm	5–15 cm	0–5 cm	5–15 cm
Soil N release rate mg N m^{-2} day^{-1}	73.67 ± 6.27^{c}	155.70 ± 15.41^{b}	77.71 ± 8.08^{c}	204.68 ± 10.53^{a}
Soil ^{15}N release rate mg ^{15}N m^{-2} day^{-1}	0.31 ± 0.02^{b}	0.48 ± 0.05^{b}	0.35 ± 0.04^{b}	0.81 ± 0.06^{a}
Specific soil N mineralization rate mg N g^{-1} soil N day^{-1}	0.60 ± 0.07^{a}	0.58 ± 0.03^{a}	0.35 ± 0.03^{b}	0.36 ± 0.03^{b}
Specific soil ^{15}N mineralization rate mg ^{15}N g^{-1} soil ^{15}N day^{-1}	0.68 ± 0.08^{a}	0.71 ± 0.14^{a}	0.39 ± 0.02^{b}	0.37 ± 0.02^{b}

Significant differences among the two soil layers and the two sites are indicated with lowercase letters (ANOVA and Tukey's honest significant differences).

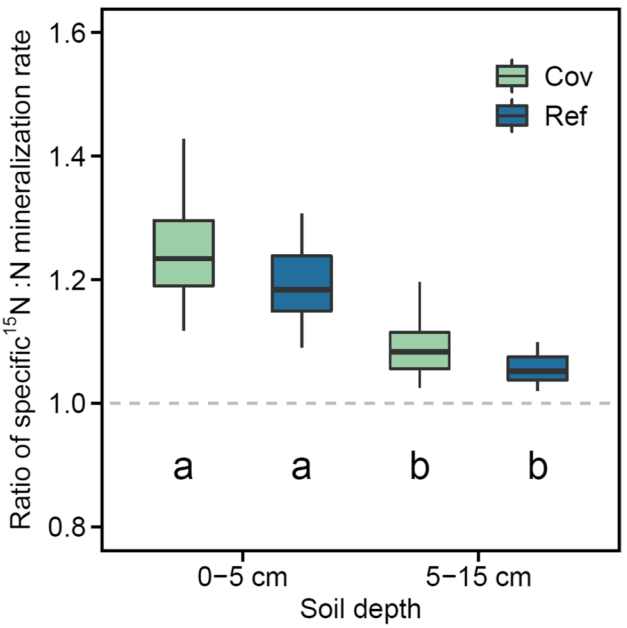

Fig. 4 The ratio of specific ^{15}N mineralization rate to specific N mineralization rate in soil layers with depth of 0–5 cm and 5–15 cm from drained peatland with (Cov) and without mineral soil coverage (Ref). This ratio was calculated from the specific extractable ^{15}N (mg ^{15}N kg^{-1} soil ^{15}N) and specific extractable N (mg N kg^{-1} soil N) from five different measurement days. Significant differences among the two soil layers and the two sites are indicated with lowercase letters

lower microbial N uptake at Ref than at Cov. In addition, soil pH of the surface layer at Cov was higher than at Ref (Table 1), which may enhance soil microbial N uptake and further induce better soil ^{15}N retention at Cov than at Ref, due to the enhanced microbial activity under high soil pH (Zhang et al. 2017).

Soil N mineralization

The laboratory incubation results showed that soil N_{r_min} and soil $^{15}N_{r\_min}$ at 0–5 cm and 5–15 cm depth were significantly higher ($p < 0.05$) at Ref than at Cov. We attribute this to the overall higher soil N content of the surface peat compared to the mineral soil cover material. In contrast, the specific soil N_{sr_min} release was higher at Cov than at Ref (Table 3), which indicates that the surface soil organic matter (SOM) at Cov was more labile compared to the Ref site. The labile SOM pool at Cov coincided with the young carbon age at Cov from the former study on the same site (Wang et al. 2021). By definition, the labile SOM pool decomposes very quickly and is easily accessible to plants and microbes (Dungait et al. 2012; Liu et al. 2017).

At both sites, the soil $^{15}N_{r\_min}$ release was faster than the N_{r_min} release (Fig. 4), implying that the added ^{15}N turnover rate was higher than the gross N turnover rate. This finding indicates that the newly applied mineral N (^{15}N and ^{14}N), after incorporation into SOM, was preferably stored in the labile soil N pool. This finding is consistent with former studies showing that the exogenous N input is mostly labile (Shevtsova et al. 2003; Mulvaney et al. 2009; Sebilo et al. 2013). This part of the soil organic N pool releases available N for plant uptake in the growing season, but likewise bears the risk of N losses to the environment.

The effect of mineral soil coverage on plant ^{15}N uptake

Over the experimental period, both sites had similar aboveground biomass; however, the aboveground plant N uptake was higher at Cov than at Ref ($p < 0.05$), suggesting that mineral soil coverage not only sustains the agricultural productivity of the drained peatland, but also increases the fertilizer N use efficiency. However, mineral soil coverage did not influence plant and root ^{15}N content. At both sites, plants took up ~ 30% of the applied ^{15}N fertilizer, similar to the results reported from a meta-analysis of ^{15}N tracer studies, which found that on average, 30% of the applied ^{15}N is taken up by plants in grassland (Templer et al. 2012). It is often assumed that plant N uptake tends to be higher with higher soil N availability (Stevens et al. 2005; Tateno and Takeda 2010). However, we found that the application of

mineral soil material, which was relatively poor in N and also released an absolutely smaller amount of N in the incubation experiments, did not reduce the plant N uptake and the plant ^{15}N recovery. The similar aboveground biomass and ^{15}N recovery might be driven by the ample amount of N supplied at both sites and the lack of any N limitations. During the experimental period, ~ 230 kg N ha^{-1} was applied equally to both sites, and the soil N_{r_min} results suggest that soil mineralization could supply further N, exceeding the demand for grass production at both sites. Therefore, as N was not limited in the system, the presence of relatively N-poor mineral soil did not impair aboveground yields.

The effect of mineral soil coverage on N loss

Fertilizer N loss reduction

The two sites received ~ 230 kg N ha^{-1} $year^{-1}$ fertilizer N input. Together, the fertilizer N input and the soil N supply largely exceed the plant N demand and consequently lead to N losses to the environment, i.e., release into the atmosphere via ammonia volatilization and denitrification as well as via leaching to the groundwater (Robertson and Vitousek 2009; Bowles et al. 2018). At Cov, less of the applied N was lost through the experimental period, considering the storage in the 0–60 cm soil layer. Thus, mineral soil coverage at this site may prevent ~ 25 kg N ha^{-1} $year^{-1}$ fertilizer N from being lost to the environment compared with Ref if no substantial leaching below 60 cm will occur.

Effects on peat decomposition

The higher soil N release from Ref at 0–15 cm soil depth (Table 3) also implies a rapid peat decomposition and peatland degradation, as the soil N losses are closely linked to the C losses from the SOM mineralization (Leifeld et al. 2020; Klein et al. 2022). However, we only have evidence for this for the topsoil, whereas the peat underneath the mineral soil coverage was not used for determining soil N_{r_min} in the incubation experiment due to the reasons below. Firstly, the soil ^{15}N signature underneath 15 cm soil layer was nearly natural abundance (Fig. 1B), which makes it impossible to determine the soil ^{15}N mineralization from the deeper soil layer. Second, the oxygen availability in the deeper soil layer is difficult to simulate in laboratory incubation, and the atmosphere oxygen availability may overestimate the soil N and ^{15}N mineralization rate from the deeper soil layer.

It may be suspected that at Cov, these subsoil organic layers may have a higher N_{r_min} release than the surface organic soil from Ref site due to two possible mechanisms. First, the covered mineral soil material revealed some N leaching (Fig. 1). This leachate from the mineral soil material may stimulate the decomposition of the peat underneath the mineral soil cover via positive priming (Kuzyakov et al. 2000). Second, the mineral cover enhanced the soil pH of the peat layers underneath (Table 1). As SOM decomposition increases with soil pH (Sinsabaugh et al. 2008), a higher potential for peat decomposition underneath the mineral soil coverage may be possible.

On the other hand, oxygen availability is vital for peat mineralization (Blodau 2002; Tiemeyer et al. 2016). For the peat layer underneath the mineral soil coverage, the oxygen availability for SOM mineralization is reduced (Jørgensen et al. 2012), leading to a presumably lower N release. In addition, the absence of fresh plant residue input into the deeper layers of the organic soil underneath might limit SOM mineralization (Song et al. 2018; Zhang et al. 2021). Fresh plant inputs are the primary source of SOM formation, which could not only determine the chemical composition of SOM, but also impact soil microbial activities. The exclusion of fresh plant inputs may lead to a N limitation for microorganisms and further limit the N mineralization for peat underneath the mineral soil coverage (Mooshammer et al. 2014). Moreover, a former study conducted at the same site found that SOC from mineral soil material contributed greatly to heterotrophic soil respiration at the Cov site (Wang et al. 2021). The contribution of the peat layer underneath the mineral soil coverage was relatively small compared to the contribution of peat C at Ref (Wang et al. 2021). The lower contribution of subsoil peat to C loss from Cov suggests that mineral soil cover might be able to reduce the peat decomposition rate despite incoming N leachate and a higher pH. However, further in situ soil profile-based SOM mineralization experiments are still needed to support this interpretation.

Conclusion

Our findings suggest that mineral soil coverage has the potential to reduce N losses (due to higher soil N retention) from drained peatland and, hence, may make agricultural production on drained peatland less harmful to the environment compared to the continued direct use of these soils. However, we would like to point out that from a nature conservation standpoint as well as climate mitigation strategy, mineral soil coverage does not replace or substitute the mitigating effect that can be achieved by rewetting. Rather, we aim to encourage further research about mineral soil coverage as a peatland management measure in regions, where peatland rewetting is not supported, be it for reasons of national food and feed provision, economic incomes, or political strategies.

Our field ^{15}N trace experiment and laboratory incubation together provide the first insight into how mineral soil coverage influences the N balance of the plant–soil system

in agriculturally managed drained peatland. Over the experimental period, mineral soil coverage of drained peatland significantly reduced the system fertilizer N loss. For the deeper peat layer, the effect of mineral soil coverage on peat decomposition and mineralization still needs to be further explored. In summary, the study suggests that mineral soil coverage, a measure used by farmers to counterbalance subsidence, provides an opportunity for reducing the environmental pollution induced by the agricultural use of drained peatland. Furthermore, the reduced N losses with mineral soil coverage from our study also highlighted the need for multiply field observation to evaluate the effectiveness of mineral soil coverage in general terms.

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Data Availability Data from this study are included in the article and supplementary material; further inquiries can be directed to the corresponding author.

Declarations

Conflict of interest The authors declare no competing interests.

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