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Evaluating peat pasture performance under elevated groundwater conditions

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ABSTRACT

Historically, Dutch peatlands have been drained for agriculture, particularly as pastures for dairy production. While drainage increases productivity, it degrades the peat layer, leading to high greenhouse gas emissions and soil subsidence. Raising the depth of the groundwater table (WT) to 20 cm below field level, contrasted to commonly drained peat at approximately 50 cm below field level, is considered an effective solution to limit peat degradation. However, it is unclear how raising the WT with an active water infiltration system (AWIS) would affect grass yield and quality, and whether dairy farming can be maintained. We studied the effect of raising WT on grass production, quality, and nitrogen utilization, using experimental plots with different WT and fertilization levels in 2020–2024. Raising WT from 50 to 20 cm below field level resulted in an average decrease of 9 % in herbage dry matter yield (DMY), with significant variation between years due to varying weather conditions. The reduction in DMY could partly be explained by the observed decrease in soil N supply in wetter conditions, due to a lower mineralization rate. Increased N fertilization could mediate the lower DMY but increases nutrient losses. Herbage crude protein concentration was moderately affected by WT, and more strongly by yearly variation. Although results are on plot level, which excludes losses originating from trafficking by machinery and animals, they indicate that raising WT is compatible with relatively high grass production and similar grass quality. Consequences on farm level need to be further explored.

1. Introduction

Globally, 12 % of peatlands have been drained, particularly to make them suitable for agricultural use (UNEP, 2022). Pristine peatland ecosystems are known to play a very important role in climate regulation, ensuring water quality and providing habitats to many different species (Farrell et al., 2021; Xu et al., 2018). In the Netherlands, 80 % of the 270.000 ha of peat soils have been drained and are used as pasture for dairy production (Poppe et al., 2021). Drainage lowers the groundwater table of peat soils, which leads to an increased load bearing capacity for machinery and animals, and allows for an improved productivity (i.e. through early spring fertilization) of the land (Wilson et al., 2016). However, the lower groundwater table results in oxidation of the peat top layer, and fertilization results in nutrient losses. As a result, instead of contributing to climate regulation, drained peat soils emit large quantities of CO_2 and N_2O (Langeveld et al., 1997; Poppe et al., 2021). The use of peatlands, often associated with intensive dairy farming practices (e.g. high fertilization levels and high stocking densities), can deteriorate water quality through leaching and run-off, which subsequently leads to a loss of biodiversity (Ramchunder et al., 2009). The loss of organic matter due to decomposition of the peat layer, moreover, leads to soil subsidence and results in high social costs to restore damaged housing and infrastructure (Brouns et al., 2015; Hoving et al., 2008; van den Born et al., 2016). According to the European Commission, only 10 % of Europe's peatlands are currently in good condition, meaning they are able to provide the important climate and water regulating functions that pristine peatlands deliver (European Commission, 2021). Therefore, there is a growing awareness to protect and restore peatlands, including those currently used for agricultural purposes. For instance, the EU Water Framework Directive and Habitat

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Directive mention the importance of peatlands as buffer habitats, and the EU Biodiversity Strategy for 2030 calls for ensuring protection and restoration of peatlands (European Commission, 2021; European Commission, n.d.), implying that previously drained peatlands should be rewetted.

Rewetting, or raising groundwater tables, reduces peat decomposition, thus lowering greenhouse gas (GHG) emissions. Recent research showed that a groundwater table of around 10–20 cm below field level results in minimal emissions of N_2O, CO_2 and methane, as compared to the current mean groundwater table of 50 cm (Evans et al., 2021; Tiemeyer et al., 2016). Raising groundwater tables and keeping them at a stable depth can be challenging however, due to fluctuations in weather conditions, within and between years (Holshof et al., 2011). In addition, several studies have found that increasing the water table through raising ditch water levels affects soil parameters and hence, has implications for grassland management and production. For instance, the mineralization rate decreases, resulting in a lower amount of available nitrogen for the grass to take up (de Vos et al., 2010). Consequently, herbage dry matter yield decreases (de Vos et al., 2010; Holshof et al., 2011; Hoving et al., 2008; Schothorst, 1982), as nitrogen can be considered the limiting nutrient for grass growth on peat soils (van der Laan et al., 2024). According to a management model for peat pastures developed by de Vos et al. (2006), raising ditch water levels from 60 to 40 cm below field level would result in a loss of 0.8 tons of dry matter grass production per hectare. Another study by Holshof et al. (2011) found that by raising the ditch water level to 30 cm below field level, grass productivity can be further impaired by a lowered load bearing capacity. This can result in grass losses of 16–20 % compared to lower water tables, due to trampling by animals and trafficking of machinery. This, in turn, affects grass utilization (de Vos et al., 2006). Additionally, the wetter conditions make grazing and mowing more difficult, which can result in sub-optimal timing of fertilization and longer intervals between cuttings, lowering not only the yield, but also the herbage quality (Holshof et al., 2011).

While previous research has focused on the effect of raising ditch water levels to 30 cm below field level on grass production and quality, it is not known whether raising groundwater tables to 20 cm below field level, which is desirable from a climate action and habitat restoration perspective (Tiemeyer et al., 2016), is still compatible with a viable dairy system. The viability of dairy farming is related to many different aspects. Grass production and quality, however, form the basis. Therefore, it is important to establish the effects of raising groundwater tables on these aspects. Furthermore, grass production and quality are also affected by management practices such as fertilization and grassland management, as well as by nutrient uptake form the soil. Previous research considered the effects of raising ditch water levels on nitrogen utilization in peat soils (Hoving et al., 2024). However, the feasibility and effects of raising groundwater tables using an active water infiltration system (AWIS) by making use of pressurized submerged drains (Hoving et al., 2023) remains unexplored. Pressurized submerged drains are drainage pipes that are placed approximately 70–80 cm below field level and connected to a water reservoir. The water table in this reservoir can be regulated using a pump, in order to manipulate the infiltration of water into the submerged drains. Depending on current groundwater tables and weather prediction, the water table in the reservoir can be increased or decreased in order to achieve the required groundwater table. Therefore, AWIS can be used to adjust groundwater tables to ensure higher and more stable levels throughout the year (Hoving et al., 2023). This is important, since a more stable groundwater table results in less emissions from the peat layer (Blodau and Moore, 2003).

Hence, the main aim of the present study was to assess the effect of raised groundwater tables using AWIS on the performance of peat pasture in terms of grass production and quality. In addition, we determined the effect of raising groundwater tables on nitrogen utilization, to better explain the underlying mechanisms that affect grass production and quality. These findings can provide valuable insights into the viability of dairy farming on peat soils with raised groundwater tables, and inform future interventions and policies on the sustainable management of peat soils.

2. Materials & methods

2.1. Area and experimental design

The study was carried out at the Knowledge Transfer Centre (KTC) Zegveld experimental farm (52°08′25.8"N 4°50′19.7"E), situated on peat soil (Terric Histosols, 37.5 % soil organic matter, 18 % sand, 25 % clay) in the western peat region (the Green Heart) of the Netherlands. This region is characterized by having relatively narrow fields (30–60 m), separated by ditches. The experiment ran for five years (2020–2024). The experimental set-up consisted of two self-contained on-farm systems under two water levels: a high-water table (HW) with a target groundwater table of 20 cm below field level, and a low-water table (LW), with a target groundwater table of 50 cm below field level. Grassland management consisted of fields that were only mown for silage production and fields that were both grazed and mown. Management on farm level was done similarly to the situation in practice (Agrimatie, 2024b), with an average annual application of 77 kg N ha^{-1} as artificial fertilizer and 173 kg N ha^{-1} as slurry, distributed in four applications over the growing season. In addition, manure from grazing cows contributed on average 41 kg N ha^{-1}.

2.1.1. Monitoring plots

In order to assess the effect of a raised groundwater table on grass production and quality, the high-water and low-water systems were equipped with monitoring plots of 10 × 10 m. In total, 15 monitoring plots were set up, with 9 plots located in the high-water system and 6 plots in the low-water system (Figure SA1). In 2023, a sixteenth plot was added to the low-water system to cover more of the variation between different fields.

Within each monitoring plot, three strips of 2.5 × 10 m were created, and these strips were fertilized with three different levels (0, 125 or 250 kg N ha^{-1} yr^{-1}) of artificial N fertilization (split over 4 applications), applied as calcium ammonium nitrate (CAN). The 250 N level corresponds to the total N application rate at system level (without manure from grazing cows), while the 125 N level was added to gain insight into the N response. P and K fertilizer were supplied in line with agricultural recommendations. At the start of each year, the plots were moved to a new location within the same field, to measure the effects of fertilizer application on the current year only and to take into account sward changes as a result of field management from the previous year. A summary of the experimental set-up can be found in Table SA1.

2.1.2. Groundwater table

The depth of the groundwater table (WT) was increased for the high-water systems (HW) and regulated along the study using an active water infiltration system (AWIS) (Hoving et al., 2023) with a target WT of 20 cm below field level. For the low-water systems (LW), the target was a ditch water level of 50 cm below field level, with no drains in place, which is in line with current practice in dairy farming. Additionally, in the HW systems, the ditch water levels were raised equal to the target WT of 20 cm below field level, to prevent circulation of water. Throughout each farming system, monitoring wells were installed to monitor WT. The monitoring plots were always within 20 m of a monitoring well. Groundwater tables were monitored to ensure whether the target WT was achieved.

2.2. Data collection

2.2.1. Groundwater table

WT was measured fortnightly in the 15 monitoring plots, by lowering a weighted measuring tape through the monitoring well positioned in the proximity of each plot. WT fluctuated between different monitoring points throughout the study period, as well as within water table treatments (see Figure SE1).

2.2.2. Herbage dry matter yield

The three strips in each plot were harvested using a Haldrup plot harvester (Løgstør, Denmark). The fresh weight of the harvested herbage was determined, after which a subsample was dried at 70 °C for 48 h to determine herbage dry matter yield.

Plots were harvested in five cuts in 2020 and 2022, and in four cuts in 2021, 2023 and 2024. The number of cuts was related to the grass growth. In 2021, the first cut was very late (start of June) as a result of a low load bearing capacity earlier in the season due to wet conditions in spring.

2.2.3. Herbage quality

Herbage N concentration (g kg^{-1} DM) was determined (Dumas, Eurofins Agro, NL) for all years, plots, cuts and fertilizer N application levels and was transformed to crude protein concentration (CP concentration, g kg^{-1} DM). Feeding value parameters including net energy for lactation (NEL, MJ kg^{-1} DM), digestible true protein (DVE, g kg^{-1} DM), and rumen degradable protein balance (OEB, g kg^{-1} DM) (Diervoeding, 2023) (NIRS, Eurofins Agro, NL) were determined for the intermediate fertilization level only. Feeding value parameters were only determined during 2022, 2023 and 2024.

2.2.4. Weather data and rainfall

Monthly average temperature and rainfall data were collected at the local rainfall station at KTC Zegveld and at the KNMI weather station in de Bilt (23 km distance) throughout the project (see Supplementary material B, Figures SB1–3). Compared to the long-term average, 2020 had a dry and warm spring (Annual average temperature of 11.7 °C, annual rainfall deficit of 115 mm), while 2021 had a relatively wet and colder spring (Annual average temperature of 10.4 °C, annual rainfall deficit of 24 mm). Spring in 2022 started out very dry in March, but rainfall increased in May, with temperatures close to the long-term average (Annual average temperature of 11.5 °C, annual rainfall deficit of 72 mm). Spring in 2023 had average temperatures and was considered very wet compared to the long-term average (Annual average temperature of 11.8 °C, annual rainfall deficit of 58 mm). Spring in 2024 had both high temperatures and high rainfall compared to the long term average (Annual average temperature of 11.8 °C, annual rainfall deficit of 21 mm) (KNMI, n.d.).

2.3. Data analysis

2.3.1. Groundwater table

The average WT per plot was calculated for each cut and for the total growing period in each year. In 2021, 2022, 2023 and 2024, there was no monitoring well close to plot 13 (LW) and therefore we excluded plot 13 from our analyses for 2021–2024. Missing WT readings for other plots (in total 119/360 readings in 2020, 18/378 readings in 2021, 19/518 readings in 2022, 12/465 readings in 2023 and 2/405 readings in 2024) were interpolated from drainpipes within the same system.

The measured WT throughout the growing season was significantly ($P < 0.001$) higher for HW compared to LW plots in all years (Fig. 1). On average, WT was −27 cm for HW and −45 cm for LW. In 2020, WT was significantly ($P < 0.001$) lower than in 2021–2023 (-42 cm and −59 cm for HW and LW in 2020 respectively, compared to an average of −24 cm for HW and −44 cm for LW across 2021–2023). In 2024, on the other hand, WT was significantly higher compared to 2021–2023 (-17 cm and −31 cm for HW and LW respectively). Overall, larger variations were found within the LW plots compared to the HW plots. Further information on WT fluctuations within the growing season can be found in Figure SE1.

2.3.2. N utilization parameters

In addition to herbage dry matter yield (DMY), several different parameters related to nitrogen utilization were calculated (Deru et al. 2019, van Eekeren et al. 2010), including N uptake (NU, kg N ha^{-1} yr^{-1}) [Eq. 1], soil N supply (SNS, kg N ha^{-1} yr^{-1}), apparent N recovery (ANR,

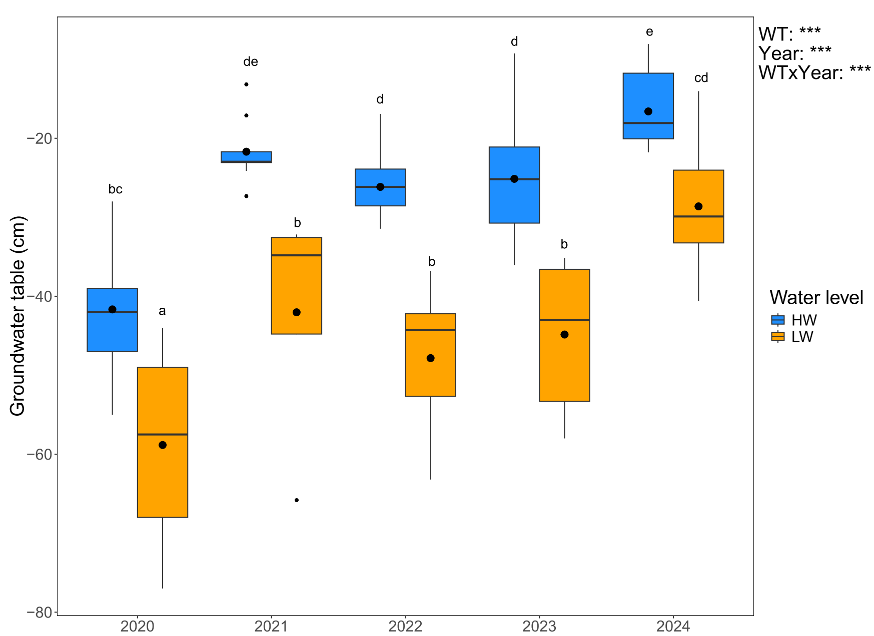

Fig. 1. Boxplot of the average measured groundwater tables (WT, in cm below field level) per plot during the growing season, for the high-water (HW, blue) and low-water (LW, orange) systems (HW, n = 9; LW, n = 5 in 2021 & 2022 and n = 6 in 2020, 2023 & 2024). Each plot value represents the average WT from measurements taken fortnightly during the growing season. The bold black dot indicates the mean, and the black horizontal line represents the median. The box spans the interquartile range (25th to 75th percentile), while the whiskers extend to the minimum and maximum values. Values with the same letters are not significantly different from each other. Significant differences are denoted by * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$.

kg N kg N applied^{-1}) [Eq. 2], and N use efficiency (NUE, kg DM kg N applied^{-1}) [Eq. 3].

$$NU = DMY * N\ concentration \tag{1}$$

Where:

NU = Nitrogen uptake by harvested herbage in kg N ha^{-1} yr^{-1}

DMY = Herbage dry matter yield in kg ha^{-1}

N concentration = Nitrogen concentration in the harvested herbage in g kg^{-1} dry matter

The soil nitrogen supply (SNS, kg N ha^{-1} yr^{-1}) was calculated as the Nitrogen uptake in the 0 kg N ha^{-1} yr^{-1} fertilizer application level.

For each plot, the apparent nitrogen recovery (ANR, kg N kg N applied^{-1}) for the 125 and 250 kg N ha^{-1} fertilizer application levels was calculated as:

$$ANR = (NU_{125N}\text{-}SNS) / 125;\ or\ (NU_{250N}\text{-}NU_{125N}) / 125 \tag{2}$$

Where:

ANR = Apparent nitrogen recovery in kg N kg N applied^{-1}

NU_{125N} = Nitrogen uptake in the 125 kg N ha^{-1} fertilizer application level

SNS = Soil nitrogen supply in kg N ha^{-1} yr^{-1}

NU_{250N} = Nitrogen uptake in the 250 kg N ha^{-1} fertilizer application level

The N use efficiency (NUE, kg DM kg N applied^{-1}) was calculated following Gastal et al. (2015) as:

$$NUE = (DMY_{125N}\text{-}DMY_{0N}) / 125;\ or\ (DMY_{250N}\text{-}DMY_{125N}) / 125 \tag{3}$$

Where:

NUE = Nitrogen use efficiency in kg DM kg N applied^{-1}

DMY_{125N} = Herbage dry matter yield at the 125 kg N ha^{-1} fertilizer application level

DMY_{0N} = Herbage dry matter yield at the 0 kg N ha^{-1} fertilizer application level

DMY_{250N} = Herbage dry mater yield at the 250 kg N ha^{-1} fertilizer application level

Lastly, average feeding values for the years were calculated based on the weighted average per cut.

2.3.3. Statistical Analysis

Initially, we set out to analyze the data with WT as a categorical factor with two levels (HW or LW). However, the measured WT showed more fluctuation from the target WT (-20 cm for HW) than expected (Fig. 1). Therefore, in addition to the analysis with categorical WT, we decided to run a regression analysis with the continuous measured average WT during the growing season. This would yield a quantitative effect of WT on plot level.

The dependent variables (i.e. DMY, SNS, ANR, NUE, feeding values) were analyzed with a Linear Mixed Model (LMM), with plot nested within field as random effect, using the lmerTest package (Kuznetsova et al., 2017) in R Studio (R Core Team, 2024). The models [Eq. 4] included all main effects and interactions (up to three-way interaction).

$$y_{ijk} = \beta_0 + \beta_1 * WT_{i\,+} + \beta_2 * Year_j + \beta_3 * N_k + \beta_4(WT_i * Year_j) + \beta_5(WT_i * N_k) + \beta_6(Year_j * N_k) + \beta_7(WT_i * Year_j * N_k) + \mu_{plot(field)} + \varepsilon_{ijk} \tag{4}$$

Where:

y_{ijk} = the dependent variable

β_0 = Overall intercept

WT_i = fixed effect of groundwater table as a factor (high (HW) or low (LW)) or as a continuous variable (in cm averaged per plot during the growing season)

$Year_j$ = fixed effect of year (factor, 2020–2024)

N_k = fixed effect of fertilization level as a factor (three levels, 0, 125 or 250 kg N ha^{-1} yr^{-1}) or continuous variable (kg N ha^{-1} yr^{-1})

$\beta_1, \beta_2, \beta_3, \beta_4, \beta_5, \beta_6, \beta_7$ = fixed effect coefficient

$\mu_{plot\ (field)}$ = random intercept for plot nested within field

ε_{ijk} = residual error term

If the linear mixed model gave a singular fit, a linear model was used and adjusted R^2 was reported, instead of conditional R^2. Since monitoring plots changed location within the same field each year, they were included as a nested random factor as repeated measurements were taken within the same fields throughout the experiment. Furthermore, repeated measures were taken within plots as well, as they were divided into three strips treated with three fertilization levels. If a factor was statistically significant as main effect and as an interaction, only the interaction was discussed. If main effects or interactions were not significant, they were removed from the model to obtain a better fit, and they were not discussed. N level was included as factor when relevant and was excluded when the N level was irrelevant to the variable, such as SNS. Data was tested for normality and heterogeneity using residual plots. Tukey's post-hoc test was performed to identify which treatments differed significantly, by using the emmeans package (Lenth, 2024). Figures for the regression analyses were made using the ggplot2 package (Wickham, 2016). Figures show the fitted regression lines for one N level only to improve clarity, but all regression lines were derived from the full model (Eq. 4). Results from the regression analyses with continuous WT are the main focus of this study, while the results from the analysis with categorical WT are shown in Supplementary material C.

3. Results

3.1. Herbage dry matter yield

Total DMY per year was significantly affected by the main effects of WT, year, and fertilization ($P < 0.001$) (Fig. 2). Overall, DMY reduced with 36 kg DM ha^{-1} yr^{-1} per cm increase in WT. As a result, at a WT of −20 cm (and fertilization of 125 kg N ha^{-1} yr^{-1}), predicted DMY ranged from 10.5 ton DM ha^{-1} yr^{-1} in 2020 to 12.4 ton DM ha^{-1} yr^{-1} in 2021. With a WT of −50 cm (and fertilization of 125 kg N ha^{-1} yr^{-1}), predicted DMY ranged from 11.6 ton DM ha^{-1} yr^{-1} in 2020 to 13.5 ton DM ha^{-1} yr^{-1} in 2021. For the analysis on system level (HW vs. LW), see Figure SC1.

Fertilization resulted in an increase in DMY of on average 10 kg DM ha^{-1} yr^{-1} for each kg of fertilizer N applied, irrespective of year or WT.

DMY per cut was significantly ($P < 0.001$) affected by WT later in the growing season, specifically in the third and fourth cut (but not in the first and second cut). In the third cut, each cm increase in WT resulted in a decrease of 12 kg DM ha^{-1}. In the fourth cut, each cm increase in WT resulted in a decrease of 20 kg DM ha^{-1}. For further details, see Supplementary material E.

The predicted DMY reduction for the target HW WT (-20 cm) compared to the target LW WT (-50 cm) was 1.0 ton DM ha^{-1}, a reduction of 9 % (Table 1). In practice, the target WT for HW and LW was not always reached (Fig. 1). We applied the regression model to predict DMY based on the observed WT for both systems, and for the LW system only, assuming that in an optimized AWIS system, target WT can be achieved (Table 1). This showed that the average predicted DMY reduction (at 125 kg N ha^{-1} N fertilization rate) at the observed WT levels was 0.7 ton ha^{-1} (-6 %), a substantially smaller reduction compared to target WT. The predicted DMY reduction was on average 1.0 ton ha^{-1} (-8 %) when target WT was achieved for the HW system. In both of the latter scenarios, there was substantial variation in DMY reduction between years.

3.2. Grass quality

3.2.1. Crude protein

CP concentration was significantly affected by the main effect of WT ($P < 0.05$) and an interaction between fertilization and year ($P < 0.001$) The reduction of CP concentration was 0.25 g kg^{-1} DM per cm increase in WT (Fig. 3). On average, at a WT of −20 cm and a fertilization of 125 kg N ha^{-1} yr^{-1}, predicted CP ranged from 184 g kg^{-1} DM in 2020 to

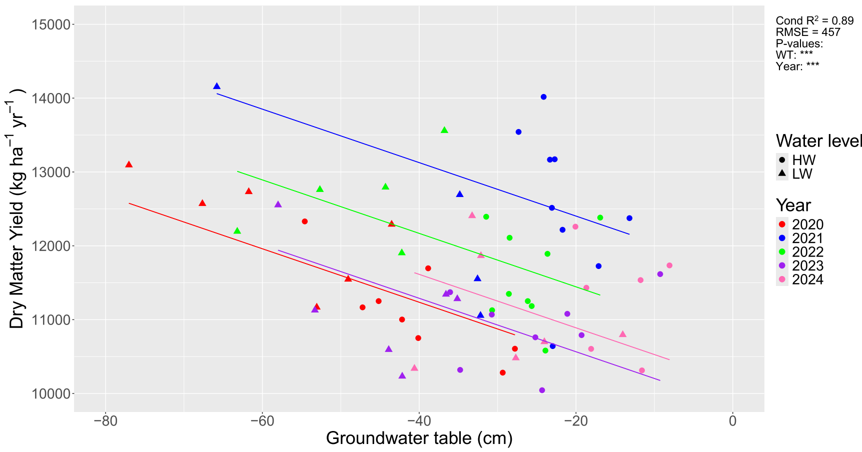

Fig. 2. The average dry matter yield in kg ha^{-1} yr^{-1} across the average measured groundwater tables (WT) in cm below field level during the growing season, in 2020–2024 for the medium level of fertilization (125 kg N ha^{-1} yr^{-1}). Symbols indicate plots within different systems (HW = high-water, LW = low-water). Significant effects are denoted by * = P < 0.05, ** = P < 0.01, *** = P < 0.001.

Table 1
Overview of the observed average groundwater table (WT) during the growing season and the predicted DMY (ton DM ha^{-1}) at 125 kg N ha^{-1} N fertilizer application rate for the LW and HW system based on the target groundwater tables (WT= −50 and −20 cm for LW and HW) and observed water tables (for LW only or for both LW and HW).

	Av. Observed water table (cm)			DMY WT: target			DMY WT: LW observed, HW target			DMY WT observed		
Year	LW	HW	%*	LW	HW	%	LW	HW	%	LW	HW	%
2020	-59	-42	-29 %	11.6	10.5	-9 %	11.9	10.5	-12 %	11.9	11.3	-5 %
2021	-42	-26	-38 %	13.5	12.4	-8 %	13.2	12.4	-6 %	13.2	12.6	-5 %
2022	-48	-26	-46 %	12.5	11.4	-9 %	12.5	11.4	-9 %	12.5	11.7	-6 %
2023	-45	-25	-44 %	11.6	10.6	-9 %	11.4	10.6	-7 %	11.4	10.7	-6 %
2024	-31	-17	-45 %	12.0	10.9	-9 %	11.3	10.9	-4 %	11.3	10.8	-4 %
Average	**-45**	**-27**	**-40 %**	**12.2**	**11.2**	**-9 %**	**12.2**	**11.2**	**-8 %**	**12.2**	**11.5**	**-6 %**

* Relative difference between HW and LW: (HW-LW)/LW x 100 %

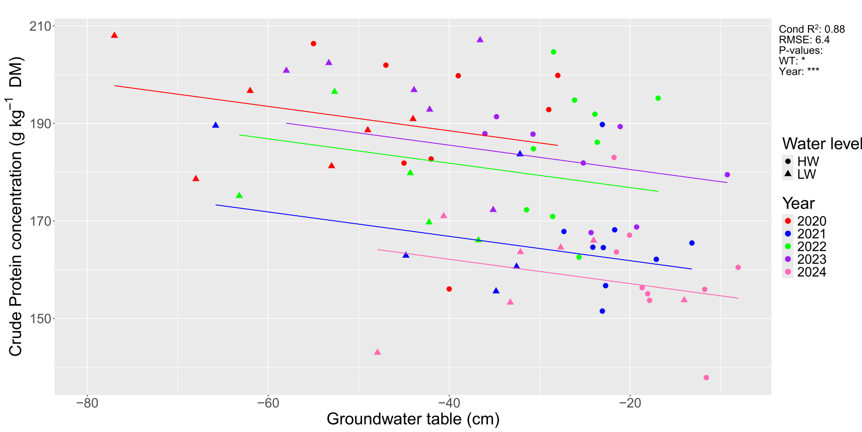

Fig. 3. The average crude protein concentration in g kg^{-1} DM across the average measured groundwater tables (WT) in cm below field level during the growing season in 2020–2024, for the medium level of fertilization (125 kg N ha^{-1} yr^{-1}). Symbols indicate plots within different systems (HW = high-water, LW = low-water). Significant differences are denoted by * = P < 0.05, ** = P < 0.01, *** = P < 0.001.

158 g kg^{-1} DM in 2024. At a WT of −50 cm, predicted CP ranged from 192 g kg^{-1} DM in 2020 to 165 g kg^{-1} DM in 2024.

3.2.2. *Other feeding values*

For the feeding values reported in this section, results were only available for 2022–2024.

Net energy for lactation (NEL) was not affected by WT. NEL was significantly affected by the main effect of year (P < 0.001), and was on average 6.2, 5.7 and 5.8 MJ kg^{-1} DM in 2022, 2023 and 2024 respectively.

True protein digested in the small intestine (DVE) was significantly affected by year (P < 0.001), but not by WT. DVE was on average 73.7, 65.8, and 65.6 g kg^{-1} DM in 2022, 2023 and 2024 respectively.

Rumen degradable protein balance (OEB) was significantly affected by an interaction between WT and year (P < 0.05). In 2022 and 2024, there was no effect of WT on OEB (on average 30 and 5 g kg^{-1} DM in 2022 and 2024 respectively), whereas in 2023, OEB was significantly affected by WT with a decrease of 0.4 g kg^{-1} DM per cm increase in WT.

3.3. Nitrogen utilization

3.3.1. Soil Nitrogen Supply

SNS was significantly affected by the main effects of WT and year (both P < 0.001). The reduction in SNS with each cm increase in WT was 1.2 kg N ha^{-1} yr^{-1} across years (Fig. 4). In 2022, SNS was significantly higher compared to the other years. As a result, at a WT of −20 cm, predicted SNS ranged from 220 kg N ha^{-1} yr^{-1} in 2024 to 279 kg N ha^{-1} yr^{-1} in 2022. At a WT of −50 cm, predicted SNS ranged from 256 kg N ha^{-1} yr^{-1} in 2024 to 315 kg N ha^{-1} yr^{-1} in 2022.

Similar to DMY, we found a contrast between the predicted reduction in SNS based on target or observed WT (or a combination, Supplementary Material F, Table SF1). Predicted SNS was on average −13 % (target WT), −12 % (observed WT for LW system) and −8 % (observed WT for both LW and HW systems).

3.3.2. Apparent Nitrogen Recovery

ANR was significantly affected by year and fertilization range (both P < 0.01) but not by WT. ANR was on average 0.58 kg N kg^{-1} N applied under the first fertilization range (0–125 kg N ha^{-1} yr^{-1}) and 0.44 kg N kg N $applied^{-1}$ under the second fertilization range (125–250 kg N ha^{-1} yr^{-1}) (average of 0.51 kg N kg N $applied^{-1}$ across both fertilization ranges). ANR varied significantly between years, with the highest average value in 2020 (0.67 kg N kg N $applied^{-1}$), followed by 2021 (0.58 kg N kg N $applied^{-1}$), 2023 (0.49 kg N kg N $applied^{-1}$), 2024 (0.44 kg N kg N $applied^{-1}$) and 2022 (0.40 kg N kg N $applied^{-1}$).

3.3.3. Nitrogen Use Efficiency

NUE was significantly affected by an interaction between fertilization range and year (P < 0.001) but not by WT. In 2020, NUE was significantly higher in the 0–125 kg N range (15.4 kg DM kg N $applied^{-1}$) compared to the 125–250 kg N range (4.8 kg DM kg N $applied^{-1}$). In the other years there was no significant difference in NUE between the two fertilization ranges (11.7, 10.3 and 7.0 and 11.4 kg DM kg N $applied^{-1}$ for 2021–2024 respectively), but overall, NUE tended to be higher in the 0–125 kg N range.

3.3.4. Nitrogen uptake

N uptake was significantly affected by the main effects of water table, year and fertilization (all P < 0.001). The reduction in N uptake was 1.4 kg N ha^{-1} yr^{-1} per cm increase in WT across years (Fig. 5). N uptake increased with 0.51 kg N ha^{-1} yr^{-1} per kg N applied, which coincides with the average value found for ANR. N uptake was significantly lower in 2024 compared to the other years. As a result, at a WT of −20 cm and a fertilization level of 125 N ha^{-1} yr^{-1}, predicted N uptake was 317 (2020, 2021, 2022, 2023), and 274 (2024) kg N ha^{-1} yr^{-1}. At a WT of −50 cm and a fertilization level of 125 kg N ha^{-1} yr^{-1}, predicted N uptake was 359 (2020, 2021, 2022, 2023), and 316 (2024) kg N ha^{-1} yr^{-1}.

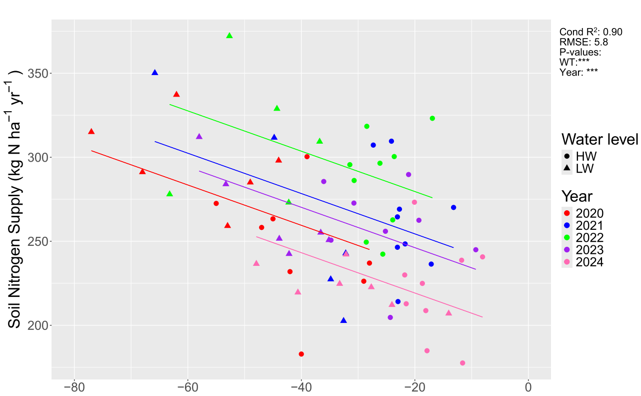

Fig. 4. The soil nitrogen supply in kg N ha^{-1} yr^{-1} across the average measured groundwater tables (WT) in cm below field level during the growing season, in 2020–2024. Symbols indicate plots within different systems (HW = high-water, LW = low-water). Significant effects are denoted by * = P < 0.05, ** = P < 0.01, *** = P < 0.001.

4. Discussion

4.1. Herbage dry matter yield

Raising the groundwater table resulted over the five years on average in a decrease of 36 kg DM ha^{-1} per cm increase in WT. Based on our results, a WT of 20 cm below field level, which is assumed to minimize GHG emissions (Tiemeyer et al., 2016), would result in a DMY of 11 ton ha^{-1} at a fertilizer N application rate of 125 kg N ha^{-1}, or a reduction of 1.0 ton ha^{-1} (9 %) compared to a WT of 50 cm below field level. Earlier research in an extensive system of Renger et al. (2002) concluded that an increase in WT would lead to a decrease in grass production, and that moving from a WT of 50 cm below field level (5 ton DM ha^{-1}) to a WT of 30 cm below field level (4.5 ton DM ha^{-1}) would result in a 10 % decrease in DMY (25 kg DM ha^{-1} per cm). Furthermore, an explorative study by Hoving et al. (2020) found that raising WT from 50 to 30 cm below field level with AWIS would lead to a loss of 1 ton DM ha^{-1} (10 %, 50 kg DM ha^{-1} per cm). This reduction is somewhat higher than our findings. However, it is important to note that our findings do not include potential losses from trafficking and trampling by machinery or cows, which may increase at higher WT (de Vos et al., 2006). Further research is needed to better understand the effects of these potential losses in combination with raising groundwater tables.

Our analysis showed that the predicted reduction in DMY for the HW plots (Table 1) using observed WT, was on average −6 %, which was smaller than the predicted DMY reduction of −9 % based on the target WT (-50 and −20 cm for LW and HW, Table 1). The achieved difference in WT was generally smaller than the target of 30 cm and was strongly affected by weather conditions (rainfall deficit) during the growing season. The predicted yield reduction at the observed WT ranged from −5 % in a relatively wet year (2021, rainfall deficit of 24 mm) when the observed difference in WT was relatively small, compared to −6 % in relatively dry years (2022, rainfall deficit of 72 mm). Holshof et al. (2011) also reported differences between individual years, with reductions ranging from 1 % to 14 % in 2005–2010 for raising ditch water levels from 50 cm to 30 cm below field level with fertilization. Overall, we found the highest DM yields in 2021, which had a relatively cold and wet spring (cumulative precipitation of 123 mm in May compared to an average of 66 mm, Figure SB2) (KNMI, 2022) and the highest measured WT. Due to these wet conditions in spring, however, the first cut was harvested two weeks later than normal, resulting in a heavy first cut (explaining the high overall production), and only four cuts in total were harvested. This resulted in a lower herbage quality as well, which will be discussed below.

Schothorst (1982) found that yield losses due to high ditch water levels could be compensated with fertilization. In our study, we found that on average DMY increased 10 kg per kg N applied. The average loss of DMY across years when moving from a WT of 50 cm to 20 cm below field level was 1.0 tons ha^{-1}. To fully compensate for that loss, an additional 100 kg N ha^{-1} would have to be applied. However, with a higher N application, both ANR and NUE declined (NUE only significantly in 2020) and both were affected by yearly weather conditions. Therefore, a higher N application will also result in a larger proportion of N not taken up by the grass, leading to increased nutrient losses to the environment through leaching and run-off, which can lead to eutrophication (van der Laan et al., 2024). In addition, the production and application of extra (artificial) N fertilizer contributes to climate change (Brentrup et al., 2004; Zak et al., 2017). Thus, if additional fertilizer N is used to counteract the yield reduction as a result of a higher WT, these negative effects could diminish the potential climate gains from raising WT (van der Laan et al., 2024). Further research should explore the

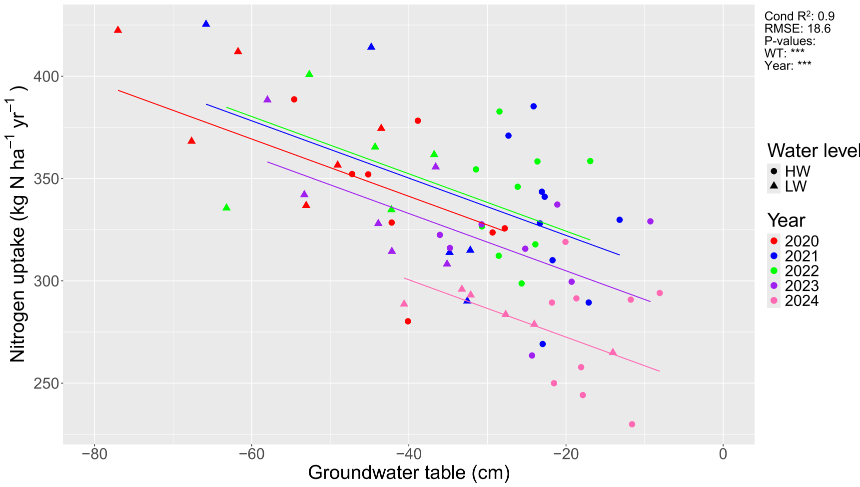

Fig. 5. The nitrogen uptake in kg N ha^{-1} yr^{-1} across the average measured groundwater tables (WT) in cm below field level during the growing season in 2020–2024, for the medium level of fertilization (125 kg N ha^{-1} yr^{-1}). Symbols indicate plots within different systems (HW = high-water, LW = low-water). Significant differences are denoted by * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$.

effects of increased fertilization with a higher WT at farm level, where the influence of grassland and animal management on eutrophication and GHG emissions are explored (Ayala et al., 2024).

4.2. Grass quality

We found an effect of WT on CP concentration, resulting in an average decrease of 0.25 g CP kg^{-1} DM per cm increase in WT. This is in line with the findings of Holshof et al. (2011), who report a CP concentration of 157 g kg^{-1} DM at a ditch water level of 30 cm below field level. Furthermore, in a mesocosm experiment by van der Laan et al. (2024) a decrease in N concentration with higher WT was observed as well. The lowest average values for CP concentration were found in 2021 (146 g kg^{-1} DM at a WT of 20 cm below field level, no fertilization), which was the year with the highest DMY. DMY was harvested in four cuts, which resulted in high yields, but lowered the herbage quality due to the age of the grass (Holshof et al., 2011).

In contrast, both NEL and DVE were only affected by year and not by groundwater table, with higher values achieved in 2022 than in 2023 and 2024. 2023 and 2024 were wetter years compared to 2022, which could have resulted in a lower DVE and NEL. OEB, on the other hand, was negatively affected by a raised groundwater table, but only in 2023, indicating that the combination of higher water tables in a wetter year might further affect feed quality. However, data on NEL, OEB and DVE was only collected in 2022, 2023 and 2024 and only at the 125 kg N ha^{-1} fertilization level, so the analysis may not account for variation across years and effects at higher N fertilization rates which are common practice in the study area, and which were significant factors for the other studied parameters.

Nonetheless, our findings show that with raised groundwater tables, the quality of the harvested herbage is only affected by a decrease in CP concentration and OEB, but average values found are not limiting for milk production and are in line with earlier research (Holshof et al., 2011). On a system level however, herbage quality might be impaired by other factors due to the wetter conditions, such as changes in botanical composition over time. In addition, sub-optimal timing of fertilization and harvesting due to restrictions as a result of a lower load bearing capacity at high groundwater tables (Janssen et al., 2023) may affect herbage quality (Holshof et al., 2011).

4.3. Nitrogen utilization

Previous studies found that reductions in DMY could be explained by decreased mineralization rates under rewetted conditions (van Dijk et al., 2009; van Kekem et al., 2004). In our study, we found that SNS was affected by WT and showed a decrease of 1.2 kg N ha^{-1} per cm increase in WT. However, the variability between years was again significant, ranging from a SNS of 280 kg N ha^{-1} (average value in 2022) to 220 kg N ha^{-1} (average value in 2024). Across years, average SNS at 20 cm below field level (247 kg N ha^{-1}) was comparable to other studies, falling within the range described by van Kekem et al. (2004), of 176 kg N ha^{-1} for wet soils to 302 kg N ha^{-1} for drained soils. Further studies found a mean of 264 kg N ha^{-1} on drained soils (Deru et al., 2019), and 252 kg N ha^{-1} for poorly drained peat soils (Vellinga and André, 1999). In addition to mineralization of the peat layer, historic additions of ditch sludge, animal slurry, and farmyard manure can contribute up to 60 % to the annual N mineralization in peat soils, leading to a high SNS, even in wetter conditions (Sonneveld and Lantinga, 2011). Furthermore, similar to findings on DMY, we found a contrast in predicted SNS with target and observed groundwater tables (Table SF1). The predicted reductions in SNS with observed groundwater tables were stronger in relatively dry years, when the observed difference in WT was larger, compared to relatively wet years.

N uptake was affected by WT and predicted to decrease with 1.4 kg N ha^{-1} per cm increase in WT. Differences in N uptake between years were primarily due to the different weather conditions. Deru et al. (2019) found an average value for N uptake of 394 kg N ha^{-1} under fertilized conditions, with a ditch water level of 49 cm below field level. To compare, in our study, with a WT of 20 cm below field level, plots had an average SNS of 247 kg N ha^{-1} but reached a N uptake of 309 and 372 kg N ha^{-1} with a fertilization rate of 125 and 250 kg N ha^{-1} yr^{-1} respectively. This indicates that, similar to DMY, the effect of increased WT on N uptake can possibly be mediated with fertilization. This is supported by Hoving et al. (2024), who found that N fertilization can compensate for a lower SNS under higher WT via a higher N uptake. However, with a lower N uptake with higher WT, an increase in N losses to the environment needs to be considered as well. Further insights are needed to determine how nitrogen is lost to the environment when WT are raised, as higher WT with increased fertilization could result in increased nutrient leaching to surface waters (van der Laan et al., 2024).

For both ANR and NUE, we expected to find an effect of groundwater table, since wetter conditions are known to affect N losses through increased denitrification and run-off (Pijlman et al., 2020; Verloop et al., 2014), potentially leading to a lower N utilization. Since we did not find an effect of WT on ANR and NUE, our results imply that N fertilization is equally efficient at higher groundwater tables, and is mainly affected by yearly weather conditions and management. The average value found for ANR (0.51 kg N kg N $applied^{-1}$) falls within the range (0.41–1.0, average 0.65 kg N kg N^{-1}) described by Deru et al. (2019) for a ditch water level ranging from 30 to 60 cm below field level. In addition, Pijlman et al. (2020) found an average ANR of 0.58 kg N kg N applied^{-1}, based on multiple studies carried out between 1992 and 2017. Yearly weather conditions did significantly affect ANR and NUE. For example, in 2024, NUE was significantly lower, which could be explained by the high temperatures (Figure SB1) and high cumulative rainfall of 623 mm during March-October (Figures SB2 and SB3)(KNMI, 2024). Increased precipitation is known to decrease mineralization rates and increase denitrification and run-off losses, (Deru et al., 2019; Zhang et al., 2020) which could have affected N availability, and ultimately lead to a decrease in NUE.

4.4. Considerations and practical implications

Previous research assessed the effects of raising ditch water levels on peat pasture performance (de Vos et al., 2006, 2010; Holshof et al., 2011). Raising ditch water levels can reduce degradation of the peat layer and subsequent soil subsidence and greenhouse gas emissions, but this method is ineffective in maintaining a stable high groundwater table, especially in summer (Hoving et al., 2008). In the current study, WT was raised using an active water infiltration system (AWIS) (Hoving et al., 2008, 2020), which uses dynamic water management by draining in wet periods and infiltrating in dry periods, resulting in a more stable WT throughout the year (Hoving et al., 2024, 2023). Yet, maintaining stable WT throughout the study period proved challenging, as indicated by the variation of WT within the HW and LW plots (Fig. 1, Table 1). Despite this challenge, the difference between HW and LW was still achieved, and the variability in WT allowed for the regression analysis with WT as a continuous variable. The regression analysis allowed for an enhanced understanding of the quantitative effect of WT per increase in cm on the studied parameters. The variation in WT, however, was different among years due to varying weather conditions, resulting in larger differences in observed WT in drier years. With continued improvement of the AWIS technology and management, WT the HW system will likely become increasingly stable at 20 cm below field level. This would further increase the variation in yield reduction between years, as illustrated by the contrasts in predicted DMY with observed WT for LW and target WT for HW (-4 % in the wet 2024 to −12 % in the dry 2020, Table 1). Furthermore, the analysis per individual cut shows that the effect of raised WT became apparent later in the growing season, in the third and fourth cut, indicated by larger differences in WT between HW and LW during summer (see Supplementary Material E, Figure SE2). This variability between years and within the growing season should be accounted for, not only for future experimental designs, but also for translating results to commercial conditions on dairy farms.

Since our results were obtained from experimental plots, our findings do not fully reflect the effect of increased WT under combined grazing and mowing conditions. Our results on DMY could be regarded as ‘potential’ yield only, since there has been no influence of trampling by cows or losses due to machinery trafficking. Harvesting in high-water fields with conventional commercial machinery can be more challenging due to the lower load bearing capacity of the soil, which can result in increased losses and sub-optimal timing of fertilization and harvest (Hoekstra et al., 2019; Janssen et al., 2023). Therefore, a comparison with further research using data of the actual harvested herbage under commercial production conditions would be valuable to determine occurring losses.

Nonetheless, our study has shown that raising groundwater tables affects grass production and quality. The average loss of 9 % for DMY is substantial, yet the average predicted DMY across the five years at a fertilizer N application rate of 125 kg N ha^{-1} remained above 11 tons per hectare, which can still be considered as highly productive (Smit et al., 2008), even when compared to other regions in the Netherlands (average Dutch DMY ranges between 9 and 11 tons per hectare (Agrimatie, 2024a)). This level of productivity would allow for continued dairy production. Yet, losses also occur during grazing (on average 15–20 %), and harvesting (on average 5 %) (Annual Nutrient Cycling Assessment (van Dijk et al., 2020)). These losses could further increase in wetter conditions, resulting in a significant impact on farm economy. Farmers would either have to import more feed, or the productivity of the herd decreases, or less cows can be kept if farmers want to remain equally self-sufficient in their feed supply from grass production. Yet, if fertilization is increased to compensate for yield losses, then additional costs and environmental impacts will need to be considered. Alternatively, farm productivity could be lowered to accommodate environmental goals, but this possibly leads to a trade-off with farm economy if no alternative business models are adopted. Thus, in addition to grass production and quality, many more aspects of farm performance should be considered when discussing the effect of raising WT, including the effects on grass intake and subsequent milk production. Farms may follow different strategies to adapt to raised WT, hence the consequences of raising WT on farm viability and environmental targets may need to be explored at farm level, while keeping different management strategies in mind. In the meantime, the findings from this study can potentially inform economic compensation schemes if WT is raised to favor climate and biodiversity targets.

Raising WT is regarded as necessary to achieve the climate goals to reduce GHG emissions from peat soils by one million tons in 2030, and to reduce soil subsidence by 50 % (Rijksoverheid, 2019), as defined in the Climate Agreement of the Dutch government. Peat pasture areas in the Netherlands are characterized by the polder landscape with ditches, designed for draining groundwater (Dosker, 2017). Therefore, it is challenging for individual farmers to raise WT on their fields, even more so when surrounding fields are still drained (Tanneberger et al., 2021). Hence, to successfully reach the targets set by the Dutch government, WT may need to be stabilized at a scale larger than farm level. In the Netherlands, the Water Boards are largely responsible for managing and regulating the national WT. The large-scale implementation of raising WT (through ditch water levels) is therefore largely dependent on policy, not on individual farmers alone. Still, based on this study, farmers would be substantially affected by raising WT. Currently, there is no real financial incentive for farmers to raise WT on their fields (Convention on Wetlands, 2021) or counter the effects of raising WT at regional level. Moreover, costs related to the installation and maintenance of the AWIS should be accounted for as such costs are also substantial (Hoving et al., 2008). In the meantime, raising WT also offers the opportunity for designing new business models where dairy farming is not only valued for the production of milk, but also for their contribution to climate and biodiversity targets, and for improving water quality and regulation (Tanneberger et al., 2021). Alternatively, CAP subsidies could be repurposed to support raising WT, as currently drainage of peatlands is still financially supported (Wichmann and Nordt, 2024). In the end, while raising WT is essential for achieving climate and biodiversity targets, it will affect dairy farmers. To ensure a successful transition towards dairy farming with raised WT, creating enabling conditions such as compensation schemes and new business models are crucial. The findings of this study can serve as a base for designing such compensation schemes, ultimately supporting farmers in making this transition a viable option.

5. Conclusion

This study aimed to assess the effect of raised groundwater tables on grass production, quality and N utilization. Raised groundwater tables affected herbage dry matter production, soil nitrogen supply, N uptake, crude protein concentration and rumen degradable protein balance. For DMY, we found an average reduction of 36 kg ha^{-1} per cm increase in groundwater table across the study years, which resulted in a 9 % loss when raising the groundwater table from 50 to 20 cm below field level. However, the effect of groundwater table on DMY was influenced significantly by yearly weather conditions. The observed difference in groundwater table between the high-water and low-water systems was generally smaller than 30 cm, and as a result, the predicted DMY reduction based on observed groundwater tables was on average 6 %. This highlights the importance of environmental variability in assessing the impact of raising groundwater tables. Even so, average yields remained above 11 ton ha^{-1} at a fertilizer N application rate of 125 kg N ha^{-1}. The decrease in DMY under raised groundwater tables can be partly explained by the observed decrease in soil nitrogen supply. No effect of groundwater table was found for ANR and NUE, indicating that N fertilization is equally efficient with raised groundwater tables. Regarding grass quality, CP concentration and rumen degradable protein balance were negatively affected by groundwater table, while NEL and DVE only showed yearly variation. The above findings are based on fenced-off experimental plots. To better understand the system-level impact of raising groundwater tables, further research should account for losses that occur during, and from sub-optimal timing of harvesting, including physical damage by machinery and grazing animals, as well as other management practices, to provide a more realistic view of the practical implications of raising groundwater tables. Raising groundwater tables is a crucial intervention for achieving the goals set by the Dutch Climate Agreement and can contribute to improving environmental sustainability of farms on peat soils. The findings from this study could be used to inform compensation schemes for farmers to encourage the transition towards dairy farming with raised groundwater tables.

CRediT authorship contribution statement

Idse Hoving: Writing – review & editing, Investigation. **Raimon Ripoll-Bosch:** Writing – review & editing, Supervision, Conceptualization. **Jeroen Pijlman:** Writing – review & editing, Investigation. **Nick van Eekeren:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization. **Nyncke J. Hoekstra:** Writing – review & editing, Supervision, Investigation, Conceptualization. **Hilde M. van Dijk:** Writing – original draft, Visualization, Formal analysis, Conceptualization.

Declaration of Generative AI and AI-assisted technologies in the writing process

During the preparation of this work the author(s) used OpenAI - ChatGTP in order to improve readability of the text. After using this tool/service, the author(s) reviewed and edited the content as needed and take(s) full responsibility for the content of the publication.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.eja.2025.127961.

Data availability

Data will be made available on request.

References

Agrimatie. (2024a). Graslandopbrengsten per ha in 2022 laag. Retrieved 10-6-2025 from ⟨https://agrimatie.nl/PublicatiePage.aspx?subpubID=2523&sectorID=2245&themaID=2754&indicatorID=3551⟩.
Agrimatie. (2024b). Stikstofgebruik bestaat voor twee derde uit dierlijke mest. Retrieved 10-6-2025 from ⟨https://agrimatie.nl/PublicatiePage.aspx?subpubID=2523&sectorID=2245&themaID=2758&indicatorID=2772⟩.
Ayala, M.C., Groot, J.C.J., Kilcline, K., Grace, C., Kennedy, J., Moran, B., de Boer, I.J.M., Ripoll-Bosch, R., 2024. Characterizing beef and sheep farming systems to customize sustainability interventions and policy implementation. J. Environ. Manag. 366, 121900. https://doi.org/10.1016/j.jenvman.2024.121900.
Blodau, C., Moore, T.R., 2003. Experimental response of peatland carbon dynamics to a water table fluctuation. Aquat. Sci. 65 (1), 47–62. https://doi.org/10.1007/s000270300004.
Brentrup, F., Küsters, J., Lammel, J., Barraclough, P., Kuhlmann, H., 2004. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. Eur. J. Agron. 20 (3), 265–279. https://doi.org/10.1016/S1161-0301(03)00039-X.
Brouns, K., Eikelboom, T., Jansen, P.C., Janssen, R., Kwakernaak, C., van den Akker, J.J.H., Verhoeven, J.T.A., 2015. Spatial Analysis of Soil Subsidence in Peat Meadow Areas in Friesland in Relation to Land and Water Management, Climate Change, and Adaptation. Environ. Manag. 55 (2), 360–372. https://doi.org/10.1007/s00267-014-0392-x.
Convention on Wetlands. (2021). Global guidelines for peatland rewetting and restoration. Ramsar Technical Report no. 11. ⟨https://www.ramsar.org/sites/default/files/documents/library/rtr11_peatland_rewetting_restoration_e.pdf⟩.
van den Born, G.J., Kragt, F., Henkens, D., Rijken, B., van Bemmel, B., van der Sluis, S., 2016. Dalende bodems, stijgende kosten: mogelijke maatregelen tegen veenbodemdaling in het landelijk en stedelijk gebied. Planbur. voor De. Leefomgeving. ⟨https://www.pbl.nl/publicaties/dalende-bodems-stijgende-kosten⟩.
van der Laan, A., van Dijk, J., Rebel, K.T., Wassen, M.J., 2024. Rewet without regret? Nutrient dynamics in fen peat exposed to different rewetting degrees. Biogeochemistry. https://doi.org/10.1007/s10533-024-01139-x.
Deru, J.G.C., Bloem, J., de Goede, R., Hoekstra, N., Keidel, H., Kloen, H., Nierop, A., Rutgers, M., Schouten, T., van den Akker, J., Brussaard, L., van Eekeren, N., 2019. Predicting soil N supply and yield parameters in peat grasslands. Appl. Soil Ecol. 134, 77–84. https://doi.org/10.1016/j.apsoil.2018.10.018.
Diervoeding, C. (2023). CVB Veevoedertabel 2023. ⟨https://www.cvbdiervoeding.nl/pagina/10081/downloads.aspx⟩.
van Dijk, W., de Boer, J., de Haan, M.H.A., Mostert, P., Oenema, J., Verloop, J., 2020. Rekenregels van de KringloopWijzer 2020; Achtergronden van BEX, BEA, BEN, BEP en BEC: actualisatie van de 2019-versie. Wagening. Plant Res. https://doi.org/10.18174/533882.
van Dijk, J., Didden, W.A.M., Kuenen, F., van Bodegom, P.M., Verhoef, H.A., Aerts, R., 2009. Can differences in soil community composition after peat meadow restoration lead to different decomposition and mineralization rates? Soil Biol. Biochem. 41 (8), 1717–1725. https://doi.org/10.1016/j.soilbio.2009.05.016.
Dosker, M.C.M.V. (2017). Veenbehoud en cultuurhistorie. Welke mogelijkheiden zijn er voor duurzaam beheer van veenweiden als waardevol cultuurlandschap? ⟨https://www.cultureelerfgoed.nl/documenten/2017/01/01/veenbehoud-en-cultuurhistorie⟩.
van Eekeren, N., de Boer, H., Hanegraaf, M., Bokhorst, J., Nierop, D., Bloem, J., Schouten, T., de Goede, R., Brussaard, L., 2010. Ecosystem services in grassland associated with biotic and abiotic soil parameters. Soil Biol. Biochem. 42 (9), 1491–1504. https://doi.org/10.1016/j.soilbio.2010.05.016.
European Commission (n.d.). Water Framework Directive. Retrieved 13-8-2024 from ⟨https://environment.ec.europa.eu/topics/water/water-framework-directive_en⟩.
European Commission (2021). Protecting our precious peat. Retrieved 13-8-2024 from ⟨https://cinea.ec.europa.eu/news-events/news/protecting-our-precious-peat-2021-05-12_en⟩.
Evans, C.D., Peacock, M., Baird, A.J., Artz, R.R.E., Burden, A., Callaghan, N., Chapman, P.J., Cooper, H.M., Coyle, M., Craig, E., Cumming, A., Dixon, S., Gauci, V., Grayson, R.P., Helfter, C., Heppell, C.M., Holden, J., Jones, D.L., Kaduk, J., Morrison, R., 2021. Overriding water table control on managed peatland greenhouse gas emissions. Nature 593 (7860), 548–552. https://doi.org/10.1038/s41586-021-03523-1.
Farrell, C.A., Coleman, L., Norton, D., Kelly-Quinn, M., Obst, C., Eigenraam, M., O'Donoghue, C., Kinsella, S., Smith, F., Sheehy, I., Stout, J.C., 2021. Developing peatland ecosystem accounts to guide targets for restoration [10.3897/oneeco.6.e76838]. One Ecosyst. 6, e76838. https://doi.org/10.3897/oneeco.6.e76838.
Gastal, F., Lemaire, G., Durand, J.-L., Louarn, G., 2015. Chapter 8 - Quantifying crop responses to nitrogen and avenues to improve nitrogen-use efficiency. In: Sadras, V.

O., Calderini, D.F. (Eds.), Crop Physiology (Second Edition). Academic Press, pp. 161–206. https://doi.org/10.1016/B978-0-12-417104-6.00008-X.
Hoekstra, N., Holshof, G., Schils, R., Philipsen, B., van Reenen, K., van Houwelingen, K., van Eekeren, N., 2019. The Effect of Kurzrasen and Strip-Grazing on Grassland Performance and Soil Quality of a Peat Meadow. Sustainability 11 (22).
Holshof, G., van Houwelingen, K., Lenssinck, F.A.J., 2011. Landbouwkundige gevolgen van peilverhoging in het veenweidegebied. Wagening. Livest. Res. ⟨http://edepot.wur.nl/184113⟩.
Hoving, I.E., André, G., Akker, J.J.H. v d, Pleijter, M., 2008. Hydrologische en landbouwkundige effecten van gebruik 'onderwaterdrains' op veengrond = Hydrological and agricultural effects of the use of submerged drains on peatsoil. Animal Sciences Group. ⟨http://edepot.wur.nl/165⟩.
Hoving, I.E., Holshof, G., Hendriks, D.M.D., 2020. Effecten vernattingsmaatregelen op veenweidebedrijven in Noord-Holland; Technische en economische consequenties en effecten op bodemdaling en broeikasgasemissie. Wageningen Livestock Research. https://doi.org/10.18174/535508.
Hoving, I.E., van Middelkoop, J.C., Massop, H.T.L., van den Akker, J.J.H., van Houwelingen, K.M., 2024. Effects of water infiltration on grass yields and soil nitrogen supply of Dutch peat soils. Grassl. Sci. Eur. 29. ⟨https://edepot.wur.nl/660620⟩.
Hoving, I.E., van Riel, J.W., Massop, H.T.L., van den Akker, J.J.H., van Houwelingen, K. M., 2023. Statistische analyse veldexperimenten waterinfiltratiesystemen in het westelijke veenweidegebied van Nederland; Rapportage veldonderzoeken die hebben plaatsgevonden in de periode 2004-2021. Wageningen Livestock Research. https://doi.org/10.18174/558180.
Janssen, P.W.L., Hoekstra, N.J., van Eekeren, N., 2023. Higher density swards have a higher load bearing capacity. Grass Forage Sci. 78 (1), 204–213. https://doi.org/10.1111/gfs.12603.
van Kekem, A., Vries, F., Meer, H., Middelkoop, J., Visser, W., & Zwart, K. (2004). Veengronden en stikstofleverend vermogen. https://doi.org/10.13140/2.1.4502.7522.
KNMI. (n.d.). Archief maand/seizoen/jaaroverzichten. Retrieved 15-8-2024 from ⟨https://www.knmi.nl/nederland-nu/klimatologie/maand-en-seizoensoverzichten/⟩.
KNMI. (2022). Jaar 2021; Gemiddeld normaal met recordaantal codes oranje en rood. Retrieved 26-8-2024 from ⟨https://www.knmi.nl/nederland-nu/klimatologie/maand-en-seizoensoverzichten/2021/jaar⟩.
KNMI. (2024). Jaar 2023; Recordwarm, recordnat en zeer zonnig. Retrieved 20-8-2024 from ⟨https://www.knmi.nl/nederland-nu/klimatologie/maand-en-seizoensoverzichten/2023/jaar⟩.
Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2017. lmerTest Package: Tests in Linear Mixed Effects Models. J. Stat. Softw. 82 (13), 1–26. https://doi.org/10.18637/jss.v082.i13.
Langeveld, C.A., Segers, R., Dirks, B.O.M., van den Pol-van Dasselaar, A., Velthof, G.L., Hensen, A., 1997. Emissions of CO2, CH4 and N2O from pasture on drained peat soils in the Netherlands. Eur. J. Agron. 7 (1), 35–42. https://doi.org/10.1016/S1161-0301(97)00036-1.
Lenth, R.V. (2024). emmeans: Estimated Marginal Means, aka Least-Squares Means. Retrieved 18-11-2024 from ⟨https://rvlenth.github.io/emmeans/⟩.
Pijlman, J., Holshof, G., van den Berg, W., Ros, G.H., Erisman, J.W., van Eekeren, N., 2020. Soil nitrogen supply of peat grasslands estimated by degree days and soil organic matter content. Nutr. Cycl. Agroecosystems 117 (3), 351–365. https://doi.org/10.1007/s10705-020-10071-z.
Poppe, K., van Duinen, L., de Koeijer, T., 2021. Reduction of Greenhouse Gases from Peat Soils in Dutch Agriculture. Eurochoices 20 (2), 38–45. https://doi.org/10.1111/1746-692x.12326.
R Core Team. (2024). R: A Language and Environment for Statistical Computing. In R Foundation for Statistical Computing.
Ramchunder, S.J., Brown, L.E., Holden, J., 2009. Environmental effects of drainage, drain-blocking and prescribed vegetation burning in UK upland peatlands. Prog. Phys. Geogr. Earth Environ. 33 (1), 49–79. https://doi.org/10.1177/0309133309105245.
Renger, M., Wessolek, G., Schwärzel, K., Sauerbrey, R., Siewert, C., 2002. Aspects of peat conservation and water management. J. Plant Nutr. Soil Sci.
Rijksoverheid. (2019). Klimaatakkoord. ⟨https://www.klimaatakkoord.nl/binaries/klimaatakkoord/documenten/publicaties/2019/06/28/klimaatakkoord/klimaatakkoord.pdf⟩.
Schothorst, C.J. (1982). Drainage and behaviour of peat soils. ⟨https://edepot.wur.nl/74505⟩.
Smit, H.J., Metzger, M.J., Ewert, F., 2008. Spatial distribution of grassland productivity and land use in Europe. Agric. Syst. 98 (3), 208–219. https://doi.org/10.1016/j.agsy.2008.07.004.
Sonneveld, M.P.W., Lantinga, E.A., 2011. The contribution of mineralization to grassland N uptake on peatland soils with anthropogenic A horizons. Plant Soil 340 (1), 357–368. https://doi.org/10.1007/s11104-010-0608-7.
Tanneberger, F., Appulo, L., Ewert, S., Lakner, S., Ó Brolcháin, N., Peters, J., Wichtmann, W., 2021. The Power of Nature-Based Solutions: How Peatlands Can Help Us to Achieve Key EU Sustainability Objectives. Adv. Sustain. Syst. 5 (1), 2000146. https://doi.org/10.1002/adsu.202000146.
Tiemeyer, B., Albiac Borraz, E., Augustin, J., Bechtold, M., Beetz, S., Beyer, C., Drösler, M., Ebli, M., Eickenscheidt, T., Fiedler, S., Förster, C., Freibauer, A., Giebels, M., Glatzel, S., Heinichen, J., Hoffmann, M., Höper, H., Jurasinski, G., Leiber-Sauheitl, K., Zeitz, J., 2016. High emissions of greenhouse gases from grasslands on peat and other organic soils. Glob. Change Biol. 22 (12), 4134–4149. https://doi.org/10.1111/gcb.13303.
UNEP. (2022). Global Peatlands Assessment – The State of the World's Peatlands: Evidence for action toward the conservation, restoration, and sustainable management of peatlands.
Vellinga, T.V., André, G., 1999. Sixty years of Dutch nitrogen fertiliser experiments, an overview of the effects of soil type, fertiliser input, management and of developments in time. Neth. J. Agric. Sci. 47 (3-4), 215–241.
Verloop, J., Hilhorst, G.J., Oenema, J., Van Keulen, H., Sebek, L.B.J., Van Ittersum, M.K., 2014. Soil N mineralization in a dairy production system with grass and forage crops. Nutr. Cycl. Agroecosystems 98 (3), 267–280. https://doi.org/10.1007/s10705-014-9610-4.
de Vos, J.A., van Bakel, P.J.T., Hoving, I.E., Conijn, J.G., 2006. Waterpas-model: A predictive tool for water management, agriculture, and environment. Agric. Water Manag. 86 (1-2), 187–195. https://doi.org/10.1016/j.agwat.2006.06.024.
de Vos, J.A., van Bakel, P.J.T., Hoving, I.E., Smidt, R.A., 2010. Raising surface water levels in peat areas with dairy farming Upscaling hydrological, agronomical and economic effects from farm-scale to local scale. Agric. Water Manag. 97 (11), 1887–1897. https://doi.org/10.1016/j.agwat.2010.06.017.
Wichmann, S., Nordt, A., 2024. Unlocking the potential of peatlands and paludiculture to achieve Germany's climate targets: obstacles and major fields of action [Policy and Practice Reviews]. Front. Clim. 6. https://doi.org/10.3389/fclim.2024.1380625.
Wickham, H., 2016. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag. ⟨https://ggplot2.tidyverse.org⟩.
Wilson, D., Blain, D., Couwenberg, J., Evans, C.D., Murdiyarso, D., Page, S.E., Renou-Wilson, F., Rieley, J.O., Sirin, A., Strack, M., Tuittila, E.S., 2016. Greenhouse gas emissions factors associated with rewetting of organic soils. Mires Peat 17 (4), 1–28. https://doi.org/10.19189/MaP.2016.OMB.222.
Xu, J., Morris, P.J., Liu, J., Holden, J., 2018. PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis. CATENA 160, 134–140. https://doi.org/10.1016/j.catena.2017.09.010.
Zak, D., Meyer, N., Cabezas, A., Gelbrecht, J., Mauersberger, R., Tiemeyer, B., Wagner, C., McInnes, R., 2017. Topsoil removal to minimize internal eutrophication in rewetted peatlands and to protect downstream systems against phosphorus pollution: A case study from NE Germany. Ecol. Eng. 103, 488–496. https://doi.org/10.1016/j.ecoleng.2015.12.030.
Zhang, X.-Y., Li, Q.-W., Gao, J.-Q., Hu, Y.-H., Song, M.-H., Yue, Y., 2020. Effects of rainfall amount and frequency on soil nitrogen mineralization in Zoigê alpine wetland. Eur. J. Soil Biol. 97, 103170. https://doi.org/10.1016/j.ejsobi.2020.103170.