Almost 40 years after raised bog restoration on black peat: How did nutrient levels in soil and water change?

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SUMMARY

Restoration of the bog known as the Leegmoor was accompanied by multiple challenging factors such as lack of a white peat layer, strong water table fluctuations and inhibited lateral water movement. The project started in 1983 and was supported by scientific monitoring. Now, almost 40 years later, the monitoring has been repeated to gauge success. As part of this initiative we investigated nutrient dynamics by monitoring soil and water chemistry (NH_4^+, NO_3^-, PO_4^{3-}, pH, N_t, P_t) and soil water content in 2019–2020 and comparing our results with data from the 1980s and 1990s. Soil water content had increased significantly. Soil NO_3^--N content had decreased significantly and was mostly below the detection limit, while pH and NH_4^+-N had not changed. Pore water PO_4^{3-}-P concentrations remained above those observed in near-natural bogs, while nutrients (except NH_4^+-N) in the surface water had decreased to levels observed in near-natural bogs. Most of the observed nutrients had decreased, some to near-natural levels, although the Leegmoor continued to receive an atmospheric supply of nutrients from adjacent agricultural areas. Our results indicate translocation of nutrients into the vegetation, which has developed substantially, but this process can be expected to end at some point.

KEY WORDS: Leegmoor, nutrient relocation, peat extraction, vegetation, nutrient concentrations

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INTRODUCTION

Globally, peatlands cover 3 % of the land area but store 30 % of soil carbon, which makes them the second most important pool of terrestrial carbon (Joosten & Couwenberg 2008, Limpens *et al.* 2008). Pristine peatlands provide various additional ecosystem services like water and nutrient retention, biological habitat and recreational areas (Sjörs 1980, Verry & Timmons 1982). For the last hundreds of years peatlands have been drained for agricultural, industrial and residential use, causing degradation. Drained peatlands can no longer fulfil their ecological roles as habitat for endangered species, water retention areas or carbon and nutrient sinks (Turner 1991, Bonn *et al.* 2014). However, during the last decades, social and scientific focus on peatland restoration has aimed to re-establish natural peatland functions (e.g. Pfadenhauer & Grootjans 1999, Vasander *et al.* 2003, Andersen *et al.* 2017).

The fact that 99 % of German peatlands no longer accumulate peat and have lost their ability to provide ecosystem services (Joosten & Couwenberg 2008, Schulz *et al.* 2019) created an urgent need for action. In particular, raised bogs in north-west Germany were severely affected by agricultural use and peat extraction. The first peat protection programme was implemented in 1981 in the state of Lower Saxony, where most (73 %) of the German bogs are situated. This programme regulates the use of bogs for industrial peat extraction and manages their restoration after extraction is terminated. Previously, agricultural after-use was common, ignoring the potential for nature conservation. The regulations for restoration were updated in 1988 by adding, among other things, a requirement that a layer of black peat at least 0.5 m thick must remain after extraction to support the re-establishment of bog vegetation (Blankenburg 2004, Caspers & Schmatzler 2009).

Pristine raised bogs are permanently waterlogged and anoxic ombrotrophic peatlands, receiving water only as precipitation with no groundwater contribution (Sjörs 1980). Consequently, they are characterised by low concentrations of plant available nutrients, low pH due to lack of calcium carbonate and, on the other hand, the presence of humic acids and cation exchange effected by bryophytes in the vegetation (Bourbonniere 2009). Nutrient fixation in the peat exceeds losses in gaseous and dissolved form owing to hampered biochemical cycling (Urban & Eisenreich 1988, Blodau 2002, Limpens *et al.* 2008). Nutrient retention is mainly controlled by plant uptake, peat formation and microbial turnover (peat mineralisation), with plant uptake prevailing under near-natural conditions (Hemond 1983, Kellogg & Bridgham 2003).

Although total nitrate and phosphate levels in bog water are usually low, microbial turnover can be rapid in an aerobic layer of topsoil close to the surface, enhancing peat mineralisation and the release of nutrients into solution for plant uptake (Bridgham *et al.* 1998). Therefore, levels of plant available nutrients like ammonium, nitrate and phosphate are usually higher in drained bogs than under pristine conditions (Brake *et al.* 1999, Frank *et al.* 2014), which presents a challenge when the aim is to re-establish natural bog habitat. Also, the water table rises seasonally due to changes in temperature, precipitation and evaporation (Bourbonniere 2009), curbing microbial activity in the aerated part of the peat column, and this may be reflected in locally and temporarily raised concentrations of plant available nutrients. Overall, the re-establishment of near-natural conditions is often hampered by non-reversible changes of physical peat properties, substantial reduction of peat thickness, and higher nutrient availability (e.g. Vasander *et al.* 2003, Triisberg *et al.* 2014, Zając *et al.* 2018).

In the case of the bog known as the Leegmoor, restoration was challenging in many ways. The thickness of parts of its remaining peat layer was 30 cm or less, so it was unclear whether this layer would provide sufficient sealing. In addition, white peat - which has favourable properties for water storage - was completely absent and it was unclear whether the remaining black peat had sufficient swelling capacity for water management.

The objective of this investigation was to evaluate the success of bog restoration at the Leegmoor after almost 40 years by assessing its soil and water chemistry and comparing our results with data collected from the same site in the 1980s and 1990s on the one hand, and from near-natural raised bogs on the other. For this purpose we monitored the concentrations of ammonium (NH_4^+), nitrate (NO_3^-), phosphate (PO_4^{3-}), total nitrogen (N_t), total phosphorus (P_t), and pH over the course of one year. Our main hypothesis was (1) that the nutrient concentrations in peat and peat waters had decreased and developed towards near-natural conditions. We assumed that the uptake of nutrients by the vegetation which had developed since the initial examination exceeded nutrient release by peat mineralisation and nutrient inputs by atmospheric deposition. Beyond that, water table levels vary greatly throughout the project area and are subject to fluctuation, thus affecting peat mineralisation and presumably nutrient retention. Therefore, we further hypothesised that the variation in water table levels causes (2) spatial differences in nutrient dynamics and (3) seasonal variability of nutrient dynamics.

METHODS

Study site

The Leegmoor (52° 59′ 34″ N, 7° 33′ 20″ E) is an oligotrophic raised bog in the Emsland region of north-western Germany (Figure 1). It developed on Pleistocene sand layers of a ground moraine from the Saalian glaciation under a humid temperate climate (Woldstedt 1954). The size of the investigation area is about 60 ha, the area of the Leegmoor is about 300 ha, and peat was formerly extracted from the entire area (Nick *et al.* 1993).

Before peat extraction, the area was already partly drained and was presumably used for buckwheat cultivation which involved regular burning of the surface peat layers. Industrial peat extraction with excavators started in 1950 when drainage ditches were cut deep into the underlying sand. At that time, there were already no remnants of white peat and 2–2.5 m of black peat was extracted. The thickness of the residual peat layer ranged from 0.2 m in the northern part of the bog to more than 2 m in the
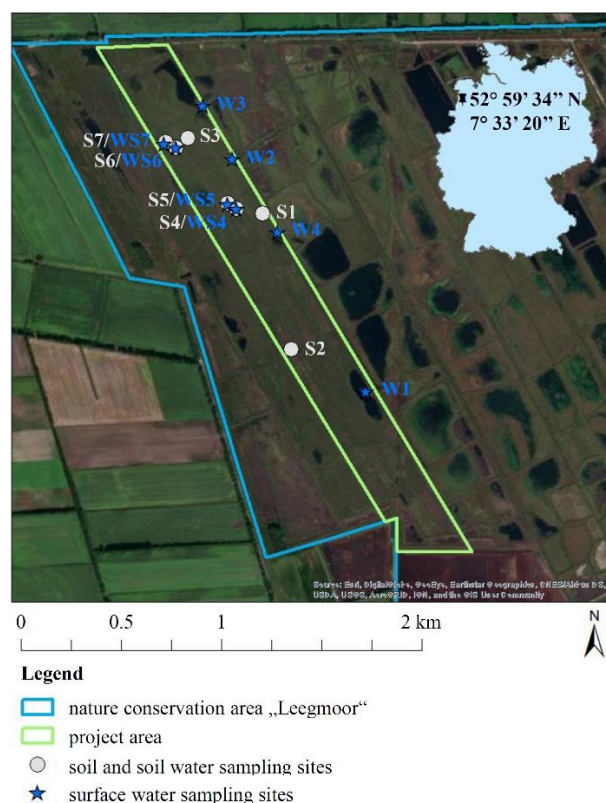

Figure 1. Map of the Leegmoor project area with sampling sites (soil and soil water sampling sites: gradient from S1 = driest to S7 = wettest site; surface water sampling sites: W1 = bog pond, W2 = ditch, W3 and W4 = intermittent ponds, WS4 – WS7 = intermittent surface water at wetter soil sampling sites).

southern part. The remaining black peat was moderately to highly decomposed (von Post H6–H9) with correspondingly low horizontal water flow and water level fluctuations > 1 m (Nick *et al.* 1993).

Restoration activities started in 1983 and lasted until 1984. Trenches were filled, the area was levelled with bulldozers, dams with overflows were built to create polders and to prevent water drainage, and a bog pond was excavated for water storage. Typical bog vegetation (*Eriophorum vaginatum*, *Eriophorum angustifolium*, *Erica tetralix*, *Calluna vulgaris* and various *Sphagnum* species) was introduced onto the bare black peat as plaggen (turf), plants or seeds. The restoration was accompanied by scientific monitoring of hydrology, meteorology, vegetation, fauna and nutrients from 1983 until 1996 (Eggelsmann & Blankenburg 1989, Nick *et al.* 1993, Gebhardt & Knabke 1994, Nick *et al.* 2001). The phase of scientific control started at the end of 2018.

Soil and water sampling
For comparability, we adopted sampling sites from the initial examination of soil nutrients conducted in 1990–1993 by Gebhardt & Knabke (1994). Altogether we sampled seven sites (Figure 1), six from the initial examination (S1 and S3–S7) and one additional site (S2). The latter was selected because it lies within a stand of heather, which was not represented in the initial examination. The sampling sites were arranged along a gradient of waterlogging from S1 (driest) to S7 (wettest). Disturbed soil cores were obtained using a boring rod for peat soils (a gouge auger that is slightly tapered towards the bottom), at monthly to bimonthly intervals from April 2019 to February 2020. Three cores were taken per sampling site. Each core was split at depth 15 cm and representative soil samples were collected from each of the two sections (depths 0–15 cm and 15–30 cm, respectively). The soil samples were placed in polyethylene (PE) bags for transportation.

For the extraction of pore water samples at the seven soil sampling sites, small holes were prepared by inserting a boring rod to the desired depth. Directly after that, a lance was inserted to the required sampling depth of 10 cm (for the 0–15 cm layer) or 25 cm (for the 15–30 cm layer). The lance was a tube with a syringe connected at its top end, through which about 150 mL of pore water was obtained via suction. Samples were collected on three occasions, in May, June and August 2020. They were transferred into PE bottles for transport.

Surface water samples were collected from eight sampling sites (Figure 1) on the same days as the soil samples. We sampled from the bog pond (W1), two smaller ponds (W3, W4) that contain water only

seasonally (intermittent ponds), a ditch (W2) stretching from the western to the eastern boundary of the investigation area, and the four wetter soil sampling sites (WS4–WS7) where the water table was above ground level during part of the year. All sites except the bog pond dried out for one or more months during the summer of 2019. For reference, on one occasion in the summer of 2019 we collected surface water samples at four other bogs in the region (Table A1 in the Appendix), which we shall refer to as hereafter as the regional reference sites. The surface water samples were transported in PE bottles.

For soil bulk density at the soil sampling sites (S1–S7), we collected undisturbed samples from soil profiles using 100 cm^3 steel rings. These samples were taken during the dry summer of 2019. Three replicates were sampled per horizon and the sampling rings were closed with lids for transport.

All samples were stored in a dark and cool storage box during transport to the laboratory, then refrigerated at 4 °C until required for further procedures. Contamination from the transport vessels (both plastic and metal) can be ruled out. Throughout the entire sampling procedure, nitrile gloves were worn.

Laboratory analyses
Unless stated otherwise, the soil samples were dried at 40 °C and passed through a 2 mm sieve, and are hereafter referred to as fine soil. Surface and soil water samples were filtered (mesh 8–12 µm for NH_4^+-N and NO_3^--N, mesh 2–3 µm for PO_4^{3-}-P analysis) the day after sampling and stored at -18 °C until measurement. Nitrile gloves were worn throughout the entire laboratory procedure.

Bulk density was determined by DIN 19672-1:1968-04. The soil samples in steel rings were dried to constant weight at 105 °C and allowed to cool in a desiccator before determining dry weight. The bulk density was calculated from sample dry weight and cylinder volume.

Soil water content was determined according to VDLUFA (1991a). 5 g of fresh soil taken from a disturbed sample was weighed, dried to constant weight at 105 °C, then allowed to cool in a desiccator before weighing to determine dry weight. Water content was calculated as the difference between fresh and dry weight.

pH was measured with a portable pH meter (Knick Portamess 911 pH). Surface water samples were measured immediately before filtration. For soil pH, 5 g of air-dry fine soil was blended with 25 mL of 0.01 M $CaCl_2$ solution in distilled H_2O and the pH value was measured after establishment of equilibrium (Blume *et al.* 2011).

Mineral nitrogen (NH_4^+-N and NO_3^--N) in the soil extract and surface water samples was analysed by fractionated distillation, according to VDLUFA (1997) and Blume *et al.* (2011). To produce soil extracts, 25 g of fresh soil was shaken for 30 minutes with 100 mL of 0.0125 M $CaCl_2$ solution, then vacuum filtered (8–12 µm) and stored in PE bottles at -18 °C until measurement. Steam distillation was conducted using the Omnilab FoodALYT D 3000, and titration with the Schott Titronic 96.

Plant available P was extracted with calcium-acetate-lactate (0.6 wt-%, pH 3.6) solution (VDLUFA 1991b). 5 g of air-dry fine soil was shaken with 125 mL of the extracting agent for two hours, filtered (2–3 µm) and stored in PE bottles at -18 °C until measurement. Phosphate (PO_4^{3-}-P) was then determined colorimetrically at 820 nm in a spectrophotometer (Shimadzu UVmin-1240) after adding 0.5 wt-% ascorbic acid and 1 wt-% ammonium heptamolybdate solution as a colouring agent. Phosphate in surface water and pore water samples was determined by the ammonium molybdate spectrometric method (DIN EN ISO 6878:2004-09) in a spectrophotometer (Shimadzu UVmin-1240).

The content of total soil nitrogen (N_t) was determined by CN elemental analysis. Air-dry fine soil from the individual samples was combined in equal parts to make up a single composite sample per sampling site and soil layer. A portion of the composite sample was ground in a ball mill and dried at 105 °C, the resulting material being referred to hereafter as powdered soil. Duplicate 2–3 mg sub-samples of the powdered soil were weighed into tin caps for analysis in a CN-Analyser (Thermo Fisher Scientific, Flash 2000).

The content of total soil phosphorous (P_t) was determined by a digestion procedure involving incineration as amended by Bleck (1965). 1 g of powdered soil was incinerated in a muffle furnace at 800 °C for 3 h, then cooked with diluted HNO_3 (volume 1:3), filtered (15 µm), blended with 0.25 wt-% ammonium vanadate solution and 5 wt-% ammonium molybdate solution, then measured in a spectrophotometer (Shimadzu UVmin-1240) at 430 nm.

The analytical approach in the initial examination (Gebhardt & Knabke 1994) was the same as described above, except that PO_4^{3-}-P was determined for the soil solution rather than for soil extracts. The samples of soil solution were produced by adding distilled H_2O to 800–1000 g of fresh soil until the soil was saturated, leaving for 24 h to equilibrate, then centrifuging and filtering (no further specification).

Data analysis

Mean and standard deviation were calculated in MS Excel (MS 365, 2021). Statistical tests were conducted in IBM SPSS Version 27. We compared our results for soil nutrient contents with the dataset from the initial examination in 1990–1993 (Gebhardt & Knabke 1994). Deviation from the normal distribution was tested using the Kolmogorov-Smirnoff test. As most of the factors measured did not follow normal distributions, we used non-parametric tests in the following.

The Wilcoxon signed-rank test was used to test the differences in soil NH_4^+-N, NO_3^--N and soil water content between the initial examination (1990–1993) and our data (2019–2020). Site S1 was excluded from the test, as it stood out in the first measurement period by having high NH_4^+-N and NO_3^--N contents, presumably due to the lack of vegetation cover (Figure 2; see Table A2 for a list of vegetation cover). Soil PO_4^{3-}-P was not measured during the initial examination, so lack of data prohibited a statistical comparison. Soil pH values were provided only as range values in the initial examination and, on that basis, statistical comparison was not possible.

In the data from the initial examination (Gebhardt & Knabke 1994), N_t was expressed as % of dry soil, and was converted to mg kg^{-1} for comparability. NH_4^+-N, NO_3^--N and P_t were provided as kg ha^{-1} and were not converted further. Soil nutrient contents from the follow-up examination were converted from mg kg^{-1} to kg ha^{-1} (for the respective 15 cm soil layers) using bulk densities (Table A3).

The only previous measurements of water nutrient concentrations and pH available for comparison were mean values and ranges for the bog pond and ditch outflow from a study conducted in 1984–1988 (Eggelsmann & Blankenburg 1989), which preceded the initial examination. Due to the lack of continuously logged data, changes could not be tested statistically.

Correlations between the waterlogging gradient and the soil nutrient contents (NH_4^+-N, NO_3^--N, PO_4^{3-}-P), as well as the pore water PO_4^{3-}-P concentration, were tested by Spearman's rank correlation test. For the water logging gradient (S1 = driest to S7 = wettest), we considered soil water content, water table at the closest measuring gauge (unpublished data provided by Geological Service Bremen, GdfB), water table measured directly at the site, and the depth of the reduced soil horizon.
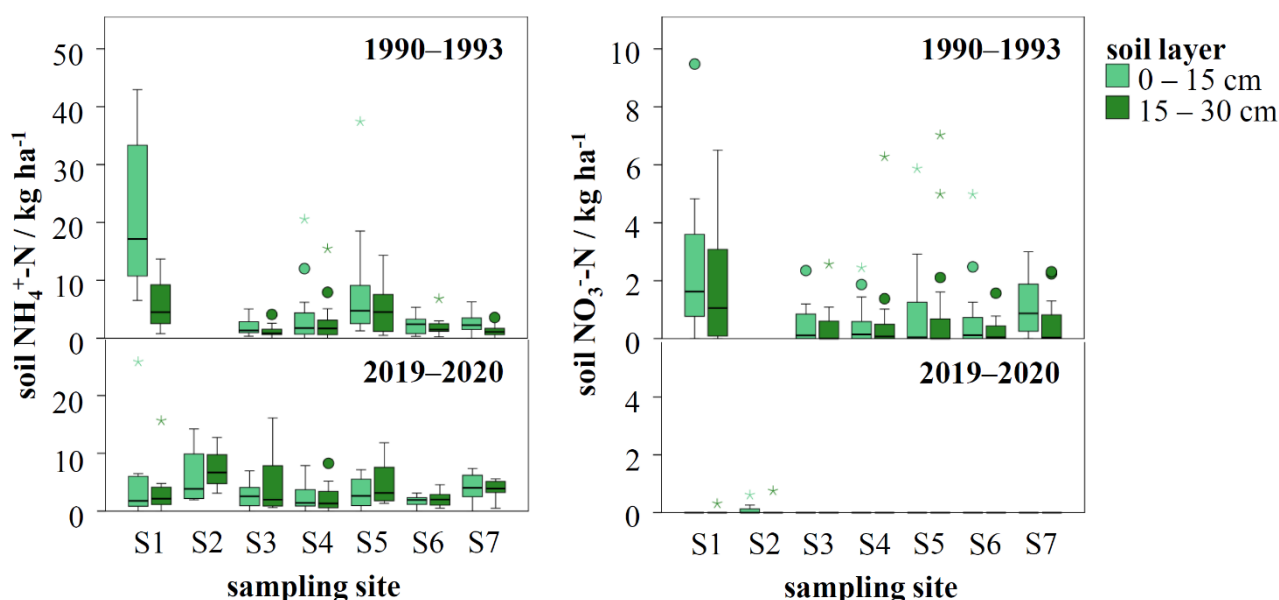
Figure 2. Boxplots of soil NH_4^+-N content (left) and NO_3^--N content (right) during the initial examination in 1990–1993 (Gebhardt & Knabke 1994) and during the follow-up examination in 2019–2020. Gradient from S1 = driest to S7 = wettest site; ° mild outliers, * extreme outliers.

RESULTS

Soil properties

In the initial examination (1990–1993), mean soil water content was 75 vol.% in the 0–15 cm layer and 73 vol.% in the 15–30 cm layer (Gebhardt & Knabke 1994). In this follow-up examination the average was 80 vol.% at 0–15 cm and 83 vol.% at 15–30 cm depth. The mean values indicated an increase in soil water content and the difference between the datasets (Table A4) was significant (Table 1).

During the initial examination the range of pH (H_2O) values was 3.2–3.9 and the range of pH ($CaCl_2$) values was 2.3–3.0 (Gebhardt & Knabke 1994). The pH (H_2O) values recorded during this follow-up examination were 3.2–3.5 in the 0–15 cm layer and 3.0–3.6 in the 15–30 cm layer; while the pH ($CaCl_2$) values were 2.3–2.9 in the 0–15 cm layer and 2.3–2.8 in the 15–30 cm layer (Table 1). Thus, the overall pH values did not differ notably between the two sampling periods.

Mean values of plant available NH_4^+-N content recorded in the initial examination (1990–1993) were 3.53 kg ha^{-1} (=11.1 mg kg^{-1}) in the 0–15 cm layer and 2.44 kg ha^{-1} (=10.2 mg kg^{-1}) in 15–30 cm layer (Gebhardt & Knabke 1994). In this follow-up examination, the average contents were 3.68 kg ha^{-1} (=17.3 mg kg^{-1}) at 0–15 cm and 4.15 kg ha^{-1} (=22.3 mg kg^{-1}) at 15–30 cm depth (Figure 2, Table 1). Although the data indicate that NH_4^+-N contents are slightly larger nowadays, the differences between the

initial examination and this follow-up examination were not significant (Table 1). In the initial examination (1990–1993), mean plant-available NO_3^--N content was 0.68 kg ha^{-1} (=2.26 mg kg^{-1}) at 0–15 cm and 0.52 kg ha^{-1} (=2.21 mg kg^{-1}) at 15–30 cm, whereas NO_3^--N was mostly below detection limit in this follow-up examination (Figure 2, Table 1). NO_3^--N occurred at a detectable level only in the driest sampling sites during the summer months. The differences were statistically significant ($p = 0.01$) for both soil layers (Table 1). Site S1, which was formerly bare soil (see Table A2), stood out in the first measurement period because of its high NH_4^+-N and NO_3^--N contents. However, both NH_4^+-N and NO_3^--N had decreased and were no longer striking amongst our results (Figure 2).

In this follow-up examination plant-available soil PO_4^{3-}-P contents were on average 2.94 kg ha^{-1} (= 32.6 mg kg^{-1}) in the 0–15 cm layer and 1.43 kg ha^{-1} (= 17.0 mg kg^{-1}) in the 15–30 cm layer and were consistently higher in the upper soil layer than in the lower one (Figure 3, Table 1). As this property was not measured during the initial examination (1990–1993), a comparison was not possible.

Neither NH_4^+-N nor NO_3^--N in soil correlated with the waterlogging gradient (Table 2). Soil PO_4^{3-}-P had a medium negative correlation ($p = 0.01$) in the 0–15 cm layer and a weak negative correlation ($p = 0.05$) in the 15–30 cm layer.

Soil N_t contents in the initial examination were on average 9967 mg kg^{-1} in the 0–15 cm layer and

Table 1. Summary of soil properties recorded in 1990–1993 (initial examination; Gebhardt & Knabke 1994) and 2019–2020 (follow-up examination), and results of Wilcoxon signed-rank test for differences between the two examinations. min = minimum; max = maximum; s = standard deviation; n = sample size. N/A - data not available; n.s. - not significant; [a] - site S1 excluded because of extraordinarily high N contents during the initial examination; [b] - data (or unit) not provided in the initial examination.

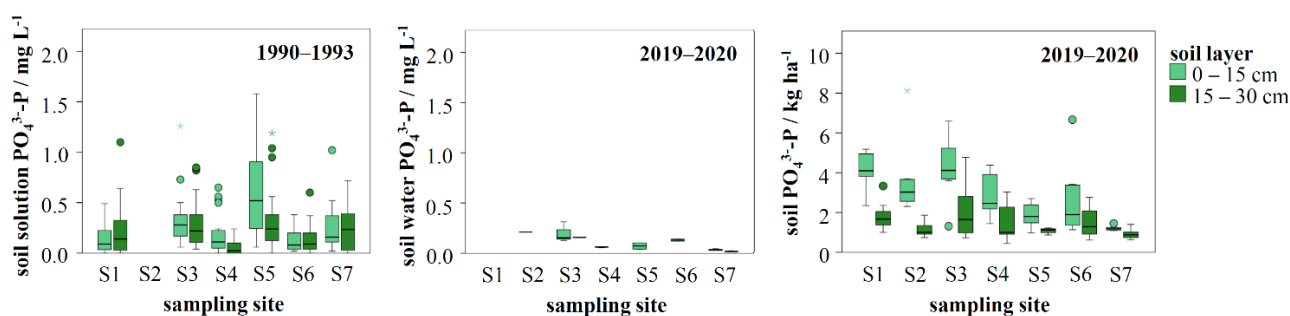| soil properties	units	soil layer (cm)	1990–1993					2019–2020					1990–1993 vs. 2019–2020
			min	max	mean	s	n	min	max	mean	s	n	
NH_4^+-N	kg ha^{-1}	0–15	0.00[a]	37.5	3.53[a]	4.66[a]	141[a]	0.00	25.9	3.68	4.23	56	n.s.
		15–30	0.00[a]	15.5	2.44[a]	3.09[a]	129[a]	0.00	16.2	4.15	3.88	56	n.s.
	mg kg^{-1}	0–15	N/A[b]					0.00	86.3	17.3	18.9	56	-
		15–30						0.00	99.3	22.3	23.6	56	-
NO_3^--N	kg ha^{-1}	0–15	0.00[a]	5.87[a]	0.68[a]	1.01[a]	141[a]	0.00	0.61	0.02	0.09	56	p = 0.01
		15–30	0.00[a]	7.03[a]	0.52[a]	1.12[a]	129[a]	0.00	0.76	0.02	0.11	56	p = 0.01
	mg kg^{-1}	0–15	N/A[b]					0.00	12.8	0.33	1.85	56	-
		15–30						0.00	20.2	0.45	2.76	56	-
PO_4^{3-}-P	kg ha^{-1}	0–15			N/A[b]			0.97	8.14	2.94	1.63	56	-
		15–30						0.45	4.77	1.43	0.81	56	-
	mg kg^{-1}	0–15			N/A[b]			10.1	118	32.6	20.0	56	-
		15–30						4.70	54.0	17.0	9.09	56	-
N_t	kg ha^{-1}	0–15			N/A[b]			2116	3343	2514	393	7	-
		15–30						1315	2750	2151	505	7	-
	mg kg^{-1}	0–15	8900	12000	9967	1084	N/A	8112	9963	8944	610	7	-
		15–30	8600	9800	9280	476	N/A	7522	9518	8215	661	7	-
P_t	kg ha^{-1}	0–15	62.6	89.9	72.4	11.4	N/A	47.5	73.8	56.6	9.33	7	-
		15–30	30.4	68.1	45.6	15.2	N/A	26.3	50.7	40.1	9.06	7	-
	mg kg^{-1}	0–15			N/A[b]			177	240	202	25.2	7	-
		15–30						131	187	155	20.3	7	-
water content	vol.%	0–15	48.1	86.6	74.7	11.6	147	67.3	89.2	80.0	4.8	56	p = 0.01
		15–30	39.9	91.1	73.3	13.7	141	71.7	90.0	82.8	4.3	56	p = 0.05
pH (H_2O)		0–15	3.2	3.9		N/A		3.2	3.5	3.4	0.1	56	-
		15–30						3.0	3.6	3.3	0.1	56	-
pH ($CaCl_2$)		0–15	2.3	3.0		N/A		2.3	2.9	2.6	0.1	56	-
		15–30						2.3	2.8	2.6	0.1	56	-

Figure 3. Boxplots of PO_4^{3-}-P concentrations in soil water and soil during the initial examination in 1990–1993 (Gebhardt & Knabke 1994) and during the follow-up examination in 2019–2020. Gradient from S1 = driest to S7 = wettest site; ° mild outliers, * extreme outliers.

Table 2. Spearman's rank correlation coefficient (ρ) values for correlations between waterlogging gradient and concentrations of soil nutrients and pore water PO_4^{3-}-P during the follow-up examination in 2019–2020. Significance levels: ** p = 0.01; * p = 0.05.

properties	soil layer (cm)	Spearman ρ
soil NH_4^+-N (kg ha^{-1})	0–15	-0.043
	15–30	-0.091
soil NO_3^--N (kg ha^{-1})	0–15	-0.192
	15–30	-0.240
soil PO_4^{3-}-P (kg ha^{-1})	0–15	-0.689 **
	15–30	-0.329 *
pore water PO_4^{3-}-P (mg L^{-1})	0–15	-0.732 **
	15–30	-0.866

9280 mg kg^{-1} in the 15–30 cm layer. The corresponding values from this follow-up examination were 8944 mg kg^{-1} (= 2514 kg ha^{-1}) in the 0–15 cm layer and 8215 mg kg^{-1} (= 2151 kg ha^{-1}) in the 15–30 cm layer. Soil P_t contents in the initial examination were on average 72.4 kg ha^{-1} in the 0–15 cm layer and 45.6 kg ha^{-1} in the 15–30 cm layer. In this follow-up examination the soil P_t contents were 56.6 kg ha^{-1} (= 202 mg kg^{-1}) in the 0–15 cm layer and 40.1 kg ha^{-1} (= 155 mg kg^{-1}) in the 15–30 cm layer (Table 1).

Soil water chemistry
In the initial examination, PO_4^{3-}-P concentration in the soil solution was on average 0.27 mg L^{-1} in the 0–15 cm layer and 0.22 mg L^{-1} in the 15–30 cm layer. In this follow-up examination PO_4^{3-}-P concentration in the pore water was on average 0.11 mg L^{-1} in the 0–15 cm layer and 0.07 mg L^{-1} in the 15–30 cm layer (Figure 3, Table 3). Thus, plant-available soil PO_4^{3-}-P

concentrations were in general higher in the upper soil layer. Disparity of extraction methods prohibited statistical comparison of results from the initial and follow-up examinations. However, if equivalence between methods is assumed, the PO_4^{3-}-P levels were lower in the follow-up study (see Table A5 for a summary of data from individual sampling sites). In 2019–2020, pore water PO_4^{3-}-P in the 0–15 cm layer showed a medium (p = 0.01) negative correlation with the waterlogging gradient (Table 2).

Surface water chemistry
In this follow-up examination NH_4^+-N concentrations were on average 0.38 mg L^{-1} for all surface water samples and 0.70 mg L^{-1} for the bog pond. Higher concentrations were measured in the bog pond during a study preceding the initial examination, which reported a mean of 1.40 mg L^{-1} and a value of 1.50 mg L^{-1} for the outflow (Eggelsmann & Blankenburg 1989). The mean NH_4^+-N concentration in surface waters of the regional reference sites was 0.61 mg L^{-1} (Figure 4, Table 4).

The surface water NO_3^--N concentrations measured in this follow-up examination were on average 0.01 mg L^{-1} for all samples and 0.03 mg L^{-1} for the bog pond. As for NH_4^+-N, the NO_3^--N concentration in the bog pond was higher during the study preceding the initial examination, with a mean of 0.30 mg L^{-1} and a value of 0.30 mg L^{-1} also in the outflow (Eggelsmann & Blankenburg 1989). The average concentration for the regional reference sites was 0.01 mg L^{-1} (Figure 4, Table 4).

The PO_4^{3-}-P concentrations in this follow-up examination were on average 0.02 mg L^{-1} for all surface water samples and 0.03 mg L^{-1} in the bog pond. Again, the study preceding the initial examination (Eggelsmann & Blankenburg 1989) revealed higher concentrations in the bog pond (mean value 0.05 mg L^{-1}) and an even higher concentration of 0.13 mg L^{-1} in the outflow. The mean PO_4^{3-}-P

Table 3. Summary of soil water PO_4^{3-}-P concentrations from the initial examination in 1990–1993 (Gebhardt & Knabke 1994) and the follow-up examination in 2019–2020. min = minimum; max = maximum; s = standard deviation; n = sample size.

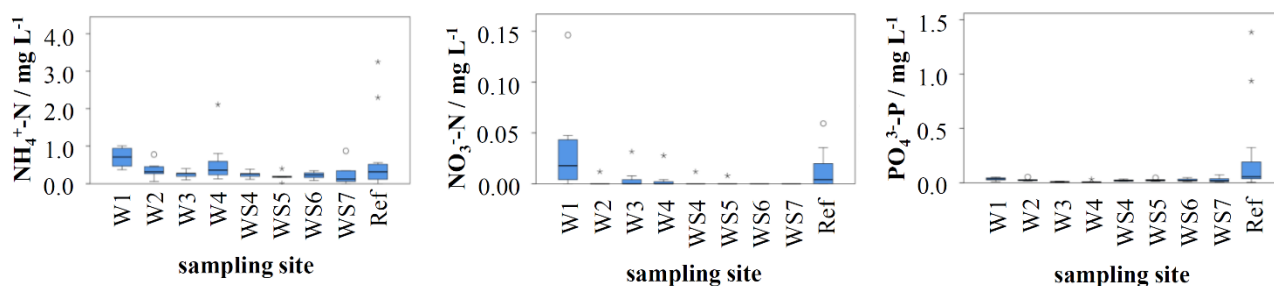| soil property	units	soil layer (cm)	1990–1993 (soil solution)					2019–2020 (pore water)				
			min	max	mean	s	n	min	max	mean	s	n
PO_4^{3-}-P	mg L^{-1}	0–15	0.00	1.58	0.27	0.29	134	0.03	0.31	0.11	0.08	14
		15–30	0.00	1.19	0.22	0.25	126	0.02	0.16	0.07	0.08	5

Figure 4. Boxplots of nutrient concentrations in surface water during the follow-up examination in 2019–2020. W1 = bog pond, W2 = ditch, W3 and W4 = intermittent ponds, WS4–WS7 = intermittent surface water at wetter sites; ° mild outliers, * extreme outliers.

Table 4. Summary of surface water properties (see Table A6) from the study preceding the initial examination in 1984–1988 (Eggelsmann & Blankenburg 1989) and the follow-up examination in 2019–2020. N/A = data not available; min = minimum; max = maximum; s = standard deviation; n = sample size.

surface water properties	units	sampling site	1984–1988	2019–2020				
			mean	min	max	mean	s	n
NH_4^+-N	mg L^{-1}	bog pond (W1)	1.40	0.37	1.01	0.70	0.25	8
		ditch, outflow (W2)	1.50	0.05	0.77	0.37	0.22	7
		Leegmoor	N/A	0.00	2.11	0.38	0.36	48
		regional reference sites	N/A	0.00	3.25	0.61	0.95	14
NO_3^--N	mg L^{-1}	bog pond (W1)	0.30	0.00	0.15	0.03	0.05	8
		ditch, outflow (W2)	0.30	0.00	0.01	0.00	0.00	7
		Leegmoor	N/A	0.00	0.15	0.01	0.02	48
		regional reference sites	N/A	0.00	0.06	0.01	0.02	14
PO_4^{3-}-P	mg L^{-1}	bog pond (W1)	0.05	0.01	0.05	0.03	0.01	8
		ditch, outflow (W2)	0.13	0.01	0.05	0.03	0.01	7
		Leegmoor	N/A	0.00	0.07	0.02	0.02	48
		regional reference sites	N/A	0.01	1.38	0.23	0.41	14
pH		bog pond (W1)	4.2	4.0	4.3	4.1	0.1	8
		ditch, outflow (W2)	4.1	4.1	4.5	4.2	0.2	7
		Leegmoor	N/A	3.8	4.9	4.1	0.2	48
		regional reference sites	N/A	3.8	5.8	4.3	0.5	14

concentration in the surface waters of the regional reference sites was 0.23 mg L^{-1} (Figure 4, Table 4), which clearly exceeded the former and present concentrations at the Leegmoor.

The range of pH values measured in this follow-up examination was 3.8–4.9 with a mean of 4.1 for all surface water samples and the bog pond (Figure 4, Table 4). pH values from the study preceding the initial examination ranged from 4.1 in the bog pond to 4.2 in the outflow (Eggelsmann & Blankenburg 1989). The pH value for surface waters of the regional reference sites was on average 4.3 and, accordingly, similar to both former and present surface water pH values at the Leegmoor.

Focusing on seasonal variability, the soil NH$_4^+$-N contents revealed different patterns during the two examination periods (Figure 5). In the initial examination, interannual fluctuations exceeded those during a single year and fluctuations in the topsoil were partly larger than in the subsoil. The latter difference was not found in this follow-up examination. In both examination periods, NH$_4^+$-N content did not follow a seasonal trend; the same was true for NO$_3^-$-N and PO$_4^{3-}$-P (data not shown).

DISCUSSION

Changes of soil and water nutrients

Neither soil pH (H$_2$O) nor pH (CaCl$_2$) values in the Leegmoor have changed notably since the initial examination. According to Blume *et al.* (2010) the soil pH (CaCl$_2$) of natural bogs typically ranges between < 3 and 4. In that regard, the Leegmoor with an averaged pH (CaCl$_2$) of 2.6 can be considered near-natural. The same is true for pH values of the Leegmoor surface waters (4.1), which are similar to values recorded during the initial examination (4.2; Eggelsmann & Blankenburg 1989) and at the regional reference sites (4.3), and within the near-natural range of pH 3.5–4.5 (Bourbonniere 2009). Hence, the Leegmoor has good status regarding pH values, which was already the case in the initial examination and was not expected to change considerably. Much more crucial, however, are the nutrient conditions.

The soil N$_t$ contents (8944 mg kg^{-1} in the 0–15 cm layer and 8215 mg kg^{-1} in the 15–30 cm layer) are characteristic for natural bog soils and consistent with other findings (Frank *et al.* 2014, Purre &
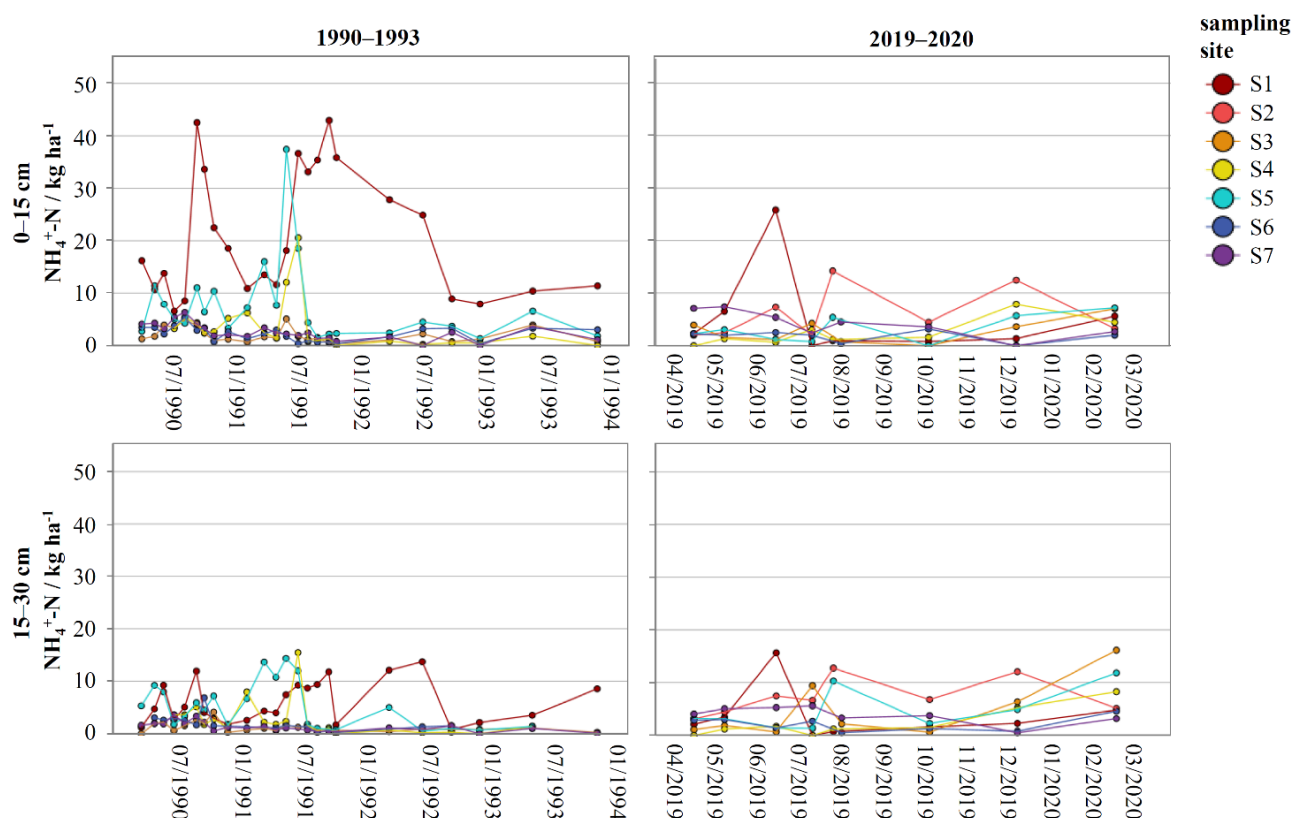
Figure 5. Scatter plots of soil NH$_4^-$-N content during the initial examination in 1990–1993 (Gebhardt & Knabke 1994) and during the follow-up examination in 2019–2020.

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Ilomets 2018). Similarly, the P_t contents (202 mg kg^{-1} in 0–15 cm and 155 mg kg^{-1} in 15–30 cm) are typical for bog soils and similar to results from other studies (Brake *et al.* 1999, Purre & Ilomets 2018, DVGW 2020). However, total nutrient amounts are inert factors that do not influence the nutrient dynamics in terms of plant uptake, leaching or gaseous losses. In this regard, mobile or plant available nutrient species need to be considered.

We found that soil NO_3^--N contents had decreased significantly (p < 0.01) since the years following restoration, from 0.7 kg ha^{-1} (0–15 cm) to not detected in most samples during this follow-up study. Changes in soil NH_4^+-N contents were not significant, although they indicated a trend towards increased amounts, indicating a shift in the N dynamics. Previous studies have shown that soil moisture has a significant effect on N dynamics. A case study on a cutover bog in south Poland showed that NO_3^--N was lower by a factor of 2 and NH_4^+-N was almost 5 times higher in wetter soils (1.5 mg kg^{-1} NO_3^--N and 77.5 mg kg^{-1} NH_4^+-N) than in drier soils (3.3 mg kg^{-1} NO_3^--N and 16.2 mg kg^{-1} NH4$^+$-N) (Zając *et al.* 2018). Similarly, Bobbink *et. al* (2012) outlined how the availability of NH_4^+-N increased under flooded conditions. On a lowland bog in Poland, Jonczak *et al.* (2015) found that water saturation hampered mineralisation and thus caused low NO_3^--N content (0.011 g kg^{-1}) at depth 0–10 cm and an even lower value (0.007 g kg^{-1}) at depth 20–30 cm. In this case, however, NH_4^+-N also decreased with depth (0.337 g kg^{-1} at 0–10 cm to 0.182 g kg^{-1} at 20–30 cm). In the Leegmoor, soil water content has increased overall since the restoration, causing more reduced conditions with oxygen depletion and also affecting the N dynamics and the proportion of N compounds. As nitrate usually exceeds ammonium under aerated conditions (Salonen 1994), the results of this study show that waterlogging has been established and this has hampered nitrification so that NH_4^+-N concentrations now substantially exceed NO_3^--N concentrations. This finding indicates a restoration success, caused by improved hydrological conditions. Soil NO_3^--N levels in the Leegmoor can be considered pristine, while NH_4^+-N levels are about twice those in natural bog (Williams *et al.* 1999).

Like the soil situation, the surface water NO_3^--N concentration in the bog pond decreased tenfold from 0.30 mg L^{-1} (Eggelsmann & Blankenburg 1989) to 0.03 mg L^{-1}. However, these concentrations were already below the maximum for near-natural bogs of 0.5 mg L^{-1} (Bourbonniere 2009) during earlier studies at the Leegmoor. The amount of NH_4^+-N in the bog pond also decreased, from 1.4 mg L^{-1} (Eggelsmann & Blankenburg 1989) to 0.70 mg L^{-1}. However, unlike

NO_3^--N, NH_4^+-N remained above the threshold (maximum 0.1 mg L^{-1}) for near-natural bogs (Bourbonniere 2009). As NO_3^--N and NH_4^+-N concentrations in the outflow were not considerably higher than in the bog pond during the earlier study by Eggelsmann & Blankenburg (1989), decline through leaching can be ruled out. Besides the effect of lowered peat mineralisation, the decrease indicates nutrient relocation into plants, particularly the dense vegetation cover that has developed from more or less bare soil since the initial examination. The regional reference sites in this study (0.61 mg L^{-1} NH_4^+-N and 0.01 mg L^{-1} NO_3^--N) had similar concentrations to the Leegmoor. In contrast, a review study of multiple near-natural and fallow bogs in north-western Germany (DVGW 2020) reported notably higher averaged concentrations of 0.2 mg L^{-1} NO_3^--N and 1.3 mg L^{-1} NH_4^+-N, showing that the levels in the Leegmoor are comparatively low on a regional scale.

Furthermore, the clear decrease of soil NO_3^--N and NH_4^+-N contents at site S1, which was bare soil during the initial examination, showed the importance of a closed vegetation cover for N dynamics. At the same time, the increase of vegetation cover implies a shift of nutrients from soil and water into plant biomass. Although the other sites were not bare in the initial examination, biomass increased at all of the sampling sites. Overall, the Leegmoor is dominated by large stands of *Molinia caerulea*, which is an indicator for degradation as it is more robust to fluctuating water table and nutrient enriched conditions and can outcompete other typical bog species (Gatis *et al.* 2019). Dominance of *Molinia caerulea* in drained and restored bogs is often connected with high atmospheric N input (Limpens *et al.* 2003, Tomassen *et al.* 2003) because this species is tolerant to nutrient enriched conditions. In north-western Germany, atmospheric nitrogen deposition is usually high owing to the presence of extensive agricultural areas. The Leegmoor supposedly has an annual nitrogen input of 24 kg ha^{-1} (UBA 2019), which exceeds the critical load of 5–10 kg ha a^{-1} of a bog ecosystem and may affect the bog typical biota (Hurkuck *et al.* 2014). This leads us to ask how long the nutrient relocation can persist, as the storage capacity of the vegetation will be depleted at some point.

Soil PO_4^{3-}-P content was significantly higher in the upper (0–15 cm) soil layer (32.6 mg kg^{-1}) than in the lower (15–30 cm) soil layer (17.0 mg kg^{-1}), supporting the assumption that mineralisation is connected to the difference in soil moisture between the sampling depths. Purre & Ilomets (2018) found much higher phosphate content (average 45.7 mg

kg^{-1}) in the upper 5 cm of an extracted and restored bog in northern Estonia (Viru). In contrast, Zając *et al.* (2018) reported lower average values for the upper 10 cm, of 23.3 mg kg^{-1} in the sector of the site with lower soil moisture and 17.9 mg kg^{-1} in the sector with higher soil moisture. If comparability is assumed despite the differing sampling depths, soil PO$_4^{3-}$-P content at the Leegmoor is within the range displayed by other studied bogs. However, it is not possible to assess changes in soil PO$_4^{3-}$-P after nearly 40 years due to the lack of initial data.

The soil water data indicate a decrease of PO$_4^{3-}$-P levels. Also, PO$_4^{3-}$-P in soil solution followed the same trend of higher levels in the upper soil layer during the initial examination of the Leegmoor. If comparability of the data for PO$_4^{3-}$-P concentration in pore water is assumed, PO$_4^{3-}$-P declined by about threefold between 1990–1993 and 2019–2020 (Table 3). The concentrations in the pore water were, however, still high in comparison with other studies. Howson *et al.* (2021) determined 0.05 mg L^{-1} (as a mean of concentrations from 20–80 cm) in a near-natural bog in Scotland, Great Britain.

The decline is also reflected in the PO$_4^{3-}$-P concentration of the bog pond, that decreased slightly from 0.05 mg L^{-1} (Eggelsmann & Blankenburg 1989) to 0.03 mg L^{-1}, which is at the threshold of near-natural bogs (Bourbonniere 2009). In the earlier study by Eggelsmann & Blankenburg (1989), concentrations in the outflow (0.13 mg L^{-1}) were considerably higher than in the bog pond, pointing towards initial leaching. Nowadays, concentrations in the outflow are not elevated and PO$_4^{3-}$-P is no longer leached out. Concentrations at the regional reference sites (average 0.23 mg L^{-1}) and in the review study by DVGW (2020) (average 0.2 mg L^{-1}) are higher. Hence, the current low P concentrations at the Leegmoor indicate a development towards near-natural conditions.

The general decrease of plant available nutrients is mostly attributed to nutrient uptake by vegetation, especially for N species, while for PO$_4^{3-}$-P leaching initially played a role as well. As the vegetation cover has increased substantially since the initial examination, it can be assumed that uptake of nutrients by plants has exceeded peat mineralisation and atmospheric deposition. This is also supported by the findings of Nordbakken *et al.* (2003), who report uptake by plants of up to 90 % of deposited N with no increase of peat N content below 5 cm depth. Immobilisation of nutrients in peat can be ruled out because no peat growth was observed at the studied sites.

All in all, the results confirm our hypothesis that the amounts of plant available nutrients in soil and surface waters (except soil NH$_4^+$-N) have generally decreased since the initial examination, and that they have remained in or developed towards a near-natural state.

Effect of waterlogging and season on the amount of plant available nutrients

We further hypothesised that differences in water table depth within the study site caused different nutrient dynamics at the sampling sites, due to enhanced mineralisation in the more aerated soils. The waterlogging gradient showed negative correlations with pore water PO$_4^{3-}$-P concentration for the upper (0–15 cm) soil layer and with soil PO$_4^{3-}$-P content for both soil layers, indicating that mineralisation and P release are enhanced under drier soil conditions. Furthermore, our finding of declining PO$_4^{3-}$-P content with increasing depth below the soil surface show that the prevention of aeration by soil moisture is a strong controlling factor for soil PO$_4^{3-}$-P mineralisation. Similar findings were presented by Zając *et al.* (2018). This indicates, once again, that waterlogging is a major factor controlling the nutrient dynamics and that optimal water management can control nutrient availability. However, no correlation was found for water and N species, although mineralisation also causes N mobilisation. This is probably due to gaseous N losses via denitrification, which are reported to be 227.7 nmol N gDW^{-1} h^{-1} under ambient conditions in Whim Bog, Scotland (van den Elzen *et al.* 2018). Thus, the hypothesis that differences in water table depth between the sampling sites caused different nutrient dynamics due to enhanced mineralisation in more aerated soils was only partly confirmed, as it was restricted to PO$_4^{3-}$-P and excluded the N species that we studied.

The assumption of varying nutrient availability due to water table fluctuations over the year was consistent with Vitt *et al.* (1995), who found that ammonium and nitrogen in bog surface water were controlled by season and peaked under warmer conditions (in spring and summer), while soluble reactive phosphorous was constant throughout the observation period. However, we did not find a seasonal trend in any of the studied nutrients. These findings were supported by data from the initial examination (Gebhardt & Knabke 1994), which showed that interannual fluctuations were higher than seasonal changes. Nevertheless, the lack of seasonal trends is also a sign of near-natural nutrient conditions due to well-adjusted water levels that remain stable in summer. Based on these data, the hypothesis that the natural water table fluctuations during the year caused seasonal variability of nutrient dynamics has to be rejected.

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AUTHOR CONTRIBUTIONS

SN and LG conceptualised the study. Data collection, sample collection and analyses were performed by SN. All versions of the manuscript were written by SN and revised by LG. Both authors read and approved the final manuscript for submission.

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Appendix

Table A1. List of regional reference sites in near-natural or restored bogs. n = number of (surface water) samples.

site name	GPS coordinates	sampling date	n
Esterweger Dose	53° 3′ 15″ N, 7° 37′ 33″ E	09 July 2019	6
Vehnemoor	53° 3′ 25″ N, 7° 59′ 33″ E	16 July 2019	5
Everstenmoor	53° 7′ 13″ N, 8° 8′ 40″ E	16 July 2019	2
Wildes Moor	53° 1′ 50″ N, 7° 27′ 29″ E	30 July 2019	1

Table A2. Plant species recorded at the soil sampling sites on the Leegmoor during the initial examination in 1990–1993 (Gebhardt & Knabke 1994) and the follow-up examination in 2019–2020. For each sampling location, species are listed in descending order of abundance. N/A = data not available.

site no.	1990–1993	2019–2020
S1	no vegetation	*Molinia caerulea, Eriophorum vaginatum, Empetrum nigrum, Erica tetralix, Vaccinium vitis-idaea*
S2	N/A	*Calluna vulgaris, Erica tetralix, Eriophorum vaginatum, Betula pubescens*
S3	*Molinia caerulea*	*Molinia caerulea, Sphagnum cuspidatum, Betula pubescens*
S4	*Eriophorum vaginatum*	*Molinia caerulea, Sphagnum fimbriatum, Sphagnum cuspidatum, Eriophorum vaginatum, Betula pubescens*
S5	*Molinia caerulea*	*Molinia caerulea, Sphagnum fimbriatum, Sphagnum cuspidatum, Eriophorum vaginatum*
S6	*Sphagnum cuspidatum*	*Molinia caerulea, Sphagnum cuspidatum, Eriophorum vaginatum, Betula pubescens*
S7	*Eriophorum vaginatum*	*Sphagnum cuspidatum, Eriophorum vaginatum, Molinia caerulea, Betula pubescens*

Table A3. Soil bulk densities (Bd), from the initial examination in 1990–1993 (Gebhardt & Knabke 1994) and the follow-up examination in 2019–2020. N/A = data not available.

| site no. | 1990–1993 | | | 2019–2020 | | | | |
	soil layer (cm)	Bd (g cm⁻³)	n	soil horizon	depth (cm)	adopted for soil layer (cm)	Bd (g cm⁻³)	n
S1	0–15	0.19	N/A	hHv	0–20	0–15	0.26	3
	15–30	0.16	N/A	hHw1	20–40	15–30	0.19	3
S2	N/A			hHv	0–18	0–15	0.14	3
	N/A			hHw	18–50	15–30	0.11	3
S3	0–15	0.18	N/A	hHv	0–30	0–15	0.18	3
	15–30	0.15	N/A			15–30		
S4	0–15	0.19	N/A	hHv	0–45	0–15	0.20	3
	15–30	0.17	N/A			15–30		
S5	0–15	0.26	N/A	hHv	0–45	0–15	0.20	3
	15–30	0.19	N/A			15–30		
S6	0–15	0.18	N/A	hHw1	0–18	0–15	0.17	3
	15–30	0.11	N/A	hHw2	18–42	15–30	0.19	3
S7	0–15	0.19	N/A	hHw1	0–22	0–15	0.18	3
	15–30	0.13	N/A	hHw2	22–36	15–30	0.15	3

Table A4. Soil properties at the individual sampling sites, from the initial examination in 1990–1993 (Gebhardt & Knabke 1994) and the follow-up examination in 2019–2020. N/A = data not available.

| site no. | soil layer (cm) | NH_4^+-N (kg ha⁻¹) | | | | NO_3^--N (kg ha⁻¹) | | | | PO_4^{3-}-P (kg ha⁻¹) | | soil water content (vol. %) | | | |
| | | 1990–1993 | | 2019–2020 | | 1990–1993 | | 2019–2020 | | 2019–2020 | | 1990–1993 | | 2019–2020 | |
		mean	s	mean	s	mean	s	mean	s	mean	s	mean	s	mean	s
S1	0–15	20.9	12.0	5.4	8.6	2.4	2.2	0.0	0.0	4.2	0.9	69	15	77	5
	15–30	6..0	4.0	3.8	5.0	1.8	2.2	0.0	0.1	1.8	0.7	80	11	82	2
S2	0–15	N/A		6.0	4.9	N/A		0.1	0.2	3.6	1.9	N/A		83	2
	15–30	N/A		7.3	3.5	N/A		0.1	0.3	1.1	0.4	N/A		87	1
S3	0–15	2.0	1.4	2.8	2.3	0.4	0.6	0.0	0.0	4.2	1.6	62	9	78	5
	15–30	1.2	0.9	4.8	5.6	0.3	0.6	0.0	0.0	2.1	1.4	62	12	77	4
S4	0–15	3.4	4.8	2.5	2.6	0.5	0.7	0.0	0.0	2.9	1.1	75	9	78	5
	15–30	2.7	3.6	2.4	2.9	0.6	1.4	0.0	0.0	1.5	0.9	81	7	83	4
S5	0–15	7.5	7.8	3.2	2.6	0.8	1.4	0.0	0.0	1.9	0.6	71	11	82	3
	15–30	5.1	4.4	4.8	4.1	0.8	1.8	0.0	0.0	1.1	0.1	75	10	83	3
S6	0–15	2.2	1.4	1.8	1.0	0.6	1.1	0.0	0.0	2.6	1.9	85	5	77	4
	15–30	1.9	1.6	2.1	1.4	0.3	0.4	0.0	0.0	1.5	0.8	78	16	82	3
S7	0–15	2.5	1.6	4.1	2.5	1.0	0.9	0.0	0.0	1.2	0.1	81	7	85	3
	15–30	1.3	0.9	3.8	1.6	0.5	0.7	0.0	0.0	0.9	0.2	73	14	85	5

Table A5. Soil water PO_4^{3-}-P at the individual sampling sites, from the initial examination in 1990–1993 (Gebhardt & Knabke 1994) and the follow-up examination in 2019–2020. N/A = data not available.

site no.	soil layer (cm)	PO_4^{3-}-P (mg L^{-1})			
		1990–1993 (soil solution)		2019–2020 (pore water)	
		mean	s	mean	s
S1	0–15	0.14	0.15	N/A	
	15–30	0.23	0.27	N/A	
S2	0–15	N/A		0.21	N/A
	15–30	N/A		N/A	
S3	0–15	0.32	0.26	0.20	0.10
	15–30	0.30	0.26	0.16	0.00
S4	0–15	0.19	0.20	0.06	0.00
	15–30	0.05	0.07	N/A	
S5	0–15	0.59	0.41	0.07	0.04
	15–30	0.33	0.32	N/A	
S6	0–15	0.12	0.11	0.13	0.01
	15–30	0.15	0.16	N/A	
S7	0–15	0.26	0.23	0.03	0.01
	15–30	0.23	0.20	0.02	0.01

Table A6. Surface water nutrient concentrations at the individual sampling sites from the follow-up examination in 2019–2020.

site no.	NH_4^+-N (mg L^{-1})		NO_3^--N (mg L^{-1})		PO_4^{3-}-P (mg L^{-1})	
	mean	s	mean	s	mean	s
W1	0.70	0.25	0.03	0.05	0.03	0.01
W2	0.37	0.22	0.00	0.00	0.03	0.01
W3	0.24	0.11	0.01	0.01	0.01	0.00
W4	0.60	0.70	0.00	0.01	0.01	0.01
WS4	0.24	0.10	0.00	0.01	0.02	0.01
WS5	0.18	0.14	0.00	0.00	0.02	0.01
WS6	0.22	0.13	0.00	0.00	0.03	0.02
WS7	0.25	0.33	0.00	0.00	0.03	0.02