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Restoring natural seepage conditions on former agricultural grasslands does not lead to reduction of organic matter decomposition and soil nutrient dynamics

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Abstract. In the central part of the Netherlands, wetland restoration projects involve the rewetting of former agricultural land, where low water levels were artificially maintained (polders). Many of these projects do not result in the expected reduction of nitrogen and phosphorus availability and subsequent re-establishment of a diverse wetland vegetation. The aim of the present study was to investigate which mechanisms are responsible for this lack of success. Thereto, we studied the effect of rewetting of former agricultural grasslands on acidified peat soil (pH = 3.5) on organic matter decomposition, nitrogen cycling and phosphorus availability in the soil for three seasons. To provide an explanation for the observed effects, we simultaneously studied a set of potentially controlling abiotic soil conditions that were expected to change after rewetting. It was found that rewetting of these grasslands with natural, unpolluted seepage water did not affect nitrogen cycling, but raised decomposition rates and almost doubled phosphorus availability. The main cause of these effects is a raise of soil pH to about 7 due to the hydrochemical composition of the soil pore water after rewetting, which reflects groundwater with high amounts of buffering ions. This effect overruled any reduction in process rates by the lowered soil redox potential. The counterintuitive finding of eutrophication after rewetting with natural and unpolluted water is considered to represent a new form of internal eutrophication, triggered by the restoration of natural site conditions of former agricultural land on acid peat soil.

Introduction

During the last decades, anthropogenic pressure on natural and semi-natural wetland ecosystems has increased considerably, mainly because of expanding human activity and associated claims on space. Especially in densely populated areas in Western Europe like the Netherlands, many wetland ecosystems are threatened by desiccation, pollution and eutrophication (Berendse et al. 1994; Beltman et al. 1996; Lamers et al. 2002). This is a serious problem, because these ecosystems have become more and more valued because of their large biodiversity, water purifying function, water storage capacity and their role as a major carbon sink in the global carbon cycle (Howard-Williams 1985; Verhoeven 1992; Oomes et al. 1996; Schlesinger 1997). Therefore, many protection and restoration projects have been set up for the conservation of these ecosystems. Natural wetland soils are generally characterised by low nutrient availability and low rates of soil processes as a result of anoxic conditions in the soil (McLatchey and Reddy 1998; Aerts et al. 1999). It is a common understanding that low nutrient availability limits plant productivity and therefore has a large impact on species composition, often favouring rare plant species (Grime 2001). Nitrogen and phosphorus are generally considered to be the most important nutrients limiting plant production in wetlands (Verhoeven et al. 1996; Aerts and Chapin 2000). Most wetland restoration projects therefore focus on restoring anoxic conditions in the soil by rewetting and thereby attempting to reduce rates of nitrogen and phosphorus cycling in order to create favourable conditions for regrowth of typical wetland plant species. Previous studies on wetland restoration have stressed the need for the restoration of both water quantity and water quality to prevent (internal) eutrophication (e.g. Verhoeven et al. 1988; Roelofs 1991; Lamers et al. 2002).

In the central part of the Netherlands, many wetland restoration projects involve the rewetting of former agricultural land, where low water levels were artificially maintained (polders). Because of the low position of these lands in the landscape, rewetting is easily achieved by abandoning active water level management, thereby restoring the naturally occurring discharge of base-rich groundwater. In this way, not only the water quantity, but also the water quality is restored. Many wetland restoration projects, however, do not result in the expected reduction of nitrogen and phosphorus availability and subsequent re-establishment of a diverse wetland vegetation (Beltman et al. 1995; Lamers et al. 2002). This stresses the need for a better understanding of the biogeochemical processes that operate in rewetted soils and the soil conditions that control them in order to predict the effects of different wetland restoration measures.

The objectives of our study were: (1) to test whether the rewetting of former agricultural land by restoration of seepage conditions results in favourable conditions for wetland vegetation regrowth by a reduction of decomposition and nitrogen and phosphorus dynamics; (2) to identify key factors that control nutrient availability during rewetting in order to provide a mechanistic explanation for the observations under 1.

To reach our first objective, we studied decomposition rates of soil organic matter and nitrogen and phosphorus dynamics in four rewetted areas and an untreated control area, which were all formerly in use for cattle breeding. These research sites are grasslands on peat soil in which the soil pH is relatively low (about 4). To reach our second objective, the effect of rewetting on potentially controlling abiotic soil conditions was studied together with the processes mentioned above. To separate effects of rewetting from the effects the disappearance of cattle from the studied sites (stop of grazing and manuring), we used a control area that was taken out of agricultural use at the same time that rewetting measures were applied in the restored areas.

Material and methods

Study area and research set-up

This study was carried out in the Vechtstreek, an extensive lowland peat area in the central part of the Netherlands in which valuable wetland nature reserve areas are alternated with large agricultural areas. The area is fed with minerotrophic groundwater from an ice pushed ridge which borders the area in the east.

Rewetting projects were set up in the Horstermeerpolder (52°14′N, 05°04′E) and the polder Bovenmeent (52°17′N, 05°07′E) by the Dutch nature management organisation Natuurmonumenten in 1998. The Horstermeerpolder is located about 2.5 m below sea level, while the polder Bovenmeent is located at 1.5 m below sea level. The soils of both areas can be characterised as a clayey peat and have a peat layer of about 2 m thickness. Rewetting was established by hydrologically isolating the restoration areas from the polders in which an artificial low groundwater level is maintained. After isolation, the natural seepage conditions of the areas caused a raise in groundwater level.

To be able to study the effects of rewetting, five research sites were selected: one dry control site and four rewetted sites. The dry control site and three of the rewetted sites were located in the Horstermeerpolder, the other rewetted site was located in the polder Bovenmeent. All sites had been in use for cattle grazing until 1998, when groundwater levels were raised to prevent water loss from adjacent wetland nature areas, except for the control site, which was taken out of use for cattle grazing without raising the groundwater level.

Within each site, 10 replicate sampling points were selected at random. All measurements described below were done at each of the sampling points to enable correlation analysis between abiotic conditions and process rates. Sampling was carried out over three seasons to account for seasonal variability. Sampling periods were May–June 2000, November–December 2000 and June 2001, further designated as spring 2000 (or 2001) and winter 2000. The spring periods were chosen to coincide with the growing peak for grassland areas (den Held and den Held 1976; Oomes 1992). The winter period was chosen to represent the peak in availability of fresh organic material.

Water variables

At all sites PVC piezometers with a screen length of 1 m were installed to a depth of 1.2 m. Groundwater levels were recorded hourly with dataloggers. Because of limited availability of dataloggers, groundwater level for one site was measured manually each week during the study period. In spring 2001, groundwater levels were measured manually each week for all sites.

Soil pore water was sampled with porous ceramic suction cups. Soil pore water was filtered over 0.45 μm Sleicher & Schull filters immediately after sampling. After filtering, 50 mL of the sample was acidified with 0.4 mL concentrated HNO_3 for cation analysis and analysis of NH_4^+, PO_4^{3-} and Si. The rest of the sample was used for anion analysis. Electroconductivity (EC) and pH were measured in the field immediately after filtering, using a WTW field kit. Water samples were kept at 4 °C until further analysis. Alkalinity was measured by titrating part of the sample down to pH 4.2 with H_2SO_4. NH_4^+, NO_3^-, PO_4^{3-}, Cl^- and SO_4^{2-} were measured on a Skalar continuous flow analyser, Na^+ and K^+ were measured on a Dionex ion chromatograph and other ions were measured on a Varian ICP. Total phenolics were measured colorimetrically, using Folin–Ciocalteaus reagent, as described in Box (1983).

A Van Wirdum classification of soil pore water, based on the ratio between Ca^{2+} and Cl^- and electroconductivity (EC), was made to study the origin of the soil pore water (van Wirdum 1991). The method classifies water samples relative to standard water samples for atmospheric water (atmocline), groundwater (lithocline) and seawater (thalassocline).

Soil variables

Soil pH was measured using a WTW pH electrode in the soil–KCl extracts that were used for determination of extractable nitrogen (see below).

Redox potential was determined at 5 cm depth with custom made platinum electrodes, a standard reference electrode and a standard millivoltmeter (Digitool digi-38). Platinum electrodes were placed into the soil one week before the actual measurement for stabilisation. Values were converted to the potential relative to the normal hydrogen reference electrode (Eh).

Extractable NH_4^+ and NO_3^- were measured in extracts of 3 g fresh weight soil (comparable to 1 g dry weight) in 25 mL of 1 M KCl, except for spring 2000, when 0.01 M $CaCl_2$ extracts were used, with a Skalar continuous flow analyser. The different extractant was used to achieve a higher yield.

Extractable PO_4^{3-} was measured colorimetrically with the phosphomolybdenum blue complex method in ammonium lactate–acetic acid soil extracts (Houba et al. 1995).

Soil processes

Decomposition rate was measured both at 0–10 cm depth and 10–20 cm depth, using the cotton-strip assay as described extensively in Harrison et al. (1988). At each sampling point, five replicate cotton-strips were inserted into the soil to 20 cm depth, one strip was immediately retrieved from the soil to serve as a control. Every following week, one of the replicate strips was retrieved from the field, to be able to monitor decay rates over time. Cotton-strips were inserted into the soil using a custom made stainless steel plate. To prevent damage to the strips by the force of insertion, a slit was cut into the soil with the insertion plate prior to insertion of the strip. Insertion was performed ensuring good contact of the cotton-strip with the surrounding soil, which is essential for the reliability of the method. Tensile strength was measured on a MECMESIN AFG1000N Advanced Force Gauge. Decomposition rates are expressed as percentage tensile strength loss (relative to the control strips) per day, during an incubation time after which all strips still had a remaining tensile strength above 30%.

N and P mineralisation were measured using a modified buried bag method (Raison et al. 1997). Duplicate soil cores of 10 cm length and 4.5 cm diameter were taken at each sampling location, using an stainless steel corer. One core was immediately transported to the lab and treated as described below. Cores for incubation were sealed in the RVS corer to prevent leaching, and incubated in the soil at its sampling location for 4 weeks. Holes were present in the top of the corer to allow gas exchange. After retrieval of the cores, coarse roots were removed from the sample, the sample was mixed thoroughly and subsamples were taken for dry weight determination and extraction and analysis of NH_4^+, NO_3^- and PO_4^{3-}. All measurements were done as described above. The difference between concentrations of the initial core and the incubated core was divided by the incubation time to calculate process rates.

Statistical analysis

The study design is considered a nested design with rewetting nested within site. Data were tested for the effect of rewetting and season, using a two-way nested ANOVA. Data was checked for equality of error variance using Levene's test and for normality with Shapiro–Wilk's test. Data with unequal error variance or a non-normal distribution were logtransformed before analysis. If transformation did not lead to equal error variance or normality, the untransformed data were used in the ANOVA and the effects of rewetting and season on these variables were tested separately with non-parametric analyses, using the Mann–Whitney *U*-test for the effect of rewetting and the Kruskal–Wallis *H*-test for the effect of season. The non-parametric tests all showed the same patterns as the nested ANOVA, except for net ammonification. Because the nested two-way ANOVA design is considered the most appropriate for representing our research set-up, only the results from this test are shown here. Because of the use of a different extractant for the extraction of NH_4^+ and NO_3^- in spring 2000 (see above), the effects of rewetting on extractable field concentrations and production rates of NH_4^+, NO_3^- and total inorganic N were analysed separately for spring 2000.

Identification of key factors

To identify key factors controlling process rates, we used a set of ordination analyses following Brown et al. (1993). First, we analysed the soil process data independently of the process data using principle component analysis (PCA). Second, we used redundancy analysis (RDA) to identify key factors from the environmental data that best explain the observed variation in the measured processes. RDA extracts patterns in processes from the variation explained by environmental factors only (direct gradient analysis), while PCA extracts patterns from all variation (indirect gradient analysis). Because it is constrained to the environmental data, RDA is generally less powerful in explaining the observed variation in process rates than PCA. However, when the measured set of environmental variables is sufficient to explain the observed variation in process rates, the results of both analyses will not differ greatly.

Based on their correlation with the extracted axes in RDA, the separate environmental variables can be grouped based on their influence on the measured process rates. These aggregates can provide more information than the effects of the separate environmental variables, especially because many soil and hydrochemical variables can show a large degree of colinearity.

Because of the large number of interaction effects between rewetting and season as observed in the ANOVA, we analysed the data separately for each season. Data were tested for normality using Shapiro–Wilk's test. Data with non-normal distribution were logtransformed before analysis. Scaling was focussed on inter-species correlations and species scores were not post-transformed. To account for the differences in units of measurement for the different variables, the species data were centered and standardised. All ordination analyses were performed using Canoco 4.5 (Ter Braak and Smilauer 2002). Correlations between environmental variables and the main axes of the ordination analyses were performed in SPSS using Pearson's correlation test. All ordination analyses showed a relatively high explanatory power of the first two axes and resulted in a very large number of significant correlations between axes and measured variables. For the identification of key factors we used a significance level for correlations of $p \leq 0.01$.

More information concerning details of ordination analyses can be found in Jongman et al. (1995) and Ter Braak and Smilauer (2002).

Results

Water variables

Groundwater levels were generally higher at the rewetted sites, but the difference in groundwater levels between control and rewetted sites varied over time (Figure 1). In winter, groundwater levels were higher for all sites.

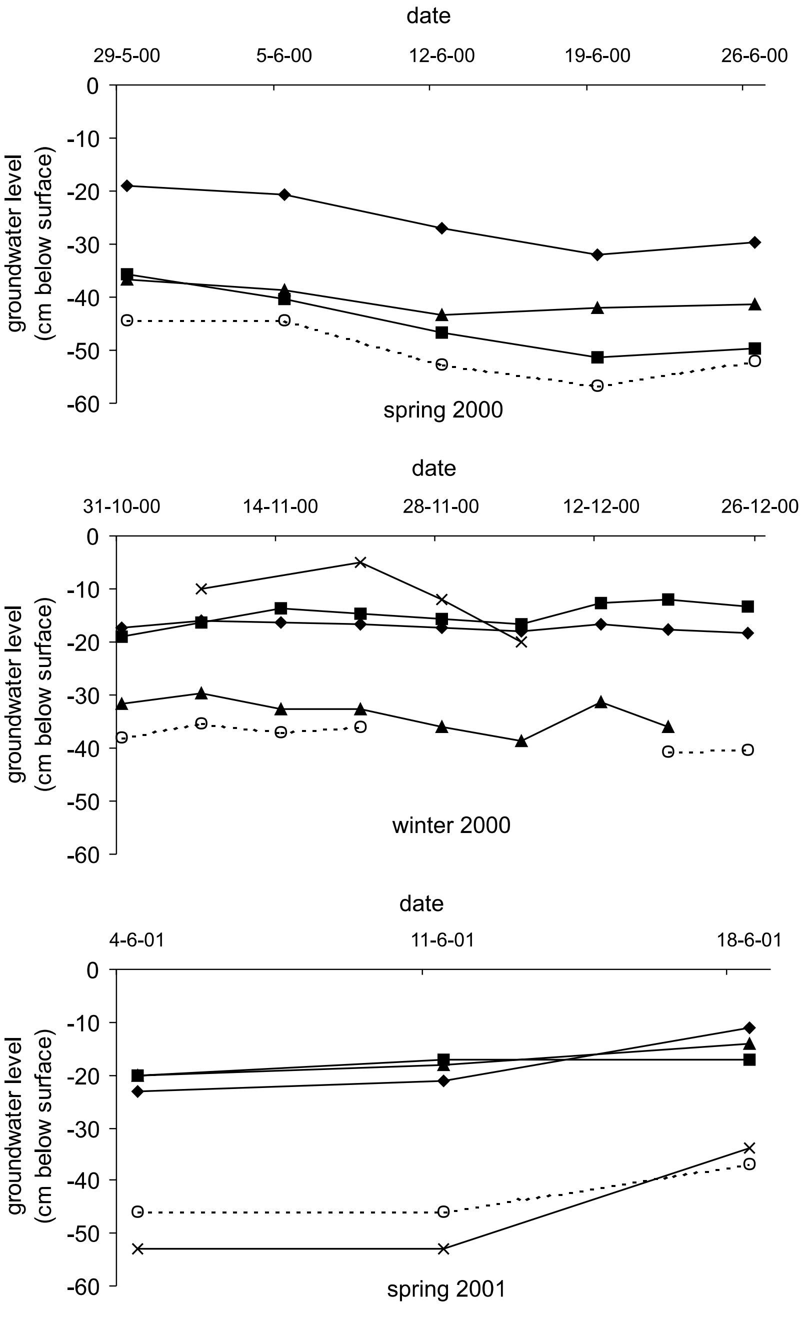


Figure 1. Groundwater levels of the research sites for the three studied seasons. Groundwater level of the control site is represented by the dotted line, solid lines with different symbols represent the different rewetted sites. Groundwater level is given in cm relative to the soil surface.

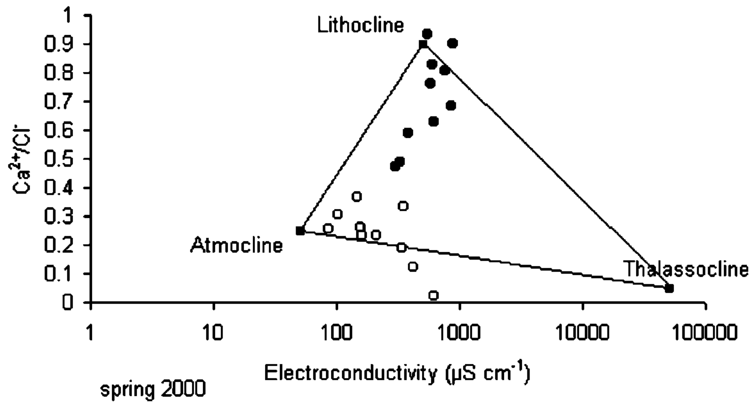
The Van Wirdum classification of soil pore water indicates that the soil pore water of rewetted sites reflected a lithocline composition, whereas the soil pore water in the control site reflected an atmocline composition (Figure 2). Thus, the soil pore water in the control site and the rewetted sites had a different origin, resulting in a different hydrochemical composition. This was the case for all seasons. An overview of the most important changes in hydrochemical characteristics of the soil pore water is given in Figure 3. Rewetting resulted in a significant rise in pH of the soil pore water from about 5 to 6.5. Soil pore water pH was significantly higher in winter (almost 6 for the control site and about 7 for the rewetted sites). Rewetting resulted in a significant raise of EC but the effect of rewetting depended on sampling season Table 1, Figure 3). EC was significantly higher in winter. Furthermore, rewetting resulted in a significant raise of concentrations of Ca^{2+}, Mg^{2+}, $Fe^{tot.}$ and $Mn^{tot.}$, ions indicative of a minerotrophic groundwater origin of the soil pore water. Ca^{2+}, Mg^{2+} and $Mn^{tot.}$ were significantly higher in winter. The effect of rewetting on Ca^{2+} and $Fe^{tot.}$ concentrations was dependent on sampling season (Table 1). Alkalinity of the soil pore water was significantly raised by rewetting and was significantly higher in winter. The effect of rewetting on alkalinity depended on sampling season (Figure 3, Table 1). Al^{3+} concentrations were significantly reduced by rewetting (Table 1, Figure 3). Concentrations in winter were significantly higher than in spring 2000, but did not significantly differ from the concentrations in spring 2001. Rewetting resulted in significantly higher concentrations of ${NH_4}^+$ in the soil pore water (Table 1). ${NO_3}^-$ concentrations, however, were significantly reduced by rewetting, but this effect depended on sampling season (Table 1). ${NO_3}^-$ concentrations were significantly lower in winter. ${PO_4}^{3-}$ concentrations in the soil pore water were significantly raised by rewetting and were significantly higher in winter (Table 1). Rewetting did not result in significant changes in soil pore water phenolics concentration (Table 1).

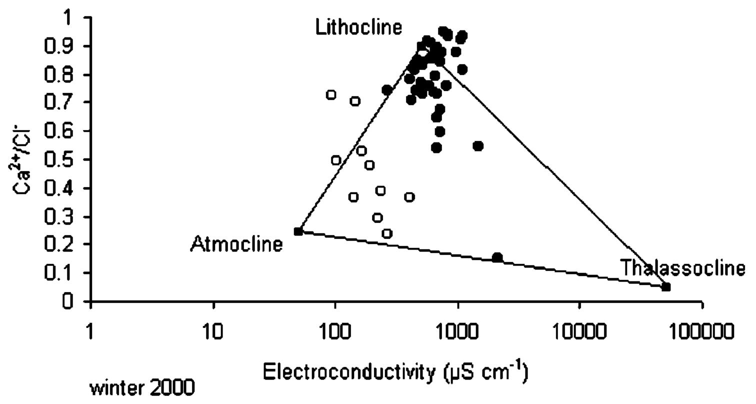
Soil variables

Rewetting resulted in a significant and strong rise in soil pH from about 3.5 to 6.5 (Figure 4, Table 1). Rewetting significantly lowered the redox potential at 5 cm soil depth (Figure 4, Table 1). The effect of rewetting on soil redox potential was dependent on sampling season. Rewetting resulted in significantly lower extractable amounts of ${NH_4}^+$ independent of the extraction medium (Figure 4, Table 1).

Process rates

Rewetting significantly raised overall decomposition rates as measured with the cotton-strip assay, both at 0–10 cm and 10–20 cm depth (Figure 5, Table 1). The effect of rewetting on decomposition rates at both depths depended on





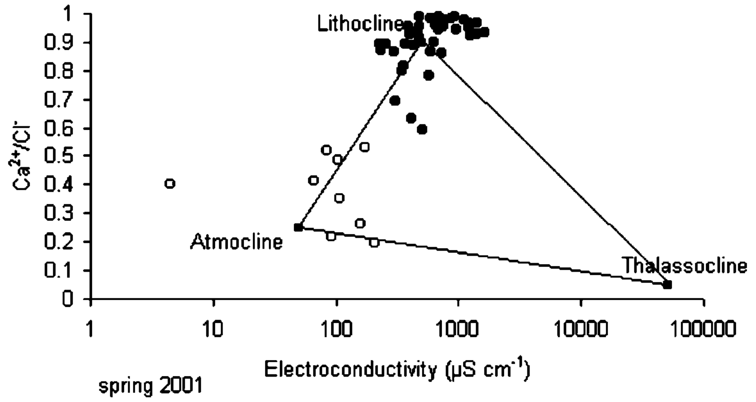


Figure 2. Van Wirdum plots of soil pore water samples for all seasons. Samples from the control site are represented by open dots, samples from the rewetted sites are represented by solid dots. Ca^{2+}/Cl^{-} is the ratio between Ca^{2+} and Cl^{-}, also called the Ionic Ratio. Position of the samples is given relative to standard reference samples of atmospheric water (atmocline), groundwater (lithocline) and seawater (thalassocline).

Table 1. Results of a nested two-way ANOVA of the effects of rewetting, sampling season and their interaction. Rewetting is nested within sampling site

	Rewetting	Season	Interaction
Water variables			
pH	***	***	
EC	***	**	**
Na	**	***	
K	*	***	
Mg	***	***	
Ca	***	***	***
NH_4	***		
Cl		***	
HCO_3	***	***	***
SO_4	***	**	***
NO_3	***	***	*
PO_4	*	***	
Si	***	***	
Al	***	*	
Fe^{tot}	***		**
Mn^{tot}	***	***	
Soil variables			
Soil redox potential	***	**	*
Soil pH	***	***	
Extractable NH_4^+	***	***	***
Extractable NO_3^-	**	**	
Soil processes			
CTSL 10	***	***	**
CTSL 20	***	***	***
N mineralisation			
Net nitrification		***	
Net ammonification	***		

*$p \leq 0.05$, **$p \leq 0.01$, ***$p \leq 0.001$.

sampling season, as shown by the significant interaction (Table 1). Decomposition rates at both depths were significantly lower in winter and the rates in spring 2000 were higher than in spring 2001. Rewetting did not lead to significant changes in net nitrification, but nitrification rates were significantly lower in winter, when there was a net loss of nitrate at both control and rewetted sites. Rewetting led to significantly lower net ammonification, but did not lead to significant changes in net nitrogen mineralisation rates and rates did not differ between seasons. Extractable PO_4^{3-} was significantly raised by rewetting.

Identification of key factors

The high process-environment correlation coefficients of the first two axis of the RDA (Table 3) and the relatively little loss of explanatory power compared

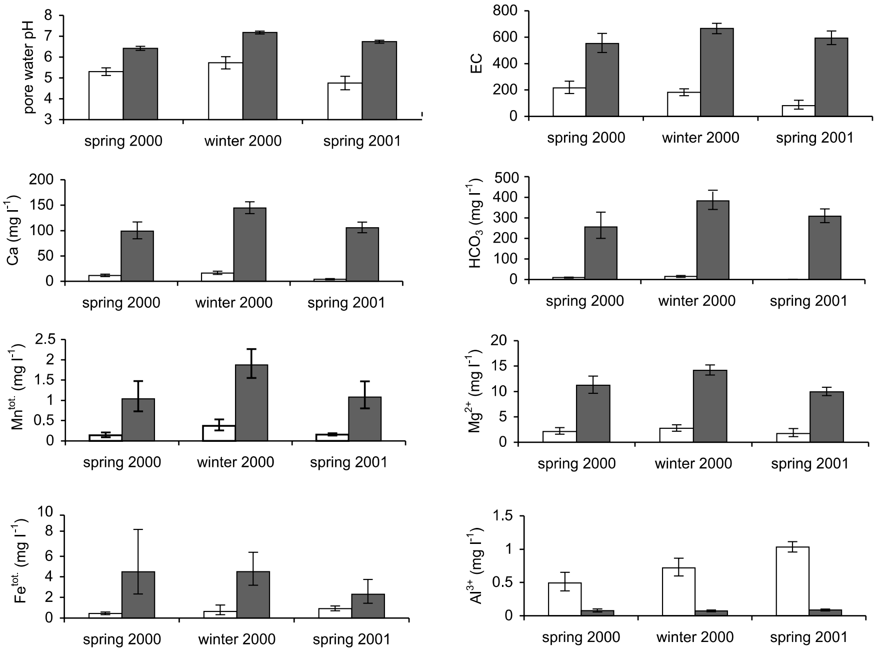


Figure 3. Selection of soil pore water characteristics of control (open bars) and rewetted (shaded bars) sites for all seasons. Concentrations are given in mg L^{-1}. Error bars represent standard error.

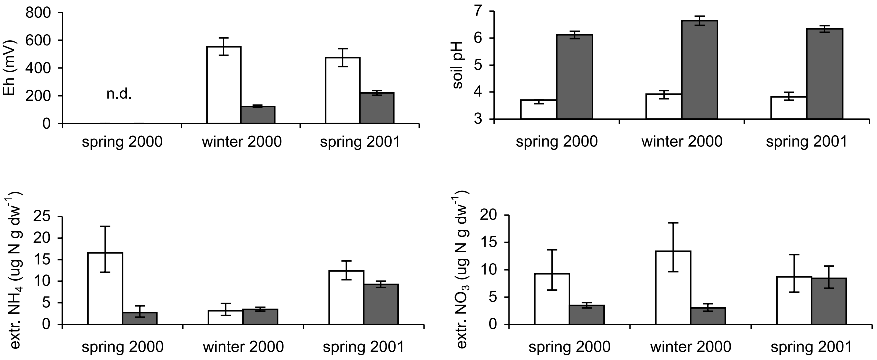


Figure 4. Soil redox potential (Eh), soil pH and extractable NH_4^+ and NO_3^- of control (open bars) and rewetted (shaded bars) sites for all seasons. N.d. indicates missing data. Error bars represent standard error. Eh is given in mV, extractable amounts of NH_4^+ and NO_3^- in μg N g^{-1} dry weight.

to the PCA of processes (Table 2) indicate that the set of environmental variables we measured is sufficient to account for a major part of the variation of the observed process rates.

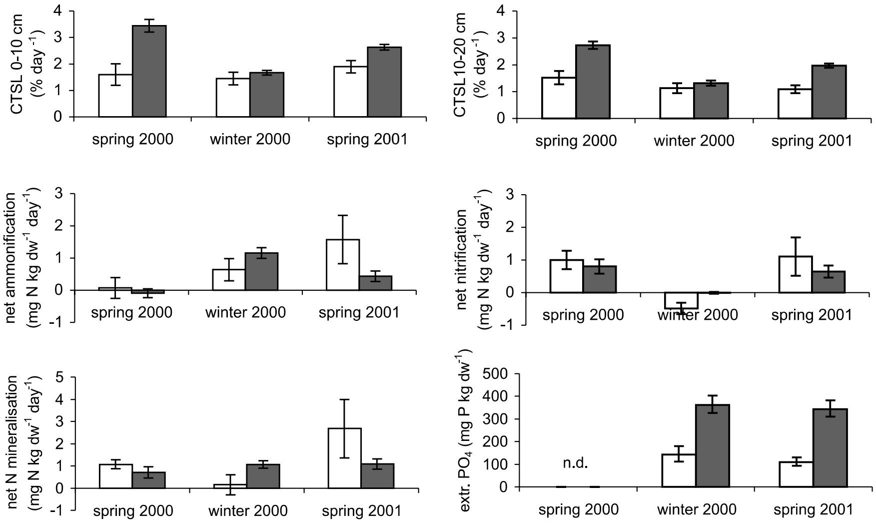


Figure 5. Soil process rates and phosphorus availability of control (open bars) and rewetted (shaded bars) sites for all seasons. Decomposition rates are given in percentage cotton-strip tensile strength loss (CTSL) per day. Net ammonification, net nitrification and net nitrogen mineralisation are given in mg N kg^{-1} dry weight day^{-1}. Error bars represent standard error.

Table 2. Percentage variance explained by the first two axes of Principal Component Analysis (PCA) and Redundancy Analysis (RDA) for all sampling seasons

	Variance explained by			
Season	PCA axis 1	PCA axis 2	RDA axis 1	RDA axis 2
Spring 2000	38.0%	28.7%	25.9%	16.9%
Winter 2000	37.6%	28.3%	24.8%	21.5%
Spring 2001	43.1%	26.7%	24.9%	15.7%

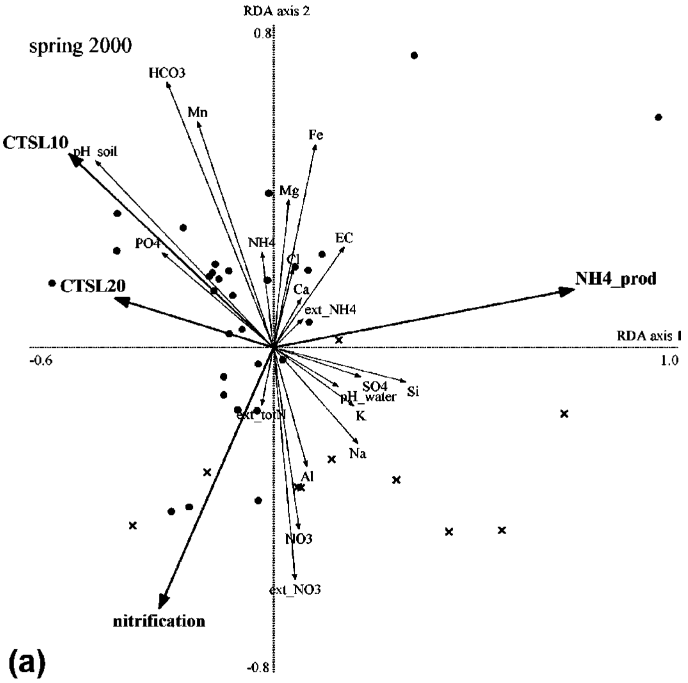
Furthermore, Figure 6a–c show that in all seasons the sampling locations from the rewetted sites (solid dots) are clearly separated from the sampling locations from the control site, indicating clear differences in process rates between the sites.

The direction of the arrows representing environmental variables are determined by the correlation of the variable with the different soil processes (Jongman et al. 1995). To investigate which (aggregates of) environmental variables are the main cause for the observed differences in process rates, correlations of the environmental variables with the main RDA axes were calculated and are summarized in Table 3. The results show that in spring 2000, the first axis was only significantly correlated with soil pH, the second axis was correlated with indicators of soil acidity and indicators of a groundwater origin

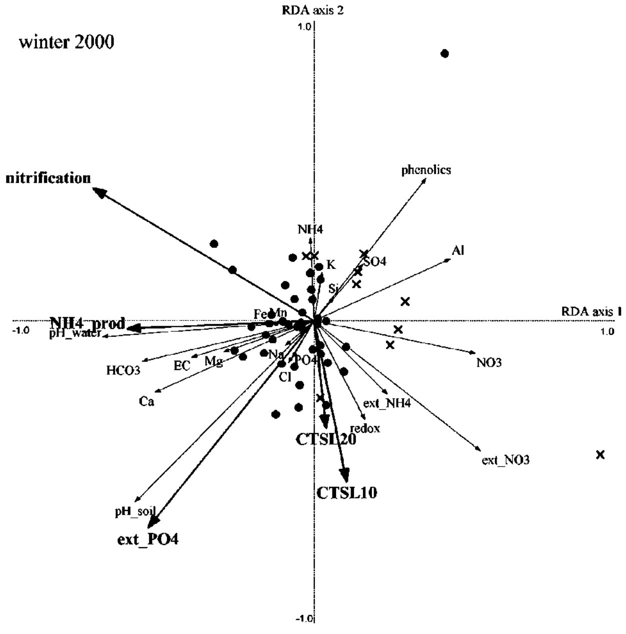
Table 3. Summary of the RDA of the total data for all seasons. Only variables that were significantly ($p \leq ; 0.01$) correlated with the axes are shown. Correlation coefficients are given in brackets

	Axis 1	Axis 2
Spring 2000		
Species–environment correlations	0.84	0.77
Correlated with	Soil pH (−0.37)	Mg (0.73) Ca (0.72) Cl (0.71) HCO_3 (0.71) EC (0.68) Al (−0.65) Fe^{tot} (0.59) Mn^{tot} (0.58) Extractable NO_3 (−0.44)
Winter 2000		
Species–environment correlations	0.92	0.82
Correlated with	Redox (0.59) Soil pH (−0.47) Pore water pH (−0.65) EC (−0.38) Ca (−0.49) HCO_3 (−0.53) NO_3 (0.50) Al (0.42) Extractable NO_3 (0.62)	–
Spring 2001		
Species–environment correlations	0.77	0.77
Correlated with	Soil pH (−0.51) Pore water pH (−0.48) Mg (0.39) HCO_3 (0.48) Al (−0.47) Phenolics (−0.43) Extractable NH_4 (−0.41) Extractable NO_3 (0.33)	Ca (−0.40)

of the soil pore water. The variation in decomposition at both soil depths was determined by both ordination axes, variation in net nitrification was mainly determined by the second axis and the variation in net ammonification was determined by the first axis. Thus, in spring 2000, the observed variation in


spring 2000
RDA axis 2
0.8
-0.8
RDA axis 1
-0.6
1.0
HCO3
Mn
Fe
Mg
CTSL10
pH soil
PO4
NH4
Cl
EC
Ca
ext_NH4
CTSL20
NH4_prod
SO4
pH_water
Si
K
Na
ext_totN
Al
NO3
ext_NO3
nitrification

(a)


winter 2000
RDA axis 2
1.0
-1.0
RDA axis 1
-1.0
1.0
nitrification
phenolics
NH4
K
SO4
Al
Si
Fe
Mn
NH4_prod
pH_water
HCO3
EC
Mg
Na
PO4
Cl
Ca
NO3
ext_NH4
redox
CTSL20
ext_NO3
CTSL10
pH_soil
ext_PO4

(b)

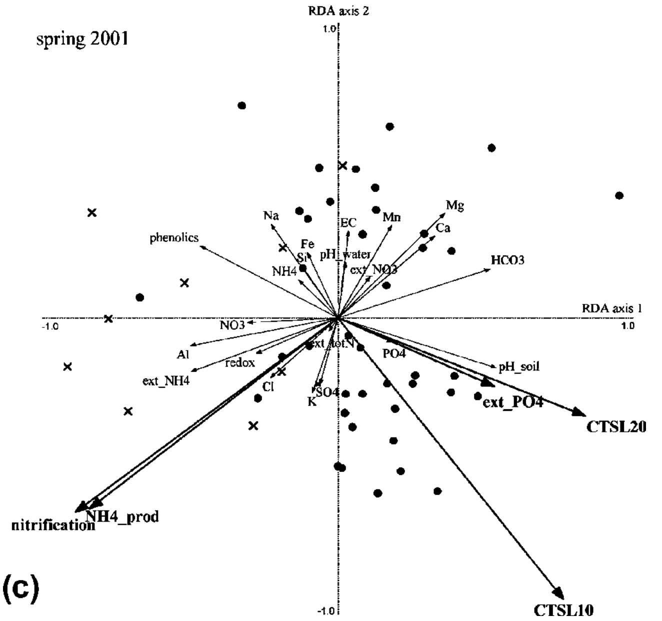


Figure 6. (a) First two ordination axes of the RDA of (a) spring 2000; (b) winter 2000; (c) spring 2001. Samples from the rewetted site are indicated with a solid dot, samples from the control site are indicated with an x. The response variables (soil processes) are indicated in bold. CTSL10 = cotton-strip decomposition at 0–10 cm soil depth, CTSL 20 = cotton-strip decomposition at 10–20 cm soil depth, NH4_prod = net ammonification, ext_ PO_4 = P availability.

process rates was mainly determined by soil acidity and the origin of the soil pore water.

In winter 2000, the first axis was highly correlated with soil redox potential, indicators for soil acidity and nitrogen availability. No correlations at the $p \leq 0.01$ level between the second ordination axis and environmental variables were found. The second axis was correlated at the $p \leq 0.05$ level with soil redox potential ($r = 0.28$) and extractable nitrate ($r = -0.37$). Variation in decomposition at both depths was determined by the second axis, net ammonification by the first axis and net nitrification and P-availability by both the first and the second axis. Thus, in winter 2000, the observed variation in process rates was not only determined by soil acidity, but also by soil redox potential and nitrogen availability.

In spring 2001, the first ordination axis was correlated with indicators of soil acidity and nitrogen availability. The second axis was only correlated with Ca^{2+} concentrations of the soil pore water. Variation in decomposition at both soil depths was determined by both ordination axes as was the variation in both net ammonification and net nitrification. P-availability was only correlated to the first axis. Thus, the observed variation in process rates were determined by soil acidity, the origin of the soil pore water and nitrogen availability.

Discussion

Environmental variables

Our results show that rewetting measures led to the expected increase in groundwater levels, although seasonal variation occasionally led to low groundwater levels even in the rewetted sites. Despite this seasonal variation, rewetting consequently led to lower soil redox potentials. In addition to this, we also found a significant change in soil pore water chemistry in the rewetted sites, associated with a change in the origin of this water. The control site receives its water mainly from rainwater input, which contains little dissolved ions and therefore has a limited buffering capacity. This lack of buffering capacity leads to a very low soil pH (around pH 3.5), for acidifying components produced by degradation processes are not neutralised. This is in contrast with the rewetted sites, which receive their water from the restored groundwater flow. This groundwater contains large amounts of calcium and bicarbonate and is well-buffered in the neutral region, leading to a near neutral soil pH in the rewetted sites.

This dramatic raise in soil pH after rewetting played a dominant role in determining soil process rates and nutrient availability. The effects were found to be different for the different soil processes under study and are discussed below.

Decomposition

Decomposition of organic material, the main process controlling nutrient availability in wetlands, was significantly stimulated by rewetting. The decomposition rates measured in the control site lie within the range of decomposition rates of natural herbaceous rich fens as reported by Verhoeven et al. (1996), but rewetting increased these rates with almost 50%. This effect was dependent on sampling season. Berendse et al. (1994) found similar patterns of decomposition in a study of rewetted humic clay grasslands. They attributed the higher decomposition rates observed in rewetted sites and its dependence on season to water limitation of the soil microbial community in the control treatment in spring. However, we did not find any indications that this has been the case in the present study. Contrary to the study of Berendse et al. (1994), we did not find higher rates of decomposition in the lower soil layer in the control site, where water is readily available throughout the year, nor did we find soil moisture contents lower than 64% in the control site (data not shown). This relatively high soil moisture content, combined with careful insertion of the cotton-strips, also excludes the possibility that changes in the physical contact between soil and cotton-strips as a result of the rewetting process could have led to the observed differences in decomposition rates. Instead, our analyses show that decomposition rates were mainly controlled by the hydrochemical composition of the soil pore water and the associated soil pH, leading to higher decomposition rates at the rewetted sites. Kok et al. (1990) and Roelofs (1991) showed that decomposition rates in acid lakes and acid lake sediments were stimulated when pH and buffering capacity were raised by the inlet of alkaline river water and this mechanism was also proposed to be effective in studies on rewetted grasslands by Oomes et al. (1996, 1997). With the results of the present study, we are able to confirm this hypothesis.

Oomes et al. (1996) expected the effect of a change in water quality on decomposition and nutrient cycling to depend greatly on local site characteristics such as depth of the water table, soil characteristics and water quality. Our results support the idea that the effect of the change in hydrochemical composition of the pore water and soil pH can be overruled by changes in the depth of the water table and the associated changes in soil redox potential. In winter, when the soil of the rewetted sites was completely waterlogged, we found that decomposition rates in these sites were reduced to levels comparable to the control site. This happened in spite of the fact that differences in hydrochemistry and soil pH were comparable to those in both spring seasons. This increased effect of water level can be attributed to the soil redox potential of the rewetted sites, which further decreased in winter. The increased importance of redox potential is supported by our RDA, in which redox potential is identified as an additional factor controlling decomposition rates only in winter.

Nitrogen cycling

Contrary to decomposition rates, net nitrification and net nitrogen mineralisation rates were not significantly affected by rewetting. Net nitrogen mineralisation rates of both rewetted and control sites were comparable to rates reported by Beltman et al. (1996) for natural rich fens in a similar area and were also comparable to rates reported for rewetted grassland mesocosms by Oomes et al. (1997).

Based on the increased decomposition rates, it was expected that nitrogen mineralisation would also be increased by rewetting. It is unlikely that increased denitrification losses, which were not measured in this study, nullified higher mineralisation rates at the rewetted sites. A study in which soil from both rewetted and control sites was incubated anaerobically with and without nitrate addition (Van Dijk et al., unpublished data), showed that denitrification in the rewetted sites is inhibited by nitrate limitation as was also found by Aerts and de Caluwe (1999) with soils from a study site close to the present one. Soil columns from the field sites of the present study that were incubated in greenhouse study under moisture conditions comparable to the field situation also showed no denitrifying activity (Van Dijk et al., in preparation).

Phosphorus availability

Phosphorus availability at the control site was within the range reported for meadows and fens on peat soil by Olde Venterink et al. (2002), but rewetting resulted in a doubling of phosphorus availability. Again, this increase could be attributed to the strong rise in soil pH associated with the groundwater origin of the soil pore water. The large amount of sorbed phosphorus in the soil is the result of decades of agricultural use of these grasslands, during which high amounts of phosphorus have been applied by manuring. Because of the acidity and high organic matter content of this soil, much of this phosphorus will have been immobilised in the soil in complexed or chelated form. Many of these complexes become unstable when soil pH is raised, thereby releasing phosphate (Stumm and Morgan 1970; Ponnamperuma 1972).

Identification of key factors

As already mentioned above, rewetting not only led to a change in water quantity, but also in water quality. Our analyses show that the observed changes in soil pore water chemistry and pH, associated with the change in water origin, are the main factors contributing to the observed rise in decomposition rates and phosphorus availability. Water quality effects leading to increased decomposition rates and increased nutrient availability described in many wetland restoration studies are generally attributed to high nutrient loadings or to internal eutrophication caused by high iron and sulphate concentrations of (polluted) river water that is used in the restoration process (e.g. Roelofs 1991; Koerselman et al. 1993; Lamers et al. 1998). In the restoration process of the present study, however, naturally occurring seepage water was used that contained very low sulphate concentrations and sulphate concentrations of the soil pore water were not different for rewetted and control sites (data not shown). Furthermore, redox potentials were not lowered enough by rewetting for sulphate reduction to occur. As long as nitrate is present in the rewetted sites, as is indicated by positive net nitrification after rewetting, this can not be the case. We therefore exclude the possibility of internal eutrophication due to increased sulphate concentrations as a result of the rewetting process.

Although redox potential of the soil was significantly decreased by rewetting, it was not identified as a key factor controlling soil process rates or phosphorus availability. Only in winter, when groundwater levels occasionally rose above the soil surface and redox potentials decreased even further, soil redox potential was identified as a major factor in controlling process rates and P availability.

Implications for wetland restoration

Previous studies on wetland restoration have stressed the need for the use of naturally occurring and unpolluted water for rewetting to prevent (internal) eutrophication (Verhoeven et al. 1988; Roelofs, 1991; Lamers et al. 2002). Surprisingly, our results indicate that restoring natural seepage conditions on former agricultural land does not necessarily lead to favourable conditions for vegetation restoration by a reduction of nutrient availability. A major problem is formed by the large amounts of phosphate that have accumulated in the soil due to the previous agricultural practice. Especially in areas where preservation of a low groundwater level has led to acidification of the top soil, restoration of a high groundwater level with alkaline seepage water will lead to a large increase in phosphate availability and eutrophic conditions. For successful restoration, additional management like removal of the top soil or mowing and removal of hay will be required to be able to reduce the amounts of available phosphate (Bakker and Olff 1995). Furthermore, the change in water quality and the associated rise in soil pH can lead to stimulation of the decomposition process rather than reducing it. This is especially the case when redox potentials in the top soil remain rather high, for instance when water levels are below surface level for a large part of the year. Stimulation of the decomposition of soil organic matter poses a serious problem, for this is the largest source of nutrients in these systems (Verhoeven 1992) and is difficult to manipulate by management.

Conclusion

We consider the counterintuitive finding of eutrophication after rewetting with natural and unpolluted water to represent a new form of internal eutrophication, triggered by the restoration of natural site conditions on former agricultural land. In analogy with 'normal' internal eutrophication, large amounts of phosphate, that have been accumulating in the soil, are released upon rewetting. However, the cause of this release changes from anthropogenic (sulphate pollution) to natural (buffering capacity of groundwater). In contrast with 'normal' internal eutrophication, phosphate release is accompanied by a stimulation of organic matter decomposition, which can add to the eutrophication by a more structural release of nutrients in the system.

Because of the natural cause of this phenomenon, the new form of internal eutrophication poses serious problems for wetland restoration projects on soils with low pH.

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