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Sulphate and bicarbonate as key factors in sediment degradation and restoration of Lake Banen

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ABSTRACT

1. In many Dutch lakes, eutrophied, alkaline and sulphate-rich river water has been used to compensate for water losses. As a result, these waters have become eutrophied, and typical macrophyte species have disappeared.
2. The contribution of increased alkalinity to the eutrophication of softwater lakes was studied. In a mesocosm experiment, two types of sediments were flooded with demineralized water containing 2 mmol L^{-1} bicarbonate ions. The upper centimetres changed from a brown soil with coarse organic particles to a fine black mud. The formation of degradation intermediates and some phosphate release were observed.
3. This degradation was more evident in sediments flooded with demineralized water containing 4 mmol L^{-1} sulphate ions. In addition, sulphate consumption, sulphide production, bicarbonate production and enhanced phosphate release were observed in the sediment.
4. The eutrophied, softwater Lake Banen has been isolated from river water inputs, and mud layers were removed to restore the formerly oligotrophic, softwater conditions.
5. Removal of the degraded sediment layer and isolation of the lake from river water prevented sediment degradation and led to a return of the endangered macrophyte communities typical of softwater lakes.

KEY WORDS: lake restoration; sediment; sulphate; bicarbonate; eutrophication; macrophytes

INTRODUCTION

In most aquatic systems, organic matter gradually accumulates in the sediment. Its rate of decay and mineralization strongly depends on sediment type and water quality. In the sediments of neutral, softwater lakes, mineralization is slow and very incomplete (Traaen, 1980). Kok and Van de Laar (1990) observed an inhibition of microbial decomposition of fresh macrophyte leaves in waters with a buffering capacity below 0.5 meq L^{-1}. This was ascribed to the accumulation of acids, produced during the degradation of organic matter. Due to the low biodegradability potential of softwater sediments, a sediment rich in organic matter is formed. In shallow lakes, these sediments are predominantly aerobic as a consequence of oxygen supply via plant roots and low oxygen consumption during mineralization. Thus, plant

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communities living in oligotrophic conditions can persist in shallow softwater lakes with organic sediments (Sand-Jensen and Søndergaard, 1979; Szmeja, 1994).

Changes in water quality may lead to changes in mineralization rates. In alkalized waters, local acidification in sediments is prevented and biodegradation rates increase (Roelofs, 1991). In limed softwater sediments, enhanced phosphate release is frequently observed, which is thought to be due to enhanced mineralization (Roelofs *et al.*, 1994; Bellemakers, 1996). High concentrations of sulphate in the water layer also stimulate mineralization; when sulphate is reduced in the sediments, alkaline conditions are generated (Cook *et al.*, 1986; Schindler, 1988). This reduction is carried out by chemo-organotrophic bacteria. Consequently, sulphate reduction is closely linked to the degradation of organic matter. In anoxic, marine sediments of sulphate-rich and alkaline seawater, sulphate-reducing bacteria are often the most important mineralizing organisms (Jørgensen, 1982).

In the Netherlands, sulphate-enriched river water from the Rhine and Meuse rivers is used on a large scale in peaty lowlands to compensate for water shortage. It has been demonstrated that this has caused enhanced mineralization of organic layers in wet grasslands and lakes (Smolders and Roelofs, 1995; Lamers *et al.*, 1998a). This may lead to serious eutrophication of the water layer and transformation of peat into reduced, partially suspended mud. On sites where this internal eutrophication has occurred, rooted macrophytes and plants from oligotrophic soils have disappeared.

River water is also widely used to compensate for water shortage in Dutch Pleistocene, sandy areas with originally soft water. It is hypothesized that both alkalinization and sulphate enrichment cause internal eutrophication in these areas. To study the susceptibility of such waters to internal eutrophication, degraded organic softwater sediments from Lake Sarsven and intact organic softwater sediments from Lake Banen were treated with sulphate- and/or bicarbonate-enriched water, and with sulphate-free, soft water. It was expected that the rate of internal eutrophication would strongly depend on the biodegradability of the sediment. Moreover, in the degraded Lake Sarsven sediment, typical signs of internal eutrophication, such as alkalinization after sulphate reduction and nutrient mobilization, would be expected to occur on a smaller scale than in the Lake Banen sediment.

Desiccation is a naturally occurring phenomenon in shallow lakes. Drainage of the environment stimulates desiccation, especially if these water losses are not compensated for by the inlet of surface water. Sulphur oxidation and acidification can occur during desiccation—for example, during extreme summer drought (Vangenechten *et al.*, 1981; van Dam and Buskens, 1993). In this study, the effects of desiccation on both sediment types were studied. Experiments on intact soil cores of sulphur-polluted freshwater wetlands demonstrated serious acidification after desiccation (Lamers *et al.*, 1998b).

Deterioration of characteristic vegetation after enhanced sulphate supply has been recorded from various wet ecosystems with a naturally inhibited mineralization (Roelofs, 1991; Lamers *et al.*, 1998b). The restoration of Lake Banen was one of the first deliberate attempts to restore an ecosystem damaged by the supply of sulphate- and bicarbonate-rich water. Degraded sediments were removed and the input of river water was discontinued. The consequences of internal eutrophication, isolation, desiccation and lake restoration measures on water and sediment chemistry and on aquatic plant growth are described.

METHODS

Study site

Two originally softwater lakes on non-calcareous sandy soils in the South East Netherlands (5°47′N, 51°16′E) were studied: Lake Banen and Lake Sarsven (Figure 1). After draining of the environment at the beginning of this century, a high water level was constantly maintained by the inlet of nutrient-, sulphate- and bicarbonate-rich River Meuse water to Lake Sarsven and further on to Lake Banen (Figure 1). The average water depths were 1.0 and 0.6 m, respectively. Both lakes have been in contact with River Meuse water for several decades. In this period, the upper sediment layer of Lake Sarsven had completely turned into a fine black mud. All softwater macrophytes had disappeared and initial growth of reeds (*Phragmites australis* (Cav.) Steud.) was followed by a decline, indicating very low soil redox potentials (Weisner and Granéli, 1989). Sediment degradation was less severe in Lake Banen, although enhanced growth of reeds and disappearance of macrophytes also occurred there. A remnant of softwater vegetation existed in open water in the isolated south-west corner (Figure 1). In 1988, the input of river water to Lake Banen was discontinued and in winter 1992/1993, the upper, degraded sediment layers and part of the helophyte vegetation was removed. After isolation, Lake Banen became completely desiccated on several occasions during the dry summers of 1989, 1990 and 1996. Lake Sarsven still received river water and did not desiccate.

Experimental design

Degraded sediment from Lake Sarsven and intact sediment from Lake Banen were gathered in spring, 1995. After careful mixing, the sediments were placed in 25 × 25-cm aquaria in an 8-cm thick layer, and three Rhizon soil moisture suction cups 10 cm long were inserted horizontally into the sediment. A multiple platinum redox electrode was inserted vertically into the sediment to measure the redox potential at 0.25, 1.25, 2.25 and 3.25 cm, using an Ag/AgCl reference electrode. The organic matter content ranged from 14 to 18% and did not change throughout the experiment. The aquaria were kept in the dark at 20°C, and were left to stabilize under demineralized water for 3 weeks. After this period, the water layer was removed and four 10-L flooding treatments were applied in triplicate: no (sodium) bicarbonate ($NaHCO_3^-$) or (sodium) sulphate ($NaSO_4^{2-}$) (A^-S^-); 2 mmol L^{-1} bicarbonate and no sulphate (A^+S^-); no bicarbonate and 4 mmol L^{-1} sulphate (A^-S^+); and 2 mmol L^{-1} bicarbonate and 4 mmol L^{-1} sulphate (A^+S^+). In addition, one group was desiccated. The desiccating, originally water-saturated sediments were totally dehydrated after 7–8 weeks. After 80 days, 10 L of demineralized water were added. In the A^- and S^- treatments, an equivalent amount of sodium chloride was used to replace the bicarbonate and/or sulphate. Every 2 weeks, sulphate and bicarbonate concentrations in the water layer were checked and, if necessary, adjusted by addition. No more than a 10% deviation from the above-mentioned concentrations was found, with the exception of the A^- treatments. Here, diffusion

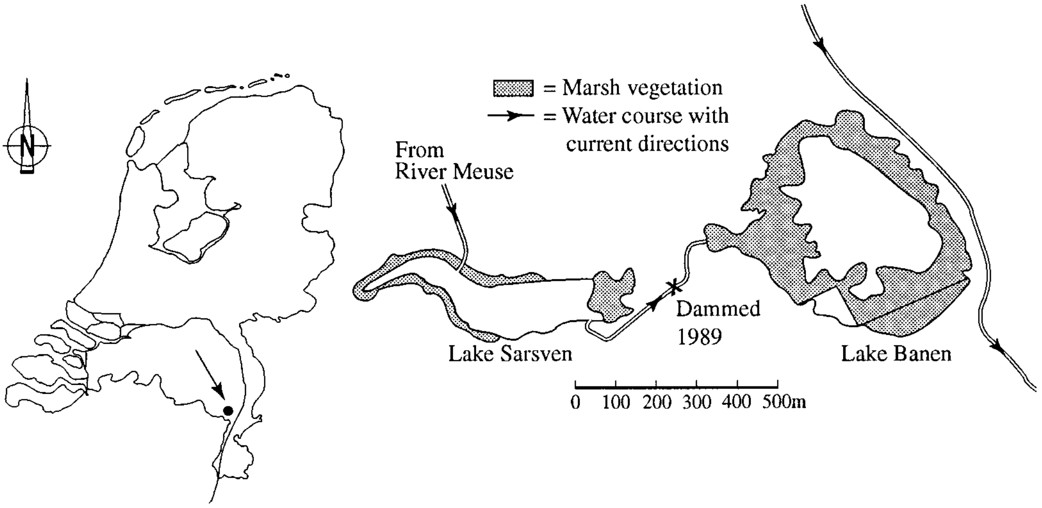

Figure 1. Location of the study site, including the River Meuse, and a map of Lake Sarsven and Lake Banen in 1989.

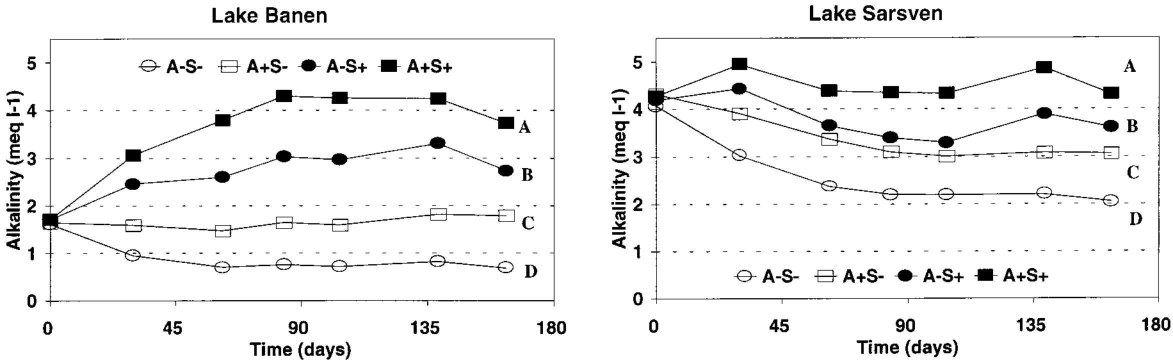

Figure 2. Changes of the alkalinity of the sediment pore water under a water layer containing: no bicarbonate or sulphate (A^-S^-); 2 mmol L^{-1} bicarbonate and no sulphate (A^+S^-); no bicarbonate and 4 mmol L^{-1} sulphate (A^-S^+); and 2 mmol L^{-1} bicarbonate and 4 mmol L^{-1} sulphate (A^+S^+). Different letters indicate significant differences ($p < 0.05$).

from the sediment led to an increase in bicarbonate in the water layer to 0.2 mmol L^{-1} (Lake Banen) and 1.0 mmol L^{-} (Lake Sarsven). For this reason, the water layer of the low bicarbonate Lake Sarsven treatment was completely replaced twice during the experiment.

Surface and sediment water were sampled monthly. The samples of the three suction cups in each aquarium were pooled. Alkalinity was directly determined by titration of 50 ml of water with 0.01 N HCl down to pH 4.2. A portion of each surface water sample was filtered through a Whatman GF/C filter (pore size, 1.2 μm). After adding 10 mg citric acid to avoid precipitation of metals, 50-mL surface and sediment samples were stored at -20°C until analysis. The concentrations of ortho-phosphate (i.e. soluble reactive phosphorus), nitrate, ammonium and chloride were measured colorimetrically with Technicon AA II systems, using ammonium-molybdate (Henriksen, 1965), hydrazinesulphate (Technicon, 1969), salicylate (Kempers and Zweers, 1986) and ferri-ammoniumsulphate (Technicon, 1969). Samples with a high dissolved organic carbon (DOC) concentration were corrected for background colour. Aluminium, iron, calcium and sulphur were analyzed using an inductively coupled plasma emission spectrophotometer (Jarell Ash Plasma 200, Instrumentation Laboratory), and potassium with a Technicon flame photometer. Extra samples were collected fortnightly and fixed immediately with sulphide antioxidant buffer to determine free sulphide concentrations. To mimic constant surface water infiltration, extra sediment pore water was collected weekly until 250 mL in total were removed. At the end of the experiment, DOC was determined. Filtered sediment and surface water samples were acidified and flushed with nitrogen to remove inorganic dissolved carbon. Following this, 5 mL of sample were evaporated and the residue was collected. The amount of carbon was measured on an Na 1500 nitrogen/carbon/sulphur analyzer (Carlo Erba Instruments). Field samples of Lake Banen water were analyzed as described above.

Data from the monthly measurements were log-transformed and tested with a two-way analysis of variance (ANOVA) with repeated measurements (SAS, 1989). Interaction effects between alkalinity and sulphate in the A^+S^+ treatment were not detected. Other results were tested with ANOVA and subsequent multiple comparisons between means were carried out.

RESULTS

Sediment degradation

Contact with sulphate-free, low-alkaline water caused a reduction in bicarbonate concentration in the intact peat sediment of Lake Banen to levels just below 1 meq L^{-1} (Figure 2), which is characteristic for

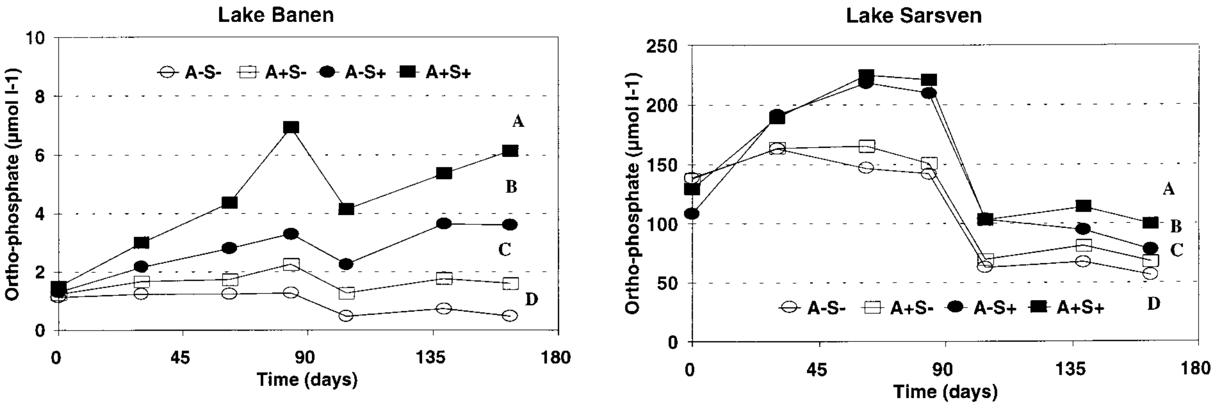

Figure 3. Development of the ortho-phosphate concentration of the sediment pore water under a water layer containing different concentrations of bicarbonate and sulphate. See legend to Figure 2 for further information. Different letters indicate significant differences ($p < 0.05$).

these sediments (Roelofs, 1983). The ortho-phosphate concentrations, which were slightly enhanced after mixing in the sediment, also dropped to characteristic values below 1 µmol L^{-1} (Figure 3). A decrease in alkalinity did not occur when the supplied water contained bicarbonate and some ortho-phosphate was mobilized. Furthermore, small increases in the concentrations of aluminium and free sulphide in the sediment pore water were observed (Table 1).

These effects became more accentuated when 4 mmol L^{-1} sulphate was present in the overlying water layer, and was strongest when both bicarbonate and sulphate were present (Figures 2 and 3). Aluminium concentrations in the sediment pore water ranged from 31 µmol L^{-1} in the A^-S^- treatment to 321 µmol L^{-1} in the A^+S^+ treatment. Free sulphide concentrations were also raised by these treatments (Table 1). A significant increase in ortho-phosphate and aluminium was also observed in the water layer of the A^+ and S^+ treatments. In both water and sediment, this increase was accompanied by an increase in the

Table 1. Some characteristics of the sediment pore water of Lake Banen and Lake Sarsven and the overlying water layer after 139 days of submergence in water containing different concentrations of bicarbonate and sulphate

	A^-S^-	A^+S^-	A^-S^+	A^+S^+
Lake Banen overlying water				
O–PO_4	1.0[a]	4.1[b]	2.6[a,b]	3.2[b]
Al	23[a]	62[a]	104[a]	101[a]
Lake Banen sediment				
Redox potential (mV)	−351[a]	−387[a]	−377[a]	−353[a]
S (including S^{2-})	63[a]	119[a]	1647[b]	1523[b]
S^{2-}	0.7[a]	1.2[a]	4.9[b]	11.3[b]
Al	31[a]	106[b]	203[c]	321[d]
Lake Sarsven sediment				
Redox potential (mV)	−406[a]	−416[a]	−398[a]	−398[a]
S (including S^{2-})	41[a]	50[a]	1874[b]	2567[c]
S^{2-}	0.3[a]	0.2[a]	1.0[b]	0.9[b]
Al	3.1[a]	4.4[a]	3.9[a]	7.5[b]

See legend to Figure 2 for further information. Concentrations are in µmol L^{-1}. Superscript letters indicate significant differences ($p < 0.05$) in the same row.

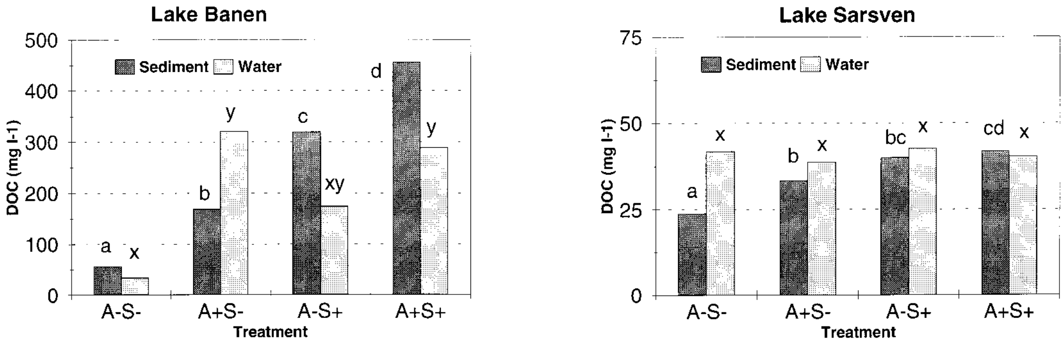

Figure 4. Dissolved organic carbon (DOC) in the water layer and the sediment pore water after 162 days of contact with water containing different concentrations of bicarbonate and sulphate. See legend to Figure 2 for further information. Different letters indicate significant differences ($p < 0.05$). Sediment and surface water data were tested separately.

DOC concentration (Figure 4). For all other parameters measured, no major differences between the treatments were observed, with the exception of a slightly increased nitrate concentration and a slightly decreased ammonium concentration in the sediments when bicarbonate and/or sulphate were added to the water layer (data not shown). The Lake Sarsven sediments were already rich in ortho-phosphate and very alkaline at the start of the experiment. Contact with water rich in sulphate and/or bicarbonate did not lead to a further increase (Figures 2 and 3). Mobilization of aluminium was minimal and only perceptible in the A^+S^+ treatment (Table 1).

The concentration of sodium chloride in the sediment became equal to the concentration in the water layer after several weeks of infiltration. However, sulphate concentrations in the sediment pore water remained lower than the added concentrations in the S^+ treatments: 1400 μmol L^{-1} in the Lake Banen sediment and 3000 μmol L^{-1} in the Lake Sarsven sediment. At the end of the experiment, these differences became less distinct (Table 1).

At the end of the experiment, sections were made of the sediment and these showed a degradation of organic matter. Sediments treated with bicarbonate- and/or sulphate-enriched water had turned into a watery, black mud in the upper centimetres, while the sediment treated with bicarbonate- and sulphate-free water still maintained the original brown colour and coarse, solid texture. At the end of the experiment, the sediments treated with bicarbonate- and/or sulphate-enriched water also showed an extremely high level of DOC (Figure 4). Part of this DOC diffused to the water layer. During the course of the experiment, the staining of the water layer screened the sediment from view in the A^+ and S^+ treatments of Lake Banen. In the Lake Sarsven sediment, however, almost no DOC production was observed and the water layer remained clear.

Sediment recovery

Contact with sulphate-free, soft water led to a limited recovery of sediment pore water quality. Alkalinity, and calcium and ortho-phosphate concentrations decreased, due to diffusion to the water layer (Figures 2 and 3). The low free sulphide and DOC concentrations suggest an inhibition of sulphate reduction and mineralization. However, the sediments remain reduced and no change in the structure of the sediments has been observed.

Desiccation

In the Lake Sarsven sediments, the redox potential increased gradually to +300 mv during desiccation (Figure 5). After subsequent flooding, alkalinity and pH dropped dramatically compared with the

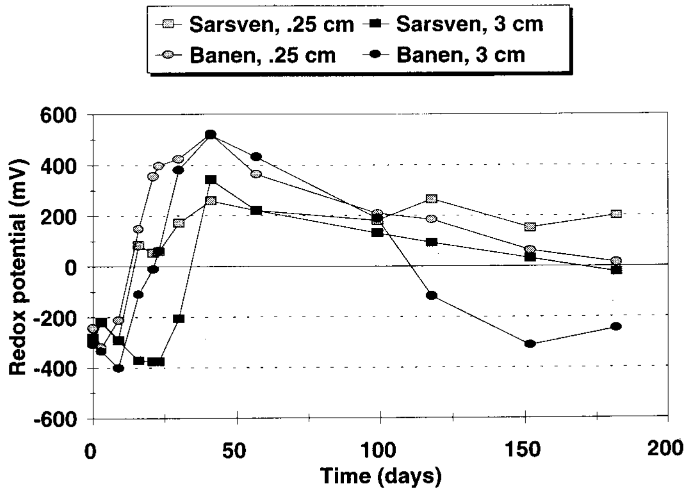

Figure 5. Redox potential of desiccated and subsequently flooded sediments of Lake Banen and Lake Sarsven, measured at 0.25 and 3.25-cm depth.

continuously submerged sediment (Table 2). Enormous concentrations of sulphate were measured and orange iron precipitates gathered on the aquarium wall. High concentrations of calcium and potassium also dissolved. During the 80 days of submergence, dissolved substances slowly diffused to the water layer. The redox potential slowly decreased, especially in the deeper sediment layers (Figure 5).

In the Lake Banen sediments, the redox potential increased rapidly to + 500 mv during desiccation. In general, the same processes occurred as in the Lake Sarsven sediments, but on a smaller scale. Therefore, no serious acidification occurred (Table 2). Ammonium disappeared during the first 20 days of emergence, before complete dehydration. No enhanced dissolution of aluminium was observed, contrary to the desiccated and acidified Lake Sarsven sediment (Table 2). After submergence, the Lake Banen sediment became more reduced than the Lake Sarsven sediment.

Table 2. Pore water characteristics of Lake Banen and Lake Sarsven sediments after 104 days of submergence in water containing no bicarbonate and no sulphate ($A^{+}S^{-}$) (Sub.), and after 80 days of gradual desiccation and 24 subsequent days of submergence in demineralized water (Des.)

	Lake Sarsven			Lake Banen		
	Sub.	Des.		Sub.	Des.	
pH	7.30	3.75	*	6.79	6.02	**
Alkalinity	2198	250	***	720	452	**
Al	2.3	23	***	24	19	*
Ca	1471	7194	***	474	1551	**
Fe	12	396	***	25	77	***
S	43	10 159	***	47	2168	***
O–PO_4	57	3.1	**	0.5	0.2	
NH_4	302	119	*	302	43	**
NO_3	1.6	4.6	**	3.0	1.8	
K	62	350	***	132	264	**

Concentrations in μmol L^{-1} or μeq L^{-1} (alkalinity). *, $p<0.05$; **, $p<0.01$; ***, $p<0.001$.

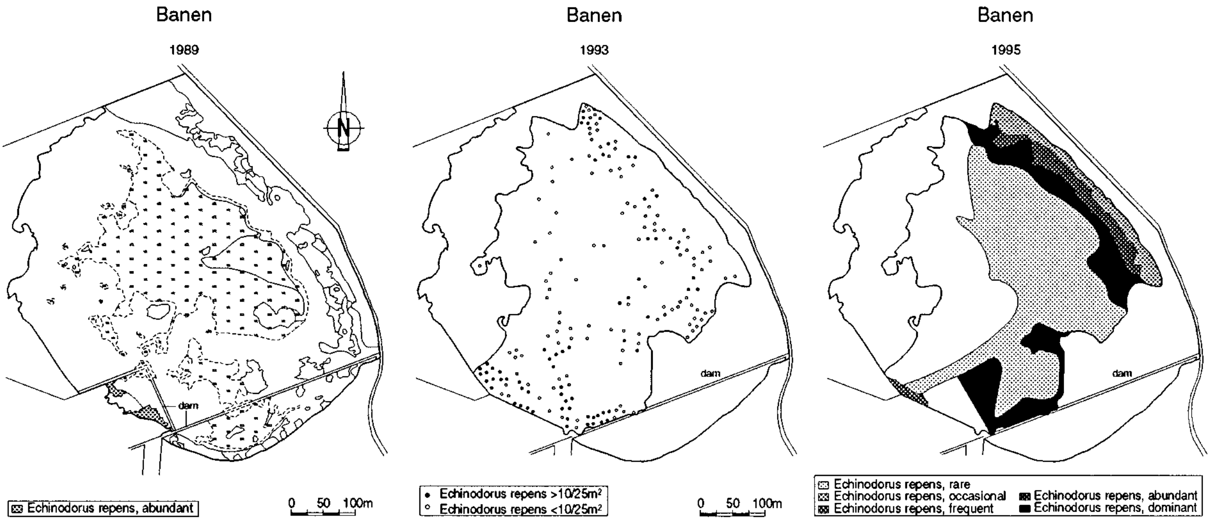

Figure 6. Distribution map of *Echinodorus repens* in Lake Banen before, immediately after and 2 years after lake restoration. In 1989, the south-west corner was isolated by a dam. This dam was removed during restoration.

Eutrophication and restoration of Lake Banen

Direct inlet of River Meuse water to originally oligotrophic, softwater lakes led eventually to the disappearance of all rooted macrophytes (Figure 6). In Lake Banen (and Lake Sarsven), several phases to this process were observed (Table 3). First, the isoetid-dominated vegetation was invaded by rooted macrophytes, including charophytes, from mesotrophic and eutrophic waters, e.g. *Elodea canadensis* Micheaux and *Utricularia australis* R.Br. Rapid reed growth was observed. After this, isoetid macrophytes disappeared, followed by the disappearance of other softwater macrophytes and the beginning of persistent algal blooms. Finally, the rooted macrophytes from eutrophic waters disappeared and the sediment turned into a fine black mud.

After ending the indirect inlet of River Meuse water in 1988, Lake Banen desiccated several times, either partially or totally, during the dry summers of 1989 and 1990. After renewed submergence, the

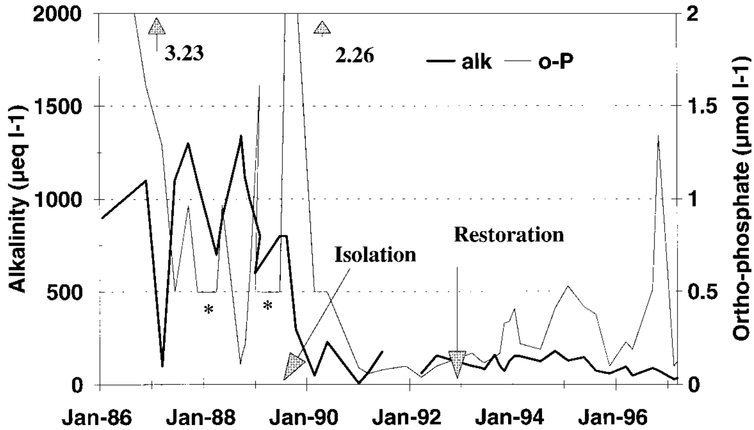

Figure 7. Alkalinity and ortho-phosphate concentrations in Lake Banen since 1986. The data from 1986–1990 were obtained partly from the Limburg Water Pollution Control Authority. *, Concentration below detection limit (1 μmol L^{-1}).

Table 3. Changes in the vegetation of Lake Banen and Lake Sarsven since the inlet of River Meuse water in both lakes during the middle of this century and since the isolation of Lake Banen from river water in 1988 and removal of degraded sediments in Lake Banen in 1993

	Lake Sarsven		Lake Banen			
	1	2	A	B	C	D
Littorelletea						
Apium inundatum (L.) Rchb.f.	x		x	x	*	x
Callitriche hamulata Koch						x
Echinodorus repens (Lam.) Kem and Reichg.			x	x	*	x
Elatine hexandra (Lapierre) D.C.						x
Eleocharis acicularis (L.) Roem & Schult.			x			x
Eleogiton fluitans (L.) Link			x	x	*	x
Hypericum elodes L.			x	x	*	x
Isoetes echinospora Durieu	x		x			x
Juncus bulbosus L.	x		x	x		x
Littorella uniflora (L.) Asch			x			x
Lobelia dortmanna L.	x		x			
Luronium natans (L.) Rafin.	x		x	x	*	x
Myriophyllum alterniflorum D.C.	x		x	x	x	x
Peplis portula (L.) D.A. Webb						x
Pilularia globulifera L.			x	x	*	x
Potamogeton gramineus L.	x		x	x	x	x
Potamogeton polygonifolius Pourr.			x	x	x	
Sparganium natans L.	x		x	x	*	x
Eutrophication indicators						
Lemna minor L.		x	x	x	x	x
Nuphar lutea (L.) Sm.				x		
Nymphaea alba L.	x	x		x	x	x
Other species						
Callitriche platycarpa Kütz					x	x
Elodea canadensis Michaux	x		x			
Hydrocharis morsus-ranae L.	x		x	x		
Lemna trisulca L.	x		x	x	x	x
Potamogeton berchtoldii Fieber				x		
Potamogeton natans L.	x		x	x	x	x
Potamogeton pusillus L.				x	x	
Ranunculus peltatus Schrank				x	x	x
Utricularia australis R.Br.				x	x	x

1 and 2, observed macrophytes in the summers of 1954 and 1955 (1), 1976 and 1985 (2); A to D, 1942 and 1950 (A), 1982 and 1985 (B), 1988 (C), 1994/1996 (D); *, restricted to the isolated south-west corner.

alkalinity of the water layer was low (Figure 7). The pH varied between 5 and 6 and large amounts of calcium and sulphate were measured. Typical softwater plants became established on the oxygenated, poorly buffered sediment. In 1992, juvenile plants of the softwater macrophytes *Isoetes echinospora* Durieu and *Elatine hexandra* (Lapierre) D.C. were found after several decades of absence. During the subsequent period of high water levels, most macrophytes disappeared once more, except in places having a thin organic layer on mineral soil.

After removal of the degraded sediment in 1993, softwater macrophytes established throughout the lake, including 12 (nationally) endangered species (Table 3). Macrophytes typical of alkaline and/or eutrophic water declined. In the proceeding years, softwater plants spread further and dominated the vegetation. The water quality remained good except for a small increase in nitrogen and phosphate concentrations after the flooding of non-restored areas of the shore during periods of very high water levels in the winters of 1994 and 1995 (Figure 7). In 1996, Lake Banen dried up in spring and did not fill again until late autumn. This caused mineralization of the sediment and temporary mobilization of nutrients. Sulphate concentrations in the sediment gradually decreased from 10 to 0.6 mmol L^{-1} in 1997, except for a small peak up to 2 mmol L^{-1} in 1996. Currently, the vegetation strongly resembles the situation before contact with River Meuse water, even after the extreme water-level fluctuations.

DISCUSSION

Enhanced bicarbonate availability in the sediment causes mobilization of phosphate and DOC production, as was observed earlier by Roelofs (1991). In his experiments, phosphate mobilization was coupled with a decrease in the sediment redox potential. As the underlying, intact sediment of Lake Banen was already strongly reduced, phosphate mobilization was more likely to be caused by an enhanced mineralization. At least part of this enhanced mineralization is caused by the activity of sulphate-reducing, minerotrophic microbes. The sulphide concentration in the sediment of the A^+ treatment was significantly higher throughout the experiment, than compared with the A^- treatment. This was not due to a difference in pH, which was between 7 and 8 in all sediments (data not shown).

Mineralization by sulphate-reducing bacteria is more obvious in sediments under sulphate-enriched water. The gradient of sulphate concentration from the water layer to the sediment suggests a continuous sulphate consumption. Based on the amount of infiltration, sulphate consumption is estimated to be 0.10 and 0.038 μmol g^{-1} fresh sediment day^{-1} in the Lake Banen and Lake Sarsven sediments, respectively. Sulphate reduction also causes the production of bicarbonate in both sediment types. The amount of sulphides produced seems small, but a large part of the sulphide probably disappeared in the gas phase or was fixed by precipitation with iron. Iron sulphide precipitation, iron exhaustion and mobilization of iron-bound phosphate in reduced sediments are well known phenomena (Caraco *et al*., 1989). After desiccation and oxidation of degraded Lake Sarsven sediments, iron appeared as a rust precipitation on the aquarium wall. Mineralization and iron-bound phosphate replacement by reduced sulphur compounds caused phosphate mobilization in the sediment. This process of internal eutrophication was already completed in the Lake Sarsven sediment before the start of the experiment. The limited DOC production and sulphate reduction show that further biodegradation of Lake Sarsven sediments is inconspicuous compared with the degradation of intact softwater sediments, equally rich in organic matter. In other words, the availability of biodegradable organic matter becomes limiting for the rate of mineralization as sediment degradation proceeds.

Internal eutrophication is not confined to the sediment; DOC and phosphate easily diffuse from the reduced sediment to the water layer, as was also observed by Roelofs (1991). The strong correlation between phosphate and DOC concentrations in the sediment and water layers suggests the complexation of both substances, as described by Boström *et al.* (1982). DOC in surface water strongly inhibits light penetration and thereby plant growth. Furthermore, a corresponding increase in aluminium concentrations in both the sediment and water layers has been observed after sediment degradation (Table 1). In general, aluminium concentrations are higher in acid waters, as can be seen in the acidified Lake Sarsven sediment after desiccation (Table 2). A rapid increase in pH can lead temporarily to very high and toxic levels of aluminium as a consequence of the formation of aluminium polymers or colloids (Rosseland *et al.*, 1992). In the present case, complexation with DOC is again more likely to be the cause of the high aluminium concentrations.

Once internal eutrophication has occurred, it is very difficult to reverse the process. In the A^-S^- treatment, comparable to shallow, soft waters, the degraded Lake Sarsven sediment loses some of its nutrients and bicarbonate, but is still nutrient-rich and strongly anaerobic, and it mixes easily with the water layer because of its fine structure. Desiccation of the degraded sediments reoxidizes the reduced sulphur compounds. Due to the accumulation of reduced sulphur compounds in the Lake Sarsven sediments, oxidation of these sediments takes more time compared with the Lake Banen sediments (Figure 5). Furthermore, sulphur oxidation leads to the consumption of alkalinity and to phosphate immobilization, but also causes acidification (Table 2). The pH drop causes mobilization of metals and, in combination with the presence of very high concentrations of sulphate, mobilization of calcium and magnesium. The increase of potassium concentrations may also be caused by an enhanced mineralization after desiccation. The same effects were observed in similar desiccation experiments by Lamers *et al.* (1998b). Once an organic-rich softwater sediment has degraded, the presence of toxic reduced substances and partially suspended sediment particles remain factors inhibiting the growth of a characteristic softwater flora and fauna.

In Lake Sarsven, all rooted macrophytes, except the very resistant *Nymphaea alba* L., disappeared after several decades of contact with River Meuse water. The most important causes are external nutrient loading, sulphide toxicity, phosphate and DOC release, algal bloom and dominance of plant species with floating leaves. In Lake Banen, partial and temporary recovery of the softwater vegetation occurred several years after isolation from River Meuse water and after some periods of desiccation. The consumption of alkalinity and ortho-phosphate immobilization in Lake Banen after desiccation, suggest the occurrence of large-scale oxidation of sulphur compounds, as observed in the desiccation experiment (Figures 2, 3 and 7). An immediate, more complete and stable recovery occurred after the removal of degraded sediments, rich in organic matter. Many of the typical softwater species had not been observed in the lake for several decades, which suggests that the presence of a vital seed bank is an important factor in lake restoration. Experience with earlier restoration projects tells us that most softwater macrophytes have such a persistent seed bank (Bellemakers, 1996). The establishment of stable macrophyte communities and the absence of algal blooms is also a result of the low alkalinity of the water layer; nutrients are fixed in organic matter in the sediment, where mineralization rates are low. In 1996, desiccation also led to the oxidation of reduced sulphate compounds, but on a smaller scale. Because of the isolation from river water, the extreme drought caused mineralization of the sediment and temporary mobilization of nutrients. The desiccation had no effect on macrophyte growth; all characteristic species persisted or were again present in 1997.

The exchange of ortho-phosphate between sediment and water is not affected by pH in soft waters with a neutral to slightly alkaline pH (Boström *et al.*, 1982). After isolation and restoration of the more alkaline Lake Cockshoot Broad (Eastern England), macrophyte establishment and growth was scarce, and phosphorus content and algal biomass were periodically high in the water layer (Moss *et al.*, 1996).

It is concluded that changes in water quality of softwater lakes can easily lead to irreversible sediment degradation, internal eutrophication and disappearance of characteristic macrophytes. However, when a vital seed bank or a remnant of the original vegetation still exists, these systems can be successfully restored by removing the degraded sediment and restoring the original water quality. The vegetation of shallow softwater lakes is less affected by prolonged periods of desiccation than by changes in water and sediment quality due to water compensation with alkaline and/or sulphate-enriched river water.

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REFERENCES

Bellemakers, M.J.S. 1996. 'Effects of liming on shallow moorland pools: a culture and a seed bank experiment', *Aquatic Botany*, **54**, 37–50.

Boström, B., Jansson, M. and Forsberg, C. 1982. 'Phosphorus release from lake sediments', *Archiv für Hydrobiologie*, Beiheft, **18**, 5–60.

Caraco, N.F., Cole, J.J. and Likens, G.E. 1989. 'Evidence for sulphate-controlled phosphorus release from sediments of aquatic systems', *Nature*, **341**, 316–318.

Cook, R.B., Kelly, C.A., Schindler, D.W. and Turner, M.A. 1986. 'Mechanisms of hydrogen ion neutralization in an experimentally acidified lake', *Limnology and Oceanography*, **31**, 134–148.

Henriksen, A. 1965. 'An automated method for determining low-level concentrations of phosphate in fresh and saline waters', *Analyst*, **90**, 29–34.

Jørgensen, B.B. 1982. 'Mineralization of organic matter in the sea bed—the role of sulphate reduction', *Nature*, **296**, 643–645.

Kempers, A.J. and Zweers, A. 1986. 'Ammonium determination in soil extracts by the salicylate method', *Communications in Soil Science and Plant Analysis*, **17**, 715–723.

Kok, C.J. and van de Laar, B.J. 1990. 'Influence of pH and buffering capacity on the decomposition of *Nymphaea alba* L. detritus in laboratory experiments: a possible explanation for the inhibition of decomposition at low alkalinity', *Verhandlungen der Internationalen Vereinigung für theoretische und angewandte Limnologie*, **24**, 2689–2692.

Lamers, L.P.M., Tomassen, H.B.M. and Roelofs, J.G.M. 1998a. 'Sulphate induced eutrophication and phytotoxicity in freshwater wetlands', *Environmental Science and Technology*, **32**, 199–205.

Lamers, L.P.M., Roozendaal, S.M.E. and Roelofs, J.G.M. 1998b. 'Acidification of freshwater wetlands: combined effects of non-airborne sulphur pollution and desiccation', *Water*, *Air and Soil Pollution*, **105**, 95–106.

Moss, B., Stansfield, J., Irvine, K., Perrow, M. and Phillips, G. 1996. 'Progressive restoration of a shallow lake: a 12-year experiment in isolation, sediment removal and biomanipulation', *Journal of Applied Ecology*, **33**, 71–86.

Roelofs, J.G.M. 1983. 'Impact of acidification and eutrophication on macrophyte communities in soft waters in the Netherlands', *Aquatic Botany*, **17**, 139–155.

Roelofs, J.G.M. 1991. 'Inlet of alkaline river water into peaty lowlands: effects on water quality and *Stratiotes aloides* L. stands', *Aquatic Botany*, **39**, 267–293.

Roelofs, J.G.M., Brandrud, T.E. and Smolders, A.J.P. 1994. 'Massive expansion of *Juncus bulbosus* L. after liming of acidified SW Norwegian lakes', *Aquatic Botany*, **48**, 187–202.

Rosseland, B.O., Blakar, I.A., Bulger, A., Kroglund, F., Kvellstad, A., Lydersen, E., Oughton, D.H., Salbu, B., Staurnes, M. and Vogt, R. 1992. 'The mixing zone between limed and acidic river waters: complex aluminium chemistry and extreme toxicity for salmonids', *Environmental Pollution*, **78**, 3–8.

Sand-Jensen, K. and Søndergaard, M. 1979. 'Distribution and quantitative development of aquatic macrophytes in relation to sediment characteristics in oligotrophic Lake Kalgeard, Denmark', *Freshwater Biology*, **9**, 1–11.

SAS 1989. *SAS/STAT Users Guide*, *Version 6*, SAS Institute Inc. Cary, N.C.

Schindler, D.W. 1988. 'Confusion over the origin of alkalinity in lakes', *Limnology and Oceanography*, **33**, 1637–1640.

Smolders, A. and Roelofs, J.G.M. 1995. 'Internal eutrophication, iron limitation and sulphide accumulation due to the inlet of river Rhine water in peaty shallow waters in the Netherlands', *Archiv für Hydrobiologie*, **133**, 349–365.

Szmeja, J. 1994. 'Dynamics of the abundance and spatial organization of isoetid populations in an oligotrophic lake', *Aquatic Botany*, **49**, 19–32.

Technicon 1969. *Technicon Auto Analyzer Methodology*, Industrial method 11 and 33-69w, Technicon Corporation, New York.

Traaen, T.S. 1980. 'Effects of acidity on decomposition of organic matter in aquatic environments', in Drabløs, D. and Tollan, A. (Eds), *Ecological Impact of Acid Precipitation*, Proceedings of an international symposium, March 1980, Sandefjord, Norway, 340–341.

Van Dam, H. and Buskens, R.F.M. 1993. 'Ecology and management of moorland pools: balancing acidification and eutrophication', *Hydrobiologia*, **265**, 225–263.

Vangenechten, J.H.D., Bosmans, F. and Deckers, H. 1981. 'Effects of short-term changes in rain water supply on the ionic composition of acid moorland pools in the Campine of Antwerp (Belgium)', *Hydrobiologia*, **76**, 149–159.

Weisner, S.E.B. and Granéli, W. 1989. 'Influence of substrate conditions on the growth of *Phragmites australis* after a reduction in oxygen transport to below-ground parts', *Aquatic Botany*, **35**, 7180.

