Preventing acidification and eutrophication in rich fens: Water level management as a solution?
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Impacts of short-term droughts and floodings in species-rich fen meadows during summer and winter; large-scale field manipulation experiments
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Casper Cusell, Ivan S. Mettrop, E. Emiel van Loon, Leon P.M. Lamers, Michel Vorenhout & Annemieke Kooijman
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Abstract

For the conservation and restoration of biodiverse rich fens, base-rich and nutrient-poor conditions are vital. In wetlands with artificially stable surface water levels, the re-introduction of temporary, short-term water level fluctuations have been postulated to restore the acid neutralizing capacity (ANC) by inundation and to reduce P-eutrophication during episodes with lower water levels.

This is the first study which tests this hypothesis in large-scale field manipulation experiments in protected base-rich fens, Calliergonella-dominated fens and Sphagnum-dominated fens. Five different experiments were performed: two weeks of raised levels (+10 cm) in a floating and a non-floating fen during winter, two weeks of high levels during summer in a non-floating fen, and two weeks of lowered levels (-15 cm) in a floating and a non-floating fen during summer.

For floating fens, both lowered and raised surface water levels in adjacent ditches did not show any effect on water tables in fens or on soil biogeochemistry. For non-floating fens, raised surface water levels led to flooding in all vegetation types, without affecting the nutrient concentrations. Although redox potentials decreased immediately in the upper part of soils, ANC was generally not enhanced in winter, due to the lack of infiltration into the waterlogged soils. In summer, in contrast, ANC increased as a result of higher temperatures and evapotranspiration, which led to enhanced infiltration of inundation water and to microbial alkalinity generation. Short-term lowering of surface water levels in summer led to lower water tables in non-floating fens, but only if precipitation rates were low. ANC and nutrient concentrations were, however, not affected at all.

Synthesis and applications: Our results show that the biogeochemical effects of short-term surface water level fluctuations strongly depend on peat buoyancy, water saturation of soils, season and weather. This explains why short-term floodings in winter are often not a suitable measure to restore the ANC of fens, while short-term floodings in summer do lead to an increase of the ANC. Short-term droughts do not affect the ANC or nutrient availability. These results are not only important for the hydrological management of fens, but also have future implications since short-term extreme weather events will occur more frequent, due to climate change.
**Introduction**

Rich fens are well-buffered and nutrient-poor peatland habitats that occur at a pH of 5.5 – 7.5 (e.g. Sjörs 1950). These species-rich fens are protected under the European Habitats Directive (transition mires, type H7140) and often comprise many threatened vascular plants and bryophytes such as *Liparis loeselii* (L.) Rich. and *Scorpidium scorpioides* (Hedw.) Limpr. In recent decades many rich fens have been lost in Europe, due to land use change (Kooijman 1992; JNCC 2007; Paulissen et al. 2013). Part of this decline is caused by natural succession to *Sphagnum*-dominated fens (e.g. Clapham 1940; Gorham et al. 1987; Kuhry et al. 1993), but anthropogenic forces, including high N-deposition rates, have presumably accelerated this succession in many cases. Although no experimental studies have been carried out yet, field studies indicate that atmospheric deposition may lead to accelerated acidification of rich fens (Gorham et al. 1987; Gunnarsson et al. 2000; Kooijman 2012). In addition, P-eutrophication can accelerate succession of P-limited rich fens to *Sphagnum*-dominated fens (Kooijman 1993a; Kooijman & Paulissen 2006).

In wetlands with strongly regulated surface water levels as a result of agriculture in the surroundings, one of the proposed management tools to counteract acidification and P-eutrophication is the re-introduction of more fluctuating surface water levels. Raised surface water levels may result in an increase of the alkalinity, pH, and/or Ca-concentrations in soil pore waters (Loeb et al. 2008a). The acid neutralizing capacity (ANC) can be increased by microbial-induced anaerobic reduction of Fe(III), SO$_4^-$ and/or NO$_3^-$ in wet soils (Gambrell & Patrick 1978; Baker et al. 1986), which will be temporary since aerobic oxidation during subsequent episodes with lower water tables in fens can lead to the opposite process of acidification (Lamers et al. 1998a; Loeb et al. 2008a). In addition, ANC can also increase more permanently by infiltration of base-rich surface water during inundation. A rise in surface water levels may, however, also result in eutrophication due to higher P- and N-inputs during flooding (Wassen et al. 1996) and/or increased internal P-mobilization by Fe(III)- and/or SO$_4^-$-reduction (Patrick & Khalid 1974; Lamers et al. 1998b). The latter process depends on the P-concentrations in the soil and its type of binding (Loeb et al. 2008b).

Episodes with lowered surface water levels will lead to reduced input of water into wetlands, because less surface water is needed to compensate for water losses. Since surface waters in Europe often contain high nutrient concentrations, due to intensive agricultural land use around wetlands, lowered water tables in fens will presumably result in a reduction of N- and P-inputs (Coops & Hosper 2002). At the same time, however, lowered water tables can also stimulate net mineralization rates, due to higher oxygen availability in fen soils (Grootjans et al. 1986; Updegraff et al. 1995; Bridgham et al. 1998; Olde Venterink et al. 2002a). Furthermore, water level drawdown may result in acidification as a consequence of aerobic oxidation processes (Lamers et al. 1998a; Lucassen et al. 2002), which are microbial-induced redox processes that consume oxygen.
All mechanisms mentioned above have been studied intensively in mesocosm and incubation experiments, but none of these studies examined the net effect of all water-level related processes in a field experiment. This is the first study in which the physical and biogeochemical responses of short-term (two weeks) surface water level rises (during winter and summer) and drawdowns (during summer) have been tested for several years in large-scale field experiments in protected base-rich fens, Calliergonella-dominated fens and Sphagnum-dominated poor fens. The questions addressed in this study were: (a) what are the changes in water table and biogeochemical responses as a result of short-term (two weeks) changes in surface water level, (b) do these responses differ between floating and non-floating fens, and (c) do the responses to raised surface water levels in non-floating fens differ between winter and summer conditions? The answers to these questions will not only be important for the hydrological management of fens, but are also likely to have future implications since short-term periods with intense precipitation or drought will occur more frequent in many parts of the world, due to climate change (e.g. Bronstert 2003; Kundzewicz et al. 2006). Our expectation for (a) was that raised surface water levels lead to an increased ANC, but also to P-eutrophication. In contrast, lowered surface water levels are expected to lead to acidification andeutrophication, due to increased mineralization rates. For (b), we expect that the effects on biogeochemistry are largest in non-floating fens, since water tables will presumably fluctuate more in these fens than in floating fens. For (c), we expect that the increase in ANC will be larger in summer than in winter floodings, because of higher infiltration rates and/or higher microbial alkalinity generation.

Materials and methods

Experimental design

Three summer-mown experimental fen sites in the Dutch Ramsar area “National Park Weerribben-Wieden” were used to determine the biogeochemical effects of short-term rises and decreases of the surface water level: a floating fen site in “De Weerribben” (WEE; 52°47’ N, 5°55’ E) and two non-floating fen sites in “Kiersche Wiede” (KW; 52°42’ N, 6°08’ E) and “Veldweg” (VW; 52°42’ N, 6°07’ E). The floating WEE-fen had a buoyant 70 – 90 cm thick peat layer, floating above a sandy substrate at 250 cm below soil surface. It comprised three vegetation types: (a) fen type where Calliergonella cuspidata (Hedw.) Loeske dominated the moss layer (Call; Caricion nigrae – Carex nigra-Agrostis canina type), (b) poor fen type where Sphagnum palustre L. and Sphagnum fallax (H.) Klinggr. dominated the moss layer (Sph; Caricion nigrae – Pallavicinio-Sphagnetum typicum type), and (c) moor type with Erica tetralix L. and S. palustre (Moor; Oxyocco-Ericion – Sphagno palustris-Ericetum type). In contrast, the non-floating KW- and VW-fens were firmly connected to the lower sandy substrate, which was found at a depth of 60 – 90 cm. In addition to the three mentioned vegetation types in the WEE-area, the KW- and VW-areas also contained
some rich fens (Scor) with *Hamatocaulis vernicosus* (Mitt.) Hedenäs (*Caricion nigrae – Carex nigra-Agrostis canina* type) or *Scorpidium cossonii* (Schimp.) Hedenäs (*Caricion davallianae – Scorpidium-Carex diandra* type), respectively.

The present surface water level of the National Park, situated below sea level, fluctuates slightly between 0.73 and 0.83 m below mean sea level (BMSL) from March to November and is maintained at 0.83 m BMSL from December to February. In this study, five experiments were conducted to evaluate the biogeochemical effects of lowered and raised surface water levels on the different fen types mentioned (Table 2.1). In the WEE- and KW-areas, experimental floodings and water level drawdowns were enabled by the construction of dams around the areas (about 5 and 35 ha, respectively) and the use of pumps. Surface water levels were raised by 10 cm for two weeks in November to 0.63 m BMSL in the floating WEE-fen (*experiment 1*) and non-floating KW-fen (*experiment 2*). These raised levels were applied in 2009 and 2010 in the WEE-fen and between 2008 and 2011 in the KW-fen. Technical constraints made it impossible to raise surface water levels in the WEE-area in 2008 and 2011. In addition, the effects of high surface water levels in summer were examined in the non-floating VW-fen (*experiment 3*) during wet periods in July 2009 and 2010. During these periods, about 50 mm of rain in two weeks (3.5 – 4 mm/day) resulted in surface water levels of 0.73 m BMSL. In this case, surface water levels were thus not manipulated by pumps and just equaled the levels in the entire National Park. Finally, surface water levels were lowered by 15 cm for two weeks in July 2009, 2010 and 2011 to about 0.98 m BMSL in the floating WEE-fen (*experiment 4*) and non-floating KW-fen (*experiment 5*).

**Table 2.1.** Schematic overview of the five field manipulation experiments.

<table>
<thead>
<tr>
<th>Area</th>
<th>Fen type</th>
<th>Month</th>
<th>Treatment of two weeks</th>
</tr>
</thead>
<tbody>
<tr>
<td>Experiment 1 Weerribben (WEE)</td>
<td>Floating</td>
<td>November</td>
<td>Raised surface water level</td>
</tr>
<tr>
<td>Experiment 2 Kiersche Wiede (KW)</td>
<td>Non-floating</td>
<td>November</td>
<td>Raised surface water level</td>
</tr>
<tr>
<td>Experiment 3 Veldweg (VW)</td>
<td>Non-floating</td>
<td>July</td>
<td>Raised surface water level</td>
</tr>
<tr>
<td>Experiment 4 Weerribben (WEE)</td>
<td>Floating</td>
<td>July</td>
<td>Lowered surface water level</td>
</tr>
<tr>
<td>Experiment 5 Kiersche Wiede (KW)</td>
<td>Non-floating</td>
<td>July</td>
<td>Lowered surface water level</td>
</tr>
</tbody>
</table>

**Sampling**

At all fen sites, five homogenous plots of 2 x 2 m were selected for each of the vegetation types present (Scor, Call, Sph and Moor). At each plot, water tables in the fen were recorded (a) two days before, (b) during and (c) two days after each experimental manipulation of the surface water level. The water tables were recorded manually.
Before and after the treatment, soil pore water samples of the upper 10 cm of the soil were collected anaerobically by connecting vacuumed plastic syringes (50 mL) to soil moisture samplers (Rhizons SMS 10 cm, Eijkelkamp Agrisearch Equipment, Giesbeek, the Netherlands). The first 10 mL of each sample was discarded to exclude stagnant sampler water. Similar samples were also collected in July 2008 to determine the initial biogeochemical conditions for all fen sites, before any treatment had started.

Since raised surface water levels led to flooding of the KW-area (experiment 4), we also collected the flooding water above the vegetation for this experiment. After one week of flooding, iodated polyethylene bottles of 100 mL were used to collect the flooding water in 2009, 2010 and 2011. At the same moment, surface water samples were taken in five adjacent ditches that supply the flooding water.

Plant species composition was also recorded in the subplot of 2 x 2 m in July 2008 (before the start of any treatment) and July 2011 (after the treatments). All vascular plant and bryophyte species were recorded, and cover values were estimated as percentages. These results will not be discussed further, since no significant developments were found during the short period of three years, except for a trend to higher abundance of *Sphagnum* spp., i.e. *S. palustre* and *S. fallax* (Cusell et al. 2013).

Chemical analyses

The pH-values of all water samples were measured, and alkalinities were determined by titration to pH 4.2, using 0.01M HCl. Next, surface water and flooding water samples were filtered through GF/C glass-fiber filters (Ø = 1.2 μm; Whatmann, Brentford, UK). Subsequently, all samples were divided into two subsamples, and 1% of concentrated HNO₃ (P.A. quality) was added to one of these subsamples to avoid metal precipitation. Both subsamples were stored in the dark at -24 °C until further analysis. Total concentrations of soluble Ca, Fe and S were measured in the acidified subsamples by ICP-OES (Optima 3000 XL, PerkinElmer, Waltham, USA). In the non-acidified subsamples, concentrations of NH₄⁺, NO₃⁻, o-PO₄ and Cl were analyzed colorimetrically by continuous flow auto-analyzers (Skalar Analytical BV, Breda, the Netherlands).

Continuous redox measurements

Continuous measurements of the redox potential (Eₚ) were conducted in the KW-fen between September 2010 and July 2012. Two fiberglass probes with platinum sensor tips (PaleoTerra, Amsterdam, the Netherlands) were permanently installed in patches with Scor-, Sph- and Moor-vegetation, and these six probes were connected to a HYPNOS III data logger (MVH Consult, Leiden, the Netherlands; Vorenhout et al. 2011). Each probe contained seven sensor tips to record the Eₚ (measured potential) at -1, -3, -5, -10, -15, -20 and -50 cm below the soil surface every 15 minutes. The Eₚ was measured as the potential between a sensor tip and a reference electrode, a 3M Ag/
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AgCl reference probe. The $E_h$ was calculated by adding a standard reference voltage and correcting for differences in pH, since pH indirectly modifies the Nernstian effect of the redox electrode:

$$E_h = E_m + E_{ref} - 59 \times (7 - \text{pH}),$$

with $E_m$ being the measured potential and $E_{ref}$ being the potential of the reference probe to the standard hydrogen probe of 228 mV.

Statistical analyses

Statistical analyses were performed in SPSS for Windows (SPSS 20.0.0, IBM, Armonk, USA). A two-way ANOVA with LSD post-hoc test was used to determine significant differences in initial water tables (relative to the fen surface) and biogeochemical conditions between fen sites and vegetation types in July 2008, before any treatment had started. Since fen sites differed in terms of biogeochemistry and the ability to float, subsequent analyses were performed separately for the five different experiments. Because the measurement plots were fixed, hence not independent over the years, a linear mixed model with year as repeated effect was used to determine the response to the fixed factors vegetation type and year (West et al. 2007). Within each year, two or three consecutive measurements were used to determine contrasts between the conditions right before, during and right after the change in surface water levels. The differences between measurements directly before and after the treatment were used as response variables in the linear mixed models. Differences between vegetation types and years, whenever significant in the mixed model, were further examined by comparing their estimated marginal means in a LSD post-hoc test.

In experiment 2, where increases of the surface water level led to flooding in the KW-area, two additional linear mixed models, each with year as repeated effect and a single predictor variable, were used. The first model used vegetation type as fixed factor to evaluate if the flooding water had a homogenous composition or differed between the vegetation types in the KW-area. The second model used a categorised value for the start water table as fixed factor to evaluate the effect of this start water table on the increase of Cl-concentrations in soil pore waters during floodings.

Results

Initial conditions

At the fen sites studied, water tables were significantly higher in base-rich Scor- and Call-vegetation than in more acidic Sph- and Moor-vegetation, with mean depths of 5 – 10 cm and 15 – 23 cm below the surface in July (Fig. 2.1, Table 2.2). As expected, initial pH-values of 5.6 – 6.3 in soil pore waters for Scor- and Call-vegetation were
Fig. 2.1. Water table (a), pH (b), alkalinity (c) and concentrations of Ca (d), Cl (e), o-PO₄ (f), NO₃ (g) and NH₄ (h) in the soil pore waters of four vegetation types (Scor = fen with a moss-layer dominated by Scorpium cossonii or Hamatocaulis vernicosus, Call = fen with a moss-layer dominated by Calliergonella cuspidata, Sph = fen with a moss-layer dominated by Sphagnum palustre, moor with Erica tetralix and Sphagnum palustre) in three fens. Sample means are given with their standard deviations (n = 5). KW = non-floating fen in Kiersche Wiede, VW = non-floating fen in Veldweg, WEE = floating fen in Weerribben. Statistical information is provided in Table 2.2.
also significantly higher than in Sph- and Moor-vegetation, where mean pH-values of about 4.7 were measured. Scor- and Call-vegetation also showed significantly higher alkalinities, Ca- and Cl-concentrations than Sph- and Moor-vegetation, with initial alkalinities of about 1000 and 200 µmol L⁻¹, Ca-concentrations of around 500 and 200 µmol L⁻¹ and Cl-concentrations of around 900 and 500 µmol L⁻¹. It was, however, remarkable that the VW-fen showed higher pH-values, alkalinities and Ca-concentrations than the other two fen sites, which was especially the case in base-rich vegetation types, as indicated by significant interaction effects of area and vegetation type. In contrast, concentrations of o-PO₄, NO₃ and NH₄ did not differ between vegetation types or fen sites. These concentrations were low in the soil pore waters of all vegetation types, with concentrations below 1, 3 and 10 µmol L⁻¹, respectively.

**Table 2.2.** Effects of fen site, vegetation type and their interaction on chemical variables in the pore water at the start of the experiment in July 2008. KW = non-floating fen in Kiersche Wiede, VW = non-floating fen in Veldweg, WEE = floating fen in Weerribben, Scor = fen with a moss-layer dominated by *Scorpidium cossonii* or *Hamatocaulis vernicosus*, Call = fen with a moss-layer dominated by *Calliergonella cuspidata*, Sph = fen with a moss-layer dominated by *Sphagnum palustre*, moor with *Erica tetralix* and *Sphagnum palustre*.

<table>
<thead>
<tr>
<th></th>
<th>Fen site</th>
<th>Veg</th>
<th>Fen site * Veg</th>
<th>KW</th>
<th>VW</th>
<th>WEE</th>
<th>Scor</th>
<th>Call</th>
<th>Sph</th>
<th>Moor</th>
</tr>
</thead>
<tbody>
<tr>
<td>water table</td>
<td>1.83</td>
<td>13.46**</td>
<td>1.57</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>b</td>
<td>b</td>
<td>a</td>
<td>a</td>
</tr>
<tr>
<td>pH</td>
<td>6.32**</td>
<td>29.46**</td>
<td>2.58*</td>
<td>a</td>
<td>b</td>
<td>a</td>
<td>b</td>
<td>b</td>
<td>a</td>
<td>a</td>
</tr>
<tr>
<td>alkalinity</td>
<td>2.46</td>
<td>21.50**</td>
<td>2.61*</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>b</td>
<td>b</td>
<td>a</td>
<td>a</td>
</tr>
<tr>
<td>Ca</td>
<td>10.44**</td>
<td>17.50**</td>
<td>4.32**</td>
<td>b</td>
<td>c</td>
<td>a</td>
<td>c</td>
<td>b</td>
<td>a</td>
<td>a</td>
</tr>
<tr>
<td>Fe</td>
<td>2.63</td>
<td>0.83</td>
<td>2.19</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
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<tr>
<td>S</td>
<td>1.64</td>
<td>1.10</td>
<td>2.32</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
</tr>
<tr>
<td>Cl</td>
<td>6.72**</td>
<td>10.50**</td>
<td>1.02</td>
<td>b</td>
<td>b</td>
<td>a</td>
<td>b</td>
<td>b</td>
<td>a</td>
<td>a</td>
</tr>
<tr>
<td>O-PO₄</td>
<td>0.32</td>
<td>0.95</td>
<td>0.62</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
</tr>
<tr>
<td>NH₄</td>
<td>0.83</td>
<td>0.94</td>
<td>1.95</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
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<tr>
<td>NO₃</td>
<td>2.20</td>
<td>1.42</td>
<td>0.62</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
</tr>
</tbody>
</table>

F-ratios with their level of significance: * P ≤ 0.05, ** P ≤ 0.01. Different letters indicate significant differences (P ≤ 0.05) between treatments, n.s. = not significant.

**Experiment 1: Raised surface water levels in a floating fen during winter**

As expected, a rise of surface water levels by 10 cm had almost no effect on the water tables in the floating fen soils (Fig. 2.2, see Table S2 in Supporting Information of online article). Along with this limited change in water tables, none of the measured biogeochemical variables was significantly changed for any of the vegetation types.
Fig. 2.2. Effect of five surface water level treatments on the water table (given in cm above/below the fen surface) in four vegetation types during the monitored years. Water tables were measured two days before (black lines at the left of each triplet), during (grey lines) and two days after the treatments (black lines at the right of each triplet). Sample means (white centers of a line) are given with their standard deviations (n = 5). Scor = fen with a moss-layer dominated by *Scorpidium cossenii* or *Hamatocaulis vernicosus*, Call = fen with a moss-layer dominated by *Calliergonella cuspidata*, Sph = fen with a moss-layer dominated by *Sphagnum palustre*, moor with *Erica tetralix* and *Sphagnum palustre*, exp. 1 = floating WEE-fen during raised surface water levels in winter, exp. 2 = non-floating KW-fen during raised surface water levels in winter, exp. 3 = non-floating VW-fen during raised surface water levels in summer, exp. 4 = floating WEE-fen during lowered surface water levels in summer, exp. 5 = non-floating KW-fen during lowered surface water levels in summer. Statistical information is provided in Table S2 (Supporting Information of online article).
Experiment 2: Raised surface water levels in a non-floating fen during winter

The rise in surface water levels by 10 cm during the treatment periods in November led to flooding in all vegetation types during all four years (Fig. 2.2, see Table S2). Water table rises were largest in 2011, when initial water tables were lowest with 5 – 15 cm below the surface, and smallest in 2009, when initial water tables were highest with levels around the fen surface. In 2009, most Scor- and Call-vegetation was even inundated at the start of the treatment. Furthermore, water tables raised significantly more in Scor- and Call-vegetation than in Sph- and Moor-vegetation.

In samples of flooding water, concentrations of the inert Cl-anion did not differ between vegetation types during any of the monitored years, and were equal to the concentrations in the adjacent ditches that supply the flooding water (Fig. 2.3). Concentrations of o-PO₄, NH₄ and NO₃ did also not differ between vegetation types and were low with values of 0.05, 3 and 2 μmol L⁻¹, respectively. In contrast, alkalinitities and Ca-concentrations did significantly differ in flooding water samples, with alkalinitities of around 900 and 500 μmol L⁻¹ and Ca-concentrations of about 500 and 200 μmol L⁻¹ above Scor- and Call-vegetation versus Sph- and Moor-vegetation. Also, the pH decreased significantly from about 7.0 in ditches to 6.4 in standing flooding water above Scor- and Call-vegetation to about 5.4 in flooding water above Moor-vegetation.

Before the floodings, Cl-concentrations in soil pore waters were lower than in the flooding water above the vegetation in all four monitored years (Fig. 2.4). However, the higher Cl-concentrations in flooding waters only led to increased pore water concentrations of Cl during the floodings of 2010 and 2011, and these effects differed significantly between vegetation types, as indicated by the interaction effect of area and vegetation type (see Table S2). In Scor- and Call-vegetation, pore water concentrations of Cl only increased significantly during the flooding of 2011, while Sph- and Moor-vegetation showed significantly increased Cl-concentrations in 2010 and 2011. This clearly shows that the flooding water did not always infiltrate. An additional analysis showed that Cl-concentrations only increased in soil pore water when the start water tables were lower than 5 cm below the soil surface (Table 2.3).

Table 2.3. Effect of water table on the Cl-infiltration into soil pore waters of the KW-fen during floodings in 2009, 2010 and 2011 (experiment 4).

<table>
<thead>
<tr>
<th>Initial water table</th>
<th>Above the surface</th>
<th>0 – 2 cm below surface</th>
<th>3 – 5 cm below surface</th>
<th>6 – 9 cm below surface</th>
<th>More than 9 cm below surface</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cl (μmol L⁻¹)</td>
<td>-64 (81)</td>
<td>53 (134)</td>
<td>46 (111)</td>
<td>282 (167)</td>
<td>185 (107)</td>
</tr>
</tbody>
</table>

Mean values and standard deviations for the differences between the Cl-concentrations just after and before the floodings. Different letters indicate significant differences between water level categories (P ≤ 0.05).
Fig. 2.3. pH (a), alkalinity (b) and concentrations of Ca (c), Cl (d), S (e), o-PO$_4$ (f), NO$_3$ (g) and NH$_4$ (h) in the surface water of adjacent ditches, which supply the flooding water, and the flooding water above four vegetation types in the KW-fen (experiment 2). See the caption of Fig. 2.1 for a description of the abbreviations. Sample means for 2009, 2010 and 2011 are given with their standard deviations (n = 15). Different letters indicate significant differences between vegetation types (P ≤ 0.05).
In line with the absence of infiltration in 2008 and 2009, flooding had almost no biogeochemical effect in 2008 and 2009. In contrast, biogeochemical effects were observed during the flooding of 2011, when infiltration occurred in all vegetation types. The response to the flooding in 2011 did, however, differ between vegetation types. Redox potentials ($E_h$) decreased almost immediately in the Sph- and Moor-vegetation, where $E_h$ decreased from about +600 to -100 mV in the upper 12 to 18 cm of the soils, respectively (Fig. 2.5). On the other hand, $E_h$ was only slightly affected in Scor-soils, because nearly the entire profile already showed anaerobic conditions, with $E_h$-values of around -200 mV before the flooding. In these soils, $E_h$ only changed slowly from around 300 to -200 mV in the upper 2 cm of the soil. In contrast, alkalinities and Ca-concentrations only increased significantly in soil pore waters of Scor- and Call-vegetation, with 350 $\mu$mol L$^{-1}$ and 150 $\mu$mol L$^{-1}$, and remained equal in soil pore waters of Sph- and Moor-vegetation, as indicated by the interaction effect of area and

**Fig. 2.4.** Effect of five surface water level treatments on the Cl-concentrations ($\mu$mol L$^{-1}$) in four vegetation types during the monitored years. Concentrations were measured two days before (black lines at the left of each triplet), during (grey lines; only in experiment 2) and two days after the treatments (black lines at the left of each triplet). See the caption of Fig. 2.2 for a description of the abbreviations. Sample means (white centers of a line) are given with their standard deviations ($n = 5$). Statistical information is provided in Table S2 (Supporting Information of online article).
vegetation type (Figs. 2.6 & 2.7, see Table S2). Finally, the flooding of 2011 had no effect on Fe-, S-, o-PO$_4$-, NH$_4$- and NO$_3$-concentrations in soil pore waters of any vegetation type (Appendix A, see Table S2).

Experiment 3: Raised surface water levels in a non-floating fen during summer

Before the start of the treatment, water tables were significantly lower in July 2010 than July 2009 (Fig. 2.2, see Table S2), with tables of 20 – 30 cm below the surface in 2010 (when the treatment was preceded by a very dry period) and tables of 3 – 20 cm below the surface in 2009. Rather heavy rainfall of 10 – 20 mm day$^{-1}$ during the first treatment week of 2009 and 2010 led to a rise of water tables by 10 – 15 cm in all soils. In 2009, this rise resulted in flooding with surface water in Scor- and Call-vegetation, while lower (surface) water levels at the start of the 2010-treatment prevented inundations.

Fig. 2.5. Redox potentials ($E_h$) in the upper 20 cm of the soil (vertical scale) in three vegetation types of the KW-fen (Scor = fen with a moss-layer dominated by *Humatocaulis vernicosus*, Sph = fen with a moss-layer dominated by *Sphagnum palustre*, moor with *Erica tetralix* and *Sphagnum palustre*) between June 16 and July 31 (2011; left), and November 1 and December 16 (2011; right). The vertical white lines indicate the initiation and end of the treatment period with lowered (-15 cm; left) and raised (+10 cm; right) surface water levels. For interpolation, ordinary kriging was applied in ArcGIS (ArcMap 10.0, ESRI, Redlands, USA).
The raised surface water levels in 2009 and 2010 had no effect on pH or o-PO$_4$$^-$, NH$_4^-$ and NO$_3^-$-concentrations in soil pore waters (Appendix A, see Table S2). In contrast, alkalinites and Ca-concentrations in soil pore waters increased during the wet periods in both years. These increases were, however, higher in 2009 than 2010 and effects differed between vegetation types, as indicated by the interaction effect of area and vegetation type (Figs. 2.6 & 2.7). In 2009, alkalinites and Ca-concentrations increased stronger in the flooded Scor- and Call-vegetation than in the non-flooded Sph- and Moor-vegetation, with increases in alkalinities of around 1900 µmol L$^{-1}$ and 300 µmol L$^{-1}$ and increases in Ca-concentrations of around 450 µmol L$^{-1}$ and 80 µmol L$^{-1}$, respectively. Non-flooded Scor- and Call-vegetation in 2010 showed significantly smaller increases in alkalinities (about 600 µmol L$^{-1}$) and Ca-concentrations (about 150 µmol L$^{-1}$), while Sph- and Moor-vegetation showed similar increases in alkalinities (about 250 µmol L$^{-1}$) and Ca-concentrations (about 80 µmol L$^{-1}$) in 2010 as in 2009. Furthermore, Cl-concentrations increased by about 300 µmol L$^{-1}$ in soil pore waters of all vegetation types during the non-flooded situation in 2010, while Cl-concentrations remained similar in 2009.

In 2009 and 2010, raised surface water levels led to decreased S-concentrations and increased Fe-concentrations in soil pore waters of all vegetation types (Appendix A, see Table S2). In all vegetation types, S-concentrations decreased with 50 – 150 µmol L$^{-1}$ in both years, while Fe-concentrations increased significantly more in 2009 than 2010, with 50 – 115 µmol L$^{-1}$ and 15 – 25 µmol L$^{-1}$, respectively.

Experiment 4: Lowered surface water levels in a floating fen during summer

Two weeks of lowered surface water levels (-15 cm) had no clear effect on the water tables in the floating soils (Fig. 2.2, see Table S2). Water tables were hardly affected by the treatments in July 2010 and July 2011, which were characterised by a precipitation surplus of 1.0 – 1.5 mm day$^{-1}$ at the end of the treatment period (weather station Marknesse: KNMI 2014). In contrast, water tables were significantly lowered in all soils after the treatment in July 2009, when there was an evaporation surplus of about 2.5 mm day$^{-1}$ during the treatment period. It should, however, be noted that it was a small decrease of only 4 cm. In addition to the limited change in water tables, none of the measured biogeochemical conditions changed in the soil pore water of any of the vegetation types (see Table S2), except that Cl- and S-concentrations decreased significantly in all vegetation types in 2011.

Experiment 5: Lowered surface water levels in a non-floating fen during summer

Before the start of the treatments, water tables significantly differed among the three monitored years (Fig. 2.2, see Table S2), with lowest levels of 20 – 30 cm below
the surface in 2010 (when the treatment was preceded by a very dry period) and significantly higher levels in 2009 (0 – -25 cm) and 2011 (+5 – -10 cm). In 2011, most Scor-vegetation was even inundated at the start of the treatment.

Lowering of surface water levels by about 15 cm only led to significantly lower water tables in July 2011 (Fig. 2.2, see Table S2). Despite the decrease of surface water levels in ditches, water tables raised in July 2009 and 2010, due to rather heavy rainfall. These raised surface water levels in 2009 and 2010 had no effect on pH and Fe-, o-PO₄, NH₄- and NO₃-concentrations in soil pore waters (Appendix A, see Table S2). The inundated locations with Scor- and Call-vegetation did, however, show significantly increased alkalinities (about 500 µmol L⁻¹) in their soil pore waters in 2009, as indicated by the interaction effect of area and vegetation type (Fig. 2.6), while Ca-concentrations did not change (Fig. 2.7) and Cl-concentrations even decreased during this treatment (Fig. 2.4).

![Fig. 2.6](image)

Fig. 2.6. Effect of five surface water level treatments on the alkalinities (mmol L⁻¹) in four vegetation types during the monitored years. Values were measured two days before (black lines at the left of each triplet), during (grey lines; only in experiment 2) and two days after the treatments (black lines at the left of each triplet). See the caption of Fig. 2.2 for a description of the abbreviations. Sample means (white centers of a line) are given with their standard deviations (n = 5). Statistical information is provided in Table S2 (Supporting Information of online article).
Although surface water levels were also raised by 4 – 6 cm after the treatment in July 2011, due to two days of rainfall (about 25 mm day\(^{-1}\)) after the end of the treatment, the lowered surface water levels did lead to lower water tables during the treatment (Fig. 2.2, see Table S2). During this treatment, water tables were lowered by 10 – 15 cm in all vegetation types. These lowered water tables led to an increase of the redox potential (E\(_h\)) from around -200 to +500 mV in the upper 5 cm of Scor-soils (Fig. 2.5). In contrast, E\(_h\) did not change in the upper 10 cm of Sph- and Moor-soils, since E\(_h\) was already above +600 mV in these topsoils before the start of the treatment. In all vegetation types, E\(_h\) decreased immediately during the two days of rainfall after the end of the treatment. However, during the episode of lowered E\(_h\), no changes in pH, alkalinity, Ca-concentrations or nutrient concentrations were observed in the soil pore waters of any of the vegetation types (see Table S2).

**Fig. 2.7.** Effect of five surface water level treatments on the Ca-concentrations (μmol L\(^{-1}\)) in four vegetation types during the monitored years. Concentrations were measured two days before (black lines at the left of each triplet), during (grey lines; only in experiment 2) and two days after the treatments (black lines at the left of each triplet). Sample means (white centers of a line) are given with their standard deviations (n = 5). See the caption of Fig. 2.2 for a description of the abbreviations. Statistical information is provided in Table S2 (Supporting Information of online article).
Discussion

Water tables in floating fens hardly depend on surface water levels

In floating fens, water tables changed only a few centimeters during raised (+10) and lowered (-15) surface water levels, but these changes were caused by weather conditions (precipitation and evapotranspiration) rather than by treatments. As hypothesized, fluctuations in surface water levels had almost no effect on water tables in floating fens with Calliergonella- and Sphagnum-dominated vegetation, since the buoyant peat followed the surface water levels. This was not only the case during short-term experiments of two weeks, but also occurred during a similar surface water level rise of three months (field observation of C. Cusell). As a result of the limited change in water tables, ANC and nutrient concentrations in soil pore waters did not change during the field experiments, not even after three months of lowered or raised surface water levels (Cusell et al. 2013).

It has, however, been reported that lowered surface water levels may lead to lower water tables in floating fens, especially when soil thickness increases (e.g. van Wirdum 1993). Similarly, it has also been shown that raised surface water levels may lead to inundations in floating fens (O’Connell 1981; Koerselman 1989; van Wirdum 1991), especially on rich fens with Scorpidium spp. (Cusell et al. 2013). Such rich fens are usually located at or below the water table, instead of clearly above, like Sphagnum-dominated fens. Although there is still debate about the origin of this inundation water, which may be seepage of surface water from beneath the floating root mat (van Wirdum 1991) or flooding of surface water (Cusell et al. 2013), it is clear that floating rich fens may get inundated when surface water levels get sufficiently high. The absence of inundation in our floating fens may thus solely be caused by the limited surface water level rise of only 10 cm and the high buoyancy of the floating fens studied, but may also reflect the absence of rich fens with Scorpidium spp. in these floating fens.

Short periods of lowered surface water levels do not lead to acidification or eutrophication

Non-floating fens did not respond uniformly to surface water level drawdowns in summer (-15 cm), since weather conditions also affected the water tables in these fens. Water tables only dropped once during the three monitored treatments, which was the only year in which the treatment period coincided with an evapotranspiration surplus. Under these conditions, water tables dropped 10 – 15 cm in non-floating fens, while levels only dropped 4 – 6 cm in floating fens. During this water table drop of about 12 cm in non-floating fens, redox potentials ($E_h$) increased from around -200 to +500 mV in the upper 5 cm of Scor-soils, which indicates that the lowered water tables led to the entry of oxygen into these soils (e.g. Gambrell & Patrick 1978; Rowell 1981). In Sph- and Moor-vegetation, oxygen availability in topsoils was already high at the start of the
treatment, as shown by the initial $E_h$-levels of above +600 mV. The intrusion of oxygen in the upper part of soils during two weeks of lowered surface water levels did, however, not lead to acidification or eutrophication. It is, however, well known that longer episodes with lowered water tables can stimulate net mineralization rates (Grootjans et al. 1986; Bridgham et al. 1998; Olde Venterink et al. 2002a) and acidification by aerobic oxidation processes (Lamers et al. 1998a; Lucassen et al. 2002).

Flooding of non-floating fens

In non-floating fens, which did not follow the surface water level, high water tables clearly led to inundation, due to fixation of these fens to the sandy substrate. This is supported by diver data (Cusell et al. 2013). The high Cl-concentrations in the inundation water compared to the soil pore waters showed that this inundation water originated from the adjacent ditches. Plots with highest water tables before the treatment, which were often dominated by Scorpidium spp. or $H. vernicosus$, showed the largest rise in water tables during these floodings. This is presumably because rich fens are usually situated in depressions which are 5 – 10 cm lower than the surface of Sphagnum-dominated vegetation, where water tables often bulge somewhat.

Effect of winter flooding on the ANC depends on infiltration rates

The absence of change in Cl-concentrations in soil pore waters during most winters, despite higher Cl-concentrations in the flooding water, shows that there was no or hardly any infiltration into the waterlogged soils. Infiltration only occurred when water tables were lower than 5 cm below the surface before the start of the flooding. This is in accordance with Hooijer (1996) and Banach et al. (2009), who also found absence of or limited infiltration of flooding water in waterlogged riverine floodplain fens.

Under non-infiltrating conditions in winter, two weeks of flooding had no effect on the alkalinity and Ca-concentration of soil pore waters. Neither infiltration of $HCO_3^-$, Ca and Mg (ANC input) nor increased anaerobic reduction rates in the peat (internal ANC generation) occurred. However, longer-term inundations during winter can lead to both forms of ANC-increase in waterlogged (rich) fen soils, as was demonstrated in a mesocosm experiment (Chapter 3).

Under infiltrating conditions during the flooding in the winter of 2011, alkalinity and Ca-concentration in soil pore water did increase by 50 – 100% during two weeks of flooding, but only in Scor- and Call-soils. Since alkalinity and Ca-concentration increased at a ratio of 2:1 in these plots, the increase in alkalinity is probably mainly caused by the infiltration of Ca- and $HCO_3^-$-rich flooding water, and not by microbial-induced anaerobic reduction processes. Despite redox potentials ($E_h$) of below -200 mV throughout the soil profile of Scor-vegetation, at which Fe(III)- and $SO_4^{2-}$-reduction will lead to internal alkalinity generation (e.g. Ponnamperuma 1984; Mitsch & Gosselink 2007), unchanged Fe- and $SO_4^{2-}$-concentrations in soil pore waters support
the idea of limited occurrence of microbial alkalinity generation. This is most probably caused by the low temperatures in winter and the subsequent low microbial activity (Loeb et al. 2008a).

For Sph- and Moor-vegetation, flooding led to an immediate decrease of $E_h$ from values around +600 mV, indicative of the presence of oxygen (e.g. Gambrell & Patrick 1978; Rowell 1981), to around -100 mV. Similar to Scor-soils, these anaerobic conditions did not result in internal alkalinity generation. However, unlike Scor-vegetation, infiltration of the flooding water did also not lead to an increase of alkalinitities and Ca-concentrations in soil pore waters of Sph- and Moor-vegetation. This was related to lower alkalinities and Ca-concentrations in the water layer above Sph- and Moor-vegetation compared to Scor- and Call-vegetation. The difference in flooding water composition at a relatively short distance (10 – 20 meters) can only be explained by the exchange of Ca$^{2+}$ for H$^+$ between flooding water and the adsorption complex of living mosses and their peat. This process has already been described regularly for non-flooded conditions in Sphagnum-dominated fens (Clymo 1963; Kooijman & Bakker 1994; Soudzilovskaia et al. 2010). Acidification (alkalinity consumption) of flooding water will mainly occur at Sphagnum-dominated sites, because adsorption complexes of Scorpidium spp. and C. cuspidata are probably already saturated with Ca before the flooding, while adsorption complexes of Sphagna often contain a fair amount of H$^+$.

**Effect of flooding on the ANC depends on season**

The increase of alkalinities and Ca-concentrations in soil pore waters after two weeks of flooding of non-floating fens with Scor- and Call-vegetation was much larger in summer than in winter, even when infiltration occurred in winter. Under summer conditions, alkalinity and Ca-concentration in Scor- and Call-soils increased by about 1900 µmol L$^{-1}$ and 450 µmol L$^{-1}$ instead of 350 µmol L$^{-1}$ and 150 µmol L$^{-1}$ in winter. This clear seasonal difference can likely be explained by higher evapotranspiration in summer, which facilitates the infiltration of base-rich flooding water.

Since alkalinity and Ca-concentration increased at a ratio of 4:1 instead of 2:1, the increase in alkalinity in summer is presumably not only caused by the infiltration of base-rich flooding water but also by other processes, i.e. evaporative concentration and microbial alkalinity generation. As long as water tables became sufficiently high (between 1 and 10 cm below the surface) in summer, raised surface water levels even led to increased alkalinities, Ca- and Cl-concentrations under non-inundated conditions. In these cases, increases in Ca- and Cl-concentrations cannot be caused by infiltration and are probably due to evaporative concentration. In addition, alkalinity production is higher during floodings in summer than in winter as a result of increased microbial alkalinity generation in the warmer peat soil, due to the absence of oxygen and use of alternative electron acceptors such as $SO_4^-$ and Fe(III). The simultaneous decrease in $SO_4^-$-concentrations ($SO_4$-reduction) and increase in Fe-concentrations (mobilization of Fe(II)) indeed support this theory (Stumm & Morgan 1996). It
must, however, be noted that increase in alkalinity by concentration and microbial-induced anaerobic reduction processes may well be temporary since aerobic oxidation during subsequent episodes with lower water tables in fens can lead to the opposite process of acidification (Lamers et al. 1998a; Loeb et al. 2008a).

**Short floodings with P-poor water do not lead to P-eutrophication**

P-eutrophication did not occur in any of the experiments. It must, however, be noted that the flooding water contained very low orthophosphate concentrations of 0.05 μmol L\(^{-1}\), excluding the occurrence of P-eutrophication by high P-inputs (external eutrophication). Other studies show that flooding with P-rich surface water may well lead to P-eutrophication in fens (e.g. Wassen et al. 1996). Although P-uptake by vegetation may somewhat mask P-eutrophication, especially in summer, the field experiments also showed no evidence of increased internal P-mobilization in waterlogged soils during two weeks of flooding. Several other experiments have, however, shown that prolonged inundation may well lead to internal P-mobilization in waterlogged soils (Patrick & Khalid 1974; Loeb et al. 2008a), especially in P-rich fens (Chapter 3) and with SO\(_4\)-rich flooding water (Lamers et al. 1998b). In the present study, the duration of two weeks seems to be sufficiently short and P-concentrations in flooding seem to be sufficiently low to induce P-eutrophication. However, P-eutrophication can certainly occur during floodings when soil quality and/or surface water quality are insufficient.

**Implications for the management of biodiverse fens**

Rich fens, which contain many threatened vascular plants and bryophytes, can only persist under well-buffered and nutrient-poor conditions (e.g. Sjörs 1950; Wheeler & Proctor 2000; Kooijman & Paulissen 2006). In wetlands with more or less stable surface water levels, the re-introduction of fluctuating levels has recently been proposed to restore the ANC and to reduce P-eutrophication, in order to conserve or restore fen biodiversity.

Our large-scale field experiments suggest that two weeks of raised surface water levels (+10 cm) may counteract acidification of base-rich fens by increasing the ANC, but only under specific conditions. A rise in surface water levels should lead to inundation. Although some studies report inundations of floating fens with *Scorpidium* spp. (O’Connell 1981; van Wirdum 1991; Cusell et al. 2013), this study showed that a limited rise in surface water levels did not lead to inundation of the floating *Sphagnum*-dominated fens. As a consequence, ANC and nutrient concentrations did not change in these floating *Sphagnum*-dominated fens. In contrast, in non-floating fens, two weeks of raised surface water levels did lead to flooding. The ANC, however, only increased when base-rich flooding water was able to infiltrate into the soil. In winter, this only occurred when the water table was lower than 5 cm below the surface before the start of the treatment. In summer, infiltration is facilitated by higher evapotranspiration,
which is supported by the outcome of long-term mesocosm experiments (Chapter 3). Furthermore, higher temperatures in summer led to internal alkalinity generation in non-floating fens as a result of Fe(III)- and SO$_4^-$-reduction, although this effect will be temporary since aerobic oxidation during subsequent episodes with lower water tables can lead to acidification (Lamers et al. 1998a; Loeb et al. 2008a).

Two weeks of moderate summer drought as a result of lowered surface water levels (-10 cm) did not have severe effect on biogeochemical conditions, and the hypothesized acidification or eutrophication did not occur. Longer periods with lowered surface water levels can, however, stimulate net mineralization and acidification. It is therefore advised to prevent longer periods of low water tables as much as possible.

Overall, it can be concluded that short periods of raised surface water level and inundation can be profitable for rich fens as long as surface waters are nutrient-poor and infiltration occurs. This management tool is most suitable for non-floating fens and may best be used in summer when temperatures and evapotranspiration are higher than in winter. Short-term periods with intense precipitation, which are very likely to occur more frequent in future, due to climate change (e.g. Bronstert 2003; Kundzewicz et al. 2006), can thus have a positive effect on rich fens, especially if it occurs in summer. On the other hand, periods with low water tables should be prevented as much as possible.

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