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Effects of long-term flooding on biogeochemistry and vegetation development in floodplains – a mesocosm experiment to study interacting effects of land use and water quality

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Abstract

The frequent occurrence of summer floods in Eastern Europe, possibly related to climate change, urges the need to understand the consequences of combined water storage and nature rehabilitation as an alternative safety measure instead of raising and reinforcing dykes, for floodplain biogeochemistry and vegetation development. We used a mesocosm design to investigate the possibilities for the creation of permanently flooded wetlands along rivers, in relation to water quality (nitrate, sulphate) and land use (fertilization). Flooding resulted in severe eutrophication of both sediment pore water and surface water, particularly for more fertilized soil and sulphate pollution. Vegetation development was mainly determined by soil quality, resulting in a strong decline of most species from the highly fertilized location, especially in combination with higher nitrate and sulphate concentrations. Soils from the less fertilized location showed, in contrast, luxurious growth of target *Carex* species regardless water quality. The observed interacting effects of water quality and agricultural use are important in assessing the consequences of planned measures for ecosystem functioning (including peat formation) and biodiversity in river floodplains.

1 Introduction

In riverine regions in Eastern Europe, both the frequency and the severity of flooding has increased in the last decades (Bronstert, 2003; Mitchell, 2003), not only in regulated rivers systems but also in more pristine rivers such as the Vistula and Odra in Poland (Kundzewicz, 2005; Kundzewicz et al., 2005). In addition, there has been a shift of the inundation period (Kundzewicz, 2005; EEA, 2007) from winter (due to snow melting) to summer (related to extreme precipitation events). These changes are probably related to climatic change resulting in an acceleration of the hydrological cycle (Milly et al., 2002; Christensen and Christensen, 2003; Kundzewicz, 2005; Kundzewicz et al., 2005). For the future, heavy summer precipitation is expected to increase in North-Eastern Europe (Beniston et al., 2007).

Because flood prevention by raising dykes seems to be insufficient in the future, new strategies have been proposed which allow creating more space for floodwater by dyke replacement and creation of secondary channels. These aim to combine several goals including safety, the restoration of other floodplain functions (land accretion, recreation, water storage), and nature restoration of both riparian wetlands and permanently flooded marshes (Smits et al., 2000; van Stokkom et al., 2005). It is, however, difficult to optimally combine all goals, and restoration projects often result in low biodiversity as a result of eutrophication (Antheunisse et al., 2006; Lamers et al., 2006). In addition, it is a problem to restore peat forming vegetation (e.g. *Carex* species), which is important to counteract the effects of land subsidence (mechanical compression and oxidation) resulting from agricultural drainage and eutrophication (Wösten et al., 1997; Schipper and McLoed, 2002). Moreover, increased decomposition has unwanted side effects such as an increase of carbon dioxide (CO_2) emission to the atmosphere and additional eutrophication (Kool et al., 2006; Lamers et al., 2006; Schipper et al., 2007).

The development of vegetation in floodplains is strongly determined by flooding characteristics (Vervuren et al., 2003; van Eck et al., 2004, 2005) and vegetation tolerance (Blom et al., 1990; van Eck et al., 2004). Oxygen deprivation, light limitation and reduced CO_2 availability restrict the metabolic efficiency (Setter et al., 1989; Mommer et al., 2005; Banach et al., 2009a) and therefore lead to biomass reduction (Pezeshki, 2001; van Eck et al., 2004). Wetland species have developed a number of adaptations (Armstrong, 1972; Justin and Armstrong, 1987; Blom and Voesenek, 1996; Mommer and Visser, 2005; Banach et al., 2009a) to cope with these stress factors including the ability to oxidize the rhizosphere (Pezeshki, 2001; Colmer, 2003).

Next, the changed biogeochemistry as affected by the combined effects of changes in water quality, soil quality and hydrological regime may also form a constraint for successful rehabilitation of riverine wetlands. The microbially governed processes in soil strongly depend on the soil aeration state, and after inundation oxygen is depleted resulting in the mobilization of reduced (often toxic) substances such as nitrite, ammonium and sulphide. As a result of iron reduction, phosphate is mobilized during flooding (Ponnamperuma, 1984; Gliński and Stępniewski, 1985; Laanbroek, 1990; Smolders et al., 2006; Banach et al., 2009b). In addition, greenhouse gases such as CO_2, nitrous oxide (N_2O) and methane (CH_4) may be produced (Yu et al., 2006). As soil microbial activity accelerates with temperature, summer inundation is expected to have much more impact on biogeochemical processes than winter inundation (Loeb et al., 2008b).

Both the composition of the flood water and soil characteristics may strongly interact with the biogeochemical effects of flooding (Swarzenski et al., 2008). Pollution with sulphate (SO_4^{2-}) may lead to increased P availability, by the interaction between produced sulphide (H_2S) and Fe-P cycling (Sperber, 1958; Roden and Edmonds, 1997; Lamers et al., 1998, 2002a; Zak et al., 2006) and by competition between SO_4^{2-} and phosphate (PO_4^{3-}) for anion binding sites (Caraco et al., 1989). As SO_4^{2-} reduction generates alkalinity, decomposition and mineralization may increase even further (Roelofs, 1991; Smolders et al., 2006). The extent of P mobilization and ammonium (NH_4^+) accumulation, and the possible additional effect of SO_4^{2-} during flooding largely depend on soil quality (Loeb et al., 2007). High levels of dissolved Fe can bind both PO_4^{3-} and H_2S, preventing P-related eutrophication and H_2S toxicity (Smolders et al., 1995; Lamers et al., 2001). The level of PO_4^{3-} mobilization has been shown to be related to the saturation of binding sites in the amorphous Fe pool rather than to the concentration of PO_4^{3-} (Young and Ross, 2001; Loeb et al., 2008a). Although phosphorous is also bound to Al and $CaCO_3$, these fractions are redox independent (Boström, 1988; Lamers et al., 2002b, 2006; Geurts et al., 2008). If high concentrations of nitrate (NO_3^-) are present, they can prevent the reduction of Fe and SO_4^{2-}, as NO_3^- is a more favourable electron acceptor acting as a redox buffer, and reduce PO_4^{3-} mobilization rates (Lucassen et al., 2004).

Both the level of eutrophication and the accumulation of potentially phytotoxic compounds may influence the vegetation of riverine wetlands by the die-off of characteristic species and development of fast-growing plants outcompeting others (Roelofs, 1991;

Lamers et al., 1998; Kotowski et al., 2006; Geurts et al., 2008). On the other hand, eutrophication may also lead to higher biomass production rates, which diminish toxicity effects due to dilution of these compounds in tissues, or due to stronger rhizosphere oxidation (Geurts et al., 2009). The development is not only important with respect to biodiversity, but also with respect to the rate of land subsidence or land accretion (Rooth and Stevenson, 2000; Rooth et al., 2003). In eutrophic systems, decomposition and land subsidence may dominate, while in less eutrophic systems, such as those dominated by *Carex* species, net carbon fixation may lead to land accretion (Portnoy, 1999).

The aim of this study was to investigate the possibilities for the creation of permanently flooded wetlands (marshes) along rivers, in relation to flood water quality (NO_3^-, SO_4^{2-}) and soil use (level of fertilization in the past). In order to study the effects under controlled conditions, a mesocosm design using intact sods was used. The results with respect to biogeochemistry (especially C, Fe, P, N and S cycling) and vegetation development will be discussed in relation to water management and nature management.

2 Materials and methods

2.1 Field information

The location where the sods were collected, Kosiorów village (51°13′ N; 21°51′ E; Fig. 1), is located close to the Chodelka River, a tributary of the Vistula River in Poland. This site was selected because of the plans of the local authorities to create a retention reservoir in this area to avoid flooding risks, as the Chodelka River is known to take back-flowing water from the Vistula during high peaks of water discharge (Banach et al., 2009b). Along this river there are several meadows which show different histories of cultivation. Two neighbouring meadows were selected of which one is heavily fertilized and mowed twice a year for hay-making (referred to as hayland, HAY), whereas the other is less fertilized and used only for grazing (pasture, PAS) at a low density of

1 animal per hectare.

Both meadows show a peaty soil type (upper 20 cm: 40–50% organic matter; with the inorganic fraction comprising 38–44% sand, 8–12% silt and 4% clay, Banach et al., 2009b), with the average water table 30 cm below soil surface. Both soils are, however, nutrient-rich, as concentrations of plant available P (Olsen P) and NO_3^- are very high in both soils (Banach et al., 2009b). In addition, total concentrations of S and Fe are high. There were significant differences between these soils with respect to moisture, organic matter content, concentrations of total S, NO_3^-, Olsen P, labile P and Fe/Al bound P fractions (Table 1). At the onset of the experiment, both locations were covered by species-rich terrestrial vegetation dominated by *Deschampsia cespitosa* L. and *Holcus lanatus* L. (Table 2).

2.2 Experimental design

For studying the effects of long-term inundation, 40 sods were collected in total, with standing vegetation. After transportation to the Netherlands in plastic containers (to avoid desiccation), the sods were placed in a greenhouse. Each sod was fitted into a separate glass container (25×25×30 cm) at an air humidity of 40–90%, under natural light and temperature conditions. The sides of the compartments were covered with black foil to avoid light influence on the sides of the soil.

Four different floodwater mixtures were prepared based on field data including a treatment with increased concentrations of nitrate (N), sulphate (S) or their combination (SN), all at the level of 1000 μmol l^{-1}. The control (Cfl) had pristine river water quality characterized by low levels of nutrients (Table 3). In addition, non flooded, moist controls were used (Cm) as a contrast. Each treatment consisted of 4 replicates which were randomly distributed over the 40 units (20 per meadow type). The sods were kept inundated at 20 cm above soil level for 9 months (January till November). Non flooded controls were watered with artificial rainwater containing 5 mg l^{-1} of sea-salt, (Wiegandt GmbH, Krefeld, Germany) in order to keep the groundwater level at 10 cm below soil surface to avoid desiccation of the sods during this period.

2.3 Measurements and chemical analyses

Soil samples were analysed for soil moisture percentage (drying samples at 105°C for 24 h) and organic matter content (loss-on-ignition, 550°C, 4 h). Levels of nutrients were examined in fresh soil samples (corrected afterwards for moisture content) by extraction: Olsen P (as an estimate of plant available P), NaCl-extractable ammonium and water extraction (Banach et al., 2009b). The concentration of amorphous iron was estimated by oxalate extraction (Schwertmann, 1964) whilst soil P fractions were estimated using the method described by Golterman (1996). In addition, total element concentrations were measured after digestion of 200 mg samples in a mixture of concentrated HNO_3 and 30% H_2O_2 (4+1 ml) using a Milestone microwave MLS 1200 Mega system (Sorisole, Italy).

Two sediment pore water samplers (Rhizon SMS-10 cm; Eijkelkamp Agrisearch Equipment, Giesbeek, the Netherlands) were placed diagonally at 5–10 cm depth connected to black silicone tubes for monitoring of sediment pore water chemistry in each container. Samples were collected anaerobically by means of 50 ml vacuumed syringes. After discarding the first 10 ml (stagnant water), collected subsamples were pooled for other measurements.

Free (dissolved) sulphide (H_2S) in sediment pore water was estimated in 10.5 ml of subsample fixed immediately after collection with 10.5 ml of sulphide antioxidant buffer (Van Gemerden, 1984). For this measurement a sulphide ion-selective Ag-electrode and a double junction calomel reference electrode were used (Roelofs, 1991).

Titration of 10 ml of sample with 0.01 M HCl down to pH 4.2 allowed us to determine alkalinity (TIM800 pH-meter with the above mentioned pH-electrode and an ABU901 Autoburette, Radiometer Copenhagen, Denmark) preceded by pH measurement. In surface water, turbidity (nephelometric turbidity units, NTU) was estimated using a WTW turbidity meter Turb550 (Weilheim, Germany). The remaining volumes were filtered over a Whatman microfiber filter type GF/C (Whatman, Brentford, UK) after which citric acid was added (to a final concentration of 0.125 $g\,l^{-1}$) to avoid precipitation of metals, and stored in 100 ml iodated polyethylene bottles at −28°C until further analysis.

The concentrations of NO_3^-, NH_4^+ and soluble reactive phosphorus (SRP) were determined by means of an Auto Analyser 3 System (Bran+Luebbe, Norderstedt, Germany) according to standard procedures (Banach et al., 2009b) followed by correction for colour (at 450 nm) caused by humic substances (Shizmadzu UV-120-01 spectrophotometer, Kyoto, Japan). The total concentrations of Fe, Ca, K, P, and S were analysed by means of inductively coupled plasma optical emission spectrometry (ICP-OES, IRIS Intrepid II, Thermo Electron Corporation, Franklin, MA, USA). At the (relatively high) concentrations used in this experiment the total S concentrations in the water layer provided a good estimate of SO_4^{2-}, because only a small percentage of the element is present in organic form. This was verified by parallel analysis of various samples for different treatments using capillary ion analysis (Waters Technologies), in which SO_4^{2-} concentrations were shown to match the total S concentrations within the uncertainty of both methods.

Total concentrations of inorganic carbon (TIC, sum of CO_2 and HCO_3^-) and CH_4 were determined by collecting pore water samples into vacuumed infusion flasks (30 ml) and correcting for the headspace volume. Concentrations of gases were measured using an infrared gas analyser (ABB Advance Optima IRGA, Zürich, Switzerland).

2.4 Vegetation description

The vegetation present on the sods was described in detail (number of individuals and their cover for each species) before the onset of submergence and at the end of the experiment. We divided the plants into 3 groups: grasses (G), *Carex* species (C) and herbs (H) (see Table 2). In addition, we determined the total cover of plants and algae (in %) during water sampling. Vegetation and algae were harvested 6 months after the onset of submergence and at the end of the study. Collected material was dried at 70°C for 48 h, weighed (dry weight) and analyzed for total concentrations of selected elements (ICP, see above) after microwave digestion (see above). Total concentrations of C and N were estimated in 2 mg of homogenized dry material using a Carlo Erba NA1500 elemental analyzer (Thermo Fisher Scientific, MA, USA). For nutrient ratios, weighted means of the separate plant groups were used.

2.5 Data analysis

All data were statistically processed by means of SPSS for Windows (SPSS 15.0, 2006, Chicago, IL, USA). Biogeochemical variables were ln(x+1) transformed in order to make the data fit better to the normal distribution and to make the variances less dependent of the sample means. Vegetation cover and algae cover were arcsin sqrt transformed and species number was log(x+1) transformed.

Relationships between variables (only for flooded) were tested by calculating Spearman's rho correlation coefficients (r_s) due to differences between sizes of variables, and regression lines were fitted (with R^2 statistic). Asterisks indicate significance of correlations: * $-p<0.05$, ** $-p<0.01$, *** $-p<0.001$.

Changes in sediment- and water-related variables as well as vegetation and algae data in time were tested in a step-wise procedure. First, a comparison of flooded versus non-flooded treatments was performed, followed by a comparison between all flooded treatments. A repeated measures ANOVA, model mixed designs (GLM 5), procedure was used in both cases for both tested meadows. If the assumption of sphericity was not met, an appropriate correction was used according to the values of the Greenhouse or Huynh-Feldt test statistics (Field, 2005). Tukey HSD (homogeneity of variances assumed) or Games-Howell procedure (homogeneity of variances not assumed) were used as post hoc tests. In addition, data from the end of the study were analyzed by means of ANOVA (with Tukey or Gamess-Howell post-hoc tests). The same test was used for plant tissue nutrient ratios.

Differences between soil characteristics were assessed using independent samples t-test. Significance was accepted at p-value ≤ 0.05. For better clarity, all data are presented as means of non-transformed variables ± standard error of the mean (SEM).

3 Results

3.1 Soil response to flooding

Initially (one week before flooding), tested soils had a pH of 5–6 with low alkalinity ($<$0.5 meq l^{-1}) and low levels of NH_4^+, Fe^{2+}, SRP and TIC in the sediment pore water. Concentrations of NO_3^- were high and differed between both meadows: around 5000 μmol l^{-1} in the hayland (HAY) and 3000 μmol l^{-1} for the pasture (PAS). Concentrations of SO_4^{2-} were below 1000 μmol l^{-1} in both soils (Figs. 2, 3).

Figures 2–5 and Tables 4–5 present the effects of the flooding on biogeochemical processes. The dissolved NO_3^- pool in sediment pore water declined by 3–5 times one week after inundation remaining significantly higher than Cm (Table 4b), without effect of soil use (p=0.72). NO_3^- concentrations in surface water differed only initially at one week after flooding due to treatment (Fig. 4, Table 4b). There was a strong nitrate reduction to levels comparable to N-poor waters (lower than 40 μmol l^{-1}), except for the end of the treatment period. Concentrations of NH_4^+ in sediment pore water showed an opposite trend in time compared to NO_3^-, interacting with soil use (TxL, Table 4b) by a strong increase to a peak of 300–600 μmol l^{-1} in HAY and 100–200 μmol l^{-1} in PAS after 16 weeks (Table 4b). In addition, NH_4^+ levels in the surface water increased (Fig. 4) differing between tested soils (Table 4b). HAY soil showed much stronger NH_4^+ mobilization (up to 50 μmol l^{-1}) then PAS ($<$20 μmol l^{-1}). Moreover, we observed a significant effect of water quality on NH_4^+ levels for HAY; the highest peak of 50 μmol l^{-1} was for SN followed by S, N (10–20 μmol l^{-1}) and Cfl ($<$10 μmol l^{-1}) treatment (Table 4b).

Concentration of SO_4^{2-} in sediment pore water differed in time interacting with flooding treatment and water quality (TxI and TxW, Table 4 and Fig. 2). Initial higher levels above 1 mmol l^{-1} in S and SN treatments were further reduced to values between 500–1000 μmol l^{-1}. We found significantly higher values for SN and S treatments in comparison to N, Cfl, and Cm, which did not differ from each other and remained below

500 μmol l^{-1}. We did not observe differences in SO_4^{2-} between soils (p=0.89, Table 4). While Cm did not show H_2S accumulation in the sediment pore water, higher levels of this compound (2–12 μmol l^{-1}) were recorded in S and SN treatments (Table 4). The concentrations of SO_4^{2-} in the surface water were related to both water composition and soil use (Table 4) decreasing from the levels of 1200–1600 μmol l^{-1} to about 400 μmol l^{-1} (S and SN treatments), especially for HAY.

Inundation led to strong mobilization of Fe^{2+} into sediment pore water after 5 weeks with a peak of 400–500 μmol l^{-1} after 9 (HAY) and 15 (PAS) weeks (Fig. 2). The concentrations of Fe^{2+} in the surface water differed between tested soils (Table 4): HAY showed stronger mobilization (160 μmol l^{-1}) than PAS (70 μmol l^{-1}). We did not detect significant differences between water quality treatments (p=0.12, Fig. 4, Table 4). There was a concomitant and rapid SRP mobilization to extremely high levels of 50–100 μmol l^{-1} one week after flooding. The amount of mobilized SRP depended on water quality; SRP levels in sediment pore water were higher for S and SN than for the others. The response differed between both tested soils: HAY showed continuous release of SRP with maximum of 200 μmol l^{-1}, while for PAS the concentration of SRP decreased after 12 weeks from 150 to 50 μmol l^{-1} at the end of the experiment, and became comparable to levels in other flooded treatments (Fig. 2, Table 4). The raised levels of SRP in the sediment pore water led to P release into the surface water above, increasing over time with differential responses for both soils. HAY treatments showed values up to 40–120 μmol l^{-1} for S and SN treatments (Fig. 4), in contrast to values of 20–50 μmol l^{-1} for HAY (S, SN). Treatments without S enrichment showed, however, much lower levels of SRP (<15 μmol l^{-1} in HAY and <2 μmol l^{-1} in PAS). Flooding resulted in an increase of water turbidity to 15–25 NTU without significant effects of the water composition (p=0.75), but with higher values for HAY than for PAS.

Inundation led to elevated levels of Ca^{2+} in sediment pore water, 2–6 times higher in comparison to about 1 mmol l^{-1} in Cm, changing over time (Fig. 3). S presence led to lower concentrations of Ca^{2+} in sediment pore water as compared to Cm and N treatments. Concentrations of Ca^{2+} in the surface water increased 2–2.5 times above the initial value of 1 mmol l^{-1} during inundation without effect of soil use (p=0.10, Table 4). However, water quality influenced Ca^{2+} levels favouring its release at low S levels (Fig. 5). Also K^+ concentrations in sediment pore water increased to 200–300 μmol l^{-1} after flooding in comparison to Cm, with higher values for HAY than for PAS (Fig. 3). In general, water quality had no significant effect of K^+ concentrations in sediment pore water (p=0.21, Table 4). Also in the surface water, K^+ showed significant changes over time (Table 4), with different patterns for both soils; there was an increase in HAY and decrease in PAS. We did not find a significant influence of water quality on K^+ mobilization to the surface water (p=0.52).

Sediment pore water pH increased very fast after inundation (Table 4a) to values of 6.5–7.5 and remaining significantly higher in comparison to Cm during the whole period. We observed a stronger increase of pH in HAY than PAS and significant differences between water qualities: SN treatment had significantly higher pH for HAY, whilst PAS showed higher pH for SN, S and N treatments in contrast to Cfl (Table 4b). The pH of the surface water rose from 7 to 7.5–8; the highest value was measured for SN (Table 4). These changes in pH coincided with strong alkalinization of the sediment pore water and surface water during flooding reaching high levels, particularly for PAS soils and SN treatments (Fig. 3, Table 4a). The increase of the alkalinity was related to the accumulation of TIC in the sediment pore water in time. Anaerobic soil conditions also led to CH_4 accumulation in sediment pore water with maximum peaks of 200–1000 μmol l^{-1} for all treatments. There were no effects of soil use (p=0.25), but CH_4 concentrations were higher in N, S and SN in comparison to Cfl (results not shown).

3.2 Vegetation response

Initially, vegetation cover was 41% for HAY and 39% for PAS (p=0.68), composed of species from different functional groups, such as grasses and herbs. In addition *Carex* species were present, mainly on PAS (Table 2).

Inundation of sods resulted in a significant decline of the vegetation in terms of cover and number of the species in each functional group over time (Table 5a). Cover of the plants decreased by 21% in HAY and 4.8% in PAS after 41 weeks in flooded sods. At this time, plants covered almost the whole surface of non flooded sods (Cm) from both meadows (98 and 93%, Fig. 6). Cm sods had a high number of individual plants for herbs, grasses (GxI) and *Carex* species. Flooding changed this composition, leading to a drastic reduction of herbs and relatively stronger development of grasses and *Carex* species (TxGxI). Moreover, species composition differed significantly (GxL) between meadows – HAY was dominated mainly by grasses and herbs with a very low number of individuals of *Carex* species, whilst PAS had much more individuals of herbs and *Carex* and a similar number of individuals of grasses compared to HAY. Observed changes in cover and species composition were not only time-related (TxI) but also depended on land use (higher for PAS, Table 5). We did not observe a significant role of water quality (p=0.18 and 0.21 for cover and biodiversity, Table 5b). There was, however, an initial stimulation of the vegetation growth in N treatments compared to Cfl control and a reduction by S and SN, followed by a decline in all treatments (data not shown) resulting in the final situation presented in Fig. 6. Above-ground total biomass was clearly influenced by interacting effects of flooding and soil use (Table 6, Fig. 6). Flooded plants from HAY had lower biomass whilst those from the pasture were comparable to Cm. Water composition had a significant effect on total biomass in HAY, where biomass was very low for the N, S and SN treatment. This adverse effect was not found for PAS.

Herbs suffered the most due to flooding, and we noticed a strong decline of *Cardamine* spp. (PAS), *Sanguisorba officinalis* (HAY), *Centaurea jacea*, *Filipendula ulmaria*, *Rumex acetosa*, *Trifolium repens*, *Veronica chamaedrys* and *Vicia* spp. (HAY and PAS). For grasses only *Anthoxanthum* spp. (HAY) and *Poa pratensis* (PAS) declined. Other species were flooding-resistant, showing either survival or even stimulation of growth (Table 2).

Above-ground biomass of grasses was significantly lower (Table 6a) at the end of the study in comparison to summer (week 24) and it was lower in flooded sods in comparison to Cm. This decline was stronger in HAY than PAS including non flooded controls. S and SN treatments had the strongest impact, followed by N, leading to significantly lower biomass of grasses for both meadows (Table 6b). Biomass of *Carex* species, which are a target for ecosystem rehabilitation, was significantly higher in flooded sods (Table 6a), especially for PAS, without significant effect of water quality (p=0.25, Table 6b). *Carex* grew well in all treatments on PAS whilst in HAY these species were present only in Cfl. Biomass of herbs declined due to inundation (Table 6a), but was not affected by water quality (p=0.97, Table 6b). However, we noticed that biomass was higher in nutrient-rich treatments in PAS in comparison to Cfl (LxW) whilst HAY, in contrast, showed reversed tendency.

Inundation of sods led to development of algae up to a cover of 100% of the water surface (Table 5b). There was a significantly higher mean overall algae cover in HAY (49%) than PAS (23%). Development of algae did, however, not depend on water quality (p=0.86, Table 5b).

The mean N:P ratio in plant tissue was 3.9 (HAY) and 2.2 (PAS) for non inundated plants (results not shown). Inundation led to significant changes of this ratio: in PAS it increased to 8.1 and in HAY to 5.4 (p<0.05). The initial P:K ratio (Cm) was 0.16 for HAY and 0.22 for PAS (p=0.15), changing to 0.19 and 0.11 after flooding. We did not observe an influence of water quality on both ratios.

4 Discussion

We showed that inundation of floodplain sediments significantly influenced both soil biogeochemistry and vegetation development, but that the severity of these redox-related changes appeared to be strongly determined by the interactions between soil characteristics, as determined by land use, and water quality.

4.1 Effects of flooding on redox-related processes

Flooding led to strong changes in soil due to a switch from aerobic to anaerobic conditions. Subsequent alternative electron acceptors (NO_3^-, Mn^{4+}, Fe^{3+} and SO_4^{2-}) were reduced leading to the sequential decrease of NO_3^- concentration, mobilization of NH_4^+, Mn^{2+}, Fe^{2+}, and SO_4^{2-} reduction (Figs. 2–3; Ponnamperuma, 1984; Gliński and Stępniewski, 1985; Laanbroek, 1990). These redox-related processes were the main cause of P eutrophication and accumulation of reduced compounds, which may both pose a threat for the biodiversity of the developing vegetation (Lamers et al., 1998; Smolders et al., 2006; Loeb et al., 2008b; Banach et al., 2009b). As the observed eutrophication was not caused by external P input (Table 3), the elevated levels of SRP must have resulted from internal mobilization of accumulated P (so-called internal eutrophication, Roelofs, 1991; Smolders et al., 2006; Banach et al., 2009b). There were two key factors involved in this process: soil characteristics and water pollution with SO_4^{2-}. The more fertilized HAY soil had higher Olsen P levels (Table 1) than PAS which could be responsible for differences between both tested soils (Fig. 2, Table 4). There are two possible sources of P in the soil: inorganically-bound and organically-bound P fractions. As an inorganic source, Fe-bound P is most of importance as SRP can be easily mobilized from this redox-sensitive fraction under anaerobic conditions due to Fe reduction (Patrick and Khalid, 1974; Caraco et al., 1989; Baldwin and Mitchell, 2000; Zak et al., 2004; Loeb et al., 2008a). Indeed, we measured increasing Fe^{2+} and SRP concentrations both in sediment pore water and surface water (Figs. 2–4). Levels of Fe^{2+} in surface water were much lower in comparison to those in sediment pore water due to oxidation of the surface water column (Loeb et al., 2007). The Fe-related P mobilization appeared to be related to the total soil Fe:P ratio (Table 1), which was around the threshold value of 12 mol mol^{-1} below which P is mobilized (Ramm and Scheps, 1997; Geurts et al., 2008). Another indicator, the Fe:PO_4 ratio in sediment pore water, correlated well with SRP in the surface water (r_s=−0.76**, R^2=0.53) in a similar way as found by Smolders et al. (2001) and Geurts et al. (2008), with a threshold value of

3–4 mol mol^{-1} below which SRP is strongly mobilized to the water layer, similar to the values found by others (Lehtoranta and Heiskanen, 2003; Zak et al., 2004). Moreover, Fe:PO_4 ratio differed significantly between tested soils and treatments, PAS having significantly higher (p<0.001) values than HAY resulting in lower P mobilization although Fe^{2+} levels of the sediment pore water were equally high (Table 4, Fig. 2).

Differences in SRP levels between both soils could additionally be explained by the role of Ca^{2+} in P binding as described by Boström et al. (1988). In our study we found significant differences in the concentrations of Ca^{2+}in sediment pore water (Table 4b) and a negative correlation (r_s=−0.49**) with SRP. As Ca^{2+} increased and SRP decreased in sediment pore water for PAS, it can be concluded that there may have been precipitation of SRP with Ca^{2+}.

4.2 Role of water quality on redox processes

Increased SO_4^{2-} concentration of the surface water led to significantly higher P mobilization interacting with soil quality, which did not occur during short-term (1 month) flooding (Banach et al., 2009b). The Fe:PO_4 ratio in sediment pore water was lower for S and SN treatments (below the threshold) suggesting additional internal eutrophication due to SO_4^{2-} influx and its reduction. Produced H_2S apparently interacted with the Fe-P cycle (Golterman, 1995; Roden and Edmonds, 1997; Smolders et al., 2006; Zak et al., 2006) and stimulated P mobilization (especially in HAY). However, both soils were Fe-rich which can explain the low H_2S concentrations in sediment pore water. Although the Fe^{2+} concentration is high, a large part is sequestered as FeS_x and not available for P binding, as indicated by the relatively low (Fe minus S) to P ratio of 6–7 (Table 1). In addition, SO_4^{2-} reduction is known to generate alkalinity, which may play a role in further nutrient mobilization (especially P) as it stimulates decomposition and mineralization (Smolders et al., 2006). This process is expected to be additionally important in this study as we recorded production of alkalinity and TIC, especially for S and SN treatments (Fig. 3, Table 4b). Unexpectedly, the presence of high concentra-

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tions of NO_3^- in the surface water did not prevent P mobilization, as is known to occur in fens related to blocking of Fe reduction by the presence of this more favourable electron acceptor (Lucassen et al., 2004).

4.3 Consequences for vegetation development

Flooding itself is a stress factor for non-wetland vegetation as it drastically changes physiological functioning of plants, such as photosynthesis, respiration and internal transport of nutrients, due to oxygen deficiency and accumulation of reduced compounds (Chen et al., 2005; Banach et al., 2009a). Herbs, the most abundant plant group on the studied meadows were most sensitive to flooding; 8 out of 26 species disappeared.

The vegetation response, however, also appeared to be strongly influenced by the interactions between soil use and water quality. There were striking differences in vegetation development between both meadows (as related to land use), with a very strong decline of the vegetation for HAY and luxurious growth of *Carex* species for PAS in N, S, and SN treatments. This may partly be related to the strong eutrophication of HAY, leading to algal development in the water layer which hampered vegetation growth. In addition, the lower redox potential related to the sequential consumption of alternative electron acceptors (inducing more severe oxygen stress) and the higher concentration of potentially phytotoxic substances in sediment pore water as a result of higher decomposition rates, including H_2S (for S-treated), nitrite (NO_2^-, for N-treated), NH_4^+, and possibly also organic acids may have influenced vegetation development (Roelofs, 1991; Armstrong et al., 1996; de Graaf et al., 1998; Lamers et al., 1998; Lucassen et al., 2003; Van den Berg et al., 2005; Koch et al., 2007). It was, however clear that land use was the main determinant for the development of target (*Carex*) vegetation, and that more eutrophic soils require surface water with low concentrations of both SO_4^{2-} and NO_3^-. The observed development of the vegetation was in strong contrast to the effects of short term flooding (Banach et al., 2009b), where all treatments showed equal reduction in vegetation cover as a result of flooding. Nutrient availability may also directly influence vegetation development and, by competition, diversity. In our study we noticed that inundation led to higher availability of both N, K, and especially P. Based on nutrient ratios in plant tissue, the vegetation on both soils appeared to be N-limited only, as N:P ratios were lower than 12–14 (Koerselman and Meuleman, 1996; Güsewell et al., 2003; Olde Venterink et al., 2003). Although the increased availability of N may have changed the vegetation composition, the negative indirect effects of eutrophication, as explained above, appeared to be far more important for vegetation development.

Plant tissue C:N ratios were 14.6 and 7.9 for HAY and PAS, respectively (results not shown). These values are far below the critical level of 30 (Scheffer et al., 2001), suggesting a relatively strong potential for the decomposition of organic material at both locations. The apparently higher rate of decomposition for HAY could be caused by differences in P availability (Verhoeven and Arts, 1992), as the C:P ratio was significantly lower ($p<0.05$) for PAS (52 instead of 127, results not shown). This stresses the fact that peat formation is not only related to the development of potentially peat forming (*Carex*) vegetation, but also to the actual decomposition rates of its litter, as determined by interacting effects of land use and water quality.

5 Conclusions

Our study showed that the effects of long-term inundation of meadows, as in projects aiming at the restoration of marshes along rivers to increase water storage capacity, are strongly determined by the interactions between land use (level of fertilization) and water quality.

Our work emphasizes the important role of land use (level of fertilization). For heavily fertilized soils, desired vegetation development only seems possible if sulphate and nitrate levels in the surface water are low. Strikingly, development of sedge fens was possible for less fertilized soils even at higher sulphate and nitrate levels, although plant biodiversity was still relatively low and peat formation is less probable due to still high levels of nutrients, presumably leading to high decomposition rates.

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Table 1. Characteristics of both tested soils (μmol l^{-1} of bulk soil except for moisture and organic matter, which are in %).

Characteristics (mean±SEM)	Soil from Hayland		Pasture		Sign.
Moisture	37	(2)	50	(2)	***
Organic matter	41	(3)	54	(1)	***
pH	6.2	(0.1)	6.4	(0.2)	NS
Total S	52 321	(4376)	40 858	(1776)	*
NO_3^{-a}	946	(107)	2236	(232)	***
NH_4^{+b}	279	(76)	180	(24)	NS
Olsen P	3818	(435)	1863	(238)	***
Total P	10 508	(886)	8833	(482)	NS
Amorphus Fe	76 671	(8668)	78 140	(3393)	NS
Total Fe	117 560	(10 480)	100 982	(5094)	NS
Fe:P	11.0	(0.9)	11.6	(0.4)	NS
(Fe-S):P	6.1	(0.6)	6.9	(0.4)	NS
Labile P fraction	90.4	(10)	15.8	(1.3)	***
Fe/Al bound P	1631	(147)	760	(37)	***
Ca bound P	3178	(417)	2981	(119)	NS
Organic P	5609	(484)	5077	(383)	NS

[a] Water extractable nitrate, [b] NaCl extractable ammonium
***$p<0.001$, **$p<0.01$, *$p<0.05$, NS not significant

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Table 2. Plant species present in sods from hayland and pasture (average abundance) at the beginning and the end the experiment. Capital letters after species name represent groups – H – herbs, G – grasses, C – *Carex* species.

Species	Family	Hayland		Pasture	
		before	after	before	after
Achillea millefolium (H)	Asteraceae	0–25	0–5	–	–
Agrostis spp. (G)	Poaceae	0–1	0–50	0	0–5
Anthoxanthum spp. (G)	Poaceae	–	–	0	0–1
Arabidopsis suecia (H)	Brassiceae	0	0–5	0–1	0–5
*Cardamine*spp. (H)	Brassiceae	0	0;<1	0–1	0
Carex hirta (C)	Cyperaceae	0	0–40	0–1	10–50
Carex acuta. (C)	Cyperaceae	0	0–50	0–1	5–50
Centaurea jacea (H)	Asteraceae	0–1	0	–	–
Deschampsia cespitosa (G)	Poaceae	5–75	0–5	5–75	0–50
Festuca rubra (G)	Poaceae	0–5	0–50	5–25	0–1
Filipendula ulmaria (H)	Rosaceae	–	–	0–1	0
Galium boreale (H)	Rubiaceae	0–5	0–10	0–5	0–10
Galium uliginosum (H)	Rubiaceae	0	0–5	0–75	0–50
Galium spp. (H)	Rubiaceae	–	–	<1	0;5
Holcus lanatus (G)	Poaceae	5–50	0–5	0	0–10
Lathyrus pratensis (H)	Fabaceae	0–1	0–5	0–1	0–1
Leontodon autumnalis (H)	Asteraceae	0	0–1	0–25	0–1
Linaria vulgaris (H)	Scrophulariaceae	0–1	0–5	0–1	0–5
Lythrum salicaria (H)	Lythraceae	0	0–40	0	0–5
Mentha arvensis (H)	Lamiaceae	0	0–5	–	–
Plantago lanceolata (H)	Plantaginaceae	0–1	0–10	0–1	0–50
Plantago major (H)	Plantaginaceae	–	–	0	0–1
Poa pratensis (G)	Poaceae	0–5	0–50	–	–
Potentilla reptans (H)	Rosaceae	–	–	5–25	0–10
Ranunculus acris (H)	Ranunculaceae	0	0–5	0	0–50
Ranunculus auricomus (H)	Ranunculaceae	0–1	0–1	0–25	0–1
Ranunculus repens (H)	Ranunculaceae	0–1	0–5	0	0–1
Rumex acetosa (H)	Polygonaceae	0–1	0	0–1	0
Sanguisorba officinalis (H)	Rosaceae	–	–	0–5	0–1
Stellaria graminea (H)	Caryophyllaceae	0–5	0–1	0–5	0–5
Taraxacum officinale (H)	Asteraceae	0–1	0–1	–	–
Trifolium repens (H)	Fabaceae	0–1	0	0–1	0
Veronica chamaedrys (H)	Scrophulariaceae	–	–	0–1	0
Vicia spp. (H)	Fabaceae	–	–	0–1	0

Table 3. Chemical composition (μmol l^{-1}) of artificial flood-water for each treatment. S: sulphate, N: nitrate, Cfl: flooded control.

Salt	SN	S	N	Cfl
$CaCl_2 \cdot 2H_2O$	610	610	610	610
KCl	240	240	240	240
$MgCl_2 \cdot 6H_2O$	75	75	75	75
Na_2SO_4	1000	1000	100	100
$NaNO_3$	1000	25	1000	25
$NaHCO_3$	2000	2000	2000	2000
NaCl	0	975	1800	2775

Table 4. Results of time (t) effects and their interactions with land use (L) for flooded versus non-flooded soils (I), examined by means of GLM5 analysis. F-ratios and their levels of significance are given (n=4). Bold values indicate significant differences.

Sediment pore water	Time (t)	$t \times$ L	$t \times$ I	$t \times$ L $\times$ I	L	I	L $\times$ I
pH	**21.50*** **	1.96^{NS}	**12.13*** **	1.25^{NS}	**4.61***	**44.19*** **	2.344^{NS}
Alkalinity	**89.55*** **	1.87^{NS}	**32.56*** **	0.53^{NS}	1.75^{NS}	**54.72*** **	0.49^{NS}
TIC	**101.47*** **	2.84^{NS}	**33.57*** **	2.24^{NS}	**12.13** **	**82.01*** **	0.02^{NS}
NO_3^-	**29.85*** **	2.67^{NS}	**9.96*** **	1.90^{NS}	2.99^{NS}	**28.92*** **	1.37^{NS}
NH_4^+	2.17^{NS}	0.10^{NS}	2.56^{NS}	0.55^{NS}	0.00^{NS}	1.30^{NS}	0.41^{NS}
SRP	**4.70** **	0.30^{NS}	**4.21** **	0.23^{NS}	2.20^{NS}	**27.71*** **	2.59^{NS}
Fe	**21.26*** **	0.65^{NS}	**5.32** **	0.17^{NS}	1.152^{NS}	**42.61*** **	0.98^{NS}
K	**5.31** **	**3.12***	**3.26***	0.55^{NS}	**8.67** **	**43.41*** **	0.14^{NS}
Ca	**3.31***	0.87^{NS}	0.34^{NS}	1.40^{NS}	3.513^{NS}	**42.54*** **	1.67^{NS}
SO_4^{2-}	**7.93*** **	1.90^{NS}	**7.51*** **	0.81^{NS}	0.54^{NS}	**4.25***	0.41^{NS}
H_2S	**5.65** **	0.38^{NS}	0.78^{NS}	0.72^{NS}	0.15^{NS}	**17.97*** **	0.00^{NS}

*** p<0.001, ** p<0.01, * p<0.05, NS not significant

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Table 5. Results of time (t) effects and their interactions with land use (L), and water quality (W) for flooded soils, examined by means GLM5 analysis. F-ratios and their levels of significance are given (n=4). Bold values indicate significant differences.

	Time (t)	$t \times$ L	$t \times$ W	$t \times$ L $\times$ W	L	W	L $\times$ W
Sediment pore water							
pH	**96.33*** **	**3.08***	1.44^{NS}	0.97^{NS}	0.29^{NS}	**5.93****	**3.35***
Alkalinity	**220.40*****	**2.85***	1.93^{NS}	0.58^{NS}	**10.21****	**3.85***	1.20^{NS}
TIC	**225.04*****	**9.80*****	0.72^{NS}	1.08^{NS}	**9.68****	0.12^{NS}	1.09^{NS}
NO_3^-	**144.81*****	**3.08***	0.88^{NS}	0.62^{NS}	0.13^{NS}	**7.73****	1.21^{NS}
NH_4^+	**13.03*****	**3.19****	1.12^{NS}	0.91^{NS}	4.08^{NS}	2.35^{NS}	1.63^{NS}
SRP	**34.26*****	**1.74****	**2.16***	1.13^{NS}	**32.53*****	**12.74*****	1.93^{NS}
Fe	**209.46*****	**2.49***	1.17^{NS}	1.45^{NS}	0.16^{NS}	1.01^{NS}	0.31^{NS}
K	**5.84****	**7.11****	0.72^{NS}	0.49^{NS}	**12.96****	1.63^{NS}	0.59^{NS}
Ca	**5.64****	**2.53***	1.61^{NS}	0.80^{NS}	**15.22****	**4.04***	0.79^{NS}
SO_4^{2-}	**30.10*****	**2.94****	**4.46*****	0.79^{NS}	0.02^{NS}	**62.77*****	0.21^{NS}
S^{2-}	**13.42*****	2.22^{NS}	1.29^{NS}	1.10^{NS}	0.68^{NS}	**30.44*****	1.48^{NS}
Surface water							
pH	**56.28*****	**2.39***	**2.84****	0.34^{NS}	3.23^{NS}	**19.87*****	0.48^{NS}
alkalinity	**148.81*****	1.99^{NS}	**7.58*****	0.57^{NS}	**8.59****	**27.09*****	0.23^{NS}
turbidity	**13.62*****	0.63^{NS}	1.37^{NS}	0.84^{NS}	**23.59*****	0.41^{NS}	2.14^{NS}
NO_3^-	**190.71*****	1.11^{NS}	1.72^{NS}	1.22^{NS}	0.07^{NS}	**14.38*****	**5.74****
NH_4^+	**2.42***	**3.20****	**1.99***	1.18^{NS}	**19.85*****	**23.53*****	**4.54***
SRP	**10.02*****	1.87^{NS}	**2.56***	0.95^{NS}	**14.37****	**9.41*****	1.31^{NS}
Fe	**93.12*****	**4.68****	**2.94****	0.88^{NS}	**8.92****	2.13^{NS}	1.22^{NS}
K	**11.82*****	**11.85*****	1.47^{NS}	0.40^{NS}	**12.18****	0.78^{NS}	0.49^{NS}
Ca	**68.73*****	**3.96****	**3.98*****	1.05^{NS}	2.86^{NS}	**5.87****	0.07^{NS}
SO_4^{2-}	**60.98*****	**2.82***	**4.18*****	1.35^{NS}	**5.01***	**194.76*****	2.49^{NS}

*** $p<0.001$, ** $p<0.01$, * $p<0.05$, NS not significant

Table 6. Statistical results for time (*t*) effects and their interactions with land use (L) for **(a)** flooded versus non-flooded soils (I), and **(b)** interactions time with land use (L), water quality (W) and group of vegetation (G) for flooded soils only examined by means of GLM5. See caption of Table 4a for more details.

(a)	Vegetation cover	Biodiversity	(b)	Vegetation cover	Biodiversity	Algae cover
t	**10.70*** **	**38.88*** **	*t*	**24.22*** **	**29.93*** **	**26.67*** **
t×G	–	**15.28*** **	*t*×G	–	**14.30*** **	–
t×L	0.72^{NS}	0.70^{NS}	*t*×L	2.74^{NS}	1.83^{NS}	0.776^{NS}
t×I	**6.49** **	**41.89*** **	T×W	1.13^{NS}	0.95^{NS}	**2.51** **
t×G×L	–	1.27^{NS}	T×G×L	–	0.67^{NS}	–
t×G×I	–	**16.37*** **	*t*×G×W	–	0.84^{NS}	–
T×L×I	0.83^{NS}	0.39^{NS}	*t*×L×W	0.44^{NS}	1.00^{NS}	1.26^{NS}
t×G×L×I	–	0.90^{NS}	*t*×G×L×W	–	**1.58***	–
G	–	**68.71*** **	G	–	**5.14** **	–
L	1.04^{NS}	**11.15** **	L	**5.60***	**6.52***	**17.45*** **
I	**53.50*** **	**72.75*** **	W	1.77^{NS}	1.56^{NS}	0.25^{NS}
G×L	–	**6.58** **	G×L	–	**7.92** **	–
G×I	–	**45.72*** **	G×W	–	1.50^{NS}	–
L×I	1.55^{NS}	1.84^{NS}	L×W	0.61^{NS}	1.23^{NS}	1.24^{NS}
G×L×I	–	2.39^{NS}	G×L×W	– 1.00^{NS} -		

– Not included in the model, *** $p<0.001$, ** $p<0.01$, * $p<0.05$, NS not significant

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Table 7. Effects of time (t), land use (L) on biomass of each group of plants caused by (a) inundation (I) and (b) water quality (W) by means of univariate ANOVA. See caption of Table 4a for more details. In addition differences between flooded treatments are presented for each group – treatments with the same letter are not significantly different.

(a)	Grass	*Carex*	Herbs	(b)	Grass	*Carex*	Herbs
t	**12.62****	0.04^{NS}	2.74^{NS}	t	**14.76*****	0.47^{NS}	**4.96***
L	**7.05***	**19.88*****	**5.29***	L	0.07^{NS}	**38.91*****	1.15^{NS}
I	**18.83*****	**5.31***	**35.22*****	W	**3.68***	1.42^{NS}	0.08^{NS}
T×L	0.74^{NS}	0.01^{NS}	0.68^{NS}	t×L	0.83^{NS}	0.40^{NS}	0.56^{NS}
t×I	0.01^{NS}	0.25^{NS}	0.18^{NS}	T×W	0.57^{NS}	0.02^{NS}	0.10^{NS}
L×I	**5.85***	3.61^{NS}	1.70^{NS}	L×W	0.64^{NS}	0.60^{NS}	**2.95***
t×L×I	2.84^{NS}	0.58^{NS}	0.02^{NS}	T×L×W	1.01^{NS}	0.03^{NS}	0.11^{NS}
				Cfl	A	A	A
				N	AB	A	A
				S	B	A	A
				SN	AB	A	A

*** $p<0.001$, ** $p<0.01$, * $p<0.05$, NS not significant

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Fig. 1. Location of Kosiorów, the sampling area (diamond) in Poland.

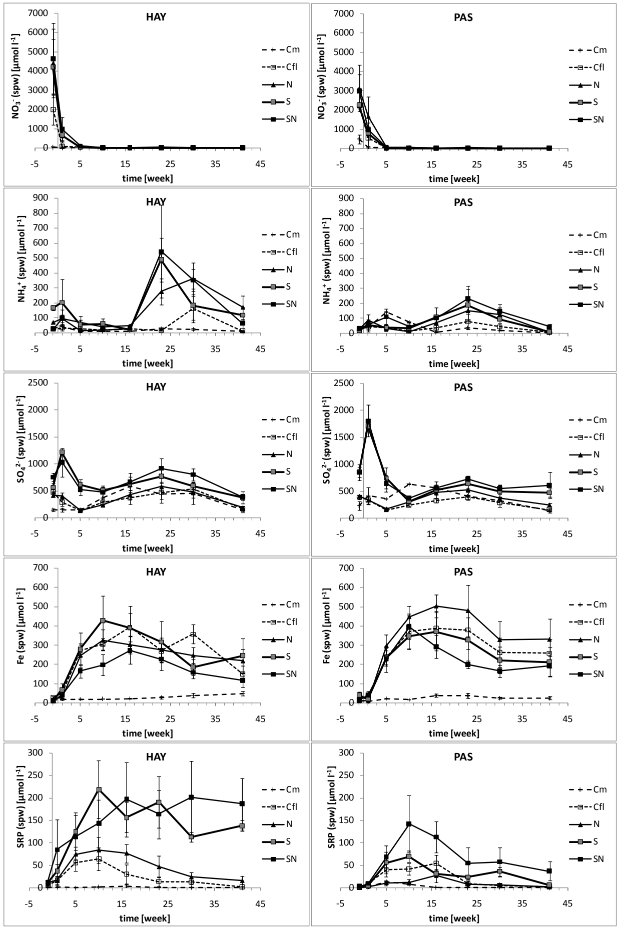

Fig. 2. Changes in concentrations of selected elements in sediment pore water (spw) – means±SEM (n=4). The week number refers to the time after the onset of flooding.

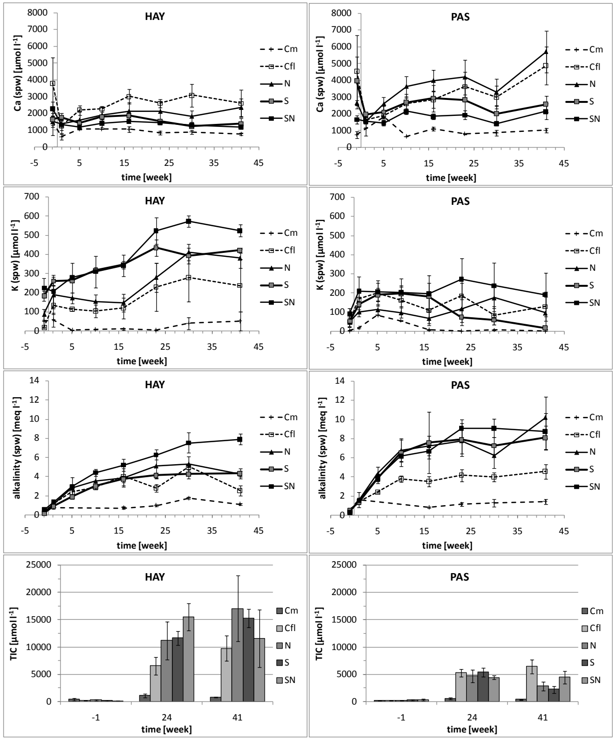

Fig. 3. Changes in concentrations of selected elements in sediment pore water (spw) – means±SEM (n=4) – continuation.

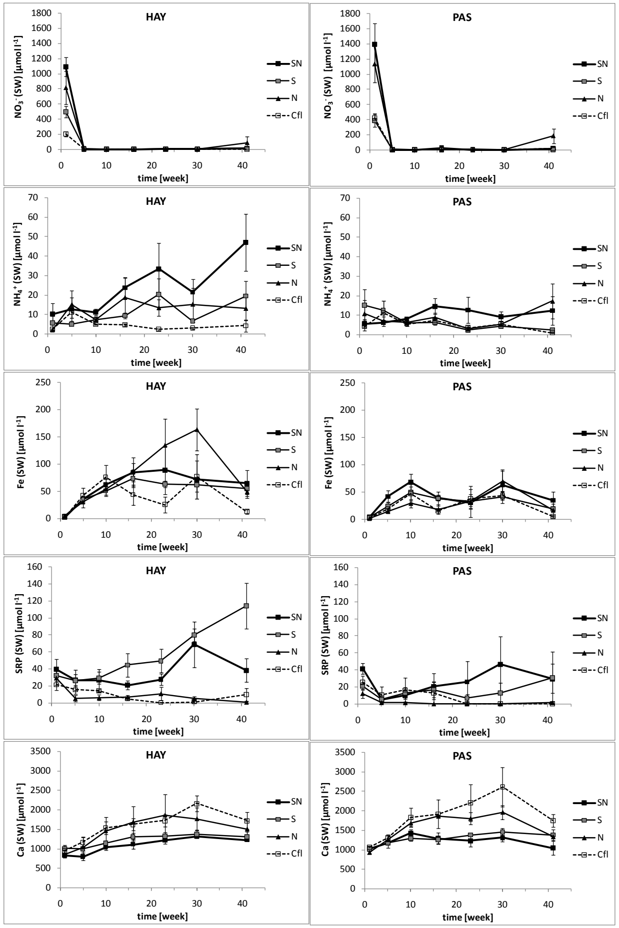

Fig. 4. Changes in concentrations of selected elements in surface water (SW) – means±SEM (n=4).

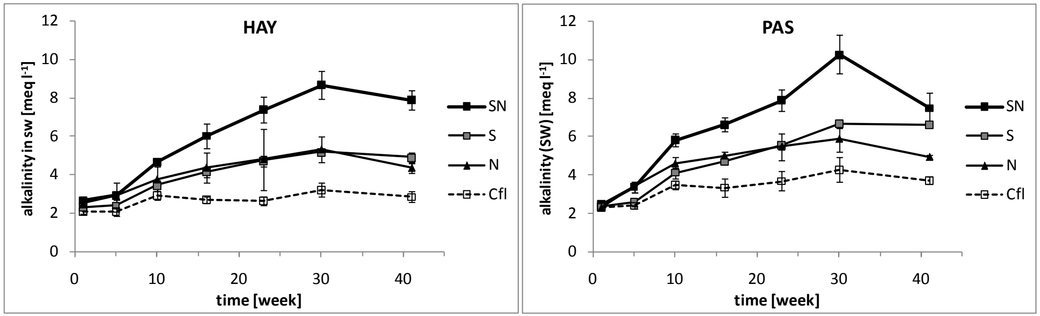

Fig. 5. Changes in concentrations of selected elements in surface water (SW) – means±SEM (n=4) – continuation.

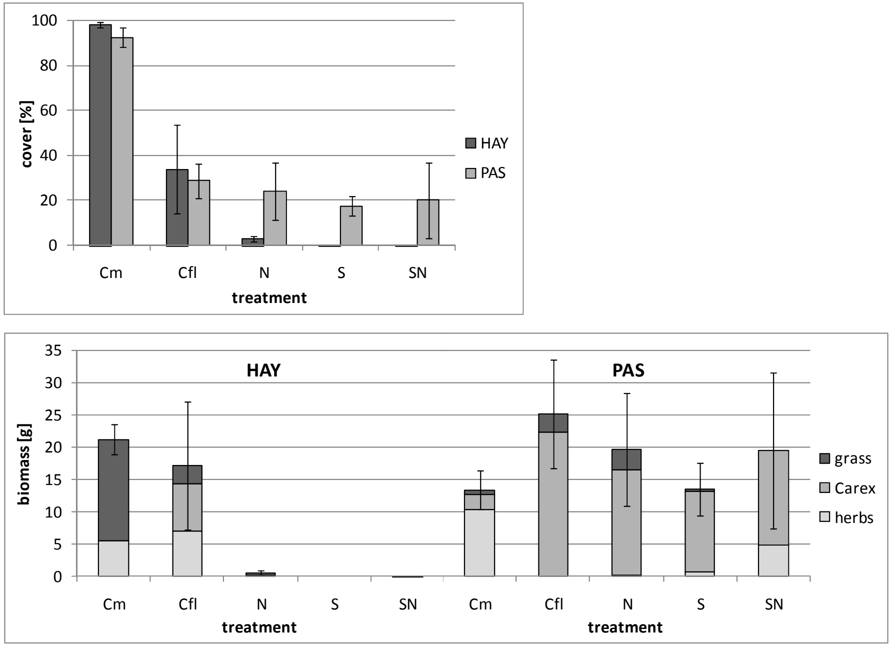

Fig. 6. Vegetation data (cover above and biomass of each group below) at the end of the inundation period (week 41) – means±SEM (n=4). Error bars of biomass represent SEM for the total biomass.