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Impact of long-term drainage on summer groundwater flow patterns in the Mer Bleue peatland, Ontario, Canada

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Abstract

Long-term impacts of a drier climate on coupled hydrology and carbon cycling in northern peatlands are poorly understood. We used a historic drainage ditch, separating an area from the main peatland, as an analogue for long-term drying in a northern temperate bog. The objective was to identify the impact of drier conditions on eco-hydrological processes and groundwater flow patterns in an area now wooded and an area that maintained bog character. Groundwater flow patterns alternated between downward flow and upward flow in the bog area and were mostly upward orientated in the wooded area. Flow patterns were in agreement with changes in post-drainage hydraulic conductivities, storage capacity of the peat and hydraulic gradients. Compared to the bog, hydraulic conductivities in the wooded area were one to three orders of magnitude lower in the uppermost 0.75 m (paired t-test, $p < 0.05$) of peat but partly higher below. Bulk density had increased and the water table level was lower and more strongly fluctuating. Our findings suggest hydraulic gradients and flow patterns have changed due to increased evapotranspiration and interception with the emergence of a tree cover. The smaller size of the now-forested area relative to the remaining bog area appeared to be important for the hydrological change. When water supply from undisturbed areas was large, enhanced runoff to the drainage channel had little effect on hydrologic patterns and vegetation pattern, whereas in the smaller, now forested area pervasive changes in vegetation and hydrologic processes occurred. This finding raises questions about tipping points with respect to the impact of drying on bog ecosystems that need to be addressed in future research.

1 Introduction

Northern peatlands cover about three percent (400 million hectare) of the world's surface (Gorham, 1991), store between 270 to 370 Pg of carbon (C), and thus represent important global C stores (Turunen et al., 2002). The response of peatlands to changes

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in the water budget is important for predicting potential feedbacks on the global C cycle (Belyea and Malmer, 2004). Hydrological fluxes and storage terms to be considered include precipitation, evapotranspiration, and groundwater seepage. All of these affect soil moisture content, water table position and hence ecological processes, such as plant succession, plant productivity, plant litter fall and peat decomposition rates (Baird et al., 2008). The interplay between these processes is an important control on the establishment of plant communities and the evolution of peatland systems (Drexler et al., 1999; Baird et al., 2008) and affects the C balance of the system (Laiho, 2006).

Groundwater flow patterns have been studied in different types of peatlands (e.g. Siegel, 1992; Waddington and Roulet, 1997; Drexler et al., 1999; Fraser et al., 2001; Baird et al., 2008). Controls on the direction and magnitude of flow have been particularly emphasized (Reeve et al., 2006). The occurrence of seasonal flow reversals has been shown to influence geochemical profiles (Devito et al., 1997; Fraser et al., 2001). Other studies focussing on local flow patterns have suggested that the plant community composition of peatland systems is affected as well (Drexler et al., 1999). Strong effects of long-term flow reversal on vegetation patterns have, for example, been identified in a bog-fen complex of the Lost River peatland, Northern Minnesota (Siegel and Glaser, 1987; Glaser et al., 1990). In this area local islands of predominantly groundwater recharge and upwelling of nutrient and alkalinity rich waters have been identified (Siegel and Glaser, 1987). These islands sustained a rich fen and Picea and Larix rich swamp vegetation within a larger Sphagnum dominated bog complex. The creation of such an area of distinctly different vegetation was identified by a sharp transition from peat moss to sedge peat at about 1160 before present (Glaser et al., 1990). The authors suggested that long-term groundwater flow reversal occurred at this time and was related to peatland growth. Given the impact of vegetation change on carbon cycling it is important to identify and characterize similar effects of groundwater flow patterns that are related to changes in the water budget of peatlands, for example by drainage and climate change.

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Long-term drainage represents to some extent an analogue for a warmer climate characterized by lower rates of groundwater renewal. Related changes in hydrologic properties and processes, such as in hydraulic conductivities and groundwater flow patterns, may also provide some insight into potential effects of climate change. Over periods of a few years, drainage resulted in a lowered water table, loss of macropores, increased bulk density, decreased hydraulic conductivities (K_h) (Silins and Rothwell, 1998; Price et al., 2003), and increased rates of organic matter decomposition (Laiho, 2006). The resulting change in peat physical properties affects the hydrological functioning of these ecosystems (Silins and Rothwell, 1998; Price et al., 2003). On the decadal time scale vegetation patterns also change in response to drainage (Talbot et al., 2010). In peatlands with wet microenvironments such as the Mer Bleue bog, Ontario, peat mosses and sedges typically dominate, whereas shrubs are more abundant in drier areas (Moore et al., 2002). Subsequent to drainage trees can become prominent, depending on the nutrient regime (Laiho, 2006). These changes in plant community affect the rate and chemical composition of litterfall and thus rates of soil organic matter decomposition (Laiho, 2006; Bubier et al., 2007). The denser canopy cover of trees and shrubs has the potential to increase evapotranspiration (ET) (Price et al., 2003; Lafleur et al., 2005; Beheim, 2006; Talbot et al., 2010), increase interception (I) (Price et al., 2003; Emili and Price, 2006) and lower groundwater recharge. In peatlands affected by persistent drainage, discharging groundwater flow may thus be more frequent and prominent and runoff may be lower during the summer season.

To improve our insight in the interactions between dryness by increased runoff, groundwater flow patterns, and vegetation change we hydrologically characterized a drained area in the Mer Bleue peatland, Ontario. Drainage by a channel of 4 km length and about 10 m width has been active for about 90 yr and created two areas of strikingly different vegetation, with an open-bog turned forest present only on one side (Talbot et al., 2010). Our objective was to identify the impact of drainage on hydraulic conductivity and groundwater flow patterns on both sides and to gain some insight on their potential importance for post-drainage vegetation development. We hypothesized that recharge

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in the smaller and now wooded part of the peatland separated by the channel would be reduced due to higher ET and *I*, and poorly permeable, more decomposed peat. The empirical work was supported by a MODFLOW groundwater model to simulate and quantitatively analyze groundwater flow.

2 Study sites, materials and methods

2.1 Study site

Field work was carried out between 6 August 2008 and 4 October 2008 at the eastern end of the Mer Bleue (MB) peatland: an open, slightly domed, acidic and ombrotrophic 28 km^2 bog, 15 km east of Ottawa, in eastern Ontario, Canada (45°30′ N latitude and 75°25′ W longitude). Mean annual temperature, precipitation and growing season length have been reported as 5.8 °C, 944 mm, and 193 days, respectively (Environment Canada, 2004). A description of the biogeochemistry and hydrology of the bog has been given by Blodau et al. (2002) and Fraser et al. (2001).

A transect covering the open bog in the west ("Bog"), drainage ditch, installed in 1922 (Talbot et al., 2010), and the now wooded bog in the eastern end ("Forest") was examined (Fig. 1). Vegetation in the "Bog" is characterized by a Sphagnum moss ground cover and an overstory dominated by evergreen and deciduous ericaceous shrubs, and a sparse cover of trees – black spruce (Picea mariana), tamarack (Larix laricina) and white paper birch (Betula populifolia) (Talbot et al., 2010). Close to the ditch, the abundance of shrubs and trees is higher, and trees and shrubs dominate in the "Forest" (Fig. 1). Aboveground biomass was between 688 ± 234 and 699 ± 234 g m^{-2} at the "Bog" sites and 3170 ± 450 and 5720 ± 1620 g m^{-2} at the "Forest" sites. Total leave area index was 1.25 to 1.58 in the bog and 5.3 to 7.25 under forest (Talbot, 2010). A detailed description of the plant distribution across the transect has been given by Talbot et al. (2010). Tree growth started after lowering of the ground water table together with increased organic matter mineralization due to aeration of former saturated peat

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(Talbot et al., 2010; Blodau and Siems, 2010). The lowered land surface has been hydrologically isolated from the open bog in the west (Fig. 1). Peat depths range from 1.85 m near the ditch to 2.5 m on the east side and > 4 m on the west side. The peat is underlain by a marine clay layer of 12–45 m thickness (Fraser et al., 2001) providing chloride that has been used as natural conservative tracer to investigate vertical transport processes in this peatland (Beer et al., 2008).

2.2 Precipitation, evapotranspiration, and hydrogeological conditions

Surrounding uplands are well drained by a network of creeks and ditches at the margins of the peatland. Thus the contributing area for groundwater recharge was considered to be solely the peatland surface (Fraser et al., 2001). Half hourly measures of precipitation (P) and latent heat flux (Q_E) were measured at a micrometeorological tower situated 1.5 km to the east in an open portion of the bog. Latent heat fluxes were obtained using the eddy covariance technique with the instrumentation and data handling procedures described in detail by Roulet et al. (2007). Briefly, 20 Hz signals of vertical wind velocity (measured with a sonic anemometer, model R3-50, Gill Instruments Ltd., England) and water vapour mixing ratio (measured with a closed-path infra red gas analyzer, model LI7000, LI-COR Inc., Lincoln, NB, USA) was used to compute Q_E. Quality control procedures described by Lafleur et al. (2005) were applied to the Q_E data. Gaps were then filled using an empirical relationship with available energy (net radiation minus the soil heat flux and changes in energy storage) parameterized using a moving window method. Evapotranspiration (ET) was calculated from Q_E, by dividing Q_E by the latent heat of vaporization.

Piezometers were constructed from 2.6 cm ID PVC pipe screened over a 10 cm interval and covered with a nylon mosquito mesh. A total of eleven piezometer nests (P1-P11, Fig. 1) were installed across the transect in hollows 0, 3, 15, 30, 60 and 200 m distance from the ditch at depths of 0.25, 0.50, 0.75, 1.0, 2.0 and 3.0 m, where peat depth permitted. Two further piezometers were installed in the ditch and at the 3 m "Forest" site at 1.75 and 2.25 m depth. Additional piezometer nests were installed; one

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at 600 m distance west of the ditch in the "Bog"; and two piezometer nests, serving as reference sites for hydraulic head measurements, at both sides at 45 m and 30 m distance perpendicular from the ditch and the transect, respectively. Hydraulic head was monitored with an electric dip stick every two to nine days. The most shallow yet water-filled piezometer was used to approximate water table position, as referred to hereafter, which was always within 15 cm of a slotted portion of a shallow piezometer. Two potentiometric water level loggers, reading every half-hour, were installed in piezometers with slotted heads at 0.75 and 2.0 m depth at the 200 m sites of the transect. An atmospheric pressure sensor allowed for barometric compensation of the water level loggers. The dominant groundwater flow direction in this study was inferred from the head differentials in the piezometer nests according to the definitions in Drexler et al. (1999). To minimize the chance of artifacts in hydraulic head due to slow equilibration, only hydraulic head readings at local temporal maxima or minima were considered for assessing vertical flow. Locations where hydraulic head decreases with depth in the peat column define areas of recharge or downward flow and locations where hydraulic head increases with depth define areas of discharge or upward flow. Locations with negligible hydraulic head changes with depth and horizontal head gradients suggest areas of lateral or horizontal flow.

Horizontal hydraulic conductivity was estimated from slug insertion after Hvorslev (1951). As the method can only be applied in the saturated zone some piezometers did not allow for a determination of K_h due to the low water table because they never contained water or fell dry at some point. Vertical hydraulic conductivity was calculated using an average anisotropy of 450, as proposed by Fraser et al. (2001) for a pristine section of the Mer Bleue bog. Porosity was determined in peat samples collected with a Finnish box corer in 10 cm intervals down to 0.7 m depth at the 60 m sites and a Russian corer at 200, 60, 30 and 3 m sites at 0.75, 1.0, 2.0 and 3.0 m depth by drying at 70 °C to constant mass. Porosity was calculated from bulk density and a specific density of the peat solids of 1.5 g cm^3 (Weiss et al., 1998).

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2.3 Sampling and analyses

The piezometers were sampled on 13 August, 5 September and 4 October, after emptying them the day before and inserting PE float levers to prevent air circulation (Heitmann et al., 2007), using a tube and 50 mL luer lock syringe, filtered (0.45 µm, nylon), and frozen for further analysis. Following further filtration (0.2 µm, nylon) pore-water concentrations of chloride (Cl^-) were analysed using an ion chromatograph (Metrohm IC-System, Metrosep Anion Dual 3 column at 0.8 mL min^{-1} flow rate, conductivity detection after chemical suppression). Detection limit was 5.6 µmol L^{-1}.

2.4 Groundwater modelling

Groundwater modelling was performed using the MODFLOW pre- and post-processor Groundwater Vistas 5. The aim of the modelling work was to simulate flow paths and to identify areas characterized by upward flow, downward flow, or alternating conditions. A steady state model to initialize the hydraulic head at the beginning of the observation period was developed first and a transient model covering the observation period from 10 August to 3 October was subsequently created. The simulations used a total of 56 stress periods (SP), with each SP representing one day. A SP is defined as an increment of time in a transient simulation during which aquifer recharge and discharge are held constant, A cell size of 3×3 m was chosen and each layer consisted of 33 rows (y-direction) in width, 133 columns in length (x-direction) and a total of 8 layers (z-direction). The layers consisted of seven peat layers and one marine clay layer (aquitard). For the uppermost peat layer a thickness of ~ 0.7 m was defined to avoid wetting/drying problems in MODFLOW in the uppermost layer due to large water table fluctuations, especially in the "Forest". The thickness of the lowest peat layer was chosen to coincide with piezometer depth, therefore ranging from ~ 0.25 to ~ 0.75 m. In general, layer thickness decreased in the vicinity of the ditch. The following boundary conditions (BC) were assigned: constant head BCs for the east and west margins (first and last column) of each layer, deduced from the potentiometric water level logger

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data, and adjusted for each SP; constant head drain BCs for the ditch in the uppermost layer deduced and adjusted for each SP from the manual measured water level; a no-flow BC for the underlying marine clay layer. Model domain thickness was set to 4 m. Active cells in the model domain were defined by the surface elevation and the elevation of the marine clay layer. K_h values (Table 1) were determined based on slug tests in the piezometers and porosity estimates were obtained from laboratory analysis of soil samples taken at the piezometer locations. Both parameters were interpolated (Kriging) for areas in between the piezometers. Due to interpolation errors, parameters for some sites were set manually. Layers were assumed to be homogenous in the y-direction. Furthermore, calculated difference between rainfall P and I and ET (P ET - I) was applied to the top layer of the transient model at the "Bog" and was assumed to be lower in the "Forest" due to greater ET and I. The changes in ET (ET "Bog" + 2 mm) and I (interception "Bog" + 0.5 mm) for the 'Forest' were based on model calibration where simulated hydraulic heads were compared to hydraulic heads measured in the field by setting calibration targets for each piezometer. Differences in a similar range in ET between mosses and trees in the growing season were found in other studies (Heijmanns et al., 2004; Bond-Lamberty et al., 2009; although the additional 2 mm in our study for a forest stand is in the upper range in the literature.

2.5 Statistics and sensitivity analyses

To examine the statistical significance of differences in the horizontal hydraulic conductivities between the "Bog" and "Forest" side for each depth (Table 3), a paired t-test was applied with a 95 % confidence interval using the statistical software *R*.

To assess the variability and uncertainty of K_h and anisotropy on model performance a sensitivity analysis was carried out (Auto Sensitivity Analysis, Groundwater Vistas 5). For each cell in the model from the channel to a distance of 60 m the K_h values used were changed using a multiplier and the sum of squares between measured and modelled hydraulic heads with variation of K_h was calculated. The sum of squares here is an indicator for the impact of variation of a parameter in a single cell on the quality of the overall model. Increasing sum of squares indicate a high sensitivity as well as deterioration of the goodness of fit. To reduce computing time, cells in distances more than 60 m from the channel were not considered. Furthermore, these cells are more influenced by the constant head model boundaries. K_h multipliers of 0.001, 0.01, 0.1, 0.5, 1, 2, 10, 100 and 1000 were used and anisotropy multipliers of 0.01, 0.1, 0.4, 0.7, 1, 1.3, 1.6, 10 and 100 were applied.

A sensitivity analysis of the impact of water input (P- ET- I) was carried out for each zone ("Bog", "Forest" and "Ditch") in the uppermost layer using multipliers of 0.01, 0.1, 0.5, 1, 2, 10 and 100 in the same way as described above.

3 Results

3.1 Porosity and bulk density

Porosity at the 60 m sites was slightly higher in the uppermost 0.5 m of peat in the "Bog" compared to the "Forest". At depths of 0.6 and 0.7 m the values were approximately equal. Mean porosity for the upper 0.7 m layer was 0.96 in the "Bog" and 0.94 in the "Forest". Porosity deeper into the peat ranged between 0.94 and 0.92 in the "Bog", and 0.95 and 0.91 in the "Forest", slightly decreasing with depth in both areas. Mean porosity in marine clay was around 0.34 ($n = 2$). Peat bulk density in the uppermost 0.5 m, was higher in the "Forest" peat than in the "Bog" peat. "Forest" bulk density peaked at 0.1 g cm^{-3} in 0.2 m depth at 2.4 times the "Bog" bulk density of 0.04 g cm^{-3} and declined to 0.09 g cm^{-3} which is 1.3 times the "Bog" bulk density of 0.07 g cm^{-3} in 0.5 m depth, until they equalled in 0.6 m depth at 0.09 g cm^{-3}.

3.2 Hydraulic conductivities

Results of the hydraulic conductivity (K_h) measurements are shown in Table 2. K_h estimates in acrotelm peat (~ 0–0.55 m) in the "Bog" ranged from 10^{-8} to 10^{-4} m s^{-1}

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($\pm 8.3 \times 10^{-5}$, standard deviation) ($n = 5$) and 10^{-8} to 10^{-6} m s^{-1} ($\pm 3.4 \times 10^{-6}$) ($n = 4$) in the "Forest". In deeper peat (> 0.55 m depth) K_h estimates ranged from 10^{-9} to 10^{-6} m s^{-1} ($\pm 1.6 \times 10^{-6}$) ($n = 14$) in the "Bog", and from 10^{-8} to 10^{-5} m s^{-1} ($\pm 1.1 \times 10^{-5}$) ($n = 12$) in the "Forest". With $K_h \ll 10^{-9}$ m s^{-1} ($\pm 4.7 \times 10^{-10}$) ($n = 7$) the underlying marine clay layer was least permeable. Under and near the ditch a clay lens seemed to be present at 2 m depth ($K_h \sim 10^{-10}$ m s^{-1} ($\pm 3.8 \times 10^{-11}$), $n = 3$), followed by amorphous clay material with higher K_h values at 2.25 and 2.5 m depths ($\sim 10^{-6}$ to 10^{-7} m s^{-1} ($\pm 3.3 \times 10^{-6}$), $n = 4$). In general, both 15 m sites were characterized by low K_h estimates (10^{-7} to 10^{-9} m s^{-1}). Compared to the "Bog" K_h values in the "Forest" were significantly lower at a depth of 0.75 m and significantly higher at a depth of 1.0 m (Table 3). No significant difference could be identified for the other depths.

3.3 Precipitation, evapotranspiration, and water table

Total precipitation (P) during the study period was 108.8 mm. P was not evenly distributed during the period with heavy rainfall occurring on 7 August (11.2 mm) and 18 August (19.0 mm), and a period with higher P from 6 September to 15 September, totaling 46.6 mm (Fig. 2). Total ET of the "Bog" was 106.8 mm, yielding a net-rainfall of only 2 mm during the observation period, which can be considered zero given the uncertainty in the quantification of P and ET. In general, water tables as approximated from the most shallow piezometer were between 7 and 15 cm below the soil surface at the "Bog" 200 m site and between 24 and 43 cm below the soil surface at the "Forest" 200 m site. Therefore, the water table fluctuated up to 8 cm in the "Bog" and 19 cm in the "Forest". Observed time lags between hydraulic responses at shallow depth (0.75 m) and deep peat (2.0 m) were short (Fig. 2).

3.4 Hydraulic heads

Continuously monitored hydraulic heads at the 200 m bog site indicated generally recharging conditions, i.e. downward directed flow, with the exception of a dry period at

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the end of September (Fig. 2). Recharge was suggested by a higher hydraulic potential at 0.75 m than at 2.0 m depth. The hydraulic dynamics differed in the “Forest”, where a discharging component, i.e. upward directed flow, occurred in periods with water-table drawdown. This was indicated by a higher hydraulic potential at a depth of 2.0 m than at 0.75 m (Fig. 2). In the “Forest” recharge only occurred during and shortly after periods with larger P when the water-table rose. The decline in hydraulic heads during late summer conditions was also much smaller in the “Bog” than in the “Forest”, especially in August when ET, measured nearby in the Bog, was still high at about 3 mm d^{-1}.

Manual monitored hydraulic heads along the transect are illustrated in Fig. 3 and depict the spatial pattern described above. In the “Bog”, differences in hydraulic head with depth were small at the 60 and 200 m sites, suggesting predominantly lateral flow in this area. Near the channel this pattern changed. The vertical hydraulic heads 15 m from the channel indicated continuous recharge with differences in hydraulic head between 0.5 and 2.0 m depth ranging from −4.6 cm to −10.3 cm. Recharge also occurred 30 m from the channel, except for a brief period at the end of August. In the ditch, hydraulic heads measured at depths of 0.5 to 1.0 m indicated discharge, except for the dry period in late August and early September, when conditions changed to recharge. Differences in hydraulic head between the open water and the hydraulic potentials at 2.25 and 2.5 m depth were around 5.0 cm and thus indicated confined conditions underneath the clay lens at around 2.0 m depth.

Vertical patterns of hydraulic heads pattern in the “Forest” were quite different. Hydraulic heads at the 15 m site indicated discharge during the observation period. At the 30 and 60 m sites discharge dominated as well until the beginning of September; recharge was observed in the beginning of October. Hydraulic heads at the 200 m site indicated discharge as well, which changed to recharge more frequently, on 20 August, 15 September, 29 September and 4 October. Hydraulic heads at the 30 m site in the “Forest” at 0.5 and 2.0 m depth indicated a strong vertical discharge reaching a maximum difference of 9.7 cm on 2 September. Hydraulic heads thus primarily indicated discharging conditions, rather than recharging as recorded in the “Bog”.

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3.5 Groundwater modelling

The MODFLOW (GW-Vistas 5) model adequately represented hydraulic heads in the flow field. The Root Mean Square Error (RMSE) between measured and observed hydraulic potentials of 0.025 m for the steady-state simulation and of 0.056 m for the transient simulation, and Nash-Sutcliffe efficiencies (E^2) of 0.994 and 0.972, respectively, confirm the realistic representations. The groundwater levels of the steady-state and the transient simulation were similar, with lower water table positions in the "Forest". As net-rainfall was not included in the steady-state simulation, hydraulic potentials differed between the two types of simulation. The steady state simulation (Fig. 5a) suggested slightly discharging hydraulic potentials in the "Forest", however, streamlines indicated that the groundwater predominantly moved laterally across the transect in the upper peat layers, whereas in the deeper peat layers movement was slightly downward. At the ditch groundwater movement was dominated by up-welling.

In the transient simulation (Fig. 5b) the water balance between P, ET, and I was included, and therefore vertical hydraulic potentials alternated between recharge and discharge in the "Bog", depending on whether P or ET was the dominant parameter for each SP. At the "Forest" discharging hydraulic potentials only changed in magnitude. Dry periods, e.g. SP 25 (4 September), resulted in flow reversals with upward orientated movement in both the "Bog and the Forest" sites within the upper peat layer. In contrast, in the deeper peat water moved downward and more slowly, with longest residence times in the "Bog" and the deepest layers. In agreement with the hydraulic head measurements the model indicated upwelling groundwater under the ditch. The horizontal runoff into the channel, assumed to be four cells (1.08 m) deep, was 4.0 L m^{-2} d^{-1} from the "Bog" and 2.2 L m^{-2} d^{-1} from the "Forest" in the steady state simulation, in line with the different ET and I.

3.6 Validation of hydraulic and modelling results

The occurrence of gradient changes with rainfall and during dry conditions and the similar temporal response of shallow and deep piezometers (Fig. 2), as well as a reasonable match between temporal dynamics of measured and modeled hydraulic heads suggested that the response time of in the piezometers was adequate to identify hydraulic gradients.

The sensitivity analysis of hydraulic conductivity revealed a low sum of square values for most cells. This suggests that the modelling results were robust with respect to uncertainty in measured and interpolated K_h values. There were some exceptions to this finding near the drainage ditch. Higher sum of square values, i.e. higher sensitivity to uncertainty in K_h, occurred in the "Bog" for small multipliers and for big multipliers in the "Forest". In the uppermost layer in the "Bog", in distance from 6 to 24 m from the channel, a sum of square for multipliers 0.5 to 0.001 were elevated. They peaked at a distance of 12 m from the ditch with a value of 10.31. At the "Forest" side, cells in this layer and distance were also sensitive to variation. Here, the sum of square value for a multiplier of 0.001 reached a maximum of 2.48 at a distance of 15 m. Multiplication of anisotropy had little impact on model performance. Highest sum of square value was 1.12 for cells at the "Forest" side. The uncertainty in our choice of anisotropy, which was not constrained by actual measurements but based on literature values from a nearby site of the Mer Bleue bog (Fraser et al., 2001) , was thus likely of little consequence. Sum of square values for recharge increased with increasing multipliers, revealing a more sensitive behaviour for recharge of the "Forest" than of the "Bog" with values of 55.4 ("Forest") compared to 7.5 ("Bog") for a multiplier of 10, and 13507 ("Forest") compared to 496 ("Bog") for a multiplier of 100.

The observed Cl^- concentrations were broadly in agreement with the hydraulic head measurements and groundwater flow modelling results (Fig. 4). Cl^- concentration peaked in the marine clay layer at 5900 µmol L^{-1}. Mean Cl^- concentrations in the "Bog" were generally lower than in the "Forest" for a given depth. The local minima

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in Cl^- concentrations of 100 to 200 µmol L^{-1} near the channel in the "Bog" indicated predominantly recharge, which is in agreement with hydraulic heads over the analysed period (Fig. 3) and modelling results. In contrast, in the forest and under the channel discharging conditions were indicated by high concentrations of up to 1700 µmol L^{-1} at a depth of 0.75 to 1.0 m. The pattern of Cl^- concentrations changed little over time, suggesting that short-term dynamics in hydraulic gradients had limited influence on the movement of solutes.

4 Discussion

4.1 Hydraulic conductivities

Prior to drainage 90 yr ago vertical profiles of hydraulic conductivities in the study area were likely quite uniform, such as described by Fraser et al. (2001) for the north-western arm of the Mer Bleue bog. In their study, which covered a transect from a beaver pond to the central bog dome, values of K_h differed little horizontally and generally decreased from 10^{-3} to 10^{-7} m s^{-1} in the acrotelm to 10^{-6} to 10^{-8} m s^{-1} in the catotelm. In our study, the decrease in K_h with depth in the "Bog" was similar to measured K_h changes reported there and in other studies (Fraser et al., 2001; Beckwith et al., 2003; Baird et al., 2008; Quinton et al., 2008; Rosa and Larocque, 2008). The vertical distribution of the K_h estimates in the "Forest", however, strongly deviated from the reported pattern. Values of K_h were significantly lower ($p = 0.0064$) at depths of 0.75 m by 1 to 3 orders of magnitude in the "Forest" compared to the "Bog" peat (Table 2, Table 3). Hydraulic conductivity is controlled by pore hydraulic radius (Quinton et al., 2008) and has thus been linked to the decomposition of fibric litter into a macroscopically amorphous peat. Drainage accelerates this process; Silins and Rothwell (1998) found an increase in peat bulk density after drainage and subsidence associated with a collapse of macropores (> 600 µm), and an increase in micropores (3–30 µm). We thus attribute the lower hydraulic conductivity in shallow peat of the "Forest" to deeper water tables post-drainage, as observed by us (Fig. 2), and the subsequent consolidation, compression and decomposition of peat (Kennedy and Price, 2005). Our findings corroborate similar reports about saturated hydraulic conductivity differences between drained and undrained peat (Silins and Rothwell, 1998; Price et al., 2003; Kennedy and Price, 2004; Whittington and Price, 2006; Strack et al., 2008).

Saturated hydraulic conductivities are also in agreement with a previous study addressing peat decomposition and humification along the transect (Blodau and Siems, 2010). A strongly lowered decomposability and advanced humification, as indicated by accumulation of aromatic and carboxylic groups, was found down to about 1.5 m peat depth near the drainage ditch, and particularly at one location in the "Forest". Changes in plant litter quality with tree growth may also have influenced this pattern (Silins and Rothwell, 1998; Laiho, 2006).

Deeper into the peat, differences in hydraulic conductivity reversed and the explanation given above does not apply. At depths of 1 m (p = 0.0295) and 2 m (not significant), saturated hydraulic conductivity was higher under forest than at the corresponding bog sites (Table 2, 3). It is plausible, albeit speculative, that the phenomenon reflects a dual porosity established with tree root growth in strongly decomposed peats (Ours et al., 1997). The development of such a dual porosity may have occurred initially after drainage but before land surface subsidence and water table rebound relative to the subsiding peatland surface restricted the growth of birch, spruce, and larch roots deeper into the – then again water saturated- peat.

A few critical points with regard to hydraulic conductivities and hydraulic parameters in groundwater modelling should be mentioned. First, hydraulic conductivity in peatlands can also reflect gas-filled porosity and bubbles occluding peat pores (Fraser et al., 2001; Baird et al., 2008). To assess the importance of this gas-filled porosity along the transect is thus important. A gas phase forms when the total gas pressure exceeds the confining hydrostatic pressure initially at CH_4 partial pressures above ca. 0.2 atm, equivalent to ca. 390 µmol L^{-1} (at 8 °C). With continuous stripping of nitrogen by ebullition of CH_4, higher partial pressures of CH_4 are required for bubble formation and release (Fechner-Levy and Hemond, 1996). In a previous investigation we found CH_4 concentrations down to peat depths of 2 m to be lower than the above mentioned threshold values at 200 m, 60 and 30 m in the "Bog" and 60 m in the "Forest" (Blodau and Siems, 2010). This finding suggests a limited or lacking contribution of bubbles to hydraulic conductivity differences at our site.

Other than the effect of gas bubbles, K_h estimates obtained by piezometer slug tests can be compromised by peat compression during insertion of the piezometers (Surridge et al., 2005; Whittington and Price, 2006). In a similar study Rosa and Larocque (2008) found slug tests to be reproducible, suggesting that a differential alteration of hydraulic properties by this approach at different sites is not very likely. Because we were interested in finding relative differences in K_h between "Bog" and "Forest", we hence consider the single piezometer slug test method appropriate for our purpose. Additional uncertainty arises from our choice of anisotropy (K_h/K_v: 450) in the model. Anisotropy in peat has been reported to be heterogeneous (Beckwith et al., 2003; Surridge et al., 2005; Rosa and Larocque, 2008) and probably differed in peat at the "Bog" and "Forest" side, owing to the degree of decomposition and the nature of new plant material added. On the other hand, the sensitivity analysis of anisotropy did show little reaction of model behaviour to changes in individual cell anisotropy values, which increases confidence in the groundwater flow modeling results.

4.2 Groundwater flow patterns

Groundwater flow patterns are controlled by hydraulic conductivities, storage capacity of the peat (Fraser et al., 2001; Reeve et al., 2006) and hydraulic gradients driven by P, ET and I. Interception I accounted for 20 % and 32 % of P in studies of forested peatlands (Price et al., 2003; Emili and Price, 2006). Fraser et al. (2001) studied groundwater flow patterns within the north-western arm of the Mer Bleue bog and found recharging conditions except when ET exceeded P in summer. Waddington and Roulet (1997) suggested that such flow reversals are driven by internal mechanisms, especially ET and lower water tables, Reeve et al. (2006); Devito et al. (1997) and Fraser et al. (2001) found vertical hydraulic reversals to be driven by groundwater recharge and ET.

In our study the small vertical flow component was controlled by meteorological conditions, depending on whether dry (discharging vertical component) or wet periods (recharging vertical component) prevailed and where a site was located along the transect. Similar to the results of Fraser et al. (2001), the 200 and 60 m sites in the “Bog” were characterized by recharge in periods with P exceeding ET and vice versa (Fig. 3); functionally this area thus maintained a dynamic that is probably typical for the Mer Bleu bog and similar systems. Closer to the ditch recharge, i.e. downward directed flow, became prevalent at the 30 m site and persistent at the 15 m site throughout the observation period. In contrast, discharge dominated at the 0, 30 m and 60 m sites into the forest. Here the water table was lower and I and ET were estimated to be higher than in the “Bog”, based on the modelling results and in line with previous work (Price et al., 2003; Heijmans et al., 2004; Lafleur et al., 2005; Beheim, 2006; Bond-Lamberty et al., 2009) (Fig. 2). Flow direction only reversed in the beginning of October and during rainfall events to recharge (Fig. 2). These patterns are supported by the chloride concentrations (Fig. 4), which indicate discharge in the “Forest” and under the channel, and increased recharge near the channel compared to the “Bog” sites that were farther away. Chloride concentration patterns were similar to those determined by Blodau and Siems (2010) two years earlier, suggesting a consistent flow pattern. Drainage and evolving tree cover have thus distinctly altered the groundwater flow patterns.

An interesting aspect was the effect of increased hydraulic conductivity on groundwater flow at depths > 1 m under the “Forest” area. Due to the inversion of the K_h profile the groundwater flow through the “Forest” catotelm was relatively more significant (Table 2 and 3). Modelled flow velocities in the catotelm, here operationally defined by peat depth > 0.55 m, were also higher than in the “Bog” (Fig. 5a and b), likely with consequences for DIC and DOC export, as was documented for a drained poor fen (Strack et al., 2008). In pristine bogs the catotelm often has been assumed to be of limited importance for groundwater movement (Siegel and Glaser, 1992; Baird et al., 2008),

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especially when underlain by sediments of low K_h (Reeve et al., 2000). It has been considered to primarily dampen groundwater dynamics in the hydrological system (Siegel and Glaser, 2006). According to our results, severe drainage involving invasion of trees may alter this role to some extent.

4.3 Ecohydrological response to drying

Our intention was to study long-term drainage as an analogue to persistently drier conditions that may occur in the future (Tarnocai, 2006). The analogue is of limited applicability as changes in the water balance will be spatially more homogenous and temporally more complex than induced by unidirectional drainage. Based on our study, little can thus be said about the effect of lower precipitation and water tables in the interior of peatlands. Many of the continental peatlands that may be potentially affected by drying, such as north of the Canadian prairies, are also already treed today (Glaser and Janssens, 1986; Vitt et al., 1994), which limits an extrapolation of our results. Nevertheless, some insight can be gained how the system of vegetation and hydrologic structures and processes reacted locally and what processes occurred that need further attention.

Long-term drainage entailed a complex ecohydrological response at the Mer Bleue bog that differed between the small eastern and the large western peatland area. Where the water table decline was strongest, near the ditch and in the now forested, small eastern area, the establishment and growth of vascular plants was triggered after drainage. This is in agreement with other sites in previous studies (Pellerin et al., 2009). Vegetation change began soon after drainage according to pollen analyses (Talbot et al., 2010). According to our modelling results, increases in ET and I, and decreases in groundwater recharge occur in the treed area in summer, otherwise the response in hydraulic heads during the measurement period and chloride concentration patterns cannot be explained. This process was potentially aided by the increased hydraulic conductivity in the catotelm. Extrapolating this pattern into the past, it is reasonable to assume that the summer discharge conditions occurred already earlier in the last

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century, at least once tree and shrub biomass came close to present levels, which are currently up to an order of magnitude larger than in the open bog (Talbot, 2010). We further assume that discharge improved conditions for vegetation change by supplying nutrients from the deeper groundwater during this period, although we cannot support this idea by measurements of nutrient fluxes. Counteracting this chain of change, subsidence and peat decomposition accelerated post-drainage (Blodau and Siems, 2010) and reduced hydraulic conductivity in the surface peat of the eastern forested peatland area and near the drainage ditch.

On the large western side the latter process apparently "plugged" the system against drainage, similarly as described for pristine bog margins by Baird et al. (2008). The impact of drainage thus remained limited to a thin zone near the drainage channel – the bog remained intact and thus showed a similar resilience as observed by Strack and Waddington (2007) following experimental drainage of a poor fen. Given that pre-drainage the entire area was likely reasonably homogeneous in terms of peat depth, vegetation, and hydrology (Talbot et al., 2010), the difference in ecohydrologic response can only be attributed to the difference in the extent of peatland area west and east of the drainage ditch (Fig. 1). Following this idea, our findings suggest that the margins of larger bogs are more resilient with respect to drier conditions in their drainage networks, for example resulting from warmer and drier climate. Ecosystem stabilization would be effective because of the larger water flux from the catchment and the protection of the bog from water loss by even lower K_h near drainage channels than have been documented under current conditions (Baird et al., 2008). Smaller peatlands may be less resilient because the water table drawdown would extend farther and may trigger growth of upland vascular plants (Moore et al., 2002; Laiho, 2006; Talbot et al., 2010). Positive feedbacks may reinforce the changes in vegetation cover and hydrologic processes that were seen at the "Forest" site. Although these interpretations are intuitive, they are speculative. Further systematic studies relating long-term drying and ecohydrological functioning across peatlands of different size will be required to substantiate

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or refute our interpretations and to identify potential thresholds upon which vegetation and ecohydrological change occurs.

Supplementary material related to this article is available online at: http://www.hydrol-earth-syst-sci-discuss.net/10/33/2013/ hessd-10-33-2013-supplement.pdf.

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Table 1. Porosity and hydraulic conductivity estimates for each layer implemented in the steady-state and transient MODFLOW models.

						Site					
Parameter	B200	B60	B30	B15	BD	D	FD	F15	F30	F60	F200
Surface elevation above ditch (m)											
	1.31	1.04	0.96	0.79	0.21	0	0.19	0.44	0.50	0.74	0.95
Porosity (–)											
Layer 1	0.96	0.96	0.96	0.97	0.99	0.99	0.98	0.94	0.94	0.94	0.94
Layer 2	0.93	0.95	0.93	0.93	0.98	0.99	0.98	0.92	0.92	0.94	0.95
Layer 3	0.94	0.94	0.93	0.93	0.98	0.99	0.98	0.93	0.93	0.94	0.94
Layer 4	0.94	0.94	0.93	0.93	0.96	0.97	0.96	0.93	0.93	0.94	0.94
Layer 5	0.93	0.93	0.92	0.92	0.94	0.95	0.94	0.93	0.93	0.91	0.93
Layer 6	0.93	0.93	0.92	0.92	0.93	0.93	0.93	0.93	0.93	0.91	0.93
Layer 7	0.93	0.93	0.92	0.92	0.93	0.93	0.93	0.93	0.93	0.91	0.93
Horizontal hydraulic conductivity ($\mathrm{m\,s^{-1}}$)											
Layer 1	4.4e^{-5}	4.7e^{-6}	8.2e^{-6}	8.0e^{-7}	2.5e^{-6}	1.0e^{-3}	9.8e^{-4}	5.0e^{-6}	5.0e^{-6}	2.5e^{-6}	1.0e^{-6}
Layer 2	1.6e^{-7}	1.0e^{-7}	3.0e^{-6}	1.3e^{-7}	2.5e^{-6}	1.0e^{-3}	2.5e^{-6}	5.0e^{-8}	6.0e^{-7}	5.0e^{-6}	5.0e^{-6}
Layer 3	8.5e^{-9}	1.5e^{-7}	1.5e^{-8}	5.0e^{-9}	8.0e^{-8}	1.0e^{-3}	1.3e^{-7}	3.0e^{-7}	4.0e^{-7}	1.3e^{-6}	3.0e^{-5}
Layer 4	3.8e^{-7}	1.5e^{-7}	1.5e^{-8}	5.0e^{-9}	1.3e^{-7}	1.0e^{-4}	1.3e^{-7}	1.3e^{-7}	4.0e^{-7}	1.6e^{-6}	3.0e^{-5}
Layer 5	3.8e^{-7}	5.9e^{-8}	1.0e^{-8}	1.0e^{-8}	1.3e^{-7}	1.0e^{-4}	3.0e^{-6}	3.0e^{-6}	2.5e^{-5}	4.9e^{-7}	5.2e^{-7}
Layer 6	1.8e^{-8}	1.0e^{-8}	1.0e^{-8}	1.0e^{-8}	1.0e^{-8}	1.0e^{-8}	1.0e^{-8}	1.0e^{-5}	2.5e^{-5}	4.9e^{-7}	5.2e^{-7}
Layer 7	1.8e^{-8}	8.0e^{-8}	8.0e^{-8}	8.0e^{-8}	8.0e^{-8}	8.0e^{-8}	8.0e^{-8}	1.0e^{-5}	2.5e^{-5}	4.0e^{-7}	5.2e^{-7}

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Table 2. Estimates of the horizontal hydraulic conductivities across the transect determined by slug tests after Hvorslev (1951).

	Horizontal hydraulic conductivity ($m\,s^{-1}$)										
Depth (m)	B200	B60	B30	B15	BD	D	FD	F15	F30	F60	F200
0.25	$2.0e^{-4}$	–	–	$7.2e^{-4}$	$> 5.0e^{-3}$	$6.9e^{-6}$	–	–	–	–	–
0.5	$4.4e^{-5}$	$4.7e^{-6}$	$6.2e^{-6}$	$8.4e^{-8}$	$2.0e^{-6}$	$1.2e^{-3}$	$4.5e^{-4}$	$2.5e^{-8}$	$1.0e^{-7}$	–	$7.6e^{-8}$
0.75	$1.6e^{-7}$	$9.6e^{-8}$	$1.9e^{-7}$	$8.9e^{-9}$	$1.4e^{-7}$	$1.2e^{-3}$	$1.8e^{-5}$	$1.6e^{-7}$	$8.3e^{-7}$	$5.2e^{-6}$	$4.5e^{-6}$
1.0	$8.3e^{-9}$	$1.5e^{-7}$	$2.3e^{-8}$	$7.3e^{-9}$	$9.1e^{-7}$	$1.3e^{-4}$	$2.9e^{-7}$	$1.8e^{-7}$	$3.7e^{-7}$	$1.6e^{-6}$	$3.0e^{-5}$
1.75	–	–	–	–	$4.9e^{-7}$	$5.1e^{-7}$	–	–	–	–	–
2.0	$3.8e^{-7}$	$4.9e^{-8}$	$2.7e^{-7}$	$1.6e^{-6}$	$3.9e^{-10}$	$4.1e^{-10}$	$4.6e^{-10}$	$5.5e^{-8}$	$2.7e^{-5}$	$4.3e^{-7}$	$5.2e^{-7}$
2.25	–	–	–	–	$1.1e^{-7}$	$6.4e^{-6}$	–	–	–	–	–
2.5	–	–	–	–	$1.0e^{-6}$	$5.9e^{-6}$	–	–	–	–	–
3.0	$1.8e^{-8}$	$1.1e^{-9}$	$1.3e^{-9}$	$5.9e^{-6}$	$3.0e^{-10}$	$2.1e^{-10}$	$2.2e^{-10}$	$3.2e^{-10}$	$5.9e^{-10}$	–	–

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Table 3. Results of a paired t-test comparing the hydraulic conductivities of the bog and forest side for each depth.

Depth	Bog	Forest	Significance (p-value)
0.5	B200, B30, B15	F200, F30, F15	No (0.9396)
0.75	B200, B60, B30, B15	F200, F60, F30, F15	Yes (0.0064)
1.0	B200, B60, B30, B15	F200, F60, F30, F15	Yes (0.0295)
2.0	B200, B60, B30, B15	F200, F60, F30, F15	No (0.3228)
3.0	B200, B60, B30, B15	F200, F60, F30, F15	No (0.9298)

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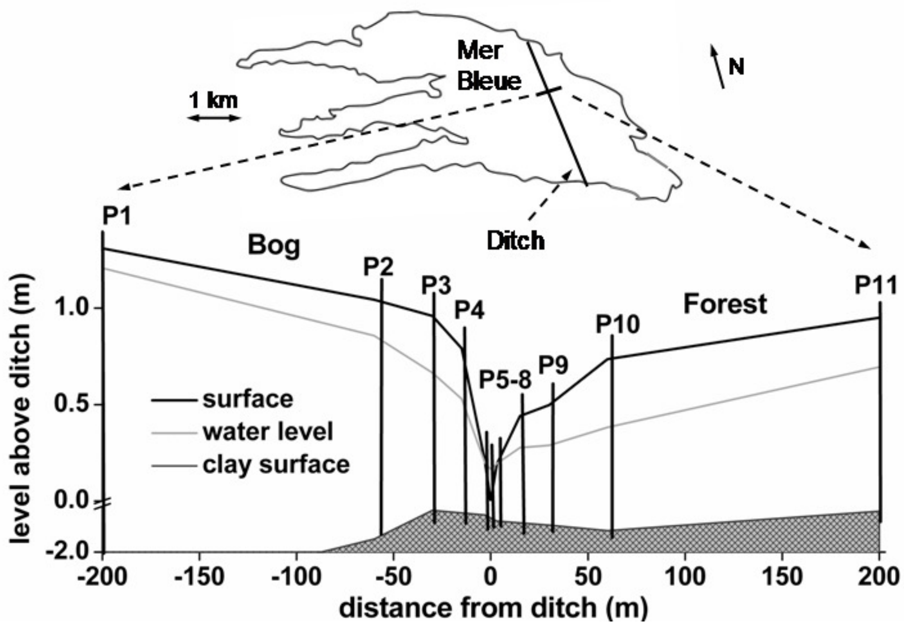

Fig. 1. Cross section of the study transect installed perpendicular to the drainage ditch in the western part of the Mer Bleue Bog, Ontario, Canada. It illustrates the position of the peat surface, piezometer nest arrangement (P1–P11), the water level and the peat-clay interface. Additional piezometer nests, serving as reference sites, were installed 600 m from the ditch in the open bog and 45 m from the ditch on both sides. Peat depth is vertically exaggerated.

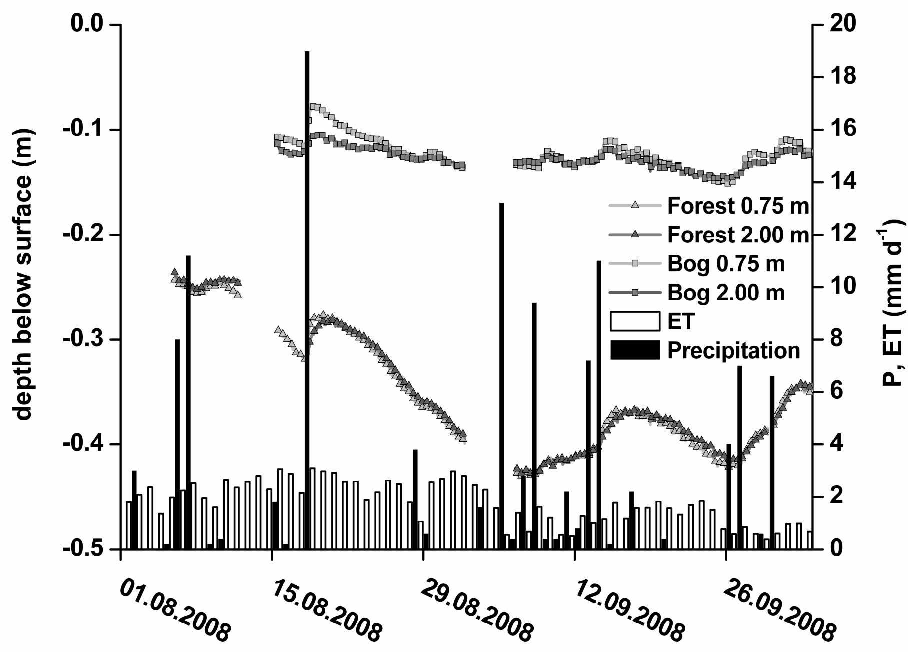

Fig. 2. Hydraulic potentials in 0.75 and 2.0 m depth of the 200 m site in the forest (light and dark grey triangles) and the bog (light and dark grey squares). P in mm d^{-1} is displayed as black bars and ET in mm d^{-1} as white bars.

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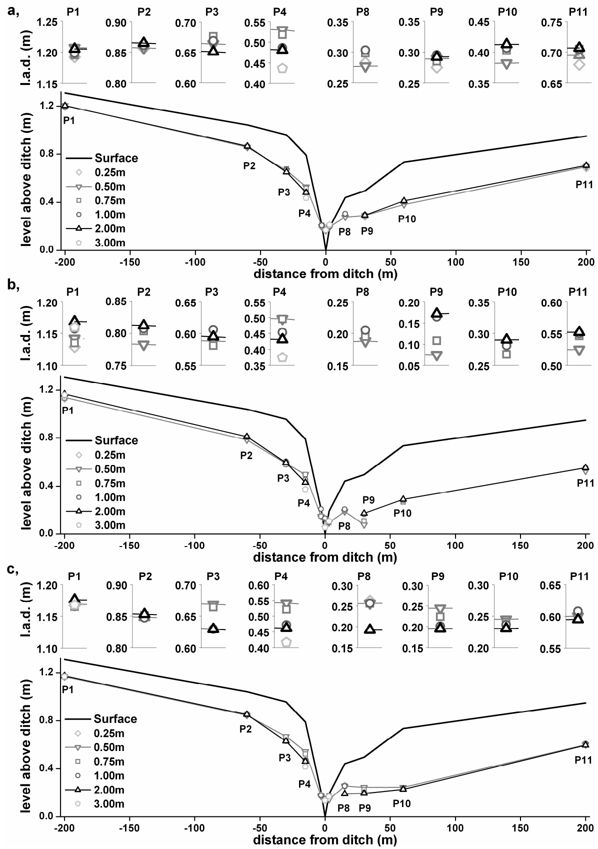

Fig. 3. Hydraulic head measurements of all piezometer nests at three different days ((a) 13 August, (b) 2 September and (c) 4 October). Hydraulic potential at 0.5 m depth is marked with grey reversed triangles and at 2.0 m depth with black triangles. Rectangle boxes are close ups of the hydraulic potentials of the piezometer nests. Peat depth is vertically exaggerated.

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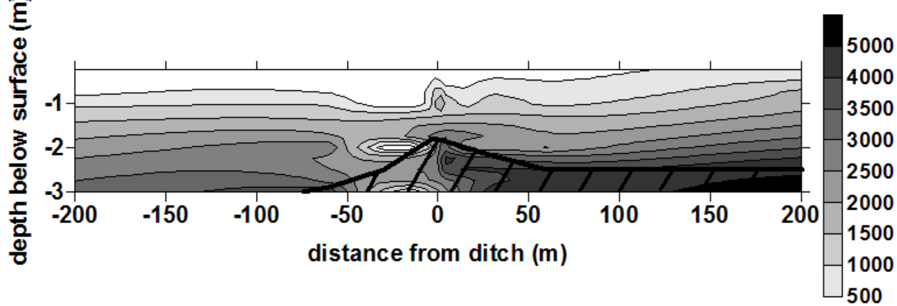

Fig. 4. Mean chloride concentrations in µmol L^{-1} in the pore water across the transect. Peat depth is vertically exaggerated.

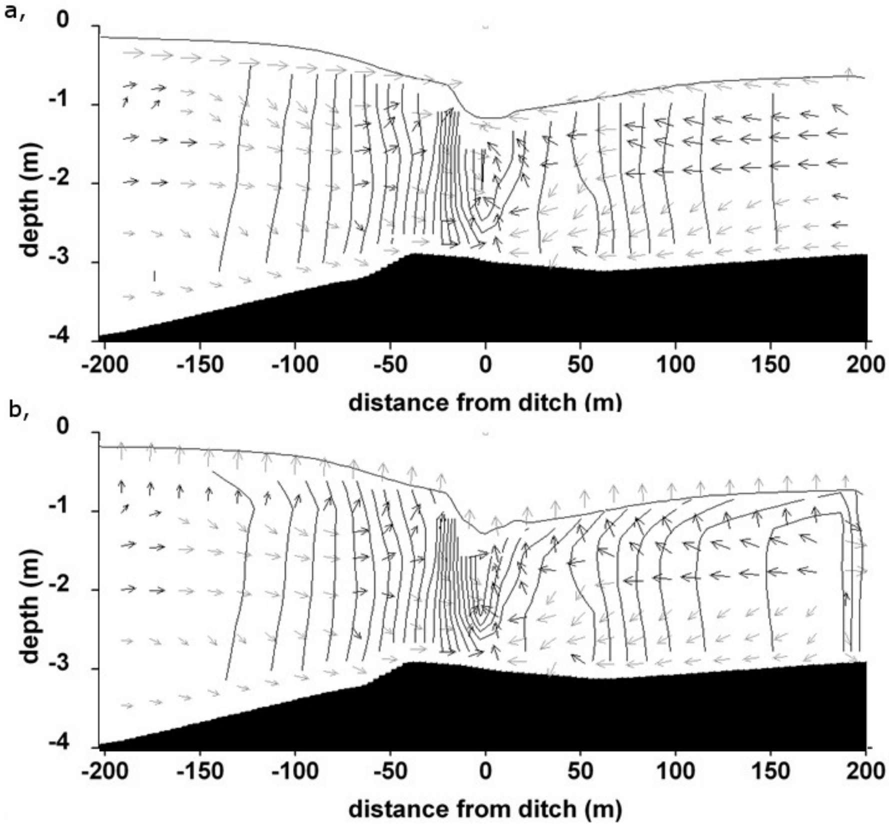

Fig. 5. (a) Results of the steady-state simulation and **(b)** of the transient simulation represented by the hydraulic conditions 4 September. Illustrated is the water table, hydraulic potentials in 0.05 m contours, flow direction (black arrows indicate upward groundwater flow and grey arrows downward flow) and flow velocity (arrow size). Peat depth is vertically exaggerated.

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