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Modelling hydrological management for the restoration of acidified floating fens

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Abstract:

Wetlands show a large decline in biodiversity. To protect and restore this biodiversity, many restoration projects are carried out. Hydrology in wetlands controls the chemical and biological processes and may be the most important factor regulating wetland function and development. Hydrological models may be used to simulate these processes and to evaluate management scenarios for restoration. HYDRUS2D, a combined saturated–unsaturated groundwater flow and transport model, is presented. This simulates near-surface hydrological processes in an acidified floating fen, with the aim to evaluate the effect of hydrological restoration in terms of conditions for biodiversity. In the acidified floating fen in the nature reserve Ilperveld (The Netherlands), a trench system was dug for the purpose of creating a runoff channel for acid rainwater in wet periods and to enable circum-neutral surface water to enter the fen in dry periods. The model is calibrated against measured conductivity values for a 5 year period. From the model simulations, it was found that lateral flow in the floating raft is limited. Furthermore, the model shows that the best management option is a combination of trenches and inundation, which gave the best soil water quality in the root zone. It is concluded that hydrological models can be used for the calculation of management scenarios in restoration projects. The combined saturated–unsaturated model concept used in this paper is able to incorporate the governing hydrological processes in the wetland root zones. Copyright © 2005 John Wiley & Sons, Ltd.

KEY WORDS floating fen; restoration; hydrology; model; ecology

INTRODUCTION

Worldwide, wetlands show a large decline in biodiversity due to drainage, eutrophication, acidification and fragmentation (Mitsch and Gosselink, 1993). To maintain wetland biodiversity, international and European agreements have been made (Ramsar, 1996) and many restoration projects of degraded wetland ecosystems have been carried out to restore biodiversity (Pfadenhauer and Klötzli, 1996). However, the success of restoration varies and is difficult to predict. In many cases, it is unclear whether the failure is caused by abiotic or biotic factors (Pfadenhauer and Klötzli, 1996).

Hydrology controls the chemical and biological processes in wetlands and may be the most important factor regulating wetland functioning and development (Mitsch and Gosselink, 1993). To maintain natural vegetation in wetlands, high water levels and the soil water quality in the root zone are important. Water may infiltrate the root zone by precipitation or by surface water due to inundation, and deeper groundwater from greater depth may also reach the root zone by upward seepage. Schot *et al.* (2004) have shown that so-called rainwater lenses in the root zone may float on the deeper groundwater. They showed that the dynamics of these rainwater lenses, and thereby of the soil water quality in the root zone, are dependent on the ecological and hydrological situation and on the soil hydrological properties.

However, whatever the wetland type, root-zone hydrological processes for both quantity and quality in the unsaturated and saturated parts of the soil must be incorporated to evaluate the effect of restoration measures.

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Hydrological models may help to simulate the effects of restoration measures on the water quality and quantity and are essential to manage and achieve successful restoration.

Until now, restoration projects have used hydrological models such as MODFLOW (Harbaugh and McDonald, 1996). However, these models inherently lack any representation of the near-surface conditions (MacAlister and Parking, 1999). Bradley and Gilvear (2000) showed that only a few two-dimensional transect model studies have been made (e.g. Young *et al.*, 1989; Zebb and Hemond, 1998). They concluded that a saturated–unsaturated model needs to be considered urgently for a representative wetland transect, because the way wetland vegetation takes up water by evapotranspiration will partly determine the pattern of seepage fluxes. Moreover, water quality is not taken into account.

Recent developments in software enable the incorporation of hydrological processes in both the saturated and unsaturated soil, for both quantity and quality. In this paper we use a combined saturated–unsaturated groundwater flow and transport model, which is able to simulate restoration measures for a case situation in so-called floating fens.

Peat has been dredged for fuel in many lowland areas of the Netherlands, resulting in open landscapes of turf ponds with narrow baulks of uncut peatland in between (Borger, 1992). These turf ponds terrestrialized again with succession stages from floating fens to complete fens with a solid peat soil. Floating fens are characterized by a species-rich vegetation with many rare species and are characterized by nutrient poor and circum-neutral pH conditions (van Wirdum, 1991; Wheeler and Shaw, 1995; Bootsma and Wassen, 1996). The main external problems of these floating fens are the decrease of calcium-rich seepage water, the increased inflow of acidifying components and nitrogen, forming acid- and nutrient-rich root-zone water. As a result, diversity of vegetation decreases. In contrast to other wetland types, the threat of drainage on vegetation is low due to the floating character of the raft resulting in constant groundwater levels.

In a restoration project of the floating fen Ilperveld in the Netherlands, it was the aim to remove the infiltrated acid rainwater from the upper water layer. Restoration measures involved both drainage of acid rainwater and enhanced intrusion of calcium-rich surface water. The effects of these measures have been monitored and evaluated in terms of vegetation development and water and soil quality composition (Beltman *et al.*, 2001; Bootsma *et al.*, 2002). However, little is known about the hydrological effects of these measures on soil water quality.

The aim of this paper is to model root-zone hydrological processes in an acidified floating fen, with the aim to evaluate the effects of hydrological management on root-zone acidity.

MATERIALS AND METHODS

Site description

The Ilperveld site is a slightly brackish peat area located 5 km north of Amsterdam in the Netherlands. The nature reserve is 1160 ha, of which 230 ha is open water. The peat layer is about 15–20 m thick. Locally, peat was excavated from the 12th until the 19th centuries, creating a landscape of turf ponds, floating rafts and baulks of uncut peatland (Figure 1). The Ilperveld is a recharge area, with a downward seepage flux of 0·1 mm day^{-1} (NITG-TNO, 2004). The surface water level is 10 cm below the fen ground surface.

In 1944, a species-rich *Pallavicinio–Sphagnetum* typicum vegetation was present (Bootsma *et al.*, 2002). Nowadays, the central part of the floating raft is isolated from the surface water and is strongly acidified due to atmospheric deposition of acidified components. The natural pH is lowered to 3·3. (see Table I). These circumstances favour *Sphagnum* and *Polytrichum* species and thereby reduce biodiversity (e.g. Bowden, 1991; Kooijman, 1992). In 1992, a restoration experiment was carried out with the aim to achieve circum-neutral conditions in the root zone. The experimental area consisted of two baulks of uncut peatland, connected by a 0·8 m thickness floating raft. This raft consists of vertical layered biomass of *Sphagnum* and *Polytrichum*. In January 1992, shallow trenches ($0{\cdot}3 \times 0{\cdot}3 \times 30$ m^3) were dug, which were connected with a supply and distribution ditch. The aim of the restoration experiment was to create a drainage flow of acid rainwater to

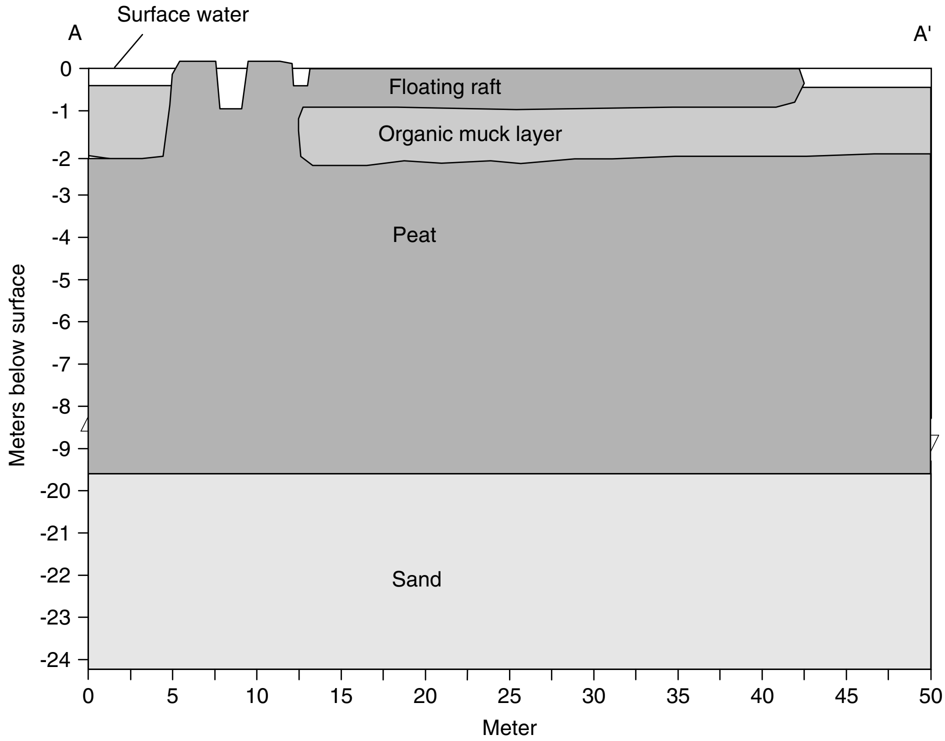

Figure 1. Cross-section A–A′ of the floating fen Ilperveld

Table I. Mean values of groundwater composition at 0.25 cm depth during the summers of 1991 and 1996. In 1996, the samples were taken 0.35 m from the trench

	pH	EC_{water} (μS cm^{-1})	Ca^{2+} (mg l^{-1})	HCO_3^- (mg l^{-1})	SO_4^{2-} (mg l^{-1})	Cl^- (mg l^{-1})	Na^+ (mg l^{-1})
1991	3·3	253	5·6	0·5	41	42	25·2
1996	5·6	1886	46·6	105	147	534	298

the trenches in wet periods and to surplus the trenches with buffered surface water, which infiltrates the root zone in dry periods. Within this experiment, six drained plots were created (Figure 2).

Data

The Royal Meteorological Institute (KNMI) in the Netherlands has measured precipitation, global radiation and temperature from 1991 to 1996. Mean precipitation was 895 mm $year^{-1}$. The winter of 1995–96 was relatively dry, resulting in a mean precipitation in 1996 of 660 mm $year^{-1}$. Potential evapotranspiration was calculated from global radiation and temperature by the Makkink equation (Makkink, 1957) and by using a combined crop factor for *Sphagnum* and *Polytrichum* (Koerselman and Beltman, 1988; van Wirdum, 1991). The throughfall factor of the vegetation was 0·9 (Koerselman and Beltman, 1988; van Wirdum, 1991), resulting in a mean throughfall rate of 806 mm $year^{-1}$.

Between 1991 and 1996, the phreatic water in the peat soil was measured in cross-section B–B′ (Figure 2) with an electrical conductivity (EC)/temperature probe (van Wirdum, 1991). Measurements were made at 10 cm intervals between 0·1 and 1·5 m below the surface. Van Wirdum (1991) showed that these EC

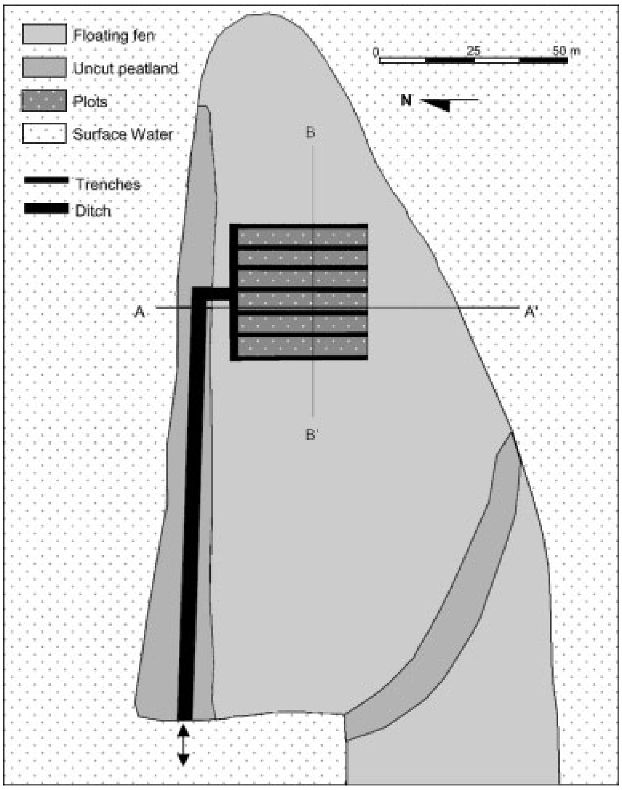

Figure 2. Schematic map of the experimental design in Ilperveld. The supply and drainage ditches and the distribution trenches between the six experimental plots were created in January 1992

measurements can be used as an indicator of the distribution of infiltrated ion-poor rainwater and ion-rich groundwater. Figure 3 shows measured scaled EC values of one drained plot during a winter, spring and summer situation. EC_{probe} measurements were linearly correlated to EC_{water} measurements. During the winter and spring situation, maximum EC values are found at a depth of 0·8 m, directly below the floating raft. During summer, maximum EC values were found near to the shallow trenches due to the infiltration of surface water with relatively high EC values. In 1996, pH was raised to 5·6 ($\sigma_{pH} = 1{\cdot}1$, $n = 18$) with an EC of 1886 μS cm^{-1} ($\sigma_{EC} = 1010$ μS cm^{-1}, $n = 18$) at 0·25 m depth, 0·35 m from the trenches. In Figure 3b and c it is shown that the lateral inflow of surface water from the trenches appears to be small, certainly at the root zone in the middle of the plot. However, below the floating raft a lateral flow seems to be more important, given a rise in EC from the bottom to the top of the system. Table I shows the chemical composition of the superficial groundwater.

Model

The numerical two-dimensional unsaturated–saturated groundwater flow and transport model Hydrus-2D (Simunek *et al.*, 1999) was used to simulate the dynamics of infiltrated rainwater, groundwater, surface water and root water uptake. Hydrus-2D uses a finite-element method to solve the partial differential equations in the saturated zone. For the unsaturated zone, Richards' equation is solved. Because of the nonlinear nature of the Richards equation, an iterative process is used until convergence is obtained. Unsaturated soil hydraulic properties are described by van Genuchten (1980) equations (van Genuchten, 1980). The calculated Makkink potential evapotranspiration is split into potential plant transpiration, soil evaporation and

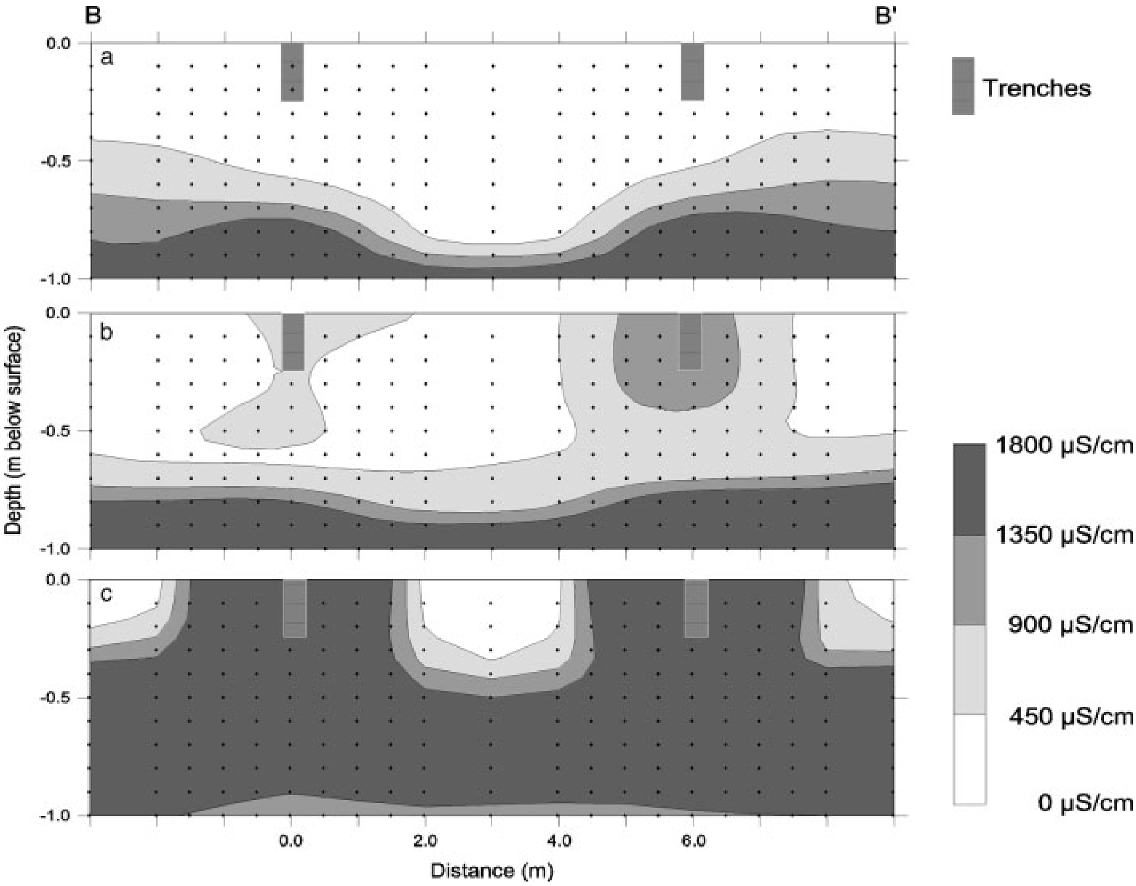

Figure 3. Interpolated EC measurements of part of cross-section B–B′; typical situations during winter (a), spring (b) and summer (c). Dots represent measurement points

interception evaporation. Potential transpiration and soil evaporation are used as input for the Hydrus model. Solute transport is considered as advective-dispersive transport in the liquid phase.

Of cross-section B–B′, one experimental plot with two trenches was modelled (Figure 4). Left and right model boundaries represent flow lines located between two trenches, making them no-flow boundaries. The lower model boundary was defined as a constant-flux boundary, assuming downward seepage at all times. The model was set to remove groundwater at the soil surface instantly (e.g. surface runoff) when the phreatic water level exceeds the soil surface. Ponding was not considered. The model trenches have a fixed-head boundary applied below the trench water level. Trench water level was set to 10 cm below the rafts' surface. Concentration (Cauchy type) solute boundary conditions of the trenches were used. The concentration of the surface water was set to 1800 µS cm^{-1} and rainwater to 0 µS cm^{-1}. Root water uptake was modelled with the Feddes water uptake model (Feddes *et al*., 1978). The standard form of this function is that water uptake is assumed to be zero close to saturation. Because mosses can also extract water from the saturated part, the function is changed in such a way that root water is extracted from both the saturated and unsaturated parts. Roots extract water with local concentration. The initial model parameters are given in Table II.

The model network consists of 3646 nodes and 7056 triangles generated by an automatic grid generator (Simunek *et al*., 1999). The distance between nodes is about 0·25 m in the centre of the section and about 0·05 m near the upper boundary and near the drainage canals. The higher network densities serve

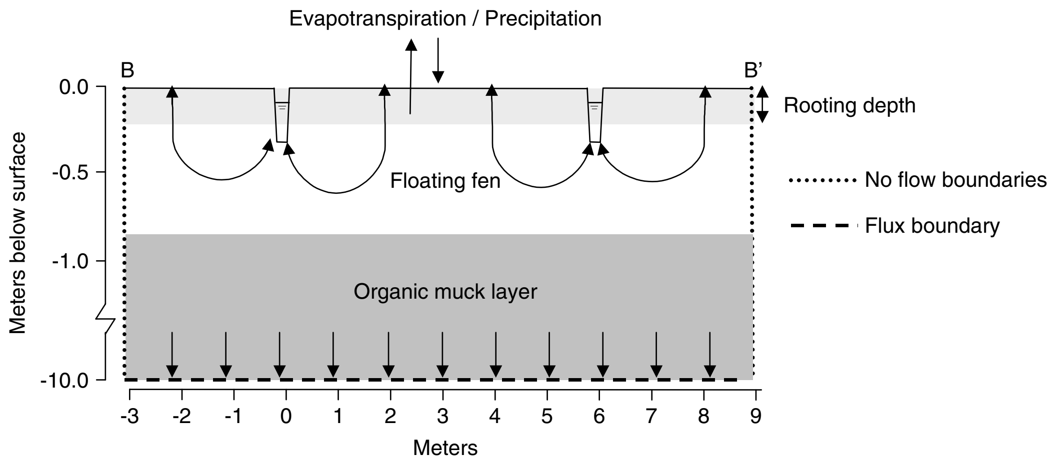

Figure 4. Two-dimensional model setup of one drained plot with two trenches. Lower model boundary condition is a flux boundary; left and right model boundaries represent flow lines located between two trenches, making them no flow boundaries

Table II. Fixed and variable parameter values and boundary conditions

Root depth (m)	0·2
K_{raft} (m day^{-1})	10–100
K_{muck} (m day^{-1})	1000
Saturated water content θ_{sat}	0·7
Lower boundary downward flux (m day^{-1})	0·1
Thickness floating fen (m)	0·8
Depth water level (m)	−0·1
Experimental design of trenches	
Depth (m)	0·3
Separation (m)	6

to minimize model errors at the contact between unsaturated and saturated groundwater and locations of concentrated groundwater in- and out-flow. The hydraulic conductivity K_{sat} of the floating raft is between 50 and 100 m day^{-1} with a saturated water content $\theta_{sat} = 0{\cdot}7$ (Koerselman, 1989; van Wirdum, 1991) and checked with the Unsaturated Soil Hydraulic Database (UNSODA; Nemes *et al.*, 2001). A thick layer of organic muck with a very high hydraulic conductivity (1000 m day^{-1}) is situated below the floating raft. Van Wirdum (1991) found K values underneath other floating rafts of between 500 and 1500 m day^{-1} based on the hydraulic gradient and the water balance. The anisotropy of the hydraulic conductivity of both layers is calibrated.

Owing to the high hydraulic conductivities, a day-to-day dynamic of infiltrating surface water and drained rainwater is expected. Therefore, daily values of precipitation and potential evapotranspiration were used between 1993 and 1996. The internal time step was between 0·1 and 0·001 days. Initial values of concentration and pressure head are found by running the model for 3 years as an initialization.

Calibration and simulation of restoration measures

In this study, we are mainly interested in the hydrological processes in the root zone. Therefore, we have only used EC cross-section measurements at 20 and 40 cm depths for model calibration of simulated concentrations. Because of model symmetry on both sides of the middle line between the trenches, calibration points were assigned at distances from the trench of 0·5, 1·0, 1·5, 2 and 3 m at depths of 20 and 40 cm, resulting in 10 calibration points of solute concentration. These modelled calibration points are compared with averaged EC measurements, which are composed from three drained plots resulting in a maximum of six measurements for every calibration point during time. In total, four time points were used for calibration: August 1993, May 1994, August 1995 and August 1996. After calibration, the model was used to calculate the following types of experimental design of restoration measures: (i) different depths of trenches; (ii) different densities of trenches; (iii) a decrease of the thickness of the floating raft by sodding, i.e. the cutting of the upper part of the sod; (iv) inundation with surface water. The average simulated concentration of the water for the total root zone C_{root} is used as a criterion to find the best restoration measure. Beltman *et al.* (1996) found in manipulation experiments that restoration of a freshwater floating raft in the Vechtplassen (The Netherlands) would not be possible with EC values smaller than 200 µS cm^{-1}. However, the floating raft of the Ilperveld is located in a slightly brackish peat area, meaning a higher EC. Moreover, the measured null situation in 1991 (Table I) for the Ilperveld was EC = 253 µS cm^{-1} ($\sigma = 69$ µS cm^{-1}, $n = 12$). Therefore, we used 500 µS cm^{-1} as the threshold criterion and calculated the cover of the root zone where this concentration is exceeded in time between 1 April and 1 October. For instance, a cover of 0·4 means that 40% of the horizontal area at the top of the system (root zone) exceeds 500 µS cm^{-1} during the growing season.

RESULTS

Calibration of drained plots

A satisfying calibration result could only be reached by adding anisotropy to the floating raft and the organic muck layer. The best calibration result was found with $K_{\text{h-raft}} = 10$ m day^{-1}, $K_{\text{v-raft}} = 50\ m\ day^{-1}$, $K_{\text{h-muck}} = 2000$ m day^{-1}, $K_{\text{v-muck}} = 1000$ m day^{-1}, so giving an anisotropy of the raft $K_{\text{h-raft}}/K_{\text{v-raft}} = 0{\cdot}2$ and of the organic muck layer $\mathrm{K}_{\text{h-muck}}/\mathrm{K}_{\text{v-muck}} = 2$. Figure 5 shows the measured versus simulated solute concentration, with $r^2 = 0{\cdot}64$. Dotted lines represent the mean standard deviation (200 µS cm^{-1}) of the measurements.

Table III shows sensitivity analyses results. Having the same anisotropy but changing $\mathrm{K}_{\mathrm{v-raft}}$ between 10 and 100 m day^{-1} only slightly changes the fit (runs 1 and 2); also, a small change in the anisotropy of the floating raft (runs 3 and 4) does not affect the model performances. However, an assumption of isotropy leads to an increase in the NRMSE to 0·36 (run 5). With these settings, the lateral influence is higher, which is

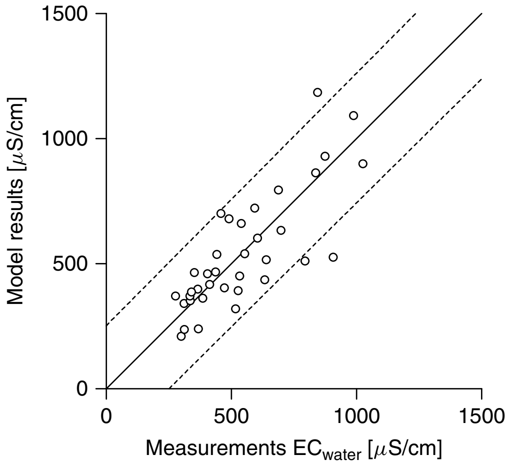

Figure 5. Measured versus modelled EC values in the root zone; r^2 is 0.64. Plotted EC measurements are composed of three drained plots resulting in a maximum of six original measurements. Mean standard deviations of the measurements are 35%, plotted as dotted lines around the 1 : 1 line

Table III. Sensitivity of model parameters. Bold means best calibration. A_{raft} is the anisotropy in the floating fen in terms of horizontal versus vertical, A_{muckis} the anisotropy of the organic muck layer. NRMSE is normalized root-mean-squared error between measurements and model results

	A_{raft}	A_{muck}	K_{raft} (m day^{-1})	NRMSE	r^2
Calibration	**0.2**	**2**	**50**	**0.25**	**0.64**
1	0·2	2	10	0·25	0·63
2	0·2	2	100	0·26	0·63
3	0·1	2	50	0·26	0·63
4	0·3	2	50	0·28	0·62
5	1·0	2	50	0·36	0·52
6	0·2	1	50	0·42	0·55
7	0·2	3	50	0·37	0·51

not confirmed by the measurements. Changing the anisotropy of the organic muck layer (runs 6 and 7) has a larger effect on the concentration in the root zone. Therefore, redistribution of water in the organic muck seems to be an important process in this system.

Owing to horizontal layering of the dead organic material, one should expect that vertical conductivity is smaller than horizontal conductivity. However, the calibration and sensitivity analyses show the opposite effect. In soil samples taken from the Ilperveld we observed no horizontal layering of *Sphagnum* and *Polytrichum* biomass, but we did in the vertical, which is probably the cause of a larger horizontal conductivity than vertical conductivity.

System dynamics

In Figure 6, the simulated solute concentrations in the root zone are shown for two observation points at 20 cm depth in which the first is 50 cm from the trench and the second is situated in the middle of the plot. Noteworthy is the highly dynamic behaviour of the concentration near the trench. Mean net recharge between 1991 and 1996, which equals precipitation minus evapotranspiration minus interception evaporation, is 300 mm on a yearly basis. Water loss due to downward seepage is 40 mm, meaning that 260 mm of rainwater will be drained to the trenches. On a yearly basis, with a trench distance of 6 m, this results in 1·5 m^3 of water per metre of trench per year being drained to the trenches. However, owing to the high saturated conductivity, a water flow due to precipitation or evapotranspiration is directly compensated by an in- or out-flow of water from the trenches. As a result, the simulated gross amount of water transported to and

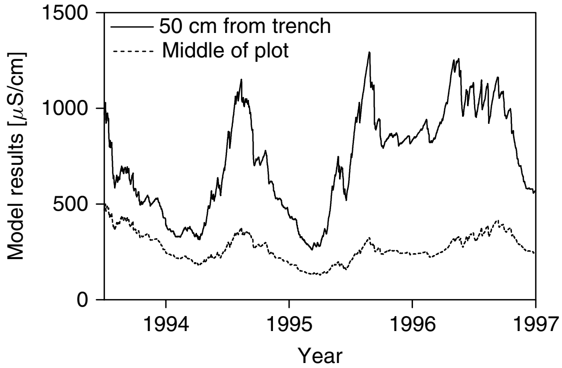

Figure 6. Simulated dynamics of the solute concentration at 50 cm from the trench and in the middle of the plot. Both observation points are at 20 cm depth

from the trenches is 7 m^3 of water per metre of trench per year. These high fluxes cause the high dynamics of solute concentration near to the trenches. In contrast, in the middle of the plot the fluxes are much lower, resulting in a much lower dynamic of solute concentration, as shown in Figure 6. In 1 day, at day number of the year (DOY) 209, a drop in concentration of 90 $\mu S\ cm^{-1}$ was modelled due to a rain event of 24 mm on this day. In contrast, in the middle of the plot, such a change would take 100 days. Figure 6 also shows the peaks of maximum concentration in 1994 and 1995 at the end of July. This implies that the concentration during the growing season, May–June, is not at its maximum. Furthermore, the solute concentration during the winter of 1995–96 remains constant at 900 $\mu S\ cm^{-1}$ at 50 cm from the trench, whereas during the other winters the concentration is decreased to 400 $\mu S\ cm^{-1}$. The high winter concentration is caused by the relatively dry winter of 1995–96: from 15 October to 15 April there was only 132 mm rain; however, in the same period in 1993–94 and 1994–95 there were respectively 620 mm and 650 mm of rain. Owing to this dry winter, less rainwater infiltrates, meaning a higher solute concentration.

Restoration measures

With the initially calibrated soil physical parameters, we have modelled different types of hydrological restoration measures. The reference situation, with trench separation, gives a mean simulated solute concentration in the root zone C_{root} of 377 $\mu S\ cm^{-1}$ with a cover of 23% (Table IV), which means that 23% of the plot area has an EC value of more then 500 $\mu S\ cm^{-1}$ during the growing season. Increasing the trench separation to 50 m (run 1) gives $C_{root} = 155\ \mu S\ cm^{-1}$, which is below the EC threshold, also at the end of the summer. The yearly dynamics of the solute concentration in systems with a large distance between trenches is low, with a maximum variation between summer and winter of 100 $\mu S\ cm^{-1}$. All solutes came originally from the organic muck layer and are transported upwards due to the root water pressure. In run 2, trench density is doubled, resulting in a much higher coverage of possible regeneration; however, in the middle of the plots, regeneration is still not possible.

In runs 3 and 4, trench depths are varied (Table IV). With deeper trenches more water is transported to and from the trenches, which results in a larger C_{root}. However, trenches at greater depth also cause a better infiltration under the root zone to the organic muck layer, meaning that the vegetation cannot extract this water. A shallower trench gives the opposite effect, with the same equal overall result. The effects of deeper or shallower trenches, however, hardly have an effect on C_{root}, which is comparable to that of the reference situation.

By cutting the sod, the thickness of the floating raft is reduced to 0·5 m (run 5, Table IV). Owing to the floating character of the raft, the surface of the raft will remain at the same level as the trench water level, i.e. 10 cm below the rafts' surface. In the model, the K_{raft} is input over 50 cm instead of 80 cm. The C_{root} increased because water from the muck layer will flow more easily from and to the trenches. Lowering the

Table IV. Types of restoration measure with calculated mean concentration of water taken up by the roots C_{root} between 1992 and 1996. Cover means the fraction in which the root zone exceeds the threshold solute concentration of 500 $\mu S\ cm^{-1}$

	C_{root} ($\mu S\ cm^{-1}$)	Cover
Reference situation	377	0·23
1: Distance trenches 50 m	155	0·0
2: Distance trenches 3 m	500	0·56
3: Trenches 0.2 m depth	347	0·22
4: Trenches 0.6 m depth	355	0·21
5: Floating fen 0.5 m thick	409	0·29
6: Surface water level −0·25 m	396	0·22
7: Inundation 1 April–1 June	646	0·86

surface water level to $-0{\cdot}25$ m (run 6), just above the bottom of the trench, results in lower groundwater levels and thereby higher drainage to the trenches, but also lower intrusion, leading to an almost similar C_{root} as found in the reference run.

Finally, for every year during the 3 months between 1 April to 1 July the plots are inundated with 1 mm day^{-1} of surface water, resulting in $C_{\mathrm{root}} == 646\ \mu\mathrm{S\ cm}^{-1}$. Only small differences were found near to the trenches, but large differences could be observed at the middle of the plot due to the measure of inundated water. As a result, coverage is nearly 90%. This is clearly the most effective restoration measure.

DISCUSSION

It is shown that the proposed combined saturated–unsaturated groundwater flow and transport model can be used to simulate the effects of hydrological restoration measures. Other types of hydrological model used for wetland restoration have important shortcomings and, thereby, give unrealistic estimates of restoration measures and management. One of the reasons for the success of our proposed model concept is the representative way of including transpiration and the root characteristics of the vegetation in the model. Owing to the fact that unsaturated zone and root water uptake were included in the hydrological model, it was found that daily values of rainfall and evapotranspiration were essential to model root-zone hydrological processes. Moreover, hydrological models that only use net recharge values can never establish the hydrological response of vegetation, due to the lack of root-zone fluxes. A second reason for the success is the use of the transport model. In the restoration project of the floating fen it was found that, even in a net infiltration area, by adding drainage and supply trenches, upward water flow during summer is found. Without having this model concept, it was not clear where the buffered solutes came from.

Furthermore, it was shown that a hydrological model could be helpful for management in wetlands. However, the conclusions in terms of successful restoration measures cannot be copied to other types of wetland. The lateral water movement in this floating raft is very small, which is in agreement with other studies. In bogs, the dominant water loss in summer is by evapotranspiration losses. Devito *et al.* (1997) show the seasonal importance of vertical flow, and Siegel *et al.* (1995) show the periodic importance of vertical flow. The hydraulic conductivity of the catotelm of raised bogs is several orders of magnitude smaller than the acrotelm (van Breemen, 1995), resulting in a more-or-less impermeable layer. As a consequence, the type of fen, bog or other wetland type can have very different patterns and rates of water fluxes dependent on the soil hydrological properties. Moreover, the success or failure of restoration measures will be dependent on the hydrological situation.

The case study of the floating fen showed that lateral movement of water in the raft is small and can only be modelled as an anisotropy in the floating raft where K_{v} is larger then K_{h}. This is not in agreement with Beckwith *et al.* (2003), who found that K_{h} was greater than K_{v} for 78% of the 400 samples investigated. However, the samples investigated consisted of moderately and poorly decomposed catotelm peat. This is in contrast to the Ilperveld site, which has a thick, living moss layer with no observed horizontal layers of decomposed material. Peat soils can have both anisotropy and spatial heterogeneity at all depths (Beckwith *et al.*, 2003). For instance, it was found that the hydraulic conductivity can vary by several orders of magnitude over just a few vertical or lateral metres (e.g. Rycroft *et al.*, 1975; Chason and Siegel, 1986; Baird *et al.*, 1997; van der Schaaf, 1999), having large effects on patterns and rates of seepage (e.g. Chason and Siegel, 1986; Baird *et al.*, 1997).

In the model simulations of the floating fen, it was found that the trench system creates a drainage flow of acid rainwater to the trenches in wet periods and improves the intrusion of buffered surface water in dry periods, resulting in a better soil water quality. This improved soil water quality resulted in a 400% increase in vegetation biodiversity of the sites near to the trenches (Bootsma *et al.*, 2002). Bootsma *et al.* (2002) found small effects of vegetation development at distances greater than 0·75 m, which agrees with the modelled soil water quality at this distance from the trench. Furthermore, in the model simulations it is shown that improvement of the soil water quality in the middle of the plots can only be reached by an increase in trench density or by inundation. A denser experimental design of trenches results in lower dynamics of water flowing from and to the trenches. Lateral flow in the middle of the plot will always remain zero. By adding buffered inundation water, the acid rainwater lens is decreased. The same effect could technically be reached by adding lime (Beltman *et al.*, 2001) at the top of the soil.

The model results show that by using trenches the maximum concentration in the root zone is found at the end of the growing season, which is not optimal for a circum-neutral ecosystem. To obtain a maximum concentration at the start of the growing season, inundation must take place during winter. Moreover, from history it is know that in the Netherlands many fens were flooded during the winter period due to insufficient pumping capacity. Furthermore, the model results show that in 1996 the conditions were much better because of the relatively dry winter, i.e. less precipitation, indicating an improvement of conditions. However, this was not observed in the vegetation development (Bootsma *et al.*, 2002).

Bootsma *et al.* (2002) have shown that the best improvement in terms of vegetation development was measured in plots with combined sod cutting and trenches. With sod cutting, the moss layer with nutrients is removed and thereby another threat to biodiversity, i.e. accumulation of nutrients, is reduced. In the middle of the sod cutting plots an increase in vegetation biodiversity was observed, whereas pH was not significantly changed (Bootsma *et al.*, 2002). This was also confirmed by the simulation results of run 5 (Table IV), where no improvement of modelled EC was observed. As a result, the positive effect of sod cutting is not based on an improvement in hydrology. Moreover, in the case of *Sphagnum* it is known that the acidification is enhanced by actively lowering the pH (Clymo, 1987; Kooijman and Bakker, 1994). So, cutting of the *Sphagnum* layer could raise the pH; however, this extra pH rise was not observed in our situation.

CONCLUSIONS

The model calibration shows that $K_{\text{v-raft}}$ is greater than $K_{\text{h-raft}}$, meaning that lateral flow to the root zone is limited in our floating rafts. Therefore, a combination of trenches and inundation will give the best result of the soil water quality in the root zone. Furthermore, it was found that transpiration and root characteristics of the vegetation must be explicitly taken into the proposed model concept to simulate the real effects of restoration measures. Understanding the hydrological situation in wetlands is a prerequisite for a successful restoration. The model concept proposed in this paper can be used to calculate the best hydrological management option to restore any specific type of wetland. Copying restoration measures to different wetlands without using a coupled saturated–unsaturated groundwater flow and transport model concept can lead to variable success. Present-day software enables better simulation of effective hydrological restoration measures and, consequently, to predict optimal conditions in the ecosystem. Hopefully, this will lead to more effective biodiversity conservation.

REFERENCES

Baird AJ, Beckwith CW, Heathwaite AL. 1997. Water movement in undamaged blanket peats. In *Blanket Mire Degradation, Proceedings*, Tallis JH, Meade R, Hulme PD (eds). Mires Research Group, British Ecological Society: Manchester.

Beckwith CW, Baird AJ, Heathwaite AL. 2003. Anisotropy and depth-related heterogeneity of hydraulic conductivity in a bog peat. I: laboratory measurements. *Hydrological Processes* **17**: 89–101.

Beltman B, van den Broek T, Bloemen S, Witsel C. 1996. Effects of restoration measures on nutrient availability in a formerly nutrient-poor floating fen after acidification and eutrophication. *Biological Conservation* **68**: 271–277.

Beltman B, van den Broek T, Barendregt A, Bootsma MC, Grootjans AP. 2001. Rehabilitation of acidified and eutrophfied fens in the Netherlands: effect of hydrologic manipulation and liming. *Ecological Engineering* **17**: 21–31.

Bootsma M, Wassen M. 1996. Environmental conditions and fen vegetation in three lowland mires. *Vegetatio* **127**: 173–189.

Bootsma MC, van den Broek T, Barendregt A, Beltman B. 2002. Rehabilitation of acidified floating fens by addition of buffered surface water. *Restoration Ecology* **10**(1): 112–121.

Borger GJ. 1992. Draining–digging–dredging; the creation of a new landscape in the peat areas of the low countries. In *Fens and Bogs in the Netherlands: Vegetation, History, Nutrient Dynamics and Conservation*, Verhoeven JTA (ed.). Kluwer Academic Publishers: Dordrecht, The Netherlands; 131–172.

Bowden RD. 1991. Inputs, outputs and accumulation of nitrogen in an early successional moss (*Polytrichum*) system. *Ecological Monographs* **6**: 207–223.

Bradley C, Gilvear D. 2000. Saturated and unsaturated flow dynamics in a floodplain wetland. *Hydrological Processes* **14**: 2945–2958.

Chason DB, Siegel DI. 1986. Hydraulic conductivity and related physical properties of peat, Lost River Peatlands, northern Minnesota. *Soil Science* **142**: 91–99.

Clymo RS. 1987. Interactions of *Sphagnum* with water and air. In *Effects of Atmospheric Pollutants on Forests, Wetlands and Agricultural Ecosystems*, Meema TCHaKM (ed.). NATO ASI Series S16; 513–529.

Devito KJ, Waddington JM, Branifireun BA. 1997. Flow reversals in peatlands influenced by local groundwater systems. *Hydrological Processes* **11**: 103–110.

Feddes RA, Kowalik PJ, Zaradny H. 1978. *Simulation of Field Water Use and Crop Yield*. Simulation Monograph. Pudoc: Wageningen.

Harbaugh AW, McDonald MG. 1996. User documentation for MODFLOW-96, an update to the USGS modular finite-difference ground water flow model. US Geological Survey Open File, 96.

Koerselman W. 1989. Groundwater and surface water hydrology of a small groundwater-fed fen. *Wetlands Ecology and Management* **1**: 31–43.

Koerselman W, Beltman B. 1988. Evapotranspiration from fens in relation to Penman's potential free water evapotranspiration and pan evapotranspiration. *Aquatic Botany* **31**: 307–320.

Kooijman AM. 1992. The decrease of rich-fen bryophytes in the Netherlands. *Biological Conservation* **35**: 139–143.

Kooijman AM, Bakker C. 1994. The acidification capacity of wetland bryophytes as influenced by simulated clean and polluted rain. *Aquatic Botany* **48**(2): 133–144.

MacAlister C, Parking G. 1999. Towards a whole-system model for the hydrology of peat mires. In *Patterned Mires and Mire Pools: Origin and Development; Flora and Fauna*, Standen V, Tallis JH, Meade R (eds). British Ecological Society: London; 116–126.

Makkink GF. 1957. Testing the Penman formula by means of lysimeters. *International Journal of Water Engineering* **11**: 277–288.

Mitsch WJ, Gosselink JG. 1993. *Wetlands*, second edition. Van Nostrand Reinhold: New York.

Nemes A, Schaap MG, Leij FJ, Wosten JHM. 2001. Description of the unsaturated soil hydraulic database UNSODA version 2·0. *Journal of Hydrology* **251**: 151–162.

NITG-TNO. 2004. *Groundwater map of the Netherlands*. Netherlands Institute of Applied GeoScience.

Pfadenhauer J, Klötzli F. 1996. Restoration experiments in middle European terrestrial ecosystems: an overview. *Vegetatio* **126**: 101–115.

Ramsar Convention Bureau. 1996. *Ramsar Convention Bureau. Wetlands and Biological Diversity: Cooperation between The Convention of Wetlands of International Importance especially as Waterfowl Habitat (Ramsar, Iran, 1971) and The Convention on Biological Diversity*, Document UNEP/CBD/COP/3/Inf.21.

Rycroft DW, Williams DJA, Ingram HAP. 1975. The transmission of water through peat. II. Field experiments. *Journal of Ecology* **63**: 557–568.

Schot PP, Dekker SC, Poot A. 2004. The dynamic form of rainwater lenses in drained fens. *Journal of Hydrology* **293**: 74–84.

Siegel D, Reeve A, Glaser P, Romanowicz E. 1995. Climate-driven flushing of pore water in peatlands. *Nature* **374**: 531–533.

Simunek J, Sejna M, van Genuchten MT. 1999. *The HYDRUS2D software package for simulating the two-dimensional movement of water, heat, and multiple solutes in variably-saturate media*, Version 1·0, IGWMC-TPS-53, International Ground Water Modeling Center, Colorado School of Mines: Golden, CO.

Van der Schaaf S. 1999. Analysis of the hydrology of raised bogs in the Irish Midlands—a case study of Ranheenmore Bog and Clara Bog. PhD thesis, Wageningen Agricultural University.

Van Breemen N. 1995. How *Sphagnum* bogs down other plants. *Trends in Ecology and Evolution* **10**(7): 270–275.

Van Genuchten MT. 1980. A closed form equation for predicting the hydraulic conductivity of unsaturated soils. *Soil Science Society of America Journal* **44**: 892–898.

Van Wirdum G. 1991. *Vegetation and hydrology of floating rich-fens*. PhD thesis, University of Amsterdam.

Wheeler BD, Shaw SC. 1995. A focus on fens—controls on the composition of fen vegetation in relation to restoration. In *Restoration of Temperate Wetlands*, Wheeler BD, Shaw SC, Fojt WJ, Robertson RA (eds). Wiley: Chichester; 33–48.

Young EG, Leeds-Harrison PB, Chapman JM. 1989. Modelling water-table movement in flat low-lying lands. *Hydrological Processes* **3**: 301–315.

Zebb PJ, Hemond HF. 1998. Hydrologic response of a wetland to changing moisture conditions: modeling effects of soil heterogeneity. *Climatic Change* **40**: 211–227.

