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# EVALUATING THE IMPACT OF RIVER RESTORATION ON THE LOCAL GROUNDWATER AND ECOLOGICAL SYSTEM: A CASE STUDY IN NE FLANDERS

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(14 figures and 3 tables)

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**ABSTRACT**. River restoration changes the interaction between groundwater and surface water. Therefore, it is expected to have an impact on ecosystems at the interface between groundwater and surface water. Quantifying and generalizing the level of change of this interaction for different hydrogeological environments is scientifically and practically challenging. In this paper we investigated the impact of different restoration measures and the effect on the interaction of the temporal resolution of the groundwater modeling methodology. The interaction is analysed in the water bodies and wetlands in the valley of the Zwarte Beek, one of the most valuable nature reserves of Flanders. In the past, several changes have been made to the river and drainage system. These adaptations are now considered to be bottlenecks in maintaining a good ecological and hydrological status of its water dependent biotopes. Hence, in the context of the EU Water Framework Directive, it is necessary to (at least partly) restore the initial natural situation. The measures proposed include the reinstatement of old meanders and the removal of a weir. By removing the weir, fish migration is again possible. Reconnecting old meanders increases the habitat diversity. We used transient groundwater modeling to evaluate the impact on the groundwater system of the wetlands. Results indicate that a peat layer, present in most of the wetland, minimizes the effects of the restoration on the groundwater table. The largest changes are confined to the areas near the old meanders and the weir. Steady-state situations do not allow a calculation of average lowest and highest groundwater levels, which are essential for simulating ecological site conditions. Hence, transient simulations with 14 days time steps are required to detect a considerably greater range of groundwater fluctuation than indicated by the seasonal simulation. It is shown that the river restoration project thus resulted in an improvement of the structure of the watercourse rather than the rewetting of the valley. We concluded also that high resolution transient groundwater modeling is an essential step towards river restoration and ecohydrological predictions.

**KEYWORDS**: Transient groundwater modeling, channel re-meandering, ecohydrological predictions, conceptual model uncertainty

## 1. Introduction

Since about 90% of the natural floodplains of Europe's rivers have been reclaimed and now lack river dynamics, nature rehabilitation along rivers is of crucial importance for the restoration of their natural function. Flood protection, self-purification of surface water, groundwater recharge, species protection and migration are all involved in this process. It is now generally recognized that rivers form natural arteries in Europe but are also of economic importance and are recognizable cultural landscapes (Pedroli et al. 2002). Creation or restoration of wet grasslands by (re)wetting is however challenging due to the high dynamics of wetland plant communities and the need for substantially raised water levels and prolonged flooding to produce significant community changes (Toogood & Joyce, 2009). Tague et al. (2008) show that restoration can have a seasonal impact with statistically significant increases in streamflow during the summer recession period and decreased groundwater table depths across a wide range of streamflow conditions. Hammersmark et al. (2009) indicate that the observed range of water-table depths in meadows can be greater due to a higher temporal resolution of water-table measurements.

A higher temporal resolution should thus be used for improved design and objective setting in future restoration projects. In contrast, steady-state groundwater modeling is often used to study flow mechanisms and impact of anthropogenic developments on the groundwater system (e.g. Van Loon et al., 2009a & b). It is therefore imperative to take into account the effect of different temporal resolutions for evaluating partial restoration of the anthropogenic interference with a river system.

The goal of this paper is to investigate the impact of the different restoration measures and the influence of the temporal resolution of the groundwater modeling methodology on the simulated interaction with the groundwater system. Moreover, it is demonstrated how a

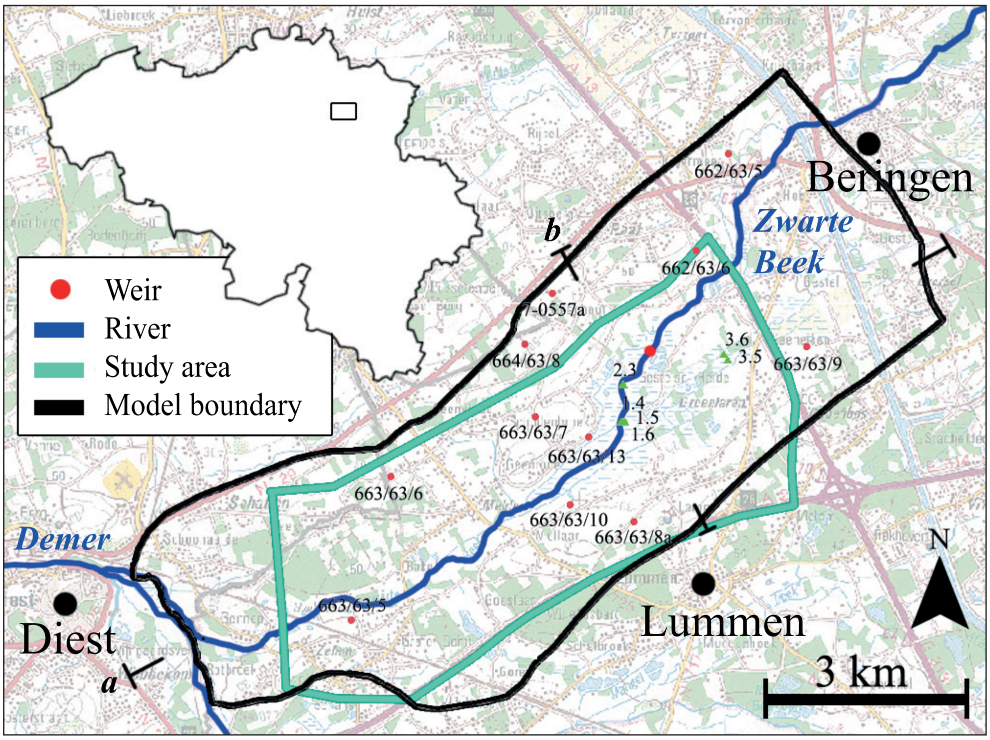


**Figure 1**. Study area, model boundaries, gauge well and geological profile locations.

seepage map and the mean lowest/highest groundwater level (respectively MLGL, MHGL) can be used for ecohydrological modelling. The latter parameters are originally defined for piezometer measurements, but can be estimated in a spatially continuous way by high resolution transient groundwater modeling.

## 2. Study area

In this study, we will focus on the valley of the Zwarte Beek (Fig. 1). This nature reserve is one of the most valuable valleys in Flanders and even in Western Europe. The importance of this area consists of the variety of rare vegetations and landscapes, and the presence of many endangered species. In the past, the stream has been adapted by the use of channeling and bank reinforcement. These measures often only displace problems and cause a reduction in habitat diversity and an impoverishment of the landscape. The European Water Framework Directive states that a good ecological status has to be achieved and maintained in each basin, or at least, if not feasible, that deterioration in status has to be prevented. It is therefore recommended to at least partly restore the initial natural situation of the valley of the Zwarte Beek. Moreover, the ecological potentials need to be investigated as the area is part of the European Network of Habitat and Bird Directives.

River restoration denotes all the measures that can be taken for restoring natural river systems, which have been degraded by human activity. In this case, one wishes to develop further the ecological potential by taking the following measures:

a) Excavate and reconnect old meanders (re-meandering)
b) Removal of a weir which acts as a fish migration bottleneck (1.5m difference in water level)
c) Use of an existing flood plain at high water levels

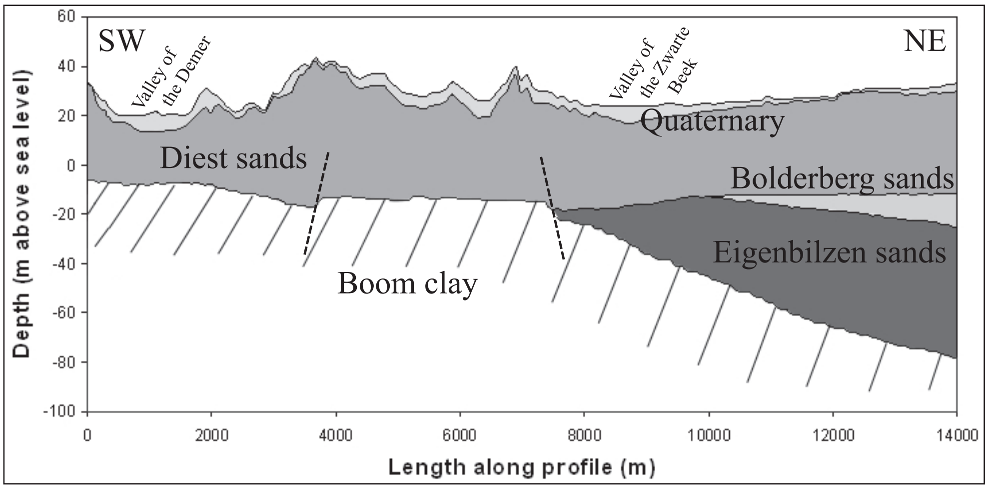


**Figure 2**. Geological profile (*a*) along the valley.

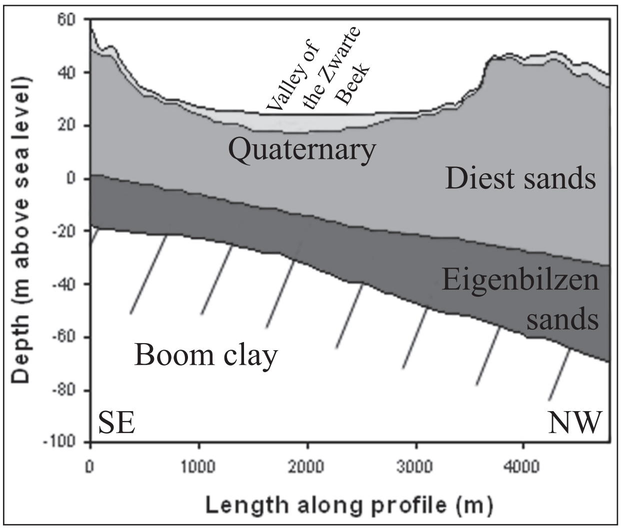


**Figure 3.** Geological profile (*b*) across the valley.

Prior to the implementation of these measures, an extensive ecohydrological study is required to assess the impact of such measures on the ecosystem. As Kasahara et al. (2009) stated, efforts to restore the vertical dimension of stream-groundwater exchange are rare but important. In addition, Staes et al. (2009) showed for an adjacent river basin that the river valley rewetting changed the saturated zone flow component, by coupling physically-based, fully distributed hydrological modelling with spatial analysis and wetland scenario generation techniques. Groundwater recharge was promoted in that case, decreasing the paved overland component of stream flow, and increasing the saturated zone flow component. Therefore, in this study fully distributed three-dimensional groundwater modeling was performed prior to ecohydrological modeling.

Concerning the hydrogeology of the study area, 4 hydrogeological units are considered based on the hydrogeological coding of the subsurface of Flanders (HCOV, Meyus et al.(2000)). Figs 2 & 3 show the geological profiles of the sections indicated in Fig. 1. A Quaternary cover is present in the entire area. However, this cover is very thin, and only plays a role for groundwater flow in the river valleys, where it is more developed, and the depth to groundwater is very small. The next unit is the Diest sands aquifer, which is of Miocene age. It consists of loose brown-green medium to coarse marine sands with high glauconite content. Both Bolderberg and Eigenbilzen sands are also of Miocene age. The Bolderberg sands consist of dark green, medium fine, slightly clayey glauconiferous sand, deposited under marine to continental conditions. The Eigenbilzen formation consists of dark green, glauconiferous, clayey fine sands with bioturbations. For an overview of the Belgian Neogene lithostratigraphic units, see Laga et al. (2001). The Boom Clay is of Oligocene age, and forms a regional aquitard. The vertical hydraulic conductivity values at the formation scale range from $3 \times 10^{-12}$ m/s to $1 \times 10^{-11}$ m/s in the locations investigated by Wemaere et al. (2008). This clay layer is about 50 m thick below the valley of the Zwarte Beek (DOV, 2009a), and can thus be considered impermeable. It occurs at depths ranging from 10 m in the south-west of the study area, to 140 m in the north-eastern part. More information on this unit can be found in Wemaere et al. (2008), and Vandenberghe et al. (2001).

## 3. Methods

The concept for the methodology is based on Shepley & Taylor (2003). A total of eight scenarios needed to be assessed: a steady-state average, summer and winter state, and a transient scenario for addressing the impact of temporal resolution, and each time, the existing situation was evaluated as well as the effect of the proposed measures. The existing situation results are then treated as the baseline prediction scenario, with which other scenarios can be compared.

### *3.1. Conceptual groundwater flow model*

Since the emphasis in this study lies on the characterisation of the impact of river restoration measures in the valley of the Zwarte Beek, some simplifications of the hydrogeological system can be made. First of all, it is sufficient to consider only the Zwarte Beek watershed from a certain distance upstream of the study area, until its confluence with the Demer (see Figs 1 & 4). The errors resulting from the consideration of this limited area will be minor in the valley when proper boundary conditions are set. The inflow of water to the system then occurs from the upstream part of the catchment, the recharge and the Zwarte Beek itself. All flow out of the system is considered to take place at the Demer, Zwarte Beek or any other draining elements like ditches or any wells present. In the vertical direction, all hydrogeological units that were described previously are considered. However, the Boom Clay serves as a boundary to the system because of its impermeability.

### *3.2. Numerical groundwater flow model*

The numerical code used in this study is MODFLOW (Harbaugh et al., 2000). GMS (EMS-I, 2009) was used as a pre- and post-processor. The model boundary conditions were determined from the results of the Central Campine System groundwater model (Meyus et al., 2004). The groundwater divides of the valley of the Zwarte Beek could be traced from particle tracking within this parent model, as shown in Fig. 4. There is only vertical flow beneath these lines, as is the case for the Demer River in the SW. They were implemented in the model as a Neumann boundary condition (Cheng & Cheng, 2005); in this case a specified flux equal to zero. In the NE, an equipotential line was traced for defining a fixed head Dirichlet boundary condition (Cheng & Cheng, 2005). These boundary conditions were applied to the entire vertical extent of the model. Because of the zero-flux across these boundaries, their steady-state character is assumed not to influence the groundwater heads in the center of the valley in the different temporal resolution scenarios. However, the fixed-head boundary in the NE,

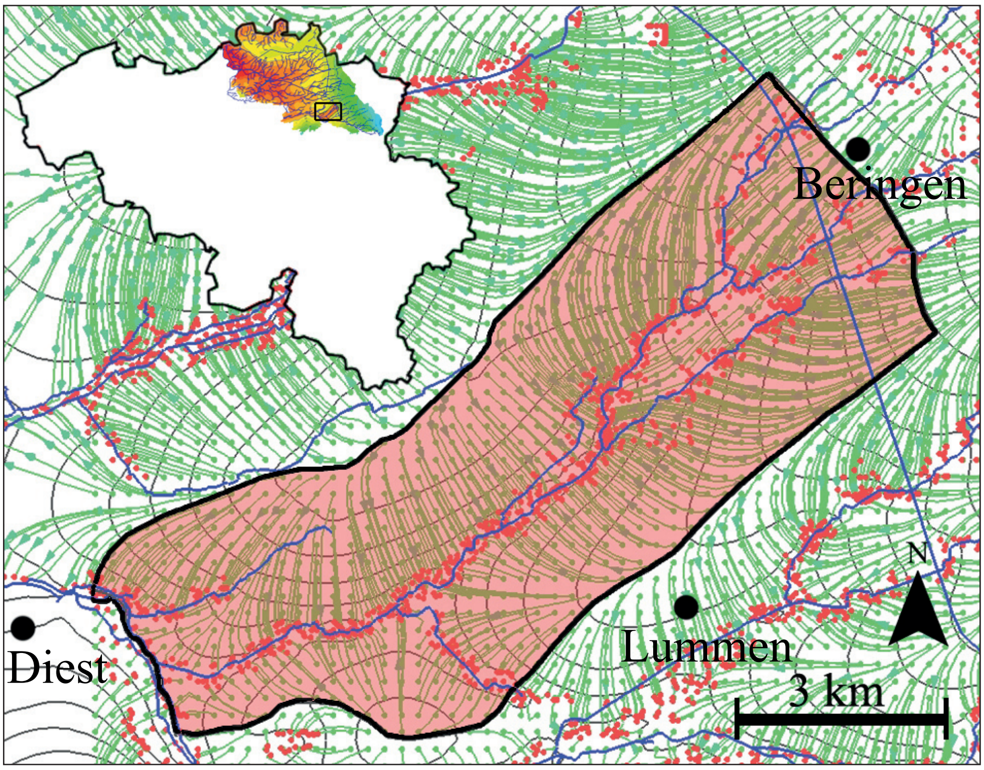


**Figure 4.** Derivation of the model boundary conditions from the parent model by particle tracking (green: starting points, red: end points).

derived from the yearly average steady-state parent model, is only used for the steady-state yearly average baseline prediction scenario. It is adapted for the winter and summer state according to time series fluctuations in the valley and interfluves, as the parent model was a yearly average steady-state model. Interpolation in time for the transient model is accomplished with a sine function between the winter and summer levels. The lower model boundary depends on the presence of the previously described impermeable Boom clay. The top of this unit can thus also be implemented as a Neumann boundary condition. The model comprises the following geological layers from bottom to top: the sands of Eigenbilzen, Bolderberg and Diest and the Quaternary layer. The grid cells have a size of 25 by 25 m and their height depends

| Stratigraphy | HCOV | 3-layer concept | 7-layer concept | |
|---|---|---|---|---|
| | | | | |
| Quaternary & Diest sands | 0100 & 0252 | Layer 1 | 1,1 | Groundwater table |
| | | | 1,2 | |
| | | | 1,3 | |
| Diest sands | 0252 | Layer 2 | 2,1 | |
| | | | 2,2 | |
| | | | 2,3 | |
| Bolderberg sands | 0253 | Layer 3 | 3 | |
| Eigenbilzen sands | 0256 | | | |
| Boom aquitard | 0300 | Lower boundary | | |

**Table 1.** Shematic representation of the model grid layers, HCOV coding and the geology for the main and the alternative conceptual model.

on the depth of the layers they represent (MODFLOWs Layer-Property-Flow package was used).

Table 1 demonstrates the division of the subsurface into 3 grid layers. The lowest one represents the sands of Eigenbilzen and Bolderberg. The bottom of the top layer corresponds to the bottom of the Quaternary or below. The second layer represents the remaining portion of the Diest sands. The upper layer only needs to be differentiated within the valley because the influence of the quaternary can be neglected outside of the valley where it is part of the unsaturated zone. The differentiation was made according to the soil types in the valley, as shown in Fig. 5. Following types were distinguished: peat, wet sand, wet loam and wet sandy loam.

The streams are all implemented using the MODFLOW Drain Package. The locations are derived from the Flemish Hydrograhical Atlas (MVG et al., 2000), a digital terrain model with horizontal resolution of 5 m, the topographical map and some terrain mapping. The water level is estimated and fixed at 1m below topography. Surface water modeling has however been conducted for the largest part of the Zwarte Beek and the Demer (Libost-Groep, 2008). The results provided the necessary information (i.e. head in the river and riverbed bottom elevation) to handle both rivers with the MODFLOW River Package. One data point is used for each 100 m of the transects. The river head is interpolated in time for the transient model with a sine function between the winter and summer values delivered by the surface water modeling. The river and drain conductances are fixed on respectively 5.0 and 2.5 (m²/d)/m. These parameters were observed to be not very sensitive in a preliminary calibration. Accordingly, the inverse procedure resulted in very large unrealistic values, hence they were fixed at

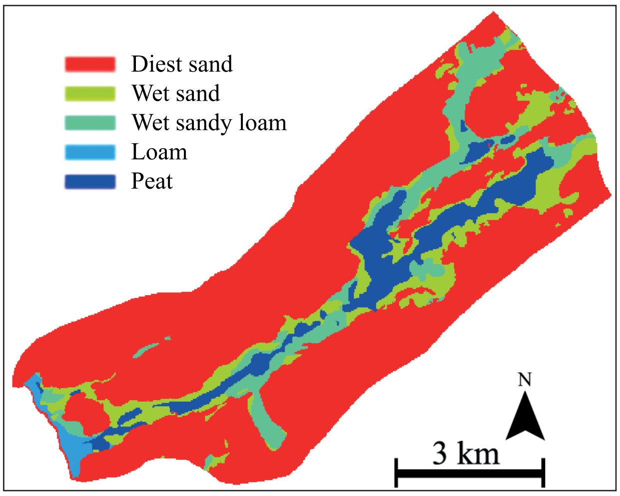


**Figure 5.** Differentiation of the upper grid layer according to soil types in the valley.

conductances, comparable to previous groundwater modeling of a similar catchment. The Albert channel is also implemented as a River because it can both act as a sink or source of water. Because of the engineered barriers however, a value of 0.01 m²/d/m was used for the channel conductance.

The groundwater recharge for the average steady-state model is implemented from the WetSpass data for Flanders (Batelaan & De Smedt, 2007). The winter and summer states are adapted with a factor according to the ratio between the average yearly flow rate in the Zwarte Beek and the winter and summer average. The transient data were calculated with SWAP (Van Dam, 2000) on the basis of precipitation data for the period 1996-2000 for different classes of average groundwater depth and soil type. This enables a net loss of water during certain periods when evapotranspiration exceeds rainwater infiltration, a phenomena that does not take place with the steady-state and seasonal temporal resolutions.

The remaining groundwater sinks in the model area are pumping wells and surface runoff. Only three wells are situated in the relevant geological layers within the model domain. Due to the small extraction rates and high hydraulic conductivity of the Diest sands, their influence is hardly visible. Surface runoff needs to be accounted for in the valley because of the marshy character. Groundwater levels are very high, and are often above the topographical surface. It is therefore necessary to specify a 2-dimensional DRAIN condition with a high conductance (10 (m²/d)/m²) at the ground surface. This ensures proper simulation of groundwater heads and fluxes in the valley.

| Model zone | Hydraulic conductivity (m/d) |
|---|---|
| Layer 1 - Peat | 0,01 |
| Layer 1 - Wet loam | 0,12 |
| Layer 1 - Wet sandy loam | 0,60 |
| Layer 1 - Wet sand | 2,98 |
| Layer 1 Diest sand & Layer 2 | 7,34 |
| Layer 3 | 1,03 |

**Table 2.** Calibrated hydraulic conductivity values for the different model zones.

The hydraulic conductivity was specified for each grid layer. The upper layer was further differentiated in the valley according to the soil types (Fig. 5). The conductivities were however coupled and treated as one variable to reduce the parameter space for calibration purposes. The proportions are 1, 10, 50, 250 for respectively peat, wet loam, wet sandy loam and wet sand. The sand of Diest in the first grid layer obviously gets the same value as the entire second layer. Vertical conductivity is taken to be 5 times smaller than the horizontal component. The values are refined by automatic calibration (PEST; Doherty, 2002), and are given in Table 2. The specific storage and effective porosity are fixed at respectively $10^{-5}$ $m^{-1}$ and 26% for the entire model domain.

The observations that were used in this study are shown in Fig. 1. In total 30 piezometers are located within the relevant strata of the model area (DOV, 2009b). A problem occurred however with 20 piezometers, which were divided over three sections perpendicular to the Zwarte Beek River. These piezometers were within 5 m distance from each other. Exact coordinates were not available for several of these piezometers because they could not be found anymore at the time of the field survey, and the measured levels were strongly dependent on the distance to the river. For instance a difference of 1 m was observed within a horizontal distance of only 10m. These strong gradients are due to a peat layer in the valley and its low hydraulic conductivity. A simple two-dimensional model confirmed that such gradients can be caused by this peat layer. Because several of these piezometers lie within one grid cell, the most representative value was chosen to address this scale-discrepancy. The piezometers furthest away from the river are the most adequate because they represent a bigger part of the grid cell than those near the river. When two wells coexisted at nearly the same location and depth, their weight for the automatic calibration was halved. By this selection procedure, only 6 of the 20 problematic piezometers were retained (indicated as triangles on Fig. 1), giving a total of 16 observation points. Another source of (soft) observational data are the marshy areas on the topographical map. The model results could repeatedly be compared with the maps so the flooded cells would more or less correspond with the indicated marshy areas.

### *3.3. Ecohydrological modeling*

We investigated the effects of changes in hydrology on vegetation using the ecohydrological model NICHE Vlaanderen (Callebaut et al., 2007). This model calculates the potentials for obtaining different groundwater depending vegetation types. It operates within a Geographical Information System (GIS) and takes into account the spatial scale and distribution patterns of a number of parameters. The NICHE Vlaanderen model consists of two main parts:

a) a series of decision rules describing the site conditions in terms of acidity, nutrient level, and moisture content;

b) a database with empirical relations between hydrological variables and presence of vegetation types.

The model input requires the following data:

a) soil type (divided in ecological categories as peat, sand, sand with high organic concentration, loam, peaty loam);

b) groundwater dynamics (defined as MLGL and MHGL in cm below soil surface);

c) seepage map (mm/day);

d) flood areas ( return period ≤ 5 years)

e) sources of nutrients e.g. atmospheric deposition, manure (kg N/ha/year);

f) management (no treatment, yearly mowing, 5-yearly mowing).

All information needs to be spatially distributed covering the whole study area because NICHE Vlaanderen uses grid cells as calculating units. As for the groundwater modeling, the calculation is done for the present state, which acts as baseline prediction scenario, as well as for the proposed measures. This makes it possible to evaluate the effects on the vegetation potentials (increase/decrease of potential locations).

## 4. Results

### *4.1. Steady-state yearly groundwater flow models*

The results for the steady-state yearly average model are presented in Fig. 6. The areas with blue triangles indicate cells where the groundwater level exceeds the surface level. This is the case in most of the valley area, indicating the marshy character of the nature reserve. This is caused by the presence of the peat layer, which is easily derived from a comparison between Figs 5 & 6. The groundwater level predictions (Fig. 7) are quite good, which is indicated by the low root mean squared error (RMSE = 0.21). The absolute differences between the observed and calculated values (Fig. 8) remain small (between 0 and 30 cm) in the valley as well as on the hillslopes.

The difference between the heads of the current and the restored state represents the impact of the river restoration on the system. The influence remains strongly focused around the course of the river. This is again due to the presence of the peat layer, whereby a strong hydraulic gradient buildup is possible. Some details of the difference map are given in Fig. 9. The effect of reconnecting an old meander results in local changes of the groundwater table of 20 to 50 cm. The removal of the weir, which currently has a height of 80 cm, has consequences of the same magnitude. The effect in the rest of the valley is restricted

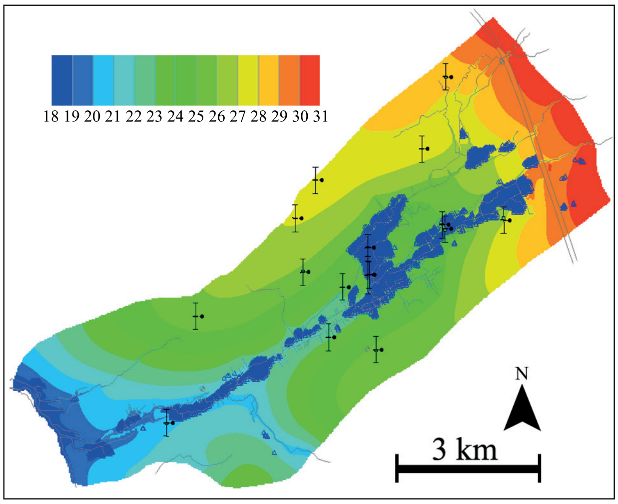


**Figure 6.** Steady-state results: Groundwater head (mTAW), for yearly average state.

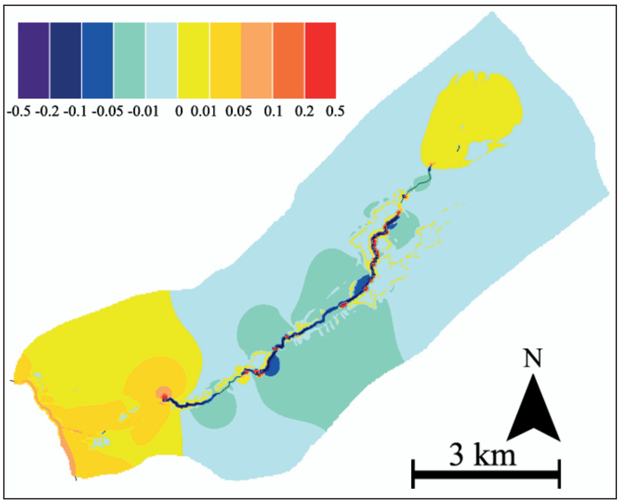


**Figure 8.** Head difference between current and restored state of the valley.

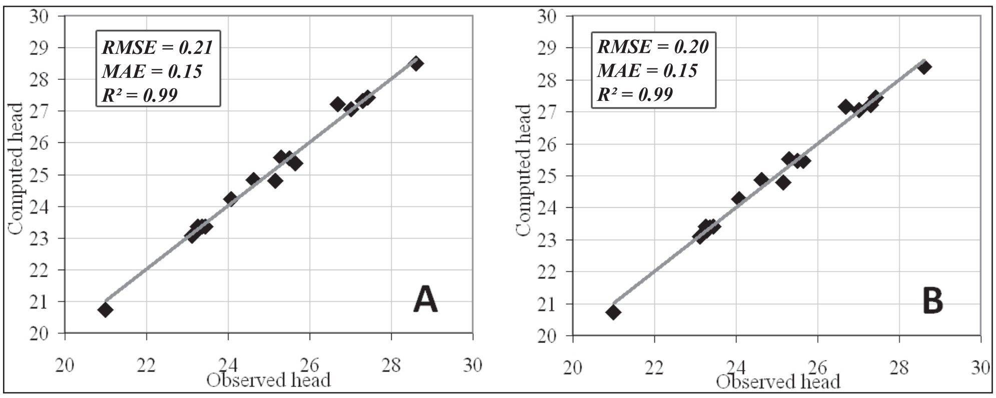


**Figure 7.** Simulated vs. observed values (mTAW) for the yearly average state. A: initial conceptual model. B: Alternative conceptual model.

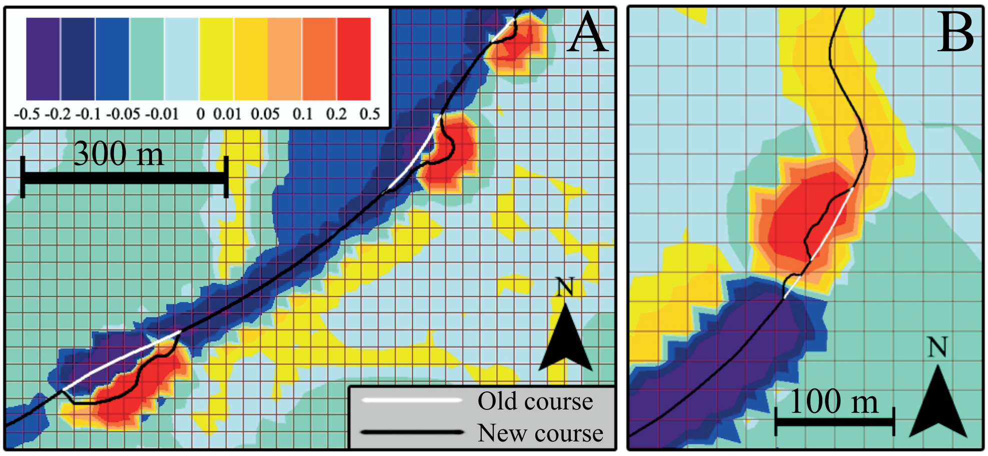


**Figure 9.** Head difference (m) between current and restored state. Left: detail of a newly joined meander. Right: location of the removed weir.

to between 0 and 5 cm. Overall, the upstream part gets a bit wetter, the downstream part a bit drier. In general, the seasonal modeling of the summer and winter states show the same patterns. However, the calculations show that the re-meandering causes a bigger change in the summer.

### *4.2. Transient groundwater flow model*

The transient calculations were done for the period 1996-2000. Stress periods were chosen to be 2 weeks, and the same timestep was used. Initial values were taken from the steady-state yearly average results. This might influence the first few weeks, however extreme values are needed to calculate the mean lowest and highest groundwater level, respectively MLGL and MHGL, hence the influence of the steady-state starting condition will be limited.

As an example, Fig. 10 shows all the observed and calculated heads for piezometer 1.6. It is important to realize that the calculated and observed data have very different timesteps. The use of a time step of two weeks in the model gives an average value, representing the entire two weeks. Head measurements occurred once every two weeks, which gives point samples that are not as representative, hence comparison should be done with care.

Another important feature is the difference in fluctuations between the steady-state and the transient results. The variation is bigger in the transient case but it remains a bit of an underestimation compared to the observations. Another remark is that the transient results are systematically lower than the steady-state results. This might have several causes. The steady-state models were calibrated with all available observations, while the first two years of the transient model (1996-1997) are relatively dry. Results for 1998 and 1999 are however also on the low side. Another cause is the use of two different sources of recharge data, a steady-state and a transient one. Thus the same time discrepancy also has an influence on the recharge.

The data needed for the ecohydrological model include MHGL and MLGL maps. These parameters are normally calculated by taking the average of 8 years of the 3 highest/lowest two-weekly observations each year. However, only 4 years could be simulated at the required temporal (~ 111 timesteps of two weeks) and spatial resolution (25x25 m, 3 grid layers, in total ~215000 grid cells) due to limited computational power (dual core, 2.4 Ghz, 3 Gb RAM). To be more specific, the used post-processor (GMS, see EMS-I (2009)) could not entirely read the MODFLOW cel centered flow output file from the transient case. The

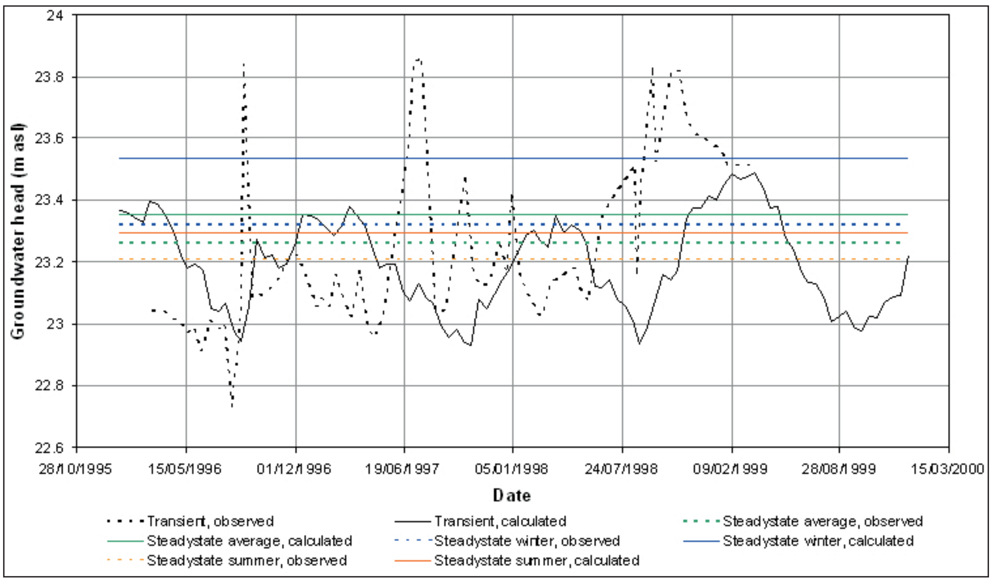


**Figure 10.** Results for GHG (cm below ground surface).

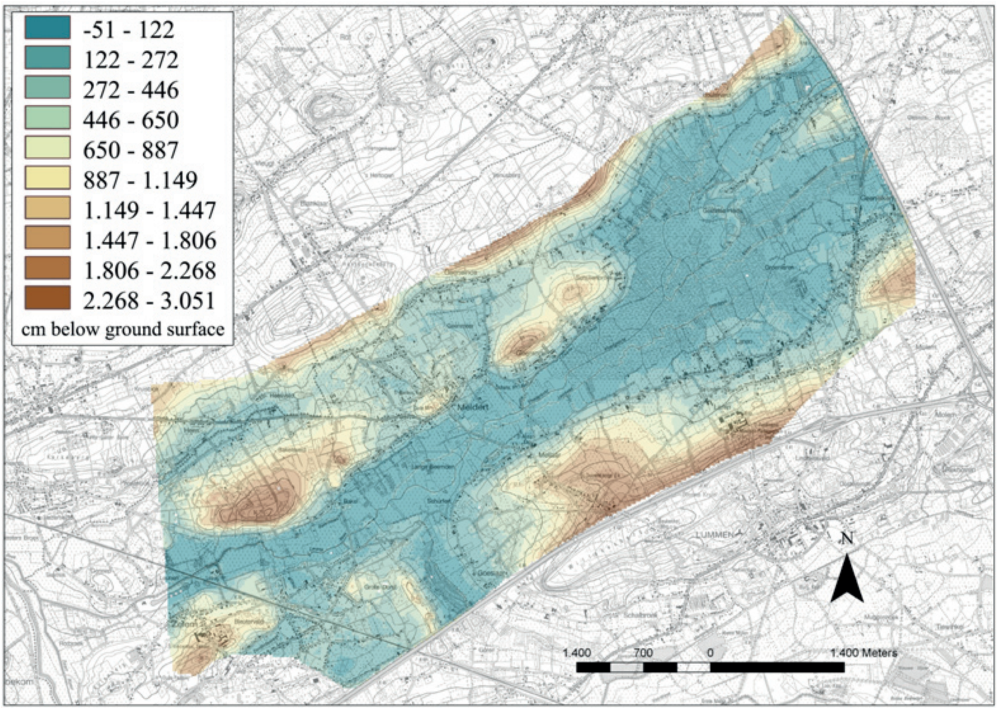


**Figure 11.** Time series syntheses of all the observed and calculated data from gauge well 1.6.

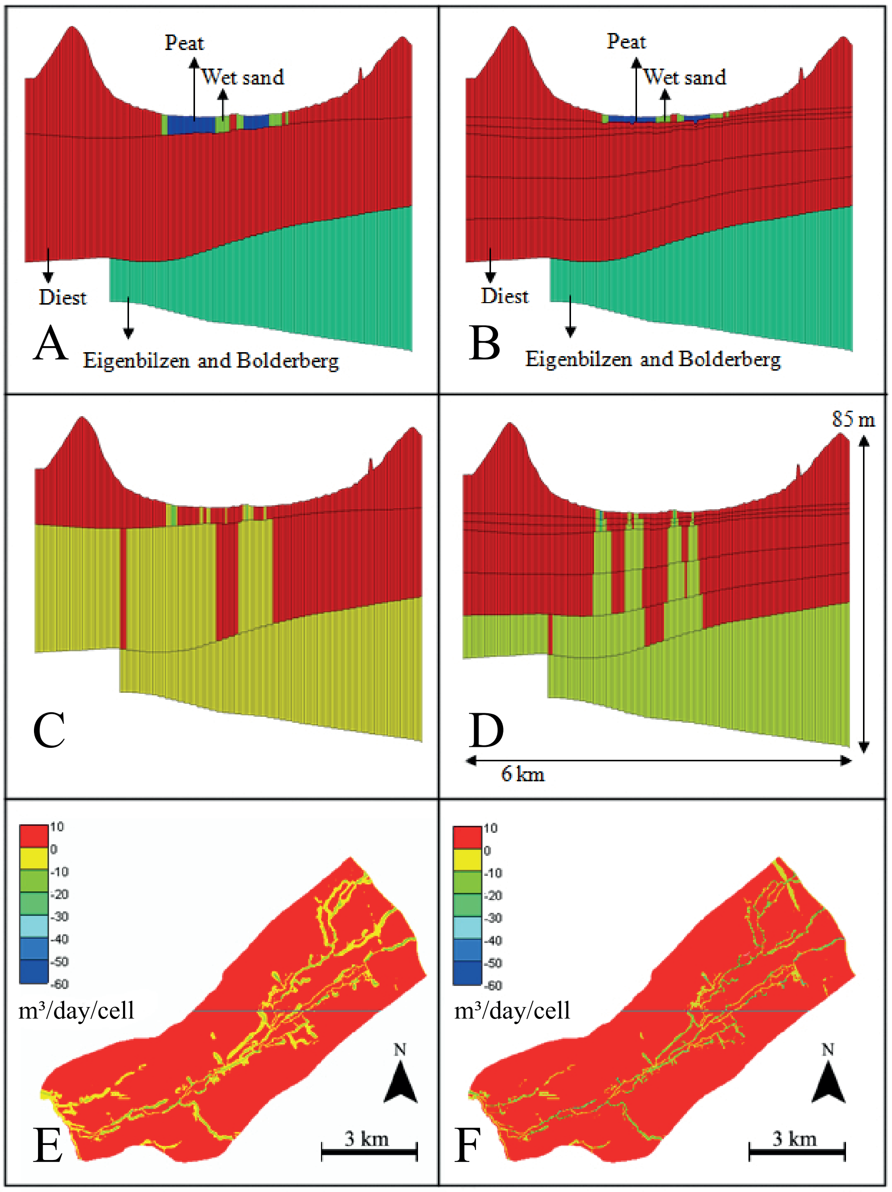


**Figure 12.** Left: 3-layer model. Right: 7-layer model. A & B: geology. C & D: flux through celbottom (m³/day/cell). E & F: plan view of the flux through the bottom of the upper grid layer (m³/day/cell).

calculated results thus do not correspond exactly to the MHGL and MLGL definitions. As an example, results for MHLG are given in Fig. 11.

### *4.3. Seepage and conceptual model uncertainty*

One of the required inputs for the ecohydrological model is a seepage map. While most of the system's parameters are clearly defined, seepage can be used in different ways for different purposes. In an ecohydrological context, it can be described as the flux of mineral-rich water, available to the plants under study. When trying to extract this information out of a groundwater model, one encounters several problems e.g. not all plant roots share the same depth; an upward flux is not necessarily rich in mineral content; plant water uptake influences the fluxes. The flux through the bottom of the first model layer was chosen as a measure for the seepage to avoid overcomplicated methods and because it is the change relative to the baseline prediction scenario that is important.

However, when looking to the results of this flux in Fig. 12, the areas with an upward flow seem to be quite narrow. This is again a consequence of the low hydraulic conductivity in the valley. The water flows along the path with the least resistance, resulting in the shortest pathway through the peat layer, which is vertical. However, to make sure that these narrow seepage areas were certainly not created by any numerical effects, an alternative conceptual model was tested. For this alternative model, the vertical discretisation was more refined (with 7 layers) as shown in Table 1 and the right section of Fig. 12. In comparison with the results from this alternative conceptual model, these small areas are in fact an overestimation of the seepage area. The difference is however not big, and though one might expect the more refined model to represent the system in a more realistic way, there is no direct evidence for this. After recalibration (all parameters stayed in the same magnitude except the Eigenbilzen and Bolderberg hydraulic conductivity, which changed about one magnitude) the RMSE value of the observed vs. calculated levels basically remained the same (0.20). Since the uncertainty introduced by the use of just one possible vertical discretisation seems to be limited, and in order to maintain a manageable model, further work was done with the original 3-layer model. The seepage map was finally calculated from the yearly average steady-state data, it is presented in Fig. 13.

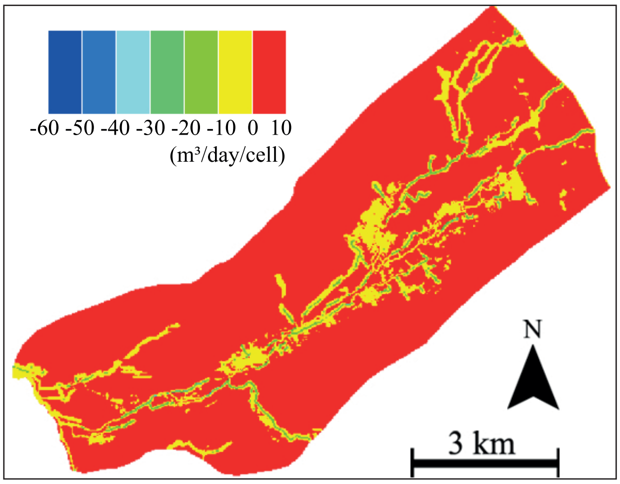


**Figure 13.** Seepage map (m³/day/cell)

### *4.4. Ecohydrological modeling*

Since the seepage area calculated from the groundwater fluxes in the groundwater model were very narrow, they did not match with the phreatophytes map, which indicated a wider zone of the valley. The seepage map resulting from the groundwater modeling was then adapted. Based on the difference between MHGL and MLGL, three seepage classes were defined, calibrated on the existing situation, and applied to the calculated scenario. A strong seepage was assumed for differences of 0-25 cm, seepage for 25-50 cm and no seepage for a difference larger than 50 cm.

The differences in groundwater levels between present state and restored scenario remain however limited (Figs

| Number of potential vegetation types per grid cell | Number of grid cells | | |
|---|---|---|---|
| | Present state | After river restoration | Difference |
| 1 | 4132 | 4201 | 69 |
| 2 | 3417 | 3425 | 8 |
| 3 | 1120 | 1124 | 4 |
| 4 | 295 | 309 | 14 |
| 5 | 281 | 266 | -15 |
| 6 | 58 | 58 | 0 |
| 7 | 2045 | 2052 | 7 |
| 8 | 41 | 28 | -13 |
| 9 | 126 | 128 | 2 |
| **Total number of grid cells** | **11515** | **11591** | **76** |

**Table 3**. Number of grid cells calculated by NICHE Vlaanderen for the present state and after river restoration, grouped by the number of potential vegetation types calculated per grid cell.

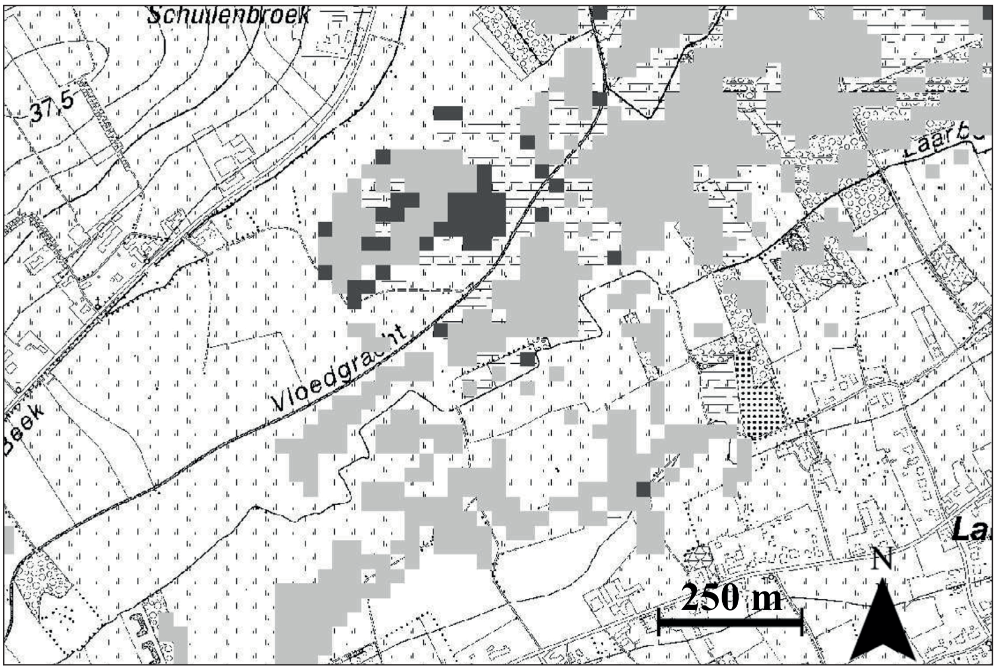


**Figure 14.** Potentials for Calthion palustris near the Vloedgracht in present state (grey) and after river restoration (dark grey).

8 & 9). Since this is one of the crucial inputs for the ecohydrological model, this limited impact is reflected in the results of the ecohydrological model as well. The increase of the number of grid cells for groundwater dependent vegetations is rather low as the total number of grid cells indicates in Table 3. For Calthion palustris (Calthion meadow) the potentials are increased near the Vloedgracht (a small brook) as illustrated in Fig. 14.

## 5. Discussion

Results of steady-state groundwater modeling indicate that a peat layer, which is present in most of the wetland, reduces the effects of the restoration on the average groundwater table position. Near the river, the annual average heads might change 20 to 50 cm. Restoring the old meanders has the effect of retaining more water in the upstream part. Downstream, the catchment gets a bit drier. In most of the area however, absolute average groundwater head changes remain smaller than 5 cm. Seepage areas are rather narrow which suggests that the marshy character mostly is a consequence of the peat retaining atmospheric water. An alternative conceptual model results in an even smaller region, it thus also supports this conclusion. The seasonal modeling results in the same relative changes as for the yearly average model. It does however provide an indication about the groundwater table fluctuation, but compared to the observations, it remains a large underestimation.

The transient modeling with time steps of 2 weeks permits the calculation of MLGL and MHGL values, which are essential for simulating ecological site conditions with the ecohydrological model NICHE Vlaanderen. The transient results show a considerable greater range of groundwater table fluctuations than the steady-state winter and summer states would predict. However, because recharge is averaged by the use of stress periods and time steps of 2 weeks, these results might still be an underestimation of the real MLGL and MHGL parameters as defined for a groundwater level time series. Comparing calculated versus observed heads shows that this is indeed the case. The temporal conceptual model thus proved to be a key issue for estimating groundwater table fluctuations. The use of high resolution transient groundwater modeling improved the estimations of groundwater level fluctuations. The calculated fluctuations however remain smaller than the observed fluctuations. The use of more densely sampled observations (e.g. daily time series) for at least a few piezometers, and the corresponding use of smaller time steps could yield better estimates of the extreme values in the future. Moreover, a longer time series would provide more information on the impact during hydrological extremes (wet and dry years).

Groundwater chemical analysis might be adequate to determine whether the seepage zones are indeed limited in areal extent. Hence, it is expected that a large part of the valley will show a strong rain water signature compared to the groundwater signature of the seepage zones. Since the occurrence of the peat layer is one of the key issues in this valley, a detailed three dimensional mapping of its extent and a characterization of its hydraulic conductivity might greatly enhance the validity of the model. The complexity of the hydraulic connection between a wetland and the underlying aquifer is also noted by Whiteman et al. (2010), who developed a procedure for the implementation of the European Water Framework Directive to determine whether a wetland is groundwater-dependent and damaged or at risk of damage as a result of groundwater quality or quantity pressures.

The influence of removing the weir in combination with the reconnection of the old meanders leads to a minimal impact on the average groundwater table and the derived MLGL and MHGL. Hence, the vegetation potentials in the valley are minimally influenced by the proposed measures. However, the proposed design meets the hydrological requirements. The fish migration bottleneck can be solved by removing the weir. The difference in water level at the weir of 1.5 m is completely resolved by reconnecting the meanders. This improvement of the morphological dynamics will be reflected in the ecological values of the water course. Management of the local hydrology and of the vegetation will very likely have a positive impact on the plant communities in the valley as well. The surplus value of this river restoration project thus is an improvement of the structure of the watercourse rather than the rewetting of the valley.

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