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Integrating Wetlands and Riparian Zones in Regional Hydrological Modeling

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Abstract: Wetlands, and in particular riparian wetlands, are at the interface between well drained land and the aquatic environment, where they control the exchange of water and related chemical fluxes from catchment areas to surface waters like lakes and streams. Integrating wetlands and riparian zones in regional hydrological modeling is challenging because of the complex interactions between soil water, groundwater and surface water. The model must be able to reproduce the special hydrologic processes in wetlands like groundwater dynamics, plant water and nutrient uptake, nutrient degradation and leaching to surface waters. An additional problem at the regional scale is the identification of riparian zones based on regionally available data.

The model used in this study is the eco-hydrological model SWIM (Soil and Water Integrated Model), in which a riparian zone and wetland module was incorporated. SWIM was chosen because it integrates the hydrological processes, vegetation, erosion and nutrient dynamics which are relevant at the watershed scale. The study shows simulation results of river discharge, groundwater dynamics and plant groundwater uptake and first results of simulated nutrient fluxes in wetlands.

Keywords: Riparian zones; wetlands; water quality; groundwater dynamics; nutrient retention

1 INTRODUCTION

The water framework directive of the European Commission demands to bring water bodies in Europe into “a good ecological status” (EC 2000). Many efforts and improvements have been done, mainly in the implementation of waste water treatment plants. But these measures only help to improve the water quality of point sources, whereas the main origin of some important contaminants are diffuse sources like atmospheric decomposition and fertilisation of crop land. Here, riparian zones and wetlands play an important role in the control of the water quality of surface water systems (Dall’O’ et al., 2001).

The paper presents an integrated catchment model with which it is possible to analyse the processes in wetlands and riparian zones in meso- to macroscale river basins, the scale relevant for water management planning and for the implementation of the water framework directive. A simple but comprehensive mechanistic wetland module was developed and coupled with the eco-hydrological model SWIM (Soil and Water Integrated Model, Krysanova et al., 1998), which integrates hydrological processes, vegetation, erosion and nutrient dynamics at the watershed scale. The reliability of the model results was tested under well defined boundary conditions by comparing the results with those from a two dimensional numeric groundwater model under steady-state and transient conditions (Hattermann et al., 2004b) as well as with observed data of a meso-scale basin, using contour maps of the long-term mean water table, observed groundwater level data and observed river discharge and nutrient concentrations.

The study area is located in the lowland part of the Elbe river basin, which is representative for semi-humid landscapes in Europe, where water availability during the summer season is the main limiting factor for plant growth and crop yields. The water and nutrient balance of the catchments is influenced by water and land use management like implementation of drainage systems, lowering of the drainage base and increased groundwater extraction. Large parts of the area have very shallow groundwater, and in particular here the water cycle is strongly influenced by water management practices like the installation of drainage systems for groundwater control (Freude, 2001, Landgraf, 2001, Bork et al., 1995).

The results of the study show that riparian zones and wetlands have a high potential to reduce the nutrient transport into surface water systems. Their impact is so large because they are at the interface between catchment and river systems, where the greater part of the nutrients in the catchment originally applied as fertilizers or mineralized from plant residues is already degraded. Restoration and management of wetlands is therefore of high priority for the control of non point source contamination of surface waters.

2 MATERIAL AND METHODS

2.1 THE MODEL

2.1.1 SWIM

The eco-hydrological watershed model SWIM integrates hydrological processes, vegetation, erosion and nutrient dynamics at the basin scale. A
three-level scheme of spatial disaggregation from basin to subbasins and to hydrotopes is used.

A hydrotope is a set of elementary units in the subbasin, which have the same geographical features like land use, soil type, and average water table depth. Therefore it can be assumed that they behave in a hydrologically uniform way (Krysanova et al., 2000). Water fluxes, plant growth and nitrogen dynamics are calculated for every hydrotope, where up to 60 vertical soil layers can be considered. The outputs from the hydrotopes are aggregated at the subbasin scale. Mean resistance time and potential retention of water and nutrient fluxes are calculated using spatial features of the hydrotopes like distance to next river, gradient of the groundwater table and permeability of the aquifer. The approach allows to consider and investigate the spatial pattern of land use and land use changes. The lateral fluxes are routed over the river network, taking transmission losses into account. Plant dynamics are simulated using a simplified EPIC approach (Williams et al., 1984). A full description of the model can be found in Krysanova et al. (1998, 2000). An extensive hydrological validation of the model in the Elbe basin including sensitivity and uncertainty analyses is described in Hattermann et al. (2004a).

2.2 THE WETLAND MODULE

Important for the investigation of meso- to macroscale river basins is to apply methods which are physically sound but simple enough to save computation time and data demand. The wetland module described here consists of two parts: one part describes the groundwater fluxes and water table dynamics, where the time scale is of days or weeks. The second part describes the nutrient fluxes and degradation, where the time scale is much larger (years and decades, sometimes centuries, because of the mean residence time of the groundwater).

Two cases have to be taken into account when calculating groundwater recharge: The first describes areas and time periods, where the groundwater table is relatively deep. SWIM uses an exponential delay function to calculate the effective groundwater recharge after drainage through the unsaturated geologic horizons from the last soil layer to the groundwater table (Arnold et al., 1993). The second case describes time periods with high recharge and areas with shallow groundwater, where the water table may rise and affect the lower soil zones. The soil is discretized in SWIM vertically into 5 cm layers. Layers \((i, i+1, \ldots)\) that are affected by groundwater are deactivated and the percolate from the layer \(i-1\) is defined as groundwater recharge. The layer is reactivated when the water table sinks.

Important for the hydrological processes and nutrient fluxes in wetlands is a good reproduction of the groundwater dynamics. Smedema & Rycroft (1983) derived a linear storage equation following the Dupuit-Forchheimer assumptions to predict the non-steady-state response of groundwater flow to periodic recharge from Hooghoudt's (1940) steady-state formula, assuming that the variation in return flow \(q\) in \(\text{mm d}^{-1}\) at time step \(t\) is linearly related to the rate of change in water table height \(h\) in m (only headlosses in horizontal direction are considered):

\[
\frac{dq}{dt} = 8\frac{T}{L^2} \frac{dh}{dt}
\]  

(1)

where \(T\) is the transmissivity in \(\text{m}^2 \text{d}^{-1}\) and \(L\) the slope length in m. If the groundwater body is recharged by deep soil percolation or another source (\(Rc\) in \(\text{mm d}^{-1}\)) and is depleted by drain discharge (\(q\)), it follows that the water table will rise when \(Rc-q > 0\) and fall when \(Rc-q < 0\). The water table fluctuations may be described as (Smedema & Rycroft 1983):

\[
\frac{dh}{dt} = \frac{(Rc-q)}{C*S}
\]  

(2)

\(S\) is again the specific yield. It follows that by assuming that the integration constant \(C = 0.8\):

\[
\frac{dq}{dt} = \frac{10*T}{S*L^2} (Rc-q) = \alpha* (Rc-q)
\]  

(3)

so that the change in drain discharge \(dq/dt\) is proportional to the excess recharge \(Rc-q\), with \(\alpha\) being the proportionality factor (reaction factor). Equation 2 can be transformed to gain the equation for return flow:

\[
q_t = q_{t-1} * \exp(-\alpha* \Delta t)
+ Rc_{\Delta t} * (1 - \exp(-\alpha* \Delta t))
\]  

(4)

Using the linear relationship between \(q\) and \(h\) (equation 1), we get:

\[
h_t = h_{t-1} * \exp(-\alpha* \Delta t))
+ \frac{Rc_{\Delta t}}{0.8*S*\alpha} * (1 - \exp(-\alpha* \Delta t))
\]  

(5)

The equations are scale independent and the spatial unit for which \(h\) and \(q\) are calculated can be either the hydrotope or the subbasin. In this study, the mean groundwater dynamics were calculated on the subbasin scale and the changes in height \((dh/dt)\) where then added to the mean water table \(\bar{h}\) of the hydrotopes \(U\) in the subbasins:

\[
\frac{dU}{dt} = \bar{h}(U) + \frac{dh}{dt}
\]  

(6)

taking into account the distance of the hydrotopes to the river, the slope length \(L\).

The factor \(q\) is a function of the transmissivity \(T\) and the slope length \(L\):
\alpha = \frac{10^9 T}{S * L^2} \tag{7}

Therefore, the reaction factor has a physical meaning, as illustrated by the comparison with the results of the numerically solved Boussinesq Equation (Hattermann et al., 2004b), where the same geo-hydrological parameters (T, L, S) were used. However, for meso- to macro-scale basins the basic geo-hydrological parameters, namely transmissivity and specific yield, are usually not available. Especially the specific yield is difficult to determine. Hattermann et al. (2004b) suggested another method to estimate the reaction factor \( \alpha \) from field observations: From Equation 5, it follows that in periods without recharge (\( RC = 0 \)):

\begin{equation}
\alpha = \frac{\ln h_{i-1} - \ln h_i}{\Delta t} \tag{8}
\end{equation}

Therefore, \( \alpha \) can be estimated directly by using observations of the groundwater head \( h \). This was done using an automatic calibration algorithm by adjusting \( T \) and \( S \) in physically sound limits. The inverse value of \( \alpha \) has the dimension of time and can be interpreted as the reaction time of the groundwater table and discharge to changes in recharge. It has a time scale of days to weeks.

While it is possible to describe water table dynamics using the mean reaction time, the time scales which have to be considered for the simulation of nutrient retention are much larger (years and decades), because the actual residence time is the crucial value which determines the intensity of degradation. According to Wendland et al. (1993), the degradation of nitrate \( N \) can be approximated by a linear decay equation, where \( \lambda \) is a function of temperature and available oxygen. The full retention of a landscape is then a function of mean residence time and degradation:

\begin{align*}
N^{\text{out}}_{s,i} & = N^{\text{in}}_{s,i} e^{-\Delta t / \beta_s} + N^{\text{in}}_{s,i} (1 - e^{-\Delta t / \beta_s}) \\
N^{\text{out}}_{i,j} & = N^{\text{in}}_{i,j} e^{-\Delta t / \beta_i} + N^{\text{in}}_{i,j} (1 - e^{-\Delta t / \beta_i}) \\
N^{\text{out}}_{b,s} & = N^{\text{in}}_{b,s} e^{-\Delta t / \beta_b} + N^{\text{in}}_{b,s} (1 - e^{-\Delta t / \beta_b}) \\
N^{\text{out}}_{b,i} & = N^{\text{in}}_{b,i} e^{-\Delta t / \beta_b} + N^{\text{in}}_{b,i} (1 - e^{-\Delta t / \beta_b})
\end{align*}

\tag{9, 10, 11}

where \( \beta \) is the mean residence time of water in a subbasin. Since SWIM distinguishes between surface flow \( s \), interflow \( i \) and base flow \( b \), each having different retention characteristics (residence time, oxygen content), there has to be one equation for each of the fluxes. The mean residence time of the water in the subbasin from hydrotrope to river \( (\chi) \) in \( \text{m s}^{-1} \) is calculated using the seepage velocity \( v_i \) \( \text{[m s}^{-1} \)\), where \( k \) in \( \text{m s}^{-1} \) is the hydraulic conductivity of layer \( z \), \( J \) the hydraulic gradient, \( dz \) in \( \text{m} \) the distance and \( n \) the number of layers:

\begin{equation}
v_i(z) = -\frac{k \cdot J(z)}{S} \tag{12}
\end{equation}

\begin{equation}
\chi = \sum_{i=1}^{n} \frac{dz_i}{v_i(z)} \tag{13}
\end{equation}

Plant uptake of water and nutrients from groundwater is only possible in times when the plant roots have excess to it and if the plant demand cannot be satisfied by soil water and nutrient recourses. A resistance function controls the ability of plant roots for water and nutrient uptake from groundwater.

### 2.3 THE BASIN

The northern lowland part of the German Elbe basin, where the model was tested in the Nuthe catchment (1998 km², see Figure 1), is climatically one of the driest regions in Germany, with mean annual precipitation of about 600 mm per year. Hence, water availability during the summer season is the limiting factor for plant growth. The lowland is formed by mostly sandy glacial sediments and drained by slowly flowing streams with broad river valleys. The upper sites with deep water tables are covered by sandy, highly permeable soils and mostly pine forests or by arable land on ground moraine with till soils that tend to have layers with lower water permeability. Valleys are covered by loamy alluvial soils with grassland and riparian forests, where the groundwater is very shallow, and arable land elsewhere. During the last two decades, decreasing water levels in rivers and groundwater have been observed (Landesumweltamt Brandenburg 2000a & 2002b). The main mean climatic and hydrologic characteristics of the study area are listed in Table 1.

![Figure 1: The location of the Nuthe basin and the observation points.](Image)
All necessary spatial information to derive the subbasin and hydrotope structure of the basins, the digital elevation model (DEM), the soil map of the State Brandenburg, the geo-hydrological map, the land use map and water table contour maps were stored on a grid format with 50 m resolution. The Nuthe basin was subdivided into 122 subbasins based on the DEM and the drainage network.

Table 1: Long term mean annual precipitation (P), mean annual temperature (T) and river discharge (Q) of the basin under study.

<table>
<thead>
<tr>
<th>basin</th>
<th>area [km²]</th>
<th>P [mm a⁻¹]</th>
<th>T [°C]</th>
<th>Q [m³ s⁻¹]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nuthe</td>
<td>1938.0</td>
<td>590.5</td>
<td>8.8</td>
<td>9.06</td>
</tr>
</tbody>
</table>

3. RESULTS AND DISCUSSION
3.1 GROUNDWATER AND RIVER FLOW DYNAMICS

First, the simulated mean annual water table depth of all subbasins in the Nuthe basins were calibrated automatically using the transmissivity in a physically sound range. The mean simulated amplitude was too high and had to be smoothed by a moderate increase in the value of specific yield (as taken from the geo-hydrological map). The Mean Absolute Error of the long term mean observed against the mean simulated water table in all subbasins was 0.026 m. The groundwater reaction factors of the subbasins had values between 0.1 (loamy sediments) and 0.3 (sandy / loamy sediments). The time dynamics of the simulated water tables in terms of rising and retention periods were not calibrated.

Figure 2 shows a comparison of five observed groundwater table hydrographs from the Nuthe basin with those simulated. The observation wells were selected in order to represent a cross section through the basin from the lowlands in the north to the hilly area in the south. Well 1 is located next to the outlet of the Nuthe river catchment. The curves show a good fit, especially for the early 1980s. The rise of the groundwater level in 1987 and 1988 is slightly overestimated by the model in subbasins 2, 4 and 5. As explained in section 1, the natural flow regime in the Nuthe basin is influenced by stream flow control (weir and reservoir management), and especially in the lowland areas the water level is controlled by land drainage. The simulated groundwater hydrographs are very similar, whereas the observations show more differences. The higher variability in the observed water levels is the result of small-scale heterogeneities in the aquifer and of precipitation events which are missing in the observed records. An even better fit would be possible by implementing additional management information. However, this was not the objective of the study. On the contrary, the study aimed at showing that a simplified model approach yields satisfactory results using commonly available data.

Figure 3 illustrates the impact of plant water uptake on the simulated water table. While the groundwater tables simulated with and without plant water uptake converge during the winter term, they separate during the vegetation period, where the plant uptake leads to a decline of the groundwater table.

The mean long term difference between the observed and simulated river discharge at the basin outlet is 3.0% for the calibration period 1981 - 1988, indicating that the water balance is correctly calculated by SWIM. The daily Nash & Sutcliff efficiency is 0.7 (only 0.54 for the validation period 1989-2000). The hydraulic regime of the Nuthe
basin is strongly influenced by water management regulations like drainage systems and weir plants, so that it is difficult to reproduce the hydrograph with higher accuracy. The summer discharge is in some years overestimated by the model (see Figure 4). This can be explained by water abstraction and regulation measures, when a minimum river flow is provided by reservoir management in dry summer periods. It is worth mentioning that the efficiency was notably higher for other meso- and macro-scale subbasins of the Elbe located in hilly and mountainous areas (Hattermann et al., 2004a).

Figure 4: Comparison of daily river flow observed and simulated (gauge Babelsberg).

Without additional plant water uptake from groundwater, the total evapotranspiration would be 24% lower, leading to an increase in river discharge of about 77%.

3.2 NITRATE CONCENTRATIONS

The nitrate concentration in the Nuthe river during the eighties was strongly influenced by point sources (irrigation of waste waters in very small areas, municipal waste waters, even direct discharge of liquid manure into surface waters), where the records are vague and incomplete, so that the comparison in Figure 5 is done for a time period in the ninetieth, where impact of point sources is very limited because of the implementation of waste water treatment plants in the basin.

Figure 5: Simulated and observed nitrate concentrations in the Nuthe river.

Diffuse sources in this study are fertilizer applications (about 180 kg/ha for winter wheat), atmospheric decompositions (about 40 kg/ha), and plant decompositions after harvest and fall. The comparison shows that the periodicity and amplitude of the observed values is mostly well reproduced by SWIM, although the difference between observed and simulated values is large especially at the end of the year. The reason is that the diffuse sources for nitrate contamination (in particular fertilization) are not very well known, because information about crop rotation schemes and fertilization regimes are not available at the regional scale. In addition, the flow regulation by dams and weirs and the drainage systems influence of course not only river discharge but also nutrient fluxes. The mean residence time of groundwater is 41 years, with a maximum of approximately 400 years. The values are in good agreement with Landesumwelamt (2002b), who estimated the nutrient loads and retention in the lowland catchments of the Elbe basin.

Figure 6: Comparison of simulated and observed nitrate concentration with and without plant water uptake from groundwater.

Figure 6 illustrates the impact of plant uptake of nitrate in riparian zones and wetlands. As shown also for the impacts of plants on the water level in Figure 3, the differences are the highest during the summer season, when plant demand is high and can therefore not be satisfied by the soil water concentrations. The difference becomes smaller during the late summer, because the total amount of available nutrients in soils and hence also the leaching of nutrients has its minimum.

Figure 7: Additional nitrate uptake by plants in riparian zones and wetlands.
Figure 7 shows a map of the additional plant nitrate uptake from groundwater in kg/ha. The values are not so large in comparison with the total plant uptake (up to 180 kg/ha). The additional uptake is only about 6% of the total uptake, but this leads to a retention of about 35.5% of the total river load. The reason is that the additional uptake happens in an area next to the surface water bodies, where the largest part of the nitrate which was originally applied by fertilizers, mineralised from plant residues and decomposed from the atmosphere is already degraded.

4 CONCLUSIONS

The simulation results indicate that relatively small parts of the total catchment area have a relatively high impact on the water and nutrient balance in the catchment (additional evapotranspiration of about 24%, additional nitrate uptake of about 6%, leading to a decrease in river discharge of about 77% and to an increase in annual river nitrate load of about 35%). Riparian zones and wetlands are buffer systems able to reduce contamination of surface waters, as long as the vegetation has access to groundwater. On the other hand, restoration of wetlands will lead to increased water losses by evapotranspiration, crucial in a region where river discharge during the summer season is only possible by water regulation through dams and weirs, and where a trend to lower annual precipitation has been observed during the last decades. It follows that water managers have to find a sensitive balance between water quality and water quantity aspects in the planning process.

5 ACKNOWLEDGEMENTS

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