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Flood induced infiltration affecting a bank filtrate well at the River Enns, Austria

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Abstract

Bank filtration employs a natural filtration process of surface water on its flow path from the river to the well. The development of a stable filter layer is of major importance to the quality of the delivered water. Flooding is expected to destabilise the riverbed, to reduce the filter efficiency of the bank and therefore to endanger the operation of water supply facilities near the riverbank. This paper provides an example of how bank storage in an unconfined alluvial aquifer causes a significant decrease of the seepage rate after a high-water event. Extensive monitoring equipment has been installed in the river bank of the oligotrophic alpine River Enns focusing on the first metre of the flow path. Head losses measured by multilevel probes throughout a year characterise the development of the hydraulic conductivity of different riverbed layers. Concentration profiles of nitrate, total ions and a NaCl tracer have been used to study infiltration rates of river water and its dilution with groundwater. Dynamic modelling was applied in order to investigate the propagation of flood induced head elevation and transport of pollutants.

Keywords: Bank filtration; Bank storage; Clogging; Stream–aquifer interaction; Alluvial aquifers; Groundwater modelling

1. Introduction

The structure of a riverbed is determined by geological conditions, by suspended sediment concentration and particle size distribution, by the flow velocity of the river (Cunningham et al., 1987) and, especially in the case of bank filtration, the infiltration velocity. Sedimentation of suspended solids is induced by gravity and forms a fine particle layer with reduced hydraulic conductivity at the stream–aquifer interface. An additional clogging mechanism is caused by a mass flux from the river into the aquifer. Fine sediments intrude into the interstitial spaces, and get strained within the pores (Boulton, 2000; Schälchli, 1992). Usually both clogging mechanisms (gravity driven sedimentation and particle intrusion by infiltration) superpose each other and create a more dense layer through which the filtrate has to pass. Not only physical filtration occurs in the upper sediment layer but also very active biological degradation processes (Brunke and Gonser, 1997; Boulton et al., 1998).

Stream stage fluctuations can serve as natural stress tests of aquifers. In the case of a complex hydrogeologic situation, statistical methods applied to a long data record are required for aquifer characterisation (Nevulis et al., 1989), whereas alluvial and glacial aquifers commonly allow an easier

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identification and calculation of river–aquifer interactions. Vekerdy and Meijerink (1998) applied one-dimensional groundwater flow equations referring to Edelmann (1947) in order to calculate the propagation of a flood induced groundwater rise in an alluvial aquifer adjacent to the Danube in Hungary. Calculated hydraulic heads showed an overestimation of the measured rise. One detected reason for this deviation was the fact that riverbed conductance was neglected in the calculations. The analysis indicates the important role of the riverbed in dampening the propagation of flood waves entering the aquifer. Conrad and Beljin (1996) specified the criterion that streambed effects cannot be neglected if the streambed conductivity is more than one order of magnitude lower than the aquifer hydraulic conductivity. This criterion was derived from comparative studies between numerical models and an analytical two-dimensional induced infiltration model (Wilson, 1993).

Investigations at the River Rur (Bucher and Denneborg, 1993) and the River Modau (Mueller and Ebhardt, 1998) in Germany demonstrated a significant increase of river leakage during floods especially of rivers with extremely clogged bottoms due to higher bank infiltration rates.

The development of the riverbed as an enduring dynamic process is determined by the boundary conditions. Abrupt changes of boundary conditions (e.g. due to the construction of reservoirs) initiate an unstable phase with major alterations of the riverbed structure. Gutknecht et al. (1998) and Sengschmitt et al. (1999) reported their detailed studies of the Danube riverbed at the reservoir of a recently built hydro-power plant near Vienna following the first impoundment. The development of riverbed layers followed a cyclic pattern. During floods the firm upper sediment layer was partly removed and afterwards overlain by loose sediments which consolidated during the subsequent base flow period.

The motivation of this research was to investigate the filter efficiency of the river bank focusing on the first metre of the flow path. The major hydraulic aspect of this paper concerns the measurement of the build up of an efficient filter layer and the question of its stability throughout a year. An experimental bank filtration well at a reservoir on the River Enns in Upper Austria was considered. The reservoir was constructed during the 1960s such that there has been considerable time to attain a steady state stream bed structure. By beginning to pump the well, the flow situation was changed and clogging of the stream–aquifer interface was initiated. Previously, Younger et al. (1993) and Doussan et al. (1994) pointed out the significance of the clogged streambed for the filtration of pollutants. High-water represents a disturbance of the balance between the riverbed and the average flow and transport conditions. The increase of the shear stress with increasing discharge is expected to destabilise the filter layer (Macheleidt et al., 2000). Further consequences of this would be intensified seepage of river water and reduced migration time of pollutants.

2. Site description and methods

The River Enns drains an average annual amount of 6.6 million m^3 water from an area of 6.080 km^2 to the Danube. The 254 km long river crosses three main geological zones, from the limestone mountains in the south to the flysch zone upstream of the city of Steyr, where the investigated filtration site is situated, to the foothills of the Alps. During the last two glacial periods a huge amount of gravel was transported from the Alps and formed terraces on both sides of the river. In the area of the considered bank filtration site the river has completely cut through the gravel layer and the impermeable flysch layer forms the river bottom (Hasenleithner et al., 1999).

The Ennskraft, an Austrian power supply enterprise which is in charge of the research project reported here, operates 10 hydropower stations along the River Enns. The investigated bank filtration well is situated about 50 m from the river bank, 750 m downstream of a hydro power plant (HPP Rosenau) and at the beginning of the 5 km long reservoir of the next power plant (HPP Garsten). No measures had been taken to seal the bank of the reservoir in the area of the study site. At this particular location the river stage is determined by the dam at a level of 302.0 m a.s.l. and shows variations in a range up to 302.4 m. During the measurement period of one year from October 1997 to October 1998 the discharge of the River Enns varied from about 70 to 540 m^3/s (Fig. 1). Only when discharge exceeded 500 m^3/s did the river stage rise significantly, reaching the annual high-water level of 302.8 m a.s.l. at a discharge of 540 m^3/s in March 1998. Since the riverbed is cut into the dense flysch zone the river water infiltrates almost exclusively through the bank and not the bottom. Between river and well the aquifer thickness is about 5 m and the total thickness of the gravel layer is 15 m.

The river water quality is relatively good compared to that of other European rivers of this size. All parameters with the exception of bacteria are within the range of the quality standards of surface water for use as drinking water recommended by the EU (75/440/EWG). Organic loading is very low and the concentration of dissolved organic carbon varies between 1 and 2 mg/l (Fig. 1). The river water is saturated with oxygen (>10 mg/l), the nitrate concentration is low (<5 mg/l) and chemical parameters like hardness (about 10 mg/l magnesium and 50 mg/l calcium) reflect the geological situation upstream. About 400,000 m^3 of sediment per year settles in the Enns reservoirs and a measured suspended solids concentration of only 8 mg/l on average passes the reservoirs. The concentration of suspended solids and discharge show hardly any correspondence with the exception of significant high-water events. At a river stage of 302.0 m a.s.l., the measured cross-section area of the river is 280 m^2 and the minimum discharge of 70 m^3/s results in a mean flow velocity of 0.25 m/s. The mean flow velocity fluctuates between this value and 1.5 m/s at a discharge of 540 m^3/s and a cross-sectional area of 350 m^2.

The groundwater quality reflects the trends of land use and intensity of agriculture along the river. The further downstream the river is, higher are nitrate and pesticides concentrations in the groundwater. In the region of the filtration site the groundwater shows nitrate concentrations near 50 mg/l while north of Steyr values reach about 100 mg/l. The enrichment of the groundwater by filtrate from the river offers a solution in order to obtain nitrate concentrations in agreement with drinking water standards.

The monitoring infrastructure involves two adjoining measurement stations installed in the riverbed and bank, respectively (Hasenleithner et al., 1999). Station 1 consists of six perforated plexiglass bores to extract sediment specimens for microbiological analysis and is presented elsewhere (Brugger et al., 2001a,b). Monitoring station 2 is a high-grade steel box with windows for the visual observation and video recording of riverbed clogging. The box, which has a flood-safe entry on the top, was installed about 1.5 m below the average river stage. One set of five probes was installed at the upstream side of the box in natural sediment and two sets of five probes (MLD) downstream in specified filter sand (0–4 mm) in order to measure hydraulic heads and to obtain water samples. These probes are arranged at depths of 0.15, 0.30, 0.50, 0.70 and 0.90 m beneath riverbed surface. Two further probes were installed between the river and the well (PLE 6 m and PLF 18 m off the river) and one probe (KHB01) 7 m on the landward side of the well (KHB02) (Fig. 2).

Water levels in the probes at station 2 correspond with the levels of graduated glass pipes within the steel box. Monitoring was carried out weekly and water samples were taken via a valve. Additionally, hydraulic heads and electrical conductivity of the bank filtration well and the river have been monitored on-line. A sodium-chloride tracer test was conducted in order to investigate the migration time from the river to the well.

3. Discussion of measurement results

3.1. Seepage and riverbed clogging

Fig. 3 shows the data from the multilevel probe MLD in the riverbed. Before the well was put into operation the levels of ground- and surface-water had been balanced and no in- or exfiltration could be observed (compare Fig. 3: no head losses between 8 and 14 October 1997). Then the pump was switched on delivering a constant 20 l/s. The relevant head losses developed at the surface of the riverbed, in the first 15 cm, respectively (i.e. the water level difference in the probes represented by the vertical distance between profile Enns and profile MLD1 in Fig. 3). Obviously the riverbed clogged in this zone which led to a major loss of the potential for infiltration flow. Hence the seepage rate reduces and as a further consequence the head losses in the deeper zones decrease (Fig. 3). The continuous development of increasing water level differences at the surface and reduced differences in deeper zones was interrupted by high-water.

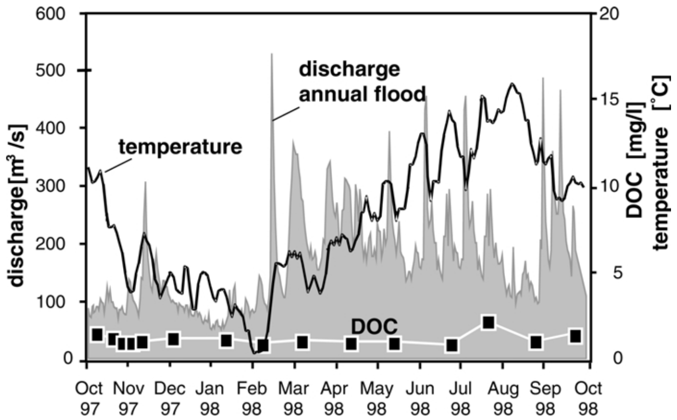

Fig. 1. Temperature, discharge and dissolved organic carbon profiles of the River Enns.

The most significant flood during the observation period occurred in March 1998 which is represented by the peak of the Enns water level in Fig. 3. During the flood the water level difference between the Enns and probe MLD5 increased from 0.2 to 0.3 m. Under the assumption of a constant hydraulic conductivity the infiltration rate increased by 50%. Reduced water level differences after the high-water indicate an abrupt decrease of the potential losses. At first inspection, the measured profiles imply that during the flood the hydraulic conductivity of the surface layer of the riverbed was reset to the state at the

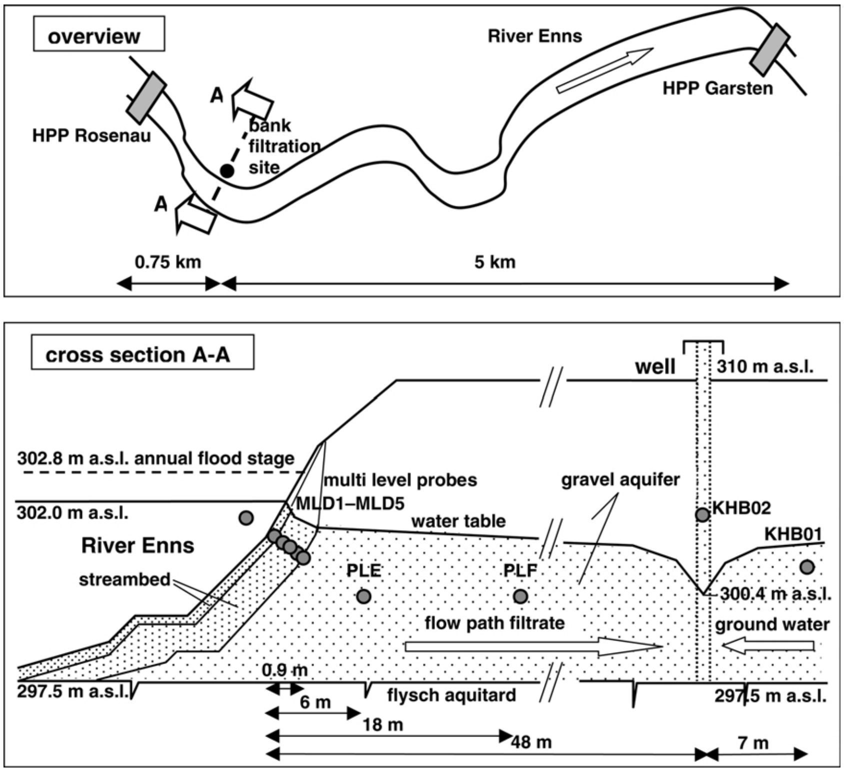

Fig. 2. Schematic overview and cross-section of the bank filtration site on the River Enns.

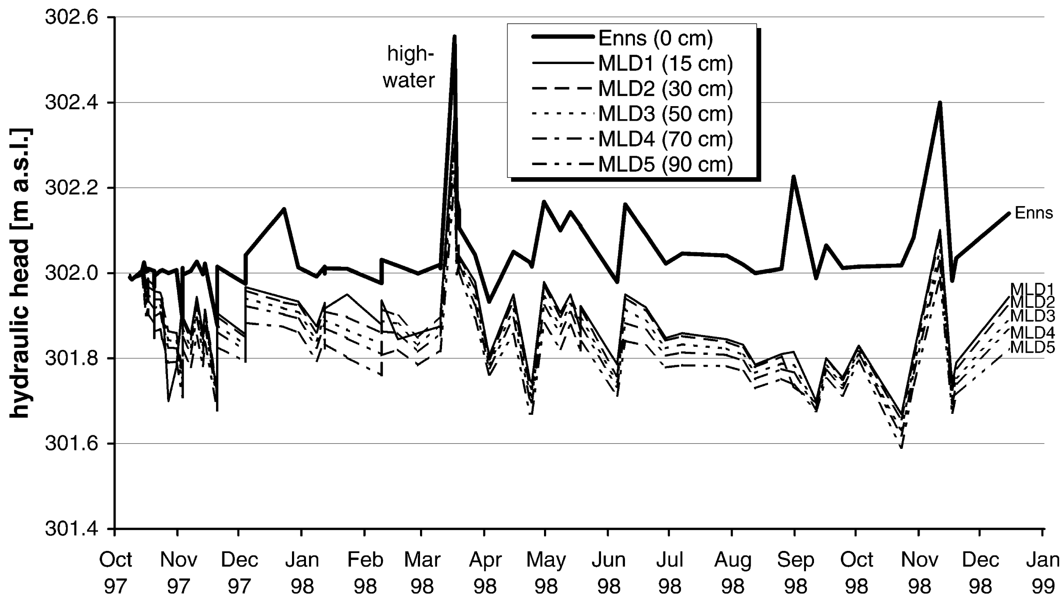

Fig. 3. Hydraulic head profiles at five different zones in the riverbed (weekly measurement).

beginning of the operation period. In fact, the dense surface layer was not denuded but the infiltration velocity was reduced. This fact is confirmed by the observation that the water level differences of the 90 cm deep zone also decreased: a zone that is hardly affected by particle intrusion. Consequently the hypotheses is stated that the measured head losses vary due to two different influences: under average discharge conditions water level differences increase because of riverbed clogging (decreasing kf in Eq. (1)). After flooding the water level differences are reduced at rather constant hydraulic conductivity because of a lower seepage rate v.

$$\frac{dH}{ds} = \frac{v}{kf} \tag{1}$$

where dH is the water level difference in m; ds, distance in m; v, Darcy flow velocity in m/s; and kf, hydraulic conductivity in m/s.

3.2. Investigating the seepage rate based on electrical conductivity measurements

The groundwater table rises immediately with the stream stage elevation. The correspondence of the subsurface water level with the surface water level is based on pressure propagation which is induced by infiltration and transmitted by the filled and released water volume of aquifer storativity (as shown in Section 4.2, a propagating concentration peak takes more than 4 days to reach the well while the propagating table elevation takes only 1 day). The water level profiles in Fig. 4 confirm the quick response of the groundwater table on the rise of the stream stage within about 1 day. In comparison with the unsteady descent of the stream stage the decrease of the groundwater table shows a smooth and delayed response. This phenomenon is referred to as delayed yield (Bear, 1987). The delay is the reason for a reduced hydraulic gradient between the river and well and consequently for a reduced seepage rate after high-water. The drop of the water level in the well before the flood and especially the fact that it occurred before the measured decrease of the river stage on 17 March could not be explained. The fact that the well water level had already risen before the flood and that the electrical conductivity of the well water increased during the flood is explained by preceding precipitation recharging the water body. In Section 4, the superimposing effects of flood induced infiltration and precipitation will be separately considered.

According to the on-line measured electrical conductivity of the river and the well water the filtrate quantity can be estimated. Monthly measurements of

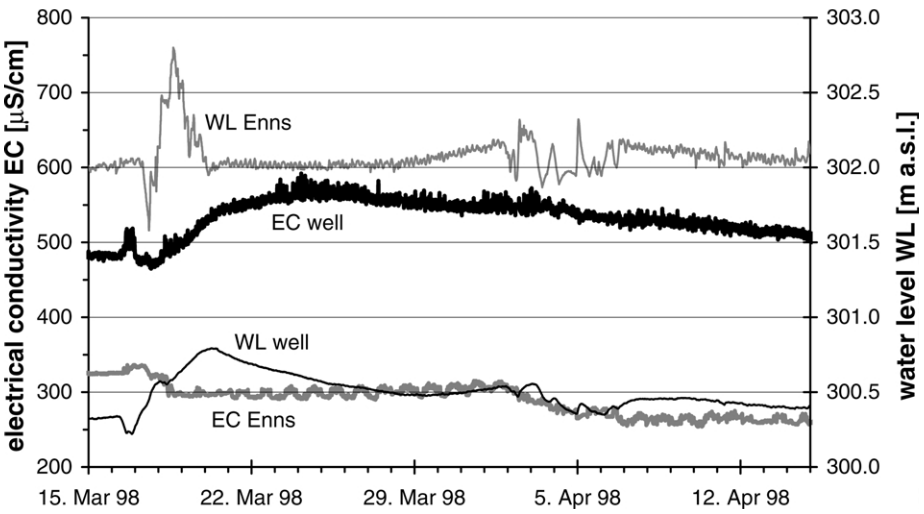

Fig. 4. Profiles of the on-line measured water level and the electrical conductivity of the River Enns and the well during the high-water period in March 1998.

the electrical conductivity in probe KHB01 near the well serve as reference values for the rather stable conductivity of the landside groundwater flux to the well. The average conductivity of the groundwater in March 1998 was about 700 µS/cm. The average rise of conductivity of the river water during the filtration process was also measured (30 µS/cm) and considered. The portion (x) of filtrate (Q_{filtrate}) in the total received well water (Q_{well}) is calculated from the electrical conductivity of the groundwater (EC_{gw}), the filtrate (EC_{filtrate}) and the well water (EC_{well}):

$$x = \frac{Q_{\text{filtrate}}}{Q_{\text{well}}} = \frac{EC_{\text{well}} - EC_{\text{gw}}}{EC_{\text{filtrate}} - EC_{\text{gw}}}. \tag{2}$$

The electrical conductivity was used as a form of natural tracer. The results are shown in Fig. 5. The advantage of the electrical conductivity is its suitability for on-line measurement (data collection every 15 min), but it is influenced by many factors, for example, precipitation. The assumption of constant electrical conductivity of the land-side groundwater flow led to an underestimation of the filtrate portion. Section 4 will present a detailed model-based investigation of the separate impacts of flood and precipitation on the groundwater flow pattern.

3.3. Investigating the seepage rate based on nitrate concentrations

As an additional tracer compound, the nitrate concentration was considered. Nitrate concentrations in the groundwater are about 10 times higher than in the river water and proved to be rather conservative under aerobic conditions (the measured maximum depletion of oxygen was 4 mg/l and of nitrate 2 mg/l on the flow path to the well during summer time). Eq. (2) was also applied to the measured nitrate concentrations (Fig. 6).

As shown in Fig. 6 the filtrate portion decreased from 65 to 51% after the high-water event. It should be noted that the nitrate concentrations were determined at the beginning and the end of the representative observation period of about 1 month. In comparison with Fig. 5, the crucial moments with the lowest filtrate portion have not been revealed. With the exception of the different frequency of sample measurement both methods of determination give similar results. The profile of the off-line measured water level difference between the River Enns and the well in Fig. 6 indicates a significant decrease in the hydraulic head of about 25 cm shortly after the flood.

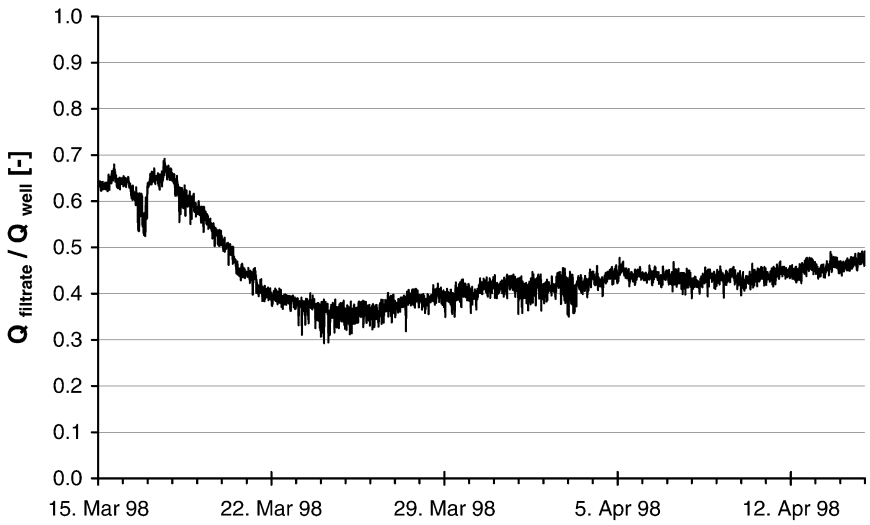

Fig. 5. Profile of the filtrate portion in the well water according to the measured electrical conductivity (Eq. (2)).

4. Numerical modelling of hydraulic head elevation and transport of pollutants by a propagating flood wave

Several studies are reported for the calculation of aquifer heads, stream bank seepage rates and bank storage that occur in response to stream-stage fluctuations. The following examples consider alluvial unconfined aquifers comparable to the field site in this paper but without a bank filtration well. Workman and Serrano (1999) applied model simulations to quantify different recharge sources of an alluvial valley aquifer focusing on the influence of floods on the total water balance during a 5-year study period. Barlow et al. (2000) developed computer programs calculating convolution integrals that represent a superposition of the system's response to individual stresses such as stream-stage fluctuation, recharge and evapotranspiration.

In the case of an aquifer under the influence of a bank filtration well, then as long as the pump induced filtrate flow dominates the flow pattern in the aquifer, no change in the flow direction will take place after flooding. Hence the water stored in the bank

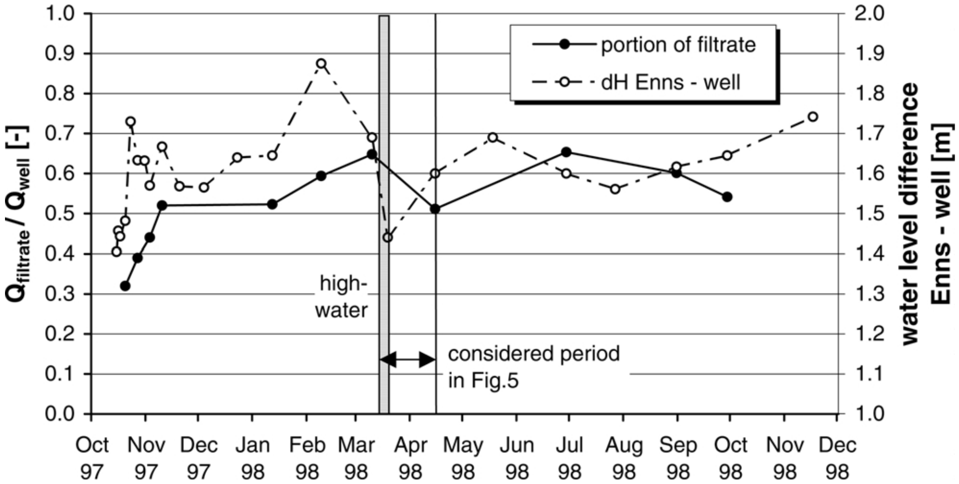

Fig. 6. The relationship between the measured hydraulic gradient and the filtrate portion determined from nitrate concentrations.

discharges to the well and not to the river. During the period of delayed discharge of stored water the groundwater table is elevated causing a decrease of the hydraulic gradient between the river and well. For this reason the seepage rate drops and the total amount of bank filtrate in the well is reduced. In order to confirm these hypotheses numerical three-dimensional modelling was applied.

4.1. Flood wave propagation at the bank filtration site considering precipitation

The hydraulic conditions at the study area were modelled using the US Geological Survey code MODFLOW (McDonald and Harbaugh, 1988). Hydraulic head data were employed to calibrate the flow model under the assumption of a homogenous aquifer (horizontal hydraulic conductivity $k_h = 3.8 \times 10^{-3}$ m/s, anisotropic ratio of vertical to horizontal hydraulic conductivity $k_v/k_h = 0.1$, porosity $\mu = 0.2$) and a riverbed leakance via a vertical boundary layer of 1 m thickness with reduced hydraulic conductivity. The hydraulic conductivity of this riverbed layer was fitted to 3.8×10^{-5} m/s and half the value in the central infiltration area 100 m up- and downstream of the well ($K_h = 1.9 \times 10^{-5}$ m/s).

For calibration of the transport model (MT3D; Zheng, 1990) a natural tracer effect was considered. Due to the large difference between nitrate concentrations in the river and the groundwater, nitrate was chosen as a calibration variable in an unsteady state simulation of the start up phase of the well operation. Obtained migration parameters matched well with values from the literature (Käss, 1992). To validate the flow and transport model a separate tracer test was conducted. 200 l of a saturated sodium chloride solution were dosed into the riverbed via a multi-level probe and on-line measurement of the electrical conductivity in the probes PLE and PLF and in the well recorded the migration of the tracer. The peak of the simulated breakthrough curve reached the well after 4.5 days and showed a good agreement with measurements (Ingerle et al., 1999).

The validated model was applied for a dynamic simulation of the flood event in March 1998. Measurement profiles of stream stage and precipitation served as the model input (Fig. 7). A rain period of 7 days caused the observed high-water of the Enns and started 4 days before the actual stream stage elevation. The calculated steady-state filtrate portion in the well at average flow conditions was calculated to be 63%. When the rain begins the filtrate portion decreases (Fig. 8). The filtrate ratio continues to decline even after the end of the rain period. A very slow system behaviour allows the minimum ratio close to 50% to be reached almost 2 weeks after the flood wave has passed. The system's recovery to steady-state conditions takes another month. Simulation results clearly indicate that the effect of precipitation dominates the effect of the stream stage elevation. A comparison with electrical conductivity measurements (Fig. 5) shows that this parameter is not reliable for the calculation of the filtrate portion after a rain period. Obviously groundwater recharge by precipitation also influences the electrical conductivity of the delivered well water due to the shallow thickness of the aquifer.

The major impact of precipitation is more precisely demonstrated by the response of the groundwater head in the well (Fig. 9). The simulation results do not consider the time lag due to the passage of rain water through the soil indicated by the measured well water level profile (daily measurement values have been used to obtain a smooth profile). After a delay of about 1.5 days precipitation substantially recharges the water body and the measurement profile approaches the simulation results. The flood wave contributes only to the very top of the head elevation in the well with a total increase of 45 cm. The decrease of the groundwater head appears as a long-term process of recovery. From the same simulation run the seepage rate of the central infiltration area is shown in Fig. 10. The filter velocity in the riverbed decreases from an initial 8.3×10^{-6} m/s due to the abstraction of a higher portion of groundwater during the rain period, and then increases sharply by almost 150% (1.2×10^{-5} m/s) before dropping to 50% of the initial value after the rain period (4.1×10^{-6} m/s). Considering a porosity of 0.2 the actual flow velocity of pore water is five times higher. The slow recovery of the seepage rate agrees with the measured head losses in the riverbed (Fig. 3).

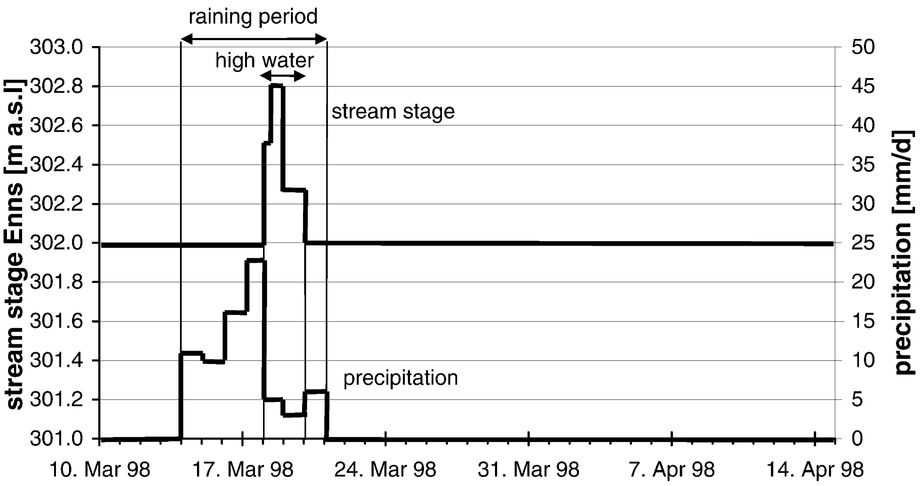

Fig. 7. Measured stream stage and precipitation profiles at the bank filtration site as a model input for a unsteady state flow and transport simulation.

4.2. Flood wave propagation at the bank filtration site without the influence of precipitation

Until now the flood event has been demonstrated by superimposed impacts of precipitation and stream stage elevation. Based on the validated model and the reference simulation both influences can be distinguished and analysed separately. The following simulation scenario assumes time limited shock loads of pollutants induced by the flood wave without any influences from groundwater recharge by precipitation. The simulation begins with steady state base-flow conditions without pollutants followed by a steep rise of the stream-stage and seepage of polluted water before returning to base-flow conditions without pollutants. The model input for the stream stage and all other hydraulic boundary conditions and parameters of the reference simulation are kept the same. The description of transformation processes is neglected but the transport of conservative pollutants

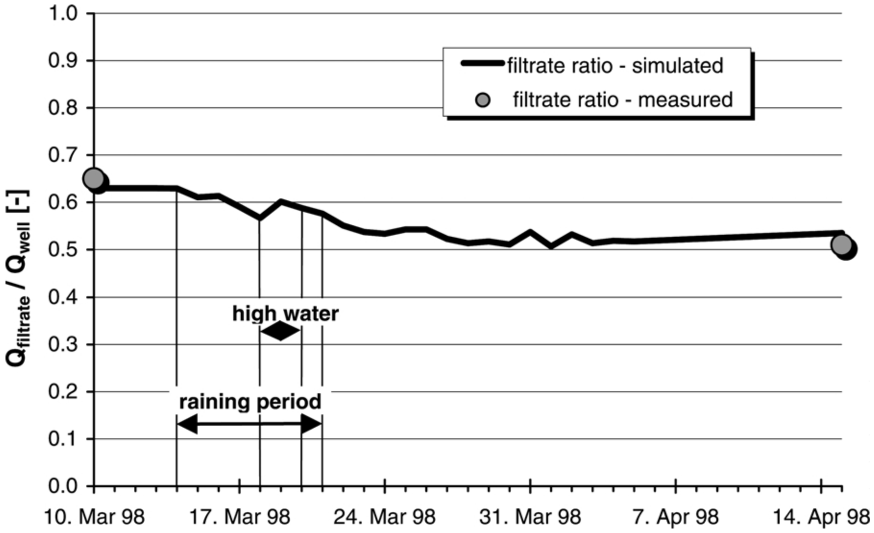

Fig. 8. Portion of filtrated river water in the well as a simulation output in comparison with values calculated from monthly nitrate measurements (see Fig. 6).

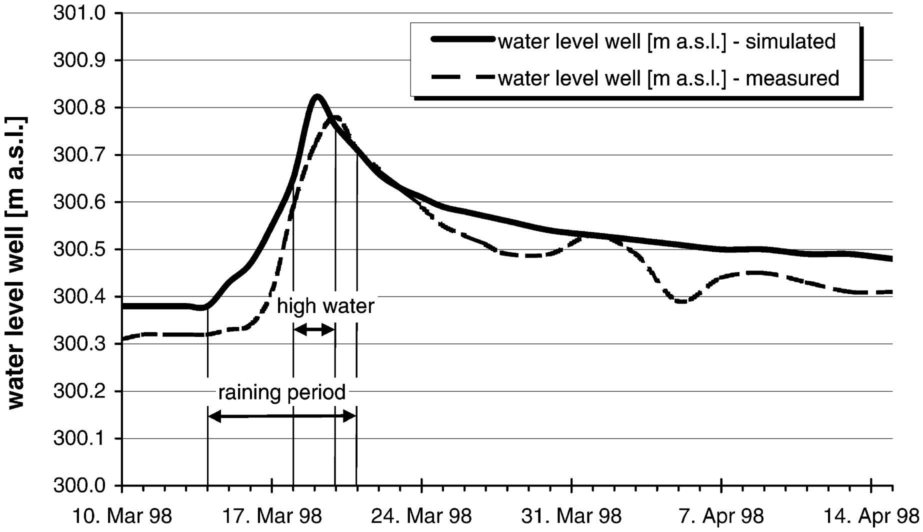

Fig. 9. Simulated and measured rise of groundwater head in the well due to precipitation and stream stage elevation.

with the assumed initial concentration of 7.5 mg/l is considered.

In Fig. 11 simulated water table profiles at various time steps are plotted versus the distance from the river. The calculated propagation of the table elevation from the river to the well is hardly recognisable as a moving wave front. Curve number 2 in Fig. 11 represents the groundwater table 24 h after the step rise of the stream stage and shows the highest level at a distance of about 85 m from the river. At a distance of 110 m the groundwater table 48 h after the step rise (curve 3) exceeds both curves 2 and 4 significantly. This indicates that it takes 2 days for the propagating wave front to reach a distance of 110 or about 55 m per day. Increasing pressure head in the unconfined aquifer causes an expansion of the water body. Water fills un- and half-saturated pores and the elevated table propagates the pressure. These results confirm the fact that pressure head propagation occurs in unconfined or phreatic aquifers much slower

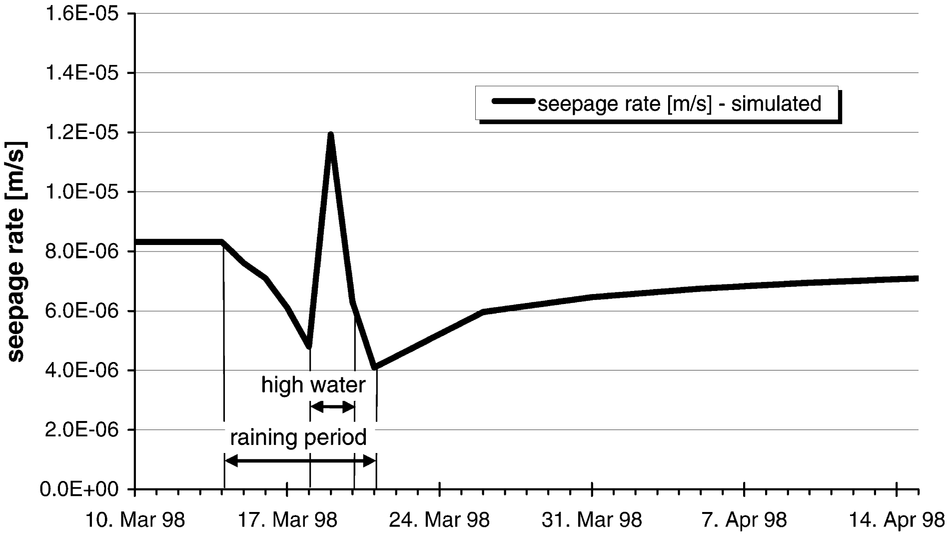

Fig. 10. Simulated seepage rate at the river–aquifer interface of the central infiltration area.

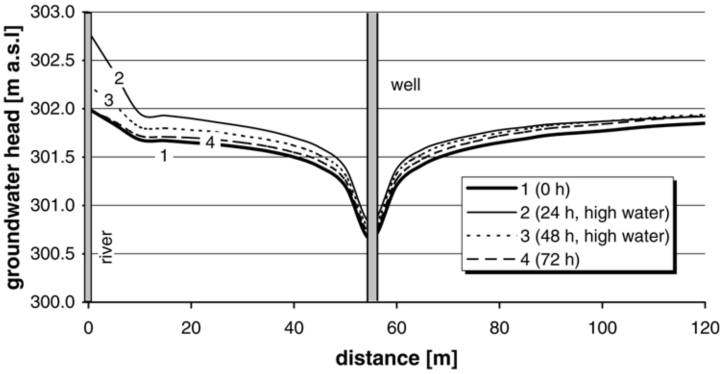

Fig. 11. Simulation of the propagation of the water table elevation induced by a flood wave.

than in confined or semi-confined aquifers. Vekerdy and Meijerink (1998) reported measured propagation velocities in a semi-confined aquifer in the Danube exceeding several kilometres a day. In such a case an expansion of the water body is hindered. Increased pressure causes only an elastic compression of aquifer material and a fast propagation of the pressure head.

Figs. 11 and 12 represent two different propagation mechanisms: water table elevation driven by pressure propagation and mass transport driven by the hydraulic gradient. At high stream-stage and corresponding hydraulic gradient the seepage rate increases (Fig. 13). The bank storage volume is filled within 1 day while the pumping rate stays constant. When the table elevation reaches the well the stream-stage drops again. From this moment on the hydraulic gradient between the river and the well is significantly decreased and the seepage rate is lower than the average rate at base-flow conditions. In comparison to the pressure induced filling of the bank-storage volume the gravity drainage of this volume is a slow process and it takes about a month for the water table in the well to reach its original value. Thus the total amount of river water delivered by the well after flooding is lower than before.

In contrast to the tracer dosage from a point source, seepage of polluted river water during flooding occurs from a line source along the river–aquifer boundary. The concentration front of infiltrated pollutants spreads on the flow path to the well, becoming indistinct by dispersion and reaches the well after a migration time of 4 days (breakthrough of the concentration peak). The dilution effect by land-side groundwater inflow significantly reduces concentration values in the well. Obviously the maximum

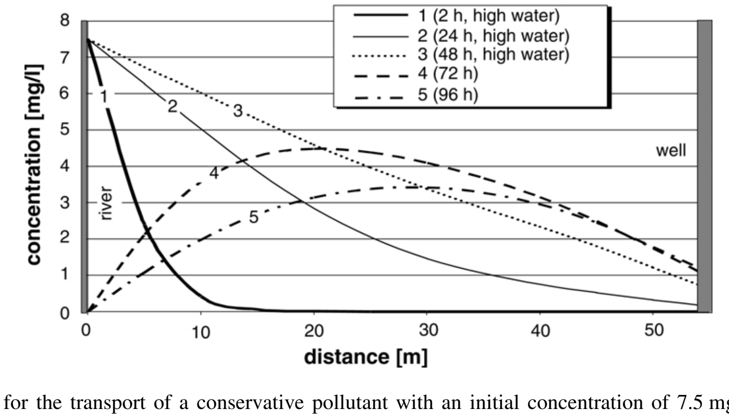

Fig. 12. Simulation results for the transport of a conservative pollutant with an initial concentration of 7.5 mg/l induced by a flood wave (duration of stream stage elevation and pollutant intrusion is 48 h). Concentration profiles at various time steps plotted versus the flow path to the well.

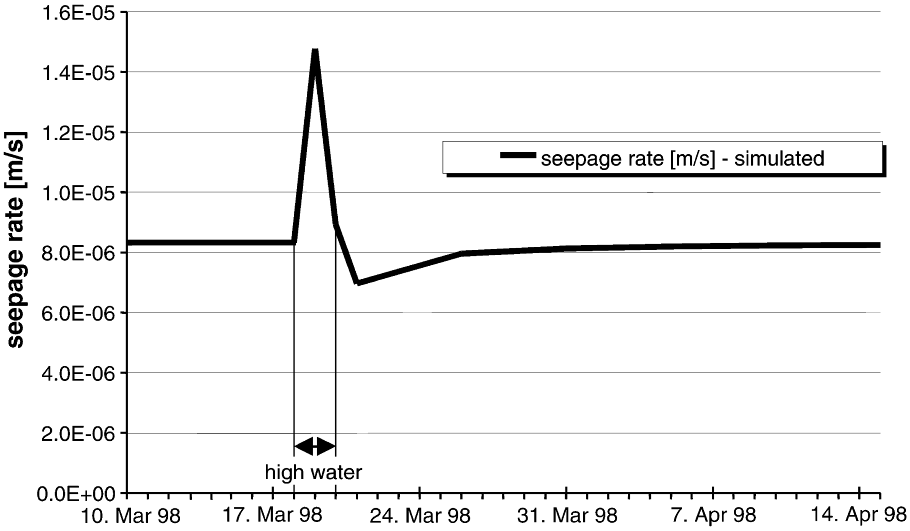

Fig. 13. Simulated seepage rate at the river–aquifer interface of the central infiltration area without the influence of precipitation.

concentration of pollutants in the well depends on the duration of the flood.

The results discussed apply to the investigated site of the Enns with an unconfined aquifer. In case of a confined or semi-confined aquifer with only elastic storage the available bank-storage volume is substantially smaller. As mentioned above the piezometric surface in the well will respond much quicker to a river-stage elevation. Without the additional buffer volume from bank-storage the well will probably be more affected during flooding. Afterwards the water table in the well will drop almost instantaneously with the stream-stage and, without storage discharge effects, the seepage will stay rather constant.

5. Summary and conclusions

Conclusions from reported river–aquifer interactions are often concerned with the question of where to situate a bank filtration well. For the example of a river reservoir, even annual floods cause only moderate stream stage elevations with limited impacts on filtrate flow patterns and, with respect to the present case study of an unconfined alluvial aquifer on the river Enns, no destruction of the filter layer at the stream–aquifer interface occurs. Despite unspectacular changes in river stage elevations, the following effects of flooding were demonstrated by measurement results and dynamic modelling:

- Immediately after flooding the portion of filtrated river water in the well decreased significantly despite the constant hydraulic conductivity of the riverbed. Two reasons were identified: groundwater recharge by precipitation and stream stage elevation during the flood increased the groundwater table. Increased groundwater head together with decreased stream stage after flooding resulted in a reduced hydraulic slope and seepage rate (about 50% of the mean value).
- Both flood induced groundwater table elevation and groundwater recharge by rain filled up the bank storage volume but was depleted by the well operation during the following weeks.
- The propagation velocity of the groundwater table elevation was about 55 m per day. The migration of infiltrated pollutants as a gravity-induced mass flow would be about four times slower. As a general recommendation for the estimation of the filtrate portion in abstract well water at other bank filtration sites, then this study showed that electrical conductivity proved to be an inconvenient parameter for mass balance calculations, since it can be significantly altered by the influence of precipitation. Nitrate appeared to be more appropriate as a natural flow tracer but only in the case of an aerobic aquifer with high

background concentrations in groundwater. Numerical modelling obviously achieves the greatest insights into infiltration flow patterns but requires detailed hydraulic head and tracer test data in order to validate the models.

Acknowledgments

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