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Effect of flood-induced chemical load on filtrate quality at bank filtration sites

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Abstract

Riparian municipal wells, that are located on riverbanks, are specifically designed to capture a portion of the river water through induced infiltration. Runoff from agricultural watersheds is found to carry enormous amounts of pesticides and nitrate. While the risk of contamination for a vast majority of sites with small-capacity vertical wells is low, potential exists for medium to large capacity collector wells to capture a fraction of the surface water contaminants during flood. Prior monitoring and current modeling results indicate that a small-capacity (peak pumpage 0.0315 m^3/s) vertical bank filtration well may not be affected by river water nitrate and atrazine even during flood periods. For a medium capacity (0.0875–0.175 m^3/s) hypothetical collector well at the same site, potential exists for a portion of the river water nitrate and atrazine to enter the well during flood periods. Various combinations of hydraulic conductivity of the riverbed or bank material were used. For nitrate, it was assumed either no denitrification occurred during the period of simulation or a half-life of 2 years. Equilibrium controlled sorption (organic carbon partition coefficient of 52 ml/g) and a half-life of between 7.5 and 15 weeks were considered for atrazine. Combinations of these parameters were used in various simulations. Peak concentrations of atrazine or nitrate in pumped water could vary from less than 1% to as high as 90% of that in the river. It was found that a combination of river stage, pumping rates, hydraulic properties of the riverbed and bank, and soil/pesticide properties could affect contaminant entry from river water to any of these wells. If the hydraulic conductivity of the bed and bank material were low, atrazine would not reach the pumping well with or without sorption and degradation. However, for moderately low permeable bank and bed materials, some atrazine from river water could enter a hypothetical collector well while pumping at 0.0875 m^3/s. It was interesting to note that doubling the pumpage of this collector well would bring in more ground water from the aquifer (with no atrazine) and thus have a lower concentration of atrazine in the filtrate. For highly conductive banks, it is possible to find some atrazine at a vertical well for a sustained pumpage rate of 0.0125 m^3/s if the effect of sorption is neglected. However, with equilibrium sorption, the concentration would be below the detection limit. On the other hand, if a collector well of capacity 0.0875 m^3/s is used at the place of the vertical well with highly conductive banks, atrazine concentration in the filtrate would be about 80% of river water even assuming equilibrium sorption and a half-life of 7.5 weeks. Remediation of river water contamination of the aquifer using ‘scavenger’ wells between the river and the pumping well(s) was not a feasible option due to the contact of the aquifer with a

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highly conductive bank at the site. However, moving the existing pumping well(s) 100 m upstream would have negligible impact from the bank-stored water.

Keywords: Flood; Riverbank filtration; Bank filtrate; Water quality; Numerical modeling

1. Introduction

Many riverbank communities develop water supplies from alluvial aquifers using vertical and horizontal (collector type) wells and these are important components of drinking water production. When wells are placed close to rivers, river water can be induced to flow to these wells under pumping stresses. Under non-pumping conditions, river water can also move towards the aquifer when the river is under flood conditions and the hydraulic gradient is from the river to the aquifer. Riverbank filtration (RBF) is a mechanism by which the permeable riverbed and underlying aquifer solids are used to attenuate dissolved and suspended contaminants in surface water as the water enters the aquifer and moves towards the pumping well(s). The attenuation of contaminants is attained through physical filtering, sorption, and degradation. In addition, dilution is possible if the concentrations of the contaminants in the ground water are lower than in the surface water. However, opposite conditions may be expected in areas where initial nitrate concentration in ground water is higher than that in the river water. It must be pointed out here that nitrate and ammonium concentrations in this paper are reported as mg/l as N. RBF has many similarities with slow sand filtration (SSF), often used for water treatment in small systems, with the added benefits of contaminant mixing with background ground water and multi-dimensional dispersion. In addition, the filter velocity in RBF could be much lower than SSF. Mikels (1992) estimated the rate of riverbed infiltration to be on the order of 2.597×10^{-3} m^3/s/ha). This appears to be much lower than the flow rate through slow sand filters (typically 0.326 m^3/s/ha).

RBF wells have been used at many riverbank communities in Europe and the United States for public and industrial water supply. Table 1 is list of selected bank filtration systems in the US and Europe. As can be found from this table, the horizontal collector wells are generally of high capacity and they are more frequently used by the US utilities. The Louisville Water Company (LWC) in Louisville, Kentucky (USA) has examined the bank filtration potential for major cities located along river valleys (J. Wang, LWC, personal communications). The LWC estimated the population that could benefit from RBF by correlating metropolitan population centers to areas where potential RBF locations can be found. According to the LWC preliminary estimates, over 68.3 million people ($>25\%$ of the total US population) could benefit from water quality improvement by the RBF process.

RBF has many benefits for water quality improvements, including removing dissolved organic carbon (DOC), particulates, pathogens, and other dissolved chemicals including pesticides and nitrate. Removal of DOC has been the focus of research for many European and US utilities from the point of view of the formation of disinfection by-products (DBPs) during the chlorination of water. Chlorine remains the major disinfectant for most US utilities and a small fraction of the European utilities. DOC, a component of the natural organic matter (NOM) is considered as the precursor to the formation of the DBPs. As a result of deficiencies to alternative disinfection, DBP precursor removal is considered the most promising solution (Clark et al., 1994). In water production, the RBF process and additional treatment unit operations within the plant contribute to NOM removal from surface water.

An extensive body of literature exists on the removal of NOM through RBF. For example, Miettinen et al. (1994) observed significant reductions in total organic carbon (TOC), non-purgeable organic compounds (NPOCs), and chemical oxygen demand of the filtrate when it was produced using a series of bank filtration wells on the lakeshore. From UV absorbance, Miettinen et al. determined that up to 87% of high-molecular weight organic fractions were removed, but little removal of low molecular weight fractions occurred. Sontheimer (1991) reported significant removal of various mass fractions of DOC

Table 1
Selected bank filtration facilities for public water supply in the United States and Europe

Site location	Well type (horizontal (H); vertical (V))	Number	Design capacity of wellfield (m^3/s)	River system
Cincinnati, Ohio	V	10	1.75	Great Miami
Columbus, Ohio	H	4	1.75	Scioto/Big Walnut
Galesburg, Illinois	H	1	0.438	Mississippi
Independence, Missouri	H[a]	1	0.656	Missouri
Jacksonville, Illinois	H	1	0.35	Illinois
Kalama, Washington	H	1	0.114	Kalama
Kansas City, Kansas	H	1	1.75	Missouri
Kennewick, Washington	H	1	0.131	Columbia
Lincoln Nebraska	H and V	2(H) + 44(V)	1.53[b]	Platte
Mt Carmel, Illinois	V	1	0.044	Wabash
Sacramento, California	H	1	0.438	Sacramento
Terre Haute, Indiana	H	1	0.525	Wabash
Louisville, Kentucky	H and V[c]	1 + 1	0.875	Ohio
Maribor, Slovenia	V	13	0.75	Drava
Mockritz, Germany	V	74	1.26	Elbe
Torgau, Germany	V	42	1.737	Elbe
Csepel Island, Budapest, Hungary[d]	V and H	280 (total)	3.47	Danube

[a] Gravel-packed laterals.
[b] Peak pumping capacity from the two horizontal wells.
[c] Vertical well is a pilot well with pumping capacity of 0.0875 m^3/s.
[d] Located south of Budapest and serves a part of the city.

from the Rhine River water at the Duisburg Wittlaer water works. For the medium mass fraction DOC (molecular weight 800–1500 g/mol), removal was nearly 70%. Ludwig et al. (1997) observed the attenuation of high-molecular weight DOC (>10,000 g/mol) along the flow direction of the Elbe river, leaving more low-mass fraction DOC in the river water. In US, Wang et al. (1995) observed a 60% reduction in TOC and a 50% reduction in UV absorbance at a monitoring well that is 9 m from the edge of the Ohio River at an experimental RBF well that was pumping at a rate of 0.0876 m^3/s. Total trihalomethanes and haloacetic acids (some of the DBPs) in the chlorinated bank-filtered water were less than 50% of that expected from settled and chlorinated river water.

Doussan et al. (1997) presented a simulation model of the conversion of the nitrogen species during bank filtration of Seine River water in France. They used redox catalyzed bacterial reactions and organic matter mineralization in transport equations. They also presented the role of infiltration velocity and dissolved carbon content on filtrate quality. It must be pointed out that the model was 1D and was developed to reproduce column experiments of von Gunten and Zobrist (1993).

RBF process also helps in the reduction in particle and pathogen count and turbidity of source water. Wang et al. (1995) observed particle removal up to 2.4 log units from the Ohio River water. Mikels (1992) reported the turbidity of pumped water from the collector well of Kalama, Washington to range from 0.3 to 0.4 nephelometric turbidity unit (NTU) and during same time the Columbia River turbidity varied between 1 and 5 NTU. The laterals of the well were located only 6 m below the riverbed. Miettinen et al. (1996) observed strong reductions in bacterial enzymatic activities after infiltration of lake water into an aquifer. They also observed a reduction in biomass production and bacterial cell counts. Havelaar et al. (1995), over a 7-month monitoring of entero- and reoviruses at a bank filtration plant, reported that bank filtration easily provided 4-log removal. They observed better (5–6 log removal) performance of bank filtration for removing F-specific RNA coliphages. Schijven and Rietveld (1997) also showed 5 log units at two RBF sites and 3–4 log units at a third site.

Wilderer et al. (1985) studied the removal of synthetic organic compounds through RBF in the lower Rhine River basin. The removal efficiency was dependent upon the biodegradability of the organic compounds and their initial concentrations. For example, they observed 71% removal of aromatic amines (initial concentration 17 μg/l), 33% removal of trichlorethene (initial concentration 1.5 μg/l), and almost no removal of chloroform (initial concentration 15 μg/l). Atrazine concentration in bank-filtered Ohio River water at Louisville (USA) was below the method detection limit (0.1 μg/l), although the unfiltered river water had concentrations above 1 μg/l (Wang et al., 1995). Sontheimer (1991) conducted some adsorption experiments with atrazine. However, he observed poor removal and attributed this to low biodegradability of the pesticide and low adsorption capacity of the sediment.

The quality of filtrate at RBF facilities depends upon numerous factors, including the source water characteristics, local geologic settings, distance of the wells from the surface water source, and their pumping rates, and ultimately the biogeochemical processes in the aquifer and the sediments at the river-aquifer interface. The source water quality is controlled by land use and climatic conditions. Spring and early summer floods in rivers passing through agricultural watersheds often contain high concentrations of farm chemicals. Contaminant concentration peaks often coincide with flow peaks (Coupe and Johnson, 1991; Demissie et al., 1996). Stamer and Wieczorek (1996) reported the concentrations for 46 pesticides at two monitoring stations in the Platte River basin, Nebraska, US. At least 10 pesticides were detected on 10 or more occasions at these two sites between the period 1992 and 1994. Cyanazine and atrazine exceeded their respective maximum contaminant levels (MCLs) in many flood occasions. Ray et al. (1998a, 2000) also observed elevated concentrations of pesticides and TOC during flood periods.

Liszewski and Squillace (1991) studied the exchange of atrazine between the Cedar River water and shallow ground water in an adjoining alluvial aquifer in Iowa, US from May through December, and more extensively during a single flooding event in March 1990. They observed stratification in atrazine concentration in the bank-stored water with the highest concentration being near the top. Several commonly used herbicides were found 100–200 m away from the riverbank at concentrations above the background levels. Thus, under low pumping stresses, a well can pick up some of these compounds. Squillace (1996) also showed that for the same site, a 2-m rise in river stage caused the bank-storage water to move 30 m laterally to the aquifer and about 4 m below the river bottom. Through modeling, he showed that the bank-stored water would take about 5 weeks to drain from the alluvial aquifer to the river after the peak flow in the river recedes. The lateral migration of bank-storage could have partly been affected by site geology where the alluvial aquifer is shallow due to the presence of low permeable layers. Schulmeyer (1995) and Verstraeten et al. (1999) have shown the potential for municipal RBF wells to contain trace concentrations of herbicides, in Cedar Rapids, Iowa and Lincoln, Nebraska, respectively. Rissati and Mehnert (1995) observed the movement of atrazine from Salt Creek towards the pumping wells of the town of Mt Pulaski, Illinois, which are low capacity wells ($0.0063–0.0126\ m^3/s$) that pumped intermittently. The aquifer at the site is thin and unconfined. However, the wells are located very close to the creek and often the floodwaters could inundate the ground around the wells.

Ray et al. (1998a,b) examined the movement of atrazine from the Illinois River towards the two RBF wells of the town of Henry, Illinois. The town has an average pumping demand of $0.0126\ m^3/s$ and each RBF well is equipped with a pump rated for $0.0312\ m^3/s$. Thus, pumping is intermittent excepting during peak demand periods. From the ground water monitoring data, it was found that there is significant storage and release of river water in the permeable bank depending upon the river stage. However, this did not have significant adverse impact on the bank-filtered water. While the river water contained atrazine in excess of 4 μg/l, the pumped water did not contain atrazine above its detection limit of 0.1 μg/l. Multi-port monitoring wells (Section 2), located within 15 m of the normal navigation pool of the river never contained atrazine in excess of 0.2 μg/l. On the contrary, a collector well for the City of Jacksonville, Illinois, approximately 210 km downstream, is located at a place where the riverbed appears to have been somewhat silted. The well has a peak pumping capacity of 0.35 m^3/s and normal pumpage is slightly below 0.3 m^3/s. The peak concentration of atrazine in the river during a flooding event in 1996 was 11.2 μg/l, however, the peak concentration in the filtrate was 1.1 μg/l and there was a lag time of approximately 2 weeks between the two peaks (Fig. 1). The laterals of the well are approximately 18 m below the bed of the river. Nitrate content of floodwater was somewhat higher than the ambient ground water and a similar small peak was observed for nitrate. In addition to sorption and dilution, DOC in river water and microbial reactions along the flow path may have some impact on individual species concentrations.

The severity of contamination of a RBF well appears to vary depending upon the site conditions, pumping stresses, and possibly the river system. As discussed earlier, biogeochemical factors in the river-aquifer system and the geohydrologic conditions of the site also contribute to contaminant transport at the RBF systems. Some of the factors that need to be addressed include the impact of flooding on the hydraulic conductivity of the sediment interface at the river-aquifer boundary and the resulting impact on contaminant migration towards the pumping wells. Further, the coupled effects of this altered hydraulic characteristics of the interface along with the biogeochemical processes in the aquifer need to be investigated. The objective of this paper is to use numerical modeling to predict chemical concentrations in filtrate water at this bank filtration well due to factors such as river stage/flow, pumpage (involving well types), riverbed conditions, and biogeochemical reactions using some of the data collected at the site and through sensitivity analyses. Numerical simulations have been conducted according to various design scenarios to examine the dissipation of concentration peaks at the pumping and selected monitoring wells.

2. Methodologies and approaches

2.1. Study site

The study site is located at Henry, a small community located on the western bank of the Illinois River upstream from the Peoria Pool. The town has three vertical wells of which two (wells 3 and 4) are located on the bank of the Illinois River at distances of 53 and 61 m, respectively, from the river's edge under normal pool elevation (134 m above mean sea level, MSL). During the study period between 1995 and 1996, floodwater reached within 3–7.6 m from these two wells. The third well (well 5) is located

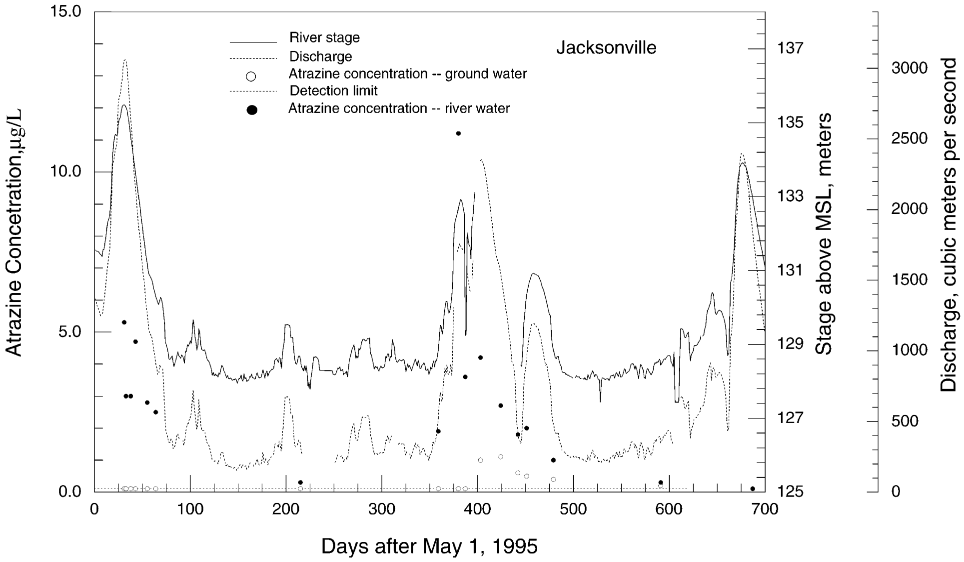


Fig. 1. Flow and atrazine concentration peaks at Jacksonville, Illinois (redrawn after Ray et al., 1998a).

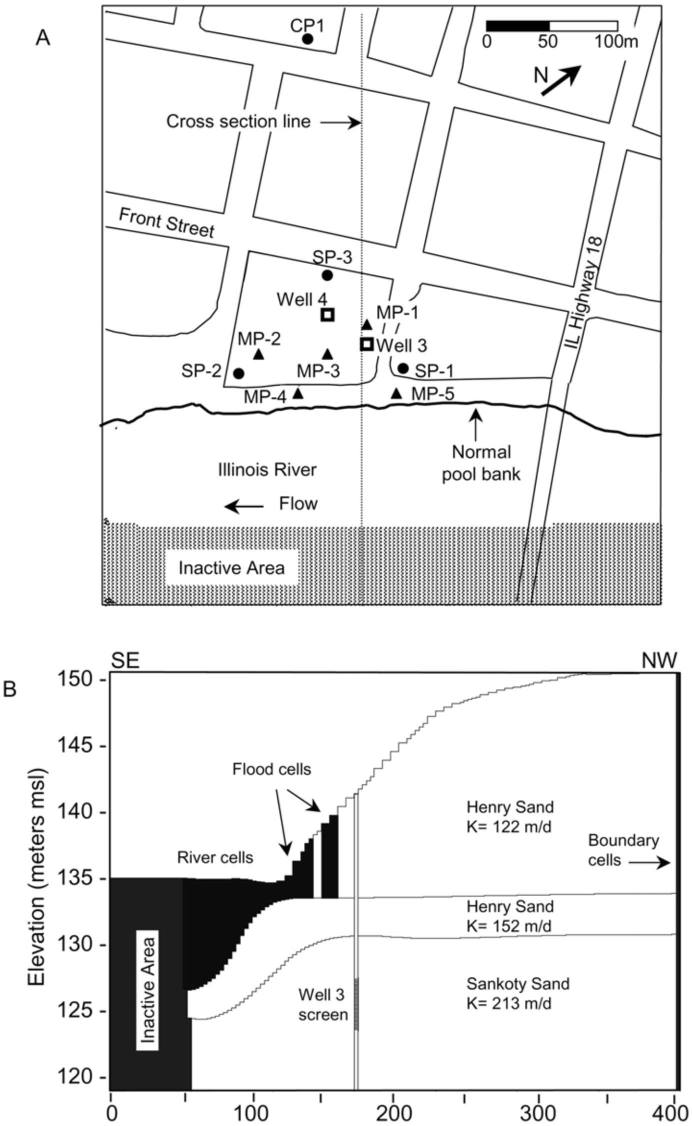


Fig. 2. Locations of pumping and monitoring wells at the Henry bank filtration site together with model grid and the vertical cross-section.

approximately 2.4 km west on higher grounds. Nitrate in ground water was high (about 8–10 mg/l as N) and in well 5, it was generally higher than that in wells 3 and 4. Nitrate in the aquifer may come from the following sources: the river, septic systems within the town or irrigated agricultural fields to the west and outside of the town, or their combinations. During and prior to 1997, individual septic systems were the primary mode of waste disposal. At present, the town has completed conversion to a sewer system with centralized treatment facility. The town uses an ion exchange system to reduce the nitrate content of finished water to between 6 and 8 mg/l as N. The utility manager prefers to use wells 3 and 4 to reduce the consumption of the ion exchange resin for removing nitrate. Wells 3 and 4 have 4.3 m screens and turbine pumps with peak pumping capacities of 0.0312 m^3/s each. Normally, one of the two wells is pumped at a given time and the system pressure controls the pumping cycle.

2.2. Data

2.2.1. Site geology

The surficial deposits at the site belong to the Mackinaw member of the Henry Formation (Willman et al., 1975) to an elevation of approximately 130 m MSL (Fig. 2). Along the river, a thin layer of Cahokia alluvium exists between depths of 0.9 and 2.1 m. The upper part of the riverbank is lined with boulders and concrete rubble. The Mackinaw member is underlain by a thick deposit of sand and gravel belonging to the Sankoty Sand member of the Banner Formation. The bedrock is Pennsylvanian shale of the Carbondale Formation and it lies approximately at 118 m MSL. A geologic cross-section of the site is presented by Mehnert et al. (1996).

2.2.2. Discharge, stage, and water table

There is an US Geological Survey gaging station at Henry which provides the local estimates of daily discharge and stage of the Illinois River. Ground water levels were measured at the pumping and monitoring wells during the sampling events between 1996 and 1997. Three single port observation wells were installed along the regional flow field upgradient of the wellfield including well CP1 shown in Fig. 2A. The regional ground water flow direction is southeast towards the Illinois River at a gradient under normal river conditions of approximately 0.001.

2.2.3. Water quality

Ground water samples were collected from the production and selected monitoring wells (Ray et al., 1998b) between June 1996 and July 1997. Three single port observation wells (SP-1 to SP-3) existed at the site. To examine the variation in ground water chemistry with depth, five multi-port observation wells (MP-1 through MP-5) were installed in November 1996. The design of these wells followed that presented by LeBlanc et al. (1991) and modified by Kelly and Ray (1999). The screen elevations for the sampling points are given in Tables 2–4.

In samples collected on June 23–24, 1997, the nitrate concentration in the ground water around the well field is very uniform between 8 and 10 mg/l except near the water table (Table 2). Likewise, the dissolved oxygen concentrations were generally around 4 mg/l and the redox potential for one of the shallow sampling points for MP-5 was close to 0 mV (Table 3). The iron concentrations were generally below 0.3 mg/l except near the water table and the same trend was found for manganese with the exception that manganese concentrations were substantially lower than the iron concentrations (Table 4). The uppermost ports of MP-4 and MP-5 are completed either in the Cahokia alluvium or the surficial layer of fill and soil. Reducing conditions exist in this zone at MP-5 as indicated by a dissolved

Table 2
Nitrate concentration (mg/l as N) from a sampling event on 6/23/97

Screen elevation (m)	MP-1	MP-2	MP-3	MP-4	MP-5	SP-1	SP-2	SP-3	CP1	Well 3
139	ns									
137.5	ns	ns	ns		ns					
136	ns	9.66	ns	ns	0.53				10.07	
134.5	6.75	8.68	10.10	1.27	7.84					
133	8.78	10.58	9.27	8.58	8.80					
131.5	8.18	10.11	9.29	9.00	9.72					
130	9.04	10.15	8.96	ns	10.25		10.29			
128.5	9.05	10.06	8.56	9.97	8.90	10.24		9.65		
127	ns	10.18	9.94	9.75	9.86					
125.5	8.98	10.37	10.10	9.78	9.55					
124		10.04	10.34	8.15						
122.5				8.81						9.17

ns—not sampled or no samples were available (dry); river stage was 135.28 m with a nitrate concentration of 4.16 mg/l as N.

Table 3
Dissolved oxygen (mg/l) and redox potential (mV) measurements from a sampling event on 6/23/97

Screen elevation (m)	MP-1		MP-2		MP-3		MP-4		MP-5	
	DO	Redox	DO	Redox	DO	Redox	DO	Redox	DO	Redox
139	ns									
137.5	ns		ns	ns	ns	ns			ns	ns
136	ns		4.58	337	ns	ns	ns	ns	0.88	0.1
134.5	3.5	309	4.97	333	3.96	352	2.00	214	ns	ns
133	ns	ns	5.1	335	4.23	352	3.10	357	4.46	358
131.5	3.65	328	4.33	341	4.21	349	3.98	354	4.15	357
130	4.61	340	5.1	337	3.93	345	ns	ns	ns	ns
128.5	4.20	336	4.25	340	4.24	342	4.24	353	4.25	354
127	ns	ns	ns	ns	2.35	336	4.28	345	ns	ns
125.5	4.27	326	4.18	344	4.26	340	5.07	342	3.66	348
124			4.23	344	4.03	337	4.68	337		
122.5							3.45	336		

ns—not sampled or not enough sample to measure the parameters; river stage was 135.28 m with a dissolved oxygen level of 8.1 mg/l.

Table 4
Total iron and manganese concentrations (mg/l each) from a sampling event on 6/23/97

Screen elevation (m)	MP-1		MP-2		MP-3		MP-4		MP-5	
	Fe	Mn	Fe	Mn	Fe	Mn	Fe	Mn	Fe	Mn
139	ns									
137.5	ns		ns		ns				ns	
136	ns		0.332	0.064	ns		ns		8.190	2.401
134.5	0.614	0.087	0.066	0.043	0.161	0.030	0.414	1.727	0.734	0.118
133	0.495	0.054	0.060	0.030	0.191	0.023	0.768	0.072	0.141	0.033
131.5	0.117	0.049	0.114	0.032	0.118	0.025	ns	ns	0.253	0.056
130	0.127	0.043	0.085	0.029	0.135	0.026	1.000	0.121	0.199	0.055
128.5	0.136	0.022	0.269	0.036	0.122	0.021	0.063	0.006	0.171	0.031
127	ns	ns	0.154	0.026	0.088	0.015	0.077	<0.006	0.380	0.050
125.5	0.146	0.031	0.097	0.022	0.099	<0.006	0.053	0.037	0.682	0.036
124			0.090	0.035	0.053	0.037	0.053	0.017		
122.5							0.051	0.126		
	SP-1		SP-2		SP-3		CP1		Well3	
139										
137.5										
136							0.719	0.079		
134.5										
133										
131.5										
130			0.140	0.050						
128.5	0.212	0.050			0.228	0.067				
127										
125.5										
124										
122.5									0.266	0.028

ns—not sampled; river level was 135.28 m with total iron and manganese concentrations of 2.38 and 0.221 mg/l, respectively.

oxygen measurement below 0.8 mg/l (Table 3), and iron concentration of 8.19 mg/l (Table 4), and an ammonium concentration of 1.17 mg/l as N. Atrazine was detected at the detection limit of 0.1 μg/l in the uppermost ports of MP-3 and MP-5 and one of the middle ports of MP-1. The Illinois River had a nitrate concentration of 4.05 mg/l as N and an atrazine concentration of over 2 μg/l.

In July 1997, pore water samples were collected from the riverbed sediment (between 0 and 1.4 m depth from land surface) at three locations on a transect from multi-port well MP-4. The three locations were 6.7, 8.9, and 10.8 m from MP-4, respectively. A well point with 30-cm screen was driven into the sediment at these locations and water samples were collected using a peristaltic pump. Near the shore, the riverbed was composed of sand. Nitrate–nitrogen concentration from the well point near the bank was 8.41 mg/l and 8.29 mg/l at depths of 0.3 and 1.0 m in the sediment, respectively. Farther offshore where the riverbed was composed of silt and clay, the nitrate–nitrogen concentration was less than 1 mg/l at depths of 0.3, 0.8, and 1.4 m. However, the ammonium concentration at 1.4 m was 11.7 mg/l as N. At the time of this investigation, the concentration of nitrate in the river was 2.69 mg/l as N. It is possible that part of the nitrate-rich ground water that discharges to the river through the sandy bank is not denitrified and that which discharges to the river through the bed sediment is denitrified in the redox environment of this layer. Further, the bed sediment (away from the bank) receives organic debris that decomposes to form ammonium nitrogen.

2.3. Model formulation

2.3.1. Models used

Three-dimensional (3D) groundwater flow at the site was simulated using the US Geological Survey model MODFLOW (McDonald and Harbaugh, 1988, later modified by Harbaugh and McDonald (1996)). Considering the depth to the water table at the study site is low and the ground is submerged when flood tops the bank, only the saturated zone was used for modeling. A 3D model was constructed for the existing vertical wells first. Then the same model was modified to include a hypothetical collector well at the site that is based on the collector well at Jacksonville (Ray et al., 1998b). Mass transport simulation was carried out using the solute transport code MT3D (Zheng, 1992). The model solves the concentration distribution of a contaminant taking into account advection, hydrodynamic dispersion, and reactions. MT3D operates on the same grid used by MODFLOW and uses the hydraulic heads and the hydraulic conductivity and effective porosity to compute ground water flow velocity. The flow velocity is used in the advection term and also with the dispersivity of the medium to compute dispersion coefficient. Reader may refer to Zheng and Bennett (1995) for details of solution mechanisms.

2.3.2. Coverage and grids

The modeled area extended landward from the edge of the Illinois by 394 m and up and down stream 424 m (Fig. 2A). The model domain was equally divided into 130 rows and 140 columns, yielding grid cells that are approximately 3 m square on the plan view and consisting of seven layers. The saturated thickness of the vertical layers ranged from 1.2 to 3.9 m. Fig. 2B shows a cross-section of the model through well 3.

For modeling the collector well an additional (0.6–1.0 m thick) layer was created from the top part of the sixth layer to accommodate the collectors. Thus, the total number of layers increased to eight in for collector well simulation. Fig. 3 shows a schematic diagram of the five laterals of the collector well. Each lateral is approximately 0.6–0.9 m thick and 3 m wide, with a length ranging from 77 to 85.7 m. In essence, the length directly beneath the riverbed and bank exceeded over 30 m for most of the laterals. In order to accommodate the typically perforated laterals, the hydraulic conductivity of the cells, where the laterals were present, was assumed to be extremely large (1.23 m/s) so that the water could flow freely along the laterals. All the five laterals were connected in the sixth layer of this new model. In order to reduce the velocity gradient and the resulting run time for the transport model, pumping from the caisson was distributed over several grid cells. In essence, the caisson served a bundle of wells each with a screen in the sixth layer where the laterals joined them. The total pumping rate was 0.0875 m^3/s.

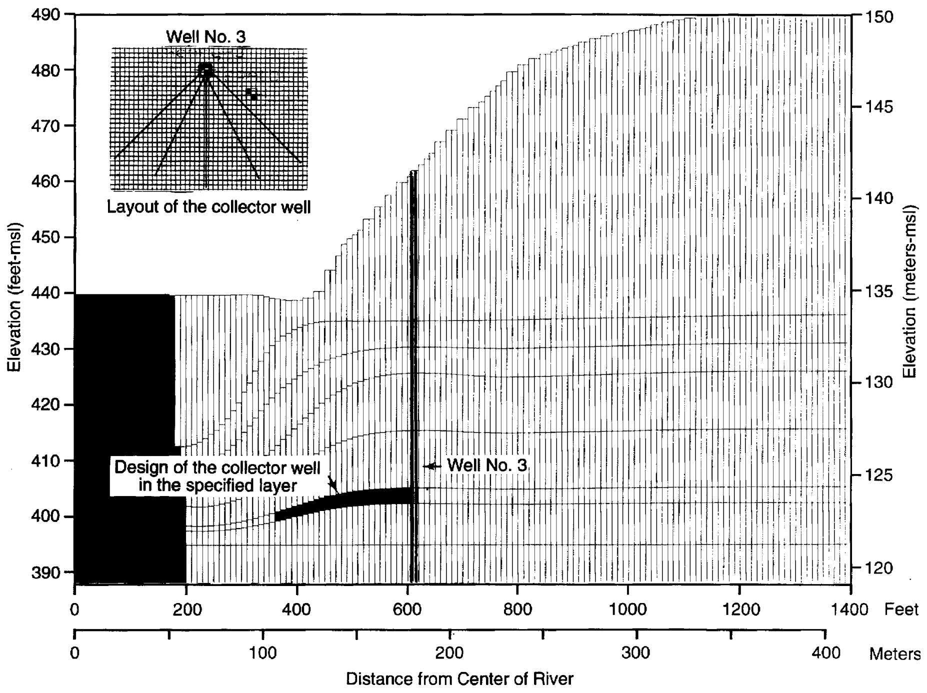


Fig. 3. Layout of the proposed horizontal well laterals at the Henry site.

2.3.3. River and flood cells

During periods of high flow, the river water inundates a grassy area between the bank and the wells, creating a shorter flow path for river water to reach the wells. Therefore, the areas on the bank that are above the normal pool elevation of river were treated as bank cells. These bank cells became active during periods of submergence, and have been designated as ‘flood’ cells. An extreme low conductance was used to make these cells ‘inactive’ during normal river stages. Flood cells include the first 3–4 rows of cells on the riverbank and a few rows of cells on the other side of a road that traverses along the riverbank are designated as flood cells. The riverbed below the normal pool level of the river was treated as river cells.

2.3.4. Parameter values

A controlled aquifer test was conducted in June 1996 at the site by pumping well 4 at a rate of 0.0312 m^3/s for 24 h. The estimated transmissivity of the aquifer was 4.3×10^{-2} m^2/s and the specific yield was 0.11. The aquifer material became coarser with depth. Based upon grain size data from drilling logs, higher conductivity layers were assigned to the bottom. The hydraulic conductivity of the first layer was 1.4×10^{-3} m/s, that for second and third layers was 1.76×10^{-3} m/s, and the remaining four layers had hydraulic conductivity of 2.47×10^{-3} m/s (Fig. 2B). For simplicity and due to the absence of site specific data, the hydraulic conductivity in the two horizontal (X and Y) directions was assumed to be isotropic in a given layer and that in the Z direction was assumed to be one-half of the aerial values for that layer. A porosity of 28% was used in the model based upon the study of Naymik and Sievers (1983) at a similar setting downstream of the river.

The hydraulic conductivity of the riverbed and steep banks (river cells) was estimated using the particle size data following a procedure similar to Vukovic et al. (1992) and the hydraulic conductivity of the areas that normally get inundated during high flows (flood cells) was estimated using a seepage meter (Lee and Cherry, 1978). Sediment core samples were collected 30, 60, and 90-cm depths from several locations on three transects between the bank and the centerline of the river and the samples were analyzed for their particle size distribution. The surficial bed materials of the river were fine sediments primarily silt and clay while the surficial bank materials were medium to coarse sand. The estimated hydraulic conductivity of the riverbed was on the order of 5.3×10^{-8} m/s, whereas that for the bank areas was 1.27×10^{-3} m/s. The bank area conductivity measured from particle size distribution appears to be close to the hydraulic conductivity of the top layer. The hydraulic conductivity of the topsoil (the grassy area that gets inundated during floods) from seepage meter experiment was 1.4×10^{-5} m/s. This value is somewhat between that for sandy bank and finer riverbed material. Using this hydraulic conductivity, one would be able to estimate that nearly 12% pumped water during June 1996 flood could have originated from the inundated areas.

High flows are expected to enhance the suspension of riverbed sediments. Coupled with higher heads and increased bank/bed conductivity, more water can move from the river to the bank filtration wells. However, river dynamics can affect the hydraulic conductivity of the riverbed materials. Utility managers have noticed less drawdown in collector well caissons for given pumping rate and river stage during a flood than the drawdown for the same rate of pumping and river stage after a flood. (Personal communications, D. Byus, Superintendent, Jacksonville Water Department, Jacksonville, IL, 1997). This could be attributed to pore clogging of riverbed/bank sediments sometime after the flood recession. Thus, it is expected that variations in bank and bed conductivity could be large due to flooding in rivers. In this research endeavor, safety concerns prevented collecting sediment core samples during flood events hence for analyzing changes in the particle size distribution. Sounding on the several transects measured before the flood with sonar showed that the riverbed was subject to scour during the flood, however, precisely matching the pre-flood transects was difficult.

Most leakage was expected to have occurred through the bank sections. The model uses riverbed conductances for each grid cell which are calculated by multiplying the hydraulic conductivity by the cell area and dividing by streambed thickness. For most practical applications, the thickness of surficial sediments rarely exceeds 0.6–0.9 m. For this work, a thickness of the fine and coarse sediment layers was assumed to be 0.9 m, each. The presence any fine bed material beneath parts of the exposed bank material (especially close to the shore line) was not known since deeper samples were not available.

Dispersion and chemical reactions were needed for MT3D simulations. A small dispersivity value (0.3 m) was used in the model. It is recognized that dispersivity depends upon the scale of the problem. Although this value is somewhat low, it was used as a conservative value that was sufficient to prevent numerical oscillations. Oshiro et al. (1993) and Hornsby et al. (1996) summarized from published results that the atrazine half-life could vary from 20 to 6900 days with most between 49 and 56 days. The type of study (laboratory versus field) and environmental conditions (aerobic versus anaerobic) are suspected to have caused some of these variabilities. Similarly, the half-life of nitrate can vary depending upon the hydrogeologic setting and other conditions (Korom, 1992). Frind et al. (1989) used a half-life of two years for denitrification in an agricultural watershed in Germany. Because of experimental difficulties and other constraints, site specific data were not available and simulation was relied upon literature derived data. For this reason, the sorption distribution coefficient and the half-life values were varied for different scenarios. An organic carbon partition coefficient of 52 ml/g was used in our calculations. In the absence of actual field data, the organic carbon in the top layer was assumed to be 0.25% and the subsequent layers was assumed to be one-half of that. Although these values seem to be low, they represent a conservative condition. For all simulations involving the vertical well, the half-life for the degradation of atrazine was considered to be about 7.5 weeks. With the exception of two scenarios, a majority of simulation scenarios for the collector well used a half-life of 15 weeks for conservative estimates. In some scenarios, the half-life of atrazine was set to be infinitely large. This was equivalent to assuming no degradation and was used for worst-case conditions. Similarly, the half-life value of nitrate was set either 2 years or an infinitely large number. Simulation scenarios for nitrate and atrazine for the vertical and the horizontal wells are presented later in Tables 7 and 8, respectively. For calculating the retardation factor, a bulk density of 1700 kg/m^3 was used.

2.3.5. Boundary conditions

The boundaries towards north and south (Fig. 2) were assumed to be no-flow boundaries since the general ground water flow direction is towards the river. The western boundary was a specified-head boundary where the hydraulic head was assigned based upon available water level data from observation well CP1 and a second observation well that is somewhat west of the model boundary. The eastern boundary was the river. Model cells to the east of the centerline of the river were treated as no-flow cells. The boundary conditions for flow included the heads in the western boundary, the river stage on the east, and aerial recharge over the top layer. Recharge to the unconfined aquifer generally takes place from the upland areas west of the site. Annual average recharge along the Illinois River valley south of Peoria has been estimated to be approximately 0.25 m (Walton, 1965). Based upon rainfall characteristics, geology, and topography, similar recharge rate is expected for this site. Though the annual average recharge is 0.25 m, season and storm events greatly influence episodic recharge events.

The boundary conditions for transport simulation were linked to the river, constant head and recharge boundaries. Two chemical constituents were simulated. The concentration of atrazine in recharge water was set to zero and the concentration of nitrate was set to 1.0 mg/l, which is typical for rainfall in the area (Bowersox, 1997). The concentrations at the western boundary varied from 9.0 to 9.5 mg/l for high nitrate case or 3 mg/l for the low nitrate case. The low nitrate case was used to observe the effect of river water nitrate on the filtrate quality some years after the conversion of the town to a centralized sewage collection system.

2.4. Simulation strategies

Simulations were conducted to evaluate the effect of flood stage and the associated changes in bank and bed conductivity, well type (hence pumping rates), and reaction rates (sorption distribution coefficient and decay rates) on the concentrations of contaminants. The results are examined at the city's pumping well (well 3) and selected observation wells (SP-1 and SP-3), representing different distances from the river's edge. It is understood that the assumption of equilibrium controlled sorption and first-order decay process may not hold for certain conditions within transient flow or near the well. However, in the absence of actual data on sorption and degradation, a range sorption and degradation values were used in our analyses (Tables 7 and 8).

2.4.1. Event studied

Floods in Illinois River generally occur in late spring and early summer months. A typical flood hydrograph is expected to last between 4 and 6 weeks depending upon the river, watershed, and precipitation conditions. The spring flood of 1997 was used in this study. The simulation period was 70 days (10 weeks) of which 6 weeks corresponded to a flood hydrograph in the river with 2 weeks of low flow prior to and after the flood each. The 6-week flood hydrograph in the river was broken into three steps, each 2 weeks long. Although daily flow data were available, daily concentration data in the river were not. Further, model input would require daily river stage and interpolated concentration data for nearly 9000 cells. This forced us to use longer time interval in the river hydrograph. By using a few key values in the rising and falling limb of the hydrograph, we were able to preserve the height and location of flow and concentration peaks. Table 5 provides the river stage,

Table 5
River stage, specified head, and assumed recharge rates used for simulation

Days	River stage above MSL (m)	Head at the western boundary (MSL) (m)	Recharge rate over the period (m/yr)
0–14	134.7	135.1	2.54×10^{-2}
14–28	136.9	136.0	0.635
28–42	138.4	136.9	1.27×10^{-2}
42–56	136.9	137.8	0.635
56–70	134.7	135.1	2.54×10^{-2}

Table 6
Concentration of nitrate and atrazine in river water and nitrate at the specified-concentration cells for simulation

Days	River water nitrate (mg/l)	River water atrazine (μg/l)	Nitrate at the western boundary (mg/l)
0–14	4	0.2	9.5
14–28	8	3	9.5
28–42	10	4	9.0
42–56	8	3	9.0
56–70	3	1	9.5

western boundary head, and the recharge rates for each of the 2-week simulation periods. It was assumed that approximately 40% of yearly recharge took place during the 70-day simulation period (which is in late spring and early summer) and the rate varied as presented in Table 5. Concentrations of nitrate and atrazine in river water and nitrate content of the specified-head (which are also specified-concentration) boundaries for the duration of simulation are presented in Table 6.

For a small number of simulations involving the collector wells, the flood simulation was carried out for additional 4 weeks (98 days) to examine the shape and the location and the concentrations of the concentration peaks realized at the pumping wells and selected observation points. The water levels at the river and the initial and the boundary conditions for this extra simulation time was kept as the fifth stage (days 56–70) of the flood hydrograph.

2.4.2. Scope

For simulations, the hydraulic conductivity of riverbed and the bank cells were allowed to vary over several log orders to examine their impact on filtrate quality. In order to present the results of the above analysis in a meaningful way, it was decided to separate the chemicals and well types. The strategy for simulation is presented in Table 7 for vertical wells and in Table 8 for collector wells. In Table 7, the bank hydraulic conductivity was varied using five sets of values (*A* through *E*) for the vertical well. In Table 8, four sets of values (*A*, *B*, *E*, and *F*) were used for the collector well. The riverbed hydraulic conductivity was assigned three values (*A*, *B*, and *C*) for vertical well simulation (Table 7) or three other values (*A*, *B*, and *F*) for collector well simulation (Table 8). Hydraulic conductivity for sets *B*, *C*, *D*, *F* were varied at each flood stage. In one simulation for the collector well, the effect of pumping stress was tested. Since the simulated concentrations can be expected to vary vertically in layered systems (in 3D simulations), the locations of observation points are quite important. For model testing or for predictive simulations, the observation points in flow and transport simulations were set at the mid-screen level for the pumping or the observation wells. For collector well simulation, the observation point was set near the center of the caisson. However, concentration in the center lateral (that goes at 90 degrees to the river, see Fig. 3) at a distance of 15 m from the caisson is later presented for some scenarios in Table 8.

The effect of cyclic pumping of the vertical well was also considered. The average daily pumping rate for simulation remained at 0.0126 m^3/s for all steps of the flood hydrographs with the exception of step 3. In step 3, well 3 was pumped at 0.032 m^3/s for 9.6 h daily (and no pumpage took place for the rest of the day) for the first 4 days of this 14-day flood stage. After that the pumping rate was set to 0.0126 m^3/s for the remainder 10 days of the time step.

For each combination of the above conditions, simulations were run for 3 or 4 scenarios for varying sorption and decay coefficients (Tables 7 and 8).

2.4.3. Initial conditions

The initial condition for flow simulation was a hydraulic head distribution from a steady-state simulation when the river stage was 134 m MSL and the western boundary was 135 m and well 3 was pumping at a rate of 0.0126 m^3/s. Hydraulic head produced from this simulation was used as the starting heads for transient simulation. For transport simulation, a solute free initial condition was considered for atrazine simulation. For nitrate, two initial conditions were considered: (a) current nitrate status

Table 7
Summary of simulations for nitrate and atrazine in the vertical wells

Bank conductivity	A						B			C			D						E					
Bed conductivity	A			B			B			C			A			B			A			B		
Peak concentration at	SP-1	W3	SP-3	SP-1	W3	SP-3	SP-1	W3	SP-3	SP-1	W3	SP-3	SP-1	W3	SP-3	SP-1	W3	SP-3	SP-1	W3	SP-3	SP-1	W3	SP-3
(a) Simulated nitrate concentration (in mg/l)																								
Case 1[a]	9.6	9.8	9.7	9.5	9.8	9.7	9.5	9.7	9.7	9.5	9.7	9.7	8.8	9.3	9.7	8.8	9.3	9.7	8.3	9.1	9.4	8.3	9.1	9.4
Case 2[b]	9.8	10.0	9.8	9.8	10.0	9.9	9.6	9.9	9.8	9.5	9.8	9.9	8.9	9.4	9.8	8.9	9.4	9.8	8.4	9.2	9.5	8.4	9.2	9.5
Case 3[c]	2.9	2.9	2.9	3.0	2.9	2.9	5.9	3.4	2.9	8.3	4.2	2.9	9.5	6.8	3.2	9.4	6.7	3.2	9.6	7.5	4.4	9.6	7.5	4.4
Case 4[d]	3.0	3.0	3.0	3.0	3.0	3.0	6.0	3.5	3.0	8.3	4.2	3.0	9.5	6.9	3.3	9.5	6.8	3.3	9.6	7.6	4.5	9.6	7.5	4.5
(b) Simulated atrazine concentration (in μg/l)																								
Case 1[e]	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Case 2[f]	BDL	BDL	BDL	BDL	BDL	BDL	0.71	BDL	BDL	2.3	0.44	BDL	3.3	1.6	BDL	3.2	1.6	BDL	3.4	2.0	0.57	2.4	2.0	0.56
Case 3[g]	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Case 4[h]	BDL	BDL	BDL	0.11	BDL	BDL	1.8	0.31	BDL	3.2	0.74	BDL	3.8	2.2	0.17	3.8	2.2	0.15	3.8	2.7	0.91	3.8	2.6	0.9

The following hydraulic conductivity values apply to the following: $A = 5 \times 10^{-8}$ m/s for all stages of the river; $B = 5 \times 10^{-8}$, 5×10^{-7}, 5×10^{-6}, 5×10^{-7}, and 5×10^{-8} m/s for the five flood stages, respectively; $C = 1.27 \times 10^{-6}$, 1.27×10^{-6}, 1.27×10^{-5}, 1.27×10^{-6}, and 1.27×10^{-7} for the five flood stages, respectively; $D = 1.27 \times 10^{-5}$, 1.27×10^{-5}, 1.27×10^{-4}, 1.27×10^{-5}, and 1.27×10^{-6} for the five flood stages, respectively; $E = 1.27 \times 10^{-3}$ m/s for all stages; $F = 5 \times 10^{-8}$, 2.5×10^{-7}, 5×10^{-7}, 2.5×10^{-7}, and 5×10^{-8} m/s for the five flood stages, respectively; W3 = pumping well No. 3 of the town of Henry; BDL = below detection limit.

[a] Case 1: initial nitrate concentration (9.8–10.4 mg/l) with half-life of 2 years.

[b] Case 2: initial nitrate concentration (9.8–10.4 mg/l) without decay.

[c] Case 3: lower initial nitrate concentration (3 mg/l) with half-life of 2 years.

[d] Case 4: lower initial nitrate concentration (3 mg/l) without decay.

[e] Case 1: equilibrium sorption and decay.

[f] Case 2: decay without sorption.

[g] Case 3: equilibrium sorption but no decay.

[h] Case 4: no sorption and no decay.

Table 8
Summary of simulated concentrations of nitrate and atrazine in the collector well

(a) Nitrate in collector wells (mg/l)

Bank conductivity	A			E			Pumping distribution (m^3/s)
Bed Conductivity	A			A			
Peak concentration at	SP-1	Caisson	SP-3	SP-1	Caisson	SP-3	
Case 1[a]	9.9	9.9	9.8	8.0	5.4	9.3	4 × 0.02185
Case 2[b]	2.9	2.9	2.9	10	9.3	2.0	4 × 0.02185
Case 3[c]				10	9.8	3.4	1 × 0.0875

(b) Pesticides in collector wells (μg/l)

Bank conductivity	A			F			B			E			Pumping distribution (m^3/s)
Bed Conductivity	A			F			B			A			
Peak concentrations at	SP-1	Caisson	SP-3	SP-1	Caisson	SP-3	SP-1	Caisson	SP-3	SP-1	Caisson	SP-3	
Case 1[d,e]	BDL	BDL	BDL	BDL	0.115	BDL	BDL	0.75[f]	BDL	3.76	3.18	BDL	4 × 0.02185
Case 2[g]	BDL	BDL	BDL	BDL	0.203	BDL	BDL	1.16[h]	BDL				4 × 0.02185
Case 3[i]							BDL	BDL[j]	BDL				1 × 0.175
Case 4[k]							BDL	BDL[l]	BDL	3.99	3.75	BDL	1 × 0.175

The following symbols apply to all: A = 5×10^{-8}m/s for all stages of the river; B = 5×10^{-8}, 5×10^{-7}, 5×10^{-6}, 5×10^{-7}, and 5×10^{-8} m/s for the five flood stages, respectively; E = 1.27×10^{-3} m/s for all stages; F = 5×10^{-8}, 2.5×10^{-7}, 5×10^{-7}, 2.5×10^{-7}, and 5×10^{-8} m/s for the five flood stages, respectively.

[a] Case 1: Current nitrate content with decay (reported values are low values).

[b] Case 2: Low (3 mg/l) with decay.

[c] Case 3: Same as Case 2 with localized pumping (0.0875 m^3/s).

[d] Case 1: with sorption and decay.

[e] For sets A–A and E–A, atrazine degradation half-life was 7.5 weeks and all others used a half-life of 15 weeks.

[f] Radial (1.26 ppb).

[g] Case 2: no sorption and no decay.

[h] Radial (1 86 ppb).

[i] Case 3: equilibrium sorption with decay—higher pumpage.

[j] Radial (0.073 ppb).

[k] Case 4: no sorption and no decay—higher pumpage.

[l] Radial (0.238 ppb).

(9.8–10.4 mg/l as N) and (b) a lowered (3 mg/l as N) nitrate representing potential conditions after the new sewage collection system has been operational for several years.

3. Results

The results of analysis, presented below, show the impact of flooding on filtrate water quality by taking into consideration the following aspects:

3.1. Model calibration

The hydraulic heads simulated from a steady-state simulation were compared against the hydraulic heads at several observation wells when the river pool was low (134.4 m) and the western boundary was set to 135 m. The pumping was simulated at 0.0126 m^3/s and an average recharge value was used. Observed water levels at SP-2, SP-3, and CP1 (see Fig. 2A for location) matched to the model data within 0.3 m. It was decided not to change hydraulic conductivity values of the individual layers. For transport simulation, the river water nitrate concentration was set to 5 mg/l (average concentration during low flow). The nitrate content at the western edge of the model was based upon the measurements at selected observation wells in this area. Initial concentrations were those observed from single and multi-port wells. Based upon steady-state pumpage and a denitrification half-life of 2 years, the model concentration in most single-port observation wells matched within 10%. The differences for the multi-port wells were within 20% of the observed data. These vertical variations are likely caused by local heterogeneities, variations in denitrification rates, dispersion scale effects, and the lack of steady-state conditions. The greatest errors occurred in the top ports of MP-4 and MP-5. No attempts were made to change the dispersivity for a better fit. For heterogeneous and non-steady systems, such differences are not unreasonable.

A calibration of the model to transient flow in the river was done using water level data for SP-1 and multi-port wells MP-3 and MP-5. Simulation was carried out for the spring flood of 1997 (April through July). Fig. 4 shows the simulated head values along with the measured values. For the most part, the heads were within 0.3 m of each other (similar to the steady-state run). The deviation for MP-3 is slightly higher than SP-1. It should be pointed out that MP-3 is quite close to the pumping wells. The pumping wells are turned off or on based upon system pressure and the pumps operate at 0.0315 m^3/s compared to calibration runs which were made for an average pumpage of 0.0126 m^3/s.

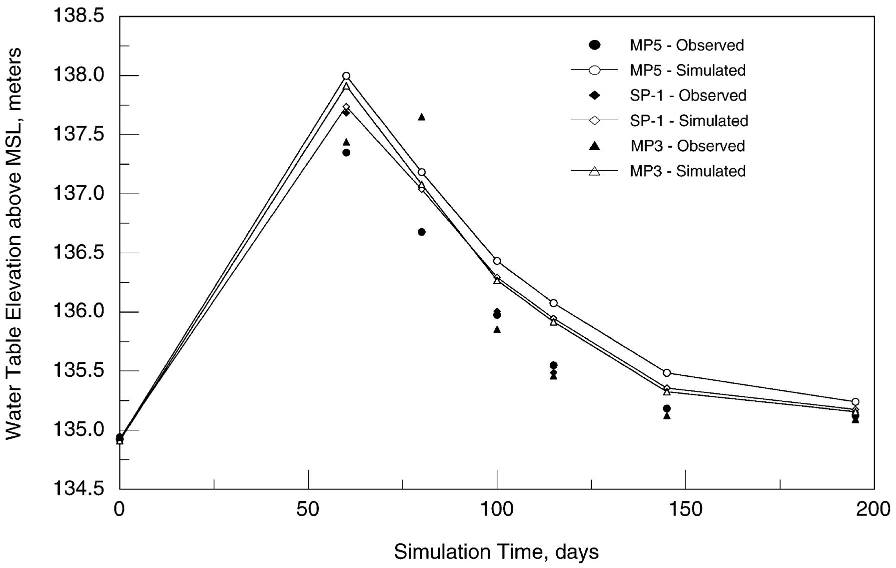


Fig. 4. Observed and simulated water table elevations for three locations at the Henry well site.

3.2. Impact of riverbed hydraulic conductivity on filtrate quality

3.2.1. Nitrate

For the vertical and collector wells, the impact of the changes in riverbed and bank hydraulic conductivity on nitrate content is presented in Tables 7 and 8, respectively, and these concentrations refer to calculated concentrations at the observation points. For high initial nitrate concentration (cases 1 and 2 in Table 7 and cases 1 in Table 8), some dilution was observable since the ground water concentrations were higher than river concentrations. For high initial concentrations of nitrate, the reported concentrations for the well, SP-1 and SP-3 are low values. These low values were generally observed after 2 weeks of simulation before the flooding began in the river. By day 14, the concentration of nitrate in the river was still 4 mg/l as N and it was much lower than the initial values in the aquifer. For the vertical wells (Table 7), comparison of bed conductivity *A* and *B* within bank conductivity *E* showed that for a large riverbank conductivity the concentrations at the pumped well and observation well SP-1 were significantly affected by the river water. This was irrespective of the case whether the riverbed conductivity remained fixed at its low value (combination of bank and bed conductivity *E*–*A*) or varied over two-log orders over the 5-stage flood simulation (combination set E-B). The peak concentrations of nitrate for cases 1 and 2 reflect the initial conditions at these observation points (SP-1, well 3 and SP-3). The reported concentrations in the table are lowest recorded concentrations prior to the rise in river stage and these values do not show significant variations between sets *E*–*A* and *E*–*B* (Table 7). This is interesting to observe that the flood has really no impact on nitrate content since the background nitrate level is higher than that in the floodwater. If the background concentration is reduced to 3 mg/l (cases 3 and 4), the peak concentration in the filtrate and at SP-1 can easily reach 9.5 mg/l, nearly the maximum observed concentration of the floodwater. It is apparent that the coarse material on the bank provides direct connection to the aquifer.

The bank materials were also assigned intermediate conductivity values (*C* and *D*), but allowed to vary with the flood stage (Table 7). This was considered a possibility for less permeable banks where floodwaters can wash off fine materials during high flows, thus enhancing the hydraulic conductivity. The bed conductivity was either fixed at the lowest value (*A*) or allowed to vary by two log orders (*B* or *C*). For the high initial background nitrate concentrations (cases 1 and 2), the minimum simulated concentration at SP-1 was slightly higher than the previous cases (sets *E*–*A* and *E*–*B*), thus showing the effect of lower dilution by floodwater. For an initial nitrate concentration of 3 mg/l as N (cases 3 and 4), the peak nitrate–N at SP-1 varied between 8.3 and 9.5 mg/l. For a higher conductive bank, the concentrations were higher. At the pumping well, these concentrations ranged between 4.2 and 6.8 mg/l, indicating the dissipation of flood impact (possibly due to dilution along the travel path).

When the hydraulic conductivity of the bank material was further reduced (set *B*) to the level of riverbed material and both were allowed to vary by two-log orders over the flood cycle, there was slight impact at SP-1, but negligible impact at the well 3. In the last simulation scenario, the bank conductivity was set at the lowest value (*A*) and the bed conductivity varied in two ways (either *A* or *B*). This is expected in cases where the flow velocity is low over a silted bank, but the bed sediment can wash off during high flow. The results indicated negligible impact on the wells (sets *A*–*A* or *A*–*B*).

For the collector well, a smaller number of nitrate simulations were conducted because of the excessive computer time needed to conduct these simulations. Overall, it was observed (Table 8) that when the bank and the bed are silted and remain the same during high flow (set *A*–*A*), the entry of the river water to the well was minimum. However, for sandy banks, there was significant influx of river water, even if the conductivity of the riverbed was low (set *E*–*A*).

3.2.2. Atrazine

For atrazine transport simulation with equilibrium sorption and literature reported half-life values, similar results were observed. For high conductive banks with the vertical well (set *E*), low levels of atrazine (still below a detection level of 0.1 μg/l) may appear at SP-1 and well 3. Without reaction, these concentrations can be significant. However, for the collector well, these concentrations can be 90–75% of river water concentration of 4 μg/l at these two locations, respectively. If both the bank and the bed conductivity were low (set $A-A$), the expected concentration at the collector well would be less than 0.02 μg/l (below detection level). If both the bank and the bed conductivity are allowed to vary by one or two log orders over the flood cycle (sets $F-F$ and $B-B$), the collector well would contain atrazine anywhere from 0.12 to 0.75 μg/l. It shows that small breakthrough of atrazine or other chemicals at collector wells is possible if the fine sediments from bank or bed get washed off.

3.3. Effect of well type and pumping rate

The pumping rates of the vertical well changed from a continuous rate to a pulse rate (as described earlier) in the third stage of the flood. This did not affect the concentration profile. In a different simulation, the pumping rate of the vertical well was increased by a factor of 3.5–0.044 m^3/s. The bank and the bed conductivity values were low and the initial background concentration of nitrate–N was 3 mg/l and (similar to set $A-A$ case 3). The simulated peak concentration of nitrate–N at well 3 was 6.9 mg/l, which was more than double the concentration of 2.9 mg/l when the well was pumping at 0.0126 m^3/s.

Pumping difference between the vertical and the higher-capacity collector wells was significant. The pumping rate for the collector well was 0.0875 m^3/s. For atrazine simulations, when the pumping rate was doubled to 0.175 m^3/s (see sets $B-B$ and $E-A$ in Table 8), the concentration in the collector well increased, depending upon the location of observation point. For the vertical well, the concentration of atrazine was below the detection limit.

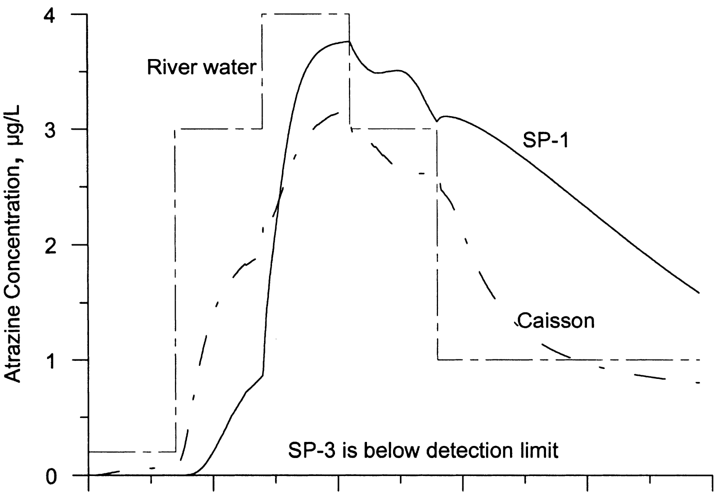
3.4. Effect of chemical reaction on atrazine transport

It is natural to expect the concentration of a non-reactive chemical be higher at the pumping well than a reactive chemical that has the same concentration in the river. Atrazine transport simulated for the vertical well (Table 7) shows four extremes where the decay rates and the organic carbon partition coefficients varied from zero to their literature reported values. Examining the calculated concentrations, it was found that including sorption and degradation as chemical reactions significantly decreased the concentration at observation points and well 3 (although all are well below the normal detection limit of 0.1 μg/l for atrazine). For the collector well simulation with a moderate conductivity bank and the riverbed (sets $B-B$ or $F-F$), the inclusion of sorption and degradation effects reduced the peak concentrations at the collector well by a factor of 2 and 7 depending upon whether it is in the well on in the radial. For high conductive banks, this ratio was small (1.1–1.2). However, the concentrations ranged from 3.18 to 3.99 μg/l.

3.5. Time lag in transport simulation

The effect of bank and bed conductivity values appeared to have some impact on travel time of the contaminant from the river water to the well(s). For example, as the bed and bank conductivity values were lowered, the rate of leakage reduced. This resulted in shifting of peaks of contaminants between the river water and the filtrate. For this reason, selected simulations were carried out for four additional weeks (98 days) at the same pumping rate of 0.0875 m^3/s. The flow and transport boundary conditions for the additional 4 weeks were kept similar to that around day 70. For a highly conductive bank with low conductivity bed (set $E-A$ in Table 8), the peak concentration of atrazine at the collector well was realized around day 42 for a case involving sorption and degradation (Fig. 5A). In river water, the peak value was reached on day 28 and remained at that level till day 42 and then dropped. For the case without equilibrium sorption and decay, the concentration peak was realized around the same time, but at higher concentrations (Fig. 5B). For SP-1, the response is similar with the exception that the effect of the river peak is more pronounced for the non-reactive case and there is a second rising limb around day 60. It must be pointed out that SP-1 is very close to the river and the water level in the river drops significantly on day 56. This causes the bank-stored water to drain to the river. At this time, the river water concentration of atrazine is low compared to the bank-storage concentrations and this results in slightly higher concentrations of atrazine at SP-1 until most of the bank-stored water drains out. Although the second peak is still visible

(A) with sorption and decay



B) without sorption and decay

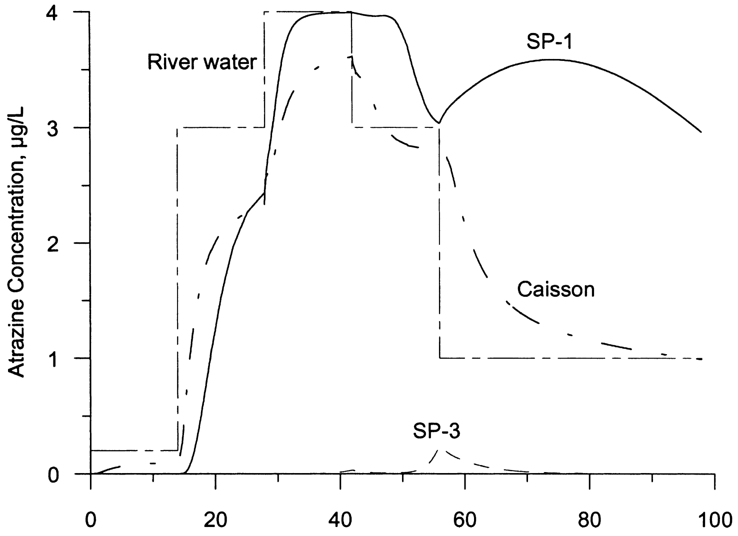


Fig. 5. Time lag between surface and ground water atrazine peaks for the collector well with highly conductive riverbank and low conductive bed (set *E–A*) with and without sorption and decay.

for SP-1 in Fig. 5A, it is much lower than non-reactive case (Fig. 5B) due to sorption and decay. The effect of river atrazine at SP-3 appears to be quite small as shown in Fig. 5. Ordinarily, one would not expect to see a peak at SP-3 when the bed or bank conductivity values are low. Since this simulation involves a highly conductive bank in contact with the aquifer (set $E-A$), bank-stored floodwater propagates beyond the pumping center. Higher than original heads were also observed at SP-3 in flow simulations.

For moderately low conductive bank and riverbed (set $B-B$) at a pumping rate of $0.0875\ \mathrm{m}^3/\mathrm{s}$, the peaks for non-reactive and reactive simulations were around day 38 and 40, respectively (Fig. 6) and peaks were much sharper. When the conductivity of the bed and bank were reduced to their lowest values and kept at that level (set $A-A$), the peaks for non-reactive and reactive transport simulations at the well were around day 55 and 65, respectively. However, the simulated concentrations were quite low in the well. This translated to about 3 weeks of time lag for reactive transport simulation. It is clear that low conductive banks and riverbed would produce a longer time lag between the river water and pumped ground water peaks. If the contaminant is reactive, the lag is further enhanced due to retardation effects of the contaminant. The peak is more attenuated for reactive contaminants than non-reactive ones.

3.6. Flooding effect on aquifer velocity profiles

River stage affected the velocity profile in the aquifer when the collector well was pumping at $0.0875\ \mathrm{m}^3/\mathrm{s}$. The velocity plots are one way to see the exchange of water between the river and the aquifer. For the bank and bed conductivity set $B-B$ (moderately conductive bed and bank), the river stage affected the direction of water flow to the well. For days 14 and 28, a large fraction of the water pumped from the well was coming from the aquifer as shown by the velocity profile. However, for high river stage (day 42), the contribution of river water towards the aquifer increased as indicated by the magnitude and direction of the velocity vectors. Fig. 7 shows the velocity plots for the top layer for days 14, 28, 42, and 70. The velocity variations among the layers were not significantly different except for the region in vicinity of the caisson in the sixth layer (that contains the laterals).

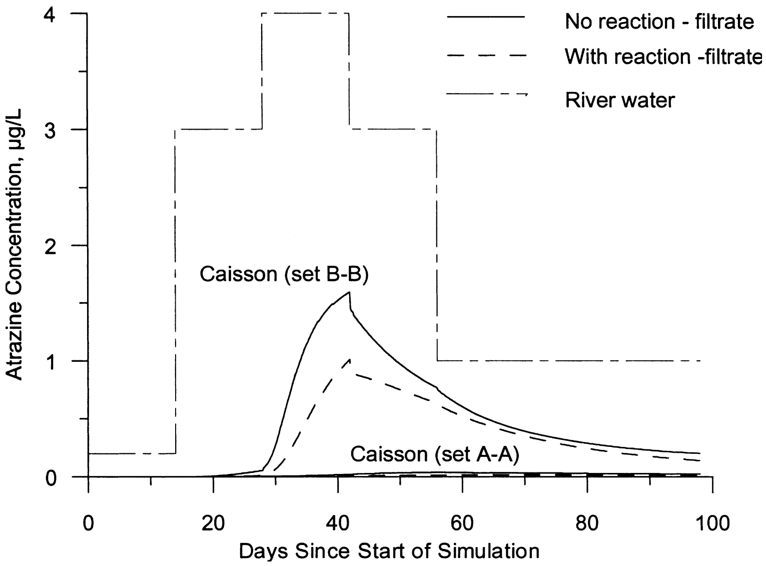


Fig. 6. Time lag between surface and ground water atrazine peaks for the collector wells with moderately conductive bed and bank material (set $B-B$).

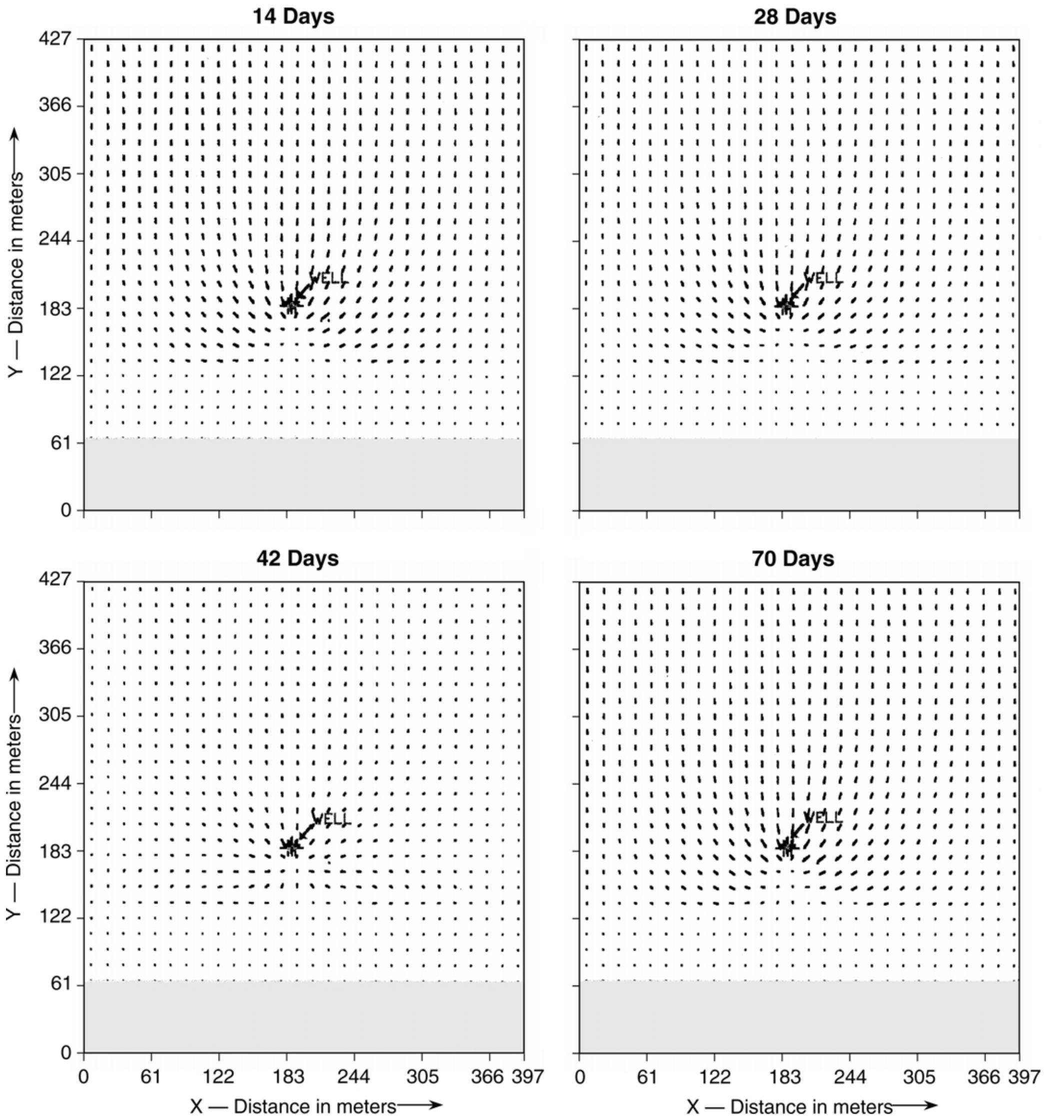


Fig. 7. Velocity profiles for the model grid for set $B-B$ for at simulations periods of 14, 28, 42, and 70 days.

4. Remediation strategies

Especially for scenarios with highly conductive riverbanks, it is probable that the pumping well will contain higher concentrations of contaminant than those with silted banks. A highly conductive bank is often encountered in a fast moving shallow river and a silted bank in slow moving deep rivers or rivers with navigation locks and dams. For the high conductive banks, two scenarios were considered. First, it was desired to create a hydraulic divide between the river and the well by placing additional pumping wells. This could retard the migration of contaminants towards the pumping well. In the second option, back up wells, further upstream from the pumping well were considered and these backup wells were pumped during flood periods. In each of these cases, it is assumed that the background nitrate concentration in the aquifer is low (3 mg/l as N) and the shock load moved through the river during floods. The concentration of nitrate in the river varied as shown in Table 6 for the passage of flood hydrograph. The bank and bed hydraulic conductivity values used for modeling are in case 1 of set E–A as shown in Table 7. It must be noted that set E is the worst-case condition for the permeable bank.

4.1. Hydraulic containment

For hydraulic containment, the simulation included a hypothetical well approximately 36 m from well 3 towards the river. This hypothetical well was screened from 129 to 120 m MSL and pumped at 0.0032 m^3/s. Such wells are often used for hydraulic containment at contamination sites and are termed as 'scavenger' wells. It was observed that the maximum concentration of nitrate–N at well 3, with the scavenger well pumping, would be about 7.7 mg/l. The corresponding concentrations at SP-1 and at the scavenger well would be 9.8 and 9.3 mg/l, respectively. However, without the scavenger wells, the peak concentration at well 3 would have been around 7.45 mg/l as N. When the pumping rate from the scavenger well was increased to 0.0063 m^3/s, the peak concentration at well 3 was 7.95 mg/l as N and the corresponding peaks at SP-1 and the scavenger wells were 9.7 and 9.4 mg/l, respectively. This rise in concentration at well 3 could be due to an enhanced movement of the river water towards the aquifer through the high conductive bank as a result of enhanced pumpage. Further, this indicates that enhanced pumping did not cause any hydraulic barrier.

In order to improve the effectiveness of the hydraulic barrier, another simulation was carried out in which well 4 was pumped (which is further away from the river). A set of 3 scavenger wells were simulated by locating them 21 m apart with the center well 30 m away from well 4. In this scenario, well 3 was not pumped and each scavenger well was pumped at 0.00315 m^3/s. The peak concentration at well 4 was 8.45 mg/l as N. The peak concentrations for each of the scavenger wells were close to 9.5 mg/l as N. It is unlikely that three scavenger wells will be used for protecting a well that pumps an average 0.012 m^3/s. To be effective, the screen zones of the scavenger wells should be located along the flow path of the contaminants from the river to the well screen. For the above case involving three scavenger wells (each 0.00315 m^3/s), all the three wells were screened for most part of the aquifer (from 134 to 120 m). It was found that the peak concentration of nitrate in well 4 would be 8.4 mg/l as N. It is apparent that remediation with scavenger well(s) is not a feasible for high conductivity bank. With well 4 pumping alone, the expected maximum concentration for this well would be 7.8 mg/l as N. This clearly indicates that at the current setting, scavenger wells would not be helpful in remediation efforts.

4.2. Well relocation

It is often thought that the bank filtration wells can be more effective if they are somewhat away from the rivers. Many European communities, with vertical wells, have kept them away (a few hundred meters more) from the riverbank if the aquifer is wide and land is available. This distance allows for the consumption of dissolved oxygen for the degradation of DOC. Once the oxygen is depleted, the anaerobic condition promotes denitrification. In the present case, only dispersion and simple first-order degradation are considered. In the model area, the aquifer is wide. However, moving the well farther from the river will be too close to the commercial district. With proper planning such a move is feasible. For simulation, a backup well, similar to well 3 was placed 100 m upgradient (westward) of the present well location. This well was allowed to pump at the same rate as well 3. During this simulation, well 3 was not pumped and was used as an observation well. The peak concentration of nitrate–N at the new well was 3.01 mg/l and that at an observation point near well 3 was about 7 mg/l. Doubling the pumping rate from this backup well increased the nitrate–N concentration from a background level of 3 to 3.5 mg/l. This increased the nitrate–N at the observation well near well 3 to around 7.8 mg/l. It is obvious that moving the well farther away from the bank would help retard the migration of contaminants.

5. Summary and conclusions

It was observed that good hydraulic connection and the close proximity of vertical wells to rivers can allow the movement of river water contaminants to the wells. The hydraulic connection between the river and the aquifer at the study site was assumed to be through the high conductivity sand that was exposed on the riverbank. In simulations, this caused most of the river water to enter the aquifer. Even under non-pumping conditions, a large mass of the river water contaminants could enter the aquifer and get stored in the aquifer. The bank-stored water slowly drains as the floodwaters recede (Fig. 5).

Pumping the water-supply well in cyclic mode did not seem to have any significant impact on reducing the concentrations of the contaminants. It was also observed that by placing one or more ‘scavenger’ wells between the water supply well and river and pumping these scavenger wells from 0.00315 to 0.0063 m^3/s each did not reduce the nitrate content of the pumping well. However, the best remedy appeared to be the pumping of a backup well another 100 m upstream of well 3.

Modeling exercises with the vertical well indicated that the atrazine concentration of water from the well would be below detection limit for cases with reactions. Atrazine in observation wells between the river and the pumping well would be below detection and the monitoring results supported these findings. However, for a large capacity collector well at this location, it is possible to have a breakthrough of atrazine depending upon bed and bank hydraulic conductivity. Even assuming the chemical to undergo equilibrium sorption and normal decay, small breakthroughs are possible for collector wells. This is probably one of the reasons why a small breakthrough at the Jacksonville’s collector well was observed (Fig. 1).

While the model data showed the possibility of a breakthrough at the pumping well for highly conductive bank without sorption and degradation, atrazine was never detected at the pumping well. Simulations involving equilibrium sorption and literature reported decay coefficients showed that atrazine content at the pumping well should be below detection level. However, further studies at the site are needed to gain a better understanding of the reaction pathways and transformation of organic and inorganic contaminants during the passage of river water in the aquifer.

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