Hydrochemical evolution of pore water in riverbed sedimentation zone during riverbank infiltration

Shuai Lu, Shenjie Li, Zhining Liu, Xinyue Gao, Lihua Zhang and Chao Sun

ABSTRACT

The riverbed sedimentation zone is an important zone of hydrochemistry, and the biogeochemical action in this zone has a significant impact on groundwater quality. As the main area where hydrochemistry occurs, studying the law of hydrochemical evolution within 1 m below the riverbed is of great significance for understanding the migration and removal of river pollutants. In this study, a combination of onsite monitoring and indoor experiments was used to analyze the variation characteristics of the hydrochemical composition of pore water during riverbank infiltration, as well as the main hydrochemical effects and influencing factors. The results show that in the process of river water infiltration, a series of redox reactions occur in the riverbed sedimentation zone, and there are differences in different infiltration depths. From 0 to 20 cm below the riverbed, strong respiration and denitrification mainly occurred. Reductive dissolution of manganese minerals mainly occurred from 20 to 60 cm, and reductive dissolution of iron minerals mainly occurred from 60 to 90 cm. River water level, dissolved organic carbon content and microbial activity had varying degrees of influence on these redox effects. The recharge of river water infiltration ensures the exploitation amount of the pumping wells, but it also leads to the increase of some components in groundwater, and the extracted water cannot be directly drunk.

Key words | groundwater source, hydrochemical evolution, influencing factors, riverbank infiltration, riverbed sedimentation zone

HIGHLIGHTS

- The main control effect of hydrochemical evolution in the riverbed sedimentation zone is redox, and there are differences in different infiltration depths.
- The redox effects were affected by river water level, dissolved organic carbon content and microbial activity.
- The recharge of river water infiltration ensures the exploitation amount of the pumping wells, but it also leads to the increase of some components in groundwater.
INTRODUCTION

As an important source of water supply, groundwater plays an important role in ensuring residents’ lives, industrial and agricultural production, and ecological environment construction (Sophocleous 2002; Bekkoussa et al. 2018; Tsanov et al. 2020). Riverside mining is an important way of groundwater mining, which can stimulate groundwater recharge. At the same time, intense and complex biogeochemical processes occur in the process of river water infiltration (Tufenkji et al. 2002; Hamann et al. 2016; Zhu et al. 2020). Due to the different physical, chemical and biological properties of river water and groundwater, obvious physical, chemical and biological gradients are often generated during riverbank infiltration, and intense and complex biogeochemical processes occur, which have an obvious impact on the quality of groundwater (Massmann et al. 2008a; Hu et al. 2016; Zhu et al. 2019). On the one hand, the viruses, microorganisms, bacteria, organic matter and heavy metals in the river are fixed and degraded, which can purify the water quality to a certain extent (Huntscha et al. 2013; Albergaro et al. 2019; Cartwright & Irivine 2020). On the other hand, it may lead to the release of toxic and harmful substances (such as arsenic, cadmium and chromium) in medium into groundwater, thus causing water pollution (Farnsworth & Hering 2011; Ahmed & Marhaba 2017; Postma et al. 2017).

The geochemical process during riverbank infiltration plays an important role in maintaining groundwater quality and ecological safety, and has become a research hotspot in many scientific fields, such as hydrology, hydrogeology and hydroecology (Brunke & Gonser 1997; Fazlalabolabadi & Golsten 2020; Jylhä-Ollila et al. 2020). As the first zone of interaction between river water and groundwater, the reactions occurring in the riverbed sedimentation zone include physical filtration, biodegradation, adsorption, chemical precipitation and redox, among which redox is the current research focus, and hydrodynamic conditions and organic matter content are two key factors for the hydrochemical evolution (Hiscock & Grischek 2002; Petrunic et al. 2005; Kim et al. 2014). Studying the hydrochemical evolution law in the riverbed sedimentation zone can better reveal the interaction mechanism between river water, groundwater and riverbed sediments, reveal a series of reactions during riverbank infiltration, and then find out which reactions are beneficial to improve groundwater quality, and which reactions degrade groundwater quality. It is very important to investigate hydrochemical evolution to improve understanding of the migration of river pollutants and their removal.

Previous studies have shown that a series of biogeochemical effects occur in the process of river water infiltration (Kedziorek & Bourg 2009; Hu et al. 2016; Lu 2018). Due to the existence of redox zoning, it has a direct or indirect impact on the migration and transformation of pollutants, and most studies have focused on the effects occurring on a regional scale (Massmann et al. 2008b; Houria et al. 2020; Lu et al. 2020). Some researchers have found that the water quality of the infiltrated river usually changes significantly in the first few meters of the seepage path, and within 1 m below the riverbed is the main zone where hydrochemistry occurs (Doussan et al. 1997; Sophocleous 2002; Burke et al. 2014). However, until now there have been few detailed studies on how the hydrochemical composition changes on this scale, and how hydrodynamic conditions and other environmental factors affect the redox zoning. Therefore, this study addressed this knowledge gap by analyzing the hydrochemical composition within 1 m below the riverbed. The results will be of great value for management plans to ensure the sustainability of drinking-water-production systems (Guiamel & Lee 2020).

Shenyang Huangjia groundwater source area is a primary low-quality high-iron–manganese groundwater area, which is prone to reductive dissolution in the nearshore zone and oxidation precipitation near the pumping wells, causing water pollution and blockage of the pumping wells, resulting in a series of problems such as reduced mining efficiency. The reductive dissolution of iron–manganese minerals will cause the release of toxic substances adsorbed on the surface and lattice of the iron–manganese minerals, and the redox effect in the riverbed sedimentation zone has a great impact on the groundwater quality (Lu et al. 2020). However, there are few studies and reports on the redox
effects and influencing factors in the primary low-quality high-iron–manganese groundwater areas. As an important zone of hydrochemistry, it is necessary to carry out related research on the riverbed sedimentation zone.

In this study, on the basis of onsite monitoring and indoor column simulation experiments, study on hydrochemical evolution of pore water in the riverbed sedimentation zone had been carried out, and the characteristics of the redox process during riverbank infiltration were analyzed, which will help to reveal the mechanism of biogeochemical evolution in the process of river water infiltration and provide a scientific basis for ensuring the safety of the groundwater source area.

MATERIALS AND METHODS

Study area description

The study area is located in the Huangjia groundwater source area, northern of Shenyang City (Figure 1). It belongs to the alluvial plain of the middle and lower reaches of Liao River, with a flat terrain and an altitude of 42–50 m. The study area is typical of temperate semi-humid monsoon climate. The mean annual temperature is 7 °C, and the mean annual precipitation is 635.5 mm. The main river in the study area is the Liao River, with a length of 3.6 km and a width of 100–300 m. The riverbed is composed of clay,
fine sand, medium coarse sand and coarse sand containing gravel. The riverbed in the area is basically stable with slight erosion.

The quaternary phreatic aquifer is the target layer for groundwater exploitation in the area. The aquifer is mainly composed of coarse sand, gravel and sand-gravel with a thickness of about 50 m. There are many iron and manganese minerals in the aquifer medium (Lu et al. 2020). There are 12 pumping wells in the area, which are arranged along the river, and the total amount of groundwater pumped is approximately 30,000 m³/d. The water level of the Liao River in the study area is always higher than the groundwater level, and the groundwater is always recharged by the river.

**Sampling and testing**

The Beeker portable sediment sampler (4 cm inner diameter) was used to collect soil samples at a distance of 3 m away from the south bank of Liao River. The sampling depth was within 90 cm below the riverbed, where the river water was 30 cm deep. The soil samples were cut at intervals of 10 cm in the anaerobic environment and sealed for preservation. After transporting them back to the laboratory, the samples were placed in 100 mL centrifuge tubes in the anaerobic environment and centrifuged for 10 min at 3,000 and 4,000 r/min, respectively, in a low-speed centrifuge. The supernatant was taken for anaerobic preservation.

When collecting river water samples, first wash the sampling bottles with the river water to be collected for more than three times, then place the sampling bottles 15 cm below the surface of the river water for collection. After the sampling, add corresponding protective agent according to the preservation requirements of different components, quickly seal it with sealing film and measure pH, oxidation-reduction potential (ORP) and dissolved oxygen (DO) on site.

A portable tester (HQ40d; Hach Company, USA) was used to test pH, ORP and DO. An ion chromatograph (881 Compact IC; Metrohm, Switzerland) was used for anion and cation analysis, and Fe²⁺ and Mn²⁺ were measured by the Hach DR2800 portable spectrophotometer. The dissolved organic carbon (DOC) was analyzed using a total organic carbon analyzer (TOC-L CPH CN200; Shimadzu, Japan). Particle analysis was performed using a sieve method.

**Experimental design**

The soil column experiment device was composed of four plexiglass columns with an inner diameter of 4 cm and a length of 70 cm. The bottom and top of the column were opened as water inlet and outlet, respectively. According to the principle of similarity ratio, the layering conditions and layer thickness of the riverbed sediments were scaled and reduced at a ratio of 1.3:1, and the columns were filled with the sediments collected in the field. From top to bottom, the column was composed of coarse sand containing gravel (10 cm), medium coarse sand (20 cm), fine sand (35 cm) and clay (5 cm). It was filled and compacted every 5 cm. In order to ensure uniform force, glass rod was used which was raised 5 cm and then fell freely 30 times. After the column was filled, ultra-pure water was used to saturate the soil for 2 days, so as to keep the DO and other indexes in the soil column stable and consistent.

According to the annual fluctuation of the water level of the Liao River measured on site, two hydrodynamic conditions were set, low water head and high water head. The outlet of the low-head bottle was 30 cm higher than the top of the column (the experiment column CI), and the high-head bottle was 60 cm (the experiment column CII). In these two cases, the seepage velocity of the water flow was 0.66 and 1.15 m/d, which were consistent with the annual data monitored in the study area (0.6–1.2 m/d) (Lu 2018).

According to the DOC annual monitoring data in the study area, the DOC content in the river water was 0.6–1.8 mmol/L (Lu 2018). The collected river water contained a certain amount of DOC (0.66 mmol/L), and adding sodium acetate can increase the content of DOC in the water. In the experiment, 41.0 mg sodium acetate was added to the 1 L river water sample (the experiment column CIII), which was equivalent to adding 12 mg C to 1 L water. The DOC content of the new water sample was 1.66 mmol/L, which can truly reflect the DOC content of river water at different periods.
The soil sample was sterilized in the oven for 4 h, and the oven temperature was set at 120 °C (the experiment column CIV). All four columns were covered with black plastic bags to avoid light.

The parameters selected for the comparative experiment are basically consistent with the data obtained from onsite monitoring in the study area during the year and can be used to study the influence of river water level, DOC content and microbial activity on the hydrochemical effects in the riverbed sedimentation zone. Four column simulation experiments are carried out at the same time, and the specific experimental design is shown in Table 1. Water samples were collected at the outlet every 24 h for testing.

RESULTS AND DISCUSSION

Hydrochemical characteristics of groundwater and river water

Hydrochemical facies

From the Piper diagram (Figure 2), we can see that the river is characterized by a Ca–Na–Cl–SO₄ type of water, and the groundwater is mainly characterized by a Ca–Na–Cl–SO₄–HCO₃ type of water, which the chemical composition of the water is similar for the river and the groundwater.

Table 1  | Experiment column control conditions

| Experiment column | Infiltration water | Whether the medium is sterilized | Water level (cm) |
|-------------------|-------------------|---------------------------------|-----------------|
| CI                | River water       | No                              | 30              |
| CII               | River water       | No                              | 60              |
| CIII              | River water (add sodium acetate) | No                             | 30              |
| CIV               | River water       | Yes                             | 30              |

Figure 2  | Piper diagram of water samples.
Hydrochemical indexes

**pH.** The pH of the river water was 8.83, and the pH of the pore water in the riverbed sedimentation zone varied from 7.34 to 7.70 (Table 2; Figure 3(a)), indicating that the aquatic acid action along the infiltration path reduced the pH during riverbank infiltration. As the depth increased, the overall pH showed a downward trend. From 0 to 20 cm below the riverbed, the pH dropped from 8.83 to 7.47, and after 20 cm below the riverbed, the pH was relatively stable. Therefore, the reaction in the shallow layer of the riverbed sedimentation zone greatly reduced the pH of the water, after which the pH changed relatively little.

**Oxidation–reduction potential.** The ORP value of river water was 145.4 mV, and the variation range of pore water was 53.9–86.5 mV (Figure 3(b)). With the increase of depth, the ORP value gradually decreased, reflecting that the relative oxidation environment changed to the relative reduction environment in the process of river water infiltration.

**Dissolved organic carbon.** DOC is the main nutrient for microbial activity in the riverbed sedimentation zone, and at the same time, it can be used as an electron donor for redox, which plays an important role in the hydrochemical evolution (Petrunic et al. 2005; Farnsworth & Hering 2011). The DOC content of river water was 0.66 mmol/L, and the variation range of pore water was 1.93–5.26 mmol/L, which was significantly higher than that of river water (Figure 3(c)). From 0 to 10 cm below the riverbed, the content rose from 0.66 to 5.26 mmol/L, reflecting that the infiltration river water dissolved the organic carbon in the sediment medium, resulting in higher organic carbon content in pore water than river water. As the depth increased, the overall content showed a downward trend, reflecting the continuous reaction of DOC as an electron donor during riverbank infiltration.

**Dissolved oxygen.** The DO content of river water (0.29 mmol/L) was higher than that in the pore water (0.034–0.11 mmol/L) (Figure 3(d)). With the increase of depth, the DO content gradually decreased, and it decreased significantly from 0 to 10 cm below the riverbed, reflecting the oxidation reaction of organic carbon with oxygen as the
electron acceptor during riverbank infiltration mainly occurred in the depth range of 0–10 cm below the riverbed, that is

\[ \text{CH}_2\text{O} + \text{O}_2 \rightarrow \text{CO}_2 + \text{H}_2\text{O} \]

\[ \text{NO}_3^- \rightarrow \text{NO}_3^- \text{.} \]  

The NO\textsubscript{3}\textsuperscript{-} content of river water was 0.28 mmol/L, and the variation range of pore water was 0–0.047 mmol/L (Figure 3(e)). As the depth increased, the NO\textsubscript{3}\textsuperscript{-} content decreased significantly, and at the depth of 20 cm, the NO\textsubscript{3}\textsuperscript{-} content rapidly decreased below the detection limit,
which reflected that from 0 to 20 cm below the riverbed, NO$_3^-$ as the electron acceptor was all reduced by denitrification, that is

$$5\text{CH}_2\text{O} + 4\text{NO}_3^- + 4\text{H}^+ \rightarrow 5\text{CO}_2 + 7\text{H}_2\text{O} + 2\text{N}_2$$

Mn$^{2+}$. The manganese content of river water was below the detection limit (Figure 3(f)). With the infiltration of river water, the manganese content first increased and then decreased. At the depth of 50 cm, the manganese content increased to 0.073 mmol/L, reflecting the occurrence of the manganese mineral dissolution reaction in the riverbed sedimentation zone, that is

$$\text{CH}_2\text{O} + 2\text{MnO}_2(s) + 4\text{H}^+ \rightarrow 2\text{Mn}^{2+} + 3\text{H}_2\text{O} + \text{CO}_2$$

Fe$^{2+}$. The iron content of river water was below the detection limit (Figure 3(g)). With the infiltration of river water, it started to rise at the depth of 60 cm, reflecting that the reduction dissolution reaction of manganese minerals was earlier than that of iron minerals, and the reduction dissolution of iron minerals occurred at a relatively deep position in the riverbed sedimentation zone, that is

$$\text{CH}_2\text{O} + 4\text{Fe(OH)}_3(s) + 8\text{H}^+ \rightarrow 4\text{Fe}^{2+} + 11\text{H}_2\text{O} + \text{CO}_2$$

**Other conventional ions.** The content of conventional ions in the riverbed sedimentation zone varies with the sampling depth, which often reflects the mixing, leaching, cation exchange, and mineral dissolution and precipitation in this zone. It is shown from Figure 5(h)–5(l) that from 0 to 10 cm below the riverbed, the content of other conventional ions in the pore water all increased, indicating that the river water had a leaching effect during the infiltration process, and some minerals had dissolved. The increase of Cl$^-$ content indicated that the chloride salt was dissolved, and $c(\text{Na}^+)/c(\text{Cl}^-) > 1$ (Figure 4(a)) indicated that not only the chloride salt was dissolved but also the silicate mineral was dissolved.

From 10 to 20 cm below the riverbed, the content of other conventional ions in the pore water all decreased. Cl$^-$ is the most stable ion in groundwater, which is not easily adsorbed by soil particles, and the solubility of chloride salt is relatively large. The decrease in Cl$^-$ content indicated that mixing had occurred. According to the lithological analysis of the riverbed sedimentation zone, in the shallow part of the riverbed, the sediments had small particle size, weak permeability, and the mixing intensity was weak.

From 20 to 90 cm below the riverbed, the Cl$^-$ and Na$^+$ contents in pore water almost unchanged, and $c(\text{Na}^+)/c(\text{Cl}^-) \approx 1$ (Figure 4(a)), indicating that Na$^+$ did not undergo cation exchange during riverbank infiltration. The contents of Ca$^{2+}$, Mg$^{2+}$, and SO$_4^{2-}$ in pore water showed similar changes with depth. The vertical distribution of SO$_4^{2-}$ content fluctuated and did not show a significant decrease, indicating that SO$_4^{2-}$ had not yet participated in the redox reaction as an electron acceptor within the range of infiltration. The contents of Ca$^{2+}$ and Mg$^{2+}$ were approximately positively correlated (Figure 4(b)), indicating that cation exchange of Ca$^{2+}$ and Mg$^{2+}$ did not occur during riverbank infiltration.
infiltration. Therefore, the variation trend of the conventional ion content from 20 to 90 cm below the riverbed was likely to be caused by mixing. The sediment particle size in this depth range was relatively large, the permeability was strong, and the mixing intensity was strong.

Main controlling effects in riverbed sedimentation zone

It can be seen from the variation characteristics of hydrochemical indexes that during riverbank infiltration, the redox reaction with DOC as the electron donor and O₂, NO₃⁻, MnO₂ and Fe(OH)₃ as the electron acceptors occurred in the riverbed sedimentation zone. From 0 to 90 cm below the riverbed, the main redox effects at different depths were different (Figure 5).

Affected by oxygen-enriched river water and redox reactions in the sedimentation zone, respiration and denitrification consumed a large amount of O₂ and NO₃⁻ in the river water from 0 to 20 cm below the riverbed, making the O₂ and NO₃⁻ contents in the pore water far less than that in the river water. From 20 to 60 cm below the riverbed, the manganese content gradually increased from below the detection limit and then decreased. The main redox effect in this area was the reductive dissolution of manganese minerals. From 60 to 90 cm below the riverbed, the iron content gradually increased from below the detection limit, while the manganese content gradually decreased. The main redox effect in this area was the reductive dissolution of iron minerals.

Influencing factors of hydrochemical evolution

River water level

The time point at which the DO content under different river water levels differed greatly was the second day after the start of the experiment, and the DO content under high river water level condition was relatively high (Figure 6(a)), indicating that the oxygen reduction rate was slower. Under low river water level condition, the DO content was only high on the first day and greatly reduced from the second day, and a large amount of DO in the column was reduced. Therefore, the river water level directly affected the rate of reduction of DO by organic carbon in the column. When the river water level was low, the water infiltration rate was relatively slow, and the water retention time was relatively long, the reaction time of reduction of DO by organic carbon was relatively sufficient, and the time required for the reduction of the same amount of DO was relatively short.

![Figure 5](http://iwaponline.com/aqua/article-pdf/doi/10.2166/aqua.2021.164/890652/jws2021164.pdf)

Figure 5 | Main controlling effects in the riverbed sedimentation zone during riverbank infiltration.
NO$_3^-$ had similar response characteristics as oxygen (Figure 6(b)). The NO$_3^-$ content was lower than the detection limit on the 14th day under high river water level condition and on the 10th day under low river water level condition. This indicated that the river water level affected the denitrification rate. Under low river water level condition, longer solute contact reaction time led to the consumption of more oxidants and the denitrification rate was faster.

Under low river water level condition, the iron–manganese minerals were reduced earlier, and the iron–manganese content in water was relatively high (Figure 6(c) and 6(d)), mainly because under this condition, the water flow velocity was slower, the water retention time in the column was longer, and the completion time of organic carbon reduction of other electron accepters was earlier, the iron–manganese minerals participated in the reaction earlier. At the same time, under low river water level condition, the reaction time of organic carbon and iron–manganese minerals was longer, resulting in higher iron–manganese content in the water.

**DOC content**

Under high DOC condition, the DO content in the column reached a low value on the first day, while under low DOC condition, the DO content reached a low value on the second day (Figure 7(a)). Therefore, other conditions being the same, the higher the DOC content, the more electrons it can provide, the more oxygen it can reduce, and the faster the environment transitions from the aerobic state to the anaerobic state.

Under high DOC condition, the NO$_3^-$ content was lower than the detection limit on the third day, while on the 10th day under low DOC condition (Figure 7(b)). The DOC content had a great impact on denitrification. Under high DOC condition, electron donors were sufficient, and NO$_3^-$ accepting electrons were reduced at a faster rate.

The iron and manganese contents under high DOC condition were higher than that under low DOC condition, and iron and manganese were detected on the second day of the
experiment under high DOC condition, while on the sixth day under low DOC condition (Figure 7(c) and 7(d)). Therefore, when other conditions were the same, the high DOC content was conducive to the reductive dissolution of iron and manganese minerals.

Microbial activity

The variation of DO content with time was almost the same under different microbial activity conditions (Figure 8(a)), indicating that only sterilization of the experimental medium had little effect on the reduction of oxygen by organic carbon. The NO$_3^-$ contents of both were lower than the detection limit on the 10th day (Figure 8(b)), indicating that soil sterilization had little influence on denitrification.

Iron and manganese were detected on the sixth day under unsterilized condition, while on the seventh day under sterilized condition, and the content was higher than that under sterilized condition (Figure 8(c) and 8(d)), indicating that microbial activity was an important factor in the reductive dissolution of iron and manganese minerals.

CONCLUSIONS

Affected by oxygen-enriched river water and redox in the riverbed sedimentation zone, strong respiration and denitrification occurred from 0 to 20 cm below the riverbed, which consumed a large amount of O$_2$ and NO$_3^-$ in the river. The reductive dissolution of manganese minerals mainly occurred from 20 to 60 cm, and reductive dissolution of iron minerals mainly occurred from 60 to 90 cm.

The water level directly affects the infiltration rate of river water and its residence time in the sedimentation zone. When the water level is high, the flow rate is faster, which reduces the residence time of the river water in the medium, thereby reducing the contact between the river water and the medium. The redox reaction time is insufficient, which slows down the consumption of DO and NO$_3^-$ in the water and reduces the reductive dissolution of iron and manganese.

As an important electron donor in hydrochemical evolution, high content of DOC is conducive to respiration and denitrification. The contents of DO and NO$_3^-$ in the water...
decrease rapidly. At the same time, it is conducive to the reductive dissolution of iron and manganese minerals, which will increase the contents of iron and manganese in water.

Whether the sediment medium is sterilized has little effect on respiration and denitrification. The reductive dissolution of iron and manganese minerals usually occurs with the participation of microorganisms, whose activity directly affects the contents of iron and manganese in pore water. After sterilization of the medium, the reductive dissolution of iron and manganese minerals occurs later, and the content in water is lower.

ACKNOWLEDGEMENTS

This work is supported by Scientific Research Initiation Funds for PhD Scholars of Hebei GEO University (BQ2019046), Teaching Development and Reform Practice Project of Hebei GEO University (2019)F03), Fund Project of Hebei Province Collaborative Innovation Center for Sustainable Utilization of Water Resources and Optimization of Industrial Structure, Basic Scientific Research Business Expenses of Colleges and Universities in Hebei Province (QN202107), Hebei Province Professional Degree Postgraduate Teaching Case Construction Project (KCJSZ2021098) and Open Topic of Hebei Key Laboratory of Geological Resources and Environment Monitoring and Protection (JCYKT202003). The authors thank the editors of AQUA and the reviewers for their thoughtful and constructive comments, which helped improve the manuscript.

DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

REFERENCES

Ahmed, A. K. A. & Marhaba, T. F. 2017 Review on river bank filtration as an in situ water treatment process. Clean Technologies & Environmental Policy 19, 349–359. https://doi.org/10.1007/s10098-016-1266-0.
Albergamo, V., Schollée, J. E., Schymanski, E. L., Helmus, R., Timmer, H., Hollender, J. & de Voogt, P. 2019 Nontarget screening reveals time trends of polar micropollutants in a riverbank filtration system. Environmental Science & Technology 53 (15), 7584–7594. https://doi.org/10.1021/acs.est.9b01750.

Bekkoussa, S., Bekkoussa, B., Taupin, J.-D., Patris, N. & Meddi, M. 2018 Groundwater hydrochemical characterization and quality assessment in the Ghriss Plain basin, northwest Algeria. Journal of Water Supply: Research and Technology – AQUA 67 (5), 458–466. https://doi.org/10.2166/aqua.2018.013.

Brunke, M. & Gonser, T. 1997 The ecological significance of exchange processes between rivers and groundwater. Freshwater Biology 37, 1–33. https://doi.org/10.1046/j.1365-2427.1997.00143.x.

Burke, V., Greskowiak, J., Asmuß, T., Bremermann, R., Taute, T. & Massmann, G. 2014 Temperature dependent redox zonation and attenuation of wastewater-derived organic micropollutants in the hyporheic zone. Science of the Total Environment 482–483, 53–61. https://doi.org/10.1016/j.scitotenv.2014.02.098.

Cartwright, I. & Irvine, D. 2020 The spatial extent and timescales of bank infiltration and return flows in an upland river system: implications for water quality and volumes. Science of the Total Environment 743, 140748. https://doi.org/10.1016/j.scitotenv.2020.140748.

Doussan, C., Poitevin, G., Ledoux, E. & Detay, M. 1997 River bank filtration: modelling of the changes in water chemistry with emphasis on nitrogen species. Journal of Contaminant Hydrology 25, 129–156. https://doi.org/10.1016/S0169-7722(96)00024-1.

Farnsworth, C. E. & Hering, J. G. 2011 Inorganic geochemistry and redox dynamics in bank filtration settings. Environmental Science & Technology 45 (12), 5079–5087. https://doi.org/10.1021/es2001612.

Fazelabдолabadi, B. & Golestani, M. H. 2020 Towards Bayesian quantification of permeability in micro-scale porous structures – the database of micro networks. HighTech and Innovation Journal 1 (4), 148–160. https://doi.org/10.28991/HiJ-2020-01-04-02.

Guaиmel, I. A. & Lee, H. S. 2020 Watershed modelling of the Mindanao River Basin in the Philippines using the SWAT for water resource management. Civil Engineering Journal 6 (4), 626–648. https://doi.org/10.28991/cj-2020-03091496.

Hamm, E., Stuyfzand, P. J., Greskowiak, J., Timmer, H. & Massmann, G. 2016 The fate of organic micropollutants during long-term/long-distance river bank filtration. Science of the Total Environment 545–546, 629–640. https://doi.org/10.1016/j.scitotenv.2015.12.057.

Hiscock, K. M. & Griscek, T. 2002 Attenuation of groundwater pollution by bank filtration. Journal of Hydrology 266, 139–144. https://doi.org/10.1016/S0022-1694(02)00158-0.

Houria, B., Mahdi, K. & Zohra, T. F. 2020 Hydrochemical characterisation of groundwater quality: Medja Plain (Tebessa Town, Algeria). Civil Engineering Journal 6 (2), 318–325. https://doi.org/10.28991/cj-2020-03091473.

Hu, B., Teng, Y., Zhai, Y., Zuo, R., Li, J. & Chen, H. 2016 Riverbank filtration in China: a review and perspective. Journal of Hydrology 541, 914–927. https://doi.org/10.1016/j.jhydrol.2016.08.004.

Huntscha, S., Velosa, D. M. R., Schrotth, M. H. & Hollender, J. 2013 Degradation of polar organic micropollutants during riverbank filtration: complementary results from spatiotemporal sampling and push-pull tests. Environmental Science & Technology 47, 11512–11521. https://doi.org/10.1021/es401802z.

Jylhä-Ollila, M., Laine-Kaulio, H., Niinikoski-Fusswinkel, P., Leveinen, J. & Koivusalo, H. 2020 Water quality changes and organic matter removal using natural bank infiltration at a boreal lake in Finland. Hydrogeology Journal 28, 1543–1557. https://doi.org/10.1007/s10040-020-02127-9.

Kedzior, M. A. M. & Bourg, A. C. M. 2009 Electron trapping capacity of dissolved oxygen and nitrate to evaluate Mn and Fe reductive dissolution in alluvial aquifers during riverbank filtration. Journal of Hydrology 365, 74–78. https://doi.org/10.1016/j.jhydrol.2008.11.020.

Kim, D. M., Yoon, S. T., Kwon, M. J., Mayer, B. & Kim, K. H. 2014 Assessing redox zones and seawater intrusion in a coastal aquifer in South Korea using hydrogeological, chemical and isotopic approaches. Chemical Geology 390, 119–134. https://doi.org/10.1016/j.chemgeo.2014.10.024.

Lu, S. 2018 Biogeochemical Process of Arsenic in Groundwater and Its Simulation Affected by Groundwater Exploitation in Riverside. PhD Thesis, Jilin University, Changchun, China.

Lu, S., Feng, X. & Su, X. 2020 Geochemical characteristics of arsenic in groundwater during riverbank filtration: a case study of Liao River, Northeast China. Water Supply 20 (8), 3288–3300. https://doi.org/10.2166/ws.2020.213.

Massmann, G., Nogezitaz, A., Taute, T. & Peckdeger, A. 2008a Seasonal and spatial distribution of redox zones during lake bank filtration in Berlin, Germany. Environmental Geology 54, 53–65. https://doi.org/10.1007/s00254-007-0792-9.

Massmann, G., Dünnbier, U., Heberer, T. & Taute, T. 2008b Behaviour and redox sensitivity of pharmaceutical residues during bank filtration – investigation of residues of phenazone-type analgesics. Chemosphere 71, 1476–1485. https://doi.org/10.1016/j.chemosphere.2007.12.017.

Petrunic, B. M., MacQuarrie, K. T. B. & Al, T. A. 2005 Reductive dissolution of Mn oxides in river-recharged aquifers: a laboratory column study. Journal of Hydrology 301, 163–181. https://doi.org/10.1016/j.jhydrol.2004.06.022.

Postma, D., Mai, N. T. H., Lan, V. M., Trang, P. T. K., So, H. U., Nhan, P. Q., Larsen, F., Viet, P. H. & Jakobsen, R. 2017 Fate of arsenic during red river water infiltration into aquifers beneath Hanoi, Vietnam. Environmental Science & Technology 51 (2), 838–845. https://doi.org/10.1021/acs.est.6b05065.
Sophocleous, M. 2002 Interactions between groundwater and surface water: the state of the science. *Hydrogeology Journal* 10, 52–67. https://doi.org/10.1007/s10040-001-0170-8.

Tsanov, E., Ribarova, I., Dimova, G., Ninov, P., Kossida, M. & Makropoulos, C. 2020 Water stress mitigation in the Vit river basin based on WEAP and MatLab simulation. *Civil Engineering Journal* 6 (11), 2058–2071. https://doi.org/10.28991/cej-2020-03091602.

Tufenkji, N., Ryan, J. N. & Elimelech, M. 2002 The promise of bank filtration. *Environmental Science & Technology* 36 (21), 422A–428A. https://doi.org/10.1021/es022441j.

Zhu, Y., Zhai, Y., Du, Q., Teng, Y., Wang, J. & Yang, G. 2019 The impact of well drawdowns on the mixing process of river water and groundwater and water quality in a riverside well field, northeast China. *Hydrological Processes* 33 (6), 945–961. https://doi.org/10.1002/hyp.13376.

Zhu, Y., Zhai, Y., Teng, Y., Wang, G., Du, Q., Wang, J. & Yang, G. 2020 Water supply safety of riverbank filtration wells under the impact of surface water-groundwater interaction: evidence from long-term field pumping tests. *Science of the Total Environment* 711, 135141. https://doi.org/10.1016/j.scitotenv.2019.135141.

First received 25 November 2020; accepted in revised form 28 April 2021. Available online 17 May 2021