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# Using hydrodynamic and hydraulic modelling to study microbiological water quality issues at a backwater area of the Danube to support decision-making

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**Abstract** The alluvial backwater areas of the Danube are valuable ecological habitats containing important drinking water resources. Due to the river regulation and the construction of power plants, the river water levels and natural dynamics of the backwater areas continuously decline, threatening their typical characteristics. The aim of this study was to evaluate how an increased connectivity of the backwater branch located in a nature-protected riverine floodplain (enabled by diverting river water into the backwater system via a weir) affects the microbiological quality of groundwater resources. The defined quality criterion was that the diversion measures must not lead to an increased detection frequency of faecal indicators in groundwater. The microbiological water quality of the Danube, its backwater branch and the groundwater was analysed from 2010 to 2013. *E. coli* was selected as bacterial indicator for recent faecal pollution. *C. perfringens* (spores) was analysed as indicator for persistent faecal pollution and potentially occurring pathogenic protozoa. We simulated the microbial transport from the Danube and the backwater river into groundwater using a 3-D unsaturated-saturated groundwater model coupled with 2-D hydrodynamic flow simulations. Scenarios for no diversion measures were compared with scenarios for an additional discharge of 3, 20 and 80 $m^3/s$ from the Danube River into the backwater branch. While the additional discharge of 20 and 80 $m^3/s$ of Danube water into the floodplain strongly improved the ecological status according to ecological habitat models, the hydraulic transport simulations showed that this would result in a deterioration of the microbiological quality of groundwater resources. The presented approach shows how hydraulic transport modelling and microbiological analyses can be combined to support decision-making.

**Keywords** Hydraulic transport model · Faecal indicators · Alluvial floodplains · Microbiological water quality · Drinking water resources

## Hydrodynamische und hydraulische Modellierung der mikrobiologisch-hygienischen Qualität als unterstützendes Werkzeug beim Schutz und Management von Trinkwasserressourcen

**Zusammenfassung** Die Flussauen der Donau sind wertvolle ökologische Lebensräume und enthalten wichtige Trinkwasserressourcen. Durch Flussregulierungen und den Betrieb von Kraftwerken nehmen der Wasserstand und die natürliche Dynamik in den angrenzenden Auengebieten kontinuierlich ab, was die natürliche Charakteristik dieser Landschaftstypen gefährdet. Das Ziel dieser Studie war es zu untersuchen, wie eine verbesserte Anbindung eines Auengebiets durch Einleitung von Donauwasser über ein Wehr die mikrobiologische Qualität der Grundwasserressourcen beeinflusst. Das Qualitätskriterium war so definiert, dass die Anbindungsmaßnahmen nicht zu einem häufigeren Nachweis von Fäkalindikatoren im Grundwasser führen dürfen. Die mikrobiologische Wasserqualität der Donau und des Seitenarms im Untersuchungsgebiet wurde von 2010 bis 2013 analysiert. *E. coli* wurde als bakterieller Indikator für rezente fäkale Verunreinigung gewählt. *C. perfringens* (Sporen) wurden als Indikatoren für persistente fäkale Verunreinigung und potenziell vorkommende pathogene Protozoen gewählt. Wir simulierten den mikrobiellen Transport von der Donau und dem Altarm im Auengebiet ins Grundwasser. Dazu verwendeten wir ein 3-D ungesättigt-gesättigtes Grundwassermodell, gekoppelt mit einem 2-D hydrodynamischen Strömungsmodell. Wir verglichen Modellszenarien ohne Anbindungsmaßnahmen mit welchen mit zusätzlichen Einleitungen von 3, 20 und 80 $m^3/s$ von der Donau in den Seitenarm. Während ökologische Habitatmodelle zeigten, dass eine zusätzliche Einleitung von 20 und 80 $m^3/s$ Donauwasser in die Au den ökologischen Zustand stark verbesserte, zeigten die hydraulischen Transportsimulationen, dass diese zu einer Verschlechterung der mikrobiologischen Qualität der Grundwasserressourcen führen würde. Der vorgestellte methodische Ansatz zeigt, wie hydraulische Transportmodellierung und mikrobiologische Analysen kombiniert werden können, um die Entscheidungsfindung bei der Planung zu unterstützen.

**Schlüsselwörter** Hydraulische Transportmodellierung · Fäkale Verschmutzung · Indikatoren · Flussauen · Mikrobiologisch-hygienische Gewässerqualität · Trinkwasserressourcen

## 1 Introduction

The wetlands of the Danube play an important role for recreation and drinking water supply. These areas offer important habitats for rare and protected wildlife and plant species and are part of a national park following the European guidelines such as 2006/105/EG. The groundwater bodies are important resources for drinking water supply. The preservation of the natural characteristics and the ecological functions of the Danube wetlands are threatened. The elevation of the Danube riverbed progressively declined due to the regulation of river water levels through the operation of power plants. The regulation of the river has further led to a declining connectivity of the backwater areas. Consequently, the groundwater levels and dynamics have declined, and the backwater areas are silting up, which causes ecological deficits. The aim of a sustainable water resource management is thus to improve the hydrological conditions and preserve the ecological functioning of such areas. One possible measure is to divert river water into the backwater branches or to improve the connectivity of the side branches. Such measures could support the development of a more dynamic alluvial backwater area with a larger biodiversity and help preserving the high value of the natural areas in the long term. Such measures, however, must be in line with the Water Rights Act (1959) and must not lead to a restriction of drinking water production. In Austria, groundwater used as drinking water resource must have sufficient quality to require no further treatment. According to the Austrian Drinking Water Ordinance (2001) and the Water Rights Act (1959), faecal pollution is not permitted in drinking water and protected groundwater resources. The compliance with this requirement is assumed fulfilled when standard faecal indicators are not detectable in 100 ml sample volume.

The wetlands of the Danube inhabit abundant wildlife including ruminants, wild boars and birds (Arnberger et al. 2009; Frühauf and Sabathy 2006; Parz-Gollner 2006). Together with the discharges of treated human wastewater along the Danube upstream, these sources potentially contribute to faecal pollution of the surface water bodies. The aim of this study was to evaluate the impact of diverting Danube water into an alluvial backwater area (BA, additionally to the normal influx via the entry point during floods, Fig. 1) on the microbiological groundwater quality based on hydrodynamic surface water and hydraulic groundwater modelling. We investigated different possible scenarios to divert Danube water into the study area. The investigated microbiological parameters were selected according to the Austrian Drinking Water Ordinance (2001). *E. coli* was selected as bacterial indicator for recent faecal pollution. *C. perfringens* (spores) was investigated as indicator to include also persistent faecal excreted pathogens, i.e. protozoan cysts and oocysts. To evaluate the effect of the diversion measures, the criterion was to achieve no increased detection frequency of the faecal indicators in the groundwater samples within the framework of the standard monitoring in comparison with the reference case of no diversion.

## 2 Materials and methods

### 2.1 Hydrological characteristics of the study site

The study area is a typical urban riverine wetland. It is located on the left side of the Danube River at the border to Vienna and is approximately 12 $km^2$ in size (Fig. 1). The Danube is regulated through the operation of a power plant at the study site. Within the city's borders, the Danube Island separates the New Danube from the Danube. Due to the regulation of the Danube, and the construction of a flood protection dam alongside the Danube, the backwater only connects with the main stream during sporadic high water events (see entry point in Fig. 1). In this case, Danube water enters the main backwater branch at its downstream end through a levee opening. When the flood reaches its peak, the flow direction reverses, and the water flows back towards the Danube River. The network consists of numerous further partly disconnected side branches and isolated ponds.

### 2.2 Microbiological analysis

*E. coli* and *C. perfringens* (spores) were analysed monthly from 1-L grab samples taken at an observation point LSW1 on the left side of the Danube from 2010 to 2012 and at observation points LSW2–9 in the backwaters from 2010 to 2013 (Fig. 1). *E. coli* was further analysed at an official bathing site of the New Danube (ND, Fig. 1) in the months from May to September from 2010 to 2013. *E. coli* and *C. perfringens* (spores) were further analysed in groundwater samples at 10 piezometers located along transects to the pumping wells from 2010 to 2013. *E. coli* and *C. perfringens* (spores) were analysed following the methods ISO 16649-2 (2001) and ISO 14189 (2013). Values reported in this paper are presented as colony forming units (CFU)/100 mL. At the New Danube (ND, Fig. 1), *E. coli* was analysed following the method ISO 9308-3 (1998). Values are presented as most probable numbers (MPN)/100 mL.

### 2.3 Groundwater flow model

For simulating the groundwater flow and microbial transport, it was important to account for the groundwater exchange with the Danube and backwaters, the 3-D groundwater flow conditions, the strongly varying topography, the geological layers and the unsaturated zone. Groundwater flow and transport were simulated using SUTRA 2.2 (Voss and Provost 2010). The 3-D governing equation for the

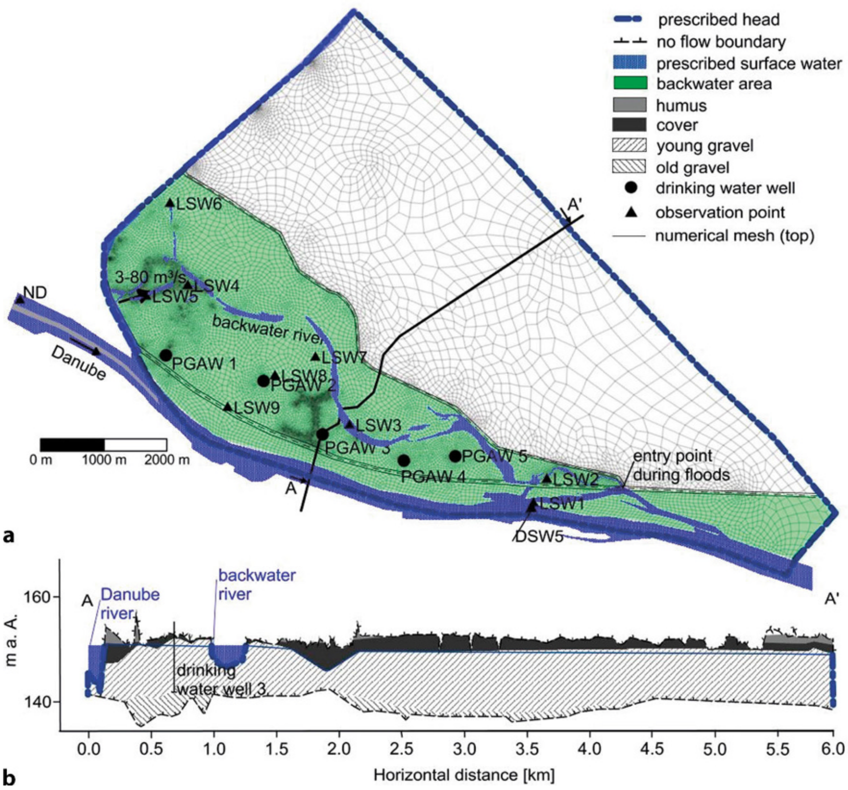


**Fig. 1** **a** Model domain and boundary conditions of the groundwater model. The numerical mesh of the groundwater model and the observation points are indicated. **b** Cross section A-A of the groundwater model

transient variably saturated water flow is

$$\left(\theta_w \rho s_{op} + \theta \rho \frac{\partial \theta_w}{\partial p}\right) \cdot \frac{\partial p}{\partial t} - \nabla\left[\frac{\rho \boldsymbol{K}(\theta_w)}{\mu}(\nabla p + \rho \cdot \mathbf{g})\right] + q_w = 0 \quad (1)$$

with the hydraulic pressure p [kg/m·s$^2$], the 3-D aquifer permeability matrix **K** [m$^2$], the gravity vector **g** [9.81 m/s$^2$], the water content $\theta_w$ [–], the water density $\varrho$ [kg/m$^3$], the specific yield $\theta$ [–], the fluid viscosity $\mu$ [kg/m·s], and $q_w$ [L/m$^3$·s] the fluid mass source. The groundwater model domain comprises an area of 52 km$^2$, limited by a Danube River stretch of 9 km length (Fig. 1). The dimensions of the aquifers were chosen large enough to avoid errors induced from boundary effects. The horizontal discretization of the numerical elements vary from 5 to 100 m. Along the banks of the Danube river and the floodplain river and in the vicinity of the drinking water wells, smaller cell sizes were used to minimize the effect of numerical dispersion on transport simulations (Derx et al. 2013). The aquifer beneath the riverbed was 10 m deep on average and was discretized into 15 layers at an interval from 1 to 90 cm in the upper layers and 0.3 to 6 m in the lower layers. The model consisted of approximately 520,000 elements in total.

The groundwater model was coupled with a 2-D hydrodynamic flow model (CCHE2D version 2.0, National Center for Computational Hydroscience and Engineering, University of Mississippi). The simulated surface water levels were set as head boundary conditions along the Danube and in the backwaters. A detailed description of the model set up and calibration is given by Frick et al. (2020). The Danube boundary was approximately 150 m wide, between the river centerline and a steep river embankment. Outside the hydrodynamic model domain, the head boundary conditions along the Danube River were set to interpolated values of the characteristic Danube water levels to the hourly-observed water levels at the Danube gauge Fischamend at river-km 1907.90 (KWD 2010). The backwater area and the Danube River bank were defined as a transition zone alternating between submerged and dry during the flood events (Fig. 1), as described by Derx et al. (2013). All other vertical head boundary conditions were interpolated from measured heads at 26 groundwater piezometers located within 5 km distance. The top layer in the land zone was set to no-flow, assuming a negligible groundwater recharge from precipitation. The bottom boundary of the domain was defined to represent the aquitard. Groundwater was abstracted from five production wells. Numerical nodes that were located within a circle of 50 m in diameter around each well at the bottom model boundary represented the horizontal collector pipes extending radially at the bottom of the aquifer. Pumping rates of 330, 250, 280, 180 and 140 l/s at wells PGAW 1–5 were assumed, respectively (Fig. 1). Transient boundary conditions were set at hourly time steps. As initial pressure heads and for simulations of steady flow conditions, simulations were performed over 1.5 years holding all boundary conditions constant.

The groundwater flow model was calibrated by adjusting the aquifer permeability within zones and fitting the simulated to the observed hourly levels at 46 groundwater piezometers distributed within the model domain. Care was taken to consolidate the same values of permeability into large regions. The aquifer permeability was set vertically anisotropic with an anisotropy ratio of 0.1 (Eq. 1). We assumed a constant specific yield of 0.15 (de Marsily 1986). Water saturation and permeability in the unsaturated zone were calculated by using the model of van Genuchten (1980) using parameter values as obtained by the Rosetta Lite program (Schaap et al. 2001) for the sand textural class of the USDA triangle. The permeability of the confined layer was set to $1 \times 10^{-12}$ m$^2$ according to results of infiltration tests and sieve analyses. To consider clogging processes, very low permeabilities of $10^{-12}$ to $10^{-13}$ m$^2$ were assigned to the uppermost 50 cm of the riverbed and riverbank in parts of the model domain, along the bed of the main and ephemeral side backwater branches according to a manual survey (Walter Reckendorfer and Raimund Taschke, personal communication). Two calibration periods were chosen, one period during mean steady flow conditions

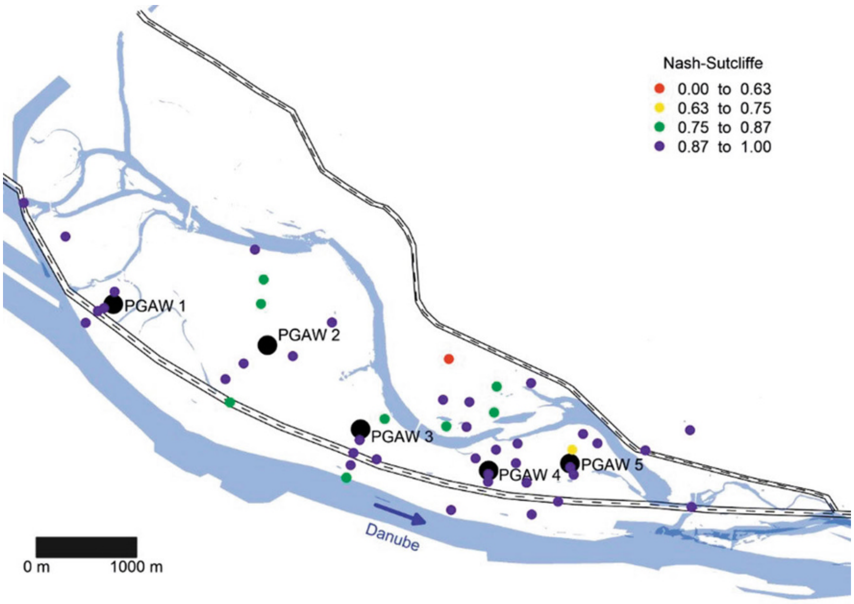


**Fig. 2** Nash-Sutcliffe coefficients of model efficiency comparing measured and simulated hourly water levels at the groundwater piezometers during the flood event from Oct. 13 to Oct. 23, 2011

and one period during a flood event in January 2011, corresponding to a return period of 9 years. The model calibration resulted in an aquifer permeability ranging from $1 \times 10^{-11}$ to $1 \times 10^{-9}\,m^2$. As a measure of model performance, we calculated the Nash-Sutcliff coefficients resulting in values above 0.87 at 78% of the piezometer locations (Fig. 2).

### 2.4 Microbial transport model in groundwater

We used the calibrated groundwater model to simulate the transport of *E. coli* and *C. perfringens* (spores) by solving the 3-D advection-dispersion equation with first order removal rates $\mu_w$ [1/d] (Voss and Provost 2010)

$$\frac{\partial\theta\theta_w\rho C}{\partial t} + \frac{\partial(1-\theta)\,\rho_s C_s}{\partial t} - \nabla(\theta\theta_w\rho \boldsymbol{D}\nabla C) + \nabla\left(\theta\theta_w\rho\boldsymbol{v}C\right) = Q_c + \theta\theta_w\rho\mathfrak{t}_w C \qquad (2)$$

with the microbial concentration in groundwater C [CFU/100 mL], the simulation time t, the soil density $\varrho_s$ [kg/m$^3$], the 3-D dispersion tensor **D** [m$^2$/s], and the pore velocity vector **v** [m/s]. The microbial removal rates [1/s] are combined filtration rates by retention and inactivation. Filtration rates of *E. coli* ranged from 0.4–2.5 1/m in sand and gravel (Artz et al. 2005; Smith et al. 1985). Using the most conservative value and a mean simulated pore velocity of $10^{-4}$ m/s during the calibration, this converts to $2.3 \cdot 0.4 \cdot 10^{-4} = 9.2 \cdot 10^{-5}$ 1/s. For *C. perfringens* (spores), the same removal rate was assumed as for *E. coli*. The inactivation rate of *E. coli* in groundwater was set to 0.18 1/d (Nasser et al. 1993). For *C. perfringens* (spores), an inactivation rate of 0 1/d was assumed. According to injection tracer tests in the field, microbial transport often follows preferential flow paths (Pang 2009). To represent this behavior and to keep the numerical dispersion small, the dispersion was set so that the criterion $\boldsymbol{v} \cdot x / \boldsymbol{D} \le 2$ was met (Kinzelbach 1987), with x [m] being the maximum extent of a numerical element. **D** is the product of longitudinal and transversal dispersivity and the pore velocity **v**. In the well capture zones of PGAW 1–5, the longitudinal and transversal dispersivity were set to 20 and 1 m, respectively. A vertical anisotropy ratio of 0.1 was assumed in consistency with the sediment origin of the aquifer (Gelhar et al. 1992).

### 2.5 Simulated scenarios

Through the combination of hydrodynamic surface water and hydraulic groundwater models, we investigated scenarios of different diversion rates into the backwater branch. The diversion measures were then evaluated based on the criterion that they must not lead to a significant ($p < 0.05$) increase in detection rates of faecal indicators in groundwater according to the Austrian Drinking Water Ordinance (2001). One possible scenario was to pipe 3 m$^3$/s surplus water of the New Danube (Fig. 1) in free descent into the backwater branch. Additionally, we investigated scenarios to connect the backwater branch with the Danube in order to restore the natural water level dynamics. The possible scenarios included an additionally discharge of 3 m$^3$/s, 20 m$^3$/s and 80 m$^3$/s of Danube water into the backwater branch via weirs. Such measures would require substantial construction works to lower the existing traverses along the backwater river, and to widen the current entry point of the Danube during floods (Fig. 1). The transport of *E. coli* and *C. perfringens* (spores) was simulated for hydrological conditions at low water level (LW), at mean water level (MW), and at mean water level plus 1 m (MW+1) (Table 1). In all scenarios, the initial concentrations of both microbial parameters were assumed to be 0 CFU/100 mL in groundwater, which complies with the measured concentrations of faecal indicators in groundwater over 97–98% of the time (Sect. 3.1). The transport simulation time was 60 days. All simulations were run on the computer aided engineering cluster (cae.zserv and phoenix2.zserv, TU Wien). The computational time for each scenario was several hours.

In order to evaluate the investigated scenarios, the modelling results are presented as difference between simulated faecal indicator concentrations with and without diverting Danube water into the backwater area. These differences are presented in ranges from $10^{-3}$ to $10^{2}$ CFU/100 mL in Figs. 4, 5, 6 and 7. An increase in faecal indicator concentrations above $10^{-2}$ CFU/100 mL would lead to higher detection rates during routine monitoring in the short term (within weeks to months), considering the frequency of standard microbiological analyses of groundwater. An increase in concentrations in the range from $10^{-3}$ to $10^{-2}$ CFU/100 mL would lead to higher detection frequencies during routine monitoring in the midterm (several months to one year). Increases in faecal indicator concentrations in the range from $10^{-3}$ to $10^{2}$ CFU/100 mL are therefore not acceptable considering the criteria of

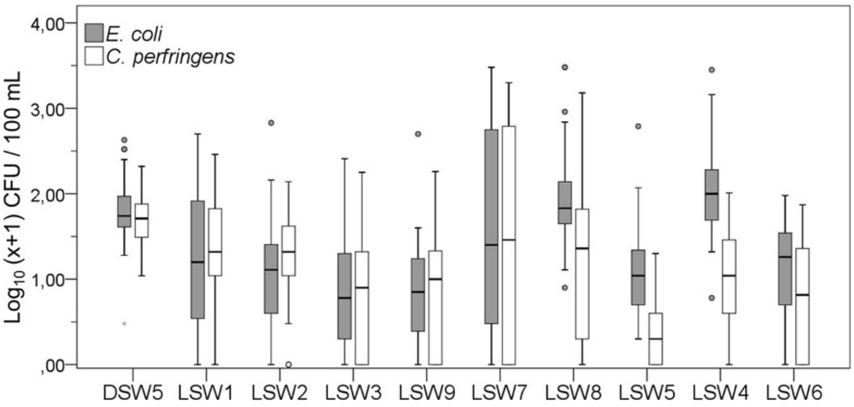


**Fig. 3** Observed concentrations of *E. coli* and *C. perfringens* (spores) at the observation points in the Danube (*DSW 5*) and in the backwater (*LSW 1–9*, Fig. 1) (DSW 5: $n=31$, LSW 1: $n=33$, LSW 2: $n=33$, LSW 3: $n=34$, LSW 4: $n=28$, LSW 5: $n=31$, LSW 6: $n=31$, LSW 7: $n=14$, LSW 8: $n=23$, LSW 9: $n=29$). The *boxes* indicate the 25th and 75th percentile, the *whiskers* indicate the 10th and 90th percentile, *circles* indicate values between the 5th and 95th percentile, *horizontal lines* indicate the median values

no deterioration of the microbiological groundwater quality. The Kruskal-Wallis test was performed to evaluate if the differences in concentrations were significant ($p<0.05$).

## 3 Results

### 3.1 Microbiological water quality

The monitored concentrations in surface water showed similar patterns for *E. coli* and *C. perfringens* (spores). The sampling sites at the Danube revealed a moderate faecal pollution level and at the New Danube a low faecal pollution level according to the classification scheme of Kirschner et al. (2009; Tables 1 and 2). The measured values were assumed as river source water concentrations in the groundwater simulations for the scenarios (Table 1).

In the main backwater branches of the study area, the faecal contamination decreased with increasing distance from the Danube (from sampling site LSW1 to LSW3, Fig. 3). The observed gradient of faecal contamination correlated with the decreasing connectivity of the sampling site with the Danube (Frick et al. 2020). At the sampling sites LSW 4, 7 and 8 representing sites with little connectivity to the Danube, a high faecal pollution level was found, with 90th percentile values from 1000 to 3200 CFU/100 mL for *E. coli*, and from 100 to 2500 CFU/100 mL for *C. perfringens* (spores), respectively (Fig. 3). These elevated concentrations presumably originated from wildlife faeces. A low level of faecal pollution was found at sampling sites LSW 5, 6 and 9, with 90th percentile values ranging from 30 to 130 CFU/100 mL for *E. coli*, and from 20 to 200 CFU/100 mL for *C. perfringens* (spores), respectively (Fig. 3).

The microbiological analysis of groundwater samples in the study area ($n=353$) resulted in concentrations of 0 CFU/100 mL in most cases, with prevalence rates of 2.5 and 2.8% for *E. coli* and *C. perfringens* (spores), respectively. The measured concentrations reached at maximum 9 CFU/100 mL of *E. coli* and 16 CFU/100 mL of *C. perfringens* (spores), respectively.

### 3.2 Scenarios of increasing floodplain connectivity

A Kruskal Wallis test showed that the simulated diversion of 3 $m^3/s$ of New Danube water into the backwater branch during LW already led to a significant deterioration of the microbiological groundwater quality as compared to the reference scenario ($\chi^2(1)=26.3$, $p=2.8\times10^{-7}$, $n=3899$, Fig. 4). Higher simulated concentrations of *E. coli* in groundwater near the drinking water wells PGAW 1, 3, 4 and 5 in comparison with the reference case would even lead to more elevated detection rates at these sites. According to the simulations, the microbiological groundwater quality deteriorates along parts of the main backwater branch, while it improves along other parts of the main backwater branch compared with the reference scenario (Fig. 4). A similar effect was shown for *C. perfringens* (spores).

When diverting 3 $m^3/s$ of Danube water into the backwater branch during LW, the microbiological ground-

**Table 1** Microbial concentrations assumed in the backwater branches during the scenarios at low water (*LW*), mean water (*MW*), and mean water plus 1 m (*MW + 1*) in the Danube. The concentrations of *E. coli* and *C. perfringens* (spores) represent the 95th percentile values of the monthly monitoring data from 2010–2013

| Scenario | Hydrological | Source water | *E. coli* [CFU/100 mL] [a][MPN/100 mL] | *n* | *C. perfringens* (spores) [CFU/100 mL] | *n* |
|---|---|---|---|---|---|---|
| 0 $m^3/s$ (reference) | LW, MW + 1 | Backwater river | 112 | 95 | 56 | 95 |
| 3 $m^3/s$ | LW, MW + 1 | New Danube | 61[a] | 37 | 59 | 28 |
| 3 $m^3/s$ | LW, MW + 1 | Danube | 330 | 33 | 181 | 33 |
| 20 $m^3/s$ | MW | Danube | 330 | 33 | 181 | 33 |
| 80 $m^3/s$ | MW | Danube | 330 | 33 | 181 | 33 |

**Table 2** Classification scheme for the microbiological contamination of surface water for the faecal indicator *E. coli* according to Kirschner et al. (2009)

| CFU/100 mL | Low | Moderate | Critical | High | Very high |
|---|---|---|---|---|---|
| *E. coli* | ≤100 | >100–1000 | >1000–10,000 | >10,000–100,000 | >100,000 |

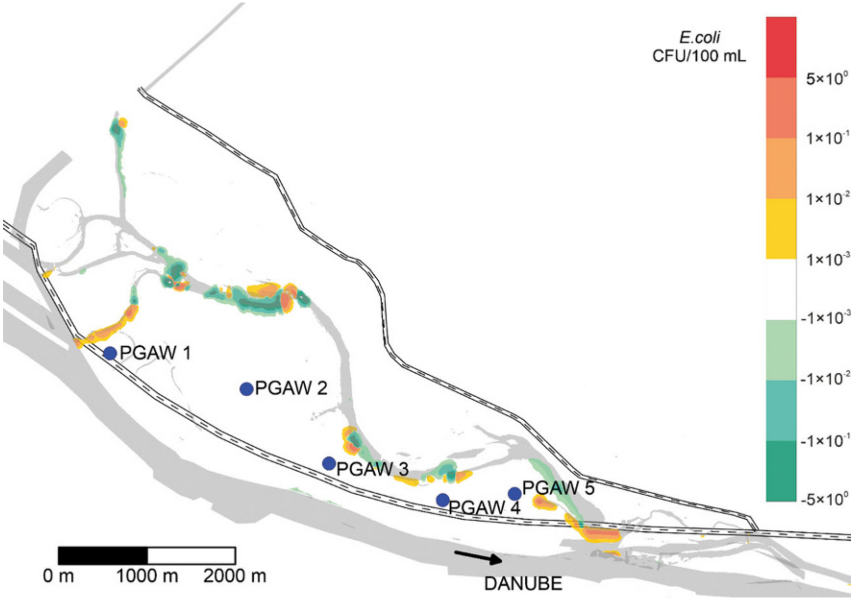


**Fig. 4** Difference between simulated concentrations of *E. coli* with and without diverting $3\,m^3/s$ of New Danube water into the backwater area at low water level (*LW*) conditions (positive/negative values indicate an increase/decrease in concentrations). Indicated are the concentrations at the bottom of the aquifer after 60 days of simulation time. The same results were obtained for *C. perfringens* (spores)

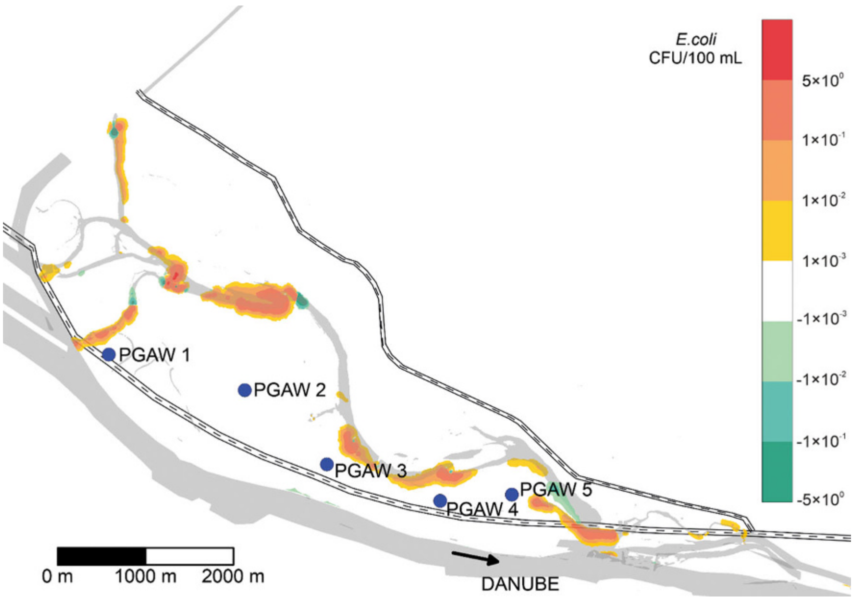


**Fig. 5** Representing Fig. 4 but for diverting $3\,m^3/s$ of Danube water into the backwater area

water quality near wells PGAW 1, 3, 4 and 5 deteriorated significantly more than if diverting New Danube water as shown by a Kruskal Wallis test ($\chi^2(1) = 938.4$, $p = 4.3 \times 10^{-206}$, $n = 5241$, Fig. 5). For all of the above scenarios, positive differences between simulated concentrations of *E. coli* with and without diverting $3\,m^3/s$ were found within the well catchment area of PGAW 1, which would therefore lead to higher detection frequencies in groundwater. Due to the higher flow rates of the main backwater branch in the scenarios than for the reference, parts of the ephemeral side branches filled up with water. Such conditions currently occur only during flood events. During MW+1, the concentrations of both faecal indicators increased even more than during LW near the main backwater branch (results not shown). The diversion of 20 to $80\,m^3/s$ of Danube water at MW likewise led to significantly increased concentrations of faecal indicators as shown by a Kruskal Wallis test in particular in the region near the main backwater branch ($\chi^2(1) = 357.4$, $p = 1.0 \times 10^{-79}$, $n = 4317$, Fig. 6, and, $\chi^2(1) = 543.7$, $p = 2.8 \times 10^{-120}$, $n = 4668$, Fig. 7). The diversion of $80\,m^3/s$ of Danube water additionally resulted in elevated concentrations of both faecal indicators near wells PGAW 4 and PGAW 5 (Fig. 7). We tested the plausibility of the model results by comparing the simulated and observed concentrations of *E. coli* at well PGWA5. According to the simulations, the concentrations of *E. coli* ranged from $1 \times 10^{-3}$ to $1 \times 10^{-2}$ CFU/100 mL at well PGWA 5 during MW+1 conditions (results not shown). The standard monitoring of *E. coli* at this well over multiple years showed concentrations in the same range only 3 to 5 times per year, which corresponds with the frequency in which this well is influenced by the Danube. The measured concentrations of *E. coli* at all other wells during these times were below the limit of detection corresponding with the simulation results.

## 4 Discussion

### 4.1 Combining hydrodynamic modelling with data from standard faecal indicator bacteria

The aim of this study was to investigate the effect of diverting river water into an alluvial floodplain area on the microbiological groundwater quality as a resource for drinking water supply. The quality criteria were chosen according to the Austrian regulations for drinking water and groundwater protection (Austrian Drinking Water Ordinance 2001; Water Rights Act 1959). We addressed this question by combining faecal indicator concentrations in surface water at detectable levels with hydrodynamic surface water flow and 3-D groundwater transport modelling. The starting point of the microbiological water quality assessment was determined by the analysed concentrations of faecal indicators in groundwater during monthly monitoring. Then, concentrations of *E. coli* and *C. perfringens* (spores) were simulated for the

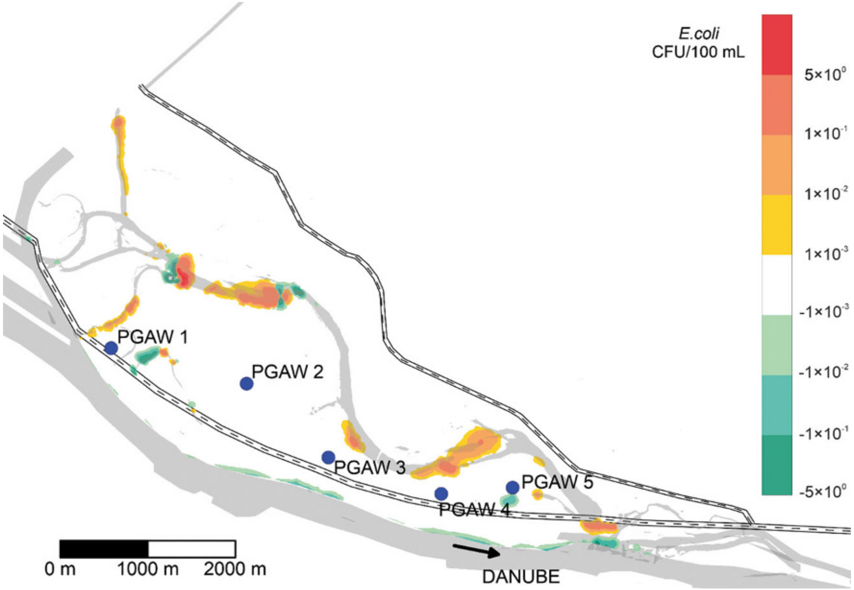


**Fig. 6** Representing Fig. 4 but at mean water level (*MW*) for diverting 20 $m^3/s$ of Danube water into the backwater area

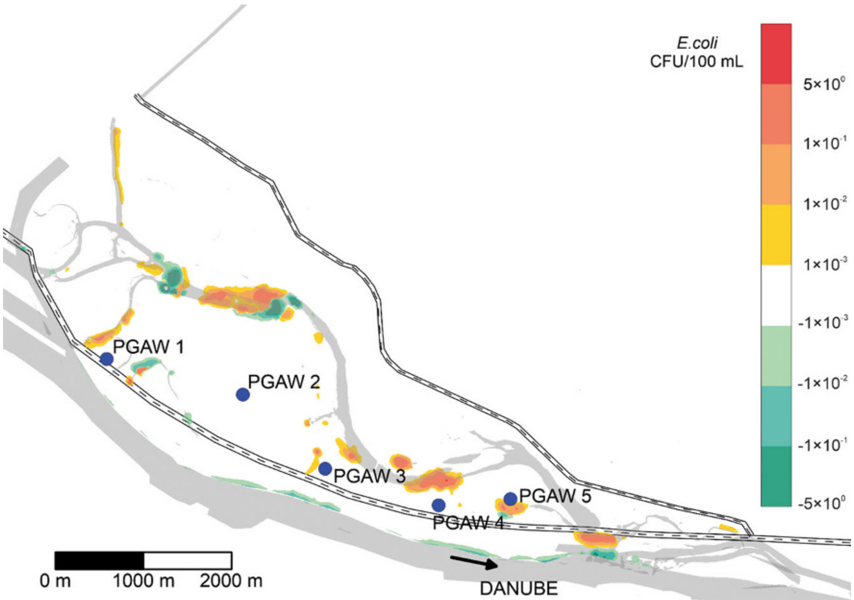


**Fig. 7** Representing Fig. 4 but at mean water level (*MW*) for diverting 80 $m^3/s$ of Danube water into the backwater area

reference case, which is no diversion of river water, and compared with the diversion scenarios of 3, 20 and 80 $m^3/s$.

The scenarios showed that diverting 3 to 80 $m^3/s$ of Danube water or New Danube water into the backwater area leads to significantly increased concentrations of faecal indicators in groundwater ($p < 0.05$), which would cause a higher detection rate during the standard monitoring. Already a diversion rate of 3 $m^3/s$ led to higher water levels and thus to side branches filling with water, which so far only filled during flood events. Measures to divert 20 to 80 $m^3/s$ of water into the backwater area would lead to increases in the water level dynamics, would homogenize the water quality and prevent the eutrophication of isolated backwater branches according to previous studies (Gabriel et al. 2014; Weigelhofer et al. 2014). Despite of this, these measures cannot be recommended from the viewpoint of groundwater as a safe drinking water resource, as the deterioration of the microbiological groundwater quality was shown to affect the drinking water well catchments. All investigated scenarios showed significantly increased concentrations of *E. coli* and *C. perfringens* (spores) in the range of $10^{-3}$ to $10^2$ CFU/100 mL ($p < 0.05$). Increased concentrations in this range would lead to higher detection frequencies during water quality monitoring programs in the short and medium term.

### 4.2 Future perspectives—linking modelling with microbial faecal source tracking and QMRA

In order to assess the safety of the drinking water resources concerning the infection protection, quantitative microbial risk assessment (QMRA) methods could be used. Further parameters to determine the origin of faecal contamination (e.g. animals versus humans) could be valuable tools to track long-term changes of the groundwater quality and to evaluate the QMRA models. These include microbial source tracking markers, bacteriophages as alternative sensitive viral indicators to detect faecal contamination, or parameters for analysing the biological stability of groundwater. Automated online techniques are becoming increasingly important to detect short time microbiological contamination as occurring during floods. Near-real time measurement techniques based on biochemical-physical online measurements are promising parameters for complementing chemo-physical online measurements as continuous process control.

## 5 Conclusion

In this study, we evaluated how an increased connectivity of a backwater branch located in a nature-protected backwater area of the Danube (enabled by diverting Danube water into the system via a weir) affects the microbiological quality of groundwater resources. While the additional discharge of 20 and 80 $m^3/s$ of Danube water into the floodplain strongly improved the ecological status according to ecological habitat models, the hydraulic transport simulations showed that this would result in a deterioration of the microbiological quality of groundwater resources. The presented approach based on microbiological analysis of the water resources combined with hydraulic flow and transport modelling is useful to support decision-making.

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