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30		Particle	
31		Given Name	**István**
32		Suffix	
33		Organization	Institute for Nuclear Research of the Hungarian Academy of Science, Department of Earth and Environmental Sciences
34		Division	
35		Address	Bem tér 18/c, Debrecen 4026
36		e-mail	
37		Family Name	**Szabó**
38		Particle	
39		Given Name	**Gergely**
40	Author	Suffix	
41		Organization	University of Debrecen
42		Division	Department of Physical Geography and Geoinformatics
43		Address	Egyetem Square 1, Debrecen 4010
44		e-mail	
45		Family Name	**Tóth**
46		Particle	
47		Given Name	**Csaba**
48	Author	Suffix	
49		Organization	University of Debrecen
50		Division	Department of Physical Geography and Geoinformatics
51		Address	Egyetem Square 1, Debrecen 4010
52		e-mail	
53		Family Name	**Posta**
54		Particle	
55		Given Name	**József**
56		Suffix	
57		Organization	University of Debrecen
58	Author	Division	Department of Landscape Protection and Environmental Geography
59		Address	Egyetem Square 1, Debrecen 4010
60		Organization	University of Debrecen
61		Division	Department of Inorganic and Analytical Chemistry
62		Address	Egyetem Square 1, Debrecen 4010
63		e-mail	
64	Author	Family Name	**Mester**
65		Particle	

66		Given Name	**Tamás**
67		Suffix	
68		Organization	University of Debrecen
69		Division	Department of Landscape Protection and Environmental Geography
70		Address	Egyetem Square 1, Debrecen 4010
71		e-mail	

75	Abstract	In the present paper we suggest a novel calibration method of the model for hydrodynamic and contaminant transport using the example of a sewage disposal site set up uninsulated in a sandy environment. With the hydrodynamic model we applied time-dependent model calculations in order to fit the individual hydrodynamic parameters. For the calibration of the transport model, sodium was chosen, which has a negligible retardation factor. We demonstrated that this approach is suitable for creating a model that provides calculated results comparable to the actually measured, experimental ones. The created model proved to be appropriate for use in the estimation of the maximal spatial extension of the contamination, which—in the case of the investigated sewage disposal site—was found to be 0.1 km^2 in the near-surface (1–3 m deep) layers, whereas it was three times higher at a depth of 40–60 m.

77	Foot note information	

The Use of Sodium to Calibrate the Transport Modeling of Water Pollution in Sandy Formations Around an Uninsulated Sewage Disposal Site

György Szabó · Éva Bessenyei · Andor Hajnal · István Csige ·
Gergely Szabó · Csaba Tóth · József Posta · Tamás Mester

Abstract In the present paper we suggest a novel calibration method of the model for hydrodynamic and contaminant transport using the example of a sewage disposal site set up uninsulated in a sandy environment. With the hydrodynamic model we applied time-dependent model calculations in order to fit the individual hydrodynamic parameters. For the calibration of the transport model, sodium was chosen, which has a negligible retardation factor. We demonstrated that this approach is suitable for creating a model that provides calculated results comparable to the actually measured, experimental ones. The created model proved to be appropriate for use in the estimation of the maximal spatial extension of the contamination, which—in the case of the investigated sewage disposal site—was found to be 0.1 km^2 in the near-surface (1–3 m deep) layers, whereas it was three times higher at a depth of 40–60 m.

G. Szabó (✉) · É. Bessenyei · J. Posta · T. Mester
Department of Landscape Protection and Environmental Geography, University of Debrecen, Egyetem Square 1, Debrecen 4010, Hungary
e-mail: szabo.gyorgy@science.unideb.hu

A. Hajnal
Department of Mineralogy and Geology, University of Debrecen, Egyetem Square 1, Debrecen 4010, Hungary

I. Csige
Institute for Nuclear Research of the Hungarian Academy of Science, Department of Earth and Environmental Sciences, Bem tér 18/c, Debrecen 4026, Hungary

G. Szabó · C. Tóth
Department of Physical Geography and Geoinformatics, University of Debrecen, Egyetem Square 1, Debrecen 4010, Hungary

J. Posta
Department of Inorganic and Analytical Chemistry, University of Debrecen, Egyetem Square 1, Debrecen 4010, Hungary

1 Introduction

The anthropogenic contamination of groundwater presents a serious problem in many locations worldwide (Datta et al. 2011; Delkash et al. 2014). Contaminants of agricultural and industrial origin can frequently be found lying behind this problem (Al-Khashman 2008); in addition, the lack of any sewerage installation, poorly insulated or even uninsulated communal septic tanks, sewage disposal sites lacking proper technological protection, and landfills for solid waste materials can also pose a threat and may seriously contaminate the groundwater (Phan et al. 2013, 2014). The rate and severity of contamination can reach such levels that in and around settlements the groundwater may become unsuitable not only for human or animal consumption but also for irrigation (Somlyódi 2002). Of the various groundwater types, shallow groundwater is the most at risk, because it is usually located near the surface and contaminants from the topsoil are first transferred here via water

infiltration (Cho et al. 2000; Kerényi 2003; Slack et al. 2005, 2007; Fejes et al. 2012).

To control the quality of groundwater and to follow the transport of possible contaminants, the operation of a monitoring network is essential. At the same time, the interpretation of the measurement results requires the formulation of a conceptual geological model. If a mathematical model based on this conceptual model is also available, it enables significant further qualitative analyses. Over the last few decades, due to the rapid development of computer science, the creation of such models—and their practical application in the solution of groundwater-protection problems—has become more and more frequent (Shahid et al. 2000; Sonneveld et al. 2010; Zhang et al. 2013). In order to protect groundwater, it is vital to ensure both the geological and the technological protection of the waste disposal sites (Panno et al. 2002, 2006; Sipos et al. 2012; Chawla and Singh 2014). This statement is supported by the research conducted by Regadío et al. (2011) and Zhan et al. (2013) in sampling locations in Spain and southeastern China, respectively, in the areas surrounding operational waste disposal sites. In India, the high arsenic (As) content of the groundwater can cause serious problems, and to understand these processes, researchers have relied on three-dimensional hydrodynamic and transport models that work with finite-difference methods, which have long been established in Japan (Nakaya et al. 2011). Chakraborty and Ghosh (2009) also worked with software that uses a finite-difference method, to model chloride and sodium transport in the vicinity of a municipal solid waste landfill.

The present research was conducted in the area of a sewage disposal site that had been operational for almost three decades and then recultivated. Based on the available measurement results, the main aim of our study was to establish a calibration procedure that would significantly improve the reliability of the contaminant transport model currently in use. A further aim was to use this newly calibrated model to assess the main characteristics of the spatial and temporal changes in water quality and the development of the clarifying process following recultivation. The latter is even more important as there is surprisingly little information available in the literature with regard to the actual size of the affected area of such a contamination source in sandy zones. The rate of water quality improvement after a recultivating intervention is also rather unclear. Thus, these investigations may provide a solid basis for the assessment of risks due to the contamination of artesian waters and also for the establishment of the need for a possible future intervention.

2 Materials and Methods

2.1 Description of the Study Area

The area we studied is found in eastern Hungary, 8 km south of Debrecen, in the administrative township of Mikepércs (Fig. 1). Regarding its geological structure, it is a S-SW sloping, sand-covered alluvial fan plain of aeolian origin. The climate of the area is warm temperate and dry temperate; the mean annual temperature is 9.8–10.0 °C, and the average annual rainfall is 550–580 mm (Dövényi 2010). Between 1982 and 2011, the hole of an abandoned sandpit was used as a disposal site for the wastewater produced by the town, the quantity of which even exceeded a daily 100 m³. This sewage disposal site had been established and used without any authorization, and had no technological protection. The sewage disposal site with a total capacity of 20,000 m³ was formed on a sandy soil structure, and under such conditions the contaminants could easily infiltrate into the groundwater. Previous studies from 2004, 2005, and 2010 justified that, as a consequence of the sewage disposal, the groundwater became heavily contaminated (Szabó et al. 2007a; Szabó et al. 2007a; **Q2** Szabó et al. 2007b). The recultivation of the sewage disposal site took place in the spring of 2011. According to the recultivation plan, the wastewater from the sewage disposal site was transferred to a sewage-treatment plant operated by the Debreceni Vízmű Zrt. (Debrecen Waterworks Ltd.). The sewage sludge was processed and rendered harmless by the AKSD Városgazdálkodási Kft. (A.K.S.D. City Management Ltd. of Debrecen). Based on the recultivation plans, the pit of the former sandpit—after being cleaned of wastewater and sewage sludge—was filled with 9000—9500 m³ of sand and 2500 m³ of organic matter rich soil (Green Side 2008). **Q3**

2.2 Sampling and Laboratory Tests

In order to collect soil and groundwater samples, a total of 12 shallow groundwater wells have been drilled since 2004 with an Eijkelkamp-type hand auger. Taking the NE-SW flow direction of the groundwater into
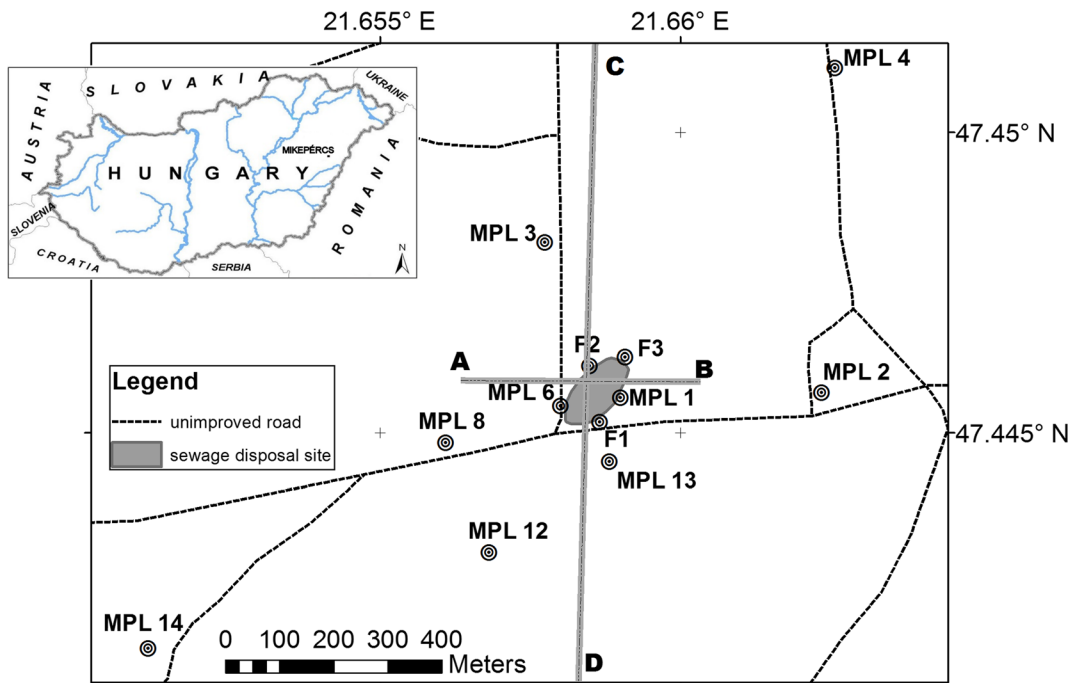
Fig. 1 The location of the study area, the sampling points, and the vertical sections (*A–B*, *C–D*)

147 consideration, the sampling locations (see Fig. 1) were
148 assigned so that the samples would provide as much
149 information as possible about the groundwater condi-
150 tions in the surroundings of the sewage disposal site.
151 The well casing was made of PVC pipes with a diameter
152 of 50 mm, the lower 1 m section of which is screened.
153 This was necessary to prevent the boreholes becoming
154 silted up when groundwater flows into the well. The
155 bottom depth of the wells was 1 m below the ground-
156 water level measured at the time of the installation
157 (Table 1).
158   Soil samples were taken from each borehole at every
159 20 cm, and were then sealed in plastic bags and trans-
160 ferred to the laboratory of the Department of Geology of
161 the University of Debrecen. The samples were exsiccat-
162 ed at 60 °C, and then their granulometric composition
163 was determined by the Köhn-pipette method (Müller
164 et al. 2009).
165   For the water sampling a peristaltic pump was used.
166 Before sampling we extracted three times the volume of
167 water originally contained in the well to obtain water
168 from the pore volume. At each sampling time one water
169 sample (250 ml) was taken from each well and the depth
170 of the groundwater level was measured (Table 1). The
171 bubble-free water samples were transferred in tightly
172 closed plastic bottles to the Geology Laboratory of the

173 University of Debrecen, where the Na concentration
174 was measured by using a PerkinElmer 3110 AAS.
175   The results of measurements were recorded in a
176 Microsoft Excel database. The statistical analyses were
177 done using SPSS 20. The maps were edited with the
178 Surfer software (version 8.0) (Golden Software LLC),
179 by Kriging interpolation.

2.3 Model Calculation   180

181 After the recultivation, we continued the investigation of
182 the level of groundwater contamination. Based on the
183 test results from three consecutive years after the
184 recultivation, no relevant changes in the groundwater
185 quality were found. We assume that in the course of the
186 three-decade operation of the sewage disposal site both
187 the sediment layers and the groundwater itself became
188 contaminated to such an extent that the long-term effect   **Q5**
189 of the operation must also be taken into consideration.

190   In order to interpret the measurement results and to
191 predict the longer-range transport of the contaminants
192 via the groundwater, we performed hydrodynamic and
193 contaminant transport model calculations. The horizon-
194 tal dimension of the modeled area was 1.8 km × 1.4 km
195 (2.52 km^2), whereas its vertical extension reached 115 m
196 above the level of the Baltic Sea. The HD72/EOV

	ID	EOV Y	EOV X	Bottom depth (cm)	Depth of groundwater (cm)			
					Mean	SD	Min.	Max.
t1.4	MPL1	846942	236792	420	149.8	45.3	57	267
t1.5	MPL2	847171	236824	370	205.4	10.8	186	224
t1.6	MPL3	846816	237084	320	238.2	26.1	167	288
t1.7	MPL4	847191	237425	420	288.1	36.4	171	350
t1.8	MPL6	846837	236745	420	326.8	24.6	273	366
t1.9	MPL8	846706	236703	340	256.3	22.7	191	302
t1.10	MPL12	846758	236518	570	384.3	59.3	301	451
t1.11	MPL13	846904	236682	400	284.8	40.6	213	360
t1.12	MPL14	846593	236312	300	193.1	46.2	99	260
t1.13	F1	846892	236763	534	216.6	44.6	91	269
t1.14	F2	846874	236844	552	229.7	57.1	131	415
t1.15	F3	846921	236889	531	339.5	37.1	253	400

Table 1 The basic properties of the monitoring wells and main statistical parameters of the depth of groundwater between 2004 and 2014

EOV Egységes Országos Vetület

coordinate system (EPSG 23700) was applied for the spatial coverages, in which one unit equals 1 m. In order to construct the digital relief map model of the surface, we used the results of our on-site measurements performed by two Trimble S9 dual-frequency, high-precision geodesic GPS instruments. The accuracy of the GPS measurements was 2 cm. The interpolation of the surface was completed with a free triangular mesh.

Based on the analysis of the boring log (Marton 2009) of the confined aquifer well with a bottom depth of 258.6 m in the vicinity of the sewage disposal site, five different geological strata were identified in the model. In the near-surface layers, the dominant fraction of the geological structure is sand; however, 54 m below the surface, 50-m-thick clayey and silty strata appear (Szabó and Szabó 2005; Marton 2009). The determination of the model's hydrodynamic factors—the diagonal

elements of the hydraulic conductivity tensor (K_{xx}, $K_{yy} \sim 10^{-5}$–10^{-7} m/s, $K_{zz} \sim 10^{-5}$–10^{-8} m/s) and the specific storage factor ($S_s \sim 10^{-5}$ m^{-1})—were based on the particle size distribution of the samples taken from the self-produced shallow groundwater wells, as well as on the data from the confined-aquifer well. For the hydraulic conductivity determination, we have used the Campbell formula:

$$K = 4 \times 10^{-5} \times e^{-6.9 f_{\text{clay}} - 3.7 f_{\text{silt}}}$$

where f_{clay} and f_{silt} are the clay and silt fractions, respectively (Campbell 1985).

Figure 2 shows the grain size distributions along the section of three selected monitoring wells (MPL1, MPL2, MPL3). All the other wells show similar distribution. The graphs show that the dominant fraction is
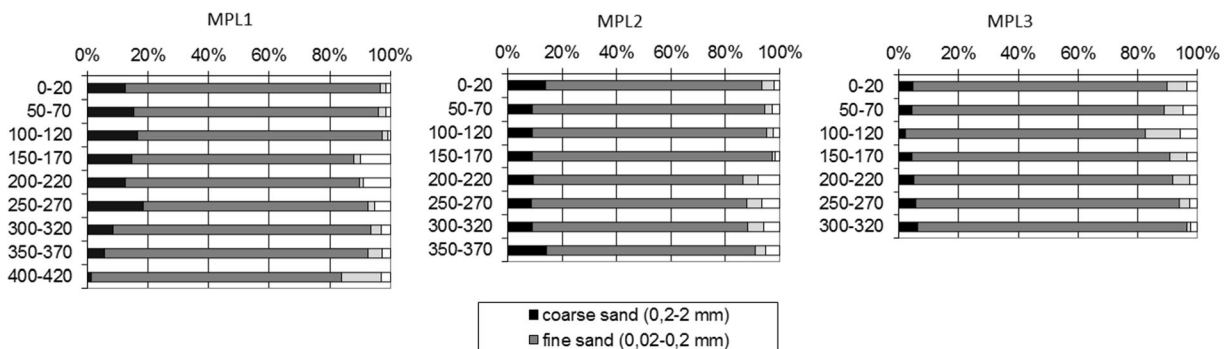
Fig. 2 The grain size distributions of the MPL1, MPL2, and MPL3 wells

fine sand (over 80 %) and the fraction of coarse sand falls below 20 %. The sum of the fractions of silt and clay, which essentially determines the hydraulic conductivity, remains below 20 % in each layer of each well.

In order to calibrate the system's hydrodynamic parameters, we first made temporally varying hydrodynamic model calculations. This was possible because the required set of rainfall and evapotranspiration input data for 3 years after the recultivation of the disposal site was available, as were the results of the hydrodynamic piezometry measurements necessary for the calibration and validation of the model. During the process of modeling, we solved Eq. (1):

$$\frac{\partial}{\partial x}\left(K_{xx}\frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_{yy}\frac{\partial h}{\partial y}\right) + \frac{\partial}{\partial z}\left(K_{zz}\frac{\partial h}{\partial z}\right) - w = S_s\frac{\partial h}{\partial t} \quad (1)$$

Q6 where w is the volumetric flux per unit volume representing the sources and sinks of water and h is the hydraulic head. During the transient calculations, the hydrodynamic potential values at the circumference of the sampling area were set so that they would approximate the water levels measured in the monitoring wells closest to the edge of the area (ID: MPL-4 and MPL-14) as much as possible. The hydrodynamic model was run by the USGS MODFLOW 2005 numerical engine (Harbaugh 2005). Figure 3 shows the input infiltration and evapotranspiration data of the time-dependent model calculations, as well as the calculated

results of the hydrodynamic potential, in comparison with the measured data from a few selected monitoring wells.

In the course of the calibration, three of the studied model parameters—the infiltrated percentage of total rainfall, the average rate of evapotranspiration, and the hydraulic conductivity of the second stratum—were set in such a way that in the other monitoring wells the sum of root–mean-squares (RMS) of the measured and calculated water levels—averaged to all sampling times— were as low as possible. In Fig. 4 a calibration curve relating to a sampling time (day 446), which is selected as an example, shows that the normalized RMS value is quite low (6 %) and the correlation coefficient approximates 1 ($r=0.966$). Thus, the model can be considered reliable.

We created the steady-state hydrodynamic model using the optimized parameters of the transient model and the average annual infiltration typical of the area. The solution of the steady-state model was used in the contaminant transport calculations.
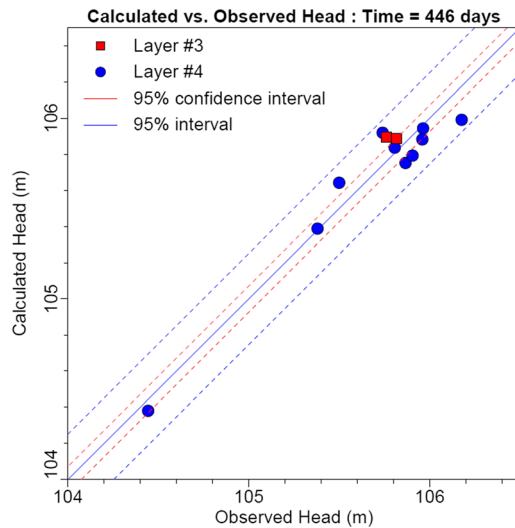
For the contaminant transport calculations, sodium was chosen because of its low retardation factor ($R\sim1.5$) (Rowe et al. 2004); i.e., it is relatively conservative, so it does not transmute during transport via groundwater, and it is poorly adsorbed (Greenwood and Earnshaw 1999; Sayyed and Bhosle 2011). Another important condition was that the given contaminant should originate primarily from the wastewater

Fig. 3 The input infiltration and evapotranspiration data of the transient model calculations and the calculated hydrodynamic potentials (continuous curves), with the water levels measured in the monitoring wells
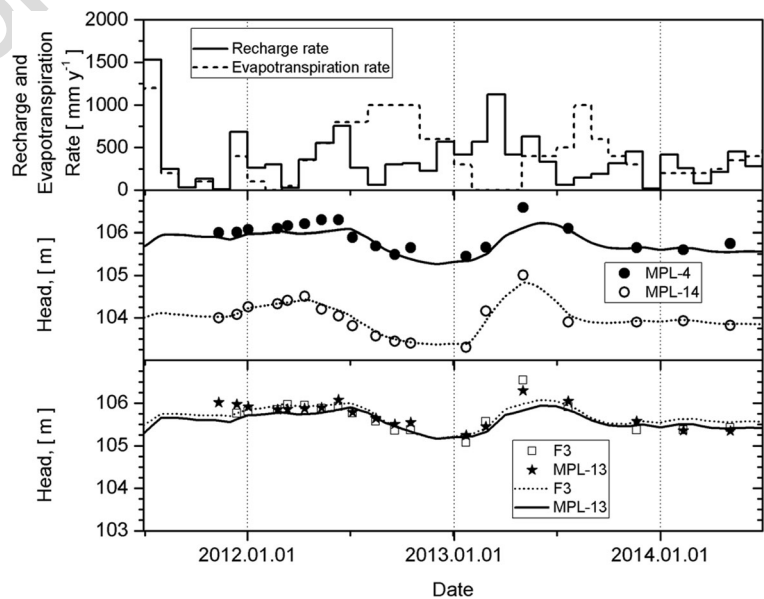
Fig. 4 One calibration curve
(relating to day 446 of sampling)
of the transient hydrodynamic
model

Max. Residual: -0.186 (m) at MPL-4/1
Min. Residual: 0.007 (m) at MPL-12/1
Residual Mean : -0.001 (m)
Abs. Residual Mean : 0.095 (m)

Num. of Data Points : 12
Standard Error of the Estimate : 0.033 (m)
Root Mean Squared : 0.111 (m)
Normalized RMS : 6.395 (%)
Correlation Coefficient : 0.966

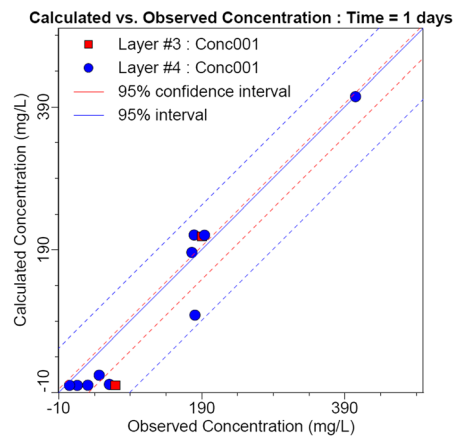collected from the disposal site. When modeling the contaminant transport, we used Eq. (2):

$$\frac{\partial(\theta C)}{\partial t} = \frac{\partial}{\partial x_i}\left(\theta D_{ij}\frac{\partial C}{\partial x_j}\right) - \frac{\partial}{\partial x_i}(\theta v_i C) \qquad (2)$$

where C is the dissolved concentration of sodium; θ is the porosity of the subsurface medium; t is time; x_i is the distance along the respective Cartesian coordinate axis; D_{ij} is the hydrodynamic dispersion coefficient tensor; and v_i is the Darcy velocity, which can be determined from the

solution of Eq. (1). We determined the diffusion constant that influences the contaminant transport as the molecular diffusion coefficient of sodium in water: 4×10^{-9} m^2/s. The longitudinal, horizontal, and vertical dispersivities were specified as 10, 1, and 0.1 m, respectively.

To calculate the contaminant transport, the MT3DMS v. 5.2 numerical transport engine (Zheng and Wang 1999) was applied. In the course of the model calibration, the parameters to be fitted were the Na concentrations in the northern and southern volumes of the disposal site. The transport calculations (Fig. 5)

Fig. 5 The calibration curve of the transport calculations

Max. Residual: -82.886 (mg/L) at MPL-13/A
Min. Residual: -1.156 (mg/L) at MPL-6/A
Residual Mean : -18.632 (mg/L)
Abs. Residual Mean : 31.134 (mg/L)

Num. of Data Points : 12
Standard Error of the Estimate : 10.687 (mg/L)
Root Mean Squared : 40.043 (mg/L)
Normalized RMS : 10.013 (%)
Correlation Coefficient : 0.965

were calibrated to the sampling time following recultivation, for which reliable measurement data were available. Based on the graph it can be established that the reliability of the transport model is acceptable, since the normalized RMS is around 10 % and the value of the correlation coefficient ($r = 0.965$) approximates to 1.

3 Results and Discussion

In the first stage we had to verify that sodium is also suitable for the calibration of the contaminant transport model in the sense that it originates primarily from the wastewater deposited at the disposal site (Szabó and Bessenyei 2013). This is because in alkaline soils the sodium contents of the groundwater may increase under natural conditions as well. However, based on the test results of the water samples originating from the monitoring wells, it can be clearly established that in the affected area of the sewage deposit it is not necessary to take into account any alkaline-formation process, since in the wells located further away from the disposal site (MPL14, MPL2, MPL4) we consistently measured quite low (under 20 mg/l) sodium content. Values greater by about one order of magnitude (150–500 mg/l) were found without exception in the wells in the immediate vicinity of the sewage disposal site (MPL6, F2, MPL1, F1, F3, MPL13) (see Figs. 1 and 6).

Regarding sodium transport, we used the model to investigate two consecutive time periods. For the first period we assumed a disposal site that had been continuously operational for three decades and filled with

contaminants. The second period refers to the disposal site that had been recultivated and filled with sand after the removal of the contaminants.

In the 30-year operational period, the hydrodynamic potential field was calculated for the static condition, with the important difference in comparison to the transient model that in the area of the disposal site we specified the water level that was available based on the measurement data from the operational period (see Fig. 7a).

In the operational period, at the border of the saturated and unsaturated zones, the maximal horizontal infiltration speed—according to calculations based on the model—was 4.2×10^{-6} m/s, whereas the maximal vertical infiltration speed was lower, at 3.1×10^{-6} m/s in areas close to the sewage disposal site (Fig. 7a). In the period following recultivation, in both the horizontal (1.3×10^{-6} m/s) and the vertical (9.2×10^{-7} m/s) infiltration speed, a certain decrease was detected, which can be explained by the cessation of the dome-like hydrodynamic potential field that came about as a consequence of the sewage disposal (Fig. 7b).

When simulating the contaminant transport, in the dome-like potential field typical of the operational period, in the northeastern 3/4 and southwestern 1/4 volumes of the disposal site, constant sodium concentrations of 200 and 500 mg/l were assumed, respectively. The sewage disposal was divided into two parts by an earth bank, and the two catch basin parts became isolated from each other whenever the water levels were low; therefore, in the southwestern basin part increased evaporation could occur, which resulted in an increase in the Na concentration (Fig. 8a).

Fig. 6 The sodium contents of the water from shallow groundwater wells, based on the measurements performed during the study period (2004–2014). The extreme values (third quartile [Q3] + 3 interquartile range [IQR]) were not plotted in the graph
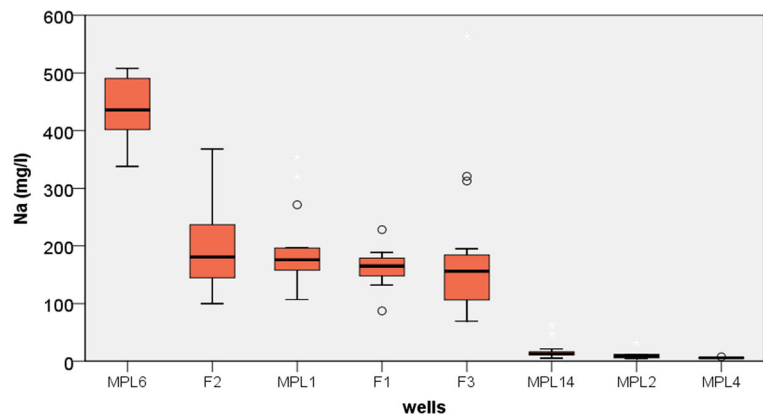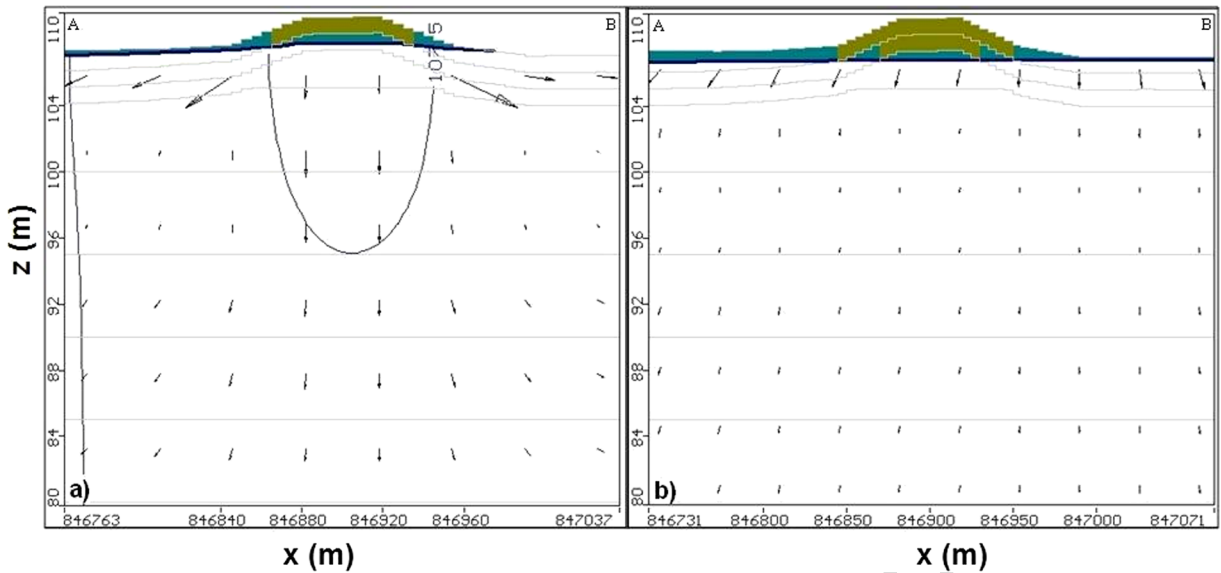
Fig. 7 **a** The W-E orientated vertical picture of the dome-like hydrodynamic potential field in the course of operation. **b** The W-E orientated vertical picture of the hydrodynamic potential field after recultivation

368   The wastewater had been disposed in the larger,
369   northeastern basin part. Consequently, the possibility
370   of a continuous thinning was present.

371   During the functional period, in the groundwater
372   dome that was being formed by the disposed wastewa-
373   ter, a radial transport of contamination can be observed;
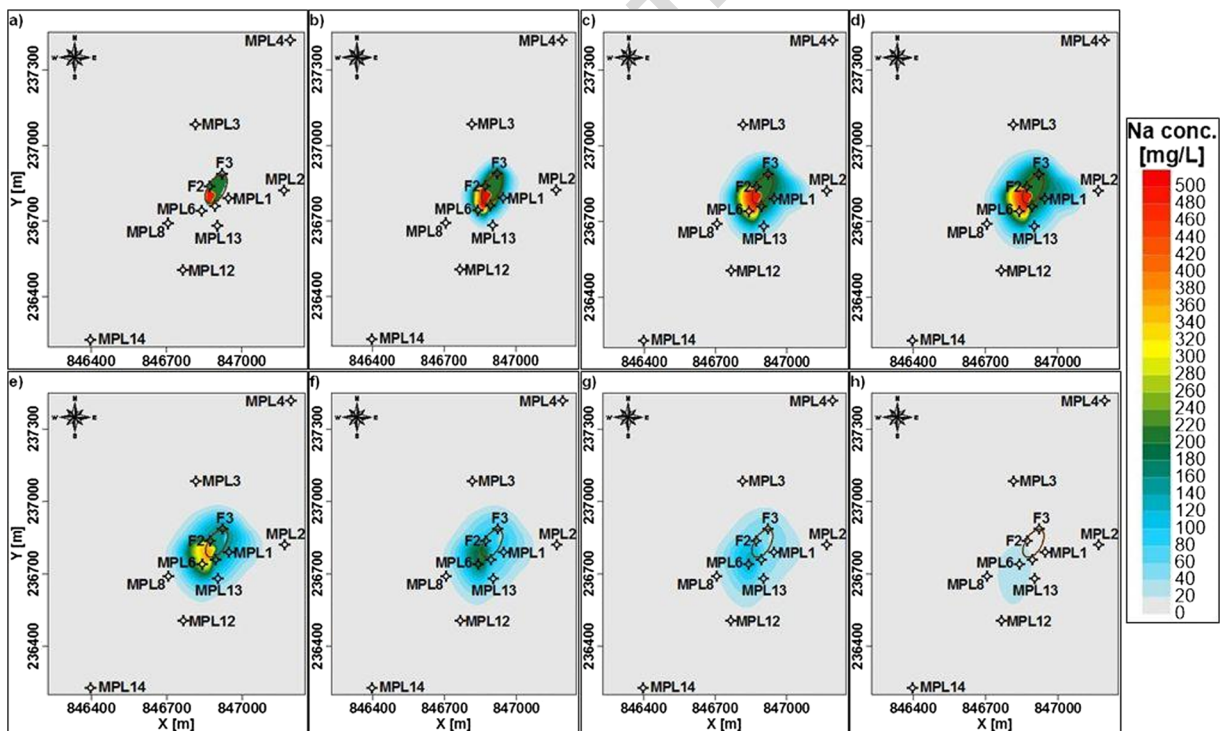
Fig. 8 The horizontal transport of Na contamination, 3 m below the surface: **a** the beginning of operation (1982); **b** 1st year of operation (1983); **c** 10th year of operation (1992); **d** end of operation, recultivation (2011); **e** 1st year after recultivation (2012); **f** 5th year after recultivation (2016); **g** 10th year after recultivation (2021); and **h** 30th year after recultivation (2041)

therefore, the NE-SW flow direction, which is characteristic under natural conditions, was hindered. The height of the groundwater dome in the surrounding areas exceeded the average natural groundwater level by 1 m on average. According to our calculations, the contamination transport at a depth of 3 m reached a distance of 50–100 m by the end of the first year of contamination (Fig. 8b). At such distances the Na concentration decreased to the level of the background concentration level (20 mg/l) typical of the area. After 30 years had passed (by the end of the operational period), at such a depth below the surface the Na contamination—according to both the model calculations and the test results of the collected water samples—could be detected at a distance of up to 200 m, and the size of the affected contamination area was 112,000 m^2 (Fig. 8d). Sodium is a conservative, i.e., enduring, contaminant, and its rate of release from the sewage disposal site is more or less time-independent; therefore, the limited horizontal dimension of the contamination suggests that in the near-surface layers the vertical component must be given greater emphasis in the transport process.

In the calculations for the 30-year period after recultivation, the starting contaminant concentration was borrowed from the concentration distribution results of the end of the first period, except for the volume size of the disposal site itself, where we defined a clean, contaminant-free territory filled with sand, since in the course of recultivation both the wastewater and the sewage sludge were removed from the disposal site and the cleaned pit was filled with pure sand. With these results we predicted the horizontal transport of the contaminants for 30 years. In this period the natural, the NE-SW groundwater flow direction already prevails. Since the resupply of contaminants has ceased, the model predicts the gradual decrease in the Na concentration. Based on the model simulations, after a year a 350–400 mg/l Na concentration can still be expected in the southwestern corner of the disposal site (Fig. 8e). This result is further supported by the 393.3 mg/l concentration value measured in the MP6 monitoring well in July 2012. According to the model, in the fifth year after recultivation the Na concentration would decrease to 250–300 mg/l (Fig. 8f), and in 10 years' time in the southwestern corner of the sewage disposal site the model predicts a mere 140–160 mg/l concentration (Fig. 8g). The model indicates that after 30 years have passed, in the areas surrounding the above-mentioned location, in a 26,000 m^2 area this value (20–40 mg/l)

will still exceed the limit values to a certain extent, but apart from this, we can base our calculations on concentrations below 20 mg/l, even for the former sewage disposal site itself. In other words, during the 30-year period following recultivation, the size of the contaminated area would gradually decrease and also slowly migrate in a SW direction (Fig. 8h).

In addition to its horizontal transport, the vertical distribution of the contamination was also modeled. Figure 9a–d demonstrates the outcome of the vertical transport of the contamination, in the 30-year time span from the start of operation as a functional sewage disposal site to the beginning of its recultivation. It can be clearly seen that the contamination already approaches the 54-m-deep water-impermeable layer in the first year (Fig. 9b). At such a depth the vertical transport of the contaminants slows down, and by the end of the operational period (i.e., in the 30th year following the start of its functioning) it reaches down to about 70 m (Fig. 9d). The highest Na concentrations can be detected under the south basin part of the sewage disposal site, due to the concentration by evaporation mentioned earlier. Concentrations higher than 400 mg/l were undetectable 40 m below the surface. It is also clearly observable that because of the continuous contaminant resupply and the water-impermeable layer found at a depth of 54 m, about 50 m below the surface a more intensive horizontal spread has developed. In this zone, the polluted water can reach distances as great as 400 m, which significantly exceeds those calculated for near-surface layers. The horizontal transport is more intensive toward the S-SW, since this fits the local groundwater flow direction.

In the first year after recultivation, no significant changes were observed. However, as Fig. 9e clearly shows, the contamination rate significantly decreased in the near-surface layers; i.e., the 400–500 mg/l Na concentration was detected only in a much narrower strip. In the 5 years after recultivation, the Na concentration reduces to below 300 mg/l in the near-surface layers even in the most heavily contaminated parts. However, in the zone lying between 30 and 40 m below the surface, below the southern basin of the sewage disposal site, the model still indicates a value of 350 mg/l (Fig. 9f). After 10 years, the contamination decreased further: the highest concentration was 290 mg/l, which appeared at greater depth, between 35 and 55 m below the surface (Fig. 9g). In the 30th year following recultivation, some contamination can still be detected, but its level never exceeds 210 mg/l. The most
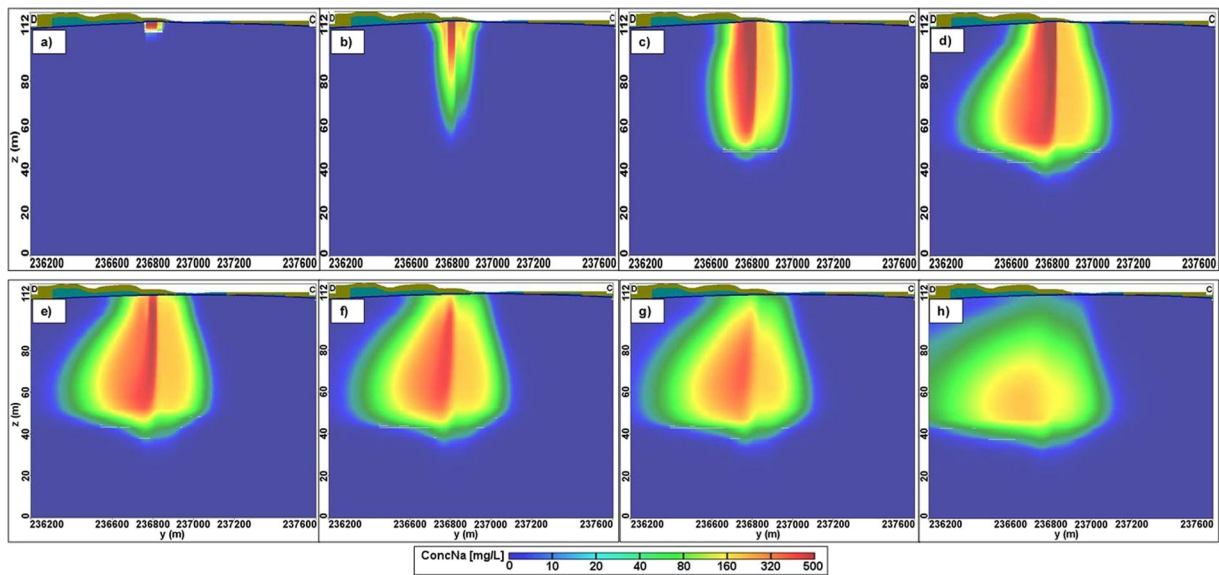
Fig. 9 The vertical transport of contamination: **a** the beginning of operation (1982); **b** 1st year of operation (1987); **c** 10th year of operation (1992); **d** end of operation, recultivation (2011); **e** 1st year after recultivation (2012); **f** 5th year after recultivation (2016); **g** 10th year after recultivation (2021); and **h** 30th year after recultivation (2041)

heavily polluted zone shifted deeper, appearing in the strip 50–60 m below the surface. However, in the near-surface layers, these values fell to around or below the background concentrations (Fig. 9h).

The migration of the contamination in a S-SW direction was considerable even in the period after recultivation. The infiltration continued to be most intensive above the water-impermeable layer that runs 54 m below the surface. After 10 years, water with a higher than 20 mg/l Na concentration reached as far as about 500 m (Fig. 9g). Unfortunately, based on this model, we could not establish precisely how far the contaminated water may travel in 30 years, because the size of the modeled area does not allow this. However, we can assume that the distance covered may be somewhere around 600–700 m. Although the dominant transport direction is S-SW, due to diffusion the contamination can spread toward the north as well.
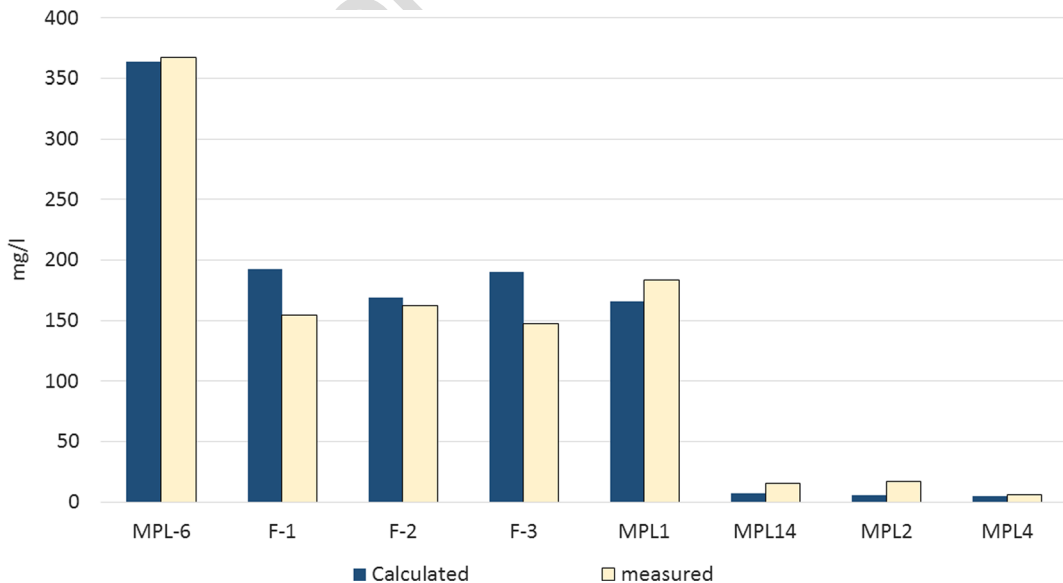
Fig. 10 The measured and the calculated values for the first year after the recultivation

However, the rate of this process is a mere 7 m per 30 years. The water-impermeable layer considerably hindered and slowed down the vertical transport of contaminants, but during the 30 years following recultivation, the contamination reached a further 6 m down (i.e., 76 m below the surface).

In order to verify the reliability of our model, we compared our measured results in the first year after the recultivation to the calculated values of the model for the end of the first year after the recultivation. In Fig. 10 it can be seen that the calculated values correspond with the measured ones, even though some differences may occur because of the uncertainty of the sampling and laboratory measurements as well as the model calculation.

4 Conclusions

In the present study we suggested a novel calibration method of the model for hydrodynamic and contaminant transport, shown via the example of a sewage disposal site set up uninsulated in a sandy environment. In the case of the hydrodynamic model, we applied time-dependent model calculations in order to fit the individual hydrodynamic parameters. For the calibration of the transport model, sodium—which has a negligible retardation factor—was used.

We proved that the Na content of the groundwater—which exceeded the 20 mg/l background concentration and close to the disposal site even reached 500 mg/l—definitely originated from the sewage disposal site. Consequently, sodium—due to its low retardation factor—proved to be a suitable element to allow us to estimate and predict the maximal spatial extension of the contamination. We established that by the end of the operational period the polluting effect of the sewage disposal site remained detectable within a 200-m range in the near-surface (1–3 m deep) layers, which is equivalent to an area slightly greater than 0.1 km^2. However, 40–50 m below the surface, the contaminated area is substantially—at least three times—greater than that measured near the surface. After recultivation the resupply of contamination ceased and according to the model, by the end of the modeled period the size of the polluted area would diminish to one-fifth in the near-surface layers and the Na concentration would decrease approximately to the level of the background concentration. However, in the deeper layers (40–50 m below surface) the Na concentration would still remain higher by about one order of magnitude.

In this pilot study we have shown that the reliability of contamination transport model for a sewage disposal site can be justified. The developed model calibration method can be profitably applied to similar relatively small-scale sites to estimate the maximal spatial extension of contamination in a sedimentary environment, and it may provide a proper basis for modeling the transport of polluting agents that are present in similar sewage disposal sites, which may pose a significantly higher risk in terms of human health and safety.

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