7

Comparison Between Aquifer Solute Concentration And Monitoring Wells

Panagiotis Anastasiadis
Technological Educational Institute of Serres,
Terma Magnisias, 62124 Serres, Greece

Pericles Latinopoulos
Aristotle University of Thessaloniki, Greece, School of Technology,
Faculty of Civil Engineering, GR-540 06 Thessaloniki, Greece

1. Introduction

Disparities between solute concentration in the aquifer and that in monitoring wells have been observed. It is important to be able to relate concentrations between monitoring well and aquifer solute concentration for design of remediation systems. Reported that solute concentrations in a monitoring well may underestimate the corresponding aquifer concentration by over an order of magnitude (Martin-Hayden *et al.*, 1991, Chiang *et al.*, 1995). Contributing factors such as well screen length, purging procedure, vertical solute concentration profile in the formation and the hydrogeological characteristics of the aquifer were cited. Factors such as chemical reactions, biological processes, adsorption phenomena and vertical groundwater flow combined with vertical and longitudinal dispersion also reported to cause the disparities between monitoring well and aquifer concentration (Barcelona and Herfrich 1986, Herzog *et al.*, 1988)

Groundwater sampling is conducted to provide accurate information of subsurface water resources. The reliable detection and assessment of groundwater contamination require minimal or no disturbance of geochemical and hydrogeological conditions during sampling. Validity of comparison between aquifer solute concentration and monitoring wells based to representative or error free sampling procedure.

In this paper our effort focused to compare solute concentration of extracted mixed-water from a partially penetrating monitoring well and calculated aquifer solute concentration as a result of numerical methods. The aquifer solute concentration calculated as a combination of a finite element method for the flow and transport simulation in a vertical section aligned with principal direction of flow and an analytical solution for describing the potential φ and the velocity field around a pumping well. On the other hand, groundwater samples are collected from partially penetrating monitoring wells and are mixed-water samples. These groundwater samples were analyzed in the laboratory using the standard methods (Hem 1995). It must be noted that this work focused to nitrate pollution control and nitrate concentration in collected samples were measured with a portable equipment of type Eijkelkamp (Marckoquant 10020) and sporadically verified using analytical methods.

The nitrate pollution is the non-point pollution problem, a significant part of which is the contamination of groundwater resources, has its main origin in agricultural practices. The use of fertilizers to supply plan nutrients has caused nitrate contamination of groundwater in numerous rural areas of the world. The increased fertilizers for the increased agricultural production must combine detrimental effect on the environment. Suggestively note that in Greece the nitrogen fertilizers increased from 4.4 Kg/ha to 100 Kg/ha in the period 1950-1990 (Ministry of Environment, Planning and Public Works 1995).

2. Site characterization

The contaminant plume considered in this paper occurs in a coastal, shallow, phreatic aquifer which is located in the south-western part of Chalkidiki Peninsula in Northern Greece and concern an area called Kalamaria Plain (Fig. 1). The aquifer under study is a part of large watershed, which drains to Thermaikos Gulf. A large part of the area is used as a agricultural land. An intense agricultural activity during the last decades sustained by heavy nitrogen fertilization has caused severe nitrate contamination in groundwater extracted from shallow irrigation wells and deeper boreholes.

The structure and hydrogeology of the system are typical of a coastal multiaquifer system in Greece. The upper part, which is the object of this study is characterized by alternating sands and gravels with small pebbles in some places. These formations are underlain by clayey layers at depths that vary from 5 to 25 meters. Geophysical prospecting methods were used in combination with lithological data from various wells in order to estimate the thickness of various layers and consistency of the geological deposits. The groundwater flow direction is towards the coast and the average gradient of the water table has been found by recent groundwater level measurements to be an order of 0.006.

A number of pumping tests performed in the aquifer leads to a range of hydraulic conductivity values between 4 and 45 m/d in the permeable strata

Finally additional data, particular from geology and hydrogeology and land use can, by found in previous publications (Latinopoulos *et al.*, 1994, Xefteris 2000, Anastasiadis and Xefteris 2001)

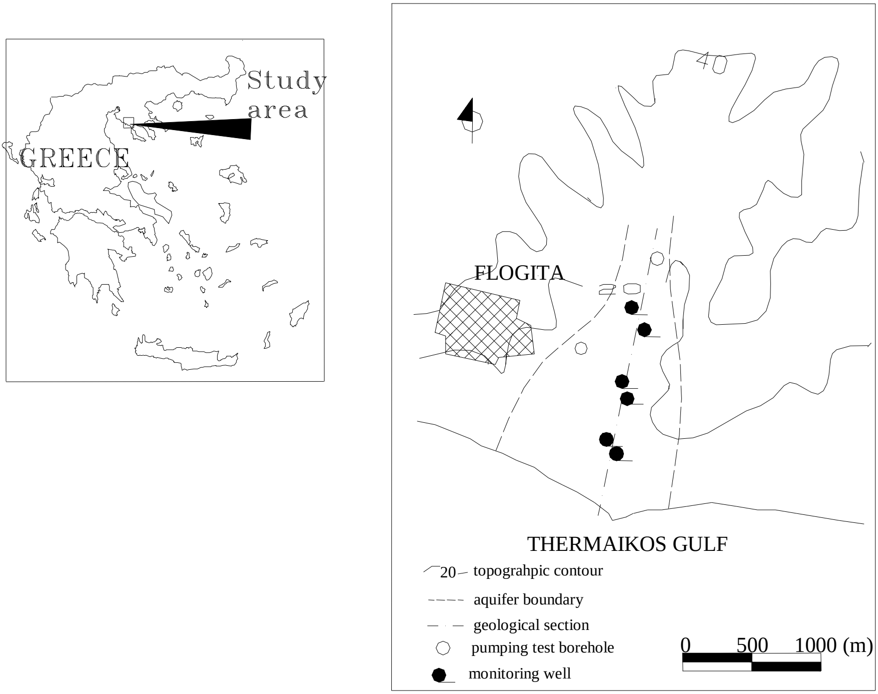

Fig. 1. Location of the study area
Rys. 1. Położenie miejsca badań

Kalamaria plain is located in the western part of Chalkidiki Peninsula, southeastern of the city Thessaloniki. The western part of the Chalkidiki Peninsula belongs to the Axios tectonic zone and is a part of Axios-Thermaikos Basin. Axios zone has been divided into three zones (Marcier 1966) and the western part of Chalkidiki Peninsula included to the eastern zone, named Peonia (Fig. 2).

Axios zone is build up of Mesozoic metamorphic rocks (epigneisses, metadiabase, schists, phyllites, quartzites and marbles), opthiolites with gabbro intrusions, granitoids, limestones, sandstones and conglomerates. In Kalamaria plain Neogene sediments have a significant extension as well as the quaternary deposits. The Neogene sediments are consisting of alternated beds of sandstones, conglomerates, marls, limestones, marly limestones, sands, and red to brick red clays. Finally, recent coastal deposits occur along the coast and they are consisting of sand and silts.

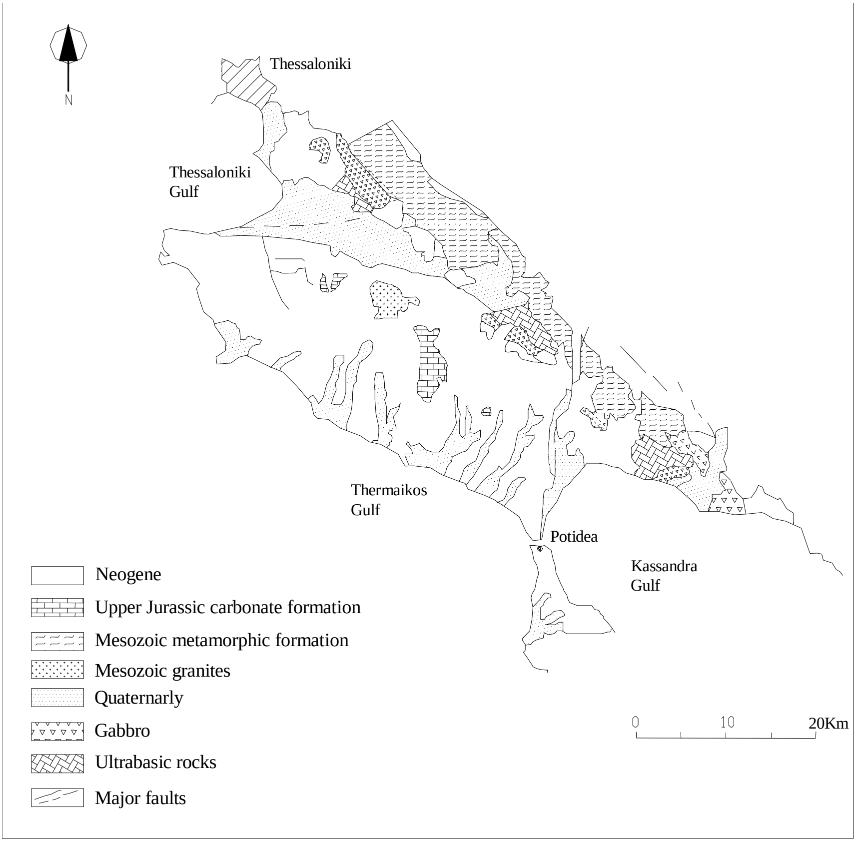

Fig. 2. Geological formation (after the geological map of Greece)
Rys. 2. Formacje geologiczne (za mapą geologiczną Grecji)

3. Solute transport analysis

The governing equation for advective-dispersive transport in a cross sectional flow system is:

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial x}\left(D_{xx}\frac{\partial C}{\partial x}\right) + \frac{\partial}{\partial z}\left(D_{zz}\frac{\partial C}{\partial z}\right) - \left(V\frac{\partial C}{\partial x}\right) - \lambda C \qquad (1)$$

where Dxx, Dzz are the dispersion coefficients, V the average pore water velocity, C the concentration, λ the decay coefficient as first order reaction (λ=ln2/t*, where t* is the half life of reactor).

In the principal direction formulation the dispersion coefficients are defined as:

$$D_{xx} = \alpha_L V + D^* \qquad (2)$$
$$D_{zz} = \alpha_T V + D^*$$

where, α_L and α_T are the longitudinal and transverse dispersivities respectively and D* is the coefficient of molecular diffusion.

Numerical Simulation and results

Numerical simulation is based on Galerkin finite element formulation (Pinder and Gray 1977, Pinder and Frind 1972) combined with a coordinate transformation which forms the common basis of the principal direction method (Frind and Germain 1986, Burnett and Frind 1987, Leismann and Frind 1989, Anastasiadis 1999)). The admissible formulations are a solid-body rotation and a symmetric deformation of the element with the local element axes remaining orthogonal. The symmetric orthogonal deformation can be handled by using an isoparametric transformation. The typical grid type for the principal direction model obtained with elements using the symmetric orthogonal deformation conforms to that of flow net consisting of streamlines and equipotential lines (or pseudopotential lines in the case of anisotropic media) meeting at right angles.

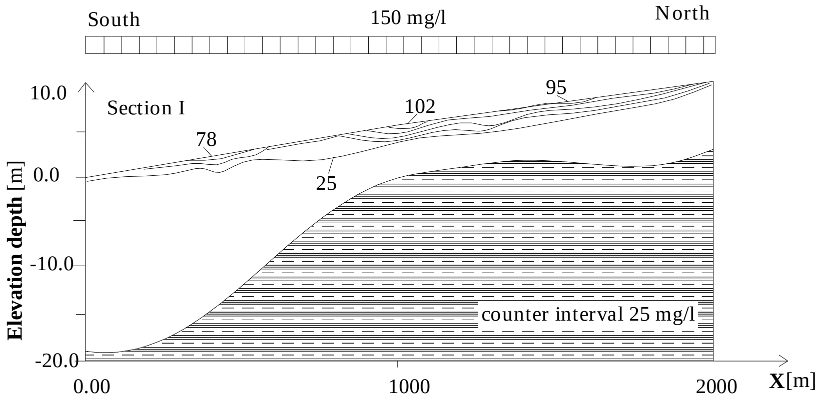

Fig. 3. Simulated nitrate concentrations in a vertical section
Rys. 3. Symulacja stężeń azotanów w przekroju pionowym

A typical data set of hydrologic and hydrogeologic data, required to solve the simulation problem. The long-term precipitation leads to an estimate of the annual uniform recharge of 100 mm. Two zone heterogeneity is assumed where sand layers are characterized by a hydraulics conductivity of 4.3 m/d and grovels by a value of 43 m/d, anisotropy is also taken into account by employing a ratio of 10:1, for the dispersivities in the principal directions x and z. The dispersivities are given the values of $\alpha L=10$ m and $\alpha T=0.001$ m using information from the available literature (Kinzelbach *et al.*, 1991, Frind *et al.*, 1990). The decay coefficient, λ is calculated with $t^*=0.5$ year. The molecular diffusion is negligible. The applied fertilizers lead to an uniform influx nitrate concentration of 150mg/l along the whole cross section of the aquifer.

In Figure 3 simulated distribution of nitrate concentrations is presented for vertical section along the principal direction of flow. Since the time period required to reach a steady state situation is of a few years only, we assume that the distributions shown in this figure are representative of the present conditions.

4. Flow and transport near a partially penetrating well

The disparity between nitrate concentration in the monitoring wells and in the aquifer will be explored through an analytical flow model for partially penetrating well and numerical results from the transport model. The integration of vertical distribution of nitrate concentration combined with the vertical distribution of hydraulic head around to partially penetrating well was estimated following next steps and assumptions.

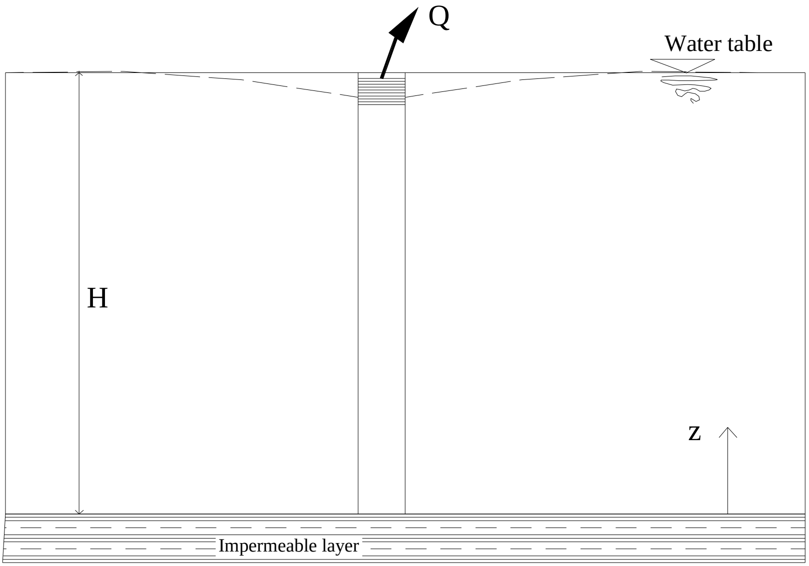

Fig. 4. Schematic of a partially penetrating well
Rys. 4. Shemat studni częściowo penetrującej

A homogeneous, horizontal isotropic aquifer is considered (Fig 4). We note that any horizontal to vertical anisotropy can be handled by the change of variables to an isotropic system with the conversion (Strack 1989):

$$Z=\beta\, Z^* \tag{3}$$

where, $\beta = \sqrt{K_H / K_V}$ with K_H and K_V are the hydraulic conductivity in horizontal and vertical direction respectively and (X,Y,Z*) are the actual coordinates and (X,Y,Z) are the coordinates in the anisotropic system. The equivalent hydraulic conductivity is given as:

$$K = \sqrt{K_H . K_V} \tag{4}$$

The analytical solution describing the potential, Φ, within an unconfined domain containing point source or point sink (Polubarinova-Kochina 1962, Muskat 1982) can be expanding for the confined case with small pumping (Q) rate or large saturation thickness (H) or large hydraulic conductivity (K). The above assumption is described with the following inequality (Mac Donald and Kitanidis, 1993):

$$q = \frac{Q}{KH^2} << 1 \tag{5}$$

where, Q is the pumping rate, H is the undisturbed saturated thickness and K is the hydraulic conductivity.

The hydraulic head in an infinite domain is given as (Polubarinova-Kochina 1962):

$$\varphi(x,y,z) = \frac{q}{4\pi}\left[\frac{1}{\sqrt{(x-x_w)^2+(y-y_w)^2+(z-z_w)^2}} + \frac{1}{\sqrt{(x-x_w)^2+(y-y_w)^2+(z+z_w)^2}}\right] \tag{6}$$

where, (x_w, y_w, z_w) is the location of a point source/sink and the system is nondimensionalized as follows:

$$x = \frac{X}{H} \qquad y = \frac{Y}{H} \qquad z = \frac{Z}{H}$$

$$\varphi = \frac{\Phi}{H} \qquad q = \frac{Q}{KH^2} \tag{7}$$

The velocities for a pumping well are given as:

$$\begin{aligned} V_x^* = K\frac{\partial \varphi}{\partial x} = -K\frac{q}{4\pi}[&\frac{(x - x_w)}{[(x - x_w)^2 + (y - y_w)^2 + (z - z_w)^2]^{2/3}} \\ &+ \frac{(x - x_w)}{[(x - x_w)^2 + (y - y_w)^2 + (z + z_w)^2]^{2/3}}] \end{aligned} \tag{8}$$

$$\begin{aligned} V_y^* = K\frac{\partial \varphi}{\partial y} = -K\frac{q}{4\pi}[&\frac{(y - y_w)}{[(x - x_w)^2 + (y - y_w)^2 + (z - z_w)^2]^{2/3}} \\ &+ \frac{(y - y_w)}{[(x - x_w)^2 + (y - y_w)^2 + (z + z_w)^2]^{2/3}}] \end{aligned} \tag{9}$$

$$V_x = \frac{V_x^*}{H} \qquad V_y = \frac{V_y^*}{H} \tag{10}$$

and the horizontal velocity of groundwater around a pumping well is:

$$V = \sqrt{V_x^2 + V_y^2} \tag{11}$$

Vertical distribution of velocities around a pumping well was determined using the above equation. Combined with the nitrate concentration as it was evaluated by numerical methods, has described the nitrate concentration in the pumping water (mixed-water).

5. Results and Discussion

A data set of hydrogeologic data, required to estimate the aquifer nitrate concentration problem. Two zone heterogeneity that is assumed can be replaced by a homogeneous with hydraulics conductivity of 25 m/d. Anisotropy is also taken into account by employing a ratio of 10:1. The pumping charge has not overcome 5 m^3/h and is valid the constrain of equation (6) for a mean saturated thickness of the shallow phreatic aquifer 10m.

$$q = \frac{Q}{KH^2} = 0.048 \langle\langle 1$$

Application of the above methodology to cross section of the study area (Fig. 1), calculated the nitrate concentrations near the locations of the monitoring wells. The comparison between the available data from the monitoring wells and the calculated nitrate concentration in the groundwater is presented in Figure 5. The extreme variations of the measurement nitrate concentration from the calculated nitrate concentration are caused mainly by the assumption of the steady-state flow in the aquifer and steady - state and uniform influx nitrate concentrations along the whole cross section of the aquifer. On the other hand it must be noted that the samples from the monitoring wells follow a random temporal distribution.

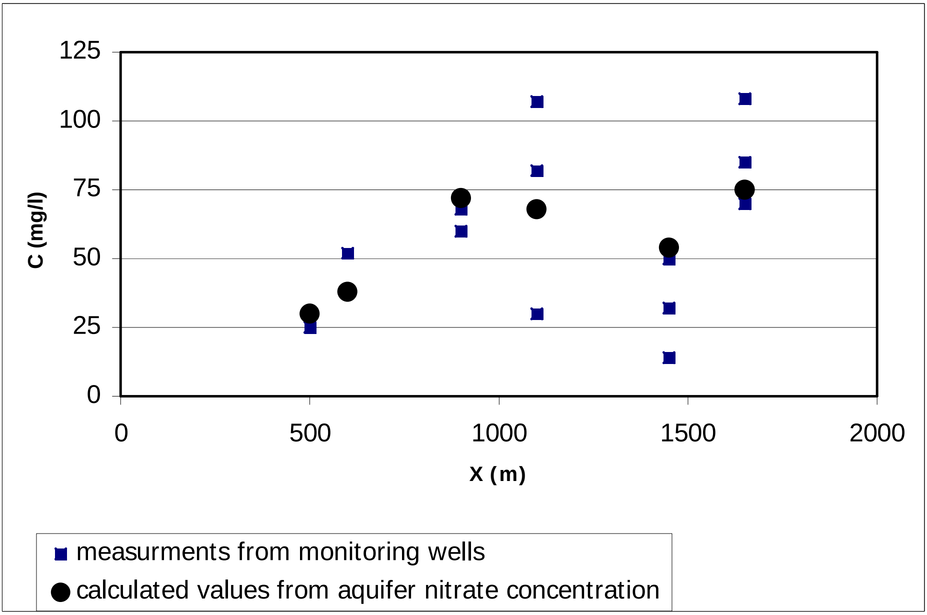

Fig. 5. Comparison results between calculated aquifer nitrate concentration and that in monitoring wells across the vertical section

Rys. 5. Porównanie wyników obliczeń stężeń azotanów z danymi ze studni monitoringowych rozmieszczonych wzdłuż przekroju pionowego

Field data from pumping wells (mixed water) is not an indicator for the vertically averaged groundwater concentration in the aquifer, but indicate the order of the pollution magnitude. The relationship between monitoring wells and aquifer nitrate concentration depends on the available field data. Nitrate as non-point or distribute pollution must be handled with a special care and from all points of view.

References

1. **Martin-Hayden, J.M., and J.W. Robbins and R.D. Dristol:** Mass balance evaluation of monitoring well purging, Part II, Field tests at a gasoline contamination site. Journal of Contaminant Hydrology, vol. 8, pp. 225-241. 1991.
2. **Herzog, B. L., S.L. Chou, J.R. Valkenburg, and R.A. Griffin:** Changes in volatile organic chemical concentrations after purging slowly recovering wells, Ground Water Monitoring Review, vol. 8 (4), pp. 93-99. 1988.
3. **Barcelona, M. J. and L.A. Helfrich:** Well construction and purging effects on groundwater samples, Environmental Science and Technology, vol 20(11), pp.1179-1184. 1986.
4. **Hem, J. D.:** Study and interpretation of the chemical characteristics of Natural Water, U.S. Geol., Survey Water Supply paper 2254. 1985.
5. Ministry of Environment, Planning and Public Works, “Greece as an ecological and cultural fund. Data-Actions and Programs for environmental protection”, Athens, 1995.
6. **Chiang C., G. Raven and C. Dawson:** The relationship between monitoring well and aquifer solute concentration, Ground Water vol. 33 (5), pp. 718-726. 1995.
7. **Frind E.O., and Germain D.:** Simulation of contaminant plumes with large dispersivy contrast: Evaluation of alternating direction Galerkin models, Water Resour. Res., 22(13), pp. 1857-1873. 1986.
8. **Leisman M.H., and E.O. Frind:** A symmetric-matrix time integration scheme for the efficient solution of adrective-dispersion problems, Water Resour. Res., vol. 25(6), 1133-1139, 1989.
9. **Burnett R. and Frind O.E.:** Simulation of Contaminant transport in three dimension 2. Dimensionality effects, Water Resour. Res., vol. 25(4), 695-705, 1987.
10. **Anastasiadis P.:** Simulation of contaminant transport. Application of principal direction technique in a shallow aquifer, 6th Inter. Confer. On Environmental Science and Technology, Samos, Sept., 1999, pp. 194-201
11. **Pinder F., and Frind E.:** Application of Galerkin’s procedure to aquifer analysis, Water Res. Res., 8(1), 108-120. 1972.
12. **Pinder F., and Gray W.:** Finite element simulation in surface and subsurface hydrology, Academic, Orlando, Fla. 1977.
13. **Anastasiadis P., and A. Xefteris:** Control of nitrogen fertilizer pollution in groundwater, Fresenius Environmental Bulletin”, vol. 10 (5), pp. 501-505, 2001.

14. **Xefteris A.:** Investigation of groundwater quality degradation in Kalamaria plain with emphasis to nitrate contamination, Phd Thesis, Dep. Civil Engn. A.U.TH Greece, 2000.
15. **Mc Donald R.T., and P.K. Kitanidis:** Modeling the free surface of an unconfined aquifer near a recirculation well, Ground Water, vol. 31(5), 774-780. 1993.
16. **Strack, O.D.L.:** Groundwater Mechanics, Prentice-Hall Inc., Enlewood Clifs, N.J. 1989.
17. **Muskat, M.:** The flow of homogeneous fluids through porous media. International Human Resources Development Corp., Boston, MA. 1982.
18. **Polubarinova-Kochina, P.Y.:** Theory of groundwater movement (Translated by J.M.R. Dewiest), Princeton Univer. Press, Princeton, N.J. 1962.
19. **Kinzelbach, W., W. Schafer, and J. Herzer:** Numerical modelling of natural and enhanced denitrification process in aquifers, Water Resour. Res., vol. 27(6) 1123-1135. 1991.
20. **Frind, O.E., W. Duynisveld, O. Strebel and, J. Boettcher:** Modelling of multicomponent transport with microbial transformation in groundwater: The Furhberg case, Water Resour. Res., vol. 26(8), pp. 1707-1719. 1990.

Porównanie stężeń substancji rozpuszczonych w formacjach wodonośnych i studniach monitoringowych

Streszczenie

Związek pomiędzy stężeniem substancji rozpuszczonych w studniach monitoringowych a w warstwie wodonośnej ma duże znaczenie przy szacowaniu wielkości zanieczyszczenia zasobów wód gruntowych. Co ważniejsze, aby ocenić wpływ odcieków z nawadnianych obszarów rolniczych, na których występuje intensywne nawożenie na studnie pobierające wodę do picia, należy skorelować stężenia azotanów w warstwie wodonośnej ze stężeniami w wodzie do picia, tak aby poprawnie ocenić realne ryzyko. Do obliczeń przepływu i prędkości wody w pobliżu częściowo penetrującej studni zastosowano dwu wymiarowy model elementów skończonych przepływu itransportu w połączeniu z rozwiązaniem analitycznym. Zaprezentowano porównanie obliczonych stężeń azotanów i zmierzonych w studniach monitoringowych. Stwierdzono różnice pomiędzy stężeniem azotanów w warstwie wodonośnej a w studniach monitoringowych.

Związek pomiędzy stężeniem azotanów w studniach monitoringowych a w warstwie wodonośnej zależy od dostępnych danych. Azotany jako zanieczyszczenie nie punktowe lub obszarowe musi być traktowane bardzo poważenie z wielu punktów widzenia.