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Hydrogeochemical computer modeling of proposed artificial recharge of the upper Santa Fe Group aquifer, Albuquerque, New Mexico

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Abstract

Recently, the City of Albuquerque began investigating the feasibility of artificial recharge of its underlying upper Santa Fe aquifer. Artificial recharge by both subsurface injection and surface infiltration are under consideration. However, this paper concerns only recharge via injection wells. Proposed recharge fluids are (1) treated effluent from the Albuquerque wastewater treatment facility and (2) Rio Grande surface water. This study used the hydrogeochemical model PHREEQE to simulate subsurface injection. As a basis for comparison, simulations were also performed with data from the successful El Paso artificial recharge project. The results suggest that subsurface injection of Albuquerque treated effluent will be successful and that subsurface infiltration of Rio Grande surface waters will probably be successful. The simulations predict no significant adverse impact on water quality as a result of artificial recharge. However, other factors that may affect potability, such as trace-metal distribution, ion exchange, bacteriological content, and the presence of potential pollutants, are not considered by the model. Site-specific pilot testing and kinetics-based computer modeling are recommended to predict the useful life of recharge facilities. Equilibrium-based models, as used in this study, can only predict whether recharge is likely to be successful. They cannot determine the potential life of a recharge facility.

Introduction

Recently, the City of Albuquerque began investigating the feasibility of artificial recharge of its upper Santa Fe aquifer. This paper presents the results of hydrogeochemical modeling performed to simulate reactions that may occur within the aquifer as a result of artificial recharge. Both artificial recharge by subsurface injection and surface infiltration are under consideration. However, this paper only concerns the feasibility of recharge via subsurface injection. Numerous potential artificial recharge sites are under consideration by the City of Albuquerque. Because the aquifer mineralogy is similar throughout the Albuquerque Basin and because permeability variations are not considered by the computer models used (Hawley and Haase, 1992), it is feasible to model the entire aquifer as a single homogenous recharge site. Permeability variations are not considered by the model.

Both treated effluent from the Albuquerque southside wastewater treatment plant and surface water from the Rio Grande north of the city have been pro-

posed as potential sources of recharge water. Chemical analyses from both of these sources were used in modeling. Surface water was modeled with no treatment and effluent was modeled as presently treated because the exact nature of any further treatment for artificial recharge is unknown. These analyses serve as "end members" for potential mixtures of the two. However, no modeling was done in this study for a blended recharge water composed of treated effluent and surface water because the likely mix is as yet undetermined and modeling all possible permutations was not feasible.

Methods

When other measures of analytical uncertainty are not available, charge balances can be calculated to determine the likelihood of major errors in an analysis. There are two commonly accepted methods of calculating charge balance errors (CBEs) for waters (Fritz, 1994). The most common is described by the following equation:

$$\% \text{CBE} = \frac{\sum z \cdot m_c - \sum z \cdot m_a}{\sum z \cdot m_c + \sum z \cdot m_a} \cdot 100 \quad (1)$$

In Equation (1), z is the absolute value of the ion's charge, and m_c and m_a are molalities of the cationic and anionic species, respectively. Other workers prefer to divide the denominator in Equation (1) by 2 in order to relate the difference between the cation and anion equivalents to the mean. The equation then becomes:

$$\% \text{CBE} = \frac{\sum z \cdot m_c - \sum z \cdot m_a}{(\sum z \cdot m_c + \sum z \cdot m_a)2} \cdot 100 \quad (2)$$

Charge balance errors of less than 5% according to Equation (1), or 10% according to Equation (2) are generally considered acceptable. Analyses for use in hydrogeochemical modeling were selected with CBEs of less than 5% as calculated by Equation (1).

PHREEQE is a computer code designed to model aqueous geochemical reactions and is based on an ion pairing model. PHREEQE was written by Parkhurst et al. (1980) and is an extremely versatile code that can simulate the following types of reactions: mixing of two solutions, titrating one solution with another solution, adding or subtracting a net stoichiometric reaction, adding a net stoichiometric reaction until the phase boundary of a specific mineral is reached, equilibrating a solution with one or more mineral or gas phas-

es, changing temperature of a solution. Several PHREEQE options were used in this study including the mixing of two solutions, and changing the temperature of a solution, and equilibration with one or more mineral phases.

Hydrogeochemical modeling

The geothermal gradient in the Albuquerque basin is 30 to 40°C/km (0.009 to 0.012°C/ft) with a mean surface temperature of 13.5°C (Reiter et al., 1986). Therefore, the expected temperature range at a depth of 3,000 ft would be from 41 to 50°C, and the expected temperature range at a depth of 5,000 ft would be from 59 to 75°C. It is unlikely that subsurface injection will occur in Albuquerque at depths greater than a few thousand feet. However, to be thorough, the effects of mixing treated effluent with two end-member ground waters were calculated over a temperature range of 25 to 75°C.

Recharge via subsurface injection was modeled with PHREEQE by first simulating the mixing of the injected water and the ground water to see if mineral precipitation might occur. After mineral saturation was examined, an additional simulation was made that equilibrated the solution with the minerals most likely to precipitate at equilibrium. The results of this second run gave the mass of each mineral likely to precipitate. From these data, the number of pore volumes of solution passing through a control volume of aquifer matrix necessary to cause a 10% reduction in porosity caused by precipitation was calculated for each significant mineral. The 10% porosity reduction was chosen only as a basis for comparison. Any other porosity reduction factor would work as well. Additional simulations were performed with data from the El Paso artificial recharge project which has been underway successfully since May 1985. The El Paso subsurface injection simulations serve as a reality check on the outcome of hydrogeochemical modeling of artificial recharge for the City of Albuquerque.

Chemical analyses

Two ground-water analyses were chosen for use in hydrogeochemical modeling: (1) an analysis with the lowest total dissolved solids (TDS) (Charles 3, 09-28-93) and (2) the analysis with the highest TDS (Coronado 1, 04-06-93). No analytical

TABLE 1—Chemical analyses used in this study. CBEs reported in this table were calculated using Equation (1).

Water	pH	Temp. °C	Eh (mv)	TDS (mg/l)	Ca (mg/l)	Mg (mg/l)	Na (mg/l)	K (mg/l)	Total Fe (mg/l)	SiO ₂ (mg/l)	HCO ₃ (mg/l)	SO ₄ (mg/l)	F (mg/l)	Cl (mg/l)	NO ₃ (mg/l)	SI Calcite	SI Hematite	% CBE
Albuquerque treated effluent (average of 5 analyses)	7.00	25.0	-	511	43.6	6.36	13.2	21.8	0.21	27.2	165.8	93.2	1.2	61	69.4	-0.64	8.57	1.4
Albuquerque Charles 3 ground-water (09-28-93)	7.81	18.4	5.26	192	34.6	2.98	15.4	1.44	0.0091	27.9	104	30.3	0.44	9.99	3.05	-0.14	15.7	-0.28
Albuquerque Coronado 1 Groundwater (02-02-92)	8.01	22.8	2.98	432	29.2	7.65	76.1	6.42	0.0326	59.6	118.8	40.5	0.98	76.8	0.58	0.06	16.5	4.8
San Felipe surface water (09-07-1990)	8.42	16.0	-	248	45.0	7.90	23.0	3.20	0.008	19.0	151	63.0	0.40	7.20	0.443	0.71	13.5	3.4
San Felipe surface water (06-04-1991)	8.24	21.0	-	160	31.0	5.30	12.0	2.00	0.04	15.0	96.0	45.0	0.30	1.80	0.354	0.16	13.5	-0.51
El Paso treated effluent (040494-9)	7.80	25.0	-	633	54.2	3.9	163.2	15.62	-	26.0	199	76.3	0.86	179.3	23.92	0.31	-	0.70
El Paso ground water EPWU-56 (09-05-1979)	8.04	25.0	-	626	50.0	9.7	150	5.7	-	37.0	110	40.0	0.54	260	4.4	0.27	-	-0.40
El Paso recovered water OB-4B (29-09-1991)	7.56	25.0	-	520	48	9.0	120	6.5	-	33	148	70	0.40	150	9.7	-0.09	-	1.4



FIGURE 1—Location map of study area showing City of Albuquerque and the USGS San Felipe gaging station on the Rio Grande.

uncertainties were available for the Albuquerque analyses. However, CBEs were computed for each analysis by Equation (1) (Table 1).

Surface-water analyses from the United States Geological Survey (USGS) San Felipe gaging station records were chosen as most appropriate for modeling. The San Felipe gaging station is north of the City of Albuquerque (Fig. 1). In a report for the United States Bureau of Reclamation (USBR), Harper (1993) stated that TDS concentrations do not vary greatly during the year except during the high-flow months of April, May, and June. Therefore, two surface-water analyses from the San Felipe gaging station were chosen. The first was from the high-flow, low-TDS time (06-04-91) and the second from a more normal flow time with higher TDS (09-07-90). No analytical uncertainties were available for these analyses.

TABLE 2—Analytical uncertainties of treated effluent as calculated from five replicates ($n = 5$).

Analysis	Mean	SD	\pm tSD (95% probability)
Bicarbonate	165.8 ppm	1.10 ppm	3.06 ppm
Calcium (Ca)	43.6 ppm	0.55 ppm	1.53 ppm
Chloride (Cl)	61.0 ppm	4.53 ppm	12.6 ppm
Fluoride (F)	1.20 ppm	0.10 ppm	0.278 ppm
Hardness (CaCO ₃)	135.0 ppm	1.00 ppm	2.78 ppm
Iron (Fe)	0.426 ppm	0.489 ppm	1.36 ppm
Magnesium (Mg)	6.36 ppm	0.090 ppm	0.250 ppm
Nitrate (NO ₃)	69.4 ppm	1.14 ppm	3.17 ppm
Potassium (K)	21.8 ppm	5.94 ppm	16.5 ppm
Silica (Si)	27.2 ppm	1.31 ppm	3.64 ppm
Sodium (Na)	103.2 ppm	2.05 ppm	5.70 ppm
Sulfate (SO ₄)	93.2 ppm	2.28 ppm	6.34 ppm

Complete chemical analyses of the City of Albuquerque treated effluent were not available. Consequently, the treated effluent was sampled, and five identical samples were analyzed by the New Mexico Bureau of Mines and Mineral Resources (NMBMMR) Chemistry Laboratory (Table 1). Analytical uncertainties determined for these analyses are shown in Table 2.

Many chemical analyses report certain parameters as being less than a given analytical detection limit. Parameters reported at the detection limit normally should not be used in hydrogeochemical modeling programs because analytical uncertainty near detection limits is often in the vicinity of $\pm 100\%$. For sparsely soluble elements such as aluminum, typically present in natural waters in concentrations between 0.1 and 1.0 $\mu\text{g/l}$ (Drever, 1988), the detection limits for some analytical techniques may be far above natural solubility. If detection limits for aluminum are entered as analytical values, often the equilibrium calculations falsely indicate that the solution is supersaturated in many species of aluminum-bearing min-

erals—including the clay minerals. Consequently, the results could be quite misleading.

When performing mixing calculations, it is important to have complete analyses of both solutions. If a value for one solute is entered for solution 1 but not for solution 2, the computer code assumes the solute concentration in the second solution to be zero. If the solute in question is really present in solution 2, the results of mixing calculations will be in error—possibly significantly.

Sensitivity analysis

Unless uncertainty inherent in hydrogeochemical modeling is quantified, data interpretation may be inaccurate. Sensitivity analyses are generally performed by changing input to a computer model and determining the resulting variation in the output. For this study, analytical uncertainties of the chemical analyses were entered to quantify the expected analytical uncertainty (precision) of the model results.

Analytical uncertainties for chemical analyses were only available for Albuquerque treated effluent (Table 2). For this reason the assumption was made that the analytical uncertainties for all of the analyses are comparable to those of the treated effluent. The NMBMMR Chemistry Lab analyzed five replicates of treated effluent. Each of these five analyses was entered into the model PHREEQE, and saturation indices were calculated for calcite and hematite. Mean and standard deviation (SD) of saturation indices (SI) were calculated for each mineral (Table 3).

For small populations ($n < 40$) the sample SD can be quite variable and possibly misleading (Weldon, 1986). A better approach is to use Student's *t*-Distribution to ascertain uncertainty levels for small populations. The procedure is to multiply the value of *t* for the given degrees of freedom ($n-1$) by the SD for a given uncertainty level. For $n = 5$, *t* at the 95% probability level is 2.78 (Dick, 1973).

The uncertainties (\pm tSD) in saturation indices shown in Table 3 apply to single solutions. For modeling runs which involve mixing two solutions, the uncertainty is greater. Errors in multiplication or division operations are generally proportional to the square root of the sum of the squares of the errors. Saturation index uncertainties for mixing runs were estimated by the following equation

$$P = \sqrt{2E^2} \quad (3)$$

where *P* is the percentage uncertainty at the 95% probability level, and *E* is the analytical uncertainty (\pm tSD) given in Table 3 for the particular mineral.

Other uncertainties, including inherent errors in the thermodynamic database, systematic (non-random) errors in the laboratory data, and density variations in the minerals themselves, may also be present. However, it is not within the scope of this study to quantify these additional uncertainties.

Modeling results are reported in numbers of pore volumes of injected artificial recharge required to cause a 10% porosity reduction in the aquifer. The equation used to calculate the number of pore volumes necessary to achieve any given percent porosity reduction caused by precipitation of a given mineral is derived as follows. Because the waters of interest are dilute, the solution density is assumed to be 1.0 g/cm³. The control volume is defined as the pore space which occupies 1,000 cm³ (i.e., contains 1 kg of water). An expression is then derived which defines how many cubic centimeters one mole of the chosen mineral occupies (molar volume)

$$\frac{W_i}{D_i} = \frac{\text{g}}{\text{mole}} \cdot \frac{\text{cm}^3}{\text{g}} = \frac{\text{cm}^3}{\text{mole}} \quad (4)$$

where W_i is the gram formula weight in g/mole and D_i is the density in g/cm³ of mineral *i*.

TABLE 3—Results of sensitivity analysis on saturation index (SI) for treated effluent ($n = 5$).

Mineral	Mean SI	SD	\pm tSD (95% probability)
Calcite	-0.6387	0.0287	0.0798
Hematite	8.8625	0.6032	1.6769

Now the relationship between a particular reduction in porosity (P_R expressed as a fraction) and the precipitate volume *V* in cm³ necessary to achieve it can be defined. Because the control volume is 1,000 cm³, then

$$V = 1000P_R \quad (5)$$

Next the number of moles occupied by *V* is determined.

$$M_V = \frac{V}{W_i/D_i} = \frac{1000 \cdot P_R \cdot D_i}{W_i} \quad (6)$$

Dividing Equation (6), which is stated for a control volume of 1,000 cm³, by the amount of mineral *i* that will precipitate P_i (moles/1,000 cm³) upon reaching equilibrium, yields the following expression for the number of pore volumes of solution P_V that must pass through the control volume in order to effect the chosen porosity reduction

$$P_V = \frac{1000 \cdot P_R \cdot D_i}{W_i |P_i|} \quad (7)$$

The absolute value of P_i is used in Equation (7) because precipitating minerals are indicated by negative signs in the model PHREEQE.

To determine the relationship between the analytical uncertainties in saturation indices and uncertainty in pore volume calculations, it is necessary to define the relationship between the two quantities. When SI = 1.0, the solution is 10x supersaturated. If SI = -1.0, then the solution is 10x undersaturated. Given that the analytical uncertainties in SI listed in Table 3 are represented by $E_{r,i}$ (\pm tSD), where *i* represents the mineral of interest, then

$$P_{h,i} = P_i \cdot 10^{E_r} \quad (8)$$

and

$$P_{L,i} = \frac{P_i}{10^{E_r}} \quad (9)$$

where P_i is moles of precipitate per kg of solution. Furthermore, since then the absolute value of the uncertainty can be calculated by

$$E_{PV} = \left| \frac{1000 \cdot P_R \cdot D_i}{W_i} \left(\frac{1}{P_{h,i}} - \frac{1}{P_i} \right) \right| \quad (10)$$

Equation (7) is used to calculate the values of pore volumes required for a 10% porosity reduction shown in the figures for the minerals of interest from the moles/kg of precipitate output by the

model. Equation (10) is used to calculate the uncertainties at the 95% confidence interval as shown by the error bars in the figures.

Results and discussion

PHREEQE is an equilibrium model. In other words, the results assume theoretical chemical-equilibrium conditions. However, most ground waters are not at equilibrium. Drever (1988) states that many natural waters may closely approach equilibrium with secondary clay minerals; however, these waters rarely achieve equilibrium with the primary igneous minerals in the aquifer matrix with the possible exception of K-feldspar.

The degree to which a water attains equilibrium with a mineral suite is a function of contact time. Minerals exhibiting relatively fast reaction kinetics, such as calcite, the clay minerals, and possibly K-feldspar, will react more rapidly, and thus equilibrium with these minerals will be reached sooner. During artificial recharge, flow rates through porous media will be more rapid than normal ground-water flow. Therefore, contact times will be shorter in artificially recharged systems than in an undisturbed aquifer. For this reason, equilibration with primary igneous minerals was not modeled in this study.

The model used in this study does not have a spatial component. Therefore, it is not possible to predict the location of mineral precipitation or the useful life of a recharge facility from model results.

PHREEQE was used to model artificial recharge by subsurface injection in two steps. First, a mixing simulation that did not allow equilibration with minerals was performed. Next, the saturation indices of all minerals in PHREEQE's database were examined to see which ones were supersaturated in the waters and therefore might precipitate. Once the supersaturated minerals were identified, PHREEQE simulations allowing mineral precipitation were performed.

In general, PHREEQE precipitates the more supersaturated minerals first. As a result, often only the most supersaturated minerals are projected to precipitate. For example, in each of the subsurface injection simulations, the waters were grossly supersaturated in hematite (Fe₂O₃) and goethite (FeOOH), with hematite consistently the most supersaturated. When runs were made equilibrating the solutions with both hematite and goethite, only hematite was predicted to precipitate at equilibrium. When simulations allowed only goethite to precipitate, no precipitation occurred. Hematite does not precipitate on short time scales because of slow reaction kinetics (Pankow, 1991). However, amorphous iron hydroxide (Fe(OH)₃) does. In interpreting these results, hematite precipitation is considered equivalent

to amorphous iron hydroxide precipitation.

Simulation of subsurface injection of treated effluent into Charles 3 and Coronado 1 ground waters predicted that only iron minerals will precipitate (Fig. 2). Calculations suggest that the maximum precipitation of iron will occur when the solution consists of 100% treated effluent and that a minimum of 1,870,000 \pm 1,760,000 pore volumes of injected fluid are required to reduce aquifer porosity by 10%. Even though the uncertainty is extremely large, there seems to be little possibility that iron precipitation could effectively plug aquifer pores unless precipitation is concentrated in a particular area, such as the well screen.

Subsurface injection of oxygenated water into ground water with a low dissolved oxygen content and also Fe⁺⁺ causes the precipitation of iron hydroxide. This reaction is often bacteriologically mediated. Iron bacteria have the ability to remain attached to a support even at high flow velocities (Appelo and Postma, 1993), and the process of injection is unlikely to dislodge these bacteria. Liang et al. (1993) proposed that natural organic matter may play a role in the reduction of Fe⁺⁺⁺ and thus in the formation of iron hydroxides. They also stated that where the dissolved oxygen content of the water is high, the rate of iron hydroxide formation is rapid, and at least some of the colloidal portion of the iron hydroxide precipitate is probably transported into the aquifer. It is likely that injecting oxygenated water into the upper Santa Fe Group aquifer will result in iron hydroxide fouling the well screens. However, the geochemical modeling described in this study does not address bacterial reduction of iron hydroxide.

Simulations of surface-water injection suggest both hematite and calcite will precipitate. Figs. 3 and 4 show the effects of hematite and calcite precipitation on porosity. The number of predicted pore volumes necessary to cause a 10% porosity reduction as a result of hematite precipitation in the aquifer ranges from approximately 9,200,000 \pm 9,010,000 to 46,000,000 \pm 45,000,000. Considering the worst case, approximately 200,000 pore volumes would be required for a 10% porosity reduction. Iron precipitation is unlikely to plug a significant volume of aquifer pore space unless precipitation is concentrated very close to the well screen.

The volume of calcite precipitation is expected to increase with temperature. The greatest likelihood of significant calcite precipitation would be at depth where the temperature is greater. Calcite precipitation during surface-water injection is projected to require a minimum between 21,000 \pm 3,500 pore volumes at 25°C and 10,200 \pm 1,700 pore volumes at 75°C (Fig. 4).

The pH governs distribution of carbonate species in waters. Consequently, pH is

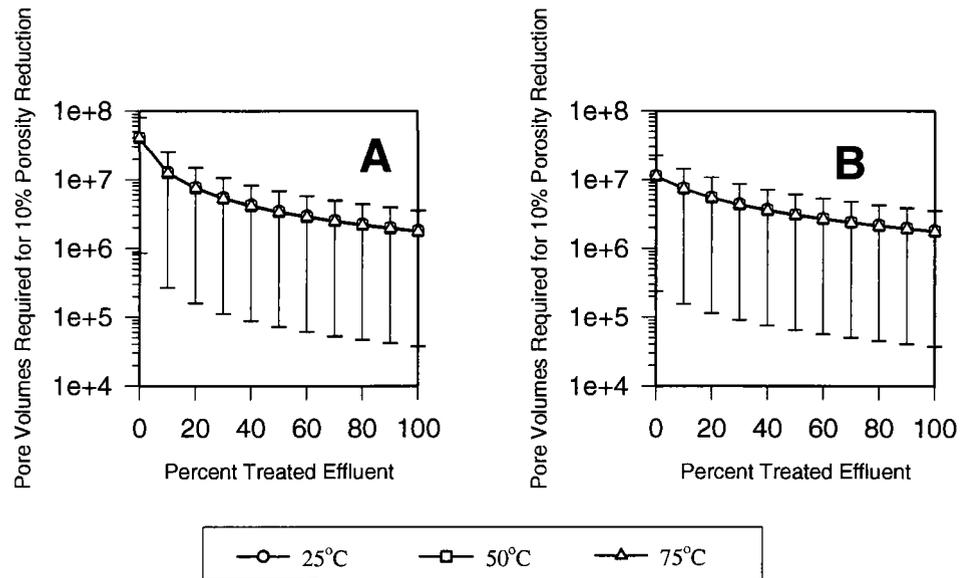


FIGURE 2—Pore volumes required to achieve a 10% porosity reduction caused by hematite precipitation during mixing of Albuquerque treated effluent and ground water from Charles 3 well (A) and during mixing of Albuquerque treated effluent and ground water from Coronado 1 well (B).

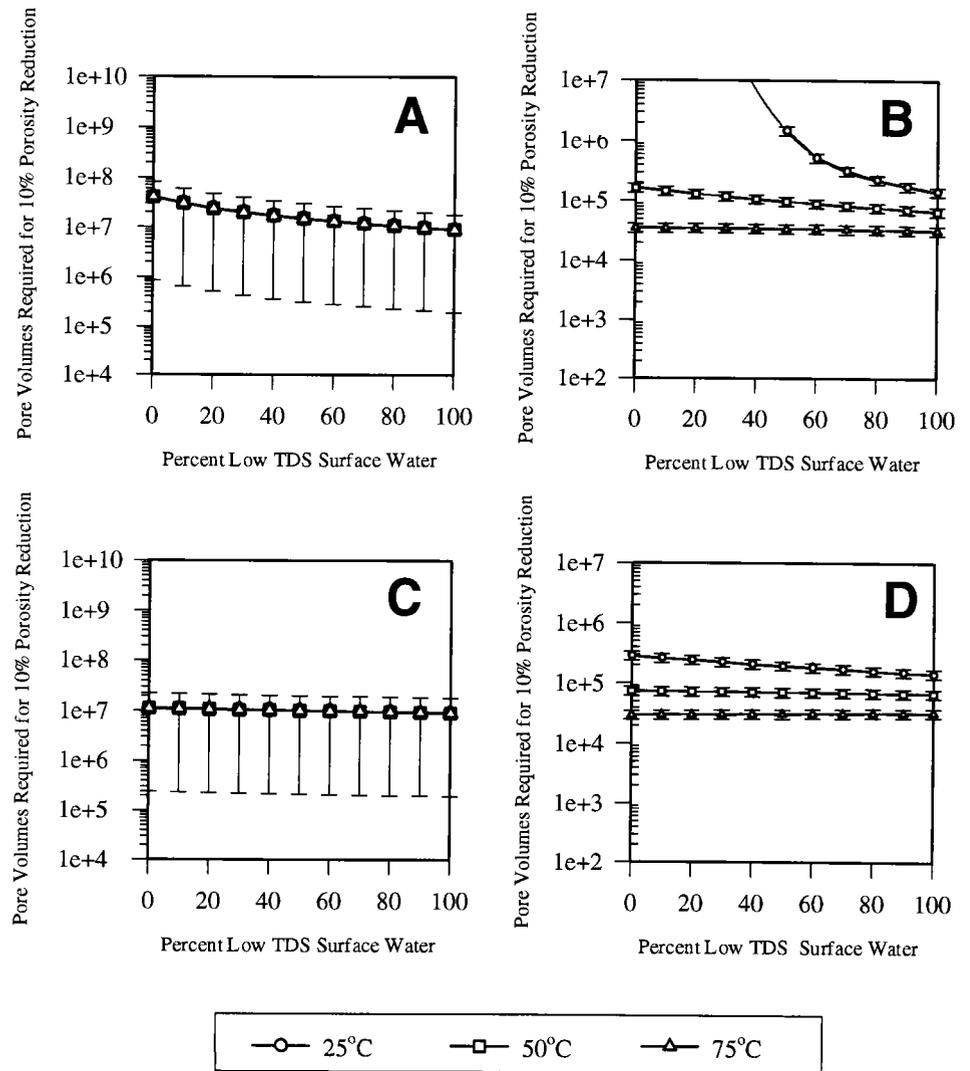


FIGURE 3—Pore volumes required to achieve a 10% porosity reduction caused by hematite precipitation (A) and calcite precipitation (B) during mixing of low TDS surface water and ground water from Charles 3 well and caused by hematite precipitation (C) and calcite precipitation (D) during mixing of low-TDS surface water and ground water from Coronado 1 well.

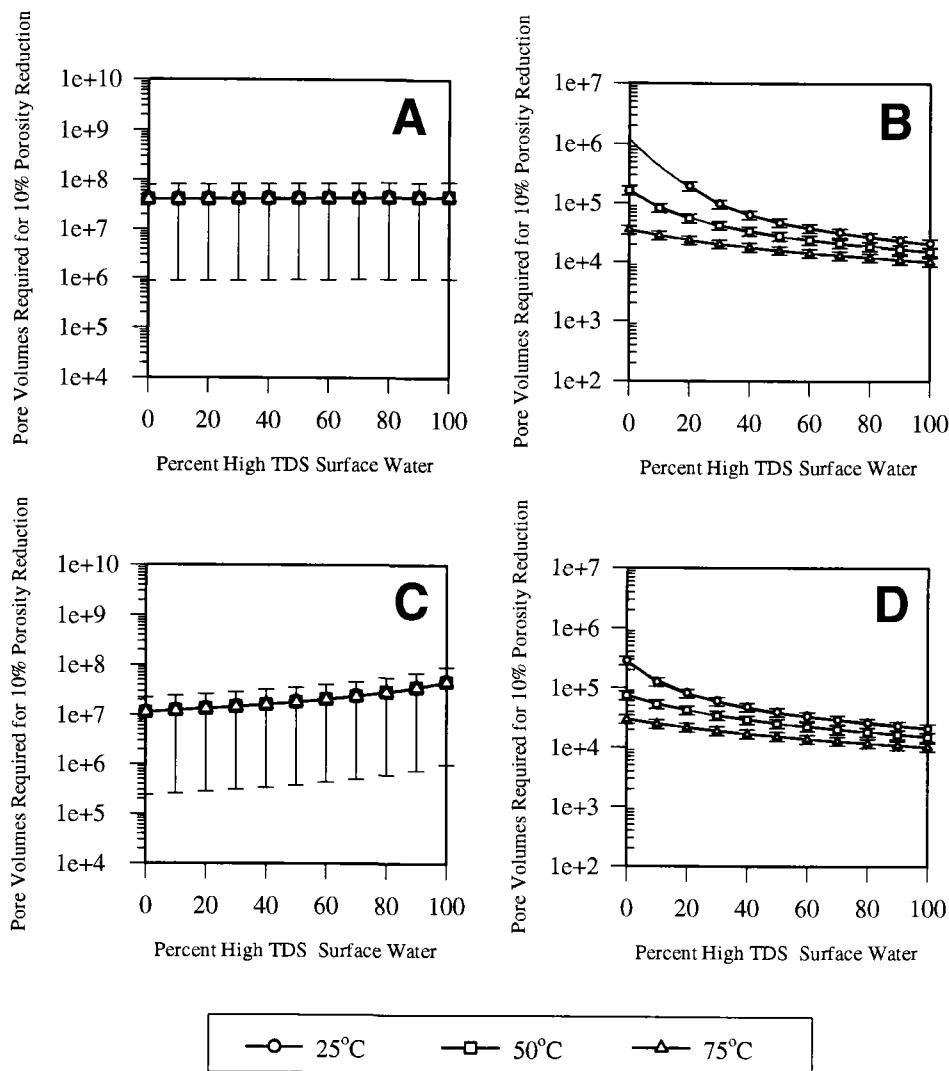


FIGURE 4—Pore volumes required to achieve a 10% porosity reduction caused by hematite precipitation (A) and calcite precipitation (B) during mixing of high TDS surface water and ground water from Charles 3 well and caused by hematite precipitation (C) and calcite precipitation (D) during mixing of high-TDS surface water and ground water from Coronado 1 well.

perhaps the most important consideration in predicting calcite precipitation. The pH values used in this study were those reported with the chemical analyses used. In each case, these were surface waters and were most likely in equilibrium with the CO_2 concentrations in the atmosphere. However, it is suggested that, once the actual injection waters are chosen, further hydrogeochemical modeling be performed to optimize pH ranges for best results during injection.

Calcite precipitation is inhibited by the presence of large amounts of Mg^{++} ions, phosphate compounds, and organic compounds in solution. Precipitation inhibition by Mg^{++} ions is apparently due to adsorption of the ion on the calcite surface, where it blocks subsequent crystal growth (Berner, 1975). Albuquerque treated effluent contains 6.36 ppm of Mg^{++} (0.262 millimoles/l) and 43.6 ppm (1.09 millimoles/l) of Ca^{++} . The molar $\text{Mg}^{++}/\text{Ca}^{++}$ ratio of the treated effluent is

0.24, whereas for seawater it is approximately 5.0. The $\text{Mg}^{++}/\text{Ca}^{++}$ ratio for Rio Grande surface water ranges from 0.29 to 0.31. It is uncertain if sufficient Mg^{++} is present in the recharge waters to significantly inhibit calcite precipitation. Pilot testing will be needed to determine this.

For phosphates, inhibition begins at concentrations of less than 1.0 ppm (Walter and Hanor, 1979; Mucci, 1986). Phosphate concentrations in Albuquerque treated effluent are approximately 0.8 ppm, and surface-water phosphate concentrations range from 0.03 to 0.09 ppm as orthophosphate. Consequently, it is possible that the presence of orthophosphate may inhibit calcite precipitation during artificial recharge in the Albuquerque Basin. For organic acids, inhibition effects begin at concentrations of less than 10 ppm (Reddy, 1977; Tomson, 1983). No information is available on organic acid concentrations in the proposed recharge waters.

Current hydrogeochemical models are incapable of considering inhibition effects on mineral precipitation caused by the presence of certain ions or acids. These inhibition effects only influence the rate of calcite precipitation and ultimately do not influence equilibrium activities.

Theoretically, the amount of precipitation that occurs in a given aquifer pore under steady state flow is a function not only of the saturation status of the solution with respect to the mineral phase, but also of how rapidly ground water passes through the pore space. For relatively short geological time scales (minutes to perhaps several hundreds or even thousands of years) precipitation reactions are often kinetically controlled, and the ground water may not attain chemical equilibrium. If water passes through the pore relatively quickly, then less of a given mineral is able to precipitate in that pore space per pore volume of water. Water injected into an aquifer passes radially out of the well. Consequently, velocity and pressure of injected water decrease with distance from the well. At some distance, where velocity has decreased and the water has a significant residence time relative to the reaction rate, a radial precipitation front may form if the aquifer is homogenous (Whitworth, 1995). The location of this precipitation front will depend on hydraulic conductivity, effective porosity, mineral saturation status and reaction kinetics, and recharge-well hydraulics. Therefore, equilibrium models, such as those used in this study, will overestimate the amount of mineral precipitation for a given control volume of aquifer where the water is moving relatively rapidly (i.e., close to the injection well), perhaps by several orders of magnitude. For this reason, precipitation volumes presented herein should be considered maximum estimates.

Different minerals may produce precipitation fronts in different locations. For example, if the rate of iron hydroxide precipitation is faster than the rate of calcite precipitation, then the iron hydroxide front would be expected to lie closer to the injection well than the calcite precipitation front (Whitworth, in press). Additionally, injected solids will tend to settle out of injected water as the pore velocity decreases radially away from the injection well. Quantification of precipitation and settling fronts should be performed on a site-specific basis; it is not within the scope of this study.

Before flow conditions for artificial injection reach steady state, mixing-front precipitation may occur. For example, if precipitation is predicted for a given mineral for mixtures between 40 and 80% of the injected fluid, this precipitation would be localized along a mobile, radial front where these mixing conditions occurred. However, as injected fluid gradually makes up greater and greater amounts of

the mixture, the importance of mixing-front precipitation should diminish. If the aquifer has a heterogeneous permeability distribution (as most do), then fingering or channeling of flow is likely to occur. If subsurface flow channeling does occur, then precipitation or settling fronts resulting from artificial injection will not be symmetrical about the injection well.

Because the City of El Paso has been successfully operating an artificial recharge project similar to the one planned by the City of Albuquerque, it is interesting model data from both areas. The first injection well for the El Paso artificial recharge project was drilled and tested prior to 1985 (Knorr and Cliett, 1985). Nine more injection wells have been drilled and placed in use since then. Full breakthrough of injected water in some water-supply recovery wells was achieved in 1990–1991 (oral communication, Paul Buszka, USGS, Indianapolis, Indiana). The El Paso artificial recharge project has successfully been underway since May 1985. The project was planned on the basis of a two-year recovery time from time of injection. All but one of the El Paso injection wells are currently in operation, although it is my understanding that the screens on all wells except the first one drilled have to be cleaned periodically, possibly because of iron hydroxide precipitation.

El Paso ground water generally has higher TDS than Albuquerque ground water. El Paso ground water ranges between approximately 500 to 800 mg/l TDS, whereas Albuquerque ground water ranges from approximately 250–400 mg/l TDS.

Simulated mixing of El Paso treated effluent and El Paso ground water suggests calcite will precipitate (Fig. 5). A 10% porosity reduction is calculated to result from injecting between 30,400 \pm 5,100 pore volumes at 25°C and 8,650 \pm 1,450 pore volumes at 75°C. However, injection of Albuquerque treated effluent is not projected to precipitate calcite. It appears that the injection of Albuquerque treated effluent should have fewer calcite precipitation problems than the El Paso artificial recharge project. Because the El Paso operation is successful, using higher-TDS water, injection of treated effluent in Albuquerque seems geochemically feasible.

Computer simulations predict that injecting surface water into the Albuquerque aquifer may lead to calcite precipitation. Calcite precipitation is projected to require a minimum between 21,000 \pm 3,500 pore volumes at 25°C and 10,200 \pm 1,700 pore volumes at 75°C to effect a 10% porosity reduction in the aquifer (Fig. 4). Thus calcite precipitation during surface-water injection should be greater than precipitation projected for injection of El Paso treated effluent. Therefore, significant pore clogging by calcite during injection of Albuquerque surface water is possible.

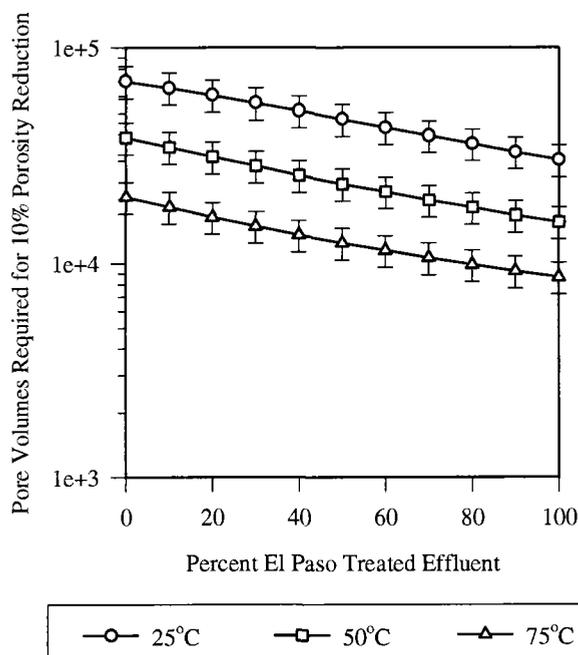


FIGURE 5—Pore volumes required to achieve a 10% porosity reduction caused by calcite precipitation during mixing of El Paso treated effluent and El Paso ground water.

Injection of treated effluent is projected to produce water with slightly higher TDS than natural ground water. For example, the projected TDS of an 80% treated effluent/20% ground water mixture is between 558 and 592 mg/l. These values are slightly above the EPA maximum contaminant level of 500 mg/l total dissolved solids for drinking water. In Albuquerque, natural ground-water TDS ranges between 192 and 432 mg/l. If additional treatment is applied to treated effluent prior to injection, the TDS of injected waters may be significantly reduced. An examination of Table 4 suggests mixing treated effluent or Rio Grande surface water with Albuquerque ground water should have no impact on the potability of the ground water. However, the model used does not consider trace metal distribution, bacteriological content, or the presence of organic pollutants that may adversely affect water quality.

Simulated ground water quality is tabulated in Table 4. The results are for an 80% injected water/20% ground water mixture. It is difficult to predict the exact mix of injected water and ground water in the aquifer resulting from artificial injection because of the presence of geologic heterogeneity. A small component of ground water is likely to remain in recovered water as result of dispersion and mixing, even after long periods of injection. Therefore the 80/20 mix was chosen arbitrarily.

Conclusions and recommendations

The projected calcite precipitation for subsurface injection of treated effluent in Albuquerque is less than that for the successful El Paso artificial recharge project.

Therefore, subsurface injection of treated effluent is not likely to cause sufficient calcite precipitation to prevent successful aquifer recharge for 10 yrs, the time the El Paso project has been successfully underway. However, subsurface injection of Rio Grande surface water is projected to cause greater calcite precipitation than the precipitation predicted for injection of El Paso treated effluent. The possibility of precipitation-induced limits to the life of surface-water injection wells does exist. Calcite precipitation increases with increasing temperature. Injection into deeper zones with warmer temperatures would be expected to produce greater precipitation of calcite. However, actual recharge waters will probably consist of a mixture of treated effluent and surface water. Therefore, the amount of calcite precipitation should be less than that predicted in this paper.

Hematite precipitation is also projected by the computer models. However, the reaction rate of hematite precipitation is quite slow, and it is more likely that bacteriologically mediated precipitation of amorphous iron hydroxide will occur. This process is capable of plugging well screens.

Site-specific pilot testing of precipitation reactions with the proposed recharge waters and/or kinetically based computer modeling is needed to ascertain the projected life of injection wells in the Albuquerque area. Equilibrium-based modeling, as done in this study, cannot make this prediction.

Mixing treated effluent or Rio Grande surface water with Albuquerque ground water is projected to have little impact on the potability of the ground water. Injection of treated effluent is expected to

TABLE 4—Simulated water-quality results. In each case the results listed here are for a mixture consisting of 80% recharge water and 20% ground water at 25°C after mineral precipitation was allowed.

Simulation	pH	TDS (mg/l)	Ca (mg/l)	Mg (mg/l)	Na (mg/l)	K (mg/l)	Total Fe (mg/l)	SiO ₂ (mg/l)	HCO ₃ (mg/l)	SO ₄ (mg/l)	F (mg/l)	Cl (mg/l)	NO ₃ (mg/l)
Subsurface injection of treated effluent into Charles 3 ground water	7.00	558	41.8	5.69	85.8	17.8	0.0002	27.4	180.5	80.7	1.05	88.3	56.2
Subsurface injection of treated effluent into Coronado 1 ground water	7.07	592	40.7	6.62	97.8	18.7	0.0041	33.7	182.9	82.7	1.16	105.3	55.7
Subsurface injection of low TDS surface water into Charles 3 ground water	7.94	195	31.2	4.84	13.0	1.89	0.0018	17.6	97.1	42.1	0.33	3.72	0.89
Subsurface injection of low TDS surface water into Coronado 1 ground water	7.96	229	30.0	5.77	25.2	2.88	→0	23.9	99.5	44.1	0.44	20.8	0.40
Subsurface injection of high TDS surface water into Charles 3 ground water	7.74	270	38.9	6.93	22.1	2.85	→0	20.8	133.0	56.5	0.41	8.05	1.33
Subsurface injection of high TDS surface water into Coronado 1 ground water	7.76	304	37.7	7.85	34.2	3.85	→0	27.2	135.5	58.5	0.52	25.1	0.47

produce recovered water with a higher TDS than Albuquerque ground water. However, the simulations do not consider trace-metal distribution, ion exchange, bacteriological content of the waters, or the presence of potential pollutants that may affect potability.

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recharge project in great detail and delineated which analyses were from unaltered ground water and which were from recovered injection water. John Hawley and Bill Haneberg aided in ways too numerous to list. This project was supported by a cooperative grant from the U. S. Bureau of Reclamation and the City of Albuquerque (#3 FC-40-14120).

References

- Appelo, C. A. J., and Postma, D., 1993, *Geochemistry, groundwater and pollution*: A. A. Balkema, Rotterdam, 536 pp.
- Berner, R. A., 1975, The role of magnesium in the crystal growth of calcite and aragonite from seawater: *Geochemica et Cosmochemica Acta*, v. 39, pp. 489-504.
- Dick, J. G., 1973, *Analytical Chemistry*: McGraw-Hill Book Company, New York, 696 pp.
- Drever, J. I., 1988, *The geochemistry of natural waters*: Prentice Hall, Englewood Cliffs, NJ, 2nd edition, 435 pp.
- Fritz, S. J., 1994, A survey of charge-balance errors on published analyses of potable ground and surface waters: *Ground Water*, v. 32, pp.539-546.
- Harper, L., 1993, Middle Rio Grande water assessment, Surface Water Quality Report: Unpublished report prepared for USBR Albuquerque Project Office, 108 pp.
- Hawley, J. W., and Haase, C. S., compilers, 1992, *Hydrogeologic framework of the northern Albuquerque Basin*: New Mexico Bureau of Mines and Mineral Resources, Open-file Report 387, 165 pp.
- Knorr, D. B., and Cliett, T., 1985, Proposed groundwater recharge at El Paso Texas: *in* Asano, T. (ed.), *Artificial recharge of groundwater*: Butterworth Publishers, Boston, pp. 425-479.
- Liang, L., McCarthy, J. F., Jolley, L.W., McNabb, J. A., and Mehlhorn, T. L., 1993, Iron dynamics: transformation of Fe(II)/Fe(III) during injection of natural organic matter in a sandy aquifer, *Geochemica et Cosmochemica Acta*, v. 57, pp. 1987-1999.
- Mucci, A., 1986, Growth kinetics and composition of magnesian calcite overgrowths precipitated from seawater; quantitative influence of orthophosphate ions: *Geochemica et Cosmochemica Acta*, v. 50, pp. 217-233.
- Pankow, J. F., 1991, *Aquatic Chemistry Concepts*: Lewis Publishers, Chelsea MI, 683 pp.
- Parkhurst, D. L., Thorstenson, D. C., and Plummer, L. N., 1980, PHREEQE—a computer program for geochemical calculations: U.S. Geological Survey, Water-resources Investigations, Report 80-96, 195 pp.
- Reddy, M. M., 1977, Crystallization of calcium carbonate in the presence of trace concentrations of phosphorous containing anions: *Journal of Crystal Growth*, v. 41, pp. 287-295.
- Reiter, M., Eggleston, R. E., Broadwell, B. R., and Minier, J., 1986, Estimates of terrestrial heat flow from deep petroleum tests along the Rio Grande rift in central and southern New Mexico: *Journal of Geophysical Research*, v. B91, pp. 6225-6245.
- Tomson, M. B., 1983, Effect of precipitation inhibitors on calcium carbonate scale formation: *Journal of Crystal Growth*, v. 62, pp. 106-112.
- Walter, L. M., and Hanor, J. S., 1979, Effect of orthophosphate on the dissolution kinetics of biogenic magnesium calcites: *Geochemica et Cosmochemica Acta*, v. 43, pp. 1377-1385.
- Weldon, K. L., 1986, *Statistics: a conceptual approach*: Prentice-Hall, Englewood Cliffs, New Jersey, 442 pp.
- Whitworth, T. M., in press, Hydrogeochemical computer modeling of proposed artificial recharge of the upper Santa Fe Group Aquifer; *in* Hawley, J. W., Haneberg, W. C., and Whitworth, T. M. (compilers), *Hydrologic investigations in the Albuquerque Basin, central New Mexico, 1992-1995*: New Mexico Bureau of Mines and Mineral Resources, Open-file Report 402, pp. 4-1-4-87. □