Groundwater Vulnerability Assessment: Hydrogeologic Perspective and Example from Salinas Valley, California

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Vulnerability of groundwater to contamination has typically been addressed by analysis or inference of near-surface hydrologic processes. Yet, in many basins like the Salinas Valley, California, shallow groundwater quality has already been degraded over large areas by nitrates, pesticides, salinity from irrigation, or other contaminants. The ultimate impact of this contamination on deeper groundwater quality during the decades, centuries or millennia to come is a highly relevant issue. We demonstrate an approach to groundwater vulnerability assessment that emphasizes important geologic features in a stochastic-geostatistical framework and incorporates information on both shallow and deep groundwater flow and contaminant transport in the context of a circulating groundwater system. The approach complements more common, shallow investigative approaches, which emphasize source inventory, soil characteristics, and vadose-zone flow and transport. Results from an assessment of groundwater vulnerability to nitrate contamination in the Salinas Valley agree with observed regional patterns in groundwater nitrate concentrations.

INTRODUCTION

Vulnerability of groundwater to contamination has typically been addressed by analysis or inference of near-surface hydrologic processes [*National Research Council (NRC),* 1993]. This is logical, as most of the contaminant sources originate at the land surface and the contaminants tend to migrate downward to the water table. Approaches to assess groundwater vulnerability have therefore focused on mapping of source locations and strengths where possible, and then somehow predicting the transport and fate of contaminants as they move through the soil zone as well as the deeper vadose zone. Difficulties with these approaches arise due to lack of adequate information on contaminant sources, lack of information on the subsurface heterogeneity of transport properties that control pathways and rates of fluid movement, and the problem of reliably modeling flow and transport in the vadose zone.

Although this shallow investigative approach is obviously valid and necessary, we assert that it furnishes only part of the information needed for vulnerability assessment – that part pertaining to impacts on water quality at the water table. Importantly, most groundwater supply wells tap aquifer zones that are 10's to 100's of meters below the water table, and groundwater travel times to such depths are commonly in the range of 10's to 1000's of years. The potential for attenuation of contaminant concentrations by both physical and chemical mechanisms en route from the water table to well intakes is therefore substantial. Clearly, assessments of vulnerability of groundwater resources to contamination require analysis of not only the vadose zone but also of the groundwater system itself. The common belief that "once the contaminant reaches the water table, the damage is already done", ignores the fact that significant changes in concentrations can occur within the deeper

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groundwater. Furthermore, in most non-point source agricultural as well as urban land-use settings, shallow groundwater quality is already being degraded over large areas by nitrates, pesticides, salinity from irrigation, and other contaminants [*Subcommittee on Ground Water*, 1989]; hence the ultimate impact on deeper groundwater quality during the decades, centuries or millennia needed for transport to well intakes is a highly relevant issue.

In this paper we outline an example of a deep groundwater vulnerability assessment in a part of the Salinas Valley, California. Using data and insights from a previous groundwater modeling study of the basin [*Durbin et al.*, 1978], geostatistical modeling of hydrostratigraphy, and a backward-in-time random walk particle method for modeling transport, we demonstrate an approach for assessing susceptibility of the groundwater to nitrate (NO_3) contamination assuming the sources are more or less pervasively distributed across the landscape. The results illustrate the importance of the geologic heterogeneity, pumping patterns, and regional groundwater migration on the timing of groundwater contamination at depths of approximately 50 and 120 m below the water table. The results also highlight the need to conceptualize and characterize the subsurface not as "soils" but as geologic media whose spatial patterns can be estimated and modeled with geologic and geostatistical techniques.

BACKGROUND ON VULNERABILITY ANALYSIS

Numerous methods have been developed for groundwater vulnerability assessments. Typically these methods fall into three categories [*NRC*, 1993]:

- Overlay and index methods that involve assigning numeric ranks and weights to various physical attributes deemed important in contaminant transport to develop a relative vulnerability score. This score is then mapped to show relative groundwater vulnerability in a region.
- Process-based simulation models that attempt to predict contaminant transport in time and space using analytical or numerical solutions to mathematical equations that represent the processes governing contaminant movement.
- Statistical methods that attempt to assess and quantify the association between measures of vulnerability and various types of information thought to be related to groundwater vulnerability.

All of these methods have benefited from the growing availability and use of geographic information systems (GIS) for managing the vast amounts of data and for coupling the data to process-based simulation models [*Corwin and Loague*, 1996]. *Corwin et al.* [1997] provide an excellent review of non-point source pollutant analyses accomplished with GIS in the vadose zone. They wisely cautioned that, despite the promise of GIS tools, the uncertainties inherent to such vadose-zone transport analyses remain formidable, particularly on the regional scale.

Attributes typically used in overlay and index methods include soil characteristics (e.g., texture, structure, thickness, and organic matter content), depth to water table, travel time to water table, general lithology and hydraulic conductivity of the saturated and unsaturated zones, topography, and groundwater recharge rates [*Vrba and Zaporozec*, 1994; *Aller, et al.*, 1987; *NRC*, 1993]. These methods tend to have greater focus on the distribution of these various attributes and less emphasis placed on processes directly controlling groundwater contamination. This, combined with uncertainties in the data themselves and in the actual relevance of each ranked factor to transport processes in the vadose zone or groundwater [*NRC*, 1993; *Loague et al.*, 1998a], make the resulting vulnerability maps highly suspect.

Other process-based simulation models have tended to be used to evaluate contaminant transport in hypothetical settings or in well-evaluated local incidences. A recent exception is the regional analysis of 1,2-dibromo-3-chloropropane (DBCP) transport in Fresno County, California by *Loague et al.* [1998b; 1998c] and *Loague and Corwin* [1998]. This unprecedented study used process-based models for both estimating rates of transport and transformation in the vadose zone (PRZM-2) and for simulating transport in groundwater. They were able to elucidate timing and mechanisms by which the contaminants migrated through the subsurface and made important inferences about DBCP application rates. Nevertheless, they too cautioned about the significant uncertainty in the model results, and their transport models could be neither calibrated nor validated because of the lack of concentration data.

Few statistical approaches to groundwater vulnerability have been developed. Included in this category, however, are geostatistical evaluations which describe the spatial distribution of process parameters that affect vulnerability [*Fogg et al.*, 1995]. Other statistical approaches, such as discriminant analysis and cluster analysis, have been used to describe relationships between soil attributes and groundwater vulnerability [*Teso, et al.*, 1988; *Troiano et al.*, 1994]. The case study discussed herein includes use of geostatistics in construction of stochastic flow and transport models of the saturated zone. According to *Corwin et al.* [1997], geostatistical-stochastic analyses of regional groundwater vulnerability are rare.

The recurring message in the literature is that the impacts of non-point pollutant sources on groundwater are

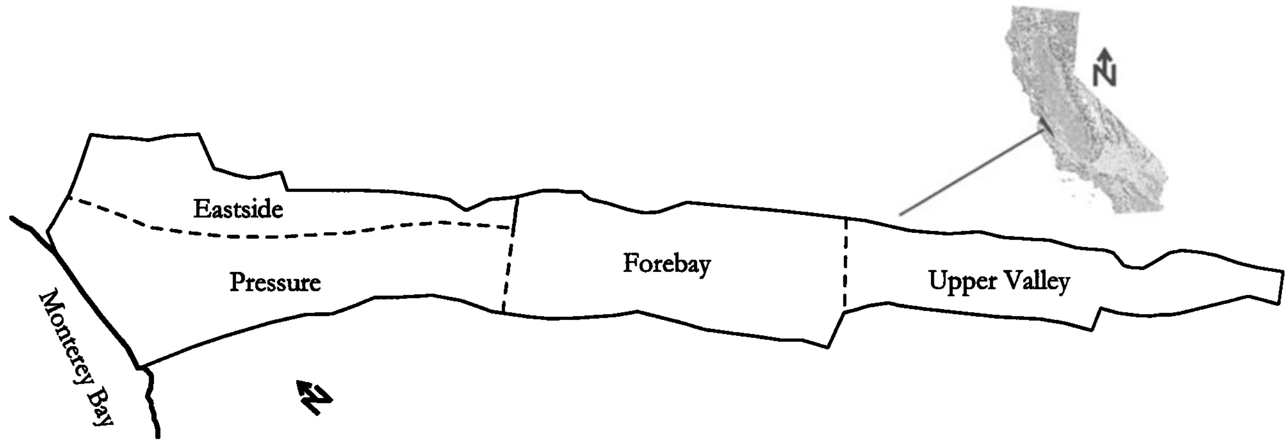


Figure 1. California Department of Water Resource's designated areas of the Salinas Valley [after *Yates*, 1988].

extremely difficult to characterize, much less predict. Furthermore, although the adoption of powerful GIS and spatial analysis techniques is leading to more realistic representations of landscape complexity, the uncertainties inherent to parameterizing and modeling the vadose zone as well as source strengths and histories have thus far precluded the routine development of process-oriented models that are sufficiently reliable to guide land-management efforts. The Salinas Valley case study provides an entirely different perspective on the problem of assessing aquifer vulnerability.

SALINAS VALLEY CASE STUDY

The Salinas Valley is located in the central coast of California (Figure 1). The valley drains northwest and is bounded to the north by Monterey Bay, the west by the Sierra de Salinas and Santa Lucia Range and the east by the Gabilan and Diablo Ranges. Figure 1 also shows the California Department of Water Resource's designated areas of the valley: the East-Side, Pressure, Forebay and Upper-Valley areas. This designation provides a convenient frame of reference for study of the valley. Results presented herein focus primarily on the Pressure and East Side areas.

Unconsolidated Salinas-River, alluvial-fan, marine clay, and wind-blown sand deposits of the Pressure and East-Side areas range from Pleistocene to recent age [*Durham*, 1974]. The Salinas River flows in a direction oriented with the long axis of the valley. Salinas River deposits are generally found toward the center of the valley. Fluvial sediments of the East Side originate from alluvial fans associated with tributaries draining uplands along the valley rim. Alluvial-fan deposits extend to the basement contact and cover much of the valley floor [*Thorup*, 1976]. A series of marine transgressions have deposited marine-clay layers that are laterally extensive in the Pressure area. These clays create the confined or semiconfined conditions in the "180 ft" (55 m) and "400 ft" (122 m) aquifers, in which most of the wells are completed.

Since about the early 1970's, nitrate concentrations in groundwater sampled from production wells of the East Side and Pressure areas of the Salinas Valley have been rising [*Snow et al.*, 1988]. Sources include irrigated crops, feedlots, greenhouses, and septic tanks [*Rolston et al.*, 1996; *Fogg et al.*, 1998], collectively forming non-point sources of nitrate over most areas in the basin. Large-scale agricultural activity began in the Salinas Valley in the early 1900's and grew at modest rates up until the 1940's, when use of irrigation water and fertilizer accelerated [*H. Esmaili and Associates*, 1978]. Nitrate concentrations in wells rose modestly from the 1950's through the 1960's and then generally increased dramatically beginning in the 1970's and 1980's.

Approach and Methods

The initial objective of this study was to elucidate the time scales over which nitrate transport is occurring so that cause-and-effect relationships between land use and groundwater quality could be better understood. That is, are the nitrates measured today from land use practices of, for example, 1945 or 1985; and if nitrate-loading rates are reduced today, how long before the beneficial effects are seen in water supply wells? As we will show, the study also produced the unanticipated benefit of revealing spatial patterns of groundwater vulnerability that match the nitrate concentration distribution reasonably well on the regional scale.

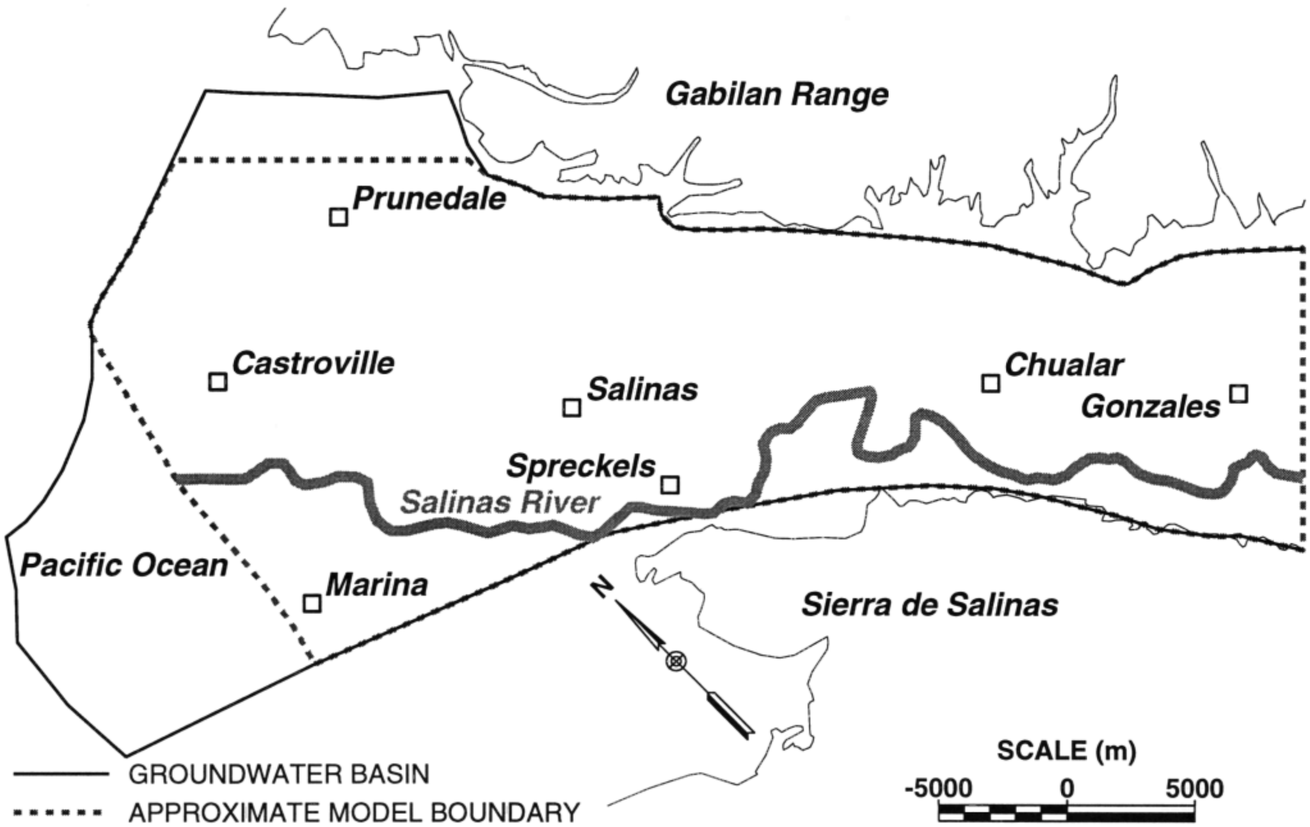


Figure 2. Model boundary and groundwater basin [after *Durbin et al.*, 1978].

The work involved both modeling of vadose-zone and regional groundwater flow and transport processes on horizontal scales of 0.7 and 40 km, respectively. In the vadose zone analysis, a detailed characterization of vadose zone conditions was performed and transport was modeled at one site. In the groundwater analysis, we constructed a regional flow model of the northern one-third of the basin ("East Side" and "Pressure" areas; Fig. 2) and used a random-walk particle technique to model nitrate transport backward in time through the saturated zone. The main goal of these analyses was to estimate, at a minimum, the approximate travel times to within plus or minus one to two decades.

Detailed analyses of water flow rates through the vadose zone were conducted by *Burow* [1993] using data on retention characteristics and permeability of the sediments obtained from more than 60 core samples [*Maserjian*, 1993], seismic characterization of the shallow stratigraphy, and two-dimensional stochastic modeling of flow and transport in the 24 to 37 m thick vadose sections. The stochastic modeling involved geostatistical indicator simulation of spatial patterns in sediment textures [*Deutsch and Journel*, 1992] which were then used as templates for assigning hydraulic properties to the models. Results indicated that the travel times through the vadose zone are on the order of 10 to 20 years. These times were consistent with measurements of bromide concentrations in the vadose zone at each site [*Burow*, 1993]. The remainder of this paper focuses entirely on the groundwater investigation.

For the groundwater transport analysis the regional hydrostratigraphy was modeled stochastically with geostatistical indicator simulation using the algorithm of *Deutsch and Journel* [1992]. To each texture type (i.e., sand, muddy sand, mud, where "mud" refers to silt and clay undifferentiated), hydraulic conductivity values were assigned based on our core measurements [*Maserjian*, 1993] and previous work on regional hydraulic properties by *Durbin et al.* [1978].

Using data on pumpage, recharge, and other boundary conditions from the previous modeling work of *Durbin et al.* [1978], a groundwater flow model was created with the computer program MODFLOW [*McDonald and*

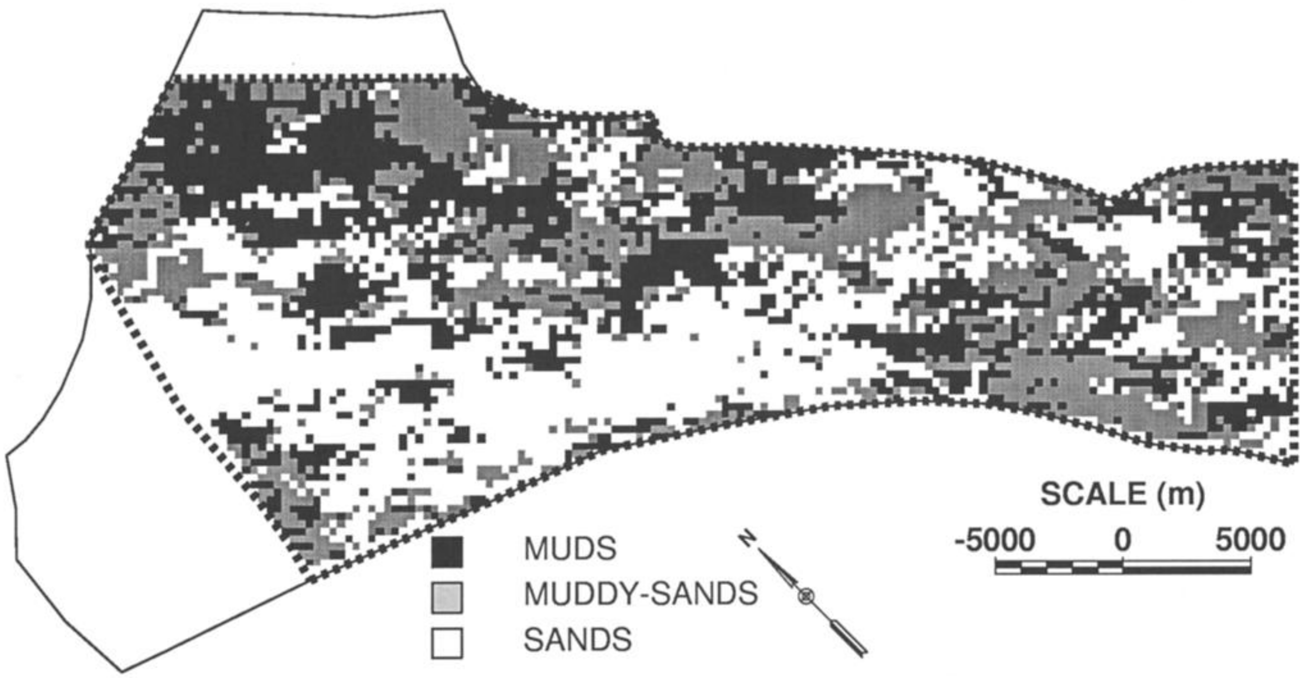


Figure 3. Horizontal cross section through a geostatistical realization of simulated hydrostratigraphy.

Harbaugh, 1988]. After minor calibration of the model to reasonably represent regional conditions, groundwater velocity fields were produced for use in the transport modeling by a random-walk particle method, which was chosen because of its computational efficiency and numerical accuracy (e.g., lack of numerical dispersion). Computational efficiency is essential for this type of regional, heterogeneous simulation because of the large number of nodes. The following sections describe details of the regional groundwater flow and transport analysis.

Geostatistical simulation of hydrostratigraphy. An example of one of 10 geostatistical realizations of the hydrostratigraphy is shown in Figures 3 and 4 with x, y, and z direction node spacings of 304.8 , 304.8 and 3.048 m, respectively, and 758,000 nodes. In this stochastic approach, each of the 10 realizations honors the data from relatively high-quality driller's log data obtained from 135 wells (Figs. 5 and 6) by a previous geologic investigation [*Salinas Valley Geologic Investigation (SVGI)*, 1960]. Spatial variability of the 3 hydrostratigraphic units, sand (includes gravel), mud (silt and clay undifferentiated), and muddy sand, varies stochastically among the realizations as a function of a variogram model that was developed from the well log data and knowledge of the regional depositional processes. The variogram model had "ranges" in the horizontal plane of 8000 m and 2000 m parallel and perpendicular to the valley axis, respectively. These fairly long correlation lengths are consistent with the well-defined laterally extensive 180-ft and 400-ft aquifers, and their confining marine clays. The variogram model had a vertical range of approximately 25 m, which is consistent with observations of thicknesses between 3 and 30 m for these geologic units.

The simulations capture several prominent features of the valley, specifically, the "180-ft" (55-m) and "400-ft" (122-m) aquifers and their confining layers (Fig. 4). The results indicate a somewhat continuous confining layer between the 180-ft and 400-ft aquifers with increasing connectivity of sand and gravel from the coast southward. As will be shown, this has a profound effect on the transport simulations. Aquifers are well defined in the pressure area and poorly defined in the East Side. The simulations, however, do not capture the complex spatial variability of the orientation of alluvial channels contained in the alluvial fans emanating from the mountain fronts. These channels trend normal to the sediment bodies deposited by the ancestral Salinas River, for which orientation of the channels is assumed to be parallel to the long axis of the Valley. *Carle et al.* [1998] illustrate this intricate interfingering of alluvial-fan and Salinas-River environments in a detailed geostatistical simulation of the hydrostratigraphic architecture of one field site in the Salinas Valley.

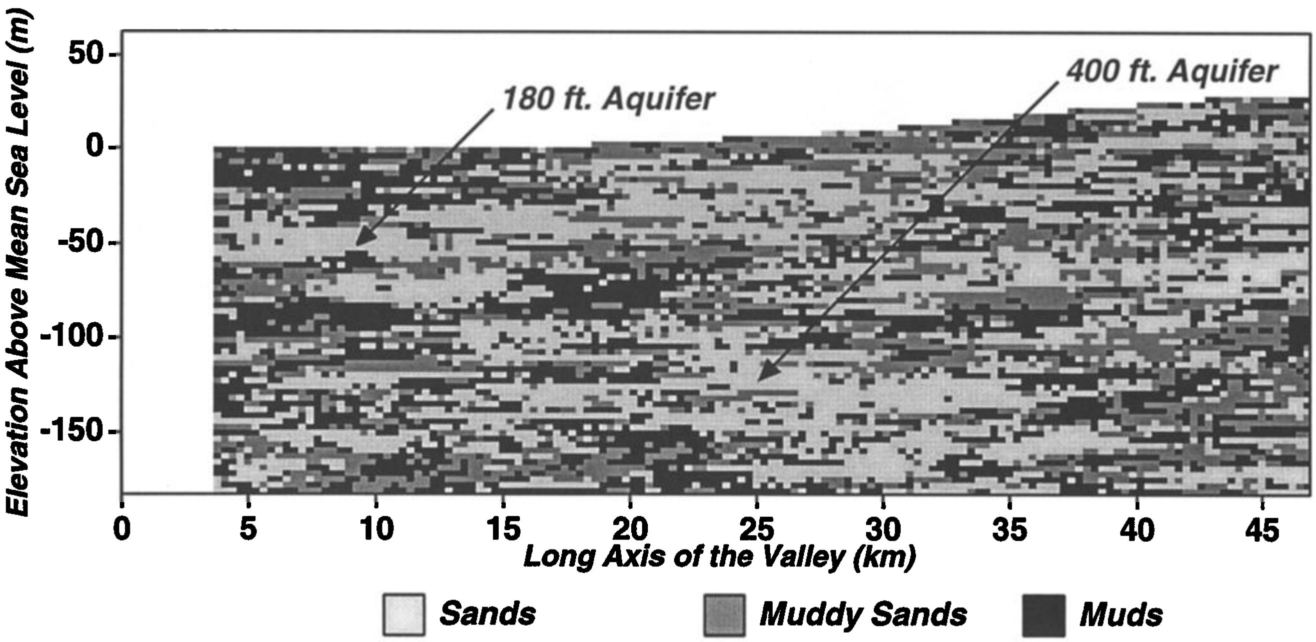


Figure 4. Vertical cross section through a geostatistical realization of simulated hydrostratigraphy.

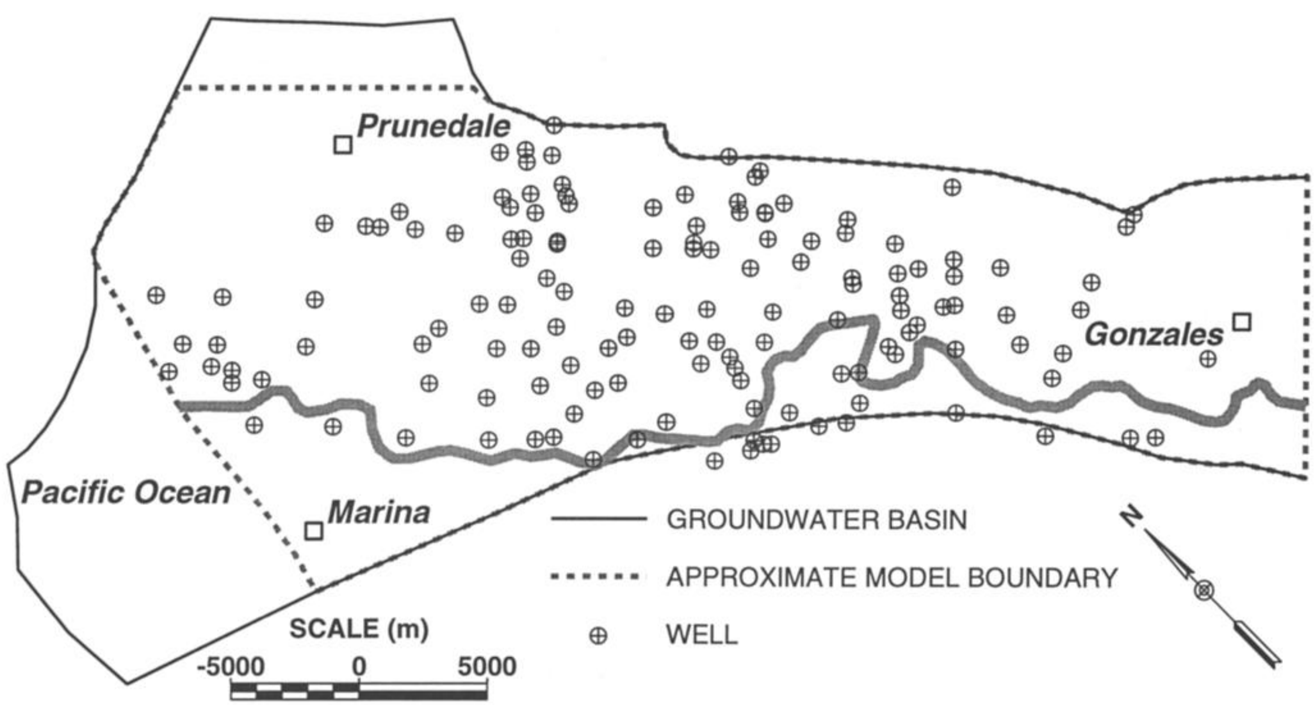


Figure 5. Location map for wells from *SVGI* [1960] used in conditional indicator simulations.

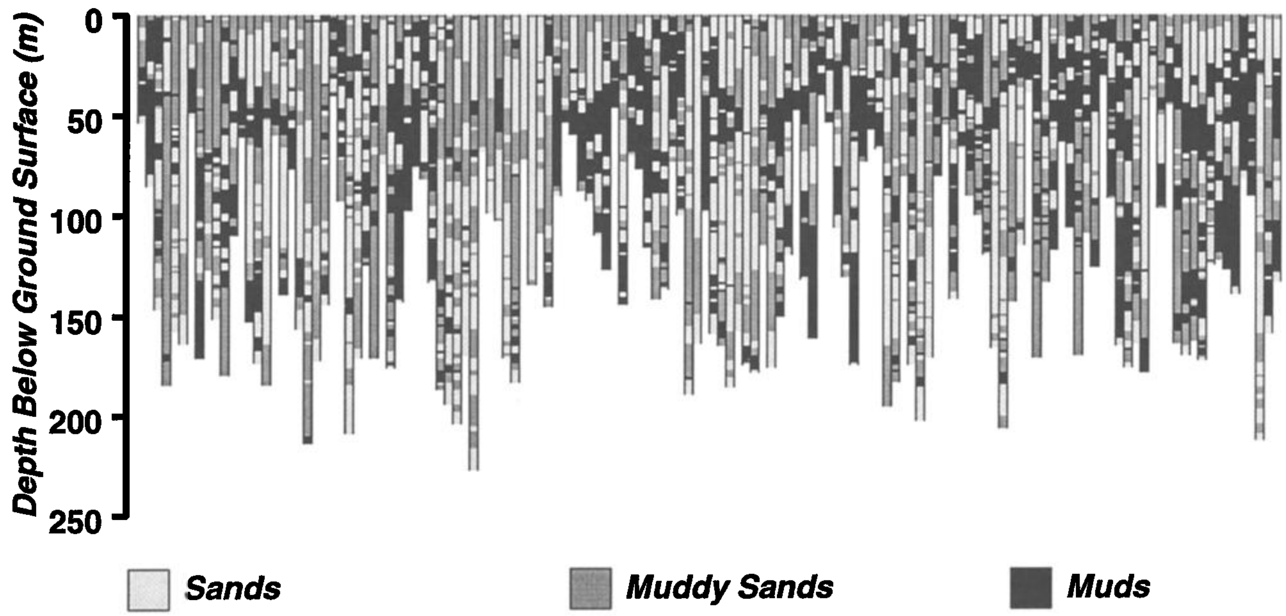


Figure 6. Textural interpretations of driller's logs from *SVGI* [1960] used in conditional indicator simulations.

In fact, there are many ways in which this characterization of hydrostratigraphy could be improved by performing more careful geologic analysis of the data and using newer techniques for modeling geologic spatial variability (e.g., *Carle and Fogg* [1996; 1997], and *Carle et al.* [1998]). Nevertheless, the models of hydrostratigraphy used here (Figs. 3 and 4) proved to be sufficient for purposes of this experiment.

Groundwater flow modeling. The objective of the flow modeling was not to mimic system behavior precisely but to estimate a steady-state, spatially varying velocity field that is representative of average conditions between 1940 and the present. The steady-state assumption was made to simplify and expedite the flow and transport modeling procedures and was considered reasonable because average water levels from 1930 to 1980 were fairly stable, with the exception of the East Side Subarea, which experienced substantial declines between the mid-1950's and mid-1960's. In addition to simulating aquifer hydraulics for each of the 10 geostatistically generated realizations of heterogeneity, scenarios with different combinations of rates of withdrawal, recharge and vertical hydraulic conductivity (K) parameter values were investigated. Hydraulic simulations provided velocity fields to the transport component of the model. The horizontal extent of the model boundary is shown in Figure 2.

Rates of recharge and discharge, including agricultural and municipal pumpage, streambed infiltration, subsurface inflow at system boundaries, infiltration of precipitation, and evapotranspiration by riparian vegetation were obtained from the modeling analysis of *Durbin et al.* [1978]. Details regarding the boundary conditions are in *Fogg et al.* [1995]. Simulation results for hydraulic heads agree well with simulations and observations published in *Durbin et al.* [1978].

Scenarios. We assumed that the potentially most significant factors affecting uncertainty of regional migration of contaminants would be the hydrostratigraphy, magnitude of the vertical conductivity, and rates of withdrawal and recharge. Consequently, hydraulic simulations considered 9 scenarios that represent all possible combinations of 3 ratios of vertical to horizontal conductivity and 3 overall rates of withdrawal and recharge (Table 1). Simulations for each scenario were performed for all 10 realizations of aquifer heterogeneity to yield a total of 90 hydraulic simulations.

Hydraulic Conductivity. Based on calibration results of *Durbin et al.* [1978], data in *Boyle Engineering Corp.* [1986], and measurements of K on core samples by *Maserjian* [1993], K values of the three hydrostratigraphic facies represented in the geostatistical simulations were assigned as follows: *sand*, 30 m/d; *muddy sand*, 3 m/d; and *mud*, 0.0003 m/d.

Clearly, K of the hydrostratigraphic units in local areas of our model may deviate considerably from the average values discussed above. Nevertheless, based on the field and laboratory measurements available, we believe our K estimates are good 'order-of-magnitude' estimates. When considering the question of deep migration of contaminants, we are concerned primarily with uncertainties in the net vertical K of the system. Therefore, in 6 of the 9 scenarios, the ratio of vertical to horizontal K, K_v/K_h, is lowered by factors of 2 and 10. The effect of the low permeability *mud* layers on the gross values of K_v is accounted for, at least to vertical resolution of 3 m, with the geostatistical realizations of the hydrostratigraphy (Figs. 3 and 4).

Transport Modeling. Transport of dissolved contaminant is described by an advection-dispersion equation (ADE)

$$\frac{\partial c}{\partial t} = -\frac{\partial v_i c}{\partial x_i} + \frac{\partial}{\partial x_i}\left(D_{ij}\frac{\partial c}{\partial x_j}\right) \tag{1}$$

where c is concentration [ML^{-3}], $\mathbf{v}$ is pore-water velocity [LT^{-1}] and $\mathbf{D}$ is a dispersion tensor [L^2T^{-1}]. Here we have assumed that spatial variations in effective porosity Θ in saturated granular porous media are generally small as compared to other parameters such that Θ may be approximated as a constant. To efficiently compute travel times, the model transport component relies on a random-walk particle method to simulate the backward-time advection-dispersion equation (BTADE) [*Uffink*, 1989; *Wilson and Liu*, 1995]

$$-\frac{\partial p(\mathbf{x},t\mid \mathbf{z},s)}{\partial t} = v_i\frac{\partial p(\mathbf{x},t\mid \mathbf{z},s)}{\partial x_i} + \frac{\partial}{\partial x_i}\left(D_{ij}\frac{\partial p(\mathbf{x},t\mid \mathbf{z},s)}{\partial x_j}\right) \tag{2}$$

where $\int_\Omega p(\mathbf{x},t|\mathbf{z},s)d\mathbf{x}$ is the probability of finding a particle in the region Ω at time t given that it was observed at location $\mathbf{z}$ at time s, $t \geq s$. In Eq. (2), p evolves in z and s and is interpreted as the probability that a particle observed at location $\mathbf{x}$ at time t originated from location $\mathbf{z}$ at time s. The approach is more general than particle-based advective backward-tracking techniques (e.g., *Newsom and Wilson* [1988] and *Bair et al.* [1990]), commonly used in the analysis of water-supply well protection zones and the design of pump-and-treat-remediation technologies, in that it includes the potentially important effects of dispersion on probability of contaminant capture and arrival.

An advantage of solving the BTADE as opposed to the ADE is that application of the latter to non-point source contamination with the random-walk algorithm would require vast numbers of particles. Furthermore, the method allows one to efficiently compute the age distribution of waters pumped by individual wells in the model.

With the BTADE experiments, we released particles from *sand* units at each of the conditioning wells (Fig. 5 and 6) at depths of approximately 55 and 122 m, corresponding to the 180-ft and 400-ft aquifers. This was repeated for each of the 10 realizations of velocity and, because the geostatistical simulations honor the well data (Fig. 5 and 6) exactly at the well locations, the particles were released from precisely the same locations and material types in each realization. The fraction of particles reaching the water table as a function of time yielded a time history of the arrival of (potentially contaminated) young water to discharge wells. The random walk code used to simulate the BTADE is a modification of the SLIM1 code developed by *Tompson et al.* [1987] and *LaBolle et al.* [1996]. Details regarding the BTADE and its numerical solution by random walks are given in *Uffink* [1989].

Parameter values and boundary conditions for transport simulations. Parameter values for the BTADE include velocities calculated from the hydraulic model, molecular diffusivity, effective porosity, and longitudinal and lateral dispersivities. The molecular diffusivity is given by the coefficient of molecular diffusion in unconfined water, approximately 1.0×10^{-5} cm^2/s, divided by the tortuosity of the medium, which we assume equals 4, to yield $D^m = 2.5\times10^{-6}$ cm^2/s. Spatial variations in effective porosity, Θ, in saturated granular porous media are generally small as compared to other parameters. We assume the effective porosity is constant and equal to 0.33. Based on effective dispersivity calculations in local-scale, higher-resolution simulation experiments of plume migration [*Fogg et al.*, 1995], an asymptotic longitudinal dispersivity of approximately 38 m was assumed. The asymptotic transverse dispersivity is assumed to be 0.38 m. Molecular diffusion is potentially important in these experiments because of the large volumes of non-aquifer materials (*muddy sands* and *muds*) within which transport is not dominated by advection and advective dispersion.

An instantaneous point source released in a particular well was simulated by maintaining, over a short time period of time, a constant number density of 10,000 particles distributed uniformly in a high-conductivity finite-difference grid-block, after which the source was eliminated.

Table 1. Specified discharge, recharge, and ratio of vertical to horizontal K, K_v/K_h for 9 hydraulic simulation scenarios.

		Discharge (ac-ft/yr)		Recharge (ac-ft/yr)		
Scenario	K_v/K_h	Agricultural	Municipal	Irrigation	Precipitation	Small Stream
1	1.0	160,000	15,000	72,000	6,000	7,000
2	1.0	116,250	15,000	52,313	6,000	7,000
3	1.0	72,500	15,000	32,625	6,000	7,000
4	0.5	160,000	15,000	72,000	6,000	7,000
5	0.5	116,250	15,000	52,313	6,000	7,000
6	0.5	72,500	15,000	32,625	6,000	7,000
7	0.1	160,000	15,000	72,000	6,000	7,000
8	0.1	116,250	15,000	52,313	6,000	7,000
9	0.1	72,500	15,000	32,625	6,000	7,000

During this time period, particles advect and disperse as a pulse into the active model domain. This was done for each well in Figure 5 at or near the appropriate depth (either 55 m or 122 m). A constant time step of 10 days was used in all transport simulations.

Arrival-Time Results

Transport simulation results were analyzed in terms of the percentage of particles released from each well that reach the water table by a given time. For example, Figures 7a and 7b plot pie charts at each well showing the percentage of particles originating from the 55 m zone breaking through the water table within 30 and 50 years of simulation (backward in time), excluding those wells for which no particles reached the water table. These results can also be viewed as the percent of water pumped by each well that is younger than the elapsed time of the backward simulation. That is, a 10 percent breakthrough at a well in the 30 year simulation of Figure 7a indicates that 10 percent of the water pumped by that well was recharged at the water table within the last 30 years.

Figures 7a and 7b are based on averages of the transport simulation results among all 10 stochastic realizations such that the particle densities represent probability that water pumped from a given well originated from various locations at the water table. Spatial patterns of young water differed little from realization to realization, possibly because of the significant amount of conditioning data (Figs. 5 and 6). This indicates that the transport model is not highly sensitive to the uncertainties in heterogeneity that are captured by the 10 realizations on the regional scale.

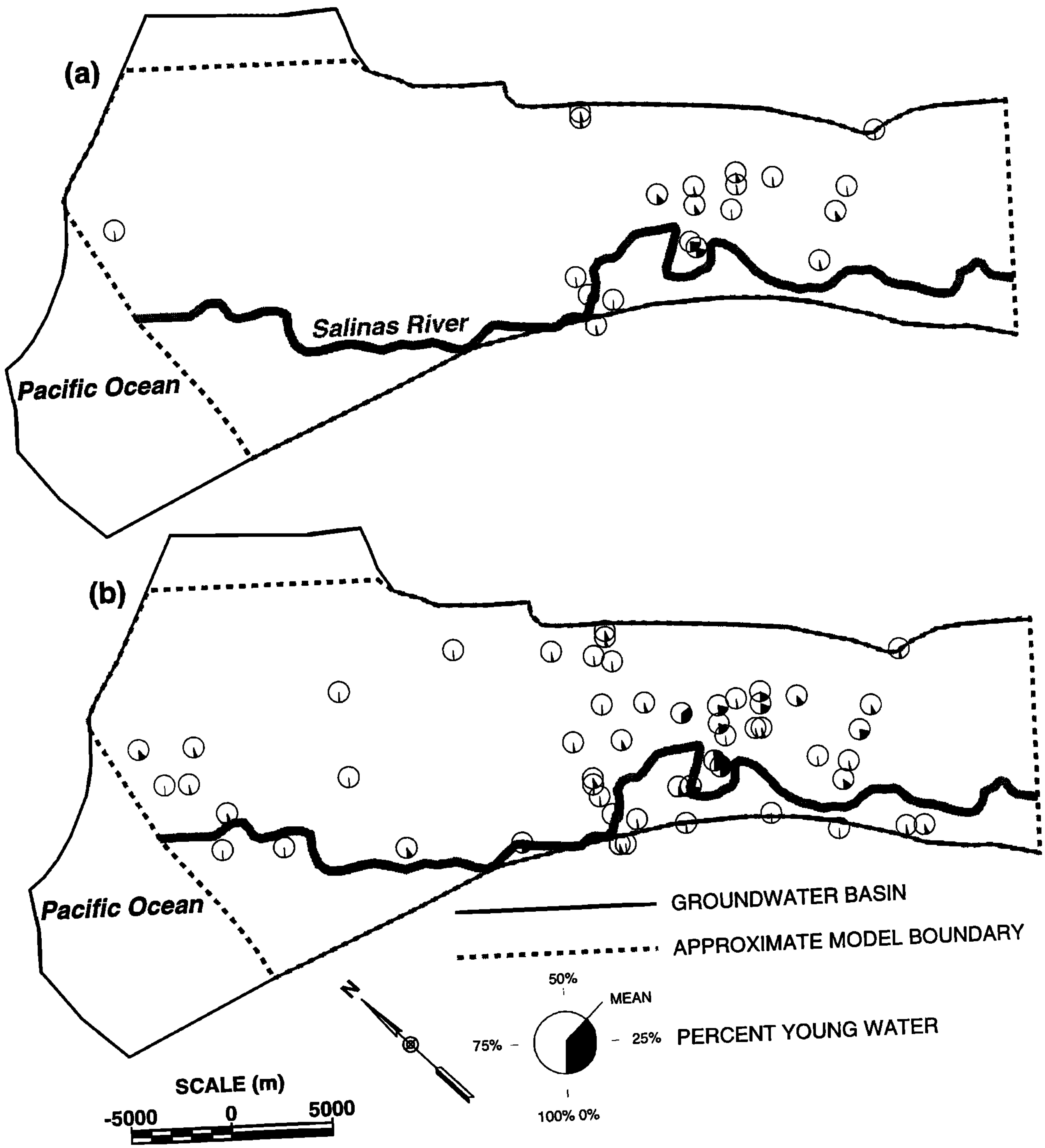
Local capture zone analysis shows that particles that reach the water table within 50 years and originate from the 55-m zone tend to breakthrough at locations within approximately a 3-km radius of the respective well. Thus, each percent young water pie chart in Figures 7a and 7b is representative of travel times in the local vicinity of that chart.

To allow for comparisons with nitrate concentration maps, Figures 7a and 7b are corrected for low-nitrate concentration water arriving from the Salinas River. In the flow model, steady-state recharge from the Salinas River is simulated by a series of constant head nodes. We defined a region of grid blocks within approximately 0.5 km of specified-head Salinas River nodes and removed particles entering this region from the simulations because the substantive amounts of water infiltrating through the Salinas River streambed are not sources of groundwater contamination. *H. Esmaili & Associates, Inc.* [1978] state: "Wells adjacent to the Salinas River on either bank contain very low nitrate concentrations and have shown little or no increases in the past."

As expected, the 50-yr backward simulation (Fig. 7b) shows a greater percent "young water" and a greater number of wells for which at least one particle reached the water table, in comparison to the 30-yr simulation (Fig. 7a). Likewise, a 10-yr simulation (not shown) produced smaller values of percent "young water" and only a few wells for which at least one particle reached the water table.

Figure 8 shows a complete arrival time distribution for 500 years of simulation at a well for scenarios 1, 2 and 3 (Table 1). These results indicate that water pumped from these wells is a mixture of ages ranging from decades to centuries or more. Furthermore, these results suggest that if contaminant-loading rates do not decline substantially, historical breakthroughs of contaminants at the wells merely represent the beginning of a gradual deterioration in groundwater quality.

Review of the breakthrough plots for individual wells (e.g., Fig. 8) showed that the relationship between pumping rates and the fraction of young water is not necessarily simple. Similar to Figure 8, many of the plots showed higher fraction of young water for intermediate pumpage than for low and high rates of pumpage. Reasons are twofold: (1) when pumping is increased, the capture zone



Figures 7a and 7b. Transport simulation results (scenario 1) showing average percent water younger than (a) 30 years and (b) 50 years in withdrawals from approximately 55-m (180-ft) depth, corrected for induced infiltration from Salinas River.

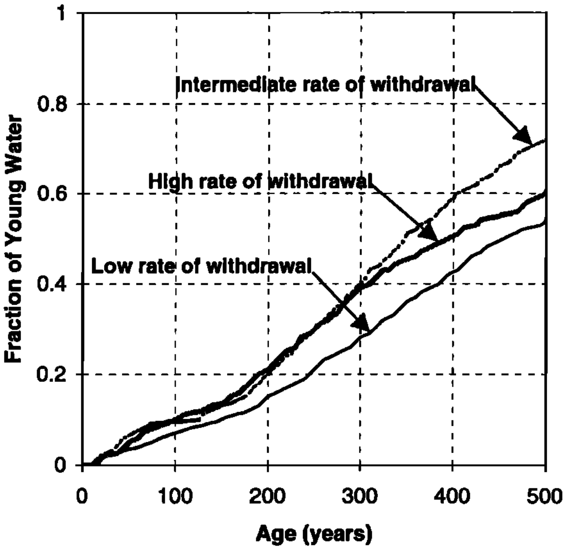


Figure 8. Cumulative arrival of mass to the water table (percent-young water pumped) versus time from one well for scenarios 1, 2, and 3 (Table 1).

expands to draw in more "old" water, thereby decreasing the *fraction* of young water as compared to scenarios having less pumpage, and (2) when pumping is increased everywhere, as in these scenarios, dynamics of the well interference can change in a complex fashion resulting in either an increase or decrease in the fraction of young water at a location in the model. Nevertheless, the higher rates of pumping do induce flow of larger *quantities* of young water to the wells.

Implications for Contaminant Transport History

Large-scale agricultural activity began in the Salinas Valley in the early 1900's and grew at modest rates up until the 1940's, when use of irrigation water and fertilizer accelerated [*H. Esmaili and Associates*, 1978]. Subsequently, nitrate concentrations in wells rose modestly from the 1950's through the 1960's and then generally increased dramatically beginning in the 1970's and 1980's. Thus, if the model indicates several decades of little to no contaminant arrival at wells and then sudden appearance of contaminant in significant quantities, this would appear to be consistent with general observations. Accordingly, the 10-year simulation showed little to no contaminant arrival, the 30-yr simulation (Fig. 7a) shows significant increases in contaminant arrival, possibly corresponding with the onset of regional nitrate impacts in the 1970's, and the 50-yr simulation (Fig. 7b) shows potential for still greater regional impact.

The 50-yr results in Figure 7b provide an opportunity to test the hypothesis that substantial nitrate loading at the water table began roughly 50 years ago, when development of large-scale irrigated agriculture began accelerating. If the spatial patterns in percent young water in Figure 7b are similar to recently measured spatial patterns in elevated nitrate concentrations, the simulations would suggest that nitrate loading at the water table began roughly 40 or 50 years ago, assuming approximately 10 yr for transport through the vadose zone [*Burow*, 1993]. Indeed, overlaying both the Figure 7b results and a nitrate groundwater concentration map for 1988 in Figure 9 shows good agreement in the regional spatial patterns.

Spatial Patterns and Hydrostratigraphy

In general, regions in Figure 9 where wells contain low nitrate levels, primarily the downstream (left) half of the study area, show little or no young water arriving within 50 years. In addition, several of the simulated hot spots correspond closely in both relative magnitude and location to observed hot spots. More specifically, the measurements show spatially sporadic, local occurrences of elevated nitrates in the downstream half of the area and a regional "plume" of high nitrates north of the Salinas River near the middle of the area (Fig. 9). Accordingly, the 50-yr simulation results show spatially sporadic, local occurrences of young water in the downstream half of the area and a relatively dense cluster of wells indicating young water that is roughly coincident with the regional "plume" near the middle of the area.

The lack of young water indicators near the northeast corner of the model, where observations show high nitrate, is caused by the lack of well data in our well log database in that area. That is, no particles were released in that area.

The lack of observed *and* simulated nitrate impacts in the downstream half of the system is clearly attributable to the presence of more extensive clay lenses and confining beds in that area. The cause in the model cannot be attributed to spatially varying pumping rates, because the pumping rates assigned to the model were fairly uniform. Furthermore, based on groundwater pumpage information in *Durbin et al.* [1978], the observed "plume" of high nitrates in the middle of the area (Fig. 9) is apparently not caused by higher rates of pumpage or more shallow pumpage in that area. The general absence of elevated nitrates immediately downstream (northwest) of the central "plume" coincides with the City of Salinas, suggesting somewhat lower rates of nitrate loading in that area.

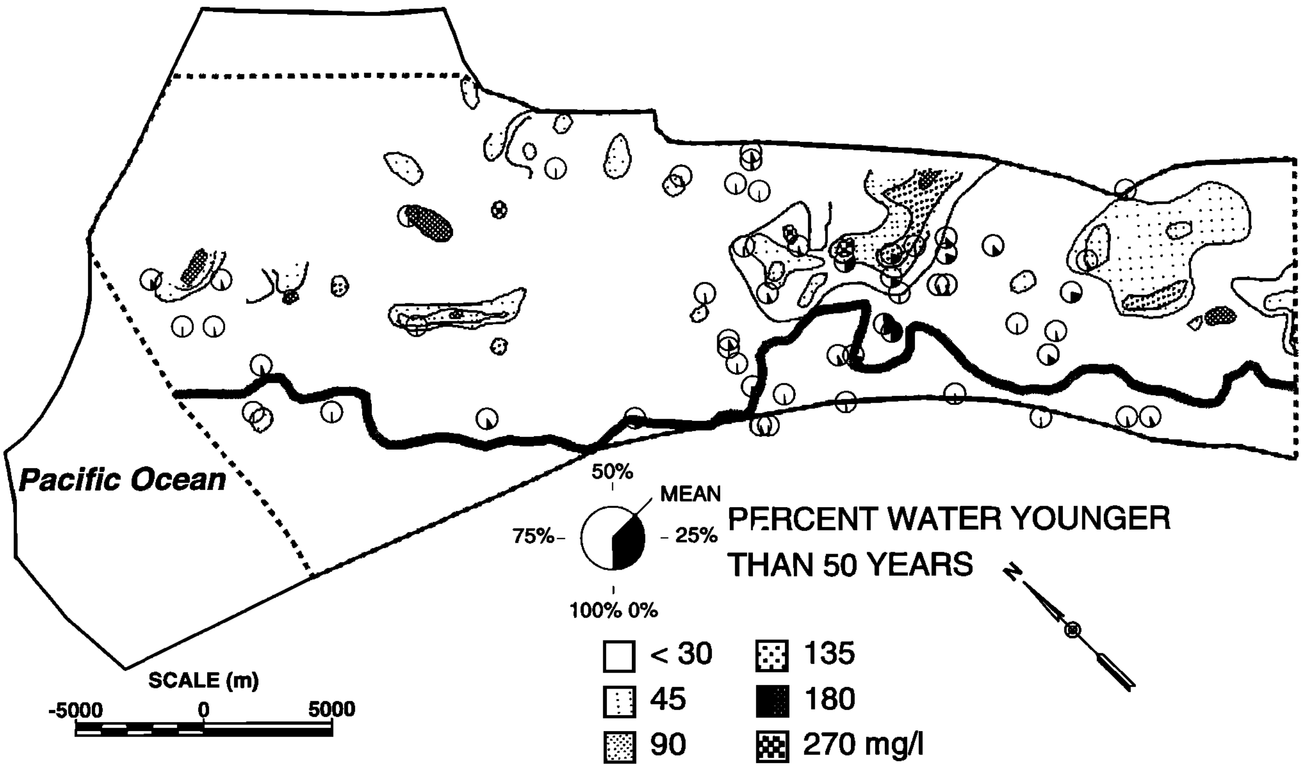


Figure 9. Transport simulation results (scenario 1) comparing average percent water younger than 50 years in withdrawals from approximately 55-m (180-ft) depth, corrected for induced infiltration from Salinas River, and observations of nitrate concentrations for 1988 from *Snow et al.* [1988].

The geostatistical simulations on which the flow simulations are based include the above-mentioned, fine-grained strata, illustrating the importance of hydrostratigraphic characterization in vulnerability analyses. Prior to these simulations, hydrologists hypothesized that the cause of the local hot spots in nitrate concentration in the downstream portion (Fig. 9) was vertical leakage along wells having poorly constructed annular seals and local spills at the surface. In contrast, the model points toward the hydrostratigraphy and the relatively rare occurrence of vertical pathways through this relatively mud-rich section as a plausible explanation.

Given the assumptions and simplifications in the flow and transport models used to simulate the system, we did not anticipate the level of agreement between observed and simulated results that is evident in Figure 9. For example, in the flow model steady-state conditions were assumed, locations (x, y, z) of pumping nodes were rough approximations, the conditional simulation of textural heterogeneity was rather approximate, and local distributions of K and its anisotropy within facies was unknown. Consequently, the degree to which the observations and simulation results agree with respect to regional presence and absence of contamination and spatial frequency is rather remarkable. Because the distribution of pumping in the model is fairly uniform and the nitrate source areas are areally pervasive (with the possible exception of City of Salinas), the most plausible explanation is that the transport is strongly governed by the hydrostratigraphy, and many of the key elements of that hydrostratigraphy are captured by the model.

Sensitivity Analysis

Effects of the different K_v values and pumping rates on the percent young water maps were tested [*Fogg et al.*, 1995] by running the different scenarios listed in Table 1 and comparing to the scenario 1 results (e.g., Figs. 7a and 7b). As expected, the times of first breakthrough were somewhat delayed and the percent young water were somewhat decreased with decreasing K_v and pumping rate. The spatial patterns in the non-zero percent, young-water locations, however, remained very similar among all the scenarios. This is encouraging because it suggests that regional patterns in travel times, or susceptibility [*NRC*, 1993] of the system to contamination, are fairly insensitive to the major model uncertainties. Scenario 1 is the most plausible because it matches the basin hydraulics (heads, recharge and discharge rates) more closely than do the other scenarios.

Simulation of Future Arrival Times

Figures 10a and 10b project percent water that is younger than 100 and 200 years, respectively, for particle releases from a 55 m depth for scenario 1. If one assumes that (1) nitrate loading rates to the water table have been consistently high since the 1940's and (2) these rates will not decline appreciably in the coming decades, the results suggest that concentrations of nitrate or other conservative contaminants will continue to increase into the future (also see Fig. 8). Similar experiments for releases from a 120-m depth below the water table, corresponding approximately to the 400-ft aquifer, indicate that the deeper aquifers would not be impacted significantly until well into the next century.

How reliable is the model for such a futuristic simulation? Much more model development and testing would be required before Figures 10a and 10b could be called bona fide predictions. Nevertheless, such simulations can be useful for elucidating time scales of water quality changes at the regional scale.

The simulations indicate a continuing increase in solute concentrations if loading rates remain constant. Thus, it appears that beneficial effects of modifications in land use today would not be felt for several decades, and nitrate concentrations may continue to rise for many years to come, regardless of changes in land use. On the other hand, the results remind us that quality of the groundwater is not sustainable under significant non-point source contamination created by current and past land use. The chances of ultimately destroying the groundwater resource would be reduced substantially by reductions in contaminant loading today.

ROLE OF THE HYDROGEOLOGIC APPROACH

The Salinas Valley case study exemplifies a "hydrogeologic approach" in that it emphasizes important geologic features and incorporates information on both shallow and deep groundwater flow and contaminant transport in the context of a circulating groundwater system. This complements the more common, shallow investigative approach, which emphasizes the source inventory, soil characteristics, and vadose-zone flow and transport. Based on the general agreement between the spatial pattern in nitrate levels and simulated groundwater ages (Fig. 9), one might conclude that the hydrogeologic approach by itself is adequate for mapping aquifer vulnerability. We caution, however, that this analysis essentially provided a susceptibility rather than a vulnerability map [*NRC*, 1993] because it neglected the effects of spatially varying sources and chemical reactions, although effects of spatially varying pumping rates were included to some extent. The good correspondence between the regional patterns in nitrate concentrations and simulated groundwater ages suggests that nitrate sources tend to be areally pervasive across much of the Salinas Valley landscape. We anticipate that contaminant sources would be more spatially variable in other basins. Nevertheless, a valid analysis of vulnerability clearly should rest on a foundation of susceptibility analysis, or subsurface water travel-time characterization.

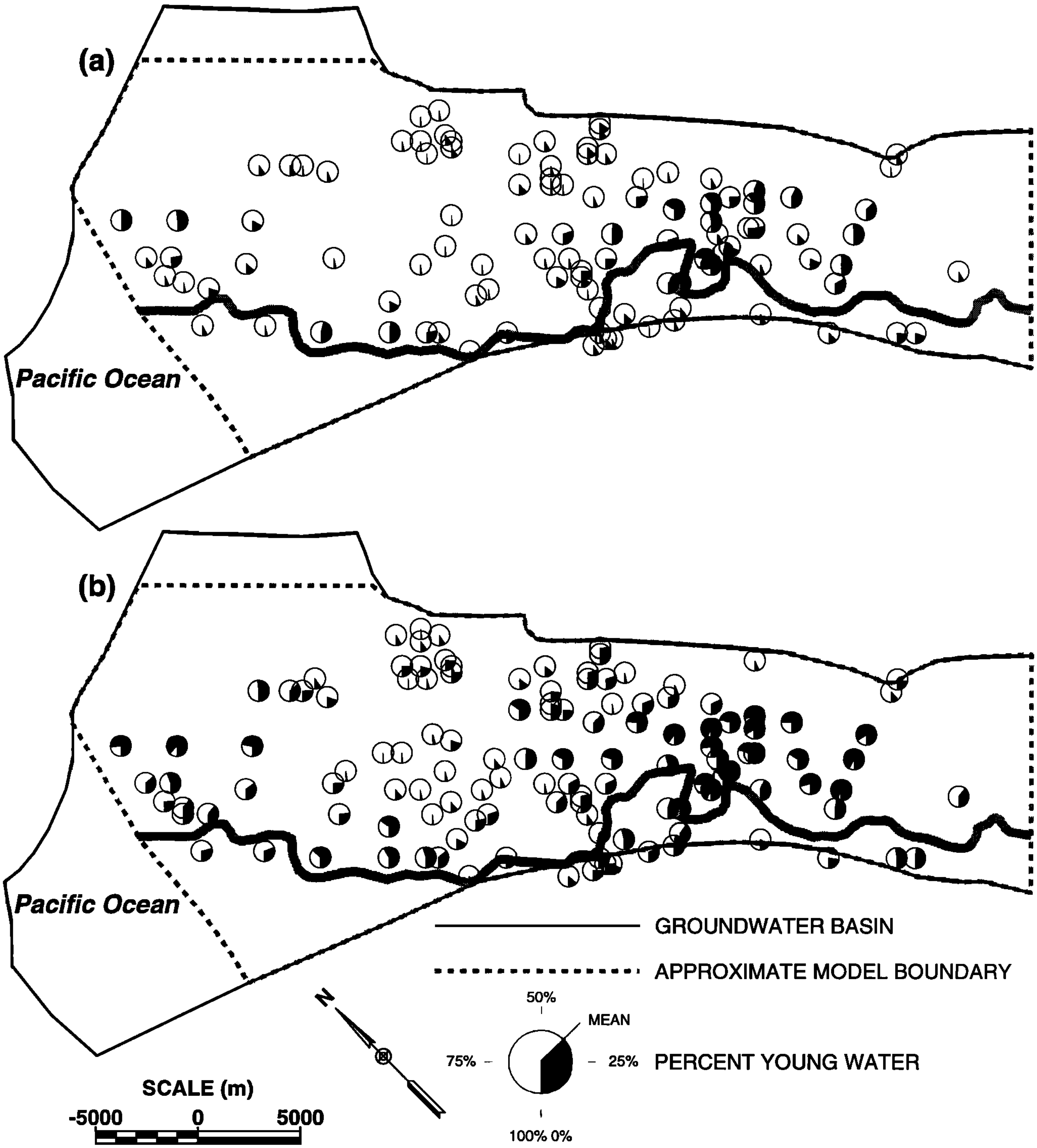
Hydrostratigraphy

The Salinas Valley example demonstrates the importance of the geology or hydrostratigraphy on vulnerability. The patchy patterns in nitrate contamination in the western part of the study area and the large "hot spot" near the middle of the system could not have been simulated without the complex hydrostratigraphy in the model.

Furthermore, we assert that hydrostratigraphic characterization is a potentially invaluable tool for vadose zone analysis, particularly in arid or semiarid basins where the water table is fairly deep (>10 m). Below the root zone in most basins, most of the porous media are not "soils," but rather geologic materials that have predictable, mapable spatial patterns which can, in turn, provide templates for estimating and scaling vadose-zone properties [*Fogg and Nielsen*, 1994; *Fogg et al.*, 1998; *Frohlich et al.*, 1995]. An unfortunate byproduct of the predominantly shallow investigative approach to groundwater vulnerability is the tendency for investigators to conceptualize the subsurface heterogeneity and processes solely in terms of the surface soils, which constitute only a tiny fraction of the system within which transport takes place.

Importance of Time

The Salinas Valley example highlights the importance of time. The particle simulations suggest that the nitrates observed in wells in 1988 originated predominantly from surface sources of the 1940's or 1950's. Groundwater flow rates in the Salinas Valley are not unusually low. In fact, flow rates are fairly high in many areas because of the abundant, coarse-grained alluvial materials. Clearly, water



Figures 10a and 10b. Transport simulation results (scenario 1) showing average percent water younger than (a) 100 years and (b) 200 years in withdrawals from approximately 55-m (180-ft) depth, corrected for induced infiltration from Salinas River.

quality in many groundwater basins, particularly the substantial groundwater resources at depths from which the most productive water supply wells produce, responds to surface phenomena on time scales of decades to centuries, not months or years. Groundwater protection plans need to consider this time lag; otherwise expectations regarding the timing of groundwater quality improvement may be unrealistic.

During most of this time lag, contaminants tend to reside in the saturated zone. Thus, the assumption that vulnerability analysis need only address the probability that contaminants will reach the water table and that the deeper groundwater system can be treated as a "black box," is often inappropriate. Contaminant concentrations can change radically due to transport and transformation processes in the saturated zone.

Differences With Shallow Approach

There exist important differences between predictive analysis of transport in the vadose-zone and groundwater zones. It is well known that vadose-zone transport processes are more complex than are saturated zone processes. Data on vadose zone parameters (e.g., retention curve and unsaturated hydraulic conductivity) and conditions (pressure head, moisture content and contaminant concentration) are seldom available, and the problem of upscaling parameters to regionally representative quantities is still a research problem at the frontiers of hydrologic science. In contrast, physics of flow of fresh, isothermal groundwater is very well understood and accurately represented with the standard groundwater flow equation. Groundwater flow parameters are frequently measured using well tests that average, or regionalize, over large volumes of the medium, lessening the need to upscale; and hydraulic head and contaminant concentration data are often relatively abundant via water level and quality measurements in wells. Furthermore, upscaling or regional parameter estimation is facilitated in groundwater analysis by hydrostratigraphic characterization that helps identify regional patterns in parameters and by the frequent ability to calibrate against extensive historical records of fluctuations in hydraulic head and concentration.

By contrasting predictive capabilities in the groundwater and vadose zones, we are not arguing that analysis of the former should take precedence over the latter. We are merely emphasizing that there is much to be learned about groundwater vulnerability by performing groundwater analysis. One should not assume that just because the groundwater system is deeper and less accessible than the vadose zone, that the groundwater hydrology is less well-understood or less amenable to characterization. Indeed, we know of no groundwater vulnerability analysis that has relied on assessment of shallow, vadose-zone processes and has been validated against actual measurements of groundwater contamination regionally. In contrast, travel time maps of the type provided herein (Figs. 7a and 7b) can potentially provide clear, verifiable indications of which areas are most vulnerable to contamination.

Extensions of Approach

To some extent, the BTADE technique for simulating solute residence times and sources can be adapted for more reactive contaminants than nitrate by including first-order decay and linear sorption, but in the backward-time sense. It is possible that more complex reactions could also be handled, but that would likely require more theoretical development. Importantly, it is also quite feasible to apply either BTADE or a forward simulation of transport implemented with the random walk particle method to a regional continuum that includes both the vadose and saturated zones. Further, the particle releases could be distributed throughout the system to accomplish a more comprehensive analysis of vulnerability.

The Salinas Valley groundwater analysis could be improved with more representative hydrostratigraphic characterization and perhaps transient modeling. If that type of analysis is combined with comprehensive source and vadose zone characterization, it appears that a groundwater vulnerability analysis of unprecedented reliability would be attainable with a moderate effort in the Salinas Valley, as well as other basins.

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