

**POTENTIAL GROUNDWATER IMPACTS FROM
CHEMIGATION**

by

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Colorado Water

Resources Research Institute

Completion Report No. 141

**Colorado
State
University**

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September 30, 1986

Grant No. 14-08-0001-G1006
Project No. 02

The research on which this report is based was financed in part by the U.S. Department of the Interior, Geological Survey, through the Colorado Water Resources Research Institute; and the contents of this publication do not necessarily reflect the views and policies of the U.S. Department of the Interior, nor does mention of trade names or commercial products constitute their endorsement by the United States Government.

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Norman A. Evans, Director

Investigation of Potential Groundwater
Contamination From Chemigation

ABSTRACT

Chemigation is the practice of applying fertilizers/pesticides through a center pivot sprinkler irrigation system. Chemigation is becoming increasingly more common and has several economic and environmental advantages. However, regulating agencies have expressed concern about the potential groundwater contamination that may result from widespread use of chemigation. The concern is that if the irrigation well fails during the chemigation cycle then the fertilizer/pesticide may be back-siphoned down the well and into the aquifer. In this study the fate in the groundwater system of these back-siphoned chemicals was investigated using a digital contaminant transport model developed at Colorado State University by the principal investigator. The model was used to simulate the effect of various breakdown scenarios, aquifer properties and stresses, quantities back-siphoned, and different chemical characteristics of the contaminant. Both conservative and nonconservative contaminant transport was modeled. In the conservative case, nitrate was simulated to represent the back flow of a fertilizer down the well. The typical chemigation system layout simulated for this case consisted of a quarter section sprinkler and a 1,000 gallon chemical storage tank. In the model simulations the removal of the nitrate from the aquifer was accomplished by restarting the irrigation well. About one and half days of pumping were required to lower the nitrate concentrations to safe levels. A delay of one week in restarting the irrigation well after breakdown increased slightly the pumping time required

to remove the nitrate from the aquifer. In the nonconservative case, a pesticide was simulated to represent the back flow of a reactive contaminant that would be adsorbed on the solid aquifer skeleton. The same chemigation system layout was used in this case as was used for the nitrate case except that a 30 gallon chemical tank was used. The adsorption of the pesticide greatly retarded the movement of the contaminant away from the irrigation well. However desorption occurred slowly and it greatly increased the pumping time required to remove the contaminant from the aquifer. For a strongly adsorbed contaminant, the movement of the contaminant away from the irrigation well was limited to only a few feet. However the pumping time required to lower the contaminant concentration to safe levels may be more than 20 days. For pesticides the effect of hydrolysis (break down of the chemical composition of the contaminant in the presence of water) may significantly decrease the time required to remove the contaminant from the aquifer. The affect of aquifer properties did not in general have a significant affect on restoration time requirements. In all cases studied, it was possible to restore the contaminant concentration in the aquifer to safe levels by restarting the irrigation well.

ACKNOWLEDGEMENTS

Funding for this project was provided by the U.S. Department of Interior, Geological Survey, through the Colorado Water Resources Research Institute, Grant No. 14-08-001-G1006. Additional financial support was also provided in part by Colorado Experiment Station project No. 1-51101 and National Science Foundation supercomputer Grant No. ECS-8515101.

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CHAPTER I
INTRODUCTION

1.1 Problem Description

An increasingly common method of applying fertilizers and pesticides to crops is through center-pivot sprinkler systems. This practice is referred to as chemigation. Shown on Figure 1 is a recommended chemigation system layout. There are several advantages in using this method of application, but it does have a disadvantage.

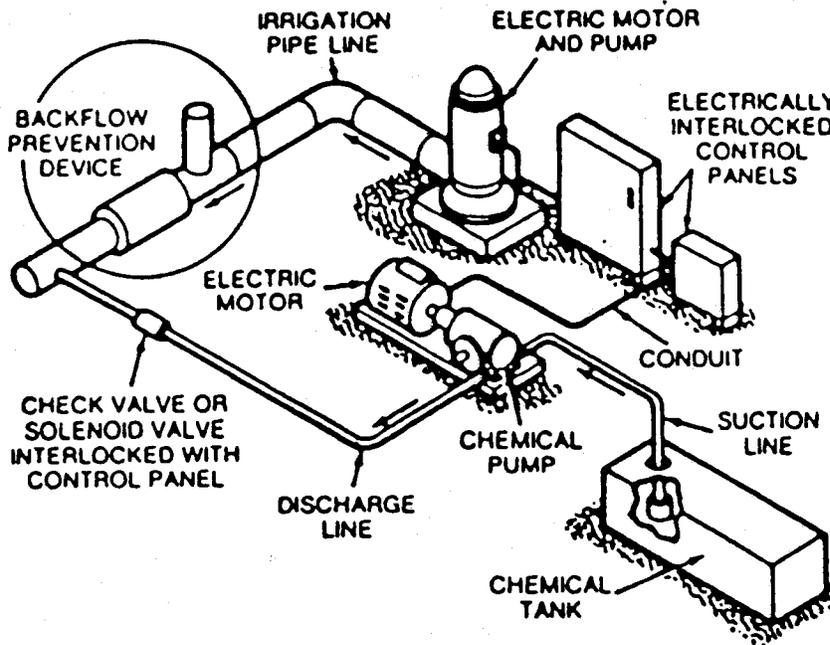


Figure 1--Recommended chemigation system layout.
(Source - American Society of Agricultural Engineers)

If the pump fails during chemigation then the fertilizer or pesticide may be back-siphoned and travel back through the piping system to the well, and may accidentally migrate into the aquifer. It is recommended that check valves be placed between the well and the point in the system where the chemical is

introduced to prevent this back-siphoning. However, there are many chemigation systems without these check valves. Even for systems with a back-flow preventative device, there is still the real possibility that the check valve will fail when the pump fails.

Consider the large number of center-pivot sprinkler systems that there are in this country. If pump failure occurs only a small fraction of the time when chemigation is taking place and if the check valve fails on only a small percentage of these cases of pump failure, then there is still the likelihood that in a typical year there are probably several occurrences when both the pump and check valve fail during chemigation and the fertilizer or pesticide is back siphoned. As the practice of chemigation becomes more common and as these systems begin to age, then the likelihood increases that system failure would result in the chemical being introduced into the aquifer. To date, an actual field case has never been reported where back flow of chemicals has occurred during chemigation. By using a numerical groundwater contaminant transport model, it has been possible to simulate the effect of various chemigation breakdown scenarios. This research considers the fate in the aquifer of these back-siphoned chemicals.

1.2 Objectives of Research

The objective of this research is to investigate the potential groundwater contamination threat posed by chemigation. Specifically, fertilizers in the form of nitrates and the pesticide Lorsban which is manufactured by Dow Chemical were studied. The nitrates act as a conservative tracer while Lorsban acts as a non-conservative contaminant in the aquifer and should have chemical characteristics typical of many other

pesticides used in chemigation. During chemigation, water pumped from the well creates a drawdown cone in the vicinity of the well, causing groundwater gradients toward the well. This tends to trap the back-siphoned chemical within the well bore. The back-siphoning of the chemical and of the water in the center-pivot piping system creates a local head in the well. If this local head in the well is above the water-level in the aquifer, then the chemical will further migrate from the well into the aquifer.

The major question addressed by this research is: if the back-siphoned chemical enters the aquifer, then will these contaminants be flushed from the aquifer when the pump on the well is restarted. Immediately after pumping is resumed, groundwater gradients toward the well will be re-established and uncontaminated groundwater outside of the affected area will begin to flow towards the well. This should cause the back-siphoned chemical that has entered the aquifer to migrate back towards the well. In theory, if the well is pumped long enough then uncontaminated groundwater from the surrounding unaffected area would eventually sweep all of the contaminants from the aquifer. In essence, even if the chemical does enter the aquifer it may be a simple process to remove it by simply restarting the pump on the well.

Alternatively, it may not be so easy to remove the chemical that has entered the aquifer. The problem is complicated by the fact that some chemicals, such as most pesticides, adsorb onto the solid aquifer material. This adsorption is beneficial in that it retards the migration of the contaminant away from the well. However, the disadvantage is that desorption can occur very slowly and can complicate the removal process. Aquifer properties, pumping rates, regional groundwater flow patterns in the

vicinity of the well, amount of chemical introduced into the groundwater system, time between when breakdown occurs and is discovered and time to restart of the well, etc. may all significantly affect the removal process. This research attempts to identify which of these factors are important and how they affect the removal process.

The specific objectives of this study are as follows. For a typical chemigation breakdown scenerio:

- (1) What is the distribution of back-siphoned chemical in the aquifer after well breakdown?
- (2) What is the time rate of change of concentration of the back-siphoned chemical in the well after pumping is restarted?
- (3) What is the time required to restore by pumping the irrigation well, the back-siphoned chemicals to safe concentration levels?
- (4) How do factors such as aquifer properties, quantity of chemical back-siphoned, chemical characteristics of back-siphoned chemical, etc. affect the removal process?

CHAPTER II
NUMERICAL MODEL

2.1 Description

A numerical groundwater transport model, GWTRAN, developed by the principal investigator, Dr. James W. Warner, as part a groundwater modeling package from Colorado State University, was used in this study. The reader interested in the mathematical development and documentation details of GWTRAN is referred to Warner(1981).

Program GWTRAN is a Galerkin finite element groundwater transport model with triangular elements and linear shape functions. The model can simulate both transient flow and transient transport and allows for both spatially varying aquifer properties and for time varying boundary and initial conditions. The model can simulate both conservative (nonreactive) transport as well as nonconservative (reactive) transport. For reactive transport the Freundlich isotherm is used. This works fairly well for most groundwater problems. This model was used to simulate typical conditions of well failure during chemigation and subsequent restart to determine the affect on concentrations of back-siphoned chemicals in the groundwater.

2.2 Finite Element Method

In the finite element method, the domain of interest is discretized into a number of subdomains called elements. Triangular elements were used here since it is possible to represent irregular boundaries and they can be concentrated in regions where rapidly varying solutions are expected, such as at a well.

Using these elements, a continuous function is replaced by values of the function that are specified at a finite number of discrete points, called nodes. The values between the nodes are then calculated using piecewise continuous interpolating functions defined for each element. The definition of head and concentration throughout the problem domain in the finite element method permits the application of variational or weighted residual techniques. The Galerkin method is based on a particular weighted residual principle which turns out to be equivalent to a variational principle, if one exists for the problem under consideration (Wang, Anderson, 1982).

The Galerkin method is applied to the groundwater flow equation and to the advection-dispersion equation for this problem where the linear differential operators L and L' (shown for one-dimension for illustrative purposes only) are defined as:

$$L(h) = T \frac{\partial^2 h}{\partial x^2} - S \frac{\partial h}{\partial t} = 0 \quad (1)$$

and

$$L'(C) = \frac{D}{R_d} \frac{\partial^2 C}{\partial x^2} - \frac{V}{R_d} \frac{\partial C}{\partial x} - \frac{\partial C}{\partial t} = 0 \quad (2)$$

where

- h = $h(x,t)$ = potentiometric head
- T = $T(x,t)$ = aquifer transmissivity
- S = $S(x,t)$ = storage coefficient
- C = $C(x,t)$ = dissolved concentration of the solute
- D = $D(x,t)$ = coefficient of hydrodynamic dispersion

R_d = retardation factor due to adsorption

$V = V(x,t)$ = average interstitial velocity.

To solve $L(h)=0$ and $L'(C) = 0$, trial solutions h' and C' are assumed which are made up of linear combination of shape functions which define the trial solution throughout the problem domain at selected points. If h' and C' were the exact solutions then $L(h') = 0$ and $L'(C') = 0$ everywhere in the problem domain. But since they are only approximate solutions, there will be an error or residual R which is defined as:

$$\begin{aligned} L(h') &= R & \text{and} \\ L'(C') &= R' \end{aligned} \tag{3}$$

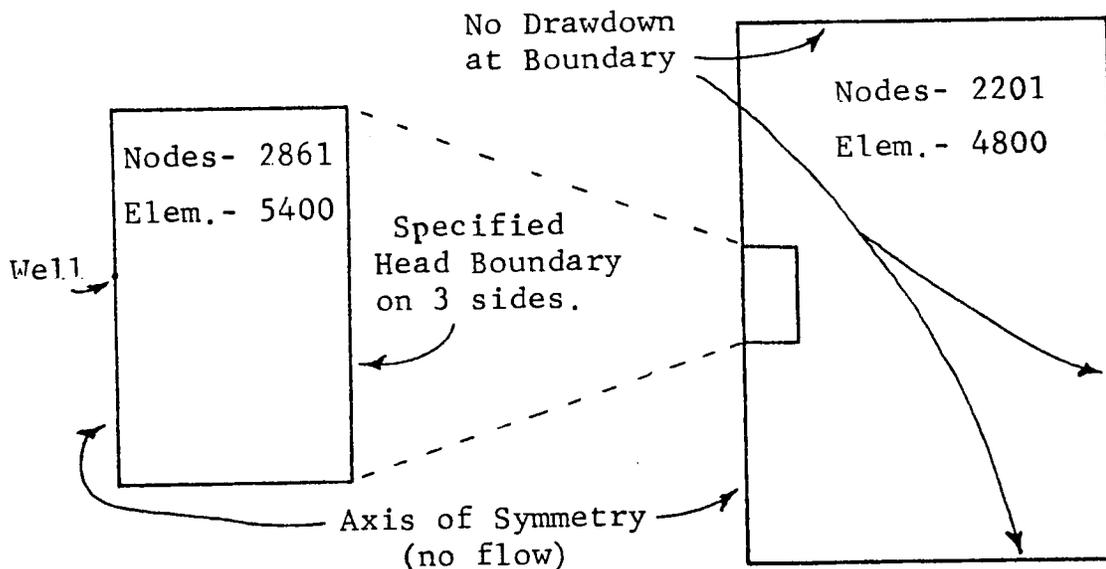
These residuals are forced to zero, in an average sense over the entire domain D using a weighted residual technique. The shape functions that are used in this finite element model are linear functions since less computational effort is required than with higher order polynomials, while still providing reliable results.

At a given time step the groundwater flow equation is solved sequentially with the advection-dispersion equation in a leap-frog solution technique. The groundwater flow equation is first solved for the head distribution in the aquifer at the specified time from which the values of the groundwater velocity and the dispersion coefficient are obtained. These values are then used in the solution of the advection dispersion equation at that time step to solve for the contaminant concentration in the aquifer.

2.3. Finite Element Mesh

Two conditions of the chemigation problem initially caused some difficulties in this study in constructing a suitable mesh. These were: (1) the boundary of the flow model should be located far enough removed from the well so as to simulate an infinite aquifer, in essence no appreciable drawdown at the boundary due to pumping of the well, and (2) a detailed mesh in the immediate vicinity of the well so as to accurately describe the distribution of contaminants near the well. The first condition required a large grid, which when coupled with the second condition of a detailed mesh around the well, resulted in a mesh with a very large number of nodes and elements.

To satisfy both of these criteria, a solution procedure using two separate meshes was used in this study. A large mesh, measuring 10,000 feet from well to boundary, and a small mesh measuring 100 feet from well to



(2a) Transport Mesh

(2b) Flow Mesh

Figure 2--Schematic of the small and large finite element meshes.

boundary, were constructed (figures 2 and 3), each satisfying one of the conditions. For the grids shown, symmetry was used and only half the flow field was modeled.

The large mesh, designated the flow mesh, has a very coarse grid arrangement and is used to calculate only the head distribution using the flow part of the model. The condition of zero drawdown at the model boundaries was met using this mesh. The smaller mesh (smaller in physical size but not in the number of nodes or elements), designated the transport mesh, was used to calculate both the potentiometric head distribution and the contaminant concentration distribution in the vicinity of the well. The boundary conditions on the smaller transport mesh were specified heads, the values of which were determined using the larger flow mesh.

2.4 Modeling Procedure

The modeling process consisted of a two-step procedure. First a model run was performed using the flow mesh to solve for the aquifer heads at the time of interest. The head values, at what were the locations of the specified head boundary for the transport mesh, were outputted after each time step. A second model run was then performed using the smaller or transport mesh for the same time period using the boundary head data generated by the larger flow mesh. At the beginning of each time step the boundary head elevation for the transport mesh was specified as a constant head condition for that time step (figure 4).

This effectively simulated the larger mesh requirement of zero drawdown at a distance of 10,000 feet from the well but then also allowed for a very fine grid spacing in the immediate vicinity of the well for calculations of

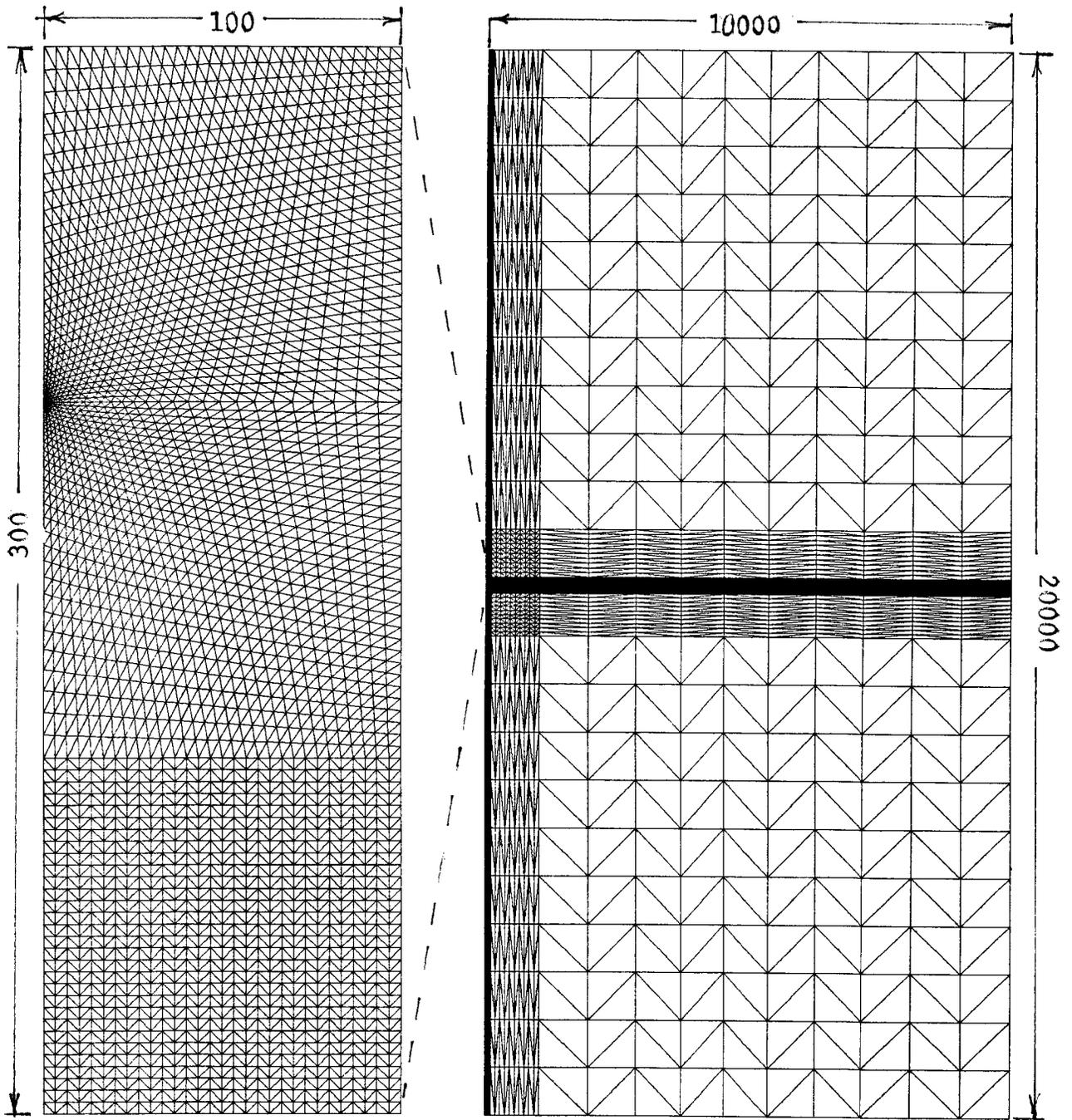


Figure 3--Discretization of the small and large meshes.

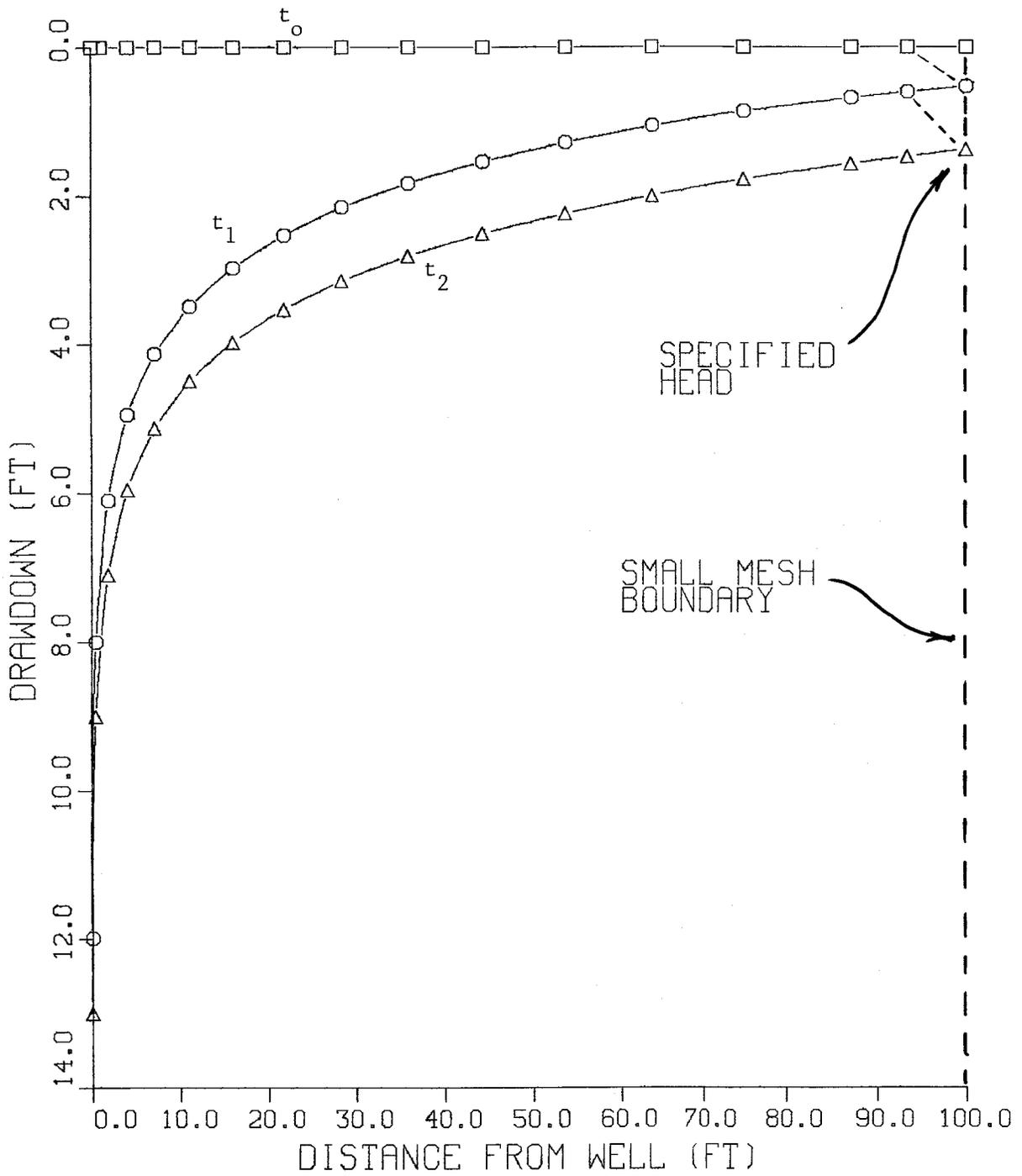


Figure 4--Illustration of specified head boundary for two time steps.

concentrations without have the fine detail at larger distances from the well where it was not needed. In this manner it was possible to have 8 nodal points within 10 feet of the well which was necessary because of the limited movement in the aquifer for the case of an adsorbing contaminant. A comparison of the head distributions for the two meshes reveals a close agreement in the region common to both meshes (figure 5).

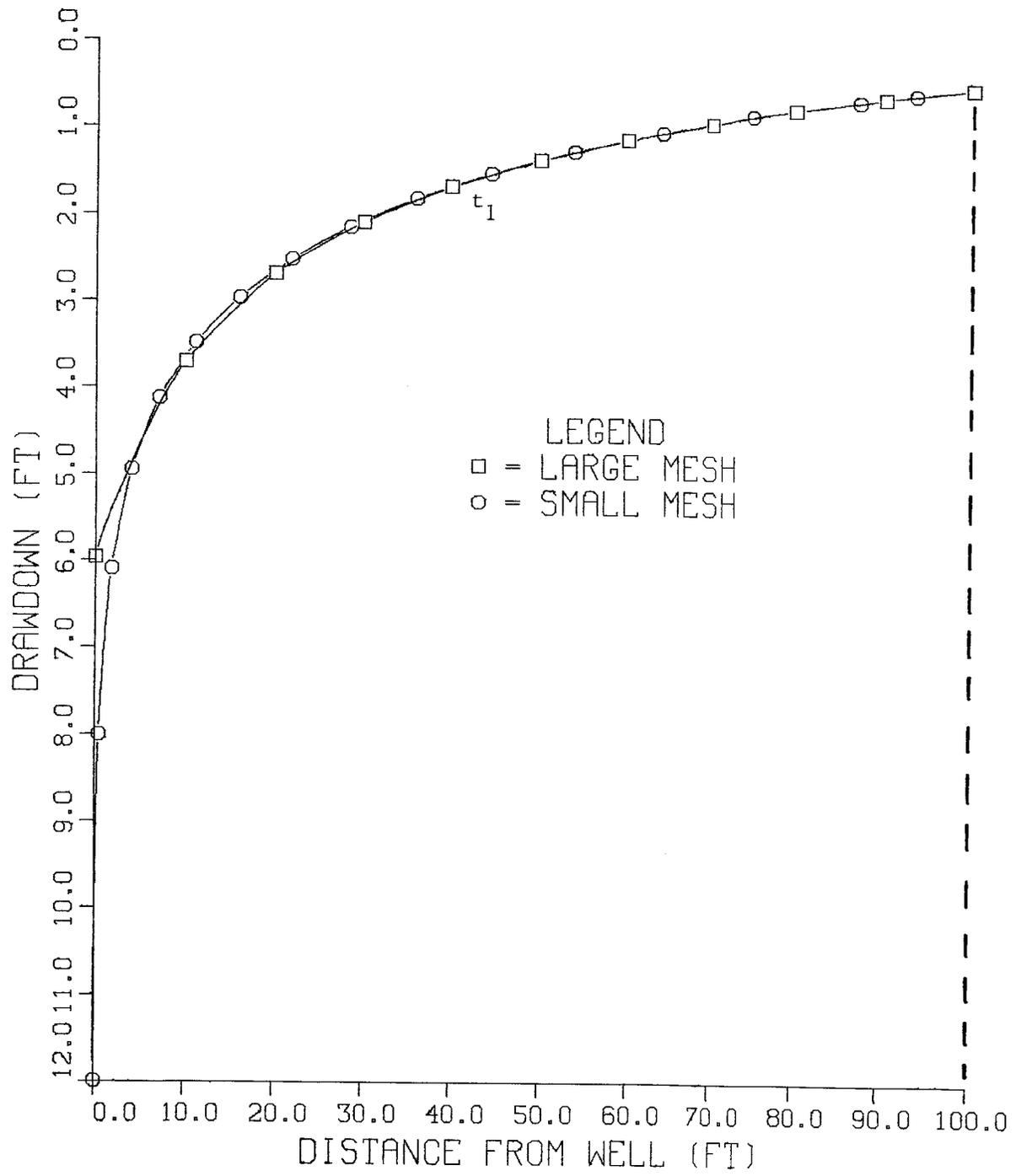


Figure 5--Comparison of drawdowns calculated using the large and small meshes.

CHAPTER III

AQUIFER CHARACTERISTICS AND CHEMIGATION SYSTEM GEOMETRY

3.1 Aquifer Characteristics

The practice of chemigation is becoming increasingly widespread particularly in the high plains region. As such, conditions typical for the Ogalla aquifer were selected for study. The aquifer data used as input into GWTRAN in this study are average values obtained from the U. S. Geological Survey High Plains project for the Ogalla Aquifer in Nebraska (Gutentag, et al., 1984). The aquifer in the local vicinity of the well, for simulation purposes, was considered homogeneous, isotropic and of infinite areal extent.

3.1.1 Saturated Thickness

The greatest percentage of the high plains aquifer is located in Nebraska, due both to the areal extent of the aquifer and also the large saturated thickness of the formation in this region. The saturated thickness of the aquifer in Nebraska ranges from 200 feet to greater than 600 feet. An average value of 300 feet was used in the model simulations.

3.1.2 Hydraulic Conductivity

The transmissivity of the Ogalla aquifer in the high plains of Nebraska ranges from 5,000 to 30,000 feet²/day. Using an average value of 16,000 feet²/day and a saturated thickness of 300 feet, yields an average hydraulic conductivity of 54 feet/day.

3.1.3 Storage Coefficient and Porosity

The range in storage coefficient for the Ogalla aquifer is from 0.00014 to 0.22. An average value of 0.15 for specific yield and effective porosity was used in the model simulations.

3.1.4 Dispersivity

A dispersivity value of five feet was in the model simulations. There is a great deal of discussion in the literature concerning dispersivity. It is generally agreed that dispersivity is scale dependent. Also numerical stability considerations for the finite element solution of the advection dispersion equation need to be considered in the selection of dispersivity. Values of dispersivity of a few feet to several hundred feet have been used in numerical groundwater transport models. Because of the small grid spacing used in the model in the vicinity of the well, then a relatively small dispersivity value was used in this study.

3.2 Chemigation System Geometry

The injection volumes used to simulate back-flow of chemicals down the well after pump failure were calculated for a center-pivot system irrigating a quarter section (130 acres) of land, (figure 6). The geometry used and other assumptions made were:

- 1) 6 inch diameter irrigation pipe,
- 2) quarter mile (1320 ft.) of pipe,
- 3) total volume of water in pipe was siphoned into well, and
- 4) total volume of chemical in the storage chemical tank was siphoned into well.

The total volume of water in the pipe is:

$$\text{Pipe volume} = \frac{\pi(.5)^2}{4} * 1320 \text{ ft} * 7.48 \text{ gal/ft}^3 \approx 2,000 \text{ gallons.}$$

In addition to the water in the center pivot piping system, there is also the volume of the chemical solution in the chemical storage tank. In the case of pesticides a small tank of about 30 gallons is commonly used in the field. For fertilizers, a larger tank of about 1,000 gallons is commonly used in the field. In the model simulations, the total volume of liquid that was back-siphoned into the aquifer was about 2,030 gallons for a pesticide and about 3,000 gallons for a fertilizer.

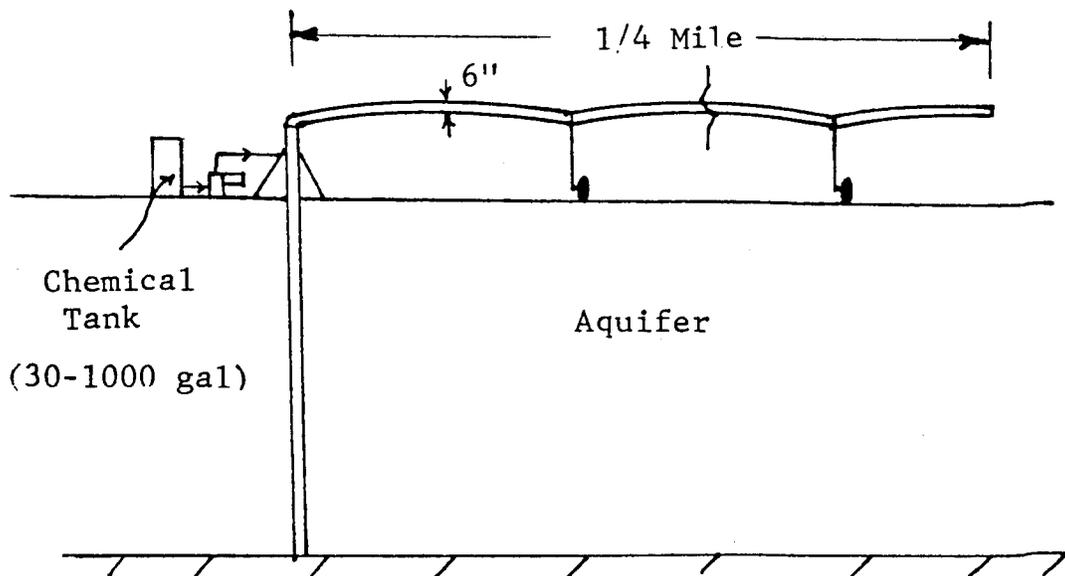


Figure 6 Center pivot system

3.2.1 Pump Failure Scenario

A simulation of the pump failure with a simultaneous failure of the back-flow check valve consists of an initial time period when the water and chemical are being back-siphoned (back flow period) into the aquifer, a time period when the pump is inoperative (breakdown period) and the contaminant is dispersed in the aquifer and migrating away from the well, and a final time period when the pump on the well is restarted (restart period) and the chemical contaminant and water from the well and surrounding aquifer are being pumped back out of the aquifer (figure 7). The length of each time period and the well pumping rates for each of the three time periods was varied in the model simulations to determine the effect on the rate of removal of the contaminant from the aquifer and on the concentration of the contaminants in the aquifer. In addition to these analyses, the aquifer properties and contaminant injection concentrations were varied, reactive and nonreactive chemical simulations were made and a regional groundwater gradient was imposed on the wellfield, to identify their individual effects on the contaminant removal process.

3.2.2 Well Pumping Rates

Most wells connected to a center pivot irrigation system are pumped at a rate of between 750 and 1,000 gallons per minute (gpm). To conservatively estimate the time required to remove the back-siphoned contaminant from the aquifer, a well discharge rate of 750 gpm was used in the model simulations. The rate at which back-siphoning of the chemicals down the well would occur, was thought to be much slower than the discharge pumping rate. A back flow rate of 125 gpm or approximately one-fifth of the

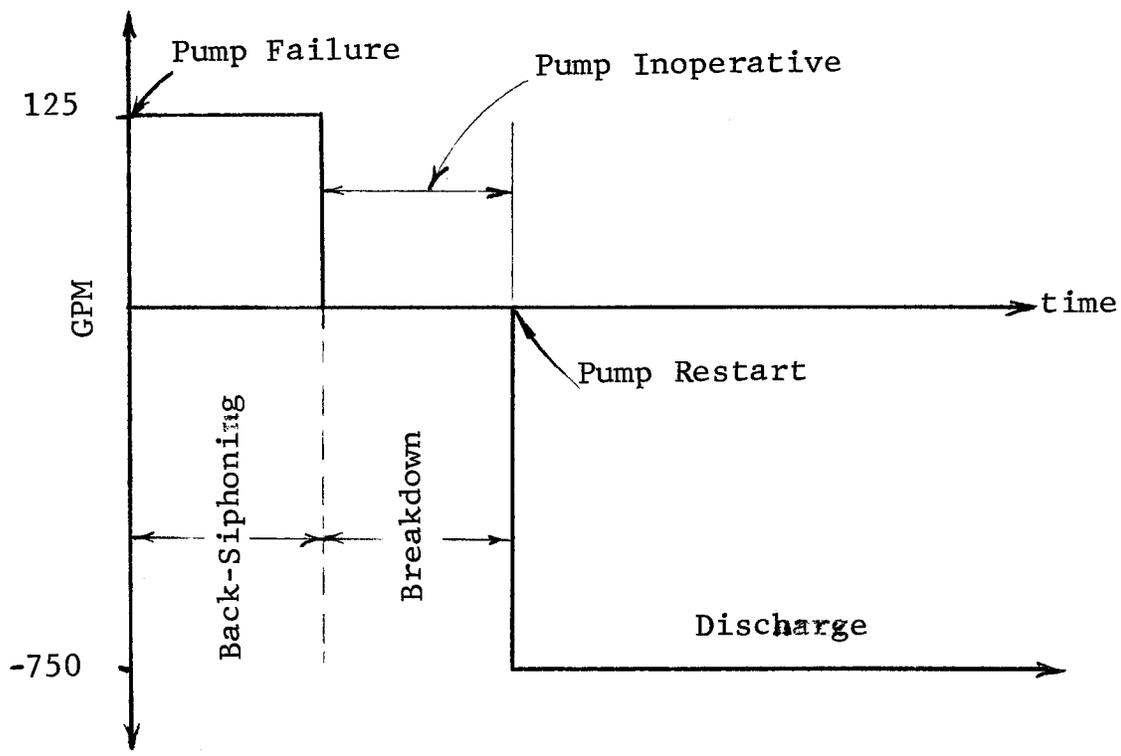


Figure 7--Pump failure scenario

well discharge rate was used. Depending on the quantity of water back-siphoned, the time required for the back flow to occur may be as long as half an hour. Because of the small diameter tubing normally used to connect the chemical storage tank to the center pivot system, the back-siphoning of the chemicals could take longer. The rate and time for back flow to occur was varied in the model simulations with almost no effect observed on the final model results. During the intermittent breakdown period, no back flow of chemicals or well pumping was simulated.

CHAPTER IV

MODEL SIMULATIONS

Two major types of model simulations were performed, conservative transport (non reactive) and nonconservative (reactive) transport. The conservative transport simulations represented the back-siphoning of a fertilizer and the nonconservative transport simulations represented the back-siphoning of a pesticide which adsorbs onto the solid aquifer material. The worst case scenario was simulated of pump failure at the beginning of a chemigation cycle. With this scenario, no cone of depression has developed due to prior pumping of the well before pump failure occurs during chemigation. In this case no drawdown cone has developed to trap the back-siphoned contaminant in the vicinity of the well. The back-siphoning of the water from the irrigation piping system and chemical storage tank, creates a local head in the well greater than the head in the aquifer in the immediate vicinity of the well. This drives the back-siphoned chemical out into the aquifer. Also since pump breakdown occurs at the beginning of the chemigation cycle, all of the chemical in the storage tank that was to be applied to the crop is also back-siphoned down the well.

4.1 Nitrate Simulation

Nitrogen is a major component of most fertilizers. The simulation of a conservative chemical such as nitrate determines the extent to which a chemical would migrate away from the irrigation well by the processes of advection and dispersion. Four different simulations were made for nitrate transport.

4.1.1 Horizontal Water Table

Three of the simulations were made with an initially horizontal water table.

4.1.1.1 Base Simulation

Using an application rate of 30 pounds per acre of Nitrogen, and a total back flow quantity of 3000 gallons, the concentration of the nitrate in the back-siphoned water will be approximately 560,000 milligrams per liter (mg/l). A back flow rate of 125 gallons per minute (gpm) was used in this simulation. After the initial back flow period, the well was shut off to simulate a period of breakdown. Most of the movement of the chemical from the well out into the aquifer occurs during the back flow period and little movement occurs during the breakdown period. The local head build up near the well dissipates very rapidly and the water table returns to nearly horizontal shortly after the end of the back flow period. Mechanical dispersion is caused by local variations in the interstitial velocity of the ground water in the aquifer. The greater the velocity the greater the dispersion. Since head buildup near the well dissipated very rapidly, aquifer velocities were small. Therefore the movement of the back-siphoned chemical out into the aquifer only occurred during the first two hours of the breakdown period, after which aquifer velocities were small and dispersion was negligible. The pump on the irrigation well was restarted at a rate of 750 gpm and operated until the concentration of the groundwater in the well returned to acceptable level of 10 mg/l (Nitrogen) or 45 mg/l (Nitrate), (figure 8).

The concentration of the nitrate at the well just prior to restarting the pump was approximately 255,000 mg/l. After just a few hours of pumping

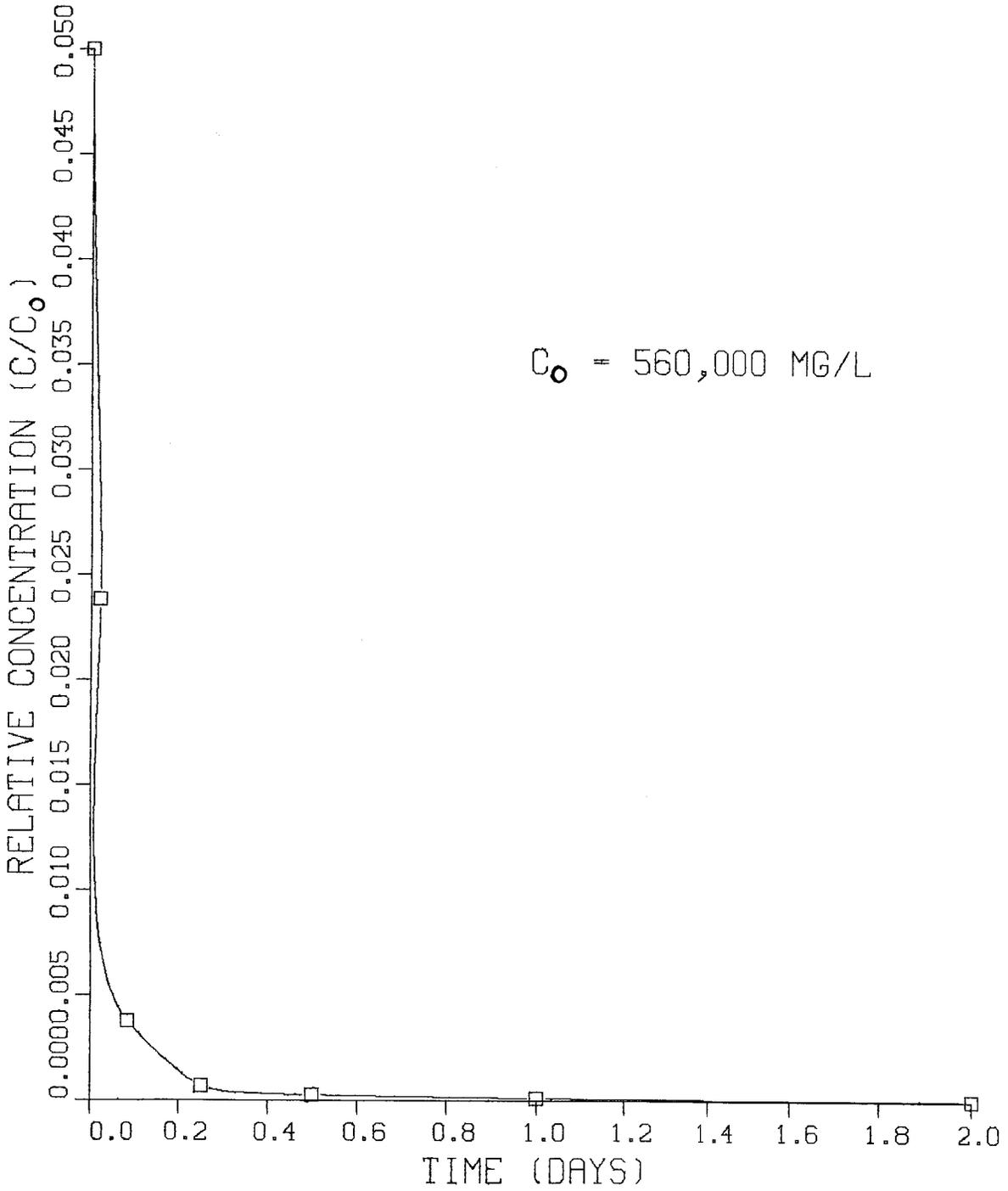


Figure 8--Plot of relative nitrate concentration at the well
verses time during restart period for base simulation.

the bulk of the nitrate is removed. As pumping continues, it is apparent that longer and longer periods of time are required to remove additional nitrate from the aquifer. The concentration distribution of the nitrate in the aquifer with time is shown in figure 9. After one-half hour of pumping the nitrate concentration has decreased from 255,000 mg/l to about 13,000 mg/l and after six hours of pumping has decreased to about 400 mg/l. After about one and half days of pumping the nitrate concentrations at the well have been lowered to the safe level of 45 mg/l (Nitrate). After four days of pumping, for all practical purposes, the nitrate has been completely removed and the aquifer restored. Table 1 summarizes the nitrate concentrations at the well during the well restart period.

4.1.1.2 Half Volume Back Flow Simulation

The possibility exists that only half the water in the center pivot irrigation system would be back-siphoned down the well. The field situation corresponding to this case is a center pivot sprinkler system that straddles a high point in the ground surface when the failure occurs. In this case, the water in the piping system on the side of the hill away from the irrigation well might not be back siphoned. Approximately the same total mass of nitrate would still be back-siphoned from the chemical storage tank. However, due to the decrease in water volume the concentration would be twice that in the previous simulation, 1,120,000 mg/l. The results of this simulation are compared with the base simulation in figures 10, 11, and, 12, and summarized in table 1.

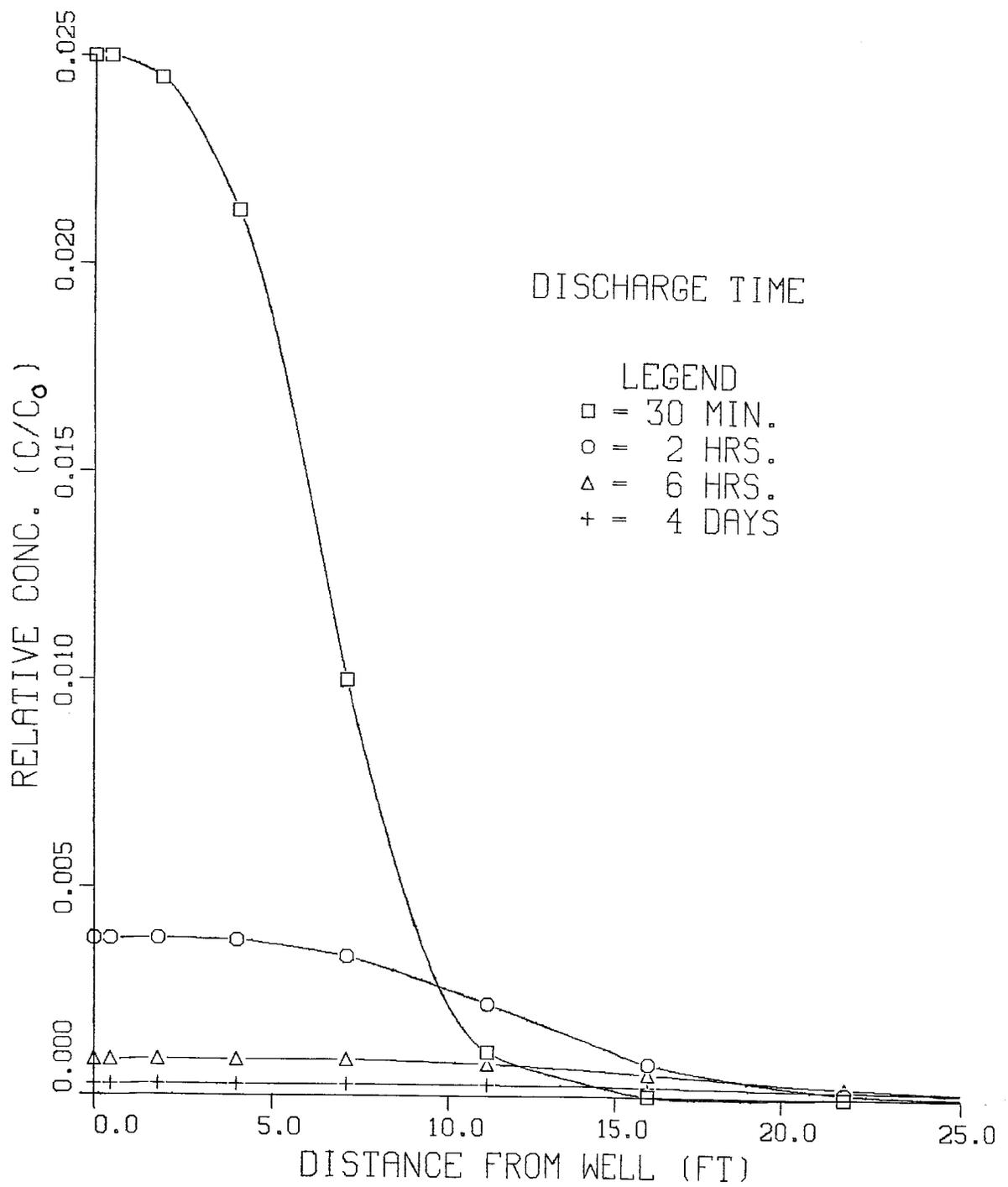


Figure 9--Plot of relative nitrate concentration verses distance from well during restart period for base simulation.

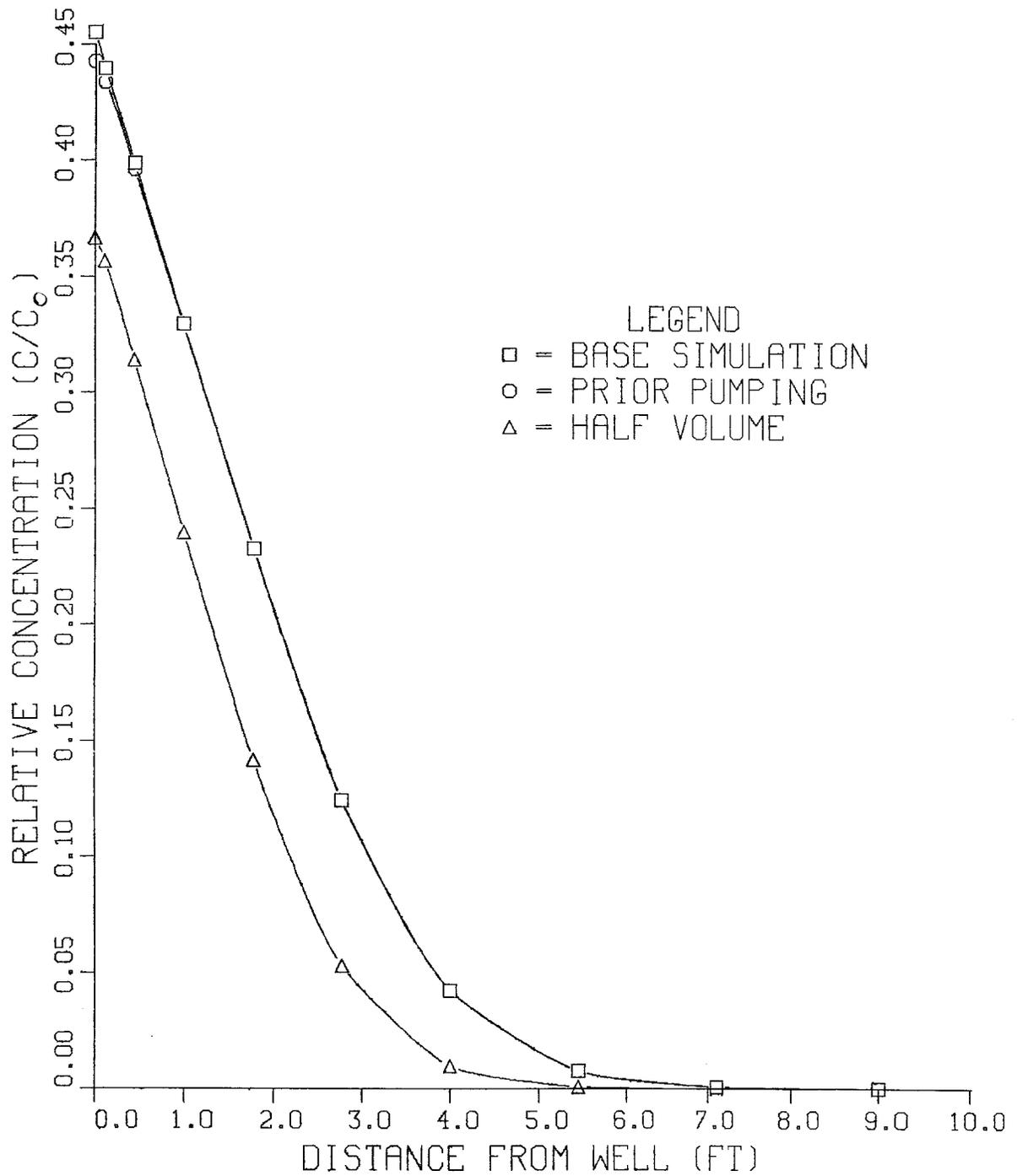


Figure 10--Comparison of relative nitrate concentration distributions versus distance from well at end of breakdown period for base, half-volume and prior pumping simulations

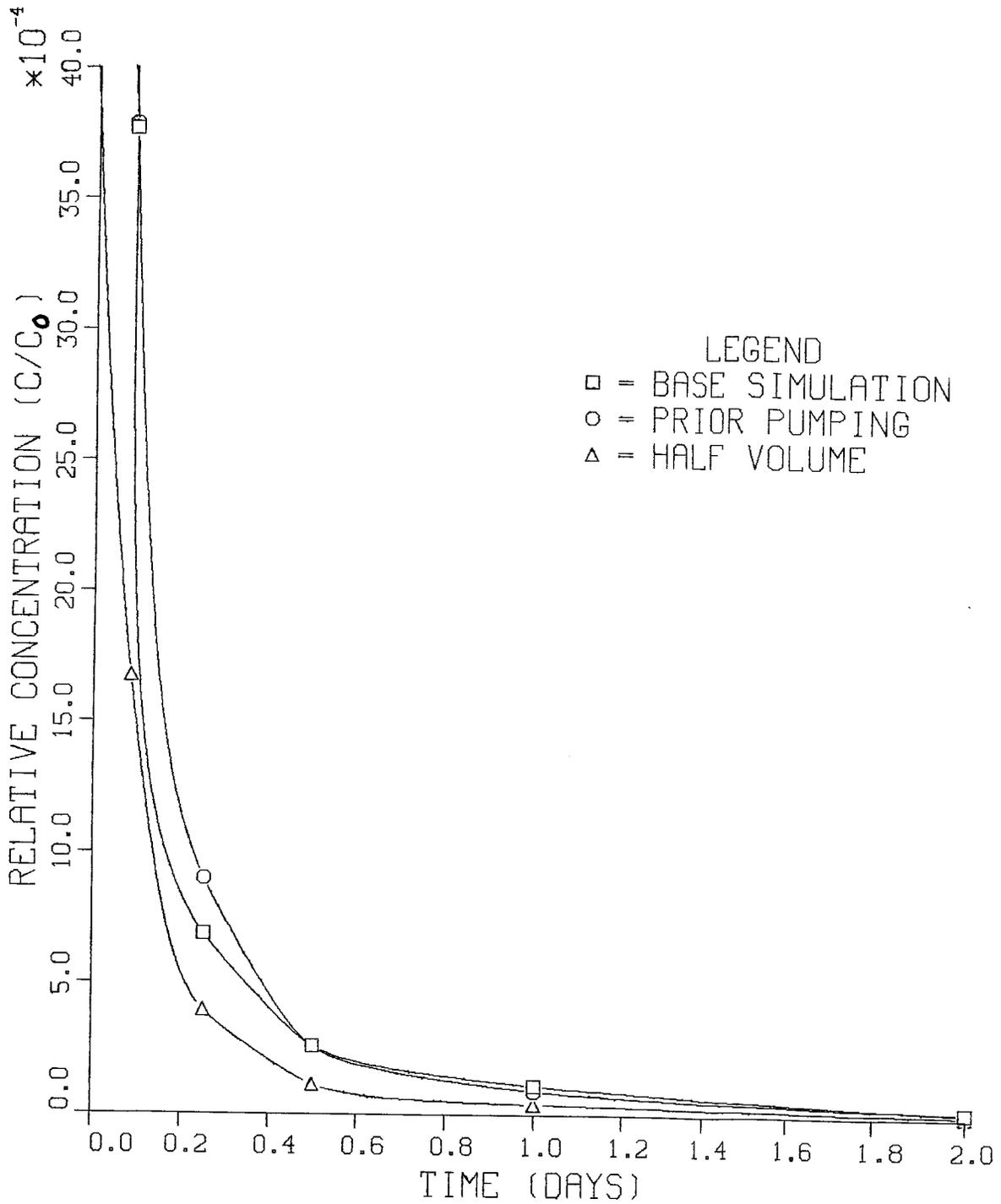


Figure 11--Comparison of relative nitrate concentration at well versus time during pump restart period for base, half-volume and prior pumping simulations.

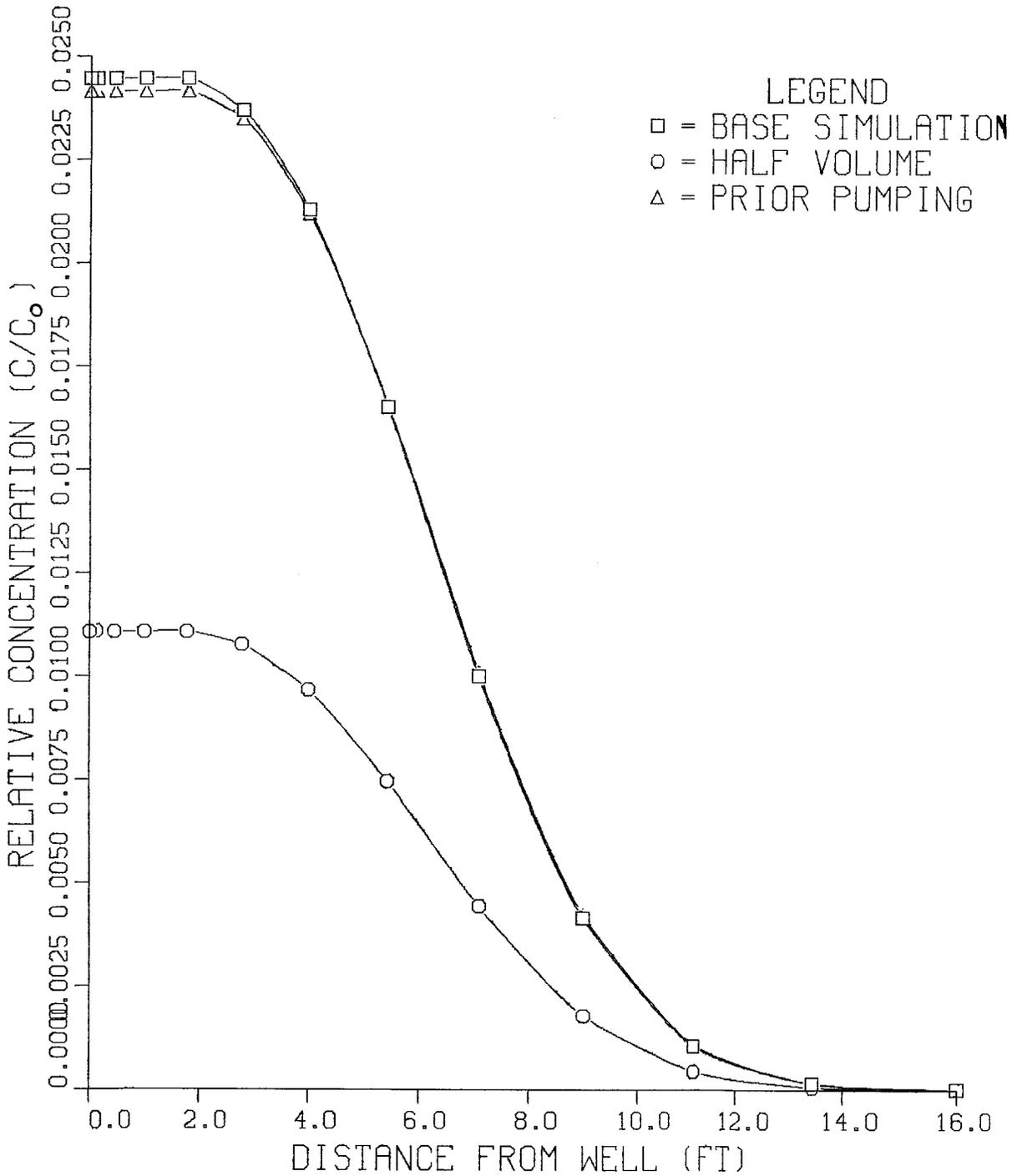


Figure 12--Comparison of relative nitrate concentration distributions versus distance from well at time 30 minutes after pump restart for base, half-volume and prior pumping simulations.

Table 1 Nitrate concentrations at well during restart period for Base, Half-Volume and Prior Pumping Simulations

Co = 560,000 mg/l for Base Simulation
 Co = 1,120,000 mg/l for Half-Volume Simulation
 Co = 280,000 mg/l for Prior Pumping Simulation

Relative Concentrations (C/Co) at the well during restart period

<u>Time</u>	<u>Base</u>	<u>Half Volume</u>	<u>Prior Pumping</u>
(hrs)			
0	.4489437	.3627190	.4331320
.5	.0238300	.0108100	.0235800
2	.0037700	.0016770	.0037880
6	.0006900	.0003970	.0009020
12	.0002600	.0001130	.0002570
(days)			
1	.0001100	.0000400	.0000900
2	.0000200	.0000100	.0000230
4	.0000000	.0000040	.0000080
7	.0000000	.0000020	.0000040

Table 1 (continued)

Actual Concentrations at the Well

<u>Time</u>	<u>Base</u>	<u>Half-Volume</u>	<u>Prior Pumping</u>
(hrs)	(mg/l)	(mg/l)	(mg/l)
0	255000	435262	123009
5	13518	12972	6696
2	2141	2012	1075
6	391	476	256
12	148	135	73
(days)			
1	62	48	26
2	11	12	7
4	0	5	2
7	0	2	1

From figures 10, 11, and 12, the relative concentration of the nitrate for the half-volume case is considerably less than the relative concentration of the nitrate for the base simulation case. However this is misleading since the concentration (C_o) of the nitrate in the back flow water for the half-volume case is 1,120,000 mg/1, double that for the base simulation case. The actual nitrate concentrations at the well are nearly identical for both the base and half-volume simulations (table 1). This is as would be expected. The concentration of nitrate in the aquifer should depend on the total mass of nitrate that is back-siphoned and not on the concentration of the nitrate in the back-siphoned water. In both cases, it takes approximately one to two days of pumping time to restore nitrate concentrations in the aquifer to acceptable safe levels.

4.1.1.3 Prior Pumping Simulation

In this simulation breakdown of the pump does not occur at the beginning of the chemigation cycle but instead occurs one-half way through the chemigation cycle. A drawdown cone has developed at the well and half of the nitrate has been applied to the crops before back-siphoning occurs as a result of the pump failure. Half of the total mass of the nitrate remains in the chemical storage tank to be mixed with the full volume of water in the irrigation pipe and then back-siphoned down the well. The concentration of nitrate in the back flow water in this simulation is 280,000 mg/1, one half the concentration of the base simulation and again only half the total mass. The results of this prior pumping simulation are compared with the base simulation in figures 10, 11, and 12, and summarized in table 1. The relative concentration plots for prior pumping simulation

are nearly identical with the base simulation (figures 10, 11, and 12). However, since only one-half the total mass of nitrate is back-siphoned, the actual concentrations for prior pumping simulation are only one-half those for the base simulation (table 1). Still about one day of pumping is required to lower nitrate concentrations in the aquifer to safe levels.

4.1.2 Nitrate Transport with a Regional Gradient

In this simulation a regional groundwater gradient of ten feet per mile was imposed in the vicinity of the well to investigate the contaminant migration which would occur under these conditions and to determine whether the contaminant could still be removed from the aquifer by restarting the pump on the irrigation well. The ten ft/mile gradient is the average groundwater gradient for the Ogalla aquifer in Nebraska and was obtained from the U.S. Geological Survey High Plains study (Gutentag, et al., 1984). With this gradient and using an average hydraulic conductivity of 54 ft/day and an effective porosity of 0.15 then the regional movement for a conservative contaminant is about 0.67 ft/day.

The same back flow and pumping rates were used in this simulation as was used in the previous simulations, but the breakdown period simulated was increased to seven days to allow the nitrate to be carried away from the well, and to simulate a period of time for the pump failure to be discovered and corrected. The concentration at the well begins to diminish immediately after back-siphoning occurs since the slug of nitrate begins to move down-gradient with the groundwater flow (figure 13). The concentration of nitrate at the well decreases from about 250,000 mg/l immediately after back-siphoning to about 16,000 mg/l at the end of the breakdown period. This is primarily due to the process of dispersion in the aquifer rather

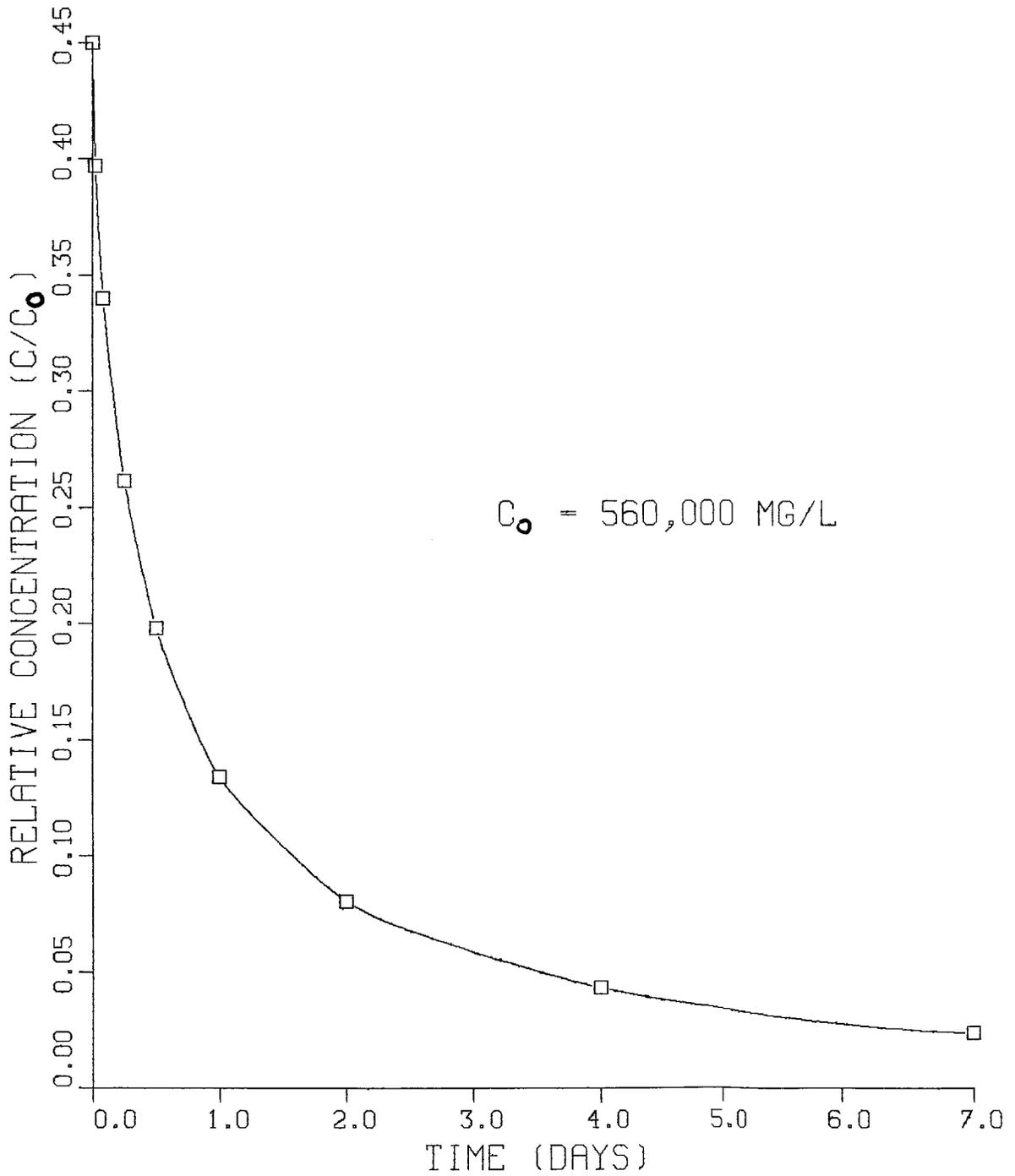


Figure 13--Relative nitrate concentration at the well versus time during back flow and breakdown periods for regional groundwater gradient simulation.

than migration of the nitrate away from the well under the imposed regional groundwater gradient. The nitrate slug has only migrated about 5 feet after 1 week (figure 14). Once the pump is restarted, the concentration in the aquifer and in the well decreases rapidly as uncontaminated water flows in from the surrounding aquifer and the nitrate is pumped back out of the well (figures 15 and 16).

In the previous simulations with no gradient, an acceptable nitrate level was reached after one and half days of pumping. With a regional gradient, three days of pumping are required before concentrations are again returned to acceptable levels of about 45 mg/l nitrate (figure 15 and 16 and Table 2). This indicates that some additional pumping will be required to remove the back-siphoned chemical if a delay of one week occurs before the pump on the irrigation well can be restarted.

4.2 Pesticide Transport

A reactive chemical such as a pesticide will adsorb onto the solid aquifer material, thereby greatly reducing its mobility in the groundwater system and impeding its migration away from the well. The pesticide Lorsban which is manufactured by Dow Chemical was chosen to be modeled in this study. Lorsban is a very common pesticide used in chemigation and has chemical characteristics typical of many other pesticides. The pesticides Arbofuran and Atrazine were also modeled.

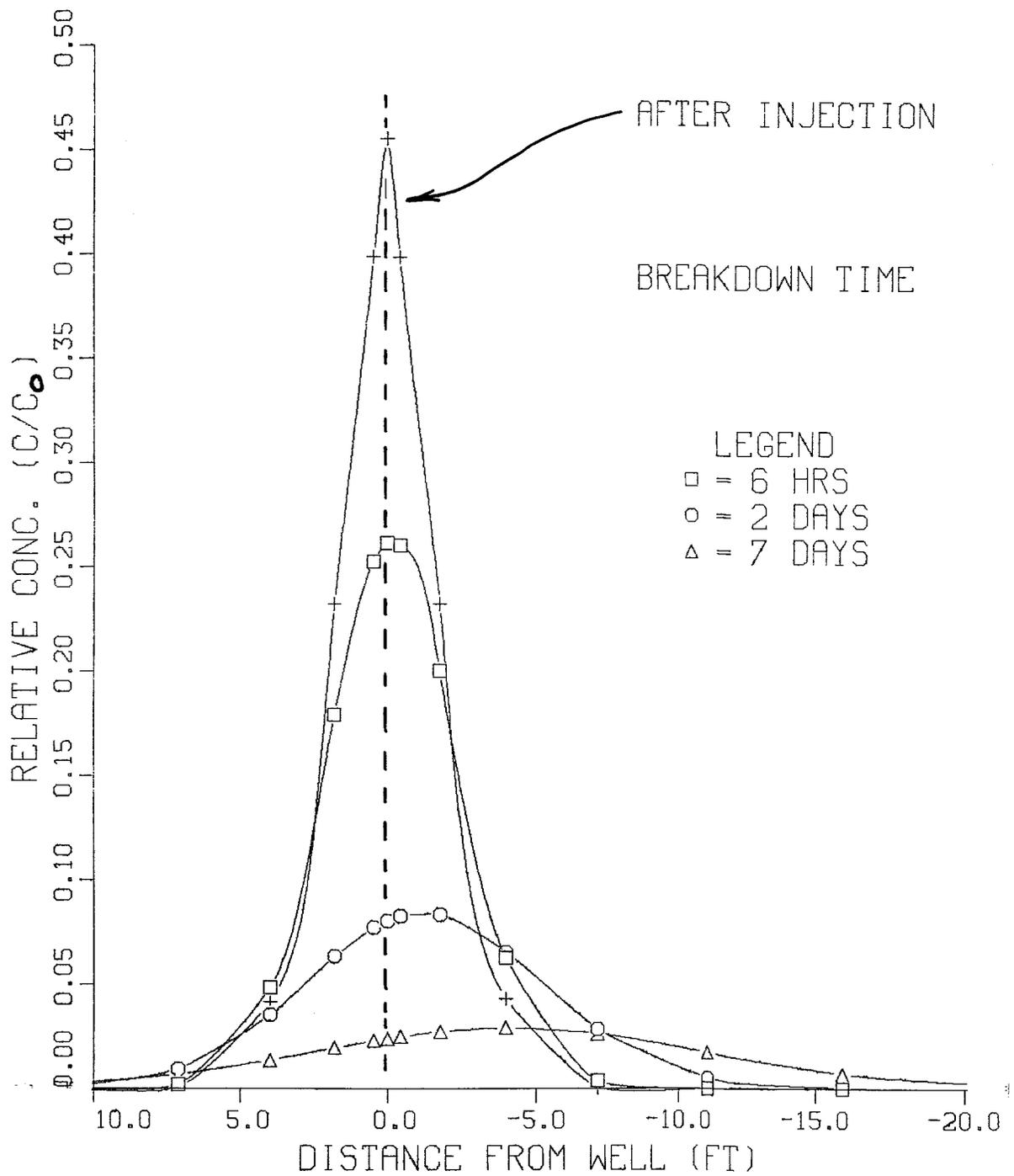


Figure 14--Relative nitrate concentration verses distance from well during back flow, and breakdown periods for regional groundwater gradient simulation.

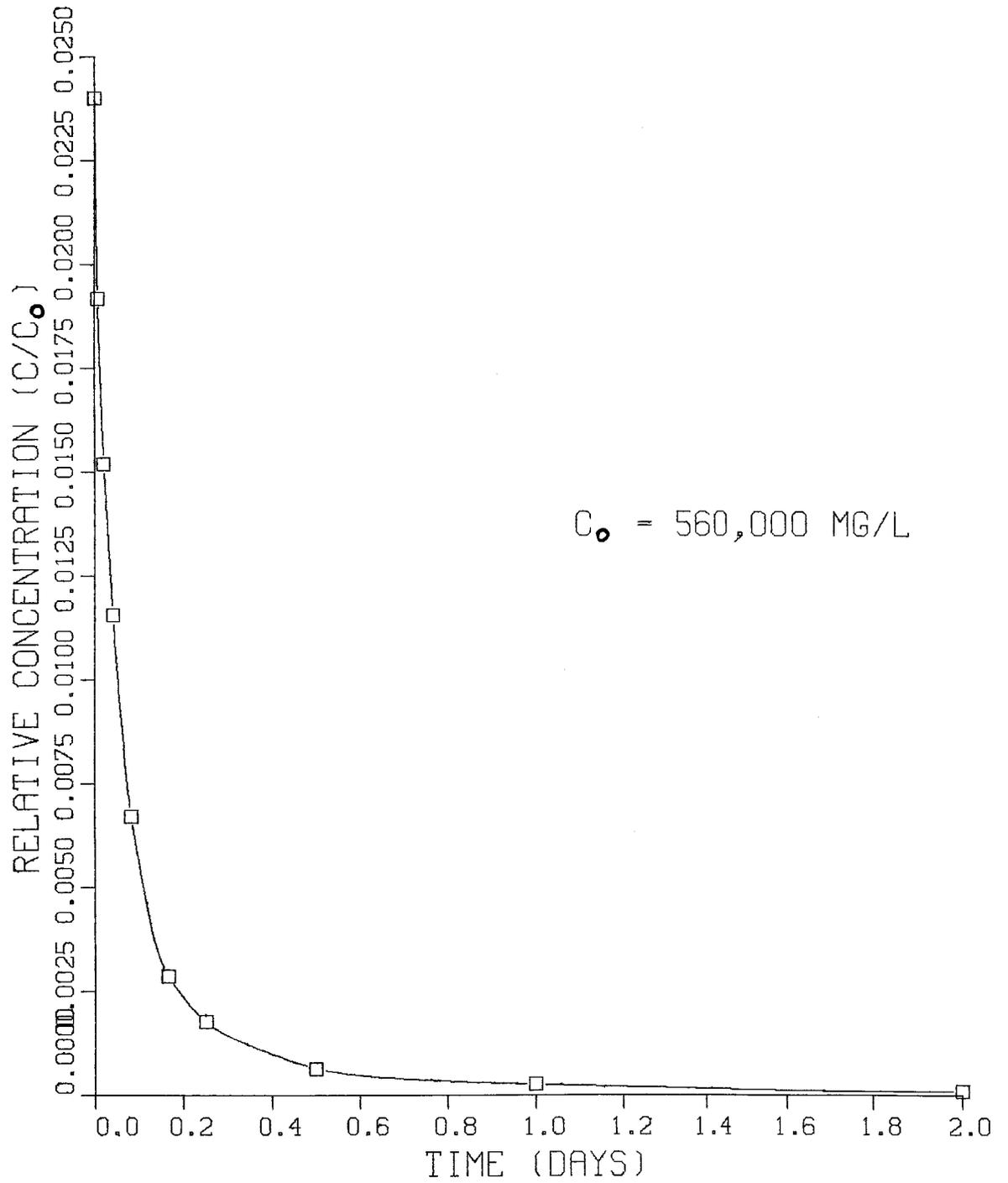


Figure 15 Relative nitrate concentration at the well versus time during restart period for regional groundwater gradient simulation.

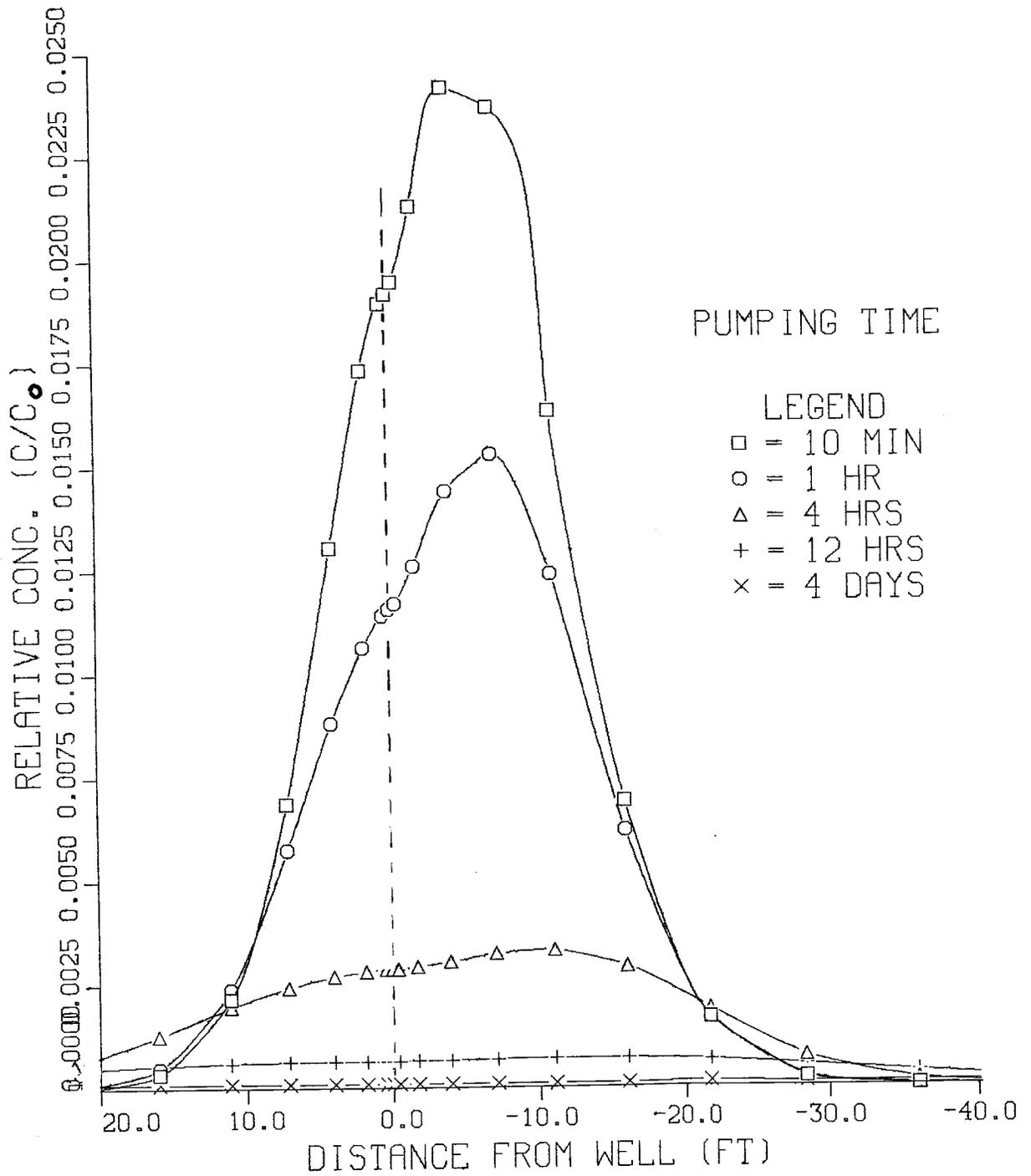


Figure 16--Relative nitrate concentrations verses distance from well during restart period for a regional groundwater gradient simulation

Table 2 Nitrate concentration at well
during restart period for regional groundwater gradient simulation

<u>Time</u> (hrs)	<u>Dissolved Concentration</u> (mg/l)
0.0	13,632
0.2	10,848
0.5	8,628
1.0	6,566
2.0	3,816
4.0	1,624
6.0	999
12.	364
24.0	147
48.0	57
96.0	11

4.2.1 Lorsban Simulation

Lorsban is a registered trademark of the Dow Chemical Company for insecticide products containing the organophosphorothioate chemical chlorpyrifos as the principal active ingredient. Studies show that it is highly adsorptive and has an average half life of 60 days in water. The application rate for Lorsban is 1 pound per acre. For a 30 gallon chemical tank along with the 2000 gallons of water in the center pivot piping system, then the concentration of Lorsban in the back-siphoned water from the irrigation system is approximately 8,000 mg/l.

In the Lorsban simulation a retardation factor (R_d) of 900 was used. See Appendix A for calculation of the retardation factor. For this retardation factor, the distance that the Lorsban would migrate in the aquifer is 1/900 of the distance that a conservative contaminant would migrate for the same aquifer properties and hydraulic conditions. Because Lorsban is so strongly adsorbed in the aquifer, a regional groundwater gradient in the vicinity of the well has negligible effect on the distribution of Lorsban in the aquifer. Similarly the length of the breakdown period has little effect on the distribution of the Lorsban in the aquifer. Due to this adsorption onto the porous media the Lorsban stays within roughly one foot of the well up to the end of the breakdown period (Figure 17). The concentration of the Lorsban in the aquifer at the well at the end of the breakdown period is approximately 330 mg/l.

When the irrigation well is restarted, the concentration again decreases rapidly at the initiation of pumping and most of the Lorsban is removed after a short period of pumping. The rate of decrease tapers off as the concentration of Lorsban becomes smaller, until, the amount of Lorsban

removed becomes very marginal as pumping continues (figures 18 and 19). The acceptable concentration level of Lorsban is ten micrograms per liter (ug/l). It requires a pumping time of 19 days for the Lorsban concentration in the water from the well to reach this limit. As stated earlier, although the Lorsban will not travel very far from the well due to adsorption onto the porous media, this property also serves to prolong the removal process since the chemical is slowly desorbing from the aquifer material during pumping. Table 3 summarizes the concentration data at the well during the restart period for the Lorsban simulation and also for the other two reactive chemical simulations.

4.2.2 Other Pesticides

The retardation factor of $R_d=900$ used in the Lorsban simulation was bracketed by simulations using R_d values of 200 and 4000 to determine what effect varying this parameter has on the cleanup process. The lower value of 200 is for a less adsorptive chemical such as Arbofuran, while the higher value $R_d=4000$ is an even more adsorptive chemical such as Atrazine. This higher value of R_d would also represent Lorsban for an aquifer material with a fractional coefficient of $f_{oc}=.04$ (see Appendix A). The concentration of the pesticide used in the backflow was again 8000 mg/l.

A comparison of the relative concentration distributions for each of the pesticides (figure 20) indicates that lower dissolved concentrations and less dispersion away from the well occurs as the retardation factor increases for the pesticide. More of the pesticide has been adsorbed onto the aquifer material for the larger retardation factor, hence, there is less

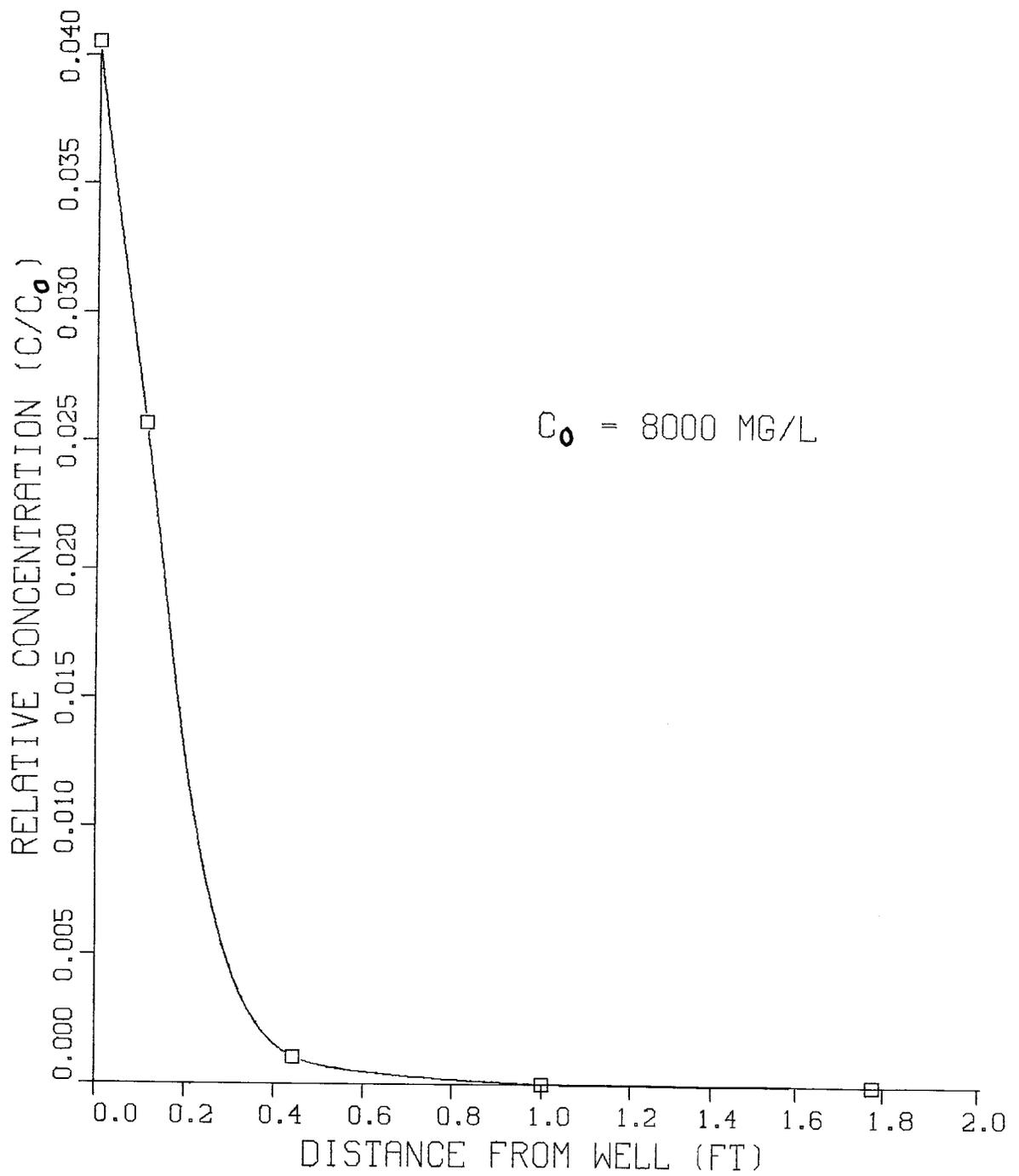


Figure 17--Relative concentration verses distance from well for the pesticide Lorsban at the end of the breakdown period.

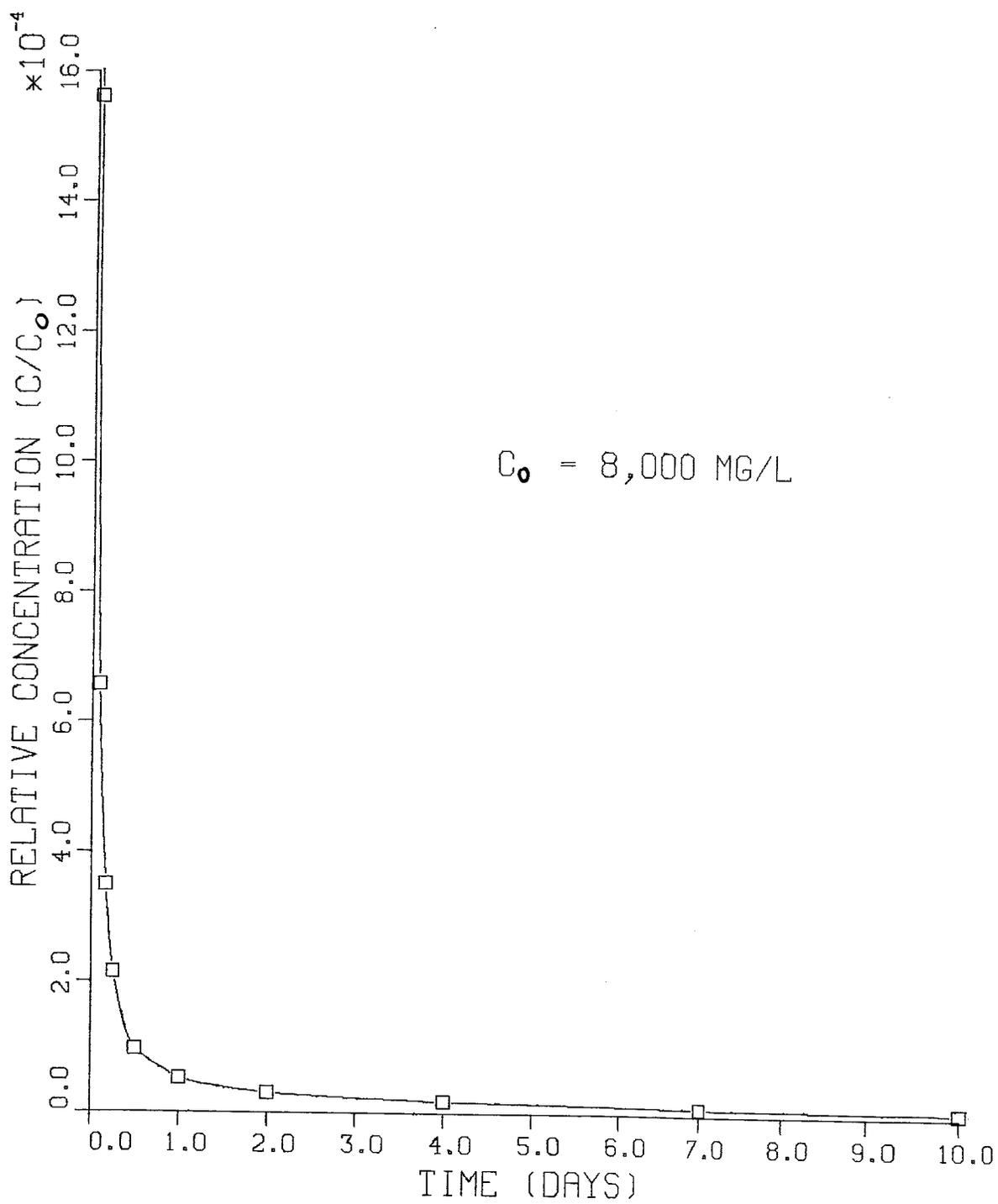


Figure 18--Relative concentration verses time at the well for the pesticide Lorsban during restart period.

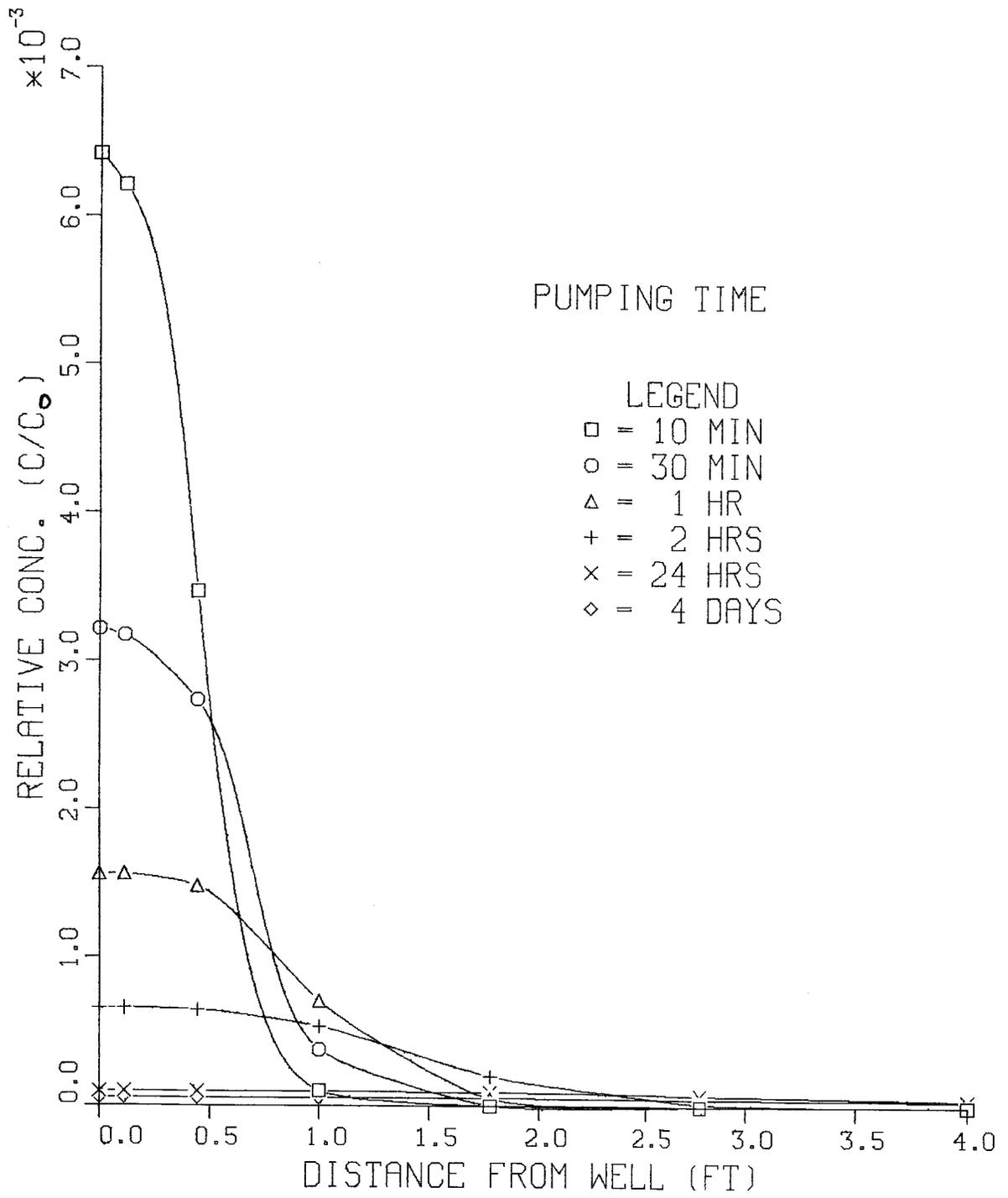


Figure 19--Relative concentration verses distance from the well for the pesticide Lorsban during restart period.

Table 3 Pesticide Concentrations at the well during restart period.

	(Arbofuran)	(Lorsban)	(Atrazine)
<u>Time</u>	<u>R_d=200</u>	<u>R_d=900</u>	<u>R_d=4000</u>
(hrs)	(mg/l)	(mg/l)	(mg/l)
0.0	585.274	328.000	176.947
0.2	83.920	51.340	34.752
0.5	40.494	26.730	16.352
1.0	19.342	12.497	7.844
2.0	7.942	5.253	3.355
4.0	4.147	2.792	1.808
6.0	2.526	1.722	1.124
12.0	1.122	0.778	0.514
(days)			
1.0	0.600	0.425	0.282
2.0	0.342	0.245	0.166
4.0	0.198	0.146	0.099
7.0	0.114	0.085	0.059
10.0	0.082	0.062	0.044
13.0	0.065	0.050	0.035
16.0	0.054	0.042	0.030
19.0	0.047	0.037	0.026

dissolved solute in the water and less pesticide available for dispersion and migration away from the well. This pattern continues after the pump on the well is restarted and the water and pesticide is pumped back out of the well, figure 21. The relative concentration at the well during the restart period, (figures 21, 22 and table 3) for the three pesticides exhibit the same initial rapid decrease in dissolved concentration immediately after restart. Again there is marginally less and less chemical removed as pumping continues due to the slow desorption of the pesticide from the aquifer material. Overall the retardation factor of the pesticide has little effect on the total pumping time required to remove the pesticide to safe levels. In all cases up to 20 days of pumping are required to remove the pesticide. In the case of Lorsban, the effect of hydrolysis (the breakdown of the chemical composition in the presence of water) would eliminate any residual pesticide amounts left in the aquifer at the end of pumping. The effects of hydrolysis was not included in any of the model simulations performed in this study.

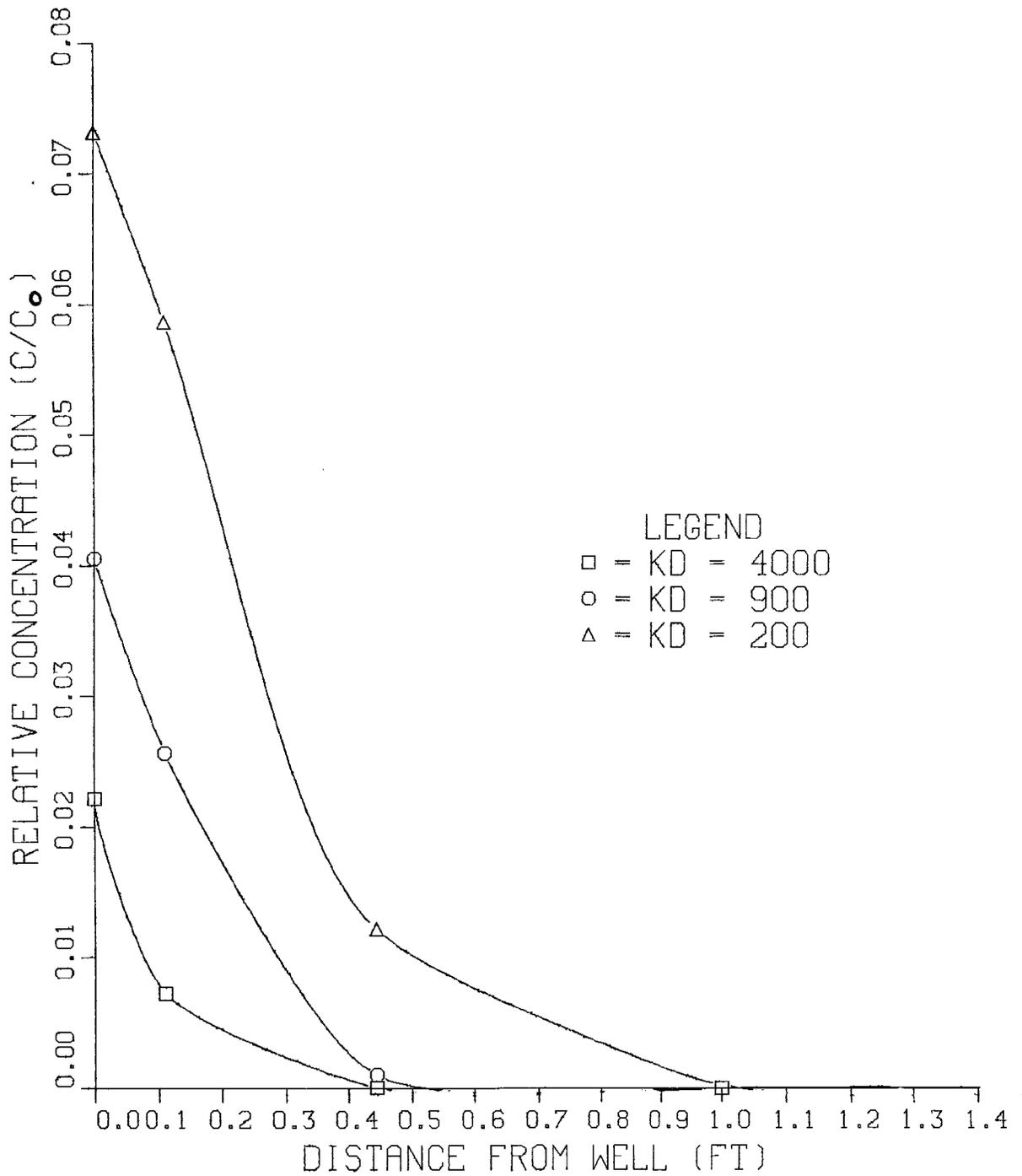


Figure 20--Comparison of relative concentrations verses distance from the well for the three pesticides at the end of the breakdown period.

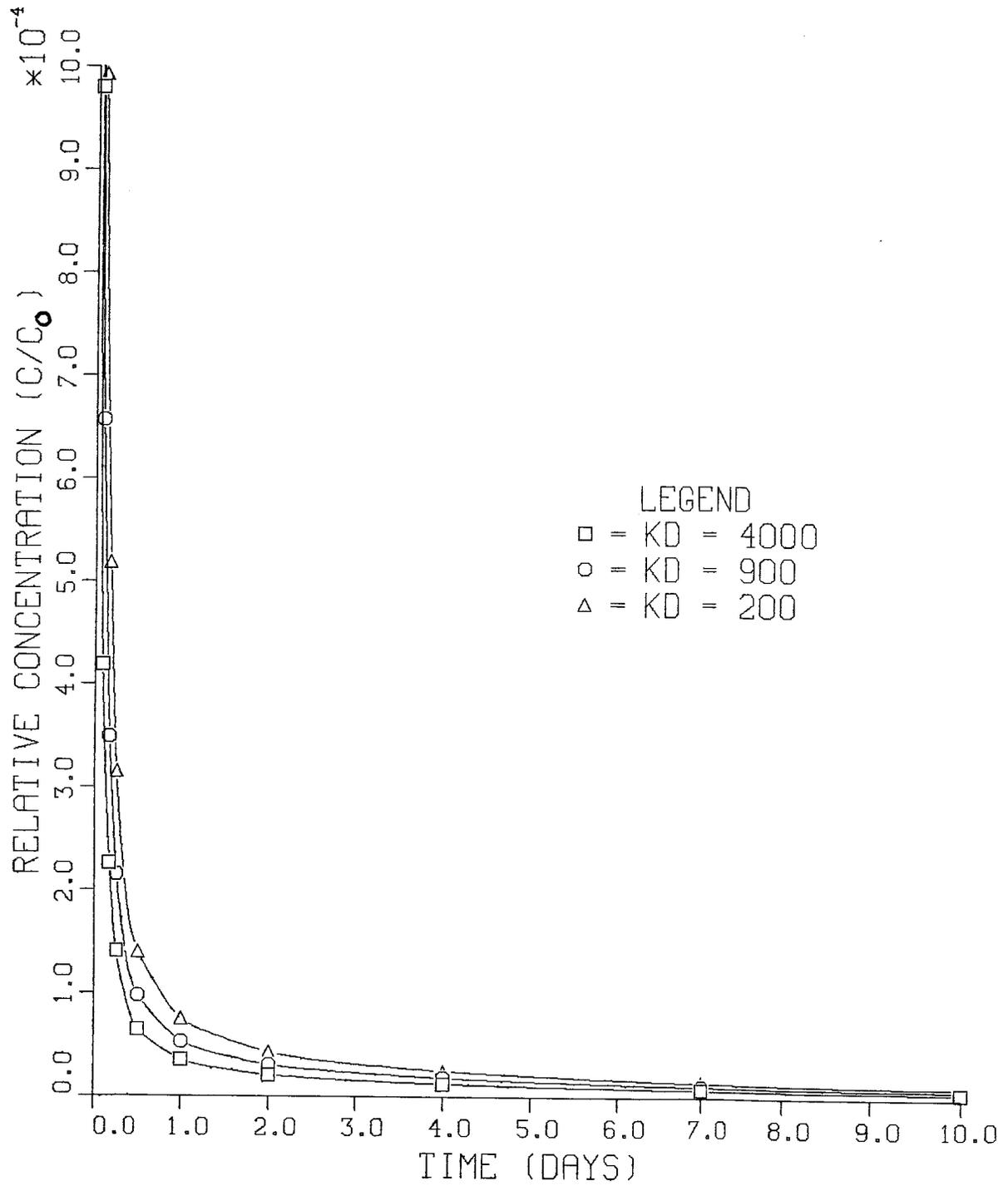


Figure 21--Comparison of relative concentrations versus time at the well for the three pesticides during the restart period.

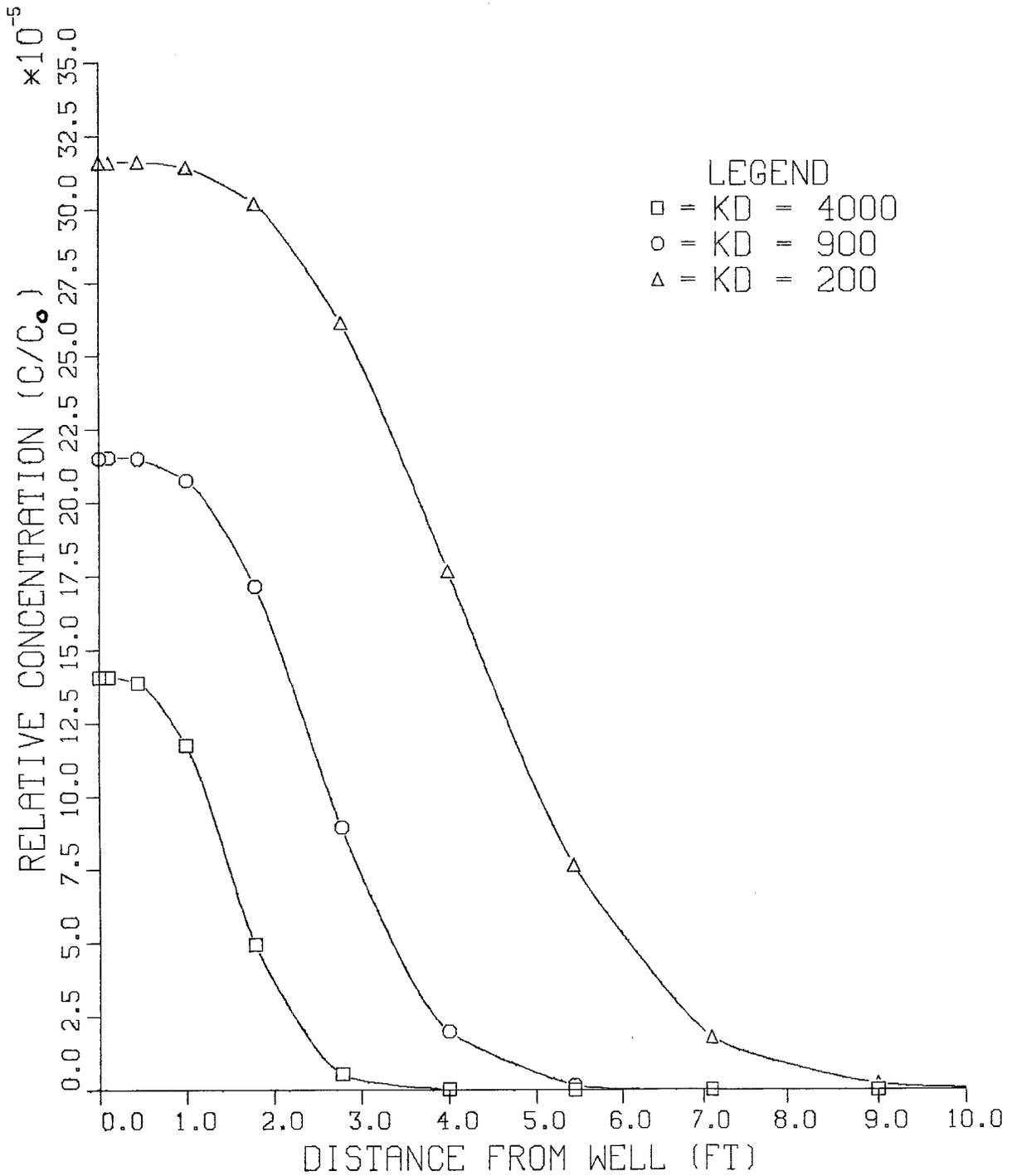


Figure 22--Comparison of relative concentrations verses distance from the well for the three pesticides during restart period.

CHAPTER V

SENSITIVITY ANALYSIS

The aquifer properties used in the model simulations in this study were obtained from the USGS High plains study and represent average values. To ascertain how the results would be affected by nonaverage values a sensitivity analysis was performed for a range of transmissivity, porosity and dispersivity values.

5.1 Transmissivity

For the high plains region of Nebraska an average value of 16,000 ft^2/day was used for transmissivity in all the previous simulations. This average value was bracketed with high and low values of 30,000 and 5,000 ft^2/day . Model simulations using the same back flow, breakdown, and restart pumping rates, volumes, and times were performed with these upper and lower values of transmissivity. There was no effect on the contaminant concentration distribution for varying transmissivity. This as would be expected since there is no change in the ground water velocities in the vicinity of the well because of continuity requirements. What was affected was the local head buildup due to back flow of the chemical and the shape of the drawdown cone due to restart of pumping.

For the lower value of transmissivity the local head buildup in the vicinity of the well due to back-flow of the chemicals and water in the irrigation piping system is more pronounced due to the decrease in hydraulic conductivity (figure 23). Similarly the drawdown cone developed during the

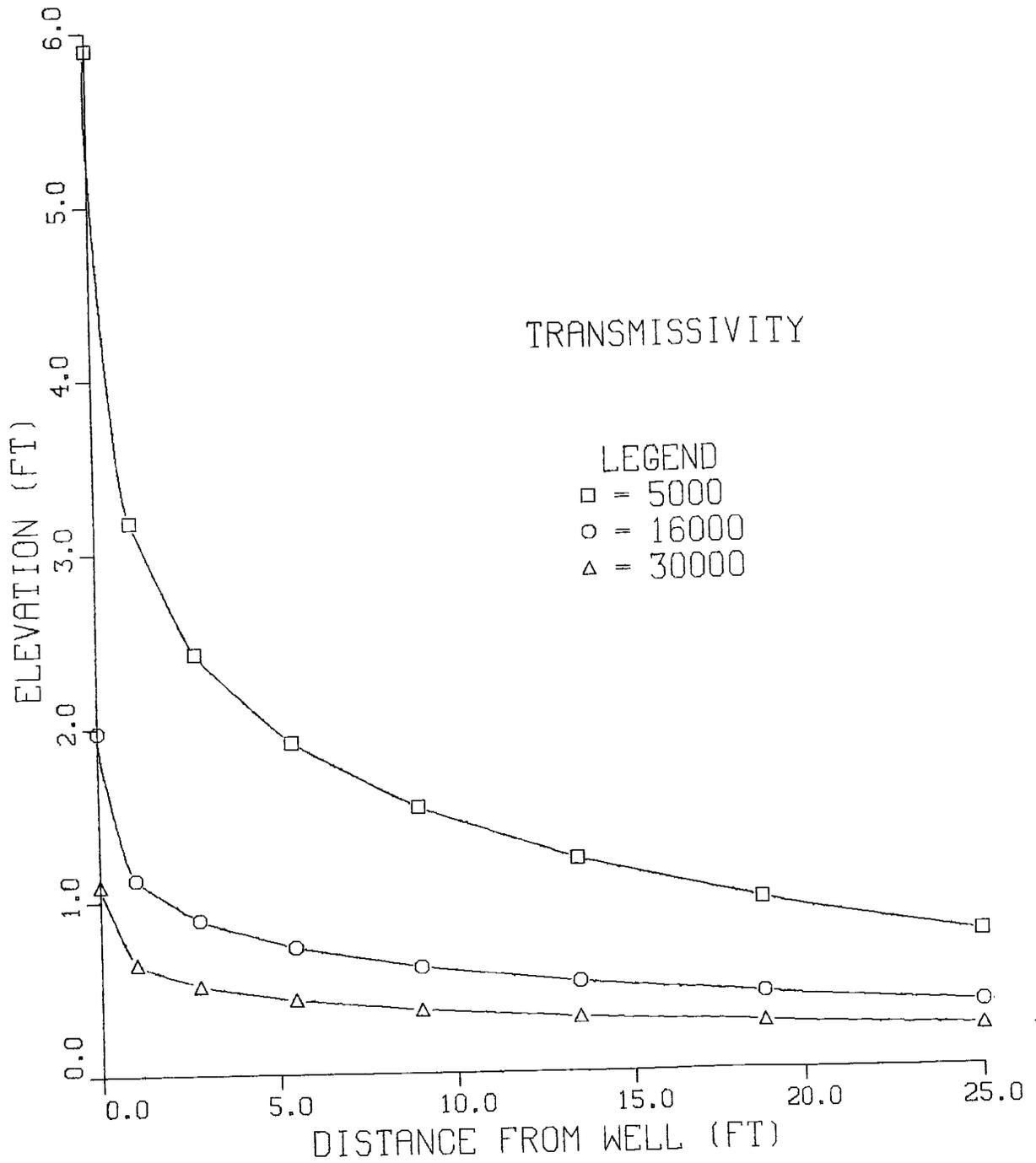


Figure 23--Local potentiometric head build up at well due to back flow of chemicals for varying transmissivity.

well restart period for the lower transmissivity is more pronounced (figure 24). The opposite situation is observed for the higher transmissivity cases. However in all cases the same quantity of water is back-siphoned down the well and the same quantity of water is removed for a given constant well discharge rate. As a result the groundwater velocities are unchanged in the vicinity of the well.

5.2 Porosity

Groundwater velocities were directly related to changes in porosity of the aquifer. A decrease in porosity, decreased the pore volume in the aquifer and the back-siphoned chemicals migrated farther out into the aquifer and contaminant concentrations in the aquifer were higher. An increase in porosity had the opposite effect. Changes in porosity had little effect on the pore volume of water that had to be removed however or on the pumping time required to restore the aquifer.

5.3 Dispersivity

A dispersivity of five feet was used in the previous simulations. This value was bracketed with high and low values of 50 and 2 feet. There is a great deal of discussion and dissension in the literature about dispersivity at this time (Cherry, Gilliam, 1982). It is generally agreed that dispersivity is a scale dependent parameter. The effect on the contaminant concentration distribution in the aquifer for varying dispersivity is shown on figures 25 and 26. With a decrease in dispersivity the chemical does not migrate as far away from the well and the concentrations at the well are

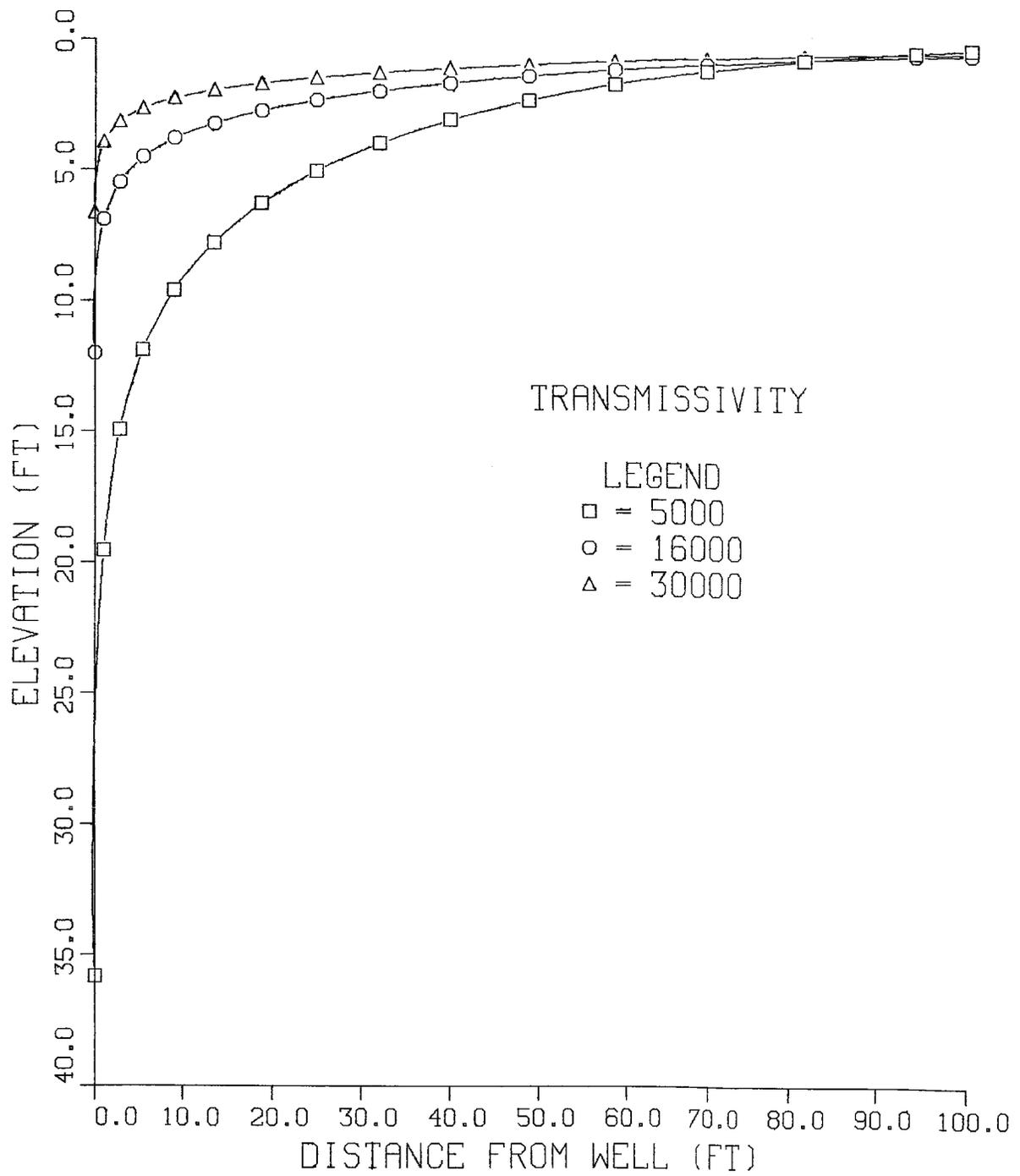


Figure 24--Drawdown cone at well due to pump restart for varying transmissivity.

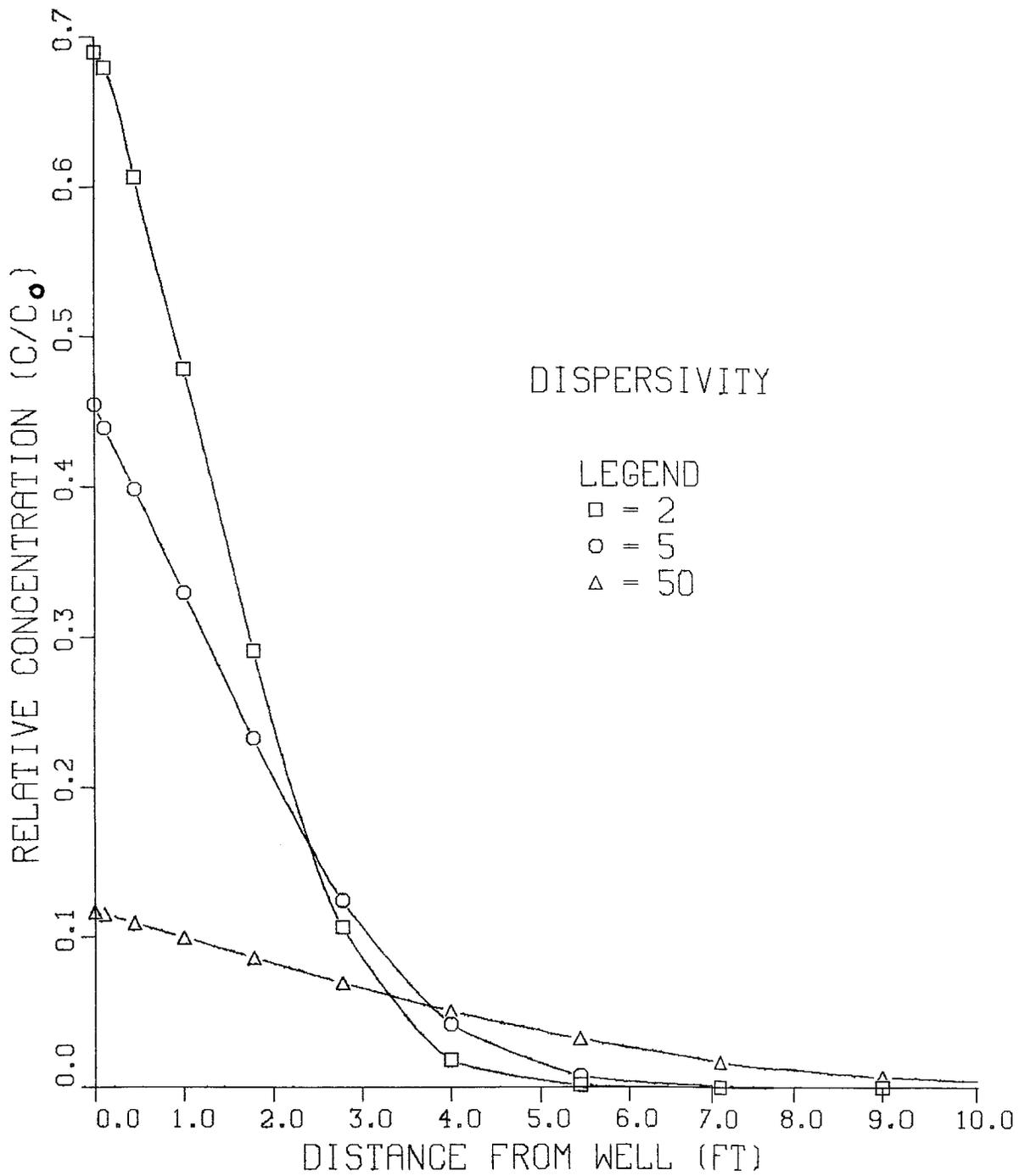


Figure 25--Relative concentration verses distance from the well for varying dispersivity for a conservative contaminant at end of breakdown period.

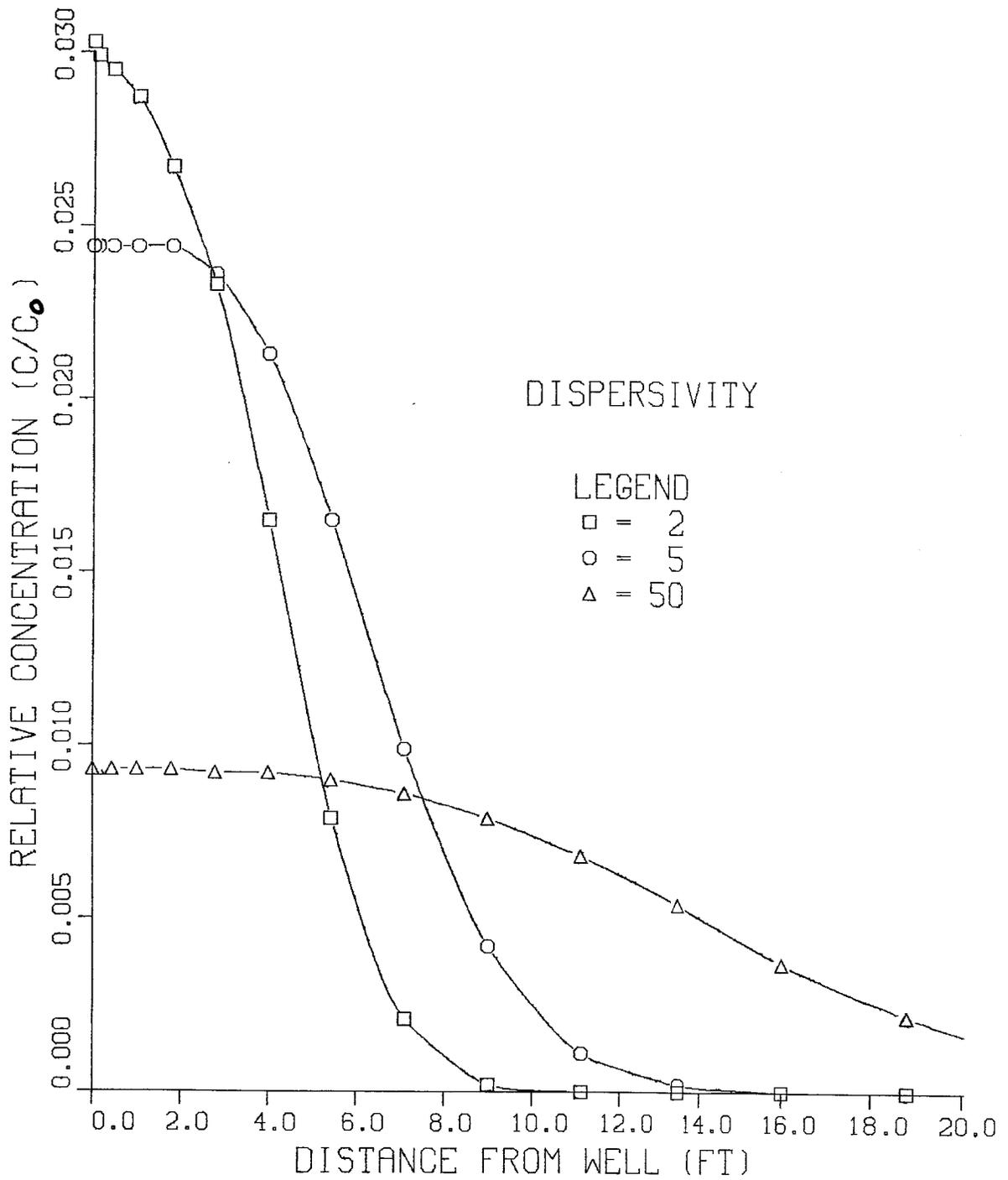


Figure 26--Relative concentration versus distance from the well for varying dispersivity for a conservative contaminant during restart period.

increased compared to those calculated using a larger dispersivity. When pumping is restarted, the contaminant front for the dispersivity of 2 feet remains much sharper where with the larger dispersivity of 50 feet the front is much more dispersed. The concentrations gradients are away from the well and the dispersivity term in the advection-dispersion equation results in some contaminant migration away from the well even during the pumping well restart period. A lower dispersivity value decreased slightly the pumping time required to remove the contaminant and a higher dispersivity value increased slightly the time.

CHAPTER VI
SUMMARY AND CONCLUSIONS

An actual field case has never been reported where back flow of chemicals has occurred during chemigation. By using the numerical groundwater transport model it has been possible to simulate the effect of various chemigation breakdown scenarios. The model was used to simulate the backflow of contaminants of varying chemical composition, quantities back-siphoned and varying aquifer properties and stresses. Each of these parameters was varied in sequence to identify the controlling factors in the contaminant removal process.

For a conservative chemical such as nitrate, restoration can be completed in the worst case scenario in about one and half days, if the well is restarted soon after breakdown occurs. For an extended breakdown period of one week it was still possible to remove the contaminant by restarting the irrigation well. In this case it may take as long as three days of pumping to lower the nitrate concentration to acceptable limit. The dispersion of the back-siphoned chemical out into the aquifer is function of dispersivity and groundwater velocities near the well. There is relatively little movement of water during the breakdown period, due to the small volume being back-siphoned. Groundwater velocities during this breakdown period are small and the effect of dispersion on the contaminant concentration in the aquifer are negligible.

The back-siphoning of a pesticide was also modeled. The main pesticide considered in this study was the pesticide Lorsban which is manufactured by Dow Chemical. Most pesticides are strongly adsorbed in the aquifer. This limits the extent of contaminant migration away from the well, but,

complicates the removal process due to the slow desorption of the contaminant which occurs after the well is restarted. In the case of Lorsban, nineteen days of pumping were required to return concentration to safe limits of 10 micrograms per liter. Pesticides which are not as strongly adsorbed as Lorsban will fall somewhere between Lorsban and nitrate in their ability to migrate away from the well and their difficulty in being removed from the aquifer. Contaminants which are more strongly adsorbed will remain closer to the well but their desorption from the aquifer material will occur more slowly.

For pesticides, their highly adsorptive characteristics precludes their migration away from the well, so that it is not imperative that the farmer achieve complete removal right away. In addition, the sixty day half life of Lorsban was not taken into consideration in the simulations. This factor would further reduce concentrations and pumping time of the Lorsban in the aquifer.

APPENDIX A
DISTRIBUTION COEFFICIENT

In the advection-dispersion equation (2) there is a retardation factor, R_d , which needs to be specified for reactive contaminants. Pesticides are examples of reactive contaminants in chemigation. For reactive contaminants, changes in concentration can occur because of the partitioning of the contaminant between the liquid and solid phases. The retardation factor is defined as

$$R_d = 1 + (\rho_b/\phi)K_d \quad (A1)$$

where

ρ_b = bulk mass density

ϕ = porosity, and

K_d = distribution coefficient

The amount of chemical that is adsorbed by the solid aquifer skeleton is commonly expressed solely as a function of the concentration of the contaminant in solution. In this case the Freundlich isotherm is used to simulate contaminant adsorption.

$$\bar{C} = K_d * C^\alpha \quad (A2)$$

where

\bar{C} = adsorbed concentration,

C = dissolved solute concentration,

K_d = distribution coefficient (dependent on contaminant species),

α = exponential coefficient (dependent on contaminant species).

For a linear isotherm $\alpha = 1$.

The distribution coefficient for organics chemical can be determined from laboratory tests using the relationship

$$K'_d = f_{oc} * K_{oc} \quad (A3)$$

where

f_{oc} = fraction of organic carbon in aquifer

K_{oc} = partition coefficient for a 100 percent organic
carbon sorbent

For the pesticide Lorsban, $K_{oc} = 6000 \text{ cm}^3/\text{gm}$ and the f_{oc} ranges from 0.01 to 0.04. This yields K'_d values of from $60 \text{ cm}^3/\text{gm}$ to $240 \text{ cm}^3/\text{gm}$.

These values of K'_d are in units consistent with the units commonly reported in the literature, where

$$K'_d = \frac{\text{mass of adsorbed solute}}{\text{mass of solute in solution}}$$

The K_d value used in the model is defined as

$$K_d = \frac{\text{mass of adsorbed solute/mass of solids}}{\text{mass of solute in solution/volume of solution}}$$

If the adsorbed contaminant concentration is expressed in the same units as dissolved contaminant concentration (mass of contaminant/volume of solution) then

$$K_d = \frac{\text{volume of solution}}{\text{mass of solids}}$$

The relationship between K_d and K'_d is then

$$K_d = \frac{K'_d * \rho_b}{\phi}$$

For the pesticide Lorsban $R_d = 900$ for $f_{oc} = 0.01$ and $R_d = 4,000$ for $f_{oc} = 0.04$.

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