Simulation of Ground-Water Flow and Application to the Design of a Contaminant Removal System, Loring Air Force Base, Maine

By J. Jeffrey Starn

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BRUCE BABBITT, *Secretary*

U.S. GEOLOGICAL SURVEY
Gordon P. Eaton, Director

For additional information write to:

District Chief
U.S. Geological Survey
450 Main Street, Room 525
Hartford, CT 06103

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CONVERSION FACTORS AND VERTICAL DATUM

Multiply	By	To obtain
mile (mi)	1.609	kilometer
square mile (mi^2)	2.590	square kilometer
inch (in.)	25.4	millimeter
inch per year (in/yr)	25.4	millimeter per year
foot (ft)	0.3048	meter
foot per day (ft/d)	0.3048	meter per day
foot squared per day (ft^2/d)	0.09290	meter squared per day
cubic foot per day (ft^3/d)	0.02832	cubic meter per day
cubic foot per second (ft^3/s)	0.02832	cubic meter per second
gallon (gal)	3.785	liter
gallon per minute (gal/min)	0.06308	liter per second

Sea level: In this report "sea level" refers to the National Geodetic Vertical Datum of 1929 (NGVD of 1929)—a geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called "Sea Level Datum of 1929."

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ABSTRACT

The fractured-bedrock aquifer underlying the former Fire Training Area at Loring Air Force Base, Maine, has been contaminated with petroleum products as a result of fire training activities. A numerical model of the ground-water-flow system near the Fire Training Area was developed to provide information for the design and operation of a contaminant removal system. The goals of the simulation modeling were to (1) determine the maximum pumping rate that could be sustained, given the constraint that water levels not rise above a specified altitude, and (2) determine the effect of seasonal variation in recharge on the ability of a transient pumping scenario to capture contaminants. A steady-state simulation model of ground-water flow was used to determine the optimal pumping rate at the site. The optimal pumping rate was 8,570 ft^3/d (44 gal/min). Monthly recharge rates were estimated for use in a transient simulation model. During a typical year, most recharge probably occurs during two periods—one during snowmelt in early spring and another, possibly less significant period, during the late fall. The transient response of the water table to 8.5 inches of recharge in April, 2 inches of recharge in October, and 0.25 inches per month for each remaining month was simulated. Fluctuations in ground-water levels caused by simulated seasonal variation of recharge would have minimal effect on the operation of the contaminant removal system because the system is not pumped when recharge is lowest, ground-water velocities are lowest, and ground-water flow past the trench is minimal.

INTRODUCTION

The fractured-bedrock aquifer underlying the former Fire Training Area (FTA) at Loring Air Force Base (LAFB) in northeastern Maine (fig. 1) has been contaminated with petroleum products as a result of fire training activities. From 1952 to 1981, 150,000 to 300,000 gal of petroleum products, such as jet fuel, waste oil, and flammable solvents, were placed in an unlined pit at the FTA and burned. It is estimated that approximately 50 percent of the liquid was not consumed and infiltrated downward to contaminate the aquifer (ABB Environmental Services, Inc., 1996). Contaminants in the aquifer are present as nonaqueous phase liquids (LNAPLs) that are less dense than water and in a dissolved, aqueous phase. As part of the Installation Restoration Program, the Air Force is conducting a pilot study to remove LNAPLs from the aquifer underlying the FTA. The LNAPL removal system consists of a blast-fractured recovery trench (150-ft long by 10-ft wide by 70-ft deep; perpendicular to the direction of ground-water flow); three extraction wells in the trench; skimmer and submersible pumps in the extraction wells; and a water-treatment system consisting of an oil-water separator, air stripper, and filters. Treated ground water is discharged to the surface downgradient from the FTA and the recovery trench. The objective of the pilot study is to remove as much of the LNAPL as possible.

During 1995-96, the U.S. Geological Survey (USGS), as part of a study done with in cooperation with the U.S. Air Force (Contract number FQMSR 96-036N), conducted geophysical studies of the fractured-bedrock aquifer (Lane and others, 1996) and, using existing hydrologic and climatic data, developed a numerical model to simulate ground-water flow near

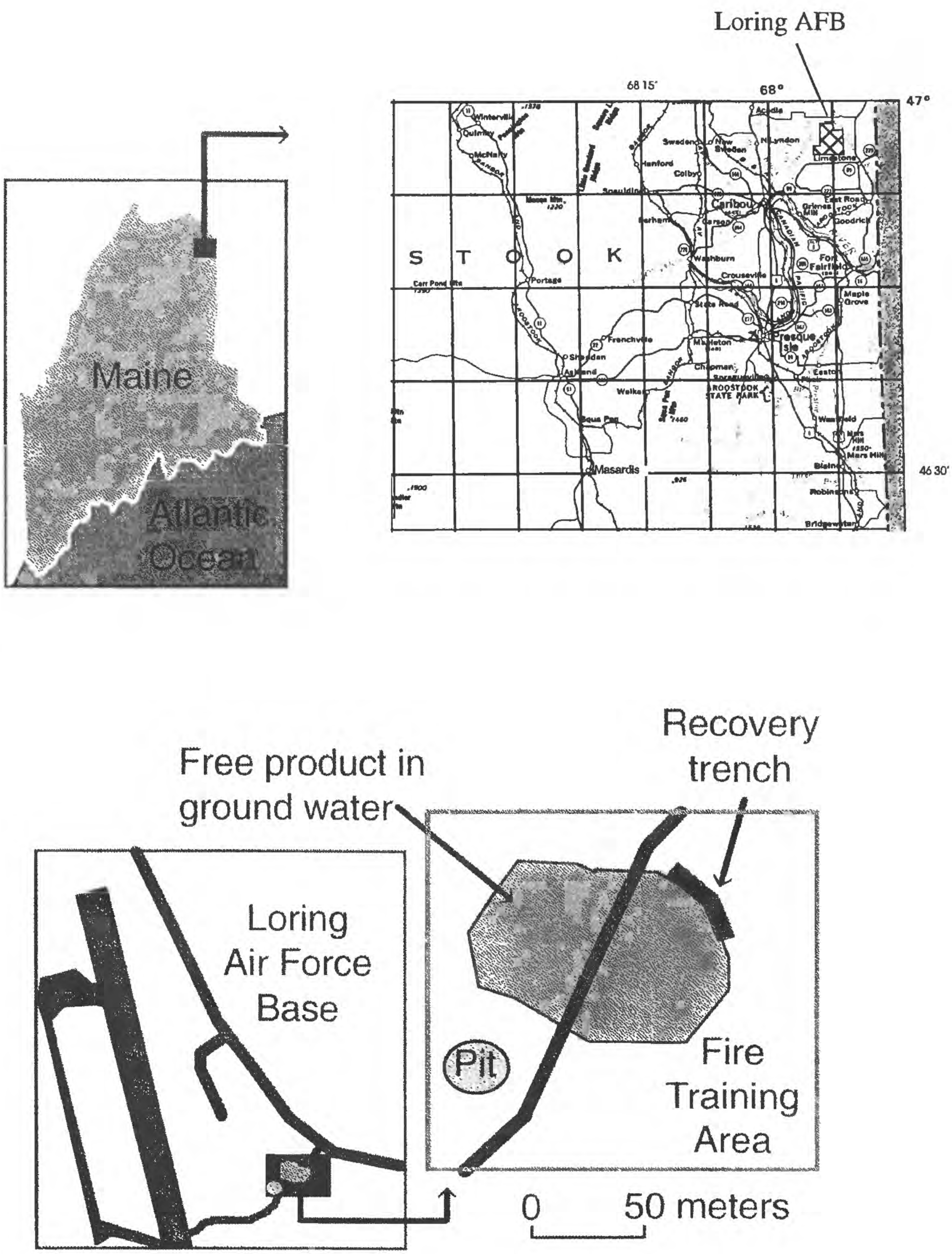


Figure 1. Location of Loring Air Force Base, Limestone, Maine. (Modified from Lane and others, 1996, fig. 1.)

the FTA. A simulation model can be used to (1) predict the response of an idealized ground-water flow system and relate that response to the aquifer at the FTA, (2) provide a starting point from which changes to the idealized system can be evaluated, and (3) assess the underlying factors that affect the design and operation of the removal system.

This report describes the simulation of ground-water flow in the FTA at LAFB and the application of simulation modeling to the design and operation of the contaminant removal system. This simulation will provide the conceptual framework to determine the optimal rate of ground-water pumping from the recovery trench, given certain constraints, and the effect of seasonal variation of recharge on ground-water flow. The simulation model is not "calibrated" and is not intended to be used to accurately predict hydraulic heads. Time and budget constraints, the difficulty of predicting ground-water-flow paths in fractured-bedrock aquifers, and the complex geology of the area near the LAFB are several of the reasons why the model was not calibrated.

The author acknowledges the assistance of the U.S. Environmental Protection Agency and the Maine Department of Environmental Protection.

SIMULATION MODELING OF GROUND-WATER FLOW

Simulation modeling was used to answer two specific questions about the design and operation of the contaminant removal system. The first question was, if all treated ground water was returned to the aquifer through injection wells upgradient from the recovery trench, what would be the maximum pumping rate that could be sustained, given the constraint that water levels not rise above a specified altitude? The reason for this concern is that if water levels rise into the unsaturated deposits above the aquifer, the water might interfere with the bioventing system already in place at the FTA. To answer the first question, steady-state ground-water flow was simulated.

The second question was, if treated ground water was not injected into the aquifer, but instead was discharged to surface water during months when it was not likely to freeze (May to October), and no pumping occurred during the cold months of the year, would LNAPL flow past the recovery trench? To answer the second question, a yearly cycle of transient ground-water flow was simulated to evaluate the effects of seasonal fluctuations in recharge on ground-water flow at the FTA.

Ground-water-flow system

The contaminated part of the aquifer is composed of a sequence of thinly interlayered limestone and calcareous phyllitic slate (Lane and others, 1996; ABB Environmental Services, Inc., 1996). The dominant unconsolidated material overlying the bedrock is ablation till, which was deposited as the glaciers melted and retreated. The water table is in the bedrock aquifer (Lane and others, 1996). Ground water and LNAPL flow through the bedrock aquifer in fractures; thus, the rate of contaminant transport is controlled by the width, orientation, roughness, connectivity, and spacing of the fractures.

In the absence of site-specific data, hydrologic boundaries to the ground-water-flow system were approximated by using selected physical characteristics from 1:24,000-scale topographic maps. Perennial streams receive ground-water discharge throughout the year and can be used to approximate discharge boundaries to the shallow ground-water-flow system. Thus, shallow ground-water flow at LAFB probably is bounded by Greenlaw Brook on the west and Butterfield Brook on the east (fig. 2). The altitude of land surface provides the driving mechanism for shallow, unconfined ground-water flow (Domenico and Schwartz, 1990, p. 243); therefore, ground water moves away from topographically high areas toward topographically low areas. Shallow ground-water flow probably is bounded by topographic divides north and south of LAFB that connect Greenlaw Brook and Butterfield Brook (fig. 2).

Aquifer properties

Hydraulic properties were estimated for four zones of the aquifer system on the basis of aquifer hydraulic testing (fig. 3; J.S. Williams, U.S. Geological Survey, written commun., 1996). Hydraulic conductivity ranged from 3.75 ft/d in the lower bedrock (zone 2) to 750 ft/d in the recovery trench (zone 4). The upper bedrock layer is equal in thickness to the depth of the trench, which is 80 ft. A lower bedrock layer is used to account for the fact that fracture width decreases with depth; the hydraulic conductivity of this layer is 3.75 ft/d (zone 2), and the layer is assumed to be 80-ft thick in the simulations. Geophysical data indicated a zone of rock surrounding the blast-fractured recovery trench (zone 3) that has a hydraulic conductivity estimated to be 75 ft/d. The aquifer was assumed to be homogeneous except for the area near the trench. The porosity of the bedrock was determined by borehole-radar techniques (Lane and others, 1996); the porosity was 0.02 in zones 1 and 2, 0.03 in zone 3 and 0.10 in zone 4.

Seasonal variation of recharge

Monthly estimates of recharge were developed to use in the simulation of seasonally varying recharge. Two independent methods were used to estimate the annual distribution of recharge, which is compared to streamflow records and ground-water hydrographs to demonstrate the reasonableness of the approach.

The first of the two methods used to estimate monthly recharge rates is a monthly water-balance model (Alley, 1984). This method is based on the potential monthly evapotranspiration, computed using the method of Thornthwaite (Veihmeyer, 1964, p. 11-26). One year (1993) was chosen for this analysis because the recharge, streamflow, and hydrograph data represent a consistent set of actual events, the characteristics of which would be lost if monthly averages were used. Mean monthly temperature and precipitation were computed from data collected at LAFB in 1993 (Susan Soloyanis, Mitretek Systems, written commun., 1996). Precipitation and streamflow records indicate that hydrologic conditions in the LAFB area in 1993 were nearly average. At the airport in nearby Caribou, Maine, average annual precipitation during 1951-95 was 37 in/yr; in 1993 it was 40 in. At the USGS streamflow-gaging station on the Aroostook River near Masardis (station 01015800), average streamflow during 1957-94 was 23 in. over the area of the basin; during 1993, streamflow was 25 in. over the area of the basin.

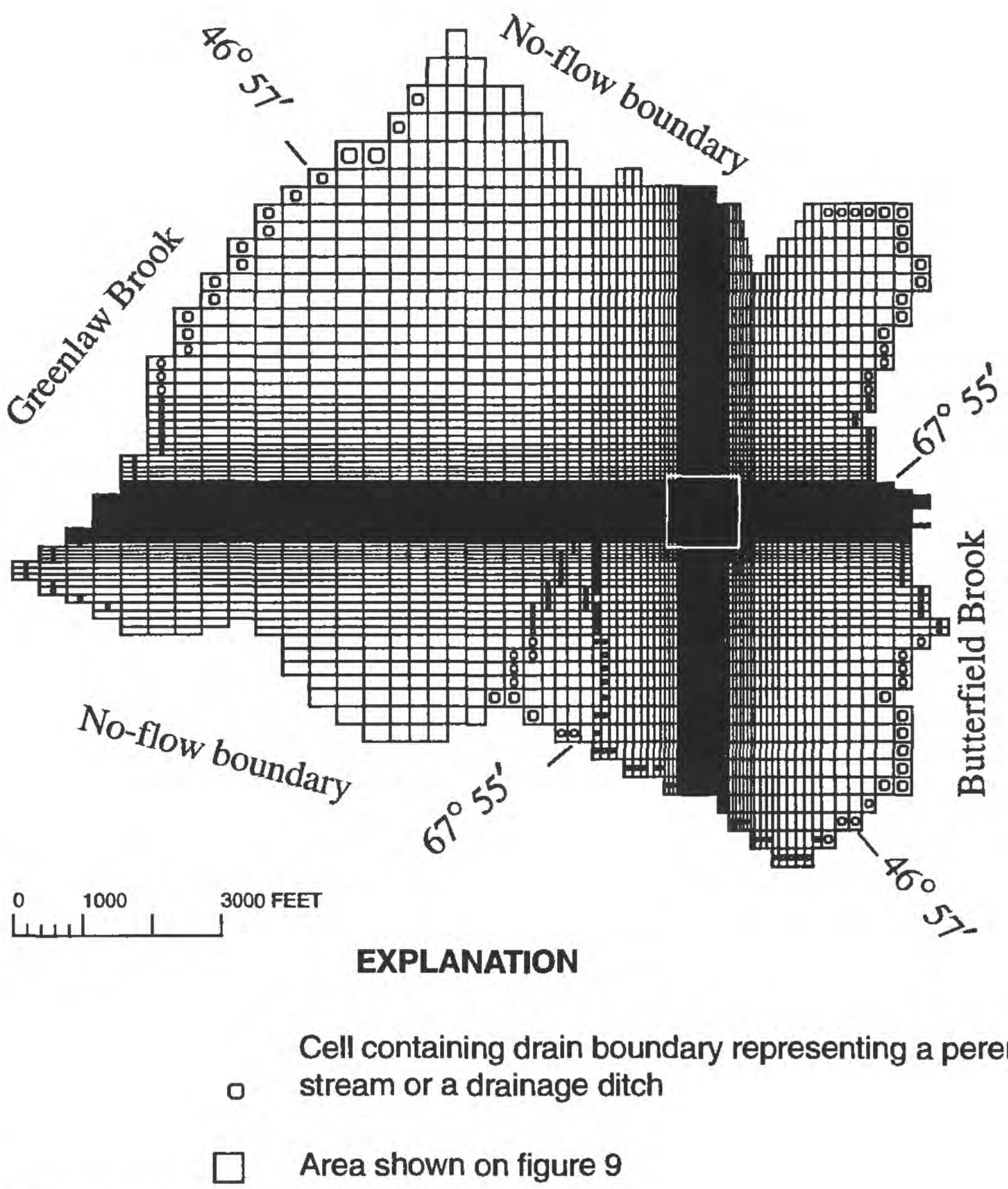


Figure 2. Model grid and boundary conditions.

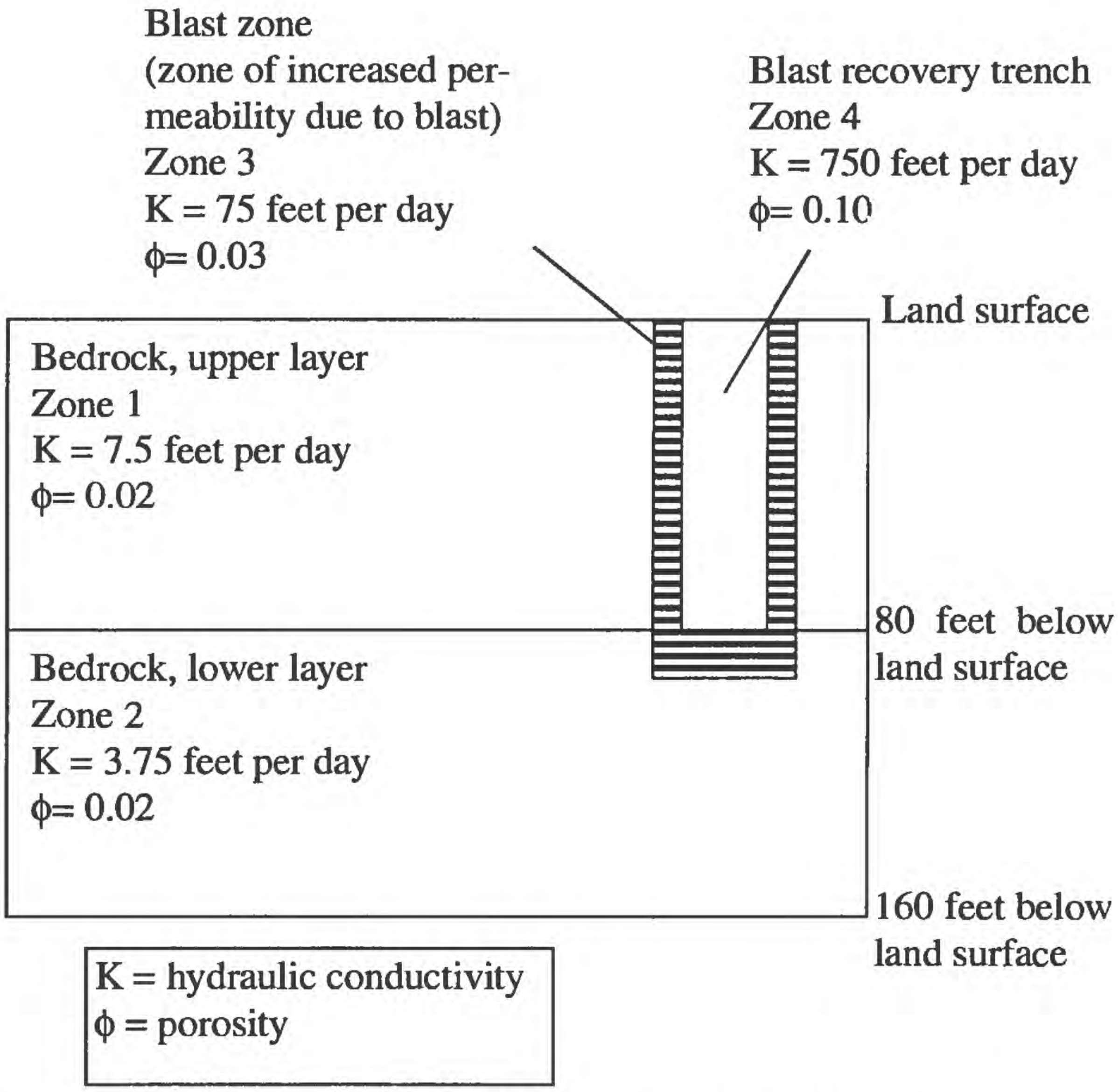


Figure 3. Aquifer properties near the Fire Training Area, Loring Air Force Base, Maine.

A modified "T model" (Alley, 1984) was calibrated to streamflow data for the Aroostook River near Masardis. The model was modified so that 80 percent of the precipitation was stored as snow during months when the mean monthly temperature was below freezing. This snow was assumed to melt over a 2-month period in the spring (April to May) when the mean monthly temperature rose above freezing. April and May were chosen as the period of melting because runoff exceeded precipitation and there clearly was an additional source of water to the Aroostook River during these two months. The percentage of snow accumulation, the maximum soil moisture content of the basin, and a factor related to the physical conditions of the basin (Alley, 1984) were adjusted during model calibration. Simulated annual runoff was 5 percent higher than actual streamflow, but even so, the model reproduced monthly runoff reasonably well.

The monthly water-balance model resulted in almost 8 in. of ground-water recharge in April when the snow melted, 1 in. of recharge in October when evapotranspiration demands were low and the ground was not yet frozen, and less than 1 in. of recharge in January, February, and March (fig. 4). The model underestimates total annual recharge because it uses mean monthly input values, even though any given month can include some days of recharge and some days of no recharge. The concentration of recharge in April is somewhat an artifact of the snow accumulation factor; other choices of model parameters that simulate runoff equally well did not produce such a large recharge event in April. A large recharge event in April will produce higher hydraulic gradients in the simulation model and a greater likelihood that LNAPL will flow past the trench than if recharge were spread over a longer period of time. Thus, this distribution of recharge produces a more conservative answer to the question being addressed by the simulation model.

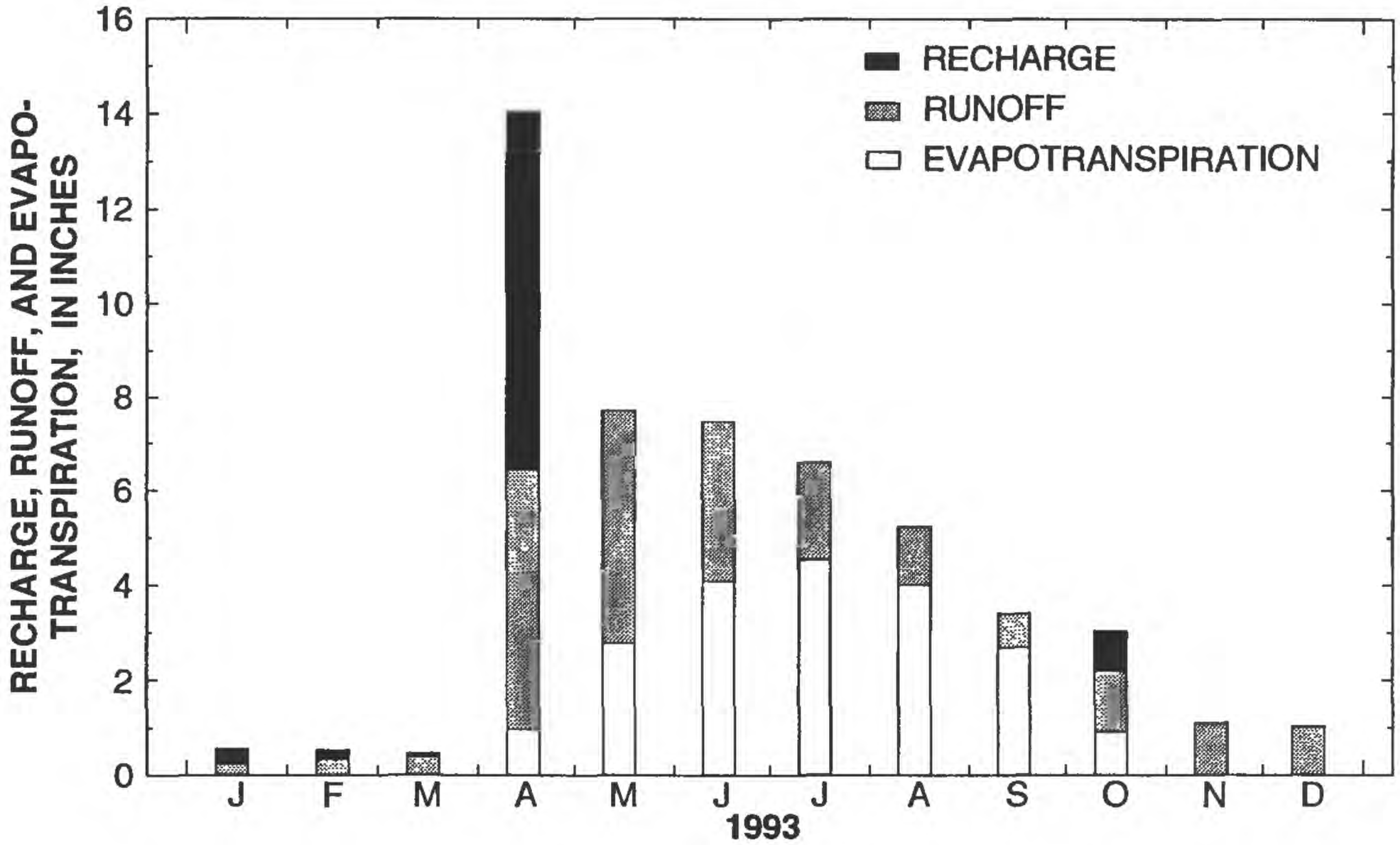


Figure 4. Monthly recharge rates determined using the water-balance method, Loring Air Force Base, Maine.

The second method used to estimate monthly recharge uses streamflow records to estimate ground-water discharge. An automated procedure (Rutledge, 1993) was used to separate ground-water recharge from the total streamflow. This procedure calculates the recession index from a large number of streamflow recessions throughout the period of record. The recession index is then used to project water-level recession after a peak on the hydrograph as if there were no subsequent recharge; the difference between two adjacent projections is equal to ground-water recharge. There can be errors in estimates of recharge during summer months because of the effects of ground-water evapotranspiration.

The streamflow-separation technique resulted in roughly the same pattern of recharge as the monthly water-balance method (fig. 5). The technique was applied to streamflow data from the USGS streamflow-gaging station on the Aroostook River near Masardis (station 01015800). Streamflow records were analyzed for each calendar year from 1958 to 1995. The results of the analysis were grouped by month to depict the median monthly variation of ground-water recharge (fig. 5). The median monthly recharge was highest in April (7 in.) and much lower in all other months. There was a period of slightly elevated recharge in October and November (2 in.). A large variation was evident in the computed recharge for May that may be related to climatic variations from year to year.

The preceding analyses are limited by the different areas the data represent and by the characteristics of the Aroostook River. The Aroostook River drainage basin above the gage covers 892 mi^2, and the precipitation and temperature data are from a single point at LAFB; therefore, the local climatic data are not necessarily consistent with the basinwide streamflow conditions. Also, the streamflow records have several deficiencies that limit the analysis. Much of the streamflow record was estimated because of ice effects, and there is some upstream regulation of the stream for power generation. In addition, Rutledge (1993) suggests that the streamflow-separation method be applied to basins 500 mi^2 and smaller, although he notes that some investigators have used the method in basins up to 2,000 mi^2.

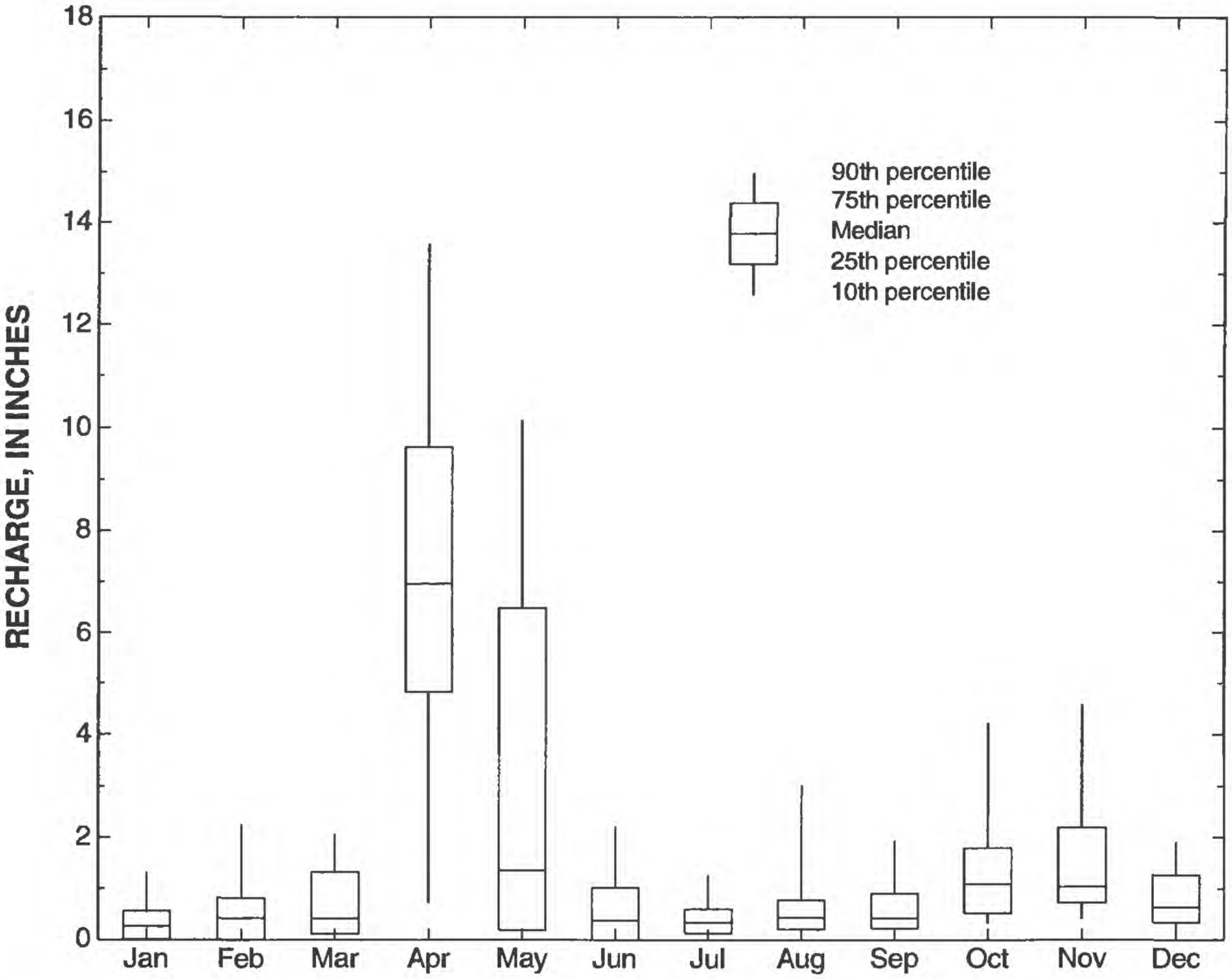


Figure 5. Monthly recharge rates determined using the streamflow-separation method, Aroostook River near Masardis, Maine, 1958-95.

The yearly distribution of recharge was compared to streamflow records and ground-water hydrographs to see if it reasonably approximated the hydrologic system. Streamflow on the Aroostook River near Masardis clearly showed an abrupt rise in April 1993 and a gradual, less prominent rise in the fall (fig. 6). Nearby ground-water levels in glacial till at Presque Isle (USGS observation well 464259067572901) show the same seasonal pattern, especially in 1990, 1992, and 1993 (fig. 7). The patterns in 1991 and 1994 were similar to those in the other 3 years, except that the October recharge rise was almost equal to the April rise.

The primary conclusion from this analysis is that during a typical year most recharge probably occurs during two periods—one during snowmelt in early spring and another, possibly less significant period, during the late fall. Based on this conclusion, an idealized distribution of annual recharge was developed for use in the simulation model (fig. 8). A small amount of recharge was added in months between major recharge periods because some recharge is likely during individual precipitation events not considered in the monthly analysis (F.P. Lyford, U.S. Geological Survey, oral commun., 1996). Total simulated annual recharge is 13 in/yr.

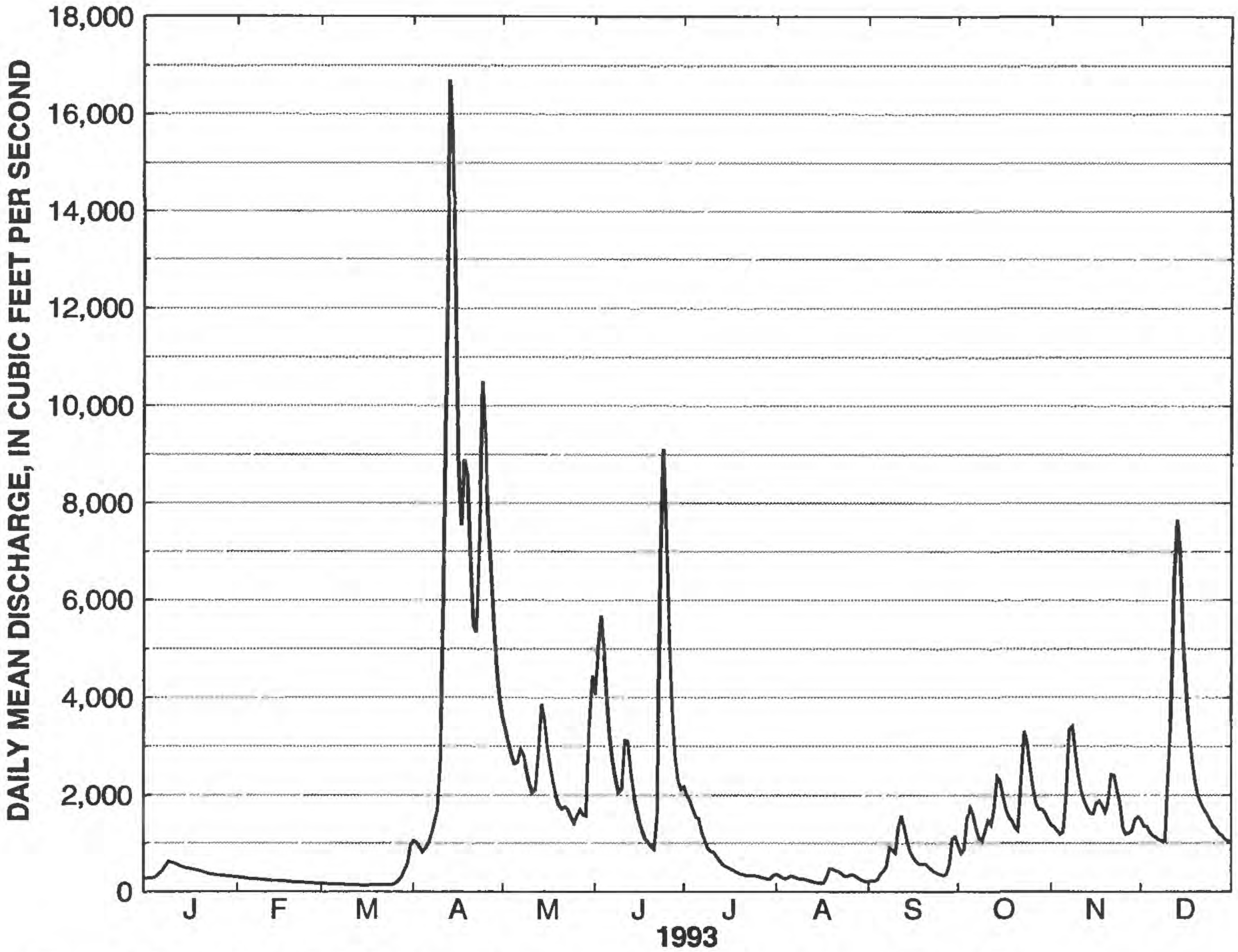


Figure 6. Streamflow at U.S. Geological Survey streamflow-gaging station, Aroostook River near Masardis, Maine (station 01015800), 1993.

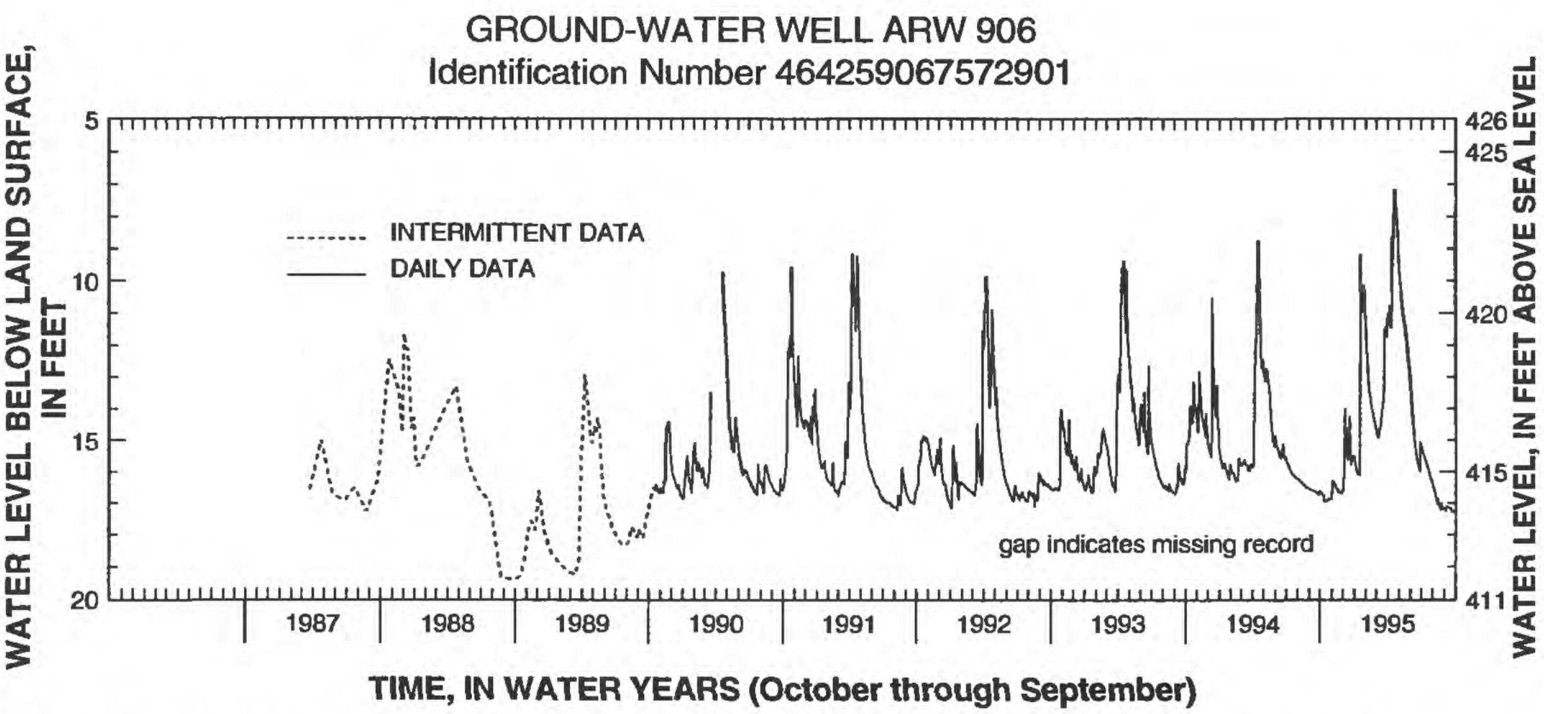


Figure 7. Ground-water levels near Presque Isle, Maine. (From Nielsen and others, 1996, p. 168.)

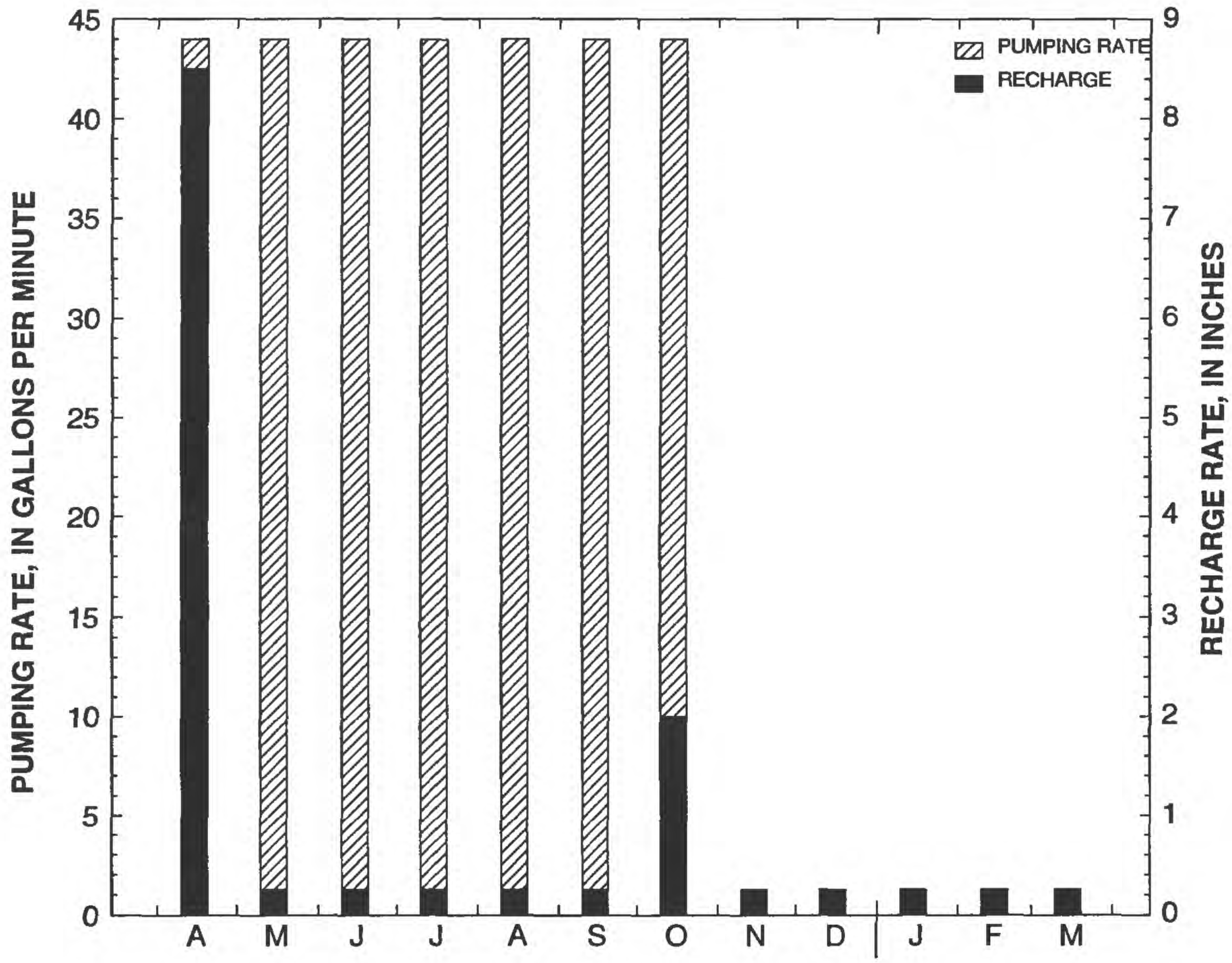
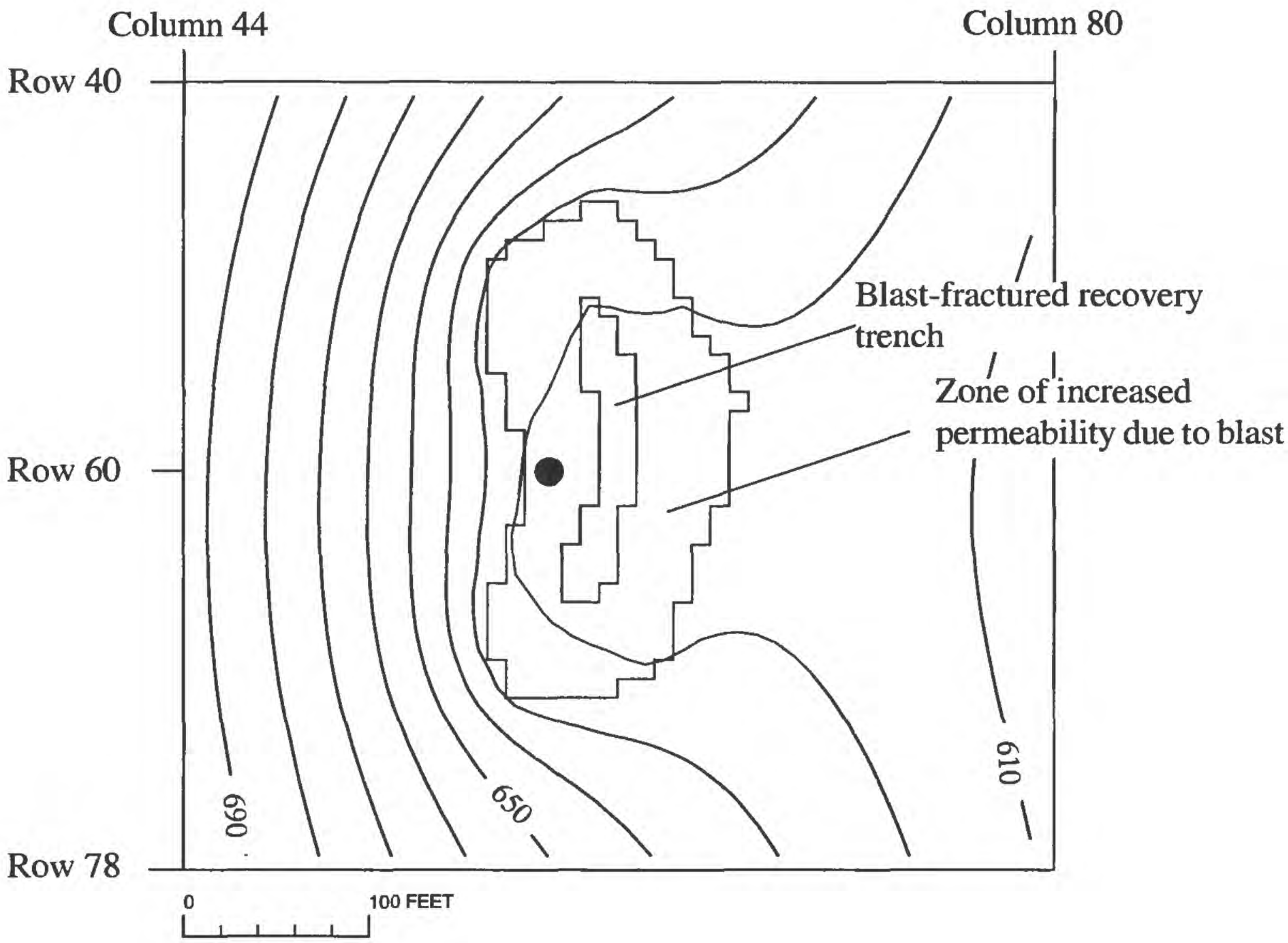


Figure 8. Pumping and recharge rates used in the Fire Training Area transient model.

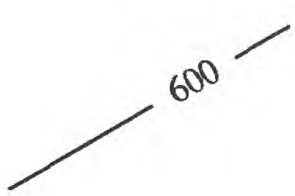
Description of simulation model

Ground-water flow was simulated using the computer code MODFLOW (McDonald and Harbaugh, 1988). MODFLOW is a three-dimensional, block-centered, finite-difference computer code that solves the ground-water-flow equation. In this study, the aquifer system was conceptualized as an isotropic, heterogeneous flow system that consists of two layers. The upper layer was simulated as a water-table layer, and the lower layer was simulated as a confined layer with constant transmissivity. Particle tracking was done using MODPATH (Pollock, 1994), which allows individual water particles to be tracked from their point of entry at the water table through a steady-state or transient flow field.

Although the area of interest for this study was near the FTA (fig. 9), the simulated area extends to the physical boundaries of the flow system (fig. 2) so that ground-water flow into the FTA would be realistic in a regional framework. The model grid is composed of variably spaced cells that range from 10 ft on each side near the FTA to 400 ft on each side far from the FTA (fig. 2). The grid is 117 rows by 109 columns. The active area of the model covers 3.3 mi^2. The grid was rotated to N50°E so that it was aligned with the principal direction of fracture orientation, as determined using square-array resistivity techniques (Lane and others, 1996).



EXPLANATION



WATER-TABLE CONTOUR-- Shows altitude of simulated water table. Contour interval is 10 feet. Datum is sea level.

● Location of hydrograph shown on figure 10.

Figure 9. Area of model grid showing steady-state simulated water-table altitudes and blast-fractured recovery trench, Loring Air Force Base, Maine (shown in inset on fig. 1).

The hydrologic boundaries to the north and south were simulated as no-flow boundaries. Greenlaw and Butterfield Brooks were simulated using the DRAIN package in MODFLOW (fig. 2). The DRAIN package allows water to flow from the aquifer to the drain cells but not in the reverse direction. The conductance of the drains was set high enough that the brooks were essentially constant heads when the water table was above the drain elevation. This feature allowed the simulation model to determine at which point the headwaters of the streams should become perennial streams.

The water-table boundary condition was simulated using the RECHARGE package in MODFLOW. An initial rate of 10 in/yr was assumed as the areally distributed rate of recharge. This value was adjusted upward to 13 in/yr so that the simulation model more closely approximated water levels at the FTA. This adjustment does not constitute a "calibration" of the model, but merely makes the steady-state water levels more consistent with field data near the FTA; the rate of recharge elsewhere in the modeled area might be different. Other investigators used a rate of 12 in/yr for a preliminary estimate for recharge at LAFB, and 8.7 in/yr was used in a Base-wide model that was calibrated to water levels measured in April 1995 (URS, Inc., written commun., 1996).

APPLICATION OF THE SIMULATION MODEL TO THE DESIGN OF A CONTAMINANT REMOVAL SYSTEM

The ground-water-flow simulation model was used to address questions about the design of the LNAPL removal system. The optimal pumping rate was determined using the method outlined below, and typical ground-water travel times were calculated using particle tracking. The effects of seasonal variation of recharge were determined by examining simulated profiles of the water table for selected times.

Optimal pumping rate

A steady-state simulation model of ground-water flow was used to determine the optimal pumping rate at the site. In this simulation, ground water will be extracted from the recovery trench, treated, and injected into the aquifer at three wells 400 ft upgradient from the trench. The injected water would enhance flushing of LNAPL by increasing hydraulic gradients toward the trench. A goal of the simulation modeling was to find the pumping rate that would keep the water level in the injection wells at the approximate altitude of the bedrock surface, 728 ft above sea level. Above this altitude, the bioventing action in the unsaturated zone would be negatively affected by rising ground-water levels. To achieve this goal, an iterative simulation strategy was adopted. In the first iteration, an arbitrary pumping rate in the recovery trench was chosen. A constant-head boundary condition equivalent to 728 ft at the injection well was placed in each cell that contained an injection well. The specified head in the injection cell was lower than 728 ft because the finite-difference scheme used in MODFLOW only allows the user to specify heads for an entire cell but not at a specific point within the cell. To allow for the averaging effect of the cell area, the head in the cell was adjusted using the following equation (Beljin, 1987):

$$h_w = h_{ij} - \frac{Q_w}{2\pi T}\ln\left(\frac{r_e}{r_w}\right), \qquad (1)$$

where

h_w is the head in the well, in feet,

h_{ij} is the average head in the cell, in feet,

Q_w is the pumping rate at the recovery trench, in cubic feet per day,

T is the transmissivity, in feet squared per day,

r_e is 0.208 times the width of the cell, in feet, and

r_w is the radius of the well in the trench, in feet.

The simulation model was run to calculate the rate at which water flowed from the constant-head boundary in order to reach a steady-state condition. In the next iteration, the flow rate at the constant-head injection wells from the previous iteration was used as the new pumping rate in the wells in the recovery trench, and the constant head in the injection well cells was adjusted based on equation (1) and the new pumping rate. The pumping rate remained constant after several iterations at a final rate of 8,570 ft^3/d (44 gal/min). Using the pumping rate of 44 gal/min, ground-water travel times between the injection wells and recovery trench ranged from 63 to 337 days; the mean travel time was 124 days.

Effect of seasonal variations of recharge on ground-water-flow paths

One option for the disposal of treated ground water is to discharge the water to streams during times when the streams are not likely to be frozen. A potential problem with this option is that when water is not being discharged it also cannot be pumped, because storage of the pumped water is impractical. When contaminated ground water is not being pumped from the aquifer, there is a chance that some LNAPL may migrate past the trench. Transient simulation modeling of ground-water flow was conducted to determine the effect on the ground-water flow system of seasonally varying pumping and recharge.

The seasonal distribution of recharge produced fluctuations in water levels through the year in the simulation model (fig. 10). Water levels decrease through late winter until the main recharge event in April. The rise during April is slightly attenuated because pumping also begins in April. After reaching their peak at the end of April, water levels decline steadily until the October recharge event. Water levels rise steadily in October until the end of the month when pumping ceases, which causes an abrupt rise. The effect of pumping extends 300 ft from the trench in the upgradient direction; at distances greater than 300 ft, the effect of pumping is minimal. Water levels reach the autumn peak in November and then begin the winter decline at the start of a new cycle.

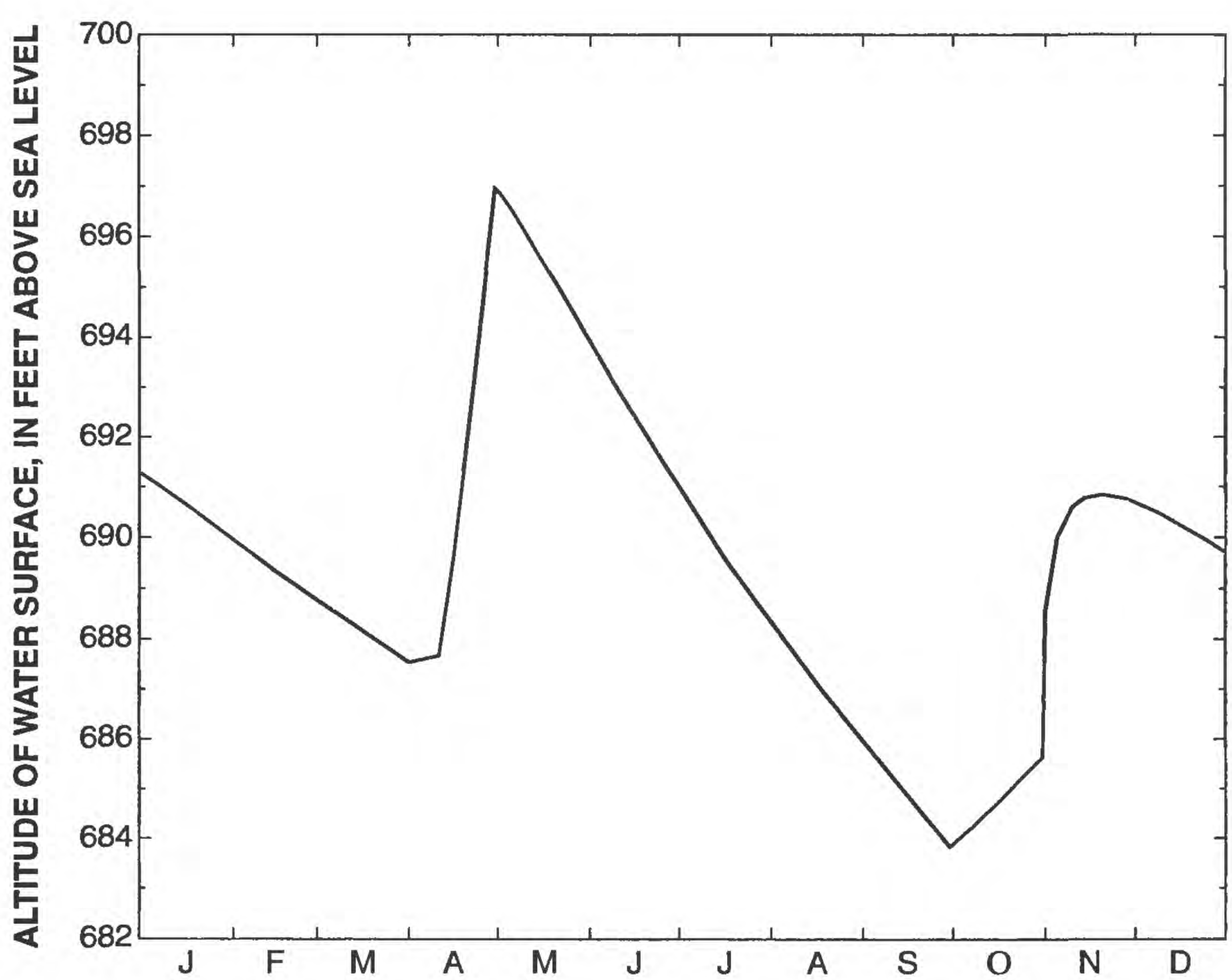


Figure 10. Simulated water levels near the trench, Fire Training Area, Loring Air Force Base, Maine. (Location of data point is shown on fig. 9.)

To assess the hydraulic control of LNAPL in the trench, the configuration of the water table was examined several times during the simulated yearly cycle (fig. 11). In the equivalent porous medium approach (discussed in the next section), a zone of mixed LNAPL and water is present above the water table (Farr and others, 1990). The flow of LNAPL in this zone is controlled by the slope of the water table. When the recovery trench is pumped, a depression forms on the water table (shown at the end of April and October on fig. 11) into which LNAPL flows and is trapped. As long as there is a depression of the water table in the trench, LNAPL will be trapped in the trench. Within days after pumping stops, the depression in the water table dissipates (shown at the end of March on fig. 11). In its place, an area of flat water-table gradient forms that is caused by the high hydraulic conductivity near the trench. Even after pumping has stopped and the depression disappears, ground-water-flow paths are bent toward the trench, although not as sharply as when the trench is pumped. When pumping resumes in April, the depression returns quickly to its maximum extent, and some LNAPL that migrated past the trench during nonpumping conditions will flow back toward the trench because of the depression in the water table. The area of reversal extends 80 ft downgradient of the trench (fig. 11).

Limitations of the simulation modeling approach

The application of the simulation modeling is limited by a number of factors; consequently, the intention of the simulation is not to predict ground-water levels or flow paths, but to show conceptually what might happen under idealized conditions. The results provide approximations of the hydrologic system that can be used, with an appropriate uncertainty factor, to evaluate some operational aspects of the contaminant removal system.

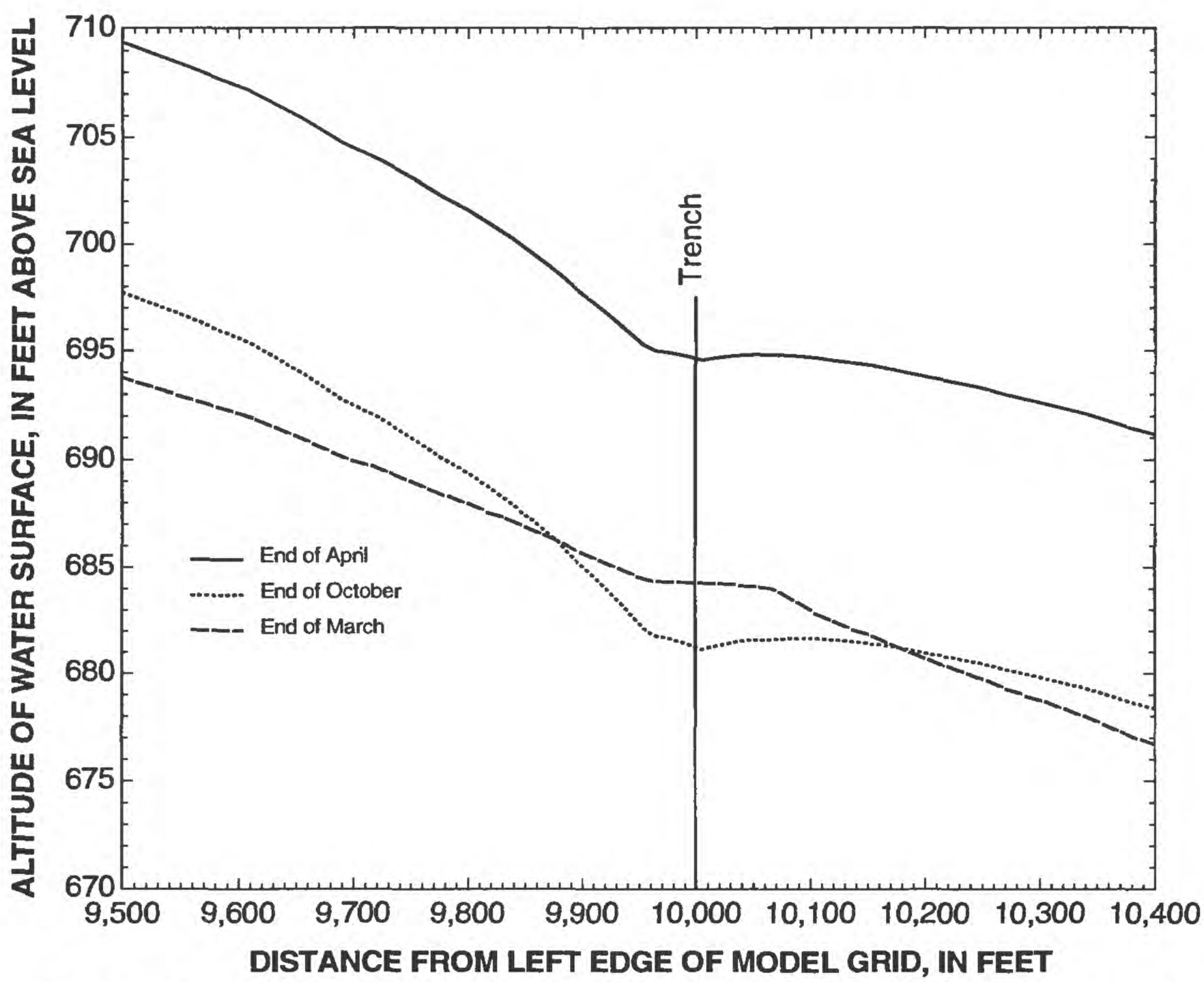


Figure 11. Simulated water-table profiles at selected times. (Profile is along row 60 of the model grid, shown in fig. 9.)

Equivalent porous medium assumption

The most important limitation of this simulation modeling is that it treats the aquifer system as a porous medium. This assumption is valid if the hydraulic properties of the aquifer are defined on an appropriate scale. For example, most aquifer properties, such as hydraulic conductivity, are defined for a representative volume of an aquifer. In a porous medium, void space is evenly distributed in a relatively small volume of aquifer, so the representative volume can be small. For the same small volume of a fractured-bedrock aquifer, void space is not evenly distributed. If the representative volume includes only unfractured rock, the hydraulic conductivity is low; if the volume includes only fractures, the hydraulic conductivity is high. As the size of the representative volume increases, the number of fractures increases and the hydraulic conductivity becomes more representative of the aquifer as a whole. This is the equivalent porous medium assumption: for a sufficiently large volume of aquifer, properties can be defined so that the position of individual fractures is not important. The size of the representative volume depends on the spacing and distribution of fractures. As the scale of the problem decreases, the importance of the location of individual fractures becomes more critical.

The simulation modeling that was done for this study covers a large area (3.3 mi^2) and the equivalent porous medium assumption probably is valid; however, the FTA is small relative to the modeled area, and the location of individual fractures is known to be important (Lane and others, 1996). The results of the simulation modeling need to be interpreted carefully near the FTA. The simulated water table (fig. 11) is for an equivalent porous medium and is presented only to show the effect of transient ground-water flow. Actual LNAPL flow paths are controlled by the pattern of fractures, but it is not possible to simulate the fracture distribution given the hydrogeologic complexity of the area. In a porous medium, the flow of LNAPL generally follows a curvilinear flow path along the water table. In a fractured medium, the flow of LNAPL may follow a rectilinear flow path through multiple fractures in many planes; thus, the length of the flow path in a fractured-rock aquifer can be longer than in an equivalent porous medium (the length of the flow path could also be shorter if the fracture was a direct connection between a recharge and discharge area). The time of travel depends on the length of the flow path, so travel times simulated using the equivalent porous medium assumption are probably too low.

Aqueous-phase flow assumption

In this study, the flow of water is simulated, not the flow of LNAPL. The flow of LNAPL differs from the flow of water because of differences in their viscosity and density. The hydraulic conductivity of the aquifer material depends on the viscosity and density of the fluid moving through it. LNAPLs are not a single compound, but a mixture of compounds, and so have characteristics that affect hydraulic conductivity nonuniformly. LNAPLs are all less dense than water, so the hydraulic conductivity with respect to LNAPL will be less than it is for pure water. On the other hand, some LNAPL is less viscous than water, and the hydraulic conductivity with respect to the LNAPL is greater than that for pure water.

The capillary forces at the water table and in fractures also can significantly affect the flow of LNAPL. The amounts of LNAPL and water present in the aquifer relative to each other depend on the interfacial tension between air and LNAPL and between LNAPL and water (Farr and others, 1990; Lenhard and Parker, 1990). In most porous aquifers, the transition between the unsaturated zone, the zone of mixed LNAPL and water, and the zone saturated with water only is smooth; there is no discrete layer of LNAPL. The effective pore size, which is analogous to the fracture width, is potentially much greater in a fractured-rock aquifer than in a porous media.

CONCLUSIONS

Injecting treated water year-round and recovering LNAPL in the trench is possible under ideal conditions; some non-ideal conditions not discussed here, such as clogging of the injection wells with iron bacteria, may make this solution impractical. The relatively low pumping rate of 44 gal/min, in addition to meeting the design constraints, would have the additional benefit of maintaining a low hydraulic gradient and avoiding stranding the LNAPL in "dead-end" fractures and other poorly connected zones in the aquifer.

Fluctuations in ground-water levels caused by seasonal variation of recharge would have minimal effect on the operation of the contaminant removal system. Ground water flows through the aquifer system most rapidly after major recharge events. Nonpumping conditions occur when recharge is minimal, ground-water velocities are lowest, and LNAPL flow past the trench is minimal. By pumping the trench immediately after significant snowmelt in the spring, and continuing to pump until ground-water levels have stabilized near their minimum in autumn, most LNAPL will be captured. When pumping resumes in the spring, some LNAPL that migrated past the trench will flow back toward the trench because of the depression in the water table caused by pumping.

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