Assessment of the Potential Effects of Phytoremediation on Ground-Water Flow Around Area C at Orlando Naval Training Center, Florida

By Keith J. Halford

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CONVERSION FACTORS, VERTICAL DATUM, ABBREVIATIONS AND ACRONYMS

Multiply	By	To obtain
	Length	
foot (ft)	0.3048	meter
	Area	
square foot (ft^2)	0.0929	square meter
square mile (mi^2)	2.590	square kilometer
	Volume	
cubic foot (ft^3)	0.028317	cubic meter
	Flow	
gallon per minute (gal/min)	0.06309	liter per second
inch per year (in/yr)	25.4	millimeter per year
	Hydraulic Conductivity	
foot per day (ft/d)	0.3048	meter per day
	Leakance	
foot per day per foot [(ft/d)/ft]	1.000	meter per day per meter
	****Transmissivity***	
foot squared per day (ft^2/d)	0.0929	meter squared per day

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows: °C=(°F-32)/1.8.

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.

***Transmissivity:** The standard unit for transmissivity is cubic foot per day per square foot times foot of aquifer thickness [(ft^3/d)/ft^2]ft. In this report, the mathematically reduced form, foot squared per day (ft^2/d), is used for convenience.

Acronyms and additional abbreviations used in report:

DCE	Dichloroethene
ET	Evapotranspiration
MODFLOW	Modular Finite-Difference Model
NTC	Naval Training Center
PCE	Tetrachloroethene
RMS	Root mean square
TCE	Trichloroethene
SR	State Road
SS	Sum of squares
VC	Vinyl chloride
VOCs	Volatile organic compounds

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Abstract

Ground-water flow through the surficial aquifer system at area C of the Orlando Naval Training Center in Florida was simulated with a three-layer finite-difference model. The model was calibrated to 80 water-level measurements from 30 wells during four synoptic surveys that were conducted between October 24, 1995, and January 31, 1997. A quantifiable understanding of ground-water flow through the surficial aquifer was needed to evaluate the potential effects of phytoremediation as a long-term, remedial-action alternative to control the discharge of contaminated ground water to Lake Druid.

An aquifer test was reanalyzed to estimate the hydraulic properties of the surficial aquifer system, which was divided into two geohydrologic units—the upper zone and the lower zone. The effect of evapotranspiration from a forested area was investigated. The analysis of ground-water flow and potential movement of contaminants within the surficial aquifer system was addressed using the calibrated model driven by a uniform recharge rate of 19 inches per year and an evapotranspiration rate of 30 inches per year from the forested area. These recharge conditions were used to simulate the advective movement of conservative contaminants through area C with MODPATH for all remedial alternatives.

Under existing conditions, phytoremediation alone at area C at Orlando Naval Training Center cannot stop the discharge of contaminants into Lake Druid because within the analyzed control volume the evapotranspirative losses (3.7 gallons per minute) are small relative to the ground-water discharge to Lake Druid (20.4 gallons per minute). The installation of a drainage ditch could redirect flow through the rooting zone of aquatic plants which would remediate the ground-water discharge. A drainage ditch would intercept flow from 0 to about 40 feet below the water table but would still allow water in the lower 20 feet of the surficial aquifer system to continue to discharge to Lake Druid.

INTRODUCTION

Inorganic and organic priority contaminants defined by the U.S. Environmental Protection Agency (USEPA, 1988) have been detected in surface water, sediment, and ground-water samples collected near an abandoned laundry facility at area C of the Orlando Naval Training Center (NTC). The Orlando NTC is located in central Orange County to the east of U.S. Highway 17-92 and north of State Road (SR) 50 (fig. 1). Chlorinated volatile organic compounds (VOCs) from dry-cleaning compounds that were spilled near the laundry facility (building 1100) are being discharged (1998) from the surficial aquifer system to Lake Druid (fig. 1). Concentrations of tetrachloroethene (PCE), trichloroethene (TCE), cis-1,2-dichloroethene (DCE), trans-1,2-DCE, 1,1-DCE, and vinyl chloride (VC) discharged to Lake Druid exceed Florida surface-water standards (U.S. Department of the Navy, 1997). The Navy installed two vertical recirculation wells in 1997 as a temporary measure to comply with Florida environmental standards, but a long-term solution is needed to attenuate contaminant concentrations to levels that comply with State regulations. As part of the Installation Restoration Program, Orlando NTC is considering phytoremediation as a long-term alternative to control.

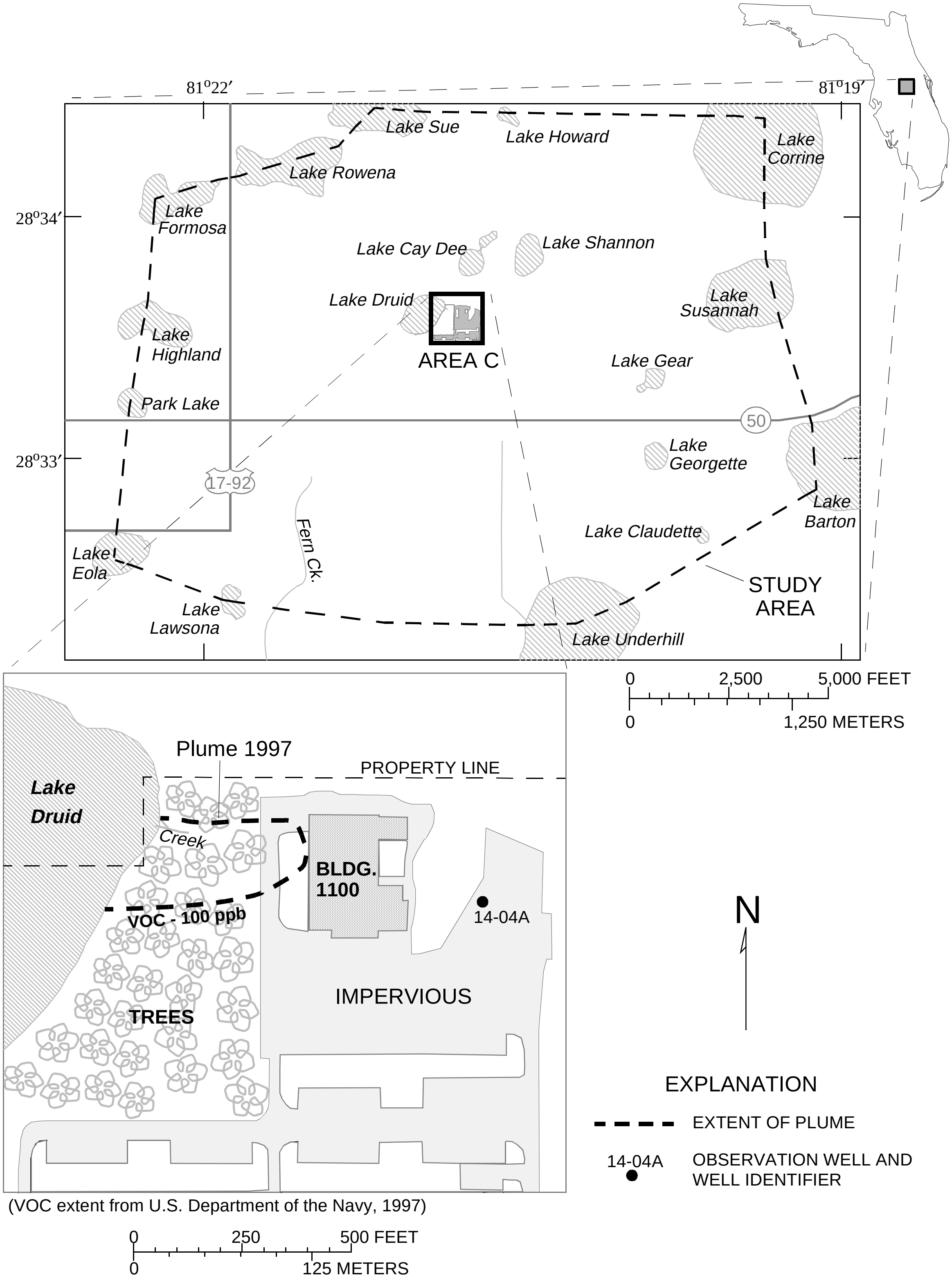

Figure 1. Location of the study area C at Orlando Naval Training Center, Orange County; building 1100; and plume of volatile organic compounds.

the discharge of contaminated ground water to Lake Druid. This effort requires a quantifiable understanding of the response of ground-water flow to current conditions and to any future stresses imposed on the surficial aquifer system. Numerical simulation provides the most tractable method of achieving this level of understanding.

Phytoremediation affects ground-water flow and reduces chlorinated solvent contaminant concentrations through the selective use of plants with high transpiration rates, high dehalogenase and oxidase activity, and a tolerance to elevated contaminant concentrations (Schnoor and others, 1995). Plants can provide hydraulic control of the contaminant plume if the transpiration rate across the phytoremediated area is sufficiently elevated above the pre-remediation transpiration rates. Ground-water contaminant concentrations are reduced by dehalogenase enzymes as the water flows through the rooting zone and is transpired by the plant (Schnoor and others, 1995).

The dissolved and degraded constituents of the dry-cleaning compounds (PCE, TCE, DCE, trans-1,2-DCE, 1,1-DCE, and VC) are the primary contaminants transported by ground water at area C of the Orlando NTC. The direction of movement of these dissolved constituents is similar to the advective flow of the ground water, as the solubility of these contaminants is usually low and the concentrations are not great enough to significantly alter the density of the ground water. These dissolved constituents sorb to the porous media of aquifers and confining units, which retards the rate of travel, but does not alter the direction of travel.

Purpose and Scope

This report presents the results of a study to estimate the potential changes to the ground-water flow system beneath area C at Orlando NTC resulting from phytoremediation. The report includes a description of the hydrogeologic framework, estimates of the lateral and vertical hydraulic conductivity of the upper zone and lower zone of the surficial aquifer system, and estimates of the recharge rate and average ground-water discharge to Lake Druid. Two remediation alternatives were investigated using a calibrated ground-water flow model.

Acknowledgments

The author extends his appreciation to Barbara Nwokike and Cliff Casey, Southern Division Naval Facilities Engineering Command; and John Kaiser, ABB Environmental Services, for assistance provided during this study.

GEOHYDROLOGY

The geologic units of interest in the study area include sediments of Holocene to Pliocene age that previous investigators have defined as the surficial aquifer system (Murray and Halford, 1996). The surficial aquifer system consists of well-sorted, fine-grained sand from 0 to 63 ft below land surface (U.S. Department of the Navy, 1997). The base of the surficial aquifer is the Hawthorn Group, which separates the surficial aquifer from the Upper Floridan aquifer. The Hawthorn Group is about 90 ft thick in the study area and consists of marine clays and discontinuous limestone stringers (Lichtler and others, 1968).

Because this study is concerned with ground-water movement near land surface, the surficial aquifer system was further subdivided into two local hydrogeologic units: the upper zone and the lower zone (fig. 2). Geologists' logs indicated very little difference throughout the geohydrologic column except for the occurrence of cemented sands between 15 and 20 ft below land surface (U.S. Department of the Navy, 1997). Differentiation of the surficial aquifer system into two zones primarily was based on the results of slug tests and a pumping aquifer test. These tests indicated that the hydraulic characteristics of the upper zone (0 to 20 ft below the water table) were measurably different from those of the lower zone (20 to 60 ft below the water table).

WATER BUDGET

A long-term water budget for the study area can be described by the following equation:

$$P + I - Q - D - ET = \Delta S \cong 0\,, \qquad (1)$$

where

P is precipitation, in inches per year;
I is irrigation, in inches per year;

Series	Formation	Lithology	Geohydrologic unit			
			Tibbals (1990)	This report	Model layers	Thick-ness, in feet
Holocene to Upper Pliocene	Undiffer-entiated surficial deposits	Well-sorted sand with a layer of finer sand between 15 and 20 feet below land surface	Surficial aquifer system	Upper zone	1, 2	20
				Lower zone	3	40
Miocene	Hawthorn Group	Clay and interbedded phosphatic sands	Intermediate confining unit		--	--

(Modified from Tibbals, 1990)

Figure 2. Generalized geologic and geohydrologic units and aquifers beneath Orlando Naval Training Center.

Q is surface-water discharge, in inches per year, which is composed of surface runoff, Q_S, and base flow, Q_B;

D is deep leakage from the surficial aquifer system to the Upper Floridan aquifer, in inches per year;

ET is evapotranspiration, in inches per year; and

ΔS is change in storage, in inches per year, which is assumed to be negligible over the long term.

Precipitation is the dominant and most variable source of water in the water budget and averages about 50 in/yr (Owenby and Ezell, 1992). Irrigation water from public supplies adds about 7 in/yr to the study area and generally is applied to lawns and ornamental plants (L.A. Bradner, U.S. Geological Survey, written commun., 1997).

Surface-water discharge, deep leakage, and evapotranspiration (ET) are the major losses from the water budget in the study area (fig. 3). Most of the surface-water discharge is routed into lakes, and the stages of these lakes are regulated by drainage wells that discharge from the lakes to the Upper Floridan aquifer. Bradner (1996) estimated that surface-water discharge through drainage wells was 20 in/yr within the study area. Deep leakage previously was estimated to be about 3 in/yr (Tibbals, 1990). The average evapotranspiration from the study area (34 in/yr) is estimated as the water added to the budget by precipitation and irrigation minus the water lost to surface-water discharge and deep leakage.

Recharge (N) is the subcomponent of the water budget that drives ground-water flow through the surficial aquifer system (fig. 3) and can be defined as: $N = P + I - ET - Q_S$ or $N = Q - Q_S + D$. The surficial aquifer system is recharged when applied water exceeds evapotranspirative losses and overcomes capillary effects in the unsaturated zone. Surface runoff (Q_S) occurs when the infiltration capacity of the soil is exceeded and additional precipitation or applied irrigation water drains directly to local streams, lakes, or depressions without infiltrating the subsurface (fig. 3). Recharge is the fraction of water reaching the water table that is not immediately extracted by evapotranspiration. Water leaves the surficial aquifer system either through evapotranspiration, base flow, or deep leakage (fig. 3).

The maximum amount of water available for recharge in the study area (23 in/yr) is the sum of the surface-water discharge and deep leakage. The actual recharge rate is expected to be less than 23 in/yr because some of the surface runoff is conveyed directly to the surface-water features and does not infiltrate into the surficial aquifer system.

The water-budget analysis provides a general estimate of the maximum amount of water that can pass through the surficial aquifer system, but cannot indicate what fraction of flow passes through the upper zone or lower zone of the surficial aquifer system. Additionally, the direction and velocity of contaminant movement from specific sites cannot be determined through a water-budget analysis. Therefore, a ground-water flow model is needed to address these more specific questions.

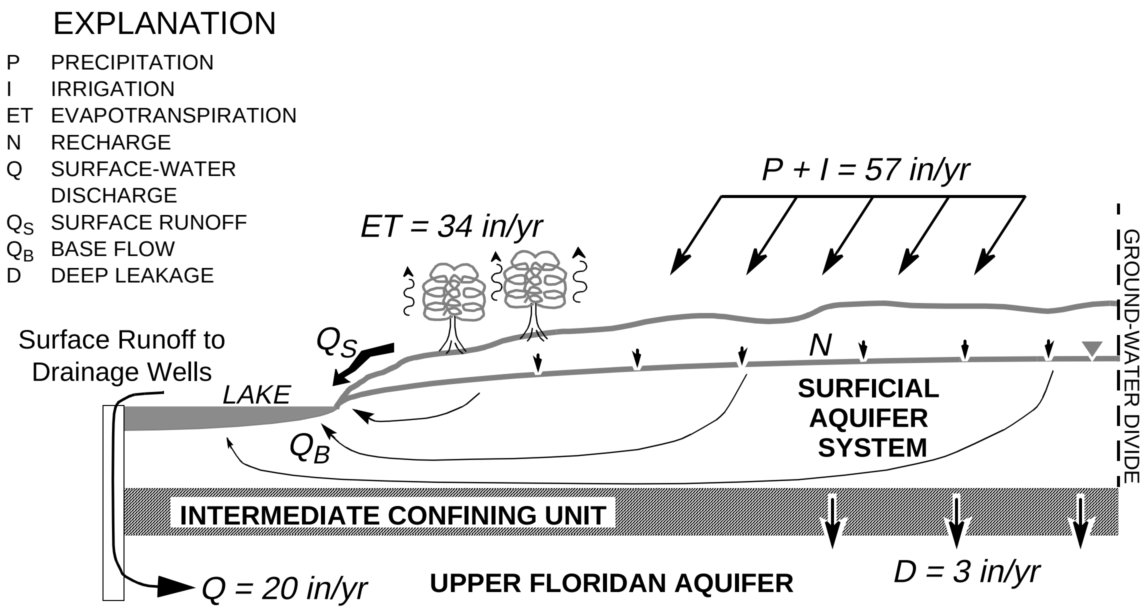

Figure 3. The water budget and its components in the study area.

SIMULATION OF GROUND-WATER FLOW IN THE SURFICIAL AQUIFER SYSTEM

A three-dimensional numerical model was used to quantitatively analyze ground-water flow and the advective transport of contaminants through the surficial aquifer system. The McDonald and Harbaugh (1988) modular finite-difference model (MODFLOW) was used to simulate flow in the surficial aquifer system and solve the governing equation:

$$\nabla \cdot (Kb\nabla h) + q + (P + I - Q_S) - ET = S\frac{\partial h}{\partial t}, \quad (2)$$

where

∇ is del, the vector differential operator;
K is hydraulic conductivity, in feet per day;
b is thickness, in feet;
h is hydraulic head, in feet;
q is a source or sink, in feet per day;
$P+I-Q_S$ is precipitation plus irrigation minus surface runoff, in feet per day;
ET is evapotranspiration, in feet per day;
S is storage coefficient in confined aquifers and the specific yield in unconfined aquifers, dimensionless; and
t is time, in days.

Description of the Ground-Water Flow Model

The study area was discretized into a rectangular grid of cells by row and column to implement a finite-difference model. The active model grid covered an area of about 6.7 mi^2 and was divided into 101 rows of 114 columns (fig. 4). Smaller cells were used near the observation wells between Lake Druid and building 1100 to avoid over-linearization in the area of interest. The model cells ranged in area from 100 to 1,000,000 ft^2 and the largest cells were in peripheral areas of little stress, away from the area of interest. Of the 34,542 model cells, 693 cells were inactive beyond the study area.

The grid was oriented along a north-south axis for simplicity because the majority of stresses or boundary conditions were not aligned along any particular axis. No measurements of anisotropy were available and a lateral anisotropy ratio of 1:1 was used for simulation. Values of aquifer hydraulic properties were assigned to the center of each cell (defined as a node) from values estimated from an aquifer test.

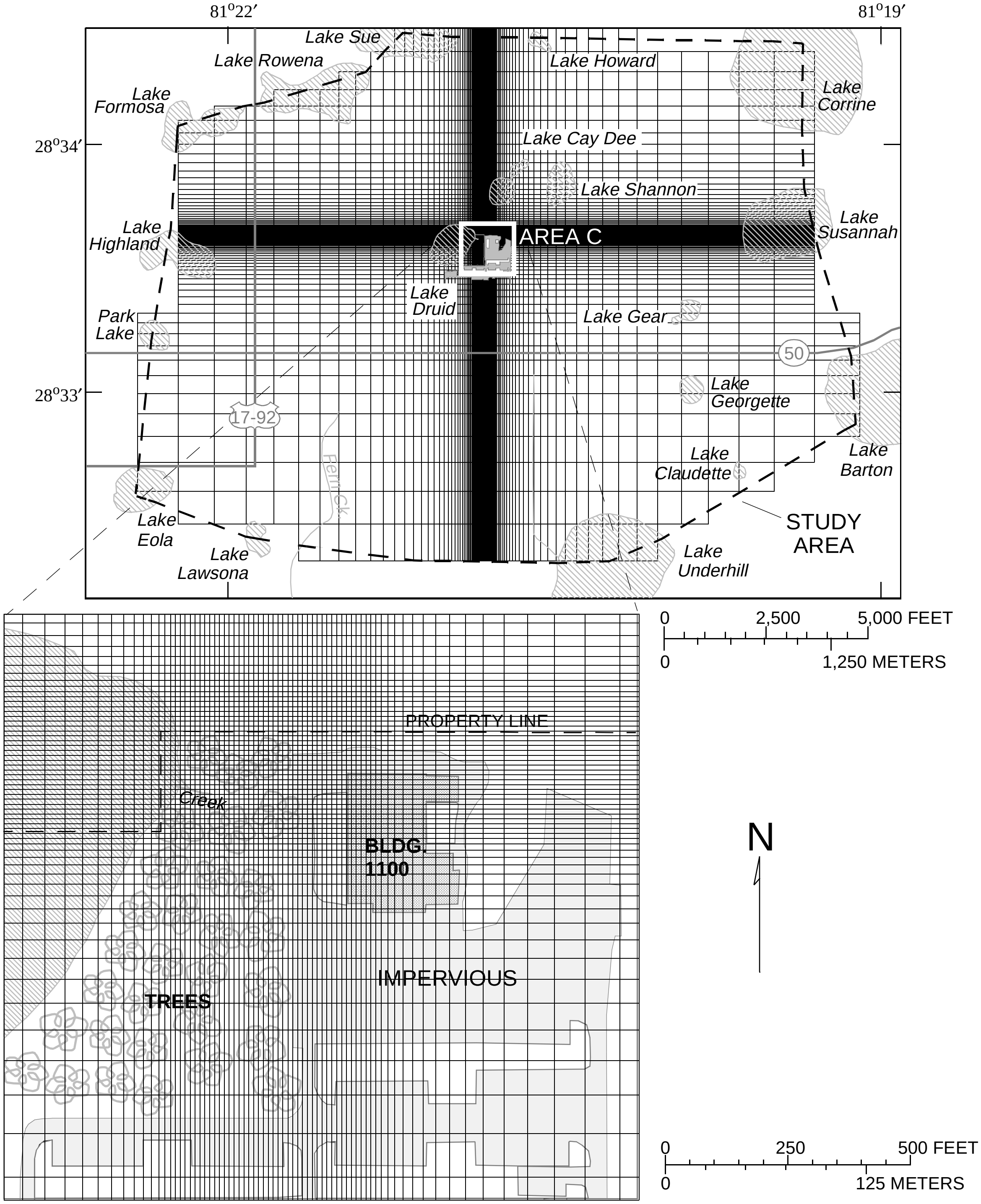

Figure 4. Location and extent of active model grid.

The model was vertically discretized into three layers to simulate the upper zone and lower zone of the surficial aquifer system. The upper zone was subdivided into two layers to facilitate the simulation of surface-water features. Vertical impedance to flow within the surficial aquifer system was simulated by assigning leakance values at each cell between model layers. Leakance is defined as the average vertical hydraulic conductivity of the aquifer between nodes divided by the vertical distance between corresponding nodes in adjacent model layers, and is in units of feet per day per foot (d^{-1}).

Hydraulic Characteristics

An aquifer test that was conducted between Lake Druid and building 1100 in area C was reanalyzed to define the hydraulic characteristics of the surficial aquifer system. The original analysis (U.S. Department of the Navy, 1996) characterized the surficial aquifer system as a Theisian aquifer with an average hydraulic conductivity of 33 ft/d. The drawdown data from this test in addition to geologic information indicated that the surficial aquifer system could be better characterized with two distinct zones (the upper zone and lower zone). Using a more complete conceptual model of the surficial aquifer system, the lateral and vertical hydraulic conductivities of the upper zone and lower zone were estimated from the reanalysis.

Water was produced from well 13-RW1 (fig. 5) during the aquifer test and the water level responses in 12 observation wells were measured and used for the analysis. The wells were spaced in plan and in a radial section as shown in figures 5 and 6. The volume of water produced was measured with an in-line totalizing flowmeter that was monitored periodically to estimate the flow rate (U.S. Department of the Navy, 1996). Well 13-RW1 was pumped at 52 gal/min for the first 3 minutes of the aquifer test and the flow rate was reduced to 39 gal/min for the remainder of the aquifer test.

Drawdown estimates were made by subtracting the measured water level from the water level just prior to stressing the aquifer. Water-level measurements from several days prior to the test suggested that any background water-level change was small relative to the effects of pumping well 13-RW1. Drawdowns were estimated for only the first 5 hours of the test because a rainfall event induced recharge slightly more than 5 hours after starting the test.

Hydraulic conductivities were estimated by fitting another ground-water flow model (the aquifer-test model) to drawdowns measured in observation wells during the aquifer test. The aquifer-test model was discretized differently than the primary flow model to account for the configuration of the observation wells and the pumped interval of well 13-RW1 (fig. 6). Complicated systems, such as the one at Orlando NTC, require a model that can account for responses to stresses from a partially penetrating well in an unconfined aquifer, the effects of stratified hydraulic characteristics, and the asymmetric influence of surface-water features on drawdown in the aquifer.

The aquifer-test model of the surficial aquifer system was discretized into 10 layers of 76 rows of 85 columns. The smallest cells were 2 ft on a side and were centered on the production well, 13-RW1. Lateral boundaries were specified as no-flow and placed more than 3,000 ft from the production well (fig. 5). Changes in the saturated thickness of the aquifer were not simulated because the maximum drawdowns of about 4 ft were small relative to the total wetted thickness of the surficial aquifer system (60 ft). The lower boundary was specified as no-flow at the contact between the surficial aquifer system and the intermediate confining unit.

Layer 1 of the aquifer-test model was only 1 ft thick to better approximate surface-water features because the creek and Lake Druid (within 50 ft of the shoreline) are very shallow features that are less than 1 ft deep. A thin upper layer better approximates the drainage of water as the water table declines because drainage adds water to the top of the surficial aquifer system. Finer vertical discretization was used in layers 4 through 8 than layers 2, 3, 9, and 10 to better simulate the end effects of the production well and contrasts between the hydraulic characteristics of the upper zone and lower zone.

The surficial aquifer system and production well were characterized with seven estimated parameters. The four primary parameters of interest were the lateral and vertical hydraulic conductivities of the upper zone and lower zone (K_{XY}-Upper, K_Z-Upper, K_{XY}-Lower, and K_Z-Lower). The remaining three estimated parameters were specific storage (S_S), specific yield (S_Y), and well-bore storage (S_{well}).

The parameters were estimated by minimizing the difference between simulated and measured drawdowns with an optimization routine (Halford, 1992) coupled to MODFLOW. Lateral hydraulic conductivity

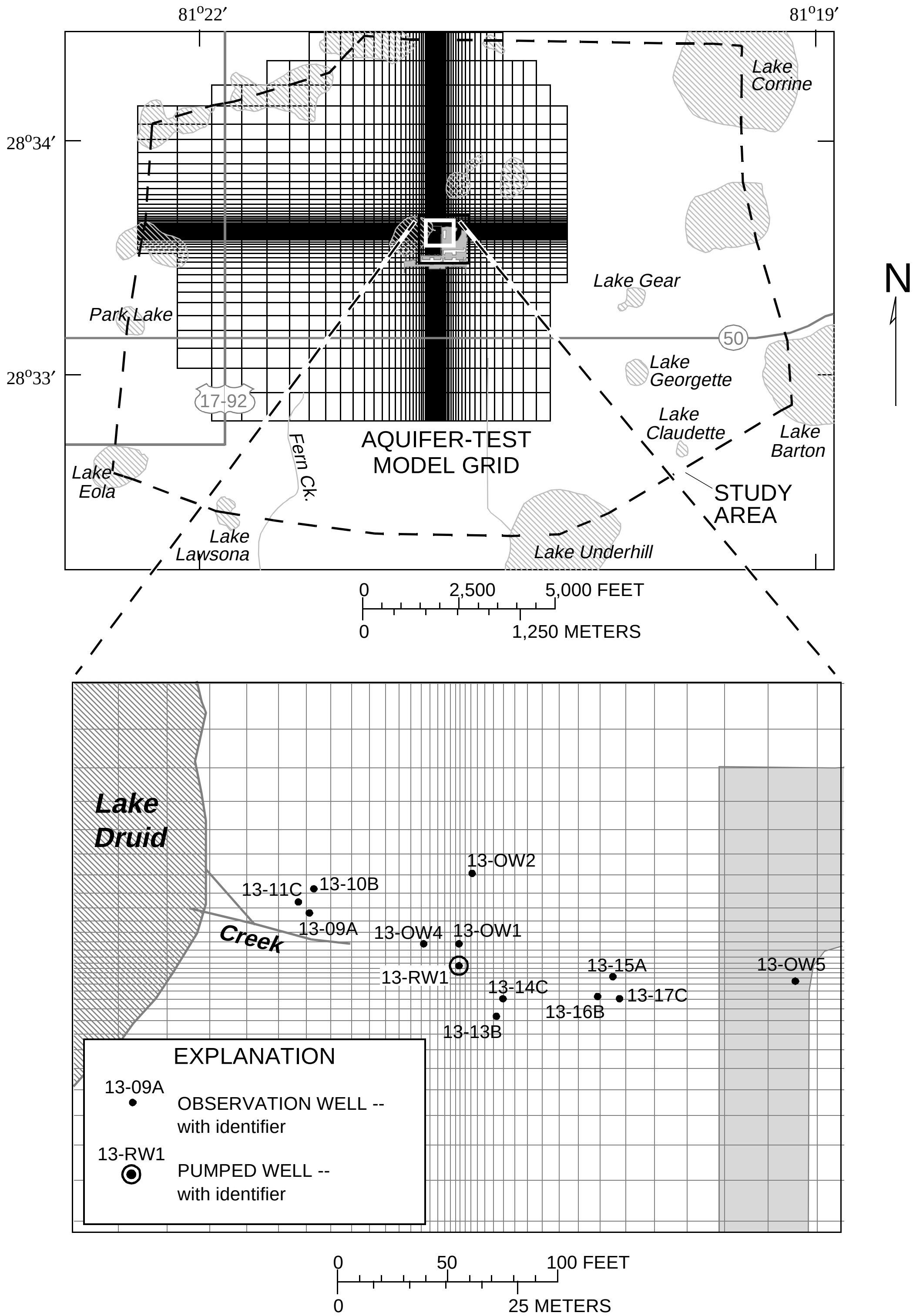

Figure 5. Location of aquifer test site, observation wells, and aquifer-test model grid at area C of the Orlando Naval Training Center, Florida.

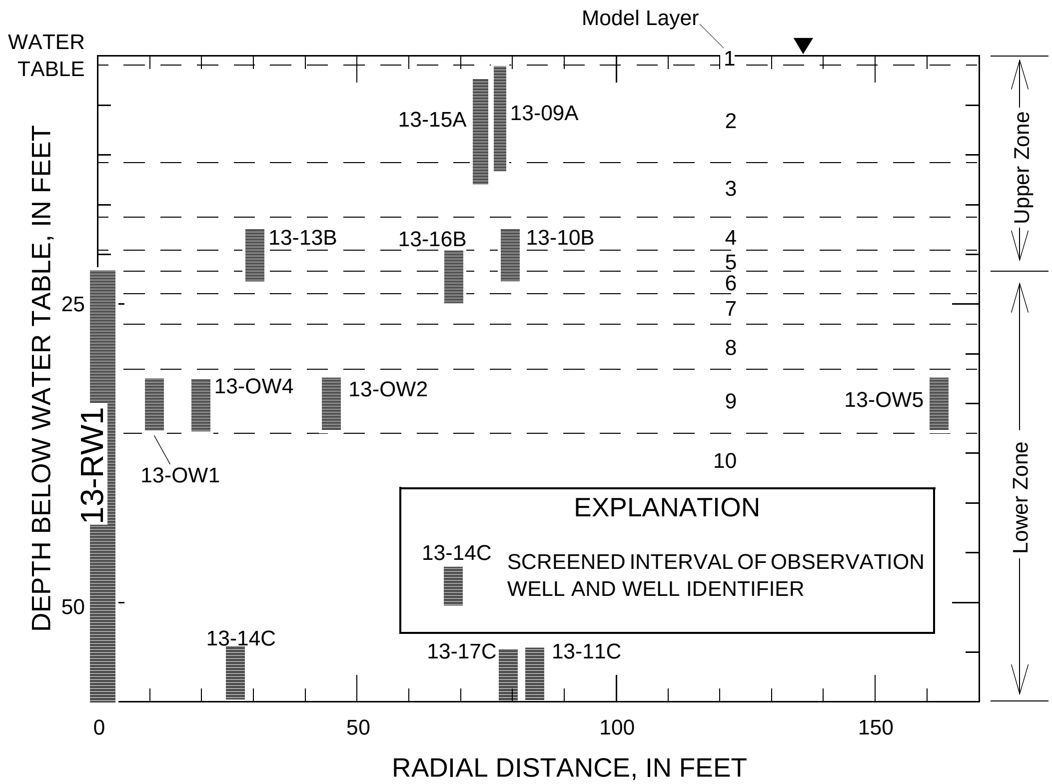

Figure 6. Model layers and well placement for the surficial aquifer system test at area C of the Orlando Naval Training Center.

estimates for the upper zone and lower zone were 10 and 40 ft/d, respectively. The average transmissivity estimate of the surficial aquifer system (1,800 ft^2/d) from this analysis is similar to the Theis analysis (2,000 ft^2/d). Vertical hydraulic conductivity estimates of the upper zone and lower zone were 3.8 and 17 ft/d, respectively. Final estimates of hydraulic conductivities and storage coefficients are listed in table 1.

K_{XY}-Upper and K_{XY}-Lower were the most highly correlated parameters (0.85) of the aquifer-test model; most of the other parameter pairs had correlation coefficients less than 0.80 (table 1). The difference between simulated and measured drawdowns was most sensitive to the estimates of K_{XY}-Lower and K_Z-Upper and least sensitive to estimates of S_{well} (table 1). As the relative sensitivity of a parameter decreases, the

Table 1. Correlation coefficients between parameters from the aquifer-test model at area C of Orlando Naval Training Center

[--, not applicable; ft/d, feet per day; 1/ft, one over feet]

Parameters	Correlation coefficients, $\rho_{i,j}$							Estimates		Relative sensitivity
K_{XY}-Lower	1.00	--	--	--	--	--	--	40	ft/d	1.00
K_Z-Upper	.85	1.00	--	--	--	--	--	3.8	ft/d	.31
S_S	.36	.30	1.00	--	--	--	--	20.E-6	1/ft	.18
S_Y	.54	.53	.06	1.00	--	--	--	.039		.13
K_Z-Lower	.38	.31	.10	.06	1.00	--	--	17	ft/d	.06
K_{XY}-Upper	.86	.79	.22	.75	.07	1.00	--	10	ft/d	.05
S_{well}	.13	.09	.59	-.02	.06	.05	1.00	.16		.01
$\rho_{i,j} = \frac{C_{i,j}}{\sqrt{C_{i,i}C_{j,j}}}$	K_{XY}-Lower	K_Z-Upper	S_S	S_Y	K_Z-Lower	K_{XY}-Upper	S_{well}			$\sqrt{C_{i,i}/C_{1,1}}$

relative uncertainty of a parameter estimate increases. The uncertainties associated with the estimates of K_{XY}-Lower and K_Z-Upper are small relative to the uncertainties associated with the estimates of K_{XY}-Upper and K_Z-Lower (table 1).

The lateral and vertical hydraulic conductivities of the upper zone and lower zone estimated with the aquifer-test model (table 1) were assumed to be the best estimates of these properties. The lateral and vertical hydraulic conductivities that were estimated from the aquifer-test model were treated as uniform, known quantities in the primary model.

Surface-Water Features

The distribution and altitude of surface-water features control the direction and rate of flow in the surficial aquifer system. The distribution of surface-water features was determined from aerial photographs (Doolittle and Schellentrager, 1978) and plans of Orlando NTC (U.S. Department of the Navy, 1997). The location of a small creek that extends approximately from well 13-RW1 to Lake Druid (fig. 5) was surveyed and the stage was determined to be equal to that of Lake Druid. The stage of Lake Druid has been measured on a monthly basis by the city of Orlando since 1957 (Kevin McCann, City of Orlando, written commun., 1997). The median stage of Lake Druid is 101.03 ft above sea level. The stage is not highly variable, as indicated by the 95 to 5 percent exceedence which ranges from 100.29 to 101.25 ft above sea level. In the remainder of the study area, the elevations of the other lakes and streams were taken from 1:24,000 scale, U.S. Geological Survey quadrangle sheets.

Interactions between the surficial aquifer system and the lakes and creeks were simulated by river nodes. The flow rate in or out of the aquifer at a river node was defined by

$$Q_B = C_{RB}(H_{RIVER} - H_{AQUIFER}) , \qquad (3)$$

where

C_{RB} is the hydraulic conductance of the riverbed, in feet squared per day;

H_{RIVER} is the average stage of the river or lake, in feet; and

$H_{AQUIFER}$ is the head in the aquifer beneath the river, in feet.

Equation 3 applies if $H_{AQUIFER}$ is greater than or equal to the assigned elevation of the bottom of the surface-water feature.

A riverbed conductance of 100,000 ft^2/d was assigned for all river nodes, based on the results of a similar study at Cecil Field NAS (Halford, 1998) that indicated the ground-water flow model was not sensitive to this parameter. Estimates of riverbed conductance were highly correlated with estimates of lateral hydraulic conductivity of the surficial aquifer, a more sensitive parameter than the riverbed conductance. This result implies that surface-water interaction is controlled by the hydraulic conductivity of the surficial aquifer rather than the riverbed conductance.

All lakes and creeks shown in figure 1 were represented in the model. A total of 1,170 river nodes was assigned to layer 1; 1,141 of these nodes simulated lakes within the study area. The river-bottom elevation for all creeks was set equal to the river stage to ensure that all simulated reaches were either gaining or inactive. For lakes, the river bottom was set far below the elevation of the water surface so that water could be gained or lost from these features.

Boundary Conditions

Proper representation of model boundary conditions is one of the most important aspects in the simulation of an aquifer system. Model boundaries are assigned to represent the actual hydrologic boundaries as accurately as possible. If model boundaries are generalized, they are placed far enough away from the influence of hydrologic stresses in the model area to minimize their effects on simulation results.

The upper boundary, layer 1, is the water table and is usually represented in MODFLOW as a free surface. Changes in the wetted thickness of the aquifer were not simulated (layer 1 had a uniform transmissivity) because the variation in water-table elevations in area C is typically less than 5 ft, which is small relative to the total wetted thickness of the surficial aquifer system (60 ft). A spatially uniform recharge rate was applied to this boundary. Although many land covers exist within the study area, insufficient water-level data and hydraulic conductivity estimates are available to differentiate the effects on recharge of these land covers.

The lower model boundary is the contact between the surficial aquifer system and the intermediate confining unit and is simulated as a specified flux boundary. Deep leakage across this boundary from the surficial aquifer system to the Upper Floridan aquifer previously was estimated to be about 3 in/yr (Tibbals, 1990). Deep leakage is not expected to be highly variable because the difference between the water table of the surficial aquifer system and the potentiometric surface of the Upper Floridan aquifer beneath area C ranges from 55 to 60 ft.

The lateral model boundaries in each layer are no-flow boundaries that cut through lakes along the periphery of the study area (fig. 4). The boundaries beneath the lakes are considered to be no-flow because most of the ground-water flow beneath a lake is vertical into or out of the lake. The boundaries between lakes are considered to be no-flow because these boundaries generally coincide with ground-water flowpaths between the lakes and water flows parallel to the boundary.

Model Calibration

Calibration is the attempt to reduce the difference between model results and measured data by adjusting model input. Calibration was accomplished in this study by adjusting input values of recharge until an acceptable calibration criterion was achieved. The "goodness" or improvement of the calibration generally is based on the differences between simulated and measured ground-water levels and stream discharges. Simulated water levels and discharges from a calibrated, deterministic ground-water model commonly depart from measured water levels and discharges, even after a diligent calibration effort. The discrepancy between model results and measurements (model error) commonly is the cumulative result of simplification of the conceptual model, grid scale, and the difficulty in obtaining sufficient measurements to account for all of the spatial variation in hydraulic properties and recharge throughout the model area.

The ground-water flow model for the Orlando NTC was calibrated to 80 water-level measurements from 30 wells. Data were collected during four synoptic surveys conducted from October 24, 1995, to January 31, 1997 (U.S. Department of the Navy, 1997). The synoptic surveys were treated as independent "snapshots" of the ground-water system. The data from these surveys were fitted to the simpler steady-state equation:

$$\nabla(Kb\nabla h) + q + N' = 0 \tag{4}$$

where

N' is the effective recharge rate during a given survey, in feet per day. The effective recharge rate (N') is the summation of precipitation, surface runoff, evapotranspiration, and water released from storage and can be expressed as:

$$N' = (P + I - Q_S) - ET - S_y\frac{\partial h}{\partial t}. \tag{5}$$

Although the effective recharge rates estimated for each synoptic survey period are not estimates of the long-term recharge rate, estimates obtained during extreme conditions can bracket the long-term recharge rate.

Some stresses must be known to calibrate a model if both recharge rates and hydraulic conductivities are simultaneously adjusted. When the use of equation 4 is appropriate, the stresses and recharge rates are proportional to the hydraulic conductivity. Usually, the lateral and vertical hydraulic conductivities are estimated when calibrating ground-water flow models of a surficial aquifer system. Commonly, stream discharge during baseflow conditions is assumed to represent the recharge rate to the aquifer during a specified period.

In a setting without stream-discharge measurements, a successful calibration strategy is based on knowing the lateral hydraulic conductivity distributions of the flow zones and estimating recharge rates during model calibration. Ground-water flow rates estimated with this approach are not dependent on the quality of stream discharge measurements and assumptions about baseflow. Instead, the accuracy of flow-rate estimates is dependent on the quality of lateral hydraulic conductivity estimates.

Calibration improvement was determined by decreases in sum-of-squares error, which is defined by:

$$SS = \sum_{k=1}^{nwl} [\hat{h}_k - h_k]^2 \quad , \tag{6}$$

where

$\hat{h}_k$ is the k$^{\underline{th}}$ simulated water level, in feet;

h_k is the k$^{\underline{th}}$ measured water level, in feet; and

nwl is the number of water-level comparisons.

Although the sum-of-squares error serves as the objective function, root-mean-square (RMS) error is reported instead because RMS error is more directly comparable to actual values and serves as a composite of the average and the standard deviation of a set. Root-mean-square error is related to the sum-of-squares error by:

$$RMS = \sqrt{\frac{SS}{nwl}} \ . \qquad (7)$$

Because measured water levels rarely coincide with the center of a cell, simulated water levels were interpolated laterally to points of measurement from the centers of surrounding cells. The simulated water levels were assumed to be part of a continuous distribution. Vertical interpolation was not considered because of the discontinuity and associated refraction of potential fields across zones of differing hydraulic characteristics.

Parameter Estimation

Model calibration is facilitated by a parameter estimation program (Halford, 1992). The parameter estimation process is initialized by using the model to establish the initial differences between simulated and measured water levels. These differences, or residuals, are then minimized by the parameter estimation program. To implement parameter estimation, the sensitivity coefficients (the derivatives of simulated water-level change with respect to parameter change) are calculated by the influence coefficient method using the initial model results (Yeh, 1986). Each parameter is charged a small amount and MODFLOW is used to compute new water levels for each perturbed parameter. The current arrays of sensitivity coefficients and residuals are used by a quasi-Newton procedure (Gill and others, 1981, p. 137) to compute the parameter changes that should improve the model. The model is updated to reflect the latest parameter estimates and a new set of residuals is calculated. The entire process of changing a parameter in the model, calculating new residuals, and computing a new value for the parameter is continued iteratively until model error or model-error change is reduced to a specified level or until a specified number of iterations are made (Halford, 1992).

A uniform effective recharge rate across the entire study area was estimated for each synoptic-survey period, so a total of four parameters was estimated. Although the evapotranspiration rate from the forested area between building 1100 and Lake Druid is of interest to this study, the effective recharge rate and associated evapotranspiration rate from the forested area could not be estimated independently. An initial recharge rate of 10 in/yr was used for all four synoptic-survey periods. Final parameter estimates were not sensitive to the initial parameter estimates.

The minimum, maximum, average, and RMS errors of the calibrated model were -0.58, 0.56, -0.04, and 0.20 ft, respectively. A more detailed listing of the error statistics by synoptic-survey period is provided in table 2. The greater number of water-level measurements available during synoptic-survey period 4 did not overly bias model calibration toward that period (table 2). The water-level residuals did not exhibit any apparent trend across the study area during any of the synoptic-survey periods. Simulated potentiometric surfaces and water-level residuals are shown for synoptic-survey period 4 alone (fig. 7) because the distribution of residuals was similar in all periods.

Table 2. Water-level error statistics and effective recharge rate estimates from calibrated Orlando Naval Training Center model by synoptic-survey period

[Minimum, maximum, average, and RMSE are in feet; n, number of samples. All effective recharge rates are in inches per year]

Synoptic-survey period	Date	n	Minimum	Maximum	Average	N′	RMSE
1	October 25, 1995	16	-0.58	0.56	-0.06	24.4	0.34
2	March 1, 1996	12	-.33	.27	-.02	17.2	.17
3	July 29, 1996	22	-.24	.30	-.03	18.8	.15
4	January 31, 1997	30	-.29	.29	-.04	14.1	.13
	Summary	80	-.58	.56	-.04		.20

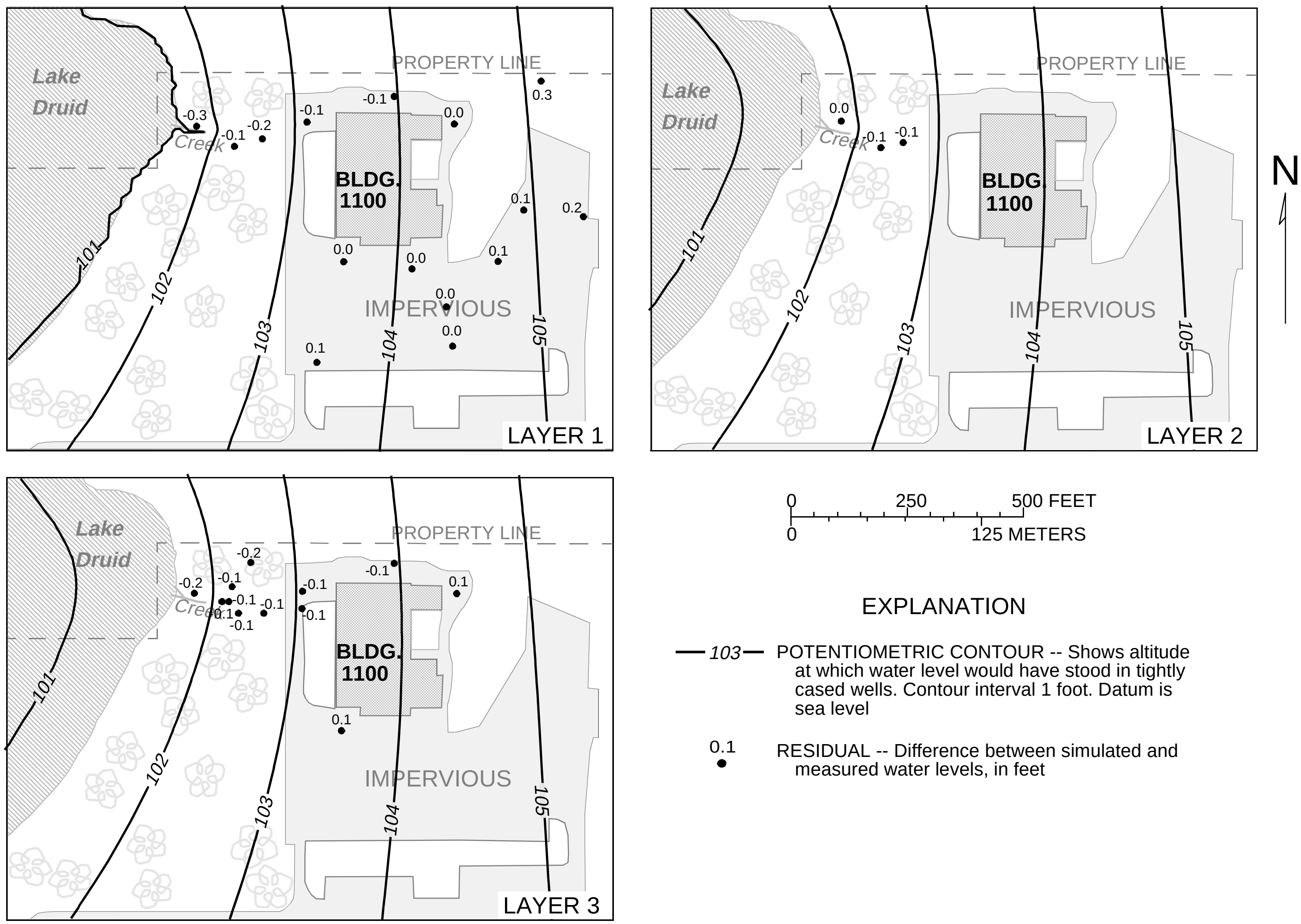

Figure 7. Simulated potentiometric surfaces of the upper zone (layers 1 and 2) and the lower zone (layer 3) on January 31, 1997.

Simulated water levels for the four synoptic-survey periods approximated the measured levels throughout the study area (fig. 8). The areal variation in observed water levels across the study area was about 7 ft. The measured temporal water-level range (101.7 to 107.6 ft above sea level) is similar to the simulated water-level range (101.4 to 108.2 ft above sea level) in area C. The water-level residuals were normally distributed and 80 percent of the simulated water levels were within 0.25 ft of the measured water levels.

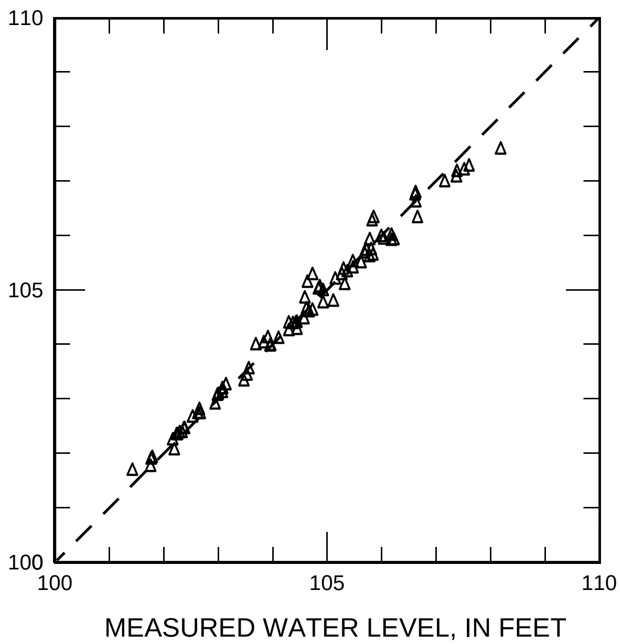

Figure 8. Comparison of simulated to measured water levels for the calibrated model.

Sensitivity Analysis

To determine how the parameters affected simulation results, each estimated effective recharge rate was varied independently from 0.5 to 2 times its calibrated value. This range was greater than the uncertainties associated with the parameters, but provided a more complete perspective on parameter sensitivity. Model sensitivity was described in terms of weighted RMS error. The sensitivity of the model to changing one parameter while all others are held at their calibrated values is shown in figure 9. Model error was determined to be most sensitive to changes in the October 25, 1995, effective recharge rate estimate, and least sensitive to changes in the January 31, 1997, effective recharge rate estimate (fig. 9). These dates correspond to the largest (24 in/yr) and smallest (14 in/yr) effective recharge rate estimates, respectively.

Estimation of the Average Recharge Rate

Instead of conceptualizing the effective recharge rates estimated for each synoptic survey period as volumetric rates, the effective recharge rates can be thought of as measures of the energy release rate or discharge from the flow system during each period. Rising or declining water levels represent increasing or decreasing rates of discharge from the surficial aquifer system. Discharge rates increase even as the surficial aquifer system is recharged, analogous to the increase in discharge from a leaky bucket as it is filled.

The relation between water level and effective recharge rate can be used to estimate the average recharge rate. Water levels are more sensitive to changes in the effective recharge rate and are more variable towards the ground-water divide and away from Lake Druid. Water levels in well 14-04A (fig. 1) were used to monitor the effective recharge rate because the well was closer to the ground-water divide than most of the other observation wells. The effective recharge rates for periods 1, 2, 3, and 4 were regressed against the water levels in well 14-04A during the respective periods (fig. 10). The effective recharge rates were correlated with the water levels (r^2=0.99), and the relation could be described by:

$$N' = 4.11h - 416.8 \tag{8}$$

where

h is the water level in well 14-04A, in feet above sea level.

The average recharge rate (19 in/yr) was estimated by using the average water level for well 14-04A (105.97 ft) in equation 8. The water levels were measured monthly from June 27, 1995, to January 31, 1997.

Evapotranspiration from Area C

The viability of phytoremediation with terrestial plants at area C is dependent on the rate of evapotranspiration from area C. If the evapotranspiration rate across the phytoremediated area is high enough, the contaminant plume will be captured by the plants and will cease to move further downgradient. Although the

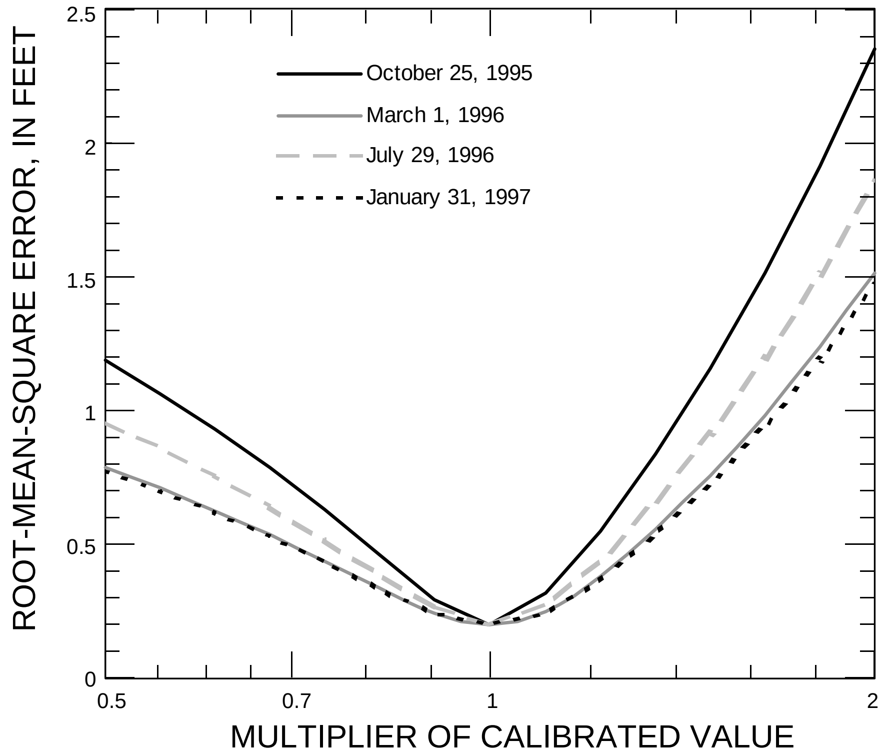

Figure 9. Model sensitivity to independent changes in selected effective recharge rates.

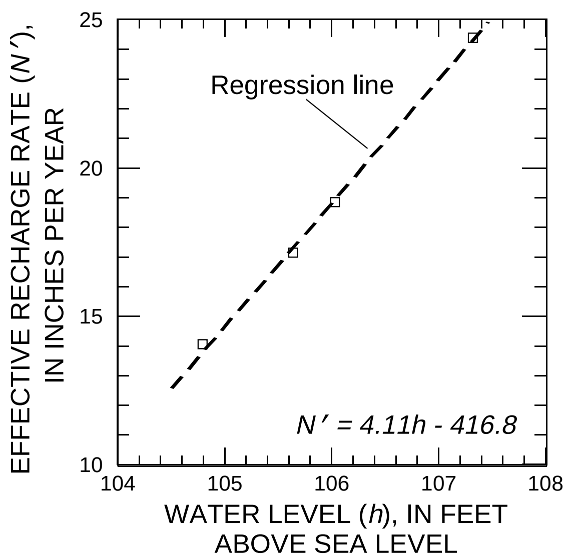

Figure 10. Relation between water levels measured in well 14-04A and the effective recharge rate estimates.

evapotranspiration rate from area C could not be estimated independently during model calibration, the maximum rate of evapotranspiration can be estimated.

Plants control evapotranspiration by restricting the amount of water available for evaporation. Consequently, the evapotranspiration rate from areally extensive vegetated surfaces (ET_{TREE}) should be less than the free water surface evaporation rate (ET_{FWS}). The simplest models of evapotranspiration treat plants as passive wicks (Morton, 1984); the water available for evapotranspiration is limited by the soil moisture content (Morton, 1984). More realistic models of evapotranspiration account for stomatal responses to external environmental factors (Monteith, 1965) that restrict water loss as the vapor pressure deficit increases.

Evapotranspiration from the ground-water flow system was simulated by applying a different effective recharge rate to the forested area between building 1100 and Lake Druid (N'_{TREE}). During periods of ground-water recession, the effective recharge rate equals the combined release of water from storage and evapotranspirative losses (eq. 5). The greatest possible effects of evapotranspirative losses from the forested area would occur if evapotranspiration from the trees is equal to the free water surface evaporation rate (ET_{FWS}) and assuming that water levels in the forested area do not decline (no water was released from storage).

Although N'_{TREE} was assumed to be equal to N' during model calibration, estimates of N'_{TREE} affect the effective recharge rate estimate throughout the remainder of the study area. The effect of evapotranspiration from area C was investigated by setting N'_{TREE} equal to ET_{FWS} for each synoptic survey and estimating N' over the remainder of the study area. In the alternative model, N'_{TREE} was assumed to be a ground-water sink during periods of recession because no water infiltrated from land surface and evapotranspirative losses were assumed to be greater than the rate of water released from storage. N'_{TREE} was simulated as a much stronger sink than what might actually occur so the results of the alternative model and the calibrated model would bracket field conditions.

Effective recharge rates for the remainder of the study area (N') were increased to compensate for the losses from N'_{TREE}. Estimates of N' for the alternative model did not differ greatly from the calibrated model and ranged from 3 to 6 percent more than N' estimates for the calibrated model (table 3). The increases in N' were proportional to the estimates of N'_{TREE}; the greatest increase in N' was 1.2 in/yr for the July 29, 1996 period, when ET_{FWS} was equal to 66 in/yr (table 3). The RMS error of the alternative model (0.25 ft) was slightly greater than the RMS error of the calibrated model (0.20 ft). The increase in RMS error suggests that N'_{TREE} was less than ET_{FWS} and, as a consequence, ET_{TREE} probably was less than ET_{FWS}.

ASSESSMENT OF THE POTENTIAL EFFECTS OF PHYTOREMEDIATION ON GROUND-WATER FLOW AND MOVEMENT OF CONTAMINANTS

The analysis of ground-water flow and potential movement of contaminants within the surficial aquifer system was addressed using the calibrated model driven by the average recharge rate of 19 in/yr. N'_{TREE} was assumed to be -30 in/yr which is about half of the maximum ET_{FWS} during the summer. These recharge

Table 3. Water-level error statistics and effective recharge rate estimates from an alternative model by synoptic-survey period

[Minimum, maximum, average, and RMSE are in feet; n, number of samples. All effective recharge rates are in inches per year]

Synoptic-survey period	Date	n	Minimum	Maximum	Average	N'_{TREE}	N'	RMSE
1	October 25, 1995	16	-0.61	0.60	-0.07	-41	25.1	0.37
2	March 1, 1996	12	-.39	.33	-.03	-50	18.0	.22
3	July 29, 1996	22	-.31	.40	-.05	-66	20.0	.23
4	January 31, 1997	30	-.32	.33	-.05	-30	14.7	.18
	Summary	80	-.61	.60	-.05			.25

conditions were used to simulate the advective movement of conservative contaminants through area C with MODPATH (Pollock, 1994) for all remedial alternatives.

Particle traveltimes and advective displacement rates are proportional to the effective porosity estimates. Effective porosity differs from total porosity in that only the interconnected pore spaces are included. If effective porosity estimates are doubled, ground-water velocities will be halved and traveltimes will double. An effective porosity of 30 percent was assumed for all particle-tracking simulations.

Existing Conditions

The water table configuration in area C is strongly influenced by Lake Druid and to a lesser extent by the unnamed creek (fig. 11) under existing conditions. The simulated lateral flow direction from building 1100 in the upper zone and lower zone of the surficial aquifer system is radially convergent on Lake Druid. Simulated lateral ground-water movement from building 1100 toward Lake Druid is 3 to 4 times faster in the lower zone (0.7 ft/d) than the upper zone (0.2 ft/d) (fig. 12).

The simulated vertical movement of ground water is downward from beneath building 1100 to the edge of the forested area and upward beneath the forested area and Lake Druid (fig. 12). The reversal in vertical movement from downward to upward is the result of the simulated evapotranspirative losses. The flow paths terminate in the unnamed creek and Lake Druid because these features receive the majority of the ground-water discharge.

The rates of ground-water movement beneath building 1100 were estimated with a small control volume that extended from Lake Druid to the eastern edge of building 1100 (fig. 11). The control volume was limited to an areal extent that was slightly larger than the contaminant plume, to better compare the rate of extraction by the remediation alternatives to the rate of ground water flow through the contaminant plume. The northern and southern boundaries of the control volume were based on flowpaths under existing conditions, so that the majority of flow would enter the control volume along the eastern boundary.

About 25.4 gal/min pass through the control volume under existing conditions (fig. 13) and 20.4 gal/min (80 percent of the flow) discharge to Lake Druid. The remainder of the flow was removed from the control volume by evapotranspirative losses (3.7 gal/min) and deep leakage (1.3 gal/min). The majority of the ground-water flow that passes through the control volume moves through the lower zone because the transmissivity of the lower zone (1,600 ft^2/d) is much greater than the transmissivity of the upper zone (200 ft^2/d).

Under existing conditions, phytoremediation alone at area C at Orlando NTC cannot stop the discharge of contaminants into Lake Druid because the evapotranspirative losses (3.7 gal/min) are small relative to the ground-water discharge to Lake Druid (20.4 gal/min). Phytoremediation still could be a viable remedial action if the contaminated ground water can be redirected to flow through the rooting zone of selected aquatic and terrestrial plants. Contaminant concentrations can be reduced in the rooting zone by dehalogenase enzymes released into the soil (Schnoor and others, 1995).

Effects of a Drainage Ditch

Some of the contaminated ground-water discharge to Lake Druid could be intercepted if a drainage ditch were dug below the elevation of Lake Druid (fig. 14). Phytoremediation would occur within the rooting zone of aquatic vegetation planted in the drainage ditch. If the water surface of the drainage ditch were maintained at the elevation of Lake Druid, the drainage ditch would significantly alter the simulated water-table configuration in area C (fig. 14). The drainage ditch would intercept flow from 0 to about 40 ft below the water table but would not intercept all the water that flows under building 1100 (fig. 15). Ground water that originates in the lower 20 ft of the lower zone beneath building 1100 would continue to discharge to Lake Druid. The drainage ditch would double ground-water velocity beneath building 1100 in the upper zone (from 0.2 to 0.4 ft/d) but would not change the ground-water velocity in the lower zone (0.7 ft/d).

The addition of a drainage ditch would increase the rate of ground-water movement through the control volume from 25.4 to 38.4 gal/min (fig. 16). The drainage ditch would capture 17.3 gal/min (45 percent) while 16.1 gal/min (42 percent) would discharge to Lake Druid. Ground-water discharge losses to evapotranspiration (3.7 gal/min) and deep leakage (1.3 gal/min) were assumed to be unaffected by the drainage ditch.

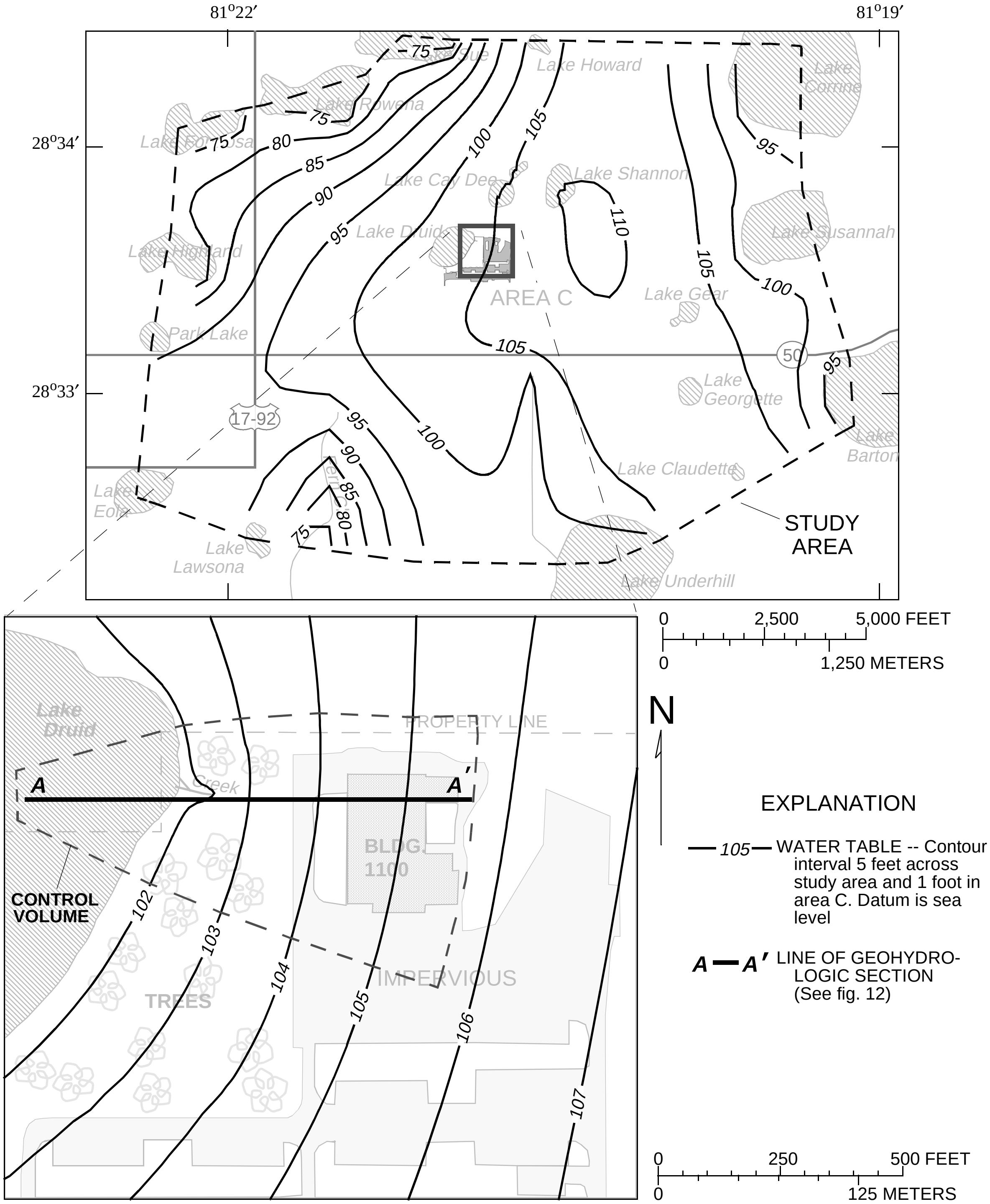

Figure 11. Simulated water table of the upper zone (layer 1) with a recharge rate of 19 inches per year and an effective recharge rate to the forested area (N'_{TREE}) of -30 inches per year.

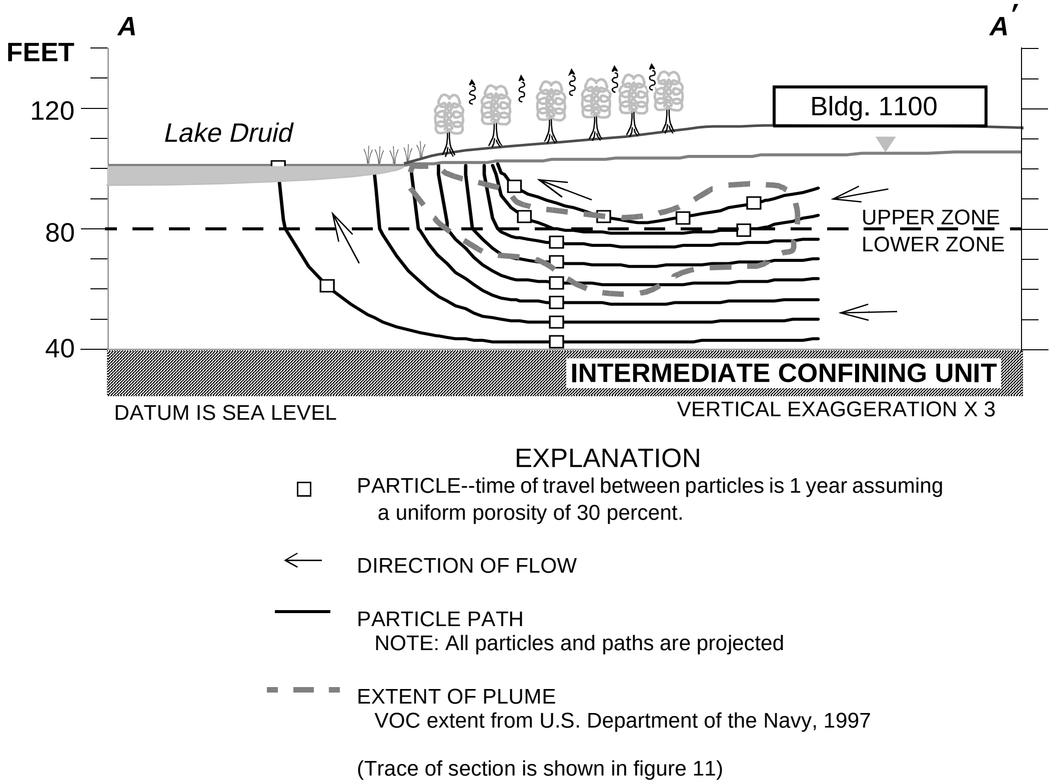

Figure 12. Flow paths from a transect through the surficial aquifer system between building 1100 and Lake Druid that were simulated with a recharge rate of 19 inches per year and an effective recharge rate to the forested area (N'_{TREE}) of -30 inches per year.

Reduction in contaminant concentrations in the drainage ditch by phytoremediation is dependent on the residence time of the water in the drainage ditch. Longer residence times allow the dehalogenase enzymes in the rooting zone more time to react and reduce the contaminant concentrations (Schnoor and others, 1995). If the wetted volume of the ditch is 15,000 ft^3 (300 ft by 10 ft by 5 ft), ground-water discharge would take about 4.5 days to displace one ditch volume. The residence time could be increased by making the drainage ditch wider or increasing the stage in the drainage ditch. Increasing the stage would reduce the flow rate into the drainage ditch, but it also would reduce the volume of the contaminant plume intercepted.

MODEL LIMITATIONS

The flow model addresses questions about the advective movement of contaminants through the surficial aquifer system beneath area C of the Orlando NTC fairly well. However, this model, or any other model, is limited by simplification of the conceptual model, discretization effects, difficulty in obtaining sufficient measurements to account for all of the spatial variation in hydraulic properties throughout the model area, and limitations in the accuracy of land surface altitude measurements.

The conceptual model has been simplified by assuming that remedial alternatives can be assessed with a steady-state ground-water flow model. However, available data indicate that between June 1995 and January 1997, water-levels fluctuated seasonally as much as 3 ft, and the surficial aquifer system infrequently approached, at best, a quasi-steady state condition. The steady-state model was adequate to compare the relative effects of the two remediation schemes on the advective movement of dissolved constituents, but was not adequate to estimate the exact advective movement of contaminants associated with any one remediation scheme.

Contaminant concentrations cannot be predicted with this model because only the advective movement of ground water and contaminants are simulated.

Prediction of contaminant concentrations requires a solute-transport model that accounts for diffusion and dispersion in addition to the advective transport of contaminants.

The model of a heterogeneous aquifer system was simplified further by the methods used to describe the spatial variability of the hydraulic conductivity distributions. The lateral and vertical hydraulic conductivity distributions of the upper zone and lower zone of the surficial aquifer system were assumed to have uniform values. The lack of sufficient measurements to account for the spatial variation in hydraulic properties throughout the model area necessitated these simplifications. Simplifying the model to this degree does not invalidate the model results, but does mean that model results should be interpreted at scales larger than the representative elemental volume of hydraulic conductivity.

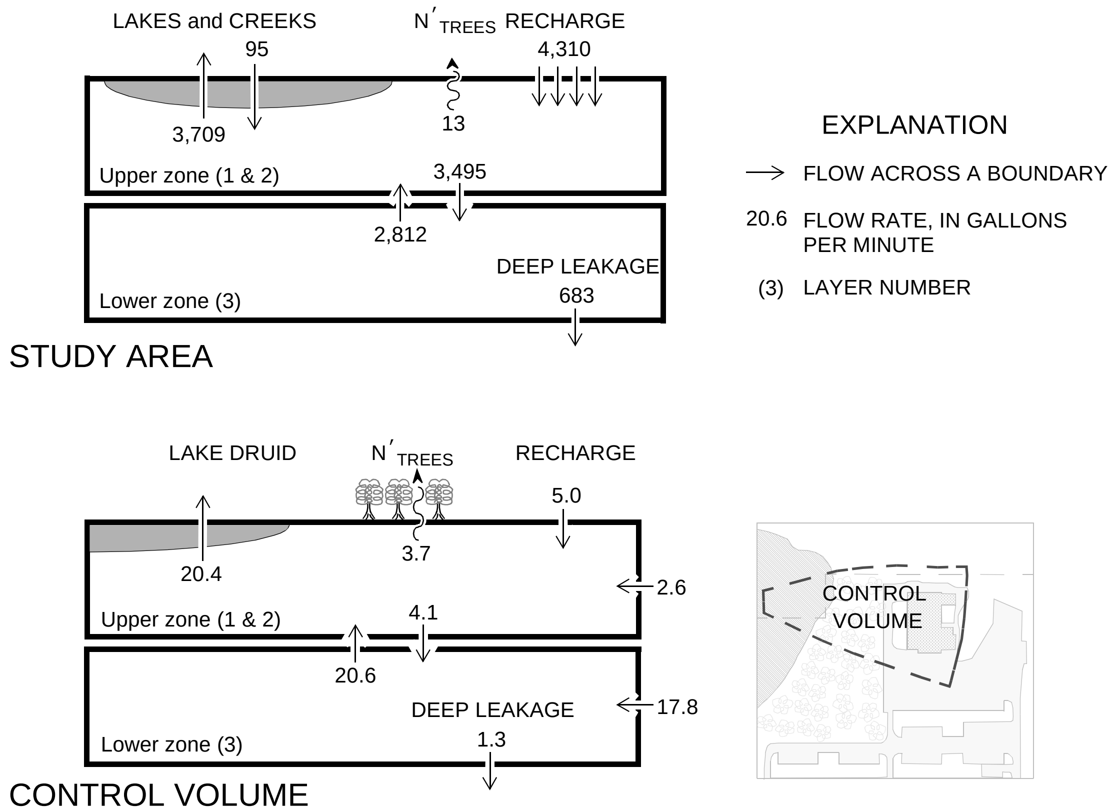

Figure 13. Simulated volumetric flow budget with a recharge rate of 19 inches per year and an effective recharge rate to the forested area (N'_{TREE}) of -30 inches per year.

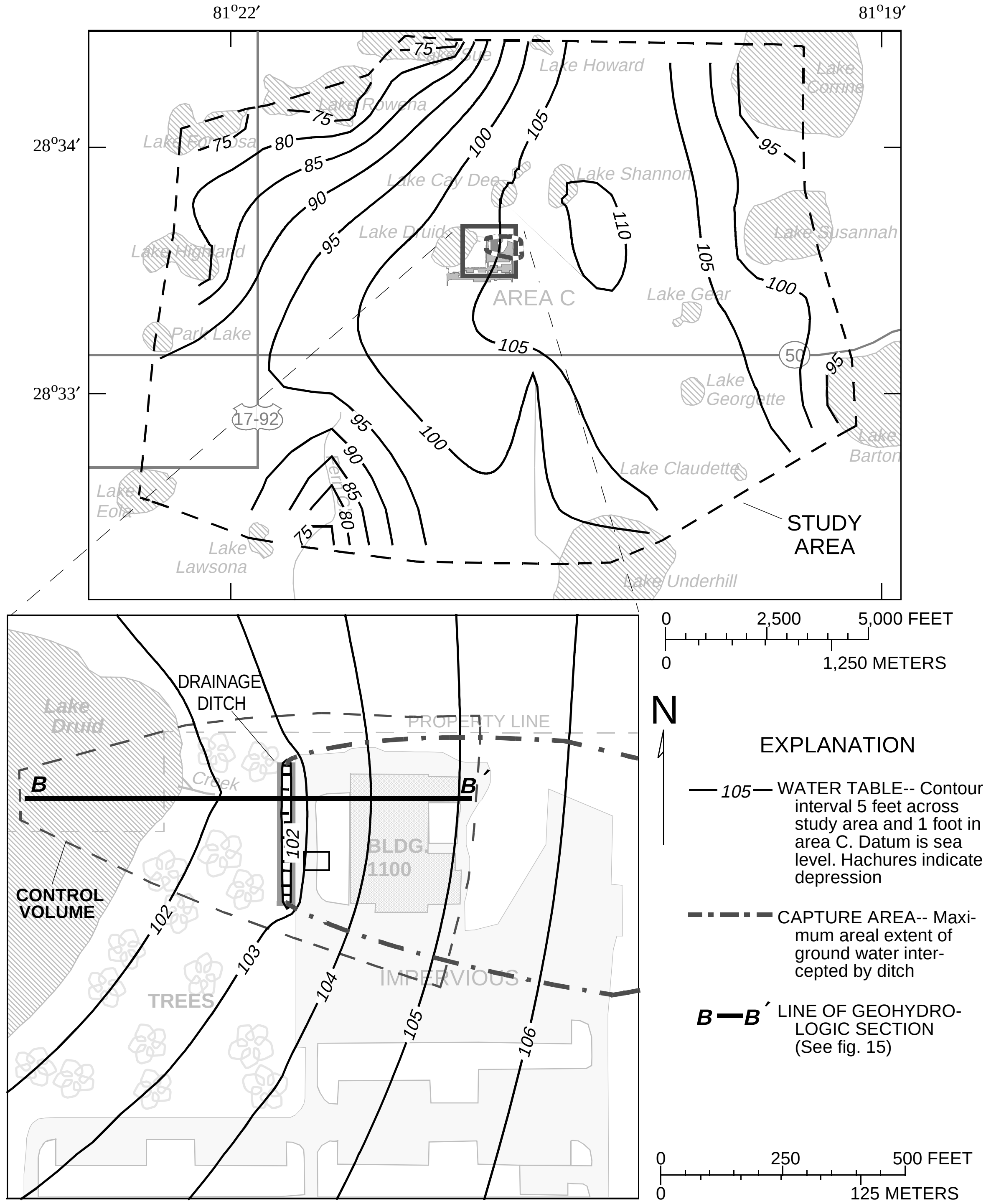

Figure 14. Simulated water table of the upper-zone (layer 1) with a drainage ditch installed, a recharge rate of 19 inches per year, and an effective recharge rate to the forested area (N'_{TREE}) of -30 inches per year.

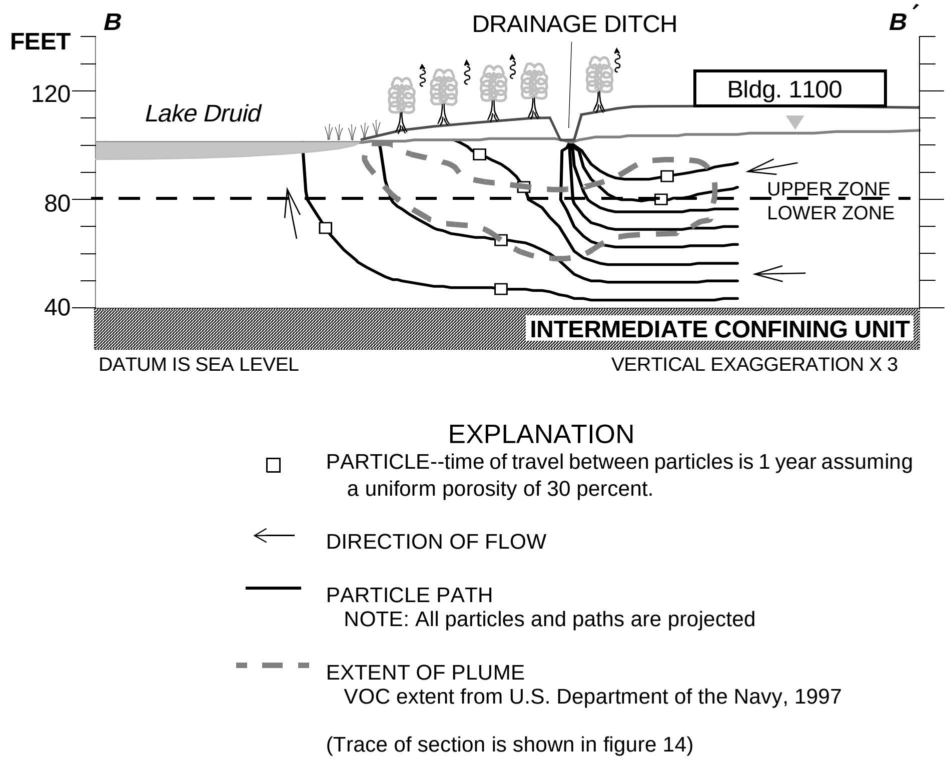

Figure 15. Flowpaths from a transect through the surficial aquifer system with a drainage ditch installed between building 1100 and Lake Druid that were simulated with a recharge rate of 19 inches per year and an effective recharge rate to the forested area (N'_{TREE}) of -30 inches per year.

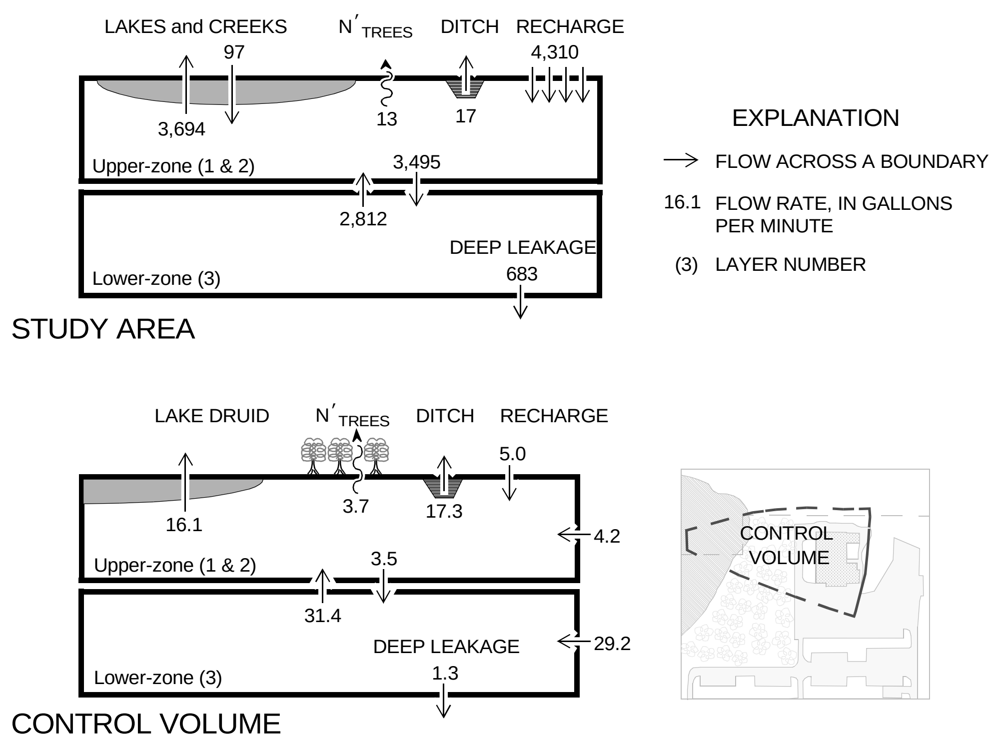

Figure 16. Simulated volumetric flow budget with a drainage ditch installed, a recharge rate of 19 inches per year, and an effective recharge rate to the forested area (N'_{TREE}) of -30 inches per year.

SUMMARY

As part of the Installation Restoration Program, the Orlando NTC is considering phytoremediation as a long-term, remedial-action alternative to control the discharge of contaminated ground water to Lake Druid and remediate the ground water. Phytoremediation affects ground-water flow and reduces chlorinated contaminant concentrations through the selective use of plants with high transpiration rates, high dehalogenase activity, and a tolerance to elevated contaminant concentrations. The evaluation of phytoremediation as a remedial action requires a quantifiable understanding of how the ground-water flow system responds to current conditions and how the system will respond to changes induced by phytoremediation schemes. Numerical simulation provides the most tractable method of achieving this level of understanding.

The surficial aquifer system and underlying Hawthorn Group include sediments of Holocene to Pliocene age. The surficial aquifer system consists of well-sorted, fine-grained sand from 0 to 63 ft below land surface, and the Hawthorn Group consists of marine clays and discontinuous limestone stringers.

Precipitation (50 in/yr) and irrigation (7 in/yr) are the sources of water in the water budget. Surface-water discharge (20 in/yr) and evapotranspiration (34 in/yr) are the major losses from the water budget in the study area. Most of the surface-water discharge is routed into lakes, and the stages of these lakes are regulated by drainage wells that discharge from the lakes to the Upper Floridan aquifer. Deep leakage across the Hawthorn Group accounts for the remaining 3 in/yr.

Ground-water flow through the surficial aquifer system was simulated with a three-layer, finite-difference model that extended vertically from the water table to the top of the intermediate confining unit. The surficial aquifer system was subdivided into two local hydrogeologic units: the upper zone and the lower zone. The hydraulic characteristics of the surficial aquifer system were estimated with an aquifer test that was conducted between Lake Druid and building 1100. The lateral and vertical hydraulic conductivity of the upper zone (layers 1 and 2) and the lower zone (layer 3) were the primary hydraulic characteristics to be determined by this test.

The distribution and altitude of surface-water features control the direction and rate of flow in the surficial aquifer system. The distribution of surface-water features was determined from aerial photographs and plans of the Orlando NTC. All lateral model boundaries in each layer were assumed to be no-flow boundaries that either coincided with surface-water features or were parallel to ground-water flow paths.

The Orlando NTC model was calibrated as a series of independent steady-state flow systems which approximated the transient system as a series of “snapshot” images. The Orlando NTC model was calibrated to 80 water-level measurements from 30 wells during four synoptic surveys that were conducted between October 24, 1995, and January 31, 1997. Model calibration was facilitated by a parameter estimation program that estimated the effective recharge rates and the vertical hydraulic conductivities.

The minimum, maximum, average, and RMS errors of the calibrated model were -0.58, 0.56, -0.04, and 0.20 ft, respectively, and the residuals did not exhibit any apparent trend across the study area. Simulated water levels for the four synoptic-survey periods approximated the measured levels throughout the approximately 7 ft range observed in the study area. Model error was determined to be most sensitive to changes in the October 25, 1995, estimate.

The effect of evapotranspiration from the forested area was investigated with an alternative model. In the alternative model, the effective recharge rate from the forested area in area C was assumed to equal free-water-surface evaporation rates for each synoptic survey and the effective recharge rate over the remainder of the study area was estimated. Effective recharge rates for the remainder of the study area were increased to compensate for evapotranspirative losses from the forested area, but did not differ greatly from the calibrated model. The RMS error of the alternative model (0.25 ft) was slightly greater than the RMS error of the calibrated model (0.20 ft). The increase in RMS error indicates that evapotranspiration from the forested area occurs at rates less than free-water-surface evaporation rates.

The analysis of ground-water flow and potential movement of contaminants within the surficial aquifer system was accomplished by using the calibrated model driven by the average recharge rate of 19 in/yr. The effective recharge rate from the forested area in area C was assumed to be a net loss rate of 30 in/yr which is about half of the maximum free-water-surface evaporation rate during the summer. These recharge conditions were used to simulate the advective movement of conservative contaminants through area C with MODPATH for the two remedial alternatives.

Under existing conditions, phytoremediation alone at area C at Orlando NTC cannot stop the discharge of contaminants into Lake Druid because within the analyzed control volume the evapotranspirative losses (3.7 gal/min) are small relative to the ground-water discharge to Lake Druid (20.4 gal/min). Phytoremediation still could be a viable remedial action if the contaminated ground water can be redirected to flow through the rooting zone of selected aquatic and terrestrial plants by means of a drainage ditch.

A drainage ditch would intercept flow from 0 to about 40 ft below the water table but would not intercept all of the water that flows under building 1100. The addition of a drainage ditch would increase the rate of ground-water movement under building 1100 by about 50 percent. The residence time of the water in a 15,000 ft^3 (300 ft by 10 ft by 5 ft) drainage ditch would be about 4.5 days. The residence time could be increased by making the drainage ditch wider or increasing the stage in the drainage ditch. Increasing the stage would reduce the flow rate into the drainage ditch, but also would reduce the volume of the contaminant plume intercepted.

The flow model addresses questions about the advective movement of contaminants through the surficial aquifer system beneath area C of the Orlando NTC fairly well. However, this model, or any other model, is limited by simplification of the conceptual model, discretization effects, difficulty in obtaining sufficient measurements to account for all of the spatial variation in hydraulic properties throughout the model area, and limitations in the accuracy of land surface altitude measurements.

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