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Predicting the Effect of Deep-Rooted Hybrid Poplars on the Groundwater Flow System at a Large-Scale Phytoremediation Site

J. J. Quinn, M. C. Negri, R. R. Hinchman, L. P. Moos, and J. B. Wozniak
Argonne National Laboratory, 9700 S. Cass Ave., Argonne, IL 60439

E. G. Gatliff
Applied Natural Sciences, Inc., 4129 Tonya Trail, Hamilton, OH 45011

* Corresponding author: John J. Quinn, Argonne National Laboratory, Environmental Assessment Division, Building 900, Argonne, IL 60439; Phone: (630) 252-5357; Fax: (630) 252-3611; E-mail: quinnj@anl.gov

ABSTRACT

Estimating the effect of phreatophytes on the groundwater flow field is critical in the design or evaluation of a phytoremediation system. Complex hydrogeological conditions and the transient water use rates of trees require the application of numerical modeling to address such issues as hydraulic containment, seasonality, and system design.

In 1999, 809 hybrid poplars and willows were planted to phytoremediate the 317 and 319 Areas of Argonne National Laboratory near Chicago, Illinois. Contaminants of concern are volatile organic compounds and tritium. The site hydrogeology is a complex framework of glacial tills interlaced with sands, gravels, and silts of varying character, thickness, and lateral extent. A total of 420 poplars were installed using a technology to direct the roots through a 25-ft (8-m)-thick till to a contaminated aquifer.

Numerical modeling was used to simulate the effect of the deep-rooted poplars on this aquifer of concern. Initially, the best estimates of input parameters and boundary conditions were determined to provide a suitable match to historical transient groundwater flow conditions. The model was applied to calculate the future effect of the developing deep-rooted poplars over a 6 year period. The first 3 years represent the development period of the trees. In the fourth year, canopy closure is expected to occur; modeling continues through the first 3 years of the mature plantation. Monthly

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estimates of water use by the trees are incorporated. The modeling suggested that the mature trees in the plantation design will provide a large degree of containment of groundwater from the upgradient source areas, despite the seasonal nature of the trees' water consumption. The results indicate the likely areas where seasonal dewatering of the aquifer may limit the availability of water for the trees. The modeling also provided estimates of the residence time of groundwater in the geochemically altered rhizosphere of the plantation.

KEY WORDS: phytoremediation, poplars, groundwater, numerical modeling, hydraulic control.

I. INTRODUCTION

Phytoremediation offers the potential for remediating groundwater and soil with the following benefits: reasonably low installation cost, remediation within a suitable time frame, low operation and maintenance costs, aesthetic value, low ecological impact, and public approval.

In the last decade, hybrid poplars have been studied to determine their ability to remove or destroy contaminants such as volatile organic compounds (VOCs). Although most of this research has been laboratory and greenhouse based, pilot- and full-scale field applications are currently underway. Researchers have found that trichloroethene (TCE), for example, is transpired unaltered through the leaves of a plant, or it is sequestered in the plant and metabolized or mineralized (Gordon *et al.,* 1997; Compton *et al.,* 1998; Newman *et al.,* 1997, 1999). TCE is also degraded through dechlorination in the root zone through the microbial activity of bacteria and fungi growing symbiotically along the roots (Brigmon, Anderson, and Fliermans, 1999; Anderson and Walton, 1995; Anderson, Guthrie, and Walton, 1993).

Other advantages of using poplars in certain phytoremediation systems include fast growth rates and the use of vast amounts of water. Poplars can achieve growth rates as high as 10 to 16 ft/yr (3 to 5 m/yr) (Chappell, 1998). While they can transpire tremendous amounts of water (Nyer and Gatliff, 1996), the rate varies, depending on climatic factors and tree density (Chappell 1998). The capability of the trees to lower the water table indicates that they have the potential to provide groundwater containment (Nyer and Gatliff, 1996; Compton *et al.,* 1998; Newman *et al.,* 1999).

Poplars are phreatophytes. These trees extend roots into the capillary fringe and can survive periods of being within the saturated zone of an aquifer as water levels fluctuate. Because phreatophytes send roots into both the vadose and phreatic zones, these plants have the potential to remediate soil, groundwater, and saturated soil media. In terms of total root length, a stand of poplars may have as much as 75,000 miles per acre (300,000 km per hectare) (Gordon *et al.,* 1997). The subsurface may consist of units of widely varying lateral or vertical extent, with gradational or sharp transitions in permeability. The fibrous nature of the roots allows the trees to penetrate and remediate both the relatively fast-flowing pore spaces and the less permeable zones. Fundamentally, this distinguishes phytoremediation from extraction wells, which remove water mainly from the most permeable aquifer media (Gatliff, 1994).

Groundwater modeling was performed in support of a large-scale phytoremediation project. The modeling focused on predicting the seasonal containment capability of a deep-rooted hybrid poplar phytoremediation system installed at Argonne National Laboratory (ANL) in June 1999. The study included a detailed analysis of subsurface hydrogeological conditions, seasonal hydrologic changes, and the seasonality of the phytoremediation system. Animated visualizations were generated to facilitate the conceptual understanding of the results. The large-scale program and the complexity of the subsurface make the site challenging in terms of its history, hydrogeology, seasonality, and implementation of the remedial technology. The methods used in this investigation may be applied to designing future phytoremediation plantations focused on groundwater.

II. STUDY AREA

Past waste disposal practices at the 317 and 319 Areas at ANL near Chicago, Illinois (Figure 1), have resulted in the contamination of groundwater by VOCs and tritium. Historical groundwater VOC concentrations near the 317 French Drain have been in the thousands and ten of thousands of parts per billion. Contaminants are transported off the ANL property at fairly low concentrations and appear at several seeps (Figure 1) in ravines of adjacent forest preserve property. In 1999, a groundwater and soil treatment program was initiated that relies on phytoremediation to (1) provide hydraulic containment in the aquifer of concern, (2) extract and transpire contaminants, (3) incorporate and/or degrade the contaminants in the biomass, and (4) cometabolize VOCs in the root zone. More than 800 trees were planted in the summer of 1999 to achieve these goals. Willows were generally planted at the surface in areas of contaminated soil. Four hundred twenty poplars (HP 510 Androscoggin Poplar and HP 308 Charkowiensis Incrassata) were installed using TreeWell® technology. These trees were planted at a 16-ft (4.9-m) spacing in 2-ft (0.6 m) diameter caisson boreholes lined with plastic sleeves in order to direct the roots to the main contaminated aquifer, exclude shallow groundwater, and optimize deep groundwater removal efficiency. This technology was necessary for implementing the phytoremediation system because of the site's hydrogeological setting. The boreholes were filled with a mixture of topsoil, sand, peat, and manure to promote root growth and tree development.

The study focused on modeling the predicted effect of the engineered deployment of the 420 specially installed poplars (Figure 2) with roots directed to a confined aquifer 25 to 30 ft (8 to 9 m) deep. An additional 389 surficially planted willows and poplars were not included in the analysis because of those trees purpose is to remediate contaminated soil or shallow groundwater that is essentially separated from the aquifer of interest.

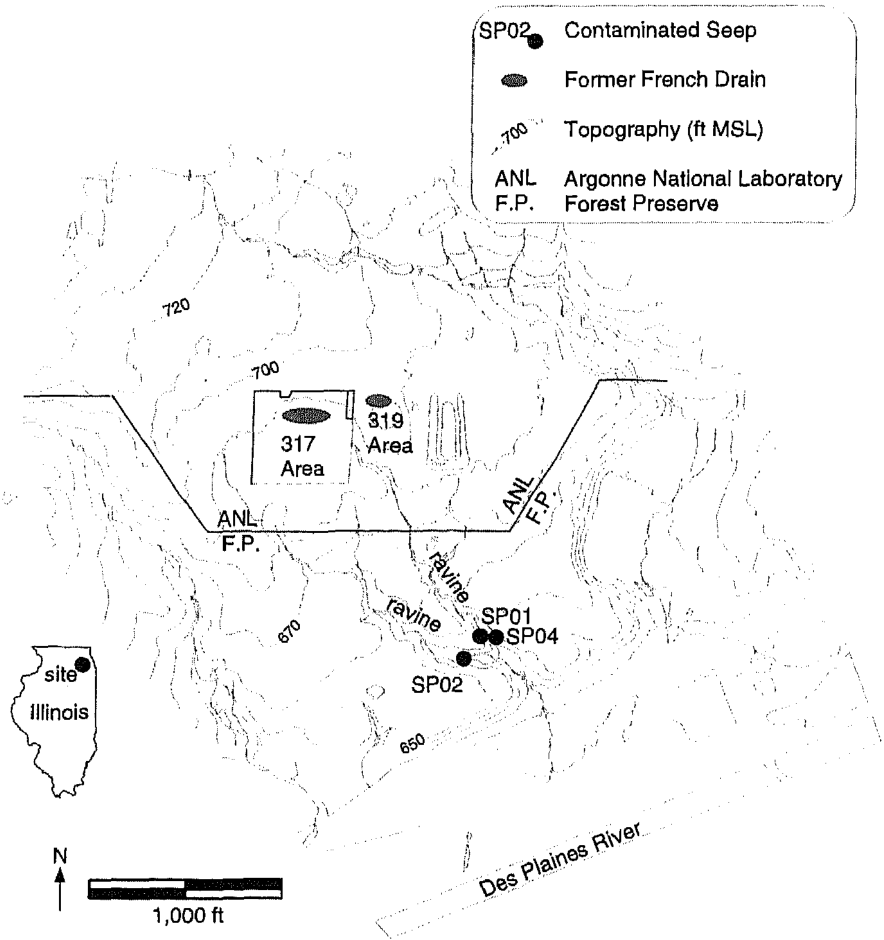

FIGURE 1. Site location and setting.

III. HYDROGEOLOGICAL SETTING AND CONCEPTUAL FLOW MODEL

The subsurface is a complex arrangement of approximately 60 ft (18 m) of glacial geologic deposits over Silurian dolomite bedrock (Figure 3). The glacial sequence is composed of Lemont Drift overlain by the Wadsworth Formation. Both units are dominated by fine-grained, low-permeability till. Permeable zones of varying character and thickness are present in each. These materials range from silty sands, to sandy, clayey gravels, to gravelly sands. In some locations, pure silt is encountered. If deep enough, this silt is saturated and assumed to play an important role in the flow of groundwater in the study area. The permeable zones vary widely in shape, including thin, lenticular, alluvial deposits; thick plugs of possible slump or channel-fill material; interfingerings; and a thick, basal, proglacial sand and gravel. In general, the permeable units are poorly sorted, and many of them may represent slope-induced mass movement, which results in transport and mixing of sediments

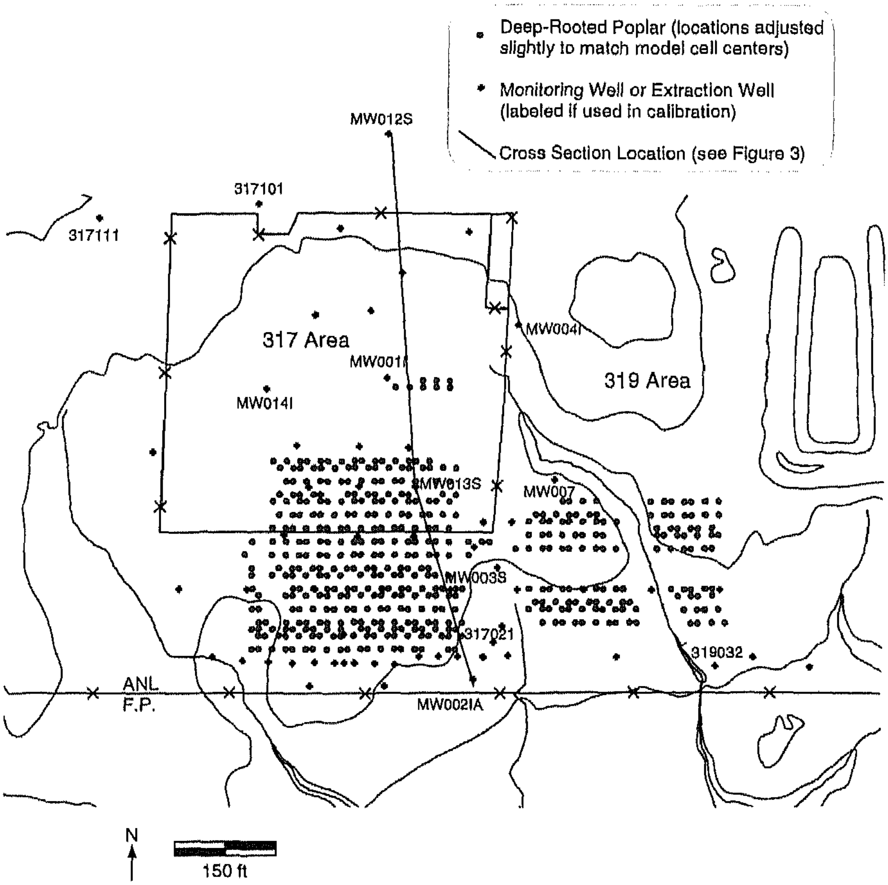

FIGURE 2. Site features and wells used in flow model calibration.

(e.g., Lawson, 1982). The sandy units have thicknesses ranging from less than 1 ft (0.3 m) to roughly 15 ft (4.5 m), and they have limited lateral extent.

The modeling was focused on phytoremediation efforts directed at the site's main contaminated aquifer of the site. Over 75 continuously sampled boreholes (Figure 2) give an indication of the structure of this unit. The depth to the top of this unit ranges from 22 to 28 ft (6.7 to 8.5 m) in the 317 French Drain area, to 22 to 34 ft (6.7 to 10.4 m) at the southern edge of the ANL site. In most locations, the confined aquifer is 3 to 10 ft (1 to 3 m) thick. Because the aquifer's top and bottom surfaces vary spatially, its thickness is variable. The unit has been delineated on the basis of stratigraphy, a southeast trend in hydraulic head data, and, where available, contaminant tracer data. This "aquifer" is best described as consisting of numerous permeable bodies of varying character and geometry that share some similarity in depth and that have some degree of hydraulic connection. The aquifer is not present everywhere, and in some locations the stratigraphic data may not support the presence of an aquifer within a reasonable depth interval (Figure 3).

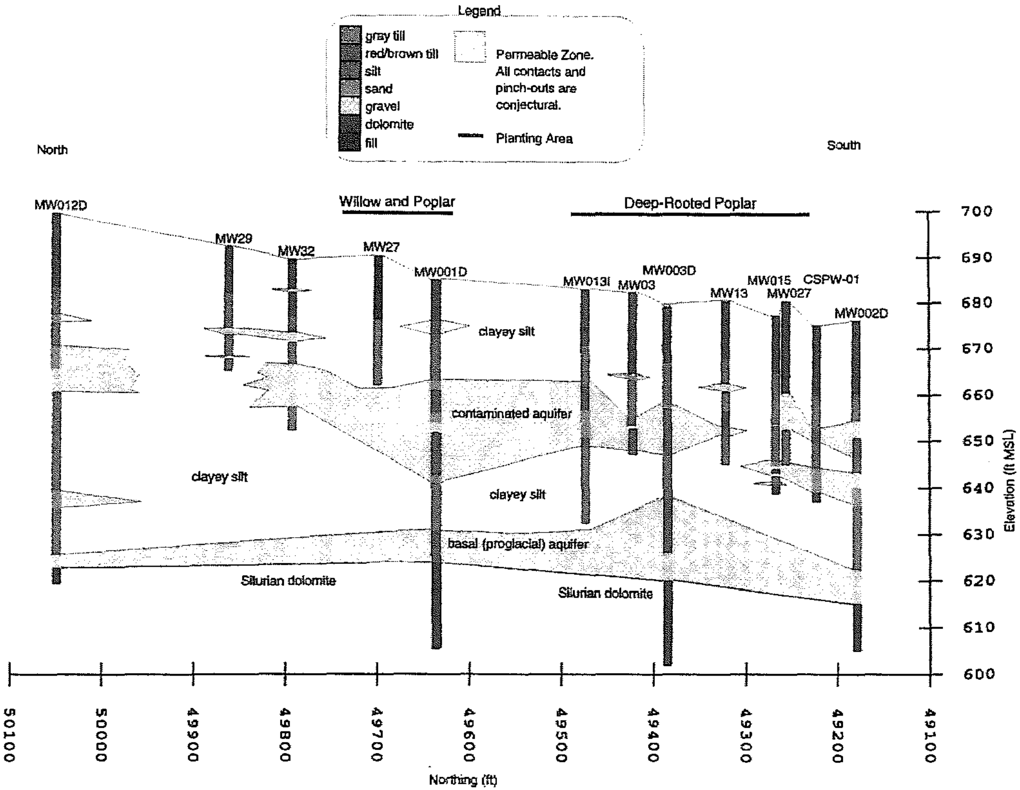

FIGURE 3. North-south cross section showing hydrogeology and planting areas.

On the basis of the southeast trend in water levels, while considering stratigraphic and well construction information, 23 monitoring wells with up to 10 years of seasonal head data were assumed to represent the main aquifer of interest, and were therefore useful in calibrating the transient flow model.

The conceptual model of groundwater flow is as follows. Groundwater in the aquifer is, in general, confined and flows to the southeast. This finding is supported by historical hydraulic head measurements and is in keeping with the notion of regional flow mimicking the topography and flowing toward the Des Plaines River valley to the southeast. The aquifer of interest is assumed to receive input from upgradient sandy units to the northwest and from infiltration from above. The recharge from infiltration may be localized and originates as seepage from shallower perched aquifers or directly from precipitation and conveyed by fractures within the overlying till. Recharge to the dolomite aquifer beneath ANL has been estimated by Walton (1965), who suggested a value of 3.3 in/yr (8.3 cm/yr). At the site scale, however, surficial recharge to the aquifer of interest is likely to be less than this value because of the predominance of low-permeability till units at the surface.

Hydraulic conductivity estimates are available in the form of a pump test and 12 slug tests. The pump test data indicated a range of 4.4 to 10.5 ft/d (1.6×10^{-3} to 3.7×10^{-3} cm/s), or an average of 8.8 ft/d (3×10^{-3} cm/s). Slug tests in the 12 appropriate monitoring wells indicate values of 0.011 to 170 ft/d (4×10^{-6} to 6×10^{-2} cm/s), with an average of 10.8 ft/d (3.8×10^{-3} cm/s). The slug tests indicate variable permeability across the site, without any trend. The permeability likely varies greatly over extremely short distances.

Most groundwater in the aquifer of interest travels southeast to the forest preserve where it discharges as seepage, either in the deep portions of the ravines or along the base of the main bluff of the Des Plaines valley. The seepage along the bluff is expected to be transient, and it may consist of a combination of localized, flowing seeps and broad, diffuse seepage that is subject to transpiration and evaporation. A minimal amount of this groundwater may recharge the dolomite aquifer.

IV. NATURAL TRANSIENT CONDITIONS

A. Modeling Approach

The U.S. Geological Survey finite-difference code MODFLOW (McDonald and Harbaugh, 1988) was selected because of its capability to address steady-state and transient flow, varying upper and lower aquifer surfaces, and aquifer input and output. MODFLOW has efficient solvers, and the code includes the capability of rewetting model cells that have been dewatered (McDonald *et al.*, 1991). To analyze and display the rate of groundwater movement, MODPATH (Pollock, 1994) was used in combination with MODFLOW flow output.

The extent of the modeling domain (Figure 4) was set reasonably far from the main area of interest (317 and 319 Areas) to minimize the effect of the boundaries on the solution of head near the planting areas. The lateral boundaries, along the northeast and southwest, are no-flow boundaries that were assumed on the basis of regional and local

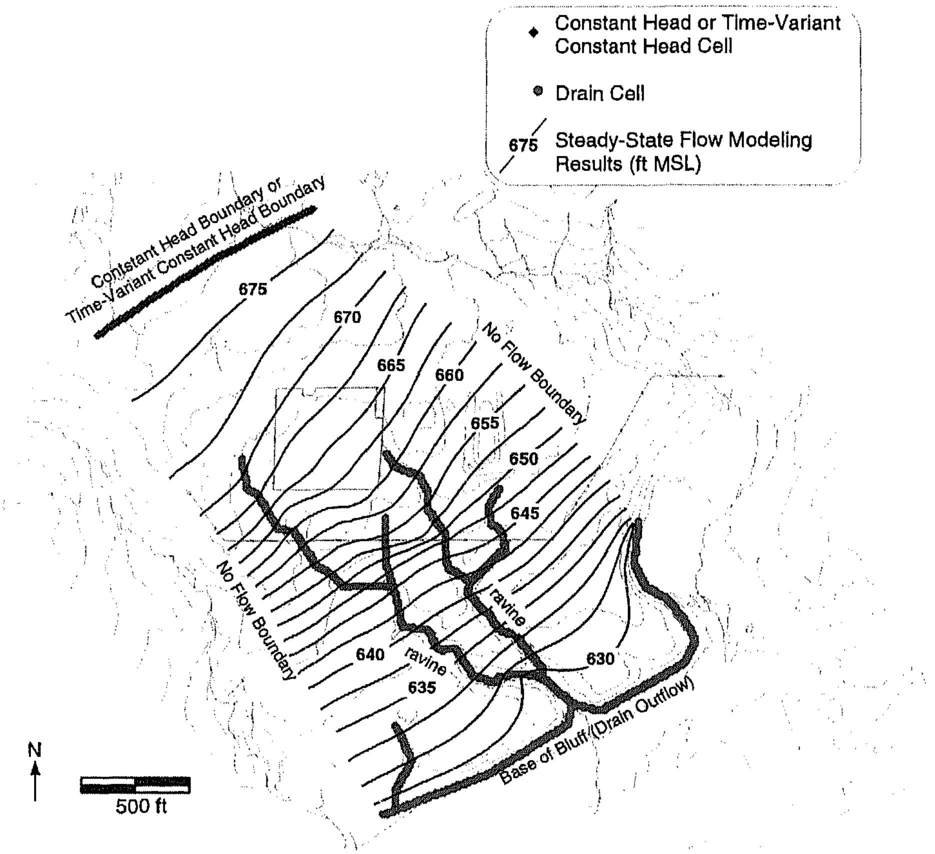

FIGURE 4. Modeling domain, boundary conditions, and steady-state results.

flow to the southeast toward the Des Plaines River valley. The boundaries were set halfway between major ravines, so that the draining effect of the ravines would be equally divided between the area of interest and adjacent watersheds.

The upgradient boundary was set about 600 ft (180 m) northwest of the study area's northwestern monitoring wells. For steady-state runs, this constant head boundary provides flux into the flow system at a distance reasonably far from the main area of interest. For transient runs, the value of the boundary was allowed to vary in time in order to replicate seasonal effects in flux from upgradient. The southern boundary was assumed to be a seepage face near the base of the main bluff of the Des Plaines River valley. Although seepage along the bluff is localized because of drift heterogeneity, the seepage was spread uniformly along the bluff because the large distance between the planting areas and the bluff minimizes the need for exact seepage locations. By assigning drain cells along the bluff, this boundary removes water from the flow system.

Drain cells also were added to remove water along ravines. Elevations of drain cells were set to the gradient of the channel, according to site topographic data. The drains will only function in the model if the hydraulic head in the aquifer at a drain location is greater than the drain elevation. In this manner, the upgradient drains do not remove water from the flow model; the downgradient drains do, however, depending on the head in the surrounding model cells.

Cells in the computational grid were a uniform 10 ft ×10 ft (3 m ×3 m); the thickness varied according to the spatially irregular upper and lower surfaces of the permeable zone. This grid spacing was small relative to the distances between monitoring wells to provide high resolution of results in the modeling domain. The design allowed the placement of each tree into a model cell at any reasonable tree spacing.

B. Results

The calibration of the model to conditions prior to any active remediation was performed by adjusting input parameters within the range of measured values and by varying boundary conditions to achieve a suitable match to existing water level data and seepage flow measurements.

On July 10, 1996, a round of water levels was measured at most wells on ANL property. However, this event was prior to the installation of many of the on-site wells and a set of mini-monitoring wells in the forest preserve. The July 10, 1996, data were compared with assorted head data sets covering the last 10 years and was determined to represent typical head conditions. The modeling was initially focused on determining appropriate parameter values and boundaries for a steady-state scenario. Recharge was varied between 1.2 to 3.3 in/yr (3 to 8.3 cm/yr); hydraulic conductivity was varied from 3 to 8.6 ft/d (1.1×10^{-3} to 3.0×10^{-3} cm/s); and the northwest constant head boundary was varied from 675 to 680 ft (205.7 to 207.3 m). The calibration was carried out to not only match the target hydraulic head surface, but to match measurements of seep discharge.

Calibrated heads are depicted in Figure 4. The inflections in the contours are due to varying transmissivity resulting from spatial variation in the thickness of the aquifer. At the low end of the ravines, the effect of the model's outflow resulted in convergence of a portion of the flowfield at the approximate location of the seeps. Calibrated values were 3 ft/d (1.1×10^{-3} cm/s) for hydraulic conductivity and 1.3 in/yr (3.2 cm/yr) for recharge. This conductivity value is in the middle of the range for silty sands (Freeze and Cherry, 1979) and therefore is appropriate as a bulk value for the poorly sorted granular materials of the aquifer of interest. The recharge value was lower than that estimated on a larger scale for the dolomite aquifer (Walton, 1965), but is reasonable for the till-covered study area. The upgradient boundary was held at 676.5 ft (206.2 m), which is appropriate on the basis of an extrapolation of the site's typical gradient.

The model's solution was also supported by calibration to available seep flow measurements. According to the simulated flow budget, drain cells remove 172 ft^3/d (4.86 m^3/d) from the two main ravines north of their confluence (Figure 4). Field measurements of the flow near seeps SP01 and SP04 during typical flow conditions indicated 94 ft^3/d (2.66 m^3/d) for the north-trending ravine. The northwest-trending ravine's flow was not measured but is likely similar in magnitude; therefore, an

TABLE 1. Calibration Statistics for Steady State Flow

Well name	Heads measured 7/10/96 (ft MSL)	Calibrated steady state heads (ft MSL)	Measured minus simulated (ft)
317111	671.12	672.20	-1.08
317101	667.17	669.46	-2.29
MW014I	665.26	665.88	-0.62
MW012S	669.19	665.09	4.10
MW001I	665.38	664.57	0.81
MW013S	663.03	662.28	0.75
MW004I	664.94	661.90	3.04
MW003S	660.24	660.63	-0.39
MW007	657.82	659.21	-1.39
319011	659.01	659.03	-0.02
317021	652.50	655.60	-3.10
MW002IA	652.83	654.55	-1.72
319032	650.89	648.09	2.80
		ME =	0.07 ft
		MEA =	1.70 ft
		RMSE =	0.58 ft

adequate match to measured fluxes was attained during model calibration. The validity of the solution was tested through particle tracing from the source areas; discharge was calculated to be at the contaminated seeps.

Calibration statistics may be calculated in several ways (Anderson and Woessner, 1992). The Mean Error (ME) is the mean difference between measured heads and simulated heads. With this method, however, negative and positive differences may balance each other and reduce the error calculation. The Mean Absolute Error (MAE) is the mean of the absolute values of the differences in measured and simulated heads. The Root Mean Squared Error (RMSE) is the average of the squared differences in measured and simulated heads.

Calibration statistics are presented in Table 1 for 13 monitoring wells completed in the aquifer of concern. The calibration error was reasonably low, especially in light of the site's hydrogeologic complexity and fairly high hydraulic gradient. In the phytoremediation zone itself, the difference between target heads and simulated values ranged from -0.6 to 0.8 ft (-0.2 to 0.2 m), indicating a strong degree of calibration in the most critical region of the modeling domain.

The sensitivity of the model with respect to input parameters was evaluated, with emphasis on determining the sensitivity of the model to two key parameters: hydraulic conductivity and recharge. During sensitivity analyses, one of these parameters was varied, while all others were held to calibrated values. Relative to the calibrated value, hydraulic conductivity was increased in various runs by factors of 1.5 to 10 and decreased by factors of 1.5 to 10. Results indicated slight decreases in heads when conductivity was increased by 50%, with dewatering of portions of the domain initiating when conductivity is increased by a factor of 2 and becoming widespread with greater increases. Decreasing the hydraulic conductivity by as little as 50% resulted in heads increased by 3 to 5 ft (1 to 1.5 m) and the onset of a groundwater divide paralleling the northwestern boundary, with reversal of flow in the northwestern portion of the modeling area.

Recharge was increased in sensitivity runs by factors of 1.1 to 2 and decreased by the same factors. Increasing recharge by only 10% was found to raise heads by about 1 ft (0.3 m). With greater increases, heads were far above target levels. Decreases in recharge of 10% resulted in heads lowered by up to 1 ft (0.3 m). However, with greater decreases in recharge, some areas went dry, but some heads actually increased slightly. The increased heads were an artifact of the modeling code, which converts dry areas to inactive. Therefore, water backed up on the upgradient sides of dewatered zones.

Following the calibration to steady-state conditions, transient model runs were performed to match the seasonal variability of the heads in the study area. Transient calculations require storage terms. The storativity, obtained from the pump test, was set to 0.0023, the average of results from Theis and Cooper-Jacob methods (Fetter, 1988). The specific yield, necessary for confined aquifer cells that convert to unconfined because of dewatering, was set to 0.15, which is appropriate for this type of poorly sorted aquifer materials (Fetter, 1988).

The transient analysis was not aimed at matching all highs and lows of the available hydraulic head data, which are influenced by short-term events, but rather at matching average, seasonal heads. To calibrate the transient model, the recharge was adjusted seasonally, and the northwestern constant head boundary was replaced by a time-varying constant head boundary. For recharge, rather than distributing a rate of 1.3 in/yr (3.2 cm/yr) evenly throughout each month, the calibrated transient model had a high rate of 2.3 in/yr (5.9 cm/yr) applied in April to July, and a low value of 0.22 in/yr (0.56 cm/yr) applied in August to November. Other months had the typical 1.3 in/yr (3.2 cm/yr) flux. The time-variant boundary was adjusted during calibration to account for the seasonal trends in heads of the study area's northwestern monitoring wells. The calibrated model had a high value of 685 ft (208.7 m) each June, and a low value of 673.5 ft (205.3 m) each November. The typical value of 676.5 ft (206.2 m) was set to each March and August. Linear interpolation was used between each specified month.

Animation files of the transient calculations may be viewed on the journal's web site. (http://www.aehs.com/journals/phytoremediation/index.htm)

V. PREDICTION OF EFFECT OF MATURE PHYTOREMEDIATION SYSTEM

Modeling Approach

The calibrated transient flow model was then used to model the effect of the phytoremediation on the groundwater flow system. The modeling covered a period of 6 years. The initial 3 years of the simulation represented the plantation's development; the following 3 years represented the first three years of the mature plantations, when the canopies have closed together and water consumption is maximized. The trees, which are planted on 16-ft (4.9-m) centers, were each placed in an individual model cell (Figure 2). Because of the difference in the model cell spacing compared with the tree spacing, the tree locations did not appear as orderly as their actual locations; however, for the purposes of the model, the location were accurately represented. To model the trees' transpiration effect, the variable recharge and upgradient boundary condition were used, as described above. The leaf-on period was assumed to be 6 months, from April through September, with water use rates assigned as in Table 2. These rates are conservative estimates derived on the basis of studies of phreatophytes, such as poplar, willow, and tamarisk (Fletcher and Elmendorf, 1955; Robinson, 1964), and on literature values for water use by another phreatophyte, alfalfa (Jensen, Burman, and Allen, 1990). These estimates are also supported by experience with the luxury water consumption conditions that are expected to occur at the site. The water usage in Table 2 is similar to the range found in studies of poplars in various growing conditions (Wullschleger, Meinzer, and Vertessy, 1998; Hinckley *et al.,* 1994).

The model incorporates the transpiration effect of the trees as shown in Figure 5. The maximum transpiration, as listed in Table 2, was attainable when the head in a model cell was at or above the top of the aquifer. Transpiration decreased linearly with decreasing head to a cutoff depth, which was assigned as the midpoint of each model cell containing a tree.

B. Results

Calculations indicated a significant, seasonal effect on groundwater flow caused by the trees. The influence of the trees was apparent as early as the summer of the second year (2001). Figures 6 and 7 illustrate representative results for a low-head period resulting from transpiration of mature trees and decreases in seasonal flux (*e.g.*, September of the fourth year, 2003) and for a high-head period resulting from recharge of the aquifer (*e.g.*, April of the fifth year, 2004), respectively. (The reader is encouraged to access the journal's web page for viewable animation files of the transient results. The animations allow visualization of the changes occurring in each month of the 3 years of tree development and in the first 3 years of maturity.) The heads and the sizes of the dewatered areas in years 4 through 6 are essentially the same, which indicates that the water withdrawal effects of the plantations are not cumulative, but rather that the mature phytoremediation system maintains the same cycle of change each year.

Because the penetration depth of roots is limited in the model, dewatering of a given model cell is not caused by a tree that may be in the cell, but by cumulative transpirative stresses that lower the water table to the elevation of the bottom of the aquifer. These dewatered areas are depicted in Figures 6 and 7 as areas bounded by polygons. The onset of dewatering coincides with the highest elevations in the bottom surface of the aquifer. Rewetting of many of the dewatered model cells occurred during recharge periods (e.g., November through April).

TABLE 2. Estimated Water Use Rates for ANL TreeWells™

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Year 0 (1999)	0	0	0	0	0	0	2	3	2	1	0	0
Year 1 (2000)	0	0	0	0	2	8.75	12.5	12.5	8.75	2	0	0
Year 2 (2001)	0	0	0	0	5	17.5	25	25	17.5	5	0	0
Year 3 (2002)	0	0	0	0	7	24.5	37.5	37.5	24.5	7	0	0
Year 4 (2003) and beyond	0	0	0	0	10	35	50	50	35	10	0	0

Note: All values are conservative estimates of average gallons per day per tree on a monthly basis (1 gallon = 3.78 liter).

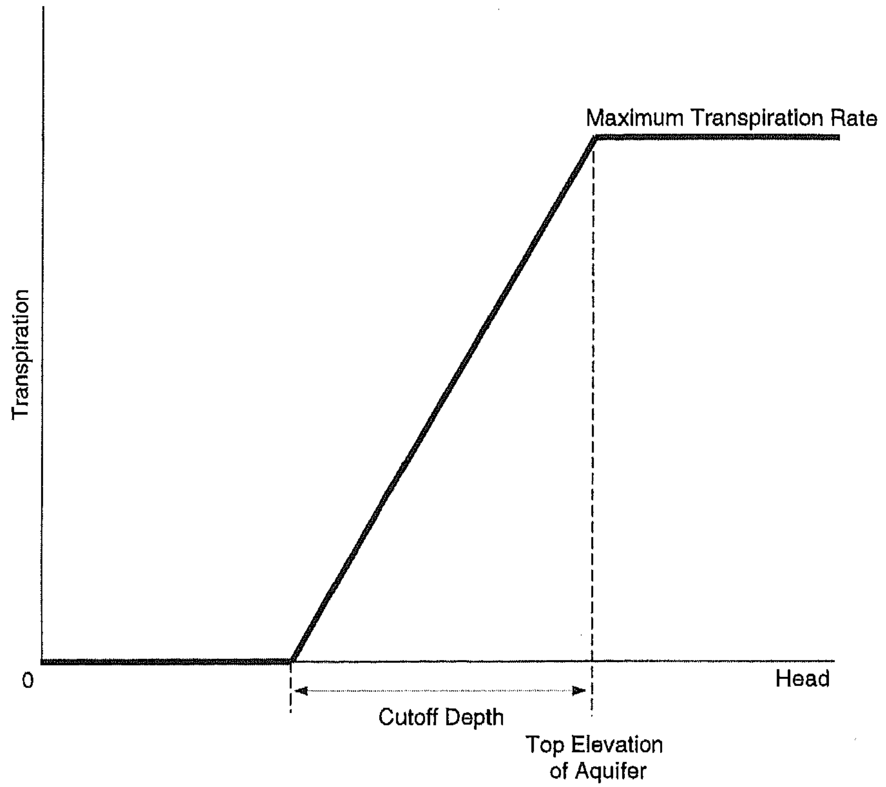

FIGURE 5. Transpiration vs. head in a model cell (modified from McDonald and Harbaugh 1988).

The calculated hydrologic budget for this simulation of 72 monthly time periods showed a maximum volumetric error (relating modeled inputs and outputs of system) of only 0.29% in any monthly period. This value indicates an accurate calculation of the modeled inputs and outputs to the system and proper convergence by the iterative solver.

A particle tracking analysis was performed to evaluate the hydraulic capture potential of the plantations and to determine the residence time of contaminated groundwater in the geochemically altered groundwater of the microbially active rhizosphere. Particle starting locations were set to the center depth of the aquifer of interest along the upgradient edges of the plantations. Starting times for particles were in January 2000, which coincides with the beginning of the 6-year simulation period. The results (Figure 8) indicate that the trees provide a large degree of hydraulic containment. In the 317 Area, most of the particles are captured; however, one particle trace skirts the edge of the 317 Area trees before essentially stagnating east of the plantation. In the 319 Area, most groundwater is captured. A small portion escapes because of the gap in the plantation along a deep, steeply banked surface drainage. The trend of this drainage happens to be aligned with the overall groundwater flow direction. Monitoring of heads over the next few years will indicate the

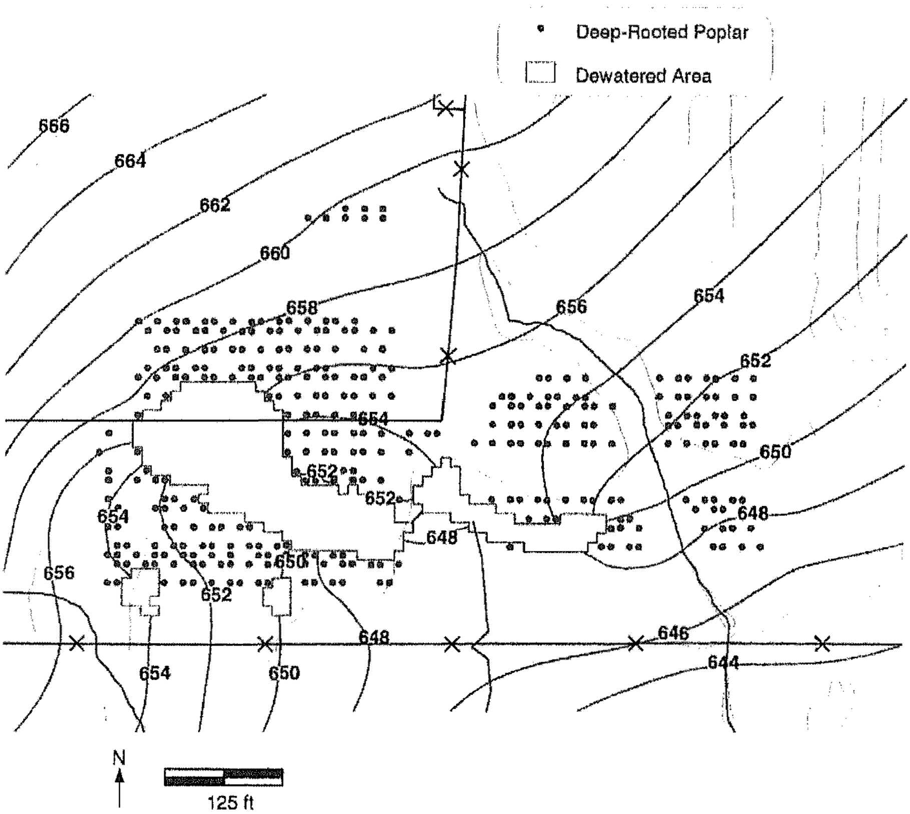

FIGURE 6. Low heads resulting from phytoremediation: September of fourth year (2003).

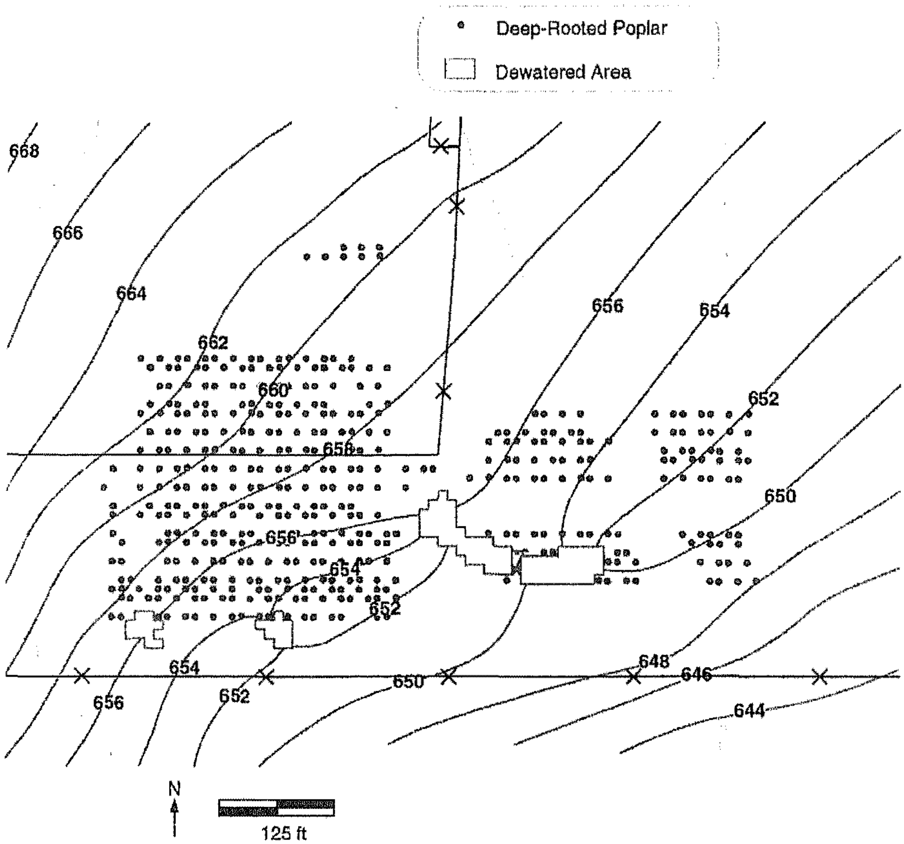

FIGURE 7. High heads during phytoremediation: April of fifth year (2004).

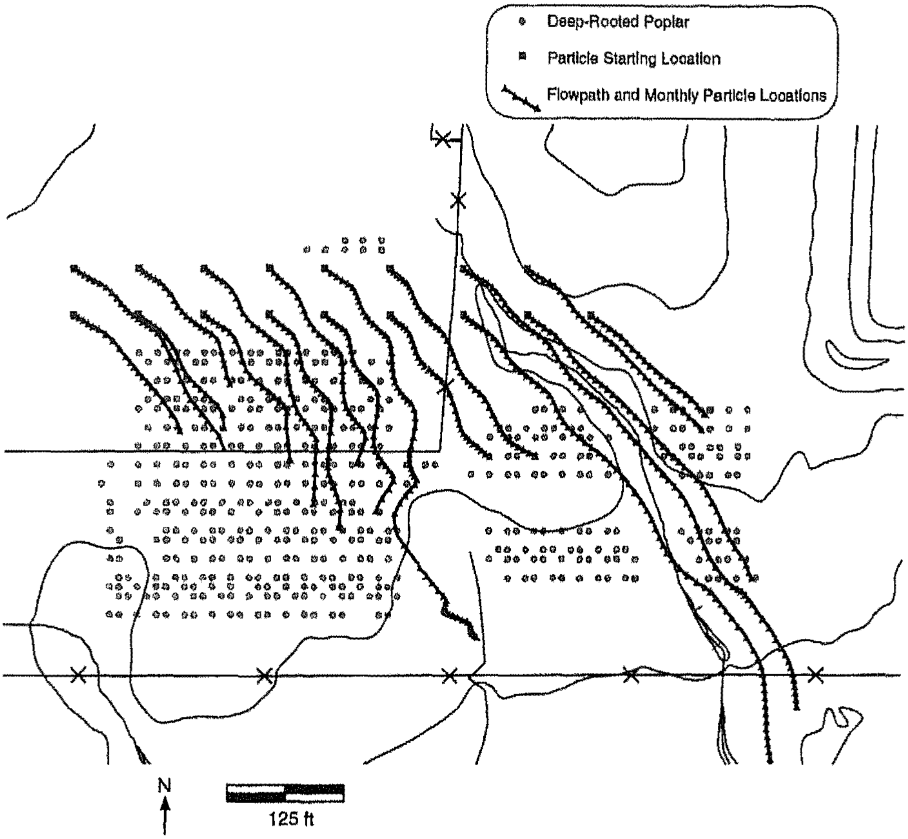

FIGURE 8. Particle tracking results for phytoremediation system (particles start in January 2000).

degree of containment and treatment being achieved, and whether the system should be modified with additional deep-rooted poplars.

Inflections along the particle traces indicated the response of the flow system to changes in the transpirative stresses and to changes in the seasonal inputs to the system. The monthly particle locations showed variable spacing, which represents relatively slower movement of groundwater during natural gradient periods (i.e., leaf-off) and relatively faster flow during withdrawal periods (i.e., leaf-on).

The monthly particle locations may be used to estimate the residence time of groundwater near the chemically altered rhizosphere. Particles started north of the main 317 planting area are in the or near rhizosphere approximately 6 to 24 months before being withdrawn by the plants. In the 319 Area, particles started north of the planting area are either removed after 3 to 10 months, or may pass through the remediation system. Those particles that escape have residence times of approximately 10 months in or near the rhizosphere of the 319 plantation. The range in these estimates is due to numerous factors: the spatially variable saturated thickness of the aquifer, the seasonal changes in groundwater flux into the system, the seasonal leaf-on and leaf-off periods, and the composite pumping effect of nearby trees.

VI. SUMMARY AND CONCLUSIONS

This study demonstrates the usefulness of numerical groundwater modeling in addressing several issues pertaining to the design or evaluation of a phytoremediation system that relies on phreatophytes. While uptake or destruction of contaminants is not explicitly addressed, the engineered system of deep-rooted poplars was predicted to provide a large degree of hydraulic control, despite seasonal variation in water use rates by the plantation. The results indicated areas that are typically dewatered because of high seasonal water use and irregular aquifer geometry; this information may be useful in the future for explaining possible different rates of tree development across the planting areas. Modeling clearly has application at phytoremediation sites for evaluating or designing a containment system with respect to factors such as tree planting density, plume width vs. groundwater flow rate, seasonal effects, residence time of groundwater within the influence of the rhizosphere, prediction of regions where seasonal dewatering may occur, and future modifications to the system design to improve the likelihood of hydraulic capture.

Future analysis at the ANL site will include a comparison of model results with measured water use by trees and measured water levels. An improved understanding of root development (i.e., lateral and downward growth) will provide a better conceptualization and implementation of roots in numerical models.

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