

HYDROCARBON SPILL EXPOSURE ASSESSMENT MODELING

Ground Water Management
Book 4 of the Series

Proceedings of the

1990

Petroleum Hydrocarbons and Organic Chemicals in Ground Water: Prevention, Detection, and Restoration

Abstract

This bound volume contains papers presented at the 1990 Petroleum Hydrocarbons Conference. The meeting was held October 31-November 2, 1990, in Houston, Texas.

The conference is annually co-sponsored by the Association of Ground Water Scientists and Engineers and the American Petroleum Institute. A combined total of 52 oral and poster presentations was presented in 1990 focusing on prevention of contamination, detection, and monitoring of hydrocarbons in soils and ground water, transport and fate, free-phase hydrocarbon recovery, defining remediation levels, remediation of residual phase hydrocarbons (by vapor extraction and bioventing), and biodegradation.

Invited presenters included Jay Lehr, National Water Well Association; Bob Hockman, Amoco Corp.; Rob Booth, Environment Canada; and Iris Goodman, U.S. EPA. Attendees represented a broad spectrum of interests and specializations.

The co-sponsors hope environmental scientists find the papers in this publication to be of great value.

Conference on Petroleum Hydrocarbons and Organic Chemicals in Ground Water-Prevention,
Detection, and Restoration (1990 ; Houston, Tex.)

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Hydrocarbon Spill Exposure Assessment Modeling

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Abstract

Hydrocarbon spills impact drinking water supplies at down gradient locations. Conventional finite difference and finite element models of multiphase, multicomponent flow have extreme requirements for both computer time and site data. Site data and the intent of the modeling often do not warrant the application of such models. An alternative approach is proposed which is based on semi-analytic models for vertical product infiltration, radial spreading on the water table, and transport of aqueous phase contaminants in the aquifer. Three individual models for these processes are linked to estimate exposure at a down gradient well. The infiltration model is based on kinematic wave theory supplemented by approximate dynamic relationships. In the vicinity of the water table the product moves laterally, and is approximated by considering the effects of buoyancy and assuming that the shape of the oil lens is given by the Ghyben-Herzberg-Dupuit assumptions. The flux of dissolved chemicals to the aquifer is related to the recharge rate, the size of the lens, the groundwater seepage velocity, and the oil-water partition coefficient of the dissolved constituents. The boundary condition for the aquifer model must be able to handle transient mass flux. This is accomplished by integrating the analytic solution of the advection-dispersion equation with the source as a time-variable boundary condition. A time record of concentration can be determined for any desired location in the aquifer. The presentation outlines the methodology and examines parameter sensitivity within the combined vadose zone, oil lens, and aquifer transport models.

Introduction

When fluids that are immiscible with water (the so-called non aqueous phase liquids or NAPLs) are released in the subsurface, they remain distinct fluids, flowing separately from the

water phase. Fluids less dense than water (L-NAPLs) migrate through the unsaturated zone, but when reaching the water table, tend to form lenses on top of the aquifer. Generally the fluids are composed of complex mixtures, so that aquifer contamination results from the dissolution of various components of the NAPL. This paper outlines the methodology for a screening model for releases or spills of L-NAPLs.

Components of the Model

The model proposed for hydrocarbon exposure assessment modeling is composed of three parts. The first two parts are transport of the hydrocarbon product downward through the unsaturated zone and subsequent radial transport along the water table. These two are simulated by the KOPT/OILENS model, where the acronym KOPT stands for Kinematic Oily Pollutant Transport. The third part of the model is transport through the aquifer of dissolved contaminants, which is simulated by TSGPLUME (Transient Source Gaussian Plume). All of the models are in the form of semi-analytical and analytical solutions of the governing equations. The conceptual basis of the models are discussed in the following paragraphs. The mathematical details of KOPT/OILENS may be found in Weaver and Charbeneau (1989) and Charbeneau et al. (1990), while those of TSGPLUME can be found in Charbeneau and Johnson (1990).

The KOPT/OILENS model is intended to address the specific problem of L-NAPL transport from the ground surface to a water table aquifer. Since ultimate interest lies with water quality, an emphasis of the model is the determination of the oil lens size and the mass flux of contaminants into the aquifer. These quantities are not predictable without considering three-phase flow in the unsaturated zone. Thus the model is intended for use in characterizing the magnitude of the source of aquifer contamination. In modeling terms, KOPT/OILENS provides a time-variable boundary condition for an aquifer model.

A chemical constituent dissolved in both the NAPL and water phases is also tracked by KOPT/OILENS. Once that chemical reaches the water table it contaminates the aquifer by contaminating the recharge water and by dissolution from the L-NAPL lens. The TSGPLUME model takes the dissolution mass flux from the KOPT/OILENS model and calculates the expected concentrations at a number of down gradient receptor wells. Notably the mass flux from KOPT/OILENS is time varying, so that the aquifer model must be capable of simulating a time varying source condition.

Usage of the Approximate Model

The purpose of the methodology described herein is use as a screening tool. For example, the model can be used to estimate the effects of L-NAPL loadings, partition coefficients, groundwater flow velocities, etc., on pollutant transport. Since approximations are used for developing the model, the model results must be viewed as approximations. If simulation of complex heterogeneous sites is needed, or other approximations made in the models are unacceptable, then a more inclusive model, such as the MOFAT code developed at VPI (Kuppusamy et al., 1987), should be used instead of, or in addition to, KOPT/OILENS and TSGPLUME.

Complex models, however, may not always be the most desirable tool for a given problem. Such models require large amounts of computer time and available memory. Further, there may be a significant investment in training users to set up the model and run it properly. Additionally, a large amount of field data is required to run such a model because the expense of running the model is not warranted if adequate site data are not available. In addition to the parameters for aqueous phase solute transport (such as hydraulic conductivity, dispersivity, sorption parameters), multiphase transport parameters are needed (interphase partition coefficients, capillary pressures and relative permeabilities) for each different zone or material present in the field. The latter properties are not well understood and are difficult to obtain for field problems. Site data is usually incomplete because of monetary, safety and regulatory limitations. Historical records of pollutant releases are often nonexistent, although such knowledge should be precisely defined in a model. Sampling limitations often result in situations where the total mass of contaminants cannot be defined. These limitations are likely to require approximations to be made even when running a complex model. Certain problems may warrant the use of alternative simplified models.

KOPT/OILENS and TSGPLUME have an advantage in some of these areas; the models execute rapidly on small computers, require little memory and are designed to be run easily. The advantage is that they are based on semi-analytic approaches, which do not require discretization of the domain nor iterative solution of the non-linear governing equations. These advantages are achieved at the cost of flexibility in accommodating heterogeneities and other phenomena. Clear recognition should be made that for the sake of efficiency and robustness, accuracy and/or the ability to simulate various situations is being given up. At some point there is a limit to the phenomena that can be treated in a simplified context; beyond that limit, the complex models must be used.

Assumptions of the Model

The following paragraphs describe the conceptual model upon which KOPT/OILENS and TSGPLUME are based. This discussion is intended to give a clear understanding of the assumptions and limitations of the models.

The flow system is idealized as consisting of a circular source region overlying an aquifer at specified depth. Although flow in the unsaturated zone is three-dimensional, the KOPT model treats transport through the unsaturated zone as one-dimensional. Lateral spreading of contaminants by capillary forces is neglected, as is spreading due to heterogeneity, since the soil is assumed to be of uniform composition. For situations where the contaminants are applied over relatively large areas, the flow becomes nearly one-dimensional in the center. For contaminant sources that are of small areal extent, the lateral transport of contaminants may be significant, and the assumption of one-dimensional flow is less applicable. In some sense, however, one-dimensional modeling leads to a conservative model as all of the pollutant is assumed to move downward. In reality, some may be left behind due to entrapment by layering or lateral spreading.

The spill or release of the L-NAPL phase may be simulated in three ways. First is a release of a known L-NAPL flux for a specified duration. The release occurs at the ground surface. Based on an approximate capillary suction relationship, some of the L-NAPL may run off at the surface if the flux exceeds the maximum effective L-NAPL conductivity. Second, a

constant depth of ponded L-NAPL, for a known duration, may also be specified. This case represents a slowly leaking tank, or a leaking tank within an embankment. Lastly, a known volume of L-NAPL may be placed over a specified depth of the soil. This last scenario represents either a land treatment operation or a landfill containing a known amount of contaminants at the beginning of the simulation.

L-NAPL phase transport occurs within the soil which contains a constant water saturation (saturation is defined as the per cent of the pore space filled by a given fluid), corresponding to the average annual recharge rate, or some other water saturation. The temporal effects of climate: temperature, rainfall, relative humidity, and wind speed are neglected, as average values of these parameters are used. The average (annual) recharge rate is used to represent the actual time dependent recharge. Justification of this approach comes from the fact that the soil moisture profile remains relatively uniform except near the surface. Much data is required to simulate the time record of rainfall events to develop the non-uniform soil moisture profile. The effort involved is not warranted in light of the assumptions used in developing the models.

In accordance with common soil science practice, the effect of the air flow on the L-NAPL phase transport is neglected. The effect of the presence of the water and air phases on the L-NAPL phase transport is included by the usage of a non-hysteretic three-phase relative permeability model. This model is a reasonable approximation of the pore-scale phenomena occurring in three-phase flow, but the actual nature of these relationships is a major cause of uncertainty in this and most other multiphase flow models. The model uses measured properties of the soil (capillary pressure curve parameters) to approximate the relative permeability. The model does not include transport in fractures or macropores.

Efficiency is achieved in running the model primarily because the gradients of the capillary pressure are neglected. This causes the governing equations to become hyperbolic equations, which can be solved by the generalized method of characteristics (Smoller, 1983). One major effect of this assumption on the simulation results is that the leading edge of the L-NAPL moving into the soil is idealized as a sharp front (Figure 1). Some experimental results show soil moisture profiles which have nearly sharp fronts. Reible and Illangasekare (1989) show infiltration results for an L-NAPL phase where the fronts are nearly sharp. For cases where the front truly spreads, Charbeneau (1984), for example, presents a theoretical proof that the mean displacement speed of the sharp and true fronts is the same. Smith (1983) presents a numerical result for water flow showing that a numerical solution of Richards equation was tracked by a sharp front solution.

Since the capillary gradient has a dramatic impact on the infiltration capacity of the soil, an approximate model (the Green-Ampt model) is used to estimate the infiltration capacity during the application of the L-NAPL phase. This gives the proper flux in the soil, given a flux or constant head ponding condition at the surface.

If a large enough volume is supplied, the L-NAPL reaches the water table. Typically this occurs in a relatively short time for L-NAPLs with high mobilities, like gasoline. If sufficient head is available, the water table is displaced downward, lateral spreading begins, and the OILENS portion of the model is triggered. OILENS is based on three major approximations. First the L-NAPL spreading is purely radial, which implies that the slope of the regional groundwater table is small enough to be unimportant for the lens motion. The second major assumption is that the thickness is determined by buoyancy only (Ghyben-Herzberg relations). Thirdly, the shape of the

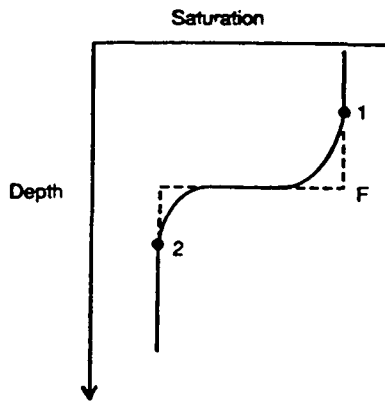


Figure 1 Idealized sharp front profile

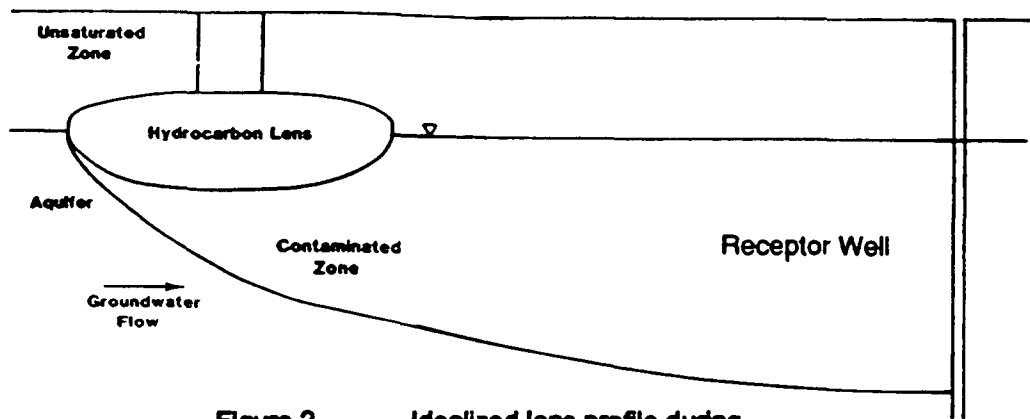


Figure 2 Idealized lens profile during hydrocarbon lens growth

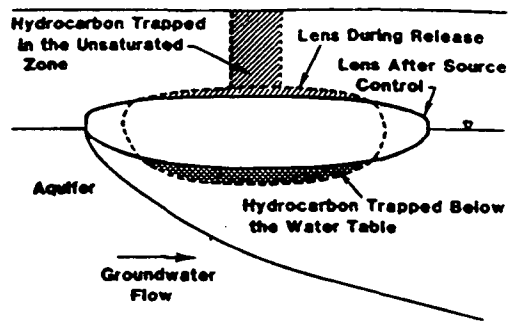


Figure 3 Idealized lens profile during hydrocarbon lens decay

lens is given by the Dupuit assumptions, where flow is assumed horizontal and the gradient is approximated by the change in head over a horizontal distance. The three major assumptions lead to a very efficient formulation of the model, which is reflected in its low computational requirements.

Typical results from OILENS include the following features. The lens thickness increases (Figure 2) during the initial phase of spreading. Initially, the L-NAPL enters the lens faster than its volume is spread radially, hence the increase in lens height. Later, the lens thins while continuing to spread laterally. Residual hydrocarbon is left both above and below the actively spreading lens during this period (indicated in Figure 3 as the lens after source control).

In KOPT/OILENS the L-NAPL is treated as a two-component mixture. The L-NAPL itself is assumed to be soluble in water and sorbing. Due to the effects of the recharge water and contact with the ground water, the L-NAPL may be dissolved. This may be significant for highly soluble L-NAPL phases. The L-NAPL's transport properties (density, viscosity, capillary pressure, relative permeability), however, are assumed to be unchanging. The second component is a chemical constituent which can partition between the L-NAPL phase, water phase and the soil, and can volatilize. This constituent of the L-NAPL is considered the primary contaminant of interest.

A kinematic approach is used within KOPT/OILENS for the dissolved chemical transport, which results in a model that neglects dispersion. The chemical motion is assumed to be due to the advection of water and L-NAPL only. The chemical, which is the second component of the L-NAPL phase discussed above, is assumed to partition between the NAPL, water and soil according to equilibrium, linear partitioning relationships. Non-equilibrium and diffusion-limited partitioning are neglected. Chemical mass flux into the aquifer comes from recharge water being contaminated by contact with the lens and from dissolution occurring as groundwater flows under the lens. The concentration of the chemical in the aquifer is limited by its water solubility. Volatilization is treated as diffusion from the top most part of the contaminated zone. The only vapor phase flux that is considered is that due to Fickian diffusion. The volatilization of the chemical from the lens is neglected.

The hydrocarbon and chemical contaminant can biodegrade within the lens, based on oxygen supply from the ground water. Both can degrade, based on their stoichiometric oxygen consumption. Degradation is neglected in KOPT, i.e., unsaturated zone transport, and reaeration of the groundwater from the unsaturated zone is neglected. Degradation in the aqueous phase is modeled in TSGPLUME using first-order decay.

The aquifer transport of the dissolved contaminant is simulated by using a two-dimensional vertically averaged analytic solution of the advection-dispersion equation. The boundary conditions are placed at the down gradient edge of the lens and take the form of a Gaussian distribution with the peak directly down gradient of the center of the lens. The peak concentration of the Gaussian distribution adjusts through time so that the simulated mass flux from the lens equals that into the aquifer. The width of the Gaussian distribution remains constant and is taken so that four standard deviations are equal to the ultimate diameter of the lens. Although the size of the lens varies with time, a constant representative lens size is used for the aquifer source condition. Note that in many cases the lens reaches its maximum size rather rapidly compared with the transport in the aquifer, so that the use of the maximum lens size will not introduce large errors.

The depth of penetration of the plume is calculated from vertical dispersion beneath the lens plus the advection flow due to infiltration through the lens, following the approach outlined in the background documents for the EPACML model (USEPA, 1988). If the calculated penetration depth exceeds the aquifer thickness, then the plume fully penetrates the aquifer and the model allows for dilution of the plume by diffuse recharge. If the penetration depth is less than the aquifer thickness, then the plume thickness is taken as the penetration depth. The thickness remains constant and the plume is not diluted by diffuse recharge, representing the case where recharge simply pushes the plume deeper into the aquifer. The analytical solution for the plume concentration at any time and location is found through use of Romberg integration.

Example Simulation

Table 1 contains the parameters for an example which will be discussed subsequently. Since there are large uncertainties associated with parameter values for subsurface models, the approach will be to investigate the sensitivity of the simulated results to parameter variations. First, a fairly detailed set of results from the models will be presented for a baseline case. Due to space limitations, parameter uncertainty will be illustrated with TSGPLUME results only.

The base case consists of a gasoline release from an above ground storage tank which falls, but the product is retained within a surrounding dike. The fate of xylene, which is assumed to compose approximately 1% of the mass, is of interest. In the base case, the radius of the source region is 4.0 meters. The gasoline is assumed ponded at an average depth of 1 cm for a period of 3 days. The water table is 10 meters below the land surface. The unsaturated zone and aquifer are composed of a clean sand.

KOPT/OILENS predicts that the 3-day duration of the release results in a total of 69.9 cubic meters of gasoline entering the subsurface. The KOPT/OILENS simulation of 1000 days, with a 5-day maximum time step, required approximately 1 minute and 49 seconds of time to run on an AST Premium 386/16 PC with an intel math co-processor and disk caching. The TSGPLUME simulation with 4 receptor well locations and 120 output times required approximately 3 minutes and 17 seconds.

Figure 4 shows total liquid profiles for several times prior to the gasoline front reaching the water table. These profiles illustrate the percent of the pore space filled by the liquids (saturation) as a function of depth for given times. The water flux of 0.001 m/d, representing the average annual infiltration rate, results in 32.6% of the pore space being filled by water. The remaining portion of the pore space is available for the gasoline. At 3 days, the end of the release, Figure 4 shows that the gasoline occupies 55.7% of the pore space from the surface to a depth of 6.18 meters. After the end of the release, the saturations decrease with time throughout the profile, as indicated for days 4 and 5. During the entire simulation, the gasoline flux decreases with time. The cumulative gasoline flux into the lens (Figure 7) shows slow drainage of the gasoline into the lens over the simulation time.

After 5.65 days, the lens begins forming at the water table. Figures 5 and 6 illustrate two configurations of the lens, corresponding to the schematic representations in Figures 2 and 3. At 7.5 days (Figure 5) the lens height is still increasing. By 50 days (Figure 6) the lens is thinning while continuing to spread radially. Figure 6 shows the extent of the gasoline that is trapped at residual saturation, after the lens has thinned. Figure 7 shows the gasoline inflow and radius as a

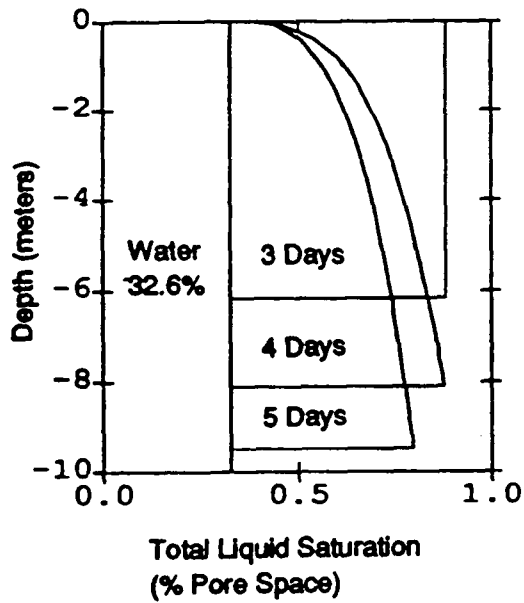


Figure 4 Total liquid saturation profiles in the unsaturated zone

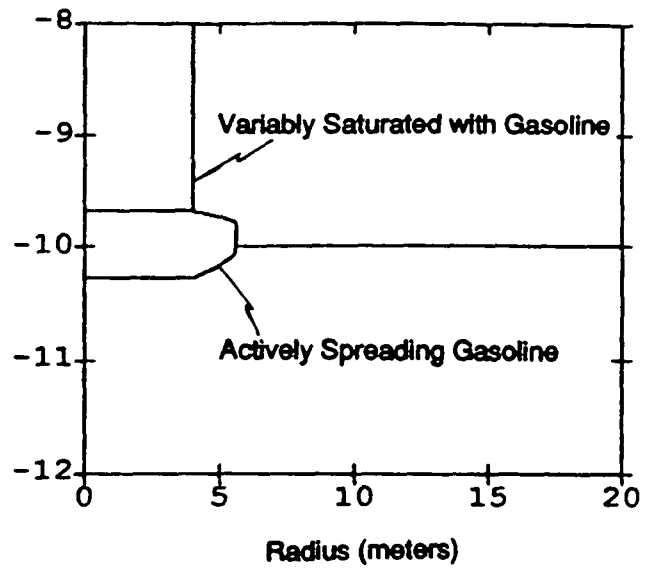


Figure 5 Gasoline lens profile at 7.5 days

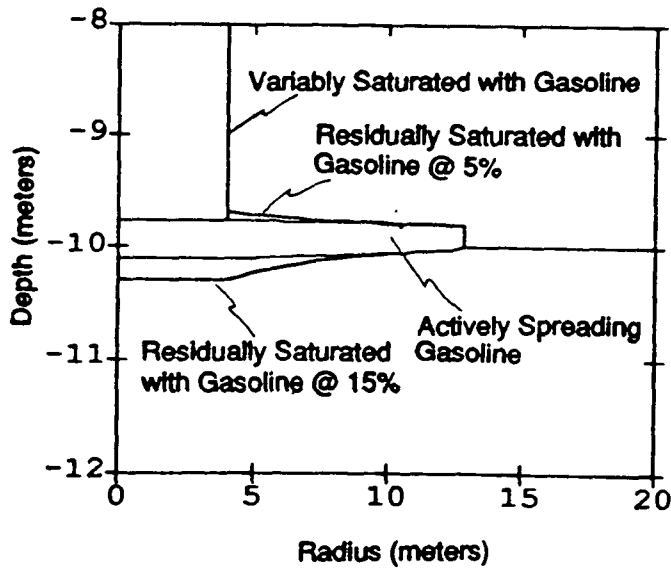


Figure 6 Gasoline lens profile at 50 days

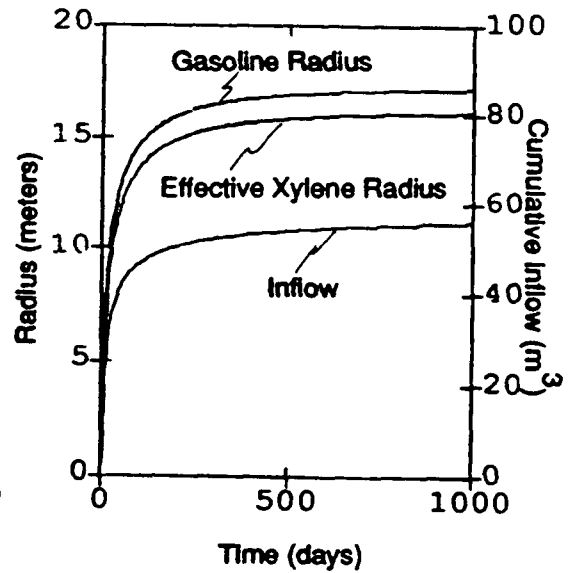


Figure 7 Gasoline radius and volume inflow with time

function of time. Note that the maximum extent of the lens is largely attained within the first 200 days of the simulation.

Table 1 Parameters for Example Simulation

Soil and Aquifer Characteristics

Saturated horizontal hydraulic conductivity (water)	5.0 m/d
Saturated vertical hydraulic conductivity (water)	1.0 m/d
Pore size distribution (Brooks and Corey)	1.0
Residual water saturation	0.10
Thickness of capillary fringe	0.10 m
Porosity	0.40
Bulk density	1.65 g/cc
Longitudinal dispersivity	10 m
Transverse dispersivity	2 m
Vertical dispersivity	0.10 m
Groundwater seepage velocity	0.25 m/d
Average water infiltration rate	0.001m/d
Aquifer thickness	5 m
Penetration Depth	2.8 m

Hydrocarbon characteristics

Density	0.70 g/cc
Dynamic viscosity	0.30 cp
Residual oil saturation above lens	0.05
Residual oil saturation below lens	0.15
Hydrocarbon solubility	200 mg/l
Surface tension	30 dyne/cm ²

Contaminant characteristics

Initial concentration in gasoline	7000 mg/l
Hydrocarbon-water partition coefficient	100
Soil-water partition coefficient	2.0 l/kg
Contaminant solubility in water	175 mg/l
Contaminant theoretical oxygen demand	0.032

The effective lens radius for the contaminant (xylene) is not necessarily the same as the lens radius. This is due to the partitioning of the contaminant between the L-NAPL, water and soil. In the lens, motion is assumed to be due to the L-NAPL only, so that there is an effective retardation of the contaminant caused by its partitioning to the water and soil. The amount of retardation depends on the L-NAPL/water and water/soil partition coefficients. Figure 7 shows the effective contaminant radius for the baseline case.

The dissolution rate of xylene into the aquifer, caused both by contact with the flowing groundwater and infiltrating water is shown in Figure 8. Initially, the rate of dissolution is low, due largely to the small radius of contact between the lens and the aquifer. As the size of the lens increases so does the dissolution rate. Later, as the water leaches the xylene out of the lens, the rate drops. Figure 9 shows the mass of xylene in the lens as a function of time. The mass of xylene contained within the lens increases for approximately the first 100 days, since the mass flux out of the lens is smaller than the xylene loading. As the lens increases in size, the mass in the lens drops due to leaching. Since mass enters the lens throughout the 1000 days depicted, the sum of the two curves in Figure 9 is not a constant.

Xylene concentration histories from TSGPLUME at receptor wells located 50, 100, 150, and 200 meters down gradient along the centerline of the plume are shown in Figure 10. As expected, dispersion decreases the peak concentrations as the distance from the source increases. At 50 meters the concentration history shows a rapid rise in concentration, followed by a long tailing at low concentration. This effect is due to the tailing in the mass flux input to the aquifer (Figure 8), which occurs even though equilibrium linear partitioning is assumed between the phases. In this example, no degradation or chemical attenuation of the xylene is assumed to occur, resulting in the long tailing of the contaminant at each location. The observed tailing is not due to non-equilibrium in partitioning, rather it is caused by the distribution of mass flux to the aquifer. Histories at the down gradient distances also show tailing, but dispersion reduces the sharpness of the rising concentration.

Parameter Sensitivity

Invariably, input parameters for subsurface models are uncertain. In this section, the effects of selected input parameter variation on exposures are assessed through the model results.

The lens source width is a parameter for TSGPLUME which must be taken as constant, despite the fact that the width varies as the lens enlarges. Figure 11 shows the effect on the estimated concentrations of using the same mass flux but with half the ultimate lens size. There is a slight increase in peak concentrations due to the narrower plume for the baseline case (see Figure 10). At 50 meters the peak concentration increases from 12.0 to 14.0 mg/l. Further down gradient, however, the effect of source size becomes negligible, as at 200 m the peak only increases from 2.97 to 3.07 mg/l.

Often the loading pattern of contaminants is a major source of uncertainty. By varying the duration of the release so that the gasoline volume varies from 49.5 to 90.0 cubic meters, the peak concentrations at 50 meters increase as illustrated in Figure 12. Increasing the volume of gasoline introduced into the subsurface increases the peak concentrations. The arrival times are unaffected in this example, because the gasoline is applied in the same way in all three cases. Since the gasoline volume increases the aquifer source radius, there may be some tendency for concentration to be reduced near the source. In this example, however, the latter effect is unimportant relative to the increased volume.

The L-NAPL/water partition coefficient, K_o , plays a major role in determining the mass flux of contaminant into the aquifer (Figure 13). As the coefficient increases, the amount of contaminant entering the water phase in a given time decreases, since the contaminant tends to

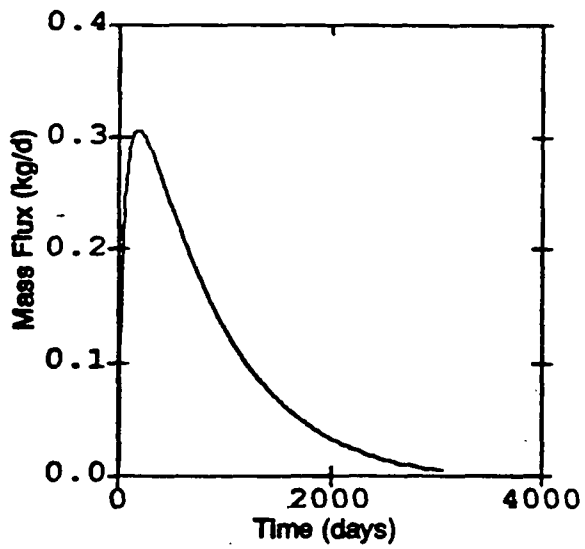


Figure 8 Xylene mass flux into aquifer

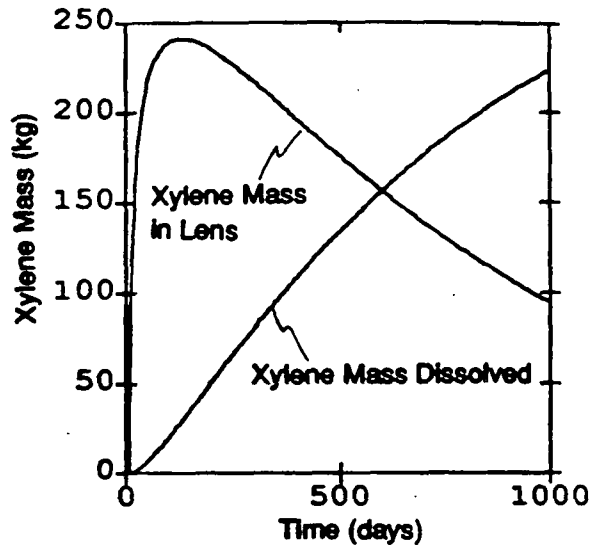


Figure 9 Xylene mass in lens with time

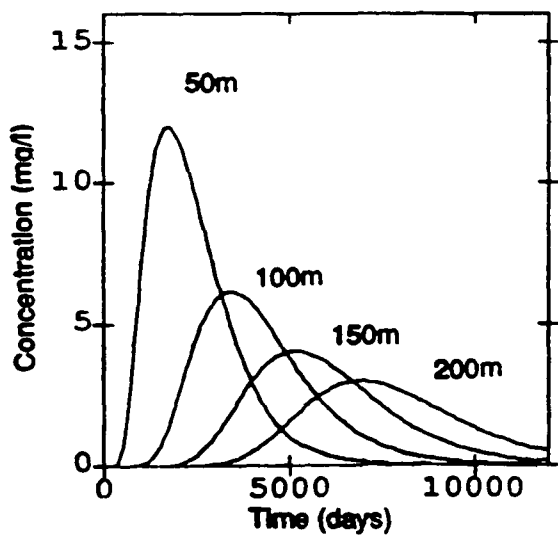


Figure 10 Estimated concentration histories for receptor wells at 50, 100, 150, and 200 meters down gradient

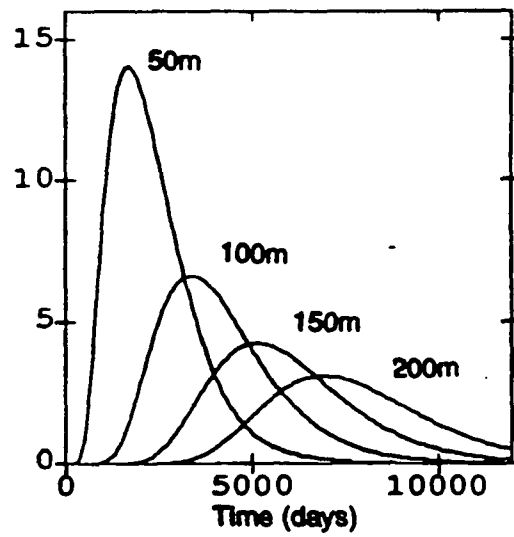


Figure 11 Source size impact on estimated concentration history at 50 meters

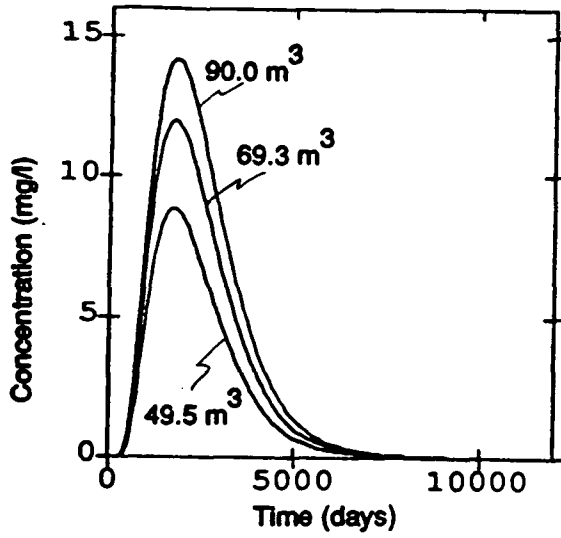


Figure 12 Gasoline loading impact on estimated concentration history at 50 meters

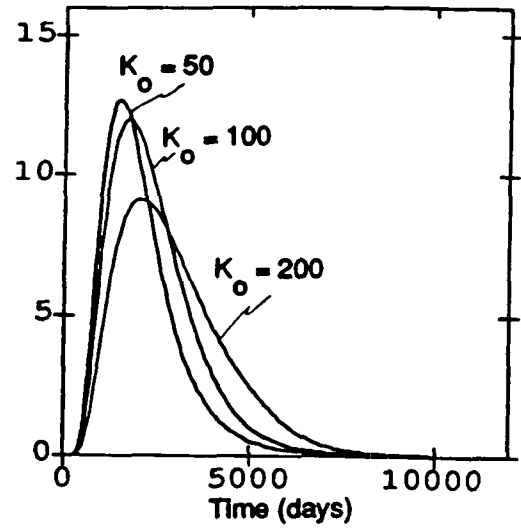


Figure 13 Gasoline/water partition coefficient impact on estimated concentration history at 50 meters

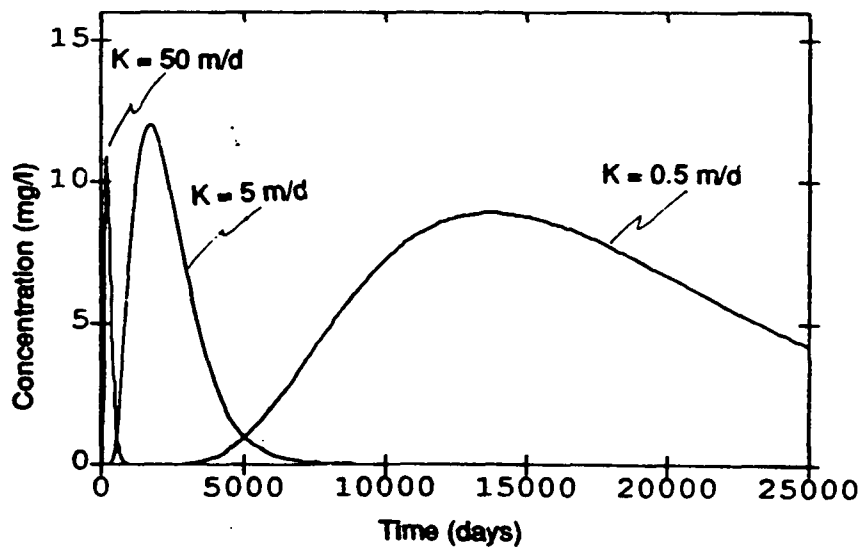


Figure 14 Hydraulic conductivity impact on estimated concentration history at 50 meters

reside within the gasoline. With enough time, however, the same mass of contaminant will leach into the groundwater. Thus the duration of the mass influx and the time to peak increase as the contaminant becomes more hydrophobic (Table 2). The effective size of the gasoline lens also increases with K_o , as more of the dissolved contaminant flows radially in the gasoline phase. Peak concentrations decrease, with increasing K_o , since the mass enters the aquifer over a longer period of time at lower mass flux. As indicated by Figures 10 and 11, the effect of increased radius also contributes to the lowering of the peak concentration with increasing K_o .

Table 2 L-NAPL/Water Partition Coefficient Effects

L-NAPL/ Water Partition Coefficient	Effective Xylene Lens Radius (meters)	Influx Duration (days)	Peak Concentration at 50 meters (mg/l)
50	14.2	1910	12.7
100	16.1	3060	12.0
200	16.9	5015	9.1

The effect of changing the aquifer hydraulic conductivity by orders of magnitude is illustrated in Figure 14. For comparison purposes, the hydraulic conductivity of the unsaturated zone is unchanged, so that the gasoline arrives at the water table at the same time in all cases. The rate at which the gasoline spreads, the mass flux of xylene to the aquifer, the peak concentration, and the advective travel time in the aquifer all strongly depend on the aquifer's hydraulic conductivity (table 3). Figure 14 shows that for the high hydraulic conductivity case ($K = 50.0$ m/d), the peak arrives fastest due to the highest seepage velocity.

The height and shape of the concentration histories depend on the distribution of xylene mass flux to the aquifer and the water flux under the lens. The xylene entering the aquifer mixes with the groundwater in a mixing zone to determine the aquifer concentration under the lens. The water volume flux through the mixing zone depends on its size, which is determined by the effective radius of the source and the penetration depth, and the seepage velocity. The combination of mass flux from the lens, seepage velocity, and mixing zone size for the high hydraulic conductivity case results in relatively low initial concentrations, compared with the other cases. At the 50 meter receptor well location, the peak concentration is lower than that for the base case ($K = 5$ m/d). The concentration history in the high hydraulic conductivity case is narrow, since the xylene enters the aquifer over a short time period. The duration is related to the seepage velocity.

At the other extreme, the low conductivity case ($K = 0.5$ m/d) gives a low mass flux from the gasoline lens over a long time period, resulting in a wide xylene concentration history. The peak concentration is relatively high, since the mass flux from the lens enters slowly moving groundwater.

Table 3 Hydraulic Conductivity Effects

Horizontal Hydraulic Conductivity	Effective Xylene Radius	Water Flux Under Legs	Maximum Mass Flux	Peak Conc. at 50 m	Approx Time of Peak
(m/d)	(meters)	(m ³ /d)	(kg/d)	(mg/l)	(days)
0.5	16.6	1.66	0.071	8.90	13700
5.0	16.1	9.02	0.306	12.0	1700
50.0	14.2	68.4	1.980	10.8	200

Conclusions

A methodology has been presented for estimating contaminant exposures resulting from spills of hydrocarbons (L-NAPLs). The KOPT/OILENS model approximates the transport of hydrocarbon and a dissolved groundwater pollutant of interest by kinematic approaches, coupled with approximate dynamic techniques when necessary. Transport along the water table is simulated by the Ghyben-Herzberg-Dupuit assumptions. Transport within the aquifer is simulated by an analytic approach for two-dimensional vertically averaged solute transport, TSGPLUME. The simulation results show tailing of concentrations in the aquifer, because of the distribution of mass flux to the aquifer. The tailing occurs even though equilibrium partitioning is assumed between all phases.

Sensitivity of estimated exposure concentrations was investigated using the models described. The greatest effect was found to be caused by varying the seepage velocity (hydraulic conductivity) of the aquifer. Peak concentration, arrival time and width of the concentration distribution are all dramatically impacted. Spill volume primarily affects the peak concentrations, while the L-NAPL/water partition coefficient affects both peak concentration and peak arrival times. Several of these results involve the effective contaminant radius, which can affect the peak concentration. As the distance to the receptor well increases, this effect becomes insignificant. Thus the assumption of constant size of the lens is relatively unimportant for TSGPLUME simulations. The results of these limited comparisons suggest that the hydraulic conductivity dominates the model results.

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