Modeling Chloride Movement in the Alluvial Aquifer at the Rocky Mountain Arsenal, Colorado

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GEOLOGICAL SURVEY WATER-SUPPLY PAPER 2044

UNITED STATES GOVERNMENT PRINTING OFFICE, WASHINGTON : 1977

UNITED STATES DEPARTMENT OF THE INTERIOR

CECIL D. ANDRUS, *Secretary*

GEOLOGICAL SURVEY

V. E. McKelvey, *Director*

Library of Congress Cataloging in Publication Data
Konikow, Leonard F.
Modeling chloride movement in the alluvial aquifer at the Rocky Mountain Arsenal, Colorado.
(Water-Supply Paper 2044)
Bibliography: p.
Supt. of Docs. no.: I 19.13:2044
1. Groundwater flow—Mathematical models. 2. Groundwater flow—Data processing.
3. Water, Underground—Pollution—Colorado—Adams Co. 4. United States Rocky Mountain Arsenal. 5. Chlorides.
I. Title. II. Series: United States Geological Survey Water-Supply Paper 2044.
GB1197.7K66 551.4'9 76-608368

For sale by the Superintendent of Documents, U.S. Government Printing Office
Washington, D.C. 20402
Stock Number 024-001-02963-4

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CONVERSION FACTORS

English units used in this report may be converted to metric units by the following conversion factors:

To convert English units	*Multiply by*	*To obtain* Metric units
Acres	4.047×10^{-3}	Square kilometers (km^2).
Feet (ft)	.3048	Meters (m).
Feet per year (ft/yr)	.3048	Meters per year (m/yr).
Feet per day (ft/d)	.3048	Meters per day (m/d).
Feet per second per foot ([ft/s]/ft)	1.0	Meters per second per meter ([m/s]/m).
Square feet (ft^2)	.0929	Square meters (m^2).
Feet squared per day (ft^2/d)	.0929	Meters squared per day (m^2/d).
Cubic feet per second (ft^3/s)	2.832×10^{-2}	Cubic meters per second (m^3/s).
Cubic feet per second per mile ([ft^3/s]/mi).	1.760×10^{-2}	Cubic meters per second per kilometer ([m^3/s]/km).
Miles (mi)	1.609	Kilometers (km).
Square miles (mi^2)	2.590	Square kilometers (km^2).

MODELING CHLORIDE MOVEMENT IN THE ALLUVIAL AQUIFER AT THE ROCKY MOUNTAIN ARSENAL, COLORADO

By LEONARD F. KONIKOW

ABSTRACT

A solute-transport model that can be used to predict the movement of dissolved chemicals in flowing ground water was applied to a problem of ground-water contamination at the Rocky Mountain Arsenal, near Denver, Colo. The model couples a finite-difference solution to the ground-water flow equation with the method-of-characteristics solution to the solute-transport equation.

From 1943 to 1956 liquid industrial wastes containing high chloride concentrations were disposed into unlined ponds at the Arsenal. Wastes seeped out of the unlined disposal ponds and spread for many square miles in the underlying shallow alluvial aquifer. Since 1956 disposal has been into an asphalt-lined reservoir, which contributed to a decline in ground-water contamination by 1972. The simulation model quantitatively integrated the effects of the major factors that controlled changes in chloride concentrations and accurately reproduced the 30-year history of chloride ground-water contamination.

Analysis of the simulation results indicates that the geologic framework of the area markedly restricted the transport and dispersion of dissolved chemicals in the alluvium. Dilution, from irrigation recharge and seepage from unlined canals, was an important factor in reducing the level of chloride concentrations downgradient from the Arsenal. Similarly, recharge of uncontaminated water from the unlined ponds since 1956 has helped to dilute and flush the contaminated ground water.

INTRODUCTION

The contamination of a ground-water resource is a serious problem that can have long-term economic and physical consequences and might not be easily remedied. Although the prevention of ground-water contamination provides the most satisfactory result (Wood, 1972), the capability to predict the movement of dissolved chemicals in flowing ground water is also needed in order to (1) plan and design projects to minimize ground-water contamination, (2) estimate spatial and temporal variations of chemical concentrations, (3) estimate the traveltime of a contaminant from its source to a ground-water sink (a discharge point, such as a stream, spring, or well), (4) help design an effective and efficient monitoring system, and (5) help evaluate the physical and economic feasibility of alternative reclamation plans for removing contaminants from an aquifer and (or) preventing the contaminants from spreading.

Reliable predictions of contaminant movement can be made only if we understand the processes controlling convective transport, hydrodynamic dispersion, and chemical reactions that affect the dissolved chemicals in ground water, and if these processes can be accurately represented in a systematic model. For a model to be usable in a variety of hydrogeologic situations, the modeling technique must be accurate, functional, and transferable. Because aquifers generally have heterogeneous properties and complex boundary conditions, quantitative predictions would appear to require the use of a deterministic, distributed parameter, digital simulation model.

This study is part of the U.S. Geological Survey's Subsurface Waste Program, the objective of which is to appraise the impact of waste disposal on the Nation's water resources. The main objective of this study was to demonstrate the applicability of the method-of-characteristics model to a problem of conservative (nonreacting) contaminant movement through an alluvial aquifer. By studying a field problem in which the effects of reactions are negligible, the effects of other processes that affect solute transport may be isolated and described more accurately. This study should serve as a basis for investigating more complex systems whose chemical reactions are significant and interact with the other processes. The purposes of this report are (1) to briefly describe the general simulation model and (2) to demonstrate its application to a complex field problem.

Because convective transport and hydrodynamic dispersion depend on the velocity of ground-water flow, the mathematical simulation model must solve two simultaneous partial differential equations. One is the equation of flow, from which ground-water velocities are obtained, and the second is the solute-transport equation, describing the chemical concentration in the ground water. Three general classes of numerical methods have been used to solve these partial differential equations: finite-difference methods, finite-element methods, and the method of characteristics. Each method has some advantages, disadvantages, and special limitations for application to field problems.

SELECTION OF STUDY AREA

The field area selected for this study is in and adjacent to the Rocky Mountain Arsenal, near Denver, Colo. (See fig. 1.) A 30-year history of ground-water contamination in this area is related to the disposal of liquid industrial wastes into ponds (Petri, 1961; Walker, 1961; Walton, 1961). The Rocky Mountain Arsenal area is well suited for this study

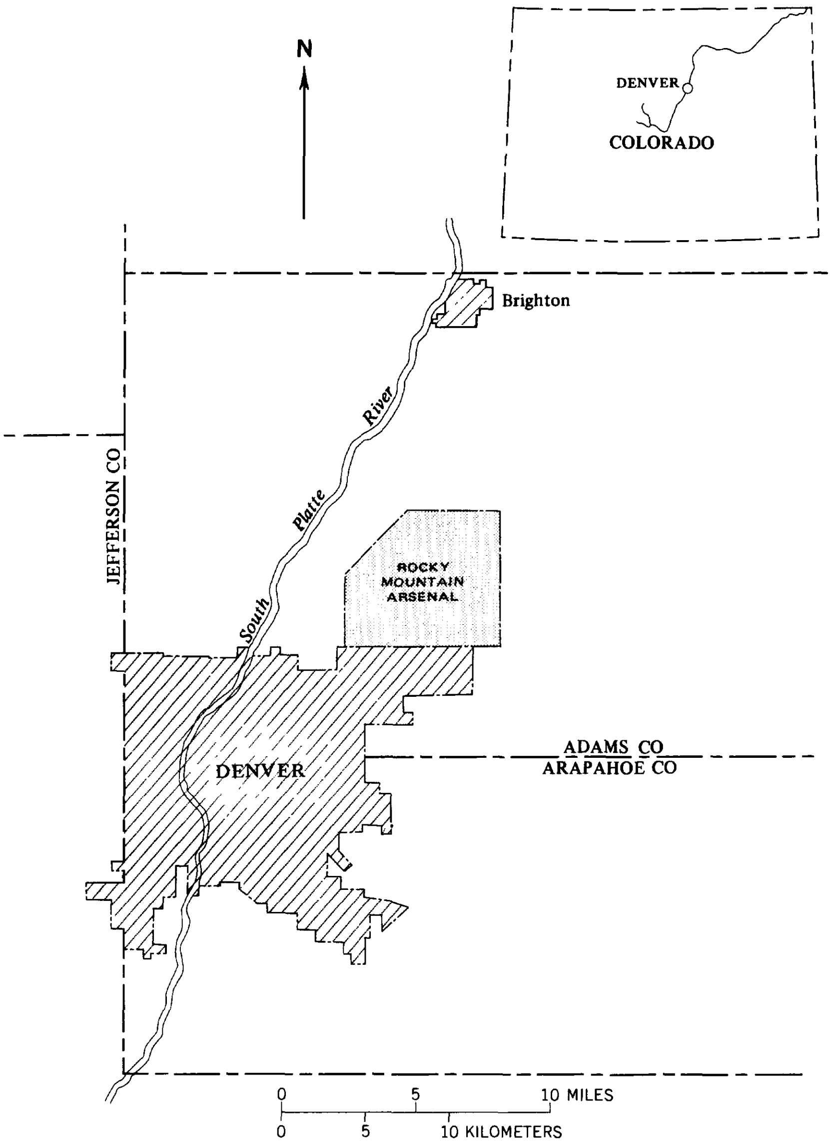


FIGURE 1.— Location of study area.

because (1) the geology and hydrology of the area are well known, (2) adequate, though limited, water-quality data are available to calibrate the mathematical model, and (3) the history of liquid waste-disposal operations at the Arsenal can be approximately reconstructed. Furthermore, the waste water has a very high chloride concentration, which can serve as a conservative tracer.

PROCEDURE OF INVESTIGATION

This investigation was conducted in three distinct phases. During the first phase, all available data were collected, interpreted, and analyzed to produce accurate, comprehensive, and quantitative descriptions for the alluvial aquifer of its (1) geologic properties and boundaries, (2) hydraulic properties, boundaries, and stresses, and (3) chemical sources and distributions over space and time. Many of the geologic and hydraulic interpretations were presented by Konikow (1975). Most chemical data are presented in this report.

During the second phase of the investigation, a steady-state flow model was developed to estimate recharge rates to the aquifer and to compute ground-water flow velocities. In the third phase of the investigation, the solute-transport model was calibrated to reproduce the observed history of ground-water contamination at the Rocky Mountain Arsenal. Much of the output from the flow model was used as input to the solute-transport model.

ACKNOWLEDGMENTS

John D. Bredehoeft, U.S. Geological Survey, and George F. Pinder, formerly with the Survey and now at Princeton University, jointly developed the original version of the solute-transport model used in this study. J.D. Bredehoeft was also instrumental both in the further development of this model and in the selection of the area for this study. Their work is gratefully acknowledged. Many data were supplied by the Rocky Mountain Arsenal, the U.S. Army Corps of Engineers, and the Colorado Department of Health, and their assistance also is appreciated.

SIMULATION MODEL

BACKGROUND

The purpose of the simulation model is to compute the concentration of a dissolved chemical species in an aquifer at any specified place and time. Changes in chemical concentration occur within a dynamic ground-water system primarily due to four distinct processes:

1. Convective transport, in which dissolved chemicals are moving with the flowing ground water.
2. Hydrodynamic dispersion, in which molecular and ionic diffusion and small-scale variations in the velocity of flow through the porous media cause the paths of dissolved molecules and ions to diverge or spread from the average direction of ground-water flow.
3. Mixing (or dilution), in which water of one composition is introduced into water of a different composition.

4. Reactions, in which some amount of a particular dissolved chemical species may be added to or removed from the ground water due to chemical and physical reactions in the water or between the water and the solid aquifer materials.

The model presented in this report assumes that no reactions occur that affect the concentration of the species of interest and that the density and viscosity of the water are constant and independent of the concentration. Robertson (1974) expanded the model to include the effects of radioactive decay and ion exchange with a linear adsorption isotherm.

The modeling technique used in this study couples an implicit finite-difference procedure to solve the flow equation and the method of characteristics to solve the solute-transport equation. The applicability of this (or any other) type of model to complex field problems can only be demonstrated by first testing it for a variety of field conditions in which observed records of contaminant movement can be compared with concentration changes computed by the model. In this manner, the accuracy, limitations, and efficiency of the method can be shown for a wide range of problems. Also, calibrating the model in an area for which historical data are available will provide insight into the use of the model in areas where few or no data are available.

FLOW EQUATION

By following the derivation of Pinder and Bredehoeft (1968), the equation describing the transient two-dimensional flow of a homogeneous compressible fluid through a nonhomogeneous anisotropic aquifer may be written in cartesian tensor notation as:

$$\frac{\partial}{\partial x_i}\left(T_{ij}\frac{\partial h}{\partial x_j}\right)=S\frac{\partial h}{\partial t}+W(x,y,t), \qquad i,j=1,2, \quad (1)$$

where

T_{ij} is the transmissivity tensor, L^2/T;
h is the hydraulic head, L;
S is the storage coefficient, L^0;
t is the time, T;
W is the volume flux per unit area, L/T; and
x, y are cartesian coordinates.

If we only consider fluxes of (1) direct withdrawal or recharge, such as well pumpage, well injection, or evapotranspiration, and (2) steady leakage into or out of the aquifer through a confining layer, streambed, or lake bed, then $W(x,y,t)$ may be expressed as:

$$W(x,y,t)=Q(x,y,t)-\frac{K_z}{m}(H_s-h), \quad (2)$$

where

Q is the rate of withdrawal (positive sign) or recharge (negative sign), L/T;

K_z is the vertical hydraulic conductivity of the confining layer, streambed, or lake bed, L/T;

m is the thickness of the confining layer, streambed, or lake bed, L; and

H_s is the hydraulic head in the source bed, stream, or lake, L.

Lohman (1972) showed that an expression for the average seepage velocity of ground water can be derived from Darcy's Law. This expression can be written in cartesian tensor notation as:

$$V_i = -\frac{K_{ij}}{n}\frac{\partial h}{\partial x_j}, \tag{3}$$

where

V_i is the seepage velocity in the direction of x_i, L/T;

K_{ij} is the hydraulic conductivity tensor, L/T; and

n is the effective porosity of the aquifer, L^0.

TRANSPORT EQUATION

The equation used to describe the two-dimensional transport and dispersion of a given dissolved chemical species in flowing ground water was derived by Reddell and Sunada (1970), Bear (1972), and Bredehoeft and Pinder (1973) and may be written as:

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial x_i}\left(D_{ij}\frac{\partial C}{\partial x_j}\right) - \frac{\partial}{\partial x_i}(CV_i) - \frac{C'W}{nb} + \sum_{k=1}^{s} R_k, \qquad i,j = 1,2, \tag{4}$$

where

C is the concentration of the dissolved chemical species, M/L^3;

D_{ij} is the dispersion tensor, L^2/T;

b is the saturated thickness of the aquifer, L;

C' is the concentration of the dissolved chemical in a source or sink fluid, M/L^3; and

R_k is the rate of production of the chemical species in reaction k of s different reactions, M/L^3T.

The first term on the right side of equation 4 represents the change in concentration due to hydrodynamic dispersion and is assumed to be proportional to the concentration gradient. The second term describes the effects of convective transport, and the third term represents a fluid source or sink. The fourth term, which describes chemical reactions, must be written explicitly for all reactions affecting the chemical species of interest. This term may be eliminated from equation 4 for the case of a conservative (nonreactive) species.

DISPERSION COEFFICIENT

The dispersion coefficient may be related to the velocity of ground-water flow and to the nature of the aquifer using Scheidegger's (1961) equation:

$$D_{ij} = \alpha_{ijmn} \frac{V_m V_n}{|V|}, \tag{5}$$

where

α_{ijmn} is the dispersivity of the aquifer, L;
V_m and V_n are components of velocity in the m and n directions, L/T; and
$|V|$ is the magnitude of the velocity, L/T.

Scheidegger (1961) further showed that, for an isotropic aquifer, the dispersivity tensor can be defined in terms of two constants. These are the longitudinal and transverse dispersivities of the aquifer (α_1 and α_2, respectively). These are related to the longitudinal and transverse dispersion coefficients by

$$D_L = \alpha_1 |V|, \tag{6}$$

and

$$D_T = \alpha_2 |V|. \tag{7}$$

After expanding equation 5, substituting Scheidegger's identities, and eliminating terms with coefficients that equal zero, the components of the dispersion coefficient for two-dimensional flow in an isotropic aquifer may be stated explicitly as:

$$D_{xx} = D_L \frac{(V_x)^2}{|V|^2} + D_T \frac{(V_y)^2}{|V|^2}, \tag{8}$$

$$D_{yy} = D_T \frac{(V_x)^2}{|V|^2} + D_L \frac{(V_y)^2}{|V|^2}, \tag{9}$$

$$D_{xy} = D_{yx} = (D_L - D_T) \frac{V_x V_y}{|V|^2}. \tag{10}$$

NUMERICAL METHODS

Because aquifers have variable properties and complex boundary conditions, exact solutions to the partial differential equations of flow (eq 1) and solute transport (eq 4) cannot be obtained directly. Therefore, an approximate numerical method must be employed.

Pinder and Bredehoeft (1968) showed that if the coordinate axes are aligned with the principal directions of the transmissivity tensor, equation 1 may be approximated by the following implicit finite-difference equation:

$$T_{xx[i-(1/2),j]}\left[\frac{h_{i-1,j,k}-h_{i,j,k}}{(\Delta x)^2}\right]$$
$$+\,T_{xx[i+(1/2),j]}\left[\frac{h_{i+1,j,k}-h_{i,j,k}}{(\Delta x)^2}\right]$$
$$+\,T_{yy[i,j-(1/2)]}\left[\frac{h_{i,j-1,k}-h_{i,j,k}}{(\Delta y)^2}\right]$$
$$+\,T_{yy[i,j+(1/2)]}\left[\frac{h_{i,j+1,k}-h_{i,j,k}}{(\Delta y)^2}\right]$$
$$=S\left[\frac{h_{i,j,k}-h_{i,j,k-1}}{\Delta t}\right]$$
$$+\,\frac{q_{w(i,j)}}{\Delta x\Delta y}-\frac{K_z}{m}\left[H_{s(i,j)}-h_{i,j,k-1}\right], \quad (11)$$

where

i,j,k are indices in the x, y, and time dimensions, respectively;

Δx, Δy, Δt are increments in the x, y, and time dimensions, respectively; and

q_w is the volumetric rate of withdrawal or recharge at the (i,j) node, L^3/T.

The numerical solution of the finite-difference equation requires that the area of interest be subdivided into small rectangular cells, which constitute a finite-difference grid. The finite-difference equation is solved numerically, using an iterative alternating-direction implicit procedure described by Pinder (1970) and Prickett and Lonnquist (1971).

After the head distribution has been computed for a given time step, the velocity of ground-water flow is computed at each node, using an explicit finite-difference form of equation 3. For example, the velocity in the x direction at node (i,j) would be computed as:

$$V_{x\,(i,j)} = \frac{T_{xx(i,j)}}{nb_{i,j}} \frac{(h_{i-1,j,k} - h_{i+1,j,k})}{2\Delta x} . \tag{12}$$

A similar expression is used to compute the velocity in the y direction.

The method of characteristics presented by Garder, Peaceman, and Pozzi (1964) is used to solve the solute-transport equation (eq 4). The development and application of this technique in ground-water problems has been presented by Pinder and Cooper (1970), Reddell and Sunada (1970), Bredehoeft and Pinder (1973), Konikow and Bredehoeft (1974), Robertson (1974), and Robson (1974). The method actually solves a system of ordinary differential equations that is equivalent to the partial differential equation (eq 4) that describes solute transport.

The numerical solution is achieved by introducing a set of moving points that can be traced with reference to the stationary coordinates of the finite-difference grid. Each point has a concentration associated with it and is moved through the flow field in proportion to the flow velocity at its location. The moving points simulate convective transport because the concentration at each node of the finite-difference grid changes as different points enter and leave its area of influence. Then, the additional change in concentration due to dispersion and to fluid sources is computed by solving an explicit finite-difference equation. In this study, four points were initially distributed in each cell of the grid.

BOUNDARY CONDITIONS

Several different types of boundary conditions can be represented in the simulation model. These include:

1. *No-flow boundary*: By specifying a transmissivity equal to zero at a given node, no flow can occur across the boundary of that cell of the finite-difference grid. The numerical method used in this model also requires that the outer rows and columns of the finite-difference grid have zero transmissivities.
2. *Constant-head boundary*: Where the head in the aquifer will not change with time, a constant-head condition is maintained by specifying a very high value of leakance (1.0 [ft/s]/ft or [m/s]/m), which is the ratio of the vertical hydraulic conductivity to the thickness of the confining layer, streambed, or lake bed. The rate of leakage is then a function of the difference between the head

of the aquifer and the head in the source bed, stream, or lake and is computed implicitly by the model.

3. *Constant flux*: A constant rate of withdrawal or recharge may be specified for any node in the model.

At any boundary that acts as a source of water to the aquifer, the chemical concentration of the source must also be defined.

DESCRIPTION OF STUDY AREA

HISTORY OF CONTAMINATION

The Rocky Mountain Arsenal has been operating since 1942, primarily manufacturing and processing chemical warfare products and pesticides. These operations have produced liquid wastes that contain complex organic and inorganic chemicals, including a characteristically high chloride concentration that apparently ranged up to about 5,000 mg/l (milligrams per liter).

The liquid wastes were disposed into several unlined ponds (fig. 2), resulting in the contamination of the underlying alluvial aquifer. On the basis of available records, it is assumed that contamination first occurred at the beginning of 1943. From 1943 to 1956 the primary disposal was into pond A. Alternate and overflow discharges were collected in ponds B, C, D, and E.

Much of the area north of the Arsenal is irrigated, both with surface water diverted from one of the irrigation canals, which are also unlined, and with ground water pumped from irrigation wells. Damage to crops irrigated with shallow ground water was observed in 1951, 1952, and 1953 (Walton, 1961). Severe crop damage was reported during 1954, a year when the annual precipitation was about one-half the normal amount, and ground-water use was heavier than normal (Petri, 1961).

Several investigations have been conducted since 1954 to determine both the cause of the problem and how to prevent further damages. Petri and Smith (1956) showed that an area of contaminated ground water of several square miles existed north and northwest of the disposal ponds. These data clearly indicated that the liquid wastes seeped out of the unlined disposal ponds, infiltrated the underlying alluvial aquifer, and migrated downgradient toward the South Platte River. To prevent additional contaminants from entering the aquifer, a 100-acre (0.405 km^2) evaporation pond (Reservoir F) was constructed in 1956, with an asphalt lining to hold all subsequent liquid wastes (Engineering News-Record, Nov. 22, 1956).

In 1973 and 1974 there were new (and controversial) claims of crop and livestock damages allegedly caused by ground water that was contaminated at the Arsenal (The Denver Post, Jan. 22, 1973; May 12,

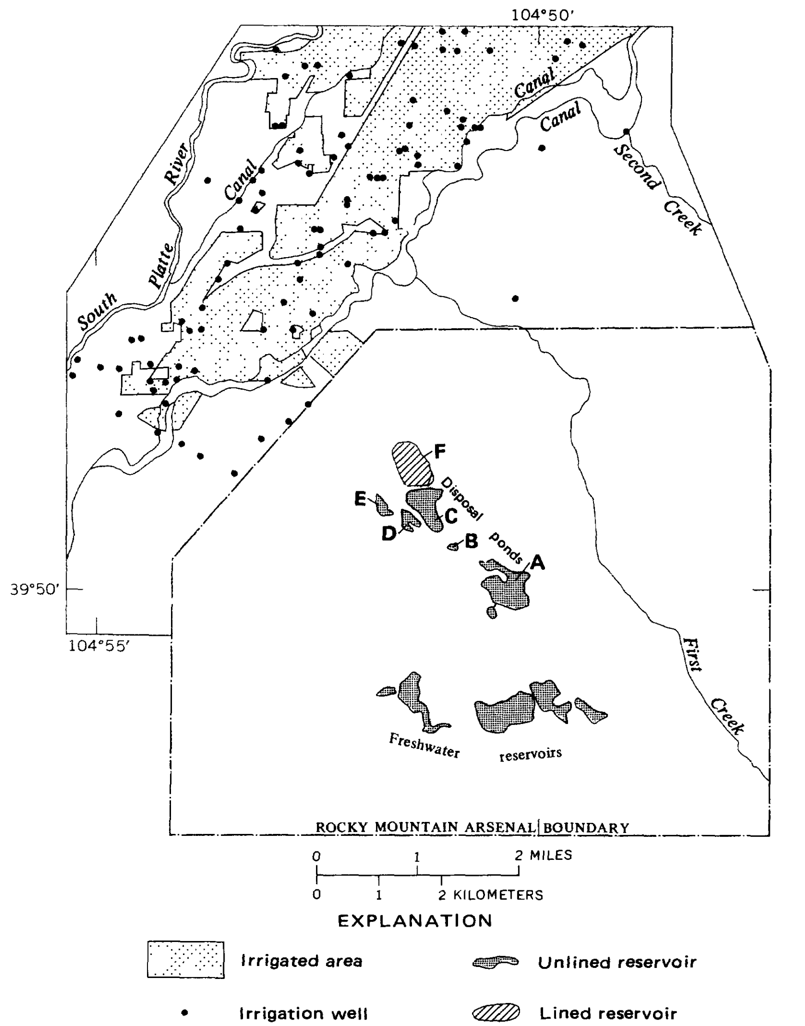


FIGURE 2.— Major hydrologic features. Letters indicate disposal-pond designations assigned by the U.S. Army.

1974; May 23, 1974). Recent data collected by the Colorado Department of Health (Shukle, 1975) have shown that DIMP (Diisopropylmethylphosphonate), a nerve-gas byproduct about which relatively little is known, has been detected at a concentration of 0.57 ppb (parts per billion) in a well located approximately 8 miles (12.9 km) downgradient from the disposal ponds and 1 mile (1.6 km) upgradient from 2 municipal water-supply wells of the City of Brighton. A DIMP concentration of 48 ppm (parts per million), which is nearly

100,000 times higher, was measured in a ground-water sample collected near the disposal ponds. Other contaminants detected in wells or springs in the area include DCPD (Dicyclopentadiene), endrin, aldrin, and dieldrin.

CONTAMINATION PATTERN

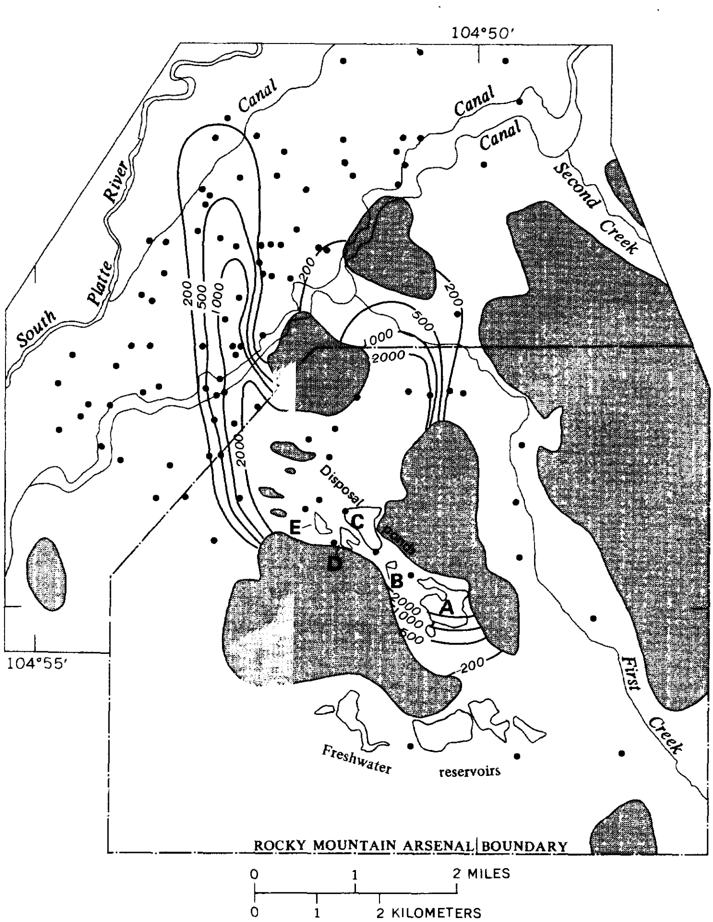
Since 1955 more than 100 observation wells and test holes have been constructed to monitor changes in water quality and water levels in the alluvial aquifer. The areal extent of contamination has been mapped on the basis of chloride concentrations in wells, which ranged from normal background concentrations of about 40 to 150 mg/l to about 5,000 mg/l in contaminated ground water near pond A.

Data collected during 1955–56 indicate that one main plume of contaminated water extended beyond the northwestern boundary of the Arsenal and that a small secondary plume extended beyond the northern boundary. (See fig. 3.) However, the velocity distribution computed from the water-table map available at that time (Petri and Smith, 1956) could not, in detail, account for the observed pattern of spreading from the sources of contamination. Because contaminant transport depends upon flow, the prediction of concentration changes requires the availability of accurate, comprehensive, and quantitative descriptions for the aquifer of its hydraulic properties, boundaries, and stresses.

HYDROGEOLOGY

The records of about 200 observation wells, test holes, irrigation wells, and domestic wells were compiled, analyzed, and sometimes reinterpreted to describe the hydrogeologic characteristics of the alluvial aquifer in and adjacent to the Rocky Mountain Arsenal. Konikow (1975) presented four maps that show the configuration of the bedrock surface, generalized water-table configuration, saturated thickness of alluvium, and transmissivity of the aquifer. These maps show that the alluvium forms a complex, nonuniform, sloping, discontinuous, and heterogeneous aquifer system.

A map showing the general water-table configuration for 1955–71 is presented in figure 4. The assumptions and limitations of figure 4 were discussed in more detail by Konikow (1975). Perhaps the greatest change from previously available maps is the definition of areas in which the alluvium either is absent or is unsaturated most of the time. These areas form internal barriers that significantly affect ground-water flow patterns within the aquifer. The contamination pattern shown in figure 3 clearly indicates that the migration of dissolved chloride in this aquifer was also significantly constrained by the aquifer boundaries.



EXPLANATION

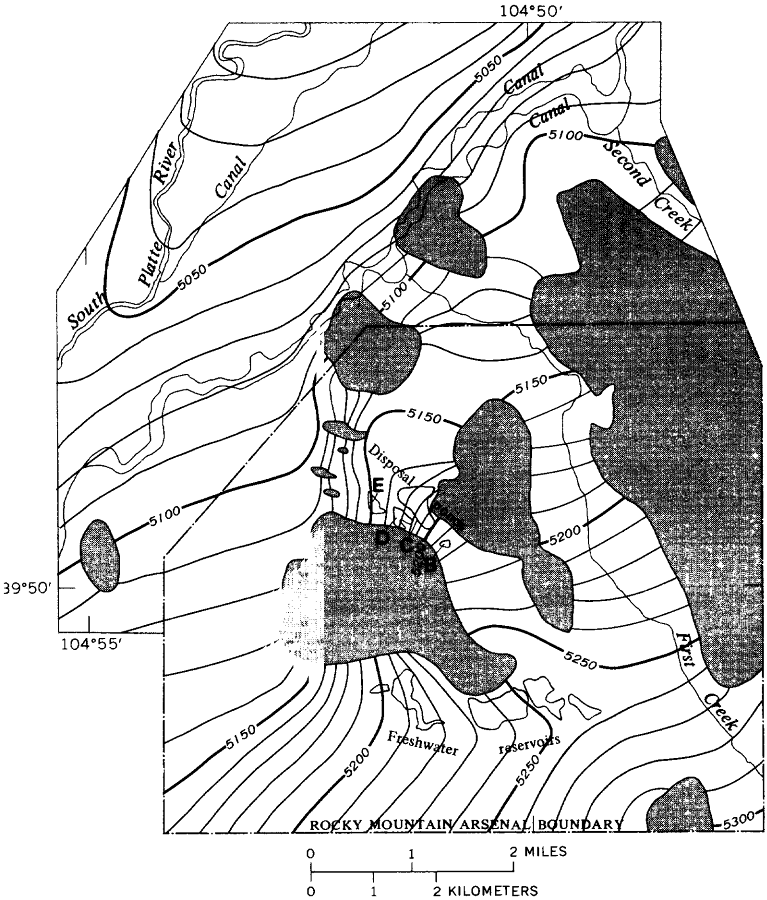
• Data point (Sept. 1955–March 1956)

—500— Line of aqual chloride concentration (in milligrams per liter). Interval variable

Araa in which alluvium is absent or unsaturated

FIGURE 3.— Observed chloride concentration, 1956.

The general direction of ground-water movement is from regions of higher water-table altitudes to those of lower water-table altitudes and is approximately perpendicular to the water-table contours. Deviations from the general flow pattern inferred from water-table contours may occur in some areas because of local variations in aquifer properties, recharge, or discharge. The nonorthogonality at places between water-table contours and aquifer boundaries indicates that the approximate limit of the saturated alluvium does not consistently represent a no-flow boundary, but that, at some places, there



EXPLANATION

—5100— WATER-TABLE CONTOUR — Shows approximate altitude of water table, 1955–71. Contour interval 10 feet (3 meters). Datum is mean sea level

Area in which alluvium is absent or unsaturated

FIGURE 4.— General water-table configuration in the alluvial aquifer in and adjacent to the Rocky Mountain Arsenal, 1955-71.

may be significant flow across this line. Such a condition can readily occur in areas where the bedrock possesses significant porosity and hydraulic conductivity, or where recharge from irrigation, unlined canals, or other sources is concentrated. Because the hydraulic conductivity of the bedrock underlying the alluvium is generally much lower than that of the alluvium, ground-water flow through the bedrock was assumed to be negligible for the purposes of this investigation.

The position of the boundary that separates the alluvial aquifer from the areas in which the alluvium is either absent or unsaturated may actually change with time as the water table rises or falls in response to changes in recharge and discharge, although the boundary was assumed to remain stationary for the model study. The effect of the changing boundary was most evident in the vicinity of pond A. A map of the water-table configuration during the period when pond A was full (Konikow, 1976) shows that during this time, there was ground-water flow from pond A to the east and northeast into the alluvial channel underlying the valley of First Creek, in addition to the northwestward flow indicated in figure 4.

APPLICATION OF SIMULATION MODEL

FINITE-DIFFERENCE GRID

The limits of the modeled area were selected to include the entire area having chloride concentrations over 200 mg/l and the areas downgradient to which the contaminants would likely spread, and to closely coincide with natural boundaries and divides in the ground-water flow system. The model includes an area of approximately 34 mi^2 (88 km^2).

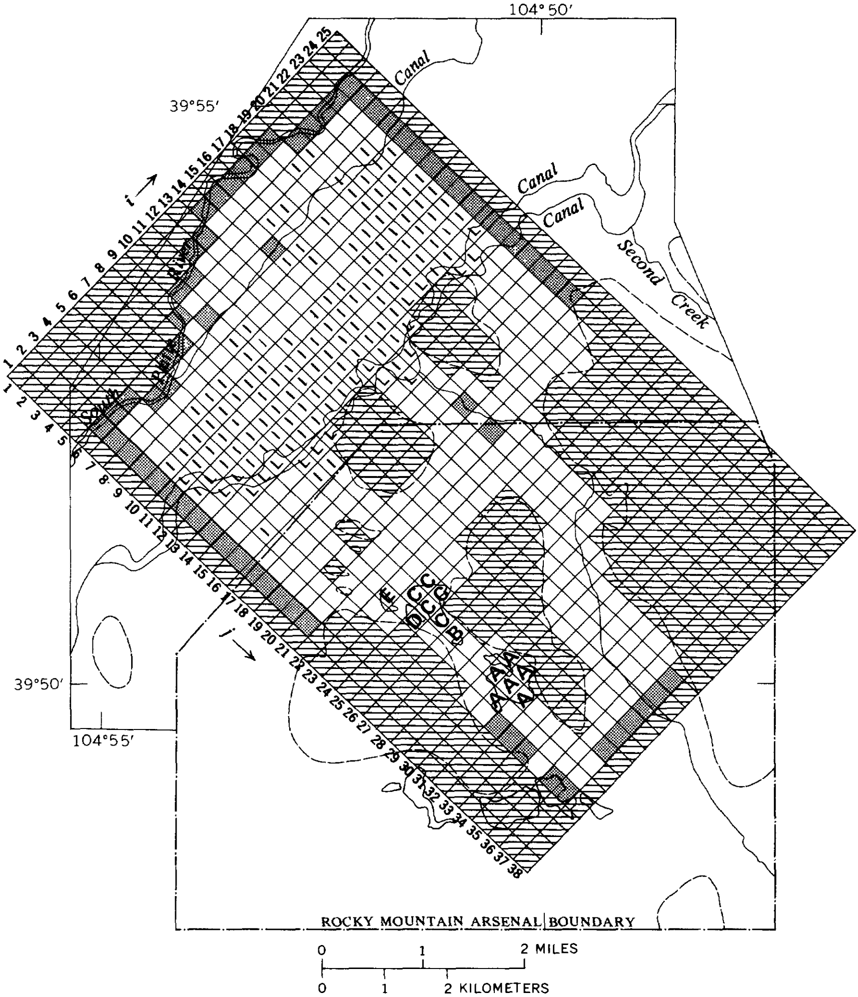
The modeled area was subdivided into a finite-difference grid of uniformly spaced squares. (See fig. 5.) The grid contains 25 columns (i) and 38 rows (j). Because of the irregular boundaries and discontinuities of the alluvial aquifer, only 516 of the total 950 nodes in the grid were actually used to compute heads (or water-table altitudes) in the aquifer. Each cell of the grid is 1,000 feet (305 m) on each side. By convention, nodes are located at the centers of the cells of the grid. All aquifer properties and stresses must be defined at all nodes of the grid.

DATA REQUIREMENTS

Many factors influence the flow of ground water and its dissolved chemicals through the alluvial aquifer near the Rocky Mountain Arsenal. To compute changes in chloride concentration, all parameters and coefficients incorporated into equations 1 and 4 must be defined. Thus, many input data are required for the model, and the accuracy of these data will affect the reliability of the computed results. The main input data requirements for modeling chloride movement in this alluvial aquifer are summarized in table 1.

AQUIFER PROPERTIES

The transmissivity of an aquifer reflects the rate at which ground water will flow through the aquifer under a unit hydraulic gradient (Lohman and others, 1972). Konikow (1975) showed that the



EXPLANATION

Zero-transmissivity cell

Constant-head cell

A Disposal-pond cell. Letter corresponds to designation in figure 2

I Irrigation-recharge cell

L Canal-leakage cell

——— Boundary of areas in which alluvium is absent or unsaturated

FIGURE 5.— Finite-difference grid used to model the study area.

transmissivity of the alluvial aquifer in this study area ranges from 0 to over 20,000 ft²/d (over 1,800 m²/d), and that the saturated thickness is generally less than 60 feet (18 m). The highest transmissivities,

TABLE 1. — *Summary of main data requirements for numerical model*

Aquifer properties	Aquifer stresses
Transmissivity	Ground-water withdrawals
Storage coefficient	Irrigation recharge[1]
Saturated thickness	Canal leakage[1]
Effective porosity	Disposal-pond leakage[1]
Dispersivity	
Boundaries	
Initial chloride concentration	

[1]Quantity and quality must be defined.

greatest saturated thicknesses, and lowest hydraulic gradients generally occur near the South Platte River in the northwestern part of the modeled area. The finite-difference grid was superimposed on the maps of transmissivity and saturated thickness presented by Konikow (1975), and corresponding values were determined for each node of the grid.

The storage coefficient of the aquifer is an approximate measure of the relation between changes in the amount of water stored in the aquifer and changes in head. Because no changes in head with time occur in steady-state flow, a value for this parameter is needed only for an analysis of transient (time-dependent) flow, which was not considered in this study.

Values of effective porosity and dispersivity of the aquifer must be known to solve the solute-transport equation. Because no field data are available to describe these parameters in this study area, values were selected by using a trial-and-error adjustment within a range of values determined for similar aquifers in other areas.

No-flow and constant-head boundaries used in this model are indicated in figure 5. Constant-head boundaries were specified where it was believed that either underflow into or out of the modeled area or recharge was sufficient to maintain a nearly constant water-table altitude at that point in the aquifer. Altitudes assigned to the constant-head cells were determined by superimposing the finite-difference grid (fig. 5) on the water-table map (fig. 4).

No data were available to describe the chloride concentrations in the aquifer when the Arsenal began its operations. Because more recent measurements indicated that the normal background concentration may be as low as 40 mg/l, an initial chloride concentration of 40 mg/l was assumed to have existed uniformly throughout the aquifer in 1942.

AQUIFER STRESSES

No direct measurements of long-term aquifer stresses were available. Hence, these factors were estimated, primarily using a mass-balance analysis of the observed flow field.

The areas that had probably been irrigated during most of the period from 1943 to 1972 were mapped from aerial photographs. These irrigated areas are shown in figure 2. In the model, irrigation was assumed to occur at 111 nodes of the finite-difference grid, which represents an area of 1.11×10^8 ft^2 (1.03×10^7 m^2).

The net rate of recharge from irrigation and precipitation on irrigated areas was estimated through a trial-and-error analysis, in which the simulation model was used to compute the water-table configuration for various assumed recharge rates. Initial estimates of net recharge were used in a preliminary calibration of the model. Transmissivity values and boundary conditions in the model were adjusted between successive simulations with an objective of minimizing the differences between observed and computed water-table altitudes in the irrigated area. The standard error of estimate (or scatter) is a statistical measure similar to the standard deviation (Croxton, 1953, p. 119). It is used here to indicate the extent of deviations between computed and observed heads. Figure 6 shows that the standard error of estimate generally decreased as successive simulation tests were made. After about seven tests, additional adjustments produced only small improvements in the fit between the observed and computed water tables.

A final estimate of the net recharge rate in irrigated areas was made using the set of parameters developed for the final test of figure 6. Figure 7 shows that the mean of the differences between observed and computed heads at all nodes in the irrigated area is minimized (equal to zero) when a net recharge rate of approximately 1.54 ft/yr (0.47 m/yr) is assumed. Also, irrigation recharge was assumed to have a chloride concentration of 100 mg/l.

The recharge rate due to leakage from unlined canals was similarly estimated to be approximately 2.37 ft/yr (0.72 m/yr), which is equivalent to 0.40 [ft^3/s]/mi (0.0070 [m^3/s]/km). The standard error of estimate in this case was about 1.3 feet (0.40 m). Canal leakage was assumed to have a chloride concentration of 40 mg/l.

Changes in the chemical concentration of ground water in irrigated areas are partly caused by the mixing (or dilution) of ground water having one concentration with recharged water having a different concentration. Because the magnitude of this change is a function of the gross recharge, rather than of the net recharge, an estimate of the gross recharge must be made. Hurr, Schneider, and Minges (1975) presented data indicating that the average rate of application of irrigation water in the South Platte River valley is about 4.2 ft/yr (1.3

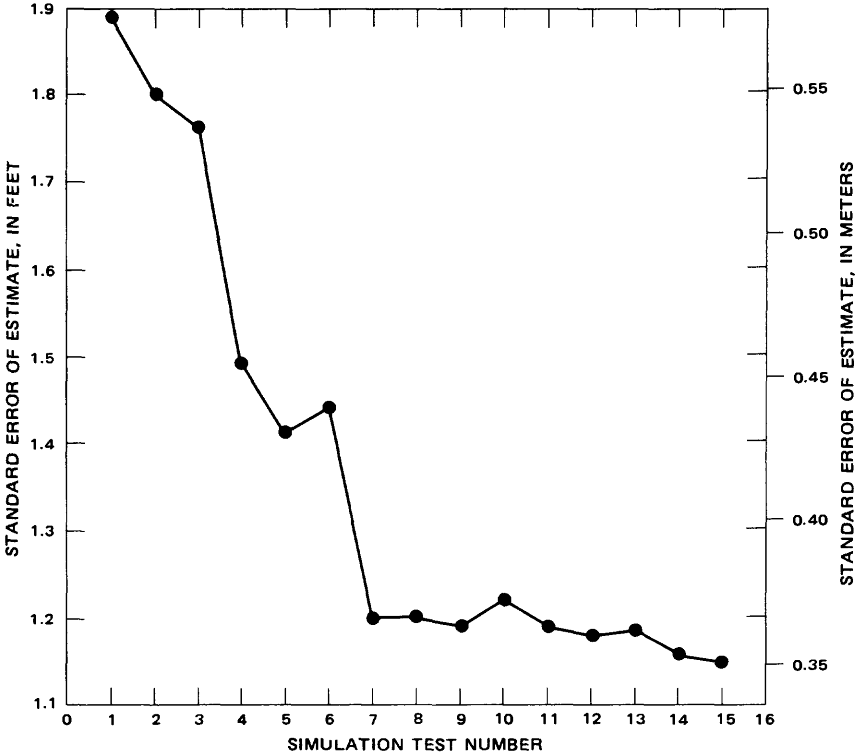


FIGURE 6.—Change in standard error of estimate for succesive simulation tests.

m/yr). Hurr, Schneider, and Minges (1975) also stated that 45 to 50 percent of the applied irrigation water is recharged to the aquifer. Thus, the gross recharge to the aquifer in irrigated parts of the study area was assumed to equal 1.9 ft/yr (0.58 m/yr).

In the study area irrigation water is derived both from surface water, diverted through canals and ditches, and from ground water, pumped from irrigation wells. The difference between the gross recharge and the net recharge, 0.35 ft/yr (0.11 m/yr), was assumed to equal the total ground-water withdrawal rate through wells.

It was estimated from data presented by Schneider (1962) and McConaghy, Chase, Boettcher, and Major (1964) that 62 irrigation wells operated in the study area during 1955–71, the period represented by the water-table map (fig. 4). Only a small number of wells were drilled after 1965 (Hurr and others, 1975, p. 5), so the estimate based on data up to 1964 is probably an accurate approximation. By multiplying the total ground-water withdrawal rate by the irrigated area and then dividing by the number of irrigation wells, the average sustained pumping rate per well is computed to be 0.02 ft^3/s (5.7×10^{-4} m^3/s).

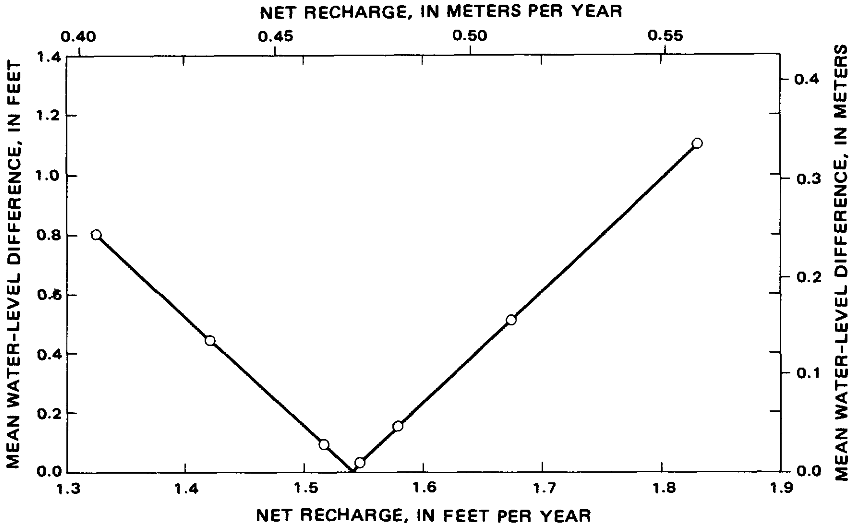


FIGURE 7.— Relation between the assumed rate of net recharge in irrigated areas and the mean difference of observed and computed water levels.

Leakage from the unlined disposal ponds at the Arsenal represents both a significant source of recharge to the aquifer and the primary source of ground-water contamination in the area. Because no records were available to describe the variations in discharge of liquid wastes to the 5 unlined ponds, the general history of their operation was reconstructed primarily from an analysis of aerial photographs, which were available in 20 sets with varying degrees of coverage during 1948–71. The summary in table 2 shows that four characteristic subperiods were identified during which the leakage rates and concentrations were assumed to remain constant for modeling purposes.

CALIBRATION OF FLOW MODEL

The flow model computes the head distribution (water-table altitudes) in the aquifer on the basis of the specified aquifer properties, boundaries, and hydraulic stresses. Because the ground-water seepage velocity is determined from the head distribution, and because both convective transport and hydrodynamic dispersion are functions of the seepage velocity, an accurate model of ground-water flow is a prerequisite to developing an adequate and reliable solute-transport model. In general, the flow model was calibrated by comparing observed water-table altitudes with corresponding computations of the model.

Insufficient field data were available to accurately calibrate a transient-flow model. However, the use of the disposal ponds varied over

TABLE 2. — *Generalized history of disposal pond operations at the Rocky Mountain Arsenal, 1943–72*

[N.A. not applicable]

Years	Average use	Computed leakage (ft^3/s)	Assumed chloride concentration (mg/l)
1943–56	A — Full	0.16	4,000
	B,D,E — Full	.18	3,000
	C — 1/2 Full	.54	3,000
1957–60	A — Empty	.0	N.A.
	B,D,E — Empty	.0	N.A.
	C — Full	1.08	1,000
1961–67	A — Empty	.0	N.A.
	B,D,E — 1/3 Full	.06	500
	C — 1/3 Full	.36	500
1968–72	A — Empty	.0	N.A.
	B,D,E — Empty	.0	N.A.
	C — Full	1.08	150

time and induced the only significant transient changes noted in the area. Several water-level measurements in observation wells at the Arsenal showed that the water table fluctuated locally by up to 20 feet (6 m), mainly in response to filling and emptying of the unlined ponds. Therefore, the hydraulic history of the aquifer was approximated by simulating four separate steady-flow periods, based on the generalized history of disposal pond operations shown in table 2.

The first period simulated was 1968–72, when it was assumed that pond C was full and that all other unlined ponds were empty. Constant-head boundary conditions were applied at the 5 nodes corresponding with pond C, and the rate of leakage from pond C was computed implicitly by the model to be about 1.08 ft^3/s (0.031 m^3/s). A comparison of the heads computed for 1968–72 with the observed water-table configuration for 1955–71 shows good agreement in most of the modeled area. The computed heads were within 2.5 feet (0.75 m) of the observed heads at more than 84 percent of the nodes. The greatest residuals (difference between the observed and computed heads at a node) were between 7.5 and 9.5 feet (2.3 and 2.9 m). Residuals in this range only occurred at less than than 1.5 percent of the nodes, and only at nodes near the disposal ponds, where the greatest variations in observed water-table altitudes had been measured. It must be emphasized that the general water-table configuration presented in figure 4 represents a composite of water-level measurements made during 1955–71 and is not necessarily an accurate representation of the water-table configuration at any specific time during that period.

The observed water-table configuration indicates that a source of recharge to the alluvial aquifer occurs in an area located approximately one-half mile (0.80 km) northeast of the center of pond C, which corresponds to the node at ($i = 10, j = 25$). This recharge might represent leakage from an unlined canal (Sand Creek Lateral) or the concentrated discharge of seepage through the unsaturated alluvium to the east and south. The model analysis indicated that an average flux of about 0.10 ft^3/s (0.0028 m^3/s) would be required to maintain the observed hydraulic gradient in this area. This average flux was thus assumed to have existed at this node from 1943–72. Because this recharge would probably be uncontaminated, it was assumed to have a chloride concentration of 80 mg/l.

Similarly, the water-table map presented by Konikow (1975) indicates that significant recharge may occur in or near the industrial area located south of pond A and north of the fresh water reservoirs. Thus, constant-head boundary conditions were applied to the three nodes located at ($i = 5, j = 32–34$). The model computed that a combined total of about 0.09 ft^3/s (0.003 m^3/s) of recharge would occur there during 1968–72. The source of this recharge could be infiltrated surface runoff from paved areas in the industrial complex. Because the chloride concentration of some ground-water samples taken in this area were slightly above normal background levels, it was assumed that any recharge from this area would have a chloride concentration of 200 mg/l.

The second period simulated was 1943–56, when pond C was assumed to leak at 50 percent of the rate computed for 1968–72. All other unlined ponds were assumed to be full during 1943–56 and were represented as constant-head boundaries in the model. Except for the changes at the disposal ponds, all other parameters and boundary conditions in the model were identical with the 1968–72 simulation. The head distribution for 1957–60 was assumed to be the same as during 1968–72 because of the apparent similar use of the disposal ponds. Therefore, the 1957–60 period did not require a separate flow simulation.

The third period simulated was 1961–67, when ponds B, C, D, and E were all assumed to leak at one-third of the rates computed for the periods when each was full. As in the second simulation period, all other parameters and boundary conditions in the model were assumed to be unchanged.

The flow model calculated a mass balance for each simulation run to check the numerical accuracy of the solution. As part of these calculations, the net flux contributed by each separate hydrologic component of the model was also computed and itemized to form a hydrologic budget for the aquifer in the modeled area. The hydrologic budget is valuable because it provides a measure of the relative importance of each element to the total budget. The hydrologic budgets for the final calibrations of the four steady-state flow models are presented in table 3. The date in table 3 indicate that the major sources of ground-water inflow are (1) infiltration from irrigated fields, (2) underflow through the aquifer into the study area, (3) seepage losses from the unlined irrigation canals, and (4) infiltration from the unlined disposal ponds. The major ground-water outflow occurs as (1) seepage into the South Platte River, (2) withdrawals from irrigation wells, and (3) underflow through the aquifer out of the study area. The computed total flux through the aquifer in the study area averages about 14 ft^3/s (0.40 m^3/s). However, most of this is flowing through the part of the aquifer north and west of the Arsenal boundary that receives most of the recharge and has the highest transmissivity.

TABLE 3. — *Elements of hydrologic budget computed by ground-water flow model*

	Computed flux[1] (ft^3/s)			
	1943-56	1957-60	1961-67	1968-72
Well discharge	−1.264	−1.264	−1.264	−1.264
Irrigation recharge	6.648	6.648	6.648	6.648
Canal leakage	1.606	1.606	1.606	1.606
Pond A	.155	.0	.0	.0
Pond B	.022	.0	.007	.0
Pond C	.542	1.083	.361	1.083
Pond D	.108	.0	.036	.0
Pond E	.050	.0	.017	.0
Freshwater reservoirs	.075	.081	.086	.081
Industrial area	.019	.094	.147	.094
Underflow across:				
Southwest boundary	4.713	4.473	4.822	4.473
Southeast boundary	.095	.089	.109	.089
Northeast boundary	−.467	−.495	−.475	−.495
South Platte River	−12.361	−12.290	−12.214	−12.290
First Creek	−.041	−.122	.014	−.122
Sand Creek Lateral	.100	.097	.100	.097
Total flux:				
Recharge	14.133	14.171	13.953	14.171
Discharge	−14.133	−14.171	−13.953	−14.171

[1]A positive value in this table indicates recharge to the aquifer in the modeled area; a negative value denotes discharge from the aquifer.

CALIBRATION OF SOLUTE-TRANSPORT MODEL

The solute-transport model applied to the Rocky Mountain Arsenal area was designed to compute changes in the chloride concentration in the alluvial aquifer during 1943–72. Heads and fluxes computed by the flow model were used as input to the transport model. A different velocity field was computed for each steady-state flow period outlined in table 2.

The solute-transport model was calibrated mainly on the basis of the chloride concentration pattern that was observed in 1956 (fig. 3). Field measurements of the effective porosity and dispersivity of the aquifer were not available, so a range of realistic values were tested in a sensitivity analysis. The computed concentrations were most sensitive to variations in the value of effective porosity and least sensitive to the transverse dispersivity. A comparison of observed and computed chloride concentration patterns indicated that an effective porosity of 30 percent and longitudinal and transverse dispersivities of 100 feet (30 m) were best.

After appropriate concentrations were assigned to all sources, and an initial background concentration of 40 mg/l was assigned to all nodes in the aquifer to represent conditions at the end of 1942, the transport model was run for a 14-year simulation period (1943–56). The model computed a chloride concentration pattern (fig. 8) that agreed closely with the observed pattern (fig. 3). The small difference in the directions of the axes of the main plumes between the observed and computed data is probably due mainly to errors in the computed flow field, rather than to errors in the transport model.

Since 1956, all disposal has been into the asphalt-lined Reservoir F, thereby eliminating the major source of contamination. However, that alone could not eliminate the contamination problem because large volumes of contaminants were already present in the aquifer. In January 1961 sufficient data were again available to contour the pattern of contamination (fig. 9). Although this is more than 4 years after the source of contamination had been apparently eliminated, only minor changes can be observed in the overall contamination pattern. These changes include a small downgradient spreading of dissolved contaminants and a significant decrease in chloride concentrations near the center of the contaminated zone. At this time the downgradient spreading was most noticeable near the northeastern limit of the contaminated zone, where a third distinct plume had formed. During 1957–72 water in the unlined disposal ponds was derived primarily from local surface runoff and canal diversions, which had relatively low chloride concentrations. Thus, much of the observed improvement in water quality near the center of the contaminated zone from 1957 to 1961 was probably the result of dilution by recharge from the former disposal ponds and from the Sand Creek Lateral.

The solute-transport model was next used to simulate the period 1957–60, using the chloride concentrations computed for the end of 1956 as initial conditions. The chloride concentrations thus computed for the end of 1960 (or the start of 1961) are illustrated in figure 10, which can be compared with the observed pattern for January, 1961 (fig. 9). The computed concentrations show the same general changes from 1956 that occurred in the observed chloride pattern. However,

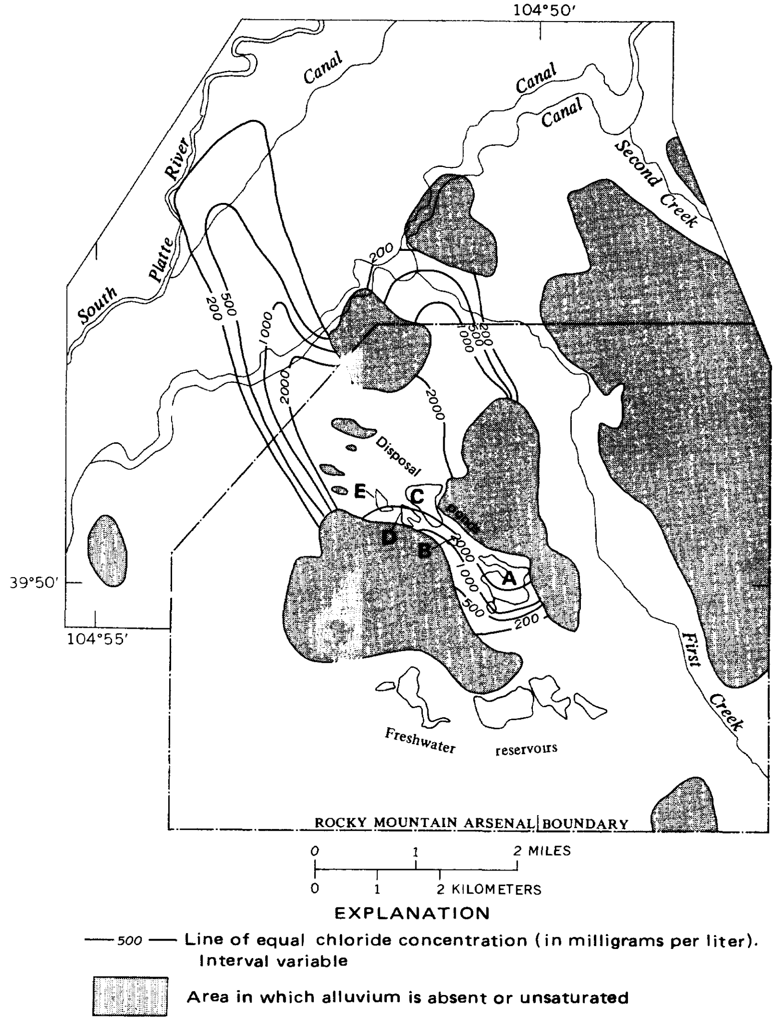
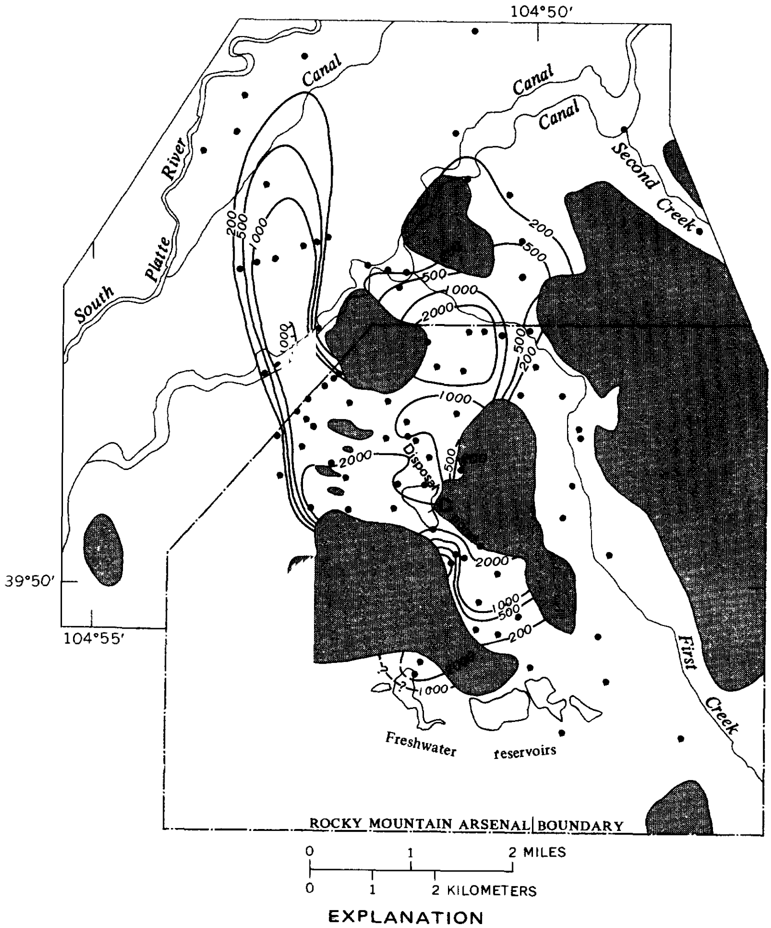


FIGURE 8.—Computed chloride concentration, 1956.

the model results indicated a more direct discharge toward the South Platte River than was observed, and the model did not indicate any spreading to the northeast to form a third plume. Some of this observed spreading may have been caused by transient changes in the flow field that could not be reproduced with the steady-state flow model.

Available data suggest that recharge of the aquifer was relatively low from 1961 to about 1968. Nevertheless, data collected in early 1969, the next time for which field data were available, do indicate the occurrence of a further significant decrease in the overall size of the



EXPLANATION

• Data point (Jan. 1961)

—500— Line of equal chloride concentration (in milligrams per liter). Interval variable

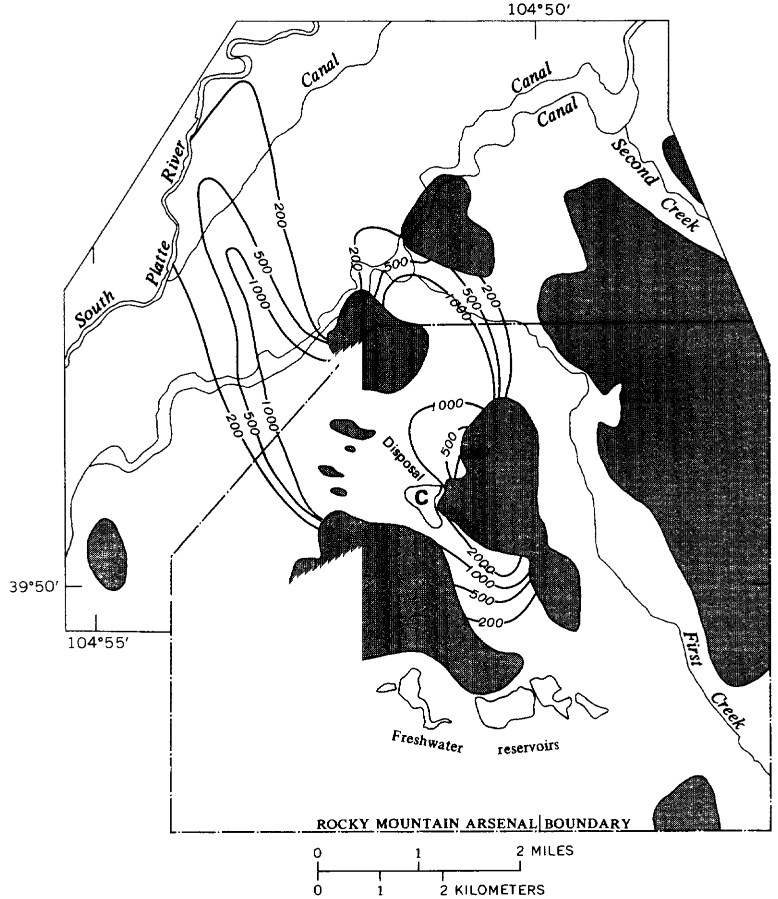
Area in which alluvium is absent or unsaturated

? Position of contour uncertain

FIGURE 9.—Observed chloride concentration, January 1961.

affected area. (See fig. 11.) Apparently, as the contaminated water continued to migrate downgradient, its chloride concentration was diminished by dispersion and dilution. Also, the concentrations decreased even more near the former disposal ponds. Chloride concentrations greater than 1,000 mg/l were now limited to only a few isolated areas.

The observed data from 1961 and 1969 also indicate that some contaminants were present in the aquifer near the freshwater reservoirs,



EXPLANATION

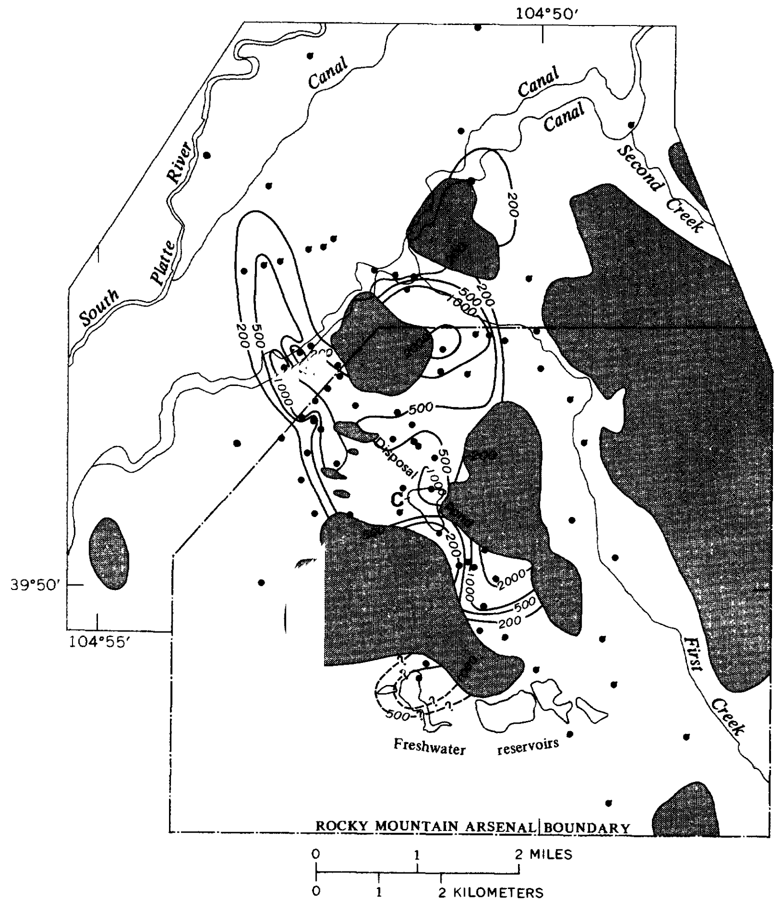
——500—— Line of equal chloride concentration (in milligrams per liter). Interval variable

Area in which alluvium is absent or unsaturated

FIGURE 10.— Computed chloride concentration at the start of 1961.

adjacent to the industrial area. It is unlikely that these contaminants were derived from the disposal ponds and, thus, were not predicted by the model. The chloride concentration pattern computed for the end of 1968 (or start of 1969), using the chloride concentrations computed for the end of 1960 as initial conditions, is presented in figure 12. For the most part, the solute-transport model has also reproduced this period of record, from 1961 to 1969, fairly well.

From about 1968 or 1969 through 1974, pond C was apparently again maintained full most of the time by diverting water from the freshwater reservoirs to the south. Available data showed that by



EXPLANATION

• Data point (Jan.–May 1969)

—500— Line of equal chloride concentration (in milligrams per liter). Interval variable

Area in which alluvium is absent or unsaturated

? Position of contour uncertain

FIGURE 11.— Observed chloride concentration, January–May 1969.

1972 the areal extent of contamination, as indicated by chloride concentration, had significantly diminished (fig. 13), and concentrations above 1,000 mg/l were now limited to just two small parts of the main zone of contamination. Because both are areas of relatively low hydraulic conductivity, it appears that low flow velocities have retarded the movement of the contaminated ground water out of or through these two areas. Chloride concentrations were almost at normal background levels in the middle of the affected area. This largely reflected

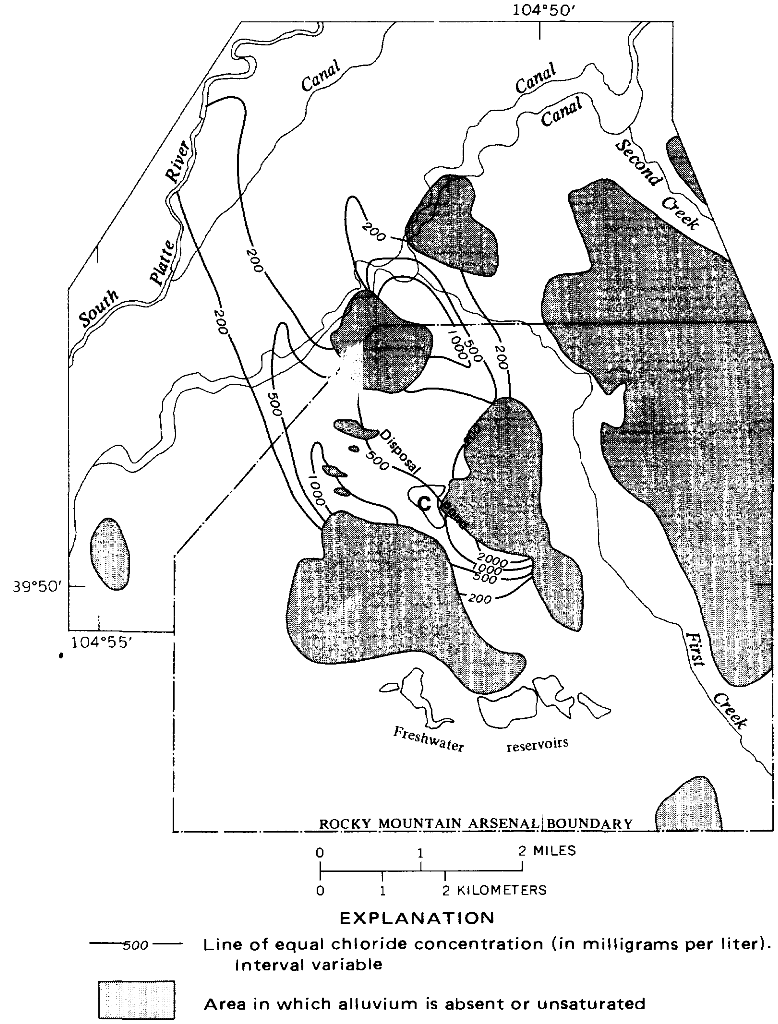
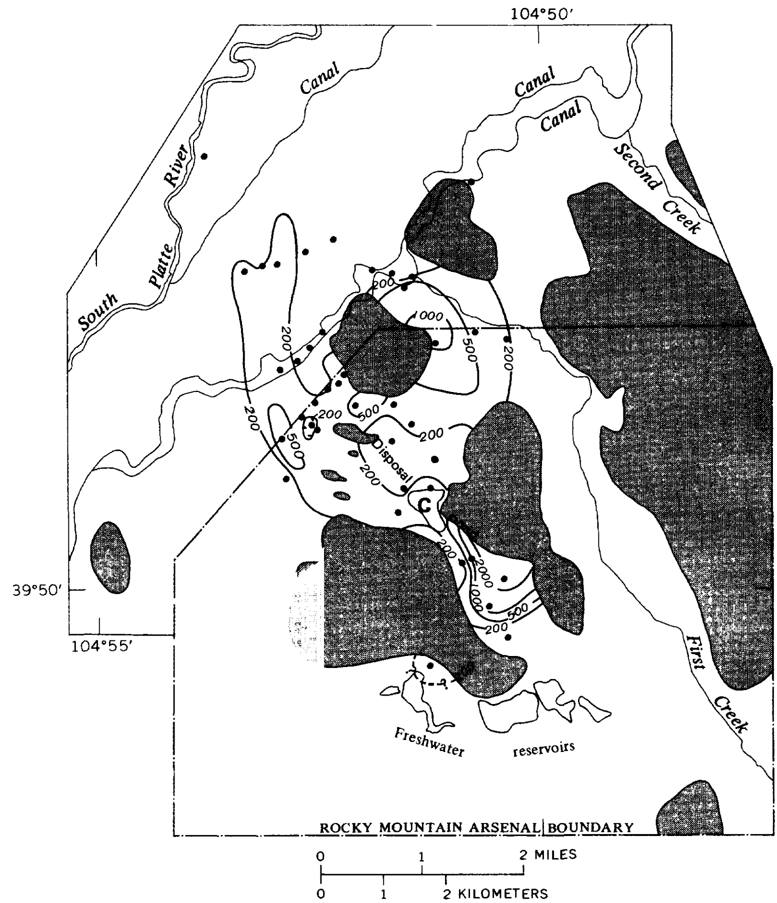


FIGURE 12.— Computed chloride concentration at the start of 1969.

the infiltration of fresh water from pond C, which had the effect of diluting and flushing the contaminated ground water.

The pattern of contamination computed for the start of 1972, by using the chloride concentrations computed for the end of 1968 as initial conditions, is presented in figure 14. The computed pattern agrees fairly well with the observed pattern (fig. 15), although the former shows somewhat longer plumes. After a 30-year simulation, the model has identified (1) the two areas where high chloride concentrations were still present, (2) the reduction in size and strength of the main



EXPLANATION

• Data point (May 1972)

——500—— Line of equal chloride concentration (in milligrams per liter). Interval variable

Area in which alluvium is absent or unsaturated

? Position of contour uncertain

FIGURE 13.— Observed chloride concentration, May 1972.

plume, and (3) the significant reduction in chloride concentration in the middle of the contaminated zone.

The simulation model computes the velocity of ground-water flow at each node of the model, but these data could not be independently verified with field data. The computed velocities ranged from less than 1.0 ft/d (0.3 m/d) to over 20 ft/d (6.1 m/d). Because the computed velocity varies greatly within the modeled area, depending on the transmissivity and hydraulic gradient, one value of velocity cannot be

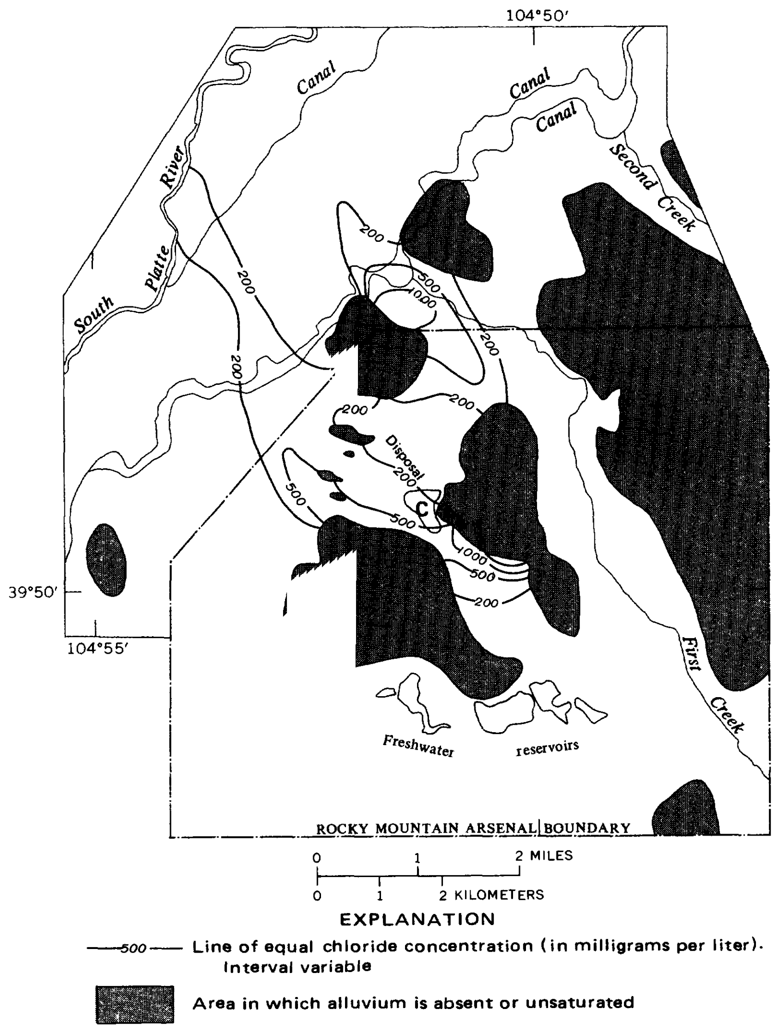


FIGURE 14.—Computed chloride concentration at the start of 1972.

used to estimate the average time of travel of dissolved chemicals between two points on a flow line. For problems where this type of information is desired, the computer program could be easily modified to provide these data.

Mass balance calculations were performed during the calibration procedure to check the numerical accuracy of the model simulations. Errors in the mass balance were always less than 1 percent for the flow model, but averaged about 14 percent for the solute-transport simulations. The latter is somewhat higher than desirable and indi-

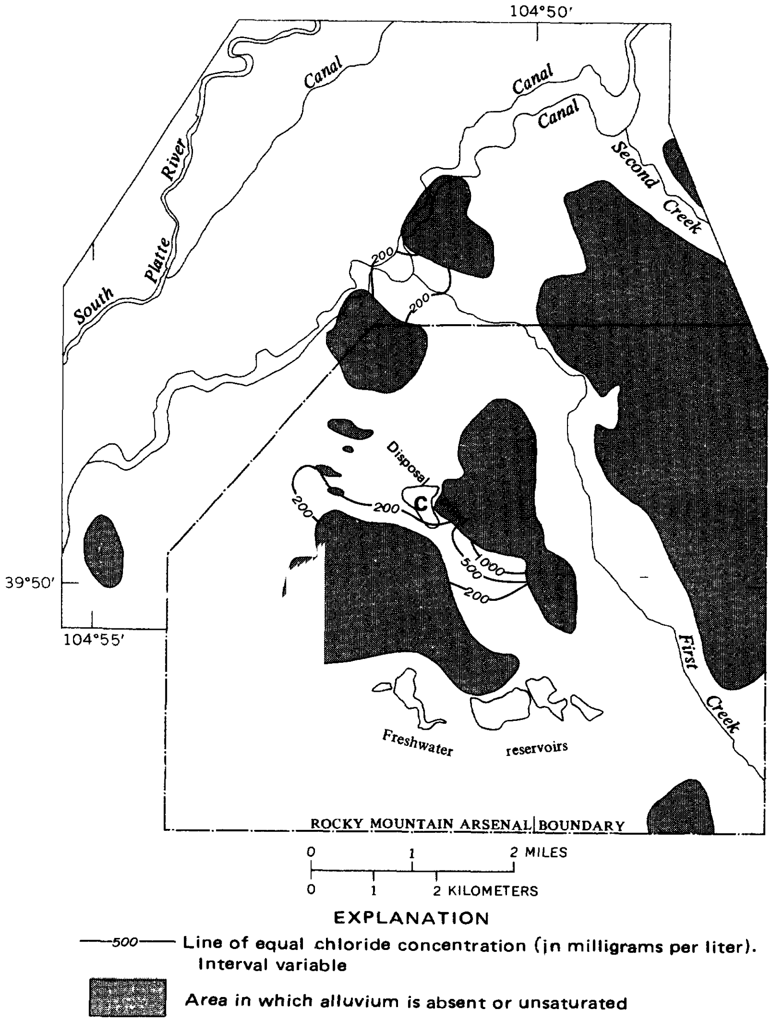


FIGURE 15.— Chloride concentration predicted for 1980, assuming that pond C is filled with fresh water during 1972–80.

cates the need for further refinements in the numerical procedure used to solve the transport equation.

PREDICTIVE CAPABILITY

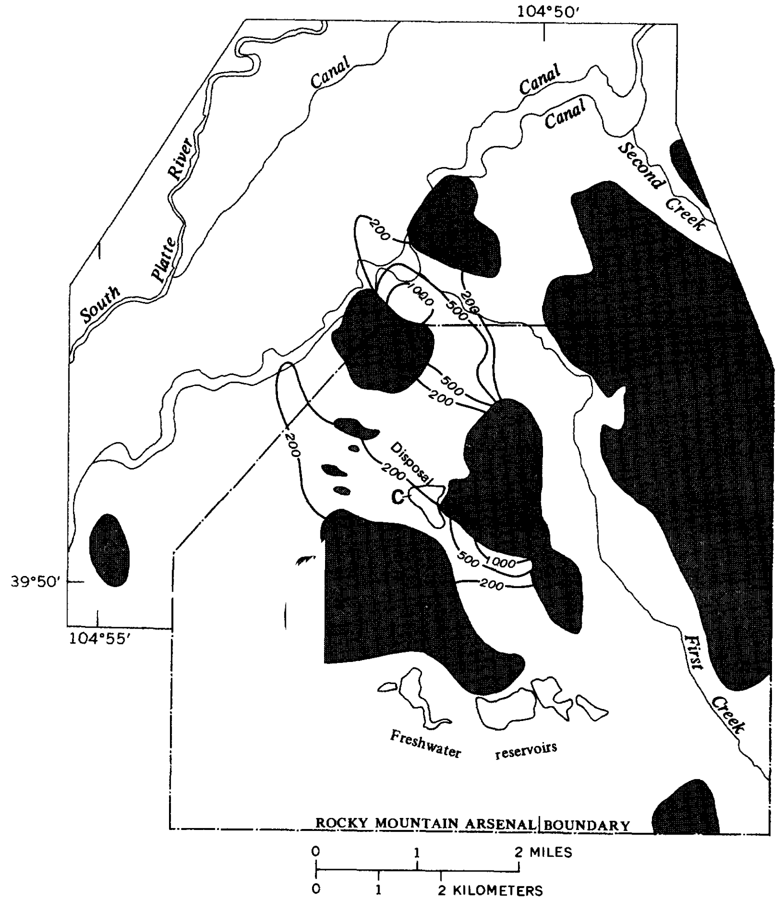
Once a model has been adequately calibrated, it can be used to predict or analyze the effects of either future or past changes in stresses or boundary conditions. The Rocky Mountain Arsenal model, which was calibrated over a 30-year historical period, appeared to be reliable enough to be used for limited predictive purposes. The predictive capability of this model can help to (1) isolate the effects of past measures, (2) evaluate the causes of present and recurring problems, and (3) predict future concentrations under a variety of assumptions. Several simulation tests were made, both to demonstrate the potential uses of this type of model and to better understand the nature of the ground-water contamination problem at the Arsenal.

Because of the relative importance of leakage from pond C (table 3), a simulation test was made to evaluate its possible effect on future chloride concentrations in the aquifer. One simulation run was made to predict the chloride concentration in 1980 if pond C were kept full of fresh water during 1972–80. These results (fig. 15) indicate that if the artificial recharge due to leakage from pond C were to continue, then in 1980 there would remain only one area of significant contamination, which would be confined entirely within the Arsenal boundaries. There would also still be one small area north of the Arsenal that would contain chloride concentrations between 200 and 500 mg/l.

The importance of the effects of artificial recharge from pond C can be illustrated by computing what the chloride concentration would be in 1980 if pond C were not kept full, and then comparing this pattern with the one in figure 15. Therefore, the chloride concentrations in 1980 were recomputed after assuming that the minimal recharge rate from pond C that was estimated for 1961–67 (table 3) had continued during 1968–80. These results are presented in figure 16 and indicate that if the artificial recharge from pond C would not occur during 1961–80, then in 1980, which is about 25 years after the sources of ground-water contamination were supposedly eliminated, there would still be two relatively large areas of contaminated ground water remaining.

A comparison of the results presented in figure 16 with those shown in figure 15 indicates that the effects of artificial recharge from pond C had significantly increased the rate of water-quality recovery in the aquifer. In addition to the effects of dilution, the recharge created a mound on the water table, which increased the hydraulic gradients, and consequently, the flow velocities. In effect, it "pushed" or "flushed" the contaminated ground water out of the area faster than would have occurred naturally. The apparent difference in the mass of chlorides present in the aquifer between figures 15 and 16 is caused by two factors. First, in figure 15 a greater percentage of the total chloride is contained in areas having a concentration less than 200 mg/l. Second, during the simulation period upon which figure 15 is based, a greater mass of chlorides has discharged from the aquifer in the modeled area, mostly to the South Platte River and as ground-water underflow across the model boundaries.

By comparing figures 15 and 16, it can also be inferred that it would probably take at least many decades for this contaminated aquifer to



EXPLANATION

—500— Line of equal chloride concentration (in milligrams per liter). Interval variable

Area in which alluvium is absent or unsaturated

FIGURE 16.—Chloride concentration predicted for 1980, assuming that recharge from pond C is minimal during 1961–80.

naturally recover its original water-quality characteristics. But it can also be inferred that appropriate water-management policies can help to reduce this restoration time to the order of years, rather than decades.

More than one-half of the total ground-water flux through the modeled area is derived from recharge from irrigation applications and canal leakage. (See table 3.) A simulation test was designed to demonstrate that these sources of recharge have an important effect on the chemical concentrations in the aquifer. A simulation run was made for 1943–56, assuming zero dilution from these recharge sources. The chloride concentrations computed for 1956 under this assumption were then compared with the computed pattern shown in figure 8. If there were no dilution from recharge, in 1956 the area contaminated with chloride concentrations greater than 1,000 mg/l would have been slightly larger than the area within the 500 mg/l isochlor shown in figure 8, and the higher concentrations would have spread to the north much earlier. Therefore, dilution from irrigation recharge and seepage from unlined canals were both important factors in reducing the level of chloride concentrations downgradient from the Arsenal.

APPLICATION TO WATER-MANAGEMENT PROBLEMS

Changes in water use or water management in an area can significantly affect both the flow and chemical quality of ground water. Because a wide variety of alternative decisions or policies are possible regarding water planning, management, and quality control, it is difficult to determine the optimum set of alternatives that will minimize detrimental effects and maximize beneficial effects. Thus, an accurate solute-transport model can be a valuable planning tool because it can help to analyze the relative sensitivity of the aquifer system to different management alternatives and demonstrate the impact of specific practices on the chemical quality of ground water.

To demonstrate this use of the model, a hypothetical change in ground-water management at the Rocky Mountain Arsenal was simulated. This illustrative example evaluates a proposal (Konikow, 1974) to maintain hydraulic sinks along those parts of the northern boundary of the Arsenal under which the plumes of contaminated ground water were moving. Construction of the hydraulic sinks could physically involve either drilling a line of wells for pumping or excavating a trench or ditch below the water table for drainage. The main purpose of the hydraulic sink would be to intercept and remove contaminated ground water from the aquifer before it migrates downgradient from the Arsenal. This proposal is evaluated here only to demonstrate the general value of the model and is intended neither to endorse this particular plan nor to suggest that any changes in water management should necessarily be implemented at the Rocky Mountain Arsenal.

The hydraulic sinks could greatly modify the heads and flow velocities in their vicinities. Therefore, these hydraulic changes would have to be computed before the effects on solute transport could be predicted. For this example problem a simplified scheme was used to generate hydraulic sinks in the flow model that was calibrated for the period 1968–72. Two sinks, A and B, were represented by imposing

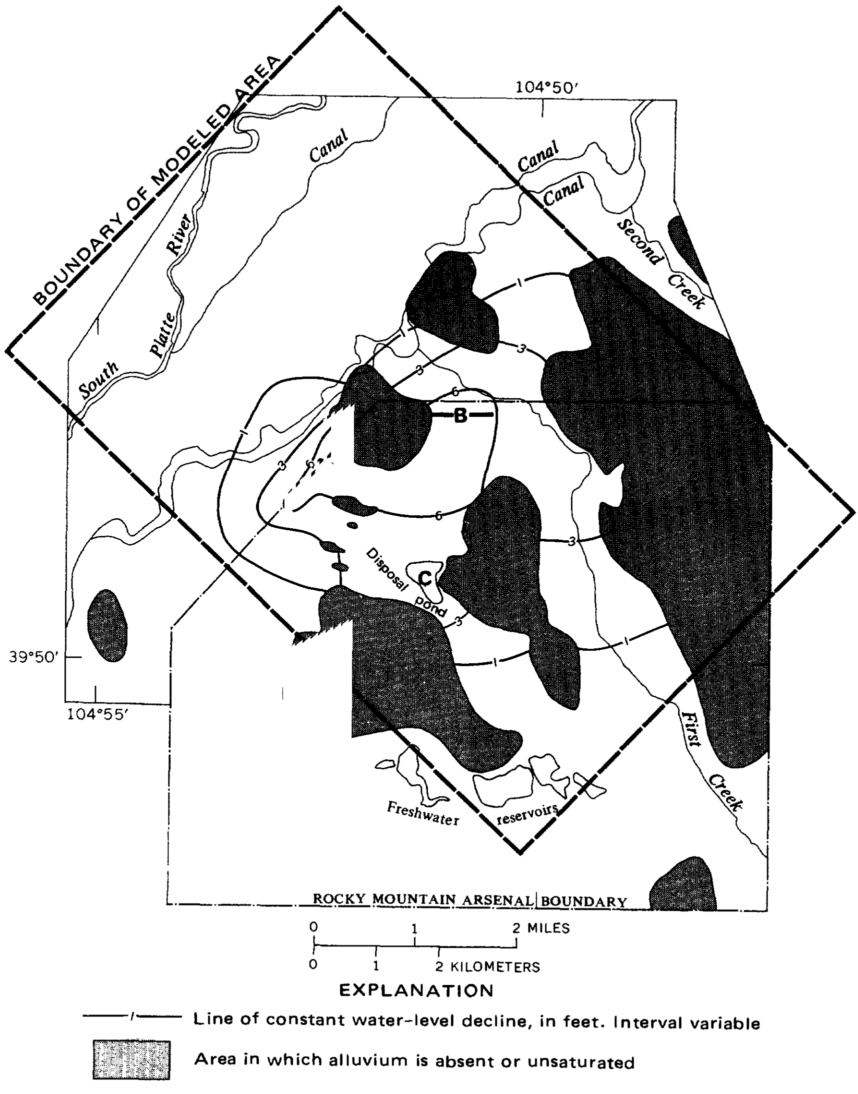


FIGURE 17.—Computed drawdown caused by maintaining two constant-head sinks along the northern boundary of the Rocky Mountain Arsenal.

constant-head boundary conditions at appropriate nodes in the finite-difference grid of the model. Sink A was along the northwestern boundary and was simulated as a constant head of 5,085 feet (1,550 m) at nodes (i = 5–9, j = 17). Sink B was along the northern boundary and was simulated as a constant head of 5,124 feet (1,562 m) at nodes (14,19), (15,20), and (16,21).

The effect of these two constant-head sinks on steady-state ground-water flow is illustrated in figure 17, which depicts the computed drawdown (water-level decline) relative to the heads computed for 1968–72. The water-level declines extend a greater distance to the south and southeast of the sinks than to the north and northwest because of the much lower availability of recharge to the south and southeast. The model computed that the discharge of sink A would be about 0.98 ft^3/s (0.028 m^3/s), whereas discharge of sink B would be about 0.15 ft^3/s (0.0042 m^3/s). The greater computed discharge of sink A reflects its location in an area of higher transmissivity and its closer proximity to the outer constant-head boundary of the model. The extent and magnitude of drawdown shown in figure 17 indicates that hydraulic gradients, and resulting flow velocities and solute transport, would be significantly affected if the hydraulic sinks were actually present.

The two hydraulic sinks were assumed to have begun operating in 1968, and pond C was assumed to have remained full after 1968. The chloride concentration pattern then computed for 1980 is shown in figure 18, which represents the combined effects of artificial recharge from pond C and artificial drainage to the two hydraulic sinks. This pattern can be compared with the one shown in figure 15, which just represents the effect of artificial recharge from pond C. The most noticeable difference is that figure 18 shows the persistence of higher chloride concentrations in larger areas downgradient (north) of the hydraulic sinks.

Although it may seem anomalous to produce less water-quality improvement with a source-sink combination than with a source alone, this occurrence is not unreasonable and can be explained with the aid of a schematic cross-section through the source of artificial recharge and the hydraulic sink. (See fig. 19.) Figure 19 shows both the original water table with recharge from the unlined pond and the new steady-state water-table position and directions of flow that were established after pumping began. The change in hydraulic gradients caused by the sink indicates that the velocity of ground-water flow would increase on the upgradient side of the sink. On the downgradient side of the sink, there would be a reversal of the hydraulic gradient within a small area near the sink. However, just beyond the area of reversal is an area where the new gradients would be much less than before, creating a zone of near stagnation in the aquifer. Because of extremely low flow velocities in this zone, any contaminants that were present in this area at the time the sink was constructed would remain in the area much longer than if no sink had been constructed.

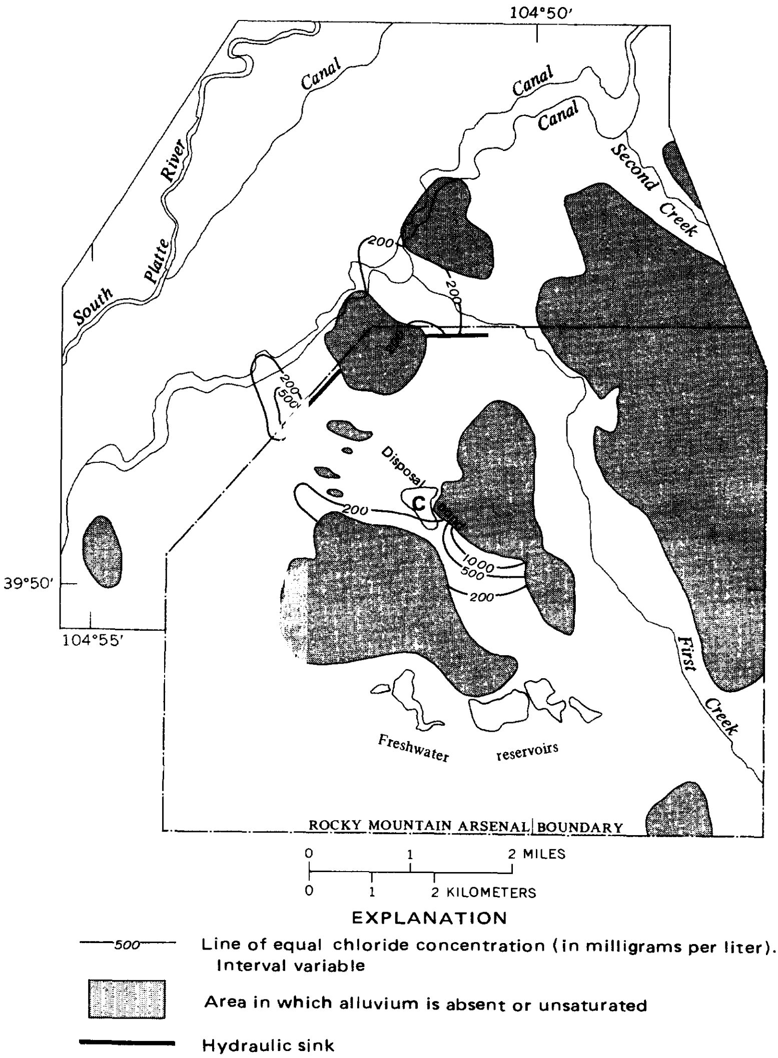


FIGURE 18.—Chloride concentration predicted for 1980, assuming that artificial recharge from pond C is coupled with drainage through two hydraulic sinks.

The construction of the hydraulic sinks would slightly increase the rate of water-quality improvement between 1968 and 1980 in the area between the source and the sinks. In fact, the results of the simulation run using sinks indicate that the chloride concentrations in the area between the source and the sinks would virtually reach an equilibrium pattern after about 10 years.

Comparison of figures 15 and 18 indicate that water management using a source-sink combination as postulated here would produce less

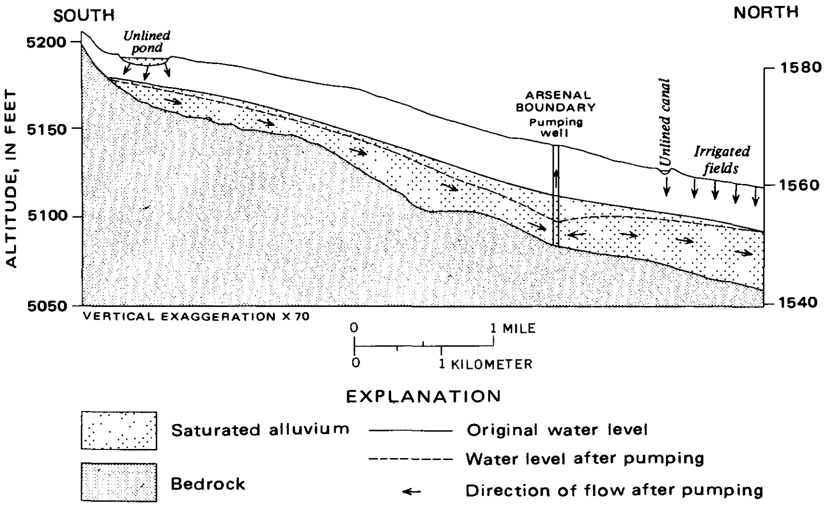


FIGURE 19.—Generalized cross-section from vicinity of source of artificial recharge through hydraulic sink (represented as a well).

desirable results overall than would a simple artificial recharge source alone. The primary reason for this appears to be that the sinks were placed near the middle of the contaminant plume. In the study area this type of scheme would be most effective if the hydraulic sinks were located at or just beyond the maximum extent of the plume. Similarly, if an artificial recharge source is to be used for flushing and dilution, it should be placed just upgradient of the contaminated zone, if possible. In general, designs would depend on the circumstances of each specific problem and should be evaluated individually.

At the Rocky Mountain Arsenal, the hydraulic sinks would provide a secondary benefit that is not apparent in the map of chloride concentration. Drainage due to the hydraulic sinks would produce water-table declines that might prevent soil salinity problems caused by high water tables in contaminated areas. This same effect would also reduce possible discharge of contaminated ground water to springs or streams, such as First Creek.

Removing pollutants from a contaminated aquifer may seem to be an almost impossible task. While this may be true for some or even most contaminated aquifers, others may be highly amenable to one or more plans for artificial reclamation that could significantly accelerate the rate of water-quality improvement in the aquifer. The feasibility of any such reclamation plan would be strongly dependent on the hydraulic properties of the aquifer and on the type and source of contamination. An accurate model of the aquifer is a prerequisite for reliably predicting changes in concentration and thereby estimating the benefits that might be derived from a given plan.

SUMMARY AND CONCLUSIONS

The movement of dissolved chemicals in flowing ground water can be simulated with a computer model that couples a finite-difference solution to the ground-water flow equation with the method-of-characteristics solution to the solute-transport equation. The usefulness of the model was illustrated by its application to a problem of ground-water contamination in the Rocky Mountain Arsenal area, Colo., where the model integrated the effects of several factors that controlled changes in chloride concentrations and successfully reproduced the record of chloride contamination observed during a 30-year period.

The method of characteristics offers one workable solution technique to the solute-transport equation. Although every ground-water contamination problem is in many ways unique, the processes controlling solute transport are the same. Thus, this method-of-characteristics model is generally applicable to a wide variety of ground-water contamination problems.

The predictive accuracy of the model is most limited by the adequacy of the input data. The results of applying the model to the ground-water contamination problem at the Rocky Mountain Arsenal indicate that where adequate hydrogeologic data are available, the model can be used to predict the rates and directions of spreading of conservative (nonreactive) contaminants from known or projected sources.

The predictive capability of the model can be helpful in designing a monitoring network. By indicating the most probable and least probable areas of future contamination and the rate of spreading, optimal locations and sampling frequencies for observation wells can be determined. The model can also indicate areas where contaminated ground water might seep into surface water.

In some cases of aquifer contamination it may be both physically and economically feasible to institute a reclamation program to improve or control the quality of ground water. Because a large variety of water-management plans can be proposed for any one problem, an accurate model of flow and solute transport in the aquifer could be an invaluable tool for planning an efficient and effective program.

Conclusions of this study that pertain specifically to the Arsenal can be summarized as follows:

1. Ground water in the alluvial aquifer in and adjacent to the Rocky Mountain Arsenal flows predominantly to the north and northwest. The alluvium generally has the greatest transmissivity, saturated thickness, and rate of recharge in the area between the South Platte River and the north and northwest boundaries of the Rocky Mountain Arsenal. Within the boundaries of the Arsenal the main source of ground-water recharge since 1943 has apparently been infiltration from the unlined disposal ponds.

2. The pattern of chloride contamination was principally controlled by the rates and directions of ground-water flow. Thus, variations in the hydraulic conductivity (or transmissivity) and boundary conditions of the aquifer directly influence solute transport. The observed contamination pattern clearly indicates that the spreading of contaminants was significantly restricted or constrained by the areas in which the aquifer is either absent or unsaturated. Model analyses support the hypothesis that dilution, both from irrigation recharge and from seepage from unlined canals, was an important factor in reducing the level of chloride concentrations in the contaminated ground water that flowed past the boundaries of the Arsenal.
3. Most of the contaminants that seeped into the aquifer were probably derived from the overflow ponds (ponds B, C, D, and E). Because the original primary disposal pond (pond A) was located in an area where the aquifer had a relatively low hydraulic conductivity, only small quantities of contaminated ground water could seep through the aquifer downgradient from pond A.
4. In 1972, approximately 16 years after the main sources of ground-water contamination had been eliminated, a large area of the alluvial aquifer still contained chloride concentrations that were significantly above normal background levels. But the magnitude and extent of contamination, as measured by the chloride concentration, had significantly diminished in comparison with observations during 1956 and 1961. By 1980 high chloride concentrations (that is, greater than 200 mg/l) will probably occur in only two comparatively small areas. One is north of the Arsenal, near First Creek, and the other is near and downgradient from the site of pond A.
5. The diluting and flushing effects of freshwater recharge from pond C contributed to a significant reduction in the concentrations and total quantity of chlorides present in the aquifer.
6. If the Rocky Mountain Arsenal or a similar industrial plant were first beginning operations today, this model could be used with reliable aquifer descriptions to predict the magnitude and extent of ground-water contamination that could be expected to result from the disposal of nonreactive liquid industrial wastes into unlined ponds. Perhaps more importantly, the model could have been used either to determine where the disposal ponds should have been located to minimize the extent of contamination or to demonstrate, in the first place, that this particular method was inappropriate for liquid waste disposal in this particular environment and that an alternative method was needed.
7. The stringent data requirements for applying the solute-transport model pointed out deficiencies in data existing at the start of this

investigation. The subsequent analysis and reinterpretation of existing hydrogeologic data led to a revised conceptual model that accounted for the effects on ground-water flow and solute transport of the areas in which the alluvium either is absent or is unsaturated most of the time.

8. The model for chloride movement cannot predict the behavior of nonconservative (reactive) chemical species. But any comprehensive study or management plan of ground-water quality at the Arsenal would need to include any such reacting species with appropriate modifications to the model.

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