In cooperation with the
NATIONAL GUARD BUREAU

Simulation of Advective Flow under Steady-State and Transient Recharge Conditions, Camp Edwards, Massachusetts Military Reservation, Cape Cod, Massachusetts

Water-Resources Investigations Report 03-4053

U.S. Department of the Interior
U.S. Geological Survey

Cover photo: A section of the firing range on Camp Edwards, Massachusetts Military Reservation, Cape Cod, Massachusetts

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By DONALD A. WALTER and JOHN P. MASTERSON

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Northborough, Massachusetts
2003

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CONTENTS

FIGURES

CONVERSION FACTORS, VERTICAL DATUM, AND ACRONYMS AND ABBREVIATIONS

CONVERSION FACTORS

Multiply	By	To obtain
acre	0.4047	hectare
cubic foot per day (ft^3/d)	0.02832	cubic meter per day
foot (ft)	0.3048	meter
foot per day (ft/d)	0.3048	meter per day
foot per day per foot (ft/d/ft)	1	meter per day per meter
foot squared per day (ft^2/d)	0.09290	meter squared per day
inch (in.)	2.54	centimeter
inches per year (in/yr)	25.4	millimeter per year
mile (mi)	1.609	kilometer

VERTICAL DATUM

Sea level: Vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD29).

ACRONYMS AND ABBREVIATIONS

AFCEE	Air Force Center for Environmental Excellence
ARNG	Army National Guard
CIA	Central Impact Area
DNT	dinitrotoluene
DRN	Drain Package
FHB	Flow and Head Boundary Package
GHB	General Head Boundary Package
HFB	Horizontal Flow Barrier
HMX	Her Majesty's Explosive
IAGWSP	Impact Area Groundwater Study Program
mg/L	milligrams per liter
MMR	Massachusetts Military Reservation
RDX	Royal Dutch Explosive
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
UXO	Unexploded ordnance

Simulation of Advective Transport under Steady-State and Transient Recharge Conditions, Camp Edwards, Massachusetts Military Reservation, Cape Cod, Massachusetts

By Donald A. Walter *and* John P. Masterson

ABSTRACT

The U.S. Geological Survey has developed several ground-water models in support of an investigation of ground-water contamination being conducted by the Army National Guard Bureau at Camp Edwards, Massachusetts Military Reservation on western Cape Cod, Massachusetts. Regional and subregional steady-state models and regional transient models were used to (1) improve understanding of the hydrologic system, (2) simulate advective transport of contaminants, (3) delineate recharge areas to municipal wells, and (4) evaluate how model discretization and time-varying recharge affect simulation results.

A water-table mound dominates ground-water-flow patterns. Near the top of the mound, which is within Camp Edwards, hydraulic gradients are nearly vertically downward and horizontal gradients are small. In downgradient areas that are further from the top of the water-table mound, the ratio of horizontal to vertical gradients is larger and horizontal flow predominates. The steady-state regional model adequately simulates advective transport in some areas of the aquifer; however, simulation of ground-water flow in areas with local hydrologic boundaries, such as ponds, requires more finely discretized subregional models. Subregional models also are needed to delineate recharge areas to municipal wells that are inadequately represented in the regional model or are near other pumped wells.

Long-term changes in recharge rates affect hydraulic heads in the aquifer and shift the position of the top of the water-table mound. Hydraulic-gradient directions do not change over time in downgradient areas, whereas they do change substantially with temporal changes in recharge near the top of the water-table mound. The assumption of steady-state hydraulic conditions is valid in downgradient area, where advective transport paths change little over time. In areas closer to the top of the water-table mound, advective transport paths change as a function of time, transient and steady-state paths do not coincide, and the assumption of steady-state conditions is not valid. The simulation results indicate that several modeling tools are needed to adequately simulate ground-water flow at the site and that the utility of a model varies according to hydrologic conditions in the specific areas of interest.

INTRODUCTION

Live-fire training activities and munitions disposal at Camp Edwards on the Massachusetts Military Reservation (MMR), Cape Cod, Massachusetts (fig. 1), have released explosive compounds into the environment. The underlying aquifer is the sole source of potable water to the residents of western Cape Cod and there is concern that migration of contaminants from Camp Edwards could adversely affect current and future water-supply resources. In 1997, the U.S. Environmental Protection Agency (USEPA) issued an administrative order that suspended live-fire training at Camp Edwards until the military could prove that current live-fire training practices do not contaminate ground water (USEPA, Region 1, Administrative Order SDWA 1-97-1030). As a result, the U.S. Army National Guard (ARNG) initiated an assessment of ground-water and soil contamination at the site.

Analysis of ground-water samples has shown that contamination from Camp Edwards has migrated in the ground water to downgradient areas. The contaminant of most concern is cyclotrimethylene-trinitramine, commonly referred to as Royal Dutch Explosive (RDX), which is a carcinogenic compound that can be transported conservatively in ground under the oxic conditions generally observed in the aquifer. RDX has been detected in ground water at distances as great as 2 mi from likely sources within Camp Edwards and at depths as great as 100 ft below the water table. Other contaminants of concern at the site include perchlorate, picric acid, dinitrotoluene (DNT), and cyclotetramethylene-tetranitramine, commonly referred to as Her Majesty's Explosive (HMX). Most of the investigations have focused on the Impact Area (fig. 1), the area most heavily used historically for training activities.

The U.S. Geological Survey (USGS) has assisted the ARNG in their investigation of ground-water contamination at the site since 1997. The role of the USGS has been to improve understanding of regional ground-water flow and to provide assistance in evaluating ground-water contamination at the site in a hydrologic context. Specifically, the USGS has (1) developed and applied steady-state regional ground-water-flow models of western Cape Cod to simulate advective transport of contaminants in the aquifer, (2) developed subregional models of specific areas to better simulate ground-water flow and advective transport in areas near ponds and municipal wells, and (3) developed and applied a transient regional model to evaluate the effects of time-varying recharge on advective transport in the aquifer. The ground-water-flow models were used to provide real-time support of field activities being conducted by the ARNG and their consultants. Simulation results helped determine locations of new observation wells, identify potential source areas for contaminants detected in the subsurface, and delineate areas contributing recharge to current and proposed municipal wells. The USGS also analyzed samples collected from a number of locations within Camp Edwards to estimate ground-water ages to assist in model validation and interpretation of field data.

Purpose and Scope

This report describes and documents USGS ground-water-flow modeling activities in support of the ARNG investigations. Specifically, the report (1) discusses the use of steady-state, regional ground-water flow models to simulate advective transport of contaminants at Camp Edwards, (2) documents the development and use of two steady-state, subregional models, and (3) documents the development and use of a transient regional model to evaluate the effect of time-varying recharge on advective transport. The report describes how the models were used to support ARNG investigations, including determination of monitoring well locations, identification of potential source areas, and delineation of areas contributing recharge to municipal wells. The report also highlights several modeling concepts that apply to simulating advective transport in unconfined aquifers, including the effects of model discretization on simulated advective transport near surface-water bodies and on simulated recharge areas to municipal wells, and the effects of time-varying recharge on advective transport, and how these effects vary within the aquifer.

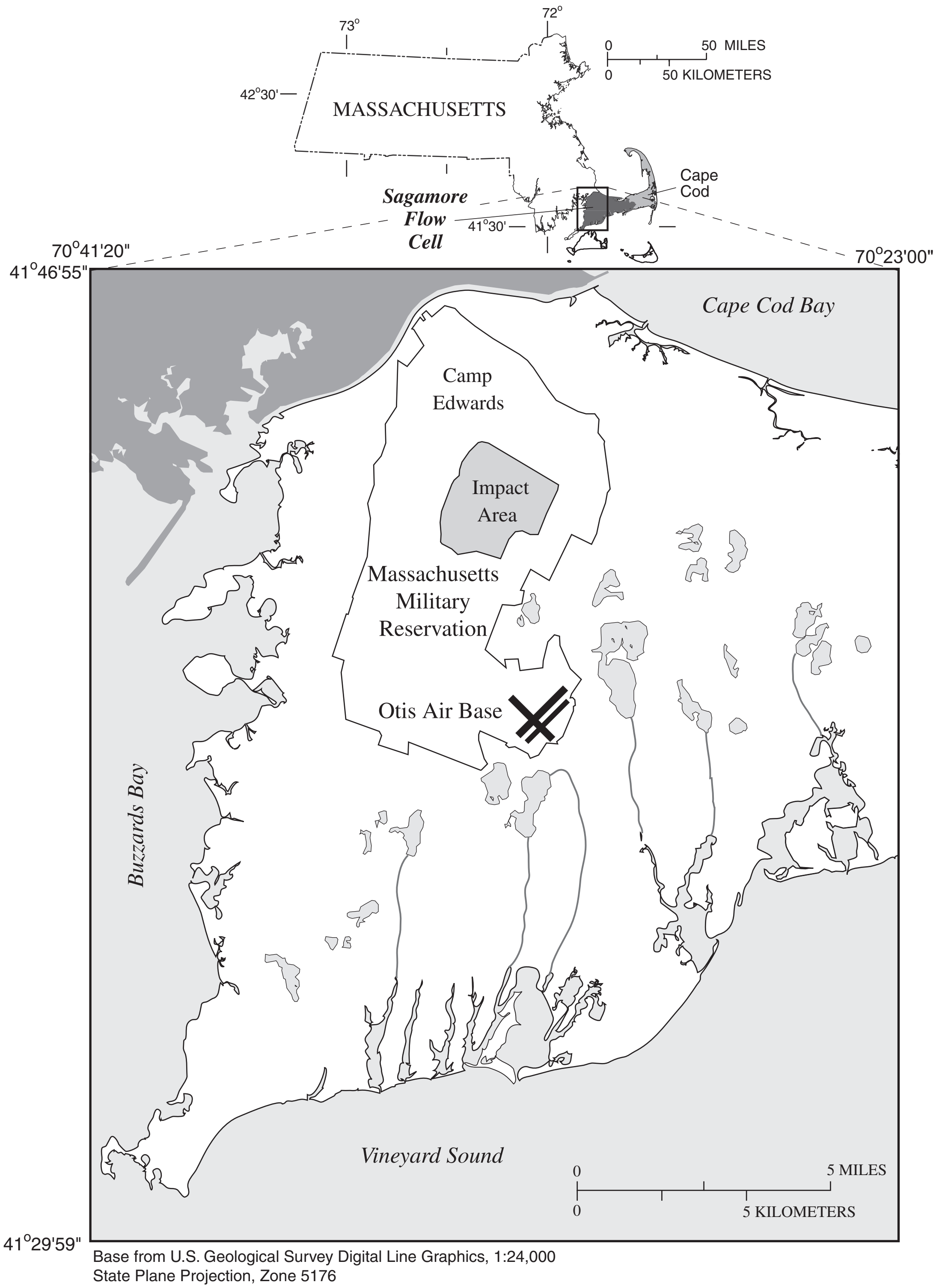


Figure 1. Location of Camp Edwards and its firing ranges, also known as the Impact Area, on the Massachusetts Military Reservation, Cape Cod, Massachusetts.

Geologic Setting

Camp Edwards is located in the northern part of western Cape Cod (fig. 1). The primary physiographic feature of the area is a broad, gently sloping glacial-outwash plain, known as the Mashpee Pitted Plain. This plain is bounded to the north and west by hummocky terrain associated with the Sandwich and Buzzards Bay glacial moraines (fig. 2) and to the east by an adjacent outwash plain.

The glacial sediments underlying western Cape Cod were deposited at the edge of retreating ice sheets during the Pleistocene Epoch, about 15,000 years ago (Oldale and Barlow, 1986). The Buzzards Bay Moraine, located to the west of the outwash plain, is an ablation moraine that likely was deposited in place by melting ice. The contact between moraine and outwash deposits likely extends beneath the outwash (B.D. Stone, U.S. Geological Survey, written commun., 1994). The origin of the Sandwich Moraine, located to the north of the outwash plain, is not as well understood. The moraine may be an ablation moraine or a tectonic moraine that consists of reworked outwash sediments pushed into place by a local readvance of the ice sheet. In the latter case, the contact between moraine and outwash deposits would extend beneath the moraine.

The glacial outwash sediments are part of a delta that was deposited in a large proglacial lake that formed at the ice margin. These sediments are glaciofluvial or nearshore glaciolacustrine in origin and consist of fine to coarse sand and gravel, which become finer-grained and thinner to the south with increasing distance from the sediment source area located near the apex of the Sandwich and Buzzards Bay moraines (fig. 2) (Masterson and others, 1997a). To the south, fine-grained glaciolacustrine sediments consisting of fine sand, silt, and clay underlie the coarser-grained glaciofluvial sediments.

The outwash plain contains numerous glacial-collapse structures. These structures, which form topographic depressions that commonly contain kettle-hole ponds, formed when buried blocks of remnant glacial ice melted, causing overlying sediments to collapse. Coarse-grained sediments that may extend to greater depths than in surrounding areas typically characterize collapse structures.

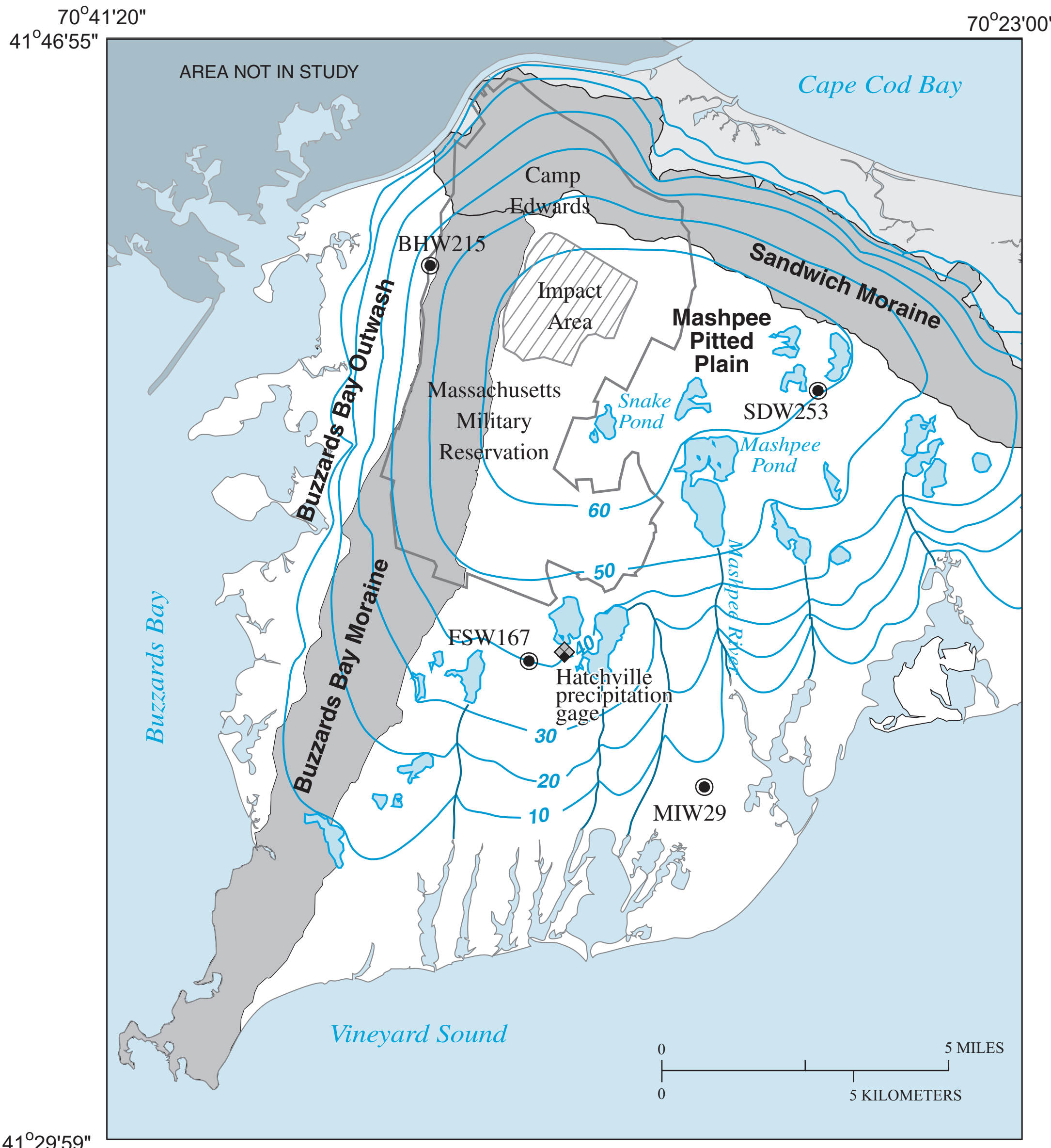
The sequence of glacial deposits on western Cape Cod ranges in thickness from 70 ft near the Cape Cod Canal to more than 400 ft along Vineyard Sound. The lowermost glacial deposits consist of basal-till deposits in most places. The unconsolidated glacial sediments are underlain by crystalline bedrock throughout Cape Cod.

Glacial outwash sediments are generally well sorted and have some degree of stratigraphic continuity, whereas the moraine deposits have a more variable lithology and, on a regional scale, generally are finer-grained than outwash deposits. Hydraulic conductivities of the glacial sediments range from about 350 ft/d for coarse sand and gravel to about 10 ft/d for silt and clay (Masterson and others, 1997b).

Camp Edwards includes areas underlain by both moraine and outwash deposits. The Impact Area is on the outwash plain near the sediment source, and thus is underlain by coarse-grained sediments characteristic of a high-energy depositional environment. The sediments generally consist of medium to coarse sand and gravel with local deposits of silt and fine sand, particularly deeper in the aquifer. The moraine deposits consist of gravel, sand, silt, and clay, have a more variable lithology, and generally are more fine-grained than the adjacent outwash deposits. Saturated thickness within the Impact Area ranges from less than 150 ft to more than 300 ft. The hydraulic conductivity of the glacial sand and gravel ranges from 200 to 350 ft/d (Masterson and others, 1997b).

Hydrologic Setting

Western Cape Cod is within the Sagamore Flow Cell (fig. 1), which is the westernmost of seven separate ground-water-flow cells on Cape Cod (LeBlanc and others, 1986). The aquifer system of western Cape Cod is surrounded mostly by salt water: Cape Cod Bay to the northeast, Cape Cod Canal to the northwest, Buzzards Bay to the west, and Vineyard Sound to the south (fig. 2). The Bass River and the adjacent Monomoy Flow Cell bound the aquifer system to the east. Recharge from precipitation is the sole source of water to the aquifer system. About 48 in. of precipitation falls annually on western Cape Cod. About half of the precipitation is lost to evapotranspiration; the remainder recharges the aquifer. A previous modeling investigation estimated that about 41 percent of ground water discharges to streams, 53 percent discharges to coastal boundaries, and the remaining 6 percent is withdrawn for water supply (Masterson and others, 1997b).



Base from U.S. Geological Survey Digital Line Graphics, 1:24,000
State Plane Projection, Zone 5176

EXPLANATION

OUTWASH SEDIMENTS

GLACIOLACUSTRINE SEDIMENTS

GLACIAL MORAINE

— 20 — WATER-TABLE CONTOUR—Shows altitude of water table, March 1993. From Savoie (1995). Contour interval is 10 feet. Vertical datum is NGVD29

◉ LONG-TERM MONITORING WELL AND IDENTIFIER

◈ HATCHVILLE PRECIPITATION GAGE

Figure 2. Surficial geology of western Cape Cod, Massachusetts, water-table-altitude contours from March 1993, and the locations of the Hatchville precipitation gage and selected long-term monitoring wells.

The aquifer system is bounded below by impermeable bedrock and at the top by the water table across which recharge enters (fig. 3). Ground water flows radially outward from a water-table mound towards discharge locations in streams and coastal embayments. The top of the mound of the Sagamore Flow Cell is within Camp Edwards; maximum water-table altitudes near the top of the mound are more than 65 ft above sea level (fig. 2) (Savoie, 1995). Water-table contours and ground-water flow patterns are strongly affected locally by numerous kettle-hole ponds. Ground water flows through these ponds and ground-water-flow paths converge in areas upgradient of the ponds where ground water discharges into the ponds and diverge in downgradient areas where pond water discharges back into the aquifer.

Water levels in the aquifer and in ponds fluctuate in response to seasonal and long-term changes in recharge rates. Pond stages in Snake Pond, which is to the southeast of the Impact Area (fig. 2), can fluctuate by more than 2 ft seasonally and by more than 7 ft between periods of drought and above-average rainfall (U.S. Geological Survey, accessed 4-12-02). In addition, the position of the top of the water-table mound likely changes with changes in recharge.

Ground-water-flow patterns have a stronger vertical component near the top of the water-table mound than in areas away from the mound. Ground-water flow is nearly vertical (downward) at the top of the water-table mound and nearly horizontal in downgradient areas of the aquifer. Flow near discharge boundaries, such as streams, ponds, and the coast, has a strong vertical (upward) component (fig. 3). Measured ground-water-flow rates in sand and gravel in an area to the south of Camp Edwards, where horizontal flow predominates, were about 1.4 ft/d (LeBlanc and others, 1991).

The top of the water-table mound is a ground-water divide from which ground water flows radially outward. As a result, ground-water flow directions within Camp Edwards differ depending on location relative to the position of the top of the mound. This radial flow field has important implications for the advective transport of contaminants from Camp Edwards. In the Impact Area, ground water flows to the northwest towards Cape Cod Canal, whereas contaminants from the southern J-Ranges Area flow southward toward Snake Pond (fig. 4). In addition, contaminants from the J-Ranges Area, which originate close to the top of the water-table mound, would be expected to move deeper in the system relative to horizontal transport distance than contaminants from sources located farther from the mound, such as Demolition Area 1 and the Impact Area (fig. 4). The effect of the radial flow field on advective transport is further complicated by changes in the mound position in response to changes in recharge conditions.

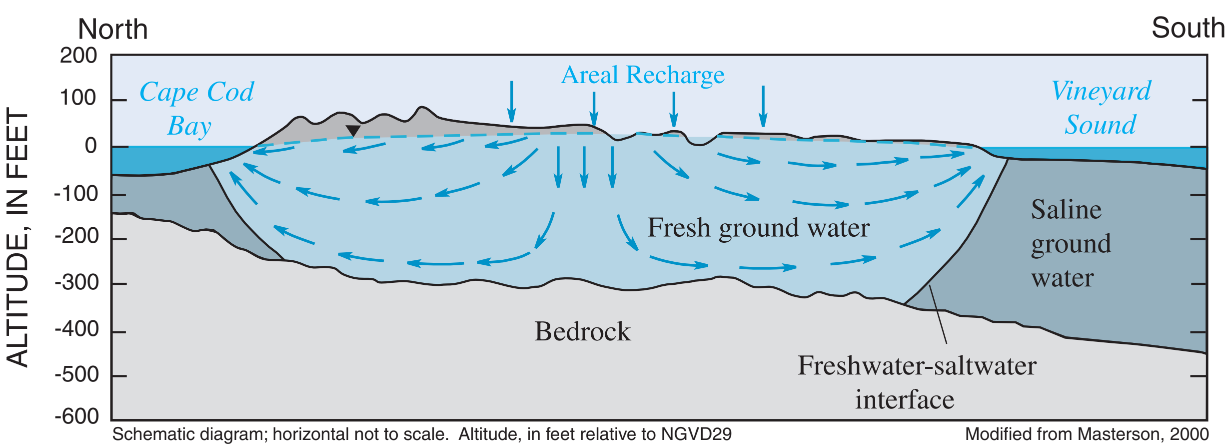
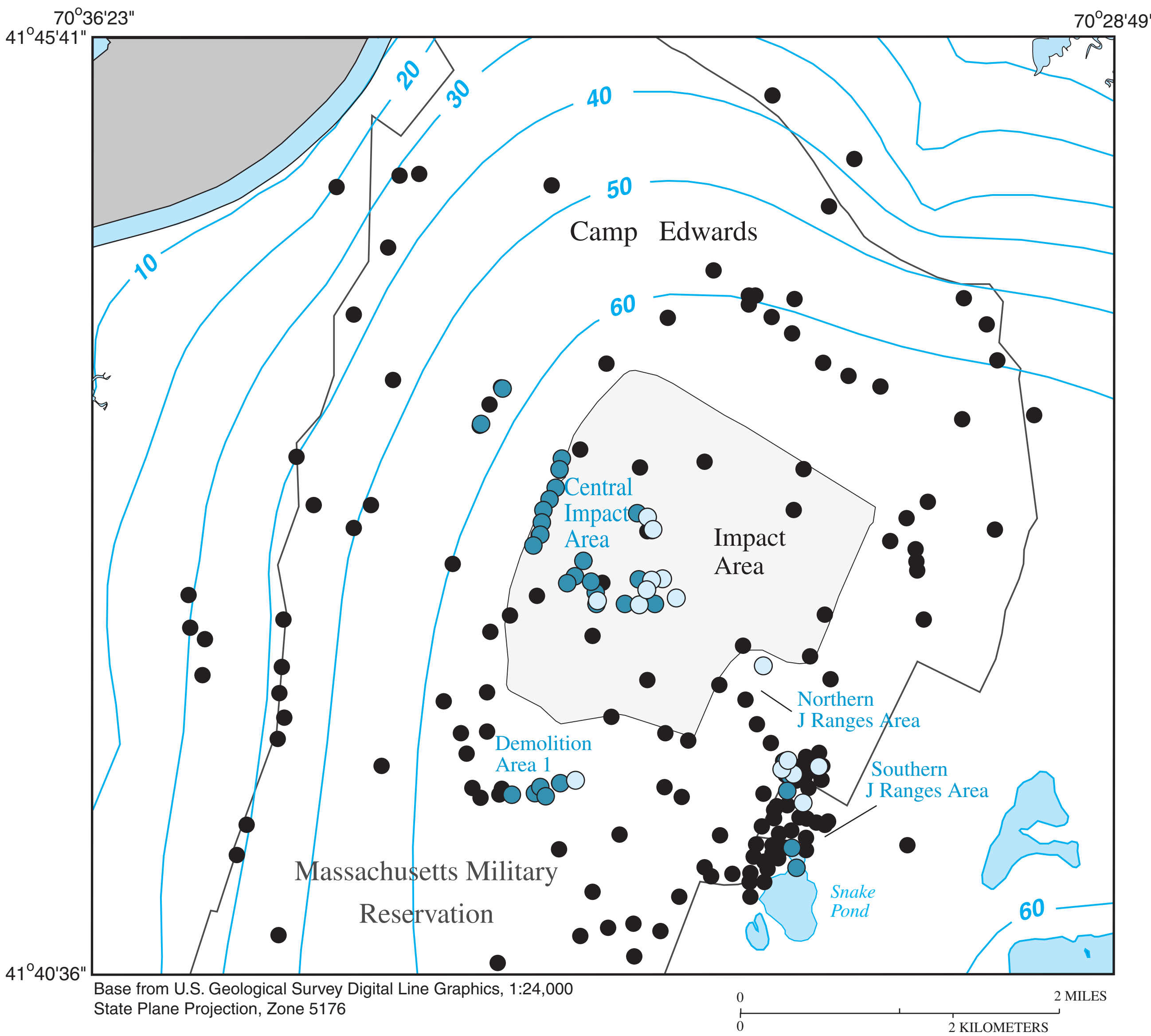


Figure 3. Generalized vertical section (vertically exaggerated) illustrating hydrologic boundaries and general flow lines in the ground-water system of western Cape Cod, Massachusetts.



EXPLANATION

—60— MODEL-DERIVED WATER-TABLE CONTOUR—Shows altitude of model-calculated water table. From 2000 Regional Model. Contour interval is 10 feet. Vertical datum is NGVD29

ROYAL DUTCH EXPLOSIVE DETECTION IN WELL—

- No detection
- In well within 10 feet of the water table
- In well more than 10 feet below the water table

Figure 4. Location of sampling locations in and around Camp Edwards, Cape Cod, Massachusetts, and wells in which Royal Dutch Explosive was detected.

Site Description and History

Military activity at the MMR, a multi-use facility that encompasses about 22,000 acres on western Cape Cod, began as early as 1911. Camp Edwards encompasses about 14,000 acres in the north-central part of the MMR and consists of the Impact Area, which is about 2,000 acres in area, surrounded by several training ranges and other facilities (fig. 1). The site was operated by the U.S. Army until about 1974 and is now used as a training facility by the ARNG. The Impact Area has been used for live-fire mortar and artillery training, and the surrounding training ranges have been used for small arms training and troop maneuvers since the mid-1930s. Other military activities at the site include ordnance training and the testing and disposal of ordnance by the military and military contractors.

As discussed previously, RDX is a contaminant of concern at the site; however, emerging contaminants, such as perchlorate, have been detected in ground water at the site. RDX was the focus of most ground-water investigations at the site at the time the modeling analyses described in this report were completed. Therefore, the focus of this report is the use of numerical models to assist in interpreting RDX contamination in the aquifer. It should be noted, however, that other emerging contaminants also have been identified as contaminants of concern in the aquifer underlying Camp Edwards.

As of June 2001, RDX had been detected in 32 of the 160 wells that had been installed and sampled as part of the investigation of ground-water contamination at Camp Edwards (fig. 4). RDX was detected in three general areas within Camp Edwards: Demolition Area 1, the J-Ranges Area, and the Central Impact Area. Figure 4 illustrates the locations of the three areas of ground-water contamination at Camp Edwards; the figure represents water-quality data available as of June 2001. Water-quality data collected since then as part of ongoing investigations at Camp Edwards are not shown in the figure. Although data collected since June 2001 have shown differences in the pattern and extent of contamination in the three areas, most observed contamination is within the same general areas. An overview of water-quality conditions in the aquifer underlying Camp Edwards, including those defined by recent (after June 2001) water-quality data, is presented below.

The area referred to as Demolition Area 1 (Demo 1) was used for training from the mid-1970s until the mid-1990s. The area, which was known as the E-2 Range until the late 1980s, was used for demolition training and the disposal of unexploded ordnance by detonation (AMEC, Inc., 2001). These activities resulted in a plume of contaminated ground water that contains RDX and extends about 5,000 ft downgradient of the source. The maximum observed concentration of 390 µg/L occurs in ground water beneath the source area. The deepest contamination is about 80 ft below the water table and occurs about 3,000 ft downgradient of the source area (AMEC, Inc., 2001).

The J-Ranges Area was used for a number of small-arms training activities from about 1935 to the mid-1980s. From the late 1960s to the mid-1980s, the western part of the J-Ranges was used for ordnance testing by military contractors. Although little is known about the exact nature of activities that occurred in this area, a number of structures that likely were used to test and dispose of ordnance have been identified. Activities in this area have resulted in contamination of the underlying ground water. Concentrations of RDX in this area are as high as 30 µg/L. Contamination emanating from the J-Ranges Area has been observed as far downgradient as Snake Pond, where RDX has been detected beneath the northern part of the pond and as deep as 120 ft below the water table. In this discussion, the northern J-Ranges Area refers to the area near the southern boundary of the Impact Area and the southern J-Ranges Area refers to the area north of Snake Pond (fig. 4).

The Impact Area has been used for live-fire mortar and artillery training since about 1940. Unexploded ordnance (UXO) is commonly observed within the Impact Area, particularly near targets. The release of explosive compounds from UXOs into the environment can occur either through low-order (partial) detonation or through deterioration of unexploded shells containing explosive compounds. A contaminant of concern in the Impact Area is RDX. The pattern of contamination, which emanates from a large number of small, isolated sources, resembles the release of contamination from a non-point source with contaminants occurring as widely distributed and sporadic detections of RDX. Sampling of soil and ground water early in the investigation indicated that most ground-water contamination was within or downgradient of the central part of the Impact Area, referred to as the Central Impact Area (CIA). Maximum observed concentrations of RDX within the CIA are about 34 µg/L. Contaminants emanating from within the CIA have been observed 3,000 ft downgradient of the Impact Area boundary and as deep as about 100 ft below the water table.

DEVELOPMENT OF MODELS

Several different numerical models were used to simulate ground-water flow in the Camp Edwards area. Steady-state regional models were developed by the USGS as part of a parallel investigation into regional sources of water to wells and natural receptors (Masterson and Walter, 2000). These models were used in this investigation to improve understanding of ground-water flow in and around Camp Edwards and to simulate advective transport at specific areas of known or suspected ground-water contamination. Two subregional models were developed as part of this investigation: a model of Camp Edwards and the surrounding area and a smaller-scale model of the southern J-Ranges Area. A transient version of the regional model was also developed and used to address advective transport within Camp Edwards under changing stress conditions.

Steady-State Regional Models

Since the investigation of ground-water contamination in and around Camp Edwards began in 1997, three successive versions of a steady-state regional model have been used to simulate advective transport in the aquifer. A regional model developed in 1993 was the first regional model developed specifically for the MMR. A second model developed in 1998 updated the 1993 model; data from the ongoing investigations by the military and their contractors was used to update hydraulic properties within the model. The latest iteration of the model, developed in 2000, updates the coastal boundaries and is the model currently being used for analysis of ground-water flow at the site. A brief synopsis of the three regional models is presented below.

1993 Regional Model

In 1993, the USGS began a cooperative investigation with the National Guard Bureau (NGB) to develop a regional model of western Cape Cod. The model was developed to improve understanding of regional ground-water-flow patterns around the MMR. The model is a finite-difference model that uses the USGS modeling program MODFLOW (McDonald and Harbaugh, 1988). A detailed documentation of the model can be found in Masterson and others (1997b). The model consists of 144 rows, 130 columns, and 11 layers and has a uniform horizontal discretization of 660 ft. The extent of the model grid and the active area of the 1993 regional model are shown in figure 5. Vertical discretization varies from 20 ft in the upper part of the model to more than 200 ft in the lowest model layer, which is bounded below by impermeable bedrock (a no-flow boundary) and has a variable thickness.

The top surface of the model was simulated as a free surface that receives recharge as the sole input of water into the model. The simulated water table cuts across model layers. It is located in layer 1 near the top of the water-table mound and falls to layer 4 near the coastline. The baseline recharge rate assigned to the model to account for recharge from precipitation is 21.6 in/yr; this value is based on previous investigations on Cape Cod (Barlow and Hess, 1993; Thornthwaite and Mather, 1957). It was assumed that 85 percent of water withdrawn for public supply is returned to the aquifer as septic system return flow. This volume of return flow was simulated as enhanced recharge in residential, non-sewered areas served by public water supplies, resulting in a spatially varying recharge rate. This areal recharge is simulated using the Recharge Package (RCH) (McDonald and Harbaugh, 1988).

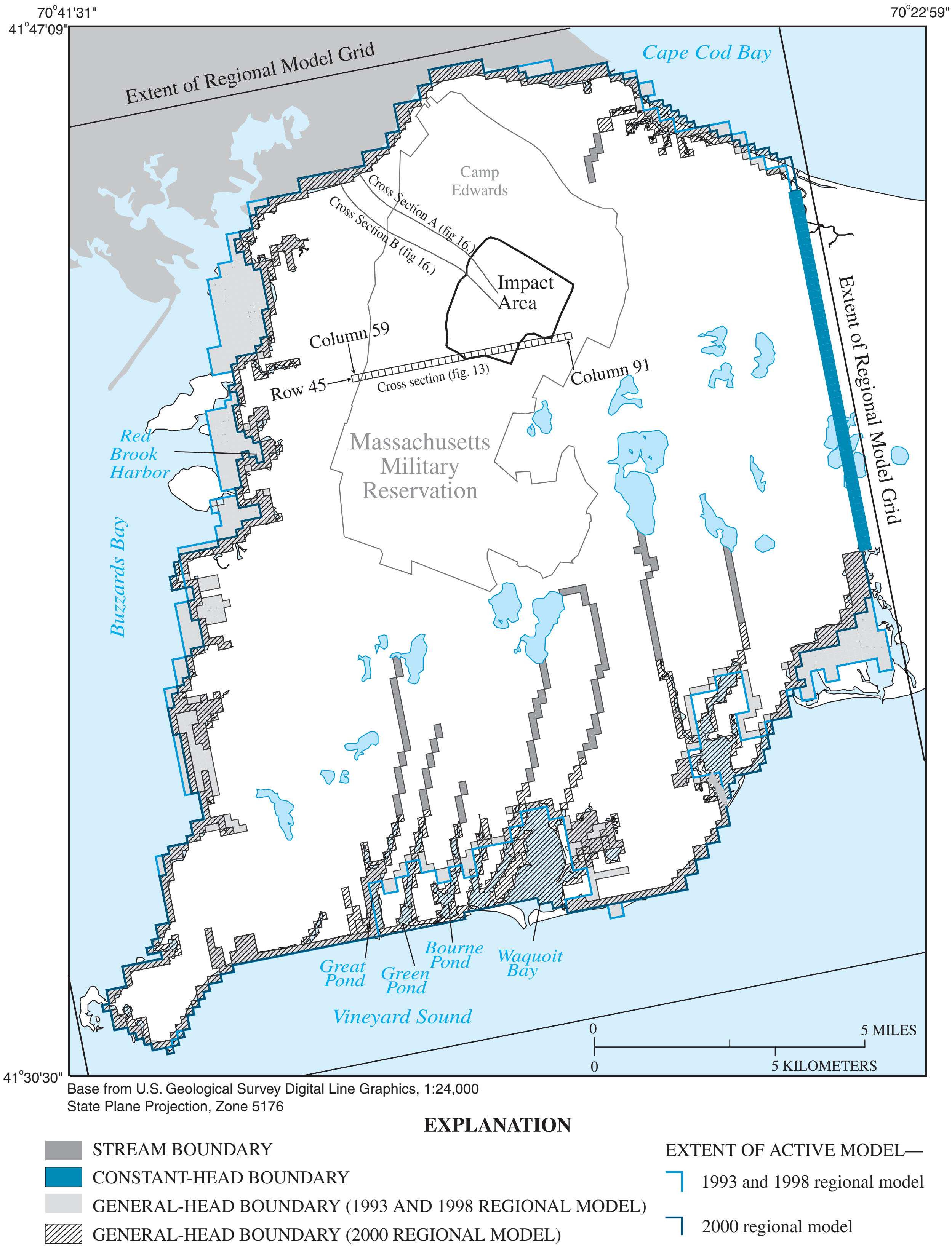


Figure 5. Regional model domains and simulated hydrologic boundaries in the 1993, 1998, and 2000 regional models of western Cape Cod, Massachusetts.

The coastal-discharge boundary in model layer 4, which corresponds to shallow, nearshore areas, is simulated using the General Head Boundary Package (GHB) (McDonald and Harbaugh, 1988) (fig. 5). The coastal boundary in lower model layers is simulated as a no-flow boundary, which results in a boundary condition in which flow is upward along a salt-water interface and discharge occurs through the seabed in nearshore areas (fig. 3). The seaward extent of the coastal boundary derives from a 1991 model of the aquifer that used the numerical model SHARP (Essaid, 1990) to simulate the position of the interface between fresh and saline water for western Cape Cod (Masterson and Barlow, 1996). The 1993 regional model is bounded to the north, south, and west by coastal boundaries (fig. 5). Along the eastern boundary, which is located far from the area of interest in the simulations, a specified head boundary derived from the 1991 regional model (Masterson and Barlow, 1996) is used (fig. 5).

Streams on western Cape Cod, where overland flow generally is negligible, receive water primarily from ground-water inflow and generally are gaining streams. In the 1993 regional model, streams are simulated by the Drain Package (DRN) (McDonald and Harbaugh, 1988) (fig. 5). This boundary condition allows ground water to discharge to the stream but does not allow streamflow to recharge the aquifer.

Glacial kettle-hole ponds in the model are simulated as areas of very high hydraulic conductivity (more than two orders of magnitude higher than aquifer hydraulic conductivities). This results in a nearly flat hydraulic gradient across the pond and is an effective way to simulate the hydraulic effect of ponds on ground-water-flow patterns in the aquifer.

A depositional model was developed for the aquifer on western Cape Cod on the basis of lithologic logs gathered through drilling in the aquifer and the pre-collapse topography of western Cape Cod (Masterson and others, 1997a). The depositional model of a glacial-lake delta bounded laterally by moraine deposits was used to determine general lithology in different areas according to location within the glacial delta. Hydraulic properties for different aquifer sediments were determined from previous aquifer-test analyses for similar hydrogeologic environments on Cape Cod and were used to assign hydraulic-conductivity values for the model domain (Masterson and others, 1997b). Hydraulic conductivity in the outwash deposits ranges from 350 ft/d for coarse sand and gravel to 10 ft/d for silt and clay and decreases with depth and to the south with increasing distance from the sediment source. Anisotropies of horizontal to vertical hydraulic conductivities range from 3:1 for coarse sand and gravel to 100:1 for silt and clay.

Pumping data compiled from local water suppliers for the period of 1986–90 was used to estimate average annual pumping rates for municipal wells simulated in the 1993 model. The Well Package (McDonald and Harbaugh, 1988) is used to simulate the withdrawal of this volume of water from the aquifer.

Hydraulic properties of aquifer sediments and leakances into streams and coastal boundaries were varied during model calibration. The model was calibrated to heads and streamflows measured as part of a synoptic-measurement event in March 1993 (Savoie, 1995), which corresponded to near-average hydrologic conditions. Contaminant plumes are good indicators of long-term average hydraulic gradients in the aquifer and mapped contaminant plumes were important calibration targets (Masterson and others, 1997b). A detailed discussion of model calibration and final model input is presented in Masterson and others (1997b).

1998 Regional Model

The regional model was updated in 1998 as part of an investigation into the source of water to municipal wells and natural receptors (ponds, streams, and coastal embayments) on western Cape Cod; this investigation was done in cooperation with the Air Force Center for Environmental Excellence (AFCEE). A detailed documentation of the 1998 regional model, which uses the USGS modeling program MODFLOW-96 (Harbaugh and McDonald, 1996), is presented in Masterson and Walter (2000). The 1998 regional model has the same horizontal and vertical discretization and the same coastal boundary condition as the 1993 regional model; however, streams in the 1998 regional model are simulated by using the Stream Routing (STR) Package (Prudic, 1989) (fig. 5). This allows simulated streams to both receive water from the aquifer (gaining conditions) and to contribute water to the aquifer (losing conditions). Although streams on western Cape Cod generally are gaining streams, streams can lose water when municipal wells induce infiltration. In addition, some streams receive water from ponds at their upstream reach, which can result in a losing condition in the stream downstream of the pond. Streamflows measured at the outlets of major ponds in March 1993 (Savoie, 1995) were used to specify streamflow input at the headwaters of streams that drain major ponds. The change in simulation approach for streams allows for a more accurate analysis of complex recharge areas that contributed water to supply wells located near streams and for a better accounting of water in the model. In the 1998 regional model, it is assumed that the volume of water specified as entering streams from pond outlets is small relative to the total volume of the pond and a corresponding volume of water is not explicitly removed from the ponds. The lower reaches of streams that were tidally influenced were represented as coastal boundaries and simulated using the GHB Package in both the 1993 and 1998 models.

Lithologic data collected since 1993 as part of ground-water investigations in the area were used to update the simulated bedrock surface and to change hydraulic properties in some parts of the model. The changes made to the model are documented in Masterson and Walter (2000).

Water-quality data collected since 1993 show that contaminant plumes extended farther downgradient than previously mapped, indicating that ground-water fluxes generally are higher and travel times faster in the aquifer than previously thought. This suggests that the aquifer generally has a higher transmissivity than was simulated in the 1993 regional model. New lithologic logs also indicate that the elevation of the bedrock surface generally is lower and that hydraulic-conductivity values generally are higher than were simulated in some parts of the 1993 regional model. The 1998 regional model was again calibrated to heads and streamflows from March 1993 and to mapped contaminant plumes. Hydraulic-conductivity values generally were increased, particularly in deeper parts of the model, and simulated baseline recharge was increased to 25.9 in/yr to increase fluxes and travel times in the aquifer. Although the distribution of hydraulic-conductivity values changed in the 1998 regional model, the values are consistent with the depositional model developed in 1993 and the same general range of hydraulic-conductivity values are used.

The 1998 regional model simulated mid-1990s and future (2020) pumping conditions. Data compiled from local water suppliers are used to estimate average pumping rates for the period 1994–96 for use in simulating current pumping stresses in the aquifer. The pumping rates and locations of future municipal wells partly derive from a separate investigation into projected water needs on western Cape Cod (Earth Tech, Inc., 1998) and are used to simulate future (2020) hydrologic conditions in the steady-state regional model. A detailed discussion of the two pumping scenarios is included in Masterson and Walter (2000).

2000 Regional Model

Another update of the regional model was done in 2000 as part of a parallel investigation of the sources of water to wells, ponds, streams and coastal embayments. The model has the same horizontal and vertical discretization as the previous two models. Lithologic logs from ongoing ground-water investigations were reviewed; however, it was determined that no significant changes to the model data sets used in the 1998 regional model were necessary. The baseline recharge rate—25.9 in/yr—and pumping stresses also were the same as those used in the 1998 regional model.

The coastal boundary was changed substantially in the 2000 regional model. The coastal boundary used in the 1993 and 1998 regional models was based on a 1991 regional model used to estimate the position of the interface between fresh and salt waters around western Cape Cod (Masterson and Barlow, 1996). The 1991 model has a horizontal discretization of 1,320 ft; the coastal boundary derived from this model is too coarse to adequately simulate many saltwater embayments along the coast. In the previous two regional models (1993 and 1998), the coastal boundary is as much as 1 mi seaward of the coastline in Buzzards Bay (western boundary) and the active model along the southern coastal boundary does not extend southward to the coast, resulting in an inaccurate representation of coastal embayments in that area (fig. 5).

A review of existing data from nearshore wells suggests that ground-water discharge in offshore areas in Buzzards Bay and Vineyard Sound is not likely. In addition, recent work by the USGS to delineate the position of the interface between fresh and salt waters beneath Red Brook Harbor, which is an embayment along the western (Buzzards Bay) coastal boundary, confirms that ground-water discharge is limited to nearshore areas (McCobb and others, 2002). The head-dependent boundary used to simulate the coast in the 2000 model more closely matches the position of the coastline and better represents nearshore coastal discharge. The modifications to the coastal discharge boundary are based on the geometry of the coastline and include adjustments to the model cells that are specified as either active, inactive, or head-dependent flux boundaries.

The areas with the most significant changes are around Great Pond, Green Pond, and Bourne Pond along the southern coast and along Buzzards Bay in Bourne (fig. 5). In the 1993 and 1998 regional models, the coastal boundary is cropped such that all ground-water discharge occurred at the landward ends of Great, Green, and Bournes Ponds, and the land spits between these embayments are not simulated (fig. 5). In the 2000 regional model, the coastal boundary conditions in this area represent the geometry of the individual embayments. The modifications include extension of the active area of the model to the southern coastline and the addition of active areas between the coastal ponds (fig. 5). The active areas between these coastal ponds have a saturated thickness of approximately 20 ft based on a field investigation conducted by the Cape Cod Commission (Thomas Cambareri, written commun., 1999).

Hydraulic conductances used to represent seabed leakances are also modified in the 2000 regional model. In the 1993 and 1998 regional models, the leakance term used in the calculation of the GHB conductance is specified as 0.2 ft/d/ft for seabed sediments. In the 2000 regional model, this value is decreased by an order of magnitude to 0.02 ft/d/ft to create the greater hydraulic gradients observed near the coast (McCobb and others, 2002). This decreased vertical leakance value is consistent with the range of seabed leakance values of 0.0001 to 0.1 ft/d/ft reported for the nearshore sediments in the Kirkwood–Cohansey aquifer system, New Jersey (Nicholson and Watt, 1997), and values of 0.01 to 1.0 ft/d/ft reported for sandy sediments, which occur over most of the Atlantic Coast Plain (Leahy and Martin, 1993).

As in the 1998 regional model, streams in the 2000 steady-state regional model were simulated by using the Stream-Routing Package. In the 2000 regional model, however, the same volumetric rate of water specified as entering streams from pond outlets is explicitly removed from the ponds by using a specified-flux boundary. This boundary condition consists of several wells distributed across the ponds that remove the appropriate volume of water. Outflows from Coonamessett, Johns, Mashpee, and Santuit Ponds were explicitly represented in the model. This approach allows for a more accurate accounting of water in the model and does not rely on the assumption that water leaving the pond through surface-water outlets is small relative to total pond outflow rates (surface and ground waters). Pond outflow was represented explicitly in the 2000 steady-state regional model to be consistent with a transient version of the model developed as part of this overall modeling effort; simulations of streamflow in the transient model required an explicit representation of the relation between pond levels and pond outflow. A detailed discussion of the approach is included in a later section.

Steady-State Subregional Models

Subregional models generally are needed to simulate advective transport in areas where there are local hydrologic boundaries such as surface-water bodies or wells. During the investigation, two subregional models were developed to simulate local ground-water flow patterns (fig. 6): (1) a subregional model of Camp Edwards was developed to better simulate areas contributing recharge to municipal wells that are downgradient of likely contaminant source areas and subsurface contaminant detections, and (2) a smaller-scale subregional model of the southern J-Ranges Area was developed to simulate advective transport near Snake Pond.

Camp Edwards Subregional Model

A subregional model of the Camp Edwards area was developed to better simulate steady-state areas contributing recharge to municipal wells in the northern part of the flow cell on western Cape Cod (fig. 6). The recharge area to a pumping well is the area at the water table across which the water that is discharging to the well originally recharged the aquifer. When a regional model is used to simulate recharge areas to municipal wells, two potential problems can arise that relate to model discretization. If the simulated volume of pumping is less than the total amount of water flux through the cell that contains the well, then the model cannot accurately represent pumping conditions and the particle-tracking methodology used to delineate recharge areas will overestimate the size of the recharge area. Also, if several wells are located in close proximity to one another (relative to model discretization), a coarsely discretized model may not adequately resolve individual recharge areas. The use of the 2000 regional model to simulate recharge areas generally is sufficient for western Cape Cod

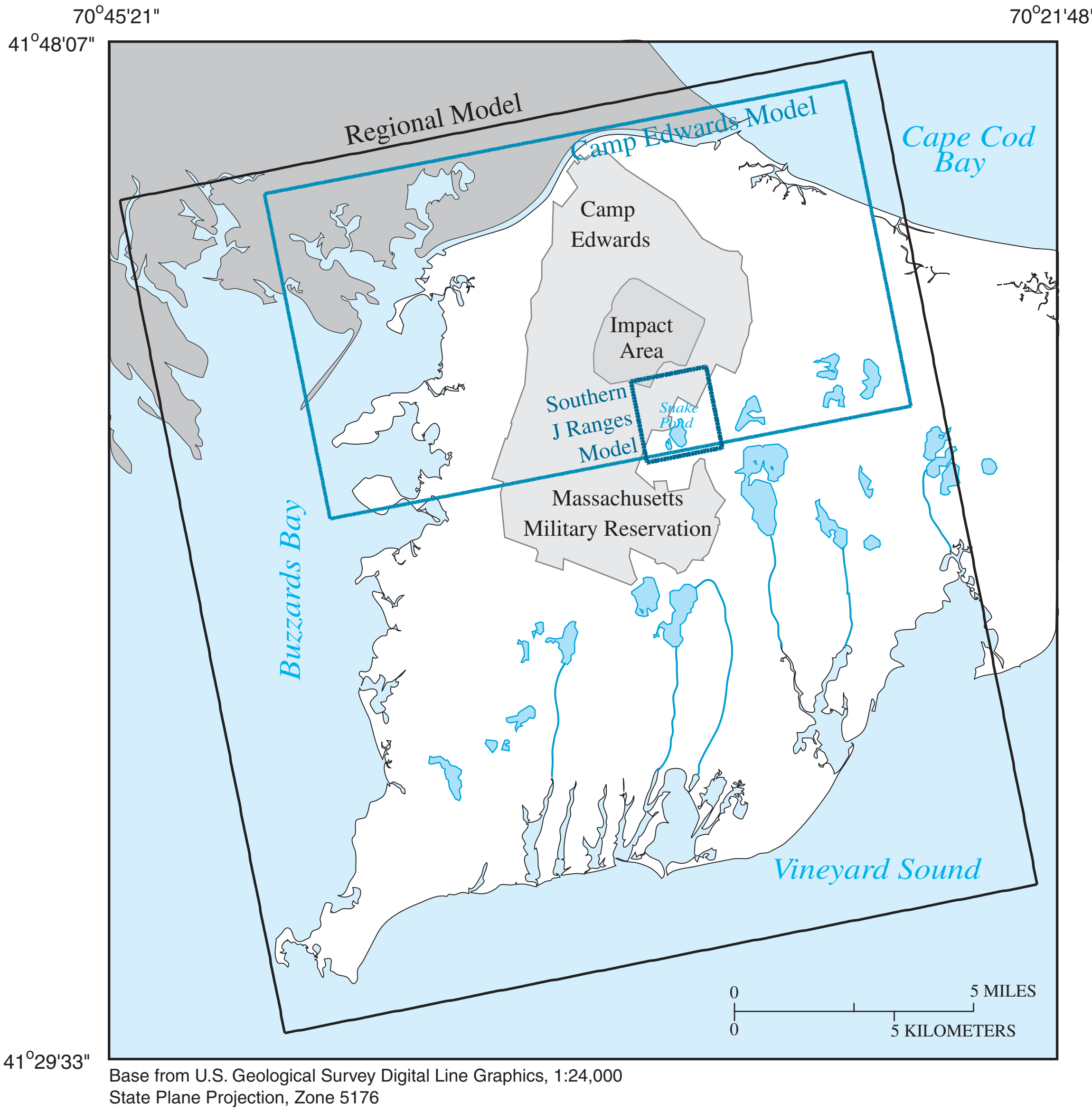


Figure 6. Extent of subregional model domains within the regional model of western Cape Cod, Massachusetts, and the subregional model of the Camp Edwards area.

(Masterson and Walter, 2000); however, the regional analysis indicates that a more finely discretized model is required to simulate recharge areas at some sites.

Grid and Boundaries

The domain of the Camp Edwards subregional model encompasses the northern part of the Sagamore Flow Cell and includes the Camp Edwards Impact Area and Training Ranges as well as areas downgradient of potential contaminant source areas on Camp Edwards (figs. 4 and 6). The model grid consists of 180 rows, 315 columns, and 20 layers. The uniform horizontal discretization of 220 ft is a ninefold increase in horizontal discretization over the regional model. The vertical discretization ranges from 10 ft in the upper 10 layers to more than 200 ft in the lowest layer, the bottom of which is a bedrock surface with a variable elevation. The vertical discretization generally corresponds to a twofold increase over that in the regional model (fig. 7*B*).

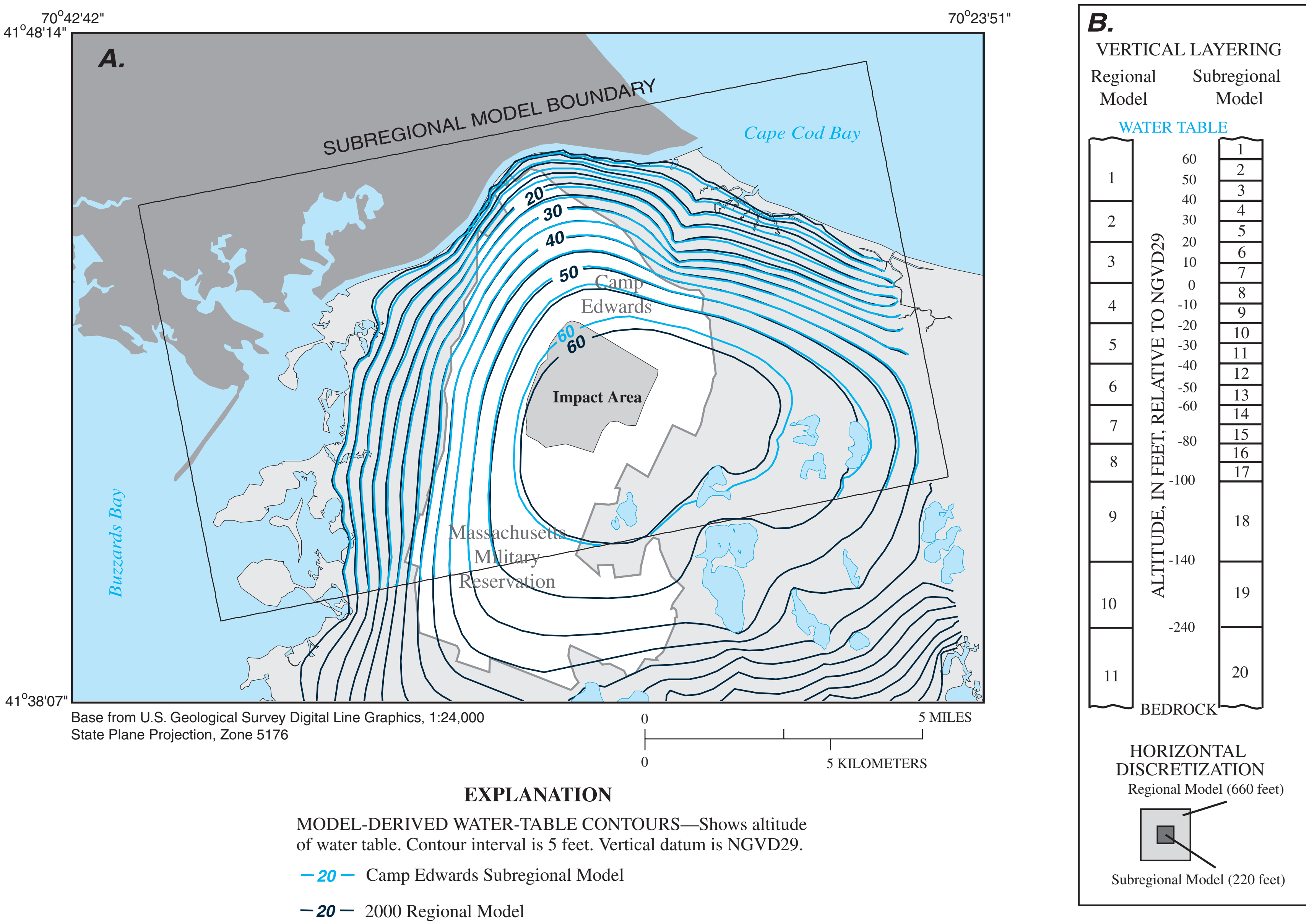


Figure 7. (*A*) Model-derived water-table contours and (*B*) vertical layering and horizontal discretization used in the regional model of western Cape Cod, Massachusetts, and in the subregional model of the Camp Edwards area, Massachusetts Military Reservation.

The subregional model uses boundary conditions derived from the 1998 regional model to insure that water fluxes through the aquifer are internally consistent between the two models. As in the regional model, the coastal boundary in the subregional model consists of a head-dependent flux boundary that is simulated using the GHB Package for MODFLOW (McDonald and Harbaugh, 1988). The same seabed conductance used in the regional model (corrected for discretization) is used in the subregional model.

Initially, two subregional models were developed that used different boundary conditions to represent internal boundaries between the regional and subregional model domains. A constant-head boundary condition was derived from the head solution produced from the 1998 regional model. The head values from the regional model were smoothed by linear interpolation to minimize the effects of the coarser regional discretization on the subregional model boundary. In addition, a specified-flux boundary condition was determined from cell-by-cell flow terms produced by the 1998 regional model. The Flow and Head Boundary Package (FHB) (Leake, 2000) was used to produce the specified-flux boundary input to the subregional model. Results from both versions of the subregional model generally are consistent and a constant-head boundary condition is used in the subregional modeling analysis.

The total net flux through the subregional model boundaries differs from the fluxes through the corresponding portion of the 1998 regional model domain by less than 1 percent. Model-calculated water-table elevations from the regional and subregional models are in close agreement (fig. 7*A*).

Aquifer Properties and Stresses

The subregional model has the same recharge rate (25.9 in/yr) and distribution of horizontal and vertical hydraulic conductivities as do the 1998 and 2000 regional models. A detailed discussion of aquifer properties used in the 1998 regional model is presented in Masterson and Walter (2000) and a discussion of the depositional model used to develop hydraulic-conductivity distributions is presented in Masterson and others (1997a). The simulated stresses representing current (1994–96) pumping conditions are the same as those in the 1998 regional model and documented in Masterson and Walter (2000). A total of 17 municipal wells are simulated in the subregional model; pumping rates range from 9,348 to 38,770 ft^3/d.

Southern J-Ranges Subregional Model

A subregional model was developed to better simulate advective transport of contaminants in the area encompassing the southern J-Ranges and Snake and Weeks Ponds, located to the south of the Impact Area (fig. 6). A number of local hydrologic boundaries limit the accuracy of the regional models in the area (fig. 8). Snake and Weeks Ponds are ground-water flow-through ponds that receive water from the aquifer in upgradient areas and contribute water to the aquifer in downgradient areas. The ponds control local ground-water-flow patterns, and an accurate simulation of pond geometries is necessary to accurately simulate advective transport around the ponds. In addition, a number of extraction and injection wells have been in operation since 1997 on the eastern side of Snake Pond to contain and remediate a nearby plume of contaminated ground water that contains benzene, toluene, and ethylene dibromide (fig. 8). The plume, known as the FS-12 Plume, emanates from the site of an aviation-fuel pipeline leak. The wells are closely spaced relative to the size of the regional model cells; the subregional model is needed to accurately simulate each well and evaluate the effects of the remediation system on ground-water-flow patterns and on the advective transport of contaminants from the southern J-Ranges Area.

The southern J-Ranges subregional model represents an update of an existing USGS subregional model developed for the area around Snake and Weeks Pond in 1997 (Walter and others, 2002). The original purpose of the model was to evaluate pond-aquifer interactions in the area under natural conditions and under stressed conditions arising from operation of the FS-12 plume-containment system.

Grid and Boundaries

The model domain of the subregional model encompasses the southern part of the J-Ranges, Snake and Weeks Ponds, and the FS-12 remediation system. The model design is based on a subregional model developed in 1997 from the 1993 regional model to

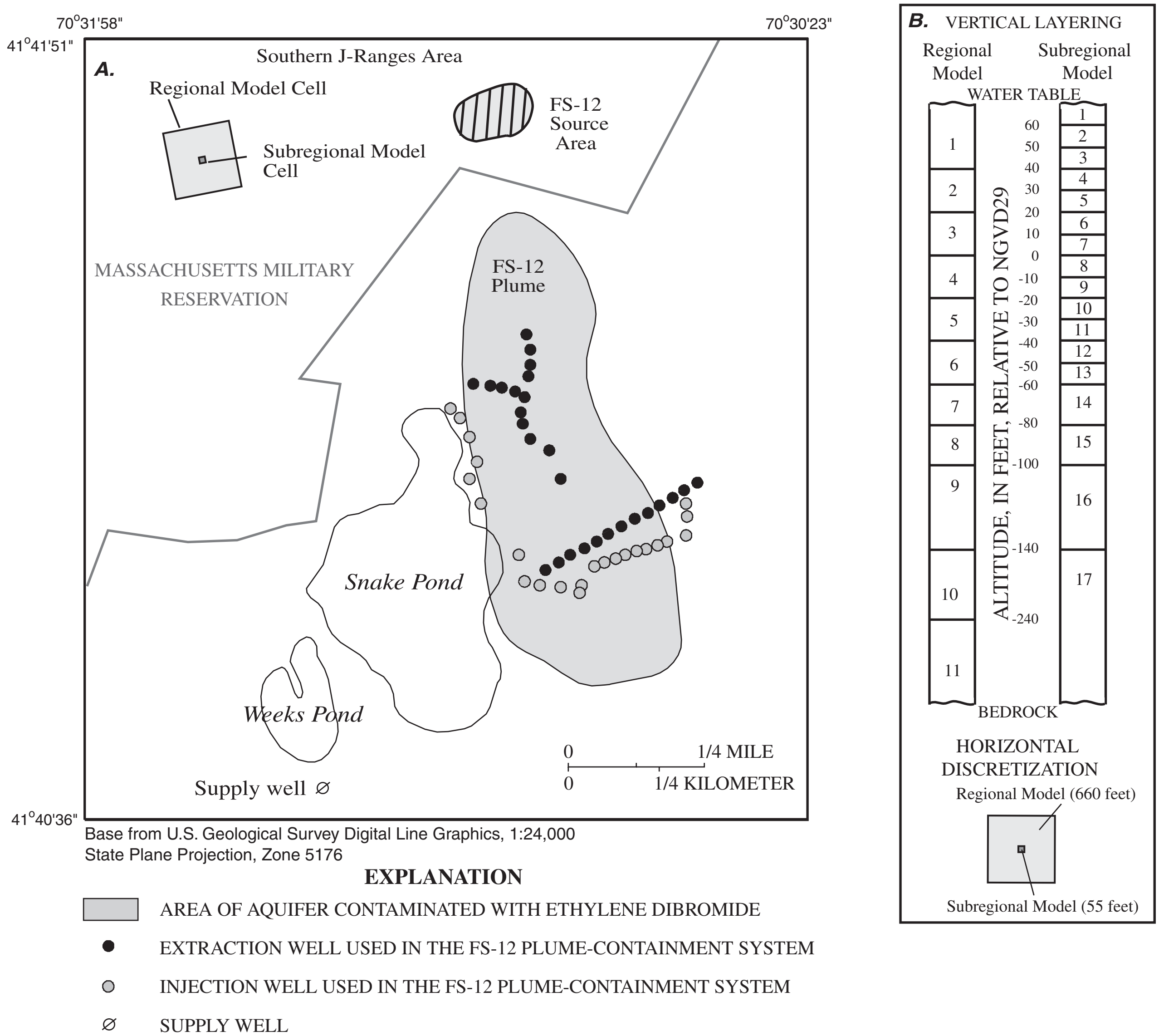


Figure 8. (*A*) Local hydrologic features and (*B*) vertical layering and horizontal discretization used in the subregional model of the southern J-Ranges Area, western Cape Cod, Massachusetts.

evaluate the hydrologic interaction between Snake Pond and the remediation system (Walter and others, 2002). The model grid consists of 168 rows, 156 columns, and 17 layers. The subregional model has a uniform horizontal discretization of 55 ft, which corresponds to 144 subregional model cells for each regional model cell within the subregional model domain (fig. 8). Vertical discretization ranges from 10 ft in the upper 13 layers to more than 200 ft in the lowest layer (fig. 8*B*).

The subregional model domain is surrounded on all sides by internal boundaries within the regional models, and there are no natural hydrologic boundaries simulated in the model. The subregional model uses a constant-head boundary condition derived from the 1998 regional model. A version of the 1998 regional model that contained a coarser representation of the remediation system was developed and its head solution was used to produce the constant-head boundaries for the subregional model. The constant heads in the subregional model were smoothed by means of node-to-node linear interpolation to minimize the effects of the coarser regional-model head solution on the subregional model solution. The fluxes through the subregional model computed on the basis of the constant head boundary were within 1 percent of fluxes through the corresponding subregion of the 1998 regional model; this agreement indicated that water fluxes through the aquifer were internally consistent between the regional and subregional models.

Aquifer Properties and Stresses

The southern J-Ranges subregional model has the same distribution of horizontal and vertical hydraulic conductivities for aquifer sediments and the same recharge rate (25.9 in/yr) as were used in the 1998 regional model (Masterson and Walter, 2000). As in the regional model, Snake and Weeks Ponds are simulated as having a high hydraulic conductivity (50,000 ft/d), which results in no effective resistance to flow within the ponds and a negligible hydraulic gradient across the ponds. The superposition of a nearly flat hydraulic gradient onto the regional hydraulic gradient causes flow lines to converge upgradient of the ponds, where the ponds receive water from the aquifer, and to diverge downgradient of the ponds, where the ponds contribute water to the aquifer. This effect on flow lines is three-dimensional and the pond effectively captures water from deep within the system. The hydrologic interaction between Snake Pond and the surrounding aquifer is discussed in detail in Walter and others (2002).

There are some differences in the representation of Snake Pond between the subregional and regional models. The smaller discretization of the subregional model allows for a more accurate representation of the geometry of the pond; the estimated volume of Snake Pond, which was determined from bathymetric data (Massachusetts Division of Fisheries and Wildlife, accessed 1-8-03) and an average pond stage of 68 ft, differs from the simulated volume by about 2 percent; this agreement indicates that the subregional model accurately represents pond size and geometry (Walter and others, 2002).

Pond-bottom sediments are more accurately simulated in the subregional model. Resistance to flow across pond bottoms was not explicitly represented in the regional model. In the subregional model, horizontal resistance to flow across the pond bottom is simulated by using the Horizontal Flow Barrier (HFB) Package (Hsieh, 1993) and vertical resistance by using specified values of vertical conductance, as represented by the VCONT parameter in MODFLOW (McDonald and Harbaugh, 1988). Values of simulated vertical hydraulic conductivity and a thickness of 10 ft were used to determine values for the VCONT parameter. Walter and others (2002) showed that the degree to which Snake Pond affected local ground-water-flow patterns was a function of the simulated permeability of pond-bottom sediments. In the subregional model, model layers 1 and 2 represent shallow nearshore areas of the pond and model layers 3 and 4 represent deeper offshore areas of the pond. Deeper offshore areas of the pond are simulated as having a pond-bottom hydraulic conductivity of 10 ft/d, corresponding to silt and clay. In shallow, nearshore areas of the pond, where most water exchange between the pond and aquifer occurs, pond-bottom sediments are assumed to consist of sand and gravel and were assigned hydraulic-conductivity values of 300 ft/d. This value is based on field observations from Snake Pond and from other ponds on western Cape Cod. A detailed discussion of the original model design, including simulation of pond-bottom sediments and the effect of this parameter on interactions between the pond and aquifer, is presented in Walter and others (2002).

The remediation system pumps, treats, and reinjects about 211,000 ft^3 of water per day through 27 extraction wells and 23 injection wells (fig. 8). The discretization of the subregional model is fine enough to simulate each well individually. The simulated stresses used to represent the remediation system currently operating near the pond were based on the final design specifications for the system (Michael Goydas, Jacobs Engineering, Inc., written commun., 2001). A public-supply well located to the south of Weeks Pond also is represented in the subregional model.

Transient Regional Models

The models discussed previously are steady-state models, which are based on the assumption that recharge rates, heads, and flows in the aquifer do not change over time. Precipitation rates on western Cape Cod can vary from about 26 to 75 in/yr and hydraulic heads in the aquifer can vary by more than 8 ft (fig. 9). Two transient versions of the regional model were

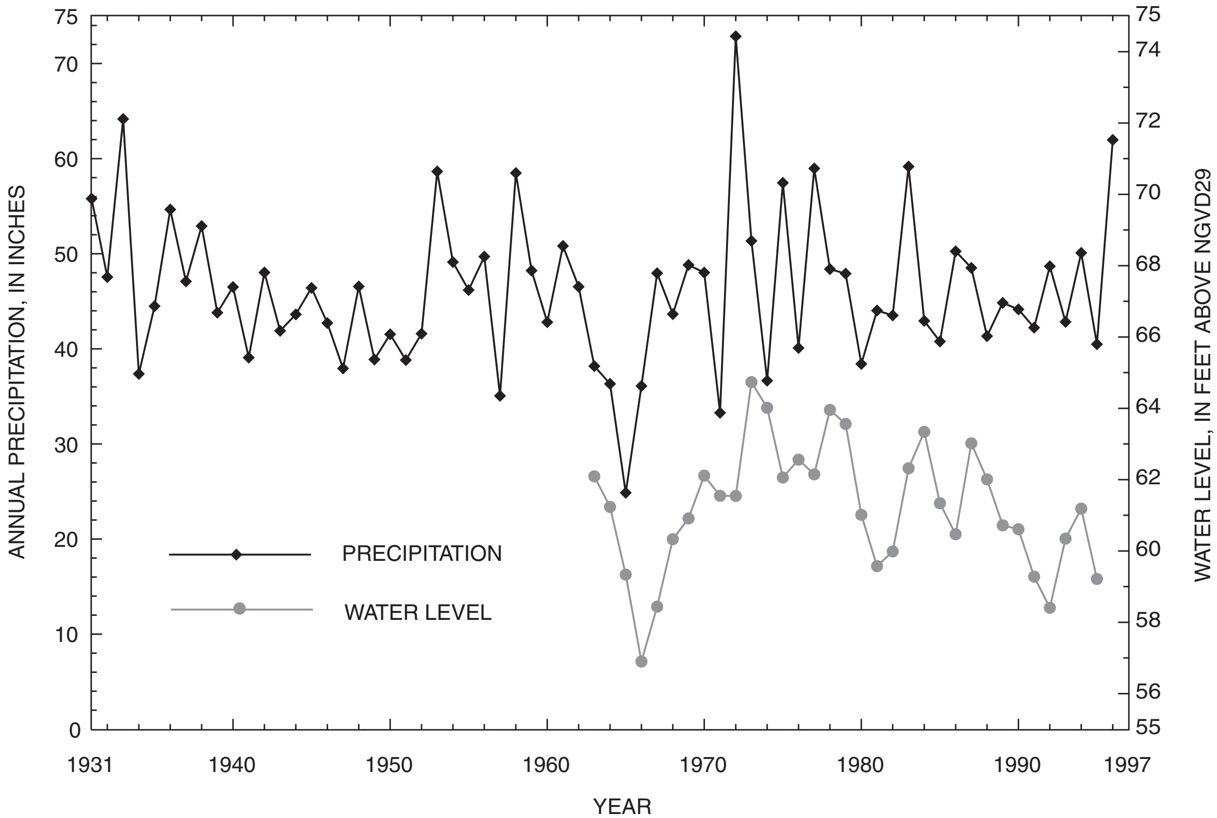


Figure 9. Changes in precipitation at Hatchville, Massachusetts, for the period 1931–96 and changes in heads in well SCW253 for the period 1963–96.

developed to incorporate seasonal and long-term changes in recharge. These models were used to evaluate the effects of time-varying recharge on advective transport in the aquifer. The same model domain, boundary conditions, and aquifer properties represented in the 2000 regional model were used in the transient models. Both transient models begin with a long (500 years) stress period in which recharge stresses are the same as in the steady-state model; this allows the model to establish a quasi-steady-state condition prior to simulation of temporal recharge stresses. In the long-term transient model, there are 59 subsequent stress periods, each 1 year long, that represent the estimated annual recharge rates for the period of 1941 to 1996 (fig. 10). In the seasonal transient model, the 100 stress periods that follow the quasi-steady-state period alternate lengths of 5 and 7 months and represent in-season and off-season recharge and pumping over a 50-year period.

The storage terms, specific yield and storage coefficient, were assigned uniformly throughout the model domain. A specific yield of 0.23 was assigned uniformly to the uppermost active model cells throughout the model domain on the basis of an aquifer-test analysis from a site to the south of the MMR (Moench, 1994). A uniform storage coefficient of 0.0002 was assigned to model cells underlying the uppermost active cells. Areas in the model that represent ponds were assigned storage coefficients of 1.0. This value was based on an analysis conducted by Barlow and Hess (1993) in an area to the south of the MMR.

Simulation of Transient Stresses

Precipitation rates at Hatchville, MA (fig. 2), measured over a period of 66 years (1931–96) were used to estimate monthly average and long-term yearly

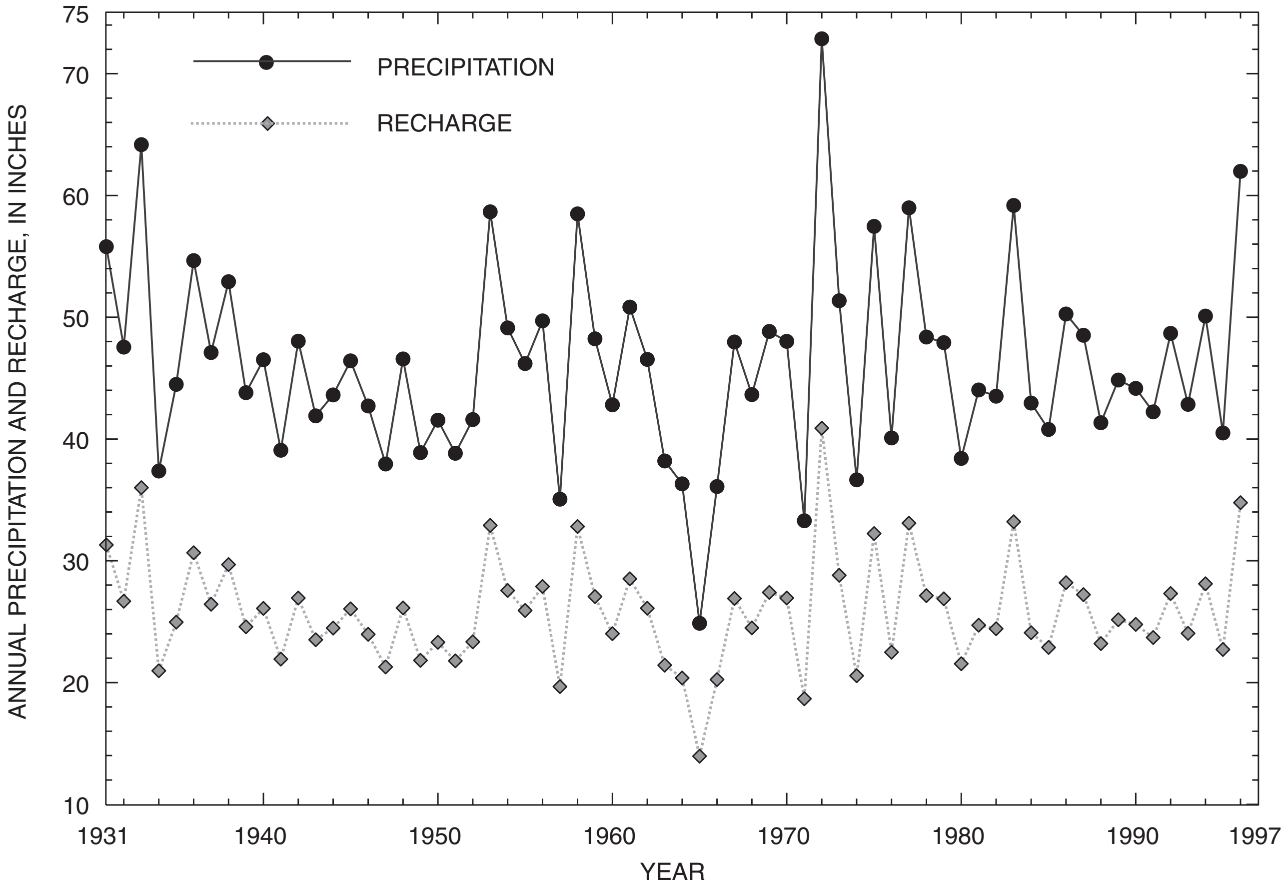


Figure 10. Precipitation and estimated recharge rates at Hatchville, Massachusetts, for the period 1931–96.

recharge rates on western Cape Cod. Measured precipitation at Hatchville has varied from a high of 74 in/yr in 1972 to a low of 26 in/yr in 1965 (fig. 9). The record includes monthly precipitation rates for each year since 1931 and was used to estimate average monthly and annual recharge rates. Long-term data regarding pumping from the aquifer was not available during this investigation and long-term changes in pumping stresses were not included in the transient model.

Long-Term Recharge and Pumping Stresses

Measured precipitation from Hatchville, MA, was used to estimate annual and monthly average recharge rates by applying analytical results from a recent investigation in a nearby basin. Barlow and Dickerman (2000) developed relationships between measured precipitation and recharge estimated from base-flow measurements over a 20-year period in the Hunt River Basin, Rhode Island, which is underlain by glacial sediments similar to those on western Cape Cod. Ratios of precipitation to recharge for each month from the Hunt River Basin were applied to the precipitation data from western Cape Cod and used to estimate monthly recharge using the long-term precipitation record. The estimated monthly recharge estimates were used to determine monthly average recharge rates for use in the seasonal transient model and annual average recharge rates for use in the long-term transient model. The estimated recharge rates, which had a long-term average of 24.6 in/yr, were normalized to a long-term average of 25.9 in/yr to be consistent with the steady-state regional model. The annual recharge rates shown in figure 10 are baseline natural rates.

The estimated annual average recharge rates were adjusted for septic-system wastewater returnflow and used to generate a simulated recharge record for the period 1941 to 1996. Changes in land use over this time period and the resulting changes in returnflow were not incorporated into the estimated recharge record. Estimated recharge rates ranged from 41.0 in/yr in 1972 to 14.0 in/yr in 1965 (fig. 10). One significant element of the constructed recharge record is a period of four years in the mid-1960s during which recharge was below average, resulting in a significant drought.

Under current (1994–96) conditions, pumping from the aquifer composes only about 6 percent of the total hydrologic budget of the aquifer and occurs primarily in the southern part of the aquifer (Masterson and Walter, 2000). It is assumed that changes in pumping over time would have little effect on regional ground-water-flow patterns in the aquifer, particularly in the northern part of the flow cell. Therefore, temporal changes in pumping rates were not incorporated in the model.

Seasonal Recharge and Pumping Stresses

The precipitation record from Hatchville, MA (fig. 10), was used to estimate long-term average recharge rates for each month in an average year. Average monthly recharge ranged from 2.3 in/yr in July to 10.8 in/yr in March. Total recharge in an average year was 24.6 in/yr. The monthly recharge rates then were normalized to an average annual recharge rate of 25.9 in/yr, which is the recharge rate used in the steady-state model. The normalized monthly recharge rates were used to estimate average in-season (mid-May to mid-October) and off-season (mid-October to mid-May) recharge rates of 14.5and 31.8 in/yr, respectively. These recharge stresses are incorporated into a seasonal transient model by developing a simulated 5-year recharge record consisting of 10 stress periods alternating in length between 5 and 7 months, representing in-season and off-season recharge.

Monthly pumping data from local water suppliers were used to partition average annual pumping rates into in-season and off-season periods of pumping. The in-season pumping rates represent the average of pumping rates from mid-May through mid-October (5 months) and the off-season pumping rates represent the average pumping rates from mid-October through mid-May (7 months). The pumping rates are incorporated into the seasonal transient model.

Limitations of Models Simulating Time-Varying Recharge

The recharge estimates for western Cape Cod are not directly based on local streamflow. The method used to estimate recharge rates from the long-term precipitation record is based on the assumption that western Cape Cod and the reference basin in Rhode Island are hydrogeologically similar. Although there are uncertainties in the recharge estimates, the method incorporates major elements of the precipitation record from western Cape Cod and is a good approximation of general recharge values and trends. Therefore, the analysis illustrates concepts related to the effect of transient recharge on advective transport in the aquifer. The results of the transient modeling analysis are intended to illustrate the general effects of time-varying recharge on ground-water-flow patterns in the aquifer and how these effects may vary within different areas of the aquifer. In the discussion of transient-model results, years are used for convenience of the discussion; these results should not be interpreted, however, as specific estimates of hydrologic conditions for the specified year. In addition, long-term changes in pumping stresses, which could affect local ground-water-flow patterns, are not included in the analysis.

Simulation of Transient Streamflow

Field data indicate that changes in recharge rates can cause pond levels to fluctuate by several feet on western Cape Cod, which can affect the amount of water discharging to streams from pond outlets. In the 2000 steady-state regional model, flows at pond outlets were specified on the basis of discharge measurements made in March 1993 (Savoie, 1995). A limitation of this approach is that these values do not change in response to changes in pond levels. For the analysis of transient flow, the relation between pond level and surface-water outflow to the adjoining stream is simulated explicitly.

The approach that is used to explicitly simulate the relation between pond levels and the resulting surface-water flow at pond outlets is to designate the model cells that contain the location of pond outlets within the ponds as the uppermost stream cells in the STR package. The streamflows to these uppermost stream cells are set to zero and the streambed hydraulic conductance is set to a high value (1×10^9 ft^2/d). The elevations specified for the stream stage, streambed bottom, and streambed top are determined by uniformly distributing pumping wells throughout the areas representing the ponds to remove from the ponds the average amount of water that is assumed to be discharging from the ponds to the adjoining streams. This set of analyses, which were done by using the steady-state regional model, were used to determine the appropriate pond level to be incorporated into the transient models.

The model-calculated, steady-state pond levels are then specified as the elevation of the stream stage (and the top of the streambed) for each model cell that includes the locations of pond outlets. The top and bottom of the streambed are set to a value 0.2 ft below these elevations to represent the structure controlling flow at the pond outlets. The method of using the resultant elevations, coupled with the extremely high hydraulic conductance, reproduces the total amount of water pumped from the pond cells as discharge to the uppermost stream cells.

The benefit of this approach is that it allows pond-level fluctuations to affect the simulated surface-water outflow to the adjoining stream. As pond levels increase in response to higher simulated seasonal recharge rates, the flow from the ponds to the streams also increases. As pond levels decrease because of decreased recharge, the surface-water outflow from the pond will decrease until the pond level drops below the specified top and bottom streambed elevation, at which time all flow from the pond stops.

A limitation of this approach is that it requires knowing the surface-water discharge at the pond outlet for average flow conditions. Limited flow data has been collected at the pond outlets. Measurements reported by Nelson (1999) of flow from Mashpee Pond to the Mashpee River in April 1994 and October 1997 suggest that the flow is in the range of 1.0 to 4.0 ft^3/s for average conditions, whereas the measured value in March 1993 was 11.9 ft^3/s. Major streams drain four ponds on western Cape Cod: Coonamessett Pond, Johns Pond, Mashpee Pond, and Santuit Pond. The values of surface-water discharge represented in the transient regional model for these ponds were 1.0, 1.5, 2.0, and 2.5 ft^3/s, respectively.

Comparison of Transient Model Results to Measured Heads

Head measurements have been collected at a limited number of monitoring wells on western Cape Cod since about 1960. Figure 11 illustrates measured and simulated heads from wells SDW253 and BHW215, which are located near Camp Edwards, and well FSW167, which is located to the south of the MMR (fig. 2); heads have been measured at well SDW253 since 1962 and at wells BHW215 and FSW167 since 1975. Calibrated head residuals (observed head-simulated head) from the 2000 steady-state regional model for wells SDW253, BHW215, and FSW167 were 0.5, 4.8, and -0.8 ft, respectively. The data indicate that simulated heads at wells SDW253 and FSW167 closely matched long-term average heads in those areas and that the simulated head at well BHW215 was 4.8 ft lower than the long-term average head at that location. Well BHW215 is located in an area of high hydraulic gradients; such areas are often difficult areas in which to calibrate regional models to measured heads.

Long-term average heads at wells SDW253 and FSW167 (simulated by using the transient model) are within about 2 ft of the average measured head (fig. 11). The average simulated head at well BHW215 is about 6.7 ft lower than the average measured heads at the well; as discussed previously, this difference likely is due to the location of well BHW215 in an area with a steep horizontal hydraulic gradient.

The model effectively represents general temporal trends in hydraulic heads as seen in the observed and simulated hydrographs from wells SDW253, BHW215, and FSW167 (fig. 11). Measured heads varied by a total of 6.7, 4.6, and 3.4 ft, respectively, over the periods of record; simulated heads over the same periods in the same wells vary by 7.2, 6.0, and 4.0 ft, respectively. The long-term drought in the mid-1960s is represented in both the observed and simulated hydrographs from well SDW253. The comparison between field-measured and simulated heads indicate that the transient model represents general hydrologic conditions observed in the field and can be used to evaluate advective transport in the aquifer under a transient flow field.

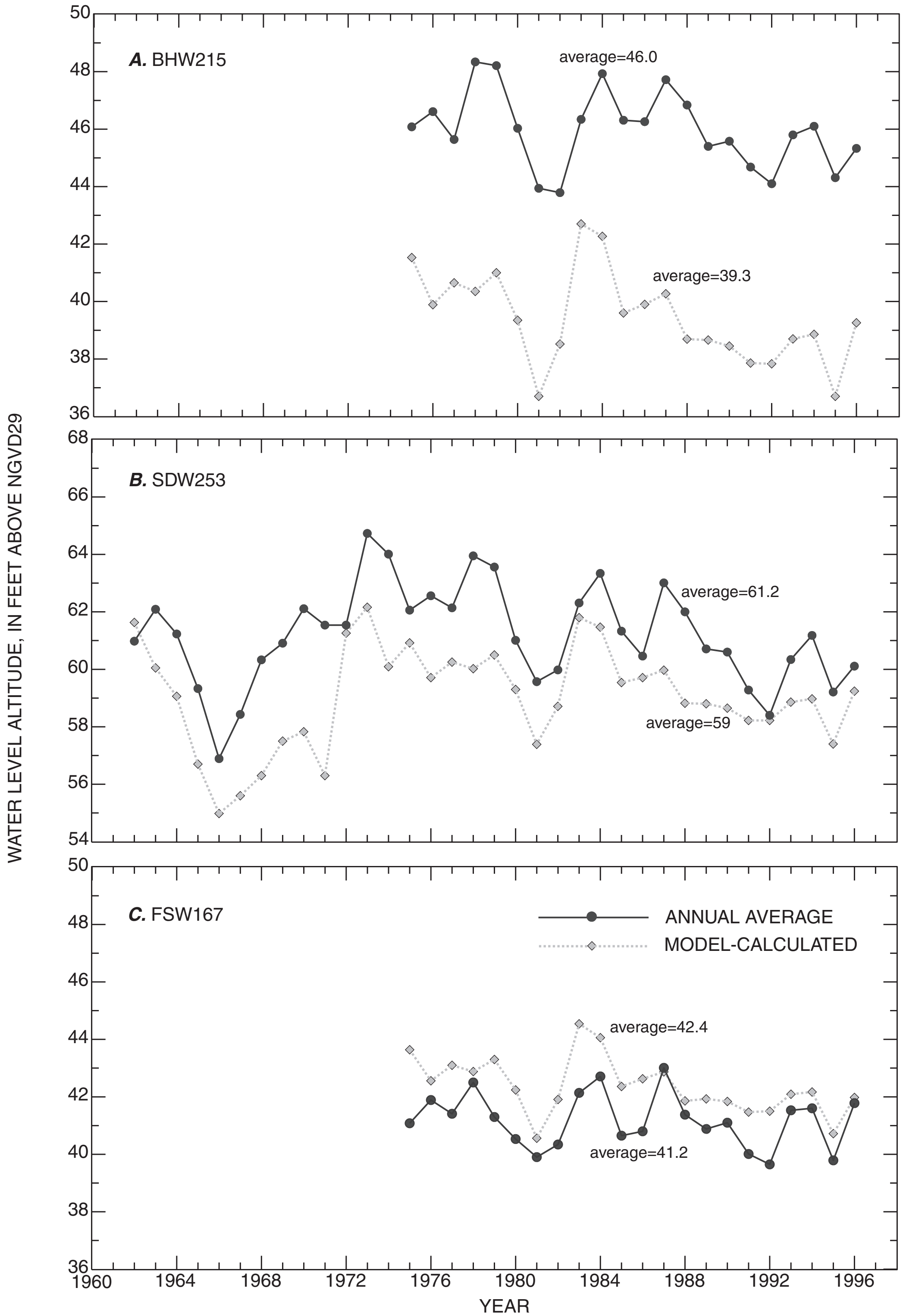


Figure 11. Changes in measured and model-calculated heads in wells (*A*) BHW215, (*B*) SDW253, and (*C*) FSW167 for the periods 1962–96, 1975–96, and 1975–96, respectively, western Cape Cod, Massachusetts.

Particle-Tracking Methodology

The flow-path analysis used to support field investigations of potential contaminant sources and their downgradient effects used the particle-tracking package MODPATH3 (Pollock, 1994). The movement of water is simulated by tracking a particle of water through the model domain according to the velocity field computed from the cell-by-cell flow terms produced by MODFLOW. The movement of a number of particles of water can be traced from their three-dimensional starting locations forward in the direction of ground-water flow (forward in time), which is referred to as forward tracking, or backward against the direction of ground-water flow (backward in time), which is referred to as reverse tracking. Forward tracking allows the user to simulate the movement of water from a particular location within the aquifer toward downgradient discharge locations. Reverse tracking allows the user to trace water from a point within the aquifer back to a starting point at the water table.

The travel times of the particles along the length of both forward and reverse tracks can be computed. The simulated travel time is proportional to the estimated porosity; uncertainties in estimated porosity translate directly to uncertainties in simulated travel times. The porosity of unconsolidated glacial sediments generally ranges from 0.3 to 0.4 (Fetter, 1988). Particle movement simulated with an estimated porosity of 0.3 will be 33 percent quicker than that simulated with the same flow-model results but with a porosity of 0.4. A porosity of 0.39 was used in this analysis; this porosity was derived from a tracer test conducted near the southern boundary of the MMR (Garabedian and others, 1991).

Particle tracking can be used to determine areas contributing recharge to pumped wells by forward tracking through the model a large number of particles, which are started at the water table. Particles that enter model cells used to represent municipal wells can be identified; the spatial distribution of the starting locations of the identified particles represents the area at the water table that contributes water to the well. The USGS software package MODTOOLS (Orzol, 1997) was used to visualize particle-tracking results. The software uses the Geographic Information System (GIS) ARC/INFO to convert output data from MODPATH3 into three-dimensional, geo-referenced images.

STEADY-STATE SIMULATIONS

Steady-state models, which simulate ground-water flow under conditions of constant recharge and pumping stresses, were used to support field investigations of potential sources of contamination at Camp Edwards. The results of simulations made with the regional models improve understanding of ground-water-flow patterns in and around Camp Edwards and allow for data collection and analysis to be done in a regional hydrologic context. Ground-water flow in areas where the utility of regional models is limited, such as near pumping wells and surface-water bodies, can be better evaluated by simulations made with the steady-state subregional models.

Regional Modeling

Regional models were used over the course of this investigation to support the field investigation in a variety of ways: to simulate the advective transport of contaminants in the aquifer and to determine the areas contributing recharge to municipal wells. For the purposes of this discussion, the 1993, 1998, and 2000 regional models are singularly referred to as the regional model.

Delineation of Ground-Water Flow and Advective Transport of Contaminants

The regional model was used to support field investigations by improving understanding of ground-water flow in the aquifer and developing the hydrologic context under which the investigation was conducted. In addition, the model supplied information regarding ground-water flow in specific areas of concern to the ARNG and its contractors.

Delineation of Ground-Water-Flow Patterns

Results of the regional model improve our understanding of ground-water-flow patterns in the aquifer underlying Camp Edwards. Figure 12 illustrates model-calculated horizontal velocity vectors in layer 3 (0 to 10 ft in elevation) of the regional model. Velocity vectors radiate outward from the top of a water-table mound located near the northern J-Ranges Area. The magnitudes of horizontal velocity vectors, as represented by the lengths of the flow vectors in figure 12, are small near the top of the mound and increase in downgradient areas. Vertical flow in the aquifer, as illustrated by flow paths along a vertical section through the aquifer (fig. 13), is greatest relative to the horizontal component of flow in areas near the top of the water-table mound, such as the northern J-Ranges Area. In these areas, flow in the aquifer is mostly downward and horizontal hydraulic gradients are small. In areas downgradient of the water-table mound, such as Demolition Area 1, the ratio of horizontal to vertical gradients is larger and horizontal flow predominates (fig. 13). Over a specified period, contaminants originating in areas dominated by horizontal flow, such as Demolition Area 1, would migrate longer distances than contaminants originating in areas near the top of the water-table mound where vertical flow predominates.

Simulation of Advective Transport to Support Field Investigations

Advective transport within the steady-state flow system is simulated by using the particle-tracking methodology applied to output from the regional model. Forward particle tracking simulates the advective transport of contaminants from subsurface detections and source areas at the water table to areas in the direction of ground-water flow (toward downgradient discharge locations). This information is useful to select the potential locations of observation wells in areas downgradient of known or possible contaminant sources. Forward tracking also estimates travel times of contaminants to wells and natural ground-water receptors. Reverse particle tracking can determine possible source areas at the water table of contaminants detected in the subsurface. A detailed documentation of all particle-track analyses done as part of this investigation is not included in this report; uses of forward and reverse particle tracking at two specific areas at Camp Edwards are described to illustrate general activities done in support of the field investigations.

In this report, comparisons are made between specific modeling analyses and water-quality data available at the time the modeling analyses were completed. Data discussed in these comparisons may not represent fully interpretations of more recent water-quality conditions (July 2002) as presented in the section “Site Description and History.”

Forward particle tracking was used to determine ground-water-flow patterns near Demolition Area 1. The source area consists of a 5-acre depression where ordnance training and disposal were conducted since the mid-1970s (AMEC, 2001). Initial sampling of the area showed that soil and ground water beneath the area were contaminated with RDX. The regional model simulates the direction of ground-water flow from the

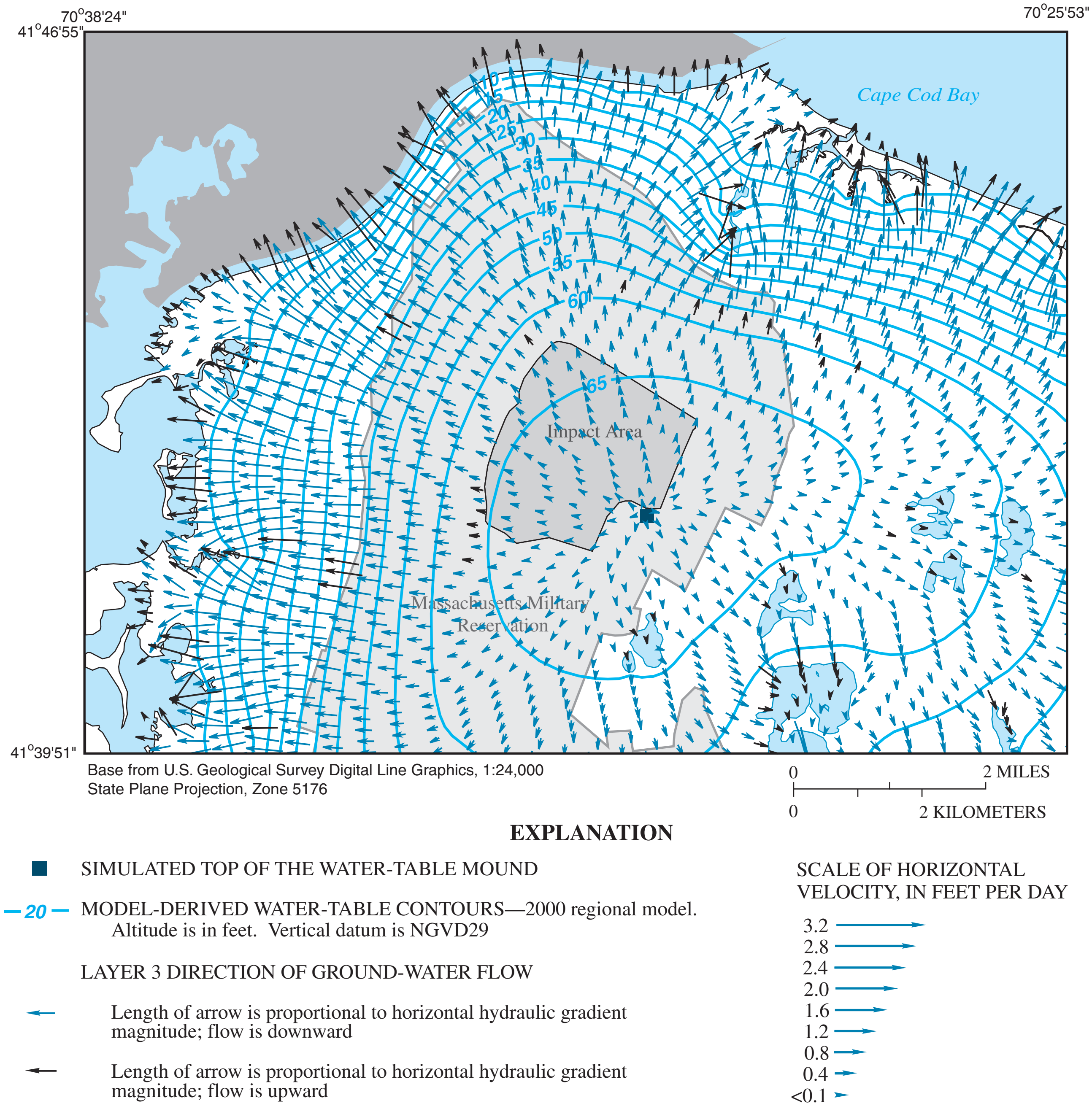


Figure 12. Model-calculated horizontal hydraulic gradients in the Camp Edwards area and the simulated location of the top of the water-table mound as determined from the 2000 regional model, western Cape Cod, Massachusetts.

water table beneath the source area; the particle tracks from the source area were used to select locations of new downgradient observation wells (fig. 14). The model with particle tracking estimates travel times along the flow paths. Sampling of water from the newly installed observation wells indicated a plume of RDX-contaminated ground water downgradient of the source area (fig. 14). RDX moves relatively conservatively in the sand and gravel material characterizing the aquifer under the MMR (AMEC, 2001). The farthest downgradient detections of contaminants were at a travel distance corresponding to about 17 years of

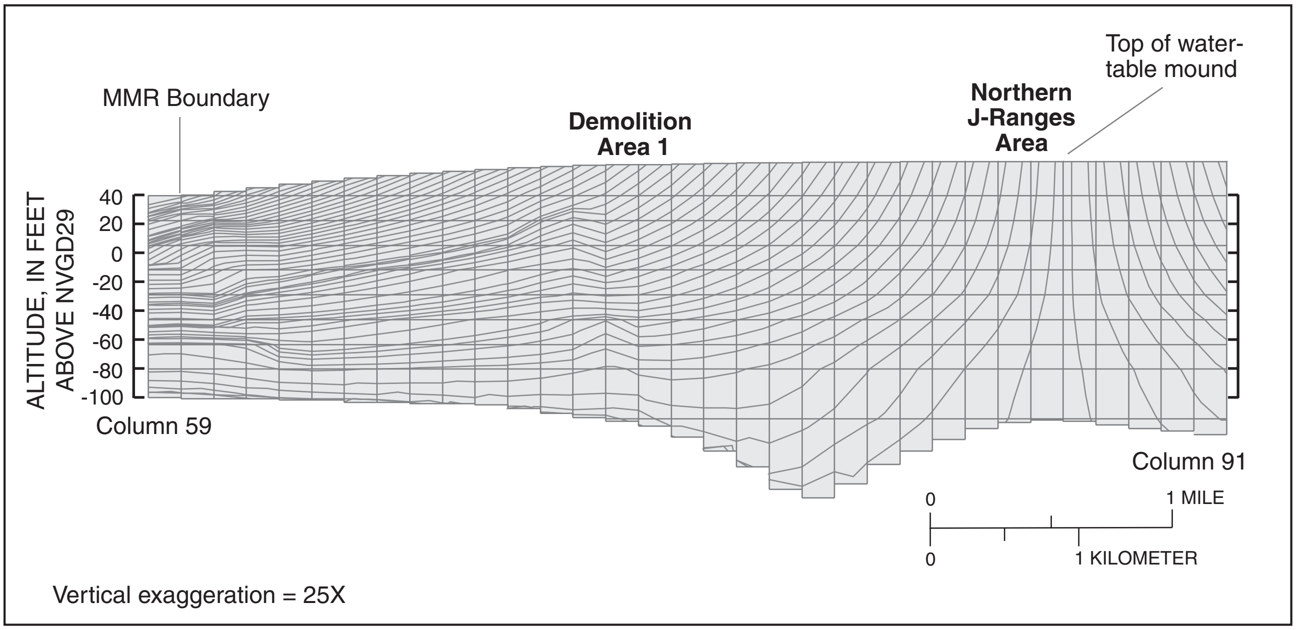


Figure 13. Flow paths calculated by the 2000 regional model along regional model row 45, and approximate locations of the northern J-Ranges Area and Demolition Area 1, western Cape Cod. Massachusetts (location of section shown in fig. 5).

transport from the source area; this travel time agrees with the known use of the demolition site. In this case the model provides an accurate tool to use to select suitable locations of downgradient observation wells and to predict the general extent of contamination downgradient of the source area (fig. 14).

Reverse particle tracking was used in the Impact Area to determine the origin at the water table of contamination detected at depth in the aquifer. Ground-water contamination in the Impact Area was thought to emanate from a number of small, isolated sources and, as a result, the area did not have large, well-defined contaminant sources. Ground-water contamination in and around the Impact Area is characterized by sporadic detections of RDX and well-defined plumes generally have not been found in the underlying aquifer. Observation wells initially were placed within and downgradient of the Impact Area and used to determine the presence of RDX and other contaminants of concern in ground water; contamination was detected in a number of wells (fig. 15). Reverse particle tracking was used to estimate the general location at the water table from which the contaminants probably originated; the contamination in the Impact Area originated in areas where a number of targets are located, including mortar ranges and an artillery-training area known as Tank Alley (fig. 15). The results indicate that the model can be used to link ground-water contamination detected at depth in the aquifer to likely source areas at the water table.

The distribution of sources within the Impact Area has resulted in sporadic detections and diffuse patterns of contamination in the subsurface. Forward and reverse particle tracking were used to determine relationships of detections at different locations and depths in the aquifer to common source areas. Two examples from within the Impact Area illustrate this application of particle tracking (fig. 16). In both examples, particle-tracking results show that several detections within the aquifer at different locations and depths are related to a common source area and may represent poorly defined plumes of contamination. The forward tracks also identify areas into which contamination will migrate with time and the ultimate discharge locations. This application of particle tracking allows water-quality data to be interpreted in a hydrologic context, particularly when source areas and histories are poorly understood.

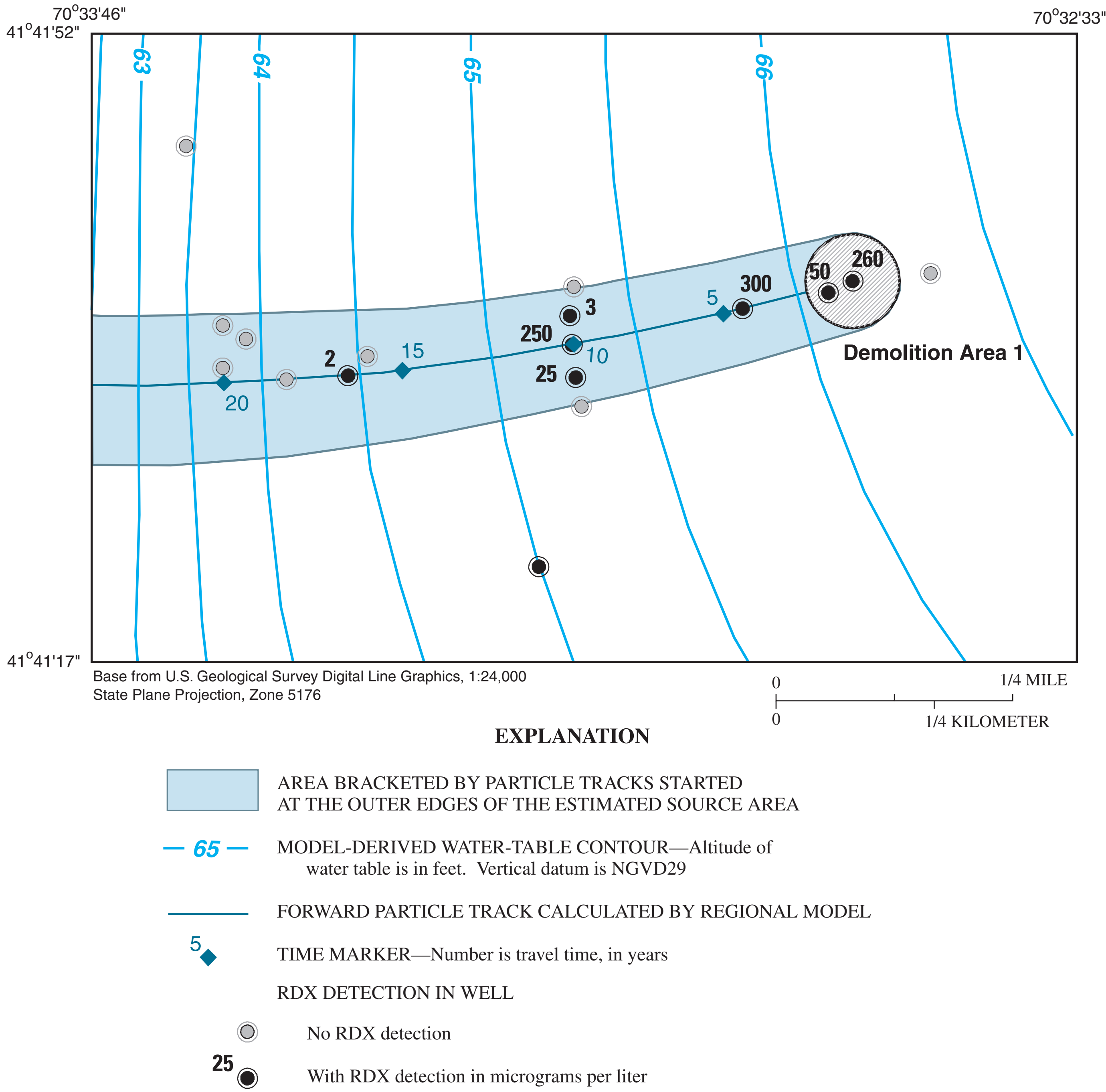
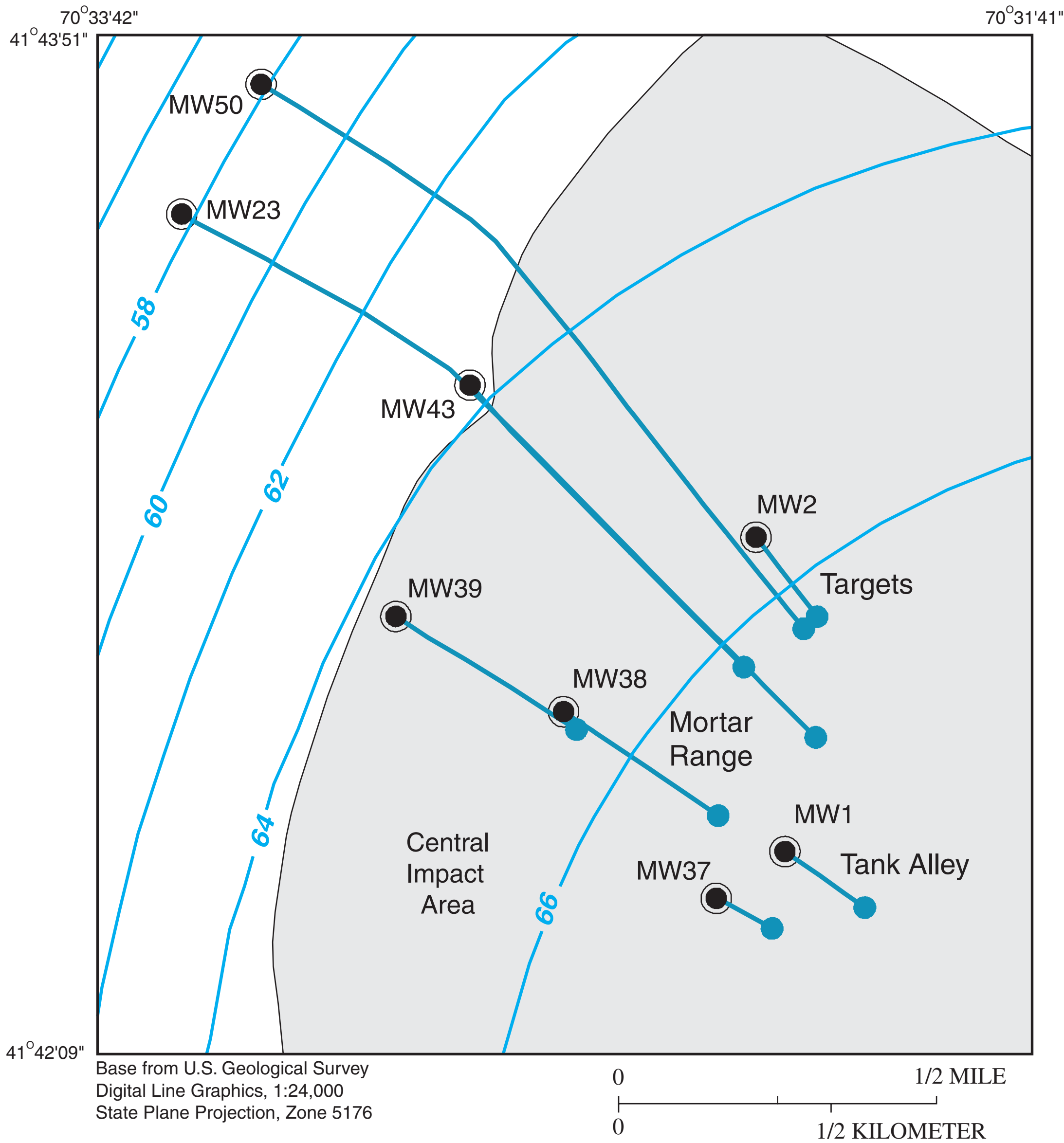


Figure 14. Advective flow paths and distribution of Royal Dutch Explosive (RDX) contamination in Demolition Area 1, 2000 regional model, western Cape Cod, Massachusetts.



EXPLANATION

MODEL-DERIVED ADVECTIVE FLOW PATH

66 MODEL-DERIVED WATER-TABLE CONTOUR—Number is altitude, in feet. Vertical datum is NGVD29

RDX DETECTION IN WELL—

Observation well with RDX detection and well identifier

Point of origin of RDX contaminants at the water table

Figure 15. Advective flow paths determined by the 2000 regional model's particle-tracking routine from locations of Royal Dutch Explosive (RDX) detections at depth in the aquifer to possible source areas in the Central Impact Area, western Cape Cod, Massachusetts.

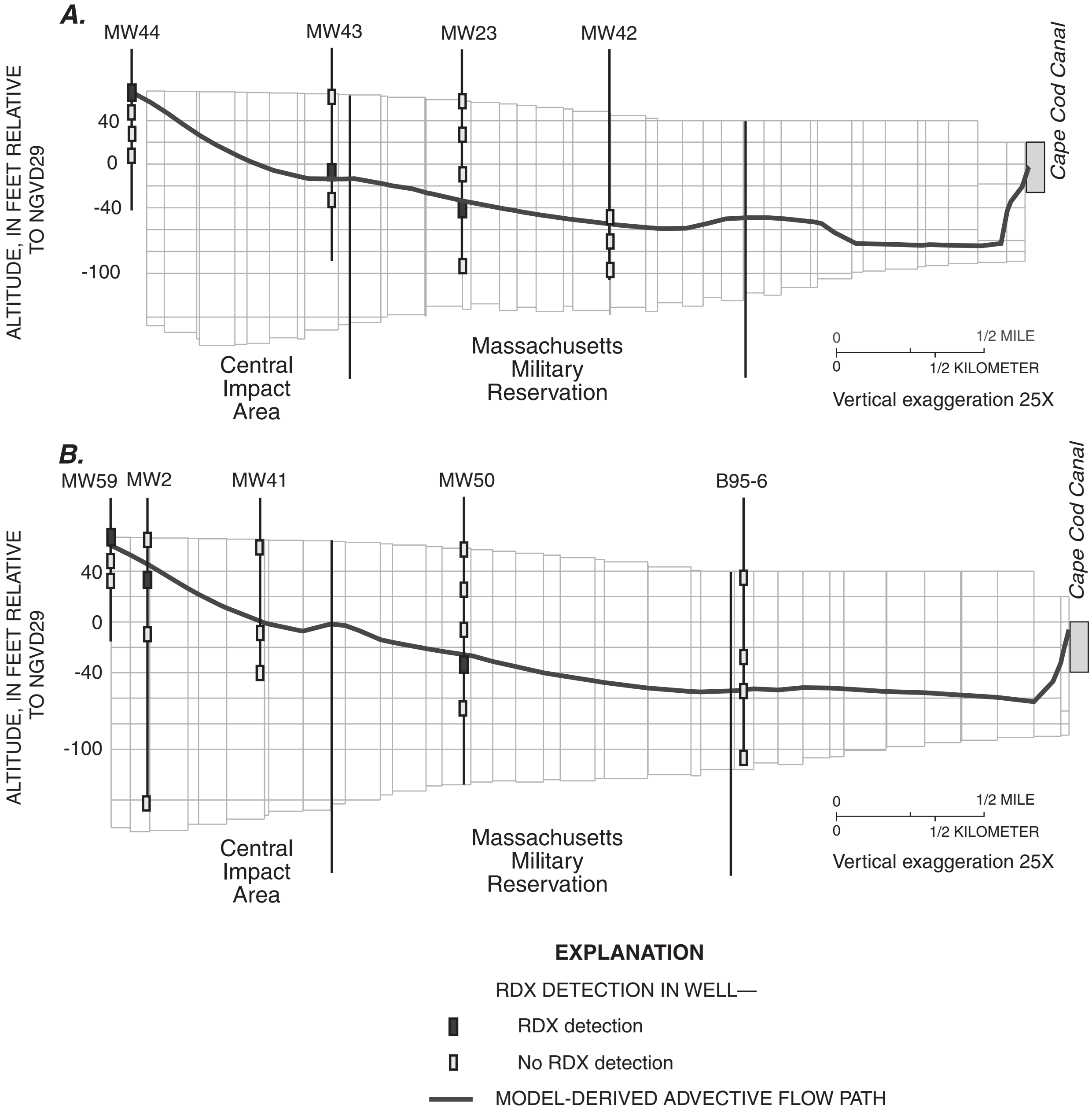


Figure 16. Advective flow paths calculated by the 2000 regional model from two monitoring wells within the Central Impact Area, (*A*) MW44, and (*B*) MW59, and the approximate positions of nearby wells contaminated with Royal Dutch Explosive (RDX), western Cape Cod, Massachusetts. The flow paths are estimates of the ultimate future discharge location of contaminants. Cross-section locations are shown on figure 5 and the monitoring wells on figure 17. Grids appear irregular because orthogonal grids are projected onto curved particle tracks.

Comparison of Particle Tracks From 1993, 1998, and 2000 Regional Models

A comparison of the results from the 1993, 1998, and 2000 regional models illustrates the general modeling uncertainties that arise from uncertainties in aquifer properties. All three models were calibrated to the same head and streamflow data from March 1993 and represent reasonable estimates of aquifer properties and recharge stresses. Although the general ground-water flow conditions are similar, the three calibrated models yield different predicted transport paths from some source areas in Camp Edwards (fig. 17).

The largest difference in predicted particle paths occurs in the northern J-Ranges Area. Under steady-state conditions, the predicted advective transport path for MW58 is to the southeast for the 1993 and 2000 regional models and to the northwest for the 1998 regional model. Although the 1998 and 2000 regional models use similar aquifer properties and recharge rates, and simulated heads, flows, and hydraulic gradients in most areas of the simulated area are similar, there is nearly a 180-degree difference in the predicted advective transport path starting in the northern J-Ranges Area. The northern J-Ranges Area is almost at the top of the water-table mound. Although the head solutions of the 1998 and 2000 regional models are similar, there is a difference of one model cell in the simulated position of the top of the water-table mound between the two models. This difference has little effect on particle tracks in downgradient locations, such as Demolition Area 1, where hydraulic gradients are more strongly influenced by the coastal boundary. However, the small shift in the position of the top of the water-table mound has a large effect on steady-state hydraulic gradient directions in the area around the top of the mound. The starting position for the northern J-Ranges particle track is northwest of the simulated position of the top of the water-table mound in the 1998 regional model and to the southeast of the simulated position in the 1993 and 2000 regional models.

The position of the top of the water-table mound has a strong control on advective transport paths. This control is stronger in areas near the top of the mound, where horizontal hydraulic gradients are nearly flat, than in areas closer to the coastal boundary, where horizontal hydraulic gradients are steeper. These results illustrate that relatively small changes in the model can alter the simulated position of the top of the water-table mound, and calibrated models with similar input parameters and simulated head solutions can produce different particle paths, particularly near the top of the water-table mound.

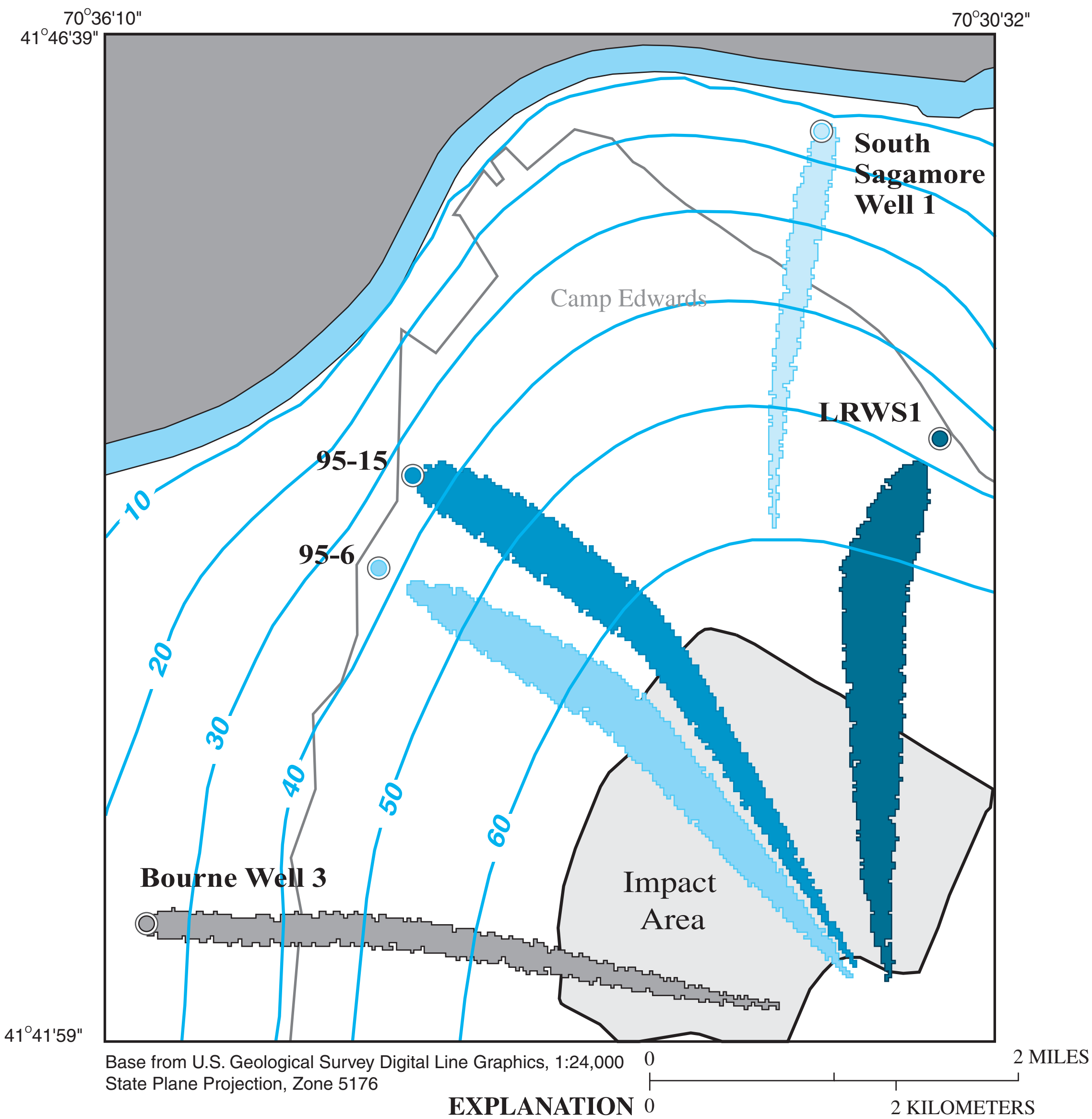
Areas Contributing Recharge to Municipal Wells

A parallel USGS investigation into the source of water to municipal wells and natural receptors on western Cape Cod (Masterson and Walter, 2000) was expanded to include proposed wells downgradient of Camp Edwards. The development of new water supplies near Camp Edwards was initiated in response to potentially adverse effects of contamination emanating from sites elsewhere on the MMR on the region's water supply. Particle tracking was used to determine the source of water to existing and proposed municipal wells near Camp Edwards (fig. 18). The delineated recharge areas to the proposed wells were used to assess the potential for contamination of the wells from source areas on Camp Edwards and to determine locations of observation wells used to monitor water quality upgradient of the wells. The proposed municipal wells were not installed, whereas wells at other locations not simulated here have been installed. The recharge areas are included here as examples of the types of modeling analyses that can be done in support of water development. A detailed discussion of the delineation of recharge areas to these wells is presented in Masterson and Walter (2000).

Figure 17. Model-derived advective flow paths from the water table at selected monitoring wells within Camp Edwards, western Cape Cod, Massachusetts. Paths were produced in simulations made with the 1993, 1998, and 2000 regional models.

Subregional Modeling

The steady-state regional model was useful in simulating advective transport, particularly in areas that were not near hydrologic boundaries, such as the Impact Area and Demolition Area 1, and in determining recharge areas to some municipal wells. Coarsely discretized regional models are not well suited, however, for simulating the recharge areas to wells when several wells are in operation in proximity to one another or for simulating advective transport near hydrologic boundaries such as ponds. In these cases, more finely discretized models are required for an accurate analysis of ground-water flow; such models are referred to in this report as subregional models. Two subregional models were developed as part of this investigation: a model of the area encompassing Camp Edwards and a model of the southern J-Ranges Area (fig. 6).



Base from U.S. Geological Survey Digital Line Graphics, 1:24,000
State Plane Projection, Zone 5176

EXPLANATION

SIMULATED AREAS AT THE WATER TABLE THAT RECHARGE WATER TO MUNICIPAL WELLS—

Existing wells
- Bourne Well 3
- South Sagamore Well 1

Proposed Wells
- 95-6
- 95-15
- LRWS1

MUNICIPAL WELL (EXISTING OR PROPOSED)

Existing wells
- Bourne Well 3
- South Sagamore Well 1

Proposed Wells
- 95-6
- 95-15
- LRWS1

Figure 18. Recharge areas to existing and proposed municipal wells downgradient of Camp Edwards, western Cape Cod, Massachusetts. Areas were delineated using the 1998 regional model and future (2020) pumping conditions.

Camp Edwards Model

The Camp Edwards subregional model is designed to better delineate the areas contributing recharge to municipal wells downgradient of Camp Edwards. The subregional model has a ninefold increase in horizontal discretization, which better represents the location of simulated wells, and a twofold increase in vertical discretization, which allows for better representation of the screened interval of the well within the simulated aquifer. One inaccuracy that can arise from using a coarsely discretized regional model to estimate recharge areas to wells occurs when the well of interest withdraws a volume of water that is small relative to the total flux of water through the model cell representing the well; such wells are known as weak sinks in the model. The size of the recharge area to a well multiplied by the recharge rate is equal to the pumping rate of the well. In a model, however, the product of the simulated recharge area and the simulated recharge rate is equal to the total flux of water through the model cell representing the well. As a result, the simulated recharge area to wells that are weak sinks in the model will be larger than the actual recharge area to the well. Wells near Camp Edwards that are weak sinks in the regional model are listed in Masterson and Walter (2000).

An example of a weak sink in the regional model is South Sagamore Well 1 (SS1), located northwest of the Impact Area. In the regional model, the specified withdrawal rate of 5,250 ft^3/d is about 75 percent of the total flux of water through the corresponding model cell. As a result, the recharge area simulated by the regional model (fig. 18) is about 33 percent larger than the recharge area calculated on the basis of the actual pumping rate. The use of a model with a smaller cell size decreases the size of the model cell representing the well and the total flux of water through the cell. This decreases the degree to which the well is a weak sink in the model and the degree to which the simulated recharge area is an overestimate of the size of the actual recharge area. Ideally, the size of the model cell should be small enough to eliminate the weak sink. Comparison of recharge areas to well SS1 indicates that the contributing area estimated by using the regional model is 36 percent larger than that delineated by using the subregional model (fig. 19). The specified withdrawal from well SS1 is 100 percent of the total flux of water through the subregional model cell; this result indicates that the recharge area delineated by using the subregional model better represents the actual size of the recharge area to the well and the weak-sink modeling problem has been eliminated.

Another inaccuracy that can arise when coarsely discretized regional models are used to delineate recharge areas to wells occurs when several wells are located in proximity to one another, relative to the size of the model cells. In these cases, coarsely discretized models will not adequately resolve recharge areas to individual wells. Examples are the simulated recharge areas to four municipal wells in the town of Bourne—Bourne Wells 1, 3, 4, and 6—to the west of Camp Edwards. Individual recharge areas simulated by the regional model (fig. 20*A*) are difficult to resolve and one of the simulated recharge areas splits. Recharge areas determined by the subregional model (fig 20*B*) indicate a distinct recharge area to each well. Both the locations and shapes of the recharge areas differ substantially between the regional and subregional models. It should be noted that Bourne Well 6 is a weak sink in the regional model. The differences in recharge areas for the well shown in fig. 20*A* and *B* are due to improved resolution in the subregional model as it relates to both well placement and representation of the well as a strong sink.

Southern J-Ranges Model

The model of the southern J-Ranges Area simulates the advective transport of contaminants in the area around Snake Pond and the treatment system installed to remediate the plume that emanates from the nearby FS-12 source area (fig. 8). The pond and the extraction and injection wells that constitute the remediation system are hydrologic boundaries that make local ground-water-flow patterns complex. The use of a regional model to simulate local head contours and advective transport paths likely would result in inaccuracies because the regional model is too coarsely discretized to adequately represent these local hydrologic features. The simulated head distribution and the predicted paths of advective transport of contaminants from known detections in the aquifer could vary substantially as a function of model discretization.

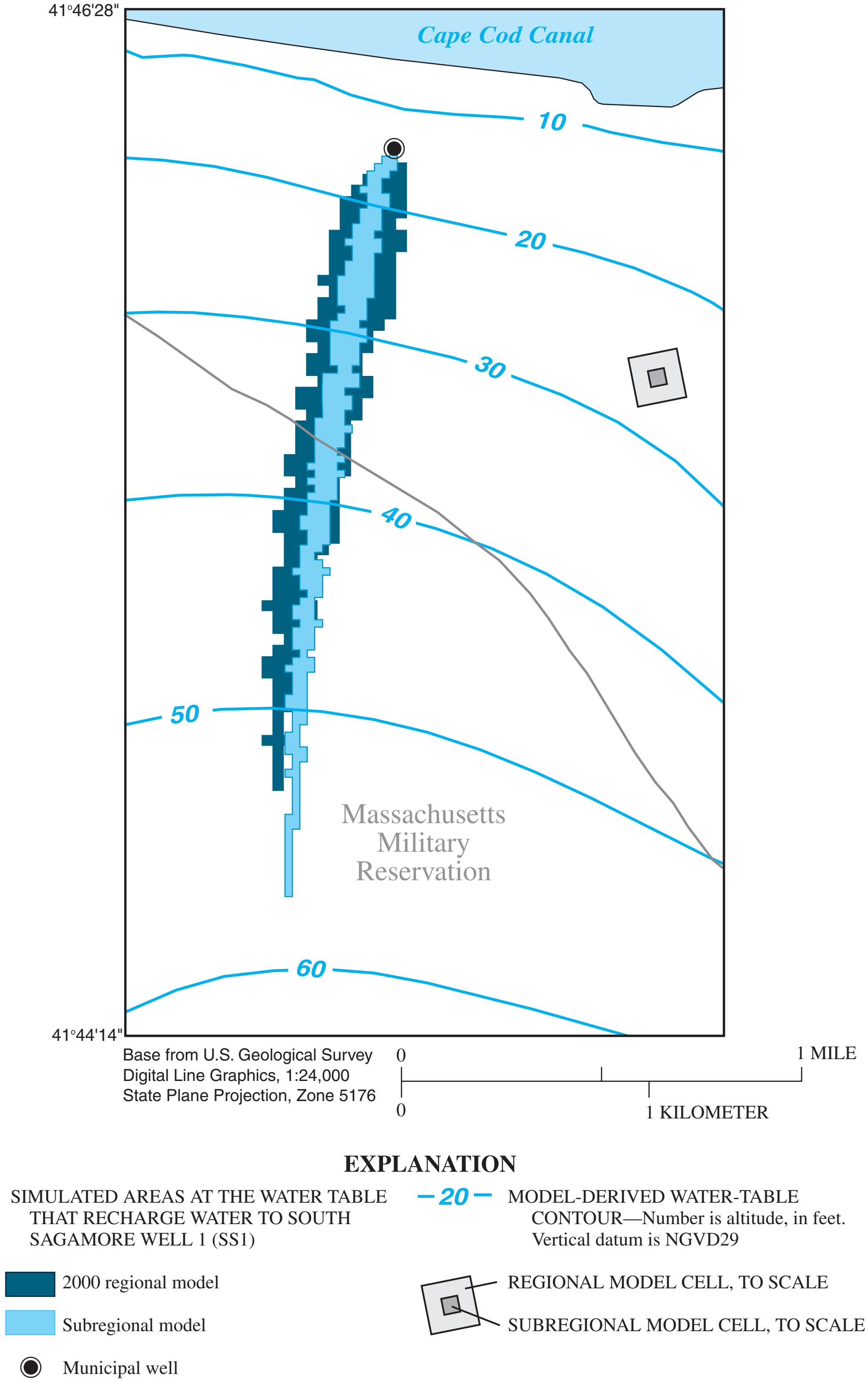


Figure 19. Areas at the water table across which water recharges the aquifer and discharges to South Sagamore Well 1, western Cape Cod, Massachusetts. Areas were simulated by the regional and subregional models.

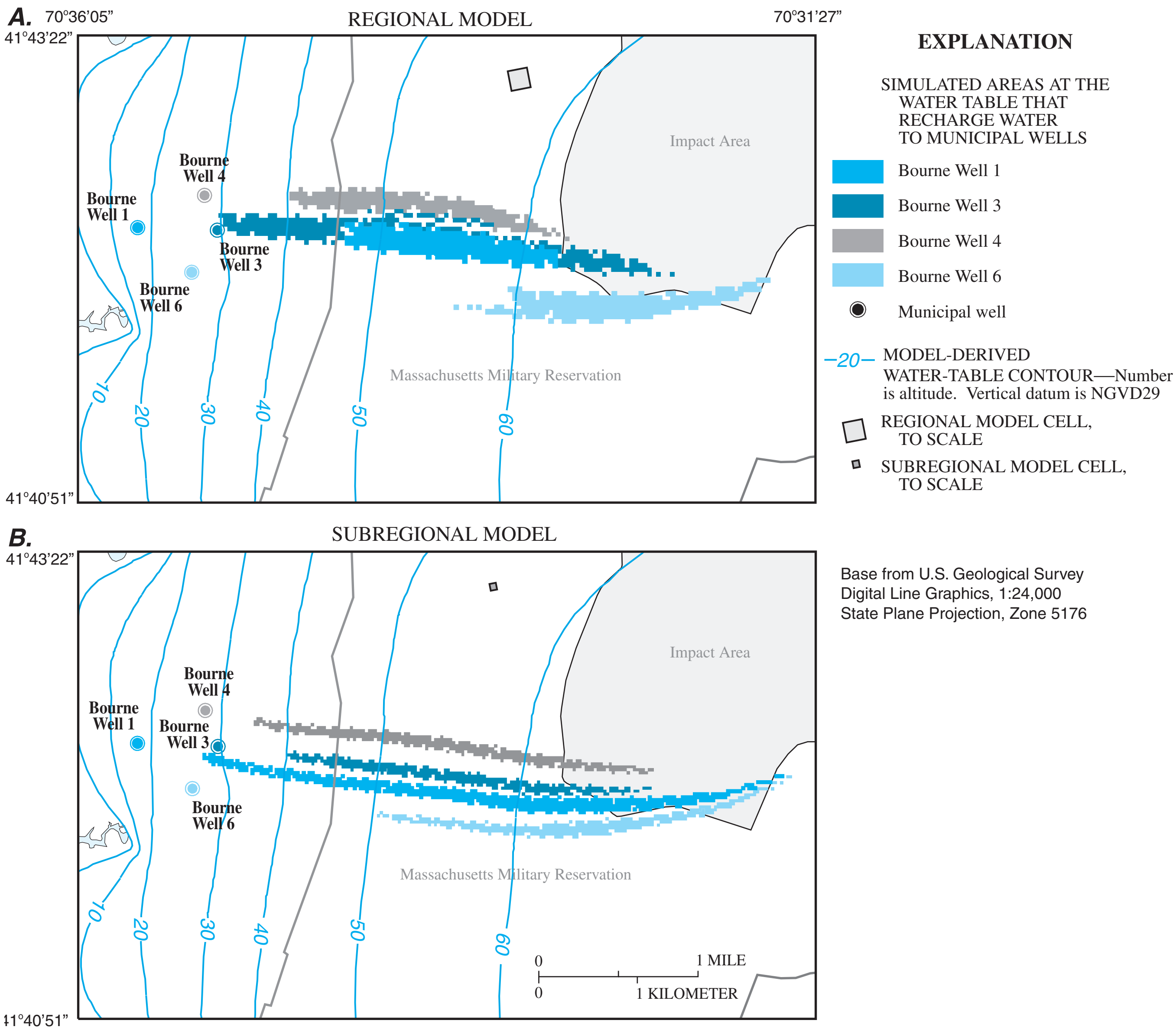


Figure 20. The effect of model discretization on simulated contributing areas to four municipal wells in close proximity to one another, by comparison of(*A*) regional and (*B*) subregional model results, western Cape Cod, Massachusetts. Vertical discretization shown in figure 7.

The subregional model and a version of the 1998 regional model that included the stresses associated with the remediation system were used to simulate hydraulic heads and contaminant transport paths from three known detections within the aquifer underlying the southern J-Ranges Area. Extraction and reinjection rates for the FS-12 plume-containment system were obtained from Jacobs Engineering, Inc., (Michael Goydas, written commun., 2001). Simulated hydraulic heads and ground-water-flow patterns differ considerably between the two models near Snake Pond and the remediation system; the subregional model represents head contours and gradients in the area with more detail than does the regional model (fig. 21). Simulation results also indicate that the predicted transport paths differ between the two models (fig. 21). In the case of contamination emanating from the area around well WT13, the regional model predicts that contaminants would not be captured by the upgradient portion of the remediation system, but would pass below the downgradient portion of the system and continue to be transported to the east of Snake Pond.

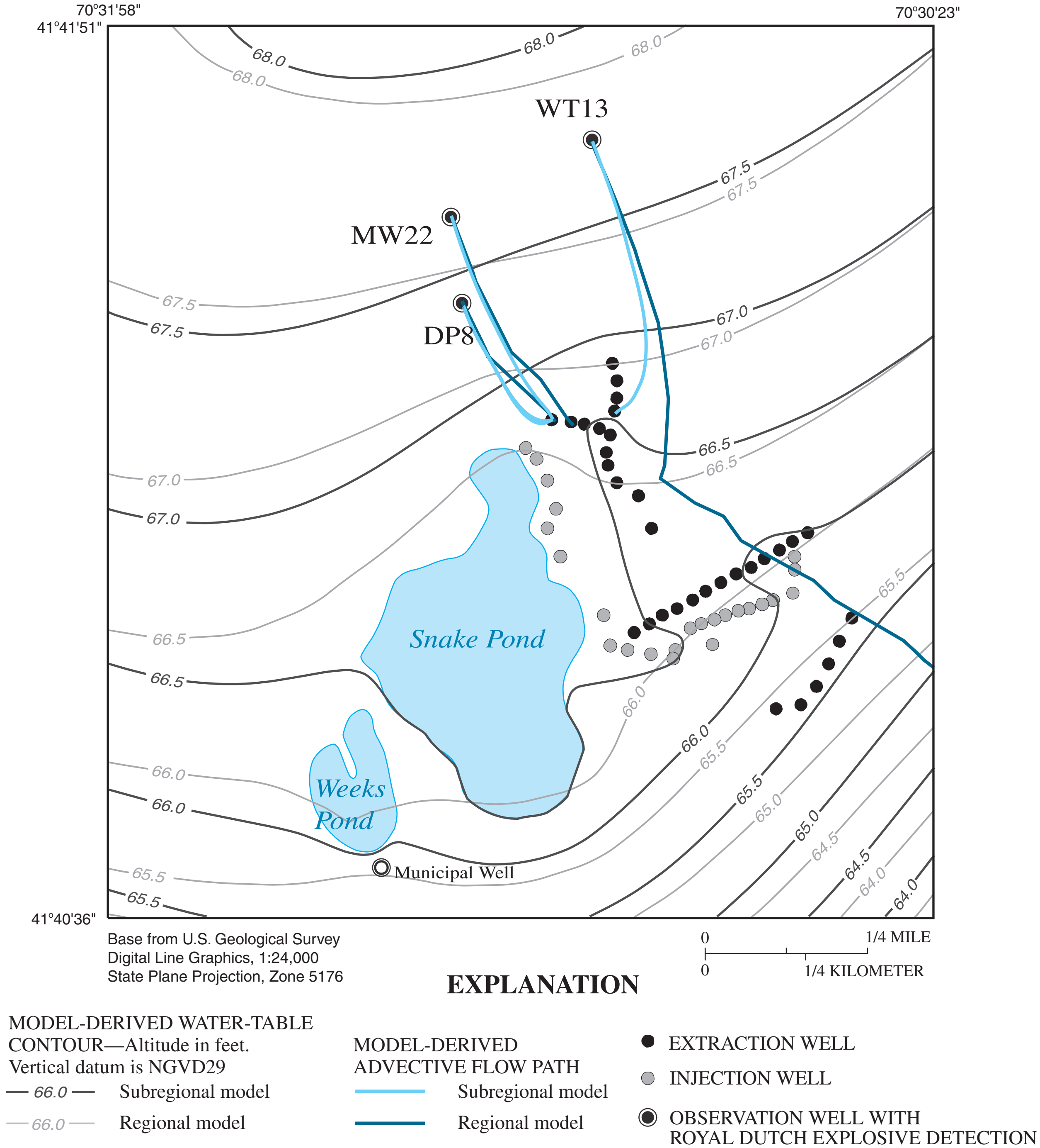


Figure 21. Advective flow paths from selected monitoring wells in the southern J-Ranges Area and simulated water-table contours calculated by the regional and subregional models, western Cape Cod, Massachusetts.

The subregional model, which gives a better representation of the extraction and injection wells, indicates that the contamination from the WT13 area would be captured by the remediation system. This example shows that conclusions based on simulation results can differ depending on model discretization, and that a regional model is not an appropriate tool for analyzing ground-water flow in areas with hydrologic features or boundaries that are small relative to model cell size.

TRANSIENT SIMULATIONS

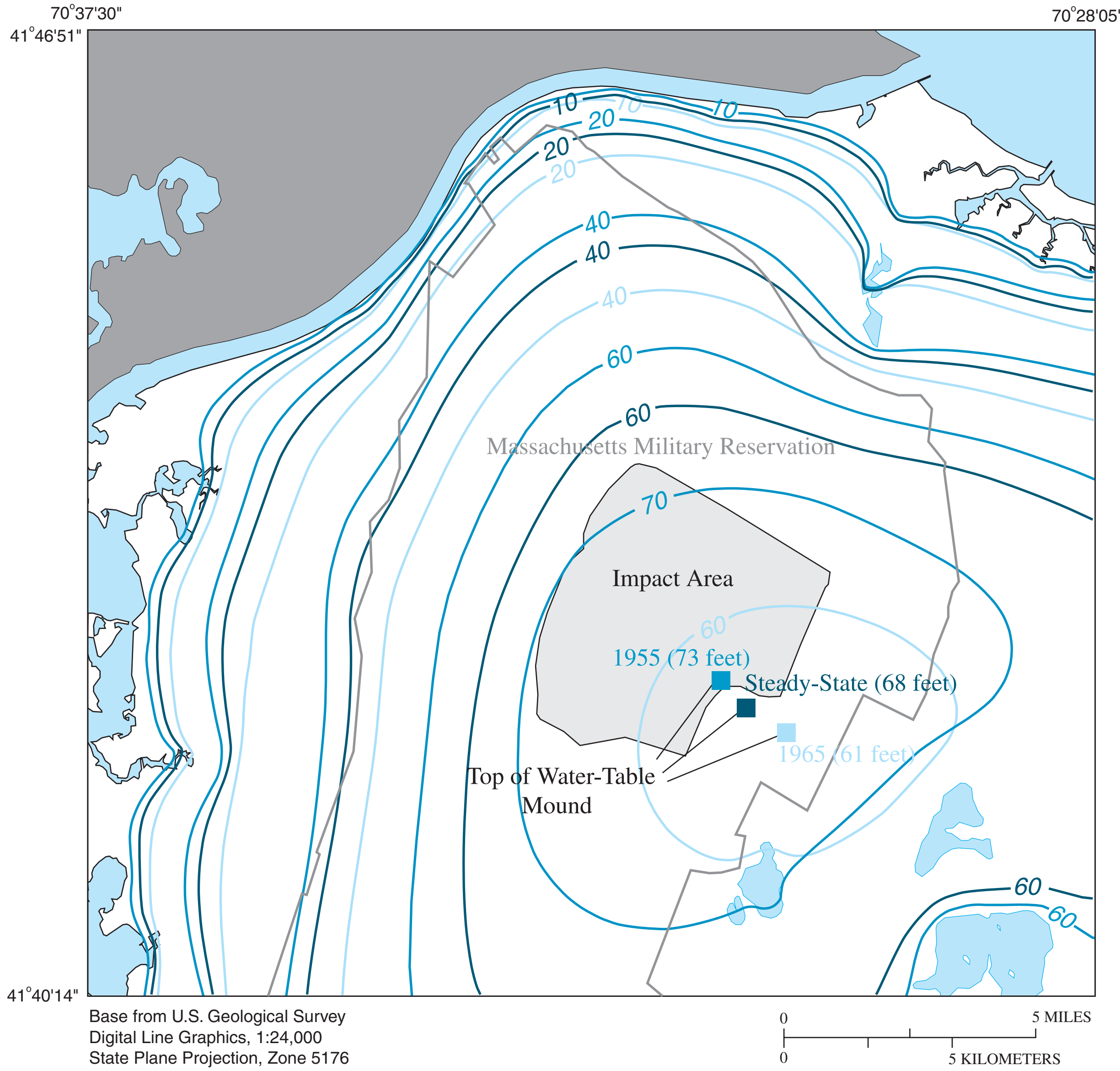
Transient versions of the regional model that incorporated seasonal and long-term variations in recharge were used to evaluate the effects of changing recharge stresses on ground-water flow in the aquifer. Results of simulations made with the transient models were used to determine if the steady-state models are valid tools for predicting advective transport paths for contaminants at Camp Edwards and if the effects of changing recharge stresses on ground-water-flow patterns were spatially variable.

Effects of Transient Recharge on Heads and Gradients

A version of the model that incorporated seasonal changes in recharge and pumping can be used to determine if these changing stresses affect ground-water-flow patterns and advective transport paths in the aquifer. Simulation results show that between in-season (mid-May to mid-October), when recharge is lowest and pumping is highest, and off-season (mid-October to mid-May), when recharge is highest and pumping is lowest, the altitude of the top of the water-table mound changes from 67 to 72 ft. The simulated top of the water-table mound does not change position relative to the regional model grid, however, and simulated hydraulic gradients in the aquifer do not change substantially. This suggests that seasonal changes in recharge and pumping regionally do not affect gradient directions and transport paths in the aquifer, particularly in areas not near ponds, such as the Camp Edwards Impact Area. This likely is due to the short time scale of the changes in stress conditions as compared to the time scale of water movement in the aquifer; with recharge over a longer time period, sufficient time is available for the shape of the water table to change. McCobb and others (1999) found that hydraulic-gradient directions did change seasonally—by about 12 degrees—upgradient of but near Ashumet Pond, which is to the south of the MMR.

A version of the regional model that incorporated the long-term changes in recharge observed on western Cape Cod can be used to determine if these changes, which are larger in magnitude and longer in duration than seasonal changes, affect ground-water flow in the aquifer. Comparisons are made between simulated hydraulic conditions in 1955, during a period when water levels were higher than average, and simulated conditions in 1965, which was during a historical drought (fig. 10). The data show that the simulated head at the top of the water-table mound changed from an elevation of 73 ft in 1955 to 61 ft in 1965; the simulated head at the top of the water-table mound under steady-state conditions was about 68 ft (fig. 22). In addition, the simulated 1955 and 1965 positions of the top of the water-table mound were about 1,500 ft apart. The model-calculated position of the water-table mound under steady-state conditions was about equidistant along a northwest-southeast trending line between the 1955 and 1965 water-table mound locations (fig. 22). This trend likely is due to the complex hydrologic boundaries of the flow system. To the south, streams extend to near the top of the mound, whereas there are no major streams to the north and west and most discharge is at the coast, at a greater distance from the mound. It should be noted that the conditions represented in figure 22 represent extremes in terms of water levels. Given the recharge record and historical water levels in the area, locations of the water-table mound probably cluster through time around the location predicted by the steady-state model and may be located near the 1955 and 1965 locations at only a few points in time, presumably during periods of extreme high or low water levels.

The changes in simulated heads in the aquifer between hydraulic conditions in 1955 and 1965 range from 0 to 12 ft and are greatest near the top of the water-table mound (fig. 23*A*). Changes in hydraulic head decrease near hydrologic boundaries where heads are constrained, such as the coastline. Comparison of simulated heads for 1955 and 1965 conditions also shows that hydraulic gradients in the aquifer change substantially in response to changes in recharge. The magnitudes of hydraulic gradients are about 0.01 to 0.3 ft/ft. The magnitudes change from 0 to 0.13 ft/ft in response to the time-varying recharge (fig. 23*B*). The changes in hydraulic-gradient magnitudes are greatest near coastal, stream, and pond boundaries. Heads at discharge boundaries, such as streams and coastal embayments, do not substantially change with time as compared to heads in the surrounding aquifer. Similarly, pond levels do not change as much as water levels in the surrounding aquifer.



EXPLANATION

MODEL-DERIVED WATER-TABLE CONTOURS—Altitude in feet.

– 20 – Steady-state 2000 regional model

– 20 – Long-term transient regional model for end of 1955

– 20 – Long-term transient regional model for end of 1965

POSITION OF THE TOP OF THE WATER-TABLE MOUND AND HEAD ALTITUDE—In feet.

Steady-state model

Transient model, end of 1955

Transient model, end of 1965

Figure 22. Water-table contours and the position of the top of the water-table mound for steady-state, high-recharge (1955), and low-recharge (1965) conditions, western Cape Cod, Massachusetts. Altitudes were calculated by the steady-state and transient regional models with respect to NGVD29.

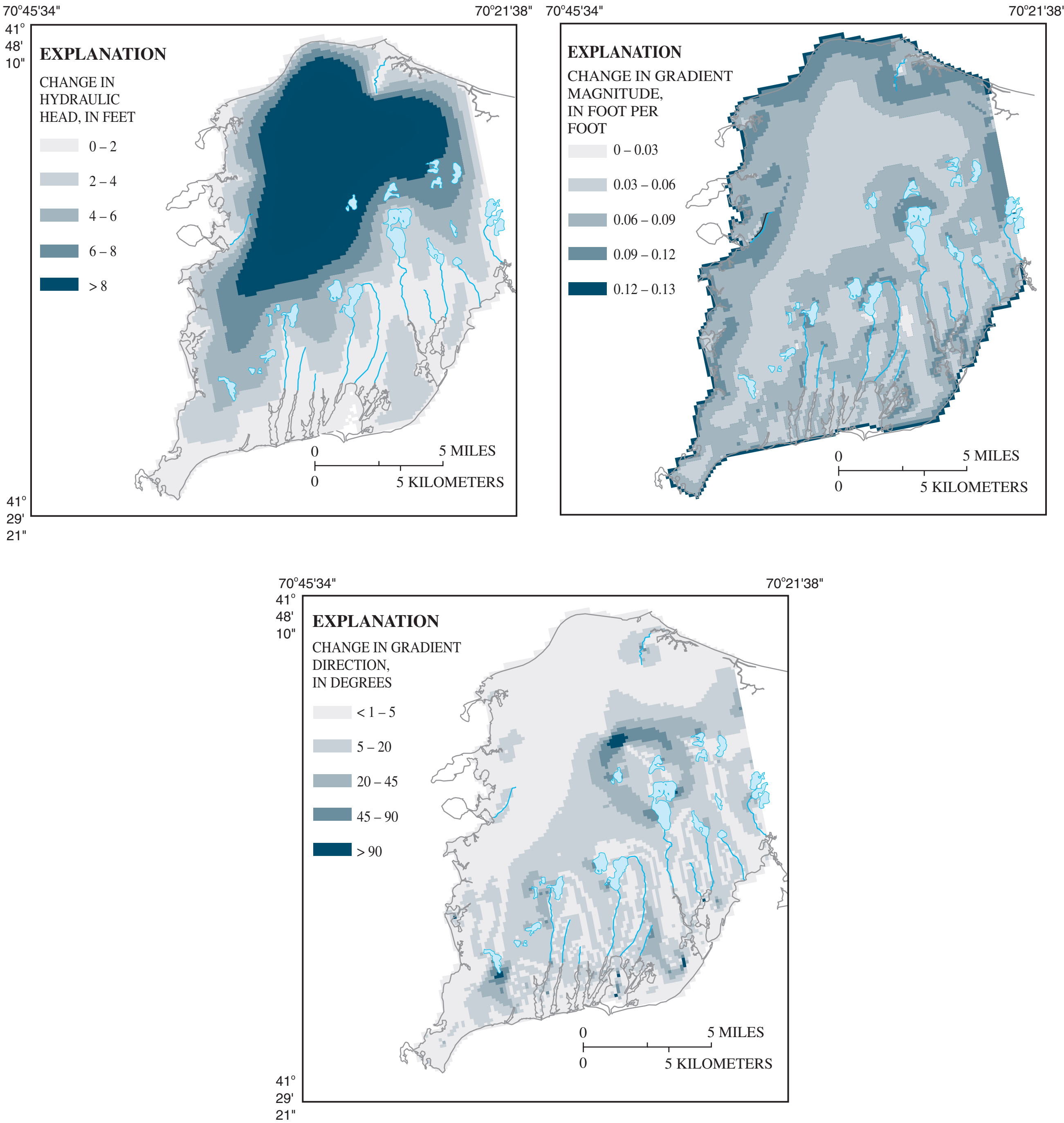


Figure 23. Simulated differences in hydraulic heads and in magnitude and direction of the hydraulic gradient between high-recharge (1955) and low-recharge (1965) conditions on western Cape Cod, Massachusetts.

As a result, the magnitudes of changes in hydraulic gradients upgradient of ponds, streams, and the coast are greater than in other areas. The change in hydraulic-gradient magnitude is smallest near the top of the water-table mound.

Hydraulic-gradient directions in the aquifer also change in response to changes in recharge rates. Between high-recharge (1955) and low-recharge (1965) conditions, changes in hydraulic-gradient direction ranged from 0 to 180° (fig. 23*C*). The greatest change in hydraulic-gradient direction occurs near the top of the water-table mound where simulated hydraulic gradients reverse direction. Large changes in hydraulic-gradient direction also occur near ponds, particularly near the boundaries between areas of pond inflow and pond outflow. Changes in hydraulic-gradient directions are small near discharge boundaries such as streams and coastal embayments. In general, there is an inverse relation between changes in hydraulic-gradient direction and magnitude (figs. 23*C* and 24); areas that show large changes in hydraulic-gradient direction between high- and low-recharge conditions are areas where corresponding changes in hydraulic-gradient magnitude are small. This inverse relationship is shown in figure 24.

Changes in hydraulic-gradient directions between high-recharge (1955) and low-recharge (1965) conditions in the northern part of Camp Edwards, including the Impact Area, Demolition Area 1, and the J-Ranges, are shown in more detail in figure 25. The effects of changing recharge rates on the hydraulic gradients in the aquifer vary spatially. The changes in hydraulic-gradient direction in the Impact Area mostly are less than 5 degrees. Hydraulic gradients in Demolition Area 1 change direction by about 13 degrees. In the J-Ranges Area, changes in hydraulic-gradient directions range from 30 to 45 degrees in the southern J-Ranges Area and exceed 135 degrees in the northern J-Ranges Area.

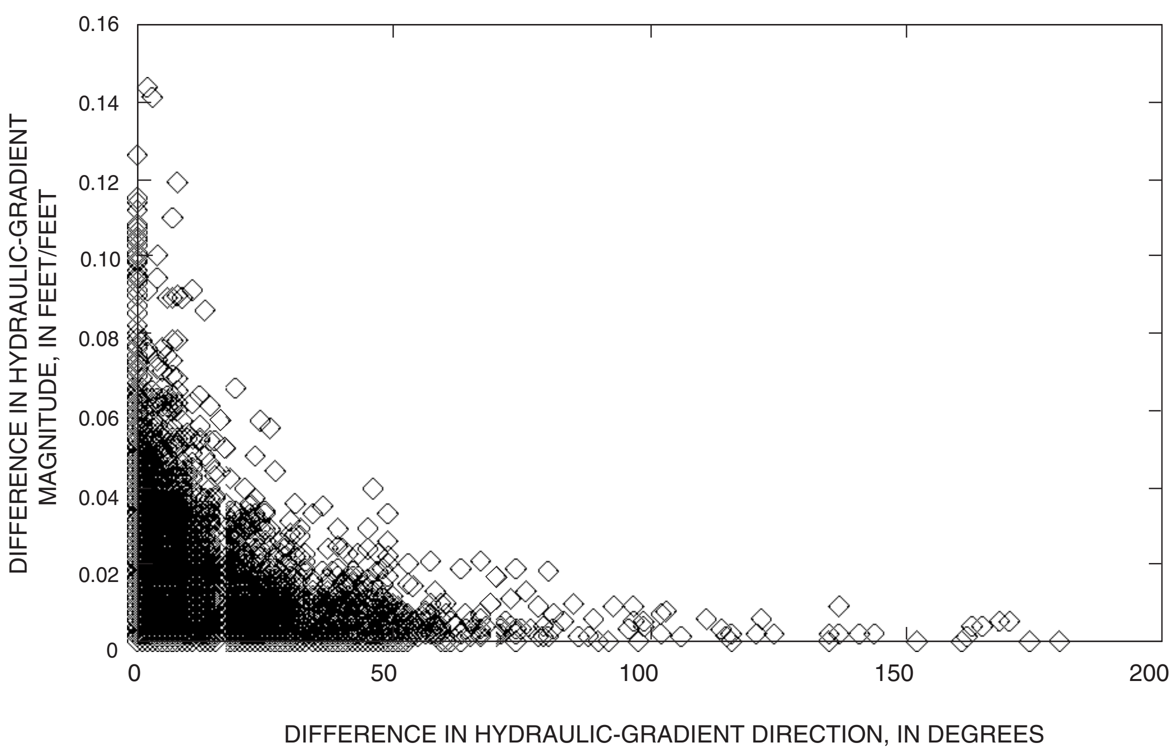


Figure 24. Relation between simulated differences in hydraulic-gradient directions and magnitudes for all active model cells for high-recharge (1955) and low-recharge (1965) conditions on western Cape Cod, Massachusetts.

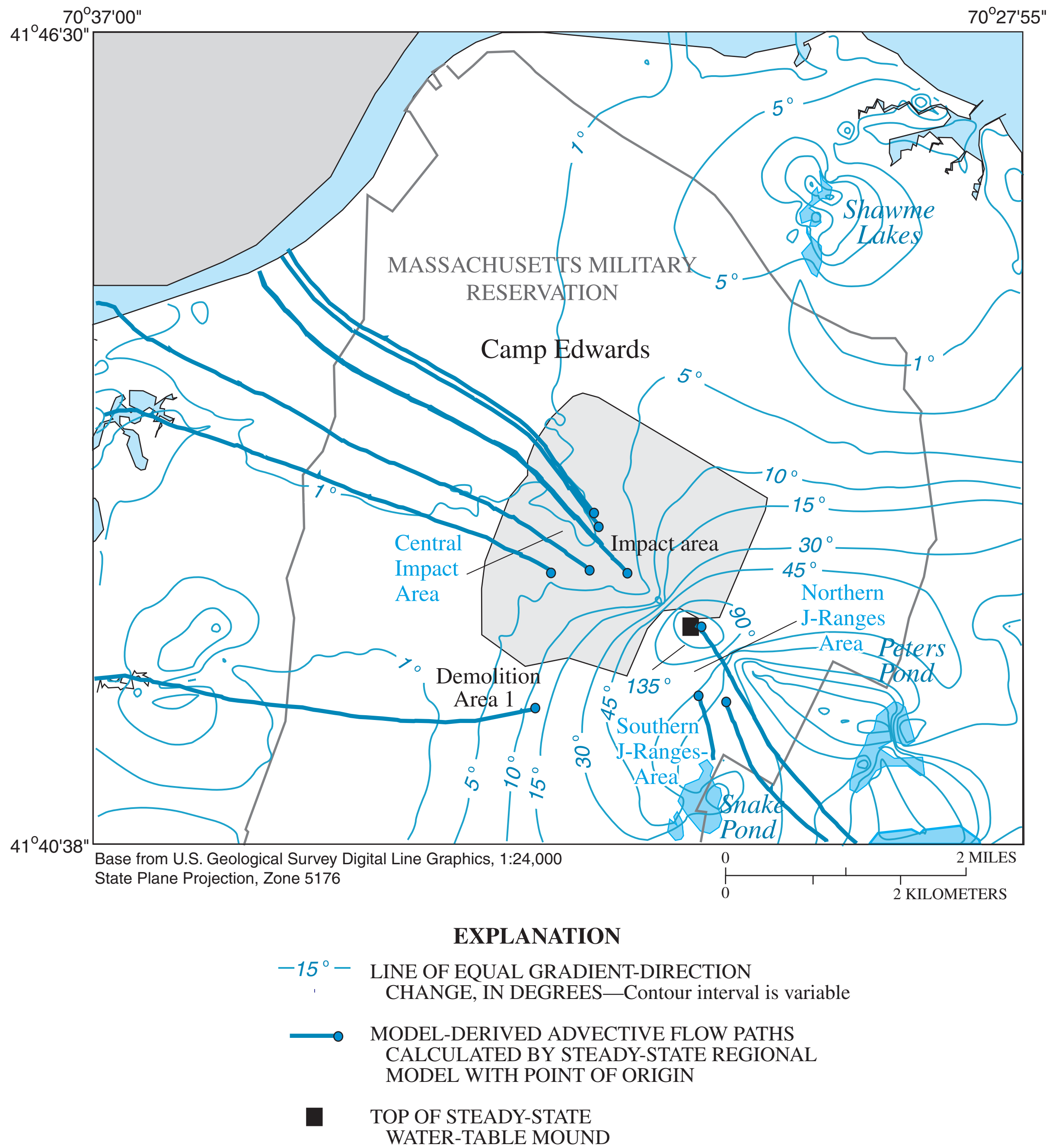


Figure 25. Changes in gradient direction between high-recharge (1955) and low-recharge (1965) conditions for the northern part of the western Cape Cod, Massachusetts, flow cell and steady-state advective flow paths from selected locations.

Effects of Long-Term Transient Recharge on Advective Transport

The transient regional models can be used to evaluate how changing stresses affect predicted advective flow paths in the aquifer. Seasonal changes in recharge stresses likely would not change predicted advective transport paths. This assumption is based on the observation that although head elevations in the aquifer change, the position of the top of the water-table mound and general hydraulic gradients in the aquifer do not change seasonally. Seasonal stress changes likely would have no effect on advective transport because the time scale of the seasonal stress changes, which is on the order of months, is small compared to the time scale of advective transport, which is on the order of decades.

The precipitation record from western Cape Cod yielded large, long-term variability in estimated recharge rates in the area (fig. 10). Long-term changes in recharge stresses likely would affect advective transport paths in some parts of the aquifer by changing hydraulic-gradient directions. The effect of long-term changes in recharge on advective flow paths likely is a function of proximity to hydrologic boundaries and to the top of the water-table mound. The long-term transient model estimates the degree to which changes in recharge affect advective flow paths in the aquifer and how predicted flow paths compare to those predicted by the steady-state model. Particles were started at the water table beneath the center of Demolition Area 1, three representative sites in the Impact Area, and three sites within the J-Ranges Area at four different times: 1955, 1965, 1975, and 1985. These times represent a general distribution of hydraulic conditions and span a period in which much of the observed ground-water contaminants were released into the environment at Camp Edwards. Particles also were tracked from the same locations under steady-state conditions. These particles were tracked forward through the modeled flow field until 1996.

Demolition Area 1 and Impact Area

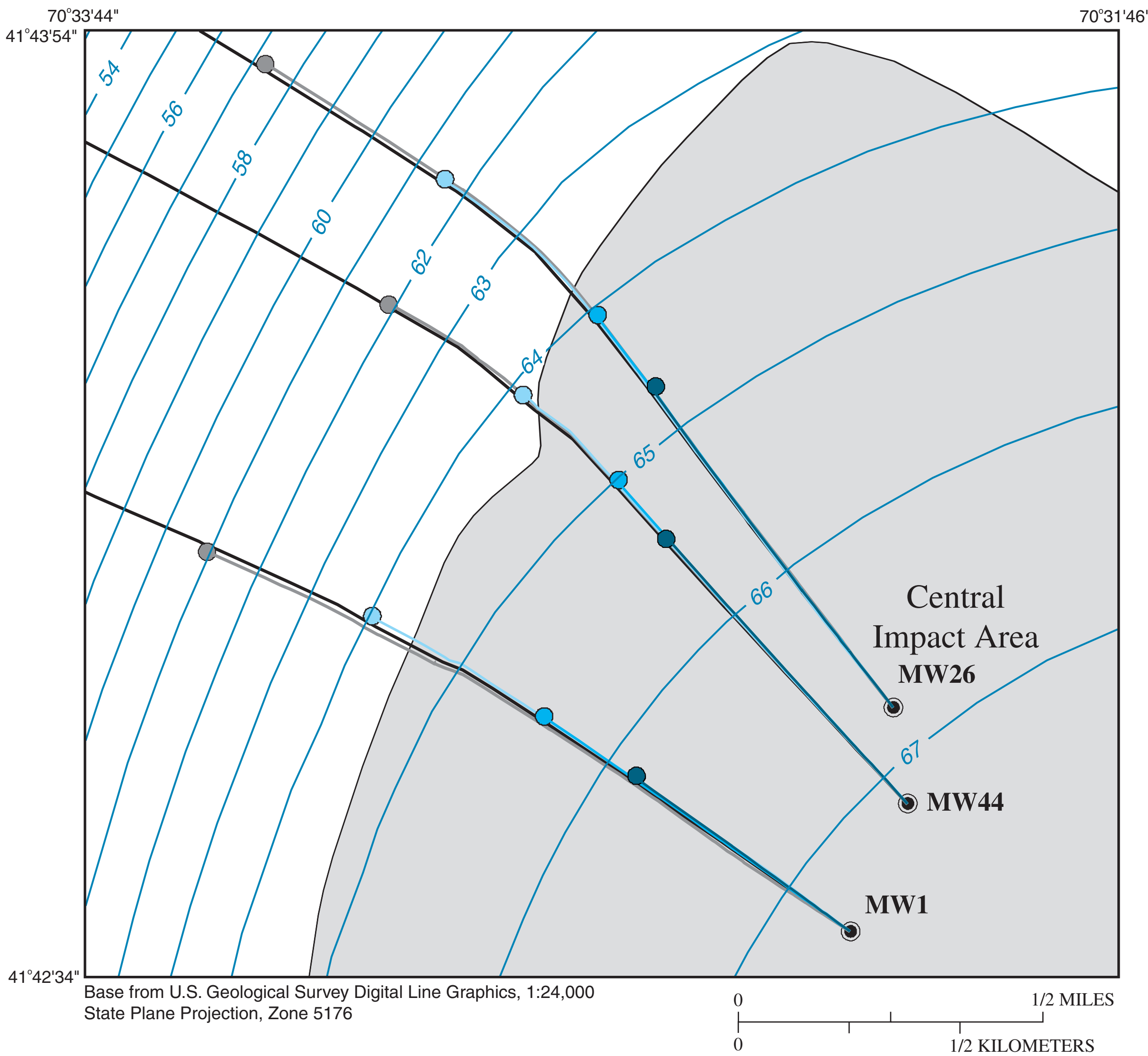
The predicted advective transport paths of contaminants starting from the water table in the Impact Area and Demolition Area 1 1955, 1965, 1975, and 1985 and stopping in 1996 are shown in figures 26 and 27, respectively. The transport distances to current (1996) locations of particles vary with recharge locations within the Impact Area. As of 1996, maximum horizontal transport distances for particles started in 1955, 1965, 1975, and 1985 were 7,300, 6,400, 4,700, and 3,900 ft, respectively. As of 1996, particles started in 1955, 1965, 1975, and 1985 at the water table beneath Demolition Area 1 had traveled 7,800, 6,000, 3,300, and 2,700 ft downgradient of the source area, respectively.

Particles started in 1955, 1965, 1975, and 1985 followed similar transport paths in the aquifer from the Impact Area and Demolition Area 1 (figs. 26 and 27). Over a transport distance of 2,000 ft, particle tracks started in 1955, 1965, 1975, and 1985 are separated by a total of about 180 ft in the Demolition Area 1 and by about 50 ft in the Impact Area. This is consistent with model results showing that, although simulated head values (fig. 23*A*) and hydraulic-gradient magnitudes (fig. 23*B*) in these areas change, hydraulic-gradient directions in the Impact Area and Demolition Area 1 do not change substantially (fig. 23*C*) with changing recharge stresses. Between 1955 and 1965, simulated hydraulic-gradient directions changed by about 13 degrees and less than 5 degrees in Demolition Area 1 and the Impact Area, respectively (fig. 25). Demolition Area 1 and the Impact Area are located about 7,700 and 5,700 ft, respectively, from the steady-state position of the top of the water table mound and about 17,300 and 20,500 ft, respectively, from the coastal boundary.

J-Ranges Area

The predicted advective transport paths of contaminants from the J-Ranges Area for 1955, 1965, 1975, and 1985 are shown in figure 28. As of 1996, particles started in 1955, 1965, 1975, and 1985 from WT13 in the southern J-Ranges Area that had not discharged to Snake Pond had traveled 7,000, 3,000, 2,400, and 1,800 ft downgradient of the source area, respectively. These transient particle tracks generally follow similar paths for all four starting years. Over a transport distance of about 2,000 ft, particle tracks from WT13 are separated by a distance of about 190 ft.

Particles started in 1955, 1965, 1975, and 1985 in the northern J-Ranges Area, which is located at or near the top of the water-table mound, had traveled only about 960, 190, 150, and 70 feet downgradient, respectively, as of 1996 (fig. 28). In this analysis, particles were started at the water table at a location coincident with the model-calculated top of the water-table mound under steady-state conditions. The simulated transport patterns are consistent with model results indicating a strong component of vertical flow and small horizontal gradients near the top of the water-table mound (figs. 12 and 13). The difference in predicted transport particle paths for the different starting dates is much greater in the northern J-Ranges Area. The predicted advective transport path is a function of the time when the particles were started at the water table; particles started at different times do not follow the same path in the aquifer. A parcel of water entering the aquifer in 1955 is transported to the southeast whereas a particle started in 1965 initially tracks to the northwest and then reverses direction and tracks to the southeast (fig. 28).



EXPLANATION

MODEL-DERIVED ADVECTIVE FLOW PATHS

- Particles started in 1985
- Particles started in 1975
- Particles started in 1965
- Particles started in 1955
- Steady-state

— 65 — MODEL-DERIVED WATER-TABLE CONTOUR—Altitude in feet. Vertical datum is NGVD29

CURRENT (1996) LOCATIONS OF PARTICLES

- Particles started in 1985
- Particles started in 1975
- Particles started in 1965
- Particles started in 1955

MW26 ◉ WELL WITH ROYAL DUTCH EXPLOSIVE DETECTION AND IDENTIFIER

Figure 26. Steady-state and transient advective flow paths of particles started in 1955, 1965, 1975, and 1985 from three locations in the Central Impact Area, western Cape Cod, Massachusetts. Paths are projected to map view. Transient flow paths end at predicted 1996 locations.

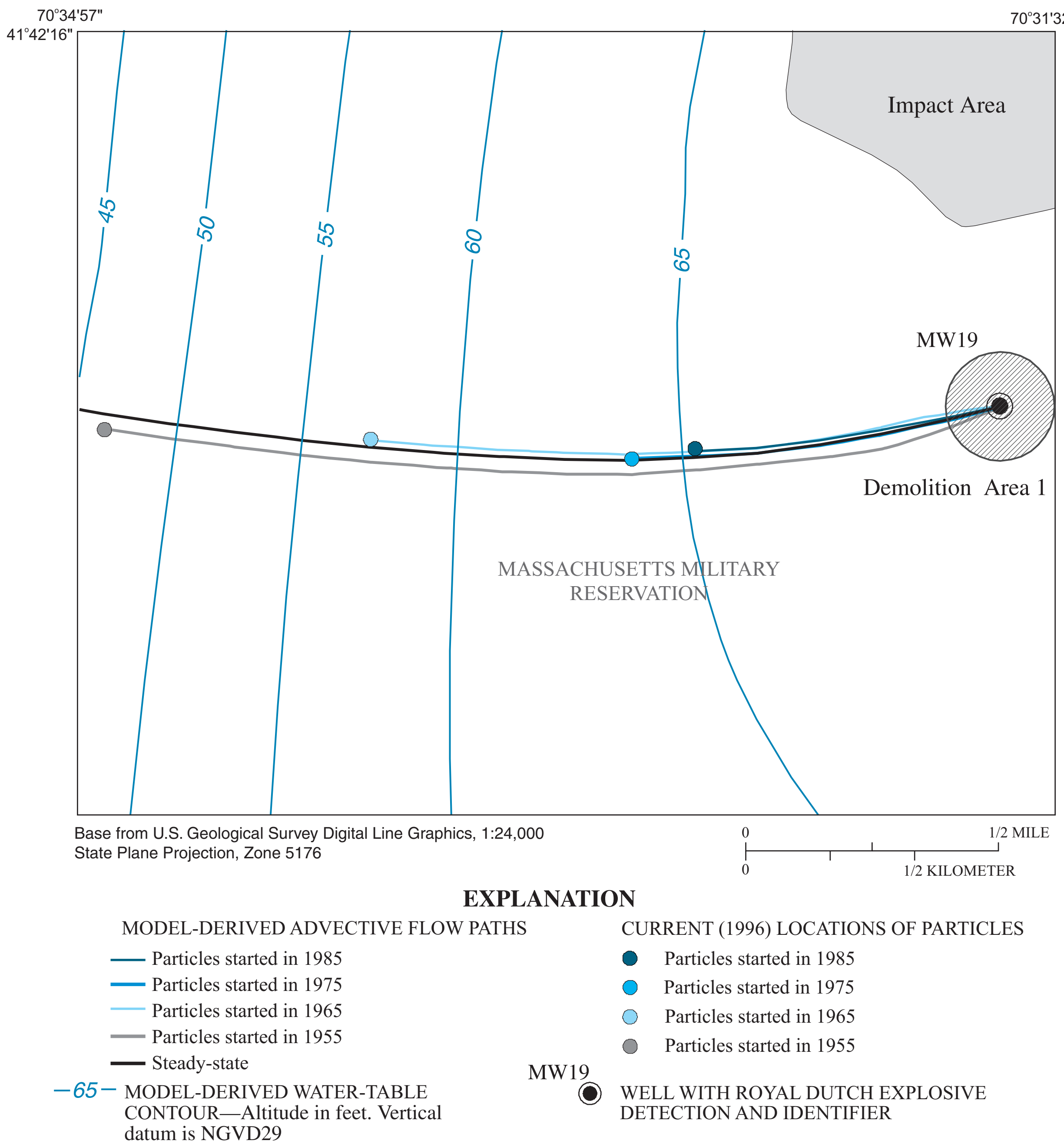


Figure 27. Steady-state and transient advective flow paths of particles started in 1955, 1965, 1975, and 1985 from Demolition Area 1, western Cape Cod, Massachusetts. Paths are projected to map view. Transient flow paths end at predicted 1996 locations.

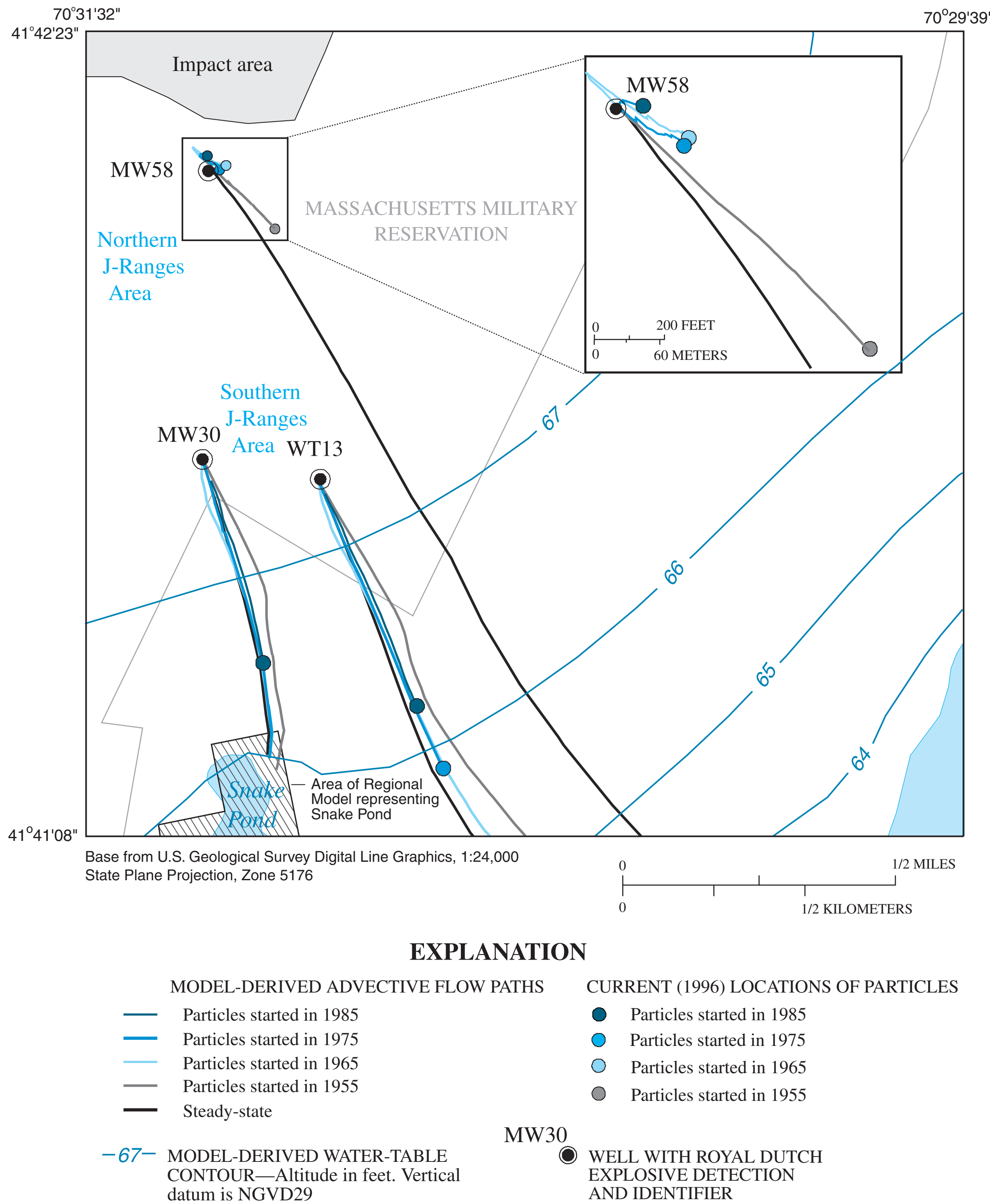


Figure 28. Steady-state and transient advective flow paths of particles started in 1955, 1965, 1975, and 1985 from three locations in the J-Ranges Area, western Cape Cod, Massachusetts. Paths are projected to map view. Transient flow paths end at predicted 1996 locations.

These tracks are consistent with model results showing that the position of the top of the water-table mound and hydraulic-gradient directions in the northern J-Ranges Area change substantially in response to long-term changes in recharge rates (figs. 22 and 25). The temporal variability seen in particle tracks in the northern J-Ranges Area is a result of changes in hydraulic gradients near the top of the water-table mound. For example, the top of the water-table mound shifted from a position northwest to a position southeast of the northern J-Ranges Area in response to a period of low recharge in the mid-1960s; gradients and flow directions in the area were to the northwest during this time. After recharge rates increased, the top of the water-table mound shifted back to the northwest and flow directions were to the southeast, which is the general flow direction under average hydraulic conditions. As a result, the particle track which was started at the water table in 1965 reversed direction. The southern J-Ranges Area is farther from the top of the water-table mound and closer to the hydrologic boundary at Snake Pond; hydraulic gradients in that area are less variable with time than in the northern J-Ranges Area. Between 1955 and 1965, simulated changes in hydraulic-gradient directions ranged from about 30 to 45 degrees in the southern J-Ranges area and exceeded 135 degrees in the northern J-Ranges Area (fig. 25).

Implications for the Use of Transient and Steady-State Models

Transient particle tracks from the Impact Area and Demolition Area 1 are in close agreement with steady-state particle tracks from the same locations (figs. 26 and 27). The assumption of a steady-state recharge stress for the aquifer is valid for particle tracking in these areas, and the steady-state regional model can be used to simulate advective transport. Likewise, transient particle tracks in the southern J-Ranges area are in general agreement with steady-state particle tracks (fig. 28); this agreement indicates that a steady-state regional model can predict general ground-water-flow directions in the aquifer with some degree of uncertainty arising from the assumption of steady-state conditions. Snake Pond locally affects ground-water-flow patterns in that area, and a coarsely discretized regional model may not be appropriate for predicting advective transport whether steady or transient conditions are assumed. This is consistent with model results showing that hydraulic-gradient directions in the southern J-Ranges Area do not change substantially with changing recharge stresses. Demolition Area 1, the Impact Area, and the southern J-Ranges area are located about 7,700, 5,700, and 3,100 ft, respectively, from the steady-state position of the top of the water-table mound. Although simulated head values and hydraulic-gradient magnitudes in these areas change, the general hydraulic-gradient directions do not change substantially in response to long-term changes in recharge.

Transient particle tracks started at different times in the northern J-Ranges Area differ substantially from the steady-state particle track from the same location (fig. 28); this difference indicates that the assumption of a steady-state recharge condition is not valid for particle tracking in that area and that the use of a the steady-state regional model could yield inaccurate results. Also, the lack of agreement between particle tracks started at different times indicates that the use of a transient model to predict advective transport accurately would need an accurate estimate of recharge rates over time. This conclusion is consistent with model results indicating that hydraulic-gradient directions near the top of the water-table mound change in response to temporal changes in recharge rates. As a result, the use of a transient model likely would have uncertainties associated with simulated stresses.

The reason that the steady-state particle track started in the northern J-Range is not bracketed by transient particle tracks started at the same location is not known. One factor may be the large sensitivity of the direction of the steady-state particle to the starting location. The particle was started in the center of the model cell representing the top of the water-table mound, which is the center of the radial flow system, and very small shifts in the starting location would cause the particle to move in very different directions. This could make steady-state particle tracks from this location suspect. Another possible factor could be that particle tracks started in the transient flow field are ultimately controlled by movement in the earliest years of simulated transport and that the years chosen for the start of the transient particle-tracking analysis were years characterized by extreme hydrologic conditions.

Particles started in the northern J-Ranges Area and at Demolition Area 1 and transported through the aquifer for 41 years travelled 960 and 7,800 ft away from their starting positions, respectively. The differences in transport distances arise from the locations of these two areas within the hydrologic system. The northern J-Ranges Area is located near the top of the water-table mound where there is a strong component of vertical flow and horizontal gradients are small. Contaminants released in this area at a specific time will not migrate as far horizontally downgradient as contaminants released at the same time in the Demolition Area 1 and the Impact Area, which are located farther downgradient in areas where the component of horizontal flow is greater.

SUMMARY AND CONCLUSIONS

Contaminated ground water emanates from a number of sources on Camp Edwards on western Cape Cod, and there is concern that contaminants could adversely affect regional water supply. The Army National Guard (ARNG) has been investigating possible ground-water contamination at the site since 1997. Three primary areas of ground-water contamination have been identified: downgradient of Demolition Area 1, in the Central Impact Area, and in the J-Ranges Area. The U.S. Geological Survey (USGS) has assisted in the investigation by developing models to simulate ground-water flow in the aquifer. As part of this effort, USGS developed regional and subregional steady-state models and transient regional models that incorporate seasonal and long-term changes in recharge. The USGS used these models to characterize the hydrologic system, simulate advective transport at specific areas of interest, delineate the recharge areas to water-supply wells, and evaluate the effects of model discretization and the assumption of steady-state hydraulic conditions on model results.

Ground-water flow in the aquifer is radially outward from a water-table mound located to the south of the Impact Area near the northern section of the J-Ranges. Vertical flow is large and horizontal hydraulic gradients are small in areas near the top of the water-table mound. Contaminants in downgradient areas, where the ratio of vertical to horizontal gradients is small and horizontal flow predominates, migrate farther in a specified period of time than contaminants in areas closer to the top of the water-table mound. Forward particle tracking was particularly useful in determining the advective transport paths of contaminants in the direction of ground-water flow in areas with well-defined source areas and histories, such as Demolition Area 1. In areas where source areas are poorly defined, such as the Central Impact Area, reverse particle tracking was used to determine potential source areas of contaminants detected in the subsurface. Particle tracking also was used to determine spatial relationships between sporadic subsurface detections and to interpret water-quality results in a hydrologic context.

The regional model also was used to delineate recharge areas to existing and proposed municipal wells. This activity was done as part of a parallel USGS investigation into the source of water to wells and natural receptors on western Cape Cod. A subregional model was needed to delineate recharge areas to some municipal wells that are either weak sinks in the regional model or located in close proximity to other pumping wells.

A subregional model was used to simulate advective transport in the southern J-Ranges Area. Snake Pond and extraction and injection wells that are part of a remediation system control the local ground-water-flow paths in the area, and simulation results indicate that the regional model is too coarsely discretized to represent the pond and wells adequately and to simulate ground-water flow accurately.

A transient version of the regional model that incorporates seasonal changes in recharge and pumping showed that simulated heads in the aquifer change seasonally, but that the location of the top of the water-table mound and hydraulic-gradient directions in the aquifer do not change seasonally. Seasonal changes in recharge stresses do not change hydraulic gradient directions and likely do not affect advective transport paths because the time scale of the changes (months) is much smaller than the time scale of advective transport (decades).

A version of the regional model that incorporates long-term changes in recharge, as estimated from a 60-year precipitation record from western Cape Cod, showed that long-term changes in recharge cause heads in the aquifer to fluctuate and hydraulic gradients to change. Heads fluctuated by up to 12 ft and the top of the water-table mound migrated nearly 1,500 ft between periods representing high-recharge (1955) and low-recharge (1965) conditions.

Downgradient of the Impact Area, Demolition Area 1, and the southern J-Ranges Area, simulated paths of advective transport do not change in response to temporal changes in recharge rates. Predicted advective transport paths from the northern J-Ranges Area do depend on when particles are started in the long-term transient model. The northern J-Ranges Area is located near the top of the water-table mound where hydraulic-gradient directions change in response to changes in the position of the top of the water table. The Impact Area and Demolition Area 1 are located in downgradient areas where hydraulic-gradient directions do not change substantially with time. In these downgradient areas, transient particle tracks agree with the corresponding steady-state particle track, whereas transient particle tracks started near the top of the water-table mound are not always consistent with the steady-state particle track. The assumption of steady-state conditions appears valid within and downgradient of the Impact Area and Demolition Area 1; however, the assumption is not valid and steady-state models cannot be used to simulate advective transport in areas near the top of the water-table mound.

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