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

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

By DONALD A. WALTER and JOHN P. MASTERSON

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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.

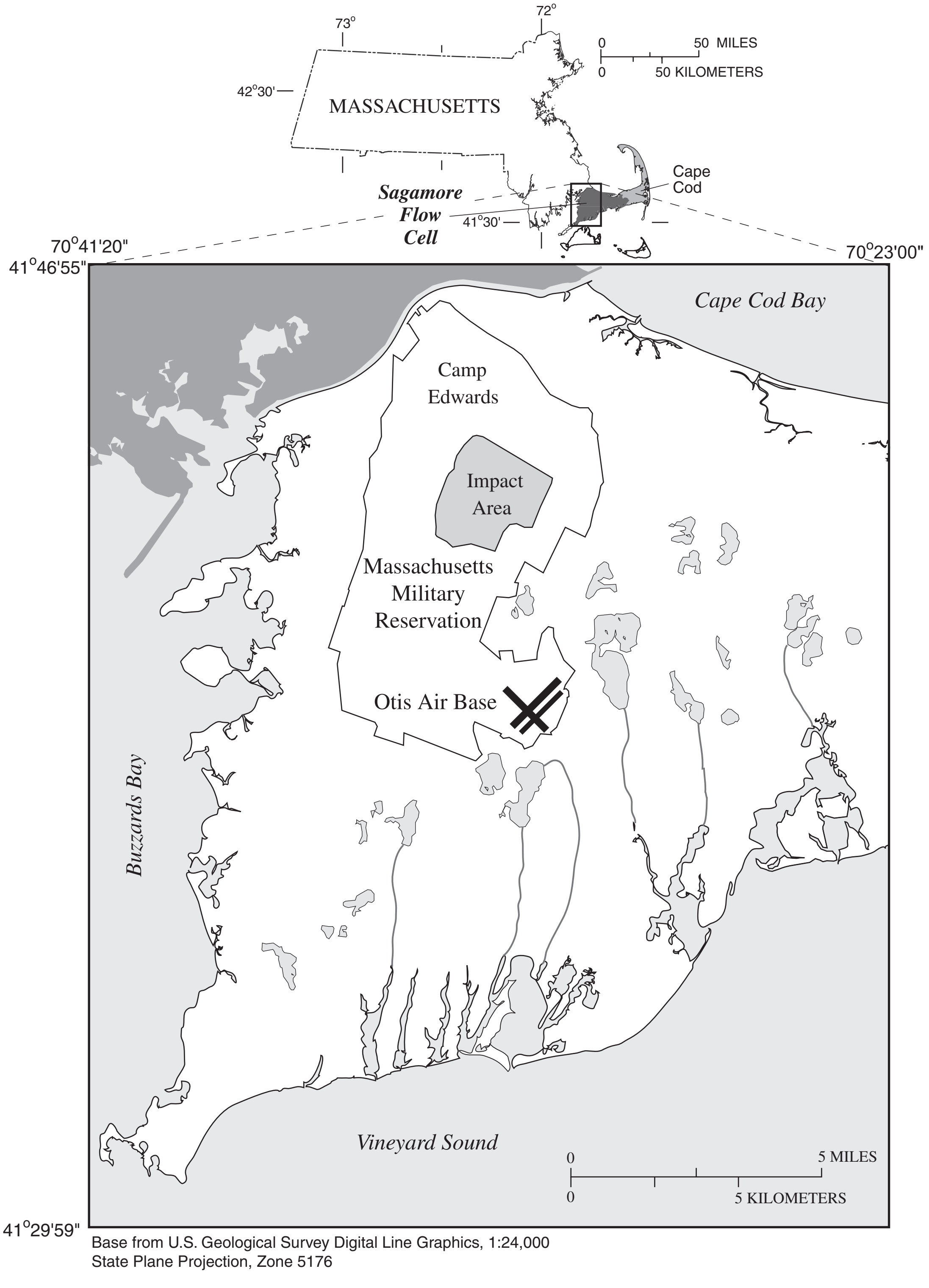


Figure 1. Location of Camp Edwards and its firing ranges, also known as the Impact Area, on the Massachusetts Military Reservation, Cape Cod, Massachusetts.

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.

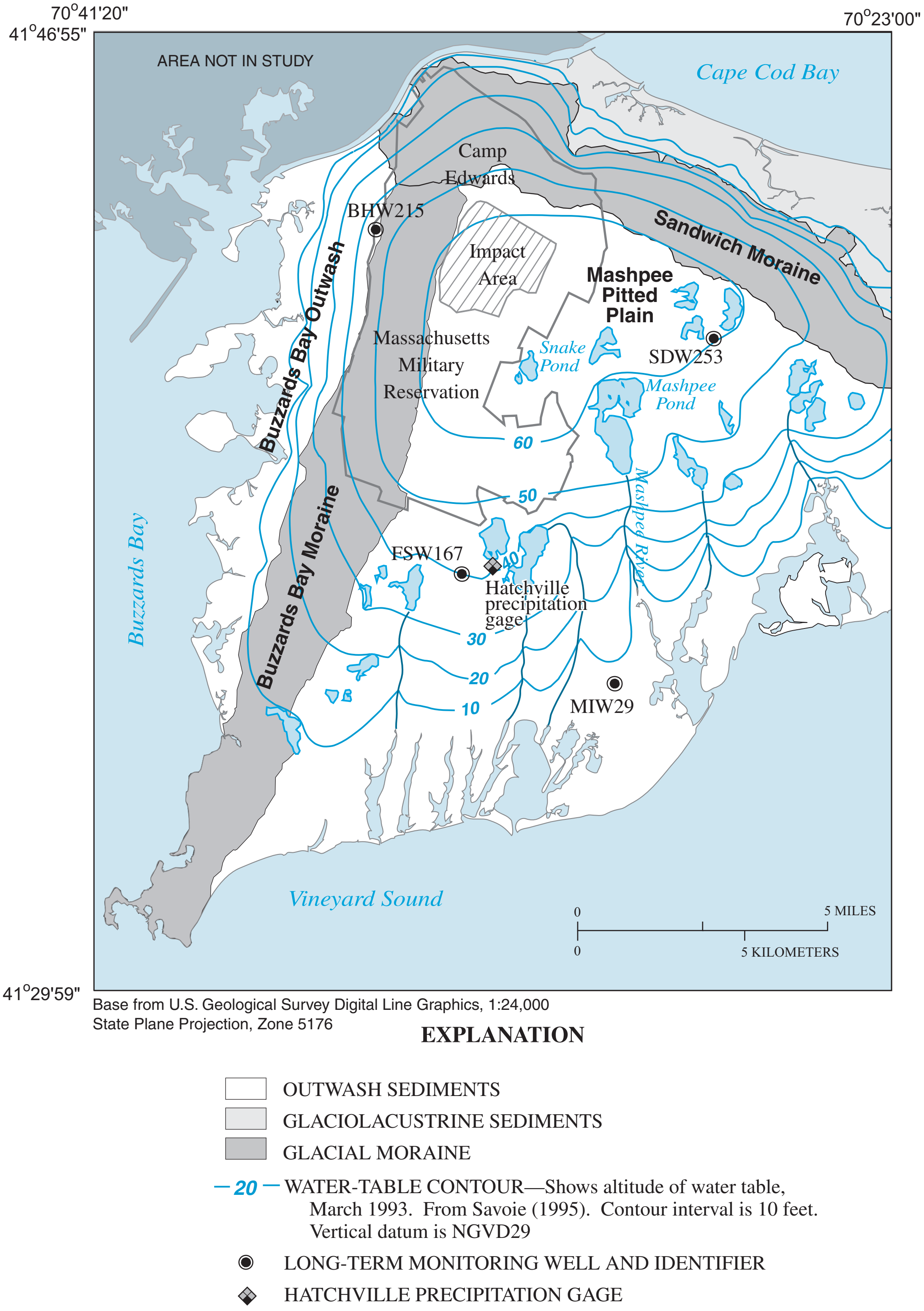
The sequence of glacial deposits on western Cape Cod ranges in thickness from 70 ft near the Cape Cod Canal to more than 400 ft along Vineyard Sound. The lowermost glacial deposits consist of basal-till deposits in most places. The unconsolidated glacial sediments are underlain by crystalline bedrock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

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.

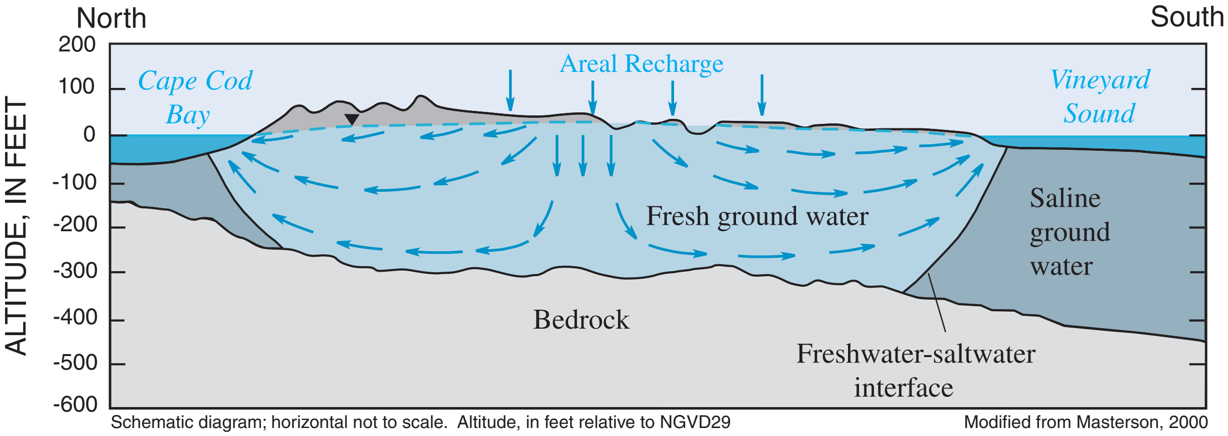
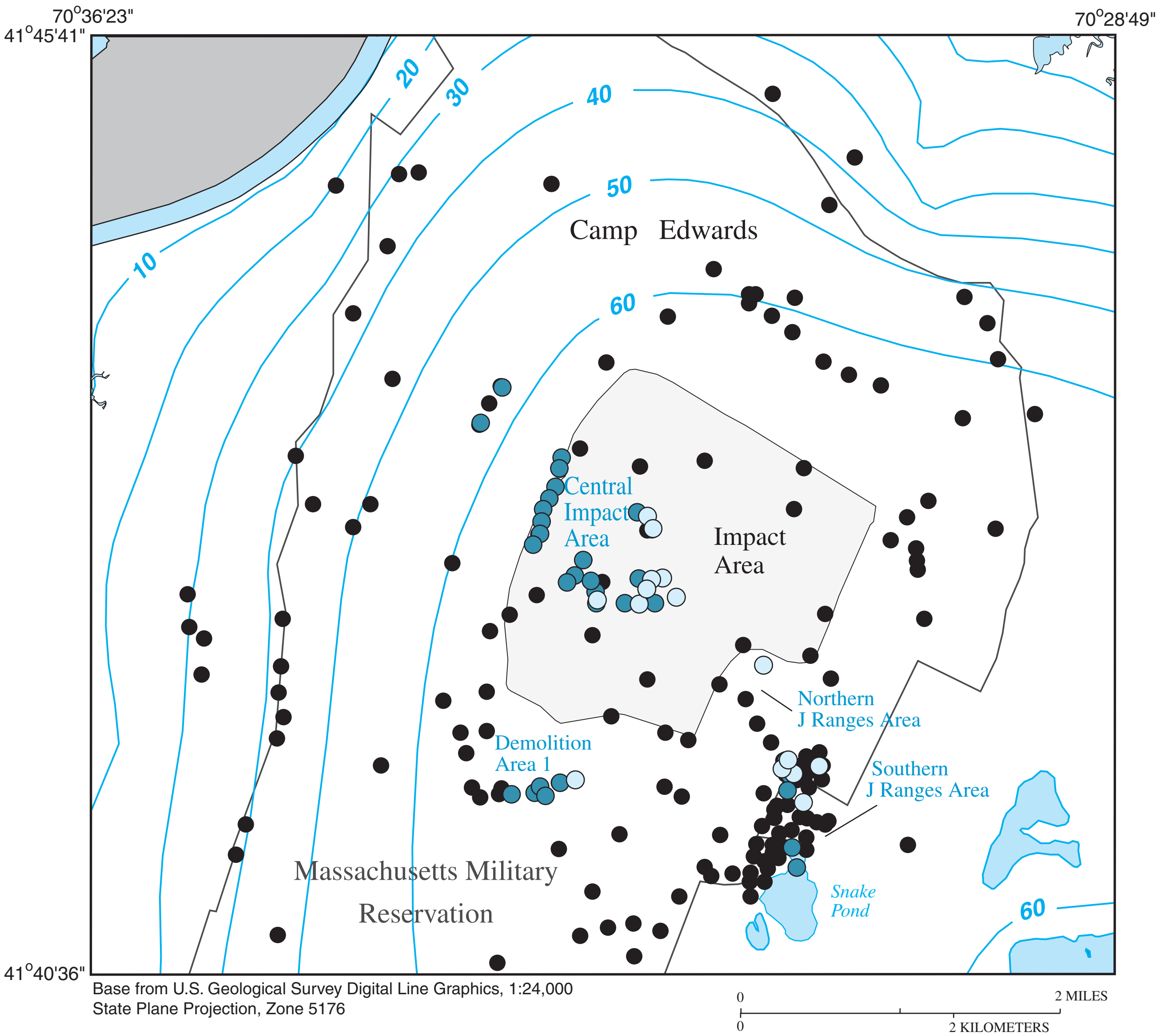


Figure 3. Generalized vertical section (vertically exaggerated) illustrating hydrologic boundaries and general flow lines in the ground-water system of western Cape Cod, Massachusetts.



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

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).

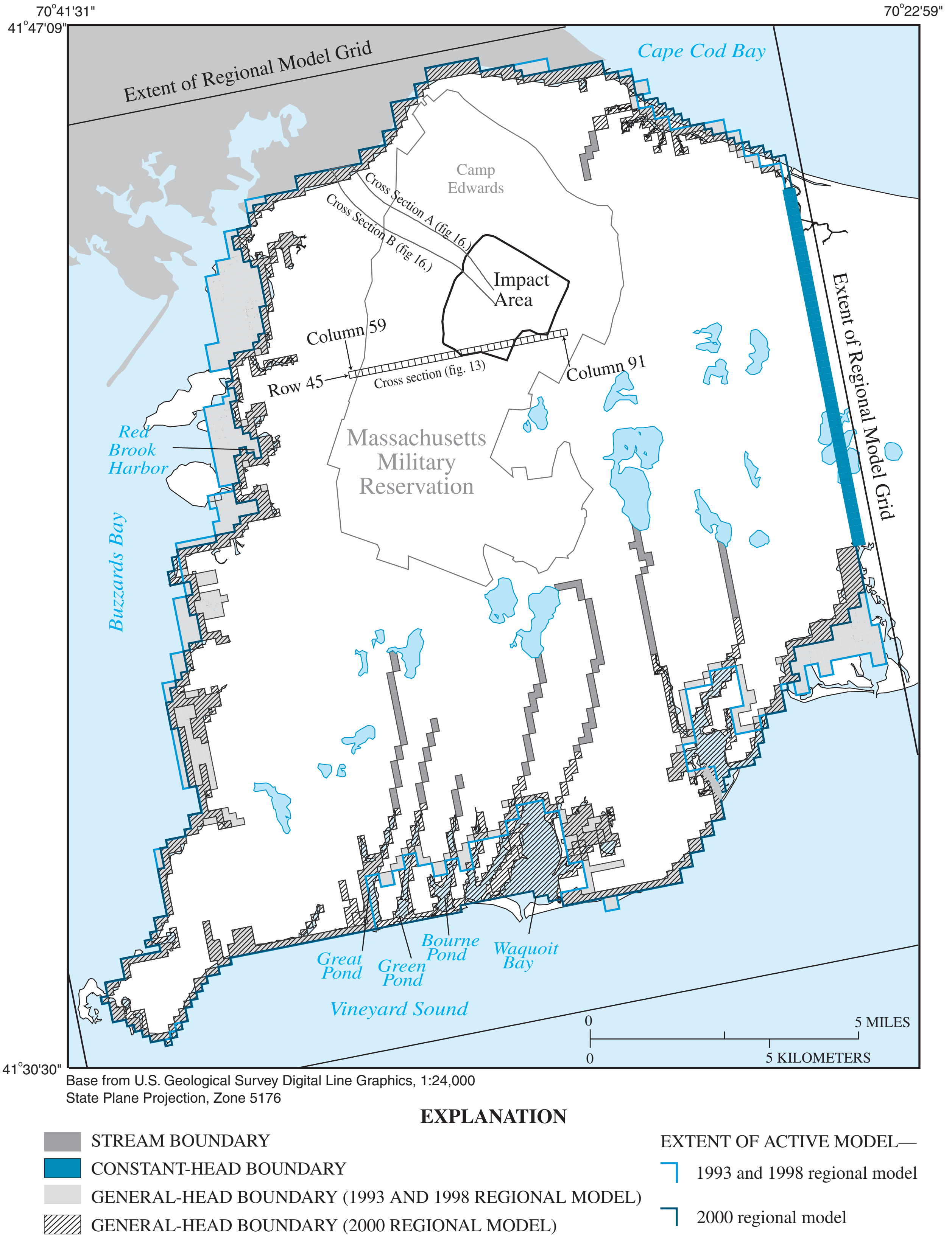


Figure 5. Regional model domains and simulated hydrologic boundaries in the 1993, 1998, and 2000 regional models of western Cape Cod, Massachusetts.

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).

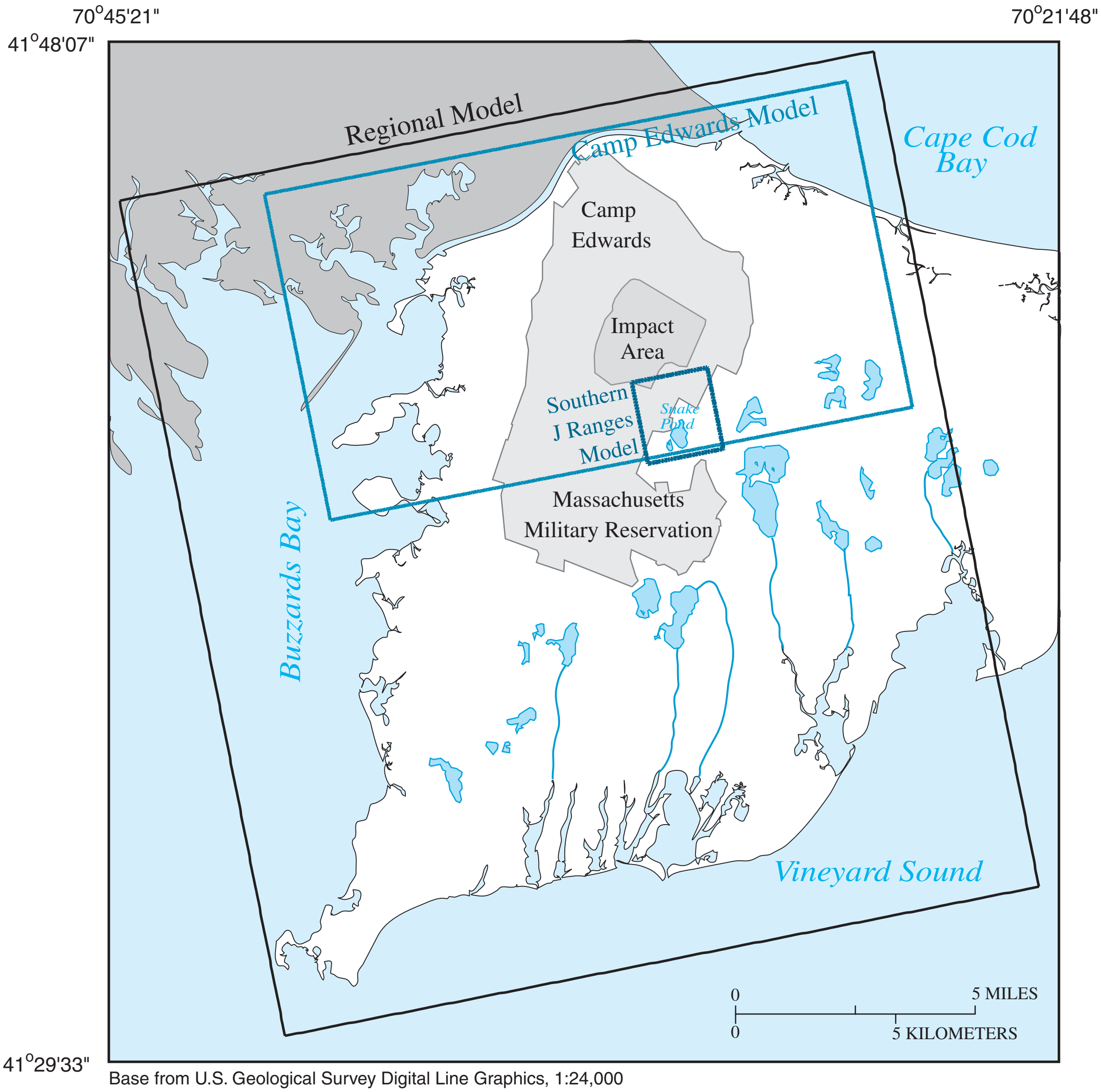
As in the 1998 regional model, streams in the 2000 steady-state regional model were simulated by using the Stream-Routing Package. In the 2000 regional model, however, the same volumetric rate of water specified as entering streams from pond outlets is explicitly removed from the ponds by using a specified-flux 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



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

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*).

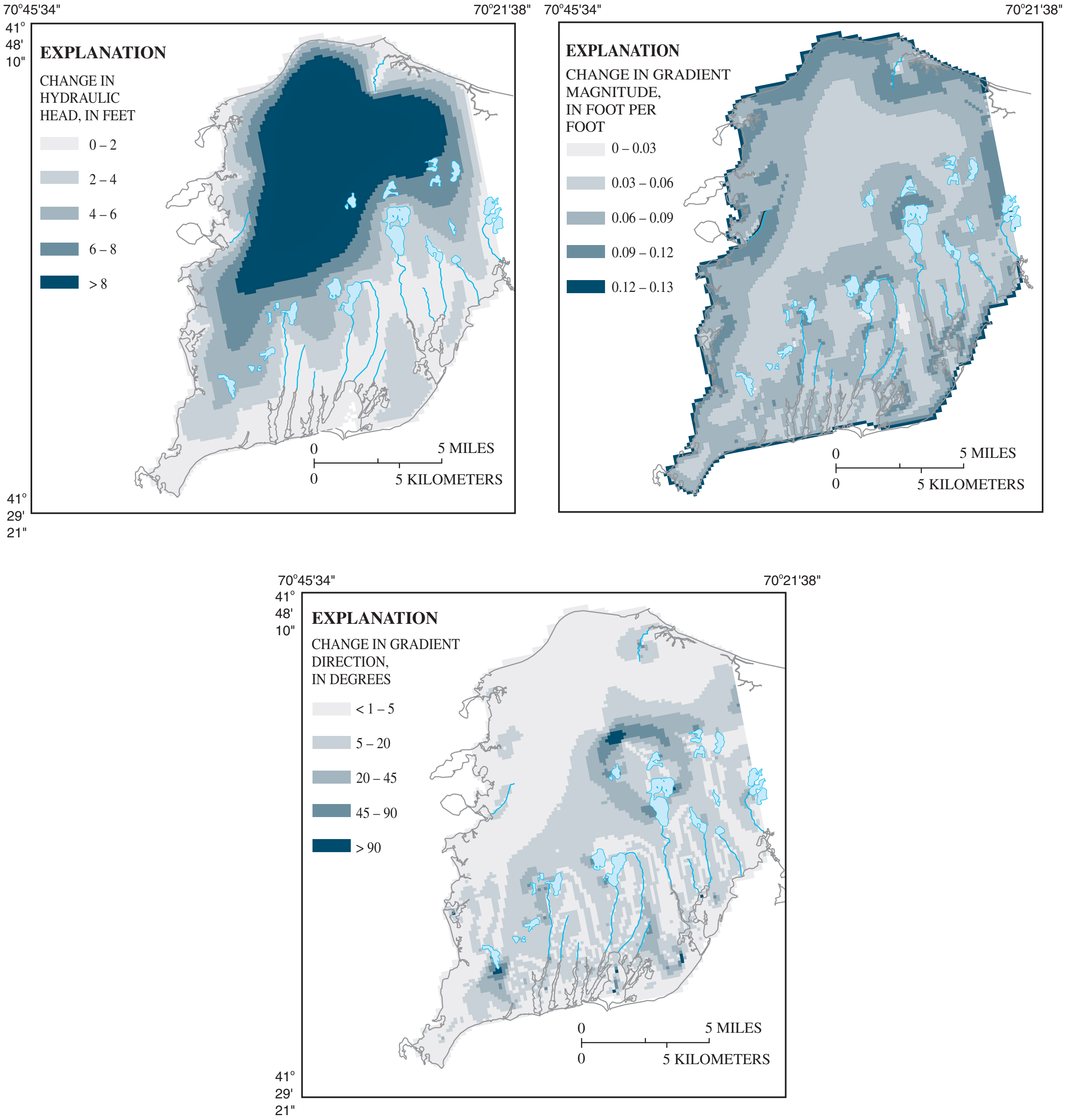


Figure 23. Simulated differences in hydraulic heads and in magnitude and direction of the hydraulic gradient between high-recharge (1955) and low-recharge (1965) conditions on western Cape Cod, Massachusetts.

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.

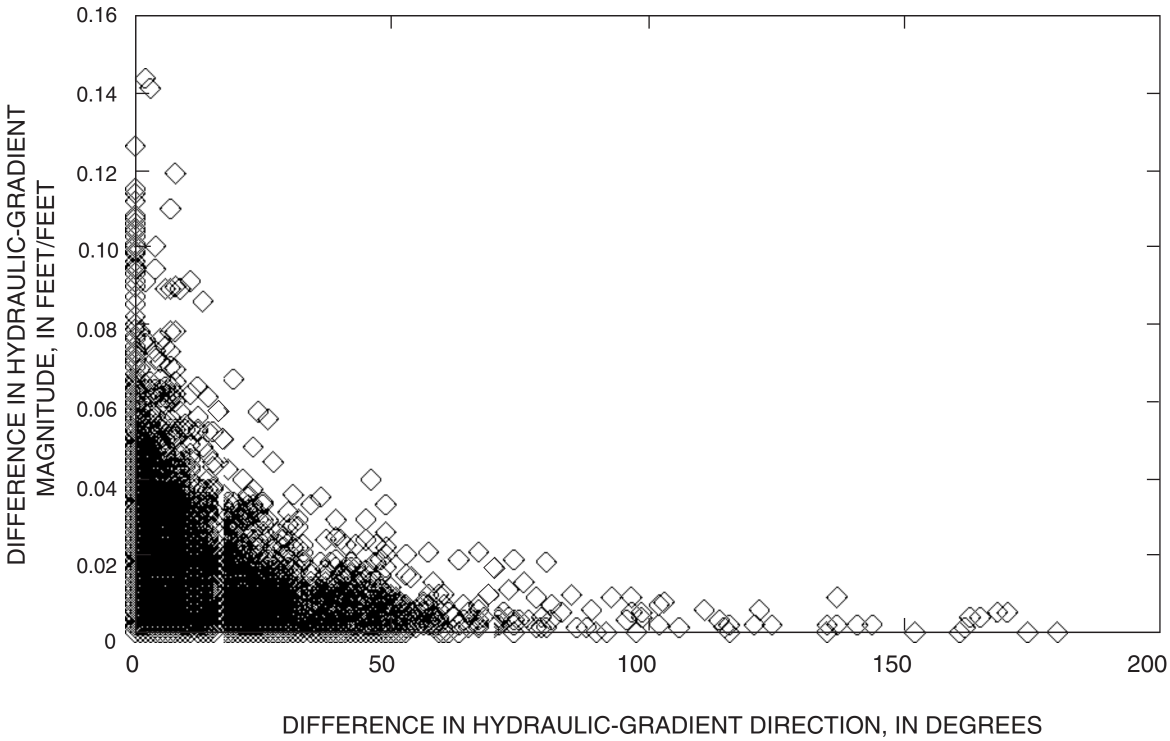


Figure 24. Relation between simulated differences in hydraulic-gradient directions and magnitudes for all active model cells for high-recharge (1955) and low-recharge (1965) conditions on western Cape Cod, Massachusetts.

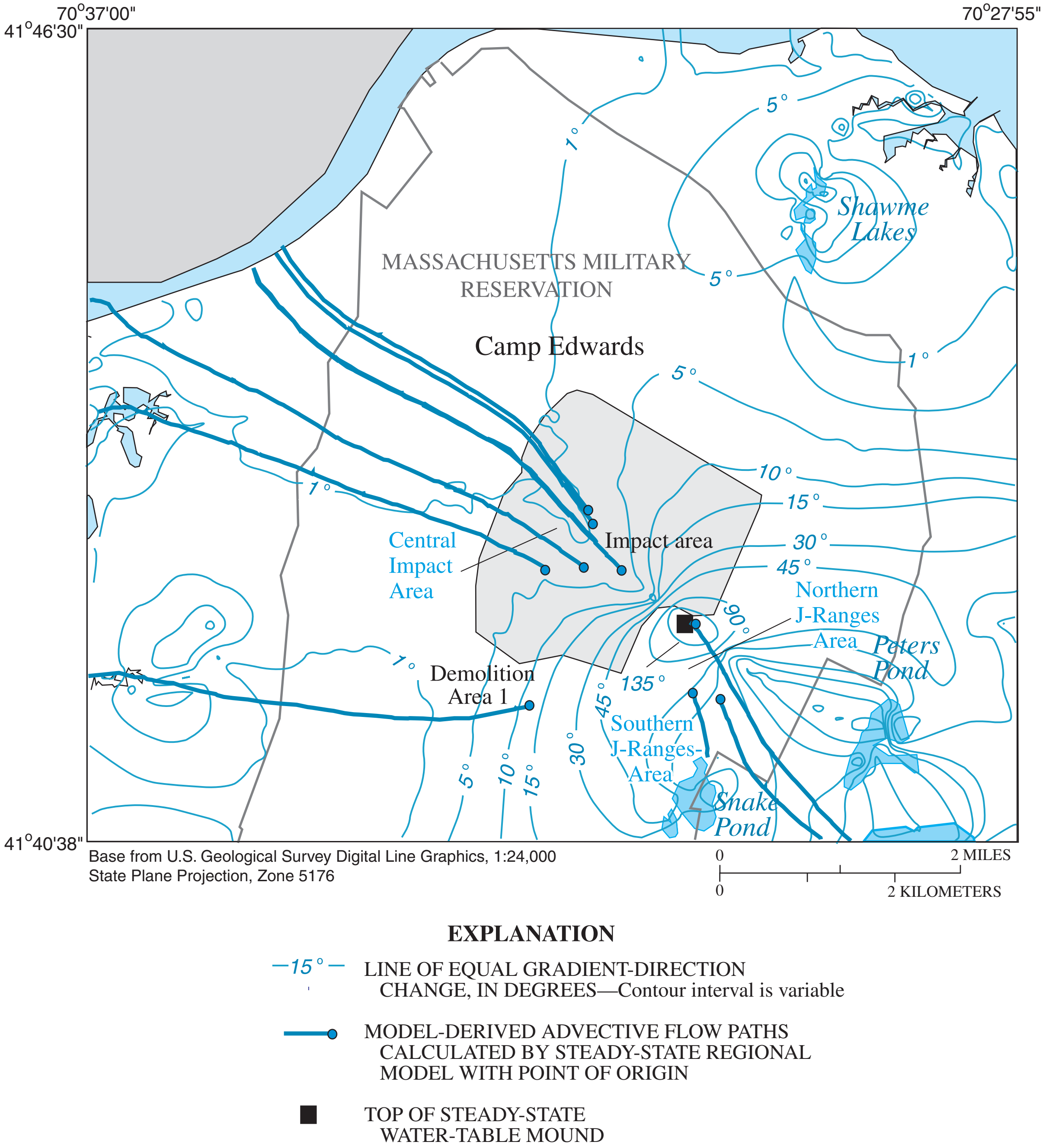


Figure 25. Changes in gradient direction between high-recharge (1955) and low-recharge (1965) conditions for the northern part of the western Cape Cod, Massachusetts, flow cell and steady-state advective flow paths from selected locations.

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).

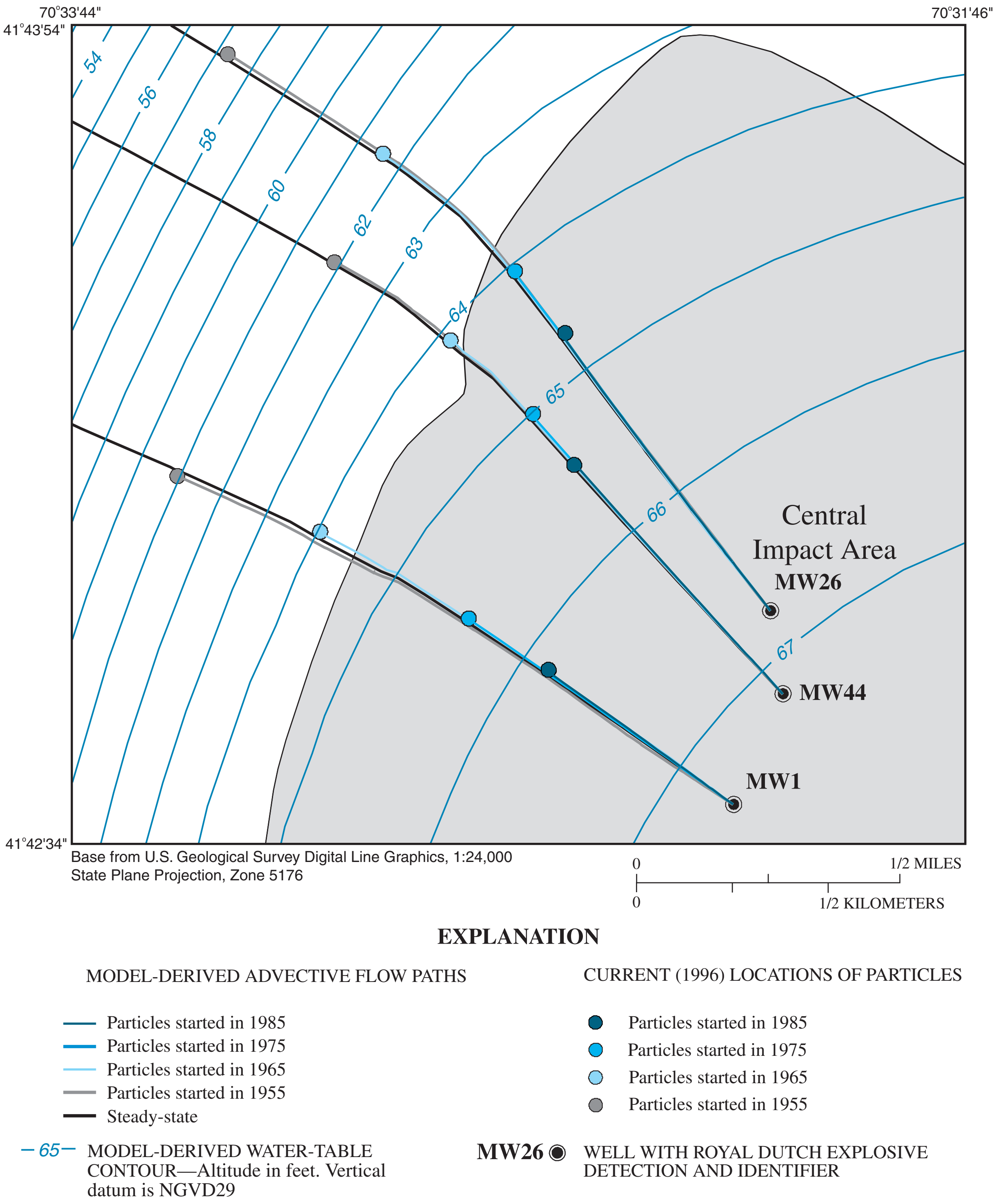


Figure 26. Steady-state and transient advective flow paths of particles started in 1955, 1965, 1975, and 1985 from three locations in the Central Impact Area, western Cape Cod, Massachusetts. Paths are projected to map view. Transient flow paths end at predicted 1996 locations.

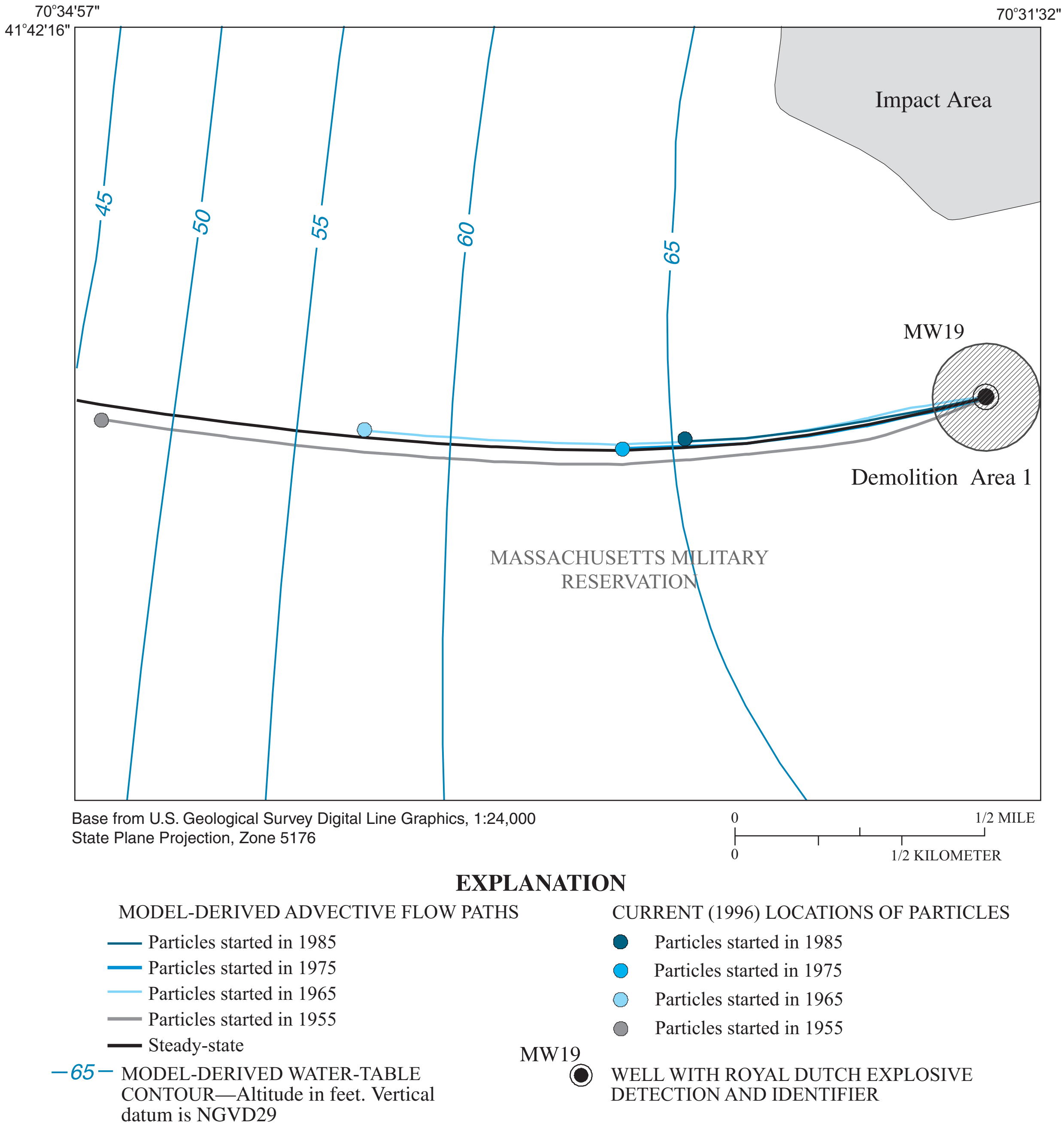


Figure 27. Steady-state and transient advective flow paths of particles started in 1955, 1965, 1975, and 1985 from Demolition Area 1, western Cape Cod, Massachusetts. Paths are projected to map view. Transient flow paths end at predicted 1996 locations.

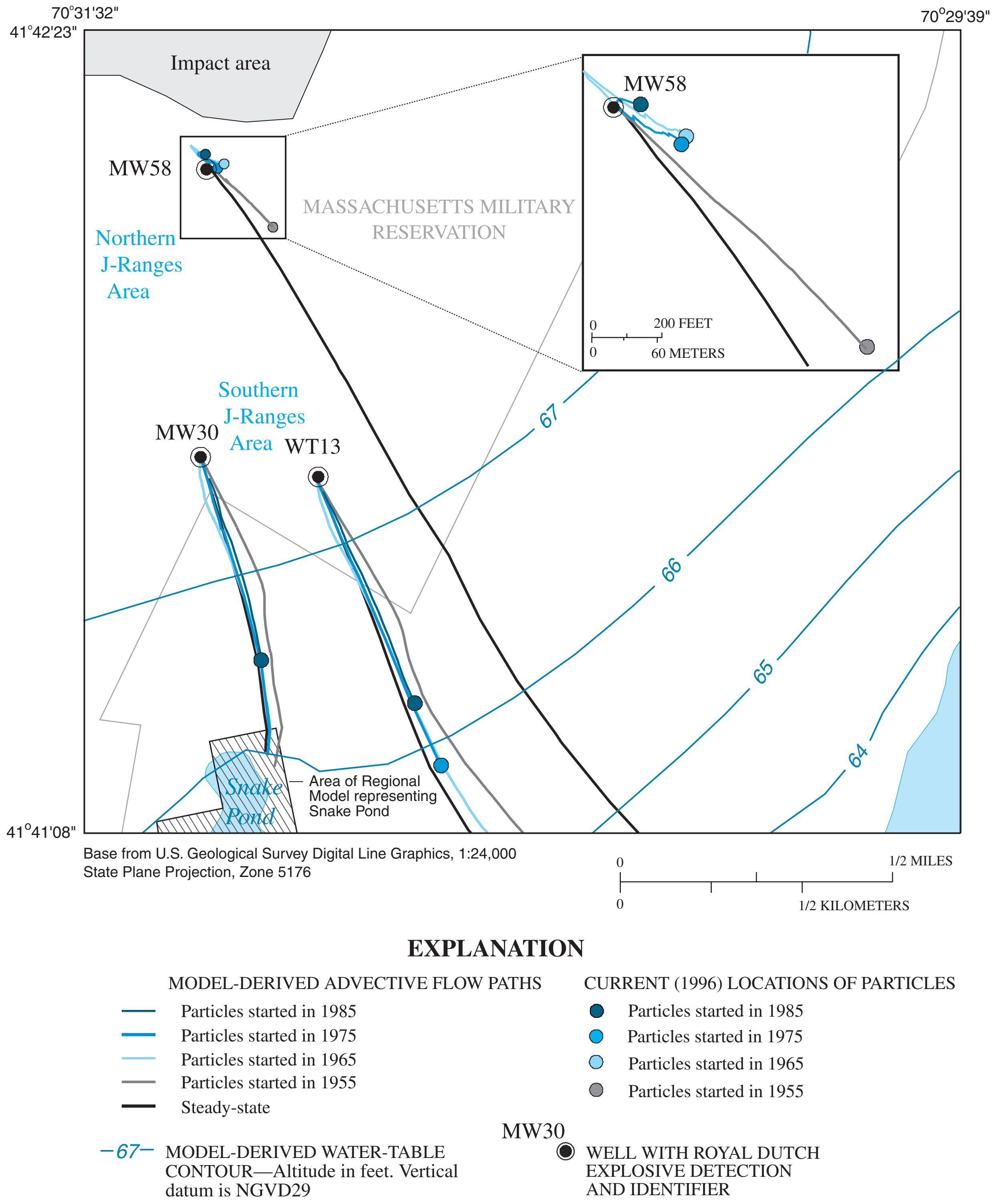


Figure 28. Steady-state and transient advective flow paths of particles started in 1955, 1965, 1975, and 1985 from three locations in the J-Ranges Area, western Cape Cod, Massachusetts. Paths are projected to map view. Transient flow paths end at predicted 1996 locations.

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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