# Simulated Interaction Between Freshwater and Saltwater and Effects of Ground-Water Pumping and Sea-Level Change, Lower Cape Cod Aquifer System, Massachusetts

By John P. Masterson

In cooperation with the
National Park Service,
Massachusetts Executive Office of Environmental Affairs,
Cape Cod Commission, and the
Towns of Eastham, Provincetown, Truro, and Wellfleet

Scientific Investigations Report 2004-5014

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

**U.S. Department of the Interior**
Gale A. Norton, Secretary

**U.S. Geological Survey**
Charles G. Groat, Director

U.S. Geological Survey, Reston, Virginia: 2004
For sale by U.S. Geological Survey, Information Services
Box 25286, Denver Federal Center
Denver, CO 80225

For more information about the USGS and its products:
Telephone: 1-888-ASK-USGS
World Wide Web: http://www.usgs.gov/



Masterson, J.P., 2004, Simulated interaction between freshwater and saltwater and effects of ground-water pumping and sea-level change, Lower Cape Cod aquifer system, Massachusetts: U.S. Geological Survey Scientific Investigations Report 2004-5014, 72 p.

# CONTENTS



## Figures

# Tables

# CONVERSION FACTORS, DATUMS, AND ABBREVIATIONS

| Multiply | By | To Obtain |
|---|---|---|
| cubic foot per second ($ft^3/s$) | 0.02832 | cubic meter per second ($m^3/s$) |
| foot (ft) | 0.3048 | meter (m) |
| foot per day (ft/d) | 0.3048 | meter per day (m/d) |
| foot per year (ft/yr) | 0.3048 | meter per year (m/yr) |
| foot squared per day ($ft^2/d$) | 0.0929 | meters squared per day ($m^2/d$) |
| gallon per day (gal/d) | 0.003785 | cubic meter per day ($m^3/d$) |
| inch (in.) | 25.4 | millimeter (mm) |
| inch per year (in/yr) | 25.4 | millimeter per year (mm/yr) |
| mile (mi) | 1.609 | kilometer (km) |
| million gallons per day (Mgal/d) | 0.04381 | cubic meter per second ($m^3/s$) |
| pounds per cubic foot ($lb/ft^3$) | 16,018 | milligrams per liter (mg/L) |

Vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29). Horizontal coordinate information is referenced to the North American Datum of 1927 (NAD 27).

| | |
|---|---|
| CCNS | Cape Cod National Seashore |
| EM | Electromagnetic |
| IPEC | Intergovernmental Panel on Climate Change |
| NPS | National Park Service |
| NTAFB | North Truro Air Force Base |
| TAZ | Traffic Analysis Zone |
| USGS | U.S. Geological Survey |
| WRAB | Eastham Water Resources Advisory Board |

# Simulated Interaction Between Freshwater and Saltwater and Effects of Ground-Water Pumping and Sea-Level Change, Lower Cape Cod Aquifer System, Massachusetts

By John P. Masterson

## Abstract

The U.S. Geological Survey, in cooperation with the National Park Service, Massachusetts Executive Office of Environmental Affairs, Cape Cod Commission, and the Towns of Eastham, Provincetown, Truro, and Wellfleet, began an investigation in 2000 to improve the understanding of the hydrogeology of the four freshwater lenses of the Lower Cape Cod aquifer system and to assess the effects of changing ground-water pumping, recharge conditions, and sea level on ground-water flow in Lower Cape Cod, Massachusetts.

A numerical flow model was developed with the computer code SEAWAT to assist in the analysis of freshwater and saltwater flow. Model simulations were used to determine water budgets, flow directions, and the position and movement of the freshwater/saltwater interface.

Model-calculated water budgets indicate that approximately 68 million gallons per day of freshwater recharge the Lower Cape Cod aquifer system with about 68 percent of this water moving through the aquifer and discharging directly to the coast, 31 percent flowing through the aquifer, discharging to streams, and then reaching the coast as surface-water discharge, and the remaining 1 percent discharging to public-supply wells. The distribution of streamflow varies greatly among flow lenses and streams; in addition, the subsurface geology greatly affects the position and movement of the underlying freshwater/saltwater interface.

The depth to the freshwater/saltwater interface varies throughout the study area and is directly proportional to the height of the water table above sea level. Simulated increases in sea level appear to increase water levels and streamflows throughout the Lower Cape Cod aquifer system, and yet decrease the depth to the freshwater/saltwater interface. The resulting change in water levels and in the depth to the freshwater/saltwater interface from sea-level rise varies throughout the aquifer system and is controlled largely by non-tidal freshwater streams.

Pumping from large-capacity municipal-supply wells increases the potential for effects on surface-water bodies, which are affected by pumping and wastewater-disposal locations and rates. Pumping wells that are upgradient of surface-water bodies potentially capture water that would otherwise discharge to these surface-water bodies, thereby reducing streamflow and pond levels. Kettle-hole ponds, such as Duck Pond in Wellfleet, that are near the top of a freshwater flow lens, appear to be more susceptible to changing pumping and recharge conditions than kettle-hole ponds closer to the coast or near discharge boundaries, such as the Herring River.

# Introduction

The ground-water lenses that constitute the Lower Cape Cod aquifer system (Nauset, Chequesset, Pamet, and Pilgrim lenses), are the sole source of drinking water for the Towns of Eastham, Wellfleet, Truro, and Provincetown, and the Cape Cod National Seashore (fig. 1). Increased land development and population growth have created concerns regarding both the quantity and the quality of ground water that is used for drinking water and that discharges to surface-water bodies and coastal areas throughout Lower Cape Cod.

These concerns described above include the effects of increased ground-water pumping on the position of the interface between freshwater and saltwater and on the amount of freshwater discharge to ponds, streams, and coastal areas. Ground-water discharge on Lower Cape Cod is the primary source of water for kettle-hole ponds and streams, and it also is a key component in the maintenance of the ecologically sensitive coastal embayments. Declines in water levels because of increases in ground-water withdrawals could have a detrimental effect on these natural resources. Small changes in water-table altitude can result in substantial decreases in ground-water discharge to streams and coastal embayments, and can substantially affect the shoreline position of the many kettle-hole ponds throughout Lower Cape Cod (Sobczak and others, 2003).

Presently (2003), only the residents of Provincetown and small portions of Truro and Wellfleet are serviced by a public-water supply system. The other residents of Lower Cape Cod obtain drinking water from shallow, small-capacity domestic-supply wells. As land development increases and wastewater continues to be returned to the aquifer through on-site, domestic septic systems, there is a growing concern that the increased amounts of non-point source contamination in the Lower Cape Cod aquifer system may adversely affect the existing water supply and may necessitate a shift from small-capacity domestic supplies to larger, more centralized municipal supplies (Sobczak and Cambareri, 1998).

Federal, State, and local officials responsible for managing and protecting water resources are concerned that a shift to large-capacity, centralized municipal supplies may create the potential for unacceptable declines in water-table and pond altitudes, decreases in ground-water discharge to streams and coastal areas, and saltwater intrusion. In response to these concerns, the U.S. Geological Survey (USGS), in cooperation with the National Park Service, Massachusetts Executive Office of Environmental Affairs, Cape Cod Commission, and the Towns of Eastham, Provincetown, Truro, and Wellfleet began an investigation in 2000 to improve the understanding of the hydrogeology of the Lower Cape Cod aquifer system and to assess possible effects of proposed water-management strategies on Lower Cape Cod.

This report describes the hydrogeology of the four flow lenses of the Lower Cape Cod aquifer system. A numerical ground-water-flow model was developed as part of this investigation to assist in the analysis of freshwater and saltwater flow for current and changing pumping and recharge conditions. Results from previous investigations that characterized the hydrogeology and ground-water flow of Lower Cape Cod, such as Guswa and LeBlanc (1985), LeBlanc and others (1986), Cambareri and others (1989), Masterson and Barlow (1996), Barlow (1996), Martin (1993), and Sobczak and Cambareri (1998), served as the foundation for the understanding of ground-water flow in the Lower Cape aquifer system. Results from these previous investigations were incorporated into the development and calibration of the ground-water-flow model developed for this investigation.

The newly released computer program SEAWAT (Guo and Langevin, 2002) was used to provide information about regional-scale flow in the ground-water-flow lenses, including regional movement of the interface separating the freshwater- and saltwater-flow systems. Although detailed analyses of local-scale hydrologic conditions were beyond the scope of this investigation, the flow model may serve as the starting point for more detailed, site-specific investigations where local-scale models may be developed.

The author thanks the members of the Lower Cape Cod Stakeholders Committee for their assistance and guidance throughout the duration of this investigation as well as the individuals from the following organizations who provided data or assisted in the aquisition of data during this investigation: Cape Cod Commission; National Park Service; Towns of Eastham, Provincetown, Wellfleet, and Truro; Barnstable County Board of Health; and Environmental Partners Group, Inc. The author also thanks USGS colleagues Ann Whealan and Timothy McCobb for their assistance in data collection and compilation, Stephen Garabedian for his guidance in solute-transport modeling, and Byron Stone for his assistance in interpreting the depositional history of the glacial sediments of Lower Cape Cod.

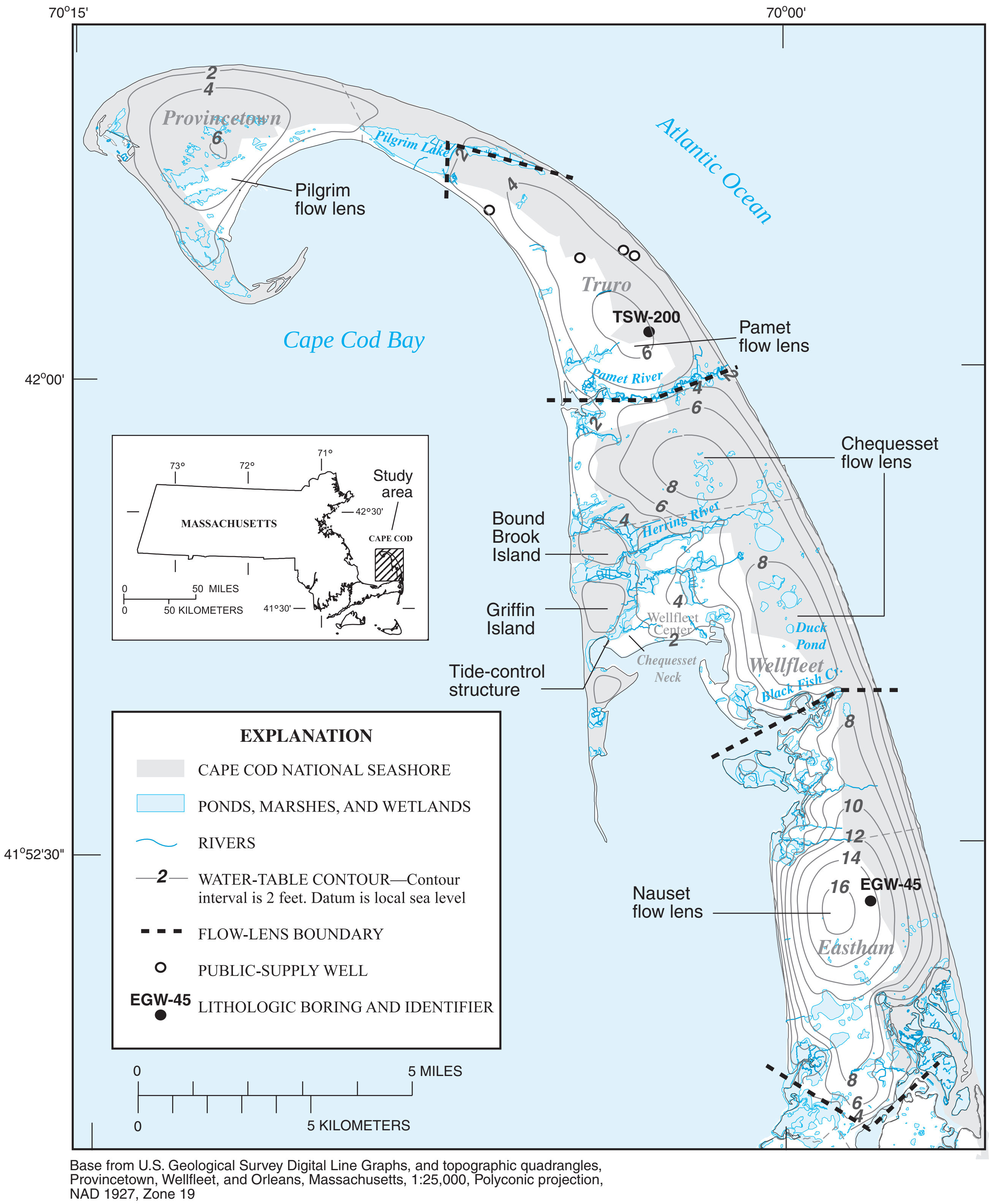



**Figure 1.** Location of the four flow lenses of the Lower Cape Cod aquifer system and model-calculated water-table contours, Cape Cod, Massachusetts.

# Geologic Setting

The glacial deposits that constitute the Lower Cape Cod aquifer system consist of sediments that range in size from clay to boulders. Approximately 15,000 years ago during the late Wisconsinan glacial stage of the Pleistocene Epoch (Oldale and O'Hara, 1984), streams flowing from the coalescing lobes of the Cape Cod Bay and South Channel glacial ice sheets deposited the glacial sediments that now constitute Lower Cape Cod (fig. 2). These surficial deposits (fig. 3) overlie Paleozoic crystalline bedrock that ranges in altitude from about 450 ft below NGVD 29 in Eastham to more than 900 ft below NGVD 29 in Truro (fig. 1) (Oldale, 1992).

## Depositional History

The sediments of the Lower Cape Cod aquifer system were deposited by meltwater from the retreating Cape Cod Bay and South Channel Lobe ice sheets as deltas prograded into a large glacial lake that formed in present-day Cape Cod Bay (Oldale, 1992). Glacial Lake Cape Cod was dammed to the south and west by the older glacial deposits of upper and middle Cape Cod and to the north and east by the ice sheets. The glacial lake grew in size in the wake of the retreating ice sheets.

The lake level of this glacial lake changed with time and that change is reflected, in part, in the altitude of the present-day land surface throughout Lower Cape Cod. The land-surface altitudes of the outwash plains represent the tops of the fluvial sediments deposited by braided rivers flowing from the ice lobes. The subsurface contact between the horizontal beds of river deposits and sloping beds of glaciolacustrine deposits indicates the lake stage that controlled the deltaic deposition. Oldale (1992) reports that the glacial lake stage was about 50 ft above the present sea level when the Wellfleet plain was deposited and less than 30 ft above present-day sea level when the Eastham plain was deposited. This deposition indicates that the stage of the glacial lake changed with time and that it was higher than the present-day sea level.

The flat surfaces of the outwash plains are altered by the numerous kettle holes that were formed as collapse structures by the melting of buried blocks of ice stranded by the retreating ice lobes. These ice blocks, stranded directly on basal till and bedrock, subsequently were buried by prograding deltaic sediments. When the buried ice blocks melted, coarse sands and gravels collapsed into the resulting depressions. The kettle holes that intercept the water table now are occupied by kettle-hole ponds.

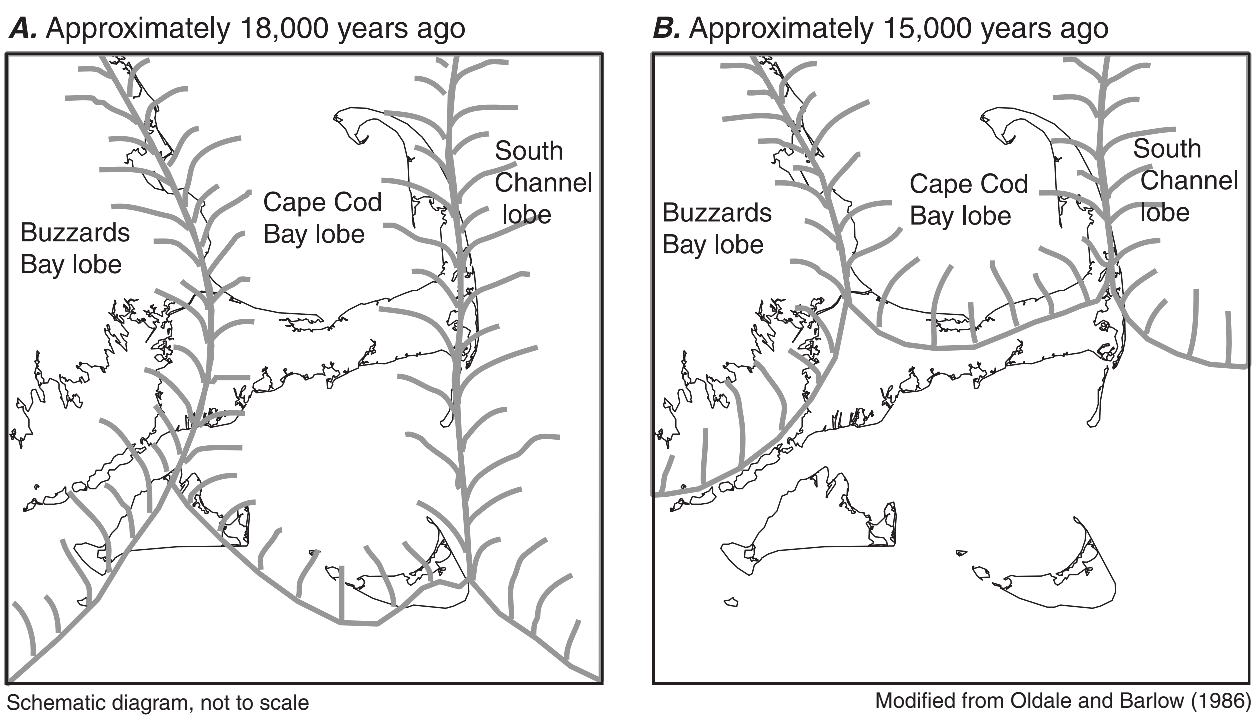



**Figure 2.** Ice recession and lobe formation in southeastern Massachusetts with respect to the present-day geography of the Cape Cod area.

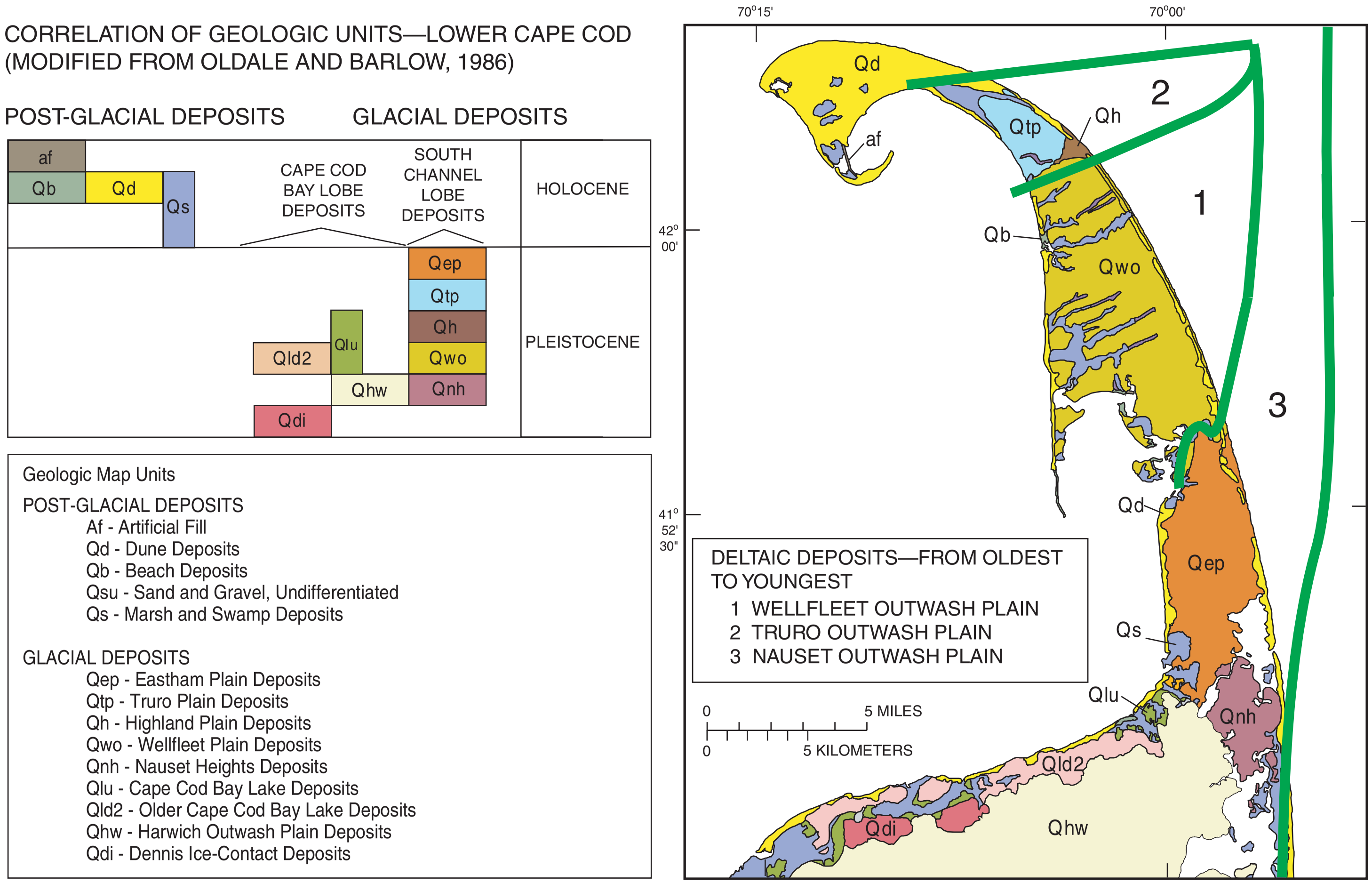



**Figure 3.** Surficial geology of Lower Cape Cod, Massachusetts (from Foster and Poppe, 2003; modified from Oldale and Barlow, 1986), and the depositional sequence of the Wellfleet, Truro, and Eastham outwash plains.

In addition, east-west-trending stream valleys, or pamets, were carved into the outwash-plain surfaces. These valleys formed possibly as a result of a process referred to as "spring sapping" (Oldale, 1992), in which ground-water springs intersect and erode up a gently sloping land surface. The gradual inland migration of the springs erodes the land and forms steep-headed channels.

The deltaic sediments deposited in this glacial lake can be divided into topset, foreset, and bottomset beds (fig. 4) (B.D. Stone, U. S. Geological Survey, oral commun., 2001). The topset beds consist of glaciofluvial outwash of coarse sand and gravel deposited by braided rivers flowing from the ice lobes. The underlying foreset beds are glaciolacustrine sediments that consist mostly of medium to fine sand with some silt and are deposited subaqueously in a nearshore lake environment. The bottomset beds are glaciolacustrine sediments that consist of fine sand, silt, and clay and are deposited in an offshore lake environment.

The sediments transported by meltwater streams flowing from the retreating ice lobes into Glacial Lake Cape Cod created three large deltas that constitute the Wellfleet, Truro, and Eastham outwash plains. The order in which these deltas were deposited was determined by the position and movement of the South Channel and Cape Cod Bay ice lobes (fig. 2). The Wellfleet plain is the oldest of the three large plains, the Truro plain is intermediate, and the Eastham plain is the youngest (fig. 3). The sediment sources for these plains generally were north and east of the present location of Lower Cape Cod in what is now the Atlantic Ocean. A smaller outwash plain, the Highland plain, formed after the Wellfleet plain but before the Truro plain in a small glacial lake between the Wellfleet plain and the South Channel ice sheet (Oldale, 1992).

Once the ice sheets retreated, sea level began to rise and erosion of the glacial Cape Cod shoreline began. The original heads of the outwash plains were eroded back to the present-day shoreline. The erosion that resulted from sea-level rise left behind the distal portions of the outwash plains that constitute three (Nauset, Chequesset, and Pamet) of the four flow lenses of the Lower Cape Cod aquifer system.

The Pilgrim flow lens was formed by the progradation of a post-glacial barrier spit of sand deposited by the prevailing ocean currents as sea level rose during the Holocene Epoch. These sands consist of eroded glacial material deposited on top of older glaciolacustrine sediments that were deposited in the offshore environment present during the deposition and subsequent erosion of the Truro plain (Zeigler and others, 1965; Uchupi and others, 1996). The Provincetown spit and resulting sand dunes continue to migrate today and parts of the outwash plains that are exposed to the Atlantic Ocean continue to be eroded at a retreat rate as high as 7 to 11 ft/yr (Oldale, 1992).

## Geologic Framework

The general trends in sediment distribution within deltaic deposits are coarsening upward and fining with distance from the sediment source. This general trend is illustrated in lithologic sections reported by Masterson and others (1997) for western Cape Cod. On Lower Cape Cod, however, the distance of the present-day outwash plains from the now-eroded sediment sources (heads of the outwash plains) is reflected in the subsurface grain-size distributions in the flow lenses and accounts for the large differences in hydraulic properties and ground-water-flow patterns between the lenses (B.D. Stone, U.S. Geological Survey, oral commun., 2001). For example, in the Pamet flow lens, which consists of Wellfleet and Truro plain deposits, the sediment source of the glacial deposits is much closer than the sediment source for the deposits of the Eastham plain that constitute the Nauset flow lens (fig. 3). The differences in distance from the sediment sources between the outwash plains is reflected in the subsurface lithology. In the Pamet lens, lithologic borings show thick sequences of fine, medium, and coarse sand and gravel that extend several hundred feet below land surface. Conversely, lithologic borings in the Nauset lens show a thin layer (less than a hundred feet) of fine, medium, and coarse sand and gravel underlain by a thick sequence of layered silts and clays.

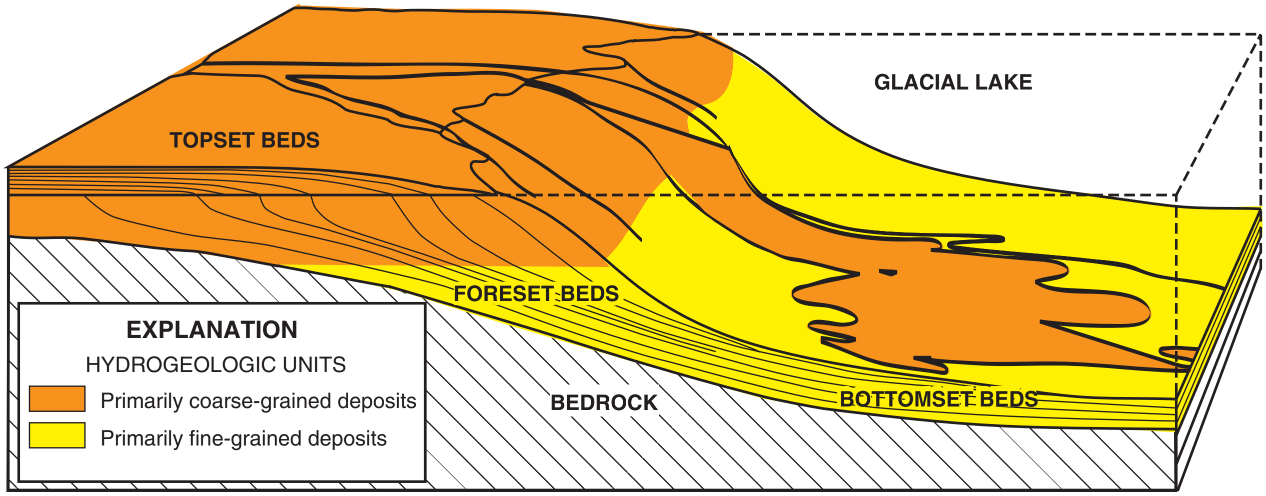



**Figure 4.** Schematic diagram of deltaic deposits prograding into a glacial lake, including topset, foreset, and bottomset deposits (modified from Smith and Ashley, 1985).

USGS Coastal and Marine scientists (Foster and Poppe, 2003) used marine seismic radar off the coasts of Cape Cod Bay and the Atlantic Ocean near North Eastham and South Wellfleet to determine the offshore extent of the glaciolacustrine deposits observed in the onshore lithologic borings. On the basis of the seismic profiles, they concluded that the thick deposits of layered silts and clays observed in the onshore lithologic boring extend beyond the eastern and western shores of the Nauset flow lens, and, therefore, are extensive beneath this flow lens.

This discussion on the glacial history and geologic setting of Lower Cape Cod is presented in this report to provide a cursory description of the geologic framework that served as the foundation for the depositional model of the glacial sediments incorporated into the ground-water-flow model developed for this investigation. For more detailed descriptions and analyses of the glacial history and geologic framework of Cape Cod, readers are referred to the following reports: Woodworth and Wigglesworth (1934); Kaye (1964); Zeigler and others (1965); Oldale and O'Hara (1984); Oldale and Barlow (1986); Oldale (1992); and Uchupi and others (1996).

# Hydrologic System

The glacial sediments that underlie the towns of Eastham, Wellfleet, Truro, and Provincetown constitute the Lower Cape Cod aquifer. The freshwater flow in this aquifer is bounded laterally and below by saltwater (fig. 5), and it often is referred to as an aquifer system because it consists of four freshwater flow lenses—Nauset, Chequesset, Pamet, and Pilgrim (Horsley and others, 1985). The flow lenses were characterized and the aquifer system was analyzed under changing hydrologic conditions by use of the ground-water-flow model developed for this investigation. A detailed discussion of the development and calibration of this model is provided in the appendix.

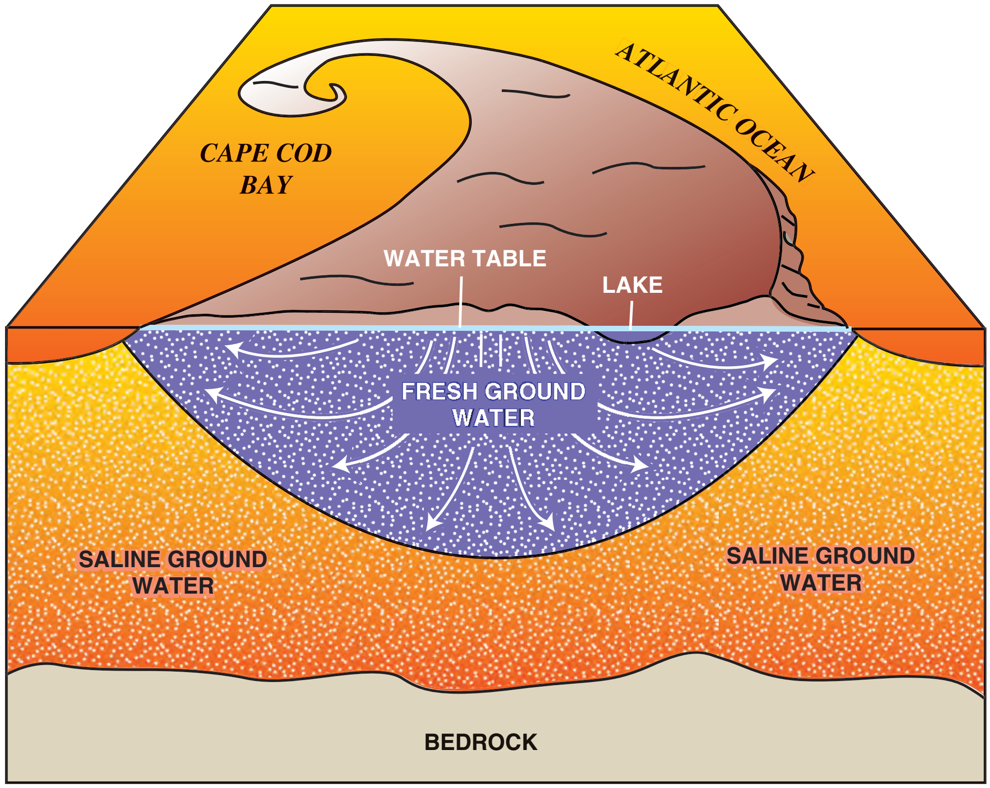



**Figure 5.** Schematic diagram of the Lower Cape Cod aquifer system, Cape Cod, Massachusetts (modified from Strahler, 1972).

## Simulation of Ground-Water Flow in the Lower Cape Cod Aquifer System

The freshwater flow lenses of the Lower Cape Cod aquifer system consist of four large water-table mounds separated from one another by inter-lens surface-water-discharge areas. Under current conditions, the four flow lenses are hydraulically independent of one another (fig. 1). Ground water flows radially from the tops of the ground-water mounds toward the coast and the inter-lens surface-water-discharge areas; flow from one lens does not discharge to another lens.

The inter-lens discharge areas are tidally affected marshes and streams under natural hydrologic conditions. However, tide-control structures installed in 1868 by the State of Massachusetts and the Towns of Provincetown and Truro to restrict the inland movement of saltwater at Pilgrim Lake and the Pamet River (fig. 1) changed these ecosystems from salt to freshwater. The Cape Cod Commission (Eichner and others, 1997) and the National Park Service have studied the effects of tide-control structures on flow in these areas and the possible hydrologic effects of returning these ecosystems to their natural state.

The primary source of freshwater for the Lower Cape Cod aquifer system is precipitation. The weather stations in Provincetown (National Oceanic and Atmospheric Administration, 2001, 1948–92) and South Truro (National Atmospheric Deposition Program Station: MA-01, 1980–2000) report an average rainfall rate of about 42 in/yr from 1948 to 2000. Previous ground-water-modeling investigations on western Cape Cod (Masterson and others, 1998) indicate that, of the 42 in/yr of rainfall, 45 percent is removed by evaporation and plant transpiration before reaching the water table. The remaining freshwater that enters the aquifer system is referred to as aquifer recharge.

Aquifer recharge rates vary from year to year in response to changes in annual precipitation. These rates also may vary from flow lens to flow lens because of differences in land use, vegetation, depth to the water table, and rainfall; however, determining the spatial and temporal changes in aquifer-recharge rates was beyond the scope of this investigation. Thus, for this investigation, it was assumed that, on average, 24 in/yr, or 55 percent of the average annual precipitation, reaches the aquifer system as recharge throughout the study area.

## Ground-Water Recharge Areas

All of the water that enters the aquifer system as recharge ultimately discharges to pumped wells, streams, coastal marshes, and beaches. Some of this water may flow through kettle-hole ponds on its way to these discharge areas. The source of water to these discharge points, or receptors, can be determined by mapping the area at the water table that, multiplied by the recharge rate, satisfies the total flow to the receptor. The concept of the source of water to a hypothetical pumped well is illustrated schematically in figure 6. This concept can be applied to any hydrologic feature that receives ground-water discharge, such as kettle-hole ponds, streams, and coastal areas (Masterson and Walter, 2000). The discharge locations of all water that enters the aquifer system can be determined once the recharge areas to all hydrologic features are delineated (fig. 7).

The size of the recharge areas to various hydrologic features is proportional to the amount of water that discharges to these features when a spatially consistent recharge rate has been applied. For instance, the areas shown on figure 7 that delineate the sources of water to Cape Cod Bay and the Atlantic Ocean represent a large percentage of the total study area. The model-calculated water budget (table 1) shows that nearly 68 percent of the total flow through the aquifer system discharges to the coast as direct ground-water discharge. About 31 percent of the total flow reaches the coast as ground-water-derived streamflow. Pumping from municipal-supply wells captures about 1 percent of the recharge, all of it from the Pamet flow lens.

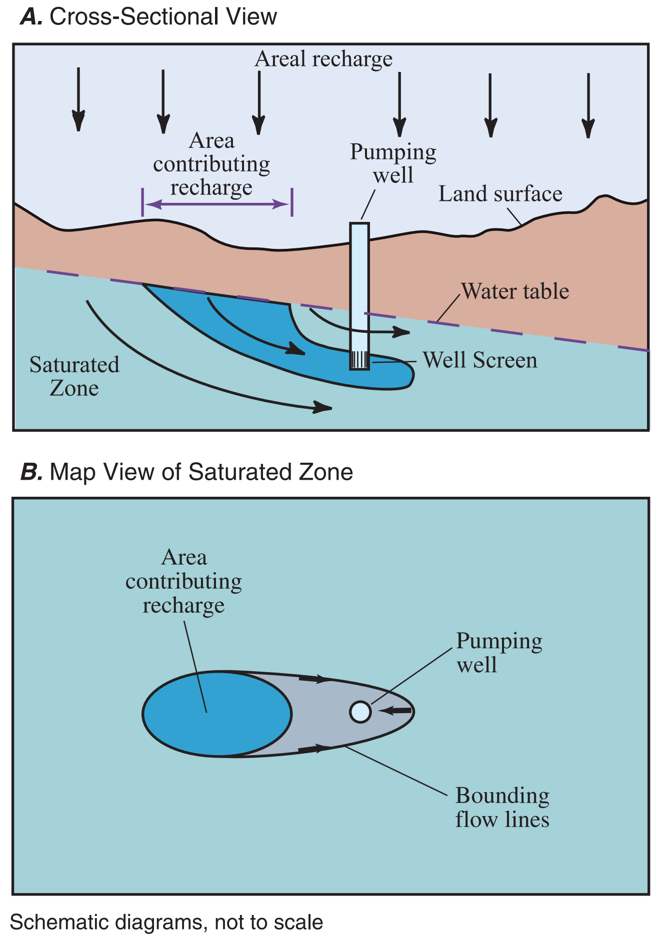



**Figure 6.** Area contributing recharge to a pumping well in a simplified, hypothetical ground-water-flow system (modified from Reilly and Pollock, 1993).

## Water Budget

All of the freshwater that flows through the Lower Cape Cod aquifer system is derived from aquifer recharge. A model-simulated recharge rate of 24 in/yr yields a total freshwater flow through the aquifer system of about 67.3 Mgal/d. An additional 0.8 Mgal/d of water is returned to the aquifer in the Pilgrim flow lens from water pumped in the Pamet flow lens and discharged to the Pilgrim flow lens as wastewater discharge from septic systems. Therefore, the total freshwater flow through the aquifer system is about 68 Mgal/d (table 1).

The total water budget for the Lower Cape Cod aquifer system can be subdivided by individual flow lenses (table 1). Subdividing the water budget by flow lens provides a better understanding of the distribution of flow to the various hydrologic features than can be obtained from the total water budget for the entire aquifer system. For instance, the total amount of ground-water discharge to streams is about 21 Mgal/d, or 31 percent of the total water budget for the Lower Cape Cod aquifer system, yet nearly 60 percent of that streamflow is in the Herring River, which bisects the Chequesset flow lens (fig. 1).

Ground-water withdrawals for public supply only account for about 1 percent of the the total water budget for the entire aquifer system; however, all of that pumping occurs in the Pamet flow lens and that pumping constitutes about 7 percent of the total budget of that flow lens. This pumped water is the primary source of drinking water for the town of Provincetown, parts of the town of Truro, and some National Park Service facilities in the Provincetown area.

A small-capacity municipal supply also is pumping water from the Chequesset flow lens. This well services approximately 30 residences in the Coles Neck area of Wellfleet and pumps on average about 7,000 gal/d. Because this well is pumping at a low rate compared to the total flow in aquifer system, it is not included in the overall water budgets for the Lower Cape Cod aquifer system or the Chequesset flow lens. This is also the case for the domestic wells from which many of the residents of Lower Cape Cod obtain their drinking water because the water pumped from and returned to the same part of the aquifer resulted in no effect on the flow system.

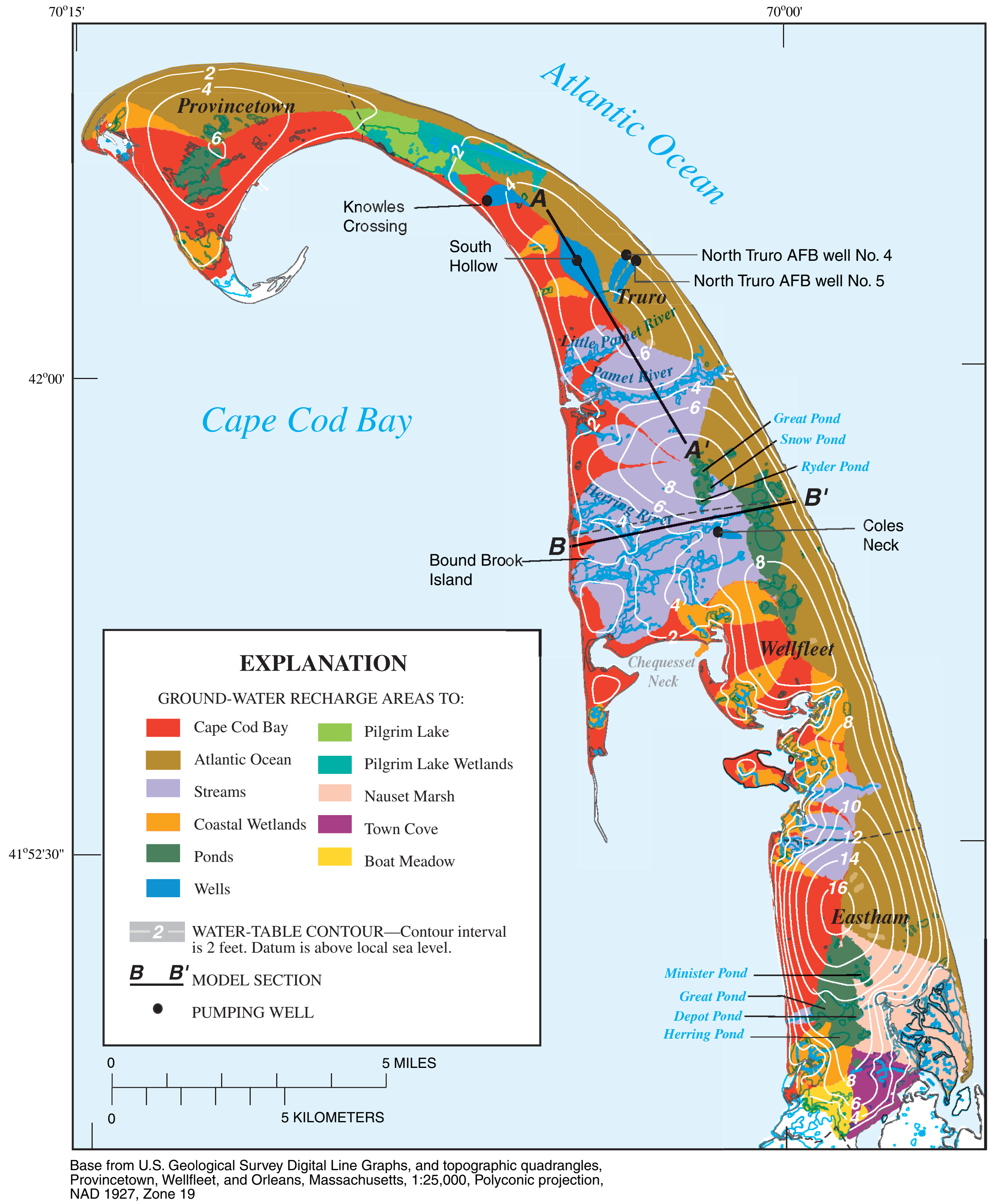



**Figure 7.** Delineation of ground-water-recharge areas to public-supply wells, ponds, streams, and coastal areas for current (2002) average pumping and recharge conditions, Cape Cod, Massachusetts. Sections *A-A′* and *B-B′* shown in figures 9 and 10, respectively.

**Table 1.** Model-calculated hydrologic budget for the four flow lenses of the Lower Cape Cod aquifer system under current (2002) pumping and recharge conditions, Cape Cod, Massachusetts.

[**Inflow:** Consists of recharge from precipitation and wastewater. All values in million gallons per day]

| Flow lens | Inflow | Outflow | | | |
|---|---|---|---|---|---|
| | | Coast | Streams | Wells | Total for each lens |
| Nauset | 19.0 | 14.0 | 5.0 | 0.0 | 19.0 |
| Chequesset | 24.2 | 11.9 | 12.3 | .0 | 24.2 |
| Pamet | 12.4 | 8.8 | 2.7 | .9 | 12.4 |
| Pilgrim | 12.5 | 11.4 | 1.1 | .0 | 12.5 |
| Total | 68.1 | 46.1 | 21.1 | 0.9 | 68.1 |

## Altitude and Configuration of Water-Table Mounds

The altitude and configuration of the water-table mounds in the Lower Cape Cod aquifer system are affected by factors such as surface-water bodies, the geologic framework, and changing pumping and recharge conditions. In general, ground water flows radially from the highest point of the water table toward the coast and the inter-lens discharge areas. Water entering near the center of the water-table mound travels deeper through the aquifer system than water that recharges near the coast (fig. 5).

The altitude of the water-table mounds differ (fig. 1) from about 16 ft above NGVD 29 in the Nauset flow lens to about 6 ft above NGVD 29 in the Pilgrim flow lens. The Chequesset flow lens, which receives the most recharge of the four flow lenses, has a maximum altitude of about 9 ft above NGVD 29; this altitude is about 7 ft lower than that of the Nauset flow lens. The differences among the maximum altitudes and configurations of the flow lenses result from differences in the distribution of surface-water bodies and in the geologic framework.

### Interaction Between Ground and Surface Waters

The water-table altitude and configuration of the Chequesset flow lens is affected more by surface-water discharge than the other flow lenses of the Lower Cape Cod aquifer system. In the Chequesset flow lens the Herring River and its tributaries (fig. 1) occupy a large area in the western part of the flow lens and drain a large percentage of the total flow in the lens toward Wellfleet Harbor as surface-water discharge. The model-calculated streamflow in the Herring River and associated tributaries is about 7.4 Mgal/d, or about 30 percent of the total flow through the Chequesset flow lens.

The Herring River and its tributaries affect the surrounding water table because the altitude of the stream-channel bottom does not change with time. The water table in the vicinity of the streams cannot rise appreciably above the stream channel because, as the water-table altitude increases in response to changing recharge conditions, ground-water discharge to the stream increases. In the absence of streams, increases in recharge would result in a corresponding increase in ground-water levels.

The Chequesset flow lens, which is separated from the Pamet flow lens to the north by the Pamet River and the Nauset flow lens to the south by Blackfish Creek, can be subdivided further. The Herring River complex subdivides the flow lens, creating two large flow lenses and three small flow lenses at Bound Brook Island, Griffin Island, and the area occupied by Wellfleet center (fig. 1). Although these five flow lenses are hydrologically independent of each other under present conditions, they were considered to be part of the larger Chequesset flow lens for the purpose of this investigation.

Kettle-hole ponds also affect the configuration of the water table in the flow lenses. The ponds, like streams, are hydraulically connected to the ground-water-flow system, causing pond levels and streamflows to fluctuate with ground-water levels. Pond levels fluctuate less than surrounding ground-water levels because ponds have substantially larger storage capacities than the aquifer. The ponds of Lower Cape Cod are areas of net recharge because the annual precipitation rate (42 in/yr) exceeds the annual potential evaporation rate from pond surfaces (28 in/yr) (Farnsworth and others, 1982) for average annual flow conditions. For periods of extended droughts, however, substantially more water can be lost from pond surfaces creating ground-water sinks during these extended drought periods. Information on how kettle-hole ponds are simulated in the ground-water-flow model is reported in the appendix.

The recharge areas for kettle-hole ponds are delineated in a manner similar to streams and coastal areas because the upgradient side of a pond acts as a ground-water discharge zone (fig. 7). Unlike the discharge into streams, however, water that discharges to a pond does not necessarily result in net removal of water from the aquifer because the water mixes within the pond and either passes through the downgradient side of the pond and re-enters the aquifer or moves directly into outflowing streams.

In the Lower Cape Cod aquifer system there are clusters of kettle-hole ponds in the center of the Chequesset flow lens and in the lower part of the Nauset flow lens (fig. 7). The pond cluster consisting of Snow, Great, and Ryder Ponds in the center of the Chequesset flow lens has smaller recharge areas than the pond cluster consisting of Minister, Great, Depot, and Herring Ponds in the lower part of the Nauset flow lens (fig. 7). This difference is due, in part, to the size and depths of the ponds, but also to the pond locations with respect to the top of the water-table mound.

Because the ponds in the Chequesset flow lens are near the top of the water-table mound, their contributing areas are small and the total flow through the ponds is low. In comparison, the ponds in the lower part of the Nauset flow have larger contributing areas and higher flow (fig. 7). This difference in the position of the ponds within the flow system may affect how much water moves through the ponds and how the ponds respond to changing stress conditions, such as changing pumping and recharge conditions.

Freshwater ponds also are in the Pilgrim lens but they are of a different origin than the kettle-hole ponds in the Chequesset and Nauset flow lenses. The ponds on the Pilgrim lens were not formed by the collapse of melting buried blocks of ice, but rather from the more recent flooding of wetlands in lowland areas where the land surface has been intercepted by the underlying water table (B.D. Stone, U.S. Geological Survey, oral commun., 2001). Recharge areas to these ponds were delineated in a manner similar to that used to calculate the contributing areas to the kettle-hole ponds. Further investigation, however, would be needed to determine if the pond-bottom sediments in the flooded wetlands differ from the pond-bottom sediments of the kettle-hole ponds. Simulated differences in pond-bottom sediments have been shown to have little effect on model-calculated pond levels, but can have a large effect on the amount of flow through the ponds, and therefore, contributing areas to ponds (Walter and others, 2002).

### Controls of Hydrogeologic Framework

Another explanation for the differences among the water-table altitudes in the flow lenses can be attributed to the differences in the hydrogeologic framework among the flow lenses. For example, the Nauset flow lens, unlike the Chequesset flow lens, consists of thick deposits of low-permeability layers of silt and clay at depth. The presence of this low-permeability material in the Nauset flow lens lowers the overall transmissivity of the aquifer and alters vertical flow paths such that proportionally more water flows through the thinner, overlying, more permeable sediments than the thicker, underlying, less permeable sediments. The focus of flow in the thinner, more permeable sediments results in higher water-table altitudes than would be expected if the more permeable sediments extended deeper into the aquifer.

## Simulated Interaction Between Freshwater- and Saltwater-Flow Systems

Freshwater flow in the Lower Cape Cod aquifer system is bounded below by saltwater rather than truncated by bedrock as is the case on western Cape Cod (Masterson and Barlow, 1996). The reason for the bounding by saltwater is that the flow lenses of Lower Cape Cod are much smaller in size than those of western Cape Cod, the depth to bedrock is greater in general under Lower Cape Cod than western Cape Cod, and because of the smaller land area, less recharge from precipitation is available to extend the freshwater lenses deep enough to intersect bedrock.

Depths to the freshwater/saltwater interface differ among the flow lenses; these depths are directly proportional to altitude of the overlying water table. If the altitude of the water table above sea level ($z_w$) is lowered by 1 ft, the depth to the freshwater/saltwater interface ($z_s$) decreases by 40 ft (Ghyben, 1888; Herzberg, 1901): $z_s = 40z_w$. The Ghyben-Herzberg relation is based on the density difference between fresh and salt waters and is a general approximation, subject to many simplifying assumptions, of the actual interaction between freshwater and saltwater flow. The relation does provide insight, however, into the differences in the depths to the freshwater/saltwater interfaces throughout the Lower Cape Cod aquifer system.

Field measurements show that the depth to the freshwater/saltwater interface is about 350 ft below NGVD 29 at test site EGW-45 in the Nauset flow lens (fig. 1) (Barlow, 1996); in the Pamet flow lens, the altitude of the water table is much lower and the depth to the interface is about 250 ft below NGVD 29 at test site TSW-200 (fig. 1) (LeBlanc and others, 1986).

Because the Ghyben-Herzberg relation is approximate and subject to simplifying assumptions, and the expense of drilling monitoring wells to the freshwater/saltwater interface throughout the study area is high, a numerical model was developed to characterize the position and movement of the freshwater/saltwater interface throughout the Lower Cape Cod aquifer system. The numerical model SEAWAT (Guo and Langevin, 2002) that simulates variable-density, transient ground-water flow in three dimensions was used for this investigation. The model development and calibration is described in detail in the appendix.

The model-calculated three-dimensional representation of the depth to the freshwater/saltwater interface shows how the depth to this interface changes throughout the Lower Cape Cod aquifer system (fig. 8). The depth to the freshwater/saltwater interface is greatest, up to 400 ft below NGVD 29, beneath the tops of the water-table mounds and shallowest at the inter-lens discharge areas.

## Effects of Surface-Water Bodies

A south-north vertical section from the center of the Chequesset flow lens across the Pamet River to the center of the Pamet flow lens (fig. 9) shows that the depth to the interface is about 300 ft below NGVD 29 beneath the center of the Chequesset flow lens. This depth decreases to about 100 ft below NGVD 29 beneath the Pamet River and then increases to the north to about 250 ft below NGVD 29 beneath the center of the Pamet lens, as calculated by the numerical model. The decrease in depth to the freshwater/saltwater interface beneath the Pamet River is a function of the low water-table altitude along the stream reach (about 2 ft above NGVD 29) and the fact that ground-water discharge to the river reaches the coast as surface-water flow and, therefore, is removed from the ground-water-flow system.

Surface-water discharge also affects the position of the freshwater/saltwater interface in the Chequesset flow lens in the vicinity of the Herring River. A simulated west-east vertical section across the Chequesset lens (fig. 10) shows an asymmetric shape to the position of the freshwater/saltwater interface with the depth to the interface greatest beneath the eastern side of the flow lens. This asymmetric geometry of the interface can be explained by the water-table configuration and the distribution of ground-water recharge areas shown on figure 7.

The map of the ground-water recharge areas for the Lower Cape Cod aquifer system (fig. 7) shows that a large area on the west side of the Chequesset flow lens contributes water to the Herring River and its tributaries. This water is lost to the ground-water-flow system because it discharges into the stream channel and is transported directly to Wellfleet Harbor as

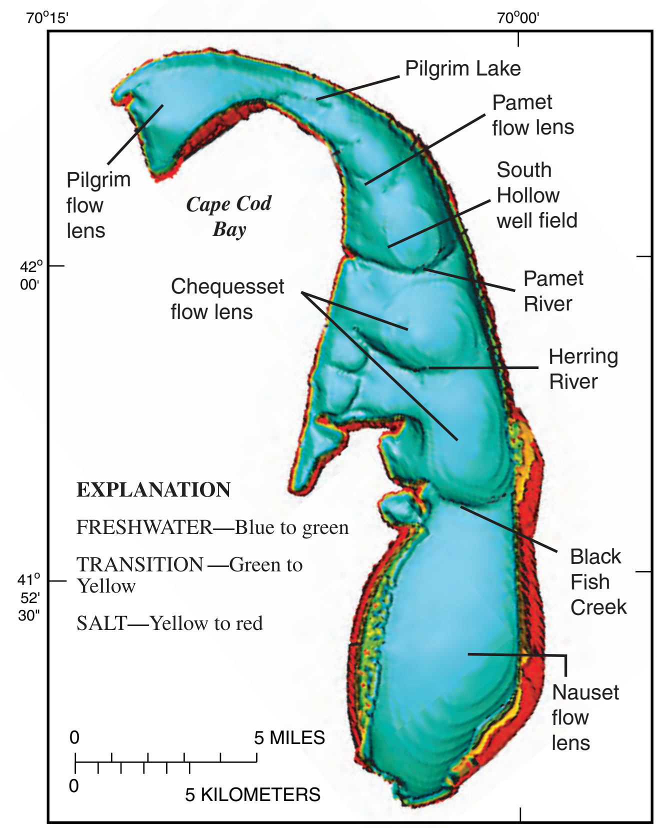



**Figure 8.** Model-calculated delineation of the boundary between freshwater and saltwater beneath the Lower Cape Cod aquifer system, Cape Cod, Massachusetts.

surface-water discharge. On the east side of the Chequesset flow lens, ground water also discharges to surface-water bodies; however, in the case of the kettle-hole ponds, most of the water flows through the ponds, re-enters the aquifer, and then discharges to the Atlantic Ocean as direct ground-water discharge. It is this difference in how the recharge reaches the coast and the effect of streams on the water-table mounds that accounts for the asymmetry in the position of the freshwater/saltwater interface in the Chequesset flow lens.

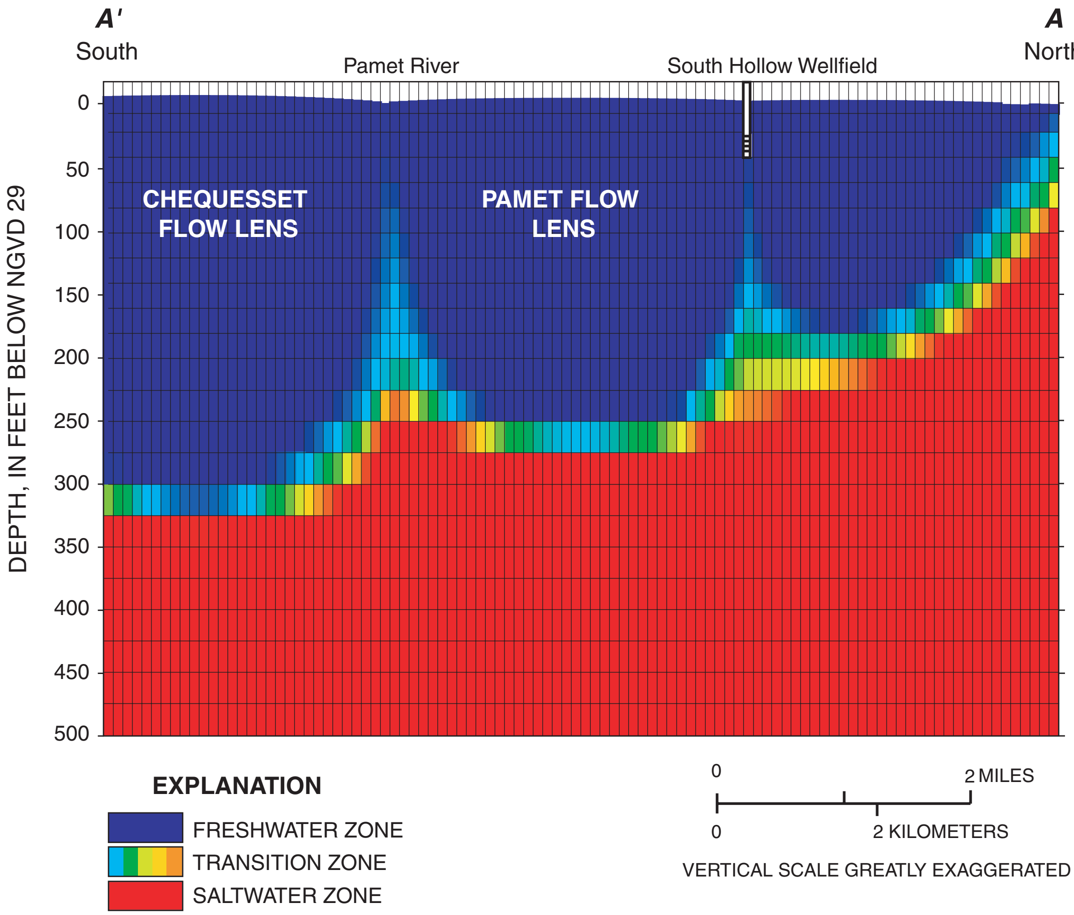



**Figure 9.** Model section showing the model-calculated boundary between freshwater and saltwater flow, Lower Cape Cod, Massachusetts. Section line *A*-*A′* shown on figure 7.

## Effects of Ground-Water Pumping

Ground-water pumping also can affect the position and movement of the freshwater/saltwater interface. Because the flow lenses are bounded laterally and underlain by saltwater, there is concern for the potential of both lateral intrusion and upconing of saltwater from pumping. A pumping well near the coast can draw saltwater in laterally and a pumping well in the center of the peninsula can pull saltwater up from below (fig. 11). Saltwater intrusion is of greater concern in the Pamet flow lens than the other lenses because nearly all of the pumping for public supply in Lower Cape Cod takes place in the Pamet flow lens.

Within the Pamet flow lens are three active well fields; Knowles Crossing, South Hollow, and the North Truro Air Force Base (fig. 12). These well fields are the primary water supply for the residents of Provincetown, parts of North Truro, and for various National Park Service facilities on the Cape Cod National Seashore. The South Hollow well field (also known as the Paul Daley well field) has been operating since 1955 and consists of eight pumping wells that currently (2002) provide, on average, 0.57 Mgal/d of water. The Knowles Crossing well field (also known as the Old Truro well field) is the oldest well field on Cape Cod and has been operating since 1907. This well field consists of two pumping wells that currently (2002) provide, on average, 0.20 Mgal/d.

The third well field in the Pamet flow lens consists of two pumping wells (No. 4 and 5) on the site of the decommissioned North Truro Air Force Base (NTAFB), which is now part of the Cape Cod National Seashore. This well field was part of the water-supply system for the North Truro Air Force Base and, since 1978, has provided water to Provincetown to help the town meet its drinking-water demand. Currently, water is being withdrawn from these wells during a 6-month period from June through November for an average daily pumping rate of 0.16 Mgal/d for the year.

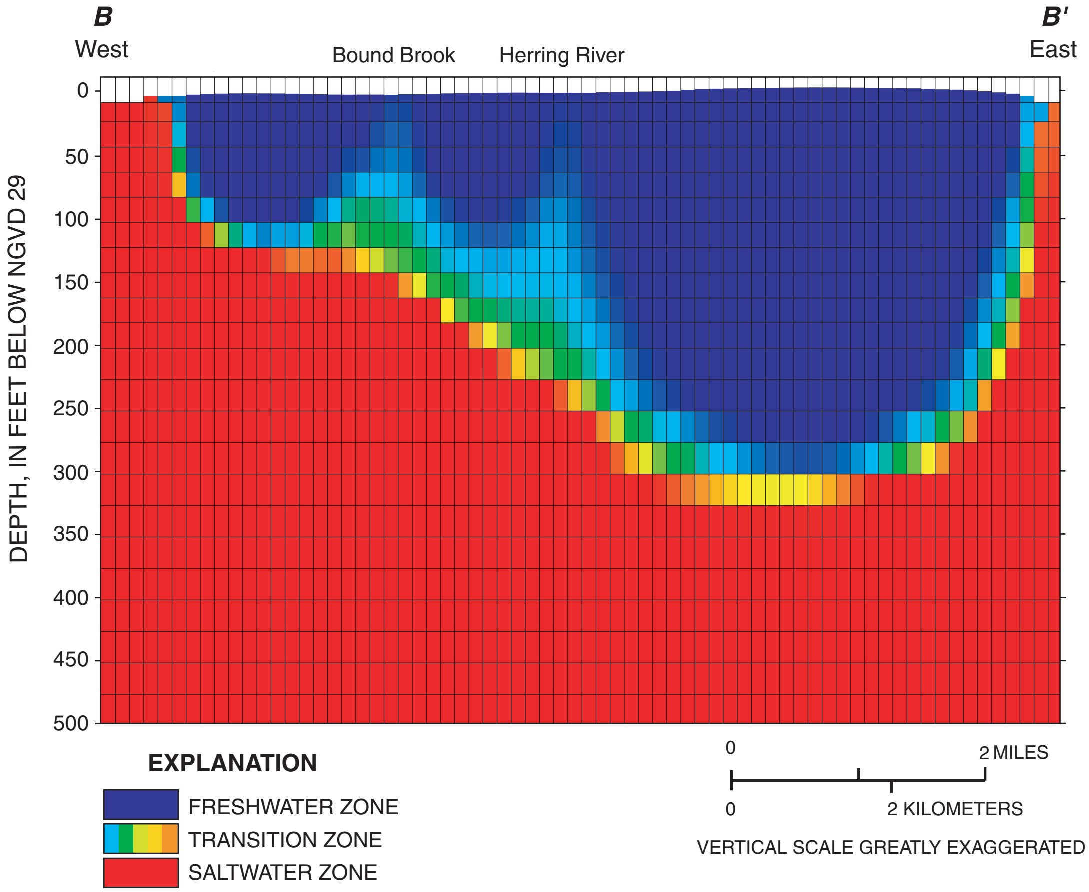



**Figure 10.** Model section showing the model-calculated boundary between freshwater and saltwater flow, Lower Cape Cod, Massachusetts. Section line *B-B′* shown on figure 7.

A fourth well field consisting of one well, Cape Cod National Seashore Site No. 4 (CCNS No. 4, fig. 12), was pumped seasonally from 1978 to 1985 to provide additional drinking water to Provincetown while pumping was reduced at the South Hollow well field. CCNS No. 4 was pumped from May to October for an average daily pumping rate of 0.25 Mgal/d for the year. Information on pumping rates and pumping duration for all of the pumping wells in the Pamet flow lens are detailed in Provincetown's water-management-planning report (Environmental Partners Group, 2002).

The threat of saltwater intrusion at the Knowles Crossing well field, which is about 1,000 ft inland of Cape Cod Bay, has concerned Provincetown water suppliers for many years. Until 1955, this well field was the primary source of water for the residents of Provincetown. As population increased, the demand for drinking water increased; the average daily withdrawal rate from this well field increased from about 0.16 Mgal/d in 1907 to 0.49 Mgal/d in 1954. This increase in water demand resulted in saltwater intrusion at the well field and a subsequent reduction to the current (2002) daily rate of 0.20 Mgal/d.

In 2000, the town of Provincetown began installing a number of deep monitoring wells at the three well fields to determine the depths of the freshwater/saltwater interface (Environmental Partners Group, 2002). Three deep monitoring wells were installed alongside the three pumping wells at Knowles Crossing. As part of this investigation, borehole geophysical electromagnetic (EM) and gamma logs were collected for the purpose of determining the depth to the freshwater/saltwater interface and for monitoring the interface position for seasonal changes.

The EM conductivity logs collected at the Knowles Crossing well field show that the depth to the transition between fresh and salt waters increases from about 60 ft below NGVD 29 at the westernmost well (KC-1) to about 100 ft below NGVD 29 at the easternmost well (KC-3) (fig. 13). The natural gamma logs (fig. 13) show the presence of fine-grained sediments that were identified by the driller to be clay (Environmental Partners Group, 2002). The clay lenses appear to have affected the position of the freshwater/saltwater interface beneath the well field such that saltwater has laterally

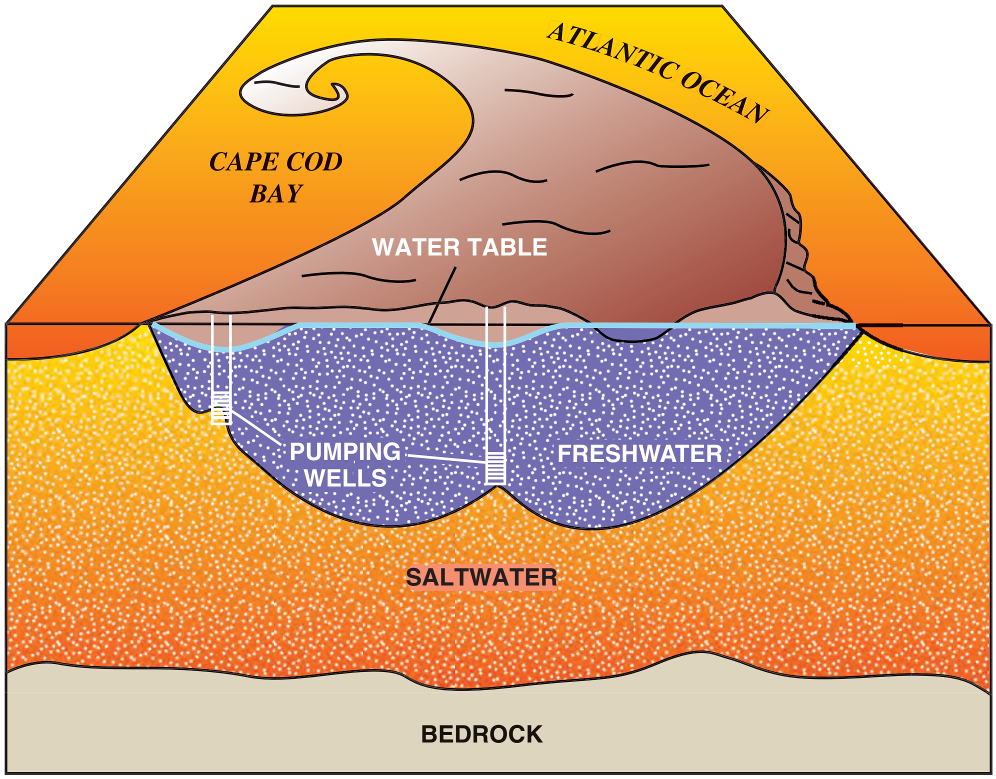


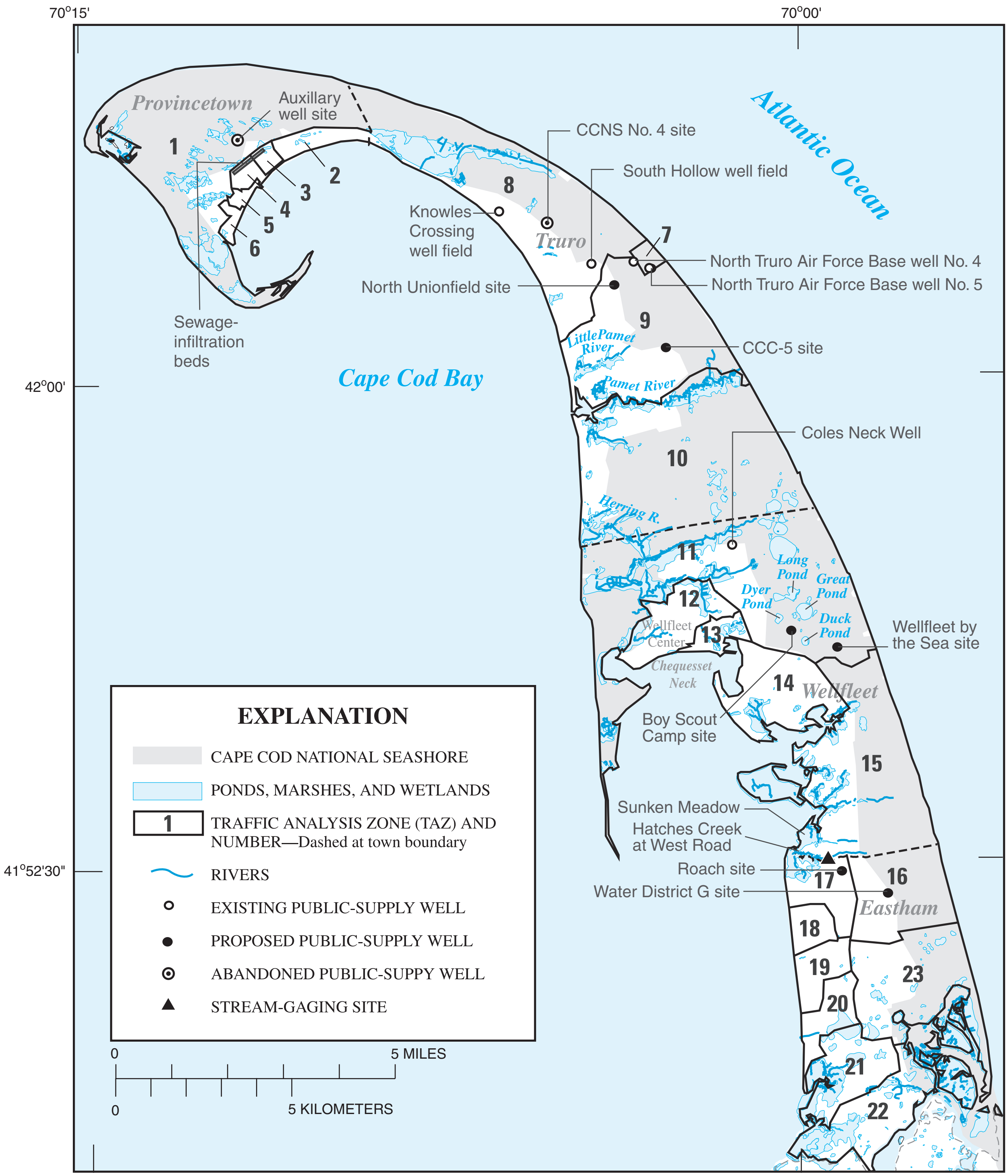

**Figure 11.** Schematic diagram of the Lower Cape Cod aquifer system showing lateral and vertical saltwater intrusion in response to ground-water pumping (modified from Strahler, 1972).

intruded above the clay lens at about 70 ft beneath pumping wells KC-1 and KC-2. This intrusion, based on the available geophysical data, is shown schematically on figure 14.

In January 2001, the USGS, in cooperation with the town of Provincetown, installed an automated, real time, water-quality-sampling system, known as "Robowell" (Granato and Smith, 2002), in two monitoring wells above and below the clay lens near the KC-2 pumping well. Results of this real-time sampling show that the specific conductance of water in the shallow well (TSW-260) was nearly twice that in the deep well (TSW-259) and that the specific conductance of water in the shallow well showed a greater response to changes in pumping and recharge than that of the deeper well (fig. 15). This information indicates that the source of saltwater intrusion at the Knowles Crossing well field was most likely the result of lateral encroachment along the top of the clay layer rather than upconing beneath the well field. In response to this information, Provincetown replaced their three-well vacuum system in 2001 with a two-well submersible system at the locations of KC-2 and KC-3 and removed KC-1, the pumping well nearest to the coast (Environmental Partners Group, 2002).

With water demand increasing and the total yield at Knowles Crossing limited by saltwater intrusion, Provincetown developed a new public-supply site in 1955 at South Hollow (fig. 12). The South Hollow well field served as the primary water-supply source for Provincetown until 1978 when gasoline was spilled in the vicinity of the well field (Environmental Partners Group, Inc., 2002). As a precautionary measure, the town limited withdrawals at the well field to avoid pumping contaminated water. This well field was either not pumped or pumped at a reduced rate from 1978 to 1985. Pumping resumed at full capacity by 1986 and currently (2002) averages 0.57 Mgal/d.

The ground-water model was used to simulate annual average pumping at the South Hollow well field from 1955 to present (2002) and calculate the change in the position of the freshwater/saltwater interface with time (fig. 16). The results indicate that, before the start of pumping in 1955, the depth to the freshwater/saltwater interface beneath the South Hollow well field was about 230 ft below NGVD 29 and that the interface has risen about 43 ft to its current position (2002) at about 187 ft below NGVD 29.




Base from U.S. Geological Survey Digital Line Graphs, and topographic quadrangles, Provincetown, Wellfleet, and Orleans, Massachusetts, 1:25,000, Polyconic projection, NAD 1927, Zone 19

**Figure 12.** Locations of existing (2002) and proposed public-supply wells and Traffic Analysis Zones, Lower Cape Cod, Massachusetts.

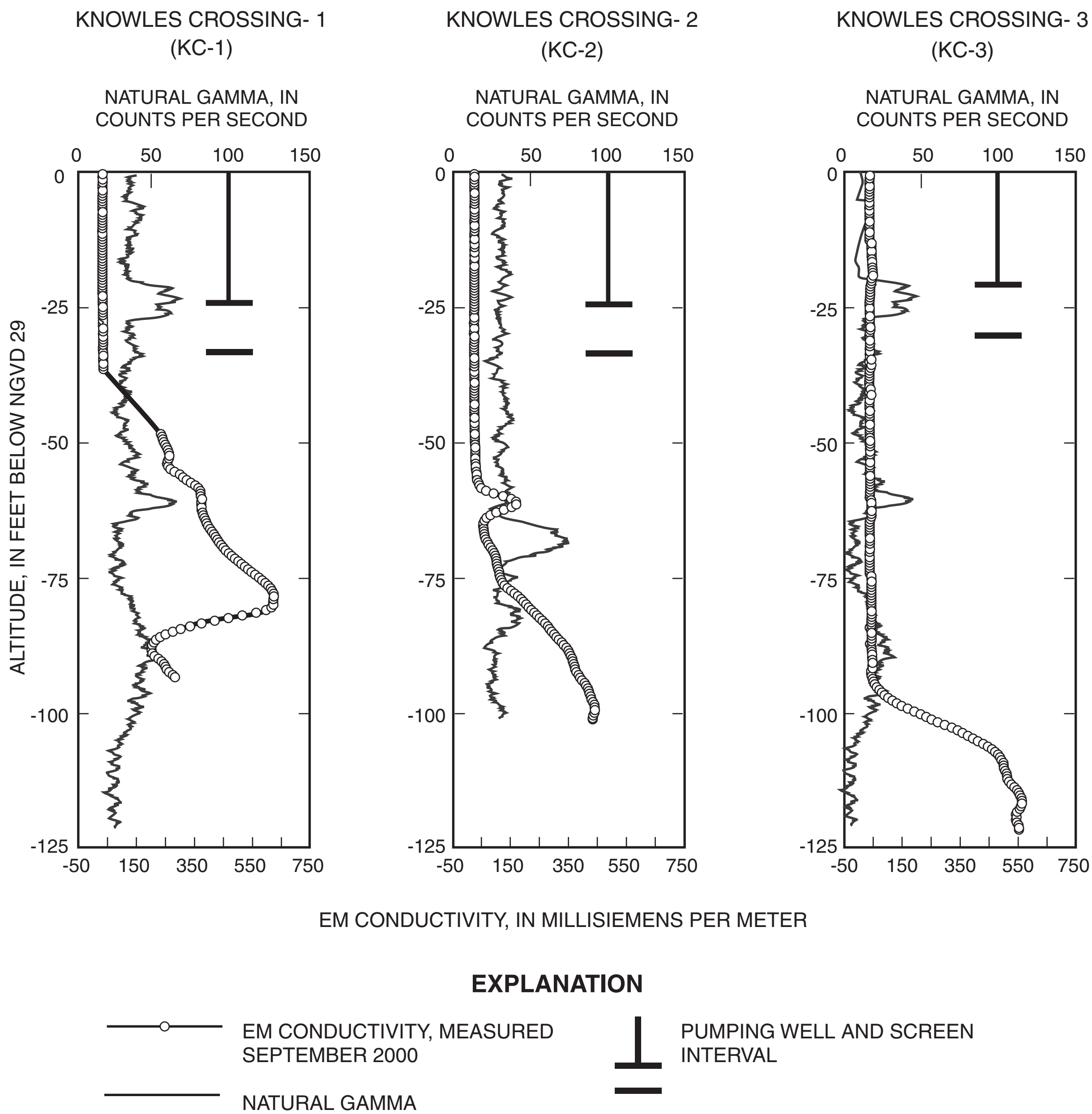



**Figure 13.** Profiles of natural gamma and electromagnetic (EM) geophysical logs at the Knowles Crossing well field, North Truro, Massachusetts, measured in September 2000.

The actual depth to the freshwater/saltwater interface in 1955, before the start of pumping, is not known, and, therefore, this depth cannot be used to determine the accuracy of the model-calculated position of the freshwater/saltwater interface for prepumping conditions. Field measurements of the freshwater/saltwater interface position were made in 2001 as part of the deep-monitoring-well-installation and sampling program and provide data on the current position of the freshwater/saltwater interface beneath the Knowles Crossing, South Hollow, and North Truro Air Force Base well fields.

At the South Hollow well field near the center of the Pamet flow lens (fig. 12), three deep monitoring wells were installed at both ends and at the middle of the well field, which consists of eight pumping wells. The EM log collected in the well at the westernmost end of the well field (SH-1) indicates a thick transition zone from about 85 ft to 175 ft below NGVD 29 (fig. 17). Conversely, the EM log collected in the well at the easternmost end of the well field (SH-3) indicates a thin transition from 165 ft to 180 ft below NGVD 29 (fig. 17).

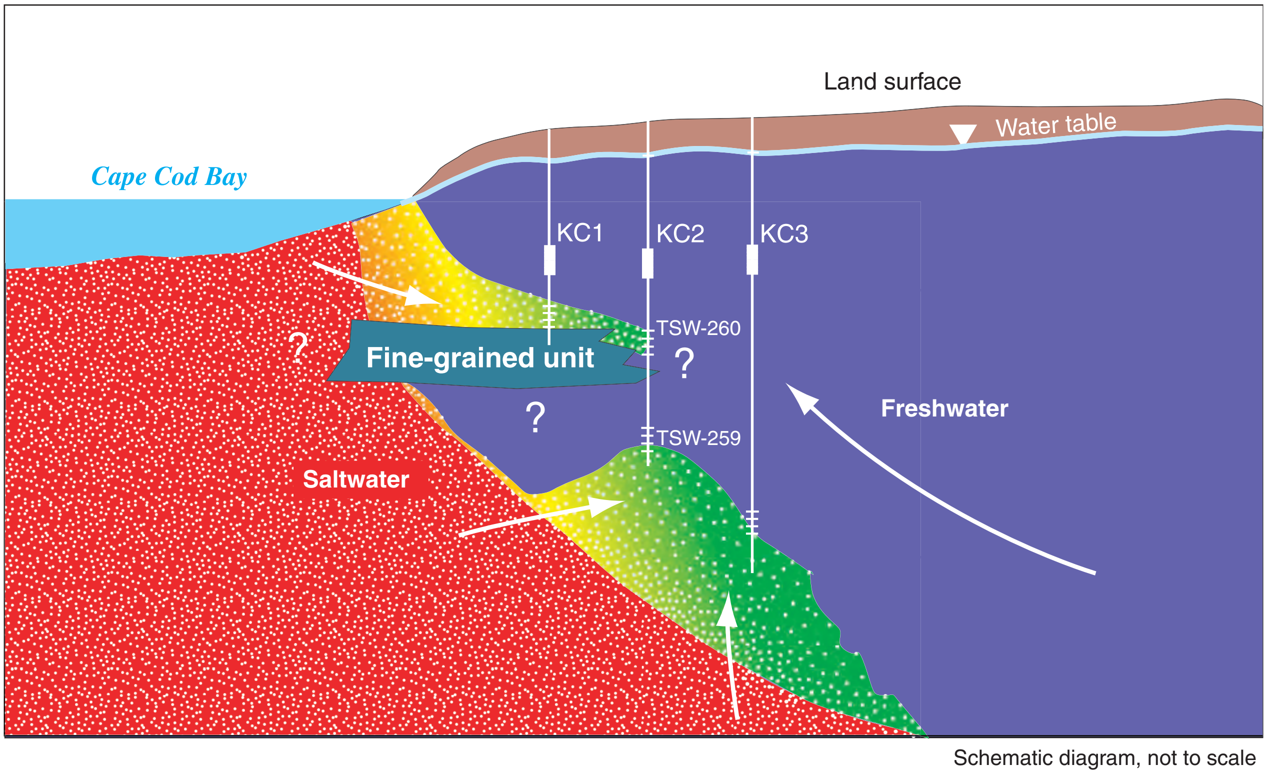



**Figure 14.** Schematic diagram showing lateral and vertical saltwater intrusion beneath the Knowles Crossing well field, North Truro, Massachusetts.

Possible explanations for the thick transition zone at SH-1 include the following. (1) The proximity of this well to Route 6 may have resulted in road salt contamination. (2) The vacuum system at this well field in operation from 1955 to 2000 may have resulted in proportionally more water pumped from the westernmost well than the other wells; this additional withdrawal of water may have caused the freshwater/saltwater interface to rise more in this part of the well field. (3) The well field is oriented within the flow lens such that the western end is closer to the coast than the eastern end and, therefore, the transition zone on the west side of the well field may be thicker and shallower than the east side.

The model-calculated position of the freshwater/saltwater interface in the three model cells representing the South Hollow well field is about 187 ft below NGVD 29 for current (2002) conditions (fig. 17), which is about 7 to 12 ft below the bottoms of the transition zones observed in monitoring wells SH-1 and SH-3. Although the model-calculated freshwater/saltwater interface position is slightly deeper than the field data indicate, the model-calculated change in this position with time (fig. 16) provides insight into how the freshwater/saltwater interface changes in response to pumping conditions. For example, at the South Hollow well field, the interface rises with time as pumping increases from 1955 to 1978 (fig. 16). In 1978, when pumping was reduced in response to the gasoline spill near the well field, the rise in the interface position stopped and the altitude of the interface began to decrease. By 1979, the pumping at the well field had stopped completely. The interface position did not return to its prepumping position, however, indicating a possible temporal lag in the reponse to the decrease in pumping (fig. 16).

Other reasons for the lag in the response include that the pumping shortfall at South Hollow from 1978 to 1984 was compensated for by pumping at CCNS No. 4 (fig. 12) and the NTAFB wells. Pumping from these wells may affect the position of the interface beneath South Hollow regionally—an effect that was not present in 1955. Another possible reason is an overall thinning of the Pamet flow lens because of the effects of sea level rise with time.

Simulation results (fig. 16) also show that if the current pumping rate of 0.57 Mgal/d were maintained for 48 years, the interface position would rise by an additional 10 ft by year 2050. These results indicate that the freshwater/saltwater interface has not yet reached equilibrium with respect to the current pumping. Alternatively, the fact that the model-calculated freshwater/saltwater interface position has not reached equilibrium with respect to current pumping rates may be an artifact of the model representation of the aquifer system,

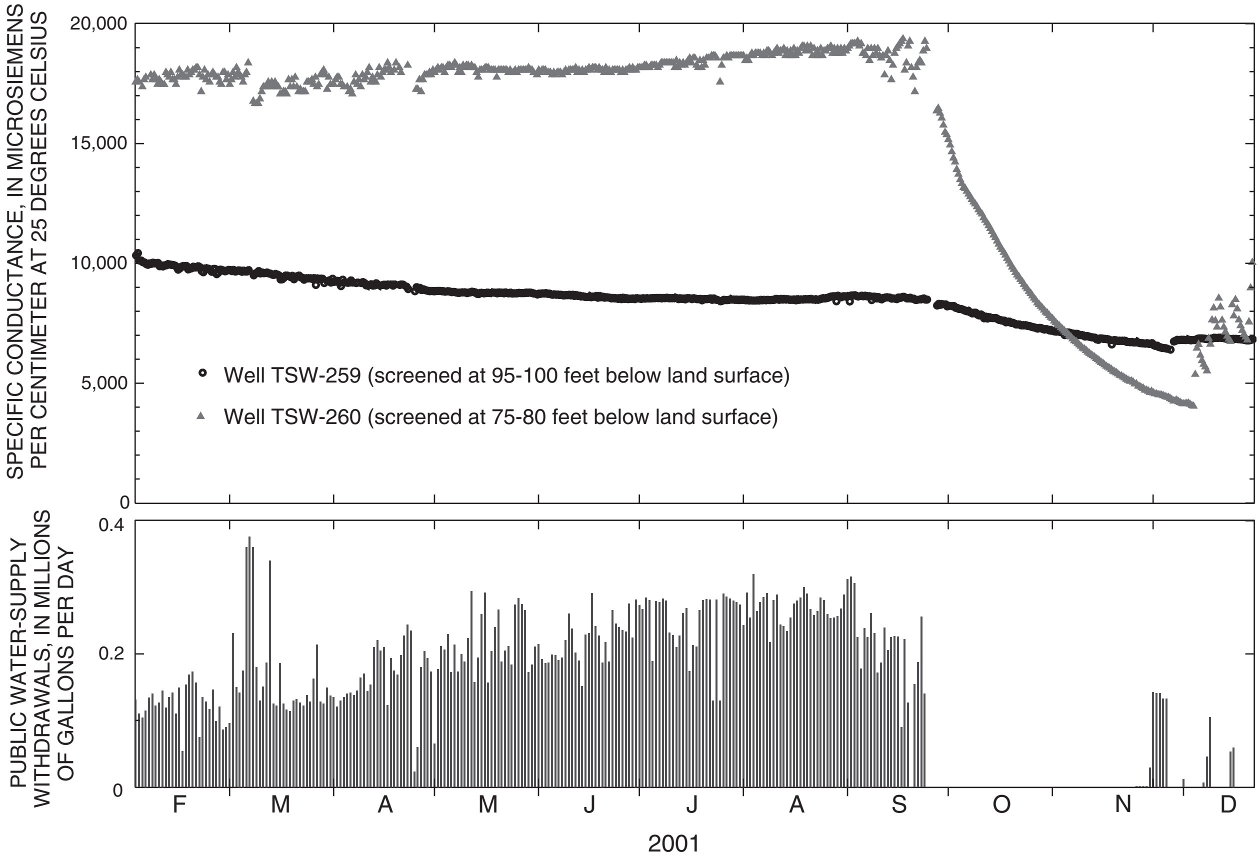



**Figure 15.** Specific conductance in monitoring wells TSW-259 and TSW-260 beneath Knowles Crossing well number 2 (KC-2) and total pumping in 2001 at the Knowles Crossing well field, North Truro, Massachusetts.

such that the simulated hydraulic properties, model-boundary conditions, and (or) the simulation of average annual pumping and recharge conditions may result in a simulated response that does not represent the aquifer system accurately. Given the lack of long-term data on the previous position and movement of the interface, however, it is not possible to assess the model-simulation accuracy with respect to future changes in the interface position.

In addition to the effects of annual average pumping rates and constant recharge rates, the possible effects of seasonal changes in pumping and recharge on the position of the freshwater/saltwater interface in the vicinity of the three pumped well fields in the Pamet flow lens were simulated. The three well fields provide different pumping conditions for which a possible seasonal effect can be evaluated. The South Hollow well field is near the center of the flow lens, and has the highest pumping rates of the three well fields: 0.9 Mgal/d in-season (May–September) and 0.34 Mgal/d off-season (October–April). The Knowles Crossing well field is located nearest to the coast (about 1,000 ft from Cape Cod Bay) and has a pumping rate of 0.31 Mgal/d in-season and 0.12 Mgal/d off-season. The North Truro Air Force Base wells are near the Atlantic Ocean and are pumped only in-season at 0.33 Mgal/d. Pumping rates for the wells were determined on the basis of historical seasonal usage apportioned from the current average daily pumping rates of 0.57 Mgal/d for South Hollow, 0.2 Mgal/d for Knowles Crossing, and 0.16 Mgal/d for the North Truro Air Force Base wells.

The effects of 5 months of low recharge and high pumping from May through September (in-season) and 7 months of high recharge and low pumping from October through April (off-season) on the position of the freshwater/saltwater interface were simulated. The annual average recharge rate of 24 in/yr was apportioned into 18 in. for the 7-month off-season period and 6 in. for the 5-month in-season period. A detailed discussion of the seasonal-varying model simulations is provided in the appendix.

The results indicate that although water levels changed in response to changing stress conditions, there was no appreciable change in the position of the freshwater/saltwater interface between in- and off-season-pumping and recharge-rate changes. This result was the same for each of the three well fields despite the different hydrologic conditions at each of the sites.

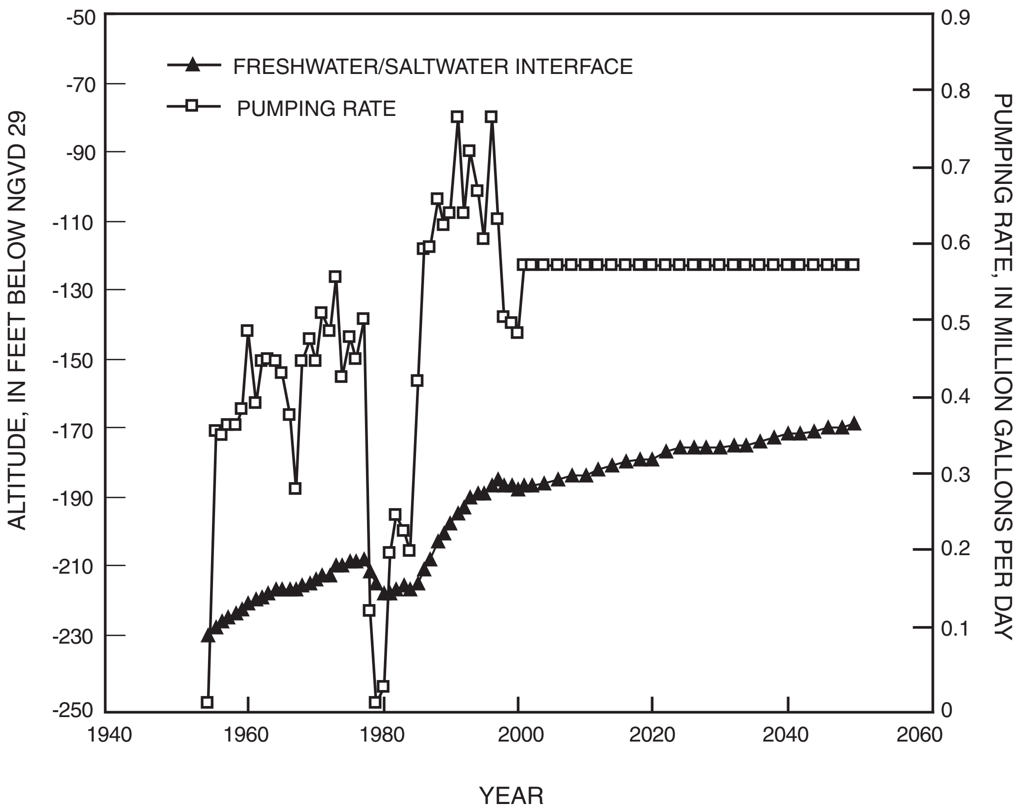



**Figure 16.** Model-calculated freshwater/saltwater interface and simulated pumping from 1955–2050 at the South Hollow well field, North Truro, Massachusetts.

Generally, it is assumed that the potential for vertical movement of saltwater beneath the well fields would increase as the salt concentrations decrease near the top of the transition zone (Reilly and Goodman, 1985). Reilly and others (1987) observed annual changes in salt concentration from 3 to 10 percent at a monitoring well beneath CCNS No. 4, a temporary supply well on the Cape Cod National Seashore that was pumped seasonally from 1978 to 1985 (fig. 12).

Model-calculated salt concentrations indicated a transition zone that is much thicker than that observed in the aquifer (fig. 17). This discrepancy of model-calculated low saltwater concentration in areas known to be freshwater is caused by numerical dispersion, or model uncertainty, which is an artifact of the coarse grid and layer spacing used in the regional model. For the purpose of this investigation the model-calculated 50-percent salt concentration (fig. 17) is assumed to be representative of the freshwater/saltwater interface. A more local-scale, subregional analysis would be required to minimize the effect of numerical dispersion and to simulate the actual distribution of salt concentrations in the aquifer explicitly for changing pumping and recharge conditions.

The absence of an appreciable seasonal change in the freshwater/saltwater interface position calculated by the regional model was confirmed by borehole EM data collected in September 2000, January 2001, May 2001, and December 2001 at the well fields. The EM data (not shown) indicate that the freshwater/saltwater interface responds slowly to changing stress conditions and the time required to adjust to dynamic equilibrium is greater than the time interval of the seasonal cycle.

A reduction in recharge was simulated for current pumping conditions to determine whether the freshwater/saltwater interface position could be affected by a prolonged drought condition. For this analysis, the annual average recharge rate of 24 in/yr was reduced by 30 percent for a 3-year period. This analysis was performed at both the South Hollow and Knowles Crossing well fields. The objective of the simulation was to determine if the depth to the freshwater/saltwater interface could be affected by the change in simulated recharge.

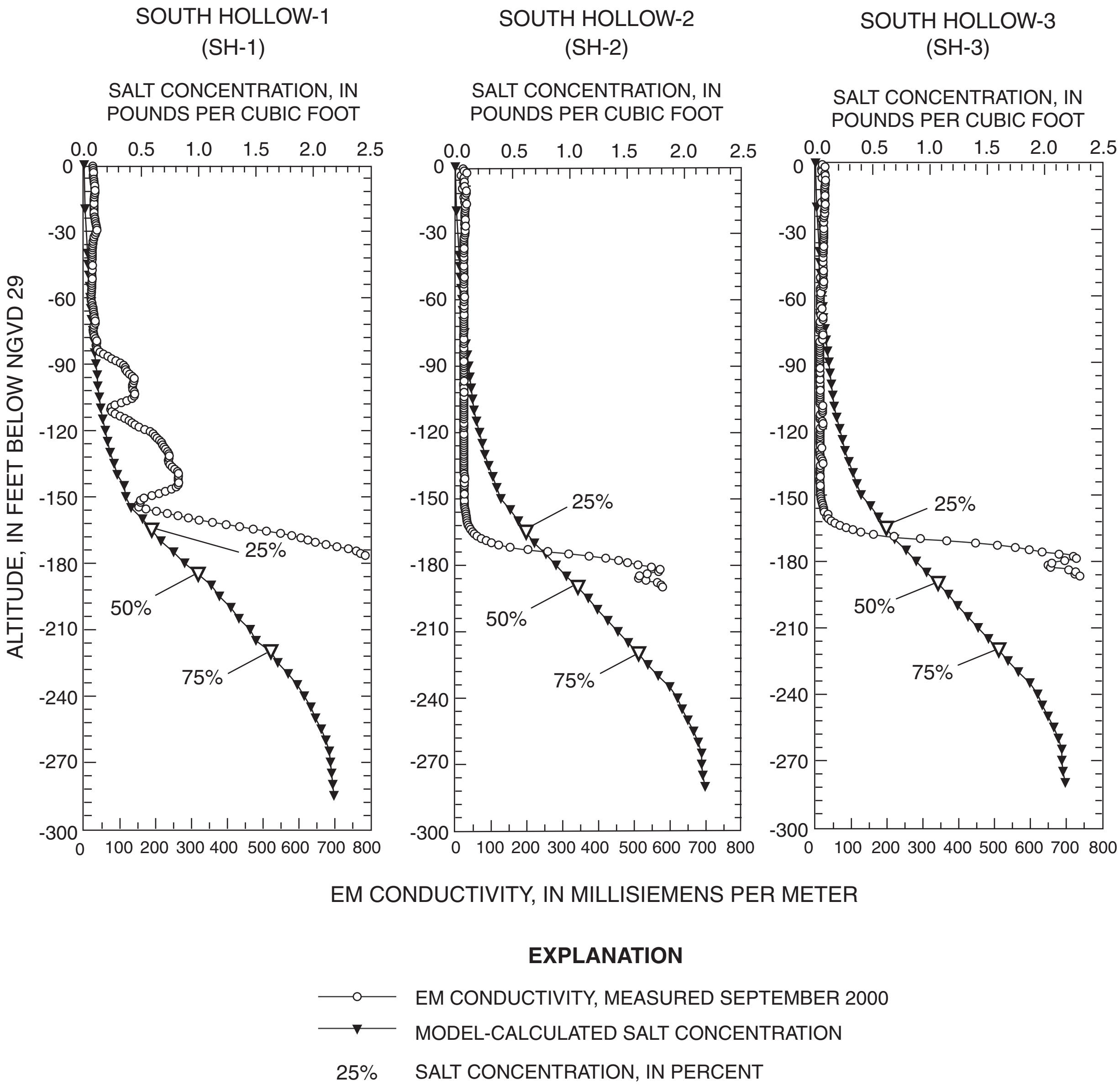



**Figure 17.** Profiles of electromagnetic (EM) logs measured in September 2000 and model-calculated changes in salt concentration for current (2002) conditions at the South Hollow well field, North Truro, Massachusetts.

The results of this simulation indicate that unlike water levels, the position of the interface responds much more slowly to the change in recharge; even with a prolonged drought condition, there was little effect on the simulated interface position at both well fields (results not shown). As is the case with seasonal fluctuations, it is possible that the zone of lower salt concentrations near the top of the transition zone and closer to the well screen may be affected more than the zone of 50-percent salt concentrations; however, this hypothesis could not be tested with the regional flow model because of the limitations imposed by the numerical dispersion that is associated with the dimensions of the model grid.

## Effects of Sea-Level Rise

Residents of coastal areas are becoming increasingly concerned about the effects of sea-level rise. The National Oceanographic Atmospheric and Administration (2003) reports a rising trend in sea level at the Boston Harbor Tidal Gage, which has been in operation since 1921, of about 0.104 in/yr (2.65 ± 0.1 mm/yr) or about 0.87 ft/100 years. The Intergovernmental Panel on Climate Change (IPCC) predicts that global seas may rise by an additional 0.5 to 3.1 ft by 2100, with a best estimate of 1.6 ft (Intergovernmental Panel on Climate Change, 2001). This rate of rise would be nearly double the rate of rise observed at Boston Harbor over the past 80 years. The primary concerns about the possible effects of sea-level rise include future higher rates of erosion than present, damage from higher storm-surge flooding (Theiler and Hammar-Klose, 2000), and landward intrusion of seawater in coastal marshes and wetlands (Donelly and Bertness, 2001).

Rising sea levels also may affect coastal aquifers, such as those of Cape Cod (Intergovernmental Panel on Climate Change, 2001; Moore and others, 1997). Nuttle and Portnoy (1992) have speculated that increases in sea level may result in higher water levels in the tidally affected streams and wetlands of Lower Cape Cod, which could affect ground-water discharge to coastal areas. An analysis of the long-term change in water level at a USGS observation well, about 1,000 ft from Cape Cod Bay near the Knowles Crossing well field, indicates an increase of about 0.1 in/yr (2.1 mm/yr) from 1950–2001 (fig. 18) (McCobb and Weiskel, 2003). This water-level trend is consistent with the trend observed at Boston Harbor, which is about 50 mi northwest of North Truro (National Oceanographic Atmospheric and Administration, 2003).

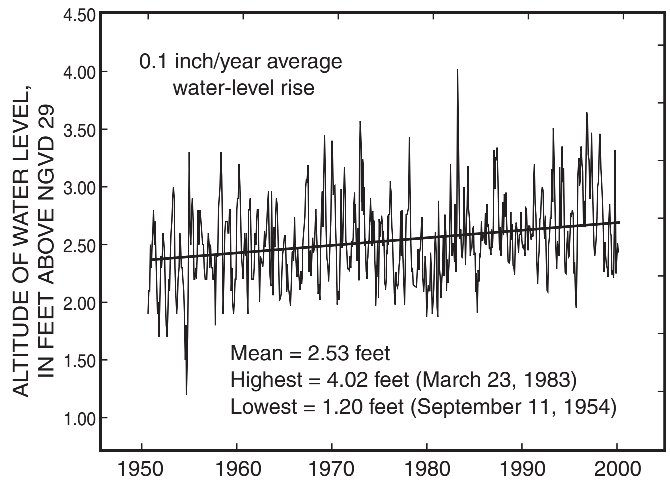



**Figure 18.** Water-table altitude at observation well TSW-1, North Truro, Massachusetts, 1950–2002 (from McCobb and Weiskel, 2003).

## Water Levels and Streamflows

Analysis of the change in water levels at the long-term monitoring wells in the study area shows varying rates of change (fig. 19). In general, the rate of water-level rise appears to be greatest in the observation wells in the Pamet and Pilgrim flow lenses, near the coast and far from any non-tidal surface-water bodies. Factors that could affect these trends, in addition to proximity to tidal waters, are proximity to pumping wells and the period of record over which the wells were measured.

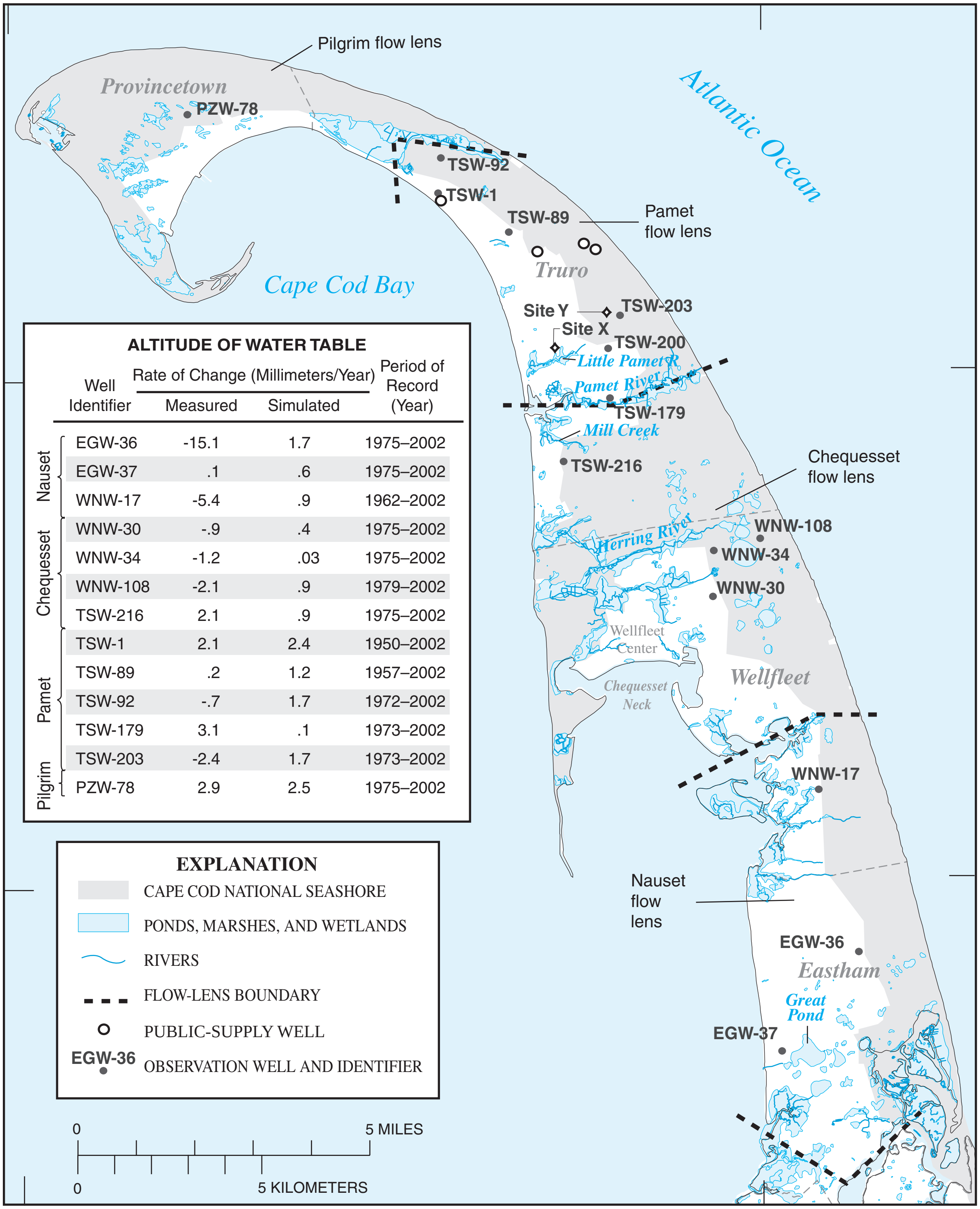
In response to these observed trends in sea-level rise and subsequent water-table rises, an annual increase of 0.104 in/yr (2.65 mm/yr) was simulated in the ground-water model from 0.0 ft in 1929 to 0.63 ft in 2002. The model then was used to assess the effects of sea-level rise on the water-table configuration, coastal discharge, and the position and movement of the freshwater/saltwater interface.

The results of the simulation compare favorably to data for two of the long-term ground-water observation sites in the Pamet (TSW-1) and Pilgrim (PZW-78) flow lenses (fig. 19). The results also indicate a water-table rise much less than the 0.104 in/yr (2.65 mm/yr) at locations near non-tidal surface-water bodies, such as throughout the Chequesset lens, which is dominated by the Herring River and numerous kettle-hole ponds. At the observation sites within the Chequesset flow lens (WNW-30, WNW-34, and WNW-108), the model-calculated changes are substantially less than those in the Pamet and Pilgrim flow lenses, but are greater than the observed negative changes (fig. 19). The difference between the simulated and observed levels may be the result of a complexity in the interactions between ground and surface waters not simulated in the flow model. Another example is in the Nauset flow lens at EGW-37, where the observed trend is 0.0043 in/yr (0.1 mm/yr) as compared to 0.022 in/yr (0.55 mm/yr) calculated by the model. This difference could be attributed, in part, to the artificial pond-level control at nearby Great Pond; this control is not represented in the ground-water model.

The model-calculated changes in water-table altitude at TSW-179 and TSW-216 are less than the observed changes at these wells (fig. 19). These differences may be the result of tidal effects that extend farther inland at Pamet River and Mill Creek than the model-simulated tidal effects.

Some other sites, such as WNW-17 and EGW-36 in the Nauset flow lens, show large declines in water-table altitudes with time rather than the rises calculated by the model. This discrepancy could be explained by local effects that are not represented in the flow model, such as ground-water pumping for water supply or irrigation at the National Park Service Headquarters near WNW-17 and the Nauset Regional High School near EGW-36.

The general trend in both the simulated and observed water-table altitude is that, for the most part, water levels are increasing with time in response to rising sea level, and the magnitude of the increase appears to be affected by the proximity to non-tidal surface-water bodies. The flow model was used to assess the regional effects of simulated sea-level rise and the local effects at or near pumping wells.




**ALTITUDE OF WATER TABLE**

| Flow lens | Well Identifier | Rate of Change (Millimeters/Year) Measured | Rate of Change (Millimeters/Year) Simulated | Period of Record (Year) |
|---|---|---|---|---|
| Nauset | EGW-36 | -15.1 | 1.7 | 1975–2002 |
| | EGW-37 | .1 | .6 | 1975–2002 |
| | WNW-17 | -5.4 | .9 | 1962–2002 |
| Chequesset | WNW-30 | -.9 | .4 | 1975–2002 |
| | WNW-34 | -1.2 | .03 | 1975–2002 |
| | WNW-108 | -2.1 | .9 | 1979–2002 |
| | TSW-216 | 2.1 | .9 | 1975–2002 |
| Pamet | TSW-1 | 2.1 | 2.4 | 1950–2002 |
| | TSW-89 | .2 | 1.2 | 1957–2002 |
| | TSW-92 | -.7 | 1.7 | 1972–2002 |
| | TSW-179 | 3.1 | .1 | 1973–2002 |
| | TSW-203 | -2.4 | 1.7 | 1973–2002 |
| Pilgrim | PZW-78 | 2.9 | 2.5 | 1975–2002 |

Base from U.S. Geological Survey Digital Line Graphs, and topographic quadrangles, Provincetown, Wellfleet, and Orleans, Massachusetts, 1:25,000, Polyconic projection, NAD 1927, Zone 19

**Figure 19.** Locations of long-term observation wells and the measured and model-calculated increase in the altitude of the water table with time, Lower Cape Cod, Massachusetts.

The regional effects of sea-level rise on the Pamet flow lens were assessed by examining the model-calculated change in water levels and position of the freshwater/saltwater interface at two locations from 1929 to 2002 (current) and from 2002 to 2050. Site X (fig. 19) is about 3,000 ft from Cape Cod Bay and adjacent to the Little Pamet River. Site Y is about 3,000 ft from the Atlantic Ocean, near well TSW-203 (fig. 19), and far from any non-tidal surface-water bodies. The model-calculated depth to the freshwater/saltwater interface and the simulated subsurface lithology are similar beneath both sites.

Results of the model simulations indicate that from 1929 to 2002 the water-table altitude at site X increased by 0.15 ft (fig. 20*A*). This increase indicates that, at this location (site X), the water table only has risen by about 0.02 in/yr (0.62 mm/yr), or about 24 percent of the 0.63 ft of regional sea-level rise. At site Y the model-calculated increase of 0.43 ft yields a water-table rise of about 0.07 in/yr (1.78 mm/yr), or about 68 percent of the 0.63 ft of regional sea-level rise (fig. 20*A*). Because the local sea-level position is simulated to be 0.63 ft higher in 2002 than it was in 1929, the increase in water level at site X of 0.15 ft relative to NGVD 29 actually is a net decline of about 0.48 ft relative to the increased sea level (fig. 20*B*). The increase in water level at site Y of 0.43 ft relative to NGVD 29 actually is a net decline of about 0.20 ft relative to the increased sea level (fig. 20*B*).

The decline in the water-table altitude relative to local sea level can be explained, in part, by the increase in ground-water pumping in the Pamet flow lens from 1929 to 2002. During this time, ground-water pumping increased from 0.30 to 0.94 Mgal/d. Nearly all of the water pumped from the Pamet flow lens is exported to Provincetown in the Pilgrim flow lens. This export of water creates the potential for an overall decline in water levels with time. If this decline were caused only by the loss of pumped water from the flow lens, then the decrease in water levels would be largest near the pumped wells in the center of the flow lens, and far from the coast or discharge areas, such as the Little Pamet River. Barlow and Hess (1993) showed that water-level declines in response to pumping near the Quashnet River in western Cape Cod were dampened by the interaction between ground and surface waters. If this dampening effect was the case in the Pamet flow lens, then the net loss at site Y, which is near a pumped well and away from surface-water discharge areas, should be greater than site X, which is near the Little Pamet River; however, the simulated net water-level decline at site X is more than double the net water-level decline at site Y. The results indicate that the difference in water-level declines at these sites may be a result of changing sea level rather than ground-water pumping.

The surface-water bodies that dampen the decline in water levels in response to increased pumping also may dampen the rise in water levels in response to sea-level rise. Ground-water-fed, or gaining streams prevent the surrounding water table from rising appreciably above the altitude of the streambed. As the water table rises in response to sea-level rise, the amount of ground-water discharge to the stream increases because the increased height of the water table adjacent to the stream results in increased streamflow rather than a higher water-table altitude at the stream.

In the Pamet flow lens, the model-calculated ground-water discharge to the Little Pamet River increased from 0.39 to 0.62 $ft^3/s$ in response to sea-level rise from 1929 to 2002. The increase in streamflow in the Little Pamet River over this period is related directly to the increased water-table altitude in the vicinity of the river. The surface-water bodies, such as the Little Pamet River, are analogous to pumping wells in the way in which they remove freshwater from the aquifer. The "pumping rates" of the streams depend upon the magnitude of water-table rise in response to the sea-level change. This "pumped" water leaves the aquifer as streamflow to the coast. The water levels have risen by a greater rate at site Y than site X probably because of the proximity of site X to the Little Pamet River.

The net decline in the water-table altitude at site Y relative to the rise in local sea level may be the result of exported pumped water from the Pamet flow lens lowering regional water levels. However, the total simulated discharge to non-tidal surface-water bodies in the flow lens increased by about 25 percent from 1929 to 2002. Therefore, this net decline also may be the result of the regional effect of streams and wetlands on the ground-water levels throughout the Pamet flow lens.

The effect of increased ground-water discharge to tidally affected surface-water bodies is negligible, especially by comparison to the effect of non-tidal surface-water bodies; it is likely that the stream stage in tidal waters will rise in conjunction with sea level at a rate equal or greater than the rise in water levels in the aquifer. Nuttle and Portnoy (1992) hypothesize that this rise in stream stage may result in an overall decrease in ground-water discharge to coastal surface-water bodies with time.

## Freshwater/Saltwater Interface

At sites X and Y (fig. 19), the water-level altitudes decreased relative to sea level in response to the rising sea level (fig. 20). The depth to the freshwater/saltwater interface decreased, or alternatively, the altitude of the freshwater/saltwater interface increased relative to sea level. The difference in the model-calculated change in altitude of the freshwater/saltwater interface between sites X and Y is controlled by the difference between the sites in the net decline in water levels relative to local sea level.

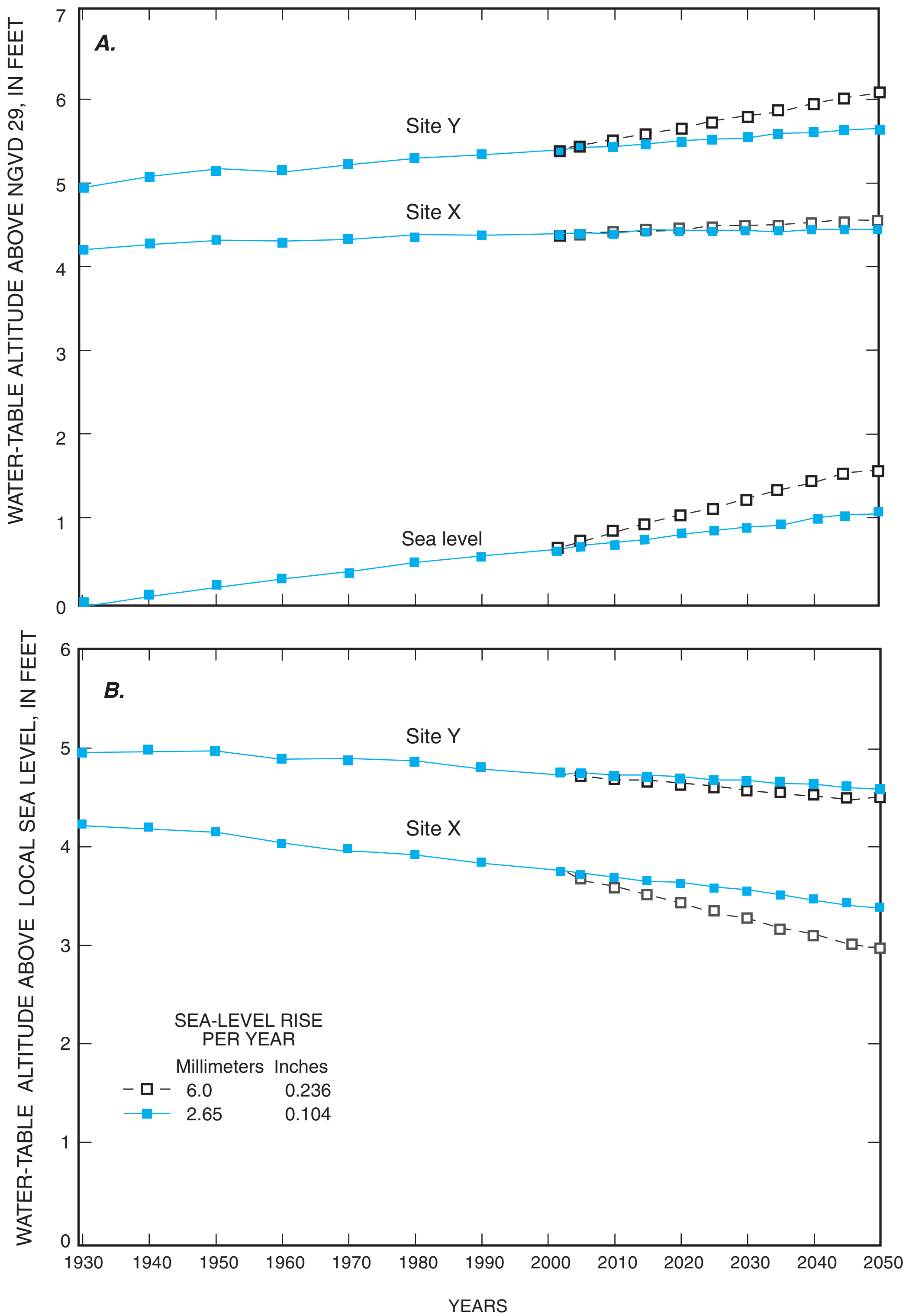



**Figure 20.** Model-calculated water-table altitude from 1929 to 2050 at sites X and Y and simulated changes in sea level *A*, above NGVD 29; and *B*, above local sea level, North Truro, Massachusetts.

The flow model was used to analyze the position and movement of the freshwater/saltwater interface beneath sites X and Y to determine the response of the interface to the rise in sea level from 0.0 ft in 1929 (NGVD 29) to its present-day (2002) altitude of 0.63 ft (NGVD 29). The position of the interface beneath site X in 1929 was calculated to be 215 ft below NGVD 29 (fig. 21). In response to sea-level rise and increased ground-water pumping, the altitude of the interface increased by about 0.37 ft by 2002. A similar increase was observed for site Y, where the altitude of the interface was calculated to be about 246 below NGVD 29 and rose by about 0.37 ft from 1929 to 2002 (fig. 21).

The lack of movement of the freshwater/saltwater interface with time in response to the change in sea level may be the result of a slow response of the interface to changing stress conditions. To determine whether the interface position is affected by the sea-level change and whether there is a lag in the response, three simulations were made: (1) the rate of sea-level rise was assumed to be zero for the next 48 years from 2002 to 2050, (2) sea level was assumed to continue to rise at 0.104 in/yr (2.65 mm/yr) from 2002 to 2050, and (3) the rate of sea-level rise was assumed to increase to 0.236 in/yr (6 mm/yr) from 2002 to 2050, as hypothesized by Donnelly and Bertness (2001).

In the first simulation described above, sea level remained at a constant altitude of 0.63 ft above NGVD 29 from 2002 to 2050 and the amount of ground-water pumping remained at the 2002 pumping rate. The water-table altitude at both sites X and Y remained constant; however, the altitude of the freshwater/saltwater interface did change beneath these sites. The altitude of the freshwater/saltwater interface increased by 1 ft beneath site Y, whereas at site X, the altitude of the interface increased by about 7 ft. The change in the position of the interface at these sites indicates that there is a lag in the reponse of the interface and that the proximity of site X to the Little Pamet River may result in more of a change in the interface position than that calculated for site Y.

In the second simulation, sea level continued to rise at the rate of 0.104 in/yr (2.65 mm/yr) from 2002 to 2050. In response to this continued sea-level rise of 0.42 ft (for a cumulative sea-level rise of 1.05 ft), the altitude of the water table at sites X and Y declined by a net of 0.34 and 0.14 ft, respectively. By 2050, the water-table altitude at site X was 3.41 ft above the local sea-level altitude of +1.05 ft NGVD 29, in comparison to an altitude of 3.75 ft above the local sea-level altitude of +0.63 NGVD 29 in 2002 (fig. 20*B*). At site Y the water-table altitude in 2050 was 4.61 ft above the local sea-level altitude of +1.05 ft NGVD 29 in comparison to 4.75 ft above the local sea-level altitude of +0.63 ft NGVD 29 in 2002 (fig. 20*B*).

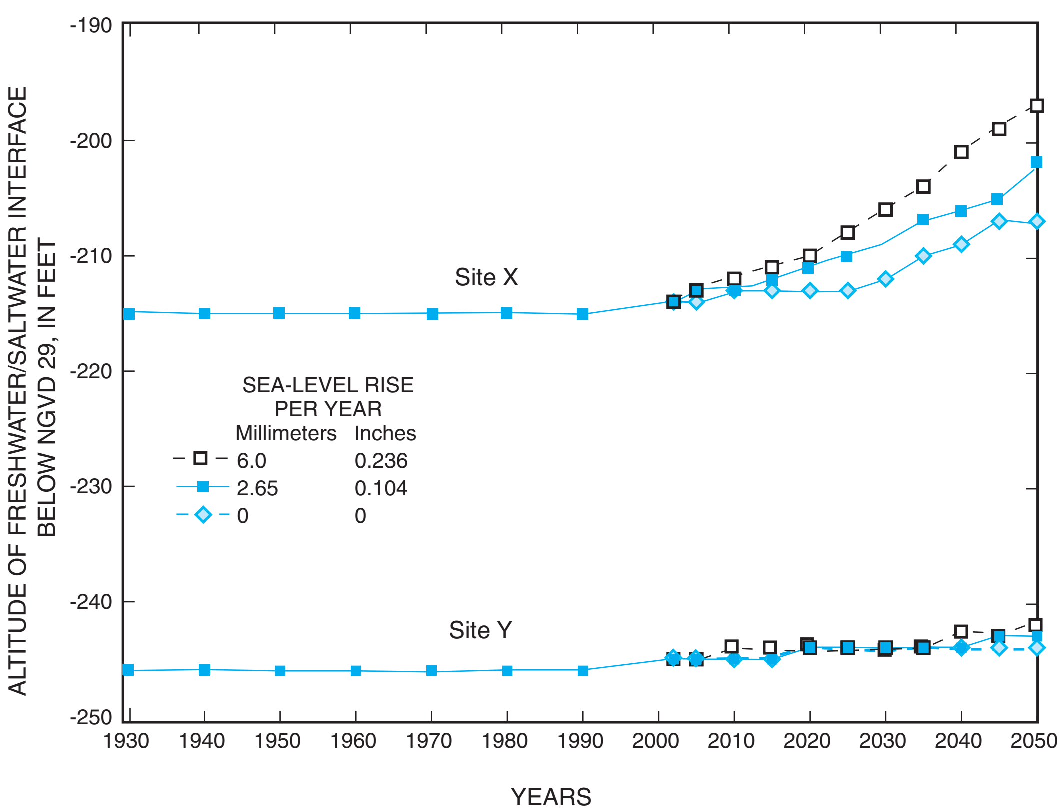



**Figure 21.** Model-calculated altitude from 1929 to 2050 of the freshwater/saltwater interface relative to NGVD 29 beneath sites X and Y, North Truro, Massachusetts.

During the period of 1929 to 2050, the rises in water-table altitudes of 0.23 ft and 0.71 ft at sites X and Y, respectively, are less than the simulated rise in sea level of 1.05 ft for that same period (fig. 20*A*). As a result, the net change in water-table altitudes at these wells relative to sea level would be about -0.82 ft for site X and -0.34 ft for site Y (fig. 20*B*). During this same period, however, the streamflow in the Little Pamet River increased two-fold from 0.39 $ft^3/s$ in 1929 to 0.78 $ft^3/s$ in 2050. It is this increase in streamflow that may account for the difference in water-level changes between sites X and Y.

The altitude of the freshwater/saltwater interface also changed beneath the sites as sea level continued to rise from 2002 to 2050. The altitude of the interface increased by 12 ft beneath site X and 2 ft beneath site Y relative to their 2002 positions. Because the sea level rose from about 0.63 to 1.05 ft above NGVD 29 over this period, the net change in altitude of the interface relative to sea level in 2050 beneath sites X and Y is 11.58 ft and 1.58 ft, respectively.

The increase in the altitude of the freshwater/saltwater interface beneath site X changed at a much greater rate from 2002 to 2050 than the rate from 1929 to 2002. From 1929 to 2002, the altitude of the interface increased by only 0.37 ft or at rate of 0.06 in/yr (1.50 mm/yr). From 2002 to 2050, the altitude of the interface increased by another 11.58 ft in only 48 years or at a rate of 2.90 in/yr (73.53 mm/yr). Of this model-calculated 11.58 ft, an increase about 7 ft can be attributed to the lag in the response to the sea-level rise from 1929 to 2002. The additional 4.58 ft of increase is a result of the continued sea-level rise of 0.104 in/yr (2.65 mm/yr) from 2002 to 2050. This rate of rise from 2002 to 2050 is about 1.15 in/yr (29.08 mm/yr) and is substantially greater than the rate of rise of 0.06 in/yr (1.50 mm/yr) from 1929 to 2002. The total increase in the altitude of the interface from 1929 to 2050 beneath site X, however, is about 11.95 ft, or 1.19 in/yr (30.10 mm/yr) in response to a decline in water-table altitude of 0.82 ft over the same period.

After accounting for the model-calculated 1 ft of rise that can be attributed to the lag in the response to the sea-level rise from 1929 to 2002 beneath site Y, the 0.58-ft increase in the altitude of the freshwater/saltwater interface from 2002 to 2050 is substantially less than that calculated for site X, but still is greater than the increase in the interface altitude of 0.37 ft for this location from 1929 to 2002. On the basis of these results and their uncertainty, however, it would appear that the position of the interface beneath this site is not substantially affected by the change in sea-level altitude over time.

In the third simulation, the potential effects of an increased rate in sea-level rise from 0.104 in/yr (2.65 mm/yr) to 0.236 in/yr (6.0 mm/yr) were assessed, as hypothesized by Donnelly and Bertness (2001). In response to this rise in the sea-level altitude from about 0.63 ft above NGVD 29 in 2002 to about 1.58 ft above NGVD 29 in 2050, the water-table altitude declined by about 0.77 ft at site X and about 0.25 ft at site Y relative to local sea level (fig. 20*A*) and the streamflow in the Little Pamet River increased to about 1.2 $ft^3/s$, nearly twice the flow in 2002.

In response to this additional increase in the rate of sea-level rise, the altitude of the freshwater/saltwater interface between 2002 and 2050 increased by 16.05 ft beneath site X and 2.05 ft beneath site Y relative to local sea level in 2050 (fig. 20*B*). Of this 16.05 ft of increase beneath site X, about 7 ft can be attributed to the lag in the response to the sea-level rise from 1929 to 2002. Therefore, the total increase in the altitude of the interface is about 9.05 ft for a sea-level rise of 0.236 in/yr (6.0 mm/yr) compared to the calculated increase in interface altitude of 4.58 ft for a sea-level rise of 0.104 (2.65 mm/yr) for the same period. Of the 2.05 ft of increase beneath site Y, about 1 ft can be attributed to the delay in the response to the sea-level rise from 1929 to 2002. Therefore, the total increase in the altitude of the interface is about 1.05 ft for a sea-level rise of 0.236 in/yr (6.0 mm/yr) compared to the calculated increase in altitude of 0.58 ft for a sea-level rise of 0.104 in/yr (2.65 mm/yr) for the same period (fig. 20*B*).

Based on the analysis described above, sea-level rise over time results in a thinning of the aquifer in the Pamet flow lens. This thinning results because the rate at which the water table increases is less than the rate at which sea level rises resulting in a net decline in the water-table altitude, which results in an increase in the altitude of the freshwater/saltwater interface. The areas where the effect on the position of the freshwater/saltwater interface is greatest are those areas near non-tidal streams and wetlands. The effect is substantially less where the water-table rise is not limited by non-tidal streams and wetlands. The rate of rise of the freshwater/saltwater interface is affected by a lag in the response of the interface to changes in water-table altitude.

The effect of sea-level rise was analyzed in the other flow lenses for the same period and a similar effect of increased water levels and streamflows was observed; however, the effect on the freshwater/saltwater interface was negligible during this period. In the Pilgrim flow lens, non-tidal surface-water discharge is negligible and the water level at PZW-78 increased by about the same rate as sea level (fig. 19). Because the water levels in this flow lens rise in conjunction with sea level, the difference between the water table and sea level does not change appreciably and, therefore, the position of the freshwater/saltwater interface remains static.

In the Chequesset flow lens, and to a lesser extent in the Nauset flow lens, there are a substantial number of nontidal surface-water discharge areas, and the rate of water-level rise in these flow lenses is much less than the rate of simulated sea-level rise (fig. 19). Although the model-calculated rate of water-level increases in these flow lenses are substantially lower than the rate of sea-level rise, there is no appreciable change in the model-calculated interface position beneath these flow lenses as was calculated beneath the Pamet flow lens. A possible reason for this difference in the response of the interface could be differences in subsurface lithology simulated in the flow lenses. It is these differences that may result in much slower response times of the freshwater/saltwater interface in the Chequesset and Nauset flow lenses than were calculated in the Pamet flow lens.

## Pumping Wells

Another hydrologic stress that affects the height of the water table, and, therefore, can be affected by the rise in sea level, is pumping wells. The South Hollow well field in the Pamet flow lens has been in operation since 1955 and currently provides, on average, over 60 percent of the nearly 1.0 Mgal/d of water pumped from the Pamet lens for public supply. The model-calculated altitude of the freshwater/saltwater interface beneath the center of this well field is about 187 ft below NGVD 29 for 2002, which is 43 ft higher than the altitude calculated for 1954 before pumping started (fig. 16).

In the first simulation, sea level was held constant at the 2002 altitude of about 0.63 ft above NGVD 29 and pumping rates were maintained at current (2002) rates from 2002 to 2050. The altitude of the freshwater/saltwater interface increased by about 10 ft beneath the South Hollow well field from 2002 to 2050. This 10-ft increase is much greater than the 1-ft increase calculated at site Y and is slightly greater than the 7-ft increase calculated at site X. The model-calculated increase in the altitude of the freshwater/saltwater interface beneath the well field from 2002 to 2050 is a delayed response to the changes in pumping from 1955 to 2002, a result of the rising sea level from 1929 to 2002, or a combined effect of both of these factors.

In the second simulation, sea level was set to rise at 0.104 in/yr (2.65 mm/yr) and pumping rates were held constant at 2002 rates. The results from this simulation indicate that by 2050 the altitude of the interface would increase by 13 ft from the 2002 position, 3 ft more than was calculated when sea level was set at the 2002 altitude of 0.63 ft above NGVD 29. This increase in the altitude of the interface is similar to the increase calculated beneath site X.

In the third simulation, the rate of sea-level rise was set to increase from 0.104 in/yr (2.65 mm/yr) to 0.236 in/yr (6 mm/yr) and pumping was held constant at 2002 rates. The altitude of the interface beneath the well field increased by 15 ft between 2002 and 2050, 2 ft more than for the rise of 0.104 in/yr (2.65 mm/yr), but less than was seen at site X. The reason for this difference in responses at the well field and site X can be attributed to the fact that the pumping rate is held constant at the well field. In the case of site X, where the ground-water discharge to the stream continued to increase in response to the rising water table, the water-table rise in the vicinity of the stream was limited. This limited rise created a greater decline in water levels relative to sea level near the stream than the declines observed at areas farther away from the stream.

With the pumping rates held constant for the 48-year simulation, the water level at the well field rose by 0.19 ft. This rise represents a decline with respect to the new sea-level position of 1.05 ft above NGVD 29 in 2050 calculated for a rate of sea-level rise of 0.104 in/yr (2.65 mm/yr). The net decline of 0.23 ft in the water-table altitude at the well field, the difference between simulated rise in sea level of 0.42 ft from 2002 to 2050, and the 0.19 ft rise in the altitude of the water table, is less than the net decline of 0.34 ft at site X near the Little Pamet River. The constant pumping rate does not affect the response of the water table to sea-level rise, whereas in the vicinity of the stream at site X, the rise in the water table is offset by the increase in streamflow. Thus, the rate of rise at the water table decreases and causes the interface to rise.

If the pumping rate at the South Hollow well field were to increase enough for the water-table altitude near the well field to remain constant even though sea level is rising, then the interface would be expected to rise at a similar rate as that observed at site X near the stream. If the pumping rate were increased enough for the water-table altitude at the well field to decline while the surrounding water table rose because of sea-level rise, then the altitude of the interface would be affected by both increased pumping and sea-level rise.

Sea-level rise and increased pumping rates have a similar effect on the relation between the yield of pumping wells and the position of the freshwater/saltwater interface. This relation may necessitate limiting pumping rates in response to sea-level rise in order to protect public-water-supply systems from saltwater contamination. Past studies to evaluate the long-term safe yield at the South Hollow well field have determined that there is the potential for saltwater intrusion if the interface rises by more than 33 percent of the distance from the predevelopment interface position to the bottom of the well screen (Camp Dresser & McKee, Inc., 1985; Environmental Partners Group, Inc., 2002). In the case of the South Hollow well field, a 33-percent change in the interface position would be a rise of 63 ft from a model-calculated predevelopment altitude of 230 ft below NGVD 29.

The model-calculated altitude of the interface beneath the South Hollow well field for 2002 is 43 ft above the predevelopment position (fig. 16) or a 23-percent change in the distance from the predevelopment interface position to the simulated altitude of the bottom of the well screen of -40 ft NGVD 29. The model simulations indicate that, during the next 48 years, the altitude of the interface may increase by as much as an additional 15 ft, depending on the rate of sea-level rise, even if pumping were held constant at 2002 rates. This additional 15-ft of rise would result in a total decrease in the depth of the interface position of 58 ft by 2050, or a 31-percent change in the distance from the predevelopment interface position to the bottom of the well screen.

On the basis of the analysis described above, the interface beneath the South Hollow well field in 2002 has risen only by 23 percent of the distance between the predevelopment interface position and the bottom of the well screen, which is much less than the 33-percent change threshold for safe yield. However, if the current (2002) pumping rates are held constant for the next 48 years, this percentage could increase to 31 percent. Presumably, if the pumping rates were increased from the current rate during this period, the percentage change could exceed the 33-percent threshold for safe yield.

# Simulation of Proposed Ground-Water-Pumping Scenarios

The flow model used in this investigation was developed to serve as a tool to assist the National Park Service and the communities of Lower Cape Cod in managing their water resources so that the future demand for water supply can be met without adversely affecting the ponds, streams, wetlands, and coastal areas throughout Lower Cape Cod. Simulations were made to evaluate whether future pumping could affect surface-water bodies, such as streams and ponds, and the position and movement of the freshwater/saltwater interface. Examples were selected to demonstrate how the flow model developed for this investigation could be used to determine the aquifer-system response to changes in pumping and recharge conditions. The proposed pumping well sites, pumping rates, and wastewater return locations simulated in the model were provided by the four towns of Lower Cape Cod through the stakeholders committee set up for this project.

## Effects on Streamflow

The residences of the town of Eastham in the Nauset flow lens (fig. 1) currently obtain nearly all of their water supply from small-capacity private drinking-water wells. Water pumped from these sites is returned to the aquifer by on-site private septic systems. Concerns over possible sources of contamination such as residential septic systems, the town landfill, and the commercial areas along Route 6, have prompted the Eastham Water Resources Advisory Board (WRAB) to consider two sites for public-supply wells to provide, on average, 1.10 Mgal/d of water (Dr. Karl Weiss, Eastham Water Resources Advisory Board, written commun., 2003).

The WRAB currently is considering two potential sites, the Roach site and Water District G site (fig. 12), to provide a total of 1.10 Mgal/d of water to meet the future water demand for the northern part of town. Given the locations of these sites, the hydrogeology of the Nauset flow lens, and the depth to the freshwater/saltwater interface, the greatest concern is the effect of pumping on Hatches Creek, which is in the northwestern part of town (fig. 12).

Hatches Creek is a small freshwater stream that flows from east to west into Sunken Meadow, a saltwater marsh along the shore of Cape Cod Bay. The model-calculated discharge to this stream under current (2002) conditions is about 0.52 $ft^3/s$ at West Road (fig. 12). The effect of pumping on streamflow in Hatches Creek at West Road was calculated for three different pumping scenarios at the proposed well sites.

First, a simulation was made with each well pumping at 0.55 Mgal/d for a total of 1.1 Mgal/d. It was assumed that 85 percent of this water was returned to the aquifer as wastewater within the areas designated as residential land use (Massachusetts Executive Office of Environmental Affairs–Community Preservation Initiative, 1999). These areas correspond with the Traffic Analysis Zones (TAZ), which partition the town into eight separate zones (fig. 12) (Landuse Collaborative, 1996), identified by the WRAB as areas likely to receive future public supply. Based on these pumping and wastewater-disposal rates, the streamflow in Hatches Creek would be expected to decrease from 0.52 to 0.15 $ft^3/s$, or about 71 percent of the total pre-pumping flow.

The reduction of streamflow described above is illustrated schematically in figure 22. If wells are pumped upgradient of a surface-water body, such as Hatches Creek, the pumping well can capture water that otherwise would discharge to the surface-water body. Ground-water discharge to Hatches Creek is reduced, thereby reducing streamflow. If the pumping from wells upgradient of discharge areas becomes large enough, ground-water flow directions can be reversed and water can be drawn directly from surface-water bodies to the pumped well (fig. 22*C*). Streamflows and pond levels then are further reduced.

Additional simulations were made with each well pumping separately to determine the effects on streamflow in Hatches Creek to changes in pumping conditions. When pumping was simulated only at the Roach site at a rate of 0.55 Mgal/d, the streamflow was reduced by about 0.23 $ft^3/s$ to 0.29 $ft^3/s$, or 56 percent of the total pre-pumping flow. When pumping was simulated only at the Water District G site at a rate of 0.55 Mgal/d, the streamflow was reduced by about 0.07 $ft^3/s$ to 0.45 $ft^3/s$, or 87 percent of the total pre-pumping flow. The effect of streamflow reduction from both wells pumped separately is 0.30 $ft^3/s$ (0.23 + 0.07 $ft^3/s$), which is 0.07 $ft^3/s$ less than the 0.37 $ft^3/s$ of reduction that was calculated when both wells were pumped simultaneously. The difference in the effect on streamflow from these simulations can be attributed to the difference in the source of water to the wells and the stream for the different pumping scenarios (figs. 23*A*–*D*).

Under pre-pumping conditions, the water discharging to Hatches Creek (0.52 $ft^3/s$) is derived from the area shown in figure 23*A*. When only the Roach site is pumped at 0.55 Mgal/d, the area contributing recharge to this well shows that the well captures water that otherwise would have discharged to Hatches Creek (fig. 23*B*). When only the Water District G site is pumped, most of the water discharging at that well does not derive from the area that contributes water to Hatches Creek; therefore, there is less effect from pumping at this site on Hatches Creek (fig. 23*C*). When both wells are pumping at 0.55 Mgal/d, the area contributing recharge to the Roach site is affected by the pumping at Water District G (fig. 23*D*). In response to the pumping at the Water District G site, the area contributing recharge to the Roach site shifts to an area that captures more water that would have otherwise discharged to Hatches Creek (fig. 23*D*). This shift may explain why the streamflow reduction is greater with both wells pumping than the arithmetic sum of the separate reductions caused by each well.

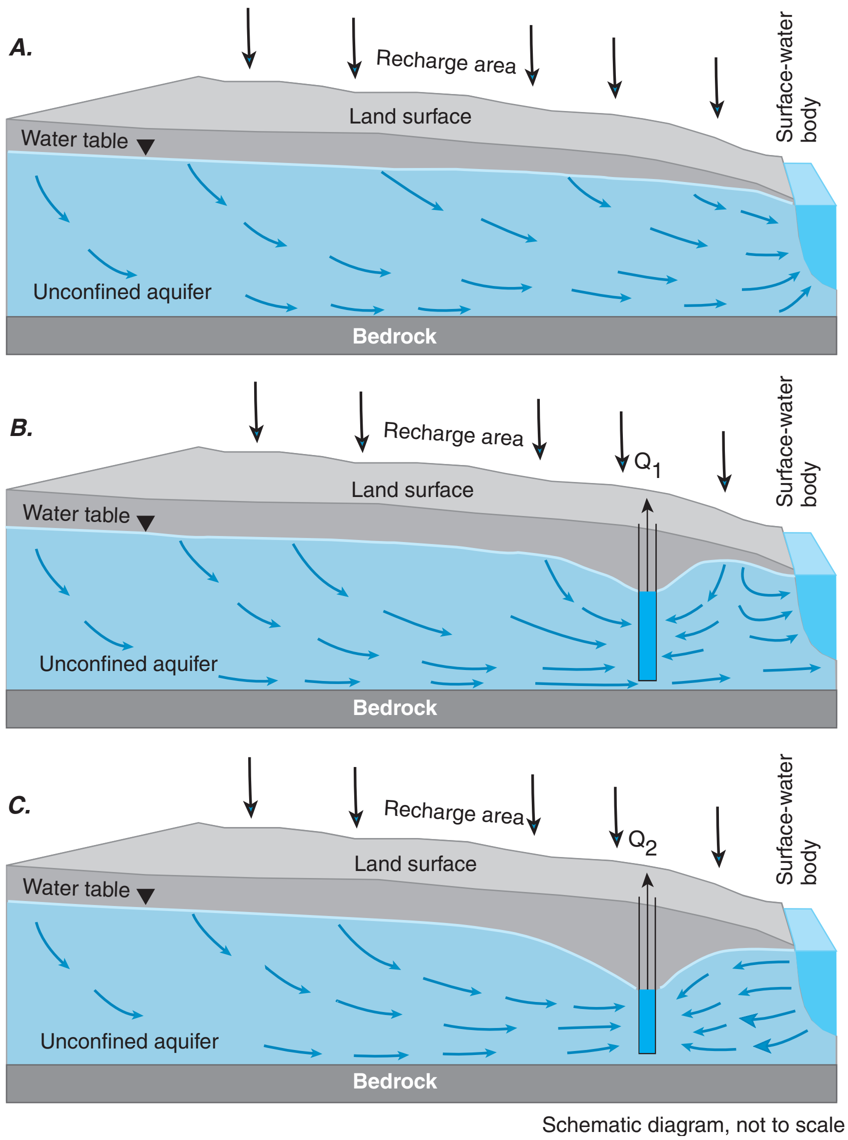



**Figure 22.** Schematic diagram of a hypothetical aquifer showing ground-water discharge to a surface-water body with *A*, no pumping; *B*, pumping at a rate such that the well would capture water that would otherwise discharge to the surface-water body; and *C*, pumping at a higher rate so that the flow direction is reversed and the well pumps water from the surface-water body (modified from Alley and others, 1999).

A final simulation was made to determine the effect of pumping from Water District G site alone at a rate of 1.10 Mgal/d on Hatches Creek. The area contributing recharge to the well at this pumping rate is double the size of the area contributing recharge to the well for the lower rate of 0.55 Mgal/d, and as a result the simulated well captures more flow that would have otherwise discharged to Hatches Creek (fig. 24). The streamflow reduction of 0.15 $ft^3/s$ (29 percent of pre-pumping flow) from the higher pumping rate at the Water District G site, however, is lower than the 0.23 $ft^3/s$ (56 percent of pre-pumping flow) from the Roach site pumping alone at 0.55 Mgal/d.

## Effects on Water Levels in Kettle-Hole Ponds

The residents of the town of Wellfleet receive all of their drinking water from the Chequesset and Nauset flow lenses. As is the case in Eastham, nearly all of Wellfleet's residents obtain their drinking water from small-capacity domestic-supply wells and the wastewater is returned to the aquifer through on-site private septic systems. The only public supply currently operating in Wellfleet is a small system that provides about 7,000 gal/d of water to the residences of Coles Neck in the vicinity of the town landfill.

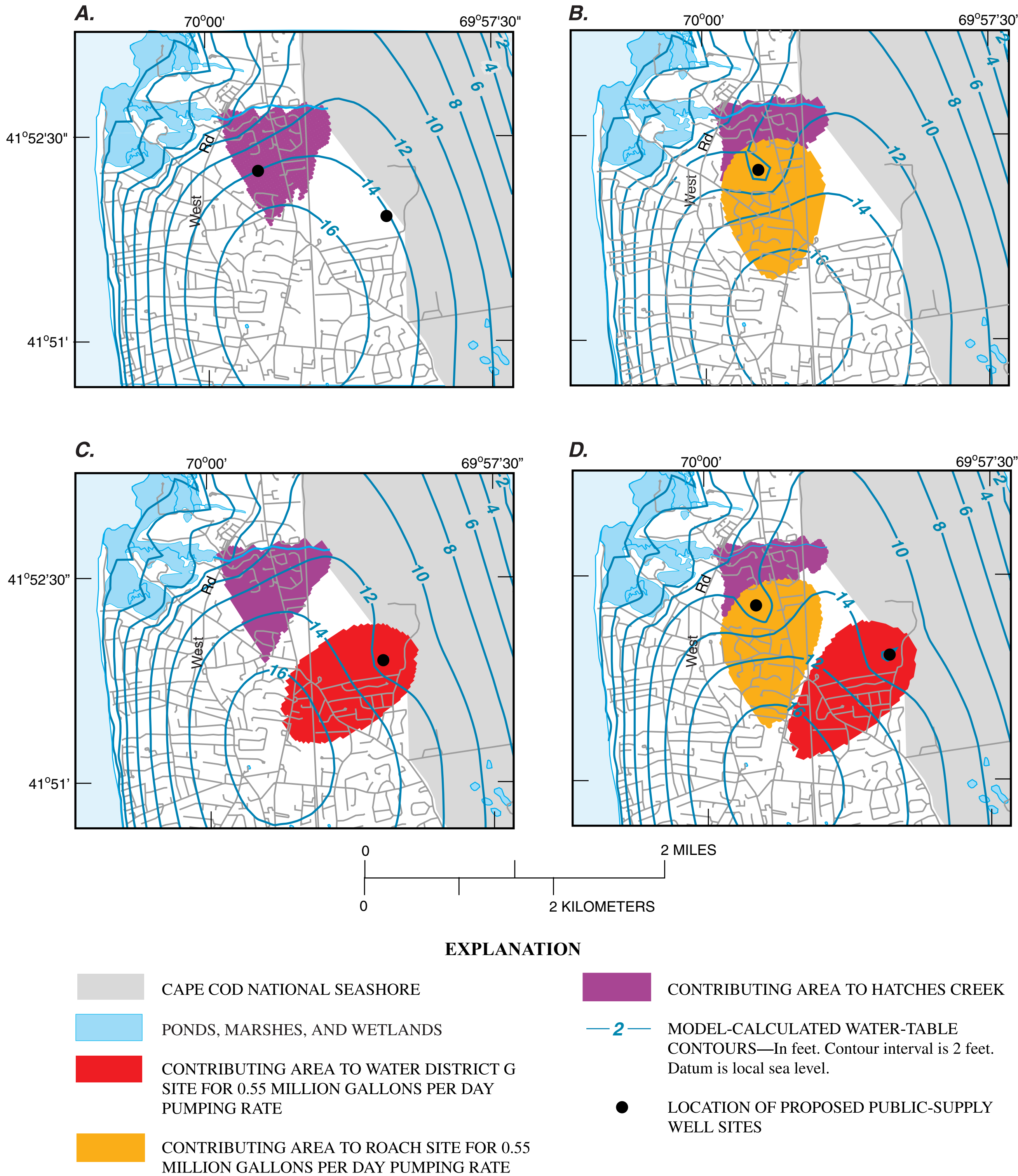



**Figure 23.** Location of model-calculated contributing areas to *A*, Hatches Creek for current (2002) conditions; *B*, Hatches Creek and the Roach site pumping at 0.55 million gallons per day; *C*, Hatches Creek and Water District G site pumping at 0.55 million gallons per day; and *D*, Hatches Creek, Water District G site, and the Roach site each pumping at 0.55 million gallons per day, Eastham, Massachusetts.

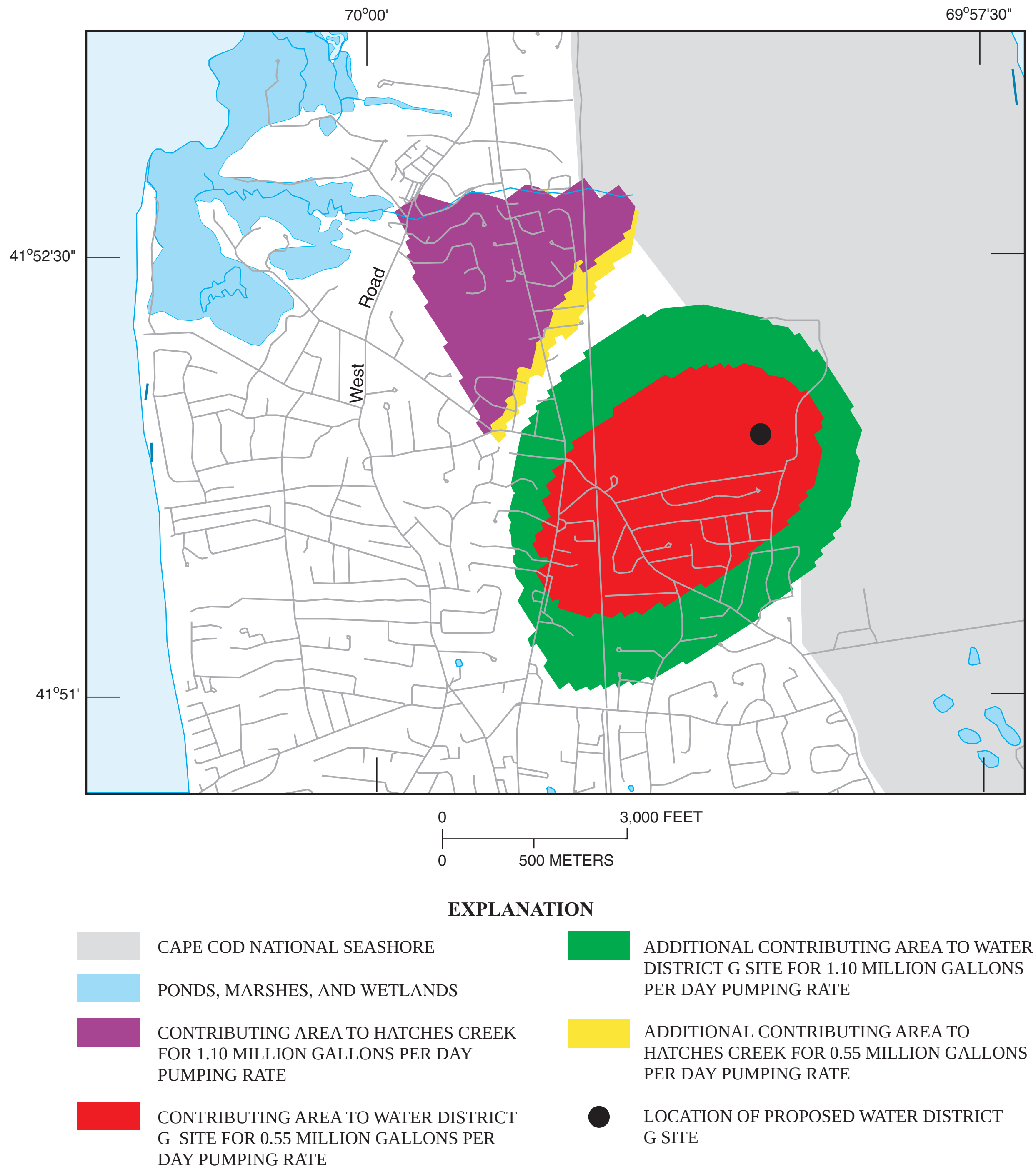



**Figure 24.** Location of model-calculated contributing areas to Hatches Creek and the proposed Water District G well pumping at 0.55 and 1.10 million gallons per day, Eastham, Massachusetts.

The town recently has begun contruction (September 2003) to expand the existing public-supply system to the municipal buildings and residences in the downtown area (TAZs 12 and 13, fig. 12) at an average daily rate of 0.30 Mgal/d. This proposed increase in public supply would be obtained from increased pumping at the Coles Neck site and the possible addition of two new well sites, the Boy Scout Camp and Wellfleet By the Sea sites (fig. 12). The town currently is conducting hydrologic investigations to determine if these three sites can be pumped at 0.1 Mgal/d each.

A simulation was made with each well pumping at a rate of 0.1 Mgal/d and about 0.255 Mgal/d of wastewater returned to the aquifer at the areas designated as residential and municipal land uses in TAZs 12 and 13 (fig. 12) (Massachusetts Executive Office of Environmental Affairs–Community Preservation Initiative, 1999). Because the potential effects of pumping at these sites on pond levels in Duck, Dyer, Long and Great Ponds (figs. 12, 25) is of great concern, determining this effect was a principal goal of this analysis. Of this group of kettle-hole ponds in the vicinity of the proposed pumping wells, Duck Pond is most likely to be affected by possible future pumping because of its proximity to the top of the water-table mound and to the proposed pumping wells.

The potential effect of pumping on Duck Pond differs from the effect of pumping on surface-water bodies shown schematically in figure 22 and on Hatches Creek (figs. 23 and 24), because Duck Pond is upgradient instead of downgradient of the proposed pumping wells. As a result, the simulated pumping wells cannot capture water that otherwise would have discharged to the pond. An analysis of the contributing areas to Duck Pond and the three pumping wells indicates that the wells do not capture any water that otherwise would discharge to the pond under current conditions (fig. 25*B*).

Although the proposed wells do not capture water that otherwise would have discharged to Duck Pond, the resulting decline in ground-water levels in response to pumping does extend to the pond. For average annual pumping conditions, the pond level would be lowered by nearly 0.4 ft (fig. 25*B*). Simulations were made to determine if the location of the pond near to the top of water-table mound may account for the seemingly large effect on the pond level by a small increase in pumping.

Previous studies on Cape Cod have shown that the natural fluctuation in water levels in response to changes in recharge generally increases from the coast to the top of the water-table mounds (Letty, 1984; Masterson and Barlow, 1996). In addition, streams also appear to limit the fluctuation of the water table in the surrounding area. Duck Pond, which is near the top of the southern water-table mound in the Chequesset flow lens (fig. 1), has a reported maximum annual fluctuation of about 1.6 ft determined from monthly measurements from 1994 to 1999 (Sobczak and others, 2003). In contrast, the maximum annual fluctuation at Gull Pond, which is close to the Herring River and connected to the nearby kettle ponds by a network of shallow, dug channels, was measured to be about 0.7 ft during that same period.

The difference in the response of the two ponds to the same changes in recharge indicate that Duck Pond is more susceptible to changes in stress conditions than ponds farther from the top of the mound, closer to ground-water discharge areas. Two simulations then were made to determine the response of water levels in Duck Pond to in-season and off-season differences in recharge and proposed pumping.

An initial simulation of off-season and in-season changes in recharge and pumping was made to determine the current range in pond level at Duck Pond for current conditions. Based on a simulated off-season recharge of 18 in. from October to April and an in-season recharge of 6 in. from May to September, the model-calculated annual fluctuation in pond levels for Duck Pond was 1.2 ft. This value is lower than the 1.6-ft range reported by Sobczak and others (2003); possible reasons for the difference include temporal discretization of the recharge or a difference between the total recharge during the measurement period (1994–99) and the assumed annual average recharge of 24 in/yr. The 1.2-ft value for the annual pond fluctuation, however, will be used in this analysis as the baseline fluctuation for the purpose of comparing the changes in pond levels for different pumping rates.

A second simulation was made with off-season and in-season changes in pumping and recharge. The annual average pumping rates of 0.1 Mgal/d at each well were apportioned between off-season and in-season pumping rates of 0.04 Mgal/d and 0.18 Mgal/d, respectively. The wastewater discharged to the aquifer was specified at 85 percent of the total pumping for each season and added to the respective recharge rates of 18 and 6 in/yr for off-season and in-season conditions.

The off-season and in-season declines in the level of Duck Pond were 0.3 and 0.5 ft, respectively, relative to the levels calculated with only temporally variable recharge. Although the model-calculated declines in pond level for off-season and in-season pumping and recharge conditions are less than half of the model-calculated annual fluctuation of 1.2 ft for current conditions, it is important to note that these reductions are superimposed on the current pond-level fluctuations. As a result, the current model-calculated pond-level altitude ranges from 10.13 to 8.94 ft above NGVD 29 and from 9.84 ft to 8.43 ft above NGVD 29 for the current and proposed pumping conditions (fig. 26).

In addition to changes in the off-season and in-season water levels, the annual fluctuation of 1.2 ft for current conditions increases to 1.4 ft for proposed pumping conditions. The areas most affected by changes in pond levels are the plant-filled shallow-water (littoral) areas of the pond. Depending on the bathymetry of the pond bottom, small changes in pond levels may substantially affect the near-shore pond ecology (Portnoy and others, 2001).

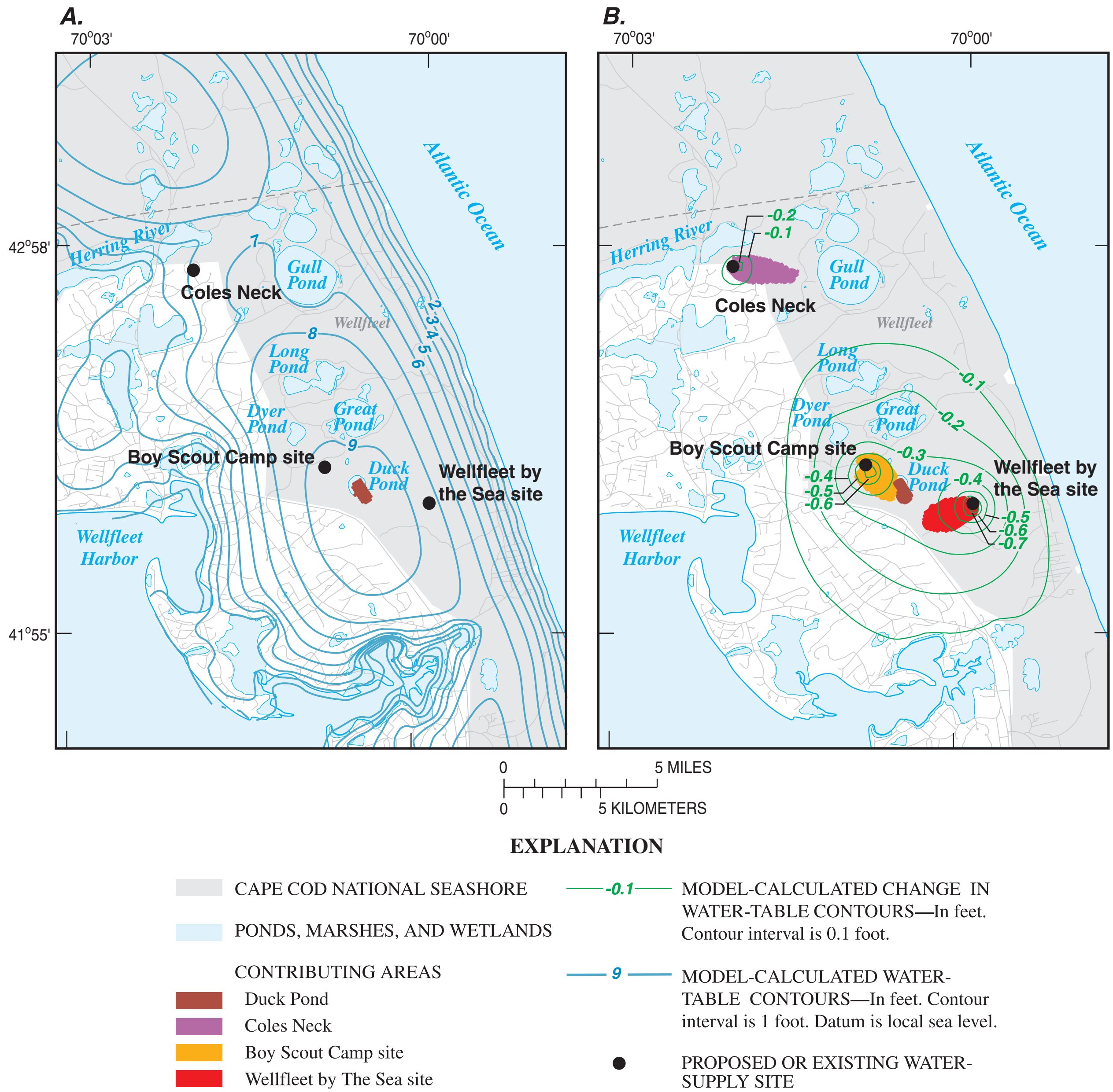



**Figure 25.** Location of *A*, model-calculated contributing area to Duck Pond and water-table contours for current (2002) conditions; and *B*, model-calculated contributing areas to Duck Pond, Coles Neck well, Boy Scout Camp site, and the Wellfleet By the Sea site, each pumping at 0.10 million gallons per day, and changes in model-calculated water levels from current (2002) conditions, Wellfleet, Massachusetts.

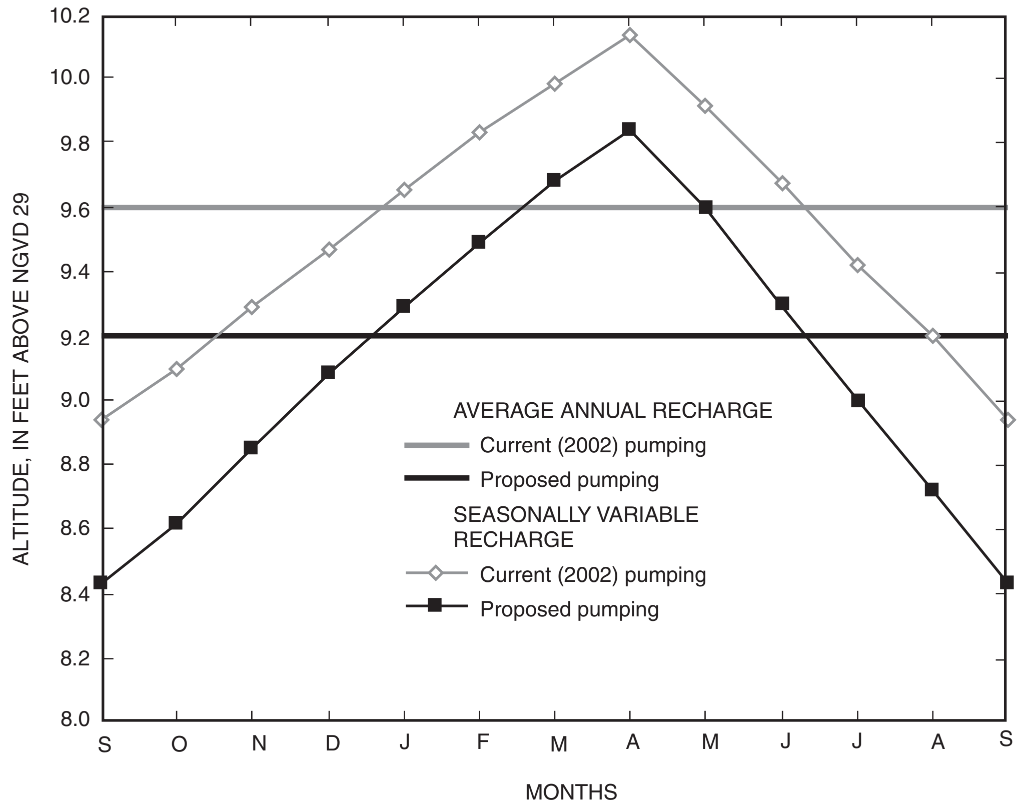



**Figure 26.** Model-calculated monthly pond-level altitudes in Duck Pond for current (2002) conditions and simulated pumping conditions of 0.10 million gallons per day at the Coles Neck well, the Boy Scout Camp site, and the Wellfleet By the Sea site, Wellfleet, Massachusetts.

## Effects on the Movement of the Freshwater/Saltwater Interface

The Pamet flow lens is the source of drinking water for the residents of Provincetown and the residents of Truro who live north of the Pamet River. Provincetown is in the Pilgrim flow lens, but the water quality in the Pilgrim flow lens is generally considered unacceptable for public supply without treatment (Environmental Partners Group, Inc., 2002). Most Truro residents in the Pamet lens obtain drinking water from private wells. The exceptions are residents and businesses along Route 6 and 6A (TAZ 8, fig. 12).

In response to concerns about increased water demand from Provincetown, the potential for future public-supply needs in Truro, and increases in the peak summer demand, the towns of Provincetown and Truro have identified two potential sites to investigate for possible future water supply, the North Unionfield and CCC-5 sites (fig. 12). In addition, the Provincetown Water Department is evaluating the potential for increasing the current pumping at the South Hollow well field from an average of 0.57 to 0.8 Mgal/d.

Simulations were conducted to evaluate the effects of future pumping on the position and movement of the freshwater/saltwater interface in the Pamet flow lens. Each of the proposed wells was pumped at 0.8 Mgal/d for 48 years (2002–50) in conjunction with the existing wells, each of which was pumped at its current (2002) rate. It was assumed that all the water pumped from these proposed sites will be used in Provincetown, and, therefore, will be removed from the Pamet flow lens and recharged to the Pilgrim flow lens through septic systems and a sewage-treatment facility.

Changes in the depth to the freshwater/saltwater interface were determined by comparing the interface positions calculated for the year 2050 for the existing wells pumping at the current rates to the positions calculated with the addition of the new wells pumping for the same period. Sea level was set at 0.63 ft above NGVD 29 (its current altitude) for the simulation period because the purpose of this analysis was to compare the effect of the different pumping scenarios on the position of the freshwater/saltwater interface.

In the first simulation, the potential well site at North Unionfield, which is about 3,000 ft south of the South Hollow well field and about 3,000 ft west of the NTAFB (fig. 12), was pumped at 0.8 Mgal/d. After 48 years of pumping, the altitude of the freshwater/saltwater interface beneath the North Unionfield well increased by about 58 ft from a model-calculated pre-pumping altitude of about 254 ft below NGVD 29. Of the 58 ft of rise, 2 ft is the response of the interface to current (2002) pumping at the existing wells and the remaining 56 ft is the result of pumping at the North Unionfield well.

The effect of pumping on the freshwater/saltwater interface extends beyond the North Unionfield site to areas beneath the existing wells at South Hollow and the NTAFB wells 4 and 5. Beneath the South Hollow well field, the altitude of the interface would increase by about an additional 17 ft above the increase of 10 ft calculated for continuing the current pumping at the existing wells for the next 48 years. Beneath NTAFB wells 4 and 5, the altitude of the interface would increase by about an additional 26 ft and 21 ft, respectively, above the increase of 8 ft calculated for each well for continuing the current pumping at the existing wells for the next 48 years. These increases are greater than those calculated for the South Hollow well field. The potential effect on the NTAFB wells is greater because the measured interface positions at these wells for current conditions are 118 ft and 67 ft below the depths of the well screens, compared to about 150 ft at the South Hollow well field.

The proximity of the North Unionfield site to the existing wells at South Hollow and the NTAFB also affects the contributing areas to these well fields. Because the North Unionfield site is upgradient of the existing wells and closer to the top of the water-table mound, pumping at the site will result in the capture of water that otherwise would have discharged to the existing wells. Thus, in order for the existing wells to satisfy their current pumping rates, these wells will begin to capture water from areas not discharging to North Unionfield. The contributing areas to these wells will shift from their current locations and bend around the contributing area to the proposed North Unionfield well site (fig. 27).

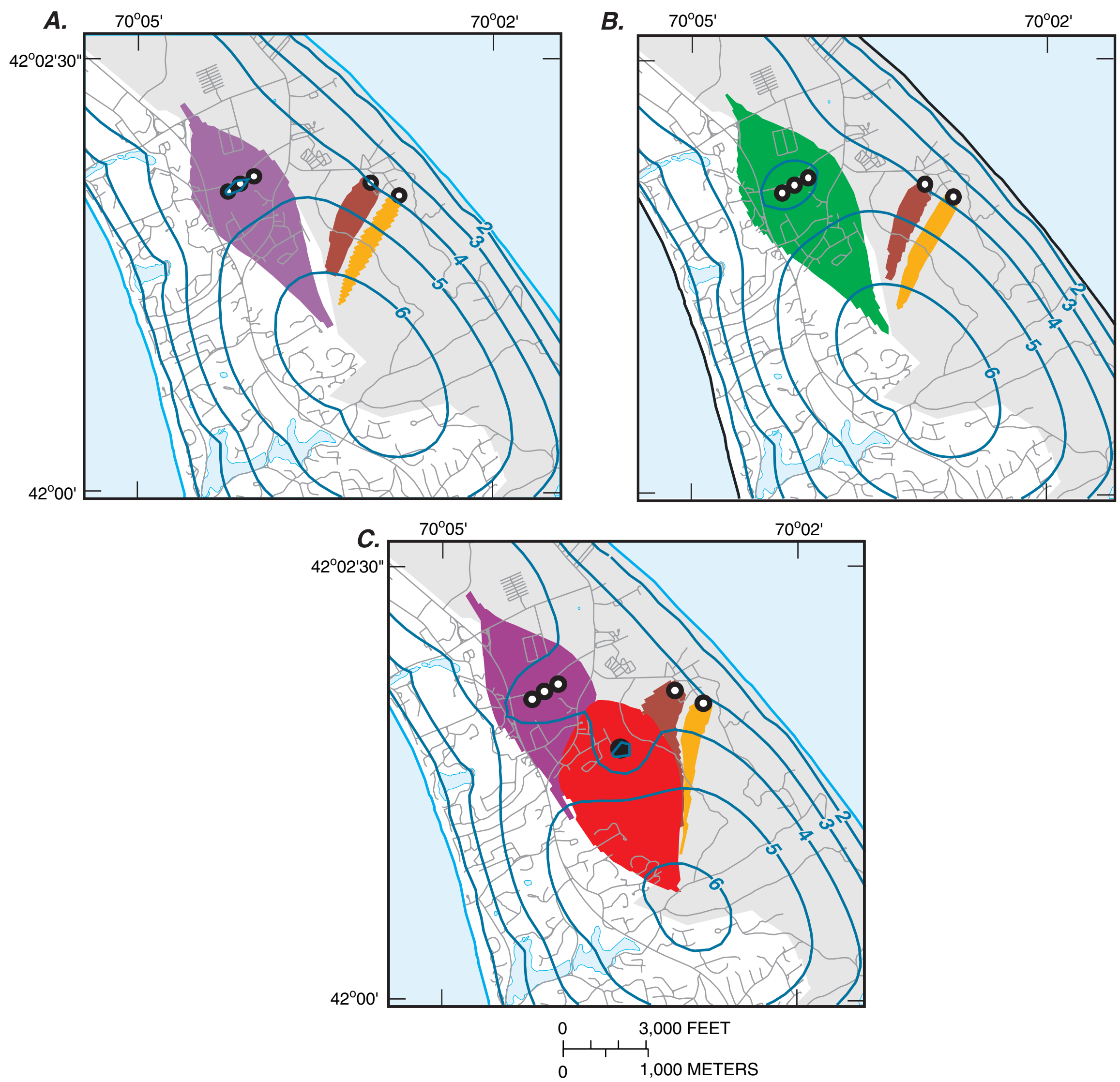
An additional simulation was made to evaluate the effect of increasing pumping from 2002 to 2050 at South Hollow from 0.57 Mgal/d to 0.80 Mgal/d with a constant sea level of 0.63 ft above NGVD 29. In this simulation, only the existing well fields at South Hollow and the North Truro Air Force Base were pumped with the South Hollow wells pumped at the increased rate of 0.80 Mgal/d. In response to the increase in pumping at the South Hollow well field, the altitude of the freshwater/saltwater interface increased by about 19 ft above the altitude calculated for a constant pumping rate of 0.57 Mgal/d for the same period. This increase is similar to the increase calculated for the well field when pumping for a constant pumping rate of 0.57 Mgal/d and for the North Unionfield site for a constant rate of 0.80 Mgal/d. The altitude of the freshwater/saltwater interface only increased by 2 ft beneath NTAFB well 4 and there was no effect calculated beneath NTAFB well 5. These calculated changes were minimal when compared to 26 and 21 ft of increase calculated for these wells, respectively, in response to pumping at the North Unionfield site.

The increase in pumping at South Hollow from 0.57 Mgal/d to 0.80 Mgal/d produced a corresponding increase in the size of the area contributing recharge to the well (figs. 27*A* and *B*); however, this increase in the area contributing recharge to the South Hollow well field did not substantially affect the areas contributing recharge to the NTAFB wells unlike the effects from pumping at the North Unionfield site (fig. 27*C*).

In the next simulation, the proposed well site at CCC-5 about 8,000 ft south of the North Unionfield site (fig. 12) was pumped at 0.80 Mgal/d. Again, sea level was assumed to be 0.63 ft above NGVD 29. After 48 years of pumping, the altitude of the interface beneath CCC-5 increased by about 14 ft from a model-calculated pre-pumping altitude of about 250 ft below NGVD 29.

The simulated effect of pumping at the CCC-5 site does not extend far enough north to affect substantially the contributing areas (fig. 28) or the freshwater/saltwater interface position beneath the existing well fields. The altitude of the freshwater/saltwater interface increased by only 1 ft at the South Hollow well field, 2 ft at NTAFB well 4, and 3 ft at NTAFB well 5. Simulated pumping from CCC-5, however, did decrease discharge at the Little Pamet River from 0.62 to 0.42 $ft^3/s$, or a 32-percent reduction in flow from current conditions. The reason for this reduction in streamflow is that pumping from the CCC-5 site captures water that otherwise would have discharged to the stream (fig. 28); this scenario is similar to the simulated pumping at the Roach site in the town of Eastham and its effect on Hatches Creek (fig. 23*B*).

On the basis of the results of these simulations, it was determined that the simulated response of the freshwater/saltwater interface to changes in pumping differed throughout the Pamet flow lens. The three pumping sites—South Hollow well field, the North Unionfield site, and the CCC-5 site—have similar pre-pumping altitudes for the freshwater/saltwater interface and yet the responses of the interface to pumping differed greatly among the sites. A possible explanation is that the altitude of the pre-pumping freshwater/saltwater interface may be insensitive to the aquifer properties used in the simulations to represent variability in the subsurface hydrogeology. The calculated response of the freshwater/saltwater interface to changes in pumping at these locations, however, may be affected greatly by simulated aquifer properties. For example, the pre-pumping altitude of the freshwater/saltwater interface beneath the South Hollow well field was calculated to be 230 ft below NGVD 29.




**EXPLANATION**

CAPE COD NATIONAL SEASHORE

PONDS, MARSHES, AND WETLANDS

CONTRIBUTING AREA TO SOUTH HOLLOW WELL FIELD FOR 0.57 MILLION GALLONS PER DAY PUMPING RATE

CONTRIBUTING AREA TO NTAFB-4 FOR 0.08 MILLION GALLONS PER DAY PUMPING RATE

CONTRIBUTING AREA TO NTAFB-5 FOR 0.08 MILLION GALLONS PER DAY PUMPING RATE

CONTRIBUTING AREA TO SOUTH HOLLOW FOR 0.80 MILLION GALLONS PER DAY

CONTRIBUTING AREA TO NORTH UNIONFIELD WELL SITE FOR 0.80 MILLION GALLONS PER DAY PUMPING RATE

— 2 — MODEL-CALCULATED WATER-TABLE CONTOURS—In feet. Contour interval is 1 foot. Datum is local sea level.

LOCATION OF EXISTING PUBLIC-SUPPLY WELL SITE

LOCATION OF PROPOSED PUBLIC-SUPPLY WELL SITE

**Figure 27.** Model-calculated water-table contours and contributing areas to *A*, South Hollow well field and the North Truro Air Force Base wells 4 and 5 for current (2002) pumping rates; *B*, South Hollow well field pumping at 0.80 million gallons per day and North Truro Air Force Base wells 4 and 5 pumping at current (2002) rates; and *C*, South Hollow well field and the North Truro Air Force Base wells 4 and 5 for current (2002) pumping rates and North Unionfield site pumping at 0.80 million gallons per day, North Truro, Massachusetts.

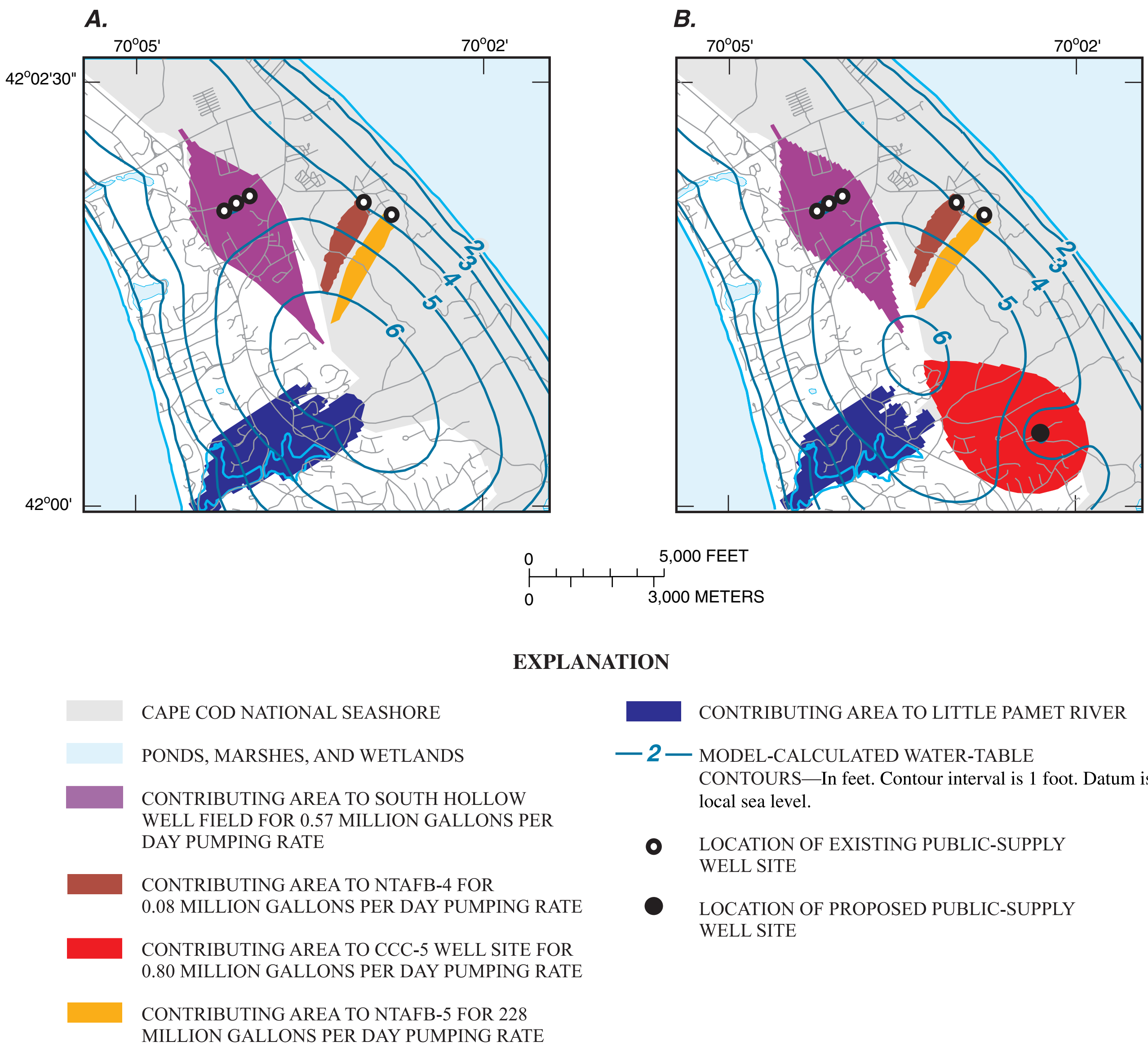



**Figure 28.** Model-calculated water-table contours and contributing areas to *A*, Little Pamet River, South Hollow well field, and North Truro Air Force Base wells 4 and 5 for current (2002) pumping rates; and *B*, Little Pamet River, South Hollow well field, and North Truro Air Force Base wells 4 and 5 for current (2002) pumping rates and CCC-5 site pumping at 0.8 million gallons per day, Truro, Massachusetts.

The altitude of the interface increased by 43 ft for the 47-year period from 1955 to 2002. This change was calculated by simulating the actual annual average pumping rates at the well field, which on average was about 0.44 Mgal/d. By comparison, the model-calculated altitude of the freshwater/saltwater interface at the North Unionfield site was 254 ft below NGVD 29 for pre-pumping conditions and increased by 58 ft with a constant pumping rate of 0.80 Mgal/d for the 48-year period from 2002 to 2050.

At the CCC-5 site, the altitude of the freshwater/saltwater interface for pre-pumping conditions was 250 ft below NGVD 29 and increased by only 14 ft for the same pumping rate and time as that of the North Unionfield site. The reason for the different responses of the interface altitude to pumping at the two sites can be attributed to the differences in lithology, and, therefore, differences in simulated hydraulic conductivity within the Pamet flow lens.

Beneath the South Hollow well field, the area in which the freshwater/saltwater interface shows the greatest response to increased pumping, the simulated transmissivity is about 35,000 $ft^2/d$. The freshwater/saltwater interface for pre-pumping conditions is 230 ft below NGVD 29 in a part of the aquifer with an assumed horizontal hydraulic conductivity of 70 ft/d and a ratio of horizontal to vertical hydraulic conductivity of 30:1. Beneath the North Unionfield site, the model-simulated transmissivity is about 22,000 $ft^2/d$ and, for current conditions, the freshwater/saltwater interface is about 254 ft below NGVD 29. At this altitude, the model-simulated horizontal hydraulic conductivity is 10 ft/d with a ratio of horizontal to vertical hydraulic conductivity of 100:1. Beneath the CCC-5 site, the model-simulated transmissivity is about 20,000 $ft^2/d$ and for current, pre-pumping conditions the freshwater/saltwater interface is about 250 ft below NGVD 29. At this depth, the model-simulated horizontal hydraulic conductivity is 10 ft/d with a ratio of horizontal to vertical hydraulic conductivity of 100:1.

An aquifer test was conducted in well TSW-200 (fig. 1) near the CCC-5 site. The results from this aquifer test indicate that the transmissivity in the upper 100 ft of aquifer is about 22,000 $ft^2/d$ with a ratio of horizontal to vertical hydraulic conductivity of 5:1 (Guswa and Londquist, 1976). At depths greater than 100 ft (about 95 ft below NGVD 29), the aquifer sediments consist of fine sand and silt. Therefore, the aquifer was assumed to have a much lower transmissivity and a much higher ratio of horizontal to vertical hydraulic conductivity at depth (Guswa and Londquist, 1976).

The measured depth to the center of the transition zone between freshwater and saltwater beneath TSW-200 is about 230 ft below NGVD 29. This measured freshwater/saltwater interface position is consistent with the model-calculated position of 247 ft below NGVD 29 beneath this well. Unfortunately, a measured response of the freshwater/saltwater interface to pumping cannot be obtained from the aquifer test to compare to the model-calculated results because the slow response in possible movement of the freshwater/saltwater interface greatly exceeds the length of time of the aquifer test. Model-simulation results indicate that the initial pre-pumping position of the interface may be insensitive to the simulated changes in subsurface lithology, as shown by the similar depths to the freshwater/saltwater interface beneath the South Hollow well field, the North Unionfield site, and the CCC-5 site despite large differences in aquifer transmissivity among the sites. The response of the freshwater/saltwater interface to changes in pumping for a given period, however, does appear to be sensitive to the simulated hydraulic properties.

The greatest response for the 48-year period (2002–50) occurs under the South Hollow well field, where it is assumed that the aquifer is more permeable at depth and the ratio of horizontal to vertical hydraulic conductivity is the lowest. The smallest response to increased pumping was calculated for the CCC-5 site, where the overall transmissivity is about 40 percent less than that simulated at the South Hollow well field. In addition, the aquifer beneath the CCC-5 site is much less permeable and the ratio of horizontal to vertical hydraulic conductivity is much higher than beneath the South Hollow well field.

Differences in aquifer transmissivity and in the hydraulic conductivity in the vicinity of the freshwater/saltwater interface, however, do not explain the difference in the responses of the freshwater/saltwater interface between the CCC-5 and North Unionfield sites. At these two sites, the simulated aquifer transmissivity values are similar (22,000 and 20,000 $ft^2/d$, respectively) and the horizontal hydraulic conductivity of the transition zones are the same, yet the model-calculated altitude of the freshwater/saltwater interface beneath the North Unionfield site increased by 58 ft, compared to only 14 ft beneath the CCC-5 site. The proximity of the North Unionfield site to the South Hollow and NTAFB wells may cause an increased effect on the altitude of the freshwater/saltwater interface beneath the well site.

Another possible explanation for the difference in the responses of the freshwater/saltwater interface may be the distribution of less permeable material beneath the pumping wells between the bottom of the well screen and the freshwater/saltwater interface rather than the overall transmissivity as determined by an aquifer test. There is a zone beneath the North Unionfield site about 90 ft thick with a simulated hydraulic conductivity value of 30 ft/d between the bottom of the proposed pumping well and the freshwater/saltwater interface. At the CCC-5 site, this zone of low hydraulic conductivity is assumed to be 150 ft thick. This zone could affect the response of the freshwater/saltwater interface to pumping at the CCC-5 site.

During a typical aquifer test, changes in water levels are measured in response to pumping over a specified period. The slow response of the freshwater/saltwater interface to changes in pumping, however, makes it impractical to measure the change in the freshwater/saltwater interface position in response to changes in pumping in the field. Therefore, analytical equations and numerical flow models that incorporate the available information on the subsurface hydrogeology must be used to predict the future effects of pumping on the movement of the freshwater/saltwater interface.

In this investigation, the assumptions about the distribution of aquifer properties throughout the flow lens are based on limited field data augmented by the relation between the regional depositional model of the glacial material and the model-simulated hydraulic-conductivity values. These results underscore the importance of developing a better understanding of the hydraulic properties of the subsurface materials through (1) well-designed aquifer tests, (2) detailed characterization of the lithology through the vertical extent of the aquifer down to the freshwater/saltwater interface, and (3) measurements of the pre-pumping depth to the freshwater/saltwater interface, prior to developing large-capacity pumping wells at the proposed well sites.

# Simulated Effects of Local Sea-Level Change Through Removal of a Tide-Control Structure

Tide-control structures were built along the mouths of river systems throughout Lower Cape Cod, Massachusetts, over the past 150 years to control mosquitoes, reclaim estuary land, and provide flood control. Roman and others (1987) determined that the tide-control structures in these river systems have resulted in the unintended effects of fish kills and increased mosquito-control problems. As a result, the National Park Service (NPS) is assessing the potential hydrological and ecological effects of removing the tide-control structures and restoring the river systems to their natural state, and will monitor the results of restoration projects to determine whether expectations are being met.

The removal of tide-control structures may affect hydraulic conditions in the upland areas adjacent to the areas of increased tidal flux. Ground-water altitudes, the depth to the interface between freshwater and saltwater, and ground-water discharge to freshwater streams may change. The effect on domestic water-supply wells in these areas will depend on well depths and the magnitude of the hydraulic changes.

A previous USGS investigation near the Herring River tide-control structure in Wellfleet, Massachusetts (fig. 1) (Fitterman and Dennehy, 1991), used surficial and borehole geophysical techniques to characterize the depth to the freshwater/saltwater interface and to assess the potential effect of the removal of the tide-control structure on the interface in the vicinity of an area of predicted increased tidal flux. Based on a predicted rise in water level of about 1.5 ft (Roman and others, 1987) and the measured position of the freshwater/saltwater interface (about 60 ft below NGVD 29) in the vicinity of Chequesset Neck (fig. 1), Fitterman and Dennehy (1991) concluded that the removal of the tide-control structure would have little effect on the total thickness of the freshwater lens.

The conclusion reached by Fitterman and Dennehy (1991), based on the present-day position of the freshwater/saltwater interface, was not supported by numerical-model simulations of the effects of removing the tide-control structure on the position of the underlying freshwater/saltwater interface. The numerical model developed in this study was used to assess the response of the flow system to moving the location of the tide-control structure from the Herring River to the upper reach of Mill Creek (fig. 29).

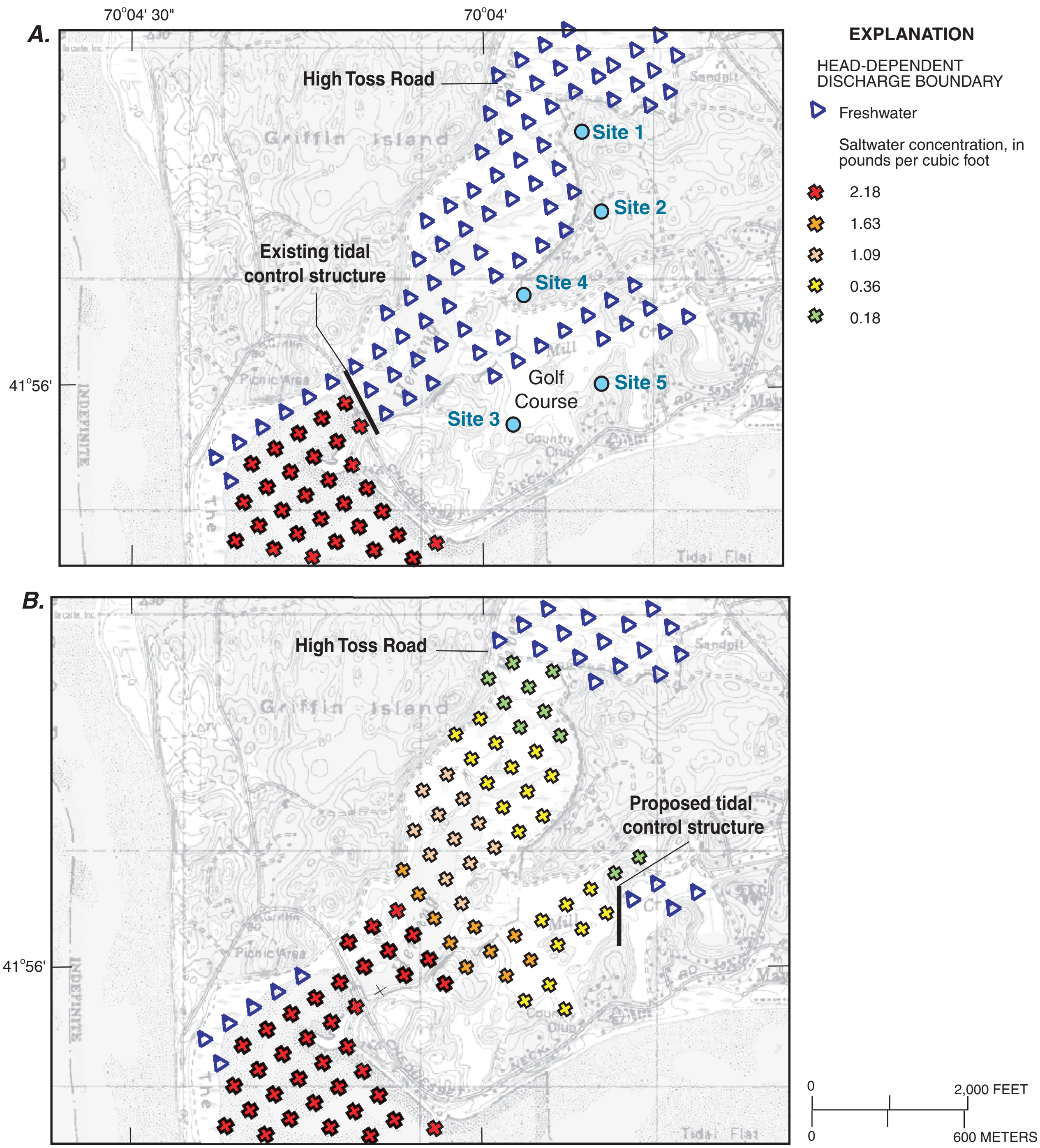
On the basis of hydrodynamic and salinity modeling, Spaulding and Grilli (2001) predicted that the removal of the two tidal gates and the complete opening of the sluice way on the tide-control structure on the Herring River [referred to as "case 5" in Spaulding and Grilli (2001)] will result in a landward flooding of seawater to an altitude at mean tide of about 2.5 ft above NGVD 29 as far inland as High Toss Road (fig. 29). Under current conditions (2002), the altitude of the Herring River is about 0.7 ft above NGVD 29 on the landward side of the tide-control structure (Spaulding and Grilli, 2001). If the structure is removed, the increases in stream stage and salinity may result in additional flooding of saline water into the lowland areas of Chequesset Neck in the vicinity of Mill Creek; these areas were salt marshes prior to the installation of the tidal restriction in 1907 (fig. 29*B*, see saltwater head-dependent discharge boundaries).

The mean sea level measured in September 2000 at the seaward side of the Herring River tide-control structure was about 1.5 ft above NGVD 29 (Spaulding and Grilli, 2001). The model was modified for this analysis by increasing the current sea-level position from 0.63 ft to 1.5 ft above NGVD 29 in the saltwater head-dependent discharge boundaries near the mouth of the Herring River (fig. 29*B*) in order to be consistent with the tidal position measured by Spaulding and Grilli (2001) at the mouth of the river. In addition, the pumping well used to irrigate the Chequesset Country Club golf course (fig. 29*A*, Site 3) was added to the model with an average annual pumping rate of 0.015 Mgal/d from a screened interval of 20 to 40 ft below NGVD 29.

A simulation was made with the golf-course well pumping at 0.015 Mgal/d and the new tidal altitude in the vicinity of the mouth of the Herring River set at 1.5 ft above NGVD 29 to establish the current position of the freshwater/saltwater interface. Similar modifications to the regional model, such as the simulated coastal boundary condition, could be made at other tidally restricted inlets and coves throughout the Lower Cape Cod study area if the tidal-altitude information was available; however, measuring these tidal positions throughout the study area was beyond the scope of the regional investigation.

Four scenarios were simulated to assess the potential effects of removing the tide-control structure on the position of the freshwater/saltwater interface in the vicinity of the Herring River. These simulations were made for a period of about 300 years from present to allow enough time for the flow system to reach equilibrium with respect to changes in the position of the freshwater/saltwater interface.

Five sites were selected at which to evaluate changes in the position of the interface in relation to the simulated current position of the interface (fig. 29*A*). Sites 1 and 2 are the locations of monitoring wells installed during the investigation conducted by Fitterman and Dennehy (1991). Two of the remaining three sites (Sites 3 and 4) are the locations of monitoring wells installed by the USGS in September 2003 to assist the NPS in monitoring possible changes in the interface from the removal of the tide-control structure.




Base from U.S. Geological Survey topographic quadrangles, Wellfleet, and Orleans, Massachusetts, Universal Transverse Mercator grid, Polyconic projection, zone 19 NAD 27, 1:25,000

**Figure 29.** Model-simulated boundary conditions with *A*, the existing Herring River tide-control structure and conditions in the Chequesset Neck area; and *B*, proposed tide-control structure and the variable saltwater concentrations used in simulations 1 and 4, Wellfleet, Massachusetts.

In the first simulation, the surface-water altitude was specified uniformly at 2.5 ft above NGVD 29 upstream in the Herring River from the tide-control structure to High Toss Road and from the mouth of Mill Creek to the proposed site of the new tide-control structure (fig. 29*B*). The increased stage of 2.5 ft above NGVD 29 is the stage calculated by Spaulding and Grilli (2001) for the removal of the two tidal gates and the complete opening of the sluice way on the tide-control structure. In addition to increasing the stage in the river and stream channels, new head-dependent boundary conditions with a water level of 2.5 ft above NGVD 29 were specified in the lowland areas that were salt marshes prior the installation of the tide-control structure in 1907 and now are occupied in part by the Chequesset Country Club golf course (fig. 29).

In the areas where increased tidal exchange was expected to increase the stream stage in Mill Creek and the Herring River and to flood the adjacent lowland areas, the specified saltwater concentrations at these head-dependent discharge boundaries were set equal to the concentrations calculated by Spaulding and Grilli (2001) (fig. 29*B*). The salt concentrations ranged from the high concentration of saltwater (2.18 lb/ft$^3$) at the tide-control structure to the low concentration of freshwater (0.0 lb/ft$^3$) near High Toss Road.

As a result of these simulated changes in stream stage and salt concentration, the model-calculated altitude of the freshwater/saltwater interface decreased between 8 and 36 ft (table 2) at the five monitoring sites relative to the simulated current conditions, despite the fact that salt concentrations shown in figure 29 were specified for areas that currently are freshwater.

In the second simulation, the stream stage of 2.5 ft above NGVD 29 was used again for the stage in Mill Creek, Herring River, and the adjacent flooded lowland areas. For the salt concentration, however, seawater was specified for all model cells assigned a head-dependent discharge boundary condition (see appendix), rather than the gradient shown in figure 29*B* and used in the first simulation.

Results from the second simulation show that the change in the specified salt concentration from that shown in figure 29*B* has a large effect on the position of the freshwater/saltwater interface. The altitude of the interface rose between 19 and 80 ft beneath the five monitoring sites (table 2) relative to the simulated current altitude.

In the third simulation, the salt concentration was specified to be the same as that of the second simulation. In this simulation, however, the stream stage in Mill Creek, the Herring River, and the adjacent flooded lowland areas was uniformly specified at 1.5 ft, the stage that would occur if the entire tide-control structure was removed, rather than only the tidal gates. In this simulation the altitude of the freshwater/saltwater interface increased between 3 and 8 ft beneath the five monitoring sites (table 2) relative to the current altitude. This effect was less than that calculated in the previous simulation with the same specified salt concentrations but a higher specified stage.

In the fourth simulation, a uniform stream stage of 1.5 ft above NGVD 29 was specified for Mill Creek, the Herring River, and the adjacent flooded lowland areas. The salt concentration specified in this simulation, however, was the gradient specified in the first simulation (fig. 29*B*). In this simulation, the altitude of the freshwater/saltwater interface increased between 5 and 10 ft beneath the five monitoring sites (table 2) relative to the simulated current altitude.

The results of the four model simulations show that the specified salt concentrations and the increased stream stage caused by the removal of the tidal gates and/or the tide-control structure can affect the simulated position of the freshwater/saltwater interface. The simulated thickness of the freshwater lens in the vicinity of the Herring River depended upon the combination of specified stream stage and salt concentration and changed considerably from the thickness simulated under current (2002) conditions.

**Table 2.** Model-calculated changes in the altitude of the freshwater/saltwater interface in the vicinity of the Herring River tide-control structure, Wellfleet, Massachusetts, in response to changes in simulated salt concentrations and stream stage

[All altitudes in feet above NGVD 29. Well site with model-cell location: Numbers in parentheses next to well-site number are model cell (row, column).]

| Simulation | Salinity | Surface-water altitude, in feet | Well site with model-cell location | | | | |
|---|---|---|---|---|---|---|---|
| | | | 1 (174, 57) | 2 (177, 56) | 3 (181, 49) | 4 (178, 52) | 5 (182, 53) |
| Current | Fresh | 0.7–1.5 | -111 | -119 | -62 | -80 | -85 |
| 1 | Variable | 2.5 | -119 | -128 | -81 | -116 | -100 |
| 2 | Salt | 2.5 | -31 | -82 | -43 | -46 | -29 |
| 3 | Salt | 1.5 | -103 | -111 | -55 | -77 | -80 |
| 4 | Variable | 1.5 | -103 | -110 | -52 | -72 | -80 |

The results from the first two simulations indicate that, with the higher value of the stage height, the position of the freshwater/saltwater interface is sensitive to the specified salt concentration. When in addition, it was assumed that the salt concentration gradually decreased landward of the tide-control structure (Spaulding and Grilli, 2001) (fig. 29*B*), the altitude of the freshwater/saltwater interface decreased. Thus, the freshwater lens became thicker in the vicinity of the Herring River. When the salt concentration was specified to equal the concentration of seawater, the altitude of the freshwater/saltwater increased. Thus, the freshwater lens became thinner in the vicinity of the Herring River.

In the third and fourth simulations, the stream stage was set at 1.5 ft above NGVD 29, the current altitude of seawater measured on the seaward side of the tide-control structure. In these simulations, it was assumed that the entire tide-control structure was removed and the stream stage as far inland as High Toss Road is the same as the current altitude of seawater. The salt concentrations were varied from a uniform concentration similar to seawater from the tidal restriction to High Toss Road (simulation 3) to the concentration gradient predicted by Spaulding and Grilli (2001) (simulation 4). In these simulations, the altitude of the freshwater/saltwater interface increased by less than 10 ft (table 2) relative to the current altitude. The difference in the salt concentration between simulations 3 and 4 did not appear to have a substantial effect on the position of the freshwater/saltwater interface.

The specified salt concentrations appeared to have the greatest effect on the simulated position of the freshwater/saltwater interface when the stream stage was set at 2.5 ft above NGVD 29, the predicted mean tide altitude if only the tidal gates were removed from the tide-control structure (Spaulding and Grilli, 2001). When the stream stage was set at 1.5 ft above NGVD 29, however, the position of the freshwater/saltwater interface did not appear to be affected greatly by the change in specified salt concentrations.

The results of these simulations indicate that setting the stream stage in the Herring River, Mill Creek, and the adjacent lowland areas at an altitude of 2.5 ft above NGVD 29 resulted in a decrease in ground-water discharge and a subsequent rise in the water table in the vicinity of the Herring River. When the specified salt concentrations were spatially variable and low, such as those predicted by Spaulding and Grilli (2001) (fig. 29*B*), and the stream stage was set at 2.5 ft above NGVD 29, the altitude of the freshwater/saltwater interface decreased. When the simulated seawater concentration was high and the stream stage was set at 2.5 ft above NGVD 29, however, the altitude of the interface rose and resulted in the thinning of the freshwater lens with time. It should be noted that this simulation represents a worst-case scenario with respect to the movement of the freshwater/saltwater interface and that the hydrodynamic modeling simulations of Spaulding and Grilli (2001) do not predict a salt concentration of seawater as far inland as High Toss Road.

When the stream stage was set at the present sea-level altitude of 1.5 ft above NGVD 29, the altitude of the freshwater/saltwater interface increased slightly from the current altitude regardless of the specified salt concentration. The reason for the small response in the interface to the increased stream stage of 1.5 ft above NGVD 29 may be that this stream stage is similar to the present water levels on both the landward and seaward sides of the tide-control structure and, thus, the changes in salt concentrations resulted in only small changes in the interface position.

# Summary

Federal, State, and local officials responsible for managing and protecting water resources of the Lower Cape Cod aquifer system are concerned that a shift to large-capacity, centralized municipal supplies from private small-capacity wells may create the potential for unacceptable declines in water table and pond altitudes, decreases in ground-water discharge to streams and coastal areas, and saltwater intrusion. In response to these concerns, the U.S. Geological Survey (USGS) in cooperation with the National Park Service, Massachusetts Executive Office of Environmental Affairs, Cape Cod Commission, and the Towns of Eastham, Wellfleet, Truro, and Provincetown began an investigation in 2000 to improve the understanding of the hydrogeology of the Lower Cape Cod aquifer system and to assess the effects of changing ground-water pumping and recharge conditions on ground-water flow in Lower Cape Cod.

A regional numerical flow model was developed as part of this investigation to assist in the analysis of freshwater and saltwater flow for changing pumping, recharge, and sea-level conditions. The model was used to determine water budgets, flow directions, and the position and movement of the freshwater/saltwater interface throughout the study area for current conditions. The model also was used to assess potential effects of increased pumping for proposed pumping well locations on ponds, streams, coastal areas, and the position of the freshwater/saltwater interface to demonstrate how the model can serve as a tool that can be used by the State and local managers to assess possible effects of proposed water-management strategies on Lower Cape Cod.

The primary results of the investigation are listed below:

- Of the approximately 68 Mgal/d of freshwater that recharges the aquifer system, about 68 percent of this water moves through the ground-water flow system and discharges directly to the coast, 31 percent reaches the coast as streamflow, and 1 percent discharges to public-supply wells. The distribution of streamflow varies greatly between flow lenses. In the Chequesset flow lens, for example, streamflow accounts for over half of the total discharge to the coast, whereas in the Pilgrim flow lens, streamflow accounts for less than 10 percent of the total discharge to the coast.
- The depth to the freshwater/saltwater interface varies throughout the study area and is largely controlled by the areal extent of the recharge areas of the flow lenses. In general, the larger the area receiving recharge, the greater the depth to the freshwater/saltwater interface; however, changes in subsurface geology or the presence of streams can affect this relation. In areas such as the western portion of the Chequesset flow lens, the depth to the freshwater/saltwater interface is shallow compared to the eastern portion of the lens where no streams are present. In the Nauset flow lens, which is underlain by thick deposits of silt and clay, the water-table altitude is higher than in the other flow lenses and the depth to the freshwater/saltwater interface is greater.
- Sea-level rise may increase water levels and streamflows throughout the Lower Cape Cod aquifer system, and yet decrease the depth to the freshwater/saltwater interface. The amount of increase in water levels in response to sea-level rise is affected by non-tidal streams because increases in streamflow limit the water-level increase. In areas where the water-level rise is limited by increased streamflow, the depth to the freshwater/saltwater interface decreases.
- Pumping from large-capacity municipal-supply wells increases the potential for effects on surface-water bodies, and, therefore, requires careful planning of future pumping and wastewater-disposal locations and rates. Pumping wells that are upgradient of surface-water bodies have the potential to capture water that would otherwise discharge to these surface-water bodies, thereby reducing streamflow and pond levels. Kettle-hole ponds, such as Duck Pond in Wellfleet, that are located near the top of a water-table mound, appear to be more susceptible to changing pumping and recharge conditions than the kettle-hole ponds closer to the coast or near streams.

# References Cited


Alley, W.M., Reilly, T.E., and Franke, O.L., 1999, Sustainability of ground-water resources: U.S. Geological Survey Circular 1186, 79 p.

Barlow, P.M., 1996, Particle-tracking analysis of contributing areas of public-supply wells in simple and complex flow systems, Cape Cod, Massachusetts: U.S. Geological Survey Water-Supply Paper 2434, 66 p.

Barlow, P.M., and Hess, K.M., 1993, Simulated hydrologic responses of the Quashnet River stream-aquifer system to proposed ground-water withdrawals: U.S. Geological Survey Water-Resources Investigations Report 93-4064, 52 p.

Cambareri, T.C., Belfit, G., Janik, D.S, Irwin, P., Campbell, B., McCaffery, D., 1989, Truro/Provincetown aquifer assessment and groundwater protection plan: Cape Cod Planning and Economic Development Commission, 65 p.

Camp Dresser & McKee, Inc., 1985, Update of the water supply plan for the Provincetown water system: Boston, MA, Camp Dresser & McKee, Inc., variously paged.

Donnelly, J.P., and Bertness, M.D., 2001, Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise: Proceedings of the National Academy of Sciences, v. 98, no. 25, p.14218–14223.

Eichner, Eduard, Cambareri, T.C., Livingston, Kenneth, Sobczak, R.V., Smith, B.S., 1997, Pamet River investigation groundwater assessment study: Cape Cod Commission, 39 p.

Environmental Partners Group, Inc., 2002, Modeling of saltwater upconing to support water management planning: Hingham, MA, Environmental Partners Group, Inc., June 12, 2002, variously paged.

Farnsworth, R.K., Thompson, E.S., and Peck, E.L., 1982, Evaporation atlas for the contiguous 48 states: National Oceanic and Atmospheric Administration Technical Report NWS 33, 26 p.

Fitterman, D.V., and Dennehy, K.F., 1991, Verification of geophysically determined depths to saltwater near the Herring River (Cape Cod National Seashore), Wellfleet, Massachusetts: U.S. Geological Survey Open-File Report 91-321, 47 p.

Foster, D.S., and Poppe, L.J, 2003, High resolution seismic reflection surveys in the near shore of Outer Cape Cod, Massachusetts: U.S. Geological Survey Open-File Report 03-235, accessed November 15, 2003, at http://pubs/usgs.gov/of/2003/of03-235

Ghyben, W.B., 1888, Nota in verband met de voorgenomen putboring nabij: Amsterdam, Tijdschrift van Let Koninklijk Inst. Van Ing.

Granato, G.E., and Smith, K.P., 2002, Robowell—Providing accurate and current water-level and water-quality data in real time for protecting ground-water resources: U.S. Geological Survey Fact Sheet FS -053-02. 6 p.

Guo, W., and Langevin, C.D., 2002, User's guide to SEAWAT: A computer program for simulation of three-dimensional variable-density ground-water flow: U.S Geological Open-File Report 01-434, 77 p.

Guswa, J. H., and LeBlanc, D.R., 1985, Digital models of ground-water flow in the Cape Cod aquifer system, Massachusetts: U. S. Geological Survey Water-Supply Paper 2209, 112 p.

Guswa, J.H., and Londquist, C.J., 1976, Potential for development of ground water at a test site near Truro, Massachusetts: U.S. Geological Survey Open-File Report 76-614, 22 p.

Herzberg, A., 1901, Die Wasserversorgung einer Nordseebaden: J. Gasbeleucht. Wasserversorg., v. 44, p. 815–819.

Horsley, S.W., Cambareri, T.C., and M. Robinson, 1985, Administrative letter to the Cape Cod Planning and Economic Development Commission: variously paged.

Intergovernmental Panel on Climate Change, 2001, The regional impacts on climate change—An assessment of vulnerability, accessed on April 30, 2003, at http://www.grida.no/climate/ipcc

Kaye, C.A., 1964, Outline of Pleistocene geology of Martha's Vineyard, Massachusetts: U.S. Geological Survey Professional Paper 501-C, p. 134–139.

Landuse Collaborative, Inc., 1996, Outer Cape capacity study: Lower Cape Water Management Task Force, variously paged.

LeBlanc, D.R., Garabedian, S.P., Hess, K. M., Gelhar, L.W., Quadri, R.D., Stollenwerk, K.G., and Wood, W.W., 1991, Large-scale natural-gradient tracer test in sand and gravel, Cape Cod, Massachusetts—1. Experimental design and observed tracer movement: Water Resources Research 27, no. 5, p. 895–910.

LeBlanc, D.R., Guswa, J.H., Frimpter, M.H., and Londquist, C.J., 1986, Ground-water resources of Cape Cod, Massachusetts: U.S. Geological Survey Hydrologic Investigations Atlas 692, 4 pls.

Letty, D.F., 1984, Ground water and pond levels, Cape Cod, Massachusetts, 1950–1982: U.S. Geological Survey Open-File Report 84-719, 81 p.

Martin, Larry, 1993, Investigation of effects of ground water withdrawals from the Pamet and Chequesset aquifers, Cape Cod National Seashore: Fort Collins, CO, National Park Service, 47 p.

Massachusetts Executive Office of Environmental Affairs–Community Preservation Initiative, 1999, accessed March 17, 2003, at http://www.mass.gov/envir/cpa

Masterson, J.P., and Walter, D.A., 2000, Delineation of ground-water recharge areas, western Cape Cod, Massachusetts: U.S. Geological Survey Water-Resources Investigations Report 00-4000, 1 pl.

Masterson, J.P., Walter, D.A., and LeBlanc, D.R., 1998, Delineation of contributing areas to selected public-supply wells, western Cape Cod, Massachusetts: U. S. Geological Survey Water-Resources Investigations Report 98-4237, 45 p.

Masterson, J.P., Stone, B.D., Walter, D.A., and Savoie, Jennifer, 1997, Hydrogeologic framework of western Cape Cod, Massachusetts: U.S. Geological Survey Hydrologic-Investigations Atlas 741, 1 pl.

Masterson, J.P., and Barlow, P.M., 1996, Effects of simulated ground-water pumping and recharge on ground-water flow in Cape Cod, Martha's Vineyard, and Nantucket Island basins, Massachusetts: U. S. Geological Survey Water-Supply Paper 2447, 79 p.

McCobb, T.M., and Weiskel, P.K., 2003, Long-term hydrologic monitoring protocol for coastal ecosystems: U.S. Geological Survey Open-File Report 02-497, 94 p.

Moore, M.V., Pace, M.L., Mather, J.R., Murdoch, P.S., Howarth, R.W., Folt, C.L., Chen, C.Y., Hemond, H.F., Flebbe, P.A., and Driscoll, C.T., 1997: Potential effects of climate change on freshwater ecosystems of the New England/Mid-Atlantic Region: Hydrological Processes, v. 11, p. 025–947.

National Oceanic and Atmospheric Administration, 2001, accessed August 15, 2001, at http://lwf.ncdc.noaa.gov/oa/pub/data/coop-precip-massachusett.txt

National Oceanic and Atmospheric Administration, 2003, accessed January 22, 2003, at http://www.co-ops.nos.noaa.gov/seatrnds.html

Nuttle, W.K., and Portnoy, J.W., 1992, Effect of rising sea level on runoff and groundwater discharge to coastal ecosystems: Estuarine, Coastal and Shelf Science, v. 34, p. 203–212.

Oldale, R.N., 1992, Cape Cod and the islands—The geologic story: East Orleans, MA, Parnassus Imprints, 205 p.

Oldale, R.N., and Barlow, R.A., 1986, Geologic map of Cape Cod and the Islands, Massachusetts: U.S. Geological Survey Miscellaneous Investigations Series Map I-1763, 1 pl., scale 1:100,000.

Oldale, R.N., and O'Hara, C.J., 1984, Glaciotectonic origin of the Massachusetts coastal end moraines and a fluctuating late Wisconsinan ice margin: Geological Society of America Bulletin, v. 95, p. 61–74.

Portnoy, J.W., Winkler, M.G., Sanford, P.R., and Farris, C.N., 2001, Kettle pond data atlas for Cape Cod National Seashore—Paleoecology and modern water chemistry: National Park Service, U.S. Department of Interior, 119 p.

Reilly, T.E., Frimpter, T.E., LeBlanc, D.R., and Goodman, A.S., 1987, Analysis of steadystate saltwater upconing with application at Truro wellfield, Cape Cod, Massachusetts: Ground Water, v. 25, no. 2, p. 82–92.

Reilly, T.E., and Goodman, A.S., 1985, Analysis of saltwater upconing beneath a pumping well: Journal of Hydrology, v. 89, p. 169–202.

Reilly, T.E., and Pollock, D.W., 1993, Factors affecting areas contributing recharge in shallow aquifers: U.S. Geological Survey Water-Supply Paper 2412, 21 p.

Roman, C.T., Garvine, R.E., Marteinsdottir, G., Nikula, B., Nuttle, W.K., and Orson, R.A., 1987, An evaluation of alternatives for estuarine restoration management—The Herring River ecosystem (Cape Cod National Seashore): New Brunswick, NJ, Center for Coastal Environmental Studies, Rutgers–The State University of New Jersey, 304 p.

Smith, N.D., and Ashley, G.M., 1985, Proglacial lacustrine environment, *in* Ashley, G.M., Shaw, J., and Smith, N.D., eds., Glacial sedimentary environments: Tulsa, OK, Society of Paleotologists and Mineralogists, Short Course No. 16, 246 p.

Sobczak, R.V., and Cambareri, T.C., 1998, Water resources of outer Cape Cod—Final report of the Lower Cape Water Management Task Force: Barnstable, MA, Cape Cod Commission, 94 p.

Sobczak, R.V., Cambareri, T.C., and Portnoy, J.W., 2003, Physical hydrology of selected vernal pools and kettle hole ponds in the Cape Cod National Seashore, Massachusetts: Barnstable, MA, Cape Cod Commission, 174 p.

Spaulding, M.L., and A. Grilli, 2001, Hydrodynamic and salinity modeling for estuarine habitat restoration at Herring River, Wellfleet, Massachusetts: Prepared for the National Park Service, U.S. Department of Interior, Wellfleet, MA, Cape Cod National Seashore, 93 p.

Strahler, A.N., 1972, The environmental impact of groundwater use on Cape Cod, Impact Study III: Orleans, MA, Association for the Preservation of Cape Cod, 68 p.

Theiler, E.R., and Hammar-Klose, E.S., 2000, National assessment of coastal vulnerability to sea-level rise; Preliminary results for the U.S. Atlantic coast: U.S. Geological Survey Open-File Report 99-593, 1 p.

Walter, D.A., Masterson, J.P., and LeBlanc, D.R., 2002, Simulated pond-aquifer interactions under natural and stressed conditions near Snake Pond, Cape Cod, Massachusetts: U.S. Geological Survey Water-Resources Investigations Report 99-4174, 35 p.

Uchupi, Elazar, Giese, G.S., and Aubrey, D.G., 1996, The late Quaternary construction of Cape Cod, Massachusetts—A reconsideration of the W.M. Davis Model: Geological Society of America Special Paper 309, 69 p.

Woodworth, J.B., and Wigglesworth, Edward, 1934, Geography and geology of the region including Cape Cod, the Elizabeth Islands, Nantucket, Martha's Vineyard, No Mans Land, and Block Island: Harvard College Museum of Comparative Zoology Memoir 52, 332 p.

Zeigler, J.M., Tuttle, S.D., Tasha, H.J., and Giese, G.S., 1965, The age and development of the Provincelands Hook, outer Cape Cod, Massachusetts: Limnology and Oceanography, Redfield Volume, p. R298–R311.

# APPENDIX: DEVELOPMENT OF GROUND-WATER MODEL

# Contents



# Figures



# Tables

# Development of Ground-Water Model

A numerical model of ground-water flow was developed to simulate freshwater and saltwater flow in the four flow lenses of Lower Cape Cod, Massachusetts. This model is based on the USGS computer program SEAWAT (Guo and Langevin, 2002), which simulates variable-density, transient ground-water flow in three dimensions. A USGS particle-tracking program (MODPATH) developed by Pollock (1994) was used to calculate the initial locations of water particles that discharge to public-supply wells, ponds, streams, and coastal areas under changing pumping and recharge scenarios.

## Model Grid

The finite-difference grid for the numerical model consists of uniformly spaced model cells that are 400 ft on a side. The grid consists of 320 rows and 110 columns and extends across all four flow lenses. The model has 23 layers that extend from the water table to a uniform depth of 500 ft below NGVD 29 with the thickness of the vertical layers ranging from 15 to 25 ft (table 1-1).

Although the unconsolidated sand-and-gravel and silt-and-clay sediments that constitute the Lower Cape Cod aquifer system extend to bedrock, the depth to bedrock is in most places much greater than the depth to the freshwater/saltwater interface. Therefore, the freshwater flow system is bounded below by the transition zone between freshwater and underlying saltwater rather than bedrock, as is the case in the upper and middle parts of Cape Cod (Guswa and LeBlanc, 1985; LeBlanc and others, 1986; Masterson and Barlow, 1996).

**Table 1-1.** Vertical layers and horizontal and vertical hydraulic conductivity for the flow model of the Lower Cape Cod aquifer system, Cape Cod, Massachusetts.

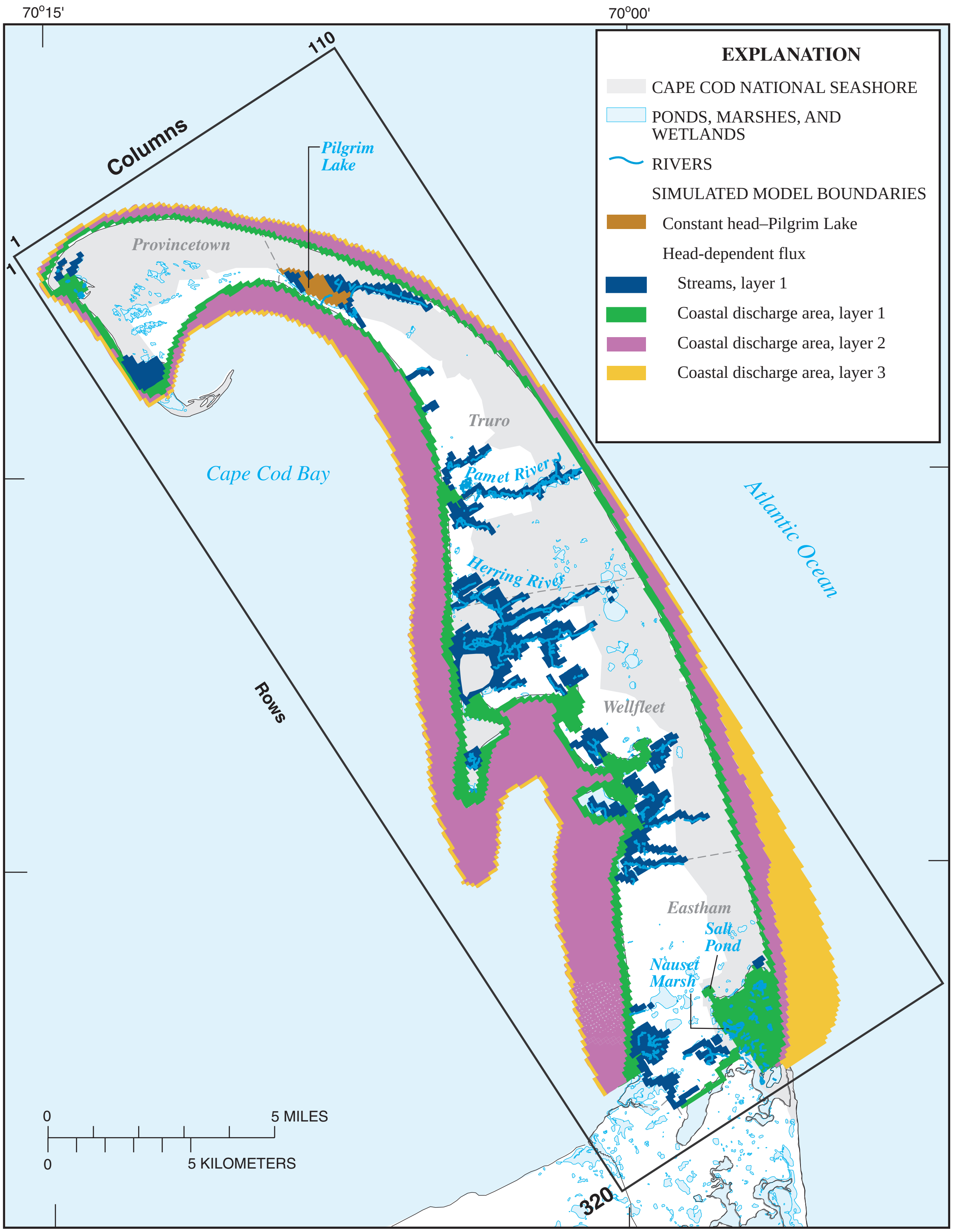
[**Altitude of layer bottom:** Altitude relative to NGVD 29. K, hydraulic conductivity; ft, foot; ft/d, foot per day]

| Model layer | Altitude of layer bottom (ft) | Horizontal K (ft/d) | Range in vertical K (ft/d) |
|---|---|---|---|
| 1 | -5 | 350–150 | 117–30 |
| 2 | -20 | 350–30 | 117–0.3 |
| 3 | -40 | 250–30 | 50–0.3 |
| 4 | -60 | 250–30 | 50–0.3 |
| 5 | -80 | 200–10 | 40–0.01 |
| 6 | -100 | 125–10 | 12.5–0.01 |
| 7 | -120 | 125–10 | 12.5–0.01 |
| 8 | -140 | 125–10 | 12.5–0.01 |
| 9 | -160 | 125–10 | 12.5–0.01 |
| 10 | -180 | 70–10 | 2.33–0.01 |
| 11 | -200 | 70–10 | 2.33–0.01 |
| 12 | -225 | 70–10 | 2.33–0.01 |
| 13 | -250 | 70–10 | 2.33–0.01 |
| 14 | -275 | 70–10 | 2.33–0.01 |
| 15 | -300 | 70–10 | 2.33–0.01 |
| 16 | -325 | 70–10 | 2.33–0.01 |
| 17 | -350 | 30–10 | 0.3–0.01 |
| 18 | -375 | 10–10 | 0.01–0.01 |
| 19 | -400 | 10–10 | 0.01–0.01 |
| 20 | -425 | 10–10 | 0.01–0.01 |
| 21 | -450 | 10–10 | 0.01–0.01 |
| 22 | -475 | 10–10 | 0.01–0.01 |
| 23 | -500 | 10–10 | 0.01–0.01 |

## Boundary Conditions

The boundaries of the numerical model of fresh ground-water flow in the Lower Cape Cod aquifer system coincide with the physical boundaries of the flow system. The upper boundary of the model is the water table, which is a free surface boundary that receives spatially variable recharge from precipitation and wastewater disposal. The lower boundary of the freshwater flow system is the transition between freshwater and saltwater; this boundary is calculated by the numerical model. The arbitrary bottom altitude of 500 ft below NGVD 29 was specified as a no-flow boundary for this analysis; this altitude generally is above the top of the bedrock surface through most of Lower Cape Cod (Oldale, 1969; B.D. Stone, U.S. Geological Survey, written commun., 1992) and yet is sufficiently deep not to affect the model-calculated position and movement of the freshwater/saltwater interface beneath the four flow lenses of Lower Cape Cod.

The lateral boundaries of the model are the coastal discharge areas that surround the study area. These discharge areas are represented as head-dependent flux boundaries that extend out into Cape Cod Bay and the Atlantic Ocean in the top three layers of the model (fig. 1-1). The distribution of these boundaries in the upper three layers is based on estimates of the offshore seabed bathymetry.




Base from U.S. Geological Survey Digital Line Graphs, and topographic quadrangles, Provincetown, Wellfleet, and Orleans, Massachusetts, 1:25,000, Polyconic projection, NAD 1927, Zone 19

**Figure 1-1.** Model extent and distribution of simulated boundary conditions of ground-water-flow model of Lower Cape Cod, Massachusetts.

Because these boundaries represent saltwater, an initial salt concentration representative of seawater (2.18 lbs/$ft^3$) was specified for these model cells. The freshwater heads specified as part of these boundary conditions were calculated as equivalent freshwater heads, thereby creating hydrostatic conditions with depth for this coastal boundary. These equivalent freshwater heads are based upon an assumed altitude of sea level. Analyses were conducted that adjusted sea level by 0.104 in/yr (2.65 mm/yr) from an assumed altitude of 0.0 ft in 1929 (NGVD 29) to an altitude of 1.05 ft above NGVD 29 in 2050. The hydraulic conductance of the seabed deposits was calculated for each model cell containing a coastal boundary as described in Harbaugh and McDonald (1996) as

$$C = \frac{(K)(W)(L)}{(M)}, \qquad (1\text{-}1)$$

where

$C$ is hydraulic conductance of the seabed ($ft^2$/d);
$K$ is vertical hydraulic conductivity of seabed deposits (ft/d);
$W$ is width of the model cell containing the seabed (ft);
$L$ is length of the model cell containing the seabed (ft); and
$M$ is thickness of seabed deposits (ft).

The vertical hydraulic conductivity ($K$) of the seabed deposits in most of the study area was assumed to be 1 ft/d, which is consistent with model simulations of similar coastal discharge areas in other areas on Cape Cod (Masterson and others, 1998). In the area occupied by Salt Pond and Nauset Marsh, it was assumed that there were thick deposits of low-permeability material (J.A. Colman, U.S. Geological Survey, oral commun., 2002) and the vertical hydraulic conductivity was set to 0.1 ft/d. The thickness of the seabed deposits was assumed to be half the thickness of the model cell containing the boundary.

In addition to head-dependent boundaries specified along coastal discharge areas, SEAWAT requires that head-dependent flux boundaries be specified along the lateral extent of the model and extended vertically throughout the model domain. These boundaries have an initial salt concentration of 2.18 lbs/$ft^3$ and equivalent freshwater heads that increase with depth on the basis of the vertical layering of the model. These boundaries are assumed to be seaward of the maximum extent of freshwater discharge and serve as a constant reservoir of saltwater required for the SEAWAT computations of the position and movement of the freshwater/saltwater interface (Guo and Langevin, 2002).

Streams and wetlands also were specified in layer 1 of the model as head-dependent flux boundaries because it is assumed that, under natural hydrologic conditions, these surface-water features receive ground-water discharge and that water is effectively removed from the ground-water-flow system once it discharges to a stream.

The altitudes of the streambeds and wetlands were estimated from 1:24,000 USGS topographic maps. The altitude estimate for coastal wetlands ranged from 2 ft to 4 ft above NGVD 29, this range is consistent with coastal marshes along Cape Cod Bay, such as Namskaket Marsh in Orleans (DeSimone and others, 1998). It was assumed for this analysis that the water levels in the streams and wetlands are approximately the same as the altitude of the streambeds and wetland surfaces because the water depths in the surface-water bodies in the study area typically are small (less than 2 ft).

The simulated hydraulic conductance was based on estimates from previous modeling investigations on Cape Cod (Guswa and LeBlanc, 1985; Barlow and Hess, 1993). The hydraulic conductance for the streambed and wetlands was calculated by the same equation (eq. 1.1) used to calculate the conductance of seabed sediments. The hydraulic conductivity was initially set at 30 ft/d; the width was 10 ft for the streams and 400 ft for the wetlands; and the length was 400 ft for both streams and wetlands.

These values were adjusted to improve the model calibration to the available field data. A previous analysis on western Cape Cod (Masterson and others, 1997) showed that large changes in estimated hydraulic conductance and streambed altitudes can affect the location and amount of ground-water discharge to surface-water bodies. Local-scale analyses of specific surface-water bodies may require more detailed hydrologic data collection than was possible for this regional analysis.

## Initial Conditions

To simulate both freshwater and saltwater flow, an initial estimate of the altitude of the interface between freshwater and saltwater must be assumed. The better this estimate is, the less will be the simulation time required to achieve a final, stable solution of the interface (Guo and Langevin, 2002). The initial estimate of the position of interface in this analysis was from the coast to a uniform depth of 250 ft below NGVD 29.

The simulation of flow and solute transport with SEAWAT requires transient conditions (Guo and Langevin, 2002). Transient simulations were made from the initial concentration until a reasonable approximation of the altitude of the freshwater/saltwater interface was achieved and the model had reached a quasi-steady-state condition with respect to simulated hydrologic conditions. This solution was used as the pre-pumping condition for subsequent analysis of changing pumping and recharge conditions from 1907 to 2050.

The simulation of solute transport was made by an implicit finite-difference solution with advective transport only (Guo and Langevin, 2002) to calculate the position and movement of the transition zone between freshwater and saltwater. This solution was necessary because of the size of the model domain in this analysis, but also results in larger numerical dispersion, or smearing of the simulated transition between freshwater and saltwater.

The resulting numerical dispersion from these simulations was similar in magnitude to the observed depth of the transition zone in the field data for non-pumping conditions (generally less than 40 ft thick). In the vicinity of simulated pumping wells, the numerical dispersion appeared to be much larger than the observed transition zone beneath the actual pumping wells (fig. 17). In these instances, the 50-percent salt-concentration contour was assumed to be a reasonable approximation of the interface between freshwater and saltwater.

## Model Stresses

The hydrologic stresses simulated in the model consist of recharge and ground-water pumping. These stresses are simulated for transient conditions from an assumed quasi-steady-state initial condition for the year 1906, the year before the start of continuous public-supply pumping on Lower Cape Cod. Simulations also were made for average annual recharge and pumping conditions, and for seasonally varying recharge and pumping conditions.

The length of the stress periods simulated in this transient model depended upon the analysis. A long initial period was simulated as part of the model-calibration process to solve for the position and movement of the freshwater/saltwater interface and to provide a best fit of model-calculated water levels and streamflow to observed data. The simulated elapsed time required for model calibration was not recorded because the iterative nature of the "trial-and-error" model-calibration process rendered this time meaningless. Once the model was calibrated, a 23-year stress period was simulated from 1907–29 for average pumping conditions followed by annual stress periods from 1930 to 2002 to simulate actual pumping conditions over that period.

Future scenarios were simulated for a 48-year-long stress period from 2002 to 2050 to assess the effects of pumping from proposed and existing pumping sites at proposed pumping rates. Because the flow model cannot simulate steady-state conditions, a long stress period of 1,500,000 days (more than 4,000 years) was simulated to provide enough elapsed time to delineate the sources of water to public-supply wells, ponds, streams, and coastal areas under quasi-steady-state conditions by using the particle-tracking method detailed in Pollock (1994).

Forty-eight monthly stress periods were used to simulate the effects of in-season and off-season pumping and recharge. The 4-year period was chosen to provide enough time to minimize the transient effects of the initial condition and the monthly stress periods were used to minimize the potential effects of time discretization on the model-calculated results. Although there were 12 monthly stress periods per year, the pumping and recharge stresses were simulated at two different rates for in-season and off-season. The off-season pumping and recharge rates were simulated for seven stress periods per year (October through April) and the in-season stress rates were simulated for five stress periods per year (May through September).

### Recharge

Recharge rates specified in the numerical model were based on estimates of precipitation and wastewater disposal through domestic septic systems and sewage-treatment facilities. The recharge estimate of 24 in/yr was based on an assumption that about 45 percent of the annual precipitation of 42 in/yr on Lower Cape Cod is lost to evaporation and plant transpiration and the remaining 55 percent enters the aquifer system as recharge, as was assumed in previous studies on western Cape Cod (Masterson and others, 1998). The average annual precipitation rate was obtained from the weather stations in Provincetown from 1948 to 1992 (National Oceanic and Atmospheric Administration, 2001), and South Truro from 1980 to 2000 (National Atmospheric Deposition Program Station: MA-01).

The simulated recharge rate was adjusted downward for ponds, streams, and wetlands to account for increased evaporation from these surface-water bodies. Farnsworth and others (1982) estimate the average free-water-surface potential evaporation from surface-water bodies throughout Cape Cod to be about 28 in/yr. The combination of this evaporation rate with 42 in/yr of precipitation results in a recharge rate of 14 in/yr from these surface-water bodies to the underlying aquifer.

An additional source of recharge is wastewater returned to the aquifer system from septic systems. Most of the communities of Lower Cape Cod obtain their drinking water from domestic wells and the resulting wastewater is returned to the aquifer system through on-site septic systems. For the purpose of this analysis, it was assumed that the water pumped from and returned to the same part of the aquifer system resulted in no effect on the flow system, and, therefore, recharge from septic wastewater was not simulated in the model.

The drinking water for the residents in Provincetown and a small portion of North Truro is provided by a centralized municipal supply and the resulting wastewater is returned to the aquifer from on-site domestic septic systems. This wastewater return results in an increase in aquifer recharge in these areas. Eighty-five percent of the pumped water was recharged to the aquifer in proportion to the areas zoned for residential land use in those Traffic Analysis Zones (fig. 12) that are on municipal supply. The 15-percent reduction accounts for the consumptive use of water. In North Truro, municipal supply was provided to TAZ 8; about 0.1 Mgal/d of additional recharge was distributed over these areas for current conditions (2002). The remainder of the municipal supply from the North Truro area is exported to Provincetown in the Pilgrim lens, where it is the primary source of drinking water for its residents. Septic-system disposal in the Pilgrim flow lens recharges an additional 0.7 Mgal/d of water to the aquifer in TAZ 2 -6 (fig. 12) for current conditions. Provincetown is in the process of sewering the downtown area and plans to dispose treated wastewater from a centralized sewage-treatment facility to infiltration beds (fig. 12). Future pumping scenarios were based on the assumption that all of the water used in Provincetown (less the 15-percent consumptive use) is returned to the Pilgrim flow lens through the infiltration beds.

Simulations for Eastham and Wellfleet also were based on future pumping and recharge scenarios. In these towns the wastewater returned to the aquifer system was apportioned by pumping rates and distributed on the basis of areas of residential and commercial land use in TAZ 16–23 in Eastham and TAZ 12–13 in Wellfleet (fig. 12). The total wastewater returned to the aquifer system in these areas was 85 percent of the water withdrawn at the proposed pumping rates for these towns.

Changes in the annual average recharge from precipitation were made to simulate seasonal variability in evapotranspiration and the effect of reduced annual precipitation in response to a prolonged drought. The distribution of in-season and off-season recharge simulated in the model was based in part on the results of model calibration and on previous analyses conducted in the northeastern United States. One example is the Toms River area in central New Jersey where it was determined that about 4 in. of the 26 in. of annual recharge occurred from May to September (Nicholson and Watts, 1997).

The average recharge in this study was divided into a 7-month period of low evapotranspiration and 5-month period of high evapotranspiration. It was assumed that that 75 percent of the total annual recharge, or 18 in., occurs during the 7-month period from October to April. It was assumed that 25 percent of the total annual recharge, or 6 in., occurs during the remaining 5-month period from May to September when evapotranspiration is high.

Wastewater and surface-water recharge rates also were varied seasonally. The wastewater disposal rates were highest for the in-season period when pumping rates were assumed to be highest. For the 7-month off-season period, wastewater disposal rates were lowest because pumping rates were lowest. All 14 in/yr of recharge into surface-water bodies was assumed to occur during the off-season period; no recharge was simulated during the in-season period.

## Pumping

The first public-water-supply system on Lower Cape Cod was installed in 1893 at the site known as the Auxillary well (fig. 12). This well was pumped at a rate of about 0.15 Mgal/d until it was discontinued in 1906 because of poor water quality in the Pilgrim flow lens (Craig Weigand, Provincetown Water Dept., written commun., 2001). Because this well had not been in operation for almost 100 years, it was not simulated in this investigation.

The earliest year of pumping simulated in this investigation is 1907, the first year that the the Knowles Crossing well field (also know as the Old North Truro well field) was in operation. This well field was developed in the Pamet flow cell to meet the demand for drinking water in Provincetown (Craig Weigand, Provincetown Water Dept., written commun., 2001).

By 2002, the town of Provincetown had developed an additional water supply at South Hollow (fig. 12), which has been used since 1955, and had used the North Truro Air Force Base wells 4 and 5 seasonally from May to October since 1978. Provincetown also used the CCNS No. 4 site (fig. 12) from 1978 through 1985 to meet the shortfall in supply caused by the gasoline spill near the South Hollow well field in 1978.

Ground-water withdrawals were simulated for the public-supply wells at their actual annual average pumping rates from 1930 to 2002 with the exception of years 1940–43 and 1950, when no data was available (table 1-2). For these years, average annual pumping rates were estimated by taking the average of the pumping from the year before and the year after the missing data.

From 1907 to 1929, the pumping from the Knowles Crossing well field was averaged and simulated as one pumping rate of 0.255 Mgal/d for this 23-year period. This was done to coincide with the model simulation of sea-level rise. For the purpose of this analysis, sea level prior to 1929 was assumed to be 0.0 ft NGVD 29 and to have risen 0.104 in/yr from 1929 to 2002.

Future pumping scenarios were simulated for a total of six additional sites in Eastham, Wellfleet, and Truro (fig. 12). The pumping rates that were simulated at these sites were varied for each scenario to determine potential effects from pumping at these sites on surface-water bodies, the position of the freshwater/saltwater interface, and the source of water to other pumping wells.

**Table 1-2.** Annual average pumping rates for Provincetown Water Department public-supply wells from 1907 to 2002, Truro, Massachusetts.

[All pumping rates are in Mgal/d. --, well not in operation]

| Year | Pumping wells | | | | Year | Pumping wells | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Knowles Crossing | South Hollow | NTAFB 4-5 | CCNS Site 4 | | Knowles Crossing | South Hollow | NTAFB 4-5 | CCNS Site 4 |
| 1907–29 | †0.255 | -- | -- | -- | 1967 | 0.416 | 0.28 | -- | -- |
| 1930 | .336 | -- | -- | -- | 1968 | .312 | .446 | -- | -- |
| 1931 | .32 | -- | -- | -- | 1969 | .349 | .476 | -- | -- |
| 1932 | .381 | -- | -- | -- | 1970 | .448 | .445 | -- | -- |
| 1933 | .424 | -- | -- | -- | 1971 | .33 | .507 | -- | -- |
| 1934 | .503 | -- | -- | -- | 1972 | .311 | .485 | -- | -- |
| 1935 | .396 | -- | -- | -- | 1973 | .295 | .556 | -- | -- |
| 1936 | .43 | -- | -- | -- | 1974 | .44 | .426 | -- | -- |
| 1937 | .398 | -- | -- | -- | 1975 | .396 | .479 | -- | -- |
| 1938 | .373 | -- | -- | -- | 1976 | .503 | .449 | -- | -- |
| 1939 | .415 | -- | -- | -- | 1977 | .497 | .5 | -- | -- |
| 1940 | *.410 | -- | -- | -- | 1978 | .489 | .12 | 0.11 | 0.2 |
| 1941 | *.410 | -- | -- | -- | 1979 | .427 | 0 | .167 | .285 |
| 1942 | *.410 | -- | -- | -- | 1980 | .409 | 0 | .138 | .342 |
| 1943 | *.410 | -- | -- | -- | 1981 | .279 | .196 | .128 | .263 |
| 1944 | .404 | -- | -- | -- | 1982 | .196 | .246 | .008 | .378 |
| 1945 | .374 | -- | -- | -- | 1983 | .267 | .224 | .102 | .267 |
| 1946 | .428 | -- | -- | -- | 1984 | .271 | .199 | .174 | .25 |
| 1947 | .517 | -- | -- | -- | 1985 | .254 | .42 | .112 | .052 |
| 1948 | .554 | -- | -- | -- | 1986 | .093 | .592 | .131 | -- |
| 1949 | .453 | -- | -- | -- | 1987 | .083 | .595 | .183 | -- |
| 1950 | *.465 | -- | -- | -- | 1988 | .062 | .656 | .109 | -- |
| 1951 | .476 | -- | -- | -- | 1989 | .1 | .622 | .107 | -- |
| 1952 | .456 | -- | -- | -- | 1990 | .084 | .64 | .108 | -- |
| 1953 | .399 | -- | -- | -- | 1991 | .093 | .764 | .08 | -- |
| 1954 | .488 | -- | -- | -- | 1992 | .151 | .64 | .1 | -- |
| 1955 | .228 | 0.364 | -- | -- | 1993 | .236 | .72 | .112 | -- |
| 1956 | .213 | .348 | -- | -- | 1994 | .266 | .666 | .123 | -- |
| 1957 | .287 | .373 | -- | -- | 1995 | .329 | .604 | .176 | -- |
| 1958 | .235 | .371 | -- | -- | 1996 | .262 | .765 | .121 | -- |
| 1959 | .284 | .384 | -- | -- | 1997 | .238 | .632 | .164 | -- |
| 1960 | .131 | .485 | -- | -- | 1998 | .268 | .5 | .163 | -- |
| 1961 | .251 | .425 | -- | -- | 1999 | .277 | .496 | .128 | -- |
| 1962 | .172 | .482 | -- | -- | 2000 | .248 | .483 | .132 | -- |
| 1963 | .267 | .466 | -- | -- | 2001 | .15 | .571 | .11 | -- |
| 1964 | .229 | .455 | -- | -- | 2002** | .2 | .57 | .165 | -- |
| 1965 | .227 | .43 | -- | -- | | | | | |
| 1966 | .285 | .378 | -- | -- | | | | | |

†Pumping rates for 23-year period averaged.

*Pumping rate estimated by averaging the 1 year before and 1 year after the year with missing pumping data.

**Estimated for 2002.

Agricultural, industrial, and small-capacity domestic pumpage was not simulated in the model because information for daily demand pumping less than 0.1 Mgal/d is not readily available, and because water pumped for these uses typically is returned to the aquifer system within the same model cells from which it is pumped. Therefore, it was assumed that pumping for these uses had a negligible effect on the aquifer system. Thus, the simulation of such small-scale pumping wells is beyond the scope of this regional analysis.

## Hydraulic Properties

The hydraulic properties required as input data for the ground-water modeling in this investigation are horizontal hydraulic conductivity, vertical hydraulic conductivity, porosity, specific yield, and storage coefficient. The determination of hydraulic properties was based largely on previous investigations throughout Cape Cod (Guswa and LeBlanc, 1985; LeBlanc and others, 1986; Barlow and Hess, 1993; Masterson and Barlow, 1996; Masterson and others, 1997)

Hydraulic conductivities initially used in the numerical model were assigned on the basis of available aquifer test, lithologic, and geologic information. The Lower Cape Cod aquifer system has few deep wells; therefore, lithologic information necessary to determine hydraulic properties at depth is limited. Hydraulic properties of the aquifer material in areas with little or no lithologic information were estimated on the basis of the geologic processes that formed the glacial sediments of the ground-water-flow system.

The relation between geologic framework and hydraulic conductivities from a similar hydrogeologic setting on western Cape Cod (Masterson and others, 1997) was used for this analysis on Lower Cape Cod. The range in hydraulic conductivities in each of the model layers is shown in table 1-1. Hydraulic conductivity for model cells containing ponds was set equal to 50,000 ft/d. Streambed leakances were determined from estimates of the vertical conductance of the streambeds.

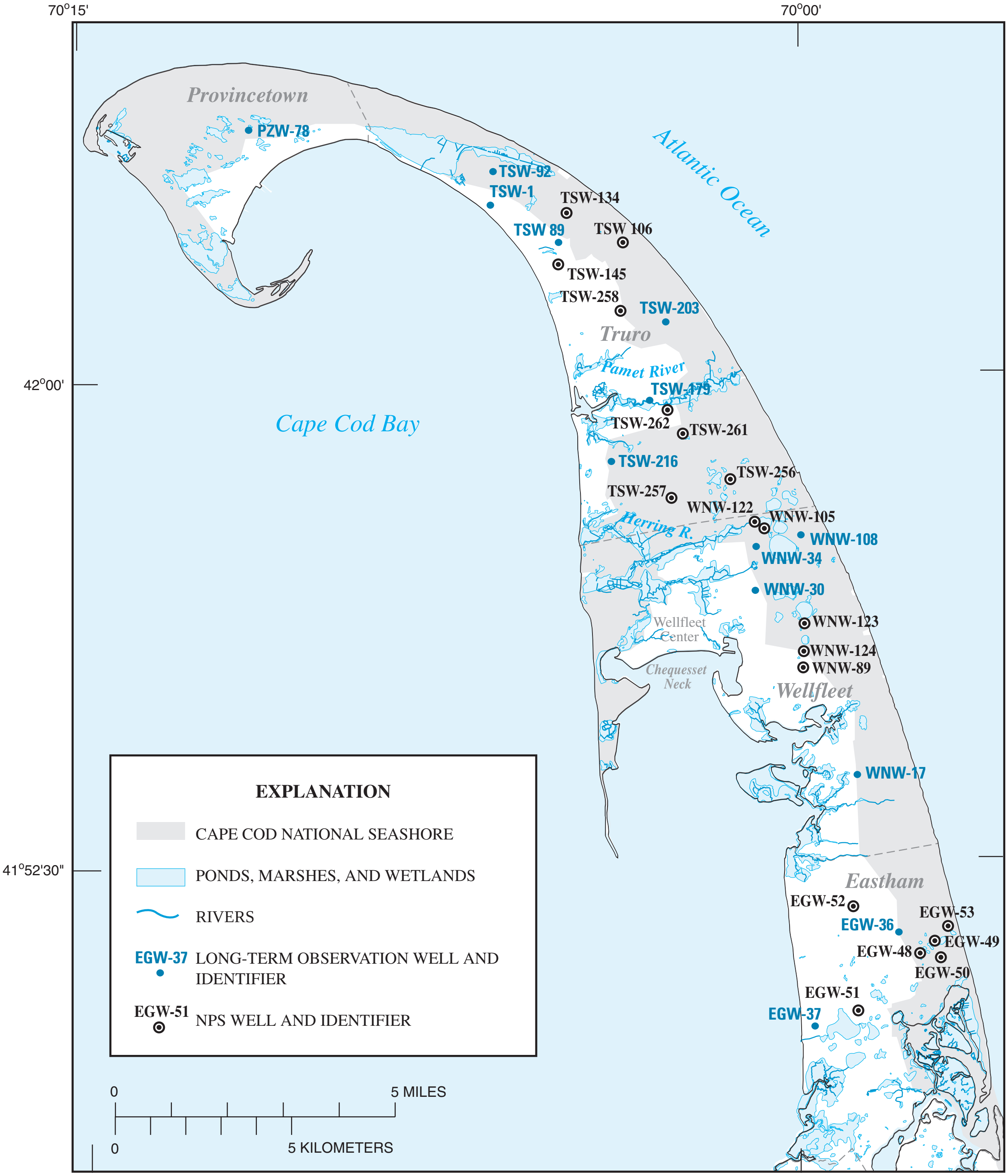
The uniform porosity value of 0.3 used in the model is consistent with previous porosity estimates throughout Cape Cod (Garabedian and others, 1988; LeBlanc and others, 1991; Masterson and Barlow, 1996). The uniform specific-yield value of 0.25 is consistent with Moench (1994) and a uniform storage-coefficient value of $1x10^{-5}$ was based on Barlow and Hess (1993). In the cells representing ponds, the porosity, specific yield, and storage coefficient were set to values of 1.0 to account for the high storage capacity assumed for the ponds.

## Model Calibration

The numerical model was calibrated to measured water levels, pond levels, streamflows, and the position of the freshwater/saltwater interface. Numerous water level measurements were available from over 50 years of data available in the USGS online database (http://waterdata.usgs.gov/nwis/gw). Streamflow data were limited to recent measurements made as part of the newly implemented National Park Service Long-Term Coastal Ecosystem Monitoring Program (McCobb and Weiskel, 2003). Data on the freshwater/saltwater interface were more difficult to obtain than water levels and streamflows because of the cost associated with drilling wells to the freshwater/saltwater interface. Available data from previous USGS studies (LeBlanc and others, 1986) and the town of Provincetown (Environmental Partners, Inc., 2002) were used for calibration as well as results from borehole geophysical logging and offshore marine resistivity surveys conducted as part of this investigation.

The water-level data used for model calibration included records from the USGS–Cape Cod Commission long-term-monitoring well network, and data from three synoptic water-level measurements on November 1975, June 2001, and May 2002. The long-term monitoring well network consists of 13 wells measured monthly or bi-monthly since 1976 (figs. 19 and 1-2). Data from this network were not extensive enough to calibrate the model adequately. The mean and median water levels from these sites, however, were used to identify the dates of near-average water levels in more extensive synoptic measurements.

Based on a comparison of the water levels measured in the long-term observation wells during more extensive synoptic measurements made in November 1975 and June 2001 with the mean and median water levels in the long-term observation wells, it was determined that these measurements were made when water levels were near-average conditions (see red symbols on fig. 1-6). The November 1975 synoptic water-level measurements were made in 46 wells; however, all but 13 of these measurements were made in the Pamet flow lens (fig. 1-3). The June 2001 measurements were made in the 13 long-term observation wells and in 19 observation wells from the newly implemented National Park Service monitoring program (fig. 1-2) (McCobb and Weiskel, 2003). In addition to the 32 water-level measurements, pond levels were measured in 10 kettle-hole ponds and streamflows were measured at 8 stream-gaging sites throughout the study area (fig. 1-4).




Base from U.S. Geological Survey Digital Line Graphs, and topographic quadrangles, Provincetown, Wellfleet, and Orleans, Massachusetts, 1:25,000, Polyconic projection, NAD 1927, Zone 19

**Figure 1-2.** Locations of long-term observation wells and Cape Cod National Seashore coastal ecosystem wells, Lower Cape Cod, Massachusetts.

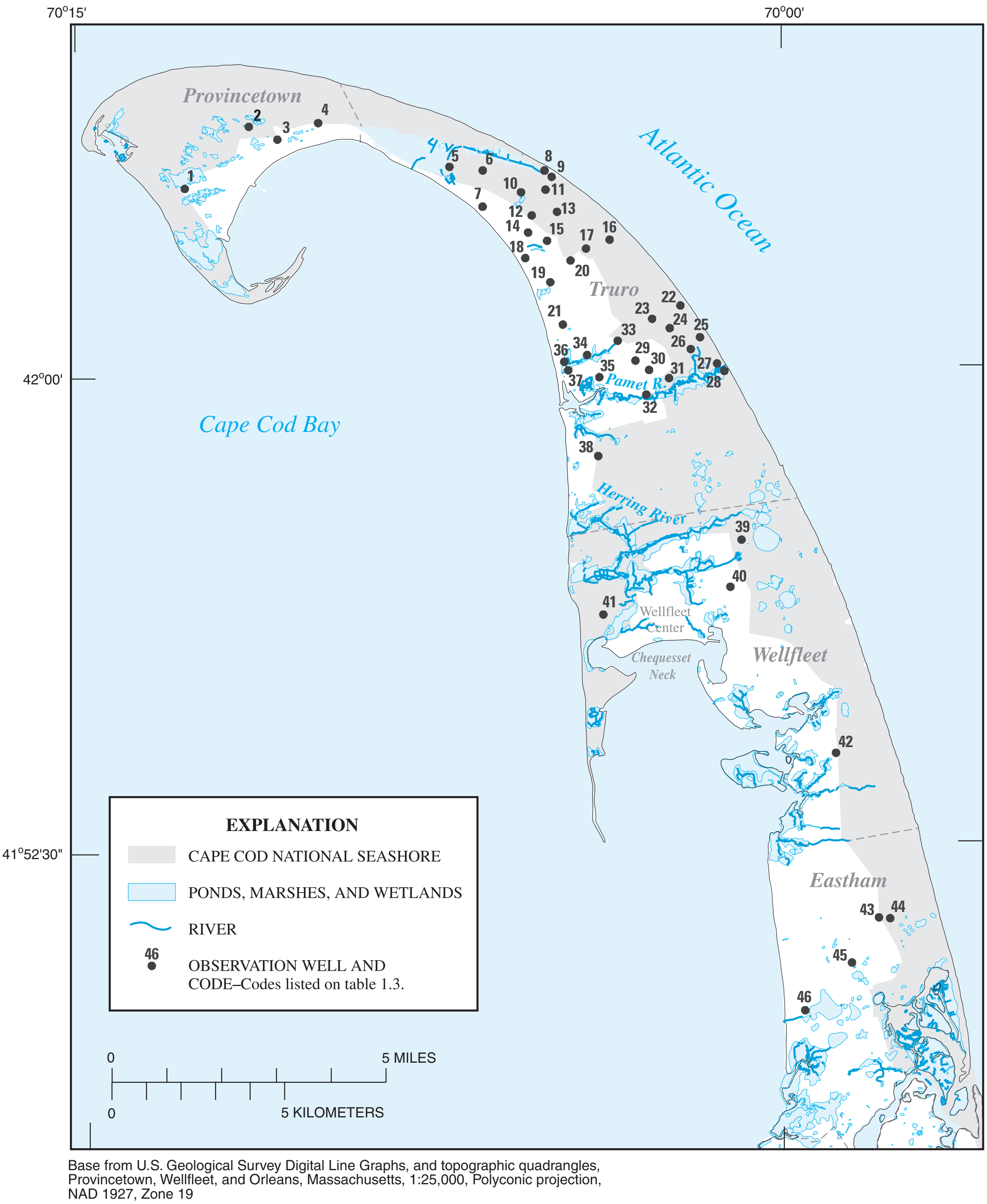
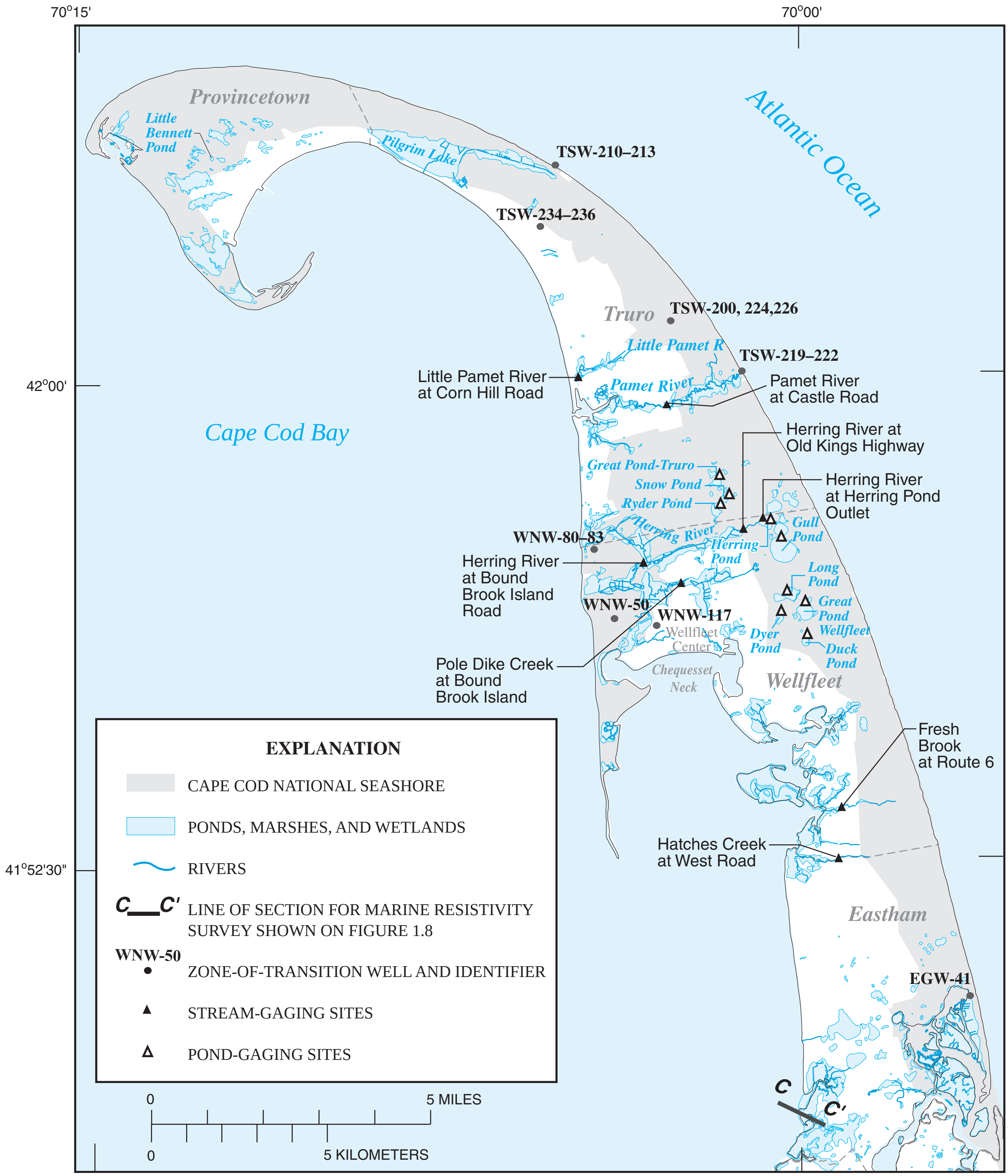



**Figure 1-3.** Locations of observation wells measured in November 1975, Lower Cape Cod, Massachusetts.




Base from U.S. Geological Survey Digital Line Graphs, and topographic quadrangles, Provincetown, Wellfleet, and Orleans, Massachusetts, 1:25,000, Polyconic projection, NAD 1927, Zone 19

**Figure 1-4.** Locations of pond and stream-gaging sites, zone-of-transition monitoring wells, and line of marine resistivity measurements, Lower Cape Cod, Massachusetts.

As part of this investigation, a new network was established with the addition of 50 new observation wells and other available wells not included previously in the November 1975 and June 2001 synoptic measurements to provide the most extensive water-level, pond-level, and streamflow-monitoring network to date. This network consists of 91 observation wells, 10 ponds, and 8 stream-gaging sites and was measured over the period of May 22–25, 2002 (fig. 1-5) (Michaud and Cambareri, 2003). These measurements, unlike the measurements made in November 1975 and June 2001, indicated conditions below the long-term average at most observation wells (fig. 1-6). Therefore, these measurements were used primarily for comparing model-calculated flow directions and the locations of the tops of the water-table mounds of the flow lenses in areas where data were previously unavailable (Scott Michaud, Cape Cod Commission, written commun., 2003).

The streamflow data available for model calibration consisted of the monthly streamflow measurements made by the National Park Service from March 2000 to September 2000 and from May 2001 to August 2001 (Evan Gwilliams, National Park Service, written commun., 2001); September 28, 2000 (McCobb and Weiskel, 2003); August 1996 (Eichner and others, 1997), and measurements made as part of the synoptic water-level measurements in May 2002.

Model calibration to the available streamflow data was complicated in that many of the streams are tidally affected and measurements of freshwater baseflow were difficult to obtain. For this investigation, the September 28, 2000, measurements (McCobb and Weiskel, 2003) and the measurements made by USGS personnel during the May 2002 synoptic event were used for model calibration.

In the case of the Pamet River, which is affected by a tide-control gate that prevents the landward flow of saltwater at high tide, measurements made by Eichner and others (1997) were adjusted by a mass-balance method in order to account for the duration in which the tide-control gate was open for each tidal cycle. A freshwater discharge of about 3.0 $ft^3/s$ was calculated for the August 1996 measurements reported in Eichner and others (1997) (S.P. Garabedian, U.S. Geological Survey, written commun., 2002).

The third set of calibration data consisted of the positions of the freshwater/saltwater interface. These data consisted of the zone-of-transition data measured at eight locations throughout the study area from August 1973 to September 1979 (LeBlanc and others, 1986) (fig. 1-4), geophysical measurements at eight deep monitoring wells at the three well fields in the Pamet flow lens from September 2000 to December 2001, and the marine resistivity measurements made as part of this investigation in March 2001 (line of section shown on fig. 1-4, section in fig. 1-8).

The goal of the model-calibration process was to develop a model that provides results that compare reasonably well with available field data, so that simulations of future conditions can be made. Hydraulic conductivity was the parameter whose values were adjusted as part of this model-calibration process. Although simulated changes in aquifer recharge would have affected the flow system and the match between the model and observed field data, the recharge rate specified in this model was held constant throughout the model-calibration process. Determining the actual temporal and spatial changes in recharge was beyond the scope of this investigation.

An initial attempt was made to calibrate the flow model with a simulated constant sea-level altitude of 0.0 ft above NGVD 29. It was determined from these initial simulations that although the model provided a reasonable approximation of the position of the freshwater/saltwater interface relative to the available field data, the long-term water-level data consistently were lower than the water levels measured in the field.

Changes in simulated hydraulic-conductivity values needed to increase model-calculated water levels resulted in an unacceptable match to data for the freshwater/saltwater interface positions beneath the pumping wells in the Pamet flow lens. Once the sea level rise of 0.104 in/yr (2.65 mm/yr) was incorporated into the simulations, the model-calculated water levels were a better match with the observed water levels (table 1-3) without a substantial change in the position of the freshwater/saltwater interface.

The model-calculated ground-water and pond levels are generally in close agreement with the average values reported on the long-term network wells, and with values measured in November 1975 and June 2001 (table 1-3). The means of the absolute error between the measured and model-calculated water levels for the average values reported for the long-term network wells, and for the measurements made in November 1975 and in June 2001 were 0.50, 0.57, and 0.72 ft, respectively; these errors correspond to about 3 to 4 percent of the total relief of the water table of the Lower Cape Cod aquifer system. The median error between measured and model-calculated water levels for these three datasets were -0.02, 0.17, and 0.07 ft (table 1-3).

A comparison between the model-calculated and measured streamflow data is presented in table 1-4. Adjustments were not made to the model to closely match this streamflow data because of the limited number of measurements and complications with obtaining reasonable flow estimates; it was impossible to determine streamflows for average conditions. Therefore, these streamflow data were not emphasized in the model-calibration process, but were used as a general guide.

The comparison between model-calculated and measured positions of the freshwater/saltwater interface is complicated by the thickness of the transition between freshwater and saltwater in the observed data and the model-calculated spreading of the interface as a result of numerical dispersion. Therefore, it is difficult to quantify an absolute altitude of the interface in the field data and in the model-calculated results.

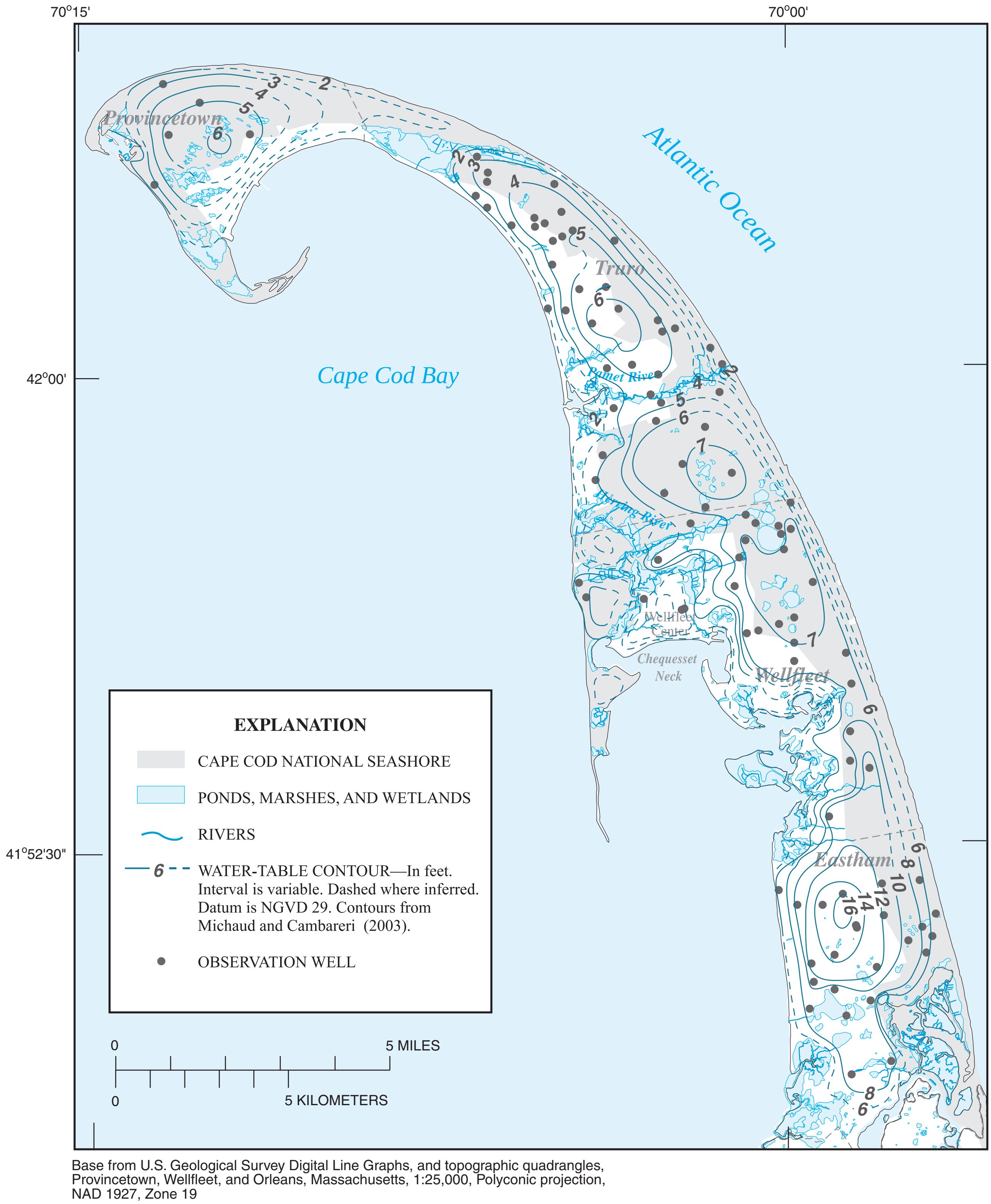



**Figure 1-5.** Locations of observation wells, water-table altitude, and configuration, May 2002, Lower Cape Cod, Massachusetts.

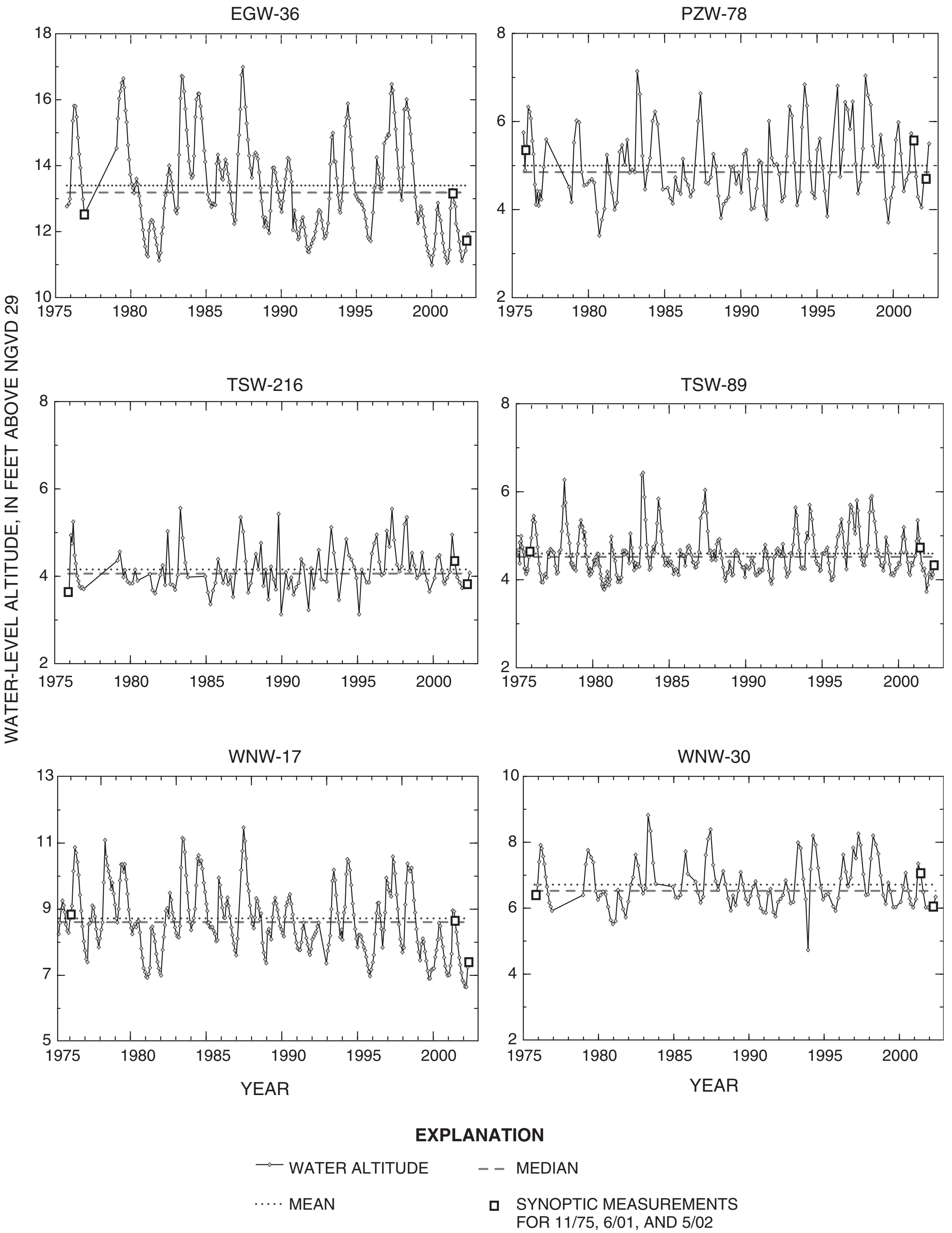



**Figure 1-6.** Water-level altitudes at long-term observation wells EGW-36, PZW-78, TSW-216, TSW-89, WNW-17, and WNW-30, Lower Cape Cod, Massachusetts. Location of wells shown in figure 1-2.

**Table 1-3.** Measured water levels for selected observation wells and kettle-hole ponds in the modeled area in November 1975, and June 2001, and the median water-level altitudes of the long-term observation wells for measured period of record and model-calculated water-level altitudes for simulated current (2002) pumping and recharge conditions, Lower Cape Cod, Massachusetts.

[**Map code, November 1975:** Map code for figure 1-3. All values in feet above NGVD 29. NI, not included in network; --, not measured]

| Wells and ponds | Model | | | Map code, November 1975 | Long-term measured | Long-term measured minus model | November 1975 measured | November 1975 measured minus model | June 2001 measured | June 2001 measured minus model |
|---|---|---|---|---|---|---|---|---|---|---|
| | Row | Column | Layer | | | | | | | |
| **Pilgrim Flow Lens** | | | | | | | | | | |
| PZW-71 | 34 | 19 | 1 | 1 | -- | -- | 4.60 | -0.14 | -- | -- |
| PZW-74 | 38 | 55 | 3 | 4 | -- | -- | 3.08 | .17 | -- | -- |
| PZW-77 | 36 | 44 | 3 | 3 | -- | -- | 3.53 | -.3 | -- | -- |
| PZW-78 | 30 | 38 | 2 | NI | 4.85 | -0.74 | 5.35 | -.25 | 5.57 | -0.02 |
| Little Bennett Pond | 30 | 28 | 1 | NI | -- | -- | -- | -- | 6.94 | 1.00 |
| **Pamet Flow Lens** | | | | | | | | | | |
| TSW-1 | 76 | 76 | 5 | 7 | 2.51 | 0.2 | 2.69 | 0.38 | 2.57 | 0.26 |
| TSW-87 | 98 | 87 | 3 | 20 | -- | -- | 3.85 | -.29 | -- | -- |
| TSW-89 | 91 | 85 | 2 | 15 | 4.52 | -.08 | 4.64 | .04 | 4.83 | .23 |
| TSW-92 | 69 | 81 | 1 | 6 | 3.39 | -.02 | 3.57 | .16 | 3.55 | .14 |
| TSW-94 | 81 | 91 | 2 | 11 | -- | -- | 4.68 | .74 | -- | -- |
| TSW-106 | 99 | 98 | 2 | 16 | -- | -- | 4.86 | 1.26 | 4.11 | .5 |
| TSW-107 | 64 | 75 | 1 | 5 | -- | -- | 2.19 | .15 | -- | -- |
| TSW-126 | 78 | 86 | 5 | 10 | -- | -- | 4.18 | -.15 | -- | -- |
| TSW-132 | 79 | 94 | 1 | 9 | -- | -- | 3.85 | 1.28 | -- | -- |
| TSW-134 | 87 | 91 | 1 | 13 | -- | -- | 4.91 | .34 | 5.34 | .77 |
| TSW-136 | 84 | 85 | 4 | 12 | -- | -- | 4.35 | -.36 | -- | -- |
| TSW-140 | 87 | 82 | 1 | 14 | -- | -- | 4.12 | -.2 | -- | -- |
| TSW-142 | 92 | 78 | 2 | 18 | -- | -- | 3.52 | .75 | -- | -- |
| TSW-145 | 96 | 82 | 2 | NI | -- | -- | -- | -- | 4.29 | .07 |
| TSW-153 | 98 | 92 | 1 | 17 | -- | -- | 5.07 | .59 | -- | -- |
| TSW-157 | 100 | 80 | 4 | 19 | -- | -- | 4.52 | .74 | -- | -- |
| TSW-162 | 110 | 77 | 1 | 21 | -- | -- | 4.32 | .11 | -- | -- |
| TSW-165 | 118 | 73 | 1 | 36 | -- | -- | 3.18 | .51 | -- | -- |
| TSW-166 | 120 | 73 | 1 | 37 | -- | -- | 3.84 | 1 | -- | -- |
| TSW-168 | 119 | 78 | 1 | 34 | -- | -- | 4.8 | .05 | -- | -- |
| TSW-170 | 121 | 86 | 1 | 33 | -- | -- | 5.84 | -.41 | -- | -- |
| TSW-173 | 122 | 103 | 2 | 22 | -- | -- | 4.37 | 1.77 | -- | -- |
| TSW-174 | 125 | 98 | 1 | 24 | -- | -- | 5.42 | .54 | -- | -- |
| TSW-176 | 127 | 87 | 1 | 29 | -- | -- | 5.44 | -.48 | -- | -- |
| TSW-177 | 125 | 78 | 1 | 35 | -- | -- | 4.26 | .03 | -- | -- |
| TSW-179 | 135 | 85 | 1 | 32 | 4.62 | 1.04 | 4.51 | .93 | 4.99 | 1.41 |
| TSW-181 | 135 | 92 | 1 | 31 | -- | -- | 4.72 | .42 | -- | -- |
| TSW-183 | 132 | 100 | 1 | 26 | -- | -- | 4.77 | .87 | -- | -- |
| TSW-184 | 131 | 103 | 1 | 25 | -- | -- | 4.45 | 1.92 | -- | -- |
| TSW-186 | 138 | 103 | 1 | 27 | -- | -- | 4.61 | 2.39 | -- | -- |
| TSW-203 | 121 | 96 | 2 | 23 | 5.72 | .08 | 5.5 | -.14 | 6.27 | .63 |
| TSW-210 | 77 | 94 | 6 | 8 | -- | -- | 3.36 | .76 | -- | -- |
| TSW-218 | 130 | 89 | 1 | 30 | -- | -- | 5.14 | -.24 | -- | -- |
| TSW-258 | 113 | 90 | 1 | NI | -- | -- | -- | -- | 7.83 | 1.49 |

**Table 1-3.** Measured water levels for selected observation wells and kettle-hole ponds in the modeled area in November 1975, and June 2001, and the median water-level altitudes of the long-term observation wells for measured period of record and model-calculated water-level altitudes for simulated current (2002) pumping and recharge conditions, Lower Cape Cod, Massachusetts.—Continued

[**Map code, November 1975:** Map code for figure 1-3. All values in feet above NGVD 29. NI, not included in network; --, not measured]

| Wells and ponds | Model | | | Map code, November 1975 | Long-term measured | Long-term measured minus model | November 1975 measured | November 1975 measured minus model | June 2001 measured | June 2001 measured minus model |
|---|---|---|---|---|---|---|---|---|---|---|
| | Row | Column | Layer | | | | | | | |
| **Chequesset Flow Lens** | | | | | | | | | | |
| TSW-216 | 141 | 67 | 1 | 38 | 4.05 | -0.55 | 3.64 | -0.96 | 4.35 | -0.25 |
| TSW-219 | 140 | 104 | 6 | 28 | -- | -- | 2.68 | .24 | -- | -- |
| TSW-256 | 160 | 89 | 1 | NI | -- | -- | -- | -- | 8.69 | .16 |
| TSW-257 | 156 | 75 | 1 | NI | -- | -- | -- | -- | 6.64 | -.81 |
| TSW-261 | 145 | 85 | 1 | NI | -- | -- | -- | -- | 7.68 | .18 |
| TSW-262 | 138 | 86 | 2 | NI | -- | -- | -- | -- | 4.21 | -.07 |
| WNW-30 | 184 | 77 | 5 | 40 | 6.53 | -.56 | 6.5 | -.59 | 7.06 | -.03 |
| WNW-34 | 176 | 85 | 3 | 39 | 7.98 | .74 | 7.88 | .64 | 8.5 | 1.24 |
| WNW-78 | 173 | 48 | 3 | 41 | -- | -- | -- | -- | 2.96 | .12 |
| WNW-89 | 206 | 79 | 2 | NI | -- | -- | -- | -- | 7.36 | -2.12 |
| WNW-105 | 176 | 85 | 1 | NI | -- | -- | -- | -- | 6.29 | -.71 |
| WNW-108 | 180 | 96 | 1 | NI | 7.52 | 1.36 | -- | -- | 8.16 | 2 |
| WNW-122 | 171 | 89 | 1 | NI | -- | -- | -- | -- | 6.64 | -.12 |
| WNW-123 | 198 | 85 | 2 | NI | -- | -- | -- | -- | 8.89 | -.33 |
| WNW-124 | 203 | 81 | 2 | NI | -- | -- | -- | -- | 9.01 | -.68 |
| Duck Pond | 200 | 83 | 1 | NI | -- | -- | -- | -- | 7.97 | -1.6 |
| Dyer Pond | 192 | 81 | 1 | NI | -- | -- | -- | -- | 9.27 | .53 |
| Great-Truro Pond | 157 | 86 | 1 | NI | -- | -- | -- | -- | 9.06 | .36 |
| Great-Wellfleet Pond | 194 | 87 | 1 | NI | -- | -- | -- | -- | 8.61 | -.25 |
| Gull Pond | 179 | 90 | 1 | NI | -- | -- | -- | -- | 6.78 | -.2 |
| Herring Pond | 172 | 90 | 1 | NI | -- | -- | -- | -- | 6.28 | -.62 |
| Long Pond | 189 | 86 | 1 | NI | -- | -- | -- | -- | 8.76 | .11 |
| Ryder Pond | 163 | 82 | 1 | NI | -- | -- | -- | -- | 8.09 | .1 |
| Snow Pond | 162 | 85 | 1 | NI | -- | -- | -- | -- | 8.74 | .37 |
| **Nauset Flow Lens** | | | | | | | | | | |
| EGW-32 | 271 | 66 | 2 | 44 | -- | -- | 12.4 | 0.17 | -- | -- |
| EGW-36 | 269 | 64 | 2 | 43 | 13.22 | -0.51 | 12.85 | -.87 | 13.21 | 0.07 |
| EGW-37 | 278 | 37 | 1 | 46 | 8.27 | -.51 | 8.34 | -.44 | 8.85 | -.51 |
| EGW-39 | 275 | 53 | 1 | 45 | -- | -- | 13.85 | -.09 | -- | -- |
| EGW-48 | 277 | 65 | 3 | NI | -- | -- | -- | -- | 10.77 | -.09 |
| EGW-50 | 279 | 71 | 1 | NI | -- | -- | -- | -- | 6.46 | -.04 |
| EGW-51 | 282 | 45 | 3 | NI | -- | -- | -- | -- | 9.27 | -.51 |
| EGW-52 | 259 | 59 | 1 | NI | -- | -- | -- | -- | 17.36 | 1.06 |
| EGW-53 | 275 | 75 | 1 | NI | -- | -- | -- | -- | 5.37 | .84 |
| WNW-17 | 234 | 77 | 3 | 42 | 8.59 | .05 | 8.82 | .28 | 8.78 | .24 |
| Mean absolute error | | | | | | 0.50 | | 0.57 | | 0.72 |
| Median error | | | | | | -0.02 | | 0.17 | | 0.07 |

**Table 1-4.** Measured discharge for selected streams in the modeled area in September 2000 and May 2002, and model-calculated streamflows for simulated current (2002) pumping and recharge conditions, Lower Cape Cod, Massachusetts.

[Locations shown on fig. 1-4. All values are in cubic feet per second. --, not available]

| Stream | September 2000 | May 2002 | Model-calculated |
|---|---|---|---|
| Little Pamet River | 1.5 | 1.2 | 0.61 |
| Herring River at pond | .22 | 1.54 | .11 |
| Herring River at Kings Highway | .22 | 1.5 | .11 |
| Pole Dike Creek | -- | 2.09 | 1.92 |
| Herring River at Bound Brook Road | 4.06 | 5.43 | 6.67 |
| Hatches Creek | .09 | .22 | .52 |
| Fresh Brook | .79 | .7 | .57 |
| Pamet River[1] | -- | -- | 2.98 |

[1]Calculated to be 3.00 cubic feet per second from August 1997 measurements.

**Table 1-5.** Measured altitude of the transition between freshwater and saltwater at selected zone-of-transition wells from LeBlanc and others (1986) and the model-calculated freshwater/saltwater interface for current (2002) pumping and recharge conditions, Lower Cape Cod, Massachusetts.

[Well locations on figure 1-4. Altitude with respect to NGVD 29, in feet. **Model 50 percent:** Model-calculated 50 percent salt concentration. ZOT, Zone of transition; --, not available]

| Wells | ZOT top | ZOT bottom | Model 50 percent |
|---|---|---|---|
| TSW-210–213 | -70 | -- | -95 |
| TSW-234–236 | -110 | -160 | -185 |
| TSW-200, 224, 226 | -170 | -250 | -245 |
| TSW-219–222 | -40 | -60 | -75 |
| WNW-80–83 | -40 | -70 | -62 |
| WNW-50 | -60 | -70 | -80 |
| WNW-117 | -55 | -75 | -115 |

For the purpose of this analysis, the 50-percent salt concentration was assumed to be a reasonable approximation of the freshwater/saltwater interface. A comparison of the model-calculated interface and the observed interface position is presented in table 1-5 for the eight zone-of-transition wells. Profiles of the model-calculated salt concentrations and the electromagnetic-conductivity geophysical logs measured in the eight deep monitoring wells at the three well fields in the Pamet flow lens are shown in figures 17 and 1-7.

Additional data used for model calibration to the position of the freshwater/saltwater interface was from the offshore marine resistivity survey conducted by the USGS Branch of Geophysical Applications and Support in March 2001 (Eric White, U.S. Geological Survey, written commun., 2001). A series of transects was made in Cape Cod Bay offshore of the Nauset flow lens to confirm the model-calculated subsurface seaward discharge of freshwater into Cape Cod Bay.

An example of the comparison between the model-calculated offshore extent of freshwater seaward discharge into Cape Cod Bay with marine resistivity results from that same area is shown in figure 1-8 and shows that the two are in close agreement. The model calculated that freshwater would discharge offshore into Cape Cod Bay because of the effect the thick deposits of low permeability silts and clay underneath the Nauset flow lens has on ground-water flow in the aquifer. The marine resistivity technique proved to be a useful means of determining the offshore seaward extent of freshwater discharge and provided the field data necessary to confirm the model results; these data could not have been readily obtained otherwise. Additional information on the marine resistivity technique for determining the offshore discharge of freshwater can be found in Manheim and others (in press; 2002).

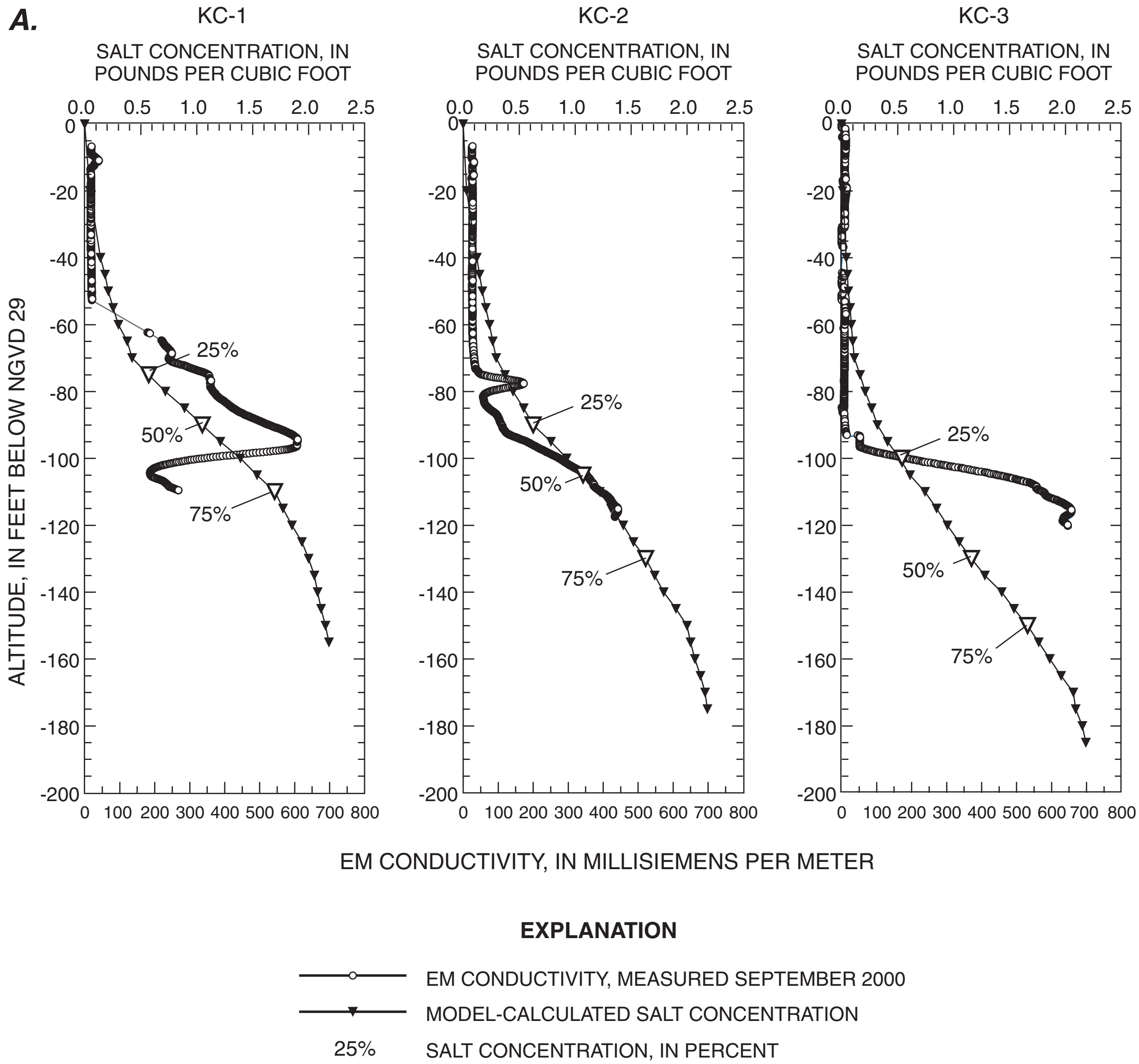



**Figure 1-7.** Profiles of electromagnetic (EM) geophysical logs at *A*, Knowles Crossing; and *B*, North Truro Air Force Base wells 4 and 5 measured in September 2000 and model-calculated changes in salt concentration with depth, Truro, Massachusetts.

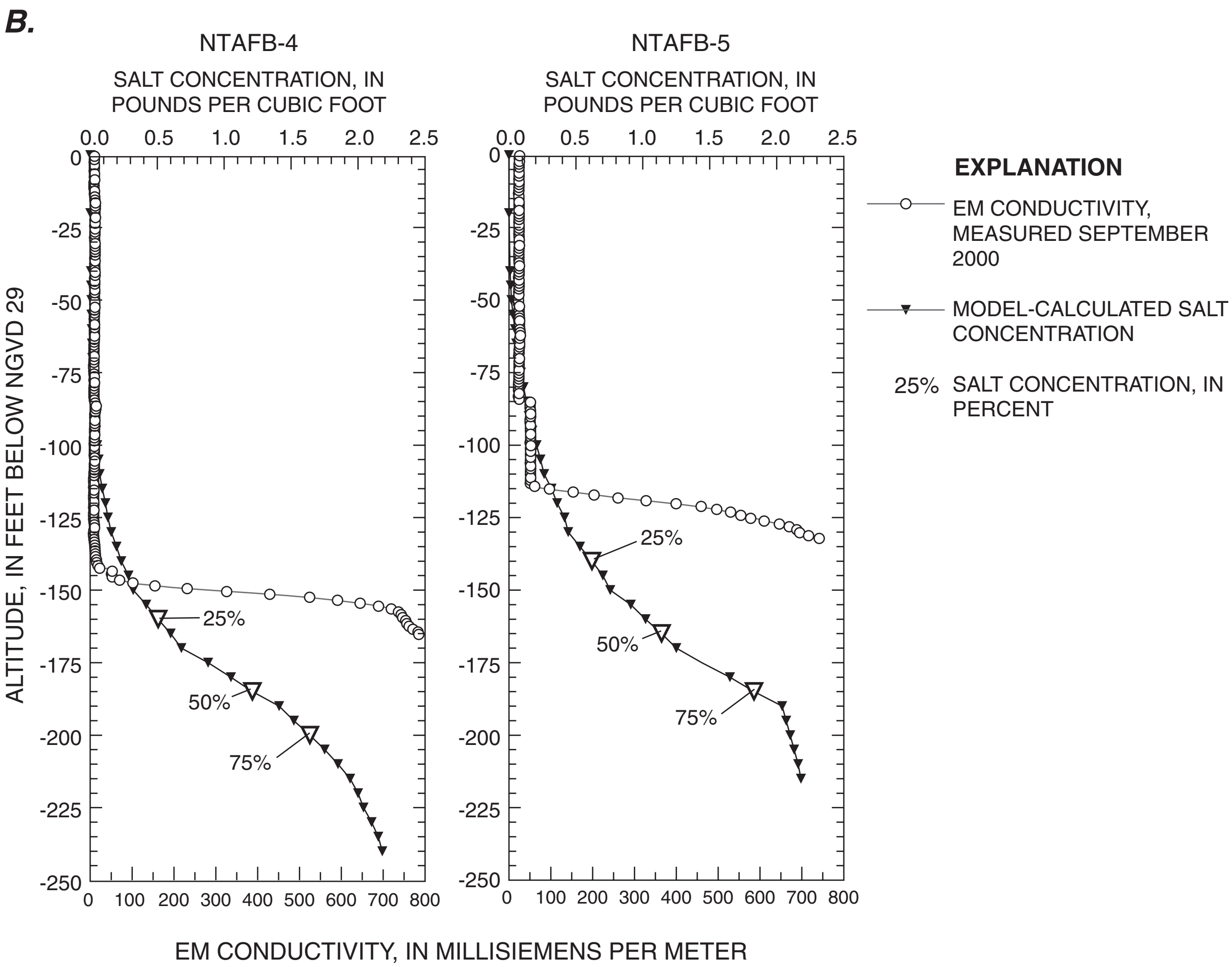



**Figure 1-7—Continued.** Profiles of electromagnetic (EM) geophysical logs at *A*, Knowles Crossing; and *B*, North Truro Air Force Base wells 4 and 5 measured in September 2000 and model-calculated changes in salt concentration with depth, Truro, Massachusetts.

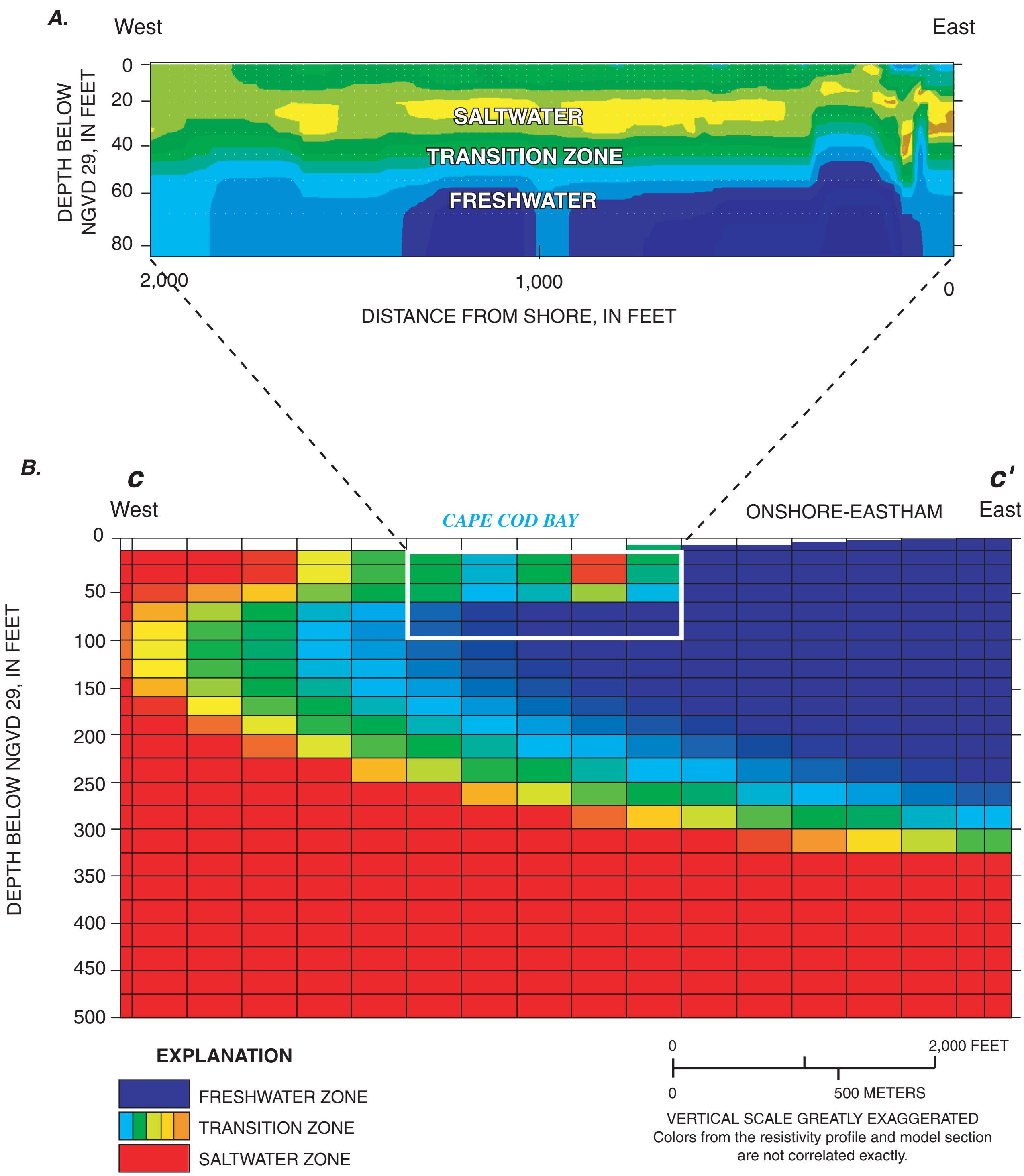



**Figure 1-8.** Cross sections showing *A*, the marine streaming resistivity profile; and *B*, the model-calculated boundary between freshwater and saltwater, Eastham, Massachusetts. Section line *C-C′* on figure 1-4.

## References Cited


Barlow, P.M., and Hess, K.M., 1993, Simulated hydrologic responses of the Quashnet River stream-aquifer system to proposed ground-water withdrawals: U.S. Geological Survey Water-Resources Investigations Report 93-4064, 52 p.

DeSimone, L.A., Howes, B.L., Goehringer, D.G., and Weiskel, P.K., 1998, Wetland plant and algal distribution in a coastal marsh, Orleans, Cape Cod, Massachusetts: U.S. Geological Survey Water-Resources Investigations Report 98-4011, 33 p.

Eichner, Eduard, Cambareri, T.C., Livingston, Kenneth, Sobczak, R.V., and Smith, B.S., 1997, Pamet River investigation groundwater assessment study: Cape Cod Commission, 39 p.

Environmental Partners Group, Inc., 2002, Modeling of saltwater upconing to support water management planning: Hingham, MA, Environmental Partners Group, Inc., June 12, 2002, variously paged.

Farnsworth, R.K., Thompson, E.S., and Peck, E.L., 1982, Evaporation atlas for the contiguous 48 states: National Oceanic and Atmospheric Administration Technical Report NWS 33, 26 p.

Garabedian, S.P., Gelhar, L.W., and Celia, M.A., 1988, Large-scale dispersive transport in aquifers—Field experiments and reactive transport theory: Cambridge, MA, Massachusetts Institute of Technology, Department of Civil Engineering, Ralph M. Parsons Laboratory Report 315, 290 p.

Guo, W., and Langevin, C.D., 2002, User's guide to SEAWAT: A computer program for simulation of three-dimensional variable-density ground-water flow: U.S. Geological Open-File Report 01-434, 77 p.

Guswa, J.H., and LeBlanc, D.R., 1985, Digital models of ground-water flow in the Cape Cod aquifer system, Massachusetts: U. S. Geological Survey Water-Supply Paper 2209, 112 p.

Harbaugh, A.W., and McDonald, M.G., 1996, User's documentation for MODFLOW-96, an update to the U. S. Geological Survey modular three-dimensional finite-difference ground-water-flow model: U.S. Geological Survey Open-File Report 96-485, 56 p.

LeBlanc, D.R., Guswa, J.H., Frimpter, M.H., and Londquist, C.J., 1986, Ground-water resources of Cape Cod, Massachusetts: U.S. Geological Survey Hydrologic Investigations Atlas 692, 4 pls.

Manheim, F.T., Krantz, D.E., and Bratton, J.F., in press, Investigations of submarine ground-water discharge in Delmarva coastal bays by horizontal resistivity surveying and ancillary techniques, *in* Ground Water Discharge to Estuarine and Coastal Ocean Environments: Ground Water.

Manheim, F.T., Krantz, D.E., Snyder, D.S., and Sturgis, B., 2002, Streamer resistivity surveys in Delmarva Coastal Bays: Proceedings Symposium on the Application of Geophysics to Environmental and Engineering Problems (SAGEEP), paper 13GSL5, 17 p.

Masterson, J.P., Walter, D.A., and LeBlanc, D.R., 1998, Delineation of contributing areas to selected public-supply wells, western Cape Cod, Massachusetts: U. S. Geological Survey Water-Resources Investigations Report 98-4237, 45 p.

Masterson, J.P., Stone, B.D., Walter, D.A., and Savoie, Jennifer, 1997, Hydrogeologic framework of western Cape Cod, Massachusetts: U. S. Geological Survey Hydrologic-Investigations Atlas 741, 1 pl.

Masterson, J.P., and Barlow, P.M., 1996, Effects of simulated ground-water pumping and recharge on ground-water flow in Cape Cod, Martha's Vineyard, and Nantucket Island basins, Massachusetts: U. S. Geological Survey Water-Supply Paper 2447, 79 p.

McCobb, T.M., and Weiskel, P.K., 2003, Long-term hydrologic monitoring protocol for coastal ecosystems: U.S. Geological Survey Open-File Report 02-497, 94 p.

Michaud, Scott, and Cambareri, T.C., 2003 Hydrogeologic investigation of the Pilgrim, Pamet, Chequesset, and Nauset Lenses—Water Table Map of the Outer Cape—May 21–23, 2002: Barnstable, MA, Cape Cod Commission, 1 pl.

Moench, A.F., 1994, Specific yield as determined by type-curve analysis of aquifer-test data: Ground Water, v. 32, no. 6, p. 949–957.

National Oceanic and Atmospheric Administration, 2001, accessed August 15, 2001, at http://lwf.ncdc.noaa.gov/oa/pub/data/coop-precip-massachusett.txt

Nicholson, R.S., and Watts, M.K., 1997, Simulation of ground-water flow in the unconfined aquifer system of the Toms River, Metedeconk River, and Kettle Creek Basins, New Jersey: U.S. Geological Survey Water-Resources Investigations Report 97-4066, 100 p.

Oldale, R.N., 1969, Seismic investigations on Cape Cod, Martha's Vineyard, and Nantucket, Massachusetts, and a topographic map of the basement surface from Cape Cod Bay to the islands: U.S. Geological Survey Professional Paper 650-B, p. B122–B127.

Pollock, D. W., 1994, User's guide to MODPATH/MODPATH_PLOT, version 3—A particle tracking post-processing package for MODFLOW, the U. S. Geological Survey modular three-dimensional finite-difference ground-water-flow model: U.S. Geological Survey Open-File Report 94-464, 234 p.