Prepared in cooperation with
Barnstable County and the Cape Cod Commission

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Scientific Investigations Report 2007–5259

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

Cover. View to the east from Salt Pond Outlet over Salt Bay Pond, Eastham, Massachusetts. Photo by Andrew Massey, U.S. Geological Survey

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DIRK KEMPTHORNE, Secretary

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Conversion Factors, Abbreviations, and Datum

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
square mile (mi^2)	2.590	square kilometer (km^2)
	Flow rate	
foot per day (ft/d)	0.3048	meter per day (m/d)
million gallons per day (Mgal/d)	0.04381	cubic meter per second (m^3/s)
inch per year (in/yr)	25.4	millimeter per year (mm/yr)
	Hydraulic conductivity	
foot per day (ft/d)	0.3048	meter per day (m/d)
	Transmissivity*	
foot squared per day (ft^2/d)	0.09290	meter squared per day (m^2/d)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

°F=(1.8×°C)+32

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

°C=(°F-32)/1.8

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 1983 (NAD 83).

Elevation, as used in this report, refers to distance above the vertical datum.

*Transmissivity: The standard unit for transmissivity is cubic foot per day per square foot times foot of aquifer thickness [(ft^3/d)/ft^2]ft. In this report, the mathematically reduced form, foot squared per day (ft^2/d), is used for convenience.

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (μg/L).

Acronyms and Abbreviations

BWDF	Barnstable Wastewater-Disposal Facility
FDM	finite-difference method
MassDEP	Massachusetts Department of Environmental Protection
MEP	Massachusetts Estuaries Project
MMR	Massachusetts Military Reservation
MOC	method of characteristics
SMAST	School of Marine Science and Technology
WDF	wastewater-disposal facility
TMDL	total maximum daily load
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
mg/L	milligrams per liter
kg/L	kilograms per liter
kg/d	kilograms per day

Use of Numerical Models to Simulate Transport of Sewage-Derived Nitrate in a Coastal Aquifer, Central and Western Cape Cod, Massachusetts

By Donald A. Walter

Abstract

The unconsolidated glacial sediments underlying Cape Cod, Massachusetts compose a regional aquifer system that is used both as a source of drinking water and as a disposal site for wastewater; in addition, the discharge of clean ground water from the aquifer system is needed for the maintenance of freshwater and marine ecosystems throughout the region. Because these uses of the aquifer conflict with one another in many areas of the Cape, local and regional planners have begun to develop sustainable wastewater plans that will facilitate the disposal of wastewater while protecting water supplies and improving the health of aquatic ecosystems. To assist local and regional planners in these efforts, the U.S. Geological Survey conducted a 2-year investigation to (1) assist local and regional planners in the evaluation of potential wastewater scenarios, (2) use results and interpretation from these analyses to develop hydrologic concepts transferable throughout the region, and (3) establish and test methods that would be of use in future evaluations.

Wastewater-disposal scenarios need to be evaluated in the context of the regional ground-water-flow system. For a given rate of disposal, wastewater from sites at or near a regional ground-water divide is transported in a wider arc of flow directions, flows deeper in the system, and contaminates a larger part of the aquifer than does wastewater discharged from sites farther from the divide. Also, traveltimes of wastewater from sites near a ground-water divide to receptors are longer (as much as several hundred years) than traveltimes from sites farther from the divide. Thus, wastewater disposal at or near a divide will affect a larger part of the aquifer and likely contribute wastewater to more receptors than wastewater disposal farther from a divide; however, longer traveltimes could allow for more attenuation of wastewater-derived nitrate from those sites.

Ground-water-flow models and particle tracking can be used to identify advective-transport patterns downgradient from wastewater-disposal sites and estimate traveltimes; however, these tools cannot predict the distribution of mass or concentrations of wastewater constituents, such as nitrate, in the aquifer. Flow-based particle-tracking analyses can be used to estimate mass-loading rates and time-varying concentrations at wells and ecological receptors by the accounting of mass-weighted particles discharging into the receptor of interest. This method requires no additional development beyond the flow model; however, post-modeling analyses are required. In addition, the method is based on the assumption that no mass is lost during transport, an assumption that likely is not valid in many systems. Solute-transport models simulate the subsurface transport of nitrate through the aquifer and predict the distribution of the mass of a solute in the aquifer at different transport times. This method does require additional model development beyond the flow model, but can predict time-varying concentrations at receptors. Estimates of mass-loading rates require minimal post-modeling analyses.

Time-varying concentrations and mass-loading rates calculated for wells in eastern Barnstable by the two methods generally were in reasonable agreement. Inherent in the flow-based particle-tracking method is the assumption that mass is conserved along a given flow line and that there is no spreading of mass in the aquifer. Although the solute-transport models also incorporate a system-wide conservation of mass, these models allow for a spreading of mass in the aquifer, and mass is not conserved along a given flow line. As a result, estimates of concentrations and mass loading rates generally were higher in particle-tracking analyses than in solute-transport simulations. Results from the two types of simulations agreed best for wells that receive large amounts of wastewater with short traveltimes (less than 10 years) because insufficient transport time likely had elapsed to make the assumption of conservation of mass along flow lines problematic. The results agreed less closely for wells that receive wastewater with long traveltimes because dispersion may have attenuated simulated concentrations. In these cases, the conservation of mass was a more problematic assumption. Given particle densities that are practicable, particle-tracking simulations may not represent advective transport in divergent flow fields with enough resolution to adequately estimate time-varying concentrations and mass-loading rates in those areas. Also, particle tracking is more likely to overpredict mass-loading rates for wells in areas where dispersion may be important, such as near the edge of a wastewater plume, at large transport distances from a source, or downgradient from a pond.

Introduction

Cape Cod, Massachusetts, (fig. 1) is undergoing rapid urbanization and development as the region transitions from a semirural region strongly dependent on agriculture and tourism to a suburban region with a large year-round population and substantial industry. Communities on Cape Cod are among the fastest growing in Massachusetts. The year-round population of the Cape has more than doubled since 1970, and some parts of the region have had population increases of more than 70 percent since 1990 (Walter and Whealan, 2004). Although population has increased substantially in most areas, large additional increases are still possible because many towns are well below projected buildout development.

The unconsolidated glacial sediments underlying Cape Cod constitute an aquifer system that is the sole source of potable water to these communities. Most of the population of Cape Cod resides over the Sagamore and Monomoy flow lenses, the largest of the six lenses that compose the regional aquifer system (fig. 1) (LeBlanc and others, 1986). Pumping from these lenses totaled about 25 Mgal/d in 2004 and is projected to increase to more than 39 Mgal/d by the year 2020. On average, about 7 percent of total recharge to the aquifer is withdrawn for water supply (Walter and Whealan, 2004); most of this pumped water (about 85 percent) is returned to the aquifer as wastewater, resulting in a total consumptive loss of about 1 percent. Most naturally recharged ground water (about 93 percent) discharges into ecological receptors, including ponds, streams, and coastal water bodies; ground-water discharge from the aquifer system supports important freshwater and marine ecosystems.

Most wastewater enters the aquifer as untreated septic-system effluent from residential areas or as treated wastewater from five wastewater-disposal facilities (WDFs) (fig. 1). The discharge of wastewater into the aquifer can adversely affect water supplies. Nitrogen, in the form of nitrate, is a common constituent in wastewater and can adversely affect human health. The discharge of sewage-derived nitrate into production wells is of concern on Cape Cod, and water suppliers monitor nitrate concentrations in production wells. The U.S. Environmental Protection Agency (USEPA) has established a drinking-water standard of 14 mg/L.

In addition to adverse effects on water supplies, wastewater can adversely affect critical habitats and ecosystems in both freshwater and saltwater. Wastewater contains nutrients such as phosphorus and nitrogen that can cause eutrophication in some waters. Several coastal water bodies on Cape Cod are considered eutrophic or mesotrophic with the primary cause attributed to the subsurface discharge of sewage-derived nitrogen into the coastal waters. Eutrophication in most coastal waters is nitrogen-limited, and the discharge of excess nitrogen into these waters can lead to the growth of harmful algae, decreases in water clarity, depletion of oxygen, and the loss of indigenous flora and fauna (School of Marine Science and Technology, 2005).

The Massachusetts Department of Environmental Protection (MassDEP) has initiated the Massachusetts Estuaries Project (MEP) to estimate the total maximum daily load (TMDL) of nitrogen for estuaries on Cape Cod. (A TMDL represents the maximum load that an individual estuary can receive without adverse effects on water quality or estuarine ecosystems.) At present (2005), several communities on Cape Cod are developing wastewater-management plans to (1) limit the discharge of nitrogen into the watersheds of sensitive coastal waters in accordance with the TMDLs established by the MEP and (2) protect current and potential water supplies from the adverse effects of wastewater disposal. Wastewater-management plans include the sewering of residential areas and the disposal of treated sewage effluent at centralized disposal locations. The redistribution of wastewater disposal can change the amount and distribution of nitrogen discharge into wells and ecological receptors and affect the hydrologic system by changing water levels, hydraulic gradients, and streamflows. Owing to the fact that communities share a regional aquifer system, the hydrologic effects of wastewater-management scenarios on individual communities are better understood in the context of the regional ground-water flow system. A regional analysis, as opposed to a town-by-town analysis, allows hydrologic analyses of water supply and wastewater-management strategies for an individual town to be done in a context that takes into account similar actions by neighboring towns. The approach also allows a sustainable balance between competing uses of the aquifer—water supply and wastewater disposal—to be achieved on a regional basis.

In 2003, the U.S. Geological Survey (USGS), in cooperation with Barnstable County and the Cape Cod Commission, began an investigation into the potential hydraulic effects of wastewater disposal on the regional aquifer system (water levels, hydraulic gradients, and streamflows) as well as on the wells and ecological receptors within the Sagamore and Monomoy flow lenses. Specifically, the objectives of the investigation were to use numerical models to (1) evaluate regional environmental effects of potential wastewater-disposal scenarios, (2) assess advective transport of sewage-derived contaminants in different hydrologic settings within the regional flow system, and (3) evaluate the use of solute-transport modeling to simulate the transport of nitrate, the wastewater constituent of most concern, in the Cape Cod aquifer system.

Purpose and Scope

This report documents hydrologic analyses of wastewater management on central and western Cape Cod. Examples are presented that illustrate the use of ground-water-flow models in evaluating the effects of wastewater disposal on the hydrologic system and the advective transport of sewage-derived contaminants from disposal areas to wells and ecological receptors. Case studies that compare and contrast advective transport of sewage-derived contaminants in different hydrologic settings are presented. The report also documents

Base from U.S. Geological Survey digital data,
Universal Transverse Mercator projection,
Zone 19, 1:24,000, 1991

Figure 1. Location of the Sagamore and Monomoy flow lenses and wastewater-disposal facilities (WDF), Cape Cod, Massachusetts.

the development of a subregional-flow and solute-transport model of an area of central Cape Cod and presents examples of the use of this tool in simulating the conservative transport of nitrate to wells and ecological receptors. Finally, the report also compares and contrasts the use of different modeling tools (flow-based particle tracking and solute-transport models) in simulating the conservative transport of sewage-derived nitrate in the Cape Cod aquifer system.

Methods of Analysis

Numerical ground-water-flow models were used to simulate the hydrologic systems of the Sagamore and Monomoy flow lenses on central and western Cape Cod. These models were developed in 2004 by the USGS, in cooperation with MassDEP. The models use the USGS software program MODFLOW-2000 (Harbaugh and others, 2000) to simulate three-dimensional ground-water flow in the region. Detailed documentation of these models, including model design, hydrologic boundaries, hydrologic stresses, aquifer characteristics, and model calibration is presented in Walter and Whealan (2004). The advective transport of contaminants in the aquifer was simulated by the USGS particle-tracking software program MODPATH (Pollock, 1994). Graphic display of simulation results was done by using a version of the GIS software application MODTOOLS (Orzol, 1997) modified to work with MODFLOW-2000. A subregional model was developed to simulate the transport of nitrate in the aquifer by use of the USGS software program MODFLOW-96 (Harbaugh, 1996) and the linked solute-transport code MT3D (Zheng, 1990). Lateral boundaries for the subregional model were derived from the regional model of the Sagamore flow lens (Walter and Whealan, 2004).

Hydrogeology

The glacial sediments beneath Cape Cod generally are sandy, highly permeable, and can exceed 500 ft in thickness in the central and western parts of the Cape. The region receives substantial rainfall, and the unconsolidated sediments are an important source of potable water for local communities. The geologic history and hydrology of Cape Cod has been documented in numerous publications, including LeBlanc and others (1986), Masterson and others (1997a), Oldale (1992), and Uchupi and others (1996).

Geologic Setting

The unconsolidated sediments underlying Cape Cod consist of gravel, sand, silt, and clay that was deposited during the Pleistocene epoch between 15,000 and 16,000 years ago (Oldale and Barlow, 1986). Sediments were deposited at or near the edges of a retreating continental ice sheet through direct deposition from the ice and proglacial deposition from meltwater. The surficial geology of western and central Cape Cod is characterized by broad, gently sloping glacial-outwash plains and hummocky terrain associated with glacial moraines (fig. 2A).

Three general types of glacial deposits occur on Cape Cod—moraine, ice-contact, and outwash. Moraines were deposited in low-energy environments at or near the edge of the ice sheets and are either ablation moraines that were deposited in place by melting ice, such as the Buzzards Bay moraine, or tectonic moraines, such as the Sandwich moraine, that consist of reworked outwash sediments pushed into place by local readvances of the ice sheets (Uchupi and others 1996; Oldale, 1992; Oldale and O'Hara, 1984). Kames are ice-contact deposits that were deposited in high-energy meltwater environments within holes in the ice. Ice-contact deposits also include sediments that were deposited by meltwater in high-energy fluvial environments near the ice margin. Outwash sediments, which compose most of the unconsolidated sediments underlying Cape Cod, were deposited by meltwater streams in depositional environments associated with proglacial lake deltas; these depositional environments are analogous to those found in present-day fluvial deltas (Oldale, 1992; Uchupi and others, 1996). Deltaic outwash sediments can be divided into three general facies: topset, foreset, and bottomset deposits (fig. 2B) (Masterson and others, 1997a). These sediments generally become finer with depth; in general, coarse-grained fluvial sand and gravel (topset sediments) overlie fine to medium sand deposited in nearshore lacustrine environments (foreset beds), which in turn are underlain by silty sand and clay deposited in offshore lacustrine environments (bottomset beds). The glacial sediments are underlain by basal till in most places (fig. 2B); basal till consists of fine-grained material that was produced by mechanical erosion of bedrock by movement of the overlying ice sheet. The unconsolidated glacial sediments are underlain by relatively impermeable crystalline bedrock.

The elevation of the bedrock surface on Cape Cod ranges from about 50 ft below NGVD 29 near the Cape Cod Canal to more than 900 ft below NGVD 29 beneath the outer part of Cape Cod. The thickness of the glacial deposits on central and western Cape Cod ranges from 70 ft near the Cape Cod Canal to more than 500 ft along Nantucket Sound (fig. 2B).

The lithology of the glacial sediments, which include moraines, kames and other ice-contact deposits, and stratified outwash, differs according to the environment in which the sediments were deposited. Deltaic outwash sediments generally are characterized by two grain-size trends: (1) a fining-down trend in which grain sizes decrease, and silt and clay content increases, with depth; and (2) a trend in which sediments become finer-grained, and coarse-grained sand and gravel deposits become thinner, with increasing distance from sediment sources (fig. 2B). Sediment sources generally were close to the present-day shore of Cape Cod Bay near the apex of the Sandwich and Buzzards Bay moraines and near the Dennis and Nauset Heights ice-contact deposits (fig. 2A),

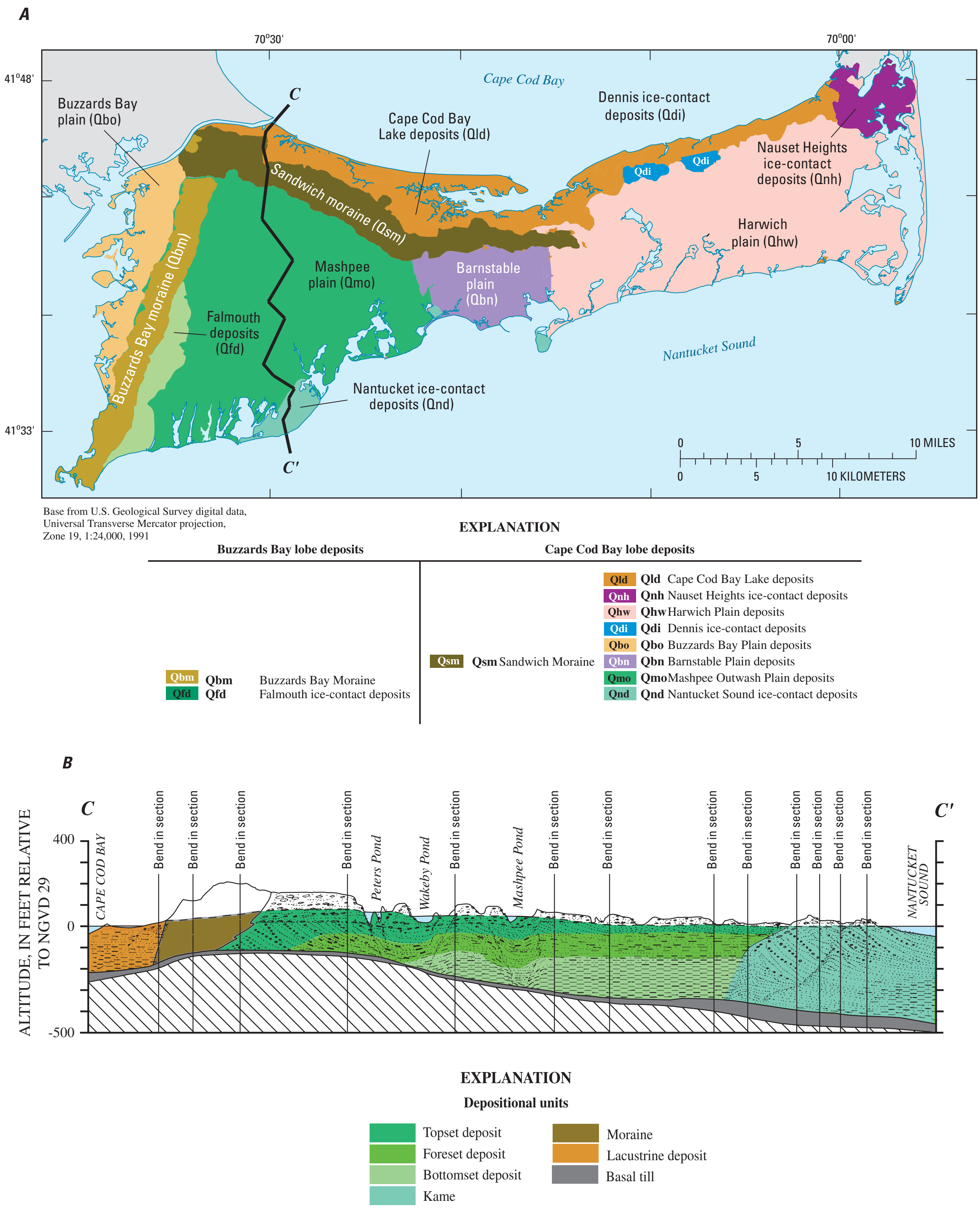


Figure 2. (*A*) The surficial geology of western and central Cape Cod, and (*B*) geologic section *C-C′* through western Cape Cod, Massachusetts.

and sediments generally become finer to the south. Moraine sediments, which consist of gravel, sand, silt, and clay, are generally poorly sorted with a highly diverse lithology. Ice-contact deposits and kame deposits are well-sorted coarse-grained sediments, and basal till is fine-grained, generally homogenous, and consists primarily of clay.

Grain-size and degree of sorting determine the water-transmitting properties of aquifer sediments. The trends in hydraulic conductivity of outwash sediments are parallel to the trends in grain size; the hydraulic conductivity of sediments generally decreases with increasing depth and with increasing distance from sediment sources, or generally southward (Masterson and others, 1997a). Previous investigations have identified general relations between sediment grain size and hydraulic conductivity, as determined from aquifer tests (Masterson and others, 1997a; Masterson and Barlow, 1994). Medium to coarse sand and gravel deposits have hydraulic conductivities that typically range from 200 to 350 ft/d. Fine to medium sands have hydraulic conductivities typically ranging from 70 to 200 ft/d. The hydraulic conductivities of very fine sand and silt typically range from 30 to 70 ft/d, and silt and clay deposits have hydraulic conductivities of between 10 and 30 ft/d. Ice-contact and kame deposits consist generally of medium to coarse sand and gravel and have hydraulic conductivities similar to those of coarse-grained outwash deposits. The lithology of moraine deposits ranges from gravel and sand to silt and clay, and moraines generally have lower average hydraulic conductivities than outwash deposits. Most areas, including moraines, have trends of decreasing hydraulic conductivity with increasing depth.

Hydrologic Setting

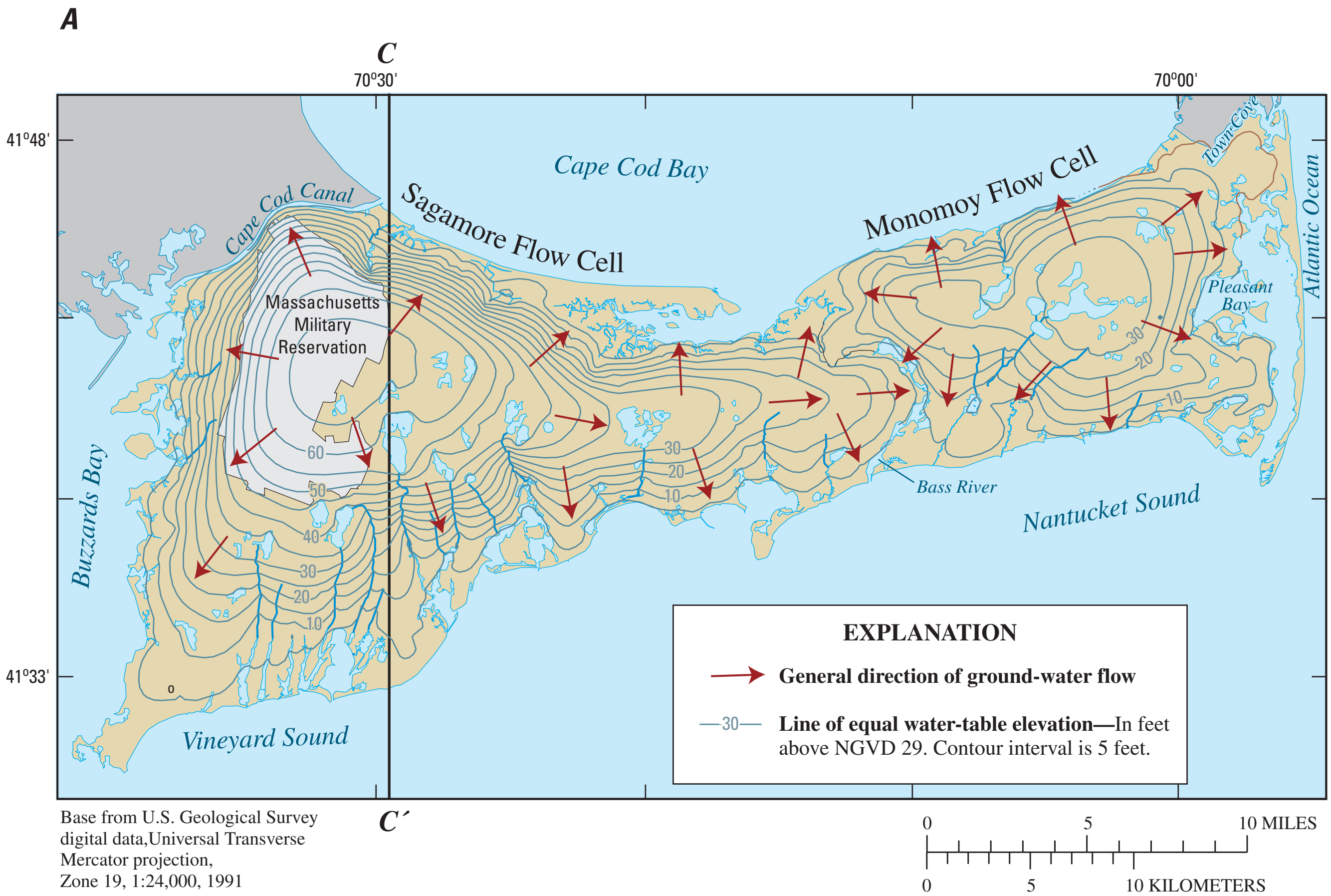
The unconsolidated glacial sediments underlying Cape Cod compose an unconfined aquifer system that is surrounded by saltwater: Cape Cod Bay to the northeast, Cape Cod Canal to the northwest, Buzzards Bay to the west, and Vineyard and Nantucket Sounds to the south. The Sagamore and Monomoy flow lenses on central and western Cape Cod are the largest and southernmost of six separate ground-water-flow lenses that underlie Cape Cod (LeBlanc and others, 1986); the two flow lenses are hydraulically separated by the Bass River (fig. 3A). The Sagamore flow lens is hydraulically separated at its northwestern extent from mainland Massachusetts by the Cape Cod Canal. The Monomoy flow lens is hydraulically separated from an adjacent flow lens by Town Cove at its northeastern extent. Each flow lens represents a distinct aquifer system that is hydraulically separate from adjacent flow lenses. The aquifer systems are bounded below by impermeable bedrock and at the top by the water table across which recharge enters (fig. 3B). Recharge from precipitation is the sole source of water to the aquifer system. About 45 in. of precipitation falls annually on Cape Cod; slightly more than half of the precipitation recharges the aquifer (LeBlanc and others, 1986). The remainder is lost to evapotranspiration; surface runoff generally is negligible owing to the sandy soils and low topographic relief of the area.

Ground water flows outward from regional ground-water divides towards natural discharge locations at streams, estuaries, and the ocean (fig. 3A). Most recharge flows through shallow sediments and discharges to streams and estuaries; ground-water recharging the aquifer near the ground-water divides flows deeper in the aquifer and discharges to the ocean (fig. 3B). Most ground water (about two-thirds) discharges into saltwater bodies. About 25 percent of ground water is discharged into freshwater streams and wetlands, and a small amount (less than 10 percent) is removed from the system for water supply (Walter and Whealan, 2004). Maximum water-table elevations of the Sagamore and Monomoy flow lenses are more than 65 and 30 ft above NGVD 29, respectively (fig. 3A). Water-table contours and ground-water-flow patterns are strongly affected locally by kettle-hole ponds because the ponds are areas in the aquifer with no effective resistance to flow. Kettle-hole ponds are flow-through ponds characterized by ground-water-flow paths converging in areas upgradient from the ponds where ground water discharges into the ponds, and diverging in downgradient areas where pond water recharges the aquifer. Some ponds have surface-water outlets where ponds drain into freshwater streams. Streams generally are areas of ground-water discharge (gaining streams) and receive water from the aquifer over most of their length. Some stream reaches may lose water to the aquifer (losing streams), particularly in areas downgradient from pond outflows.

Owing to high recharge rates (over 25 in/yr) and the generally high permeability of the aquifer sediments, advective transport likely is the dominant component of contaminant transport in the aquifer (LeBlanc, 1984). Some contaminant plumes around the Massachusetts Military Reservation (MMR) on western Cape Cod (fig. 1) have migrated more than 4 mi downgradient since the mid-1950s. Ground-water velocities of more than 1.5 ft/d have been observed at the USGS Toxics Substances Research Site at the former sewage-treatment facility at the MMR (fig. 1) (LeBlanc and others, 1991); ground-water velocity is a function of aquifer porosity, which is about 0.35 near the MMR (Garabedian and others, 1985).

The rate of advective transport in the aquifer is a function of the locations of source areas relative to regional ground-water divides; water recharged near divides, where ground-water flow is more vertical, moves more slowly than water recharged farther from divides, where horizontal flow dominates (fig. 3B) (Walter and Masterson, 2003; Walter and others, 2004). Traveltime, defined as the total time it takes water to move from recharge locations at the water table to natural discharge locations, is longest near regional ground-water divides and ranges from essentially zero adjacent to discharge boundaries to hundreds of years near ground-water divides (Walter and others, 2004).

About 7 percent of the water recharging the Sagamore and Monomoy aquifer system is removed for water supply (Walter and Whealan, 2004). Most of this water is returned to



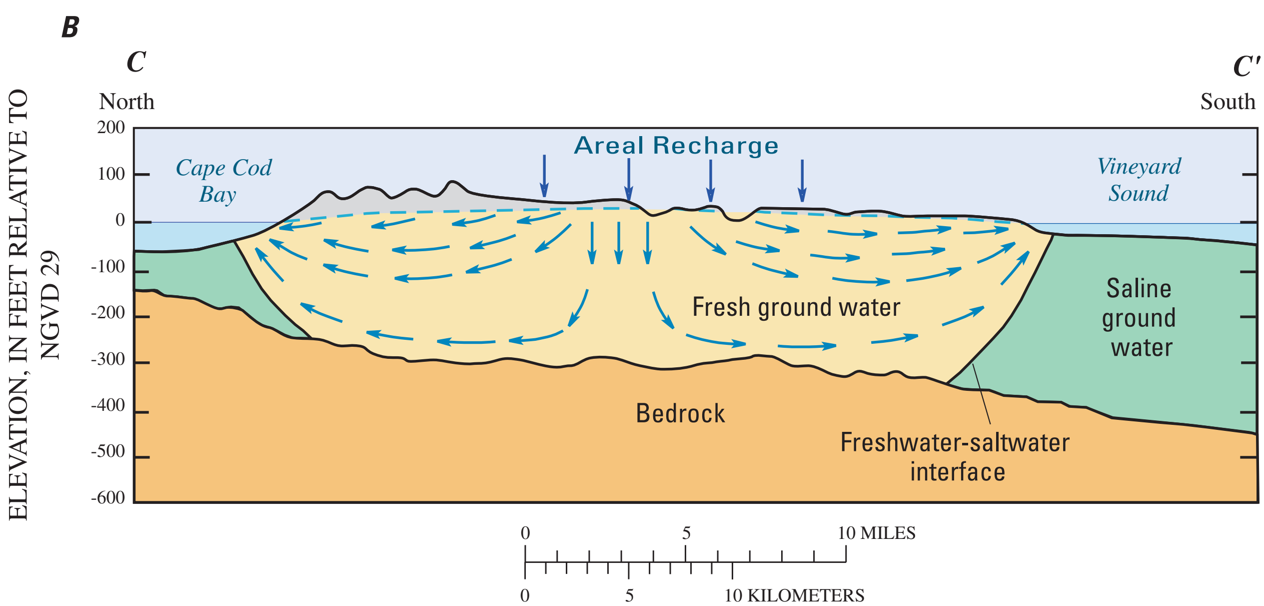


Figure 3. (*A*) The regional water table and general flow paths for central and western Cape Cod and (*B*) generalized vertical ground-water flow, western Cape Cod, Massachusetts.

the system as disposed waste-water, either as dispersed septic-system discharge or as point discharges to the aquifer at sewage-treatment plants. Although most of the pumped water is returned to the aquifer—a process that creates a regional mass balance—the water is usually disposed of at some distance from where it was withdrawn, particularly in areas served by public water supply. Large-capacity production wells decrease ground-water levels and can affect natural resources by drying vernal pools, drawing down ponds, and decreasing streamflows by changing hydraulic gradients and either intercepting water that would have discharged to a surface-water body or by inducing infiltration from the stream (Masterson and others, 2002). In the vicinity of large wastewater-disposal facilities, ground water can form a mound, and wastewater disposal can increase streamflows.

Nitrogen Transport in Ground Water

Nitrogen occurs in natural waters primarily in the form of nitrate (NO_3^-), ammonia (NH_4^+), dissolved nitrogen gas ($N_2(g)$), or organic nitrogen; the predominant species of inorganic nitrogen in ground water is determined by the pH and redox conditions of the water (Hem, 1981). Ammonia is a reduced form of nitrogen that strongly sorbs to aquifer sediments, whereas nitrate, an oxidized form of nitrogen, is an anion that does not sorb to aquifer sediments and is transported conservatively in ground water (Hem, 1981). LeBlanc (1984) reported that nitrate was the predominant form of nitrogen in treated sewage effluent at the MMR wastewater-treatment facility; Weiskel and Howes (1992) found that most nitrogen emanating from septic systems in a sandy coastal aquifer in mainland southeastern Massachusetts also was in the form of nitrate.

Nitrate can be lost or attenuated in an aquifer through the process of denitrification in which nitrate is reduced to nitrogen gas (Hem, 1981). In most aquifers the denitrification process is coupled with the oxidation of organic carbon present in aquifer sediments. The sandy aquifer underlying Cape Cod generally is oxic and poor in organic carbon (LeBlanc, 1984), and therefore, nitrate attenuation through denitrification may be limited by available carbon. Weiskel and Howes (1992) found that a total of about 26 ±10 percent of nitrogen (primarily as nitrate) in septic-system effluent was lost during transport over small transport distances (less than about 600 m). The mechanism for the loss, however, is unknown; much of the nitrate may have been lost within the septic systems or in the underlying unsaturated zone and not in the aquifer. Most data suggest that the oxic, carbon-poor conditions in the Cape Cod aquifer generally would be favorable for the persistence of nitrate in the subsurface.

The one-dimensional subsurface transport of a solute can be described as (modified from Fetter, 1988):

$$\frac{\partial C}{\partial t} = \underbrace{D_L\left[\frac{\partial^2 C}{\partial x^2}\right]}_{\text{(Dispersion)}} + \underbrace{D_d\left[\frac{\partial^2 C}{\partial x^2}\right]}_{\text{(Diffusion)}} - \underbrace{v_x\left[\frac{\partial C}{\partial x}\right]}_{\text{(Advection)}} - \underbrace{\left[\frac{B_d}{n}\right]\left[\frac{\partial C_s}{\partial t}\right]}_{\text{(Sorption)}} + \underbrace{\left[\frac{\partial C}{\partial t}\right]_{rxn}}_{\text{(Reactions)}} \qquad (1)$$

where

C	=	concentration of dissolved solute,
t	=	time,
D_L	=	longitudinal dispersivity,
D_d	=	diffusivity,
v_x	=	linear ground-water velocity,
B_d	=	bulk density of aquifer sediments,
n	=	porosity,
C_s	=	concentration of sorbed solute, and
rxn	=	subscript indicating chemical or biological reactions.

Advection refers to transport of a solute with average ground-water flow; dispersion, a process that occurs in all natural aquifers, refers to a spreading of solute mass with respect to the path of average ground-water flow owing to aquifer heterogeneity and preferential flow. Ground-water-flow models are mathematical representations of the aquifer that account for advective transport only. In highly permeable, well sorted aquifers, such as that which underlies Cape Cod, advection generally is the dominant transport mechanism, and ground-water models can be used to simulate this part of the transport process. Solute-transport models, such as those applied in this investigation, supply the additional terms for sorption, diffusion, and simple one-component reactions like decay. In the case of nitrate, which is conservative and nonreactive, the last two terms of the equation (sorption and reactions) likely are negligible, and diffusion generally is negligible in advection-dominated systems. The result is a transport equation in which the change in concentration with time is a function of advection and dispersion. It should be noted that the assumption of conservative transport not only is generally supported by field data, but also results in worst-case estimates of mass-loading rates; therefore, this assumption is favorable in formulating strategies for adequate resource protection.

Numerical Models

The set of modeling tools used in the analyses documented in this report include recently (2004) developed regional models of the Sagamore and Monomoy flow lenses as well as a subregional model of parts of Barnstable and Yarmouth on central Cape Cod (fig. 4A). The numerical models are discretized mathematical representations of the aquifer system that can simulate hydrologic conditions (heads and flows) in the aquifer and the movement of water through the aquifer.

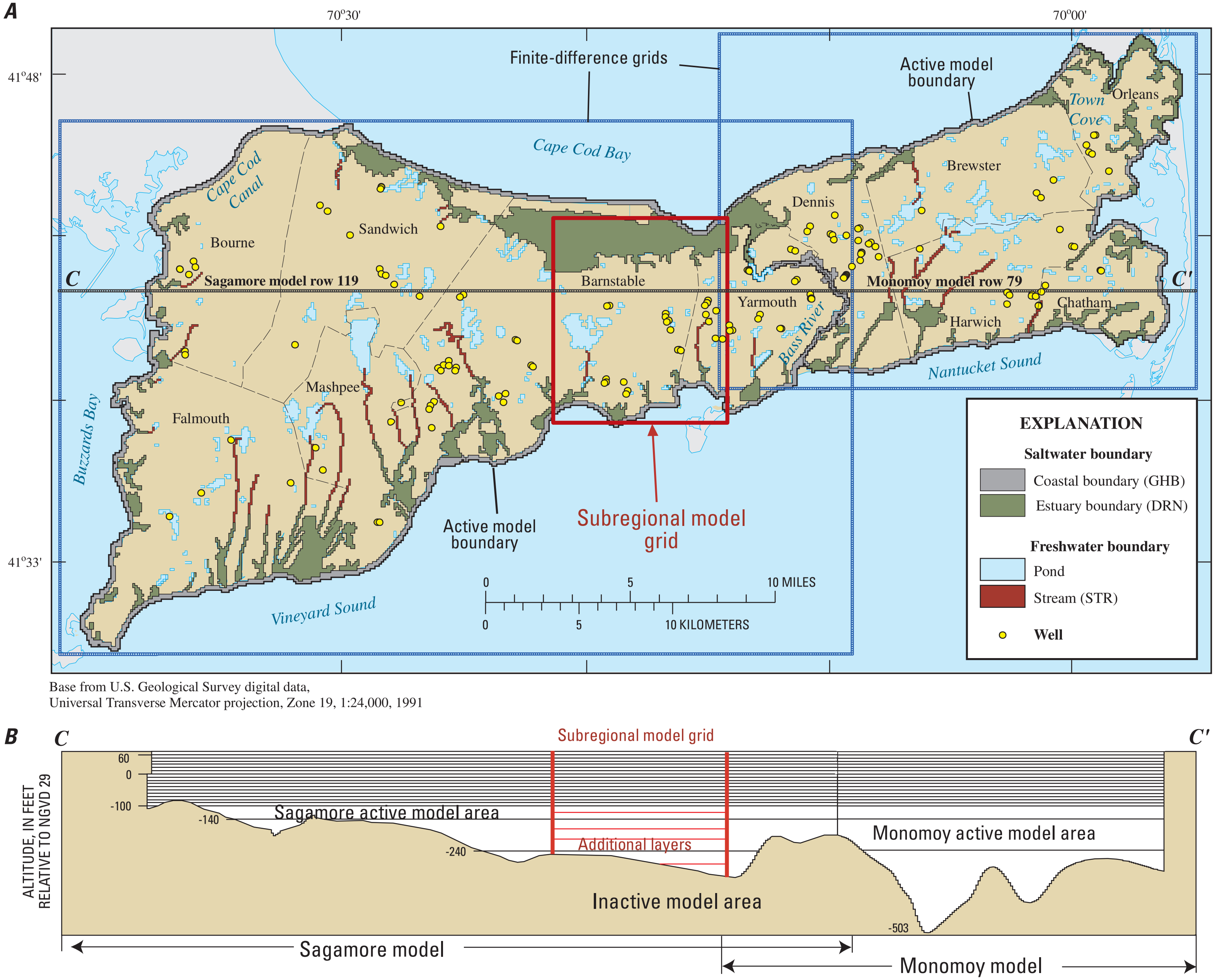


Figure 4. (*A*) Finite-difference grids and simulated hydrologic boundaries, and (*B*) vertical discretization for regional models of Sagamore and Monomoy Flow lenses, Cape Cod, Massachusetts.

Regional Models

The regional models of the Sagamore and Monomoy lenses are documented in detail in Walter and Whealan (2004), Appendix 1. These models were previously used to delineate recharge areas to wells and ecological receptors on Cape Cod (Walter and others, 2004) as well as to evaluate the effects of time-varying pumping and recharge on the regions' hydrologic system.

Regional Model Design

The models extend from the Cape Cod Canal in Bourne to Town Cove in Orleans; the boundary between the active parts of the two model domains is the Bass River in Yarmouth (fig. 4A). The finite-difference ground-water modeling program MODFLOW-2000 (Harbaugh and others, 2000; McDonald and Harbaugh, 1988) was used to simulate the ground-water systems of the two flow lenses. The Sagamore model has 246 rows and 365 columns with a total active modeled area of 246 mi^2, and the Monomoy flow model has 164 rows and 220 columns with a total active modeled area of 106 mi^2. Both models have a uniform horizontal discretization of 400 by 400 ft; the grids are coincident where the models overlap (fig. 4A). Both models have 20 layers with thicknesses ranging from a maximum of 10 ft in the top 17 model layers to more than 200 ft in the deepest layer (fig. 4B).

Estuaries, open coastal waters, and streams are represented as head-dependent flux boundaries in the models (fig. 4A). Open coastal waters, saltwater estuaries, and some freshwater wetlands were simulated by using the General Head Boundary (GHB) and Drain (DRN) packages in MODFLOW-2000 (Harbaugh and others, 2000; McDonald and Harbaugh, 1988). Freshwater streams and some freshwater wetlands were simulated by using the Streamflow Routing Package (STR) (Prudic, 1989) which allows for an accounting of streamflow and pumping-induced streamflow depletion. Freshwater ponds, which were represented as regions of high hydraulic conductivity (100,000 ft/d), were active parts of the aquifer, and, therefore, pond levels fluctuate in response to changing hydraulic stresses.

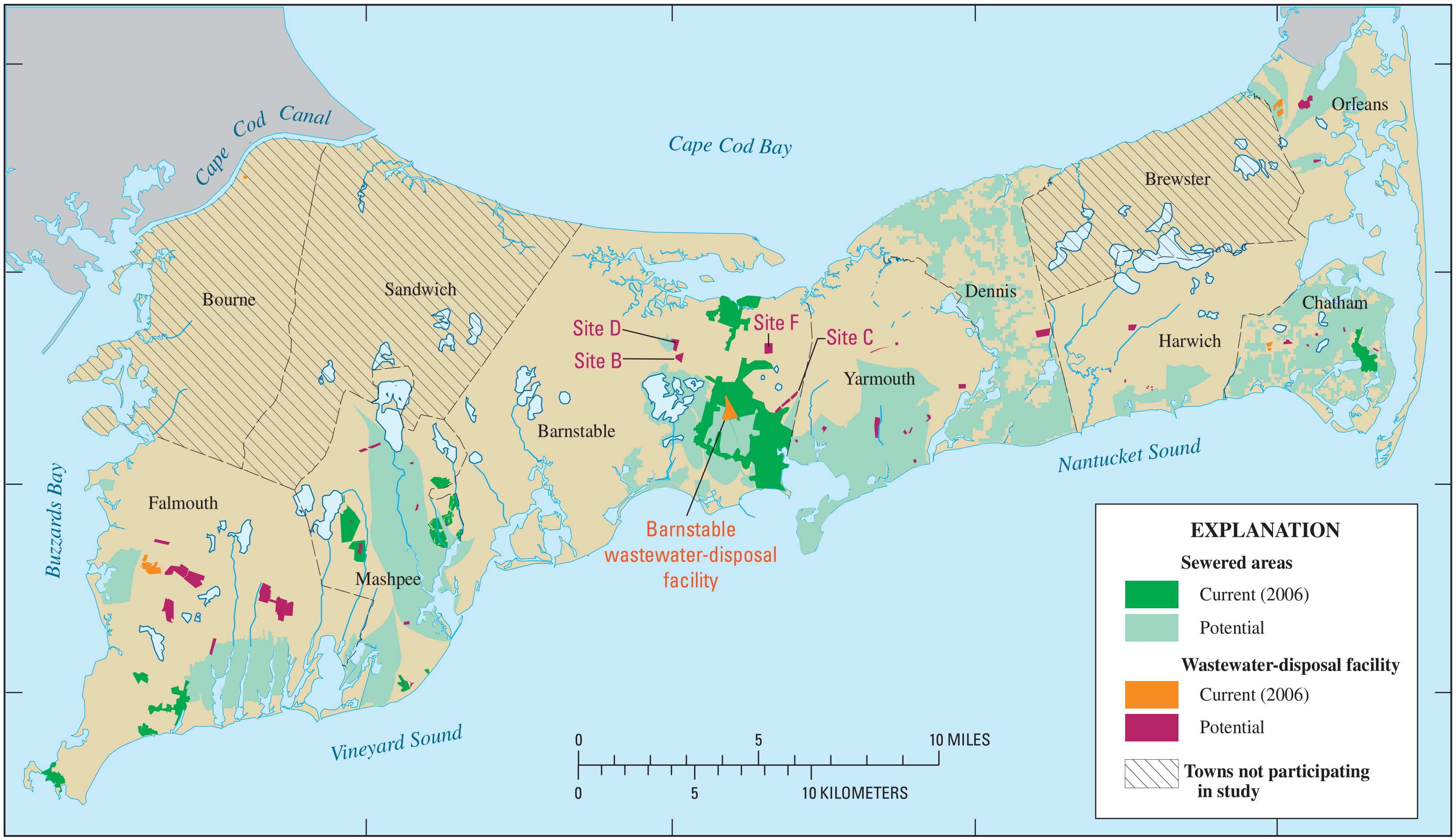
The only input of water into the model is recharge from precipitation. A natural recharge rate to aquifer sediments of 27.1 in/yr was specified; this value was obtained from long-term records from the precipitation gage at Hatchville, Mass. (fig. 1), was adjusted during model calibration, and is consistent with recharge values used in previous investigations on Cape Cod and southeastern New England (Barlow and Dickerman, 2000; Desimone and others, 2001; Masterson and others, 1997b; Walter and Masterson, 2003). Natural recharge to surface-water bodies was adjusted according to the receptor: 16 in/yr to ponds representing net recharge after pan evaporation (Farnsworth and others, 1982) and no recharge to wetlands, which likely are areas of net ground-water discharge. Recharge was further adjusted to account for septic-system return flow in residential areas. Because parcel-scale water-use data were not available, the volume of generated wastewater was determined from the total volume of ground water withdrawn by each town; a consumptive loss of 15 percent was assumed, and the remaining volume was evenly distributed across model cells representing nonsewered residential areas. Recharge also was adjusted in areas of current wastewater-disposal facilities (WDFs) in Barnstable, Chatham, Falmouth, Orleans, and at the MMR (fig. 1). In 2004, about 156 production wells operated on central and western Cape Cod (Walter and Whealan, 2004). Withdrawal of water at production wells was simulated using the Well (WEL) Package (McDonald and Harbaugh, 1988). The regional models of the Sagamore and Monomoy flow lenses simulate current ground-water withdrawals of about 17.3 Mgal/d and 7.6 Mgal/d, respectively. The models also simulate future (2020) ground-water withdrawals of about 27.4 and 11.6 Mgal/d, respectively, for the two flow lenses.

Aquifer properties were estimated from lithologic logs and from previously developed depositional models of central and western Cape Cod (Masterson and others, 1997b; Byron Stone, U.S. Geological Survey, written communication, 2002). Hydraulic conductivities varied spatially with depth and ranged from 350 ft/d for coarse sand and gravel to 10 ft/d for silt and clay (Walter and Whealan, 2004). The steady-state regional models were calibrated on the basis of measured long-term water levels and streamflows as well as delineated contaminant plumes as indicators of advective flow paths. Initial estimates of recharge and intrinsic aquifer properties were adjusted during the calibration process to achieve an acceptable match between observed and simulated hydrologic conditions at the calibration points (Walter and Whealan, 2004). Steady-state models were used in the analysis of advective transport. Although recharge and pumping vary through time and versions of the regional models that incorporate these time-varying stresses have been developed, steady-state models are adequate for the simulation of advective flow because the time scale of the variability of these stresses is small compared to the time scale of transport in the aquifer (Walter and Masterson, 2003).

Modifications to the Regional Models

The hydrologic boundaries, intrinsic aquifer properties, and natural recharge rates described in Walter and Whealan (2004) also were used in this investigation. Modifications were made to human-influenced recharge stresses simulated in the regional models. Changes in pumping rates were minor and were based on additional information made available from local water suppliers during the investigation.

The amount and spatial distribution of recharge representing septic-system return flow also was changed. The additional recharge representing septic-system return flow was removed from some areas to simulate the diversion of water from these areas to areas of centralized sewage disposal (fig. 5). Recharge was enhanced in areas of the model grid that represent possible locations of centralized wastewater disposal, simulat-



Base from U.S. Geological Survey digital data,
Universal Transverse Mercator projection,
Zone 19, 1:24,000, 1991

Figure 5. Current and potential sewered areas and areas of centralized wastewater disposal, central and western Cape Cod, Massachusetts.

ing the disposal of wastewater onto sand-infiltration beds at these locations (fig. 5). Each modeled scenario represented an adjustment of model recharge according to this method. Owing to the fact that septic-system return flow was estimated from general residential land use as opposed to actual parcel-scale water use, the volume of water made available through the conversion of on-site septic disposal to sewering often did not equal the volume of water specified for disposal at centralized locations. In these cases, return flow in nonsewered areas was adjusted to maintain a mass balance of water (adjusted for consumptive loss) within each town. In all cases, the discrepancy was less than 1 percent of total recharge in the model, and this simplification did not affect simulated advective flow paths in the aquifer.

Subregional Model

A subregional model was developed as part of this investigation to simulate ground-water flow and transport in the eastern part of Barnstable (fig. 4A). This area was chosen because the area is the most densely populated part of Cape Cod, and wastewater disposal is an issue of great concern in Barnstable. The town is in the process of developing a wastewater-management plan; at the time of this investigation (2006), the community had already identified four potential wastewater-disposal sites as wells as additional areas to be sewered.

Model Design

The model grid, which includes the eastern part of Barnstable and the western part of Yarmouth, encompasses an area of about 43.2 mi^2 on central Cape Cod (fig. 4A). The model has 376 rows, 320 columns, and 26 layers; horizontal discretization is a uniform 100 ft on a side. Layer thicknesses from the water table to 100 ft below NGVD 29 do not exceed 10 ft; this area of the model is of most interest because wells and hydrologic boundaries as well as most ground-water flow are within the upper part of the aquifer. Six layers were added to the subregional model below the elevation of 100 ft below NGVD 29: layer 18 in the regional model (100 to 140 below NGVD 29) was equally subdivided into two layers, layer 19 (140 to 240 ft below NGVD 29) was subdivided into three layers, and layer 20 (240 to 516 ft below NGVD 29) in the regional model was subdivided into two layers. The finer vertical discretization was designed to limit vertical numerical dispersion at depth in the aquifer to the extent practicable. Most wells and all hydrologic boundaries are within the top 17 layers where vertical discretization is 10 ft or less.

The modeled area includes outwash and ice-contact deposits to the south and moraine and lacustrine deposits in the north, along Cape Cod Bay (fig. 2). Hydraulic conductivities of the outwash sediments range from 220 ft/d in the shallow part of the aquifer to 10 ft/d in the basal sediments, and hydraulic conductivities of ice-contact deposits decrease from 260 to 10 ft/d with depth; the finer-grained moraine and lacustrine deposits have hydraulic conductivities ranging from 100 to 10 ft/d at depth. The hydrologic-boundary geometries used in the regional model were incorporated into the subregional model, so that the two models were consistent (fig. 6A). Boundary leakances also were the same: 0.2 ft/d for open coastal waters (GHB boundaries), 0.1 ft/d for estuaries and salt marshes (DRN boundaries), and 1.0 ft/d for streams (STR boundaries) (Walter and Whealan, 2004).

The subregional model was linked to the regional model of the Sagamore flow lens by constant-head boundaries along the eastern, western, and northern edges of the subregional model; the version of the regional model used to generate boundary conditions was modified to have the same 26 layers as in the subregional model. For each subregional model run, hydraulic stresses were modified in the regional model to match those to be simulated in the subregional model. The resulting regional head distribution was used as a specified head boundary in the subregional model. On the basis of 2004 stresses, simulated heads in the subregional model closely agreed with heads simulated by using the regional model (fig. 6A); the two sets of heads were within 0.3 ft of one another (or less than 1 percent of the total hydraulic gradient) within the subregional-model domain. The hydrologic budget of the subregional model closely matched the hydrologic budget of the coincident part of the regional model (fig. 7). Most budget components—recharge to the aquifer and discharge into streams, estuaries, and open coastal waters—were within 1 percent of their regional-model equivalents; flows across the constant-head boundaries differed by about 5 percent from flows across the coincident boundaries in the regional model.

Model Stresses

The distribution of recharge was the same in both the regional and subregional models; recharge zones included aquifer sediments, ponds, and wetlands (fig. 6B). Recharge values were set to match the values used in the regional model: 27.1 in/yr to aquifer sediments, 16 in/yr to surface-water bodies, no recharge to wetlands, and additional recharge to aquifer sediments in residential areas representing septic-system return-flow. Twenty-five production wells are within the subregional-model domain, 17 in Barnstable and 8 in Yarmouth (fig. 6A). In 2006, the wells withdrew a total of 4.81 Mgal/d and are projected to withdraw about 6.59 Mgal/d in the future (2020). Simulated pumping rates in 2006 were 5 percent higher than the 2004 pumping rates reported in Walter and Whealan (2004); the projected 2020 pumping rate is about 5 percent lower than the 2020 rates reported in Walter and Whealan (2004).

The recharge distribution includes areas of current and potential sewering and five locations of possible centralized wastewater disposal, including the Barnstable wastewater-disposal facility (BWDF) currently in operation and four candidate sites (sites B, C, D and F) for disposal as identified by

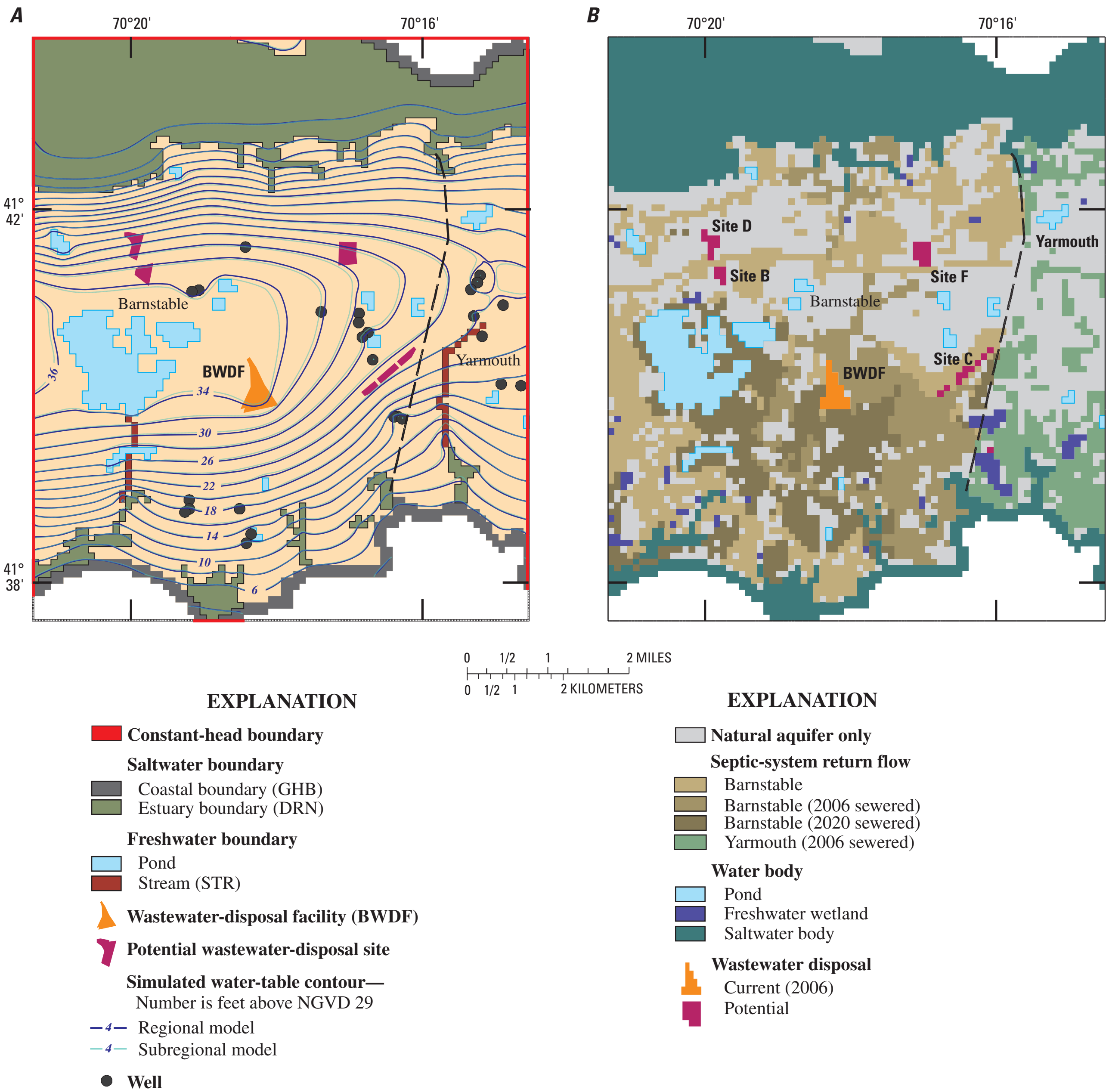


Figure 6. (*A*) Hydrologic boundaries in the subregional model and simulated water-table contours from the regional and subregional models and (*B*) recharge zones in the subregional model, Barnstable and western Yarmouth, Cape Cod, Massachusetts.

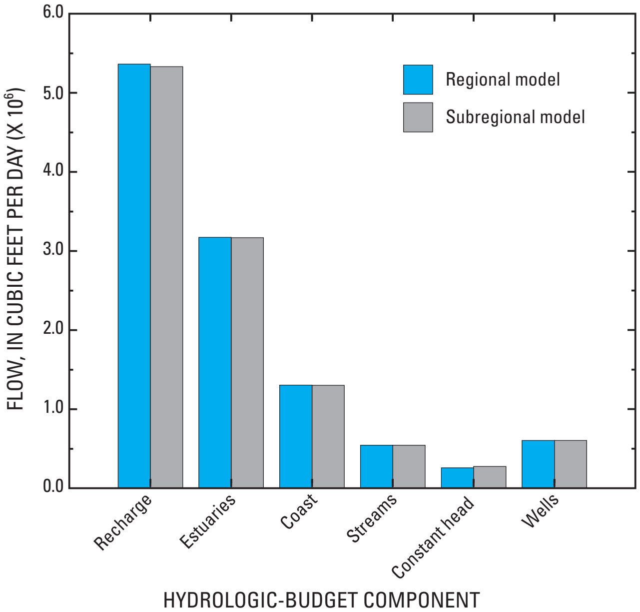


Figure 7. Simulated hydrologic-budget components in relation to flow from the subregional model and the coincident parts of the regional model.

In addition to the physical boundary conditions in the flow model, concentration boundaries were required for the solute-transport simulations. Areas of wastewater disposal (fig. 6B) were simulated as a source of nitrate that was continuous over the period of simulation. Treated wastewater was assumed to have a nitrate concentration of 10 mg/L (Thomas Cambareri, Cape Cod Commission, written commun., 2005). Areas of septic-system return flow also were represented as a continuous source of nitrate. Septic-system effluent was assumed to have a nitrate concentration of 35 mg/L (Weiskel and Howes, 1992) and was adjusted to account for dilution by natural recharge. Inflow concentrations in residential areas (fig. 6B) ranged from about 3.7 mg/L in Barnstable to about 6.9 mg/L in Yarmouth.

The transport equation can be solved by using either the finite-difference method (FDM) or particle-based methods such as methods of characteristics (MOC). The FDM was used for these simulations because the method provides good conservation of mass and is computationally efficient. In advection-dominated systems, particle-based methods generally have less numerical dispersion; however, conservation of mass can be poor. The use of a mass-balance method, such as FDM, in advection-dominated systems can result in some numerical dispersion; however, the problem can be minimized by using numerical models with finer discretization. Numerical dispersion can be approximated as a function of model discretization, velocity, and time-step length with the equation

regional and local planners (Cape Cod Commission, written commun., 2004) (fig. 6B). Sewered areas and areas of centralized wastewater disposal are represented as different recharge zones that allow for the simulation of different combinations of sewering and disposal of different volumes of wastewater at the designated sites. The total volumes of wastewater disposal simulated in the different scenarios for this area ranged from 1.5 to about 7.0 Mgal/d.

Solute Transport

The subsurface transport of nitrate was simulated by using the three-dimensional transport code MT3D (Zheng, 1990). MT3D simulates several components of transport, including advection, mechanical dispersion, and diffusion, and can also simulate some single-species chemical reactions, including contaminant decay and various types of linear and nonlinear sorption. In the case of nitrate, which is conservative and nonreactive (Hem, 1981), the most important components of transport are advection and, to a lesser extent, mechanical dispersion. MT3D is linked to MODFLOW and uses steady-state cell-by-cell flows and velocities from the flow model to simulate the movement of nitrate through the aquifer. The simulation outputs are dissolved and sorbed concentrations of the solute of interest for cells within the active model domain at different transport time steps. This information can be used to develop maps of solute concentrations as well as concentration breakthrough curves.

$$\alpha = \frac{\Delta x}{2} + \frac{v\Delta t}{2}, \qquad (2)$$

where

α = dispersivity,
x = cell length,
v = velocity, and
t = the time-step length (Fletcher, 1991).

With a cell length of 100 ft, a velocity of 1 ft/d, and a time step of 4 d, parameters that are consistent with the subregional model, the approximate numerical dispersion is 52 ft. The Peclet number, which is defined as the ratio of cell length to dispersivity, is about 2. For advection-dominated systems, a Peclet number of less than about 4 indicates that the FDM is appropriate for use in solving the solute-transport equation.

Simulation of Nitrate Transport

The regional models were used to address two sets of questions regarding the regional effects of wastewater disposal: (1) the hydrologic effects of sewering and centralized wastewater disposal, including changes in water levels in wells and ponds and in streamflows, and (2) the discharge of sewage-derived contaminants, particularly nitrate, into wells and ecological receptors. The regional effects of wastewater-disposal scenarios in 8 of the 11 towns on the Sagamore and

Monomoy flow lenses were analyzed as part of this investigation (fig. 5). Because each town had multiple disposal sites as well as a range of possible wastewater-disposal volumes, multiple scenarios were analyzed as part of the investigation. In this report, a modeling analysis of a single hypothetical scenario of nitrate transport is presented to feature hydrologic concepts transferable to other scenarios and to document methods used in the investigation.

Advective Transport and Discharge to Wells and Ecological Receptors

Particle tracking was used to simulate the transport of sewage-derived contaminants through the aquifer from source areas to discharge locations at wells and hydrologic boundaries. Particle-tracking algorithms use model-calculated cell-by-cell flows to track the movement of particles of water through the subsurface. Advective transport, which is the predominant component of transport in the aquifer, is adequate for representing the movement of conservative, nonreactive solutes with ground-water flow if no loss of mass can be assumed. For this analysis, the contaminant of most concern in wastewater is nitrate, which generally is conservative and nonreactive in ground water. As a result, it can be assumed that nitrate is transported primarily by advection, and therefore, flow-based particle tracking can be used to simulate the transport of nitrate through the subsurface and to estimate the potential of effects of sewage-derived nitrate on wells and ecological receptors. Nitrate can be attenuated by denitrification in the presence of organic carbon (Colman and others, 2004), and some nitrate attenuation is assumed in estimating TMDLs for estuaries on Cape Cod (School of Marine Science and Technology, 2005). The aquifer sediments on Cape Cod, however, are generally carbon poor (LeBlanc, 1984), and it is assumed that nitrate is transported conservatively. If some nitrate is attenuated, the estimates of mass-loading rates to receptors in this analysis would represent worse-case conditions.

The movement of wastewater through the aquifer was quantified by assigning particles at wastewater-disposal sites and tracking the particles to their discharge locations. An accounting of particles was used to estimate the volumes of wastewater discharging into wells and receptors; the specified volume of wastewater and the distribution of assigned particles were uniform within disposal areas so that an equal volume of wastewater could be assigned to each particle and the volumes of wastewater discharging to receptors could be estimated from particle counts. These estimates were used, in turn, to estimate mass-loading rates of nitrate by assigning concentrations to the wastewater and assuming that no mass is lost during transport.

The hypothetical scenario features full sewering of residential areas within Barnstable and the disposal of 5.36 Mgal/d of the generated wastewater at the BWDF and at sites B, C, and F (figs. 5 and 6B). Disposal volumes at the four sites are as follows: 1.70 Mgal/d at the BWDF, similar to current flows, and 1.22 Mgal/d each at potential sites B, C, and F. It should be noted that this scenario was and is not under consideration by local planners. However, this hypothetical scenario is similar in terms of locations and volumes of disposal to actual scenarios analyzed and illustrates the types of analyses done as part of this investigation as well as concepts generally pertinent to wastewater disposal in the region.

Discharge to Wells and Ecological Receptors

The water table in east Barnstable and west Yarmouth features an east-west trending ground-water divide; ground-water flow, and therefore advective transport, is away from the divide. Under 2004 stresses, which include pumping and a loading rate of 1.7 Mgal/d of wastewater at the BWDF, the divide trends east-west and is closer to coastal boundaries to the north than those to the south (fig. 8A). The divide deviates to the south near the BWDF owing to mounding associated with wastewater disposal at the site. The areas on the water table that contribute recharge to wells and natural receptors—ponds, streams, and the coast—extend upgradient from the receptors in the direction of the regional ground-water divide (fig. 8A). These contributing areas can change in response to changing hydraulic stresses such as centralized wastewater disposal; any wastewater within the contributing area to a receptor will contribute water to that receptor.

For the hypothetical scenario, the simulated disposal of an additional 3.66 Mgal/d of wastewater (1.22 Mgal/d at sites B, C, and F) changed water levels and hydraulic gradient directions and magnitudes in the aquifer as well as the position of the regional ground-water divide (fig. 9). The divide shifted to the north in response to disposal at sites B and F (fig. 9) relative to its position (assuming 2006 hydraulic stresses) (fig 8A). Sites B and F were near but to the north of the divide when the sites were not in operation and were on top of the divide when the sites were in operation (fig. 9). The hydraulic effects of disposal generally are greater near divides because hydraulic gradients are less steep than in areas closer to discharge boundaries (figs. 6A and 8). As an example, mounding at the water table exceeded 6 ft at sites B and F—near the ground-water divide—whereas the same amount of wastewater disposal at site C, an area of steeper hydraulic gradients farther from the divide, resulted in mounding of about 2 ft (fig. 9).

Sewage-derived contaminants were transported away from the source areas and discharged into wells, ponds, streams, and coastal water bodies (fig. 9). The names of these receptors and their corresponding codes are presented in table 1. Of the 5.36 Mgal/d of disposed wastewater, most (about 70 percent) discharged directly into 10 different coastal-water bodies (fig. 10A). These water bodies (fig. 9) represent estuaries and open coastal-water bodies for which TMDLs will be evaluated as part of the Massachusetts Estuaries Project (Brian Howes, School of Marine Science and Technology (SMAST) and Cape Cod Commission, written commun., 2001). About

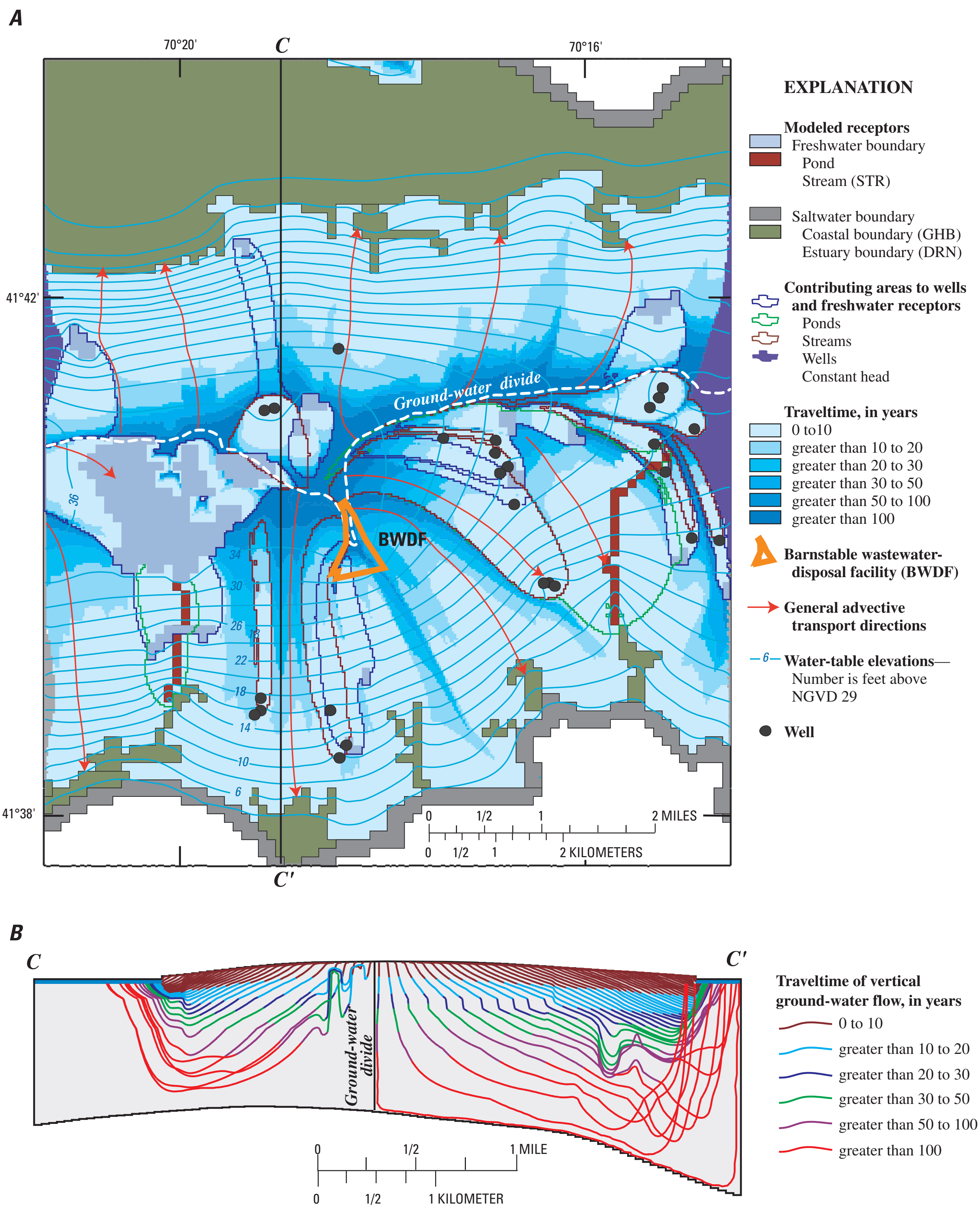


Figure 8. (*A*) Simulated water-table elevations, traveltimes, and general advective-flow paths, and (*B*) generalized vertical ground-water flow for unstressed conditions for Barnstable and western Yarmouth, Cape Cod, Massachusetts.

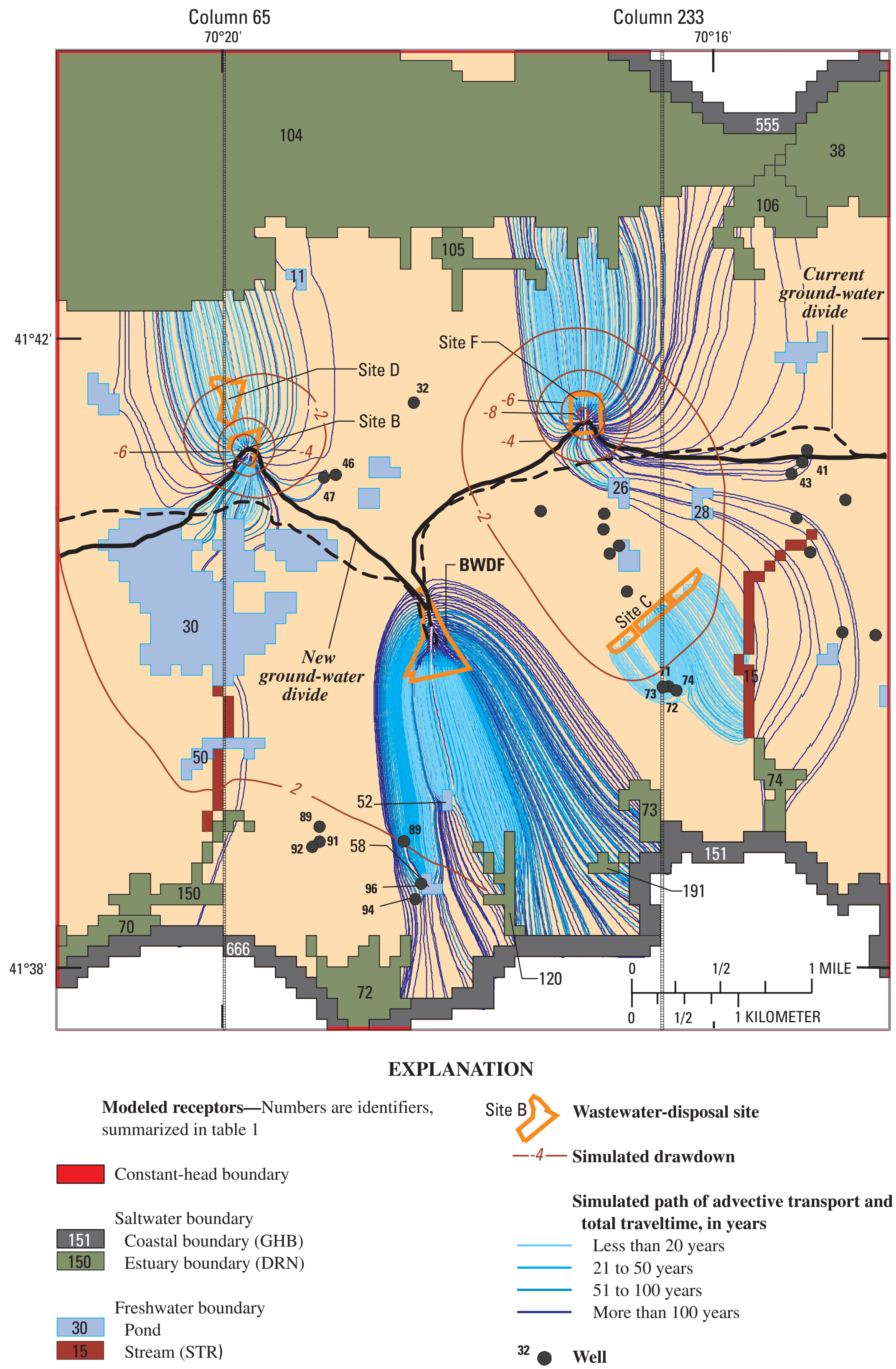


Figure 9. Simulated drawdowns and simulated particle tracks representing advective-flow paths downgradient from wastewater-disposal sites for a hypothetical wastewater-disposal scenario, Barnstable, Cape Cod, Massachusetts.

12 percent of the wastewater discharged into a stream and entered the coastal waters as surface-water inflow. Generally, the direct discharge of sewage-derived nitrogen into coastal waters is more problematic than wastewater that reaches the coast as streamflow (Colman and others, 2004). The remaining 18 percent of the wastewater was intercepted by production wells. About 13 percent of the wastewater flowed through freshwater ponds prior to later discharge into coastal waters or wells. Ponds are considered to be effective at removing nitrogen from the system. Also, pond ecosystems generally are phosphorus-limited and, therefore, are not as sensitive to the adverse effects of sewage-derived nitrogen.

Given the location and volume of wastewater disposal for this scenario, Barnstable Harbor (code 104, fig. 9) received the largest input of sewage-derived nitrogen expressed as volume of discharging wastewater, about 1.7 Mgal/d. Discharging wastewater exceeded 0.5 Mgal/d in Stewarts Creek (code 120) and Nantucket Sound (code 666) (fig. 10A). Generally, open coastal water bodies, such as Nantucket Sound, can more readily assimilate nitrogen owing to greater circulation of water and, as a result, are more favored as receptors of wastewater than are more constricted inland estuaries (Edward Eichner, Cape Cod Commission, oral commun., 2005). In addition to total volumes, the source of discharging wastewater is important to local and regional planners in evaluating the viability of wastewater-disposal sites. Barnstable Harbor received large volumes (more than 0.5 Mgal/d) of wastewater from both sites B and F, whereas most other estuaries received wastewater from a single disposal site (fig. 10A). Also, some sites, such as site B, contributed large amounts of wastewater to a single coastal receptor, whereas other sites, such as the BWDF and site F, contributed smaller volumes to a larger number of receptors (fig. 10A).

Simulated wastewater discharge from the BWDF as well as from proposed sites B and F flowed through freshwater ponds prior to discharging into downgradient wells and ecological receptors (fig. 10B). The largest input of wastewater, about 0.22 Mgal/d, was into Fawcetts Pond (code 52) downgradient from the BWDF. The wastewater-discharge rate exceeded 0.1 Mgal/d to three of the seven ponds that received wastewater in this scenario. No wastewater from these three sites discharged into a freshwater stream. About one-half of the wastewater from site C (about 0.66 Mgal/d) discharged into Phlashes Brook (code 15). No wastewater from site C passed through ponds or discharged directly at the coast.

The discharge of wastewater into a well can adversely affect water quality in the well and can be detrimental to human health. Eleven production wells received wastewater from either the BWDF or from a proposed site; some wastewater from all four sites discharged into a well. Wastewater-discharge rates exceeded 0.1 Mgal/d to 4 of the 11 wells: codes 94, 71, 73, and 74 (fig 10C; table 1). Well 74 received the most wastewater, about 0.25 Mgal/d. Three of these four wells are downgradient from site C; the remaining well 94 is downgradient from the BWDF. Wastewater from sites B and

Table 1. Wells, ecological receptors, and corresponding codes in modeled area of western and central Cape Cod, Massachusetts.

[Ecological receptors are ponds, streams, and coastal water bodies]

Name	Code
Coastal water bodies	
Hyannis Inner Harbor, Barnstable	73
Mill Creek, Barnstable	74
Great Marshes, Sandwich/Barnstable Harbor	104
Maraspin Creek, Barnstable	105
Mill/Short Wharf,Barnstable	106
Stewarts Creek, Barnstable	120
Upper Bumps River	150
Lewis Bay Proper	151
Coast Cape Cod Bay	555
Coast, Nantucket Sound	666
Ponds	
Hinckley Pond	11
Israel Pond	26
Lamson Pond	28
Wequaquet Lake	30
Long Pond	50
Fawcetts Pond	52
Simmons Pond	58
Streams	
Phlashes Brook	15
Wells	
HIGGINS CROW #3a,Yarmouth	41
HIGGINS CROW #1,Yarmouth	43
RT 132 WELL #3,BFD	46
Rt. 132 Well #4,BFD	47
MAHER #3,BWC	71
MAHER #1a,BWC	72
MAHER #1,BWC	73
MAHER #2,BWC	74
Straightway 2,BWC	89
SIMMONS POND W,BWC	94
HYANNISPORT W,BWC	96

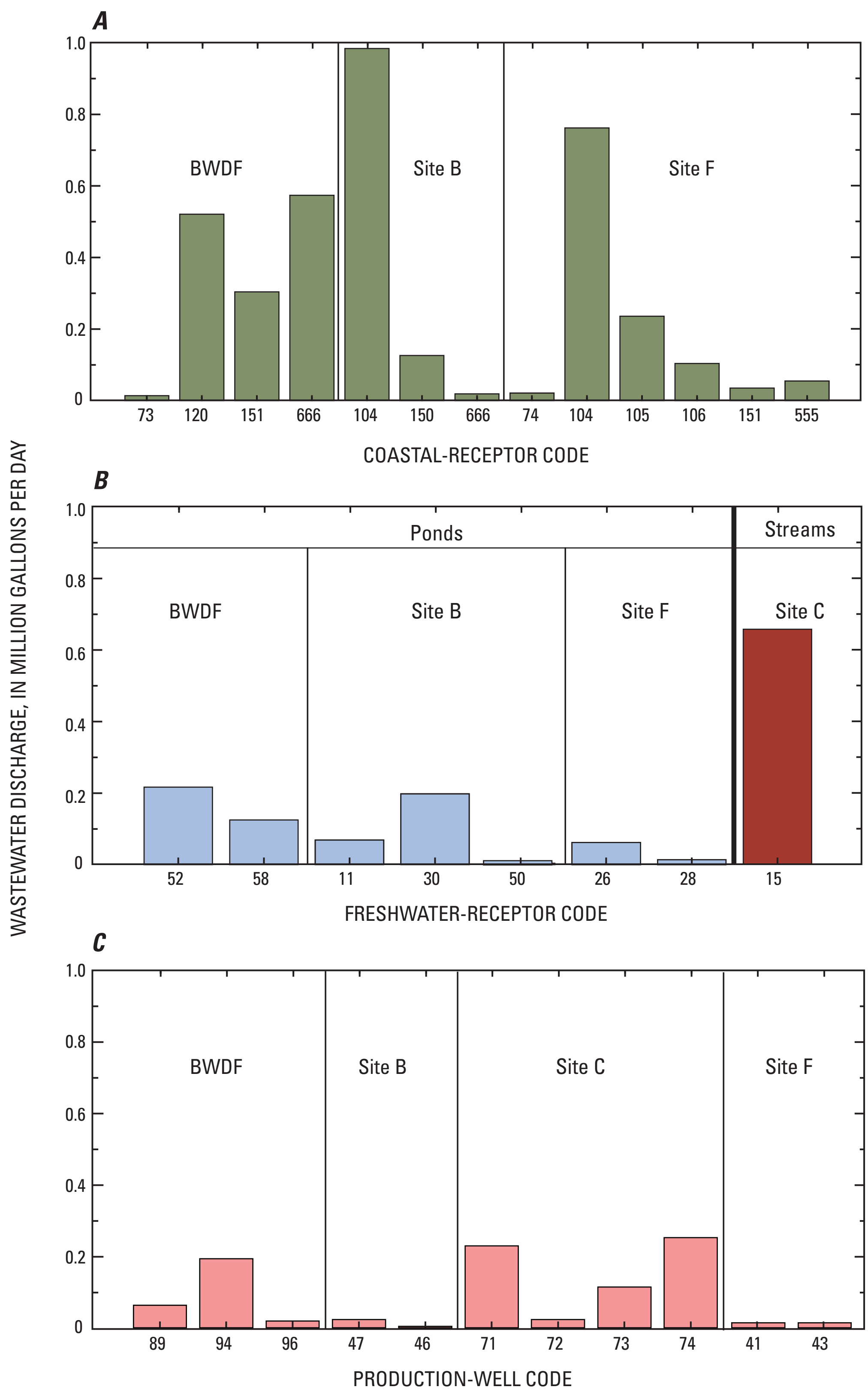


Figure 10. Model-estimated wastewater discharge rates into (*A*) coastal water bodies, (*B*) ponds, streams, and (*C*) production wells for a hypothetical wastewater-disposal scenario, Barnstable and Yarmouth, Cape Cod, Massachusetts.

F discharged into two wells each; however, the volumes of discharge were low (0.02 Mgal/d or less).

The use of numerical models to estimate wastewater-loading rates to wells and natural receptors provides several benefits in evaluating wastewater-disposal scenarios. Modeling analyses can differentiate wastewater that discharges into receptors of greater concern, such as inland estuaries and wells, from wastewater that discharges into receptors of less concern, such as ponds, streams, or open coastal waters. Modeling analyses with a sufficient level of detail also can help determine the viability of proposed wastewater-disposal sites on the basis of possible ecological effects on sensitive receptors.

Advective Transport in Different Hydrologic Settings

Patterns of advective transport would be expected to differ in different hydrologic settings, particularly on the basis of the location of a potential disposal site relative to ground-water divides and hydrologic boundaries. In areas near divides, hydraulic gradients are small, and flow can diverge in different directions. Also, vertical flow is enhanced and velocities are smaller than in downgradient areas closer to hydrologic boundaries, where hydraulic gradients are steeper and flow directions more constrained.

Flow is outward from the regional divide towards hydrologic boundaries. Water enters coastal estuaries either as direct ground-water discharge or as streamflow. Some water is extracted for water supply, and recharge within contributing areas to ponds flows through the ponds prior to discharging into estuaries, streams, or wells (fig. 8A). Traveltimes, which are defined as the total transport time from recharge at the water table to discharge at a hydrologic boundary, are important considerations in the evaluation of wastewater disposal because nitrogen attenuation is assumed to increase with longer traveltimes (Colman and others, 2004; Valiela and others, 2000). Wastewater disposal that contributes water to an estuary with traveltimes longer than 10 years is assumed to pose less ecological risk than disposal of wastewater with traveltimes shorter than 10 years (School of Marine Science and Technology, 2005). Traveltimes to the coast are largest (longer than 100 years) for recharge that originates near the divide because velocities are smaller there, and these areas are farther from the discharge locations (fig. 8A); traveltimes for recharge that originates close to discharge locations are small (less than 1 year). Traveltimes to coastal discharge boundaries are shorter than 10 years in about 58 percent of the total water-table area that contributes water to the discharge boundaries. For ponds and streams, the percentages of contributing area for which recharge traveltimes are shorter than 10 years are 82 and 72 percent, respectively. About 57 percent of water captured by production wells traveled less than 10 years from the recharge area at the water table.

Vertical flow paths and subsurface traveltimes under unstressed conditions (no pumping or return flow) are shown in figure 8B. In areas close to the divide, vertical gradients are enhanced relative to horizontal gradients, and advective flow paths are more downward than areas nearer the coast where vertical gradients are lower and flow paths are more horizontal. Flow is locally upward at coastal-discharge locations. This general pattern differs in the vicinity of local hydrologic features, such as ponds, streams, and wells; some perturbations are related to changes in lithology within the simulated aquifer. The particle paths shown in figure 8B were weighted with equal volumes of recharge water so that the volumes around the particles are analogous to stream tubes, and therefore, qualitatively represent ground-water fluxes with depth. Fluxes and velocities are higher nearer the coast as illustrated by the larger density of particles in those areas; conversely, fluxes and velocities near the ground-water divide are lower. In some areas, such as immediately below the divide, fluxes are small, and solutes move more slowly through that part of the aquifer; this pattern is reflected in the regional distribution of traveltimes (fig. 8A); long traveltimes (longer than 100 years) for recharge that originates near the regional ground-water divide are a result of transport through deep areas of the aquifer with small ground-water fluxes.

In the hypothetical scenario, wastewater disposal at three sites was on or close to the regional ground-water divide (BWDF and sites B and F) and at one site farther from the divide (site C) (fig. 9A). Hydraulic conditions at the three sites were reflected in the advective flow paths emanating from the source areas. Along the ground-water divide at sites B and F, simulated particle tracks fanned out in a wide arc of flow directions (fig. 9A). At site B, particle paths were both to the north into the Barnstable Harbor (code 104) estuarine system as well as to the south into Wequaquet Lake (code 30) and the coastal boundaries near Nantucket Sound (code 666). Similarly, advective flow from site F discharged to coastal boundaries to the north and south and to production wells to the east. In contrast, flow paths from site C had a smaller range of flow directions; wastewater discharged directly into Phlashes Brook (code 15) and into production wells directly downgradient from the site. Flow paths from the BWDF were to the south, but particle paths fanned out in a wider range of flow directions than particles from site C owing to the proximity of the regional ground-water divide to the BWDF source area (fig. 9A).

Traveltimes to coastal receptors differed by source area and receptor (fig. 11A). Median traveltimes from the BWDF source area to coastal receptors ranged from 22.7 years for Stewarts Creek (code 120) to 133 years for Nantucket Sound (code 666) (fig. 11A). The difference in traveltimes is a function of the setting of these hydrologic boundaries in the regional flow system. Stewarts Creek is an inland estuary that intercepts shallow ground water from a part of the aquifer with higher fluxes and velocities. Conversely, Nantucket Sound is the outer boundary of the flow system and captures ground

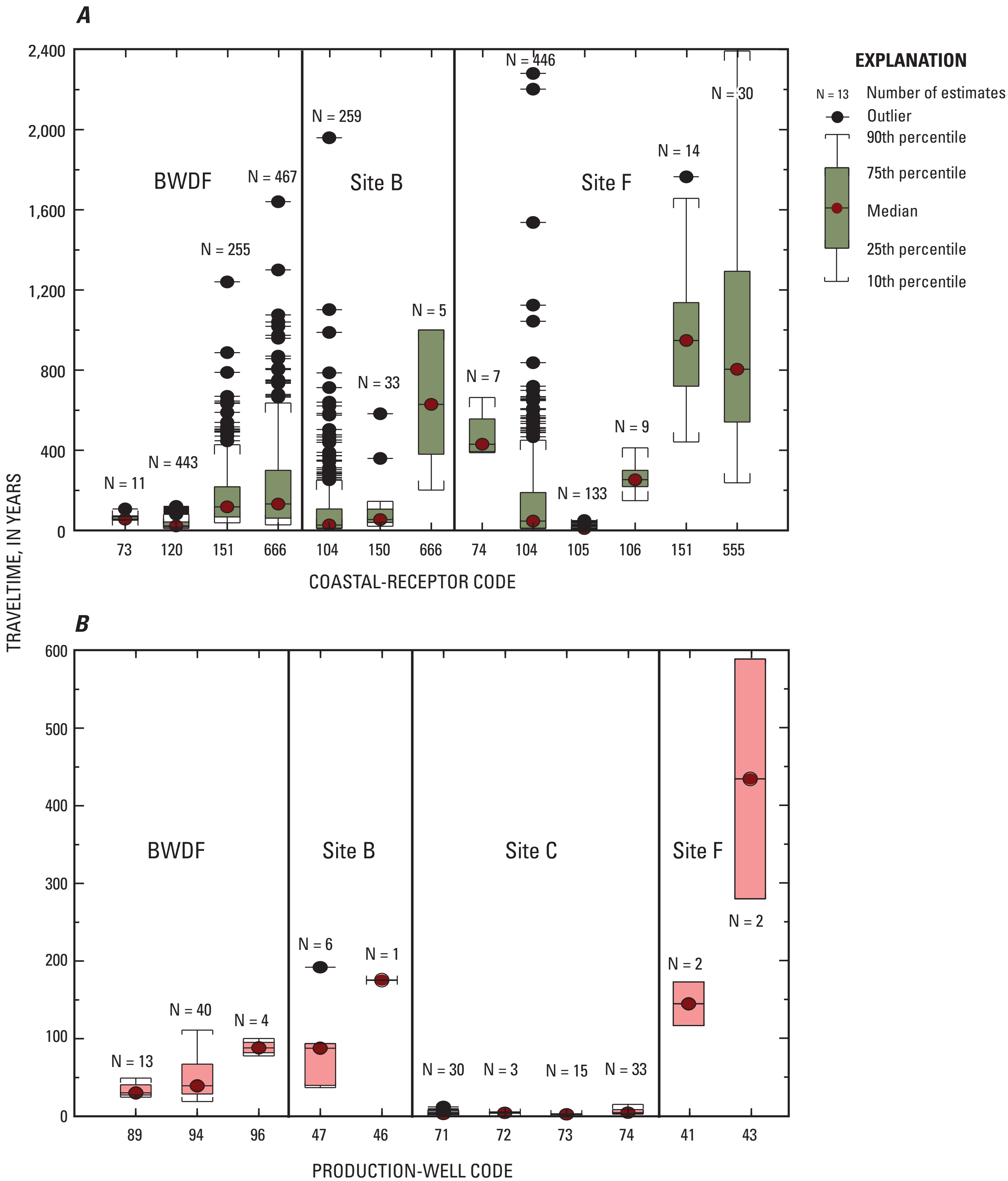


Figure 11. Model-estimated traveltimes to (*A*) coastal water bodies and (*B*) production wells, Barnstable and Yarmouth, Cape Cod, Massachusetts.

water from deeper in the flow system; this ground water recharged the aquifer through a part of the source area closer to the divide and traveled deeper in the aquifer at a slower velocity than water recharged near the coast (fig. 3B). Stewarts Creek and Nantucket Sound received similar volumes of wastewater from the BWDF; about 0.52 and 0.58 Mgal/d, respectively (fig. 10A). The traveltimes, however, indicated that Stewarts Creek may be more adversely affected by sewage-derived nitrate than Nantucket Sound owing in part to shorter traveltimes and, presumably, less subsurface nitrate attenuation.

Most wastewater from sites B and F was transported to the north and discharged into Barnstable Harbor (code 104) (fig. 10A). Median traveltimes for wastewater discharged from the two source areas to Barnstable Harbor were 26.8 and 47.4 years, respectively. Both sites were at the ground-water divide and flow fanned out in a number of different directions (fig. 9A). As a result, traveltimes of wastewater from sites B and F to individual receptors also differed widely (fig. 11A). Wastewater from site B discharging into Nantucket Sound (code 666) had a median traveltime of more than 600 years; wastewater from site F discharging into Lewis Bay (code 151) had a median traveltime of about 900 years. In both cases, the wastewater flowed very deep into the system and discharged along the south shore. Most receptors had outliers with very large traveltimes (longer than 1000 years); these outliers represent particles that moved deeply through near-stagnant parts of the aquifer with little ground-water flux and slow velocities.

Traveltimes to production wells also differed by source area and individual well. Source areas at the BWDF and site C discharged large volumes of wastewater to wells (fig. 10C). For wastewater originating at the BWDF, median traveltimes ranged from 30.3 years (well 89) to 88 years (well 96) (fig. 11B). At site C, which discharged nearly half of its generated wastewater into four production wells, median traveltimes were all shorter than 5 years, owing to the shallower flow paths and faster velocities in that part of the aquifer as well as to the proximity of the wells to the source area (fig. 11B). As an example of the importance of traveltimes, well 94 received nearly 0.2 Mgal/d of wastewater from the BWDF, similar to the volumes received by wells 71 and 74 downgradient from site C; however, the median traveltime of wastewater received in well 94 is about 40 years compared to fewer than 5 years for wells 71 and 74 (fig. 11B). Based on the assumption that nitrate is attentuated over long transport times, the results of this simulation suggest that more nitrate would be attenuated prior to wastewater discharge into wells downgradient from the BWDF source area, and that wells downgradient from site C would be more adversely affected by wastewater disposal. Only small volumes of wastewater originating from sites B and F were captured by production wells. Traveltimes ranged from 86 to more than 400 years, indicating that the wells capture wastewater that had traveled along deep flow paths from the source areas.

Ground-water fluxes and velocities depend on location within the aquifer, and wastewater discharged in different hydrologic settings would be transported at different velocities. To generate a three-dimensional image of the velocity field within the aquifer, velocity vectors were computed for each segment of each particle track from the four simulated source areas to the discharge areas (fig. 12A). At the BWDF and at sites B and F near the ground-water divide, median velocities were less than 0.5 ft/d. At site C farther from the divide and in a shallower, more horizontal flow regime, the median velocity was 1.7 ft/d. Median total traveltimes of wastewater originating from the three source areas near the divide (the BWDF and sites B and F) were 66, 31, and 48 years, respectively, whereas the median traveltime at site C farther from the divide was less than 10 years (fig. 12B).

Vertical flow paths along two columns in the model from sites in two different hydrologic settings—sites B and C—are shown in figure 13; the sites are at similar distances from hydrologic boundaries and receive the same volume of wastewater. For these simulations, no pumping was simulated at wells 71, 72, 73, 74, 46, and 47 so that advective transport would proceed to natural hydrologic boundaries from both sites; it should be noted that these stresses were different than in the scenario shown in figure 9, and therefore, flow lines in the two scenarios would not coincide. Flow paths from site B at the top of the ground-water divide were very deep—to near the bottom of the aquifer—and large volumes of the aquifer received wastewater. Flow was predominantly to the north; however, some flow was also to the south into Wequaquet Lake. Wastewater entering the lake also discharged from the lake and proceeded to the coast, but it was assumed that any associated nitrogen was attenuated within the pond. Some particles underflowed the lake and discharged into coastal boundaries along the south shore, but only after traveltimes longer than 100 years (fig. 13A).

From site C, flow paths were shallow and a smaller volume of aquifer received wastewater. Traveltimes, however, were significantly shorter (fig. 13B). Traveltimes of wastewater discharged from site B were longer and more variable than traveltimes of wastewater from site C; the minimum, maximum, and median traveltimes from site B were about 8, 2095, and 32 years, respectively. The median traveltime of wastewater discharged from site C was about 10 years, and the range was about 3 to 72 years. This illustrates the importance of hydrologic setting in considering the downgradient proximity of receptors of concern and evaluating total transport time to the receptors from proposed wastewater-disposal sites.

Solute-Transport Simulations

The use of a solute-transport code has some advantages for evaluating the transport of wastewater-derived nitrate from source areas to receptors. Simulations with such a code can (1) estimate the distribution of nitrate in the aquifer, expressed as a concentration, (2) evaluate changes in nitrate concentrations over time at receptors, and (3) simulate the effects of mechanical dispersion on nitrate transport in the aquifer.

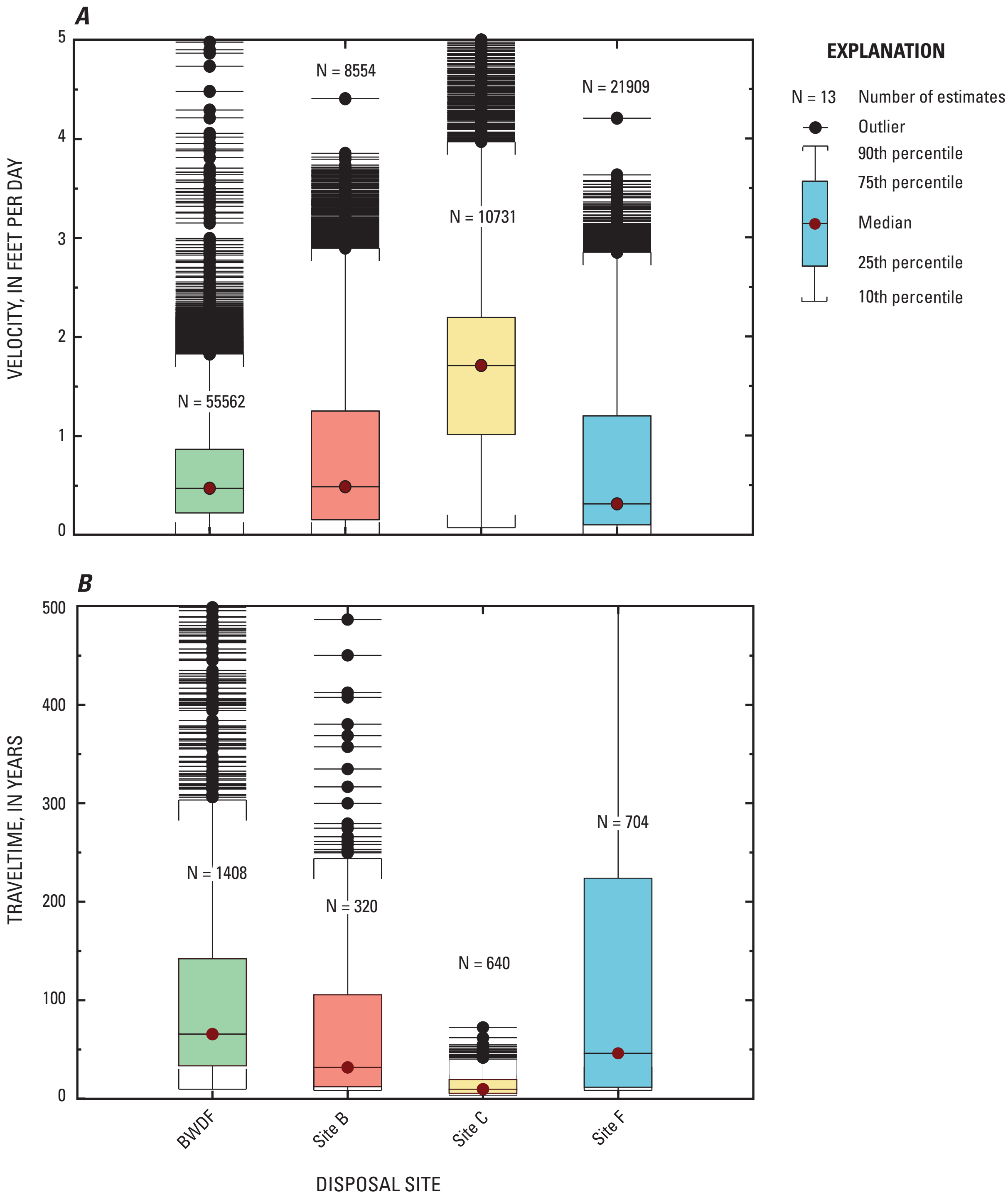


Figure 12. (*A*) Ground-water velocities and (*B*) traveltimes for advective-transport paths from four wastewater-disposal sites for a hypothetical wastewater-disposal scenario, eastern Barnstable, Cape Cod, Massachusetts.

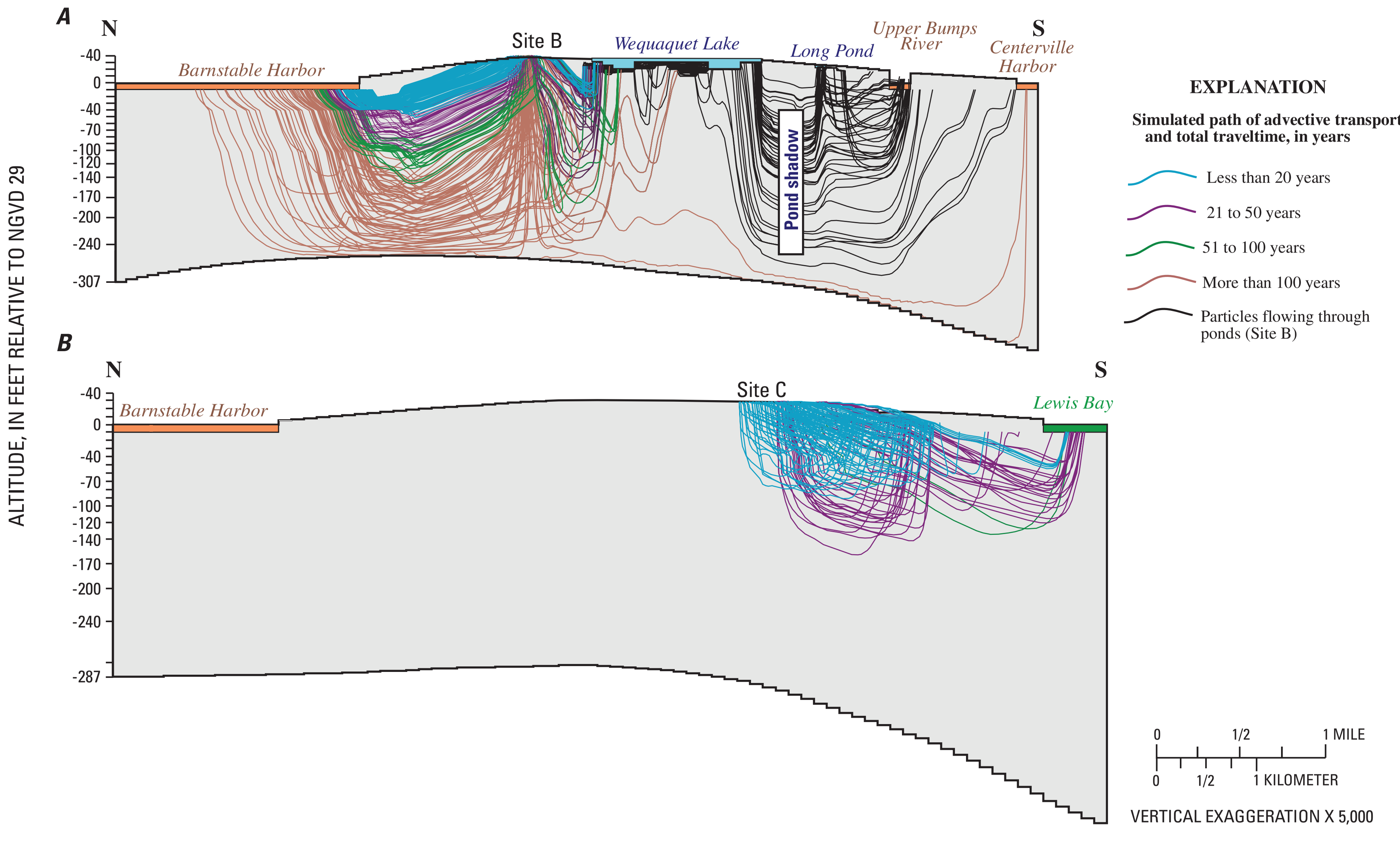


Figure 13. Vertical particle tracks representing advective-transport paths under the condition of no pumping downgradient from (*A*) site C and (*B*) site B near the Barnstable Wastewater Disposal Facility, Cape Cod, Massachusetts.

Particle tracking also can yield insight into the distribution of nitrate in the aquifer; however, the information from particle-tracking simulations is qualitative. As an example, the particle paths shown in figures 8 and 13 illustrate that a larger portion of the aquifer received wastewater from site B than from site C; wastewater from site B flowed across a wider arc of flow directions horizontally and much deeper in the aquifer vertically. The particle tracks, however, do not indicate the distributions of solute mass or of concentrations, which are of importance in evaluating the potential effects of wastewater disposal on ground-water quality. In addition, solute-transport models can more readily be used to evaluate time-varying concentrations and mass-loading rates at downgradient receptors. Although particle tracking can be used to quantify mass-loading rates at receptors by using mass-weighted particles, these methods often involve additional postmodeling analyses. The use of volume- or mass-weighted particles in modeling analyses for ground water on Cape Cod has been reported previously in Barlow (1997) and Walter and others (2002).

Solute-transport models can be used to represent the subsurface distribution of nitrate in the aquifer. Simulated nitrate concentrations downgradient from the BWDF at different horizontal sections through the nitrate plume are shown in figs. 14A-F. The concentration of nitrate in the discharged wastewater was simulated as 10 mg/L; after 100 years of disposal, and assuming conservative transport, the maximum extent of nitrate concentration exceeding 1 mg/L was in model layer 18 (at 110 ft below NGVD 29) (fig. 14D). Maximum nitrate concentrations in this layer exceeded 8 mg/L at the coast. The distribution of nitrate is controlled in part by hydrologic boundaries. As an example, Fawcetts Pond (code 52) is a flow-through pond that focuses ground-water flow, and as a result, the patterns of nitrate concentrations are affected by the pond (figs. 14B–F).

Nitrate concentrations along vertical sections (column 147 and row 304, fig. 14A) are shown in figures 14G and H. Concentrations exceeded 1 mg/L to an elevation of more than 300 ft below NGVD 29 (layer 24) beneath the disposal beds, near the bedrock surface; this pattern was consistent with the location of the BWDF near the regional ground-water divide where vertical flow paths were enhanced (fig. 14B). Nitrate concentrations along the coast exceeded 1 mg/L to a depth of about 250 ft below NGVD 29. Nitrate concentrations exceeding 1 mg/L were simulated at shallower depths in the aquifer near hydrologic boundaries than in the aquifer farther from the boundaries; this result indicates upward flow toward those boundaries. Examples are Fawcetts Pond (fig. 14G), Stewarts Creek, and Nantucket Sound (fig. 14H).

The solute-transport model also can be used to evaluate nitrate concentrations in receptors that receive wastewater discharge. Nitrate concentrations vary with time and with the location of a receptor relative to a source area (fig. 15). At Fawcetts Pond, concentrations increased to 8.1 mg/L within about 45 years after discharge from the BWDF; the pond is close to the BWDF source area (fig. 14A). At well 89, which is near the western edge of the nitrate plume (fig. 9), concentrations increased to about 1.3 mg/L within about 65 years. Concentrations were constant after 100 years of transport; this result indicated that a sufficient transport time had elapsed to establish a steady-state condition with respect to nitrate concentration. At Stewarts Creek, concentrations increased to about 7.9 mg/L after 100 years of transport (fig. 15); although concentrations were still increasing after that time, the reduced rate of increase indicates a near-steady-state condition. At Nantucket Sound, concentrations increased to about 3.2 mg/L after 100 years of transport. The continuing increase in concentration indicates that a steady-state condition had not been reached and that concentrations would continue to increase for transport times beyond 100 years.

After 100 years of transport, the distribution of mass downgradient from the four wastewater-disposal sites is a result of advective transport patterns, and therefore the hydrologic setting of the disposal site. As an example, nitrate concentrations exceeded 0.1 mg/L over a large area downgradient from site F (fig. 16). Because site F is near the regional ground-water divide, discharge from this site generated a wide arc of advective flow paths from the site (figs. 9 and 16). Beneath site B, also near the divide, concentrations exceeding 9 mg/L extended to near bedrock (the full saturated thickness of the aquifer) owing to enhanced vertical flow beneath the site (figs 13A and 17A). At site C, farther from the ground-water divide concentrations exceeded 0.1 mg/L over a smaller area than at sites B and F, reflecting more constrained advecive flow paths (figs. 9 and 16). Flow from this site was more horizontal and advective-transport paths were shallower; concentrations exceeding 9 mg/L extended to about 70 ft below NGVD 29 (or the upper 25 percent of the saturated thickness) after 100 years of transport (figs. 13B and 17B). Although advective-transport paths did not extend deeper than about 140 ft below NGVD 29 (or the upper 28 percent of the saturated thickness) (fig. 13B), simulated concentrations beneath site C exceeded 0.1 mg/L to the depth of bedrock (fig 17B). It is likely, however, that there is no nitrate mass at depth in the aquifer and that the simulated concentrations were caused by numerical dispersion. Vertical discretization is larger at depth in the model, and therefore numerical dispersion, which is a function of model-cell size, would be expected to be enhanced there. Although mass fluxes at depth in the aquifer are small, this simulated scenario illustrates the need to consider numerical dispersion when evaluating simulated concentrations.

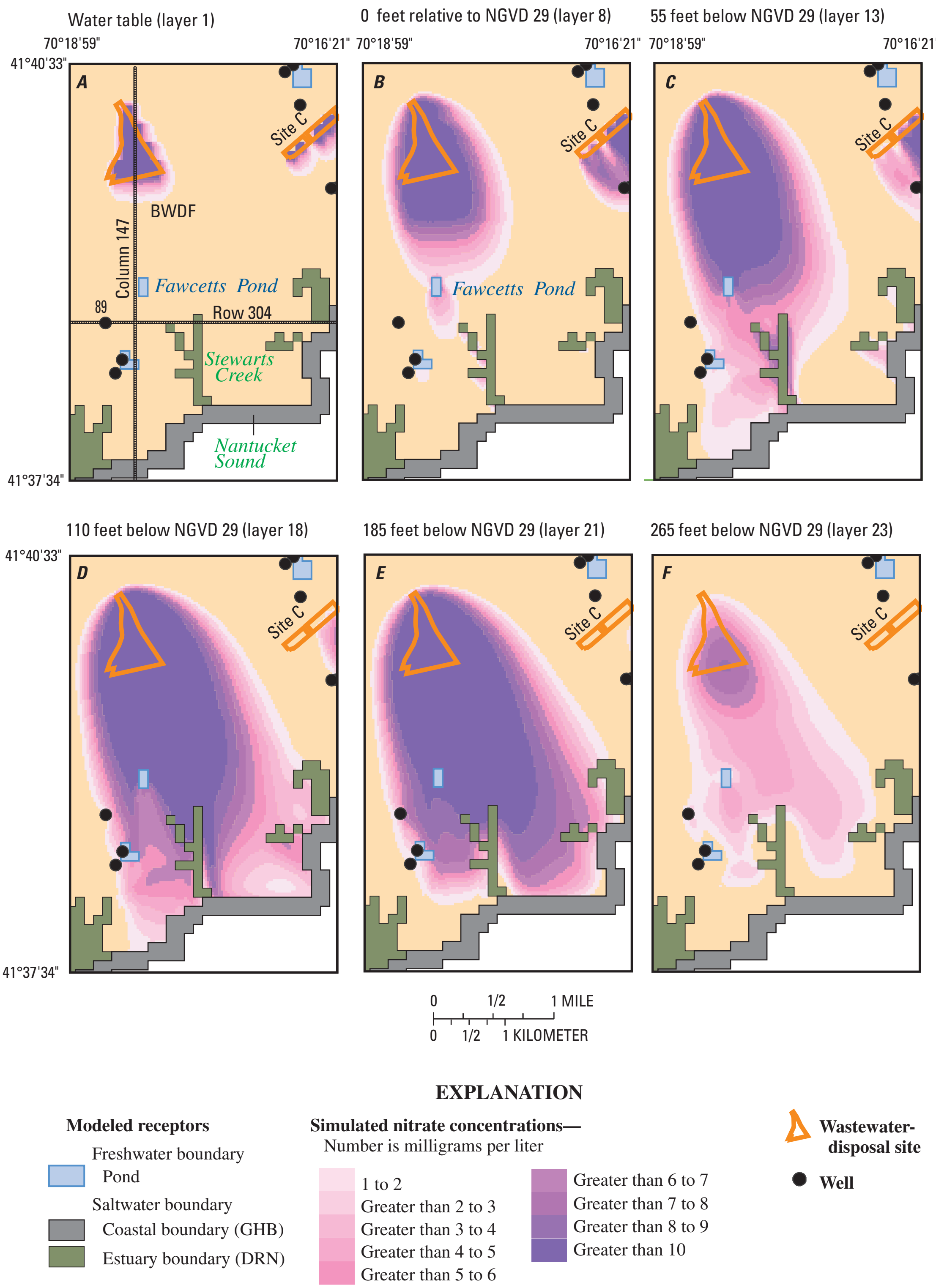


Figure 14. (*A-F*) Simulated nitrate concentrations downgradient from the current wastewater-disposal facility for six horizontal sections and (*G-H*) simulated nitrate concentrations along vertical column 147 and row 304.

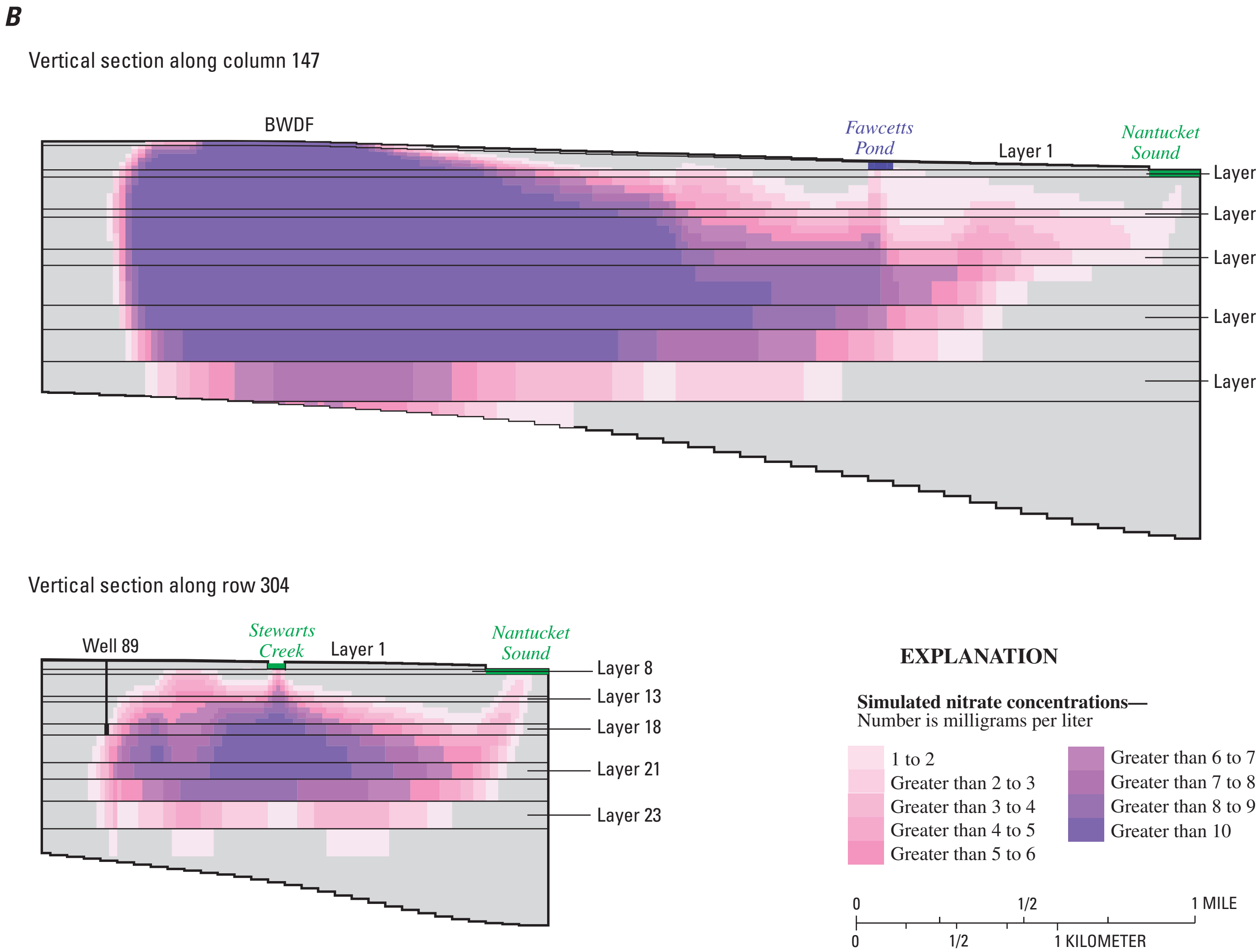


Figure 14. (*A-F*) Simulated nitrate concentrations downgradient from the current wastewater-disposal facility for six horizontal sections and (*G-H*) simulated nitrate concentrations along vertical column 147 and row 304.—Continued

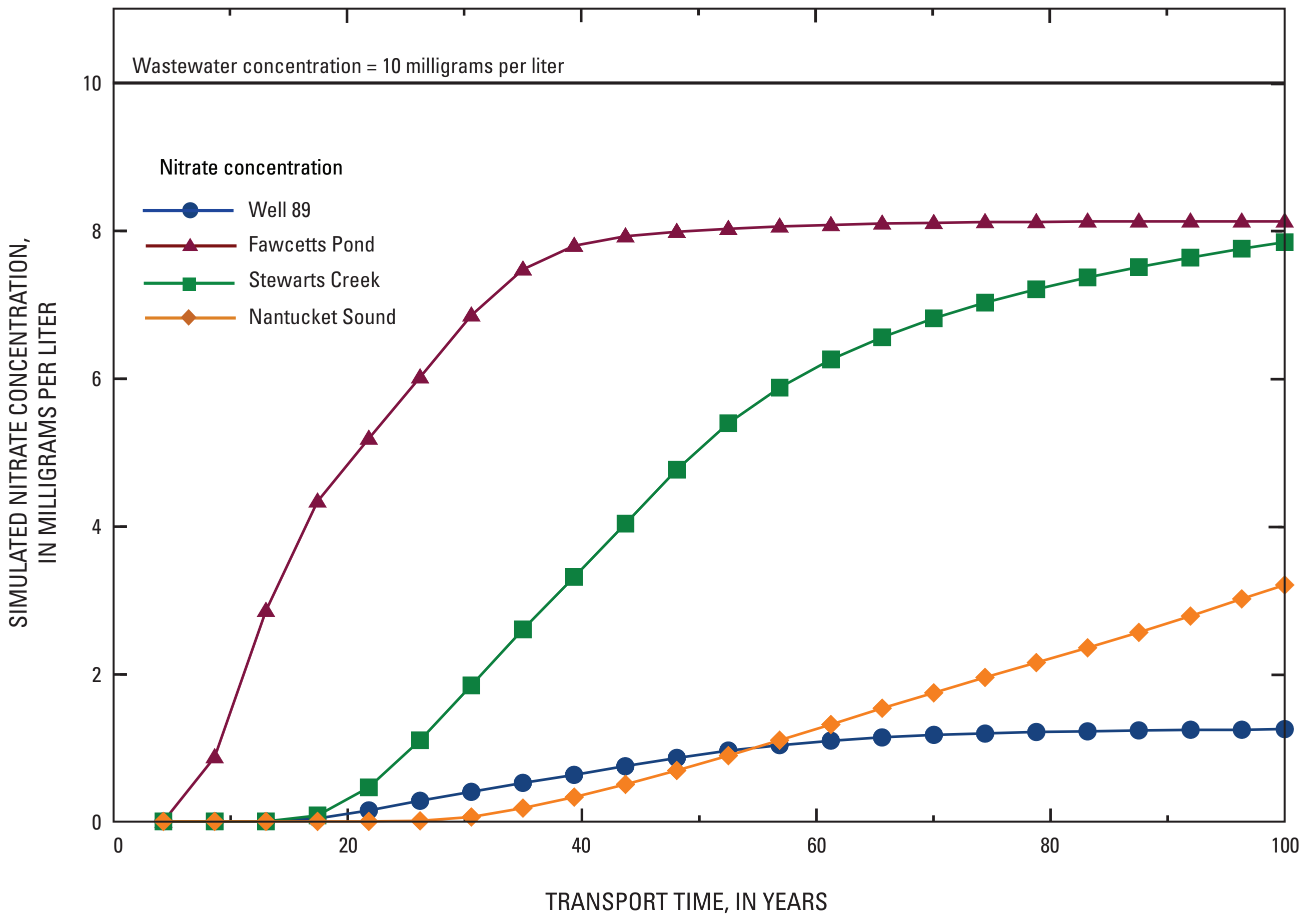


Figure 15. Simulated time-varying nitrate concentrations at well 89, Fawcetts Pond, Stewarts Creek, and Nantucket Sound, Cape Cod, Massachusetts.

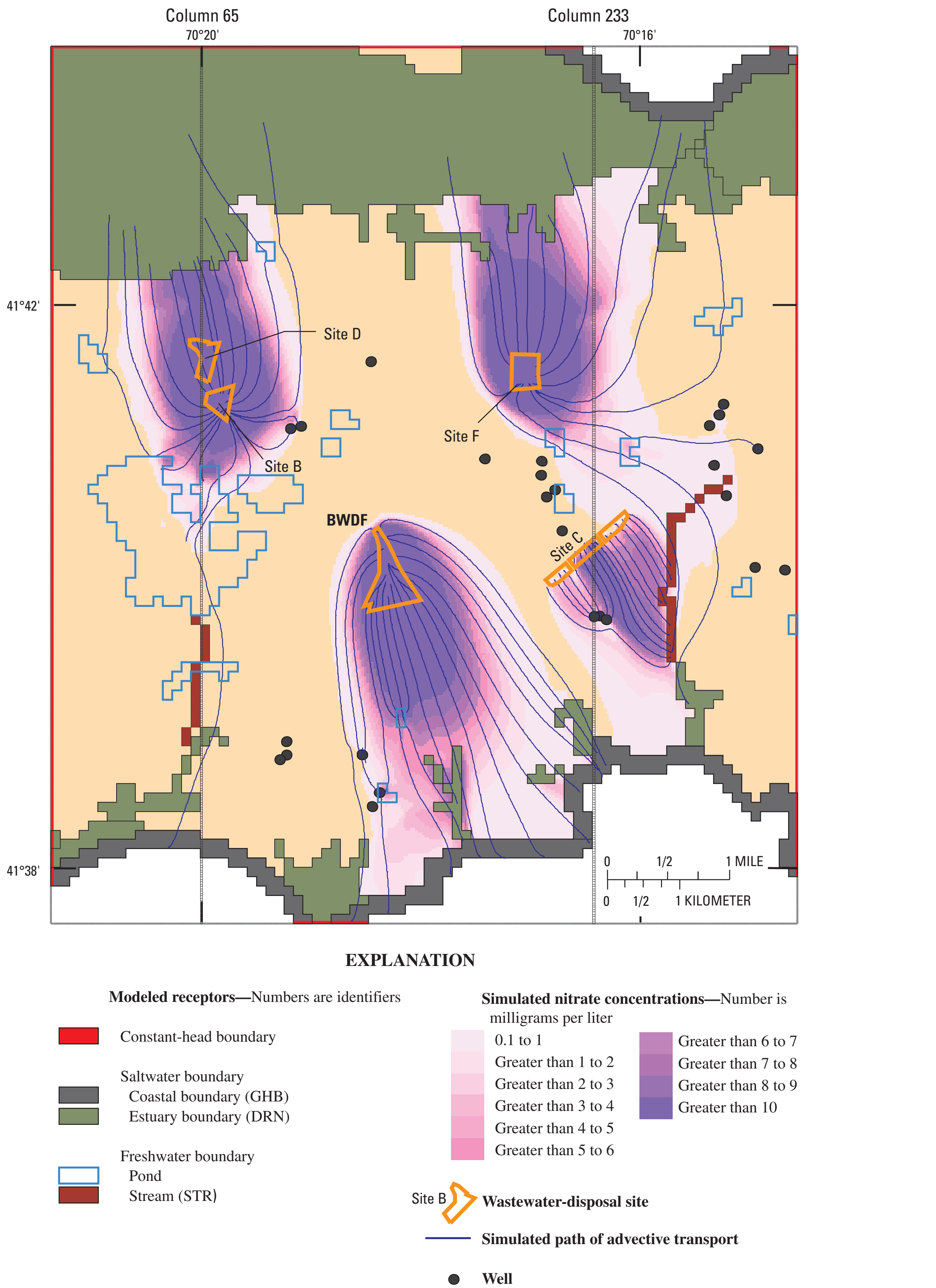


Figure 16. Simulated nitrate concentrations downgradient from five wastewater-disposal sites and simulated advective-flow paths in model layer 13 (50 to 60 ft below NGVD 29) near the Barnstable Wastewater-Disposal Facility, Cape Cod, Massachusetts.

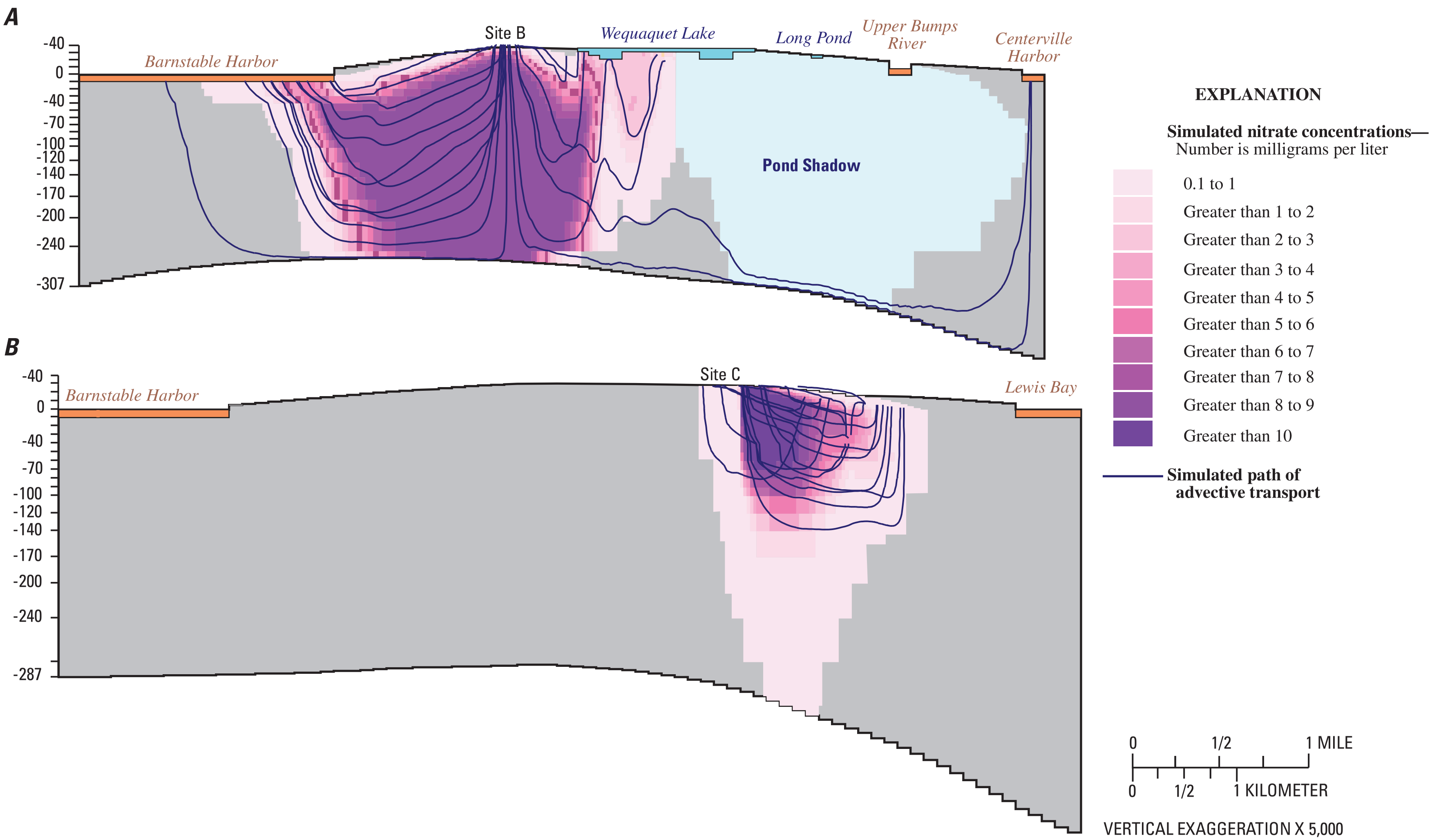


Figure 17. Nitrate concentrations and particle tracks representing advective-transport paths downgradient from (*A*) site B and (*B*) site C near the Barnstable Wastewater-Disposal Facility, Cape Cod, Massachusetts.

Comparison to Advective-Transport Simulations

Both solute-transport and particle-tracking methods can simulate the movement of a conservative solute, such as nitrate, through the aquifer. Particle-tracking methods use mass-weighted particles to simulate the movement of a solute and can estimate mass-loading rates at receptors through an accounting of particles (fig. 18A). This method requires no additional model development beyond development of the flow model but does require postmodeling analyses to convert particle track information into useful estimates of mass and concentration. In addition, representing mass by discrete particles nay not yield sufficient resolution for some scenarios, and the assumption that there is no dispersion or loss of mass during transport is not valid in most systems. Conversely, solute-transport methods explicitly simulate concentration and mass at all points within the model domain at each transport time step and can simulate transport processes, such mechanical dispersion and sorption; however, this approach requires additional model development and the assembly of additional input-data sets. In addition, numerical dispersion needs to be considered when interpreting model results. Particle-tracking results in complete conservation of mass along flow lines (fig. 18A), whereas solute-transport methods allow for spreading of mass between flow lines (transverse and vertical dispersion) (fig. 18B). Solute-transport methods also allow for the spread of mass in the direction of flow (longitudinal dispersion). Although mass is conserved for the entire system in the case of nonreactive solutes such as nitrate, the inclusion of dispersion in the transport process can result in changes in mass along flow lines (fig. 18B). The choice of method should be based on the types of questions being addressed and the level of resolution and accuracy needed.

Particle tracks from the four wastewater-disposal sites and simulated nitrate concentrations in model layer 13 (50 to 60 ft below NGVD 29) are shown in figure 16; it should be noted that the comparison is between mass in a single model layer and a projection of three-dimensional particle tracks onto a two-dimensional map surface. The spatial distribution of particle tracks generally corresponds to the distribution of mass simulated by the solute-transport model (fig. 16). Thus, the two methods delineate the same general areas of the aquifer that receive wastewater-derived nitrate; however, the particle tracks intrinsically do not include any quantitative information regarding the distribution of mass. The agreement between simulated vertical particle tracks and nitrate mass near sites B and C is also close (figs. 17A and B). Both also delineate the same general volumes of the aquifer that receive

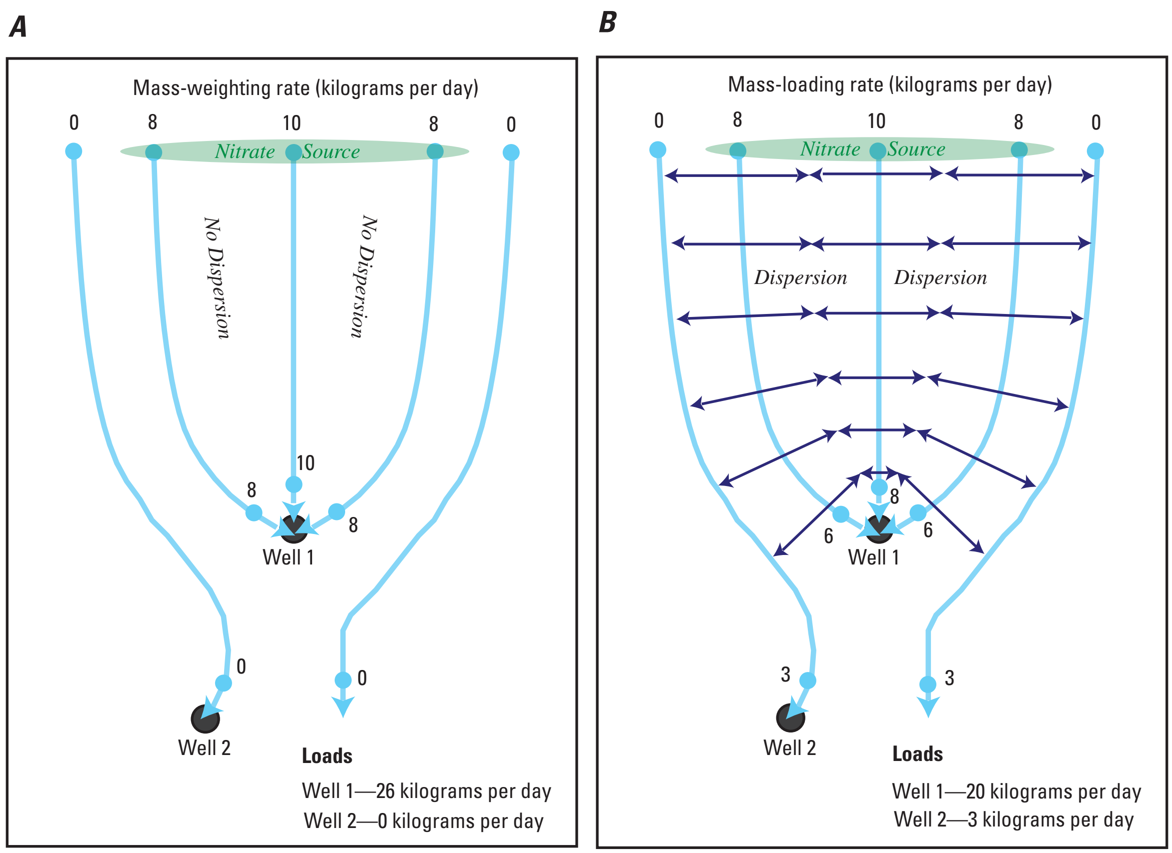


Figure 18. Mass-loading rates estimated from (A), flow-based particle tracking and (B) solute-transport modeling at two downgradient production wells, Cape Cod, Massachusetts.

wastewater-derived nitrate. The spreading of concentrations between 0.1 and 1 mg/L below the particle tracks to the bedrock surface beneath site C was likely caused by numerical dispersion (fig. 17B).

Both particle-tracking and solute-transport methods can provide estimates of mass-loading rates into wells and ecological receptors. Flow-based particle-tracking methods calculate mass-loading rates by using a set of discrete particles to represent the load of wastewater being discharged into the aquifer. From the known concentration of nitrate in the wastewater, a mass can be assigned to each particle, and the fate of the nitrate can be determined by an accounting of particles that discharge into each receptor. If the distribution of particles representing the loading at the site is uniform, the load associated with each particle is determined by multiplying the concentration by the wastewater-discharge rate and dividing that result by the number of particles. Important considerations to make when estimating mass-loading rates by particle tracking include (1) the need to use a sufficiently large number of particles to represent the advective-flow system adequately and minimize the amount of mass represented by each particle and (2) the possibly invalid assumption that mass does not change along flow lines because of dispersion. Solute-transport methods calculate mass-loading rates into a receptor by multiplying the simulated concentration by the ground-water flux into the receptor.

An important concern in Barnstable and western Yarmouth, the most densely populated area on Cape Cod, is the potential for the discharge of wastewater-derived nitrate into production wells from one current and several potential wastewater-disposal sites. For the hypothetical scenario described earlier, the particle-tracking method simulated a range of mass-loading rates to the wells of 10.1 kg/d (well 74) to 0.1 kg/d (well 46) (fig. 19). Well 74 is directly

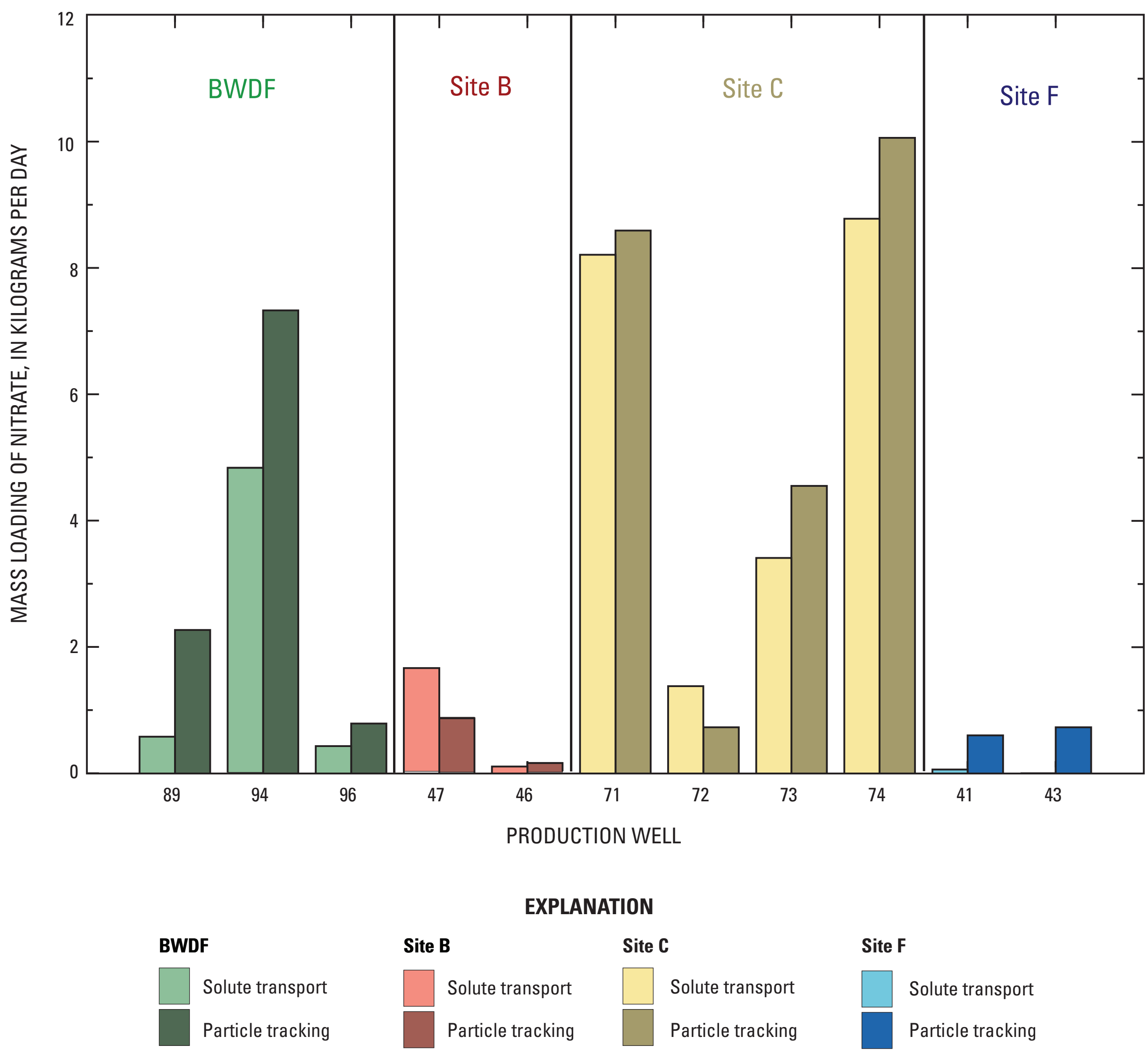


Figure 19. Nitrate loads estimated by flow-based particle tracking and solute transport modeling in wastewater intercepted by 11 production wells, Cape Cod, Massachusetts.

downgradient from proposed site C and would receive about half of its water from wastewater discharge from the site (fig. 9). Well 46, to the east of Site B (fig. 9), is not in the part of the aquifer that would receive significant amounts of wastewater, as indicated by advective flow paths; as a result, the mass-loading rate is small. Mass-loading rates estimated by the solute-transport model ranged from 8.8 kg/d at well 74 to 0.1 kg/d at wells 41, 43, and 46 (fig. 19). Advective flow paths indicate that wells 41 and 43, to the east of site F (fig. 9), are in a part of the aquifer that would receive only small amounts of wastewater.

Mass-loading estimates calculated by the two methods generally were in agreement (fig. 19). Estimates by particle-tracking methods generally exceeded estimates by the solute-transport method; this was the case for 9 of the 11 wells. This likely was the result of the complete conservation of mass along flow lines inherent in the particle-tracking method. The numerical dispersion created by the solute-transport model would spread mass spatially and thus decrease concentrations in some areas; particle tracking does not account for dispersion and the resulting local change in mass. The results calculated by the two methods agreed best for the wells downgradient of site C. The median traveltimes for wastewater intercepted by these wells were small (about 5 years), and it is likely that, given the short transport time, little spreading of mass occurred prior to discharge into the wells. Thus, the assumption of complete conservation of mass along flow lines was not as important for the simulation of wastewater transport to these wells. The largest discrepancy was for well 94 where mass-loading estimates by particle-tracking and solute-transport methods were 7.3 and 4.8 kg/d, respectively. The well is directly downgradient from the BWDF (fig. 9), and the mass-loading rate into the well was represented by a larger count of particles. In this case, the discrepancy likely was a result of the presence of Simmons Pond (code 58) just upgradient from the well (fig. 9). In the particle-tracking simulation, particles flowed through the pond and into the well with no loss of mass along flow lines represented by the movement of particles. In the solute-transport simulation, mass decreased locally within the pond owing to enhanced dilution and dispersion. These processes resulted in lower concentrations and, therefore, lower mass-loading estimates downgradient at the well.

For wells 72 and 47, mass-loading estimates by the solute-transport model were higher. Well 72 is downgradient of site C in close proximity to three other wells. Because the well is screened 20 ft directly below well 73, the particle-tracking method may not represent advective flow paths with sufficient resolution to produce reliable estimates for each well. If the two wells are combined, mass-loading estimates by the particle-tracking and solute-transport methods are similar: 5.3 and 4.8 kg/d, respectively. Well 47 is east of site B in an area that would receive a small portion of the wastewater (fig. 9); the mass-loading rate to this well is represented by a small count of particles. In these cases, particle-tracking methods based on particle densities that are practicable may not represent advective flow paths with enough resolution, which would be particularly important in areas of strongly diverging flow paths, such as near sites B and F (fig. 9). As a result, particle-tracking methods may have limitations for wells with small mass-loading rates in strongly diverging flow fields given particle densities that are practicable.

Mass-loading rates estimated from mass-weighted particles represent a steady-state condition in which concentrations have reached a maximum and do not change further over time. In some cases, it is useful to determine the amount of time before wastewater disposal affects a well or ecological receptor. This determination is done by quantifying how concentrations change with time prior to reaching a steady-state condition. Solute-transport simulations yield nitrate concentrations for each time step directly from model output. At wells 71 and 74, concentrations began to increase within about 2 years of transport and reached a final steady state after about 13 and 24 years, respectively (figs. 20A and 20B). Because wells 71 and 74 are about 2,100 ft directly downgradient from site C, they would receive a large amount of wastewater, and as a result, concentrations would reach a steady-state condition quickly. Well 94 is at a greater distance, about 7,800 ft, downgradient from the BWDF (fig. 9). Nitrate concentrations in wastewater intercepted by this well did not increase until after about 20 years of simulated transport from the BWDF and then increased gradually to a near-steady-state condition after about 100 years (fig. 20C). These results are consistent with the location of the well downgradient from the disposal site and near the edge of the simulated nitrate plume. At well 47 to the east of site B, concentrations began to increase after about 15 years and were still increasing steadily after 100 years of transport (fig. 20). These results also were consistent with traveltimes estimated by the flow model (fig. 11). The median traveltime for wastewater arriving at wells 71 and 74 is less than 10 years; thus, concentrations would increase to a steady-state condition more quickly at these wells than concentrations in wastewater intercepted by wells 94 and 47, which capture flow from deeper in the aquifer and have median traveltimes of 45 and 90 years, respectively (fig. 11).

Particle-tracking methods also can estimate time-varying concentrations by accounting for the traveltimes of individual mass-weighted particles discharging into each receptor; these traveltimes and time-specific particle counts can be converted to concentrations by dividing the calculated mass-loading rate by the water flux through the well. Because the process involves several postmodeling analyses, it is not as straightforward as the direct simulation of concentrations by the solute-transport method. Time-varying concentrations simulated by the particle-tracking method for wells 71 and 74 were similar to the concentrations simulated by the solute-transport model (figs. 20A and 20B). For well 94, however, the steady-state concentration estimated by the two methods differed by about 2.5 mg/L owing to the assumption inherent in the particle-tracking method that no loss of mass occurs along flow lines. This assumption probably affected the results

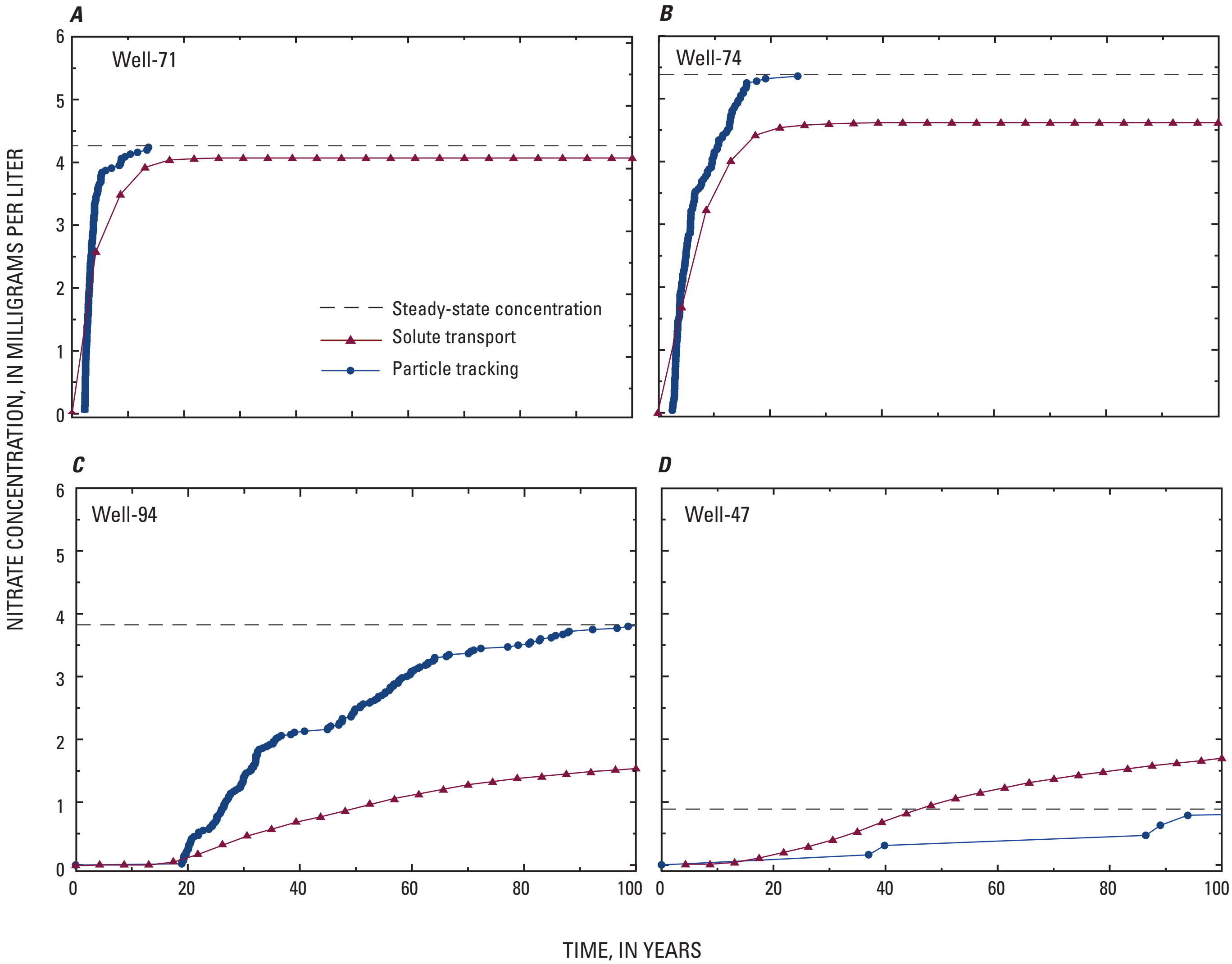


Figure 20. Time-varying and steady-state nitrate concentrations in wastewater intercepted by four production wells, calculated by solute-transport and particle-tracking methods, Cape Cod, Massachusetts.

for well 94 because the well is far from the source area and downgradient of a flow-through pond; both of these factors likely caused a local decrease in mass due to enhanced dispersion. For well 47, the two methods gave similar results, but the solute-transport method predicted larger concentrations than the particle-tracking method. In this case, estimates of time-varying concentrations were based on a small number of particles; the results suggest that the particle-tracking method does not sufficiently define the advective-flow patterns to the well given particle densities that are practicable. Although the particle-tracking method does yield similar time-varying concentrations in some cases, the additional effort and possibly invalid assumptions involved in the approach make the choice of solute-transport methods preferable.

The effect of dispersion on mass-loading rates and how those effects differ spatially are shown conceptually in figures 18A and B, in which a flow field is represented as discrete flow lines. Figure 18A illustrates a hypothetical particle-tracking scenario in which mass-loading rates were estimated at two wells downgradient from a continuous source of nitrate. Particles within the source area are weighted with 8 to 10 kg/d of nitrate, and particles from outside the source area are not weighted with any mass. An accounting of particles at the two wells shows that well 1 downgradient from the source receives a total of 26 kg/d of mass, whereas well 2 to the right of the source area receives no mass. If transverse dispersion allows for mass to spread between adjacent flow lines, mass decreases along the center flow line and increases along flow lines near the edges. The resulting mass-loading rates at wells 1 and 2 are 20 and 3 kg/d, respectively (fig. 18B). Although mass is shifted within the system, the total mass is conserved (26 kg/d). This simplified example illustrates how dispersion can result in different distributions of estimated concentrations and mass within an aquifer.

Effects of Dispersion

Advection, which refers to the movement of a solute with ground-water flow, is the dominant component of transport in most sandy coastal aquifers, including the Cape Cod aquifer system, where recharge rates and flow velocities are high; dispersion, however, is an additional component of transport that can affect the movement of nitrate through these aquifers. Dispersion refers to the differential movement of a solute through the aquifer as a result of heterogeneities within the aquifer matrix and, like heterogeneity, is scale dependent and is expressed in units of length. In a perfectly homogenous system, dispersivity is near zero; as the degree of heterogeneity increases, dispersivity increases. For one-dimensional transport through a homogenous sediment column, outflowing concentrations would increase to the initial inflowing concentration instantaneously at a time determined by the water velocity. If the sediments are heterogeneous, dispersion would cause outflowing concentrations to increase gradually to the inflowing concentration; the rate of the increase is a function of the dispersivity of the sediments. Longitudinal dispersion refers to a spreading of mass in the direction of ground-water flow; transverse and vertical dispersion refer to a spreading of mass orthogonal to the ground-water-flow direction. Longitudinal dispersion is typically much larger than the other two components.

Estimates of dispersivity increase with increasing transport distance. Horizontal transport distance, defined as the distance between the waste-disposal sites and discharge locations, ranged from about 1 to 3 mi for the nitrate plumes simulated in this analysis. This range of distances is referred to as plume-scale transport distances. On the basis of an empirical relationship derived from literature values (Luckner and Schestakow, 1991), longitudinal dispersivity ranges from 42 ft to 55 ft for plume-scale transport distances of 7,500 and 13,500 ft, respectively. Spitz and Moreno (1996) suggest a longitudinal dispersivity of about 90 ft for a plume-scale transport distance of about 13,000 ft. Longitudinal, transverse, and vertical dispersivities of 35, 3.5, and 0.35 ft best matched plume geometries at the MMR on western Cape Cod (U.S. Air Force Center for Environmental Excellence, 2000). The approximate numerical dispersivity generated by the solute-transport model—as estimated from equation (1)—is about 55 ft. This similarity suggests that numerical dispersion alone, a modeling artifact, may reasonably represent the general scale of longitudinal dispersion over plume-scale transport distances in the aquifer.

Dispersion can affect simulated concentrations and estimated mass-loading rates of nitrate at wells and ecological receptors that receive wastewater. For a mass of solute released continuously from a source over time, the effect of dispersion on concentrations at a given time and receptor would depend on the location of the receptor relative to the source, including both the downgradient distance from the source and the location of the receptor relative to the center of advective transport from the source. The effects of dispersion also would differ if, at a given time, steady-state concentrations had been reached at a given location. For a continuous source of solute, the effect of longitudinal dispersion generally would be to decrease the time of transport to a receptor and thereby increase the rate at which concentrations increase prior to reaching steady state. The effects of transverse and vertical dispersion would be to spread mass over a larger volume of aquifer and thus to decrease steady-state concentrations at receptors directly downgradient from source areas (near the centers of nitrate plumes) and increase steady-state concentrations in areas near the edges of the plumes.

The effect of dispersion on time-varying concentrations in four wells is shown in figure 21. Longitudinal dispersivities of 20, 50, 100, 200, and 500 ft were simulated; these values represent mechanical dispersion in addition to numerical dispersion (about 50 ft). In all cases, transverse and vertical dispersivities were specified as being one and two orders of magnitude lower, respectively, than longitudinal dispersivity. In general, the results show that the simulated dispersivity does not appreciably affect the simulated concentration. It should be noted that a dispersivity of zero represents numerical dispersion only. At wells 71 and 74, concentrations during 100 years of transport did not change substantially with increasing dispersivity (figs 21A and B). For dispersivities between 0 and 200 ft, the steady-state concentrations in wells 71 and 74 differed only by about 0.04 and 0.08 mg/L, respectively. For a dispersivity of 500 ft, steady-state concentrations were about 0.2 and 0.3 mg/L lower than concentrations simulated for the other dispersivities in wells 71 and 74, respectively. In these cases, sufficient time and transport distance may not have elapsed to allow dispersion to appreciably affect concentrations simulated at the wells. For well 94, nitrate concentrations for dispersivities between 0 and 100 ft differed by about 0.46 mg/L after 100 years of transport, a difference of about 33 percent (fig. 21C). Time-varying concentrations in well 47 increased with increasing dispersivity for about the first 40 years of transport. For transport times of about 50 to 100 years, concentrations for dispersivities between 0 and 200 ft were similar. For dispersivities of less than 200 ft, concentrations in wells 94 and 47 were still increasing after 100 years of transport.

For the largest dispersivity simulated (500 ft), steady-state concentrations in wastewater in wells 71 and 74 were about 0.2 and 0.3 mg/L lower, respectively, than concentrations for dispersivities of less than 200 ft. The lower steady-state concentrations likely were a result of an enhanced spreading of mass at the larger transverse and vertical dispersivities (50 and 5 ft). For wells 94 and 47, a steady-state concentration was reached when a dispersivity of 500 ft was simulated (fig. 21C and D). For well 94, concentrations for longitudinal, transverse, and vertical dispersivities of 500, 50, and 5 ft, respectively, were similar to those for dispersivities of 200, 20, and 2 ft for about the first 50 years of transport. Between transport times of 50 and 100 years, concentrations for a longitudinal dispersivity of 500 ft neared a steady-state

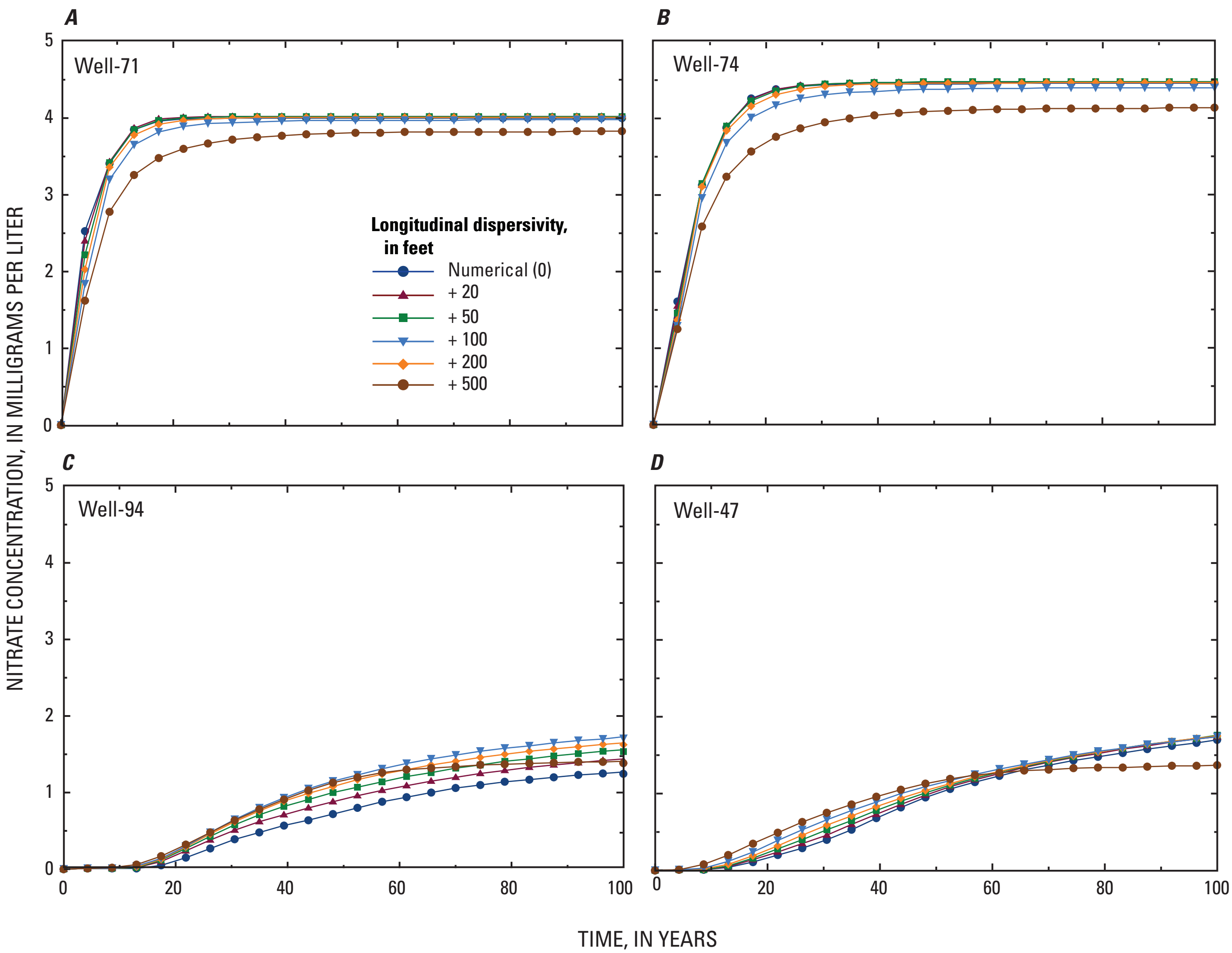


Figure 21. Time-varying nitrate concentrations in wastewater intercepted by four production wells under simulated dispersivities of 0, 20, 50, 100, 200, and 500 ft., Cape Cod, Massachusetts.

condition, whereas concentrations for the lower dispersivities continued to increase (fig. 21C). This effect is illustrated more clearly for well 47. Concentrations for a longitudinal dispersivity of 500 ft were larger than concentrations for lower dispersivities for the first 50 years of transport. After 50 years of transport, concentrations assuming a dispersivity of 500 ft reached steady state, while concentrations for lower dispersivities were still increasing; the concentration assuming a dispersivity of 500 ft was about 0.4 mg/L lower than concentrations assuming lower dispersivities after 100 years of transport (fig. 21D). Simulation results indicated that the effects of longitudinal dispersivity caused early-time concentrations to increase with increasing dispersivity; however, at very high values of dispersivity, concentrations reached steady state. By that time in the transport process, the enhanced transverse and vertical spreading of mass may have depleted the mass transported along the central path directly downgradient from the source and thus lowered the concentration below the steady-state value reached at the lower dispersivities. The results indicate that the effects of dispersion on nitrate concentrations are enhanced in wells (1) where sufficient transport times have elapsed, (2) near the edges of a plume, or (3) downgradient from flow-through ponds.

Limitations of Analysis

Several limitations apply to estimates of mass loading by particle-tracking (flow-based) and solute-transport methods. The analyses presented in this report were done with deterministic ground-water-flow models in which hydraulic conductivities, recharge, and boundary leakances were determined by trial-and-error calibration to observations of heads, streamflows, and advective flow, as indicated by contaminant plumes. The model is assumed to represent the flow system adequately because (1) the distribution of hydraulic conductivity is based on a model that incorporates an understanding of the depositional history of the glacial sediments in the region, (2) values of hydraulic conductivity, boundary leakance, and recharge are reasonable values consistent with previous investigations, and (3) the simulated results reasonably match observations of heads, streamflows, and advective transport (Walter and Whealan, 2004). The flow model produces a simulated flow field that is nonunique in the sense that different model configurations could produce similar head distributions. Particle-tracking results would be expected to differ as aquifer properties and recharge were changed; however, if the resulting head distribution was reasonably calibrated to observed conditions, particle tracks likely would be similar for different combinations of aquifer properties and recharge. In addition, flow lines in the aquifer are strongly affected by hydraulic boundaries, such as ponds, streams, and coastal waters; because boundary geometries are static, however, simulated particle tracks and estimates of advective flow likely would be similar for a range of model inputs.

Although advective-flow paths are likely to be reasonably robust for a reasonable range of model inputs, simulated traveltimes can be affected by several factors. If hydraulic conductivity and recharge rates were increased in a correlated manner relative to their values in a baseline simulation to produce a similar calibrated head distribution, ground-water fluxes would increase and traveltimes would decrease. Another factor affecting traveltimes is aquifer porosity. The uniform value of 0.35 used in these analyses was based on field-scale tracer tests on western Cape Cod (Garabedian and others, 1985); the porosity of glacial sand and gravel likely ranges between 0.25 to 0.4 (Freeze and Cherry, 1979). The linear velocity of ground water is inversely proportional to porosity; decreasing the porosity would increase velocity and decrease simulated traveltimes.

Additional considerations are necessary when particle-tracking methods are used to estimate concentrations and mass; uncertainties in simulated results can arise from the representation of volumes of water or mass by too small a number of discrete particles. If the number of particles is small, the particles may not represent the flow field at a resolution adequate to estimate traveltimes to a receptor. Even if adding particles does change the estimated mean traveltime to a receptor, the means should generally be similar because the position of the receptor in the flow system would have the most control on the age of water intercepted by the receptor. It is advisable to choose the largest number of particles that is practicable with the use of particle-tracking methods.

Another assumption in these analyses is that of steady-state flow based on constant recharge. Although actual recharge varies both monthly and from year to year, hydraulic-gradient directions in the aquifer generally do not vary over time owing to the small time scale of recharge variability (months) compared to the much larger time scale of advective transport (multiple years). Walter and Masterson (2003) compared particle tracks on northwest Cape Cod through both steady-state and transient flow fields and found that the particle tracks were nearly identical. The results of this comparison indicated that it was reasonable to simulate steady-state advective transport even though recharge was known to vary through time. Ground-water withdrawals also change seasonally and over multiple years. These seasonal changes and longer-term trends also would be expected to alter hydraulic gradients; however, these changes likely would be limited to areas close to the wells.

The solute-transport methods used in these analyses also rely on intercell flows produced by the ground-water-flow model. As a result, estimated concentrations, mass-loading rates, and ground-water velocities are susceptible to the same uncertainties as the same quantities calculated from particle tracks produced by the flow model. Additional uncertainties include (1) the effects of dispersion on the transport of mass, (2) the possibility of processes affecting nitrate transport that are not accounted for in the solute-transport model, and (3) estimated nitrogen inputs, which will depend on nitrogen-treatment levels at specific wastewater-treatment plants in the proposed source areas.

Summary

Cape Cod, a rapidly urbanizing region in southeastern Massachusetts, faces several water-resource issues. The unconsolidated sandy aquifer underlying the region is the sole source of potable water to local communities and is the primary receptor of the region’s wastewater. Ground-water discharge from the aquifer supports ecosystems in freshwater ponds and streams and in coastal waters.The disposal of municipal wastewater from the region into the aquifer has created the potential for adverse effects of wastewater disposal on production wells and ecological receptors. Nitrogen, in the form of nitrate, is a wastewater constituent of particular concern. Marine and estuarine ecosystems are sensitive to inputs of wastewater-derived nitrogen because excess nitrogen in marine waters can cause algal blooms, degradation of water clarity and, ultimately, loss of critical habitats, such as sea-grass beds. In areas where wastewater-disposal sites are near production wells, the discharge of nitrate into the wells can be harmful to human health.

Many of the communities in the region are in the process of developing wastewater-management strategies that will allow for the sustainable use of the aquifer as both a source of potable water and a receptacle of wastewater and will maintain healthy aquatic ecosystems. To support these efforts, the Massachusetts Department of Environmental Protection is currently (2006) developing estimates of the total maximum daily loads of nitrogen into individual estuaries and coastal waters. In 2003, the U.S. Geological Survey began a cooperative investigation with Barnstable County and the Cape Cod Commission to describe the effects of wastewater disposal in the context of the regional-flow system. An important component of the investigation was the use of numerical models to simulate the transport of nitrate from areas of centralized wastewater disposal to wells and natural receptors, such as ponds, streams, and coastal waters. Possible wastewater-disposal scenarios, including sewering and centralized disposal, from individual towns were simulated using regional flow models of central and western Cape Cod. The dual purpose of this modeling support was to (1) assist communities in the initial evaluation of possible wastewater-disposal scenarios and (2) develop a regionally applicable understanding of wastewater disposal in different hydrologic settings.

Numerical models of ground-water flow can be of use in the evaluation of wastewater-disposal scenarios by simulating both the hydraulic effects of wastewater disposal on water levels and streamflow and the advective transport of wastewater-derived nitrate through the aquifer. Of particular concern to local and regional wastewater planners are the potential mass-loading rates of nitrate into wells and ecological receptors—ponds, streams, and coastal waters. Particle-tracking methods use model-calculated cell-by-cell flows to track particles of water through the subsurface. These methods can quantify wastewater discharge to receptors by an accounting of volume-weighted particles discharging to the receptors. These estimated discharge volumes can be converted to mass-loading rates by multiplying these volumes by an assumed nitrate concentration in the treated wastewater.

The utility of flow models in evaluating wastewater-disposal scenarios is illustrated by a hypothetical scenario in eastern Barnstable. This area, which is the most urbanized part of Cape Cod, has an operating wastewater-disposal facility—the Barnstable Wastewater Disposal Facility (BWDF)—and four potential new wastewater-disposal sites (sites B, C, D, and F). The area also includes several production wells and sensitive ecological receptors. The hypothetical scenario includes the sewering of eastern Barnstable and the disposal of 1.7 Mgal/d of wastewater at the BWDF and 1.22 Mgal/d at sites B, C, and F. Although this hypothetical scenario was not analyzed as part of modeling support to the community, the volumes and locations of wastewater disposal are consistent with scenarios done as part of that modeling support, and the results illustrate the types of analyses that would be useful in evaluating actual scenarios.

The ground-water-flow system is characterized by an east-west trending regional ground-water divide; ground-water flow and advective transport is away from the divide toward downgradient discharge locations. Near the divide, vertical flow is enhanced and flow lines are deep in the system. Farther from the divide, ground-water flow is more horizontal and generally shallower. Traveltimes in the aquifer, referring to the elapsed time between recharge at the water table and discharge from the system, range from essentially instantaneous next to discharge locations to more than 100 years near regional ground-water divides.

According to the simulation results, wastewater would be transported by advection away from the current and proposed disposal sites toward discharge locations, and several sensitive receptors in this part of Cape Cod would receive some wastewater: 10 coastal water bodies, 1 stream, 7 ponds, and 11 production wells. The wastewater-disposal sites are in different hydrologic settings within the ground-water-flow system. The BWDF and sites B and F are at or near the regional ground-water divide, and site C is farther from the divide. Wastewater originating from source areas near the ground-water divide would be transported over a wider arc of flow directions and deeper into the aquifer than wastewater emanating from the source area farther from the divide. These advective-transport patterns indicate that wastewater disposed near the divide would likely contaminate a larger volume of the aquifer. Of the 5.36 Mgal/d of discharged wastewater, about 70 percent discharged into coastal waters, 12 percent into streams, and 18 percent into production wells; about 13 percent of the wastewater flows through freshwater ponds prior to discharge from the system. The volumes of wastewater that discharged into individual coastal water bodies ranged from about 0.01 to nearly 1 Mgal/d. Discharge volumes into both ponds and wells ranged from about 0.01 to more than 0.2 Mgal/d. About 0.65 Mgal/d of wastewater discharged into Phlashes Brook, the only stream that received wastewater. Wells, ponds, streams, and coastal waters differ in their capacity to assimilate wastewater-derived nitrate, and therefore calculating wastewater-discharge volumes to individual receptors from each individual source area is useful in evaluating the potential for adverse effects of wastewater disposal on the receptors. Because freshwater ecosystems are not as sensitive to nitrogen as are marine ecosystems, ponds and streams are receptors of less concern than are coastal receptors. Also, open coastal waters are of less concern than inland estuaries because open coastal waters have more tidal flushing and circulation. Wells are receptors of concern because of the potential for adverse effects on human health.

Nitrate may be attenuated over long periods of subsurface transport primarily by the process of denitrification, and traveltimes also are important in evaluating the potential for adverse effects on receptors that receive wastewater-derived nitrate. Traveltimes in the aquifer range from essentially instantaneous near discharge boundaries to hundreds of years near regional ground-water divides. Traveltimes of wastewater discharged from the source areas differ according to hydrologic setting. For the hypothetical scenario, wastewater discharged from source areas near ground-water divides were as large as several hundred years, whereas traveltimes for wastewater discharged from the source area farther from regional ground-water divides—where horizontal ground-water velocities are higher—were as small as 3 years. If pumping is removed from the scenario and all wastewater is allowed to discharge to natural receptors, median traveltimes of wastewater from the BWDF and sites B and F, near the regional divide, are 66, 31, and 48 years, respectively. The median traveltime of wastewater from site C farther, from the divide, to natural receptors is about 10 years. As an example of the importance of traveltimes in the analysis of regional wastewater disposal, both Stewarts Creek and Nantucket Sound receive a substantial amount of wastewater (more than 0.5 Mgal/d) from the BWDF; however, the median traveltime of wastewater to Stewarts Creek is about 20 years, whereas the median traveltime to Nantucket Sound is about 130 years. As a result of the longer traveltime to the Sound, the potential for nitrate attenuation during transport to this receptor is greater.

A subregional solute-transport model of eastern Barnstable was developed and used to simulate nitrate transport resulting from the hypothetical wastewater-disposal scenario; results from the solute-transport simulations were compared with those from flow-based particle-tracking analyses to evaluate the utility of using particle-tracking and solute-transport methods for calculating mass-loading rates to wells and ecological receptors. Flow-based particle tracking can be used to simulate advective-flow directions and traveltimes, and the analyses require no further model development beyond the ground-water-flow model; however, the method does not yield substantial quantitative information regarding the distribution of mass in the aquifer. The approach represents movement of a mass of nitrate with discrete mass-weighted particles and, thus, is based on the assumption, likely to be invalid in most systems, that dispersion does not occur and mass is conserved along flow lines. Also, the use of flow-based particle tracking to estimate mass-loading rates and time-varying concentrations at receptors requires a number of additional postmodeling analyses. In contrast, solute-transport methods require additional model development but explicitly simulate the transport of nitrate through the subsurface and better estimate mass-loading rates and time-varying concentrations at receptors with no postmodeling analyses. Also, solute-transport models can simulate the effects of other components of transport, such as dispersion, on nitrate transport. Numerical dispersion, which is a function of model discretization, should be considered if solute-transport models are used to simulate the movement of nitrate. In advective-dominated aquifer systems such as that on Cape Cod, models with a sufficiently small discretization can be chosen to minimize the potential for numerical dispersion to be greater than actual dispersion in the aquifer.

Mass-loading rates to 11 production wells estimated from both particle-tracking and solute-transport methods generally were in good agreement. Mass-loading rates calculated by particle-tracking methods generally were higher than rates calculated by solute-transport methods owing to the assumption that no mass is lost during transport. Results for wells with high mass-loading rates and short traveltimes agreed best. More solute mass is conserved if nitrate concentrations reach steady state quickly with little time for mass to be lost during transport. In these cases, the invalid assumption that mass is conserved along flow lines is less important. In some cases, the two methods yielded substantially different results. Well 94 is downgradient from a pond, and the enhanced dispersion of nitrate by the pond as represented by the solute-transport model causes concentrations and, therefore, mass-loading rates in wastewater intercepted by the well to be lower than those estimated by the particle-tracking method. The mass-loading rate calculated by the solute-transport model exceeded the particle-tracking estimate for wastewater received in well 47. The well is east of source area B in a diverging flow field near the ground-water divide. It is likely that the particle-tracking method does not represent advective flow, which was mostly to the north, with sufficient resolution to simulate nitrate transport in the diverging flow field and that the required particle density may be higher than that which is practicable. Flow-based particle tracking also can estimate time-varying concentrations by a cumulative accounting of particles arriving after different traveltimes at a well. For wells with large mass-loading rates and short traveltimes, the two methods yielded similar results. At wells 94 and 47, time-varying concentrations estimated by particle tracking were substantially different from those simulated by solute-transport modeling; however, the particle-tracking method did accurately represent the elapsed time before steady-state concentrations were reached.

Transport in the Cape Cod aquifer is dominated by advection, which refers to the movement of a conservative solute, such as nitrate, with ground-water flow. Another component of transport, however, is dispersion, which refers to the effect of aquifer heterogeneity on the differential transport of a solute through the aquifer and the resultant spread of mass in the aquifer; both aquifer heterogeneity and dispersion are scale dependent. Numerical dispersion is an artifact of the solute-transport model and is a function of ground-water velocities and the spatial and temporal discretization of the model. The subregional model of Barnstable has an approximate numerical dispersivity of about 50 ft; because this value is consistent with a range of reasonable plume-scale dispersivities for the Cape Cod aquifer, the model may reasonably represent dispersive transport with no additional simulated dispersivity. The sensitivity of simulated concentrations to dispersivity was evaluated for a range of values from 20 to 500 ft. In general, time-varying concentrations at wells were not greatly affected by dispersion. Concentrations in wastewater received in wells 71 and 74 reached a similar steady-state value for dispersivities between 20 and 500 ft owing to the close proximity of the wells to the site C source area and the small traveltimes of nitrate from the source area to the wells. For wells 47 and 94 farther and at greater transport distances from source areas, the effect of dispersion was more pronounced owing to the fact that the cumulative effect of dispersion was greater over the longer transport times and distances.

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