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The Simulated Effects of Wastewater-Management Actions on the Hydrologic System and Nitrogen-Loading Rates to Wells and Ecological Receptors, Popponesset Bay Watershed, Cape Cod, Massachusetts

Scientific Investigations Report 2013–5060

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

Front cover photograph. Northward view of the Eel River in East Falmouth, Cape Cod, Massachusetts. The Eel River, which is typical of the estuaries along the southern shore of Cape Cod, is a long, narrow saltwater embayment, referred to locally as a coastal pond, that occupies a drowned glacial valley in the Mashpee outwash plain. Photography by Denis LeBlanc, June 20, 2013.

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U.S. Department of the Interior
SALLY JEWELL, Secretary

U.S. Geological Survey
Suzette M. Kimball, Acting Director

U.S. Geological Survey, Reston, Virginia: 2013

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Suggested citation:
Walter, D.A., 2013, The simulated effects of wastewater-management actions on the hydrologic system and nitrogen-loading rates to wells and ecological receptors, Popponesset Bay Watershed, Cape Cod, Massachusetts: U.S. Geological Survey Scientific Investigations Report 2013–5060, 62 p., at http://pubs.usgs.gov/sir/2013/5060/.

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Conversion Factors, Datum, and Abbreviations

Inch/Pound to SI

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	4,047	square meter (m^2)
acre	0.4047	hectare (ha)
square mile (mi^2)	259.0	hectare (ha)
	Flow rate	
foot per day (ft/d)	0.3048	meter per day (m/d)
cubic foot per day (ft^3/d)	0.02832	cubic meter per day (m^3/d)
inch per year (in/yr)	25.4	millimeter per year (mm/yr)
million gallons per day (Mgal/d)	0.04381	cubic meter per second (m^3/s)
	Application rate	
pounds per acre per year [(lb/acre)/yr]	1.121	kilograms per hectare per year [(kg/ha)/yr]

Vertical coordinate information is referenced to North American Vertical Datum of 1988 (NGVD 88).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

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

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

Loads are given in kilograms per day (kg/d) and kilograms per year (kg/yr).

List of Abbreviations

FDM	finite-difference method
GIS	geographic information system
MassDEP	Massachusetts Department of Environmental Protection
MEP	Massachusetts Estuaries Project
MMR	Massachusetts Military Reservation
MPA	Mashpee Planning Area
TMDL	Total Maximum Daily Loads
UCC	Upper Cape Cooperative
USGS	U.S. Geological Survey
WTF	wastewater-treatment facility

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The Simulated Effects of Wastewater-Management Actions on the Hydrologic System and Nitrogen-Loading Rates to Wells and Ecological Receptors, Popponesset Bay Watershed, Cape Cod, Massachusetts

By Donald A. Walter

Abstract

The discharge of excess nitrogen into Popponesset Bay, an estuarine system on western Cape Cod, has resulted in eutrophication and the loss of eel grass habitat within the estuaries. Septic-system return flow in residential areas within the watershed is the primary source of nitrogen. Total Maximum Daily Loads (TMDLs) for nitrogen have been assigned to the six estuaries that compose the system, and local communities are in the process of implementing the TMDLs by the partial sewering, treatment, and disposal of treated wastewater at wastewater-treatment facilities (WTFs). Loads of waste-derived nitrogen from both current (1997–2001) and future sources can be estimated implicitly from parcel-scale water-use data and recharge areas delineated by a groundwater-flow model. These loads are referred to as "instantaneous" loads because it is assumed that the nitrogen from surface sources is delivered to receptors instantaneously and that there is no traveltime through the aquifer. The use of a solute-transport model to explicitly simulate the transport of mass through the aquifer from sources to receptors can improve implementation of TMDLs by (1) accounting for traveltime through the aquifer, (2) avoiding limitations associated with the estimation of loads from static recharge areas, (3) accounting more accurately for the effect of surface waters on nitrogen loads, and (4) determining the response of waste-derived nitrogen loads to potential wastewater-management actions.

The load of nitrogen to Popponesset Bay on western Cape Cod, which was estimated by using current sources as input to a solute-transport model based on a steady-state flow model, is about 50 percent of the instantaneous load after about 7 years of transport (loads to estuary are equal to loads discharged from sources); this estimate is consistent with simulated advective traveltimes in the aquifer, which have a median of 5 years. Model-calculated loads originating from recharge areas reach 80 percent of the instantaneous load within 30 years; this result indicates that loads estimated from recharge areas likely are reasonable for estimating current instantaneous loads. However, recharge areas are assumed to remain static as stresses and hydrologic conditions change in response to wastewater-management actions.

Sewering of the Popponesset Bay watershed would not change hydraulic gradients and recharge areas to receptors substantially; however, disposal of wastewater from treatment facilities can change hydraulic gradients and recharge areas to nearby receptors, particularly if the facilities are near the boundary of the recharge area. In these cases, nitrogen loads implicitly estimated by using current recharge areas that do not accurately represent future hydraulic stresses can differ significantly from loads estimated with recharge areas that do represent those stresses. Nitrogen loads to two estuaries in the Popponesset Bay system estimated by using recharge areas delineated for future hydrologic conditions and nitrogen sources were about 3 and 9 times higher than loads estimated by using current recharge areas; for this reason, reliance on static recharge areas can present limitations for effective TMDL implementation by means of a hypothetical, but realistic, wastewater-management action. A solute-transport model explicitly represents nitrogen transport from surface sources and does not rely on the use of recharge areas; because changes in gradients resulting from wastewater-management actions are accounted for in transport simulations, they provide more reliable predictions of future nitrogen loads.

Explicitly representing the mass transport of nitrogen can better account for the mechanisms by which nitrogen enters the estuary and improve estimates of the attenuation of nitrogen concentrations in fresh surface waters. Water and associated nitrogen can enter an estuary as either direct groundwater discharge or as surface-water inflow. Two estuaries in the Popponesset Bay watershed receive surface-water inflows: Shoestring Bay receives water from the Santuit River, and the tidal reach of the Mashpee River receives water (and associated nitrogen) from the nontidal reach of the Mashpee River. Much of the water discharging into these streams passes through ponds prior to discharge. The additional attenuation of nitrogen in groundwater that has

passed through a pond and discharged into a stream prior to entering an estuary is about 3 kilograms per day.

Advective-transport times in the aquifer generally are small—median traveltimes are about 4.5 years—and nitrogen loads at receptors respond quickly to wastewater-management actions. The simulated decreases in nitrogen loads were 50 and 80 percent of the total decreases within 5 and 15 years, respectively, after full sewering of the watershed and within 3 and 10 years, for sequential phases of partial sewering and disposal at WTFs. The results show that solute-transport models can be used to assess the responses of nitrogen loads to wastewater-management actions, and that loads at ecological receptors (receiving waters—ponds, streams or coastal waters—that support ecosystems) will respond within a few years to those actions.

The responses vary for individual receptors as functions of hydrologic setting, traveltimes in the aquifer, and the unique set of nitrogen sources representing current and future wastewater-disposal actions within recharge areas. Changes in nitrogen loads from groundwater discharge to individual estuaries range from a decrease of 90 percent to an increase of 80 percent following sequential phases of hypothetical but realistic wastewater-management actions. The ability to explicitly represent the transport of mass through the aquifer allows for the evaluation of complex responses that include the effects of surface waters, traveltimes, and complex changes in sources. Most of the simulated decreases in nitrogen loads to Shoestring Bay and the tidal portion of the Mashpee River, 79 and 69 percent, respectively, were caused by decreases in the nitrogen loads from surface-water inflow.

Introduction

Estuaries and inshore coastal waters adjacent to urbanized and developing areas are at risk for eutrophication arising from excess nutrients from onshore sources and the resultant degradation of water quality and clarity, harmful algal blooms, and loss of critical marine habitat. These inland marine waters often have limited exchange with open coastal waters and receive freshwater from groundwater discharge and surface-water inflows that introduce excess nutrients from anthropogenic sources into the estuary. The nutrient generally of most concern to marine ecosystems is nitrogen, which can be introduced into estuaries and inshore coastal waters from a number of onshore sources, including road and roof runoff, fertilizers, and wastewater. The economic and recreational importance of estuaries and inshore coastal waters, which provide habitat for commercial and game fish, makes eutrophication and habitat loss an increasing concern to local communities as well as the regulatory community.

Cape Cod is a rapidly developing area in the coastal plain of southeastern Massachusetts (fig. 1). The population of Cape Cod more than doubled between 1970 and 2010 with the densest development generally in areas near the shore (Cape Cod Commission, 2012). As a result, many estuaries on Cape Cod have been adversely affected by the discharge of nitrogen from sources in the associated watersheds. Eel-grass beds, an important habitat for fish and shellfish populations, have decreased substantially in most inshore waters since 1950 and have been absent from several estuarine systems on Cape Cod since the mid-1990s (University of Massachusetts-Dartmouth School of Marine Science and Technology, 2012a). In 2001, the Massachusetts Department of Environmental Protection (MassDEP) initiated the Massachusetts Estuaries Project (MEP) in coordination with the University of Massachusetts-Dartmouth School of Marine Science and Technology to assess the ecological health of the Commonwealth's coastal waters and to determine Total Maximum Daily Loads (TMDLs) for nitrogen that would establish conditions suitable for the growth of eel grass in individual estuarine systems.

The MEP approach, referred to as the "Linked Watershed-Embayment Management Modeling Approach," links watershed (groundwater-flow) models and embayment (hydrodynamic) models in conjunction with the collection of water-quality and ecological data to develop the TMDLs. Detailed discussions of the application of the approach to a number of estuarine systems on Cape Cod are presented in a series of reports published between 2003 and 2010 (University of Massachusetts-Dartmouth School of Marine Science and Technology, 2012). As part of the approach, recharge areas (watersheds) delineated by using a groundwater-flow model are mapped to detailed spatial data representing current (1997–2001) nitrogen sources and used to estimate loads from these sources to the estuary; these modeled loads are used as inputs in a hydrodynamic model of the estuary, which is calibrated to represent current water-quality conditions within the estuary (Howes and others, 2004). The ecological health of the estuary is evaluated, and the rate of nitrogen loading that the system can assimilate and still support the growth of eel grass is estimated. The same watershed-based method is used to estimate loads resulting from a planned wastewater-management action, generally consisting of collection (through sewering), treatment, and centralized disposal of wastewater. Finally, the same set of hydrodynamic and water-quality models is used to determine if the future nitrogen load would result in water-quality conditions that are suitable for the growth of eel grass, indicating compliance with the TMDL.

Popponesset Bay is an inland estuary on the south shore of western Cape Cod that has been adversely affected by development within the estuary's watershed (fig. 1). The estuary has a maximum depth of about 15 feet (ft); daily tidal fluctuations in the estuary are between 1 and 1.5 ft (Howes and others, 2004). In 1951, there were about 97 acres of eel-grass habitat within the estuarine system, which has a total area of about 700 acres; eel grass was absent from the system in surveys conducted in 1995 and 2001 (Howes and others, 2004). As part of the MEP, TMDLs for nitrogen recently (2004) have been developed for the individual receptors that compose the Popponesset Bay system; a detailed discussion of

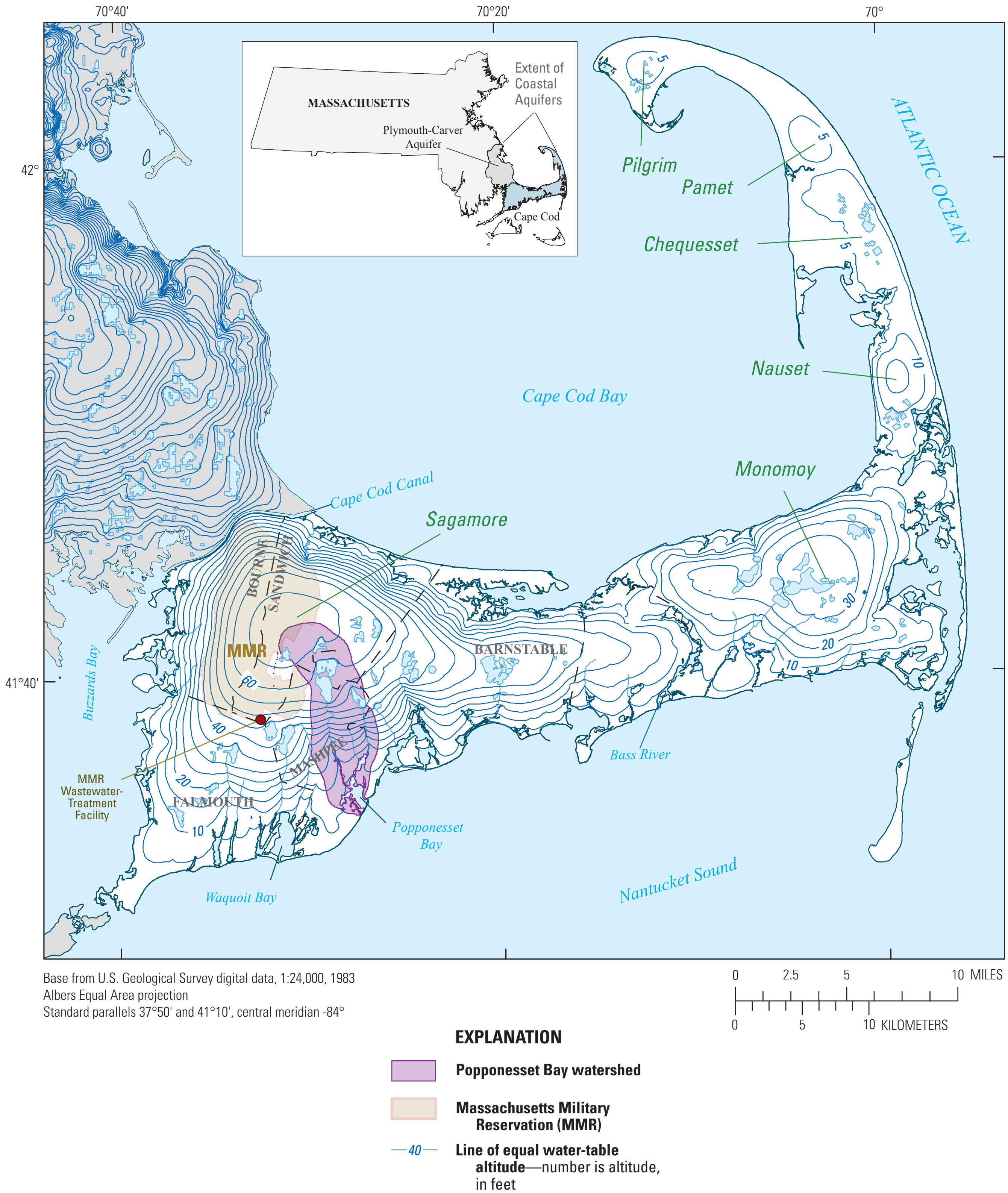


Figure 1. Extent of coastal aquifers in southeastern Massachusetts and hydrography of Cape Cod and location of the Popponesset Bay watershed, western Cape Cod, Massachusetts. Green labels identify flow lenses.

the application of the MEP approach to the Popponesset Bay system is presented in Howes and others (2004).

The response of nitrogen loads to the estuary is of interest to local communities that must implement the TMDLs. The Town of Mashpee, which had the largest population increase on Cape Cod between 2000 and 2003 and encompasses most of the Popponesset Bay watershed, is evaluating wastewater-management actions to reduce nitrogen loads into the Popponesset Bay system in compliance with the TMDLs. Terrestrial nitrogen loads estimated by using the MEP approach represent what are defined as "instantaneous" loads: the load estimated by mapping model-delineated watersheds to spatial data on current or future nitrogen sources is assumed to be the same as the load discharging to the estuary at that time, and, therefore, the approach does not explicitly account for the subsurface transport of nitrogen through the aquifer from surface sources to the ecological receptor—pond, stream, or estuary. Although the use of instantaneous loads may be suitable for estimating current nitrogen loads and the development of TMDLs, the approach makes it difficult for decisionmakers to estimate the time-varying response of nitrogen-loading rates to the complex, competing effects of sewering and centralized disposal resulting from a wastewater-management action.

The U.S. Geological Survey (USGS), in cooperation with the MassDEP, has evaluated calculations by solute-transport models of the responses of both the hydrologic system and loads of waste-derived nitrogen to wells and ecological receptors within the Popponesset Bay system resulting from a set of hypothetical wastewater-management actions similar to those being considered by local communities. The purposes of the analysis are to (1) develop methods for incorporating detailed spatial data on nitrogen sources into a modeling analysis, (2) determine how the hypothetical wastewater-management actions could affect water levels and streamflows, (3) evaluate how quickly the rate of nitrogen discharge into receptors could change in response to wastewater-management actions, and (4) demonstrate the applicability of these methods to other estuarine systems in southeastern Massachusetts for enhancing implementation of the TMDLs developed by the MEP. The analysis has the ancillary benefit of assisting the Town of Mashpee in better understanding the dynamics of their coastal-aquifer system and protecting local marine ecosystems. The methods presented in this report also will have applicability to estuarine systems in similar hydrologic environments in southeastern Massachusetts and elsewhere.

Purpose and Scope

This report discusses the effects of a set of wastewater-management actions—consisting of the collection through sewering, treatment, and centralized disposal of wastewater—on the hydrologic system and ecological receptors within the Popponesset Bay watershed. Two sets of effects are discussed: (1) the effects on water levels, streamflows, and recharge areas and (2) the effects on time-varying rates of nitrogen loading to supply wells and ecological receptors. The report also documents the numerical models and methods developed to support the analysis and provide information on how similar approaches could be applied to other estuarine systems.

The effects of changing anthropogenic stresses—sewering, disposal at wastewater-treatment facilities (WTFs), and pumping—on water levels and streamflows are presented graphically and discussed. Maps show the effects of such stresses on recharge areas to selected ecological receptors, and the report includes discussions of how changes in recharge areas can affect estimates of nitrogen-loading rates from spatial data on nitrogen sources and implications for the use of static recharge areas in TMDL implementation.

The methods used to incorporate detailed nitrogen-source data into a solute-transport modeling analysis are documented in the report, including comparisons of rasterized data used for model inputs and original vector data to quantify the accuracy of the representation of detailed nitrogen sources in a solute-transport model. The amount of nitrogen loading to wells and ecological receptors—ponds, streams, and estuaries—simulated by using a solute-transport model is presented by using maps and graphs. The analyses in this report address loads of nitrogen only from wastewater sources because (1) wastewater is the largest source of nitrogen in coastal watersheds on Cape Cod, and (2) management of wastewater is most effective for nitrogen-load reduction. The report includes comparisons between loads simulated by using a solute-transport model and those estimated by using model-delineated recharge areas mapped to spatial data on nitrogen sources. The report presents and discusses the responses of nitrogen-loading rates at wells and ecological receptors to different wastewater-management actions, including (1) no action, (2) complete removal of nitrogen through sewering and disposal outside of the area, and (3) a two-phase combination of sewering and centralized disposal of treated wastewater within and near the watershed. Finally, the applicability of the methods to other estuarine systems is discussed, as are limitations of the analysis, to allow proper application of the method.

Hydrogeology

Cape Cod is underlain by unconsolidated sediments that generally are highly permeable and nearly 1,000 ft thick in some areas. The region receives substantial rainfall, and the unconsolidated sediments compose the sole source of potable water for the region's communities. The Cape Cod aquifer system consists of six separate flow lenses: Sagamore, Monomoy, Nauset, Chequesset, Pamet, and Pilgrim (fig. 1). Each flow lens represents a distinct aquifer system that is hydraulically separate from adjacent flow lenses. Popponesset Bay and its watershed are in the south-central part of the Sagamore flow lens—the largest and westernmost of the flow lenses (fig. 1). The geologic history and hydrology of Cape Cod have been documented in numerous publications, including LeBlanc and others (1986), Masterson and others (1997a), Oldale (1992), and Uchupi and others (1996).

Geologic Setting

The glacial sediments, which consist of gravel, sand, silt, and clay and are underlain by crystalline bedrock, were deposited 15,000–16,000 years ago within and near the margins of retreating continental ice sheets (Oldale and Barlow, 1986; Uchupi and others, 1996). The altitude of the bedrock surface underlying the glacial sediments ranges from about 50 ft below North American Vertical Datum of 1988 (NAVD 88) near the Cape Cod Canal to more than 900 ft below NAVD 88 beneath the outer part of Cape Cod; the bedrock surface is as deep as 500 ft below NAVD 88 along the shore of Nantucket Sound and beneath Popponesset Bay (Fairchild and others, 2013). The surficial geology of Cape Cod is characterized by broad, gently sloping outwash plains and hummocky terrain associated with glacial moraines and ice-contact deposits. Outwash sediments, which compose most of the glacial sediments underlying Cape Cod, were deposited in fluvial and lacustrine depositional environments associated with proglacial lake deltas analogous to those seen in present-day fluvial deltas (Oldale, 1992).

Outwash sediments are broadly divided into three depositional units: coarse-grained sand and gravel deposited in meltwater streams (topset beds), fine to medium sands deposited in nearshore lacustrine environments (foreset beds), and fine sand and silt deposited in offshore lacustrine environments (bottomset beds) (Masterson and others, 1997a). Geologic contacts generally are gradational, and heterogeneities are common, including lenses of silt and clay within the shallow, generally coarse-grained sediments—possibly representing deposition locally within the fluvial outwash plain—and lenses of coarse sediments within deep, generally fine-grained sediments, possibly representing deposition by gravity flows. Moraines were deposited in low-energy environments and generally are finer grained and less sorted than are outwash sediments; ice-contact deposits were deposited within high-energy fluvial environments beneath and inside the ice sheets and generally are coarser grained than are the outwash deposits. The Popponesset Bay watershed is within the Mashpee outwash plain on western Cape Cod (fig. 2A); the outwash plain is the largest on Cape Cod and is bounded on the north by the Sandwich moraine and on the west by the Buzzards Bay moraine (fig. 2A). Outwash sediments underlying the Mashpee outwash plain generally become finer grained with depth and to the south with increasing distance from the sediment source near the present-day Cape Cod Canal. The western part of Popponesset Bay is underlain by ice-contact deposits (fig. 2A).

The water-transmitting properties of the glacial aquifer sediments are determined primarily by grain size and degree of sorting. The trends in hydraulic conductivity of outwash sediments result from the trends in grain size; the hydraulic conductivity of sediments generally decreases with depth and with increasing distance from sediment sources, or generally southward (Masterson and others, 1997b). Previous investigations have identified general relations between sediment grain size and hydraulic conductivity as determined from aquifer tests (Masterson and others, 1997b; Masterson and Barlow, 1994). Medium to coarse sand and gravel deposits have hydraulic conductivities that range from about 200 to 350 feet per day (ft/d). Fine to medium sands have hydraulic conductivities typically ranging from 70 to 200 ft/d. The hydraulic conductivity of very fine sand and silt typically ranges from 30 to 70 ft/d; silt and clay deposits have hydraulic conductivities from about 10 to 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. Moraine deposits have a variable lithology—ranging from gravel and sand to silt and clay—and generally have lower average hydraulic conductivities than outwash deposits. Most areas, including moraines, have trends of decreasing hydraulic conductivity with depth.

Hydrologic Setting

The unconsolidated glacial sediments underlying Cape Cod compose an unconfined aquifer system that is bounded below by relatively impermeable bedrock, above by the water table, and laterally by salt water: Cape Cod Bay to the northeast, Cape Cod Canal to the northwest, Buzzards Bay to the west, and Nantucket Sound to the south (fig. 1). The Sagamore flow lens on central and western Cape Cod is the largest and westernmost of the six separate groundwater-flow lenses that underlie Cape Cod; the flow lens is hydraulically separated at its northwestern extent from mainland Massachusetts by the Cape Cod Canal and from the adjacent flow cell by the Bass River at its eastern extent. Recharge from precipitation is the sole source of water to the aquifer system. About 45 inches (in.) of precipitation falls annually on Cape Cod; slightly more than half of the precipitation recharges the aquifer across the water table (LeBlanc and others, 1986). The remainder is lost to evapotranspiration; surface runoff is negligible owing to the sandy soils and low topographic relief of the area.

Groundwater flows outward from regional groundwater divides towards natural discharge locations at streams, coastal estuaries, and the ocean (fig. 1). On western Cape Cod, the groundwater flows outward from the northwestern part of the Sagamore flow lens (fig. 2A). Most groundwater flows through shallow sediments and discharges to streams and estuaries; groundwater recharging the aquifer near groundwater divides flows deeper in the aquifer and discharges to the ocean (fig. 2B). Most groundwater discharge (about two thirds of the total) discharges into salt-water bodies. About 25 percent of groundwater 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). Water-table altitudes on the Sagamore flow

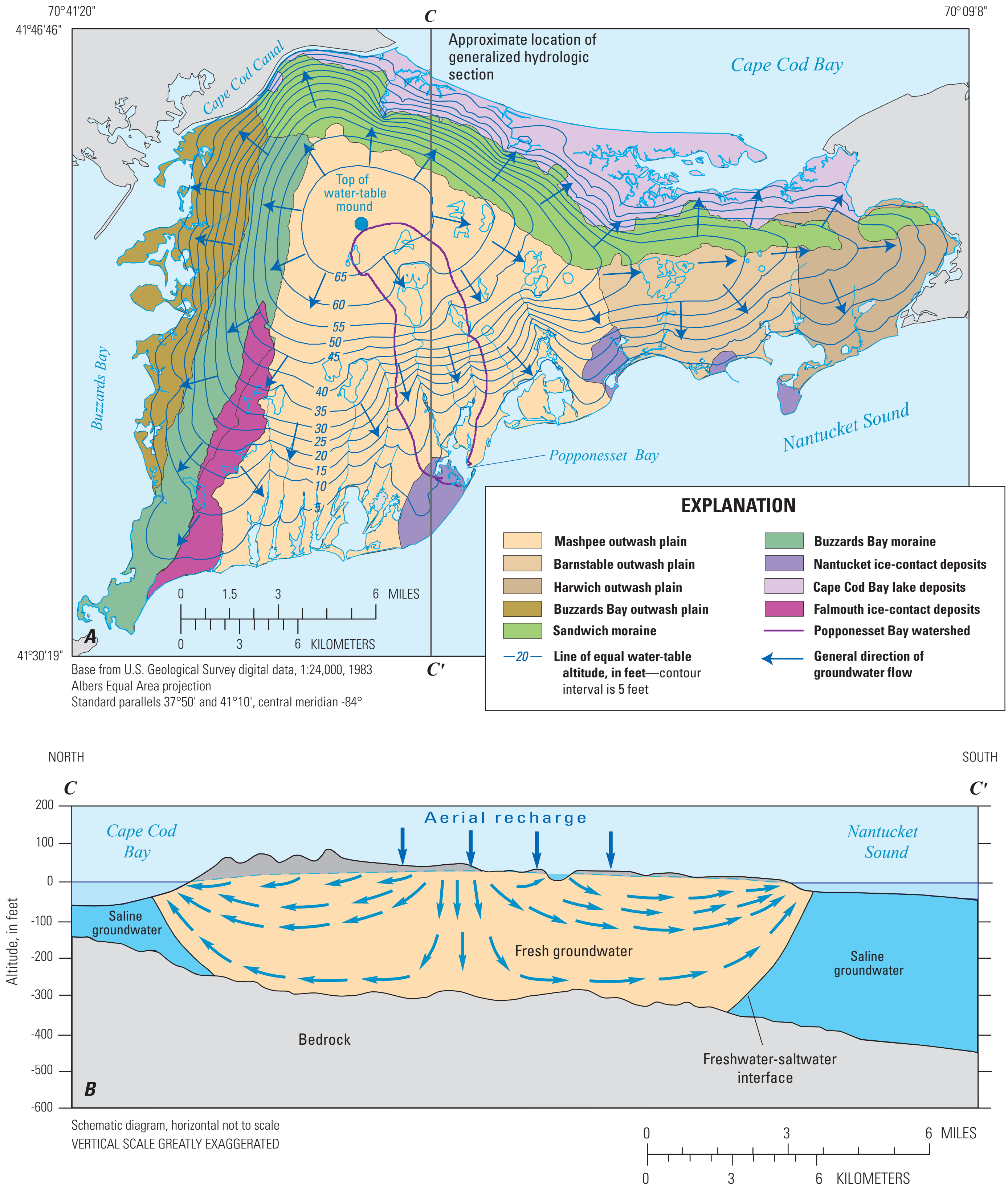


Figure 2. *A,* Surficial geology, water-table altitudes, general groundwater-flow directions, and location of Popponesset Bay on the Mashpee outwash plain and *B,* generalized north-south hydrologic cross section through western Cape Cod, Massachusetts.

lens exceed 65 ft above NAVD 88 (fig. 2A). Water-table contours and groundwater-flow patterns are strongly affected locally by numerous kettle-hole ponds; these are flow-through ponds that focus groundwater flow (Walter and others, 2002). Groundwater-flow paths converge in areas upgradient of the ponds, where groundwater discharges into the ponds, and diverge in downgradient areas, where pond water recharges the aquifer. Some ponds have surface-water outlets that drain into freshwater streams. Streams generally are areas of groundwater discharge (gaining streams) and receive water from the aquifer. Some stream reaches may lose water to the aquifer (losing streams), however, particularly in areas downgradient from pond outflows.

About 7 percent of the water recharging the Cape Cod aquifer system is removed for water supply (Walter and Whealan, 2004). Most of this water is returned to the system as disposed wastewater, either as dispersed septic-system return flow or as point discharges to the aquifer at WTFs. Although most of the pumped water is returned to the aquifer, the water is usually disposed in areas away from where the water was withdrawn, particularly in areas served by public water supply. Large-capacity supply wells decrease groundwater levels and can affect natural resources by drying vernal pools, drawing down ponds, and decreasing streamflows by changing hydraulic gradients and either intercepting groundwater that would have discharged to a stream or by direct infiltration of water from the stream. In the vicinity of large wastewater-disposal facilities, groundwater can become mounded, and streamflows can increase locally as a result.

The Popponesset Bay watershed extends from Nantucket Sound to the top of the regional water-table mound on the Sagamore lens (fig. 2). Groundwater-flow directions in the watershed generally are southward from the top of the regional water-table mound to discharge locations at streams and Popponesset Bay (fig. 3). The northern part of the watershed has several large kettle-hole ponds: Mashpee-Wakeby Pond, Peters Pond, Snake Pond, and Santuit Pond (fig. 3). Freshwater streams—Mashpee River, Quaker Run, and Santuit River—are the predominant hydraulic features within the central part of the watershed. The Popponesset Bay system, to the south, consists of six estuaries defined by hydrodynamic, water-quality, and ecological criteria: Popponesset Bay, Popponesset Creek, Ockway Bay, Mashpee River (tidal), Shoestring Bay, and Pinquickset Cove (fig. 3) (Howes and others, 2004). Groundwater-flow patterns in the northern part of the watershed are affected by kettle-hole ponds, and groundwater-flow directions in the central part of the watershed converge towards discharge areas at freshwater streams; nearer the coast, groundwater-flow patterns are controlled by the geometry of coastal water bodies, which receive direct groundwater discharge (fig. 3).

Recharge Areas to Wells and Ecological Receptors

The area at the water table that contributes recharge to a receptor is referred to as the recharge area to that receptor. Natural receptors include coastal waters (estuaries and open coastal waters), streams, and ponds; water can also discharge into pumped wells. Ponds differ from other natural receptors in that water entering ponds can reenter the aquifer to later discharge into one of the other types of receptors. Recharge areas form a mosaic that is a function of hydraulic gradients and the locations and types of receptors. Walter and others (2004) present a regional mosaic of recharge areas to major receptors for Cape Cod as well as examples of recharge-area mosaics for surface-water-dominated and groundwater-dominated estuarine watersheds.

The recharge mosaic indicates the mechanisms by which anthropogenic compounds can be introduced into an estuarine system (fig. 4). Water entering the aquifer from recharge areas to estuaries, along with any associated anthropogenic compounds, discharges directly to those estuaries, whereas water and associated compounds entering from recharge areas to streams discharge to those streams and enter estuaries as surface-water inflow. Water entering recharge areas to ponds, along with associated compounds, discharges into the ponds and can reenter the aquifer as pond-water outflow. The water and associated compounds can then either discharge directly to a coastal-water body or into a stream, entering the estuaries respectively as groundwater discharge or surface-water inflow. In some cases, ponds have surface-water outflows, and some water and associated compounds discharge directly from the ponds into streams, entering the estuaries with surface-water inflow (Walter and Masterson, 2011).

The Popponesset Bay watershed is a surface-water-dominated system owing to the numerous kettle-hole ponds and freshwater streams within the watershed (fig. 3). Areas contributing recharge directly to ponds, streams, and estuaries compose 40, 33, and 27 percent, respectively, of the Popponesset Bay watershed (fig. 3). Recharge and associated nitrogen discharge from the southern part of the watershed directly to an estuary; from the central part of the watershed into freshwater streams and then the estuaries as surface-water inflow; and from the northern part of the watershed into kettle-hole ponds, the aquifer, freshwater streams, and finally estuaries as surface-water inflow. Some water and associated nitrogen discharging into Mashpee-Wakeby Pond and Santuit Pond leave the ponds as surface-water outflow into the Mashpee River and Santuit River, respectively. Some water leaves the watershed through pumped wells. These mechanisms control the introduction of nitrogen into the Popponesset Bay estuarine system and could affect the potential for the attenuation of nitrogen concentrations prior to entering the system.

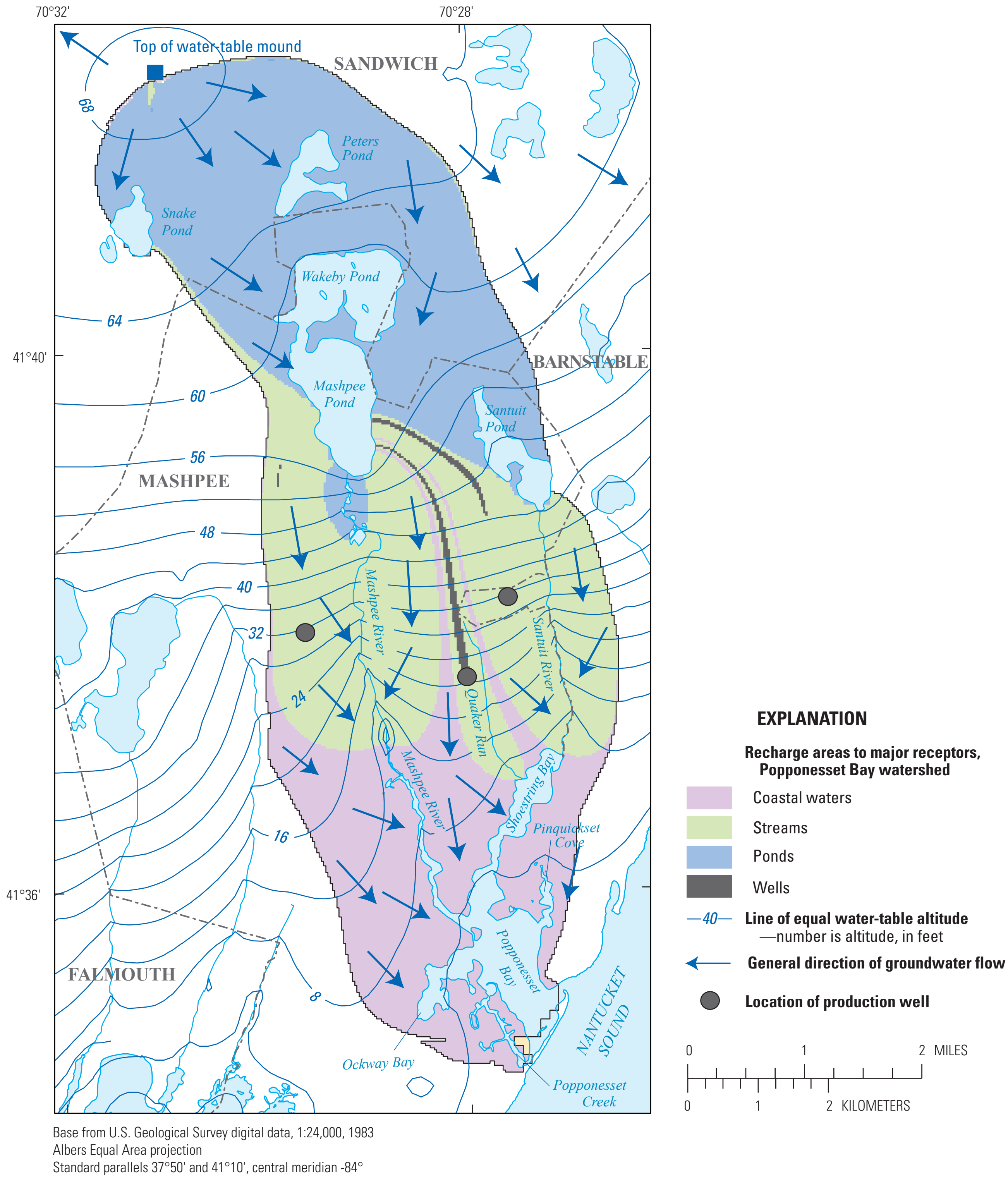


Figure 3. Water-table altitudes, directions of groundwater flow, and recharge areas to wells and major ecological receptors in the Popponesset Bay watershed, western Cape Cod, Massachusetts.

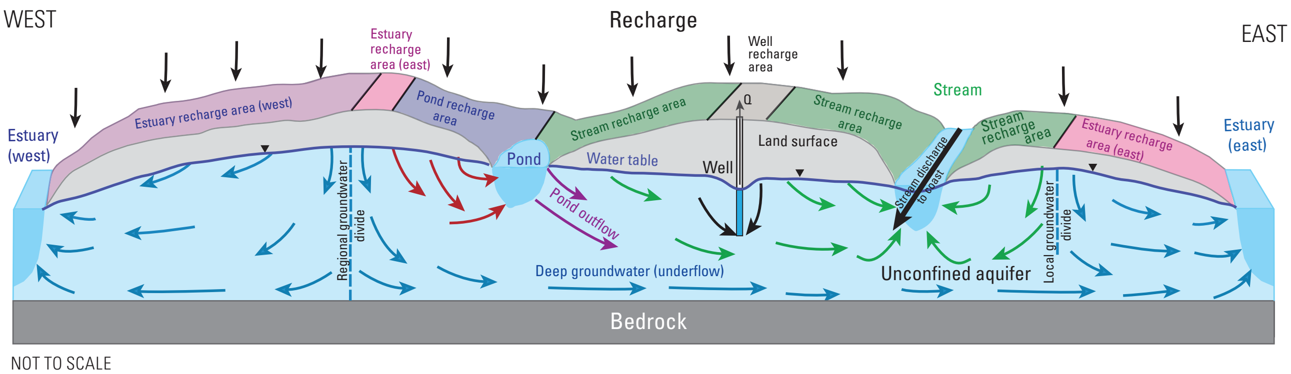


Figure 4. Different types of receptors, recharge areas, and mechanisms of freshwater discharge in coastal watersheds typical of Cape Cod, Massachusetts (modified from Walter and others, 2004).

Nitrogen in the Cape Cod Aquifer

Nitrogen is the element of most concern in the waters around and on Cape Cod owing to its role as a limiting nutrient for algal growth in marine and brackish waters and the adverse ecological and economic effects of nitrogen-related eutrophication in some estuaries. Nitrogen is introduced into the groundwater system from both natural and anthropogenic sources; on Cape Cod, the largest source is wastewater. Nitrogen generally is transported conservatively from surface sources to receptors under the oxic conditions that prevail in the aquifer.

Sources of Nitrogen

Natural sources of nitrogen in coastal watersheds on Cape Cod include leaf fall and atmospheric deposition. In the 16 estuaries for which analyses have been completed by the MEP (2012), recharge through natural areas contributed an average of 3.2 percent of the total nitrogen load and did not exceed 7 percent of the total nitrogen load in any estuary on Cape Cod (http://www.oceanscience.net/estuaries/reports.htm, accessed March 18, 2012, Ocean Science, 2012). Howes and others (2004) reported that leaf fall likely is not a significant component of the nitrogen load to Popponesset Bay. Atmospheric deposition directly onto receiving waters is the largest natural source contributing nitrogen to estuaries, although atmospheric-deposition rates are affected by the anthropogenic release of nitrogen into the atmosphere. Atmospheric-deposition rates between 1910 and 1995 on Cape Cod ranged from 0.9 to 4.0 kilograms per hectare per year (kg/ha/yr), with an increase of about 0.26 kg/ha/yr per decade (Bowen and Valiela, 2000). A concentration of about 1.09 milligrams per liter (mg/L) in precipitation is used by the MEP to estimate loads of wet atmospheric deposition onto ecological receptors (Howes and others, 2003). The median proportion of total nitrogen load resulting from wet atmospheric deposition is 12.0 percent for the 16 watersheds for which MEP analyses have currently been completed (2012). The proportion for individual estuarine systems ranged from less than 1 percent to more than 40 percent and is a function of the size of a coastal water body relative to the size of the watershed (http://www.oceanscience.net/estuaries/reports.htm, accessed March 18, 2012).

Anthropogenic sources of nitrogen include wastewater, fertilizers, and runoff from impervious surfaces. In groundwater in glacial aquifers that underlie agricultural areas, the primary source of nitrogen to pumped wells is nitrogen fertilizers (Warner and Arnold, 2010), and fertilizer application has resulted in the contamination of glacial aquifers in many areas of the United States (DeSimone and others, 2009). In urbanized, nonagricultural areas, such as Cape Cod, fertilizer application generally is limited to lawns and golf courses. In the 16 coastal watersheds for which MEP analyses have been completed, the median proportions of total nitrogen loads to estuaries resulting from fertilizer application and runoff from impervious surfaces were 5.0 and 6.0 percent, respectively (http://www.oceanscience.net/estuaries/reports.htm, accessed March 18, 2012).

Wastewater is the largest source of nitrogen to glacial aquifers and ecological receptors on Cape Cod; the mean and median proportions of total nitrogen loads in the 16 watersheds for which MEP analyses have been completed were 72.1 and 72.5 percent, respectively (http://www.oceanscience.net/estuaries/reports.htm, accessed March 18, 2012). Wastewater sources included both untreated return flow from septic systems and the discharge of treated wastewater from WTFs. Only about 15 percent of residential areas on Cape Cod currently are sewered, and the largest source of wastewater-derived nitrogen is septic-system return flow. A mean of about 92 percent of the total load of wastewater-derived nitrogen for the 16 coastal watersheds is currently from septic-system return flow (http://www.oceanscience.net/estuaries/reports.htm, accessed March 18, 2012).

Land uses within the 19-square-mile (mi^2) watershed to Popponesset Bay include residential commercial, industrial, agricultural, and recreational, and open space (fig. 5A). Commercial and industrial land uses compose about 6 and 1 percent of the watershed, respectively; the remaining nonresidential land uses, most of which are undeveloped or open space, constitute about 46 percent of the watershed (fig. 5A). Residences, which occupy about 47 percent of the area (fig. 5A), compose the largest single land use in the watershed.

Residential land use provides a potential source of nitrogen from septic-system return flow (Persky, 1986) and, as part of the MEP process, parcel-scale water-use data were compiled in cooperation with local communities to estimate the distribution of return flow—and associated nitrogen—within the watershed. About 33 percent of the watershed receives septic-system return flow (fig. 5B). In 2003, about 32,300 kilograms per year (kg/yr) of nitrogen from wastewater sources discharged into the Popponesset Bay system, representing about 66 percent of the total nitrogen load to the system (Howes and others, 2003). In addition to septic-system return flow, wastewater-derived nitrogen is introduced into the watershed at four small wastewater-treatment facilities (fig. 5B). Anthropogenic nitrogen loads to the Popponesset Bay system from fertilizer application and impervious surfaces were 5 and 6 percent, respectively; nitrogen loads to the Popponesset Bay system from natural sources—wet atmospheric deposition and recharge onto natural areas—were 17 and 5 percent, respectively (Howes and others, 2003).

Transport of Nitrogen

Background, or natural, concentrations of nitrogen generally are less than 0.1 mg/L in the Cape Cod aquifer (LeBlanc, 1984; Savoie and others, 2012; Barbaro and others, 2013). Concentrations of nitrogen in septic-system effluent can exceed 30 mg/L as N, and concentrations exceeding 10 mg/L have been observed in groundwater near residential areas (Weiskel and Howes, 1992). Advective transport generally is the dominant component of transport for conservative solutes such as nitrate in the aquifer because of high recharge rates and the generally high permeability of the aquifer sediments (LeBlanc, 1984). Groundwater velocities of more than 1.5 ft/d have been observed at the USGS Toxics Substances Hydrology Research Site near the former wastewater-treatment facility (WTF) at Massachusetts Military Reservation (MMR) (fig. 1) (LeBlanc and others, 1991). The rate of advective transport in the aquifer is related to the location of a source area relative to regional groundwater divides and discharge locations; near divides, where horizontal gradients are small and downward components substantial, groundwater flow is more vertical and slower than flow recharged farther from divides, where horizontal flow predominates (fig. 2B) (Walter and Masterson, 2003b; Walter and others, 2004). Traveltimes, defined as the total times required for water to move from recharge locations at the water table to natural discharge locations, are largest for groundwater flow that originates near regional groundwater divides and range from essentially zero adjacent to discharge boundaries to hundreds of years near groundwater divides (Walter and others, 2004).

Nitrogen in natural waters occurs primarily as nitrate (NO_3^-), ammonium (NH_4^+), dissolved nitrogen gas (N_2(g)), or as organic nitrogen; the predominant species of inorganic nitrogen in groundwater is determined by the pH and redox conditions of the waters (Hem, 1985). LeBlanc (1984) reported that nitrate was the predominant form of nitrogen in treated sewage effluent at the MMR wastewater-treatment facility, and nitrate emanating from the site has been transported, primarily by advection, more than 7 miles (mi) downgradient from the source (Barbaro and others, 2013). Denitrification occurs within parts of the plume characterized by high organic carbon and low dissolved oxygen concentrations; nitrogen is primarily in the form of ammonia in anoxic parts of the plume (Barbaro and others, 2013). In addition, Weiskel and Howes (1992) found that most nitrogen emanating from septic systems in a sandy coastal aquifer in southeastern Massachusetts was in the form of nitrate.

Denitrification, in which nitrate is reduced to nitrogen gas, can remove or attenuate nitrate in an aquifer (Hem, 1985) and in most aquifers is coupled with the oxidation of organic carbon present in aquifer. The glacial aquifer underlying Cape Cod generally is oxic and contains little organic carbon (LeBlanc, 1984), and, therefore, nitrate attenuation through denitrification is likely to be limited. 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 and the conservative transport of nitrate with groundwater flow (advection) from surface sources to ecological receptors.

Reducing conditions favorable for denitrification may occur in (1) septic-system return flow prior to recharge at the water table, (2) large plumes downgradient of WTFs and landfills, (3) local-scale reducing environments in the aquifer, (4) muddy, organic-rich bottom sediments in estuaries, and (5) surface-water bodies. Weiskel and Howes (1992) found that a total of about 16 to 36 percent of nitrogen (primarily as nitrate) in septic-system effluent was lost during transport over small distances (less than about 2,000 ft), mostly within the septic systems or in the underlying unsaturated zone prior to recharge at the water table. Local environments can be reducing in deep parts of the aquifer, suggesting the potential for denitrification and nitrogen attenuation over long transport distances and times (Colman and others, 2004). Kroeger and Charette (2008) report geochemical conditions favorable for and evidence of denitrification within estuarine muds in similar hydrogeologic settings. However, the importance of these attenuation processes in the Cape Cod aquifer is unknown, and estimates of nitrogen loads to estuaries by the MEP are based on the assumption of no attenuation in aquifer or estuarine sediments, so that the highest plausible load estimates will be used to implement nitrogen TMDLs.

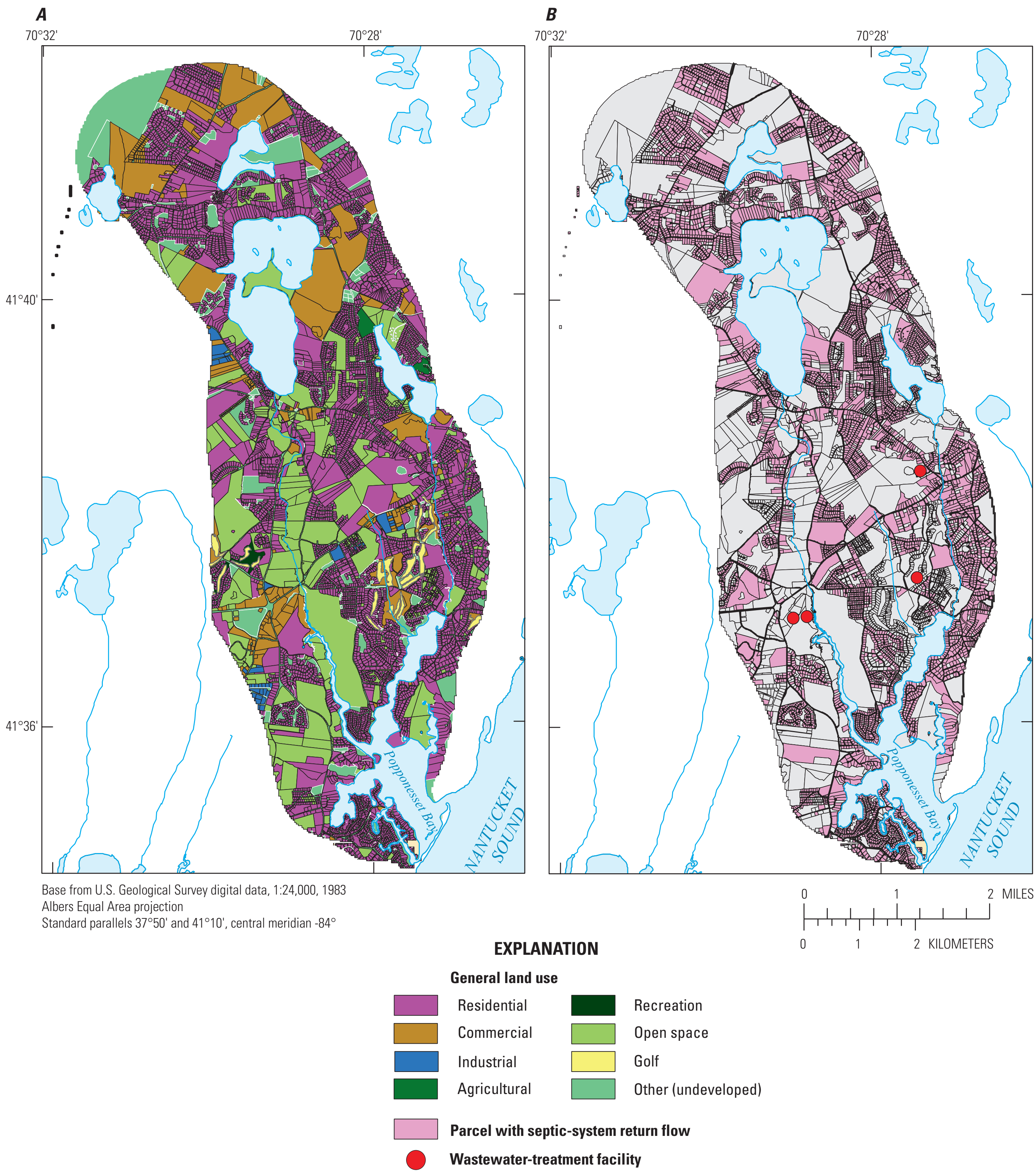


Figure 5. *A,* Land use and *B,* the locations of wastewater-treatment facilities and parcels served by septic systems within the Popponesset Bay watershed, western Cape Cod, Massachusetts.

Nitrogen in coastal watersheds is attenuated within surface waters prior to entering coastal receiving waters; attenuation results from geochemical conditions favorable for denitrification and uptake of nitrogen by biota within the surface waters. Current nitrogen-load estimates by the MEP are based on the assumption that 50 percent of nitrogen is attenuated within ponds and 30 percent in streams; the attenuation is accounted for by decreasing nitrogen sources within the recharge areas to surface waters—50 and 30 percent from recharge areas to ponds and streams, respectively (Howes and others, 2003).

Methods of Analysis

Groundwater-flow and single-phase solute-transport modeling techniques were used to simulate the effect of wastewater-management actions on nitrogen loading to the Popponesset Bay estuarine system. First, an existing regional groundwater model was modified to facilitate the simulation of nitrogen transport (conservatively as nitrate) within the Popponesset Bay watershed; second, a subregional model of the watershed was developed and linked to the modified regional model (fig. 6A); and third, the subregional model was linked to a solute-transport code to allow for the simulation of nitrogen transport in the aquifer to wells and ecological receptors. A Geographic Information System (GIS) was used to convert detailed data on nitrogen sources into model inputs.

Groundwater-Flow Modeling

An existing regional model of the Sagamore flow lens previously developed to determine the hydrologic budget and the effects of current and future well withdrawals on surface waters and to delineate recharge areas to wells and ecological receptors was repurposed in support of the MEP (Walter and Whealan, 2004; Walter and others, 2004). A detailed discussion of the model design and calibration is presented in Walter and Whealan (2004). The existing regional model was calibrated to current hydrologic conditions and includes stresses similar to current stresses. In addition, the existing regional model also was the basis for previous work to delineate recharge areas to receptors as part of the MEP process. Therefore, it was considered reasonable to modify the regional model to facilitate simulations of nitrogen transport discussed in this report.

Regional-Model Design

The regional model of the Sagamore flow lens extends from the Cape Cod Canal to the Bass River (fig. 1) and uses the finite-difference groundwater-modeling program MODFLOW–2005 (Harbaugh, 2005) to simulate groundwater flow in the aquifer. The model has 246 rows and 365 columns with a uniform horizontal discretization of 400 ft and a total active modeled area of 246 mi^2 (fig. 6A). The original model has 20 layers with thicknesses ranging from 10 ft in the top 170 ft of saturated thickness to more than 200 ft in the deepest layer (fig. 6B).

Surface waters are represented as head-dependent flux boundaries, with open coastal waters represented by the GHB Package and saltwater wetlands and some freshwater wetlands by the DRN Package (Harbaugh and others, 2000) (fig. 6A). Freshwater streams and some freshwater wetlands were simulated by the STR package (Prudic, 1989), which allows for an accounting of streamflow and pumping-induced streamflow depletion. Freshwater ponds are simulated as active parts of the aquifer, and simulated pond levels can change in response to changing stresses. Ponds are represented as areas of very high hydraulic conductivity (100,000 ft/d) within the aquifer; the HFB Package (Hsieh and Freckleton, 1993) is used to represent pond-bottom sediments implicitly.

A natural recharge rate onto aquifer sediments of 27.1 inches per year (in/yr) is specified as the sole source of water to the aquifer by the RCH Package (Harbaugh and others, 2005). This value was obtained from long-term precipitation records, was adjusted during model calibration (Walter and Whealan, 2004), and is consistent with previous investigations on Cape Cod and southeastern New England (Barlow and Dickerman, 2000; DeSimone and others, 2001; Masterson and others, 1997b; Walter and Masterson, 2003b). Recharge onto surface-water bodies was 16 in/yr, representing the difference between precipitation and pan evaporation (Farnsworth and others, 1982). Recharge was further adjusted to account for septic-system return flow in residential areas. Parcel-scale water-use data were not available during development of the regional model (2002), so the volume of generated wastewater was determined from the total volume of groundwater withdrawn by each town; a consumptive loss of 15 percent was applied, and the remaining volume was evenly distributed across model cells representing nonsewered residential areas. Recharge also was adjusted in areas of existing wastewater-disposal facilities (Walter and Whealan, 2004). Withdrawal of water from 95 production wells was simulated by using the WEL Package (McDonald and Harbaugh, 1988) and average pumping rates for 2003 (Walter and Whealan, 2004). Pumping at three proposed (2003) wells also was represented in analyses presented in Walter and Whealan (2004); these wells, which currently (2012) are in operation, are referred to as the Upper Cape Cooperative wells (fig. 6A) and were installed in the northern part of the MMR to offset any possible losses of potable water related to contamination from sources on the MMR.

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 commun., 2002). Hydraulic conductivities varied spatially and with depth and ranged from 350 ft/d for coarse sand and gravels to 10 ft/d for silt and clay (Walter and Whealan, 2004). The steady-state regional models were calibrated by using measured long-term water levels and streamflows; in addition, delineated contaminant plumes in and around the MMR to the west of

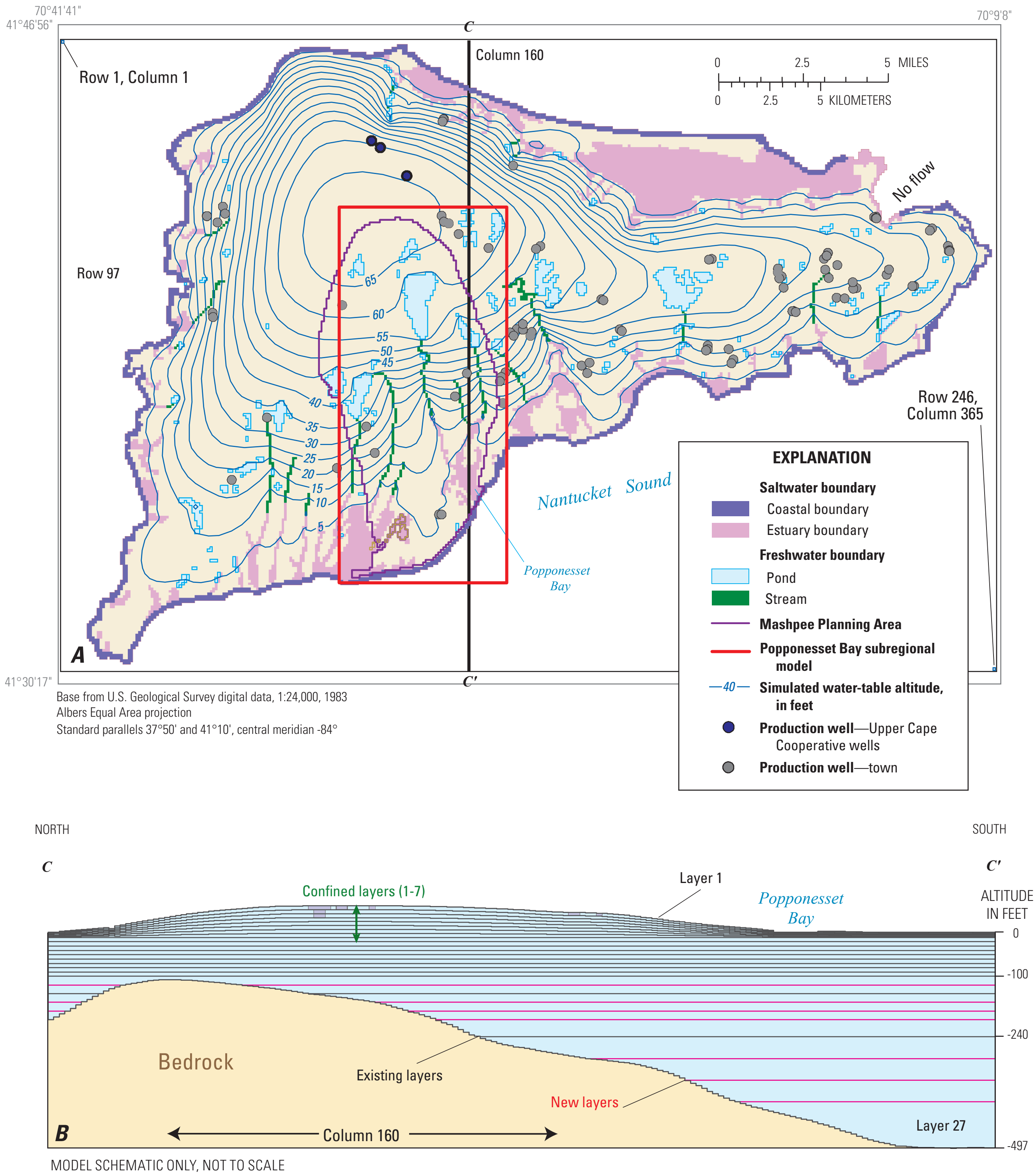


Figure 6. *A,* Extent of the regional model of the Sagamore flow lens with hydraulic boundary conditions, locations of production wells, and simulated water-table altitudes, and *B,* section showing vertical discretization of the original and revised models along column 160 transecting Popponesset Bay, western Cape Cod, Massachusetts.

the Popponesset Bay watershed were used as indicators of advective flow paths to adjust hydraulic parameters to better match regional hydraulic gradients (Walter and Whealan, 2004). 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). Although recharge and pumping stresses vary through time, and versions of the regional models that incorporate these time-varying stresses are available, steady-state models likely are adequate for the simulation of advective flow because the time scale of the variability of these stresses is shorter than the time scale of transport in the aquifer (Walter and Masterson, 2003).

Modifications to the Regional Model

Two sets of modifications were made to the regional model to facilitate better simulation of nitrogen transport: (1) conversion of the model to a linear (confined) configuration and (2) addition of model layers to increase the vertical discretization at depth. The existing model is an unconfined (non-linear) model that has seven layers with uniform thicknesses of 10 ft above NAVD 88—these layers can go dry as the simulated head falls below the bottom of a layer. Although this model configuration is sufficient for the simulation of groundwater flow, the resultant nonlinearity can be problematic for solute-transport simulations. The model was converted to a confined configuration—or linearized—by changing layer tops and bottom to conform to the simulated water table. For a given set of stresses, the water-table altitude was first simulated by using the unconfined configuration. In each model cell, the bottoms of the 7 layers above sea level were spaced equally between the simulated water-table altitude and 10 ft below sea level (NAVD 88) (fig. 6B), and the layers were represented as confined. Model boundaries, which were assigned to different layers on the basis of altitude, were all assigned to the top layer by using the same altitude as in the original model.

The original regional model of the Sagamore flow lens has a uniform discretization of 10 ft for the top 17 layers, which were above -100 ft (Walter and Whealan, 2004). In the modified regional model, layer bottoms were adjusted for the top 7 layers as part of the linearization process, so that the thicknesses of the 8 layers between the water table and an altitude of -10 ft (as described above) would be variable but less than 10 ft. The next 10 layers down to -100 ft have a uniform thickness of 10 ft in both the original and linearized models (fig. 6B). The bottom 3 layers in the original model, which can be truncated by the bedrock surface, were subdivided into 10 layers in the modified regional model to decrease layer thicknesses and thus the amount of numerical dispersion in the solute-transport simulations. The maximum thicknesses of the bottom layers in the original and modified regional models are 257 and 107 ft, respectively, in the deepest part of the simulated aquifer along regional model column 160 underlying Popponesset Bay (fig. 6B).

Although the bottom altitudes of the layers in the two model versions differ for the top seven layers, the resulting combined transmissivity of the layers is the same for both model configurations; likewise, subdivision of the bottom 4 layers into 11 layers does not change the total transmissivity in that part of the aquifer. As a result, the unconfined and confined model solutions are essentially identical. The simulated mean head difference between the confined and unconfined models at 301 head-observation locations (Walter and Whealan, 2004) was about 0.02 ft; the simulated fluxes at hydrologic boundaries were within 1 percent for each boundary type (estuary, coast, and streams) (fig. 7A). Both confined and unconfined models were run and compared for all scenarios to ensure consistency between the models.

Subregional Model Development

A subregional groundwater model was developed that encompasses about 5 mi^2 on western Cape Cod. The model domain includes the entire Popponesset Bay watershed and extends from near the top of the water-table mound (the regional divide for the Sagamore lens) at its northern extent to Popponesset Bay and Nantucket Sound at its southern extent (fig. 6A). The subregional model has 592 rows and 264 columns that coincide with the regional-model grid, but with a uniform horizontal discretization of 100 ft. The model has 27 layers with the same vertical discretization as the modified regional model (fig. 6B). The hydrologic boundaries in the subregional model coincide with those of the regional model (fig. 8). The finer horizontal discretization could have facilitated a more detailed representation of hydraulic boundaries; however, coincident boundaries were chosen to allow for consistency between the analyses presented here and analyses done as part of a previous modeling effort in support of the MEP (Walter and others, 2004). The only change in hydrologic boundaries was for freshwater streams, which were represented as streams by using the STR Package (Prudic, 1989) in the regional model and as drains in the subregional model by using the DRN Package. The change in the stream-simulation method minimized overly complex input data files arising from the subregional discretization; because streams on Cape Cod generally are gaining features that receive groundwater discharge, the representation of streams as drains was appropriate. The subregional model was linked hydraulically to the regional model with a specified-head boundary along its external boundary by using the CHD Package (Harbaugh and others, 2005); the specified heads along the external boundary are the same as the heads in the regional model at coincident cells.

Hydraulic stresses (recharge and pumping) and intrinsic properties—hydraulic conductivity and boundary leakances—are the same in the subregional model and coincident parts of the regional model. Hydraulic conductivities of the shallowest outwash and ice-contact deposits that underlie

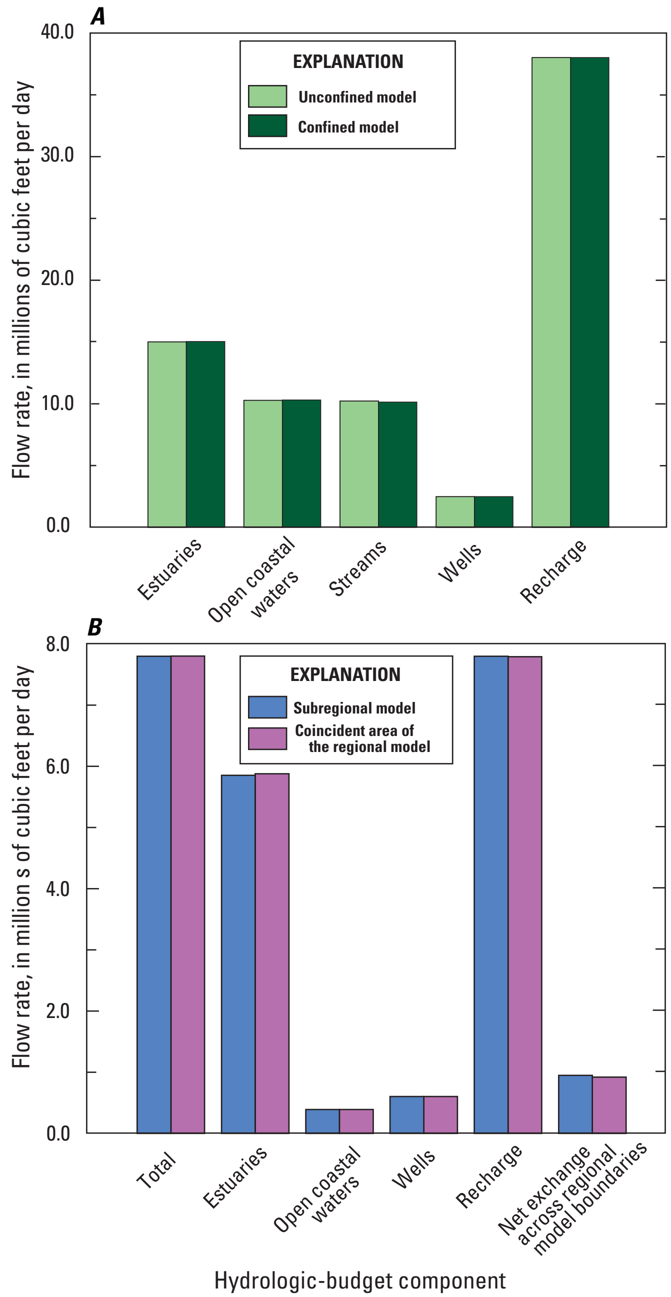


Figure 7. Comparison of hydrologic budgets between *A,* the unconfined (original) and confined (modified) regional models, and *B,* a subzone of the modified regional model and the entire subregional model of the Popponesset Bay watershed, western Cape Cod, Massachusetts.

the Popponesset Bay watershed and comprise coarse fluvial sand and gravel range from about 300 ft/d in the northern part of the watershed to about 180 ft/d near Nantucket Sound. Hydraulic conductivities in the deepest sediments that underlie the watershed and comprise fine sand, silt, and clay deposited in low-energy lacustrine environments are about 10 ft/d. The regional and subregional models yield essentially identical solutions because the subregional model is consistent with the coincident part of the regional model and linked to it by a consistent specified-head boundary. Simulated water-table altitudes and gradients in the subregional model and the coincident part of the regional model also are essentially identical (fig. 8). Similarly, recharge and fluxes across hydrologic boundaries and subregional model boundaries agree within 1 percent (fig. 7B).

Simulation of Solute Transport

Sources of anthropogenic contaminants on Cape Cod include point sources—such as landfills, WTFs, and industrial sites—and nonpoint sources, such as septic systems. Contaminants are transported as dissolved solutes with groundwater flow from the water table underlying surficial sources to wells and ecological receptors. As an example, contaminant plumes around the MMR on western Cape Cod (fig. 1) have migrated several miles downgradient since the mid-1950s from a number of sources on the facility (U.S. Air Force Center for Environmental Excellence (AFCEE), 2000; AFCEE, written commun., 2008). The one-dimensional subsurface transport of a solute can be expressed as

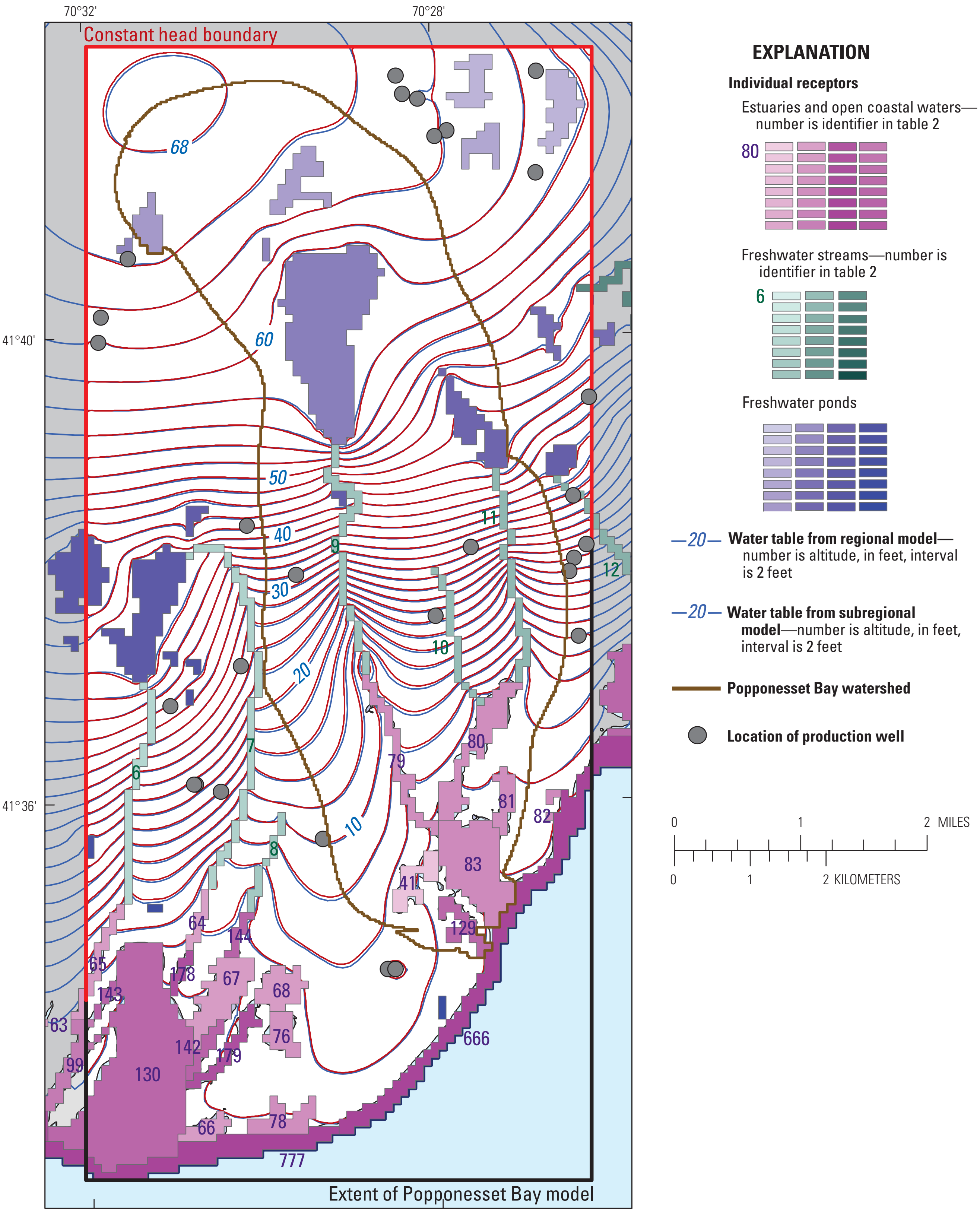
$$\underbrace{\delta C/\delta t = D_L\,(\delta^2 C/\delta x^2)}_{\text{Dispersion}} + \underbrace{D_d\,(\delta^2 C/\delta x^2)}_{\text{Diffusion}} -$$

$$\underbrace{v_x\,(\delta C/\delta x)}_{\text{Advection}} - \underbrace{(B_d/n)\,(\delta C_{(s)}/\delta t)}_{\text{Sorption}} + \underbrace{(\delta C/\delta t)_{rxn}}_{\text{Reactions}}, \quad (1)$$

where

C = concentration of dissolved solute,
t = time,
D_L = longitudinal dispersivity,
D_d = diffusivity,
v_x linear groundwater velocity,
B_d = bulk density of aquifer sediments,
n = porosity,
$C_{(s)}$ = concentration of sorbed solute, and
rxn = subscript indicating chemical or biological reactions (modified from Fetter, 1992).

In the case of nitrate, which is conservative and nonreactive in oxic, carbon-poor aquifers, the last two terms of the equation (sorption and reactions) can be ignored, and diffusion generally is insignificant 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. Advection refers to transport of a solute with groundwater flow; dispersion, which occurs in all natural aquifers, refers to the spreading of solute mass caused by aquifer heterogeneity and preferential flow paths relative to average groundwater flow.



Base from U.S. Geological Survey digital data, 1:24,000, 1983
Albers Equal Area projection
Standard parallels 37°50' and 41°10', central meridian -84°

Figure 8. *A,* The subregional model of the Popponesset Bay watershed including hydrologic boundaries and simulated water-table altitudes for the regional and subregional models, western Cape Cod, Massachusetts.

Simulation of Advective Transport

Groundwater-flow models are mathematical representations of advective transport based on the assumption that mass is conserved along flow lines in an aquifer. Advection generally is the dominant transport mechanism in highly permeable, well-sorted aquifers, such as that which underlies Cape Cod, and, therefore, groundwater-flow models can be used to approximate transport. The use of particle tracking to represent the transport of nitrate in the Cape Cod aquifer implicitly is presented in Walter (2006). The advective transport of contaminants in the aquifer was simulated by using the USGS particle-tracking software algorithm MODPATH (Pollock, 1994), which uses cell-by-cell flows from the groundwater-flow model to track the movement of a particle or a field of particles through the simulated aquifer. Particle-tracking results were geoprocessed and displayed graphically, in part, by using the USGS postprocessing software MODTOOLS (Orzol, 1997).

Particle tracking was used to delineate recharge areas to wells and ecological receptors by seeding a field of particles at the water table and tracking them forward to a receptor. Particles entering a receptor of interest were identified; the starting locations of those particles were processed into georeferenced polygons by using a GIS. Traveltimes of individual particles from the water table to the receptor were processed into separate polygons. The porosity of aquifer sediments, to which velocities and traveltimes are inversely proportional, was specified as 0.3, typical of sandy glacial sediments.

Simulation of Mass Transport

Solute-transport codes explicitly represent the transport of a solute through the aquifer by solving independently for concentrations and, with additional terms, can account for dispersion, diffusion, sorption, and simple one-component reactions during transport (eq. 1). Nitrate is assumed to be the predominant nitrogen species given the geochemical conditions in the aquifer; sorption and reactions are assumed not to be important components of transport. Furthermore, diffusion is assumed to be negligible compared to advection and dispersion. The assumption of conservative transport also maximizes plausible estimates of mass-loading rates, an assumption that may be useful for adequate resource protection.

The solute-transport capabilities of particular interest in this analysis are (1) the explicit simulation of the transport of mass through the aquifer and surface waters, (2) the direct estimation of mass-loading rates at receptors, and (3) accounting for changing hydraulic gradients and dynamic recharge areas. The subsurface transport of nitrate was simulated by using the three-dimensional transport code MT3DMS (Zheng, 1999). MT3DMS simulates several components of transport, including advection, dispersion, and diffusion, and can simulate some single-species chemical reactions, including contaminant decay and various types of linear and nonlinear sorption. MT3DMS is linked to MODFLOW by the LMT6 Package (Zheng and others, 2001) and uses cell-by-cell flows and velocities from the flow model to simulate the movement of nitrate through the aquifer. The model calculates dissolved and sorbed concentrations of the solute of interest at cells within the active model domain at specified transport time steps. This information can be used to develop maps and cross sections of solute concentrations, breakthrough curves, and estimates of time-varying mass-loading rates at receptors. Mass-loading rates of nitrate at wells and ecological receptors were determined by using the Transport Observation (TOB) Package (Zheng, 2010).

The most important components of transport are advection and, to a lesser extent, dispersion for conservative solutes like nitrate. Dispersion, like heterogeneity, is scale dependent and generally increases with increasing transport distance. Dispersivity is near zero in a perfectly homogenous system and increases as the degree of heterogeneity increases. The spreading of mass in the aquifer affects concentrations and mass-loading rates at receptors. Longitudinal dispersion refers to a spreading of mass in the direction of groundwater flow and is the largest dispersive component; transverse and vertical dispersion refer to spreading of mass along the two axes orthogonal to the groundwater-flow direction and are much smaller than the longitudinal component. Based on an empirical relationship derived from literature values, longitudinal dispersivity ranges from 42 to 55 ft for transport distances of 7,500 and 13,500 ft, respectively (Luckner and Schestakow, 1991). Spitz and Moreno (1996) suggest longitudinal dispersivities of about 90 ft for a plume-scale transport distance of about 13,000 ft. AFCEE (2000) found that longitudinal, transverse, and vertical dispersivities of 35, 3.5, and 0.35 ft best matched plume geometries at a similar transport distance at the MMR. Transport distances from sources to receptors in the Popponesset Bay watershed typically are estimated to be less than 10,000 ft from recharge areas to major receptors (fig. 3); thus, dispersivity between 35 and 90 ft may be applicable at the watershed scale on the basis of literature values.

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

$$\alpha = \frac{\Delta X}{2} + \frac{V \Delta T}{2} \quad , \tag{2}$$

where

α is numerical dispersion,

ΔX is cell length,

V is velocity, and

ΔT is 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 days, 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. The criteria used to determine model discretization include the need to (1) minimize numerical dispersion (through minimal spatial and temporal discretization) and (2) limit the size of the model to make the transport simulation tractable. The finite-difference discretization of 100 ft used in the subregional model produces a reasonable numerical dispersion according to literature values, and still allows for a tractable model size for transport simulations—about 4.2 million cells. Numerical dispersion, as an artifact of a finite-difference solution, is a behavior equivalent to dispersion within an aquifer (Bear, 1979, p. 262). The approximate numerical dispersion in the solute-transport model—52 ft estimated from equation 1—is within the range of literature values between 35 and 90 ft for transport distances typical for the Popponesset Bay watershed. This estimate indicates that numerical dispersion alone may reasonably represent the general scale of real dispersion over watershed-scale transport distances in the aquifer. As with particle-tracking analyses, a porosity of 0.3 was specified for the solute-transport simulations. The effect of simulated dispersivities on time-varying nitrate concentrations and loads to receptors in the western Cape Cod aquifer system is presented in Walter (2006). It should be noted that dispersion affects the rate of change of concentrations and loads at a receptor, but not the steady-state loads, which are attained after sufficiently long transport times.

Hydrodynamic mixing of solutes within ponds cannot be explicitly represented in the transport model but can be implicitly represented by enhanced dispersion through the ponds. Ponds are represented as areas of high hydraulic conductivity and, as a result, flow is focused through the ponds. Open surface waters have an effective porosity of 1; however, the porosity of the ponds was specified to be the same as the aquifer (0.3) to increase velocities. The larger velocities result in more dispersion within the pond, approximating mixing.

Simulation of Nitrogen Sources and Wastewater-Management Actions

A number of potential wastewater-management actions are being evaluated to implement TMDLs for nitrogen in the Popponesset Bay watershed and the eastern part of the Waquoit Bay watershed; these two areas compose a 39-mi^2 area referred to as the Mashpee Planning Area (MPA) (figs. 6, 9A). Potential wastewater-management actions involve the collection (through sewering), treatment, and disposal of the treated wastewater at WTFs. The simulation of wastewater-derived nitrogen transport from sources to receptors and of the response of the system to wastewater-management actions requires a detailed understanding of point (WTF) and nonpoint (septic-system) sources within the watershed and the possible effects of changes in these sources on nitrogen-loading rates to the estuarine system.

As part of the MEP process, parcel-scale water-use data were compiled for the Popponesset Bay watershed in cooperation with local communities to estimate the distribution of return flow and associated nitrogen within the watershed. The database includes water-use data from the three towns in the watershed (Barnstable, Mashpee, and Sandwich) for different periods for each town between 1997 and 2001; the data were linked to digital parcel and tax-assessor data from 2000 and 2001 to quantify the amount and location of septic-system return flow (Howes and others, 2003). Parcel-scale water-use data were obtained from the Town of Mashpee (Jeff Gregg, GHD, Inc., written commun., August 15, 2011) and are consistent with data developed by the MEP. Septic-system return flow was estimated from the parcel-scale water-use data by applying a consumptive loss of 10 percent to be consistent with the MEP analysis (Howes and others, 2003) and was normalized to the areas of the parcels to convert return-flow volumes into rates (in feet/day). About 33 percent of the watershed currently (1997–2001) receives return flow; rates per parcel within the MPA range from 0.0002 to 0.8 ft/d (fig. 9A).

A GIS was used to convert the spatial distribution of return-flow rates, referred to as vector data (fig. 9A), into a rasterized form suitable for input into the groundwater model. The process consisted of three steps: (1) mapping the vector data to a data raster coincident with the model grid, (3) calculating the area-weighted mean return flow within each raster cell, and (2) mapping the rasterized data to the model grid (figs. 9B, 10). A data raster, as used in this report, refers to the spatial representation of data as values in a continuous grid of square cells. The result is a raster of area-weighted mean return-flow rates that is coincident with both the regional and subregional model grids (fig. 9B). The area-weighted mean for each raster cell is overlaid onto the model grid and converted into Cartesian coordinates for input into the model. The same procedure was used to estimate area-weighted mean flows representing disposal at WTFs.

This process was done for both the regional and subregional model grids to ensure consistent return-flow stresses between the two models. Return flow and disposal at WTFs are represented in the models as enhanced recharge—the sum of the natural recharge and the area-weighted mean return flow in each model cell. A version of the regional model was produced in which return flow in the MPA as represented in Walter and Whealan (2004) was removed and replaced with recharge representing the area-weighted return flow in each cell; return flow was unchanged outside the planning area

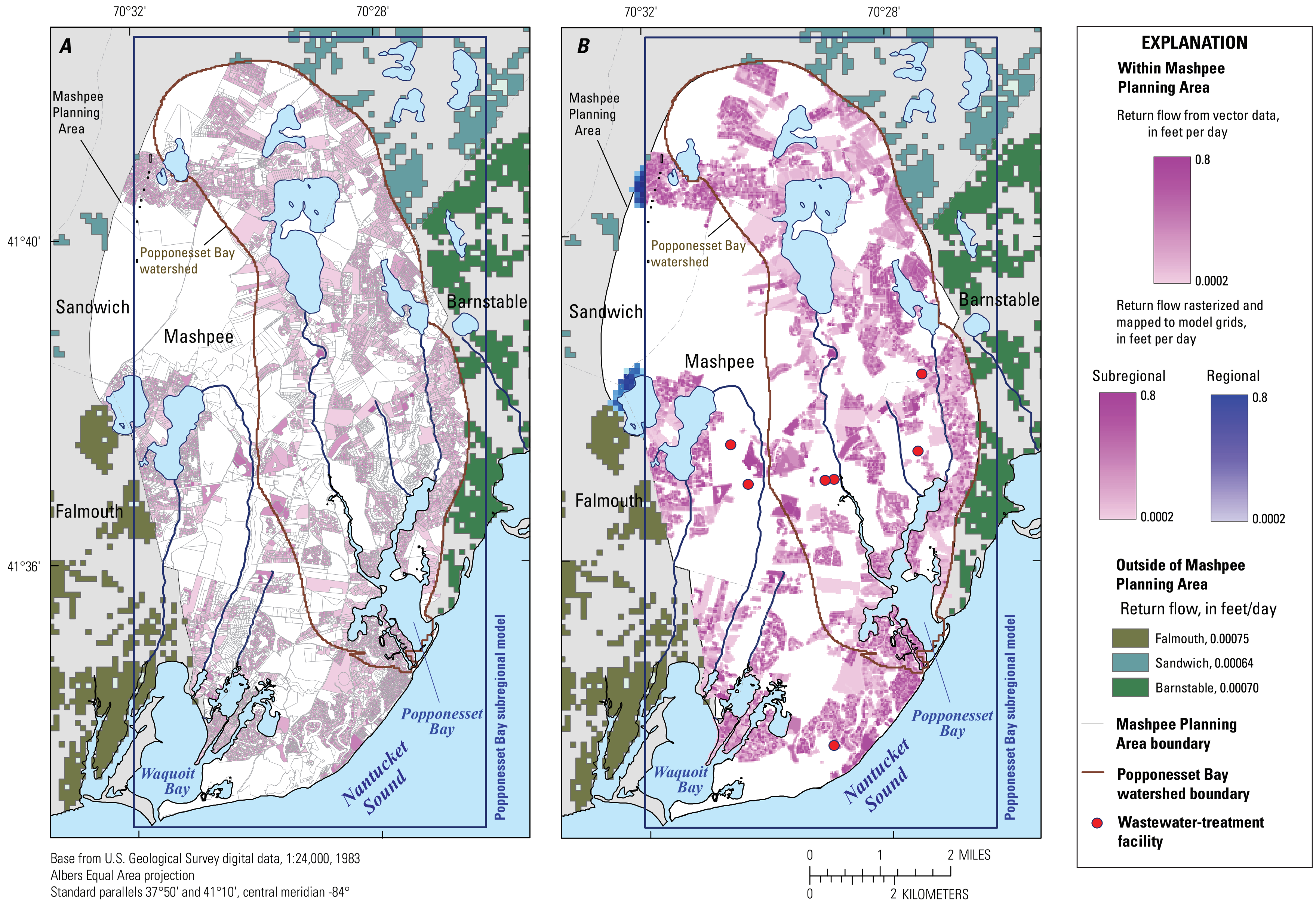


Figure 9. *A,* Return-flow rates from MEP vector data and *B,* the rasterized version of the data mapped to the regional and subregional model grids for the Mashpee Planning Area, western Cape Cod, Massachusetts.

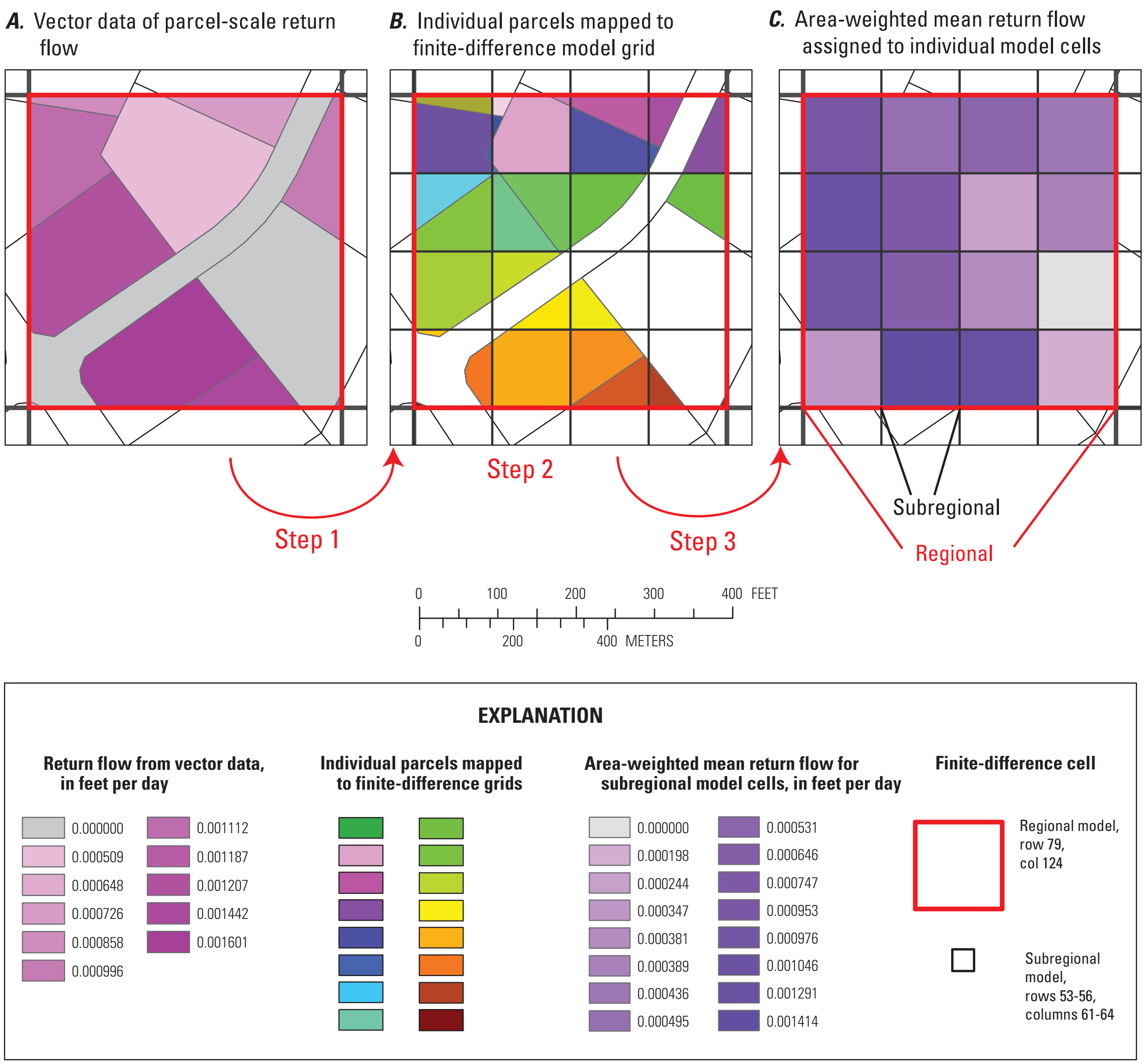


Figure 10. Examples of *A,* vector data of parcel-scale return flow, *B,* vector data mapped to a raster coincident with a model grid, and *C,* area-weighted mean return flow mapped to the model grid.

(fig. 11). The same approach was used to specify enhanced recharge in the subregional model, resulting in a more detailed recharge distribution (fig. 11). Although enhanced-recharge rates may differ between the regional and subregional models at a specific point, the enhanced-recharge rate in a regional model cell is the same as mean rate for the sixteen subregional cells within the cell (fig. 10C).

Currently (1997–2001) about 188,500 cubic feet per day (ft^3/d) of septic-system effluent are recharged into the aquifer; an additional 21,770 ft^3/d of treated wastewater are discharged at seven WTFs; four of the facilities are in the Popponesset Bay watershed (fig. 9B). Two sets of wastewater-management actions were evaluated: (1) full sewering of the MPA and disposal outside of the MPA and (2) two sequential phases of hypothetical but realistic actions that combined partial sewering and centralized disposal of most of the treated wastewater at WTFs within the MPA.

The first set of wastewater-management actions specifies the collection of all return flow (about 188,500 ft^3/d), the disposal of the treated wastewater offshore or on land at locations outside of the MPA, and a maintenance of disposal at existing WTFs (21,770 ft^3/d); the offshore disposal of treated wastewater is prohibited by law, however, and this set of wastewater-management actions is designed only to illustrate the response of the system to a full removal of septic-system return flow.

The first hypothetical phase of the second set of actions specifies the collection of 135,934 ft^3/d from sewering and the disposal of 97,532 ft^3/d (72 percent) of treated wastewater at seven WTFs in the MPA and the remaining 28 percent at unspecified locations outside of the MPA. The second phase specifies additional development including a buildout scenario as well as additional sewering and centralized disposal at WTFs. The net return flow would decrease by 121,581 ft^3/d as a result of sewering, and the net rate of treated-wastewater disposal would increase by 196,198 ft^3/d in the MPA as a result of additional development. This set of wastewater-management actions is realistic because the sewerage locations for untreated wastewater and the disposal sites for treated wastewater, in addition to the volumes of both, are similar to those being considered for the implementation of nitrogen TMDLs for ecological receptors within the MPA (Jeff Gregg, GHD, Inc., written commun., August 15, 2011).

Recharge inputs representing the collection of return flow through sewering and the centralized disposal of treated wastewater were developed for each set of wastewater-management actions by the same approach that was used to produce recharge inputs for current (1997–2001) conditions. The combined volumes of return flow and wastewater from the vector data summarizing wastewater-return flows (including septic systems and WTFs) and the input volumes representing those stresses in the model as enhanced recharge were within 1 percent for current conditions and all phases of wastewater management (fig. 12).

The parcel-scale water data compiled by the MEP (fig. 9A) was used to estimate the nitrogen load to the aquifer associated with each parcel. Septic-system effluent was assumed to have a nitrogen concentration of 35 mg/L as N and the nitrogen loss to be about 25 percent in the septic system and unsaturated zone prior to recharge at the water table (Weiskel and Howes, 1992); the assumed nitrogen concentration of 26.25 mg/L as N at the water table is consistent with the concentration assumed by the MEP in their nitrogen-load estimates (Howes and others, 2004). The area-weighted mean volume of septic-system return flow determined by the rasterization process (fig. 10) and the assumed concentration of 26.25 mg/L as N were used to calculate the nitrogen load associated with the estimated return flow. For current (1997–2001) conditions, the nitrogen-loading rate from septic-system return flow ranged from about 0.0002 to 1.3 kilograms per day (kg/d) (fig. 13) and had the same spatial distribution as the rasterized return flow (fig. 9B). Rasterized return flows were used to estimate parcel-scale nitrogen loads for the second wastewater-management action consisting of partial sewering, disposal at WTFs within the MPA, and additional development (Jeff Gregg, GHD, Inc., written commun., August 15, 2011) (figs. 14A and B).

The nitrogen loads at WTFs were determined by using the specified volume of wastewater disposal to determine nitrogen-loading rates. At the seven existing WTFs (fig. 13), a nitrogen concentration in treated wastewater of 10 mg/L as N was assumed for current conditions and the first phase of wastewater management and a concentration of 3 mg/L for the second phase of wastewater management; nitrogen concentrations in treated wastewater at the two planned WTFs (fig. 14) were assumed to be 3 mg/L for the facility in the Popponesset Bay watershed (the Rock Landing WTF) and 10 mg/L for the facility outside the watershed (not shown on fig. 14) (Jeff Gregg, written commun., GHD, Inc., August 15, 2011). The total recharge onto each model cell (fig. 11) and the nitrogen-loading rate (including both septic-system and WTF components) (figs. 13 and 14) were used to determine the concentration of nitrogen flowing into each model cell. The inflowing concentration was specified in MT3DMS as a specified-concentration boundary condition at the water table (recharge) to simulate the transport of nitrogen through the aquifer to wells and ecological receptors.

A consistent set of groundwater-flow modeling analyses was done for each hydrologic condition representing a wastewater-management condition—current, full sewering, and two phases of partial sewering and disposal in the MPA. A set of regional-model inputs for recharge was developed from the parcel-scale return-flow data, as described above, and used in an unconfined-model simulation with the modified, 27-layer vertical discretization; simulated heads from that simulation were used to convert the regional model to a linearized (confined) model. Results from the confined regional-model simulation were rediscretized to produce model inputs for the subregional model and to develop the specified-head boundary condition linking the regional and subregional models. The subregional model was then run as steady state with the recharge inputs developed for the subregional grid from the parcel-scale return-flow data.

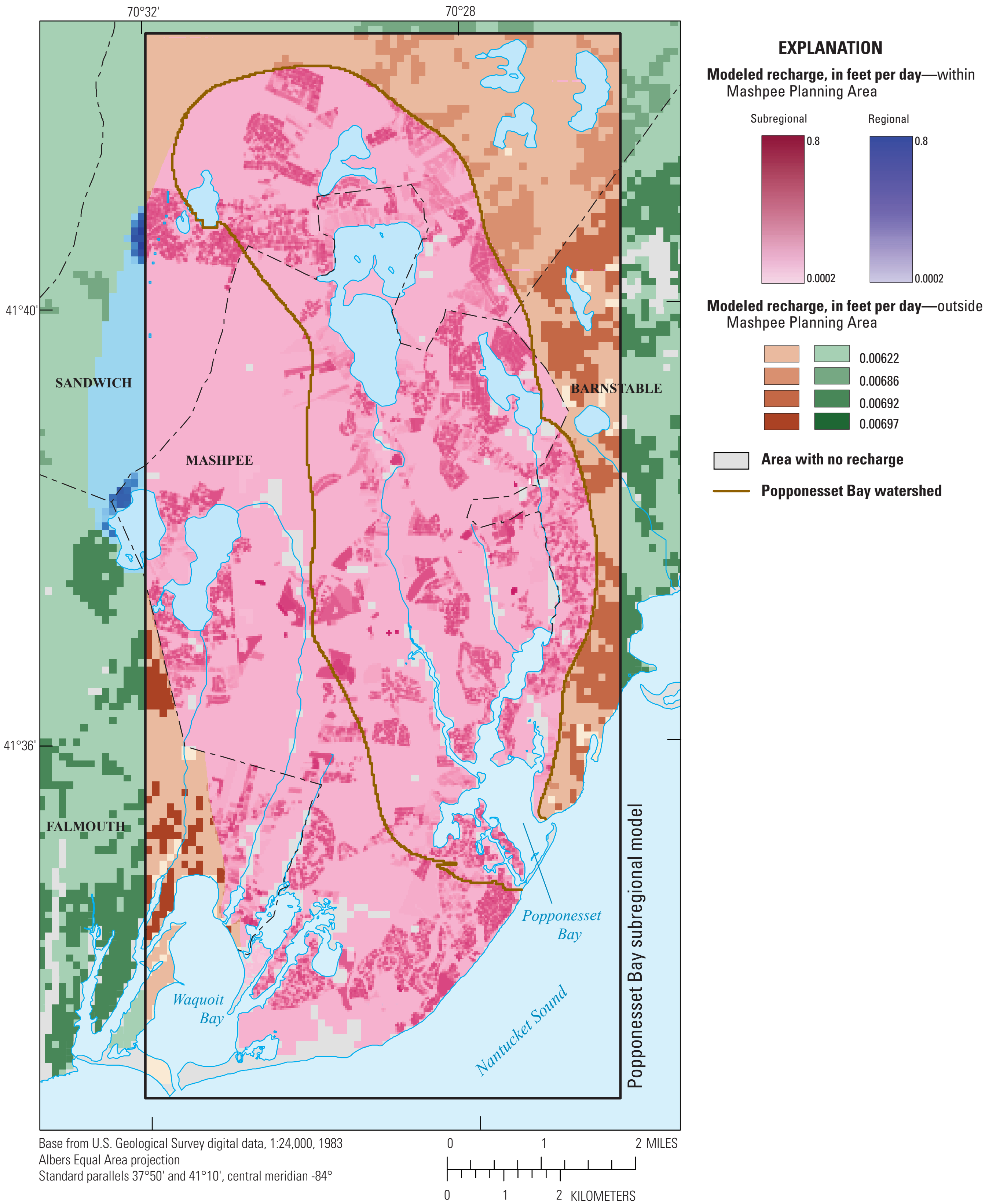


Figure 11. Distribution of simulated current recharge rates representing septic-system and treated-wastewater return flow derived from Massachusetts Estuarine Project data for the Mashpee Planning Area, western Cape Cod, Massachusetts.

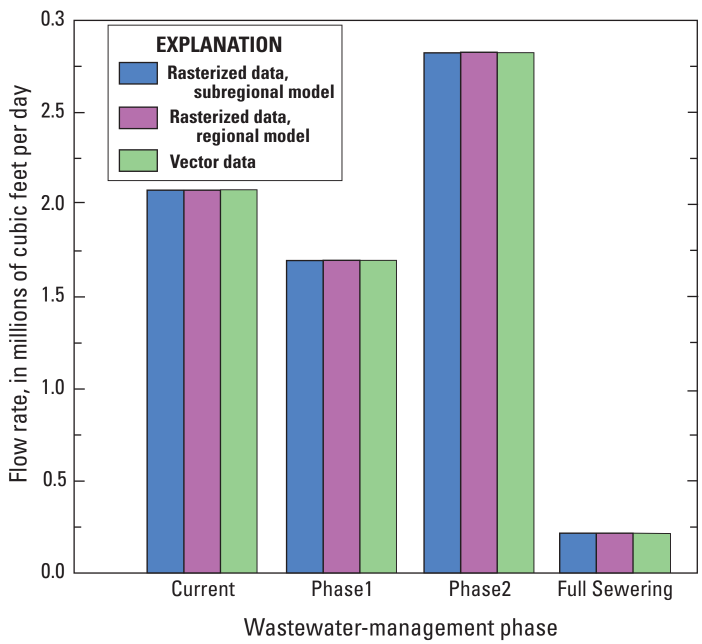


Figure 12. Wastewater return-flow volumes from Mashpee Planning Area vector data and rasterized data used as input to both the regional and subregional models, Mashpee Planning Area, western Cape Cod, Massachusetts.

The steady-state subregional models were run for a stress-period length of 100 years to facilitate the solute-transport simulations. The subregional groundwater-flow model is the basis of the solute-transport simulations, which use cell-by-cell flows from the subregional model. A uniform transport time-step length of 1 day was used for all simulations. The specified-concentration boundary at the water table (representing the recharge of wastewater) for current (1997–2001) conditions (fig. 13) was simulated for a transport period of 100 years starting with an initial condition of no waste-derived nitrogen in the aquifer. Simulations for all wastewater-management actions were also run for 100 years with the corresponding specified recharge concentrations (figs. 14A and B). Scenarios representing full sewering and the first phase of partial sewering and disposal in the MPA included an initial condition representing 30 years of transport under current (1997–2001) nitrogen-loading rates (fig. 14A); the initial condition for the second phase of partial sewering and disposal in the MPA represented 30 years of transport under phase-1 loading rates (fig. 14B).

The use of a 30-year transport time under current loading from surface sources to calculate current nitrogen concentrations in the aquifer and loading to receptors was arbitrary. The 30-year transport time was used because (1) inspection of traveltimes in the watershed suggests that loads after 30 years approach steady state (within about 80 percent), and (2) the 30-year time period generally is consistent with the pace of recent development in the region. It should be noted that although the use of different transport times from current sources in an initially pristine aquifer would result in differing estimates of current (1997–2001) nitrogen loads from those sources, the response times of nitrogen loads at receptors to wastewater-management actions and the effects of simulated attenuation in surface waters—the primary purpose of the analysis—would be similar regardless of the simulated current loads.

Effects of Wastewater-Management Actions on the Hydrologic System

Anthropogenic stresses, such as withdrawal of water from pumped wells and the return of the water to the aquifer as enhanced recharge from septic systems or WTFs, can affect the local hydrologic system. Effects of anthropogenic stresses on the hydrologic system of Cape Cod include changes in water levels in the aquifer, pond stages, and base flow in streams (Walter and Whealan, 2004). Hydraulic gradients in the aquifer are affected (Walter and Masterson, 2004) and, as a result, recharge areas to ecological receptors also can be affected. On average, about 23.7 million gallons per day (Mgal/d; about 3.2 million ft^3/d) of water recharges the aquifer through the Popponesset Bay watershed; about 3 percent of the recharge is from wastewater return flow at septic systems or WTFs. About 70 percent of the water discharges to streams or through pond outflows, and about 27 percent discharges to the Popponesset Bay estuarine system. The remaining 3 percent is withdrawn from wells (table 1).

Water Levels and Streamflow

Wastewater-management actions can affect both water levels and streamflows by altering the distribution of return flow through sewering and disposal at WTFs. Sewering in highly urbanized watersheds overlying a glacial aquifer on Long Island in the 1940s resulted in water-level declines of more than 6 ft and decreases exceeding 65 percent in base flow to streams (Scorca, 1986). Similar hydrologic effects will also likely occur on Cape Cod, although the region generally is much less urbanized, and the effects would be less than those in more populous, urbanized regions. Septic-system return flow and disposal compose about 3.3 percent of total recharge within the MPA (table 1). As a result, wastewater-management actions have a small effect on groundwater fluxes to ecological receptors—ponds, streams, and estuaries—within the Popponesset Bay watershed (fig. 15A). Recharge in the watershed resulting from three wastewater-management actions (full sewering and two sequential phases of partial sewering and disposal at WTFs) is within 3 percent of current

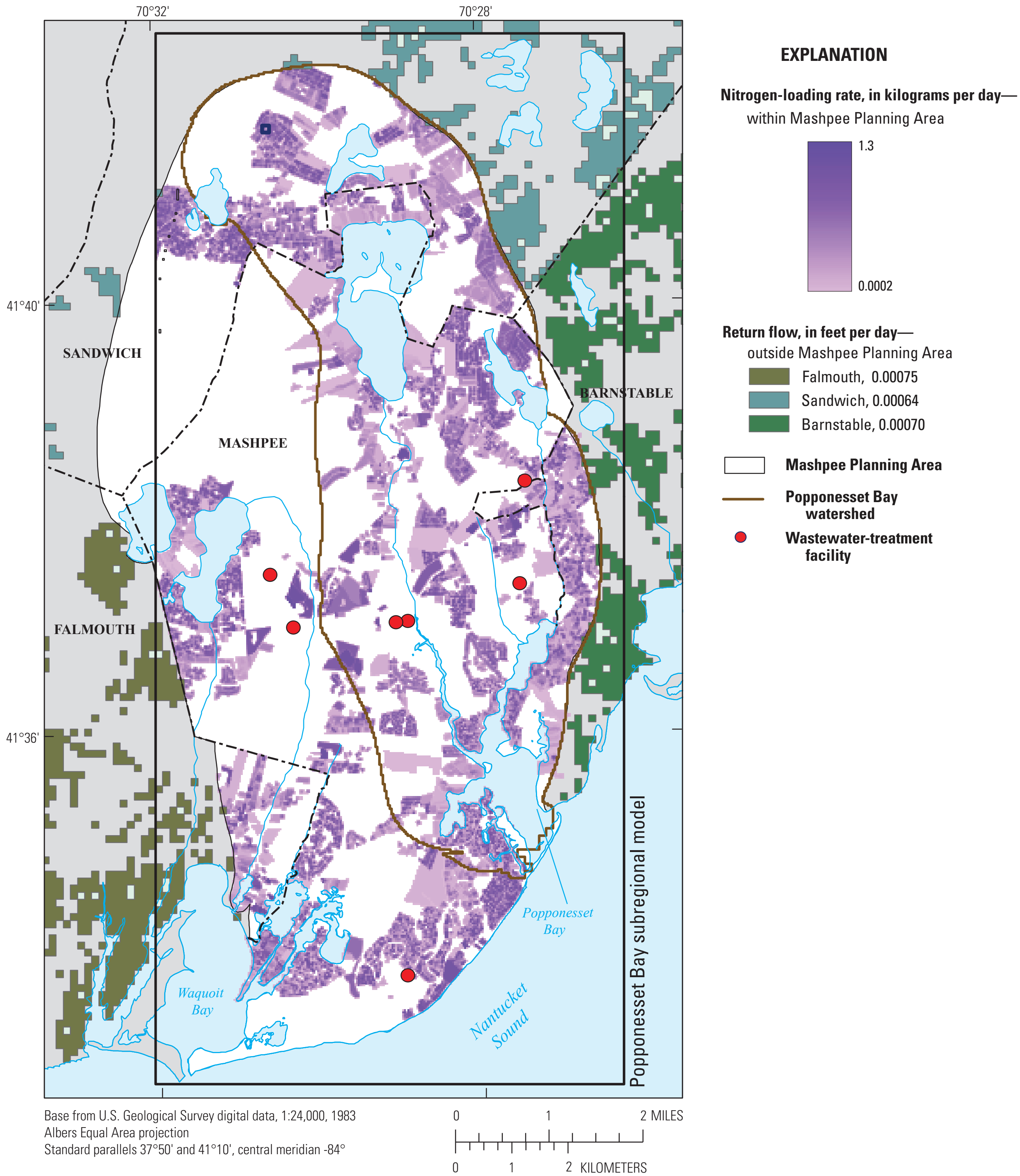
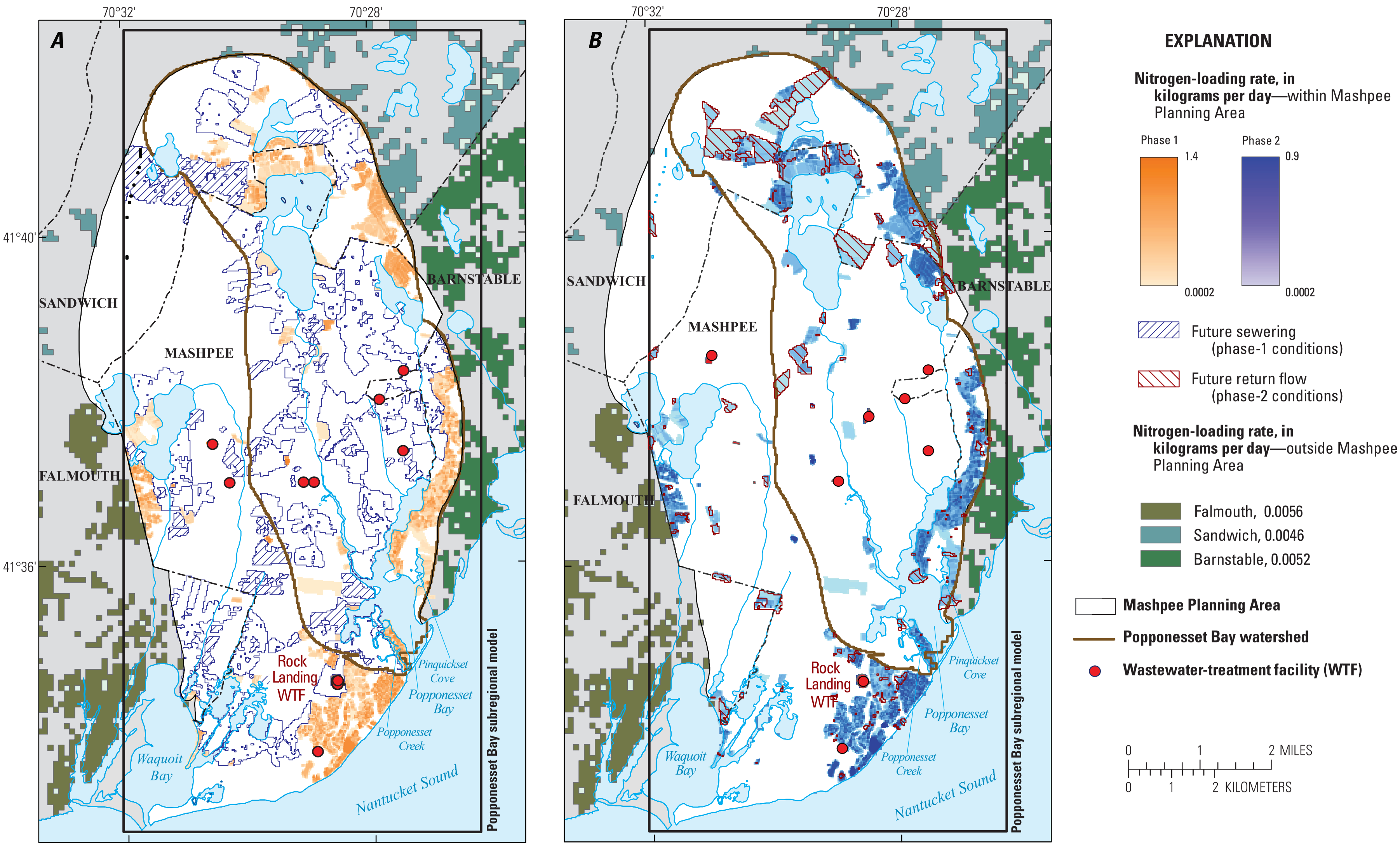


Figure 13. Distribution of nitrogen-loading rates associated with current wastewater return flow in the Mashpee Planning Area, western Cape Cod, Massachusetts.



Base from U.S. Geological Survey digital data, 1:24,000, 1983
Albers Equal Area projection
Standard parallels 37°50' and 41°10', central meridian -84°

Figure 14. Distribution of nitrogen-loading rates associated with wastewater return flow for *A,* the first phase and *B,* the second phase of partial sewering and disposal at wastewater treatment facilities (WTFs) in the Mashpee Planning Area, western Cape Cod, Massachusetts.

Table 1. Simulated hydrologic budgets and septic-system return flow and disposal at wastewater-treatment facilities for current (1997–2001) conditions and potential future conditions resulting from hypothetical wastewater-management actions in the Popponesset Bay watershed and the Mashpee Planning Area, western Cape Cod, Massachusetts.

[All values are in Mgal/d, million gallons per day; WTF, wastewater-treatment facility; --, not applicable]

Components	Current conditions	Wastewater management		Full sewering
		Phase 1	Phase 2	
A. Simulated hydrologic budgets (Mgal/d)				
Popponesset Bay watershed				
Recharge	23.659	23.411	23.750	23.021
Wells	0.624	0.624	0.624	0.624
Stream discharge	14.720	14.619	14.751	14.579
Pond outflow	1.935	1.706	1.755	1.616
Coastal discharge	6.356	6.441	6.598	6.178
B. Wastewater-management action (Mgal/d)				
Popponesset Bay watershed				
Septic-system return flow	0.794	0.203	0.261	0.000
Disposal at WTFs	0.117	0.136	0.379	0.117
Sewering (change from current conditions)	--	-0.591	-0.531	-0.794
Disposal at WTFs (change from current conditions)	--	0.019	0.262	0.000
Mashpee Planning Area				
Septic-system return flow	1.391	0.378	0.484	0.000
Disposal at WTFs	0.163	0.892	1.632	0.163
Sewering (change from current conditions)	0.000	-1.013	-0.907	-1.391
Disposal at WTFs (change from current conditions)	0.000	0.729	1.469	0.000

recharge for current conditions (fig. 15A); groundwater discharges to ponds, streams, and estuaries are within 2.9, 1.6, and 2.4 percent of current conditions, respectively, for the three wastewater-management actions (fig. 15A). Groundwater discharge to all major receptors decreases after full sewering and disposal outside of the MPA (fig. 15A). For sequential phases of partial sewering and disposal at WTFs, groundwater discharge to streams decreases from sewering, whereas groundwater discharge to estuaries increases through disposal at WTFs (fig. 15A), particularly disposal at the proposed Rock Landing site (figs. 14A, B).

Groundwater discharge decreases to all individual ecological receptors (streams and estuaries) following full sewering (fig. 15B) because all collected return flow is transferred outside of the watershed. Groundwater discharge to streams decreases slightly (less than 4 percent) following two sequential phases of partial sewering and disposal at WTFs (fig. 15B). Changes in groundwater discharge to individual estuaries vary following the two phases of wastewater-management actions. Groundwater discharge to Popponesset Bay, Popponesset Creek, and Ockway Bay increased by 8.4, 28.0, and 14.8 percent, respectively, because of the proximity of the Rock Landing site, whereas discharge to Pinquickset Cove was almost unchanged (less than 0.2 percent) because the distribution of return flow within the watershed generally was unchanged (figs. 14A and B).

Full sewering and the removal of collected return flow from the watershed results in a small—about 3 percent—decrease in recharge within the MPA. As a result, water-table altitudes and hydraulic gradients are essentially the same as those for current conditions in most of the watershed (fig. 16A). The decrease in water-table altitudes (drawdown) exceeds 0.2 ft around Snake Pond near the top of the water-table mound (fig. 16B). The mean drawdown from current conditions is 0.07 ft and exceeds 0.1 and 0.2 ft in 39.1 and 4.3 percent of the MPA, respectively.

Total recharge within the MPA differs by less than 1 percent between current conditions and conditions following the two sequential phases of partial sewering and disposal at WTFs and, as a result, water-table altitudes remain similar to current conditions in most areas (fig. 16A). Water-table altitudes and hydraulic gradients differ the most around an area of potential disposal at the Rock Landing site, to the west of Popponesset Bay, where most of the treated wastewater

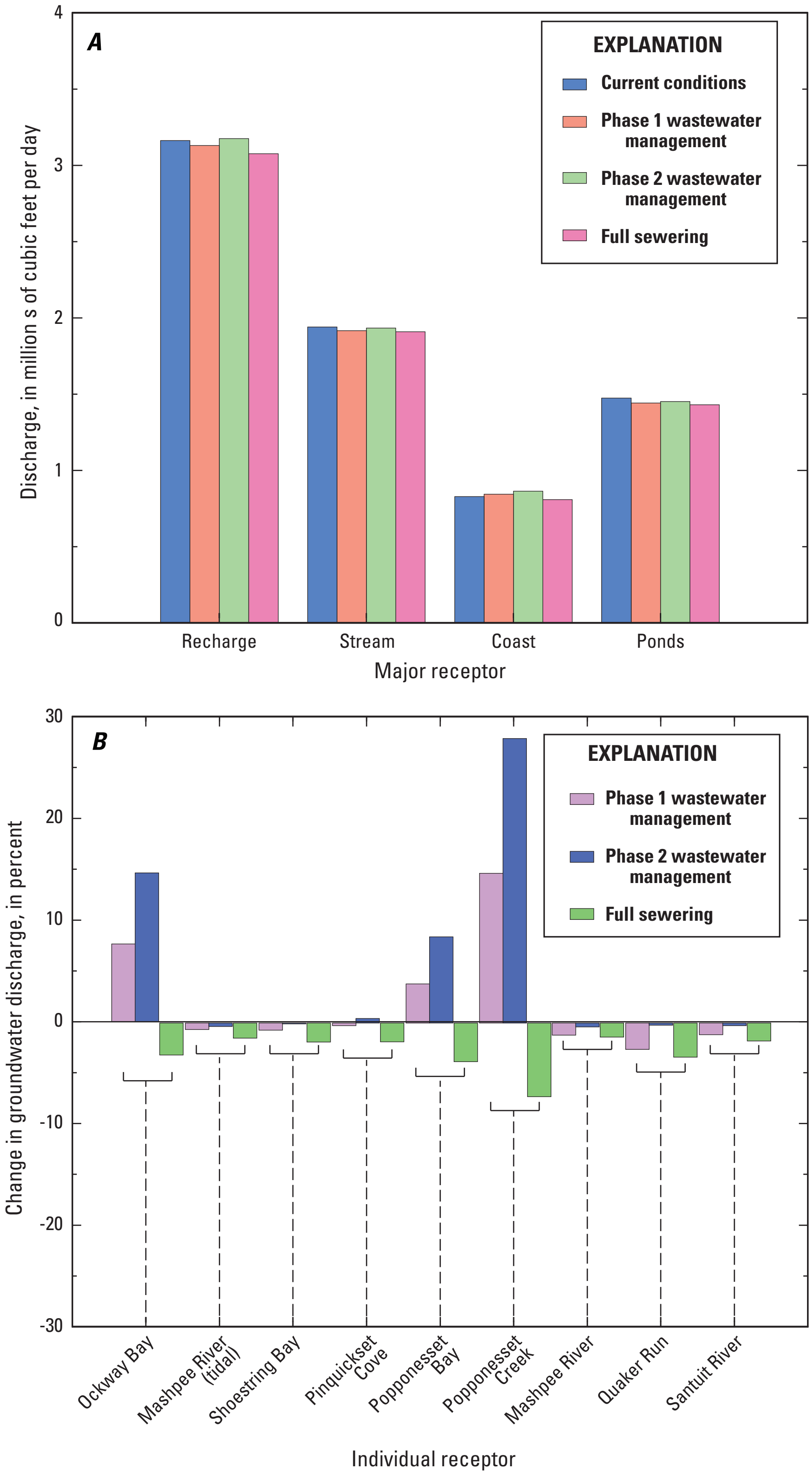


Figure 15. *A,* Groundwater-discharge to major receptors for current wastewater conditions and three simulated wastewater-management actions, and *B,* changes in groundwater discharge from current conditions for the three simulated wastewater-management actions, Popponesset Bay watershed, western Cape Cod.

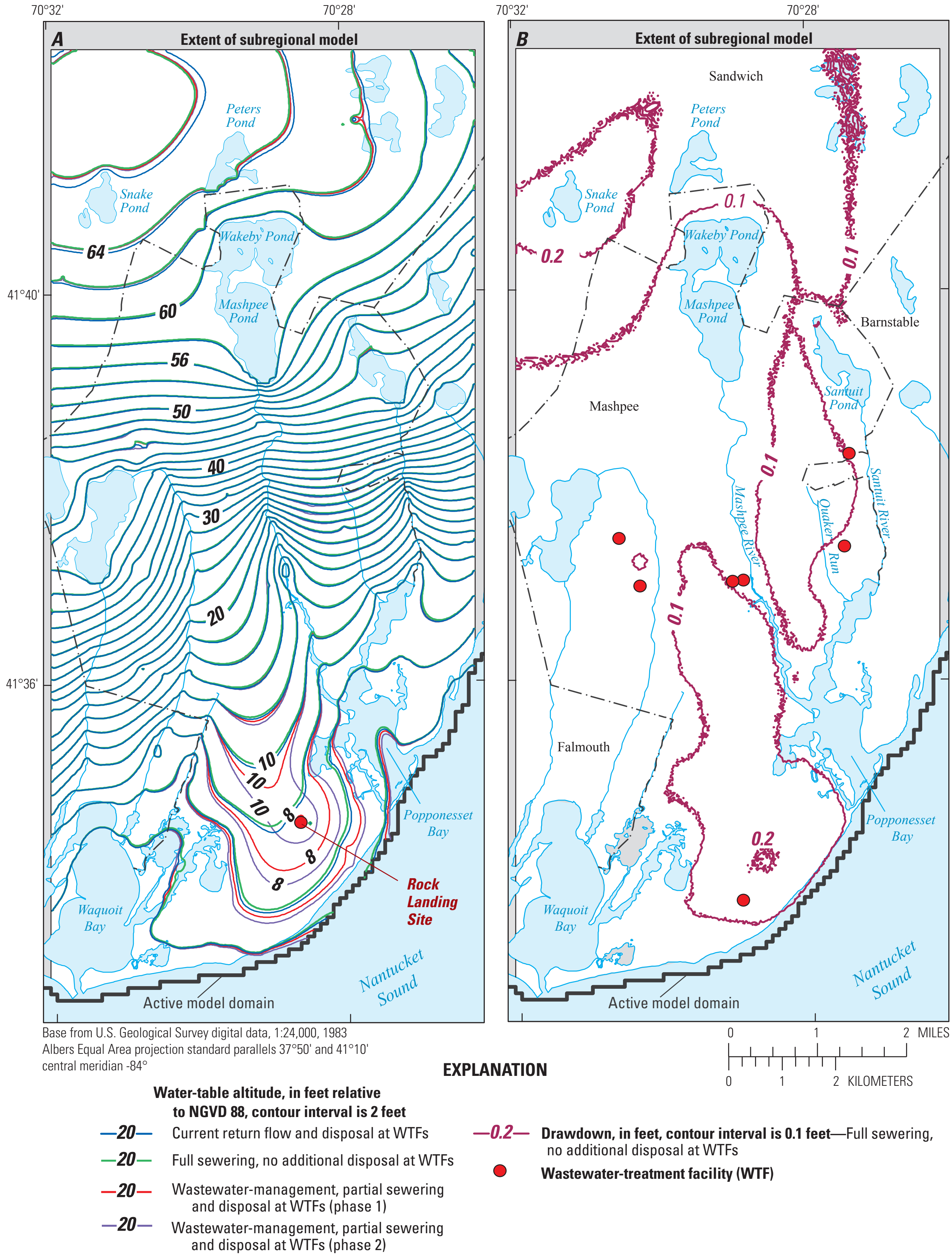


Figure 16. *A,* Water-table altitude for current conditions and potential future conditions resulting from three wastewater-management actions, and *B,* drawdown resulting from full sewering within the Mashpee Planning Area, western Cape Cod, Massachusetts.

is discharged in this hypothetical scenario (fig. 16A). The two phases of wastewater management result in drawdown in newly sewered areas and increases in water-table altitudes (mounding) near WTFs (fig. 17). After the first phase of partial sewering and disposal at WTFs, water-table altitudes decrease as a result of sewering across 79.7 percent of the MPA with a maximum decrease of 0.21 ft near Snake Pond (fig. 17A). Water-table altitudes increase near WTFs across 13.7 percent of the MPA; mounding exceeds 3.2 ft at the Rock Landing site (fig. 17A). After the second phase of partial sewering and disposal at WTFs, water-table altitudes decrease across 14.8 percent of the MPA with a maximum decrease of more than 0.7 ft (fig. 17B). Water-table altitudes increase over 24.1 percent of the MPA near WTFs; the largest increase exceeds 5.4 ft near the Rock Landing site (fig. 17B). Water-table altitudes in the remaining areas are essentially unchanged.

Recharge Areas to Receptors

Watershed boundaries in areas dominated by surface-water drainages, such as those provided by valley-fill aquifers, are determined by topography and generally do not change in response to changing hydrologic conditions. In groundwater-dominated systems, watersheds are determined by the configuration of the water table, and the resultant hydraulic gradients can change in response to changes in hydrologic conditions. Anthropogenic stresses, such as pumping at wells and return flow at septic systems or WTFs, can alter water-table altitudes and hydraulic gradients (fig. 16A) and, therefore, can potentially change the recharge areas to ecological receptors.

Septic-system return flow composes about 3 percent of total recharge within the MPA. Removal of return flow through full sewering results in water-table altitudes and hydraulic gradients that are similar to those for current conditions; as a result, recharge areas to wells and ecological receptors are essentially the same for the two hydrologic conditions (fig. 18A). The two sequential phases of partial sewering and disposal at WTFs change recharge by less than 1 percent from current conditions (fig. 15A) and, as a result, water-table altitudes and hydraulic gradients are similar to those for current conditions in most areas; an exception is the area near the Rock Landing site (fig. 18B), where most of the collected return flow is treated and discharged. The recharge areas to ponds, streams, and wells are essentially the same for current conditions and conditions following the two sequential phases of wastewater management. The recharge areas to coastal receptors differ between the two conditions to the west of Popponesset Bay, where gradients are affected by disposal at the Rock Landing site (fig. 18B), but otherwise, are the same for current and future conditions.

The recharge areas to Ockway Bay and Popponesset Creek are the most affected by discharge at the Rock Landing site and the resulting change in hydraulic gradients (fig. 19). The recharge areas to the two receptors increased by 19 and 21 percent, respectively, following the two phases of wastewater-management actions. The recharge areas extend farther to the west toward the Rock Landing site than the recharge areas for current conditions. The recharge area to Ockway Bay expands about 2,000 ft farther to the south, and the recharge area to Popponesset Creek expands about 1,200 ft farther to the southwest, as a result of disposal at the Rock Landing site (fig. 19). The larger recharge areas are consistent with the increases in groundwater discharge to the two receptors of 14.8 and 28.0 percent, respectively (fig. 15B). The groundwater-discharge rate to a receptor is a function of the size of the recharge area and recharge rates within the area. The results indicate that, given the volume of redistributed return flow relative to natural recharge to the aquifer, the removal of return flow through sewering does not greatly affect hydraulic gradients and recharge areas, whereas disposal at WTFs can affect hydraulic gradients and recharge areas—the effects are functions of the volume of disposal, hydrologic setting, and proximity of the discharge to a recharge-area boundary.

Pumping from wells for water supply represents another hydraulic stress that can affect hydraulic gradients and potentially recharge areas to ecological receptors. The Upper Cape Cooperative (UCC) wells are on the MMR in Sandwich to the north of the Popponesset Bay watershed (fig. 6A); the wells are an auxiliary source of water to communities near the MMR during dry periods and were pumped at a rate of 1.7 Mgal/d in 2003 (Walter and Whealan, 2004). Operation of the UCC wells, which are near the top of the water table mound, decreases water levels in the upper part of the Popponesset Bay watershed by about 1.5 ft, changes hydraulic gradients, and shifts the regional groundwater divide as much as 1,000 ft to the south (fig. 20). As a result, the northern extent of the Popponesset Bay watershed shifts southward, and the recharge area to the ponds in the northern part of the watershed decreases in size by about 8 percent (fig. 20). Groundwater fluxes to ponds, which are functions of the sizes of the recharge areas, decrease by about 11 percent. The recharge areas to streams and estuaries in the watershed generally are unaffected by operation of the UCC wells.

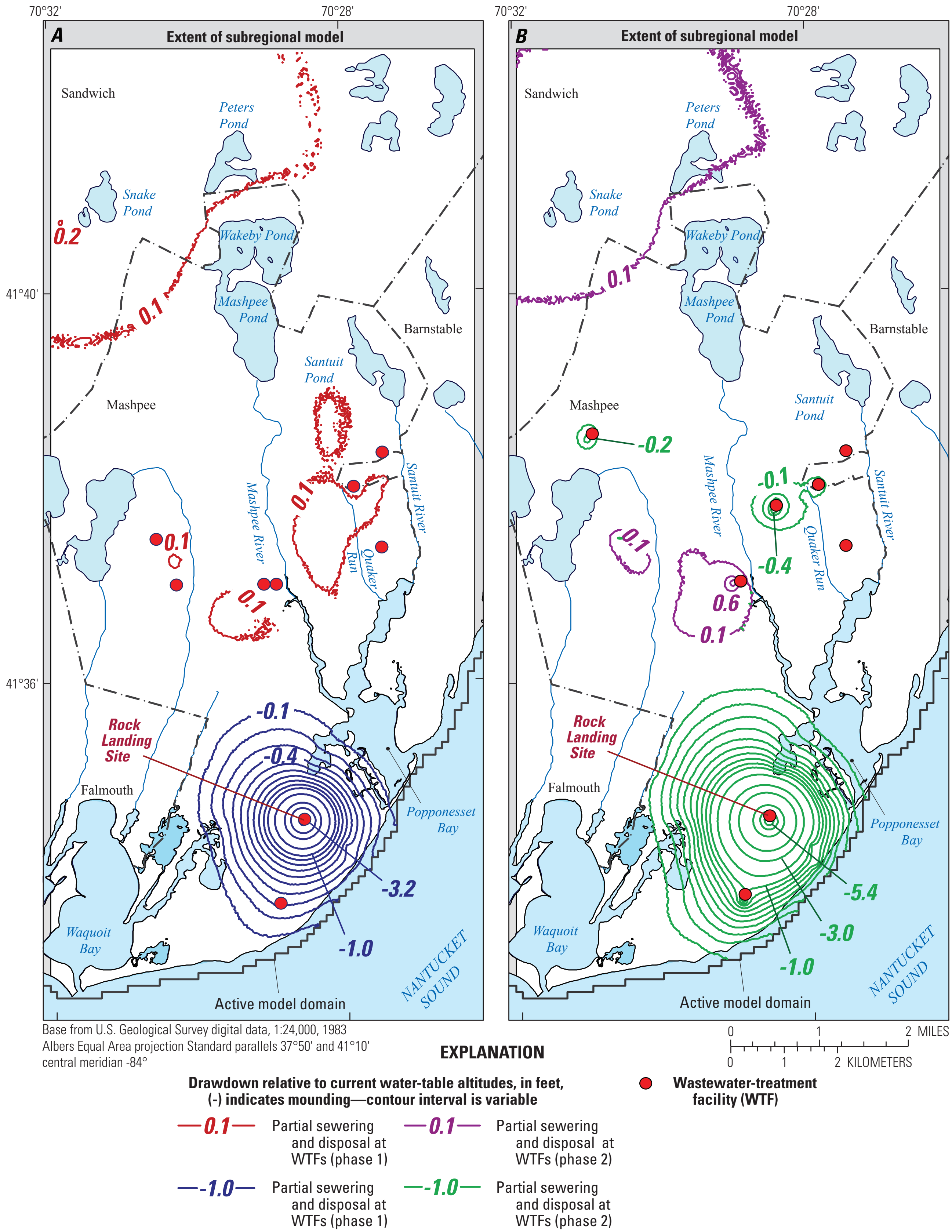


Figure 17. Drawdown resulting from *A,* the first phase and *B,* the second phase of partial sewering and disposal at wastewater treatment facilities (WTFs) within the Mashpee Planning Area, western Cape Cod, Massachusetts.

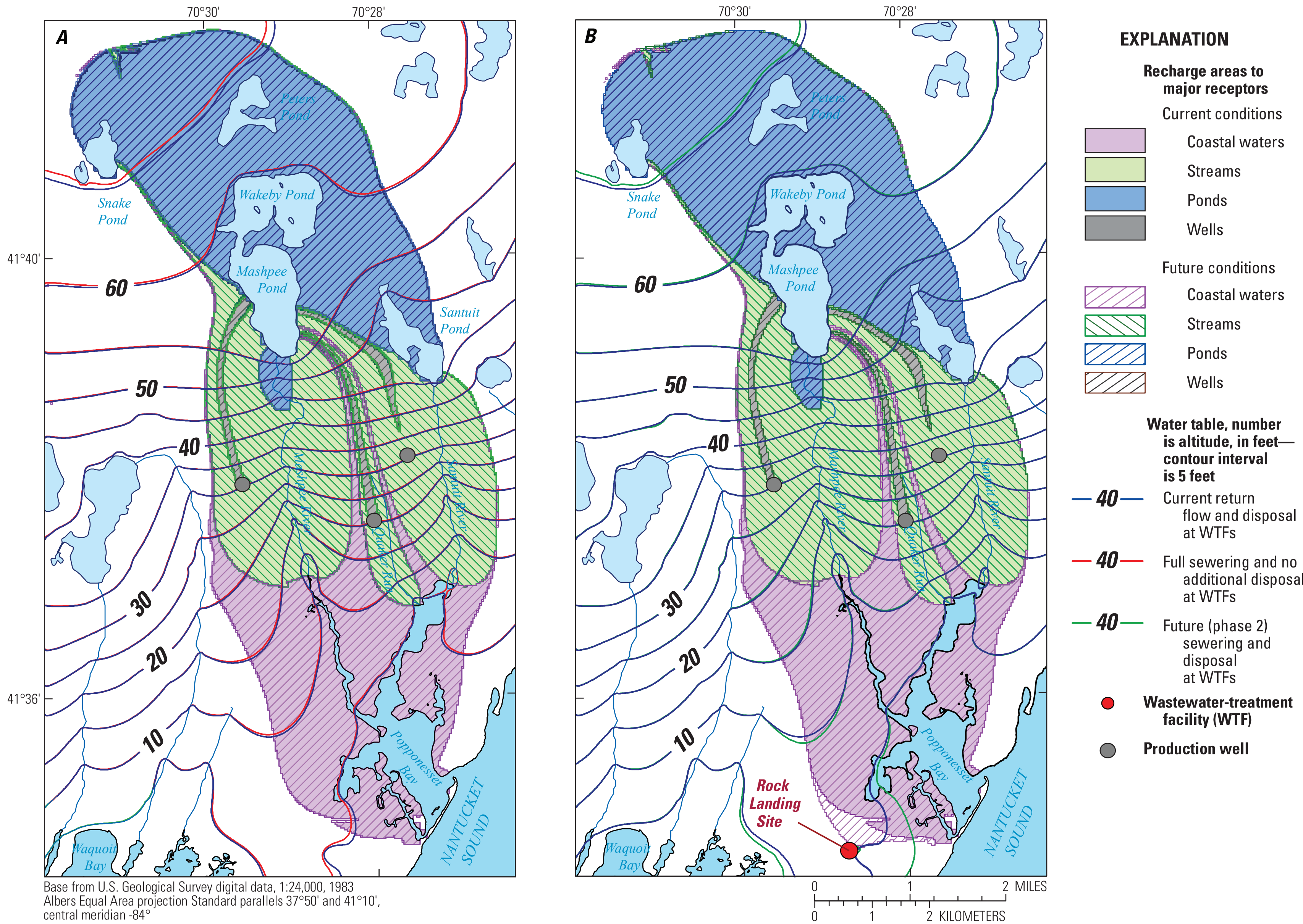


Figure 18. Recharge areas to wells and ecological receptors for current conditions and *A,* full sewering and *B,* two sequential phases of wastewater management, Popponesset Bay watershed, western Cape Cod, Massachusetts.

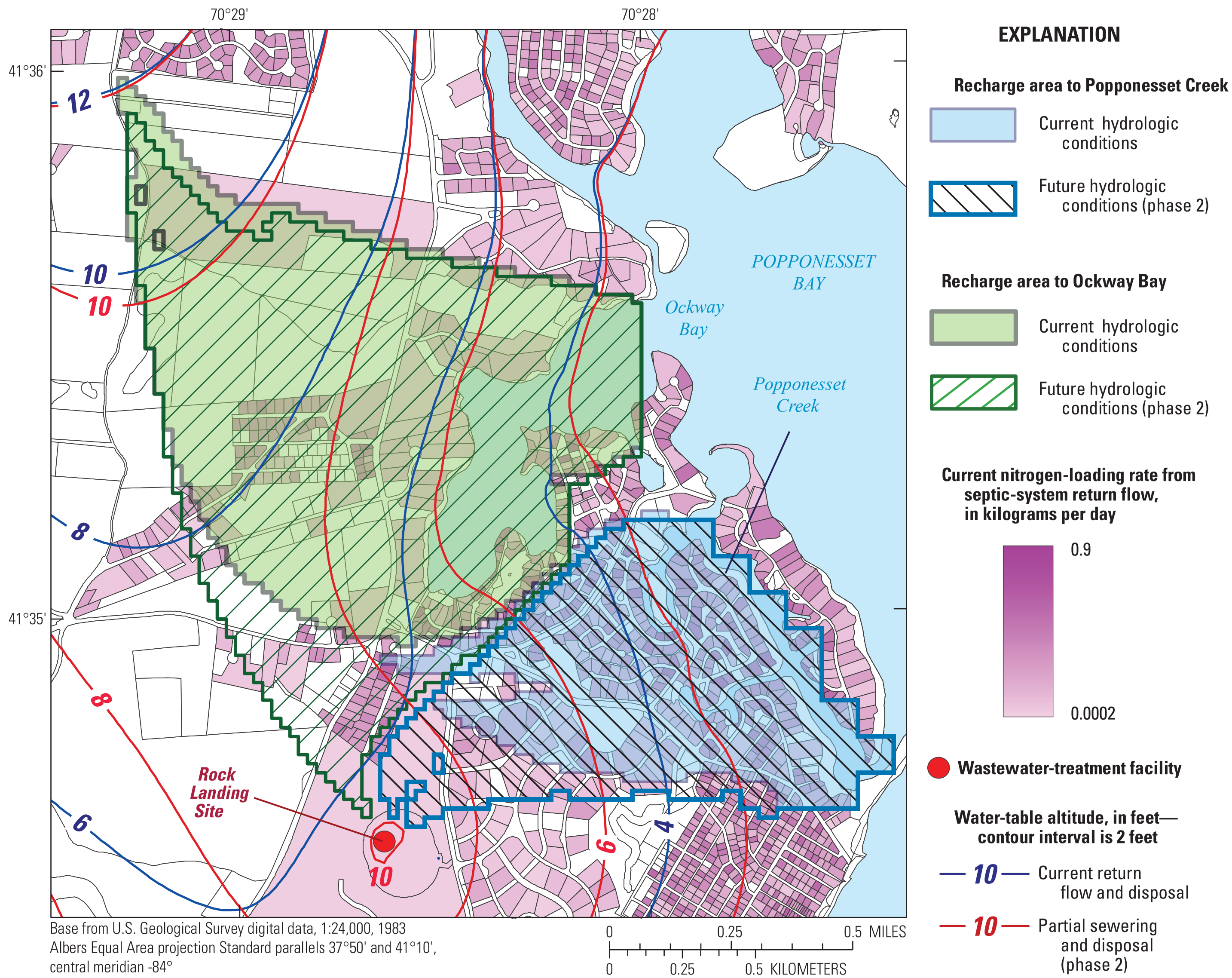


Figure 19. Recharge areas to Popponesset Creek and Ockway Bay for current hydrologic conditions and potential future conditions resulting from two phases of wastewater management following partial sewering and disposal, and current nitrogen-loading rates from parcels within the Mashpee Planning Area, western Cape Cod, Massachusetts.

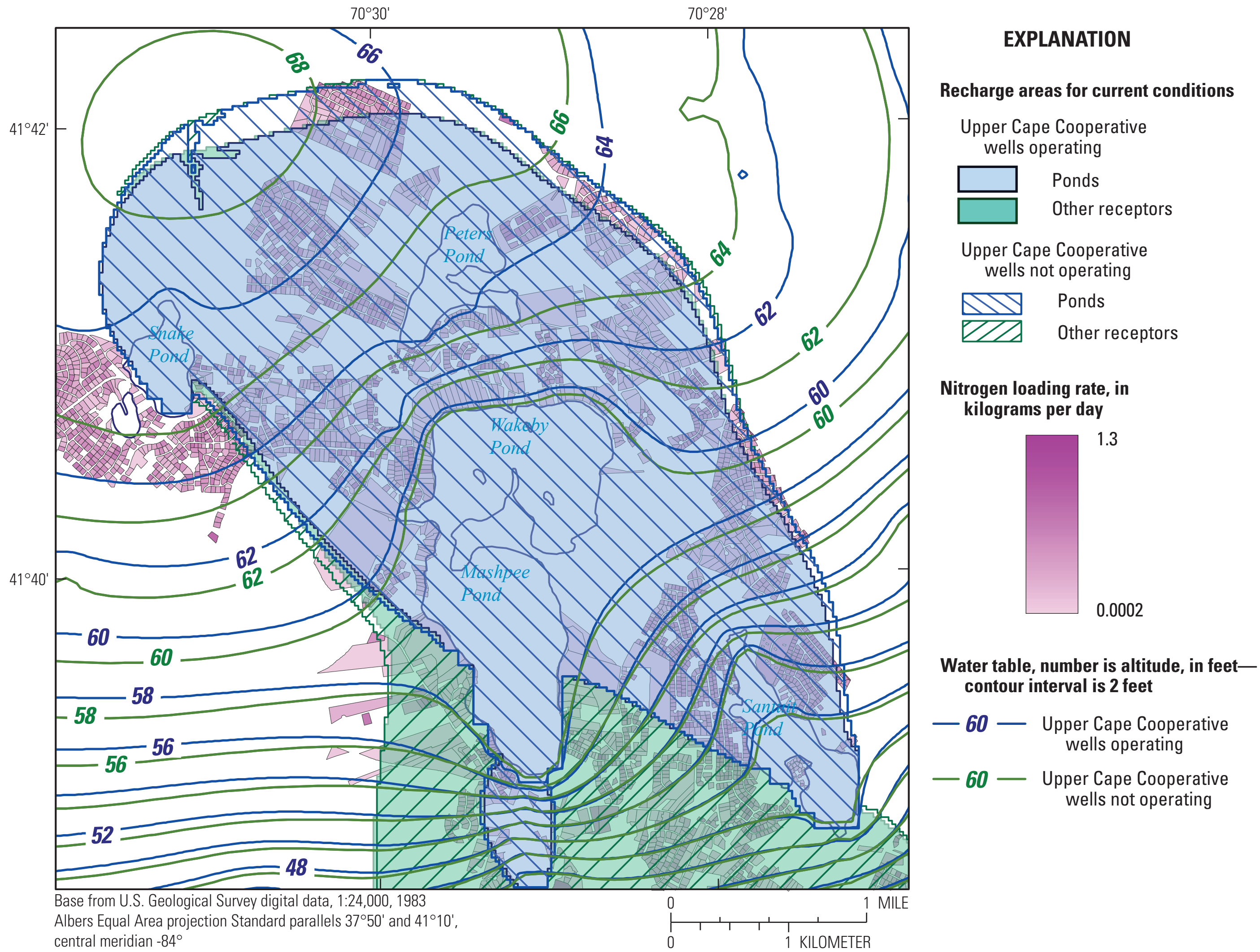


Figure 20. Recharge areas to ponds in the Popponesset Bay watershed with operation of the Upper Cape Cooperative wells (current conditions) and without operation of the wells (potential future conditions), western Cape Cod, Massachusetts.

Effects of Wastewater-Management Actions on Nitrogen Loads to Receptors

Daily loads of waste-derived nitrogen to ecological receptors can be determined in two ways: (1) implicitly by using model-delineated recharge areas and vector data on nitrogen sources and (2) explicitly by using a solute-transport model that represents the same sources as inflowing concentrations at the water table. Nitrogen loads estimated from recharge areas are referred to as "instantaneous" loads because loads from sources at the land surface (recharging at the water table) are assumed to be equal to those discharging to the receptor at the same time. The method does not take into account subsurface transport times or changes in recharge areas in response to hydrologic stresses.

Nitrogen Loads Estimated by Using Recharge Areas

Estimates of waste-derived nitrogen loads obtained by using recharge areas involves three steps: (1) identification of individual ecological receptors within an estuarine system, (2) delineations of recharge areas to those receptors, (3) mapping of delineated recharge areas to spatial data regarding sources of nitrogen, and (4) summation of loads from individual sources in the recharge areas. The identification of individual estuarine receptors typically is based on a set of hydrodynamic, water-quality, and ecological criteria; recharge areas to those identified receptors are determined by using a groundwater-flow model and particle tracking as discussed in the section entitled "Simulation of Advective Transport." Recharge areas are mapped to spatial (vector) source data, and the loads are estimated by using a GIS.

Recharge Areas and Traveltimes

The Popponesset Bay system is separated into six separate estuaries—Popponesset Creek, Ockway Bay, the tidal portion of the Mashpee River, Shoestring Bay, Pinquickset Cove, and Popponesset Bay—and three freshwater streams—Mashpee River, Quaker Run, and Santuit River (fig. 21). The sizes of the individual recharge areas to the estuaries range from 0.35 mi^2 for Popponesset Creek to 1.83 mi^2 for the tidal portion of the Mashpee River; recharge areas to streams range in size from 0.60 mi^2 for Quaker Run to 8.11 mi^2 for the Mashpee River (fig. 21).

Traveltimes within the Popponesset Bay watershed range from essentially instantaneous at discharge boundaries to more than 100 years, but generally traveltimes from the water table to the receptors in most parts of the watershed are small (fig. 22). The median traveltime to all receptors (including ponds) is about 4.5 years; the median traveltimes to estuaries, streams, and ponds are 5.7, 3.7, and 4.5 years, respectively (fig. 23). Traveltimes from the water table to a receptor can be divided into specific intervals to allow nitrogen loads to be estimated for these specific periods of advective transport. As an example, nitrogen loads in MEP analyses are differentiated for areas with traveltimes of greater than or equal to and less than 10 years. In the Popponesset Bay watershed, 63.4 percent of the traveltimes to estuaries are less than 10 years; 76.4 and 65.8 percent of the traveltimes to streams and ponds, respectively, are less than 10 years (fig. 22). Traveltimes are analogous to advective-transport times; nitrogen transported conservatively from sources in most areas reaches a receptor within 10 years. Recharged water that flows deeper into the system can require longer traveltimes to reach a receptor. Traveltimes exceed 100 years from 8.9, 2.8, and 3.3 percent of the recharge areas to estuaries, streams, and ponds, respectively (fig. 23). The percentage of long traveltimes from the recharge areas to estuaries is largest (fig. 23) because estuaries, as the most downgradient component of the flow system, generally receive the deepest (and oldest) recharged water. This water comprises water recharged close to the regional groundwater divide, near the edge of the watershed boundary, and between recharge areas to streams (fig. 22).

Estimates of Nitrogen Loads for Current and Future Wastewater Conditions

Model-delineated recharge areas and traveltimes can be used in conjunction with spatial data on nitrogen sources to estimate daily nitrogen loads by adding loads from individual sources within an entire recharge area or within specific ranges of traveltimes. The recharge area to the tidal portion of the Mashpee River is the largest area of direct groundwater discharge to an estuary within the Popponesset Bay watershed. Traveltimes from the water table to this receptor range from essentially instantaneous at the estuary to more than 100 years from near the western edge of the recharge area. Current (1997–2001) nitrogen loads in septic-system return flow from individual parcels range from 0.0002 to 0.8 kg/d (fig. 24). The total load of waste-derived nitrogen to the tidal Mashpee River is 8.9 kg/d, determined by summation of the loads from the nitrogen sources within the recharge area by a GIS. Traveltimes from the water table in about 62.8 percent of the recharge area to the estuary are 10 years or less (fig. 24). Of the total waste-derived nitrogen load to the estuary, about 56 percent originates within an area with a traveltime of 10 years or less to the estuary. It should be noted that the estuary also receives a load of waste-derived nitrogen from surface-water inflow from the Mashpee River (fig. 24).

Loads of waste-derived nitrogen vary among receptors as functions of the sizes and the unique distributions of nitrogen sources within the recharge area to the receptor. For current (1997–2001) conditions, the largest loads of waste-derived nitrogen to estuaries are for the tidal portion of the Mashpee River and Shoestring Bay (fig. 25), which also have the two

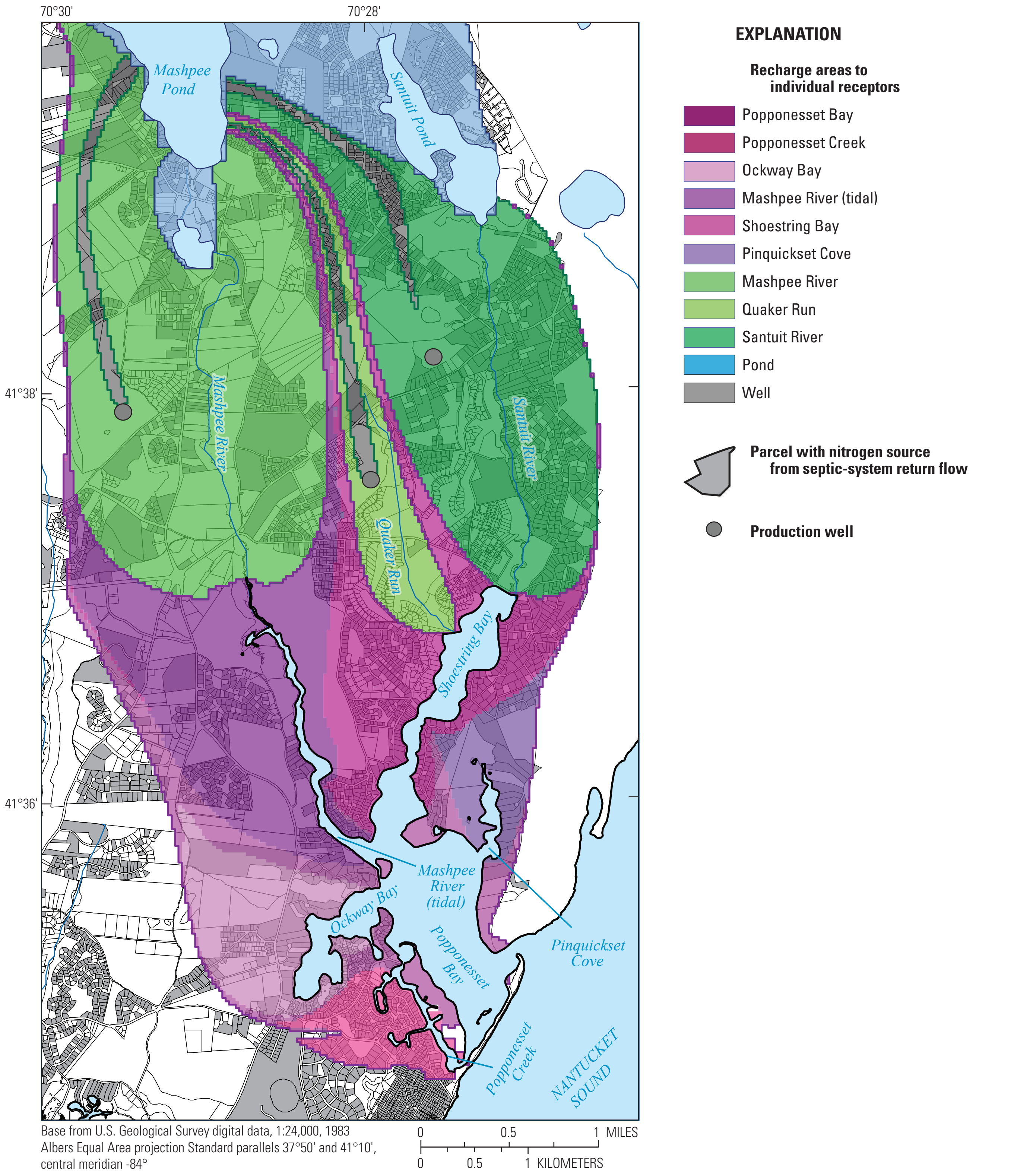


Figure 21. Distribution of recharge areas to wells and individual ecological receptors—streams and estuaries—and parcels containing sources of septic-system return flow, Popponesset Bay watershed, western Cape Cod, Massachusetts.

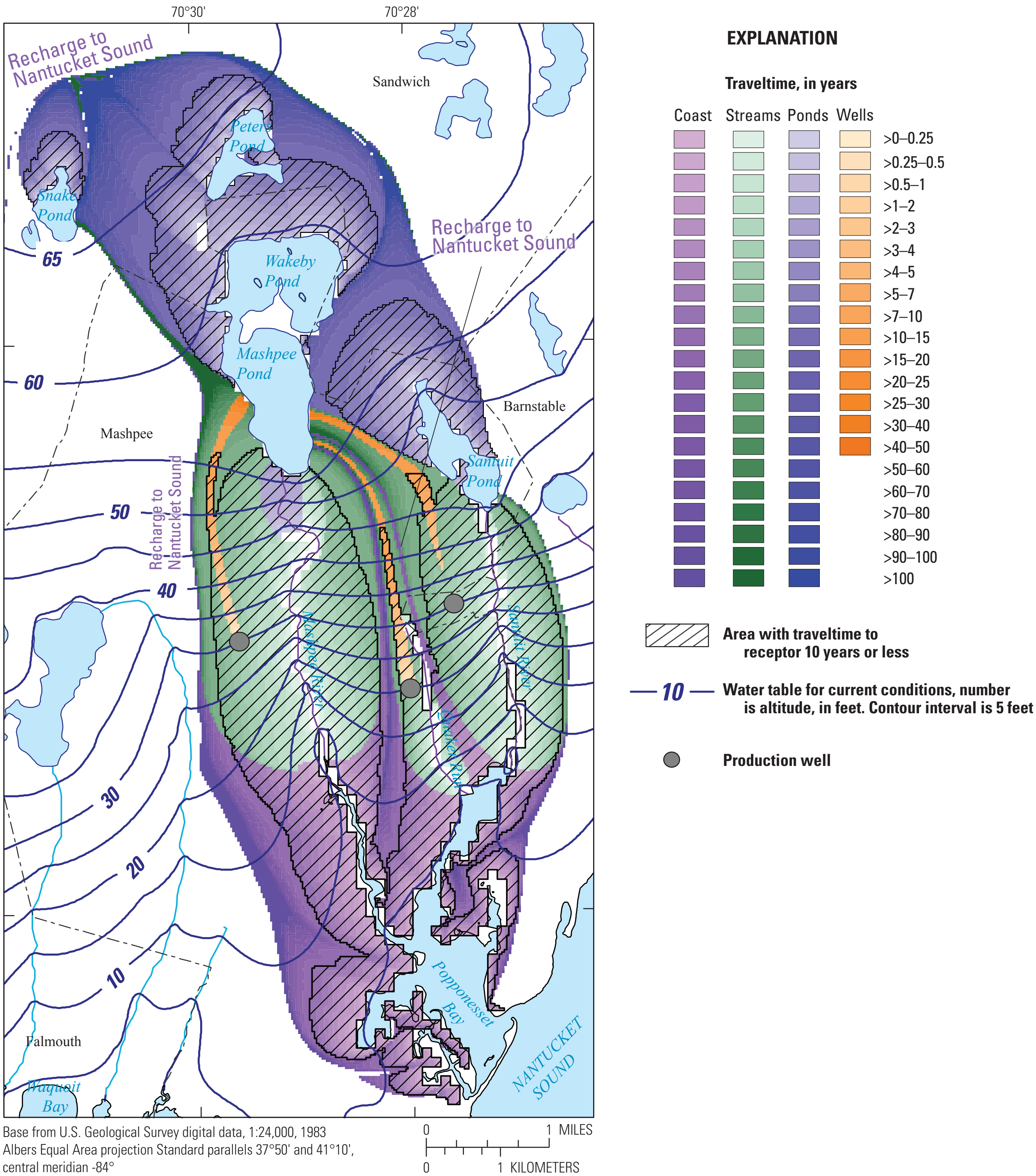


Figure 22. Traveltimes to wells and ecological receptors within the Popponesset Bay watershed for current conditions, Popponesset Bay watershed, western Cape Cod, Massachusetts.

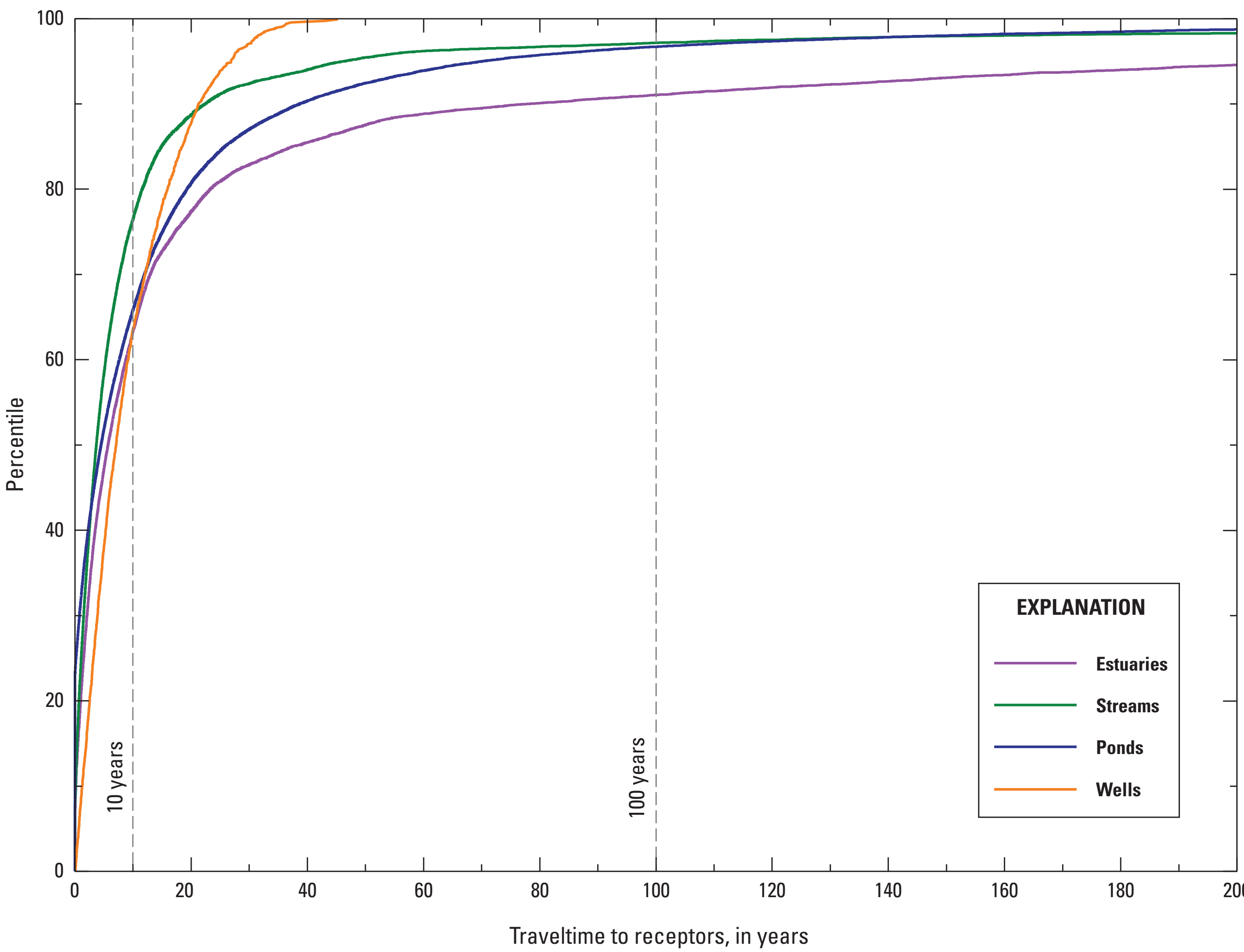


Figure 23. Distribution of traveltimes from the water table to wells and ecological receptors, Popponesset Bay watershed, western Cape Cod, Massachusetts.

largest recharge areas (fig. 21); both estuaries also receive an additional nitrogen load from surface-water inflows. The load of waste-derived nitrogen for traveltimes less than 10 years to Shoestring Bay is larger, despite the smaller recharge area, because of intense development in the recharge area (fig. 21). The smallest load of waste-derived nitrogen for traveltimes less than 10 years is to Pinquickset Cove, which has the smallest recharge area and relatively sparse residential development (fig. 21). The largest load of waste-derived nitrogen to a freshwater stream for traveltimes less than 10 years is to the Santuit River (fig. 25). Although the recharge area to the Santuit River is about 40 percent smaller than the recharge area to the Mashpee River, there are more sources of waste-derived nitrogen within its recharge area (fig. 21).

The total load of nitrogen to the estuary derives from all of the surficial sources within the watershed and is referred to as the “instantaneous” load in MEP analyses. The portion of the total nitrogen load that discharges within a certain period of transport is a function of the distribution of traveltimes and nitrogen sources within the recharge area. Most of the total load of waste-derived nitrogen travels from the water table to an estuary within 10 years (fig. 25) because most of the recharge areas are within a 10-year traveltime to a receptor (fig. 22). Also, residential development generally is densest nearer to the coast (fig. 21). The portion of nitrogen loads arriving within 10 years of traveltime varies among individual estuaries. Loads from about 70 percent of the recharge area to Popponesset Creek arrive within a 10-year traveltime, but more than 90 percent of the total load of waste-derived nitrogen to the estuary arrives within a traveltime of 10 years (fig. 25). Most of the load of waste-derived nitrogen to Pinquickset Cove—about 67 percent—is from source areas with traveltimes greater than 10 years (fig. 25) because of the large undeveloped areas near the shore (fig. 21). About 84 percent of the total load of waste-derived nitrogen arrives at the Santuit River within a traveltime of 10 years or less. Only

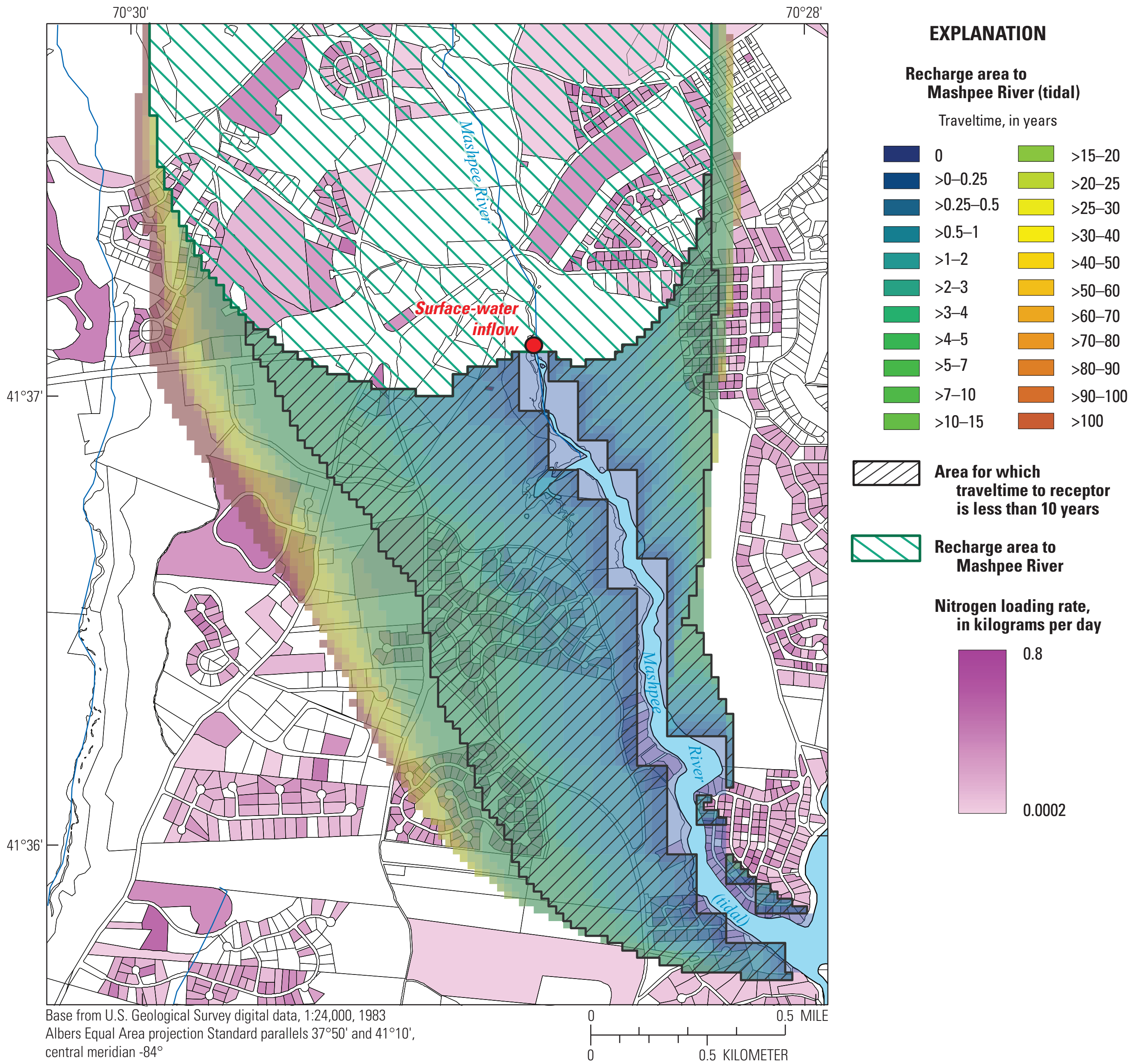


Figure 24. Distribution of traveltimes and nitrogen-loading rates within the recharge area to the tidal portion of the Mashpee River for current wastewater conditions, Popponesset Bay Watershed, western Cape Cod, Massachusetts.

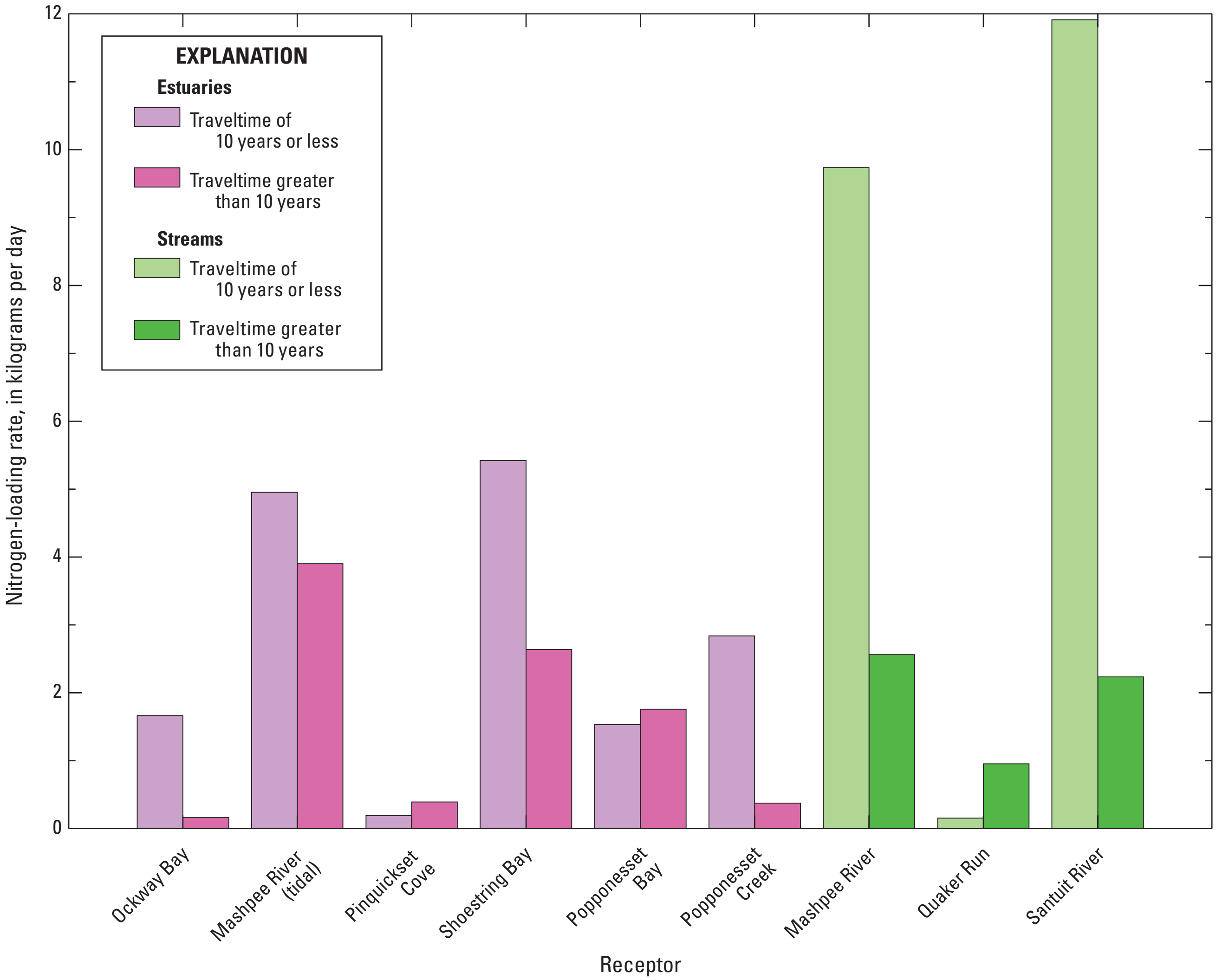


Figure 25. Nitrogen-loading rates from recharge areas to individual ecological receptors—streams and estuaries—estimated for current (1997–2001) conditions, Popponesset Bay watershed, western Cape Cod, Massachusetts.

about 13.9 percent of the total load to Quaker Run is from areas with traveltimes of 10 years (fig. 25) because of the large amount of undeveloped land near the stream (fig. 21).

The same methods were used to estimate loads resulting from hydrologic stresses caused by a set of wastewater-management actions by using spatial data on potential future sources of waste-derived nitrogen in recharge areas. Full sewering within the MPA resulted in a 94.7-percent decrease in the total load of waste-derived nitrogen to wells and ecological receptors (fig. 26); the remaining nitrogen load was from existing WTFs. Full sewering resulted in no load of waste-derived nitrogen to wells and ponds because septic-system return flow, which would be completely removed by the action, was the only source of waste-derived nitrogen in recharge areas to those receptors. Loads to estuaries and streams decreased by about 90.2 and 93.0 percent, respectively, as a result of full sewering. The first phase of partial sewering and disposal at WTFs resulted in a 70.2-percent decrease in the total load of waste-derived nitrogen to receptors; loads of waste-derived nitrogen to estuaries decreased the most, by about 67.0 percent. Waste-derived nitrogen loads to streams and ponds decreased by 77.1 and 63.1 percent, respectively (fig. 26). The second phase of wastewater management resulted in a 22.0-percent increase relative to the first phase in total loads of waste-derived nitrogen to receptors because of the additional residential development within the watershed. Nitrogen loads to estuaries increased by about 10.3 percent; nitrogen loads to streams and ponds increased by 24.4 and 33.8 percent, respectively (fig. 26).

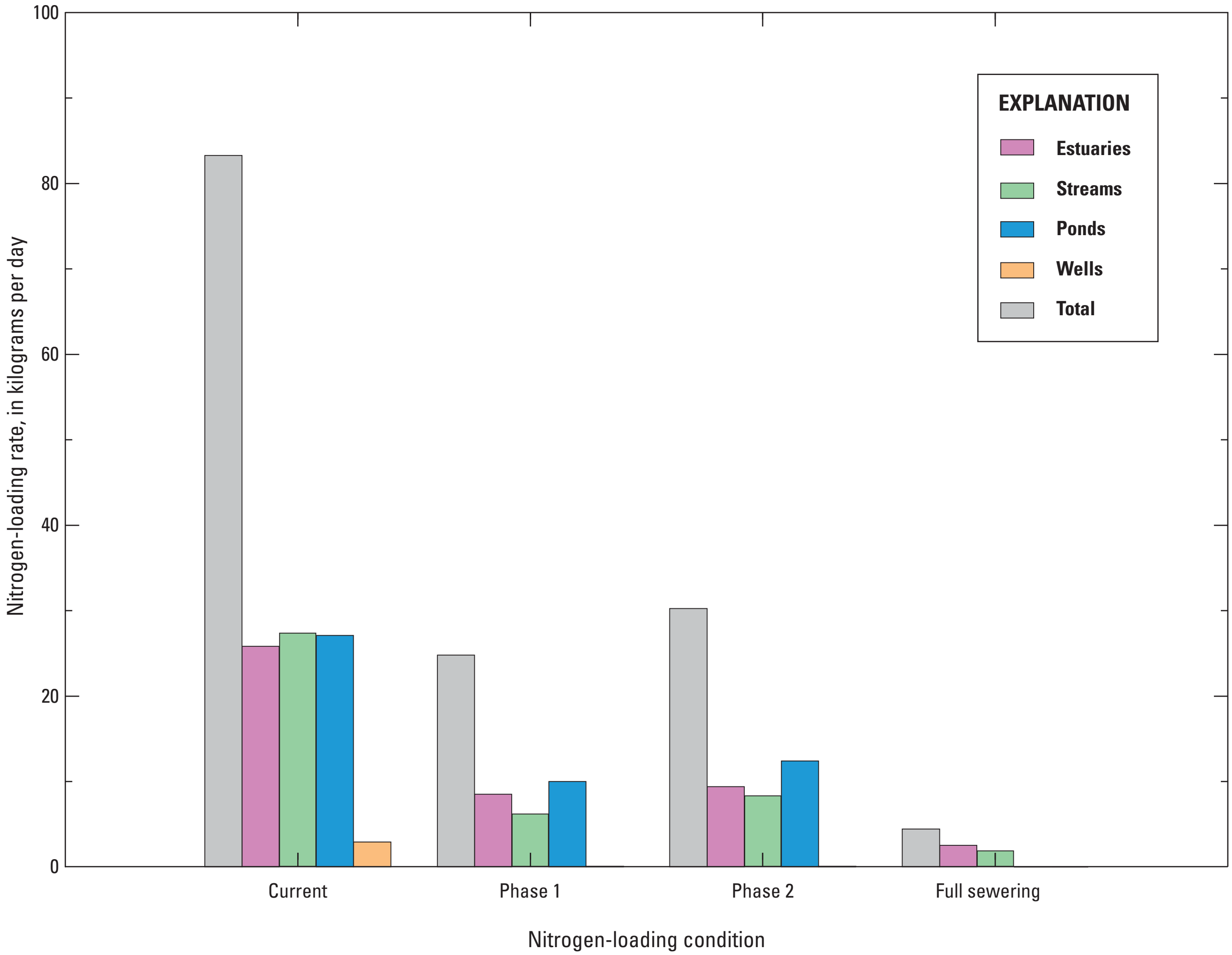


Figure 26. Nitrogen-loading rates estimated for recharge areas to wells and major ecological receptors under current (1997–2001) conditions and conditions resulting from full sewering and two phases of partial sewering and disposal at wastewater treatment facilities, Popponesset Bay watershed, western Cape Cod, Massachusetts.

The Effect of Dynamic Recharge Areas on Estimates of Nitrogen Loads

Recharge areas in groundwater-dominated aquifers, such as on Cape Cod, are functions of hydraulic gradients and can change in response to disposal at WTFs (fig. 19) and pumping from production wells (fig. 20). A change in a recharge area could result in a different estimated nitrogen load, depending on the amount of change in the recharge area and the unique distribution of nitrogen sources within near the recharge area. Operation of the UCC wells shifts the regional groundwater divide southward. As a result, the northern extent of the Popponesset Bay watershed shifts southward by as much 1,000 ft, and the recharge area to the ponds is smaller by about 8 percent (fig. 20). The result is a decrease of about 9.5 percent in the total current (1997–2001) load of waste-derived nitrogen to ponds in the northern part of the watershed.

Hypothetical disposal at the Rock Landing site, to the west of Popponesset Bay, increases the sizes of the recharge areas to Popponesset Creek and Ockway Bay and extends them southward in the direction of the site (figs. 19, 27). Using the recharge areas delineated for current (1997–2001) conditions and current sources of waste-derived nitrogen, so that recharge stresses are consistent with return-flow rates, results in loads to Popponesset Creek and Ockway Bay of about 3.22 and 1.83 kg/d, respectively. The hypothetical wastewater-management action includes the sewering of almost all of the recharge areas to the two estuaries, changing the hydraulic stresses and nitrogen sources (fig. 27). Using the future nitrogen sources and recharge areas for current conditions results in waste-derived nitrogen loads to the two estuaries of 0.44 and 0.08 kg/d, which are equal to decreases from current loading of about 86 and 96 percent, respectively. Using future nitrogen sources and recharge areas

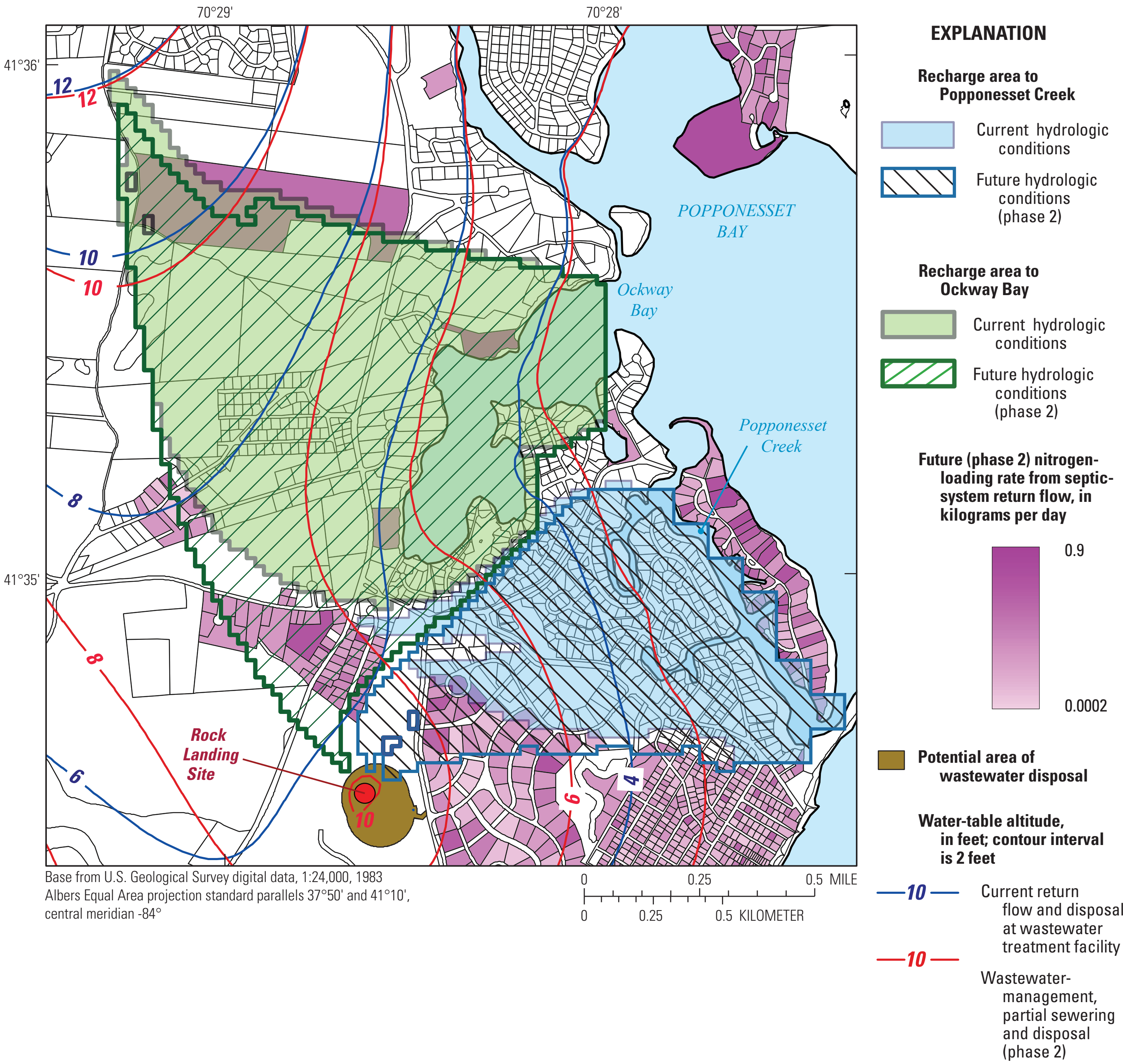


Figure 27. Recharge areas to Popponesset Creek and Ockway Bay for current hydrologic conditions and potential future conditions resulting from two phases of wastewater management following partial sewering and disposal, and future nitrogen-loading rates from parcels within the Mashpee Planning Area, western Cape Cod, Massachusetts.

for hydrologic stresses consistent with those future sources, however, yields nitrogen-load estimates for Popponesset Creek and Ockway Bay of 1.45 and 0.77 kg/d—3.3 and 9.6 times larger than the estimates obtained by using current recharge areas. Disposal at the Rock Landing site, which is close to the western boundary of the watershed, would raise water levels in the area by about 5 ft, change hydraulic gradients, and expand the recharge areas to include unsewered areas assumed to be outside of the watershed (figs. 5, 27). The nitrogen loads estimated by using recharge areas consistent with the estimated changes in nitrogen sources are nonetheless much less than current loads—about 55 and 58 percent less, respectively, for Popponesset Creek and Ockway Bay.

The use of recharge areas that are inconsistent with the nitrogen sources that cause a change in hydrologic stresses has the potential to be problematic if the stressed area is close to the boundary of a watershed. Estimates of current loads obtained by using recharge areas for current conditions, as has been done by the MEP, likely are valid. Furthermore, partial sewering and disposal at other WTFs did not affect recharge areas, and, therefore, presumably nitrogen-load estimates to the other four estuaries and three streams in the Popponesset Bay watershed, suggesting that the use of current recharge areas to estimate future loads probably is reasonable in those cases. However, when recharge areas are used to implicitly estimate nitrogen loads to ecological receptors, recharge areas must be consistent with wastewater-management actions if those actions or changes in groundwater withdrawals from pumped wells could alter hydraulic gradients and recharge areas. Inconsistencies are most likely to be problematic if large point sources, such as large WTFs, are proposed for locations near watershed boundaries.

Estimates of Time-Varying Nitrogen Loads by Using a Solute-Transport Model

The use of recharge areas to implicitly estimate waste-derived nitrogen loads from surface sources has limitations that include (1) the reliance on static recharge areas to represent dynamic conditions, (2) inherent limits in the ability of the resultant instantaneous analysis to assess the temporal response of the system to changing nitrogen sources, and (3) the needs to delineate potentially large numbers of recharge areas and to use a GIS for additional postprocessing to determine loads. Solute-transport models, on the other hand, explicitly represent the transport of nitrogen from sources to receptors. Modeled quantities include time-varying concentrations within the aquifer and at receptors and mass-loading rates determined by adding the products of concentrations and flow rates for groups of model cells representing a receptor. The only preprocessing required is the conversion of source information into the representation of modeled sources.

Estimates of Current Loads

Current (1997–2001) nitrogen loads can be explicitly simulated by using nitrogen sources rasterized from parcel-scale water-use data as the source term in a solute-transport model (fig. 9B). An inherent assumption of this approach is that current residential land uses—and the associated nitrogen sources—have not changed historically, and that current loads can be simulated by using the current nitrogen sources and the assumed initial condition wherein the aquifer contains no nitrogen. The simulated loads will approach a steady-state load approximating the instantaneous loads over sufficiently long transport times. Although residential development has changed over time, there is little historical data available regarding residential water use at the parcel scale. Given the short travel-times in the aquifer, current (1997–2001) loading rates can be assumed to be reasonable approximations of historical loading rates; this assumption also is inherent in the use of current recharge areas to estimate instantaneous loads. Concentrations of nitrogen in the aquifer between 0 and -10 ft, the range which generally corresponds to the altitude of groundwater discharging into coastal waters, exceed 5 mg/L as N in several areas after 30 years of transport (fig. 28A); patterns of concentrations generally are similar to the distribution of septic-system sources (fig. 13). Concentrations generally decrease with depth (increasing vertical transport distance from surface sources), but exceed 1 mg/L at an altitude of -100 ft in several areas (fig. 28B). Most nitrogen generally is in shallow parts of the aquifer (less than about -100 ft) after 30 years of transport from surficial sources to receptors (fig. 28C). Nitrogen is transported to deeper parts of the aquifer (altitude higher than -100 ft) primarily downgradient of ponds where the aquifer is recharged by pond water, and downward vertical gradients are greater; nitrogen is transported downward almost to the bedrock surface downgradient of Mashpee Pond (fig. 28C). Nitrogen concentrations downgradient of ponds—areas referred to as pond shadows—generally are more homogenous than concentrations upgradient of ponds because of mixing within the ponds (fig. 28C).

The simulated nitrogen load to a receptor would approach the instantaneous load over sufficiently long transport times, with some negligible difference caused by transverse and vertical dispersion—the rate at which the simulated load approaches the instantaneous load would be a function of groundwater velocities and likely would be greater in advection-dominated systems. The combined simulated load to all receptors in the Popponesset Bay watershed is within ranges of 50 percent of the instantaneous load estimated from sources in recharge areas after 7 years and 80 percent after 30 years (fig. 29) under the assumption that the aquifer initially contains no nitrogen. The rapid rate at which simulated loads approach instantaneous loads estimated from recharge areas suggests that, given the absence of historical land-use data, it is reasonable to use (1) current (1997–2001) sources for estimating current loads to receptors and (2) instantaneous loads estimated from recharge areas as approximations of current loads. Changes in nitrogen loads differ by the type of receptor. Loads to estuarine receptors reach 50 and 80 percent of the instantaneous loads estimated from recharge areas after about 7 and 25 years; similarly, loads to wells reach 90 percent of instantaneous loads estimated from recharge areas within 50 years (fig. 29). Nitrogen loads to streams exceed those estimated from direct recharge areas to the streams within 12 years and, after 100 years of transport, are 65 percent greater (fig. 29). This excess is a result of the additional load of nitrogen entering the streams from groundwater that has flowed through ponds prior to discharge and is not accounted for in the model-calculated recharge areas to the streams.

Effect of Surface Waters

Most (about 70 percent) of recharged water in the Popponesset Bay watershed discharges into a stream or pond (fig. 18A). As a result, attenuation of nitrogen concentrations in surface waters affects nitrogen loads to estuaries. Streams, estuaries, and wells are features where groundwater leaves the aquifer system, whereas ponds are flow-through features that receive water and associated nitrogen from recharge areas upgradient of the pond and contribute water and associated nitrogen to the aquifer downgradient from the ponds. In some cases, water and associated nitrogen may also exit a pond as surface-water outflow (Walter and Masterson, 2011). About 40 percent of the Popponesset Bay watershed is within the recharge area to a pond (fig. 18A), and about 34 percent of the total waste-derived nitrogen from current (1997–2001) sources in the watershed (79.1 kg/d; fig. 29) passes through a pond prior to discharging from the system at a stream, estuary, or well (26.9 kg/d). After 100 years of transport, about 0.01 and 0.21 kg/d of that nitrogen load is discharging to estuaries and wells, respectively (fig. 30A), representing less than 0.1 percent of the total mass of nitrogen passing through ponds. About 15.1 kg/d nitrogen (56 percent of the total) passing through ponds is discharging to streams after 100 years of transport (fig. 30A); another 6.2 kg/d (23 percent) is discharging into streams as surface-water outflow directly from the pond (fig. 30A).

After 100 years of transport, about 20 percent of the total discharging load of waste-derived nitrogen (represented as a mass-loading rate) flowing through a pond is not yet discharging to a receptor and is entrained in the aquifer (fig. 30A). Flow-through ponds cause steep vertical gradients that are upwards in areas upgradient and downward in areas downgradient from the ponds. As a result, advective flow paths downgradient from a pond pass deeper into the aquifer where groundwater fluxes are smaller and traveltimes are longer (Walter and others, 2003). Simulated concentrations at an altitude of -90 to -100 ft indicate areas with higher concentrations (exceeding 0.5 mg/L as N) downgradient from Mashpee Pond, Peters Pond, and Santuit Pond (fig. 28B); in these areas, pond water containing nitrogen has recharged the aquifer at the pond bottom and flowed to substantial depth. The large traveltimes of this deep water are the reason why a significant amount (about 20 percent) of the solute mass passing through the ponds has not arrived at a receptor after 100 years of transport; over sufficiently long transport times, the mass entrained deeper in the aquifer will discharge into a receptor, and nitrogen loads to the receptors will approach the instantaneous load.

Streams can receive water and an associated load of waste-derived nitrogen through (1) groundwater originating from direct recharge at the water table, (2) groundwater originating from a flow-through pond, and (3) surface-water inflow from a pond. The portion of the total nitrogen load from each component is a function of the hydrologic setting of the stream (fig. 30B). In the Popponesset Bay watershed, the simplest example is Quaker Run, which has no surface-water inflow and receives almost all (more than 99 percent) of its load of waste-derived nitrogen directly from groundwater recharged at the water table (fig. 30B). The Mashpee and Santuit Rivers both receive loads of waste-derived nitrogen from ponds through surface-water and groundwater inflows. The Santuit River, which receives the largest load of nitrogen of any receptor in the watershed (fig. 25), receives most (about 64 percent) of its nitrogen load from groundwater recharged directly at the water table. Nitrogen from Santuit Pond composes the remaining 36 percent of the load to the river—comprising about 24 percent groundwater originating from the pond and 12 percent from surface-water outflow from the pond. About half (51 percent) of the nitrogen load to the Mashpee River is from Mashpee Pond—about 40 percent from groundwater and 14 percent from surface-water outflow (fig. 30B). The remaining 49 percent is directly from groundwater recharged at the water table.

There likely is a loss of nitrogen in fresh surface waters resulting from biological processes; in determining loads to estuaries, the MEP assumes a loss of 50 percent within ponds and 30 percent within streams (Howes and others, 2003). The loss can be represented by using either of two methods: adjusting (1) the source terms or (2) loads at the receptors by a desired loss term. A simple loss term can be incorporated into estimates of nitrogen loads to receptors by decreasing the loads associated with sources within a recharge area by the desired amount. The same result can be achieved in a solute-transport model by adjusting the simulated loads at the receptor by the same loss term and thus requiring no delineation of the recharge area or other source adjustments. With an assumed loss of 50 percent in ponds and 30 percent in streams, the loads simulated by adjusting source terms by the appropriate loss term for the corresponding recharge areas are the same as the loads at the receptor adjusted directly by the same loss term at the receptor (fig. 31A). The effect of mass moving through more than one pond was not evaluated as part of this analysis, but the same concept of additive loss likely would apply. As an example, nitrogen entering the aquifer upgradient of Peters Pond (the northernmost pond in the Popponesset Bay watershed; fig. 3) would flow through the pond prior to reentering the aquifer and discharging into

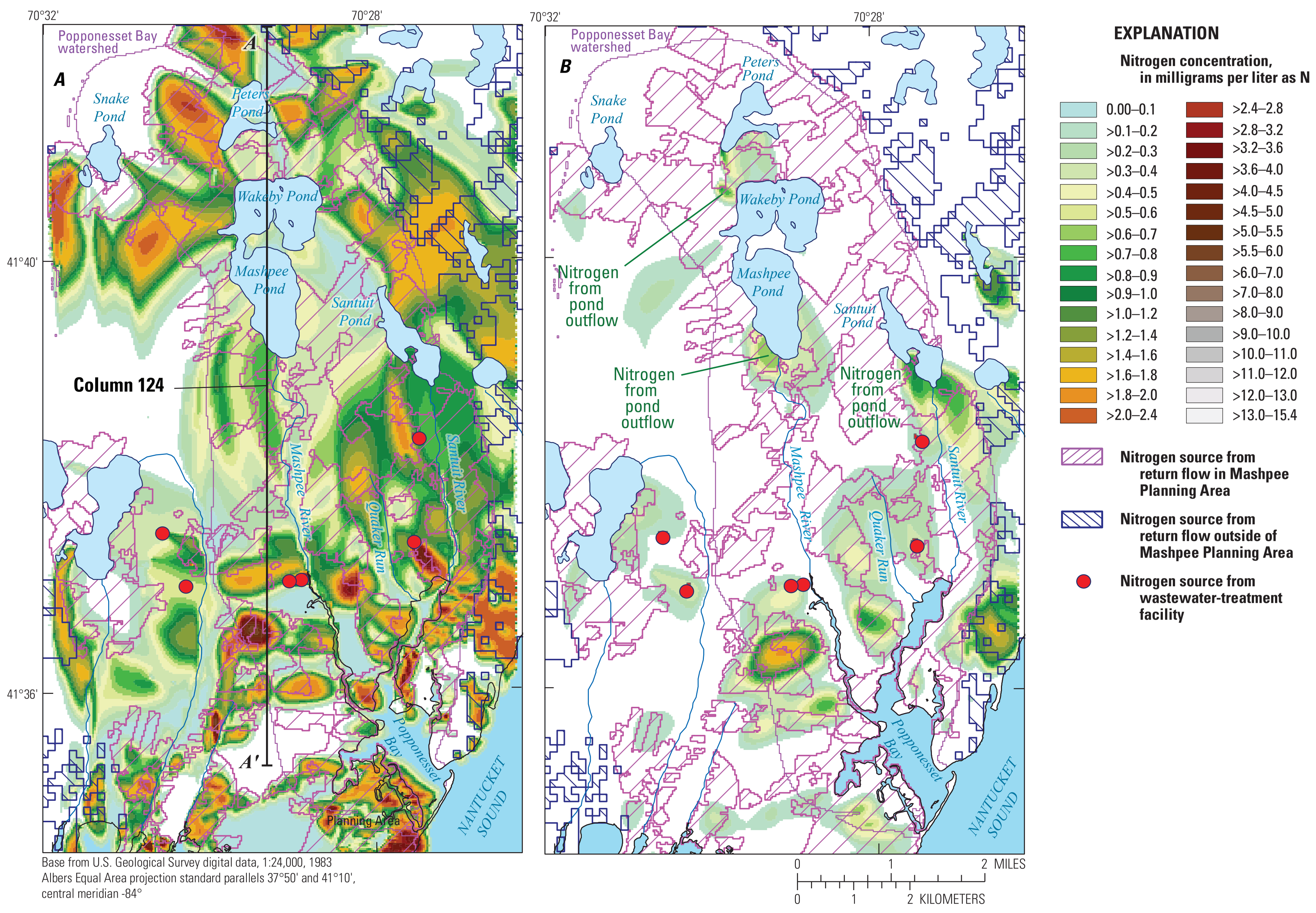


Figure 28. Simulated nitrogen concentrations after 30 years of transport from current nitrogen sources for altitude ranges *A,* 0 to -10 feet, *B,* -90 to -100 feet, and *C,* along vertical section A–A′, near Popponesset Bay watershed, western Cape Cod, Massachusetts. Bedrock topography from Fairchild and others, 2013.

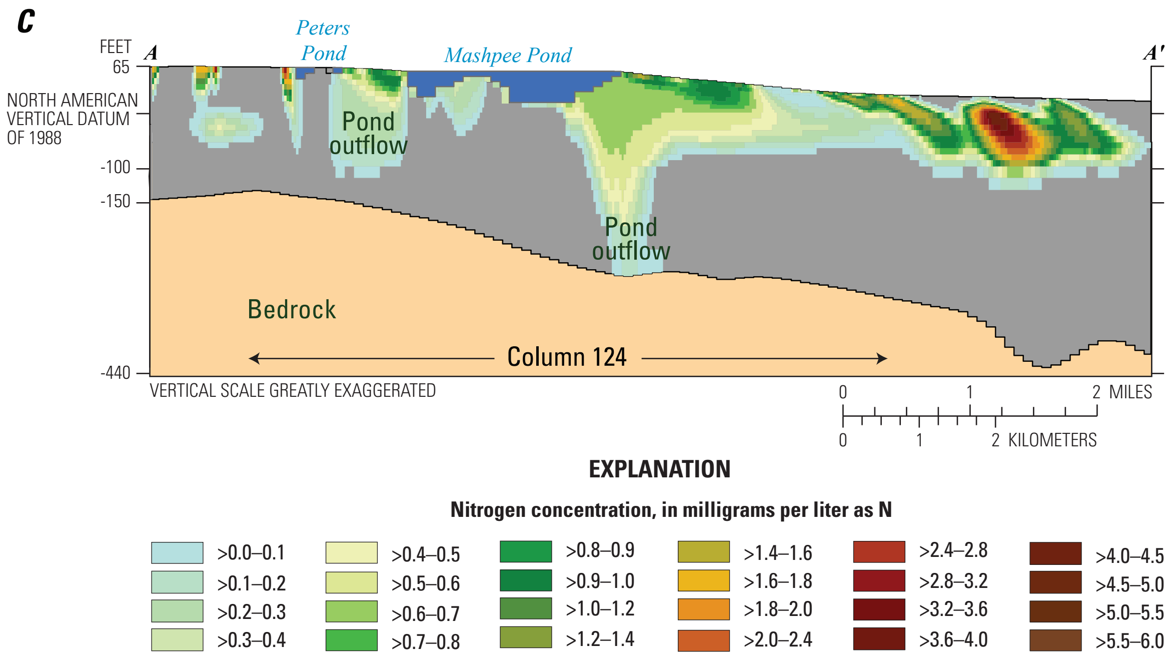


Figure 28. Simulated nitrogen concentrations after 30 years of transport from current nitrogen sources for altitude ranges *A,* 0 to -10 feet, *B,* -90 to -100 feet, and *C,* along vertical section A–A′, near Popponesset Bay watershed, western Cape Cod, Massachusetts. Bedrock topography from Fairchild and others, 2013.—Continued

Mashpee—Wakeby Pond; the sum of the loss terms for the two ponds could apply to that component of the nitrogen load.

Attenuation of nitrogen in surface waters—ponds and streams—was implicitly represented in MEP load calculations as assumed loss factors of 0.5 and 0.3, respectively. The potential for loss in fresh surface waters makes it advantageous to explicitly account for the mechanisms by which nitrogen enters an estuary. As an example, the Mashpee River receives most of its load of waste-derived nitrogen from recharged groundwater (fig. 30B). The load of nitrogen directly from groundwater recharged at the water table (about 49 percent) can be decreased by 30 percent (as assumed by the MEP) to implicitly represent attenuation in the river; the load of nitrogen in groundwater that passed through a pond prior to discharging into the river could be decreased by 50 percent (as assumed by the MEP) to represent attenuation in the pond and, if the loss terms are assumed to be sequential, decreased by an additional 30 percent to represent the final degree of attenuation in the stream. With assumed losses of 50 and 30 percent for nitrogen originating within recharge areas to ponds and streams, respectively (consistent with MEP assumptions), the attenuation of nitrogen entering the tidal portion of the Mashpee River as surface-water inflow (the difference between attenuated and unattenuated loads) is estimated as 10.5 kg/d (40 percent) after 100 years of transport (fig. 31B). The nitrogen load can be attenuated by an additional 2 kg/d if an additional loss of 30 percent is assumed for nitrogen in groundwater entering the stream that had originated from a pond—a total loss of 65 percent for that component of the load (fig. 31B).

Assessing System Response to Wastewater-Management Actions

Changes in loads of waste-derived nitrogen at receptors in response to wastewater-management actions are functions of advective traveltimes in an advection-dominated aquifer (fig. 23); these traveltimes are small (less than 10 years from the water table to receptors) in most of the Popponesset Bay watershed (fig. 22). Nitrogen concentrations in the elevation horizon between 0 and -10 ft are less than 0.2 mg/L as N in most of the watershed after 30 years of transport following full sewering and removal of the collected and treated wastewater from the watershed (fig. 32). Concentrations exceed 5 mg/L as N downgradient from existing WTFs and 0.5 mg/L downgradient from Peters Pond and Mashpee Pond (fig. 32). The nitrogen downgradient from the ponds—one of the components of nitrogen loads to the Mashpee River—is dissolved in deep groundwater recharged from the ponds; traveltimes for groundwater flowing deeper in the aquifer are longer, and the nitrogen load to streams from groundwater that has passed through a pond is still increasing after 90 years of loading at the assumed rate for current sources (1997–2001) (fig. 30B).

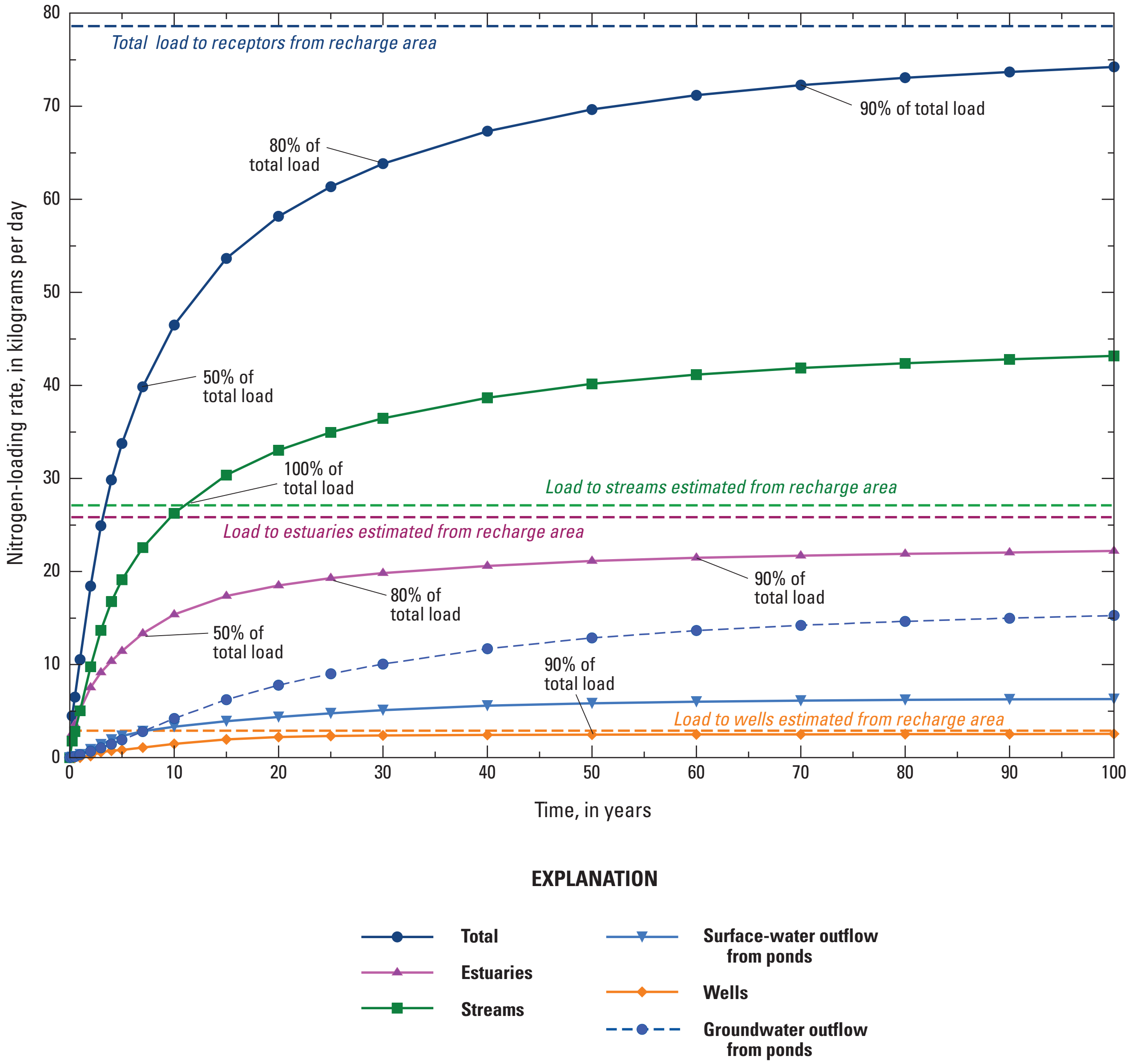


Figure 29. Time-varying loads of waste-derived nitrogen to wells and ecological receptors over 100 years of transport from current nitrogen sources, Popponesset Bay watershed, western Cape Cod, Massachusetts. %, percent.

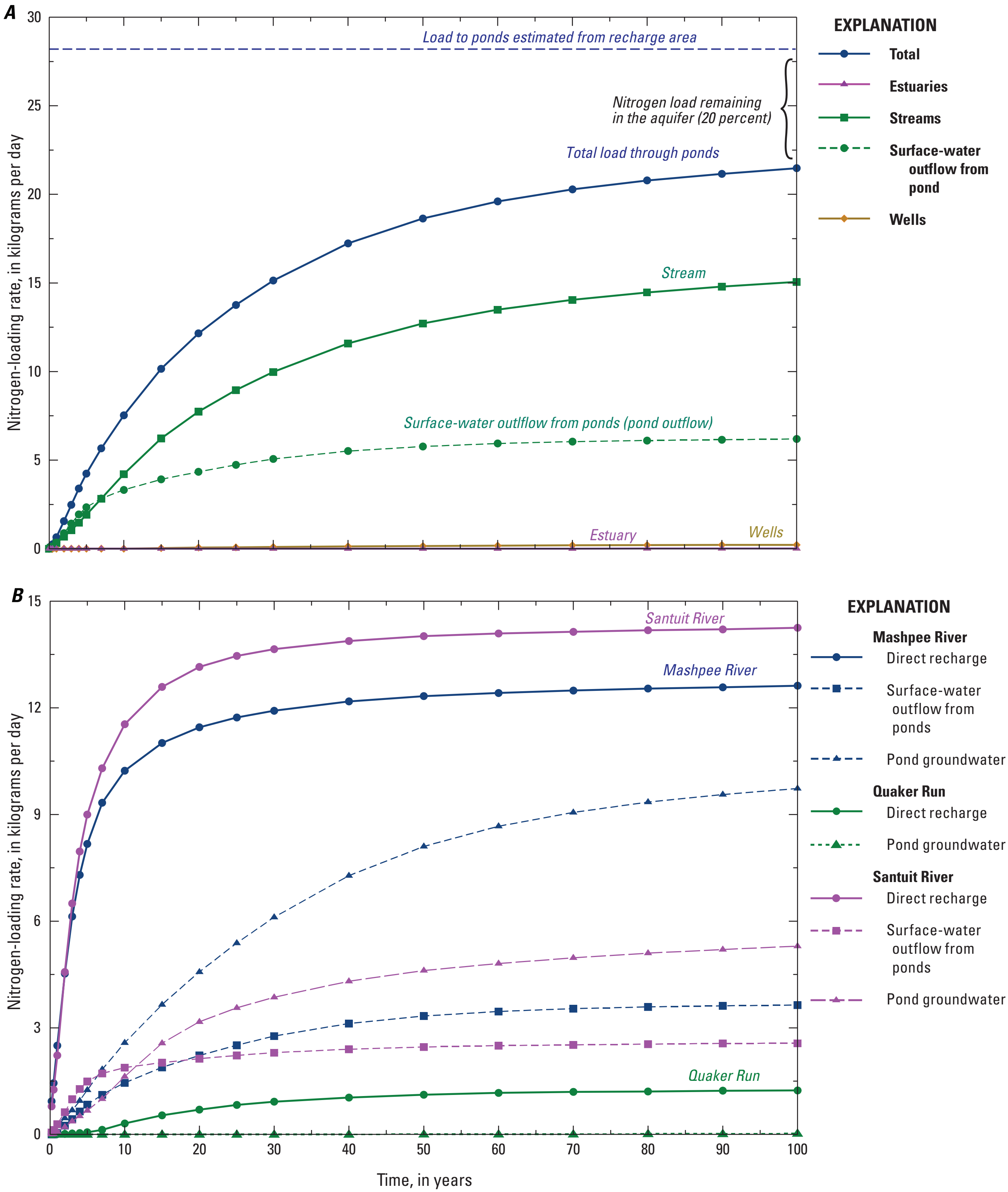


Figure 30. *A,* Time-varying loads of waste-derived nitrogen that have passed through ponds prior to discharging at receptors and *B,* components of nitrogen loads to three streams, Popponesset Bay watershed, western Cape Cod, Massachusetts.

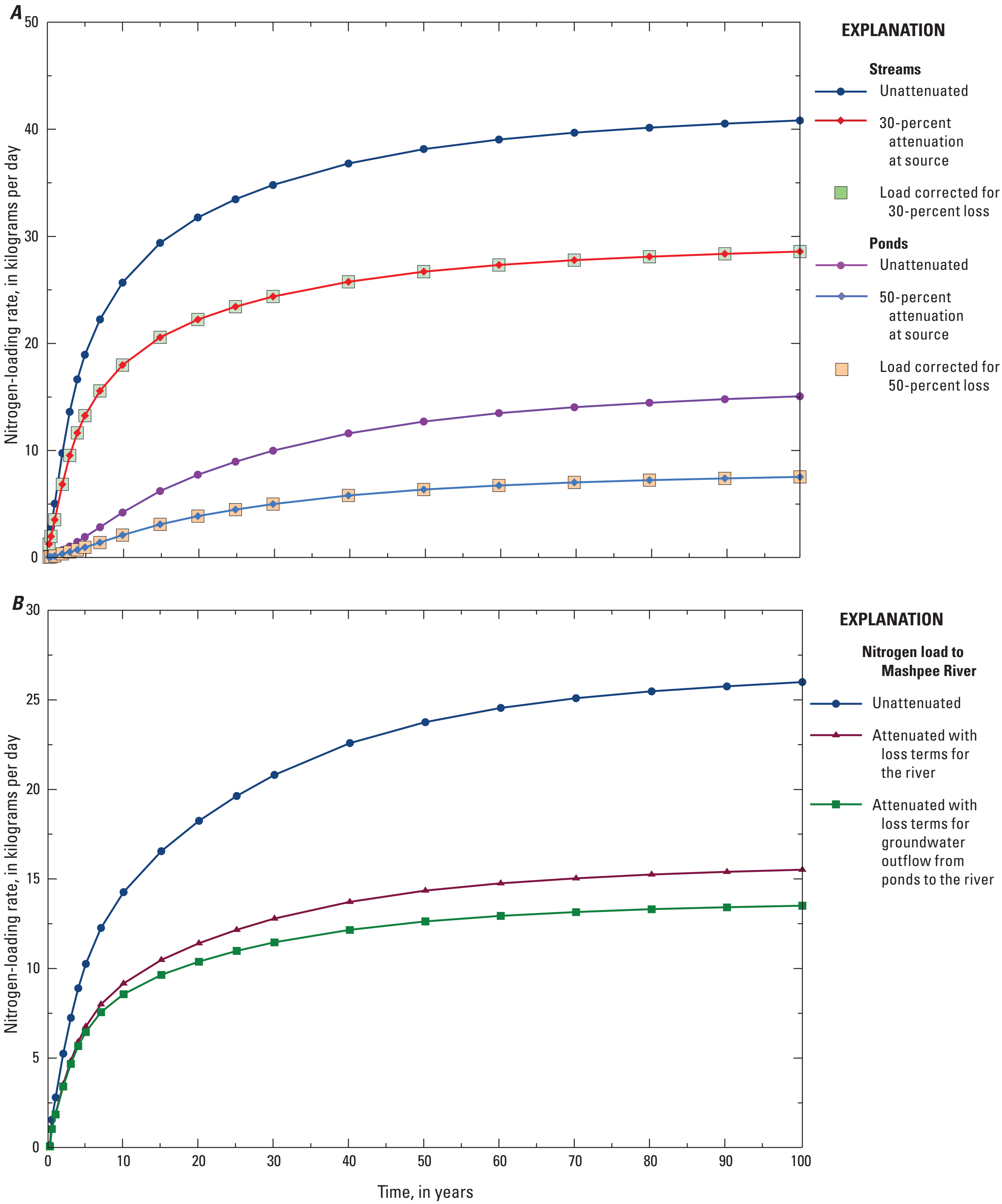


Figure 31. *A,* Unattenuated and attenuated loads to streams and ponds calculated by using loss terms and *B,* nitrogen loads to the tidal reach of the Mashpee River calculated under different loss assumptions, Popponesset Bay watershed, western Cape Cod, Massachusetts.

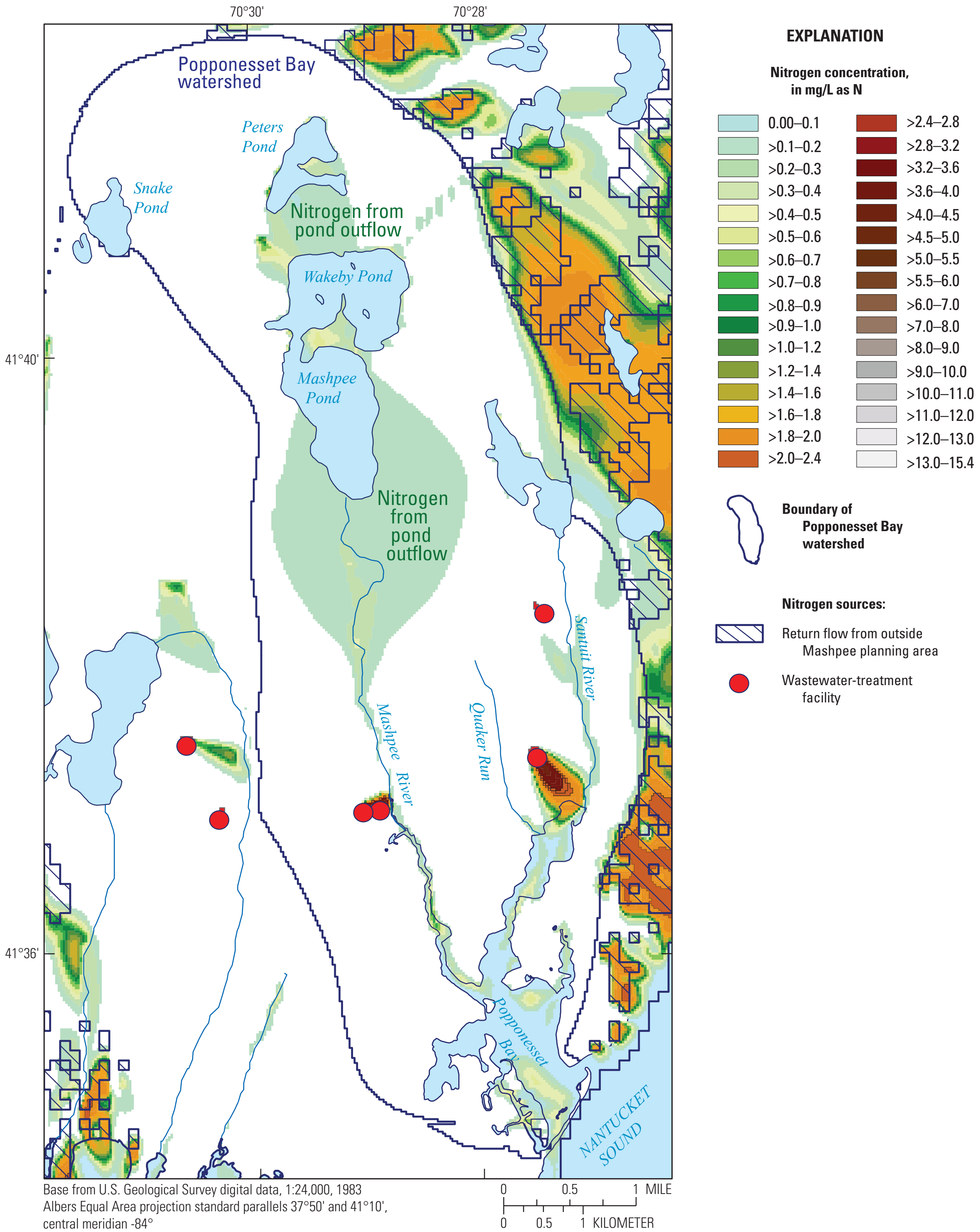


Figure 32. Concentrations of nitrogen at an altitude between 0 and -10 feet following 30 years of full sewering in the Mashpee Planning Area, western Cape Cod, Massachusetts.

The response of the system to wastewater-management actions, including nitrogen loads to receptors, involves adjusting nitrogen sources to reflect the action and using the simulated concentration field after a desired period of transport from pervious sources as the initial condition for the solute-transport simulation. The load to receptors in the Popponesset Bay watershed decreases from about 64 to 6 kg/d after 90 years of transport following full sewering of the MPA (fig. 33). About 50 percent of the total decrease occurs by about 5 years after full sewering and 80 percent by about 15 years after full sewering (fig. 33). These results are consistent with the median advective traveltimes from the water table to receptors, which range from 3.7 years for streams to 5.7 years for estuaries (fig. 23). It should be noted that the predicted loads presented in figures 33–36 are not corrected for surface-water attenuation; these predicted (unattenuated) loads can be corrected by a specified loss term as discussed in the section entitled “Effect of Surface Waters.” Correcting for loss in surface waters proportionally decreases the response curves but does not affect the proportional response of loads to wastewater-management actions.

The response of nitrogen loads at receptors to partial sewering and disposal within the planning area is similar to that for full sewering (fig. 34). Loads of waste-derived nitrogen decrease from about 64 to 27 kg/d after 90 years of transport and 30 years after implementation of the second phase of wastewater management; these rates of decrease approximate a near steady state with respect to nitrogen-loading rates. About 50 percent of the total decrease occurs within 3 years and about 80 percent within 10 years after implementation of the first phase of wastewater management (fig. 34). Thirty years after the phase-1 action has been in place, the total load of waste-derived nitrogen has decreased to about 26 kg/d. Loads of waste-derived nitrogen increase slightly—to about 27 kg/d—after implementation of the second phase of wastewater management; the second phase includes additional return flow from new residential development (fig. 14B). The load of waste-derived nitrogen to the coast (Nantucket Sound) increases following the two phases of wastewater management from about 12 to 21 kg/d. Open coastal waters have a large amount of hydrodynamic circulation and flushing, and, therefore, nitrogen inflow to the coastal waters likely does not adversely affect ecosystems in the waters around Cape Cod; Howes and others (2004) report that Nantucket Sound has a much lower mean nitrogen concentration than the Popponesset Bay estuarine system.

Wastewater-management and TMDL implementation, as currently (2003–2012) evaluated on Cape Cod, are based on the assumption that open coastal waters, like Nantucket Sound, are suitable as receiving waters for treated wastewater. As a result, the direct-recharge areas to open coastal waters, such as Nantucket Sound, are generally assumed to be suitable locations for the future disposal of treated wastewater. Discharge to these areas effectively transfers the potential nitrogen load from more sensitive receptors, such as streams and inland estuaries, directly to open coastal waters.

The load of waste-derived nitrogen comes from both septic-system return flow and treated wastewater (with lower nitrogen concentrations); partial sewering and disposal from WTFs in the watershed can partially offset each other, with sewering decreasing the load and disposal at WTFs within the watershed increasing the loads. The desired result is a net decrease in the total load to a rate that is compliant with a given TMDL. The two phases of wastewater management decrease the nitrogen load to receptors by about 37 kg/d (fig. 35A), whereas disposal at WTFs within the watershed results in an increase of about 4 kg/d. The load of nitrogen from WTFs is about 29 percent of the total load to receptors within the Popponesset Bay watershed (fig. 35A) after 90 years of wastewater management if treatment at WTFs is assumed to reduce nitrogen concentrations in the collected wastewater to 3 mg/L. Estuaries are the type of receptor that receives the largest amount of treated wastewater—about 55 percent of the total load after 90 years of wastewater management (fig. 35A, B); the load of treated wastewater to streams is about 18 percent of their total load from WTFs and septic systems after the same time interval (fig. 35B). Determining the relative effects of sewering and disposal at WTFs can allow planners to determine the efficacies of these actions in lowering effluent concentrations at WTFs if larger decreases in nitrogen loads are needed for TMDL compliance.

Examination of total loads to different types of receptors improves understanding of the dynamics of the aquifer system as it relates to nitrogen transport and the response to wastewater-management actions; however, acceptable loads to individual estuaries, which are based on a set of water-quality, ecological, and hydrodynamic criteria established by the MEP, are needed to evaluate TMDL compliance. The MPA includes the 6 estuaries within the Popponesset Bay system and 10 of the 15 estuaries within the eastern part of the Waquoit Bay system (table 2); the location of each modeled estuary is shown in figure 8 (corresponding identifiers are listed in table 2). The response of nitrogen-loading rates from direct groundwater discharge varies among the estuaries (fig. 36) and is a function of the hydrologic setting of the estuary, traveltimes within the recharge area, and the unique set of current and future nitrogen sources within the recharge area. The nitrogen loads presented in figure 36 are from direct groundwater discharge and do not include nitrogen loads from surface-water inflows.

In the Popponesset Bay system, most of the estuaries show similar changes in nitrogen loads in response to the two phases of wastewater management: a decrease following implementation of sewering in the first phase of wastewater management followed by a small increase after implementation of the second phase because of additional residential development (fig. 36A). The load to Pinquickset Cove increases steadily but remains less than 1 kg/d 90 years after implementation of the wastewater-management actions (fig. 36A) probably because the recharge area to the estuary is small and relatively undeveloped with no planned sewering (fig. 14A). Nitrogen loads decrease the most (by about 90 percent) in the tidal portion of the Mashpee River

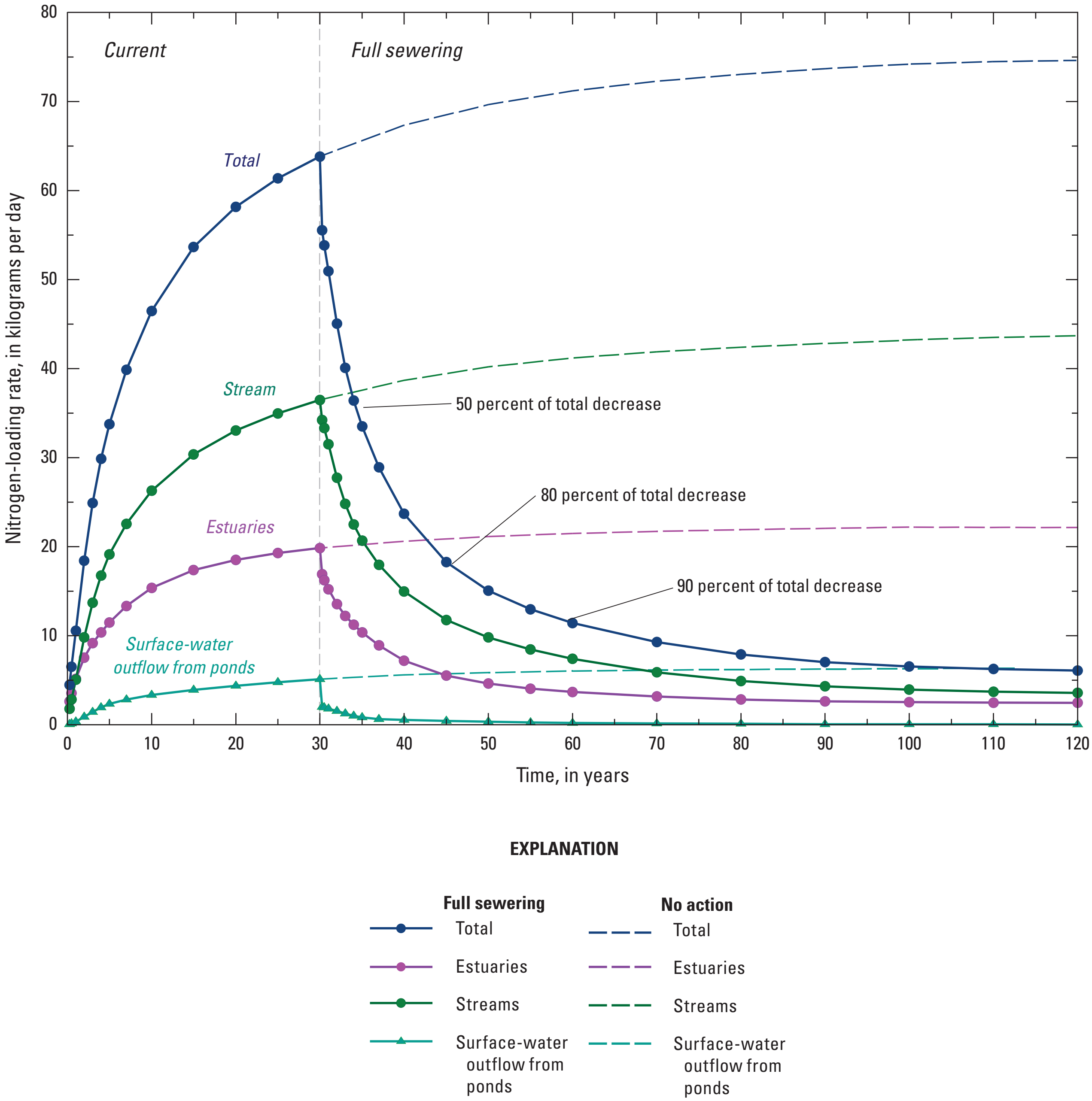


Figure 33. Changes in loads of waste-derived nitrogen to wells and ecological receptors in the Popponesset Bay watershed following full sewering of the Mashpee Planning Area, western Cape Cod, Massachusetts.

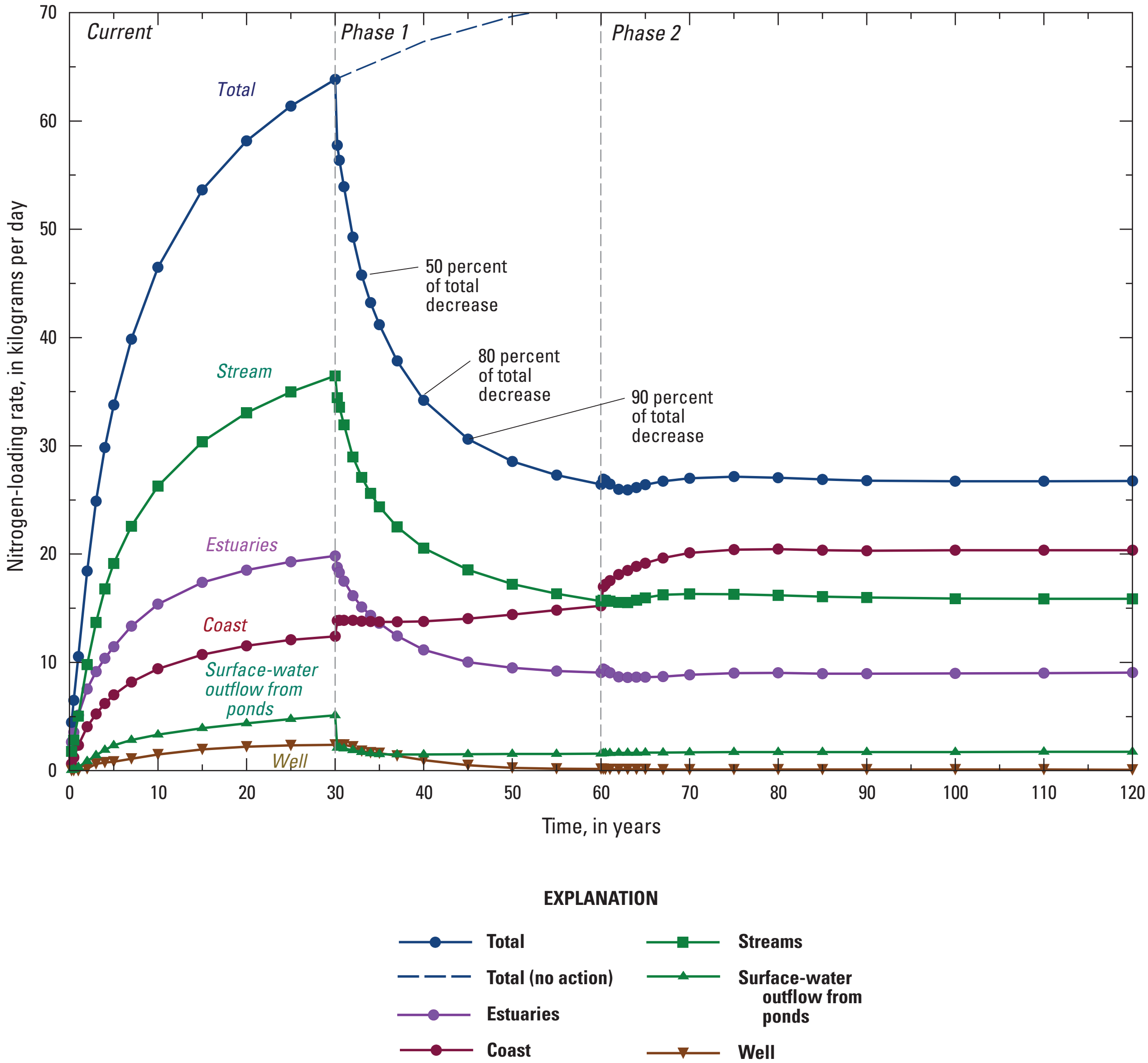


Figure 34. Changes in loads of waste-derived nitrogen at wells and ecological receptors in the Popponesset Bay watershed following two phases of partial sewering and disposal of treated wastewater at wastewater-treatment facilities within the Mashpee Planning Area, western Cape Cod, Massachusetts.

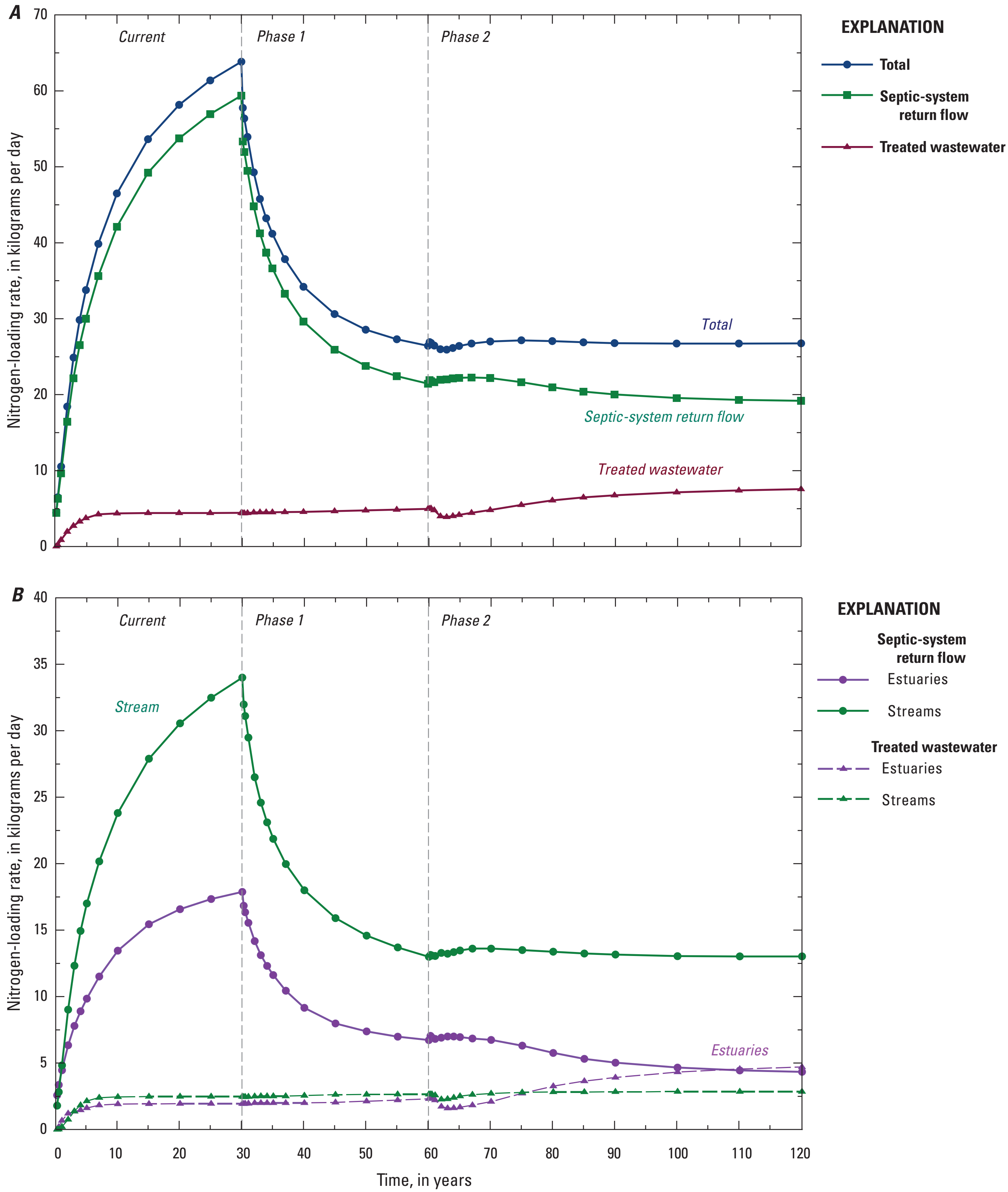


Figure 35. Changes in nitrogen loads in wastewater from partial sewering, septic-system return flow, and disposal at wastewater-treatment facilities in the Mashpee Planning Area for *A,* all receptors and *B,* streams and estuaries in the Popponesset Bay watershed, western Cape Cod, Massachusetts.

Table 2. Names and corresponding identifiers for freshwater streams, estuaries, and open coastal waters that receive nitrogen from terrestrial sources, Popponesset Bay watershed and the eastern part of Waquoit Bay watershed, western Cape Cod, Massachusetts.

[Nantucket Sound, identifier 666/777, is the only open coastal-water receptor and is not listed; identifiers are shown on figure 8]

Identifier	Name	Watershed	Mashpee Planning Area
Freshwater streams			
6	Childs River	Childs River	
7	Quashnet River	Quashnet River	
8	Little River	Little River	
9	Mashpee River	Mashpee River	X
10	Quaker Run	Quaker Run	X
11	Santuit River	Santuit River	X
12	Marston Mills River	Marston Mills River	
Estuaries			
41	Ockway Bay	Popponesset Bay	X
63	Eel Pond	Waquoit Bay	
64	Upper Quashnet River (tidal)	Waquoit Bay	X
65	Childs River (tidal)	Waquoit Bay	
66	Sage Lot Pond	Waquoit Bay	X
67	Hamblin Pond	Waquoit Bay	X
68	Great River	Waquoit Bay	X
76	Jehu's Pond	Waquoit Bay	X
78	Flat Pond	Waquoit Bay	X
79	Mashpee River (tidal)	Popponesset Bay	X
80	Shoestring Bay	Popponesset Bay	X
81	Pinquickset Cove	Popponesset Bay	X
82	Rushy Marsh Pond	Three Bays	
83	Popponesset Bay	Popponesset Bay	X
99	Seapit River	Waquoit Bay	
129	Popponesset Creek	Popponesset Bay	X
130	Waquoit Bay	Waquoit Bay	
142	Little River (tidal)	Waquoit Bay	X
143	Upper Seapit River	Waquoit Bay	
144	Red Brook	Waquoit Bay	X
178	Lower Quashnet River	Waquoit Bay	X
179	Lower Great River	Waquoit Bay	X

(fig. 36A). The load to the estuary decreases following implementation of the first phase of sewering, but more gradually than at other estuaries (fig. 36A), partly because most of the residential areas to be sewered (fig. 14A) are in areas with longer traveltimes (fig. 24). The result is a slower response of loads to sewering within the recharge area. Also, loads decrease substantially following implementation of the second phase of wastewater management (fig. 36A) as a result of the cessation of wastewater discharge at a WTF within the recharge area (fig. 14B). The responses of nitrogen loads to estuaries in the eastern part of the Waquoit Bay system, where current loading rates generally are lower, are similar: a large decrease following the first phase of wastewater management and little change resulting from the second phase (fig. 36B). The first phase would include nearly full sewering of the recharge areas to those estuaries (fig. 14A).

Two estuaries in the Popponesset Bay system, the tidal reach of the Mashpee River and Shoestring Bay, receive loads of waste-derived nitrogen from surface-water inflows—77 and 75 percent, respectively—prior to the implementation of wastewater-management actions. The total load of waste-derived nitrogen to the tidal portion of the Mashpee River decreases by about 17.4 kg/d after 90 years of wastewater management (fig. 37A). Decreases in loads from direct groundwater discharge and surface-water inflow from the Mashpee River compose about 31 and 61 percent of the total decrease in nitrogen load, respectively; the remaining 8 percent is from a decrease in the load from surface-water outflow from Mashpee Pond (fig. 37A). Wastewater-management actions result in a total decrease in nitrogen load to Shoestring Bay of about 13.4 kg/d (fig. 37B). About 21 percent of the total decrease is from a decrease in the load from direct groundwater discharge (fig. 36B). Decreases in loads in surface-water inflow from the Santuit River and Quaker Run compose about 68 and 6 percent, respectively, of the total decrease; the remaining decrease—about 5 percent—is from surface-water outflow from Santuit Pond (fig. 37B). Decreases in loads to surface waters account for 79 and 69 percent, respectively, of the total decrease in loads to Shoestring Bay and the tidal portion of the Mashpee River.

Explicitly representing the mass of nitrogen in a transport simulation allows for attenuated loads to be determined for a given set of assumed loss factors; the model-calculated loads can be adjusted by loss factors to implicitly represent attenuation in surface waters. With losses of 50 percent in ponds and 30 percent in streams, which are consistent with MEP assumptions, the attenuated loads (uncorrected for pond groundwater) to the tidal portion of the Mashpee River and Shoestring Bay are both about 4 kg/d lower than the unattenuated load after the 30 years of partial sewering and disposal at WTFs (phase 1; fig. 37). The loss of nitrogen in groundwater that has passed through a pond prior to discharge into a stream would include both loss terms—for a total of loss of about 65 percent. Including the additional loss for groundwater that has passed through a pond prior to discharge would decrease the load of waste-derived nitrogen to the tidal portions of the Mashpee River and

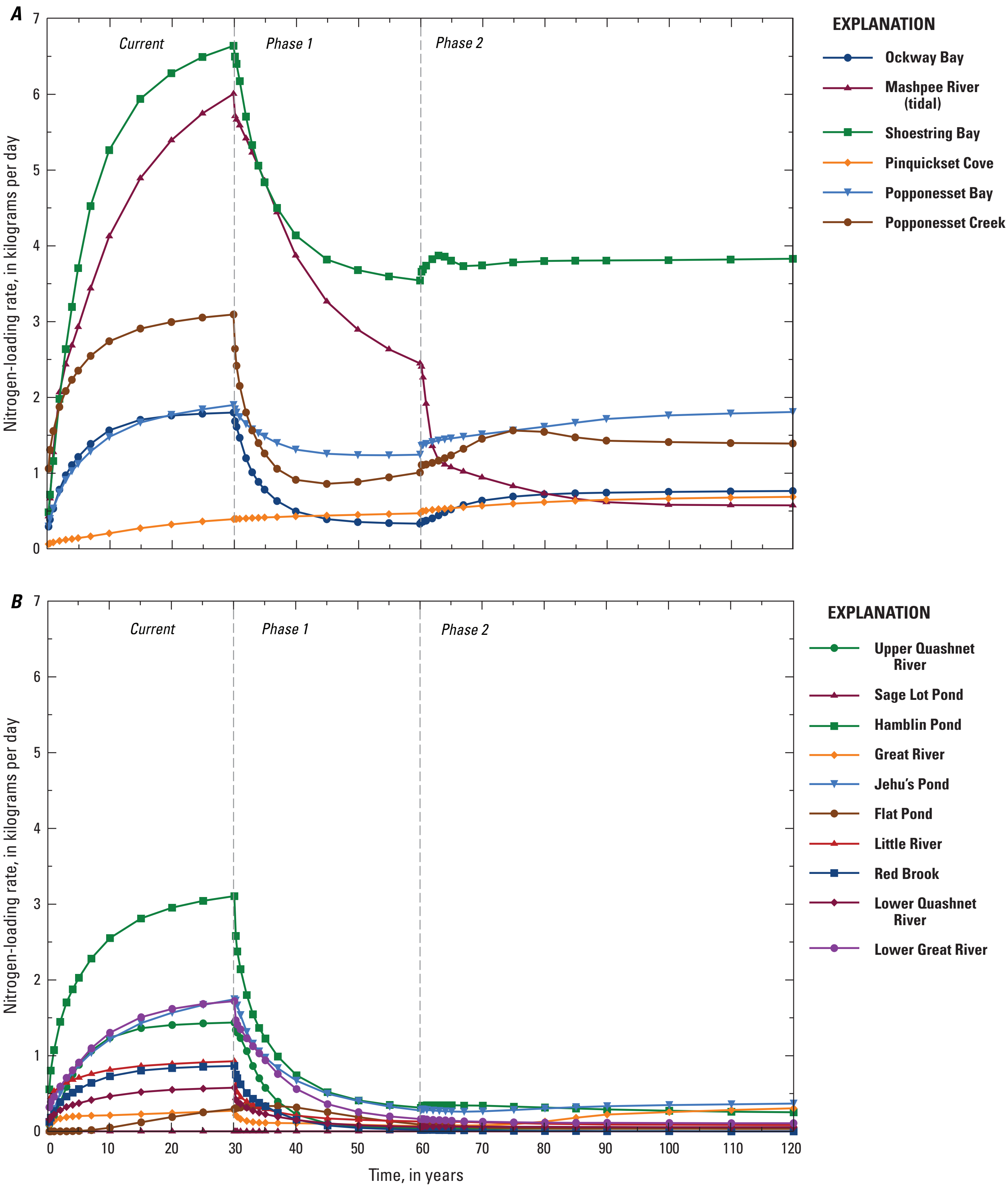


Figure 36. Changes in nitrogen loads from direct groundwater discharge resulting from partial sewering and disposal at wastewater-treatment facilities in the Mashpee Planning Area for individual receptors in the *A,* Popponesset Bay system and *B,* the eastern part of the Waquoit Bay system, western Cape Cod, Massachusetts.

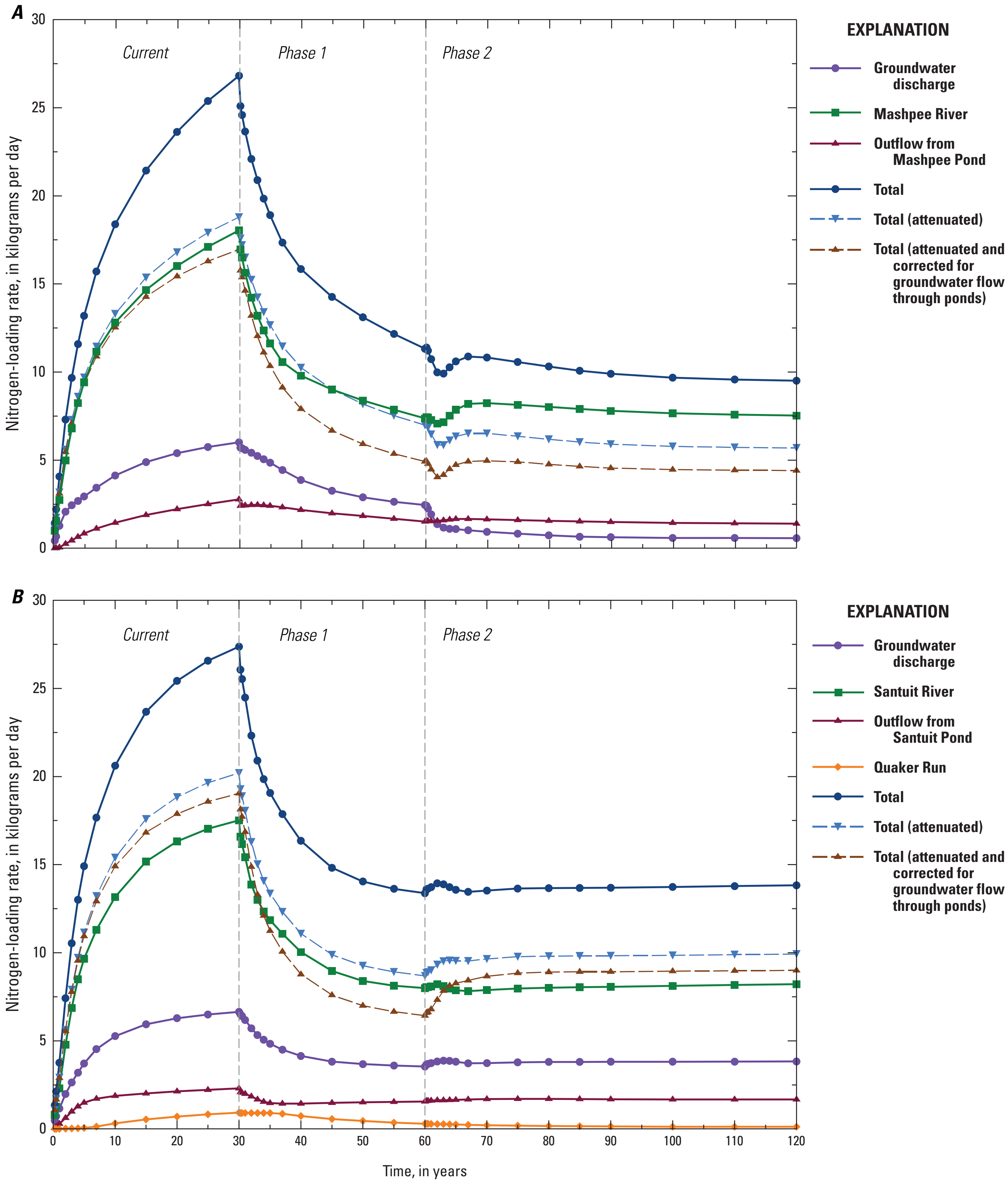


Figure 37. Changes in nitrogen-loading rates from septic-system return flow and treated wastewater in response to two sequential phases of partial sewering and disposal at wastewater-treatment facilities for different load sources in *A,* the Mashpee River and *B,* Shoestring Bay, Popponesset Bay watershed, western Cape Cod, Massachusetts.

Shoestring Bay by an additional 2.3 and 2.1 kg/d, respectively (fig. 37).

Limitations of Analysis

Sources of uncertainty associated with predictions of waste-derived nitrogen loads to receptors and of changes in those loads in response to a set of wastewater-management actions arise from (1) conceptual and numerical models of groundwater flow, (2) simplifying assumptions of conservative transport and values of transport parameters, and (3) limitations in the accuracy of data related to nitrogen sources and characteristics. The existing regional model was useful in determining the effect of time-varying recharge and pumping on the surface waters and the sources of water to pumped wells on western Cape Cod (Walter and Whealan, 2004) and recharge areas to ecological receptors—ponds, streams, and estuaries (Walter and others, 2004). With some modifications, namely an increase in vertical discretization, the model also allowed for the development of more finely discretized subregional models capable of simulating the transport of nitrogen in the aquifer. The existing regional groundwater-flow model used in this analysis was calibrated by trial and error to observations of water levels and streamflows; natural recharge, hydraulic conductivity, and boundary leakances were adjusted manually until there was a reasonable match to observed water levels and streamflows (Walter and Whealan, 2004). Changes in these three parameters could affect advective transport patterns and, therefore, predictions of the physical transport of nitrogen from sources to receptors. Because the calibrated flow model reasonably matched observed hydrologic conditions, model-calculated predictions of hydraulic gradients and advective-flow patterns were assumed to be reasonable.

The assumed quantities pertaining to nitrogen transport, such as the rates of dispersion and attenuation and the representation of surface waters, are considered valid for this analysis as discussed in the text. The effects of these assumptions, which were designed to be consistent with the MEP process, on nitrogen transport and loads to receptors could be readily evaluated with sensitivity analyses or the use of multiple-model realizations, but these processes were beyond the scope of this effort. An example of the effect of assumed dispersivities on nitrogen transport and loads is presented in Walter (2006). Also, physical and chemical processes that could affect nitrogen transport and attenuation in local environments, such as ponds and organic estuarine muds, were beyond the scope of and not explicitly represented in this analysis; those processes could be more explicitly simulated using a reactive-transport model.

Predictions of waste-derived nitrogen loads also are affected by transport parameters, such as porosity and dispersivity. Groundwater velocities are inversely proportional to porosity, which was assumed to be 0.3 in these analyses. A range in porosity of 0.2 to 0.4 would represent a twofold difference in velocities and transport times from the water table to receptors. Changes in the transport time affect the rate at which the load of nitrogen at a receptor changes in response to a change in nitrogen sources; however, the total mass discharging to the receptor after sufficiently long transport times would be the same. Dispersivity describes the effect of heterogeneity on the spreading of mass in an aquifer and is a function of aquifer characteristics and transport distance. Numerical dispersion refers to the spreading of simulated mass that arises from the numerical solution of the transport equation in discretized space beyond the amount of spreading explicitly related to heterogeneity. Longitudinal dispersion—in the direction of groundwater flow—is the largest component of numerical dispersion. In this analysis, a model-cell discretization was chosen that resulted in a longitudinal dispersion generally within the range of dispersivities reported for similar transport distances in other studies. Longitudinal dispersion affects the rate at which nitrogen loading changes in response to a change in nitrogen sources, but like porosity, does not affect the total mass discharging to a receptor over sufficiently long transport times.

Differences between predicted and actual loads of waste-derived nitrogen to a receptor also can arise from the assumptions in the analyses regarding nitrogen sources and transport processes. The assumption assumed to be reasonable in a generally oxic aquifer like that on Cape Cod that nitrogen is transported conservatively as nitrate allows for the use of groundwater and solute-transport models to explicitly represent mass transport and loading rates. Attenuation processes, such as denitrification, may occur locally in the aquifer and will result in actual loads that are lower than those predicted by the models; these processes, which generally operate within fine-grained estuarine muds that are characterized by reducing environments, further decrease actual loads to an estuary.

Assumptions are also made with regard to the definitions of nitrogen sources within a watershed. Little direct information regarding the amount and character of septic-system return flow is available, and as a result, the nitrogen input from return flow is estimated implicitly from water-use data, which are often combined from different periods. Nitrogen concentrations in septic-system return flow were assumed to be equal to 35 mg/L for these analyses, and losses associated with attenuation in the unsaturated zone were represented implicitly as loss factors applied to surface sources. Inaccuracies in all of these assumptions propagate into the model-calculated estimates of nitrogen-loading rates at receptors. An additional uncertainty arises from the assumption that the calculated loads are in equilibrium with the loads originating from the defined sources. It was assumed that current sources reasonably represent the current distribution of mass in the aquifer, thereby discounting temporal changes in nitrogen sources, for which there generally was little information; changes in nitrogen sources resulting from wastewater-management actions were also assumed to be instantaneous. Although both of these assumptions are approximate, the small traveltimes in this aquifer suggest that the two assumptions are reasonable.

Summary and Conclusions

The discharge of excess nitrogen into estuaries on Cape Cod has led to the degradation of water quality and clarity, algal blooms, and the loss of critical eel-grass habitat in several estuarine systems. The largest source of nitrogen is from wastewater, primarily from residential septic systems. The Massachusetts Estuaries Project (MEP) was initiated by the Massachusetts Department of Environmental Protection (MassDEP) to establish Total Maximum Daily Loads (TMDLs) for nitrogen in estuaries and assist local communities in the restoration of marine ecosystems. The Popponesset Bay ecosystem consists of six estuaries that receive excess nitrogen primarily from residential septic systems within its 19-mi^2 watershed. TMDLs for nitrogen have been developed for the system, and the town of Mashpee currently (2012) is developing wastewater-management strategies to implement those TMDLs.

Development of the TMDLs included the implicit estimation of current nitrogen loads from parcel-scale water-use data and model-delineated recharge areas; these loads are referred to as "instantaneous" loads. A hydrodynamic model used the nitrogen loads to estimate the resultant water-quality conditions at specific locations within the estuary; this process is referred to as the "Linked Watershed-Embayment Management Modeling Approach." Implementation of the TMDLs involved determination of nitrogen loads from the same set of model-delineated recharge areas for a set of nitrogen sources altered or emplaced through wastewater-management actions. The same linked-model approach was used to determine if a specific wastewater-management action would result in a nitrogen load and resultant water-quality conditions in the estuary suitable for the growth of eel grass. The use of static recharge areas delineated by a groundwater-flow model to estimate nitrogen loads introduces limitations, including the inability to (1) account for the effects of dynamic recharge areas on nitrogen loads, (2) explicitly represent the potential effects of surface-water bodies on nitrogen loads, and (3) evaluate the time-varying responses of waste-derived nitrogen loads to wastewater-management actions. Solute-transport models, which explicitly represent the transport of mass through an aquifer from sources to receptors, can enhance implementation of the TMDLs by (1) eliminating the reliance on static recharge areas, (2) more explicitly representing the transport of mass through surface waters, and (3) allowing for the evaluation of changing waste-derived nitrogen loads in response to wastewater-management actions.

The U.S. Geological Survey, in cooperation with MassDEP, has evaluated the efficacy of using a solute-transport model to assist in the implementation of the TMDLs for nitrogen in the Popponesset Bay ecosystem. Parcel-scale water-use data assembled by the MEP were used to represent current nitrogen sources within the watershed. Three other source terms were developed representing full sewering of the watershed and two sequential phases of hypothetical wastewater management within the Mashpee Planning Area (MPA), which is composed of the Popponesset Bay watershed and parts of the Waquoit Bay watershed. The nitrogen-source terms were used as input into a subregional model of the watershed designed to represent the conservative transport of nitrate in the aquifer and linked to an existing regional model of western Cape Cod. The subregional model was used to evaluate the effects of different wastewater-management actions—hypothetical but consistent with those that could be considered for the watershed—on water levels, groundwater-discharge rates, recharge areas, and time-varying loads of waste-derived nitrogen to wells and ecological receptors.

Wastewater-management actions within the MPA included sewering, treatment, and disposal at wastewater-treatment facilities (WTFs). The resulting changes in hydrologic conditions included changes in water levels, groundwater discharges to streams and coastal waters, and hydraulic gradients. Changes in hydraulic gradients, in turn, can change recharge areas to wells and ecological receptors. Significant findings are summarized below.

- Return flow from septic systems is a dispersed hydraulic stress and represented only about 3 percent of recharge within the MPA. Groundwater discharge to ponds, streams, and estuaries in the Popponesset Bay watershed changed by less than 3 percent as a result of full sewering. The mean water-level decrease in the watershed as a result of full sewering was 0.07 ft; the maximum drawdown exceeded 0.2 ft near the northern extent of the watershed.

- The centralized disposal of treated return flow at WTFs is a more focused hydraulic stress that can cause localized water-level increases and changes in hydraulic gradients. Hypothetical disposal at a WTF site west of Popponesset Bay resulted in a water-level increase of more than 5 ft after two sequential phases of wastewater management and larger hydraulic gradients near Popponesset Bay. Groundwater discharges to two estuaries near the WTF increased by 15 and 28 percent as a result of the disposal of treated wastewater at this site.

- Recharge areas to receptors changed very little as a result of sewering because changes in the dispersed stress do not significantly change hydraulic gradients. However, the disposal of large amounts of treated wastewater at a WTF can cause large localized changes in water levels and hydraulic gradients. As a result, recharge areas to nearby receptors can change. Simulated recharge areas to two estuaries increased by about 20 percent and expanded 1,200–2,000 ft upgradient as a result of hypothetical disposal at a WTF site west of Popponesset Bay. Recharge areas also are affected by large groundwater withdrawals from wells; simulated pumping of three wells north of the watershed shifted the regional groundwater divide southward and decreased the recharge areas to ponds in the watershed by about 8 percent.

Estuaries receive waste-derived nitrogen from direct groundwater discharge and, in some cases, surface-water inflow from streams. Ponds are hydrologic features that both receive water from and contribute water to the aquifer; as a result, some waste-derived nitrogen passes through ponds prior to discharge into estuaries. Recharge areas to estuaries, streams, and ponds respectively compose about 27, 33, and 40 percent of the watershed. Wastewater-management actions that included sewering and centralized disposal at WTFs redistributed nitrogen sources within the watershed and changed loads of waste-derived nitrogen to receptors. Significant findings are summarized below.

- Loads of waste-derived nitrogen resulting from a set of wastewater-management actions can be implicitly estimated on the basis of recharge areas. However, recharge areas delineated on the basis of current hydrologic conditions were different from recharge areas delineated on the basis of hydrologic stresses and nitrogen sources resulting from wastewater-management actions. In most areas of the Popponesset Bay watershed, hydraulic gradients and recharge areas did not change substantially in response to wastewater-management actions, and inconsistencies between current and future recharge areas and nitrogen sources likely did not affect estimates of waste-derived nitrogen loads. Hydraulic gradients and recharge areas to two estuaries did change in response to hypothetical disposal at a WTF site to the west of Popponesset Bay. Loads of waste-derived nitrogen to these two estuaries estimated by using recharge areas consistent with the simulated hydraulic gradients were 3 to 9 times larger than loads estimated by using current-condition recharge areas.

- Current nitrogen loads explicitly simulated by using a solute-transport model were similar to those estimated on the basis of recharge areas over sufficiently long transport times when simulated loads approach a steady-state condition with respect to with surface sources. Total nitrogen loads estimated from current sources by a solute-transport model were within 50 and 80 percent of the instantaneous loads by about 7 and 30 years, respectively. These results are consistent with the small advective traveltimes in the aquifer—median traveltimes were less than 6 years to ponds, streams, and estuaries—and suggest that the use of current-condition recharge areas is suitable for estimating current nitrogen loads to receptors.

- Most (about 70 percent) of recharged water in the Popponesset Bay watershed enters the estuaries as surface-water inflow, with about 40 percent of recharged water flowing through a pond prior to discharging into a stream or estuary. Surface waters attenuate nitrogen; attenuation losses are assumed to be 30 and 50 percent in streams and ponds, respectively, for evaluating TMDL compliance. Explicitly representing the transport of mass improved the estimation of attenuation within surface waters. After 100 years of transport from current sources, about 56 percent of the simulated mass of nitrogen passing through a pond discharged to a stream, and about 23 percent left the pond as surface-water outflow to streams; less than 0.1 percent discharged directly to estuaries. The remaining 20 percent of the nitrogen daily load passing through a pond was still stored in deeper parts of the aquifer after 100 years of simulated transport but would be expected to eventually discharge to a receptor after sufficiently long transport times. Accounting for the mass of nitrogen passing through ponds prior to discharging to a stream resulted in total attenuated loads to the estuaries that were lower than loads determined without accounting for that process.

- The response of waste-derived nitrogen loads at receptors to wastewater-management actions is a function of traveltimes, which generally are small in the Cape Cod aquifer—median traveltimes were less than 6 years to the natural receptors. Full sewering of the MPA decreased loads to receptors in the Popponesset Bay watershed from about 64 to 6 kg/d. About 50 and 80 percent of the total decrease in nitrogen load occurred within 5 and 15 years, respectively, following implementation of full sewering. Nitrogen loads to receptors decreased from about 64 to 27 kg/d following the implementation of two sequential phases of partial sewering and disposal at WTFs; the decrease in loads was about 50 and 80 percent of the final amount within 3 and 10 years, respectively, of complete implementation of the two phases.

- The response of an individual estuary is a function of traveltimes to the estuary, the mechanisms by which nitrogen enters the estuary, and the specific wastewater-management actions implemented within the recharge area to the receptor. Changes in nitrogen loads from direct groundwater discharge to individual estuaries in the Popponesset Bay watershed ranged from a decrease of 90 percent to an increase of 80 percent depending on the wastewater-management actions that were implemented. Two estuaries in the Popponesset Bay ecosystem also receive loads from surface-water inflows. Most of the 69- and 79-percent decreases in the total nitrogen loads to these estuaries was from decreases in loads delivered by the surface-water inflows.

Solute-transport models explicitly represent the mass transport of nitrogen from sources at the water table to receptors. The detailed parcel-scale data used to characterize nitrogen sources can be accurately represented as a source term in the models, and the effects of changes in the source terms on nitrogen loads to receptors of interest can be evaluated. This method can improve implementation of TMDLs for nitrogen in estuaries on Cape Cod by (1) avoiding the limitations associated with implicit methods for evaluating nitrogen loads, (2) improving the method used to incorporate nitrogen attenuation in surface waters, and (3) providing an understanding of the complex response of the system to wastewater-management actions.

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